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Agricultural nonpoint source pollution management: Water quality impacts of Balm Road Treatment Marsh, Hillsborough County, Florida

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Agricultural Nonpoint Source Pollution Management: Water Quality Impacts of Balm Road Treatment Marsh, Hillsborough County, Florida

by

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ABSTRACT

Balm Road Treatment Marsh is a 12 ha constructed wetland treatment system in south-central Hillsborough County, Florida created to improve water quality in Bullfrog Creek and ultimately Tampa Bay. The treatment system was designed to treat runoff from approximately 741 ha of upstream agricultural land prior to discharging into the creek, with the primary goals of reducing sediment and nutrient loads. Water quality data from four sites on Bullfrog Creek were analyzed to determine impacts to ambient water quality and pollutant load reductions downstream. Results were compared to the performance of other wetlands to treat both nonpoint and point source pollution. Impacts to ambient water quality in the creek were found to be minimal, if any, and although significant load reductions were found downstream, they could not be attributed to wetland treatment affects with confidence. In general, nonpoint source pollution, particularly from agriculture, was found to be treated less effectively than point sources. The importance of monitoring the performance of stormwater projects while employing a strategic sample design and including receiving water impacts is highlighted.

Chapter 1

Introduction

Balm Road Treatment Marsh is 12 ha constructed surface-flow wetland system in south-central Hillsborough County, Florida created to improve water quality in Bullfrog Creek and ultimately Tampa Bay (Figure 1). The treatment system is located near the headwaters of Bullfrog Creek, which has been partially diverted to flow through the wetland along with any overland runoff from the upper parts of the watershed. Bullfrog Creek then empties into Tampa Bay approximately 32 km downstream. The treatment system was designed to treat runoff from approximately 741 ha of upstream agricultural land prior to discharging into the creek, with the primary goal of reducing sediment and nutrient loading to Tampa Bay while improving water quality in Bullfrog Creek (Figure 2). The system was constructed in 2004 through a joint effort between Hillsborough County and the Southwest Florida Water Management District (SWFWMD). This research uses water quality data from Bullfrog Creek upstream and downstream from the treatment system to examine its affects on the water quality in Bullfrog Creek and loadings to Tampa Bay. The treatment performance of this treatment wetland system is compared to other performance data available in the literature to determine whether constructed wetland treatment systems are a useful tool in managing agricultural nonpoint source pollution.

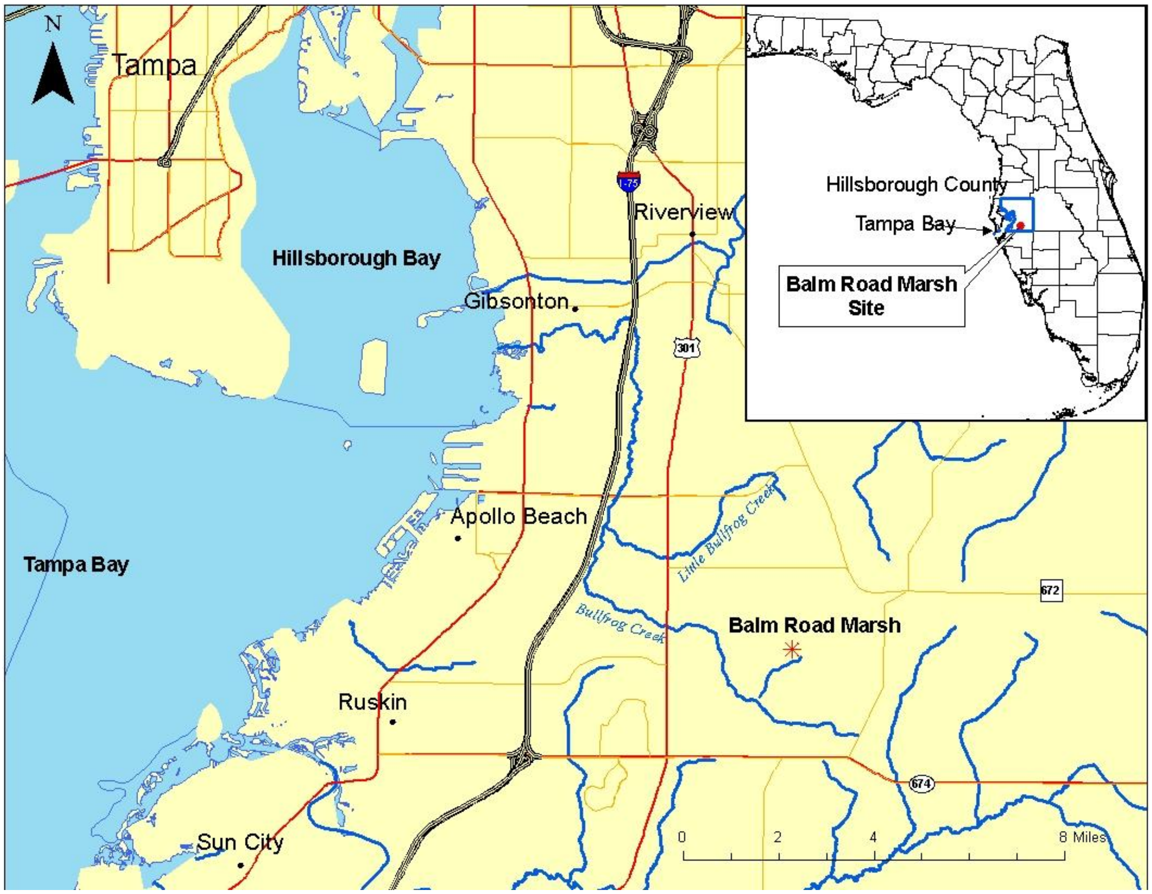

Figure 1. Location of Balm Road Marsh.

This document outlines the research in its relevant context. Background information is presented including the current status of water quality in the United States and Florida. The role of agricultural nonpoint source pollution is discussed along with detailed impacts of nutrients and sediments on water resources. A brief history of related policy, both at national and state levels, is then outlined. Wetlands as pollution treatment systems are discussed including history and processes. To conclude the background section, the design of Balm Road Treatment Marsh is described.

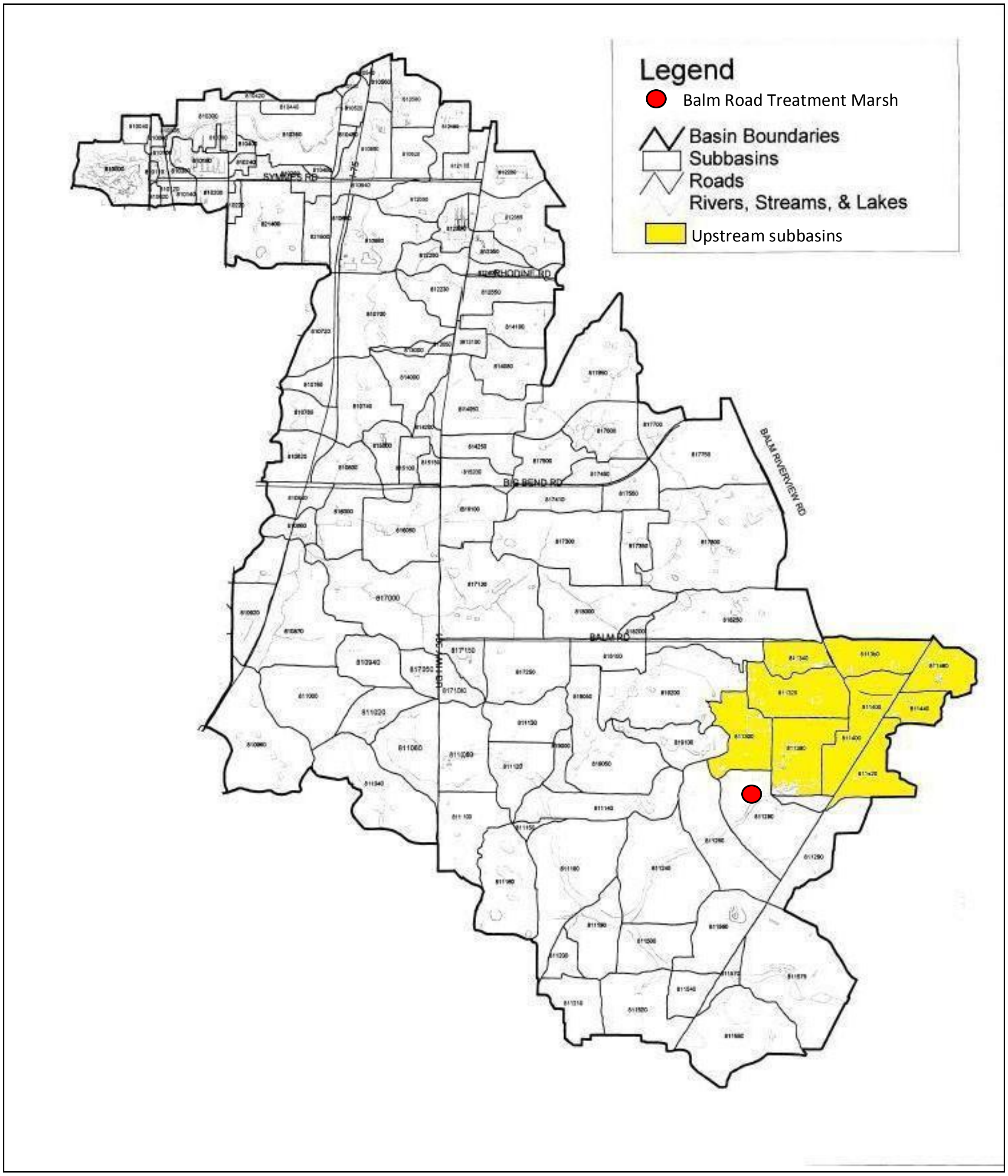

Figure 2. Bullfrog Creek watershed. Subbasins that drain to Balm Road Marsh are highlighted in yellow. Adapted from Dames & Moore, 2000.

A review of the literature as related to constructed treatment wetlands follows. Literature reviewed includes treatment wetland performance investigations, studies determining factors affecting performance, literature on processes and design, and sources for data analysis reference. This, along with the background section, set the framework for the research.

The study purpose is to determine the water quality impacts of Balm Road Treatment Marsh in order to gain a better understanding of the performance of constructed treatment wetlands for agricultural pollution. The specific research questions regarding the treatment system are presented as follows: What are the resulting ambient water quality impacts on Bullfrog Creek? Was there a subsequent pollutant load reduction to Tampa Bay? How does the performance of constructed wetlands used to treat agricultural pollution compare to wetlands used to treat other pollution? The comparisons and questions are intended to help solve the overarching problem of whether or not constructed wetlands are appropriate for agricultural pollution management.

The study area is described including climate, soil, land use, and hydrology. Next, the specific research methods are outlined. This includes sections on sample design and data collection and data analysis. Finally, the results and conclusions are discussed which include the determination of impacts to water quality in Bullfrog Creek, load reductions to Tampa Bay, and the discussion of treatment wetlands as potential management strategies for agricultural nonpoint source pollution.

Chapter 2

Background

Water Quality in the United States and Florida

The United States Environmental Protection Agency (EPA) has reported that approximately 44% of river reaches, 64% of lake area, and 30% of estuarine area assessed do not fully meet their water quality standards (EPA, 2009). The state of Florida reports similar results with 32% of stream reaches, 64% of lake area, and 98% of estuarine area not meeting water quality standards (FDEP, 2008). These numbers can be seen in Figures 2 and 3. Improving surface water quality has been a national goal in the United States since the passage of the Clean Water Act (CWA) in 1972. Although the CWA was largely successful in reducing point source pollution, nonpoint source pollution remains the major cause of water body degradation. Nutrients, sediment, bacteria, metals and oxygen depleting substances have been found to be the most common causes of water body impairment. The leading source of these pollutants is from urban and agricultural runoff, known as nonpoint source pollution (EPA, 2002). In fact, agricultural nonpoint pollution has been identified as the number one source of water quality impairments to streams and lakes in the United States (Parry, 1998; EPA, 2009).

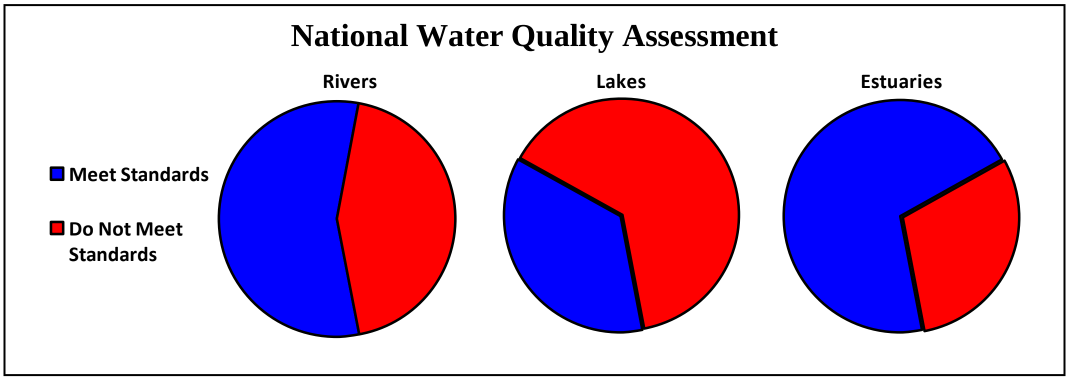

Figure 3. EPA national water quality assessment. Waters that do not meet the standards for their designated uses shown in red (EPA, 2009).

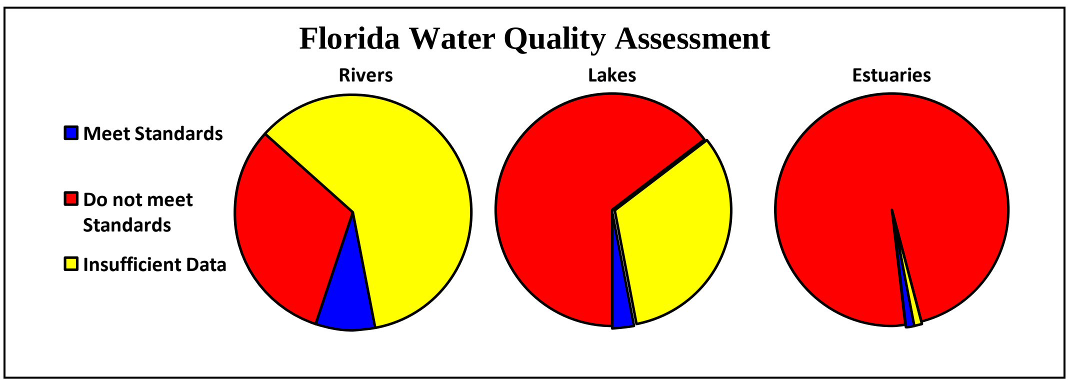

Figure 4. Florida water quality assessment. Waters that do not meet the standards for their designated uses shown in red (FDEP, 2008).

Water Quality in Tampa Bay

Tampa Bay is Florida's largest open water estuary spanning over 1,000 square kilometers. The primary source of nitrogen, the bay's target pollutant of concern, is from urban and agricultural runoff. In fact, nonpoint source pollution accounted for 63% of nitrogen loading to the bay from 1999-2003, nearly half of which is from agricultural lands (TBEP, 2006). Total nitrogen loading to the bay from nonpoint sources for this time period was approximately 2,321 metric tons per year, total phosphorus was 747 metric tons per year, and totals suspended solids was 37,068 metric tons per year (TBEP, 2005). The major contributing basin of concern for the proposed research is the Coastal Hillsborough Bay basin, which includes the Bullfrog Creek basin. The Coastal Hillsborough Bay basin represents only 7.5% of the Tampa Bay watershed area (FDEP, 2001). Estimated loading to this basin for the same five year period was 465 metric tons per year of total nitrogen, which represents approximately 20% of loadings to the Tampa Bay watershed. It was estimated that 50% of the load for this basin was from nonpoint sources (TBEP, 2005).

Agricultural Nonpoint Source Pollution

Approximately 50 to 70% of water bodies assessed have been found to be adversely affected by agricultural nonpoint source pollution (Ritter & Shirmohammadi, 2001). Agricultural runoff carries sediments from erosion resulting from row crops and overgrazing as well as nutrients, primarily nitrogen and phosphorus, originating from fertilizer application. Approximately 30% of phosphorus and 18% of nitrogen applied to agricultural land in the form of fertilizers is utilized in plant production (Isermann, 1991; Carpenter, 1998). The remaining nutrients either runoff to surface water or accumulate in agricultural soils, which may eventually erode and also runoff to surface water. Nitrogen export from agricultural land also occurs through leaching and infiltration which eventually deposits nitrogen to ground and surface waters. Nutrients also accumulate in a similar manner from animal waste and manure (Carpenter *et al*, 1998).

Soil erosion is the source of 99% of the total suspended solids (TSS) loads found in water bodies (Ritter & Shirmohammadi, 2001) and sediments and the pollutants attached to sediments are the most widespread source of pollutants in surface waters of the United States (Gianessi & Peskin, 1989). Sediments affect water bodies by degrading wildlife habitat, decreasing water storage capabilities, and may result in the need for costly dredging activities (Ritter & Shirmohammadi, 2001). Increased sediment loads also interfere with recreational use and cause water clarity problems, decreasing the aesthetic value of water bodies (USDA, 1997). Sediments are also harmful to aquatic organisms, result in temperature changes, and cause oxygen depletion. Effects on benthic invertebrates and algae populations vary from reduced growth rates to mortality (Hynes, 1970; Newcombe & MacDonald, 1991). Increased suspended sediment loads can cause a reduction in fish growth rate and disease resistance, modify migration patterns, reduce the number of organisms available for fish to feed on, interfere with fishing activities, and can be lethal at higher concentrations (Newcombe & MacDonald, 1991).

In addition to the direct effects of suspended sediments, soil particles also degrade water quality by transporting other pollutants to surface waters. Phosphorous, nitrogen, and pesticides bind to soil particles on agricultural land and are washed into waterways after irrigation or rain events (Ritter & Shirmohammadi, 2001). Soil erosion accounts for 80% of the total phosphorous and 73% of the total Kjeldahl nitrogen found in waterways of the United States (USDA, as cited by Ritter & Shirmohammadi, 2001).

Nutrients transported to surface waters either attached to soil particles or dissolved in runoff have been identified as the number one cause of impairment by nonpoint source pollution in lakes and estuaries (Baker, 1992). The primary nutrients of concern are nitrogen and phosphorus (Carpenter *et al*, 1998). While nitrogen can be toxic to humans at certain concentrations, phosphorus is not considered to be directly toxic to humans or animals. Rather than toxicity concerns, water bodies are listed as impaired for excessive nutrients because they lead to accelerated eutrophication, or excessive plant growth (EPA, 1999). Phosphorus is the limiting nutrient in the majority of freshwater lakes, and nitrogen is generally the limiting nutrient for estuaries (Baker, 1992). In Florida, some regions are composed of soils with large deposits of phosphorus, and nitrogen becomes the limiting factor for lakes in these regions (Florida Lakewatch, 2000). Eutrophication of water bodies in the United States is a growing problem that accounts for about 50% of impaired lake area and 60% of impaired river reaches (Carpenter *et al*, 1998). According to the University of Florida, 57% of Florida lakes are considered either eutrophic or hypereutrophic (UF IFAS, 2009).

Eutrophication is a process caused by increased nutrient loads in a water body that results in excessive algae and plant growth (Correll, 1998). Plants and animals require nutrients for growth, and nitrogen and phosphorous occur naturally in aquatic environments at levels below 0.3 and .01 mg/L, respectively. When nitrogen and phosphorus are introduced into aquatic ecosystems above these natural levels, plant production increases which can lead to eutrophication (EPA, 1999). Eutrophication can severely impact a water body's ability to attain its designated use standards. The most obvious impact is that the overgrowth of algae and aquatic weeds impairs the fisheries, aquatic life, recreation, and drinking water supply uses. In addition, increased decomposition of dead plant matter results in oxygen shortages which can cause fish kills (Carpenter et al, 1998). Eventually oxygen in the bottom of lakes can become depleted which leads to toxic releases from sediments affecting the fisheries and aquatic life uses (EPA, 1999). Drinking water supplies are impaired by cyanobacteria blooms that result from eutrophication. Excessive algae cause foul tastes and smells in drinking water, can clog water treatment plant filters, and form potentially carcinogenic trihalomethane during the chlorination process. Excessive plant growth and odors also interfere with recreational uses such as swimming, fishing, and boating (EPA, 1999).

Agricultural Pollution Legislation

Although the Clean Water Act (CWA) was mainly targeted at point source pollution, nonpoint source pollution was addressed as well. Section 208 called for the development of watershed management plans and all sources, including nonpoint sources, were to be included in the plans (Malik, 1994). States were directed to identify and control nonpoint source problems and to implement appropriate controls; however, due to the prevalence and severity of point source pollution problems, nonpoint sources were routinely overlooked (Adler *et al*, 1993).

Section 303 of the CWA outlined the Total Maximum Daily Load (TMDL) program. The program called for states to identify waters that do not meet water quality standards, determine the maximum pollutant loads that would bring water quality to standards, and to develop basin management action plans to implement the TMDL. The TMDL was to be split between all sources, both point and nonpoint (Houck, 2002). The program moved very slowly until more recent years, but implementation plans are currently being developed that will push pollution reduction strategies.

In 1987, Congress passed amendments to the CWA including section 319 which set up state programs to address nonpoint source pollution problems. States were directed to identify sources of nonpoint source pollution and implement management programs to control the sources that included best management practices (BMPs), or land-use controls and land-management practices (Malik, 1994). Management practices can be either structural or managerial in nature. Examples of managerial BMPs for agricultural pollution control include rotational grazing, nutrient management, pesticide management, and conservation tillage. Structural BMPs include the use of treatment lagoons or ponds, terraces, and sediment basins (EPA, 2003). The most efficient and accepted approach by land owners to control agricultural pollution is a combination of these BMPs along with offsite natural or constructed wetlands located in various areas throughout the watershed designed to receive nonpoint source pollution from larger areas (Hammer, 1992).

In 1999, many years following the passage of the CWA, Florida Legislature enacted the Florida Watershed Restoration Act (FWRA) in order to establish the TMDL program in accordance with the federal requirements (Section 403.067, Florida Statutes). The Florida Department of Environmental Protection (FDEP) was authorized as the lead agency in determining impaired waters and TMDL development. The Department of Agriculture and Consumer Services (DACS) was established as the lead agency responsible for FWRA enforcement involving agricultural nonpoint source pollution. Under the FWRA, DACS may develop and adopt BMPs to meet the load allocations for agricultural nonpoint source pollution resulting from TMDLs (UF IFAS, 2005).

Treatment Wetlands for Agricultural Pollution Management

Natural wetlands have been used for wastewater discharge sites for at least one hundred years in some locations around the world. However, their water quality benefits were not recognized until monitoring of some of these natural wetlands began in the 1960s (Kadlec & Knight, 1996). The first constructed wetland was designed to receive wastewater and underwent extensive scientific investigations beginning in 1952 (Kadlec & Knight, 1996; Campbell & Ogden 1999). Widespread use of constructed wetlands for wastewater treatment began in the United States in the 1970s. Industrial stormwater and process water began to be treated by constructed wetlands in 1975 and in the 1980s constructed wetlands were beginning to be designed for urban stormwater treatment (Kadlec & Knight, 1996). The use of treatment wetlands for nonpoint sources can be more complex than their use for point source pollution. For example, storms can have a large effect on their pollutant removal efficiency. High flows into the wetland can severely impair pollutant retention and can even cause release of nutrients (Mitsch & Gosselink, 2000). As the construction of treatment wetlands increased, so did the research and understanding of their processes and functions in regard to water treatment (Campbell & Ogden, 1999). Constructed wetlands have the benefits of being self sustaining and having relatively low maintenance requirements (Kadlec, 2001). However, the increasing popularity of using constructed wetlands for water quality treatment can be primarily attributed to their efficiency in pollutant reduction and relatively low cost (Hammer, 1992). The success of using treatment wetlands to treat point sources and later nonpoint sources, has led to interest in their use to treat agricultural runoff (Kovacic *et al*, 2000). In fact, wetlands have been recognized as potentially the most cost effective pollutant sinks in many agricultural landscapes (van der Valk & Jolly, 1992). Despite the increased use and recognized importance of treatment wetlands in agricultural pollution control, few studies have been published on wetland effectiveness in reducing agricultural runoff pollution in the United States (Mitsch & Gosselink, 2000).

Treatment wetland processes

Pollutant removal in treatment wetlands occurs by a variety of physical, chemical and biological processes. Wetlands have important characteristics that influence their pollutant reduction capabilities. The gas exchange rates between wetland soils and the atmosphere are very low due to the fact that they are usually inundated or at least saturated, which causes wetland sediments to be mostly anaerobic (Mitsch & Gosselink, 2000; Bix, 1993). This causes organic material to accumulate on top of the bottom sediments because decomposition is significantly slowed in anaerobic conditions. In, addition, because wetlands are generally fairly heavily vegetated, there is an overabundance of organic material within wetland systems. The layer of organic matter on the wetland bottom combined with the vegetation provides a large surface for microbial growth (Bix, 1993). Although sediments are highly anaerobic, a very thin oxidized layer is usually present on the surface of the soil. This layer contributes to sediments having a high oxidation-reduction potential which is important in the chemical transformations that occur in wetlands (Mitsch & Gosselink, 2000; Bix, 1993). This combination of characteristics gives wetlands their high capability of transforming nutrients (Bix, 1993).

Suspended solids are removed by the purely physical processes of sedimentation and filtration (Bix, 1993). Although resuspension may be common in some shallow lakes and floodplain wetlands, sedimentation is generally an irreversible process in most wetlands, including constructed wetlands (Johnston, 1991). In addition to the natural process that occurs to remove sediments in wetlands, many constructed wetlands are designed with some type of sediment basin or mechanical pretreatment unit to remove sediments before they even enter the wetland (Bix, 1993, Higgens *et al*, 1993).

The processes involved in nitrogen removal in wetlands include ammonification or mineralization, nitrification, and denitrification. Ammonification refers to the series of biological transformations that convert organic nitrogen to ammonia which occurs when organic matter is decomposed by microorganisms (Kadlec & Knight, 1996; Ritter & Shirmohammadi, 2001). Nitrification then takes place as ammonia is oxidized to nitrate by microbes in the aerobic zone. Nitrates can either be immediately assimilated by plants or microbes, or are converted into nitrogen gas by microbes in the anaerobic zone through a process called denitrification (Bix, 1993; Mitsch & Gosselink, 2000). Nitrification plays a significant role in a wetland's ability remove nitrogen from water as it releases the gas into the atmosphere (Mitsch & Gosselink, 2000).

Phosphorus retention in wetlands can occur as either short-term or long-term storage. Although a large number of temporary phosphorus storage processes and transfers occur within a wetland, the primary process involved in permanent phosphorus removal is soil sorption (Kadlec & Knight, 1996). This occurs through adsorption, complexation, and precipitation with aluminum, iron, calcium and clay minerals present in wetland sediments (Bix, 1993). However, the capacity of wetland soils to sorb phosphorus is highly variable and may only last a short period of time. Phosphorus that is attached to sediment particles is lost through the physical process of sedimentation. Phosphorus removal also occurs through plant uptake, however, it has been suggested that this should not be considered a long-term retention process (Kadlec & Knight, 1996).

Balm Road Treatment Marsh

Pictures of Balm Road Treatment Marsh can be found in Appendix A and show many of the features described in this section. The treatment system was designed as a series of shallow vegetated cells located in the floodplain on the northwest side of Bullfrog Creek. It has a wetland to watershed ratio of 2%, which is the recommended minimum for successful treatment of pollutants (Carleton *et al*, 2001). The system receives flow diverted from the creek near the end of McGrady Road. A diversion ditch and two structures were constructed to route the water from the creek into the system. Two existing channels in the creek diverge near the south end of McGrady Road. A diversion structure was placed in each of the two channels, so that water enters the system from each channel of Bullfrog Creek. The structures are constructed of sheet pile and slotted to provide base flow to the historic creek channel. The constructed diversion ditch is approximately 2.7 m deep and begins at the previously existing western channel and flows approximately 400 m west to the sedimentation basin.

The sedimentation basin is approximately 4.6 m deep at its deepest point, 91.4 m wide at the bank and sloping to 7.6 m wide at the bottom of the pond. A series of four cells are separated by berms. The system was designed to avoid "dead zones", or areas of no flow. Water flow between the four cells is maintained by 1.2 m diameter pipes. The system was designed so that the majority of dry season flow and at least the first flush of runoff from the upstream watershed resulting from storms are diverted into the wetland. This was accomplished by placing structures in the existing channels that would be overtopped during the 100-year flood event, thus do not increase the 100-year peak water elevations. Another important design element was ensuring embankments were protected from erosion and overtopping during the 100-year flood, while providing adequate treatment time during periods of low flow. At the time of design, the normal pool elevation was expected to be approximately 25 m NGVD during the dry season with small fluctuations following minor rainfall events. During the wet-season, elevations were expected to fluctuate somewhat above the dry season elevation. At 25 m NGVD, the maximum water depth in the ponds would be approximately 0.5 m deep in the few deep water areas. In May 2005 staff gauges were installed in cell number three and four and water levels were recorded during monthly sampling events. The mean water level from the period of observations from May 2005 to September 2007 was 25.7 m, and the lowest observation which occurred during the dry season of 2006 was 25.3 m. At 25.7 m, the maximum depths were approximately 1.2 m, with average depths at approximately 0.5 m. Depths are an approximation based on design plans; actual depths may very due to possible soil swelling and lift after saturation (Kadlec & Wallace, 2009). Water levels in the wetland remained higher than anticipated.

The original planting plan called for low elevations of the four cells (24.5 – 24.7 m NGVD) to be planted with groupings of spatterdock (*Nuphar luteum*). These elevations would be approximately 0.5 m deep under normal conditions, but may be submersed in up to 2.0 m of water during seasonal high stages. The upper portions of the anticipated dry-season seasonal high water elevations (25.0 – 25.3 m) were planted with pickerelweed (*Pontederia cordata*) and arrowhead (*Sagittaria latifolia*) with smaller amounts of sawgrass (*Cladium jamaicense*) and fireflag (*Thalia geniculata*). Areas above the dry season high water elevation, but within the anticipated wet-season normal pool elevations (25.3 – 25.9 m), were dominated by pickerelweed and maidencane (*Panicum hemitomon*) along with several other herbaceous species and some trees and shrubs. Upper elevations of the wetland area (25.9 – 26.8 m) were planted to resemble a pine flatwoods community with species such as slash pine (*Pinus elliottii*), wax myrtle (*Myrica cerifera*), and saw palmetto (*Serenoa repens*). These elevations were expected to only be inundated for very short periods of time following large storm events.

By August, 2005, many of the plants had been destroyed by nutria and apple snails. Replanting of the site was completed by December, 2006. Areas throughout the four cells with no coverage remaining were planted with spikerush (*Eleocharis intersticta*) in elevations of 24.7 – 25.3 m and spikerush (*Eleocharis intersticta*), bulrush (*Scirpus validus*), and maidencane (*Panicum hemitomon*) in elevations of 25.3 – 25.8. These plants were chosen based on their ability to withstand apple snail infestations. Nutria were trapped and removed from the site. A site visit on September 26, 2009 revealed there had been a major shift in vegetation. As seen in Appendix A, water paspalum (*Paspalum repens*) dominated every pond along with the submerged invasive species hydrilla (*Hydrilla verticillata*). There were still small amounts of pickerelweed, maidencane, duck potato, spike rush, and arrowhead remaining and the non-native wild taro (*Colocasia esculenta*) and torpedo grass (*Panicum repens*) were becoming established. Typically, displacement of planted species by other species will not alter treatment efficiency (Kadlec & Wallace, 2009). However, the lack of established vegetation during much of the period of study could be a factor in performance.

Annual load reduction estimates for the treatment system were made prior to construction using a model developed for Hillsborough County. The model calculations were based on EPA approved runoff calculations, which were developed using land use and soils data for the project area. Event mean concentrations of parameters were developed from NPDES permit sampling performed by the County. For the purposes of the model run, it was assumed that all of the pollutants from the modeled drainage basin enter the creek and are routed through the wetland system. The wetland system was identified in the model as a wet detention best management practice (BMP) with removal efficiencies estimated using previous field and literature research collected by SWFWMD and Environmental Research and Design (ERD), Inc. (M. Moore personal communication, September 12, 2000). Load reductions were estimated at 85% TSS, 30% TN, and 65% TP which equaled 125,060 kg TSS, 8,700 kg TN, and 13,690 kg TP per year.

Chapter 3

Review of the Literature

Wetland Treatment Performance

A review of the literature revealed that the performance of many types of treatment wetlands have been assessed in a variety of studies from around the world. For example, Yang *et al* (1995) studied the removal efficiency of a vegetated subsurface flow bed used to treat municipal wastewater in Shenzhen, China. Monthly samples were taken at the inflow and outflow of the wetland for a period of three years and the data were analyzed to determine the percent reduction for a suite of parameters. The removal efficiencies found were 92.6% total suspended solids, 23.2% total nitrogen, and 30.6% total phosphorus. The results were used for a comparative study with other similar wetlands. The studied wetland was highest in total suspended solid removal, but much lower than the highest performing wetland in nitrogen and phosphorus removal.

In Estonia, three different types of treatment wetlands were studied. A vertical-flow sand/plant filter, a semi-natural wet meadow, and a drainage channel planted with macrophytes which were all designed to treat wastewater were sampled on a monthly basis for several parameters (Mander & Mauring, 1997). Nitrogen removal efficiencies ranged from 36-67% and phosphorus removal ranged from 69-74%. Statistical analysis included using the Student's *t*-test, Kruskal-Wallis test and the Pearson's correlation technique to compare results for the different types of wetlands. The method of data analysis used in this study was examined for its applicability to the present study on Balm Road Treatment Marsh as described by Zar (1984).

A surface flow wetland, which is the same type of wetland as Balm Marsh, was studied in Italy (Borin *et al*, 2001). Although this wetland was designed to treat agricultural waste water, it is much smaller than Balm Road Treatment Marsh, and receives less water from a smaller agricultural area. Nitrogen was the only water quality parameter analyzed and it was sampled on a daily basis. Reductions were found to be almost 90%.

In Thailand, a constructed wetland was studied to determine its efficiency for removing pollutants from seafood industry wastewater (Yirong & Puetpaliboon, 2004). The wetland consisted of a series of ponds with differing process designs with the final pond in the series designed as a free water surface wetland. Samples were collected once per week for a period of only four months after approximately one year of the wetland becoming operational. Nitrate concentrations were found to be higher at the wetland outflow than the inflow, however total Kjeldahl nitrogen removal was 56%. Suspended solids removal was 95%.

In Polk County, Florida, a natural cypress dome was studied that has been used to treat municipal wastewater since 1985 (Martin *et al*, 2001). Water quality was monitored on a monthly basis at the inflow, center, and outflow of the wetland for a period of eight years, which allowed for the evaluation of long-term performance. Average removal efficiencies for the eight years were 38% total suspended solids, 90% total nitrogen, and 48% total phosphorus based on mass.

Although treatment wetland performance has been studied around the world, performance varies due to several site specific factors including wetland design, soil, plant species and number, fauna, hydrology, climate, receiving water, and source water. This creates difficultly in using the results from a particular study to assess another wetland (Kadlec & Knight, 1996; Borin *et al*, 2001; Carleton *et al*, 2001). Even though the performance of a variety of treatment wetlands is well represented in the literature, there are fewer studies that describe the performance of constructed wetlands to treat nonpoint source pollution, and fewer that focus specifically on agricultural runoff. Even results from the few existing studies cannot be used to accurately characterize the performance of a different constructed treatment wetland for agricultural runoff, because the available data contain no clear performance trends based on characteristics (Kadlec & Wallace, 2009). Studies that were found that address agricultural runoff focus on pollutant removal efficiencies, and not the overall affects to downstream ambient water quality (Koskiaho *et al*, 2003; Kovacic *et al*, 2000; Tanner *et al*, 2005). Receiving water impacts appear to be lacking for all wetland types and pollution sources.

Factors Affecting Performance

The factors causing variability in the performance of treatment wetlands have been studied. Kuehn and Moore (1995) compare data from constructed wetlands treating pulp mill effluent for reduction in biochemical oxygen demand and total suspended solids. Ponds were constructed with varying retention times and vegetation and a replicate pond was constructed for each, so that there were pairs of nearly identical ponds for comparison. Samples were taken from the inflow and outflow of each of the ponds and the resulting data compared. The results showed that similar pairs of ponds had very low performance variability. Significant variation occurred between all other ponds. The factors leading to variation included vegetation type and retention time as well as variation over time according to the season. Other studies of comparable ponds have shown similar results (Gearheart, 1992). These studies demonstrate some of the important factors affecting variability in the performance of treatment wetlands and support the fact that wetland performance results cannot be extended across wetlands.

Carleton *et al* (2001) compared pollutant reduction efficiencies from forty nine wetland systems used to treat direct stormwater runoff flows or runoff impacted surface water. When the results from all forty nine wetlands were combined and compared to values reported for wastewater treatment wetlands, nitrogen removal efficiencies were very similar. Stormwater treatment wetlands, however, showed much higher variability than wastewater treatment wetlands, which is generally expected due to the nature of stormwater and variable flows. Removal rate constants for several parameters were calculated and compared to those constants reported in the literature for wastewater treatment wetlands and found to be similar. This study suggests that it is reasonable to expect stormwater treatment wetlands to have removal rate constants similar to wastewater removal rate constants, which have been extensively studied and published in the literature compared to stormwater removal rate constants. The rate constants can be used in determining the pond area needed to achieve a specific reduction of pollutants by a stormwater wetland.

A number of studies have explored the phosphorus retention capacity and removal efficiency of treatment wetlands (Liikanen et al, 2004; Moustafa, 1999; Novak et al, 2004; Casey & Klaine, 2001; Richardson, 1985; Dierberg, 2001). Liikanen *et al* demonstrates the importance of soil characteristics in phosphorus removal efficiency. Soil properties were studied before the construction of the treatment wetland and used to determine its ability to retain phosphorus. Soil samples were used in laboratory studies to determine their ability to remove phosphorus and water samples were taken at the inflow and outflow of the wetland once it was operational to determine its efficiency. The study found that if soils on the wetland project site contain high amounts of phosphorus, it is essential to remove the soils prior to construction because they can lower phosphorus removal of the wetland. This research is significant in that it demonstrates the importance of soil characteristics in phosphorus removal.

Another factor involved in phosphorous retention capacity of wetlands is the extractable aluminum content of the soil (Richardson, 1984). Soils from a wide range of natural wetlands were sampled to determine their phosphorus sorption capacity. Actual measurements of phosphorus exports from the same wetlands correlated to soil sorption capacities. The sorption capacity was then compared to other soil characteristics such as percent organic matter, pH, and extractable aluminum, iron, and calcium. Statistical analysis showed a direct correlation between the amount of extractable aluminum present and soil sorption capacity. This study reconfirms the importance of soil characteristics in phosphorus removal efficiencies. The data also indicated that initial phosphorus removal rates of a wetland may be followed by large exports of phosphorus within a few years.

There are other factors influencing phosphorus retention in treatment wetlands as demonstrated by Moustafa (1999). Moustafa examined data from approximately one hundred wetlands to determine their phosphorus loading rates, morphology, and hydrological characteristics. The research found that water depth plays a key role in phosphorus retention and showed that shallow water depths within a wetland increase the amount phosphorus removal. Phosphorus removal efficiencies were also demonstrated to be a function of water and phosphorus loading rates into the wetland. The relationship can be used to predict phosphorus removal efficiencies.

An in-stream wetland that receives water from an agriculturally intensive subwatershed in North Carolina was examined for phosphorus retention (Novak *et al*, 2004). Weekly samples for dissolved phosphorus were taken along with flow data to determine inflow and outflow dissolved phosphorus load estimates and retention and release rates. Water column dissolved phosphorus samples were also collected at two points within the wetland along with soil samples that were analyzed for phosphorus. The data were then used to determine the sorption or desorption tendency of the wetland sediments by comparing the water and soil samples. Water column sediments were also sampled and analyzed for dissolved phosphorus. The data were used to produce dissolved phosphorus concentration profiles under varying management conditions, including flooding, draining and shifts in dissolved phosphorus concentrations. These results can be used to determine optimal ranges for variables that affect phosphorus retention including residence time and sediment surface area. An important conclusion drawn from this research is the fact that this particular wetland did not provide effective long-term dissolved phosphorus retention. The results here indicate that long-term detention in phosphorus laden wetlands may be unlikely. If inflow phosphorus concentrations are reduced resulting in less phosphorus present in the water column than the underlying sediments, the sediments may release phosphorus resulting in higher phosphorus discharges than inflows, creating a negative phosphorus retention rate.

Phosphorus release from underlying sediments can negatively impact treatment wetland removal rates.

Nitrogen retention by wetlands has been studied as well. A study by Felberova *et al* reported seasonal variations in nitrogen retention (1993). Removal efficiencies were determined to be greater in the summer months. Nitrogen retention was also shown to be affected by plant species. A constructed wetland that received wastewater treatment flow was designed with four subsurface horizontal flow treatment beds. Pairs of beds were planted with a different wetland species. The treatment beds with different species showed varying removal efficiencies, while similar beds displayed similar results. This study was important in describing factors that affect nitrogen retention in wetlands.

A more thorough investigation of vegetation and temperature effects on nitrogen removal efficiency was performed by Bachand and Horne (2000). The study was intended to determine the design features of a constructed treatment wetland that may contribute to increased nitrogen removal performance. Species were planted in six treatment cells; two cells contained only bulrush (*Scirpus* spp.), two cells only cattail (*Typha* spp.) and the last two cells contained a combination of the two. The cells received water with nitrogen concentration similar to that from a wastewater treatment plant. Water samples were collected at the inflow and outflow of the cells on varying frequency, at times as often as every day. Plant and soil samples were also taken and all three sample types were analyzed for nitrogen concentration. Nitrogen removal rates between cells with different plant composition showed significant differences. The mixed vegetation displayed the greatest efficiency followed by the cattail and the bulrush species. The study was combined with a thorough review of the literature to make detailed suggestions of vegetation composition for the most efficient nitrogen removal. By comparing nitrogen concentrations, it was found that sediment and plant uptake only accounted for a fraction of the nitrogen removed from the water column, concluding that denitrification was the primary responsible process. It was further concluded that dissolved oxygen concentrations and nitrate availability did not affect denitrification, but that water temperature was likely the most influencing factor. This research suggests that vegetation effects, water temperature, and seasonal variations should all be taken into consideration when examining nitrogen removal efficiency. There appears to be a general consensus in the published literature that pollutant removal efficiencies show seasonal variation. This suggests that data should be examined on a seasonal basis in addition to long-term comparisons.

Processes and Design

Chemical, physical and biological cycles and processes in treatment wetlands are important factors in pollutant removal. Kadlec (1999) presents some of these cycles and describes their effects on pollutant removal. For example, solar radiation drives photosynthesis influencing plant processes on an annual cycle. Pollutant uptake and burial is regulated by the biogeochemical cycle and rain and evapotranspiration influence the wetland water budget which in turn affects pollutant removal. Due to many of the cycles involved, nitrogen and phosphorus removal may vary seasonally due to temperature dependent processes.

Hammer has published a substantial amount of work on treatment wetlands in peer-reviewed journals as well as written and edited books on the topic, especially concerning treatment wetland design (Hammer, 1989a; Hammer 1989b; Hammer, 1992;

Hammer 1994; Hammer 1997). His work was reviewed extensively and incorporated into the present research, an example of which is presented here. Hammer (1992) provides good background information in the historical use of wetlands, both natural and constructed, for their water treatment capabilities. He also discusses the four principle components in the pollution reduction functions of wetlands – vegetation, water column, substrates, and microbial populations. Hammer then presents a detailed discussion of designing treatment wetlands for livestock wastewater treatment. This includes the use of an optional settling basin just upstream of the wetland to remove solids, site selection criteria, the required treatment area, suggested number of treatment cells, cell shape, water control structures, pond bottom and liners, and vegetation. The above criteria are then adjusted and presented along with additional recommendations for adapting the design for pasture or crop field runoff. The design details of the Balm Road Treatment Mars were evaluated and compared against design criteria found in the literature.

Kadlec and Knight (1996), a chemical engineer and a wetland ecologist, have both been studying treatment wetlands since 1970. They combined their efforts in 1996 to produce the first engineering design manual for treatment wetlands. Most of the literature published since this book, reference the manual at least once, and it was referred to often for this research. Although the work is primarily focused on treatment of wastewater, rather than nonpoint source pollution, the underlying concepts are generally the same. Topics included in this work range from wetland structure and function, soils, hydrology, microbial communities, plants, wildlife, effects on water quality with detailed processes, modeling efforts and values for rate constants and regression parameters, wetland design, operation and maintenance, and case studies.

Although the text presents only a limited amount of information on monitoring and performance determination, which is the main focus of the present research, the information presented in the text was necessary to present the research in its relevant context.

Data Analysis

Data analysis for the determination of treatment wetland performance found in the existing literature has relied primarily on the comparison of inflow and outflow constituent concentration averages sometimes combined with discharge data to find a concentration reduction or mass removal (Kuehn & Moore, 1995; Yang *et al*, 1995; Mander & Mauring, 1997; Borin *et al*, 2001; Martin *et al*, 2001; Yirong & Puetpaiboon, 2004). However, outflow pollutant concentration and discharge data for Balm Road Treatment Marsh are not available and the current research focus is the affect on receiving water quality. There has been a vast array of literature published on water quality data analysis which was examined in relation to the present research (Hirsch *et al*, 1982; van Belle & Hughes, 1984; Helsel, 1987; Lettenmaier, 1998; Berryman *et al*, 1998; Loftis *et al*, 1991; Hirsch *et al*, 1991; Harcum *et al*, 1992). An important consideration in this research was the use of parametric verses nonparametric statistical analysis which is discussed at length. Nonparametric methods have distinct advantages when analyzing data without normal distributions and many outliers. This literature was the basis for choosing statistical methods for data analysis to determine impacts to ambient water quality data.

Methods of calculating pollutant loads under typical conditions where discharge data are available at near continuous intervals, but water quality data are collected less frequently have been studied extensively (e.g., Dolan et al, 1981; Walling & Webb, 1981; Ferguson, 1987; Richards & Holloway, 1987; Cohn et al, 1989; Preston et al, 1989; Kronvang & Bruhn 1996). The methods used to produce load estimates using limited water quality data can be split into three general categories: averaging approaches, regression models, and ratio estimators. Averaging is considered to be the simplest approach and is based on some form of average used in calculations with available discharge and water quality data (Preston et al, Richards 1996). There have been a number of different averaging approaches suggested with varying degrees of accuracy and precision (Dolan et al, 1981; Walling & Webb, 1981; Preston et al, 1989). Although it has been found that regression and ratio methods are often more accurate than averaging methods, they frequently lack precision and produce inconsistent results. Some averaging methods, although they often greatly underestimate loads, tend to be fairly precise among estimates and may be the more appropriate choice in certain situations (Walling & Webb, 1981; Richards, 1996). These studies were used in determining the most appropriate method for estimating pollutant loads.

Chapter 4

Research Design

Problem Statement

The purpose of this study was to determine the water quality impacts of Balm Road Treatment Marsh in order to gain better understanding of the performance of constructed treatment wetlands for agricultural pollution management.

Research Questions

Three research questions were answered in order to address the problem statement. What were the resulting ambient water quality impacts of Balm Road Treatment Marsh on Bullfrog Creek? Was there a subsequent pollutant load reduction to Tampa Bay? How does the performance of constructed wetlands used to treat agricultural pollution compare to wetlands used to treat other pollution? These answers aided the determination of whether or not constructed treatment wetlands are appropriate for agricultural pollution management, which in turn will help water resource managers design effective pollution reduction strategies for agricultural nonpoint source pollution.

Study Significance

As previously noted, agricultural nonpoint source pollution is the number one source of water quality impairments to most surface water in the United States (Parry, 1998). It is therefore imperative to find effective tools and management practices to reduce pollution from this source in order to ensure water bodies meet their designated standards. Surface water is important for use as a source for drinking water, navigation, recreation, and habitat for wildlife and fish among others. Meeting quality standards for these uses is dependent on effective management practices that lead to maintaining and improving water quality. The proposed research will address a specific management practice that is being used with increasing frequency, but for which there is little information concerning its effectiveness (Mitsch & Gosselink, 2000). Van der Valk and Jolly (1992) found that studies which address the effectiveness of constructed wetlands as nutrient sinks are one of the most important research needs regarding the use of wetlands to treat agricultural pollution. This research is an important step in filling the information gap that exists on the effectiveness of constructed wetlands to reduce agricultural nonpoint source pollution. In addition, the information on overall affect on receiving water bodies is limited. This is of particular importance when the treatment objective is to improve water quality in receiving waters, for example to meet water quality standards. The information on pollutant removal efficiency of wetlands available in the literature rarely includes overall affects on downstream water quality.

Chapter 5

Study Area

Location

The Bullfrog Creek basin is 100 square kilometers located between the Alafia and Little Manatee Rivers in southern Hillsborough County. It drains to the Hillsborough Bay segment of Tampa Bay just south of the Alafia River (Dames and Moore, 2000). The basin has been grouped with the Coastal Hillsborough Bay major basin for loading estimates to the Bay (TBEP, 2005). The basin's elevations range from 44 m NGVD in the east with rapid declines to sea level moving west to the bay (Dames and Moore, 2000).

Balm Road Marsh is located near Bullfrog Creek's headwaters in the upper portions of the Bullfrog Creek basin (Figures 1 and 2). The 12 ha treatment system was built on the southeast corner of a 121.4 ha portion of county land. The Balm Road property's elevation ranges from approximately 30.5 feet NGVD at the high end near the upland areas to less than 19.8 m NGVD in the stream channel located in the west end of the site (Ayres, 2000).

Climate

The area climate is subtropical, with long humid summers and mild short winters. The majority of rainfall occurs between the summer months of June and September as seen in Figure 5. Rainfall is highly variable both spatially and temporally with the majority of rain resulting from isolated summer thunderstorms. Intense rainfall may result from hurricanes, tropical storms, or tropical depressions. Winter rainfall is light (Dames & Moore, 2000). Historical data retrieved from the nearest Southeast Regional Climate Center weather station located in Parish, Florida, reveal that the average maximum summer temperature is approximately 33° C with an average minimum of 22° C. Winter average maximum is 23° C and average minimum is 11°C.

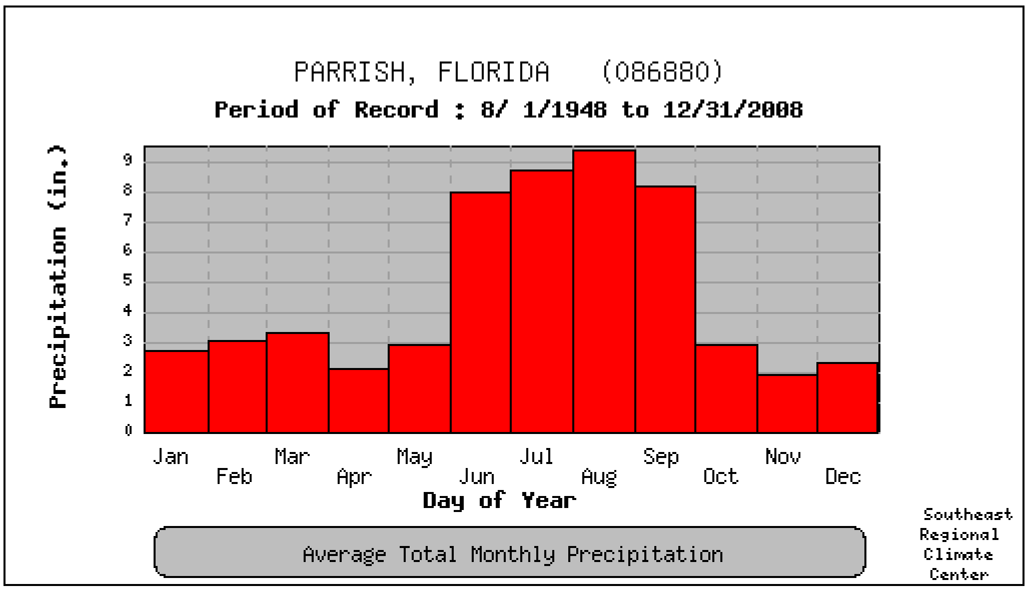

Figure 5. Average Monthly Precipitation in Parish, Florida (SRCC, 2007).

Soil

The dominant soil type in the Bullfrog Creek basin is Myakka, which is a fine, poorly drained sand with no or extremely low slopes. Bullfrog Creek and its tributaries are dominated by Winder fine sands, which is frequently flooded and either flat or nearly flat. The dominant hydrological soil group is D in the naturally undrained condition and B where the soils have been artificially drained. Group D soils are described as having high runoff potential and low infiltration rate. They are mostly shallow clays with a high water table. Group B soils have a moderate infiltration rate. They are moderate to deep, with a moderately fine to moderately course texture, and are moderately well drained (Dames and Moore, 2000).

The soil survey for the Balm Road property is shown in Figure 6 (USDA, 2006). The highest elevations on the site consist of mostly Archbold fine sand, labeled 3 on the map, and some Pomello fine sand (41). Ayres studied historical aerial photographs and adjacent undisturbed habitat to determine that this area formerly supported a scrub habitat (2000). Myakka fine sand (29), which is generally associated with pine flatwoods, covers almost half of the property area and the majority of the actual wetland site. Other soils found on the property include Basinger, Holopaw, and Samsula soils (5) located in the natural flatwoods pond on site, St. Johns fine sand (46) which is typically found in areas of natural overland flow, and Winder fine sand (60) found in the Bullfrog Creek floodplain (Ayres 2000; USDA 2006).

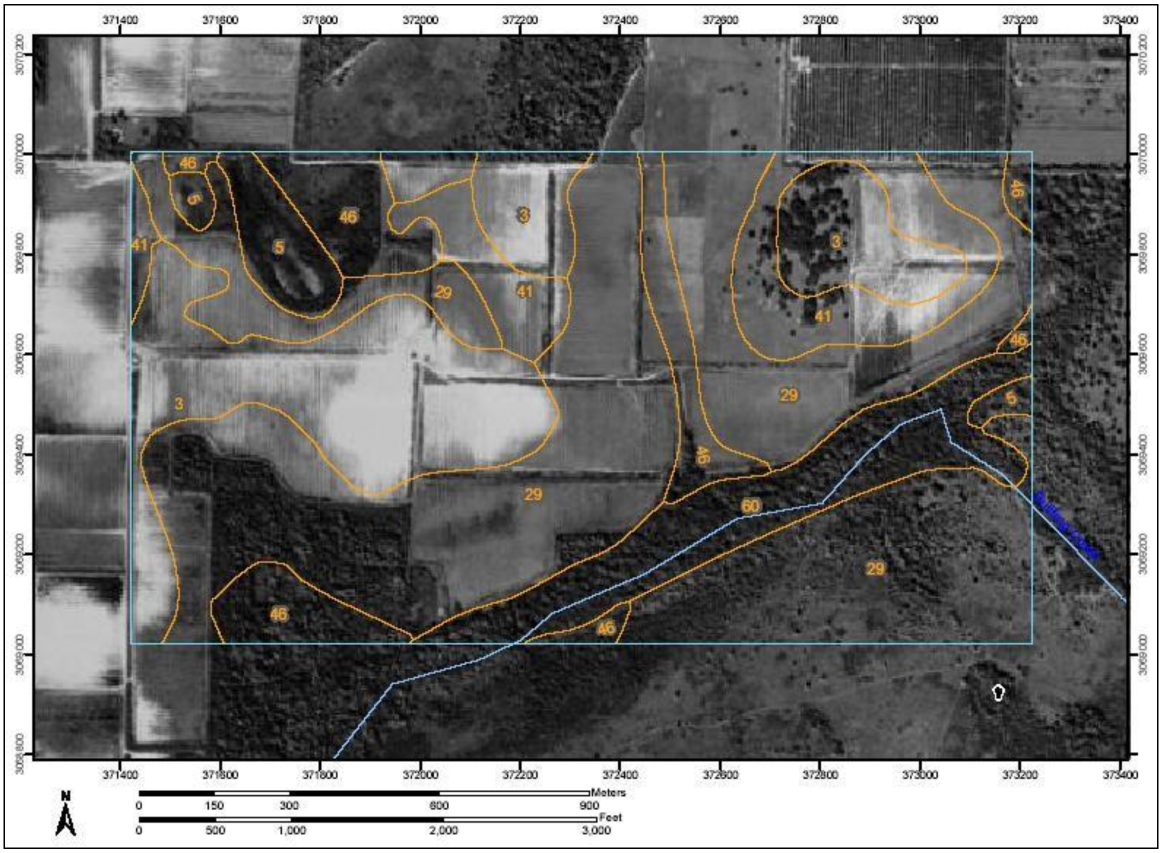

Figure 6. Soil Map for Balm Road Treatment Marsh site (USDA, 2006).

Land Use

The Bullfrog Creek basin consists of 65% agricultural lands including field and row crops, citrus, and pasture. Residential is the second highest land use which comprises 8% of the total area. Other minor land uses include natural lands and industrial. However, future land uses are projected to be primarily residential with agricultural lands being quickly developed into residential areas. (Dames and Moore, 2000).

The Balm Road property is mostly uplands with some natural wetlands. The land was previously converted to row crops, which involved the removal of native vegetation, grading, and the construction of an extensive network of drainage ditches throughout the surrounding uplands. One of the larger ditches, just upstream of the site and parallel to McGrady Road, receives runoff from a few hundred hectares of pasture and citrus groves. Prior to construction of the treatment marsh, the entire property was used for cattle grazing (Aryes 2000). The area upstream to the inflow of the marsh site consists of approximately 741 ha of land used primarily for pasture, citrus groves, and tropical fish farms, and a few single-family residential areas.

Hydrology

The major conveyance in the Bullfrog Creek basin is Bullfrog Creek. The creek has several tributaries from the east, with the largest being Little Bullfrog Creek (Figure 3). The creek flows from the southeast to the northwest, with the longest segment flowing directly to the north. The flow is relatively quick in the lower reaches and slow in the wetland sections in the upper reaches and near the headwaters (Dames and Moore, 2000).

Detailed hydrologic studies and modeling have been performed for the Bullfrog Creek/Wolf Creek Watershed and were later modified by Ayres for use specific to the Balm Road property (Dames & Moore, 2000; Ayres, 2000). Ayres found that the 2.33 year storm event has a peak flow rate of 13 m^3/s with a 26.93 m stage and the 100 year storm event has a peak flow rate of 45 m^3/s with a 27.57 m stage at the marsh site.

Chapter 6

Methods

Sample Collection and Laboratory Analysis

Water quality data from four locations on Bullfrog Creek were analyzed to answer the research questions. In order to establish base line conditions, ambient water quality monitoring on Bullfrog Creek began six years prior to the construction of Balm Road Treatment Marsh in 1998. The first water quality sample collection site on Bullfrog Creek was located just upstream of the proposed inflow to the marsh system at the end of McGrady Road. For the purpose of this research, this site is called Upstream. This site continued to be monitored throughout the construction phase and post-construction until the end of the study. The Upstream site was located downstream from a culvert on Bullfrog Creek after merging with a drainage ditch. The area was wide and water flow slowed and created a small pool between the upstream and a second downstream culvert. The creek split here and water either flowed through the first diversion structure continuing down the first branch of Bullfrog Creek, or down a canal which led to the second diversion structure. The second structure diverted baseline flow to a second branch of Bullfrog Creek. All other flows went though the treatment system.

The second sample collection site of interest was monitored beginning in 2001, over two years before construction of the treatment system was completed. The site was located on Bullfrog Creek just downstream from the planned wetland discharge. The site was approximately 1.3 km downstream from the Upstream site and after the treatment system was complete, it included both the untreated baseline flow though Bullfrog Creek and the treated wetland discharge. The data collected here represents the overall impacts of the treatment system to Bullfrog Creek. This site is named Downstream 1.

Additional water quality sample sites were located further downstream from the wetland in order to monitor the resulting changes to the creek's ambient water quality. The second downstream site was located in Bullfrog Creek Scrub, a 650 ha nature preserve approximately 9 km downstream from the treatment system. This site is named Downstream 2. Water quality monitoring began at this site in August of 2002 and continued through the end of the study.

The final monitoring site, Downstream 3, was located at a United States Geological Survey (USGS) flow gauge, and is the only site with flow rate data for the creek. The site is located approximately 12.5 km downstream from the treatment system at Big Bend Road. Monitoring here began in 1998 and continued until study completion.

The site locations can be found in Figure 7, with the Upstream and Downstream 1 sites just above and below the area labeled Balm Road Marsh and the other downstream sites located further downstream from the marsh. Sampling for all four sites was conducted on a monthly basis until the project concluded in September of 2007. Samples and field measurements were collected by the Southwest Florida Water Management District (SWFWMD). Samples for the final four months of monitoring were collected by the researcher, and previous data were collected by other SWFWMD staff. Monthly sampling was scheduled at the convenience of SWFWMD staff, so it usually occurred on a different day every month without regard to previous rainfall. Therefore, some sampling events may have occurred immediately following storm events while others occurred during extended dry periods and sampling intervals vary month to month. Each site was sampled within a few hours on the same day.

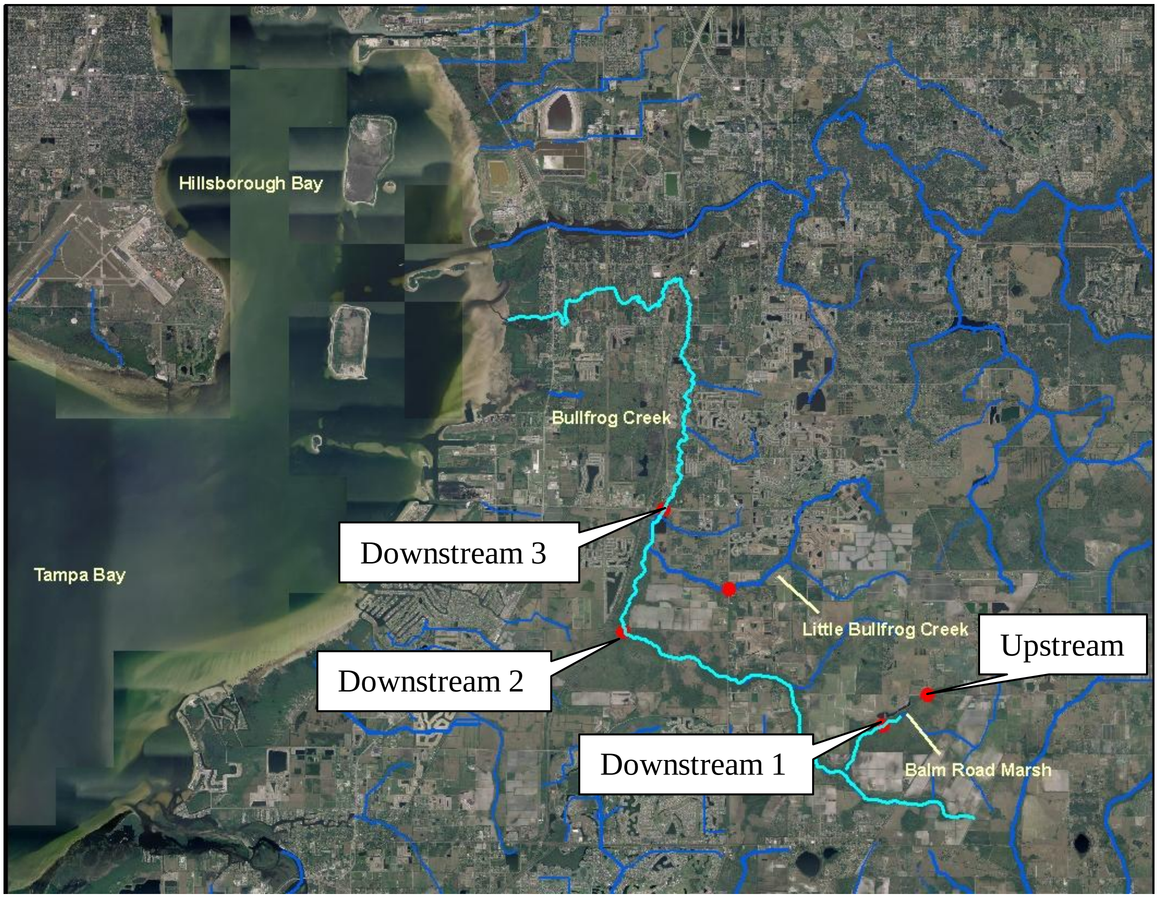

Figure 7. Bullfrog Creek water quality sample sites. Sites are symbolized as red dots along Bullfrog Creek. The sites of interest for the proposed research are labeled as Upstream, Downstream 1, Downstream 2, and Downstream 3.

Monthly water quality measurements included a suite of parameters. Field measurements were taken using a YSI 6 Series Sonde and included temperature, specific conductance, dissolved oxygen, pH, total (stream) depth, and sample depth. Samples for laboratory analysis were then collected following the Florida Department of Environmental Protection's Standard Operating Procedures (FDEP, 2004). Samples were collected at half of the total depth at the sample site. Nutrient samples were immediately preserved with sulfuric acid to a pH of less than 2. All samples were immediately put on ice for preservation. Samples were transported to the SWFWMD laboratory in Brooksville, Florida for analysis of total suspended solids, nitrogen, ammonia, nitrate, nitrate/nitrite, phosphorus, orthophosphate, chlorophyll-a, chlorophyll-b, chlorophyll-c, phaeophytin, turbidity, total coliform, and fecal coliform. Sampling, analyses, and associated tasks were performed in accordance with federal (USEPA), state (FDEP), and regional (SWFWMD) quality assurance requirements. The SWFWMD laboratory is certified by the National Environmental Laboratory Accreditation Program (NELAP) under the Florida Department of Health for all parameters analyzed. The list of SWFWMD NELAP certified methods can be found in the FDEP NELAP-Certified Laboratories Database available online (FDEP, 2009b).

The parameters of interest for the present research are total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP). TSS is the measure of suspended material present in a sample and includes sediments and other particulates. Nitrogen and phosphorus are present in surface waters in a variety of forms and TN and TP includes each of these forms (Florida Lakewatch, 2000). Table 1 lists the detection limits, units of measure, and methods used by SWFWMD for each analysis which can be found at the original sources (EPA, 1983; Greenburg *et al*, 1992).

Parameter	Detection Limit	Units	Method
Total Suspended Solids	0.01	mg/L	S.M. 18th ED. 2540 D
Total Nitrogen	0.16	mg/L	E.P.A. 353.2
Total Phosphorus	0.03	mg/L	E.P.A. 365.1

Table 1. Detection limits, units, and methods for parameters of interest.

Analytical results were received from the laboratory in the form of a hardcopy report and entered along with field measurements into Excel worksheets by the researcher or other SWFWMD staff. These Excel files were used as the source for all analysis for this research. The field data were also entered into a separate spreadsheet and sent via email to the SWFWMD laboratory staff to be combined with laboratory data and uploaded to the state and federal storage and retrieval databases called STORET. Raw data can be retrieved from either FDEP STORET using organization identification code 21FLSWFD or SWFWMD's Water Management Information System (WMIS) (FDEP, 2009c; SWFWMD, 2009). The station names in the databases will not match those used here, so identification numbers are listed in Table 2.

Station Name	Station ID	Dates Available
Upstream	17927	4/1998 – 9/2007
Downstream 1	17982	4/1998 – 9/2007
Downstream 2	17737	8/2002 – 9/2007
Downstream 3	17925	12/2002 – 9/2007

Table 2. Station ID numbers and available dates for retrieval from online databases.

Discharge, or flow rate, is measured by a United States Geological Society (USGS) gaging station on Bullfrog Creek at the Downstream 3 sample site (Figure 8). The gaging station on Bullfrog Creek is a real-time system that sends instantaneous discharge data to USGS via satellite. Discharge is monitored indirectly and calculated using stage height and the predetermined rating curve for this location. Stage height is recorded by a stilling well which consists of a float inside a vertical pipe attached to a bridge on Old Big Bend Road where it crosses Bullfrog Creek. The float is attached by a pulley to a data logger and satellite (USGS, 2009b). Data was downloaded from the USGS Instantaneous Data Archive, site number 02300700 / Bullfrog Creek near Wimauma FL (USGS, 2009a).

Figure 8. Picture of USGS Gaging Station. This site is number 02300700 Bullfrog Creek Near Wimauma, FL. 9/26/2009.

Data Organization

Total nitrogen, total phosphorus, and total suspended solids data for the Upstream and Downstream 1-3 sites were extracted from the existing data set. Data were grouped into three time periods: baseline, pre, and post (Figure 9). The baseline data were not used in the research due to the lack of data at two of the four stations. The pre phase represented the time period prior to the treatment system becoming fully operational. The post phase represented the time period after the treatment system was fully operational. Pre and post data were then further split into wet season and dry seasons. Wet and dry season determinations were based on historical rainfall data from the Southeast Regional Climate Center's data collection site at nearby Parish, Florida (Figure 5). For the purposes of this research the wet season was from June to September, and the dry season was from October to May.

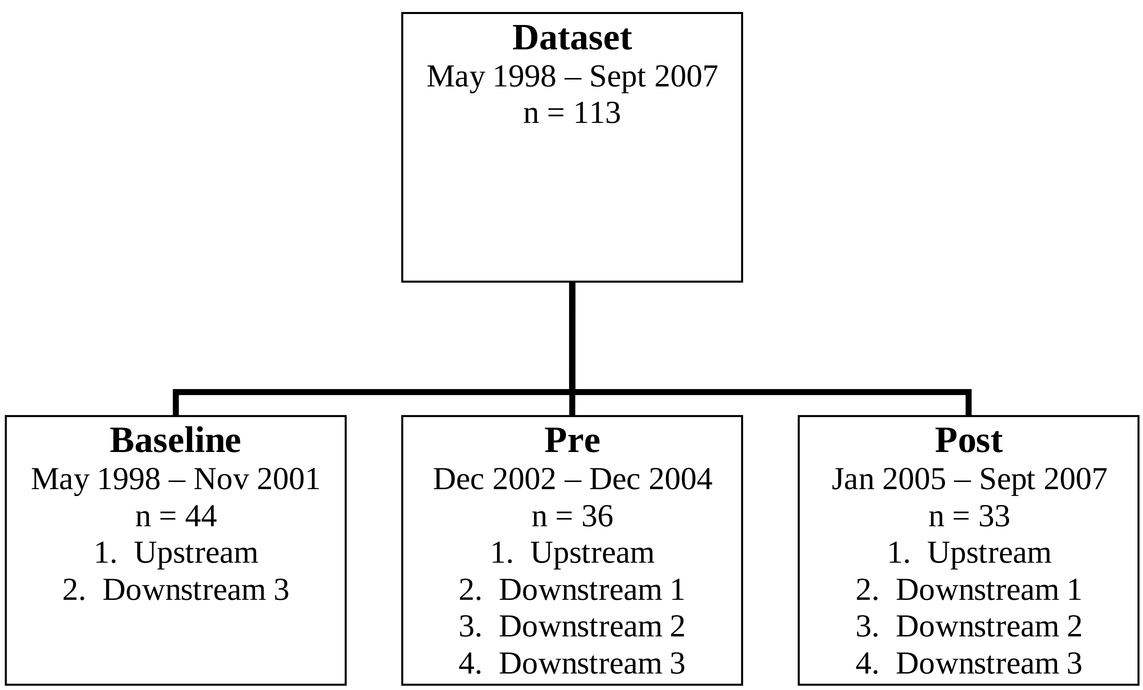

Figure 9. Data grouping diagram. The dataset was split into three subsets: baseline, pre, and post.

The baseline dataset consisted of data from the Inflow and Downstream 3 sites collected from May 1998 to November 2001. During this time period a total of 44 water quality samples were collected from the both the Inflow site and Downstream sites. No samples were collected at the other two sites during this time. These data were not used due to lack of available data at the Downstream 1 and 2 sites.

The pre dataset consisted of data collected in December 2001 through December 2004. During this period, 36 sample events occurred at all sites except Downstream 2 where 25 samples were collected. The treatment system became fully operational in late December 2004, so samples collected prior represent untreated conditions in the creek. This dataset was analyzed to find overall median TN, TP, and TSS as well as wet and dry season medians. These data were used in comparisons with data from the post phase from each site to determine water quality impacts of the treatment system to Bullfrog Creek.

The post dataset included data collected since January 2005, after the treatment system was fully operational. Thirty three samples were collected for each of the four sites. Data from the Downstream 1-3 sites during this time period reflect the impacts of the treatment system. Data were used to determine the treatment system's overall impact to ambient water quality in Bullfrog Creek and load reductions to Tampa Bay. Both overall and wet and dry season medians were found for both the Upstream and Downstream 1-3 sites. The post Upstream and Downstream 1-3 datasets were compared to both the corresponding pre datasets as well as the post Upstream in order to determine the water quality impacts of Balm Road Treatment Marsh on Bullfrog Creek.

Downstream 3 site data were also combined with discharge data and pre/post comparisons were made to determine load reductions to Tampa Bay.

Statistical Analysis

The software package PASW 18.0 (formerly SPSS) was used for all statistical analyses. Seasonal means and other descriptive statistics for each dataset were determined. The comparisons between datasets to determine treatment impacts on water quality and pollutant loads were then made. Like most water quality data, the datasets were not normally distributed and log transformations were not appropriate due to the presence of heavy tails. Histograms for each dataset can be found in Appendix B. Due to the lack of a normal distribution and presence of outliers, nonparametric methods were chosen for statistical analysis (Hirsch *et al*, 1982).

The Mann-Whitney test was used to compare the pre/post medians for each sample site. The test determines whether or not the datasets come from different populations by comparing medians and determined whether or not the values were larger in the pre Downstream 1-3 datasets when compared to the post Downstream 1-3 datasets. If the pre Downstream concentrations are found to be significantly larger than the post Downstream concentrations, a reduction in pollutant concentration may be attributed to Balm Road Treatment Marsh. Wet season, dry season and overall TP, TN, and TSS for each dataset were compared following this example as depicted in Figure 10.

Data from the Upstream site were compared to data from the Downstream sites 1-3 for the post-treatment system time period. The Wilcoxon matched-pairs signed rank test was used for these comparisons (Figure 10). This test is similar to the Mann-Whitney test, except that it compares the differences in the paired medians to determine which come from a larger population. If the Upstream site has significantly higher pollutant concentrations than the Downstream sites, a reduction in pollutant concentration may be attributed to Balm Road Treatment Marsh.

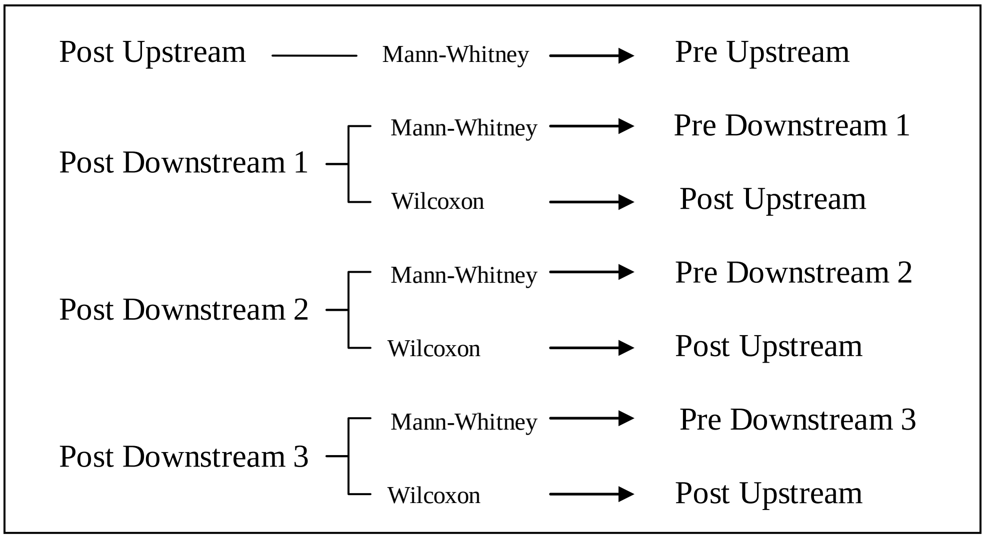

Figure 10. Diagram depicting sample comparisons using nonparametric tests. Each dataset on the left was compared to the datasets on the right using the method listed.

In addition to the treatment system's direct affect on Bullfrog Creek, load reduction estimates to Tampa Bay were determined by comparing annual pollutant loads at the Downstream 3 site both pre and post treatment system. Instantaneous discharge data from the USGS stream gaging station at the Downstream 3 site are available online in fifteen minute increments. Sampling times were recorded to the nearest five minutes, so water quality data were paired to a discharge rate within five minutes of the sampling time.

The aim of this research is to accurately detect the change in load rather than to quantify the actual load, so precision is more important than accuracy, and it was determined that averaging was the most appropriate technique for load estimation.

Walling and Webb (1981) analyzed six averaging techniques and found two that provide the most consistent results. Both methods were used to estimate pre and post average annual loads at the Downstream 3 site and produced similar results. The results from Walling and Webb's Method One are presented in order to determine load reductions to Tampa Bay (1981):

$$\text{annual load} = K(\sum^{n}_{i=1} C_i/n)(\sum^{n}_{i=1} Q_i/n)$$

where:

annual load = estimated annual load (kg/year)

K = conversion factor to take account period of record and weight units

(60*60*24*365*0.000001)

C_i = instantaneous concentration associated with individual samples (mg/L)

Q_i = instantaneous discharge at time of sampling (L/sec)

n = number of samples

Wet and dry season loads were calculated for each year during both the pre and post time periods using the formula above. The seasonal mean concentration for the corresponding phase was used when monthly water quality data were missing. An overall annual load was found by adding the time weighted wet and dry season loads for each year. This method assumes that the values of concentration and discharge associated with the individual monthly samples may be averaged to provide representative mean values for the associated time of record.

Chapter 7

Results and Discussion

Water Quality Descriptive Statistics

Overall pollutant concentration descriptive statistics including the minimum, maximum, median, mean and standard deviation are found in Table 3 for each sample site with pre and post phases combined.

	N	Minimum	Maximum	Median	Mean	Standard Deviation
Total Suspended Solids (mg/L)						
Upstream	112	0.20	32.72	2.44	4.20	5.43
Downstream 1	68	0.87	22.10	3.25	4.81	4.21
Downstream 2	59	0.16	9.95	0.54	1.22	1.91
Downstream 3	111	0.50	45.25	3.20	4.73	6.01
Total Nitrogen (mg/L)						
Upstream	112	0.25	3.94	0.90	0.93	0.60
Downstream 1	68	0.22	3.05	0.87	0.96	0.50
Downstream 2	59	0.30	1.22	0.55	0.58	0.20
Downstream 3	110	0.11	3.33	0.72	0.83	0.42
Total Phosphorus (mg/L)						
Upstream	113	0.04	0.58	0.11	0.15	0.11
Downstream 1	68	0.05	0.64	0.12	0.16	0.10
Downstream 2	59	0.12	0.60	0.26	0.29	0.13
Downstream 3	111	0.08	0.60	0.24	0.26	0.11

Table 3. Overall descriptive statistics for entire dataset available at each site.

TSS means ranged from 1.22 mg/L at Downstream 2 to 4.80 mg/L at Downstream 1. TN means ranged from 0.58 mg/L at Downstream 2 to 0.96 mg/L at Downstream 1. TP means ranged from 0.15 mg/L at Upstream to 0.30 at Downstream 3. For comparison, typical statewide values are provided in Table 4. Boxplots are shown in Figures 11-13.

Parameter (mg/L)	10th	Median	90th
TSS	2	7	26
TN	0.5	1.2	2.7
TP	.02	.09	.89

Table 4. Typical statewide percentile values for Florida streams.

Median TSS values fall below the 50th percentile in statewide comparisons, but maximums at two of the four site fall above the 90th percentile. Median TN values fall below the 50th percentile in statewide comparisons, however maximum values at three of the four sites fall above the 90th percentile. Total phosphorus medians and maximums fall above the 50th percentile.

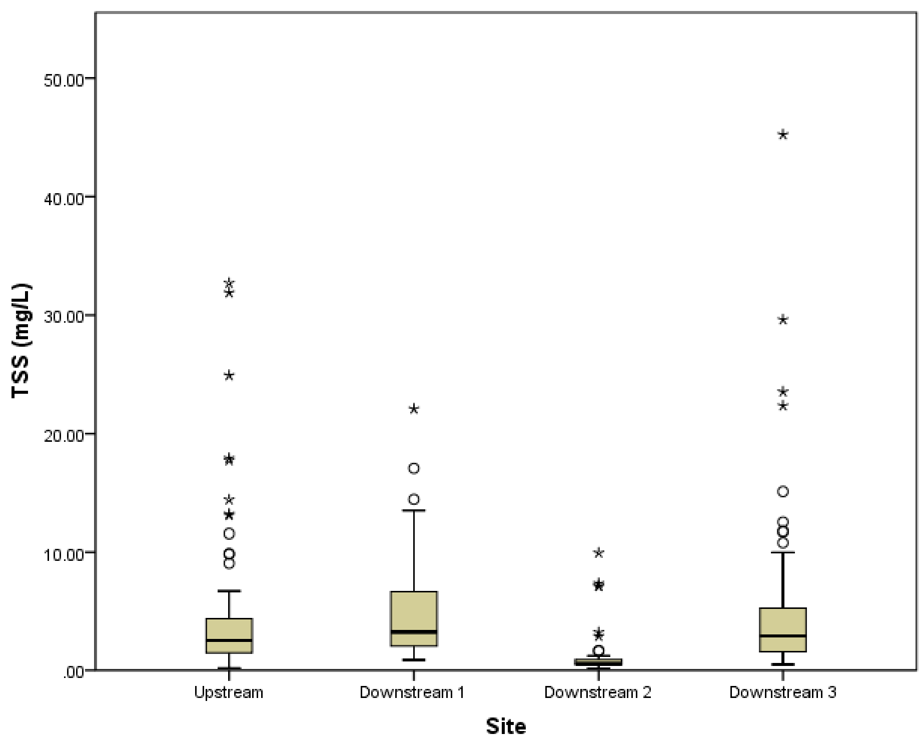

Figure 11 . TSS dataset boxplot.

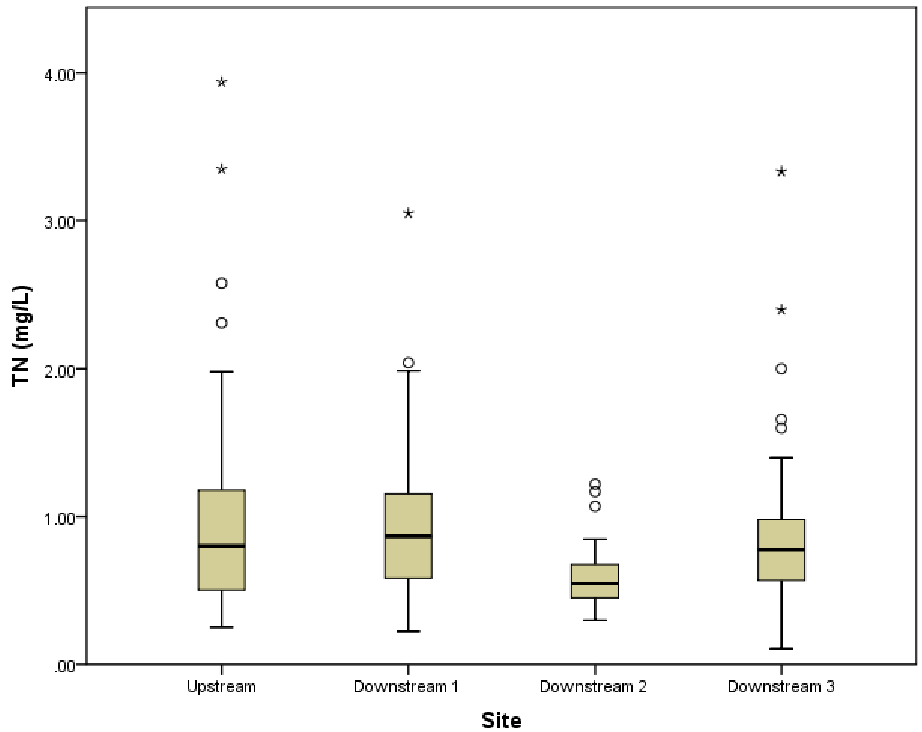

Figure 12 . TN dataset boxplot.

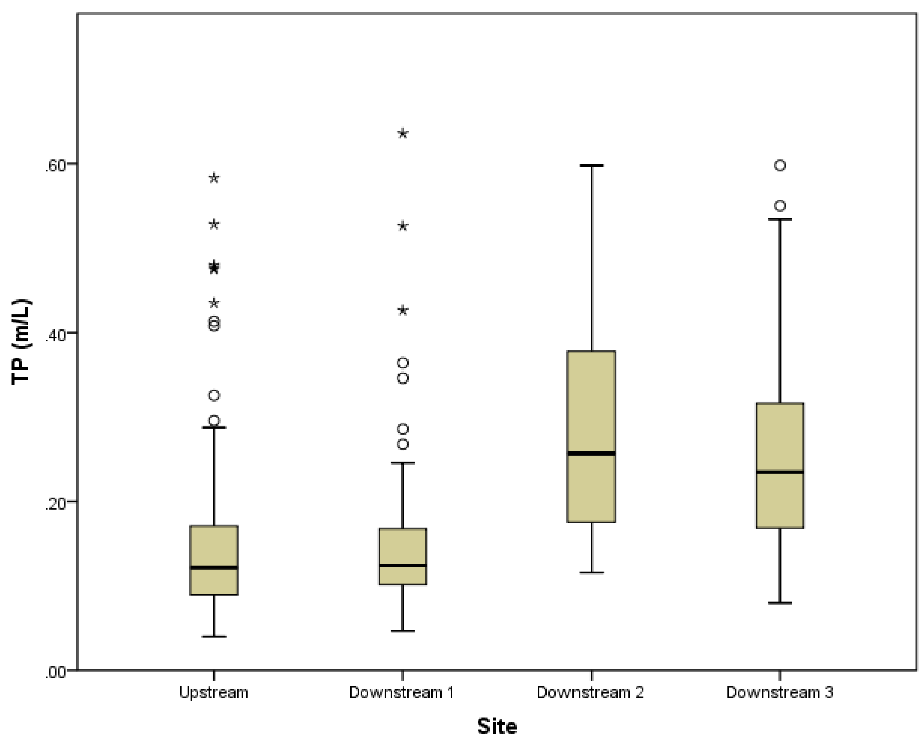

Figure 13 . TP dataset boxplot.

According to the boxplots there are many outliers, displayed as circles, and extreme values, displayed as stars, for most of the datasets. Outliers are more than 1.5 times the interquartile range and extreme values are greater than 3 times the interquartile range. The data are bound at the minimum detection limit for the given parameter and contain occasional high values, which makes the datasets highly skewed with a non-normal distribution. These are common characteristics of water quality data (Helsel, 1987). For more information on the distributions, see the histograms in Appendix B.

Descriptive statistics were also found after splitting data into pre and post phases for both combined seasons and wet and dry seasons. Table 5 and Table 6 contain wet and dry season descriptive statistics. Boxplots for combined, wet and dry seasons are displayed in Figures 14-19.

Site	Season	N	Minimum	Maximum	Median	Mean	Standard Deviation
			Pre TSS (mg/L)				
Upstream	Wet	12	0.64	17.92	4.09	5.09	4.98
Upstream	Dry	24	0.80	17.73	2.50	3.30	3.41
Downstream 1	Wet	11	2.42	14.44	4.90	5.98	3.33
Downstream 1	Dry	24	0.87	12.25	2.20	2.97	2.56
Downstream 2	Wet	8	0.50	2.89	0.66	1.04	0.85
Downstream 2	Dry	18	0.50	7.36	0.60	1.24	1.67
Downstream 3	Wet	11	1.42	12.52	4.60	5.27	3.06
Downstream 3	Dry	23	1.10	29.63	2.88	5.04	6.84
			Pre TN (mg/L)				
Upstream	Wet	12	0.52	1.85	1.26	1.28	0.43
Upstream	Dry	24	0.25	3.35	0.91	1.02	0.72
Downstream 1	Wet	11	0.58	1.54	1.19	1.11	0.35
Downstream 1	Dry	24	0.22	3.05	0.57	0.86	0.65
Downstream 2	Wet	8	0.55	0.83	0.74	0.73	0.09
Downstream 2	Dry	18	0.35	1.22	0.54	0.63	0.26
Downstream 3	Wet	11	0.59	1.12	0.90	0.92	0.15
Downstream 3	Dry	23	0.11	3.33	0.69	0.86	0.61
			Pre TP (mg/L)				
Upstream	Wet	12	0.11	0.53	0.18	0.25	0.15
Upstream	Dry	24	0.05	0.30	0.11	0.12	0.07
Downstream 1	Wet	11	0.14	0.64	0.17	0.23	0.14
Downstream 1	Dry	24	0.05	0.36	0.11	0.12	0.06
Downstream 2	Wet	8	0.32	0.54	0.46	0.45	0.08
Downstream 2	Dry	18	0.12	0.60	0.20	0.25	0.13
Downstream 3	Wet	11	0.16	0.55	0.39	0.37	0.18
Downstream 3	Dry	23	0.11	0.51	0.23	0.24	0.10

Table 5 . Wet and dry season descriptive statistics for the pre phase.

Site	Season	N	Minimum	Maximum	Median	Mean	Standard Deviation
Post TSS (mg/L)							
Upstream	Wet	12	0.63	13.2	2.40	3.85	3.75
Upstream	Dry	20	0.55	6.48	1.64	2.39	1.79
Downstream 1	Wet	12	1.36	17.07	4.52	6.37	4.81
Downstream 1	Dry	21	0.92	22.10	4.26	5.39	5.20
Downstream 2	Wet	12	0.16	7.11	0.75	1.24	1.86
Downstream 2	Dry	21	0.16	9.95	0.50	1.27	2.46
Downstream 3	Wet	12	0.87	10.80	5.42	5.57	3.31
Downstream 3	Dry	21	0.67	7.27	1.65	2.40	1.70
Post TN (mg/L)							
Upstream	Wet	12	0.36	1.57	0.84	0.86	0.42
Upstream	Dry	21	0.33	1.98	0.70	0.86	0.47
Downstream 1	Wet	12	0.56	1.99	0.93	1.10	0.50
Downstream 1	Dry	21	0.40	1.88	0.81	0.91	0.34
Downstream 2	Wet	12	0.49	0.85	0.63	0.65	0.11
Downstream 2	Dry	21	0.30	0.70	0.44	0.44	0.11
Downstream 3	Wet	12	0.53	1.32	0.82	0.89	0.24
Downstream 3	Dry	21	0.35	0.98	0.54	0.59	0.19
Post TP (mg/L)							
Upstream	Wet	12	0.11	0.58	0.18	0.23	0.14
Upstream	Dry	21	0.40	0.12	0.80	0.79	0.22
Downstream 1	Wet	12	0.12	0.53	0.16	0.22	0.13
Downstream 1	Dry	21	0.05	0.35	0.10	0.12	0.07
Downstream 2	Wet	12	0.29	0.48	0.31	0.38	0.07
Downstream 2	Dry	21	0.13	0.36	0.20	0.21	0.06
Downstream 3	Wet	12	0.24	0.44	0.38	0.32	0.07
Downstream 3	Dry	21	0.08	0.31	0.16	0.17	0.06

Table 6 . Wet and dry season statistics for the post phase.

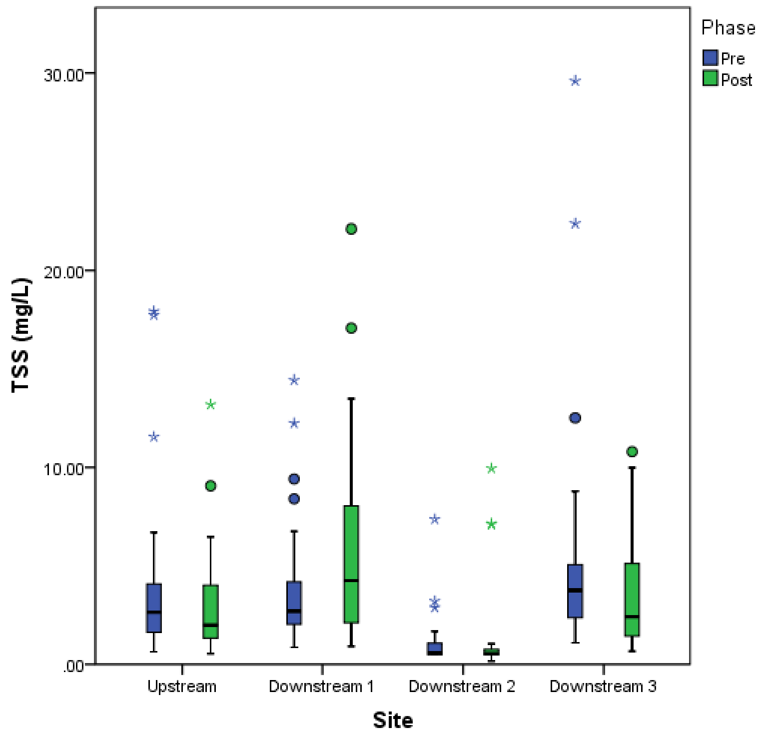

Figure 14 . TSS box plot by phase with seasons combined.

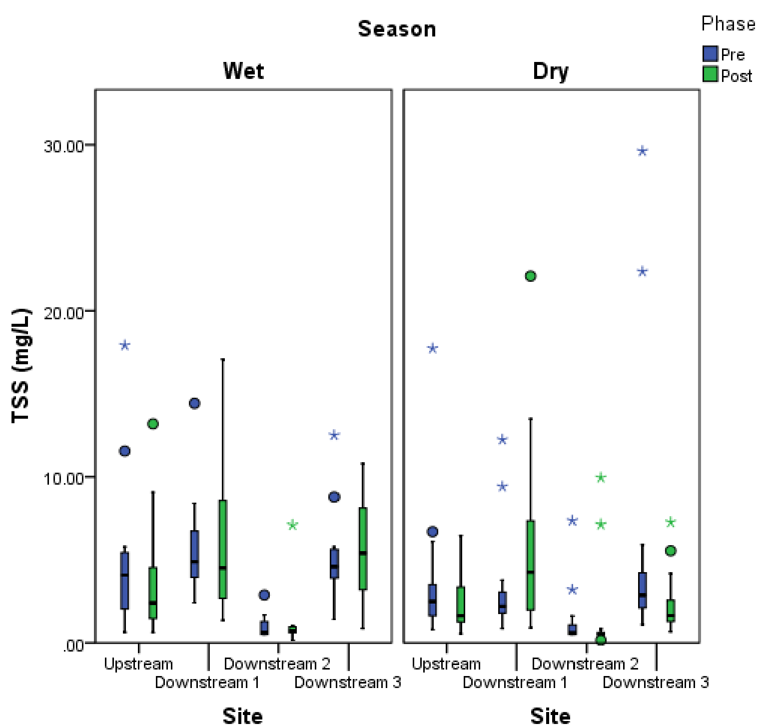

Figure 15. TSS box plot by phase and season.

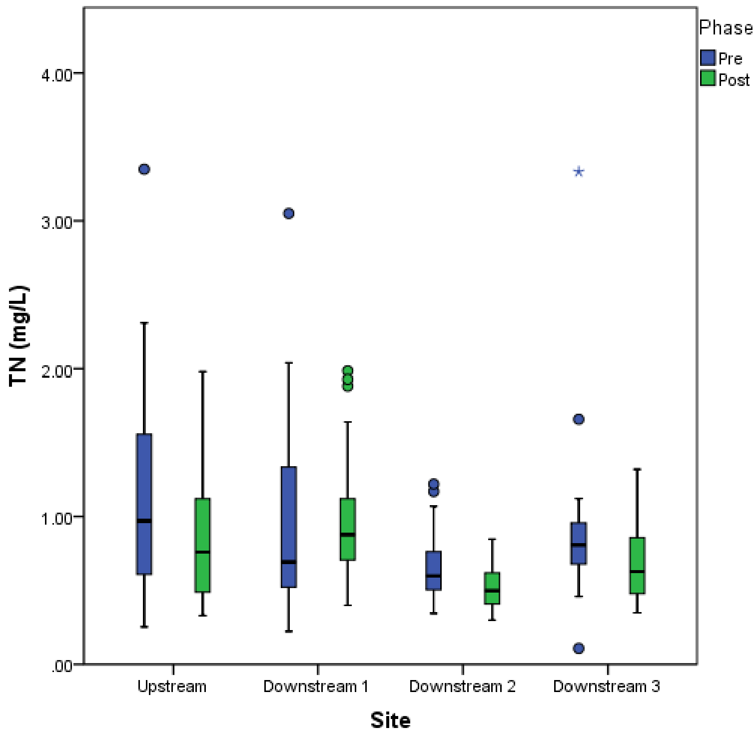

Figure 16. TN box plot by phase with seasons combined.

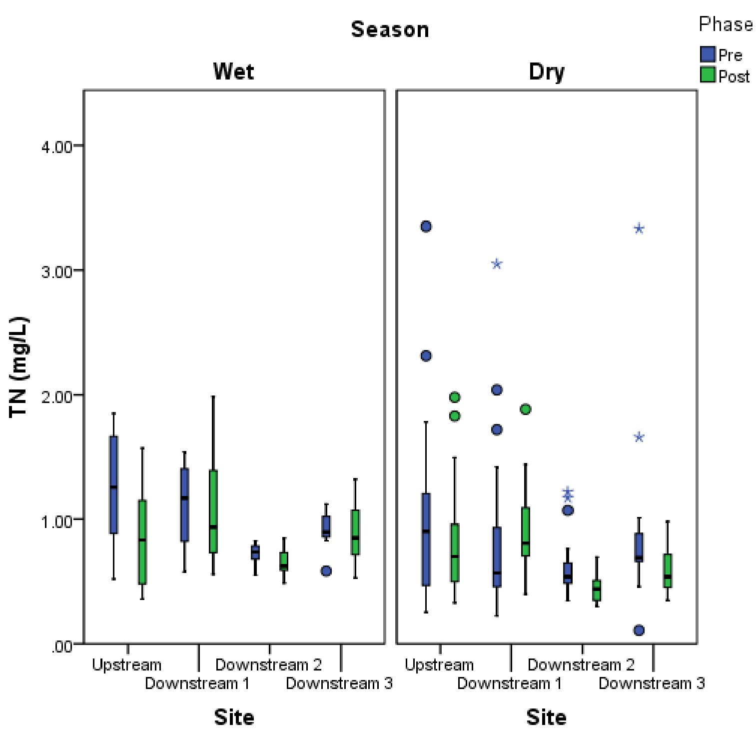

Figure 17. TN boxplot by phase and season.

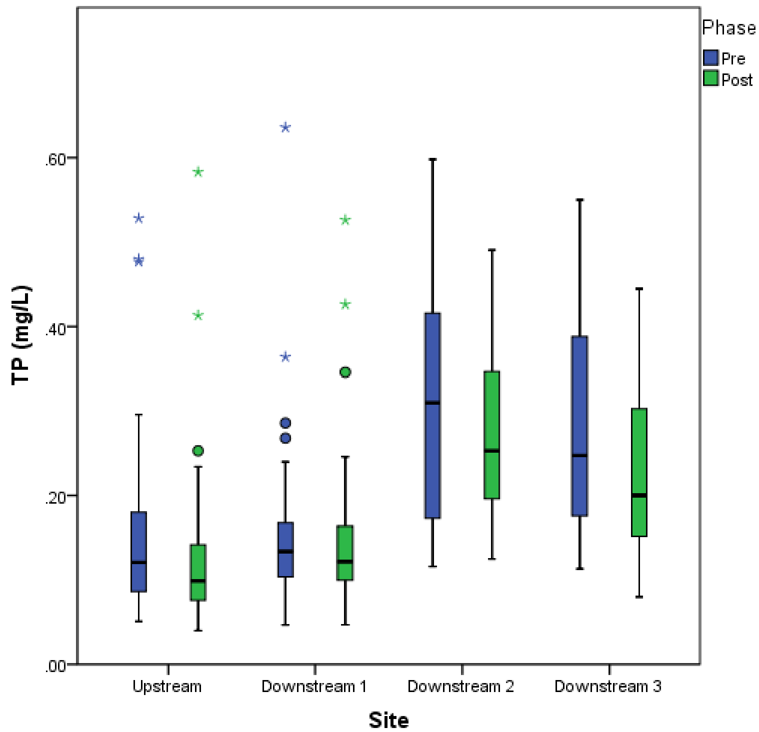

Figure 18. TP box plot by phase.

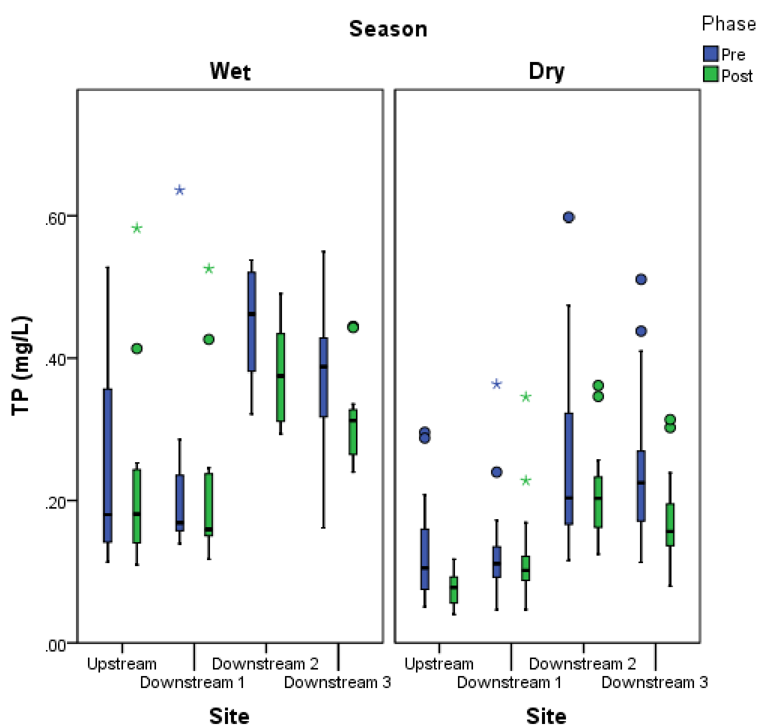

Figure 19. TP box plot by phase and season.

Pre TSS means ranged from 1.17 mg/L at the Downstream 2 site and 5.11 mg/L at the Downstream 3 when data from both season were combined. Post TSS mean range increased to 1.26 mg/L at Downstream 2 and 5.75 mg/L at Downstream 1. Pre TN ranged from 0.66 mg/L at Downstream 2 to 1.10 mg/L at the Upstream site. Post TN mean range decreased to 0.52 mg/L at Downstream 2 to 0.99 mg/L at Downstream 1. Pre TP means ranged from 0.16 mg/L at the Downstream 1 site to 0.31 mg/L at Downstream 2. Post TP mean ranges decreased to 0.13 mg/L at the Upstream site and 0.27 mg/L at Downstream 2. All wet season means were higher than dry season means except Downstream 2 pre and post TSS and post Upstream TP.

Discharge and Precipitation

Annual average discharge at the USGS gaging station located at the Downstream 3 site were found for the period of study (Figure 20). For this purpose the period of study was considered to be the entire years from 2002 to 2007, even though December 2001 water quality data is included in pre phase and the final three months of 2007 are not included the post phase water quality data. Monthly total precipitation data from SWFWMD’s nearby Romp 49 Balm Park Supervisory Control and Data Acquisition (SCADA) site was used to find yearly totals (Figure 21). The site is located approximately 3.2 km northwest of the Upstream site in the Bullfrog Creek watershed. Data was downloaded from SWFWMD’s Water Management Information System available online (SWFWMD, 2009).

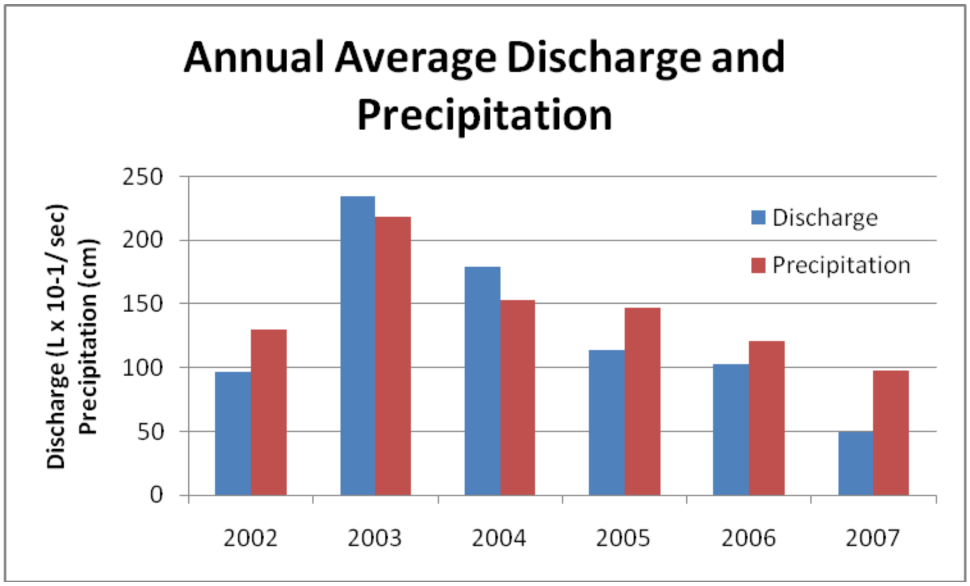

Figure 20. Annual average discharge and total precipitation.

As expected, discharge and precipitation appear to be directly related. Average discharge and total precipitation were greater during the pre (2002-2004) than the post period (2005-2007). The average discharge for the pre phase was approximately 1,700 L/sec and decreased to 885 L/sec during the post phase. Average precipitation for the pre phase was 167 cm and decreased to 122 cm during the post phase. Annual average precipitation for the area based on historical data is 138 cm. (SRCC, 2007). Precipitation not only affects discharge, but is also important when examining water quality data, particularly for pollutants whose primary source is from non point source pollution.

Ambient Water Quality Impacts to Bullfrog Creek

Pre-Post Comparisons. Mann-Whitney comparisons of the pre and post phase conditions at each sample site on Bullfrog Creek were made for each parameter with both the seasons combined and the data split into wet and dry seasons. Statistically significant changes ($p < 0.05$) in pollutant concentration were found for several sites during comparisons of the pre and post phases (Table 7).

Parameter and Season	N Pre/Post	Median Concentration (mg/L) Pre	Post	z	Significance (*p*)
			Upstream Site		
TSS Combined	36/32	2.65	1.99	-1.364	.173
TSS Wet	12/12	4.09	2.40	-0.246	.356
TSS Dry	24/20	2.50	1.64	-1.320	.187
TN Combined	36/33	0.97	0.76	-1.562	.118
TN Wet	**12/12**	**1.26**	**0.84**	**-0.431**	**.024/decrease**
TN Dry	24/21	0.91	0.70	-0.466	.641
TP Combined	36/33	0.121	0.099	-1.808	.071
TP Wet	12/12	0.180	0.181	-0.677	.908
TP Dry	**24/21**	**0.106**	**0.078**	**-2.663**	**.008/decrease**
			Downstream 1 Site		
TSS Combined	35/33	2.71	4.26	-1.319	.187
TSS Wet	11/12	4.90	4.52	-.246	.805
TSS Dry	24/21	2.20	4.26	-1.433	.152
TN Combined	35/33	0.69	0.88	-1.184	.236
TN Wet	11/12	1.26	0.84	-.431	.667
TN Dry	24/21	0.57	0.81	-1.718	.086
TP Combined	35/33	0.134	0.122	-0.558	.577
TP Wet	11/12	0.169	0.160	-.677	.498
TP Dry	24/21	0.112	0.102	-1.126	.260
			Downstream 2 Site		
TSS Combined	26/33	0.70	0.54	-1.342	.180
TSS Wet	8/12	0.80	0.75	-.155	.877
TSS Dry	18/21	0.68	0.50	-1.88	0.60
TN Combined	**26/33**	**0.57**	**0.50**	**-2.512**	**.012/decrease**
TN Wet	8/12	0.72	0.63	-1.466	.143
TN Dry	**18/21**	**0.54**	**0.44**	**-2.832**	**.005/decrease**
TP Combined	26/33	0.298	0.253	-.939	.348
TP Wet	8/12	0.416	0.376	-1.929	.054
TP Dry	18/21	0.204	0.203	-.662	.508
			Downstream 3 Site		
TSS Combined	34/33	3.70	2.42	-1.467	.142
TSS Wet	11/12	4.60	5.42	-0.369	.712
TSS Dry	**23/21**	**2.88**	**1.65**	**-2.197**	**.028/decrease**
TN Combined	**34/33**	**0.83**	**0.61**	**-2.163**	**.031/decrease**
TN Wet	11/12	0.90	0.82	-0.492	.622
TN Dry	**23/21**	**0.70**	**0.54**	**-2.620**	**.009/decrease**
TP Combined	34/33	0.248	0.200	-1.913	.056
TP Wet	11/12	0.388	0.313	-1.477	.140
TP Dry	**23/21**	**0.225**	**0.157**	**-2.585**	**.010/decrease**

Table 7. Mann-Whitney results using PASW. Pre and Post comparisons for each sample site on Bullfrog Creek. Note: $\alpha = 0.05$ for all tests.

Reductions were found at the Upstream site for wet season TN and dry season TP. No statistically significant changes were found at the Downstream 1 site. The Downstream 2 site showed a decrease in overall TN along with reduced TN concentrations during the dry season. The Downstream 3 site showed reductions during the dry season in TSS. TN reductions were seen when the seasons were combined with a significant reduction during the dry season being the major contributor. TP demonstrated reduced concentrations during the dry season.

Although some statistically significant reductions were found for TN, TSS, and TP at the Downstream 2 and 3 sites, it is difficult to attribute the reductions to Balm Road Treatment Marsh with confidence for several reasons. First, although reductions were found at some of the downstream sites, significant reductions were also found at the Upstream site for TN and TP. The upstream site acts as a control site, receiving no influence from the treatment system. Reductions found at this site, without treatment system impacts, lend to the possibility that factors other than the treatment system may have impacted reductions at the downstream sites as well. Also, the reductions in input to the treatment system suggest that results may be due in part to decreased inputs and not treatment of TN and TP.

Second, no reductions were found at the Downstream 1 site. Significant reductions at this site would have provided evidence for positive impacts to ambient water quality. The site was located only a few hundred meters downstream from the treatment system, and the contributing drainage basin is only slightly larger than that of the wetland and the Upstream site. Not finding reductions at this site gives rise to the possibility that reductions found at the other downstream sites were due to factors in their contributing basins independent of treatment system impacts.

Downstream 2 and 3 sites were located approximately 9 and 12.5 km downstream from the treatment system. There are numerous other factors that may affect water quality this far downstream. The area of the contributing watershed is much greater at these points along the creek and any changes within the watershed could affect water quality downstream. The drainage basin for the Downstream 3 site is 75.4 km^2, compared to the Upstream site basin which is only 7.4 km^2, or approximately 10% of that for the Downstream 3 site (USGS, 2009). An examination of the drainage basin was made using the FDEP Map Direct Consolidated Application available online (FDEP, 2009a). Approximately sixty-five National Pollutant Discharge Elimination System (NPDES) stormwater permits were issued within the contributing watershed to the Downstream 3 site. All sixty-five of these permits were for construction sites greater than 0.4 ha. Only one of the sixty-five sites was in the treatment system's drainage basin. The search revealed only two permitted wastewater discharges in the Downstream 3 basin and both were well downstream from the treatment system. Land use in the larger Downstream 3 basin is similar to that of the treatment system basin, mainly agriculture with only slight increases in residential and suburban areas. However, the small differences in land use correlate to the numerous NPDES stormwater permits for construction in the Downstream 3 watershed that could lead to increased pollutant inputs at the downstream sites. These differences make it difficult to correlate results from these sites to impacts from the treatment system. If the construction activities or wastewater discharges produce greater pollutant outputs during the pre treatment system phase, reduced pollutant concentrations would be observed during the post phase independent from treatment system impacts. These observations were based on visual examination of the permitted facility locations on a map, along with contours, flow lines, and land use and not geospatial analysis, so numbers are approximate.

Third, reduction in pollutant concentration at the downstream sites may have been due to changes in precipitation, rather than the treatment impacts of Balm Road Treatment Marsh. Figure 20 in the previous section showed that both annual average discharge and annual precipitation were less during the post than during the pre phase. As discussed earlier, the primary source of TSS, TP, and TN pollution in Bullfrog Creek is from agricultural nonpoint source pollution. The pollutants are picked up from the surrounding landscape by runoff and washed into the creek. Less precipitation in the post time frame could be the cause of lower pollutant concentrations due to less storm events to carry pollutants to the creek.

There is no strong evidence from the Mann-Whitney results that water quality in Bullfrog Creek was positively impacted by Balm Road Treatment Marsh. However, not finding significant impacts to ambient water quality at this site does not necessarily imply the treatment system is unsuccessful in treating the pollutants. The treatment system was designed to capture flows resulting from storms while leaving a fairly stable baseline flow through the creek. The wetland receives the first flush after a storm that would be expected to be high in sediments and nutrients and also captures the less pollutant concentrated waters that may be experienced during longer rain events. This water would dilute pollutant concentrations in the creek and since the wetland receives this water rather than the creek, pollutants may be more concentrated at the Downstream 1 site after some storms when compared to pre treatment system conditions. The wetland significantly altered the hydrology in the upper portions of Bullfrog Creek. The hydrological impact may have masked the pollutant reductions. Pollutant loads, however, may still be reduced, but this does not aid in determining ambient water quality impacts of the treatment system to Bullfrog Creek.

In addition, ambient water quality monitoring was not designed to establish the performance of the treatment wetland. Stormwater monitoring at the inflow and the outflow would have more accurately determined the pollutant reduction of the treatment system. Although it may be reasonable to expect pollutant reductions year-round because water from the creek is always flowing through the system, the system was designed primarily to reduce pollutant loads from agricultural runoff. Therefore, the only way to accurately measure the effectiveness of the system would be through stormwater monitoring, rather than ambient water quality monitoring.

Upstream-Downstream Comparisons. Wilcoxon matched-pairs signed-rank tests were performed to compare ambient water quality between the Upstream site and downstream sites on the creek during the post phase. Recall that the Upstream site is located near the headwaters of the creek and upstream of the treatment system, the Downstream 1 site is just downstream from the system, and the other two sites are further downstream. Both overall and wet and dry season comparisons were made. Statistically significant differences ($p < 0.05$) between the Upstream and other sites were found for the majority of the comparisons (Table 8).

Parameter and Season	N Upstream/Test Site	Median Concentration (mg/L) Upstream	Test Site	z	Significance (*p*)
		Downstream 1 Site			
TSS Combined	**32/33**	**1.99**	**4.26**	**-2.599**	**.009/increase**
TSS Wet	12/12	2.40	4.52	-1.334	.182
TSS Dry	**20/21**	**1.64**	**4.26**	**-2.203**	**.028/increase**
TN Combined	33/33	0.76	0.88	-1.832	.067
TN Wet	12/12	0.84	0.84	-1.961	.050
TN Dry	21/21	0.70	0.81	-0.852	.394
TP Combined	**33/33**	**0.099**	**0.122**	**-2.251**	**.024/increase**
TP Wet	12/12	0.181	0.160	-.471	.638
TP Dry	**21/21**	**0.078**	**0.102**	**-3.215**	**.001/increase**
		Downstream 2 Site			
TSS Combined	**32/33**	**1.99**	**0.54**	**-3.889**	**.000/decrease**
TSS Wet	**12/12**	**2.40**	**0.75**	**-3.059**	**.002/decrease**
TSS Dry	**20/21**	**1.64**	**0.50**	**-2.576**	**.010/decrease**
TN Combined	**33/33**	**0.76**	**0.50**	**-3.940**	**.000/decrease**
TN Wet	**12/12**	**0.84**	**0.63**	**-1.883**	**.060/decrease**
TN Dry	**21/21**	**0.70**	**0.44**	**-3.493**	**.000/decrease**
TP Combined	**33/33**	**0.099**	**0.253**	**-4.708**	**.000/increase**
TP Wet	**12/12**	**0.181**	**0.376**	**-2.746**	**.006/increase**
TP Dry	**21/21**	**0.078**	**0.203**	**-4.015**	**.000/increase**
		Downstream 3 Site			
TSS Combined	32/33	1.99	2.42	-1.047	.295
TSS Wet	12/12	2.40	5.42	-1.490	.136
TSS Dry	20/21	1.64	1.65	0.000	1.000
TN Combined	**33/33**	**0.76**	**0.61**	**-3.788**	**.000/decrease**
TN Wet	12/12	0.84	0.82	-1.334	.182
TN Dry	**21/21**	**0.70**	**0.54**	**-3.667**	**.000/decrease**
TP Combined	**33/33**	**0.099**	**0.200**	**-4.530**	**.000/increase**
TP Wet	**12/12**	**0.181**	**0.313**	**-2.353**	**.019/increase**
TP Dry	**21/21**	**0.078**	**0.157**	**-3.980**	**.000/increase**

Table 8. Wilcoxon matched-pairs signed-rank test results using PASW. Post-construction comparisons between the Inflow site and other sites on Bullfrog Creek downstream from the treatment system. Note: $\alpha = 0.05$ for all tests.

Pollutant concentrations were found to be greater at the Downstream 1 site for overall TSS and TP as well as dry season TSS and TP. There were differences for all test parameters between the Upstream and Downstream 2 site. TSS and TN for both overall and individual seasons were less, and TP for both overall and individual seasons were greater at the Downstream 2 site. Combined season TN and dry season TN were less at the Downstream 3 site than the Upstream site. Both combined and individual season TP increased at the Downstream 3 site.

The increased pollutant concentrations at the Downstream 1 site provides evidence that the treatment wetland had a negative impact on ambient water quality in Bullfrog Creek. There are two possible explanations: either the treatment wetland was exporting TSS and TP or something else caused the higher concentrations at the Downstream 1 site. It is highly possible that the wetland exported TP. This occurrence has been extensively noted in the literature. Soil properties prior to construction of a wetland can influence phosphorus removal. The Balm Road Marsh property was previously used for agriculture, so it is likely that the soils were high in phosphorus. High phosphorus content in soils can impact the wetland's ability to remove phosphorus (Liikanen et al, 2004). When phosphorus concentrations are low in the wetland inflow, sediments may release phosphorus back into the water column (Novak *et al*, 2004). Phosphorus retention is dependent on water depth and according to staff gauge measurements in the wetland, Balm Road Marsh water levels remained higher than anticipated. Phosphorus retention decreases as water depth increases, so higher than anticipated water levels could have diminished phosphorus removal efficiency (Moustafa, 1999). Additionally, research has shown that long-term phosphorus retention may not occur in wetland systems and storage may be only temporary (Richardson, 1984; Kadlec & Knight, 1996).

The wetland may also be exporting suspended solids. Particulates are removed from the water column by sedimentation that occurs as the water slows down in the sedimentation basin and three wetland cells. Shorter residence times may affect the amount of particulates that settle. Flowing waters pick up sediments by erosion and resuspension of bottom sediments, which are high energy actions that are unlikely to occur in the slow moving waters of the treatment wetland. However, wind and wave action have been shown to cause resuspension of sediments in shallow lakes and could have similar affects in wetlands (Kadlec &Wallace, 2009). The presence of emergent vegetation reduces resuspension by wind and waves (Horpilla & Nurminen, 2001). However, the establishment of vegetation in the Balm Marsh Treatment System had early setbacks and replanting was necessary to overcome the effects of nutria and exotic apple snails. The treatment cells still contain large open water areas, as seen in the photographs in Appendix A. Wind and wave action in the wetland could have been a factor in increased TSS concentrations at the Downstream 1 site.

Additionally, TSS measurements do not include only sediments. Other particulates including suspended algae and other organic material are included in TSS measurements. If phytoplankton is being exported from the system, it will appear in TSS results downstream (Mays, 2001). The final cell in the wetland has a large area of open water with vegetation only around the perimeter. This configuration is susceptible to high algae production which may have influenced TSS measurements at the Downstream 1 site (Kadlec & Wallace, 2009). Examining available aerials from the SWFWMD General Map Viewer, the FDEP Map Direct, and Google Earth revealed several algae blooms over the years in various cells. In 2005 there was an algae bloom throughout the entire wetland seen mainly along the shorelines (Appendix A, Figure A-2), 2007 aerials revealed what appeared to be algae mats in cell one, 2008 aerials showed a large bloom in the sedimentation basin, and 2009 aerials showed a bloom in the sedimentation basin and cells one, two, and possibly three. These observations support the possibility that algae may have contributed to TSS downstream from the wetland. In addition, a strong positive correlation between post phase Downstream 1 chlorophyll-a and TSS is evidence that algae exports influenced TSS values ($r_{s(32)} = 0.70$, $p < 0.05$).

Finding no reduction in TN is not unexpected based on inflow concentrations to the treatment system. Median TN values at the Upstream site were 0.9 mg/L, which is below the 50^{th} percentile for typical statewide stream concentrations (Table 4). There is strong evidence that wetlands either pass through or produce a background level of approximately 1-2 mg/L of organic nitrogen and up to 2.5 mg/L TN. Outflow concentrations will likely be as high as 2.5 mg/L, therefore inputs of 0.9 mg/L would not be expected to be affected by treatment and may actually increase to background levels (Kadlec & Wallace, 2009).

TSS reductions could be affected in a similar manner; the inflow concentrations are so low there is little room for improvement. During the post phase, median TSS was 1.99 mg/L which is in the 10^{th} percentile for streams in the state. This low concentration of suspended sediments is difficult to improve upon. This condition does not, however, apply to TP. Median TP at the Upstream site during the post phase was 0.11 mg/L which is above the 50^{th} percentile in statewide comparisons (FDEP, 2000). High removal efficiencies have been found at lower inflow concentrations for other constructed wetlands treating agricultural runoff, for example 80% reduction at 0.075 mg/L inflow, 76% reduction at 0.075 mg/L, and 58% reduction at 0.067 mg/L (Tanner *et al*, 2003 and SWFWMD unpublished data as cited in Kaldec & Wallace, 2009). Background concentration in the southeast is approximately 0.01 mg/L; therefore there was large margin available for TP improvement by wetland treatment.

As discussed in the previous section, examination of ambient water quality data is not the preferred method of determining treatment efficiency. Low median inflow TSS and TN does not represent the entire range of conditions that occur in the stream. As previously stated, monthly sampling occurred at the convenience of SWFWMD staff, without regard to precipitation patterns. It is likely that peak influx of sediments and nutrients, which would be expected to occur with storm events, were not captured in the dataset. Nonpoint source pollutants are typically found in highest concentrations during the first flush of a storm event after an extended antecedent dry period. Stormwater sampling would be a more appropriate choice to capture peak performance of a treatment system designed to treat pollution resulting from runoff. Flows enter the wetland year round regardless of precipitation, but the highest concentrations and therefore the best opportunity for large reductions, occur with storm events. Low inflow concentrations and poor performance during ambient monthly sampling events do not indicate poor performance over the entire range of conditions. Large amounts of pollutants may have been retained from storm flows; however the monitoring scheme was not designed to capture performance under these conditions. Interestingly, increased TSS and TP at the Downstream 1 site only were only found during the dry season. No statistically significant changes were found during the wet season.

Another possibility is that the increased TSS and TP were not exported from the wetland. There is one small tributary to Bullfrog Creek in between the Upstream and Downstream 1 site. The tributary serves a small drainage basin of mainly agricultural land and some upland forest. The water quality of the tributary is unknown, but based on the small contributing basin and similar land use to the Upstream site basin, the pollutant loads would be expected to be much smaller than those from the larger basin that contributes to the wetland. However, with no data to confirm this, the possibility remains that the tributary could contribute significantly to pollutant concentrations at the Downstream 1 site.

There were both increases and reductions of pollutant concentrations at the Downstream 2 and 3 sites compared to the Upstream site. The decreased TSS and TN concentrations are not likely due to the treatment system because there were no reductions found immediately downstream from the system at the Downstream 1 site. The reductions must have been due instead to other factors. Two possibilities are the dilution by downstream tributaries or attenuation through physical or chemical processes and assimilation as the pollutants travel downstream. Increased TP at the Downstream 2 and 3 sites may be due in part to exports from the treatment system; however concentrations are higher than at the Downstream 1 site, so phosphorus loading from either runoff from the surrounding watershed or the permitted point sources must be involved as well.

Loading Impacts to Tampa Bay

Pollutant load reductions were found at the Downstream 3 site as seen in Table 10 and Figure 21.

Year	TSS Load (kg/year)	TN Load (kg/year)	TP Load (kg/year)
Wet Season			
2002	29507	5607	2035
2003	81795	14234	6439
2004	340379	54711	21791
2002–2004 AVG	150560	24851	10088
2005	87283	11183	4100
2006	110763	17411	5350
2007	11935	2722	1041
2005-2007 AVG	69994	10439	3497
Reduction	**80566**	**14412**	**6591**
% Reduction	**54**	**58**	**65**
Dry Season			
2002	69898	11623	2776
2003	44472	9043	2962
2004	103564	14406	4604
2002–2004 AVG	72645	11691	3447
2005	42556	7346	2565
2006	4763	1805	498
2007	26189	7263	1862
2005-2007 AVG	24502	5471	1642
Reduction	**48142**	**6220**	**1806**
% Reduction	**66**	**53**	**52**
Combined Seasons			
2002	99404	17230	4811
2003	126266	23277	9401
2004	443943	69117	26395
2002–2004 AVG	223205	36541	13536
2005	129839	18529	6665
2006	115525	19216	5848
2007	38124	9985	2902
2005-2007 AVG	94496	15910	5139
Reduction	**128709**	**20631**	**8397**
% Reduction	**58**	**56**	**62**

Table 10. Average annual load reductions at the Downstream 3 site.

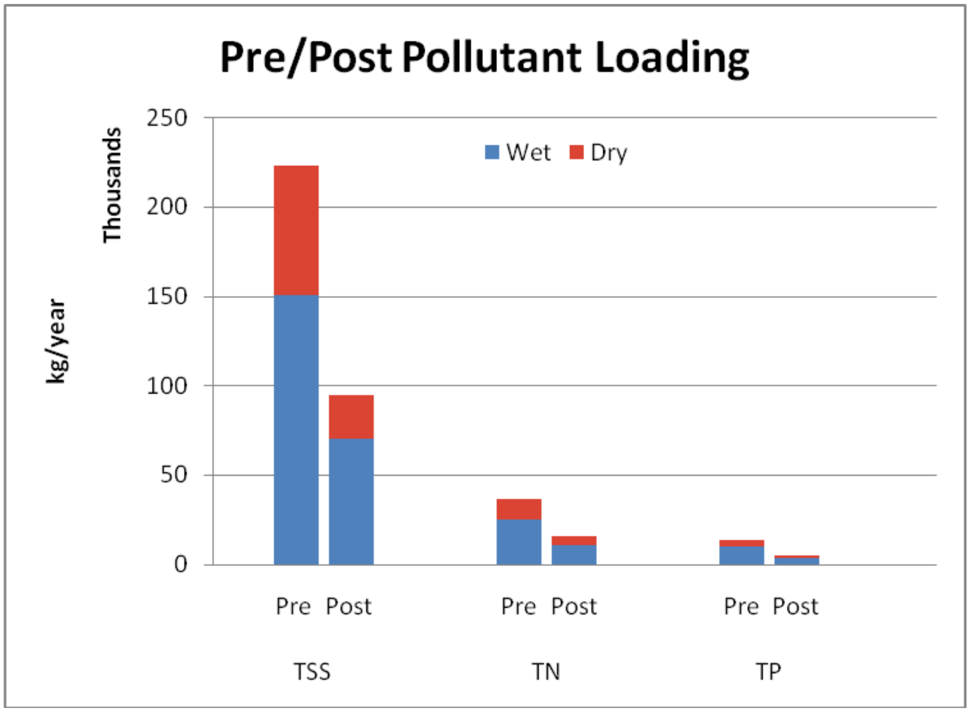

Figure 21. Pre/Post pollutant load reductions at the Downstream 3 site.

When comparing the pre and post average annual loads, reductions were 58% for TSS, 56% for TN and 62% for TP. This translates into reductions of approximately 129,000 kg TSS, 20,600 kg TN and 8,300 kg TP entering Tampa Bay each year. The calculated reduction for TSS is very near the estimated reduction based on modeling performed prior to construction of the wetland. The estimated reductions were 125,060 kg TSS, 8,700 kg TN, and 13,690 kg TP per year. However, loads were calculated based on a method to provide precision and not accuracy. The method chosen has been found to underestimate loads by as much as 80%, so the similarities between estimated and calculated load reductions may be misleading (Walling & Webb, 1981). In addition, estimated load reductions were based on modeled inflow and typical performance data and calculated load reductions were based on loads pre and post treatment wetland at a site downstream from the treatment system. These differences make comparing calculated and estimated load reductions problematic. The majority of the pollutant loading occurred during the wet season for each parameter. This is typical for pollutants whose major source is runoff from the surrounding watershed. There were load reductions in the post phase both during the wet and dry season.

Figures 22 and 23 demonstrate how discharge, precipitation, and pollutant concentration reductions contributed to load reductions. Monthly total precipitation data from SWFWMD's nearby Romp 49 Balm Park SCADA site was used to find wet season and dry season annual averages.

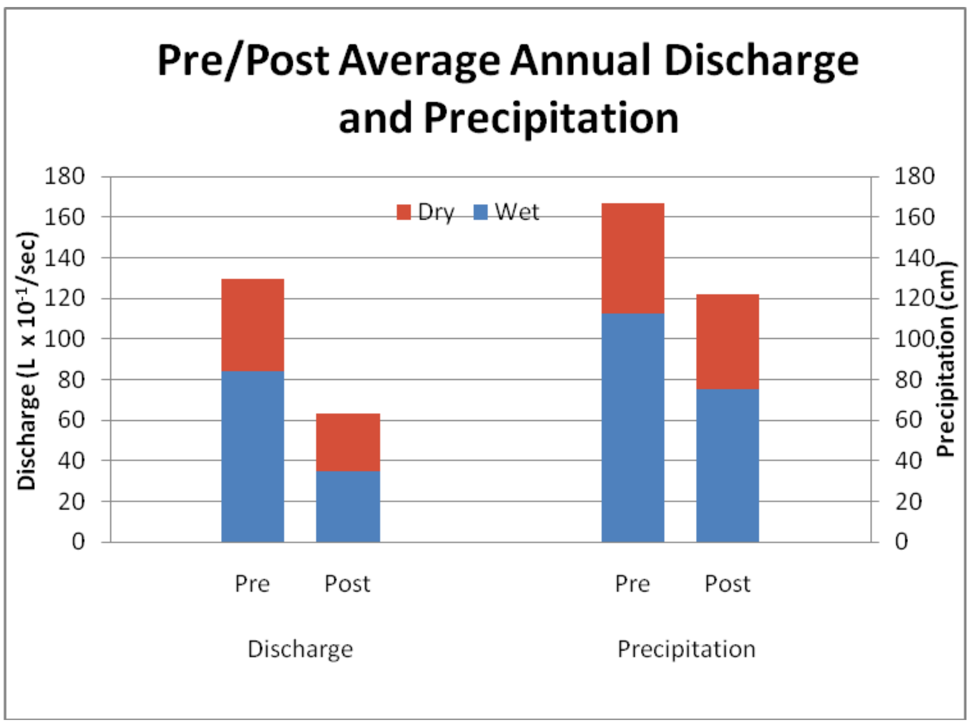

Figure 22. Pre/Post mean discharge and annual average precipitation. Discharge is from the USGS gauging station at the Downstream 3 site and precipitation is from the SWFWMD site at nearby Balm Park.

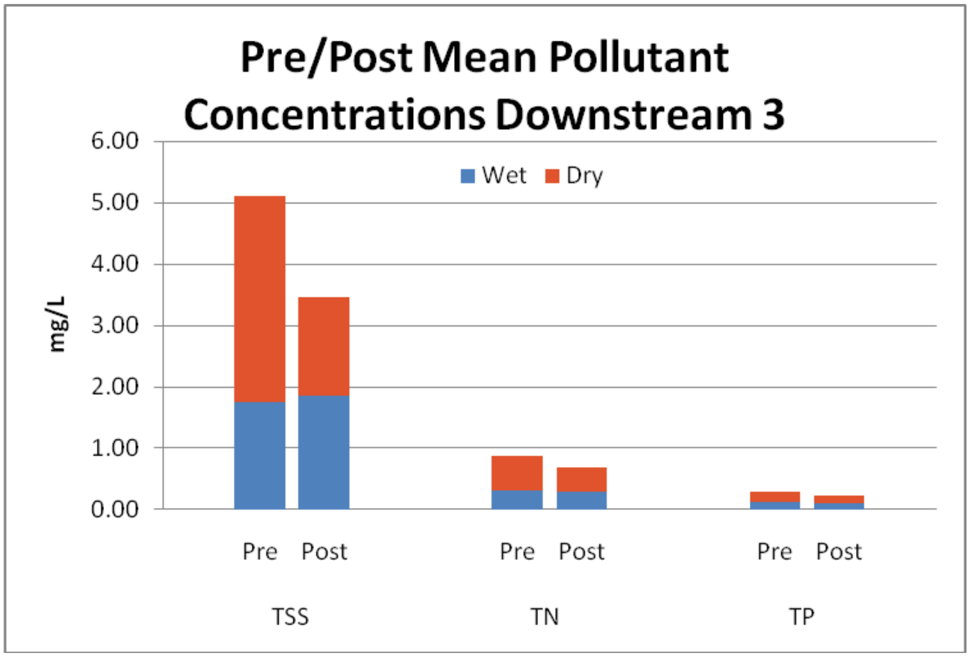

Figure 23. Pre/Post mean pollutant concentration at the Downstream 3 site.

Discharge at the Downstream 3 site decreased during the post phase. This is likely due to a decrease in precipitation during the post phase as depicted in Figure 23. Precipitation is the annual average amount for the three years during the post and pre phase, while discharge is the mean instantaneous flow at the time each sample was taken, so it is not expected that the differences in pre and post discharge and precipitation would be proportional. Both reductions in pollutant concentration and discharge during the post phase contributed to load reduction as shown in Figures 22 and 23. As shown in both Figure 23 and the Mann-Whitney results in Table 7, wet season concentrations for all three constituents remained largely unchanged, therefore wet season load reductions can be attributed mainly to decreased flow through the Downstream 3 site during the post phase, and not treatment system affects. However, dry season pollutant reductions did contribute to overall load reductions for all three parameters.

Figure 24 is provided to compare mean pollutant concentration reductions at the Downstream 3 site to reductions at the Upstream site.

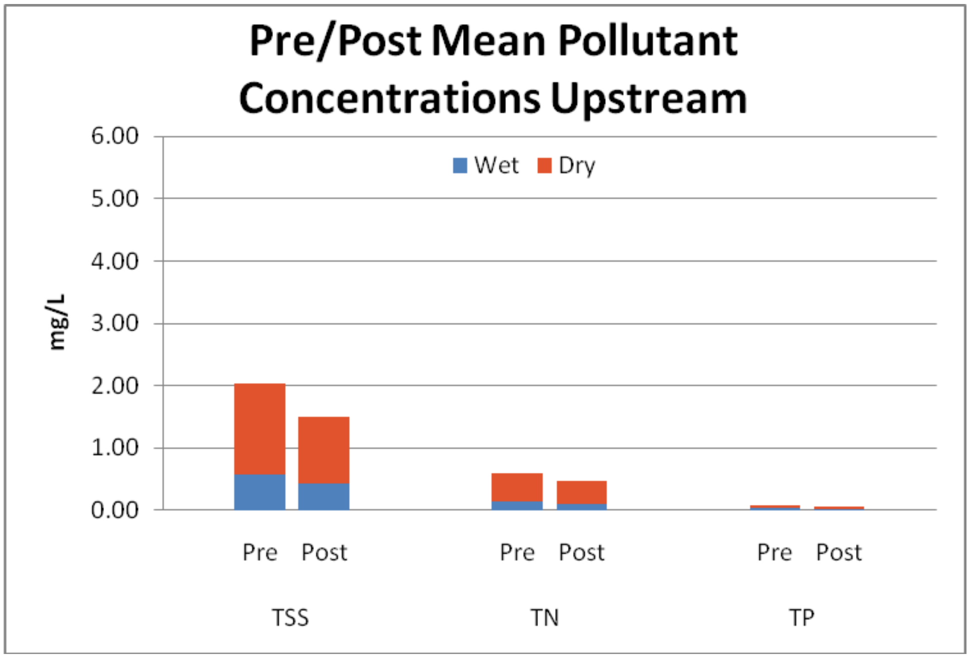

Figure 24. Pre/Post mean pollutant concentration at the Upstream site.

From the graphs in Figures 23 and 24 it appears that there were pollutant reductions at both the Inflow and Downstream 3 site when comparing pre and post phases, indicating that Downstream reductions may have been at least partially due to reductions in pollutant inputs to the treatment system, rather than affects of the treatment system. However, there were reductions during the post phase for more parameters at the Downstream 3 site than the Upstream site. The Mann-Whitney test results in Table 7 indicated that only TN during the dry season and TP during the wet season had statistically significant reductions at the Upstream site when comparing pre and post phases, whereas TSS during the dry season, both overall and dry season TN, and dry season TP demonstrated significant reductions at the Downstream 3 site. Therefore, input reductions did not contribute to all reductions at the Downstream site.

The results do not implicitly indicate that the treatment wetland was responsible for the load reductions found. Decreased discharge during the post phase, rather than a reduction in pollutant concentration may be the primary cause. In addition, decreased concentration during the dry season at the Downstream 3 site, which contributed to decreased loads, was not necessarily due to treatment by the wetland as discussed in the previous sections.

Comparison to other Wetlands

Kadlec and Wallace (2009) compiled wetland treatment performance data for constructed systems designed to treat agricultural runoff. They found pollutant reductions by comparing mean pollutant concentration entering the wetland with mean pollutant concentration leaving the wetland following storm events. The mean pollutant reductions from the compiled data were 52% TSS, 30% TN, and 22% TP. TSS reductions were present for all fourteen wetlands examined, while two out of nineteen experienced TN increases, and four out of twenty-four had TP increases. The largest TN increase reported was 11% and the largest TP increase reported was 76%.

Assuming the increase in pollutant concentrations were due to wetland impacts and not the small tributary or other factors, the performance of Balm Road Treatment Marsh can roughly be compared to other treatment wetlands found in the literature. TSS and TP medians at the Downstream 1 site were found to be higher than at the Upstream site during the post phase according to Wicoxon tests (Table 8). Although not statistically significant, both median and mean TN were also higher at the Downstream 1 site than the Upstream site. Direct comparisons to the reductions found by Kadlec and Wallace are difficult because the Downstream 1 site is composed of both wetland discharge and base flow through Bullfrog Creek and sampling was conducted without regard to storm events, but are provided to give a general idea of where Balm Road Treatment Marsh falls among other wetlands. Balm Road Treatment Marsh reductions are in Table 10.

Parameter	Mean Upstream (mg/L)	Mean Downstream 1 (mg/L)	Reduction (%)
TSS	2.94	5.75	-96
TN	0.86	0.98	-14*
TP	0.133	0.154	-16

Table 10. Pollutant reductions for Balm Road Treatment Marsh. Negative reductions represent increases. *TN changes were found to be statistically insignificant using the Wilcoxon matched pairs test.

All reductions in Table 10 are negative indicating that there were actually increases for each parameter. TP increase is well within range of that reported by Kadlec and Wallace and TN is nearly within range (2009). TSS concentration almost doubles and there were no reported increases for other wetlands.

Reductions from wetlands designed to treat agricultural nonpoint source pollution can be compared to reductions reported for wetlands treating other source water to aid in determining whether constructed wetlands are a good option for treating agricultural runoff. When Balm Road Treatment Marsh reductions are combined with those reported by Kadlec and Wallace, mean reductions are 42% TSS, 28% TN, and 20% TP with ranges from -96 to 97% TSS, -14 to 67% TN, and -76 to 60% TP (2009). However, since

Balm Road Treatment Marsh reductions were not based on stormwater sampling, they were not included in the comparisons in Table 11.

	Mean % Reduction in Constituent Concentration		
	Ag Runoff	Urban Runoff	Wastewater
TSS	52	64	72
TN	30	35	53
TP	22	44	56

Table 11. Wetland performance by source water using values from the literature.

Urban stormwater treatment by constructed wetlands was reported as having mean reductions of 64% TSS, 35% TN and 44% TP (Kadlec & Wallace, 2009). Wastewater treatment by both constructed and natural wetlands, including municipal and industrial wastes, has been reported as having average concentration reductions of 72% TSS, 53% TN, and 56% TP (Kadlec & Knight, 1996). Agricultural runoff treatment by constructed wetlands appears to be less effective than treatment of pollutants in urban stormwater or wastewater. Nonpoint source pollution, whether from agricultural or urban sources, is not treated as effectively as wastewater. The decreased reductions for nonpoint source pollution are likely due to the fact that the amount of water and pollutant concentration entering the system is highly variable over time due to the dependence on precipitation. Municipal and industrial wastewater typically has a fairly constant flow rate and pollutant concentration. The reasons for differences between agricultural and urban runoff treatment efficiency are unknown. The research on agricultural runoff treatment by constructed wetlands is limited and the reductions were calculated based on the performance of only 12 to 24 wetlands, varying based on parameter. Performance of urban runoff treatment was based on only 19 wetlands, while wastewater treatment performance was based on 48 to 71 wetlands. More research is needed to more accurately characterize the performance of wetlands treating nonpoint source pollution and determine factors affecting performance.

Chapter 8

Conclusions

Summary

The goal of this research was to determine the water quality impacts of Balm Road Treatment Marsh in order to gain better understanding of the performance of constructed treatment wetlands for agricultural pollution management. In order to accomplish the research goal, three questions were posed: What were the resulting water quality impacts of Balm Road Treatment Marsh to ambient conditions in Bullfrog Creek? Was there a subsequent pollutant load reduction to Tampa Bay? How does the performance of constructed wetlands used to treat agricultural pollution compare to wetlands used to treat other pollution? It was proposed that answering these questions would help determine whether or not constructed treatment wetlands are appropriate for agricultural pollution management, which would assist water resource managers in designing effective pollution reduction strategies for agricultural nonpoint source pollution.

Beneficial ambient water quality impacts of Balm Road Treatment Marsh to Bullfrog Creek appear to be minimal, if any. No significant changes in pollutant concentration could be found immediately downstream from the treatment wetland when comparing pre and post treatment wetland pollutant concentrations. When comparing data from upstream and downstream of the treatment wetland, some of the pollutants were actually more concentrated downstream. Pollutant reductions were found at sites several kilometers downstream from the treatment system, however due to the distance from the treatment system and large increase in contributing drainage basin to these sample sites, there are too many uncertainties to attribute the reductions to the treatment system with any confidence.

Results show large reductions in loads to Tampa Bay, but again there is not enough evidence to attribute the reductions to Balm Road Treatment Marsh. The load reductions may be due in part to decreased pollutant inputs at the headwaters of the creek and therefore fewer pollutants entering the treatment system. Load reductions were a function of both decreased pollutant concentrations and discharge, but only discharge impacted wet season load reductions.. Decreased concentrations may have been due to factors in the contributing basin rather than treatment system affects.

Pollutant reduction percentages of Balm Road Treatment Marsh were all negative, indicating there was actually an increase in pollutant concentrations downstream from the system. When comparing reported treatment wetland pollutant reductions for agricultural runoff to those of urban runoff and wastewater, agricultural runoff treatment is less effective than treatment of other pollution sources. The decreased treatment efficiency, along with the increased possibility of pollutant exports, could lead to the conclusion that constructed wetlands may not be the best option for treating agricultural nonpoint source pollution. However, available data for removal efficiencies of agricultural runoff treatment are limited and more research should be conducted before drawing conclusions. In addition, Balm Road Treatment Marsh data were not optimal for making the comparisons, since outflow data was partially composed of flows that were not treated by the wetland, and only ambient data, rather than stormwater data were available. In addition, more research needs to be done to determine why pollutant removal is less efficient and whether new technology or improved design can improve treatment.

Data Limitations and Future Needs

Unfortunately, the sample design was not optimal for determining the efficiency of the treatment system and additional sampling needs to be performed to successfully answer the research questions. Some of the proposed changed to the sampling design can be accomplished in future studies; however some of the elements recommended should have been included in the original design prior to construction of the treatment system.

Although water quality at the inflow to the treatment system was known due the close proximity of the Upstream sample site, the treatment system discharge water quality was unknown. An additional sampling site at the outflow of the wetland, prior to merging with Bullfrog Creek would have provided important information. In addition to being able to determine overall impacts to Bullfrog Creek through evaluation of the Downstream 1 site data, the treatment system impact on water flowing through the wetland could have been determined.

In addition, flow rate data at the treatment system inflow and outfall would have allowed for calculation of pollutant mass reduction. This could have been expressed as a percentage which would have allowed for additional comparison to values found in the literature for a variety of treatment system types and pollutant sources. This information would aid in determining the effectiveness of wetland treatment of agricultural nonpoint source pollution compared to other sources. A future study can be designed incorporating this site and flow rate information.

Water quality data from the small tributary to Bullfrog Creek located between the Upstream and Downstream 1 site would have provided essential information regarding the water quality impacts to Bullfrog Creek. In order to make conclusions about the impacts of the treatment system to Bullfrog Creek, it must be assumed that the influence of the tributary was minimal. More data is needed to support or contradict this assumption. Support of the assumption would have increased confidence in the conclusion that the treatment system did not positively impact ambient water quality in Bullfrog Creek immediately downstream. If data contradicted the assumption, overall impacts to the creek still could not have been determined with confidence. A future study could be designed incorporating data from the tributary. However, the main goal of the treatment system project was to decrease loading to Tampa Bay, which could have been accomplished even though significant improvements to Bullfrog Creek were not found.

Since the treatment system was designed to achieve the greatest pollutant reductions following storm events, stormwater sampling would have added valuable information. Automatic sampling devices installed at the Upstream site and Downstream 1 sites both during the pre and post phases would have allowed for a more complete analysis of impacts to the creek and loading to Tampa Bay. The samplers could be programmed to begin sampling after a specified amount of precipitation was detected. A preprogrammed volume of water per unit time would be collected, and if flow is measured as well, a flow-weighted composite sample would be analyzed for the parameters of interest. These data could have been compared both using the Mann-Whitney test to compare Downstream 1 pre and post data and the Wilcoxon matched pairs test to compare Upstream and Downstream 1 data during the post phase. It is expected that these analyses would have detected the greatest pollutant reductions. Future studies could incorporate stormwater sampling into the design; however Wilcoxon pre-post comparisons would not reflect the updated sample design.

Recommendations for Project Managers

Although no absolute conclusions were made as to the effectiveness of constructed wetlands for agricultural nonpoint source pollution treatment, the research still provides important lessons for nonpoint source pollution managers regarding impact and pollution reduction studies. Careful planning and sample design is important prior to spending large sums of money and several years collecting data to determine project impacts. In this study, monthly water quality samples were collected and analyzed for a period of nine and a half years. Although the first few years prior to construction of the treatment system were necessary to collect baseline information to aid in design, the designs were complete mid 2002, and the remaining five years were to monitor changes in ambient water quality. With the correct sample design, the project time period and number of samples could very likely have been reduced to counter the additional costs associated with additional sample sites and equipment needed to collect stormwater samples and measure flow. Monthly grab sampling is typical for ambient water quality monitoring, but perhaps not the best choice for impact studies.

One aspect of this study that does not often occur in effectiveness studies is the overall ambient water quality impact to receiving water bodies. Typically the treatment system is studied as separate and complete system and overall impacts to in-stream water quality (or other affected water bodies) are overlooked. If reducing downstream pollutant concentration is a project goal, for example to meet water quality standards, ambient water quality impacts should be studied in addition to pollutant load reductions. If major improvements only occur following storm events or if the treatment system alters flows so that pollutant concentrations are affected, improvements may not necessarily be detected in ambient water quality data. Monitoring impacts to downstream conditions is an important aspect of effectiveness studies that is often overlooked. This research demonstrates that it is an important component to aid in pollution management decisions.

Although this research did not come to definitive conclusions regarding the water quality impacts in Bullfrog Creek by the treatment system, it does appear that ambient water quality was not positively impacted. This demonstrates the importance of selecting treatment options. It has been shown that constructed wetlands do not always perform as expected, and pollutants, especially phosphorus, have been shown to be exported under some conditions. Often, stormwater treatment projects are constructed with no subsequent effectiveness studies. This research demonstrates the importance of such studies in order to fill the existing data gap, especially in treating agricultural and other nonpoint source pollution. The information will help managers select appropriate treatment options to successfully reduce pollution and limit the misuse use of resources.

This research demonstrates that constructed wetland systems to treat agricultural nonpoint source pollution may not be as effective as wetlands designed to treat other sources of pollution. Additionally, pollutant exports from these systems are possible. Although more research is needed, managers may choose to other options for reducing agricultural nonpoint source pollution until more research becomes available. For example using BMPs on individual farms to reduce the amount of pollutants reaching streams may be a better option than treatment within the watershed.

Balm Road Treatment Marsh was not found to positively impact ambient water quality in Bullfrog Creek, and although there was a significant load reduction of nutrients and TSS to Tampa Bay, it could not be attributed to treatment by the wetland with confidence. However, the sample design was lacking, and more research is recommended before final conclusions as to the success of treatment and impacts to water quality are drawn. The proposed future research will produce results that can be effectively compared with pollutant removal efficiencies of wetlands to treat other sources of pollution found in the literature. The comparisons will be useful in the determination of the appropriateness of using constructed wetlands to treat agricultural nonpoint source pollution. This research demonstrated the importance of monitoring the performance of pollution management projects, strategic sample design, and including receiving water impacts in monitoring studies while adding to the limited existing information of the effectiveness of using constructed treatment wetlands to manage agricultural nonpoint source pollution.

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Appendices

Appendix A

Pictures of Balm Road Treatment Marsh

A-1. Balm Road Marsh Property aerial, 2004.

A-2. Balm Road Marsh Property aerial, 2005.

A-3. Balm Road Treatment Marsh Sedimentation Basin 9/26/2009.

A-4. Balm Road Treatment Marsh Cell #1 9/26/2009.


A-5. Balm Road Treatment Marsh Cell # 2 9/29/2009.


A-6. Balm Road Treatment Marsh Cell #3 9/26/2009.

A-7. Balm Road Treatment Marsh Cell #4 9/26/2009.

A-8. Upstream sampling site 9/26/2009. Maintenance crews had recently removed sediments and hydrilla from the creek bed.

A-9. Diversion structure on the left and canal to Balm Road Treatment Marsh on the right 9/26/2009.

A-10. Diversion structure allowing base flow to Bullfrog Creek 9/26/2009. All additional flows are directed through the canal on the left that flows to Balm Road Marsh.

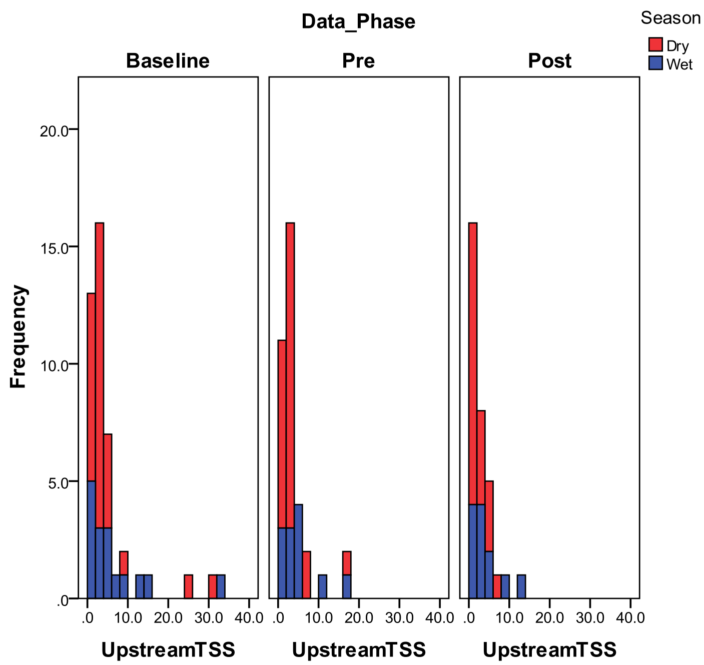
A-11. Treatment system outfall structure in cell #4 9/26/2009.

A-12. Treatment system outfall 9/26/2009. Merges with Bullfrog Creek approximately 200 m downstream.

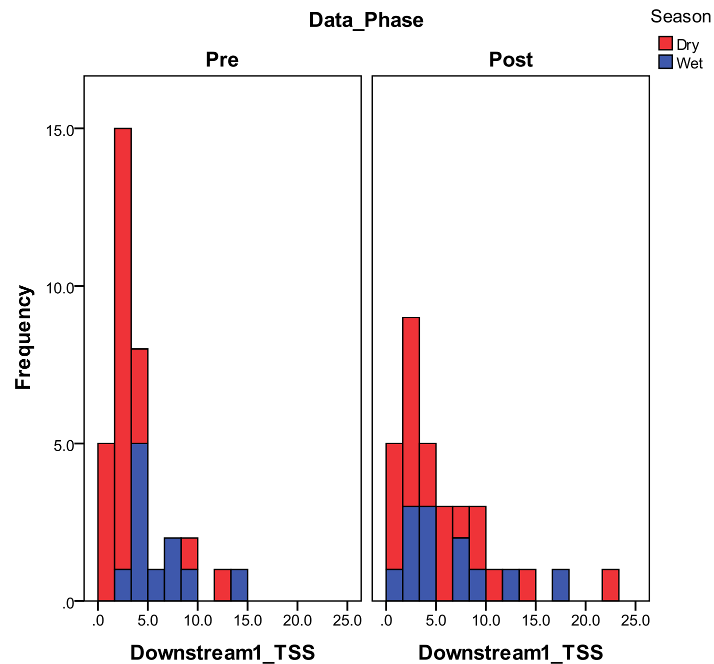
A-13. Looking upstream on Bullfrog Creek from the Downstream 1 sample site 9/26/2009. Bullfrog Creek on the right merges with the treatment system outflow on the left.

Appendix B

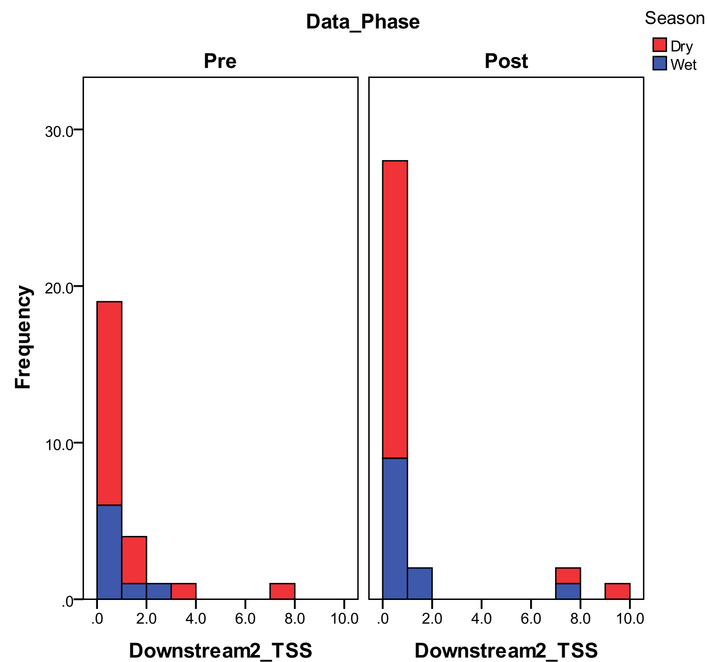
Histograms



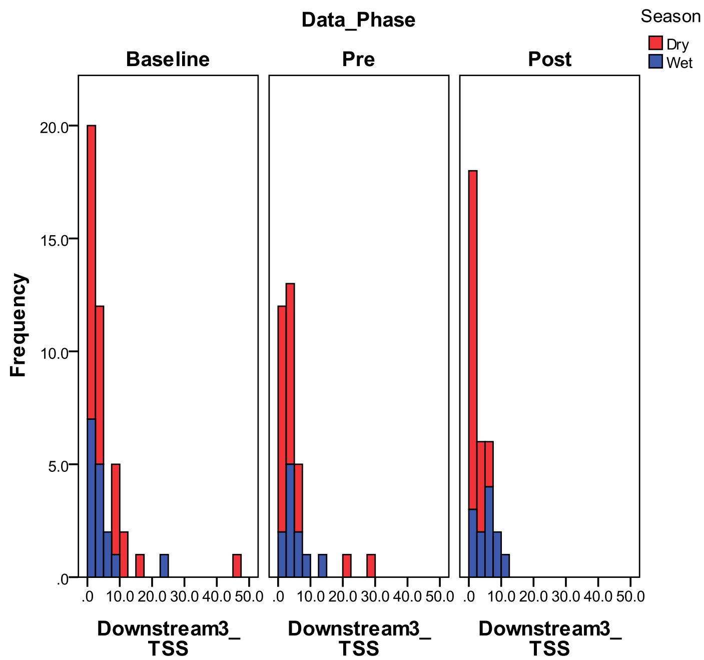
B-1. Upstream TSS Histograms.



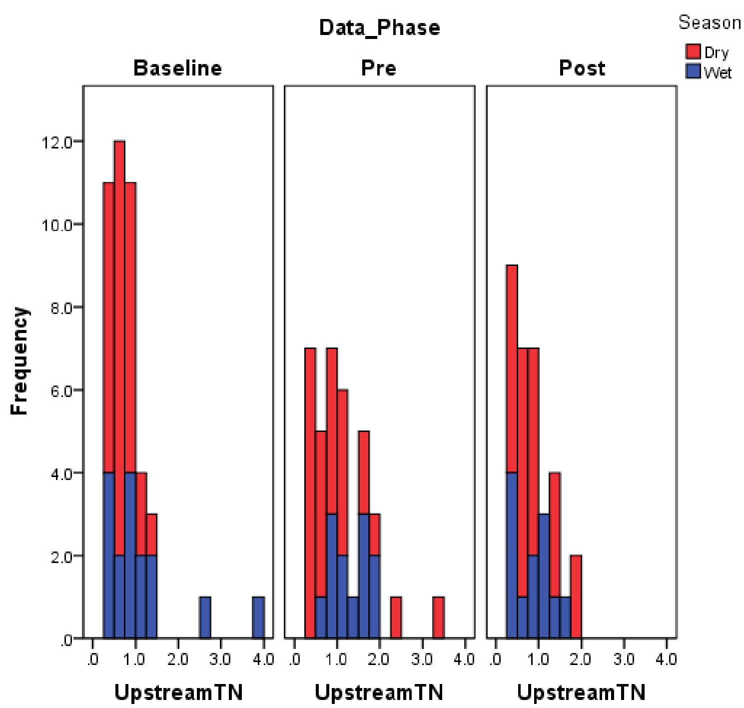
B-2. Downstream 1 TSS Histogram.



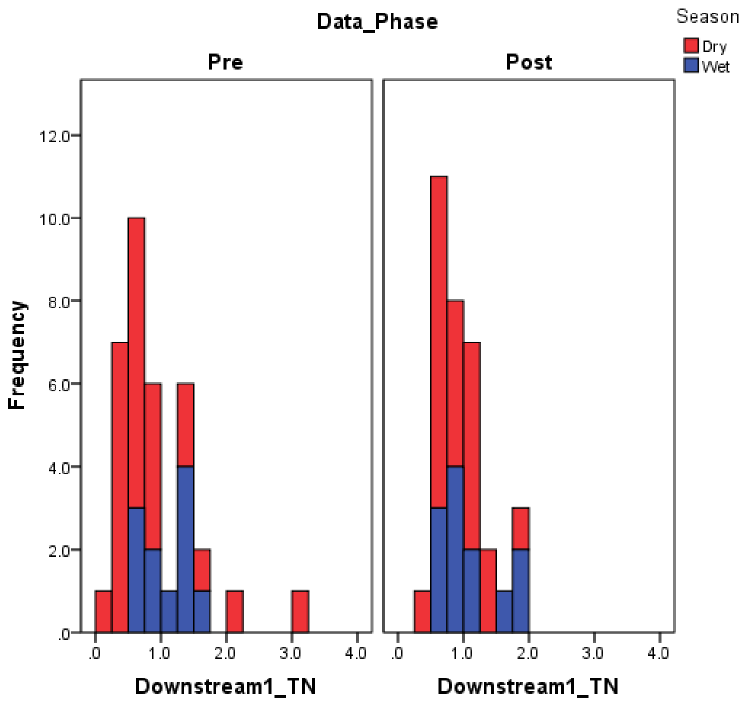
B-3. Downstream 2 TSS Histogram.



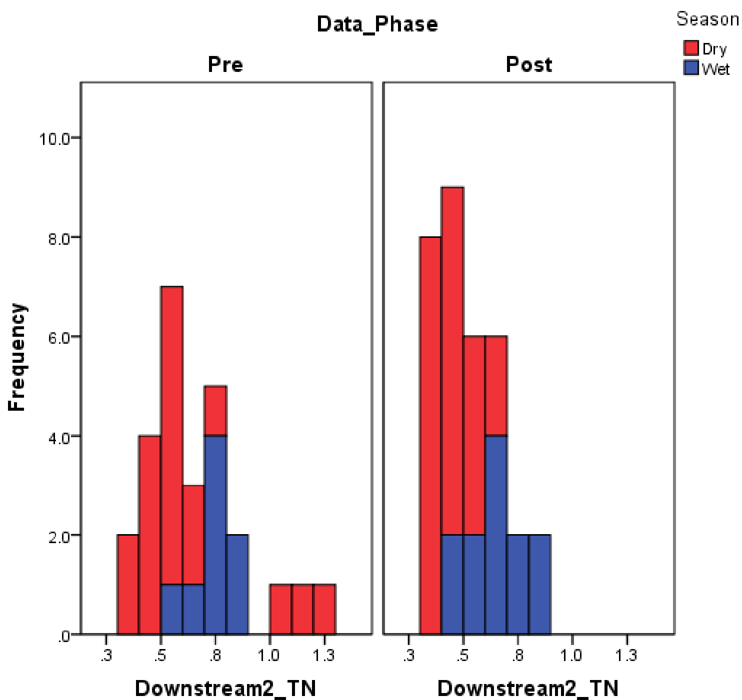
B-4. Downstream 3 TSS Histogram.



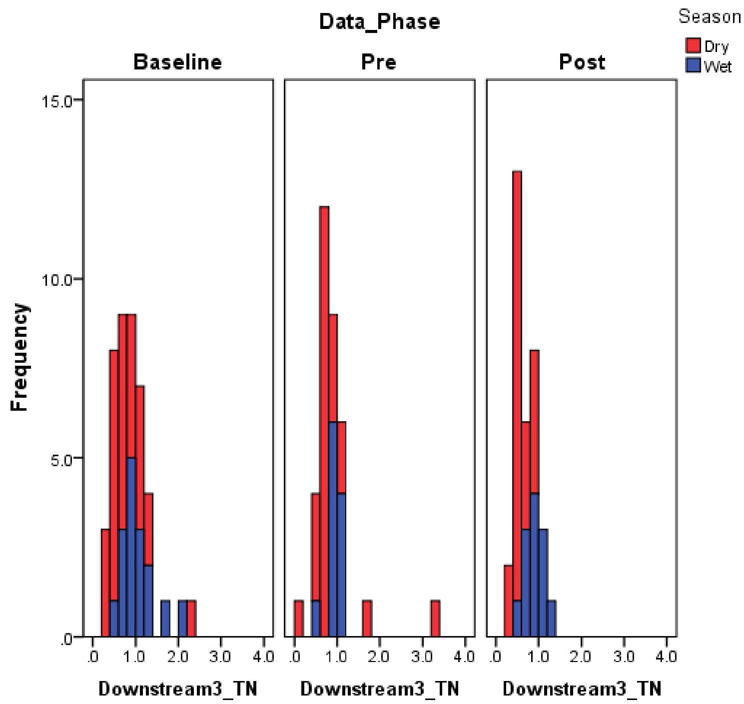
B-5. Upstream TN Histogram.



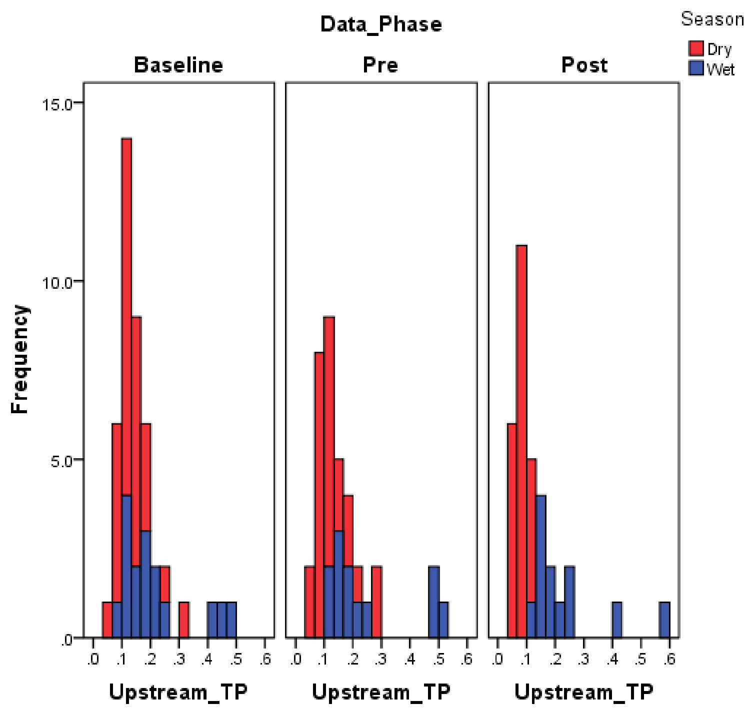
B-6. Downstream 1 TN Histogram.



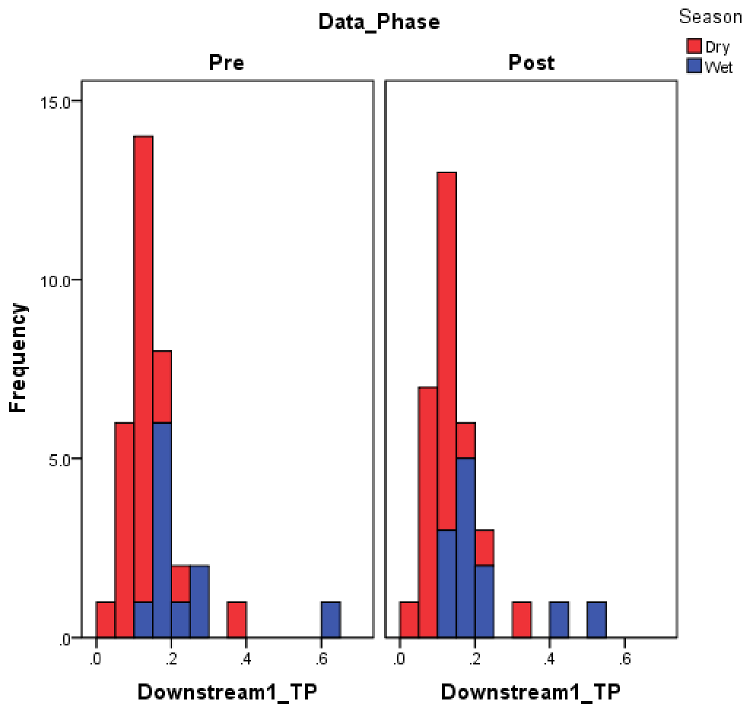
B-7. Downstream 2 TN Histogram.



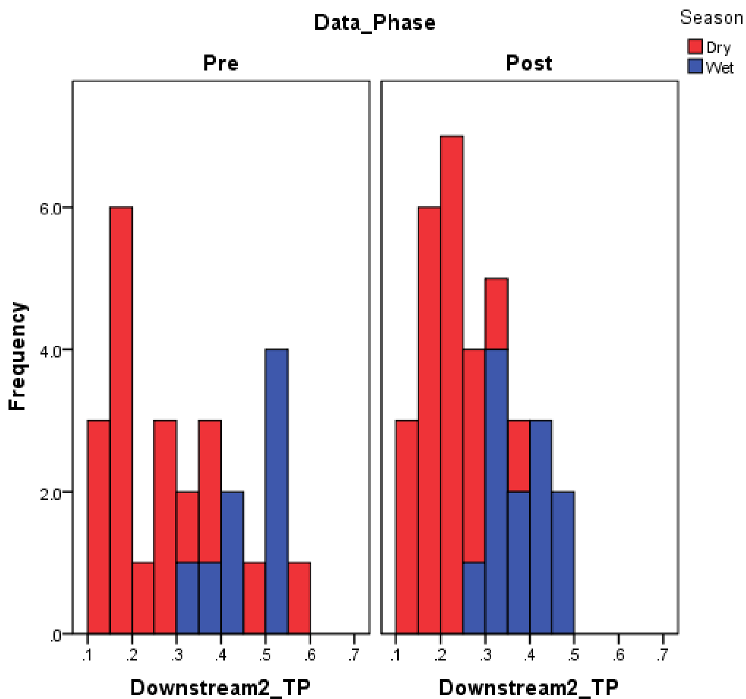
B-8. Downstream 3 TN Histogram.



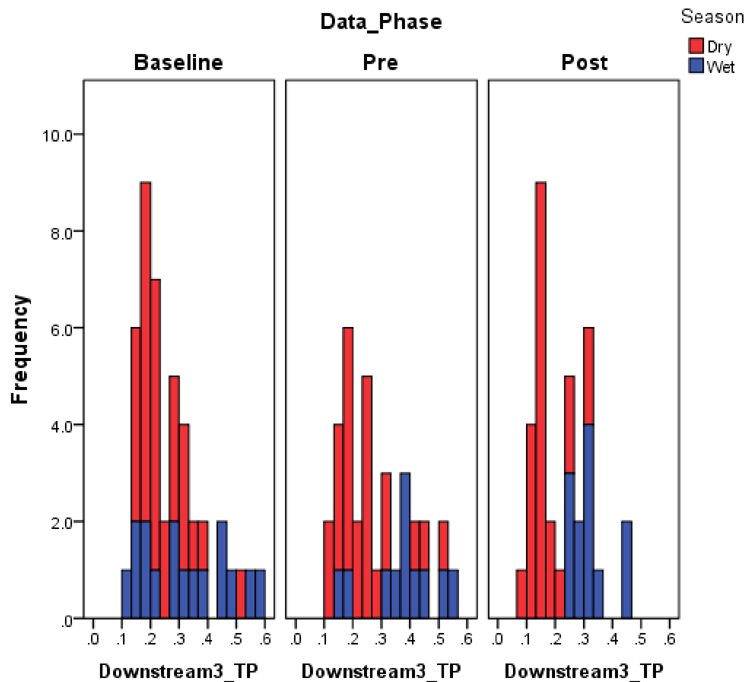
B-9. Upstream TP Histogram.



B-10. Downstream 1 TP Histogram.



B-11. Downstream 2 TP Histogram.



B-12. Downstream 3 TP Histogram.