Summer 8-2009

Factors Influencing the Environmental Quality of the Bay of Saint Louis, Mississippi and Implications for Evolving Coastal Management Policies

Pradnya Ankush Sawant
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The University of Southern Mississippi

FACTORS INFLUENCING THE ENVIRONMENTAL QUALITY OF THE BAY OF SAINT LOUIS, MISSISSIPPI AND IMPLICATIONS FOR EVOLVING COASTAL MANAGEMENT POLICIES

by

Pradnya Ankush Sawant

A Dissertation
Submitted to the Graduate Studies Office
of The University of Southern Mississippi
in Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy

Approved:

[redacted]

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ABSTRACT

FACTORS INFLUENCING THE ENVIRONMENTAL QUALITY OF THE BAY OF SAINT LOUIS, MISSISSIPPI AND IMPLICATIONS FOR EVOLVING COASTAL MANAGEMENT POLICIES

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The Bay of St. Louis, MS is a small northern Gulf of Mexico estuary that has been identified by the Mississippi Department of Environmental Quality (MDEQ) as an impaired waterbody for its designated uses, mainly due to the presence of pollutant pathogens. A systematic study of this estuary was important to understand the behavior and responses of the bay to several natural and anthropogenic forcing factors. A 14-month long study (bimonthly sampling) to evaluate its environmental quality was undertaken from April 2003 to May 2004. Environmental quality was defined as "the health of an ecosystem characterized in terms of water clarity, ability to support aquatic life, nutrient concentrations, and phytoplankton biomass."

Water column temperature, salinity, pH, DO, and turbidity were measured. Surface samples were analyzed for dissolved inorganic nutrients (nitrate, nitrite, ammonium, soluble reactive phosphate, and silicate) and chlorophyll *a* concentrations. Weather parameters including air and dew point temperature, relative humidity, PAR, solar radiation, and wind speed were measured. Total precipitation, river discharge, and gage height data were also obtained. Similarly, Land use and Land cover (LULC) data from the watershed of the estuary was also included in this study. Parameters such as concentrations of dissolved inorganic nitrogen (DIN), dissolved inorganic phosphates

(DIP), dissolved oxygen (mg L^{-1}) (DO), chlorophyll *a*, and turbidity were selected as indicators of environmental quality. An Environmental Quality Index (EQI) was developed for this ecosystem using the selected indicators and suitable reference values. Based on the EQI, an environmental quality report card was created as an evaluation tool for this estuary. Spatial interpolation techniques were applied to present the variability in the environmental quality graphically in the form of maps using GIS software. Data from previous studies conducted separately between 1977 and 1998 were compared to understand the factors influencing the longer-term environmental quality of this estuary.

Based on the EQI and the indicator parameters selected for this study, it was found that the environmental quality of the Bay of St. Louis was not "impaired" during the 2003-2004 study period. Precipitation, river discharge, winds, and tides were determined as the primary factors influencing changes in the environmental quality of the bay. Significant spatial and seasonal variability in the environmental quality was observed due to changes in nutrient (DIN and DIP) and sediment loads. The spatial variability was due to increased nutrient concentrations at locations close to point sources than other areas in the bay. River and bayou mouths, wastewater outfalls, and the Mississippi Sound were determined to be the point sources of nutrients to the bay. The Mississippi Sound and Bayou Portage were identified as the major sources of DIP to this estuary. Spatial variability in nutrient concentrations in the bay was also related to the extent of urban and agricultural land uses in the surrounding sub watersheds. Temporal variability in the environmental quality was due to significant differences in nutrient concentrations and turbidity observed during different seasons. Increased nutrient concentrations (particularly DIN) and turbidity were observed during periods of increased rainfall and riverflow (Spring and Summer), whereas, increased DIP concentrations were observed during periods of low riverflow and high wind speeds (Fall).

Variability in the environmental quality of the bay was also seen over different data periods from 1977 to 2004. The environmental quality of the estuary varied over the years in response to shifts in climate patterns/interannual oscillations such as the El Niño Southern Oscillation (ENSO). Although significant changes in the LULC in the watershed (due to increasing population and increases in urban and agricultural uses) were observed, a declining trend in the environmental quality was, however, not observed over the years.

A management plan for the Bay of St. Louis must be designed to include three key components: a comprehensive suite of indicators with suitable target values, effects of changes in activities and developments in the watershed, and effects of natural shifts in climate patterns. It is imperative that management programs are based on sound science, detailed study, and regular monitoring of this dynamic environment. Equally important is participation and coordination between scientists, land managers, coastal managers and user groups. Finally, effective dissemination of information (such as the use of a GIS-based Environmental Quality Report card), communication with all stakeholders, and timely review and improvisation of implemented programs is crucial.

ACKNOWLEDGEMENTS

In completion of this study, I have received help and co-operation from several individuals and institutions. At the outset, I acknowledge with a deep sense of gratitude, the guidance and encouragement I received from my advisor, Professor Donald G. Redalje, under whose able tutelage I had the privilege of doing my research. His valuable comments and expert advice, I believe, have enriched the study.

I am also grateful to all committee members for the interest they have taken in my work. Their searching questions and constructive suggestions helped me to fill up the gaps in my dissertation. I sincerely thank Steven Lohrenz, Vernon Asper, Patrick Biber (USM Gulf Coast Reasearch Laboratory), and George Raber (USM Department of Geography).

I would like to thank all participants of the EEQ project (MDEQ-CIAAP-NOAA) 2003-04, for helping with data collection and laboratory analysis. I am particularly thankful to Egan Rowe, Megan Natter, and Ray Pluhar for excellent team work and more importantly for their undying support and friendship. The field and lab work during this project would not have been possible without the technical help and guidance provided by Merrit Tuel and Erik Quiroz. My thanks also go to Erin Kirk and Rebecca Shilling for their help in the laboratory.

I also wish to acknowledge all personnel at the offices of USGS, EPA-GMPO, MDEQ, MDMR, Hancock County Board of Supervisors for providing timely information and data for this study.

I acknowledge with gratitude the financial assistance (teaching assistantship) that I received from the Department of Marine Science. I am thankful to the other faculty members of the Department, particularly Karen Orcutt, who took keen interest in my work and encouraged me. My thanks are also due to the staff of the Department for their ungrudging assistance whenever I needed it.

I wish to acknowledge the valued support I received from my other colleagues and friends in the Department. I wish to thank Kevin Mahoney, Xuemei, Sergey, Nadya, and Jenny. I will never forget their co-operation and friendship. There are a host of others who remained unnamed but who helped me in small measures in successful completion of this study. My thanks to all of them.

To my family, I thank them for their support and sacrifices. I shall for ever remain indebted to them.

Needless to say even though I have taken help and guidance from a number of persons, the responsibility for facts, analysis and conclusion presented in this study is entirely mine.

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LIST OF ABBREVIATIONS AND SYMBOLS

BSL	Bay of Saint Louis
Chl *a*	Chlorophyll *a*
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic Phosphate
DO	Dissolved Oxygen
ENSO	El Nino Southern Oscillation
EQI	Environmental Quality Index
GCRL	Gulf Coast Research Laboratory
JR	Jourdan River
MDEQ	Mississippi Department of Environmental Quality
NTU	Nephelometric Turbidity Units
PAR	Phostosynthetically Active Radiation
SOI	Southern Oscillation Index
SRWMD	Southern Regional Wastewater Management District
UNCED	United Nations Conference on Environment and Development
USACE	United States Army Corps of Engineers

USEPA .... United States Environmental Protection Agency

WR .... Wolf River

CHAPTER I

INTRODUCTION

Increasing population and rapid urban development along the coastlines of the world have subjected the coastal ecosystems to immense pressure and acute environmental degradation. Owing to these factors, wise coastal zone management practice has become one of the most urgent needs of our time. Developing an appropriate management plan, however, involves a careful study of the various factors that play a major role in altering these environments. It also requires an integrated approach towards maintaining the health of these ecosystems (Cicin-Sain and Knecht 1998). A wise management plan must include the concerns of all the stakeholders and provide protection to the natural resources while promoting sound economic growth. Such an integrated plan needs to be based on good science and a detailed study of the long-term variability observed in these dynamic systems. The plan must also take into account the responses of the coastal environments to changing variables such as land use distribution and changes in climate. Coordination between the monitoring efforts, scientific research and analysis, and management is essential to fulfill the shared objective of sustaining our coastal resources.

The water quality of the Gulf Coast estuaries is affected by pathogens, oxygen depleting substances, metals, and nutrients (USEPA 2001). The sources of most of these stressors are municipal point sources, urban runoff, storm sewers, atmospheric deposition, agriculture, and industrial discharges (USEPA 2001). The physical factors that control the mixing and transport processes in these ecosystems are tides, winds, rainfall, evaporation, fresh-water inflows, and anthropogenic alterations to the estuaries and their watersheds (Solis and Powell 1999). Unprecedented population growth, point and non-point sources of nutrients and contaminants, wetland losses, sediment contamination, poor benthic conditions, and high expression of eutrophication are some of the major common stress factors in the Gulf Coast estuaries (USEPA 2001, 2005).

The Bay of St. Louis in Mississippi is a small estuary in the north-central Gulf of Mexico that has a total area of about 40 km^2 (GCRL 1978) (Figure 1.1). It connects to the Mississippi sound with a narrow passage of about 3 km. It is a shallow (average depth is 1.5 m), vertically-mixed, micro-tidal estuary. Circulation in the bay is mainly driven by wind stress and diurnal tides. The primary sources of fresh water to this estuary are the Jourdan River, flowing into the western part of the bay and the Wolf River on the north-eastern side. Numerous bayous, mainly, Portage, De Lisle, and Mallini bayous, and streams drain directly into the bay. The bay has several point and non-point sources of nutrients, organic matter and sediments. The point sources include the rivers, streams and bayous (e.g., Wolf and Jourdan rivers) as well as four sewage treatment outfalls, a gaming facility, and a titanium dioxide plant. The non-point sources are in the form of old and leaking septic tanks as well as runoff from agricultural and several other anthropogenic activities that occur within the watershed of the bay.

Problem Statement

The designated uses classified for this bay by the State of Mississippi, as per the requirements of the Clean Water Act (1972), are shellfish harvesting and primary contact recreation (MDEQ 2003). However, the Mississippi Department of Environmental Quality (MDEQ) had listed this estuary as “impaired for its designated uses” under the ‘Mississippi Section 303 (d) List of Water bodies’ (MDEQ 2004). The bay was thus

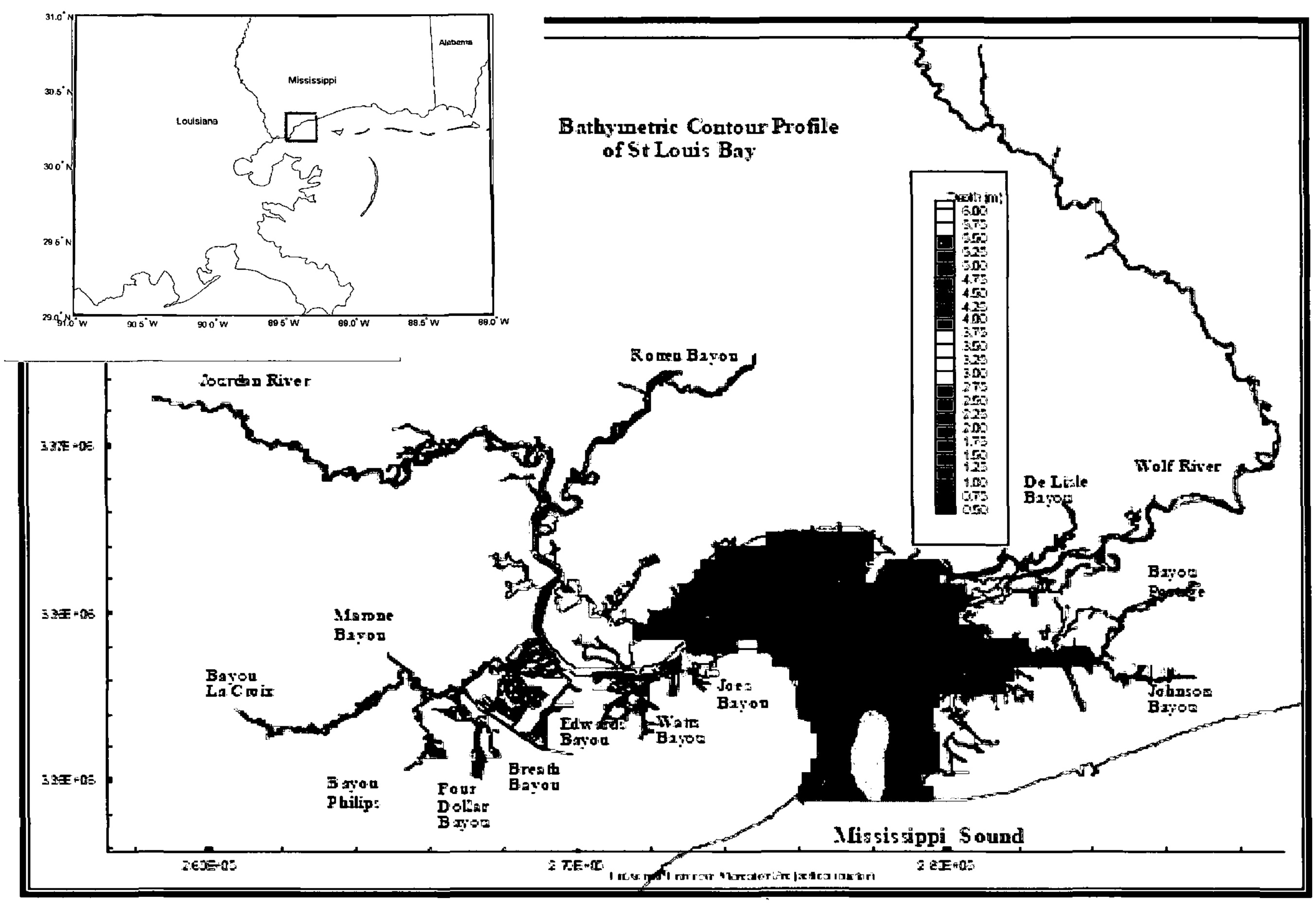

Fig. 1.1 Location of the Bay of St. Louis, MS and Bathymetric Contour Profile. A northern Gulf of Mexico estuary, it is shallow (mean depth =1.5m) and mixed vertically. The fresh water sources are the Jourdan and Wolf rivers and several bayous that empty into the bay. The Mississippi Sound is the main source of saline water to the bay. Profile Figure Source: MDEQ 2001.b. Fecal coliform TMDL for St. Louis Bay, Jourdan River (Phase Two) and Wolf River (Phase Two).

identified due to the presence of pathogens indicated by fecal coliform bacteria as well as due to its proximity to waste water sources, both of which were violations of shellfish harvesting and contact recreation uses (MDEQ 2004). Although the MDEQ later developed Total Maximum Daily Load (TMDL) for pollutant pathogens, certain areas of the bay continued to be listed as impaired for its designated uses and were subjected to further monitoring (MDEQ 2004 and 2005). These segments of the bay were considered to be impaired due to nutrient and organic enrichment and low concentrations of dissolved oxygen. The entire Bay of St. Louis may, however, not reflect poor water quality at all times and at all locations and therefore could not be entirely categorized as impaired. Certain parts of the bay were likely to have acceptable (or better) water quality and could then be allowed for upward classification based on the criteria used for assessing the environmental health of the system (MDEQ 2001.b.). Similarly, significant variability in environmental quality of the bay has been observed in a previous study (Phelps 1999). In order to understand the dynamics of this system and the reasons behind the spatial and temporal variations, it was necessary to explore the various small and large-scale factors affecting the environmental quality of this bay.

Definitions: For the purpose of this study, "Environmental Quality" was defined as the health of an ecosystem characterized in terms of water clarity, ability to support aquatic life, nutrient concentrations, and phytoplankton biomass. "Impaired" was defined as the state of a system, which fails to meet the established criteria and set management objectives.

The total area of the Bay of St. Louis watershed is about 2,117 km^2 and it spans partly or entirely over five different counties of Mississippi (MDEQ 2001.b.). The land use distribution in the drainage areas of the Jourdan and Wolf rivers includes forests, pastures, wetlands, croplands, and barren and urban areas (MDEQ 2001.b.). However, the land use and land cover (LULC) distributions in the watershed have changed over the past thirty years due to urban development and changes in anthropogenic activities (USACE 2003). The land cover distribution has shifted from forest type to croplands or urban types (USACE 2003). Changes in LULC in the watershed can have a significant impact on the water quality of rivers and estuaries due to an increase in the inflow of nutrients and organic matter from the watershed (Dauer et al. 2000; D'Elia et al. 2003; Weller et al. 2003). Changes in LULC in the watershed areas were therefore considered while studying and monitoring the environmental quality of this estuary.

The Gulf of Mexico estuaries are also affected by the shifting climate patterns (Lipp et al. 2001; Scavia et al. 2002; Schmidt et al. 2004). Climate change is mainly characterized in terms of meteorological parameters such as temperature, precipitation, and wind speed as well as by the statistical properties such as means, extremes, or cycles of varying periodicity (Wohl et al. 2000). Different climate patterns and phases occur due to the variability in the ocean-atmosphere interactions. These large-scale oscillations of decadal/multi-decadal time scales are global phenomena. Climate patterns such as the Pacific decadal Oscillation (PDO), North Atlantic Oscillation (NAO), El Niño-Southern Oscillation (ENSO), and Atlantic Multidecadal Oscillation (AMO) are some of the major oceanic phenomena associated with global atmospheric anomalies (Rasmusson and Wallace 1983; Wang et al. 1999; Wohl et al. 2000; Enfield et al. 2001).

Climate change effects in the Gulf of Mexico area are seen in terms of higher than average rainfall (Scavia et al. 2002) and changes in the frequency and severity of winter storms and hurricanes (NAST 2001). The Gulf coast experiences a large number of winter storms and high rainfall during the El Niño conditions (Rasmusson and Wallace 1983). The changes in precipitation, fresh water inflow, and the strength and timing of the river runoff can affect the water quality of the estuaries by causing changes in the delivery of sediments, nutrients, and contaminants (NAST 2001). Lipp et al. (2001.b.), in their study conducted in Tampa Bay, Florida, found considerable changes in the water quality in relation to the ENSO variability. There was a significant increase in the fecal pollution levels during the El Niño winter and fall periods and a significant decrease during strong La Niña winter and fall periods in relation to the normal phase conditions (Lipp et al. 2001.b.).

In order to identify the impacts of the different climate patterns on the environmental quality of the Bay of St. Louis, it was important to study the relationships between the environmental quality parameters that have been used commonly and the different indices of climate variability. A preliminary study conducted in the Bay of St. Louis, MS, demonstrated a significant relationship between the Southern Oscillation Index (SOI) and some of the environmental quality parameters (Redalje et al. 2004). The environmental quality of the entire bay was found to vary significantly due to winter storm events (Redalje et al. 2003). The bay-wide average N:P ratios (moles) approached Redfield ratio only during normal phase years based on the SOI. These ratios were greater significantly than the Redfield ratio during La Niña events and were below the Redfield values significantly during El Niño events (Redalje et al. 2004).

Significance of the Study

The Bay of St. Louis system is important for its ecological and economic purposes. Similarly it has social, recreational, and tourism value. It is used commonly for water sports, recreational fishing, and occasional shellfish harvesting. It is, however, under the influence of a population that is growing rapidly (US Census Bureau 2006). Assessing and monitoring the water quality and the health of this ecosystem is imperative to maintain its ecological, social, and economic value. It was therefore important to evaluate the overall health of this system as well as to study the variability in the behavior of this estuary. Environmental quality was defined as the health of an ecosystem characterized in terms of water clarity, ability to support aquatic life, nutrient concentrations, and phytoplankton biomass. The principal purpose of this study was to obtain an insight on how this ecosystem functioned and responded to changing land use patterns, increased anthropogenic pressures, and shifting climate regimes. It was crucial to base such efforts on rigorous scientific study of this estuary. In order to achieve a better understanding of the uncertainties involved in the processes of this dynamic ecosystem, a study of the longer-term (over years or decades) responses of the bay to the natural and anthropogenic changes was essential. Understanding the trends in the responses of the bay over a long-term period will be useful to further implement a better monitoring and management program for this estuary. Given the anthropogenic alterations in the watershed and the natural variability in climate, there were a number of concerns regarding the health of the bay and the changes undergone in the past thirty years that remained to be addressed. The following questions were asked in this study:

1. Is the environmental quality of the bay impaired based upon the parameters that define environmental quality in this study? If yes, then to what extent is it impaired?
2. What specific factors play a major role in affecting (improve or deteriorate) the environmental quality of the bay? Are these factors correlated to each other?
3. Are any spatial variations in the environmental quality observed throughout the bay?
4. Has the environmental quality of the bay changed over the years? Are any specific trends or patterns found?
5. What parameters could be used in the future to determine if the bay was impaired or not impaired?
6. What efforts have been taken so far to regulate the environmental health of the bay?
7. What needs to be done to manage the Bay of St. Louis effectively?

Previous Research and Monitoring Programs

Various efforts have been carried out in the past to understand the dynamics of the bay as well as to assess and regulate the environmental quality of this estuary. The Gulf Coast Research Laboratory (GCRL) conducted a short study in the bay in 1973 as part of the cooperative Gulf of Mexico Estuarine Inventory and Study, Mississippi (Christmas 1973). An environmental baseline survey of the bay in 1977-78 was another study conducted by the GCRL. The survey was carried out in order to assess the potential impacts of the DuPont titanium dioxide plant, presently located along the northern shore of the bay (GCRL 1978).

Environmental monitoring projects were conducted by The University of Southern Mississippi, Department of Marine Science (USM/DMS) in 1995-96 and 1997-98 (Redalje and Rasure, unpub. 1995-1996; Phelps 1999; Holterman 2001). The most recent USM/DMS project titled 'Evaluating Environmental Quality for the Bay of St. Louis, MS' (EEQ) was carried out from April 2003 to May 2004 and funded by the NOAA Coastal Impact Assistance Program (CIAP) via the MDEQ (project number MS.23.15).

In addition to the research studies, several monitoring and regulatory programs to protect and manage the environmental health of the bay have reportedly been carried out by the state and federal agencies. The MDEQ completed TMDL allocations of fecal coliform for the Bay of St. Louis, Jourdan River, and Wolf River soon after the bay and the two rivers were listed as impaired due to the presence of pathogens (MDEQ 2001.b.). The effort was undertaken in order to reduce the fecal coliform loads to the bay and improve the water quality, which in turn may allow for upward classification of the bay and include shellfish harvesting as one of its primary designated uses. Certain other segments of the Bay of St. Louis continued to be monitored by the MDEQ (MDEQ 2004 and 2005). Similarly, the bay waters are monitored regularly by the Mississippi Department of Marine Resources (MDMR) and the United States Geological Survey (USGS) (MDMR 2004). The USGS has continuous stream flow and water quality monitoring stations that measure gauge height, temperature, and salinity at different locations in and around the bay (USGS 2004). The MDEQ adopted wastewater regulations for National Pollutant Discharge Elimination System (NPDES) permits, Underground Injection Control (UIC) permits, state permits, water quality based effluent limitations and water quality certification in 1982 that were last amended on October 25, 2001 (MDEQ 2001.a.). The report on the NPDES and storm water management programs described the required regulations that have to be followed in order to protect the water quality of the state. Similarly, the water quality criteria developed and adopted by the State of Mississippi and approved by the Environmental Protection Agency (EPA) are followed to maintain and regulate the water quality of the intrastate, interstate and the coastal waters of Mississippi (MDEQ 2003).

Specific Objectives

1. The primary objective of this study was to determine the status of health of the Bay of St. Louis and find out if the environmental quality of this estuary (as defined above) was impaired. In the process, the goal was to determine the major factors that affected the health of the bay.
2. Another objective was to detect the spatial and temporal variations in the environmental quality of the bay that may have occurred over the years and to examine the long-term trends (over 30 years) in the responses of the bay to the varying natural and anthropogenic influences.
3. The final objective of this study was to provide a science-based management option that can be implemented readily. The goal was to develop an environmental quality evaluation card that can help assess and monitor the health of the bay and may be used as an effective management tool.

Research Hypotheses

Based on the results obtained from analyzing the available data and applying the appropriate statistical tests, the following hypotheses were tested:

1. H_0 -The environmental quality of the bay, considered as one system, is impaired.
2. H_0 - The environmental quality is similar at all locations within the bay including closer to the point-sources of nutrients and organic matter and does not vary over time (i.e. no spatial or temporal variability in environmental quality is observed in the bay).
3. H_0 - The environmental quality of the bay over the years is not affected significantly by the following factors:
    a. Variability in climate measured in terms of climate indices such as the SOI
    b. Changes in land use and land cover in the watershed.

CHAPTER II

BACKGROUND

Changes in Land Use and Land Cover (LULC) in the Watershed

The Bay of St. Louis watershed spans upto five different counties including the Hancock, Harrison, Pearl River, Stone and Lamar (Figure 2.1). The total drainage area for this estuary is approximately 2,117 square kilometers (MDEQ 2001.b.). The human population in the watershed area of the Bay of St. Louis increased by 64 % in the thirty years between 1970 and 2000 (Figure 2.2) (U.S. Census Bureau 2006). Similarly, modifications in anthropogenic activities led to changes in land use and alterations to land cover in the area during this period (1972-2000) (Figure 2.3). In Hancock County, developed land area increased by 79.6 %, agricultural land increased 25.5 %, while forests, wetlands, and other natural cover decreased by 5.7% between 1972 and 2000 (USACE 2003). Similar changes were observed in Harrison County where, urban areas increased 40.8%, agricultural cover grew by 71.8 %, while natural cover decreased by 9.6 % (USACE 2003). The Bay of St. Louis watershed, overall, lost 4.4 % of natural cover including wetlands, grasslands, forests and other such areas, while developed urban areas increased by 70 % (USACE 2003). Agricultural cover decreased by 4.7% between 1992 and 2000. However, agriculture increased by 13.7 % since 1972 (USACE 2003). Inland fresh water areas in both counties surrounding the bay as well as in the watershed remained the same (USACE 2003). Within the Bay of St. Louis watershed, several subwatershed areas exhibited different extents of land use and land cover (Figure 2.4 and Figure 2.5) (USACE 2003). The largest urban (9,031 acres) and wetland (9,934 acres) areas were found in the Bayou La Croix sub watershed (USACE 2003). The Upper

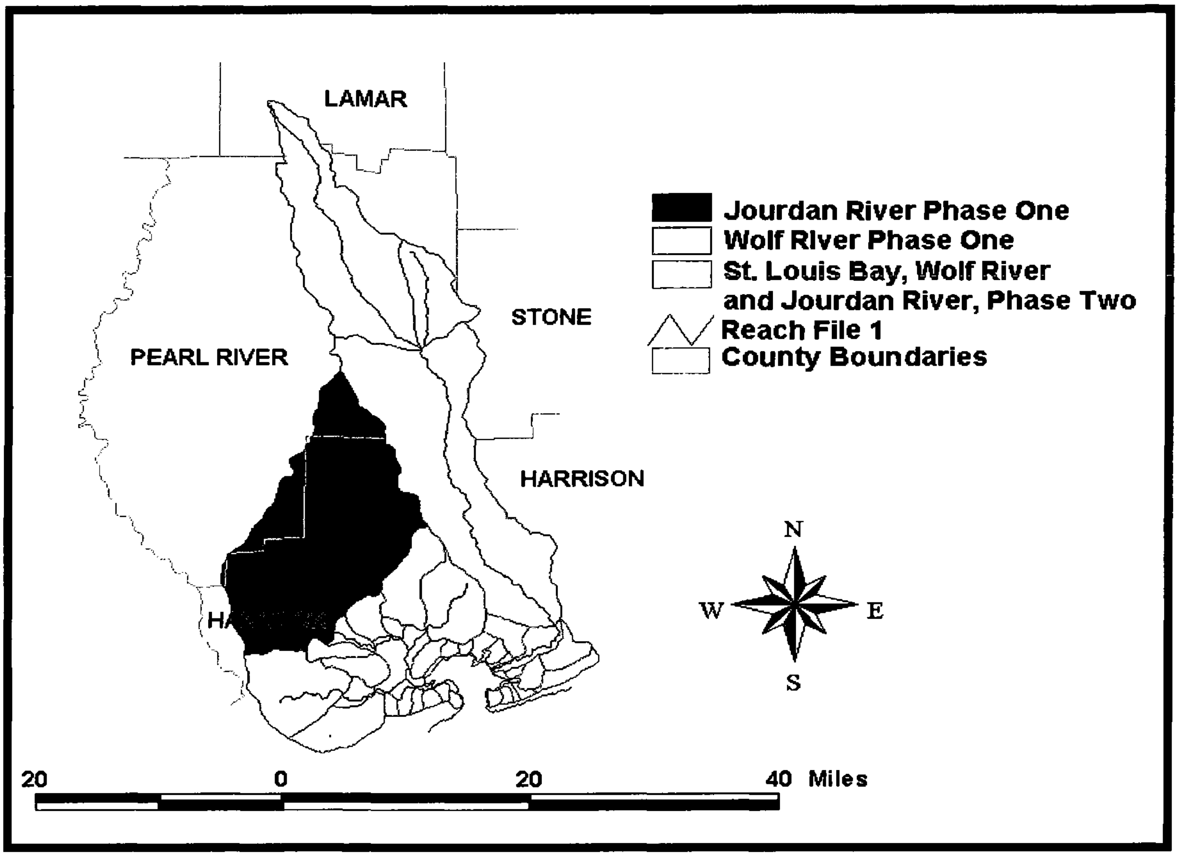

Figure 2.1 Map of the Bay of St. Louis Watershed. The BSL watershed extends into five different counties including the Hancock, Harrison, Pearl River, Stone and Lamar. Figure Source: MDEQ 2001.b. Fecal coliform TMDL for St. Louis Bay, Jourdan River (Phase Two) and Wolf River (Phase Two).

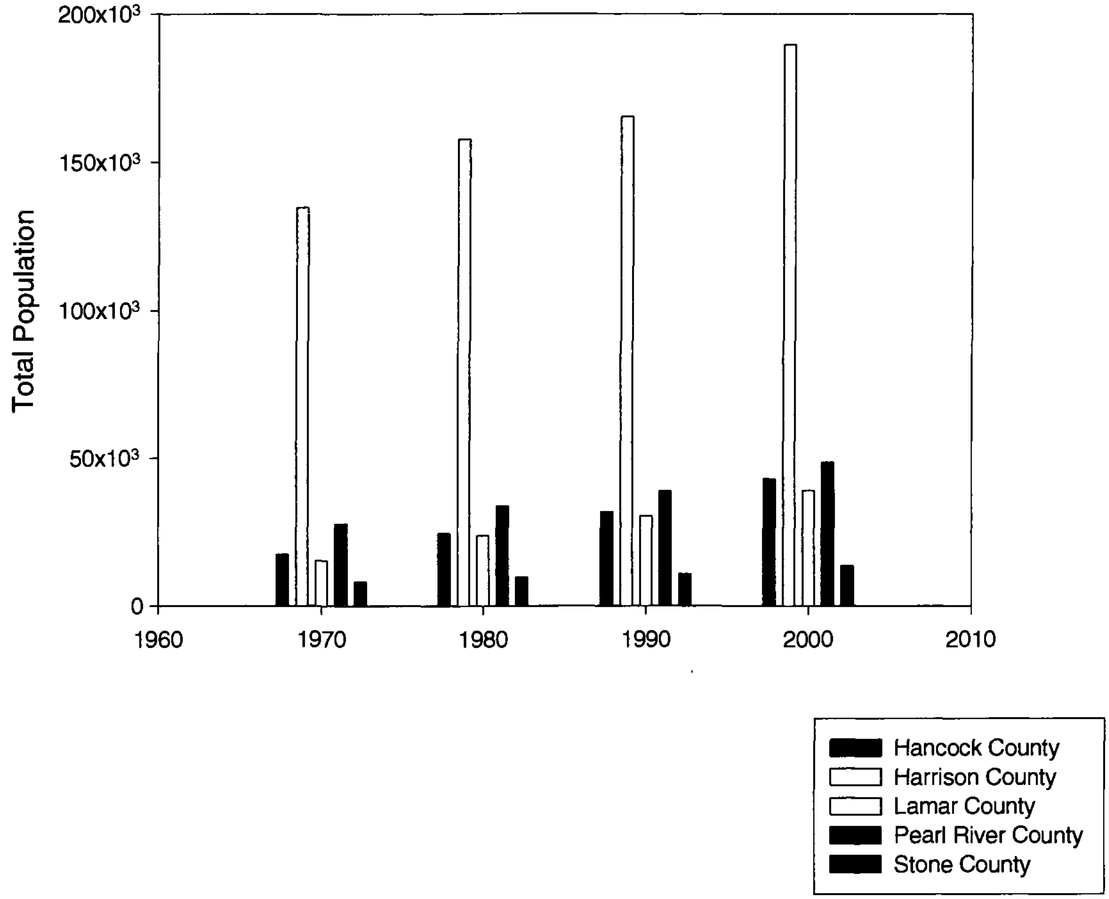

Figure 2.2 Changes in Total Human Population in Five Counties Constituting the Watershed of the Bay of St. Louis Recorded Since 1970 Until 2000. Data obtained from U.S. Census Bureau. Total population increased in all counties over thirty years. Harrison County had the largest population while Stone County had the least populated area.

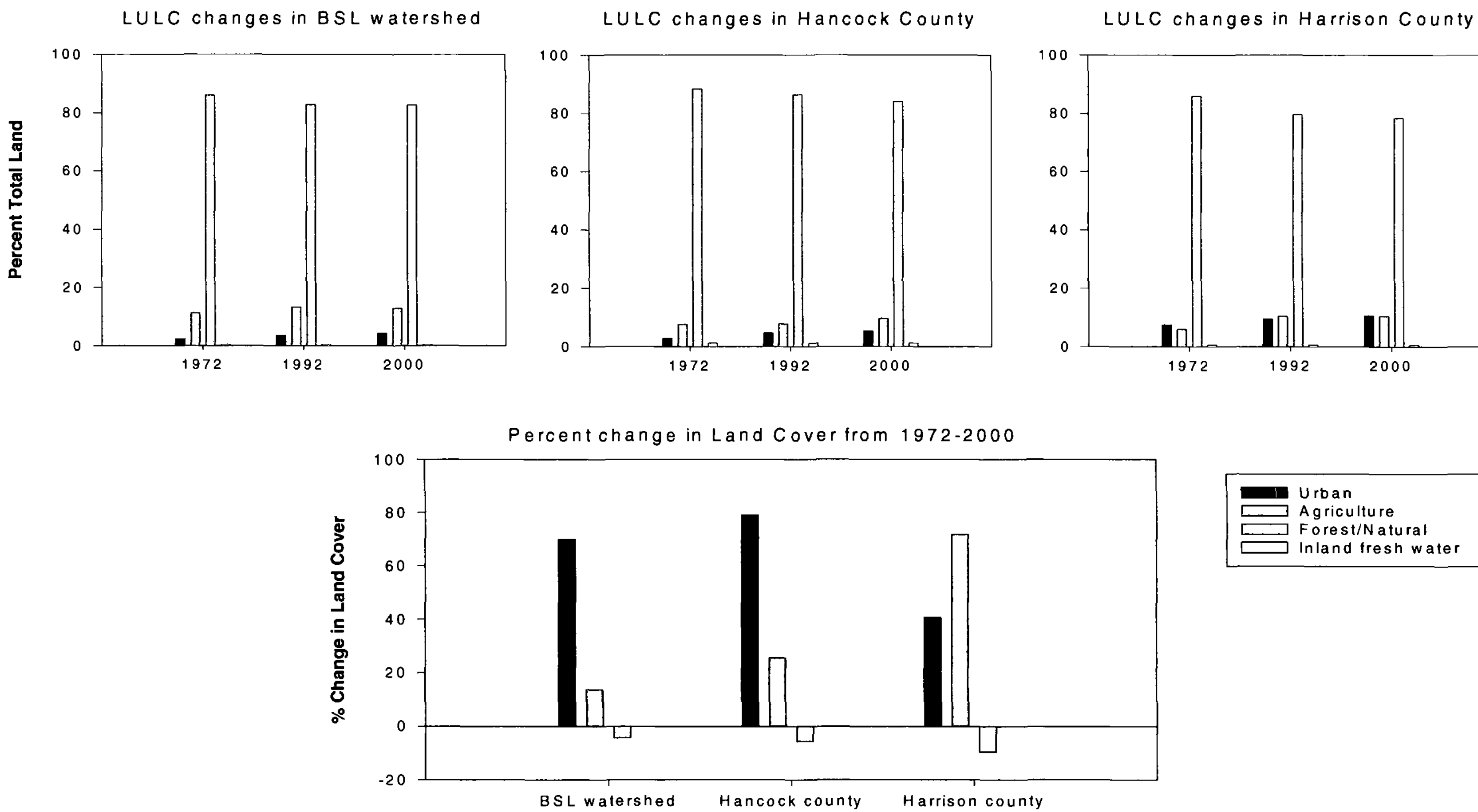

Figure 2.3 Percent Total Land Use Land Cover (LULC) in the Entire Watershed of the Bay of St. Louis and in Hancock and Harrison Counties Between 1972 and 2000. Data source, USACE 2003. Urban and agricultural use increased while forest cover decreased in all three regions over a period of thirty years. Highest increase in urban use was seen in Hancock County. Agricultural use in Harrison County increased more than urban use in thirty years.

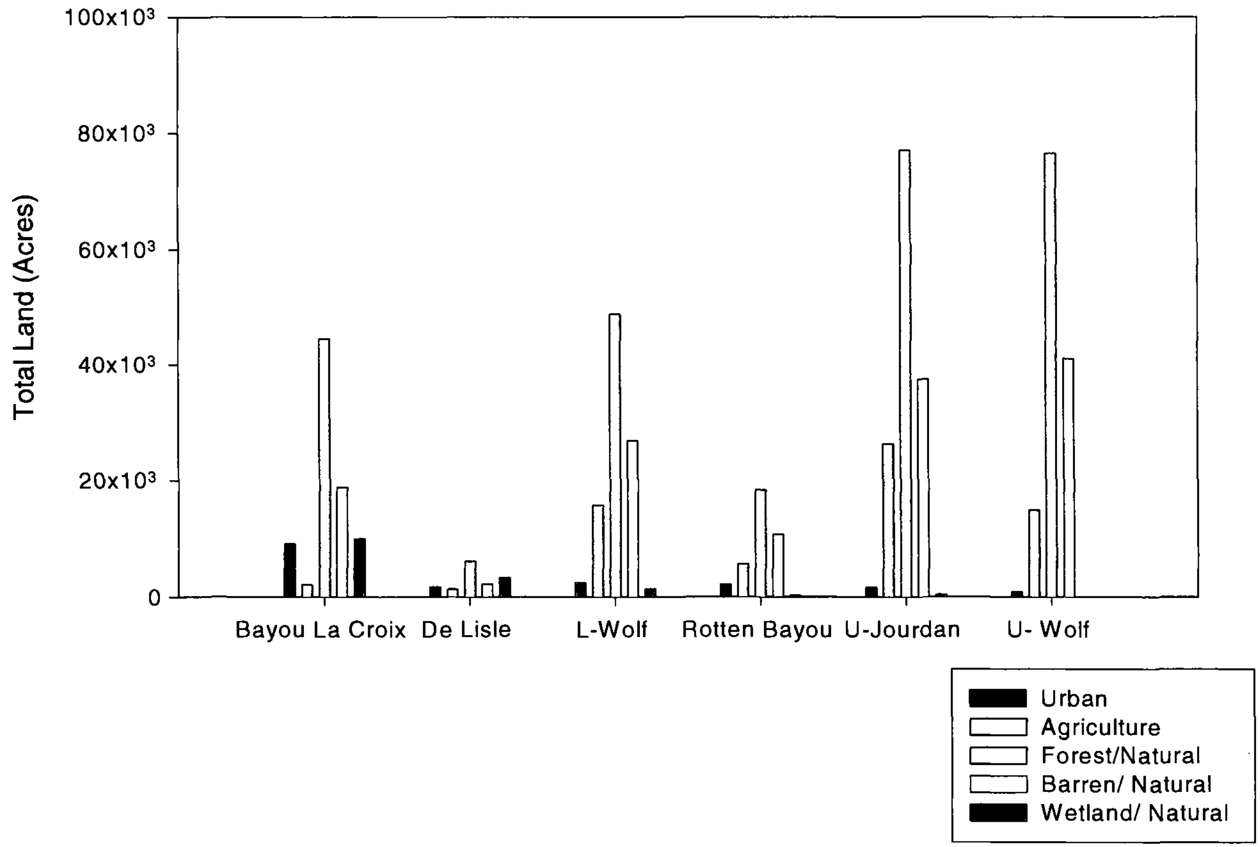

Figure 2.4 Total Land Areas in Each Subwatershed of the Bay of St. Louis Covered by Different LULC Types Such as Urban, Agriculture, Forest, Barren and Wetland as Recorded in 2000. Data source, USACE 2003. The subwatersheds include Bayou La Croix, De Lisle, Lower Wolf River (L-Wolf), Rotten Bayou, Upper Jourdan River (U-Jourdan), and Upper Wolf River (U-Wolf). All subwatersheds had higher forest cover and than urban and agricultural use.

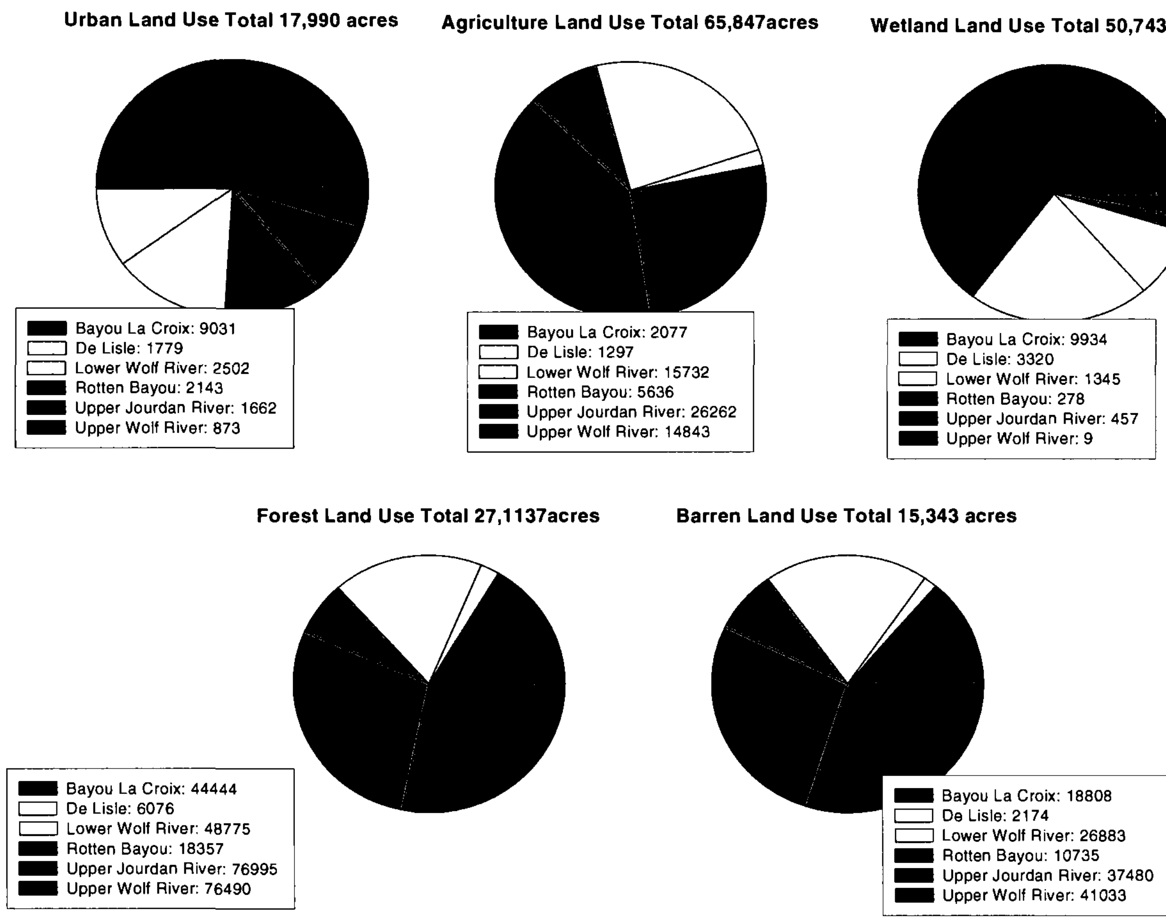

Figure 2.5 Extent of Each Land Use Land Cover Type Observed in Every Subwatershed of the Bay of St. Louis. Data source, USACE 2003. Urban and wetland areas were largely found in Bayou La Croix, higher agriculture and forest use in Upper Jourdan River, while largest barren land cover was found in Upper Wolf River subwatershed.

Jourdan River subwatershed had the most agricultural (26,262 acres) and forest land use (76,995 acres) (USACE 2003). The Upper Wolf River subwatershed had the second largest forest cover (76,490 acres), largest barren land use (41,033 acres), and smallest urban area (873 acres) (USACE 2003).

Point source discharges and runoff from the Bayou La Croix, Rotten Bayou, and Upper Jourdan river subwatersheds enter the bay from the western side, while the discharges from the De Lisle and Upper and Lower Wolf River subwatersheds flow into the north-northeastern bay (USACE 2003). Subwatersheds on the western side of the bay together had larger urban (12836 acres) and agricultural (33975 acres) uses than those on the northeastern side (USACE 2003) (Figure 2.6). Similarly large differences existed in types of wastewater systems between the Hancock (western bay) and Harrison counties (eastern bay). Several homes continued to be unsewered in the northern region of Hancock County (GMPO 2001). Total number of households connected to septic tank systems instead of centralized or public wastewater systems was higher in Hancock County (47.8 %) as compared to Harrison County (18.3 %) (USACE 2003). Also, the sewer collection systems in Hancock County had inflow and infiltration problems that can lead to an increase in the nutrients, pollutants, and fecal coliform concentrations (GMPO 2001).

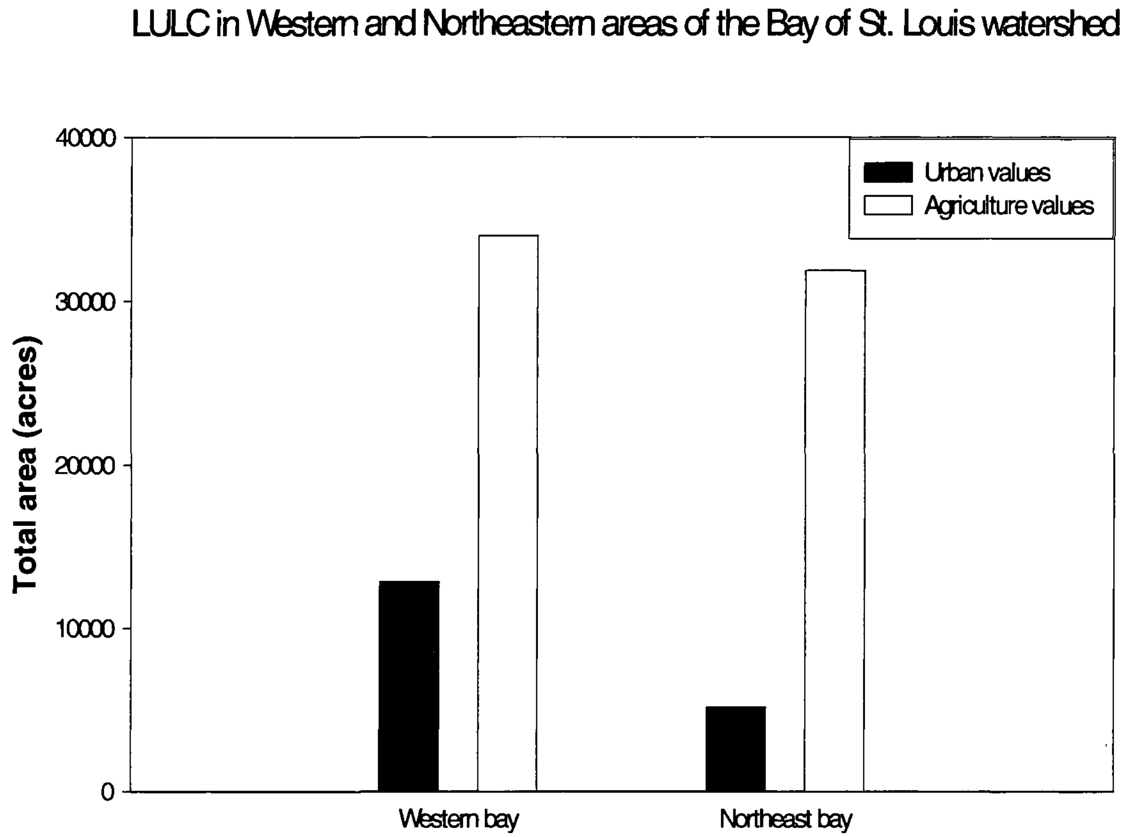

Figure 2.6 Comparison of Land Use in Two Different Areas of the Bay of St. Louis. Data source, USACE 2003. The subwatersheds surrounding and draining into the western areas had higher urban and agricultural use than those draining into the eastern areas of the Bay of St. Louis.

APPENDIX

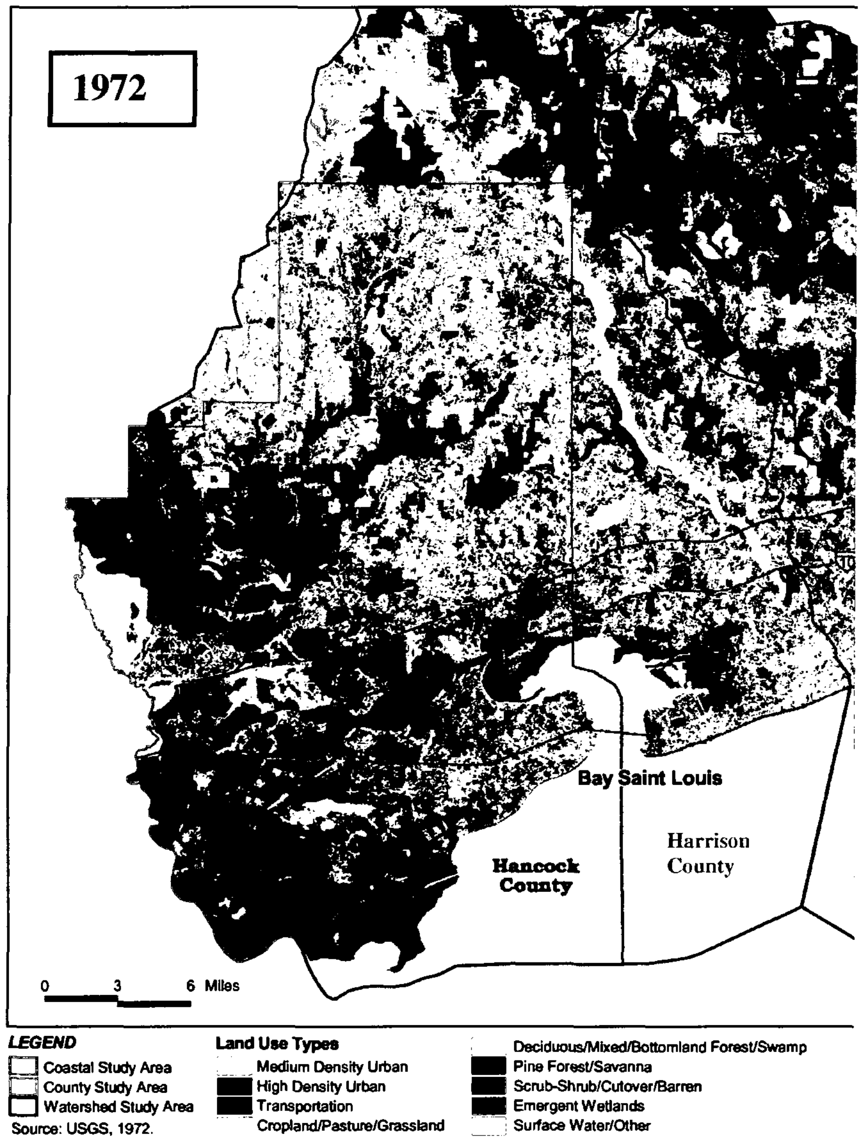

Figure 2A.1 Land Use and Land Cover in the Bay of St. Louis Watershed in 1972. The Bay of St. Louis watershed area, outlined in blue extends into the Hancock, Harrison, Stone, Pearl River, and Lamar counties. Upper watershed area in Lamar County is not shown. Forest cover was the largest type of LULC in this watershed. Figure source: USACE 2003.

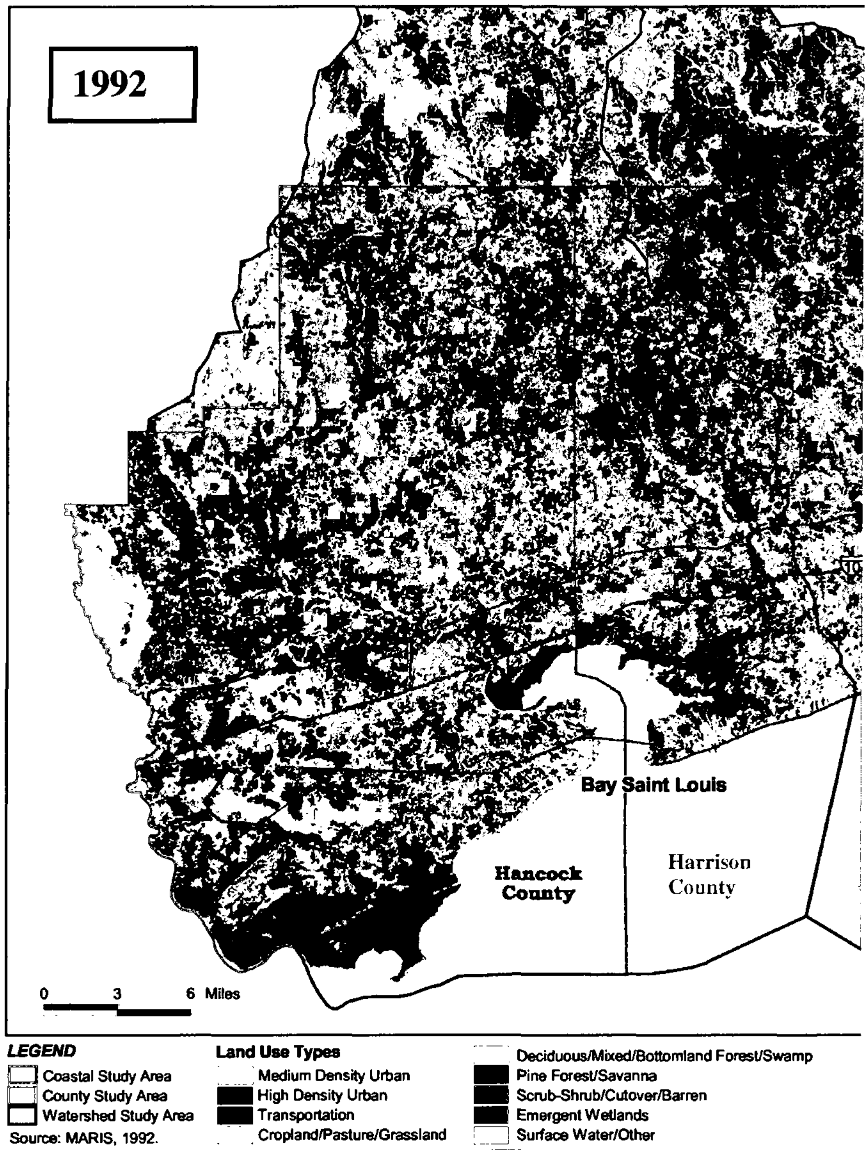

Figure 2A.2 Land Use and Land Cover in the Bay of St. Louis Watershed in 1992. The Bay of St. Louis watershed area, outlined in blue extends into the Hancock, Harrison, Stone, Pearl River, and Lamar counties. Upper watershed area in Lamar County is not shown. Figure source: USACE 2003.

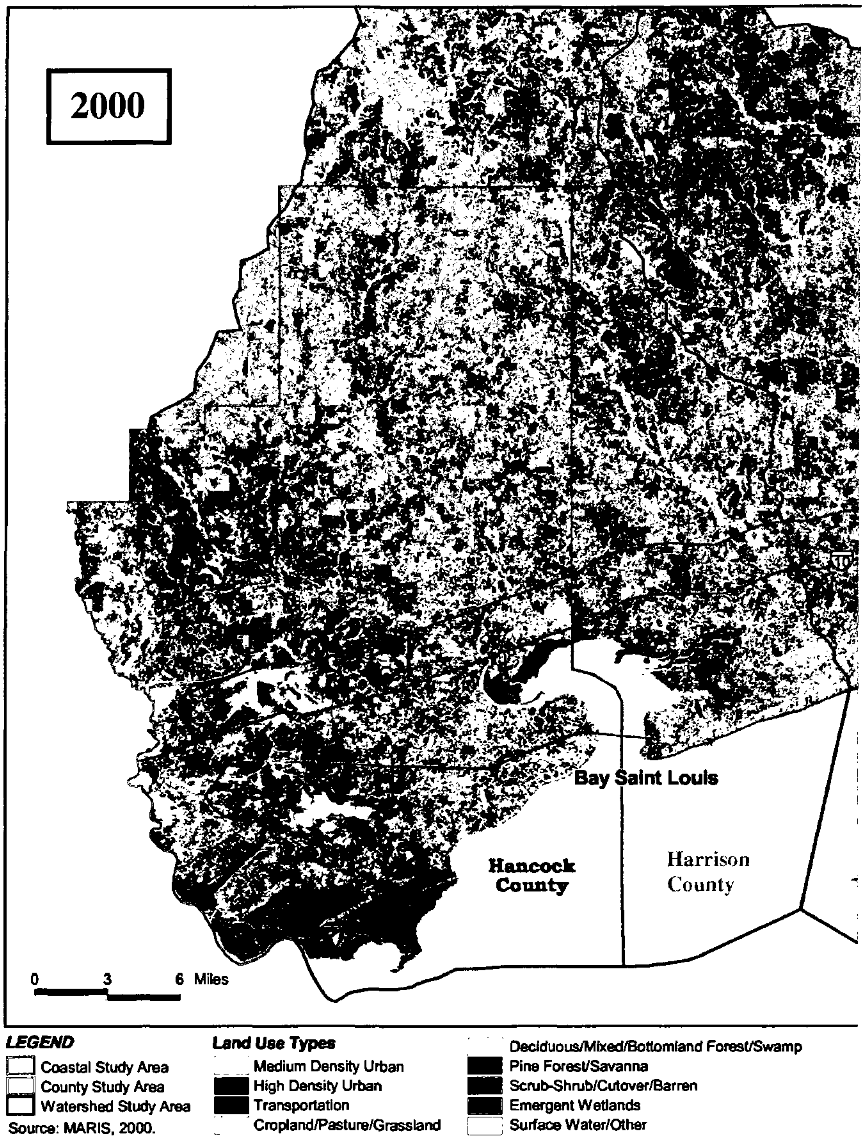

Figure 2A.3 Land Use and Land Cover in the Bay of St. Louis Watershed in 2000. The Bay of St. Louis watershed area, outlined in blue extends into the Hancock, Harrison, Stone, Pearl River, and Lamar counties. Upper watershed area in Lamar County is not shown. Between 1972 and 2000, the urban land cover increased in this watershed by 70%, while agricultural cover increased by 12.7%. Natural land cover including forests, wetlands, and grasslands decreased by 4.4% in the Bay of St. Louis watershed since 1972. Figure source: USACE 2003.

CHAPTER III

MATERIALS AND METHODS

Data Acquisition

2003-2004 Sampling Period

Sampling for the year 2003-2004 was carried out from April 2003 to May 2004 as part of the EEQ project. In order to obtain representative data, an adaptive sampling technique was used wherein the sampling stations were strategically placed close to known point sources of nutrients, organic matter and sediments. Ten sampling stations were established to represent the various point sources, a north-south transect, and mixed zones (Figure 3.1 and Table 3.1). Two stations were established on the western side of the bay that represented the point sources. Station 1 was located at the mouth of the Jourdan River, a source of fluvial input, runoff from the western subwatersheds, and the discharge from sewer outfalls (The Southern Regional Wastewater Management District, which empties into the Edward's bayou that joins the Jourdan River and the Diamondhead Water and Sewer District) (Figure 3.1). Station 6a was located farther away from station 1 on the western shore of the bay and was closer to a drainage ditch (Figure 3.1). The north and north-eastern shore of the bay were represented by stations 2, 4 and 5 (Figure 3.1). Station 2 was located near the outfall of a titanium dioxide plant. Station 4 was located at the mouth of the Wolf River thus representing the fluvial inputs and runoff from the Wolf River watershed. Station 5 was located near the mouth of Bayou Portage, which received sewer discharge from the Long Beach-Pass Christian municipal sewer treatment plant and several other commercial and industrial dischargers.

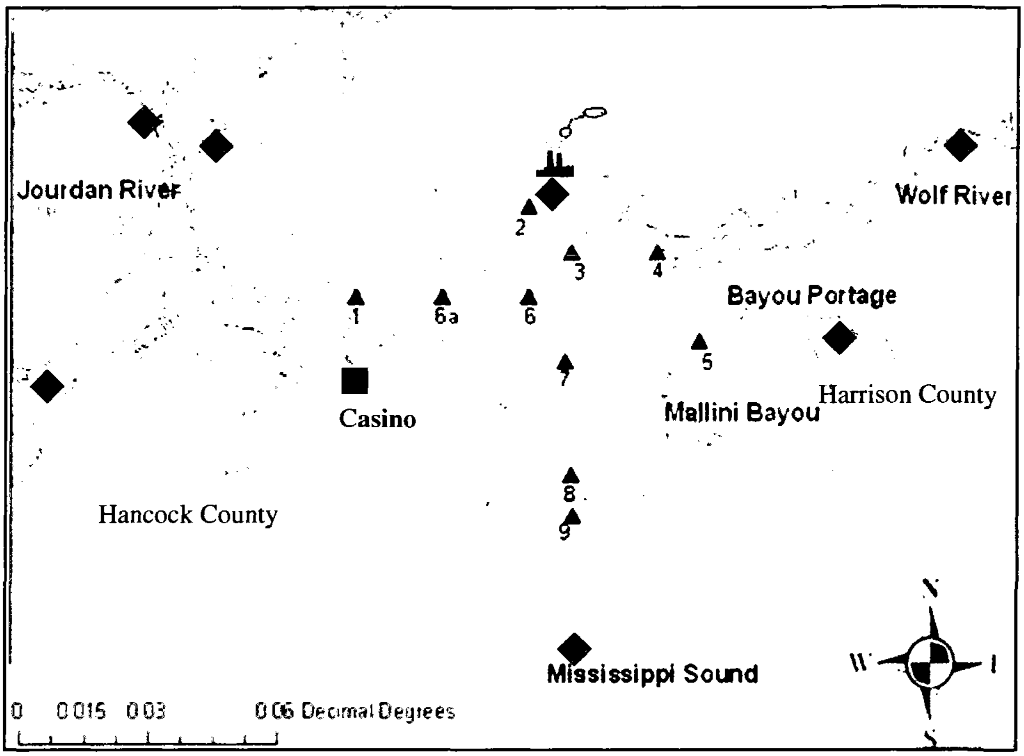

Figure 3.1 Map of the Sampling Locations in the Bay of St. Louis, MS. Sampling stations are denoted by black triangles and marked with station numbers. The vertical blue line running through the center is the county line dividing the Hancock and Harrison counties. The major sources of fresh (Jourdan and Wolf rivers) and saline water (Mississippi Sound) are denoted by red diamond symbols. The Jourdan River enters the estuary on the western side, the Wolf River on the northeast, and the Bayou Portage empties into bay on the eastern side. The bay joins the Mississippi Sound to the south by a narrow passage. Adaptive sampling method was used wherein some of the sampling stations were strategically placed close to known point sources of nutrients, sediments, and pollutants (Please refer Table 3.1 for station characteristics). The identified point sources included the rivers, streams, and bayous, four sewage treatment outfalls (denoted by dark brown diamond symbols), a gaming facility (denoted by a purple rectangle), and a titanium dioxide plant (denoted by a factory symbol). The non-point sources were in the form of old and leaking septic tanks as well as runoff from agricultural and several other anthropogenic activities that occur within the watershed of the bay. Note: The stations and other facilities shown on this map are marked based on their approximate geographical locations.

Table 3.1 Location Characteristics of Sampling Stations. Adaptive sampling technique was used wherein each station was strategically placed close to known sources of nutrients, sediments, and pollutants or to represent mixed waters.

Stations	Location Characteristics
1	At the mouth of the Jourdan River, near a gaming facility
2	Near the outfall of the titanium dioxide plant
3	Near grassy point, close to the outfall of the titanium dioxide plant and lies along the north-south transect
4	At the mouth of the Wolf River
5	At the mouth of Bayou Portage
6	Middle of the bay: represented mixed waters and lied within the north-south and east-west transects
6a	In the path of the outflow of the Jourdan River along the western shore, lied close to the residential shoreline and near a drainage ditch
7	Middle of the bay: represented mixed waters and lied within the north-south and east-west transect
8	Middle of the bay: located in the narrow passage connecting to the Mississippi sound (between the Highway 90 and railroad bridges)
9	The southernmost station located in the Mississippi Sound, at the mouth of the bay

The north-south transect was formed by stations 2, 3, 6, 7, 8, and 9 (Figure 3.1). Station 9 was the southern most sampling station and was located at the mouth of the bay in the Mississippi Sound. Sampling at each station was carried out twice every month once each during the outgoing and the incoming tide for the entire period of fourteen months except during April 2003. Only one sampling was carried out in April 2003, and it was done during an outgoing tide.

In situ profiles of temperature, salinity, dissolved oxygen, pH, and turbidity were obtained using Yellow Springs Instruments (YSI) 6000 UPG, multi-parameter, water quality monitor that had depth, temperature, conductivity, pH, dissolved oxygen (DO), and turbidity (turbidimeter calibrated with formazin providing measure of suspended particles only and not dissolved material) sensors on it. For each of the above parameters, an average of measurements taken from 0m to 0.5m was used as surface data for this project. All data were measured from April 2003 to May 2004. Turbidity data were not available from November 2003 to January 2004 due to failed turbidity sensor. Surface water samples were collected in one liter Nalgene bottles at each sampling station and stored in a cooler until filtration in the lab. Each water sample was filtered (47mm Whatman GF/F) and further analyzed in the laboratory for nutrient and plant pigment concentrations. The samples collected from the bay were filtered onto 47mm Whatman GF/F filters to determine the plant pigment concentrations using High Performance Liquid Chromatography (HPLC). Samples were also filtered onto 25mm GF/F filters from June 2003 onwards to measure chlorophyll *a* concentrations fluorometrically using the procedures described by Parsons et al. (1984). The procedures were slightly modified for the extraction process. A sonic dismembrator and a vortex mixer were used instead of a motorized homogenizer before centrifugation. The samples were immersed in 5 mL of 90% acetone in 10 mL centrifuge tubes kept in a dark box. The dark box was placed into a freezer overnight for pigment extraction. Samples from June 2003 to October 2003 were analyzed using the calibrated Turner Designs Model 10 R using the following equations:

$$\text{Chl } a\ (\mu gL^{-1}) = F_D \times (\tau/\ \tau\text{-}1) \times (R_B - R_A)\ v/V$$

$$\text{Phaeo-pigment } (\mu gL^{-1}) = F_D \times (\tau/\ \tau\text{-}1) \times (\tau \times R_A - R_B)\ v/V$$

where F_D was the door factor for each door setting, τ was a constant, 2.05(the mean maximum ratio of R_B/R_A), R_B and R_A were the fluorescence before and after the addition of acid respectively, v was the volume of 90% acetone extract in mL, and V was the volume of filtered sample water in liters.

Samples from November 2003 to May 2004 were measured using the calibrated Model 10- AU-005-CE field fluorometers and the following equations:

$$\text{Chl } a\ (\mu gL^{-1}) = (K \times F_m \times v \times (F_o - F_a))/\ V_f \times (F_m - 1)$$

$$\text{Phaeo-pigment } (\mu gL^{-1}) = (K \times F_m \times v \times (F_m \times F_o - F_a))/\ V_f \times (F_m - 1)$$

where K was a constant, 9.8838×10^{-4}, F_m was a constant, 1.84 (the mean maximum ratio of F_o / F_a), F_o and F_a were the fluorescence readings before and after acidification respectively, v was the volume of 90% acetone extract in mL, and V_f was the volume of filtered sample water in liters.

All filtered surface samples were analyzed for five different nutrients including Nitrate, Nitrite, Ammonium, Orthophosphate, and Silicate. The nutrient concentrations were determined for small volumes (10 mL) with three analytical replicates. All the nutrients except ammonium were analyzed using the methods described by Parsons et al.

(1984). Ammonium was determined using the fluorometric technique described by Holmes et al. (1999). The nitrate reduction column of the Lachat AE QuickChem flow injection analyzer with random access sampler was used for the nitrate analysis. Nitrate, Nitrite and Ammonium were summed to determine Dissolved Inorganic Nitrogen (DIN) while the measured Dissolved Inorganic Phosphorus (DIP) was Soluble Reactive Phosphorus (SRP) or orthophosphate. DIN data were available from June 2003 onwards since ammonium was not analyzed during April and May 2003. Nutrient concentrations, below detection levels, were considered zero and included in the data.

Weather data for the this project were obtained using a HOBO weather station deployed near Mallini Point in Pass Christian, MS from July 2003 to May 2004. These data included atmospheric pressure, temperature, precipitation, humidity, Photosynthetically Active Radiation (PAR), full spectrum solar radiation, wind speed, wind gust, and wind direction. Although precipitation data were collected from the HOBO weather station, the data were not used for this study due to the intermittent failure of the rain gauge. Precipitation and other real-time data such as the river flow data were obtained from the USGS continuous stream flow and water quality monitoring stations located at Wolf and Jourdan rivers and the Mississippi sound near the Bay of St. Louis (USGS 2004). Stream flow data were not recorded continuously at the Jourdan River station and therefore only the river discharge data recorded at the USGS Wolf River station near Landon, MS were used for this study. Gage height data were obtained from the USGS station at the Bay-Waveland Yacht Club in Bay St. Louis, MS. This USGS station was close to sampling station 6a of this study. Gage height at the time of sampling individual stations was calculated by interpolation using the start and end times of sampling. LULC data were obtained from the published literature of the U.S. Army Corps of Engineers, Mobile District (USACE 2003). The total population data was obtained from the US Bureau of the Census website (U.S. Census Bureau 2006).

Previous Data Sets

Data for the year 1977-78 were obtained from the published report of the study conducted by the GCRL (GCRL 1978). Twenty stations were sampled during this period, only 10 of which were comparable to the sampling stations used in the USM/DMS studies (1995-96, 1997-98, and 2003-04) based on their geographical locations. However, only statistical data such as the annual average, standard deviation, range and number of data points were available for individual stations for the 1977-78 study. Most physical parameters such as temperature, salinity, pH, dissolved oxygen, and turbidity were similar to the latter studies. However, raw data for DO was not available for this study. Turbidity was measured nephelometrically in this study but was reported as JTU (or Jackson Turbidity Units) instead of NTU (Nephelometric Turbidity Units) (GCRL 1978). Therefore the turbidity data from this study were considered suitable to be used to compare with the turbidity measurements (reported in NTU) from the latter studies. All nutrients including ammonium, nitrate, nitrite and orthophosphates were reported as μg-at L^{-1} (GCRL 1978). All nutrients except ammonium were determined colorimetrically, ammonium was analyzed using the ion selective electrodes (GCRL 1978). Chlorophyll *a* concentrations were reported in units of mg m^{-3}, which could be compared to the chlorophyll data in the latter studies (GCRL 1978).

Adaptive sampling techniques were also used for the previous USM/ DMS projects (1995-1996 and 1997-1998). However, only nine locations were sampled during these studies, all of which were part of the 2003-04 sampling program. Station 6a was not sampled during the 1995-96 study while station 6 was excluded during the 1997-98 study. All the nutrients which were determined colorimetrically and the chlorophyll *a* concentrations, determined fluorometrically, were analyzed using the techniques described in Parsons et al. (1984) (Redalje and Rasure, unpub. 1995-1996; Phelps 1999). The Yellow Springs Instruments (YSI) 6000 UPG, multi-parameter, water quality monitor was used during both the studies to measure the *in situ* temperature, salinity, dissolved oxygen, pH, and turbidity at sampling locations. Turbidity data were, however, not available for the 1995-96 study.

For a comparative study, the total precipitation data for all data periods (1977-2004) were obtained from the National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center station located in Hattiesburg, MS. The river discharge data for all years (1977-2004) were obtained from the USGS continuous stream flow and water quality monitoring station located at Wolf River near Landon, MS. The SOI data were obtained from the Commonwealth of Australia, Bureau of Meteorology, where the final SOI values are calculated by multiplying the index values (standardized anomaly) by 10 so that SOI ranges from -35 to +35.

Developing a Report Card

An evaluation system for the Bay of St. Louis was developed based on the guidelines of the Moreton Bay and Chesapeake Bay Report Cards (Healthy Waterways 2002; Integration and Application Network 2003). Several physical, chemical and biological parameters commonly measured during all the sampling periods were representative of the state of health of the ecosystem. Since these parameters were quantifiable, they could be used effectively as indicators of the environmental quality of the bay. In order to select the indicators, it was important to confirm that these parameters met the objectives of the management plan. The following management objectives were identified for the Bay of St. Louis in this study:

1. Clear water = Less turbidity: Turbidity measurements determine the water clarity that will further determine the productivity and the health of the estuary.
2. Reduced nutrients = Reduced N and P loads: Dissolved Inorganic Nitrogen (DIN) ($NO_3 + NO_2 + NH_4$) and Dissolved Inorganic Phosphorus (DIP) are the major nutrients that cause nutrient enrichment problems and therefore are required to be monitored and regulated.
3. Support aquatic life = More Dissolved oxygen: Dissolved oxygen is important for survival of all aquatic life.
4. No algal blooms = Controlled phytoplankton concentrations: Measurement of chlorophyll *a* concentrations is a representation of the measure of the phytoplankton biomass in the waters. It is important to control the factors that may lead to an increase in primary production, which in turn may trigger algal blooms. Thus in order to maintain a healthy ecosystem it was essential to monitor for excessive algal biomass.

The reference values set for each indicator were representative of the values found or expected in pristine or largely unmodified subtropical environments (USEPA 1999 and Commonwealth of Australia 2002) (Table 3.2). The measured values from the available data sets had to be either equal to or greater than (for dissolved oxygen) or lower than

Table 3.2 Reference Values for Environmental Quality Indicators Adapted from USEPA 1999 and Commonwealth of Australia 2002. The reference values are representative of values found or expected in pristine or largely unmodified subtropical environments.

Environmental Quality Indicator	Reference Value
DIN	2 μM
DIP	0.2 μM
Chl *a*	15 μg/L
Turbidity (NTU)	10
Dissolved Oxygen	2 mg/L

Table 3.3 Evaluation Criteria for the Environmental Quality Indicators. Environmental quality at a particular location did not meet a certain management objective if the measured values were higher than the upper threshold value (except for DO). It exceeded the management objective if the measured values were lower than the lower threshold value (except for DO).

No.	Environmental Quality Indicator	Does not meet Objective	Meets Objective/ Reference values	Exceeds Objective
1	DIN	> 4 μM	2 - 4 μM	< 2 μM
2	DIP	> 0.4 μM	0.2 – 0.4 μM	< 0.2 μM
3	Chl *a*	>20 μg/L	15 - 20 μg/L	< 15 μg/L
4	Turbidity (NTU)	>15	10 - 15	< 10
5	Dissolved Oxygen	< 2 mg/L	2-5 mg/L	> 5mg/L

(for nutrients and chlorophyll *a* concentrations and turbidity) the set criteria to meet or exceed the management objectives (Table 3.3).

Once the criteria for environmental quality were established (i.e. selecting the environmental quality indicators and suitable reference values for each indicator), depending on whether the dataset values met the criteria or not, the environmental quality was determined as Does not meet Objective, Meets Objective, or Exceeds Objective for a particular location. Numerical values were assigned to these grades to develop an Environmental Quality Index (similar to Ecosystem Health Index; Healthy Waterways 2002; Jones et al. 2004) for the entire bay (Table 3.3). Sum of the numerical values was used as a rank on a scale of 1 to 10 for a particular location (Table 3.4). An overall grade from A to F based on the rank scale was then assigned to these locations (Table 3.4). Similarly, an overall grade based on the rank scale was assigned for the entire bay as well.

Data Analysis

Preliminary statistical analysis techniques such as measures of central tendency, measures of dispersion, and graphical presentations were applied to all of the available data. Similarly, the data were tested for normality before conducting further analyses and statistical tests. Non-parametric statistical analysis techniques were applied to the data set if the data were not distributed normally. The spatial and temporal variations in the environmental quality of the bay and the selected parameters were determined using the analysis of variance (ANOVA) on Ranks, particularly the Kruskal-Wallis test, or the Mann-Whitney Rank Sum test depending upon the distribution of the data. The significance of interrelationships between the chosen variables and their relation to the

Table 3.4 Color-Coded Legend for EQI Values that Range from 0.00 to 2.00. These values are based on environmental quality grades of 'Does not meet Objective" (Red), "Meets Objective" (Orange), or "Exceeds Objective" (Green) for every indicator at a particular location. Higher EQI values and the green color denote good conditions, while lower EQI values and the red color denote bad conditions.

Environmental quality grade	***Color***	***EQI value***
Exceeds Objective		**2.00**
Meets Objective		**1.00**
Does not meet Objective		**0.00**

Table 3.5 Color-Coded Legend for Final Environmental Quality Index (EQI) Ranks. Total EQI scores range from 0 to 10 on the final EQI rank scale and are sum of EQI values for all indicator parameter assigned at each sampling station. Letter grades from A to F are assigned for different scores on the rank scale. The letter grade A and green color indicate good conditions while the grade F and brown color indicate impaired conditions.

Environmental Quality	**Index**	**Final EQI Ranks**	**Letter Grade**
GOOD		**7.5 – 10.0**	**A**
ACCEPTABLE		**5.0 – 7.5**	**B**
POOR		**2.5 – 5.0**	**C**
IMPAIRED		**0.0 – 2.5**	**F**

responses of the bay were verified using Spearman rank correlation analysis. The results from the 2003-04 sampling period were described using several different analyses. Temporal variability in the weather and environmental quality parameters was determined using the Kruskal-Wallis One Way ANOVA on ranks test. Correlations between the environmental quality parameters and the weather parameters were determined using the non-parametric Spearman rank correlation test. All environmental quality data were also analyzed for spatial variability. This was done using the Kruskal-Wallis One Way ANOVA on ranks test. Spatial and seasonal variability in the environmental quality indicators were described using spatial interpolation analysis in GIS. Spatial distributions of each environmental quality indicator were graphically presented with the help of GIS maps of the Bay of St. Louis developed for each season.

Annual environmental quality index values for the indicator parameters were assigned for each station and presented in a tabular form. A total EQI value for each station was calculated by adding up the EQI values of all parameters at each station. A final (average) EQI value for all stations was calculated by adding the total EQI values of all stations divided by the number of stations.

Data from the previous sampling periods were described and compared with the 2003-04 data period. Mean and standard error values of indicator parameters were used for all data periods. These data were also analyzed for temporal variability using the Kruskal-Wallis One Way ANOVA on ranks test. Significant correlations between precipitation, river discharge, and nutrient concentrations and the Southern Oscillation Index (SOI) values were determined using spearman rank correlations tests. Environmental Quality Index (EQI) values for each indicator parameter were calculated for all data periods. In order to compare the differences in environmental quality between different climate regimes, the final EQI values for each indicator parameter were calculated for each station sampled during the four data periods. These final EQI values for each parameter were presented in separate tables comparing all data periods.

Geographic Information Systems (GIS) Mapping: Graphical Presentation of the Data

GIS mapping is an effective technique to synthesize multi-dimensional data sets and perform spatial statistical analyses (Integration and Application Network 2003). It is also a tool that facilitates geographical and conceptual linking of individual data points as well as creation of statistically valid spatial interpolations (Integration and Application Network 2003; Jones et al. 2004). GIS mapping was used in this study to display the geographic information such as the study area and sampling locations and also to present the spatial distribution of the data. It was also used to demonstrate the trends and changes in parameters over different seasons. Similarly, the annual environmental quality status of the Bay of St. Louis for 2003-04 was also presented with the help of the GIS mapping techniques. ESRI software programs Arc View 3.3 and Arc GIS 9, ArcMap 9.1 were used for GIS mapping in this study. A GIS layer of the Bay of St. Louis, MS (a map outline) was digitized using Arc View 3.3 to create a shape file of the bay. Ten sampling stations were added to the layer of the bay polygon using ESRI software Arc GIS 9, ArcMap 9.1. Data from the ten sampling points (stations) for each of the five indicator parameters were used to obtain the respective values for the entire bay. This was done using the Krigging interpolation method. Individual prediction maps were created for each parameter, which provided the predicted values for every point in the bay. It is, however, important to note that creating contour maps based on ten points (insufficient data) are often subject to unavoidable artifacts. Scarcity of data points led to generation of unsupported results. Therefore, a respective standard error map was also created for every prediction map. Since contour maps were created using a computer program, prediction results with high standard errors were masked manually in all the maps presented in this document.

CHAPTER IV

RESULTS

Overview

Environmental quality and weather parameters for the Bay of St. Louis were measured for fourteen months (twice a month) during 2003-04. The 2003-04 data were compared with previous datasets from three other sampling periods (1977-78, 1995-96, and 1997-98). All data were analyzed for spatial and temporal variability including variability on the scale of shifting climate regimes. The data were then evaluated for environmental quality based on the Environmental Quality Index developed for this estuary Seasonal variability in 2003-04 was seen in weather parameters such as air temperature, air pressure, dewpoint temperature, relative humidity, wind speed and direction, PAR and solar radiation, total precipitation and river discharge. Significant spatial variability was observed only in few parameters such as salinity, pH, NO_3+NO_2, DIN, and DIP concentrations. Temporal variability was observed in the environmental quality parameters including water temperature, salinity, pH, DIN, DIP, silicates, chlorophyll *a,* phaeopigments, turbidity, and DO concentrations. Environmental quality indicator parameters were correlated to changes in weather and physical conditions, and thus seasonal variability was seen in all indicator parameters (DIN, DIP, chlorophyll *a*, turbidity, and DO). Spatial and temporal variability in all parameters were determined using the Kruskal-Wallis one-way analysis of variance on ranks test. Annual environmental quality for the entire bay was "Good" marginally. Significant temporal differences in environmental quality indicators were also observed between the different data periods (variability was determined using the Kruskal-Wallis one-way analysis of variance on ranks test). The highest nutrient concentrations were observed during 1995-96 while maximum turbidity was recorded during 1997-98. The nitrogen nutrient indicator was correlated significantly to river discharge over all data periods. The average discharge from Wolf River was correlated significantly to SOI during 1977-2004. Environmental Quality Index (EQI) values calculated for each data period indicated that the environmental quality in the bay mostly deteriorated during the 1995-96, was mostly "Acceptable" during 1977-78 and 1997-98, and was mostly "Good" during the 2003-04 data period.

2003-2004 Sampling Period

Weather Conditions

Weather data such as air pressure, temperature, dew point, relative humidity, solar radiation, PAR, wind and gust speeds, wind direction and rainfall were obtained from the HOBO weather station from July 2003 to May 2004. During this period, daily average air temperature ranged from 3.6°C to 29.5°C. Highest daily average temperature was recorded in late summer in the month of August, while lowest daily average temperature was recorded in winter in January 2004 (Figure 4.1). Temperatures were high in the summer and early fall months (July to October) and generally low in late fall and winter (November to February). Air temperatures gradually increased again in late winter and the following spring (February to May 2004). The trend seen in air temperature was also observed in the dew point data (Figure 4.1). The highest daily average dew point temperature (25.9 °C) was recorded in August 2003, while the lowest daily average frost point (-8.8 °C) temperature was observed in January 2004. Relative humidity changed in correlation with both air temperature and dew point (Figure 4.1). Highest daily average relative humidity (100.41%) was recorded in February 2004, whereas lowest daily average relative humidity (27.09 %) was recorded in April 2004, when dew point and air temperatures were on a rise. Atmospheric pressure also changed in relation to air temperature (Figure 4.2). Daily average air pressure increased to its highest recorded value of 1033.59 mbar in winter (January 2004), while the lowest daily average pressure (1005. 53 mbar) was recorded in April 2004.

PAR and Solar radiation were found to co-vary and were correlated significantly to each other ($r = 0.939$, $n = 315$, $p < 0.01$). Total daily PAR was highest (32.4 mol $m^{-2} d^{-1}$) in May 2004and the lowest (0.36 mol $m^{-2} d^{-1}$) in the summer in July 2003 (Figure 4.3). Frequent peaks in PAR, also however, occurred in July and October 2003 and in February-March 2004. Highest total daily solar radiation values (114×10^3 W/m^2) were recorded in October 2003 while lowest solar radiation, like PAR, was also measured in July 2003. Wind data recorded since July 2003 indicated an increase in wind speed during May 2004, with the highest daily average wind speed of 1.84 ms^{-1} reported that month (Figure 4.4). Lowest daily average wind speed was measured in February 2004. Wind direction during the entire period was mainly onshore, with most frequent winds from South and South-East. Fresh water input to the bay varied over the fourteen months of sampling. Precipitation in the area increased during June-July 2003 and February and May 2004 (Figure 4.5). A corresponding increase in river discharge was observed in July 2003, the month of highest daily average river flow (62.07 m^3s^{-1}) and again in February and May 2004 (Figure 4.6).

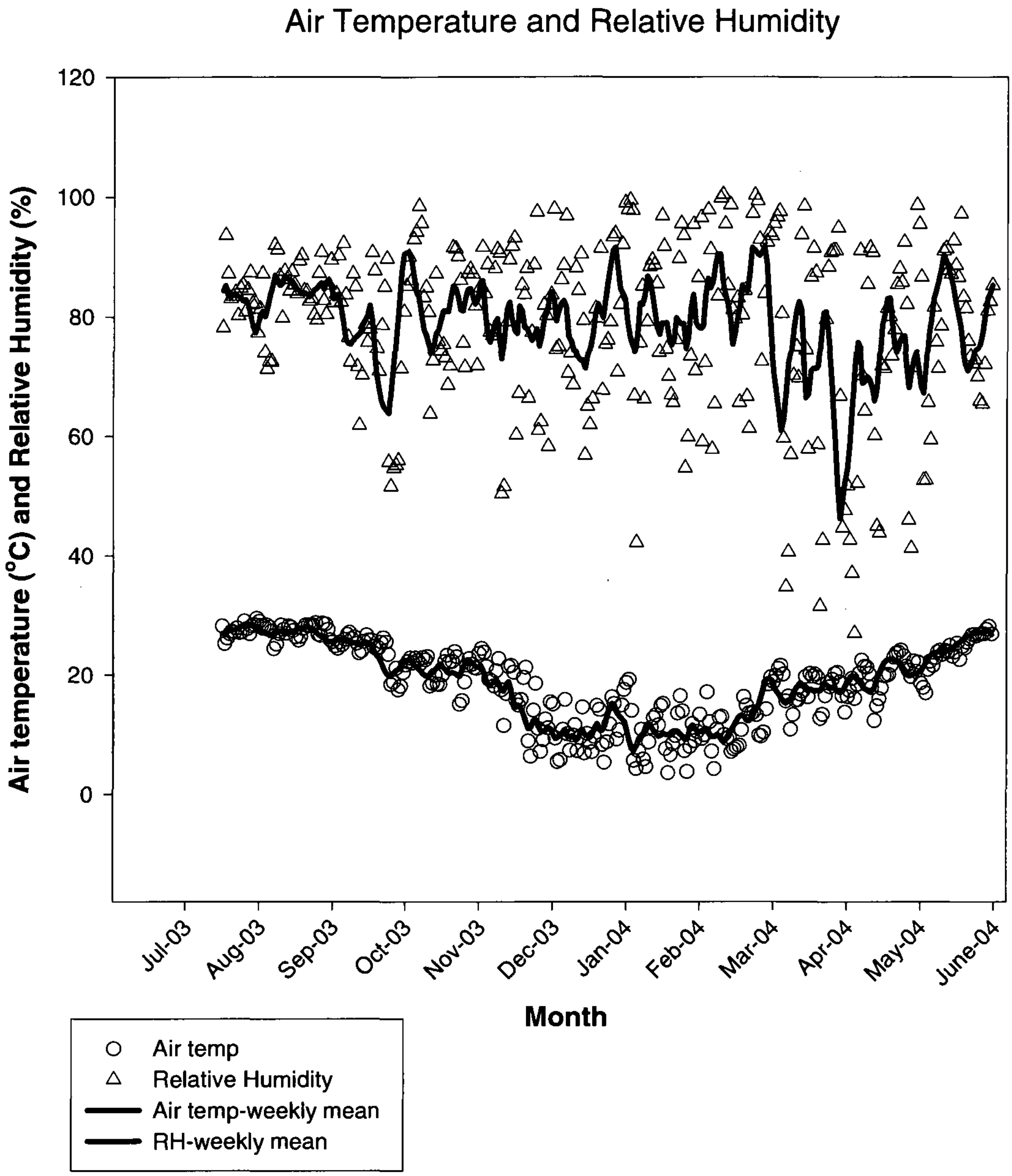

Figure 4.1 Average Daily Air Temperature and relative Humidity Recorded Every Day from July 2003 to May 2004 at HOBO Weather Station near Mallini Point, MS. Circles denote average air temperature for each day, while the triangles denote average daily relative humidity. Solid black and blue lines are the weekly means of air temperature and relative humidity respectively. Highest air and dew point temperatures were in summer while coldest temperatures were observed in winter. Relative humidity was highest in the winter in February 2004 and lowest in April 2004.

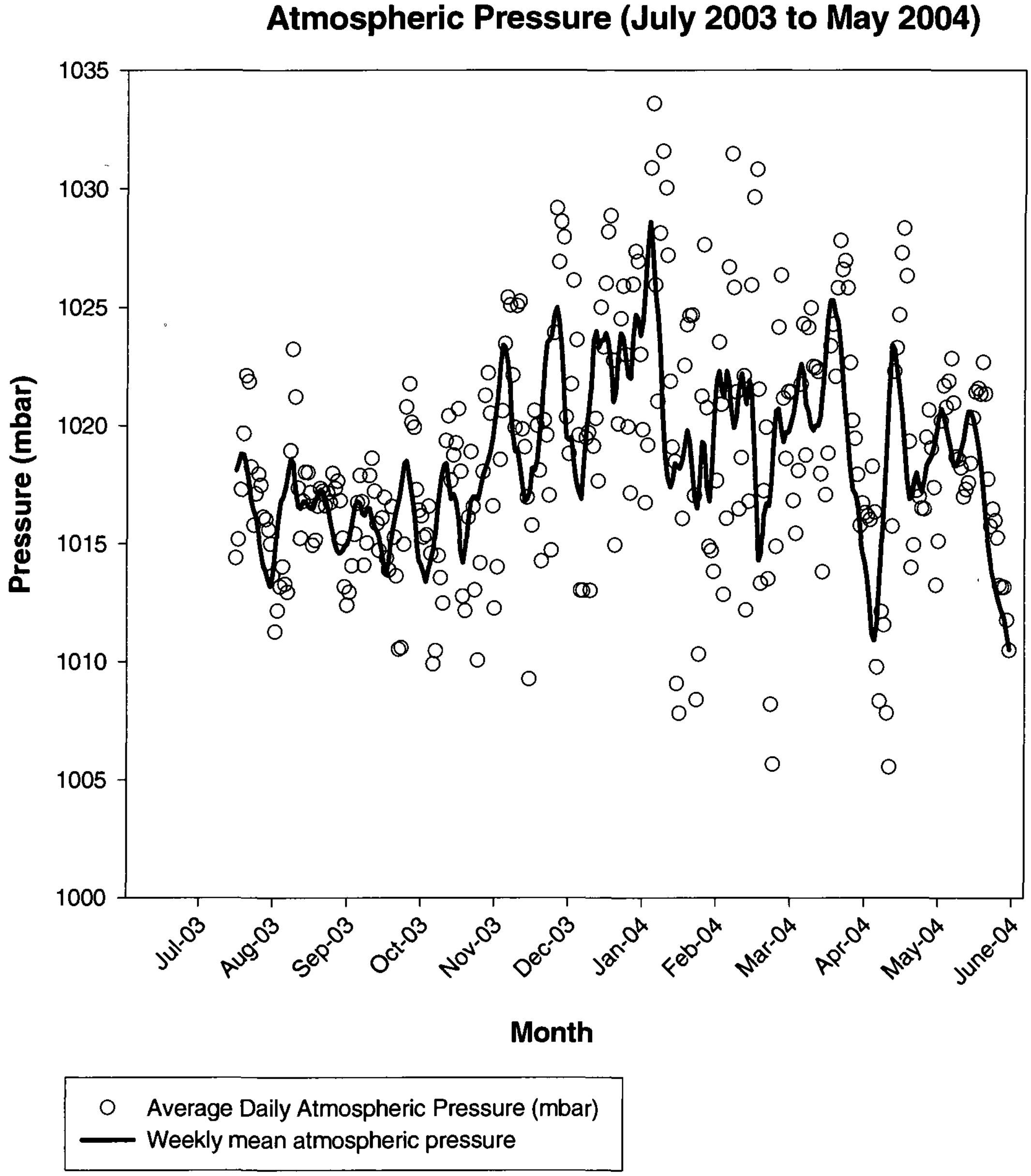

Figure 4.2 Average Daily Atmospheric Pressure Recorded Every Day from July 2003 to May 2004 at HOBO Weather Station near Mallini Point, MS. Circles denote average air pressure for each day. Solid line represents the weekly mean atmospheric pressure. Air pressure increased to its highest recorded value in winter, in January 2004 while the lowest atmospheric pressure was recorded in spring in April 2004.

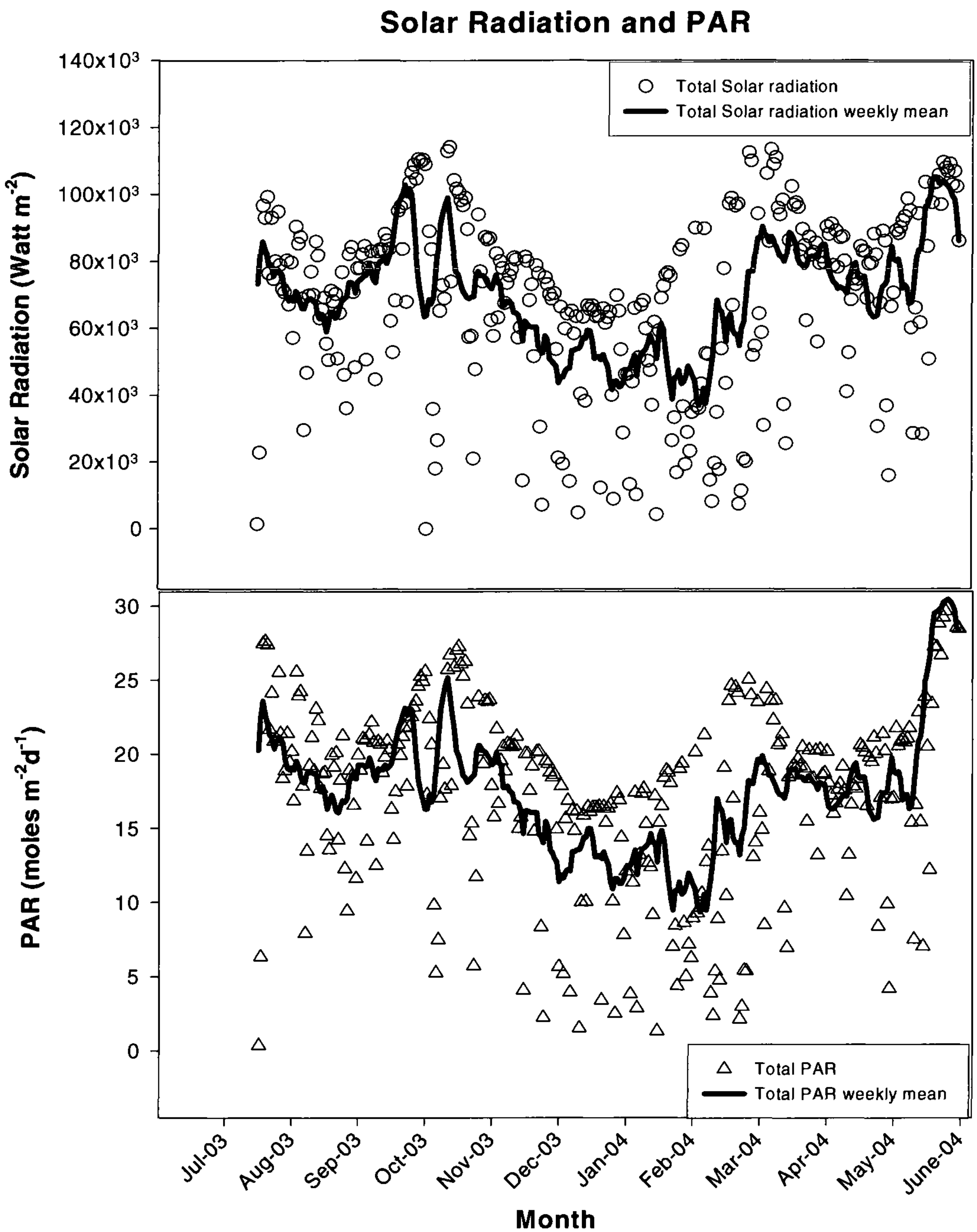

Figure 4.3 Total Daily Solar Radiation and Photosynthetically Active Radiation (PAR) Recorded from July 2003 to May 2004 at HOBO Weather Station near Mallini Point, MS. Circles denote total daily solar radiation, while triangles denote total daily PAR. Solid black and blue lines are the weekly means of solar radiation and PAR respectively. Highest PAR and solar radiation were measured in May 2004 and October 2003 respectively. The lowest values were recorded in the summer in July 2003. Values of both parameters also increased in July and October 2003 and in February-March 2004.

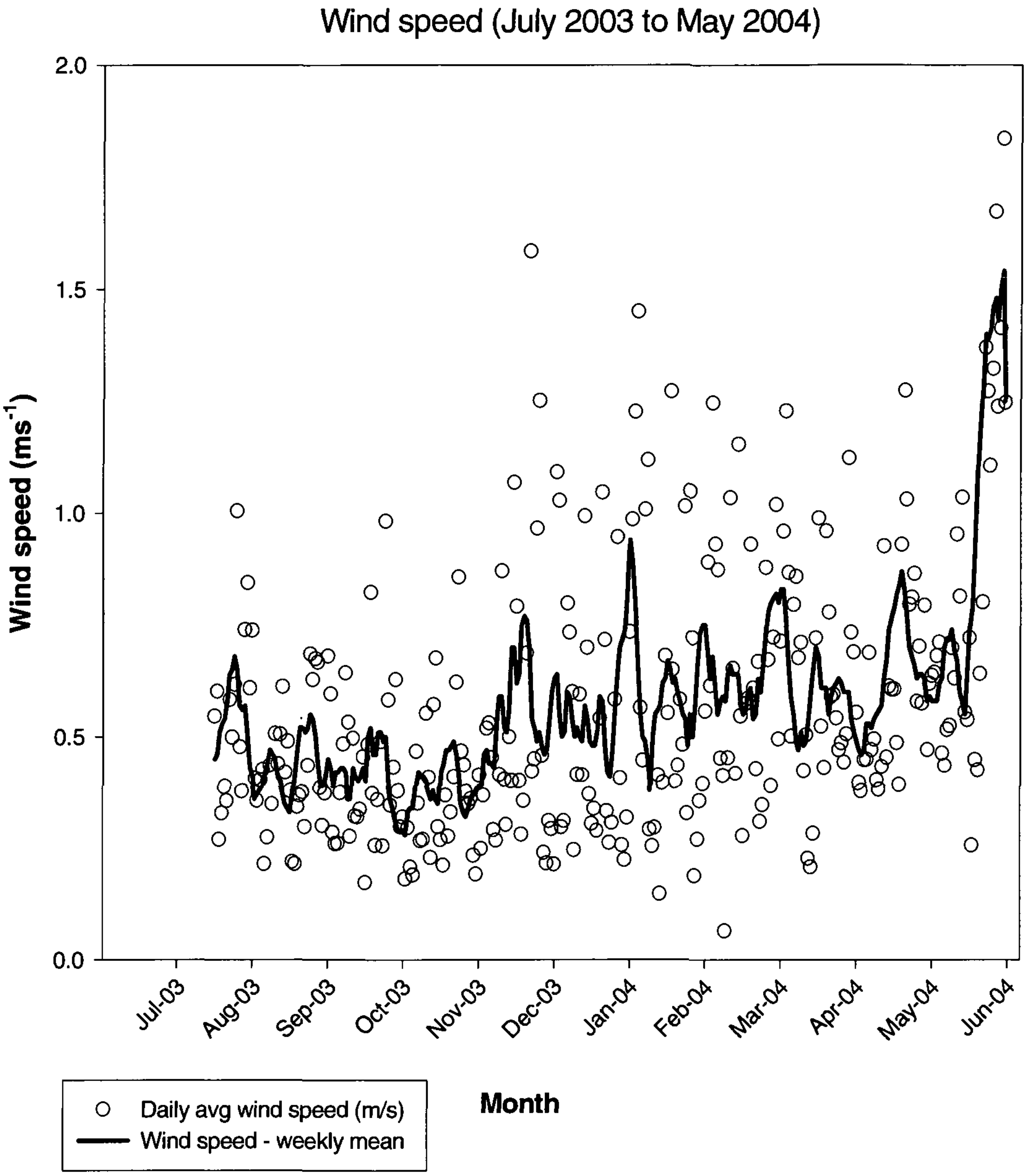

Figure 4.4 Average Daily Wind Speed Recorded Every Day from July 2003 to May 2004 at HOBO Weather Station near Mallini Point, MS. Circles denote average wind speed for each day. Solid line represents the weekly mean wind speed. Highest wind speed of 1.84 ms^{-1} was recorded in spring, in May 2004. Lowest wind speed was measured in February 2004.

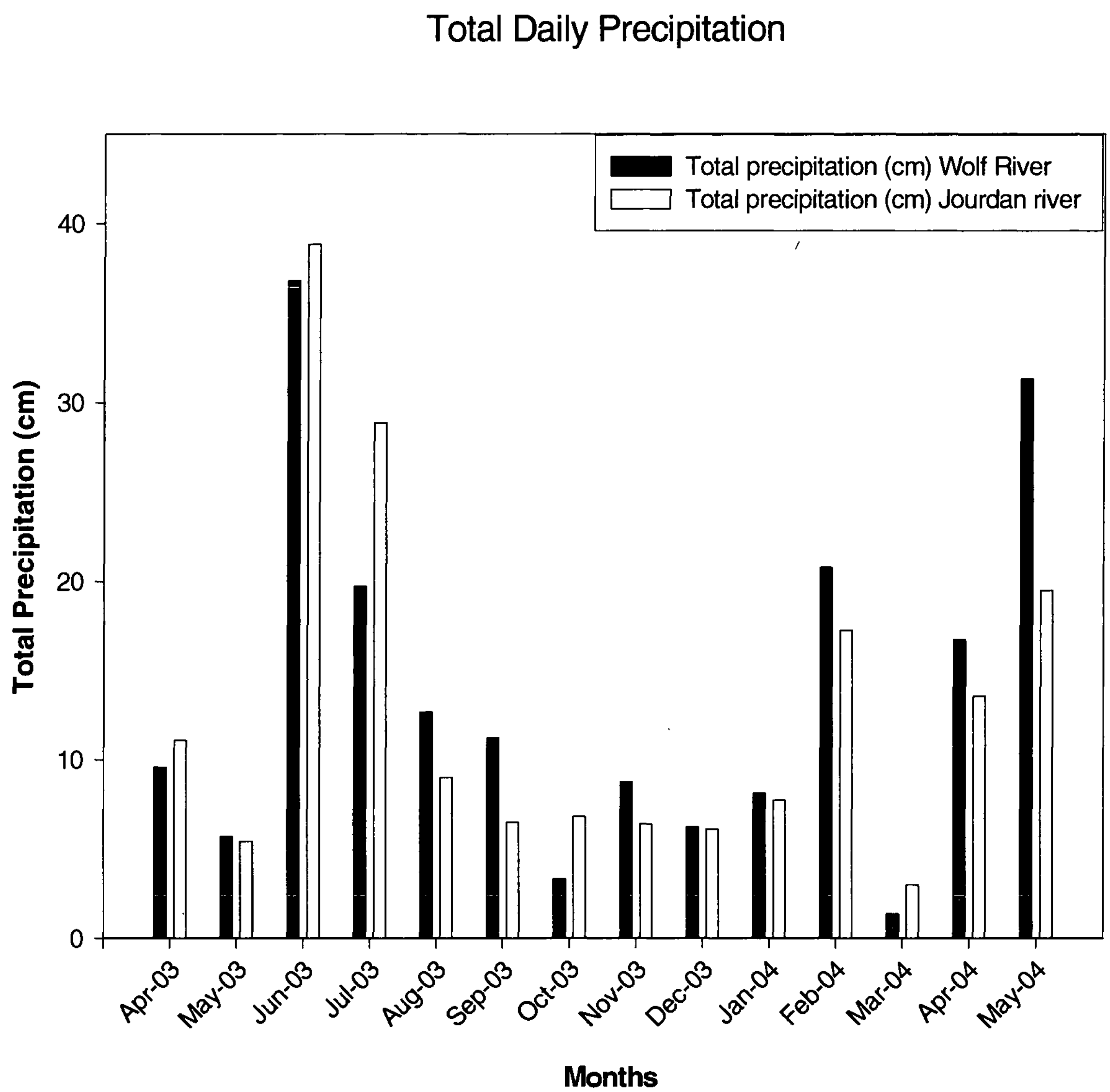

Figure 4.5 Total Daily Precipitation (cm) Measured at two Different Sites, USGS Jourdan River Station near Bay St. Louis, MS and USGS Wolf River Station near Landon, MS from April 2003 to May 2004. Increased precipitation in the area was recorded in June-July 2003 and again in February and May 2004 at both the USGS sites.

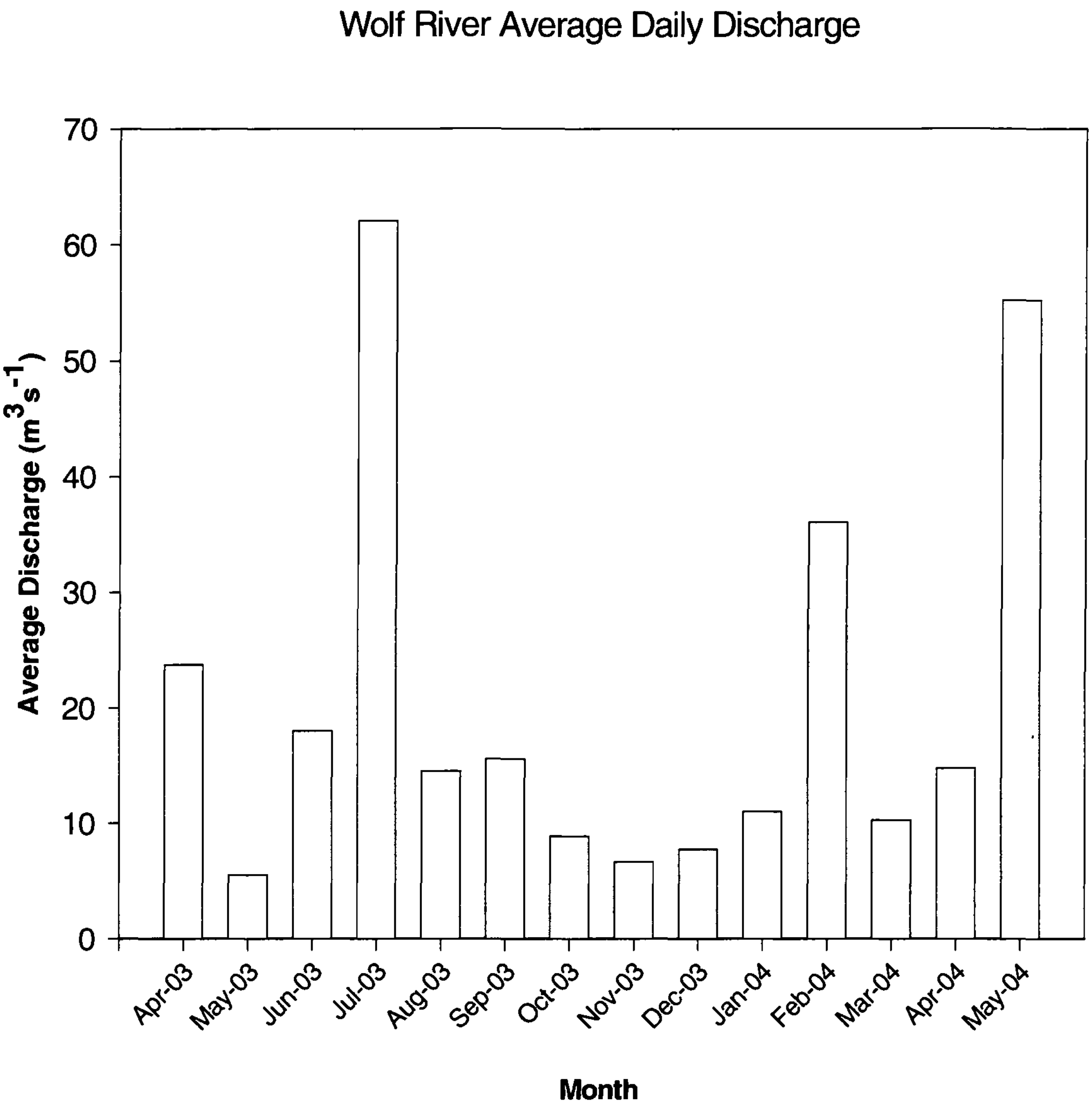

Figure 4.6 Average Daily Discharge for Wolf River Measured at USGS Station near Landon, MS. Increase in river discharge followed an increase in total precipitation. Highest river flow was measured in July 2003. River discharge was also high in February and May 2004.

Temporal Variability in Environmental Quality Parameters

Significant temporal changes were also seen in the environmental quality parameters measured twice every month from April 2003 to May 2004 in the bay (Table 4.1). A seasonal change in water temperature was observed as waters got warmer in summer and cooler in winter. Water temperature over the year was in the range of 9.7°C to 33.0°C (Figure 4.7). Maximum and highest median temperature (32.1°C) were recorded in June 2003. Lowest median (10.2 °C) and minimum water temperatures were reported in February 2004. Water temperature in the bay was correlated significantly with weather parameters such as air temperature ($r = 0.931$, $n = 190$, $p < 0.01$), dew point ($r = 0.804$, $n = 190$, $p < 0.01$), relative humidity ($r = 0.463$, $n = 190$, $p < 0.01$), total daily precipitation (JR) ($r = 431$, $n = 260$, $p < 0.01$), total daily precipitation (WR) ($r = 0.284$, $n = 260$, $p < 0.01$), and daily average discharge (WR) ($r = 0.148$, $n = 260$, $p < 0.05$). Water temperature was correlated negatively with atmospheric pressure ($r = -0.736$, $n = 190$, $p < 0.05$), gage height ($r = -0.197$, $n = 230$, $p < 0.01$), salinity ($r = -0.366$, $n = 260$, $p < 0.01$), pH ($r = -0.378$, $n = 260$, $p < 0.01$), and DO ($r = -0.505$, $n = 260$, $p < 0.01$). Salinity ranged from 0.01 to 26 (Figure 4.8). Highest median salinity (19.14) was observed in November 2003. Lowest median value of salinity (0.28) occurred in July 2003. Low median salinity values over time coincided with high river flow in months of July 2003 and May 2004. Salinity had positive correlations with gage height ($r = 0.318$, $n = 230$, $p < 0.01$), pH ($r = 0.423$, $n = 260$, $p < 0.01$), and DO ($r = 0.237$, $n = 260$, $p < 0.01$). Salinity was, however, correlated negatively with turbidity ($r = -0.505$, $n = 204$, $p < 0.01$), total daily precipitation (JR) ($r = -0.188$, $n = 260$, $p < 0.01$), total daily precipitation

Table 4.1 Kruskal-Wallis One-Way Analysis of Variance on Ranks for Determining Temporal Variability in Environmental Quality Parameters Measured at Each of the Ten Locations Twice Every Month During the Sampling Period (2003-04). Kruskal- Wallis statistic H, degrees of freedom (df) and probability (p value) are presented. Probability values <0.05 (highlighted in bold) indicate a significant difference in measured parameters between one or more stations. Significant temporal variability was observed in all measured parameters including temperature, salinity, pH, DIN, DIP, silicate, chlorophyll *a*, phaeopigment, and DO concentrations at all stations in the bay.

Parameter	K-W statistic (H)	Degrees of freedom (df)	Probability (p)
Temperature	264.25	26	**0.001**
Salinity	211.06	25	**0.001**
pH	133.96	25	**0.001**
DIN	109.59	23	**0.001**
DIP	106.64	26	**0.001**
Silicate	209.50	26	**0.001**
Chlorophyll *a*	111.55	22	**0.001**
Phaeopigments	145.66	22	**0.001**
Turbidity	135.61	20	**0.001**
DO	227.68	25	**0.001**

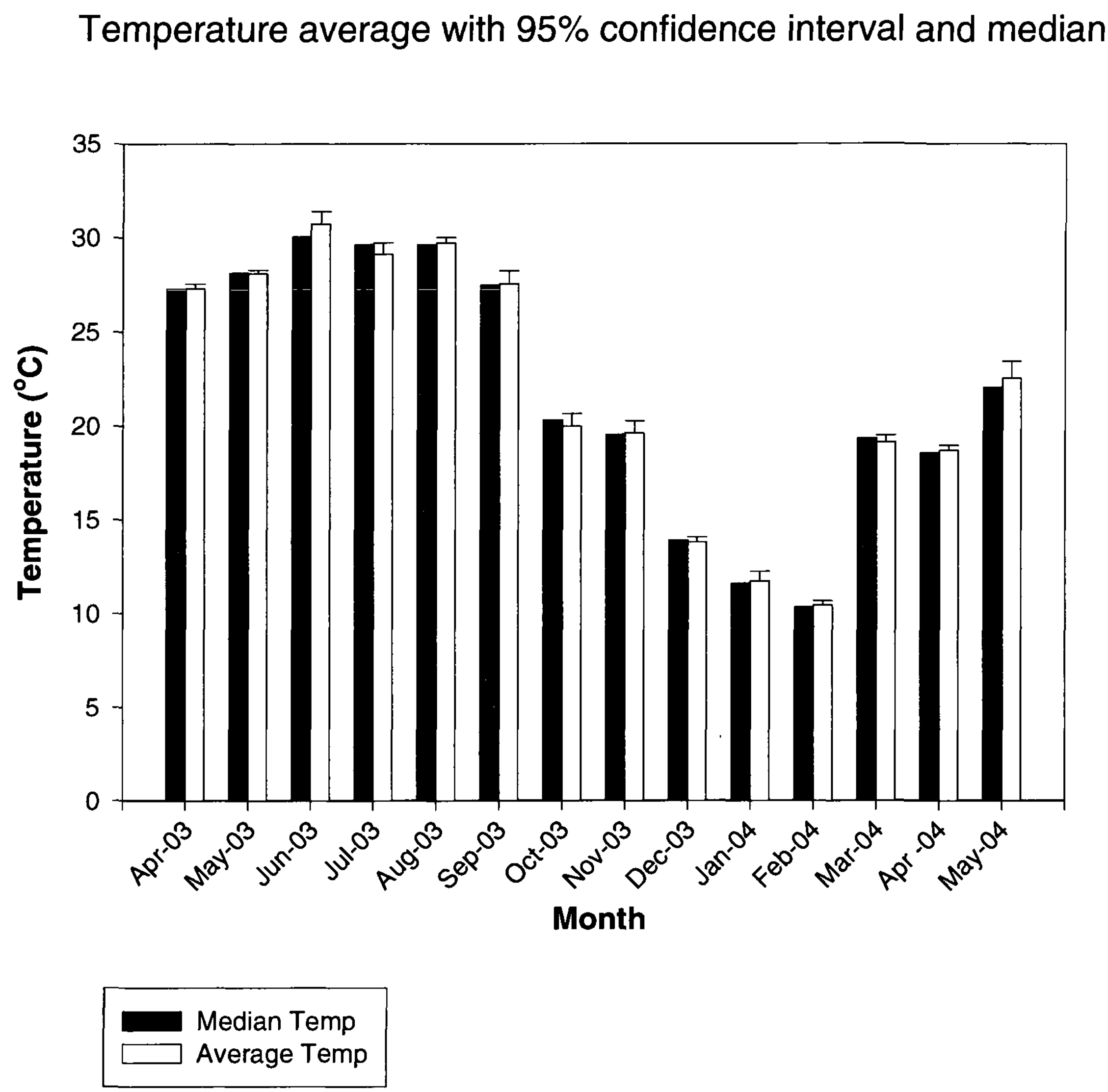

Figure 4.7 Vertical Bar Plot of Water Temperature (°C) Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Maximum and highest median temperatures were recorded in summer in June 2003. Lowest median and minimum water temperatures were measured in the winter in February 2004. **Note:** The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing individual sampling days.

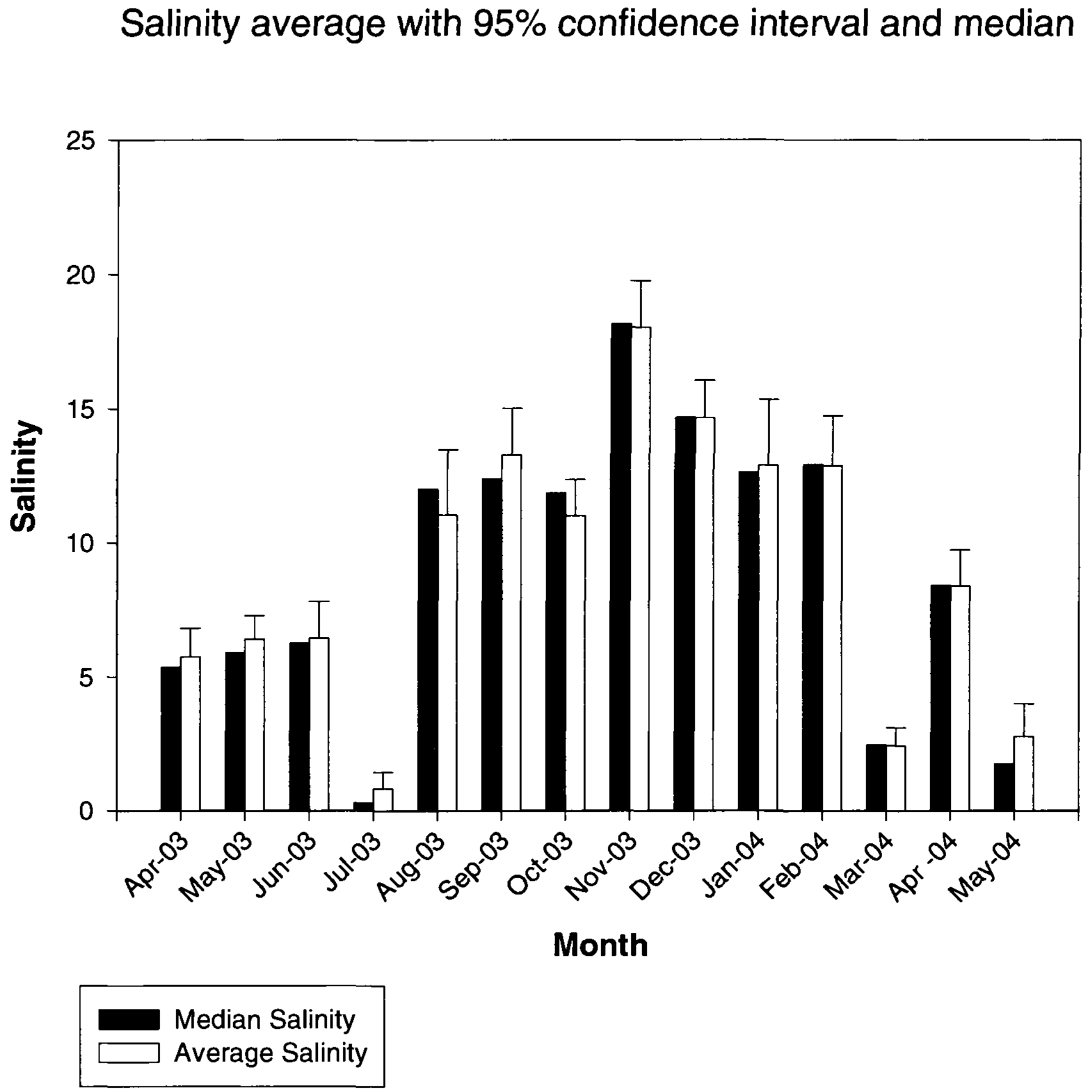

Figure 4.8 Vertical Bar Plot of Salinity Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Bay-wide salinity values were lowest in July 2003 followed by May 2004, the months of highest discharge. Highest median salinity was measured in November 2003. **Note:** The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing individual sampling days.

(JR) ($r = -0.188$, $n = 260$, $p < 0.01$), and daily average discharge (WR) ($r = -0.413$, $n = 260$, $p < 0.01$). Highest significant correlation of salinity with daily average discharge was one day after rainfall ($r = -0.450$, $n = 260$, $p < 0.01$).The pH in the bay varied from slightly acidic in summer (lowest pH of 5.57 recorded in July 2003) to 8.35 in the spring (March 2004), the highest pH value measured in 2003-04 (Figure 4.9). Lowest median pH (6.74), like salinity, was also recorded in July 2003 while highest median pH (7.96) was recorded in February 2004. Also, pH was correlated negatively with average daily discharge (WR) ($r = -0.252$, $n = 260$, $p < 0.01$) and correlated positively with gage height ($r = 0.285$, $n = 230$, $p < 0.01$).

Significant temporal variability was also observed in nutrient concentrations in the bay. Maximum DIN concentration (12.32 μM) was measured in August 2003 (Figure 4.10). This increase in DIN concentrations followed an increase in river discharge. The highest median value of DIN (7.00 μM) occurred in May 2004. Lowest median DIN concentration (0.03μM) was observed in January 2004. DIN concentrations were correlated positively with average daily wind speed ($r = 0.256$, $n = 200$, $p < 0.01$), water temperature ($r = 0.167$, $n = 230$, $p < 0.05$), average daily discharge ($r = 0.395$, $n = 240$, $p < 0.01$), and turbidity ($r = 0.412$, $n = 175$, $p < 0.01$). DIN concentrations were correlated negatively with salinity($r = -0.520$, $n = 230$, $p < 0.01$), pH($r = -0.421$, $n = 230$, $p < 0.01$), and gage height ($r = -0.214$, $n = 230$, $p < 0.01$).

DIP concentrations, on the other hand, increased in late fall with maximum (2.63 μM) and highest median concentrations (0.86 μM) observed in November 2003 (Figure 4.11). Lowest median DIP concentration (0.01 μM) occurred in January 2004. Thus, the

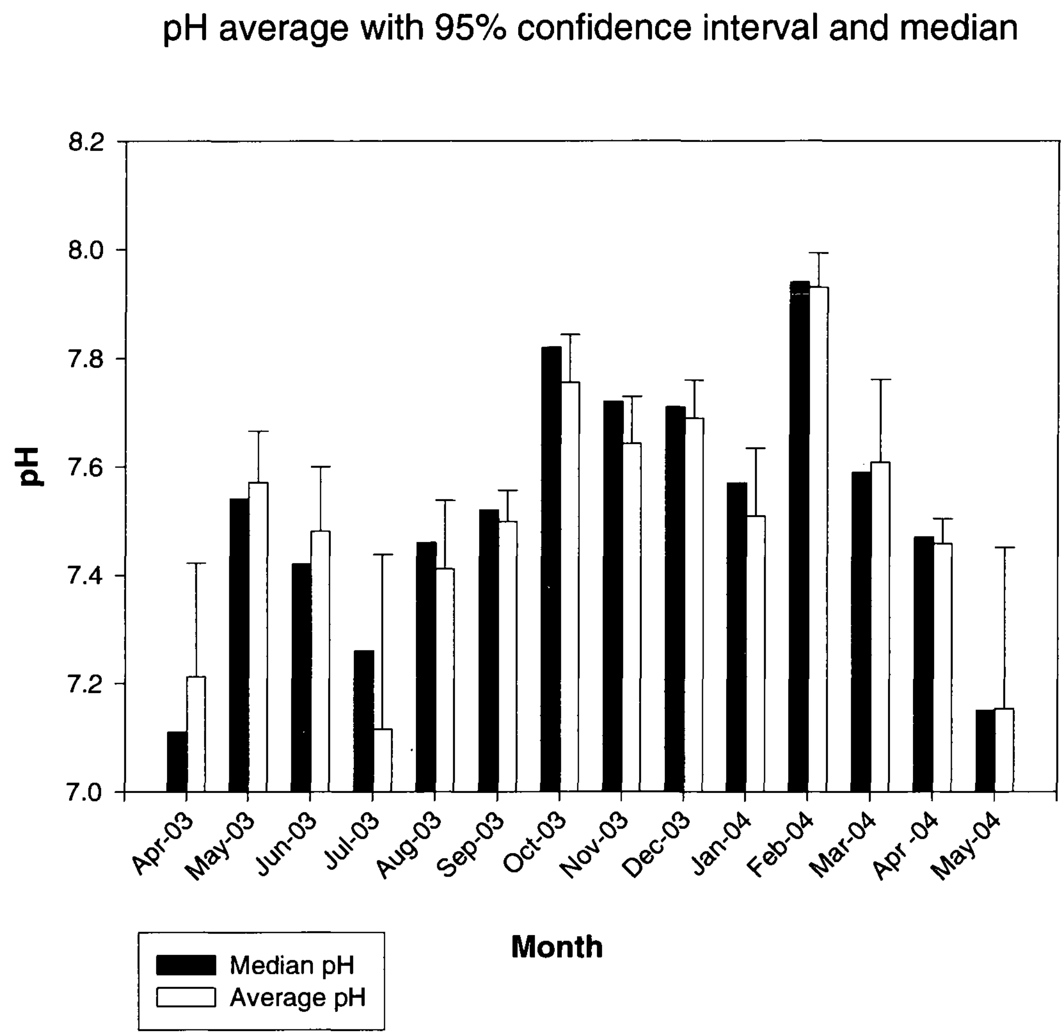

Figure 4.9 Vertical Bar Plot of pH Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Minimum pH and lowest median pH (among individual sampling days) were recorded in July 2003. The highest median pH was measured in February 2004. **Note:** The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing individual sampling days.

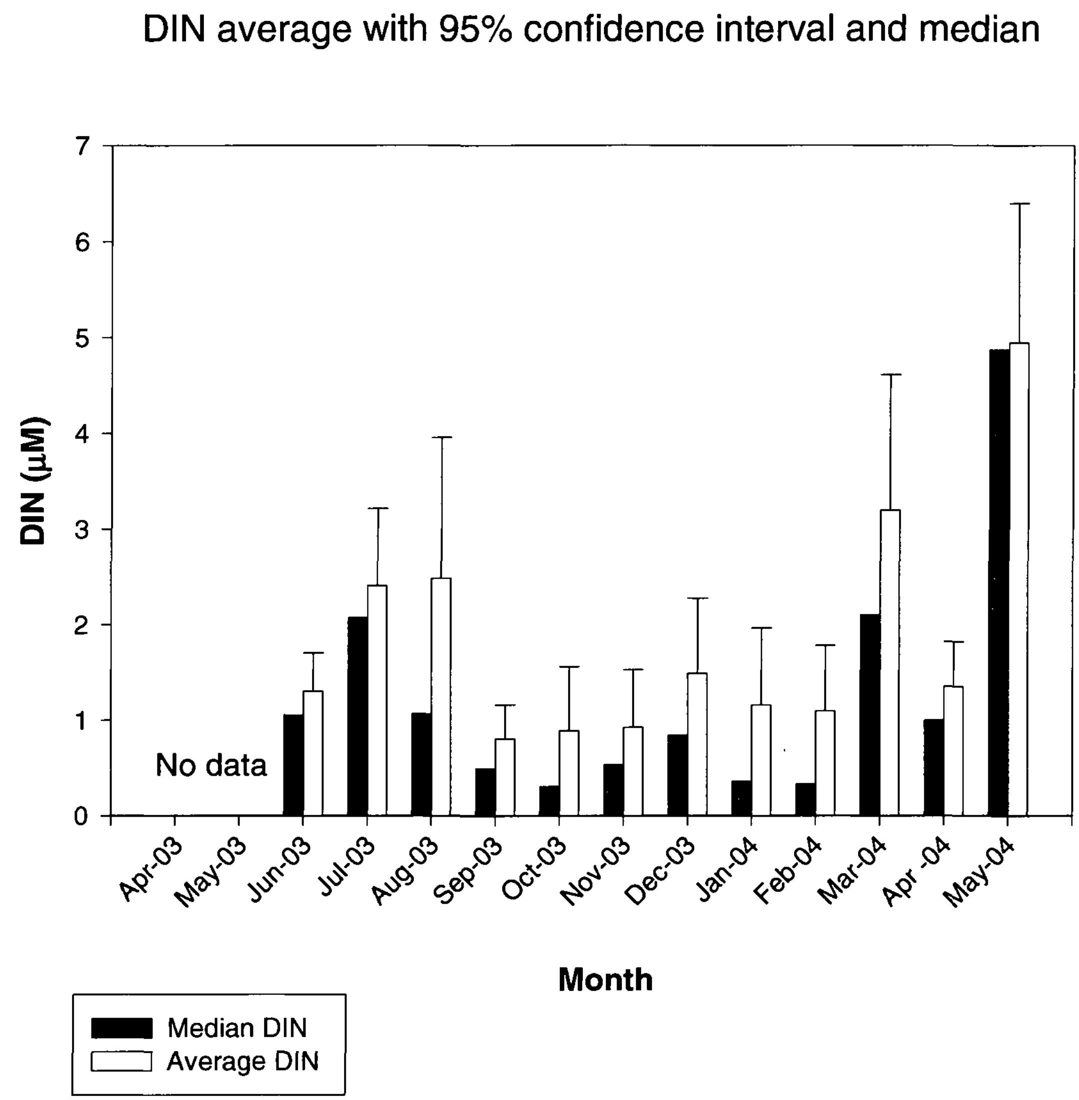

Figure 4.10 Vertical Bar Plot of DIN Concentrations Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. The highest median DIN concentration was measured in May 2004 followed by July 2003, months of high river flow. Maximum DIN concentration was measured in August 2003. The lowest median concentration of DIN (among individual sampling days) was observed in winter in January 2004. **Note:** Data were note available for the months of April and May 2003. The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing individual sampling days.

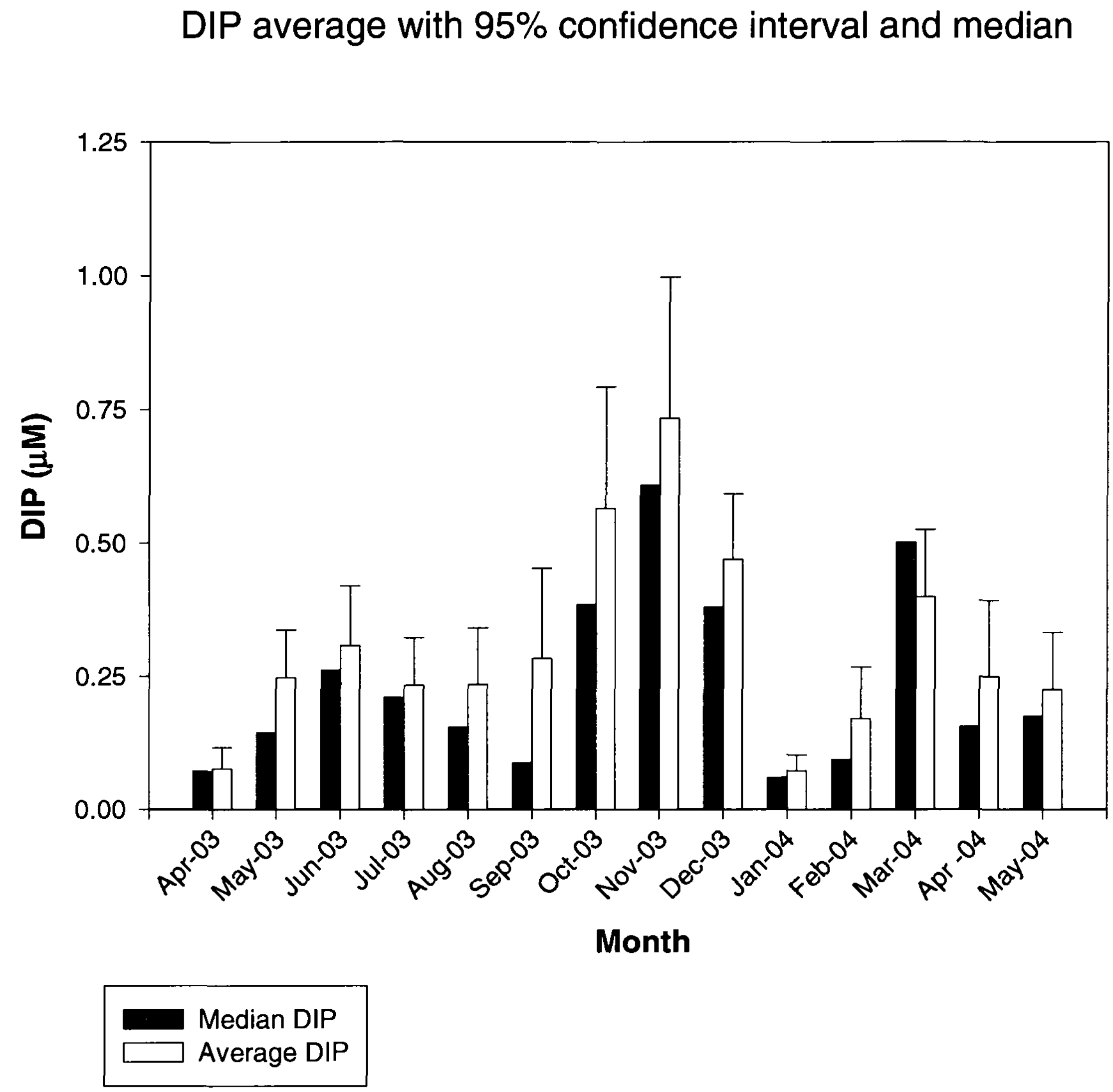

Figure 4.11 Vertical Bar Plot of DIP Concentrations Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Maximum and highest median DIP concentrations were measured in November 2003. The lowest median DIP concentration was measured in the winter in January 2004. **Note:** The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing all individual sampling days.

lowest median concentrations for both nutrients (DIN and DIP) were observed in winter, in January 2004. Significant positive correlations of DIP concentrations were observed with salinity($r = 0.334$, $n = 260$, $p < 0.01$), pH ($r = 0.259$, $n = 260$, $p < 0.01$), wind speed ($r = 0.216$, $n = 200$, $p < 0.01$), and gage height ($r = 0.195$, $n = 240$, $p < 0.01$). DIP concentrations were correlated negatively with average daily discharge (WR) ($r = -0.148$, $n = 270$, $p < 0.05$).

Silicate concentrations in the bay ranged from below detection levels to 199.42 μM (Figure 4.12). The maximum and the highest median (147.78 μM) silicate concentrations were observed in April 2003. Lowest median silicate concentration of 4.89 μM was measured in March 2004. Silicate concentrations were below detection limits in May 2004. Silicate concentrations in the bay were correlated positively to DIN concentrations ($r = 0.241$, $n = 240$, $p < 0.01$), chlorophyll *a* and phaeopigments ($r = 0.311$, $n = 230$, $p < 0.01$and $r = 0.217$, $n = 230$, $p < 0.01$ respectively), water temperature($r = 0.439$, $n = 260$, $p < 0.01$), and daily average discharge ($r = 0.166$, $n = 270$, $p < 0.01$). Negative correlations of silicate concentrations were observed with DIP concentrations ($r = -0.355$, $n = 270$, $p < 0.01$), salinity ($r = -0.373$, $n = 260$, $p < 0.01$), DO concentrations ($r = -0.322$, $n = 260$, $p < 0.01$), pH ($r = -0.318$, $n = 260$, $p < 0.01$), turbidity ($r = -0.218$, $n = 204$, $p < 0.01$), and gage height ($r = -0.337$, $n = 200$, $p < 0.01$).

Chlorophyll *a* values varied significantly throughout the year (Figure 4.13). Concentrations ranged from 0.12 μgL^{-1} to 56.08 μgL^{-1}, with the highest value observed in summer (July 2003). Highest median chlorophyll *a* concentration (10.12 $\mu g\ L^{-1}$) occurred in June 2003 while the lowest median value (0.28 μgL^{-1}) was seen during winter in January 2004. Concentrations of chlorophyll *a* increased during periods of high DIN concentrations and high river discharge. Lowest median chlorophyll *a* values occurred in winter (January 2004), also the period of lowest median phosphate and nitrogen nutrient concentrations. Phaeopigment concentrations also decreased significantly in the winter with lowest median concentration (0.95 μgL^{-1}) measured in January 2004. The highest median (16.97 μgL^{-1}) and maximum (29.02 μgL^{-1}) phaeopigment concentrations, however, were observed in March 2004. Chlorophyll *a* concentrations had significant positive correlations with water temperature ($r = 0.426$, $n = 220$, $p < 0.01$), PAR ($r = 0.142$, $n = 200$, $p < 0.05$), wind speed ($r = 0.148$, $n = 200$, $p < 0.05$), daily average discharge (WR) ($r = 0.286$, $n = 230$, $p < 0.01$), and DIN concentrations ($r = 0.231$, $n = 240$, $p < 0.01$). Significant negative correlations of chlorophyll *a* concentrations were found with salinity ($r = -0.197$, $n = 220$, $p < 0.01$) and pH ($r = -0.290$, $n = 220$, $p < 0.01$).

Phaeopigment concentrations were also correlated positively with water temperature ($r = 0.271$, $n = 220$, $p < 0.01$), turbidity ($r = 0.219$, $n = 165$, $p < 0.01$), PAR ($r = 0.155$, $n = 200$, $p < 0.05$) and solar radiation ($r = 0.168$, $n = 200$, $p < 0.05$). Phaeopigments were correlated negatively with salinity ($r = -0.339$, $n = 220$, $p < 0.01$), DO ($r = -0.436$, $n = 220$, $p < 0.01$), gage height ($r = -0.280$, $n = 230$, $p < 0.01$), and total daily precipitation (WR) ($r = -0.143$, $n = 230$, $p < 0.05$).

Turbidity in the bay increased to its highest median value (29.43) in May 2004, also the month of high river flow and highest wind speed. Turbidity values throughout the bay ranged from 0.5 to 89 NTU (Figure 4.14). Maximum turbidity was observed in April 2004, while minimum turbidity was reported in June 2003. Lowest median turbidity (2.11) was measured in June 2003. Turbidity was correlated positively with

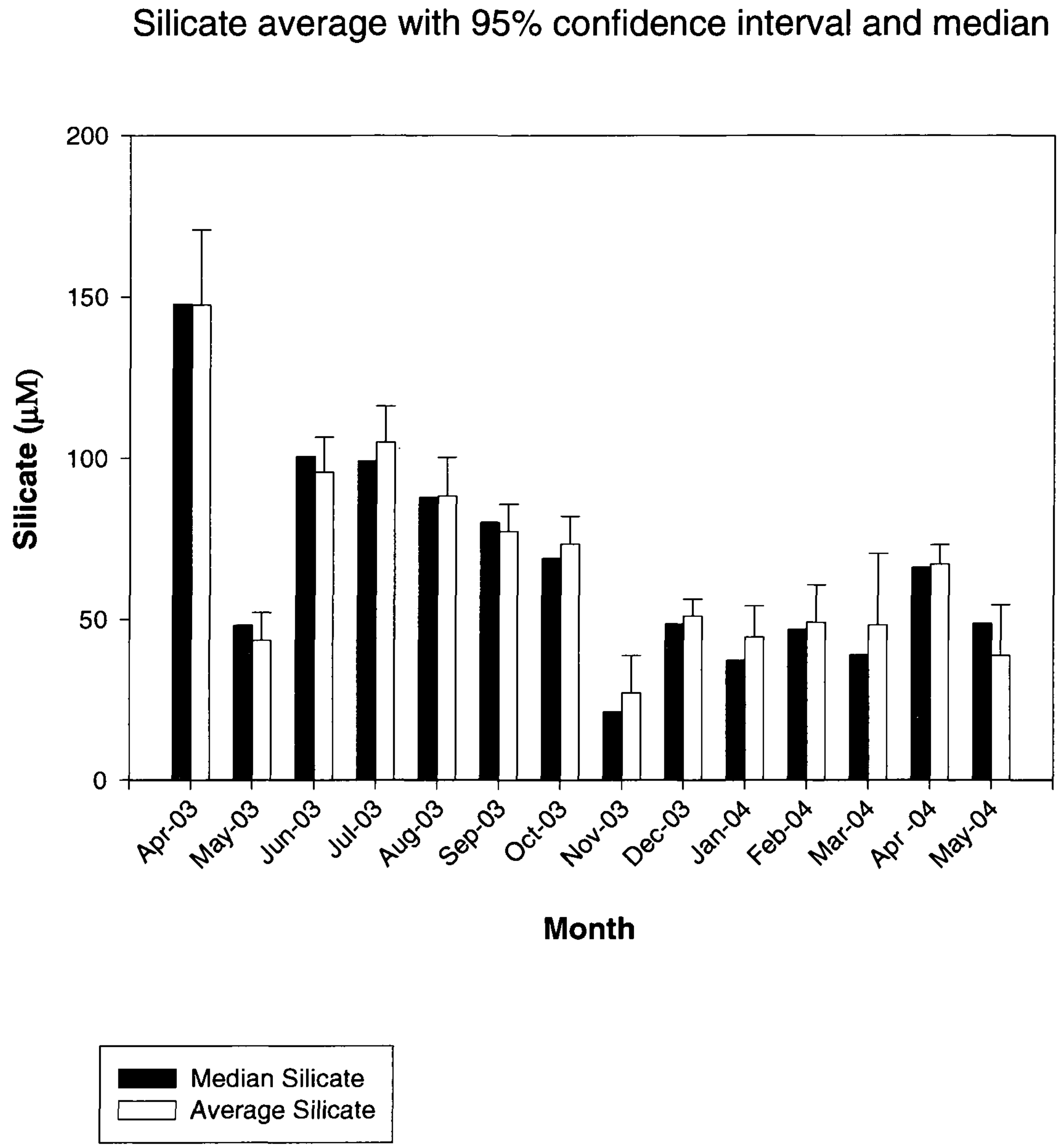

Figure 4.12 Vertical Bar Plot of Silicate Concentrations Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Highest median and maximum silicate concentrations were measured in April 2003. The lowest median value (among individual sampling days) was observed in March 2004, while silicate concentrations were minimum (below detection) in May 2004. **Note:** The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing all individual sampling days.

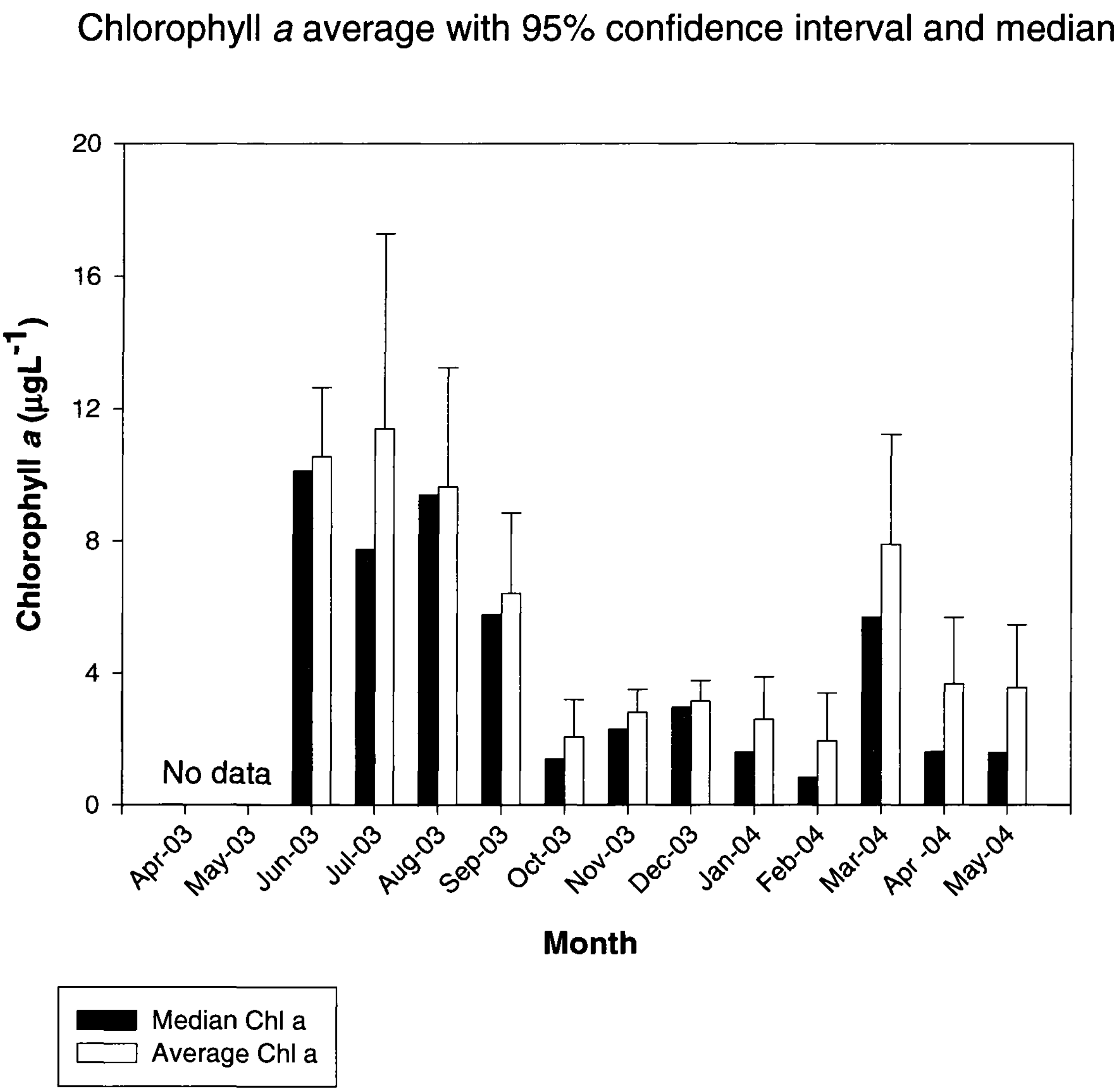

Figure 4.13 Vertical Bar Plot of Chlorophyll *a* Concentrations Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Highest median chlorophyll *a* concentrations were observed in June 2003, while lowest median concentrations ((among individual sampling days) were measured in January 2004. Maximum chlorophyll *a* concentration was measured in July 2003. **Note:** Data were note available for the months of April and May 2003. The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing all individual sampling days.

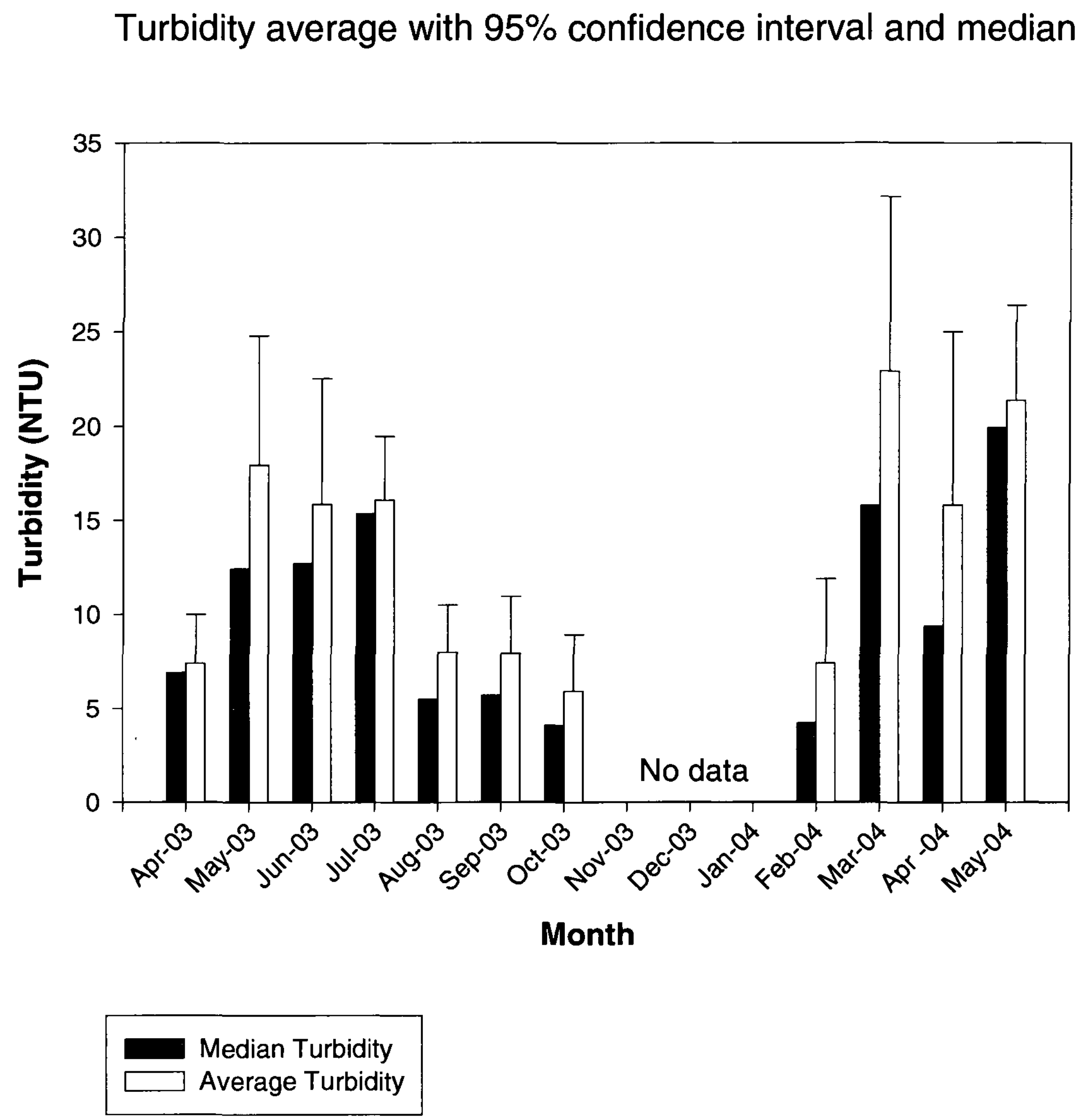

Figure 4.14 Vertical Bar Plot of Turbidity Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. Median turbidity values were high in the months of high river discharge and wind speed. Highest median turbidity was measured in May 2004, while maximum turbidity was observed in April 2004. Minimum and lowest median turbidity values (among individual sampling days) were measured in June 2003. **Note:** Data were not available from November 2003 to January 2004. : The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing all individual sampling days.

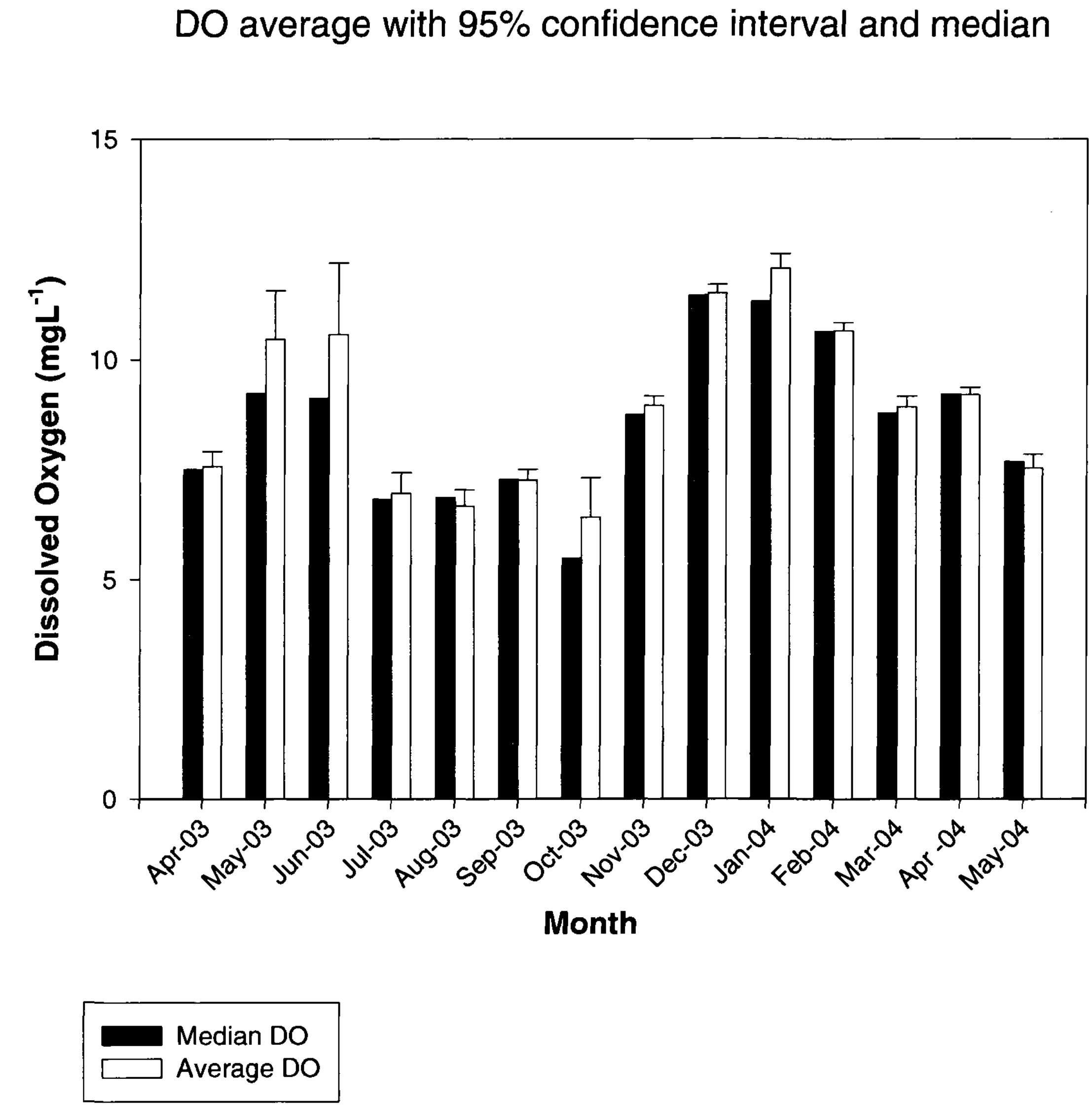

Figure 4.15 Vertical Bar Plot of DO Concentrations Measured Each Month. Plot includes data collected every month (two sampling days combined) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95% confidence interval of the mean for each month. The highest median DO concentration was measured in January 2004. Lowest median DO concentration (among individual sampling days) occurred in October 2003. : The median values presented in the graph are median of monthly data, where two sampling days are combined and are not the same values as the median of data representing all individual sampling days.

wind speed and gust speeds ($r = 0.340$, $n = 137$, $p < 0.01$ and $r = 0.169$, $n = 137$, $p < 0.05$) and total daily precipitation (WR) ($r = 0.284$, $n = 260$, $p < 0.01$).

DO concentrations in the bay changed in inverse relation to water temperature (Figure 4.15). Concentrations ranged from 4.54 mg L^{-1} (141.88 μmol kg^{-1}) to 22.3 mgL^{-1} (696.90 μmol kg^{-1}). (Note: This value was high but was not considered an outlier since no technical errors were found in the sensor. Supersaturation was often observed in the bay). Highest median DO value (12.10 mg L^{-1} or 378.14 μmol kg^{-1}) occurred in winter in January 2004, thus corresponding with cooler temperatures. Lowest median DO value (4.73 mg L^{-1} or 147.82 μmol kg^{-1}) occurred in fall in October 2003. The maximum DO concentration was measured in June 2003. Positive correlations of DO concentrations were observed with salinity ($r = 0.237$, $n = 260$, $p < 0.01$), pH ($r = 0.252$, $n = 260$, $p < 0.01$), and gage height ($r = 0.228$, $n = 230$, $p < 0.01$), while negative correlations were found with water temperature ($r = 0.284$, $n = 260$, $p < 0.01$), total daily precipitation (recorded at Jourdan river and Wolf river) ($r = -0.256$, $n = 260$, $p < 0.01$ and $r = -0.318$, $n = 260$, $p < 0.01$ respectively) and average daily discharge ($r = -0.158$, $n = 260$, $p < 0.05$).

Spatial Variability in Environmental Quality Parameters

Environmental quality parameters measured at every station were individually tested for spatial variability using the Kruskal-Wallis Analysis of Variance (ANOVA) test (Table. 4.2). As expected in an estuarine environment, spatial variability was observed in salinity. Lower values were found at stations closer to river mouths (Stations 1 and 4) and higher salinities were seen at stations in and near the Mississippi Sound (Stations 8 and 9). Salinity ranged from 0.01 to 26 (Figure. 4.16). Highest median

Table 4.2 Kruskal-Wallis One-Way Analysis of Variance on Ranks for Determining Spatial Variability in Environmental Quality Parameters Measured at Each of the Ten Locations Over the Sampling Period (2003-04). Kruskal-Wallis statistic H, degrees of freedom (df) and probability (p value) are presented. Probability values <0.05 (highlighted in bold) indicate a significant difference in measured parameters between one or more stations. Significant spatial variability was observed in salinity, pH, and NO_2+NO_3, DIN and DIP concentrations, whereas, temperature, turbidity, ammonium, silicate, chlorophyll *a*, phaeopigment, and DO concentrations did not differ significantly throughout the bay.

Parameter	K-W statistic (H)	Degrees of freedom (df)	Probability (p)
Temperature	0.35	9	1.00
Salinity	27.92	9	**0.001**
pH	60.68	9	**0.001**
NO_2+NO_3	31.33	9	**0.001**
Ammonium (NH_4)	13.30	9	0.150
DIN	34.95	9	**0.001**
DIP	67.72	9	**0.001**
Silicate	16.69	9	0.054
Chlorophyll *a*	13.75	9	0.132
Turbidity	11.05	9	0.272
DO	3.66	9	0.931
Phaeopigments	7.45	9	0.591

salinity (13.81) and maximum salinity were observed at station 9. Station 1 had the lowest median salinity of 4.56 followed by station 4 (6.51). Minimum salinity was recorded at station 4 followed by station 1 (salinity = 0.02). Measured values of pH varied significantly between stations (Figure. 4.17). The highest median pH (7.79) and maximum pH (8.35) were measured at station 9, while the lowest median pH (7.30) was observed at station 6a. Minimum pH (5.57) was measured at station 1 near the mouth of the Jourdan River.

Nutrient concentrations in the bay also showed significant variability ($p < 0.05$) between sampling locations. NO_2+NO_3 concentrations varied significantly throughout the bay (Table. 4.2). The highest median NO_2+NO_3 concentration (2.64 μM) and maximum NO_2+NO_3 concentration (8.89 μM) were observed at station 1, while lowest median NO_2+NO_3 concentration (0.22 μM) was observed at station 8. A similar trend in variability among sampling stations was observed in DIN concentrations, since ammonium concentrations were not different significantly between stations. Total DIN concentrations ranged from 0 μM to 12.32 μM with highest values found along the western shore of the bay (Stations1and 6a) (Figure 4.18). High values were also found at station 4. The maximum DIN (12.32 μM) as well as highest median DIN concentrations (3.30 μM) were observed at station 1. The lowest median value (0.56 μM) occurred at station 8. The median concentration of DIN at station 1 was five times that at station 8. DIP concentrations ranged from 0 μM to 2.63 μM (Figure 4.19). The maximum DIP value was observed at station 5. High DIP values were also measured at stations 6a, 7, 8, and 9. The median DIP concentration at station 5 (0.60 μM) was also the highest amongst all locations in the bay. Station 3, on the other hand, had the lowest median DIP

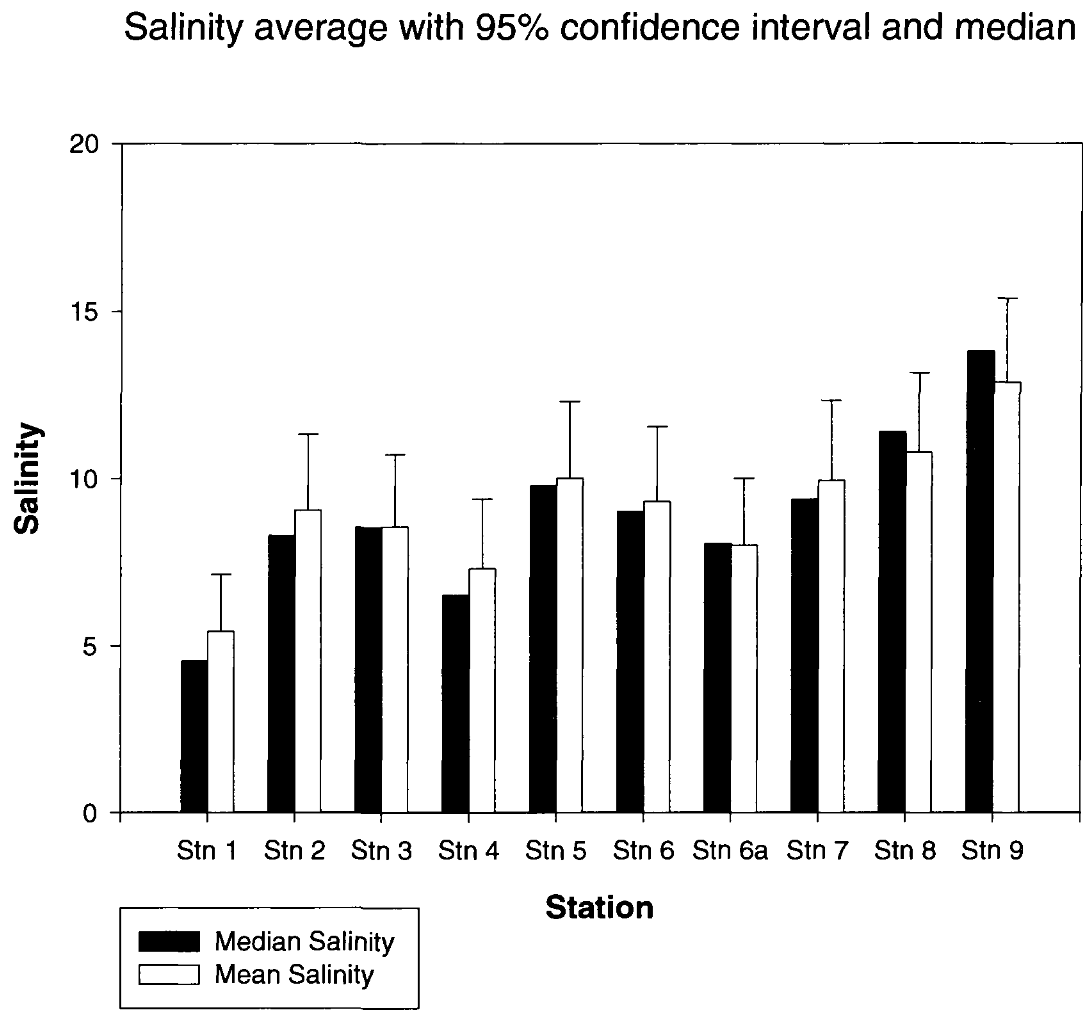

Figure 4.16 Vertical Bar Plot of Salinity Measured at Each Station. Plot includes data collected every sampling day (two days per month for fourteen months) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95 % confidence interval of the mean for each station. Highest median and maximum salinities were measured at station 9, while the lowest median salinity was recorded at station 1. Near zero salinities were recorded at both stations 1 and 4.

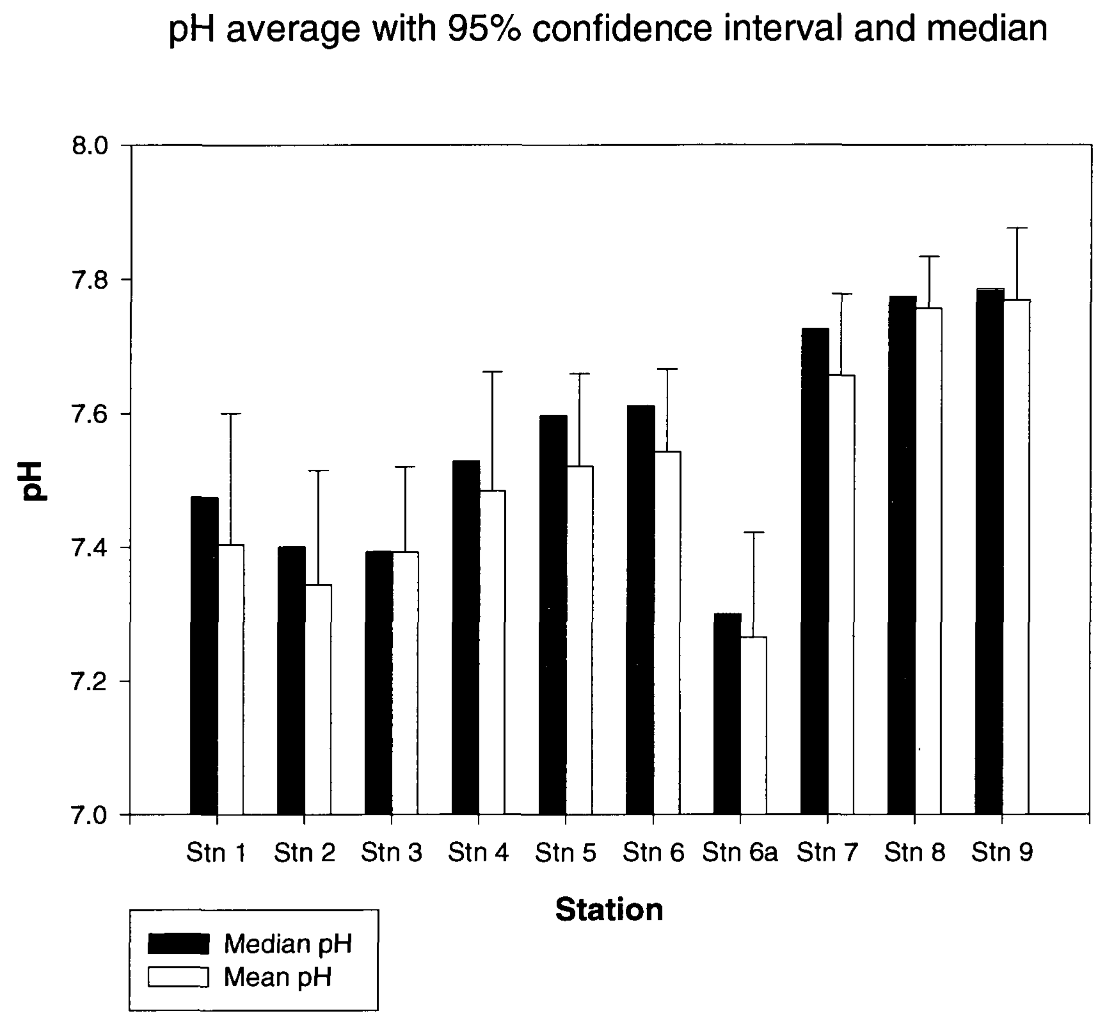

Figure 4.17 Vertical Bar Plot of pH Measured at Each Station. Plot includes data collected every sampling day (two days per month for fourteen months) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95 % confidence interval of the mean for each station. Maximum and highest median pH was recorded at station 9, while the lowest median pH was measured at station 6a. Minimum pH was recorded at station 1.

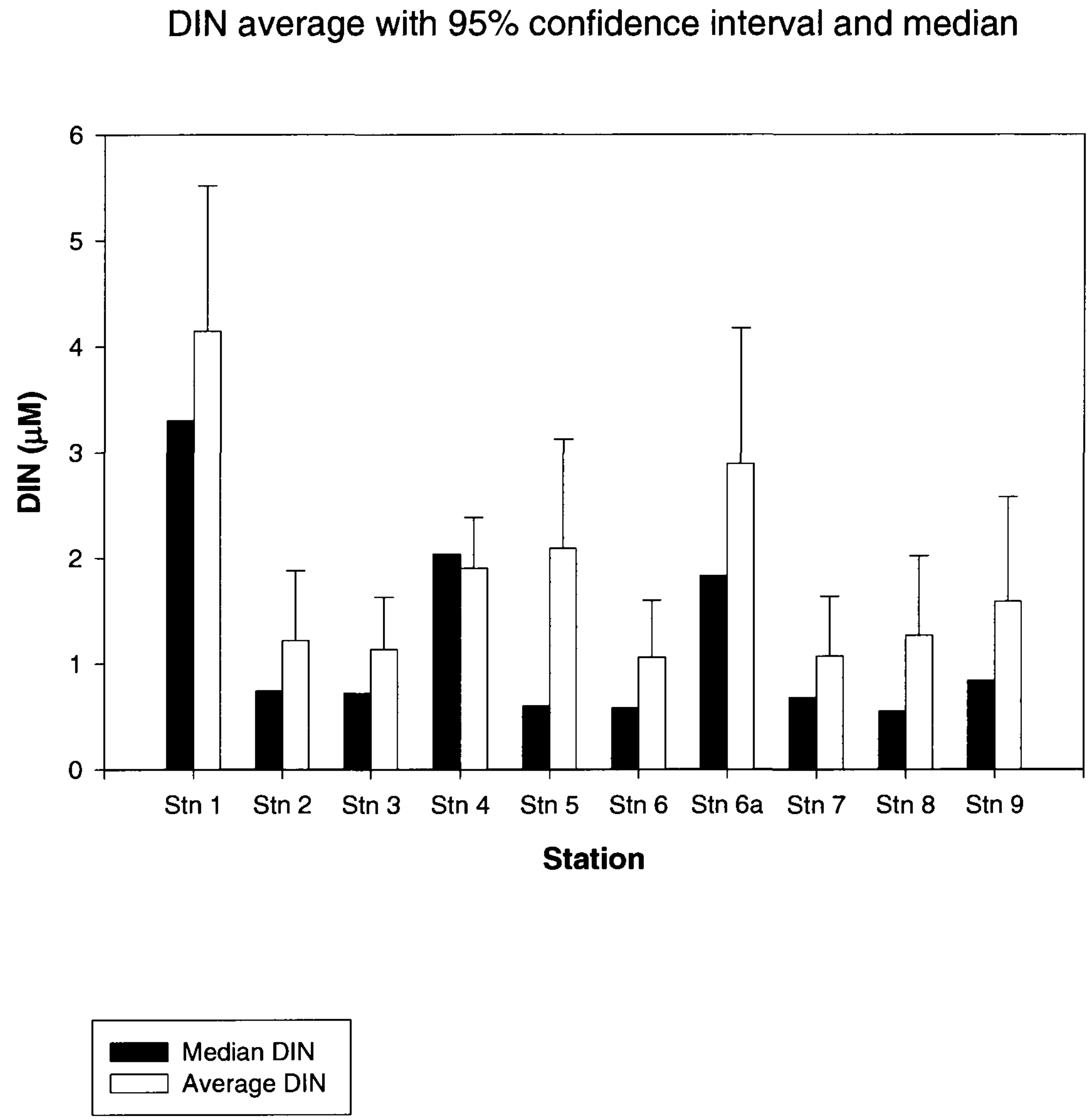

Figure 4.18 Vertical Bar Plot of DIN Concentrations Measured at Each Station. Plot includes data collected every sampling day (two days per month for fourteen months) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95 % confidence interval of the mean for each station. Maximum and highest median DIN concentrations were observed at station 1, while the lowest median DIN concentration was measured at station 8.

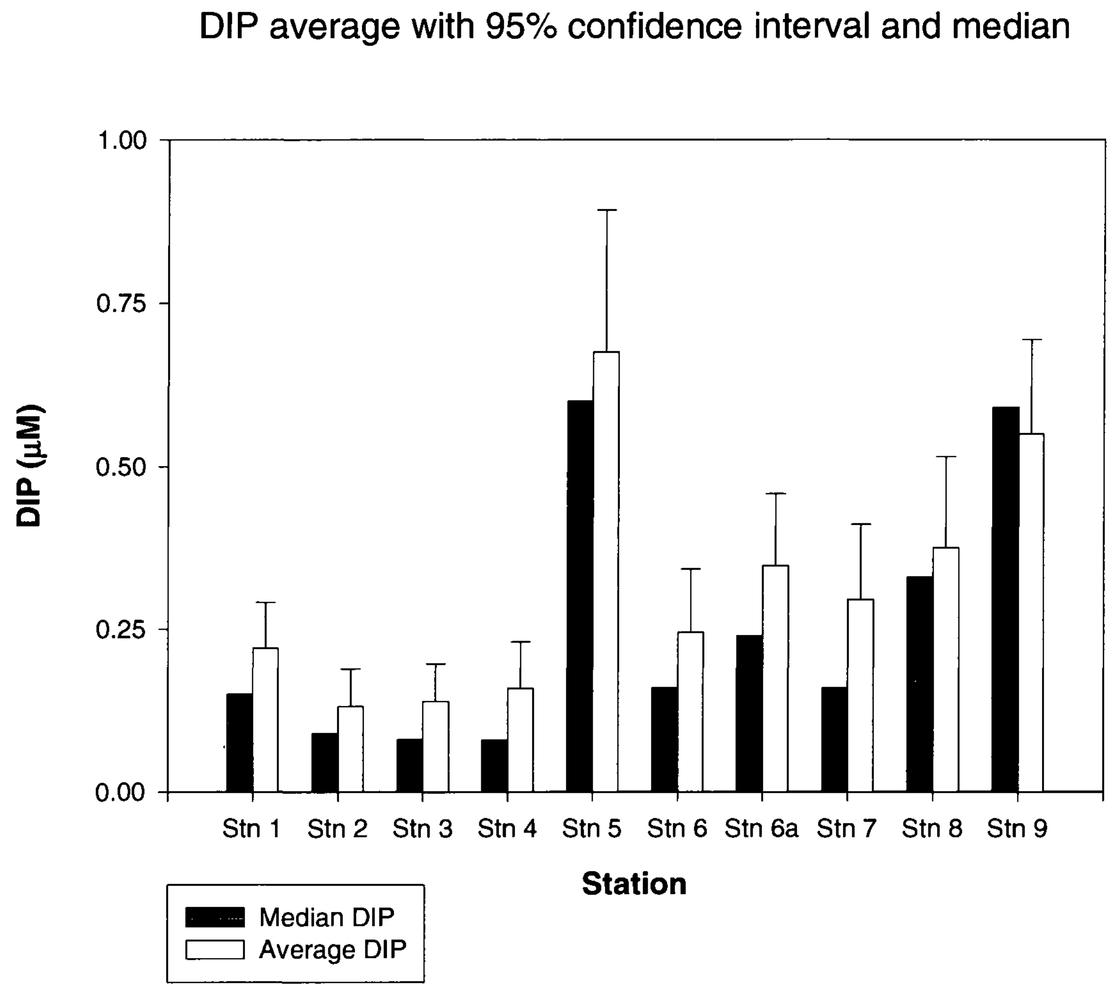

Figure 4.19 Vertical Bar Plot of DIP Concentrations Measured at Each Station. Plot includes data collected every sampling day (two days per month for fourteen months) at each of the ten stations in 2003-04. The filled black bars represent the median of the data. The white bars denote the mean of the data with error bars showing 95 % confidence interval of the mean for each station. Highest median and maximum DIP concentrations were observed at station 5 followed by station 9. Lowest median DIP concentration was measured at station 3.

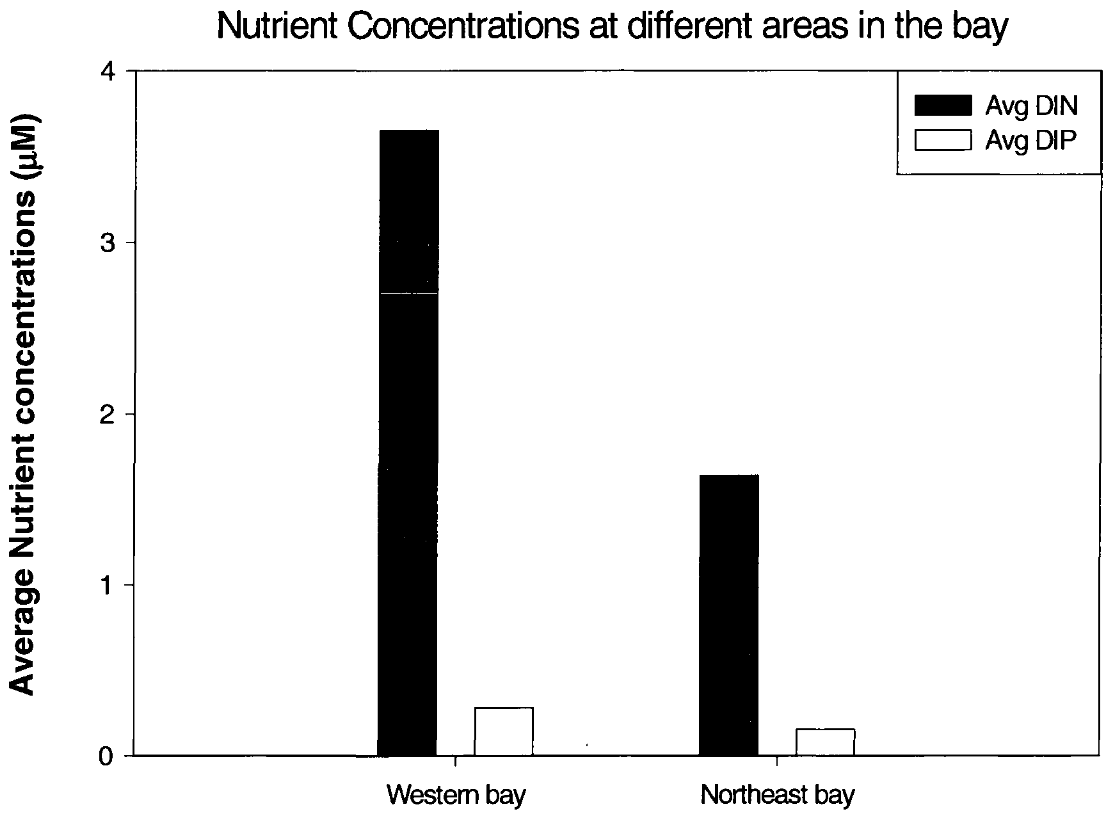

Figure 4.20 DIN and DIP Concentrations Averaged over the Entire Sampling Period of 2003-04 at Two Different Areas in the Bay Representing the Subwatersheds Surrounding and Draining into the Western and the Eastern Areas of the Bay of St. Louis. Average DIN and DIP concentrations at stations 1 and 6a (western bay) were twice of that observed at stations 2 and 4 (northeastern bay). Note: The N:P ratios at both areas were less than the Redfield ratio of 16. The N:P values at the western bay were 13.0 and those at the eastern bay were 10.6.

concentration of 0.08μM. Significant differences in nutrient concentrations were observed at stations representing the western (Stations 1 and 6a) and northeastern (Stations 2 and 4) sub-watersheds of the bay (Figure. 4.20). Average DIN and DIP concentrations at stations 1 and 6a were twice of that observed at stations 2 and 4. Other parameters such as temperature, turbidity, silicate, chlorophyll *a*, phaeopigment, and DO concentrations measured over fourteen months were not different significantly between the ten sampling stations.

Environmental Quality in 2003-04

The five parameters selected as indicators of environmental quality were correlated significantly to changes in the physical and weather conditions over time and space (Tables 4.3. a. and b.). Spatial interpolation analysis of each of the five indicator parameters (DIN, DIP, Chlorophyll *a*, Turbidity and DO) resulted in graphical representation of spatial and temporal variability in the environmental quality of the bay (Figures 4.21 to 4.43). Seasonal variability in nutrient concentrations was observed at all stations in the bay. DIN concentrations averaged seasonally exceeded the threshold values of 4μM during summer (July and August) 2003 and spring 2004 (March and May 2004). High DIN concentrations during these seasons were mainly observed at stations 1 and 6a. DIN concentrations were acceptable (2-4μM) or exceeded management objective (< 2μM) during the fall and winter of 2003. High DIP concentrations (> 0.4μM) in the bay were measured at stations 5 and 9 during summer 2003 and at stations 1, 5, and 9 in spring 2004. High concentrations at station 5 were also observed in winter 2003 (December and February 2003). Average DIP concentrations in fall 2003 exceeded the threshold value of 0.4μM at stations 5, 6, 7, 8, and 9. Average DIP concentrations were

Table 4.3.a Spearman Rank Correlation Analyses for Environmental Quality Indicator Parameters and Physical Forcing Factors in the Bay for 2003-04. Correlation coefficients highlighted in bold are significant at 0.01 level (2-tailed) while others are significant at 0.05 level (2-tailed). N is the number of data points. NS indicates no significant correlations between variables. Environmental quality indicators such as nutrients, chlorophyll *a*, turbidity and DO were correlated significantly to one or more physical parameters such as temperature, salinity, wind speed, precipitation, river discharge, and tidal forcing (gage height).

Rho	*DIN*	*DIP*	*Chl a*	*Turbidity*	*DO*	*Temp*	*Salinity*	*Wind speed*	*Rain Jourdan River*	*Rain Wolf River*	*Flow Wolf River*	*Gage height*
DIN		NS	**0.231** N = 230	**0.412** N = 175	NS	0.167 N = 230	**-0.520** N = 230	**0.256** N = 200	NS	NS	**0.395** N = 240	**- 0.214** N = 230
DIP	NS		NS	NS	NS	NS	**0.314** N = 250	**0.201** N = 181	NS	NS	NS	0.160 N = 220
Chl a				NS	NS	**0.426** N = 220	**-0.197** N = 220	0.148 N = 200	NS	NS	**0.286** N = 230	NS
Turbidity					NS	NS	**-0.505** N = 204	**0.340** N = 137	NS	0.139 N = 204	NS	NS
DO						**-0.505** N = 260	**0.237** N = 260	NS	**-0.256** N = 260	**-0.318** N = 260	-0.158 N = 260	**0.228** N = 230
Temp							**-0.366** N = 260	NS	**0.431** N = 260	**0.284** N = 260	0.148 N = 260	**-0.197** N = 230
Salinity								**-0.340** N = 190	**-0.188** N = 260	NS	**-0.413** N = 260	**0.318** N = 230
Wind speed									NS	NS	NS	NS
Rain Jourdan River										**0.830** N = 270	**0.426** N = 270	NS
Rain Wolf River											**0.235** N = 270	**0.170** N = 240
Flow Wolf River												NS

Table 4.3.b Spearman Rank Correlation Analyses for Physical Forcing Factors in the Bay (Such as Water Temperature, Salinity, Wind Speed, Precipitation, River Discharge, and Gage Height) and Weather Parameters for 2003-04. Correlation coefficients highlighted in bold are significant at 0.01 level (2-tailed) while others are significant at 0.05 level (2-tailed). N is the number of data points. NS indicates no significant correlations between variables. Physical parameters in the bay were correlated significantly to weather parameters such as atmospheric pressure, air temperature, dew point, and relative humidity.

Rho	*Atmospheric pressure*	*Air temp*	*Dew point temp*	*Relative humidity*	*Wind speed*	*Rainfall (WR)*	*River flow*	*Water temp*	*Salinity*
Atmospheric pressure		**-0.480** N=315	**-0.502** N=315	**-0.299** N=315	NS	**-0.342** N=315	NS	-0.736 N=190	NS
Air temperature			**0.929** N=315	**0.176** N=315	NS	0.114 N=315	0.144 N=313	**0.931** N=190	NS
Dew poin temperature				**0.494** N=315	NS	**0.268** N=315	**0.183** N=313	**0.804** N=190	NS
Relative humidity					-0.137 N=315	**0.473** N=315	NS	**0.463** N=190	NS
Wind speed						**0.167** N=315	0.133 N=313		
Rainfall (measured at WR)							**0.198** N=425	**0.284** N=260	**-0.188** (JR) N=260
River flow								0.148 N=260	**-0.413** N=260
Water temperature									**-0.366** N=260

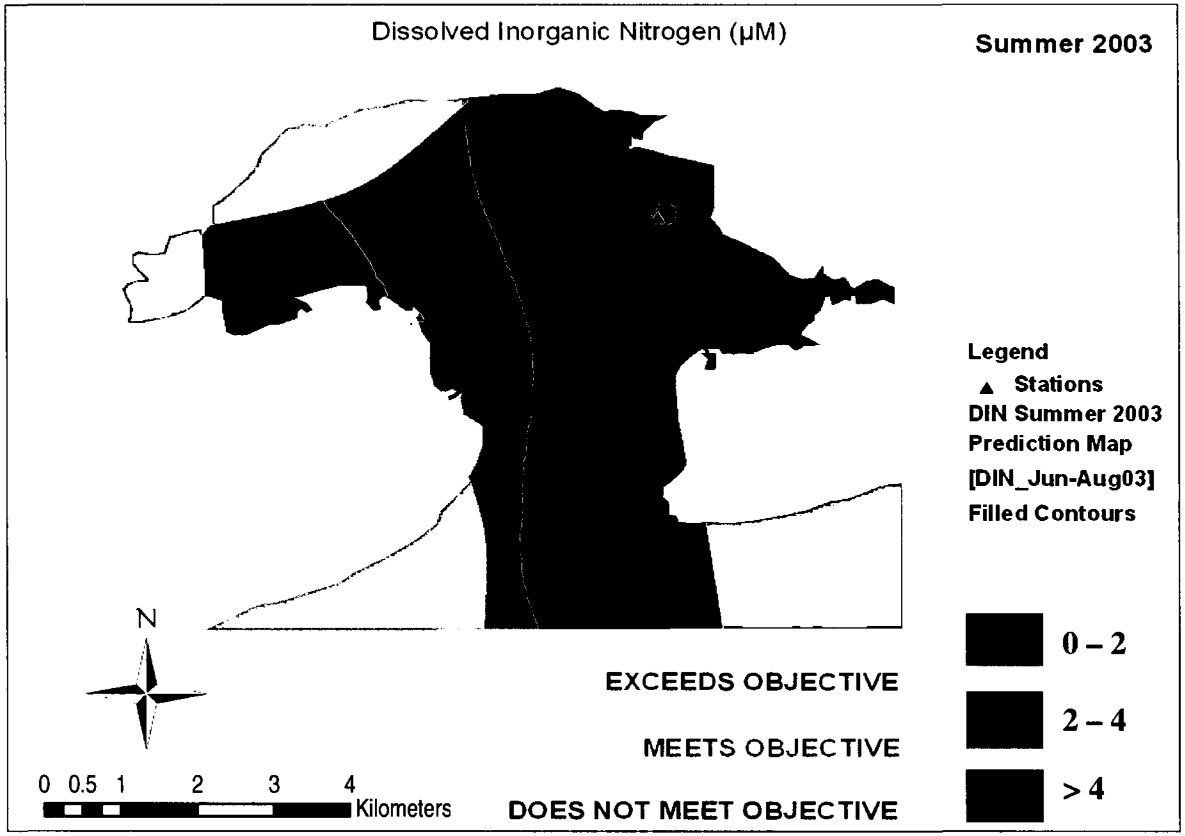

Figure 4.21 Spatial Distribution of Measured and Spatially Interpolated DIN Concentrations Averaged over Summer 2003 in the Bay of St. Louis. Areas colored in red did not meet the objective of reduced nutrients (> 4 µM). The orange areas met the objective (2- 4 µM). Areas colored in green exceeded the objective (< 2 µM). Excessive DIN concentrations were observed at the western end (station 1) near Jourdan River, while concentrations decreased towards the eastern part of the bay.

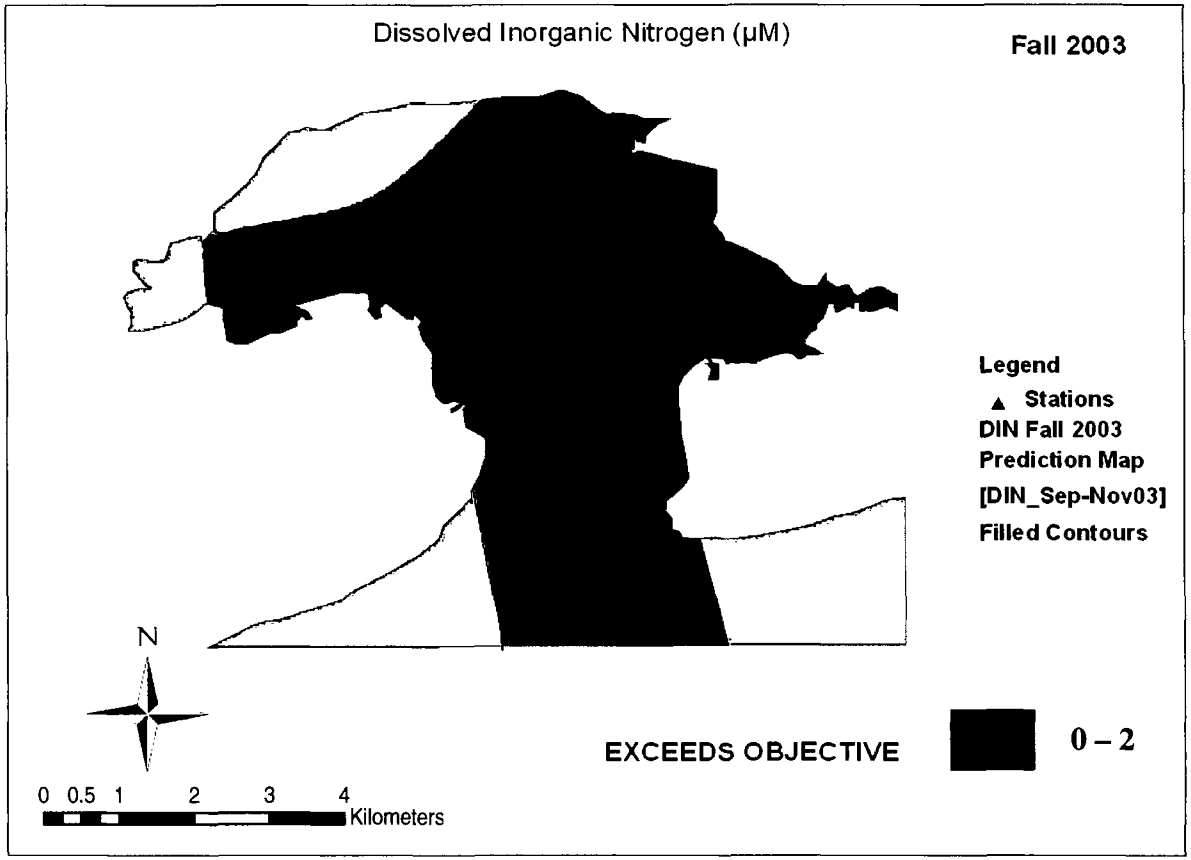

Figure 4.22 Spatial Distribution of Measure and Spatially Interpolated DIN Concentrations for Fall 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of reduced nutrients (< 2 µM). All areas in the bay exceeded the objective.

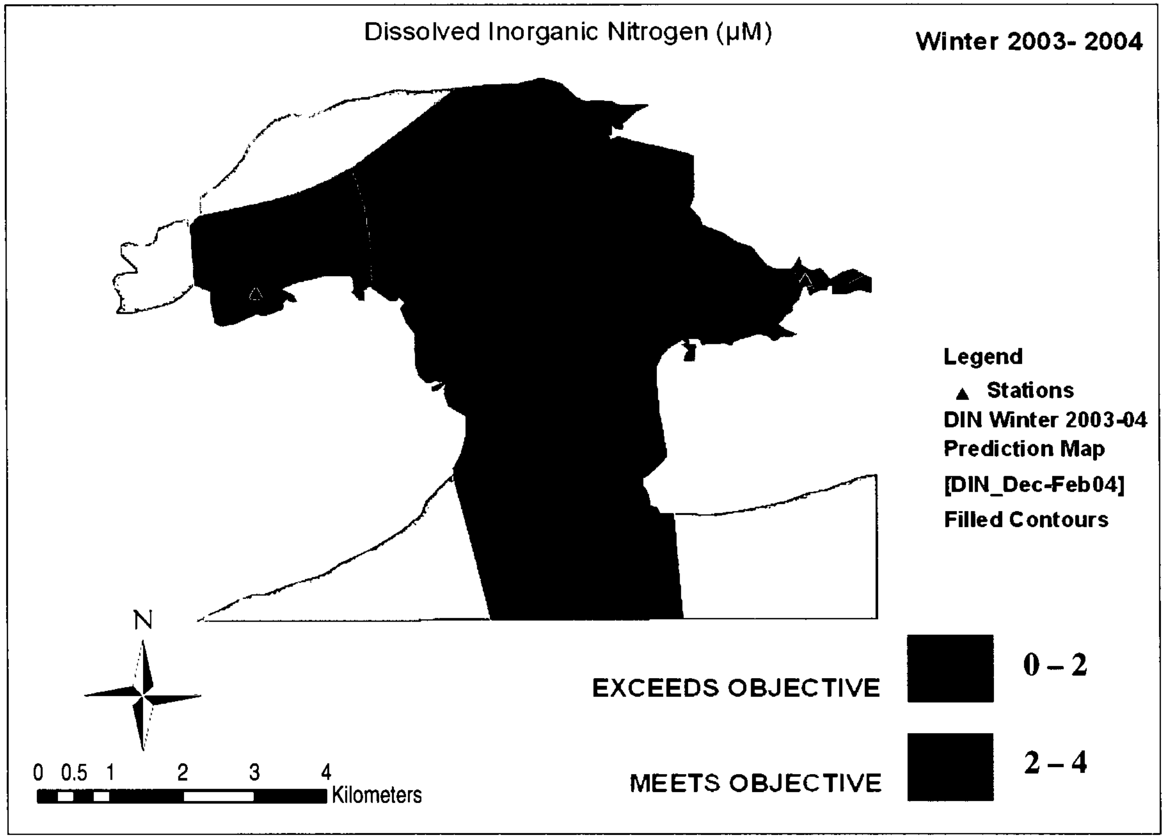

Figure 4.23 Spatial Distribution of Measured and Spatially Interpolated DIN Concentrations for Winter 2003-04 in the Bay of St. Louis. Areas colored in orange met the objective of reduced nutrients (2- 4 µM). Areas colored in green exceeded the objective (< 2 µM). Values within orange color code are graded from light to heavy shades for lower to higher values. DIN concentrations were low in most of the bay except in areas near Jourdan River and Bayou Portage, where the concentrations were in the acceptable range.

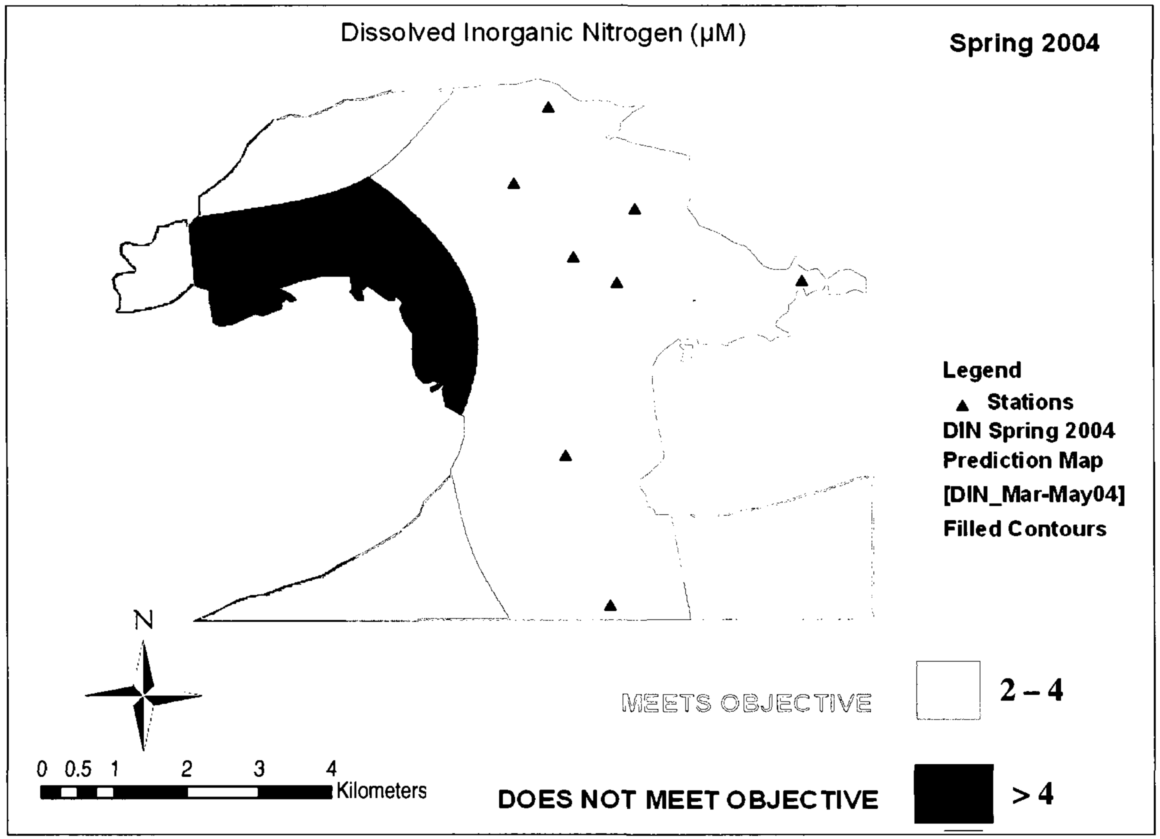

Figure 4.24 Spatial Distribution of Measured and Spatially Interpolated DIN Concentrations for Spring 2004 in the Bay of St. Louis. Areas colored in red did not meet the objective of reduced nutrients (> 4 µM). Areas colored in orange met the objective (2- 4 µM). Values within the orange color code are graded from light to heavy shades for lower to higher values. DIN concentrations throughout the bay were within the acceptable range except in the western part near Jourdan River, where excessive concentrations were observed.

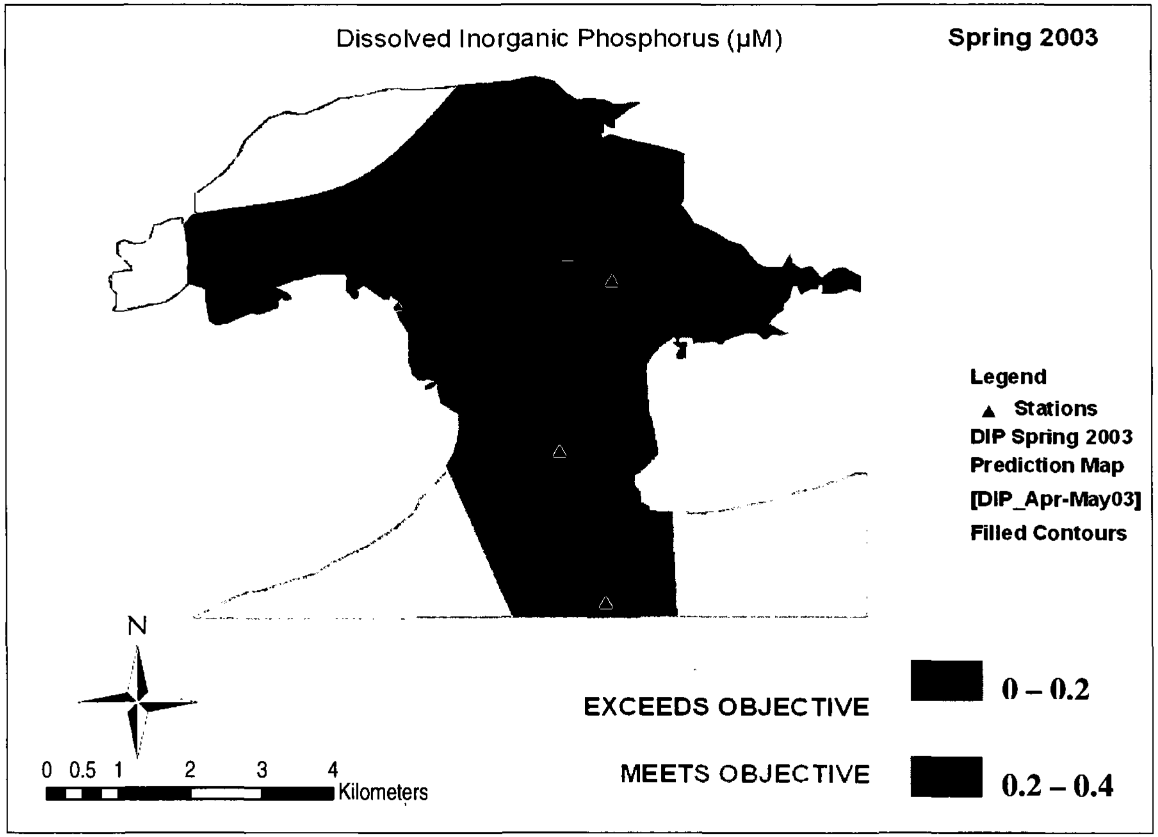

Figure 4.25 Spatial Distribution of Measured and Spatially Interpolated DIP Concentrations Averaged over Spring 2003 in the Bay of St. Louis. Areas colored in orange met the objective (0.2 – 0.4 μM) of reduced nutrients. Areas colored in green exceeded the objective (< 0.2μM). Values within the green color code are graded from light to heavy shades for lower to higher values. DIP concentrations were higher but at acceptable levels at areas closer to the mouth of the bay. Low DIP concentrations were observed in areas closer to the rivers and Bayou Portage.

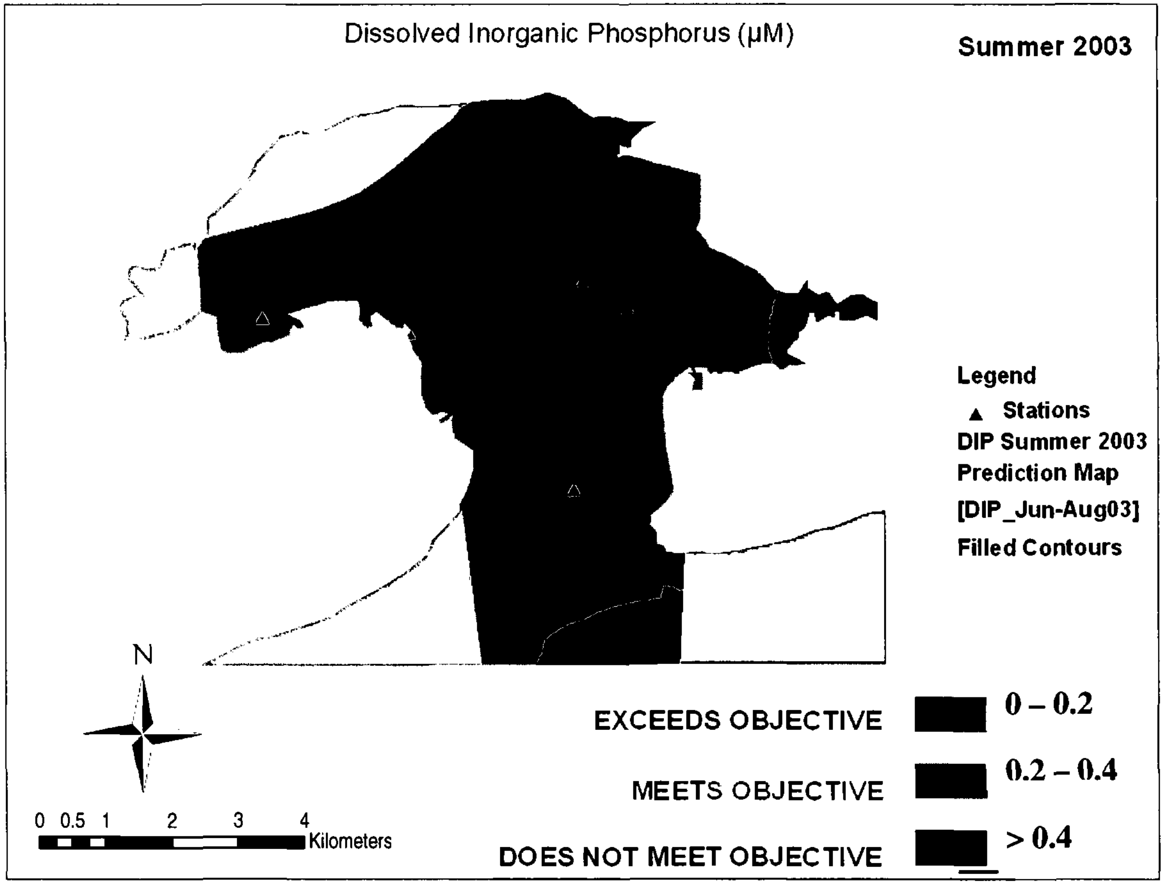

Figure 4.26 Spatial Distribution of Measured and Spatially Interpolated DIP Concentrations Averaged over Summer 2003 in the Bay of St. Louis. Areas colored in red did not meet the objective of reduced nutrients (> 0.4 µM), while the orange areas met the objective (0.2 – 0.4 µM). Areas colored in green exceeded the objective (< 0.2 µM). Values within a color code are graded from light to heavy shades for lower to higher values. Excessive DIP concentrations were observed in areas near the mouth of the bay in the Mississippi Sound and near Bayou Portage. Areas close to the northern shore and the Wolf River had the lowest DIP concentrations.

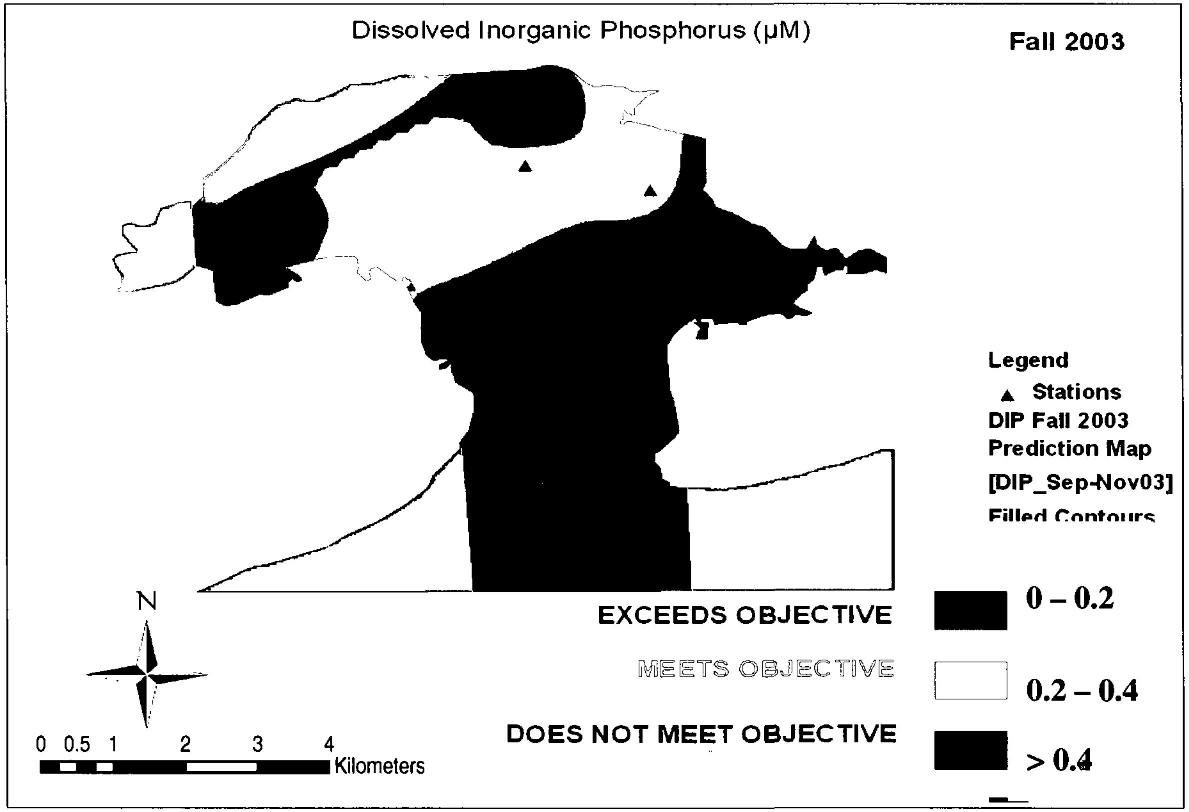

Figure 4.27 Spatial Distribution of Measured and Spatially Interpolated DIP Concentrations for Fall 2003 in the Bay of St. Louis. Areas colored in red did not meet our objective of reduced nutrients (> 0.4 µM), orange areas met the objective (0.2 – 0.4 µM). Areas colored in green exceeded the objective (< 0.2 µM). Areas within the red color code are graded from light to dark shades for lower (poor) to higher (impaired) values. Excessive DIP concentrations were observed in central bay (stations 6 and 7), near Bayou Portage (station 5), and near the mouth of the bay in the Mississippi Sound (Stations 8 and 9).

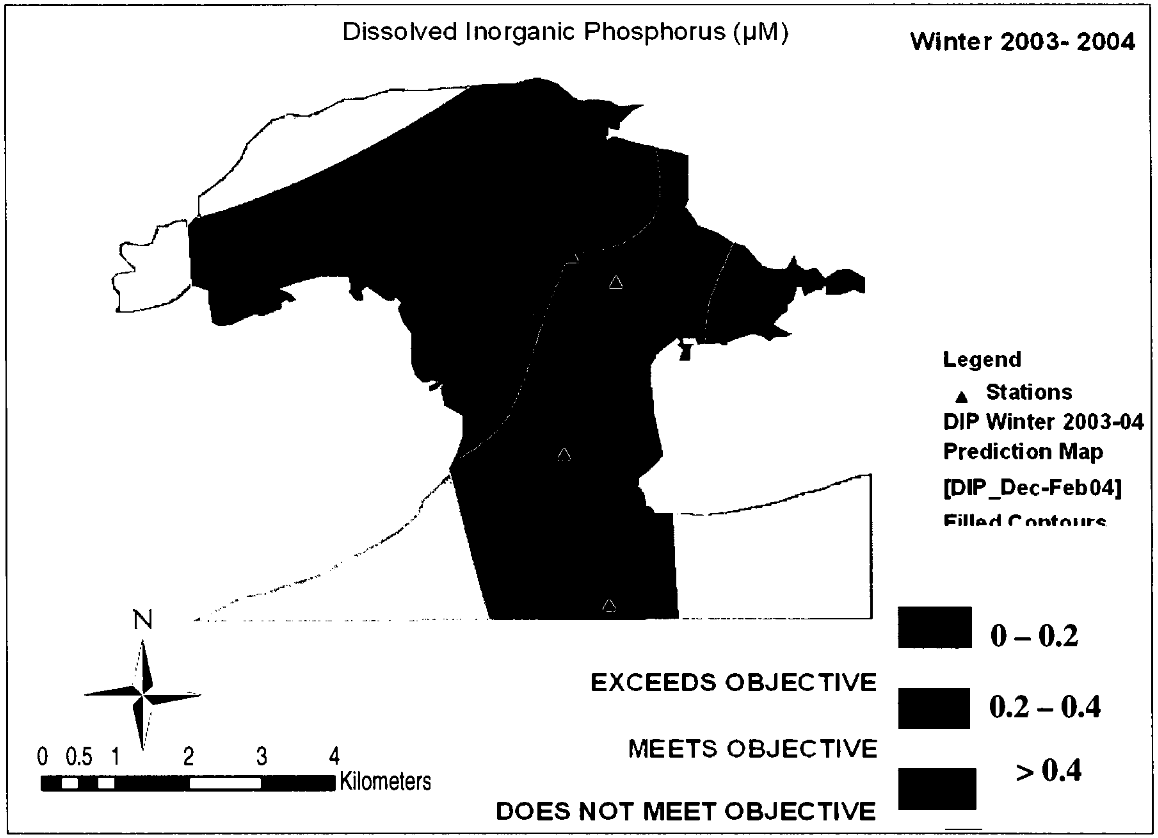

Figure 4.28 Spatial Distribution of Measured and Spatially Interpolated DIP Concentrations for Winter 2003-04 in the Bay of St. Louis. Areas colored in red did not meet our objective of reduced nutrients (> 0.4 µM), orange areas met the objective (0.2 – 0.4 µM). Areas colored in green exceeded the objective (< 0.2 µM). Values within the orange color code are graded from light to heavy shades for lower to higher values. Excessive DIP concentrations were observed at areas near Bayou Portage, while concentrations were at acceptable levels along the eastern part and near the mouth of the bay.

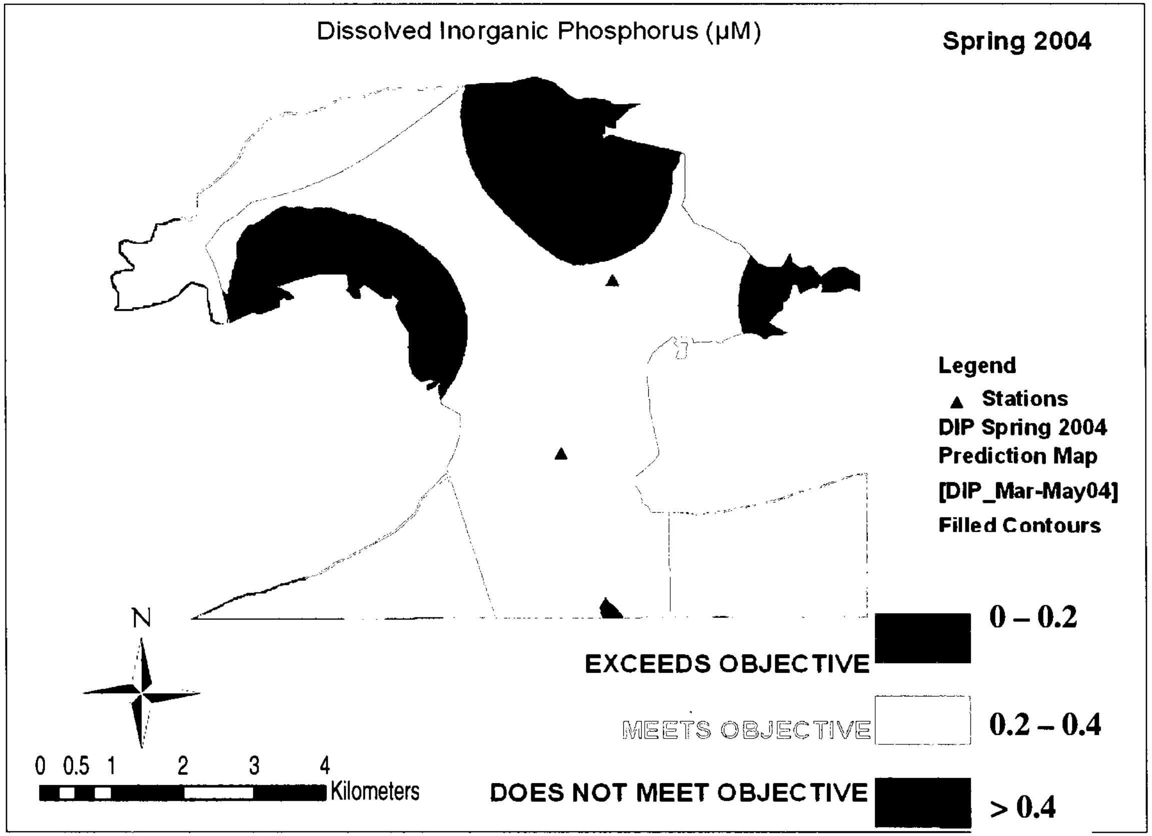

Figure 4.29 Spatial Distribution of Measured and Spatially Interpolated DIP Concentrations for Spring 2004 in the Bay of St. Louis. Areas colored in red did not meet our objective of reduced nutrients (> 0.4 µM), orange areas met the objective (0.2 – 0.4 µM). Areas colored in green exceeded the objective (< 0.2 µM). Excessive DIP concentrations were observed at the western (near Jourdan River) and eastern (Bayou Portage) ends of the bay. Low DIP concentrations were observed in the north-eastern region, near Wolf River including stations 2, 3, 4, 6, and 7. Concentrations were in the acceptable range in the central areas and closer to the mouth of the bay.

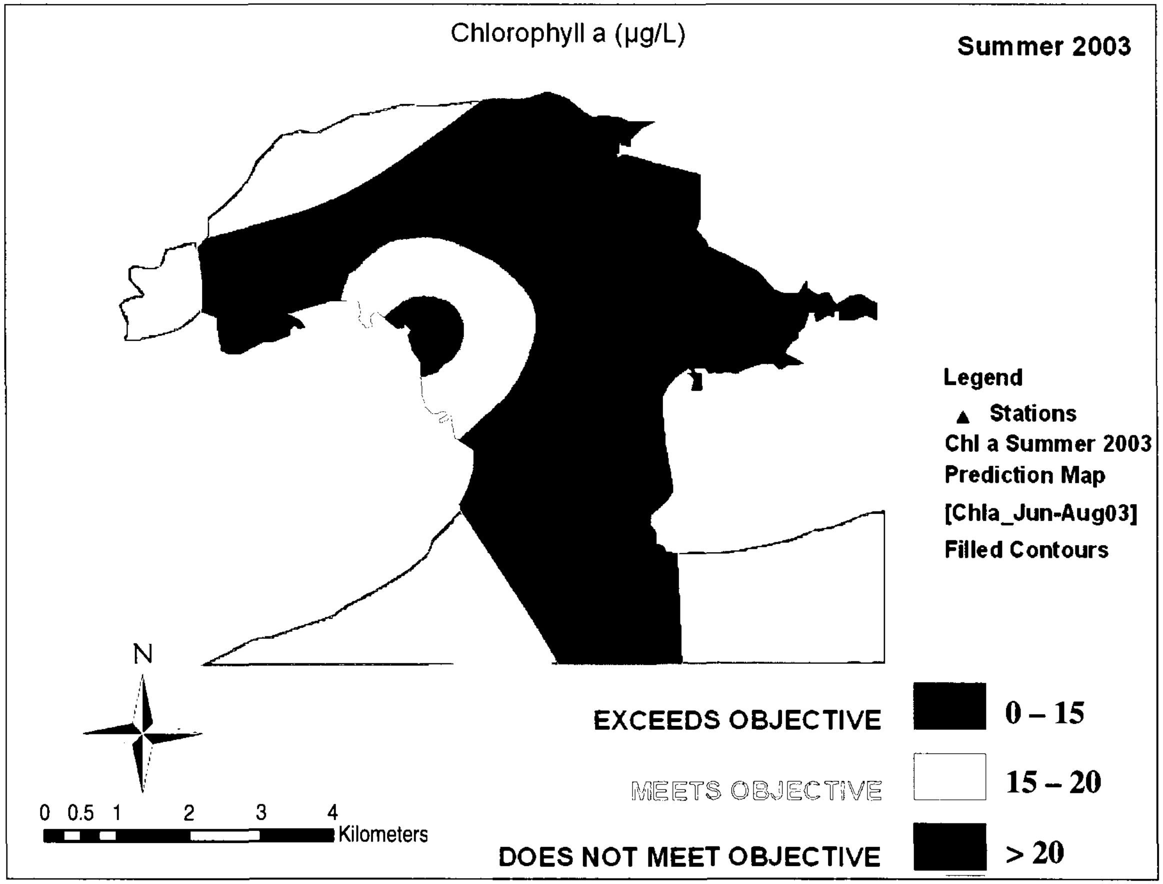

Figure 4.30 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Chlorophyll *a* during Summer 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of no algal blooms (< 15 μgL^{-1}). Areas colored in red did not meet the objective (> 20 μgL^{-1}), while orange areas met the objective (15-20 μgL^{-1}). All areas in the bay exceeded the objective except those close to station 6a, where excessive to acceptable concentrations of chlorophyll *a* were observed.

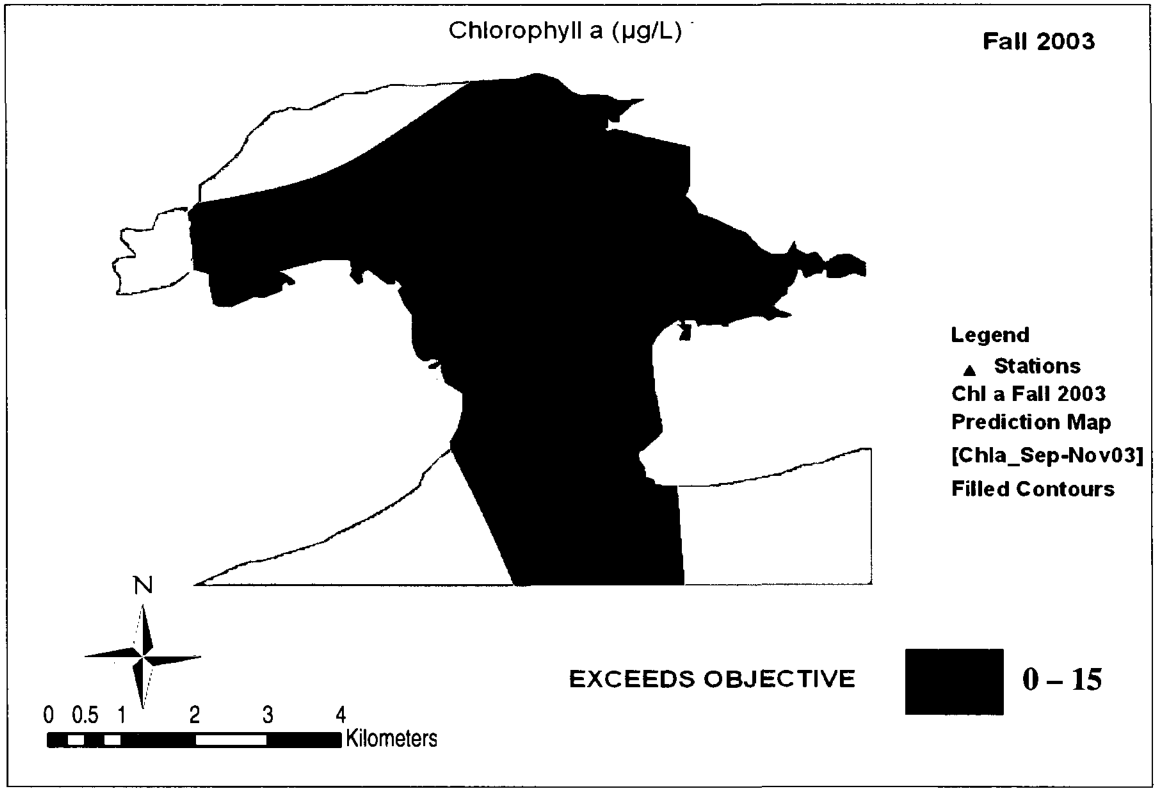

Figure 4.31 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Chlorophyll *a* for Fall 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of no algal blooms (< 15 μgL^{-1}). The entire bay exceeded the objective.

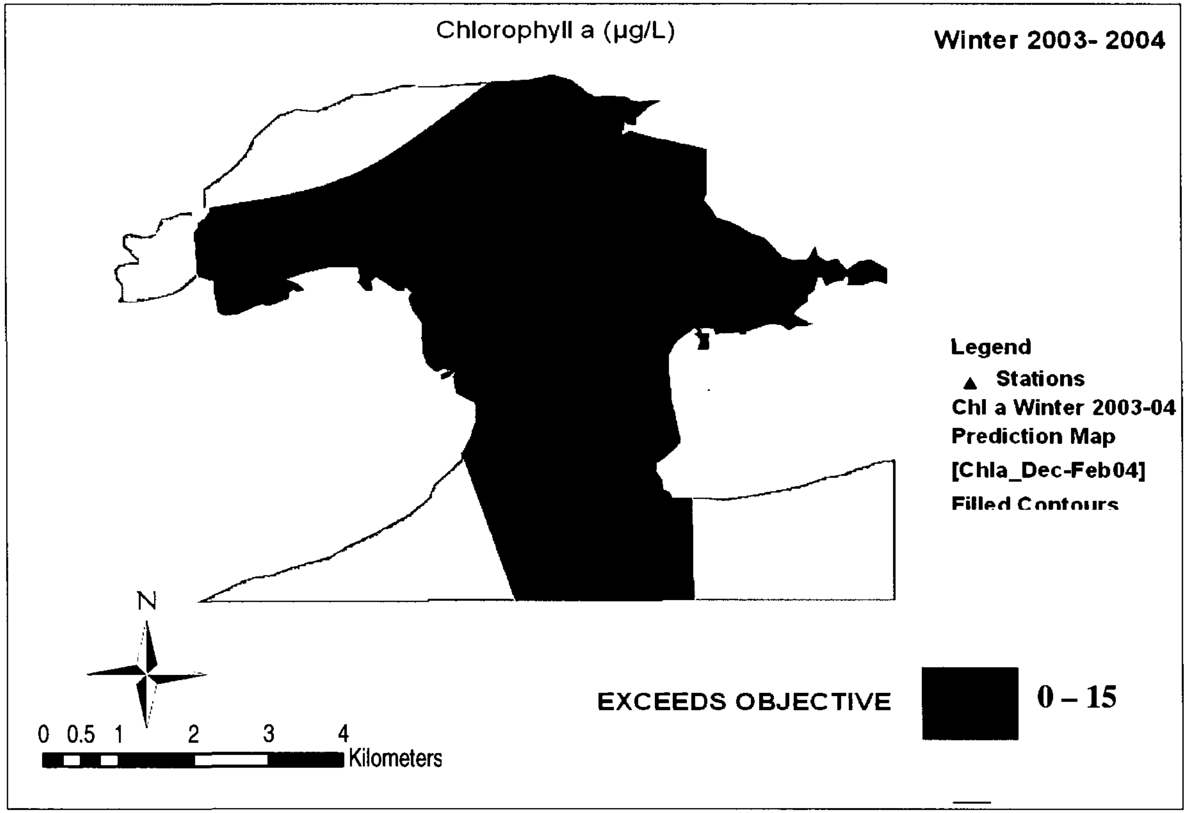

Figure 4.32 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Chlorophyll *a* for Winter 2003-04 in the Bay of St. Louis. Areas colored in green exceeded the objective of no algal blooms (< 15 μgL^{-1}). The entire bay exceeded the objective.

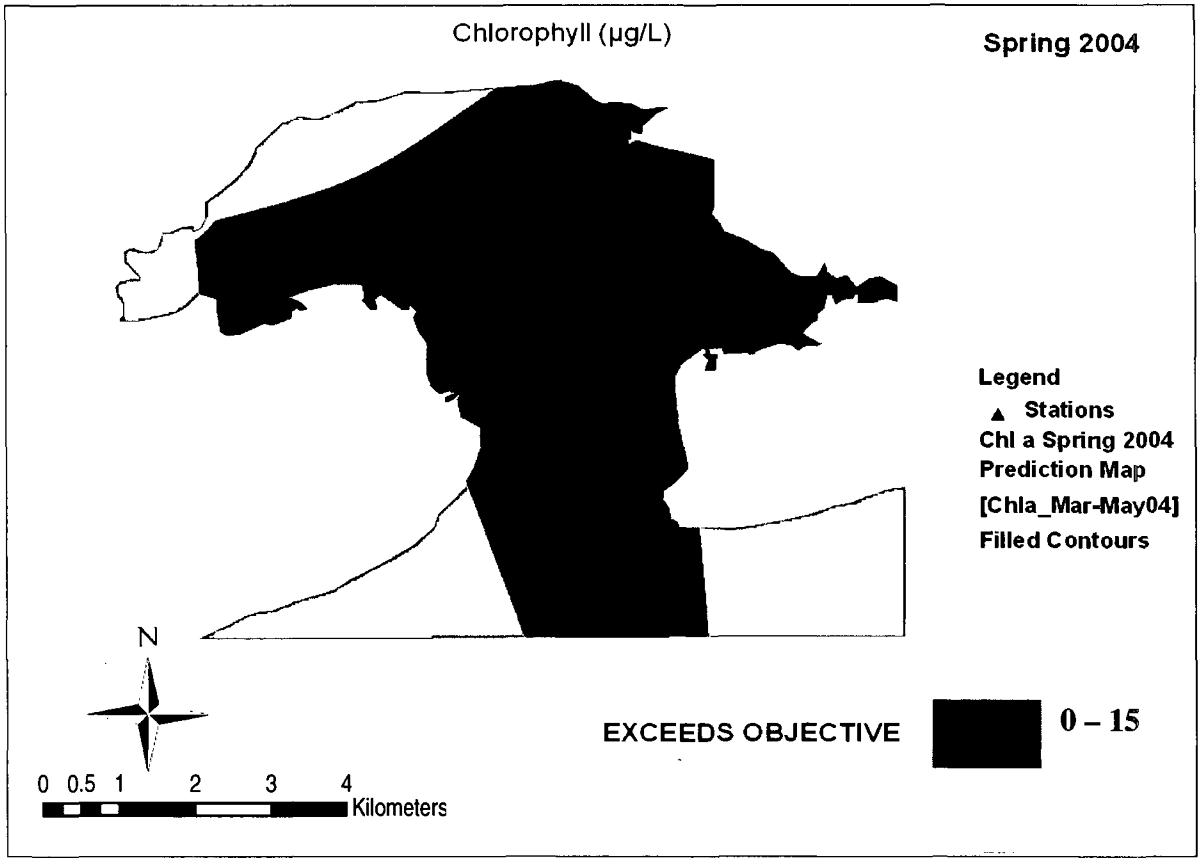

Figure 4.33 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Chlorophyll *a* for Spring 2004 in the Bay of St. Louis. Areas colored in green exceeded the objective of no algal blooms (< 15 μgL^{-1}). The entire bay exceeded the objective.

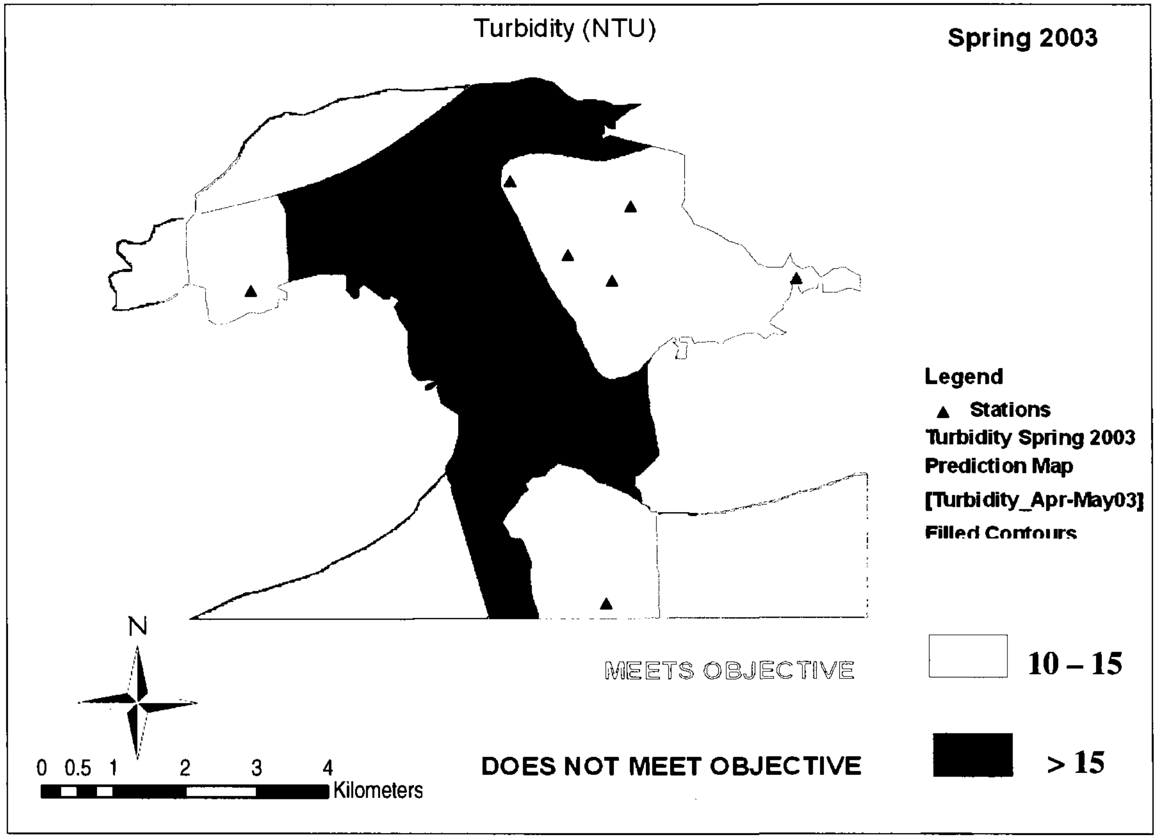

Figure 4.34 Spatial Distribution of Measured and Spatially Interpolated Turbidity Values for Spring 2003 in the Bay of St. Louis. Areas colored in red did not meet the objective of clear waters (> 15 NTU), while the orange areas met the objective (10 – 15 NTU). Areas within the red color code graded from light to dark shades represent poor to highly impaired areas. Turbidity was highest at station 6a. High turbidity values were observed between stations 2 and 8 along the western region. All other areas in the bay were within the acceptable range of values for turbidity.

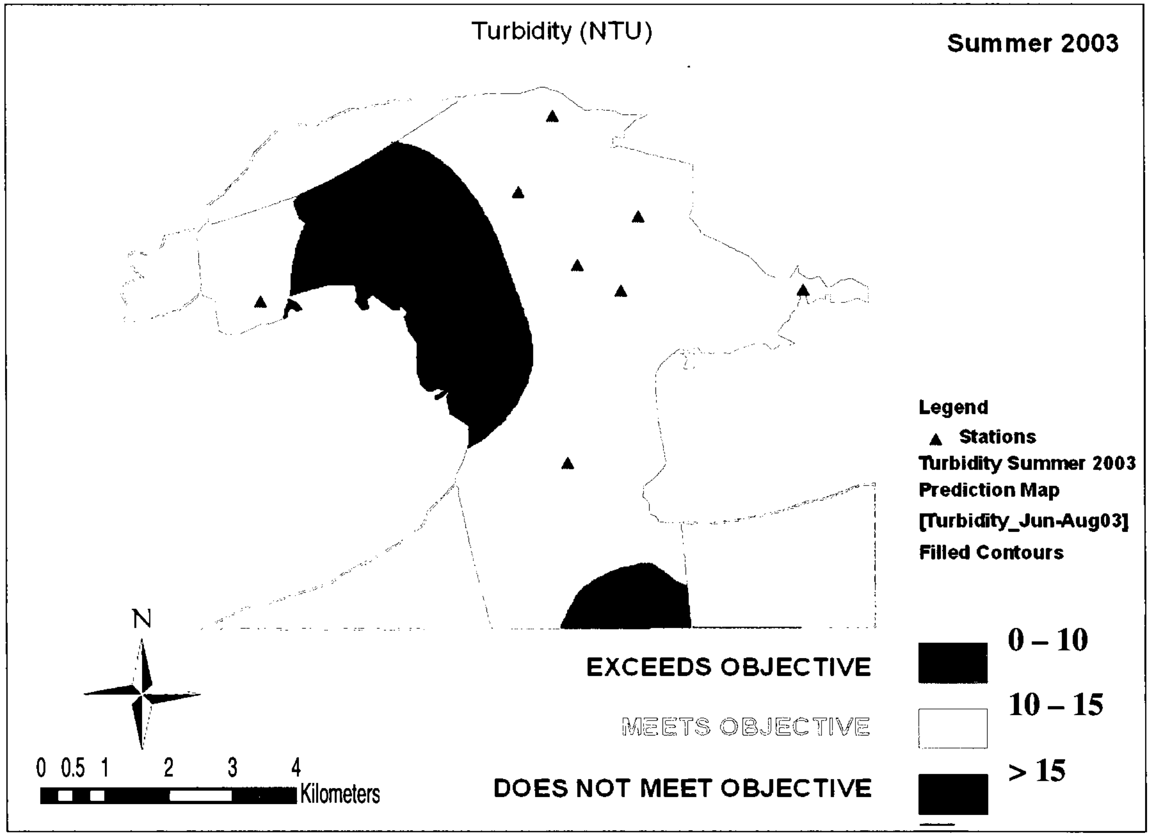

Figure 4.35 Spatial Distribution of Measured and Spatially Interpolated Turbidity Values for Summer 2003 in the Bay of St. Louis. Areas colored in red did not meet the objective of clear waters (> 15 NTU), orange areas met the objective (10 – 15 NTU). Areas colored in green exceeded the objective of clear waters (< 10 NTU). High turbidity values were observed in the western region closer to station 6a. Turbidity levels were within the acceptable range for other areas in the bay and were lowest near station 9 in the Mississippi Sound.

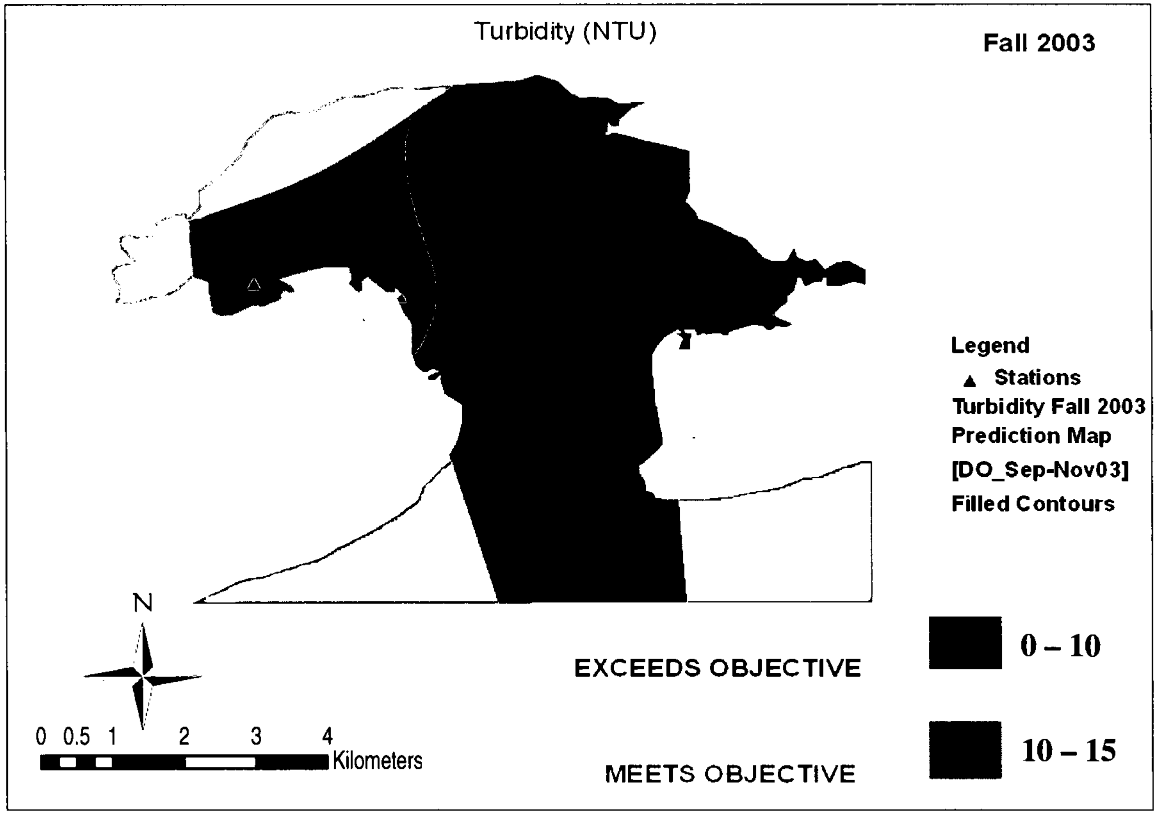

Figure 4.36 Spatial Distribution of Measured and Spatially Interpolated Turbidity Values for Fall 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of clear waters (< 10 NTU); orange areas met the objective (10 – 15 NTU). Acceptable levels of turbidity were observed in the western areas close to Jourdan River (stations 1 and 6a). Turbidity values were low in the remaining part of the bay.

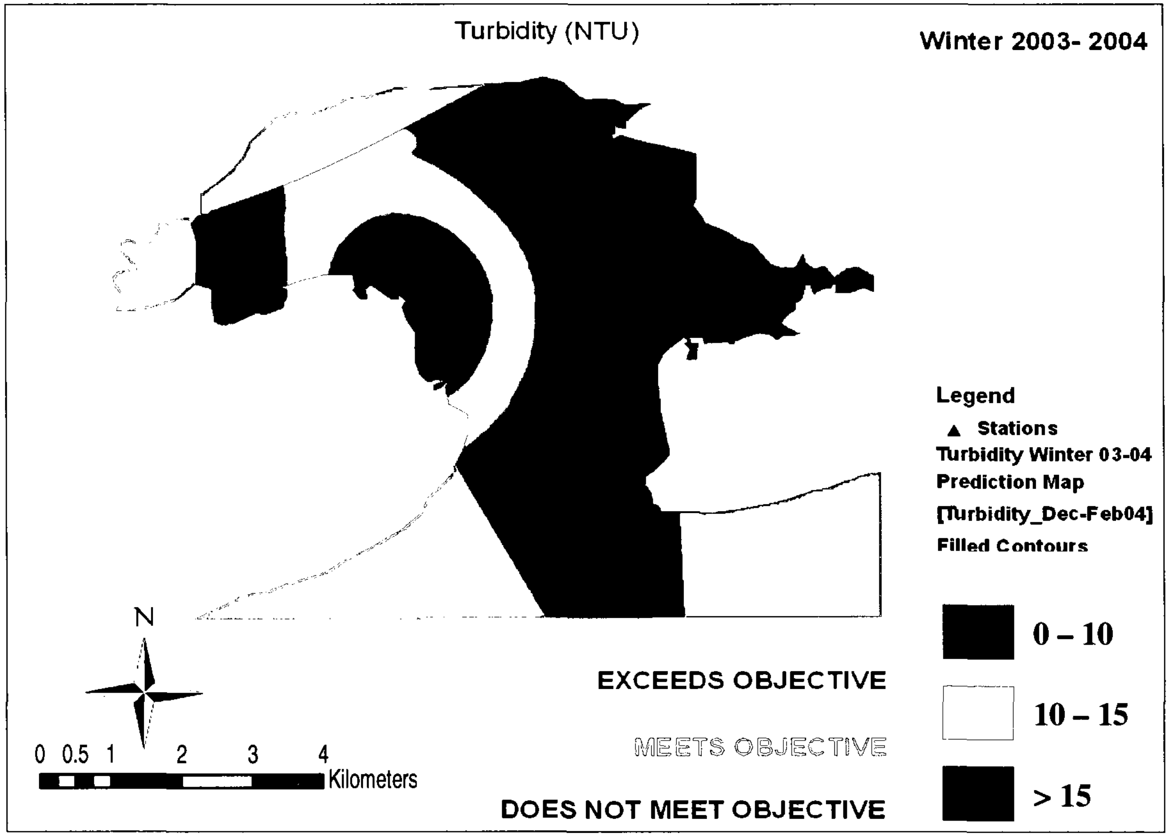

Figure 4.37 Spatial Distribution of Measured and Spatially Interpolated Turbidity Values for Winter 2003-04 in the Bay of St. Louis. Areas colored in red did not meet the objective of clear waters (> 15 NTU), orange areas met the objective (10 – 15 NTU). Areas colored in green exceeded the objective of clear waters (< 10 NTU). Turbidity values throughout the bay were low except along the western shore near station 6a, where excessive turbidity was observed.

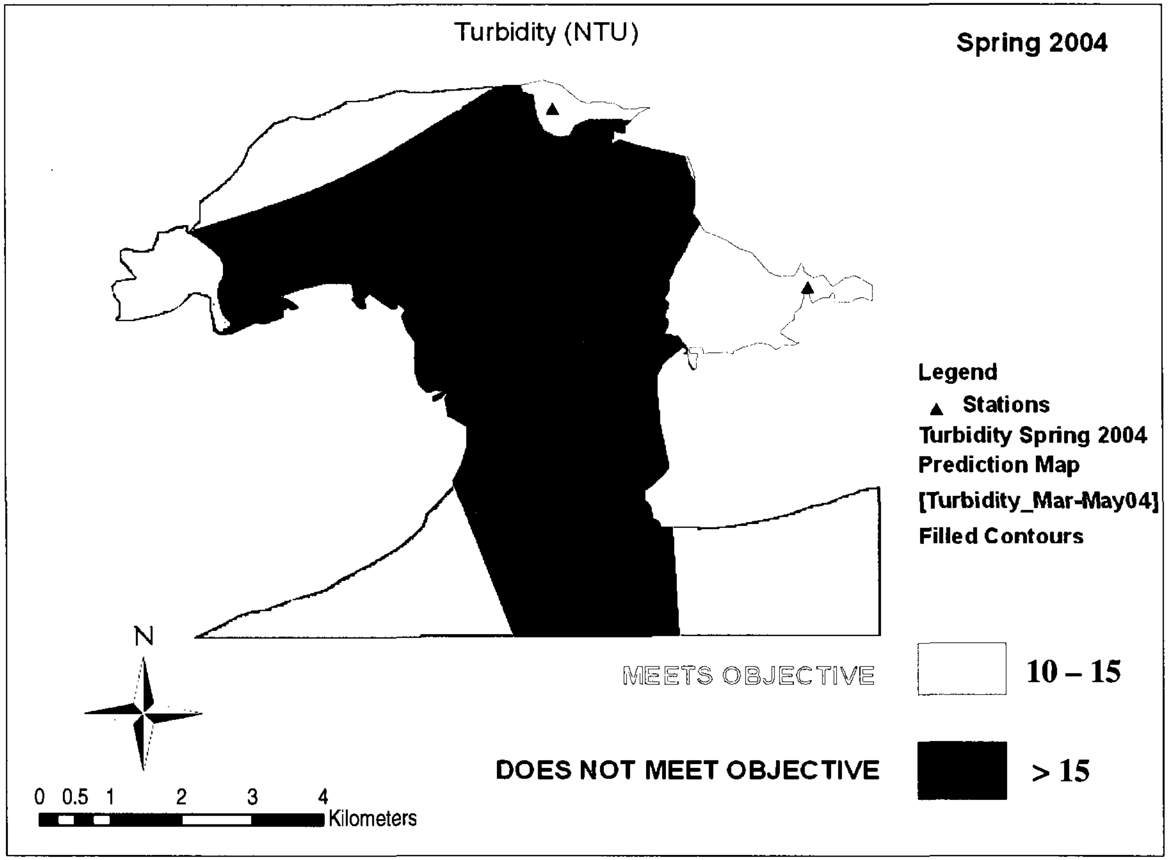

Figure 4.38 Spatial Distribution of Meaured and Spatially Interpolated Turbidity Values for Spring 2004 in the Bay of St. Louis. Areas marked red did not meet the objective of clear waters (> 15 NTU). Orange areas met the objective (10 – 15 NTU). Areas within the red color code graded from light to dark shades represent poor to highly impaired areas. Excessive turbidity was observed throughout the bay except at station 2 and near Bayou Portage, where turbidity levels were acceptable.

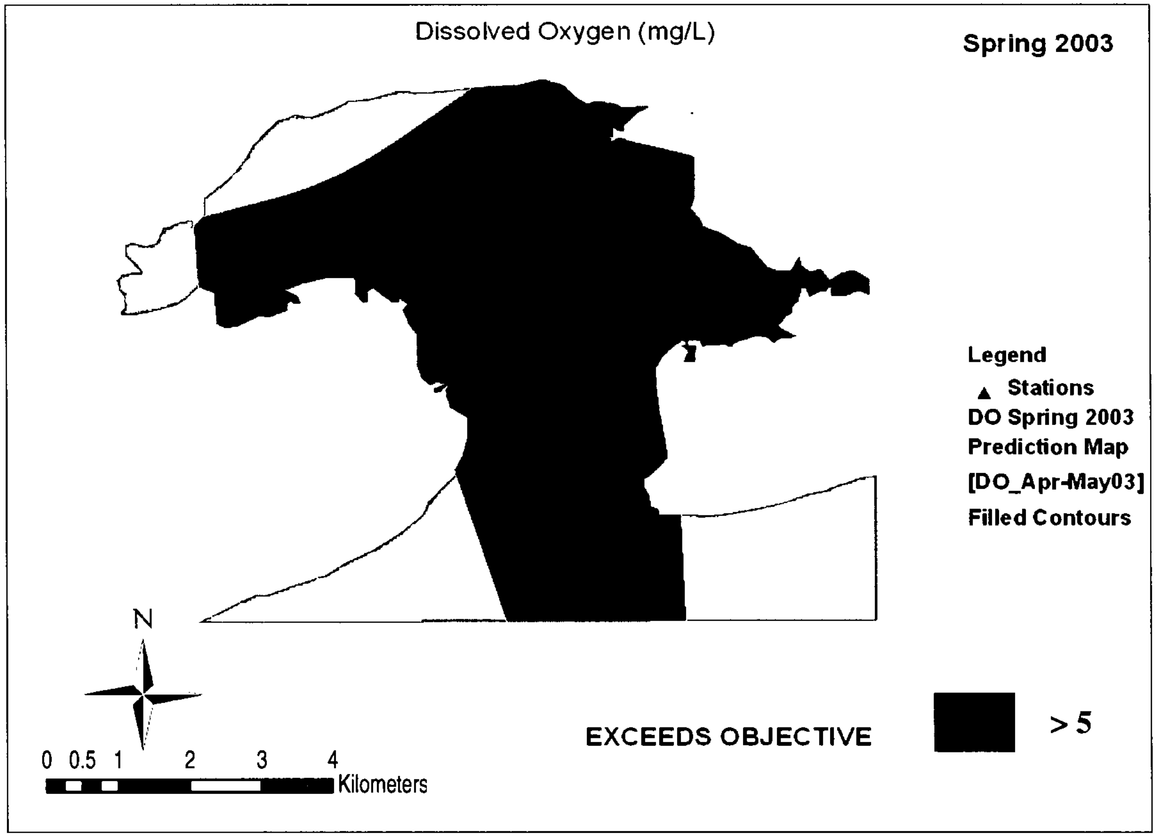

Figure 4.39 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Dissolved Oxygen (DO) for Spring 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of supporting aquatic life (> 5 mg L^{-1}). The entire bay exceeded the objective.

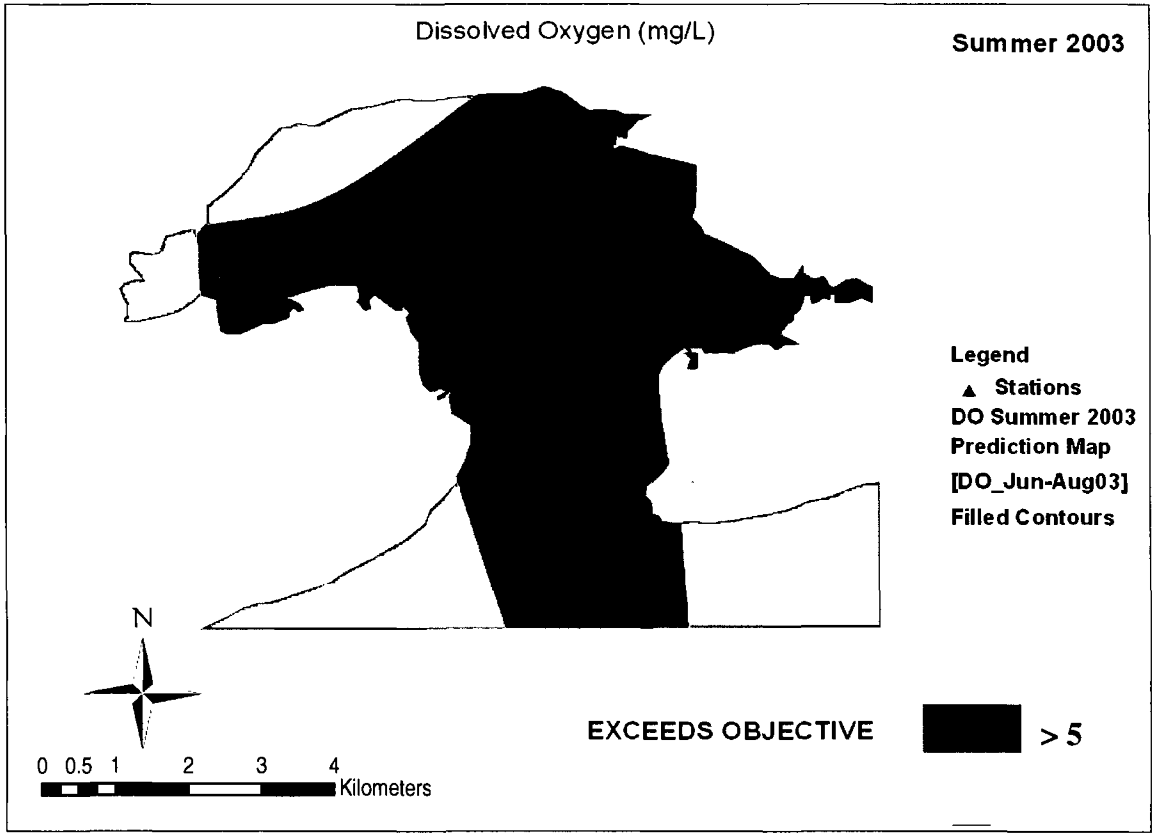

Figure 4.40 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Dissolved Oxygen (DO) for Summer 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of supporting aquatic life (> 5 mg L^{-1}). The entire bay exceeded the objective.

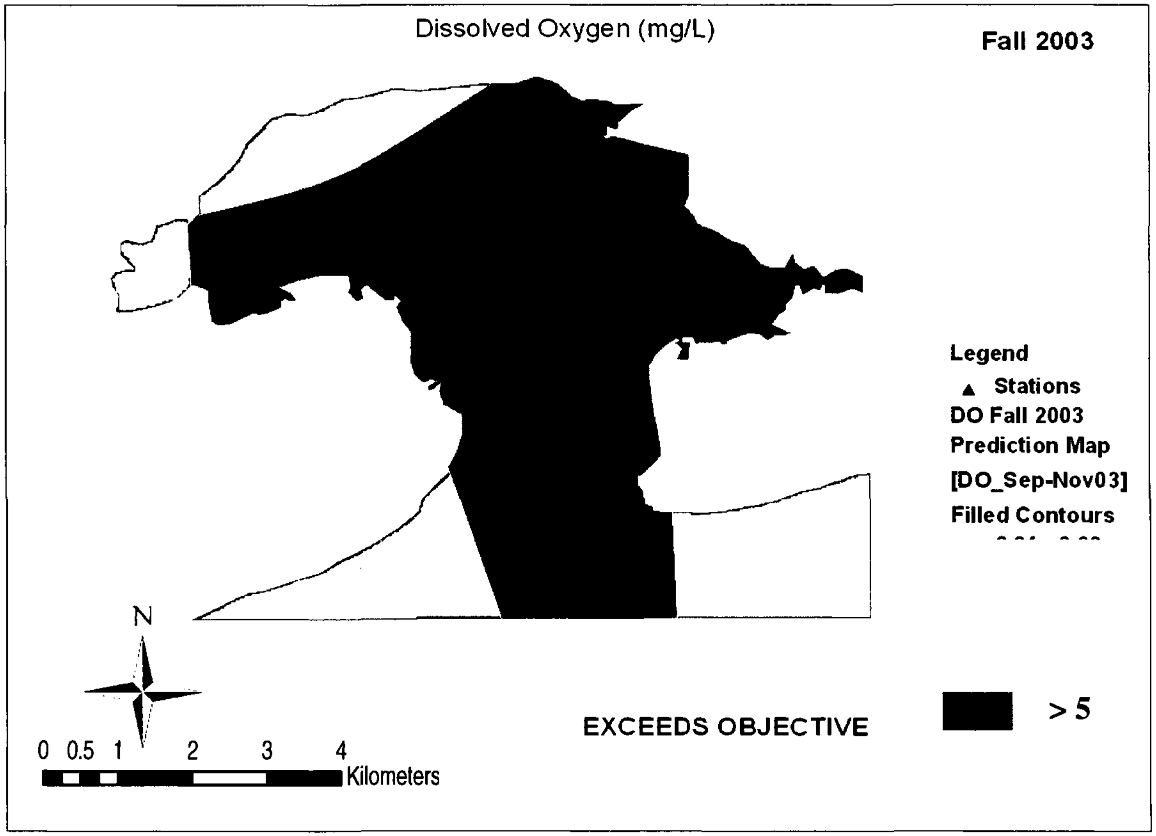

Figure 4.41 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Dissolved Oxygen (DO) for Fall 2003 in the Bay of St. Louis. Areas colored in green exceeded the objective of supporting aquatic life (> 5 mg L^{-1}). The entire bay exceeded the objective.

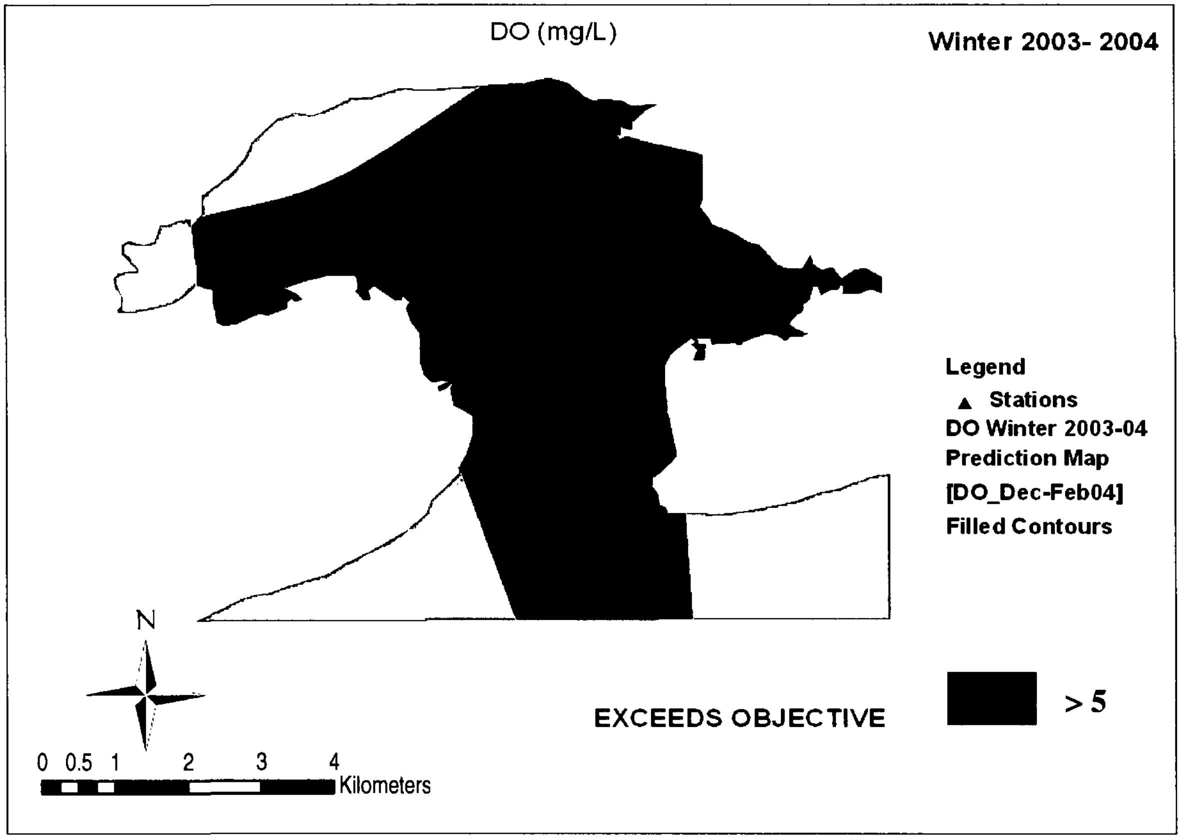

Figure 4.42 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Dissolved Oxygen (DO) for Winter 2003-04 in the Bay of St. Louis. Areas colored in green exceeded the objective of supporting aquatic life (> 5 mg L^{-1}). The entire bay exceeded the objective.

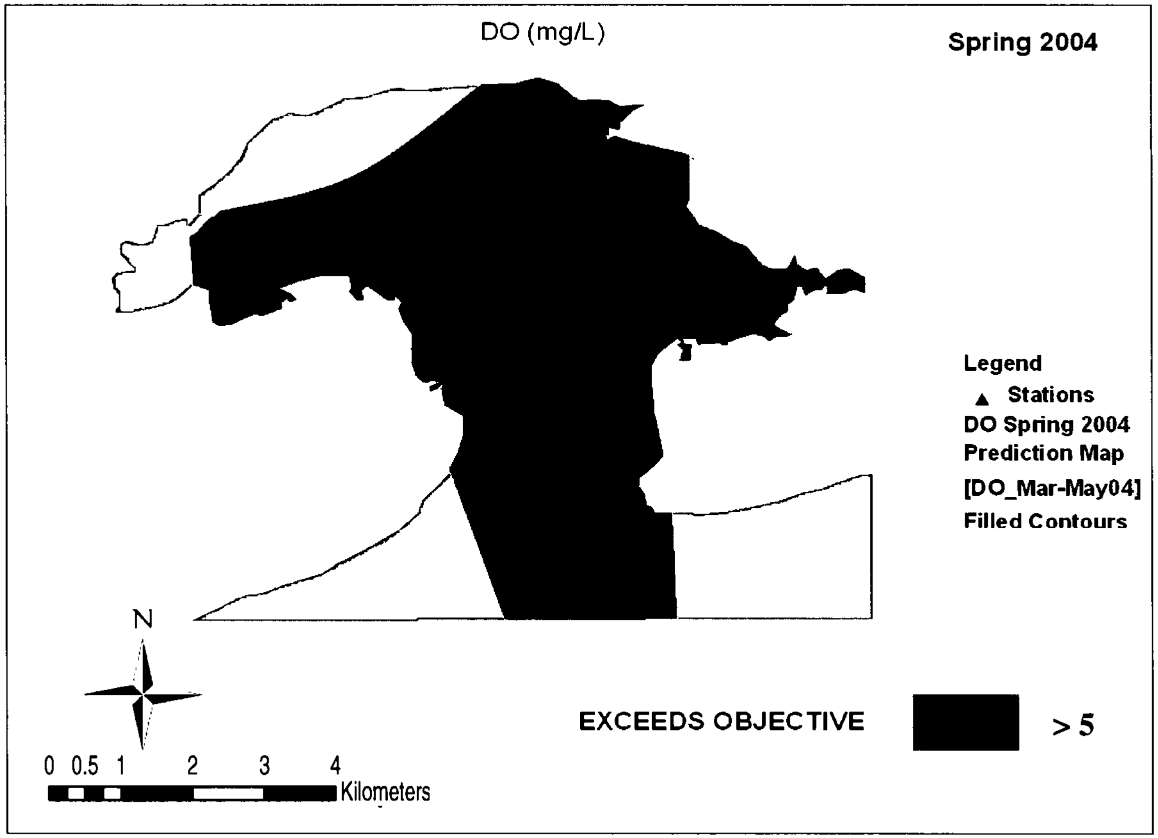

Figure 4.43 Spatial Distribution of Measured and Spatially Interpolated Concentrations of Dissolved Oxygen (DO) for Spring 2004 in the Bay of St. Louis. Areas colored in green exceeded the objective of supporting aquatic life (> 5 mg L^{-1}). The entire bay exceeded the objective.

less than 0.4μM at all stations in spring 2003.

Chlorophyll *a* concentrations averaged seasonally were higher than the threshold value (20 μgL^{-1}) in summer 2003 at station 6a. Average chlorophyll *a* concentrations in the bay were less than 20 μgL^{-1} during fall and winter 2003 as well as in spring 2004. Turbidity values averaged over individual seasons exceeded the threshold value (15 NTU) at station 6a in spring, summer, and winter 2003.

Average turbidity values in spring 2003 were also high at stations 2 and 8. Average turbidity values in spring 2004 were higher than 15 NTU at all stations except stations 2 and 5. DO concentrations in the bay exceeded the threshold value (2 mgL^{-1} or 62.50 $\mu mol\ kg^{-1}$) at all stations during the entire year.

The environmental quality for each season was determined based on the average EQI (Environmental Quality Index) values. The average EQI scores were calculated by adding up the total EQI values (based on Table 3.4) for all parameters for every sampling station and dividing by the number of stations. Based on the final average EQI scores, it was found that the environmental quality of the bay was partly compromised (Acceptable) during the spring and the summer (grades "B") seasons (Table 4.4). However, the environmental quality of the bay was never found to be poor or impaired during any season. (Note: The low total EQI score during spring 2003 was the result of the unavailability of DIN and chlorophyll data during that season. Also, the field sampling for this study was started in the last week of April 2003, therefore, only three samplings were carried out that season). Environmental quality of the bay was "Good" (grade "A") during both the fall and winter seasons.

Table 4.4 Final Report Card and Environmental Quality Index (EQI) Values for Each Season During 2003-04 for Every Station in the Bay of St. Louis. Average EQI values for the entire bay ranged from 4.2 to 8.9 and were based on environmental quality grades of "Good" to "Impaired" as shown in the legend above. Environmental quality was "Good" during fall and winter while it ranked as "Acceptable" during the spring (2004) and summer seasons (See Appendix A). Note: Spring 2003 score was not included since DIN and chlorophyll data were not available during that season. Also, the field sampling for this study was started in the last week of April 2003, therefore, only three samplings were carried out that season.

Environmental Quality Grade	Index	Final EQI Ranks	Letter Grade
GOOD		7.5 – 10.0	A
ACCEPTABLE		5.0 – 7.5	B
POOR		2.5 – 5.0	C
IMPAIRED		0.0 – 2.5	F

Stations	Latitude	Longitude	Summer 2003	Fall 2003	Winter 2003	Spring 2004
1	30.3407	-89.3554	6	7	9	5
2	30.37218	-89.3085	9	8	10	8
3	30.35926	-89.314	9	9	10	7
4	30.35503	-89.2947	8	9	9	7
5	30.34283	-89.2681	7	7	7	6
6	30.34686	-89.3046	8	9	9	6
6a	30.33847	-89.3322	4	8	8	4
7	30.34257	-89.2976	8	8	9	6
8	30.31344	-89.3061	8	8	9	6
9	30.28795	-89.2987	6	8	9	6
Total EQI			73	81	89	61
Final Average EQI			7.3	**8.1**	**8.9**	6.1
Final Grade			B	**A**	**A**	B

Annual environmental quality

Bay-wide DIN concentrations over the entire sampling period were low (< 4μM) at all stations except at stations 1 and 6a, where the concentrations were between 2-4μM. DIP concentrations in the bay averaged over the fourteen months were high (> 0.4μM) only at stations 5 and 9. DIP concentrations were in the acceptable range (0.2-0.4μM) at stations 1, 6, 6a, 7, and 8. DIP concentrations at stations 2, 3, and 4 were less than 0.4 μM. Chlorophyll *a* concentrations at all stations were below $20 \mu g L^{-1}$. Turbidity values were high (> 15 NTU) at station 6a and were less than 10 NTU at station 5. Turbidity values were between 10 -15 NTU at all other stations. Dissolved oxygen concentrations were above 5 $mg L^{-1}$ throughout the bay.

Environmental Quality Index (EQI) values assigned to each sampling station for every parameter ranged from 1.00 to 2.00 for stations 1 through 4 and for stations 6 and 7; and from 0.00 to 2.00 for stations 5, 6a, and 9 (Table 4.5). An EQI value of zero was assigned to Stations 5 and 9 for high (> 0.4 μM) DIP concentrations. Station 6a was assigned an EQI value of zero for high (> 15 NTU) turbidity values. Total EQI scores for stations 2, 3, 4, 6, 7, and 8 were between 8.00 and 9.00, which ranked as "Good" on the environmental quality index. Stations 1, 5, 6a, and 9, however, had total EQI scores between 6.00 and 7.00 and therefore were ranked as "Acceptable" on the final EQ Index. Overall, EQI value for the entire bay averaged to 7.60 and as a result environmental quality of the bay was considered as marginally "Good", thus obtaining the grade "A" (See Appendix A).

Table 4.5 Report Card and Environmental Quality Index (EQI) Values for Each Indicator Parameter Assigned for the Entire Period (2003-04) for Every Station in the Bay of St. Louis. EQI values range from 0.00 to 2.00 and are based on environmental quality grades of "Excellent", "Good", and "Poor" as shown in the legend above. Stations 2, 3, 4, 6, 7, and 8 ranked "Good", while stations 1, 5, 6a, and 9 ranked as "Acceptable" on the final Environmental Quality Index. Overall, EQI value for the entire bay averaged to 7.60 and ranked as "Good".

Environmental quality grade	*Color*	*EQI value*
Excellent **Exceeds Objective**		**2.00**
Good Meets Objective		**1.00**
Poor **Does not meet Objective**		**0.00**

Environmental Quality Grade	Index	Final EQI Ranks	Letter Grade
GOOD		7.5 – 10.0	A
ACCEPTABLE		5.0 – 7.5	B
POOR		2.5 – 5.0	C
IMPAIRED		0.0 – 2.5	F

Stations	Latitude	Longitude	EQI value-DIN	EQI value-DIP	EQI value-Chlorophyll *a*	EQI value-Turbidity	EQI value-DO	Total EQI value	Final Grade
1	30.3407	-89.3554	1.00	1.00	2.00	1.00	2.00	7.00	B
2	30.37218	-89.3085	2.00	2.00	2.00	1.00	2.00	9.00	A
3	30.35926	-89.314	2.00	2.00	2.00	1.00	2.00	9.00	A
4	30.35503	-89.2947	2.00	2.00	2.00	1.00	2.00	9.00	A
5	30.34283	-89.2681	1.00	0.00	2.00	1.00	2.00	6.00	B
6	30.34686	-89.3046	2.00	1.00	2.00	1.00	2.00	8.00	A
6a	30.33847	-89.3322	1.00	1.00	2.00	0.00	2.00	6.00	B
7	30.34257	-89.2976	2.00	1.00	2.00	1.00	2.00	8.00	A
8	30.31344	-89.3061	2.00	1.00	2.00	1.00	2.00	8.00	A
9	30.28795	-89.2987	1.00	0.00	2.00	1.00	2.00	6.00	B
Final Average EQI								**7.60**	**A**

Comparisons with Previous Datasets (1977-78, 1995-96, and 1997-98)

Weather and environmental quality data similar to those obtained during 2003-04 were available from previous years for three different periods. These data were also analyzed for spatial and temporal variability and were compared with the 2003-04 dataset. Highest total daily precipitation amongst all data periods was recorded during 1997-98 (January 1998, 34.57 cm) followed by February 2004 (28.85 cm) and July 2003 (26.06 cm) (Figure 4.44). Total precipitation generally increased in spring during all data periods. High rainfall was also observed in the winters of 1977-78 (January 1978) and 1997-98 (January 1998). There was also an increase in rainfall during summer of 1997-98 (July 1997) and 2003-04 (June 2003). Thus, high total precipitation was observed in winter and spring of 1977-78, in spring of 1995-96, in summer, winter and spring of 1997-98, and in summer and spring of 2003-04. River discharge records were similar to the seasonal patterns of total daily precipitation (Figure 4.45). Average daily discharge was highest in January 1998 (93.19 m^3s^{-1}) over all data periods. High river flow was generally observed in spring during all periods. River discharge was also high during the winters of 1977-78 (January 1978) and 1997-98 (January 1998). An increase in river flow was also seen during summer of 2003-04 (July 2003). Seasonal peaks in precipitation and discharge occurred during different months (seasons) in different data periods.

Nutrient concentrations in the bay changed significantly over the different study periods (Table 4.6). Total DIN concentrations measured during 1995-96 were significantly higher ($p < 0.01$, $df = 3$, $H = 23.21$) than other years (Figure 4.46). Bay-wide averaged values during this period ranged from 8.93 µM to 11.42 µM. Lowest DIN concentrations were observed in 2003-04, when bay-wide averaged concentrations were found in the range of 0.80 μM to 5.12 μM. DIN concentrations during all four study periods increased during the months of increased river discharge. A significant difference ($p < 0.01$, df = 3, H = 11.83) among the four data periods was also observed in measured concentrations of DIP (Figure 4.47). The highest DIP values were observed again in the 1995-96 study period. Concentrations averaged over all stations in the bay in 1995-96 ranged from 0.37 μM to 1.56 μM. The lowest DIP values were reported during the year 1997-98 and were found in the range of 0.12 μM to 0.42 μM. Average DIP concentrations increased in the fall and decreased in the winter during both the 1997-98 and 2003-04 study years.

Chlorophyll *a* values averaged across the bay ranged from 4.30 μgL^{-1} to 18.90 μgL^{-1} during 1977-78, the data period with the highest chlorophyll *a* concentrations (Figure. 4.48). Chlorophyll *a* concentrations in the estuary changed significantly ($p < 0.01$, df = 3, H = 13.74) over time. Low mean chlorophyll *a* values were observed in both 1995-96 (3.44 μgL^{-1} to 6.88 μgL^{-1}) and 1997-98 (1.82 μgL^{-1} to 6.35 μgL^{-1}) data periods. Significant differences in turbidity ($p < 0.01$, df = 2, H = 11.99) were observed in the bay between the three study periods 1977-78, 1997-98, and 2003-04 (Figure 4.49). The highest mean turbidity was reported in 1997-98 when values ranged from 6.54 NTU to 56.25 NTU. Maximum turbidity recorded during this study period occurred in January 1998, also the month of high river discharge. The lowest mean turbidity levels were seen in 1977-78 study period when values ranged from 3.57 NTU to 12.60 NTU.

There were no statistically significant differences in mean DO concentrations between the three different data sets: 1995-96, 1997-98, and 2003-04 (Figure 4.50).

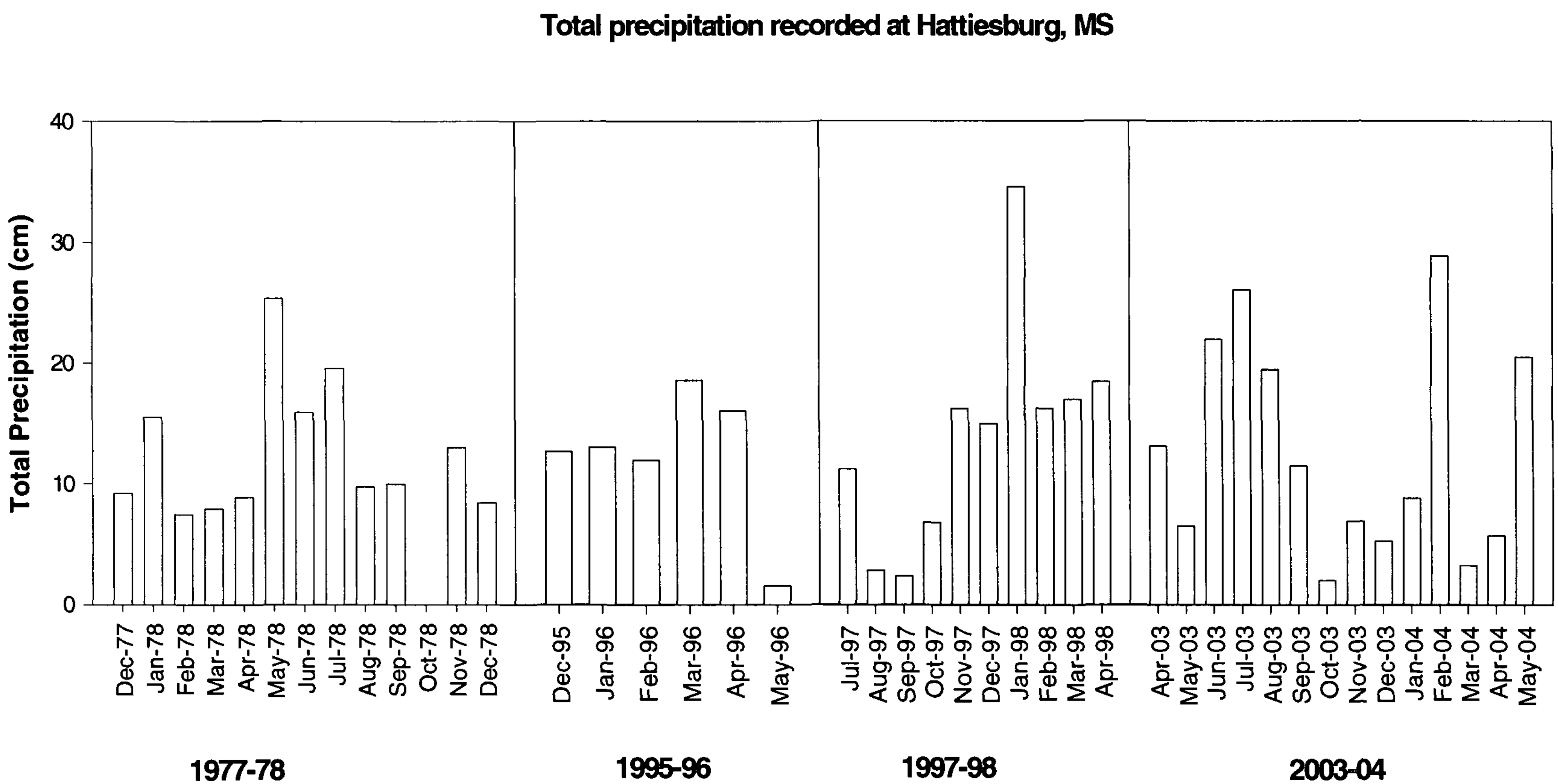

Figure 4.44 Total Monthly Precipitation (cm) Measured at Hattiesburg, MS over the Four Data Periods, 1977-78, 1995-96, 1997-98, and 2003-04. Data obtained from the National Climatic Data Center (NCDC) of NOAA. Highest total daily precipitation amongst all data periods was recorded during 1997-98 (January 1998) followed by February 2004 and July 2003.

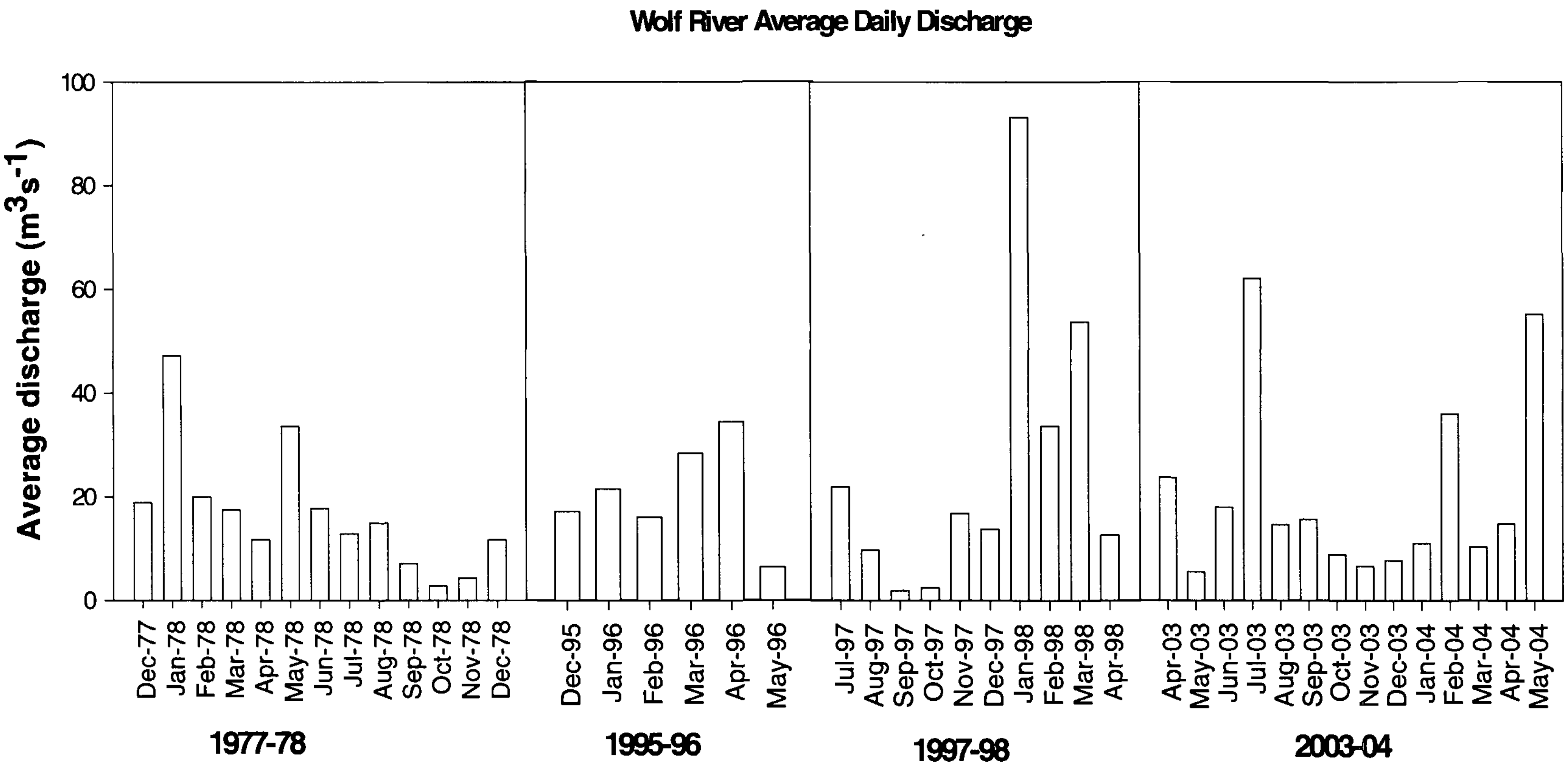

Figure 4.45 Average Daily Discharge for Wolf River Measured over the Four Data Periods at USGS Station near Landon, MS. Highest average daily discharge among all data periods was recorded in January 1998 (93.19 m^3s^{-1}). High river flow was observed in spring during all periods.

Table 4.6 Kruskal-Wallis One-Way Analysis of Variance on Ranks for Determining Temporal Variability in Bay-Wide Averages of Environmental Quality Parameters Measured over Four Different Data Periods (1977-78, 1995-96, 1997-98, and 2003-04). Kruskal- Wallis statistic H, degrees of freedom and p values are presented. Probability <0.05 (highlighted in bold) indicate a significant difference between one or more study periods. Significant variability was observed in DIN, DIP, and Chlorophyll *a* concentrations as well as in turbidity values between the different data periods. DO concentrations, however, did not significantly change over the different sampling periods.

Indicator	K-W statistic (H)	Degrees of freedom (df)	Probability (p)
DIN	23.21	3	**0.001**
DIP	11.83	3	**0.008**
Chlorophyll *a*	13.74	3	**0.003**
Turbidity	11.99	2	**0.002**
DO	0.53	2	0.768

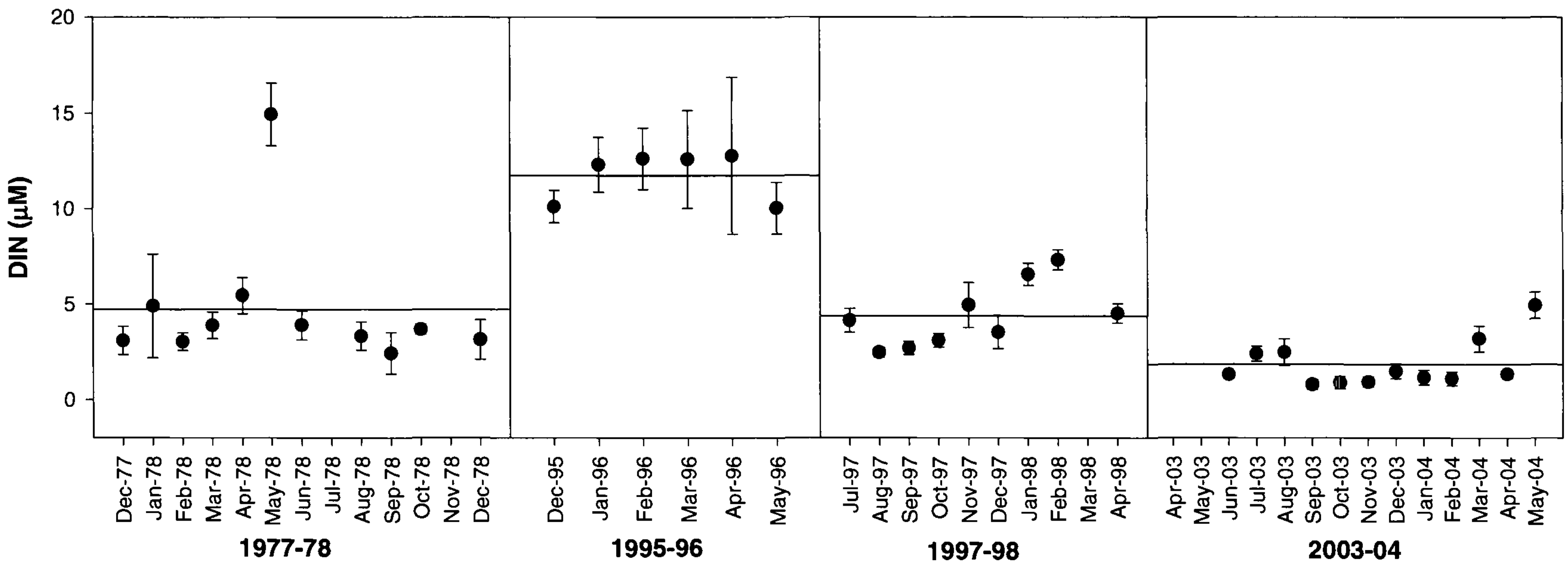

Figure 4.46 Mean and Standard Error of DIN Concentrations Averaged over All Sampling Locations in the Bay for Every Month of Sampling of Each of the Four Data Periods. Black circles denote the mean value of DIN concentrations measured at all locations during each month. Error bars are standard errors of the mean. Red horizontal line denotes the average concentration over an entire data period. Total DIN concentrations measured during 1995-96 were significantly higher ($p < 0.01$, $df = 3$, $H = 23.21$) than other years. Bay-wide averaged values during this period ranged from 8.93 µM to 11.42 µM. Lowest DIN values were observed in 2003-2004.

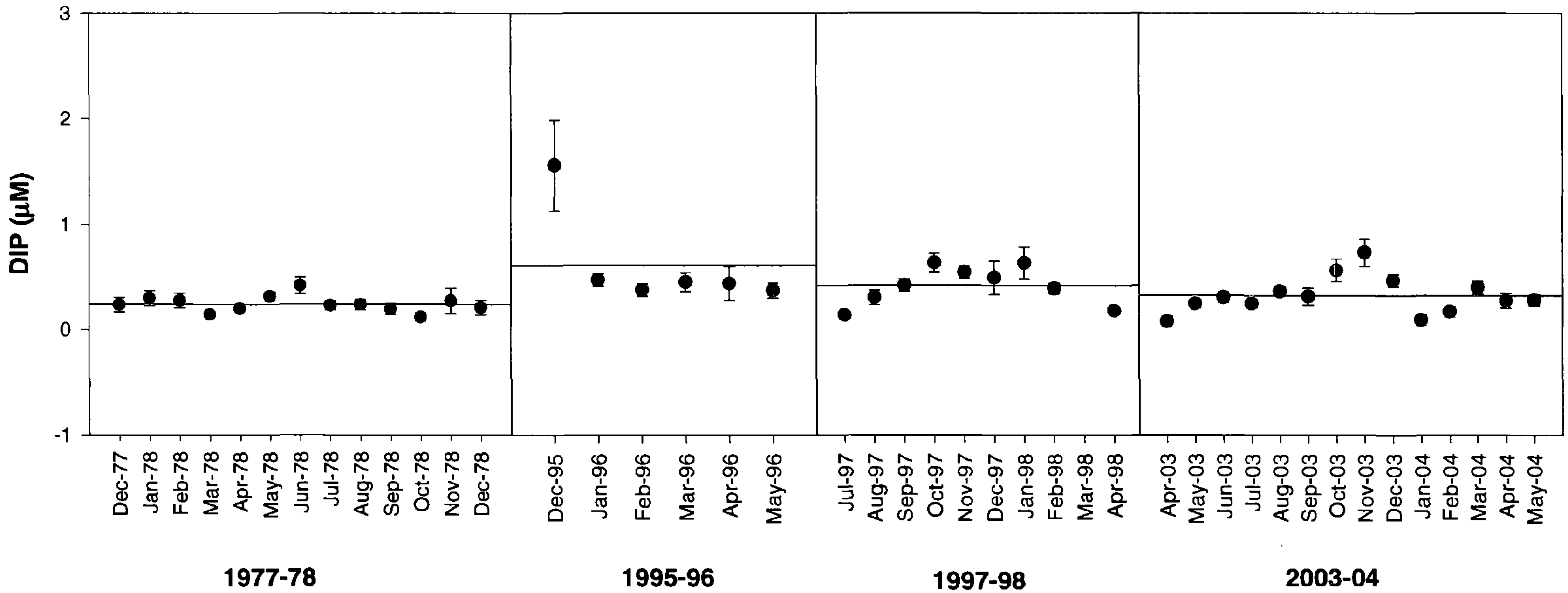

Figure 4.47 Mean and Standard Error of DIP Concentrations Averaged over All Sampling Locations in the Bay for Each Month of Sampling in the Four Data Periods. Black circles denote the mean value of DIP concentrations measured at all locations during each month. Error bars are standard errors of the mean. Red horizontal line denotes the average concentration over an entire data period. Highest DIP values were observed again during the 1995-96 study period. Concentrations averaged over all stations in the bay, in 1995-96, ranged from 0.37 µM to 1.56 µM. Lowest DIP values were reported during the year 1997-98 and were found in the range of 0.12 µM to 0.42 µM.

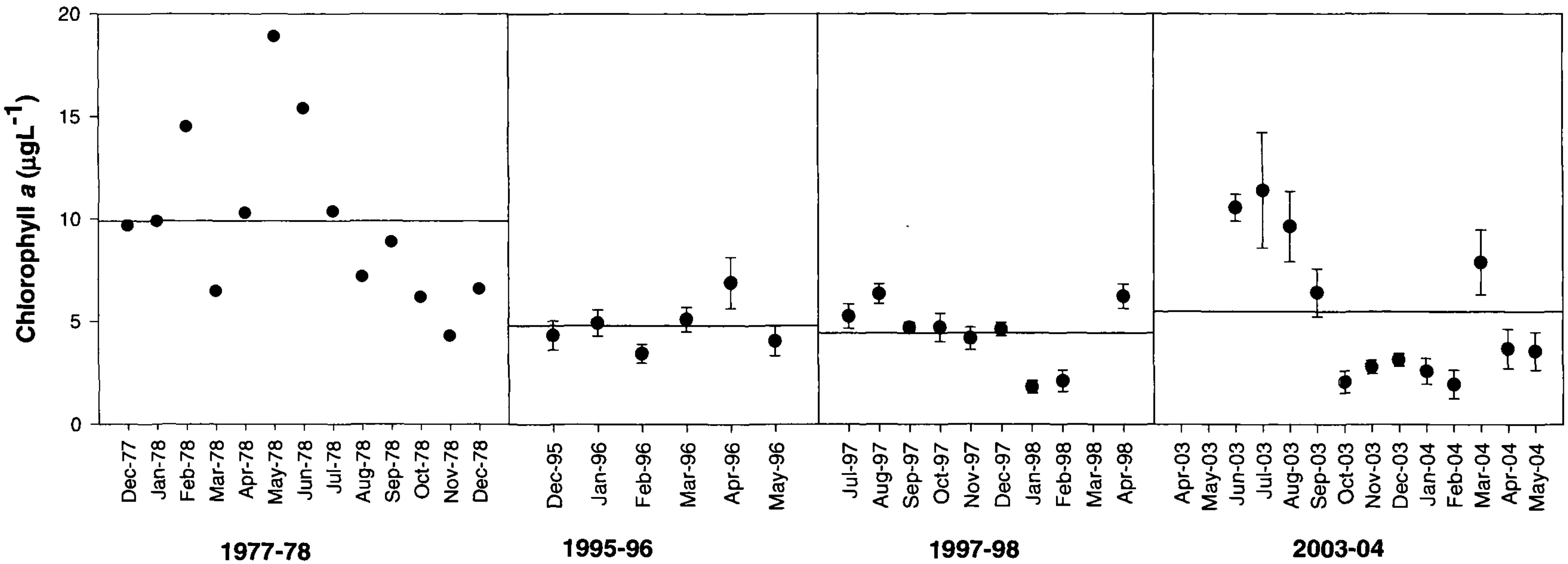

Figure 4.48 Mean and Standard Error of Chlorophyll *a* Concentrations Averaged over All Stations in the Bay for Every Month of Sampling During the Four Data Periods. Black circles denote the mean value of chlorophyll *a* concentrations measured at all locations during each month. Error bars are standard errors of the mean. Red horizontal line denotes the average concentration over an entire data period. Note: Standard error values were not available for the 1977-78 data set. Highest chlorophyll *a* concentrations were observed during 1977-78. Lowest mean chlorophyll *a* values were observed in both 1995-96 (3.44 μgL^{-1} to 6.88 μgL^{-1}) and 1997-98 (1.82 μgL^{-1} to 6.35 μgL^{-1}) data periods.

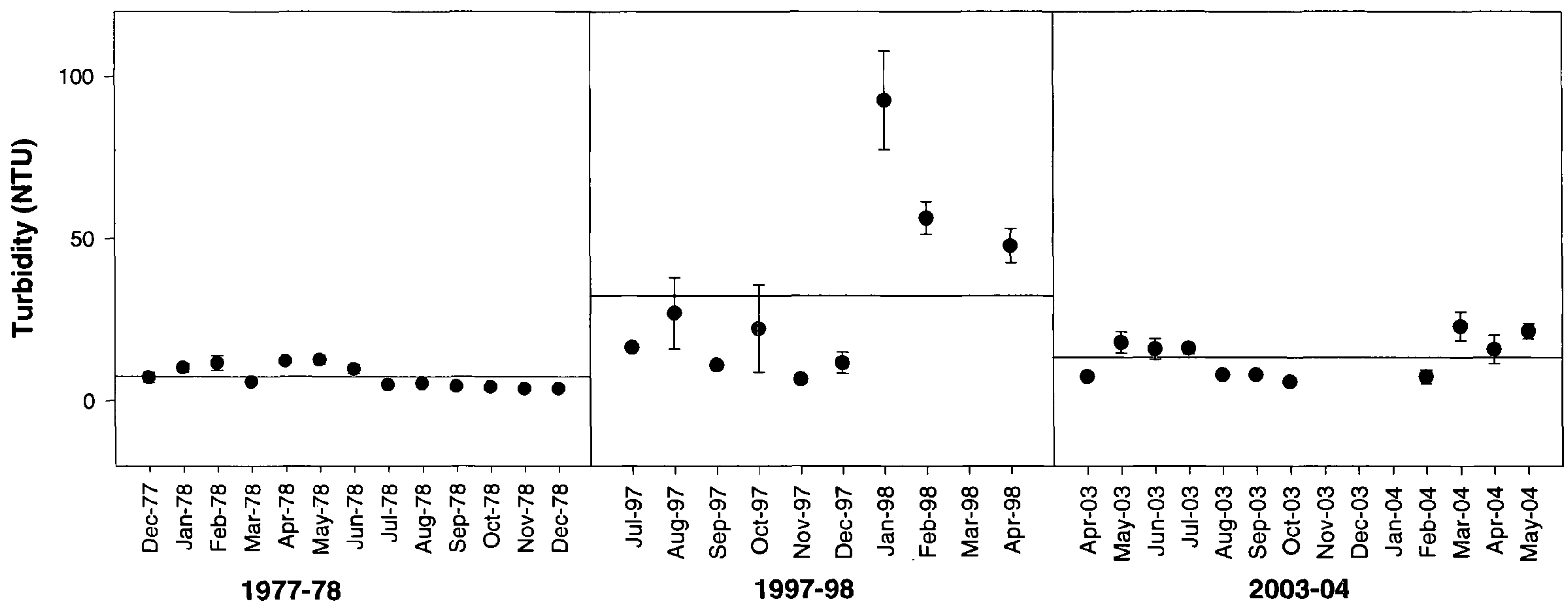

Figure 4.49 Mean and Standard Error of Turbidity Values Presented as a Bay-Wide Average for Each Month of Sampling for the Three Data Periods. Black circles denote the mean turbidity values measured at all locations during each month. Error bars are standard errors of the mean. Red horizontal line denotes the average value over an entire data period. Highest mean turbidity was reported in 1997-98 when values ranged from 6.54 NTU to 56.25 NTU Note: Data were available for only three of the four study periods. 1995-96 turbidity data were not available. Lowest mean turbidity levels were seen in 1977-78 study period when values ranged from 3.57 NTU to 12.60 NTU.

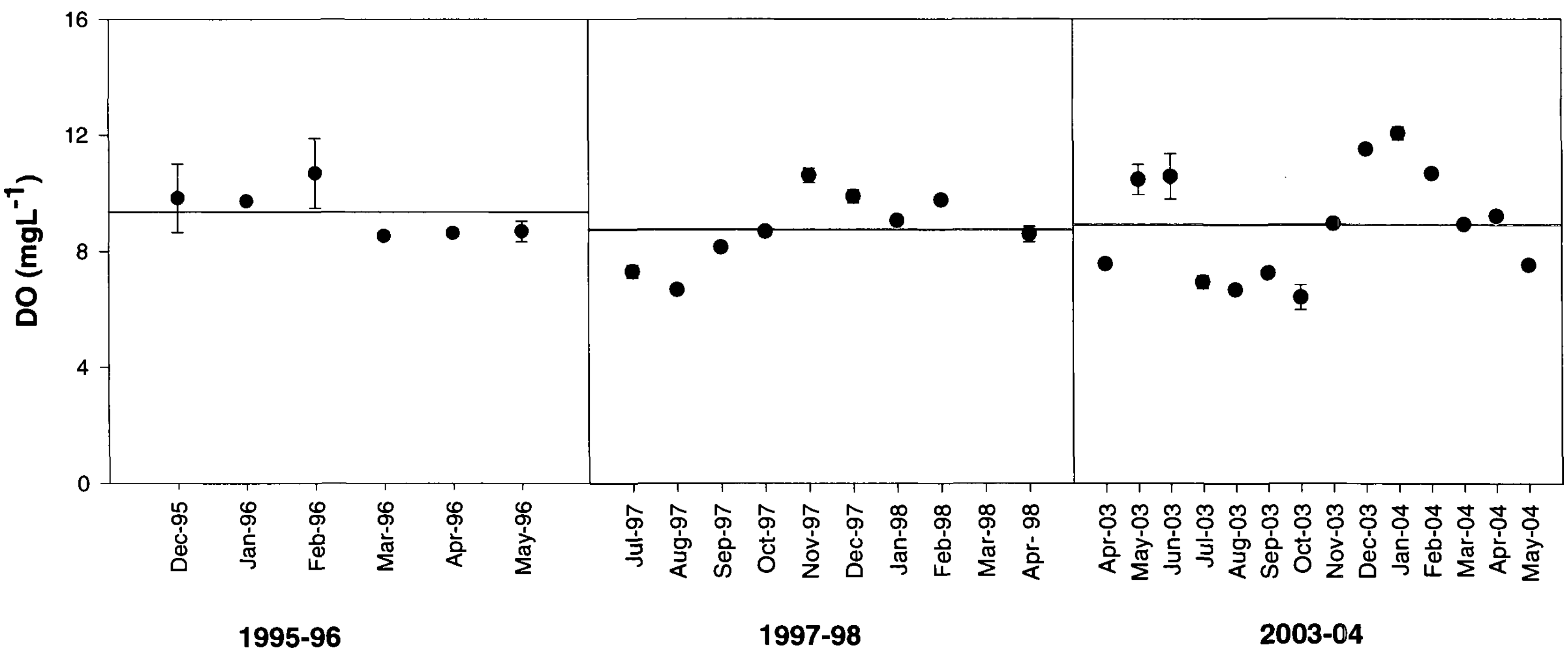

Figure 4.50 Mean and Standard Error of DO Concentrations Presented as a Bay-Wide Average for Each Month of Sampling for the Three Data Periods. Black circles denote the mean turbidity values measured at all locations during each month. Error bars are standard errors of the mean. Red horizontal line denotes the average concentration over an entire data period. No statistically significant differences in mean DO concentrations were observed between the three different data sets, 1995-96, 1997-98, and 2003-04. Values during these three study periods were in the range of 6.41 mg L^{-1} to 12.06 mg L^{-1}. Note: Data were available for only three of the four study periods. DO concentrations data were not available for 1977-78.

Average values during these three study periods were in the range of 6.41 mg L^{-1} (200.32 μmol kg^{-1}) to 12.06 mg L^{-1}(376.89 μmol kg^{-1}). Both the lowest and the highest values in the range were recorded in the year 2003-04.

Climate Oscillations and Environmental Quality

The four different data periods fell under different climate regimes. Based on the Southern Oscillation Index, 1995-96 was a La Niña year (high positive SOI values), 1997-98 data period was in the El Niño phase (high negative SOI values), and 1977-78 and 2003-04 were normal years (SOI values ranging between 0-10, based on the SOI index calculated by the Commonwealth of Australia, Bureau of Meteorology; where the final values are calculated by multiplying the index values (standardized anomaly) by 10 so that SOI ranges from -35 to +35) (Figure 4.51). Precipitation patterns in the region varied during different data periods with highest precipitation recorded during the El Niño phase and least precipitation during the La Niña year. Distinct patterns were also observed in average discharge from the Wolf River, with the highest discharge observed during the 1997-1998 El Niño phase. Similarly, DIN concentrations were correlated significantly with average discharge from Wolf river ($r = 0.384$, $p < 0.05$, $N = 38$) for all data periods.

Environmental Quality Index Values for Each Data Period

EQI values for DIN concentrations were zero (indicating poor environmental quality) at stations 1, 5, and 6a during both 1977-78 and 1997-98 data periods (Table 4.7). EQI values were also zero at station 4 during 1977-78 and at stations 7 and 9 during 1997-98. EQI values for DIN concentrations during 1995-96, however, were zero at all stations in the bay,while they were higher than zero during 2003-04. EQI values for DIP

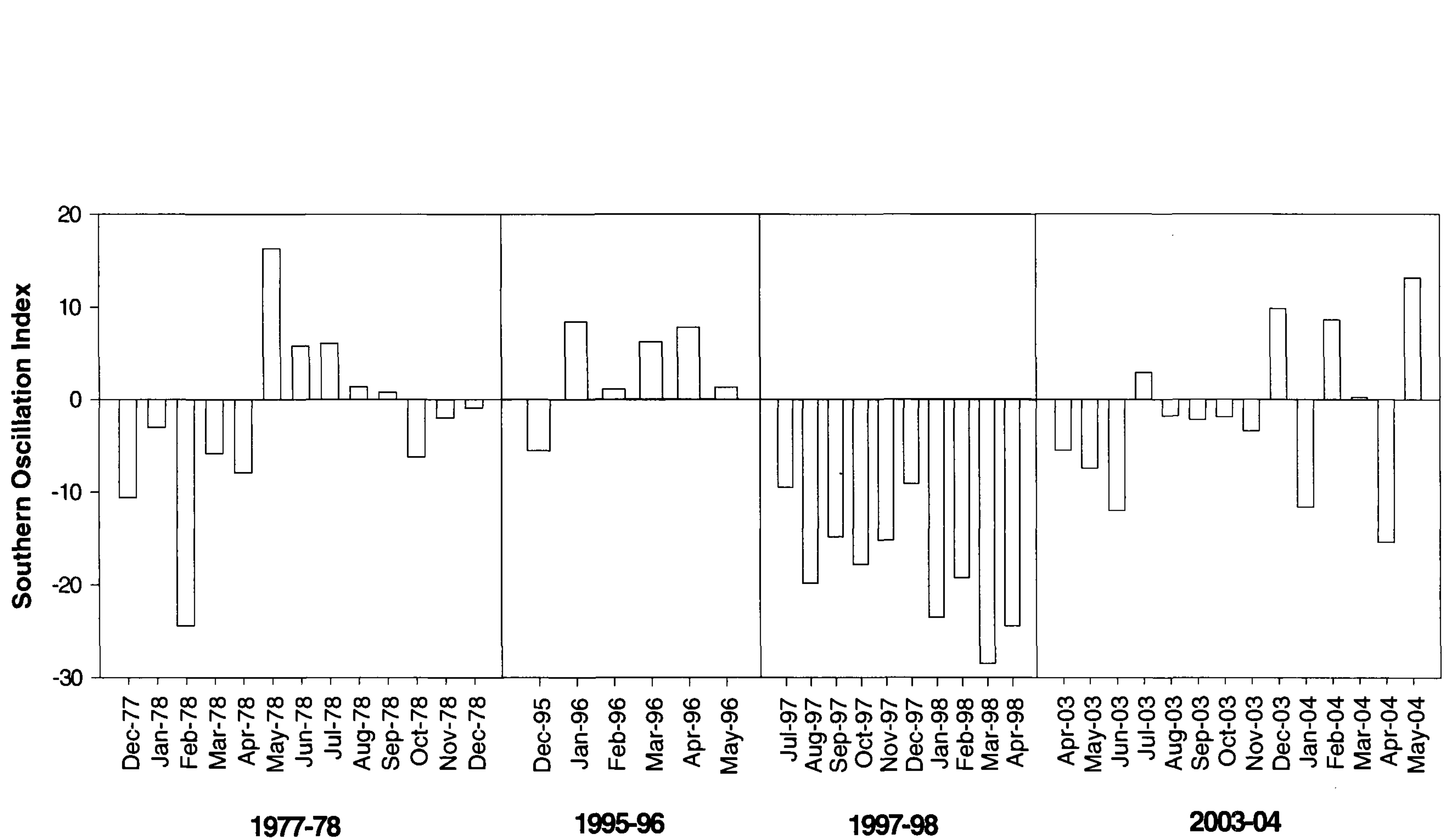

Figure 4.51 Southern Oscillation Index (SOI) Calculated from Monthly (or Seasonal) Fluctuations in Mean Sea-Level Pressure Differences between Tahiti and Darwin, Australia. Data Obtained from Commonwealth of Australia, Bureau of Meteorology. Note that the final SOI value is calculated by multiplying the index value (standardized anomaly) by 10 so that SOI ranges from -35 to +35. High positive values indicate La Niña, high negative values indicate El Niño, while values close to zero indicate normal conditions. 1995-96 was a La Niña year, 1997-98 was in the El Niño phase, and 1977-78 and 2003-04 were normal years.

concentrations were zero at station 5 during all data periods (Table 4.8). The EQI values were also zero at stations 1, 8, and 9 during 1997-98 and at station 9 during 2003-04. EQI values for DIP concentrations during 1995-96, however, were zero at all stations except station 4. (Note: Data were not available for stations 8 and 9 during 1977-78. Data were also not available for station 6a during 1995-96 and for station 6 during 1997-98 since those stations were not sampled during the respective periods).

EQI values for chlorophyll *a* and dissolved oxygen were above zero during all data periods (Tables 4.9 and 4.11). (Note: Chlorophyll *a* data were not available for stations 1, 4, 5, 6, 8 and 9, while dissolved oxygen data were not available for all stations during 1977-78. Both chlorophyll *a* and dissolved oxygen data were not available for station 6a during 1995-96 and for station 6 during 1997-98 since those stations were not sampled during the respective periods). EQI values for turbidity were zero at all stations during 1997-98 and at station 6a during 2003-04 (Table 4.10). EQI values for turbidity were above zero at all stations during 1977-78. (Note: Turbidity data were not available for stations 8 and 9 during 1977-78 and for all stations during 1995-96. Data were also not available for station 6 during 1997-98 since that station was not sampled during that period).

Table 4.7 EQI Values for Average DIN Concentrations Observed During the Four Different Data Periods. EQI values for DIN concentrations during 1995-96 were zero at all stations, indicating that the objective of reduced nutrients was not met at any station in the bay for that year. EQI values were also zero at stations 1, 4, 5, and 6a during 1977-78; and at stations 1, 5, 6a, 7, and 9 during 1997-98. Note: ND = No data. Data were not available for stations 8 and 9 during 1977-78; for station 6a during 1995-96 and station 6 during 1997-98.

Stations	Latitude	Longitude	1977-78	1995-96	1997-98	2003-04
1	30.3407	-89.3554	0	0	0	1
2	30.37218	-89.3085	1	0	1	2
3	30.35926	-89.314	1	0	1	2
4	30.35503	-89.2947	0	0	1	2
5	30.34283	-89.2681	0	0	0	2
6	30.34686	-89.3046	1	0	ND	2
6a	30.33847	-89.3322	0	ND	0	1
7	30.34257	-89.2976	1	0	0	2
8	30.31344	-89.3061	ND	0	1	2
9	30.28795	-89.2987	ND	0	0	2

Table 4.8 EQI Values for Average DIP Concentrations Observed During the Four Different Data Periods. EQI values were zero at station 5 during all data periods. EQI values for DIP concentrations during 1995-96 were zero at all stations except station 4. EQI values were also zero at stations 1, 5, 8, and 9 during 1997-98; and at stations 5 and 9 during 2003-04. Note: ND = No data. Data were not available for stations 8 and 9 during 1977-78 and for station 6a during 1995-96 and for station 6 during 1997-98.

Stations	Latitude	Longitude	1977-78	1995-96	1997-98	2003-04
1	30.3407	-89.3554	1	0	0	1
2	30.37218	-89.3085	2	0	1	2
3	30.35926	-89.314	2	0	1	2
4	30.35503	-89.2947	1	1	1	2
5	30.34283	-89.2681	0	0	0	0
6	30.34686	-89.3046	2	0	ND	1
6a	30.33847	-89.3322	2	ND	1	1
7	30.34257	-89.2976	2	0	1	1
8	30.31344	-89.3061	ND	0	0	1
9	30.28795	-89.2987	ND	0	0	0

Table 4.9 EQI Values for Average Chlorophyll *a* Concentrations Observed During the Four Different Data Periods. EQI values were above zero at all stations during all data periods. Note: ND = No data. Chlorophyll *a* data were not available for stations 1, 4, 5, 6, 8 and 9 during 1977-78. Data were also not available for station 6a during the 1995-96 and for station 6 during 1997-98 sampling periods.

Stations	Latitude	Longitude	1977-78	1995-96	1997-98	2003-04
1	30.3407	-89.3554	ND	2	2	2
2	30.37218	-89.3085	2	2	2	2
3	30.35926	-89.314	2	2	2	2
4	30.35503	-89.2947	ND	2	2	2
5	30.34283	-89.2681	ND	2	2	2
6	30.34686	-89.3046	ND	2	ND	2
6a	30.33847	-89.3322	2	ND	2	2
7	30.34257	-89.2976	2	2	2	2
8	30.31344	-89.3061	ND	2	2	2
9	30.28795	-89.2987	ND	2	2	2

Table 4.10 EQI Values for Average Turbidity Observed During the Four Different Data Periods. EQI values were zero at all stations during 1997-98, indicating that the objective of clear waters in the bay was not met that year. EQI values were above zero at all stations during 1977-78, indicating that the objective of clear waters was met that year. EQI values for turbidity were zero only at station 6a and were above zero at all other stations during 2003-04. Note: ND = No data. Turbidity data were not available for all stations during 1995-96. Data were also not available at stations 8 and 9 during 1977-78 and at station 6 during 1997-98.

Stations	Latitude	Longitude	1977-78	1995-96	1997-98	2003-04
1	30.3407	-89.3554	2	ND	0	1
2	30.37218	-89.3085	2	ND	0	1
3	30.35926	-89.314	2	ND	0	1
4	30.35503	-89.2947	2	ND	0	1
5	30.34283	-89.2681	2	ND	0	2
6	30.34686	-89.3046	2	ND	ND	1
6a	30.33847	-89.3322	2	ND	0	0
7	30.34257	-89.2976	2	ND	0	1
8	30.31344	-89.3061	ND	ND	0	1
9	30.28795	-89.2987	ND	ND	0	1

Table 4.11 EQI Values for Average DO Concentrations Observed During the Four Different Data Periods. EQI values were above zero at all stations during all data periods. Note: ND = No data. DO data were not available for all stations during 1977-78. Data were also not available for station 6a during the1995-96 and for station 6 during the 1997-98 sampling periods.

Stations	Latitude	Longitude	**1977-78**	**1995-96**	**1997-98**	**2003-04**
1	30.3407	-89.3554	ND	2	2	2
2	30.37218	-89.3085	ND	2	2	2
3	30.35926	-89.314	ND	2	2	2
4	30.35503	-89.2947	ND	2	2	2
5	30.34283	-89.2681	ND	2	2	2
6	30.34686	-89.3046	ND	2	ND	2
6a	30.33847	-89.3322	ND	ND	2	2
7	30.34257	-89.2976	ND	2	2	2
8	30.31344	-89.3061	ND	2	2	2
9	30.28795	-89.2987	ND	2	2	2

Calculation of Residence Time for the Bay of St. Louis

The annual average river discharge data available for both the rivers (USGS 2004) indicated that the Jourdan River streamflow was greater than the Wolf River by a factor of 1.5. Assuming that the daily average discharge of the Jourdan River was one and a half times that of the Wolf River, the total daily average freshwater discharge into the bay would be about 50 m^3s^{-1} under normal (average rain) conditions. The total daily average discharge from both rivers during the low flow conditions would be about 6 m^3s^{-1}, while that under the extreme high flow period such as storm events would be about 800 m^3s^{-1}. Considering the total daily average discharge from both rivers during three different scenarios: the low-flow period or the minimum flow recorded, the average-flow period, and the maximum discharge measured (such as that recorded during a storm event), the freshwater displacement time for the Bay of St. Louis could be calculated as follows:

The fresh water replacement time in the estuary can be calculated as:

$$FRT = \left(\frac{Ss - Se}{Ss}\right) * \frac{V}{Qf}$$

Where,

FRT – Freshwater replacement time

Ss – Salinity at the seaward end (average) = 12.9

Se – Salinity within the estuary (average) = 9.1

V – Volume of the estuary = 60,000,000 cubic m (59816585 m^3)

Qf_a – Total freshwater input (average flow) = 50 m^3s^{-1}

Qf_b – Total freshwater input (maximum flow) = 799 m^3s^{-1}

Qf_c – Total freshwater input (low flow) = 6 m^3s^{-1}

$$FRT = \left(\frac{12.9 - 9.1}{12.9}\right) * \frac{59816585}{50} = 4.1 \text{ days}$$

$$FRT = \left(\frac{12.9 - 9.1}{12.9}\right) * \frac{59816585}{799.25} = 0.3 \text{ days}$$

$$FRT = \left(\frac{12.9 - 9.1}{12.9}\right) * \frac{59816585}{6.25} = 32.7 \text{ days}$$

CHAPTER V

DISCUSSION

Overview

The environmental quality for the entire bay during 2003-2004 was found to be marginally "Good". The contradiction of this result with that of the MDEQ, which listed the bay as impaired, arose due to the selection of different parameters to indicate environmental quality. It is essential to use a comprehensive suite of indicators that can represent all components of the ecosystem for effective assessment of environmental quality. The major factors influencing changes in the environmental quality of the Bay of St. Louis were rainfall, streamflow, tidal exchange and wind. The environmental quality indicators were correlated significantly to these weather and physical factors and varied spatially and temporally. The indicator values at different locations varied due to the proximity of the locations to point sources of nutrients as well as due to the runoff from the subwatersheds entering into the bay along those locations. The indicator values at individual locations varied during different times of the year due to changes in precipitation, stream flow, and wind stress and direction. The environmental quality of the bay changed over four data periods (1977-78, 1995-96, 1997-98, and 2003-04), which fell under different phases of the ENSO. This was due to the variability in indicator parameter values that occurred as a result of changes in local weather and physical conditions (such as changes in precipitation, river discharge, and wind patterns) in response to the changes in climate patterns (ENSO phases). Differences in land use in the subwatersheds also affected the input and delivery of nutrients at different locations in the bay, thus affecting the overall environmental quality of the estuary. This relationship between increased urban and agricultural land use in the subwatershed and increased nutrients in the bay was, however, not seen over the long-term period (four different data periods). This was likely in part due to the daily tidal flushing, restorative actions in the watershed, and the impact of climate variability. The growing population and increasing urbanization in the watershed along with the natural variability in climate can, however, pose a greater threat to the environmental quality of the estuary. Integration of information about the behavior and responses of the bay to these stimuli requires extensive scientific evaluations as well as active communication between the land (urban development) and coastal managers. Based on the results of this study, following recommendations are made for effective management of the Bay of St. Louis: increased monitoring at point sources and during events of increased discharge (on seasonal and inter-annual/decadal scales), inclusion of indicators representing the structure and functioning of the entire ecosystem, implementation of science-based tools such as the spatially explicit index and environmental quality report card developed in this study, coordinated efforts with land managers overlooking developments in the watershed, integration of information regarding variability in climate, promotion of cooperation and active exchange of ideas between all stakeholders (including regulatory agencies, scientists, and the community), and initiation and development of a comprehensive and continually evolving, flexible and adaptive management program.

Factors Influencing the Environmental Quality of the Bay of St. Louis

Factors Affecting Circulation of Nutrients and Pollutants

The environmental quality of the Bay of St. Louis appeared to be impacted mainly by changes in nutrient concentrations and turbidity values. These changes occurred primarily due to the following factors:

- Precipitation and River discharge
- Tidal exchange
- Wind

Changes observed in the delivery and movement of nutrients and sediments in the bay were largely due to variability in rates of fresh water input to the estuary. DIN concentrations and turbidity values in the Bay of St. Louis exceeded target values during periods of increased stream flow following high precipitation events. DIN concentrations in excess of 5μM and turbidity values higher than 20 NTU were observed in May 2004, which was the period of increased river discharge as recorded at Wolf River near Landon, MS. The impact of freshwater discharge on the estuarine environmental quality was most evident in areas near the river mouths. Increased DIN concentrations were recorded at stations located closer to the Jourdan River along the western shore of the bay. High median DIN concentrations in the range of 2μM to 3μM were observed at stations representing the Jourdan and Wolf rivers. Median DIN concentrations at other locations in the bay remained lower than those at river mouths, thus indicating that the river discharge was the primary source of DIN to the Bay of St. Louis.

DIP concentrations, on the other hand, were higher in the vicinity of Bayou Portage and at the mouth of the bay adjacent to the Mississippi Sound. The Mississippi

Sound was found to be an important source of phosphates to the bay. The high DIP concentrations in the Sound in this region were likely due to the phosphate rich waters coming from the East carried by the prevailing westward flowing currents. The Mississippi Sound receives discharge waters from several streams, bayous, bays, and rivers that drain the increasingly developing watersheds of the northern Gulf States. Several commercial and industrial dischargers including fertilizer and concrete manufacturing plants, where phosphate compounds are produced or used in large quantities, are located along the coast in the states of Mississippi and Alabama. Although there have been reports of increased phosphate discharges from some of these industries in the past, no direct evidences linking these dischargers and the high PO_4 concentrations in Mississippi Sound were investigated in this study. The source of high DIP concentrations ($> 0.6\mu M$) measured at the mouth of Bayou Portage was also most likely the discharge from the concrete facility located on Bayou Portage in Pass Christian, MS (a company named Gulf Coast Pre-Stress Inc.), in addition to the sewage input from domestic and commercial dischargers. The DIP concentrations measured at the station located in the Sound were always higher than in rest of the bay except for at the mouth of Bayou Portage. During the incoming tide, the DIP rich waters from the Sound entered the bay and joined the waters flowing out from Bayou Portage creating a PO_4 rich area near the inlet. Thus, the tidal exchange between the waters of the Mississippi Sound and those of the Bay of St. Louis was equally important in affecting the estuarine environmental quality.

The diurnal tides introduce waters from the Mississippi Sound and replace the fresh water in the bay. The Gulf of Mexico tides are generally known to be small in amplitude (microtidal, 15-30 cm). Based on the gage height data recorded at the Bay Waveland Yacht Club, Bay St. Louis, MS, the average tidal height in the bay ranged from 40-60 cm (See Appendix B). In the absence of wind influence, the tidal reach and the residence time of the pollutants introduced by the tidal waters and by the rivers and bayous depended upon the discharge of the two rivers. The amount of time required for the flushing of the entire or most of the bay was related to the amount of freshwater replaced within the estuary. The residence time for pollutants introduced into the bay from the rivers, as calculated in this study, varied from about 0.33 days (maximum flow conditions such as storm events) to over a month (32.7 days representing low-flow or minimum flow period) depending upon the streamflow conditions. The average fresh water replacement time calculated for the estuary was 4.1 days during normal stream flow conditions ($50 m^3 s^{-1}$). This was, of course, assuming that the nutrient and pollutant inputs from other sources such as the bayous, surface runoff or ground water discharge were minimal compared to the total discharge from both rivers and that there was no uptake, attachment, or adsorption of nutrients and pollutants within the bay during that period.

In addition to the freshwater and tidal flows, circulation in the Bay of St. Louis was also affected by wind flow and wind direction. This shallow estuary (mean depth =1.5m) was well-mixed vertically due to wind-driven circulation. Most frequent daily wind direction during the sampling period was onshore (South-East) (See Appendix B). Daily average wind speed ranged from 0.07 ms^{-1} recorded in winter (February) to 1.84 ms^{-1} in spring (May). Increased wind speeds in spring (May 2004) were associated with increased precipitation events such as storms, which led to higher fresh water discharge and therefore increased flushing of the bay. Intense offshore winds, if occurring during storm events associated with frontal passages from the north may lead to increased flushing of the bay by pushing the estuarine waters farther out into the Sound, thus reducing the residence time of pollutants in the bay. Increased wind-driven mixing coupled with wave-induced currents could cause the estuarine waters to be forced offshore into the MS Sound or pushed farther inwards into the bay depending on the wind intensity and direction, either in coordination with the tides or against the tidal direction (Cobb and Blain 2002). Cobb and Blain (2002) demonstrated the movement of passive particles between the Sound and the bay using climatologies and an iterative hydrodynamic-wave coupled model and found that the wave-induced out-flowing current was stronger significantly than the tidally-induced flood currents. Strong wave-induced inflowing currents were observed along the sides of the inlet and the tracers (passive particles) near the sides moved into the bay with those currents, some of which were flushed out along the out-flowing current (Cobb and Blain 2002). However, while the tracers near the middle of the inlet were forced out into the Sound along the out-flowing current, other tracers remained trapped in a strong eddy on the right side of the inlet and did not exit the bay (Cobb and Blain 2002).

Lateral mixing and circulation due to high onshore winds (summer and fall) may cause the retention of surface waters even during an outgoing tide. Such observations were made by Caffrey and Day (1986) in the Fourleague Bay, LA, when river water from the Atchafalaya was piled up along the coast directed away from the bay during periods of steady southeasterly winds. On the other hand, the bay waters can be pushed farther out into the Sound during periods of high offshore winds such as prior to frontal passages from the north (generally late winter and spring). However, during the 2003-04 study, the most frequent wind direction was South-East and onshore winds were most prevalent during all seasons. The influence of tidal exchange and river discharge on the bay waters was seen in terms of changes in nutrient concentrations in the bay.

Major Sources and Flow Paths of Nutrients and Other Contaminants to the Bay

Point sources:

- STPs and wastewater outfalls
- River and bayou mouths (The mouths of rivers and bayous emptying into the bay were considered to be point sources for the purpose of this study, since these rivers, streams, and bayous were identified to be definite sources of nutrients/pollutants entering into the bay).
- Mississippi Sound

The total N and P loadings to the bay were not known. Although nutrient loads from the two rivers or the sewage outfalls were not recorded, the nutrient loadings from the sewage treatment plant outfalls in the Bay of St. Louis could be estimated using available discharge load data. The point source loadings from all sewage treatment plants (STPs) that discharged into the bay had a monthly average discharge load of 2.5 to 5 million gallons per day (MGD) (GMPO 2001). However, in the absence of tertiary treatment plants, the N and P loads from these outfalls were not monitored at these facilities. The monthly average ammonium loads from the largest discharger SRWMD (5MGD) were about 37 kg d^{-1} or 2 mgL^{-1}(GMPO 2001). Therefore, the annual average ammonium discharge was 13,592 kg. Assuming that the mass ratios of ammonium and nitrate to total STP N load (based on the composition typical of secondary-treated sewage treatment plant loads using CSIRO SER Model II calculations for point source loads (CSIRO Australia 2003) are 50% each, the total discharge from the 5MGD treatment plant could be estimated to be 27,185 kg N yr^{-1}. Although not accurate, these calculations provide a rough estimate of the N and P loads received by the bay. Since, the SRWMD plant had the highest monthly average flow of 5MGD, it could be assumed that this was the highest N load discharged from a single STP and similar or lesser loads were discharged from the rest of the STPs that emptied into the bay. Also, the highest DIN concentrations in the bay (highest median value of 3.30 μM) were consistently observed in the area near the mouth of the Jourdan River, which received waters from the SRWMD outfall via Edward's bayou. In comparison, N loads received by the Choptank River estuary, a subestuary of the Chesapeake Bay, from a 3.13 MGD plant average over 55, 000 kg N yr^{-1}, while the discharge from a 1.58 MGD plant averages about 21,000 kg N yr^{-1} (Staver et al. 1996; Jones et al. 2004). The estimated total nitrogen loading to the Bay of St. Louis from the highest discharger (5MGD plant) is almost half of that received by the Choptank River estuary from a 3.13 MGD plant. The difference between the nitrogen loadings is likely due to the larger agricultural and urban landuse in the Choptank River watershed (Jones et al. 2004).

Non point sources:

In addition to the point sources (wastewater outfalls and river and bayou mouths), non point sources such as surface runoff, groundwater discharge, and atmospheric deposition are also important sources of pollutants to an estuary. Although extensive sampling was not conducted in this study to find the effects and contributions of the major diffuse sources, surface runoff is known as a prominent source from urban watersheds while ground water discharge is known to be a major source of nutrients and pollutants from urban and agricultural watersheds (Corbett et al. 1999). In a study using atmospheric deposition data collected at a site nearer to the Bay of St. Louis watershed, it was estimated that the annual wet deposition concentrations of nitrogen on croplands in the Bay St. Louis watershed were between 0.5mgL^{-1} and 1.5 mgL^{-1} as NO_3 and varied from 0.14mgL^{-1} to 0.42 mgL^{-1} of ammonium, while organic Nitrogen ranged from 0.17 mgL^{-1} to 0.44 mgL^{-1} (Kieffer 2002). Thus, the total N deposition on the croplands could be estimated to be between 0.8 mgL^{-1} and 2.4 mg L^{-1}. Although, the contribution of atmospheric deposition to the total nutrient load (in terms of percentage) were not known, such estimates provide a general idea of the kind and amount of nutrient loads that may be introduced into the bay via atmospheric deposition. It is also possible that direct atmospheric deposition may not be a significant contribution of nutrients as compared to the total terrestrial discharges of nutrients into the bay. Staver et al. (1996) found that direct atmospheric nitrogen input to the Choptank River estuary was less than twenty five percent of total diffuse-source N loadings and was a minor component relative to the terrestrial N input. However, further studies focused on investigating the flux of nutrients via atmospheric deposition, surface runoff, and groundwater discharge to the Bay of St. Louis will be useful in assessing the quantitative significance of each of the diffuse sources.

Spatial and Temporal Variability in Environmental Quality

Spatial Variability

Salinity in the bay was lowest near the river mouths and highest in the Mississippi Sound and was correlated to gage height. Such changes in salinity were expected in an estuary, where the Jourdan and Wolf rivers were the major sources of fresh water, while the saline waters from the Mississippi Sound entered the bay due to tidal forcing. Spatial variability was observed in certain environmental quality indicators during this study.

The nutrient concentrations at stations closer to the point sources were consistently higher than those at locations away from the point sources. The DIN concentrations during 2003-2004 were higher significantly at stations near the river mouths (station 1 and station 4) and sewage and wastewater outfalls (station 6a) than the rest of the bay. Similar observations of higher nutrient concentrations at river mouths were made in an earlier study in the bay (GCRL 1978) as well as in other shallow micro-tidal Gulf estuaries such as the Apalachicola Bay, the Atchafalaya Delta estuarine complex including the Vermillion Bay and the Cote Blanche Bays, and the Fourleague Bay (Pennock et al. 1999; Lane et al. 2002; Caffrey and Day 1986). Pennock et al. (1999) have shown that fresh water inputs are primary sources of nutrients in the river dominated estuaries of the Gulf of Mexico. Nitrate concentrations in the Atchafalaya delta estuarine complex regions decreased with distance from the river mouth and were significantly lower than those in the Atchafalaya River throughout the year except during spring (Lane et al. 2002). Nitrate is removed from estuarine environments by several processes including dilution, denitrification, chemical reduction, biological uptake, or burial. Lane et al. (2002) estimated that nearly 41% to 47% of nitrate in the Atchafalaya River was removed before reaching the Gulf waters. The circulation patterns in the Bay of St. Louis, however, allow for flushing of the nitrate rich waters from the rivers along the western and eastern shores into the Mississippi Sound.

The DIP concentrations in the bay were significantly higher on the eastern side near Bayou Portage (location of a sewage outfall) and at the station located in the Mississippi Sound. The DIP concentrations in the bay regions closer to the Mississippi Sound were also mostly higher during high/incoming tides and were correlated positively with salinity and gage height. These observations indicated that the Bayou Portage outfall and the Mississippi Sound were the most important sources of DIP to the bay. Higher nitrate, ammonium, and orthophosphate concentrations at the mouth of the rivers and the Bayou Portage were also observed in a previous study conducted by the GCRL (1978) in the bay. However, inspite of the high orthophosphate concentrations observed at the mouth of the two rivers and at Bayou Portage during the previous study, the values of orthophosphate were not different significantly throughout the bay (GCRL 1978). Also, DIP concentrations measured during that study were correlated negatively with salinity except at Bayou Portage (GCRL 1978). This indicated that the Mississippi Sound is a relatively new (less than thirty years) source of DIP to the bay. Several developments along the northern Gulf of Mexico such as increasing population (increased sewage pollution) and the rise in commercial and industrial dischargers including concrete and phosphate manufacturing units may have attributed to the higher DIP concentrations in the Mississippi Sound over the years. The significant spatial differences in DIP concentrations during 2003-04 may have been due to an increase in the phosphate inputs from Bayou Portage and the Mississippi Sound over the years as compared to the inputs from the rivers. And although increased sewage and industrial waste were likely the major causes, the exact sources of these increases in DIP concentrations in Bayou Portage and the Mississippi Sound were not determined in this study. Increases in dissolved phosphorus concentrations at higher salinities can also occur due to desorption process (Nedwell et al. 1999). Froelich (1988) suggested that the release of phosphates from fluvial inorganic suspended particles can lead to a 2-5 fold increase in total dissolved phosphorus load to the sea. Lane et al. (2002) found benthic remineralization to be the major source of higher phosphate concentrations in the Atchafalaya estuarine regions compared to the Atchafalaya River. However, the release of DIP may be limited or inhibited by lower particulate levels or masked by other processes such as biological uptake in some estuaries (Nedwell et al. 1999). The total phosphorus concentrations in Fourleague Bay, LA were not correlated with the river flood cycle and were not found to have changed significantly throughout the year (Madden et al. 1988).

Interestingly, differences in nutrient concentrations (DIN and DIP) were also observed between the western (areas close to Jourdan River) and northeastern (Upper-Wolf River side) regions of the bay. Higher nutrient concentrations were observed around the western areas that received discharge from the Jourdan River, which drained the subwatersheds with higher urban and agricultural use (Bayou La Croix, Rotten Bayou, and Upper Jourdan River subwatersheds). The nutrient concentrations in the northeastern bay that received runoff and discharge from subwatersheds with lower urban and agricultural use (De Lisle, Upper Wolf River, and Lower Wolf River) were lower significantly than the western bay. This suggested that differences in land use and land cover in the watershed affected the variability in concentrations of nutrients in the bay and therefore the environmental quality of the bay. Higher export of nutrients from the watershed with higher agricultural and urban land use occurs due to increased application of fertilizers for agricultural purposes as well as development of landscaped recreational areas such as golf courses and parks in urban communities. Similarly, deforestation and loss of wetlands along with increased runoff from urban residential and commercial areas with larger impervious surfaces, sewage and waste disposal facilities, and from animal farms can lead to increased export of nutrients into the surrounding waterbodies. Such differences in the nutrient and sediment inputs due to differences in the extents of urban and agricultural uses in the watersheds have been observed in several other studies (Jones et al. 2004; Dauer et al. 2000; Interlandi et al. 2003). Jones et al. (2004), in their pilot study in the Chesapeake Bay area found that the ecosystem health of the Choptank River was lower than that of the Patuxent River based on indicators such as the total nitrogen, total phosphorus, δ^{15}nitrogen, chlorophyll *a*, and dissolved oxygen concentrations and secchi depth. The differences in the ecosystem health were found to be largely due to the differences in the land uses between the two river watersheds (Jones et al. 2004). The Patuxent River watershed had large forested and extensive urban areas, with few sewage treatment plants located downstream of the river (Jones et al. 2004). The Choptank River, on the other hand, was mainly a large agricultural watershed with moderate urban use and had several sewage treatment plants located along the entire length of the river (Jones et al. 2004). Such differences in land use and land cover were observed in the Jourdan River and Wolf River watersheds, where higher urban and agricultural use is found in the Jourdan River watershed than the Wolf River drainage area (Figures 2.4 and 2.5). It is important to monitor the development in the watersheds and understand and address the sources and paths of nutrients into the bay in order to improve the environmental quality.

An important observation regarding the nutrient concentrations in the Bay of St. Louis was that the measured values were an order of magnitude lower than those found in other larger temperate estuaries (Weller et al. 2003; Boynton et al. 1995). Pennock et al. (1999) observed that such differences may be due to the relatively pristine nature of the river systems (as compared to the river systems of larger temperate estuaries with highly developed watersheds) that deliver nutrients to the river-dominated estuaries of the Gulf of Mexico such as the Bay of St.Louis (Pennock et al. 1999). However, higher nutrient concentrations (average DIN concentrations > 50μM) have been observed in neighboring Gulf of Mexico estuaries that receive fresh water input from larger river systems such as the Atchafalaya River (Caffrey and Day 1986; Madden et al. 1988; Lane et al. 2002). Under the circumstances of rising population and rapid urban development, anthropogenic activities in the watershed of the Bay of St. Louis may have to be monitored closely to prevent deterioration of the environmental quality of the bay in the future.

Significant spatial differences in other environmental quality indicators such as chlorophyll *a,* turbidity, and DO concentrations were not observed over the entire sampling period. However, spatial differences in these parameters were observed on certain individual sampling days. Changes in chlorophyll *a* concentrations representing algal biomass did not generally affect the overall environmental quality since the concentrations were mostly below bloom levels. The chlorophyll *a* concentrations were found to be higher than $20\mu gL^{-1}$ only once during the summer of 2003 at station 6a. This station was the shallowest station sampled and was also located close to sources of nutrients; a drainage ditch and mouth of the Jourdan River. High DIN concentrations were generally observed at station 6a. Also, although it was the station with highest turbidity values, turbidity at this location was lowest during summer and, therefore, allowed for more light penetration. Pennock et al. (1999) indicated that the algal biomass in the river-dominated estuaries of the Gulf was regulated by light and nutrient availability and shorter residence times. The increase in algal biomass at this station was, therefore, likely due to the greater availability of both nutrients and light. Thus, there is a possibility that an increase in nutrient availability (via increased discharges from sewage outfalls or increased runoff) with a simultaneous decrease in turbidity during summer will lead to increased chlorophyll concentrations ($>50\mu gL^{-1}$) and can result in algal blooms in the Bay of St. Louis.

The Bay of St. Louis was generally found to be turbid and significant spatial differences in turbidity were not observed over the course of this study. This was largely because of resuspension of sediments occurring throughout the shallow estuary due to wind mixing. Turbidity was, however, correlated negatively with salinity and decreased towards the mouth of the bay as salinity increased. This indicated that higher turbidity in low-salinity waters was mainly due to increased input of suspended material from river discharge. This observation was consistent with other studies carried out in similar shallow Gulf of Mexico estuaries such as the Fourleague Bay and the Atchafalaya Delta estuaries, where the Atchafalaya River was found to be the primary source of sediments to the bays (Lane et al. 2002; Caffrey and Day 1986). This observation was, however, in contrast to those of the study conducted in the bay during 1977-78 (GCRL 1978). The turbidity in the bay was then found to vary in positive correlation with salinity, indicating that the primary influence during that time was due to the incoming tides and increased suspended particle input from the Mississippi Sound (GCRL 1978). This spatial difference in turbidity between the two studies may have been due to the increase in the input of sediments and other suspended material from the rivers over the years (turbidity measured in this study was a measure of suspended particles). It is likely that changes in land use and land cover such as increased deforestation in the subwatersheds led to increased runoff and higher suspended matter in the river discharge. The turbidity levels in the bay during 2003-04 were always high at station 6a, which was also the shallowest station sampled. The bottom sediments in shallow waters are easily stirred up by wind mixing, which, in addition to river discharge, was an important factor in increasing the turbidity at station 6a.

Decreases in dissolved oxygen concentrations below 2 mgL^{-1} can lead to hypoxic conditions in an estuary. However, the DO concentrations at all stations in the bay were always above the upper threshold value of $5mgL^{-1}$ (often saturated or at times supersaturated) and hence did not impact the environmental quality negatively. Occurrence of hypoxic conditions in the Bay of St. Louis is not likely as vertical stratification was never observed in this shallow estuary, which remained well-mixed due to wind. Hypoxia in the Gulf of Mexico estuaries is known to occur due to vertical stratification, eutrophication, or a combination of both stratification and eutrophication (USEPA 2005). Although the Gulf coast estuaries are shallow and micro tidal, several of these systems exhibit a seasonal bottom hypoxia that is highly variable and can change over very short periods of time (Engle et al. 1999; Turner et al. 1987). Thus, the environmental quality of the bay during 2003-04 was mainly influenced by three (DIN, DIP and turbidity) of the five indicators since chlorophyll *a* and DO concentrations were mostly within acceptable ranges and close to pristine levels.

Temporal Variability

Changes in the values of the environmental quality indicators over time were related to changes in wind speed, precipitation and river discharge. Nutrient concentrations in the bay varied temporally and were correlated to changes in the meteorological parameters. The DIN concentrations in the bay increased significantly during the spring and summer seasons. These were also the periods of increased river discharge that impacted the environmental quality of this system. The increased precipitation and the resultant increase in river discharge and runoff introduced nutrients (particularly DIN) and sediments from the watershed to the bay. Similar observations were made in a previous study conducted in the bay, where concentrations of nitrate, the dominant form of inorganic nitrogen nutrients in the estuary, were higher following heavy rains in the spring (GCRL 1978). Similar increases in DIN concentrations during spring have been observed in other temperate and Gulf of Mexico estuarine systems and are known to be a characteristic of river-dominated estuaries (Schubel and Pritchard 1986; Madden et al. 1988; Caffrey and Day 1986).

DIP concentrations in the bay were not correlated to seasonal changes in precipitation and river discharge. This observation was again consistent with that of the previous study conducted by GCRL (1978). However, DIP concentrations were correlated positively to wind speed, salinity, and gage height. The primary influencing factor for observed temporal variability in DIP was the retention of higher salinity tidal waters introduced from the Mississippi Sound during low discharge periods. Although increased wind speed was mostly associated with increased rainfall and corresponding increases in river flow, which delivered nutrients to the bay, high winds also likely allowed for lateral mixing and circulation of the phosphate rich Bayou Portage and tidal waters within the bay. Increases in salinity occurred during periods of low river flow (e.g. Fall season). The increased DIP concentrations during periods of higher salinity in the bay is an indication that the DIP input from the rivers was not significant compared to the input from Bayou Portage and especially, the Mississippi Sound. The Mississippi Sound and Bayou Portage were thus identified as major sources of DIP to the bay during 2003-04. Similar observations were made in the Atchafalaya River delta estuarine complex, where significantly higher phosphate concentrations were found in the estuarine regions compared to the river during the fall season (Lane et al. 2002). Also, based on the circulation model applied to the Bay of St. Louis in their study, Blain and Veeramony (2002) expected tidal forcing to be dominant during the low river flow conditions. It is also likely that the average Wolf River flow (30 m^3s^{-1}) may allow only gradual dispersion of particles into the offshore waters, while retaining most of the waters within the bay (Blain and Veeramony 2002). Thus, the DIP concentrations were found to be high during the periods of low or average riverflow when the phosphate rich waters from the Mississippi Sound and Bayou Portage were retained within the estuary. Similarly, the increase in precipitation and river discharge likely caused flushing of the DIP rich waters coming from Bayou Portage and the Mississippi Sound out of the bay during the Spring and Summer. Significant flushing of the bay may occur due to river discharge during the maximum river flow conditions such as episodic storm events (Blain and Veeramony 2002) as well as during increased winds associated with such events. The temporal variability in nutrient concentrations in the bay that resulted from seasonal changes in river discharge is a commonly observed phenomenon in river-dominated estuaries of the Gulf of Mexico (Pennock et al. 1999).

The chlorophyll *a* concentrations in the bay also changed seasonally and were correlated to temperature and river discharge, which explained the high chlorophyll *a* concentrations during summer when DIN concentrations in the bay were high and turbidity was low, which allowed light penetration. Similar observations of increased chlorophyll *a* concentrations during summer, the period of higher nutrient and light availability, were made in other Gulf of Mexico estuaries such as the Fourleague Bay and the Atchafalaya Delta estuarine complex (Lane et al. 2002; Madden and Day 1992). Increased chlorophyll *a* concentrations in the plume regions and the Vermilion and Cote Blanche Bay regions of the Atchafalaya Delta estuarine complex also coincided with low total suspended solids concentrations during summer (Lane et al. 2002).

Turbidity in the bay increased during the month of highest wind speed and high river discharge (May of 2004). This suggested that the estuary was affected by wind mixing and also by river flow which introduced new sediments as well as caused sediment resuspension in the water column, which led to an increase in turbidity (turbidity measured in this study was a measure of suspended particles and did not include measure of dissolved material such as CDOM). This observation was consistent with the findings of a previous study in the bay (GCRL 1978). Although, the overall turbidity values did not change significantly over the year during the GCRL (1978) study, the suspended solid concentrations in the bay were found to have increased during periods of increased rainfall and river discharge (GCRL 1978).

The DO concentrations also changed seasonally with temperature and were correlated positively with salinity. The DO concentrations increased during periods of high salinity and low precipitation and river discharge. This observation was in contrast to that made in the Perdido Bay system (Livingston 2001). Livingston (2001) found that the surface DO concentrations in the Perdido Bay were related positively to the nutrient inputs to the bay, which increased during the periods of high riverflow. The DO concentrations in the Bay of St. Louis were, however, not correlated with nutrient inputs, which also increased during periods of high river discharge. The DO concentrations in the bay increased instead due to the cooler temperatures during winter, when low rainfall and river discharge were recorded. DO values in the bay were always above the required 5 mgL^{-1} threshold and were never found to be below the required concentrations, including during the events of increased rain and riverflow.

Periods of high winds, precipitation, and river discharge such as summer 2003 and spring 2004 were determined to be the periods of poor environmental quality mainly due to increased nutrient concentrations and turbidity values. This was consistent with the results of previous studies conducted in the bay (Phelps 1999). Phelps (1999) found that the environmental quality of the bay deteriorated during events of increased fresh water input associated with episodic storms in the 1997-98 study period.

Environmental Quality and Climate Variability

The four different data periods used for comparison purposes in this study represented different phases of the Southern Oscillation Index. The 1977-78 and 2003-04 were the Normal years, during which the SOI values ranged between both positive and negative numbers on the index (-10 to +10). The 1995-96 period was a La Niña phase during which the SOI values were highly positive, while the 1997-98 was an El Niño year when the SOI values were highly negative. It was interesting, however, that the SOI values during the 2003-04 sampling period were mostly negative than positive, therefore displaying El Niño like conditions. The highest total precipitation and average river flow was observed during the El Niño year, followed by the 2003-04 data period. The lowest total rainfall and average river discharge were recorded during the La Niña year. It is important to note that there was consistent rainfall (about 15 cm) during all months (except May) of that sampling period (1995-96).

The values of indicator parameters changed significantly during all the data periods following changes in precipitation and river flow during different phases of the SOI. Based on the observations from the 2003-04 study and the Phelps (1999) study, it was expected that the nutrient concentrations and turbidity would be highest during the periods of highest rain and river flow, such as during the El Niño sampling period (1997-98). Highest mean nutrient concentrations were, however, observed during the La Niña period (1995-96). This was not consistent with the observations from the 2003-04 sampling period, when increased nutrient concentrations were associated with periods of increased river discharge. This could have been due to the high intensity of the fresh water inflow during the El Niño year that caused flushing of all the river-borne nutrients out of the bay. Pennock et al. (1999) suggest that the low residence times during high fresh-water input along with the shallow nature of the Gulf of Mexico estuaries result in flushing of nutrients through these systems out into the near-coastal waters. Therefore, accumulation of nutrient rich waters is not generally observed during periods of higher river discharge. Whereas, during the La Niña sampling periods, although all of the land-based discharge flowed into the bay during the rainy months, the intensity of the flow was not high enough to cause flushing of the nutrients out of the bay. This allowed for higher residence times, and therefore, increased nutrient concentrations were observed during that period.

Nutrient concentrations in the bay were expected to increase over the data periods from 1977 to 2004 as human population and urban and agricultural land use increased with time. This was based on the observation that the nutrient concentrations in the areas that received runoff from the subwatersheds with higher urban and agricultural use were higher than those areas that received runoff from the less developed subwatersheds. The nutrient concentrations in the bay, however, were not found to increase over the years with increases in urban and agricultural use and human populations as was expected based on the earlier observation and other studies (Dauer et al. 2000; Interlandi et al. 2003). The lowest DIN concentrations were, in fact, measured during the latest (2003-04) sampling period. This could have been in part, in addition to the daily tidal flushing, due to the restorative actions taken towards reducing pollution in the watershed (including improved sewage treatment or other actions, not evaluated in this study) and also due to high precipitation and river flow recorded that year. During the 2003-04 sampling period, rainfall and river discharge conditions were similar to the El Niño conditions. Therefore, nutrient concentrations observed in the bay were not high since most of the river-borne nutrients were discharged (flushed out) into the Mississippi Sound similar to what was observed during the 1997-98 (El Niño year) data period. This similarity in conditions between the 1997-98 and 2003-04 data periods was also observed in the N: P ratios calculated for the bay. The N:P ratios during 2003-2004, a normal year based on the SOI, were less than the Redfield ratio and were comparable to the low N:P ratios observed during the El Niño years (Redalje et al.`2004).

Mean chlorophyll *a* concentrations were highest during 1977-78, during which the nutrient concentrations and turbidity were low. It is likely that the high values were due to discrepancies in methods used for chlorophyll estimation (spectrophotometric vs fluorometric detection) in the 1977-78 study as compared to studies from the other data periods (1995-96, 1997-98, and 2003-04). Although, nutrient concentrations were highest during the La Niña year, chlorophyll concentrations were found to be low. This could have been due to low PAR resulting from increased cloud cover and/or high turbidity during the rainy months that formed most of that sampling period. However, both PAR and turbidity data were not available for this period to verify or confirm the explanation.

Highest mean turbidity in the bay was observed during the El Niño year (1997-98), which was the period of highest river flow. This was consistent with the observed pattern of increased turbidity during events of episodic storms and increased winds, precipitation and river flow as seen in this research (2003-04) and in the previous studies by Phelps (1999) and the GCRL (1978).

Dissolved oxygen concentrations in the bay did not change significantly throughout all data periods. These results confirmed the earlier observations that the bay was not affected by low-oxygen conditions generally. The shallow and well-mixed (due to wind) nature of this estuary, in addition to the daily tidal flushing, allows the estuarine waters to remain well oxygenated.

Similar to the 2003-04 sampling period, nutrient concentrations and turbidity were the only indicator parameters that changed significantly to influence the environmental quality of the bay during all of the previous study periods (1977-78, 95-96, and 1997-98). Within the restrictions of available mean data for all data periods, environmental quality in the bay appeared to be "Poor" during 1995-96, the La Niña period due to high concentrations of nutrients, was marginally "Acceptable" in 1997-98, the El Niño period due to lower nutrient concentrations but high turbidity, was mostly "Acceptable" during 1977-78, and was "Good" during 2003-04 sampling period. These results are supported by other studies conducted in similar environments where, significant changes in coastal water quality were seen during different climate phases (Lipp et al. 2001. a; 2001. b). Lipp et al. (2001. a; 2001. b) used bacterial and enteroviral indicators and found variability in water quality in relation to the ENSO variability in two separate studies conducted in the Charlotte Harbor and the Tampa Bay estuaries in Florida. There was a significant increase in fecal pollution levels during the El Niño winter and fall periods and a significant decrease during strong La Niña winter and fall periods in relation to the normal phase conditions (Lipp et al. 2001. b). The changes in water quality parameters during different ENSO phases were correlated significantly to changes in precipitation and the corresponding streamflow (Lipp et al. 2001. a; 2001. b; Schmidt 2004). Similar correlations between environmental quality indicators and changes in weather parameters such as rainfall and riverflow, due to changes in SOI values, were also observed in this study. DIN concentrations and turbidity were correlated significantly to river discharge during all data periods. The environmental quality of the bay, similar to the Florida estuaries, was compromised during the El Niño phase due to increased levels in turbidity. However, the environmental quality in the Bay of St. Louis deteriorated due to nutrients during the La Niña phase and not the El Niño period as observed in the Tampa Bay and Charlotte Harbor estuaries (Lipp et al. 2001. a; 2001. b). This difference in the influence of the two ENSO phases on the environmental quality between the estuaries may be due to the differences in the catchment areas as well as flushing times of the estuaries. As mentioned before, the smaller area and shallower depths of the Bay of St. Louis, in addition to the daily tidal influence, led to the flushing of river borne nutrients out of the bay during the high precipitation El Niño periods. Thus, although shifting climate patterns influenced this estuary in terms of increased winds, precipitation, and river flow as seen in other systems, the impacts of the different climate patterns on the environmental quality were not always similar and did not always occur during the same phases as those observed in other estuaries (Lipp et al. 2001. a; 2001. b; Schmidt 2004). This indicates that the responses of individual systems need to be studied independently and the environmental quality must be monitored accordingly.

Environmental Quality and Changes in LULC

Differences in urban and agricultural use in the subwatersheds affected the input of nutrients into the different areas of the bay as mentioned above. Increased nutrients in the western areas of the bay may be associated with increased urban and agricultural land use in the respective subwatersheds. The differences in nutrient concentrations in different regions of the bay could also have been due to the differences in wastewater disposal systems between the Hancock (western bay) and Harrison counties (eastern bay). Several homes continued to be unsewered in the Hancock County (GMPO 2001). An increase in nutrients and fecal coliform concentrations in the surrounding bayous and therefore the Bay of St. Louis may be caused by failed septic tanks, percolation and runoff (GMPO 2001). Similarly, sewer collection systems in Hancock County are also known to have inflow and infiltration problems that may again cause an increase in the concentrations of nutrients, fecal coliforms, and other anthropogenic pollutants (GMPO 2001).

Considerable changes in LULC were observed over the years spanning all study periods. Urban and agricultural use in the Bay of St. Louis watershed increased between 1970 and 2000 (USACE 2003). The total human population in the Bay of St. Louis watershed also increased (by 64%) during those thirty years (U.S. Census Bureau 2006). Increases in human population and urban land use over time are known to have negative impacts on the water quality of rivers and estuarine systems (Dauer et al. 2000; D'Elia et al. 2003; Interlandi et al. 2003; Weller et al. 2003). Dauer et al. (2000) have shown in their study that the water quality, sediment quality, and the condition of benthic communities were affected by anthropogenic changes in urban and agricultural uses in the Chesapeake Bay watershed. The benthic biotic integrity was correlated negatively with human population density and nutrient loadings, while sediment contaminants and low dissolved oxygen events were positively correlated with both population density and urban land use (Dauer et al. 2000). Similarly, the total nitrogen concentrations in the water column were correlated positively with agricultural land use in the watershed (Dauer et al. 2000). Based on the aforementioned results, it was expected that the increase in urban and agricultural use over the years in the Bay of St. Louis watershed would lead to an increase in the input of nutrient concentrations and sediments to the bay. However, such a trend was not observed in the data. This may be due to several factors such as the differences in population density, watershed area, land use, especially urban and agricultural use in the watersheds, the bathymetry and hydrography, and the flushing times of the Chesapeake Bay and the Bay of St. Louis systems. Increased accumulation of pollutants over longer periods of time is not likely in this small, shallow, vertically-mixed estuary with daily tidal flushing and relatively less developed watersheds. Also, certain restorative programs and management actions taken throughout the watershed (such as the replacement of septic tanks with sewer systems) may have led to positive changes in the environmental quality downstream over the years. However, evaluations of such specific changes were not investigated in this study.

The influence of climate variability (such as the ENSO) on this ecosystem could be another major reason for not seeing the expected trend in the relationship between changes in LULC and environmental quality over the years. The observed changing intensities of precipitation and river flow due to changes in climate patterns altered the delivery of nutrients and sediments to the bay over time. Increased nutrient loads from the rivers could be carried out into the offshore waters during events of severe storms or El Niño like conditions, thus maintaining low nutrient concentrations in the bay. Therefore, alterations to land use and land cover in the watershed, although known to deteriorate the environmental quality, may not affect the bay over the time scales on which climate shifts occur. Thus, shifts in climate patterns play a major part and may either nullify or exacerbate the negative impacts of alterations in the watershed on the environmental quality of the estuary.

The negative impacts of altering LULC, however, could be reduced by effective management in the watershed. As newer developments and better decisions in regulating urban expansion and pollution are made, it is possible that fewer impacts of urbanization and population growth will be seen on ecosystems. However, natural variability in climate could continue to bring about significant variability in the responses of these ecosystems. It is therefore imperative to consider natural changes in climate in addition to anthropogenic factors while addressing long-term regulation and management concerns for this estuary.

Environmental Quality Index and Report Card

Water quality index or report card based evaluation programs have been used by state and federal agencies to report the water quality and/or environmental conditions of local coastal ecosystems (USEPA 1999; 2001; 2005). The development of an index that included reference values suitable for near pristine subtropical environments allowed for an appropriate and detailed evaluation of the environmental quality of the Bay of St. Louis. Implementation of such a spatially explicit index along with the report card was an effective tool that provided both criteria and threshold values to classify impaired areas easily. Such report cards can be generated on an annual, seasonal or monthly basis for regular and timely assessment of the environmental quality of the bay. Also, since the index parameters and reference values can be modified based on the changing climate and land use conditions, it allows for adaptive and flexible approaches for better and efficient management. The GIS-based evaluation system also allowed for better translation and communication of recorded data to managers, scientists and stakeholders. Such a management tool, although it has been applied in the past to many systems such as the Moreton Bay in Australia and the Chesapeake Bay, as well as the EPA-National

Coastal Condition assessment areas, it has never before been developed for or applied to the Bay of St. Louis or other such small subtropical systems.

Applying the EQI and developing the Report Card for the Bay of St. Louis has provided information for better assessment of problem areas. Areas of significance, those which will have to be monitored closely in the future included the areas close to point sources, especially near the mouth of Jourdan River and along the western region for high (exceeding target values) DIN input as well as the area extending from near the mouth of Bayou Portage out to the sound for increased (exceeding target values) DIP levels. It was also established that frequent monitoring/assessments will be required during episodic events of increased river discharge that lead to increased flux of nutrients and pollutants in the bay occurring on a seasonal scale such as during the Spring and Summer. Adaptive or flexible monitoring for different parameters/pollutants during diverse conditions will be essential to adjust to the different phases of changing climate patterns. For example, increased turbidity was found to be one of the major causes of poor environmental quality during the El Niño phase, whereas increased nutrient inputs were the concern during the La Niña phase. Similarly, changing landscapes and landuse patterns such as an increase in urban and agricultural use accompanied by a decrease in forest cover will lead to changing fluxes of nutrients, sediments and pollutants. Such variations in pollutant levels will require flexible monitoring and adaptive assessment and management programs. Implementation of a report card based tool developed in this study can allow for such effective management.

GIS Mapping

The GIS-based seasonal maps developed for the bay were useful in presenting the variability in the environmental quality of the bay over space and time. The method of spatial evaluation of environmental quality is also a useful tool for summarizing and communicating the status of an estuary in relation to its management objectives (Pantus and Dennison 2005). The evaluation card developed for this study was based on transforming the measured and predicted values into single number scores (or ranks) depending upon whether or not these values met the specified objectives. Pantus and Dennison (2005) have shown that this method allows for comparisons of different parameters on a single standard scale of compliance with the objectives. Application of such indices into a report card integrated with the GIS-based maps is becoming a commonly used tool for monitoring and managing estuarine ecosystems effectively (Integration and Application Network 2003; Abal et al. 2001; Jones et al. 2004). A pilot study in the Chesapeake Bay region used similar tools to compare the ecosystem health of the Patuxent and the Choptank rivers using indicators such as DO, secchi depth, chlorophyll *a*, total nitrogen and phosphorus, and δ^{15}nitrogen (Jones et al. 2004). The results of the study by Jones et al. (2004) were presented in terms of spatially explicit report card to provide a timely feedback on the health of the system and allow concentration of the management and research efforts on specific target areas. Similarly, the management approach including the Ecosystem Health Index and an environmental report card developed for the Moreton Bay in Australia led to significant improvements in ecosystem health as well as reduced expenses in water quality management (Abal et al. 2001). GIS-based maps of environmental quality for different seasons developed in this study facilitated instantaneous interpretation of the health status of the bay on both spatial and temporal scales. Such spatially and temporally explicit indices and report cards can provide unequivocal translation of scientifically rigorous data, thus allowing better interpretation of results for managers and stakeholders (Jones et al. 2004).

Implications for Evolving Coastal Management Policies

Defining "Impaired"

The Bay of St. Louis has been listed as an impaired water body for its designated uses of shellfish harvesting and contact recreation for several years (MDEQ 2004; 2005; 2006). This classification is mainly based on the detected presence of coliform and fecal coliform bacteria as well as due to its proximity to a waste water source. Certain bayous such as the Bayou La Croix, Mallini Bayou, Cutoff Bayou, and Rotten Bayou that empty into the bay or into the Jourdan River, that enters the bay, have also been listed as impaired due to nutrient and organic enrichment and low concentrations of dissolved oxygen (MDEQ 2004; 2005; 2006). The indicator parameters used by the MDEQ to identify the environmental quality of the bay have been coliform and fecal coliform bacteria, nutrients, and DO concentrations. The indicator parameters used in this study included DIN, DIP, chlorophyll *a*, and DO concentrations as well as turbidity. These parameters were based on the specific management objectives identified for the bay in this study instead of following the standard regulatory parameters used by state agencies listed above. Based on the selected indicators the environmental quality of this estuary was not found to be impaired. This contradiction in the results between the MDEQ studies and this study are due to the differences in the parameters used as indicators of environmental quality. The indicators and the reference values identified in this study can be used as additional tools along with the commonly used regulatory parameters to manage this estuary effectively. Significant information about the environmental quality of the bay was obtained based on the indicator parameters used in this study. It was found that the nutrient concentrations and turbidity played a major role in altering the environmental quality of the system. Whereas, changes in chlorophyll *a* and dissolved oxygen concentrations did not affect the environmental quality significantly since the concentrations of these parameters always met or exceeded the management objectives.

Since the use of different indicators (nutrients, chlorophyll, turbidity, and DO vs. fecal coliform) led to different conclusions about the environmental quality of the bay, it is important to classify the indicators under different categories, such as water quality, sediment quality or biotic integrity based on their specific purposes. It will also be beneficial to use a comprehensive suite of indicator parameters (biotic and abiotic) representing each of these categories, to get a better idea of the status of the overall environmental health of the estuary. For water and sediment quality assessments, anthropogenic pollutants such as dioxins, heavy metals, hormones or other such chemically derived compounds can be used in addition to the primary indicators such as nutrients, chlorophyll *a*, turbidity, and DO. Inclusion of pathogen indicators such as fecal coliforms is also essential. Similarly, biotic indicators of pelagic and benthic community structures for trophic and species diversity need to be used in addition to the selected parameters for better assessment of the environmental quality of this estuary. Elston et al. (2005) had identified dioxins and heavy metals as sediment and shellfish contaminants that affected the health of the bay and led to the violation of the designated uses of the bay as well. Indicators of such toxins and chemical pollutants must be identified and monitored regularly to control sediment and shellfish toxicity in the bay. Similarly, the sources and the extent of distribution of anthropogenic pollutants can be determined by identifying and monitoring indicator parameters of urban pollution. Several studies have shown the importance of sewage mapping techniques using $\delta^{15}N$ indicators or caffeine as indicators of the impact of human activities on ecosystems (Siegener and Chen 2002; Ferreira 2005; Jones et al. 2004). Several such factors can provide additional information about the health of this estuary and developing an index based on multiple indicators such as those mentioned above can help identify the precise status of environmental quality of this estuary at any given time.

Ecosystem Health: The Primary Designated Use

Most of the state and local efforts for improving the environmental quality of the estuary are focused on maintaining the designated uses of the waterbody and are driven by the necessity to maintain the uses and benefits for human activities and impacts on human health. The premise and methodology used in this research, on the other hand, was an attempt to move in the direction of understanding the value of the overall health of the system. The study was designed to detect the behavior of the system and identify the factors that made a significant impact not only on human health but on the health of the estuary. Such monitoring and observation programs are essential to ensure the sustainability of our ecosystems so that severe events such as eutrophication, toxic algal blooms, and hypoxia are detected early on and disastrous outcomes such as complete collapse of the ecosystem can be prevented. Making that shift in our approach from the "anthropocentric" to "ecosystem-based" is imminent in coastal management. There have been glaring examples of several fisheries that have collapsed in many parts of the world following a rise in fishing efforts because the management focus was solely on stock assessments or total catch and not on ecological stability or where the exploitation of natural resources continued without attempting to understand beforehand how the environments worked resulting in collapsing ecosystems. It is important to implement the lessons learned from fisheries management into coastal management. Our effort to broaden the management focus from exclusively human health to a more holistic ecological health will require us to consider "ecosystem health" as the primary designated use for all ecosystems. And although the concept of ecosystem health largely remains ill-defined and obscure, it can best be described in terms of the response of the structural and functional components of the system to natural and anthropogenic impacts (Coates et al. 2002). Thus, the use of multiple indicators representing all aspects of an ecosystem including water quality, sediment quality, as well as biotic integrity, can allow for better assessment of an environment that is naturally varying and dynamic in nature.

Cinderella Estuaries

Water quality index or report card based evaluation programs have been used by state and federal agencies to report the water quality and/or environmental conditions of local coastal ecosystems (USEPA 1999; 2001; 2005; 2008). Although the water quality indicator parameters used by such agencies are similar to those used in this study, the reference values for most indicators, especially nutrients, are generally representative of cool temperate estuaries with large developed watersheds (Jones et al. 2004; USEPA 1999; 2001; 2005; 2008). The reference values for DIN and DIP concentrations identified for the Gulf of Mexico estuaries by the USEPA, including the subtropical and tropical systems, ranged from 0.1 to 0.5 mgL^{-1} (7 μM -36 μM) for DIN and 0.01 to 0.05

mgL^{-1} (0.3 μM -1.6 μM) for DIP concentrations (USEPA 1999; 2001; 2005; 2008). However, the nutrient concentrations averaged over all data periods (1977-2004) in the Bay of St. Louis were lower than 5 μM (2 μM) for DIN and 0.3 μM for DIP. Thus, the standard nutrient criteria developed for the Gulf of Mexico estuaries by the USEPA did not reflect the average nutrient condition of the bay in the absence of cultural impacts (USEPA 2001. a.) and could not be applied to this particular estuary. Nutrient concentrations in tropical and subtropical estuaries are expected to be lower, especially during summer, due to lower freshwater input and rapid use of dissolved nutrients by phytoplankton (USEPA 1999; 2005). Therefore, an environmental quality index based on threshold values specific to small subtropical estuaries with less developed watersheds was required for the bay. The development of an index that included reference values suitable for near pristine subtropical environments such as the Bay of St. Louis allowed for an appropriate and detailed evaluation of the environmental quality of this estuary. Further studies on the functioning and responses of such smaller subtropical systems with lesser developed watersheds are essential to develop reference values representative of these "Cinderella estuaries." Increased attention and effort based on an ecosystem point of view and integrated management plans tailored for individual systems are essential for sustaining these pristine, but insufficiently documented, environments. Current evaluation programs in the Bay of St. Louis are based on looking at shorter time scales and are focused only on one or select parameters such as fecal coliform pollution directed mainly towards protection of the oyster reefs (MDEQ 2001. b; 2003). Although, shellfish harvesting is the most economically significant use of this waterbody, the overall health of this ecosystem, in terms of its biotic integrity, structure and functioning, and resilience and sustainability must also be addressed. A scientifically driven monitoring program is necessary to establish an efficient management plan and to develop regulatory procedures for any ecosystem (D' Elia et al. 2003). The present monitoring efforts of state and federal agencies are limited in space, time, and objectives. An increase in the number of sampling stations, increased frequency of sampling, and a comprehensive suite of indicator parameters representing both biotic and abiotic components of the ecosystem are required to be included in the monitoring plans. Similarly, flexibility and adaptability in management procedures in accordance with changing conditions in the environment will be essential. This will allow for better and early detection of the wide-ranging changes in the environmental quality of the bay which can lead to timely and effective restoration of the health of this system.

Implementation of such system-specific management programs must, however, be a bottom-up approach. The best judges of the characteristic requirements for management of a particular system are most likely its local residents, scientists and lawmakers. An effort where the local scientists, regulatory agencies, and community come together to develop monitoring plans, maintain essential data and records over time, and implement improvised, evolving, and adaptive management programs is critical. Such intrinsic and integrated efforts are effective means of protecting local environments. A generic framework designed for ecosystem management will be limited in its applications for individual systems that vary in geography, climate, watershed activities, and landcover. System-specific monitoring and management is required for estuaries like the Bay of St. Louis rather than following a standardized format that may not be adequate in its applicability. Behavioral changes and subtle variations specific to a system have to be studied individually in order to be able to regulate and manage that system effectively. Adaptive management plans need to be developed to accommodate variability in these dynamic systems with respect to growing population and related changes in land use and landcover in the watershed. A system that is pristine in the present may not continue to be so in the future with increasing urbanization and population demands. A gradually adaptive and evolving management plan with predictive facilities will be required in order to avoid sudden and drastic changes or complete collapse of the system. The current approach of managing our ocean and coastal ecosystems is that "we manage by crisis" (Muller-Karger 2006). This should no longer be the scenario in the field of coastal management and taking an educated preemptive approach must be the norm so as to be able to remediate situations before a "crisis" occurs.

Efforts at addressing some of the above-mentioned problems are slowly gaining ground on a national scale. Bricker et al. (2007) suggest that estuaries with similar responses, sensitivities, or functional characteristics must be assessed as a group for effective management of all systems. Development of standard monitoring protocol by for nationally comparable results is also recommended (Bricker et al. 2007). It is crucial that such recommendations are followed and implemented readily for effective management of our coastal systems.

Implications for Policy: "In sync" with National and International Concerns

The major development in ocean and coastal management on the international scene was the drafting of an action-plan, Agenda 21 at the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro in 1992. In Chapter 17 of Agenda 21, a comprehensive framework of the central concepts of sustainable and integrated management of the ocean and coastal areas was provided (UNCED 1992). On the other hand, the Pew Oceans Commission (2003) and the US Commission on Ocean Policy (2004) released reports to provide guidance on national policy issues regarding ocean and coastal management. Following were the recommendations emphasized largely in all of these documents regarding coastal management:

- Integrated management and sustainable development
- Ecosystem-based approach
- Watershed perspective
- Impacts of changes in climate

Impacts of variability in climate.

It was evident from this study that understanding the long-term dynamics and responses of the bay was an essential part of assessing the environmental health of the bay. It is, therefore, imperative to monitor not just the seasonal variations but also the inter-annual or decadal changes in the environmental quality and develop a management plan based on such observations. Nutrient and pollutant loads must be controlled differently during periods of differing intensities of river flow and therefore during different phases of climate variability. For example, nutrient and pollutant loads should be reduced during conditions like the La Niña phase of ENSO, when nutrient concentrations were high within the estuary due to less flushing of the bay. Whereas, turbidity can be the major cause of poor environmental quality in the bay during periods of increased river discharge such as the El Niño phase. Also climate change effects in the Gulf of Mexico region are predicted to be seen in terms of higher than average rainfall

(Scavia et al.2002) and changes in frequency and severity of hurricanes and winter storms (NAST 2001).

Policy implication: Management programs implemented in the Bay of St. Louis in the future must include and account for the "impacts of variability in climate on appropriate time scales" on the environmental quality of these systems.

Watershed perspective.

It was also evident that point source loadings in different regions of the bay must be regulated differently owing to the significant spatial variability observed in nutrient concentrations. The spatial variability in nutrient concentrations occurs largely due to differences in anthropogenic activities in the subwatersheds of the Bay of St. Louis. Increased nutrient concentrations in the western part of the bay were related to the larger urban and agricultural areas in the Jourdan River watershed as compared to the mainly agricultural and less urbanized Wolf River watershed. Expansion and growth of agricultural and urban land use in the watershed may lead to a corresponding increase in nutrient loads to the bay. Since landuse decisions affect the environmental quality of the coastal waters, it is crucial to consider such potential impacts in addition to the socioeconomic factors before the selection of sites and designs of new developments (U.S. Commission on Ocean Policy 2004). This entails changes in land use policies such as reforms in zoning and building laws in the watershed. Although, the bay was not found to be affected by acute conditions such as eutrophication or hypoxia/anoxia or harmful algal blooms, the conditions may change over time with growing population and urbanization. It is critical to be able to make amendments in the management plans (both watershed and coastal) accordingly in order to maintain the environmental quality of the estuary. Environmental standards and management solutions must therefore be flexible and will need to be modified over time with changing land use and population growth in the region. Similarly, indicator parameters and reference conditions may have to be reevaluated based on the developments and regulatory changes taking place in the subwatersheds. In addition, groundwater input of nutrients to the estuarine system from the watershed will have to be monitored routinely and must be included as a necessary parameter in the ecosystem management strategy.

Policy implication: Effective management of an estuary cannot be achieved by ignoring the developments in its watershed. Similarly, development activities in the watershed must not be planned without an input from coastal managers. Management options for the Bay of St. Louis have to be coordinated with the developments, policies, and regulatory actions employed in the entire watershed and vice versa.

Ecosystem-based approach.

Indicator parameters selected in this study were different than those used by the state regulatory agencies (e.g. MDMR, MDEQ), which led to different conclusions about the environmental quality of the bay. Focusing on any one aspect of an ecosystem and neglecting others can be misleading and may not provide accurate information about the health status of that environment. In the event of higher significance placed on economic and human health benefits derived from a system in addition to high costs involved in broad-scale monitoring, the importance of sustaining an entire ecosystem is often lost. It is, however, crucial to include the indicators of structural, functional, and biotic integrity of an ecosystem (such as those mentioned earlier) in the monitoring and management plans to ensure sustainable environments for the future.

Policy implication: It is imperative to develop a comprehensive ecosystem-based management program for the Bay of St. Louis, wherein all components important to the structure and functioning of the ecosystem such as the physical, chemical, and biological factors are considered.

Integrated management and sustainable development.

An adaptive, integrated, and ecosystem–based approach is crucial to developing sound coastal policies (UNEP/GPA 2006; U.S. Commission on Ocean Policy 2004; UNCED 1992). One of the major limitations of management programs of waterbodies that span several regulatory territories is the fragmented effort at managing the system. All stakeholders representing both the watershed and coastal areas, including federal, state, and local governments, local scientists, and private and public user groups, must be involved in a collective endeavor of managing the estuary. For example, comprehensive interagency coordination (including land and coastal managers) is required to control point source loadings (such as by developments of TMDLs or initiation of tertiary treatments at wastewater treatment plants) as well as minimize the effects of non point source pollution over time. Point and non-point source pollution leads to unfavorable environmental quality conditions in the bay that need to be addressed by a joint and integrated effort. Such an approach is also imperative to building sustainable ecosystems. Similarly, managing an ecosystem cannot be limited to particular space or time scales since ecosystems function on the basis of interrelated parameters and processes. Sharing and integrating the knowledge about changing climate, information about altering watersheds, and the use of management objectives-based tools (such as the one developed in this study) is essential. Equally important is the co-ordination and cooperation amongst the efforts of the regulatory agencies, academia, and other stakeholders. This approach can be a useful and effective way of monitoring, managing and communicating the health of this estuary for years to come.

Policy implication: Comprehensive and coordinated management efforts that are integrated and adaptive in nature and involve all stakeholders is the foremost requirement of any and every coastal management program. Such efforts must be encouraged in the Bay of St. Louis management initiative.

The results of this research, about an estuary as obscure on the national map as the Bay of St. Louis, reiterate the importance of the recommendations made in Agenda 21 and by the US Commission on Ocean Policy. Based on the results of this study, it can be concluded that these recommendations are not just valid on a broader scale for larger waterbodies but equally relevant to the smallest of the systems. The findings of this study have proved (and added to the growing evidence) that these recommendations have universal applicability. The policy propositions made in this study are, therefore, significant not only for the Bay of St. Louis, but all other estuaries and must be considered to promote changes in the way we manage our coastal environments.

CHAPTER VI

SUMMARY AND CONCLUSIONS

Precipitation, river discharge, tidal input, and wind forcing were identified as the primary factors influencing the environmental quality of the Bay of St. Louis. The overall environmental quality of the entire bay was not "impaired" as reported by the state agency (MDEQ 2004, 2005) but marginally "Good" on the Environmental Quality Index (EQI) during the 2003-04 sampling period. Selection and use of different indicator parameters were the cause of discrepancies in the results of the two studies. Better understanding of the environmental conditions in the estuary can be achieved by combining a suite of indicator parameters representing the entire ecosystem. Based on the parameters and criteria selected for this study, the environmental quality of the bay deteriorated significantly only due to increases in nutrient concentrations and turbidity during certain periods. The chlorophyll *a* and DO concentrations in the bay were generally within acceptable ranges and close to pristine levels. Thus, while eutrophication, algal blooms, and hypoxia are found to be common occurrences in several estuaries including few Gulf of Mexico estuaries, the Bay of St. Louis was not found to be affected by such events.

Spatial variability was observed in the environmental quality of the bay. This was mainly due to differences in concentrations of nutrients at different locations in the bay. These differences were dependent on the proximity of locations to point sources of nutrients. Poorer environmental quality (based on the EQI) was observed at areas closer to the point sources of pollutants than seen in areas farther away from the point sources. River and bayou mouths, sewage and wastewater outfalls, as well as the Mississippi

Sound were recognized as major point sources of nutrients to the bay. The Mississippi Sound and Bayou Portage were identified as important point sources of DIP to the bay.

Temporal variability in the environmental quality of the entire bay was also observed during 2003-2004 due to changes in weather and physical conditions. The periods of high winds, precipitation, and river discharge were determined to be the periods of poorer environmental quality. Increases in DIN concentrations and turbidity (exceeding target values) were seen at several locations during such periods due to increased input of nutrients and sediments via river discharge and surface runoff. DIP concentrations were higher (exceeding target values) during periods of higher wind speeds and salinities such as during the Fall season. Increased onshore winds, reduced flushing, and tidal forcing are some of the plausible causes of higher input and retention of phosphate rich waters from the Mississippi Sound and Bayou Portage during the low rainfall and low river flow periods.

Shifting climate patterns such as the El Niño Southern Oscillation (ENSO) affected the local weather conditions, particularly the primary influencing factors (such as changes in amount and intensities of wind, rainfall, and streamflow), and thus also influenced the environmental quality of the estuary. Increased precipitation and river discharge were recorded during the El Niño period, while lower than average rain and river flow conditions were observed during the La Niña sampling period. The environmental quality of the bay, however, deteriorated during the La Niña phase and was rated "Poor" (1995-1996 sampling period). The environmental quality of the bay improved during the El Niño ("Acceptable" during 1997-1998) and Normal ("Acceptable" during 1977-1978 and "Good" during 2003-2004) phases of the Southern

Oscillation Index. Although increased nutrient concentrations were associated with periods of high river flow, this was not observed during the El Niño phase. This was likely due to the high intensities of rain and river discharge that may have caused flushing of the river borne nutrients out into the Mississippi Sound, thus resulting in low nutrient accumulation but higher turbidity within the bay. The intensities of rainfall and riverflow during the La Niña period, were however, not as high as during the El Niño conditions which resulted in retention of nutrients within the bay.

Spatial variability in the environmental quality of the bay was also observed in relation to differences in LULC in the subwatersheds. Nutrient concentrations in different areas of the bay varied due to the differences in land use and land cover in the respective subwatersheds that were drained into those areas of the bay. Higher nutrient concentrations were observed at locations in the bay that were close to point sources that drained /lied within subwatersheds with higher agricultural and urban land use. Changes in environmental quality due to changes in LULC over the years were, however, not observed. The environmental quality of the bay was expected to decline with increasing population and urban and agricultural development in the watershed. However, a declining trend in the environmental quality over the years was not observed and was likely due to a combination of factors such as restorative management actions in the watershed and flushing rates of pollutants in the estuary.

Although the environmental quality index (EQI) was developed using only a few indicator parameters, it was more functional in terms of its application to this specific estuary than the USEPA indices developed for the Gulf of Mexico estuaries (USEPA 2005, 2008). The EQI and the target values developed for this estuary reflected the nutrient concentrations of small, shallow subtropical estuaries with less developed watersheds instead of those of the cool temperate estuaries with highly developed and urbanized watersheds. The GIS-based maps and the environmental quality report card developed in this study provided an easy demonstration and simplified communication of the spatial and seasonal variability in the environmental quality of the bay. The spatial maps indicated that the nutrient concentrations will have to be monitored at the river mouths, especially the Jourdan River for nitrogen nutrients and near Bayou Portage for high DIP inputs. Similarly, due to seasonal variability in environmental quality, it will be necessary to monitor the loads of nitrogen nutrients and sediments during periods of high rainfall and riverflow. Such maps along with the EQI-based report card developed in this study are important and essential tools of better communication and effective management of coastal systems such as the Bay of St. Louis.

The findings of this research can be incorporated to develop a management program for the Bay of St. Louis that is suitable for this estuary and can be implemented effectively. Policies that dictate current management practices need to be changed to accommodate the essential forcing factors. Inclusion of the following aspects in their monitoring and management plans is highly recommended to the managers of the Bay of St. Louis:

- Selection and use of a comprehensive suite of indicator parameters (and their target values) that represent structural, functional, and biotic integrity of the entire ecosystem
- Effects of daily or seasonal changes in the primary influencing factors (wind, tidal forcing, rainfall, and river flow) identified in this study

- Effects of short and long-term (interannual to interdecadal) changes in the primary influencing factors occurring due to natural shifts in climate such as the ENSO studied in this research
- Effects of changing LULC and anthropogenic activities and developments in the watershed
- Integrated coordination between land and coastal managers, and co-operative efforts within and between agencies, scientists, and all stakeholders
- Effective communication of information using tools such as the EQI and the GIS-based Report card developed in this study

Based on the results of this study, there are several important factors that should be considered for designing a management program for this estuary and similar ecosystems. An adaptive approach which can allow changes or incorporation of additional indicator parameters as well as provide flexibility in implementation of regulations is required. Different areas in the bay may need to be monitored for pollutant loads differently and several anthropogenic activities in the respective subwatersheds may have to be regulated to control these inputs. As mentioned above, areas close to point sources, such as the river mouths, sewer outfalls, Bayou Portage will have to be monitored closely for increased pollutant inputs. Also, as the Jourdan and Wolf River watersheds continue to grow in human population and urban and agricultural use, the pollutant loads in the bay will have to be regulated (e.g. by developing TMDLs for identified pollutants or beginning tertiary treatment at wastewater plants) in accordance with the development in those watersheds. An integrated approach, which includes coordinated efforts between land and coastal managers, will be required to achieve better environmental quality of the estuary. Involvement of coastal managers in planning and regulating urban development in the watershed is necessary for sustaining the health of coastal systems.

Since events of poorer environmental quality were observed during periods of increased precipitation and river discharge, the Bay of St. Louis and activities in the watershed can be monitored and managed (such as regulation of nutrient and sediment loads using TMDLs) differently during different weather conditions or different seasons of the year. However, the changes in the intensities of wind speed, precipitation, and river discharge during different climate regimes are important factors to consider and should be taken into account. While the increase in precipitation and discharge during the spring and summer weather conditions of 2003-2004 affected the environmental quality of the bay negatively in terms of higher nutrient inputs, increases in riverflow during other years may not result in similar impact on the environmental quality of the bay. The high flow conditions, such as those observed during the El Niño phase (1997-98, the sampling period of highest total precipitation and average rainfall), did not have a negative impact on the environmental quality in terms of increased nutrient concentrations in the bay. The increases in the intensities of rain and riverflow during that year, instead, resulted in the flushing of nutrients out of the bay. Whereas, the environmental quality of the bay was compromised due to high nutrient concentrations in the bay during the rainy months of the La Niña phase (1995-96), the sampling period with the lowest total precipitation and average rainfall measured. However, the turbidity levels in the bay were high during all such periods due to the increased wind intensities associated with events of increased precipitation such as storms. Thus, the environmental quality will have to be regulated differently during the wet weather conditions depending upon the different phases of influencing climate patterns. A cautious approach must be taken while drawing long-term regulatory policies concerning pollutant loads from point and non-point sources. Shifts in local weather patterns due to natural oscillations in climate play an important role in the delivery and retention of pollutants in the bay. Also, the negative or positive effects of alterations in the watershed on the environmental quality of the bay may be superimposed or overridden by the effects of changes in global climate patterns. A comprehensive suite of indicators will have to be used to monitor environmental quality, prevent violations of the designated uses, and in the process allow for better environmental conditions. Several indicators of water quality, sediment toxicity, and biotic integrity can be monitored and added to the environmental quality index developed in this study. An environmental quality index that integrates all known influencing parameters will be useful to evaluate the overall environmental health of the bay effectively. The use of GIS-based maps and a report card of periodic changes in the environmental quality can be a simple and efficient tool to disseminate information about the conditions of the estuary to all stakeholders including the managers, scientists, and the general community.

Effective management of the Bay of St. Louis can be achieved and implemented via a five-fold approach:

1. Getting together all stake-holders especially scientists, land managers, coastal managers, and user groups
2. Continuing to study the system using sound science and incorporating the results of this research and several other scientific studies conducted in the bay

that have identified sources of impacts, changes in behavior and responses of the bay over short and long time scales

3. Developing a monitoring and management program that is customized for the system using the results of the science-based studies and that is integrated, and adaptive in nature

4. Evaluating the success of implemented programs by assessing outcomes in terms of benefits of services and the quality of the environment

5. Using the information and feedback obtained to formulate further strategies and improvise management options.

Future Work

A comprehensive index that includes more representation of influencing pollutants than those used in this study can be developed in the future. Elston et al. (2005) had identified dioxins and heavy metals as sediment and shellfish contaminants that affected the overall health of the bay. The pathogen pollutants have been a significant concern for shellfish harvesting in this estuary (MDEQ 2000). An appropriate indicator species selected for pathogens can be included in the index in the future. Understanding the relationship between the activities in the watershed and the environmental quality of the bay will require further study involving sampling upriver as well as measurements of specific indicators of anthropogenic activities and developments. Several studies have shown the importance of using caffeine as an indicator and chemical tracer of urban pollution (Siegener and Chen 2002; Ferreira 2005). This, in addition to the sewage mapping techniques using $\delta^{15}N$ (Jones et al. 2004) can help determine the sources and extent of the distribution of anthropogenic pollutants in the bay. This can help further our knowledge about the causes of decline in the environmental quality and isolate the sources for better management. Reducing non-point source inputs will require the knowledge of leaching rates from agricultural lands in the watershed, soil erosion rates from forested or mangrove areas, and measurements of inputs of nutrients and other pollutants into the subsurface flow system. Understanding that will allow better regulation of the activities (for example, control the application rates of fertilizers or limiting the development of concrete pavements/ impervious surfaces). Pathogen, chemical, and anthropogenic pollutant indicators added to the present environmental quality index can provide a better understanding of the environmental condition of this estuary. This will allow incorporating water quality, sediment quality as well as biotic integrity aspects into an index that can then truly represent the overall environmental health of the bay. An increase in the frequency of sampling within the bay and establishing additional monitoring stations both within the bay and upstream of the rivers and bayous entering the bay as well as the Mississippi Sound will be required. Simultaneous monitoring of the environmental quality of the rivers and bayous that discharge into the bay and of the Mississippi Sound will provide a better understanding and regulation of the sources of pollutant inputs into the bay. Similarly, the relationship between the variability in the environmental quality and the changes in weather parameters during different climate regimes can further be applied to predict environmental quality in the future. Such recommendations of applying the knowledge of climate forecasts combined with the information about impacts of altered watersheds to develop environmental quality forecast models have been made in the past (Lipp et al. 2001a). The ability to forecast can then allow the managers to regulate point source loadings and prepare for severe conditions beforehand. A more integrated approach including effects of changes in land use and changing climate regimes is required in estuarine systems with developing watersheds (Interlandi 2003). Such changes in our approach towards managing the health of this ecosystem can provide for a healthier, sustainable, and certainly a more productive resource for all stakeholders to use.

APPENDIX A

A STANDARD ERROR MAP, ANNUAL EQI MAP AND SEASONAL EQI VALUES FOR 2003-2004

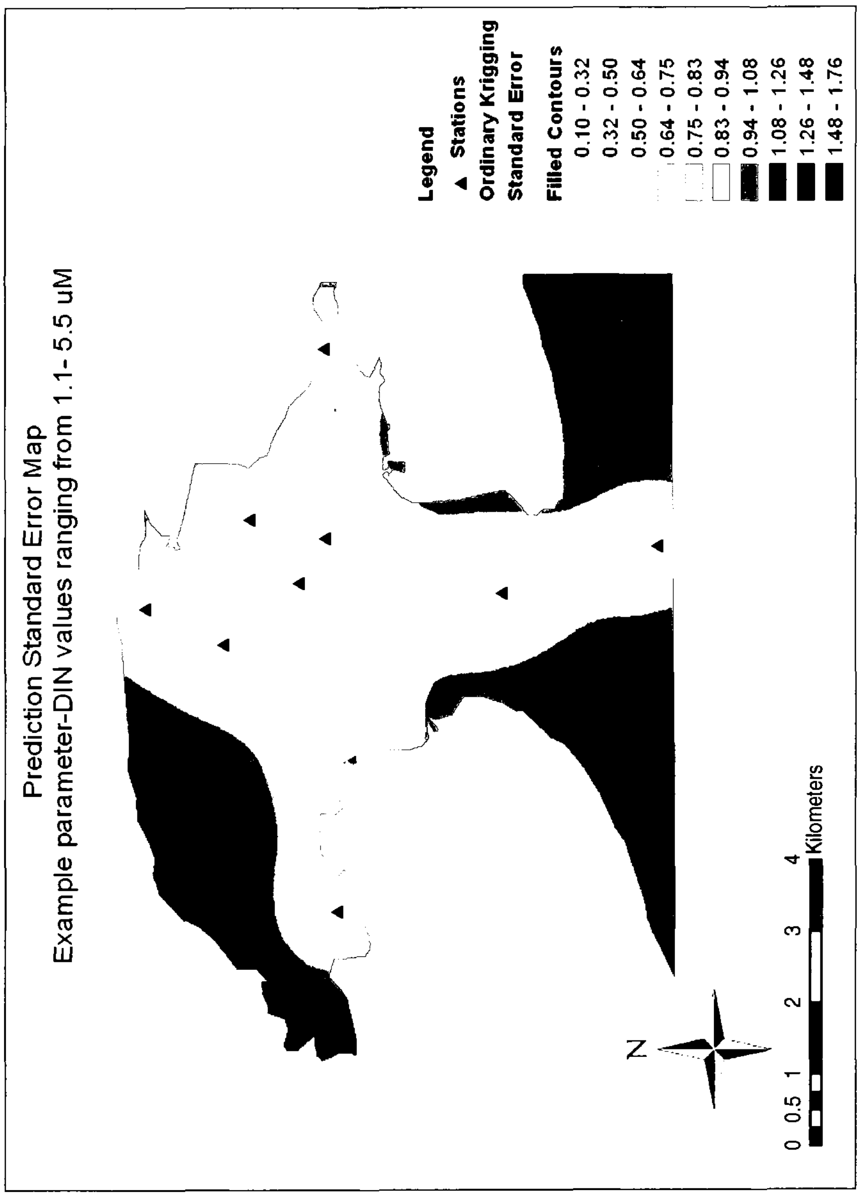

Figure A.1 An example of a Spatial Standard Error Map using DIN as the Sample Parameter. Standard error maps were created with prediction maps of indicator parameters. Standard error for any value increases in relation to the distance of a point from the sampling location. Since same stations were occupied throughout the study period, the standard error maps are similar for all parameters measured over the entire sampling period.

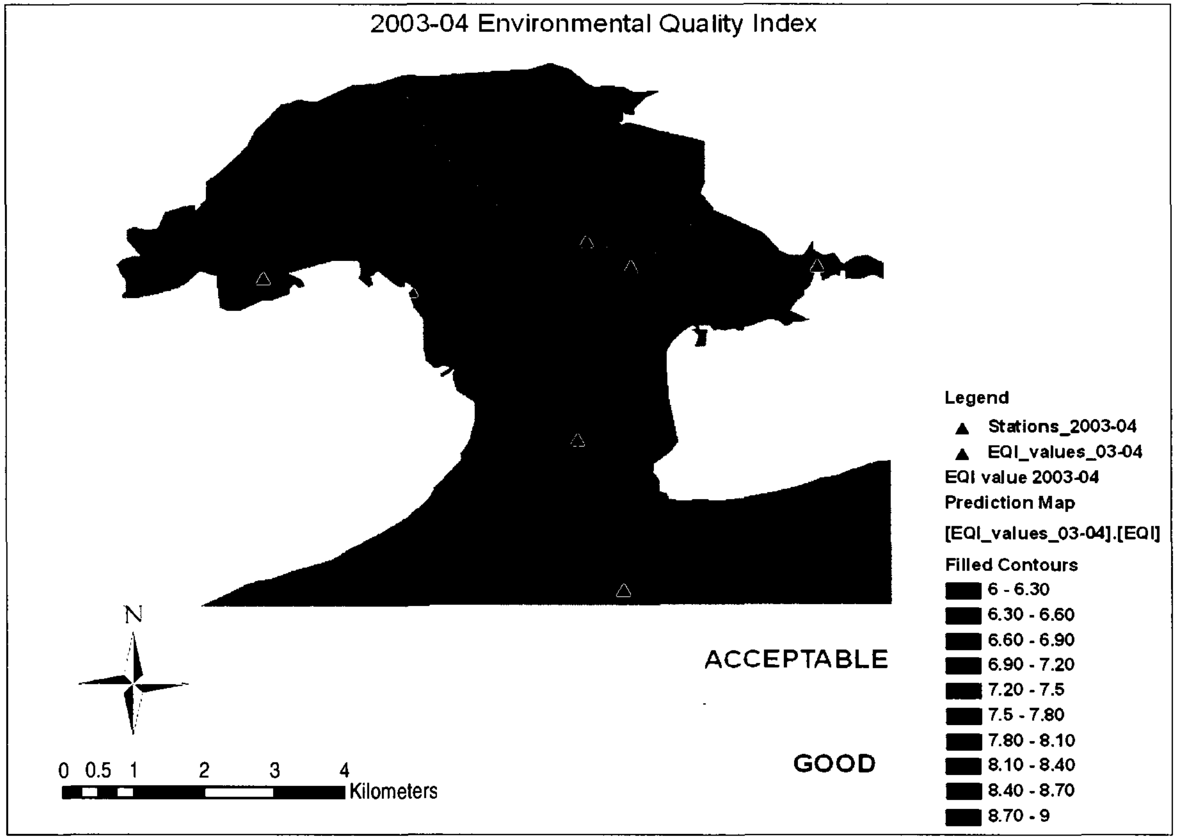

Figure A.2 Spatial Distribution of Calculated and Spatially Interpolated Environmental Quality Index (EQI) Values for the Bay of St. Louis for the 2003-2004 Data Period. Environmental quality of the bay was considered as marginally "Good" for the year. EQI value for the entire bay averaged to 7.60, thus obtaining an "A" grade. (Refer to the standard error map above)..

Table A.1 EQI Values Calculated for All Indicator Parameters for Each Sampling Station for Spring 2003. Final Average EQI for this season was 4.2 which ranked as "Poor" on the EQ Index. However, this result cannot be confirmed since DIN and chlorophyll data were not available during that season. Also, the field sampling for this study was started in the last week of April 2003, therefore, only three samplings were carried out that season.

Stations	Latitude	Longitude	EQI value-DIN	EQI value-DIP	EQI value-Chlorophyll *a*	EQI value-Turbidity	EQI value-DO	Total EQI value
1	30.3407	-89.3554	ND	2	ND	1	2	5
2	30.37218	-89.3085	ND	2	ND	0	2	4
3	30.35926	-89.314	ND	2	ND	1	2	5
4	30.35503	-89.2947	ND	2	ND	1	2	5
5	30.34283	-89.2681	ND	1	ND	1	2	4
6	30.34686	-89.3046	ND	1	ND	1	2	4
6a	30.33847	-89.3322	ND	1	ND	0	2	3
7	30.34257	-89.2976	ND	1	ND	1	2	4
8	30.31344	-89.3061	ND	1	ND	1	2	4
9	30.28795	-89.2987	ND	1	ND	1	2	4
Final Average EQI								**4.20**

Table A.2 EQI Values Calculated for All Indicator Parameters for Each Sampling Station for Summer 2003. Final Average EQI for this season was 7.3 which ranked as "Acceptable" on the EQ Index.

Stations	Latitude	Longitude	EQI value-DIN	EQI value-DIP	EQI value-Chlorophyll *a*	EQI value-Turbidity	EQI value-DO	Total EQI value
1	30.3407	-89.3554	0	1	2	1	2	6
2	30.37218	-89.3085	2	2	2	1	2	9
3	30.35926	-89.314	2	2	2	1	2	9
4	30.35503	-89.2947	1	2	2	1	2	8
5	30.34283	-89.2681	2	0	2	1	2	7
6	30.34686	-89.3046	2	1	2	1	2	8
6a	30.33847	-89.3322	1	1	0	0	2	4
7	30.34257	-89.2976	2	1	2	1	2	8
8	30.31344	-89.3061	2	1	2	1	2	8
9	30.28795	-89.2987	1	0	1	1	2	6
Final Average EQI								**7.3**

Table A.3 EQI Values Calculated for All Indicator Parameters for Each Sampling Station for Fall 2003. Final Average EQI for this season was 8.1 which ranked as "Good" on the EQ Index.

Stations	Latitude	Longitude	EQI value-DIN	EQI value-DIP	EQI value-Chlorophyll *a*	EQI value-Turbidity	EQI value-DO	Total EQI value
1	30.3407	-89.3554	1	1	2	1	2	7
2	30.37218	-89.3085	2	1	2	1	2	8
3	30.35926	-89.314	2	1	2	2	2	9
4	30.35503	-89.2947	2	1	2	2	2	9
5	30.34283	-89.2681	1	0	2	2	2	7
6	30.34686	-89.3046	2	1	2	2	2	9
6a	30.33847	-89.3322	2	1	2	1	2	8
7	30.34257	-89.2976	2	0	2	2	2	8
8	30.31344	-89.3061	2	0	2	2	2	8
9	30.28795	-89.2987	2	0	2	2	2	8
Final Average EQI								**8.1**

Table A.4 EQI Values Calculated for All Indicator Parameters for Each Sampling Station for Winter 2003. Final Average EQI for this season was 8.9 which ranked as "Good" on the EQ Index.

Stations	Latitude	Longitude	EQI value-DIN	EQI value-DIP	EQI value-Chlorophyll *a*	EQI value-Turbidity	EQI value-DO	Total EQI value
1	30.3407	-89.3554	1	2	2	2	2	9
2	30.37218	-89.3085	2	2	2	2	2	10
3	30.35926	-89.314	2	2	2	2	2	10
4	30.35503	-89.2947	1	2	2	2	2	9
5	30.34283	-89.2681	1	0	2	2	2	7
6	30.34686	-89.3046	2	1	2	2	2	9
6a	30.33847	-89.3322	2	2	2	0	2	8
7	30.34257	-89.2976	2	1	2	2	2	9
8	30.31344	-89.3061	2	1	2	2	2	9
9	30.28795	-89.2987	2	1	2	2	2	9
Final Average EQI								**8.9**

Table A.5 EQI Values Calculated for All Indicator Parameters for Each Sampling Station for Spring 2004. Final Average EQI for this season was 6.1 which ranked as "Acceptable" on the EQ Index.

Stations	Latitude	Longitude	EQI value-DIN	EQI value-DIP	EQI value-Chlorophyll *a*	EQI value-Turbidity	EQI value-DO	Total EQI value
1	30.3407	-89.3554	0	1	2	0	2	9
2	30.37218	-89.3085	1	2	2	1	2	10
3	30.35926	-89.314	1	2	2	0	2	10
4	30.35503	-89.2947	1	2	2	0	2	9
5	30.34283	-89.2681	1	0	2	1	2	7
6	30.34686	-89.3046	1	1	2	0	2	9
6a	30.33847	-89.3322	0	0	2	0	2	8
7	30.34257	-89.2976	1	1	2	0	2	9
8	30.31344	-89.3061	1	1	2	0	2	9
9	30.28795	-89.2987		1	2	0	2	9
Final Average EQI								**6.1**

APPENDIX B

TIDE. GAGE HEIGHT. AND WIND DIRECTION DATA FROM 2003-2004

EEQ Data-
(April 2003-May 2004)

Sampling Date	Sampling Station	Tide	Gage Height (m)	Wind Direction (Mode)
4/30/03	1	Outgoing	1.27	No Data
4/30/03	2	Outgoing	1.25	
4/30/03	3	Outgoing	1.24	
4/30/03	4	Outgoing	1.23	
4/30/03	5	Outgoing	1.20	
4/30/03	6	Outgoing	1.21	
4/30/03	6a	Outgoing	1.28	
4/30/03	7	Outgoing	1.19	
4/30/03	8	Outgoing	1.19	
4/30/03	9	Outgoing	1.19	
5/15/03	1	Incoming	No Data	No Data
5/15/03	2	Incoming		
5/15/03	3	Incoming		
5/15/03	4	Incoming		
5/15/03	5	Incoming		
5/15/03	6	Incoming		
5/15/03	6a	Incoming		
5/15/03	7	Incoming		
5/15/03	8	Incoming		
5/15/03	9	Incoming		
5/29/2003	1	Outgoing	No Data	No Data
5/29/2003	2	Outgoing		
5/29/2003	3	Outgoing		

5/29/2003	4	Outgoing		
5/29/2003	5	Outgoing		
5/29/2003	6	Outgoing		
5/29/2003	6a	Outgoing		
5/29/2003	7	Outgoing		
5/29/2003	8	Outgoing		
5/29/2003	9	Outgoing		
6/3/2003	1	Incoming	No Data	No Data
6/3/2003	2	Incoming		
6/3/2003	3	Incoming		
6/3/2003	4	Incoming		
6/3/2003	5	Incoming		
6/3/2003	6	Incoming		
6/3/2003	6a	Incoming		
6/3/2003	7	Incoming		
6/3/2003	8	Incoming		
6/3/2003	9	Incoming		
6/24/2003	1	Outgoing	0.57	No Data
6/24/2003	2	Outgoing	0.58	
6/24/2003	3	Outgoing	0.59	
6/24/2003	4	Outgoing	0.59	
6/24/2003	5	Outgoing	0.60	
6/24/2003	6	Outgoing	0.60	
6/24/2003	6a	Outgoing	0.56	
6/24/2003	7	Outgoing	0.60	
6/24/2003	8	Outgoing	0.60	
6/24/2003	9	Outgoing	0.59	
7/8/2003	1	Outgoing	0.61	No Data
7/8/2003	2	Outgoing	0.62	

7/8/2003	3	Outgoing	0.63	
7/8/2003	4	Outgoing	0.64	
7/8/2003	5	Outgoing	0.66	
7/8/2003	6	Outgoing	0.65	
7/8/2003	6a	Outgoing	0.60	
7/8/2003	7	Outgoing	0.68	
7/8/2003	8	Outgoing	0.69	
7/8/2003	9	Outgoing	0.70	
7/16/2003	1	Incoming	0.61	No Data
7/16/2003	2	Incoming	0.60	
7/16/2003	3	Incoming	0.59	
7/16/2003	4	Incoming	0.59	
7/16/2003	5	Incoming	0.60	
7/16/2003	6	Incoming	0.58	
7/16/2003	6a	Incoming	0.63	
7/16/2003	7	Incoming	0.57	
7/16/2003	8	Incoming	0.56	
7/16/2003	9	Incoming	0.55	
8/6/2003	1	Outgoing	0.41	South
8/6/2003	2	Outgoing	0.43	South
8/6/2003	3	Outgoing	0.44	South
8/6/2003	4	Outgoing	0.45	South
8/6/2003	5	Outgoing	0.48	South
8/6/2003	6	Outgoing	0.46	South
8/6/2003	6a	Outgoing	0.39	South
8/6/2003	7	Outgoing	0.49	South
8/6/2003	8	Outgoing	0.50	South
8/6/2003	9	Outgoing	0.52	South
8/12/2003	1	Incoming	0.72	South

8/12/2003	2	Incoming	0.71	South
8/12/2003	3	Incoming	0.71	South
8/12/2003	4	Incoming	0.69	South
8/12/2003	5	Incoming	0.67	South
8/12/2003	6	Incoming	0.69	South
8/12/2003	6a	Incoming	0.73	South
8/12/2003	7	Incoming	0.66	South
8/12/2003	8	Incoming	0.64	South
8/12/2003	9	Incoming	0.58	South
9/4/2003	1	Outgoing	0.75	North
9/4/2003	2	Outgoing	0.77	North
9/4/2003	3	Outgoing	0.78	North
9/4/2003	4	Outgoing	0.80	North
9/4/2003	5	Outgoing	0.82	North
9/4/2003	6	Outgoing	0.81	North
9/4/2003	6a	Outgoing	0.74	North
9/4/2003	7	Outgoing	0.84	North
9/4/2003	8	Outgoing	0.85	North
9/4/2003	9	Outgoing	0.86	North
9/11/2003	1	Incoming	0.52	South-East
9/11/2003	2	Incoming	0.50	South-East
9/11/2003	3	Incoming	0.50	South-East
9/11/2003	4	Incoming	0.50	South-East
9/11/2003	5	Incoming	0.49	South-East
9/11/2003	6	Incoming	0.49	South-East
9/11/2003	6a	Incoming	0.53	South-East
9/11/2003	7	Incoming	0.49	South-East
9/11/2003	8	Incoming	0.48	South-East
9/11/2003	9	Incoming	0.49	South-East

10/2/2003	1	Incoming	0.58	North
10/2/2003	2	Incoming	0.61	North
10/2/2003	3	Incoming	0.63	North
10/2/2003	4	Incoming	0.64	North
10/2/2003	5	Incoming	0.65	North
10/2/2003	6	Incoming	0.64	North
10/2/2003	6a	Incoming	0.56	North
10/2/2003	7	Incoming	0.66	North
10/2/2003	8	Incoming	0.68	North
10/2/2003	9	Incoming	0.68	North
10/14/2003	1	Outgoing	0.30	South-East
10/14/2003	2	Outgoing	0.29	South-East
10/14/2003	3	Outgoing	0.30	South-East
10/14/2003	4	Outgoing	0.32	South-East
10/14/2003	5	Outgoing	0.34	South-East
10/14/2003	6	Outgoing	0.33	South-East
10/14/2003	6a	Outgoing	0.27	South-East
10/14/2003	7	Outgoing	0.35	South-East
10/14/2003	8	Outgoing	0.36	South-East
10/14/2003	9	Outgoing	0.37	South-East
11/18/2003	1	Outgoing	1.53	South-East
11/18/2003	2	Outgoing	1.55	South-East
11/18/2003	3	Outgoing	1.55	South-East
11/18/2003	4	Outgoing	1.56	South-East
11/18/2003	5	Outgoing	1.58	South-East
11/18/2003	6	Outgoing	1.57	South-East
11/18/2003	6a	Outgoing	1.53	South-East
11/18/2003	7	Outgoing	1.59	South-East
11/18/2003	8	Outgoing	1.60	South-East
11/18/2003	9	Outgoing	1.60	South-East

11/22/2003	1	Incoming	0.76	East
11/22/2003	2	Incoming	0.75	East
11/22/2003	3	Incoming	0.74	East
11/22/2003	4	Incoming	0.74	East
11/22/2003	5	Incoming	0.73	East
11/22/2003	6	Incoming	0.73	East
11/22/2003	6a	Incoming	0.77	East
11/22/2003	7	Incoming	0.73	East
11/22/2003	8	Incoming	0.72	East
11/22/2003	9	Incoming	0.72	East
12/2/2003	1	Outgoing	0.98	South
12/2/2003	2	Outgoing	0.98	South
12/2/2003	3	Outgoing	0.98	South
12/2/2003	4	Outgoing	0.98	South
12/2/2003	5	Outgoing	0.98	South
12/2/2003	6	Outgoing	0.98	South
12/2/2003	6a	Outgoing	0.98	South
12/2/2003	7	Outgoing	0.97	South
12/2/2003	8	Outgoing	0.97	South
12/2/2003	9	Outgoing	0.97	South
12/5/2003	1	Incoming	0.86	North
12/5/2003	2	Incoming	0.86	North
12/5/2003	3	Incoming	0.86	North
12/5/2003	4	Incoming	0.85	North
12/5/2003	5	Incoming	0.85	North
12/5/2003	6	Incoming	0.85	North
12/5/2003	6a	Incoming	0.86	North
12/5/2003	7	Incoming	0.85	North
12/5/2003	8	Incoming	0.84	North

12/5/2003	9	Incoming	0.84	North
1/23/2004	1	Outgoing	0.53	North-East
1/23/2004	2	Outgoing	0.54	North-East
1/23/2004	3	Outgoing	0.54	North-East
1/23/2004	4	Outgoing	0.55	North-East
1/23/2004	5	Outgoing	0.97	North-East
1/23/2004	6	Outgoing	0.55	North-East
1/23/2004	6a	Outgoing	0.53	North-East
1/23/2004	7	Outgoing	0.57	North-East
1/23/2004	8	Outgoing	0.58	North-East
1/23/2004	9	Outgoing	0.59	North-East
1/30/2004	1	Incoming	0.96	East
1/30/2004	2	Incoming	0.93	East
1/30/2004	3	Incoming	0.92	East
1/30/2004	4	Incoming	0.91	East
1/30/2004	5	Incoming	0.88	East
1/30/2004	6	Incoming	0.90	East
1/30/2004	6a	Incoming	0.98	East
1/30/2004	7	Incoming	0.87	East
1/30/2004	8	Incoming	0.85	East
1/30/2004	9	Incoming	0.84	East
2/4/2004	1	Incoming	0.78	East
2/4/2004	2	Incoming	0.77	East
2/4/2004	3	Incoming	0.76	East
2/4/2004	4	Incoming	0.75	East
2/4/2004	5	Incoming	0.74	East
2/4/2004	6	Incoming	0.75	East
2/4/2004	6a	Incoming	0.79	East
2/4/2004	7	Incoming	0.75	East

2/4/2004	8	Incoming	0.75	East
2/4/2004	9	Incoming	0.76	East
2/9/2004	1	Outgoing	0.76	South-East
2/9/2004	2	Outgoing	0.75	South-East
2/9/2004	3	Outgoing	0.75	South-East
2/9/2004	4	Outgoing	0.74	South-East
2/9/2004	5	Outgoing	0.73	South-East
2/9/2004	6	Outgoing	0.73	South-East
2/9/2004	6a	Outgoing	0.77	South-East
2/9/2004	7	Outgoing	0.72	South-East
2/9/2004	8	Outgoing	0.72	South-East
2/9/2004	9	Outgoing	0.72	South-East
3/5/2004	1	Incoming	0.94	South
3/5/2004	2	Incoming	0.92	South
3/5/2004	3	Incoming	0.91	South
3/5/2004	4	Incoming	0.90	South
3/5/2004	5	Incoming	0.87	South
3/5/2004	6	Incoming	0.88	South
3/5/2004	6a	Incoming	0.96	South
3/5/2004	7	Incoming	0.86	South
3/5/2004	8	Incoming	0.85	South
3/5/2004	9	Incoming	0.85	South
3/8/2004	1	Outgoing	0.65	North
3/8/2004	2	Outgoing	0.65	North
3/8/2004	3	Outgoing	0.66	North
3/8/2004	4	Outgoing	0.66	North
3/8/2004	5	Outgoing	0.67	North
3/8/2004	6	Outgoing	0.67	North
3/8/2004	6a	Outgoing	0.64	North

3/8/2004	7	Outgoing	0.68	North
3/8/2004	8	Outgoing	0.68	North
3/8/2004	9	Outgoing	0.69	North
4/2/2004	1	Outgoing	-0.27	South-East
4/2/2004	2	Outgoing	-0.30	South-East
4/2/2004	3	Outgoing	-0.31	South-East
4/2/2004	4	Outgoing	-0.32	South-East
4/2/2004	5	Outgoing	-0.34	South-East
4/2/2004	6	Outgoing	-0.33	South-East
4/2/2004	6a	Outgoing	-0.24	South-East
4/2/2004	7	Outgoing	-0.35	South-East
4/2/2004	8	Outgoing	-0.37	South-East
4/2/2004	9	Outgoing	-0.36	South-East
4/5/2004	1	Incoming	0.28	South-East
4/5/2004	2	Incoming	0.26	South-East
4/5/2004	3	Incoming	0.24	South-East
4/5/2004	4	Incoming	0.23	South-East
4/5/2004	5	Incoming	0.20	South-East
4/5/2004	6	Incoming	0.22	South-East
4/5/2004	6a	Incoming	0.29	South-East
4/5/2004	7	Incoming	0.19	South-East
4/5/2004	8	Incoming	0.18	South-East
4/5/2004	9	Incoming	0.17	South-East
5/1/2004	1	Outgoing	0.15	South-East
5/1/2004	2	Outgoing	0.15	South-East
5/1/2004	3	Outgoing	0.15	South-East
5/1/2004	4	Outgoing	0.15	South-East
5/1/2004	5	Outgoing	0.16	South-East
5/1/2004	6	Outgoing	0.15	South-East

5/1/2004	6a	Outgoing	0.15	South-East
5/1/2004	7	Outgoing	0.16	South-East
5/1/2004	8	Outgoing	0.15	South-East
5/1/2004	9	Outgoing	0.16	South-East
5/17/2004	1	Incoming	0.67	South-East
5/17/2004	2	Incoming	0.64	South-East
5/17/2004	3	Incoming	0.64	South-East
5/17/2004	4	Incoming	0.63	South-East
5/17/2004	5	Incoming	0.62	South-East
5/17/2004	6	Incoming	0.63	South-East
5/17/2004	6a	Incoming	0.67	South-East
5/17/2004	7	Incoming	0.61	South-East
5/17/2004	8	Incoming	0.61	South-East
5/17/2004	9	Incoming	0.60	South-East

APPENDIX C

SPEARMAN RANK CORRELATIONS		NO2NO3	PO4	NO2	NO3	NH4	DIN	SiO2	Chlorophyll
NO2NO3	Correlation Coefficient	1.000	-0.005	0.481	0.977	0.432	0.837	0.271	0.085
	Sig. (2-tailed)	.	0.931	0.000	0.000	0.000	0.000	0.000	0.202
	N	269	269	269	269	239	239	269	229
PO4	Correlation Coefficient	-0.005	1.000	0.006	0.004	0.140	0.019	-0.355	-0.033
	Sig. (2-tailed)	0.931	.	0.925	0.952	0.030	0.769	0.000	0.619
	N	269	270	270	270	240	240	270	230
NO2	Correlation Coefficient	0.481	0.006	1.000	0.355	0.085	0.330	0.084	0.035
	Sig. (2-tailed)	0.000	0.925	.	0.000	0.188	0.000	0.167	0.594
	N	269	270	270	270	240	240	270	230
NO3	Correlation Coefficient	0.977	0.004	0.355	1.000	0.442	0.843	0.267	0.094
	Sig. (2-tailed)	0.000	0.952	0.000	.	0.000	0.000	0.000	0.154
	N	269	270	270	270	240	240	270	230
NH4	Correlation Coefficient	0.432	0.140	0.085	0.442	1.000	0.781	0.134	0.243
	Sig. (2-tailed)	0.000	0.030	0.188	0.000	.	0.000	0.038	0.000
	N	239	240	240	240	240	240	240	230
DIN	Correlation Coefficient	0.837	0.019	0.330	0.843	0.781	1.000	0.241	0.231
	Sig. (2-tailed)	0.000	0.769	0.000	0.000	0.000	.	0.000	0.000
	N	239	240	240	240	240	240	240	230
SiO2	Correlation Coefficient	0.271	-0.355	0.084	0.267	0.134	0.241	1.000	0.311
	Sig. (2-tailed)	0.000	0.000	0.167	0.000	0.038	0.000	.	0.000
	N	269	270	270	270	240	240	270	230
Chlorophyll	Correlation Coefficient	0.085	-0.033	0.035	0.094	0.243	0.231	0.311	1.000
	Sig. (2-tailed)	0.202	0.619	0.594	0.154	0.000	0.000	0.000	.
	N	229	230	230	230	230	230	230	230

Phaeopigments	Correlation Coefficient	0.244	0.072	0.323	0.191	0.003	0.118	0.217	-0.040
	Sig. (2-tailed)	0.000	0.278	0.000	0.004	0.963	0.074	0.001	0.551
	N	229	230	230	230	230	230	230	230
Temperature (C)	Correlation Coefficient	-0.102	-0.071	-0.128	-0.071	0.246	0.167	0.439	0.426
	Sig. (2-tailed)	0.102	0.253	0.039	0.252	0.000	0.011	0.000	0.000
	N	260	260	260	260	230	230	260	220
Salinity	Correlation Coefficient	-0.367	0.334	-0.481	-0.315	-0.412	-0.520	-0.373	-0.197
	Sig. (2-tailed)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003
	N	260	260	260	260	230	230	260	220
DO %	Correlation Coefficient	-0.398	0.099	-0.343	-0.357	-0.157	-0.212	-0.243	0.178
	Sig. (2-tailed)	0.000	0.111	0.000	0.000	0.017	0.001	0.000	0.008
	N	260	260	260	260	230	230	260	220
DO mg/L	Correlation Coefficient	-0.180	0.028	-0.113	-0.171	-0.108	-0.097	-0.322	-0.014
	Sig. (2-tailed)	0.004	0.652	0.070	0.006	0.102	0.145	0.000	0.833
	N	260	260	260	260	230	230	260	220
pH	Correlation Coefficient	-0.304	0.259	-0.132	-0.284	-0.371	-0.421	-0.318	-0.290
	Sig. (2-tailed)	0.000	0.000	0.033	0.000	0.000	0.000	0.000	0.000
	N	260	260	260	260	230	230	260	220
Turbidity (NTU)	Correlation Coefficient	0.206	0.078	0.327	0.165	0.362	0.412	-0.218	0.108
	Sig. (2-tailed)	0.003	0.270	0.000	0.018	0.000	0.000	0.002	0.168
	N	204	204	204	204	175	175	204	165
Pressure (mbar)	Correlation Coefficient	0.090	-0.017	0.342	0.050	-0.033	0.052	-0.099	-0.043
	Sig. (2-tailed)	0.204	0.811	0.000	0.478	0.647	0.463	0.164	0.542
	N	199	200	200	200	200	200	200	200

Rain (mm)	Correlation Coefficient	-0.160	-0.036	-0.341	-0.123	0.074	-0.051	0.033	0.116
	Sig. (2-tailed)	0.024	0.616	0.000	0.084	0.300	0.478	0.647	0.101
	N	199	200	200	200	200	200	200	200
PAR (uE)	Correlation Coefficient	0.244	0.003	0.346	0.222	0.067	0.204	0.396	0.136
	Sig. (2-tailed)	0.001	0.962	0.000	0.002	0.348	0.004	0.000	0.055
	N	199	200	200	200	200	200	200	200
Solar Radiation (W/m^2)	Correlation Coefficient	0.277	-0.054	0.452	0.220	0.064	0.190	0.389	0.131
	Sig. (2-tailed)	0.000	0.444	0.000	0.002	0.366	0.007	0.000	0.064
	N	199	200	200	200	200	200	200	200
Wind Speed (m/s)	Correlation Coefficient	0.075	0.216	0.228	0.079	0.390	0.256	-0.124	0.148
	Sig. (2-tailed)	0.290	0.002	0.001	0.267	0.000	0.000	0.080	0.036
	N	199	200	200	200	200	200	200	200
Gust Speed (m/s)	Correlation Coefficient	0.002	0.232	0.226	0.008	0.313	0.170	-0.086	0.080
	Sig. (2-tailed)	0.976	0.001	0.001	0.911	0.000	0.016	0.225	0.261
	N	199	200	200	200	200	200	200	200
Air Temp (C)	Correlation Coefficient	0.052	0.032	-0.178	0.097	0.225	0.143	0.228	0.236
	Sig. (2-tailed)	0.462	0.651	0.012	0.173	0.001	0.044	0.001	0.001
	N	199	200	200	200	200	200	200	200
Dew Point (C)	Correlation Coefficient	-0.021	-0.046	-0.380	0.066	0.201	0.139	0.076	0.177
	Sig. (2-tailed)	0.771	0.518	0.000	0.352	0.004	0.050	0.284	0.012
	N	199	200	200	200	200	200	200	200
RH (%)	Correlation Coefficient	-0.067	0.008	-0.478	0.020	0.219	0.115	-0.346	-0.035
	Sig. (2-tailed)	0.350	0.907	0.000	0.781	0.002	0.106	0.000	0.619
	N	199	200	200	200	200	200	200	200

Gauge Height (Feet)	Correlation Coefficient	-0.342	0.195	-0.218	-0.293	-0.081	-0.214	-0.337	-0.049
	Sig. (2-tailed)	0.000	0.002	0.001	0.000	0.219	0.001	0.000	0.461
	N	239	240	240	240	230	230	240	230
Total daily pptn JR day0	Correlation Coefficient	0.066	-0.067	-0.028	0.065	0.134	0.103	0.010	0.125
	Sig. (2-tailed)	0.284	0.269	0.643	0.286	0.038	0.111	0.866	0.058
	N	269	270	270	270	240	240	270	230
Total daily pptn WR day0	Correlation Coefficient	-0.065	-0.078	-0.096	-0.065	0.044	-0.043	-0.106	-0.046
	Sig. (2-tailed)	0.286	0.201	0.116	0.286	0.499	0.507	0.083	0.483
	N	269	270	270	270	240	240	270	230
Daily avg discharge WR day0	Correlation Coefficient	0.367	-0.148	0.123	0.369	0.406	0.395	0.166	0.286
	Sig. (2-tailed)	0.000	0.015	0.044	0.000	0.000	0.000	0.006	0.000
	N	269	270	270	270	240	240	270	230

		Phaeo	Temp (C)	Salinity	DO %	DO mg/L	pH	Turbidity (NTU)	Pressure (mbar)	Rain (mm)	PAR (uE)	Solar Rad (W/m^2)
NO2NO3	Correlation Coefficient	0.244	-0.102	-0.367	-0.398	-0.180	-0.304	0.206	0.090	-0.160	0.244	0.277
	Sig. (2-tailed)	0.000	0.102	0.000	0.000	0.004	0.000	0.003	0.204	0.024	0.001	0.000
	N	229	260	260	260	260	260	204	199	199	199	199
PO4	Correlation Coefficient	0.072	-0.071	0.334	0.099	0.028	0.259	0.078	-0.017	-0.036	0.003	-0.054
	Sig. (2-tailed)	0.278	0.253	0.000	0.111	0.652	0.000	0.270	0.811	0.616	0.962	0.444
	N	230	260	260	260	260	260	204	200	200	200	200
NO2	Correlation Coefficient	0.323	-0.128	-0.481	-0.343	-0.113	-0.132	0.327	0.342	-0.341	0.346	0.452
	Sig. (2-tailed)	0.000	0.039	0.000	0.000	0.070	0.033	0.000	0.000	0.000	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
NO3	Correlation Coefficient	0.191	-0.071	-0.315	-0.357	-0.171	-0.284	0.165	0.050	-0.123	0.222	0.220
	Sig. (2-tailed)	0.004	0.252	0.000	0.000	0.006	0.000	0.018	0.478	0.084	0.002	0.002
	N	230	260	260	260	260	260	204	200	200	200	200
NH4	Correlation Coefficient	0.003	0.246	-0.412	-0.157	-0.108	-0.371	0.362	-0.033	0.074	0.067	0.064
	Sig. (2-tailed)	0.963	0.000	0.000	0.017	0.102	0.000	0.000	0.647	0.300	0.348	0.366
	N	230	230	230	230	230	230	175	200	200	200	200
DIN	Correlation Coefficient	0.118	0.167	-0.520	-0.212	-0.097	-0.421	0.412	0.052	-0.051	0.204	0.190
	Sig. (2-tailed)	0.074	0.011	0.000	0.001	0.145	0.000	0.000	0.463	0.478	0.004	0.007
	N	230	230	230	230	230	230	175	200	200	200	200
SiO2	Correlation Coefficient	0.217	0.439	-0.373	-0.243	-0.322	-0.318	-0.218	-0.099	0.033	0.396	0.389
	Sig. (2-tailed)	0.001	0.000	0.000	0.000	0.000	0.000	0.002	0.164	0.647	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
Chlorophyll	Correlation Coefficient	-0.040	0.426	-0.197	0.178	-0.014	-0.290	0.108	-0.043	0.116	0.136	0.131
	Sig. (2-tailed)	0.551	0.000	0.003	0.008	0.833	0.000	0.168	0.542	0.101	0.055	0.064
	N	230	220	220	220	220	220	165	200	200	200	200

Phaeopigments	Correlation Coefficient	1.000	0.271	-0.339	-0.460	-0.436	-0.106	0.219	-0.206	-0.163	0.180	0.168
	Sig. (2-tailed)	.	0.000	0.000	0.000	0.000	0.116	0.005	0.003	0.021	0.011	0.018
	N	230	220	220	220	220	220	165	200	200	200	200
Temperature (C)	Correlation Coefficient	0.271	1.000	-0.366	-0.062	-0.505	-0.378	-0.020	-0.736	0.345	0.024	-0.002
	Sig. (2-tailed)	0.000	.	0.000	0.320	0.000	0.000	0.775	0.000	0.000	0.741	0.977
	N	220	260	260	260	260	260	204	190	190	190	190
Salinity	Correlation Coefficient	-0.339	-0.366	1.000	0.311	0.237	0.423	-0.505	-0.087	0.302	-0.318	-0.429
	Sig. (2-tailed)	0.000	0.000	.	0.000	0.000	0.000	0.000	0.232	0.000	0.000	0.000
	N	220	260	260	260	260	260	204	190	190	190	190
DO %	Correlation Coefficient	-0.460	-0.062	0.311	1.000	0.816	0.190	-0.074	0.226	0.040	-0.385	-0.399
	Sig. (2-tailed)	0.000	0.320	0.000	.	0.000	0.002	0.294	0.002	0.583	0.000	0.000
	N	220	260	260	260	260	260	204	190	190	190	190
DO mg/L	Correlation Coefficient	-0.436	-0.505	0.237	0.816	1.000	0.252	-0.048	0.573	-0.234	-0.252	-0.222
	Sig. (2-tailed)	0.000	0.000	0.000	0.000	.	0.000	0.496	0.000	0.001	0.000	0.002
	N	220	260	260	260	260	260	204	190	190	190	190
pH	Correlation Coefficient	-0.106	-0.378	0.423	0.190	0.252	1.000	-0.109	0.286	-0.029	-0.083	-0.149
	Sig. (2-tailed)	0.116	0.000	0.000	0.002	0.000	.	0.121	0.000	0.693	0.254	0.040
	N	220	260	260	260	260	260	204	190	190	190	190
Turbidity (NTU)	Correlation Coefficient	0.219	-0.020	-0.505	-0.074	-0.048	-0.109	1.000	0.287	-0.205	0.185	0.281
	Sig. (2-tailed)	0.005	0.775	0.000	0.294	0.496	0.121	.	0.001	0.016	0.030	0.001
	N	165	204	204	204	204	204	204	137	137	137	137
Pressure (mbar)	Correlation Coefficient	-0.206	-0.736	-0.087	0.226	0.573	0.286	0.287	1.000	-0.540	0.414	0.420
	Sig. (2-tailed)	0.003	0.000	0.232	0.002	0.000	0.000	0.001	.	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200

Rain (mm)	Correlation Coefficient	-0.163	0.345	0.302	0.040	-0.234	-0.029	-0.205	-0.540	1.000	-0.547	-0.555
	Sig. (2-tailed)	0.021	0.000	0.000	0.583	0.001	0.693	0.016	0.000	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200
PAR (uE)	Correlation Coefficient	0.180	0.024	-0.318	-0.385	-0.252	-0.083	0.185	0.414	-0.547	1.000	0.946
	Sig. (2-tailed)	0.011	0.741	0.000	0.000	0.000	0.254	0.030	0.000	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200
Solar Radiation (W/m^2)	Correlation Coefficient	0.168	-0.002	-0.429	-0.399	-0.222	-0.149	0.281	0.420	-0.555	0.946	1.000
	Sig. (2-tailed)	0.018	0.977	0.000	0.000	0.002	0.040	0.001	0.000	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200
Wind Speed (m/s)	Correlation Coefficient	0.048	0.020	-0.340	-0.219	-0.071	0.024	0.340	0.035	-0.109	0.186	0.185
	Sig. (2-tailed)	0.501	0.787	0.000	0.002	0.334	0.742	0.000	0.627	0.125	0.008	0.009
	N	200	190	190	190	190	190	137	200	200	200	200
Gust Speed (m/s)	Correlation Coefficient	0.113	0.002	-0.269	-0.248	-0.094	0.138	0.169	0.009	-0.138	0.185	0.146
	Sig. (2-tailed)	0.110	0.973	0.000	0.001	0.199	0.058	0.048	0.899	0.051	0.009	0.039
	N	200	190	190	190	190	190	137	200	200	200	200
Air Temp (C)	Correlation Coefficient	0.350	0.931	-0.188	-0.516	-0.812	-0.383	-0.063	-0.669	0.275	0.131	0.096
	Sig. (2-tailed)	0.000	0.000	0.009	0.000	0.000	0.000	0.463	0.000	0.000	0.065	0.175
	N	200	190	190	190	190	190	137	200	200	200	200
Dew Point (C)	Correlation Coefficient	0.219	0.804	-0.083	-0.395	-0.675	-0.306	-0.119	-0.659	0.393	-0.127	-0.217
	Sig. (2-tailed)	0.002	0.000	0.253	0.000	0.000	0.000	0.165	0.000	0.000	0.074	0.002
	N	200	190	190	190	190	190	137	200	200	200	200
RH (%)	Correlation Coefficient	0.005	0.463	0.026	-0.065	-0.268	-0.168	0.042	-0.567	0.439	-0.599	-0.672
	Sig. (2-tailed)	0.944	0.000	0.719	0.371	0.000	0.021	0.625	0.000	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200

Gauge Height (Feet)	Correlation Coefficient	-0.280	-0.197	0.318	0.315	0.228	0.285	-0.070	0.118	0.152	-0.491	-0.618
	Sig. (2-tailed)	0.000	0.003	0.000	0.000	0.000	0.000	0.359	0.096	0.031	0.000	0.000
	N	230	230	230	230	230	230	175	200	200	200	200
Total daily pptn JR day0	Correlation Coefficient	-0.113	0.431	-0.188	-0.107	-0.256	-0.114	0.084	-0.422	0.742	-0.491	-0.476
	Sig. (2-tailed)	0.087	0.000	0.002	0.085	0.000	0.066	0.230	0.000	0.000	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
Total daily pptn WR day0	Correlation Coefficient	-0.143	0.284	0.023	-0.149	-0.318	0.014	0.139	-0.494	0.818	-0.574	-0.565
	Sig. (2-tailed)	0.030	0.000	0.713	0.016	0.000	0.819	0.048	0.000	0.000	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
Daily avg discharge WR day0	Correlation Coefficient	0.086	0.148	-0.413	-0.308	-0.158	-0.252	-0.003	0.063	0.202	-0.084	-0.108
	Sig. (2-tailed)	0.192	0.017	0.000	0.000	0.011	0.000	0.969	0.374	0.004	0.238	0.127
	N	230	260	260	260	260	260	204	200	200	200	200

		Wind Sp (m/s)	Gust Sp (m/s)	Air Temp (C)	Dew Pt (C)	RH (%)	Gauge Ht (Ft)	Total daily pptn JR	Total daily pptn WR	Daily avg discharge WR (m^3/s)
NO2NO3	Correlation Coefficient	0.075	0.002	0.052	-0.021	-0.067	-0.342	0.066	-0.065	0.367
	Sig. (2-tailed)	0.290	0.976	0.462	0.771	0.350	0.000	0.284	0.286	0.000
	N	199	199	199	199	199	239	269	269	269
PO4	Correlation Coefficient	0.216	0.232	0.032	-0.046	0.008	0.195	-0.067	-0.078	-0.148
	Sig. (2-tailed)	0.002	0.001	0.651	0.518	0.907	0.002	0.269	0.201	0.015
	N	200	200	200	200	200	240	270	270	270
NO2	Correlation Coefficient	0.228	0.226	-0.178	-0.380	-0.478	-0.218	-0.028	-0.096	0.123
	Sig. (2-tailed)	0.001	0.001	0.012	0.000	0.000	0.001	0.643	0.116	0.044
	N	200	200	200	200	200	240	270	270	270
NO3	Correlation Coefficient	0.079	0.008	0.097	0.066	0.020	-0.293	0.065	-0.065	0.369
	Sig. (2-tailed)	0.267	0.911	0.173	0.352	0.781	0.000	0.286	0.286	0.000
	N	200	200	200	200	200	240	270	270	270
NH4	Correlation Coefficient	0.390	0.313	0.225	0.201	0.219	-0.081	0.134	0.044	0.406
	Sig. (2-tailed)	0.000	0.000	0.001	0.004	0.002	0.219	0.038	0.499	0.000
	N	200	200	200	200	200	230	240	240	240
DIN	Correlation Coefficient	0.256	0.170	0.143	0.139	0.115	-0.214	0.103	-0.043	0.395
	Sig. (2-tailed)	0.000	0.016	0.044	0.050	0.106	0.001	0.111	0.507	0.000
	N	200	200	200	200	200	230	240	240	240
SiO2	Correlation Coefficient	-0.124	-0.086	0.228	0.076	-0.346	-0.337	0.010	-0.106	0.166
	Sig. (2-tailed)	0.080	0.225	0.001	0.284	0.000	0.000	0.866	0.083	0.006
	N	200	200	200	200	200	240	270	270	270
Chlorophyll	Correlation Coefficient	0.148	0.080	0.236	0.177	-0.035	-0.049	0.125	-0.046	0.286
	Sig. (2-tailed)	0.036	0.261	0.001	0.012	0.619	0.461	0.058	0.483	0.000
	N	200	200	200	200	200	230	230	230	230

Phaeopigments	Correlation Coefficient	0.048	0.113	0.350	0.219	0.005	-0.280	-0.113	-0.143	0.086
	Sig. (2-tailed)	0.501	0.110	0.000	0.002	0.944	0.000	0.087	0.030	0.192
	N	200	200	200	200	200	230	230	230	230
Temperature (C)	Correlation Coefficient	0.020	0.002	0.931	0.804	0.463	-0.197	0.431	0.284	0.148
	Sig. (2-tailed)	0.787	0.973	0.000	0.000	0.000	0.003	0.000	0.000	0.017
	N	190	190	190	190	190	230	260	260	260
Salinity	Correlation Coefficient	-0.340	-0.269	-0.188	-0.083	0.026	0.318	-0.188	0.023	-0.413
	Sig. (2-tailed)	0.000	0.000	0.009	0.253	0.719	0.000	0.002	0.713	0.000
	N	190	190	190	190	190	230	260	260	260
DO %	Correlation Coefficient	-0.219	-0.248	-0.516	-0.395	-0.065	0.315	-0.107	-0.149	-0.308
	Sig. (2-tailed)	0.002	0.001	0.000	0.000	0.371	0.000	0.085	0.016	0.000
	N	190	190	190	190	190	230	260	260	260
DO mg/L	Correlation Coefficient	-0.071	-0.094	-0.812	-0.675	-0.268	0.228	-0.256	-0.318	-0.158
	Sig. (2-tailed)	0.334	0.199	0.000	0.000	0.000	0.000	0.000	0.000	0.011
	N	190	190	190	190	190	230	260	260	260
pH	Correlation Coefficient	0.024	0.138	-0.383	-0.306	-0.168	0.285	-0.114	0.014	-0.252
	Sig. (2-tailed)	0.742	0.058	0.000	0.000	0.021	0.000	0.066	0.819	0.000
	N	190	190	190	190	190	230	260	260	260
Turbidity (NTU)	Correlation Coefficient	0.340	0.169	-0.063	-0.119	0.042	-0.070	0.084	0.139	-0.003
	Sig. (2-tailed)	0.000	0.048	0.463	0.165	0.625	0.359	0.230	0.048	0.969
	N	137	137	137	137	137	175	204	204	204
Pressure (mbar)	Correlation Coefficient	0.035	0.009	-0.669	-0.659	-0.567	0.118	-0.422	-0.494	0.063
	Sig. (2-tailed)	0.627	0.899	0.000	0.000	0.000	0.096	0.000	0.000	0.374
	N	200	200	200	200	200	200	200	200	200

Rain (mm)	Correlation Coefficient	-0.109	-0.138	0.275	0.393	0.439	0.152	0.742	0.818	0.202
	Sig. (2-tailed)	0.125	0.051	0.000	0.000	0.000	0.031	0.000	0.000	0.004
	N	200	200	200	200	200	200	200	200	200
PAR (uE)	Correlation Coefficient	0.186	0.185	0.131	-0.127	-0.599	-0.491	-0.491	-0.574	-0.084
	Sig. (2-tailed)	0.008	0.009	0.065	0.074	0.000	0.000	0.000	0.000	0.238
	N	200	200	200	200	200	200	200	200	200
Solar Radiation (W/m^2)	Correlation Coefficient	0.185	0.146	0.096	-0.217	-0.672	-0.618	-0.476	-0.565	-0.108
	Sig. (2-tailed)	0.009	0.039	0.175	0.002	0.000	0.000	0.000	0.000	0.127
	N	200	200	200	200	200	200	200	200	200
Wind Speed (m/s)	Correlation Coefficient	1.000	0.964	0.012	-0.032	-0.030	-0.009	-0.030	-0.110	0.087
	Sig. (2-tailed)	.	0.000	0.866	0.657	0.672	0.905	0.673	0.120	0.219
	N	200	200	200	200	200	200	200	200	200
Gust Speed (m/s)	Correlation Coefficient	0.964	1.000	-0.006	-0.056	-0.077	0.034	-0.090	-0.147	0.000
	Sig. (2-tailed)	0.000	.	0.933	0.434	0.280	0.634	0.205	0.038	1.000
	N	200	200	200	200	200	200	200	200	200
Air Temp (C)	Correlation Coefficient	0.012	-0.006	1.000	0.868	0.427	-0.314	0.254	0.229	0.265
	Sig. (2-tailed)	0.866	0.933	.	0.000	0.000	0.000	0.000	0.001	0.000
	N	200	200	200	200	200	200	200	200	200
Dew Point (C)	Correlation Coefficient	-0.032	-0.056	0.868	1.000	0.720	-0.101	0.413	0.402	0.459
	Sig. (2-tailed)	0.657	0.434	0.000	.	0.000	0.156	0.000	0.000	0.000
	N	200	200	200	200	200	200	200	200	200
RH (%)	Correlation Coefficient	-0.030	-0.077	0.427	0.720	1.000	0.325	0.582	0.597	0.495
	Sig. (2-tailed)	0.672	0.280	0.000	0.000	.	0.000	0.000	0.000	0.000
	N	200	200	200	200	200	200	200	200	200

Gauge Height (Feet)	Correlation Coefficient	-0.009	0.034	-0.314	-0.101	0.325	1.000	-0.001	0.170	-0.068
	Sig. (2-tailed)	0.905	0.634	0.000	0.156	0.000	.	0.985	0.008	0.294
	N	200	200	200	200	200	240	240	240	240
Total daily pptn JR day0	Correlation Coefficient	-0.030	-0.090	0.254	0.413	0.582	-0.001	1.000	0.830	0.426
	Sig. (2-tailed)	0.673	0.205	0.000	0.000	0.000	0.985	.	0.000	0.000
	N	200	200	200	200	200	240	270	270	270
Total daily pptn WR day0	Correlation Coefficient	-0.110	-0.147	0.229	0.402	0.597	0.170	0.830	1.000	0.235
	Sig. (2-tailed)	0.120	0.038	0.001	0.000	0.000	0.008	0.000	.	0.000
	N	200	200	200	200	200	240	270	270	270
Daily avg discharge WR day0	Correlation Coefficient	0.087	0.000	0.265	0.459	0.495	-0.068	0.426	0.235	1.000
	Sig. (2-tailed)	0.219	1.000	0.000	0.000	0.000	0.294	0.000	0.000	.
	N	200	200	200	200	200	240	270	270	270

		NO2NO3	PO4	NO2	NO3	NH4	DIN	SiO2	Chlorophyll
NO2NO3	Correlation Coefficient	1.000	-0.005	0.481	0.977	0.432	0.837	0.271	0.085
	Sig. (2-tailed)	.	0.931	0.000	0.000	0.000	0.000	0.000	0.202
	N	269	269	269	269	239	239	269	229
PO4	Correlation Coefficient	-0.005	1.000	0.006	0.004	0.140	0.019	-0.355	-0.033
	Sig. (2-tailed)	0.931	.	0.925	0.952	0.030	0.769	0.000	0.619
	N	269	270	270	270	240	240	270	230
NO2	Correlation Coefficient	0.481	0.006	1.000	0.355	0.085	0.330	0.084	0.035
	Sig. (2-tailed)	0.000	0.925	.	0.000	0.188	0.000	0.167	0.594
	N	269	270	270	270	240	240	270	230
NO3	Correlation Coefficient	0.977	0.004	0.355	1.000	0.442	0.843	0.267	0.094
	Sig. (2-tailed)	0.000	0.952	0.000	.	0.000	0.000	0.000	0.154
	N	269	270	270	270	240	240	270	230
NH4	Correlation Coefficient	0.432	0.140	0.085	0.442	1.000	0.781	0.134	0.243
	Sig. (2-tailed)	0.000	0.030	0.188	0.000	.	0.000	0.038	0.000
	N	239	240	240	240	240	240	240	230
DIN	Correlation Coefficient	0.837	0.019	0.330	0.843	0.781	1.000	0.241	0.231
	Sig. (2-tailed)	0.000	0.769	0.000	0.000	0.000	.	0.000	0.000
	N	239	240	240	240	240	240	240	230
SiO2	Correlation Coefficient	0.271	-0.355	0.084	0.267	0.134	0.241	1.000	0.311
	Sig. (2-tailed)	0.000	0.000	0.167	0.000	0.038	0.000	.	0.000
	N	269	270	270	270	240	240	270	230
Chlorophyll	Correlation Coefficient	0.085	-0.033	0.035	0.094	0.243	0.231	0.311	1.000
	Sig. (2-tailed)	0.202	0.619	0.594	0.154	0.000	0.000	0.000	.
	N	229	230	230	230	230	230	230	230

Phaeopigments	Correlation Coefficient	0.244	0.072	0.323	0.191	0.003	0.118	0.217	-0.040
	Sig. (2-tailed)	0.000	0.278	0.000	0.004	0.963	0.074	0.001	0.551
	N	229	230	230	230	230	230	230	230
Temperature (C)	Correlation Coefficient	-0.102	-0.071	-0.128	-0.071	0.246	0.167	0.439	0.426
	Sig. (2-tailed)	0.102	0.253	0.039	0.252	0.000	0.011	0.000	0.000
	N	260	260	260	260	230	230	260	220
Salinity	Correlation Coefficient	-0.367	0.334	-0.481	-0.315	-0.412	-0.520	-0.373	-0.197
	Sig. (2-tailed)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003
	N	260	260	260	260	230	230	260	220
DO %	Correlation Coefficient	-0.398	0.099	-0.343	-0.357	-0.157	-0.212	-0.243	0.178
	Sig. (2-tailed)	0.000	0.111	0.000	0.000	0.017	0.001	0.000	0.008
	N	260	260	260	260	230	230	260	220
DO mg/L	Correlation Coefficient	-0.180	0.028	-0.113	-0.171	-0.108	-0.097	-0.322	-0.014
	Sig. (2-tailed)	0.004	0.652	0.070	0.006	0.102	0.145	0.000	0.833
	N	260	260	260	260	230	230	260	220
pH	Correlation Coefficient	-0.304	0.259	-0.132	-0.284	-0.371	-0.421	-0.318	-0.290
	Sig. (2-tailed)	0.000	0.000	0.033	0.000	0.000	0.000	0.000	0.000
	N	260	260	260	260	230	230	260	220
Turbidity (NTU)	Correlation Coefficient	0.206	0.078	0.327	0.165	0.362	0.412	-0.218	0.108
	Sig. (2-tailed)	0.003	0.270	0.000	0.018	0.000	0.000	0.002	0.168
	N	204	204	204	204	175	175	204	165
Pressure (mbar)	Correlation Coefficient	0.090	-0.017	0.342	0.050	-0.033	0.052	-0.099	-0.043
	Sig. (2-tailed)	0.204	0.811	0.000	0.478	0.647	0.463	0.164	0.542
	N	199	200	200	200	200	200	200	200

Rain (mm)	Correlation Coefficient	-0.160	-0.036	-0.341	-0.123	0.074	-0.051	0.033	0.116
	Sig. (2-tailed)	0.024	0.616	0.000	0.084	0.300	0.478	0.647	0.101
	N	199	200	200	200	200	200	200	200
PAR (uE)	Correlation Coefficient	0.244	0.003	0.346	0.222	0.067	0.204	0.396	0.136
	Sig. (2-tailed)	0.001	0.962	0.000	0.002	0.348	0.004	0.000	0.055
	N	199	200	200	200	200	200	200	200
Solar Radiation (W/m^2)	Correlation Coefficient	0.277	-0.054	0.452	0.220	0.064	0.190	0.389	0.131
	Sig. (2-tailed)	0.000	0.444	0.000	0.002	0.366	0.007	0.000	0.064
	N	199	200	200	200	200	200	200	200
Wind Speed (m/s)	Correlation Coefficient	0.075	0.216	0.228	0.079	0.390	0.256	-0.124	0.148
	Sig. (2-tailed)	0.290	0.002	0.001	0.267	0.000	0.000	0.080	0.036
	N	199	200	200	200	200	200	200	200
Gust Speed (m/s)	Correlation Coefficient	0.002	0.232	0.226	0.008	0.313	0.170	-0.086	0.080
	Sig. (2-tailed)	0.976	0.001	0.001	0.911	0.000	0.016	0.225	0.261
	N	199	200	200	200	200	200	200	200
Air Temp (C)	Correlation Coefficient	0.052	0.032	-0.178	0.097	0.225	0.143	0.228	0.236
	Sig. (2-tailed)	0.462	0.651	0.012	0.173	0.001	0.044	0.001	0.001
	N	199	200	200	200	200	200	200	200
Dew Point (C)	Correlation Coefficient	-0.021	-0.046	-0.380	0.066	0.201	0.139	0.076	0.177
	Sig. (2-tailed)	0.771	0.518	0.000	0.352	0.004	0.050	0.284	0.012
	N	199	200	200	200	200	200	200	200
RH (%)	Correlation Coefficient	-0.067	0.008	-0.478	0.020	0.219	0.115	-0.346	-0.035
	Sig. (2-tailed)	0.350	0.907	0.000	0.781	0.002	0.106	0.000	0.619
	N	199	200	200	200	200	200	200	200

Gauge Height (Feet)	Correlation Coefficient	-0.342	0.195	-0.218	-0.293	-0.081	-0.214	-0.337	-0.049
	Sig. (2-tailed)	0.000	0.002	0.001	0.000	0.219	0.001	0.000	0.461
	N	239	240	240	240	230	230	240	230
Total daily pptn JR day0	Correlation Coefficient	0.066	-0.067	-0.028	0.065	0.134	0.103	0.010	0.125
	Sig. (2-tailed)	0.284	0.269	0.643	0.286	0.038	0.111	0.866	0.058
	N	269	270	270	270	240	240	270	230
Total daily pptn WR day0	Correlation Coefficient	-0.065	-0.078	-0.096	-0.065	0.044	-0.043	-0.106	-0.046
	Sig. (2-tailed)	0.286	0.201	0.116	0.286	0.499	0.507	0.083	0.483
	N	269	270	270	270	240	240	270	230
Daily avg discharge WR day0	Correlation Coefficient	0.367	-0.148	0.123	0.369	0.406	0.395	0.166	0.286
	Sig. (2-tailed)	0.000	0.015	0.044	0.000	0.000	0.000	0.006	0.000
	N	269	270	270	270	240	240	270	230

		Phaeo	Temp (C)	Salinity	DO %	DO mg/L	pH	Turbidity (NTU)	Pressure (mbar)	Rain (mm)	PAR (uE)	Solar Rad (W/m^2)
NO2NO3	Correlation Coefficient	0.244	-0.102	-0.367	-0.398	-0.180	-0.304	0.206	0.090	-0.160	0.244	0.277
	Sig. (2-tailed)	0.000	0.102	0.000	0.000	0.004	0.000	0.003	0.204	0.024	0.001	0.000
	N	229	260	260	260	260	260	204	199	199	199	199
PO4	Correlation Coefficient	0.072	-0.071	0.334	0.099	0.028	0.259	0.078	-0.017	-0.036	0.003	-0.054
	Sig. (2-tailed)	0.278	0.253	0.000	0.111	0.652	0.000	0.270	0.811	0.616	0.962	0.444
	N	230	260	260	260	260	260	204	200	200	200	200
NO2	Correlation Coefficient	0.323	-0.128	-0.481	-0.343	-0.113	-0.132	0.327	0.342	-0.341	0.346	0.452
	Sig. (2-tailed)	0.000	0.039	0.000	0.000	0.070	0.033	0.000	0.000	0.000	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
NO3	Correlation Coefficient	0.191	-0.071	-0.315	-0.357	-0.171	-0.284	0.165	0.050	-0.123	0.222	0.220
	Sig. (2-tailed)	0.004	0.252	0.000	0.000	0.006	0.000	0.018	0.478	0.084	0.002	0.002
	N	230	260	260	260	260	260	204	200	200	200	200
NH4	Correlation Coefficient	0.003	0.246	-0.412	-0.157	-0.108	-0.371	0.362	-0.033	0.074	0.067	0.064
	Sig. (2-tailed)	0.963	0.000	0.000	0.017	0.102	0.000	0.000	0.647	0.300	0.348	0.366
	N	230	230	230	230	230	230	175	200	200	200	200
DIN	Correlation Coefficient	0.118	0.167	-0.520	-0.212	-0.097	-0.421	0.412	0.052	-0.051	0.204	0.190
	Sig. (2-tailed)	0.074	0.011	0.000	0.001	0.145	0.000	0.000	0.463	0.478	0.004	0.007
	N	230	230	230	230	230	230	175	200	200	200	200
SiO2	Correlation Coefficient	0.217	0.439	-0.373	-0.243	-0.322	-0.318	-0.218	-0.099	0.033	0.396	0.389
	Sig. (2-tailed)	0.001	0.000	0.000	0.000	0.000	0.000	0.002	0.164	0.647	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
Chlorophyll	Correlation Coefficient	-0.040	0.426	-0.197	0.178	-0.014	-0.290	0.108	-0.043	0.116	0.136	0.131
	Sig. (2-tailed)	0.551	0.000	0.003	0.008	0.833	0.000	0.168	0.542	0.101	0.055	0.064
	N	230	220	220	220	220	220	165	200	200	200	200

Phaeopigments	Correlation Coefficient	1.000	0.271	-0.339	-0.460	-0.436	-0.106	0.219	-0.206	-0.163	0.180	0.168
	Sig. (2-tailed)	.	0.000	0.000	0.000	0.000	0.116	0.005	0.003	0.021	0.011	0.018
	N	230	220	220	220	220	220	165	200	200	200	200
Temperature (C)	Correlation Coefficient	0.271	1.000	-0.366	-0.062	-0.505	-0.378	-0.020	-0.736	0.345	0.024	-0.002
	Sig. (2-tailed)	0.000	.	0.000	0.320	0.000	0.000	0.775	0.000	0.000	0.741	0.977
	N	220	260	260	260	260	260	204	190	190	190	190
Salinity	Correlation Coefficient	-0.339	-0.366	1.000	0.311	0.237	0.423	-0.505	-0.087	0.302	-0.318	-0.429
	Sig. (2-tailed)	0.000	0.000	.	0.000	0.000	0.000	0.000	0.232	0.000	0.000	0.000
	N	220	260	260	260	260	260	204	190	190	190	190
DO %	Correlation Coefficient	-0.460	-0.062	0.311	1.000	0.816	0.190	-0.074	0.226	0.040	-0.385	-0.399
	Sig. (2-tailed)	0.000	0.320	0.000	.	0.000	0.002	0.294	0.002	0.583	0.000	0.000
	N	220	260	260	260	260	260	204	190	190	190	190
DO mg/L	Correlation Coefficient	-0.436	-0.505	0.237	0.816	1.000	0.252	-0.048	0.573	-0.234	-0.252	-0.222
	Sig. (2-tailed)	0.000	0.000	0.000	0.000	.	0.000	0.496	0.000	0.001	0.000	0.002
	N	220	260	260	260	260	260	204	190	190	190	190
pH	Correlation Coefficient	-0.106	-0.378	0.423	0.190	0.252	1.000	-0.109	0.286	-0.029	-0.083	-0.149
	Sig. (2-tailed)	0.116	0.000	0.000	0.002	0.000	.	0.121	0.000	0.693	0.254	0.040
	N	220	260	260	260	260	260	204	190	190	190	190
Turbidity (NTU)	Correlation Coefficient	0.219	-0.020	-0.505	-0.074	-0.048	-0.109	1.000	0.287	-0.205	0.185	0.281
	Sig. (2-tailed)	0.005	0.775	0.000	0.294	0.496	0.121	.	0.001	0.016	0.030	0.001
	N	165	204	204	204	204	204	204	137	137	137	137
Pressure (mbar)	Correlation Coefficient	-0.206	-0.736	-0.087	0.226	0.573	0.286	0.287	1.000	-0.540	0.414	0.420
	Sig. (2-tailed)	0.003	0.000	0.232	0.002	0.000	0.000	0.001	.	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200

Rain (mm)	Correlation Coefficient	-0.163	0.345	0.302	0.040	-0.234	-0.029	-0.205	-0.540	1.000	-0.547	-0.555
	Sig. (2-tailed)	0.021	0.000	0.000	0.583	0.001	0.693	0.016	0.000	.	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200
PAR (uE)	Correlation Coefficient	0.180	0.024	-0.318	-0.385	-0.252	-0.083	0.185	0.414	-0.547	1.000	0.946
	Sig. (2-tailed)	0.011	0.741	0.000	0.000	0.000	0.254	0.030	0.000	0.000	.	0.000
	N	200	190	190	190	190	190	137	200	200	200	200
Solar Radiation (W/m^2)	Correlation Coefficient	0.168	-0.002	-0.429	-0.399	-0.222	-0.149	0.281	0.420	-0.555	0.946	1.000
	Sig. (2-tailed)	0.018	0.977	0.000	0.000	0.002	0.040	0.001	0.000	0.000	0.000	.
	N	200	190	190	190	190	190	137	200	200	200	200
Wind Speed (m/s)	Correlation Coefficient	0.048	0.020	-0.340	-0.219	-0.071	0.024	0.340	0.035	-0.109	0.186	0.185
	Sig. (2-tailed)	0.501	0.787	0.000	0.002	0.334	0.742	0.000	0.627	0.125	0.008	0.009
	N	200	190	190	190	190	190	137	200	200	200	200
Gust Speed (m/s)	Correlation Coefficient	0.113	0.002	-0.269	-0.248	-0.094	0.138	0.169	0.009	-0.138	0.185	0.146
	Sig. (2-tailed)	0.110	0.973	0.000	0.001	0.199	0.058	0.048	0.899	0.051	0.009	0.039
	N	200	190	190	190	190	190	137	200	200	200	200
Air Temp (C)	Correlation Coefficient	0.350	0.931	-0.188	-0.516	-0.812	-0.383	-0.063	-0.669	0.275	0.131	0.096
	Sig. (2-tailed)	0.000	0.000	0.009	0.000	0.000	0.000	0.463	0.000	0.000	0.065	0.175
	N	200	190	190	190	190	190	137	200	200	200	200
Dew Point (C)	Correlation Coefficient	0.219	0.804	-0.083	-0.395	-0.675	-0.306	-0.119	-0.659	0.393	-0.127	-0.217
	Sig. (2-tailed)	0.002	0.000	0.253	0.000	0.000	0.000	0.165	0.000	0.000	0.074	0.002
	N	200	190	190	190	190	190	137	200	200	200	200
RH (%)	Correlation Coefficient	0.005	0.463	0.026	-0.065	-0.268	-0.168	0.042	-0.567	0.439	-0.599	-0.672
	Sig. (2-tailed)	0.944	0.000	0.719	0.371	0.000	0.021	0.625	0.000	0.000	0.000	0.000
	N	200	190	190	190	190	190	137	200	200	200	200

Gauge Height (Feet)	Correlation Coefficient	-0.280	-0.197	0.318	0.315	0.228	0.285	-0.070	0.118	0.152	-0.491	-0.618
	Sig. (2-tailed)	0.000	0.003	0.000	0.000	0.000	0.000	0.359	0.096	0.031	0.000	0.000
	N	230	230	230	230	230	230	175	200	200	200	200
Total daily pptn JR day0	Correlation Coefficient	-0.113	0.431	-0.188	-0.107	-0.256	-0.114	0.084	-0.422	0.742	-0.491	-0.476
	Sig. (2-tailed)	0.087	0.000	0.002	0.085	0.000	0.066	0.230	0.000	0.000	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
Total daily pptn WR day0	Correlation Coefficient	-0.143	0.284	0.023	-0.149	-0.318	0.014	0.139	-0.494	0.818	-0.574	-0.565
	Sig. (2-tailed)	0.030	0.000	0.713	0.016	0.000	0.819	0.048	0.000	0.000	0.000	0.000
	N	230	260	260	260	260	260	204	200	200	200	200
Daily avg discharge WR day0	Correlation Coefficient	0.086	0.148	-0.413	-0.308	-0.158	-0.252	-0.003	0.063	0.202	-0.084	-0.108
	Sig. (2-tailed)	0.192	0.017	0.000	0.000	0.011	0.000	0.969	0.374	0.004	0.238	0.127
	N	230	260	260	260	260	260	204	200	200	200	200

		Wind Sp (m/s)	Gust Sp (m/s)	Air Temp (C)	Dew Pt (C)	RH (%)	Gauge Ht (Ft)	Total daily pptn JR	Total daily pptn WR	Daily avg discharge WR (m^3/s)
NO2NO3	Correlation Coefficient	0.075	0.002	0.052	-0.021	-0.067	-0.342	0.066	-0.065	0.367
	Sig. (2-tailed)	0.290	0.976	0.462	0.771	0.350	0.000	0.284	0.286	0.000
	N	199	199	199	199	199	239	269	269	269
PO4	Correlation Coefficient	0.216	0.232	0.032	-0.046	0.008	0.195	-0.067	-0.078	-0.148
	Sig. (2-tailed)	0.002	0.001	0.651	0.518	0.907	0.002	0.269	0.201	0.015
	N	200	200	200	200	200	240	270	270	270
NO2	Correlation Coefficient	0.228	0.226	-0.178	-0.380	-0.478	-0.218	-0.028	-0.096	0.123
	Sig. (2-tailed)	0.001	0.001	0.012	0.000	0.000	0.001	0.643	0.116	0.044
	N	200	200	200	200	200	240	270	270	270
NO3	Correlation Coefficient	0.079	0.008	0.097	0.066	0.020	-0.293	0.065	-0.065	0.369
	Sig. (2-tailed)	0.267	0.911	0.173	0.352	0.781	0.000	0.286	0.286	0.000
	N	200	200	200	200	200	240	270	270	270
NH4	Correlation Coefficient	0.390	0.313	0.225	0.201	0.219	-0.081	0.134	0.044	0.406
	Sig. (2-tailed)	0.000	0.000	0.001	0.004	0.002	0.219	0.038	0.499	0.000
	N	200	200	200	200	200	230	240	240	240
DIN	Correlation Coefficient	0.256	0.170	0.143	0.139	0.115	-0.214	0.103	-0.043	0.395
	Sig. (2-tailed)	0.000	0.016	0.044	0.050	0.106	0.001	0.111	0.507	0.000
	N	200	200	200	200	200	230	240	240	240
SiO2	Correlation Coefficient	-0.124	-0.086	0.228	0.076	-0.346	-0.337	0.010	-0.106	0.166
	Sig. (2-tailed)	0.080	0.225	0.001	0.284	0.000	0.000	0.866	0.083	0.006
	N	200	200	200	200	200	240	270	270	270
Chlorophyll	Correlation Coefficient	0.148	0.080	0.236	0.177	-0.035	-0.049	0.125	-0.046	0.286
	Sig. (2-tailed)	0.036	0.261	0.001	0.012	0.619	0.461	0.058	0.483	0.000
	N	200	200	200	200	200	230	230	230	230

Phaeopigments	Correlation Coefficient	0.048	0.113	0.350	0.219	0.005	-0.280	-0.113	-0.143	0.086
	Sig. (2-tailed)	0.501	0.110	0.000	0.002	0.944	0.000	0.087	0.030	0.192
	N	200	200	200	200	200	230	230	230	230
Temperature (C)	Correlation Coefficient	0.020	0.002	0.931	0.804	0.463	-0.197	0.431	0.284	0.148
	Sig. (2-tailed)	0.787	0.973	0.000	0.000	0.000	0.003	0.000	0.000	0.017
	N	190	190	190	190	190	230	260	260	260
Salinity	Correlation Coefficient	-0.340	-0.269	-0.188	-0.083	0.026	0.318	-0.188	0.023	-0.413
	Sig. (2-tailed)	0.000	0.000	0.009	0.253	0.719	0.000	0.002	0.713	0.000
	N	190	190	190	190	190	230	260	260	260
DO %	Correlation Coefficient	-0.219	-0.248	-0.516	-0.395	-0.065	0.315	-0.107	-0.149	-0.308
	Sig. (2-tailed)	0.002	0.001	0.000	0.000	0.371	0.000	0.085	0.016	0.000
	N	190	190	190	190	190	230	260	260	260
DO mg/L	Correlation Coefficient	-0.071	-0.094	-0.812	-0.675	-0.268	0.228	-0.256	-0.318	-0.158
	Sig. (2-tailed)	0.334	0.199	0.000	0.000	0.000	0.000	0.000	0.000	0.011
	N	190	190	190	190	190	230	260	260	260
pH	Correlation Coefficient	0.024	0.138	-0.383	-0.306	-0.168	0.285	-0.114	0.014	-0.252
	Sig. (2-tailed)	0.742	0.058	0.000	0.000	0.021	0.000	0.066	0.819	0.000
	N	190	190	190	190	190	230	260	260	260
Turbidity (NTU)	Correlation Coefficient	0.340	0.169	-0.063	-0.119	0.042	-0.070	0.084	0.139	-0.003
	Sig. (2-tailed)	0.000	0.048	0.463	0.165	0.625	0.359	0.230	0.048	0.969
	N	137	137	137	137	137	175	204	204	204
Pressure (mbar)	Correlation Coefficient	0.035	0.009	-0.669	-0.659	-0.567	0.118	-0.422	-0.494	0.063
	Sig. (2-tailed)	0.627	0.899	0.000	0.000	0.000	0.096	0.000	0.000	0.374
	N	200	200	200	200	200	200	200	200	200

Rain (mm)	Correlation Coefficient	-0.109	-0.138	0.275	0.393	0.439	0.152	0.742	0.818	0.202
	Sig. (2-tailed)	0.125	0.051	0.000	0.000	0.000	0.031	0.000	0.000	0.004
	N	200	200	200	200	200	200	200	200	200
PAR (uE)	Correlation Coefficient	0.186	0.185	0.131	-0.127	-0.599	-0.491	-0.491	-0.574	-0.084
	Sig. (2-tailed)	0.008	0.009	0.065	0.074	0.000	0.000	0.000	0.000	0.238
	N	200	200	200	200	200	200	200	200	200
Solar Radiation (W/m^2)	Correlation Coefficient	0.185	0.146	0.096	-0.217	-0.672	-0.618	-0.476	-0.565	-0.108
	Sig. (2-tailed)	0.009	0.039	0.175	0.002	0.000	0.000	0.000	0.000	0.127
	N	200	200	200	200	200	200	200	200	200
Wind Speed (m/s)	Correlation Coefficient	1.000	0.964	0.012	-0.032	-0.030	-0.009	-0.030	-0.110	0.087
	Sig. (2-tailed)	.	0.000	0.866	0.657	0.672	0.905	0.673	0.120	0.219
	N	200	200	200	200	200	200	200	200	200
Gust Speed (m/s)	Correlation Coefficient	0.964	1.000	-0.006	-0.056	-0.077	0.034	-0.090	-0.147	0.000
	Sig. (2-tailed)	0.000	.	0.933	0.434	0.280	0.634	0.205	0.038	1.000
	N	200	200	200	200	200	200	200	200	200
Air Temp (C)	Correlation Coefficient	0.012	-0.006	1.000	0.868	0.427	-0.314	0.254	0.229	0.265
	Sig. (2-tailed)	0.866	0.933	.	0.000	0.000	0.000	0.000	0.001	0.000
	N	200	200	200	200	200	200	200	200	200
Dew Point (C)	Correlation Coefficient	-0.032	-0.056	0.868	1.000	0.720	-0.101	0.413	0.402	0.459
	Sig. (2-tailed)	0.657	0.434	0.000	.	0.000	0.156	0.000	0.000	0.000
	N	200	200	200	200	200	200	200	200	200
RH (%)	Correlation Coefficient	-0.030	-0.077	0.427	0.720	1.000	0.325	0.582	0.597	0.495
	Sig. (2-tailed)	0.672	0.280	0.000	0.000	.	0.000	0.000	0.000	0.000
	N	200	200	200	200	200	200	200	200	200

Gauge Height (Feet)	Correlation Coefficient	-0.009	0.034	-0.314	-0.101	0.325	1.000	-0.001	0.170	-0.068
	Sig. (2-tailed)	0.905	0.634	0.000	0.156	0.000	.	0.985	0.008	0.294
	N	200	200	200	200	200	240	240	240	240
Total daily pptn JR day0	Correlation Coefficient	-0.030	-0.090	0.254	0.413	0.582	-0.001	1.000	0.830	0.426
	Sig. (2-tailed)	0.673	0.205	0.000	0.000	0.000	0.985	.	0.000	0.000
	N	200	200	200	200	200	240	270	270	270
Total daily pptn WR day0	Correlation Coefficient	-0.110	-0.147	0.229	0.402	0.597	0.170	0.830	1.000	0.235
	Sig. (2-tailed)	0.120	0.038	0.001	0.000	0.000	0.008	0.000	.	0.000
	N	200	200	200	200	200	240	270	270	270
Daily avg discharge WR day0	Correlation Coefficient	0.087	0.000	0.265	0.459	0.495	-0.068	0.426	0.235	1.000
	Sig. (2-tailed)	0.219	1.000	0.000	0.000	0.000	0.294	0.000	0.000	.
	N	200	200	200	200	200	240	270	270	270

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