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Correlation of Water Quality Indicators for Coastal Marshes

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THE FLORIDA STATE UNIVERSITY
FAMU - FSU COLLEGE OF ENGINEERING

CORRELATION OF WATER QUALITY INDICATORS FOR
COASTAL MARSHES

By
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ABSTRACT

Florida provides some of the most diverse and beautiful environments in the United States, with northwest Florida being a nearly undisturbed bounty of lush lands, springs, and other geological and ecological features. Increases in population have recently started to jeopardize some of Florida's prized areas, including the Ochlocknee-St Marks Basins. Development and agriculture has started to affect watersheds and waterbodies. This study has evaluated the water quality of coastal marshes at the St. Marks national Wildlife Refuge in terms of coliform and chlorophyll *a*. With these constituents, dissolved oxygen (DO), temperature, pH, conductivity, and total suspended solids were evaluated. The study took place with data from January 2009 to June 2009. The study site of this research, St. Marks National Wildlife Refuge, is located 25 miles south of Tallahassee along the Gulf Coast of Florida. To analyze for coliform, 100ml samples were filtered on cellulose nitrate membranes and subsequently incubated. After 24hrs the fecal coli forms were counted and recorded. Measuring the chlorophyll *a* was done taking a volume of water from a particular depth and filtering it through a fine glass fiber filter to collect all of the particulate material great than 1 micron. To extract the chlorophyll, acetone/alcohol was used as a solvent and then measured using an optical kit and Luminometer utilizing a chlorophyll *a* standard. Effective diffusion was then measured in the lab using a diffusion coefficient in distilled water. This process was also observed using a Hamamatsu high sensitivity monochrome digital camera in a dark room, with the sample being placed in a petri dish with native soil. Chlorophyll *a* adsorption was measured by introducing the sample to a column filled with goethite-coated silica sand. Transport parameters were obtained by fitting the experimentally obtained Chlorophyll *a*

breakthrough data using an implicit, finite-difference scheme. All the parameters were optimized by minimizing the sum of squared differences between observed and fitted concentrations using the nonlinear least-square method. Some key observations over the sample period include the dropping of DO as the summer months began indicating increased microbial activity with temperature. TKN and Chlorophyll were observed to increase as temperature increased. When observing Chlorophyll as a function of TKN, the Pearson Correlation and the plot indicate that with increased TKN more microbiological activity occurs – such as with plants, algae, etc. Interestingly when Chlorophyll was plotted as a function of DO, the low DO encourages microbial growth, however only to an extent. The point can be made that less polluted water has less microbial activity. The correlation between coliform and turbidity showed an increase of coliform with NTU. Chlorophyll as a function of turbidity showed no correlation. However as a first order function, $d[\text{Chlorophyll } a]/d[\text{Turbidity}]$ displayed a linear relationship with turbidity, indicating the correlation of chlorophyll *a* variation with respect to turbidity. When conductivity was plotted against pH, the variability of conductivity and pH was not significantly different. Chlorophyll *a* also had a correlation with pH. With the increase of pH, chlorophyll *a* exponentially decreased. This research shows the increased ability to forecast and assess risk presented by development and agriculture of coastal environment based on physical and chemical variations in coastal wetland ecosystems.

CHAPTER ONE

BACKGROUND

The Ochlockonee–St. Marks Basins, which include the Lower Ochlockonee River Basin and the Apalachee Bay–St. Marks River Basin, are located in the Big Bend region of northwestern Florida and southern Georgia (Figure 1). This is one of the most pristine areas of Florida, with a much lower population density and fewer stresses on the environment than in other parts of the state (Bonvechio et al. 2005). Vast areas of land remain relatively undeveloped and many waterbodies have not been significantly modified. More than one-third of the basin group is protected as publicly owned conservation lands. Consequently, rivers, streams, lakes, estuaries, and springs continue to provide great recreational opportunities to area residents and critical habitat for wildlife (Fu and Winchester 1994). In some areas, however, urban and residential development, agricultural practices, and other influences threaten the health of watersheds and have adversely affected waterbodies. Extending from southern Georgia through Florida to the Gulf of Mexico, the Ochlockonee-St. Marks Basins drain a combined area of more than 3,600 square miles. Although they exist as two distinct hydrologic basins, the Ochlockonee-St. Marks Basins both drain to Apalachee Bay in the Gulf of Mexico. The physiography, geology, ecology, and socioeconomic settings in both basins are similar (O'Brien and Williams 2002).

More than 57 percent of the land area in the two basins consisted of natural forest and managed pines and more than 22 percent was made up of wetlands. Agriculture

(primarily pastureland) occupied approximately 8 percent of the basins. Row croplands made up less than 1 percent of the basin area south of the Florida–Georgia state line (Korajkic et al. 2009). However, there is a far larger agricultural area in the Georgia portion of the Ochlockonee River Basin. Approximately 7 percent of the Ochlockonee-St. Marks Basins was classified as urban (Young and Laerm 1993). Two large lakes in the two basins are artificially impounded. Lake Talquin, the area’s largest lake, is an impoundment of the Ochlockonee River that was originally created to provide hydroelectric power. It detains water from the Ochlockonee River and smaller streams in the upper part of the basin (Young and Laerm 1993). Lake Munson, south of Tallahassee, is maintained as an impoundment to provide floodwater equalization, protecting residential areas to the south. It receives urban stormwater via the city’s network of stormwater ditches. Water control structures on Lake Miccosukee, Lake Iamonia, and the Lake Lafayette chain artificially control water levels.

Figure 1. The Ochlockonee–St. Marks Basins Map

1.1 Surface Water Resources

The Ochlockonee-St. Marks Basins contain numerous surface waterbodies, including lakes, streams, wetlands, and springs. Surface water hydrology in the St. Marks Basin differs from that of the Ochlockonee Basin. The Ochlockonee Basin is an integrated, alluvial drainage system almost entirely drained by the Ochlockonee River and its tributaries (Parauka et al. 2011). From the Florida–Georgia state line, the Ochlockonee River flows southward and southwestward across northwestern Leon County and eastern Gadsden County before it reaches the Lake Talquin reservoir (Fu and Winchester 1994). Before it reaches the lake, the Ochlockonee River flows at an annualized average rate of 1,530 ft³/s (Parauka et al. 2011). The Apalachee–St. Marks Basins are disjointed, with no integrated drainage from most of the northern part of the basins, a short coastal drainage system composed of local streams draining coastal regions, and the St. Marks River being the only continuous integrated drainage system. The St. Marks Basin contains two primary rivers, the St. Marks River and its major tributary, the Wakulla River.

Only two surface waterbodies in the Ochlockonee-St. Marks Basins are Class I or potable waters. These are Quincy Creek, a tributary of the Little River in the area north of Lake Talquin in the Ochlockonee Basin, and its tributary, Holman Branch. Quincy Creek in Gadsden County is used as a primary supply of water for the city of Quincy. Areas of Apalachee Bay along the Wakulla County coastline and Ochlockonee Bay within both Wakulla and Franklin Counties are designated as Class II or shellfish propagation/harvesting waters because they include commercially viable oyster beds. All other waters in the Ochlockonee and St. Marks Basins are designated Class III or

recreation/propagation/fish and wildlife waters (Perez-Dominguez et al. 2012). There are no Class IV or V waters in the basins.

1.1.1 Urban Streams

Tallahassee, the region's largest city in size and in population, is located between the two rivers and its urban and suburban areas lie in both basins. Networks of channelized urban streams and ditches provide stormwater drainage from the Tallahassee area and discharge to area lakes. The ditches receive heavy loads of stormwater runoff from roadways, parking lots, commercial and industrial sites, and other paved surfaces in the city.

1.1.2 Wetlands and Estuaries

Wetlands cover approximately 18 percent of the Ochlockonee-St. Marks Basins. In the basins, wetlands exist as near-coastal marshes, in river floodplains, in areas of poor soil drainage adjacent to lakes, and in poorly drained swamps and "bays" of the Apalachicola Coastal Lowlands. Most of the wetland areas are expansive, internally drained "bays" within the fl at Apalachicola Coastal Lowlands (Anderson and Lockaby 2011).

The major estuary is Apalachee Bay, a large, open, shallow bay that receives water from the Ochlockonee and St. Marks Rivers, local tidal creeks and drains, and submarine springs. Apalachee Bay supports a variety of aquatic habitats, including salt marshes, seagrass beds, oyster bars and reefs, and hardbottom reefs (Stoner 1980). It also contains part of the most extensive span of seagrass beds in the country: the Big Bend Seagrasses Aquatic Preserve. This area is the largest aquatic preserve in Florida, encompassing about

450,000 acres of seagrass beds and salt marsh that extend southward and eastward from the St. Marks River to the Withlacoochee River (Valentine and Heck 1991). Apalachee Bay is an important recreational and commercial fishery in the region.

1.2 Ground Water Resources

1.2.1 Aquifers

In the Ochlockonee-St. Marks Basins, ground water for potable supply, irrigation, and industrial uses is obtained from the Floridan aquifer, a vast carbonate aquifer system that exists throughout most of Florida and extends into other southern states. The Tallahassee municipal well field is the primary user of ground water from this aquifer. Throughout most of the area, the Floridan aquifer is highly productive and provides ample quantities of water (Morrissey et al. 2010). However, west of the Ochlockonee River in Gadsden County, the limestone units of the Floridan aquifer are less porous and permeable, the aquifer does not produce very well, and the amount of ground water available for irrigation and municipal water supply is restricted. Water quality in the Floridan aquifer west of the Ochlockonee River is also of poorer quality, having a higher dissolved mineral content (Ward et al. 1992).

1.2.2 Ground Water–Surface Water Interactions

Ground water–surface water interaction is important in this part of the state. In the Tifton Uplands/Tallahassee Hills, north of the Cody Scarp, the Floridan aquifer is under artesian pressure, confined by a thick overburden of sediment that affords a degree of protection against pollution (Brown and Reece 1978). However, breaches in the confining layer,

mainly sinkholes, provide pathways for recharge and the introduction of contaminants. Springs provide the most direct and obvious connection between ground water and surface water. Wakulla Springs, one of the largest freshwater springs in the world, is the site of a state park.

In recent years, nitrate has become a contaminant of concern in ground water in many parts of Florida (Heffernan et al. 2012). It has been documented that elevated concentrations of nitrate in ground water commonly result from agricultural and urban land use practices in ground water recharge areas. Overall, nitrate levels in wells of the Ochlockonee-St. Marks Basins are considerably lower than the drinking water standard and currently cause no public health concerns, except in isolated instances (Ritorto et al. 2009). However, increasing nitrate concentrations in Floridan aquifer springs in the basins are a significant ecological concern. Water quality data indicate that the nitrate concentration in the springs has increased from between 0.10 mg/l and 0.33 mg/l in the 1970s to approximately 1 mg/L in recent years (Heffernan et al. 2012; Smith and Swarzenski 2012).

Water use in the two basins averaged more than 75 million gallons per day with more than 80 percent of the water consumed from ground water sources (Corbett et al. 1999). The main uses of water in the two basins are 39% for public supply and 28% for irrigation. Most ground water use is in the St. Marks Basin, which is 61% with the largest amount used for public supply. Most surface water use is in the Ochlockonee Basin, which is 93% with the largest amount used for irrigation (Tan and Shih 1991).

1.3 Watershed Management Activities

Over the years, management plans and activities in the basins have been implemented to eliminate wastewater discharges; reduce discharges of polluted stormwater from urban and agricultural areas; and protect, preserve, and restore special areas.

1.3.1 Elimination of Wastewater Discharges to Surface Water

The City of Tallahassee constructed a land application system for disposal of municipal wastewater in the late 1970s and eliminated the direct discharge of municipal sewage to Munson Slough in the early 1980s. This has been the most significant step toward improving water quality in Lake Munson. During the same period, Wakulla County created a municipal wastewater treatment facility (spray irrigation) and required coastal residents and businesses in the communities of Panacea and Ochlockonee Bay to connect to the central system and discontinue the use of individual septic tanks. More recently, the city of Havana implemented a similar spray irrigation facility to eliminate wastewater discharge to surface water.

1.3.2 Tallahassee Area Stormwater Management

The City of Tallahassee Stormwater Management Division and the Leon County Public Works Department have carried out most of the stormwater management activities for the basin to date. Activities have focused on implementing drainage improvements, carrying out stormwater retrofits, and restoring Tallahassee lakes. It has long been documented that municipal wastewater discharges have degraded urban stormwater channels and Lake Munson. The cessation of sewage discharges gave some relief to these waterbodies.

However, not until the 1980s was urban stormwater pollution addressed as a significant concern for water quality (Entry 2012). Since then, state and local governments and the general public have become more aware of the need for stormwater treatment, and there are increasing numbers of stormwater projects as well as use of nonstructural management practices. The nonstructural stormwater practices include public information, staff training, incident reports, and other activities to address and prevent polluted stormwater.

1.3.3 Local Stormwater Activities and Projects

In accordance with the 1990 Tallahassee–Leon County Comprehensive Plan, the city has constructed 11 regional facilities throughout the community that incorporate both water quality treatment as well as flood prevention into their design. Development guidelines and permitting requirements to prevent or minimize stormwater contamination also resulted from the Comprehensive Plan and the stormwater ordinance. Stormwater Pollution Reduction Program (SPRP), a monitoring program and development of a computer model to simulate existing and future pollutant loadings, has been established for the development of a problem prioritization system to identify areas with the optimum potential for stormwater treatment.

1.3.4 Ground Water Protection

As required by the Tallahassee-Leon County Comprehensive Plan, an Aquifer Protection Program was established to protect wellheads, delineate high-recharge areas and areas most susceptible to ground water contamination, and identify contaminated areas. The

City of Tallahassee Aquifer Protection Program is responsible for implementing the 1992 Aquifer Recharge/Wellhead Protection Ordinance. Projects in which the city has been involved include a karst inventory of Leon County and a wellhead protection assessment and modeling project funded by the EPA.

1.3.5 Agricultural Best Management Practices

The FWRA authorizes the Florida Department of Agriculture and Consumer Services (DACS) to develop interim measures and agricultural best management practices (BMPs). Additional authority for agricultural BMPs is provided in legislation on nitrates and groundwater, the Lake Okeechobee Protection Program, Agricultural Water Conservation, and Florida Right to Farm Act Amendments.

1.4 Watershed Modeling

A watershed model is essentially a series of algorithms applied to watershed characteristics and meteorological data to simulate naturally occurring land-based processes over an extended period, including hydrology and pollutant transport (Srivastava et al. 2007). Many watershed models are also capable of simulating in-stream processes using the land-based calculations as input. Receiving water models are composed of a series of algorithms applied to characteristics data to simulate flow and water quality of the waterbody (Qiu et al. 2007). The characteristics data, however, represent physical and chemical aspects of a lake, river, or estuary. These models vary from simple 1-dimensional box models to complex 3-dimensional models capable of

simulating water movement, salinity, temperature, sediment transport, and water quality (Easton et al. 2008).

1.4.1 Physical Domain

The physical domain likely is the most essential factor when it comes to the selection of the physical domain. The physical domain center on the effort of modeling — most often either the receiving water itself or a combination of the contributing watershed and the receiving water (Kumar et al. 2009). The model domain selection process is depended on what is wanted and the conditions when the stream displays impairment. For a stream dominated by point source inputs that exhibits impairments under only low flow conditions, a steady-state approach is typically used. This method centers on only the input and receiving of water process during the conditions set forth by the participant.

For water bodies affected additionally or primarily by rainfall-driven flow and pollutant contributions, or by sediment oxygen demand, a dynamic approach is recommended. Dynamic models consider time-variable nonpoint source contributions from a watershed surface or subsurface, or throughout the water column of a receiving water body. Some models consider monthly or seasonal variability, while others enable assessment of conditions immediately before, during, and after individual rainfall events (Golden et al. 2012).

1.4.2 Source Contributions

Primary sources of pollution to a waterbody must be considered in the model selection process. Accurately representing contributions from permitted point sources and nonpoint source contributions from urban, agricultural, and natural areas is critical in properly representing the system and ultimately evaluating potential load reduction scenarios.

1.4.3 Constituents

Another important consideration in model selection and application is choosing appropriate constituents to simulate. Choice of state variables is a critical part of model implementation. The more state variables included, the more difficult the model will be to implement and calibrate. However, if key state variables are omitted from the simulation, the model might not simulate all necessary aspects of the system and might produce unrealistic results. A delicate balance must be met between minimal constituent simulation and maximum applicability.

1.4.4 Model Selection

Establishing the relationship between water quality targets and source loading is a critical component of watershed modeling (Lin et al. 2010; Zheng et al. 2011). It allows for the evaluation of management options that will achieve the desired source load reductions. The link can be established through a number of techniques, ranging from qualitative assumptions based on sound scientific principles to sophisticated modeling techniques. The Environmental Fluids Dynamic Code (EFDC), a general-purpose modeling package for simulating one- or multi-dimensional flow, transport, and bio-geochemical processes

in surface water systems including rivers, lakes, estuaries, reservoirs, wetlands, and coastal regions, is popularly utilized (Peng et al. 2011; Wu and Xu 2011). In addition to hydrodynamic, salinity, and temperature transport simulation capabilities, EFDC is capable of simulating cohesive and non-cohesive sediment transport, near field and far field discharge dilution from multiple sources, eutrophication processes, the transport and fate of toxic contaminants in the water and sediment phases, and the transport and fate of various life stages of finfish and shellfish (Liu et al. 2008).

The structure of the EFDC model includes four major modules: (1) a hydrodynamic model, (2) a water quality model, (3) a sediment transport model, and (4) a toxics model. The EFDC hydrodynamic model is composed of six transport modules including dynamics, dye, temperature, salinity, near field plume, and a tracer module which simulates the movement of neutrally buoyant drifters released in each three-dimensional model cell at specified time sequences. EFDC has been extensively tested, documented, and applied to environmental studies world-wide (Jeong et al. 2010; Liu and Huang 2008).

1.4.5 Land Use Representation

The land use representation provides the basis for distributing soils and pollutant loading characteristics throughout the basin. The watershed model requires a basis for distributing hydrologic and pollutant loading parameters to appropriately represent hydrologic variability throughout the basin, which is influenced by land surface and subsurface characteristics. It is also necessary to represent variability in pollutant loading, which is

highly correlated to land practices. The basis for this distribution was provided by land use coverage of the entire watershed. Although the multiple categories in the land use coverage provide much detail regarding spatial representation of land practices in the watershed, such resolution is unnecessary for watershed modeling if many of the categories share hydrologic or pollutant loading characteristics. Therefore, many land use categories were grouped into similar classifications, resulting in a subset of 4 categories for modeling (Doe et al. 1996; Lin et al. 2007). These land use data provide a foundation upon which the significance of nonpoint sources can be estimated.

1.4.6 Pollutant Representation

Based on a review of previous studies, impairment source information, and water quality monitoring data, possible nonpoint sources include agriculture, urban/residential areas, derelict lands, and illicit marina discharges (Lee et al. 2010). There are currently no traditional permitted point sources in the watershed. The primary pollutant represented in the watershed model to estimate oxygen-demanding substance loadings is BOD. Loading processes for pollutants were represented for each land unit. These modules simulate the accumulation of BOD on the land surface and removal during overland flow, which is simulated as being removed at a related volume of water flowing over the land surface (Mishra et al. 2009). Although atmospheric deposition may be an issue, it was not explicitly simulated in the watershed model. It was, however, represented implicitly in the model through use of the land use- and pollutant-specific accumulation rates.

1.4.7 Meteorological Representation

Meteorological data are a critical component of the watershed model, and appropriate representation of precipitation, wind speed, potential evapotranspiration, cloud cover, temperature, and dew point are required to develop a valid modeling system (Khalili et al. 2011). These data provide necessary input to the algorithms for hydrologic and water quality representation. Meteorological data have been accessed from a number of sources in an effort to develop the most representative dataset for the watershed (Shirinian-Orlando and Uchrin 2007).

1.4.8 Waterbody Representation

Waterbody representation refers to simulate flow and pollutant transport through streams and rivers.

1.4.9 Water Quality Model Structure

Dissolved oxygen concentration is governed by various physical, chemical, and biological factors such as BOD and benthic oxygen dynamics (vanderPerk 1997). Wind can impact the reaeration rate which controls the exchange of oxygen between atmosphere and water. High BOD usually causes low DO because of the oxygen demand used by bacteria to decompose the organic matter (Varis et al. 1994).

1.4.10 Initial and Boundary Conditions

For the dynamic model, initial conditions provide a starting point for the model to march forward through time. The major boundary conditions for the model consist of

upstream/lateral boundary conditions, surface boundary conditions, and open boundary conditions. The upstream/lateral boundary conditions include the freshwater inflow and the associated temperature, DO, and BOD.

CHAPTER TWO

INTRODUCTION

St. Marks National Wildlife Refuge was established in 1931 to provide wintering habitat for migratory birds. Encompassing 68,000 acres spread out between Wakulla, Jefferson, and Taylor counties along the Gulf Coast of northwest Florida, St. Marks National Wildlife Refuge is one of the oldest refuges in the National Wildlife Refuge System and is also home to a diverse community of plant and animal life (Baird et al. 1998). The topography is relatively flat, with soils and woodlands interspersed with ponds and sawgrass sloughs (Figure 2). Elevations range from the open water of Apalachee Bay and the barrier beaches to approximately 30 feet about mean sea level. In addition, there are seven rivers and numerous creeks crossing the refuge. The climate is moderate, with temperatures ranging from 21 to 96 degrees, and with an annual rainfall of 55 inches. The major components of St. Marks National Wildlife Refuge are coastal marshes, characterized by shrubs and grasses (Figure 3). The coastal marshes also provide other important services such as flood control, water quality improvement, shore erosion protection besides wildlife habitat (Oviatt et al. 1977). Water is the essential element that dominates marsh habitats. Marshes and other wetland ecosystems undergo cyclical periods of flooding, drought, and deterioration during a five to twenty year cycle. Water levels in marsh environments are continuously fluctuating and individual marshes have unique water level patterns (Price 1990). Marshes are generally wetter in the summer and drier in the winter, although water levels generally are higher in northern Florida because of greater rainfall.

Figure 2. St. Marks National Wildlife Refuge

Figure 3. Shrubs and Grasses of St. Marks National Wildlife Refuge

Water quality is an important factor in nutrient and detrital distribution in coastal marshes, which is regulated by the Clean Water Act and the United States Environmental Protection Agency. The adequacy of techniques for water quality monitoring of coastal marshes has received a great deal of attention in recent years because coastal marshes are suspected of being degraded by changes to their natural water sources, pollutants and sediment from runoff, and encroachment by invasive species (Reed 1995). Effective coastal marsh management (including source water protection) requires detailed knowledge of the factors affecting water quality indicators. Nutrient levels currently are the most important considerations because they are related to eutrophication (Martin et al. 2009). Fertilizers, septic effluent leachate from coastal development, animal wastes, and atmospheric deposition are all sources of nitrate loading. Most of the water comes from seawater in the lower salt marsh, but the upper marsh is often heavily influenced either by abundant freshwater from the land drainage and flooding or the occasional catastrophic storms that floods the entire marsh with seawater.

2.1 Objectives

If the water quality of coastal marshes is affected by outside factors, the study of certain indicators will lead to the prevention and if necessary the proper action in the case of an epidemic. The purpose of this study is to evaluate water quality of coastal marshes at St. Marks National Wildlife Refuge in terms of coliform and chlorophyll *a*, along with along with dissolved oxygen, temperature, pH, conductivity, and total suspended solid (TSS). St. Marks National Wildlife Refuge was sampled from January 2009 through June 2009. Initial data analysis revealed an interesting disparity in the different water quality

indicators. In coastal lagoons, a spatial and temporal sequence in the composition, abundance and biomass of the microalgal communities is commonly observed (Baird et al. 1998), due to the occurrence of extreme abiotic gradients over the year (Midwood and Chow-Fraser 2012). Depending on the relative abundance of the major algal assemblages in wetlands (phytoplankton, epiphyton, epipelon and metaphyton), chlorophyll measurements may provide a useful estimate of algal biomass (Dubois et al. 2012). Biomass estimation could potentially be useful to assist with biofuels. For this research, chlorophyll *a*, a type of chlorophyll that is most common and predominant in all oxygen-evolving photosynthetic organisms such as higher plants, red and green algae was also monitored. Coliform and chlorophyll *a* were further correlated to water physical and chemical parameters based on synthetically correlative principles.

CHAPTER THREE

MATERIALS AND METHODS

The study site of this research, St. Marks National Wildlife Refuge, is located 25 miles south of Tallahassee along the Gulf Coast of Florida. The easternmost of the refuge, the St. Marks Unit, is generally bounded by the Aucilla River to the east and by the Wakulla and St. Marks Rivers to the west. The Wakulla Unit of the refuge runs west from the Wakulla and St. Marks Rivers to Spring Creek Hwy. The westernmost section, or the Panacea Unit, extends west from Spring Creek Hwy to the Ochlockonee River and Ochlockonee Bay.

3.1 Sampling

From January 2009 through June 2009, the St. Marks Unit was sampled along the coast and the sampled water was analyzed for physical and chemical properties such as temperature, pH (pH Meter), turbidity (Hach Turbidity Meter), conductivity (Conductivity Meter), dissolved oxygen (DO) (DO Meter), total suspended solids (TSS) (Standard Method) (Clescerl et al. 1998), COD (Standard Method) (Clescerl et al. 1998), BOD₅ (Standard Method) (Clescerl et al. 1998), total Kjeldahl nitrogen (Standard Method) (Clescerl et al. 1998), total phosphorous (Standard Method) (Clescerl et al. 1998), coliform and chlorophyll *a* (Figure 4). The membrane filtration method was used to analyze fecal coliform (Method 9222D) (Clescerl et al. 1998). Specifically, 100 ml of water samples was aseptically filtered on cellulose nitrate filtration membranes (0.45 µm pore size, Whatman, UK). The membranes were then placed on m-FC Agar plates with

rosolic acid (BD, Franklin Lakes, NJ) and incubated at 44.5°C for 24 hr. Colonies of fecal coli forms were counted and recorded as colony-forming units (CFU) per 100 ml. To measure chlorophyll *a*, a volume of water from a particular depth is filtered through a fine glass-fiber filter to collect all of the particulate material greater than about 1 micron (1/1000th of a millimeter) in size. The chlorophyll-a in this material was extracted with a solvent (acetone or alcohol) and was measured using chlorophyll optical kit (P/N 7000-961) and a TD-20/20 Luminometer against liquid primary chlorophyll *a* standard (P/N 10-850) (Turner Designs, Sunnyvale, CA).

Figure 4. Sampling at St. Marks National Wildlife Refuge

3.2 Diffusion

In the Refuge, chlorophyll *a* is able to diffuse. The effective diffusion coefficient of chlorophyll *a* was measured in the lab using a diffusion coefficient in distilled water. A

known concentration of chlorophyll *a* is placed in a diffusion cell immersed in distilled water. A magnetic stirrer was used and the chlorophyll *a* concentration was monitored during the progress of diffusion over time. This process was also observed using a Hamamatsu high sensitivity monochrome digital camera C4505-50 in a dark room (Type U-1) (Figure 6), with the sample being placed in a Petri dish with native soil. Images from the camera are attached in **Appendix A**. The chlorophyll *a* concentration was fitted in the following equation (Fukuda and Kawai 1995):

$$\frac{\partial C}{\partial t} = D_{\text{eff}} \frac{\partial^2 C}{\partial r^2} \quad (1)$$

where *C* (mmol/L) is the concentration of chlorophyll *a* in solution, *t* is the time, *D_{eff}* is the effective diffusion coefficient and *r* is the radius of the diffusion process. The initial and boundary conditions as follows:

$$C(r, t = 0) = \begin{cases} C_0 & \text{for } r = 0 \\ 0 & \text{for } r > 0 \end{cases} \quad (2)$$

$$\left. \frac{\partial C}{\partial z} \right|_{z=L} = 0 \quad (3)$$

where *C₀* is initial put chlorophyll *a* concentration, *L* is the radius of the reactor. The analytical solution of the diffusion equation is given as:

$$C(x, t) = \frac{M}{\sqrt{4\pi Dt}} \exp\left(-\frac{x^2}{4Dt}\right) \quad (4)$$

The *D_{eff}* was then estimated from by fitting the experimental data to above equation by the least-square minimization using Powell's method (Li et al. 2011). The effect of

temperature on the diffusion coefficient was theoretically estimated based on the Stokes–Einstein relation (Pottier 2005):

$$D(T) = \frac{(273.15 + T)\mu(T_0)}{(273.15 + T_0)\mu(T)} D(T_0) \quad (5)$$

where $D(T)$ and $D(T_0)$ are diffusion coefficients at temperatures T and T_0 ($^{\circ}\text{C}$), respectively, and $\mu(T_0)$ and $\mu(T)$ are dynamic viscosities of water at temperatures T and T_0 , respectively. The structure of chlorophyll *a* is illustrated in Figure 5.

Figure 5. Molecular Structure of Chlorophyll *a*

3.3 Pearson's Correlation

The correlation of chlorophyll *a* with orthophosphate and TKN was described by the Pearson product-moment correlation coefficient, or “Pearson's correlation”, which was obtained by dividing the covariance of the two variables by the product of their standard deviations:

$$\rho_{x,y} = \frac{\sum_{i=1}^n (x_i - \bar{x})(y_i - \bar{y})}{(n-1)S_x S_y} = \frac{\sum_{i=1}^n (x_i - \bar{x})(y_i - \bar{y})}{\sqrt{\sum_{i=1}^n (x_i - \bar{x})^2 \sum_{i=1}^n (y_i - \bar{y})^2}} \quad (6)$$

where $\bar{x}$ and $\bar{y}$ are the sample means of X and Y and S_x and S_y are the sample standard deviations of X and Y.

Figure 6 Hamamatsu Camera C4505-50

3.4 Chlorophyll *a* Adsorption

To investigate Chlorophyll *a* adsorption, the water sample was introduced to a column filled with goethite-coated silica sand. Goethite, a frequent and abundant form of iron oxide in the soil and sediments, is an important component influencing Chlorophyll *a* adsorption in natural aquatic environments or in the soil. Goethite has been extensively used in adsorption studies, in part because it is stable and can be easily produced in the laboratory. Goethite was prepared as described by Schwertmann et al. (Schwertmann et al. 1985). Briefly, 1.0M ferric nitrate was mixed with 1.0M KOH (1:9, v/v) and aged for 21 days at 25°C. This suspension was then washed extensively with de-ionized water via centrifugation. The rinsed solid was re-suspended in 0.4M HCl. After washed and dialyzed against de-ionized water, it was freeze dried to obtain crystalline goethite. Thus obtained goethite was then coated on silica sand following the method of Schwertmann et al. (Schwertmann et al. 1985). Specifically, goethite were mixed with silica sand (1:5, w/w) in 0.01M NaNO₃ solution (pH 7.5) and shaken for 48 hrs. Coated silica sand was then washed with 0.1M NaNO₃ (pH 7.0) via centrifugation. After rinsed with de-ionized water, coated silica sand was oven-dried at 110°C. Goethite coating was determined by dissolving coated silica sand in HNO₃ (95%) and HF (40%) (2:1, v/v). Specific surface areas of goethite-coated silica sand were also measured by the surface area analyzer was determined to be 318 m²/g.

Column experiments were conducted using an acrylic column (5 cm × 15 cm, Kimble-Kontes, Vineland, NJ) to investigate Chlorophyll *a* adsorption on the goethite-coated silica sand. The column was oriented vertically and sealed at the bottom with a custom

frit to permit the flow of water and retain the media. Goethite-coated silica sand was packed in the column through CO₂ solvation to eliminate air pockets. Prior to starting each experiment, approximately 100 pore volumes of nano-pure de-ionized water was eluted through the column by a peristaltic pump to stabilize the column. Before the introduction of water sample into the column, a conservative tracer of chloride was introduced to the column to estimate the porous media properties. After the tracer study, two pore volume of water sample was pumped into the column at a flow rate of 0.56 ml/min. The column was then flushed with nano-pure de-ionized water alone for up to 50 pore volumes until no Chlorophyll *a* could be detected in the elution. Elution was collected by a fraction collector and was measured for Chlorophyll *a*. A breakthrough curve was generated and mass balance analysis was performed. For the column experiment, three runs were performed, and the inconsistency of breakthrough curves was within 5% (95% CI). As a control experiment, transport of Chlorophyll *a* in uncoated silica sand was also conducted.

Figure 7. Column Experiment Setup of Chlorophyll *a* Adsorption

Under saturated conditions, Chlorophyll *a* transport in goethite-coated silica sand is controlled by both equilibrium adsorption and kinetic deposition, which can be described by (Saiers et al. 1994) [advection-dispersion equation]:

$$(1 + \frac{\rho_b(1-\theta)}{\theta} K_d) \frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial x^2} - v \frac{\partial C}{\partial x} - \mu C \quad (7)$$

where C is the Chlorophyll *a* concentration in the solution (g/m^3); t is the elapsed time (sec); ρ_b is the sediment bulk density (g/m^3); K_d is the partitioning coefficient of Chlorophyll *a* between the aqueous phase and porous media (m^3/g); θ is the porosity of the porous media (m^3/m^3); D is the longitudinal dispersion coefficient (m^2/sec); x is the coordinate parallel to the flow (m); v is the pore velocity (m/sec); and μ is the first order Chlorophyll *a* deposition coefficient on goethite-coated silica sand (sec^{-1}).

$[1 + \frac{\rho_b(1-\theta)}{\theta} K_d]$ is defined as the retardation factor, R , which is an indicator of the “lag” of phosphorus transport due to reversible adsorption.

Transport parameters in equation (1) were obtained by fitting the experimentally obtained Chlorophyll *a* breakthrough data using an implicit, finite-difference scheme. All the parameters were optimized by minimizing the sum of squared differences between observed and fitted concentrations using the nonlinear least-square method.

CHAPTER FOUR

RESULTS

4.1 Water Quality Parameters

The diversity and abundance of biological communities are determined by the physical environment. In this research, the water quality parameters including physical, chemical and biological factors of the environment such as water temperature, turbidity, conductivity, pH, dissolved oxygen, TSS, TKN, coliform and chlorophyll *a* were monitored. Physical and chemical data can be commonly utilized as indicators to evaluate the extent and severity of any environmental stressors, which may impact the biological communities. Most of these factors were related to the temperature. Throughout the course of monitoring, the temperature was in the range from 38°F to 86°F (Figure 8).

Figure 8. Water Temperature of the Experimental Period

Dissolved oxygen was obviously impacted by the temperature, which ranged 3.89 mg/l to 7.6 mg/l during the monitoring period. Usually, high dissolved oxygen values indicated healthy and stable environments, which can support a diversity of aquatic organisms. The pH range of the refuge during the sapling period was in the range of 7.1 to 8.25 (Figure 9). pH is a measure of the hydrogen ion concentration, or acidity of water. Therefore, the pH was in the neutral range.

Figure 9. Dissolved Oxygen and pH of the Experimental Period

Turbid resulted from soil erosion, urban runoff, algal blooms and bottom sediment disturbances. At high levels of turbidity, water becomes warmer as suspended particles absorb heat from the sun. There was a general trend that turbidity increased with the increase of water temperature. Dissolved oxygen also decreased with the increase of turbidity. Turbidity led to a decreased light penetration, thereby decreasing photosynthesis, which in turn reduced dissolved oxygen concentrations. High

temperatures, which may result from high turbidity, decrease the amount of gases that can be dissolved in water. There was a general trend that turbidity increased during the sampling period, during which water temperature increased accordingly (Figure 10).

Figure 10. Turbidity of the Experimental Period

BOD levels were typically less than 5 mg/l during the sampling period, indicating generally low levels of organic loading. Total suspended solids showed some peak values up to 12 mg/l during the sampling period, following precipitation events (Figure 11). The coliform up to 10 CFU was observed. The most important contributing factor to the microbial numbers was perhaps the wildlife sources, with flocculation coincident with the flooding event.

Figure 11. Coliform and Total Suspended Solid of the Experimental Period

Salinity is the total of all salts dissolved in water. The salt content of water affects the distribution of plant and animal communities in an aquatic system, based on the amount of salt they can tolerate. Salinity changes daily with the tides or seasonally with the changing environmental conditions. Salinity can also decrease during major storms events that result in a lot of precipitation such as a hurricane. Salinity can be represented by conductivity. In this research, conductivity was generally high, with an average value exceeding 200 micromhos (umho/cm).

Figure 12. Conductivity of the Experimental Period

Chlorophyll *a*, the green pigment that is responsible for a plant's ability to convert sunlight into the chemical energy needed to fix CO₂ into carbohydrates, provides a reasonable estimate of algal biomass. Chlorophyll *a* in this research was in the range of 2 µg/l to 6 µg/l and varied dramatically during the experimental period (Figure 13). TKN is the total concentration of organic nitrogen and ammonia. At St. Marks National Wildlife Refuge, the TKN was around 1 mg/l. The TKN variation with respect to the date was not significant.

Figure 13. Chlorophyll *a* and TKN of the Experimental Period

4.2 Chlorophyll *a* Diffusion Coefficient

To estimate the Chlorophyll *a* diffusion coefficient, the chlorophyll *a* concentration was fitted in equation (1) to estimate the effective diffusion coefficient D_{eff} . The analysis of the diffusion problem indicated that the characteristic time scale was in the order of a few minutes. Based on this, only the first 2 min of data were used to estimate the diffusion coefficient.

By and large the diffusion process could be well described with equation (1) and the estimated D_{eff} of chlorophyll *a* was in the order of $10^{-9} \text{ m}^2/\text{s}$. The D_{eff} of chlorophyll *a* increased with increasing temperature. The observed temperature effect on the diffusion coefficient was consistent with what would be expected based on equation (5).

Figure 14. Hamamatsu C4505-50 Showing Chlorophyll Diffusion

4.3 Chlorophyll *a* Sorption on goethite-Coated Silica Sand

Tracer (nitrate) transport was studied before Chlorophyll *a* transport experiments in both uncoated silica sand and goethite-coated silica sand. For both cases, nearly all the input tracer was eluted from the column. The tracer breakthrough curve was characterized by a breakthrough front and an elution tail. The lasting tail of the tracer breakthrough curve indicated possible retardation of nitrate in the column. The tracer breakthrough curve was simulated with equation (7). During the model simulation, the retardation factor was set to 1.0, i.e., $K_d = 0$. This was based on the consideration that the tracer should not be retarded in the media as the tracer was assumed not to adsorb in the media. In addition, the deposition coefficient μ was set to zero, i.e., no retention of the tracer in the media. This was true since nearly all the inputted tracer was eluted from the column at the end of the transport experiments. During the simulation, the velocity was fixed at 0.058 cm/min

and the initial D was set as $8.00 \text{ cm}^2/\text{min}$ for both uncoated and goethite-coated silica sand. After the simulation, D was determined to be $9.96 \text{ cm}^2/\text{min}$ for uncoated silica sand and $11.8 \text{ cm}^2/\text{min}$ for goethite-coated silica sand. These D values were then used for the simulations of Chlorophyll a transport in the corresponding media. From the tracer study, the porous media porosity was found to be 0.49 for uncoated silica sand and 0.47 for goethite-coated silica sand.

Chlorophyll a breakthrough curves were fitted well with the equation (7). The accuracy of Chlorophyll a transport modeling was expressed by the sum of the squared differences between observed and fitted concentrations. The mean square for error of nitrate transport was 0.976 and the mean square for error of Chlorophyll a transport was in the range from 0.890 to 0.990. There was a small retardation and minimal deposition when Chlorophyll a was transported in uncoated silica sand ($R = 1.21$ and $\mu = 1 \times 10^{-7} \text{ min}^{-1}$) (Figure 15). When transported in goethite-coated silica sand, however, more Chlorophyll a was retarded as evidenced by the delayed breakthrough front ($R = 3.24$). There was also Chlorophyll a retention in the column as the peak value of the breakthrough curve was much lower than that of the uncoated silica sand. Chlorophyll a transport breakthrough curve also had a long elution tail. The long-lasting tails of the breakthrough curve indicated kinetic-controlled Chlorophyll a deposition in the column. Chlorophyll a adsorption in goethite-coated silica sand was thought to occur owing to the replacement of OH- groups with Chlorophyll a on the goethite surface. Mass removal was estimated.

Figure 15. Chlorophyll *a* Transport in Uncoated and Goethite-Coated Silica Sand

CHAPTER FIVE

DISCUSSION

5.1 Water Quality Parameter Correlations

Apalachee Bay is home to bottlenose dolphins, brown pelicans, wintering redhead ducks, and a rich diversity of marine life. Another vital habitat are the hardwood swamps, which support wood ducks, night herons, black bears, otter, and raccoons, etc. The extensive pine woodlands offer food and cover for turkeys, white-tailed deer, bluebirds, fox squirrels, gopher tortoises, Bachman's sparrows, red-cockaded woodpeckers, pine warblers, and many more species. Located in Apalachee Bay, St. Marks is well-known as an oasis of different habitats for wildlife. Natural salt marshes, tidal flats, and freshwater impoundments in St. Marks National Wildlife Refuge attract thousands of waterfowl, shorebirds, wading birds and other animals (Baird et al. 1998). The open marshes and swamps also provide a home to over 2,000 American alligators. In addition, the salt marshes are a valuable nursery area for fish, shrimp, and shellfish and provide protection during storms to shorebirds and other migrants (Sabat 2000; Wieski et al. 2010). Therefore, St. Marks National Wildlife Refuge contains a diverse variety of environmentally sensitive habitats that provide food, cover, and protection for many species of wildlife. Of the more than 300 species of birds recorded on the refuge, 98 species nest here. Some 19 species of ducks and two species of geese may be seen from mid-November through January. Refuge inland hardwood swamps also support other birds as well as a broad range of mammals including the Florida Black Bear, white-tailed deer, otter and raccoon.

The correlations of physical, chemical and biological water quality parameters were explored using regression analysis. Dissolved oxygen decreased with the increase of temperature (Figure 16).

Figure 16. Dissolved Oxygen as a Function of Temperature

Based on data presented in Figure 16, Pearson correlation between oxygen and temperature was obtained to be -0.640. The negative value indicated that dissolved oxygen had an inverse relationship with the temperature. The simulation also showed that the correlation was significant at the 0.01 level.

Table 1. Correlations of Dissolved Oxygen with Temperature

Correlations			
		Temperature	DO
Temperature	Pearson Correlation	1	-.640**
	Sig. (2-tailed)		.001
	N	25	25
DO	Pearson Correlation	-.640**	1
	Sig. (2-tailed)	.001	
	N	25	25

** . Correlation is significant at the 0.01 level (2-tailed).

TKN and Chlorophyll *a* also showed correlation with temperature, i.e., both increased with the increase of temperature (Figure 11 and Figure 18). Phytoplankton growth is dependent mainly on adequate light intensity and nutrients availability. During the summer, with the increase of temperature, phytoplankton growth was more obvious, resulting in increased chlorophyll *a* observation at St. Marks National Wildlife Refuge.

Figure 17. TKN as a Function of Temperature

Figure 18. Chlorophyll *a* as a Function of Temperature

The decomposition of dead plants and animals and the fecal waste from living animals release nitrate into the aquatic system. Nitrogenous compounds may be toxic to aquatic organisms (Paungfoo et al. 2007). Nitrogen content influenced the chlorophyll *a* and consequently the plant functions. Plant leaves exhibit a structural and functional acclimation of the photosynthetic apparatus to the light intensity experienced during their growth (Argenta et al. 2003). Nitrogen supply has large effect on leaf growth because it increases the leaf area of plants and, on that way, it influences on photosynthesis. Photosynthetic proteins represent a large proportion to total leaf nitrogen (Shadchina and Dmitrieva 1995). Chlorophyll *a* content is approximately proportional to leaf nitrogen content. As demonstrated in this research, chlorophyll *a* had a correlation with TKN (Figure 19). The Pearson correlation between chlorophyll *a* and TKN was found to be 0.204 (Table 2). Since the Sig (2-tailed) value was 0.328, which was greater than 0.05,

the variability was about the same. This meant that the variability of chlorophyll *a* and TKN was not significantly different. One might expect to see a positive proportional relationship between TKN and chlorophyll *a*.

Figure 19. Chlorophyll *a* as a Function of TKN

Table 2. Correlations of Chlorophyll *a* with TKN

Correlations		TKN	Chlorophyll a
TKN	Pearson Correlation	1	.204
	Sig. (2-tailed)		.328
	N	25	25
Chlorophyll a	Pearson Correlation	.204	1
	Sig. (2-tailed)	.328	
	N	25	25

Chlorophyll *a* was also a function of dissolved oxygen. As shown in Figure 20, chlorophyll *a* concentration increased with the increase of dissolved oxygen until 4.5

mg/l, after which chlorophyll *a* concentration decreased with the increase of dissolved oxygen. Another possibility is excessive richness of nutrients, also known as eutrophication.

Figure 20. Chlorophyll *a* as a Function of Dissolved Oxygen

Coliform levels were closely linked with variation in turbidity (Figure 21). Elevated coliform (e.g., *Escherichia coli*) bacterial concentrations indicated increased animal waste contamination potential and associated risk of pathogen exposure. The Pearson correlation between coliform and turbidity was found to be 0.633 (Table 3). Since the Sig (2-tailed) value was 0.001, which was greater than 0.05, the variability was about the same. This meant that the variability of coliform and turbidity was not significantly different. The direct correlation between turbidity and coliform indicated that as turbidity increased, so did coliform bacteria. The result was significant in obtaining an accurate estimate for coliform before waiting the 24 hour incubation time the procedure required.

Figure 21. Coliform as a Function of Turbidity

Table 3. Correlations of Coliform with Turbidity

Correlations		coli form	Turbidity
coli form	Pearson Correlation	1	.633**
	Sig. (2-tailed)		.001
	N	25	25
Turbidity	Pearson Correlation	.633**	1
	Sig. (2-tailed)	.001	
	N	25	25

** . Correlation is significant at the 0.01 level (2-tailed).

Chlorophyll *a* showed no direct correlation with turbidity (Figure 21). However, $d[\text{Chlorophyll } a]/d[\text{Turbidity}]$ displayed a linear relationship with turbidity (Figure 22), indicating the correlation of chlorophyll *a* variation with respect to turbidity.

Figure 22a. Chlorophyll *a* as a Function of Turbidity

Figure 22b. $d[\text{Chlorophyll } a]/d[\text{Turbidity}]$ as a Function of Turbidity

Conductivity increased linearly with the increase of pH, although the increase was not significantly pronounced (Figure 23). The Pearson correlation between conductivity and pH was 0.266 (Table 4). Again, since the Sig (2-tailed) value was 0.199, which was

greater than 0.05, the variability was about the same. This meant that the variability of conductivity and pH was not significantly different.

Figure 23. Conductivity as a Function of pH

Table 4. Correlation of Conductivity with pH

Correlations			
		Conductivity umhos/cm	pH
Conductivity umhos/cm	Pearson Correlation	1	.266
	Sig. (2-tailed)		.199
	N	25	25
pH	Pearson Correlation	.266	1
	Sig. (2-tailed)	.199	
	N	25	25

Chlorophyll *a* also had a correlation with pH. With the increase of pH, chlorophyll *a* exponentially decreased (Figure 24).

Figure 24. Chlorophyll *a* as a Function of pH

5.2 Chlorophyll *a* Diffusion Coefficient

Chlorophyll *a* measurements have historically provided a useful estimate of algal biomass and its spatial and temporal variability. Measuring chlorophyll concentrations as an indicator of algae biomass may provide one tool to assess coastal water quality and its change over time (Mauromicale et al. 2006). Chlorophyll concentrations can provide guidance for decision making in the context of environmental management. Phytoplankton as chlorophyll-containing organisms is the first step of production in most marine processes and food chains (Argenta et al. 2003). Nutrient enhancement in the seawater due to the discharge of agricultural, industrial, and urban wastes threatens the coastal environment. Increasing concentrations of chlorophyll-*a* in seawater, in reaction to the elevation of nutrient supply can have severely damaging effects on the marine

environment. In this research, seasonal variability of the chlorophyll *a* concentrations was examined using field observations.

Chlorophyll *a* concentrations can act as an indicator of phytoplankton abundance and biomass in the coastal waters. It is natural for levels of chlorophyll *a* to fluctuate over the different seasons (Shadchina and Dmitrieva 1995). Chlorophyll *a* is one of the vital requirements for marine ecosystems. The estimated chlorophyll *a* diffusion coefficient can present useful knowledge about distribution of chlorophyll *a* concentration in the coastal waters. Due to low values of salinity, variation of density is in high agreement with changes of temperature. In this research, the characteristic time scale of chlorophyll *a* was found to be in the order of a few minutes. Therefore, the spreading of chlorophyll should be an issue in coastal wetlands.

5.3 Chlorophyll *a* Adsorption

Chlorophyll adsorption on goethite-coated silica sand was an instantaneous process. As discussed previously, the cations on goethite-coated silica sand should play a more important role in Chlorophyll *a* adsorption. As demonstrated by the column experiments (integration of the breakthrough curve), around 68% of Chlorophyll *a* can be retained through adsorption by goethite. The retention of Chlorophyll *a* on goethite-coated silica sand was attributed to the adsorption since there was no precipitation during the transport experiments.

CHAPTER SIX

CONCLUSIONS

There are numerous sources of fecal coliform bacteria contamination, including wildlife at St. Marks National Wildlife Refuge. A long history of illness outbreaks and epidemics has demonstrated a relationship between the presence of fecal coliform bacteria and the presence of illness-causing viruses and bacteria, which makes coliform estimation very important. At the same time, chlorophyll *a* measurements can provide a useful estimate of algal biomass and its spatial and temporal variability, which can provide guidance for decision making in the context of environmental management. Nutrient enhancement in the seawater due to the discharge of agricultural, industrial, and urban wastes threatens the coastal environment, which can be represented by chlorophyll *a*. Research, water quality parameters at St. Marks National Wildlife Refuge were monitored to predict the correlations of wetland ecosystems. Coliform and chlorophyll *a* showed correlations with temperature, dissolved oxygen, turbidity and pH, etc. The results of this study increase our capability to forecast and assess risk of coastal environment based on basic physical and chemical variations in coastal wetland ecosystems. While some systematic studies have been undertaken, there continues to be a need for long-term water quality monitoring related to the water quality of coastal wetlands. This research could provide important information in the event of an outbreak. Although the study period is relatively short; in the case of an outbreak the conditions of the water quality indicators at that time could be compared during the ones of this sample period.

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BIOGRAPHICAL SKETCH

BACKGROUND

Environmental engineer with 3 years experience in a variety of tasks including environmental remediation, hydraulics and hydrology, permitting, construction oversight, and quality assurance. Experience in managing and conducting a variety of engineering and scientific projects. Have diverse experience and project background for multi-party and multi-discipline projects.

EDUCATION

Florida State University

M.S. in Environmental Engineering, anticipated graduation Fall 2012

Florida State University

B.S. in Environmental Engineering April 2011

Florida State University

B.S. in Criminology Dec. 2008

PROFESSIONAL EXPERIENCE

HANDEX CONSULTING AND REMEDIATION (current employer)

Tallahassee, FL

Staff Engineer – 4/11 to present

Work closely with FDEP for the cleanup of sites contaminated from petroleum storage systems. Work with PE's, write reports, assist with system design and other engineering tasks. Perform field work including groundwater and soil sampling, site assessment, and construction oversight.

Florida State University (current employer)

Tallahassee, FL

Teaching Assistant – 8/10 to present

Perform teaching assistant activities such as grading homework assignments and projects, creating handouts for the class, offering office hours to help students, and proctoring exams.

Classes for TA: Geotechnical Design, Hydraulics, Hydraulic Engineering, AutoCAD.

GEOSYNTEC CONSULTANTS

Jacksonville, FL

Engineering Intern – 3/11 to 3/12

Perform vapor study on a soil vapor extraction system cleaning up dry cleaning solvent PCE. Learned how to operate mechanical remediation systems. Worked under PE.

FLORIDA DEPARTMENT OF ENVIRONMENTAL PROTECTION

Tallahassee, FL 32399

Engineering Specialist I – 4/10 to 4/11

Responsibilities include working directly under a Professional Engineer in the department of Waste Management, in the Bureau of Petroleum Storage Systems. Track over 750 systems in the state of Florida that remediate sites contaminated with hydrocarbons. Including air sparge, soil vapor extraction, chemical injection, and others.

MEMBERSHIPS/CERTIFICATIONS/AWARDS

Florida Water Environment Association, FAMU/FSU Chapter, Co-Founder

Tau Beta Pi, Engineering Honor Society, Member

American Society of Civil Engineers, Member

Engineers Without Borders, Member

Florida Bright Futures

Seminole Torchbearers, Member

American Society of Microbiology – Southeastern Conference 2010, 2011 – Travel Award & Participant