Status and Trends of Eutrophication, Pathogen Contamination, and Toxic Substances in the Barataria-Terrebonne Estuarine System

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Publication # 22
Barataria-Terrebonne National Estuary Program
July 1995

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Acknowledgements

This project was funded by the Barataria-Terrebonne National Estuary Program through an Interagency Agreement No. 24400-94-17 between the Louisiana Department of Environmental Quality and the Center for Coastal, Energy and Environmental Resources of Louisiana State University. LSU CCEER provided the overall management for the project under the direction of Paul Kemp and Ivor van Heerdon (initial Program Manager), with the assistance of Dianne Lindstedt and Cathy Brignac (initial Program Coordinator).

We are especially grateful to the Designated Reviewer for Water Quality, Kerry M. St. Pé for his guidance, helpful insight, and avenues to data and information. We appreciate the help of the BTNEP program office from Steve Mathies, Richard DeMay, Tracy Kling, and Stevie Smith.

Employees of the Louisiana Department of Environmental Quality were especially helpful in the acquisition of data and reports: Emilise Cormier, Jan Boydstun, and Joe Holmes.

Unpublished data were generously shared by Tom Soniat, Nicholls State University, and Chris Madden, Horn Point Environmental Laboratory.

Fish kill data and information concerning fish kill data sets were obtained from Richard Bejarano, Harry Blanchette, Agent Frazier, John Hawk, Mathew D. Henrich, John A. Plumb, Agent Ogé, Hershel Morris, and Bobby Simoneaux. John K. Dameier and Quay Dortch provided a number of valuable insights and ideas about how to treat the data. Chris Piehler provided much help and access to the LDEQ data set. Jamison Lowe Higgins helped in our understanding of fill kill data quality, database structures and how the information can be useful.

Revisions to the report were facilitated by the comments of Paul Conzelman, John Day, Dennis Demcheck, Larry Hartzog, Sue Hawes, Albert Hindrichs, Barbara Keeler, Paul Kemp, Kerry St. Pé, and George Townsley.

We thank the LSU CCEER Cartography group, headed by John Snead, and the LSU CCEER Editorial group, headed by Michael Coffey, for their assistance in the generation of the final report.

Citation:

Rabalais, N. N., Q. Dortch, D. Justic', M. B. Kilgen, P. L. Klerks, P. H. Templet, R. E. Turner, B. Cole, D. Duet, M. Beacham, S. Lentz, M. Parsons, S. Rabalais, and R. Robichaux 1995. Status and Trends of Eutrophication, Pathogen Contamination, and Toxic Substances in the Barataria and Terrebonne Estuarine System. BTNEP Publ. No. 22, Barataria-Terrebonne National Estuary Program, Thibodaux, Louisiana, 265 pp. plus Appendices.

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EXECUTIVE SUMMARY
Introduction

The Barataria-Terrebonne estuarine system has been affected by diverse human activity. As in other estuarine systems, Barataria-Terrebonne has been a waste repository for point and nonpoint source inputs of nutrients and contaminants as well as atmospheric deposition. Ecosystem responses to water quality variability may be a consequence of natural and human-induced loadings from the estuarine watershed, the periphery (including the Mississippi and Atchafalaya rivers and the nearshore coastal waters), and the atmosphere. The cumulative effects of multiple stressors may be manifested in several ways, including elevated chlorophyll levels, noxious and toxic algal blooms, integrative ecosystem measures such as dissolved oxygen concentration, contaminant levels in water, sediment or organisms, elevated fecal coliform levels in water, shellfish bed closures, and fish kills.

The Barataria-Terrebonne National Estuary Program (BTNEP) is one of several programs administered under a federal-state partnership with the U.S. Environmental Protection Agency (EPA). A BTNEP Management Conference composed of over 100 citizens identified seven priority issues affecting the Barataria-Terrebonne estuarine system: hydrologic modification, sediment reduction, habitat loss, eutrophication, pathogen contamination, toxic substances, and living resources. To answer questions concerning these issues and to develop a management plan, data were compiled to document the status and trends of three priority issues: eutrophication, toxic substances, and pathogen contamination. Status and trends determination is difficult without an adequate data base. The data must be temporally and spatially complete enough to identify the natural variability of the system and to document deviations from the mean condition. For trends analyses, data records of 15 years or longer were analyzed for significant changes over time. Status was determined from data circa 1990 to present.

Classifying the causes for poor water quality or impaired ecosystem health is difficult. A single indicator may or may not provide adequate assessment, and combinations of indicators often provide a better assessment of environmental health. The data bases do not necessarily cover all problems in the system. A pollutant of concern or indicator of eutrophication may still exist but not be documented in the data. Features of the system may be changing but cannot be identified without a long-term data set.

A downward trend in a pollutant or contaminant or an upward trend in an indicator of improved ecosystem health indicates water quality control measures are probably having the desired effect. Several examples show that positive changes in water quality can occur in response to human intervention. A continuance of the management practices that contribute to improved water quality is the recommended action. In addition, management of resources should seek to minimize the inputs of various pollutants (nutrients, toxic substances, and pathogens) to the environment while balancing the requirements for ecosystem level health.

The water quality issues examined in this report were restricted to those issues that have, or are likely to have, an effect on water quality today. Additional issues may arise in the near future, with positive or negative effects on water quality. What is clear is that water quality will continue to change in the decades ahead through human influences.

Sources of Nutrients, Toxic Substances, and Pathogen Contamination

Watershed

Present sources of pollutants in the Barataria-Terrebonne estuarine system (exclusive of atmospheric deposition) are mostly generated within the system because no major rivers or streams flow into the estuary. Some Mississippi River water is siphoned into the Barataria basin at Naomi and Pointe a la Hache, and a diversion pumps river water into Bayou Lafourche at Donaldsonville. River flow reaches the lower bounds of the estuary via Fourleague Bay and western Terrebonne marshes, through the various passes of the bird-foot delta, and from the nearshore coastal waters. The magnitude of freshwater flows and maintenance of a fresh/saline gradient is controlled by the seasonal rainfall pattern.

Major and minor industrial point discharges are numerous and cumulatively account for large volumes of petroleum hydrocarbons, metals, and radionuclides to portions of the study area. Municipal point sources include high-volume sewage discharges from municipalities and smaller volumes from subdivisions and rural communities. Septic tanks, sewage/stormwater overflow, unsewered communities, pasturelands, and marshes are also contributors of nutrients, organic matter, and fecal coliform contamination. A frequent avenue for the addition of pollutants is by sewage/stormwater overflow or, as is the case for several areas, pumped stormwater drainage systems.

Mississippi and Atchafalaya Rivers

The Mississippi River watershed, the largest in the United States, encompasses 41% of the area of the conterminous 48 states. Major alterations in the main river channel, widespread landscape alterations in the watershed, and anthropogenic additions of nitrogen and phosphorus have resulted in dramatic water quality changes this century. Since the 1950s, the riverine nitrogen and phosphorus loadings to offshore waters have doubled and silicate concentrations declined by about 50%. Nitrate concentrations appear to have stabilized, but trends are masked by increased variability in the 1980s data. The seasonal variability of nutrient concentrations also has shifted. The present conspicuous spring peak in nitrate concentration in the river was not evident before the 1950s. The dissolved N:P:Si ratios of the Mississippi Delta thus changed during the last several decades to result in increased eutrophication and hypoxia in the Mississippi River delta bight primarily from changes in nitrogen loadings. The lower

Mississippi River receives numerous inputs from a wide variety of petrochemical and chemical industries, numerous municipal wastewater discharges, and the integration of multiple inputs from the watershed.

Nearshore Coastal Waters

The nearshore coastal waters adjacent to the Barataria-Terrebonne estuarine system are influenced by the freshwater, nutrient, and sediment fluxes from the Mississippi and Atchafalaya rivers. In turn, there is exchange of the nearshore coastal waters and its constituents with the lower portions of Barataria and Terrebonne bays, with the influence being greater on Barataria Bay.

Eutrophication

Eutrophication is the effect of natural or artificial addition of nutrients to water bodies. Eutrophication of estuarine ecosystems is a natural process that has been greatly accelerated by human activity. Not all nutrient increases are detrimental, but it is generally perceived that aquatic systems are limited in their assimilative capacities and that the effects of eutrophication are deleterious. It is largely accepted that increased nutrient loadings lead to increased primary productivity manifested in high chlorophyll concentrations. Shifts in phytoplankton species composition occur in response to nutrient increases and/or changes in the ratios of the essential nutrients: nitrogen, phosphorus, and silica. Noxious, harmful, or toxic algal species are becoming increasingly common in coastal waters as a result of nutrient alterations. Furthermore, where eutrophication occurs, hypoxia or oxygen depletion and sometimes fish kills often follow as a consequence of an increase in organic loading.

Nutrients and Chlorophyll *a*

Long-term changes in nutrients within the Barataria and Terrebonne estuaries were few, and inconsistent patterns occurred within one site. The few apparent changes were a slight decline in the nitrate+nitrite concentrations and/or total Kjeldahl nitrogen but increases in total phosphorus. Statistically, water quality trends indicated by nutrient concentrations were virtually non-existent. Nutrient concentrations have been used as diagnostic parameters of estuarine condition. Water bodies in Barataria-Terrebonne were classified across the full spectrum from healthy to fair–to–medium to high nutrient concentrations.

Algal growth bioassays indicate that the southern portions of Terrebonne Bay and the nearby coastal waters vary between nitrogen and a nitrogen+phosphorus limitation. The nutrient ratios from north-to-south in Barataria Bay indicate a change from phosphorus limitation in the freshwater headwaters to nitrogen in the south. Silica, required for diatom growth but usually omitted from agency water quality monitoring, was occasionally found to limit phytoplankton growth in algal growth bioassays. Phytoplankton growth also may be limited by the available light. Although this limitation may be significant in the turbid estuaries, phytoplankton production continues often at high rates. Water residence times in water bodies also influence phytoplankton growth.

Phytoplankton biomass may be roughly equated to photosynthetic pigments represented by chlorophyll *a* (Chl *a*). Data from Barataria Bay north-to-south show, in 1994 at least, high Chl *a* concentrations (100 $\mu g\ l^{-1}$) in the north during winter and a high spring peak in the southern area (40 $\mu g\ l^{-1}$). Lower values are found in Terrebonne Bay compared to Barataria Bay during 1982–1983.

Comparative data for chlorophyll *a* biomass for several areas within the Barataria and Terrebonne basins over several decades provide very strong evidence that eutrophication has occurred. Chlorophyll *a* has shown a five-fold increase in Lac des Allemands in the past 30 years, and the Barataria Bay samples indicate a doubling in the last 20 years. Samples from lower Terrebonne Bay indicated a doubling from the mid-1970s to the early 1980s.

Sediment records provide useful information on water quality changes. Sediment cores from sites in Barataria and Terrebonne salt marshes show a coincidental rise and fall between the remnants of diatoms (or a surrogate for phytoplankton biomass) and local fertilizer use. The effects of agricultural fertilizers appear significant and greater than the effects of population growth. Nonpoint runoff is a significant source of nutrients affecting the estuaries, even though they have relatively large wetland areas to buffer the nutrient loading. The sediment record indicates that the wetlands incompletely buffer the effects of increased nutrient loading on water quality and that the ability of the wetlands to absorb additional amounts of nutrients is much less than 100%.

Dissolved Oxygen

Oxygen-depleted waters are obvious manifestations of nutrient enrichment and organic loading. Decreases in oxygen content with respect to a saturation value can be attributed to the decay of organic matter, but not all oxygen depletion results from nutrient and/or organic enrichment. Low oxygen levels are more likely where stratification prevents re-aeration to bottom waters, where chemical oxygen demand is high when respiration of plants and phytoplankton during the dark cycle is extensive— especially in submerged aquatic vegetation beds and naturally stagnant waters. Several sites show an increase in oxygen saturation with time, over the last 17 years, but the majority display no trend. In the case of Bayou Black at Gibson, the frequency of hypoxic events has increased in spite of the increasing trend in oxygen saturation. Almost half of the locations still exhibit < 30% saturation in more than 50% of the cases (= poor water quality due to organic loading). Oxygen saturation values in the lower Mississippi River show a highly significant trend of increase at most stations, which correlates well with decreasing surface biological oxygen demand.

Historical and limited data for bottom-water dissolved oxygen indicates that oxygen depletion is likely to occur in poorly flushed environments with high organic loading, in deeper channels and bayous, in canals adjacent to produced water discharges, and in water bodies receiving high chemical oxygen demands or organic loading from sewage or other wastewater outfalls. Bottom waters severely depleted in oxygen are seasonally dominant features of the Louisiana and Texas continental shelf adjacent to the deltas of the Mississippi and Atchafalaya rivers. The areal extent of bottom-water hypoxia in mid-summer may cover up to 9,500 km^2 with the spatial configuration varying interannually. Movement of these waters onshore as a result of wind shifts is often the cause of massive fish kills.

Toxic and Noxious Phytoplankton

Blooms of toxic and noxious phytoplankton, or "red-tides" as they used to be called, are natural phenomena that may be exacerbated by several human activities such as nutrient pollution, aquaculture, and shipping. Such blooms have a variety of impacts from human illness and death, due to consumption of primarily, shellfish contaminated with toxins produced by certain algae, to mortality of other organisms at higher trophic levels, including commercially important species, and to loss of recreational and aesthetic value from water discoloration and unpleasant odors. In Louisiana there have been no known human health problems from consumption of algal toxins in shellfish or fish. In fact, it has generally been believed that the low salinity of Louisiana estuaries prevents the growth of all toxic algal species. However, there are published accounts of red tides in the region. Further, increasing coastal eutrophication has been accompanied by increases in toxic and noxious algal blooms in other ecosystems. Nutrient inputs to the Mississippi delta bight have increased since the 1950s to result in enhanced eutrophication. Changes in chlorophyll *a* concentrations suggest that similar increases in eutrophication have occurred in the estuaries. Thus, toxic and noxious algal blooms may be more likely now than in the past.

A variety of potentially toxic and noxious phytoplankton are observed in Bayou Little Caillou in the Terrebonne Bay estuary and in Fourleague Bay. A much larger variety of species and higher abundances of most species are observed in the offshore zone. Two taxa are especially abundant and occur frequently: *Pseudo-nitzschia* spp. (selected varieties cause amnesic shellfish poisoning [ASP]) and *Gymnodinium sanguineum* (discolored water, association with low oxygen, and ichthyotoxin producer). *Pseudo-nitzschia* spp. occur commonly in the nearshore coastal waters, and *Gymnodinium sanguineum* occurs frequently in the estuary.

There have been no documented human health impacts, but water discoloration events have been reported. Some blooms have extended for long distances along and across the shelf and have been associated with fish kills. A particularly insidious dinoflagellate, dubbed the "phantom killer," recently has been implicated in many unexplained fish kills in low salinity, highly eutrophied water bodies. Although its occurrence in Louisiana waters is unknown, it has been associated with estuarine fish kills from Delaware to Alabama.

The Barataria-Terrebonne estuarine system is at risk from problems caused by toxic and noxious phytoplankton. The greatest threat to human health is ASP; diarrhetic Shellfish

Poisoning (DSP) is the other possible threat. Abundances of *Psuedo-nitzschia* spp. can exceed thresholds that have led to toxic shellfish in other areas. None of the species known with certainty to cause DSP occur in this region; however, the toxins associated with DSP have been measured in oysters from Mobile Bay. Several species of *Dinophysis* and *Prorocentrum,* which are observed in the study area, have been implicated in causing DSP. Those species are more common in the offshore zone to suggest that potential threats to human health will be from consumption of oysters grown at higher salinities. Several algal species are linked to Paralytic Shellfish Poisoning (PSP)—the most common and most serious toxic algal problem—but the evidence for PSP agents in the estuary is tenuous. However, the release of untreated ballast water elsewhere has led to a global epidemic of PSP that could spread to this area.

Contaminants

Numerous potential sources of toxicants exist within the estuarine system: pesticides and herbicides, inputs from a few industries along the Mississippi River, light industry and domestic inputs from population centers, storm and urban runoff, atmospheric deposition, drilling fluids and produced waters from the oil and gas extraction industry, oil spills, and inputs from the Mississippi and Atchafalaya rivers through various avenues.

Toxic contaminants in the Barataria-Terrebonne estuarine system include elements (especially metals), organometals (e.g., tributyltin), radionuclides, and organic contaminants. Important among the latter are the chlorinated aromatic compounds (including PCBs), the chlorinated hydrocarbons (including DDT and many other pesticides), and the polycyclic aromatic hydrocarbons (or PAHs). The factors that determine risks to people and the ecosystem include toxicity, concentration, bioavailability, and persistence. Environmental contaminants may be very stable, toxic at low concentrations, and bioavailable. Moreover, several may have carcinogenic effects. These characteristics increase the likelihood of toxic effects in the environment and on human health. Water, sediment, and tissue samples provide different information on pollution. Contaminant levels in water are likely to fluctuate much more than in sediment or organisms. Many pollutants have an affinity for particles, so concentrations in sediment are usually magnitudes greater than those in water. Organisms differ in (1) the extent to which a contaminant is accumulated and (2) their capacity to detoxify contaminants. Biota sampling is likely to provide valuable information, but data on sediment and water are also needed to complete the status of environmental contamination.

Discharges

Toxic discharges to the Mississippi River have decreased since a 6-yr record high in 1987, but the decline is erratic. The discharges average 150–200 million pounds of toxic chemicals in 1992 for the stretch of the river from Old River junction to the Gulf of Mexico. Some toxic releases are decreasing with time, while others are increasing. There is an apparent increase over time of the toxic releases to the estuarine system; however, most of this increase may be from more rigorous reporting. Significant quantities of produced waters (oil-field brine) are discharged into the system at numerous locations. These effluents are elevated in salinity, trace metals, radionuclides, and many organic compounds, primarily petroleum hydrocarbons. Localized (within 300 m) or broad (up to 1,000 m) impacts of produced waters are identifiable adjacent to many of the discharges, e.g., contaminated waters and sediment, impacted biological communities, and bioaccumulation of contaminants in oysters. Elevated levels of contaminants are accumulated to depth in the sediment and will likely continue to pollute adjacent environments and affect benthic infauna. The system is particularly vulnerable to releases of oil-field fluids because of the numerous storage vessels, production facilities, and miles of pipelines, flowlines, and injection lines. With the decline in oil and gas production, the increase in injection lines and aging infrastructure, and residual contamination, resource management should strive to minimize further impacts to the environment from oil and gas activities.

Status and Trends of Contaminants

A comparison of discharges of chlorinated hydrocarbons and the ambient water quality gives a consistent picture of reduced discharges of these contaminants to the Mississippi River and their detection. The regulatory function seems to be having the desired result; however, river discharges are still among the highest in the United States and should continue to be reduced. The length of the Mississippi River is contaminated with agrochemicals, including herbicides and pesticides. Many of these compounds are transported to the lower Mississippi River and into the lower reaches of Barataria and Terrebonne bays.

Trends in contaminants within the basin are inconsistent where the data sets overlap in time and specific pollutants. Possible reasons for inconsistencies include differences in variables analyzed, differences in periods compared, and disparate areas covered by the data sets. Overall, it appears that levels of some contaminants in water (arsenic, cadmium, and lead at specific locations) have increased over the last decade, while others (copper, chromium, and mercury) have decreased with the trends varying among sites. Cases where contaminant levels have decreased appear more numerous than cases where contaminants increased to indicate that the overall trend is favorable. Oyster tissues showed statistically significant increases in silver, cadmium, copper, and lead from 1986 to 1990. Statistically significant decreases were evident for arsenic, dieldrin, chlordane, lindane, and low molecular weight PAHs and high molecular weight PAHs.

The status analyses did not reveal any major problems with respect to contaminants in the Barataria and Terrebonne basins; however, some minor contamination problems were evident. The Louisiana Department of Environmental Quality (LDEQ) data showed that levels of cadmium, copper, and lead in water from Bayous Segnette and Choctaw, and the Lower Grand River did occasionally reach levels that exceed EPA criteria for the protection of aquatic life, while elevated levels of arsenic and mercury in water may possibly affect human health (through the ingestion of water and/or fish consumption). The Estuarine Monitoring and Assessment Program–Estuaries (EMAP-E) data indicated possible problems for many of the 18 sites covered by the current data set. For some sites (notably Lakes Verret and Palourde, and to a lesser extent Bayou Terrebonne), this is reflected in sediment toxicity and sediment contaminant levels. This may indicate a general environmental impact. For the sediment at these sites, elevated levels of arsenic, nickel, dieldrin, endrin, and chlordane may be responsible for possible effects. For another group of sites (Barataria and Terrebonne bays, Lake Felicity, and Bayou Terrebonne), potential problems are associated with the human consumption of fish. Fish tissue contaminant analyses point to arsenic and several pesticides (mirex, aldrin, dieldrin, and heptachlor) as potential health problems associated with the fish consumption (specifically hardhead and gafftopsail catfish, which were the most commonly collected species). Levels in shrimp samples did not exceed health criteria. Mussel Watch data pointed out problems for arsenic, cadmium, and copper in oysters from the Oyster Bayou and Tiger Pass sites (as well as dieldrin in Tiger Pass oysters) to indicate contaminants in the Mississippi and Atchafalaya rivers are responsible for the elevated concentrations in oysters. In addition, high levels of organotin in oysters collected in Barataria Bay at Middle Bank should be a cause for concern.

The sites for which contamination problems are most evident from analyses of the various data sets, are Bayous Segnette and Choctaw, Lower Grand River, Lakes Verret and Palourde, Oyster Bayou, and Tiger Pass. These sites all fall on the periphery of the study area. The interior of the basins appears relatively clean, with the exception of produced water discharges and other possible contaminant sources not covered by the state and federal monitoring programs that form the basis of this review. Contamination sources for most of the sites on the periphery are not readily apparent, i.e., a combination of point and nonpoint sources. For Oyster Bayou and Tiger Pass the elevated contaminant levels are likely to be related to the inflow in these areas of water from the Atchafalaya and Mississippi rivers, respectively. This also means that in cases where Mississippi River water will be diverted into marshes as a marsh restoration method, pollutant levels might increase and should be monitored.

Contamination problems were identified in all of the environmental components monitored: water, sediment, and fish and shellfish. Contamination was found at various sites and for disparate contaminants. The occurrence of elevated metal levels in water of various waterways (especially Bayous Segnette and Choctaw, and Lower Grand River), sediment toxicity at various sites (in e.g., Little Lake, Bayou Terrebonne, and Lake Verret), elevated levels of metals and pesticides in sediment (especially Lakes Verret and Palourde), elevated levels of arsenic in catfish at several sites, and elevated levels of some contaminants in oysters (especially at Tiger Pass and Oyster Bayou) indicate contamination is fairly widespread. Contamination should therefore be a source of concern, though none of the contamination problems appeared serious enough to warrant immediate and drastic actions.

Fish Kills

An approach to understanding the effects of contaminants or poor water quality is to compile information on fish kills. Fish kills are a clear sign of acute stress. The source of the stress, however, may be anthropogenic or natural, or a complex combination of natural and human-induced factors. Fish kill data are useful in many ways but provide only partial or conservative information on the spatial and temporal dimensions of potential problems. For many reasons, fish kill data are far from complete and are inconsistently collected. A data base is only as worthwhile as the quality of data entered; we are cautious in applying too much emphasis to this analysis. However, some trends have emerged from the analyses.

- Most kills occur in the warmest months of the year.
- Naturally occurring events dominate the fish kills.
- Mortalities attributable to storm events comprise a significant percentage of the total number of reported fish killed.
- "Hot spots," or problem areas, and recurring kills include oil-field–related activities, marine transportation, dredging, pesticides from agricultural runoff, herbicide spraying of water hyacinths, release of chemicals used in fish farming, releases of incompletely treated effluent from fish-processing plants, paper mills, and sugarcane factories, and sewage treatment facilities.
- Fish kills related to pesticide use have increased. (A measure of this is attributable to the fact that fish kill investigations are more sophisticated recently).

Pathogen Contamination

The waters of the estuarine system contain productive shellfish growing areas. The potential for contamination with human fecal pathogens, as reflected in the occurrences of microbial indicators (i.e., fecal coliform bacteria) and marine pathogens, poses a risk to public health and affects the oyster industry. Only fecal coliform indicators are considered a "water quality" issue. However, naturally occurring marine bacteria in the family *Vibrionaceae* have the ability to cause human disease under certain conditions.

Fecal Coliform Indicators

Overall, there are no statistically significant trends in most probable number (MPN) fecal coliform counts over the last 15 years in the Barataria or Terrebonne estuaries in the eight sites selected for trends analysis. For the two river stations, only the east bank at the Pointe a la Hache Ferry Landing shows a significant decrease in MPN fecal coliform levels for the period 1980–1994.

Natural Marine Pathogens

There were 134 illnesses between 1980 and 1994 because of eight species of the natural marine genus *Vibrio*. *Vibrio* infections were the only cause of death from contact with marine waters or ingestion of raw seafood. There were 14 *Vibrio*-related deaths in the 15-yr period; all were in high-risk individuals with underlying illness. It appears the number of *Vibrio*-related illnesses increased in 1986; however, this is most likely because *V. vulnificus* illnesses became an issue with the press and the oyster industry. Potential *V. vulnificus* cases have been actively investigated in many shellfish-producing states since 1986. From that time, the levels of all *Vibrio* infections seem stable. *Vibrio*-related illnesses are not a major contaminant problem, but educational efforts should be continued to inform high-risk consumers and recreational users of the estuaries of the potential risk of infection, and especially of the risk of fatal wound infection or primary septicemia from eating raw seafood like oysters.

Pathogens from Fecal or Sewage Pollution

No cases of enteric bacterial illness were reported for 1980–1994. In 1982, the Louisiana Department of Health and Hospitals, Office of Public Health (LDHH, OPH) reported one outbreak of approximately 500 cases of mild Norwalk-like viral gastroenteritis (from human fecal pollution) associated with raw oyster consumption. Following this outbreak, a new system of seasonal classification for shellfish growing waters was implemented in 1982. A parishwide sewerage system also was installed for Terrebonne Parish. Since then, there have been no additional reported incidences of sewage-related illnesses from shellfish or seafood consumption.

The current system of seasonal classification of oyster growing waters has been extremely effective in preventing sewage-related contamination in oyster growing waters. The system is based on the fecal coliform indicator, which is not an effective indicator of human fecal viruses like Norwalk virus, but it is also based on temperature, which is somewhat effective for controlling human enteric viruses. The restrictiveness has been questioned but has undoubtedly also been responsible for the lack of illness from properly harvested waters.

Shellfish Bed Closures

It is difficult to draw conclusions concerning the trends in shellfish bed closures over the last 15 years because there are no available data to determine percentages of closures at any time. Visual observation of the closure lines from 1983 and 1994 indicate there are no greater areas of closure.

Recommendations or Management Concerns

Data, Information, and Monitoring Inadequacies

The data to evaluate how much the study area has changed in the past 20 years are amazingly sparse and need to be supplemented with current data. Needed is a re-survey of the water quality monitoring stations in mid-estuary sampled in the late 1960s and early 1970s. These surveys can serve to update our knowledge of the health of the estuaries and can be used as a basis for understanding management options if coordinated with other field research and monitoring programs. Important parameters missing from water quality monitoring are chlorophyll *a* and silicate.

The analyses of toxic and noxious phytoplankton point to several potential problems. It would be far better to know if these constitute real threats, so by appropriate monitoring and management, serious human health and economic consequences can be avoided. Two types of complementary information are necessary:

- Phytoplankton species distributions and an understanding of the environmental factors that regulate the occurrence of toxic and noxious phytoplankton, and

- Toxin analyses on oysters and other indicator species.

Algae can cause fish kills without being at high enough concentrations to discolor the water. Thus, the number of fish kills related to toxic and noxious algae may be underestimated. During investigations of fish kills where the cause is not immediately apparent, low oxygen is a suspected cause, or if discolored water is present, samples for identification of phytoplankton should be taken.

Considering the potential effects of arsenic and mercury, the LDEQ water quality program would greatly benefit from enhancing its current analytical capabilities for these elements. Although there were no identified problems with mercury contamination, continued monitoring of levels in water seems warranted, however, because it is highly toxic and because levels in water occasionally appear to exceed water quality criteria. Arsenic was identified as a problem contaminant in several areas. Considering the potential for human health effects, additional monitoring of contaminants in catfish is warranted. In addition, further research is needed to address the cause and effects of high butyltin levels in oysters in Barataria Bay at Middle Bank. A combination of methods, sediment contaminants, sediment toxicity, organism contaminant levels, and ambient water quality, would be an ideal program for identifying contamination problems.

Management to Ameliorate Eutrophication Impacts

Reducing the effects of eutrophication is the basis for numerous legislation. However, some less obvious implications are related to the interrelationship of policies of nutrient control in fresh water and the impact, or lack of impact, on coastal systems. The management of eutrophication on a national scale has not sufficiently integrated freshwater and estuarine systems. The primary nutrient targeted to improve water quality in freshwater systems is usually phosphorus. However, coastal systems are usually thought to be nitrogen limited, at least part of the time, and this includes the Barataria and Terrebonne estuaries. Sewage treatment upstream does not necessarily equate to controlling nutrient loading to downstream estuaries. The minimization and mitigation of nutrient consumption seem a less prudent management policy than reduction. If source quantities are reduced, then managing subsequent problems becomes easier.

River Diversions and Consequences

There are various plans and discussion of plans for substantial alteration of the freshwater inflows to these estuaries, and others in Louisiana. The proposed diversions move water with a nutrient concentration that is, in general, much higher in nutrients than the receiving water. The combination of even a 20% increase in freshwater inflow that contains very much higher concentrations of nutrients will very likely cause further eutrophication in the Barataria watershed. River concentrations of nitrate+nitrite are ten times greater than the proposed receiving waters of Barataria Bay, and silicate concentrations are twice as high. The observed changes of eutrophication are a two- to five-fold increase in chlorophyll levels over the last 30 years; the trend will likely continue upward. Other potential harmful effects of eutrophication include increase of incidence of hypoxia and anoxia and increase in catastrophic events like fish kills and toxic and noxious phytoplankton blooms.

Ideally, nutrient inputs should be minimized. The ability of the wetlands to assimilate additional nutrients is already less than 100% as evidenced from sediment cores that record the responses of diatom production over decades to changes in nutrient loadings. Nutrients may be absorbed or released by wetlands when water flows over or through them. The amount and direction of these exchanges are determined by the wetland type, the hydrologic regime, and the nutrient concentration and composition. Louisiana wetlands are not uniform in their uptake and release of nutrients. Swamps appear to take up some nitrogen and phosphorus forms, whereas wetlands with emergent macrophytes release these nutrients.

The nitrogen uptake rate for swamps in the Barataria basin can be used to calculate the hypothetical maximum uptake of total nitrogen from projected river diversions. If all swamp area in the basin (64,462 ha) were to have water diverted over it at the same rate as the experimental study and at the same nitrogen removal rate, then only 10% of a 1% diversion of the Mississippi River average flow would be removed. The application of these few measurements into management principles generally applicable to river diversions should be done cautiously, if only because scaling from small experimental study areas to a river diversion has not been done for Louisiana.

While toxic discharges affect ambient water quality—especially in the Mississippi River—they are not an overriding issue of concern to the health of most of the area. Exceptions occur on the periphery where point and nonpoint sources are most likely to impact water and sediment quality or where routine monitoring does not cover other possible contaminant sources such as produced water discharges. Elevated toxic contaminant levels in oysters from Oyster Bayou and Tiger Pass are likely related to the inflow from the Atchafalaya and Mississippi rivers, respectively. The fecal coliform levels in the lower Mississippi River range in the 200s–300s MPN FC/100 ml, which is well over the 14 MPN acceptable level for fecal coliforms in oyster growing waters.

The potential benefits of freshwater diversions may outweigh water quality issues but need to be weighed against potential increases in eutrophication and its effects, contaminant levels, and fecal coliform levels. Predicting and monitoring the impacts of diversions on water quality should be incorporated in the planning, construction, and operation stages for adaptive management that will minimize the potentially harmful effects.

INTRODUCTION

The Barataria-Terrebonne estuarine system has been affected by diverse human activity. As in other estuaries, Barataria-Terrebonne has been used as a waste respository for point and nonpoint source pollutants, including domestic and industrial discharges and agricultural runoff. It also has experienced large-scale hydrologic and sedimentologic modification. Eutrophication, toxic substances, and pathogen contamination have been identified as priority issues related to water and sediment quality problems in the Barataria-Terrebonne estuarine system. The Barataria and Terrebonne basins are bordered by agricultural lands, urban areas, and rural communities and receive nutrient and contaminant loadings from multiple sources, including atmospheric deposition. Significant industrial and municipal effluents enter the Mississippi River between Baton Rouge and New Orleans, and lesser amounts enter the lower Atchafalaya River. Ecosystem responses to water quality variability in the estuaries may be a consequence of natural and human-induced loadings from the estuarine watershed and offshore component.

A gradient exists along the axis of the Barataria estuary in salinity, nutrient concentrations, phytoplankton biomass and production, and turbidity. A similar, but not as dramatic gradient occurs along the axis of Terrebonne Bay. The relative inputs of nutrient loadings and sources change along the same gradients away from municipal and agricultural sources on the upper end, to the plume of the Mississippi River and Louisiana Coastal Current on the offshore end. The influence of the Mississippi River and coastal boundary diminishes to the west off Terrebonne Bay—where the influence of the Atchafalaya River dominates in the westernmost portion of the Terrebonne basin—but not to a large extent in the coastal boundary. The three major nutrient loadings (atmosphere, within basin, and oceanic) occur at different places and times within a year and have changed historically at different scales and times this century.

The estuarine system is subject to a combination of point and nonpoint source inputs of nutrients and contaminants as well as atmospheric deposition. Point discharges include municipal wastewater treatment plants, National Pollutant Discharge Elimination System (NPDES) permitted discharges, and Louisiana Department of Environmental Quality (LDEQ) permitted discharges (e.g., produced water discharges). Nonpoint sources include agricultural runoff and stormwater pump discharge. Cumulative effects of these multiple stressors may be manifested in several ways, including for example, phytoplankton or bacterial community composition, water quality indicators such as high chlorophyll levels, integrative ecosystem measures such as oxygen concentration, contaminant levels in water, sediment or organisms, pathogen contaminant levels in shellfish, and fish kills.

Objectives

The Barataria-Terrebonne National Estuary Program (BTNEP) is one of several programs administered under a federal-state partnership with the U.S. Environmental Protection Agency (EPA). A BTNEP Management Conference composed of over 100 citizens identified seven priority issues affecting the Barataria-Terrebonne estuarine system: hydrologic modificaiton, sediment reduction, habitat loss, eutrophication, pathogen contamination, toxic substances, and living resources. To answer questions concerning these issues and to develop a management plan, data were compiled to assess their current status and historical trends.The goal of this report is to document the status and trends of three priority issues: eutrophication, toxic substances, and pathogen contamination. Appropriate literature and data sets were assembled to define the nature and extent of each issue. Where appropriate, data were analyzed to identify trends in changes of key parameters. Empirical and logical relationships were developed between key parameters and relevant environmental and system-level characteristics to develop interrelationships between water and sediment quality changes and changes in inputs, landscape usage, and anthropogenic impacts. Finally, recommendations were developed for addressing problems in the priority issues and ecosystem level management.

Study Area

The Barataria-Terrebonne estuarine system (figure 1) covers an area of approximately 6,500 mi^2 and comprises water bodies and wetlands within the Mississippi delta plain filling the interdistributary basins between the two active deltas of the Mississippi River and the Mississippi River proper and the Atchafalaya River. The system contains those tidally influenced environments delimited by the west bank levees of the Mississippi River to the north and east. Western boundaries are the Atchafalaya Bay, Atchafalaya River, and the Atchafalaya Basin East Guide Levee. The Barataria-Terrebonne estuarine system begins on the north near Morganza in Pointe Coupee Parish. The southern boundary is the Gulf of Mexico. Because some of the water bodies such as Caillou Bay are broadly open to the Gulf, this boundary in places is arbitrary.

The estuarine system consists of two reasonably discrete basins. In the Barataria basin, drainage eventually enters Barataria, Caminada, or Bastian bays; drainage in the Terrebonne basin ultimately enters Timbalier, Terrebonne, Caillou, or Fourleague bays or Lake Pelto. The natural levees along Bayou Lafourche provide a barrier to interchange between the two basins, but a variety of canals, including the Gulf Intracoastal Waterway, allow some exchange.

The estuary is physiographically diverse. Terrain within the system varies from alluvial flood plains in the north to expansive coastal marshes in the south. The entire area is interlaced with relict distributaries (bayous) and their associated natural levees, which are the remnants of past deltas. A large number of shallow water bodies dominates the landscape.

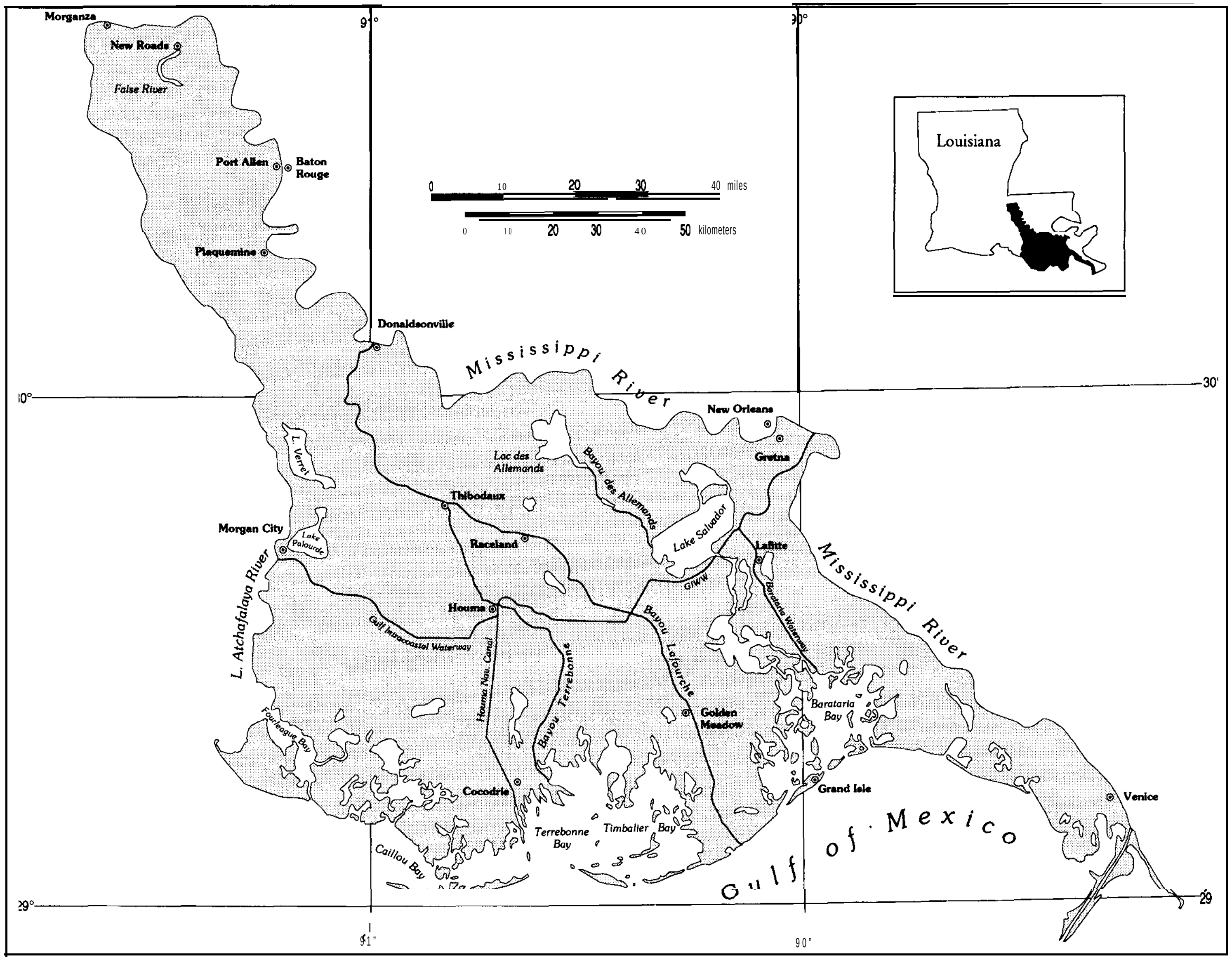


Figure 1. Map of the Barataria-Terrebonne estuarine system.

The system contains 19 major lakes ranging in surface area from 2.2 mi^2 to 70 mi^2. Several natural and human-made waterways transect the estuary. These streamsand bayous have low gradients, sluggish flow, and long retention times. The Gulf Intracoastal Waterway transverses the Barataria-Terrebonne estuarine system, connecting the Mississippi River though the Algiers and Harvey locks in the east to the Atchafalaya River at Morgan City in the west. Other major waterways and passes include the Houma Navigation Canal, Barataria Bay Waterway, Quatre Bayou Pass, Pass Abel, Bayou Segnette Waterway, Bayou Terrebonne, Lafourche-Jump Waterway, Empire Waterway, and Caminada Pass. Natural waterways typically run south to the Gulf, and canals provide east-west access. Such alterations have greatly modified the natural hydrology of the area.

Constituents from water bodies outside the study area have a known and significant impact on the waters of the two basins. These areas include the lower Atchafalaya River, the lower Mississippi River, and the Louisiana bight and nearshore coastal waters.

Sources of Nutrients, Toxic Substances, and Pathogen Contamination

Watershed

The sources and activities that affect the water quality of an estuary are varied and many. Sources are often divided into point and nonpoint sources, but the division between them often becomes blurred. The present sources of pollutants in the Barataria-Terrebonne estuarine system (exclusive of atmospheric deposition) are generated mostly within the system because no major rivers or streams flow into the estuary. Some Mississippi River water is siphoned into the Barataria basin at Naomi and Pointe a la Hache. Bayou Lafourche receives an approximate discharge of 11.2 cm s^{-1} of Mississippi River water at a pumping station at Donaldsonville (Doyle 1969). A large portion of the Mississippi River flow reaches the lower bounds of the estuary via Fourleague Bay, Oyster Bayou (Atchafalaya), and through the various passes of the bird-foot delta (Mississippi). Once offshore, riverine fluxes of nutrients, fresh water, sediment, and contaminants may be exchanged with the lower ends of the estuarine system through tidal exchange. The tidal water entering the estuary is nearly always measurably diluted with river water.

The magnitude of freshwater flows and maintenance of a fresh/saline gradient is controlled by the seasonal rainfall pattern (Reed et al. 1995). The average annual rainfall is about 150 cm with about 75 cm evaporated yearly. The surplus of 75 cm infiltrates the soil or runs off through the estuary.

Major and minor industrial point sources require NPDES permits from EPA and/or permits from LDEQ, and regulatory controls are in place to control the pollutants that are discharged. Produced water discharges, once excluded from NPDES permitting and regulated by LDEQ in their permitting program, are now regulated by EPA under a General Permit system (see p. 110). These discharges are numerous and cumulatively discharge large volumes of petroleum-derived hydrocarbons, metals, and radionuclides to portions of the Barataria-Terrebonne estuary. Nonpoint sources contribute pollutants, nutrients, or pathogens to receiving rivers and streams at numerous and widespread locations rather than at a single discharge point. A frequent avenue for the addition of pollutants to the Barataria-Terrebonne system is by sewage/stormwater overflow or, as is the case for several areas, pumped stormwater drainage systems.

Impairment of water bodies by pathogen indicators (fecal coliform bacteria) is a major contributor to the impaired use of Louisiana's waters. Municipal point sources are cited as the leading cause of impairment due to high fecal coliform levels. These sources include high-volume sewage discharges from municipalities, and smaller volumes from subdivisions and rural communities. Septic tanks, sewage/stormwater overflow, unsewered communities, pastures and range lands, and wetlands also are contributors of fecal coliform contamination. Sewage sources contribute to the nutrient enrichment and organic carbon loading of estuaries. Larger wastewater treatment facilities handle toxic sources.

Agricultural sources, including cropland, pastureland, feedlots, aquaculture, and silviculture, are also a major source of impairment among Louisiana water bodies. Agricultural runoff can result in nutrient enrichment and organic loading. The rapid rise in fertilizer application in the United States since 1930 appears to have stabilized in the 1980s and early 1990s (figure 3 in Starke et al. 1994; also, see p. 30). Fertilizer use in the Terrebonne basin is similar to levels in the 1930s–1950s but has increased substantially since then in the Barataria basin (see p. 50). Fertilizer applications in Jefferson and St. Charles parishes also have increased.

Atmospheric inputs of pollutants are considered a nonpoint source and may reach the watershed and estuaries via precipitation or dry deposition. Pollutants in atmospheric deposition originate from the combustion of fossil fuels, incineration, and also emissions of nitrous oxides and ammonia from agriculture. Areas of the United States with the greatest rainfall and atmospheric pollution have the greatest amounts of nitrogen and other constituents deposited from the atmosphere. In general, the Barataria-Terrebonne system ranks in the third quartile for estimated atmospheric deposition of nitrogen compared to the remainder of the United States (Puckett 1987), but quantification of atmospheric input to the Barataria-Terrebonne estuarine system is lacking.

Mississippi and Atchafalaya Rivers

The Mississippi River watershed, the largest in the United States, encompasses 41% of the area of the conterminous 48 states. The Mississippi River system ranks among the world's top ten rivers in discharge (580 $km^3\ yr^{-1}$) and sediment yields (210 x 10^6 $t\ yr^{-1}$) to the coastal ocean (Milliman and Meade 1983). Major alterations in the main river channel and widespread landscape alterations in the watershed along with anthropogenic additions of nitrogen and phosphorus have resulted in dramatic water quality changes this century (Turner and Rabalais 1991a). Approximately one-third of the flow of the Mississippi River system enters the Gulf via the Atchafalaya River. Of the remaining discharge from the Mississippi

River Delta proper, approximately 53% flows westward onto the Louisiana shelf (U.S. Army Corps of Engineers 1974). The peak in flow usually occurs in March–May. Although flow is reduced in summer, large-scale circulation patterns often retain the fresh water on the shelf. Freshwater inflow from the Mississippi and Atchafalaya is a major feature of the Louisiana shelf, even though it is a fairly open system.

Since the 1950s, the riverine nitrogen and phosphorus loadings to offshore waters have doubled, and silicate concentrations declined by about 50% (see p. 22). Nitrate concentrations appear to have stabilized, but trends are masked by increased variability in the 1980s data. The Mississippi River below St. Francisville receives numerous inputs from a wide variety of petrochemical and chemical industries and numerous municipal wastewater discharges as well as the integration of inputs from the watershed above.

Water from the Mississippi and Atchafalaya rivers reaches the Barataria-Terrebonne estuarine system minimally through small diversions at Naomi and Pointe a la Hache and the diversion at Bayou Lafourche and maximally at the terminus of their deltas. Fourleague Bay, the western Terrebonne marshes on the west, and the southeastern portion of the system within the birdfoot delta receive the most direct freshwater inflows. Discharge onto the shelf from both deltas rapidly forms the Louisiana Coastal Current, which flows on average westward along the Louisiana coast. Wind reversals in late spring and summer tend to reverse the flow so that much of the freshwater content is held on the shelf.

Mississippi River water quality and its associated freshwater, nutrient, sediment, and contaminant load currently affects the water quality of the Barataria-Terrebonne system. Further importance in understanding the role of river water quality is necessitated by planned management strategies for the basins, which include the introduction of sediment and water from the Mississippi River.

Nearshore Coastal Waters

The nearshore coastal waters adjacent to the Barataria-Terrebonne estuarine system are influenced by the freshwater, nutrient and sediment fluxes from the Mississippi and Atchafalaya rivers. In turn, there is exchange of the nearshore coastal waters with the lower portions of Barataria and Terrebonne bays, with the influence being greater on Barataria Bay. The implications of water exchange with nutrients and phytoplankton species is discussed in the section on eutrophication. The presence of an extensive oxygen-depleted bottom water layer also is implicated in massive fish kills. An approximate 10-m water depth contour to the southern extremes of Barataria and Terrebonne bays constitutes the offshore boundary of the study area (figure 2).

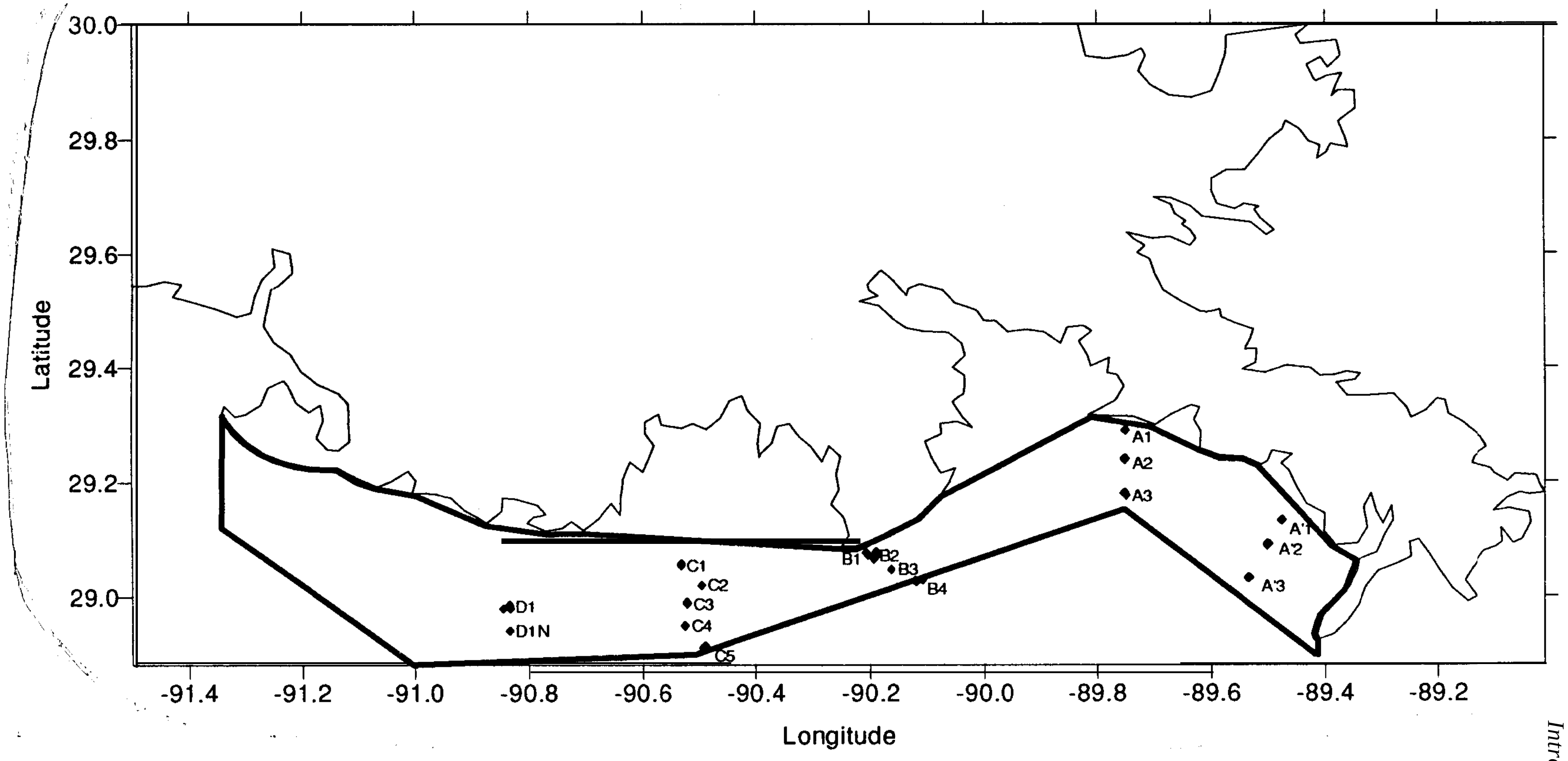


Figure 2. Outline of the nearshore coastal portion of the Barataria-Terrebonne estuarine system.

Issues

Eutrophication

Eutrophication is the effect of natural or artificial addition of nutrients to water bodies (Rohlich 1969). Eutrophication of estuarine ecosystems is a natural process that has been greatly accelerated by human activity. Most of the waters of the middle and upper Barataria-Terrebonne estuarine system are eutrophic (Conner and Day 1987). There is strong evidence despite high natural variability that eutrophication, as measured by chlorophyll *a* biomass, has increased in recent decades in the Terrebonne and Barataria watersheds (see p. 45). Eutrophic waters are characterized by frequent algal blooms, fish kills caused by low levels of dissolved oxygen, and increasingly the potential for noxious and toxic phytoplankton blooms. Domestic wastes are an increasing nutrient problem because of diffuse inputs from unsewered communities and inadequate municipal sewage systems. Urban runoff is a significant pollution source from cities on the west bank of the Mississippi River and the cities of Houma and Thibodaux. Agricultural practices that are widespread within both basins, predominantly in Barataria, are significant sources of nutrients. Atmospheric deposition is another, as yet unquantified, source of nutrients.

Toxic Substances

Numerous potential sources of toxicants exist within the estuarine system: pesticides and herbicides from agriculture (particularly citrus and sugar cane) located on the natural levees along the Mississippi River and major bayous; herbicides used in aquatic weed control; inputs from a few industries along the Mississippi River; light industry and domestic inputs from population centers; storm and urban runoff; atmospheric deposition; drilling fluids and produced waters from the oil and gas extraction industry; oil products spills; runoff and leachates from hazardous waste sites; and inputs from the Mississippi River itself. The greatest inputs of toxic substances into the Barataria-Terrebonne estuarine system are likely associated with discharges along the eastern margins of the Barataria basin because of the high concentration of heavy industries, large urban centers, and agricultural areas along the river corridor. There are hundreds of oil and gas wells in numerous fields in the estuarine and wetland environments of the Barataria-Terrebonne system. Petroleum activities, oil products spills and in-place contaminants have caused loss of designated uses in nine waterbody segments of the Barataria basin and 21 in the Terrebonne basin because of levels of "oil and grease" (LDEQ 1993a). Pesticide concentrations in excess of EPA and U.S. Army Corps of Engineers (USACE) criteria have been recorded in several water bodies. The use of azinphos methyl, an organophosphate pesticide, lead to several large fish kills in 1991–1992, following runoff from sprayed sugarcane fields into streams (St. Pé and DeMay 1993).

Pathogen Contamination

The potential for contamination of the important shellfish growing areas with human fecal pathogens, as reflected in the occurrence of microbial indicators (i.e., fecal coliform bacteria), and marine pathogens poses public health risk and affects the Louisiana oyster industry. Septic tanks and urban and agricultural runoff constitute diffuse sources of potential pathogens. In the Terrebonne basin, 14 towns have known septic tank problems. Twenty-three waterbody segments in this basin indicated partial loss of their designated uses because of pathogen contamination, mainly resulting from unsewered communities, but urban runoff contributes to ten of these segments and agricultural runoff to four. In the Barataria basin, nine waterbody segments indicated pathogen contamination: nine resulting from unsewered communities, two from agricultural runoff, and four from urban runoff (LDEQ 1993a).

Methods

Appropriate literature and data sets were assembled to define the nature and extent of each priority issue. Literature reviews were of limited use in assessing status and trends but did provide a framework for data interpretation. Considerable routine water quality monitoring data reside with the LDEQ, the U.S. Geological Survey (USGS) Water Resources for Louisiana, USACE, and the Louisiana Department of Health and Hospitals (LDHH). Data from more directed federal programs such as NOAA's National Status and Trends Program, Mussel Watch Project, and the EPA Estuarine Monitoring and Assessment Program–Estuaries (EMAP–E) provided more specific information but usually over a less-intense spatial and temporal scale. Most of the large data sets are available from the EPA STORET system or commercial vendors. Several other data sets were examined and used if appropriate data were found. Data compiled specifically for this project will be housed at the Barataria-Terrebonne National Estuary Program office.

Determination of status and trends is difficult without an adequate data base. The data must be temporally and spatially complete enough to identify the natural variability of the system and to document deviations from the mean condition. For many trends analyses, data records of 15 years or longer were analyzed for significant changes over time. The more current status of the Barataria-Terrebonne system was determined from data circa 1990 to present. The results for this report are based on data available for analysis. Statistically significant trends or differences are those highlighted in the report. Site-specific studies or studies with limited temporal resolution were inappropriate to the overall objectives of this report.

Notes

The data bases do not necessarily cover all problems in the system. A pollutant of concern or indicator of eutrophication may still exist but not be documented in the data. Features of the system may be changing but cannot be identified without a long-term data set. Also, a single indicator may or may not provide an adequate assessment of status and trends of the priority issues. Combinations of indicators provide a better assessment of the ecological health of the system. Single indicators need to be examined within the context of the others.

It is also important to note that a downward trend in an indicator of pollution does not mean that the particular pollutant or indicator of poor ecosystem health is not a problem for the Barataria-Terrebonne estuarine system. The downward trend indicates that some pollution abatement regulations or water quality control measures are probably having the desired effect. Alternatively, these changes are possibly the result of economic changes, including alternatives or reduced profit. Several examples show that positive changes in water qality can occur in response to human intervention. A continuance of the management practices that contribute to improved water quality is the recommended action.

Other priority issues were identified by BTNEP for assessment of status and trends. These included hydrologic modification, reduction in sediment availability, habitat loss/modification, and living resources. An additional study was conducted of the land-use and socio-economic status and trends of the Barataria-Terrebonne estuarine system. Individuals interested in more details of the other priority issues are encouraged to pursue the reports (see General References, p. 241). This report, however, is specific to the priority issues of eutrophication, toxics, and pathogen contamination. Other aspects of water quality or modifications to the ecosystem that might affect water quality will be addressed by the BTNEP Comprehensive Conservation & Management Plan (CCMP). Many of these issues were beyond the scope of this report, or data were not available to address the problems.

Water quality issues examined in this status and trends report were restricted to those issues that have, or are likely to have, an effect on water quality today. There are additional issues that may arise in the near future. Some of these additional issues are listed in table 1. This list is neither all-inclusive nor are all consequences known. There may be either positive or negative effects on water quality. Some will have local consequences and others cumulative or regional consequences. The management concerns may vary depending on the water quality issue, the driving force, and the degree of changes anticipated. What is clear is that water quality will continue to change in the decades ahead as a result of human influence.

Table 1. Checklist of future issues affecting water quality.

Issue	Outline
Marsh Management	hydrologic change within and between wetlands; vegetation management; aquaculture use of lands
"Red Mud"	wetland disposal of potentially toxic materials (by-products of aluminum extraction)*; under review by Louis. DNR Coastal Restoration Program
Large River Diversions	wetland gain and loss issues, pathway alterations of constituents in the river; flushing of estuary; this involves larger diversions than presently planned, e.g., Davis Pond
Urban Expansion	wetland dredge and fill; sewage and landuse changes
Agriculture Policy	nonpoint runoff and landuse changes
Bayou Lafourche Diversion	increased diversion of Mississippi River water into the BTNEP area
Regulation	public policy evolution; value judgments and cumulative impact management; short- and long-term management choices
Global Climate Change	loss of wetlands; weather changes, including number and strength of strong weather patterns
Continued Decline in Oil and Gas Recovery	maintenance dredging changes; closing old fields; aging infrastructure with increased incidence for breakdowns in safety equipment and conduits

* see (Gambrell and Prasana 1994, Kong and Mendelssohn 1994, Gatliff ND) for further information.

EUTROPHICATION

Marine pollution concerns in the United States often have focused heavily on toxic substances (heavy metals, petroleum hydrocarbons, and other organic compounds), the effects of specific discharges (power plants, industrial plants, municipal waste waters), and dumping activities (sewage sludge, dredged material, and industrial wastes). Other less identifiable sources of pollutants enter rivers, estuaries, and coastal areas from urban runoff, agricultural runoff, rural communities, and the atmosphere. Many of these nonpoint sources are additive to point sources and contribute labile organic material, nutrients, and chemicals to the receiving water bodies. There is increasing concern in the United States and other nations that eutrophication of coastal waters from multiple sources may be having pervasive effects on living resources.

Nutrient Enrichment

There has been considerable discussion and usually lack of agreement on the definition of "eutrophication." For this report, the definition that will be used was developed at a National Academy of Science (NAS) symposium on eutrophication in 1967 (NAS 1969), i.e., *Eutrophication is a natural or artificial addition of nutrients to bodies of water and to the effects of added nutrients.* Not all nutrient increases are detrimental. An adequate supply of essential nutrients is required to support food webs, and intentional nutrient additions have been shown to increase fish stocks in some experimental systems. It is generally perceived, however, that aquatic systems are limited in their assimilative capacities and that the effects of eutrophication are deleterious.

Eutrophication has been a worldwide phenomena of aesthetic, social, health, environmental, and economic concern for many years and is the subject of numerous scientific and management reports and meetings. Eutrophication affects coastal habitats through changes in food source quantity and quality, habitat suitability, alterations of predator-prey relationships, catastrophic events, ecosystem predictability, and intraspecific competition. We seem to know more about what is on this list, which is seemingly all-encompassing, than of the scientific understanding of the subtle interactions. We also have limited experience treating the symptoms. However, we have the experience to recognize the validity of the list, even though it has not really been compiled for coastal systems in a way comparable to freshwater systems.

The nutrient inputs and concentrations within an estuarine body or coastal area originate from sources outside the system (allochthonous) and from within the system (autochthonous). It is largely accepted that increased allochthonous loadings of nutrients can lead to increased primary production (Nixon et al. 1986, Oviatt et al. 1986, Malone 1987). The quantification of these relationships and paths of nutrient uptake and regeneration are not, however, adequately known. Several studies suggest that recycled nutrients account for a greater percentage of the ambient nutrient concentration than the "new" load entering the system each year (Fisher and Doyle 1987). Although autochthonous loadings are essentially unmanageable, an understanding of their contribution is necessary in understanding the effects of all sources of nutrient inputs.

Nitrogen and phosphorus are the primary anthropogenic nutrient inputs of concern in coastal waters. Nitrogen is considered the most important nutrient in marine and coastal waters, and phosphorus, in freshwater systems. Nitrogen and phosphorus are important in estuaries, depending on the season, total nutrient loadings, and various physical and chemical conditions. Silica, which is delivered by rivers and altered as an indirect consequence of phosphorus loading, varies in its availability and may strongly influence the deleterious effects of eutrophication. The relative proportion of nutrients will likely vary down the stem of the estuary, and the subsequent responses of phytoplankton concentrations also will change. Other chemical constituents such as carbon, trace metals, dissolved organic compounds (e.g., amino acids), and chelators also may be important in specific systems and under certain conditions. Physical features of the water body (e.g., light or temperature) also control the expression of nutrient additions (i.e., primary production and chlorophyll *a* biomass).

The factors that affect water quality of an estuarine basin are many. The rise in nutrient loading from sewage, industry, and urban and agricultural runoff into coastal habitats is widespread (Nehring 1984, Fransz and Verhagen 1985, Rosenberg 1985, Lancelot et al. 1987, Andersson and Ryberg 1988, Wulff and Rahm 1988, Turner and Rabalais 1991b). A major global source of the new nitrogen and phosphorus entering estuaries is mostly from fertilizer applications (Newbould 1989) and to a lesser extent, the combustion of fossil fuels. Atmospheric sources of nitrous oxide and ammonium also are associated with nitrogenous fertilizer application and livestock wastes in areas of intensive stock holding (Morris 1991, Matthews 1994). Sewage outfalls are a dramatic, but only symptomatic fruition, of the more basic rise in consumption of basic commodities, and of the need to dispose of the undesirable by-products. There would be no need for disposal if the products were not grown, produced, and consumed. Sewage treatment systems, however, are in themselves problematic because of varying levels of operation efficiency. There are a large number of home sewerage package units that are very poorly operated and a fewer but significant number of larger systems with severe problems. World fertilizer consumption has been increasing since World War II, when industrial processes evolved for fixing atmospheric nitrogen into fertilizers and mining of P-rich mineral deposits expanded rapidly. Additional nutrients are released through devegetation, farming, soil erosion, weathering, etc. A linear relationship between fertilizer application and water quality is not always expected because of the interaction of various ecosystem components and subsequent adjustments by microorganisms, in particular (see for example, Aber et al. 1989).

Effects of Eutrophication

The potential and observed consequences of nutrient enrichment for coastal habitats include changes in food source quantity and quality, habitat suitability, alterations of predator-prey relationships, catastrophic events, ecosystem predictability, and intraspecific competition (table 2). Increased phytoplankton production and biomass are the likely result in an otherwise nutrient-limited estuarine or coastal food web. Although there are no phytoplankton indicator species of incipient or advanced stages of coastal eutrophication presently identifiable, several researchers (e.g., Smayda 1990, Cherfas 1990, Cadée 1990) have identified a significant community-structure shift occurring globally at the phylogenetic level in response to coastal nutrient enrichment (see p. 80). Smayda (1990) further notes that this phylogenetic shift has been toward increased abundance and seasonal dominance of flagellates and non-motile, nannoplankton chrysophytes and in some cases, N-fixing bluegreen algae. Many of these species may be noxious, harmful, or toxic in increased concentrations, and their sinking and decomposition in the water column or at the seabed may contribute to increased hypoxic/anoxic episodes.

Diatoms are thought to provide the primary energy source for traditional food webs that support teleosts as top predators. The abundance of coastal diatoms is influenced by the silicon supplies and the ratio of silica to nitrogen and phosphorus. Diatoms out-compete other algae in a stable and illuminated water column of favorable silicate concentration. When nitrogen increases and silicate decreases, flagellates may increase in abundance to form blooms (Officer and Ryther 1980). In particular, noxious blooms of flagellates are becoming increasingly common in coastal systems (many examples are in Shumway 1990). Zooplankton—the main consumers of whole diatoms and a staple of juvenile fish—are thus affected by these nutrient changes in a cascading series of interactions. Furthermore, where eutrophication occurs, hypoxia often follows, presumably as a consequence of this increase in organic loading. Supportive evidence of this benthic-pelagic coupling is the observation of Cederwall and Elmgren (1980), who demonstrated a rise in macrobenthos around the Baltic islands of Gotland and Oland that they attributed to eutrophication (Nehring 1984). The widespread occurrence of changes in nutrient loading to coastal zones also has changed the annual and seasonal variability of nutrient concentrations. A change in the timing of the spring bloom that supports fish entering estuaries to feed during critical recruitment periods also may be an important consequence of eutrophication.

Oxygen-depleted waters are obvious manifestations of nutrient enrichment. Where eutrophication occurs, oxygen depletion often follows as a consequence of the increase in organic loading that is stimulated by increased nutrients. Excessive production of organic material in surface waters may sink to the lower water column or seabed either directly, as grazed material, or advected. Decomposition of these materials may lead to oxygen depletion.

Table 2. Examples of coastal eutrophication and its effects (adapted from Turner and Rabalais 1991b).

Area	Probable or Observed Effect[1]	Reference
Adriatic Sea	ox.; turb.; food chain Justic´ 1991, Faganeli et al. 1985,	Krstulovic´ and Solic´, 1990; Stachowitsch 1986
Baltic Sea, incl. Kattegat and Skagerrak	ex. al.; food chain; ox., macrophyte gain; incr. sec. prod.	Rosenberg 1985, 1986; Rosenberg and Loo 1988; Cederwall and Elmgren 1980; Andersson and Rydberg 1988; Ankar 1980
Bayou Texar, FL	ox., food chain, hlth., nox.	Moshiri et al. 1981
Chesapeake Bay	food chain, incr. prod., ox.	Seliger et al. 1985
Great South Bay, Long Island, NY	ex. al.	Ryther 1954
Lac de Tunis, Tunis, Tunisia	ex. al.; ox.; macrophyte loss; macrophyte gain; food chain	Kelly and Naguib 1984
Mississippi R. Delta Bight, U.S.	changes in area and extent of hypoxia	Rabalais et al. 1991
Southern Bight of the North Sea	ex. al.; incr. prod., food chain; nox.; incr. sec. prod. Westernhagen and Dethlefson	van Bennekom et al. 1975, Beukema 1991, van 1983, Lancelot et al. 1987
Tampa Bay, FL	macrophyte loss; turb.; hlth.; ox.; nox.; food chain	Johansson and Lewis 1991 Santos and Simon 1980

[1]Key to abbreviations:

ex. al.	= excessive algal growth (including filamentous and attached)
food chain	= food chain alterations affecting important fisheries species, including fish kills, loss of benthic organisms
incr. al. prod.	= increased primary productivity
incr. sec. prod.	= increased secondary productivity, including benthos
nox.	= noxious algal blooms
hlth.	= health problems with seafood consumption
macrophyte loss	= loss of important macrophytes
macrophyte gain	= gain of macrophytes
macrophyte inv.	= invasion of undesirable macrophytes
ox.	= low oxygen levels
turb.	= increased turbidity from phytoplankton growth

Eutrophication may cause the loss of emergent and submerged macrophytes that limit fisheries species during critical recruitment periods. It is well established that certain coastal fisheries species seem to require a physical structure to escape from predators while young. Where the area of estuarine macrophytes declines or improves, fisheries harvest is observed to respond proportionally (e.g., Turner and Boesch 1987). The loss of seagrass beds following decreased water clarity is often observed (Cambridge and McComb 1984, Cambridge et al. 1986), leading to the conclusion that the potential harvest of dependent fisheries will probably decline. Such subtle changes are difficult to detect without substantial amounts of long-term data (two examples are in Turner and Boesch 1987). Numerous examples exist of the impacts of low oxygen conditions on reduced benthic fauna, in terms of reduced abundance or decreased diversity. Areas impacted by low oxygen are a concern with respect to fisheries resources either because of direct mortalities, avoidance of hypoxic regions, altered migration, reduction in available habitat, changes in food resources, and increased susceptibility to predators (including humans).

To adequately assess eutrophication it would be necessary to consider nutrient concentrations and ratios over the appropriate spatial and temporal scales as well as the numerous indicators of eutrophication (e.g., chlorophyll *a* biomass, turbidity, algal composition, noxious phytoplankton, oxygen depletion, etc.). Assessment of changes in nutrients and/or the effects of nutrient additions requires a data base with an adequate temporal resolution over a long time. This review focuses on the nutrient structure and some of the more obvious indicators of nutrient enrichment. Experimentally or empirically derived relationships between changes in nutrients and biological and/or chemical effects provide a solid basis for determining how nutrient enrichment affects estuarine and coastal systems. The more insidious effects (e.g., altered food webs, habitat suitability, shifts in ecosystem trophic structure, altered predator-prey interactions) are not as straightforward, and complex interactions obscure direct lines of evidence.

Background Information

The most comprehensive historical data on Louisiana's estuaries including those in the BTNEP were collected by the Louisiana Wild Life and Fisheries Commission in their estuarine inventories (Barrett et al. 1978). Significant sources of information concerning the Barataria basin are Seaton (1979), Conner and Day (1987), Madden et al. (1988), Childers and Day (1990 a, b), Witzig and Day (1983), and Roemer (1989). Less has been published concerning the Terrebonne basin with the exception of the Fourleague Bay area (Caffrey and Day 1986; Madden 1986, 1992; Randall and Day 1987; Madden et al. 1988; Teague et al. 1988; Childers and Day 1990 a, b; works in progress of J. Day, R. Shaw, L. Rouse, R. Twilley, M. Dagg, and B. McKee).

Barataria basin receives nutrient loadings on the same order of magnitude as Fourleague Bay influenced by the Atchafalaya River, but the source is different (Madden et al. 1988).

Barataria basin is bordered by agricultural and urban areas, and its chemistry is impacted by high-anthropogenic nutrient loading in runoff. Hopkinson and Day (1979) estimated mean total nitrogen and phosphorus concentrations entering the upper Barataria basin at 279 and 18 µM, respectively. The altered hydrography of the upper basin with its extensive channelization has been implicated in reducing the natural wetlands' ability to assimilate high influxes of nutrients (Gael and Hopkinson 1979). Nutrient concentrations are higher in the water column in channelized areas (Kemp and Day 1984), and receiving waters that drained the areas of highest canal density were the most eutrophic waters of the basin (Seaton and Day 1979).

The lower end of the Barataria basin is influenced by the Gulf of Mexico waters and subsequently the plume of the adjacent Mississippi River. Ho and Barrett (1975, 1977) measured pulses of fresh water carrying nitrate, phosphate, and silicate through the mouth of Barataria Bay during spring flooding. The water quality in Barataria Bay may be affected by changes in Mississippi River water quality because of its relatively large freshwater inflow and evidence that salinity in Barataria Bay is inversely related to river discharge (Wiseman and Swenson 1987, Wiseman et al. 1990).

There is a gradient along the axis of the Barataria estuary of nutrient concentrations, phytoplankton production, chlorophyll biomass, and turbidity. Witzig and Day (1983) developed a trophic state index to classify the water bodies in the Barataria basin with respect to level of eutrophication, based on quarterly collections of water quality data from 24 stations from August 1986 to August 1987. Water bodies in the upper to middle portion of the basin were the most eutrophic.

The northern portion of the Terrebonne/Timbalier drainage system is composed mostly of high land broken by bayous, canals, and swamps that is utilized for agriculture, pastureland and an ever increasing amount of urban development. Extensive marshes border the southern land extremities and intermingle gradually with an extensive estuarine system, namely Terrebonne and Timbalier bays and Lake Pelto. The sources of eutrophication in Terrebonne and Timbalier bays is similar to other basins. In the upper basin agricultural runoff is primarily from sugarcane fields. Industrial waste seems to be the source of extremely high phosphorus inputs, particularly in the Lake Verret region (Craig and Day 1977). Municipal sewage from the Houma area is also a source of high nutrient input. Several studies were conducted in the eastern portion of Timbalier Bay from August 1972 to January 1974 in conjunction with an analysis of the effects of oil drilling and production (Ward et al. 1979). These included hydrographic profiles, water column nutrients, and sediment nutrients along a transect from the upper bay to the intersection with the Gulf of Mexico. A year-long study (October 1982–October 1983) of water column characteristics was conducted along a Terrebonne Bay transect and into adjacent Gulf of Mexico waters (Dagg ND).

Seasonal variations in nutrient concentrations in the oil and gas production study (June 1972–March 1974) (Burchfield et al. 1979) were generally comparable to data collected by Barrett et al. (1978). Nitrogen levels peaked during the winter months, increased in July in response to heavy summer rainfall, decreased slightly in October, and then increased during the following winter. The highest phosphate levels were observed during the fall, then decreased and remained relatively constant throughout the remainder of the study. Silica concentrations were highest during October 1972 and 1973 and minimal during the winter months. Price (1979) suggested that there is less seasonal and spatial variability than is typical of many other bays along the central Gulf coast; however, despite the paucity of streams entering the bay and its relative isolation from the Mississippi River, it was responsive to the flooding of the Mississippi River in July 1973.

Flow of the Atchafalaya River strongly influences spatial and seasonal patterns of nutrients, suspended sediments, and primary production in Fourleague Bay. The estuary receives most of this sediment input and high loadings of nutrients during spring. During low flow in summer and fall, Gulf of Mexico waters dominate Fourleague Bay, and increased water clarity provided an opportunity when riverborne and regenerated nutrients can be exploited maximally by phytoplankton. Even during periods when river discharge is expected to dominate, climatic conditions can have a significant effect (Caffrey and Day 1986).

N:P ratios in Fourleague Bay waters average about 30:1, but upper bay ratios are often >100:1 (Madden et al. 1988), thus indicating a potentially large deficiency of phosphorus relative to nitrogen. The seasonal decline in nitrate input with reduced river discharge is followed by a reduction in the N:P ratio; although, it rarely falls to 16:1 in the upper bay. During summer and fall there is a major shift in the ratio of inorganic nitrogen to phosphorus in the lower bay. During the spring flood, nitrate concentration in upper Fourleague Bay averages 120 µM—in lower Fourleague Bay, 15 µM. During low flow, nitrate concentration in the upper bay averages 30 µM and in the lower bay is often <1 µM.

Nutrients and Chlorophyll *a*

Introduction

This section of the report is a review of the status and trends of nutrients and chlorophyll *a*, a measure of phytoplankton abundance, in the Barataria and Terrebonne estuarine system. Phytoplankton, by virtue of their great diversity, rapid growth rate (often only several days or less), and importance to the estuarine food chain are useful indicators of estuarine condition. If nutrients are excessively abundant and in appropriate balance with one another, then eutrophication may occur.

Phytoplankton require nitrogen and phosphorus for growth in approximately a proportion of 16 nitrogen atoms for every 1 phosphorus atom (the Redfield ratio). In general, where the N:P atomic ratio is less than 10 or greater than 20, nitrogen or phosphorus respectively, the supply may be insufficient to allow for optimal growth rates. Nitrogen is commonly thought to limit phytoplankton growth in coastal and oceanic waters (e.g., Harris 1986, Valiela 1984). However, not all coastal systems are nitrogen limited (e.g., the Huanghe in China is phosphorus limited, Turner et al. 1990), nor is changing nutrient loading the only factor influencing phytoplankton growth (Skreslet 1986). Marine phytoplankton also may respond differently to nutrient additions if introduced gradually or suddenly with changing flushing rates or salinity, and with cell density (Sakshaug et al. 1983, Sommer 1985, Suttle and Harrison 1986, Turpin and Harrison 1980). The abundance of coastal diatoms is also influenced by the silicon supplies, whose Si:N atomic ratio is about 1:1 (the Redfield ratio). Alterations in silica supply and the ratio of Si to the other limiting nutrients may lead to species shifts and increased incidence of noxious and toxic phytoplankton, as described above.

Data Sources and Variability

Data sources for this report are from university, state and federal agencies that span the period from the late 1960s to 1994 for the Barataria and Terrebonne estuaries. Figure 3 is a map for all important locations mentioned in the text. Not all of these data are reliable or useful (see below). The major data types sought are data on nutrient concentration (nitrate, nitrite, ammonia, silicate, and phosphate) and indicators of phytoplankton abundance (primarily chlorophyll *a* but also oxygen saturation and secchi disk depth). There are two studies that sampled the Barataria estuary from the freshwater end member to the opening at the Gulf of Mexico. One was by Seaton (1979, quarterly sampling, 72 stations per year), and the other is an ongoing LSU project (R. Turner and N. Rabalais, monthly sampling, 444 stations per year). No comparable data exist for all sub-basins of Terrebonne Bay, and it would be difficult to attain given the various north-to-south levees dissecting the watershed. Water quality data for Fourleague Bay are mostly from LSU researchers (e.g., Caffrey and Day 1986, Randall 1986, Randall and Day 1987, Teague et al. 1988, Madden et al. 1988, Madden 1992). The Barataria Bay estuary has a distinct connection to the nearshore waters, whereas the Terrebonne Bay estuary has many. The circulation patterns in the Terrebonne Bay estuary are less complicated than in the Barataria Bay estuary, which is therefore less likely to be sampled effectively with the same amount of funding or with the same scientific return on sampling effort.

In addition to the 1994 survey of Barataria Bay, there are three additional studies of multiple stations sampled for at least one year at quasimonthly intervals: two coastwide studies by the Louisiana Wild Life and Fisheries Commission for the periods 1967–69 and 1974–1976 and a 12-month survey of Terrebonne Bay for 1982–1983.

Additional data are from point sampling from water quality monitoring stations maintained by USGS and LDEQ. Most of these sampling stations are located at accessible points where a vehicle can be driven and thus do not represent the open water bodies normally considered an estuary. This means only some representative parts of the estuarine environment are sampled. Further, some of these sampling stations were discontinued within the last decade and are of limited use. We narrowed the trends analysis at water quality monitoring stations to records greater than five years of

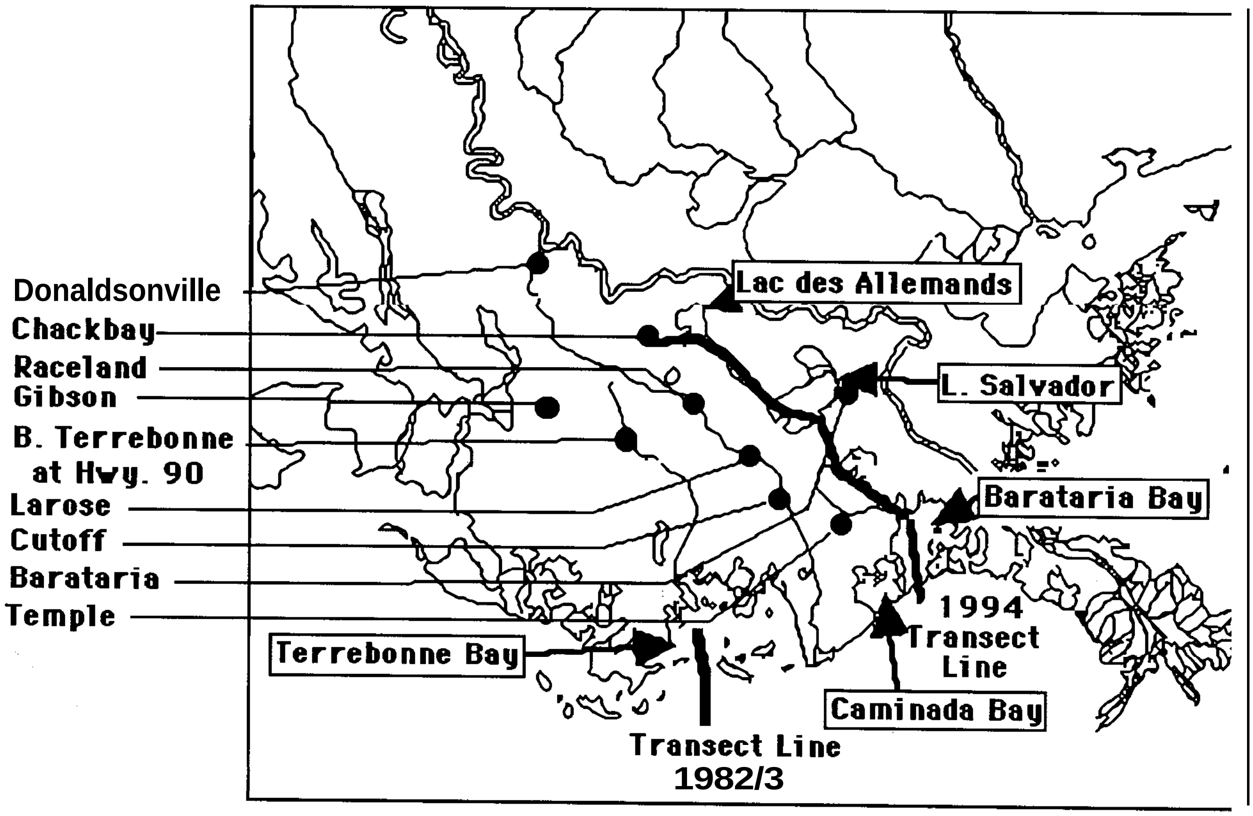


Figure 3. Location map for the sites discussed in the text. The Mississippi River levee forms the eastern boundary. A former tributary of the Mississippi River, the Lafourche delta distributary, divides the Terrebonne and Barataria bay estuaries. The Atchafalaya River levee forms the western boundary. The major entrance to the Barataria Bay is between the barrier islands of Grand Isle and Grand Terre (50 m depth). Several tidal passes form the southern boundary of Terrebonne Bay.

continuous data.

Not all of these data are reliable. Chlorophyll *a* data may represent active or inactive pigment forms. The methodology is often not indicated. Some of the water quality analyses are known to be suspect either because of delayed analyses, improper storage, or lack of adequate chemical standards. However, the data entry control appears to be excellent. No clear examples of mis-entered data were found in over 50 water quality stations and other data sets, representing more than 10^5 data points.

The sampling months and years vary among the identified data sources, and the spatial coverage is not consistent, as might be expected. The sampling density is an important consideration for two reasons. First, considerable variability occurs within the estuary along the salinity gradients and across the width of the estuary. Salinity samples taken different distances away from the axis running between the major freshwater and in a north-to-south direction) are not similar. Samples taken near each other on that salinity axis show major rises and decreases in concentration (see examples below). Constituents in water quality samples taken from bayous entering Lac des Allemands show great variability along the stream length and are usually in much higher concentrations than in Lac des Allemands. Second, the natural changes in hydrologic forces (e.g., tides, rainfall, fronts, and storms) provide additional variability in important mixing and dilution functions affecting water column and wetland habitats. Phytoplankton abundance (indicated by chlorophyll *a*) and nutrient concentrations (which phytoplankton take up) are influenced by these changes in mixing factors.

Temporal variability in the data arises from atmospheric, freshwater, and offshore sources. Storm fronts pass through about 50 times annually and every 4–10 days between October and April. These fronts drain and fill the estuary by changing water levels two to five times the mean tidal range. Water levels are at a seasonal low in winter and high in summer, implying greater tidal exchange during the summer when the freshwater inflow from the northern portion of the watershed is at a seasonal low. This natural variability in the physical environment exists with seasonal variability in the biological environment. Consequences of the historically sparse sampling scheme and this natural variability are several. The major concern, however, is that the *signal,* if any exists, may be hidden among the noise of this variability.

Nutrients

Mississippi River

The nutrient content of the Mississippi River is changing mostly from land-use practices occurring far from the estuary. The Mississippi River watershed, like other developed rivers, has undergone cultural eutrophication and has had consequential impacts on the continental shelf ecosystem (Turner and Rabalais 1991a, 1994). Since the 1950s, the riverine nitrogen and phosphorus loadings to offshore waters have doubled (figure 4). Nitrate concentrations appear to have stabilized, but trends are masked by increased variability in the 1980s data. Silicate concentrations declined by about 50% (figure 4), while phosphorus loadings have doubled (figure 5). The average annual nitrate concentration in the river is positively related to nitrogen fertilizer use. Only about 20% of that applied is required to leave the site of application and enter the aquatic system to account for the observed changes in the river. However, and in contrast to the changes to nitrogen, the average annual silica concentration is *inversely* related to phosphorus fertilizer use. Presumably this result is because phosphorus stimulates diatom growth and diatom tests sink, thus storing dissolved silica in sediment that would otherwise move downstream (Schelske et al. 1983, 1988). The seasonal variability of nutrient concentrations also has shifted. The present conspicuous spring peak in nitrate

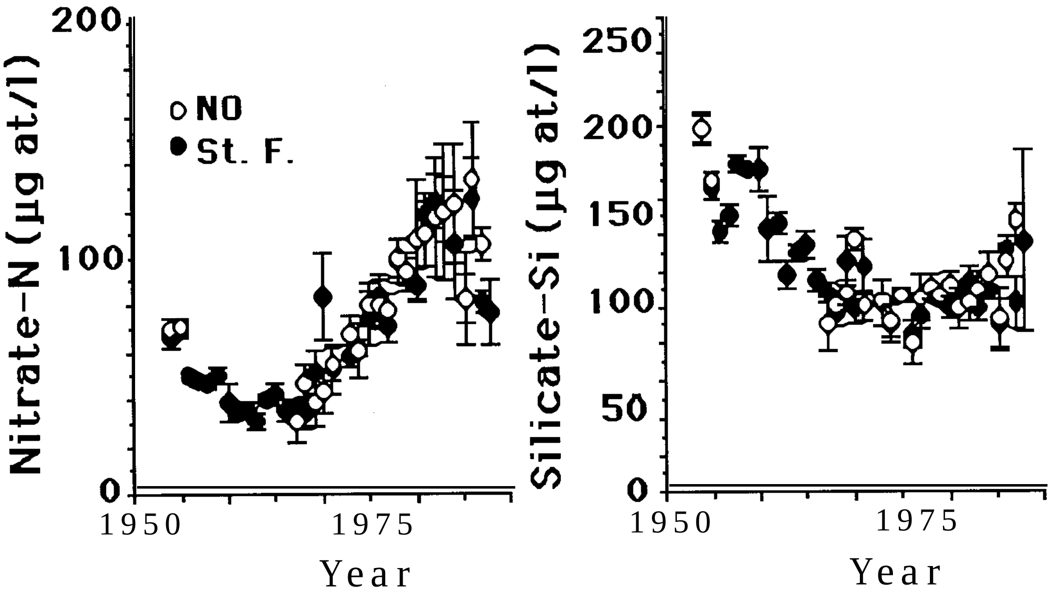


Figure 4. The annual average concentrations of silicate and nitrate in the Mississippi River at St. Francisville, Louisiana (St. F.) and at New Orleans, Louisiana (N.O.) (adapted from Turner and Rabalais 1991a).

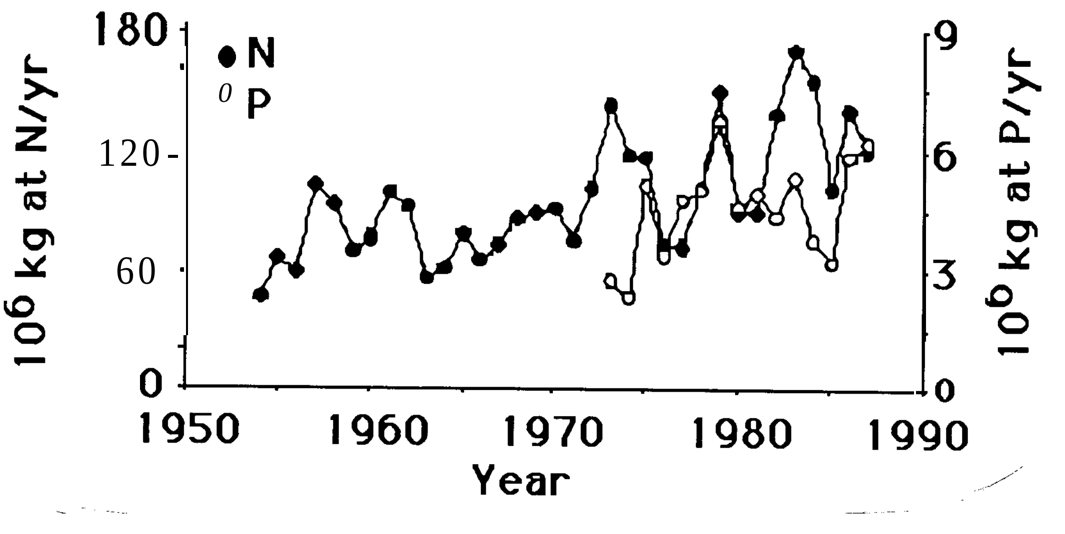


Figure 5. Nitrogen and phosphorus loading from the Mississippi River to the coastal zone (from Turner and Rabalais 1991a).

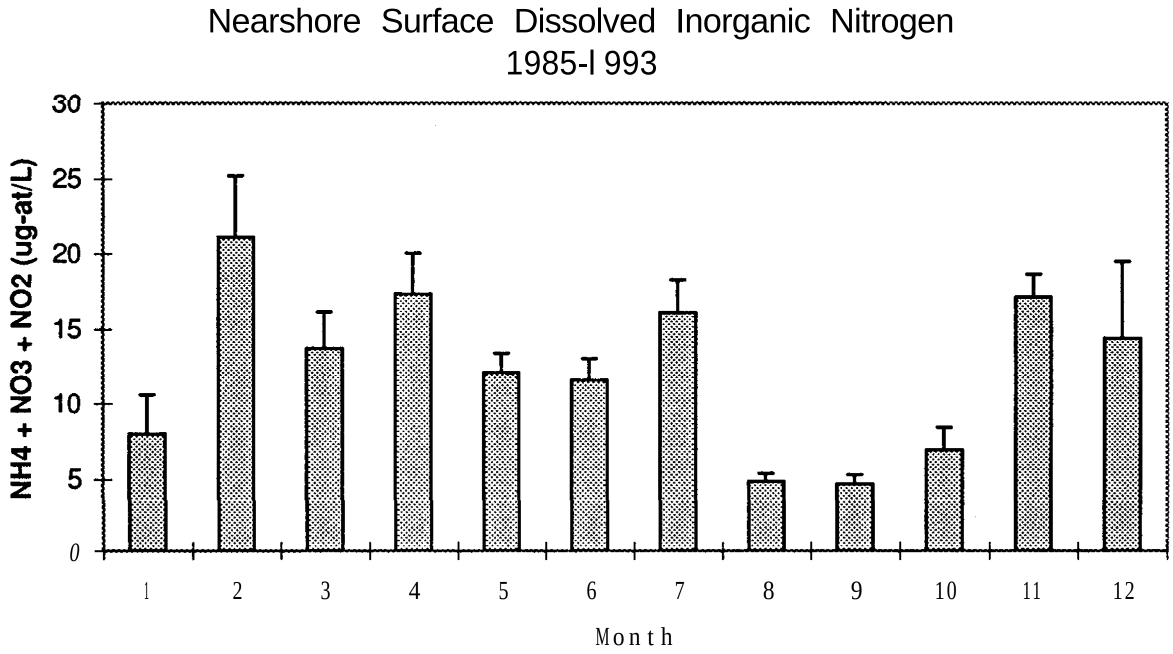
concentration in the river was not evident before the 1950s, and the silicate peak occurring at the turn of the century is now greatly reduced. Hypoxia (# 2 mg l^{-1}) in bottom waters on the continental shelf is widespread during the summer (>9,500 km^2) (Rabalais et al. 1991a) and probably larger, more severe, and longlasting as a result of higher nutrient loadings.

The dissolved N:P:Si ratios of the Mississippi River Delta thus changed along the freshwater-to-seawater gradient during the last two decades. Although the silicate concentration in the river declined, *net* Si uptake along the estuarine mixing zone appears to be the same or higher in the 1990s compared to that found in the 1960s (Turner and Rabalais 1995). The effects of these changes on the continental shelf have not been fully explored but are under continuing investigation. An analysis of the diatom, foraminifera, and carbon accumulation sedimentary records supports the inference of increased eutrophication and hypoxia in the Mississippi River delta bight primarily because of changes in nitrogen loadings from agricultural sources (Turner and Rabalais 1994, Rabalais et al. in press).

Indications that these riverine water quality changes (arising through mixing at the tidal passes) may be seen in an estuary downstream from the river mouth are based on knowledge of the mixing regime and on strong inference. The net productivity of nearshore coastal waters lags one month behind the long-term average peak of Mississippi River flow (Justic' et al., 1993). Long-term seasonal peaks of dissolved inorganic nitrogen (DIN) levels and chlorophyll *a* biomass in offshore coastal waters parallel the flow of the Mississippi River and its freshwater and nutrient content (figure 6). The salinity of estuaries near the Mississippi River is strongly influenced by the offshore freshwater content (figure 7) to suggest that significant amounts of constituents other than salts (e.g., plankton, nutrients, and sediment) are transported into and out of the Barataria and Terrebonne estuaries during tidal exchanges.

Barataria Bay from Lac des Allemands to Grand Isle

Nutrient concentrations and ratios change along a north-to-south gradient within the Barataria basin. Nutrient concentrations may exhibit the lowest concentration seaward (figure 8)**.** Because concentrations tend to decrease as they enter open-water areas and then rise as water continues seaward, fast regeneration of nutrients is indicated. The relationship of nutrient concentrations and salinity show additional patterns (figure 9). The concentration changes are quite large in the freshwater section of the estuary, particularly for N:P atomic ratios. As the freshwater nutrient sources are initially diluted with the seawater endmember, phosphorus is taken up faster than nitrogen, and the N:P atomic ratio rises. This indicates P limitation of growth. As water moves down the estuary seaward, the N:P ratio declines and indicates increasing N limitation. In the open waters of Barataria Bay, the N:P ratio begins to increase, indicating additional P limitation in the nearshore coastal waters entering the bay. Both N and P limitation of phytoplankton growth have been observed in the lower portion of the Terrebonne and



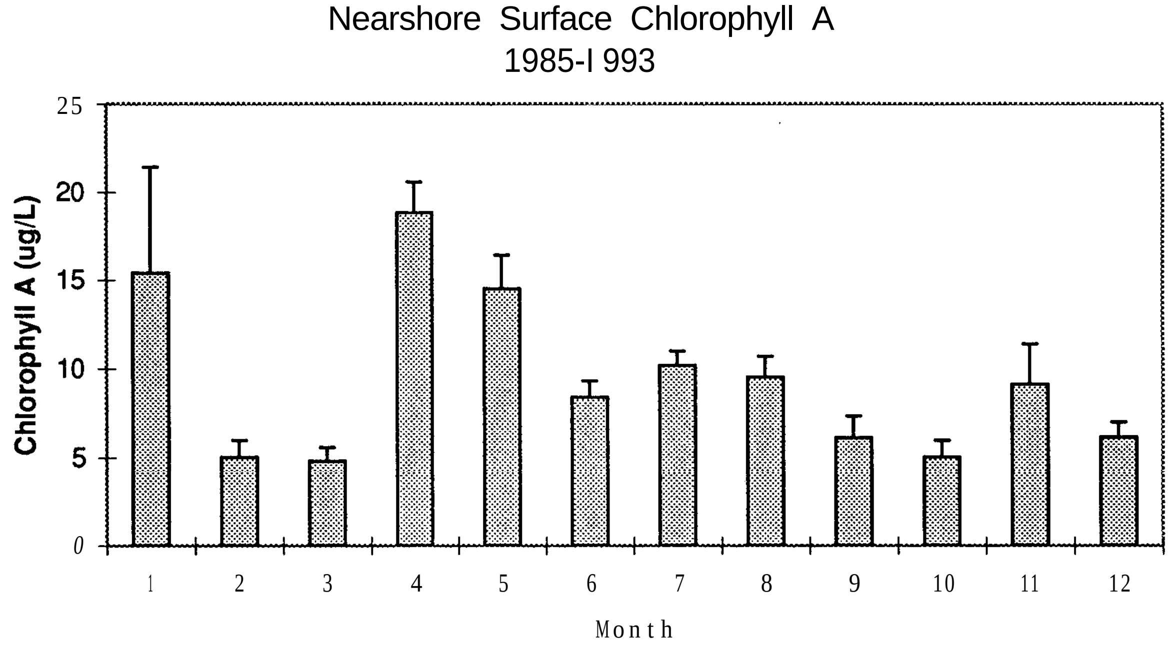


Figure 6. Long-term averages (19851993) of surface nearshore coastal waters for dissolved inorganic nitrogen (NO_3+NO_2+NH_4) and chlorophyll ***a*** biomass. Stations indicated in figure 2; means are heavily influenced by the western end of the study area and the summer months.

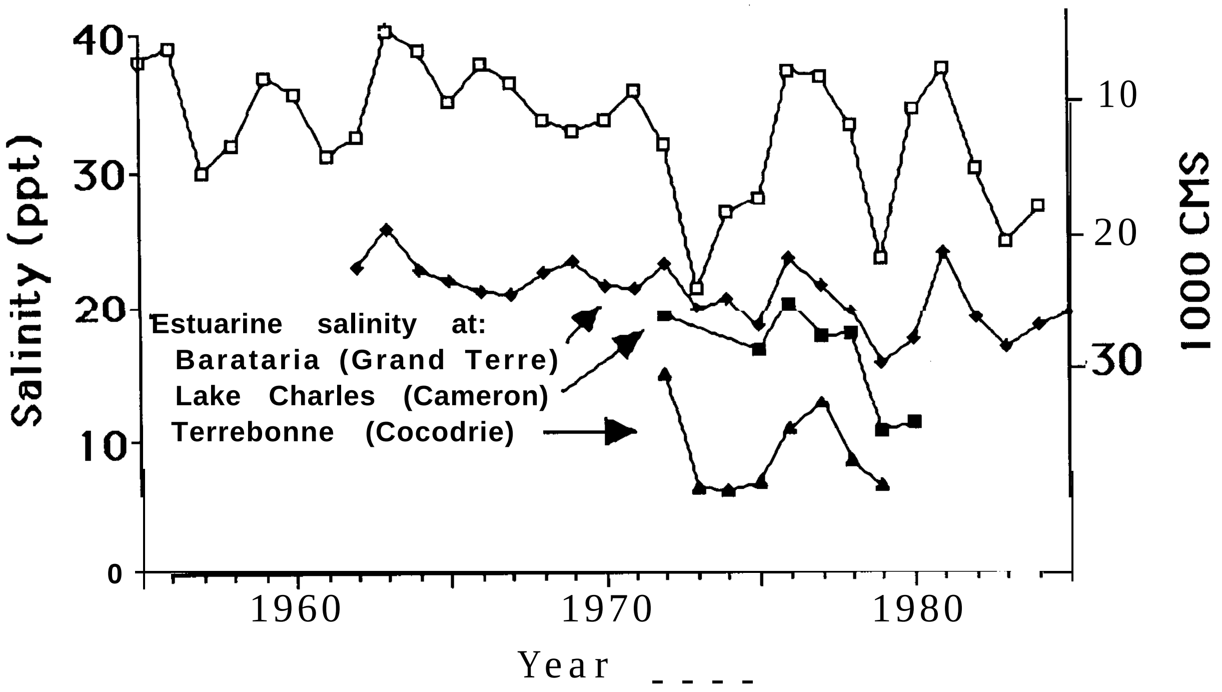


Figure 7. Time series plots of the combined annual mean flow of the Mississippi and Atchafalaya Rivers and plots of mean annual salinity from selected Louisiana Wildlife and Fisheries sampling stations. River flow is in thousands of cubic meters per second (CMS) with a 15,000 CMS offset and flipped vertically. Adapted from Wiseman et al. (1990).

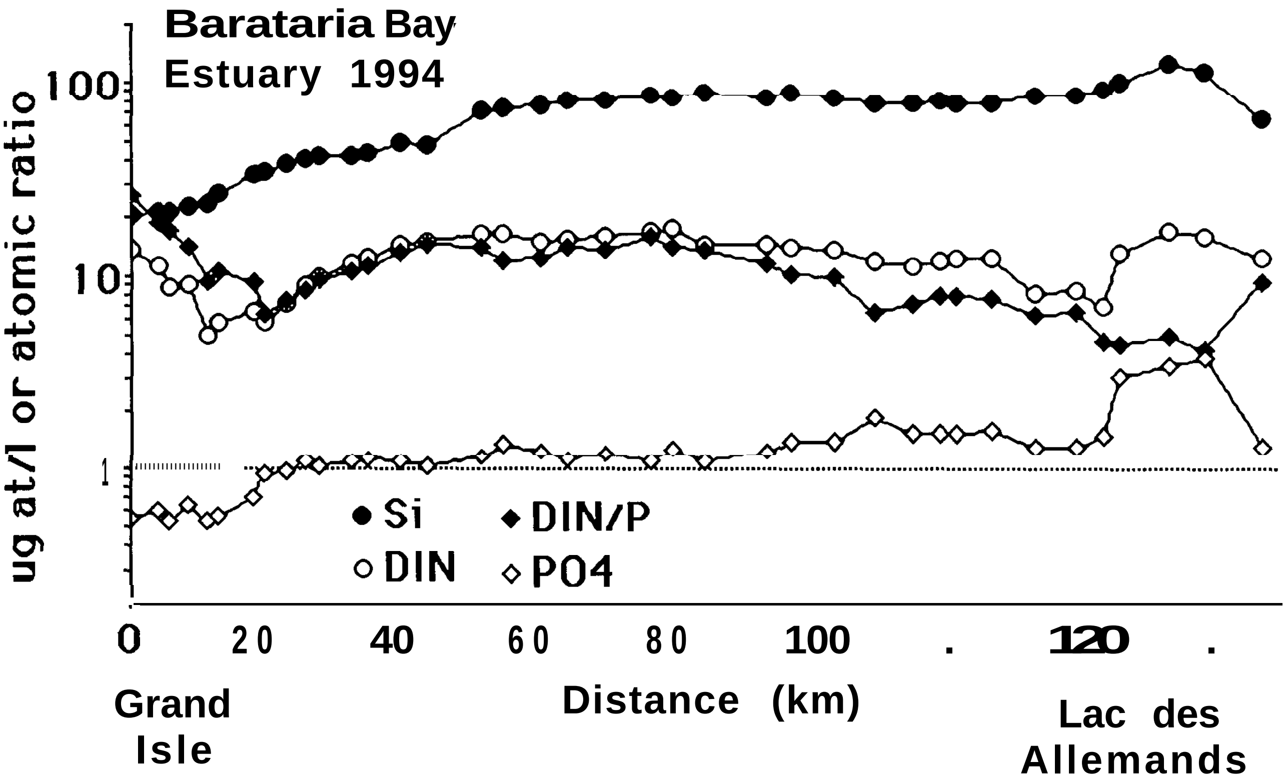


Figure 8. The average annual nutrient and nutrient ratios along a north-to-south transect in Barataria Bay for 1994.

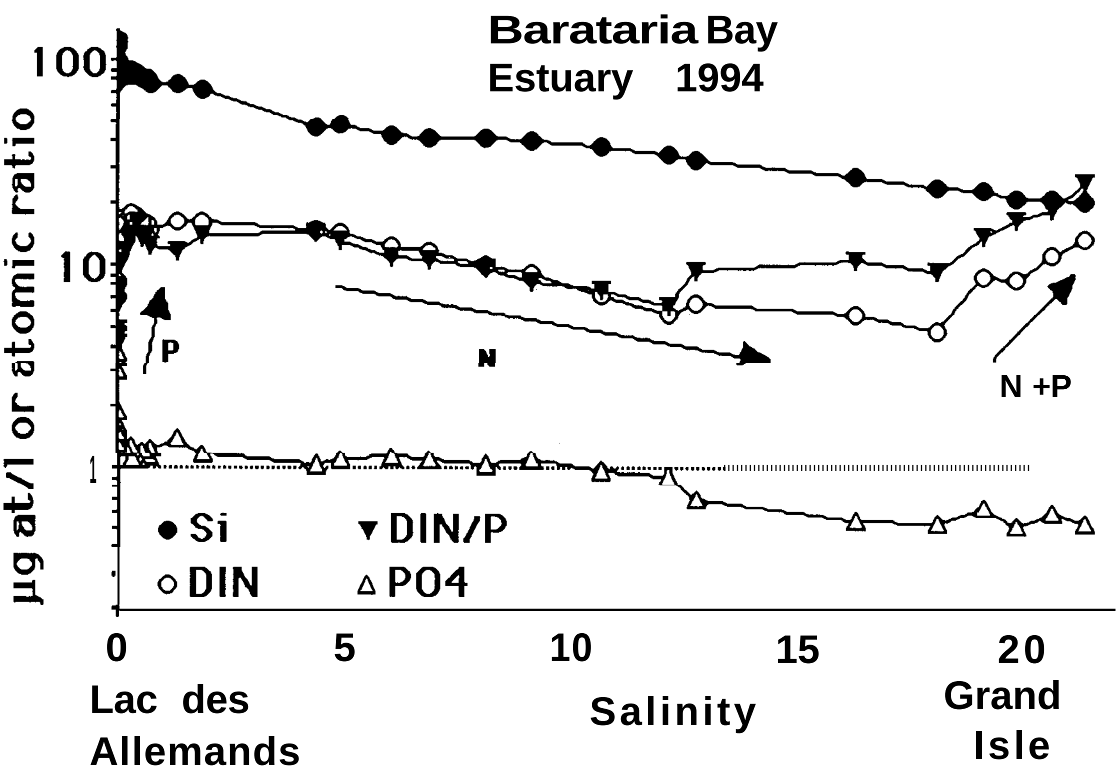


Figure 9. The variations in the average (monthly averages) nutrient and nutrient ratios versus salinity along a north-to-south transect in Barataria Bay for 1994. The arrows and element indicate the nutrient most likely to limit phytoplankton growth.

Barataria estuaries (see section on Algal Growth Limitations), and nitrogen sources exist for uptake from offshore and freshwater sources. However, the direct relationship between chlorophyll *a* concentration and nitrate uptake suggests a dominant nitrogen limitation of phytoplankton growth at higher salinities. The loading from upland sources has changed the past few decades not only because of a general population increase but also from agricultural sources (figure 10). There are no other significant terrestrial sources of nutrients to the estuary. Mississippi River inputs to Bayou Lafourche are mostly confined by natural and human-made levees. Atmospheric inputs are likely substantial but remain undocumented.

It was not possible to determine the long-term trends in nutrient concentrations in these bays because either the signal:noise ratio was too high, or there were no trends. However, nitrogen concentrations have been measured in the tributaries entering Lac des Allemands in the northern Barataria Bay watershed. An analysis of these records shows a decline in nitrate+nitrite concentrations and in the concentration of Total Kjeldahl nitrogen (figures 11 and 12) and a slight increase in Total P (figure 13), but none of these trends are statistically significant (table 3). Light penetration improved in the same interval, but there was no significant change in the concentration of total carbon. Stations at Grande Bayou near Highway 20, Bayou Barataria at Barataria, Louisiana, and Little Lake at Temple, Louisiana, also showed few changes in nutrient concentrations (table 3).

Bayou Lafourche

Bayou Lafourche is a boundary between the Barataria and Terrebonne estuaries. Its narrow former distributary levees are densely populated, and the center channel is used for navigation, drinking water supplies, and waste disposal. The concentration of Total P and nitrate+nitrite declines as the diverted Mississippi River water moves from Donaldsonville southward towards Cutoff, whereas the concentration of Total Kjeldahl nitrogen increases in the same direction (figure 14). Apparent changes were a slight decline in nutrient concentrations over time and a slight improvement in light penetration.

Bayou Terrebonne

Bayou Terrebonne, located near Houma, is a thin urban island amidst rural swampy landscape. Statistically the water quality trends are virtually non-existent (table 3). Although the oxygen saturation appears to have had lower values in the 1960s and 1970s (figure 15), there was no statistically significant trend in hypoxic conditions (< 2 mg l^{-1}). Also there were no significant trends in the nutrient concentrations. The absence of oxygen will normally be accompanied by an increase in nutrients that are released from the anoxic sediment or from decomposition in situ. Thus changes in nutrient concentrations might increase under eutrophic conditions, which adds to the variability in the record.

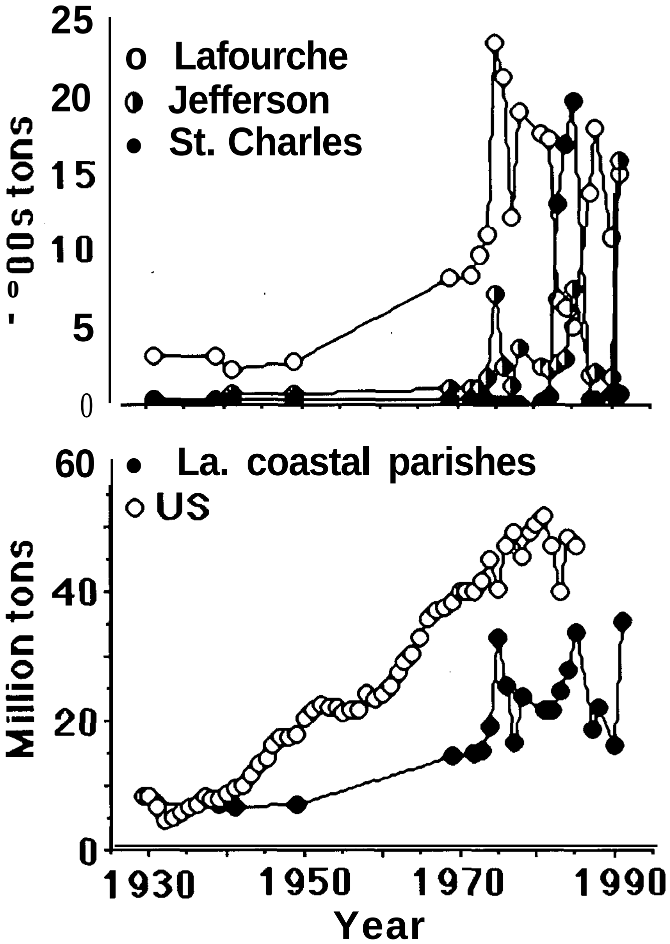


Figure 10. The annual variability in fertilizer consumption for the three major parishes of the Barataria Bay estuary (upper panel) and U.S. and La. coastal parishes (lower panel). Note the lack of synchrony among parishes.

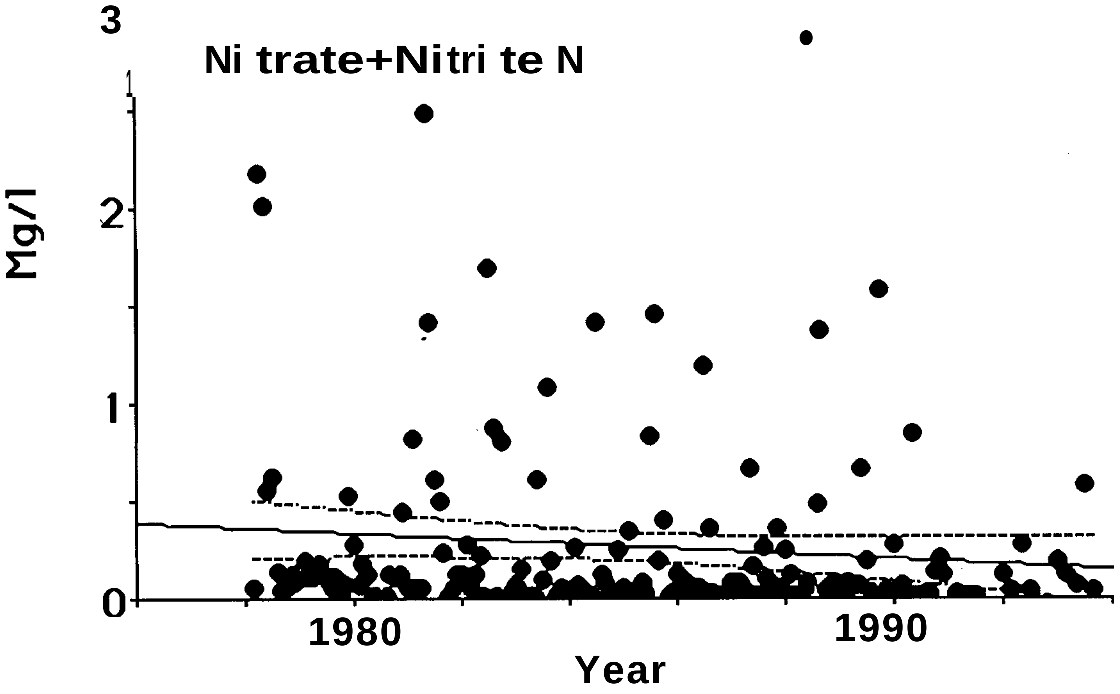


Figure 11. The long-term water quality records at Bayou Chevreuil at Chackbay, La. for nitrate and nitrite. A linear fit with 95% confidence boundaries for the slope is shown.

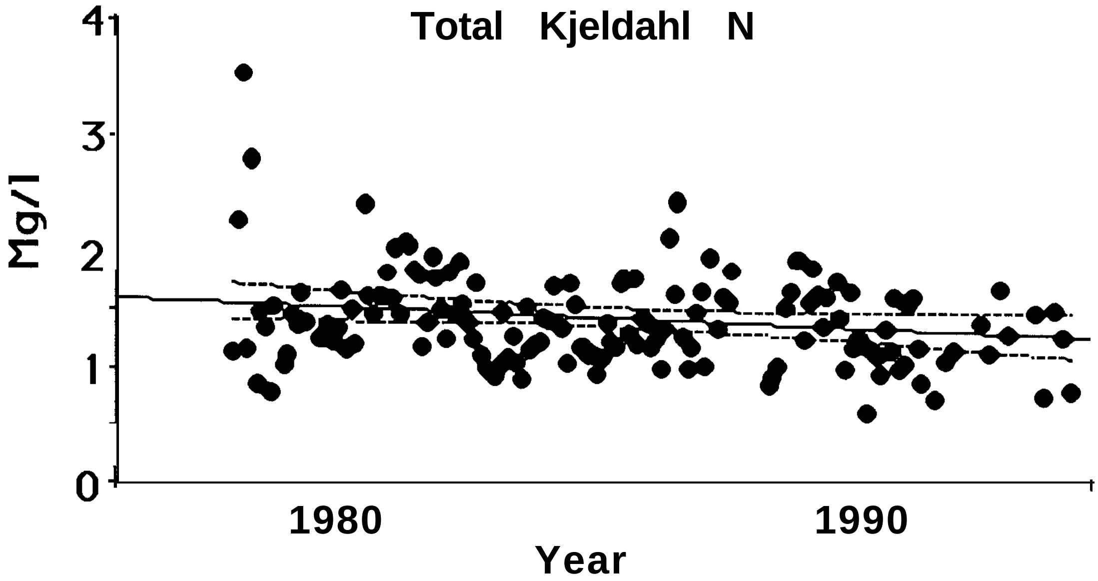


Figure 12. The long-term water quality records at Bayou Chevreuil at Chackbay, La. for total Kjeldahl nitrogen. A linear tit with 95% confidence boundaries for the slope is shown.

Table 3. Water quality–monitoring stations with significant changes in light penetration, oxygen saturation and nutrients for records > 5 years. The values are for the level of significance where $p \leq 0.05$. A "+" or "-" in parentheses indicates the slope of the linear regression.

Station	Years	SD	%DO	KJN	NO3+2	TP	TC
Bayou Chevreuil at Chackbay = "Chegby" in USGS data	1978–93	0.013 (+)	0.05 (+)	ns	ns	ns	ns
Grand Bayou near Chackbay = "Chegby" in USGS data	1978–91	ns	0.002 (+)	0.0001 (-)	0.0001 (-)	0.003 (-)	ns
Grand Bayou near Highway 20	1958–78	nd	0.0008 (-)	nd	nd	nd	nd
Bayou Barataria at Barataria	1973–81	nd	0.001 (-)	nd	nd	nd	nd
Little Lake at Temple, LA	1980–93	ns	ns	0.0002 (-)	ns	0.0002 (-)	ns
Bayou Lafourche at:							
Donaldsonville	1978–91	0.002 (+)	ns	0.001 (-)	ns	ns	ns
Raceland	1978–93	0.0012 (+)	ns	ns	0.08 (-)	ns	ns
Larose	1978–91	0.0002 (+)	0.001 (-)	ns	0.0002 (-)	nd	0.003 (+)
Cutoff	1978–91	ns	0.002 (+)	0.0001 (-)	0.003 (-)	0.003 (-)	ns
Lower end near Gulf of Mexico	1973–81	nd	0.0001 (-)	nd	nd	nd	nd

Table 3. Continued.

Bayou Terrebonne at Hwy.90	**1958–93**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**
Bayou Black near Gibson	**1958–91**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**
Bayou Boeuf at Bayou Chene	**1973–81**	**nd**	**ns**	**nd**	**nd**	**nd**	**nd**
Caminada Pass SE Grand Isle	**1973–81**	**nd**	**0.0001 (-)**	**nd**	**nd**	**nd**	**nd**
Caminada Bay at Bay Lizette	**1973–81**	**nd**	**0.0001 (-)**	**nd**	**nd**	**nd**	**nd**
Summary:							
number of sites with data		**9**	**15**	**9**	**9**	**8**	**9**
number higher		**4**	**3**	**0**	**0**	**0**	**1**
number lower		**0**	**6**	**4**	**4**	**3**	**0**
no change		**5**	**6**	**5**	**5**	**5**	**8**

Codes:
SD=secchi disk depth
%DO=dissolved oxygen saturation
KJN=Kjeldahl nitrogen
NO3+2=dissolved nitrate and nitrite
TP=total phosphorus
TC=total carbon
nd=no data
ns=not significant at $p \leq 0.05$

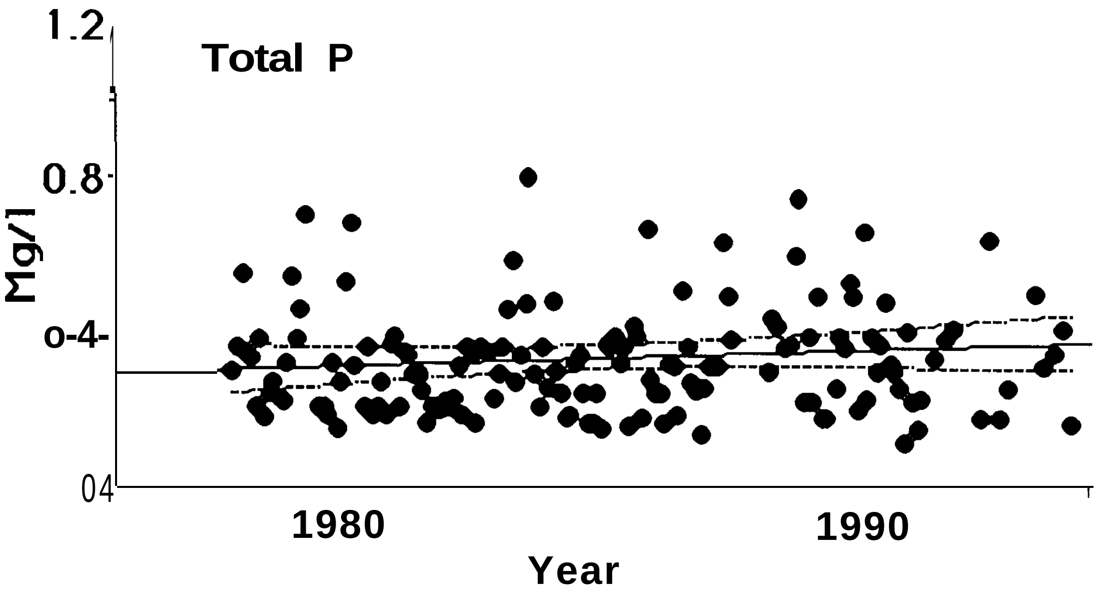


Figure 13. The long-term water quality records at Bayou Chevreuil at Chackbay, LA for phosphate. A linear fit with 95% confidence boundaries for the slope is shown.

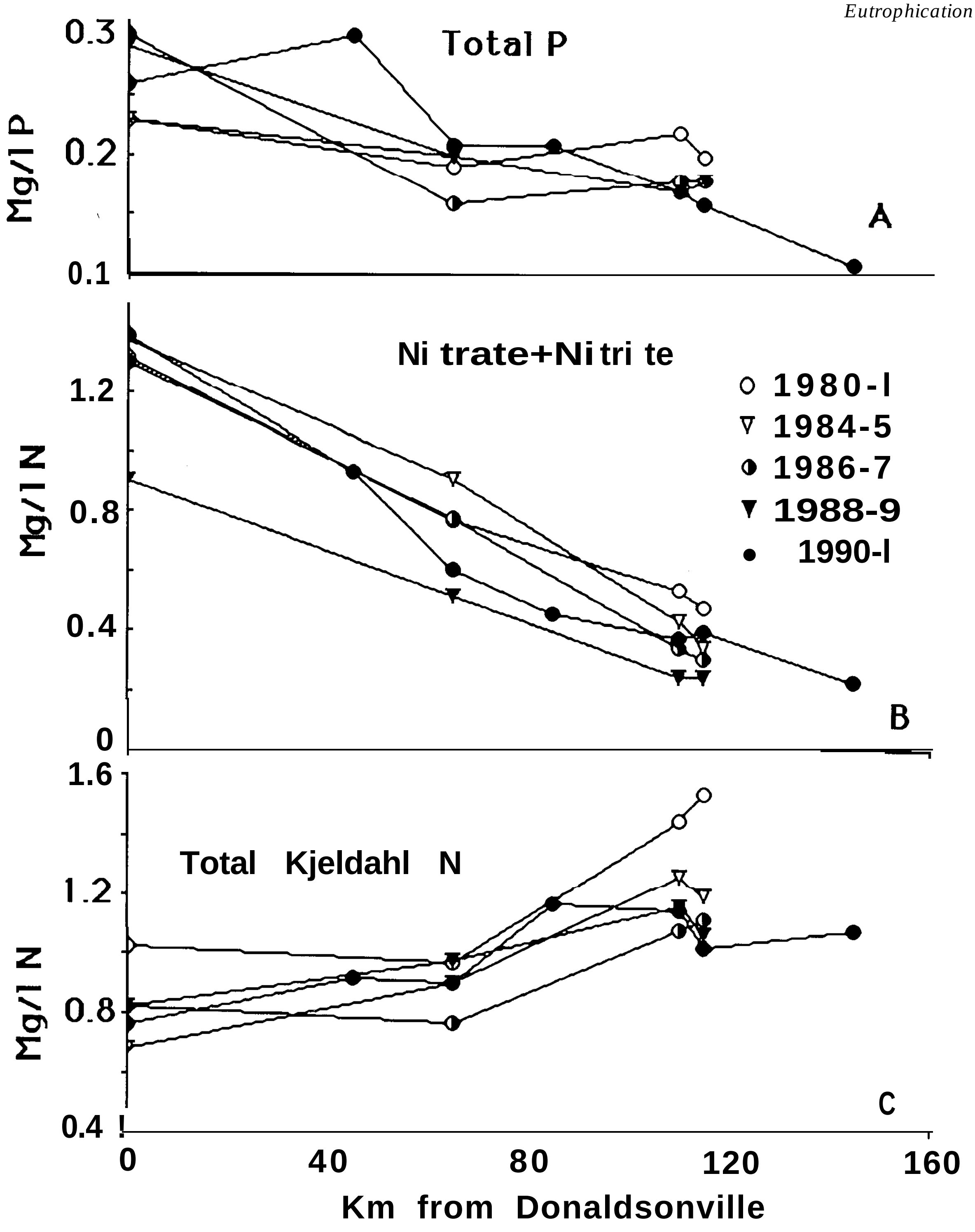


Figure 14. The concentration of nutrients in Bayou Lafourche from north to south for different time intervals. A two-year average for five time periods is shown. A. total phosphorus. B. nitrate+nitrite. C. total Kjeldahl nitrogen.

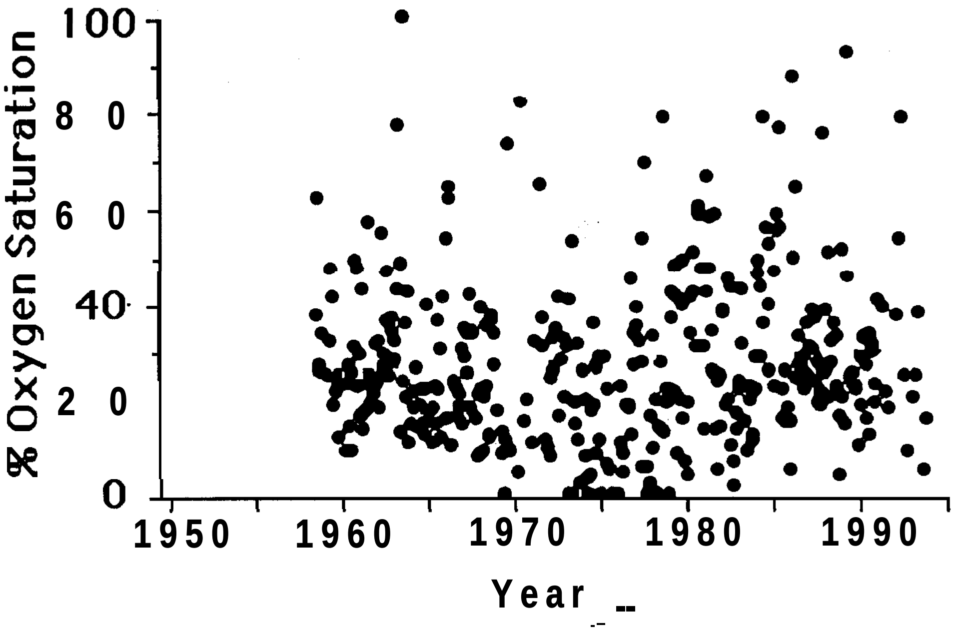


Figure 15. Oxygen saturation for Bayou Terrebonne at Highway 90.

Western Terrebonne

There were no comparable data (i.e., continuous record > 5 years, circa 1979–1991) in the USGS and LDEQ data bases for this portion of the study area.

Atchafalaya River

Approximately one-third of the Mississippi River discharge joins that of the Red River to form the Atchafalaya River. An analysis of long-term nutrient changes in the Atchafalaya River was conducted for data at Simmesport and Morgan City, Louisiana (Turner and Rabalais 1991a). The concentrations of nitrate, silicate, and total phosphorus at Morgan City were 69, 94, and 130%, respectively, of that in the Mississippi River at St. Francisville for the combined data for 1973–1987. Differences between nutrient concentrations in the Atchafalaya and Mississippi rivers are presumably a consequence of the Red River (which mixes with the Mississippi to form the Atchafalaya) compared with that in the Mississippi.

Summary of Changes and Status

There are 15 stations identified with long-term records of nutrient concentrations for the study region (table 3). Long-term changes were few, and inconsistent patterns occurred at one site. Nutrient concentrations could change for a variety of reasons, including changes in loading, removal during eutrophication, release during decay and as a consequence of eutrophication, and lack of uptake because one limiting nutrient was missing. In general, the monitoring station records are sparsely located across the study area and confined to easily sampled locations—not in open waters where the maximum uptake occurs. [Other indicators of changes in status of eutrophication are evident in chlorophyll levels (p.45) and indicators of diatom productivity in dated sediment cores (p. 49).]

Nutrient concentrations have been used as diagnostic parameters of estuarine condition. A NOAA/EPA study (NOAA/EPA 1990) classified estuaries into low, medium, and high nutrient conditions based on the following concentrations:

	low (mg l^{-1})	medium (mg l^{-1})	high (mg l^{-1})
Total Kjeldahl Nitrogen (sum of ammonia and organic N = TKN)	< 0.1	0.1–1.0	> 1.0
Total Phosphorus (sum of inorganic and organic P = TP)	< 0.01	0.01–0.1	> 0.1

Nitrate concentrations for saline waters were grouped into the following four conditions for the Chesapeake Bay (Fedkiw 1991):

	Nitrate N (mg l^{-1})
Healthy	0.6
Fair	0.6–1.0
Fair to Poor	1.0–1.8
Poor	> 1.8

All of the monitoring stations included in this review would be classified as either medium or high nutrient conditions under the NOAA classification scheme. All have samples that are sometimes in the high category during part of the year. Most of the monitoring stations in non-freshwater sites examined would be classified as healthy to fair using the Chesapeake Bay criteria.

Atmospheric Influences

Natural and industrially produced nitrogen is recycled through the atmosphere and into water supplies. The worldwide rise in nitrogen concentration in rainfall is symptomatic of the extent of the human impacts on the environment. The importance of atmospheric inputs of essential nutrients has only recently been understood but is clearly important and changing. For example, the inputs from precipitation and river runoff are equal in Chesapeake Bay (Correll and Ford 1982), an estuarine system often described as eutrophic. Nitrogen concentration in precipitation has recently increased in the eastern United States and in Europe (e.g., Likens and Borman 1979, Brimblecombe and Pitman 1980). It is therefore likely that most coastal landscapes have had significantly increased nitrogen loading this century. In addition, there is a change this century from viewing eutrophication as a local and perhaps point-source problem manageable on a regional scale to a phenomenon that is the cumulative result of many small actions throughout the world and whose scale of management is vastly expanded (and more expensive).

Comparison of Nitrogen Loadings

A preliminary nitrogen budget of the Barataria Bay watershed shows that annual loadings to the estuary from atmospheric sources are approximately one-half of loadings from agricultural applications. In contrast, the nitrogen movement during tidal exchanges at the passes is higher by more than 1,000 times. Transfer efficiency (elemental uptake) of these nutrient loading is unknown. Despite the apparent availability of these seaward sources, a retrospective analysis of sediment cores from salt marshes of the Barataria, Terrebonne, and St. Bernard estuaries indicates that variability in diatom accumulations (measured as biogenic silica, BSi) is coincidental with changes in fertilizer loadings within the basin (see Sedimentary Record). The offshore signal in these cores appears masked by the onshore signal. However, these cores were taken from that portion of the estuary closest to fertilizer application sites. Samples collected closer to the tidal passes or nearer the eastern boundary where there are fewer agricultural fertilizer sources may reveal a pattern of BSi accumulation that indicates an offshore source of nutrients.

Estimates of Algal Growth Limitation

The nutrients identified as potentially limiting to phytoplankton growth and biomass accumulation are many and generally include four groups: (1) nitrogen, phosphorus, and silica, (2) trace metals, (3) vitamins, and (4) chelators controlling the availability of other elements or compounds. Approaches to determine the nutritional limits to phytoplankton growth and accumulation may be grouped into three categories: chemical assays of water (e.g., inorganic N:P ratios), bioassays, and physiological (cellular) assays. There are advantages and disadvantages to using each approach. Physiological assays may be time consuming and insufficient by themselves, whereas chemical determinations are relatively simple. Two notable problems with the chemical approach are that (1) many algae have such a high affinity for nitrogen and phosphorus that, if limiting, the amounts may be below detection by usual analytical techniques, and (2) the algae may modulate "both their internal nutrient quota and their maximum short term uptake rates in response to variations in external nutrient concentrations" (Morel 1987). In addition nutrient limitation may not be interpreted correctly if another nutrient is limiting algal growth or inhibition by the added nutrient occurs. Constructing mass balances or loading rates for coastal systems are extremely difficult in part because of the paucity of good data, the confounding effects of physical processes (e.g., flushing and tidal excursions), large size, or if estuarine boundaries are poorly defined.

Algal bioassays have been conducted within the Terrebonne and Barataria estuaries using the deletion and addition approach (Turner unpublished). Additional but less comprehensive information is available from analyses of the nutrient ratios. The southern portions of Terrebonne Bay and the nearby coastal waters appear to vary between a nitrogen and a nitrogen+phosphorous limitation (figure 16). Airplane Lake in southern Barataria Bay appears to be primarily nitrogen limited. The nutrient ratios from the transect running north to south in Barataria Bay indicated a change from phosphorous limitation in the freshwater headwaters to nitrogen in the south. For example, the N:P atomic ratio rises, on average, as water enters the freshwater Lac des Allemands, and then downstream towards Lake Salvador (figure 8). This pattern indicates that phosphorus is taken up preferentially over dissolved nitrogen. In the southern, more saline end within Barataria Bay, the N:P atomic ratio drops to indicate a significant preferential uptake of dissolved nitrogen over dissolved phosphorus. Further, the rise and fall in nitrogen is proportional to the fall and rise, respectively, in chlorophyll *a* (figure 17).

The apparent phosphorous limitation in fresh water may be widespread across the Louisiana coastal zone. Figure 18 shows the relationship between the annual average total

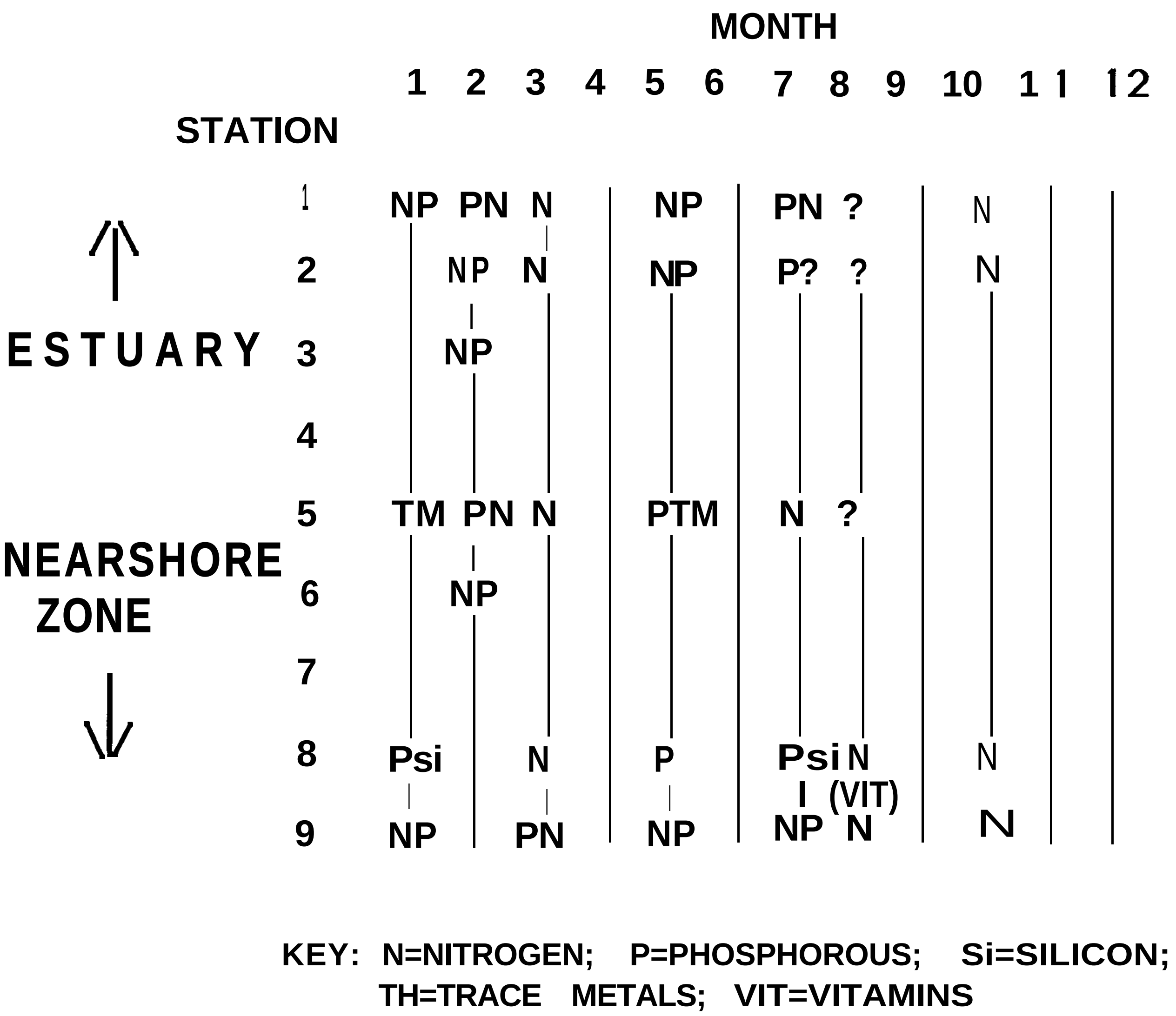


Figure 16. Nutrient limitation experiment results for lower Terrebonne. The methodological protocol is in Turner et al. (1990).

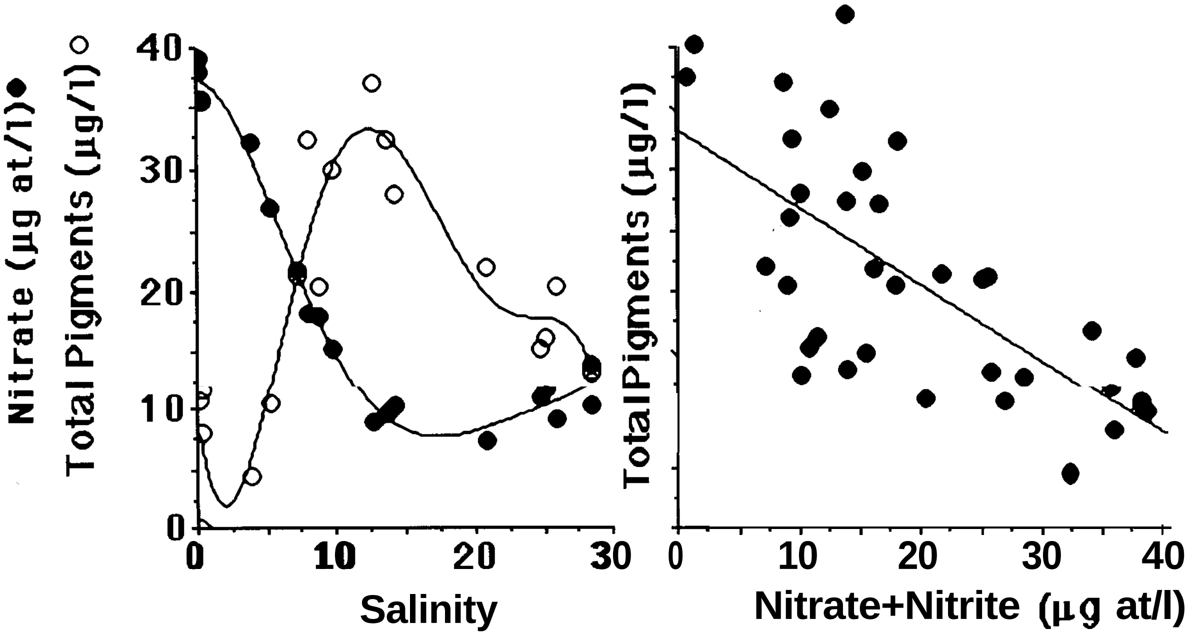


Figure 17. Left panel: Chlorophyll *a* pigment and nitrate concentrations versus salinity along a transect in the lower estuary of Barataria Bay (salinity range l-30 ppt) in January 1994. Right panel: The relationship between nitrate and pigment concentration from Barataria Bay entrance at Grand Isle, to the freshwater end member of Lac des Allemands in January 1994. A linear fit of the data is shown.

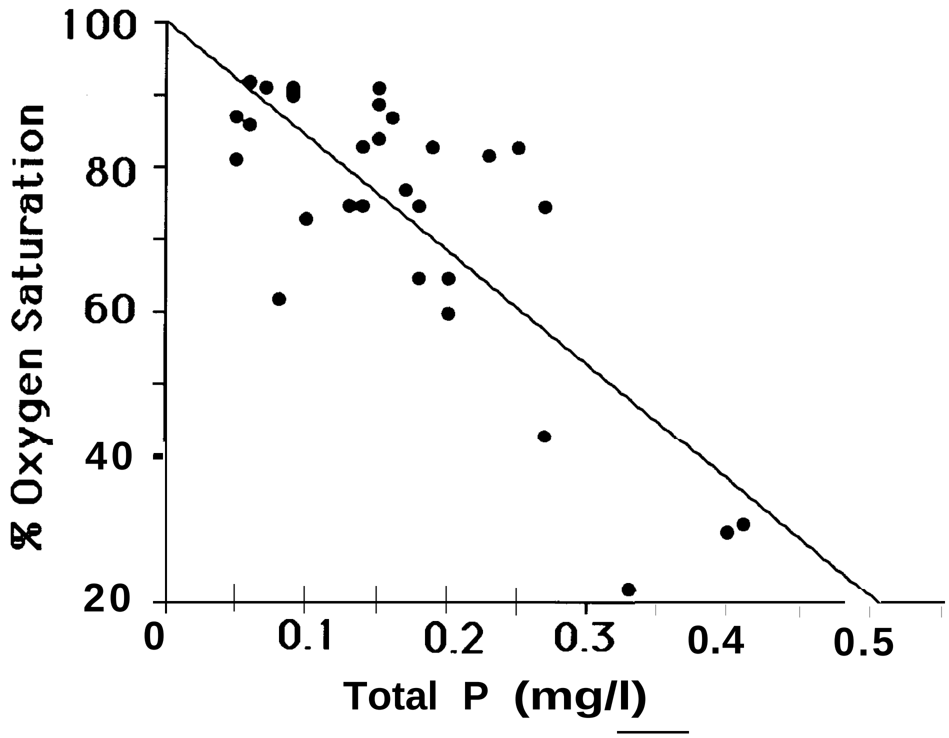


Figure 18. The relationship between the annual average oxygen saturation versus total phosphorus (TP) for 1988-1989 for coastal stations sampled by LDEQ. TP may be an indicator of sewage (possible) or the RQ is different. The stations are mostly from low salinity (<5 ppt) waters.

P concentration and the percentage oxygen saturation at all LDEQ stations with a salinity less than 1 ppt. An alternative interpretation is that the P concentration is proportional to sewage loading.

It should be noted, however, that other nutrients also may limit phytoplankton accumulation in the coastal zone. Silica is required for diatom growth (it forms part of the frustules). Silicate, in particular, occasionally was found to limit phytoplankton growth in a few algal growth bioassays. Also, dissolved silicate was effectively stripped from the water column in Lac des Allemands in winter (figure 19). Silicate is usually omitted from agency water quality monitoring, so there are very little data on its seasonal cycle.

Phytoplankton growth also may be limited by the available light. Although this limitation may be significant, phytoplankton production continues. For example, Fourleague Bay is one of the most turbid of the Barataria and Terrebonne sub-estuaries yet has a very high phytoplankton production rate (400 gC m^{-3} yr^{-1}; Madden 1992). This seemingly unlikely outcome is a result of the rapid mixing of phytoplankton in a well-mixed and shallow-water column. Until light becomes very limiting, phytoplankton growth continues—albeit at less than maximal rates.

Water turnover rate is also an important parameter in determining growth rates. If phytoplankton grow in a water body with sufficiently long residence times (assuming that zooplankton grazing is not important), then nutrients can still be depleted because of phytoplankton uptake. Fourleague Bay has very short residence time compared to Barataria Bay or Lac des Allemands (i.e., a few days to weeks, compared to many months). In Fourleague Bay the water residence times are very short, so only occasionally are nutrients completely depleted and the average standing stock of algal biomass is high but not extremely high. Even in this sub-estuary with high turbidity and short residence times, chlorophyll concentrations as high as 135 µg l^{-1} have been observed (Madden 1992) to indicate that phytoplankton blooms occur. Other parts of the Barataria-Terrebonne area have lower turbidity and longer residence times (lower water turnover). In these areas, light limitation will be even less likely to prevent increased algal biomass and the concomitant increased likelihood of toxic and noxious algal blooms if nutrient loadings increase.

Chlorophyll *a*

Phytoplankton production of biomass is a major source of the organic material consumed in aquatic systems. Phytoplankton biomass may be roughly equated to photosynthetic pigments represented by chlorophyll *a* (Chl *a*). The ratio of Chl *a*:carbon varies depending on the nutritional status of the cells, light, and species composition and accessory pigments like Chl *b, c,* and carotenoids also are important. Chl *a* is now easily measured using a variety of techniques that discriminate between the metabolically active form and the metabolically inactive Chl *a* molecule without the Mg ion in the phytol ring. Measurements of Chl *a* in the 1950s and 1960s in the study area did not distinguish between the active and inactive forms of Chl *a*.

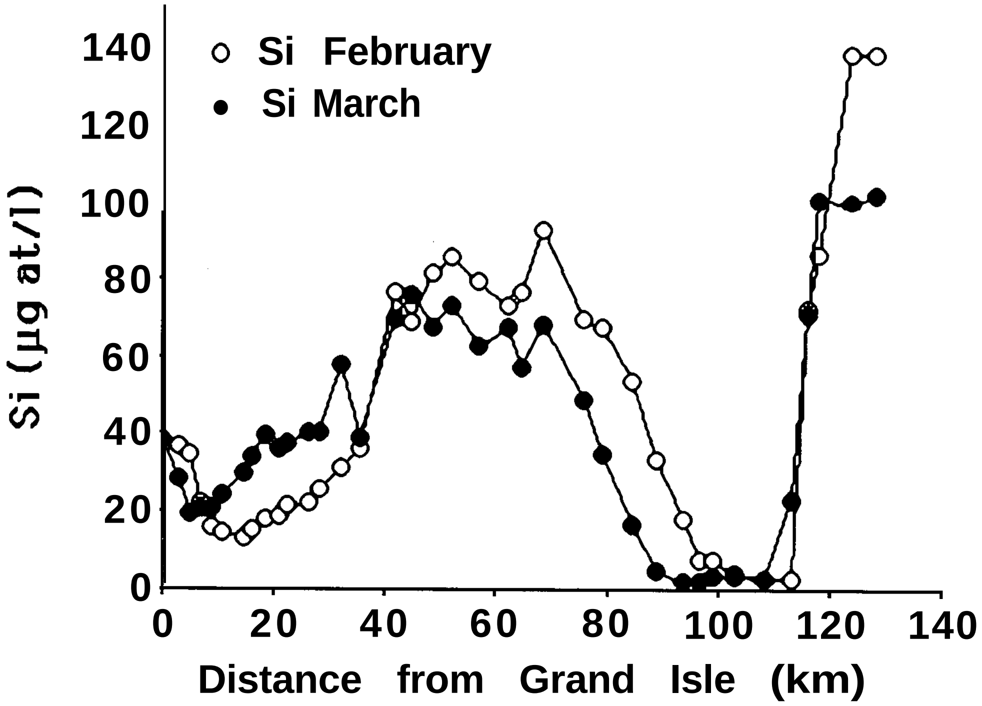


Figure 19. An example of silicate depletion in the upper end of Barataria Bay during February and March 1994.

In general, the amount of Chl *a* in the upper part of the basin is higher than at the estuarine entrance (see also figure 6 for nearshore coastal waters). An example is shown in figure 20 for the Barataria watershed. An annual average for 12 samples at 37 stations is depicted together with an estimate of the variability. Water entering from Bayou Chevreuil at Chackbay, Louisiana, into Lac des Allemands quickly accumulates Chl *a* entering the lake. As water leaves the lake Chl *a* concentration decreases. The same pattern of rise and fall entering and leaving an open water body occurs twice more at Lake Salvador and Barataria Bay. A reasonable interpretation of these events is that water turnover within the open water bodies is slower than in the drainage channels that dilute the growth of phytoplankton, thus tipping the balance between accumulation and loss. Note that most of the Chl *a* is not the degraded or inactive form of Chl *a* (pheopigments) but the active form (figure 20). Further, the rapid decline in Chl *a* as water leaves the lake implies a rapid turnover of phytoplankton cells. A similar pattern is found for important nutrients.

The data are averages but illustrate the variability within the watershed along the transect and throughout the year. The variability at the estuarine entrances to Terrebonne and Barataria bays further illustrates this variability (figure 21). Data from the same Barataria Bay north-to-south transect shown in the previous figure illustrate, in 1994 at least, the very high Chl *a* concentrations (100 µg l^{-1}) in the northern part of the watershed in the winter and a big spring peak in the southern area (40 µg l^{-1}) during the spring. Lower values are found in Terrebonne Bay compared to Barataria Bay during 1982–1983.

There is, therefore, considerable variability within small distances and relatively short time periods. This variability makes the detection of trends more difficult as the data set size is smaller. However, there are several data sets of sufficient size to make comparisons. The Louisiana Department of Wildlife and Fisheries (LDWF) (Harrison) made measurements of Chl *a* in the 1964–1966 era in the southern portions of Barataria and Terrebonne estuaries; Dagg (No Date) reported on Chl *a* concentration for samples taken in 1982–1983 in Terrebonne estuarine entrance; Seaton (1979) for a few samples in Barataria Bay; and unpublished data from Turner and Rabalais for the entire Barataria transect (1994). Data for Lac des Allemands include those of Lantz (1979) for 1964–1966, Butler (1975) for 1973–1974, Seaton (1979) for 1976, and Turner and Rabalais (unpublished).

The trends at all three locations show a large rise in Chl *a* (figure 22). The Lac des Allemands samples show a five-fold increase in the past 30 years, and the Barataria Bay samples indicate a doubling in the past 20 years. Samples from lower Terrebonne Bay indicate a doubling from the mid-1970s to the early 1980s. These results are very strong evidence that despite the natural variability, eutrophication of the Terrebonne and Barataria watersheds has occurred in recent decades.

Sedimentary Record

Sediment accumulates each year under normal circumstances and contains remnants of materials in the overlying water. This sediment record, when properly dated and

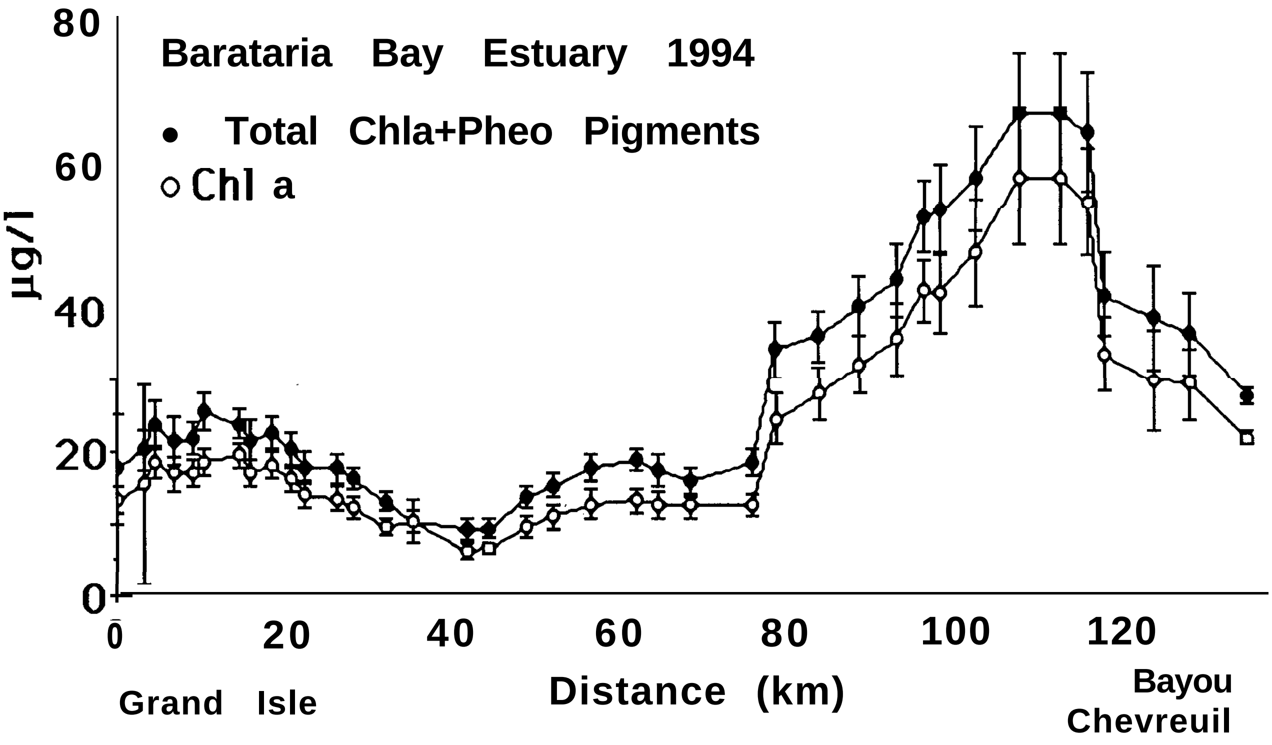


Figure 20. The average annual chlorophyll ***a*** concentration along a north-to-south transect in Barataria Bay. See figure 3 for a location. The standard error (± 1) is shown for monthly samples. The 0 km mark is the entrance to Grand Isle and the transect's end point is Bayou Chevreuil at Chackbay, LA.

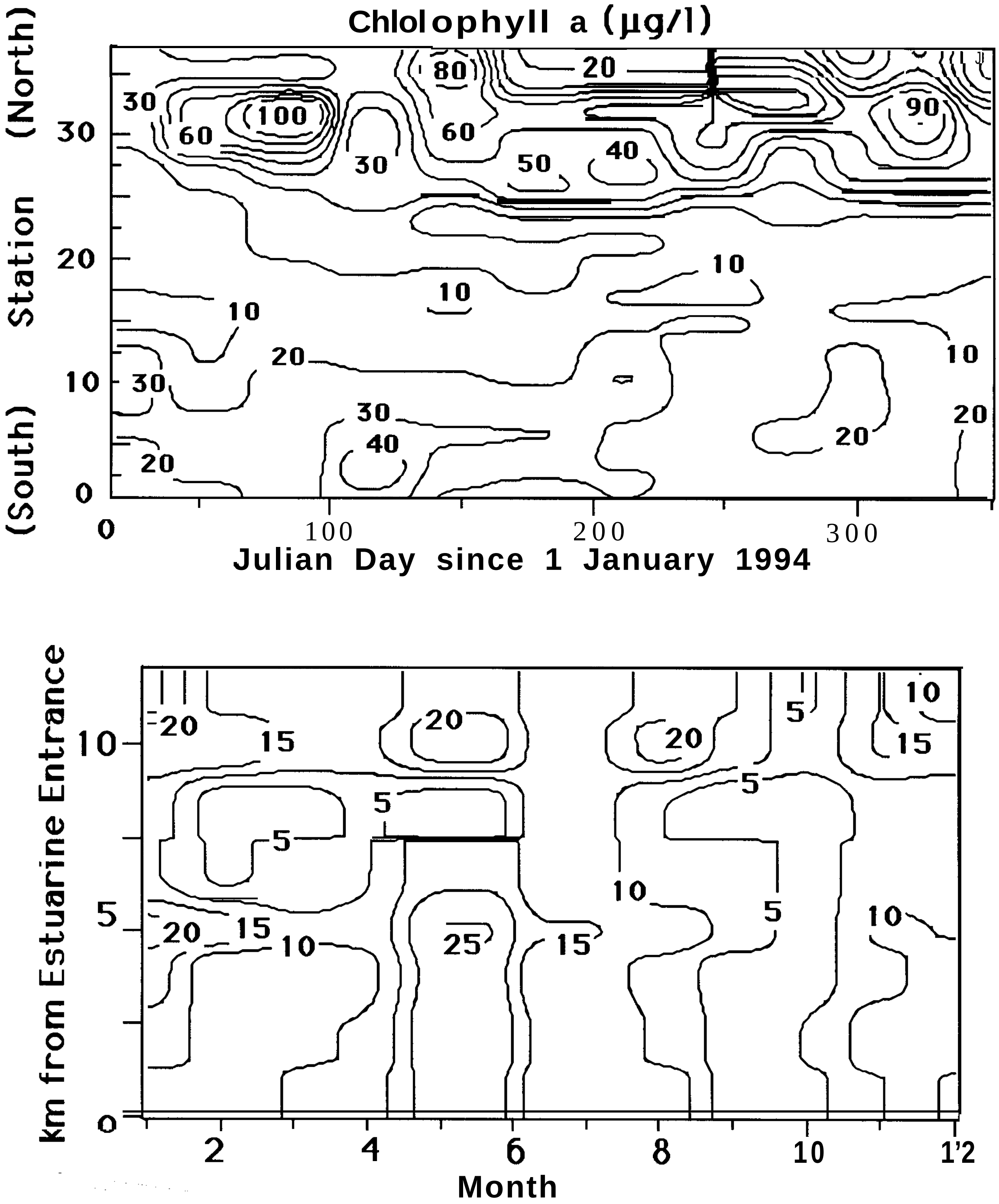


Figure 2 1. Contour plots of the monthly chlorophyll *a* concentration along a north-to-south transect in Barataria for 1994 (upper panel) and Terrebonne Bay for 1982-1983 (lower panel).

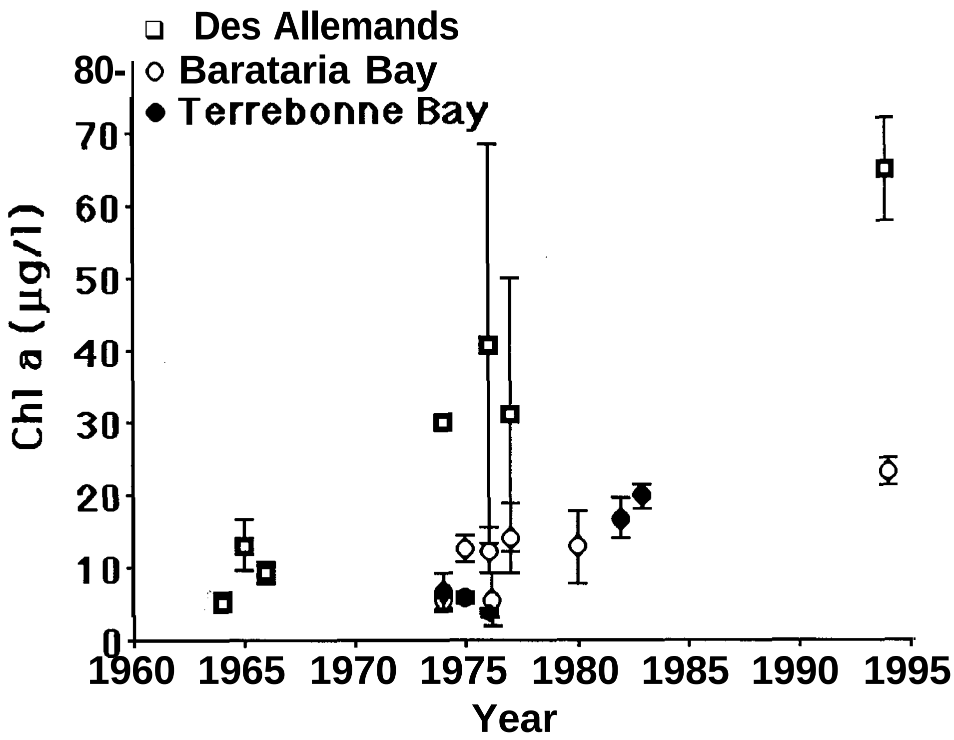


Figure 22. The long-term average annual chlorophyll ***a*** (total chlorophyll ***a*** and pheopigments, where measured) for the Lac des Allemands, southern Barataria Bay, and southern Terrebonne Bay. The error bars are ±1 standard error for monthly samples. Note: data for Lac des Allemands were erroneously reported as mg l^{-1} rather than as µg l^{-1} in original the report. Data are from Lantz (1979), Butler (1975), Barrett et al. (1978), Seaton (1979), Turner and Rabalais (unpubl. data), and Dagg (ND).

analyzed, appears as a useful relatively inexpensive and potentially reliable reservoir of information on water quality changes, especially in the absence of long-term water quality monitoring results. Schelske, Conley, and their colleagues demonstrated the utility of using this approach in the Great Lakes, and there are numerous paleolimnological studies of freshwater lakes and estuaries such as the Chesapeake Bay (e.g., Brush and colleagues; Brush 1984, 1989; Brush and Davis 1984). An examination of the continental shelf sediment near the Mississippi River Delta revealed parallel changes in riverine water quality and the record of silica preserved in diatom frustules (measured as biogenic silica, or BSi; Turner and Rabalais 1994). In other words, diatom remains in offshore sediment increased in proportion to nutrient loading from the nearby Mississippi River.

Sediment cores were collected by Turner, Rabalais, and Dortch from wetland sites in Barataria and Terrebonne salt marshes in 1991 and dated using gamma spectroscopy (unpublished). Appropriate corrections were made to account for core compaction during field collection, sectioning shavings, and subsampling. Accumulations of biologically bound silica (BSi) were used as a surrogate for diatom remains. In general, the %BSi is highest in the most recent core segment, especially since 1920, but recently stabilized or declined in Barataria Bay (figure 23). Each of the estuarine sediment samples showed a coincidental rise and fall between %BSi and local fertilizer use. There was no apparent coupling between the %BSi in the offshore sediment compared to the onshore sediment. A key indicator period is the 1900–1940s era, when the offshore had a conspicuous rise (there are data from another six cores not shown) and fall that was absent in the onshore (estuarine) samples. Effects of agricultural fertilizers appear to be significant and greater than the effects of population growth. Nonpoint runoff is a significant source of nutrients affecting these estuaries; even though, they have relatively large wetland areas to buffer the nutrient loading.

The results lead to several conclusions:

(1) Variations in local land-use patterns have relatively fast (<1 year) effects on estuarine water quality. The nutrients from land runoff vary from year to year, but the loadings from sewage would not. Water quality changes (represented by the relative diatom abundances) fluctuated annually with land-cover changes and fertilizer applications. This result implies that nonpoint runoff is the main factor responsible for the variations in diatom accumulation.

(2) Wetlands very incompletely buffer the effects of increased nutrient loading on water quality. If wetlands were able to absorb the runoff from land-use changes, then the annual variability would be much less or unnoticed.

(3) The wetland sediment record is useful to reconstruct water quality information in the absence of water quality records.

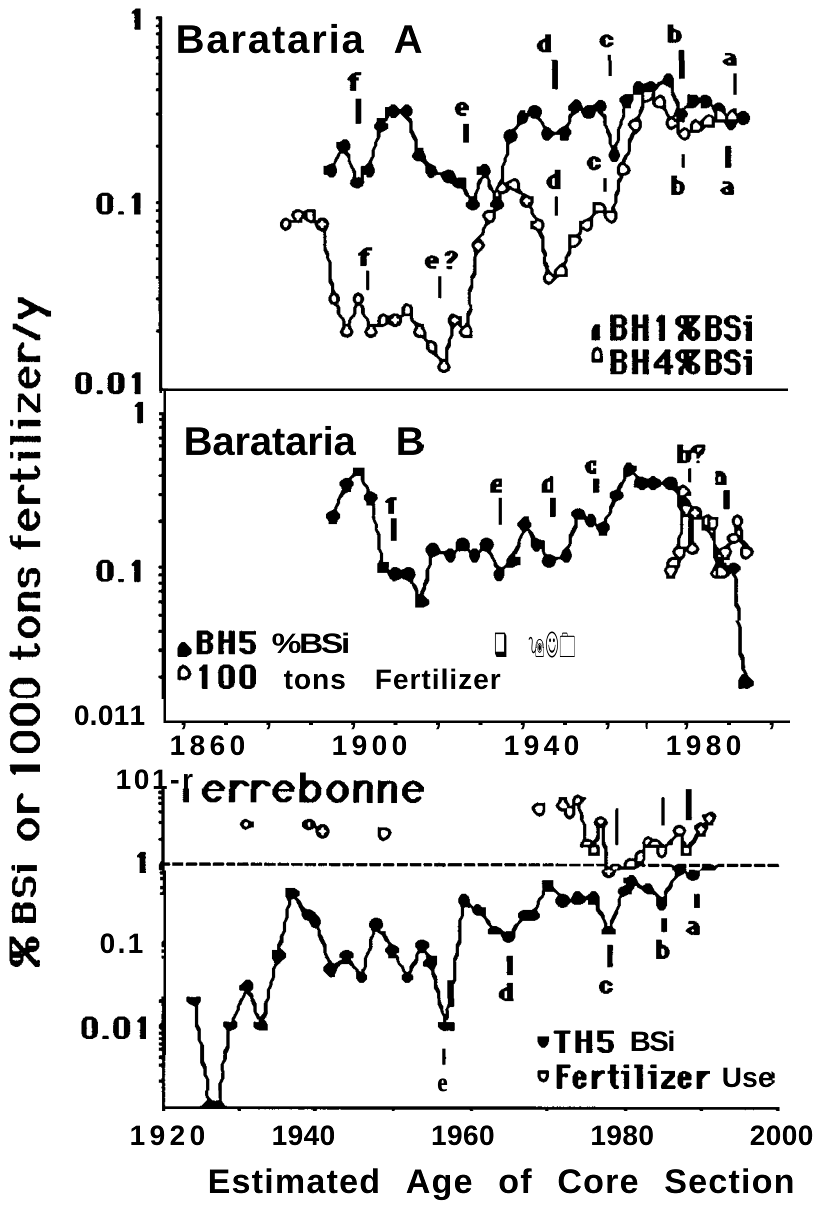


Figure 23. Biogenic silica accumulation from dated sediment cores collected in the Barataria Bay and Terrebonne Bay estuaries. Note the similarity of changes occurring between nutrient loading from fertilizers and BSi accumulations within the core sections. The changes in BSi downcore are not coincidental with the variability in BSi accumulation in cores immediately offshore of the Barataria Bay entrance, suggesting differences in nutrient controls in estuarine and offshore environments. From Turner and Rabalais, manuscript.

Wetland Removal of Nutrients

Nutrients may be absorbed or released by wetlands when water flows over or through them. Important factors that determine amount and direction of these exchanges include the wetland type, the hydrologic regime, and the nutrient concentration and composition.

Louisiana wetlands are not uniform in their uptake and release of nutrients that strongly influence phytoplankton growth rates, for example, nitrate and phosphate. Although there are no data on silicate exchange rates between the water column and wetlands, land surfaces are generally considered sources of silicate, not sinks. Swamps appear to take up some nitrogen and phosphorus forms, whereas wetlands with emergent macrophytes release these nutrients (table 4). Exceptions are the brackish and saline marshes of western Terrebonne Parish that appear to release phosphate; the same marsh type, however, in Barataria Bay absorbs phosphate. The researchers who developed the data suggested that this difference in phosphorus release was related to the Atchafalaya River discharges into the western Terrebonne marshes.

How water moves within and through wetlands determines several types of the nutrient exchanges with the overlying water. If the residence time is short, there is less time for nutrient uptake and release. The recent experimental release of Mississippi River water through the Bonnet Carré spillway resulted in virtually no changes in dissolved nitrogen and phosphorus as water moved from the diversion site to Lake Pontchartrain, probably because the residence time from release to lakeshore was about one day. Uptake is favored by high residence times, unless deoxygenation occurs (anaerobic conditions greatly favor ammonia and phosphate release, for example). Further, the higher the loading rate, the lower the efficiency of removal in overland flow systems. (Mitsch and Gosselink 1993, provide a review of nitrogen and phosphorus removal in freshwater systems). Finally, water flow through (or under) a wetland has a higher removal rate of some nitrogen and phosphorus forms than water flow over a wetland.

The nitrogen uptake rates for swamps in the Barataria basin were estimated for one loading rate by Kemp et al. (1985). The removal efficiency was 26% and 41% of total nitrogen and total phosphorus, respectively, or 3.87 and 1.7 g $m^{-2} yr^{-1}$. The application of these few measurements into management principles generally applicable to river diversions should be done cautiously, if only because scaling up from small experimental study areas to a river diversion has not been done for Louisiana. This is not to say that river diversions should not be attempted but rather that realistic calculations of nutrient removal should. Consider that the total area of swamp in Barataria basin is about 64,462 ha. If this swamp area were to have water diverted over it at the same rate as in the Kemp et al. (1985) study and at the same total nitrogen removal rate, only 10% of a 1% diversion of the Mississippi River average flow would be removed. Because the emergent wetlands are a net source of dissolved nitrogen, any of the unabsorbed dissolved nitrogen will likely not be taken up once it leaves the swamp. Observations on the effects of the planned diversion at Davis Pond will shed light on the net transport rates and effects on phytoplankton in the downstream water bodies.

Table 4. Import (I) and export (E) of nitrogen and phosphorus from wetlands through overland flow (g element m^{-2} y^{-1}).

Location	Wetland Type	DN	TN	DIP	TP
Fourleague Bay[1]	Fresh	E-NO_3 E-NH_4 E-TKN		E	E
Bayou Chevreuil[2]	Swamp	I-NO_3 I-NH_4 E-DON	I-3.87	E-OP-0.1 E-TP-0.19	I-1.71
Barataria Bay[3]	salt and brackish	E-NO_3 E-NH_4 E-DON	E-TKN	I	E
Fourleague Bay[3]	salt and brackish	E-NO_3 E-NH_4 E-DON	E-TKN	E	E
Bonnet Carré[4]	fresh	no change		no change	

[1]Stern et al. (1986)
[2]Kemp et al. (1985)
[3]Childers and Day (1990 a, b)
[4]unpublished USGS report, based on trial river diversion in 1993/4

River Diversions

River diversions are planned or being built that will bring large quantities of water from the Mississippi River into Barataria Bay. USACE estimates that the maximum flow for three Louisiana river diversions (Davis Pond, Caernarvon, and Bonnet Carré) will be 10,650, 8,000 and 30,000 cfs, respectively, and occur mostly during the spring. Of these, only Davis Pond goes directly into the Barataria estuarine watershed. The total maximum flow of the three diversions (48,650 cfs) is 1% of the average flow for the year (4.6×10^6 cfs, 1954 through 1988) and only 0.2% for the Davis Pond diversion. There is a direct connection between river nutrient loading and the hypoxic zones on the Louisiana shelf, so these river diversions might be considered as a possible management tool to decrease organic loading to the offshore waters and thereby raise oxygen concentrations in offshore bottom waters. However, the amounts of river water to be diverted are so small relative to the size of the discharge that it will have an insignificant effect on the size, frequency, and duration of oxygen depletion within offshore bottom waters.

The importance of the Davis Pond diversion and future diversions to Barataria can be evaluated in terms of the quantity and content of water brought into the Barataria estuary. It may be a significant influence on the algal community for several reasons related to the volume of water relative to present freshwater inflow and the concentration of important growth limiting nutrients within it. The river flow within the Mississippi River main channel is 80 times larger than the freshwater inflow into the Barataria watershed (NOAA 1987), which is almost exclusively from local rainfall. Diverting one percent of the river flow into the watershed will almost double the freshwater budget. The concentration of various nutrients in the river is also much higher in Mississippi River water than in the estuary (figure 24), especially in the spring when diversions are at their maximum. For example, nitrate concentrations in the river during the spring are 20–50 times higher than the average for all of the Barataria estuary for nitrate and 1–2 times higher for silicate. Silicate is important because of probable silicate limitation of phytoplankton production in Lac des Allemands during winter 1994. Nitrogen limitation of phytoplankton growth appears probable all year, based on the nutrient growth limitation experiments in Terrebonne Bay and the nutrient ratios and changes in nutrients from north to south in the Barataria Bay estuary. Further, these additions of nutrients limiting phytoplankton growth will occur in a system with already high-standing crops of algal biomass.

The combination of even a 20% increase in freshwater inflow that contains much higher concentrations of nutrients will very likely cause further eutrophication in the Barataria watershed; although, their effects may be coincidental with the seasonal timing of the diversions. This result is likely in light of the apparent effects of fertilizers and the documented increase in phytoplankton biomass (measured as chlorophyll *a*). Because the smaller additions of fertilizers have a demonstrable effect on primary production, it is likely that the ability of wetlands to absorb the additional amounts of nutrients in river water is already much less than 100%. These concerns are summarized in table 5. A checklist of issues affecting diversions and the possibility for wetlands to buffer the

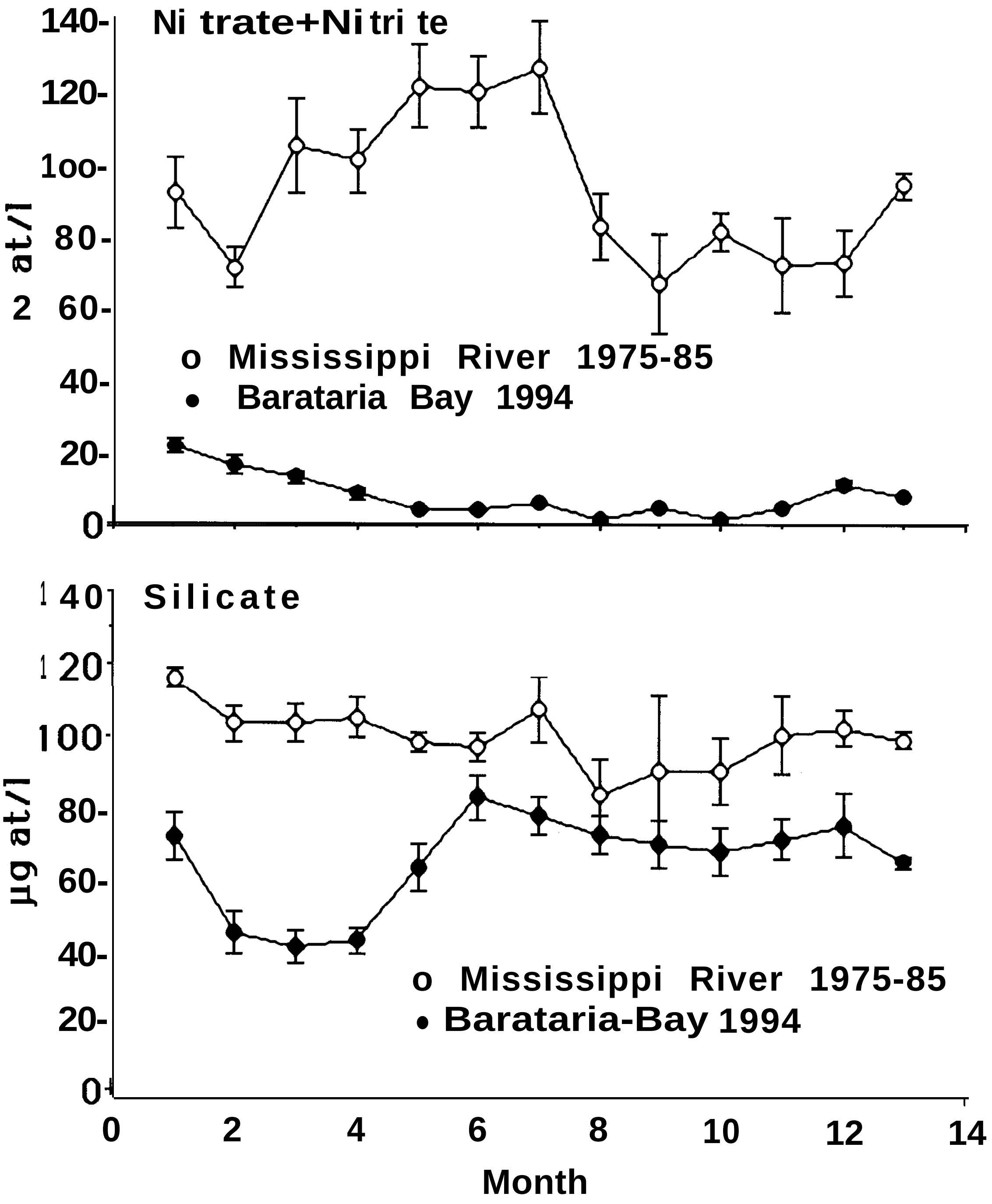


Figure 24. The average concentration of nitrate and nitrite (upper panel) and silicate (lower panel) along the 1994 transect running north to south in Barataria Bay (see figure 3) compared to the seasonal concentration of that nutrient in the Mississippi river at St. Francisville (average of 1975-1 985).

Table 5. Comparison of nutrients and freshwater inflow volume in diversions moving riverwater to Barataria Bay during spring.

Issue	Observation
1. Riverwater nutrient concentrations compared to that in the Barataria watershed samples in 1994:	
(a) Nitrate	80 times higher
(b) Silicate	1–2 times higher
2. Freshwater inflow into Barataria Bay watershed	
1% of Mississippi River flow	double freshwater inflow

potential eutrophication from higher nutrient loadings is provided in table 6.

Finally, the ameliorating effect of overland flow through the wetlands is likely to be less than optimal because the diversions take place when overland flow is at a seasonal average not high point (figure 25) and because nutrient regeneration is significant. The consequence to the estuary is an increased likelihood of ever higher algal blooms, perhaps toxic blooms (see p. 80), than heretofore experienced in the already eutrophic waters and changes in estuarine food webs.

Recommendations

Reducing the effects of eutrophication is the basis for numerous legislation. However, some less obvious implications are related to the interrelationship of policies of nutrient control in fresh water and the impact or lack of impact on coastal systems. The management of eutrophication on a national scale has not sufficiently integrated freshwater and estuarine systems. The primary nutrient targeted to improve water quality in freshwater systems is usually phosphorus and is based on the numerous excellent laboratory and field studies of the stimulatory effect of phosphorus on freshwater ecosystems. However, coastal systems are usually thought to be nitrogen limited, at least part of the time, and this includes the Barataria and Terrebonne Bay estuaries.

Control of nutrient inputs common to freshwater and coastal systems is sewage treatment in general. But, as is shown for the Mississippi River (Turner and Rabalais 1991a), the terrestrial system is very leaky, and treatment does not mean a reduction of loading to the estuary via water and precipitation. So a second understated issue,

Table 6. Checklist of issues for river diversions.

Issue	Favored by or indicated by	Louisiana Experience
Algal Growth	short freshwater residence time (days)	3–20 days, Fourleague Bay 180 days, Barataria estuary
	high Chl *a* or primary production	yes, Fourleague Bay (both) yes, Barataria Bay yes, Bonnet Carré diversion study
	low turbidity	30 cm, Fourleague B. (Randall 1986) 47 cm, Fourleague B. (Madden 1992) 49 cm, B. Des Allemands 1994 64 cm, lower Barataria Bay 1994
	higher nutrient loading	Bonnet Carré 1993/4 resulted in higher Chl *a* levels
Nutrient Removal by wetlands	Long contact time (days)	1 day, Bonnet Carré (very short)
	Sufficient area	restricted/limited by existing upland development and landower concerns; not extensive relative to loading rates
	higher loading = less retention	not documented in Louisiana, but is a general experience nationwide
	nitrate and phosphate removal	export from all but swamps and uplands

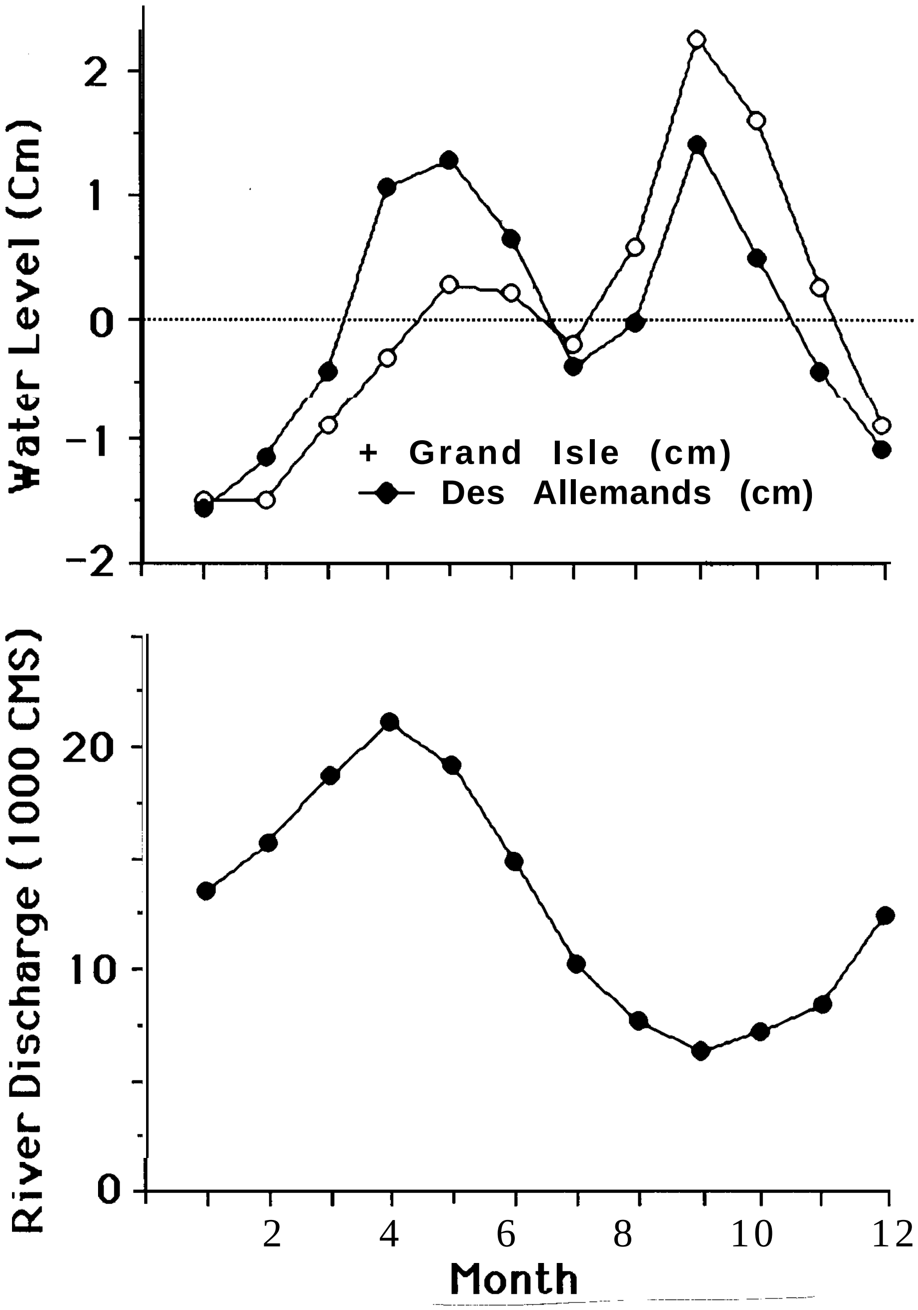


Figure 25. Variations in the average monthly water level at Grand Isle and Lac des Allemands, relative to the annual average (from Turner 1991) and the timing of the average spring peak discharge of the Mississippi River at Tarbert Landing.

therefore, is that sewage treatment upstream does not necessarily equate to controlling nutrient loading to downstream estuaries.

Third, minimization and mitigation of nutrient consumption seem a less-prudent management policy than reduction. The ecosystem is simply too leaky to control all flows of important nutrients from consumption to arrival in the estuary.

Some options for controlling nutrient loadings to the estuary are listed in table 7. Reducing the quantities used lessens subsequent efforts to mitigate, treat, or manage impacts. Best management practices for point and nonpoint sources are known contributors for subsequent treatment and control of those nutrients released and are not part of this report. There is no one magical, simple, and easily applied biological or management technique but rather a variety of approaches that have been successfully applied elsewhere. However, these approaches become more difficult and expensive as quantities increase. If source quantities are reduced, then managing subsequent problems becomes easier.

The data to evaluate how much these two estuaries have changed in the past 20 years is amazingly sparse and needs to be supplemented with current data. Needed is a re-survey of the water quality monitoring stations in mid-estuary that were sampled in the late 1960s and early 1970s (primarily by the Louisiana Wild Life and Fisheries Commission). These surveys should be conducted at least monthly over a 24-month period. They can serve to update our knowledge of the health of the estuaries and be used as a basis for understanding management options if coordinated with other field research and monitoring programs.

There are various plans and discussion of plans for substantial alteration of the freshwater inflows to these and other estuaries in Louisiana. The proposed diversions move water with a nutrient concentration that is in general much higher than the receiving water. We should conduct a comprehensive evaluation of the effects of nutrient loading from river diversions to the coastal zone. This evaluation should include field studies of replicated nutrient additions and be integrated with the monitoring sampling presently underway or planned (by various agencies and as recommended above).

Dissolved Oxygen

Introduction

Dissolved oxygen is essential to the metabolism of most aquatic organisms. As such, it is also one of the fundamental parameters for characterization of trophic conditions in aquatic ecosystems. Oxygen is abundant in the atmosphere and dissolves readily in water. Oxygen solubility is affected in a nonlinear way by temperature, salinity, and atmospheric pressure. In short, cold water absorbs more oxygen than does warm water, salinity decreases solubility, and pressure increases it. Equilibrium oxygen content, however, is rarely achieved in nature because of biological production and uptake.

Oxygen-depleted waters are obvious manifestations of nutrient and organic carbon

Table 7. Options for the control of nutrient loadings to watersheds (adapted from Meybeck et al. 1989).

	Control Option		
Problem	Source Reduction	Management Option	Treatment Option
Point discharges			
Sewage:	pretreatment of industrial wastes	Separate sewage and storm waters	Conventional treatment; nitrification/denitrification; physical separation; land disposal
Industry	Reduce product loss	Good housekeeping process changes	Chemical reaction; biological treatment
Nonpoint sources			
Fertilizers	Regulate rate, frequency and amount used	Minimize runoff and leaching	Collect runoff and hold in ponds
Farmlands	Use manure for fertilizer	Reduce erosion and runoff by vegetation or buffer strips, cultivation practices	Collect and treat
Forest lands	Regulate rate, frequency and amount used	Vegetation buffer strips	Collect and treat where possible
Urban lands	Recycling	Recycling	Collect and treat

enrichment. Decreases in oxygen content with respect to a saturation value can be attributed to the decay of organic matter through bacterial action and respiration of plants and animals. Oxygen concentrations may be severely reduced (hypoxia or anoxia) in waters rich in labile organic matter, especially in stratified water columns where the depletion rate of oxygen is greater than the re-aeration of the water column from surface oxygen production or diffusion. In contrast, a surplus of dissolved oxygen with respect to the saturation value is almost exclusively the consequence of photosynthesis. Oxygen saturation values of 200% or higher, for example, are frequently measured in productive surface waters during daylight hours. In aquatic ecosystems, therefore, variability of dissolved oxygen in time and space provides an insight into two fundamental ecosystem processes: production and decay of organic matter.

In addition to oxygen concentration and saturation, biological oxygen demand (BOD) is also frequently used in limnological studies. This method measures the total oxygen uptake in a dark bottle stored at ambient temperature over a specific period of time, usually five days. It provides an estimate of the total community respiration rate and, as such, it is useful for assessing the availability of labile organic matter in natural waters. Indirectly, the BOD method may be used to determine the presence of toxic substances which inhibit the respiration process.

Dissolved oxygen values are especially useful for assessing long-term natural and anthropogenic impacts on aquatic ecosystems. There are several important reasons for this. First, the methodology for determining dissolved oxygen concentration in natural waters, the Winkler titrametric method, has remained fundamentally unchanged since 1888. Recent introduction of oxygen probes still permits a full comparison because the probes are usually calibrated with oxygen saturation in air at a given temperature and barometric pressure but cross-calibrated using the Winkler titrametric method. Most dissolved oxygen kits used today for monitoring employ the azide-iodide modification of the original Winkler method, and results are the same as the wet titration method. In contrast to the Winkler method, methods for determination of dissolved nutrient concentrations, pigment concentrations, and primary productivity, for example, have changed substantially during the last several decades, which makes comparison of long-term data records difficult or impossible. Second, the systematic error of the Winkler method ranges from 0.1% to 5%, which is significantly lower when compared to most other analytical methods.

It should be pointed out that not all oxygen depletion results from nutrient and/or organic carbon enrichment. In many deep channels, density stratification controlled by temperature and/or salinity differences may prevent the re-aeration of bottom waters without increased organic loading. Frequently, however, the oxygen demand in bottom waters of deeper channels, especially near industrialized and urbanized areas, may be related to chemical oxygen demand or organic loading from sewage outfalls. In submerged aquatic vegetation beds (e.g., seagrass meadows or heavily vegetated creeks and bayous), a normal diel cycle of oxygen concentration will produce low oxygen levels in the early morning hours following extensive respiration of the vegetation during the dark cycle. In other instances, the movement of naturally stagnant swamp waters into other water bodies following a flushing event may contribute temporarily to a low oxygen condition.

A word of caution should be given regarding the use of dissolved oxygen data in water quality–monitoring data sets. Continuous oxygen measurements (e.g., Summers and Engle 1992) indicate diel variability in dissolved oxygen levels in most estuaries. Water quality monitoring conducted primarily during daylight hours may bias distributions of dissolved oxygen measurements to the higher values when photosynthesis rates are greater. Also, most monitoring data represent surface water samples that are likely to have higher dissolved oxygen concentrations than waters at depth. Thus, most monitoring data will demonstrate oxygen values (and subsequent determinants of water quality) at the high end (or better water quality) of the distributions rather than depict the lower values for a diel period or a clearer representation of the entire diel cycle.

Data Sources

The most extensive dissolved oxygen data are collected by LDEQ Office of Water Resources in support of their annual Water Quality Inventory, USGS, and USACE. All data are submitted to the EPA STORET system.

Dissolved oxygen data are for surface waters. The values are reported as mg l^{-1}, or percent oxygen saturation. Most of the data contained in the LDEQ Water Quality Monitoring Inventory are collected during daylight hours (see note of caution above). Because LDEQ data are collected on a set schedule, the stations monitored are visited within an estimated 1-hr span of the previous collection. In other words, collection time is consistent, and the data are conducive to comparisons for long-term trends. Surface water BOD data are also routinely collected by USGS and USACE. Data were collected monthly or bi-monthly from 1978 to 1993, depending on the data set, from a series of stations across the Barataria-Terrebonne estuarine system. Stations with a 15-yr period of data were used in the trend analysis. Classification of surface waters (i.e., status, see below) was based on 1990–1993 data.

Bottom water dissolved oxygen data are not routinely collected by state or federal agencies, except for selected stations by LDWF in its monitoring work for the Louisiana Offshore Oil Port (LOOP). These data were not readily available for analyses. Other data are collected as part of special studies by state and federal institutions and research institutions. A long-term data set for nearshore coastal waters is held by Rabalais et al. (LUMCON) as part of their hypoxia studies. A summary of these historical data is provided.

Dissolved Oxygen in Surface Waters: Status and Trends

Classification of Surface Waters of the Barataria-Terrebonne Estuarine System based on Dissolved Oxygen Saturation and BOD

There are several classification schemes dealing with the water quality of surface waters. We can distinguish between two fundamentally different systems: limnosaprobity and toxicity. Limnosaprobity classifies surface waters primarily based on the organic content, production rate, and decomposition of organic matter, and the resulting community structure. The system of toxicity classifies surface waters based on concentrations of various toxic substances and their inhibitory effects manifested at the community level.

In limnosaprobic classification surface waters are usually subdivided into classes I (oligotrophic), II ($–mesosaprobic), III ("–mesosaprobic), and IV (polysaprobic). Class I refers to the highest quality water, which is well saturated in oxygen and has a low organic content. Class IV comprises heavily polluted waters, where pollution is most often caused by organic loading from anthropogenic sources. Importantly, different limnosaprobic systems that have appeared in the literature have included oxygen saturation and biological oxygen demand as classification factors. For the purpose of this project we adopted the system of water quality composed by Sladecek (1973). This is a remarkably versatile system frequently used in limnological studies. The classification criteria are explained in table 8.

On the basis of oxygen saturation averages and 95% confidence limits for the period 1990–1993, we classified 27 localities in the system. The majority of the sites falls into category IV according to this classification (table 9). Two sites, however, may be classified as oligotrophic (class I): Lac des Allemands and Little Lake. A multivariate analysis of water quality parameters (total organic nitrogen, total phosphorus, Secchi disk depth, chlorophyll *a*, and total inorganic nitrogen for quarterly sampling between August 1976 and August 1977) was used to classify 19 water bodies in the Barataria basin (Seaton 1979, Witzig and Day 1983). Only three sites from their study overlapped with the analysis in table 9, and the classifications were inconsistent for Lac des Allemands and Little Lake. The classification for Bayou Chevreuil, however, was consistent. Classification into Class I (oligotrophic) or Class II $–mesosaprobic) (table 9) is likely complicated by either high frequency of oxygen supersaturation events or high chlorophyll *a* values. Comparisons of water bodies percentages within classes for the two approaches are inappropriate because of the differences in sampling stations, the data used for classification, and the time periods.

A substantially lower number of BOD data is available for the period 1990–1993 versus oxygen saturation data. According to adopted criteria for BOD_5 (table 8), seven analyzed sites may be subdivided into classes II and III (table 10).

Table 8. Criteria for classification of surface waters based on oxygen saturation and biological oxygen demand (modified from Sladecek 1973).

Water Quality Class	Indicator	
	Oxygen saturation (%) (lower 95% probability level)	5-d BOD (mg/l) (upper 95% probability level)
I (oligosaprobic)	> 70	< 3
II ($–mesosaprobic)	50–70	3–5.9
III ("–mesosaprobic)	30–49	6–12
IV (polysaprobic)	< 30	> 12

Table 9. Classification of surface waters of the Barataria-Terrebonne estuarine system based on oxygen saturation values for the period 1990–1993. Water quality criteria are explained in table 8.

Station	Water Quality Class			
	I	II	III	IV
Lac des Allemands north of Raceland*^				
Little Lake at Temple				
Lake Verret at Attakapas Landing near Georgia*				
Houma Navigation Canal at Bayou Petit Caillou				
Bayou Lafourche at Cut Off				
Lower Grand River at Bayou Sorrel				
Bayou Segnette south of Westwego				
Bayou Segnette near Barataria				
Bayou Lafourche at Larose				
Bayou Lafourche at Lockport				
Bayou Grand Caillou at Dulac				
Bayou Des Allemands at Des Allemands				
Bayou Black at Gibson				
Bayou Black west of Houma				
Bayou Chavreuil near Chackbay				
Bayou Des Familles at Christmas Rd.				
Bayou Folse north of Houma				

Bayou Lafourche at Raceland				
Bayou Lafourche at Thibodaux				
Bayou Petite Caillou south of Houma				
Bayou Segnette near Westwego				
Bayou Terrebonne at Houma				
Grand Bayou at Grand Bayou				
Grand Bayou near Chackbay				
Kenta Canal northwest of Crown Point				
Kenta Canal west of Crown Point				
Millaudon Canal near Westwego				

*the classification is uncertain because of the high frequency of oxygen supersaturation events and/or high chlorophyll values (see figure 26).

^Lac des Allemands does not exhibit high frequency of supersaturation (see figure 26) because data are usually collected in early morning before maximum photosynthesis rates occur.

Table 10. Classification of surface waters of the Barataria-Terrebonne estuarine system based on BOD_5 values for the period 1990–1993. Water quality criteria are explained in table 8.

Station	Water Quality Class I	II	III	IV
Bayou Des Familles at Christmas Rd.				
Kenta Canal west of Crown Point				
Millaudon Canal near Westwego				
Bayou Segnette near Barataria				
Bayou Segnette south of Westwego (2.9 miles)				
Bayou Segnette south of Westwego (4.6 miles)				
Kenta Canal northwest of Crown Point				

Long-term Trends in Surface Oxygen Content and BOD

Dissolved oxygen is especially useful for assessing long-term changes in aquatic ecosystems because the measurement method has remained unchanged since 1888. Also, oxygen data are often normally distributed and variance of the sample is independent of the mean. These features are important prerequisites for the analysis of trend.

Methods. Prior to the trend analysis, it is important to eliminate local variability and short-term fluctuations in the data. We decided, therefore, to exclude stations with a total sampling record less than 15 years. Using this approach, 19 localities—10 in the Barataria-Terrebonne estuarine system and 9 on the Mississippi River—were selected for the analysis (figure 27). A standard linear regression model was used (example in figure 28). Models having a probability value >0.01 were rejected. The 95% confidence limits for the mean response are plotted as the dotted lines closest to the regression line. Also, the 95% prediction limits for the expected values are plotted as the pair of dotted lines farthest from the regression line. See p. xx for a more detailed description of statistical methods.

Barataria-Terrebonne Estuarine System. Results of trend analyses for the 10 stations in the system are summarized in table 11 (long-term trends for all stations are given in appendix A). Most sites in the Barataria-Terrebonne system show an increase in oxygen saturation with time, ranging between 0.13 and 1.16 % yr^{-1}. Because of a large variability

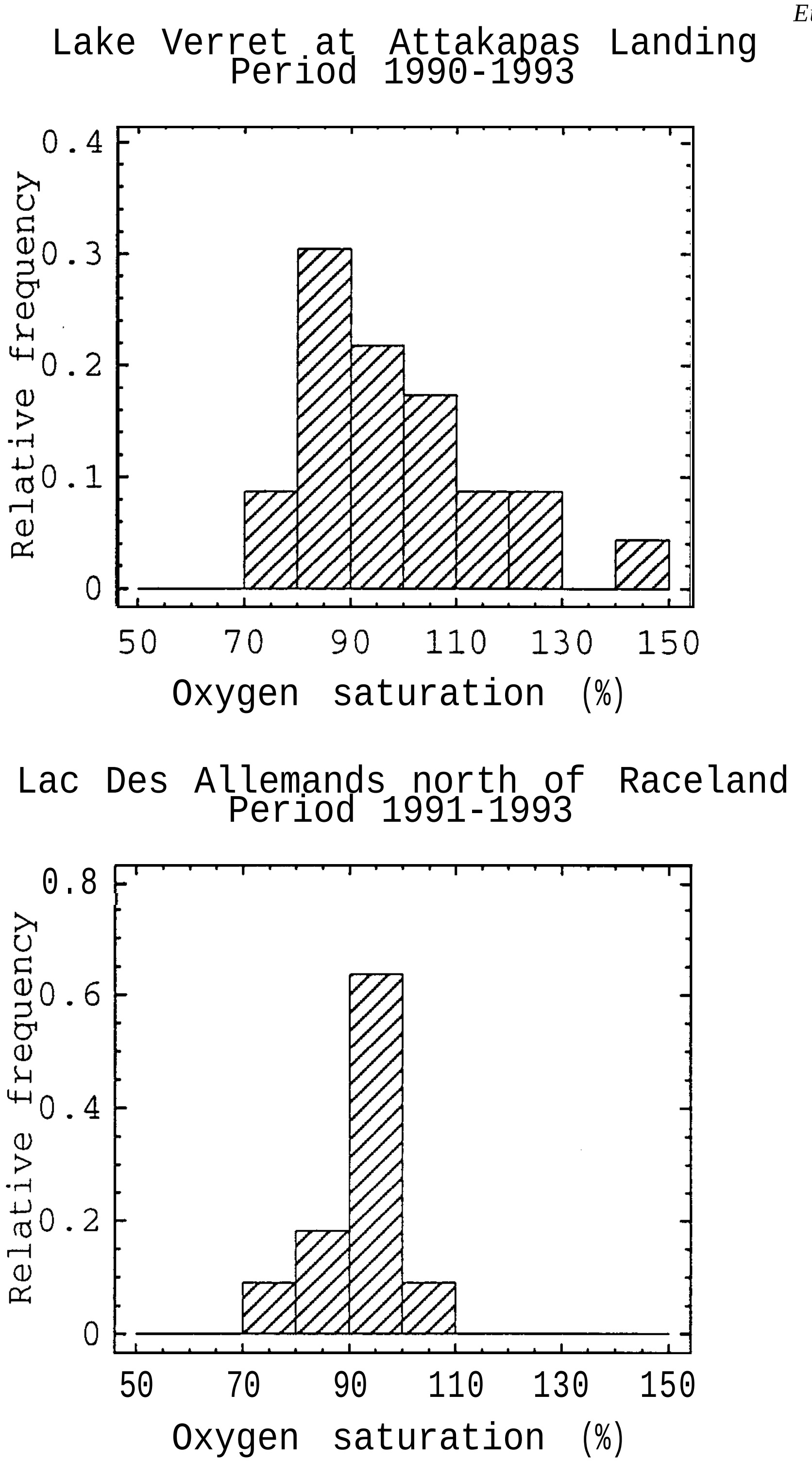


Figure 26. Frequency distribution of oxygen saturation at Lake Verret (upper panel) and at Lac des Allemands (lower panel) for periods indicated. Lac des Allemands does not exhibit high frequency of supersaturation because data are usually collected in early morning before maximum photosynthesis rates occur.

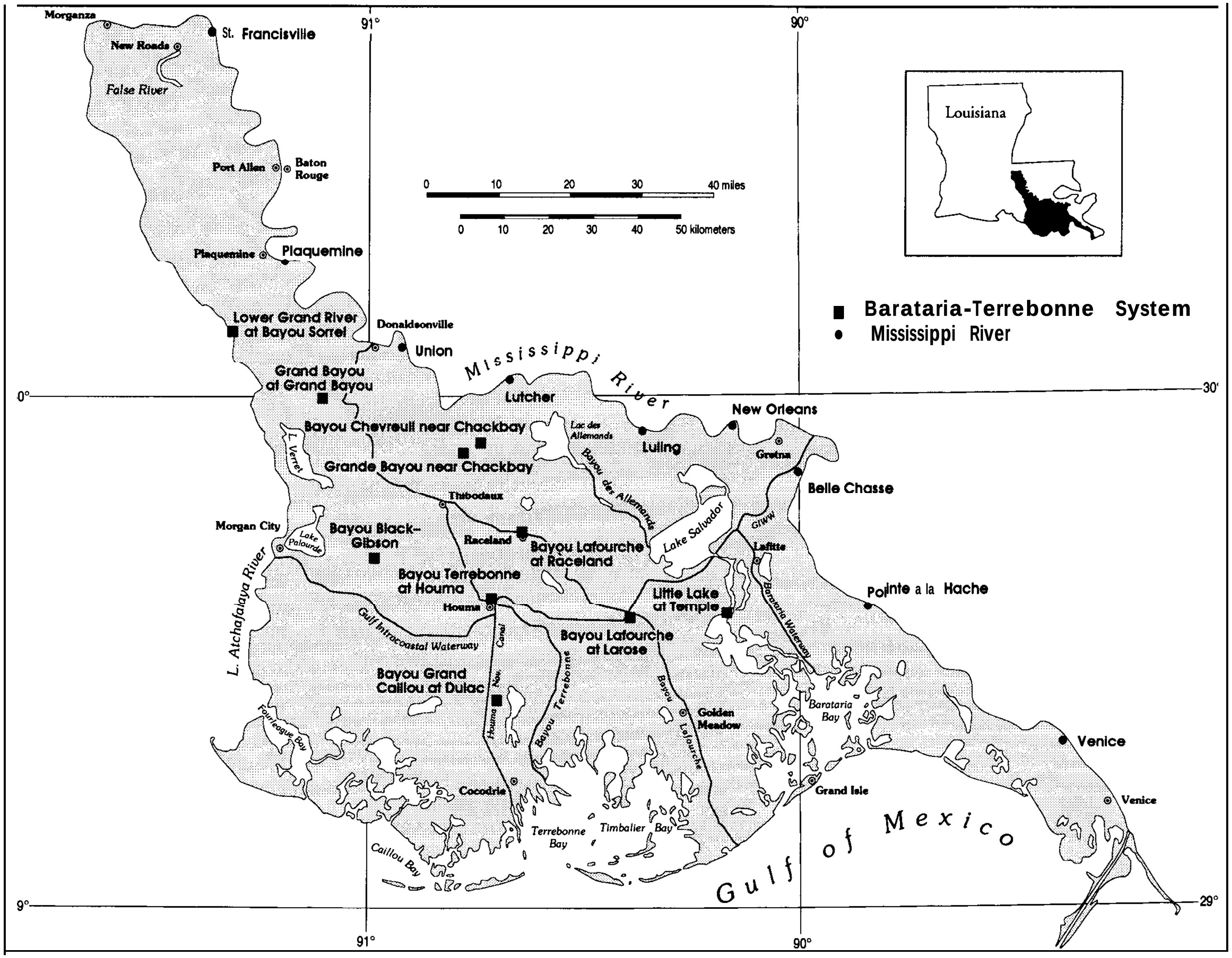


Figure 27. The area of study with stations indicated for total dissolved oxygen record of 15 years or longer.

Table 11. Long-term trends in oxygen saturation for selected water bodies within the Barataria-Terrebonne estuarine system and selected stations on the Mississippi River. The trends are computed from the data collected between 1955 and 1993; NS denotes the trends that are not significant at $p = 0.01$.

Station	Trend (% O_2 saturation yr^{-1})	Significance Level
Barataria-Terrebonne System		
Little Lake at Temple	0.68	$p < 0.01$
Bayou Lafourche at Larose	0.7	$p > 0.01$ (NS)
Bayou Lafourche at Raceland	0.24	$p > 0.01$ (NS)
Bayou Grand Caillou at Dulac	1.16	$p < 0.01$
Bayou Black at Gibson	0.44	$p < 0.01$
Bayou Chevreuil near Chackbay	0.62	$p > 0.01$ (NS)
Lower Grand River at Bayou Sorrel	-0.27	$p > 0.01$ (NS)
Bayou Terrebonne at Houma	0.13	$p > 0.01$ (NS)
Grand Bayou at Grand Bayou	-0.23	$p > 0.01$ (NS)
Grand Bayou near Chackbay	-0.43	$p < 0.01$
Mississippi River		
St. Francisville	-0.03	$p > 0.01$ (NS)
Plaquemine	0.19	$p < 0.01$
Union	0.17	$p > 0.01$ (NS)
Lutcher	0.31	$p < 0.01$
Luling	0.2	$p < 0.01$
New Orleans	0.37	$p < 0.01$
Belle Chase	0.65	$p < 0.01$
Pointe a la Hache	0.93	$p < 0.01$
Venice	0.28	$p < 0.01$

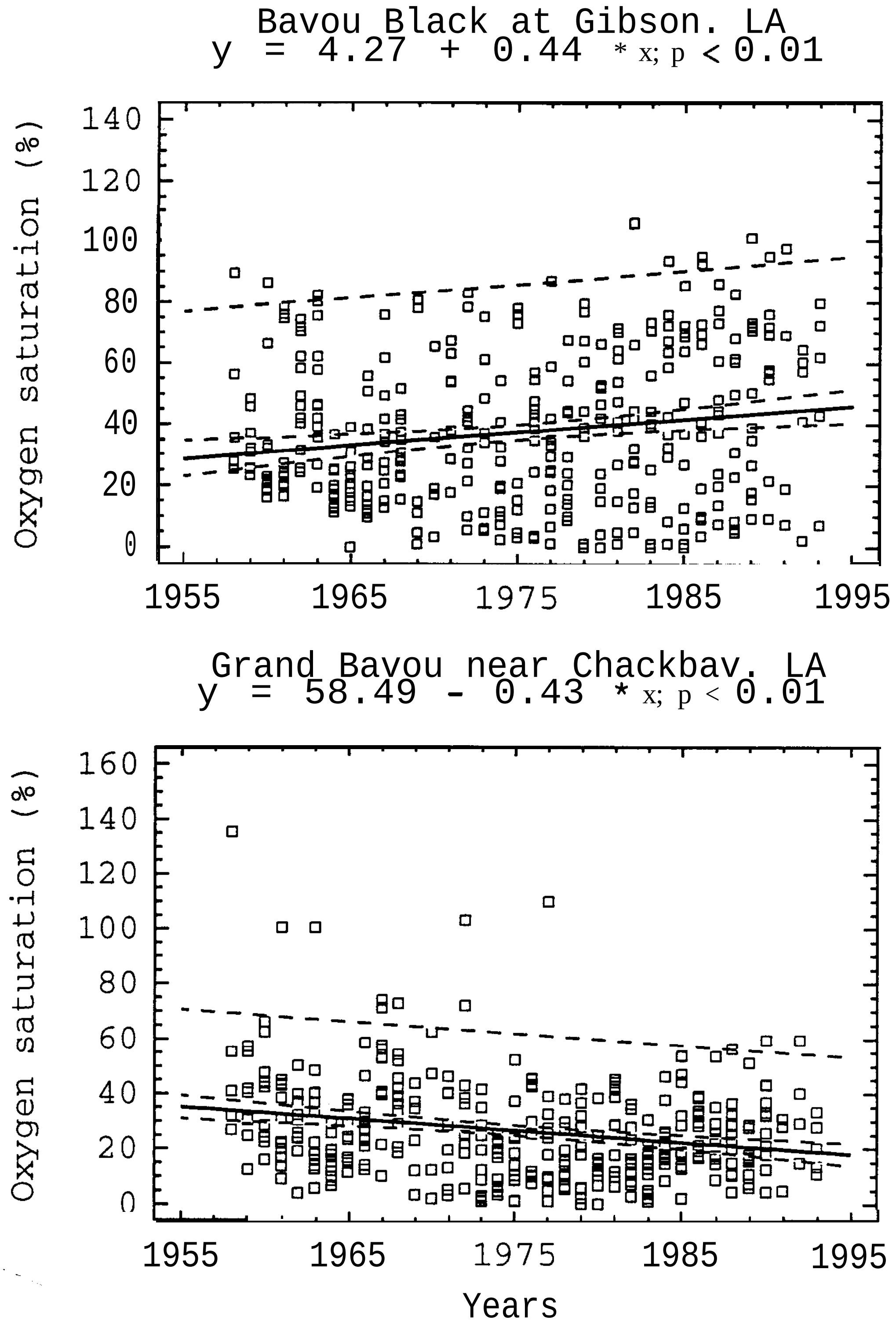


Figure 28. Long-term trends in surface oxygen saturation for BayGBlack at Gibson and Grand Bayou near Chackbay.

in the data, however, most of the observed trends are not statistically significant. Surface oxygen saturation increased significantly in three localities: Little Lake at Temple, Bayou Black at Gibson, and Bayou Grand Caillou at Dulac. In contrast, a significant trend of decrease is inferred for Grand Bayou near Chackbay (table 11). Examples of two significant yet opposite trends are shown in figure 28**.** In the case of Bayou Black, statistical data analysis showed that although the average oxygen saturation has increased between 1958 and 1993, the frequency of hypoxic events also has increased (figure 29).

Mississippi River. Oxygen saturation values in the lower Mississippi River depict a highly significant trend of increase at all stations except St. Francisville and Union. The rate of change inferred from significant linear models ($p < 0.01$) is high and ranges between 0.19% and 0.93% yr^{-1} (table 11). The highest recovery rate is observed at station Pointe a la Hache (figure 30). Interestingly, this long-term increase in oxygen saturation in the lower Mississippi River correlates well with the decreasing surface BOD (figure 31). The observed trends in BOD_5 are quite similar at all stations analyzed: Plaquemine, Union, Luling, New Orleans, and Venice. An average slope coefficient is around -0.06 mg O_2 l^{-1} yr^{-1}.

From the general theory of oxygen dynamics in flowing waters, it is inferred that the highest impact of organic loading from a point source is evident in oxygen saturation at a certain section of the river downstream from the same source (figure 32). Distance from the source of organic matter at which the oxygen minimum will develop depends on several factors, where flow rate, concentration of labile organic matter in riverine water, ambient oxygen concentration, and temperature are the most important. Inversely, if organic loading from a certain point source is gradually decreased over time we would expect to see the highest recovery rate at the same section of the river. This general theory could explain observed differences in oxygen recovery rates in the lower Mississippi River. It is obvious from figure 31 that the highest rates of increase in oxygen saturation are observed at several stations downstream from New Orleans. Thus, we can hypothesize that this differential oxygen recovery could have been the consequence of better efficiency of sewage treatment plants in the New Orleans area. An overall trend of decrease in BOD values in the lower Mississippi River, and at stations downstream from New Orleans as well, seems to support the above hypothesis.

Dissolved Oxygen in Bottom Waters

Data on the dissolved oxygen content of the waters of the Barataria-Terrebonne estuary are limited. None are available for trend analysis nor are adequate data available to assess status.

Hewatt (1950) collected data from 60 stations in Barataria Bay, mostly during April–July from 1945 to 1947. Of 95 measurements, only one fell below 2 mg l^{-1}. Barrett et al. (1978) recorded no instances of dissolved oxygen less than 2 mg l^{-1} in either Barataria or Terrebonne Bay for the period October 1974–September 1976. Faunal studies of dredged canals were conducted in Terrebonne Bay October 1972–September

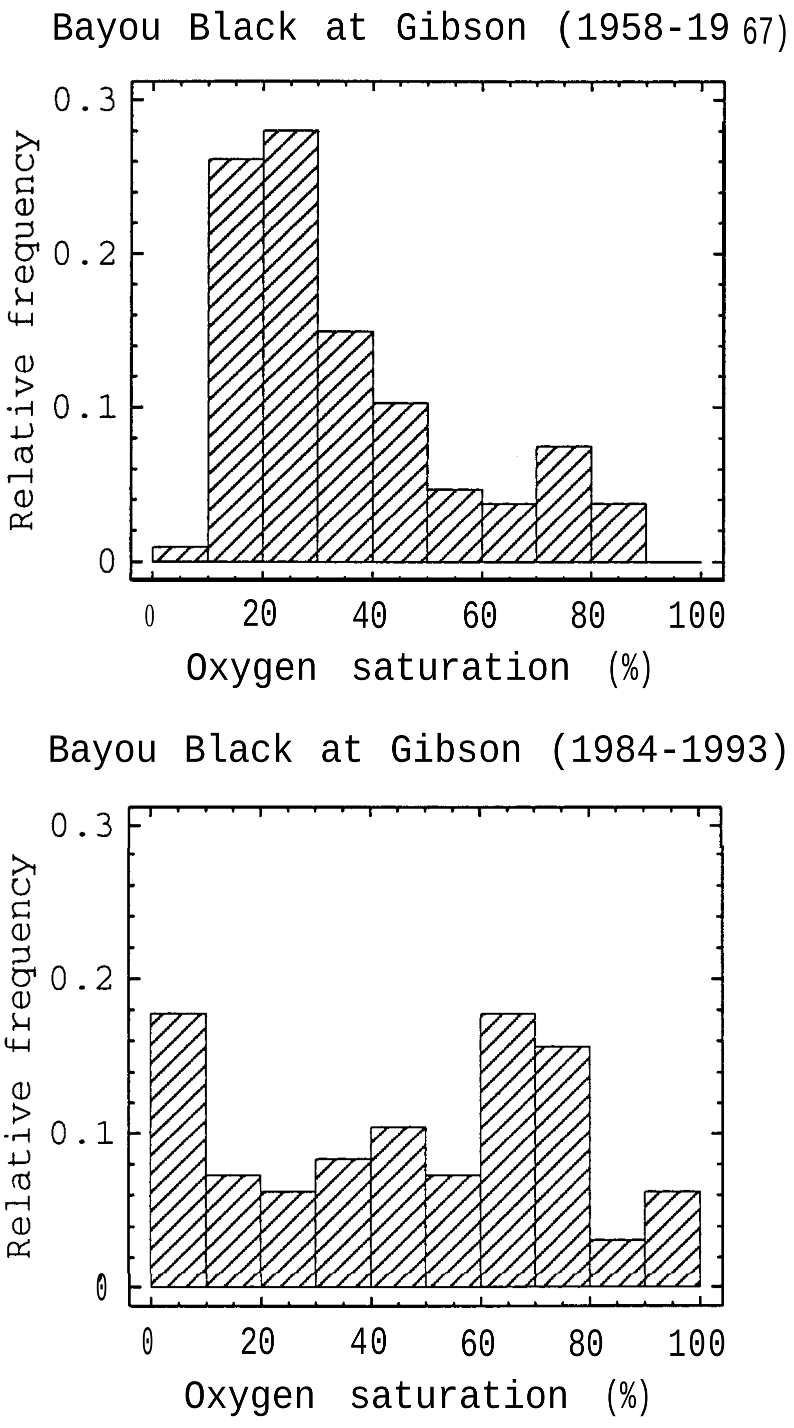


Figure 29. Frequency distribution of oxygen saturation for Bayou Black at Gibson for 1958-1967 (upper panel) and for 1984-1993 (lower panel).

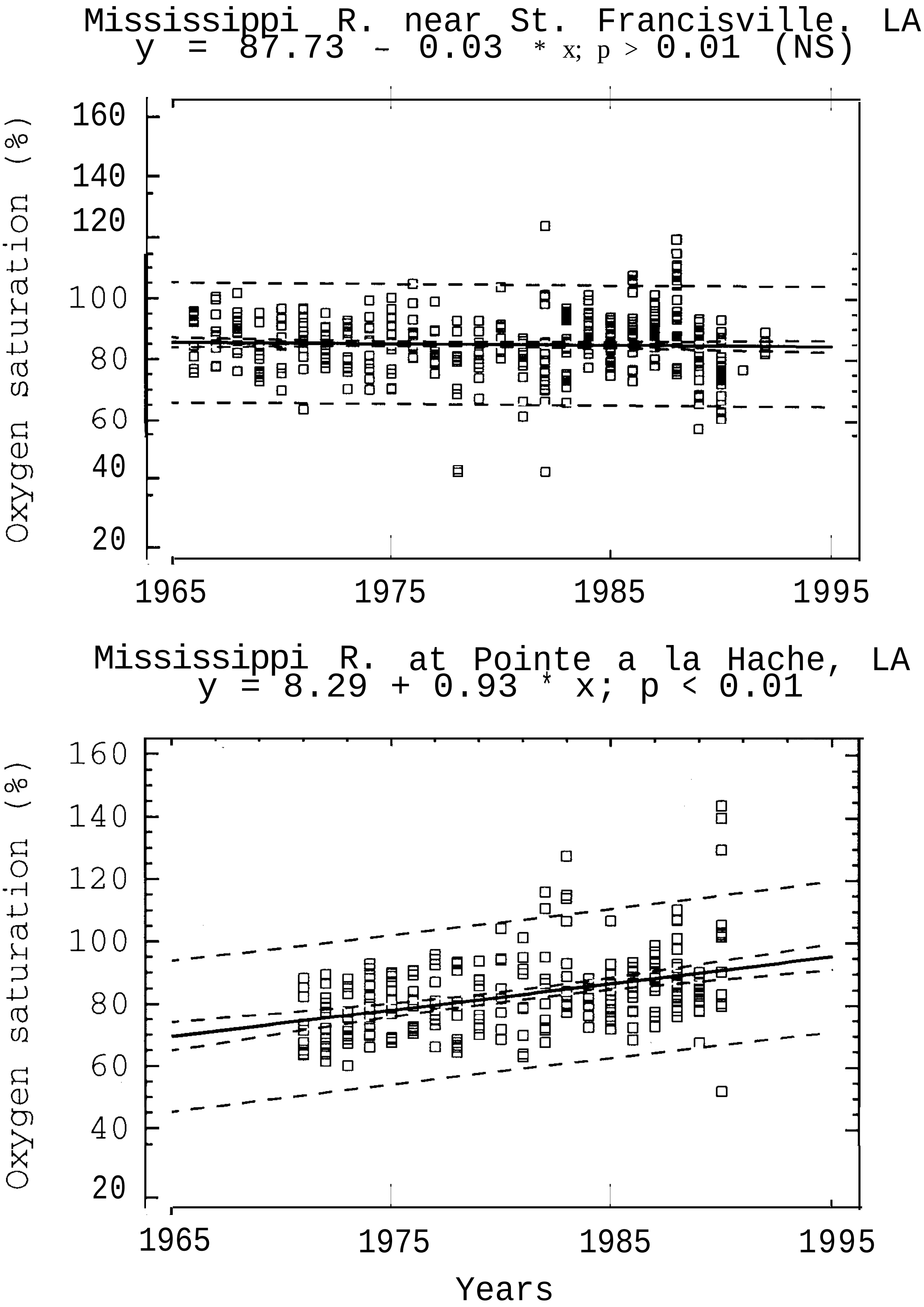


Figure 30. Trends in surface oxygen saturation for two stations on the lower Mississippi River, St. Francisville and Pointe a la Hache.

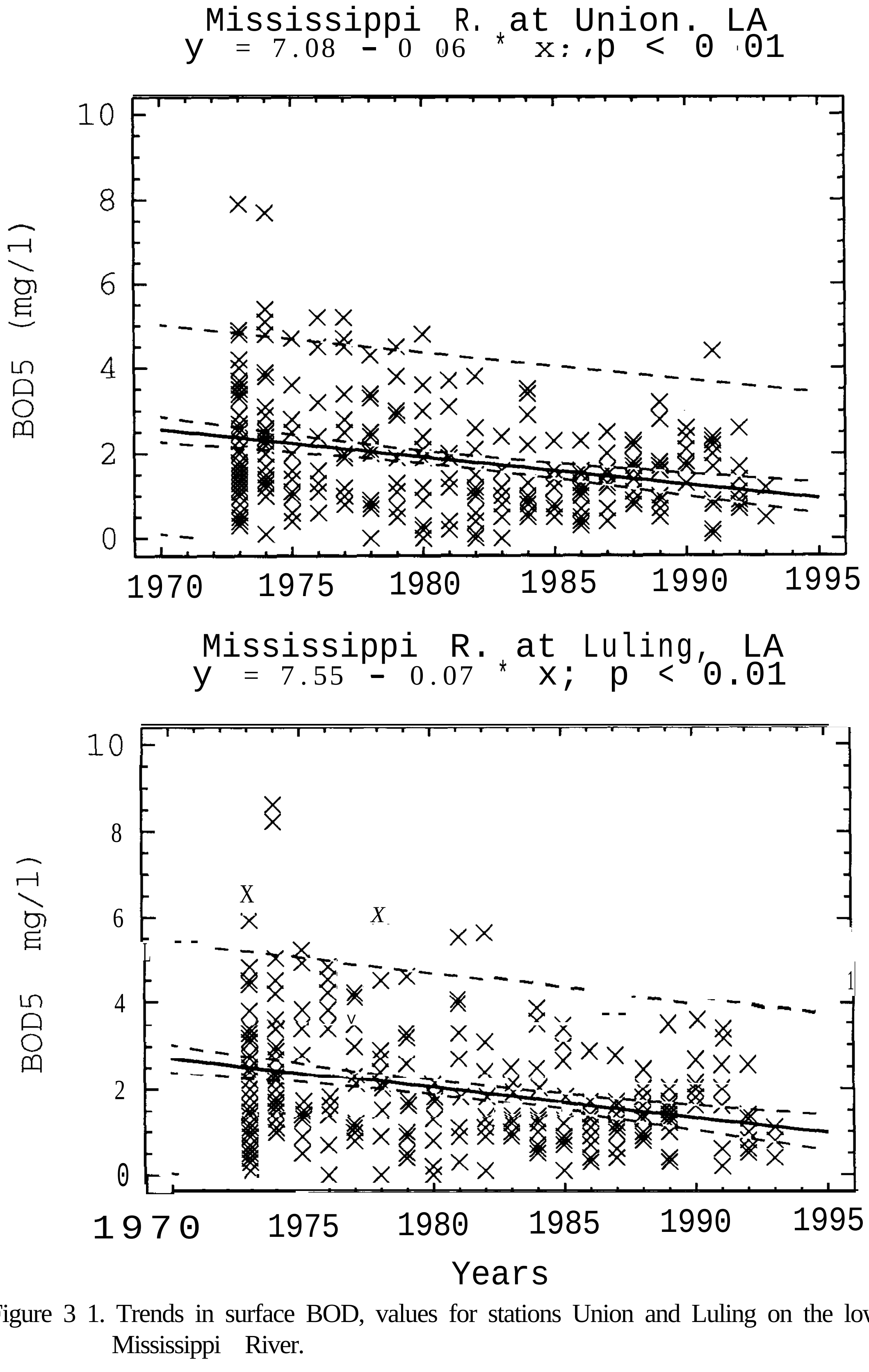


Figure 3 1. Trends in surface BOD, values for stations Union and Luling on the lower Mississippi River.

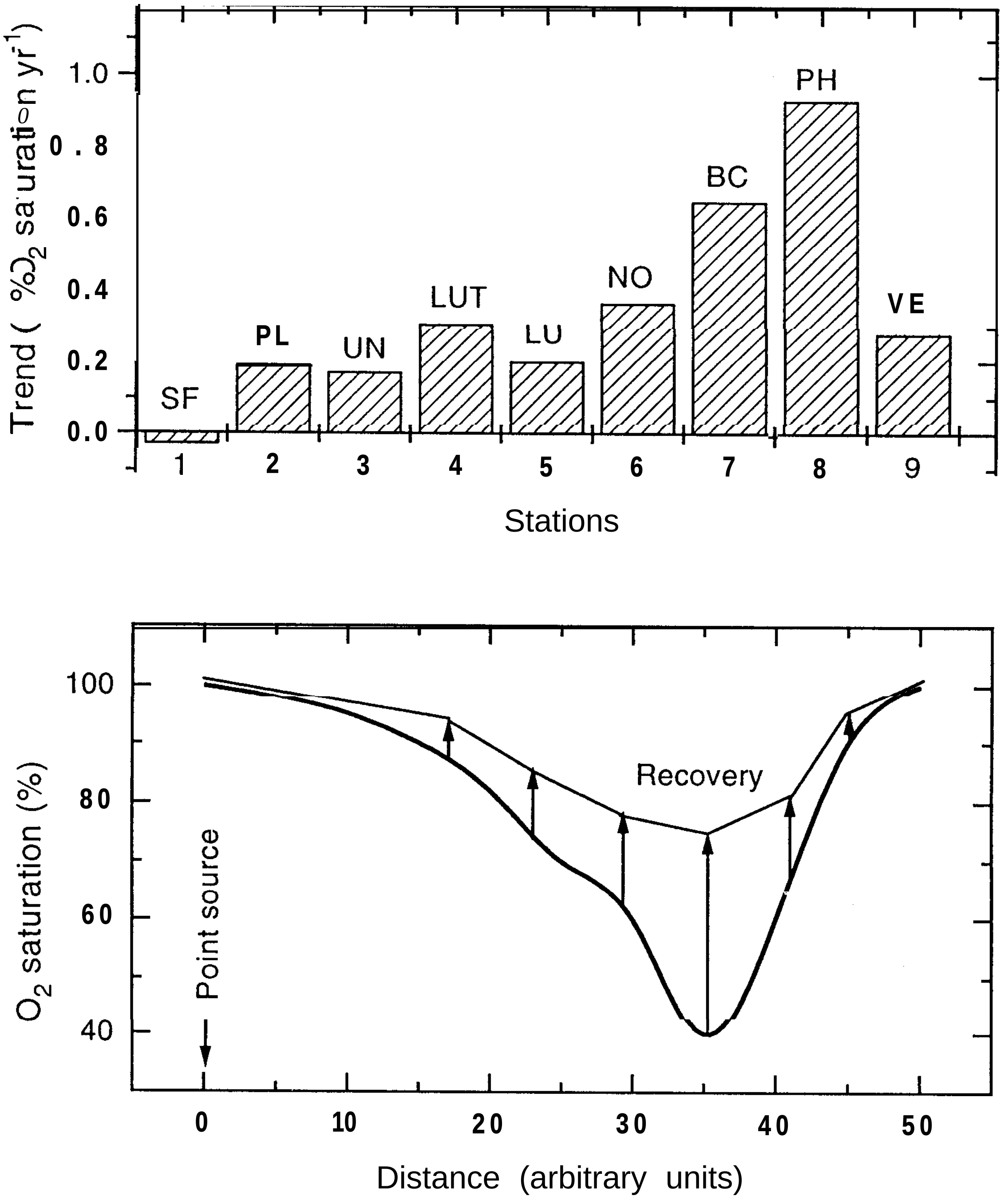


Figure 32. Comparison of trends in surface oxygen saturation for selected stations on the lower Mississippi River (upper). The letter codes at the top of the vertical bars refer to the stations St. Francisville, Plaquemine, Union, Lutcher, Luling, New Orleans, Belle Chase, Pointe a la Hache, and Venice. The observed differences in oxygen saturation trends among stations may be a consequence of reduction in organic loading from major point sources along the river (lower). See text for further explanation.

1974 (Adkins and Bowman 1976). No values of dissolved oxygen (1 ft below the surface) less than 2.5 mg l^{-1} were recorded; the 2.5 mg l^{-1} value was near Terrebonne Bayou in a closed canal. Williams (1956) examined water quality in the Grand Bayou Blue section of upper Timbalier Bay in December 1954–August 1955. Dissolved oxygen values (taken 1 m below the surface) were very low in August. All of the stations in the Williams (1956) study were in dredged canals or narrow bayous where flow is expected to be reduced and stagnation may occur during hot weather. Near bottom–water oxygen concentrations substantially below 2 mg l^{-1} were found in some canals adjacent to a produced water effluent (Boesch and Rabalais 1989b, Rabalais et al. 1991b); dissolved oxygen levels returned to ambient levels at 250–500 m from the point of lowest oxygen concentrations.

Seaton (1979) profiled dissolved oxygen values for 24 stations in the Barataria basin quarterly from August 1976 through August 1977. Of these stations, only those in the northernmost part of the basin (6 of 7 stations from Lac des Allemands and northward) displayed dissolved oxygen values below 2 mg l^{-1}. Low oxygen values were usually in the lower water column, but for some, very shallow stations were evident in the upper water column. Frequency of occurrence of values below 2 mg l^{-1} ranged from 20% to 60% for the six stations in upper Barataria that experienced hypoxia (Seaton 1979).

Dissolved oxygen measurements were made in the eastern portion of Timbalier Bay in conjunction with the Offshore Ecology Investigation (Price 1979). Hydrographic profiles were conducted along a transect from the upper bay to the intersection with the Gulf of Mexico from August 1972 to June 1974. Mean dissolved oxygen concentrations ranged from slightly less than 6 mg l^{-1} to almost 11 mg l^{-1} except for the July 1973 cruise when the mean value was near 3.6 mg l^{-1}; these low oxygen levels coincided with the flood of the Mississippi River in 1973 (Price 1979). The highest readings, $\geq$13 mg l^{-1}, occurred in the winter months; the lowest was 1.3 mg l^{-1} in July 1973.

Oxygen-depleted bottom waters are seasonally dominant features of the Louisiana and Texas continental shelf adjacent to the deltas of the Mississippi and Atchafalaya rivers (Rabalais et al. 1991a). The areal extent of bottom-water hypoxia in mid-summer may cover up to 9,500 km^2 with the spatial configuration varying interannually. Bottom waters severely depleted in oxygen often infringe on the seaward boundary of the Barataria and Terrebonne estuaries (figure 33). The movement of these waters onshore as a result of wind shifts is often the cause of massive fish kills. More frequent sampling off Terrebonne Bay indicates that hypoxic bottom waters form as early as February and persist as late as October, with widespread, persistent, and severe hypoxia/anoxia from mid-May to mid-September (figure 34).

Conclusions

(1) Based on surface oxygen saturation during the period 1990–1993 for the 27 studied localities in the Barataria-Terrebonne estuarine system, 2 stations (7% of sample) are classified as oligotrophic waters (class I). Eleven stations (11%) belong to the β–mesosaprobic class (class II); 8 (26%) belong to the

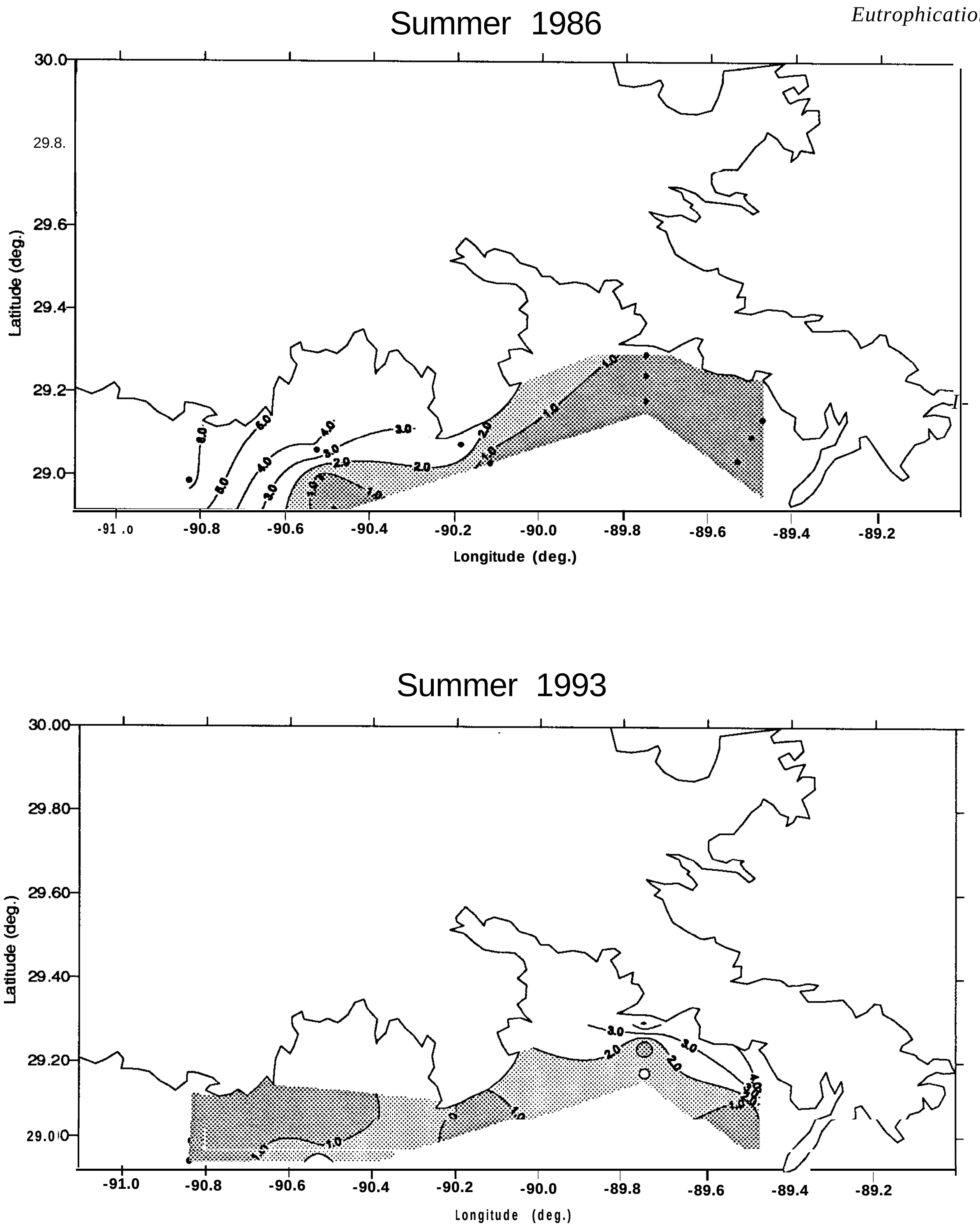


Figure 33. Distribution of dissolved oxygen in nearshore coastal waters mid-summer for July 1986, and July 1993.

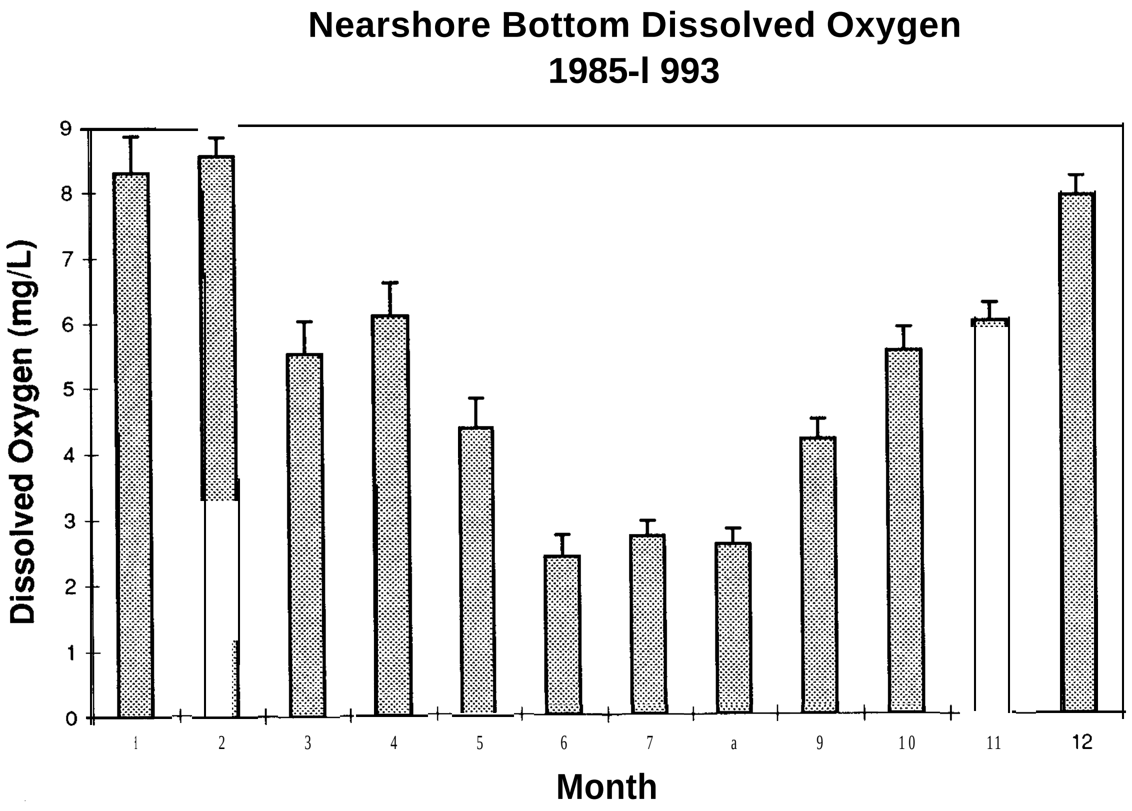


Figure 34. Long-term seasonal averages for bottom water dissolved oxygen concentrations in nearshore coastal waters, primarily off Terrebonne Bay, but across the study area (see figure 2) for 1985-1993.

"–mesosaprobic waters (class III); and 15 stations (56%) have the poorest water quality in this classification and belong to the polysaprobic class (class IV).

(2) Based on surface oxygen BOD_5 values for the period 1990–1993, the seven studied localities in the Barataria-Terrebonne estuarine system are nearly equally divided between the \$–mesosaprobic class (class II) and the "–mesosaprobic class (class III); three stations (43%) and four stations (57%), respectively.

(3) Several Barataria-Terrebonne estuarine stations show an increasing trend in surface oxygen saturation during the period 1955–1993. This suggests that there has been an overall reduction in organic loading. In the case of Bayou Black, however, the frequency of hypoxic events has increased in spite of the increasing trend in oxygen saturation. Also, in spite of the increasing oxygen saturations, 15 of the analyzed 27 localities still exhibit < 30% saturation in over 50% of the cases. Thus, it appears that heterotrophic processes still predominate over autotrophic processes, and the recovery rate is slow.

(4) Stations in the lower Mississippi River below St. Francisville show a highly significant trend of increase in surface oxygen saturation for the period 1970–1993. This coincides well with the overall reduction in BOD_5 values in the lower Mississippi River. The highest oxygen recovery rate is observed at two stations downstream from New Orleans, Belle Chase, and Pointe a la Hache. This suggests that observed changes in the lower Mississippi River may largely be because of better efficiency of sewage treatment plants, especially in the New Orleans area.

(5) Historical and limited data for bottom-water dissolved oxygen indicates that oxygen depletion is likely to occur in poorly flushed environments, in deeper channels and bayous where stratification may develop, in canals adjacent to produced water discharges, and in water bodies receiving high chemical oxygen demands or organic loading from sewage or other wastewater outfalls.

Toxic and Noxious Phytoplankton

Introduction

Species Composition

In freshwater systems, the phytoplankton assemblage itself is indicative of certain types and severity of eutrophication. In marine and estuarine systems, perhaps because eutrophication of these areas is a more recent phenomenon, changes in phytoplankton species composition indicative of a change from naturally productive to highly eutrophied have not yet been determined. The limited phytoplankton data for the Barataria-Terrebonne estuarine system are concentrated in the marine and estuarine areas. It is qualitatively similar to that from other nearby regions (Housley 1976; Maples 1982, 1983; Fay and Schnitzer 1984; Simmons and Thomas 1962), but these also may be highly eutrophied. There are insufficient species data for the freshwater areas to assess their trophic status. There is an extensive USGS data set, collected from 1974 to 1982, but the data are only reported to genus level and are thus unsuitable for this analysis.

Toxic and Noxious Phytoplankton

Blooms of toxic and noxious phytoplankton, or "red-tides" as they used to be called, are natural phenomena that may be exacerbated by several human activities such as nutrient pollution, aquaculture, and shipping (Hallegraeff 1993). Such blooms can have various impacts from human illness and death—from consumption of primarily shellfish contaminated with toxins produced by certain algae—to mortality of other organisms at higher trophic levels, including commercially important species, to loss of recreational and aesthetic value because of water discoloration and unpleasant odors. The economic impact of these problems can be substantial and sometimes regionally devastating (e.g., Shumway 1990, Taylor 1993). Worldwide, phytoplankton from disparate taxa can cause toxic and noxious algal blooms, but dinoflagellates are the most frequent culprits.

In Louisiana there have been no known human health problems from consumption of algal toxins in shellfish or fish. In fact, it has generally been believed that the low salinity of Louisiana estuaries prevents the growth of all toxic algal species. However, there are published accounts of red tides in the region (Perry et al. 1979, Eleuterius et al. 1981, Maples et al. 1981, Perry and McLelland 1981). Further, increasing coastal eutrophication has been hypothesized to lead to increases in toxic and noxious algal blooms (Smayda 1989a, b; Shumway 1990; Hallegraeff 1993). Nutrient inputs to the shelf adjacent to the estuarine area have increased since the 1950s (Turner and Rabalais 1991a) to result in enhanced eutrophication (Turner and Rabalais 1994). Indirect evidence suggests that similar increases have occurred in the estuaries; although, recently nutrient concentrations have remained constant. Thus toxic and noxious algal blooms may be more likely now than in the past.

Data Sources

Historical

Historical data on phytoplankton abundance are quite limited. Some data are available only as species lists (Day et al. 1973, Conner et al. 1987) for which the original data, sampling locations, and dates are no longer available. Other data are available in a form that cannot be compared with the rest of the quantitative data because chains rather than individual cells were counted (Green 1975). However these data, plus two studies with some abundance data (Hart 1979, Fucik and El-Sayed 1980), can be compiled into a species list to determine which species were present in the past (table 12). Two studies contain semi-quantitative abundance data (table 13); only the most abundant species are listed (Fucik and El-Sayed 1980, Hart 1979) and in one case the data are averaged over the entire study period (Hart 1979). The original raw data are available in neither case.

Recent

Data collected between 1989 and 1994 are available for three areas (table 14): Fourleague Bay, Bayou Little Caillou in the Terrebonne Bay estuary, and the continental shelf adjacent to the estuarine system area out to 10-m water depth (=Offshore, figure 35). In Fourleague Bay samples were taken from three to six stations on six dates in 1990 and 1991. The sites were chosen to cover the full range of salinities observed on each date (Dortch and Madden unpublished). In the Terrebonne Bay estuary the same three sites were sampled weekly during 1993 and 1994 (figure 36). The sites are located over oyster reefs and were chosen because they represent a range of salinities and distance from the open bay (Dortch and Soniat unpublished). Both estuarine sampling sites are extremely environmentally variable and presumably represent areas with low- water residence times. Offshore sampling occurred sporadically from February through November with the greatest number of samples in April (n=83) and July (n=128).

Phytoplankton in the water were preserved in 0.5% glutaraldehyde and refrigerated for 1–24 hr. The samples were size fractionated by filtration onto 0.2, 3, and 8 µm polycarbonate filters, with 0.03% proflavine hemisulfate used to stain the latter two fractions, and the filters were mounted in immersion oil (Murphy and Haugen 1985, Shapiro et al. 1989). The 0.2–3 µm fraction was counted immediately, the 3–8 µm fraction was counted immediately, if possible, and, if not, refrigerated and counted as soon as possible. The > 8 µm fraction was frozen and counted later. All samples were counted using an Olympus BH2-RFCA epifluorescence microscope with blue and green excitation light and as necessary transmitted light. The phytoplankton were identified to the nearest possible taxon.

Assessing Potential Problems

Determining which species are a potential threat is difficult because the list of known or suspected toxic and noxious phytoplankton grows yearly, the taxonomy is continuously undergoing revision, and an organism that causes a problem in one place may not in another. Further, the information is not housed in a single location. Consequently, potential impacts of species observed in the Barataria-Terrebonne estuarine system are summarized in table 15 and detailed documentation of sources is provided in appendix B. Table 15 contains only those species for which there is some documented evidence that they are a human health threat, can cause mortality of

Table 12. Historical data on presence of toxic and noxious phytoplankton species, based on species lists and quantitative data.

Location		Barataria Bay Area	Laurier Bay Leeville field Bayou Ferblanc	Lac des Allemands Lake Salvador Lake Cataouatche	Airplane Lake	Offshore	Offshore
Dates		Before 1987	April, 1977– April, 1978	Before 1987	Before 1987	May,1973– Mar, 1974	June,1972– Jan., 1974
Salinity Range (ppt)		NA	13–19	0–10?	6?–22?	8–40?	15–33
Taxon	*Alexandrium monilatum*	*				*	*
	Anabaena flos-aquae			*			
	Ceratium spp.		*			*	
	C. furca					*	
	C. fusus	*					*
	C. hircus	*	*		*	*	*
	C. trichoceros	*					
	C. tripos		*				
	C. vulgar	*					
	Dinophysis sp.		*				
	D. caudata	*	*				
	Gonyaulax sp.		*		*	*	*
	G. diegensis						*
	G. fragilis						*
	G. polygramma						*
	G. turbynei						*
	Gymnodinium brevis				*[1]		
	G. sanguineum		*			*	
	Oscillatoria sp.					*	
	Prorocentrum sp.		*				
	P. compressum		*		*	*	*
	P. gracile	*					*
	P. maximum	*			*		
	P. micans		*		*		
	Pseudo-nitzschia spp.	*	*			*[2]	*
	Scrippsiella trochoidea						*
Source		Day et al. 1973	Hart 1979	Conner et al. 1987	Conner et al. 1987	Green 1975	Fucik & El-Sayed 1980

[1]Identification uncertains since it is never found at salinities <21E/oo and the optimum for growth is 33–35E/oo (Aldrich and Wilson 1960).

[2]Listed only as *Nitzschia*. Green (pers. comm.) identified as *N. pungens*, now classified as a species of *Pseudo-nitzschia*.

Table 13. Historical data on abundance of toxic and noxious phytoplankton species.

Location	Laurier Bay		Leeville Oil Field		Bayou Ferblanc		Offshore
Dates	May 1977–April 1978[1]		May 1977–April 1978[1]		May 1977–April 1978[1]		June 1972–Jan. 1974[2]
Salinity (ppt)	13 ± 1.1		15.0 ± 1.0		19.0 ± 0.9		15–33
Abundance	Mean (Cells/L)	% Rel	Mean (Cells/L)	% Rel	Mean (Cells/L)	% Rel	% Freq
Taxon							
Alexandrium monilatum							5
Ceratium fusus							5
Gonyaulax sp.							10
G. diegenesis							5
G. fragilis							5
G. polygramma							5
G. turbynei							10
Prorocentrum compressum	1.19×10^5		8.4×10^4		1.3×10^5		25
Prorocentrum gracile							5
Prorocentrum micans	7.0×10^4		7.7×10^4		8.5×10^4		
Pseudo-nitzschia spp.		1.8		2.0		1.4	65
Scrippsiella trochoidea							5
Source	Hart 1979		Hart 1979		Hart 1979		Fucik and El-Sayed 1980

1 Two stations at each site, sampled monthly. Abundance reported only for species present in 1/3 of samples. % Rel=abundance relative to diatoms.

2 Two stations sampled twelve times during period, data only for 10 most abundant species. Slightly outside of offshore BTNEP region, but only historical quantitative data for offshore region. Percent frequency (% Freq) = # samples with organism/total # samples.

Table 14. Current status of toxic and noxious phytoplankton in the BTNEP area.

Taxon	Offshore <10 m		Bayou Little Caillou Terrebonne Bay Estuary		Fourleague Bay	
	Max # (cells/L)	Freq[1] (%)	Max # (cells/L)	Freq[1] (%)	Max # (cells/L)	Freq[1] (%)
Alexandrium monilatum	3.16×10^4 $>10^6$	0.52 S[2]	0	0	0	0
Ceratium spp.	2.60×10^5	31.7[3]	1.35×10^4	10.2[4]	4.00×10^2	3.8[5]
Dinophysis caudata	6.21×10^4	20.9	0	0	0	0
D. ovum	2.71×10^4	3.1	0	0	0	0
Gonyaulax spp.	8.40×10^3	2.9	0	0	0	0
Gymnodinium sanguineum	2.03×10^4	12.0	7.08×10^4	31.0	3.28×10^4	30.8
Heterosigma cf. *akashiwo*	9.82×10^5 $>10^6$	1.0 S[2]	8.12×10^3	0.5	0	0
Lingulodinium polyedra[6]	$>10^6$	S[2]	0	0	0	0
Noctiluca sp.	1.64×10^3	0.5	2.98×10^5	S[2]	0	0
Oscillatoria spp.	1.11×10^8	6.8	7.86×10^5	0.5	nd	nd
Prorocentrum compressum	4.06×10^5	50.0	2.86×10^2	0.5	1.00×10^3	15.4
P. micans	3.25×10^4	20.2	1.35×10^4	6.5	1.00×10^3	7.7
P. minimum	5.34×10^4	2.9	2.03×10^4	3.2	3.25×10^3	15.4
Pseudo-nitzschia spp.	1.02×10^8	50.5	1.09×10^5	16.2	1.71×10^4	11.5
Scrippsiella cf. *trochoidea*	2.67×10^5 $>10^6$	1.3 S[2,6]	3.27×10^4	0.5	0	0
Number of Samples	382		216		26	
Sampling Period	4/12/89–9/13/94		1/28/93–4/1/93 and 8/26/93–11/17/94		1/25/90–8/22/91	
Sampling Interval	Sporadic		Weekly		Seasonally	
Salinity Range (ppt)	0–36		0–23		0–20	

[1]% Frequency (% Freq) = # samples with organism/total # samples x 100
[2]S=sporadic bloom with discolored water
[3]*C. furca, fusus, pentagonium, trichoceros, tripos*
[4]*C. furca, fusus, pentagonium*
[5]*C. furca*
[6]Slightly outside of BTNEP area, but a potential threat

Table 15. Potential impacts of toxic and noxious algal species observed in recent or historical samples from BTNEP area, based on data from other areas (see appendix B). A ? on potential impacts indicates that information for this taxon is not conclusive. A ? on cyst formation indicates that data for this taxon are uncertain or there is no information for this species, but others in genus do form some type of cyst.

Taxon	Potential Impacts					Forms Cyst
	Human Health	Higher Trophic Level			Discolored Water	
		Shellfish[1]	Fish	Other		
Alexandrium monilatum (=*Gonyaulax monilata*)		X	X	X	X	X
Anabaena flos-aquae				X		
Ceratium furca	DSP[2]?				X	
C. fusus		X				
C. tripos		X[3]				
Dinophysis caudata	DSP?					X?
Dinophysis ovum	DSP?					X
Gonyaulax polygramma		X[3]	X[3]	X[3]	X	X?
Gonyaulax spp.[4]	PSP[5]	X?	X?	X?	X?	X?
Gymnodinum breve (=*G. brevis*, =*Ptychodiscus brevis*)	NSP[6]		X		X	X
G. sanguineum (=*splendens*)		X	X		X	X
Heterosigma akashiwo			X		X	X
Lingulodinium polyedra (=*Gonyaulax polyedra*)	PSP?		X		X	X
Noctiluca spp.		X	X			
Oscillatoria spp.			X	X	X	
Prorocentrum compressum (=*Exuviella compressa*)	DSP?					X?
P. gracile					X	X
P. micans	PSP?				X	X
P. minimum	DSP? PSP? VP[7]	X	X		X	X?
Pseudo-nitzschia spp. (=*Nitzschia pungens*; (*N. seriata*)	ASP[8]			X		
Scrippsiella trochoidea (=*Peridineum trochoideum*)			X[3]		X	X

[1]Includes bivalves and shrimp
[2]Diarrhetic Shellfish Poisoning: gastrointestinal distress but not fatal; also a tumor promoter (Hallegraeff 1993)
[3]Problems due to low oxygen rather than toxins
[4]Without species identification not possible to asses impact; many of members of genus (now renamed), associated with PSP
[5]Paralytic Shellfish Poisoning: numbness, paralysis, respiratory failure and death (Hallagraeff 1993)
[6]Neurotoxic Shellfish Poisoning: respiratory irritation, tingling, weakness, cramps, not fatal (Hemmert 1975)
[7]Venerupin poisoning: liver injury and death (Okaichi and Imatomi 1979)
[8]Amnesiac Shellfish Poisoning: short-term memory loss, seizures, death (Hallegraeff 1993)

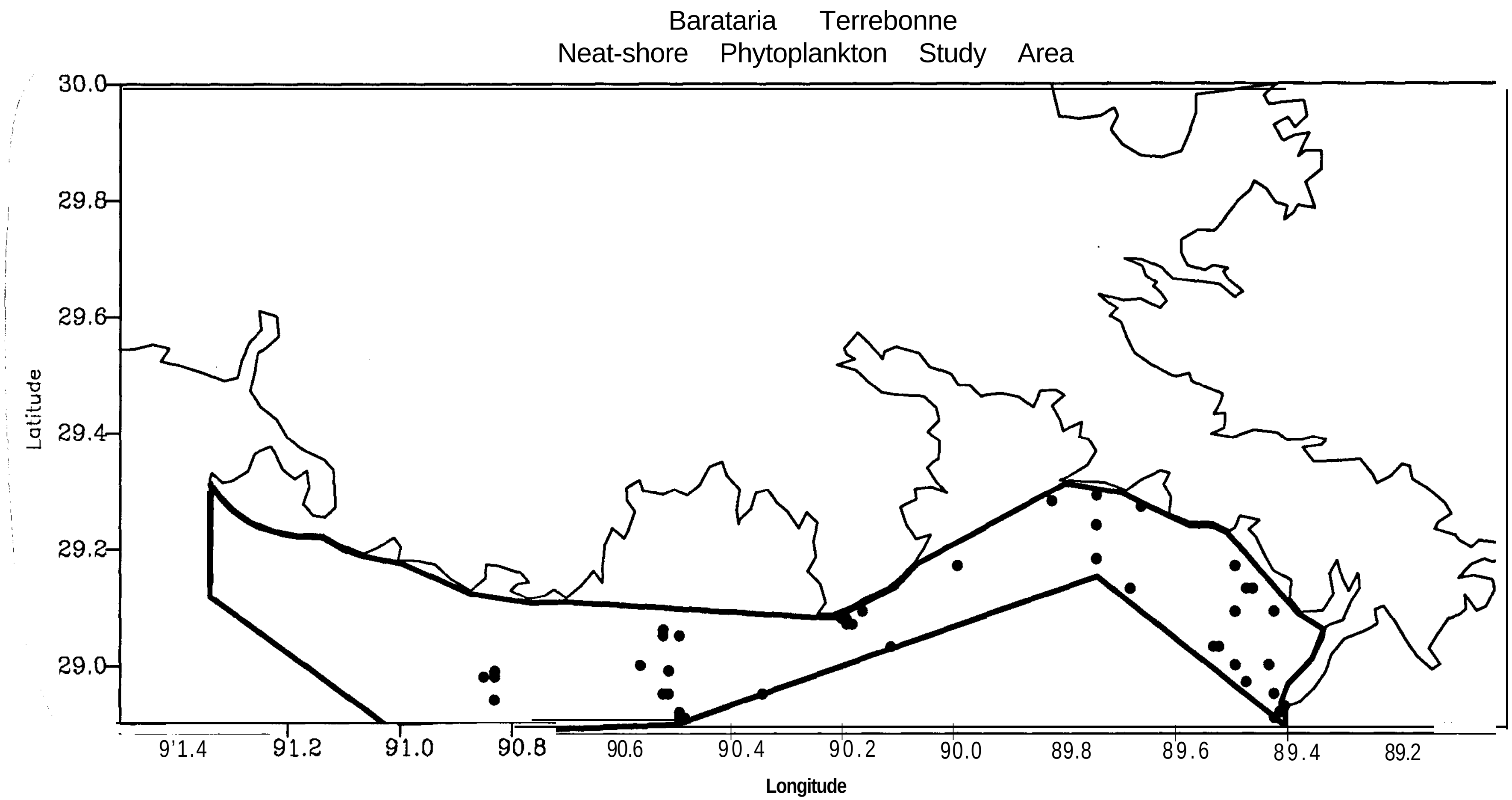


Figure 3 5. Outline of nearshore coastal portion of the Barataria-Terrebonne estuarine system with "Offshore" phytoplankton sampling stations indicated.

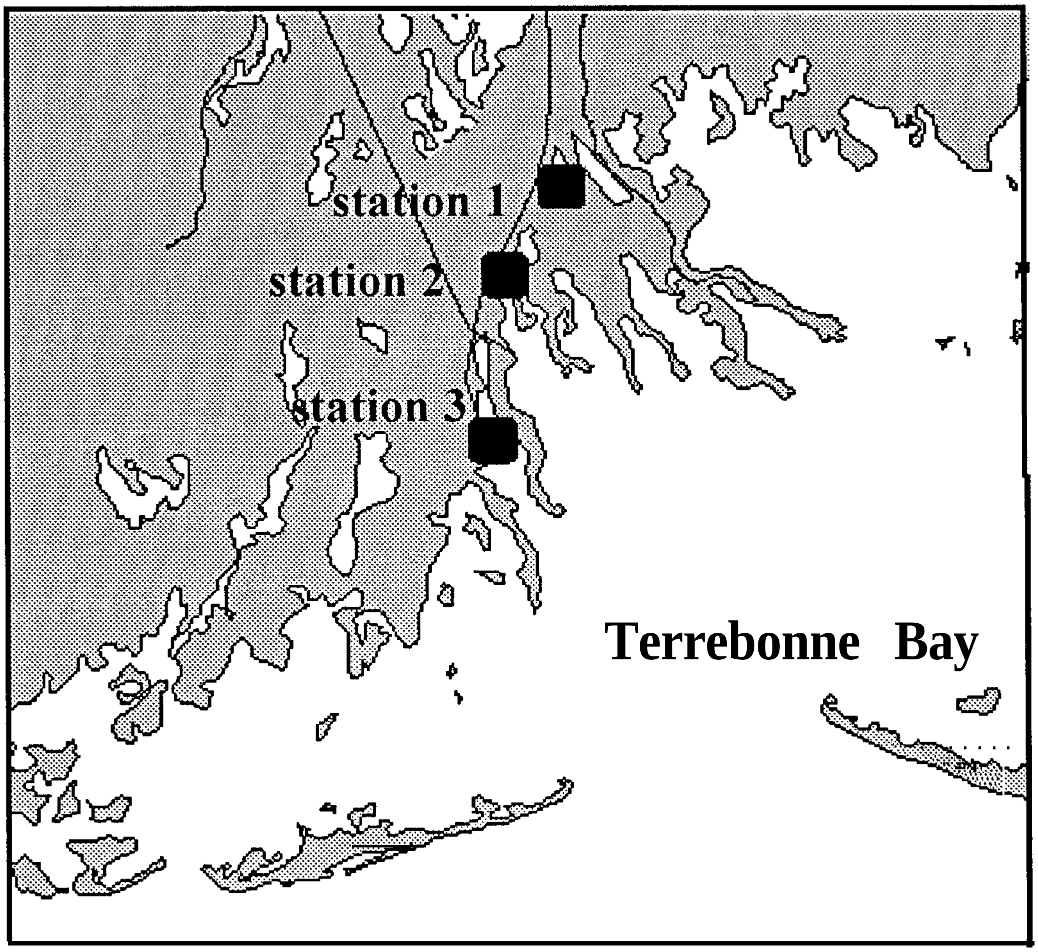


Figure 36. Location of three sampling stations along Bayou Petit Caillou for phytoplankton species composition.

organisms at higher trophic levels, or cause severe episodes of water discoloration. Tables 12–14 also contain data on related species (same genus) for which there are no documented problems because of the uncertainties in determining exactly which are problem species.

Some taxa have been left out of tables 12–15 that also might be considered toxic and noxious phytoplankton:

(1) *Skeletonema costatum,* which is the numerically most abundant diatom on the shelf (Dortch unpublished), makes up the bulk of the phytoplankton sinking to the bottom and may be one of several major causes of hypoxia on the shelf (Dortch et al. 1992). It is also abundant in all estuarine samples. However, it is a normal component of the plankton, and neither causes water discoloration nor is toxic.

(2) The numerically most abundant phytoplankton, especially in the summer, are small coccoid cyanobacteria, both on the shelf (Dortch 1994) and in the estuaries (Mire et al. 1995, Dortch and Madden unpublished). There is a growing awareness that cyanobacteria produce toxins, but to date there is no evidence that the small coccoid cyanobacteria are a problem.

(3) A number of tiny phytoflagellates have caused serious problems with water discoloration, fish kills, mortality of invertebrates, and in some cases die-offs of seagrasses in Narragansett Bay, Long Island Sound, Laguna Madre, and Scandinavian coastal waters (Cosper et al. 1990, Graneli et al. 1993, Stockwell et al. 1993). The traditional counting methods used for the Barataria-Terrebonne estuarine system historical data would not have counted them. The method used for the current data counts them in broad categories because they are extremely difficult to identify to the species or in some cases genus level. However, if numbers in the broad categories had indicated a bloom, further taxonomic study would have been made. Thus, these organisms have not been a threat in this area; although, they may have been present in low numbers.

Occurrence of Toxic and Noxious Phytoplankton

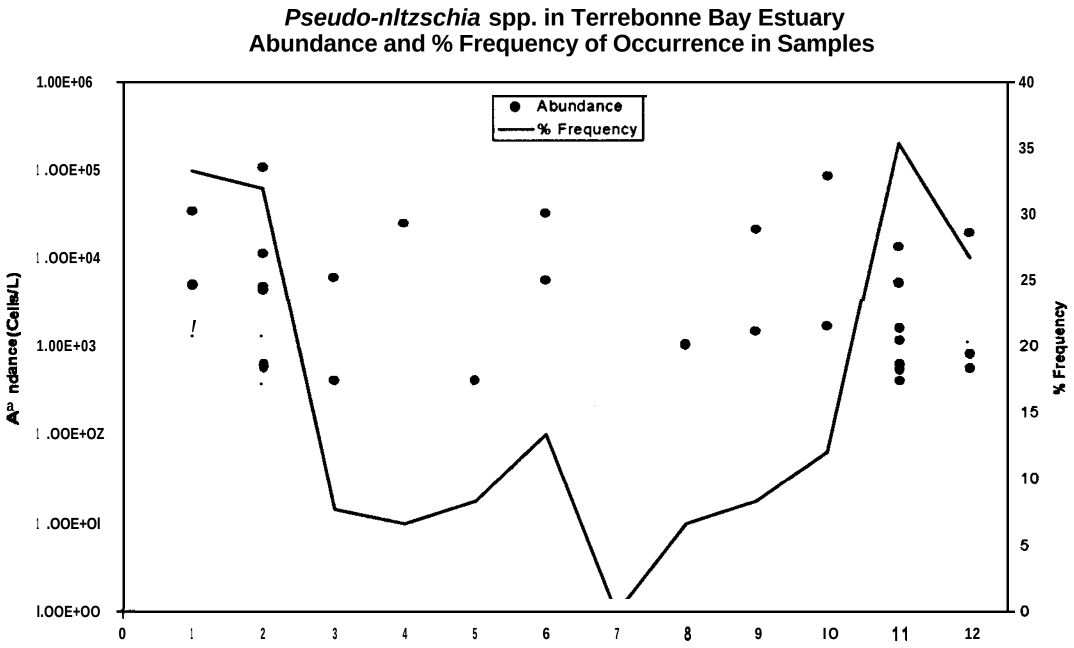
Historical data (tables 12 and 13) indicate the presence of a variety of potentially toxic and noxious algal species within the Barataria-Terrebonne estuarine system as early as the 1970s. These include species that can have human health impacts, affect organisms at higher trophic levels, and cause water discoloration (table 15). These data do not represent a complete listing of species present in the past because they focus only on the numerically most abundant organisms. Incomplete species lists and lack of abundance data make it impossible to determine in general if the number of species present or their abundance has increased.

It is evident that more potentially toxic and noxious phytoplankton were present in higher salinity waters than at very low salinities. Further, the presence of *Pseudo-nitzschia* spp., whose potential toxicity was unknown at the time of these earlier studies, was noted in most studies (tables 12 and 13).

A variety of potentially toxic and noxious phytoplankton is observed in Bayou Little Caillou in the Terrebonne Bay estuary and in Fourleague Bay (table 14). More species appear to be present in samples from Bayou Little Caillou, but this may be because the numbers of samples differ from each location. In contrast, a larger variety of species and higher abundance of most species are observed in the offshore zone. Some of the maximum concentrations on the shelf are extremely high. The greater variety of species and the much higher concentrations of most species on the shelf could be because most are transported into the estuary from the shelf or because the water residence times in the two estuarine areas are extremely short. Data are missing from larger water bodies, especially the open bays. Linking the nearshore waters of the continental shelf and the estuaries, these areas would have longer water residence time to allow blooms to develop during optimal conditions for a particular species over an extended period. Thus, the incidence of toxic and noxious phytoplankton blooms in the estuary may be underestimated by the data presented here.

Two taxa are especially abundant and occur frequently in the estuary and on the shelf: *Pseudo-nitzschia* spp., which causes Amnesic Shellfish Poisoning (ASP) a serious threat to human health (table 15); and *Gymnodinium sanguineum*, associated with fish kills and possibly problems with oyster recruitment (table 15). *Pseudo-nitzschia* spp. reach high concentrations in the estuary and on the shelf, but they are more abundant and occur frequently in samples on the shelf. On the shelf there is a strong seasonal cycle to the abundance with peaks in April and September (figure 37) and decreased frequency of occurrence in summer. In the estuary there is no seasonal variation in abundance; although, it also occurs less frequently in summer (figure 37). In contrast *G. sanguineum* occurs much more frequently in the estuary where it shows a pronounced seasonal maximum in summer (figure 38). The relationship between salinity and abundance for each species (Robichaux and Dortch submitted, Dortch et al. in prep) suggests that *Pseudo-nitzschia* spp. grow best on the shelf, whereas *G. sanguineum* may grow better in the estuary.

Several toxic and noxious phytoplankton species present in the Barataria-Terrebonne estuarine system (table 14) produce a cyst or resting stage (table 15). Cysts can sink to the bottom where they remain until conditions are suitable for growth. Re-seeding from cysts can occur in shallow estuarine or shelf waters. It provides a mechanism for retaining toxic and noxious species in an area where they have been recently introduced. For example, a bloom of *Heterosigma* cf. *akashiwo* in 1993, which can produce potent ichthyotoxins, was the first recorded occurrence of this species in this area. Because it produces a benthic resting stage (Tomas 1978, Imai et al. 1993), it will probably become endemic, and future blooms can be expected.



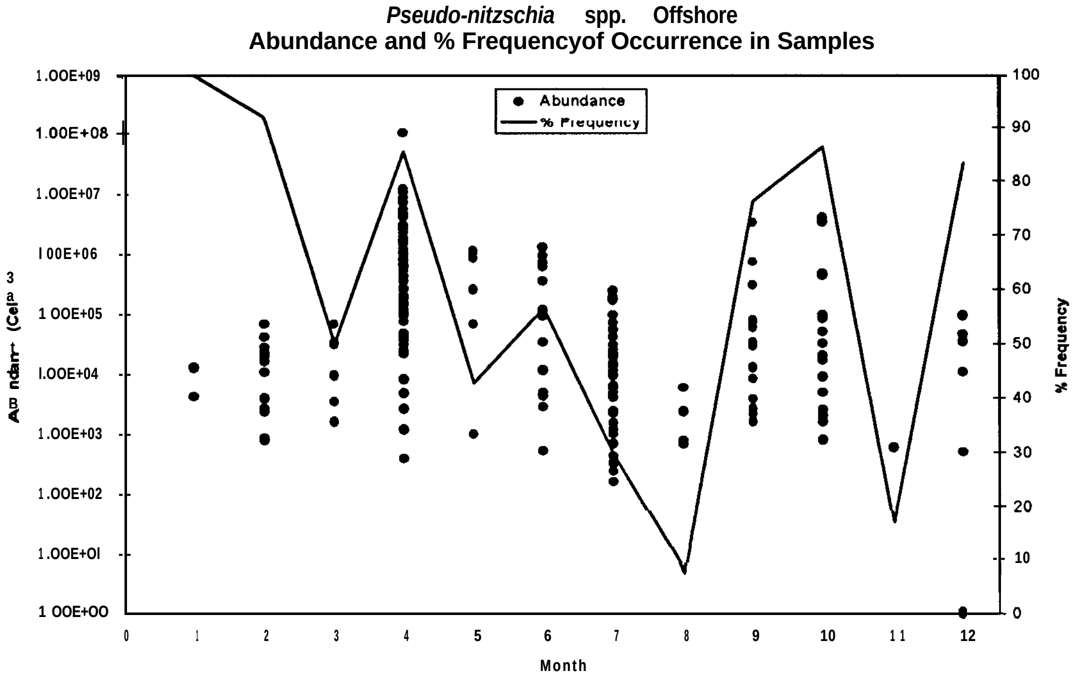
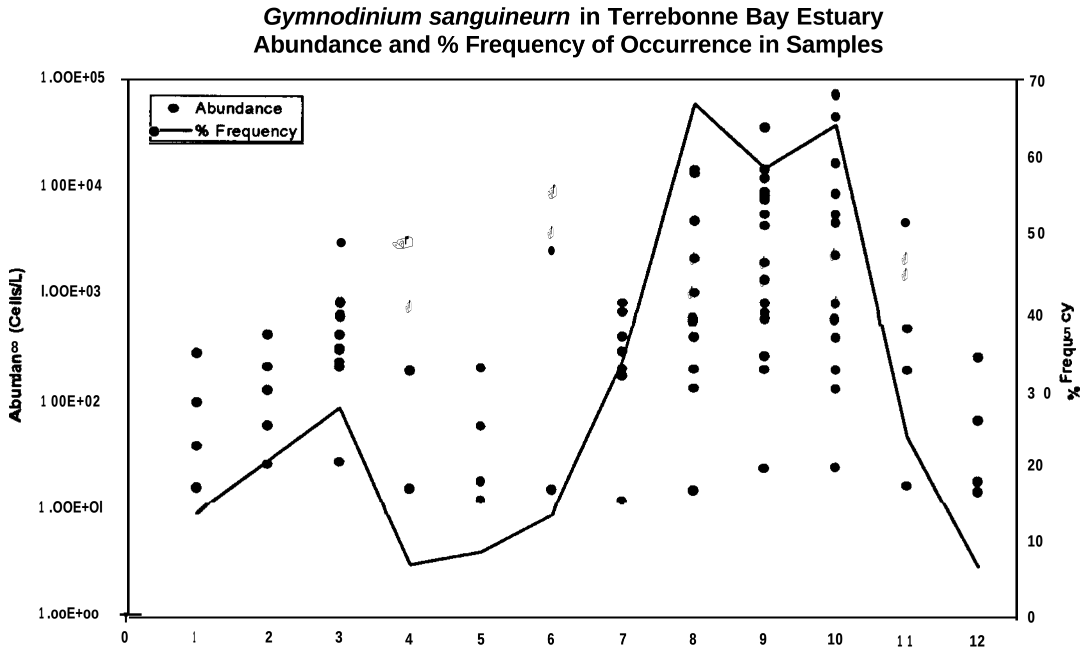


Figure 37. *Pseudo-nitzschia* spp. abundance and % Frequency of Occurrence (# samples with *Pseudo-nitzschia* spp. present/total # samples x 100) in Bayou Little Caillou in the Terrebonne Bay estuary and Offshore of the BTNEP area to water depths of 10 m. Note that one was added to all abundance data so that zero values could be plotted on a log scale.



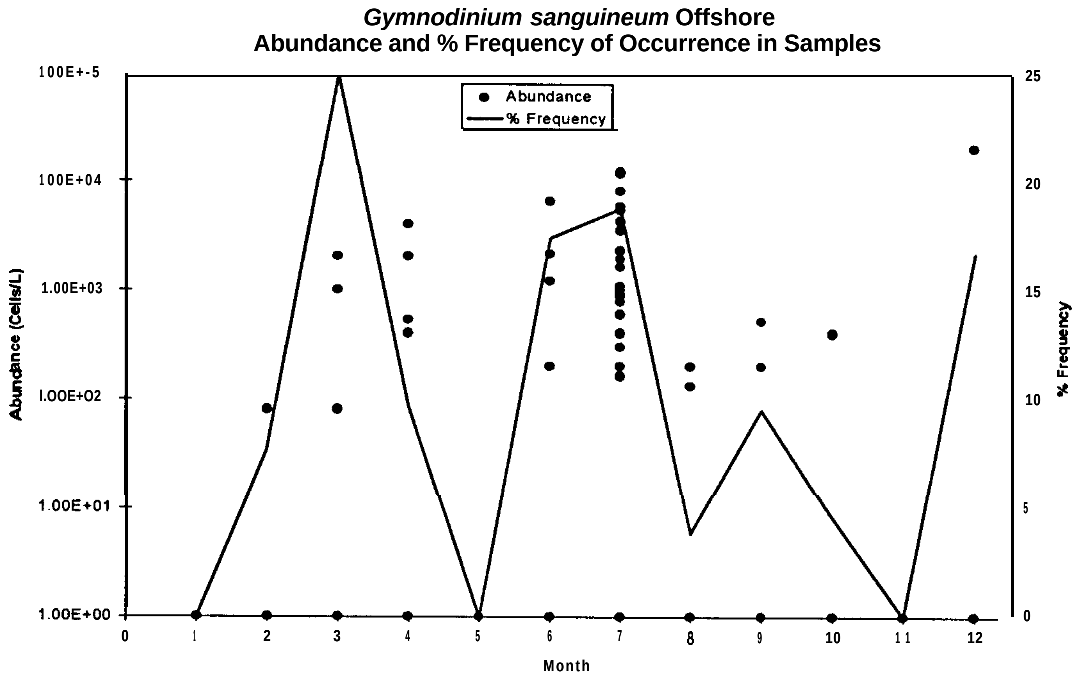


Figure 38. *Gymnodinium sanguineum* abundance and % Frequency of Occurrence (# samples with G. *sanguineum* present/total # samples x 100) in Bayou Little Caillou in the Terrebonne Bay estuary and Offshore of the BTNEP area to water depths of 10 m. Note that one was added to all abundance data so that zero values could be plotted on a log scale.

Documented Impacts

There have been no documented human health impacts. However, the toxins causing Diarrhetic Shellfish Poisoning (DSP) have been measured at low levels in oysters from Mobile Bay (Dickey et al. 1992), which is an estuary similar to the Barataria-Terrebonne estuarine system. The symptoms of DSP are similar to the gastrointestinal distress caused by more common viral pathogens; isolated incidents, however, could have been overlooked.

Water discoloration events have been documented for this region (table 14). The causative agents include *Alexandrium monilatum, Lingulodinium polyedra, Heterosigma* cf. *akashiwo, Noctiluca* sp., and *Scrippsiella* cf. *trochoidea*. Other events have occurred in similar nearby environments, involving *Gymnodinium sangineum* (Harper and Guillen 1989, Dortch and Robichaux submitted), *Prorocentrum minimum* (Maples et al. 1981, Perry and McLelland 1981), and *Oscillatoria erythraea* (Eleuterius et al. 1981). Some of these blooms have extended for long distances along and across the shelf and have been associated with fish kills (Harper and Guillen 1989, Robichaux and Dortch submitted). In most instances the highest numbers were recorded usually in nearshore areas of the shelf. Because data are completely lacking for the open bays of the study area, the extent to which these blooms are transported between bays and offshore is unknown. However, in several instances where a bloom was occurring offshore, high numbers of the same organism were observed well into the estuary on Bayou Little Caillou.

In the analysis of fish kills (see pg. 206), eight were related to algal blooms and low dissolved oxygen. No identification of the algae was made, so it is impossible to determine if species producing ichthyotoxins were present. Some species such as *Gymnodinium sanguineum* may be associated with low oxygen and also produce ichthyotoxins (Robichaux and Dortch submitted) so the cause of the fish kill is often difficult to determine.

The cause of another 12 fish kills was undetermined. Although these may be unrelated to toxic algae, algae can cause fish kills without being at high enough concentrations to discolor the water. A particularly insidious dinoflagellate, dubbed the "phantom killer," has recently been implicated in many unexplained fish kills in low-salinity, highly eutrophied water bodies (Burkholder et al. 1992). This organism is difficult to identify because (1) special techniques are required, (2) it is present on the fish or in the water for only a short time, (3) it is present in cysts in sediment, and (4) it has multiple and morphologically dissimilar life stages. Its occurrence in Louisiana waters is unknown, but has been associated with estuarine fish kills from Delaware to Alabama (Toffer et al. 1995).

Potential Impacts

No Increase in Nutrient Inputs

Nutrient levels in the Barataria-Terrebonne study area appear to have remained steady for the last 15 years (table 3), while other indicators of eutrophication appear to have increased (figures 22 and 23)**.** Even with no increase in nutrient inputs and eutrophication, this region is still at risk from problems caused by toxic and noxious phytoplankton. Assessing the degree of risk in detail requires more information than is currently available, but some problems appear more likely than others.

The greatest threat to human health is ASP. Concentrations of *Pseudo-nitzschia* spp. in the coastal zone off Barataria and Terrebonne bays are the highest ever observed; even the lower concentrations in the estuaries are high enough to cause problems elsewhere (Dickey et al. 1992, Garrison and Walz 1993). Not all species of *Pseudo-nitzschia* are toxic, but distinguishing species requires Scanning Electron Microscopy (SEM), a technique not routinely applied. One sample taken just outside the study area was examined by SEM and contained *P. pseudodelicatissima* (Dortch et al. in prep), which is sometimes toxic (Martin et al. 1990). Toxic *P. multiseries* and non-toxic *P. pungens* have been isolated in Galveston Bay (Fryxell et al. 1990). Thus, it is likely that toxic forms will occur in the study area at times. More information about its toxicity and population dynamics are required to determine if monitoring is necessary to insure public health. This genus has been a major component of phytoplankton in many areas for a long time with no problems just as it has been in the Barataria-Terrebonne area (tables 12 and 13). Since the first episode of ASP in 1987 with 107 people ill and 3 dead (Todd 1993), the toxins have been measured at levels that threaten human health and kill birds, fish, shellfish, and crabs, necessitating the closure of commercial and recreational harvesting of a number of species, on the east and west coasts of the United States and Canada (Villac et al. 1993 a, b). Because there has been no problem with ASP in this region historically, does not mean that one cannot develop.

DSP and the possibly related venerupin poisoning are other human health threats. None of the species known with certainty to cause DSP inhabit this region. Several species of both *Dinophysis* and *Prorocentrum*, which are observed in the study area, have been implicated in causing DSP; although, the link has not yet been demonstrated conclusively. Species implicated in DSP are more common in the offshore zone, to suggest that if a threat to human health exists, it will be from consuming oysters growing at the highest salinities.

Several species in table 14 have been linked to Paralytic Shellfish Poisoning (PSP, table 15), the most common and most serious toxic algal problem in other regions, both from human health and economic perspectives. The evidence is tenuous, and at this time PSP seems unlikely in this region. However world wide toxic species are suddenly appearing and causing severe problems in areas where they have never been seen. PSP has spread to many regions, and there is growing suspicion that cysts are being transported to new areas in ship ballast water (Hallegraeff 1993). Because LOOP is located just offshore of the Barataria-Terrebonne area and New Orleans is a major U.S. port, introduction of new species into this area by this means is not unlikely. Most ballast water introductions have involved PSP-causing organisms.

The potential impact on higher trophic levels consists of two types. The first involves the well-documented mass mortality of fish and other organisms as a result of blooms and toxin production or oxygen depletion. These types of events are already known to occur in this system, although underestimated, and they are likely to continue. Secondly, toxic species also affect ecosystem trophodynamics by reducing fecundity, survival, and recruitment and increasing mortality at all levels of the food web (Smayda 1992). These effects are not as readily apparent as mass mortality, and the impacts are generally unknown. For example, *Gymnodinium sanguineum* has been associated with lack of recruitment or mortality in juvenile oysters (Woelke 1961, Cardwell et al. 1979, Bricelj et al. 1992). A bloom of *G. sanguineum* during a critical period in the oyster life cycle could reduce yields of oysters with no obvious mass mortality event.

Increasing Nutrient Inputs

Despite high turbidity, phytoplankton growth in the Barataria-Terrebonne estuarine system may be nutrient limited, and increased nutrient input may enhance phytoplankton growth in general (see pg. xx). The most likely scenario for increasing nutrient inputs is diversion of fresh nutrient-rich Mississippi River water into the system. The impact will depend on the time, location, and magnitude—in terms of freshwater and nutrients—of the input. In general, unless the nutrient levels of the inflowing waters are lower than those in the estuary or the nutrient levels in the estuary are so high that nutrients are never limiting, increased incidence of toxic and noxious algal blooms can be expected. The worst-case scenario would be to create conditions within the estuary or a part of the estuary similar enough to those on the shelf that the types and numbers of toxic and noxious algal species in the estuary resemble those on the shelf (i.e., more species and much higher abundances). The potential human health and economic consequences would be magnified because of the proximity to commercially harvested shellfish and nursery grounds for many commercially and recreationally important species. For example, *Pseudo-nitzschia* spp. reach their highest levels every April on the shelf, probably in response to high river runoff, whereas levels are much lower in the estuary where there is no seasonal pattern to the abundance. Large inputs of nutrients and fresh water in the spring to parts of the estuary with long water residence times might result in blooms of potentially toxic *Pseudo-nitzschia* spp. similar to those now seen in the nearshore zone.

Ideally, nutrient inputs to the estuary should be minimized. However, with an adequate understanding of the phytoplankton dynamics in a particular area, provided by baseline studies, and if necessary continued monitoring of phytoplankton and toxin levels, flows could be managed to minimize the occurrence of toxic and noxious algal blooms in the estuary.

Recommendations

1. **Determine specific threats posed by toxic and noxious phytoplankton.** The analysis presented here points to several potential problems. It would be far better to know if these constitute real threats so serious human health and economic consequences can be avoided via appropriate monitoring and management. A history without problems related to toxic and noxious phytoplankton is not indicative of future developments, as has happened recently in many other areas of the world. This is especially true for areas that may be impacted by freshwater diversions. Two types of complementary information are necessary:

    a. Phytoplankton species distributions and an understanding of the environmental factors that regulate the occurrence of toxic and noxious phytoplankton

    b. Toxin analyses on oysters and other indicator species

2. **Identify algae associated with fish kills.** Algae can cause fish kills without being at high enough concentrations to discolor the water. Thus, the number of fish kills related to toxic and noxious algae may be underestimated. During investigations of fish kills where the cause is not immediately apparent, low oxygen is a suspected cause, or discolored water is present, samples for identification of phytoplankton should be taken as soon as possible using appropriate methods.

3. **Efforts to reduce the risk of introducing new species of toxic and noxious phytoplankton from ballast water must be undertaken.**

Introduction of new species via ballast water is a real threat. Of particular concern are ships entering Southwest Pass of the Mississippi River and because of prevailing water movements, seeding the entire Louisiana coastal zone. Studies are underway to develop practical means of treating ballast water to avoid seeding areas with new organisms. As soon as these methods are available, their use should be required because they may protect against importation of other exotic species as well as toxic algae. Meanwhile, there are voluntary conventions requesting that ships not take on ballast water in areas experiencing red tides. These measures may be ineffective; therefore, monitoring is essential, especially if any red tides are reported.

CONTAMINANTS

Introduction

Types of Contaminants

The types of toxic contaminants found in the Barataria-Terrebonne estuarine system include elements (especially metals), organometals (e.g., tributyltin), radionuclides, and a long list of organic contaminants. Important among the latter are the chlorinated aromatic compounds (including PCBs), the chlorinated hydrocarbons (including DDT and many other pesticides), and the polycyclic aromatic hydrocarbons (or PAHs). Factors that determine risk to people and the ecosystem include toxicity, concentration, bioavailability (a term describing to what extent organisms can take up these pollutants), and persistence (reflecting recalcitrance to environmental biodegradation either in the environment itself or by organisms). Environmental contaminants may be very stable, toxic at low concentrations, and bioavailable. Moreover, several may have carcinogenic effects. These characteristics increase the likelihood of toxic effects in the environment as well as on human health. Our analyses will mainly concentrate on metals, chlorinated aromatics, and polycyclic aromatic hydrocarbons because they tend to fall in this high-risk group and are the contaminants most often analyzed by environmental protection and regulatory agencies.

Presence of Contaminants in Water, Sediment, and Organisms

For toxic chemicals to be present in water they must possess some degree of solubility. Many of the metals and organic contaminants can be found in water; however, concentrations in water tend to be low because many of these compounds have a low solubility, or are particle reactive. They are therefore generally found attached to sediment particles in the water column or in bottom sediment. This means that concentrations measured in water samples will be lower if water samples are filtered prior to pollutant analyses. As of 1991, LDEQ began filtering their water samples used for the determination of metal levels. This change reflects the scientific consensus that dissolved element levels are generally better indicators of toxic effects than the combined levels of dissolved metals and those associated with particles. The presence of contaminants in sediment provides an additional pathway for entrance in aquatic food chains. The accumulation of contaminants in sediment has the advantage that long-term records can potentially be obtained through the sampling and analysis of sediment cores. Dated sediment cores have proven very informative of the history of estuarine and coastal sediment contaminants (e.g., Trefry et al. 1985, Owens and Cornwell 1995). Few long-term records of toxics in sediment exist for the Barataria-Terrebonne area, but several indicators of current levels do.

Toxics such as metals, chlorinated aromatics, and some polycyclic aromatic hydrocarbons are stable and bioavailable. These compounds, therefore, tend to accumulate in fish and shellfish. Bivalves, such as oysters, generally accumulate high levels of contaminants, making it important to monitor contaminants in these organisms. This will reduce the chance of human health problems associated with the consumption of shellfish as well as provide valuable information on the health of the environment. Moreover, mercury and many of the organic compounds are lipid (fat) soluble and may biomagnify in organisms where pollutant concentrations are higher in subsequent levels in the food chain. The pesticide, DDT, is a classic example of the latter. Mercury accumulation in fish may be a problem in parts of Louisiana north of the Barataria-Terrebonne area. Bioaccumulation and biomagnification thus result in pollutant concentrations being higher in aquatic organisms than in the surrounding water.

Water, sediment, and tissue samples provide disparate information on pollution. Contaminant levels in water are likely to fluctuate much more than in sediment or organisms, with fluctuations brought about by immediate input changes (such as run-off following a heavy rain) or by other changes in water chemistry. Consequently, sampling designs that investigate contaminant concentrations in water tend to involve a much higher sampling frequency than programs aimed at sediment and organisms. Also, changes in concentrations of water pollutants do not necessarily mean that the same changes will occur in pollutant accumulation and in the severity of toxic effects in organisms. Pollutant bioaccumulation from water depends on a variety of factors. The recently increased emphasis on pollutants present in the dissolved phase (rather than those associated with particles in the water column) generally strengthens the relationship between levels of contaminants in water and those in organisms present in these waters. Sediment integrates pollutant levels over time such that contaminant levels in sediment tend to show much less fluctuation than levels in the water column. This also reduces the need for frequent sampling. In addition, many pollutants have a high affinity for particles, such that concentrations in sediment are usually orders of magnitude above those in water. This facilitates the analysis of pollutants in sediment and reduces the chance of erroneous results due to artificial contamination (e.g., during sampling or laboratory analyses). The ready availability of standard reference sediment (both freshwater and estuarine sediment with specified levels of many pollutants, for example, from the National Institute of Standards and Technology) also facilitates the maintenance of tight quality assurance and quality control programs in analytical laboratories. However, pollutant levels in sediment, like pollutant levels in water, suffer from a possible lack of a tight relationship between contaminant levels in sediment and concentrations and effects in sediment-dwelling organisms or bottom-feeding fish. The analyses of pollutant levels in organisms has the advantages that bioaccumulation of contaminants generally leads to much higher concentrations in these organisms than in water, that organisms integrate pollutant levels over time, that reference materials (such as oyster tissue) are readily available, and most importantly, that effects and contaminant levels in organisms are assessed directly. Especially where organisms are directly consumed by humans (e.g., fish and finfish), direct relevance to human health is easily demonstrated. However, interpreting data on contaminant levels in organisms is not straightforward. Organisms differ in the extent to which a contaminant is accumulated and in their capacity to detoxify contaminants. Detoxification may result in low body burdens of the original contaminants, while metabolites of these contaminants in the organisms may cause toxic effects. Sampling of biota is thus likely to provide very valuable information on environmental contamination, though data on contaminant levels in sediment and water are also needed for a complete picture on the status of environmental contamination.

Data Sources (General)

While there are numerous sources of water quality data available for the Mississippi River and the Barataria and Terrebonne basins, many of them are limited in area, time, and types of pollutants measured. A simple status can be determined from the limited studies, but determining trends requires chemical specific data collected consistently for an area over a reasonably long period.

For the Mississippi River, discharge data is from the EPA Toxics Release Inventory (TRI) for water, which includes data reported annually by the manufacturing industry in Louisiana. However, the effect of these discharges on river water quality is best reflected by ambient (in-river) water quality data. Discharge data for the two basins also is taken from TRI. Data on other point source discharges include the larger volume wastewater treatment effluents and site-specific produced water (oil-field brine) discharges.

The most comprehensive and consistent sets of ambient water quality data for the study area are collected by LDEQ and include nine sampling stations in the river adjacent to the study area and numerous stations within the Barataria and Terrebonne basins. Site-specific, long-term ambient water quality data are available from the USGS Water Resources data for Louisiana. Sediment and organism contaminant levels and sediment toxicity results are available from federal programs that assess environmental pollution levels across all U.S. estuaries.

Data were obtained from several sources (see References and specific sections below), evaluated for usefulness, and summarized if relevant. A screening process was used to determine if the data were pertinent, extended over a sufficient time period, were collected consistently, and met quality control standards.

Sources of Contaminants

Toxics Release Inventory

Mississippi River

TRI data can be used to establish short-term trends in toxic chemical discharges to the Mississippi River. The study area covers discharges from Old River junction south to the Gulf of Mexico. This includes the vast majority of toxic discharges to the river in Louisiana. The TRI discharge data begin in 1987 and extends to the present although 1992 was the most recent year available at the time of this analysis (table 16). Although erratic, toxic discharges to the river have been decreasing since 1987 (figure 39). A major contributor to discharges is rainfall runoff from the massive gypsum stacks near Geismar, Louisiana, which are the result of phosphate fertilizer manufacture. A heavy rainfall year will have higher toxic discharges. As figure 39 indicates, 1988 and 1989 were low discharge years because of low rainfall. The discharges are averaging 150–200 million pounds of toxic chemicals by 1992, but a major reduction is expected in the 1994 data because the gypsum stacks are now covered by a clay cap. A 75% reduction of the toxics discharged from the gypsum stacks is expected.

The chemicals being discharged to the river in quantity are shown in figure 40. Ammonia, methanol, and chlorinated hydrocarbons (summed over all of chlorinated hydrocarbons on the TRI list) are the three types of chemicals discharged in largest quantities. Acids leached from the gypsum are omitted from figure 40 because they obscure other chemicals. In addition, as discussed above, the gypsum stack runoff is being largely eliminated through pollution prevention methods. Neither ammonia nor methanol are highly toxic; although, ammonia contributes to the nutrient levels in the river (discussed elsewhere in this report). Ammonia also seems to be increasing over time (figure 41). Ammonia sources are the nitrogen fertilizer manufacturing plants along the river. The methanol releases over time are highly variable (figure 42), and the trend is insignificant. Metals discharged to the river average about 25,000 lbs per year but do not show significant trends (figure 43). Lead and its compounds are decreasing significantly (figure 44) and presently averaging about 500 lb/yr. Phenol and naphthalene discharges, by-products of petroleum-related activities, are declining over time (figures 45 and 46, respectively). The declines are significant at the 90% confidence level. The sum of chlorinated hydrocarbons discharged to the river yearly are shown in figure 47. They are decreasing consistently and significantly over time and have been reduced from 110,000 lbs in 1987 to approximately 20,000 lbs by 1992.

Table 16. Dischargers to the Mississippi River adjacent to the BTNEP study area for 1992.

Facility	Discharges (lbs.)	Facility
Agrico Chemical Co. Hahnville	286,000	Georgia Gulf Corp.
Agrico Chemical Co. St. James	83,615,400	Georgia-Pacific Corp
Agrico Chemical Co. Uncle Sam	57,825,523	ICI Americas Inc.
Air Products & Chemicals	18,181	James River Paper Co. Inc.
Allied-Signal Inc.	54,807	Kaiser Aluminum and Chemical
AMAX Metals Recovery Inc.	4,680	LaRoche Chemicals Inc.
American Cyanamid Co.	297,671	M-I Drilling Fluids Co.
AMPRO Fertilizer Inc.	34,500	Marathon Oil Co.
Arcadian Fertilizer L.P.	37,671,015	Melamine Chemicals Inc.
Ashland Chemical Inc.	5,369	Michoud Assembly Facility
Baroid Drilling Fluids Inc.	250	Mobile Oil Corp.
BASF Corp.	14,848	Monsanto Co.
BF Goodrich Co.	250	NALCO Chemical Co.
Borden Chemicals & Plastics	138,072	Occidental Chemical Corp. Convent
PT Oil Co.	230,433	Occidental Chemical Corp. Taft
BTI Plaquemine	20	Placid Refining Co.
CF Industries Inc.	934,905	Pioneer Chlor Alkali Co.
Chevron Chemical Co.	253	Rhone-Poulenc
CIBA-GEIGY Corp	157,652	Rubicon Inc.
COSMAR Co.	236	Schuylkill Metals Corp.
DELTECH Corp	649	Shell Chemical Co.
Dow Chemical Co.	369,687	Shell Oil Co.
DSM Copolymer Inc.	3,260	Star Enterprises Inc.
DuPont	5	Triad Chemical
DuPont Burnside Plant	400	Union Carbide Corp.
Ethyl Process Development	597	Union Texas Prods. Corp.
Exxon	312,704	Uniroyal Chemical Co. Inc.
Ferro Corp.	11,255	Vulcan Materials Co.
Formosa Plastics Corp.	831	

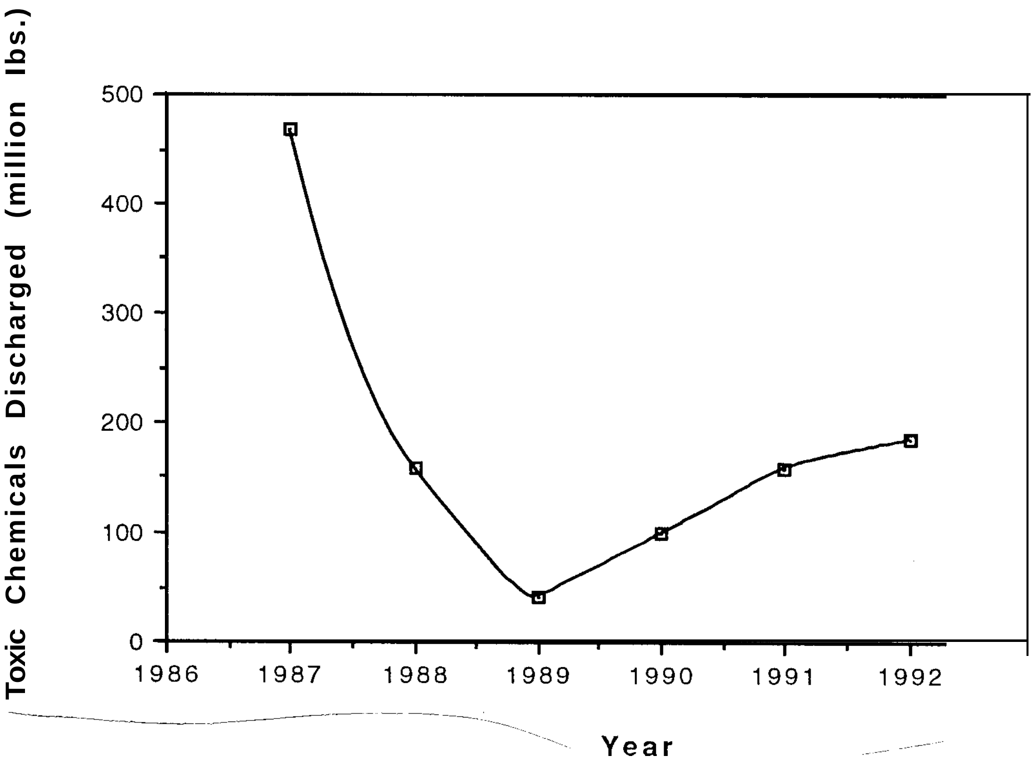


Figure 39. Toxic chemicals discharged to the Mississippi River (adjacent to the BTNEP area) by year.

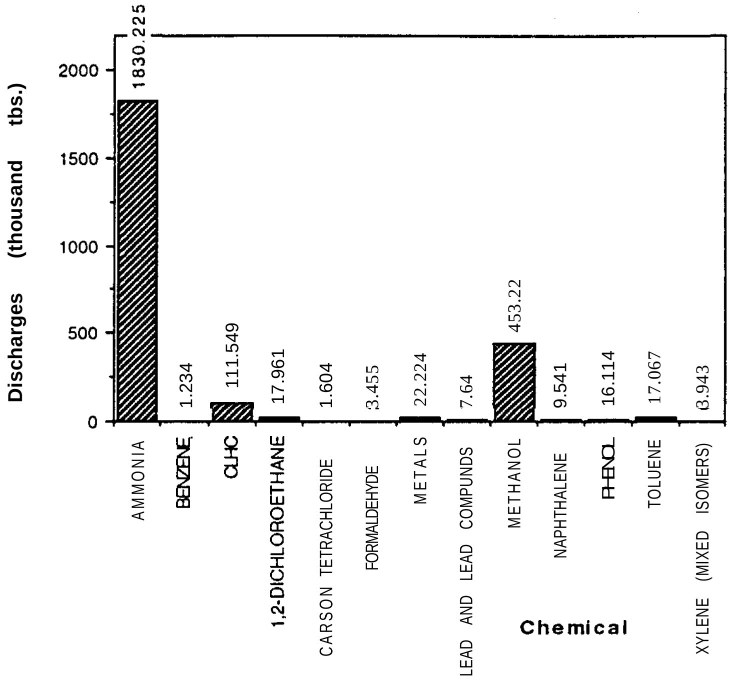


Figure 40. Toxic chemicals discharged to the Mississippi River (adjacent to the BTNEP area) in 1987.

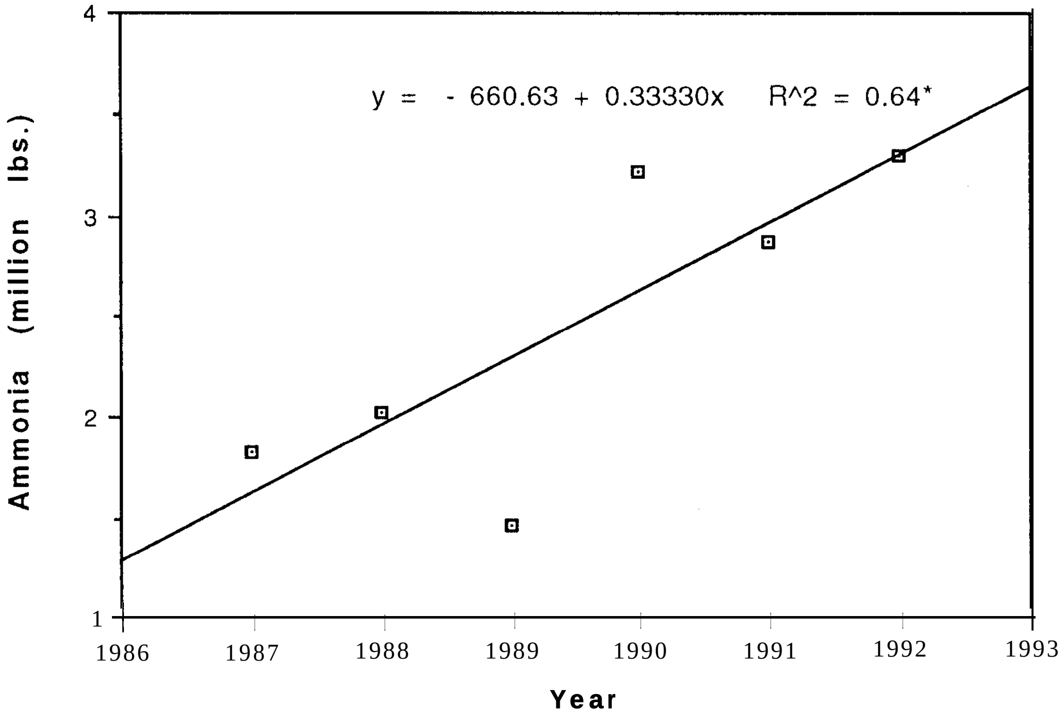


Figure 41. Ammonia discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<O.OOl.

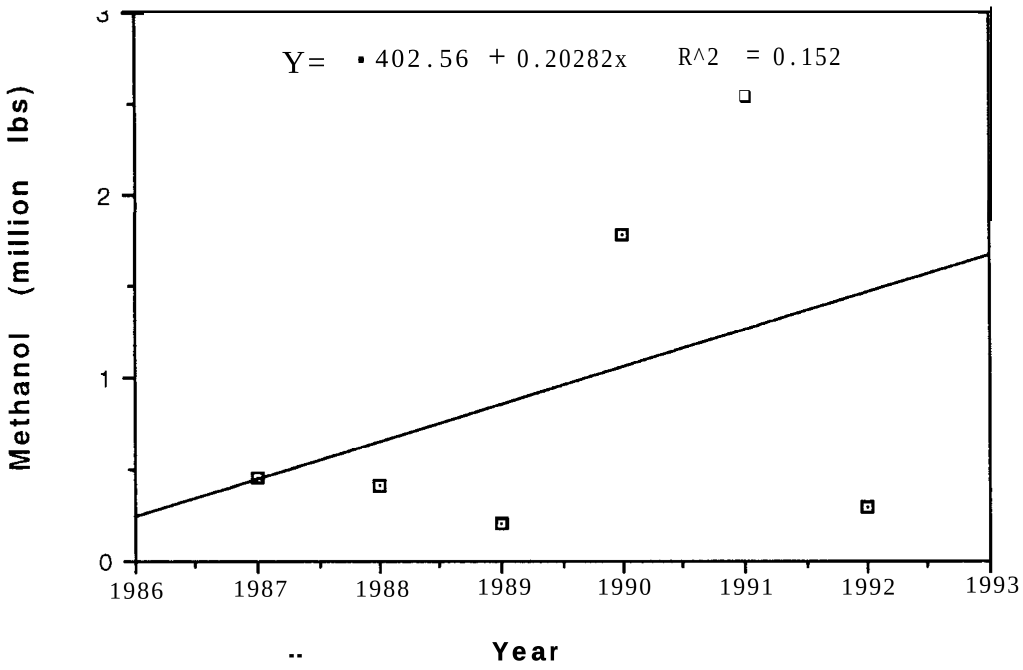


Figure 42. Methanol discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: ~ for p<0.01, * for 0<0.05 ** for p<0.01 and *** for p<O.OOl.

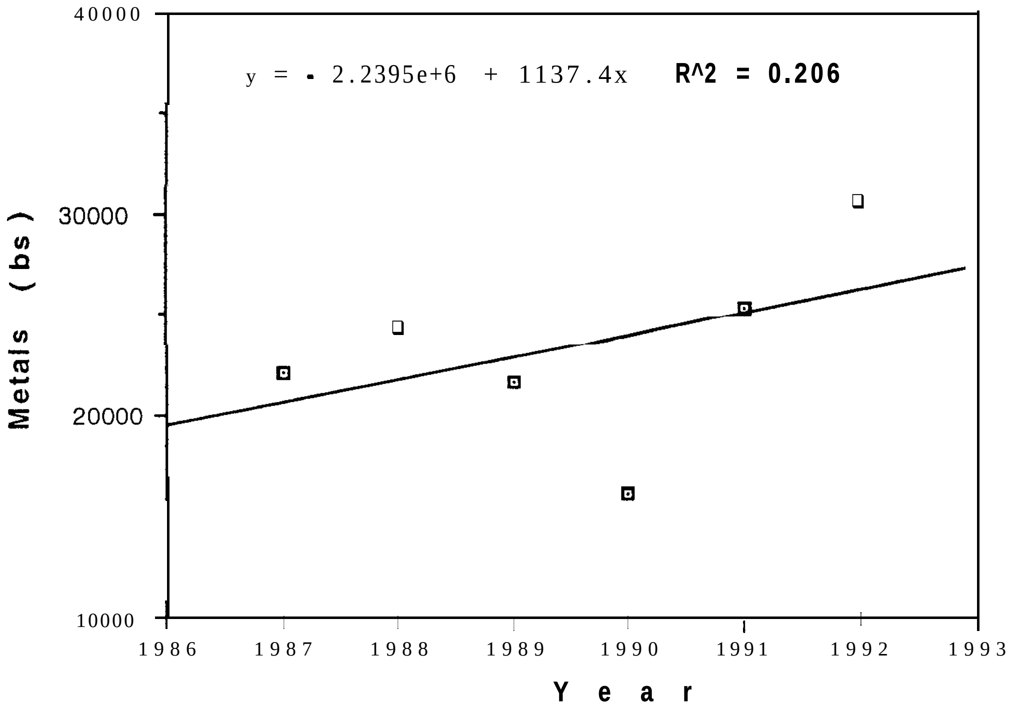


Figure 43. Metals discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: ~ for p<0.01, * for p<0.05, ** for p<0.01 and *** for p<O.OOl.

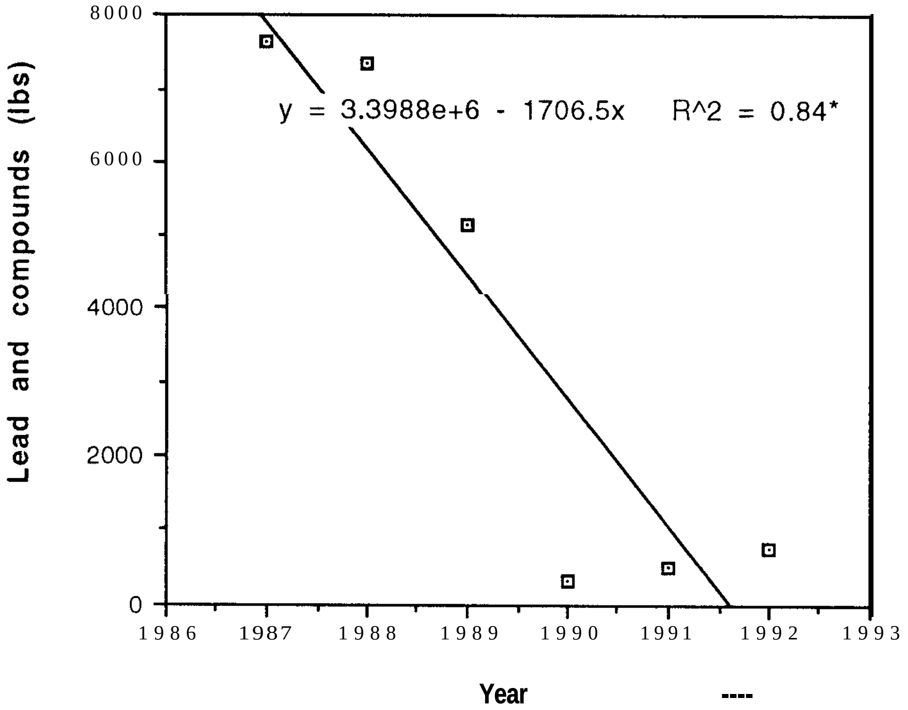


Figure 44. Lead (and its compounds) discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: -for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<O.OOl.

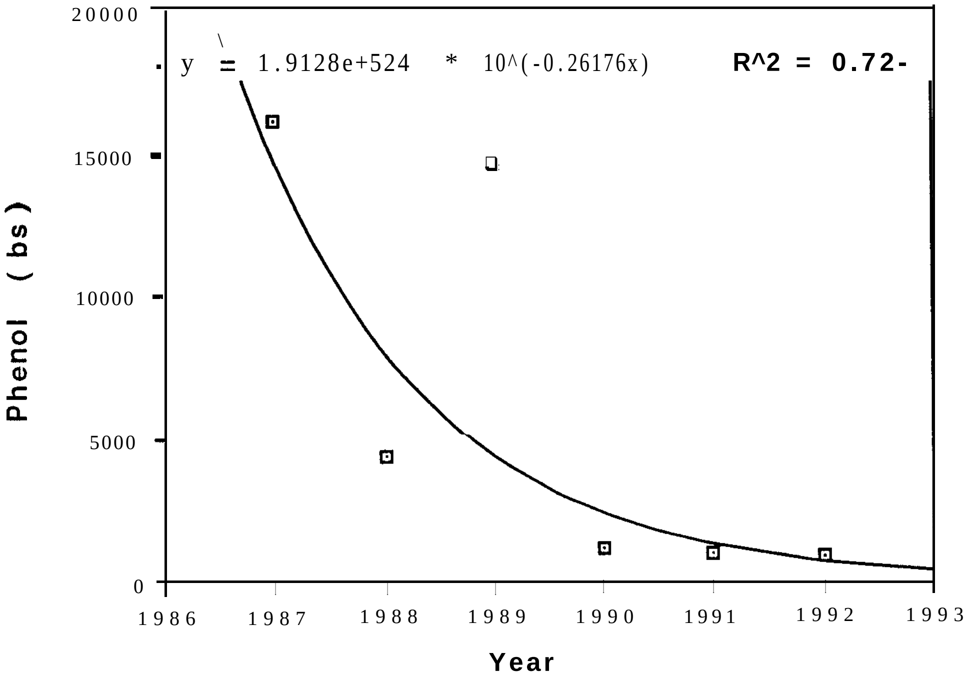


Figure 45. Phenol discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<0.001.

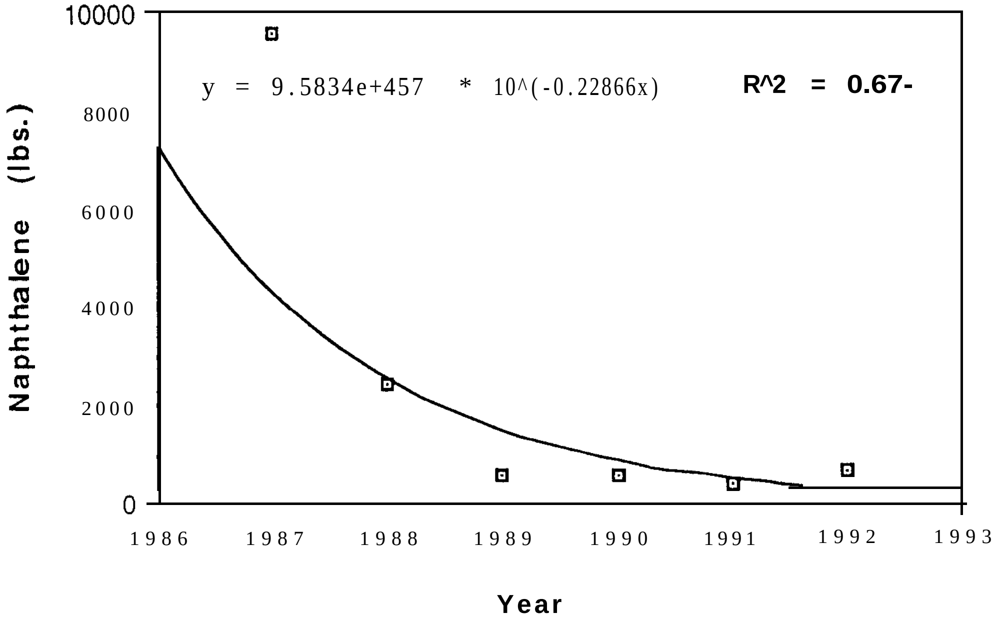


Figure 46. Naphthalene discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for o<0.01 and *** for p<0.001.

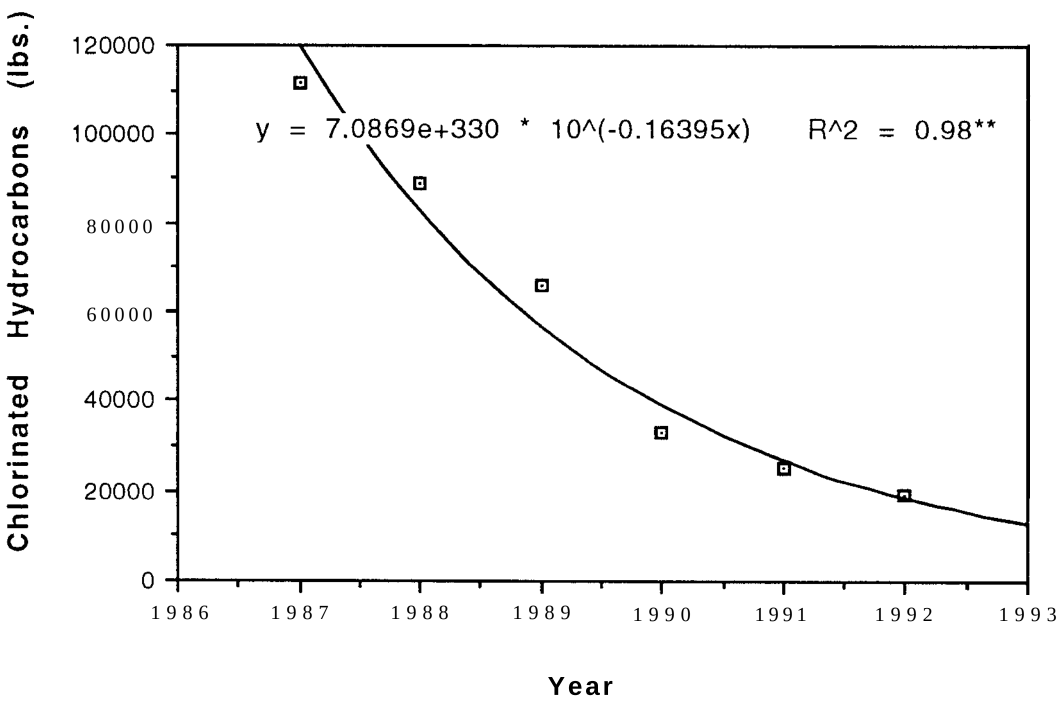


Figure 47. Chlorinated hydrocarbons discharged to the Mississippi River (adjacent to the BTNEP area) by year. Trends that are significant at the 95% confidence level are coded as: ~ for $p<0.1$, * for $p<0.05$, ** for $p<0.01$ and *** for $p<0.001$.

Barataria and Terrebonne Basins

With the exception of two firms located in Port Allen that discharge into the Intracoastal Waterway, all of the companies releasing toxics into the estuary are located in the southern part of the estuary in Lafourche and St. Mary parishes (figure 48). In Lafourche Parish, the Forty Arpent Canal is the receiving stream; in St. Mary Parish, Bayou Boeuf and the Intracoastal Waterway are the receiving streams. Table 17 shows yearly toxic releases to the two basins in the estuary.

It should be noted that the number of companies reporting releases is nearly consistent across years and the increase in observations is the result more of reports being submitted (each representing a different chemical) than new companies operating. The large increase observed from 1990 to 1991 is largely because of one company and one chemical that although reported in previous years increased in releases in 199 1. Table 18 summarizes releases by chemical and year.

The sum of all toxic releases to the estuary is shown in figure 49 by year. There is an apparent increase over time as stated above; most of this is believed to be due to more rigorous reporting under TRI regulations. Figure 50 shows the amounts released of the

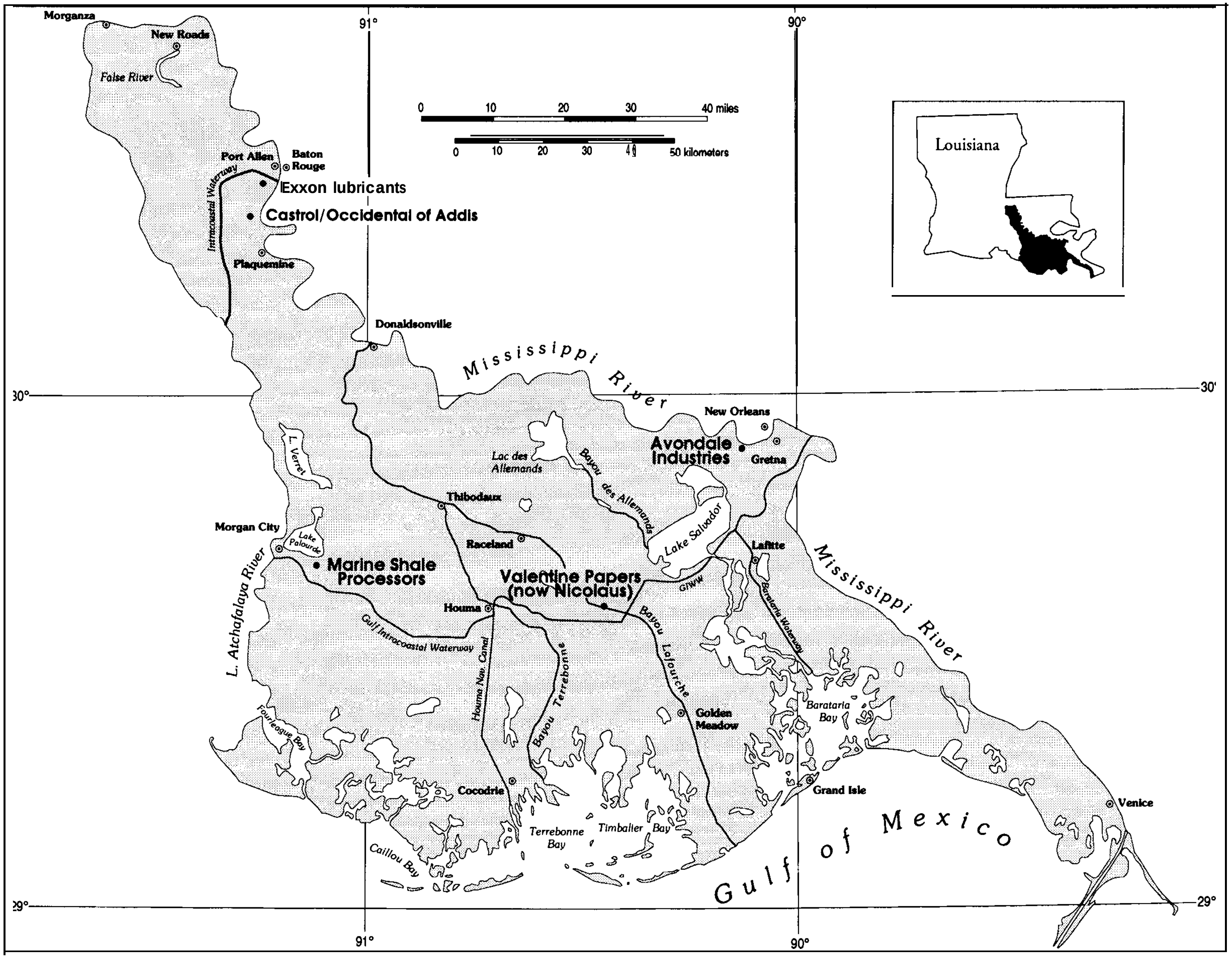


Figure 48. Sources of toxic discharges into the Barataria and Terrebonne basins.

Table 17. Toxic releases to Barataria and Terrebonne estuaries (source: U.S. EPA Toxics Release Inventory).

Year	Number of Reports	Total Releases (lbs)
1987	5	914
1988	6	836
1989	5	280
1990	10	1,014
1991	12	2,117
1992	18	2,662

Table 18. Toxics released to the Barataria and Terrebonne estuaries by chemical (lbs) and year.

Chemical (lbs)	1987	1988	1989	1990	1991	1992
Formaldehyde	79	54	16	250	1850	2150
Phenol	85	32	8	255	252	255
Sodium Hydroxide	250					
Sulfuric Acid	250			5		
Aluminum Oxide		250				
Chlorinated Hydrocarbons			6			20
Metals	250	500	250	509	31	227
Phthalate				1		
Toluene					1	5
Xylene					1	
Ethyl Benzene						5
Total (lbs)	914	836	280	1020	2135	2662

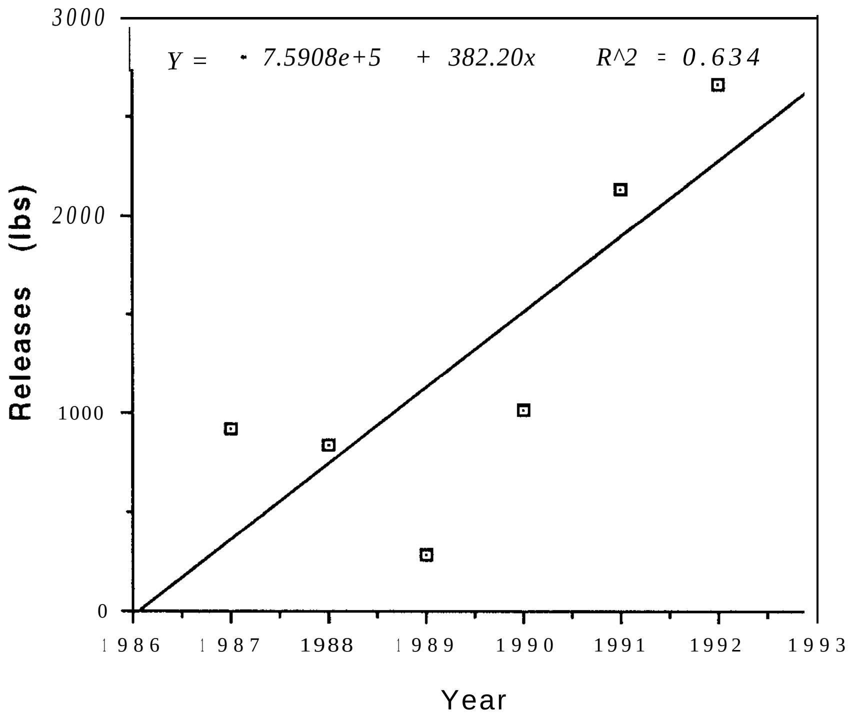


Figure 49. Total toxics released into the Barataria and Terrebonne basins (source: U.S. EPA Toxics Release Inventory). Trends that are significant at the 95% confidence level are coded as: ~ for $p<0.1$, * for $p<0.05$ ** for $p<0.01$ and *** for $p<0.001$.

major compounds. Formaldehyde is released in the largest amounts followed by phenols, metals, and chlorinated hydrocarbons; however, these figures are relatively low at least when compared with discharges to the Mississippi River. The discharge of approximately 2,600 lbs of chemicals, predominately formaldehyde which is biodegradable, to an area as large as the estuary may not exceed the assimilation capacity. There are relatively few dischargers in the Barataria and Terrebonne estuaries because it has not been a preferred site for industrial facilities. Most of the major industries in the study area are sited on the Mississippi River because of its transportation ease, water supply, and other advantages. The time trend of formaldehyde releases are shown in figure 5 1, which depicts a significant increase over time. Figure 52 shows an increasing trend for phenols and a decreasing trend for metals over time; however, considerable scatter is present in the metals data, and the trend is not significant.

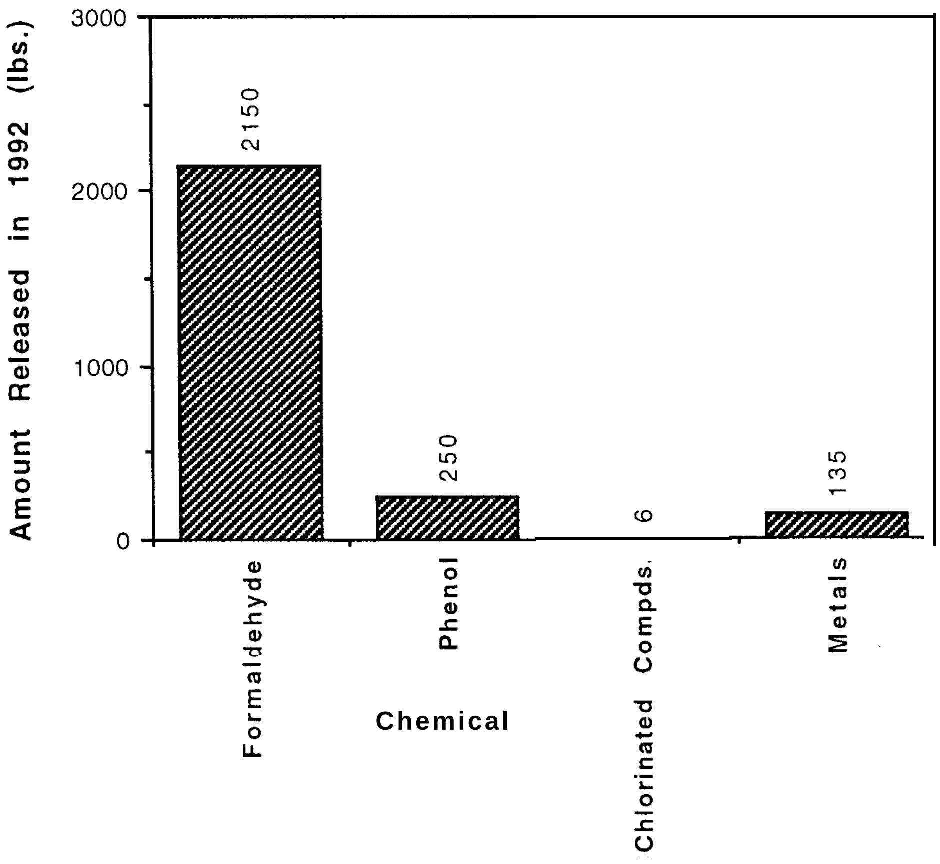


Figure 50. Amount of toxic chemicals released into the Barataria and Terrebomre basins combined, by individual chemical or group of chemicals (source: U.S. EPA Toxics Release Inventory).

Produced Water Discharges

During the production of crude oil, condensates, or natural gas, water that is brought to the surface with the product stream is removed. The separated produced water (or oil-field brine) is re-injected into a well, either for disposal or enhanced recovery of hydrocarbons, or is discharged into surface waters of the ocean or coastal area, which is common practice during production in the northern Gulf of Mexico region.

Produced water discharges are now permitted by EPA under a General Permit system. EPA has issued NPDES General Permits for produced water discharges in the “Upland” and “Coastal” categories. The Coastal NPDES General Permit became effective February 8, 1995. Under the EPA definition, the inner boundary of the “coastal” area is not a single line but is any body of water landward of the territorial sea. The “upland” subcategory

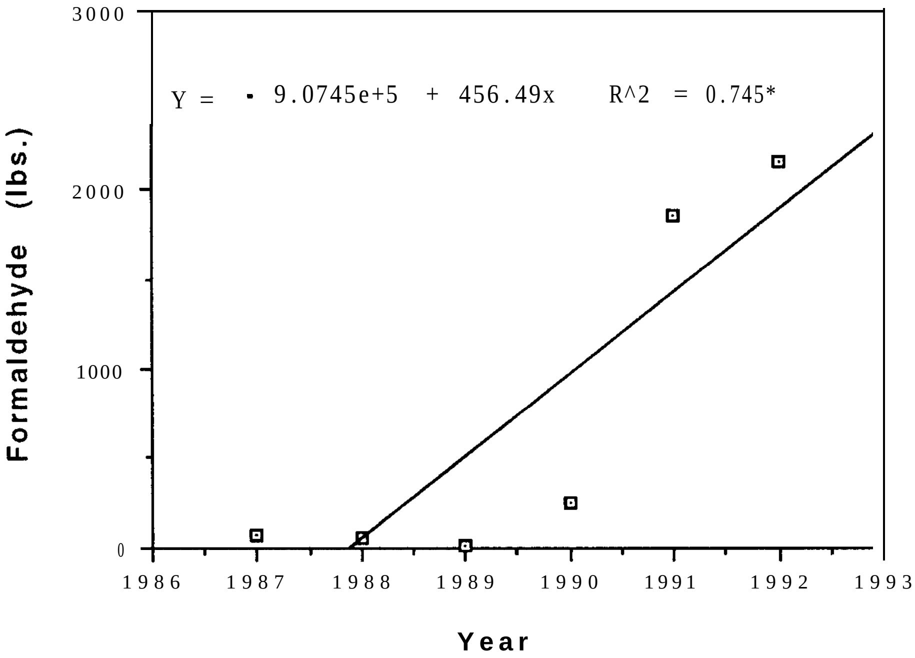


Figure 5 1. Formaldehyde releases to the Barataria and Terrebonne basins from 1987 to 1992 (source: U.S. EPA Toxics Release Inventory). Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<0.00l.

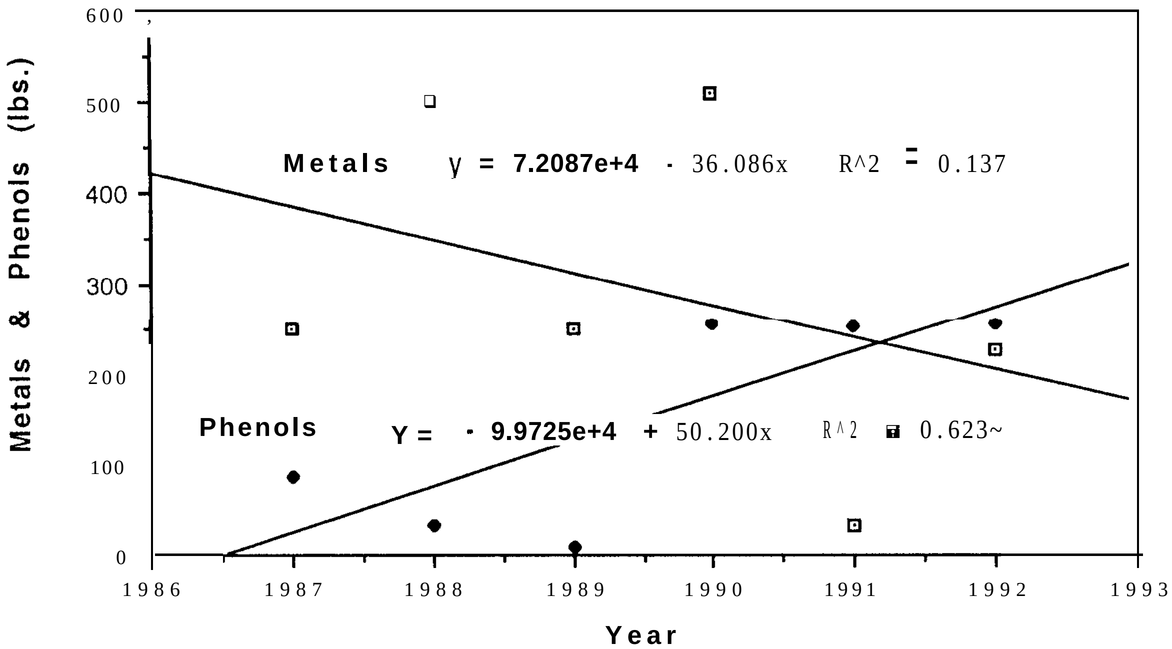


Figure 52. Metals and phenols discharged to the Barataria and Terrebonne basins from 1987 to 1992 (source: U.S. EPA Toxics Release Inventory). Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<0.00l.

would be any land-based petroleum extraction facility. These EPA permits call for no discharge of produced water, except for a few facilities (5–6), located in the lower reaches of the Mississippi and Atchafalaya rivers (i.e., below the latitude of Morgan City, which must file for individual permits). A no-discharge compliance must be met no later than January 1, 1997, unless the state requires an earlier compliance.

Eventually, produced water discharges will be non-existent. Results from several studies of the fate and effects of produced water discharges, however, hold valuable information for current and future resource management. The following sections detail the sources; fate and effects are provided on p. 181–185.

Discharge Points and Volumes

The total volume of produced waters discharged into Louisiana state waters averages near 1.8–1.9 million bbl/day (Boesch and Rabalais 1989a, Rabalais et al. 1991b). If offshore waters within the Louisiana territorial sea are excluded, the total volume discharged to estuarine waters is 1.7 million bbl/day. Included in the discharges within state waters are discharges originating on the federally controlled Outer Continental Shelf (=250,000 bbl/day). Current estimates for "State Waters" (total) is 1.5 million bbl/day (D. Hale, LDEQ, pers. comm.). The reason for the discrepancy is unclear; therefore, a more thorough analysis of permit status, locations, and volumes is warranted. The majority of produced water discharge sites (by number) and volume occur within the Barataria-Terrebonne estuarine system or in the Atchafalaya or Mississippi rivers, which affect the periphery of the study area (figures 53–55).

Chemical Constituents

Produced waters are elevated in salinity, ranging from near the value of seawater (35 ppt) to as high as 200 ppt (Rabalais et al. 1992). Sulfide concentrations can be high (120 µg-atoms S l^{-1}), elevated (3 to 50 µg-atoms S l^{-1}), or non-detectable.

Produced waters contain many species of organic compounds, primarily petroleum hydrocarbons, but also partially oxidized organics. Similar compounds are found in each of the effluents, but the relative proportions differ by facility and time (detailed results in Rabalais et al. 1991b). The components representing the highest proportion of the organic load in effluents studied by Rabalais et al. (1991b) were the aliphatic fatty acids and the aromatic acids. The compounds are water soluble and likely to be diluted in the water column. Saturated hydrocarbons were the next highest in concentration, but these compounds are the least toxic fraction of crude oil and very susceptible to microbial degradation in the environment. The volatile hydrocarbons (next highest contribution of identified organics) are highly water soluble and are acutely toxic to organisms exposed to high concentrations. Their long-term fate, however, is to be volatilized or diluted and dispersed in the water column. The polynuclear aromatic hydrocarbons (PAH) contributed the lowest proportion of the total identifiable hydrocarbons. This fraction,

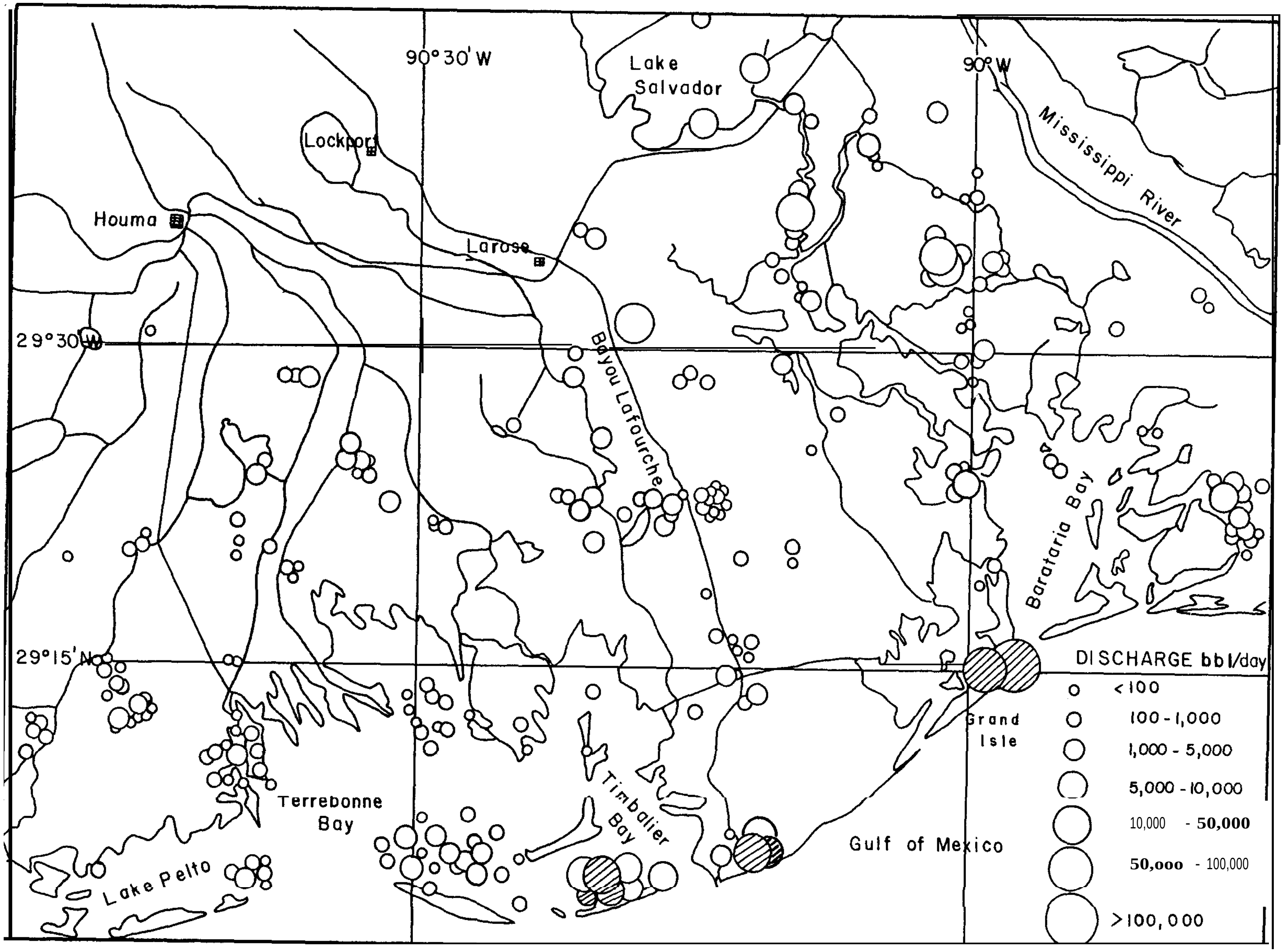


Figure 53. Location of produced water discharges and their relative size for portions of the Barataria and Terrebonne estuaries (from Boesch and Rabalais 1989a).

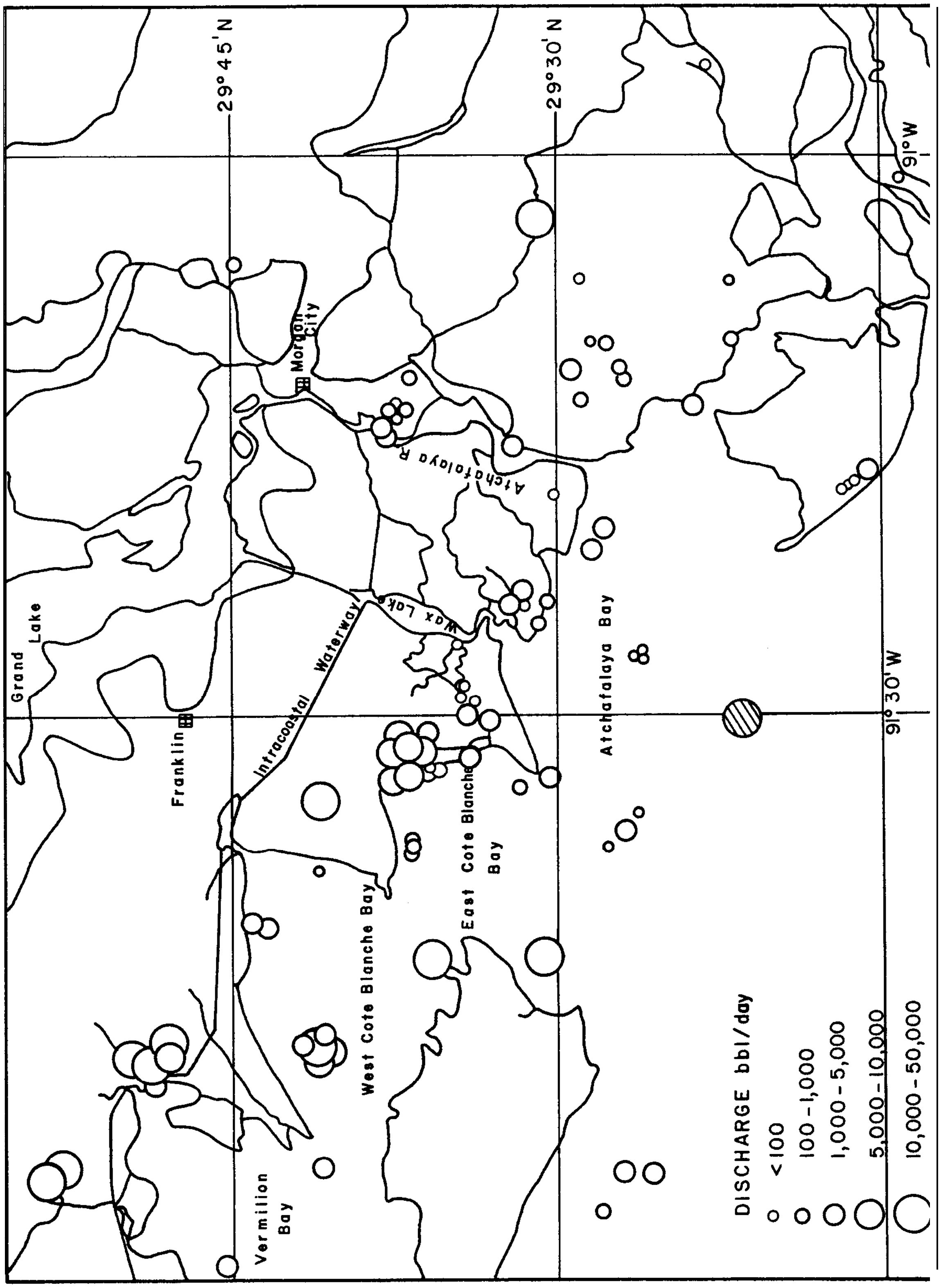


Figure 54. Location of produced water discharges and their relative size for portions of the Terrebonne estuary and the Atchafalaya River and Wax Lake Outlet areas (from Boesch and Rabalais 1989a).

however, is the heaviest, most toxic, and environmentally stable fraction of crude oil. Crude oil toxicity is a reflection of its aromatic content, primarily the alkyl-substituted naphthalenes and phenanthrenes (National Research Council 1985, Boesch and Rabalais 1987). PAHs and their alkylated homologs are the most likely components of produced water discharges to be incorporated into sediment because of low water solubilities and high sorption coefficients.

Produced water effluents generally have high concentrations of trace metals. The most abundant in the effluents sampled by Rabalais et al. (1991b) were barium, vanadium, and nickel. Zinc, copper, and chromium also were found in higher than ambient concentrations in most of the discharges. Cadmium, mercury, and lead were detected in various effluents. In most cases, the concentrations of all nine metals exceed the levels normally found in fresh water and seawater by a factor of 1,000 (Forstner and Wittman 1983).

Produced waters from the sites examined by Rabalais et al. (1991b) had total radium (^{226}Ra plus ^{228}Ra) activities approximately 150–1,500 times higher than natural waters (Reid 1983). All discharges sampled had total radium (two thirds are ^{226}Ra) activity in excess of the ~55 pCi activity designated by EPA as hazardous waste.

Estimated Loadings of Selected Produced Water Constituents

Table 19 lists the amounts of produced water from oil and gas operations in the Barataria and Terrebonne basins and estimates of the associated pollutants being discharged. The hydrocarbons discharged yearly in produced waters were estimated from Boesch and Rabalais (1989a) findings for hydrocarbon concentrations in produced waters from Conoco's Grand Isle oil field. Hydrocarbons used to calculate loadings were benzene, toluene, ethylbenzene, xylenes, phenol, p-cresol, m,o-cresol, acids, and naphthalene. Note that above it was stated that constituents and concentrations of constituents vary by discharge; also identifiable hydrocarbons make up a small percentage of the total hydrocarbon suite. The estimates thus provide a minimal estimate of the total petroleum hydrocarbons discharged.

Estimated concentrations for metals in table 19 are those routinely sampled by LDEQ as part of the Ambient Water Monitoring Program: copper, cadmium, lead, chromium, and arsenic. While mercury is a criterion metal for LDEQ, sampling of produced waters in the Boesch and Rabalais (1989a) study found mercury concentrations to be generally below detection level. [Note, that Rabalais et al. (1991b) reported detectable mercury in some produced water effluents.] The estimated amounts of metals would be higher if other metals found in produced water (iron, aluminum, barium, and others) were included. The metal concentration estimates were calculated by averaging over metal concentrations in six produced water samples, two Conoco discharges, and four Exxon discharges on Grand Isle (Boesch and Rabalais 1989a).

To estimate naturally occurring radioactive material (NORM) releases into the basins, total produced water releases were multiplied by the average ^{226}Ra content of produced

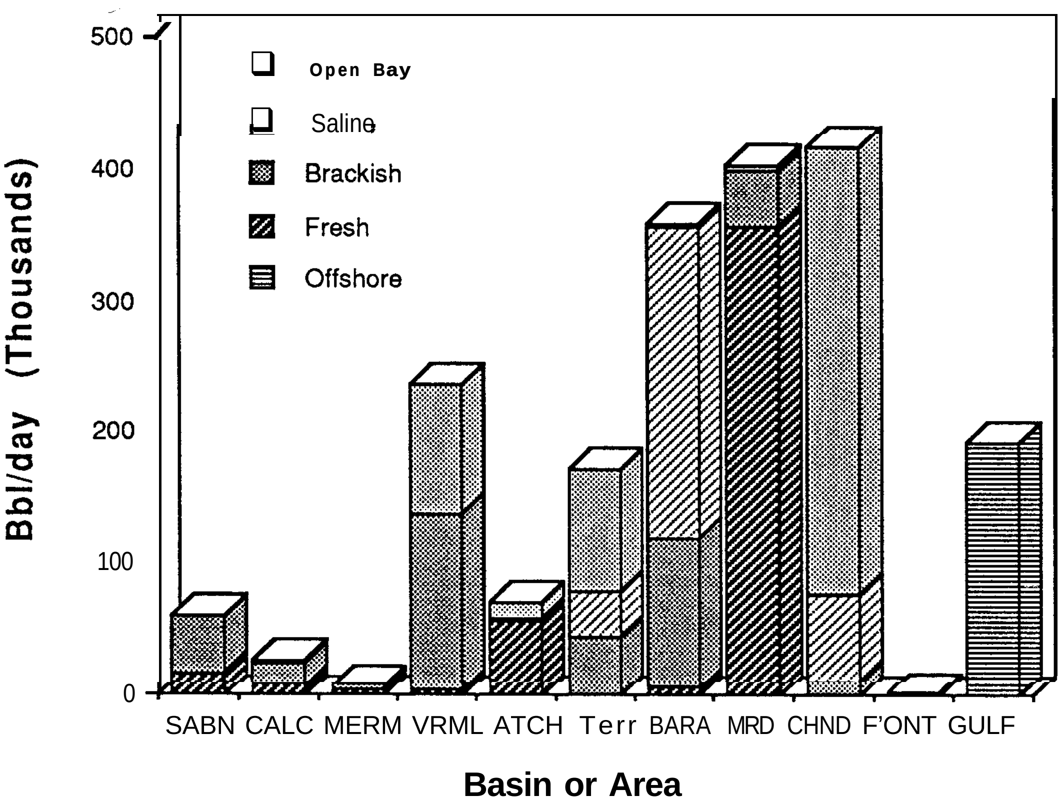


Figure 55. Volumes (by habitat type) of produced water discharges for major hydrologic areas in Louisiana (from Boesch and Rabalais 1989a).

waters from nine Gulf Coast sites sampled by Reid (1983) and cited in St. PC (1990). The reader should be aware that radium concentrations are normally reported in pico curies (pCi), which are 1 X 10^{-12} curies. The amount of the radium reported in table 19 is in curies, which is a large amount of radioactive material.

Municipal Wastewater Discharges

Publicly Owned Treatment Works (POTW) that discharge over one million gallons per day are to include priority pollutants (toxics) in their monthly Discharge Monitoring Report (DMR) to LDEQ. While at least some of the treatment stations in Morgan City, Houma, and Thibodaux meet this volume criterion, a review of the DMRs (they were unavailable for Morgan City) revealed no reported toxic discharges. These files are available for public review at LDEQ but have not been summarized for easy use. For the most part they document chlorine discharges, expected constituents from water treatment plants.

Table 19. Produced water discharges and estimated load of selected pollutants by basin.

	Units	Barataria Basin	Terrebonne Basin	Total of Both Basins
Total Discharge*	bbl/day	363,054	173,656	536,710
Contaminants:				
Hydrocarbons*~	lbs/yr	479,635	229,422	709,057
Metals				
Copper*	lbs/yr	10,411	4,980	15,391
Cadmium*	lbs/yr	1,655	792	2,447
Lead*	lbs/yr	10,287	4,921	15,208
Chromium*	lbs/yr	1,361	651	2,012
Arsenic*	lbs/yr	10,829	5,180	16,009
Total of 6 Metals	lbs/yr	34,543	16,524	51,067
Radioactivity^	Radium 226 Ci/yr	5.27	2.52	7.79

*Boesch and Rabalais 1989a
^St. Pé, 1990
~based on benzene, toluene, ethylbenzene, xylene, phenol, m,o-cresols, acids, and naphthalene; i.e. not total hydrocarbons

Unpermitted Discharges, Accidental Spills

The Barataria-Terrebonne estuarine system is particularly vulnerable to release of oil-field fluids because of the numerous storage vessels, production facilities, and miles of pipelines, flowlines and injection lines within its borders. The petroleum industry infrastructure within the system is enormous. An indication of its magnitude is in the value of severance taxes and royalties paid to the state in 1987 from oil and gas produced within the estuarine system, which was slightly over $221 million or 30% of the total paid statewide (Roemer 1989). Many spills are classified as accidental, or equipment, storage tank, or pipeline failures. The biggest source of major oil spills is from pipelines and storage tanks, which are plentiful in the study area (G. Rainey, Minerals Management Service, pers. comm.). The likelihood of breaches in pipelines and other product lines increases with time as more facilities are required to install produced water injection lines, as the infrastructure of the oil-field ages, and as coastal landscape alterations and subsidence expose and stress pipelines. Abandoned or neglected facilities create spills from leaking storage tanks and flow lines or inadequate containment. These facilities, if out of compliance, can be designated "orphan wells" under Louisiana Department of Natural Resources (LDNR) regulations and cited under other regulations of LDEQ, EPA, and the U.S. Coast Guard (USCG).

Oil, diesel, fuel oil, and mixed oil/water spills are common in the Barataria-Terrebonne estuarine system. The petroleum in these spills is a complex mixture of hydrocarbons that can be toxic to plants and animals impacted by a release or spill. In the case of produced waters, the brine composition of the effluent also may affect vegetation from excessive salt. Exact numbers of petroleum releases or spills are difficult to obtain because no single agency maintains spill data for the area.

There are indications that spills are becoming more numerous or may be increasing in magnitude of impact. The New Orleans and Morgan City Coast Guard Marine Safety Offices are the two busiest spill response districts in the United States (K. M. St. Pé, pers. comm.). While some of the responses in the New Orleans office are in the Mississippi River and some of the Morgan City office responses are west of the Barataria-Terrebonne estuarine system, the high response rates are an indication of how many spills occur. There has been an increase in the amount of personnel effort spent on oil spill response from 1990 to 1994 with a tripling of effort since 1990 for the Bayou Lafourche Regional Office (K. M. St. Pé, pers. comm.), which is one of the three LDEQ regions covering the study area.

Extracting data from the EPA National Response Center or the USCG Marine Safety Office concerning oil products spills is a lengthy, bureaucratic procedure. We obtained data on oil, diesel, fuel oil, and mixed oil/water spills from the LDEQ Surveillance Office (C. Piehler, pers. comm.) for the period September 1989–May 1994. The data were reported by date, facility, parish, waterbody, amount, and spill type. Conversions were made to convert to gallons per spill. Produced water spills greater than one barrel (42 U.S. gallons) are required to be reported to LDEQ; however, the volume spilled is not easily determined because these fluids

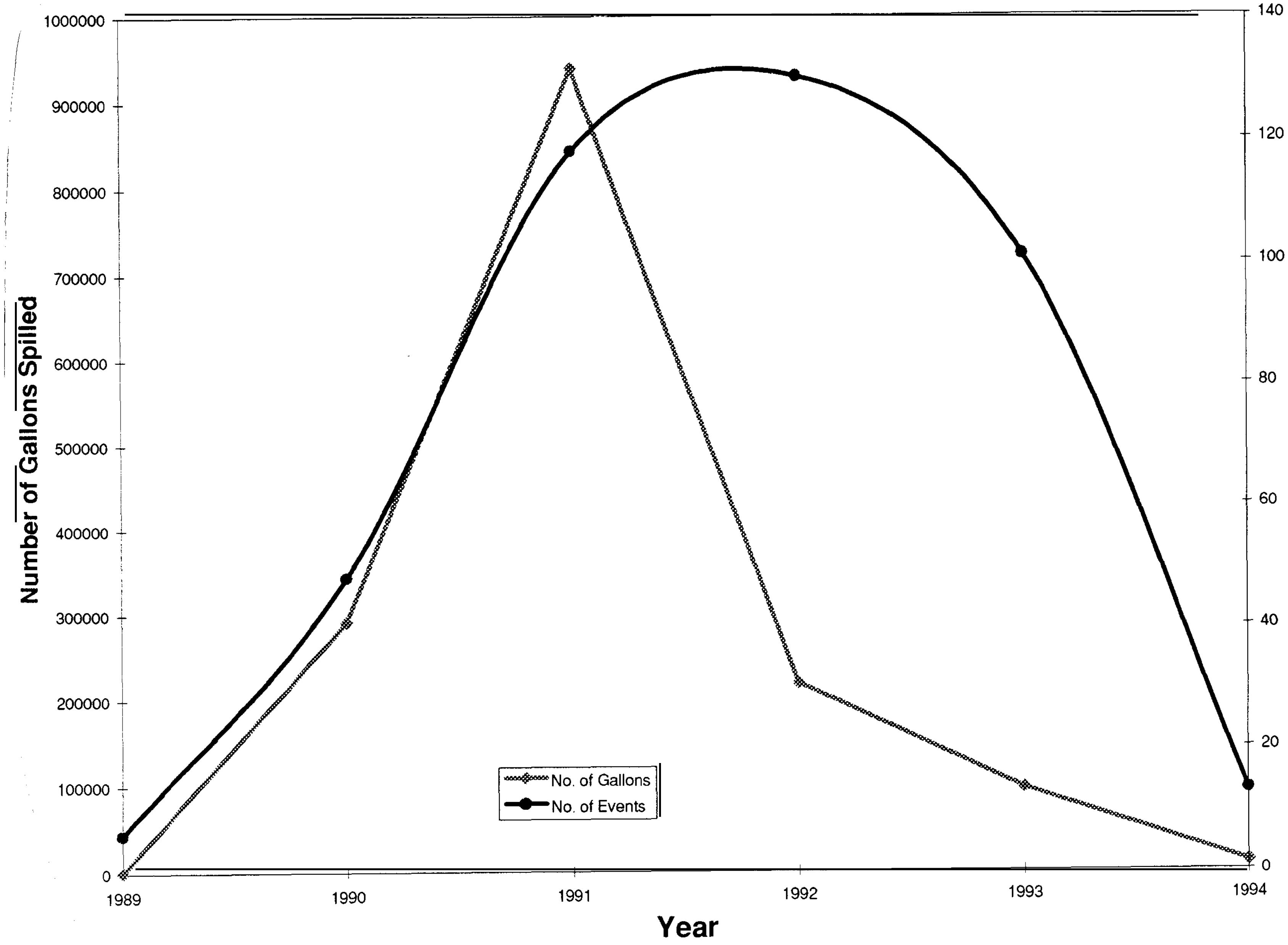


Figure 56. Number of events and gallons of oil products spilled per year for September 1989 through May 1994. Note that 1989 and 1994 are not complete years. Data from LDEQ, Surveillance.

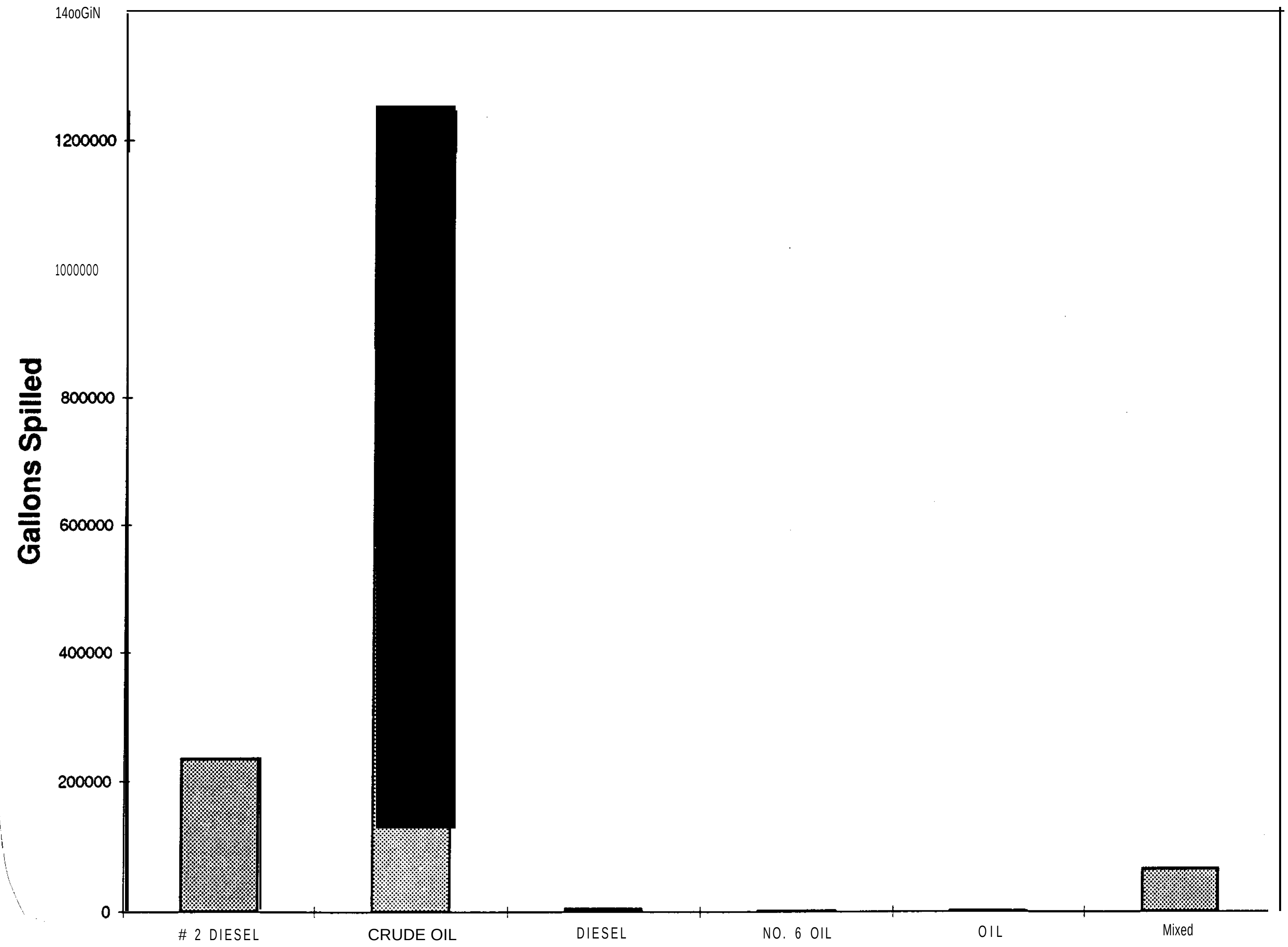


Figure 57. Number of gallons of oil products spilled by product type for September 1989 through May 1994. Note that 1989 and 1994 are not complete years. Data from LDEQ, Surveillance.

Percentage of Gallons of Oil Products Spilled per Parish

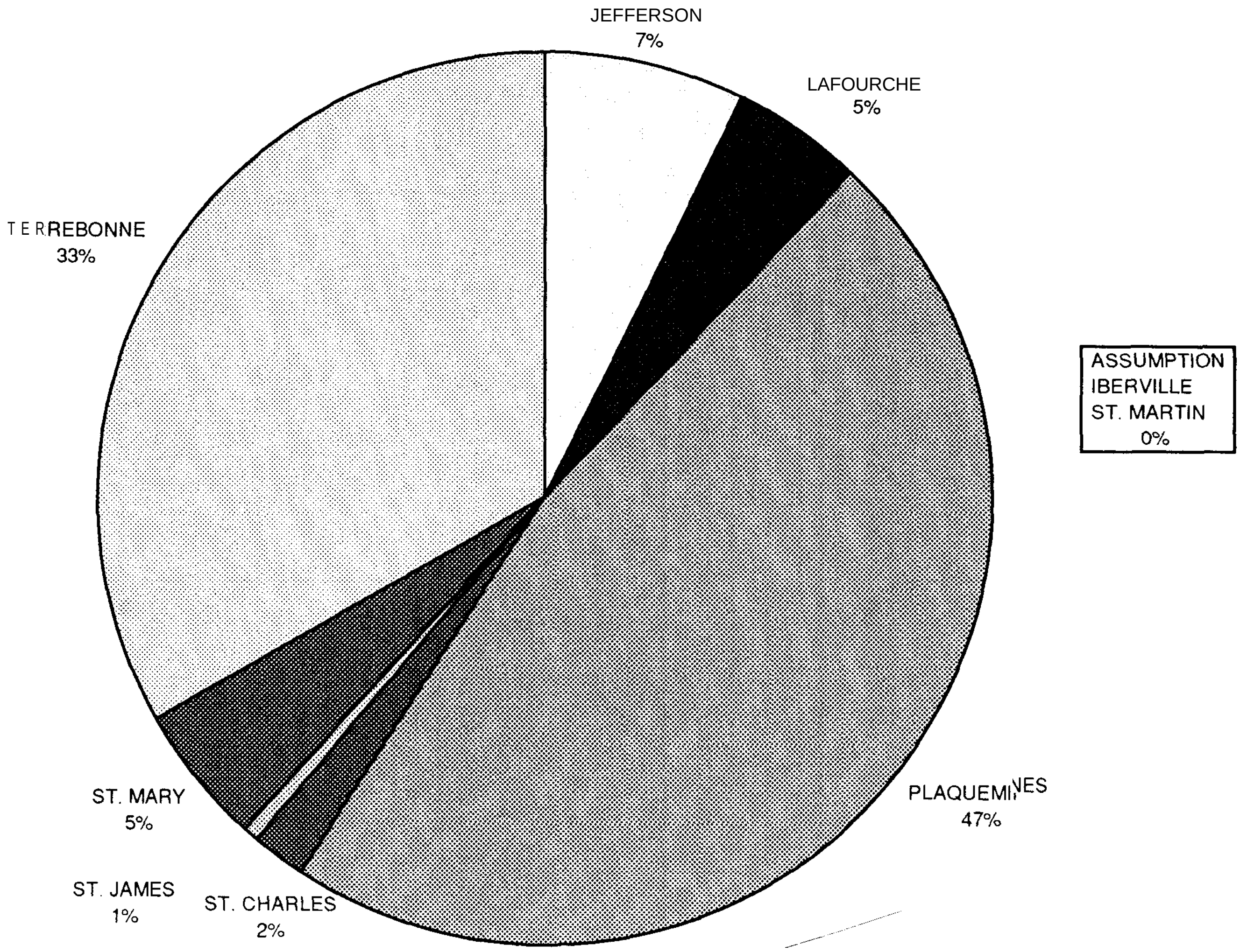


Figure 58. Frequency distribution of gallons of oil products spilled by parish with the Barataria-Terrebonne estuarine system. Data from LDEQ, Surveillance.

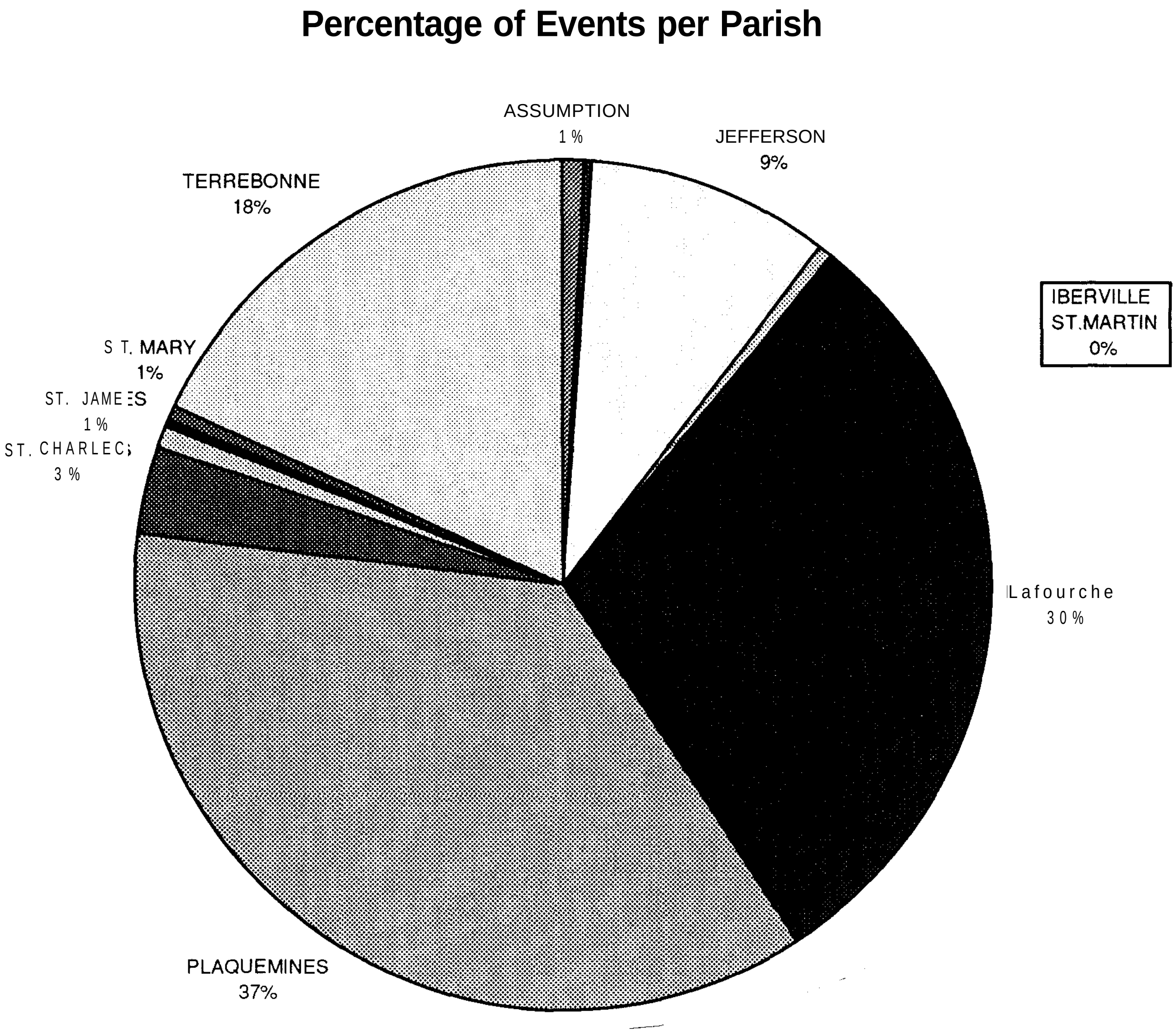


Figure 59. Frequency distribution of events of oil products spilled by parish within the Barataria-Terrebonne estuarine system. Data from LDEQ, Surveillance.

detected, the percent of "detects" out of total samples analyzed can be calculated. This technique has been used by LDEQ in other studies where very small concentrations are prevalent. Figure 60 shows the percent detections of volatile chlorinated hydrocarbons averaged over all nine river stations by year. Because we are now examining concentrations of chlorinated hydrocarbons in river water, the flow volume of the river becomes a dominant factor, and in dry years, such as 1988, less water is available for dilution, and concentrations (and percent detects) rise dramatically. In normal and high river flow years, the percent detection is expected to drop as figure 60 indicates.

Another trend of interest is that of the ambient percent detection of chlorinated hydrocarbons as a function of location on the Mississippi River. Locations are measured by River Mile (RM) distance above Head of Passes, which is near the mouth of the river. Figure 61 shows the percent detection trend of chlorinated hydrocarbons as a function of river mile by year for 1987-1990. Three of the years show significant trends. The highest

Figure 60. Average % detections of volatile chlorinated hydrocarbons averaged over all nine river stations by year.

Figure 60. Average % detections of volatile chlorinated hydrocarbons averaged over all nine river stations by year.

percent detection occurs at the lowest RM, i.e., the downstream locations have the highest percent detections and presumably the highest concentrations of chlorinated hydrocarbons as we would expect if chemicals were being added in the study area as the discharge data show. The highest concentrations occurred in 1988, the lowest river flow year, also as expected. The most recent data, 1990, have the lowest percent detection, which tracks the discharge reductions shown in figure 61.

We also can examine the trend of percent detection by year for several stations of interest. The two stations nearest the mouth of the Mississippi River, RM 105.8 and RM 115.2, show declining percent detection over time (figure 62), and the trend for RM 115.2 is significant at the 90% level. Station RM 155.6 shows a significant counter trend, i.e., increasing percent detection of chlorinated hydrocarbons over time (figure 63). This could result from increasing discharges slightly upstream of the station, but determining the cause is beyond the scope of this study.

The trends analysis of the Mississippi River using discharge (see above) and ambient water quality data gives a consistent picture of reduced discharges to the river and reduced detection of chlorinated hydrocarbons in river water. The regulatory function seems to be having the desired result; however, discharges to the river are still among the highest in the United States and should continue to be reduced.

Agricultural Chemicals in the Mississippi River

USGS Studies

Agricultural chemicals deposited in the Mississippi River drainage basin impact the river and in turn the Barataria and Terrebonne estuaries. Determining the fate of the thousands of tons of pesticides deposited annually to the Mississippi River drainage basin is difficult. USGS estimates annual application for the herbicides, atrazine, alachlor, cyanazine, and metolachlor. Battaglin et al. (1993) combined these estimates with river sampling results to model transport of herbicides as a percentage of application. They estimate that 1.6% of atrazine and cyanazine, 0.8% of metolachlor, and 0.2% of alachlor applied from April 1991 to March 1992 made its way to rivers in solution.

Pereira et al. (1989) reported that some of the chemicals unaccounted for in river solution became soil bound, thus hindering their potential for biodegradation. Pereira et al. show concentrations in sediment of eight pesticides for two 1987 study periods. Though further study is recommended, western Gulf of Mexico circulation trends (Pereira and Hostettler 1993) indicate possible deposition into Barataria and Terrebonne bays of contaminated sediment that reach the mouth of the Mississippi River. Bioaccumulation in the food chain also is indicated by detection of agrochemicals in Mississippi River catfish.

Pesticide concentration data from water samples indicate temporal and spatial variability. Stream loads are generally higher in May and June, coincidental with "flushing" of pesticides by spring rains (Pereira and Rostad 1990). During this time, the

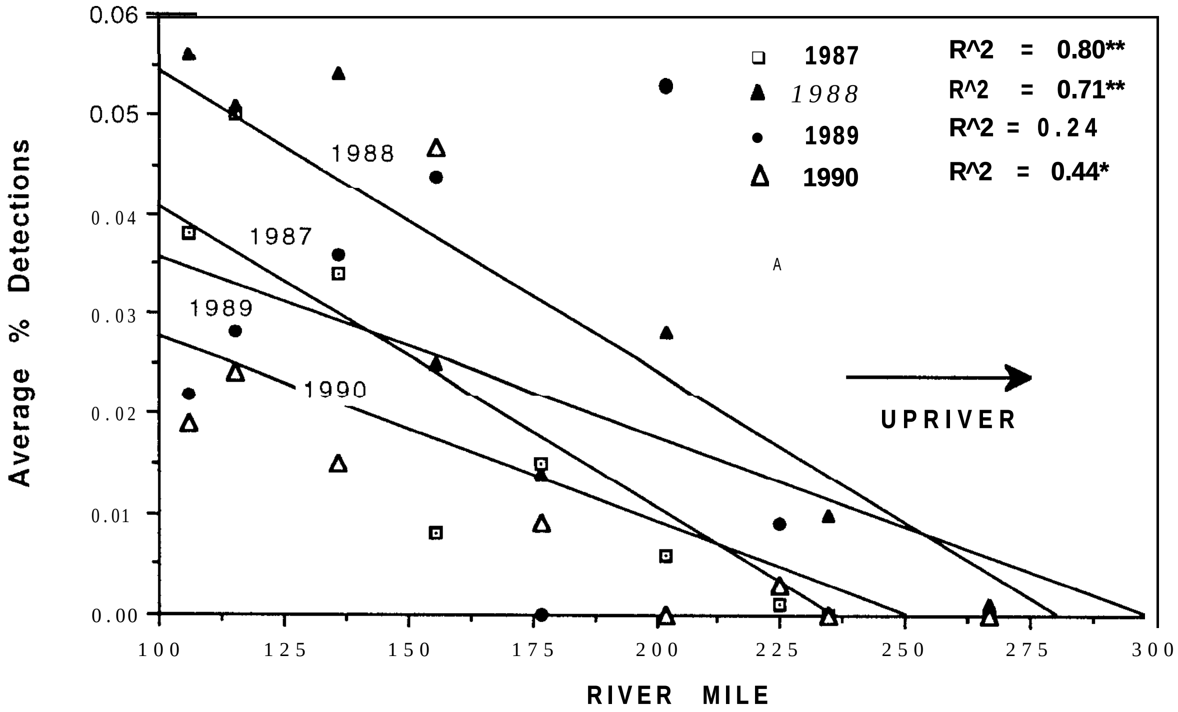


Figure 61. Trend of average % detections of chlorinated hydrocarbons as a function of river mile by year for 1987-1990. Trends that are significant at the 95% confidence level are coded as: ~ for P<0.1, * for p<0.05 ** for p<0.01 and *** for p<0.00l.

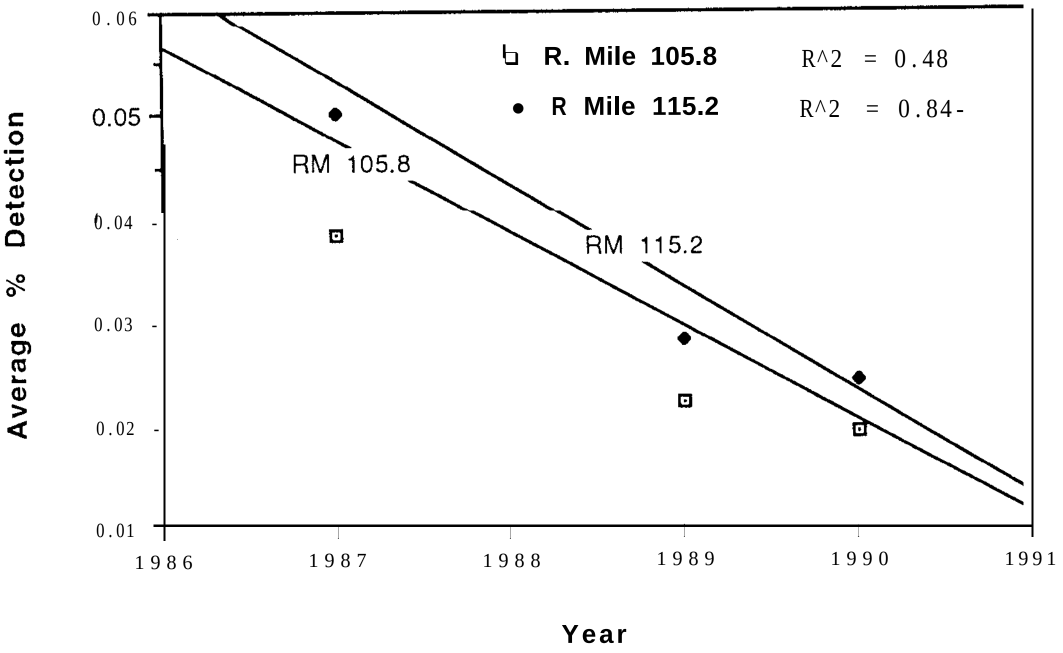


Figure 62. Comparison of average % detections of volatile chlorinated hydrocarbons by year for the two stations closest to the mouth of the Mississippi River at RM 105.8 and RM 115.2. Trends that are significant at the 95% confidence level are coded as: ~ for P<0.1, * for p<0.05, ** for p<0.01 and *** for p<0.001.

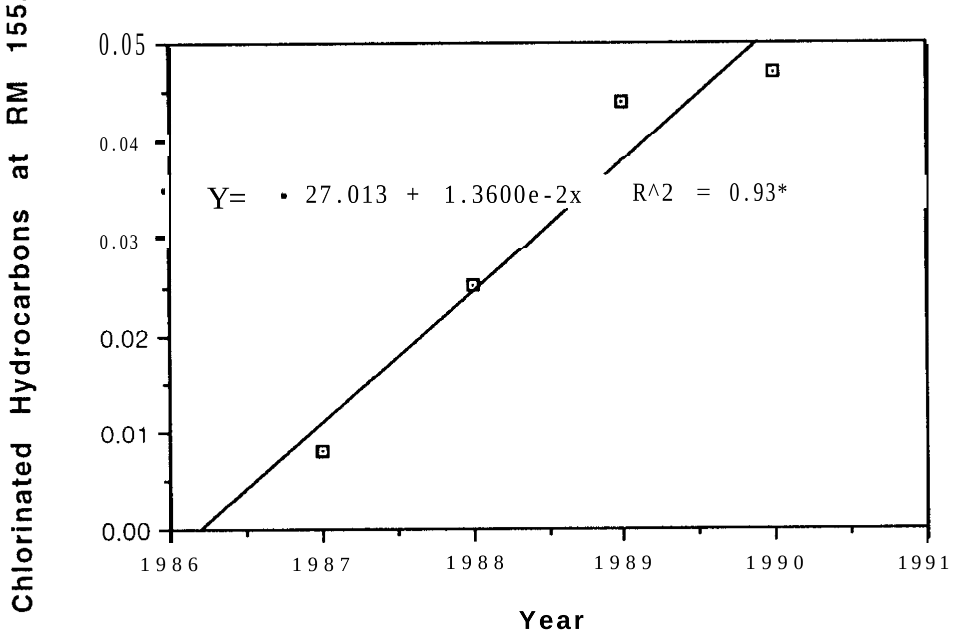


Figure 63. Trend of average % detections of chlorinated hydrocarbons by year for RM 155.6 by year for 1987-1990. Trends that are significant at the 95% confidence level are coded as: ~ for P,0.1, * for p<0.05, ** for p<0.01 and *** for p<0.001.

highest concentrations for most pesticides are observed upstream of the study area where river discharge is smaller and pesticides applied in the Midwest enter the system. Some pesticides show an increased concentration downstream of the convergence of the Yazoo River, which drains rice and cotton farmland. Pereira and Rostad (1990) estimate that 25–30% of the pesticide load enters the Atchafalaya River as 30% of the Mississippi's discharge is diverted to account for pesticide load decreases downstream of the Old River Outflow Channel. Pereira and Hostettler (1993) explained the year-round detection of pesticides results from pesticide-contaminated alluvial aquifers releasing their load as the river approaches base flow in fall and winter.

Environmental Working Group

In October 1994, the Environmental Working Group released a report on herbicides in drinking water from the Mississippi River. The nonprofit organization employed data from USGS, the Ciba Corporation, and the Jefferson Parish Water Quality Laboratory to estimate cancer risk from herbicides in drinking water. The report, entitled *Tap Water Blues*, faults EPA for what it finds to be dangerously weak standards for cancer-causing herbicides and recommends limiting these toxins at their source. The Jefferson Parish Water Quality Laboratory reported detections for each of the five herbicides: atrazine, alachlor, cyanazine, metolachlor, and simazine, in 83–93% of the 200 available samples. The study reiterates the seasonal variability that results in higher contamination levels during summer months and the potential for increased risk from the simultaneous exposure to more than one herbicide and their various metabolites.

For the purpose of their study, the Environmental Working Group applied federal pesticide standards for food to drinking water. It found Louisiana residents who use the Mississippi River as their drinking water source to be exposed to a cancer risk that is 8.4 times the EPA acceptable level of one in one million. For children and infants, the estimated risk is higher still. The fate of these herbicides as they enter the Gulf is not well known. The effects on the Barataria-Terrebonne estuarine system is not documented, but the detection of herbicides in Jefferson Parish water is a cause of concern for estuary management.

Basinwide Analysis of Toxics

Trends

Under its Ambient Water Monitoring Program (reported in the Water Quality Inventory) LDEQ maintains a network of sampling stations in the two basins where samples are collected regularly, usually monthly, and analyzed for a number of constituents. The constituents of interest for this basinwide analysis are the six toxic metals: arsenic, cadmium, chromium, copper, lead, and mercury. In 1980 and 1981, the earliest years used in this study, seven stations were sampled in Barataria basin and six stations in Terrebonne basin. By 1992–93, there were 11 in Barataria and 23 in Terrebonne. An analysis over time by station would only allow a look at the seven long-term Barataria stations and the six long-term Terrebonne stations, which is presented elsewhere in this report.

By averaging over all stations available for a given sampling period (i.e., each data point is averaged over two years), the basinwide average of each metal for a sampling period can be obtained. A disparate number of stations in different years means the basin average for a certain year is more or less representative—the more stations, the better representative the average. However, there is no indication the average data for any year are not representative of the basins. The reader also should remember that LDEQ changed its handling of the metal samples so that the 1993 data are for dissolved metals rather than total metals.

Figure 64 shows the basinwide averages for copper by 4-yr intervals; the fitted curve depicts the average across both basins. Copper concentrations in ambient water have been declining since the early 1980s, possibly earlier, and are now fairly low. Despite the change in chemical analyses from total to dissolved in 1991, the concentrations of the three metals are declining smoothly even without the 1993 data. Figure 65 shows the Barataria basin averages for lead, arsenic, and chromium, which also are declining. Figure 66 shows the basin averages decline over time in Terrebonne basin for the same metals. As before, the basin yearly averages are declining. In addition, a t-test shows that the differences across basins are insignificant for the metals. Therefore, the analysis of other metals are averaged over both basins in the estuarine system. Figure 67 shows the system averages of cadmium and mercury over time. While cadmium is declining over time, mercury is remaining fairly constant at about 0.25 $\mu g\ l^{-1}$. It is unclear why mercury does not decline over time as most metals do. Figure 68 represents the average over the six metals and all stations by year.

Metals in Excess

To test for uniformity across the basins, individual stations can be examined for their deviations from the mean. We can construct table 20 using a standard: if a station's concentration averages are twice the standard deviation above the mean, then the metal can be considered "excessive." Table 20 shows that in 1981 three stations in Barataria basin had excessive metals (mercury, copper, lead, and arsenic), while only one station in Terrebonne had an excessive metal (mercury). By 1993 the trend reversed and Terrebonne basin now has six stations showing excessive metals while Barataria has only two. However, Bayou Segnette near Westwego in Barataria basin is high in four metals (arsenic, lead, cadmium, and copper), which may be due to urban runoff because this station is near the west bank of the Mississippi River in the developed area of Jefferson Parish. The stations in Terrebonne with high metals are widely scattered, but most are

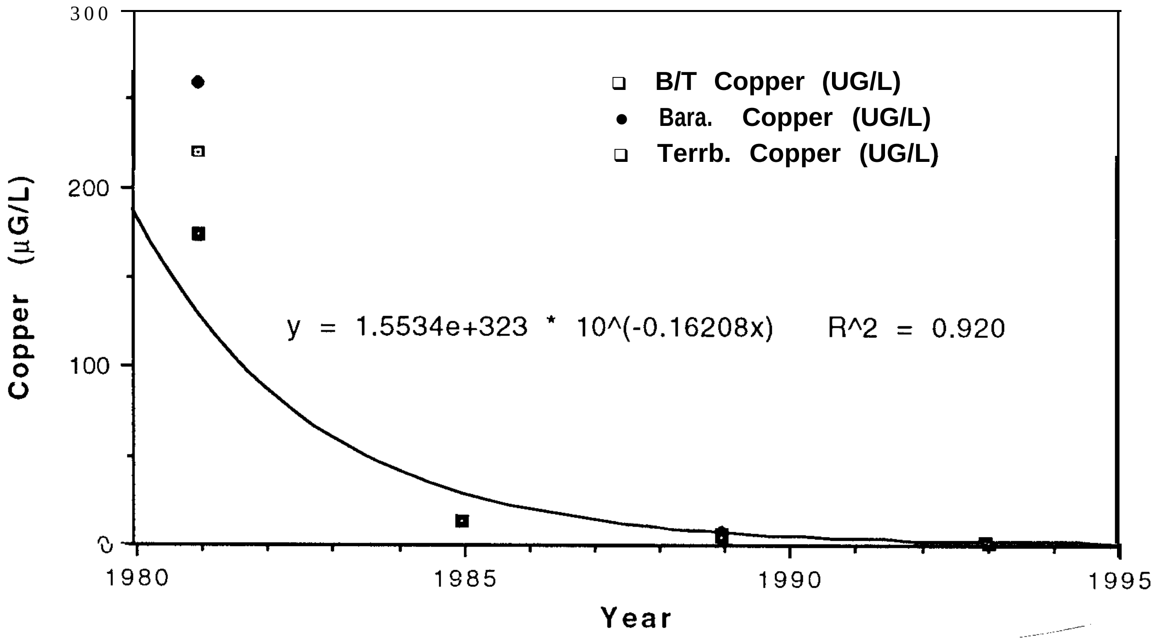


Figure 64. Basinwide ambient copper in water, averaged across all stations within each basin and for basins combined (n varies by year, n varies between basins), by sampling period (4-yr interval). Trends that are significant at the 95% confidence level are coded as: ~ for $p<0.1$, * for $p<0.05$ ** for $p<0.01$ and *** for p<0.00l.

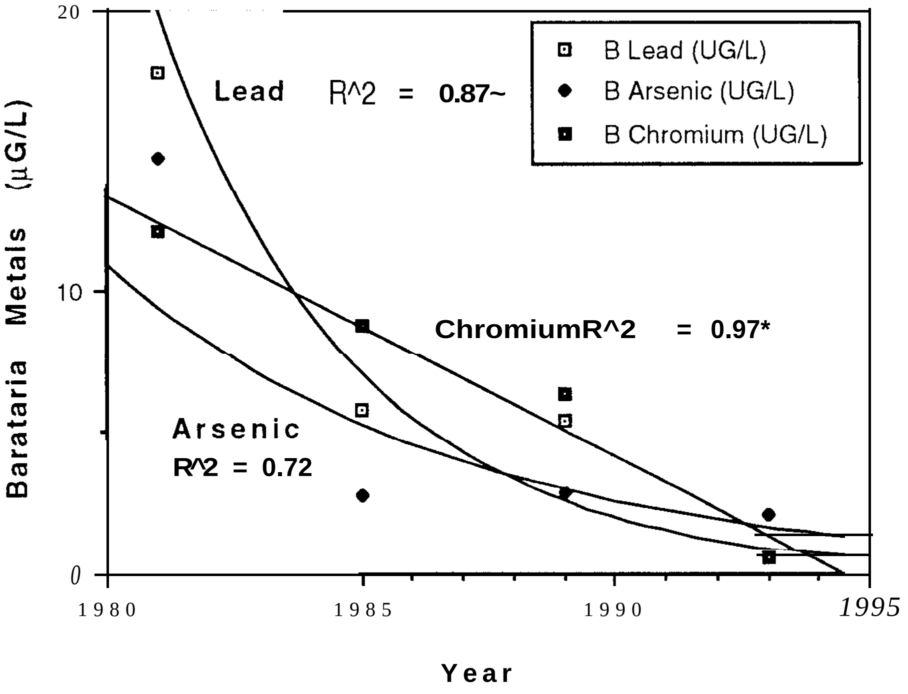


Figure 65. Basinwide ambient lead, arsenic and chromium, averaged across all stations within Barataria basin (n varies by year), by sampling period (4-yr interval). Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<0.001.

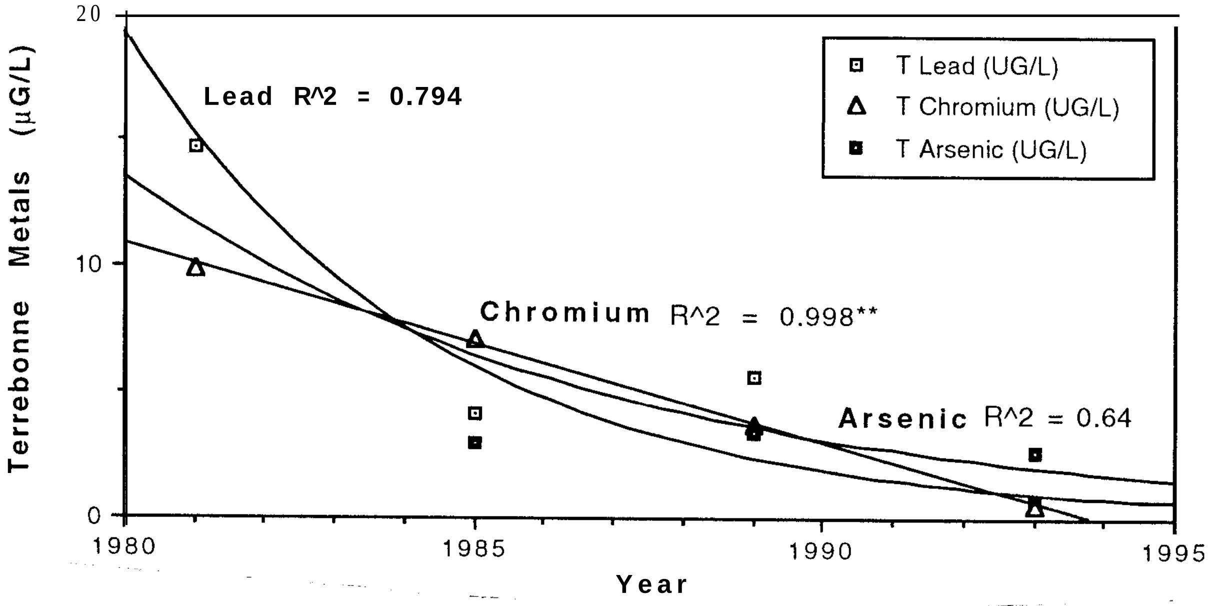


Figure 66. Basinwide ambient lead, arsenic and chromium, averaged across all stations within Terrebonne basin (n varies by year), by sampling period (two years combined), by sampling period (4-yr interval). Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05, ** for p<0.01 and *** for p<0.001.

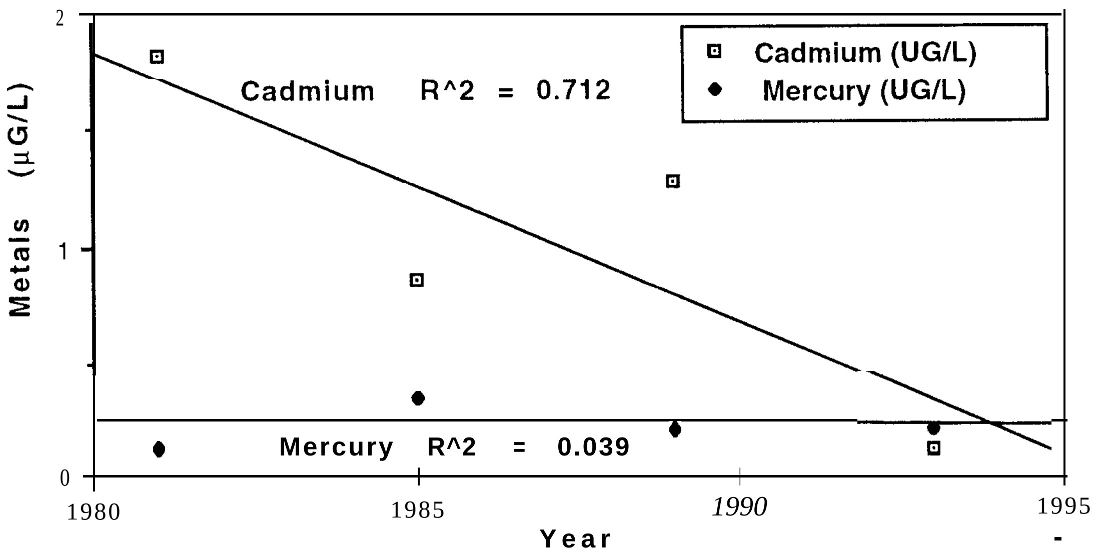


Figure 67. Basinwide ambient cadmium and mercury, averaged across all stations for both Barataria and Terrebonne basins combined (n varies by year, n varies between basins), by sampling period (4-yr interval). Trends that are significant at the 95% confidence level are coded as: ~ for p<0.1, * for p<0.05 ** for p<0.01 and *** for p<0.001.

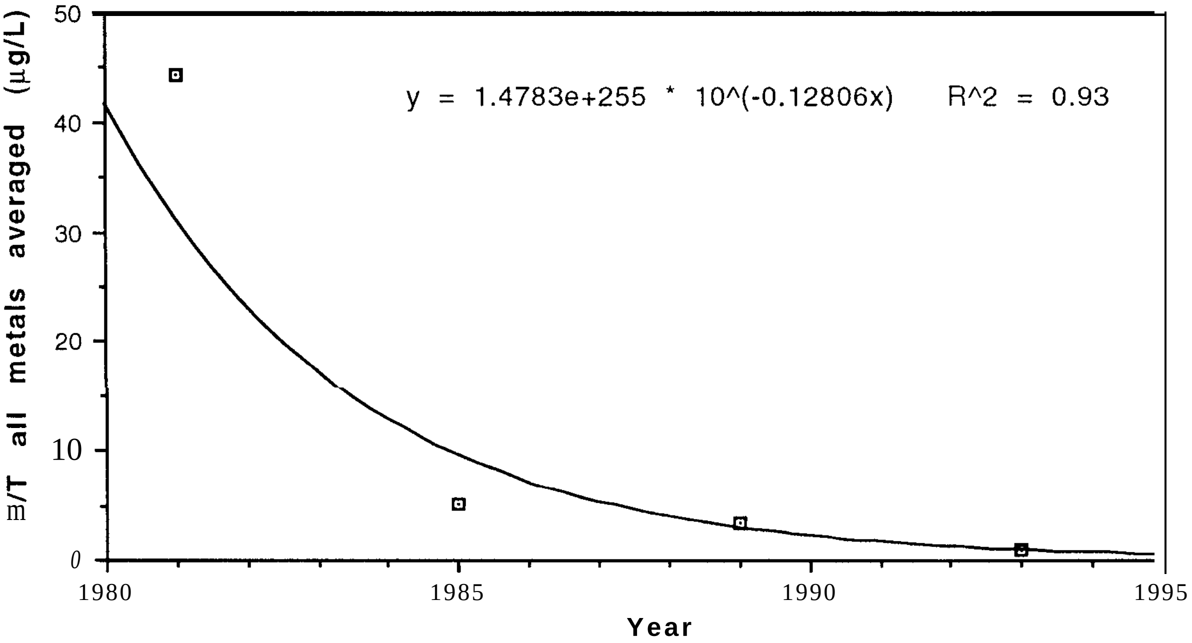


Figure 68. Basinwide ambient total metals (copper, lead, arsenic, chromium, cadmium and mercury), averaged across all stations for both Barataria and Terrebonne basins combined (n varies by year, n varies between basins), by sampling period (4-yr interval). Trends that are significant at the 95% confidence level are coded as: -for $p<0.1$, * for $p<0.05$ ** for $p<0.01$ and *** for $p<0.001$.

near developed areas, which also may indicate nonpoint sources as the cause of the metal concentrations.

Conclusions

Toxic discharge data for the estuary show an increase over time that may represent a real increase or merely an increase in reporting rigor. Toxics releases above the assimilation capacity of the estuary can have a negative impact. Efforts should be made to reverse the trend of increasing releases if indeed it represents actual increases rather than more rigorous reporting. Ambient metals in water appear to be declining uniformly in the estuary, except for mercury. Because mercury is a strong bioaccumulator, efforts should be made to lower concentrations to the natural background, if indeed, they are currently above background. Some stations in the basins in 1993 have metals concentrations that may indicate they are excessive when compared to basin averages. The causes of these elevated levels should be investigated. (See below for a more detailed temporal and spatial analysis of ambient metal concentrations.)

Table 20. Metals in excess* at stations in the Barataria and Terrebonne basins.

Basin	Station Number	Year	Location	Excessive* Metal
Barataria	0023	1981	Bayou Lafourche near Donaldsonville	Mercury, Copper
Barataria	0038	1981	Bayou Lafourche at Cutoff	Lead
Barataria	0008	1981	Little Lake at Temple	Arsenic
Terrebonne	0037	1981	Houma Navigation Canal at Bayou Petit Caillou	
Barataria	0295	1993	Bayou Lafourche near Golden Meadow	Chromium
Barataria	0296	1993	Bayou Segnette	Arsenic, Lead,
Terrebonne	0080	1993	Lower Grand River at Bayou Sorrel	Copper
Terrebonne	0144	1993	Lake Verret at Attakapas Landing	Lead
Terrebonne	0335	1993	False River south of New Roads	Arsenic
Terrebonne	0336	1993	Bayou Choctaw west of Port Allen	Copper
Terrebonne	0345	1993	Bayou Chauvin near Houma	Arsenic
Terrebonne	0348	1993	Bayou Grand Caillou south of Houma	Copper

*Excessive metals are twice the standard deviation above the mean

Status and Trends of Contaminants in the Barataria-Terrebonne Estuarine System

Data Sources

Various data sources were initially considered for the analyses of historical trends and current status. Trend analyses require a sampling program extending over a long period (e.g., five or more years), while recent data are useful to assess current status. These requirements limited the useful data to a few fairly large data sets. Each of the data sets used for the analyses of status and trends is described below. Data sets that, for various reasons, did not appear well suited for this particular project were LDEQ's data on organic pollutants in water, sediment and tissue samples, data on contaminants for Goose Bayou collected by the Jefferson Parish Department of Public Works, and data on copper and zinc in oysters from a study conducted by the Gulf Coast Shellfish Sanitation Research Center (Kopfler 1966).

Louisiana Department of Environmental Quality's Water Quality Monitoring Program

The LDEQ data set contains data on total levels of six elements (arsenic, cadmium, chromium, copper, mercury, and lead) in water for the period 1978 to present. Water samples were consistently collected at a depth of one meter. This program's overall data sets date back as far as 1958 for some sites, but no pollutant data were collected until 1978. Beginning 1991, dissolved metal levels have been determined instead of total metal levels. This change reflects the scientific consensus that dissolved element levels are generally better predictors for toxic effects than total element levels (combining dissolved element levels with those associated with particles such as suspended sediment). In 1991 many new sampling sites were added, and some of the old sites were dropped. Because of this change in the data set, data from the period 1978–1990 were used for trend analyses, while data from the period 1991–1994 (data up to March were available) were used for the determination of status. Since 1988, sampling occurred mostly on alternate months, while before that time sampling was accomplished monthly (with some gaps). Mercury analyses were performed on a less-regular basis than those for the other elements.

Sites used in the trends analyses are listed in table 21, and their geographic location is shown in figure 69. For the Barataria basin, all seven sites for which data were collected in the period 1978–1990 were used in trends analyses. In the Terrebonne basin, two sites (Lake Verret at Attakapas Landing near Georgia and Lake Verret near Pierre Part) were added to the sampling program as of January 1987. These two sites were not used in the trends analyses because of insufficient temporal coverage, so data from six sites in the Terrebonne basin were used for these analyses.

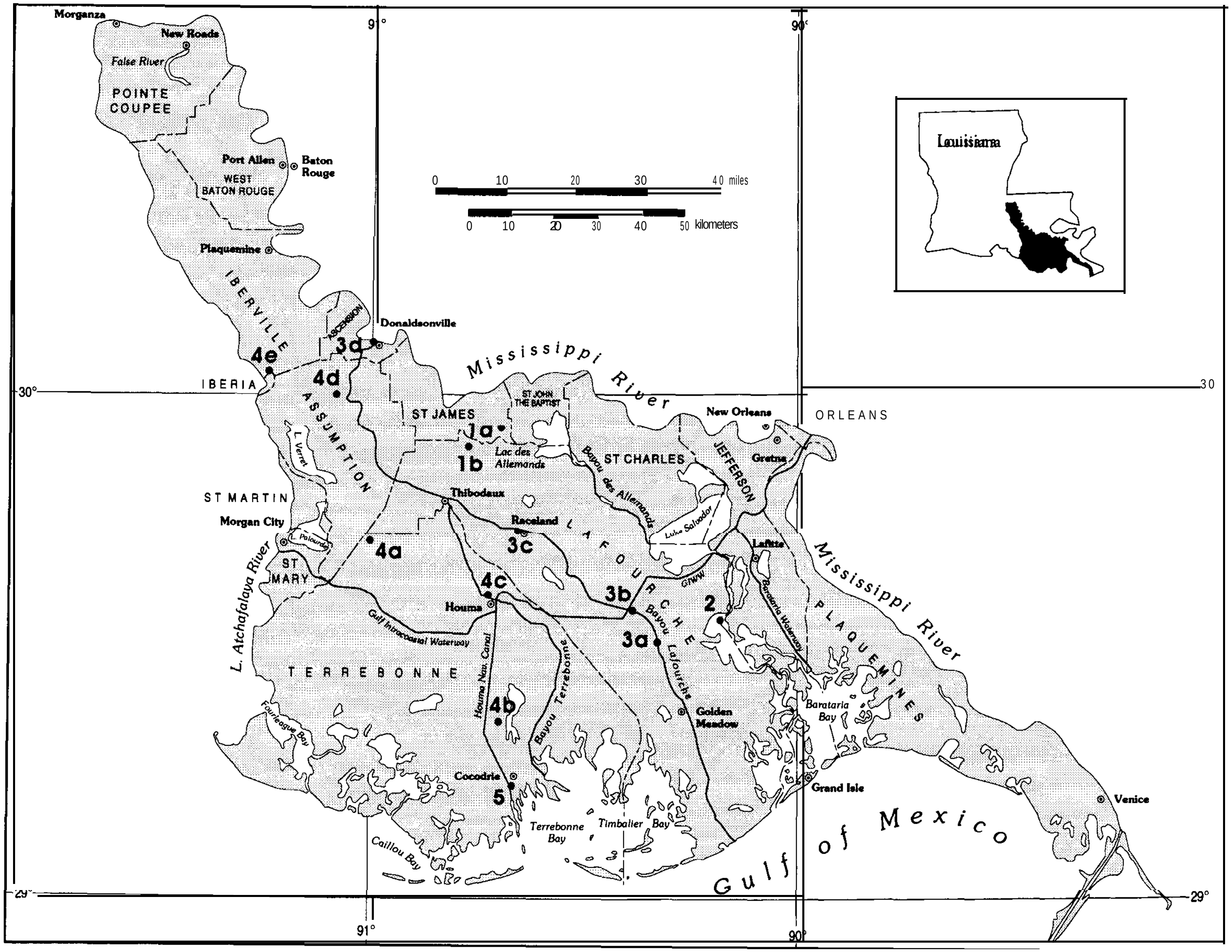


Figure 69. Location of LDEQ's water quality monitoring stations used in analyses for temporal trends in contaminant levels in water of Barataria and Terrebonne basin water bodies (see table 21 for listing of site names.

Table 21. Listing of LDEQ's water quality monitoring stations used in the analyses of temporal trends in levels of contaminants in water of Barataria and Terrebonne basin water bodies. Site numbers correspond to those on the map in figure 69. (Numbers for latitude and longitude are xx°xx'xx")

	site	latitude	longitude	name used sites
Barataria Basin				
Bayou Chevreuil near Chackbay	1a	295442	904345	Chackbay
Grand Bayou near Chackbay	1b	295327	904703	
Little Lake at Temple	2	293400	901000	Little Lake
Bayou Lafourche at Cut Off	3a	293244	902021	
Bayou Lafourche at Larose	3b	293420	902302	Bayou Lafourche
Bayou Lafourche at Raceland	3c	294340	903531	
Bayou Lafourche near Donaldsonville	3d	300548	910000	
Terrebonne Basin				
Bayou Black at Gibson	4a	294105	905955	
Bayou Grand Caillou at Dulac	4b	292258	904255	
Bayou Terrebonne at Houma	4c	293555	904307	Terreb. Bayous
Grand Bayou at Grand Bayou	4d	300055	910752	
Lower Grand River at Bayou Sorrel	4e	300940	912014	
Houma Navigational Canal near Cocodrie	5	291400	904000	HNC

Sites used for the status determinations on this data set are listed in table 22; their locations are shown in figure 70. This data set contains dissolved element levels for 11 sites in the Barataria basin and 23 sites in the Terrebonne basin.

U.S. Geological Survey Water Resources Data for Louisiana

This large USGS data set contains data on pollutant levels for the period 1975–1994. As this monitoring program appears aimed at addressing specific questions at specific locations and times, it is not ideally suited for investigating general trends or assessing the contamination status for a wide geographic area. Most of the data were collected for the period 1975–1981. For this period, sampling frequencies among nine sites in the Barataria and Terrebonne basins ranged from 1 to 21 sampling dates per year. Consequently, only two sites (Bayou Lafourche at Larose and Mississippi River Southwest Pass) had sufficient observations to permit the use of these data for trends analyses. Other considerations (e.g., contaminant concentrations relative to detection limits) have resulted in limitations for this data set to levels of total recoverable levels of arsenic, copper, nickel, lead, and zinc.

NOAA National Status and Trends Program

Two programs collecting environmental pollutant levels reside in the NOAA National Status and Trends Program: the Mussel Watch Project (MWP) and the National Benthic Surveillance Project (NBSP). MWP has been analyzing sediment and bivalves, while NBSP has analyzed benthic fish and sediment. Available data from the NBSP data set for the area under investigation are limited to two sites and three years (1984–1986), while the sediment component of the MWP data set is limited to eight sites in the Barataria-Terrebonne area and a maximum of two years (not extending beyond 1988) for each of these sites. This limits use of the National Status and Trends Program data to the bivalve tissue data on contaminants. For the Louisiana area, the bivalve sampled under this program is the oyster *Crassostrea virginica*.

Three replicates of composite samples of soft tissue of 20 individual oysters (7 cm– 10 cm) were analyzed for the period 1986–1991, while only one composite sample was collected starting in 1992. Data from 1994 are not yet available. The following determinations have been made on the oyster tissue samples:

major elements: aluminum, iron, manganese, and silicon;

trace elements: antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, tin, and zinc;

6 forms of DDT and its metabolites;

18 chlorinated pesticides other than DDT, including 9 PCBs (18 in samples collected in 1988);

18 polyaromatic hydrocarbons (PAHs).

Tissue lipid content also was determined.

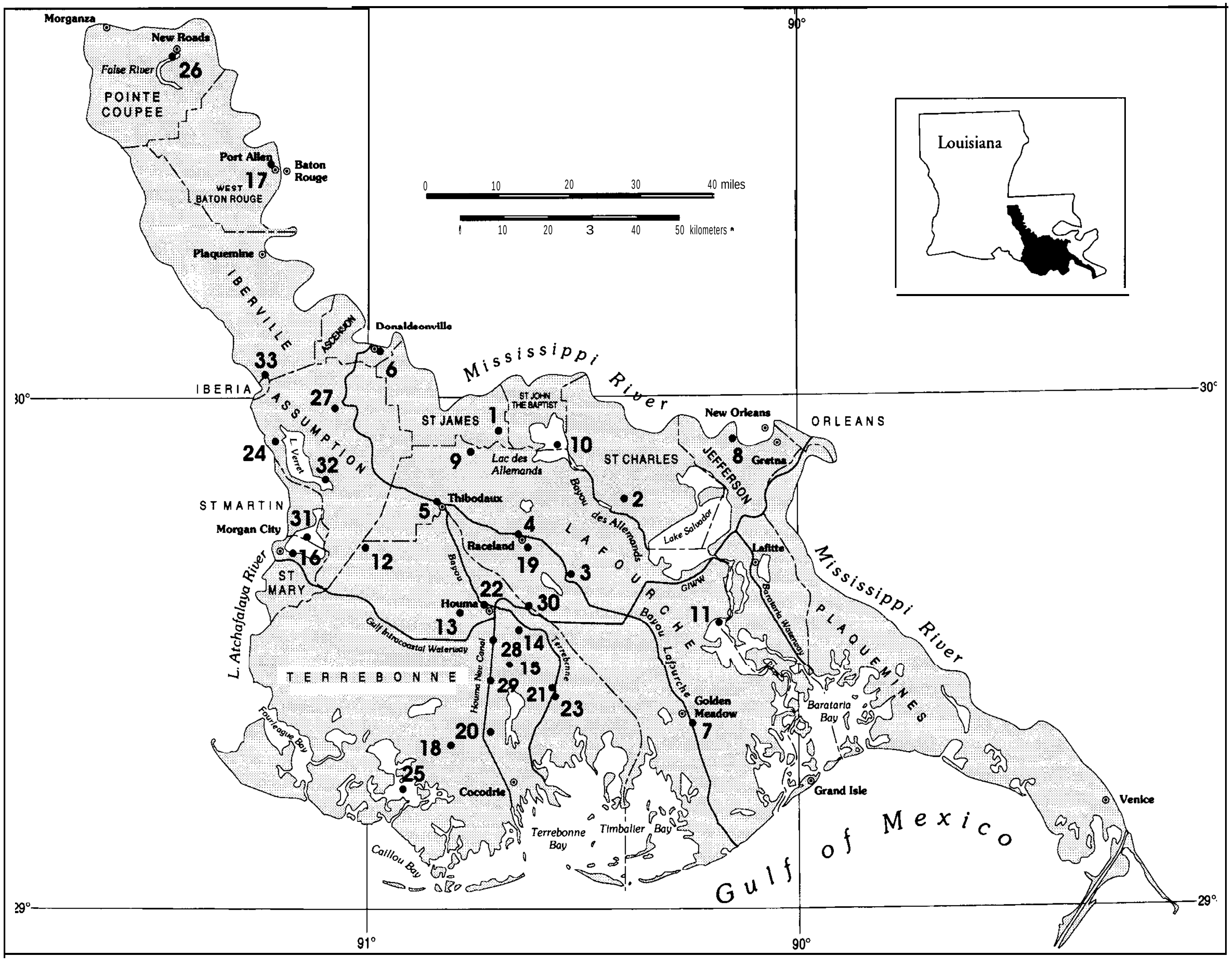


Figure 70. Location of LDEQ's water quality monitoring stations used in status analyses for contaminants in water of Barataria and Terrebonne water bodies (see table 22 for listing of site names).

Table 22. Listing of LDEQ's water quality monitoring stations used in status analyses for contaminants in water of Barataria and Terrebonne basin water bodies. Site numbers correspond to the numbers in figure 70. (Numbers for latitude and longitude are xx°xx'xx")

Barataria Basin	Site	Latitude	Longitude
Bayou Chevreuil near Chackbay	1	295442	904345
Bayou Des Allemands at Des Allemands	2	294926	902730
Bayou Lafourche at Lockport	3	293848	903212
Bayou Lafourche at Raceland	4	294340	903531
Bayou LaFourche at Thibodaux	5	294756	904905
Bayou Lafourche near Donaldsonville	6	300548	910000
Bayou Lafourche near Golden Meadow	7	292036	901446
Bayou Segnette near Westwego	8	295351	900924
Grand Bayou near	9	295327	904703
Lac Des Allemands north of Raceland	10	295357	903325
Little Lake at Temple	11	293400	901000

Terrebonne Basin	Site	Latitude	Longitude
Bayou Black at Gibson	12	294105	905955
Bayou Black west of Houma	13	293433	904836
Bayou Chauvin near Houma	14	293315	903938
Bayou Chauvin south of Houma	15	292810	903921
Bayou Chene southeast of Morgan City	16	293825	910552
Bayou Choctaw west of Port Allen	17	302635	912049
Bayou Dularge south of Houma	18	291906	905140
Bayou Folse north of Houma	19	294213	903722
Bayou Grand Caillou south of Houma	20	292257	904253
Bayou Petit Caillou south of Houma	21	292856	903445
Bayou Terrebonne at Houma	22	293555	904307
Bayou Terrebonne southeast of Houma	23	292852	903318
Belle River north of Morgan City	24	295431	911258
Caillou Lake south of Houma	25	291501	905517
False River south of New Roads	26	303645	912833
Grand Bayou at Grand Bayou	27	300055	910752
Houma Navigation Canal near Houma	28	293404	904255
Houma Navigation Canal south of Houma	29	292304	904346
Intracoastal Waterway east of Houma	30	293433	903619
Lake Palourde near Morgan City	31	294151	910559
Lake Verret at Attakapas Landing near Georgia	32	295030	910556
Lower Grand River at Bayou Sorrel	33	300940	912014

Data on contaminants in oysters have been used for status and trends analyses. Trends analyses were made on the data from the period 1986–1993, while status was determined from the data covering the period 1990–1993. The MWP data set contains data on eight sites in the Barataria-Terrebonne area. Data from Barataria Bay/Turtle Bay cover only 1988 and were therefore not used in analyses. The other seven sites are listed in table 23, and their geographic locations are shown in figure 71.

Environmental Monitoring and Assessment Program, EMAP-Estuaries, Louisiana Province

The Environmental Monitoring and Assessment Program is aimed at evaluating the condition of the estuarine resources in the Louisiana Province. The large data set contains data on 198 sites, of which 18 fall in the Barataria-Terrebonne estuarine system. The available data reflect sampling that was done during the summers of 1991 and 1992 (data for 1993 have not yet been released). The probability-based sampling design of this program means that different sites were sampled in the two years. Also, the data set contains one data point per variable per site, limiting the scope of statistical analyses that can be performed on this data set. The two-year duration of the current data base is insufficient for trends analysis. However, the data are well suited for the assessment of the current pollution status. The data set combines assessment of sediment toxicity (using the mysid *Mysis bahia* and the amphipod *Ampelisca abdita*), pollutant concentrations in sediment and pollutant tissue levels in fish and shrimp (all at the same sites). The following contaminants have been measured in shrimp and several species of fish: metals/elements (aluminum, arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, tin, and zinc), 15 pesticides, and 21 congeners of PCBs.

Contaminants measured in surface sediment include 27 alkanes; 43 PAHs; 20 congeners of PCBs; tetrabutyltin, tributyltin, dibutyltin, and monobutyltin; 23 pesticides; and the elements aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, tin, and zinc. To limit the size of the data set, the following variables were excluded from the analyses: iron (on the basis of its low toxicity), toxaphene, mirex, and alpha- and beta-endosulfan. The latter were all present below their detection limits (0.10, 0.17, 0.25, 0.25, and 0.25 ng g^{-1}, respectively).

By design, spatial coverage is limited to the estuarine portion of the study area. The 18 sampling sites in the Barataria-Terrebonne area are shown on figure 72; site names are listed on the tables that describe the results of the status assessments for this data set.

Statistical Treatment

Analyses were carried out using statistical analysis and graphics programs Statgraphics Plus version 6 (Manugistics 1992), Deltagraph Pro 3.1.1 (DeltaPoint 1993), and StatView 4.2 (Abacus Concepts).

Table 23. National Status & Trends Program's sites in the Barataria and Terrebonne basins for which contaminant levels in oyster tissues have been collected as a part of the Mussel Watch Project. Sampling sites were not combined, as an ANOVA on log-transformed data revealed a significant difference among these sites. Site numbers correspond to those on the map of figure 71, while site names are used in subsequent figures and graphs.

Site	analyses used in	site name designation	map site #
Atchafalaya Bay, Oyster Bayou	status & trend	ABOB	1
Caillou Lake, Caillou Lake	status & trend	CLCL	2
Terrebonne Bay, Lake Barre	status & trend	TBLB	3
Terrebonne Bay, Lake Felicity	status & trend	TBLF	4
Barataria Bay, Bayou St. Denis	status & trend	BBSD	5
Barataria Bay, Middle Bank	status & trend	BBMB	6
Mississippi River, Tiger Pass	status & trend	MRTP	7

Detection Limits

A general problem in the analyses and evaluation of pollutant data is the presence of values below the detection limit. Often these values are set to 0 in data reports. However, this leads to an underestimation of true pollutant levels. Values reported to be "less than detection limit" were therefore set midway between 0 and the reported detection limit. For example, if the detection limit for a compound was 10 ng l^{-1}, a value "less than detection limit" was replaced by a value of 5.0001 ng l^{-1} (the 0.0001 was used to differentiate this value from a value reported as 5.0). This reduces the chance of underestimating actual pollutant levels. However, it does lead to an overestimation for those contaminants present at concentrations usually less than the detection limit. Variables generally present below the detection limit of the used methodology were therefore usually excluded from analyses.

Data Transformation

Before application of quantitative statistical techniques, the data set must conform to a series of rules. These are that (1) the data follow a normal distribution, (2) the variance of the sample is independent of the mean, and (3) components of the variance are additive. Most of the data sets are not normally distributed. Also, the variance frequently increases with the mean. One way of overcoming these problems is to transform the raw data. There are several different transformations commonly applied such as the logarithmic transformation or the square root transformation. In this study we used the $\log_{10}$ transformation. This transformation greatly improved the normality of the data and

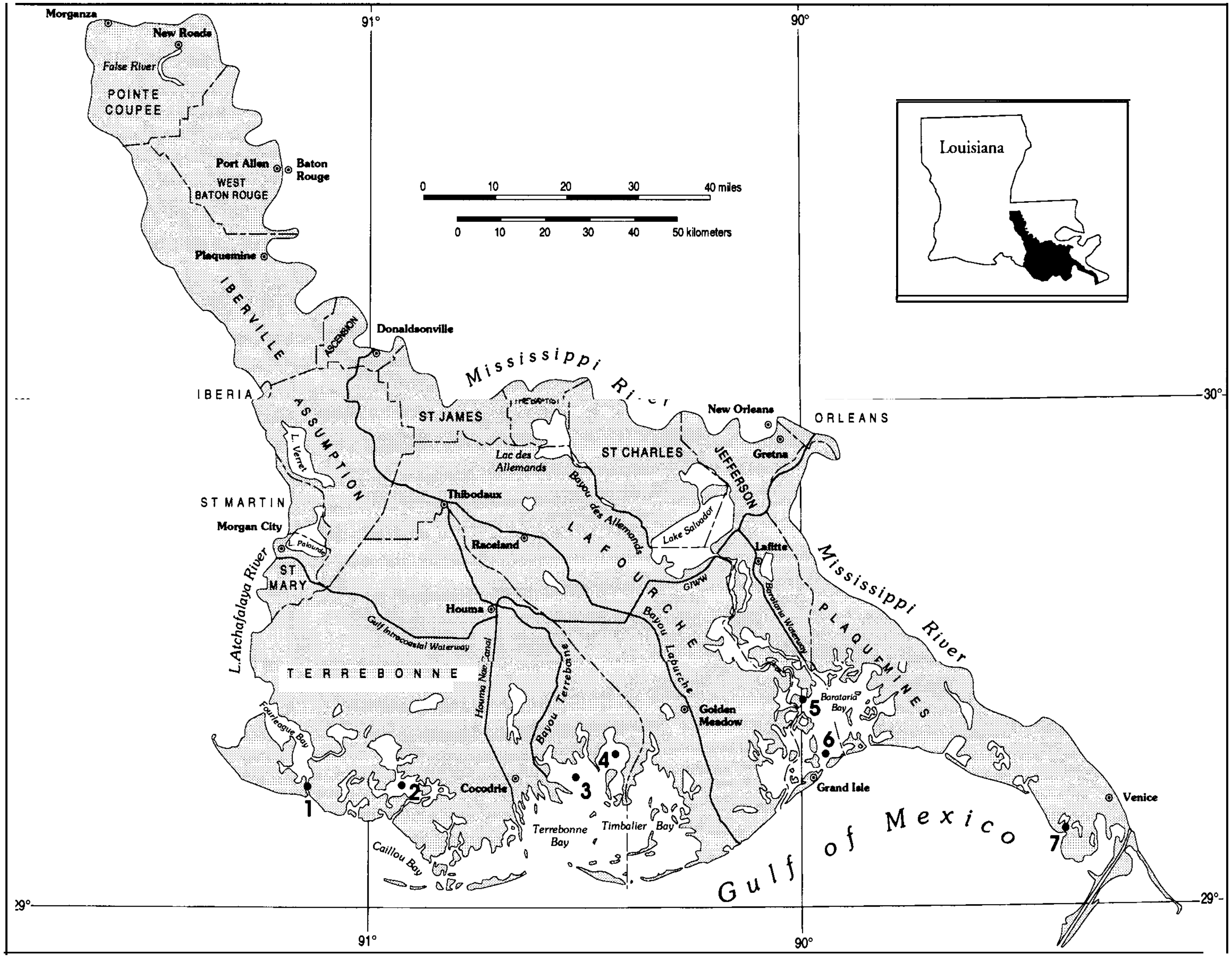


Figure 71. Location of NOAA Mussel Watch Project stations used in analyses for status and trends in contaminant levels in oysters (see table 23 for listing of site names).

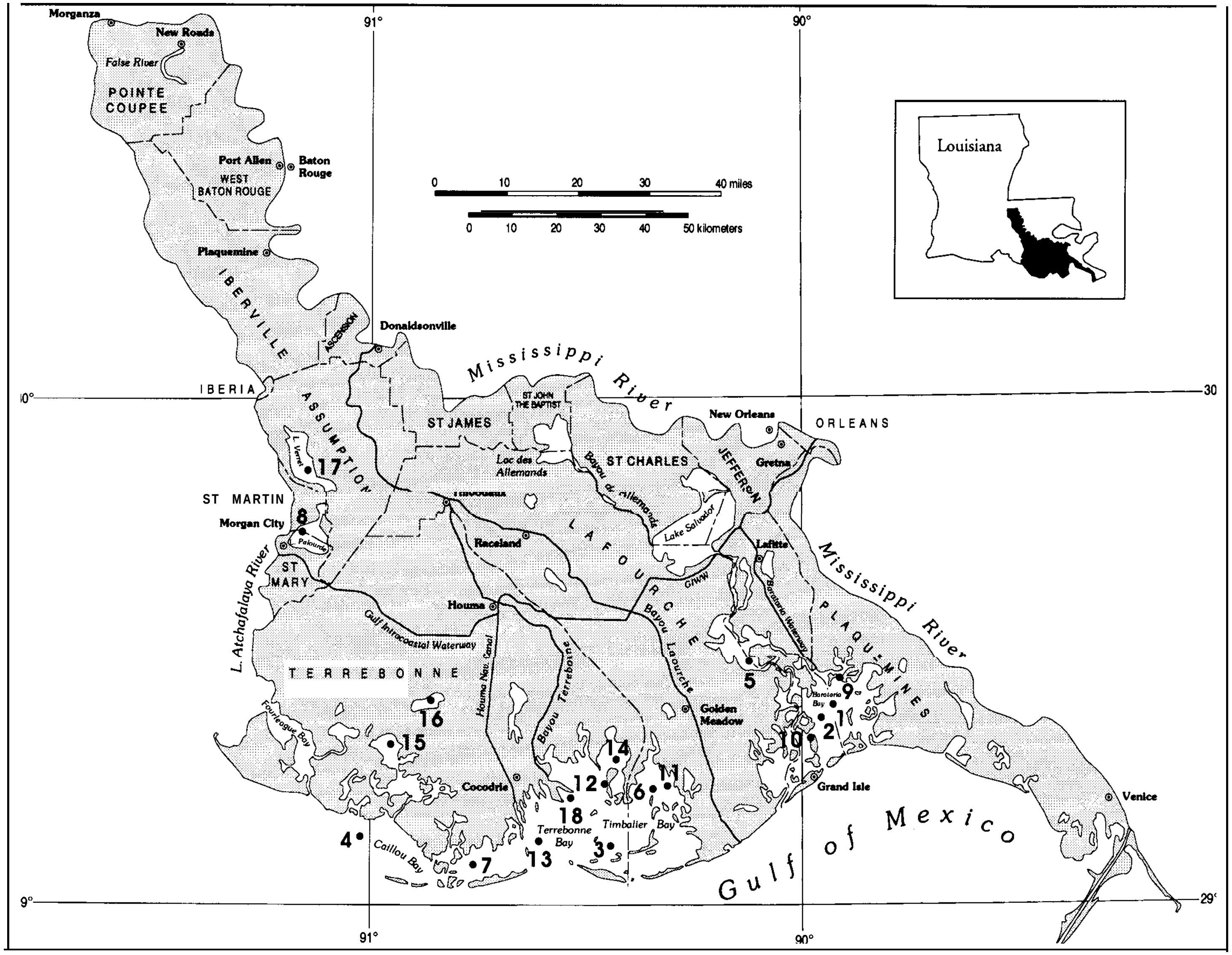


Figure 72. Location of EMAP-E (Louisiana Province) stations used in analyses for status in sediment toxicity as well as in contaminant levels in sediment and (shell) fish. Sites 1-8 were sampled in 1991; sites 9-18 were sampled in 1992.

rendered the variance largely independent of the mean. The normal probability plot and a Chi-square test were used subsequently to verify the normality of the transformed data. An example of the transformation procedure is illustrated in figures 73 and 74. Figure 73 shows a normal probability plot of Cu levels in water (from the LDEQ data set) to indicate a very poor fit to a normal distribution. Figure 74 shows that the data fit the normal distribution more closely following the $\log_{10}$ transformation.

Descriptive Statistics and Data Grouping

A variety of summary statistics are routinely calculated for the raw and transformed data. These statistics include the average, median, mode, geometric mean, variance, standard deviation, standard error, minimum, maximum, range, lower and upper quartiles, interquartile range, skewness, kurtosis, and coefficient of variation. The average, median, mode, and geometric mean measure the central tendency of the data, while the variance, standard deviation, range, and interquartile range measure spread or dispersion. The skewness coefficient measures the data symmetry. The kurtosis coefficient reveals how flat or steep the data distribution is with respect to Gaussian or normal distribution.

To obtain more representative data sets and to reduce the number of analyses run, data from several sites within the same hydrologic unit (e.g., lake, bayou, canal) were combined (this was the case for the Bayou Lafourche sites in the LDEQ data set). Also, sites within the same basin for which contaminant levels did not differ significantly (in ANOVA on log-transformed data) were combined. The lack of significant difference between the sample means at the $p = 0.05$ level, as determined by a one-way analysis of variance or a paired t-test, was used as a criterion for this procedure.

In the section on the determination of the status of pollutant levels, recent data (since 1990 or 1991, depending on data set specifics) were pooled. Following $\log_{10}$-transformation, these data were subjected to an analysis of variance to determine if there were differences among sites. Where sites differed significantly, post-hoc comparisons (Fisher's PLSD test) were run to determine which sites differed from other ones.

In addition to combining sites for trends analyses, several variables were combined to keep the number of variables to be included in the analyses to a manageable level. The large number of PCBs (18 or 21 congeners in the data sets) were combined into one value. Similarly, PAHs (with data sets containing, e.g., the 21 priority pollutants in this group) were combined into low-molecular-weight PAHs (lmw-PAHs) and high-molecular-weight PAHs (hmw-PAHs).

Time Series Analyses

Stations with total sampling record less than 13 years of data were excluded to eliminate local variability and short-term fluctuations. For LDEQ's water quality data, two periods in the 13-year record were analyzed separately for trends because of apparent

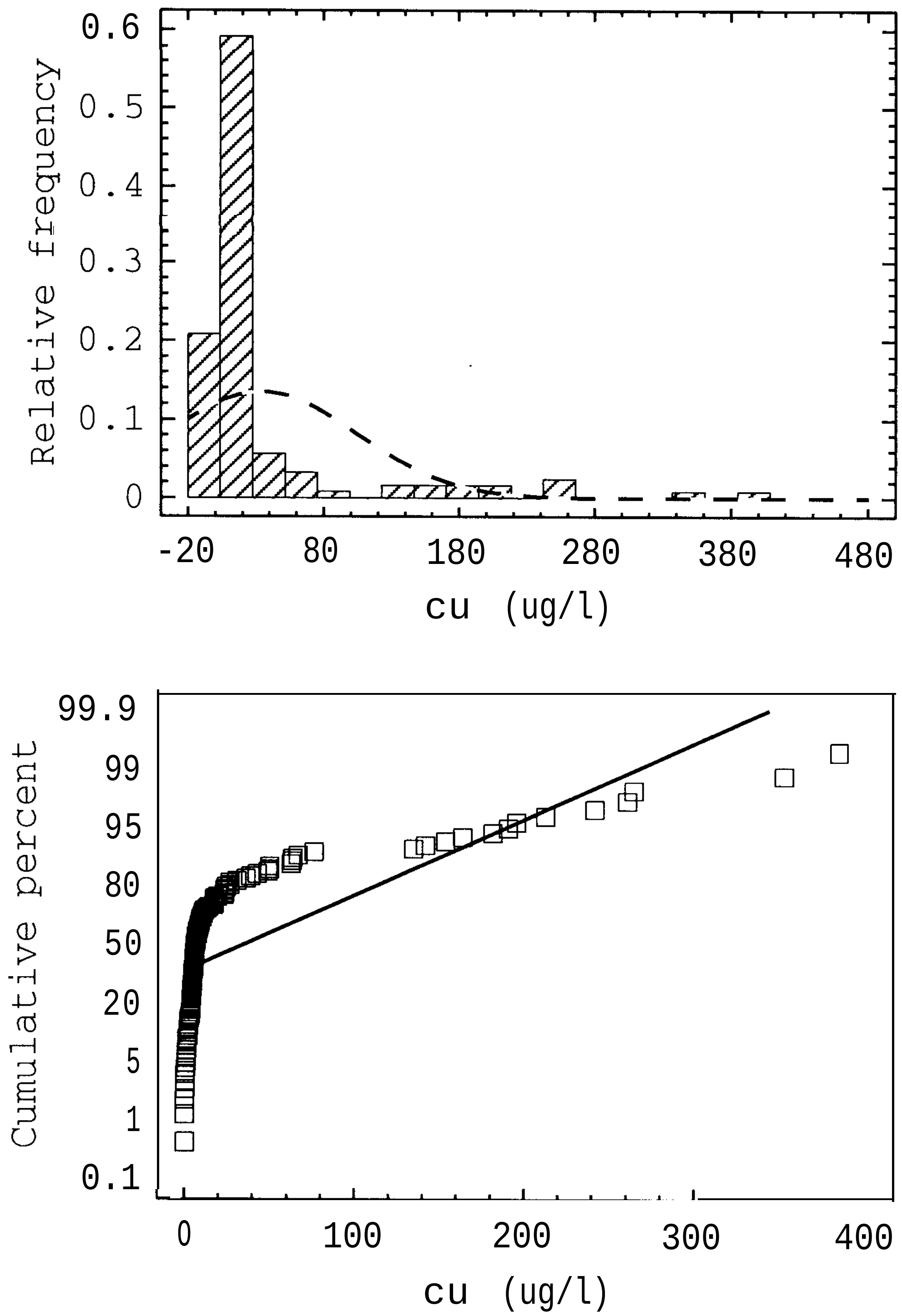


Figure 73. Relative frequency histogram (upper panel) and normal probability plot (lower panel) of dissolved copper (Cu) concentrations in water samples in the Houma Navigation Canal near Cocodrie (period 1978-1990 from the LDEQ Water Quality Monitoring Program). Significant deviations from the normal distribution are indicated in both plots.

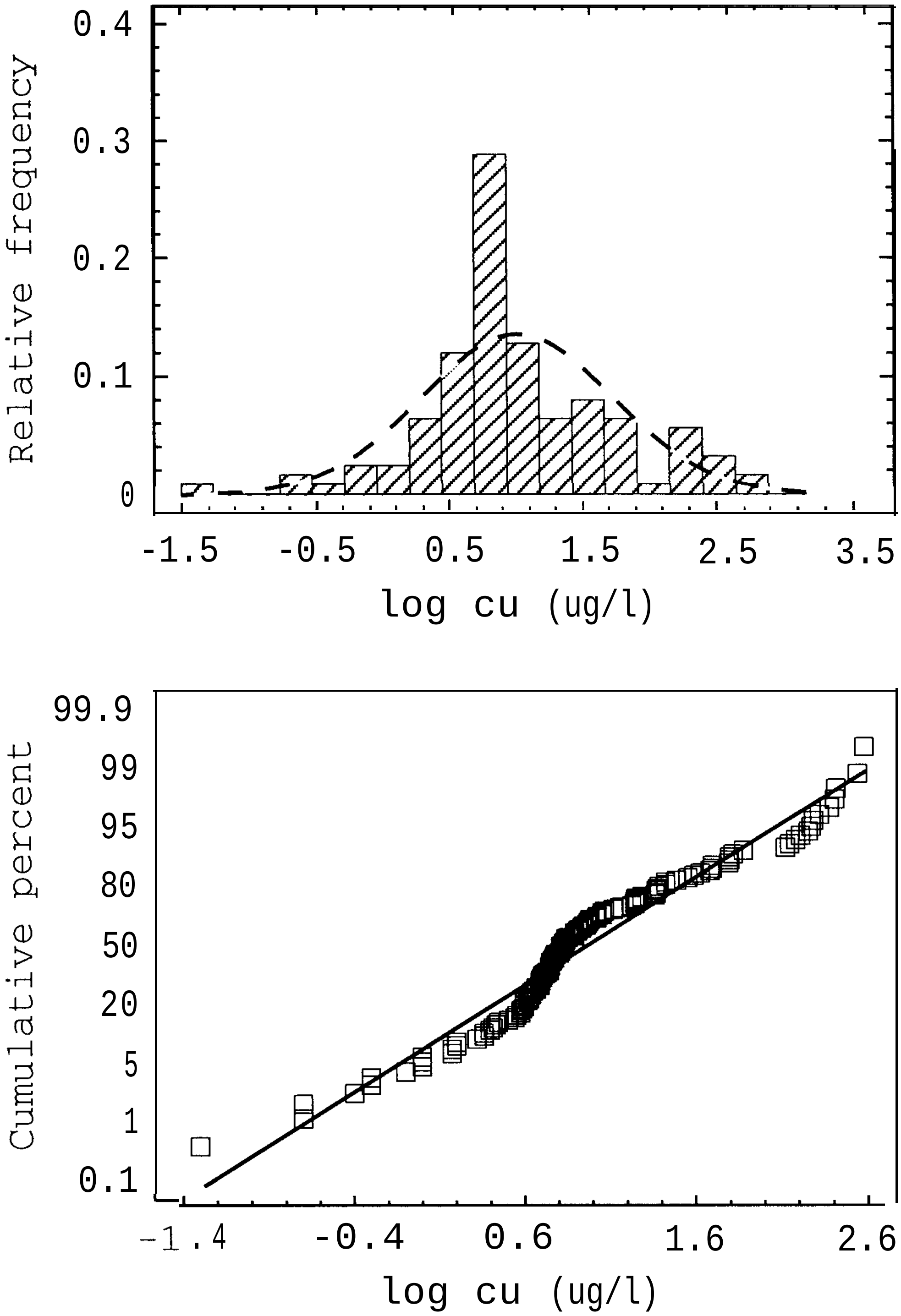


Figure 74. Relative frequency histogram (upper panel) and normal probability plot (lower panel) of transformed data (log„) of dissolved copper (Cu) concentrations in water samples in the Houma Navigation Canal near Cocodrie (period 1978-1990 from the LDEQ Water Quality Monitoring Program). A good agreement with the normal distribution is indicated.

changes around 1982–1983. A standard linear regression model was used to determine temporal trends:

$$Y_t = a + b\,t$$

In the above equation Y_t denotes the value of dependent variable at time t, t (= time) is the independent variable, b is the slope coefficient, and a is the intercept. In assessing the significance of the trend we used a significance level of 0.05. Models having the probability value (p) for the condition [H_0: $b = 0$] greater than 0.05 were rejected. Varying levels of significance are identified at $0.05<p>0.01$, $0.01<p>0.001$, and $p<0.001$.

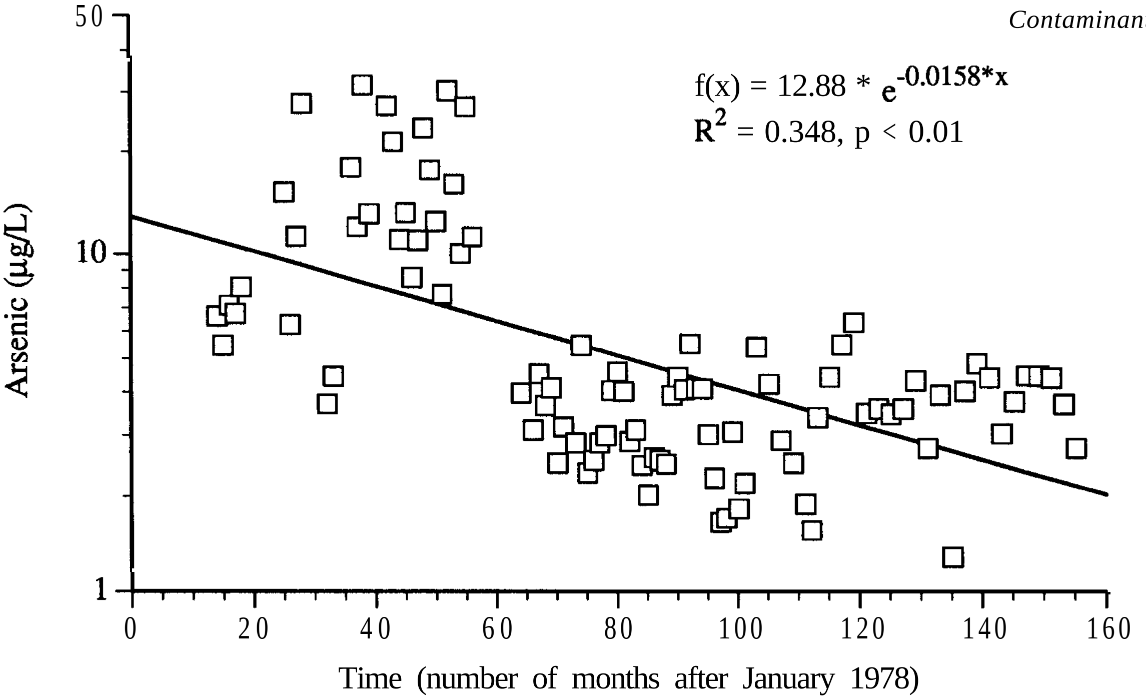
An example of the trends analysis is given in figure 75. In data plots, the solid line denotes the trend (= regression line). Full statistics for this regression model are given in table 24.

Historical Trends in Estuarine Contaminants

Three of the data sets described above were used for the trends analyses. These are the LDEQ data set, USGS data set, and Mussel Watch data set. These data sets differ in temporal coverage and sampling sites as well as environmental components sampled. The LDEQ and USGS data sets consist of contaminant levels in water, while the Mussel Watch data set consists of contaminant levels in oysters and sediment. (See Introduction for a discussion of information provided by different types of samples.)

Pollution Trends Based on LDEQ's Water Quality Monitoring Program

Results from the regressions of levels of six elements against time are shown in table 25. Part A of this table shows that all elements tend to show a statistically significant decline over the period 1978–1990. Two graphs representative of this pattern are shown in figure 75. An inspection of the graphs produced in the statistical analyses indicated that patterns were not necessarily consistent over the 13-year period. Drastic changes apparently occurred around 1982–1983. This is especially evident from the arsenic data shown in figure 75. Because the exact nature and cause of these sudden changes are unclear at this point (and may be due to changes in equipment and methodology implemented at LDEQ; Elaine Sorbet, pers. comm.), analyses were run separately for the periods 1978–1982 and 1984–1990. Results of these regression analyses are shown in parts B and C of table 25. The strong negative trends in element levels appear in part caused by the sudden reduction around 1982. Up to 1982, several elements (As, Cu, and to some extent Cd) showed a statistically significant trend of increasing levels. For As, the large drop around 1982 appears to have been sufficient for an overall significant decrease over the period 1978–1990. Neither of the periods 1978–1982 and 1984–1990 by themselves show a significant decline in As. However, the other five elements tend to



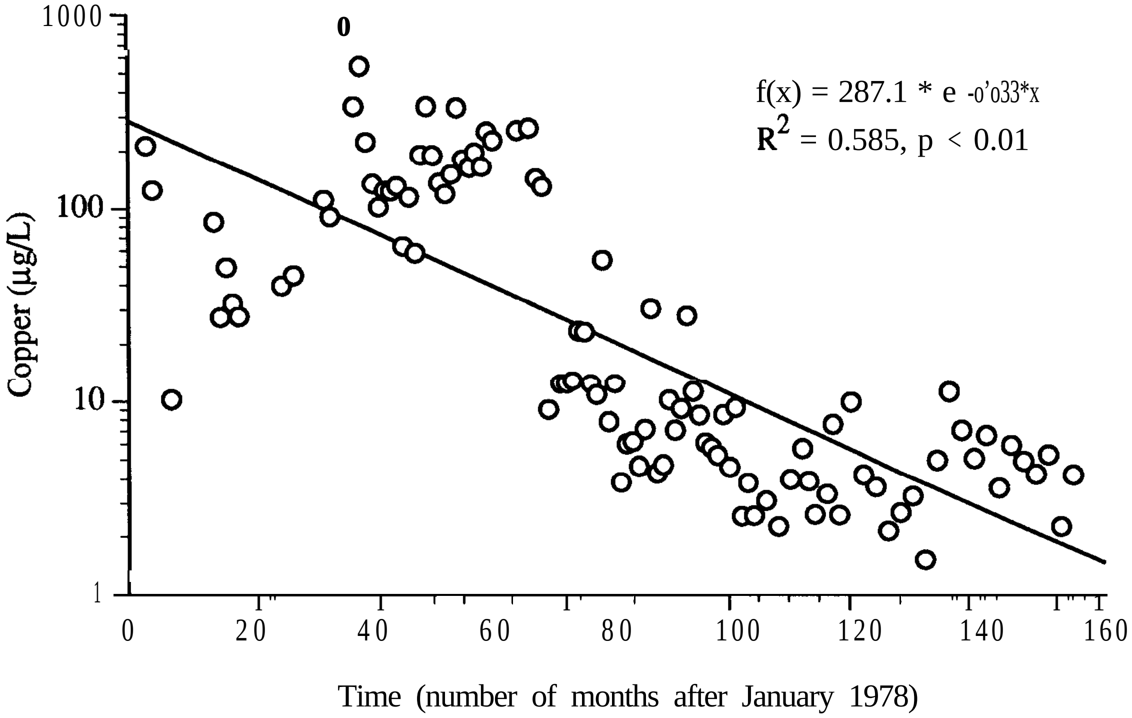


Figure 75. Concentrations of arsenic and copper (dissolved+particulate) in water samples collected in five Terrebonne sites under the LDEQ Water Quality Monitoring Program, between 1978 and 1990. The regression line depicts the statistical trend over the total time period.

Table 24. Full statistics for the trend analysis of Cr concentration in Bayou Lafourche (pooled data from four sites, period 1978–1990).

Regression analysis–Linear model: Y = a + bx

Dependent variable: LOG10 (Cr) Independent variable: TIME

Parameter	Estimate	Standard Error	T Value	Prob. Level
Intercept	1.23832	0.050178	24.6785	0.00000
Slope	-0.003877	0.0006066	-6.39217	0.00000

Analysis of variance

Source	Sum of squares	Df	Mean square	F-ratio	Prob. level
Model	3.299224	1	3.299224	40.85979	0.00000
Residual	9.44717	117	0.08075		
Total (corr.)	12.74639	118			

Correlation coefficient = - 0.508759 R-squared = 25.88 percent

Stnd. error of est. = 0.284157

Table 25. Results of analyses for trends on LDEQ's data on total concentrations of six elements in water. Values shown are results from regression analyses on elemental concentrations (log transformed) versus time, for the period 1978 through 1990. Regression coefficients are shown as (-) or (+) where p > 0.05, and + or -, ++ or -- and +++ and --- for 0.05 >p>0.01, 0.01>p>0.001 and p<0.001, respectively. Results are shown for analyses using data from 1978-1990, 1978-1982, and 1984-1990. Data were collected generally on a monthly or bimonthly basis.

a. Using all data (1978–1990):

	As	Cd	Cu	Cr	Hg	Pb
Chackbay	---	---	---	(-)	---	---
Little Lake	---	---	---	---	(-)	---
B. Lafourche	---	---	---	---	--	---
Terr. Bayous	---	---	---	---	--	---
HNC	---	---	---	---	(-)	---

b. Using data from the first period (1978–1982) only:

	As	Cd	Cu	Cr	Hg	Pb
Chackbay	+++	(+)	+	(+)	*	(+)
Little Lake	+++	(+)	+++	(-)	(-)	(+)
B. Lafourche	+	(-)	+++	(-)	(-)	(+)
Terr. Bayous	++	++	+++	(+)	*	(+)
HNC	+	(+)	++	(-)	(-)	(+)

** insufficient data*

c. Using data from the second period (1984–1990) only:

	As	Cd	Cu	Cr	Hg	Pb
Chackbay	(+)	(+)	-	(-)	---	(+)
Little Lake	(-)	(-)	(-)	-	(-)	--
B. Lafourche	(+)	(-)	---	-	---	--
Terr. Bayous	(+)	(-)	---	(-)	---	(+)
HNC	+	--	---	--	(-)	(-)

show a significant decline for the period 1984–1990 by itself, albeit more variable among sites and less pronounced than for the entire 1978–1990 period.

Pollution Trends Based on USGS's Water Resources Data for Louisiana

Results of regression analyses using this data set are shown in table 26. Significant regressions were obtained only for the Bayou Lafourche site, for arsenic and lead. The significant increase in lead over the period 1975–1981 is consistent with a tendency for an increase in lead for the LDEQ data covering the period 1978–1982. The decrease in arsenic levels at the Lafourche site does not agree with the LDEQ data. This discrepancy could be due to the difference in years covered (1975–1981 *vs*. 1978–1982) because their surveys are based on different sites or possibly chance or other factors.

Pollution Trends Based on Oyster Tissue Data from the NOAA Mussel Watch Project

Results of regression analyses using this data set are shown in table 27. Some examples of regressions for specific pollutants and specific sites are shown in figure 76. There is considerable variation among sites, yet several distinct patterns emerge from this data set. Oysters sampled over this period showed the tendency of statistically significant increases in silver, cadmium, copper, and lead. Statistically significant decreases were evident for arsenic, dieldrin, chlordane, lindane, and lmw- and hmw-PAHs.

Conclusions on Trends in Contaminant Levels

The three data sets used show apparent inconsistencies in trends where the data sets overlap in temporal coverage and coverage of specific pollutants. Possible reasons for apparent inconsistencies between the LDEQ and USGS data were discussed earlier. For the metals covered by both data sets, significant trends for As, Cd, Cu, and Pb in the Mussel Watch data were opposite from those that emerged from the LDEQ water data. It is unclear why trends differed among the data sets. Differences between the two data sets could be related to differences in variables analyzed. For example, if the amount of suspended sediment in the water has decreased over the last decade, total metal levels in the water are likely to have declined on the basis of that fact alone, and dissolved metal levels might have stayed the same or increased. Because oysters are likely to take up metals mostly from the dissolved phase, metal accumulation by oysters might have increased. Another possibility for the differences in general outcomes could be related to the different areas covered by the two data sets. The LDEQ sites fall mostly in the upper part of the Barataria-Terrebonne area, while the Mussel Watch sites fall in the lower section of the area (see figures 69 and 71). The Mussel Watch sites could potentially be exposed to water from a different source (e.g., Mississippi River water finding its way back into the estuary) than the LDEQ sites. Overall, it appears that levels of some

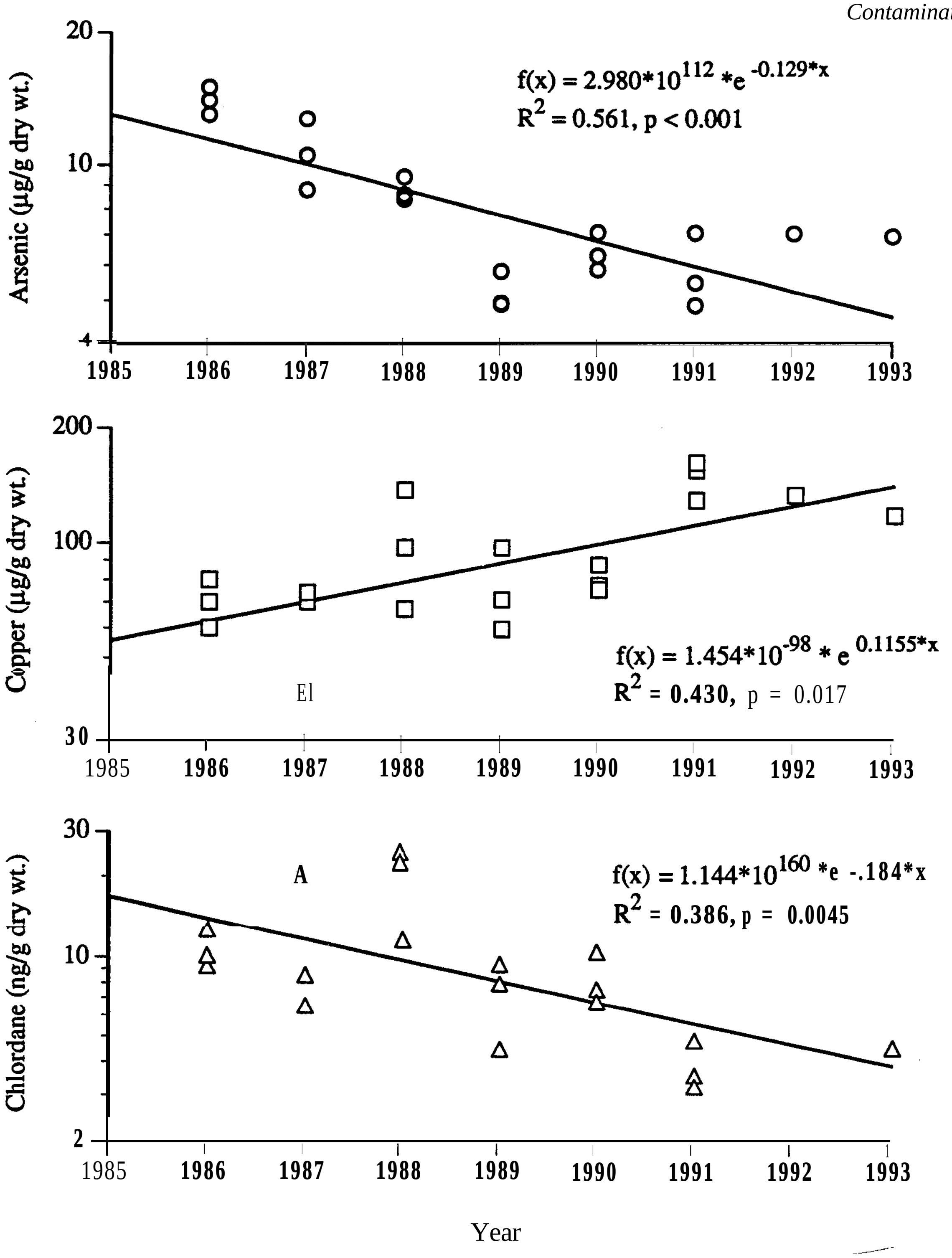


Figure 76. Examples of temporal trends in contaminant levels in oyster tissues sampled under the Mussel Watch Project. Data shown are concentrations of Arsenic and copper at site TBLF, and concentrations of chlordane at site BBSD.

Table 26. Results of analyses for trends on pollutant levels in water (total levels) determined in USGS's water quality monitoring program. Values shown are results from regression analyses on toxicant concentrations (log transformed) versus time, for the period 1975 through 1981. Data for Cd and Cr were not used as values were generally close to detection limits. To depict both sign (pos. or neg.) and significance of regression coefficients, these are reported as (+) or (-) where $p > 0.05$, and **+** or **-** for $0.05 > p > 0.01$. Data were collected generally 4-12 times per year for the Larose site and 4–21 times per year for the SW Pass site.

	Bayou Lafourche at Larose	Mississippi River SW Pass
As (arsenic)	**-**	(-)
Cu (copper)	(+)	(+)
Ni (nickel)	(-)	(-)
Pb (lead)	**+**	(+)
Zn (zinc)	(-)	(-)

contaminants have increased over the last decade (while others have decreased) with the trends varying among sites in the Barataria-Terrebonne estuary. Cases where contaminant levels have decreased appear more numerous than cases where contaminants increased to indicate that the overall trend is a favorable one.

Status of Estuarine Contaminants

Three of the data sets described earlier were used to assess the current status. These are two of the data sets used in the trends analyses (data from LDEQ's water quality monitoring program and data from NOAA's Mussel Watch project), and one data set not used in the trends analyses (data from the Louisiana Province of the EMAP-E program). As was the case for the trends analyses, these data sets differ in many respects. The environmental components sampled differ among the various programs. The LDEQ data used for status analyses consist of dissolved contaminant levels in water samples. The Mussel Watch data and the EMAP-E data contain tissue contaminant data (though for different aquatic species), while the EMAP-E data also provide information on sediment contaminant levels and sediment toxicity (the latter obtained in laboratory bioassays conducted with field-collected sediment). Contaminants analyzed in samples also differ among the data sets. The LDEQ pollutant data are limited to six elements; the full NOAA data set covers 16 elements; and 51 organic pollutants (several of the latter were combined to allow more general conclusions to be drawn from our analyses); while the

Table 27. Results of analyses for trends on Mussel Watch's data on concentrations of pollutants in oyster tissues. Values shown are results from regression analyses on toxicant concentrations (log transformed) versus time, for the period 1986 through 1993, with the following exceptions:

- N 1986–1990 for Ag and Cr, 1986–1991 for Mn (discontinued in respect. 1990 and 1991)
- N 1989–1993 for GPAHs, as detection limits were high initially and new PAHs were added to the list up to 1989.
- N 1987–1993 for GPCBs, as a change in listing (by Cl level 6 by congener) was made in 1987.

Regression coefficients are shown as (-) or (+) where p > 0.05, and + or -, ++ or -- and +++ and --- 0.05 >p>0.01, 0.01>p>0.001 and p<0.001, respectively. Data were collected generally on a yearly basis, with 3 replicates in 1986–1991 and 1 replicate per site since 1992.

	ABOB	CLCL	TBLB	TBLF	BBSD	BBMB	MBTRP
Ag (silver)	+	+	++	(+)	(+)	(+)	(+)
As (arsenic)	(-)	(-)	--	---	---	(-)	(-)
Cd (cadmium)	(+)	+	(+)	(+)	+	+	(+)
Cr (chromium)	(+)	(+)	+	(-)	(+)	(+)	(-)
Cu (copper)	(+)	+	+	++	+	(+)	(+)
Hg (mercury)	(+)	(+)	(+)	(-)	(+)	(+)	(+)
Mn (manganese)	(+)	(-)	(+)	-	(+)	(-)	(+)
Ni (nickel)	(-)	(+)	(+)	(-)	(-)	(+)	(+)
Pb (lead)	(+)	++	(+)	(-)	++	(+)	(-)
Se (selenium)	(+)	(+)	(+)	(+)	(+)	(+)	(-)
Zn (zinc)	(+)	(+)	(+)	(+)	+	(-)	(-)
GOrganotins	(-)	(-)	(-)	(-)	-	(-)	(+)
GPCBs (18 congeners)	(-)	(-)	(-)	(+)	(-)	(-)	(-)
G"DDT"s (incl. DDD, DDE)	---	(-)	(-)	(-)	(-)	(-)	(+)
Dieldrin	---	(-)	(-)	(-)	(-)	--	(+)
Chlordane	---	--	--	-	--	---	(-)
Lindane	--	(-)	(-)	(-)	(-)	-	(-)
Mirex	(+)	+	(+)	(+)	(+)	(+)	(+)
GlmwPAHs (2, 3-ring PAHs)	--	(-)	-	(+)	---	(-)	(-)
GhmwPAHs (4,5, 6-ring PAHs)	(-)	(-)	-	(+)	---	(+)	(-)

full EMAP-E data set covers 12 elements and 36 organics for the tissue data and 15 elements, four organo-metals and 113 organic contaminants. For analytical purposes, specific organics were again combined in this data set.

The time periods covered by these data sets used for status analyses also differed among the data sets. In principle, data from 1990 to present were used to determine the current status. However, availability of recent data differed among the various data sets, while specific characteristics for the disparate data sets resulted in the exclusion of some data. The LDEQ data used for status analyses are from samples collected between 1991 and March 1994 (1990 data could not be used as analyses were changed from total to dissolved element levels starting 1991). Available EMAP-E data cover 1991 and 1992 (this program started in 1991), while the Mussel Watch data cover 1990 through 1993. The number of sampling sites and the distribution of these sites over the Barataria-Terrebonne estuarine system also differ among the sites (see figures 70–72). All these differences among the data sets have to be taken into consideration when trying to derive general conclusions regarding the current status of the environmental contamination in the Barataria and Terrebonne basins.

To determine the relevance of specific contamination levels, contaminant concentrations were compared to specific "health criteria." Contaminant concentrations were compared to EPA's quality criteria for water (both criteria for the protection of aquatic life and their uses, as well as human health criteria; EPA 1986). Contaminant levels in sediment were compared to guideline values reported by Long et al. (1995) and Long and Morgan (1990) (the latter obtained from Summers et al. 1993). Contaminant levels in organisms were compared against criteria based on FDA and international limits listed in Summers et al. (1993). It has been recognized that many of these criteria levels may be too high for the adequate protection of all organisms. However, trying to establish more specific and possibly more rigorous criteria is beyond the scope of this report.

Data transformations were sometimes necessary when comparing contaminant levels in organisms (oysters, fish, shrimp) to health criteria levels. Direct comparisons were not possible when concentrations were expressed in different units (specifically tissue dry weight *vs.* tissue wet weight). A factor of 8 was used for converting concentrations based on wet weight into concentrations based on dry weight. This value is the average for wet/dry weight ratios reported for oysters (Zumada and Sunda 1982, Wright et al. 1985).

Pollution Status Based on LDEQ's Water Quality Monitoring Program

Concentrations of the six elements of this data set, as well as results from the post hoc pairwise comparisons when an overall ANOVA showed a significant difference among sites, are shown in figures 77–82.

Barataria basin. The following conclusions can be drawn from the data on the 11 Barataria basin sites:

Chromium and mercury do not differ significantly among water samples from these sites. Chromium (averaging about 0.5 $\mu g\ l^{-1}$) is present at concentrations well below EPA's water quality criteria. Mercury may be present at concentrations above the chronic freshwater and marine water quality criteria of 0.012 and 0.025, respectively. However, Hg in these samples is often present below the analytical detection limit (0.2 µg l^{-1}; an order of magnitude above the criteria levels). These observations weaken the inferences that can be made about Hg. One measurement of 0.3 µg l^{-1} Hg indicates that Hg may be a problem in the study area. Because the criteria for Hg are based on a worst-case scenario (where all the mercury is present as the extremely toxic methylmercury), the occurrence of levels exceeding the criteria does not necessarily translate into an inadequate protection of aquatic life (EPA 1986).

For all four elements (As, Cd, Cu, Pb) for which concentrations differ among sites, concentrations at Bayou Segnette are significantly above those at all other sites. The mean arsenic (As) concentration at Bayou Segnette is 4.9 µg l^{-1}, with a maximum concentration reported of 72 µg l^{-1}. Arsenic toxicity depends on the form in which it is present (arsenic-III, arsenic-V, or organic arsenic). National water quality criteria for the protection of aquatic organisms and their uses are available only for the As-III oxidation state. Freshwater aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentration does not exceed 190 µg l^{-1}, while this value for saltwater aquatic organisms is 36 µg l^{-1} (EPA 1986). For the Bayou Segnette site, the freshwater criteria are more appropriate than the marine ones because Cl concentration for 1991 at this site averaged 111 mg l^{-1} (LDEQ 1992). Consequently, no effects on aquatic organisms and their uses are expected for As. However, As is a carcinogen in humans. Current toxicological models used for regulatory purposes assume the absence of a threshold for carcinogens such that ambient water concentrations should be zero. EPA has determined the additional risks of cancer in humans due to elevated levels of arsenic in water (for ingestion of water and consumption of fish combined, as well as for consumption of fish only.) The calculated incremental increase of cancer risk of 10^{-5} over a human lifetime corresponds to an As concentration in water of 0.175 µg l^{-1}, based on the human consumption of fish and ignoring the direct accumulation from water (EPA 1986). Consequently, levels of As (especially at Bayou Segnette) may be a human health problem in spite of the fact that levels are not exceptionally high.

The cadmium (Cd) concentration in Bayou Segnette averages 0.65 µg l^{-1}, with a maximum recorded value of 4.80 µg l^{-1}. EPA's freshwater acute criterion for Cd is 3.9 µg l^{-1}, with a freshwater chronic criterion of 1.1 µg l^{-1} (EPA 1986) to indicate a potential problem with Cd at this site. The EPA criteria are hardness dependent. At an average hardness of 210 mg l^{-1} reported at this site for 1991 (LDEQ 1992), a 4-day average concentration of 2.03 µg l^{-1} should not be exceeded more than once per three years for aquatic organisms and their uses to be affected unacceptably. Cd levels at Bayou Segnette exceeding 4.0 µg l^{-1} were reported twice during the approximate 3-year period covered by our analysis (albeit for durations unknown) to indicate a potential problem for Cd at this site.

Copper (Cu) levels at Bayou Segnette averaged 6.6 µg l^{-1}, with a maximum recorded value of 23.9 µg1^{-1}. The EPA's general freshwater and chronic criteria for this metal are

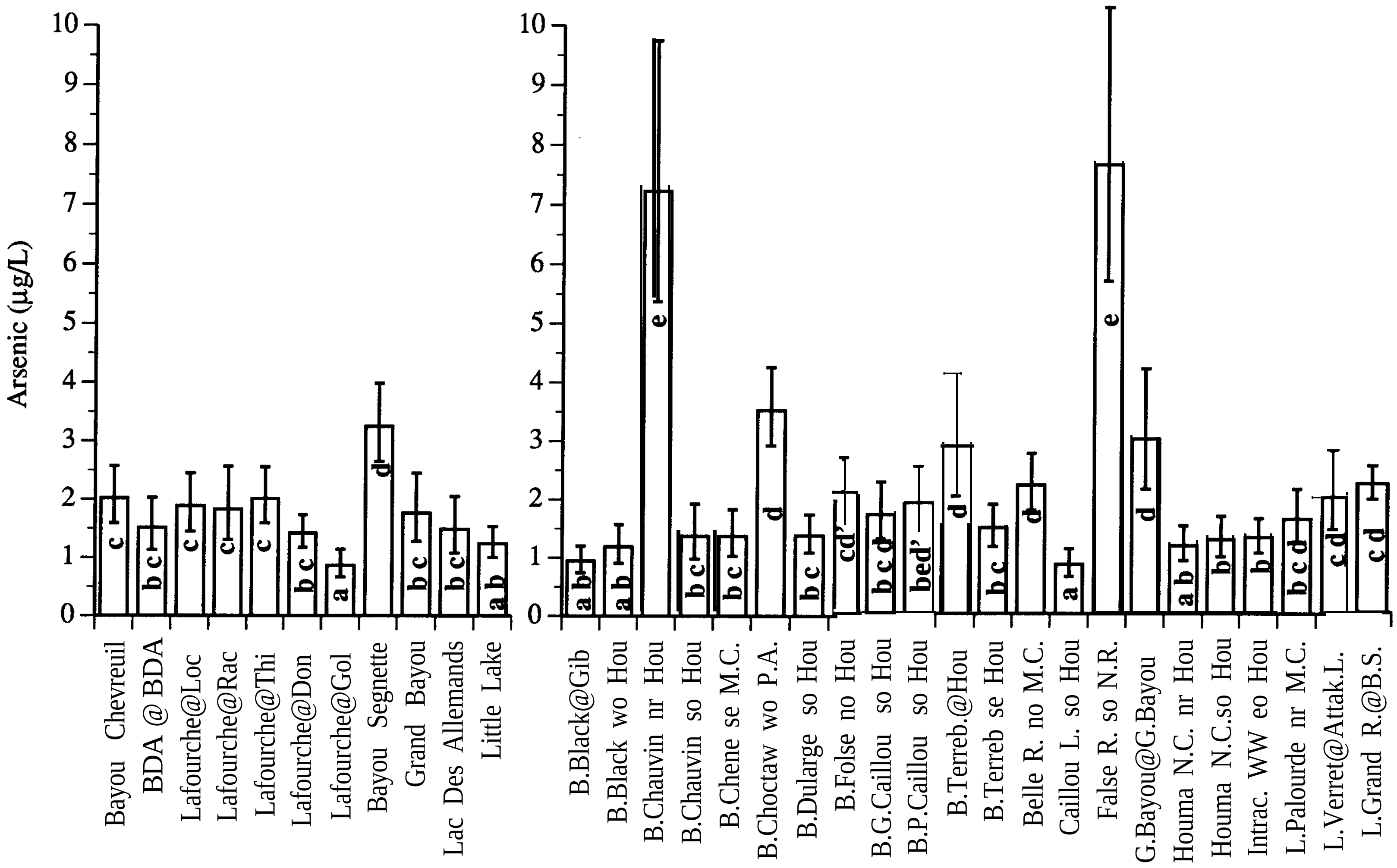


Figure 77. Dissolved arsenic concentrations in water samples collected at Barataria sites (left panel) and Terrebonne sites (right panel) under the LDEQ Water Quality Monitoring Program since 1991. Error bars are 95% conficence limits for the mean. Columns with the same letter (within a panel) were not significantly different in post-hoc comparisons (Fisher's PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

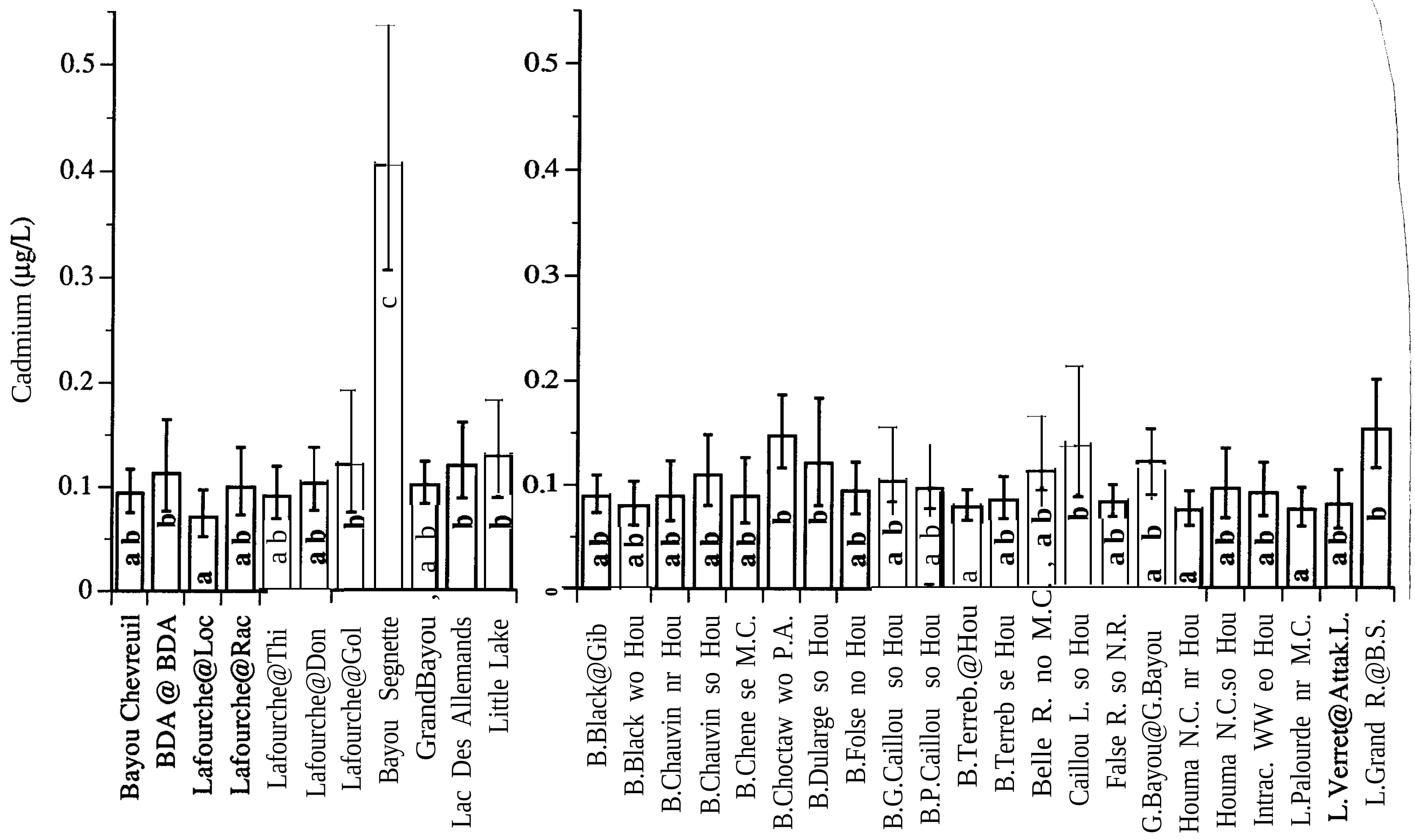


Figure 78. Dissolved cadmium concentrations in water samples collected at Barataria sites (left panel) and Terrebonne sites (right panel) under the LDEQ Water Quality Monitoring Program since 1991. Error bars are 95% conficence limits for the mean. Columns with the same letter (within a panel) were not significantly different in post-hoc comparisons (Fisher's PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

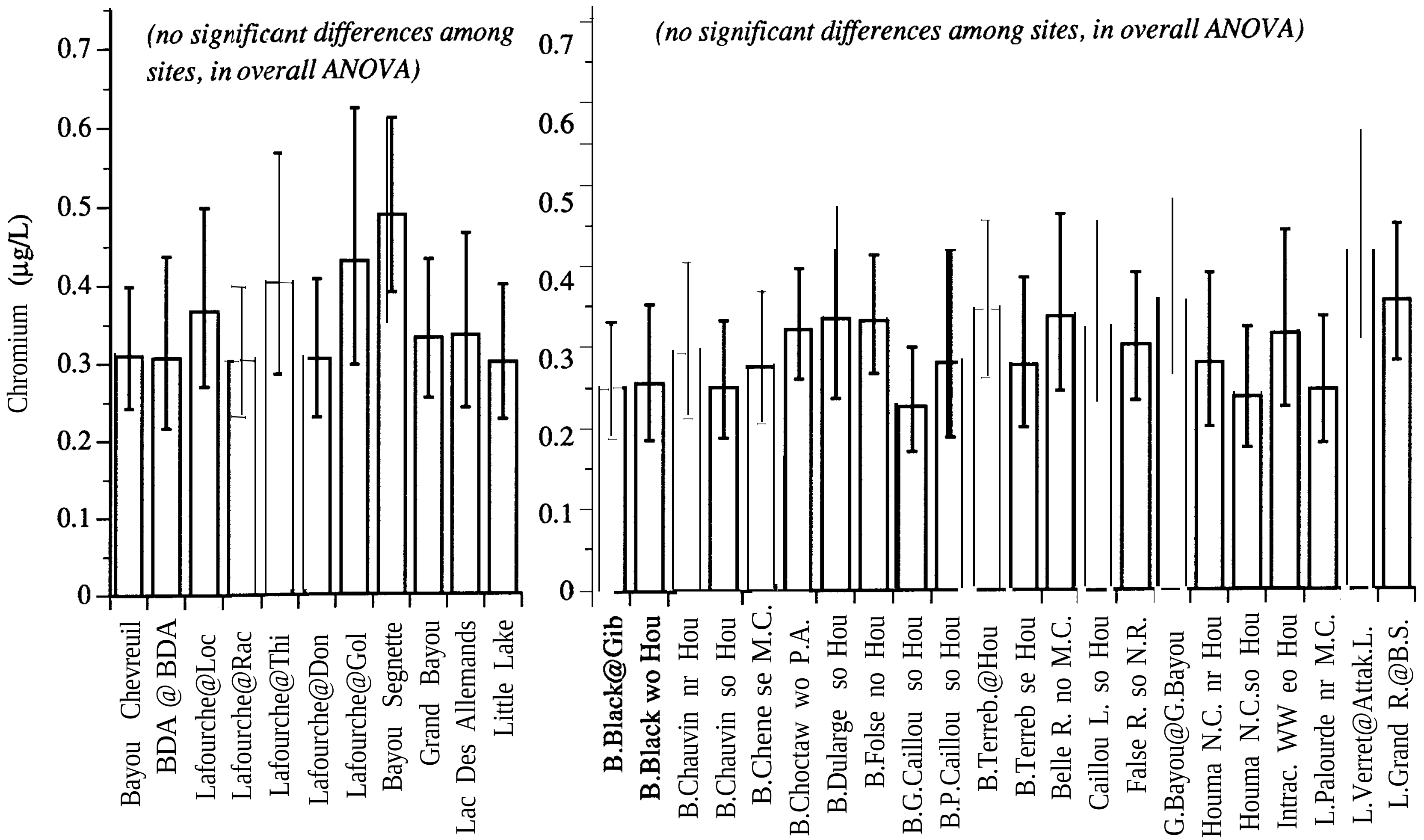


Figure 79. Dissolved chromium concentrations in water samples collected at Barataria sites (left panel) and Terrebonne sites (right panel) under the LDEQ Water Quality Monitoring Program since 1991. Error bars are 95% conficence limits for the mean. Columns with the same letter (within a panel) were not significantly different in post-hoc comparisons (Fisher's PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

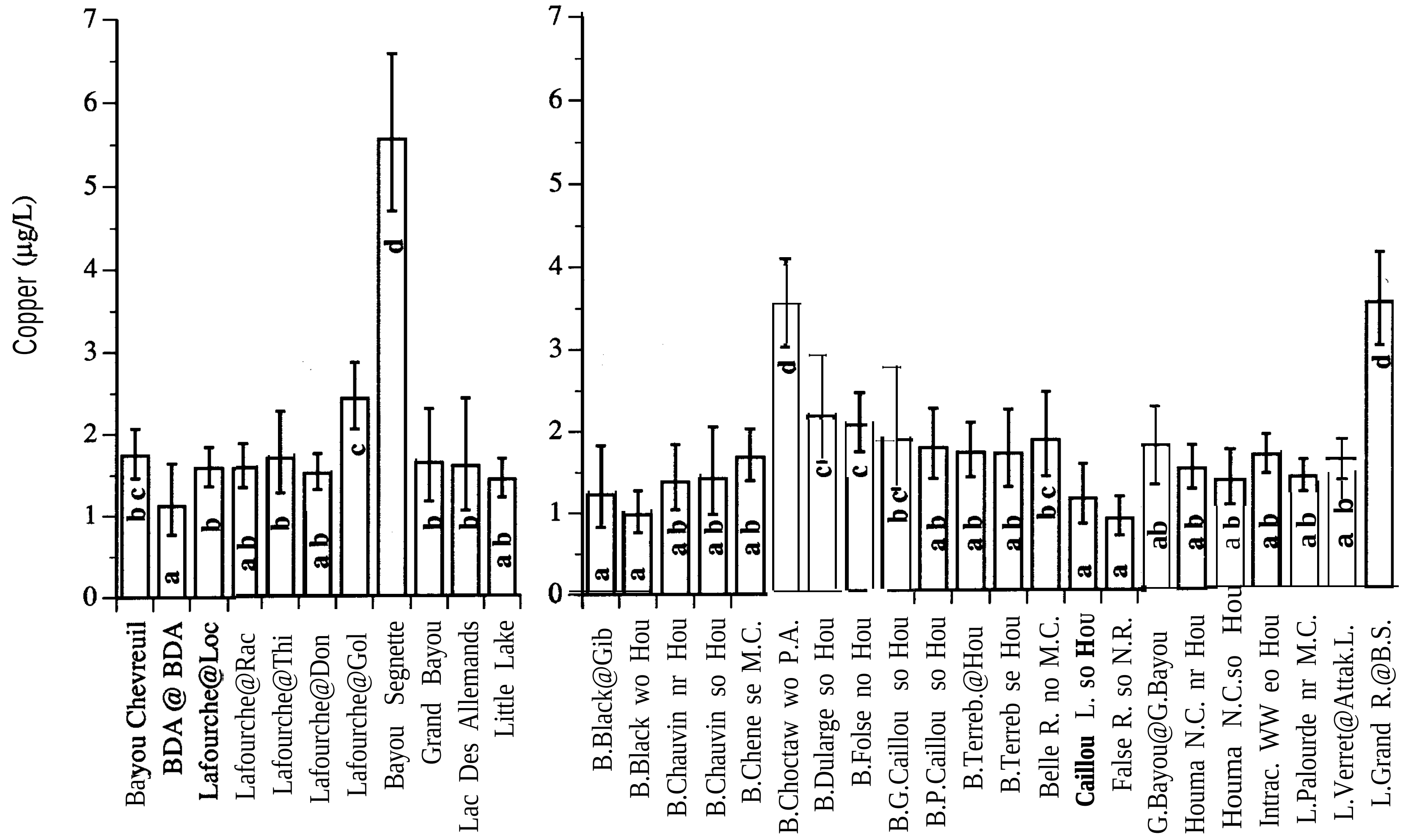


Figure 80. Dissolved copper concentrations in water samples collected at Barataria sites (left panel) and Terrebonne sites (right panel) under the LDEQ Water Quality Monitoring Program since 1991. Error bars are 95% conficence limits for the mean. Columns with the same letter (within a panel) were not significantly different in post-hoc comparisons (Fisher's PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

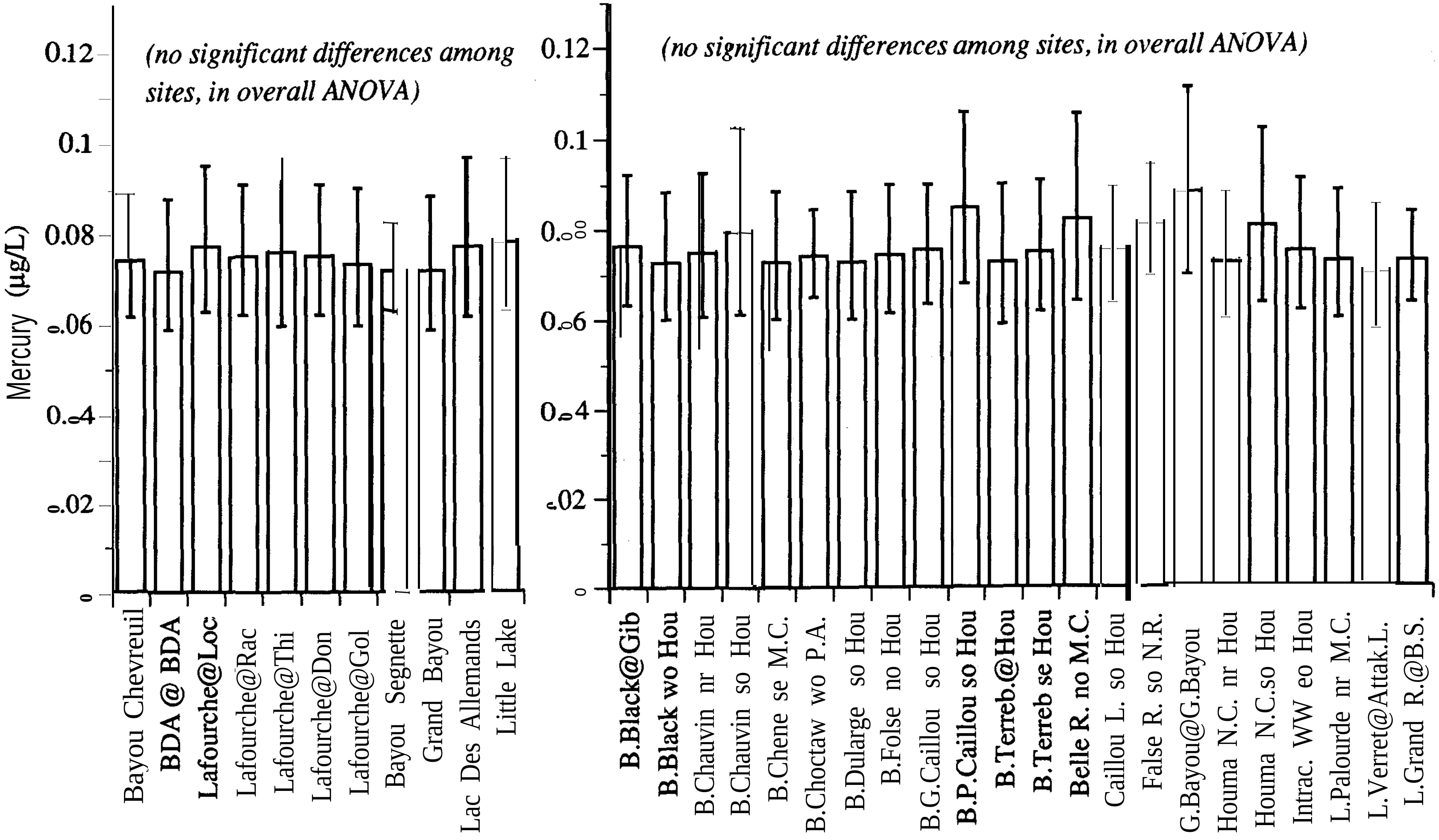


Figure 81. Dissolved mercury concentrations in water samples collected at Barataria sites (left panel) and Terrebonne sites (right panel) under the LDEQ Water Quality Monitoring Program since 1991. Error bars are 95% conficence limits for the mean. Columns with the same letter (within a panel) were not significantly different in post-hoc comparisons (Fisher's PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

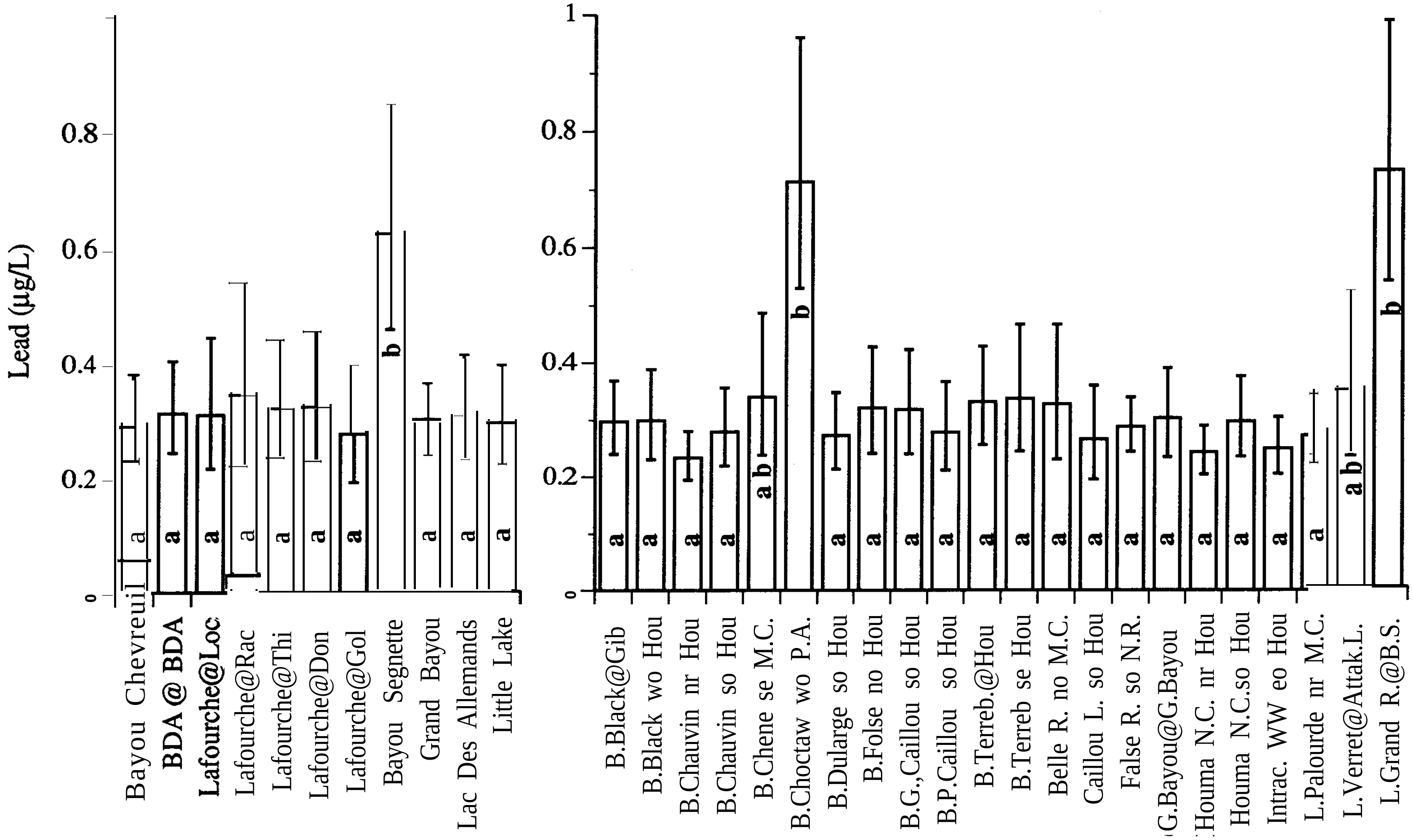


Figure 82. Dissolved lead concentrations in water samples collected at Barataria sites (left panel) and Terrebonne sites (right panel) under the LDEQ Water Quality Monitoring Program since 1991. Error bars are 95% conficence limits for the mean. Columns with the same letter (within a panel) were not significantly different in post-hoc comparisons (Fisher's PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

respectively 18 and 12 µg l^{-1}. Taking water hardness dependence of this criteria into consideration translates into a 4-day average concentration not to be exceeded more than once per three years of 22.3 µg l^{-1}, at a hardness of 210 mg l^{-1}. The EPA criteria for copper may not be currently in effect because they are listed as "suspended, canceled or restricted" (EPA 1986). However, values reported for Bayou Segnette approach water quality criteria levels to indicate that copper levels at this site may constitute a water quality problem.

Lead (Pb) levels at Bayou Segnette average 1.1 µg l^{-1}, with a maximum level reported of 6.60 µg l^{-1}. The EPA's general freshwater and chronic criteria for this metal are respectively 82 and 3.2 µg l^{-1}. These concentrations are again hardness dependent, such that the 4-day average concentration not to be exceeded more than once per three years comes out to 8.2 µg1^{-1} at a hardness of 210 mg l^{-1}. These values indicate that Pb levels at this site occasionally may affect aquatic organisms and their uses.

Terrebonne basin. The following conclusions can be drawn from the data on the 22 Terrebonne basin sites:

Similar to the situation for the Barataria basin sites, Cr and Hg levels do not differ significantly among the various sites. Chromium concentrations are again well below EPA's water quality criteria (EPA 1986) for either hexavalent or trivalent chromium (Cr speciation not determined by LDEQ). An interpretation of the importance of Hg levels again suffers from the problem that EPA's water quality criteria are below LDEQ's method detection limit.

Arsenic concentrations are highest at Bayou Chauvin near Houma and False River south of New Roads. The highest levels here are higher than those reported for the Barataria basin, with means (and maxima) at the two Terrebonne sites of respectively 8.9 (21.2) and 11.6 (31.2) µg l^{-1}. Firm conclusions with respect to impacts on aquatic organisms and their uses are not possible, as EPA's water quality criteria have been derived separately for As-III and As-V. Maximum As levels at the False River site are not far removed from the lowest observed effect level for freshwater chronic exposure to pentavalent arsenic of 48 µg l^{-1} (EPA 1986), but the LDEQ data do not distinguish between the two oxidation states. Similar to the situation at Bayou Segnette, As levels are well beyond the 0.175 µg l^{-1} corresponding to a 10^{-5} incremental increase of cancer risk in humans via fish consumption (EPA 1986).

Cadmium concentrations do not vary much among the sites (though some statistically significant differences are present) and concentrations are lower than those reported for Bayou Segnette. Concentrations equal to or exceeding the 1.1 µg l^{-1} general freshwater chronic criterion are reported for 10 samples. The 4-day average concentration not to be exceeded more than once per three years comes out to 1.49 µg l^{-1} at a hardness of 141 mg l^{-1} (the hardness average for the four most-polluted sites in the Terrebonne basin during 1991). This concentration was exceeded six times among all sites during the 3-year period covered by the data.

Copper concentrations were highest in Bayou Choctaw west of Port Allen and in the Lower Grand River at Bayou Sorrel. Mean (and maximum) levels at these sites are 4.0 (13.3) and 4.2 (27.2) µg l^{-1}; somewhat less than values reported for Bayou Segnette. However, levels do occasionally exceed the general freshwater acute and chronic criteria of respectively 18 and

12 µg l^{-1} to indicate that copper levels may at times be problematic. The 4-day average concentration not to be exceeded more than once per three years for water with a hardness of 140 mg l^{-1}, calculated to be 15.76 µg l^{-1}, was exceeded once among the two sites with the highest copper levels.

Lead levels in water show the same pattern as observed for copper, with mean (and maximum) levels of 1.2 and 1.3 µg l^{-1} at respectively the Bayou Choctaw and Lower Grand River sites being significantly higher than at most other sites. These lead concentrations are similar to values reported for Bayou Segnette. A comparison of the maximum levels at the two sites (respectively 9.6 and 7.5 µg l^{-1}) with the general freshwater chronic criterion of 3.2 µg l^{-1} indicates that Pb may occasionally be problem at these sites. The hardness-dependent criterion for a 4-day average not to be exceeded more than once per three years was calculated to be 4.88 at a hardness of 140 mg l^{-1}. This value was exceeded five times at the two sites with the highest Pb levels.

Pollution Status Based on EMAP-E (Louisiana Province) Monitoring Program

In principle, this data set is ideally suited to determine the pollution status of the Barataria and Terrebonne basins because it combines sediment toxicity, contaminant concentrations in sediment and contaminant tissue levels in fish and shrimp (at the same sites). However, spatial coverage is limited to the coastal section of the basins (see figure 72). As explained in the description of the data sets (see p. 141) sites are visited only once, and no replicate sampling is done, so standard statistical analyses such as ANOVA cannot be used (e.g., for deciding if there are real differences among sites).

Sediment toxicity. Statistically significant mortality occurred in sediment from 9 of the 18 sites for either one or both of the test species used (see table 28). Significant sediment toxicity was observed in 11 of the 36 bioassays. Significant mortalities exceeding 10% were reported for sediment from Little Lake, Bayou Terrebonne, Lake Palourde, Lake Verret, Lake Pelto, and one site in Terrebonne Bay. Of these sites, statistically significant mortalities exceeding 20% (the criterion for toxicity used by the EMAP-E project, Summers et al. 1993) were reported for Little Lake, Bayou Terrebonne, and Lake Verret. Though these data by themselves provide no information on the cause of the toxic effects, the bioassay results indicate that sediment toxicity is a problem for the Barataria-Terrebonne estuarine system.

Sediment contaminant levels. Sediment levels of elements and organic contaminants (PCBs, PAHs, pesticides) are shown in table 29. To determine the significance of the observed contaminant levels, values were compared to the criteria developed by Long and co-workers (Long et al. 1995, Long and Morgan 1990; latter cited in Summers et al. 1993). None of the values exceeded Long and Morgan's criteria for 50% effects ("effects likely"), though several exceed 10% effects criteria ("effects possible"). The 10% effects

Table 28. Sediment toxicity results from experiments conducted with two bioassy species. Sediment was collected under the EMAP Program. See figure 72 for location of sample sites.

		Sediment bioassay results			
		Ampelisca abdita		*Mysis bahia*	
Site name	site #	test survival (% of control)	Sign. less than control?	test survival (% of control)	Sign. less than control
Barataria Bay	1	102	N	117	N
Barataria Bay	2	103	N	93	Y
Barataria Bay	9	108	N	107	N
Barataria Bay	10	103	N	100	N
Little Lake	5	55	Y	107	N
Lake De Cade 1	6	91	Y	100	N
Lake Felicity	14	99	N	100	N
Lake Mechant	15	98	N	97	N
Lake Palourde	8	92	Y	87	Y
Lake Verret	17	97	N	66	Y
Bayou Terrebonne	18	74	Y	104	N
Caillou Bay	4	97	N	97	N
Lake Pelto	7	87	Y	93	Y
Lake Raccourci	6	95	N	100	N
Terrebonne Bay	3	81	Y	100	N
Terrebonne Bay	11	99	N	97	N
Terrebonne Bay	12	92	Y	96	N
Terrebonne Bay	13	92	N	107	N

Table 29a. Concentrations of specific metals in sediments (in µg/g dry weight), determined under the EMAP Program (1991 & 1992). Concentrations in bold exceed guideline values (see footnote) reported by Long et al. (1995). Guideline values are effects range-low (ERL; "effects possible") and effects range median (ERM; "effects likely"). Also shown in the footnote (for those metals for which ERL values are approached or exceeded) are incidences of effects normally associated with metal concentrations falling between ERL and ERM values.

	site #	Ag	As	Cd	Cr	Cu	Hg	Mn	Ni	Pb	Sb	Se	Sn	Zn
Barataria Bay	1	0.40	7.50	0.15	48.0	10.6	0.027	200.0	16.2	18.8	0.30	0.60	0.90	42.0
Barataria Bay	2	0.11	5.20	0.12	48.0	8.6	0.027	263.0	18.1	13.8	0.40	0.20	1.20	48.0
Barataria Bay	9	0.11	7.36	0.32	55.6	11.4	0.039	142.3	**21.3**	16.3	0.54	0.45	0.97	65.5
Barataria Bay	10	0.11	7.92	0.26	45.9	12.7	0.043	266.8	19.2	13.0	1.06	0.36	1.26	68.0
Little Lake	5	0.13	4.40	0.14	33.0	7.2	0.016	303.0	16.1	13.1	0.30	0.30	1.30	51.0
Lake De Cade	16	0.13	7.38	0.32	44.6	13.4	0.04	380.7	18.8	18.9	0.47	0.49	1.20	74.6
Lake Felicity	14	0.13	8.81	0.50	69.2	17.4	0.066	513.3	**25.2**	23.9	0.60	0.77	1.29	92.3
Lake Mechant	15	0.10	5.87	0.31	42.1	9.3	0.041	268.0	16.7	17.5	0.34	0.29	0.86	63.2
Lake Palourde	8	0.15	**9.40**	0.39	62.0	17.7	0.058	516.0	**29.1**	21.8	0.90	0.40	1.80	87.0
Lake Verret	17	0.19	**8.55**	0.26	67.1	21.1	0.086	778.6	**35.4**	28.7	0.58	0.41	2.78	124.5
Bayou Terrebonne	18	0.13	**8.77**	0.50	77.2	23.7	0.048	616.4	**34.2**	21.8	0.96	0.60	1.64	107.3
Caillou Bay	4	0.12	5.40	0.16	54.0	12.5	0.029	375.0	**21.2**	16.0	0.30	0.30	1.50	69.0
Lake Pelto	7	0.13	5.90	0.15	51.0	9.2	0.026	278.0	13.5	13.0	0.80	0.30	1.10	54.0
Lake Raccourci	6	0.12	5.50	0.19	55.0	11.6	0.036	360.0	**21.2**	17.9	0.80	0.40	1.50	68.0
Terrebonne Bay	3	0.17	7.50	0.15	46.0	11.2	0.031	331.0	18.6	17.4	0.80	0.30	1.20	63.0
Terrebonne Bay	11	0.11	7.27	0.22	43.4	10.8	0.053	549.2	16.7	14.0	0.12	0.37	1.12	71.3
Terrebonne Bay	12	0.12	**25.94**	0.41	55.3	16.9	0.045	590.4	**23.9**	21.0	0.71	0.63	1.36	84.9
Terrebonne Bay	13	0.09	5.57	0.17	30.4	7.1	0.031	272.4	14.4	15.2	0.20	0.05	0.90	55.7
ERL guideline value:		*1.00*	*8.2*	*1.2*	*81*	*34.0*	*0.15*	*n.a.*	*20.9*	*46.7*	*n.a.*	*n.a.*	*n.a.*	*150*
ERM guideline value:			*70*						*51.6*					
% effects at intermed. conc.:			*11%*		*21%*					*17%*				

n.a.: not available

Table 29b. Concentrations of specific organic contaminants in sediments (in ng/g dry weight), determined under the EMAP Program (1991 & 1992). None of the concentrations exceed guideline values reported by Long et al. (1995), for those contaminants for which these values have been determined (see footnote). Guideline values are effects range-low (ERL; "effects possible").

	site #	mono- butyltin	di- butyltin	tri- butyltin	total butyltin	total alkanes	total isoprenoids	lmw PAHs	hmw PAHs	total PAHs	total PCBs
Barataria Bay	1	n.d.	n.d.	n.d.	n.d.	2108	12.3	10.0	42.1	52.2	0.260
Barataria Bay	2	n.d.	n.d.	n.d.	n.d.	1684	25.8	11.8	31.7	43.4	0.200
Barataria Bay	9	2	1	1	4	3810	198.3	41.8	92.1	133.9	0.805
Barataria Bay	10	2	1	1	4	5498	360.5	66.1	115.5	181.6	1.778
Little Lake	5	n.d..	n.d.	n.d.	n.d.	2057	19.2	11.8	42.7	54.5	0.170
Lake De Cade	16	1	1	1	3	4661	333.0	50.3	109.1	159.5	0.702
Lake Felicity	14	4	2	3	9	9583	369.3	78.7	135.4	214.1	2.049
Lake Mechant	15	2	2	2	6	5633	342.9	62.9	103.3	166.2	0.806
Lake Palourde	8	n.d.	n.d.	2	2	5130	89.8	40.8	163.2	204.0	3.840
Lake Verret	17	1	2	2	5	8725	430.3	60.4	409.9	470.3	1.375
Bayou Terrebonne	18	1	1	1	3	5837	270.9	33.2	28.3	61.5	0.946
Caillou Bay	4	n.d.	n.d.	n.d.	n.d.	1195	14.6	14.0	51.4	65.4	0.580
Lake Pelto	7	n.d.	n.d.	n.d.	n.d.	2101	24.6	14.0	51.8	65.8	0.350
Lake Raccourci	6	n.d.	n.d.	n.d.	n.d.	2675	25.4	29.4	82.2	111.6	0.940
Terrebonne Bay	3	1	1	n.d.	2	1710	29.3	20.2	107.9	128.1	11.930
Terrebonne Bay	11	3	3	2	8	4850	302.3	99.9	97.9	197.8	3.032
Terrebonne Bay	12	2	1	2	5	3890	179.8	33.1	81.1	114.2	0.635
Terrebonne Bay	13	1	1	1	3	1753	123.5	33.6	61.6	95.2	0.697
ERL guideline value:		*n.a.*	*n.a.*	*n.a.*	*n.a.*	*n.a.*	*n.a.*	*552*	*1700*	*4022*	*23*

n.d.: not detected. n.a.: not available

Table 29c. Concentrations of specific pesticides in sediments (in ng/g dry weight), determined under the EMAP Program (1991 & 1992). Concentrations in bold exceed guideline values (see footnote) reported by Long & Morgan, 1990 (indicated by *; values from Summers et al., 1993) and Long et al., 1995 (indicated by ~). Guideline values are effects range-low (ERL; "effects possible") and effects range median (ERM; "effects likely").

	site #	aldrin	dieldrin	endrin	total BHC	chlordane	total DDT & metab.s	hexachloro-benzene	lindane	total nonachlor
Barataria Bay	1	0.010	n.d.	n.d.	0.010	0.020	0.070	0.010	0.010	n.d.
Barataria Bay	2	0.010	n.d.	n.d.	0.010	0.030	0.090	n.d.	0.010	0.010
Barataria Bay	9	0.015	0.005	0.004	0.063	0.049	0.077	n.d.	0.038	0.006
Barataria Bay	10	n.d.	n.d.	n.d.	0.334	n.d.	n.d.	n.d.	n.d.	n.d.
Little Lake	5	0.010	n.d.	n.d.	n.d.	n.d.	0.160	n.d.	n.d.	n.d.
Lake De Cade	16	n.d.	n.d.	n.d.	0.204	n.d.	0.449	n.d.	n.d.	n.d.
Lake Felicity	14	0.028	**0.033**	n.d.	0.205	0.126	0.113	0.002	0.159	0.008
Lake Mechant	15	0.025	0.014	n.d.	0.108	0.021	0.086	0.001	0.084	0.003
Lake Palourde	8	n.d.	n.d.	**0.110**	0.060	**0.500**	0.650	n.d.	n.d.	0.170
Lake Verret	17	0.015	**0.025**	n.d.	0.208	0.338	0.304	0.010	0.184	0.083
Bayou Terrebonne	18	n.d.	n.d.	n.d.	0.251	n.d.	n.d.	n.d.	n.d.	n.d.
Caillou Bay	4	0.010	**0.020**	n.d.	n.d.	n.d.	0.390	0.010	n.d.	n.d.
Lake Pelto	7	n.d.	n.d.	n.d.	0.100	n.d.	0.070	n.d.	n.d.	n.d.
Lake Raccourci	6	n.d.	n.d.	n.d.	n.d.	n.d.	0.080	n.d.	n.d.	n.d.
Terrebonne Bay	3	n.d.	n.d.	n.d.	n.d.	0.080	n.d.	n.d.	n.d.	0.080
Terrebonne Bay	11	n.d.	n.d.	n.d.	0.294	n.d.	0.182	0.053	n.d.	n.d.
Terrebonne Bay	12	0.019	0.008	0.003	0.065	0.030	0.064	0.001	0.034	0.010
Terrebonne Bay	13	0.008	0.006	n.d.	0.071	0.048	0.054	0.001	0.059	0.008
ERL guideline value:		*n.a.*	*0.02**	*0.02**	*n.a.*	*0.5**	*1.58~*	*n.a.*	*n.a.*	*n.a.*
ERM guideline value:			*8**	*45**		*6**				

n.d.: not detected. n.a.: not available criteria are shown at the bottom of table 29. Pollutant levels exceeding the "10% effects" or ERL guideline were As (at Lake Palourde, Lake Verret, Bayou Terrebonne, and one of the Terrebonne Bay sites), Ni (at one of the Barataria Bay sites, Lake Felicity, Lake Palourde, Lake Verret, Bayou Terrebonne, Lake Raccourci, and one of the Terrebonne Bay sites), dieldrin (at Lake Felicity, Lake Verret, and Caillou Bay), endrin (at Lake Palourde) and chlordane (again at Lake Palourde). In addition, the chromium concentration in the Bayou Terrebonne site approached that ERL guideline value. Tributyltin (TBT) concentrations in sediment samples did not exceed the 5 ng g^{-1} cited by Summers et al. (1993) as a clear indicator of conditions degraded by TBT. Sites with more than two variables exceeding the "10% effects" criteria were Lakes Verret and Palourde, both located in the central western part of the Barataria basin. Except for Lake Felicity and Caillou Bay, the presence of pollutant levels in excess of the criteria values agree well with sediment toxicity (see above).

Tissue contaminant levels. Concentrations of elements and organic contaminants reported for shrimp and fish (several species of catfish and Atlantic croaker) are reported in table 30. Not all sites are equally represented. For example, no samples were collected from Lake Palourde or Little Lake, while only one sample (consisting of two individuals of one catfish species) was collected from Lake Verret. This is unfortunate, considering the potential for contaminant problems expected for these sites on the basis of sediment toxicity and sediment contaminant levels.

The following conclusions can be drawn on the basis of a comparison of these tissue contaminant levels with FDA and international health criteria (keeping in mind that criteria have not been established for many of the contaminants). Levels in shrimp and Atlantic croaker did not exceed the health criteria listed by Summers et al. (1993). However, these levels were exceeded in several catfish samples. Levels of arsenic exceeding health criteria were found in catfish (hardhead and/or gafftopsail) from Barataria Bay (one of the four sites there), Lake Felicity, Bayou Terrebonne, and two of the four sites in Terrebonne Bay. On the basis of EPA established limits for contaminants for human health risk for carcinogens (Stober 1992, cited in Summers et al. 1993), an 10^{-6} incremental risk for As occurs at a tissue level of <1 µg As g wet weight^{-1}. Assuming a wet:dry weight ratio of 8, this translates into a health criterion of <8 µg g dry weight^{-1}. This value was exceed in many (7 out of 17) of the catfish samples. For the organic contaminants, an aldrin value corresponding to a 10^{-6} incremental cancer risk was exceeded for an Atlantic croaker sample from Lake Pelto. The same croaker sample had dieldrin levels exceeding the 10^{-6} incremental risk value for this pesticide. A heptachlor epoxide value of 17.6 ng g dry weight^{-1} in a Terrebonne Bay catfish sample also exceeded the 10^{-6} incremental risk value. These data indicate that contaminant levels in fish tissues (especially catfish) occasionally reach levels that may be problematic.

Table 30a. Contaminant levels in edible portions of fish and shrimp collected under the EMAP Program. Metal concentrations in µg/g dry weight, PCB concentrations in ng/g dry weight. Each value is based on 1 sample (# fish or shrimp per sample shown in table). Missing values indicate that contaminants were not detected. Values in bold exceed health criteria levels.

site name	site #	species	# in sample	Ag	Al	As	Cd	Cr	Cu	Hg	Ni	Pb	Se	Sn	Zn	PCBs
Barataria Bay	1	Gafftopsail catfish	n.r.*													43.4
		White shrimp	n.r.													
Barataria Bay	2	Hardhead catfish	n.r.	0.16	3.2	15		0.1	1.8	0.025	0.4		0.7	1.41	50.7	37.8
Barataria Bay	9	Hardhead catfish	7	0.06		**24**		0.4		3.477			1.3	1.5	18.4	20.0
		Atlantic croaker	2													103.8
		Shrimp sp.	1		14.7	2.1	0.1	0.4	1.6	0.105	0.4		1.4	3.54	18.6	7.6
Barataria Bay	10	Hardhead catfish	5	0.37				0.1		0.688	0.8	0.1	2.4	3.92	25.4	32.6
Lake De Cade	16	Atlantic croaker	10	1.32	10.9	0.5	0.5	1	2.1	0.231	0.3	0.6	0.6	1.87	21.9	14.2
	16	Shrimp sp.	4		4.3	0.6		0.3	4.8	0.101	0.4	0.1	0.4	1.31	42.3	4.5
Lake Felicity	14	Hardhead catfish	1	0.21		**47.9**		0.2		0.445	0.4	0.2	2.4	3.51	32.9	20.7
		Gafftopsail catfish	6	0.87	19.4	0.4	0.7	0.9	2.1	0.065	0.6	0.1	1.1	3.48	13.1	58.4
		Atlantic croaker	10	1.57	17.4	1.1	0.7	0.8	1	0.571	0.6	0.9		1.64	13.5	15.2
		Shrimp sp.	2		8.5	0.9		0.3	5.8		0.2	0.1	0.2		12.8	3.0
Lake Mechant	15	Hardhead catfish	3													33.1
		Gafftopsail catfish	10	0.56	14.9	0.2		0.1	1.6	0.110	0.1	0.4	0.6	1.22	2.4	41.0
		Atlantic croaker	8	0.13	11.9	0.6	0.4	0.9	1.4	0.157	0.4	0.2	0.3	1.09	15.3	9.6
Lake Verret	17	Blue catfish	2	1.43	12.6	0.5	0.1		3.2	0.132	0.7	0.2	0.9	1.63	14.3	8.1
B. Terrebonne	18	Gafftopsail catfish	4			**20.3**		0.4		0.154	0.7	0.3	1.8	1.36	15.9	22.6
		Atlantic croaker	5													
		Shrimp sp.	10	0.1	12.7				5.5	0.120	0.6		0.9	0.06	5.7	2.0
Caillou Bay	4	Hardhead catfish	n.r.		6.3	1.1	1.5	0.2	2.3	0.082	0.3		0.2	0.66	59.7	32.4
Lake Pelto	7	Hardhead catfish	n.r.	1.53	9			0.2	0.8	1.217			0.2	0.56	30.2	38.9
		Gafftopsail catfish	n.r.							0.156						65.4
		Atlantic croaker	n.r.	0.54	18.7	1.6		0.5	0.4	0.455			0.8	0.94	26.8	94.3
Lake Raccourci	6	Gafftopsail catfish	n.r.							0.001						32.0
		Atlantic croaker	n.r.	0.25	5.2	10.3		0.1	1.4	0.003	0.3	0.1	0.8	0.42	31.9	18.6
Terrebonne Bay	3	Hardhead catfish	n.r.	0.05		3.9		0.6	1.4	0.015	0.1		0.1		28.6	80.4
Terrebonne Bay	11	Hardhead catfish	1	0.06		**21.7**		0.4		0.549			1.4	0.14	21.2	19.6
		Gafftopsail catfish	3													15.7
		Atlantic croaker	10	1.14	6.9	0.2	1	1.1	1.3	0.247		0.5	1.4	2.83	20.4	19.4
		Shrimp sp.	9	0.28	14	0.8		0.4	8.7	0.127	0.9	0.1	1.1	0.68	9.9	2.8
Terrebonne Bay	12	Gafftopsail catfish	11	0.44		**49.1**		0.1		0.284	0.3	0.2	1.5	0.07	71.3	20.3
		Atlantic croaker	10		12	0.3	0.6	0.5	1.6	0.067	0.9		1.2	2.51	11.2	21.9
		Shrimp sp.	2		9.5	5	0.1	0.5	4.4	0.095	0.4		3.4	1.62	23.7	9.1
Terrebonne Bay	13	Hardhead catfish	2	0.04	0.2	7.2		0.7		0.491			2.1	0.65	15.7	24.5
health criteria: 500				*n.a.**	*n.a.*	*16*	*4*	*8*	*120*	*8*	*n.a.*	*4*	*8*	*n.a.*	*480*	

* n.r.: not reported n.a.: not available

Table 30b. Contaminant levels (ng/g dry weight) in edible portions of fish and shrimp collected under the EMAP Program. Each value is based on 1 sample (# fish or shrimp per sample shown in table). Missing values indicate that contaminants were not detected. Hexachlorobenzene, toxaphene and lindane were omitted (not detected in any samples).

site name	site#	species	# in sample	DDD	DDE	DDT	aldrin	chlor-dane	diel-drin	endo-sulfan	endrin	hepta-chlor	heptachlor-epoxide	mirex	trans-nonachlor
Barataria Bay	1	Gafftopsail catfish	n.r.*	30.6	1.7	18.3		6.3	3.4					69.1	5
	1	White shrimp	n.r.											5.2	
Barataria Bay	2	Hardhead catfish	n.r.	14.7		13.4		4.4						5.8	2.1
Barataria Bay	9	Hardhead catfish	7		3.0	2.0								4	
	9	Atlantic croaker	2	12.2	9.9	3.7		3.7		4		11.3	4.1	5.7	
	9	Shrimp sp.	1	8.3		18.5						2.05		4.05	
Barataria Bay	10	Hardhead catfish	5	1.9	13.7	23.2								4.7	1.5
Lake De Cade	16	Atlantic croaker	10			4.8									
	16	Shrimp sp.	4			16.1								9.3	
Lake Felicity	14	Hardhead catfish	1		10.2	5.8								6.3	1.6
	14	Gafftopsail catfish	6		31.8	42.8								108.4	
	14	Atlantic croaker	10			4.6									
	14	Shrimp sp.	2			53.1								5.6	
Lake Mechant	15	Hardhead catfish	3		4.5							7.5			
	15	Gafftopsail catfish	10	9.2	53.4	38.2		8.0	3.8		2.7			103.2	4.1
	15	Atlantic croaker	8		1.1	10.1								3.8	
Lake Verret	17	Blue catfish	2		5.8	5.3		0.9						62.5	
B. Terrebonne	18	Gafftopsail catfish	4		6.7	6.1								1.6	
	18	Atlantic croaker	5												
	18	Shrimp sp.	10			8.3								3.1	
Caillou Bay	4	Hardhead catfish	n.r.	21.4		14.7		1.4	3.7					4	1.7
Lake Pelto	7	Hardhead catfish	n.r.	3.6		4.0	3.7		1.5				1.7	5.9	
	7	Gafftopsail catfish	n.r.	63.1		23.0		5.9						10.9	6.2
	7	Atlantic croaker	n.r.	12.9	3.2	11.1	15.9	8.6	11.5		11.8		2.2	21.6	3.7
Lake Raccourci	6	Gafftopsail catfish	n.r.	26.0		12.6		3.9						1.9	3.3
	6	Atlantic croaker	n.r.	3.0		6.5		1.9	1.6					5	
Terrebonne Bay	3	Hardhead catfish	n.r.	12.5	7.7	23.2		8.4			2.6		17.6		
Terrebonne Bay	11	Hardhead catfish	1			3.2								4.4	
	11	Gafftopsail catfish	3		6.7	3.5									
	11	Atlantic croaker	10			17.4									
	11	Shrimp sp.	9			14.3								4	
Terrebonne Bay	12	Gafftopsail catfish	11	1.6	10.6	5.2								3.6	1.2
	12	Atlantic croaker	10			18.2									
	12	Shrimp sp.	2			8.2								2.6	
Terrebonne Bay	13	Hardhead catfish	2		3	25.5								4.7	
health criteria:				*40000*	*40000*	*40000*	*2400*	*2400*	*2400*	*n.a.**	*2400*	*2400*	*2400*	*800*	*n.a*

* n.r.: not reported n.a.: not available

Pollution Status Based on Mussel Watch Project

The seven Mussel Watch sites in the Barataria and Terrebonne basins all fall in the southern part of the basins (see figure 71). Because this data set contains several samples per site (initial sampling consisted of three composite samples per site, while recent sampling has been reduced to one composite sample per site), analyses were made to compare contaminant levels among the sites. Results on tissue levels in oysters are reported in figures 83–89.

Concentrations of silver, cadmium, and copper were all highest at the Atchafalaya Bay at Oyster Bayou (ABOB) and the Mississippi River Tiger Pass (MRTP) sites. No health criteria are available for silver. For cadmium, average concentrations at these two sites exceed the health criterion of 4 µg g dry wt^{-1}. Similarly, mean copper levels at these sites exceed the health criterion of 120 µg g dry wt^{-1}.

Arsenic concentrations were highest in oysters from ABOB and Barataria Bay at Middle Bank (BBMB). Average As levels (as well as individual values) at these sites were below the health criterion of 16 µg g dry wt^{-1}. However, concentrations are sometimes above the <10 µg g dry wt^{-1} criterion for a 10^{-6} incremental cancer risk.

Chromium levels were highest at Terrebonne–Bay Lake Barre (TBLB) but well below health criteria levels.

Mercury levels were highest for BBMB and TBLB but did not reach health criteria levels.

Levels of manganese were highest at TBLB. No health effect criteria are available for this element.

Nickel concentrations did not differ significantly among the sites. Observed values did not exceed health criteria levels for a 10^{-6} incremental cancer risk.

Lead levels showed minor differences among some sites. Concentrations averaged about 5 µg g dry wt^{-1}, well below the health criterion listed by Summers et al. (1993).

Selenium levels were highest at the ABOB site. Concentrations did not approach health criteria levels.

Zinc concentrations showed some minor variations among sites. Concentrations at all sites averaged about 2,000 µg g dry wt^{-1}; well above the health criterion of 480 µg g dry wt^{-1} reported by Summers et al. (1993). The fact that zinc levels are high across the board indicates that this might not be due to pollution.

Organotin concentrations at the BBMB site were about an order of magnitude higher than at the next highest site (MRTP). An earlier report (Garcia-Romero et al. 1993) also reported the high butyltin concentrations at this site and showed that these levels appeared to increase over the three years covered by that study (in contrast to the situation at most other sites). No health criteria were found for the butyltins. However, the high butyltin concentrations at the MRTP site should be a cause for concern.

Levels of PCBs, DDT, dieldrin, and chlordane were all highest at the MRTP site. Levels of these organic contaminants did not exceed the regular health criteria listed by

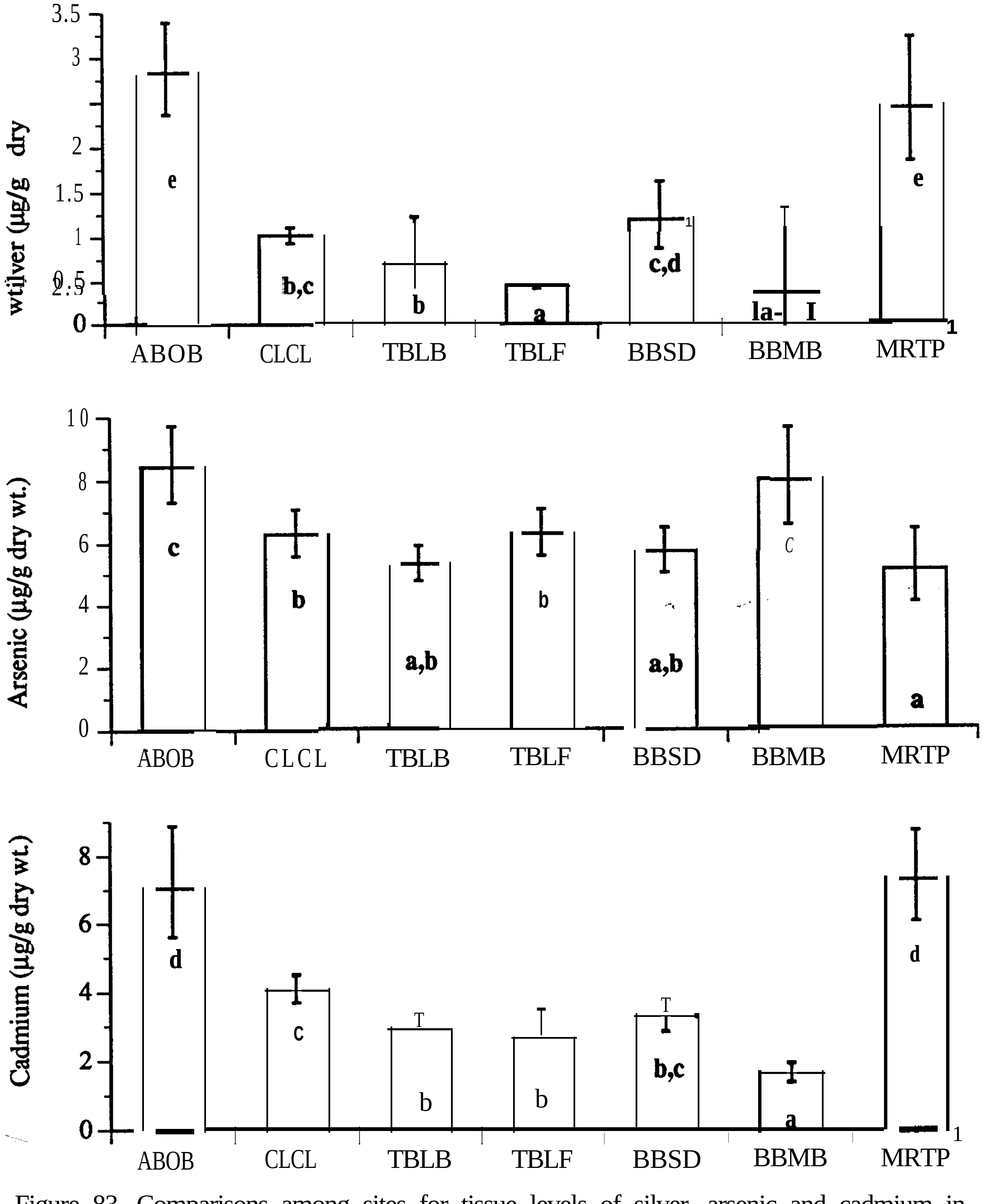


Figure 83. Comparisons among sites for tissue levels of silver, arsenic and cadmium in oysters collected under the Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

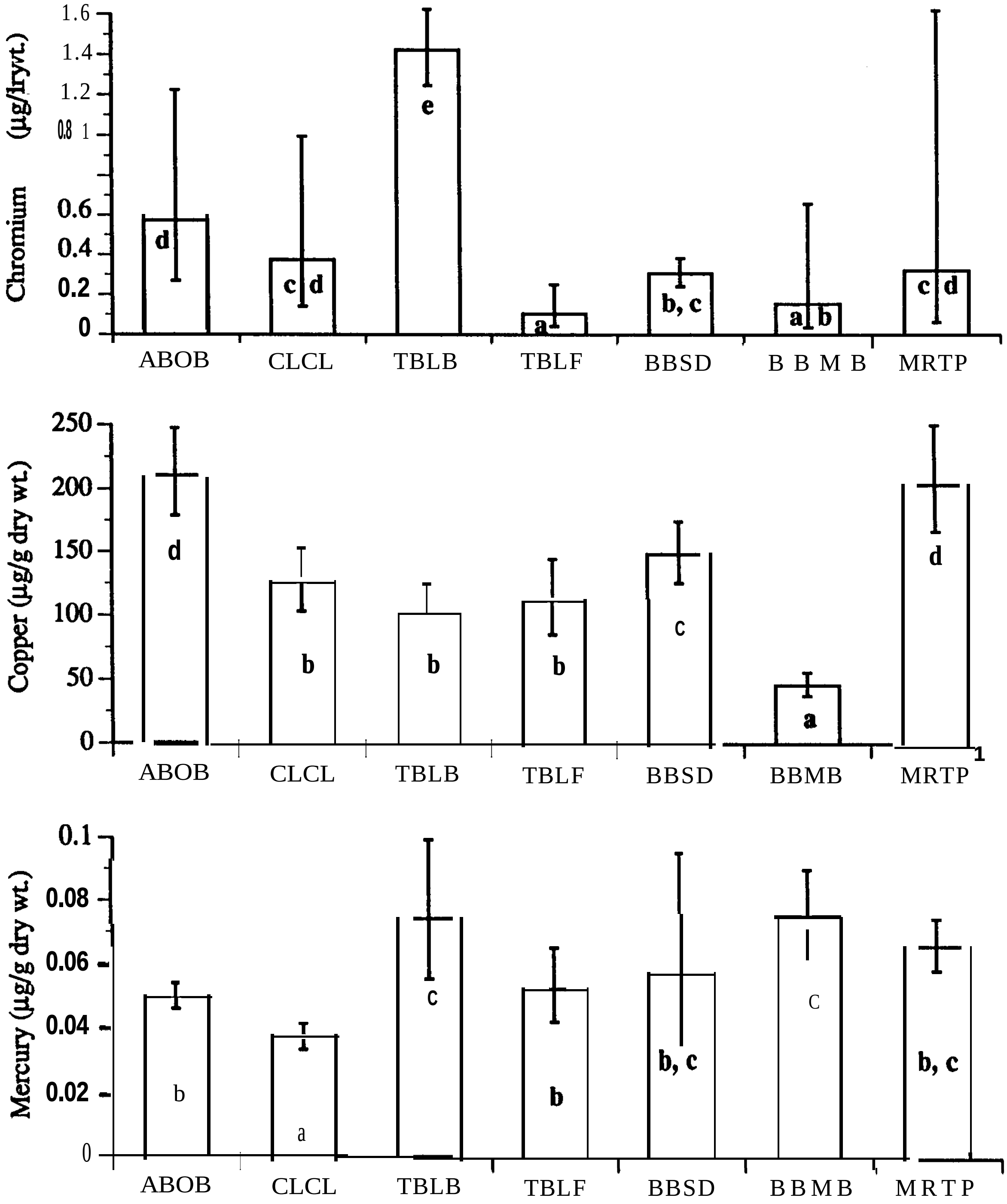


Figure 84. Comparisons among sites for tissue levels of chromium, copper and mercury in oysters collected under the Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

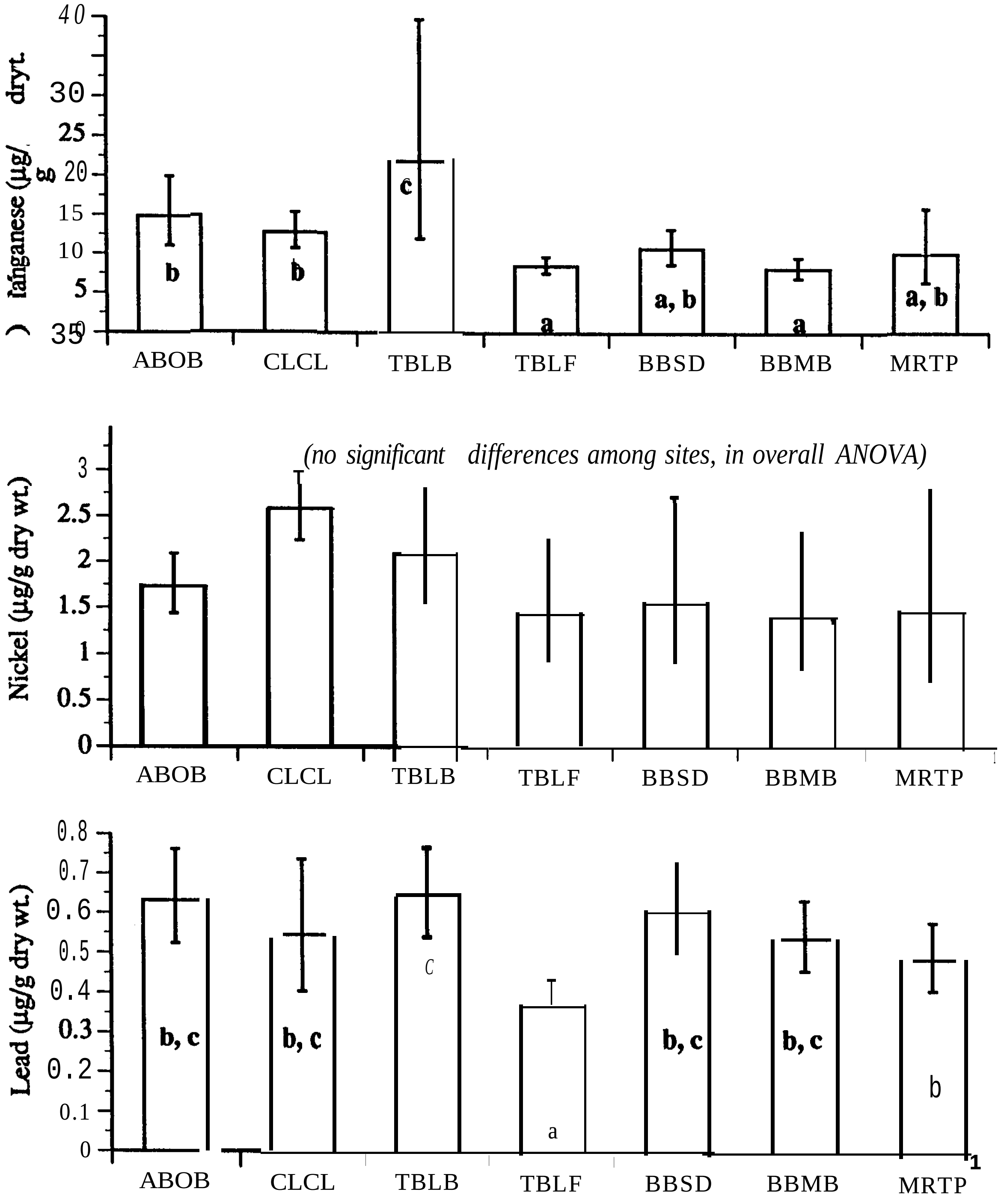


Figure 85. Comparisons among sites for tissue levels of manganese, nickel and lead in oysters collected underthe Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

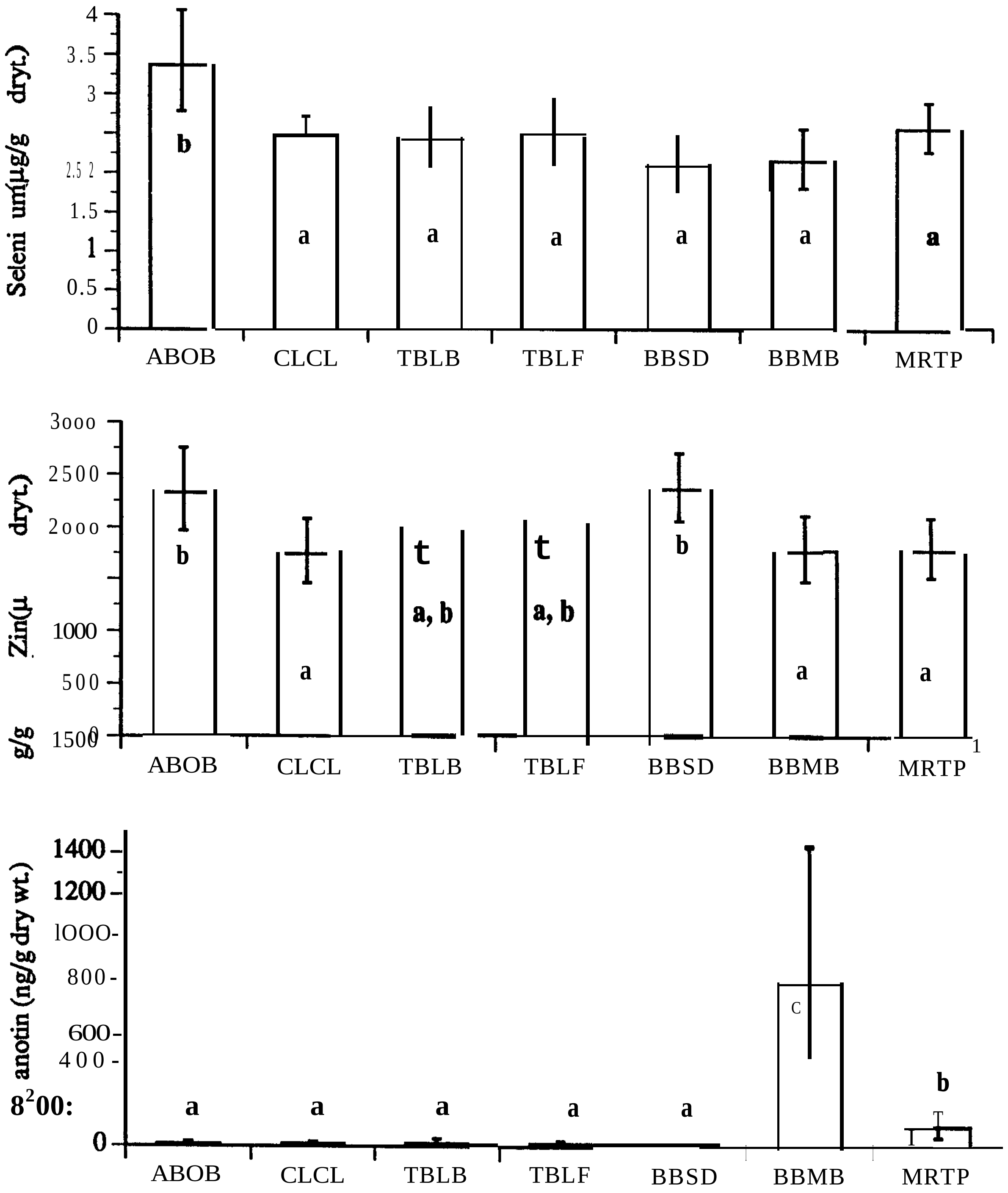


Figure 86. Comparisons among sites for tissue levels of selenium, zinc and organotin in oysters collected under the Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

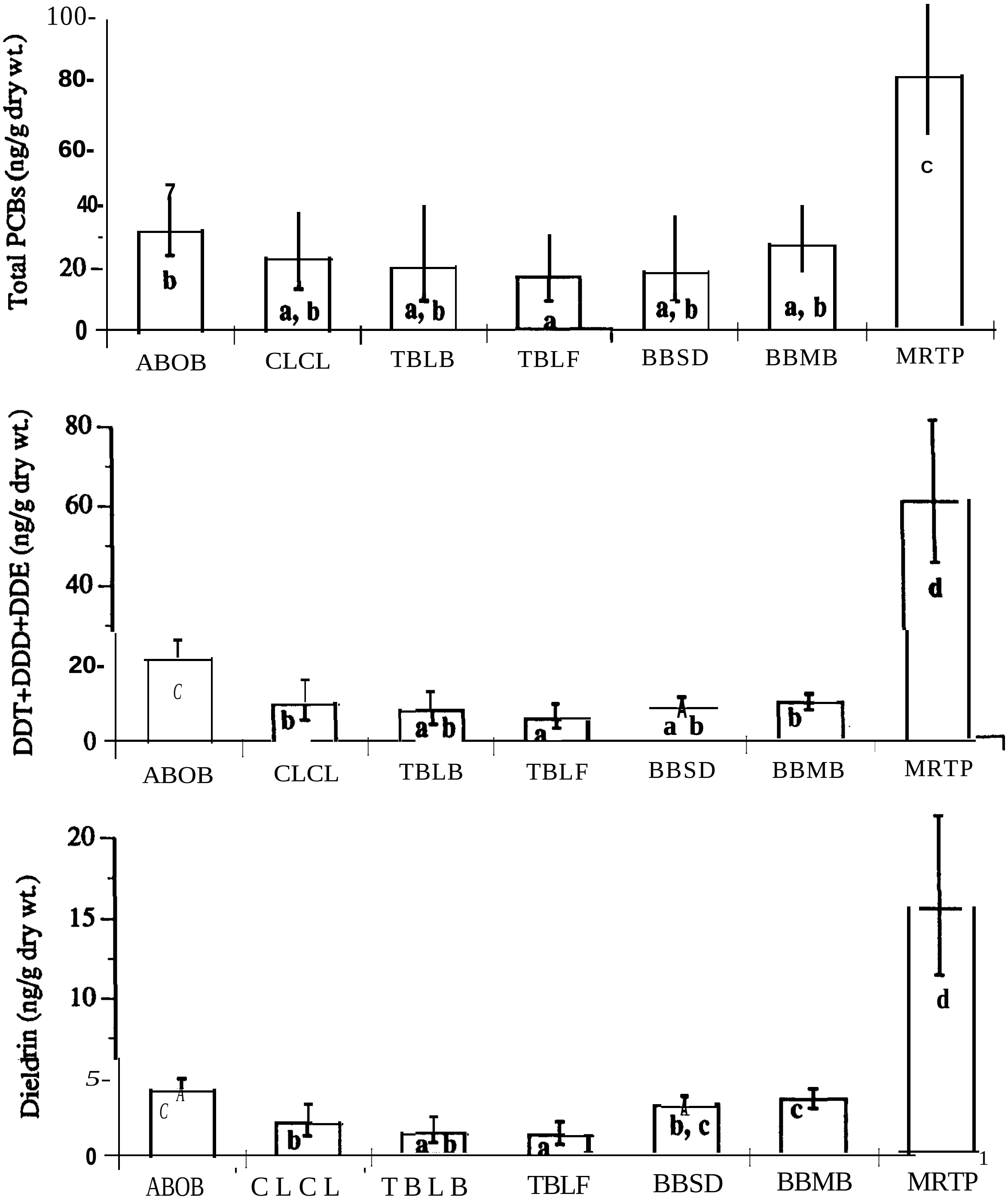


Figure 87. Comparisons among sites for tissue levels of Total PCBs, DDT+DDD+DDE and Dieldrin in oysters collected under the Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

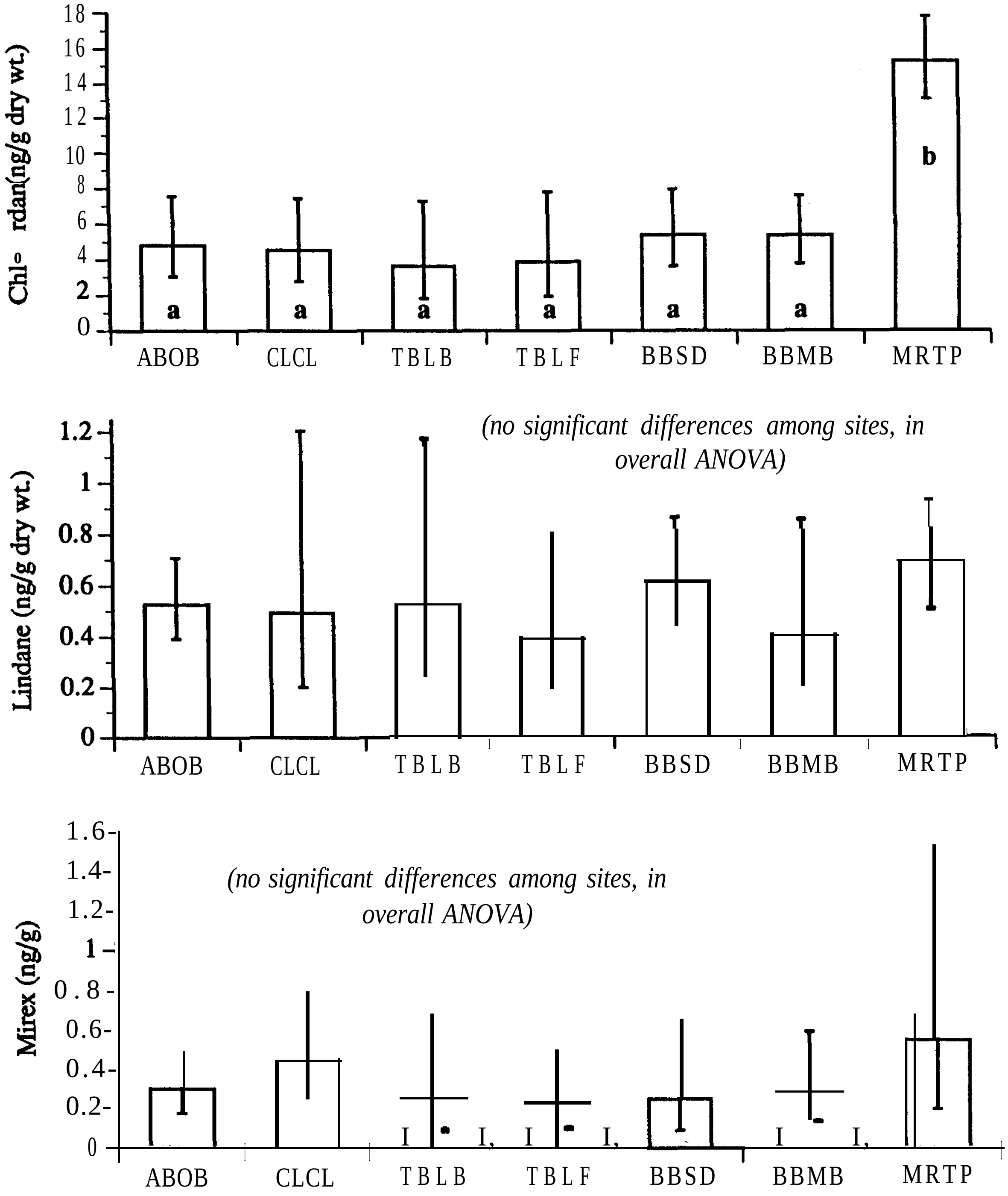
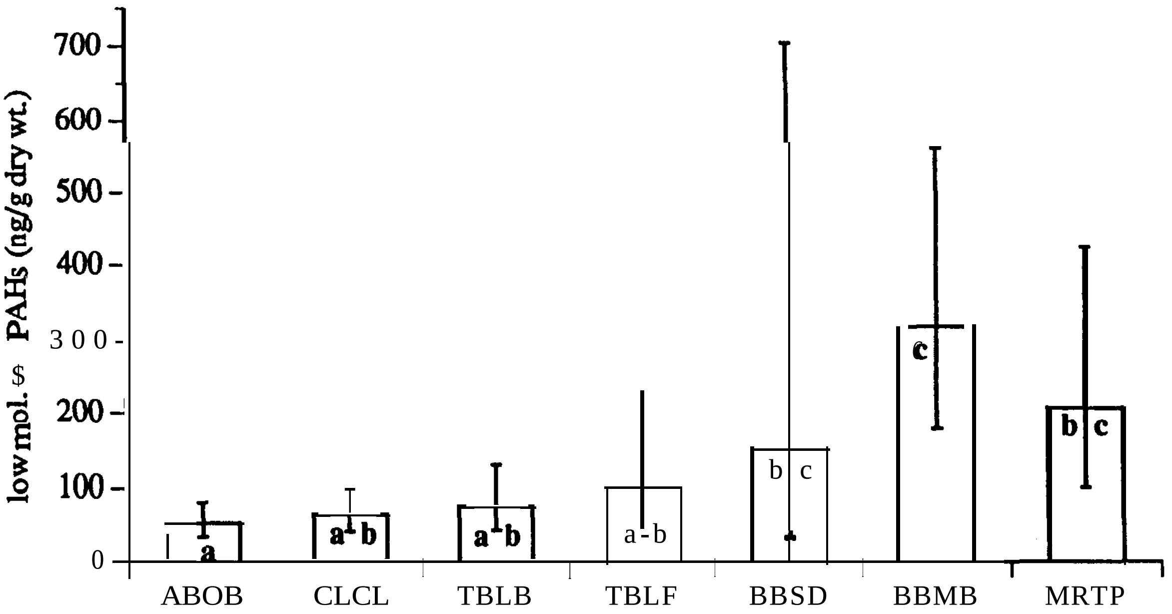


Figure 88. Comparisons among sites for tissue levels of Chlordane, Lindane and Mirex in oysters collected under the Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.



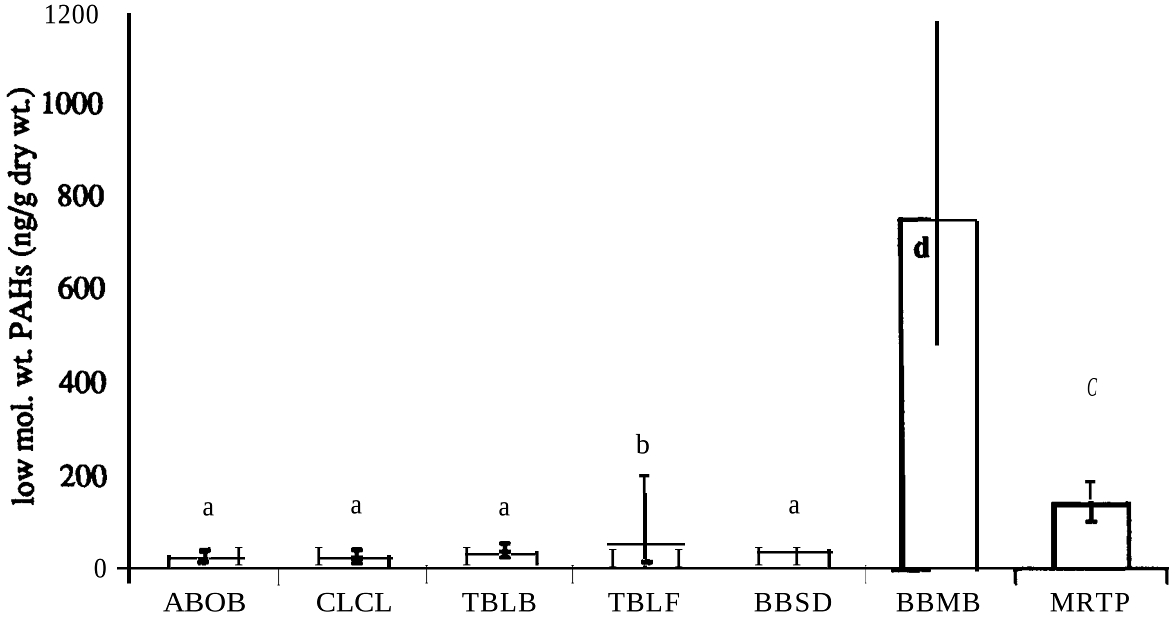


Figure 89. Comparisons among sites for tissue levels of low and high molecular weight PAHs in oysters collected under the Mussel Watch Project. Error bars are 95% confidence limits for the mean. Columns with the same letter are not significantly different in post-hoc comparisons (Fishers PLSD test). Computations and analyses used log-transformed data, while results were backtransformed for reporting purposes.

Summers et al. (1993). Dieldrin levels did exceed the value corresponding to a 10^{-6} incremental cancer risk.

Levels of lindane and mirex did not differ among sites, while values for neither of these pesticides exceeded health criteria.

PAH levels were highest MRTP, BBMB, and BBSS. No health criteria were available for comparison.

Conclusions on Status of Contaminant Levels

The status analyses did not reveal any major problems with respect to contaminants in the Barataria and Terrebonne basins; however, some minor contamination problems were evident. The LDEQ data showed that levels of Cd, Cu, and Pb in water from Bayou Segnette, Bayou Choctaw, and the Lower Grand River occasionally reached levels exceeding EPA criteria for the protection of aquatic life, while elevated levels of As and Hg in water may possibly affect human health (through the ingestion of water and/or consumption of fish). The EMAP-E data indicated possible problems for many of the 18 sites covered by the current data set. For some sites (notably Lakes Verret and Palourde and to a lesser extent Bayou Terrebonne), this is reflected in sediment toxicity and sediment contaminant levels. This may indicate a general environmental impact. For the sediment at these sites, elevated levels of arsenic, nickel, dieldrin, endrin, and chlordane may be responsible for possible effects. For another group of sites (Barataria and Terrebonne bays, Lake Felicity, and Bayou Terrebonne), potential problems are associated with the human consumption of fish. Fish tissue contaminant analyses point to arsenic and several pesticides (mirex, aldrin, dieldrin, and heptachlor) as potential health problems associated with the consumption of fish (specifically in hardhead and gafftopsail catfish, which were the most commonly collected species). Levels in shrimp samples did not exceed health criteria. The Mussel Watch data pointed out problems for As, Cd, and Cu in oysters from the Oyster Bayou and Tiger Pass sites (as well as dieldrin in Tiger Pass oysters) to indicate that these contaminants in the Mississippi and Atchafalaya rivers are responsible for the elevated concentrations in oysters. In addition, high levels of organotin in oysters collected in Barataria Bay at Middle Bank should be a cause for concern.

Impacts of Produced Water Discharges

Several factors determine the fate and effects of produced waters in coastal environments and organisms (Rabalais et al. 1992). These include the volume and composition of the discharge and the hydrologic and physical characteristics of the receiving environment. Comparisons of the produced water discharges can be approached by calculations of loadings of particular contaminants. A lower volume discharge may be compensated for by a high concentration of a single constituent, or a contaminant may not be particularly concentrated in the effluent, but the volume may be large and the total loading similarly large. The hydrology and sediment will influence the dilution potential of the receiving environment to the discharge. Dilution of water-soluble contaminants would be influenced primarily by the volume of the receiving waters, the current velocity, and the potential for resuspension of sediment. Dispersion of sediment-adsorbed contaminants would be influenced by bed shear stress, sedimentation rates, and the grain size distribution of surface sediment. Receiving environments can be classified with a high, medium, or low dilution potential. A number of studies of the fate and effects of produced water discharges (Boesch and Rabalais 1989a, b; St. Pé 1990; Rabalais et al. 1991b; API 1991) have been conducted throughout the Barataria and Terrebonne estuaries (see Rabalais and St. Pé 1991), across the full range of loadings of contaminants and dilution potential of the receiving environments. A summary of the results follows; readers are referred to the listed reports for more details.

Dispersion of Effluents

Because produced water effluents have salinity levels in excess of that of ambient sea water and may act as a dense plume on discharge into receiving waters, the salt content of the bottom waters may be used as a conservative tracer of the brine plume. Clear density plumes (up to 800 m) are identifiable only adjacent to large volume discharges, or in receiving environments with limited flows (up to 300 m). The St. Pé (1990) study in brackish water systems with limited flow demonstrated that chlorinities of overlying water and sediment were highest near the discharge point, and sediment demonstrated a greater impact further from the point of discharge than the overlying water. St. Pé (1990) further demonstrated that the produced water influences on chloride concentrations of the receiving water body were considerably more apparent in bottom sediment in comparison to those effects measured in the water column. The elevated salinity values above that of the ambient water observed across a suite of sites, however, was within the range of the physiological capabilities of euryhaline estuarine organisms.

Other tracers of produced waters (e.g., volatile hydrocarbons) are water soluble and not conservative, but their presence in near-bottom waters indicates the extent of the produced water plume to at least the distance they were detected. The detection of volatile hydrocarbons to considerable distances from produced water effluents (300 m–750 m) occur when the loadings are high, and/or the dilution potential of the receiving environment is low. Where the dilution potential of the environment is medium, volatile hydrocarbons are detected near the discharge point and at uniformly low levels distant from the discharge or undetected in highly dispersive environments.

Sediment Contamination

The zone of sediment contamination extends to greater distances where the volume of discharge is large, the loadings of petrogenic-origin hydrocarbons are high, and the receiving environment is characterized by restricted water exchange. Regardless, some level of sediment contamination is evident adjacent to most produced water discharge sites. The zones of alkylated PAH contamination may extend up to 1,000 m–1,300 m from the discharge, but more typically range from 200 m to 500 m from the discharge. The presence of high concentrations of produced water–derived hydrocarbons to depths of 25 cm–30 cm in vertical sediment cores at some of the discharge sites indicates the long-term accumulation of these contaminants and their resistance to degradation. Similar accumulations in surficial sediment, and particularly sediment to depth, adjacent to a discontinued produced water discharge identified the long-term (at least three years) persistence of produced water origin hydrocarbons that continued to impact the benthic infaunal community. Neither metals nor radionuclides were particularly good tracers of produced water–origin hydrocarbons in the environment; although sediment at three of the four sites in the St. Pé (1990) study demonstrated a ^{226}Ra signal.

Biological Assessments

Impacts to benthic macroinfaunal communities adjacent to produced water discharges are seen as reduced abundances of organisms and reduced diversity. Benthic community impacts are severe near some discharges, localized when observed, and are not always evident. Benthic fauna are severely reduced or absent where the threshold of certain contaminants (such as volatile hydrocarbons and alkylated PAHs) are exceeded. The extent of benthic community impacts may reach as far as 800 m from the discharge. Additionally, accumulated pollutants adjacent to a discontinued discharge continued to adversely affect benthic communities at least three years after the effluent was stopped. Several bioaccumulation studies (Boesch and Rabalais 1989a, Rabalais et al. 1991b, St. Pé 1990) have demonstrated the clear potential for uptake and accumulation of produced water-origin contaminants (particularly the PAHs) by oysters, in close proximity to the discharges but also to great distances from the discharges (350 m–1,000 m). Limited accumulation of radium was observed.

The potential for elevated salinities in produced waters to affect organism survival was studied by St. Pé (1990). Mysids and sheepshead minnows both exhibited acute toxicity to the effluents, but it was apparently because of a component of the effluent other than salinity. Significant toxicity of sediment was also evident in mortality rates of the burrowing amphipod *Hyalella azteca*.

Cumulative and Future Impacts

Point source discharges of produced waters are abundant in the Barataria-Terrebonne estuarine system. Individual effluents may be low volume or well over 50,000 bbl/day. Localized (within 300 m) or broad (up to 1,000 m) impacts are identifiable adjacent to many of the discharges, e.g., contaminated sediment (surficial and to depth), impacted biological communities, and bioaccumulation of contaminants in tissues of filter-feeding organisms (oysters and mussels). In some receiving environments (well flushed, dispersive) faunal impacts may not be identifiable and contaminant levels are relatively low. The additive, or cumulative, effect of the identifiable, localized impacts attributable to produced water discharges is more difficult to define. Equally difficult is the extrapolation of site-specific impacts to ecosystem level impacts. With over 400 discharges accounting for over 900,000 bbl/day effluent in the Barataria-Terrebonne estuarine system, the cumulative effects cannot be inconsequential. While regulations will eventually end produced water discharges into most state surface waters, elevated levels of produced water origin contaminants will remain in sediments adjacent to the discharge points and potentially be resuspended and transported from the area due to natural events, boat traffic, or mobilized during dredging activity. In the only known study of a discontinued produced water discharge, contaminated sediment adjacent to the abandoned facility continued to impact benthic infauna up to three years from cessation of the effluent (the only duration examined). The potential for continued and cumulative impacts to biological communities (infauna and filter feeders) is real.

Recommendations

Produced water discharges at the surface should be phased out as the law requires. The amounts of toxic pollutants from produced waters far exceeds any other source. Injection into deep wells is a viable alternative and is the preferred disposal method in the remainder of the United States. The most recent deadline for phase out is 1997, and there should be no further extensions. In preparation for the regulations and/or poor economics of production and treatment, operations have been discontinued at several produced water facilities. The long-term impacts of sediment contamination, including biological impacts, of accumulated and resistant produced water-origin hydrocarbons should be determined. A detailed, computerized system should be developed (portions of Barataria-Terrebonne estuarine system are already on a GIS at LDEQ) to locate the produced water discharge points, historic and current, for future management considerations. Projections of contaminant levels could be made for all discharge points based on knowledge derived from the several fate and effects studies listed above. Management considerations should include decisions to clean up and restore the most severely polluted sites and/or to minimize future recontamination of overlying waters and adjacent areas through events that would resuspend the contaminated sediment. Additionally, efforts must be made to maintain the piping or other conduits used to transfer produced waters to injection wells for subsurface disposal. Produced water leaks often remain undetected until the overlying vegetative growth is irreversibly impacted (K. M. St. Pé, pers. comm.). Proper preventative maintenance to these lines is critical to maintenance of healthy wetland vegetation.

The receiving environments for the produced water discharges include open waters, well-flushed bayous, poorly flushed canals, dead-end canals, vegetated marshes, and bare sediment adjacent to water bodies. Many of the discharge facilities were built on or adjacent to wetland habitats. In many cases, open pits were used as holding facilities or separation ponds during the produced water treatment. In recovery of land following removal of facilities, habitats may or may not be restored as closely as possible to pre-existing wetland habitats. With the decline in oil and gas production and the termination of surface discharges of produced waters, a legacy of facility infrastructure will remain in Louisiana coastal habitats. While not a direct issue relevant to this review, restoration of these sites should be considered within the larger context of habitat and hydrological modifications of the Barataria-Terrebonne estuarine system (see Reed et al. 1995). Restoration relevant to the issue of contaminants is the accumulated produced water contaminants and their future management to minimize further impacts to the environment.

Identification of Water Quality Problem Areas

LDEQ Determinations as Affected by Toxicants or Sediment Contaminants

Within the estuarine basins, LDEQ has determined that Bayou Barataria Waterway (Intracoastal Waterway to Bayou Rigoletts), Bayou Rigoletts, and Bayou Perot to Little Lake are contaminated by toxics in the water column or in sediments (LDEQ 1994). In Harvey Canal, situated on the west bank of the Mississippi River in Jefferson Parish and inside the study area, phenols and volatile organics were found in the water, and sediment was heavily contaminated with metals, PAHs, and PCBs (LDEQ 1994b). The Harvey Canal near the Mississippi River is an industrial area that includes ship repair facilities. Many of these problem areas could be due to nonpoint source runoff.

While LDEQ is monitoring groundwater from several wells with known historical organic and/or metal contamination, these are all outside of and generally north of the study area. None of the 16 swimming and fish consumption advisories occur within the Barataria-Terrebonne area; although one of them, Devil's Swamp and Lake to the northwest of Baton Rouge, is in contact with the Mississippi River during flood stages of the river.

Results from Status and Trends Analyses

Stations with excessive metals in the water, two in Barataria and six in Terrebonne, are identified in table 20. The increasing discharges of total toxic chemicals over time to the estuary may be a cause for concern if the trend continues. Chemical amounts are still low for a large basin but could become a problem without due vigilance. Formaldehyde is the principle component. The amounts of hydrocarbons, metals, and radium discharged to the estuary with produced waters is troublesome.

In the Mississippi River, the station at RM 155.6 exhibited a consistent increasing trend of detection of chlorinated hydrocarbons in river water. The increases in the amount of ammonia discharged to the Mississippi River over time may be a cause for concern, especially when the already large nutrient load of the river is considered.

The sites for which contamination problems are most evident from analyses of the various data sets are Bayous Segnette and Choctaw, Lower Grand River, Lakes Verret, and Palourde, Oyster Bayou, and Tiger Pass. As can be seen in figure 90, these sites fall on the periphery of the Barataria-Terrebonne area. The interior of the basins appears relatively clean, with the exception of produced water discharges and other possible contaminant sources not covered by the state and federal monitoring programs that form the basis of this review. Contamination sources for most of the sites on the periphery are not readily apparent, i.e., a combination of point and nonpoint sources. For Oyster Bayou and Tiger Pass the elevated contaminant levels are likely related to the inflow in these areas of water from the Atchafalaya and Mississippi rivers, respectively. This also means that in cases where Mississippi River water will be diverted into marshes as a marsh restoration method, pollutant levels might increase and should be monitored. A recommended avenue of investigation would be to determine trends in relevant contaminants in the Mississippi River (i.e., elevated in waters, sediment and organisms of the estuary) and assess to what extent water quality in the interior of the basins would be affected by the input of Mississippi River water.

Toxic chemicals have been found in Mississippi River fish and organisms by the LDEQ Mississippi River Toxics Inventory Project. Toxics detected in fish and crustaceans were pesticides, metals, volatile organics including chlorinated hydrocarbons, PCBs, and base/neutrals. The concentrations of toxic chemicals found do not necessarily pose a problem to the fish or shellfish but rather to organisms including people higher in the food chain that consume them.

Contamination problems were identified in the environmental components monitored: water, sediment, and fish and shellfish. This indicates that these problems are real. Contamination is found at various sites and for various contaminants. The occurrence of elevated metal levels in water of various waterways (especially Bayous Segnette and Choctaw and Lower Grand River), sediment toxicity at various sites (in e.g., Little Lake, Bayou Terrebonne, and Lake Verret), elevated levels of metals and pesticides in sediment (especially Lakes Verret and Palourde), elevated levels of arsenic in catfish at several sites, and elevated levels of some contaminants in oysters (especially at Tiger Pass and Oyster Bayou) indicate that contamination is fairly widespread in scope. Contamination should therefore be a source of concern, though none of the contamination problems appeared serious enough to warrant immediate and drastic actions. Butyltin levels (most likely resulting from the application of TBT as an antifoulant on boats) may constitute a problem in Barataria Bay at Middle Bank.

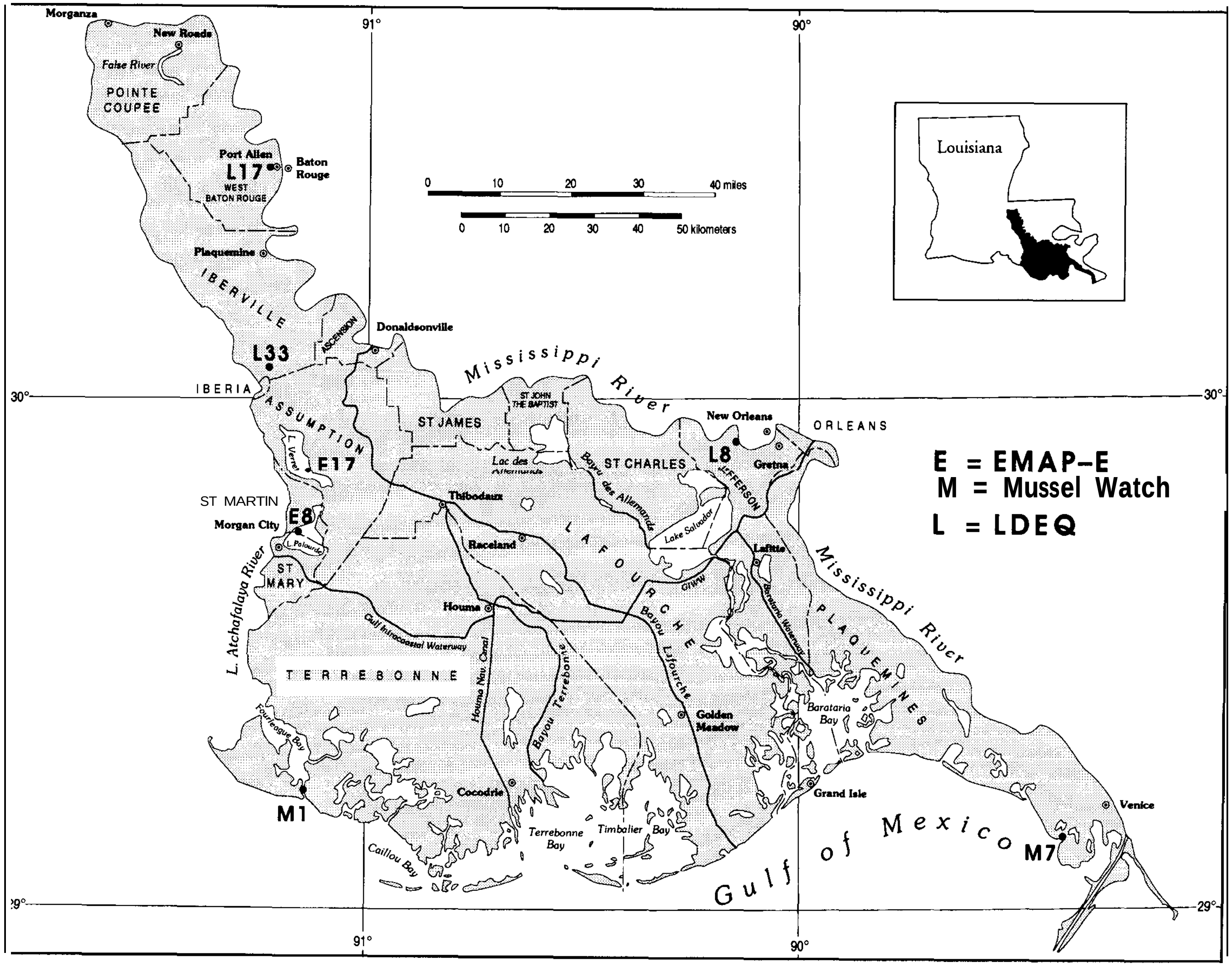


Figure 90. Location of sampling stations for which status analyses indicate contaminant levels of concern in either water, sediment, or (shell) fish. Prefixes to the numbers indicate specific monitoring programs: L for LDEQ stations, M for Mussel Watch stations, and E for EMAP stations.

Relationship Between Sources and Indicators of Ecosystem Health

It is difficult to relate contaminant sources (be it as a quantification of discharges or determinations of contaminant levels in the abiotic components such as water or sediment) with ecosystem health. This difficulty arises from the many variables that influence the ultimate effects of pollutants (e.g., behavior of the pollutant in the environment, an organism's normal sensitivity, pollutant metabolism by the organism, exposure of the organism to other stressors such as pathogens, etc.). However, laboratory studies have firmly established the relationships between pollutant levels in the environment and effects of these pollutants on organisms. This information has for instance resulted in the establishment of EPA water quality criteria for effects on aquatic organisms and their uses as well as for effects on humans (through fish consumption and water ingestion). Also, criteria have been developed for sediment levels in pollutants (e.g., Long and Morgan's criteria for sediment degradation based on pollutant levels in sediment, see Summers et al. 1993). Moreover many field studies have shown links between environmental pollutant levels and ecosystem effects (e.g., fish tumors in fish inhabiting polluted areas).

Some of the data sets available for the Barataria-Terrebonne area investigated pollutant levels in an environmental component (e.g., water). Where levels exceed health criteria, impacted ecosystem health may be expected (considering that the health criteria are based on levels of individual pollutants, whereas organisms in polluted areas are generally exposed to many pollutants simultaneously). However, information on contaminant levels alone is insufficient to accurately predict toxic effects. A much stronger case for a relationship between elevated pollutant levels and ecosystem health can be made when a multi-tiered approach is used. The "sediment quality triad," where sediment contaminants are determined together with sediment toxicity and effects on the composition of the communities residing in this sediment, is such an approach (Chapman et al. 1986). A similar approach is used in the EMAP-E study, where the analysis of sediment contaminants is combined with sediment toxicity and the analysis of contaminants in fish and shrimp, with all variables being determined at the same sites. This study thus provided very strong evidence that contaminants are having some (albeit not major) effects on ecosystem health at several of the sample sites. The presence of elevated levels of contaminants in oysters does not mean that the sites where these oysters were collected are impacted by the pollutants. Further study would be needed. However, human consumption of these organisms does mean that human health is an issue where pollutant levels exceed health criteria as was the case in some instances.

Management Recommendations

Data and Monitoring Inadequacies

Considering the potential effects of arsenic and mercury, the LDEQ water quality program would greatly benefit from enhancing its analytical capabilities for these elements. A change in the methodology used for chromium also seems beneficial. EMAP and Mussel Watch data did not point to any major problems with respect to the mercury accumulation in sediment or fish and finfish. Continued monitoring of Hg levels in water seems warranted, however, by its high toxicity and by the fact that levels in water occasionally appear to exceed water quality criteria. To make this monitoring more meaningful, Hg detection limits should be brought in line with current water quality criteria levels. Recent technological advancements make this feasible. Hg detection limits have already been reduced to 0.050 ppb. However, further reductions would still be meaningful (and possible), since the lowest ambient water quality criterion for Hg is 0.012. The determination of organic forms of mercury (e.g., methylmercury) also should be considered, as current scientific evidence indicates that Hg toxicity greatly depends on the form in which it is present.

It is well established that arsenic speciation affects its toxicity. Moreover, EPA's water quality criteria have been established separately for As(III) and As(V). Information from the water quality sampling program would greatly benefit from determining levels of both oxidation states of As. In addition, Cr analyses would improve by examining at speciation for this element, though this review has not identified chromium as a likely problem for the Barataria-Terrebonne area. Similar to the situation for As, Cr is present in the environment in two oxidation states (Cr[III] and Cr[VI]) that differ in toxicity. Chromium is much more toxic in its hexavalent state than in its trivalent state. An optimal sampling program would distinguish between these two forms of Cr.

The EMAP-E program is in principle ideally suited for identifying contamination problems. However, its usefulness for the Barataria-Terrebonne estuarine system is limited to the estuarine section of the basins. No similar program is available for the other areas. Our abilities to monitor contamination would greatly benefit from a long-term program that also incorporated analyses of contaminants into environmental compartments other than water because concentrations in water only point to the possibility of contaminant problems in organisms (including humans). Similar to the EMAP program, the inclusion of pollutant analyses of sediment and aquatic organisms would be very valuable. Bivalves are well suited for such biomonitoring because they are easily collected and remain resident in one particular area. Oysters are well suited for the estuarine areas where they occur naturally. For the more northern parts of the area, the zebra mussel (*Dreissena polymorpha*) would be well suited. This small bivalve, which has recently been introduced into North America, has made its way down the full length of the Mississippi River and will in all likelihood be rather common in most parts of Louisiana very soon. This mussel has recently been included in the Mussel Watch project for the Great Lakes area and is used in biomonitoring programs in western Europe. Data interpretations can thus build on a large data set available for this bivalve.

Considering the potential for human health effects, additional monitoring of contaminants in catfish is warranted. In addition, further research is needed to address the cause and effects of high butyltin levels in oysters in Barataria basin at Middle Bank.

The samples taken from the ambient water quality monitoring stations in the basins and in the Mississippi River are not normally analyzed for toxic compounds with the exception of metals. [Exceptions are Mississippi River water at St. Francisville and Pointe a la Hache for testing of organic compounds on a monthly basis.] An organic chemical toxic screen should be carried out on a selected portion of the samples so that toxic chemicals will be reported if they are present. Failing that, a Total Organic-Halide (TOX) analysis could be carried out for screening and if any are detected then a more detailed speciated analysis would be called for. The TOX test is easy and relatively inexpensive.

Implications for Diversions of Mississippi River Water into the Barataria and Terrebonne Basins

Toxic discharges affect ambient water quality, especially in the Mississippi River; however, they are not an overriding issue of concern to the health of most of the Barataria-Terrebonne estuarine system. Exceptions occur on the periphery of the system where unidentified contamination sources as well as waters from the Mississippi and Atchafalaya rivers may impact water and sediment quality. The areas near the river deltas are also the areas where selected contaminant levels in oysters are elevated above the mean for most stations within the estuary. This review indicates that contaminant levels are especially elevated in areas near the periphery of the study area and that the interiors of the basins are relatively uncontaminated (with the exception of produced water discharges and other possible contaminant sources not covered by the state and federal monitoring programs that form the basis of this review). In the best-of-all-worlds, pollution can be reduced further in the estuary and river. Water quality in the Mississippi River assumes greater importance when diversion of river water is viewed as a management strategy. Toxic discharges to the river are declining but are still high relative to other rivers. In cases where Mississippi River water is planned for diversion into marshes as a marsh restoration method, pollutant levels might increase. A careful consideration of the benefits and ecological implications of indirect deleterious effects should be undertaken.

FISH KILLS

Introduction

An approach to understanding the effects of contaminants or poor water quality is to compile information on fish kills, which are a clear sign of acute stress. The source of the stress, however, may be anthropogenic, natural, or a complex combination of natural and human-induced factors. Assessments based solely on fish kills provide only partial and conservative information on the spatial and temporal dimensions of potential problems. When adequate data exist, a temporal record may be used to evaluate evidence of trends in water quality. Fish kill data are useful because many states compile the information, and there is the potential to identify problems and/or changes in water quality where the data set is adequate.

Fish kills can be related to specific human events such as a chemical spill or chlorine discharge from wastewater treatment plants. By-catch from trawling or loss of a net set during fishing operations may result in large numbers of dead fish. Events may be linked to natural phenomena such as oxygen depletion resulting from sustained periods of hot weather coupled with low flow conditions. Sustained freezing temperatures may cause many fish kills as well as flushing low oxygen waters and/or organic rich materials from swamps during hurricanes. Low oxygen levels also may result from a combination of natural conditions and human-related factors such as algal blooms stimulated by nutrients introduced from nonpoint sources. Nutrient enrichment can be implicated in the production of toxic algal blooms, some of which kill fish.

Data Overview

Fish kill data for the area have been and continue to be collected by several agencies and institutions: LDEQ, LDWF, the Louisiana Department of Agriculture and Forestry (LDAF), NOAA Strategic Environmental Assessment Division, EPA, LSU School of Veterinary Medicine, and Auburn University's Southeast Fisheries Disease Project (table 31). Currently, fish kill data for the Barataria-Terrebonne estuarine system exist in several forms: most are agency annual reports or field and laboratory investigation documents compiled by government and university scientists. Reportedly, fish kill data bases exist at LSU School of Veterinary Medicine; however, the information was not made available. The EPA fish kill reporting program is a continuation of the U.S. Public Health Service program that tracked events from 1960 to 1971. State participation in the EPA program was voluntary and has declined significantly since 1979. The EPA data formed about a

Table 31. Fish kill data sets for Barataria-Terrebonne estuarine system.

Description	Location	Collector	Period of Record	Contact	Format
Large file folder of kill reports	LDEQ	LDEQ	1981–1994	Chris Piehler 504-765-0671	Documents
FISHKILL.SUM	LDEQ/ LUMCON	LDEQ	1981–1994	Chris Piehler 504-765-0671	WordPerfect
B-T_FISH.XLS derived from FISHKILL.SUM	LUMCON	LDEQ	1980–1994	Ben Cole 504-851-2800	Excel
Large file folder of kill reports	LDWF N.O.	LDWF	1981–1986	Richard Bejarano 594-568-5685	Documents
Xerox of field reports	LDWF B.R.	LDWF	1991–	Harry Blanchette 504-765-2800	Documents
Kill investigations (dubious value)	LDWF study areas	LDWF	early 1980s	Harry Blanchette 504-765-2800	Documents
GULFFISH.DBF	LUMCON	NOAA	1980–1989	Jamison Lowe 301-713-3000	Dbase
Fish kill data base in GULFFISH.DBF		EPA		Nina Harlee 202-382-7071	
Cultured fish disease/kill	LSU	Vet School	1990–	John Hawk	Case studies, data base
LDAF, LDEQ, LDWF, EPA	LSU	Vet School	1990–	Jay Means	Case studies, data base
LDAF Investigations	LDAF B.R.	LDAF	1991–	Bobby Simoneaux 504-925-3763	Annual reports
SE Fish Disease Project studies reports	Auburn	Auburn	1970–1990	John Plumb 205-844-9215	Case studies, annual

EPA–Environmental Protection Agency
NOAA– National Oceanic and Atmospheric Administration
LDAF– Louisiana Department of Agriculture and Forestry
LDEQ– Louisiana Department of Environmental Quality
LDWF– Louisiana Department of Wildlife and Fisheries
LSU– Louisiana State University

LUMCON– Louisiana Universities Marine Consortium

third of the information in the NOAA data set (Lowe et al. 1991). The most complete data sets are those of LDEQ and NOAA, which were used in this analysis. The LDEQ data were incorporated into the NOAA data set (Lowe et al. 1991), which summarized data from 1980–1989. The LDEQ data cover fish kill events from 1980 to September 1994. A comparison and compilation of LDEQ and NOAA data sets form the basis for this analysis.

For many reasons, the data presented here are in complete. Agency and institutional funding, missions, and personnel are constantly evolving; thus, temporal consistency in data collection are lacking. Training, equipment, and numbers of investigating personnel have not always been as good as would be desired. During the 1990s, the quality of fish kill event investigations in Louisiana has improved markedly. A data base is only as worthwhile as the quality of data entered; we are cautious in applying too much emphasis to this analysis.

Events that occur near densely populated areas are more likely to be reported than events in remote areas. Many significant natural and anthropogenic fish kills simply are not reported. Shrimp trawling is a common activity, and by-catch, which is generally discarded, is often a high percentage of what is collected in the trawl. LDEQ investigates fish kills that are likely pollution related. If the cause, on investigation, is shown to be a result of trawl discard, it is recorded as such. Kills of this type, however, are not routinely investigated, and this category is probably under-represented in the fish kill data. Finfish and crawfish are farmed, and oysters are cultivated on many privately held agricultural and water-bottom acres. Events that occur on private property are not always reported to state or federal agencies. Farmed fish are often held at much higher population densities than wild fish. High nutrients and organic carbon loading can cause stressful conditions in culture ponds where low oxygen may develop and where toxins and diseases can be transmitted rapidly. No records of farmed finfish or oyster mortalities were in the data available for this report.

Data Sources, Quality, and Processing

Fish kill data reported in this document were acquired from two data sets:

- NOAA's Fish Kills in Coastal Waters, 1980–1989 (Lowe et al. 1991), designated as GULFFISH.DBF
- LDEQ Fish Kill Data (WordPerfect Document), provided by Chris Piehler, designated as FISHKILL.SUM.

Of the 172 Louisiana events in GULFFISH.DBF, 62 occurred in the Barataria-Terrebonne estuary. Of the 699 fish kills in the Louisiana FISHKILL.SUM, 167 occurred in the study area. Although EPA data are included in GULFFISH.DBF, all data for the Barataria-Terrebonne area in GULFFISH.DBF were derived from LDEQ investigation reports (J. A. Lowe pers. comm.).

There are disparities between Event Dates in the two data sets. On some occasions, when the same event is recorded in both data sets, there are differences in values for Total Number of Fish Killed (for example, 5,843 killed–LDEQ, 332 killed–NOAA, 01/05/84 Mississippi River Tenneco plant). Both data sets have errors; GULFFISH.DBF is cleaner, but FISHKILL.SUM includes data from many events not reported in GULFFISH.DBF. Correspondingly, 17 of the kills reported in GULFFISH.DBF were not reported in FISHKILL.SUM. GULFFISH.DBF is a data set with a structure that was well thought out and thus useful for descriptive statistics. FISHKILL.SUM has fewer data fields than GULFFISH.DBF and is a preliminary draft with data missing from many fields (to be formalized as a data set for LDEQ in summer 1995, C. Piehler pers. comm.).

In both data sets, there are problems with quantification of how many fish were killed. In some cases, this is because it is difficult to count decomposed fish. In others, it is a waste of resources to quantify non-water quality–related fish kills. A number, or estimated number, of fish killed was not entered for 50 of the 189 kills reported in FISHKILL.SUM. In 27 events where variations of "N/A" or "none observed" were entered, the count was considered to be zero. In GULFFISH.DBF there were no data in the Total Killed field for 24 of 62 Barataria-Terrebonne events. Kill Number values in FISHKILL.SUM were re-formatted in this report to use for descriptive statistics (table 32).

FISHKILL.SUM (because of its longer record) was used as the primary data set to form the basis for B-T_FISH.XLS used in this report. NOAA's Fish Kill Inventory, Worksheet for Classifying Sources and Causes (J. A. Lowe pers. comm.) was used to code and describe land use, source, event, cause, and specific pollutant data. Code events were expanded to include "non event" (no fish observed) and "other" (mechanical kill, no investigation and bacterial infection). Code cause was expanded to include "fishing activities," "herbicides," "no investigation/event not confirmed," and "mechanical." Data from the species field of FISHKILL.SUM were used coding Land Use as "coastal waters–fresh" or "coastal waters–saline." "agriculture," "fishing," "fish processing," "oil-field–related" and "marine transportation" were added to the descriptions in the source field. The data presented in this report should be considered preliminary draft data that will not be considered complete until reviewed, clarified, and verified by LDEQ.

Results

At least 188 fish kills were investigated and reported in the Barataria-Terrebonne estuarine system during July 1980–September 1994 (the location for 60 events in FISHKILL.SUM could not be identified). Figure 91 summarizes the fish kill events and numbers of fish killed.

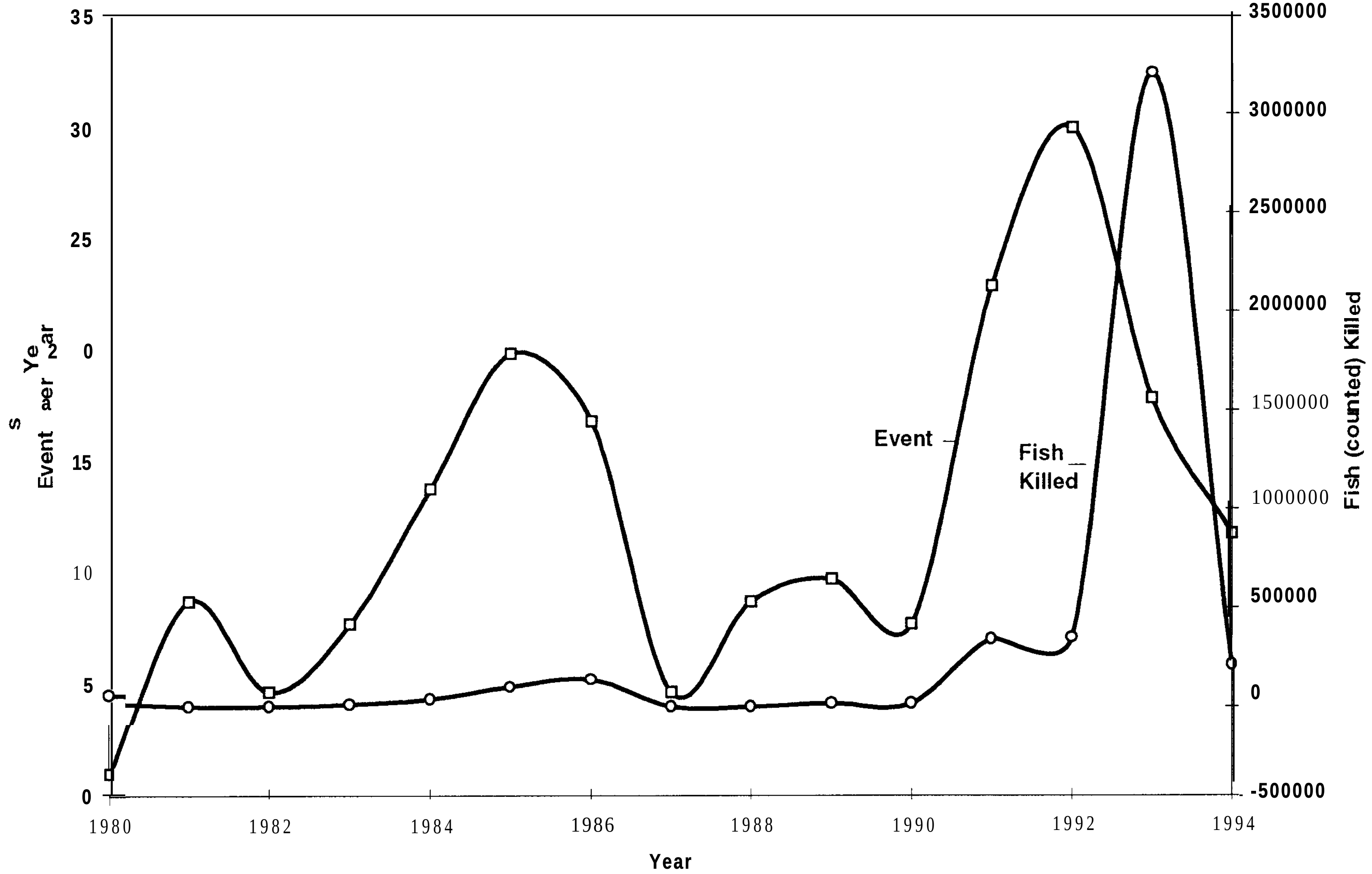


Figure 91. Number of fish kill events and number of killed fish reported for period 1980-l 994 for the Barataria-Terrebonne estuarine system (data from B-T-FISH.XLS).

Table 32. Examples of protocol used to revise entries in the NUMBER of fish killed field (developed by B.Cole for B-T_FISH.XLS).

Category	Number Designation
several hundred	= 300
< 2,000	= 2,000
> 100,000	= 100,000
1,000 - 5,000	= 3,000
10,000 estimated	= 10,000
approx. 1,000	= 1,000
ca 1,000	= 1,000
est. 30,000	= 30,000
160 tons	= 1,547,616
abundant, few, minor, numerous, small number, unspecified, too old to count, etc.	= not given
N/A, none observed, thousands reported, etc.	= 0

Highs and lows in fish killed and events are likely related to reporting effort. Secondly, as mentioned above, the Barataria-Terrebonne fish kill data set is incomplete, and descriptive statistics may be skewed to emphasize events where more information was supplied. For example, in 1985 eight fish kill events resulting from a tropical storm and two hurricanes (Danny and Juan) were reported. In 1992, twelve fish kills were reported to be caused by Hurricane Andrew. The total number of fish killed was not estimated for four of the eight kills associated with 1985 storms. Total number counted for Juan equals 1,003. After Hurricane Andrew a value was estimated for each of the 12 events. The total number for Andrew in the Barataria-Terrebonne estuarine system equaled 262,054. It is probable that significant numbers of events occurred in Hurricane Andrew that were not recorded. As an indication of this, LDWF estimated 182 million fish were killed in the adjacent heavily impacted Atchafalaya basin. It is also probable that more fish died in the Golden Meadow area following Hurricane Juan than the 5,000 reported by LDEQ at Wilkinson Canal and Bayou Folse (B. Cole pers. observ.). Similarly, during January 1988 and December 1989 week-long freezing temperatures resulted in thousands of fish killed. Currently, there are no freeze-related fish kills in the data set.

The single event with the greatest number of fish killed occurred in Empire Canal September 28, 1993; the number of fish estimated in the report was 160 tons. Twenty eight of the 188 fish kills in the Barataria-Terrebonne area were considered major (figure 92). The events all had numbers of fish killed $10,000. Many of the kills were natural events: storms, algal blooms, turnover, "low dissolved oxygen," and arguably, parasites, and bacterial infections. The 10 remaining major kills resulted from lost sets, fish plant discharges, rig moves, pesticides, and herbicides. In some cases pesticide releases into water bodies can be caused by heavy rainfall too soon after application. "Low dissolved oxygen" designations also can result from high BOD loadings from wastewater effluents, killing of water hyacinths in the waterways, and movements of large boats and barges through canals.

One type of "major" event that does not appear to be included in the LDEQ data set is massive kills that result from anoxic bottom waters in nearshore areas encroaching on the barrier islands and lower parts of the system. Seasonally severe oxygen depletion occurs in bottom waters most summers off the Louisiana coast and is particularly well developed off the estuarine system. The distribution of these low dissolved oxygen waters is usually in water depth of 5 m–30 m. Following a wind from the north, however, upwelling favorable conditions are created along the coast. Anoxic bottom waters, often high in hydrogen sulfide, move closer to shore, trap and kill fish. An event in late August 1990 killed an estimated 100,000–150,000 fish near Grand Isle. Similarly, the movement of Hurricane Andrew onto the Louisiana coast in late August 1992 created another massive fish kill along Point au Fer.

Most of the fish kills in the basins occur in the warmest months of the year. Most events (31%) occurred in August, the hottest month, followed by July and September, each with 12% of the events (figure 93). The importance of temperature is even more dramatic when the numbers of fish killed are sorted by month; the majority of fish were killed in August–October (figure 93).

Causes of Fish Kills

Data were sorted by "cause," the reasons the fish died, and are shown in figures 94 and 95. The most commonly cited cause of kills was "low dissolved oxygen." Low dissolved oxygen is more often a symptom than a cause, and the designation appears to have been used as a catch-all term when there was a lack of information. The next five most common causes for events were storms, wastewater, pesticides, unknown, and fishing activities. "Fishing activities" include trawl cull and lost menhaden sets. (If trawl cull is a fish kill event, it has been vastly under-reported.) "Pesticide" kills usually result from agricultural runoff. "Herbicide" kills are caused by the practice of spraying water hyacinths that decay and use up oxygen. Rotenone use by fish farmers and fisheries-monitoring personnel caused "organic" kills. Oil spills and refinery leaks were grouped under "petroleum." Dredging, marine transportation, and movement of drilling barges resulted in events that were characterized as being caused by "sedimentation." "Nutrients" described a kill thought to have been caused by sewage. The "mechanical" kill was caused when fish were trapped by the intake of a power plant. Wastewater from

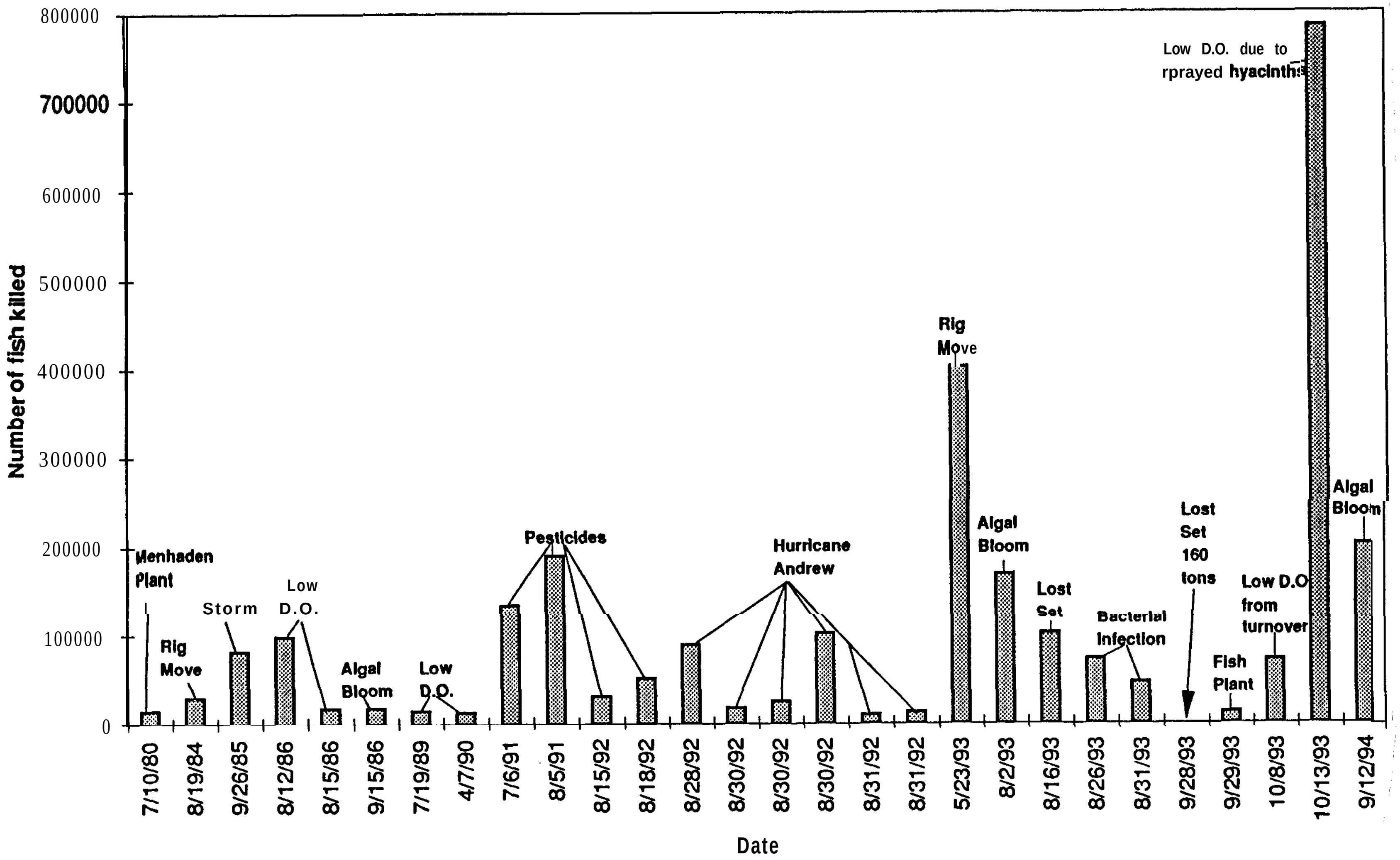
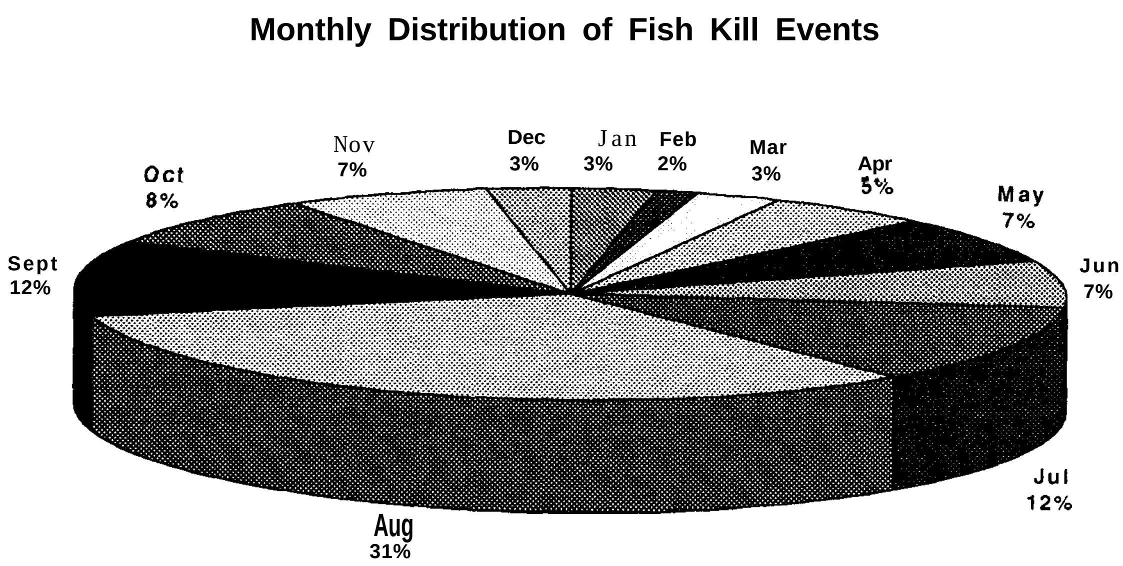


Figure 92. Causes of major fish kill events in the Barataria-Terrebonne estuarine system for period 1980-l 994 (data from B-T-FISH.XLS).



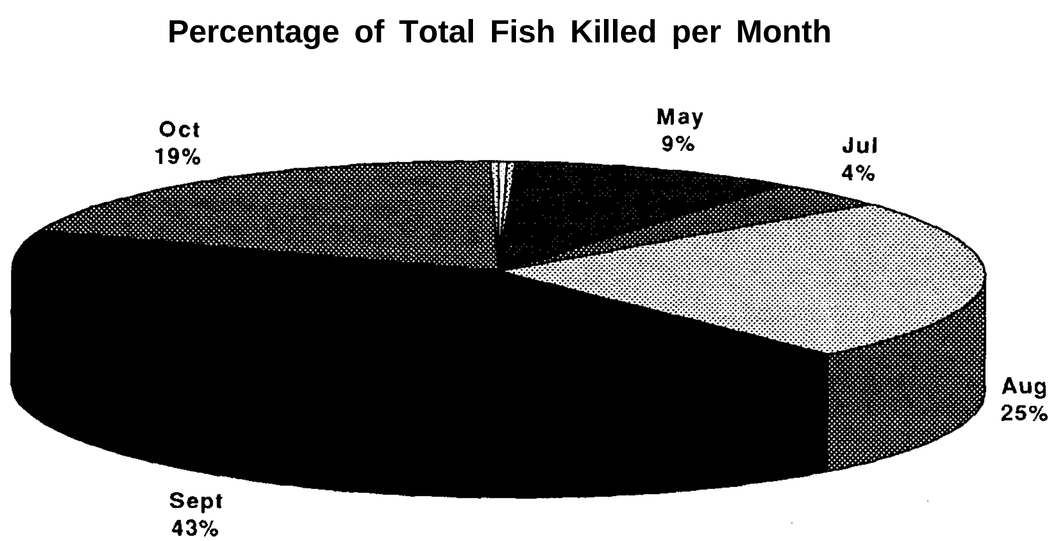


Figure 93. Percentages of fish kill events (upper panel) and number of killed fish reported by month for period 1980-l 994 for the Barataria-Terrebonne estuarine system (data from B-T-FISH.XLS).

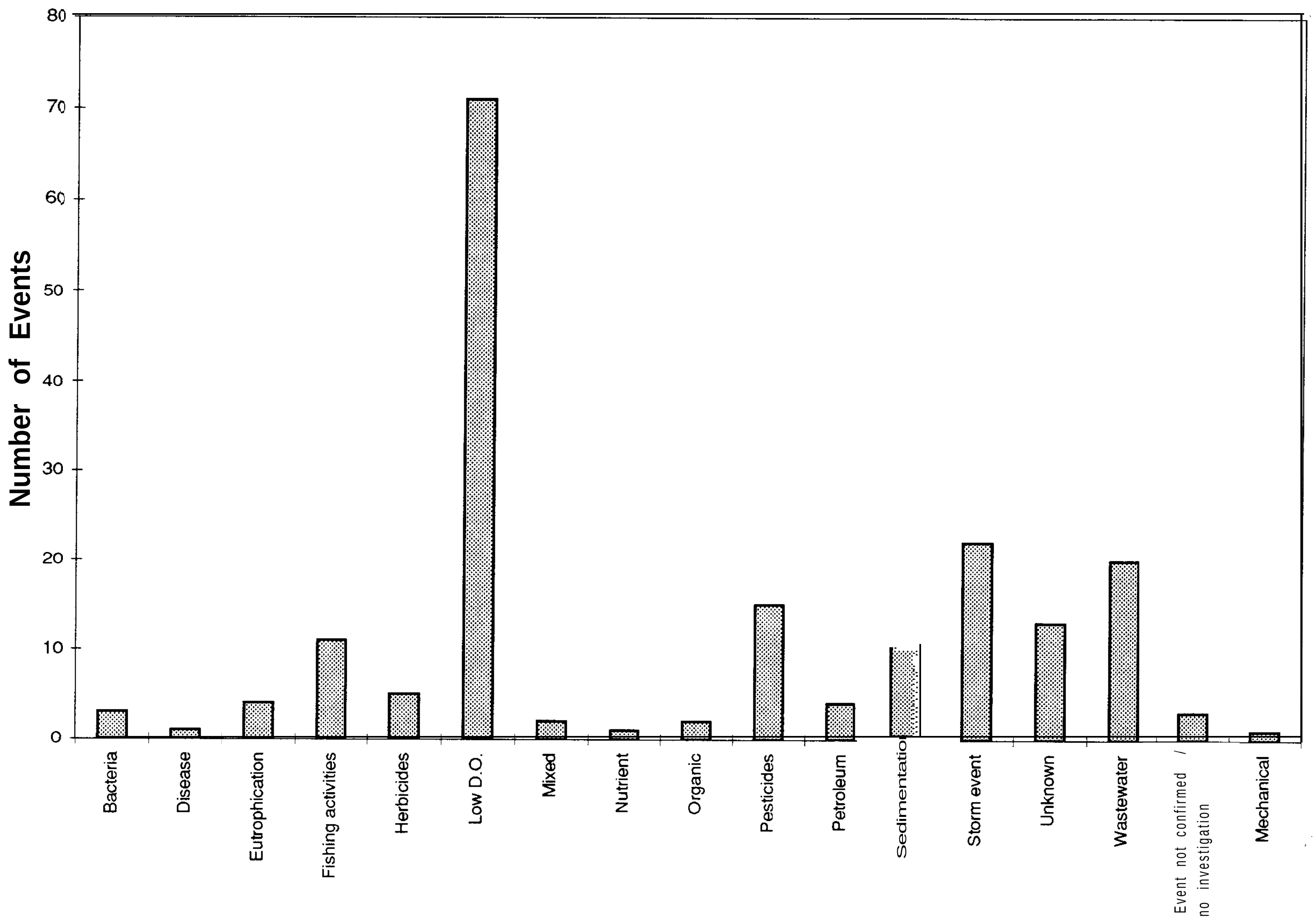


Figure 94. Distribution of causes of fish kill events in the Barataria-Terrebonne estuarine system, by number of fish kill events for period 1980–1994 (data from B-T_FISH.XLS).

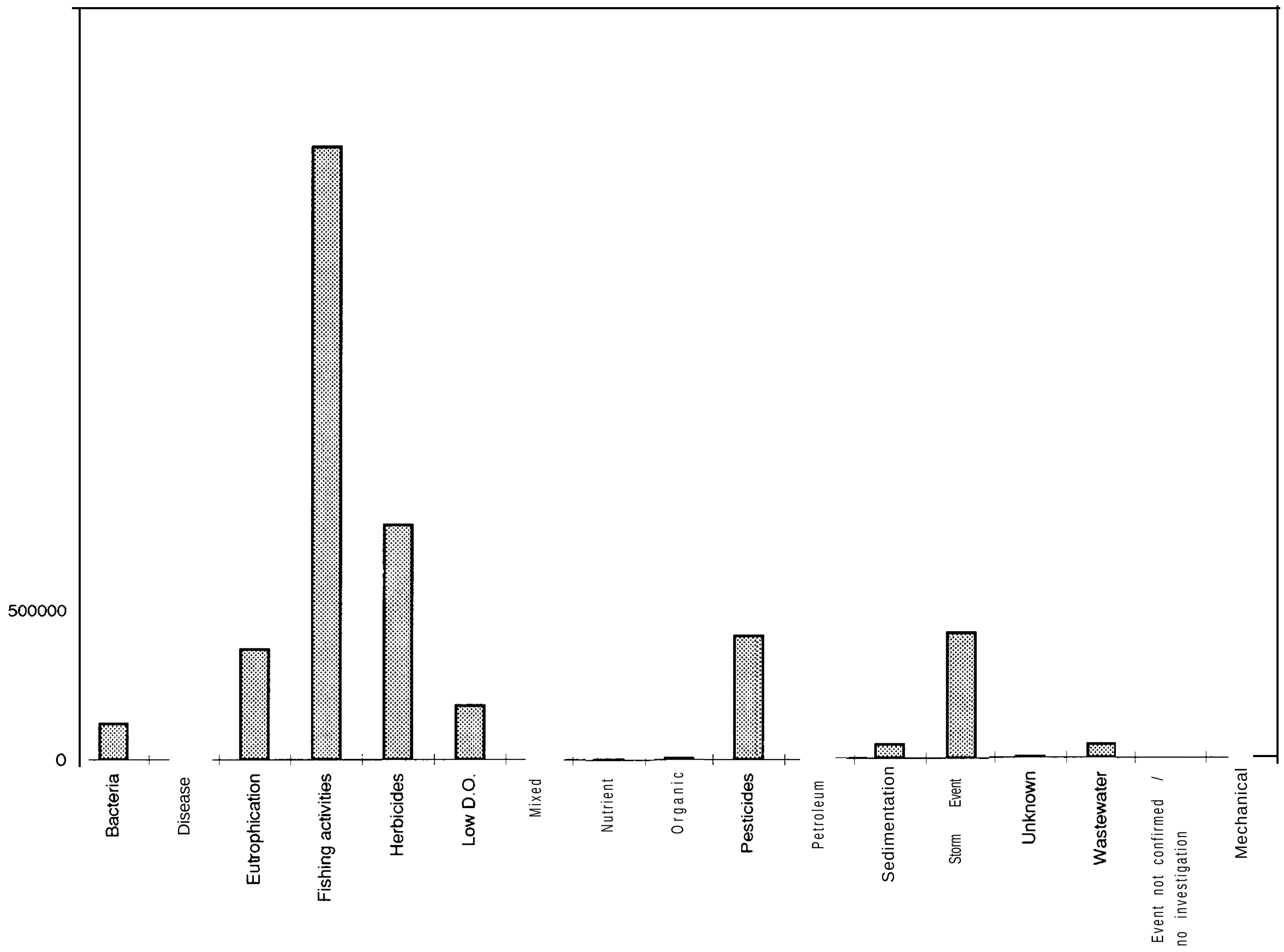


Figure 95. Distribution of causes of fish kill events in the Barataria-Terrebonne estuarine system by number of killed fish reported for period 1980–1994 (data from B-T_FISH.XLS).

fish processing plants, sugar refineries, and sewage treatment facilities were grouped under "wastewater."

Fishing activities, herbicides, storm events, pesticides, and algal blooms, in descending order, were the causes of the greatest numbers of fish killed. A single lost menhaden purse seine set was calculated to have caused an event where 1.5 million fish were lost. Two other lost sets were estimated to have involved 100,000 and 400,000 fish. Algal blooms caused two kills of 167,000 and 200,000 fish. A mixed-cause event (with a primary cause of azinphos-methyl and a secondary cause of low dissolved oxygen from sugar refinery wastewater BOD) resulted in 189,000 fish killed. The largest pesticide kill was 133,837 fish followed by kills of 50,000 and 30,000. Storms, especially hurricanes, were likely under-reported.

Events

Fish kills were coded as "event" types, used to describe what action occurred that introduced the pollutant into the water body, and are summarized in table 33. Events coded as fishing accounted for the greatest number of fish killed (over 2 million, 47% of the total). Natural and spraying events comprised 21% and 18%, respectively. Runoff and runoff/routine release codes combined described 9% of mortalities. Interestingly, the number of fish killed from the 22 wastewater discharges was 41,000, or only 1% of the total counted in the data set.

Sources

Coding data by sources identified the specific water body, site, farm, plant, refinery, etc. from which the pollutant was introduced. The list of source possibilities from the NOAA worksheet for classifying sources was expanded for the purposes of this report to include the following categories: agriculture, fishing, fish-processing, oil-field–related, marine transportation, and herbicide.

Source data from the total 188 events were divided into two subsets: water body and anthropogenic. Waterbody sources were the default code and used in cases of natural kills or when not enough information about a point source was available. One-hundred-eighteen events and 1,090,346 mortalities were coded as waterbody sources. It was possible to ascribe sources related to human activities to 70 events (table 34). When numbers of fish killed were evaluated, it was determined that 75% (3,344,797 fish) of the total mortalities were anthropogenic. Thirteen fish kill events resulted from agricultural runoff. Fishing was named as the source of 12 events, of which three were lost menhaden sets with a combined total of more than 2 million fish killed.

Table 33. Description of fish kill events, occurrence, and number of fish killed per event type (data from B-T_FISH.XLS).

Event	Description	Number of Events	Number of Fish Killed
Unspecified	Insufficient or non-sufficient data	48	150,797
Natural	Algal blooms, stratification, turnover, storms	42	931,658
Wastewater discharge	Planned releases from discrete containment areas, e.g., holding ponds, or industrial and and sewage treatment	22	41,188
Runoff	Pollutants (mostly pesticides) carried into water bodies by rainfall	14	224,728
Unknown	Insufficient or non-sufficient data	12	7,215
Dredging or drilling	Waterway dredging, drilling barge moves (5), brine discharge, propeller dredging in shallow water	11	45,200
Fishing	Lost menhaden sets (4), trawl bycatch	11	2,048,525
Non event	Investigation made, but no fish observed	8	0
Spraying	Liquid pollutant applied to a waterbody, e.g., rodeo, rotenone, herbicides used on water hyacinths	6	786,213
Other	Power plant intake, sites not investigated	5	151
Spill	Pollutant released directly into waterbody, e.g., oil spill, oil refinery leak, rotenone used in a pond then discharged into a water body	4	10,343
Drawdown	Canals pumped down and anoxic water introduced into another water body	2	1,025
Accidental release	Aromatic dispersant at a drilling operation	1	100
Runoff, wastewater discharge	Caused by pesticides primarily and sugar refinery discharge secondarily	1	188,000

Table 34. Anthropogenic sources of fish kills (data in B-T_FISH.XLS)

Anthropogenic Source	Events per Source	Total Killed per Source
Fishing	12	2,048,525
Herbicide	5	785,913
Agriculture	13	224,728
Agriculture/sugar refinery	1	188,000
Oil-field related	8	38,300
Fish processing	8	35,029
Marine transportation	6	8,000
Oil refinery	1	5,843
Farm pond	2	3,800
Sewage treatment plant	5	3,550
Sugar refinery	7	2,609
Fish pond	1	500
Power plant	1	0

Specific Pollutants

If the data were provided, the category "specific pollutant" names the agent that caused the fish to die. Of the 188 fish kill records, 119 had information on the specific pollutant (summarized in table 35). Specific pollutant data can be useful when quantified patterns can be seen such as recurring events. Although little can be done about natural "pollutants," details related to repeated kills caused by specific agents can be useful in formulating water quality management and enforcement policies.

Land Use

Data were coded for land use to describe the land-use type from which the pollutant originated. The following categories were used:

- agricultural–land used for the production of food, including sugar cane and fish and crawfish ponds;
- industrial–land used for industries and refineries;
- urban–land uses such as sewage treatment plants, power plants, and drainage canals;
- impoundments–land used as a reservoir, lake, canal, or ditch;
- coastal waters [fresh]–fresh running water bodies including bayous and rivers;
- coastal waters [saline]–salt, or brackish waters including estuaries and nearshore waters; and
- unknown.

Whether coastal waters were coded fresh or saline was determined by data in the "species" field.

Coastal waters, impoundments, or unknown accounted for 142 events, or 89% of the counted fish killed. Most of events in the data set with natural and anthropogenic causes were coded as land uses but were more properly water bodies. (Important recurring kills, such those resulting from dredging, fishing activities, herbicide application, marine transportation, oil spills, etc., and natural kills, such as storms, algal blooms, bacterial infections, etc., are not easy to distinguish under these broad land-use types). With the remaining codes (agricultural, industrial, mining, and urban), the usefulness of land-use designation can be seen. There were 18 agricultural kills involving rotenone or pesticides. Land uses of menhaden plants, shrimp and seafood processing plants, and sugar and oil refineries were the designations responsible for 16 fish kill events. There were nine urban-related and three mining-related fish kills. All of these kills were associated with specific sites that could be designated "hot spots." Because the land-use designation agricultural-industrial refers to a single event that occurred at the juncture of Himalaya Canal and Bayou Louis in August of 1991 (described as having been caused by "pesticides primarily, low dissolved oxygen due to sugar mill discharge secondarily"), it

Table 35. Fish kill events identified by specific pollutants (data from B-T_FISH.XLS).

Specific Pollutant	Occurrences
Bacterial infection	3
Parasites	1
Algal bloom/eutrophication	8
Fishery discards	7
Lost menhaden sets	4
Herbicides	5
High BOD input	3
Canal drawdowns	1
Low dissolved oxygen	2
Stratification	1
Trapped on power plant intake filter	1
Aromatic Dispersant OSD-700	1
Soap spill/fishery discards?	1
Rotenone	2
Pesticides	15
Crude oil and chlorides	1
Leak from refinery	1
Oil spill	1
Oil brine discharge	1
Dredging	2
Movement of drilling rig	5
Marine transportation	3
Storm event	23
Sewage	3
Menhaden plant discharge	2
Sugar refinery plant discharge	7
Sewage treatment plant discharge	5
Seafood processing plant discharge	6

would be reasonable to conclude that 90% of the fish killed in connection with "hot spots" were agriculture related.

Conclusions and Recommendations

Compiling fish kill data is difficult for many reasons, including inconsistent reporting, shared agency responsibility, higher numbers of kills reported near higher population areas, varied formats, gaps in data due to program lapses, inconsistent content, data format (i.e. not in computer data bases), agency priorities, changes in investigating personnel, etc. Fortunately, the quality of fish kill investigations and documentation has improved with time in Louisiana, particularly since 1991. Caution should be used when interpreting data based on fish kill investigation reports.

Even though the data available do not present a complete inventory of fish kill incidents in the Barataria-Terrebonne estuarine system, the data can be used to quantify certain aspects of kills, such as seasonality and primary causes and to identify problem areas or pollutants. Although many fish kills are the result of natural phenomena, kills with anthropogenic sources are potentially preventable. Identifying and documenting these problems is a first step in preventing and minimizing future fish kills.

Conclusions

- Fish kill data can be used as indicators of environmental stress.
- Most kills occur in the warmest months of the year.
- Naturally occurring events dominate Barataria-Terrebonne estuarine system basin fish kill events.
- Mortalities attributable to storm events comprise a significant 10.5% of the total number of reported fish killed.
- "Hot spots," or problem areas, and recurring kills include oil-field–related activities, marine transportation, dredging, pesticides from agricultural runoff, herbicide spraying of water hyacinths, accidental release of chemicals used in fish farming, planned and accidental releases of untreated water from fish-processing plants, paper mills and sugar cane factories, and sewage treatment facilities.
- LDEQ data indicate that kills related to pesticide use have increased. (A measure of this is definitely attributable to the fact that fish kill investigations, analyses, and documentation have become much more sophisticated in recent years).
- Interestingly, the number of fish killed from the 22 routine release events was 41,000 fish, only 1% of the total fish counted killed in the data set.
- Sugar refinery kills occur in the fall months of October–December when sugar cane is usually processed, but two kills were noted in May.
- Seafood plant discharge related kills occurred between April and September.

- Fourteen of 15 pesticide kills occurred in July and August.
- Hyacinth herbicide kills spanned the months of June–October with three of six occurring in August.

Recommendations

- Update, verify, clarify, and computerize the LDEQ fish kill data set.
- Expand the LDEQ data set to include as much information as possible about the kill event and to make it as comparable as possible to the NOAA Fish Kill Inventory.
- Continue current efforts to investigate fish kills.
- Identify water bodies with recurring fish kill events
- Identify sources and anthropogenically caused fish kills.
- Determine the significance of fish kills in the Barataria-Terrebonne estuarine system to overall fish population levels.
- Reduce toxics, chemical oxygen demand, or biological oxygen demand in discharges that are consistently implicated in fish kills.
- Apply better agricultural management practices to reduce the number of fish kill events associated with pesticide runoff.
- Apply mitigative measures to identifiable causes of fish kills, e.g., specific effluents.
- During investigations of fish kills where the cause is not immediately apparent, low oxygen is a suspected cause, or discolored water is present, samples for identification of phytoplankton should be taken.

FECAL COLIFORM INDICATORS, PATHOGENS, PUBLIC HEALTH, AND SHELLFISH BED CLOSURES

Introduction

The rich brackish waters of the Barataria-Terrebonne estuarine system buffer the Gulf of Mexico from the coastal freshwater areas and are among the most productive waters in the world. They have provided a rich environment for all inhabitants and users of their resources, including the earliest human coastal dwellers. Molluscan shellfish and other renewable fishery resources have generally been abundant in these areas, have always provided an important source of food, and have remained vitally important to the traditions and socioeconomic welfare of the coastal communities in this system.

The loss of the valuable heritage and renewable resources of the Louisiana wetlands has long been the subject of tremendous concern. Two factors of concern in addressing microbiological water quality in the estuaries and decreased resources or resource use are (1) the intrusion of high-salinity waters from the Gulf and (2) possible increased point and nonpoint source sewage or fecal pollution runoff from the coast (Kilgen 1991).

These two factors also can be contributing sources of the two main groups of water-borne pathogens and fecal coliform indicator bacteria in the estuarine system: (1) the naturally occurring marine bacteria belonging to the family *Vibrionaceae* (*Vibrio, Aeromonas,* and *Plesiomonas* species) and (2) human fecal pollution that may contain enteric bacteria (*Salmonella, Shigella,* and *Campylobacter* species) and human enteric viruses (Norwalk or related agents, hepatitis A, and enteral hepatitis E).

Illnesses from these microorganisms may result from wound infections from primary or secondary recreational contact with marine waters (*Vibrionaceae* only) or from consumption of contaminated estuarine waters, raw seafood contaminated with pathogens from growing waters, or cooked seafood re-contaminated with raw seafoods (*Vibrionaceae* and fecal bacteria or viruses) (NAS 1991).

The Barataria and Terrebonne basins have approximately 240,000 acres of private oyster leases (68% of total state private leases). Traditionally, this area has been an important oyster-producing area, averaging about 8 million pounds of oyster meat per year until 1990. There has been an increase in leased acreage over this time period but with no increase in production. Production from the state's public grounds also has been declining (Perret and Melancon 1991). Among many other economic and environmental factors, this has been due in large part to lack of sales from loss of public confidence in oyster products from the Gulf coast since 1988 (Kilgen 1993).

Louisiana has traditionally been the number one producer state of shell-stock oysters in the United States, averaging approximately 12 million pounds of meats per year from 1980 to 1989. In a state economically depressed by the decline in production of non-renewable oil and gas resources, the traditional renewable resources and heritage of the fishing industry become more important.

In the last few years, however, extensive press campaigns questioning the safety of domestic seafood—especially raw molluscan shellfish—have created national public concern. As a result, large retail buyers eliminated oysters entirely in 1988, and three states, including Louisiana, require warning labels on raw oyster products for high-risk consumers. Demand and resultant production of oysters dropped 41% nationally from the high in 1982 to 1991. Oyster production in Louisiana dropped 45% from 1988 to 1991 (13.3 million pounds of meat to 7.3 million pounds of meat) and concurrently dropped 32% in dockside price (Kilgen 1993). Between 1982 and 1991, per capita use of domestic eastern oysters was down by 60%. This meant losing former oyster consumers (Roberts 1992).

Oyster processors and wholesalers reported that as much as 50% of their wholesale business had been lost in 1990, with sales down two-thirds and prices approximately one-half of what they were two years previous. This was attributed to loss in consumer confidence in oyster products with the main contributing factor the negative press concerning the safety of oysters (Kilgen 1993). The extensive negative press and resultant public fear of molluscan shellfish were related to the natural marine bacteria, *Vibrio vulnificus*. This species is present in all marine waters, with extremely high numbers in warm temperatures, but is not associated with serious illness in normal healthy individuals. In rare cases, however, *V. vulnificus* and some of the other *Vibrio* species may cause primary septicemia in 1:10,000 high-risk immunocompromised individuals with diseases of the liver, blood, stomach, or immune system. Once a *Vibrio* septicemia from consumption of raw seafood or contact of wounds with marine waters is contracted by a high-risk patient, the expected mortality rate is 50%. This alarming fact and the extreme nature of the clinical pathology of this disease have been sensationalized by the press and other groups, and have caused the consuming public to incorrectly assume that anyone may be at considerable risk from this high-mortality *Vibrio* septicemia through consumption of molluscan shellfish.

To compound this problem, the U.S. Food and Drug Administration (USFDA) proposed a possible seven-month ban April–October on sale of Gulf Coast oysters for raw consumption because of the potential risk to those few individuals with the above underlying illnesses. Because the state of Louisiana has traditionally been the main producer of raw shell-stock oysters not only on the Gulf Coast but also in the entire country, this would essentially destroy the Louisiana oyster industry.

Although the salt water from the Gulf and runoff from the coast can contribute to the potential fecal coliform indicator and pathogen load in the estuaries, only fecal coliform indicators are considered a "water quality" issue. However, the increase of salinity with saltwater intrusion can increase the levels of naturally occurring marine bacteria in the family *Vibrionaceae*—some of which have the ability to cause human disease under certain circumstances (CDC 1989; Colwell 1984; Rodrick 1990; NAS 1991; Kilgen 1991, 1995; Rippey 1994). Increases in salinity also bring in oyster predators that increase oyster mortality and push productive areas toward the coast.

At the same time, increases in coastal populations throughout the country generally result in increases in human sewage runoff into the estuaries. The 1990 National Shellfish Register of Classified Estuarine Waters documented that increased coastal development between 1985 and 1990 resulted in a 15% increase in closure of shellfish growing waters and an overall trend of decreased open areas throughout the country. This was attributed to increases in fecal coliform indicator levels above the allowable growing-water standard for shellfish harvesting. The main sources of increased fecal coliform contamination were identified as urban runoff and malfunctioning or non-existent septic systems in coastal dwellings (USDC 1991).

The history of the program for the classification of oyster growing waters in the Barataria-Terrebonne estuarine system and in the entire state dates from the early 1960s. The Louisiana Department of Health and Hospitals (LDHH) historically took about 700 water samples across the entire coast and analyzed them for most probable number fecal coliform (MPN FC)/100 ml. LDWF decided on closure area boundaries and enforced the closure laws. This system was based on areas that had chronic conditions of high fecal coliform levels and were nearly always closed. Some areas were closed during periods of abnormally high FC counts following events such as hurricanes. Most of Terrebonne Parish oyster areas were open, and Sister Lake, a state seed ground, was producing about 2,000 sacs of oysters per day in November 1982 when 500 cases of gastroenteritis implicating Sister Lake oysters forced the immediate closure of about 500,000 acres of oyster fishing grounds, including nearly all of the Terrebonne Parish oyster-growing areas (Kilgen and Kilgen 1990).

This event prompted the development of the current "seasonal" closure lines for oyster-growing waters. Before, little statistical methodology was employed. Areas were historically "good" or "bad" based on water samples (Kilgen and Kilgen 1990, Hemphill 1994).

Data Sources

Several data bases for pathogen assessment in the estuary are potentially useful; others are not useful because they do not have historical continuity. Many studies provide "snapshot" pictures of the microbiological condition of areas in the estuarine system for a one- or two-year study. Some of them are listed in the Barataria-Terrebonne Basin Data Inventory (Barataria-Terrebonne National Estuary Program 1991b). Some of the special studies took monthly samples at the same sites. Some sites were sampled quarterly, and some were sampled once to obtain a representation of a larger area.

The oldest and most temporally complete data bases are from LDHH and LDEQ. The most potentially useful data bases and other data sources for this study were the following:

Historical Data Bases

(1) Oyster Water Monitoring Program, Louisiana Department of Health and Hospitals. 1980–1994 data base (K. Hemphill and S. Shah pers. comm.).

Monthly water samples for MPN fecal coliform, salinity and temperature are taken. There are 117 samples sites in basin 02 (Barataria) and 201 in basin 12 (Terrebonne).

The data selected for this study were a 15-year continuous data base 1980–1994 at four sites within each basin. These sites, selected through input by members of the Water Quality Team, LDHH Oyster Water Monitoring Program and LDEQ Water Monitoring Program, include:

Terrebonne basin–LDHH sites 1 (Catfish Lake), 83 (Bay Cocodrie), 134 (Sister Lake), and 168 (Hell Hole Lake).

Barataria basin–LDHH sites 10 (Bay Jacques), 32 (Bay Adams), 53 (Bay San Bois), and 70 (St. Mary's Point)

(2) LDHH, Office of Public Health, Department of Epidemiology, Annual Reports, 1984–1993.

(3) LDHH, Office of Public Health, Department of Epidemiology. Reported incidence of water-borne diseases in the Barataria and Terrebonne estuaries (Dr. L. McFarland and S. Wilson pers. comm.).

(4) LDEQ, Water Quality Monitoring Program data. Monthly data since 1988 of subsegment water bodies.

Other Data Sources

The other data sources and references are listed in the References.

Discussion of Issues

Fecal Coliform Indicators, Shellfish Bed Closures, and Public Health Risk

Scientists who work in the applied field of environmental microbiology have faced the ongoing problem of determining the best scientific methods to quantify levels of pollution of shellfish-growing estuaries with pathogens from human wastes and assessing the actual risk to the health of the general public from this type of pollution. In addition, they are now trying to evaluate the ecology of the naturally occurring *Vibrio* organisms in the marine environment to understand why they can be present in extremely high numbers (10^3 - 10^6 MPN/g) in shellfish in warm marine waters, and yet be potentially deadly in only a few rare cases.

These tasks have been extremely difficult for many reasons, including (1) the lack of a specific indicator of human enteric viruses that actually cause raw oyster-associated gastroenteritis; (2) if there were a very specific microbiological indicator, the ecology of coastal waters does not lend itself to consistent predictability or modeling of spatial and temporal levels of microorganisms. Microorganisms are never homogeneously distributed in the marine environment, and their presence and persistence are determined by the interaction of many environmental and physicochemical parameters; (3) the lack of simple, rapid, economical methods for rapid pathogen identification in shellfish and growing waters; (4) the lack of consistency between levels of fecal indicators and pathogens in overlying growing waters and shellfish themselves. This is a problem because government policy requires testing of growing waters for classification instead of the product itself; (5) the lack of understanding of the ecology, distribution, serology, and molecular biology of *Vibrio* organisms in the marine environment and their interactions with shellfish and certain few immunocompromised individuals (Kilgen 1995).

The current indicator bacterial standard in estuarine waters was established to prevent human sewage contamination in shellfish, which may be consumed raw, and thus intended to protect the public health. Shellfish are filter feeders and can concentrate bacteria and viruses found in their growing waters (USFDA 1993a, b). However, as noted previously, water-borne pathogens may originate from two main sources: the naturally occurring marine bacteria of the family *Vibrionaceae* and fecal pollution that may contain enteric bacteria and human enteric viruses.

The use of bacterial indicators to predict human fecal pollution in estuarine waters was the premise of the first shellfish management program developed by the U.S. Public Health Service in 1925, and also for the current program (USFDA 1993a, b). It is also the premise for EPA recreational contact water quality standards (Cabelli et al. 1983, EPA 1985).

Coliforms and fecal coliforms are a very heterogeneous population of bacteria present in high numbers in raw sewage and have been considered "indicators" of the possible bacterial and viral human enteric pathogens that also may be found in feces. The allowable numbers of indicator bacteria in shellfish growing waters was established in the earlier decades of the 20th century and was based on the relationships between total coliform bacterial indicators to *Salmonella* bacterial pathogens in human sewage–polluted growing waters and shellfish during typhoid fever epidemics. Later studies concluded that fecal coliform bacteria were more accurate indicators of fecal contamination than total coliforms, and the allowable numbers were extrapolated to the current standard of 14 most probable number fecal coliforms (MPN FC)/100 ml of growing waters with not over 10% of samples exceeding 43 MPN FC/100 ml (Hunt and Springer 1978, USFDA 1993a, b). Unfortunately, this standard was extrapolated from very old and very limited epidemiological data from the early part of the century. Because of the lack of current epidemiological data regarding true human health risk, this allowable level of 14 MPN/100 ml fecal coliforms in growing waters was arbitrarily extrapolated. This has resulted in a lack of confidence in the fecal coliform indicator by the scientific, regulatory, and business communities as a true indicator of human fecal pollution and public health risk from water or shellfish (Elliot and Colwell 1985, Kilgen 1989, NAS 1991).

The basic requirements for an ideal indicator have been discussed at length (Elliot and Colwell 1985, Kator and Rhodes 1994). Several indicator organisms have been proposed in the past for detection of fecal pollution in fresh, brackish, estuarine, and sea waters. However, no single indicator organism exists for determination of public health risk in waters or seafood (Elliot and Colwell 1985, Kator and Rhodes 1994). The current fecal coliform indicator standard in growing waters and guideline for oyster meats have been seriously questioned in the last decade.

Some of the more serious problems with this indicator of fecal contamination include:

(1) Non-*E. coli* fecal coliforms and even non-sewage–related bacteria may predominate in the fecal coliform population analyzed by APHA approved methods (Kilgen et al. 1988, Paille et al. 1987).

These non-*E. coli* fecal coliforms can be found in shellfish, sediment, and the water column, especially in warm temperatures. In Louisiana oysters during the warm months, *Klebsiella pneumoniae* isolates accounted for 86% of the non-*E. coli* fecal coliforms and often outnumbered *E. coli* 1,000:1 (Kilgen et al. 1988, Paille et al. 1987). Similar studies by USFDA concurred that Gulf Coast oysters harvested from approved growing waters in summer months may contain excessively high levels of non-*E. coli* fecal coliforms and not represent a health hazard.

(2) The fecal coliform indicator for waters does not indicate the presence of non-sewage–related naturally occurring aquatic bacterial pathogens (Elliot and Colwell 1985, Rodrick 1990).

Members of the family *Vibrionaceae* are natural aquatic organisms generally not associated with fecal contamination. The National Advisory Committee for Microbiological Criteria for Foods recommended that the current Aerobic Plate Count method for shellfish should include a modified aerobic plate count procedure that encourages the growth of

Vibrio spp. (NACMCF 1992). USFDA has recently proposed a seven-month ban on the raw bar sale of oysters from the Gulf of Mexico to prevent high-risk individuals from eating them and possibly contracting a primary septicemia from *V. vulnificus*. This is a clear example of setting government policy because of public perception and pressure from the press and other groups without sound science. Because the target group of high-risk individuals is clearly defined, a comprehensive education program is more logically and economically sound. The proposed policy would have a devastating economic impact on the Gulf Coast oyster industry in general, and particularly in the Barataria and Terrebonne estuaries.

(3) The fecal coliform indicator does not correlate with the presence of human enteric viruses, which are the pathogens most commonly associated with sewage contamination of waters and seafood (Elliot and Colwell 1985, Kilgen et al. 1988, Kilgen 1989, NAS 1991).

A constant and predictable relationship does not exist between fecal coliform indicators and enteric viruses in waters or shellfish. For these reasons, a National Indicator Study (NIS) was initiated in the summer 1987 to determine the best regulatory indicator(s) of sewage-associated disease risk from the consumption of raw shellfish (Kilgen 1989). An excellent review book on environmental indicators and shellfish safety was developed in this study (Hackney and Pierson 1994).

The most promising enteric virus indicator has been male specific or F+ RNA bacteriophage. The F+ RNA phages have similar survival and persistence in the environment and in sewage treatment as the human enteric viruses of concern in shellfish (Norwalk and HAV). Members of the RNA F+ or male specific coliphage have four serotypes: I, II, III, and IV (Furuse 1987). Furuse (1987) in Japan, Havelaar et al. (1990) and Havelaar (1991, 1993) in the Netherlands, and later Sobsey et al. (1994) in New Jersey, studied the prevalence of each serotype in human and animal feces and raw human sewage collected from treatment plants, and from nonpoint drainage sources of human and animal sewage. It has been generally demonstrated by these studies that serotypes I and IV were most often associated with animal sources of *E. coli* sewage pollution, and serotypes II and III were most often associated with human sources of *E. coli* sewage pollution. F+ RNA phage were generally more numerous in animal feces than in human. It was concluded that F+ RNA phage are rare in human feces.

In a study of the distribution of F+ RNA coliphages in raw sewage from treatment plants in various countries throughout the world, group III was most prevalent, followed by group II and then group I. Group IV had only two isolates out of over 1,800 tested. Group I serotype also was found in human and animal waste water in the Netherlands by Havelaar et al. (1993). They found serogroup II in waste water of human origin but not in human feces. Because of these results, Havelaar et al. concluded that F+ phage were an excellent indicator of human or animal sewage or wastewater pollution sources of waters rather than fecal pollution of waters**.** Sobsey (1994) showed in a study of various sources of New Jersey waste water that the expected serotypes of groups II and III were generally more associated with human sewage or wastewater sources and that groups I and IV were more associated with animal sewage sources. However, one human sewage point source did have a large number of group I isolates.

Kilgen et al. (1994) conducted a 12-month fecal coliform monitoring, identification and assessment study of selected pump stations within the Barataria and Terrebonne estuaries. Study sites were selected to assess possible contamination to the lower oyster growing waters of the estuaries by pumping stormwater nonpoint source runoff into the lower estuaries. It was determined that the most important sampling sites would be pumping stations in the upper estuarine areas above the LDHH oyster water sampling sites.

Twenty-three sites were identified for fecal coliform and physicochemical monitoring and assessment and 14 sites for F+ RNA phage identification and assessment. The sites were representative of human and animal sewered and non-sewered areas, and mixed human and animal areas. However, none of the sites were point sources of pollution. All were nonpoint human sewered (municipal plant or septics) or unsewered, and animal sources. These sources were mainly the intake and discharge sides of strategically located pumping stations that drain large rural areas of the lower inhabited sections of the Barataria and Terrebonne estuaries. The city of Houma was the only municipal area where pumping station sites were selected. Two freshwater diversions on the Mississippi River at Naomi and West Pointe a la Hache, the Houma Navigation Canal at the Intracoastal Waterway, Bayou Famille at Crown Point, and Hell Hole Canal near the lower end of Bay Junop were non-pump station sites.

Overall, the microbiological fecal coliform and physicochemical results from the 23 sites did not show any statistical correlations. Of all the sites that were evaluated, the Lafitte pump station that discharged into The Pen showed the greatest wastewater impact from the surrounding non-sewered human dwellings. The coliform levels were the highest and the dissolved oxygen concentrations the lowest (many microbiologically anaerobic). It was also the only site where the drainage area to the pump station had higher salinity on the intake drainage side than at the discharge side into The Pen.

Of the F+ RNA phage types that were identified from all 14 sites during the 12-month sampling period, 76.5% were type I, 22% were type II, and only 1.24% were type III. No type IV RNA F+ phage were isolated. The only phage site that had an overall greater percentage (60%) of type II (human source) RNA F+ phage than type I (40%) (human and/or animal) was the Bayou Dulac pump station on Bayou Grand Caillou at Dulac. The source at the intake side of this pump station is a mixture of human septics, human unsewered dwellings, and pasture (Kilgen et al. 1994).

Another site showing high percentages of type II RNA F+ phage was the pump station at Industrial Blvd. in Houma (33.3% type II, 11.1% type III, and 55.6% type I). This site is considered a municipal human sewered site. However, all pump stations drain storm water runoff from the surrounding areas. This does not preclude animal wastewater runoff to the area. The intakes of the pump stations at Woodlawn Ranch Road, at Golden Meadow, at Bayou L'Ours, and at The Pen also showed high percentages of type II (human sewage associated) RNA F+ phage. These sites all represent storm water runoff from municipal (Woodlawn) and human septics or human unsewered areas (Golden Meadow, The Pen). Bayou L'Ours was originally selected as a pasture runoff site. The runoff into the intake side of the pump station there was mainly from pastures in the area. There were, however, some houses at the site.

The only other sites selected as animal sites were the discharge side of the pump from Citrus Lands at Wilkinson Canal in Barataria and Hell Hole Canal at the lower end of Terrebonne. These areas have traditionally had very high levels of fecal coliform indicators. The high levels in Wilkinson have been attributed to the pasture runoff. The high levels at Hell Hole Canal cannot be explained. At the Wilkinson site, only one RNA F+ phage was isolated in the entire 12-month sampling period. None were isolated from Hell Hole. This could be an indication that although the fecal coliform indicators are high in the area, there is no real sewage or wastewater discharge to the area. These phage are only associated with human or animal sewage, wastewater, and graywater discharges (Havelaar 1993). Sobsey (1994) also reported that he could not find RNA F+ phage in cattle feces or ditch water in cattle pastures.

The discharge from the Mississippi River freshwater diversion at Naomi yielded only two RNA F+ type I phage during the entire sampling period. However, the highest MPN FC/100 ml from this river discharge site was 490 MPN/100 ml in July 1993. This seems to indicate that the area is assimilating any extremely high levels of fecal coliforms before they reach the oyster growing areas—at least at Naomi.

Water-borne Microbial Pathogens and Public Health Risk

Two main sources of water-borne pathogens in estuarine waters are of importance in discussing and evaluating public health risk from water-borne pathogens. These are (1) naturally occurring marine bacteria, some of which may be pathogenic under certain circumstances, and (2) bacterial and human viral pathogens from fecal pollution of estuarine waters.

Only the potential enteric pathogens from fecal or sewage sources are addressed through water quality regulatory indicator standards for primary and secondary recreational contact or for shellfish growing estuarine waters (APHA 1985, Cabelli 1983, NAS 1991, USFDA 1993a, b). The naturally occurring marine bacteria do not correlate to standard bacterial indicators of fecal pollution and are not enumerated for regulatory purposes (Colwell 1984, Rodrick 1990, NAS 1991, USFDA 1993a, b).

The first route of infection is by direct contact (wound or other) with naturally occurring pathogens in estuarine waters through primary or secondary marine contact. All wound infections are from contact with marine waters containing the *Vibrio* pathogens.

The second route of infection is from consumption of sewage-contaminated estuarine or marine waters through primary recreational contact, or consumption of raw or re-contaminated seafoods containing either naturally occurring marine pathogens or enteric pathogens from fecal pollution of harvest waters. Table 36 summarizes the routes of infection from water-borne pathogens in estuaries.

Natural Marine Pathogens

Most of the free-living marine bacteria that may be capable of causing human disease under certain circumstances belong to the family *Vibrionaceae*. This family includes the genera *Vibrio, Aeromonas* and *Plesiomonas*. These bacteria are not associated with fecal pollution and do not correlate with standard fecal coliform bacterial indicators. They are free living in estuarine and marine waters, and increase in culturable numbers with increase of salinity (5 ppt–25 ppt) and temperature (>15EC) (Colwell 1984, Rodrick 1990, NAS 1991).

About 11 of the 66 species of vibrios in marine environments can cause illness in humans under certain circumstances. The most pathogenic is *Vibrio cholerae. V. cholerae* and *V. parahaemolyticus* are the only two species that have caused reported outbreaks involving more than two individuals. An epidemic of *V. cholerae* has been ongoing in South America for several years and has killed thousands of individuals. The other species of vibrios cause isolated and sporadic incidents of disease in single individuals and overwhelmingly in high-risk individuals who have predisposing underlying disease of the liver, blood, stomach, or immune system, which may include one or a combination of the following underlying illnesses: liver disease, alcoholism, diabetes, peptic ulcer renal disease, gastric surgery, heart disease, hematologic disease, immunodeficiency (including AIDS), cancer, or chemotherapy (LDHH, OPH 1993; Rodrick 1990; NAS 1991; USFDA *V. vulnificus* Workshop 1994, pers. comm.).

The most potentially serious of the *Vibrio* species for persons with underlying illness (high-risk individuals) is *V. vulnificus*; although, all vibrios are potentially dangerous to high-risk individuals. *V. vulnificus* illnesses have probably been reported in highest numbers in these individuals because of its normal ubiquitous presence in all marine waters and at extremely high levels (10^2–10^3/ml) in estuarine waters with temperatures >25EC (Kaysner et al. 1987, Oliver et al. 1983, Rodrick 1990).

Illness from naturally occurring *Vibrio* species can be divided into three main disease types (table 37): (1) mild to severe gastroenteritis, (2) wound infections, and (3) primary septicemia (Rodrick 1990, NAS 1991, USFDA *V. vulnificus* Workshop 1994, pers. comm.).

Mild to severe gastroenteritis and wound infections may occur in normal and high-risk individuals. However, all illness is more prevalent in high-risk individuals. Primary septicemia is associated with infections of *V. vulnificus* in high-risk individuals only. Primary septicemia can result from consumption by a high-risk individual of raw seafoods containing *V. vulnificus* normally found in marine waters in very high numbers.

Table 36. Routes of infection from water-borne pathogens in estuaries (from Kilgen 1991).

1. Wound or other infections from primary or secondary contact with estuarine or marine waters

   All from naturally occurring *Vibrio* species in estuarine waters—mainly *V. vulnificus* in hig- risk individuals

2. Consumption of marine waters or seafoods contaminated with pathogens from harvest waters

   Mainly from human fecal pollution of estuarine harvest waters

   a. Human enteric viruses

      Human Norwalk and Norwalk-like viruses (number one cause of all seafood-associated illness), hepatitis A virus, and enteral hepatitis E (very rare)

   b. Enteric bacteria

      Salmonella, Shigella, Campylobacter

   Naturally occurring *Vibrionaceae* marine bacteria

   More prevalent in high-risk individuals

   Mild to severe gastroenteritis mainly in high risk individuals

   Most severe possibility is primary septicemia from *V. vulnificus* in hig- risk individuals only (50% mortality)

   Usually involves time/temperature abuse or re-contamination of cooked seafood with raw seafood or estuarine water

Incidences of mortalities associated with water-borne pathogens in estuarine waters are generally only from *Vibrio* primary septicemia and wound infections, mainly *V. vulnificus*, and only in high-risk individuals. However, the associated mortality in these high-risk individuals with *Vibrio* primary septicemia is as high as 50% (LDHH, OPH 1994; NAS 1991; Rippey 1994; USFDA *V. vulnificus* Workshop 1994, pers. comm.).

A Centers for Disease Control (CDC) survey in Florida showed that 25% of the adult population ate raw oysters. Of these individuals, 2.5% of them knew they had liver disease. Only 7% of the high-risk patients were warned by medical personnel not to eat raw oysters. Only 16% of liver transplant patients were warned (USFDA *V. vulnificus* Workshop 1994, pers. comm.). The following facts were also obtained from the USFDA *Vibrio vulnificus* Workshop (1994, pers. comm.):

- Persons who became ill from *V. vulnificus* by eating raw oysters purchased them from these sources: 85% restaurant, 11% retail, 4% wholesale. None were self-harvested.

Table 37. Types of disease from naturally occurring *Vibrio* species.

	Normal Individuals	High-Risk Individuals (with underlying illness)
Gastroenteritis[a]	+	++
Wound Infections[b]	+	++
Primary Septicemia[c]	-	++[1]

[a] Gastroenteritis (GI)—Very minor GI infection, 48–72 hours. No deaths. Also, mainly in high-risk individuals**.** Some very few reported in persons with no known underlying illnesses. However, when GI illness occurs, other enteric pathogens are very frequently isolated with *V. vulnificus*. These may be from other food sources.

[b] Wound Infections (WI)—These can occur in high-risk and normal individuals. Approximately 80% of the patients are high risk. This is a very serious infection, with a 25% mortality rate in high-risk individuals. However, in rare cases, it can even be fatal to normal individuals. Normal individuals may also have infections from wound contact with seawater severe enough to require amputation. Wound infections account for about 1/3 of the total cases of *V. vulnificus,* but only 8% of the total fatalities. (This may be due to the fact that many normal individuals get wound infections, and their possibility of death is extremely small.) In Florida from 1981 to 1987, there were significantly more patients with wound infections from contact with seawater than primary septicemias from consumption of raw shellfish (USFDA *V. vulnificus* Workshop 1994, pers. comm.).

[c] Primary Septicemia (PS)—Disease of bacteria growing in the blood. Only high-risk individuals get this type**.** This is the most serious.

[1] Once a high-risk individual contracts this form of illness, the mortality rate is about 50% and can be as rapid as two days following infection. CDC data shows 100% of primary septicemia patients had underlying illnesses (USFDA *V. vulnificus* Workshop 1994, pers. comm.).

- CDC data showed 100% of primary septicemia patients had underlying illnesses.

- 99.9% of individuals who were high risk did not get *V. vulnificus* infections from eating raw oysters. (The approximate risk to high-risk oyster eaters is only 1 in 10,000 for primary septicemia.)

- There are no documented cases of anyone dying from eating commercially shucked oysters. In the only possible CDC case documented, the person also ate raw shell-stock.

Pathogens from Fecal or Sewage Pollution

Increases in coastal pollution and resultant runoff into the estuaries can bring enteric bacterial and viral pathogens from fecal or sewage sources. Epidemiological data suggest that these are mainly human enteric viruses from human fecal pollution (Cabelli et al. 1983; CDC 1989; Dufour and White 1985; NAS 1991; LDHH, OPH 1993; Rippey 1994).

The current microbiological fecal coliform indicator standards in shellfish and growing waters are extrapolated from standards set in the 1920s to protect the public from typhoid fever epidemics (NAS 1991, USFDA 1993a, b). Results from studies in the last decade have indicated that these existing microbiological indicator standards and thus the classification of shellfish growing waters may no longer be valid (Cole et al. 1986, Elliot and Colwell 1985, Gerba 1988, Kilgen and Cole 1990, NAS 1991). This is also true for primary and secondary recreational contact in marine or estuarine waters (Cabelli et al. 1983). Although the current fecal coliform indicator was developed to prevent enteric bacterial illness, current epidemiological data indicate human enteric viruses rather than bacterial pathogens are of main concern in fecal contamination of waters and shellfish, and that a constant and predictable relationship does not exist between fecal coliform indicators and enteric viruses in estuarine waters and shellfish (Cabelli et al. 1983, Elliot and Colwell 1985, Kilgen 1989, NAS 1991).

The public health risk of fecal material from animal sources versus human sources also has always been in question. There is a great need for research in this area to assess the human health risks from wild and domestic animal runoff. Although animals can carry some bacterial pathogens, these have generally not been documented to be associated with shellfish-borne illnesses (CDC 1989, NAS 1991, Rippey 1994).

Data Analysis and Results

Status and Trends of Fecal Coliform Indicators

The LDHH, OPH, OWMP has a total of about 700 water sampling stations for oyster growing areas across the Louisiana coast. The Terrebonne basin (12) has about 200 sampling stations, and the Barataria basin (02) has about 117 sampling stations.

Four strategically placed LDHH sampling sites in each of the two estuaries were selected for trends analyses. The locations of these LDHH stations are shown in figure 96. Comments regarding the status of each site and the potential impacts on its water quality were provided by Ken Hemphill and Dan Mathews of the LDHH, OPH, OWMP. All LDHH sites in the Barataria and Terrebonne basins also were analyzed for overall trends from 1980–1994. In addition, two LDEQ Mississippi River water sampling stations on the east and west bank at the Pointe a la Hache freshwater diversion were included.

Fifteen years (1980–1994) of geometric mean MPN FC/100 ml data were statistically analyzed by regression correlations to determine any significant trends in increase or decrease of FC. The data were provided by the LDHH, OPH. Each year from 1980 through 1994 was divided into the two seasons identified by the results of Kilgen (1994) (figures 97 and 98). These seasons were April–October (summer season) and November–May (winter season).

Results of the regression correlations are summarized in table 38 and the overall Barataria and Terrebonne basins trends are shown in figures 99 and 100. Figures for all LDHH station data are provided in appendix C. Overall, there are no statistically significant trends at the 95% confidence level in fecal coliform MPN counts over the last 15 years in the Barataria or Terrebonne estuaries. Only the east bank of the Mississippi River station at Pointe a la Hache shows a significant downward trend.

Terrebonne Basin

(1) Hell Hole Lake–LDHH site number 168: This site has been historically closed but with no identified sources of fecal coliforms, with the possible exception of wild birds or other animals. It will open for the first time May–August 1995. The trend for MPN FC/100ml is slightly down but not significant.

(2) Sister Lake–LDHH site number 134: This site is a state seed ground and a highly productive commercial oyster-producing area. It is closed six months of the year for management purposes. It is open May–August and September–October. The trend MPN FC/100ml is slightly down for November–March but not significant. The trend from 1980 to 1994 from April to October is level.

(3) Bay Cocodrie–LDHH site number 83: This area has been historically closed since the 1970s and is not highly commercial. However, it was considered important because it is impacted by campsites in the area. The trend for MPN FC/100ml is down for April–October but not significant. The trend for November to March is nearly level.

(4) Catfish Lake–LDHH site number 1: This area is a commercial oyster harvesting site. It is only open in the summer season. The trend for MPN FC/100ml is very slightly up for April to October but not significant. The trend for November to March is level.

Table 38. Summary of regression trends analyses of geometric mean data points of samples for each year, 1980–1994 (through 1990 for Mississippi River) (12 months per year, some years with less). Regression model used is $Y_t = a + b\ t$, where Y_t is the annual geometric mean of MPN FC/100 ml at time t, b is the slope coefficient and a is the intercept. Significance at the 95% confidence level is indicated by *. Plots of all LDHH station data are provided in appendix C.

Site	DHH Site Number	Period	Trend	df	Regression Equation	r
Terrebonne Basin						
All LDHH Stations	-	Apr–Oct	Down	13	Y_t = 15.00-0.49t	0.264
		Nov–Mar	Leve	12	Y_t = 132.20-0.30t	0.034
Hell Hole Lake	168	Apr–Oct	Down	8	Y_t = -19.61+6.19t	0.077
		Nov–Mar	Down	8	Y_t = 46.50-1.92t	0.071
Sister Lake	134	Apr–Oct	Level	12	Y_t = 4.10-0.07t	0.038
		Nov–Mar	Down	11	Y_t = 20.91-0.80t	0.080
Bay Cocodrie	83	Apr–Oct	Down	13	Y_t = 16.83-0.69t	0.268
		Nov–Mar	Level	14	Y_t = 51.17+0.19t	0.0008
Catfish Lake	1	Apr–Oct	Up	12	Y_t = 3.94+0.12t	0.044
		Nov–Mar	Level	10	Y_t = -16.26+3.16t	0.472
Barataria Basin						
All LDHH Stations	-	Apr–Oct	Level	13	Y_t = 14.15-0.35t	0.117
		Nov–Mar	Down	12	Y_t = 145.91-1.80t	0.214
St. Mary's Point	70	Apr–Oct	Down	12	Y_t = 6.62-0.31t	0.284
		Nov–Mar	Down	10	Y_t = 15.30-0.56t	0.073
Bay San Bois	53	Apr–Oct	Up	13	Y_t = 5.85+0.19t	0.056
		Nov–Mar	Down	12	Y_t = 38.63-1.65t	0.189
Bay Adams	32	Apr–Oct	Level	13	Y_t = 2.49+0.01t	0.004
		Nov–Mar	Level	13	Y_t = 5.07-0.03t	0.004
Bay Jacques	10	Apr–Oct	Level	13	Y_t = 14.24-0.20t	0.012
		Nov–Mar	Level	13	Y_t = 18.47+0.07t	0.001
Mississippi River						
West Bank Ferry Landing Pointe a la Hache		All months combined	Down	9	Y_t = 722.11-46.61t	0.462
East Bank Ferry Landing Pointe a la Hache		All months combined	Down	9	Y_t = 635.78-41.55t	0.646*

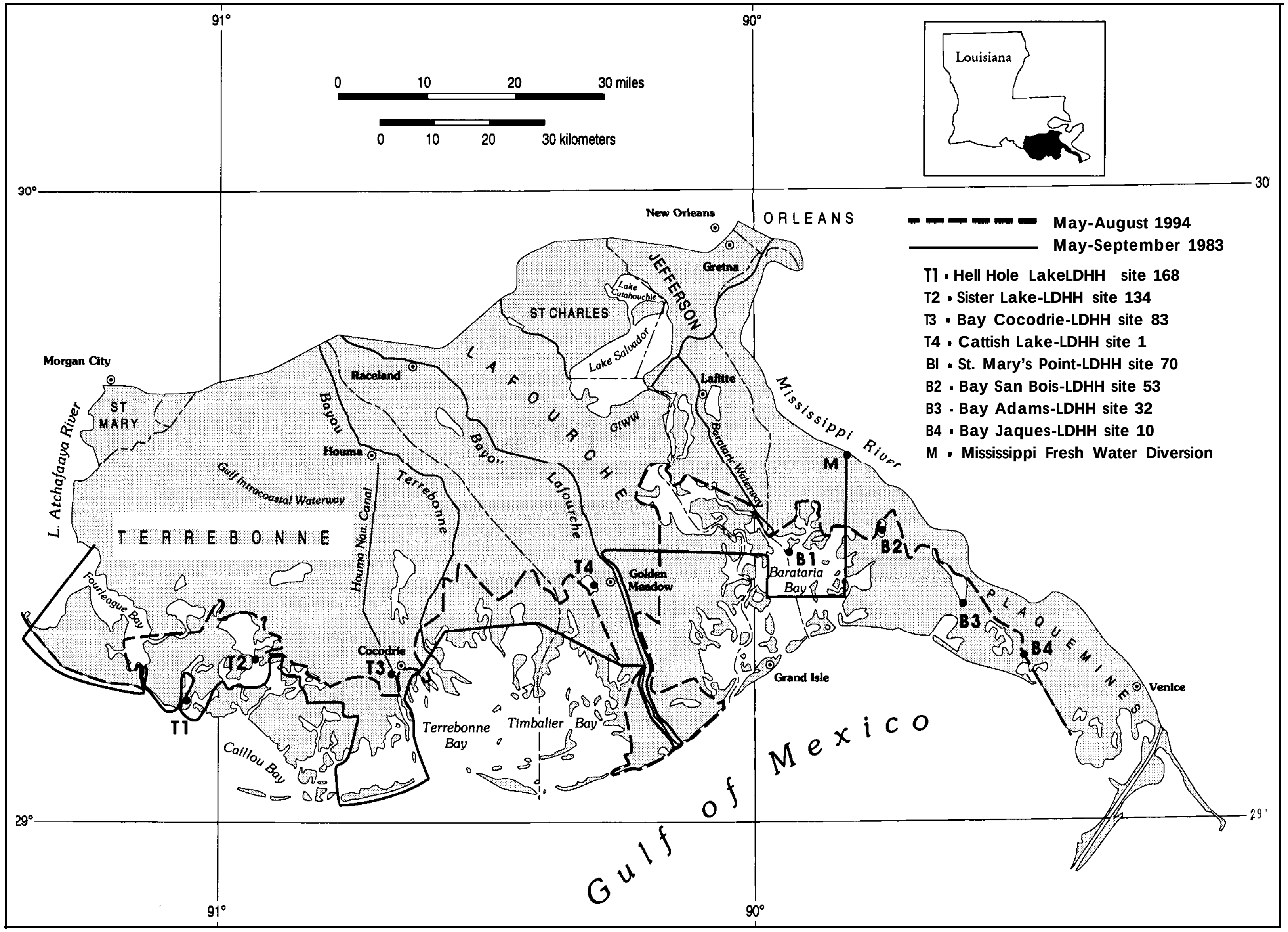


Figure 96. Selected LDHH sites in Barataria and Terrebonne and shellfish growing lines for 1983 versus 1994.

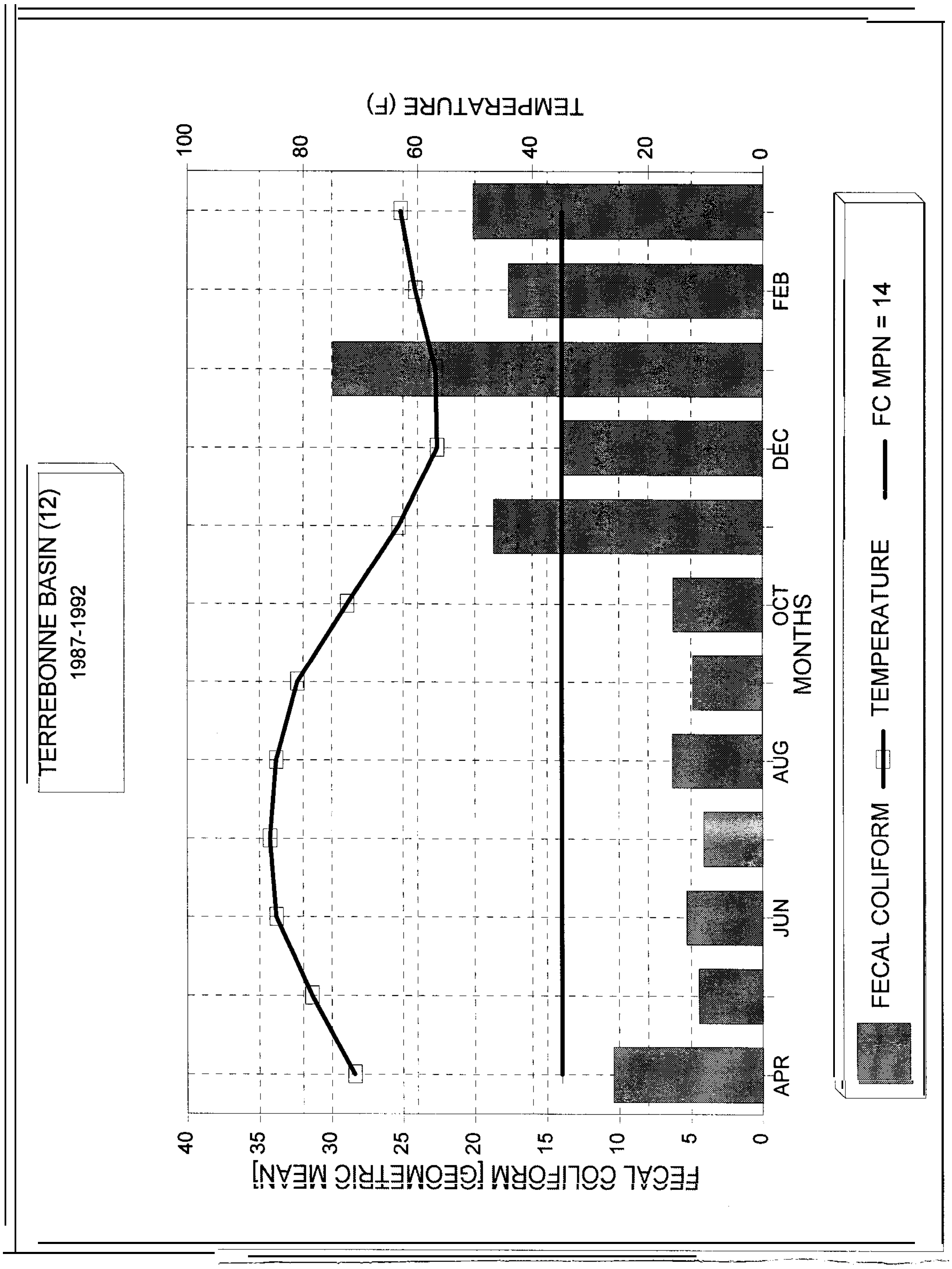


Figure 97. Seasonal monthly FC and temperature for Terrebonne basin (1987–1992) (from Kilgen 1994).

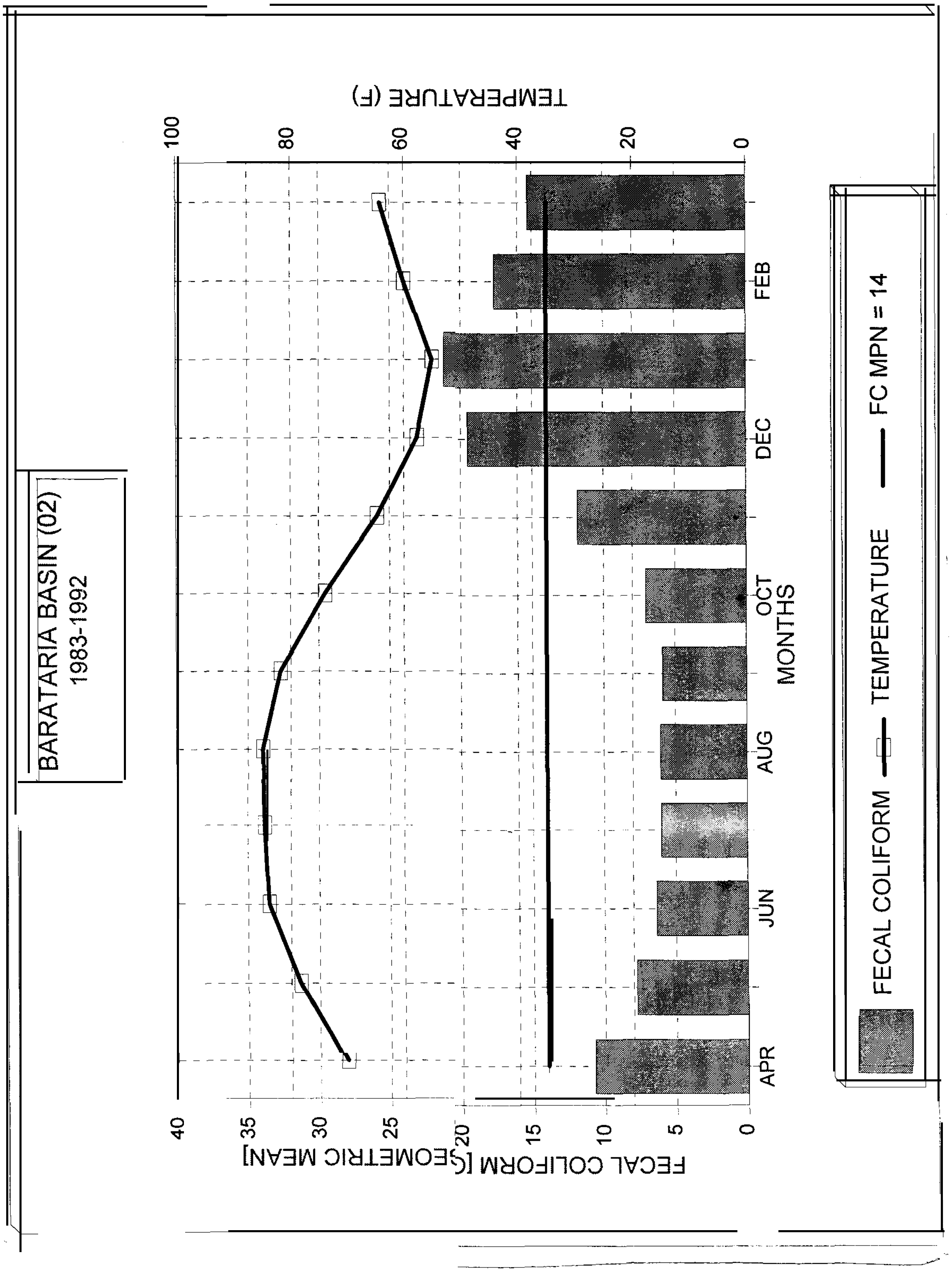


Figure 98. Seasonal monthly FC and temperature for Barataria basin (1983–1992) (from Kilgen 1994).

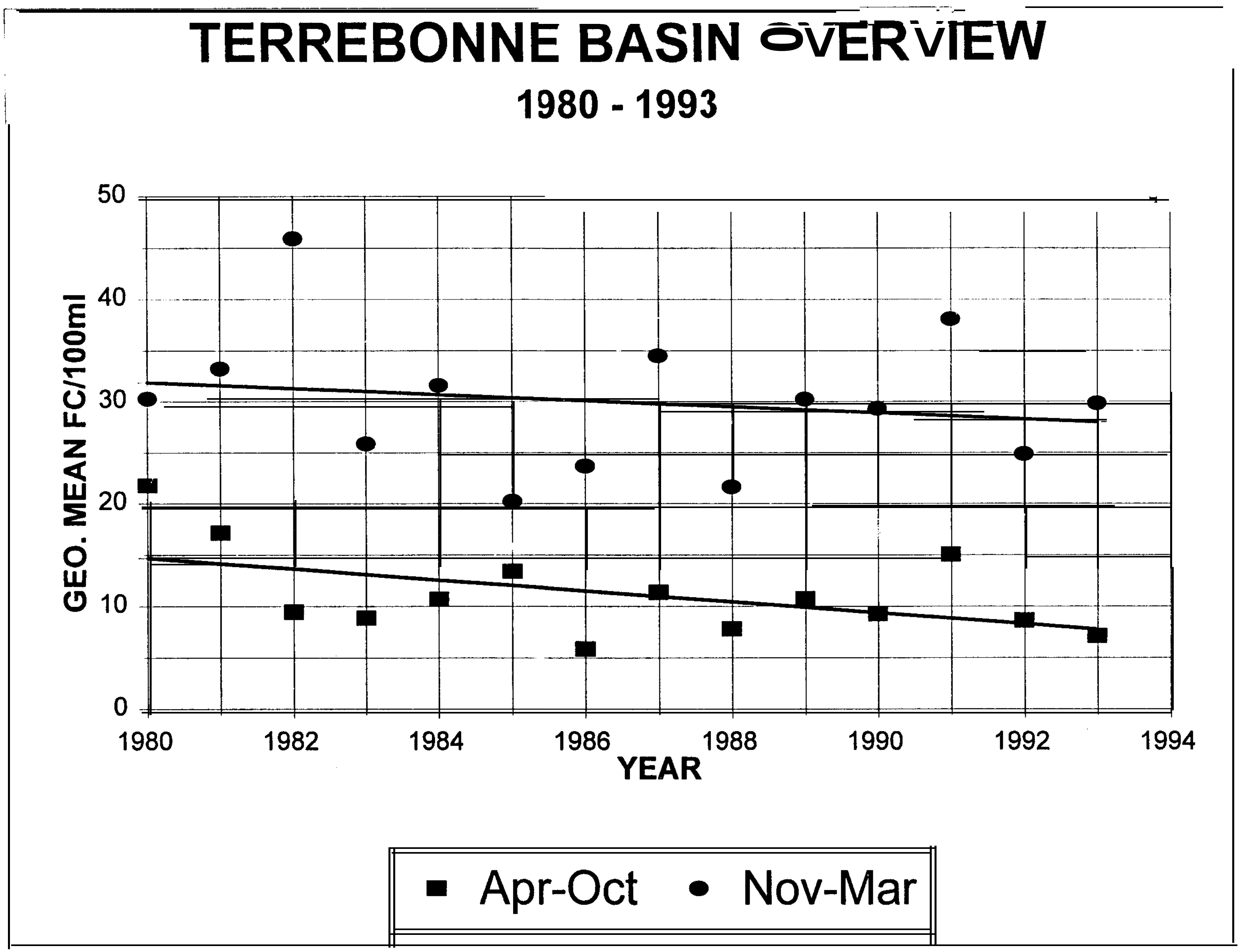


Figure 99. Results of regression correlations of fifteen years of geometric mean MPN FC/100 ml data for all LDHH sampling sites in the lower Terrebonne basin for the periods April–October and November–March.

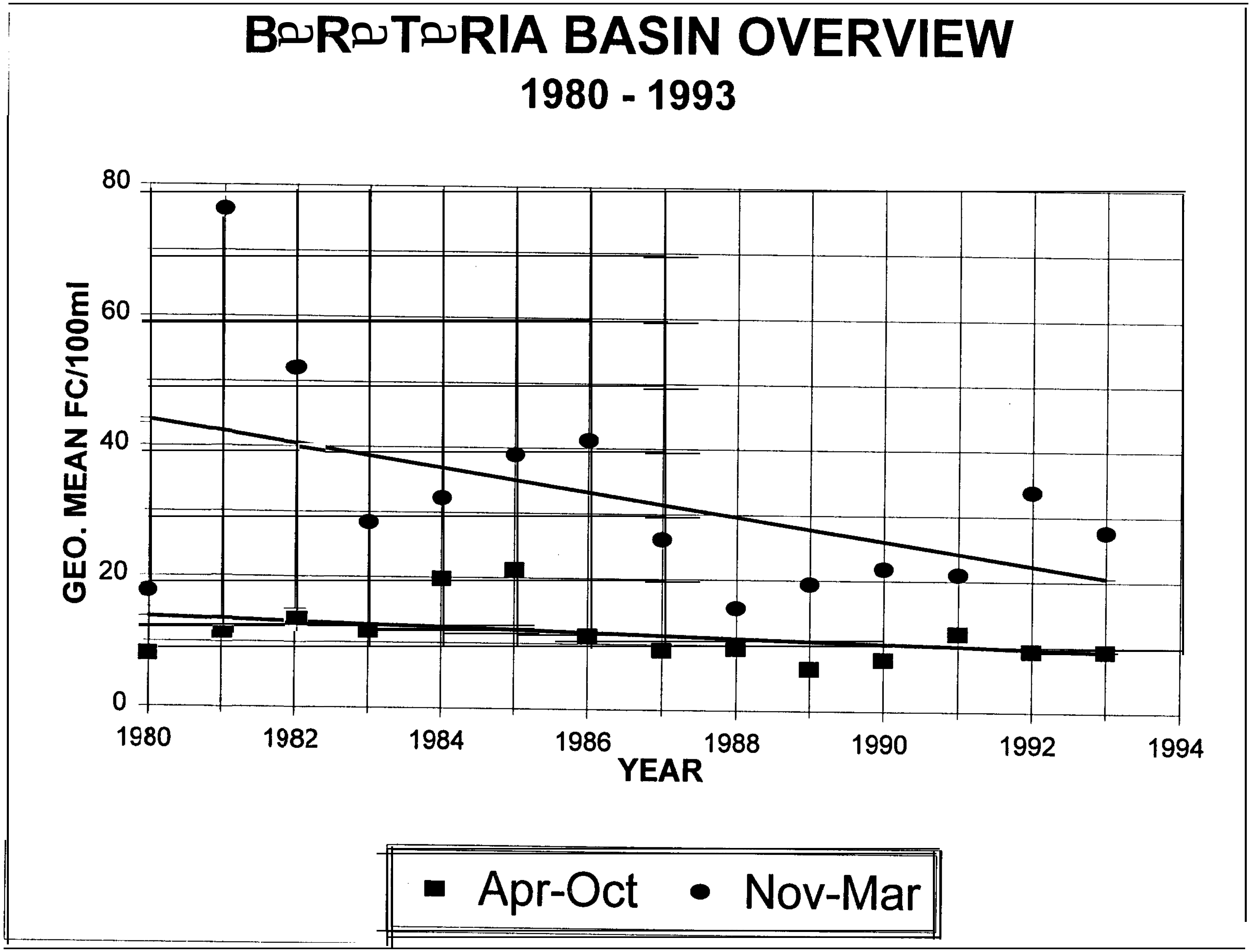


Figure 100. Results of regression correlations of fifteen years of geometric mean MPN FC/100 ml data for all LDHH sampling sites in the lower Barataria basin for the periods April–October and November–March.

Barataria Basin

(1) St. Mary's Point–LDHH site number 70: This site is a highly commercial oyster growing and harvesting area that is impacted by the Barataria Waterway. It is also impacted by the Naomi freshwater diversion from the Mississippi River, which was a site for the study of Kilgen et al. (1994) on the assessment of pumping stations and freshwater diversions for fecal coliform and virus contamination to the oyster growing areas. The trend for MPN FC/100ml is down but not significant.

(2) Bay San Bois–LDHH site number 53: This is a highly commercial area. It opens in the summer period (May–August) and closes September–April. It is impacted by several pumping stations along the Mississippi River and the freshwater diversion at West Pointe a la Hache. The trend for MPN FC/100ml is up for April–October and down for November–March but were not significant.

(3) Bay Adams–LDHH site number 32: This is also a highly commercial oyster-producing area. Southern winds in the summer keep salt water in the bay. It is closed in winter because of low tides and water from the Empire locks when they open for traffic. There were no visible trends in these two seasons for this site.

(4) Bay Jacques–LDHH site number 10: This site is also highly commercial for oyster production. It is impacted by the Mississippi River and the pumping station at Triumph. It is closed in winter. There were no visible trends in these two seasons for this site.

Mississippi River

In addition to analyzing data from the Barataria and Terrebonne estuaries, we examined Mississippi River data for sites near freshwater diversions that could impact the oyster growing areas. LDEQ had only one water quality fecal coliform site in the Mississippi River at a freshwater diversion. It was at the ferry landings on the west and east bank of the river at Pointe a la Hache. This data base covered 1980–1990. There was a downward trend in the geometric mean MPN FC/100ml on the west bank of the river at the ferry landing, but it was not statistically significant at the 95% level. The downward trend for the east bank of the river at the ferry landing was statistically significant at the 95% confidence level (but not at the 99% level).

Status and Trends of Microbial Pathogens and Associated Public Health Issues

Status and Trends of Natural Marine Pathogens

There were no reported illnesses from *Aeromonas* or *Pleisomonas* species in the Barataria and Terrebonne estuaries from 1980 to 1994 (LDHH, OPH 1993). In fact, only 18 cases of seafood-related *Pleisomonas* and seven cases of seafood-related *Aeromonas* were reported in the United States between 1978 and 1990 (NAS 1991, Rippey 1994).

The *Vibrio* species involved in a total of 134 reported illnesses in the Barataria and Terrebonne estuaries between 1980 and 1994 included *V. vulnificus, V. parahaemolyticus, V. cholerae* non-01, *V. hollisae, V. mimicus, V. alginolyticus, V. fluvialis,* and *V. damsiela*. Appendix C contains the data on all *Vibrio* illnesses for 1980–1994. Figure 101 shows the percentage of reported illnesses from each species of *Vibrio* for the period (see also table 39), and figure 102 shows the summary of the disease types for all species of vibrios. When all eight species of *Vibrio* are evaluated for disease types, the gastrointestinal disease is most common (42%). Wound infections also are very common (34%). Primary septicemia only occur in *V. vulnificus* infections in high-risk individuals (17%). The remaining 7% represented infections of other organisms or unknown etiologies. Table C1 in appendix C provides all the *Vibrio* data. Figure 103 shows the disease types for *V. vulnificus* infections from 1980 to 1994 in the Barataria and Terrebonne basins. As expected, 100% of the patients with the most serious primary septicemia had an underlying illness. There was a 58% mortality rate for these individuals. This is the expected mortality for a *Vibrio* primary septicemia in a high-risk patient. There was only 2% gastroenteritis reported in patients with 100% underlying illness. Wound infections comprised 55% of the infections, and only 74% of the patients had underlying illness. As discussed above, wound infections in saltwater environments are potentially fatal even in normal, healthy individuals.

In general, the most potentially serious species for normal individuals with no underlying illness is *V. cholerae* 01. No cases of *V. cholerae* 01 were reported for this period. Fourteen mortalities were reported from the Barataria and Terrebonne estuaries for the period 1980–1994 (table 39). Eleven were from *V. vulnificus* primary septicemia in high-risk individuals. One death was from *V. parahaemolyticus* in a patient with underlying illness, but the disease route was unknown. One death resulted from a wound infection with *V. cholerae* non-01 in a high-risk individual. The last death resulted from a gastrointestinal infection of a high-risk individual with *V. hollisae*.

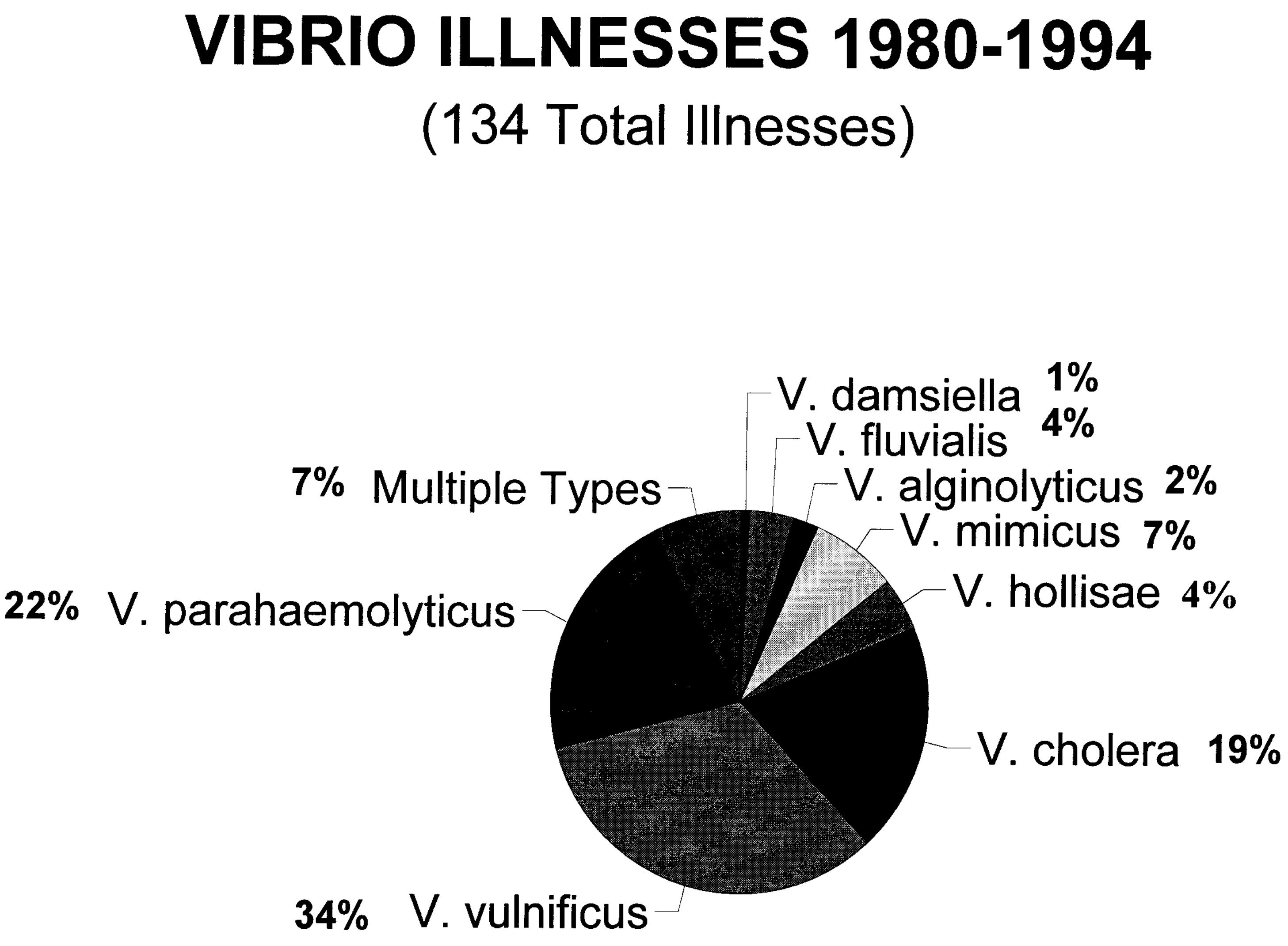
Status and Trends of Pathogens from Fecal or Sewage Pollution

In the United States for the last 10 years, very few cases of illness from enteric bacteria were reported to be associated with seafood consumption, and most of those

Table 39. Vibrio illnesses in the Barataria and Terrebonne estuaries from 1980 to 1994.

Year	VULN	PARA	N-01	HOLL	MIMI	ALGI	FLUV	DAMS	MULT	Totals
1980	4	0	0	0	0	0	0	0	0	4
1981	8	0	0	0	0	0	0	0	0	8
1982	2	0	0	1	0	0	0	0	0	3
1983	1	0	1	0	0	0	0	0	0	2
1984	1	0	0	0	0	0	0	0	0	1
1985	0	0	0	0	0	0	0	0	1	1
1986	1	3	2	0	1	1	0	0	2	10
1987	1	1	5	0	0	0	0	0	0	7
1988	2	3	2	3	1	1	0	0	1	13
1989	3	3	5	0	1	0	1	0	0	13
1990	2	2	4	1	1	0	0	0	0	10
1991	5	3	2	0	1	0	0	0	1	12
1992	4	4	0	0	2	0	0	0	2	12
1993	6	2	2	0	3	1	2	1	2	19
1994	4	8	3	1	0	0	2	0	1	19
Totals	44	29	26	6	10	3	5	1	10	134
Deaths	11* All PS	1 UK	1 WI	1 GI						14 deaths in 15 yrs

*All deaths were in individuals with underlying illness (LDHH, OPH, 1995, S. Wilson, pers. comm.)
PS = Primary Septicemia
UK = Unknown
WI = Wound Infection
GI = Gastrointestinal



ure 101. Summary of percentage of reported illnesses from each species of *Vibrio* for the period 1980–1994.

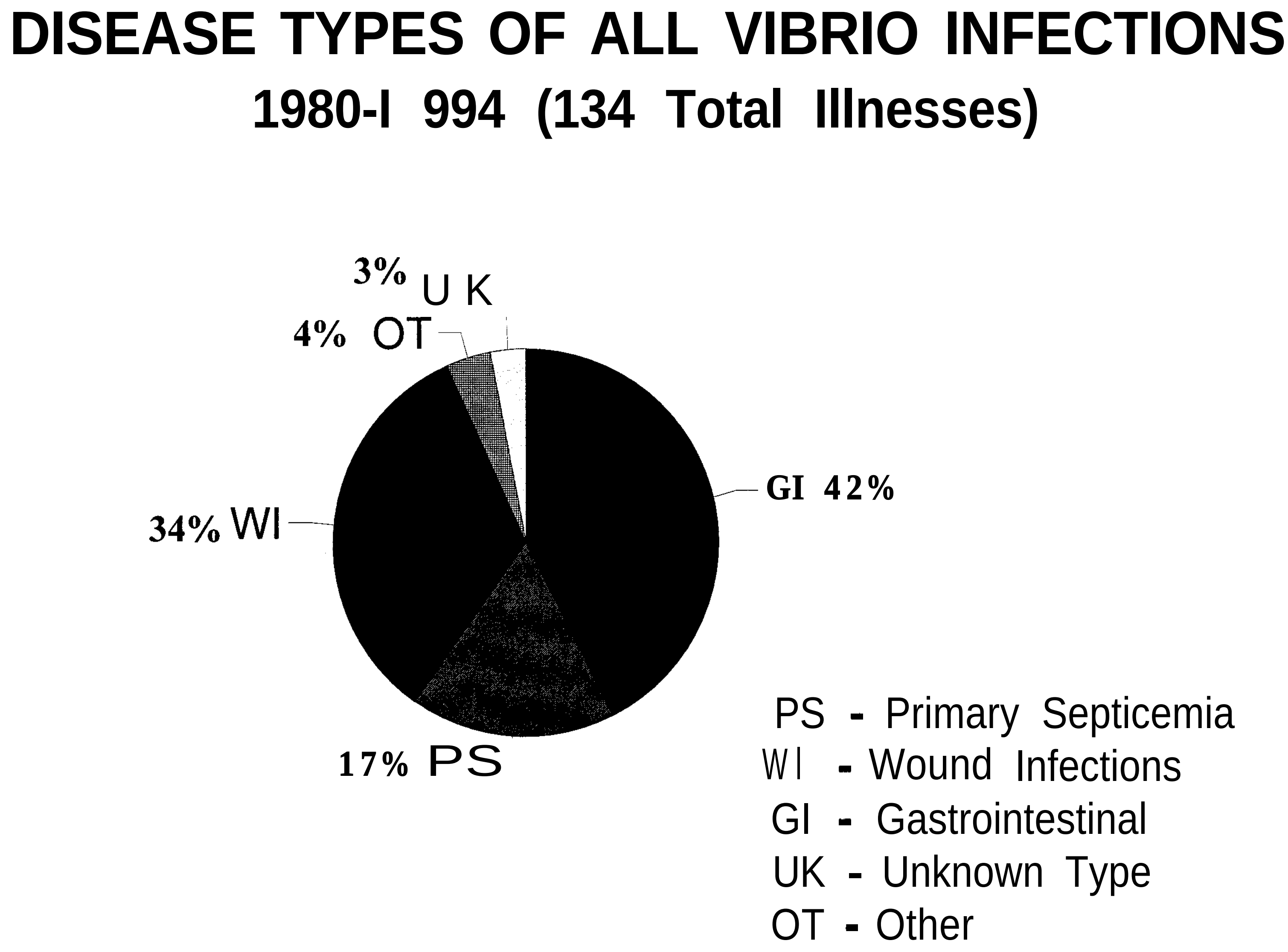


Figure 102. Disease types of all Vibrio infections (134 total; 8 species) for the period 1980-1994.

cases were from re-contamination by food handlers (CDC 1989, NAS 1991, Rippey 1994). No cases of enteric bacterial illness were reported from the Barataria and Terrebonne estuaries.

The most serious potential human enteric viruses are hepatitis A virus (HAV) and only very rarely in this country, enteral hepatitis E. There have been only one or two cases of enteral hepatitis E in the United States in the last 10 years (NAS 1991), and none were reported from Louisiana. Cases of HAV related to consumption of raw seafood contaminated with human sewage have been decreasing in the last 15 years, with the exception of outbreaks in Florida in 1988, 1989, and 1990 (NAS 1991, Rippey 1994). There have been no reported cases of seafood-related HAV in Louisiana in the last 15 years (LDHH, OPH 1993, 1995; NAS 1991; Rippey 1994).

Infections from consumption of fecally contaminated marine waters during primary contact recreation (swimming) are rare, and none were reported by LDHH Department of Epidemiology from 1980 through 1994 (LDHH, OPH 1995).

A National Academy of Sciences (NAS) report on the safety of seafood concluded that most illnesses from fecal pollution of estuarine waters result from consumption of raw shellfish or other seafoods contaminated with the human enteric viruses, Norwalk and Norwalk-like agents of gastroenteritis. These are very mild illnesses with 12–24-hour duration and have no associated mortality. Norwalk virus is considered the number one cause of all seafood-related illness, mainly from consumption of raw shellfish contaminated with human sewage (NAS 1991). Norwalk virus is also a common cause of gastroenteritis from swimming in sewage-polluted marine waters (Cabelli et al. 1983).

In 1982, the LDHH, OPH reported one outbreak of approximately 500 cases of mild 12–24-hour gastroenteritis associated with raw oyster consumption in the Barataria and Terrebonne estuaries (CDC 1989; Kilgen and Kilgen 1989, 1990; NAS 1991; Rippey 1994). The cases were officially reported as being of "unknown etiology" to the CDC and the USFDA New England Technical Services Unit (CDC 1989, Rippey 1994), but the clinical symptoms were strongly suggestive of human Norwalk virus etiology. Growing water samples taken from Barataria Bay and Sister Lake, where the implicated shellfish were harvested, were tested for human enteric viruses. Norwalk virus cannot be cultured in the laboratory, but other culturable human enteric viruses were isolated from the growing waters during the outbreak period (Cole et al. 1986, Kilgen et al. 1988).

Following this sewage-associated viral outbreak, a new system of seasonal classification for shellfish growing waters was implemented in 1982. A parishwide sewerage system was installed for Terrebonne Parish. Since that time, there have been no additional reported incidences of sewage-related illnesses from shellfish or seafood consumption in the Barataria and Terrebonne estuaries (Kilgen and Kilgen 1989, 1990; LDHH, OPH 1993; Rippey 1994).

In November 1993, the Louisiana Office of Public Health reported a multi-state outbreak of Norwalk virus gastroenteritis linked to Louisiana oysters from the Grand Pass/Cabbage Reef area east of the Mississippi River. These oysters were not from Barataria or Terrebonne but were harvested from a pristine area in Mississippi Sound

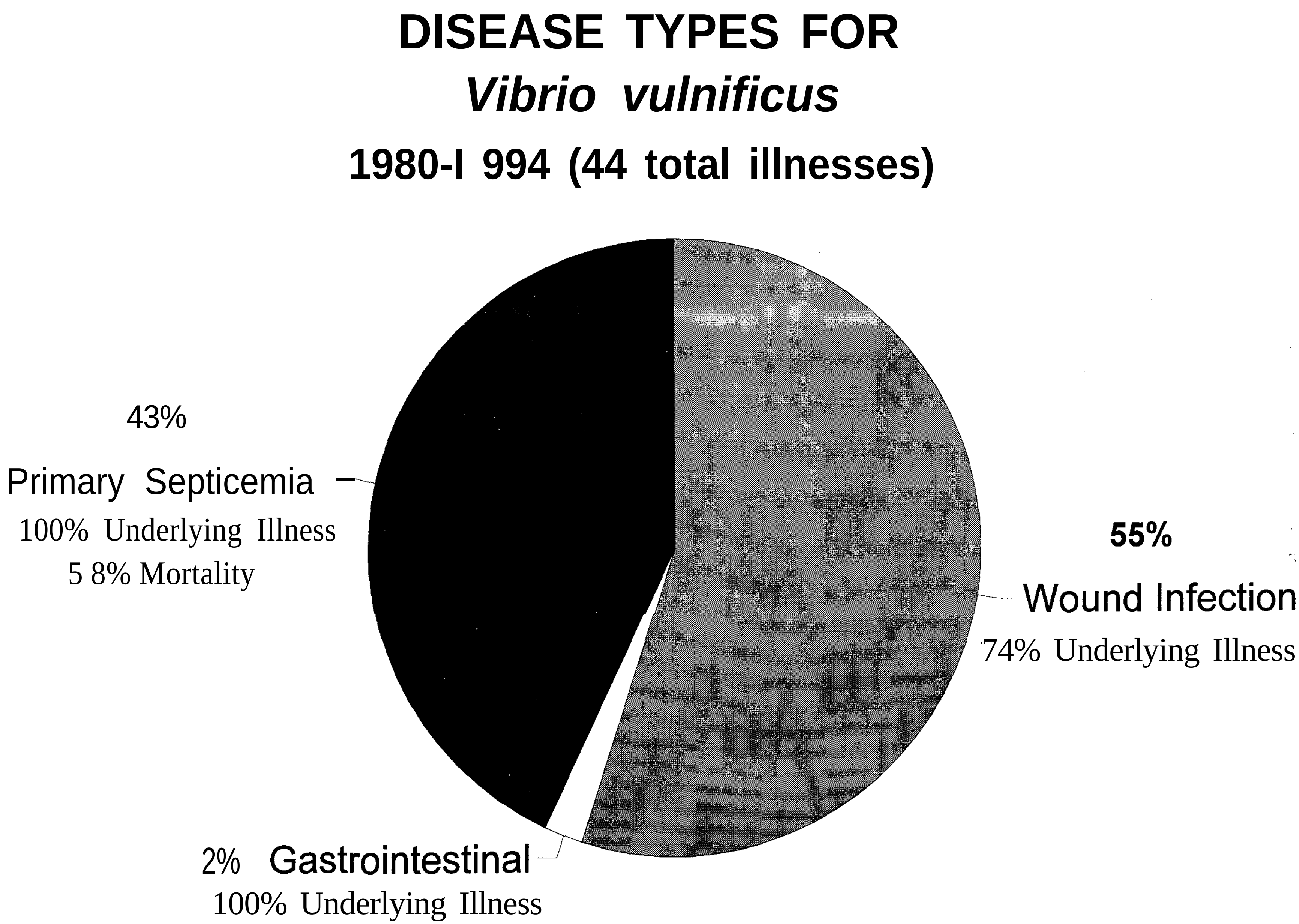


Figure 103. Disease types for Vibrio *vulnificus* infections for the period 1980-1994 (11 deaths from primary septicemia).

with approved fecal coliform levels. Subsequent epidemiological investigations by the Louisiana Office of Public Health and the CDC documented that overboard sewage disposal from oyster harvesters on boats was the source of the enteric virus (Kohn et al. 1995). Because this outbreak occurred at the peak oyster sales before Thanksgiving, the economic impact to the oyster industry and to Louisiana was tremendous.

Status and Trends of Shellfish Bed Closures

The basic approach for the current system of classification of shellfish growing areas in Louisiana was established in 1984. This system had a conditional management system of four classification periods using a 10-year data base of fecal coliform levels: November–February, March–April, May–August, and September–October. Rudimentary statistical analyses on fecal coliform (FC) data indicated higher FC levels in the colder months of the year and improved water quality during the summer months. Rather than opening areas as soon as the FC counts dropped to acceptable levels, a management decision was made to close areas for an entire season to insure safe oyster growing-water quality (Hemphill 1994).

In 1990, the USFDA refused to accept this system of classification. The LDHH, OPH, OWMP agreed to examine a two-season classification system utilizing a five-year data base for water sample analyses that showed frequency distributions of the percent of samples greater and less than 43 MPN FC/100ml that were collected in the most adverse weather conditions (Hemphill 1994).

A study of the criteria used in Louisiana's four-season conditional management areas, relative to the National Shellfish Sanitation Program manual requirements, was conducted for Barataria, Terrebonne, and Pontchartrain basins to determine whether a two-season classification was feasible (Kilgen 1994). The results from this study clearly showed a two season range of geometric mean MPN FC/100ml (figures 97 and 98). The two seasons were based on FC counts inversely correlated with temperature.

This study resulted in the implementation of the current two-season, four-period classification of molluscan shellfish growing waters. The four periods of the 1984 scheme are incorporated into seasons of "summer and winter." The "summer" season is from April through October, and the "winter" season is from November through May. The FC data analyzed for the Kilgen (1994) report used this breakdown. However, the official two-season classification program currently in effect is November–April and May–October (Hemphill 1994). These each contain two periods:

(1) November–April is divided into
  a) November–February
  b) March–April
(2) May–October is divided into
  a) May–August

b) September–October

The placement of the classification line that delineates approved oyster harvesting areas is determined using a 10-year data base from each sample station, which is examined for various statistical parameters, including (1) the percentage of samples greater than 43 MPN FC/100ml, (2) the number of samples greater than 43 MPN FC/100ml, and dates of their collection, (3) estimated 90th percentile, (4) geometric mean MPN FC/100ml, (5) median MPN FC/100ml, (6) mode value, and (7) standard deviation. In addition, the largest MPN FC/100ml value greater than 43 and the lowest MPN FC/100ml also are examined. Finally, a test of means of wet/dry and rising/falling tide data is analyzed to determine if either would have an adverse affect on open stations. If a condition is identified as being more adverse to the station, the station is then reclassified under the identified parameter. This procedure was developed in collaboration with the USFDA Northeast Technical Service Unit (Hemphill 1994).

There are no data available to determine percentages of oyster-bed closures at any time during one of the classification periods. This is due to many factors, including the fact that many growing areas are closed if there is no commercial oyster harvest because those areas are not sampled. Also, oyster growing areas can be closed immediately during an emergency situation like a hurricane or an outbreak of oyster-associated illness from an identified area.

The only method available to examine the trends of oyster-bed closures over the last 10 years was to compare the placements of the classifications lines from the oldest available map with closure lines, which was May–September 1983, with a similar period of the classification lines for 1994. Figure 96 shows these classification lines. It appears from the closure lines from 1983 and 1994 that there is no increase in area of closures over this period.

Conclusions and Recommendations

Fecal Coliform Indicators

Overall, there are no statistically significant trends in increases or decreases in fecal coliform MPN counts over the last 15 years in the Barataria or Terrebonne estuaries in the four sites selected for trends analysis. Some sites had a slightly upward or downward trend others remained level for the period. Only the east bank of the Mississippi River at the Pointe a la Hache East Bank Ferry Landing showed a significant decrease in MPN fecal coliform levels for the period 1980–1994. Only Catfish Lake in April–October and Bay San Bois in April–October showed slight upward trends in MPN fecal coliform counts; however, it would be hard to conclude that either area had an increase in fecal coliform loading. Summary regressions of all four sites within each of the Barataria and Terrebonne basins also were insignificant for any change with time. It is recommended that the LDHH, OPH, OWMP data base is a good background and baseline data source to evaluate future status and trends of fecal coliforms in the estuaries.

Natural Marine Pathogens

There were 134 illnesses in the Barataria and Terrebonne estuaries between 1980 and 1994 due to eight species of the natural marine genus *Vibrio*. The main conclusions from the analysis affecting these illnesses are:

- *Vibrio* infections were the only cause of death from contact with marine waters or ingestion of raw seafoods.
- There were 14 *Vibrio*-related deaths in the 15-yr period; all were in high-risk individuals with underlying illness.
- 11 of the high-risk deaths were from *V. vulnificus* primary septicemia.
- 100% of all *V. vulnificus* infected individuals had underlying illness.
- The largest percentage of *Vibrio*-related illness was gastrointestinal (42%)
- Wound infections accounted for 34% of the *Vibrio* illnesses and primary septicemia accounted for 17% of illnesses.
- With the exception of wound infections and gastrointestinal infections, all individuals infected with *Vibrio* species had underlying illness.

Overall, it appears that the number of *Vibrio*-related illnesses increased significantly in 1986; however, this is most likely because *V. vulnificus* illnesses became an issue with the press and the oyster industry at this time. Potential *V. vulnificus* cases were actively investigated in many shellfish-producing states since 1986. From that time, all *Vibrio* infections in Barataria-Terrebonne seem level. The deaths from *Vibrio* infections in the Barataria and Terrebonne basins averaged one per year for the last 15 years.

It is recommended that the state continue its educational efforts to inform high-risk consumers and recreational users of the estuaries of the potential risk of infection from the natural marine vibrios, and especially of the risk of fatal wound infection or primary septicemia from eating raw seafoods like oysters. This educational information should be non-sensational and should make it absolutely clear that the risk to high-risk consumers of raw seafoods is 1 in 10,000 of contracting a potentially fatal (50%) primary septicemia from *V. vulnificus* or one of the other *Vibrio* species. Normal healthy individuals have no risk of contracting a high-mortality primary septicemia from raw seafoods. The most serious risk to normal healthy individuals from the naturally occurring marine vibrios is contracting a serious wound infection. Death from a wound infection in a healthy individual would be very rare, but it could be possible in extreme cases that amputation would be necessary to prevent mortality.

Pathogens from Fecal or Sewage Pollution

The main conclusions for the period 1980–1994 are:

- No cases of enteric bacterial illness were reported.
- In 1982, the LDHH, OPH reported one outbreak of approximately 500 cases of mild 12–24-hour Norwalk-like viral gastroenteritis (from human fecal pollution) associated with raw oyster consumption in the Barataria-Terrebonne estuarine system.
- Following this sewage-associated viral outbreak, a new system of seasonal classification for shellfish growing waters was implemented in 1982. A parishwide sewerage system also was installed for Terrebonne Parish.
- Since that time, there have been no additional reported incidences of sewage-related illnesses from shellfish or seafood consumption.

It is recommended that the current system of seasonal classification of oyster growing waters has been extremely effective in preventing sewage-related contamination in oyster-growing waters of the estuarine system. The classification system is based on the fecal coliform indicator, which is an ineffective indicator of human fecal viruses like Norwalk virus, but it also is based on temperature, and this is somewhat effective for controlling human enteric viruses. The seasonal closures are more restrictive in winter when viruses like Norwalk have a greater incidence and longer persistence. It is less restrictive in the summer when there is less incidence and shorter persistence. The restrictiveness has been questioned but has undoubtedly also been responsible for the lack of illnesses from properly harvested waters.

Shellfish Bed Closures

It is difficult to draw conclusions concerning the trends in shellfish bed closures over the last 15 years because there are no data available to determine percentages of oyster-bed closures at any time. This is due to many factors, including the fact that many growing areas are closed if there is no commercial oyster harvest because those areas are not sampled. Also, oyster growing areas can be closed immediately during an emergency, like a hurricane or outbreak of oyster-associated illness. Visual observation of the closure lines from 1983 and 1994 indicate that there are no greater areas of closure. The greatest differences appear to be that all of the Barataria basin was open to the east of Barataria Bay in the 1983 summer closure. The closure line for the summer of 1994 extends west of the river almost to Venice. More of Barataria Bay itself was closed in 1983 than 1984, and Lake Pelto was closed all the way to Isles Dernieres in 1983.

It is recommended that a data base be developed to determine the numbers of actual shellfish-producing acres closed by fecal coliforms. An economic impact statement based on lost-harvest income for these closed areas also should be developed.

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Appendix A

Long-Term Trends in Surface Water Oxygen Saturation and BOD_5 Values

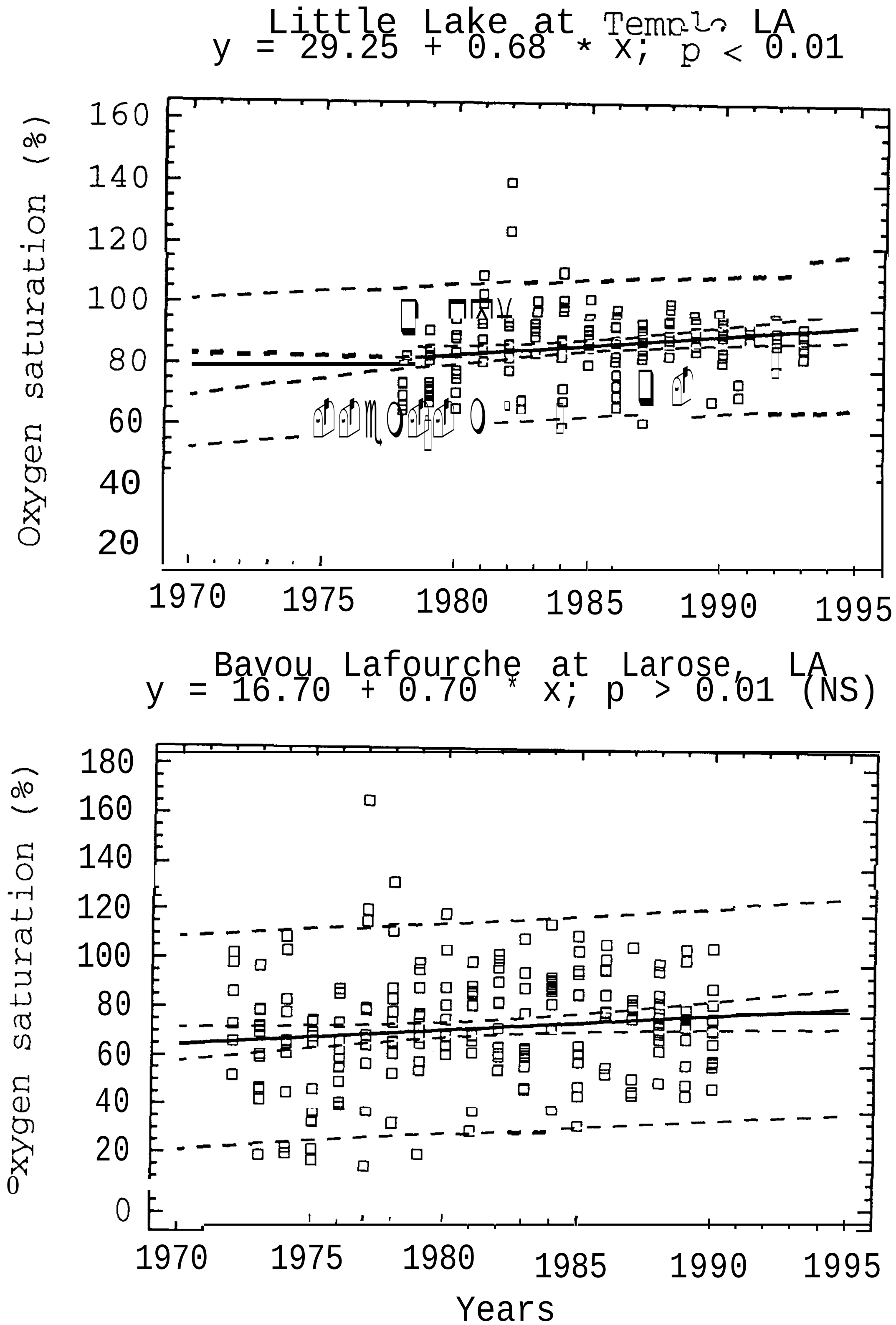
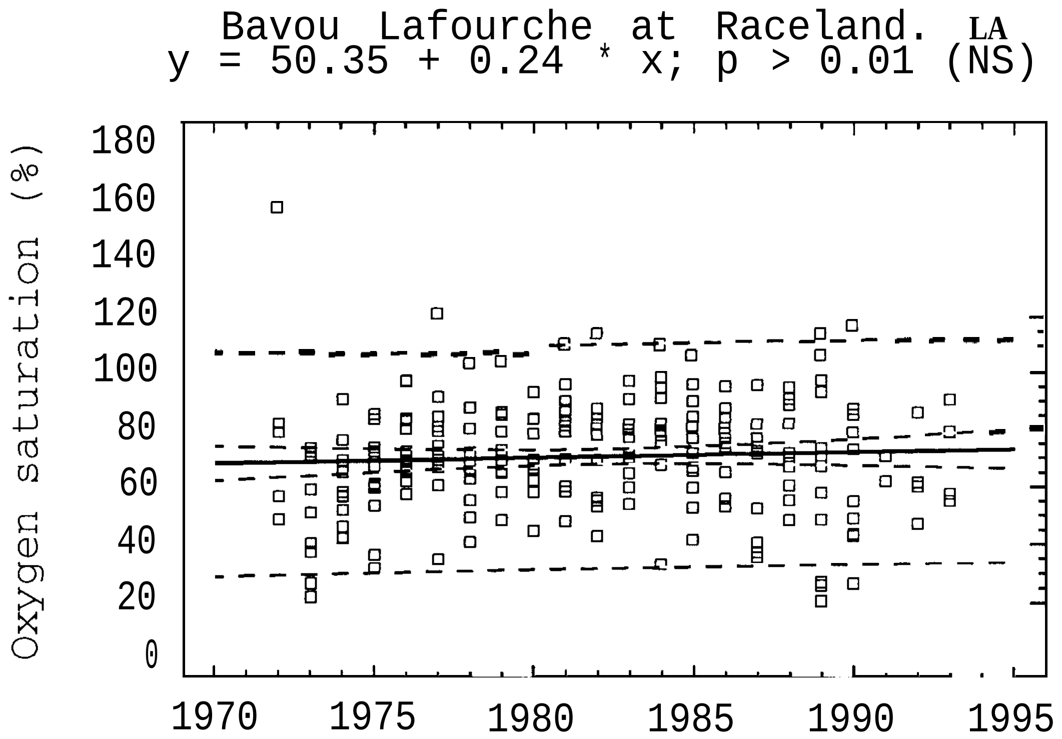


Figure Al. Long-term trends of surface water oxygen saturation for Little Lake at Temple and Bayou Lafourche at Larose.



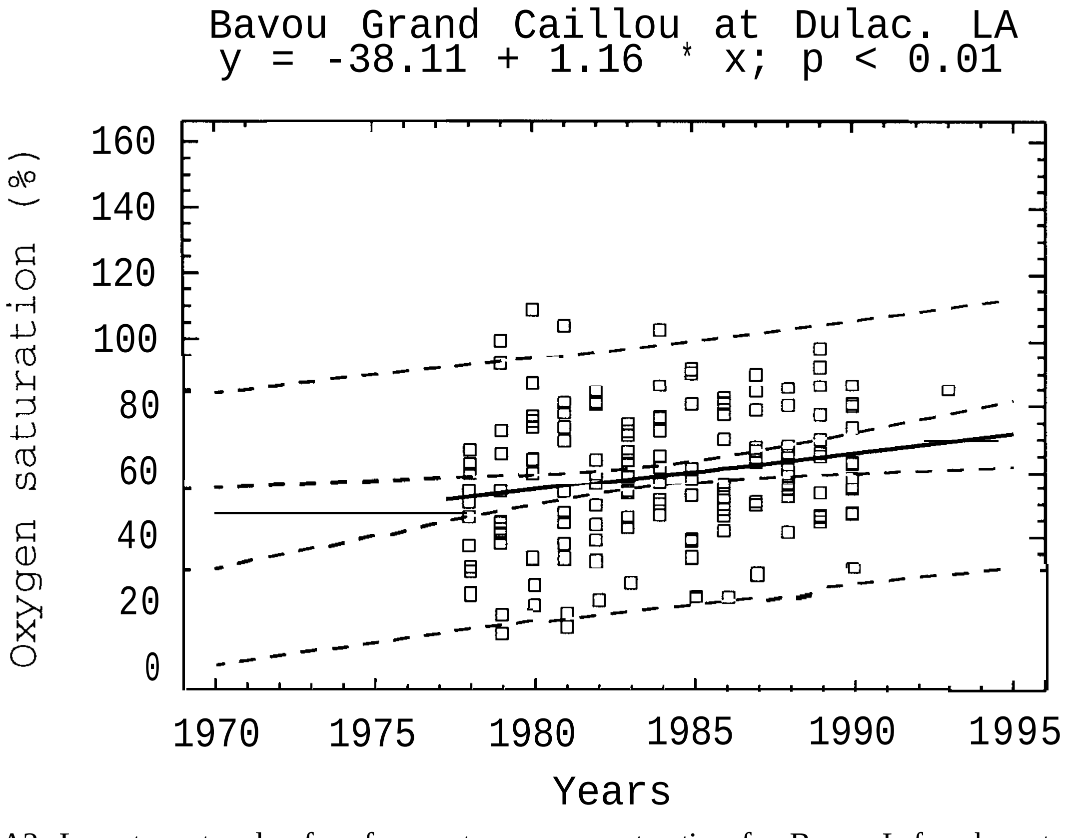


Figure A2. Long-term trends of surface water oxygen saturation for Bayou Lafourche at Raceland and Bayou Grand Caillou at Dulac.

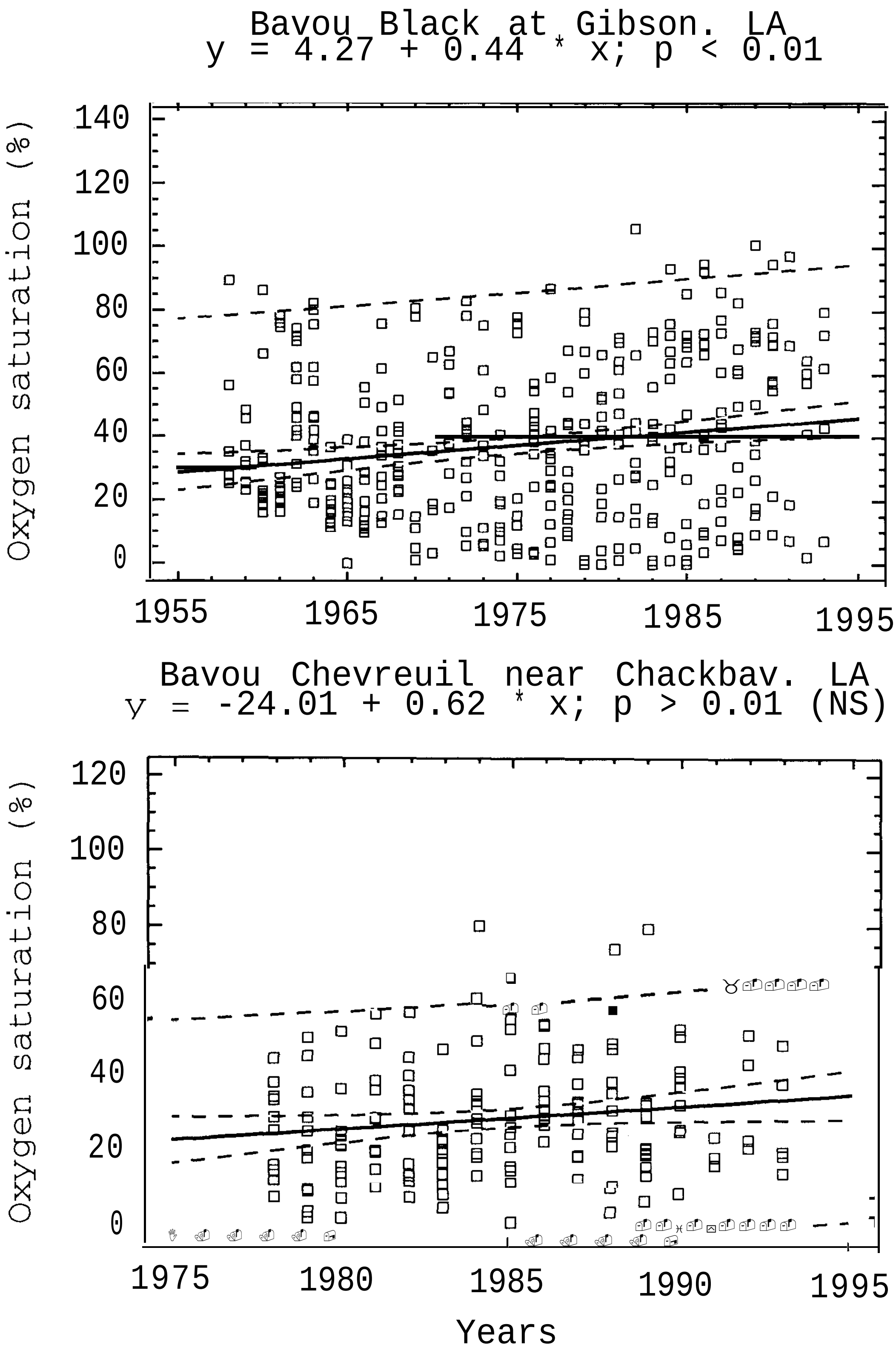


Figure A3. Long-term trends of surface water oxygen saturation for Bayou Black at Gibson and Bayou Chevreuil near Chackbay.

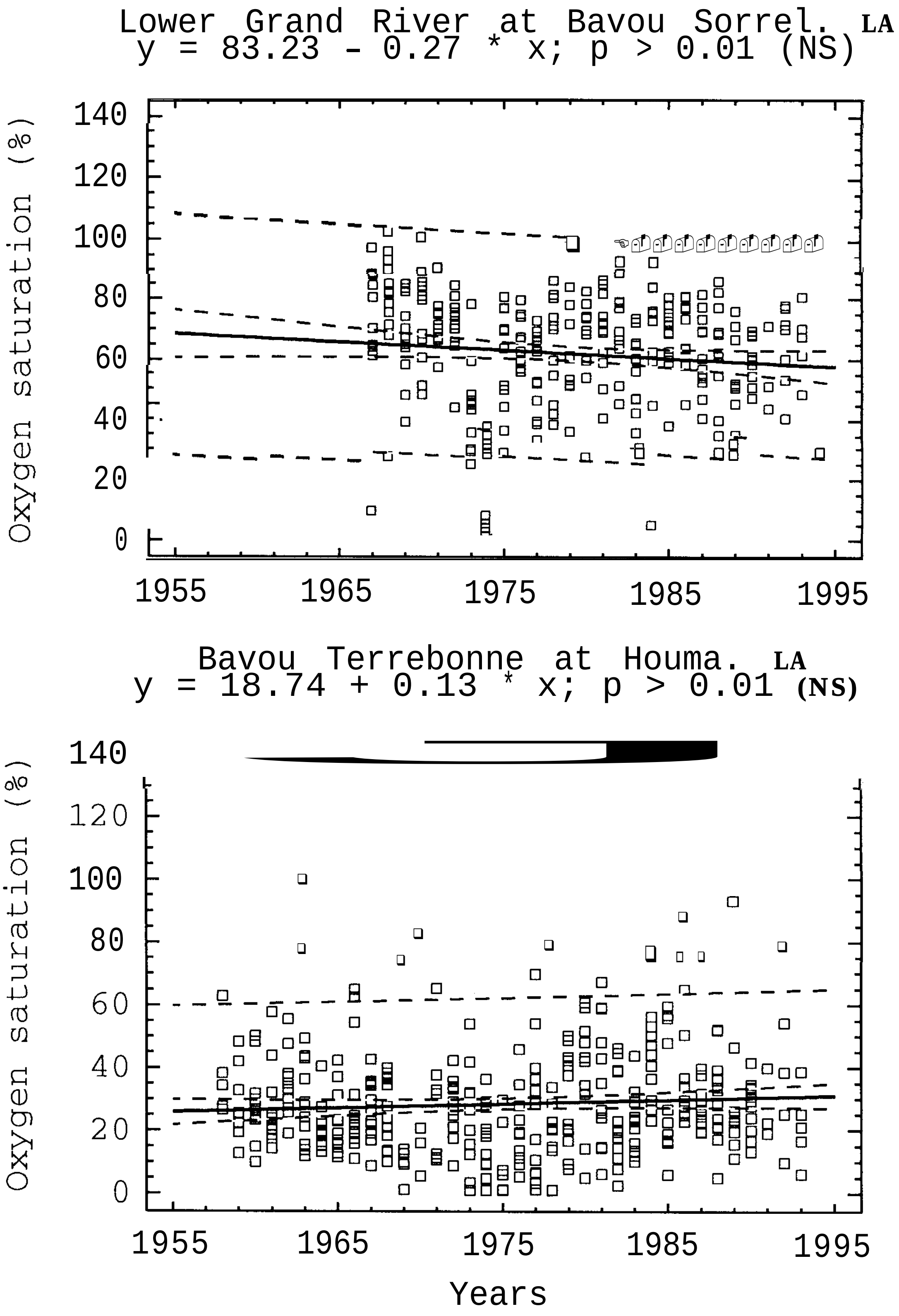


Figure A4. Long-term trends of surface water oxygen saturation for Lower Grand River at Bayou Sorrel and Bayou Terrebonne at Houma.

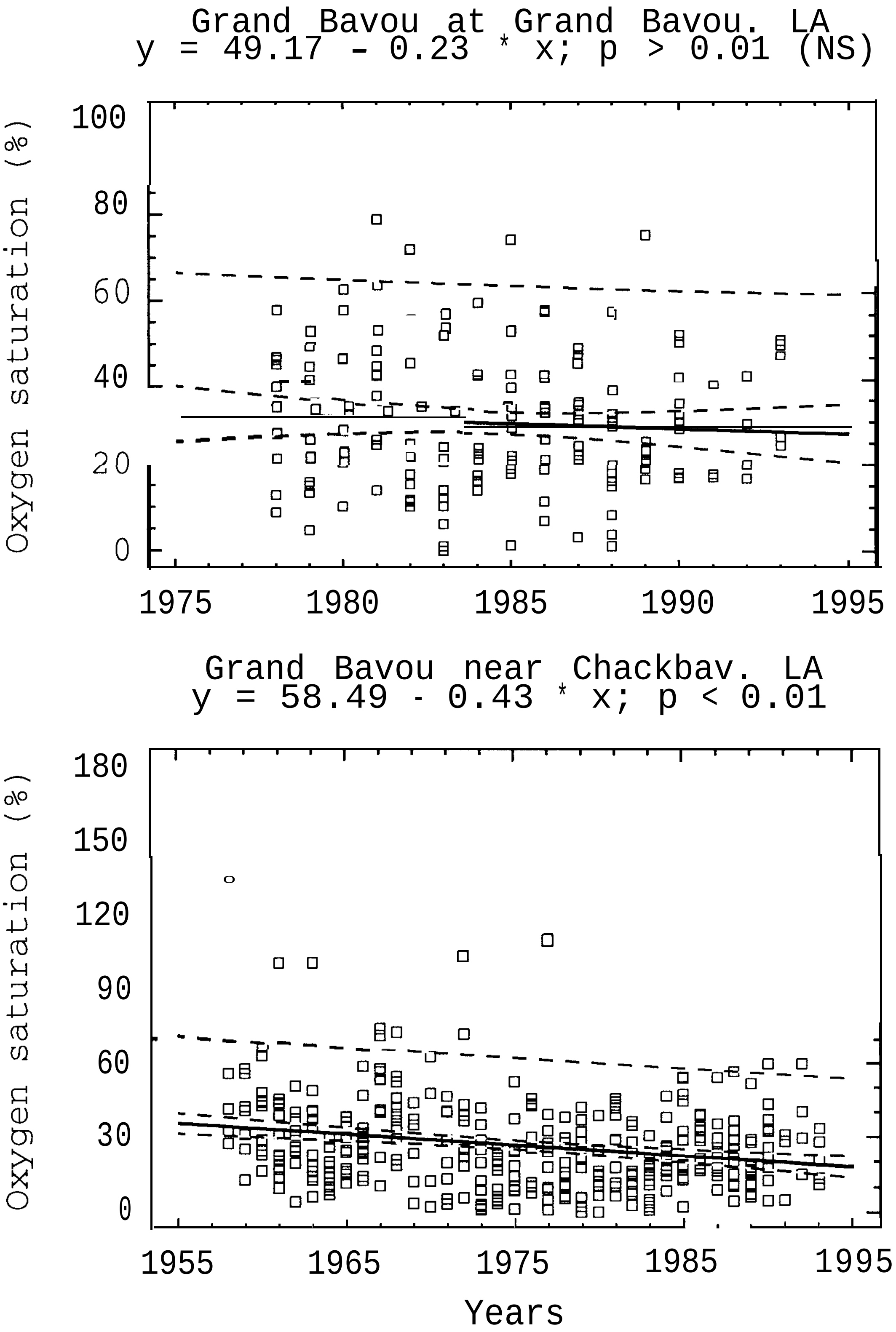


Figure A5. Long-term trends of surface water oxygen saturation for Grand Bayou at Grand Bayou and Grand Bayou near Chackbay.

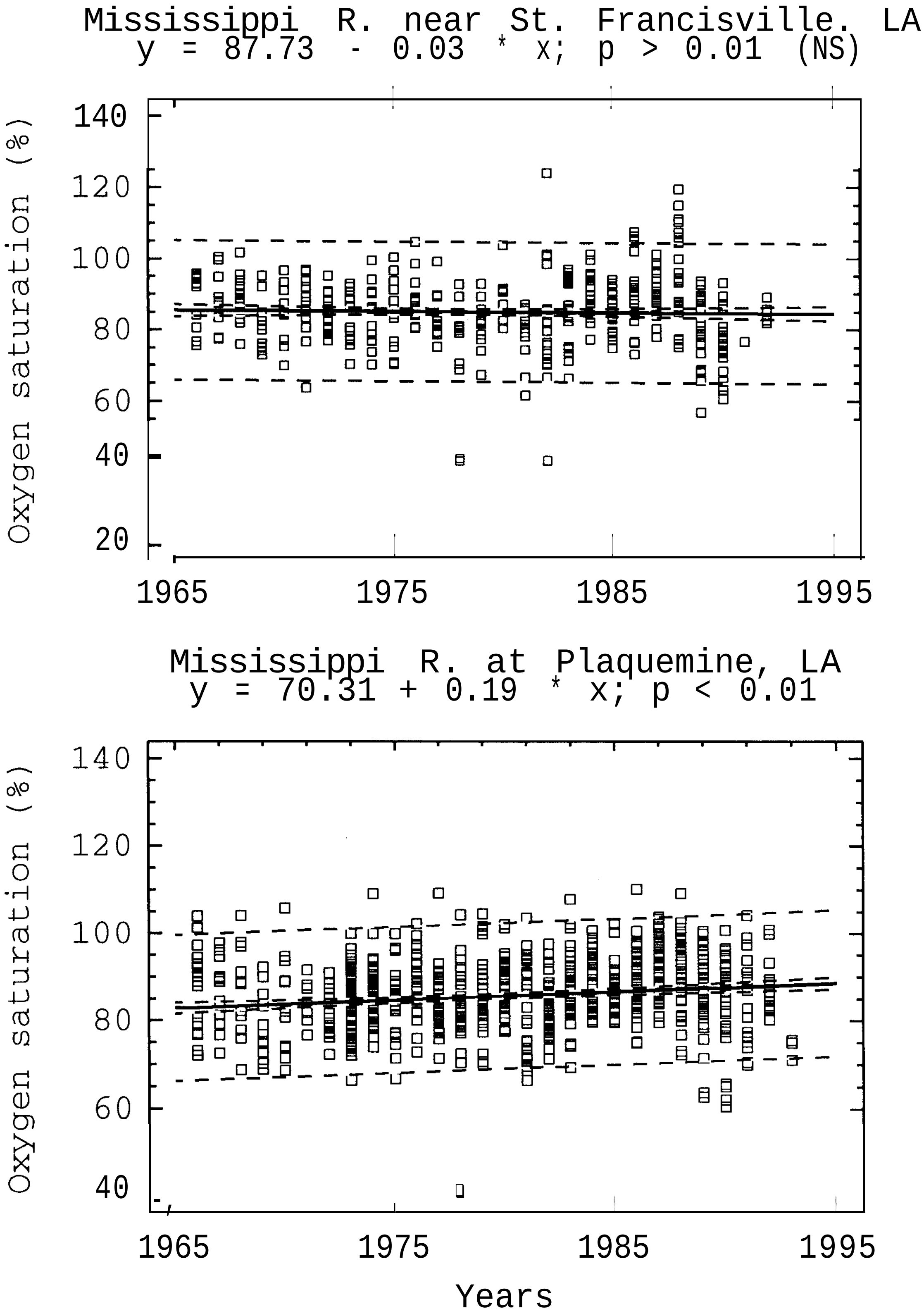


Figure A6. Long-term trends of surface water oxygen saturation for the Mississippi River near St. Francisville and at Plaquemine.

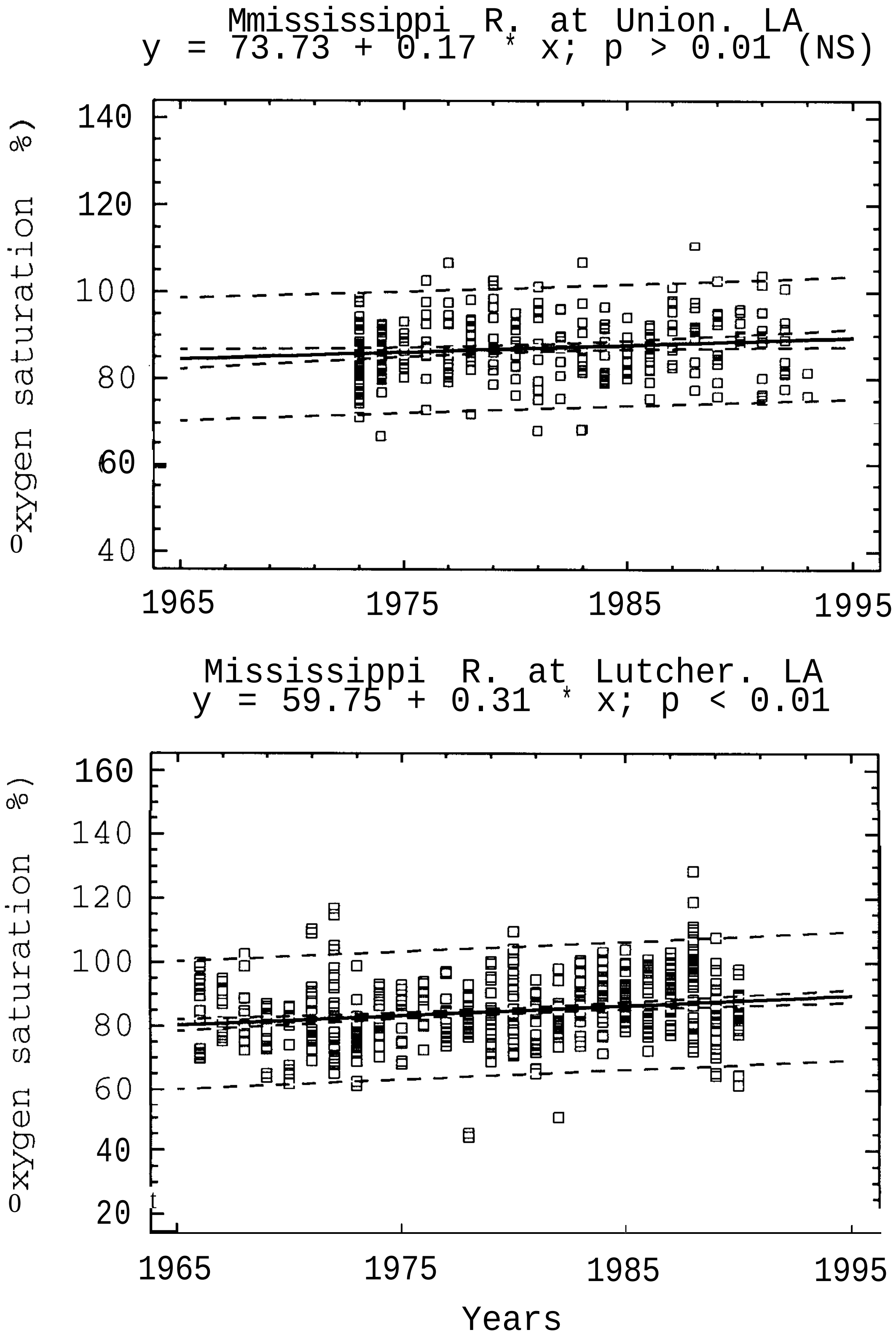


Figure A7. Long-term trends of surface water oxygen saturation for the Mississippi River at Union and at Lutcher.

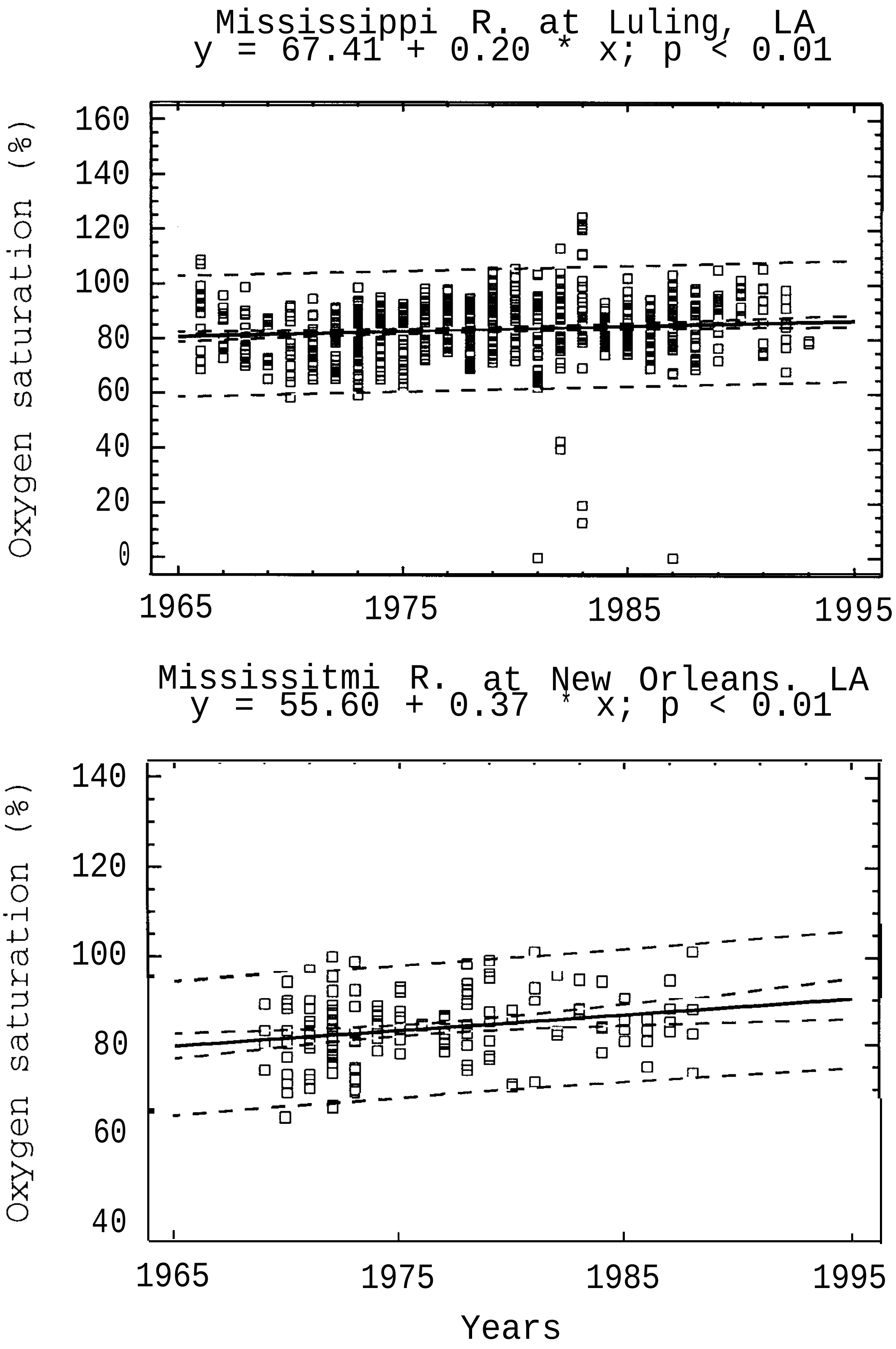


Figure AS. Long-term trends of surface water oxygen saturation for the Mississippi River at Luling and at New Orleans.

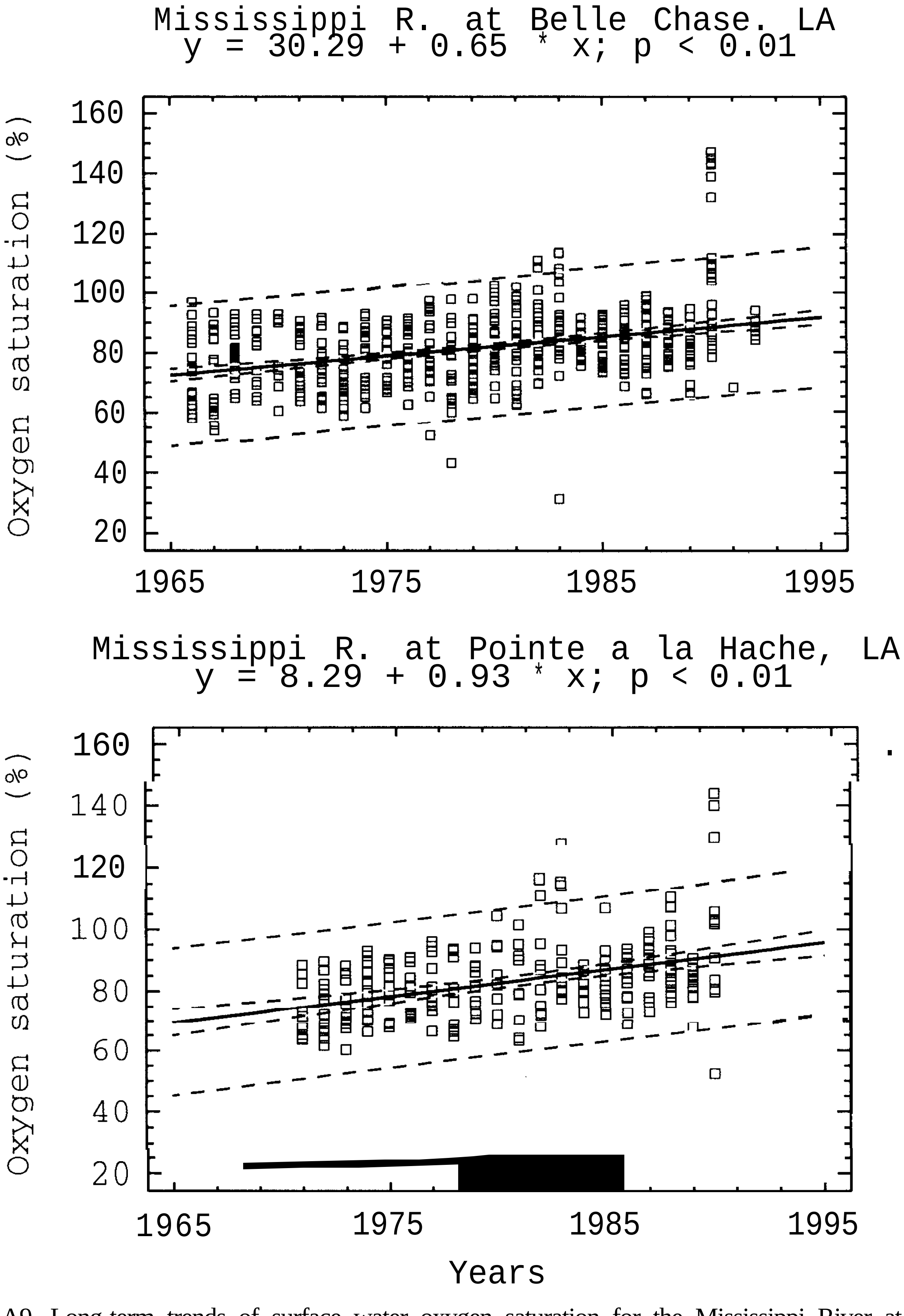


Figure A9. Long-term trends of surface water oxygen saturation for the Mississippi River at Belle Chase and at Pointe a la Hache.

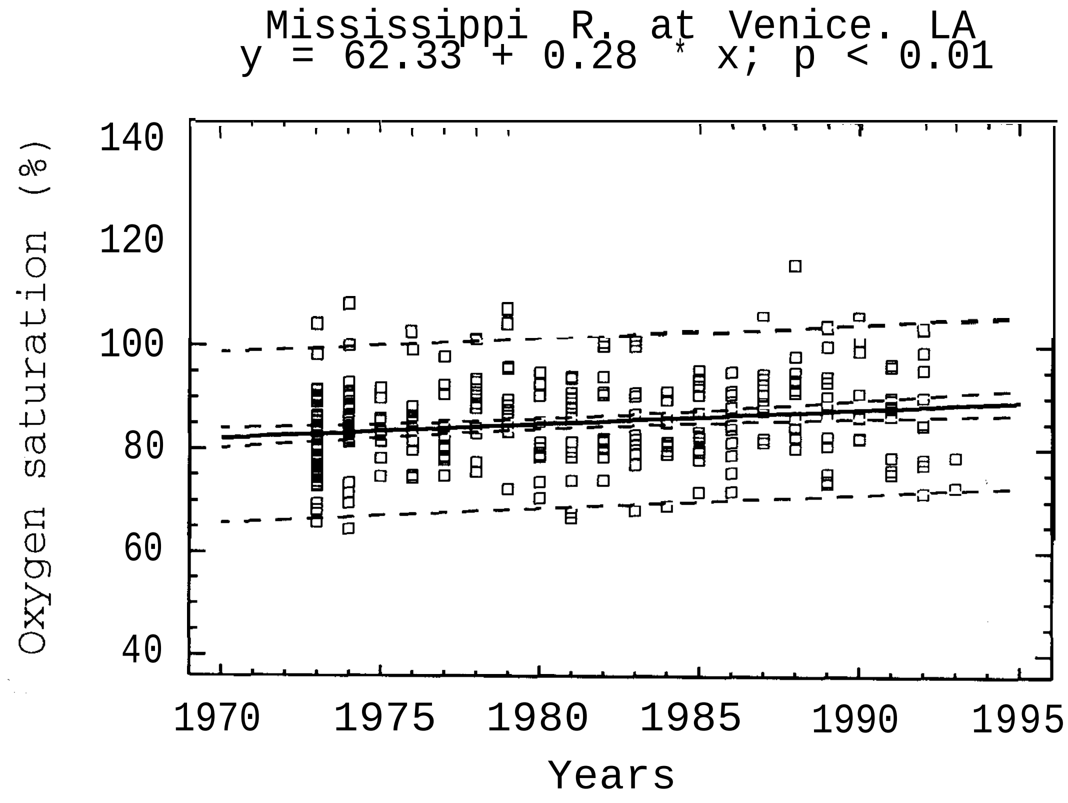


Figure Al 0. Long-term trends of water oxygen saturation for the Mississippi River at Venice.

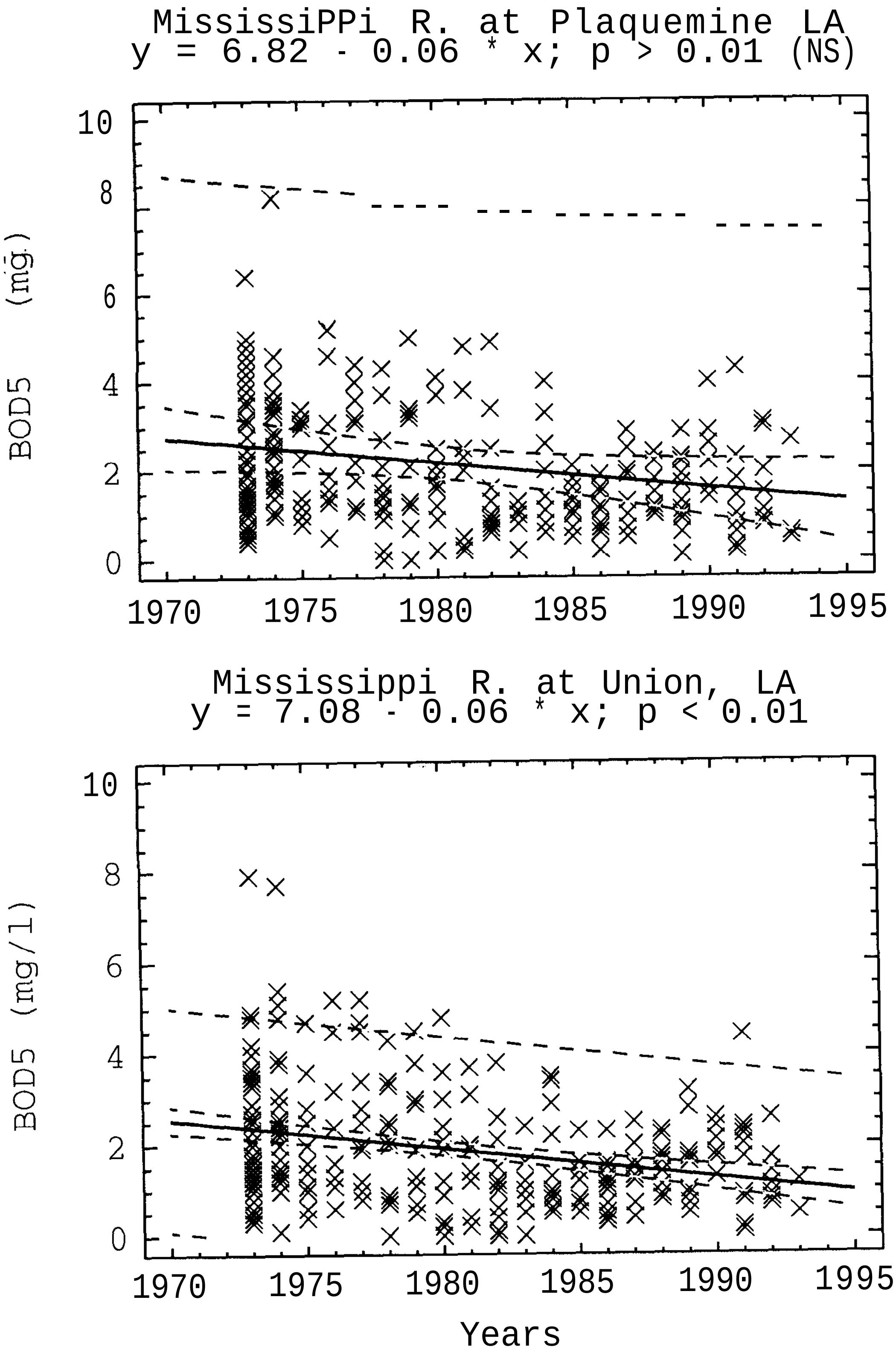


Figure Al 1. Long-term trends of surface water oxygen saturation for stations in the Mississippi River at Plaquemine and at Union.

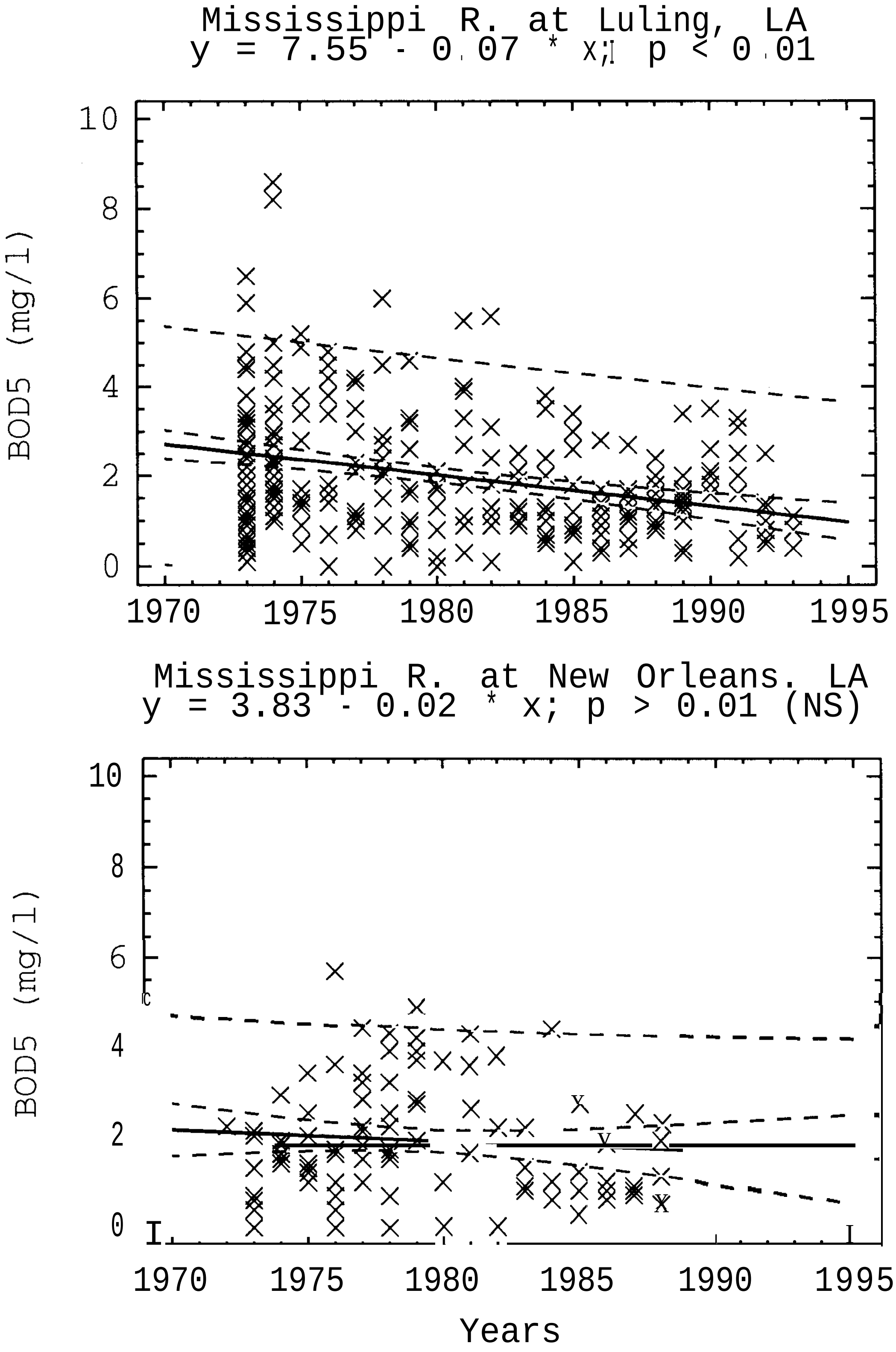


Figure A12. Long-term trends of surface water oxygen saturation for stations in the Mississippi River at Luling and at New Orleans.

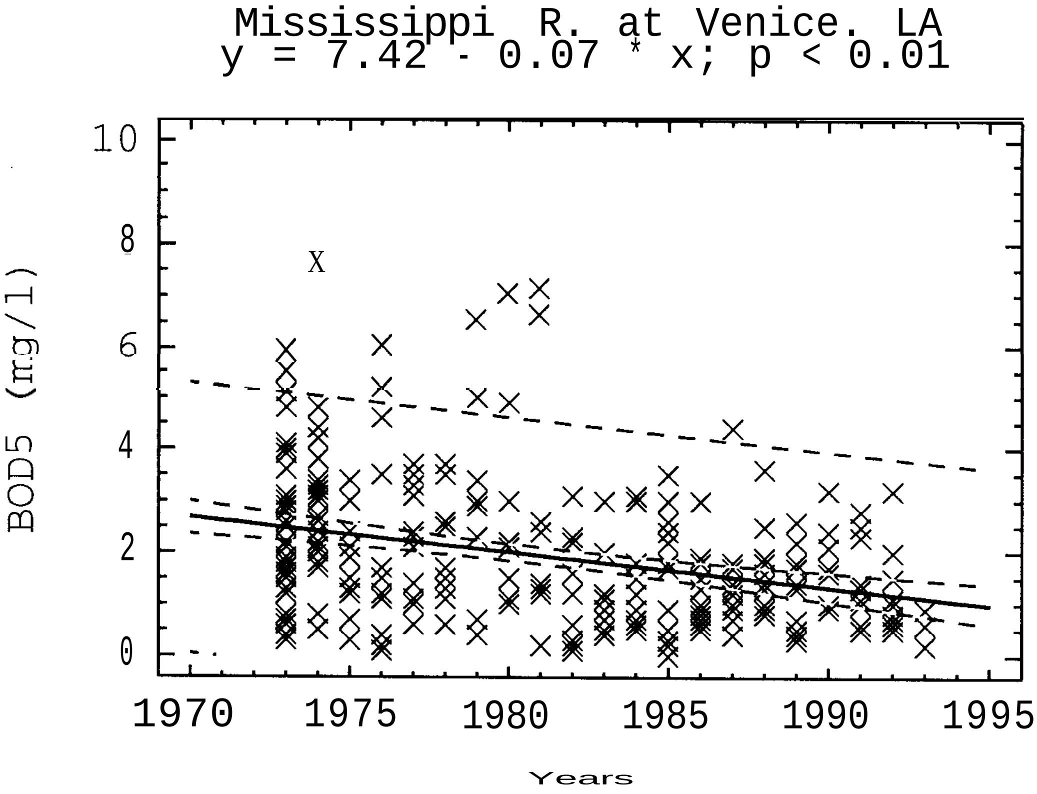


Figure Al 3. Long-term trends of surface water oxygen saturation for stations in the Mississippi River at Venice.

Appendix B

Impacts of Toxic and Noxious Phytoplankton in the BTNEP Area

Table B1. Documentation of potential impacts of toxic and noxious phytoplankton found in the BTNEP area. See table 15 for explanations of abbreviations and alternate names of taxa. Toxicity is highly variable between strains, even in well known highly toxic species. Part of the difficulty is the necessity of relating toxicity of species in mixed natural populations with impacts, often using data collected after the event. Many species cannot be cultured and tested directly. Only those references documenting an impact are included in this table (i.e., references citing no impact are not included), but where there is some doubt about the link between the organism and the impact, a question mark is placed next to the impact. For the taxa with well documented impacts, only review papers are cited. Cyst formation, referring to any resting stage, is not well documented for most species. References are included for the same genus if there are no data for the same species, as indicated. Cyst formation is assumed if vegetative cells can be cultured from sediment originally lacking vegetative cells.

Taxon	Potential Impact	Reference	Cyst	Reference
Alexandrium monilatum	Ichthyotoxic	Connell & Cross 1950	Yes	Walker & Steidinger 1979
		Howell 1953		
		Gates & Wilson 1960		
		Ray & Aldrich 1967		
		Aldrich et al. 1967		
		Sievers 1969		
		Wardle et al. 1975		
	Toxic to invertebrates	Sievers 1969		
		Wardle et al. 1975		
	Toxin lyses mammalian red blood cells	Bass et al. 1983		
	Stop oyster feeding	Ray & Aldrich 1967		
	Water discoloration	Connell & Cross 1950		
		Howell 1953		
		Wardle et al. 1975		
		Perry et al. 1979		
Anabaena flos-aquae	Mammalian neurotoxins	Sivonen et al. 1990		
		Rapala et al. 1993		

Table B1. Continued.

Taxon	Potential Impact	Reference	Cyst	Reference
Anabaena flos-aquae (cont.)	Mammalian hepatoxins	Sivonen et al. 1990		
Ceratium furca	DSP?	Horstman (in Shumway 1990)		
	Water discoloration	Blasco 1975		
		Horstman 1981		
		Wong 1989		
		Pitcher 1993		
		Soucek & Marshall 1993		
		Yuzao et al. 1993		
Ceratium fusus	Larval oyster mortality?	Cardwell et al. 1979		
Ceratium tripos	Shellfish death due to hypoxia	Mahoney & Steimle 1979		
Dinophysis caudata[1]	DSP? and produces tumor promotor[2]?	Kurunasagar et al. 1989	Yes[3]	Bardouil et al. 1991
		Maranda & Shimizu 1987		Cannon 1993
		Freudenthal & Jijima 1988		Moita & Sampayo 1993
		Lee et al. 1989		MacLachlan 1993
		Cembella 1990		
		Masselin et al. 1992		
		Sedmark & Fanuko 1992		
		Boni et al. 1993		
		Loggia et al. 1993		
Dinophysis ovum[1]	DSP? and produces tumor promotor[2]?	Maranda & Shimizu 1987	Yes	Cannon 1993
		Freudenthal & Jijima 1988		
		Suganuma et al. 1988		
		Kurunasagar et al. 1989		
		Lee et al. 1989		
		Cembella 1990		
		Masselin et al. 1992		

Sedmark & Fanuko 1992

Table B1. Continued.

Taxon	**Potential Impact**	**Reference**	**Cyst**	**Reference**
Dinophysis ovum (cont.)		Boni et al. 1993		
		Loggia et al. 1993		
Gonyaulax polygramma	Fish kills due to hypoxia	Grindley & Taylor 1962	Yes[3]	Matsuoka et al. 1989
		Ferraz-Reys et al. 1979		
	Shellfish death due to hypoxia	Grindley & Taylor 1962		
		Ferraz-Reys et al. 1979		
	Water discoloration	Grindley & Taylor 1962		
		Bodeanu & Usurelu 1979		
		Ferraz-Reyes et al. 1979		
		Subramanian 1985		
		Hallegraeff 1992		
		Tseng et al. 1993		
Gonyaulax[4] spp.	PSP?	Shumway 1990	Yes?	Matsuoka et al. 1989
		Hallegraeff 1993		
		Steidinger 1993		
Gymnodinium breve	NSP	Steidinger 1993	Yes	Walker 1982
	Water discoloration	Steidinger & Joyce 1973		
Gymnodinium sanguineum	Associated with fish kills	Harper and Guillen 1989	Yes	Voltolina 1993
		Texas Parks and Wildlife Department 1994		Robichaux & Dortch in press
	Associated with oyster mortality?	Woelke 1961		Steidinger et al. in press
		Cardwell et al. 1979		
		Bricelj et al. 1992		
	Water discoloration	Blasco 1975, 1979		

		Horstman 1981		

Table B1. Continued.

Taxon	Potential Impact	Reference	Cyst	Reference
Gymnodinium sanguineum (cont.)		Robinson & Brown 1983		
		Harper & Guillen 1989		
		Mendez 1993		
		Soucek & Marshall 1993		
		Texas Parks and Wildlife Department in press		
Heterosigma akashiwo	Ichthyotoxic	Black et al. 1991	Yes	Tomas 1978
		MacKenzie 1991		Imai 1993
		Chang et al. 1993		
		Clemente & Lembeye 1993		
		Honjo 1993		
		Yang et al. 1993		
	Water discoloration	Wang 1991		
		Hallegraeff 1992		
		Honjo 1993		
		Park et al. 1989		
		Yuzao et al. 1993		
		Taylor et al. 1994		
Lingulodinium polyedra	PSP?	Schradie & Bliss 1962	Yes	Matsuoka et al. 1989
		Bruno et al. 1990		Marsovic 1989
	Inability to support larval anchovy growth	Huntley 1989		
	Water discoloration	Blasco 1975, 1977		

Noctiluca spp.	Fish kills	Legovic et al. 1991 Adnan 1989 Devassy 1989

Table B1. Continued.

Taxon	Potential Impact	Reference	Cyst	Reference
Noctiluca spp. (cont.)	Damage to prawn mariculture	Chen & Gu 1993		
	due to hypoxia	Suvapepun 1989		
	Water discoloration	Hortsmann 1981		
		Adnan 1989		
		Devassy 1989		
		Wong 1989		
		Uhlig & Sahling 1990		
		Garate-Lizarraga 1991		
		Hallegraeff 1992		
		Chen & Gu 1993		
		Mendez 1993		
		Soucek & Marshall 1993		
Oscillatoria spp.	mammalian neurotoxin	Hawser et al. 1991		
		Endean et al. 1993		
	toxic to calanoid copepods	Hawser et al. 1992		
	water discoloration	Eleutrius 1981		
		Adnan 1989		
		MacLean 1989		
		Wong 1989		
		Hallegraeff 1992		
		Yuzao et al. 1993		
Prorocentrum compressum	DSP? and tumor promoter[2]?	Freudenthal & Jijina 1988	Yes[3]	Bhaud et al. 1988
				Faust 1990, 1993
				Cannon 1993
Prorocentrum gracile	Water discoloration	Tseng et al. 1993	Yes	Cannon 1993

Table B1. Continued.

Taxon	Potential Impact	Reference	Cyst	Reference
Prorocentrum micans	PSP?	Pinto & Silva 1956	Yes	Bhaud et al. 1988
		Horstman 1981		Cannon 1993
	Shellfish mortality	Horstman 1981		
	Water Discoloration	Blasco 1975		
		Hortsman 1981		
		Park et al. 1989		
		Wong 1989		
		Munoz et al. 1991		
		Pitcher et al. 1993		
		Yuzao et al. 1993		
Prorocentrum minimum	DSP? and tumor promoter[2]?	Tangen 1983	Yes[3]	Faust 1990, 1993
	VP	Nakazima 1968		Bhaud et al. 1988
		Okachi & Imatomi 1979		Caonon 1993
	Shellfish mortality	Dodge 1993		
		Luckenback et al. 1993		
		Wikfors & Smolowitz 1993		
		Wikfors et al. 1993		
	Fish kills	Rabbani et al. 1990		
		Ho & Hodgkiss 1993		
	Water discoloration	Perry & McLelland 1981		
		Maples et al. 1981		
		Kimor et al. 1985		
		Wong 1989		
		Jochem 1990		
		Rabani et al. 1990		
		Dodge 1993		

Table B1. Continued.

Taxon	Potential Impact	Reference	Cyst	Reference
Prorocentrum minimum (cont.)		Edler et al. 1993		
Pseudo-nitzschia spp.	ASP	Fryxell et al. 1990		
		Villac et al. 1993 a, b		
		Todd 1993		
Scrippsiella troichoidea	Fish kills due to hypoxia	Maclean 1989	Yes	Matsuoka et al. 1989
		Hallegraeff 1992		
	Water discoloration	Hortsman 1981		
		Clement & Guzman 1989		
		Maclean 1989		
		Hallegraeff 1992		

[1]Many *Dinophysis* implicated in okadaic acid production, the toxin which causes DSP and promotes tumorgenesis. Because no species can be grown in culture, direct proof requires monospecific blooms. Neither *D. caudata* nor *D. ovum* have bloomed elsewhere, so they are untested, although Kurunasager et al. 1989 implicates *D. caudata*.

[2]Okadaic acid promotes tumor formation in laboratory studies (Suganuma et al. 1988), but has not been linked to tumors in humans.

[3]Same genus but not same species.

[4]Many causing PSP renamed *Alexandrium.*

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Appendix C

Fecal Coliform Indicators and *Vibrio* Data for BTNEP

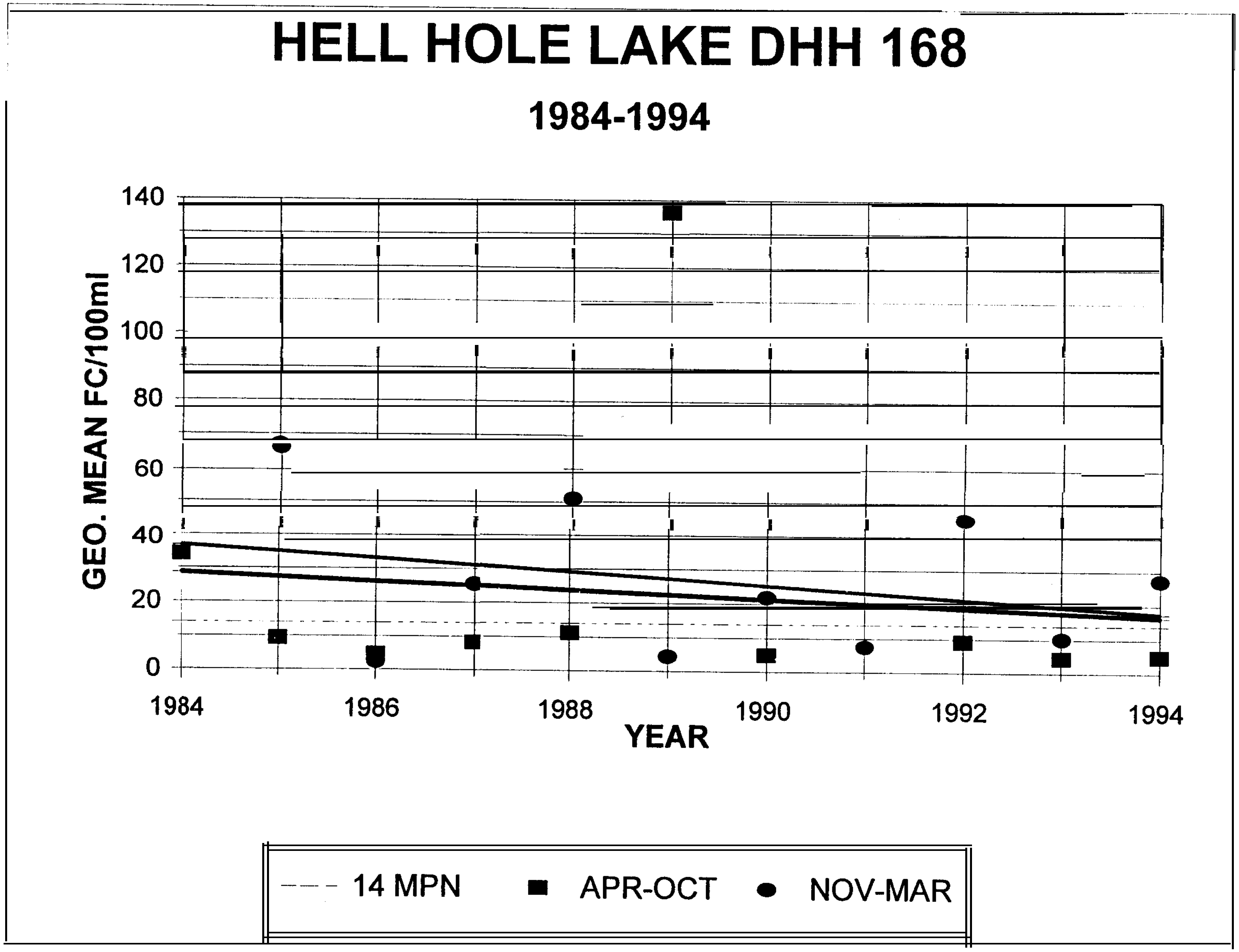

Figure C1. Regression analyses of geometric mean MPN FC/100 ml by year (two seasons) for Hell Hole Lake.

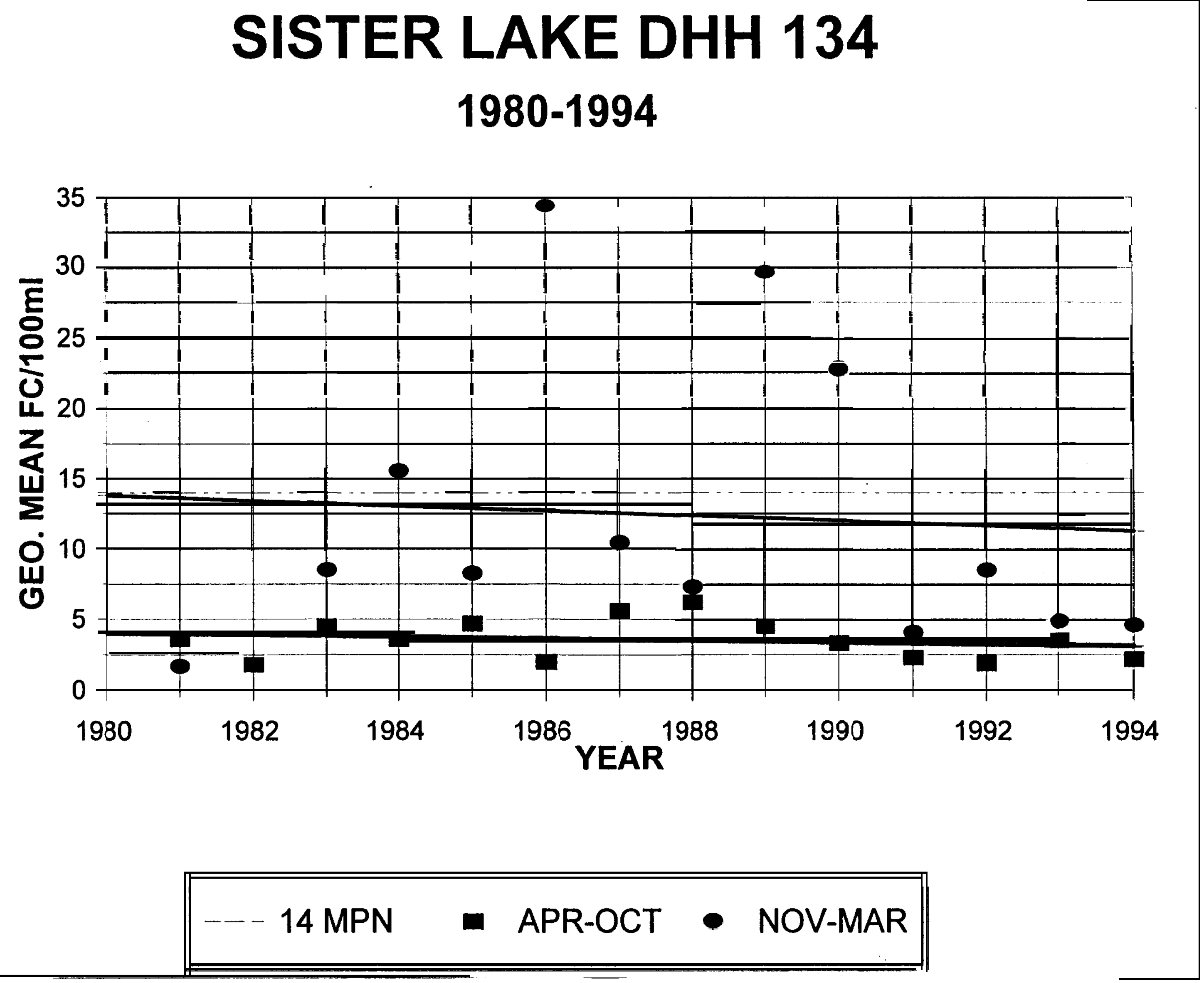

Figure C2. Regression analyses of geometric mean MPN FC/100 ml by year (two seasons) for Sister Lake.

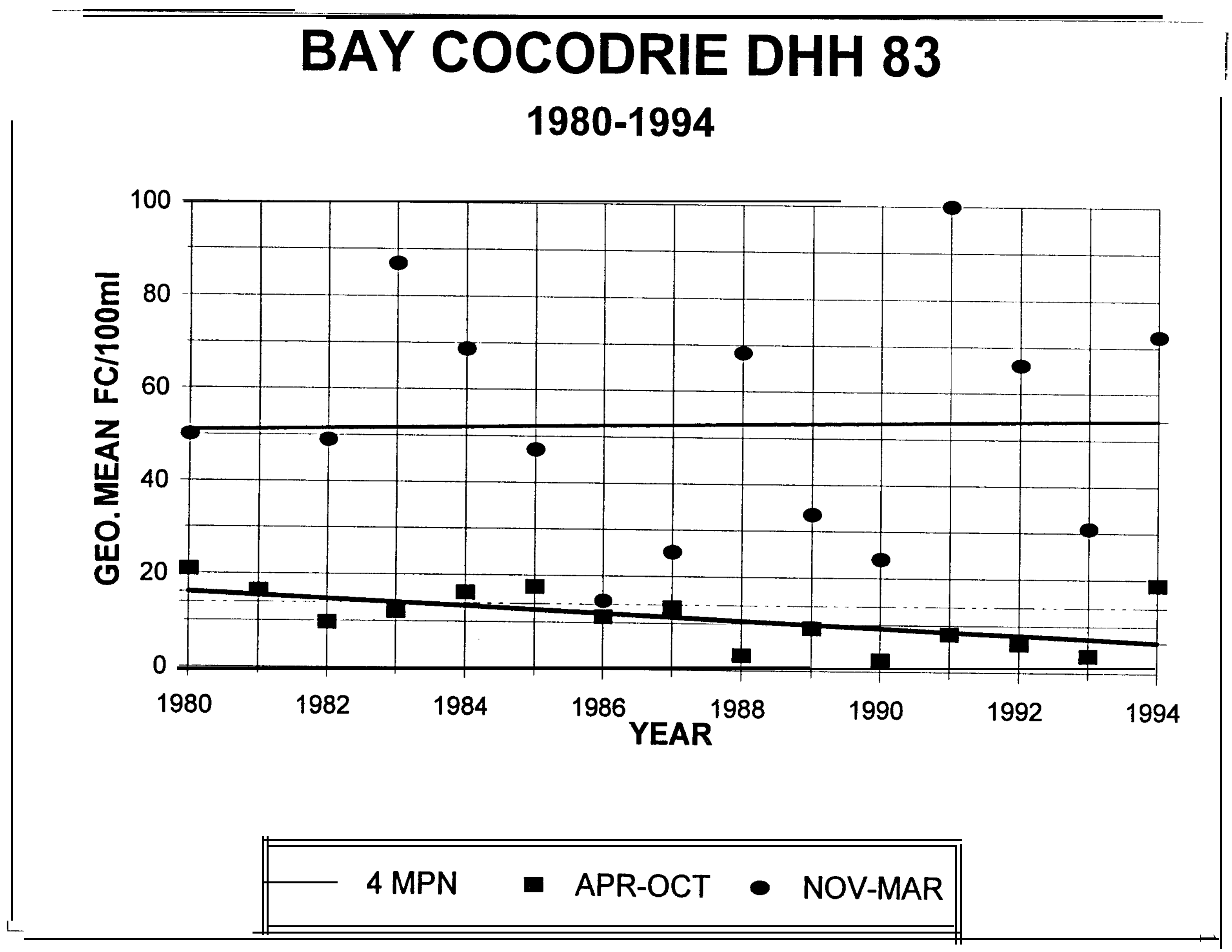

Figure C3. Regression analyses of geometric mean MPN FC/100 ml by year (two seasons) for Bay Cocodrie.

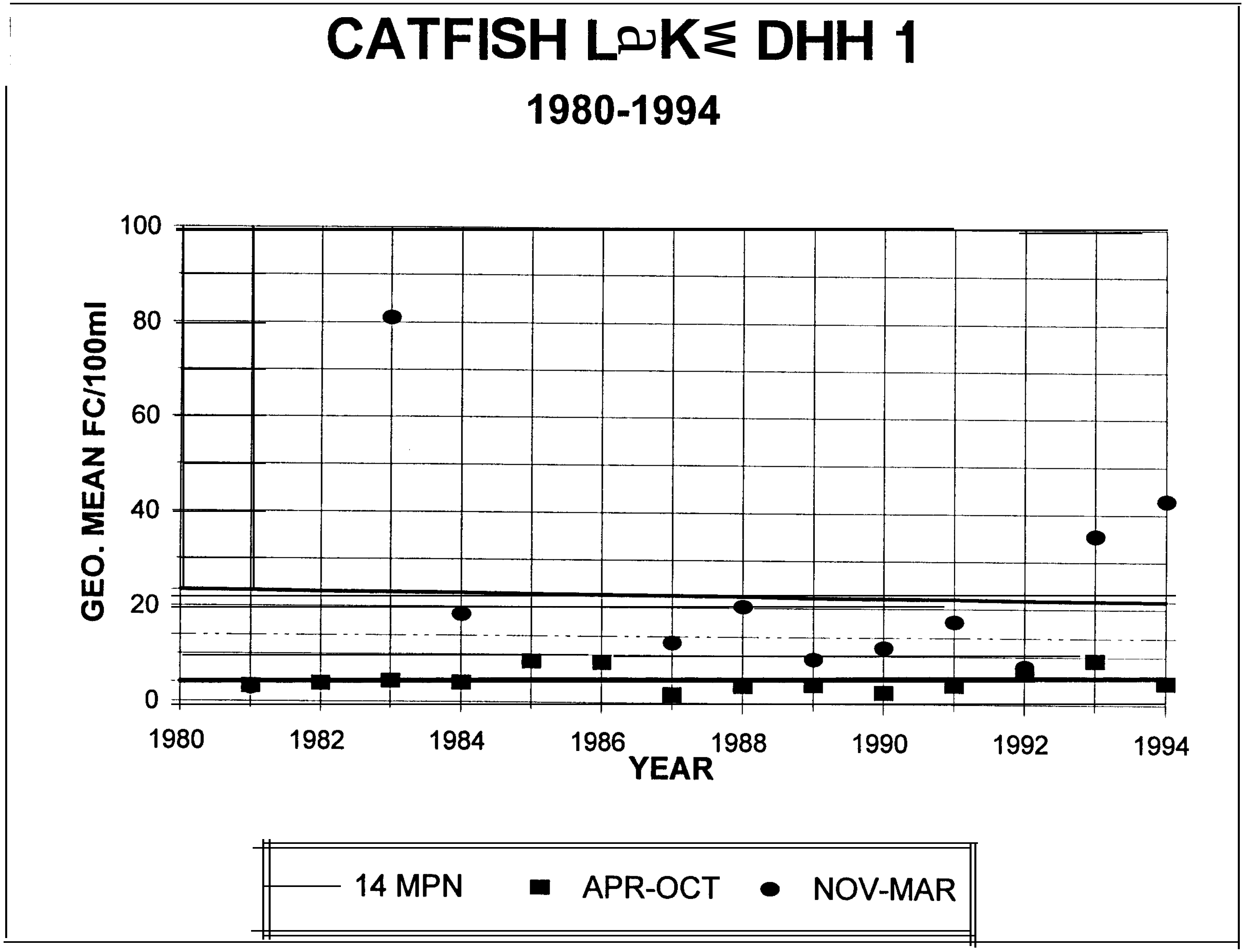

Figure C4. Regression analyses of geometric mean MPN FC/100 ml by year (two seasons) for Catfish Lake.

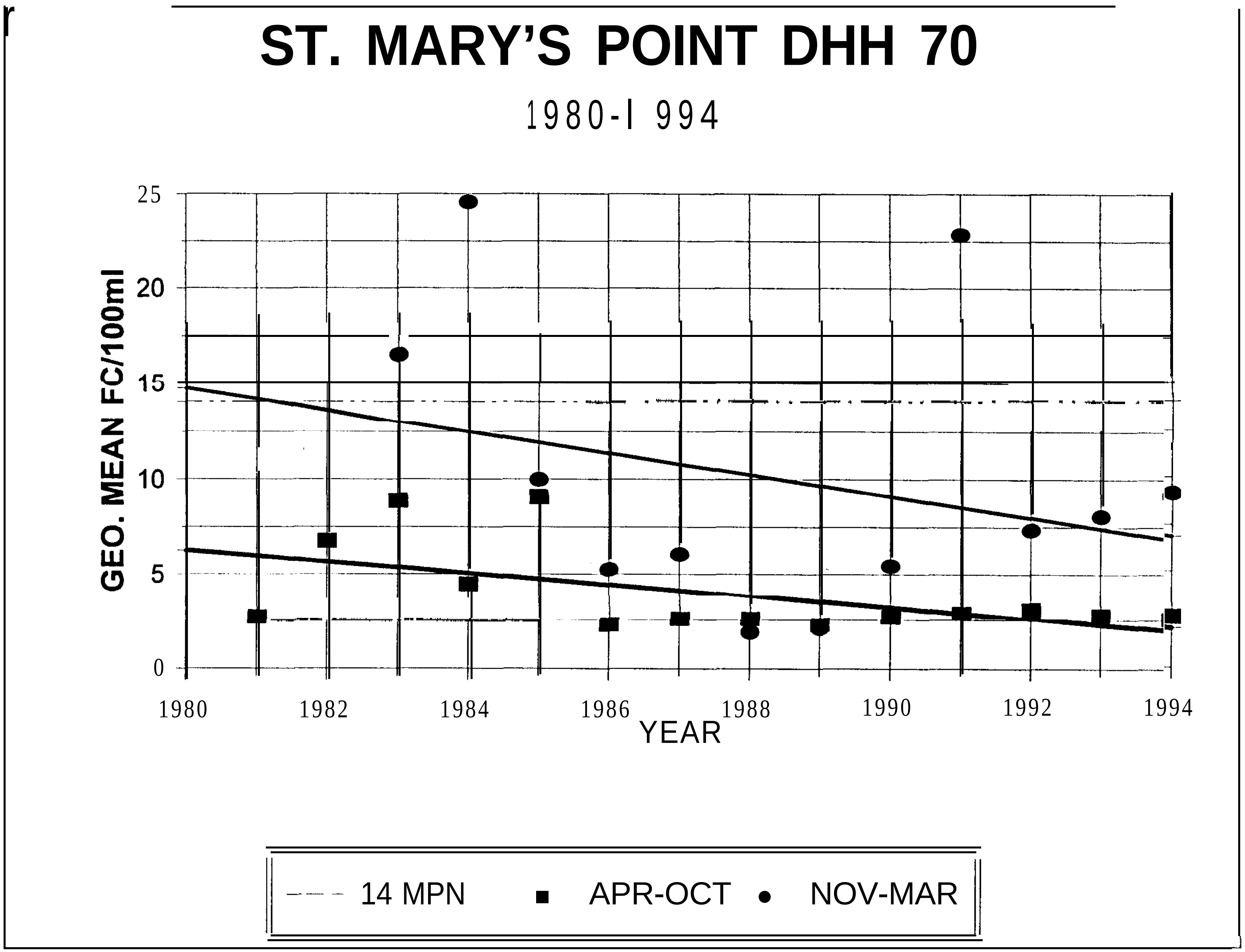

Figure C5. Regression analyses of geometric mean MPN FC/lOO ml by year (two seasons) for St. Mary's Point.

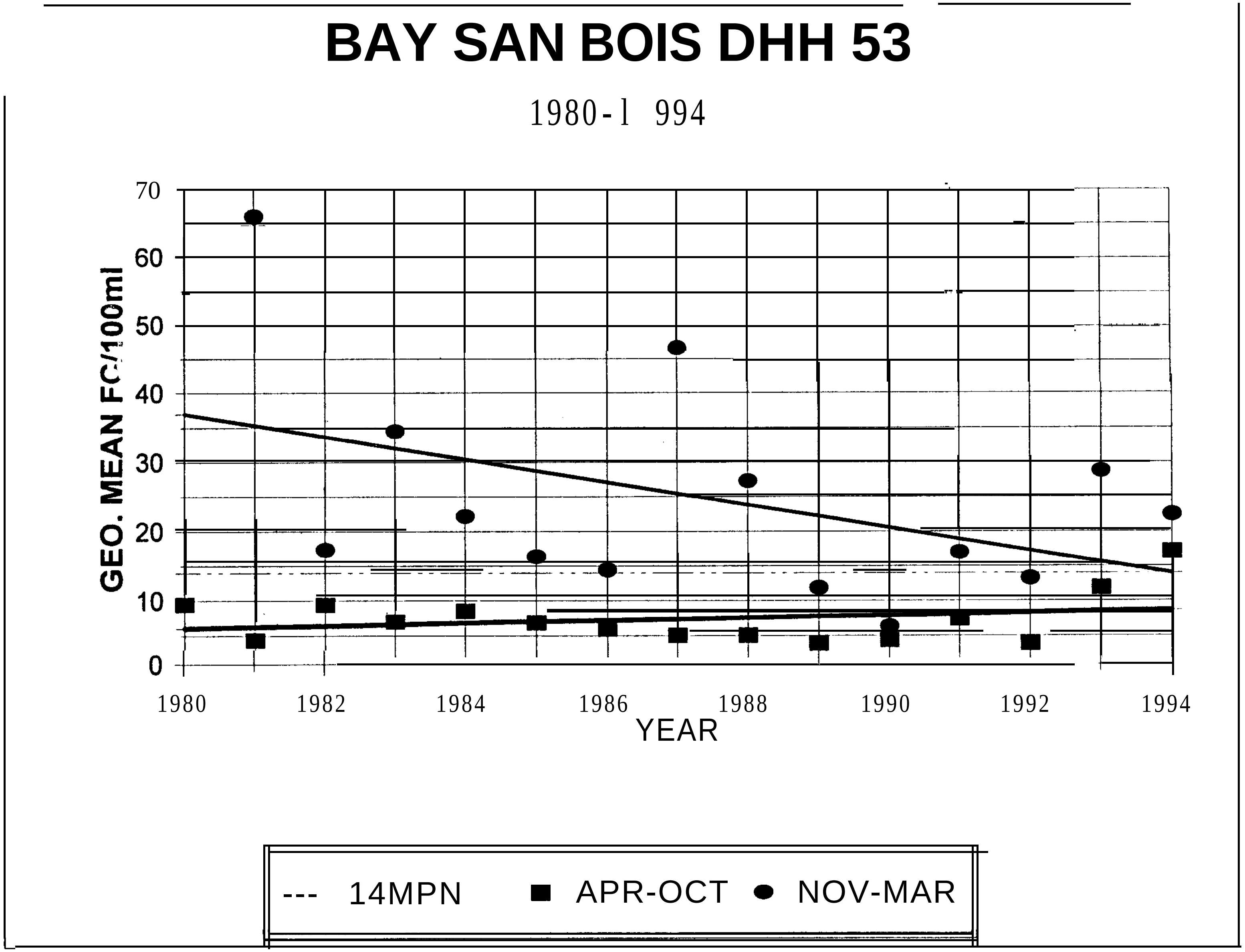

Figure C6. Regression analyses of geometric mean MPN FC/100 ml by year (two seasons) for Bay San Bois.

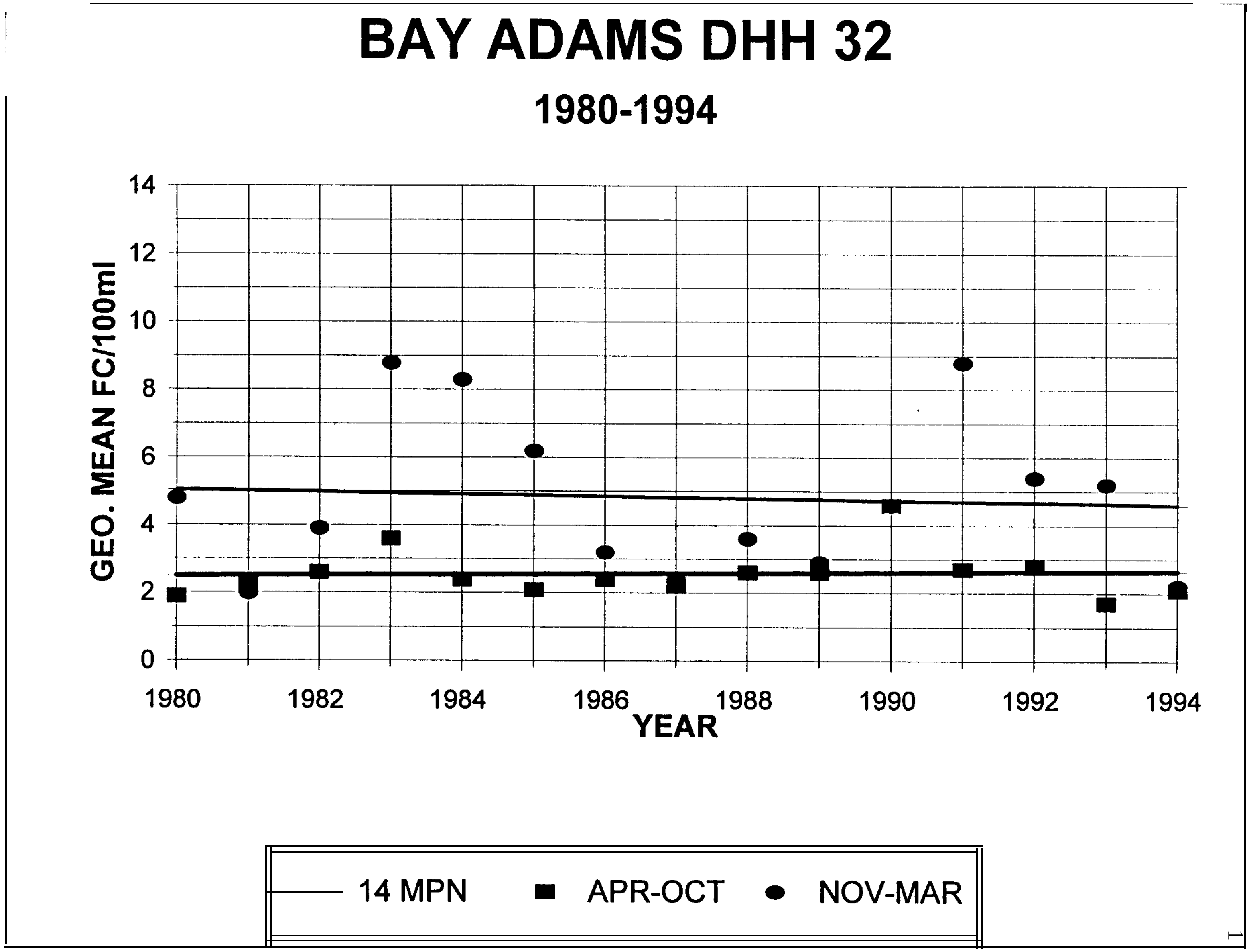

Figure C7. Regression analyses of geometric mean MPN FC/100 ml by year (two seasons) for Bay Adams.

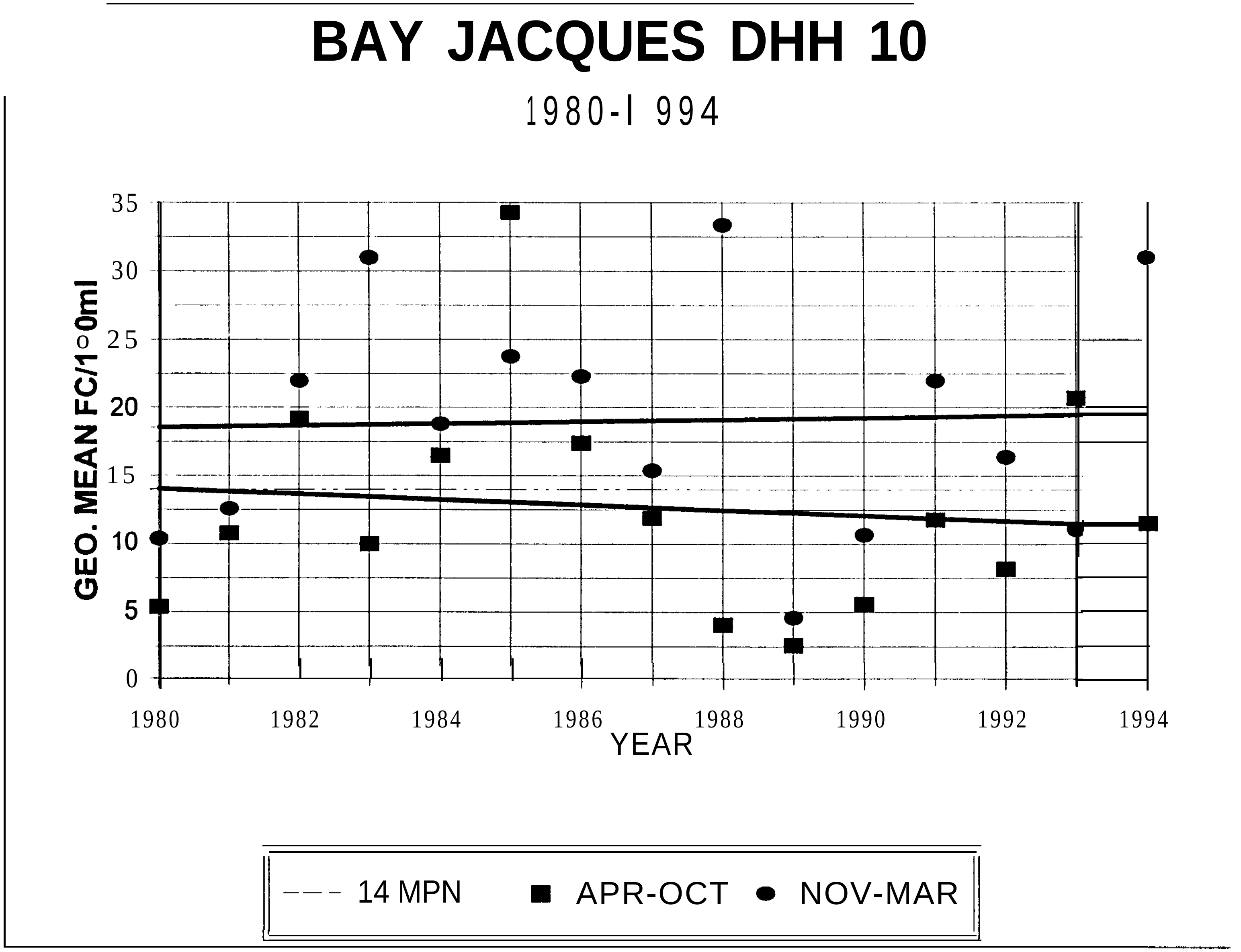

Figure C8. Regression analyses of geometric mean MPN FC/lOO ml by year (two seasons) for Bay Jacques.

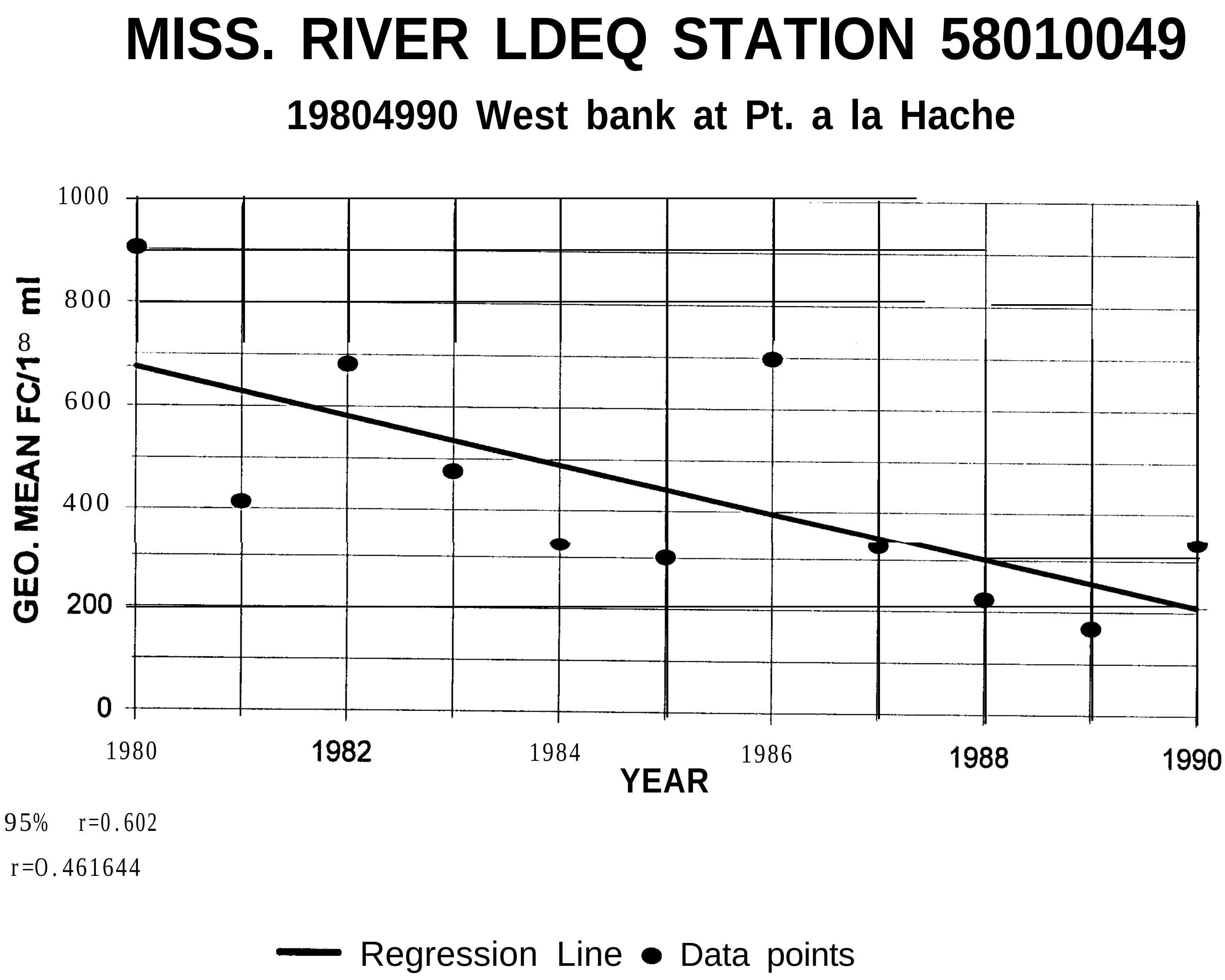

Figure C9. Regression analyses of geometric mean MPN FC/100 ml by year for the west bank of the Mississippi River at Pointe a la Hache.

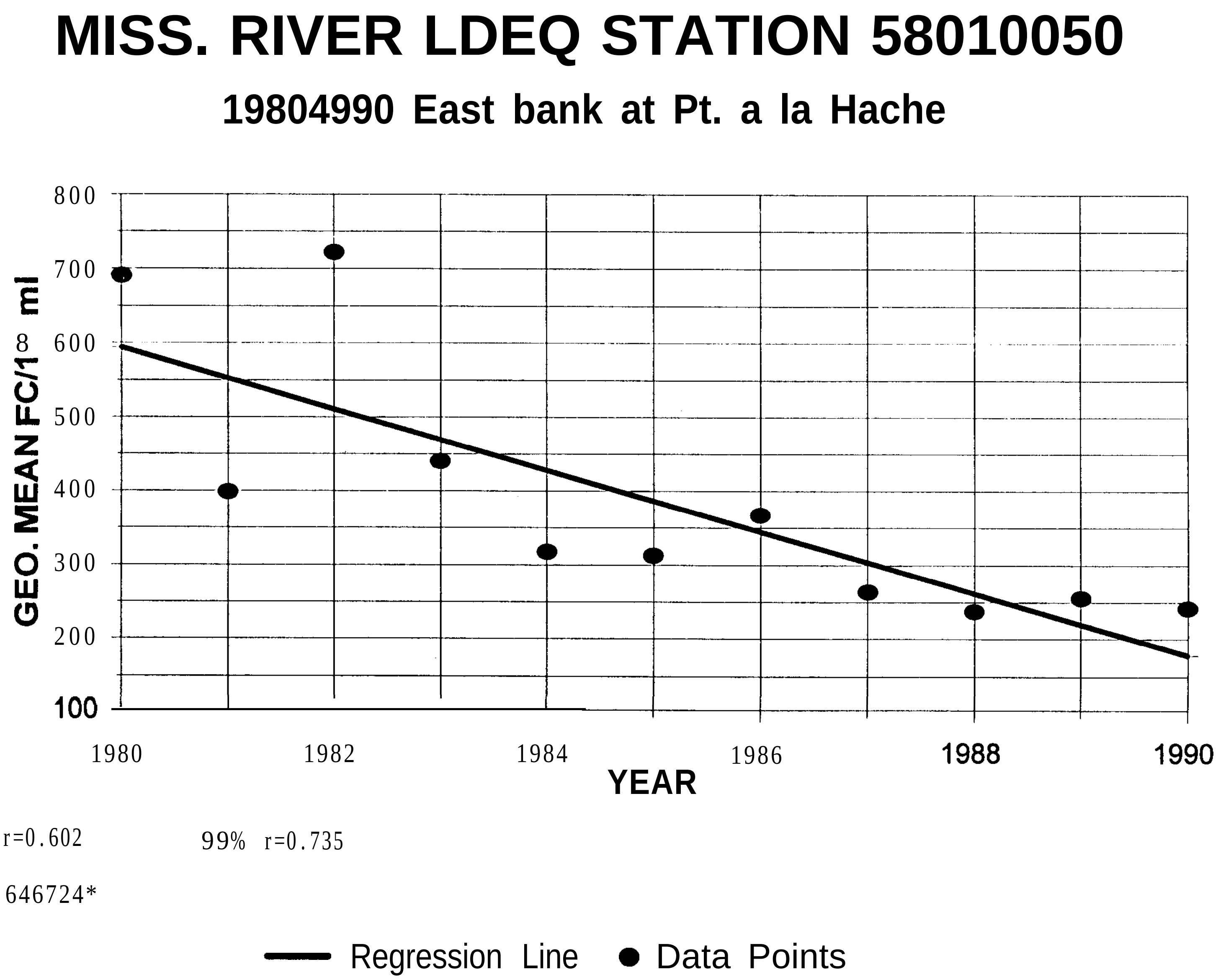

Figure C 10. Regression analyses of geometric mean MPN FC/l 00 ml by year for the east bank of the Mississippi River at Pointe a la Hache.

Table C 1. Vibrio data for the Barataria and Terrebonne estuaries (from LDHH, OPH, Dept. of Epidemiology, S. Wilson, pers. comm, March 24, 1995).

1980

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
VULN	WI	Y	Y	Y	Y	Y	S
VULN	PS	Y	N	U	*	*	D
VULN	WI	Y	Y	N	N	N	S
VULN	WI	Y	Y	U	*	*	S

1981

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
VULN	GI	Y	U	Y	Y	Y	S
VULN	PS	Y	N	Y	Y	Y	D
VULN	PS	Y	N	U	U	U	S
VULN	WI	Y	Y	Y	N	N	S
VULN	WI	N	Y	N	N	N	S
VULN	PS	Y	Y	U	*	*	D
VULN	PS	Y	N	U	*	*	D
VULN	PS	Y	N	Y	Y	Y	D

1982

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
HOLL	PS	Y	U	U	*	*	S
VULN	PS	Y	N	Y	Y	Y	D
VULN	PS	Y	N	U	*	*	S

1983

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
NO01	WI	N	Y	U	*	*	U
VULN	PS	Y	*	Y	Y	Y	D

1984

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
VULN	WI	U	Y	U	*	*	S

1985

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
PARA/VULN	**WI**	**Y**	**Y**	**N**	**N**	N	S

Codes:

DIAGNOSIS	**OUTCOME**
WI = Wound Infection	S = Satisfactory
PS = Primary Septicemia	D = Death
GI = Gastrointestinal	U = Unknown
UK = Unknown	
OT = Other	

Table C 1. Continued.

1986 TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
NO01	GI	U	U	U	*	*	S
NO01	GI	U	U	Y	Y	Y	S
PARA	WI	Y	Y	U	*	*	S
ALGI/PARA	WI	N	N	N	N	N	S
PARA	GI	U	U	Y	Y	Y	S
PARA	GI	Y	U	U	*	*	U
ALGI	WI	U	U	U	*	*	U
ALGI/PARA	GI	U	N	Y	Y	Y	S
MIMI	GI	Y	N	N	N	N	U
VULN	PS	Y	U	U	*	*	D

1987 TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
NO01	PS	U	U	U	*	*	U
NO01	GI	U	U	Y	Y	Y	U
NO01	PS	U	U	U	*	*	U
NO01	GI	U	U	U	*	*	U
NO01	GI	N	U	Y	Y	Y	U
PARA	UK	U	U	Y	N	N	U
VULN	WI	N	Y	U	*	*	S

1988 TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
NO01	GI	U	U	U	*	*	U
NO01	WI	Y	U	U	*	*	D
PARA	GI	U	U	U	*	*	U
PARA	GI	N	N	Y	Y	Y	S
HOLL	GI	N	Y	Y	Y	Y	S
HOLL	GI	Y	U	U	*	*	D
ALGI	WI	N	N	Y	Y	N	S
HOLL	GI	N	N	Y	Y	Y	S
MIMI	GI	Y	N	N	N	N	S
PARA	UK	Y	U	U	*	*	D
VULN	WI	Y	U	U	*	*	S
DA/PA/VU	WI	Y	Y	N	N	N	S
VULN	WI	N	Y	N	N	N	S

Codes:

DIAGNOSIS	OUTCOME
WI = Wound Infection	S = Satisfactory
PS = Primary Septicemia	D = Death
GI = Gastrointestinal	U = Unknown
UK = Unknown	
OT = Other	

Table C 1. Continued.

1989

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
PARA	GI	N	N	Y	Y	Y	S
PARA	GI	U	U	Y	Y	U	S
PARA	GI	Y	U	N	N	N	S
NO01	GI	N	N	Y	Y	Y	S
NO01	GI	Y	Y	Y	N	N	S
NO01	GI	Y	N	Y	N	N	S
NO01	GI	N	Y	Y	Y	N	S
NO01	GI	Y	Y	Y	Y	Y	S
MIMI	GI	Y	Y	Y	Y	Y	S
FLUV	GI	Y	N	Y	N	N	S
VULN	WI	Y	Y	U	*	*	S
VULN	PS	Y	N	Y	Y	Y	D
VULN	WI	Y	Y	Y	N	N	S

1990

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
HOLL	UK	N	U	U	*	*	S
NO01	GI	Y	Y	Y	N	N	S
NO01	GI	Y	N	Y	N	N	S
MIMI	*	U	U	U	*	*	*
NO01	GI	N	N	Y	Y	Y	S
PARA	WI	N	Y	Y	N	N	S
NO01	GI	N	Y	Y	U	U	S
PARA	GI	N	N	Y	Y	Y	S
VULN	WI	Y	Y	Y	N	N	S
VULN	WI	Y	Y	N	N	N	S

1991

TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
NO01	GI	Y	Y	Y	N	N	S
NO01	PS	Y	N	U	N	N	S
MIMI	GI	Y	N	Y	N	N	S
PARA	WI	Y	Y	N	*	*	U
MIMI/NOO	GI	Y	N	Y	Y	Y	U
PARA	GI	N	Y	Y	Y	Y	S
PARA	GI	N	Y	Y	Y	Y	U
VULN	PS	Y	Y	N	*	*	S
VULN	WI	Y	Y	U	U	U	S
VULN	PS	Y	Y	Y	N	N	S
VULN	PS	Y	Y	Y	N	N	S
VULN	PS	Y	Y	U	U	U	D

Codes:

DIAGNOSIS

WI = Wound Infection

PS = Primary Septicemia

GI = Gastrointestinal

GI = Gastrointestinal

UK = Unknown

OUTCOME

S = Satisfactory

D = Death

U = Unknown

Table C 1. Continued.

1992 TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
PARA	WI	N	Y	N	*	*	U
VULN	PS	Y	Y	U	*	*	S
PARA	GI	N	N	Y	U	U	U
VULN	PS	Y	N	Y	N	N	S
MIMI	GI	N	N	Y	N	N	S
PARA/VULN	WI	Y	Y	Y	N	N	S
VULN	WI	Y	Y	Y	N	N	S
MIMI	GI	N	N	Y	Y	Y	S
VULN	WI	Y	Y	Y	N	N	S
PARA	OT	N	Y	N	*	*	S
DA/PA/VU	WI	Y	Y	U	*	*	S
PARA	GI	Y	N	N	*	*	S

1993 TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
ALGI	GI	U	N	Y	N	N	U
PARA	WI	N	N	Y	N	N	S
VULN	WI	Y	Y	N	*	*	S
VULN	WI	Y	Y	N	*	*	S
NO01	GI	Y	Y	Y	N	N	S
DAMS	WI	N	Y	Y	U	U	S
MIMI	GI	N	Y	Y	N	N	S
FLUV	GI	Y	Y	Y	Y	Y	S
VULN	WI	Y	Y	N	*	*	S
NO0 1NULN	WI	N	Y	N	*	*	S
FLUV	GI	Y	N	N	*	*	S
NO01	OT	Y	Y	N	*	*	S
VULN	WI	Y	Y	Y	U	U	S
VULN	WI	Y	Y	N	*	*	S
VULN	PS	Y	N	Y	Y	Y	S
MIMI	GI	N	Y	Y	Y	Y	S
HOLL/PARA	GI	Y	Y	Y	Y	Y	S
MIMI	GI	Y	Y	Y	Y	Y	S
PARA	GI	N	N	Y	Y	U	S

Codes:

DIAGNOSIS
WI = Wound Infection
PS = Primary Septicemia
GI = Gastrointestinal
UK = Unknown
OT = Other

OUTCOME
S = Satisfactory
D = Death
U = Unknown

Table C 1. Continued.

1994 TYPE	DIAGNOSIS	UNDERLYING	EXPOSURE	SHELLFISH	OYSTERS	RAW	OUTCOME
PARA	GI	N	N	Y	Y	N	S
VULN	WI	N	Y	N	*	*	S
HOLL	GI	Y	Y	Y	Y	Y	S
PARA	OT	Y	Y	N	*	*	S
NO01	GI	N	Y	Y	N	N	S
FLUV	OT	U	U	U	*	*	U
PARA	WI	N	Y	N	*	*	S
VULN	WI	Y	Y	N	*	*	S
VULN	WI	Y	Y	N	*	*	S
AL/DA/PA	WI	Y	Y	N	*	*	S
NO01	GI	Y	Y	Y	Y	Y	S
PARA	GI	Y	U	U	*	*	S
PARA	WI	N	Y	N	*	*	S
NO01	OT	Y	N	Y	Y	Y	S
FLUV	GI	Y	N	U	*	*	S
PARA	WI	U	Y	Y	Y	Y	S
VULN	PS	Y	U	U	*	*	D
PARA	WI	Y	Y	Y	Y	Y	S
PARA	GI	U	U	U	*	*	S

Codes:

DIAGNOSIS	OUTCOME
WI = Wound Infection	S = Satisfactory
PS = Primary Septicemia	D = Death
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