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Innovative Pollutant Load Monitoring

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INNOVATIVE POLLUTANT LOAD MONITORING

by
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ABSTRACT

Modern streamflow measuring equipment, water quality sampling techniques and a better understanding of pollutant washoff are continuously being developed as today's society is in critical need of improving water management, minimizing developmental impacts and preventing environmental hazards. In particular, the study of the spatial, temporal and volumetric characteristics of annual pollutant loading caused by variations in precipitation, land use and other anthropogenic factors is of great significance due to their relation to future global water demands.

The research presented here falls in three parts. In the first part of the dissertation, an acoustical Doppler velocity profiler installed in a submerged concrete channel is proposed to continually measure the annual fluctuation in streamflow levels down to dry channel conditions. The tailwater influenced, intermittent streamflow conditions for the City of Kissimmee, Florida were selected for the evaluation of this approach under a 3-year study from 2006 to 2008. The performance of these concrete channels was systematically evaluated by comparisons with established field measurement techniques over various stream configurations and flow conditions.

The second part of this research investigates the dynamics of flood wave detection with respect to enabling an automatic water quality sampler to start collecting samples. The main focus was on the accurate detection of flood waves in the absence of rainfall and the presence of fluctuating baseflows and stream stages. In the 3-year study, it was shown that a dual parameter trigger, utilizing independent measuring equipment, resulted in accurate flood wave detection with minimal false triggering of the autosampler. In addition, an incremental or percent

deviation from a moving average of stage or flow proved to be a more consistent indicator for the presence of a flood wave.

In the third part of this work, the frequency of water quality sampling and the associated level of detail for sampling of rainfall events were investigated with respect to accurately depicting annual pollutant loads. It was found that the seasonal variations in baseflow pollutant loads are not accurately represented by current 4-quarter grab sampling. Also, significant pollutant loading within rainfall events may not be captured by only performing grab sampling during baseflow conditions. In addition, although increased pollutant concentrations were observed within the initial 30 minutes of the flood wave, their actual loadings did not represent a significant impact on the annual pollutant loads. A biweekly grab sampling frequency was found to be adequate in many cases to depict the annual pollutant loads, but depending upon the targeted constituent and particular streamflow condition, rainfall event sampling might also be necessary. The results of this research complemented with other studies will promote better understanding of intermittent streamflows, accurate flood wave detection, and assessment of annual pollutant loads to our nation's waterbodies.

To my family:

My sons, Christopher, Sean and Justin whom I love very deeply. I pray that my accomplishments inspire them to develop the vast potential within themselves and that they always strive to do the right thing.

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CHAPTER ONE: INTRODUCTION

Research Needs

Florida is surrounded by water, and its many internal lakes and rivers have long been recognized for their excellent fishing and boating. This notoriety draws land developers to the lake shores to establish residential and commercial infrastructure. This land development brings with it flood plain alteration, water level stabilization, and increased nutrients, which cause adverse impacts to our lakes. In response, the United States Environmental Protection Agency (EPA) passed the Federal Clean Water Act (CWA) in 1972, which set the framework for the water quality standards for the entire United States. As a result of the CWA many point sources were eliminated, but in the process it became apparent that nonpoint source loads represented even more of a threat. To further study the physical and chemical characteristics of urban runoff the Nationwide Urban Runoff Program (NURP) was established in 1978. This research lead to a series of management options, named Best Management Practices (BMPs), which proposed various structural and non-structural methods to reduce nutrient loads. But the research and data collection on the effectiveness of these systems to remove nutrients is in its infancy.

Monitoring the pollutants, which enter the tributaries of our receiving waterbodies is essential to understanding the initial pollutant loading and the effectiveness of any nutrient removal systems. Effective pollutant monitoring approaches include the accurate measurement of streamflow during baseflow conditions and in response to rainfall events. The concentrations of each constituent within the stream during both baseflow and rainfall events are also required. These individual streamflow and concentration measurements are combined and then summed to obtain the annual pollutant loads to the receiving waterbody. The results of this annual pollutant

loading analysis are used to determine Event Mean Concentrations (EMC) for various land uses and to evaluate the effectiveness of nutrient removal systems.

Accurately determining pollutant loading is difficult because the existing EMCs for land uses vary by region, are dependent on local watershed characteristics and are affected by the BMPs in place. Several factors also impede the physical measurement of pollutant loads using monitoring stations. The spatial and temporal distributions of rainfall lead to difficulties in identifying the presence and size of pollutant waves. Seasonal climatic variability also makes it difficult for the monitoring equipment to accurately measure annual flow conditions. Watershed characteristics affect the response of excess runoff from rainfall events and induce non-uniform flow conditions. Finally, anthropogenic effects interrupt natural streamflows, increase sedimentation and increase debris flow.

Numerous studies have suggested that pollutant loads can be determined by spreadsheets using regression equations and empirical EMC values. Other studies, which also rely on empirical EMC values, incorporate computer models to determine pollutant loads. These computer models typically use synthetic rainfall distributions in the absence of historical data to determine streamflow rates. Both of these empirical EMC methods are usually used for planning level analysis. More accurate physically based studies use United States Geological Survey (USGS) stage-discharge rating curves in conjunction with water quality grab samples to depict the streams annual pollutant loading.

Recently, pollutant loads have been determined by using automatic sampling equipment linked with stage measuring recorders to obtain continuous, rating curve based, streamflow data. These automatic samplers use either local rainfall depths or stream stage levels as triggers to obtain runoff pollutant concentrations from rainfall events. More advanced pollutant monitoring

stations measure in-situ stream velocity using Acoustical Velocity Meters (AVM) mounted on the stream banks or Acoustical Doppler Profilers (ADP) mounted within the streamflow. These stations use a stage-area rating curve with continuous stream velocities to provide more accurate streamflow measurements under non-uniform flow conditions.

Methods currently being developed to determine pollutant loads include non-contact approaches such as land based Photo Image Velocimetry (PIV) or spaced based Synthetic Aperture Radar (SAR). Both of these systems are used to provide continuous streamflow measurements based on the movement of surface waves or foam. Limitations to these methods include cloud cover or low lighting conditions, which prevent measurements from being collected. Research is also underway to determine pollutant concentrations from satellite imagery, but it is still in the experimental stages and decades away from implementation.

There is a gap between the availability of accurate pollutant load data for local land uses and the physically based models used to determine the Total Maximum Daily Loads (TMDLs) for receiving waterbodies. Within this gap is a need to improve the accuracy of annual streamflow measurements and to develop a rainfall event trigger to start the automatic samplers at the front of the pollutant wave.

The objective of this research was to demonstrate an innovative approach to water quantity and water quality data collection that would generate accurate and effective measurements using automatic monitoring stations. This data was collected throughout the City of Kissimmee, Florida to determine pollutant loadings into the tributaries of Lake Tohopekaliga. These monitoring sites were located such that inflows from outside the city limits could be isolated and external pollutant loads quantified. Also, additional internal monitoring sites were established to determine the pollutant loads of internal sections of the city.

Hydrologic Cycle

On earth, water exists in a liquid, solid or vapor form. In the liquid form it creates the oceans, seas, lakes, rivers and groundwater. In the solid state it exists as ice and snow cover. The atmosphere contains water in its vapor form. The energy from the sun puts all of this water into motion. This circulation of the earth's water from the land to the sky and back again is called the 'hydrologic cycle'. Figure 1 shows a depiction of the various stages in the hydrologic cycle. This process places the oceans, rivers, clouds and rain in a never-ending state of change. The total amount of water on the earth and in its atmosphere does not change but the form of water is in a continuous motion.

Although other processes in the hydrologic cycle aid in nutrient transport, the surface runoff process is the primary nutrient transport mechanism. The importance placed on the surface runoff process is due to particles of sediment being dislodged from the earth (erosion) and carried with the water until it is deposited into the receiving waterbody (sedimentation). Therefore the rate of weathering and erosion from the soils in the contributing watershed directly affects the nutrient concentrations in receiving waterbody. In fact, land use alterations in the watershed can actually serve as early warning indicators for environmental impacts to a lake (USEPA, 1996).

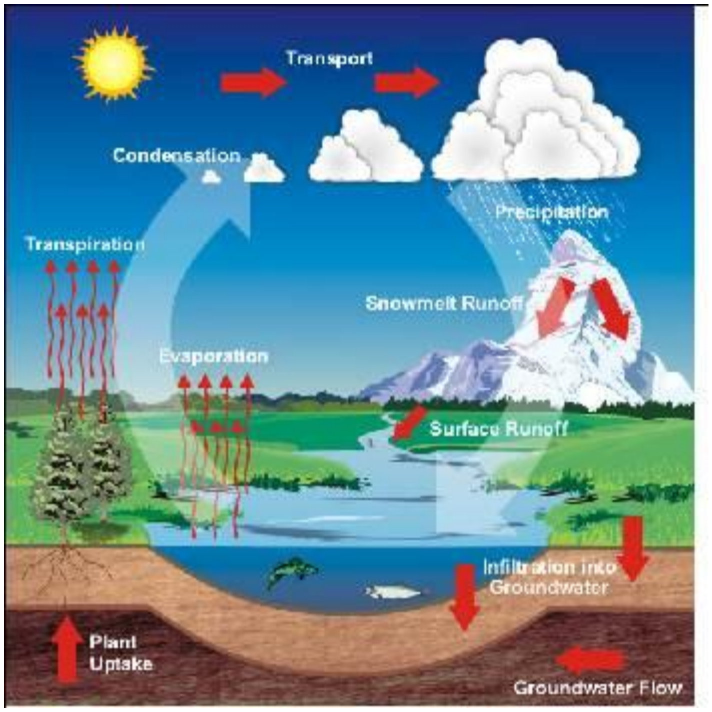

Figure 1: Hydrologic CycleUrban Development

The altering of the natural environment during the urbanization of watersheds can cause harmful side effects such as decreased infiltration of rainfall, increased runoff volumes and increased occurrences of flooding. These hydrologic factors lead to streambank erosion, which is the main transport mechanism for pollutant export to receiving waterbodies (Schueler, 1987). The influx of these nutrients carried by the runoff from developed watersheds can lead to algae blooms in receiving waterbodies.

The process of nutrient enrichment to our lakes is the most widespread water quality problem in the United States. There are two main types of these nutrient sources. The first is the point source such as a wastewater treatment plant. These point sources are easy to identify and their pollutant loads are relatively easy to quantify. The second type, referred to as non-point sources are more difficult to identify and quantify. These non-point sources come from multiple

sources including sanitary sewer leaks, septic system leachate, lawn fertilizers, agricultural wastes, highway runoff, urban development and wildlife.

The leachate from septic tank systems, runoff from highways, and agricultural land wastes provide excess nutrients such as nitrogen and phosphorus. Residential areas also contribute nitrogen and phosphorus from lawn fertilizers, grass clippings, leaves, and animal wastes. Industrial areas contain nitrogen and phosphorous in cleaning chemicals or degreasers. The quantity of these nutrient contributions is dependent upon local human population densities and the type of land use (Klein, 1975). In terms of water quality, nutrients are considered pollutants when their concentrations are sufficient enough to allow excessive growth of aquatic plants, particularly algae.

Limnology

Large quantities of nitrogen and phosphorus transported by surface runoff can enrich the nutrient levels of a receiving lake. The classification of this degree of nutrient enrichment is called 'Eutrophication' and can also be used as a measure of lake health. Eutrophication is broken down into three classifications, or levels based on nutrient concentrations. The first classification is called 'Oligotrophic' and has very low levels of nutrients, very little organic material along the lake bottom, and high levels of dissolved oxygen near the lake floor. 'Mesotrophic' lakes are the second classification with moderately enriched nutrient levels and have a natural accumulation of sediments and a normal growth of aquatic vegetation is occurring. The final classification is called 'Eutrophic', which are highly nutrient enriched, have an accumulation of organic sediments, and low levels of dissolved oxygen in water near the lake bottom. Eutrophic lakes typically have high concentrations of algae or aquatic vegetation and

also differ from oligotrophic and mesotrophic lakes in the type of vegetation and animal life that can exist in the lake.

There are also different schemes to classify the quality of lakes relative to one another. Recently, the most common method of classifying lakes is by the Trophic State Index (TSI) created by Einar Naumann. The trophic state refers to the degree or amount of enrichment, or eutrophication, the lake has with the nutrients in the water. The trophic state number is a measure of lake productivity regarding biomass, which is directly related to nitrogen and phosphorous levels. Higher nutrient levels lead to higher productivity of biomass, which in turn means a higher trophic state. Although the trophic state focuses on nutrient levels to measure plant growth, other components of the lake ecosystem, such as zooplankton concentrations are affected as well by plant growth thereby making this a good indicator of lake health.

Fish need dissolved oxygen in the water to survive. Lakes obtain their dissolved oxygen from either the atmosphere or the photosynthesis by aquatic plants. When excessive nutrients are introduced into the lake the production, death and decay of phytoplankton is increased to a level to produce the algae mats. Not only does the decay of plankton decrease the dissolved oxygen levels but the algae mats that are typically produced in the process allow very little sunlight to reach the plants. This reduced sunlight reduces or in severe cases even stops the photosynthesis process and thereby prevents the production of dissolved oxygen. When this occurs there is not enough dissolved oxygen produced during the day to compensate for normal daily uses by fish, plants and bacteria. If this condition continues until the dissolved oxygen is depleted then the fish will suffocate. In shallow ponds that are heavily vegetated and have high levels of decomposing organic matter this can occur in only a few days.

Other conditions reduce the oxygen in the water which accelerates the effects of nutrient loading to a lake. Dissolved oxygen levels are at their highest on sunny days late in the afternoon after a long period of photosynthesis. When the sun sets the production of oxygen ends, but the oxygen consumption still continues. Therefore the photosynthesis during the day must be great enough to supply the demand during the night. Cloudy weather during the day will reduce the amount of dissolved oxygen generated by photosynthesis.

Although the light from the sun is beneficial to dissolved oxygen production, the heat from the sun can create a temperature difference in the water. This temperature difference causes a stratification of the lake water with the less dense warmer water remaining at the surface and the cooler, denser water forced to the bottom. These temperature differences between surface and bottom layers may be up to 10 to 15°F. The surface water layer typically has enough dissolved oxygen, however the bottom layer will often have little or none due to the consumption of dissolved oxygen by bacteria breaking down organic matter. If any significant, sudden mixing of these two layers occurs by wind or wave action, then the oxygen deficient bottom water can cause the ponds overall dissolved oxygen to drop drastically. This condition is called 'inversion' and is a common reason for fish kills in small ponds with heavy sudden inflows.

The effects of this eutrophication process are more pronounced in watersheds with nutrient rich, heavily urbanized surface runoff. However, some studies have shown significant impacts to aquatic life in ponds with less than 10 percent urbanization. In Maryland a study was conducted on 27 small watersheds having similar physical characteristics, but varying land uses. The findings indicated aquatic life problems when at least 12 percent of the watershed was impervious and severe aquatic life problems were noted after the imperviousness reached 30 percent (Klein, 1975). Also, nineteen wetlands impacted by varying levels of urbanization were

studied in New Jersey by Ehrenfeld and Schneider. The findings showed a significant increase in nutrient impacts to the wetlands from all of the urban runoff. Finally, a study conducted by the Northeastern Illinois Planning Commission found that a majority of streams with watersheds having population densities greater than three hundred people per square mile showed signs of significant impairment.

The primary nutrient criteria variables of concern in over enrichment are nitrogen and phosphorus (EPA, 1999). Vollenweider's advances in limnology and lake management following many years of experience dealing with temperate climes and freshwater lakes has developed a general rule-of-thumb about eutrophication with regards to nutrients. Ambient total phosphorus (TP) concentration of greater than about 0.15mg/L and or total nitrogen (TN) of about 1.5 mg/L is likely to cause blue-green algal bloom problems during the growing season. This over enrichment leads to lake quality degradation in the form of low dissolved oxygen, fish kills, algal blooms, expanded macrophytes, increased sedimentation, and shifts in both flora and fauna.

Regulations

In an attempt to prevent the degradation of the lakes in country, the EPA passed the Federal Clean Water Act (CWA) in 1972 which set the framework for the water quality standards for the entire United States. As a result of the CWA, many point sources were eliminated, but in the process it became apparent that nonpoint source loads represented more than 65 percent of pollutants entering our nation's waterbodies (Rushton and Dye, 1993, Livingston, 1985). Research that began prior to the adoption of the CWA documented that a large source of nonpoint pollution is the runoff from urban and industrial areas (Whipple and

Hunter, 1977). The Nationwide Urban Runoff Program (NURP) was established in 1978 to collect basic data on the physical and chemical characteristics of urban runoff across the country (EPA, 1983).

A series of management options, named Best Management Practices (BMPs) were developed to control the pollutants transported in urban runoff (Schueler, 1987). These BMPs can be either maintenance or development practices that do not include the construction of a permanent stormwater management structure like street sweeping or Low Impact Development (LID) which are referred to as "non-structural" or they can be actual ponds, swales or physical processes which are referred to as "structural". The effectiveness of each of these BMPs varies according to the targeted pollutant, pollutant concentration, and site conditions.

Although this framework has existed for over thirty years, it has only been through the recent creation of the Florida Watershed Restoration Act in 1999 that a quantifiable stormwater quality criterion was established. This criterion is defined by the Total Maximum Daily Load (TMDL) levels, which will be set for each impaired waterbody in the State of Florida. These TMDLs have been incorporated into the Environmental Protection Agency's (EPA's) National Pollutant Discharge Elimination System (NPDES) permitting requirements and are managed by the Florida Department of Environmental Protection (FDEP).

These requirements all fall within the framework of the original Clean Water Act [40 CFR Part 130] established back in 1972 which in its current form requires each State to identify waters within its boundaries not meeting water quality standards applicable to the water's designated uses. This list of identified waters, referred to as the 303(d) list, must be submitted to the EPA for review and approval. The "listed" waters identified by the State are prioritized for Total Maximum Daily Loads (TMDL) development based on factors described in CWA

regulations, such as the use of the water and the severity of pollution. A separate TMDL is established for each pollutant at a level necessary to attain the applicable water quality standards taking into account seasonal variations and a margin of safety. The TMDL establishes allowable loadings of pollutants for a water body based on the relationship between pollution sources and in-stream water quality conditions. With this information, States can establish water-quality based controls to reduce pollution from both point and nonpoint sources to restore and maintain the quality of their water resources (USEPA, 1996).

These regulations and procedures are useful in identifying whether a tributary or its receiving waterbody is impaired, and if not, how to have it delisted. But what if the tributary is found to contribute pollutants to its receiving waterbody and is actually impairing the health of the lake? A means of accurately identifying these harmful nutrients entering the waterbody must be implemented so that the lake can be restored back to a more natural state.

Stormwater Monitoring

The two most common approaches to water quality monitoring in a watershed are the influent-effluent constituent monitoring approach and the watershed monitoring approach. The three commonly used types of watershed approaches are upstream-downstream, before and after, and paired watershed (Coffey, 1993). These watershed monitoring approaches are typically used only when the physical constraints of a site do not permit the adoption of an influent-effluent approach. However, these watershed approaches are useful in wide scale applications to evaluate the effectiveness of nonstructural BMPs such as street sweeping.

In contrast, the influent-effluent approach is the most effective method for estimating the pollutant removal efficiency of an individual, structural BMP. This is because pollutant removal

efficiencies are based on calculating the difference between influent and effluent loads (Urbonas, 1994). Since the locations of the sampling points are immediately upstream and downstream of the BMP, it makes it possible to isolate the pollutant loads for the mass balance calculations. This simplicity in evaluating BMPs is not the only benefit of the influent-effluent approach. The monitoring costs are substantially less since very few additional environmental factors need to be factored into the overall evaluation to determine the BMPs effectiveness. Also, the time needed for monitoring can be substantially less and, since it is an isolated analysis, the evaluation results of a particular BMP can be extrapolated to other local systems. One drawback to the influent-effluent approach is the difficulty of establishing any downstream benefits of the BMP without additional data being collected from the receiving waterbody.

Once the monitoring approach is selected there are numerous ways to actually collect and prepare the samples. The two most commonly used samples types are flow-proportional and flow-weighted. The flow-proportional sample is the most common type of composite sample. It consists of constant sample volumes taken at time intervals which are spaced in proportion to the volume of flow passing by the collection point. The flow-weighted sample are a series of samples taken at equal time increments which are composited in proportion to the volume of flow since the last time the sample was collected.

For this research, a set of core indicators (e.g., water quality parameters) for the study area, which includes physical, chemical, and biological attributes of the tributaries, were defined. The core indicators were selected to reflect general parameters of the water resources field so they can be used to assess attainment of applicable water quality standards throughout the basin. These indicators are monitored to assure that the fundamental parameters that affect the impairment of water quality in an aquatic environment are accurately assessed.

First Flush

For baseflow, a representative grab sample is collected for water quality analysis. For rainfall events, the concentrations for not only a flow-weighted composite sample are measured, but an additional sample for the first flush of pollutants is collected for analysis. The concept of first flush was initially established almost a century ago when horses were the primary mode of urban transport (Metcalf and Eddy, 1916). Referred to as "first foul flush", it described the initial rainfall volume that transported horse fecal solids from roads into the receiving streams.

The first flush phenomenon is typically associated with smaller watersheds with relatively high impervious surfaces. During the higher intensities of initial rainfall, pollutants deposited prior to the storm are washed off in high concentrations. Although first flush can occur in larger watersheds with less impervious surfaces, the longer travel times allow runoff from adjacent areas to comingle and can reduce the affect (Kim et al., 2005). There is also a seasonal first flush affect, which refers to storms that occur at the end of a dry season and wash off a disproportionally larger mass of pollutants that have collected due to the lack of rainfall (Kim et al., 2004).

The time interval for the first flush can vary with each watershed, but a period of 30 minutes is typically used for the initial volume of concentrated pollutant runoff. In a study performed by Kim et al. (2007), it was found that the accumulated pollutants were essentially washed off from the paved surfaces within the initial 15 to 20 minutes of rainfall. In the same study, the final concentration of all pollutants dropped as much as 50% from their initial first flush levels due to wash off effects.

The first flush affect is a mass limited pollutant transport process where concentrations are controlled by the amount of pollutants available for transport according to the following equation (Sheng et al., 2008).

$$\Delta M_T = M_0(1 - e^{-K_1 V_T}) \qquad \text{Eqn. 1}$$

Where: ΔM_T is the mass transported, M_0 is the initial mass available for transport, K_1 is a transport rate constant and V_T is the transport volume.

When the rainfall amount is sufficient to where the wash-off rate is no longer a critical factor in limiting pollutant transport, it becomes a flow limited transport process. Under flow limited transport, the wash-off rate is unrelated to the amount of mass remaining on the surface and follows Equation 2 (Sheng et al., 2008).

$$\Delta M_T = K_0 V_T \qquad \text{Eqn. 2}$$

Where: ΔM_T is the mass transported, K_o is the transport rate constant and V_T is the transport volume.

Most watersheds start off as mass limited, but as the rainfall amounts significantly increase, they typically transition into flow limited transport. For larger and more complex watersheds the first flush effect is short lived to the point of even non-existent (Sansalone and Cristina, 2004). When cumulative mass versus cumulative volume curves are constructed, mass limited pollutant transport tend to follow a first-order exponential pattern whereas flow limited pollutant transport exhibited a linear pattern (Sheng et al., 2008).

Thirteen separate urban watersheds with distinct types of residential and industrial development were selected for a stormwater runoff monitoring study on first flush effects (Lee et

al., 2002). A distinct mass limited transport pattern of first flush was found for the pollutants of all 38 storms monitored; however, no correlation was found between the first flush phenomenon and antecedent dry weather period. The results of this study can be found in Figure 2 and 3.

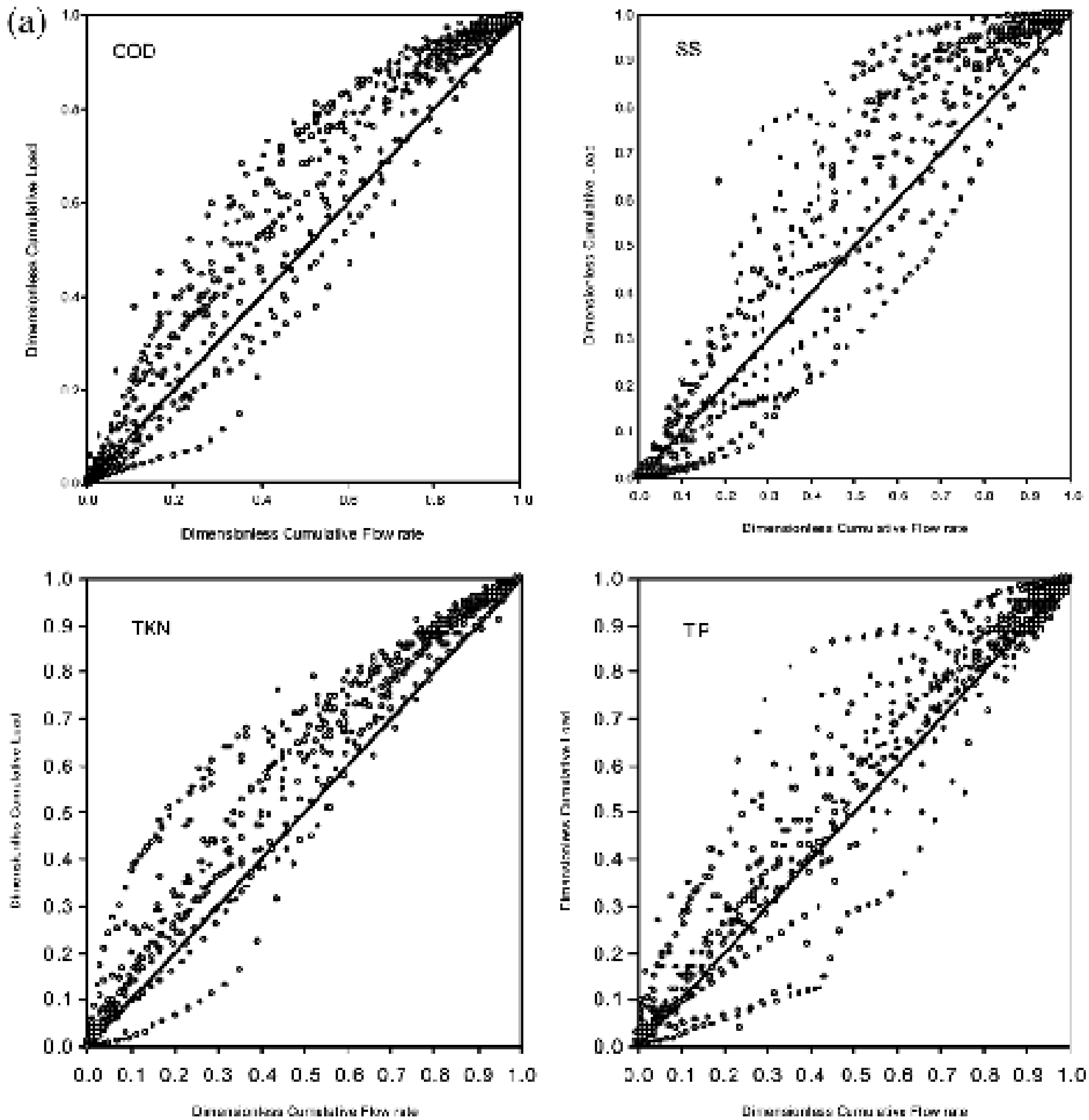

Figure 2: Cumulative Flow Rate vs. Cumulative Load (1 of 2) (Lee et al., 2002)

(b)

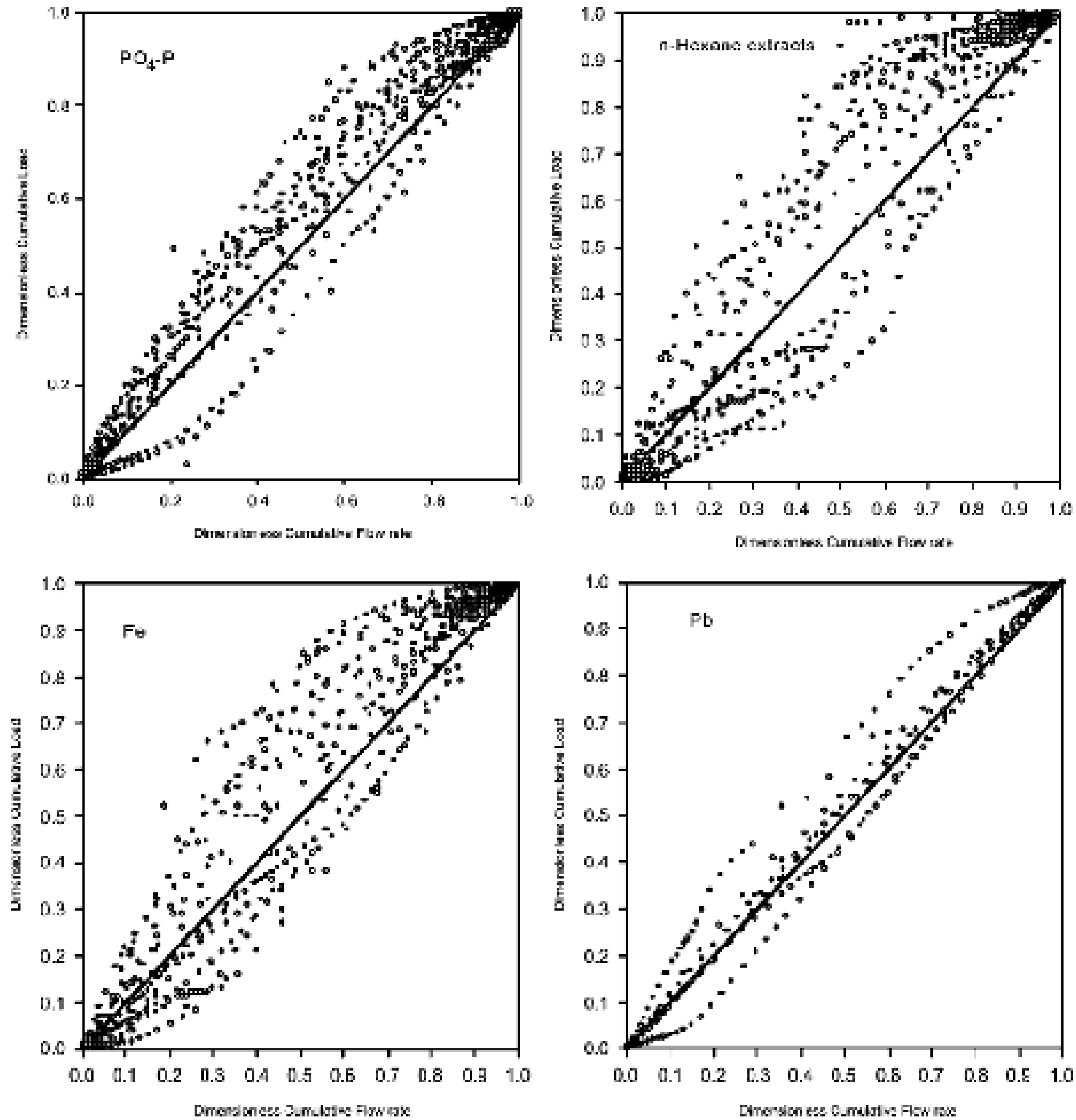

Figure 3: Cumulative Flow Rate vs. Cumulative Load (2 of 2) (Lee et al., 2002)

Mass Balance

The total mass of nutrients being transported during an interval of time is called the nutrient load. An analysis of this nutrient loading to determine the loss or gain of mass between two points is called a mass balance. The United States Environmental Protection Agency proposed two different mass balance methods for computing nutrient removal efficiency in a lake. The first method, called the average event mean concentration efficiency ratio (E_{emc}), uses an average of the event mean concentrations from all of the samples distributed over the sum of the sample volumes. The (E_{emc}) is expressed as percentages and is computed as follows:

$$E_{emc} = (1 - AEMC_{out} / AEMC_{in}) \times 100 \qquad \text{Eqn. 3}$$

Where: AEMC is the average event mean concentration and the subscripts "out" and "in" refer to outlet and inlet, respectively.

Loads are computed as the product of event mean concentrations and the associated volume. Since the average event mean concentration efficiency method averages all of the event volumes, it gives equal weight to each storm event.

The second method, called the summation of loads efficiency ratio (E_{sol}), sums the product of each sample volume multiplied by its corresponding event mean concentration. The (E_{sol}) is expressed as percentages and is computed as follows:

$$E_{sol} = (1 – SOL_{out} / SOL_{in}) \times 100 \qquad \text{Eqn. 4}$$

Where: SOL is the summation of loads and the subscripts "out" and "in" refer to outlet and inlet, respectively.

Loads are computed as the product of event mean concentrations and the associated volume, but unlike the average event mean concentration method, sample data is required for each events input and output loads.

Although, both of these methods are independent of the number of samples collected and assume their results represent the storms that normally occur in the region, the summation of loads method also assumes the collected samples represent all significant input and output loads (Martin, 1986). A comparison of these two methods found them to yield similar results, with the average event mean concentration method producing slightly lower values (Martin, 1986). Even though the average event mean concentration method is capable providing efficiencies of BMPs, the summation of loads method was found to be a better measure of the overall efficiency of a BMP (Martin, 1986).

Pollutant Load Measurement

In the absence of physical measurements, pollutant loads for various land uses are typically estimated based on EMC values developed in other regions. These pollutant loads are routed through the study area using either a simple weighted spreadsheet or more rigorously through the use of hydrologic computer simulation. In either case, the estimated EMC values are usually adjusted to account for different watershed characteristics and varying levels of pre-treatment, however, they often fail to represent the actual local pollutant loads.

In order to obtain a more accurate representation of the local pollutant loads, the actual flow rates and concentrations should be measured. The concentrations within the streamflow can be measured manually using grab samples or remotely by using automatic sampling equipment (autosamplers). Typically, grab sampling is used to collect baseflow concentrations and either

grab sampling or autosamplers are used to collect rainfall event concentrations. Stream flow rates can be estimated indirectly using stage data or measured directly using a combination of velocity and stage data.

The simplest direct stream flow measurements are volumetric methods such as the "bucket and stopwatch method" (Hauer and Lamberti, 1996) or "dilution streamflow gauging" (Noppeney and Kranenburg, 1989), but they are not suited for continuous streamflow measurement. In addition, they are only applicable for relatively steady-flow conditions without tailwater influences. More practical approaches for streamflow measurement make use of continuous stage-gauging data. In fact, most historical streamflow records are not based on direct measurement of streamflow, but are derived from continuous measurements of stream stage (Hirsch and Costa, 2004). Continuous stage data is usually collected by hydraulic reactions which are typically measured within a stilling well to minimize the effects of turbulence or wave action.

The availability of continuous stage data allows the streamflow to be continuously measured with the installation of weirs or flumes. Weirs are typically used in irrigation channels because they are ideally suited for low head-loss conditions where the discharges are very low. Whereas flumes are installed to measure larger ranges of flows without introducing large head losses. In either structure, streamflow is proportional to the height of water built up on the upstream side of the weir or flume. Although weirs and flumes can both provide continuous streamflow measurements from shallow flows down to even dry conditions, they both are highly vulnerable to clogging from debris, and high tailwater conditions complicate if not negate the streamflow measurements.

A method that makes use of continuous stage data to measure streamflow without the clogging potential of weirs is the "slope-area method" (Kuusisto, 1996). This method determines the difference in water surface elevations from upstream and downstream gauge stations and divides them by the distance between the gauge stations to yield the energy slope in Manning's equation.

$$Q = \frac{cAR^{2/3}S^{1/2}}{n} \qquad \text{Eqn. 5}$$

Where: Q is the streamflow, A is the cross-sectional area, R is the hydraulic radius, S is the energy slope, n is the streambed roughness and c is a constant.

Because Manning's equation is based on the kinematic assumption, the friction forces must be in complete balance with gravitational forces. Although this method is continuous and can accommodate very shallow depths, the required steady, uniform flow conditions rarely occur in natural streams especially when the streams have shallow slopes and are impacted by variable downstream conditions.

When accurate values of the streambed roughness are not available, but accurate, discrete streamflow is measured, then the "stage-discharge method" (USGS, 1982; Maidment, 1993) can be used. This method is also based on Manning's steady, uniform flow assumptions; however, instead of calculating the energy slope it assumes a one-to-one relationship between stream stage and discharge. The graphical presentation of a rating curve has the following form:

$$Q = a(H - H_0)^b \qquad \text{Eqn. 6}$$

Where: Q is the discharge, H is the stream stage, a and b are constants, and H_0 is the stage at which the discharge is zero.

Although this "stage-discharge method" has limitations similar to the "slope-area method", it does introduce the concept of determining streamflow using current velocity measurements.

The discrete streamflow measurements used in conjunction with stream stage data to form the rating curve are typically obtained using the "velocity-area principle". This approach uses the continuity equation to determine the streamflow (Q) as a function of the cross-sectional flow area (A) and the mean velocity ($\bar{u}$):

$$Q = A\bar{u} \qquad \text{Eqn. 7}$$

If the flow is steady, fully turbulent and one-dimensional, then the velocity profile can be approximated by the log-law-of-the-wall (Chow, 1959). This approximation yields a one-dimensional velocity profile, which varies as a parabola from zero at the stream bottom to a maximum near the surface. With this profile, the mean velocity along a vertical line can be determined by taking the average of the velocities at two-tenths and eight-tenths depths below the stream surface. Since the velocity is not constant across the width of the stream, the velocity measurements must also be made at multiple points across the width of the stream. For this procedure to be accurate, the streamflow must be relatively constant throughout the period it takes to conduct these velocity measurements.

Although the discrete velocity measurements for the "stage-discharge method" are conducted during a steady flow condition, the same stage level can actually have different flows depending on whether the stream stage is rising or falling during a flood wave. This condition is called a "looped rating curve" (USGS, 1982). Loops in the rating curves can also be created

when stream gauging stations are affected by variable downstream conditions such as a fluctuating tailwater. Since using rating curves to determine the streamflow produces significant errors, approaches using continuous velocity measurements to estimate streamflows have become more popular.

Determining streamflow based on continuous velocity measurements removes the limitation of natural streams subjected to varying tailwater conditions. If accurate, shallow to dry streamflow measurements can be provided by one of these continuous velocity and stage measurement approaches, then the annual TMDL pollutant loads during both baseflow and rainfall events can be determined. The main obstacle to overcome in these approaches is to obtain accurate measurements in combination with an uninterrupted flow of data.

Continuous velocity measurements can be made using methods in which the equipment either does or does not actually contact the stream. Non-contact methods estimate flows based on stream surface velocities such as “Particle Image Velocimetry” (PIV), which analyzes video tapes for the movement of naturally occurring foam as a tracer (Creutin et al., 2003). Another non-contact method, called “Surface Radar”, uses aerial mounted, low-power radars to measure the doppler shift of surface waves, which are analyzed with first-order Bragg scattering (Teague et al., 2005). A similar aerial radar technology , called “Space Radar Altimetry”, uses along-track interferometric synthetic aperture radar (along-track InSAR) to measure the doppler shift of surface waves from a space mounted platform (Romeiser, 2008). The main advantages of these non-contact methods are the elimination of equipment clogging from debris and disruption of flow by interference from equipment. However, the drawbacks of these methods include inadequate lighting, effects of surface winds, inadequate reference points and changes in stream bathymetry. These drawbacks affect both accuracy and the continuous flow of data.

The simplest direct stream contact method is the “Surface Float Method”, which estimates the mean stream velocity using a partially submerged float between two known points. In addition to obviously not meeting the continuous measurement requirement, this method can also be significantly affected by surface winds (Kuusisto, 1996). Acoustical Doppler equipment mounted on a floating platform, called “BoogieDopp” (BD), have been used successfully to measure the stream velocity in small, shallow rivers (Cheng and Gartner, 2003), but they are more applicable to obtaining discrete streamflow measurements in lieu of continuous data.

Permanently mounted “Acoustical Doppler Profilers” (ADP) provide a continuous velocity measurement along the channel axis by using two acoustical beams pointed upstream and downstream, respectively (Ward et al., 2007). Although this method provides both accurate and continuous velocity measurements, it requires a minimum flow depth of two inches over the instrument called “blanking distance” to operate properly. In addition to this limitation, undesirable acoustic energy, called “side lobes”, occur naturally and propagate from the transducer and angle away from the main beam which can lead to inaccuracies in shallow streamflow measurements.

The only method currently in practice that is capable of continuously measuring streamflow down to dry channel conditions without clogging is the “Stage-Discharge Method”. That also explains why it is the most commonly used method despite its limited applicability. The ADP provides the most accurate, continuous streamflow measurements, however it has certain physical measurement limitations.

Factors that Affect Pollutant Load Measurement

Pollutant load monitoring can be affected by numerous factors including atmospheric conditions, watershed characteristics and anthropogenic impacts. Global atmospheric conditions such as the El Ninõ / Southern Oscillation (ENSO) are caused by water temperatures in the eastern Pacific Ocean along the equator rising by as much as ten degrees (Brolley et al., 2007). These elevated water temperatures peak around Christmas, increasing rainfall around the South American region and create a shift in world-wide weather patterns. The resulting impact of this altered weather pattern typically lowers the likelihood of hurricane activity in Florida; however, it is not always the case. During a strong El Ninõ season in 1992, the category five Hurricane Andrew impacted South Florida. The opposite global effect known as La Ninã creates a higher likelihood of hurricane activity in Florida.

Regional atmospheric conditions such as the Humid Subtropical climate produce significant amounts of precipitation year-round within Florida. In fact, Central Florida has the highest annual number of days with thunderstorms in the United States (Williams et al., 1992). Large storm fronts traveling from west to east, called “Westerlies”, deposit rain from November through April and the migration of the inter-tropical convergence zone creates convective rainfall from May through October (Baigorria et al., 2007). Hurricanes also exert an infrequent but significant influence on seasonal and annual rainfall totals. Another significant influence with and opposite to annual rainfall totals are periods of dry days, which occur during late fall and early spring.

Localized atmospheric conditions can create large variations in spatial and temporal rainfall distributions. Florida regularly experiences seabreeze (SB) fronts and rainfall-induced, outflow boundaries (OB) due to its unique geographical and meteorological conditions. The

interactions between SB and OB, coupled with their associated convergence, provide lift to initiate convection (Shepherd et al., 2001). Lines of single-cell and multi-celled thunderstorms can form squall lines, containing newer cells on the leading edge of the front and weakening cells on the trailing edge (Yuter et al., 1994). Heavy rain and frequent lightning within these fronts typically lasts anywhere from 1 to 2 hours and can contribute up to 40% of Florida's total annual rainfall (Shepherd et al., 2001).

The rainfall in Florida is not only temporally distributed, but also has a strong spatial distribution. Heymsfield et al. (2000) found that more than three-quarters of the storm cells from convective precipitation in Florida are less than 3 mi. and Goldhirsh and Musiani (1986) suggest that the median convective cell size is slightly over 1 mi. The intensity of rainfall within these convective cells also varies exponentially in space throughout the width of their fronts due to varying vertical velocities in the form of strong updrafts and downdrafts.

Pollutant load monitoring can be affected by various watershed characteristics such as the response of runoff from excessive rainfall called "lag time". The area, shape, slope and surface cover of the watershed affects the time it takes for the excess rainfall to reach a channelized flow. A related watershed affect called "travel time" depends upon the conveyance between the channelized flow and the sample measuring point. The watersheds lag time and travel time are also a function of the intensity, duration and spatial coverage of the rainfall. As the basin area, land use and physical terrain vary between watersheds, the shape of the runoff hydrograph can be significantly altered.

The geological properties of the watershed can also have a significant effect on pollutant load monitoring. A major component of pollutant transport pertains to watersheds with relatively fine soils that are destabilized by intense rainfall resulting in nutrient rich sediments

being conveyed into streams. The volume of pollutants and size of sediment transported to the stream are directly related to rainfall intensity. Depending on the soil properties of the receiving stream, it also can be source of sediment transport through the process of scouring. In fact, scouring causes the cross-section of many alluvial streams to be significantly altered throughout the year.

The complexity of a watershed can also affect pollutant load monitoring. The runoff hydrograph from a simple watershed will respond significantly different than a watershed with multiple subbasins. Lateral inflows into the stream from multiple subbasins can cause a deviation from a simple rainfall driven runoff hydrograph of a single basin. The convolution of runoff from the individual subbasins can lead to multiple peaks in the streamflow hydrograph even when impacted by relatively stable rainfall intensities. Given the spatial and temporal variations of rainfall, the streamflow hydrograph in complex watersheds can be very irregular in comparison to a simple watershed hydrograph.

The effects of anthropogenic activities within the watershed, such as increasing impervious surfaces and improving hydraulic performance, combine to reduce infiltration and increase excess rainfall runoff rates. Both of these impacts lead to reduced stream baseflows and increased flood waves. Stormwater management systems constructed to minimize downstream hydrologic impacts from these effects can reduce stream baseflows even further and elevate tailwaters.

Pollutant loads are increased through urbanization in the form of highways, buildings and agriculture. Vehicular activity is a major source of zinc, copper, lead and hydrocarbons from brake wear and leaking engine oils (Adams et al., 2007). Roof surfaces contribute microbiological contaminants from bird feces and organic matter from atmospheric deposition.

Pathogens are associated with septic tank leachate and agricultural runoff which is a public health hazard. Agricultural and residential developments increase nutrient loadings such as nitrogen and phosphorous. Solid waste and small debris are generated by most urban development which is conveyed to receiving streams during heavy rainfall.

<u>Dissertation Objectives and Organization</u>

The main goal of this research is to develop an innovative monitoring approach that can collect accurate, continuous water quantity and water quality data. This research also intends to verify the suitability of existing water quality monitoring methods to predict the annual impacts of pollutant wash-off loads for both baseflows and rainfall events. The results of this research would also provide more accuracy in EMC values and complement the EPA stormwater quality database in Florida. The specific objectives of this study are:

1. Develop a monitoring approach to continuously measure year-round water quantity in shallow to dry streamflow subjected to non-uniform and sediment rich flow conditions (Chapter 2).

   Accurate water quantity and water quality measurements are needed to determine actual pollutant loading to the tributaries of our nation's lakes. The increased impervious surfaces from urbanization have caused intermittent streams to become very shallow or even dry during baseflow conditions. These very shallow stream depths hinder the ability to obtain the needed continuous, accurate velocity measurements. Many pollutant load studies, like the Nanticoke River in 2004, have been forced to stop collecting samples because the water levels were too low (Andres et al., 2007). Low stream depths also can make it difficult to grab water quality

samples during non-rainfall events. Additionally, the construction of stormwater management systems to reduce downstream flooding lowers the baseflow even more and also increases tailwater effects, which complicate flow measurement.

The increased runoff rates caused by urbanization have also lead to increases in erosion and sediment transport. This erosion has affected the morphology of many streams, which make it difficult to maintain a constant stream crosssection. A stable stream crosssection is necessary for the stage-area relationship to determine accurate flow measurements. Flow measurements are further impacted by this suspended sediment as it settles to the stream bottom and contributes to clogging of the in-situ velocity equipment.

2. Develop a trigger to enable an autosampler at the start of a pollutant wave, in absence of rainfall, and subjected to seasonal fluctuations in baseflow and tailwater (Chapter 3).

   Accurate water quality measurements during a rainfall event require the triggering of the automatic sampler to occur at the start of the pollutant wave. Extreme variability in the temporal rainfall patterns of Florida can affect the lag time and travel time of runoff hydrographs during storm events. These effects on the runoff hydrographs, combined with urbanization and complex watershed configurations, can delay the response of streamflow from rainfall inputs and create multiple imbedded pollutant waves. Current autosampling triggers require a threshold of rainfall depth, stream stage or flow rate to be established prior to the occurrence of the rainfall event. The resulting erratic behavior in flow levels and discharge rates of these pollutant waves

often causes the autosampler to either falsely trigger or entirely miss the pollutant wave.

Florida rainfall patterns can also have extreme spatial variability, which can generate a significant pollutant wave at an upstream station while depositing little to no rainfall at a downstream monitoring station. Unless an effective, non-rainfall autosampler trigger is installed at the downstream monitoring station, the pollutant wave will not be measured. Autosamplers triggered by only rainfall rates will obviously not detect the pollutant wave in these situations. To solve this problem, some autosamplers are triggered by change in stage or flow rate, but large seasonal fluctuations in baseflow rates and tailwater elevations also make these triggers ineffective.

3. Develop local land use based pollutant concentration loads based on actual wash-off response from baseflow and rainfall event sampling to determine an effective water quality sampling frequency (Chapter 4).

   The current database of pollutant loadings is insufficient to determine the impact of a particular land use, surface cover or management practice for many of our nation's lakes. Accurate water quality measurements for various land uses are needed to determine the actual pollutant loadings from our local watersheds. Due to the effect of spatial and temporal rainfall distributions in Florida on pollutant wash-off rates, year-round samples must be collected from both baseflow and rainfall events. In addition, the water quality grab sampling must be frequent enough to capture the fluctuations in pollutant concentrations of the stream baseflow.

> For larger, more complex watersheds, the rainfall event sampling must include an analysis of the first flush of pollutant wash-off in addition to a composite sample of the runoff into the receiving streams. This first flush can be represented by the initial 30 minutes of runoff volume; however, the variability of storm duration can be problematic for the accurate collection of composite samples. Water Quality sampling of the rainfall event must collect sufficient volumes for sample analysis while ensuring the entire pollutant wash-off is captured for an accurate flow-weighted composite.

Prior to data collection, the locations and configurations of each monitoring site had to be determined based on the topographic, hydrologic and land use characteristics of the study area. Each location for sample collection also needed to be evaluated to determine the appropriate water quality sampling method and corresponding collection apparatus. A Pilot Study was conducted on one of the internal wet detention ponds within the City of Kissimmee to verify the effectiveness of the grab sampling methods used in this innovative monitoring approach, the results of which are provided in Appendix F.

Study Area

The study area encompasses the corporate limits of the City of Kissimmee located in Osceola County, Florida which has a population of approximately 55,000 residents. Adjacent portions of Osceola and Orange counties were also included in this study to define the points where stormwater flows in to and out of the City of Kissimmee. This area was chosen because of its significant contribution of water flow into Lake Tohopekaliga. Lake Tohopekaliga is located in the upstream portion of the Upper Kissimmee Watershed.

The study area encompasses approximately 20 square miles of surface area with a relatively flat topography and poorly drained soils. A mixed land use of residential, commercial and agricultural can be found throughout the City of Kissimmee. Stormwater runoff in the city is conveyed to Lake Tohopekaliga by 6 distinct tributaries which receive flow from the runoff of their respective watersheds.

Shingle Creek is the largest of these tributaries, which has its headwaters in Orange County and discharges along the western side of the City of Kissimmee into Lake Tohopekaliga. Shingle Creek is mostly rural and the lower portions, which flow through the City of Kissimmee, are undeveloped wetland floodplains. The second largest tributary flowing through the City into the lake is Mills Slough, which is located towards the east side of the city and has its headwaters in southern Orange County. Bass Slough is located at the eastern side of the City of Kissimmee and has its headwaters in northern Osceola County. Both Mills Slough and Bass Slough are mostly residential land uses. East City Ditch, West City Ditch and Downtown Area are the final 3 tributaries and have their headwaters completely inside the city limits. East City Ditch is a mixture of residential and light commercial land use. West City Ditch is a mixture of residential and light industrial land use. The Downtown Area has mostly a light commercial land use. The watersheds and land uses are depicted in Figure 4.

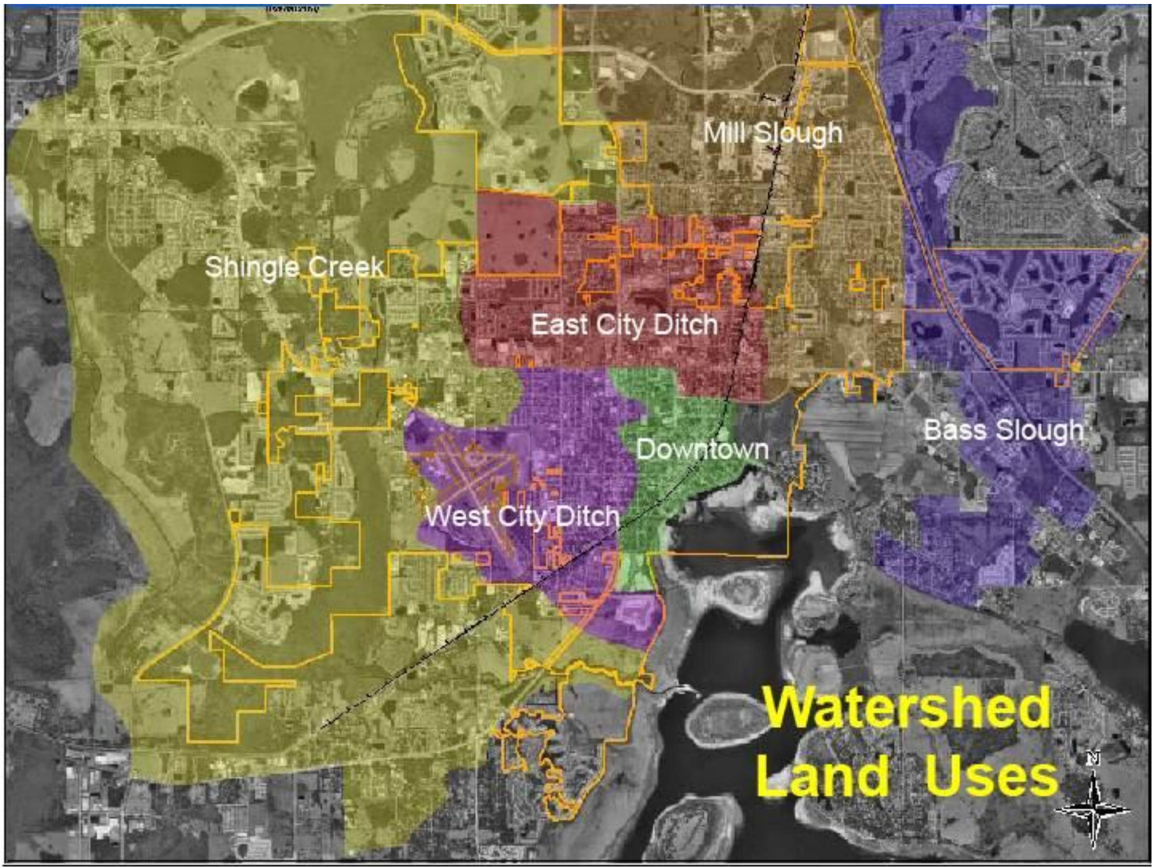

Figure 4: Watershed Land Uses

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CHAPTER TWO: CONTINUOUS DRY STREAM FLOW MEASUREMENT

Introduction

Fishing and boating in our nation's internal lakes and rivers have long attracted land developers in their vicinity to establish residential and commercial infrastructure. The altering of the natural environment during the urbanization of watersheds can cause harmful side effects. Building impervious structures reduces the area available for soil infiltration, which increases the quantity of stormwater runoff. Altering the ground slope and surface cover of the watershed reduces the time-of-concentration, which increases peak runoff rates. These two factors cause accelerated streambank erosion which is the main transport mechanism for pollutant export to receiving waterbodies (Schueler, 1987). The influx of these pollutants carried by the runoff from developed watersheds can lead to severe water quality impacts. The magnitude of water quality impacts are more pronounced in watersheds with nutrient rich, heavily urbanized surface runoff. However, some studies have shown significant impacts to aquatic life in ponds with less than 10 percent urbanization (Klein et al., 1975).

Nutrients carried by the runoff from urban development can result in large blooms of microscopic algae (phytoplankton) in the lake. As these algae die and settle to the bottom of the lake their decomposition depletes the lake's dissolved oxygen. Not only does the decay of plankton decrease the dissolved oxygen levels but the algae mats that are typically produced in the process allow very little sunlight to reach the plants. This reduced sunlight lowers, or in severe cases even stops the photosynthesis process and thereby prevents the production of dissolved oxygen. When this occurs there is not enough dissolved oxygen produced during the day to compensate for normal daily uses by fish, plants and bacteria. If this condition continues

until the dissolved oxygen is depleted the fish will suffocate. In shallow ponds that are heavily vegetated and have high levels of decomposing organic matter this can occur in only a few days.

In recognition of these pollutant impacts, the Environmental Protection Agency (EPA) is continually identifying impaired waterbodies and establishing Total Maximum Daily Load (TMDL) levels for each of them. These TMDLs set allowable loadings of pollutants for a waterbody based on the relationship between pollution sources and in-stream water quality conditions. Given this information, EPA establishes water quality based controls to reduce pollution from both point and nonpoint sources to restore and maintain the quality of the nation's water resources (USEPA, 1989). To establish these water quality pollutant loads, EPA requires not only water quality data but an accurate measurement of the corresponding flow at the monitoring locations. These accurate flow measurements need to be collected over a wide range of stream flow conditions which vary throughout the year with changing season.

The rates of rainfall during storm events are generally more intense than during non-storm events and typically give rise to higher than average combinations of pollutant concentrations and flows. However, the runoff volume from storm events typically represents only a small portion of the total annual volume of water flowing into our receiving waterbodies. Since pollutant loading is a result of both concentration and volume, we cannot limit our focus to only storm events. The year-round base flow with its corresponding concentration is a significant component of pollutants reaching our lakes and needs to be included to adequately assess the health of our lakes.

Urbanization has created an additional problem to overcome. Over time, the extensive paving of pervious surfaces in conjunction with hydraulic improvements to existing channels has caused many streams that were naturally perennial because of prolonged baseflow to become

intermittent. The United States Geological Survey (USGS) conducted a survey in 1990 of more than 7,000 gauging stations across the nation to assess the range of flows and conditions in which their current meters would operate accurately. At the time of the survey, over three-quarters of the discharge measurements made that year were in depths shallow enough to wade with the mean depth of all measurements being less than two feet (Fulford, 1992). That year almost eleven percent of the measurements made were not reliable because the stream was too shallow to trust the results. Although new technological breakthroughs in velocity meters have been made since the 1990's, the latest devices still cannot directly measure velocity down to the extremely shallow depths found in many baseflow conditions.

Background

Since streamflow is measured in units of volume per time, the most direct approach would be to simply determine the time it takes to fill a container of known volume. This method is referred to as the "bucket and stopwatch method" (Hauer and Lamberti, 1996). Although this direct volumetric analysis is accurate and straight forward, it is limited to small, clear, confined, cascading, steady-flowing streams and does not provide the continuous, instantaneous streamflow data necessary for meeting the TMDL requirements.

An alternative method called "dilution streamflow gauging" (Noppeney and Kranenburg, 1989) involves injecting a tracer into the stream and determining the flow based on dispersive properties at a downstream sampling point. This tracer can be a chemical or dye which is injected as either an instantaneous "slug load" or applied continuously. Dilution streamflow gauging can be used on small, shallow streams, however it is usually restricted to steady-flow

conditions since a predictable estimation of the velocity is necessary to accurately determine the turbulent mixing length.

The previous volumetric methods are not suited for continuous streamflow measurement. In addition, they are only applicable for relatively steady-flow conditions without tailwater influences. More practical approaches for streamflow measurement make use of continuous stage-gauging data. In fact, most historical streamflow records are not based on direct measurement of streamflow, but are derived from continuous measurements of stream stage (Hirsch and Costa, 2004). Continuous stage data is usually collected by hydraulic reactions which are typically measured within a stilling well to minimize the effects of turbulence or wave action.

The availability of continuous stage data allows the streamflow to be continuously measured with the installation of a weirs or flumes. Weirs are typically used in irrigation channels because they are ideally suited for small, low head-loss conditions where the discharges are very low. Whereas flumes are installed to measure larger ranges of flows without introducing large head losses. In either structure, streamflow is proportional to the height of water built up on the upstream side of the weir or flume. Although weirs and flumes can both provide continuous streamflow measurements from shallow flows down to even dry conditions, they both are highly vulnerable to clogging from debris and high tailwater conditions complicate, if not negate, their streamflow measurements.

A method that makes use of continuous stage data to measure streamflow without the clogging potential of weirs is the “slope-area method” (Kuusisto, 1996). This method determines the difference in water surface elevations from upstream and downstream gauge

stations and divides them by the distance between the gauge stations to yield the energy slope in Manning's equation.

$$Q = \frac{cAR^{2/3}S^{1/2}}{n} \qquad \text{Eqn. 8}$$

Where Q is the streamflow, A is the cross-sectional area, R is the hydraulic radius, S is the energy slope, n is the streambed roughness and c is a constant.

Because Manning's equation is based on the kinematic assumption, the friction forces must be in complete balance with gravitational forces. Although this method is continuous and can accommodate very shallow depths, the required steady, uniform flow conditions rarely occur in natural streams especially when the streams have shallow slopes and are impacted by variable downstream stream conditions.

When accurate values of the streambed roughness are not available, but accurate, discrete measurements of the streamflow are, then the "stage-discharge method" (USGS, 1982; Maidment, 1993) can be used. This method is also based on Manning's steady, uniform flow assumptions, but instead of calculating the energy slope it assumes a one-to-one relationship between stream stage and discharge. The graphical presentation of a streams stage–discharge relationship is called a "rating curve" and has the following form:

$$Q = a(H - H_0)^b \qquad \text{Eqn. 9}$$

Where Q is the discharge, H is the stream stage, a and b are constants, and H_0 is the stage at which the discharge is zero. Although this "stage-discharge method" has limitations similar to

the “slope-area method”, it does introduce the concept of determining streamflow using current velocity measurements.

The discrete streamflow measurements used in conjunction with stream stage data to form the rating curve are typically obtained using the “velocity-area principle”. This approach uses the continuity equation to determine the streamflow (Q) as a function of the cross-sectional flow area (A) and the mean velocity ($\bar{u}$):

$$Q = A\bar{u} \qquad \text{Eqn. 10}$$

If the flow is steady, fully turbulent and one-dimensional, then the velocity profile can be approximated by the log-law-of-the-wall (Chow, 1959). This approximation yields a vertical velocity profile which varies as a parabola from zero at the stream bottom to a maximum near the surface. With this profile, the mean velocity along a vertical line can be determined by taking the average of the velocities at two-tenths and eight-tenths depths below the stream surface. Since the velocity is not constant across the width of the stream, the velocity measurements must also be made at multiple points across the width of the stream. For this procedure to be accurate, the streamflow must be relatively constant throughout the considerable amount of time it takes to conduct these velocity measurements.

Although the discrete velocity measurements for the “stage-discharge method” are conducted during a steady flow condition, the same stage level can actually have different flows depending on whether the stream stage is rising or falling during a flood wave. This condition is called a “looped rating curve” (USGS, 1982). Loops in the rating curves can also be created when stream gauging stations are affected by variable downstream conditions such as a fluctuating tailwater. Since using rating curves to determine the streamflow produces significant

errors, approaches using continuous velocity measurements to find streamflows have become more popular.

Determining streamflow based on continuous velocity measurements removes the limitation of natural streams subjected to varying tailwater conditions. If accurate, shallow to dry streamflow measurements can be provided by one of these continuous velocity and stage measurement approaches, then the annual TMDL pollutant loads during both baseflow and rainfall events can be determined. The main obstacle to overcome in these approaches is to obtain accurate measurements in combination with an uninterrupted flow of data.

Continuous velocity measurements can be made using methods in which the equipment either does or does not actually contact the stream. Non-contact methods estimate flows based on stream surface velocities such as “Particle Image Velocimetry” (PIV), which analyzes video tapes for the movement of naturally occurring foam as a tracer (Creutin et al., 2003). Another non-contact method, called “Surface Radar”, uses aerial mounted, low-power radars to measure the doppler shift of surface waves which are analyzed with first-order Bragg scattering (Teague et al., 2005). A similar aerial radar technology , called “Space Radar Altimetry”, uses along-track interferometric synthetic aperture radar (along-track InSAR) to measure the doppler shift of surface waves from a space mounted platform (Romeiser, 2008). The main advantages of these non-contact methods are the elimination of equipment clogging from debris and disruption of flow by interference from equipment. However, the drawbacks of these methods include inadequate lighting, effects of surface winds, inadequate reference points and changes in stream bathymetry. These drawbacks affect both accuracy and the continuous flow of data.

The simplest direct stream contact method is the “Surface Float Method” which estimates the mean stream velocity using a partially submerged float between two known points. In

addition to obviously not meeting the continuous measurement requirement, this method can also be significantly affected by surface winds (Kuusisto, 1996). Acoustical doppler equipment mounted on a floating platform, called “BoogieDopp” (BD), have been used successfully to measure the stream velocity in small, shallow rivers (Cheng and Gartner, 2003), but they are more applicable to obtaining discrete streamflow measurements in lieu of continuous data.

Permanently mounted “Acoustical Velocity Meters” (AVM) provide a continuous velocity measurement along the channel axis by using two acoustical beams pointed upstream and downstream, respectively (Ward et al., 2007). Although this method provides both accurate and continuous velocity measurements, it requires a minimum flow depth of two inches over the instrument called “blanking distance” to operate properly. In addition to this limitation, undesirable acoustic energy, called “side lobes”, occur naturally and propagate from the transducer and angle away from the main beam which can lead to inaccuracies in shallow streamflow measurements.

The only method currently in practice that is capable of continuously measuring streamflow down to dry channel conditions without clogging is the “Stage-Discharge Method”. That also explains why it is the most commonly used method despite its limited applicability. The AVM provides the most accurate, continuous streamflow measurements, however it has certain physical measurement limitations. This paper studies the permanent mounting of an AVM in a submerged concrete “U-Channel” to eliminate the limitations of its current application. The success of this approach is based on producing accurate, continuous streamflow measurements down to the extremely shallow depths in order to meet the annual baseflow constrictions of the TMDL requirements in natural intermittent streams.

U-Channel

The AVM was mounted it into a thirty-two by eight feet concrete “U-Channel” box, which is two feet deep and submerged eighteen inches below the streambed. A six feet wide, six inch deep notch was cut into the top of the upstream and downstream ends of the “U-Channel” and the AVM was mounted on a platform elevated six inches above the “U-Channel” bottom. The longitudinal edges outside of the “U-Channel” were filled-in to prevent by-pass flow. This configuration was not only designed to solve the AVM’s primary “blanking distance” limitation by providing a constant one-foot depth during dry streamflow conditions, but it also reduces, if not eliminates, many other measurement issues.

The “U-Channel” provides a stable, uniform cross-section and a constant defined channel roughness. With routine maintenance, elevating the AVM six inches from the structure bottom prevents data loss from sediment covering the instrument. The width of the structure at low flows significantly reduces the effects of “side lobe” interference. The notches on the upstream and downstream ends of the “U-Channel” direct lower flows through the center of the structure where the AVM is mounted. As can be seen in Figure 5, the straight and symmetrical cross-sectional shape of the “U-Channel” yields a maximum velocity which is located at the middle of the structure (Chen and Chiu, 2004). Finally, even when the stream is flowing at very shallow depths, the flow depth in the “U-Channel” is relatively large so the one-sixth power law assumptions for turbulent flow are still valid (Hubert and Chanson, 2004).

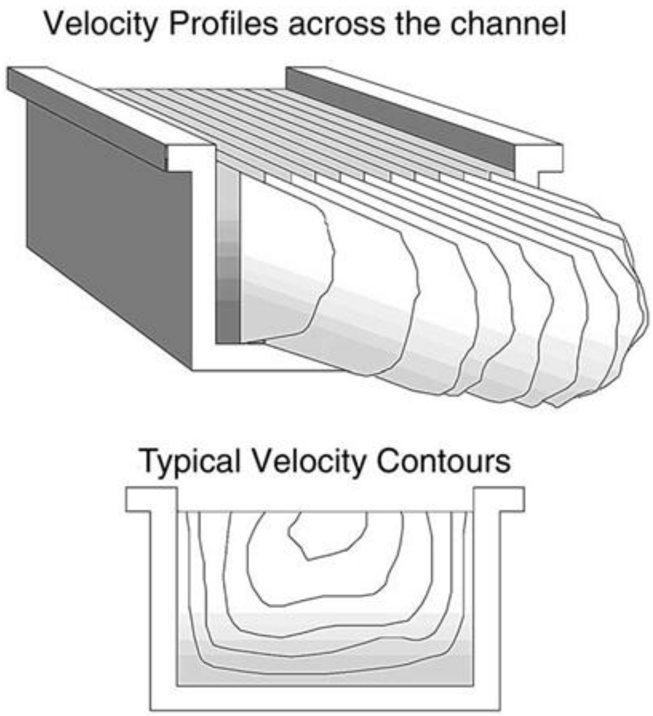

Figure 5: Velocity Profiles (Jones, 2002)

Study Sites

Seven of the nineteen sites within a twenty square-mile area of the City of Kissimmee, Florida were chosen to study the installation of an AVM within the concrete “U-Channel”. These sites were constructed on four tributaries of Lake Tohopekaliga. The South Florida Water Management District controls the water level of Lake Tohopekaliga and it is varied seasonally to provide flood control and supply water. This Central Florida region is a low-lying terrain subject to intense tropical rainfalls and downstream tailwater effects. These watershed and atmospheric characteristics create a wide range of streamflow conditions throughout the study area ranging from dry (no flow) to deep tailwater submerged flow.

Three of these seven “U-Channel” study sites (sites 06, 07, and 10) were constructed within midstream of a long reach of a man-made channel. The downstream reaches of these

channels are at a relatively high grade and are significantly upstream of any tailwater effects. Site 04 and site 08 were located immediately downstream and upstream of a broad crested weir, respectively. These weirs were constructed for stormwater management purposes to regulate the discharge from an upstream waterbody. The flows from site 08 were governed by the free discharge of the downstream weir, whereas site 04 has the potential for occasional tailwater impacts. The remaining two study sites (sites 05 and 09) were installed upstream of a discharge-controlled waterbody. The flows through these "U-Channels" are influenced by their downstream water levels. The locations of these sites within the City of Kissimmee are presented in Figure 6.

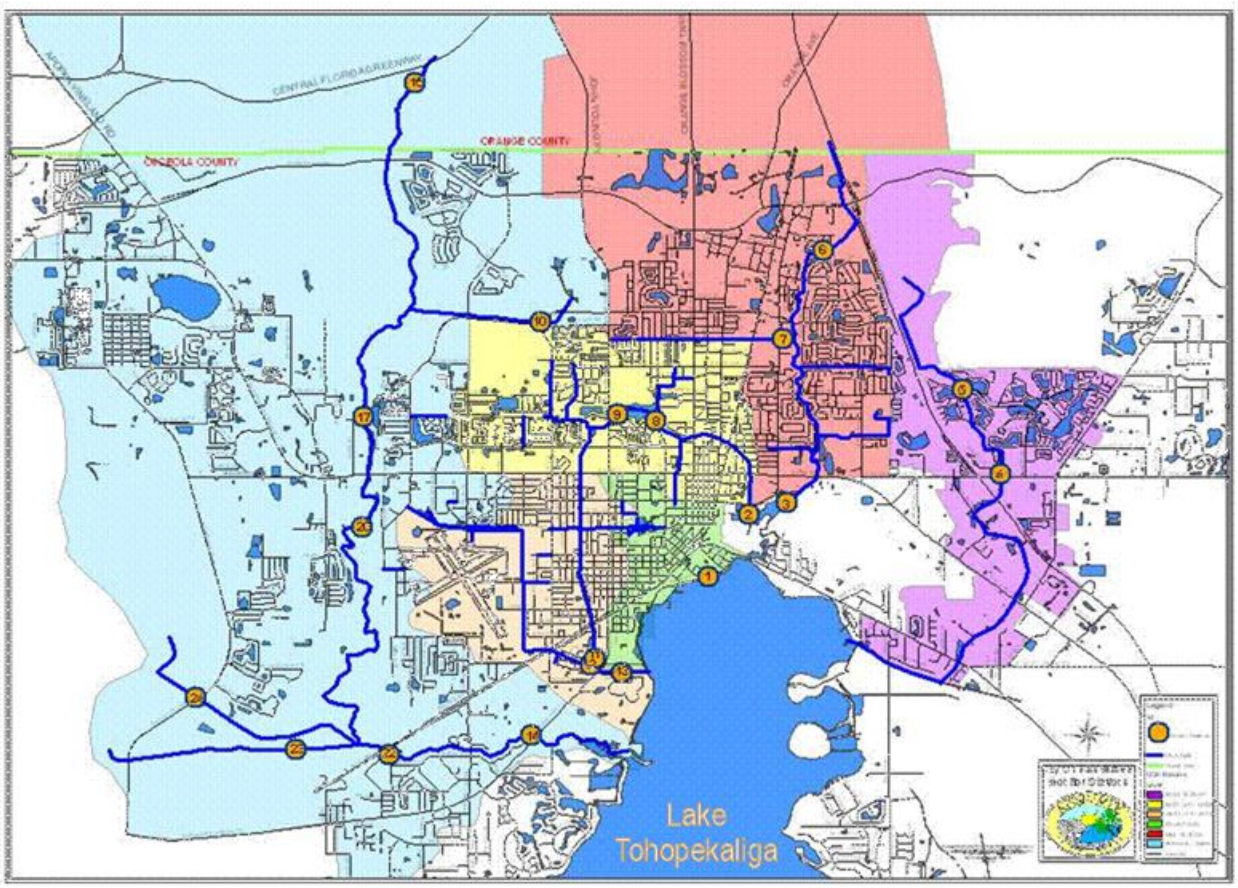

Figure 6: Study Sites

Each "U-Channel" site was equipped with a pressure transducer for stage measurement and an Argonaut-SW (Shallow Water) produced by YSI, Inc. for acoustic velocity measurement. The pressure transducer was set to measure stage at five minute increments. The Argonaut-SW was set to measure continuously and report a moving five minute average. This data was collected by a YSI Econet unit which transmitted it to an remote electronic data storage site. Construction of the sites was completed during the fall of 2005. This study is based on the five minute data from the nine sites and discrete streamflow measurements during the period from January 2006 through December 2008.

Methodology

The maximum stream velocity (u_{max}) measured in the "U-Channel" by the AVM is related to the mean stream velocity ($\bar{u}$) by the following relationship (Chiu, 1996).

$$\frac{\bar{u}}{u_{max}} = \frac{e^M}{e^M - 1} - \frac{1}{M} = \Phi \qquad \text{Eqn. 11}$$

Where *M* is the cross-section and Φ is a constant. This relationship is only based on natural stream factors such as energy slope and stream roughness. Therefore the value of Φ is valid throughout all ranges of discharge and stage as long as the channel properties remain stable (Chen and Chiu, 2002). In addition, research indicates that if the AVM's beam was slightly off of the actual location of the maximum velocity, it will not have a significant effect on the constant, Φ (Chiu et al., 2005). Thus, positioning of the AVM at the center axis of the "U-Channel" will yield accurate measurements of the maximum velocity throughout a wide range of flows.

Since the AVM measures the maximum stream velocity and the "velocity-area principle" (Eqn. 10) uses the mean stream velocity to determine streamflow, the value of Φ must be determined. This was accomplished by "Velocity Indexing", which relates the actual streamflow determined by discrete streamflow measurements to the streamflow measured in the "U-Channel" by the AVM.

Stage was measured within the "U-Channel" by the transducer and the corresponding velocity was measured by the AVM. These stage measurements were used to determine the cross-sectional flow area from the known stage-area relationship of the "U-Channel". Discrete streamflow measurements were also conducted at a location immediately upstream and downstream of the "U-Channel" coinciding with the time of the stage and AVM measurements. Using Equation 10 and the cross-sectional area, the discrete velocity was determined for the "U-Channel". A scatter plot of AVM (maximum) and discrete (mean) stream velocity measurements was prepared and a best fit line for the data yielded the constant, Φ.

Scatter plots of the discrete streamflow measurements were also made simply with respect to stage data. The best fit line for these plots yield equations for the rating curves according to Equation 9. These equations provide the widely accepted "Stage-Discharge" relationships for the "U-Channels". Even though steady, uniform conditions are rarely present in dry and very shallow streams, these relationships represent a benchmark of the most commonly used approach currently in practice.

Results

The integration of the AVMs within the "U-Channels" was accomplished in the fall of 2005. All but one of the sites were able to collect continuous stage and velocity data every five

minutes throughout the three year study period. Site 06 was subjected to vandalism and as a result was unable to function when discrete measurements were scheduled. Scatter plots of stage versus discrete measured streamflow were prepared for the remaining six study sites. Due to budget constraints, the "Stage-Discharge" calculations were based on a minimum of seven discrete streamflow measurements. In practice these measurements would continue throughout the life of the project to refine the relationship. However, sufficient combinations of stage and streamflow were encountered during the seven samples to yield sufficient resolution.

The data pairs of discrete streamflow as a function of measured stage were fit to the power curve of Equation 9 using the following relationships for variables *a* and *b.*

$$a = exp\left[\frac{\sum \ln Q_d}{n} - b\frac{\sum \ln S}{n}\right] \qquad \text{Eqn. 12}$$

$$b = \frac{\sum(\ln Q_d)(\ln S) - (\sum \ln Q_d)(\sum \ln S)/n}{\sum(\ln S)^2 - (\sum \ln S)^2/n} \qquad \text{Eqn. 13}$$

Where Q_d is the discrete streamflow, *S* is the stage and *n* is the number of data pairs. The scatter plot and corresponding best fit curve for Site 05 is presented in Figure 7. By inspection of this rating curve it is obvious that the discrete streamflow measurements do not fall along the best fit curve.

A "Goodness of Fit" procedure was used to compare the accuracy of indirect streamflow estimation determined from using only the stream stage data to the actual streamflow measured discretely. The goodness of fit is measured by the coefficient of determination, R^2 (Eqn. 14), which is an indicator of how well the rating curve fits the actual values. If the stage, *S*, produces a streamflow from the rating curve that is perfectly correlated to the discrete measured mean streamflow, Q_d, then R^2 is equal to one. In contrast, an R^2 value of zero means that there is no

correlation. An R^2 value of less than 0.9 corresponds to a rather poor fit of data to the rating curve.

$$R^2 = \frac{[\sum(\ln Q_d)(\ln S) - (\sum \ln Q_d)(\sum \ln S)/n]^2}{[\sum(\ln S)^2 - (\sum \ln S)^2/n]\,[\sum(\ln Q_d)^2 - (\sum \ln Q_d)^2/n]} \qquad \text{Eqn. 14}$$

The coefficient of determination, R^2, for Site 05 (Figure 7) is almost zero (0.032) which indicates a very poor correlation of the rating curve with the actual streamflow. Site 05 is a very shallow stream which is located immediately upstream of control structure and exhibits significant tailwater effects. It is reasonable to expect the resulting non-uniform flow conditions to cause this poor correlation when only stage is considered.

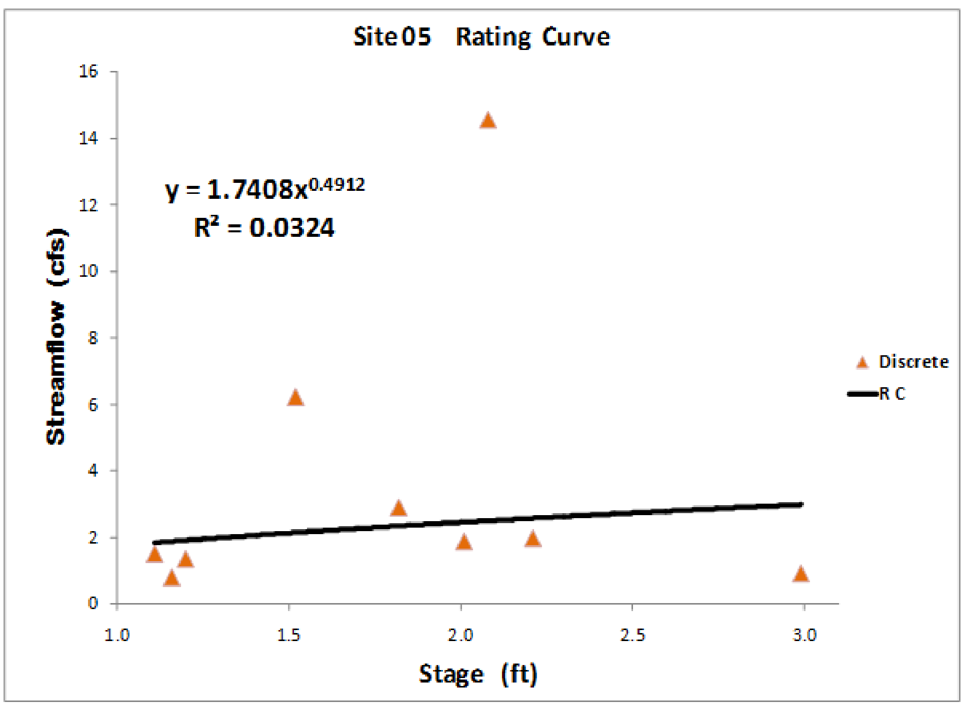

Figure 7: Rating Curve for Site 05

Similar rating curves were generated for the remaining five study sites. The flows in Site 07 and Site 10 were not influenced by significant downstream tailwater effects and exhibited relatively steady, uniform conditions. The R^2 values for the rating curves of site 07 and site 10 were 0.873 and 0.914, respectively. These relatively good correlations demonstrate the applicability of using only stage data to predict steady, uniform streamflows.

Site 04 and site 08 were located immediately downstream and upstream of a broad crested weir, respectively. These weirs were constructed for stormwater management purposes to regulate the discharge of an upstream waterbody. The flows from site 08 were governed by the free discharge of the weir, whereas site 04 was occasionally influenced by significant downstream tailwater effects. The R^2 values for the rating curves of site 04 and site 08 were 0.254 and 0.768, respectively. The poor correlation for Site 04 is due to tailwater effects since it is positioned downstream of the weir. The correlation of Site 08 was expected to be much closer to 1.0 since the downstream weir creates a very stable stage-discharge relationship. The reason for the poorer correlation at Site 08 was due to floating debris blocking the weir which caused varying stage elevations for similar discharges.

The remaining study site (Sites 09) was installed upstream of a discharge-controlled waterbody. The flow through this “U-Channel” was significantly influenced by downstream tailwater effects and was rarely uniform. The R^2 value for the rating curve of site 09 was 0.478. Since this site exhibits similar flow conditions to Site 05 (tailwater influences), this poor correlation should be expected. The stream stage at Site 09 is not as shallow as Site 05 which dampened out the impact of the fluctuating tailwater depth on the R^2 value.

The purpose of this study was to find an approach to increase the correlation between data measured in the field and the actual streamflow. Mounting the AVM within the “U-

Channel" provided the ability to measure stage and velocity down to very shallow stream depths. To determine whether these measurements were more accurate than the rating curves it was necessary to first evaluate the best fit for the linear relationship of the constant, Φ (Eqn. 11).

The data pairs of discrete stream velocity as a function of AVM measured velocity were fit to the linear relationship of Equation 11 using the following relationship for the constant Φ.

$$\Phi = \frac{\sum(V_d)(V_A) - (\sum V_d)(\sum V_A)/n}{\sum(V_A)^2 - (\sum V_A)^2/n} \qquad \text{Eqn. 14}$$

Where V_d is the discrete stream velocity, V_A is the AVM velocity and n is the number of data pairs. The coefficient of determination, R^2, for correlation of the constant Φ was also determined based on the following equation.

$$R^2 = \frac{\sum(V_d)(V_A) - [(\sum V_d)(\sum V_A)/n]^2}{[\sum(V_A)^2 - (\sum V_A)^2/n]\,[(\sum V_d{}^2) - (\sum V_d)^2/n]} \qquad \text{Eqn. 15}$$

This correlation measures how well the constant operator, Φ, applied to the maximum stream velocity measured by the AVM, V_A, compares to the actual stream velocity determined by the discrete measurements, V_d. The scatter plot and corresponding best fit curve for the constant, Φ, at Site 05 is presented in Figure 8. By inspection it is obvious that the discrete stream velocity measurements do not completely align with the best fit line, but the R^2 value of 0.900 demonstrates a relatively good correlation for the constant.

The remaining five study sites were plotted in a similar manner to determine the constant relationship between mean and maximum stream velocity. The velocities measured by the AVM and the corresponding stages were applied to Equation 10 to determine the AVM measured streamflows. Figure 9 presents a comparison of the discrete measured streamflows to both the

streamflows determined for the rating curves using stage only and the streamflows based on including the velocities measured by the AVM.

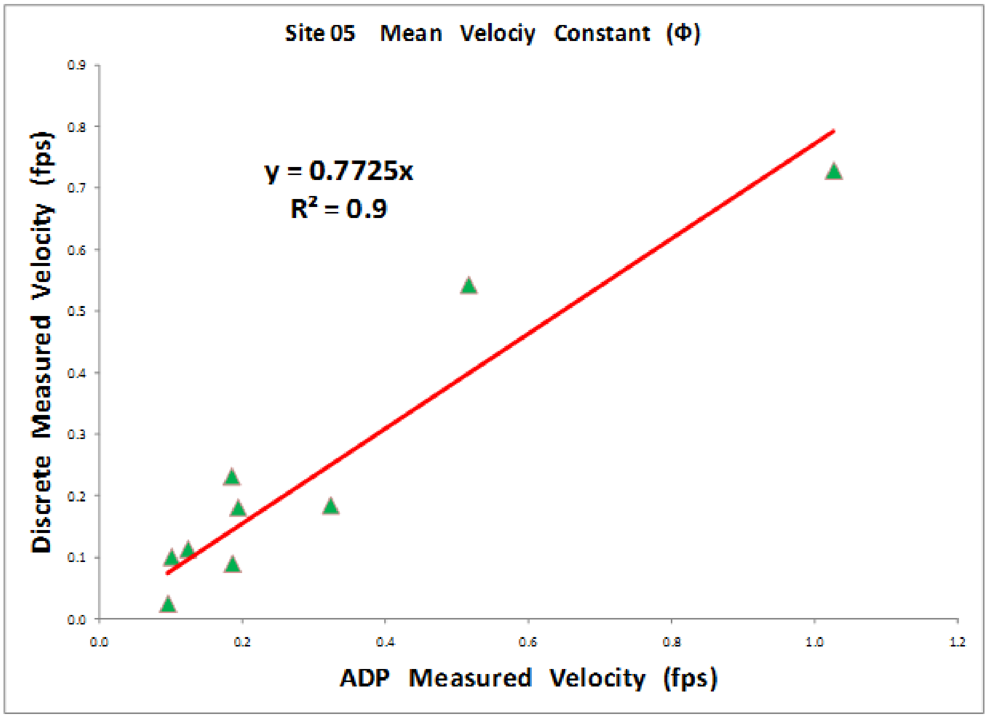

Figure 8: Mean Velocity Constant for Site 05

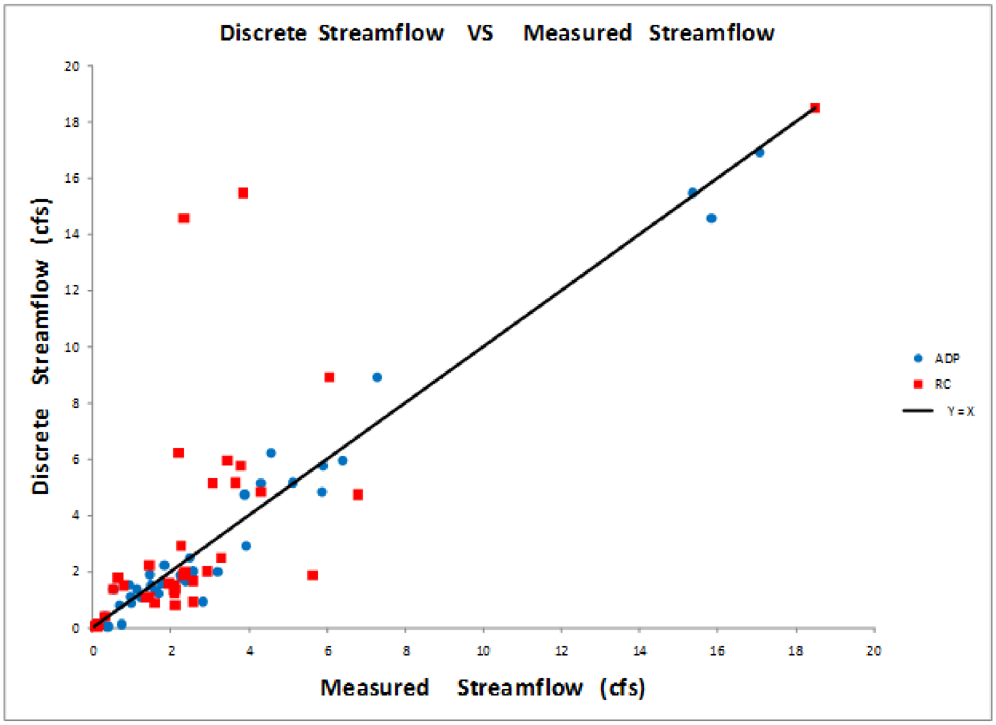

Figure 9: Comparison of Rating Curve and AVM Streamflows

The better correlation of the “U-Channels” (AVM) in comparison to the “Stage-Discharge” relationships (RC) are represented by how much closer they tend to align with the actual discrete streamflow measurements (X = Y line) in Figure 9. This is most pronounced in the region of lower streamflows where the shallow depths are more often impacted by tailwater effects which cause significant inaccuracies in the rating curves. A summary of the goodness of fit results for all of the study sites is presented in Table 1. The correlation for the rating curves ranged from a low of 0.032 at Site 05 to the highest value of 0.914 at Site 10. The corresponding correlation for the "U-Channels” ranged from a low of 0.865 at Site 07 to the highest value of 0.998 at Site 04. All of the rating curve correlations were improved by use of the “U-Channels” except for Site 07 which remained relatively unchanged. Since there are no significant tailwater effects at Site 07 it is reasonable to expect little difference in the two approaches. For the

remaining sites, where slight to significant deviations from uniform flow conditions were encountered, increases in correlation were obtained.

Table 1: Goodness of Fit for the U-Channel

SITE	RATING CURVE		U-CHANNEL		
	MEASUREMENTS	R^2	MEASUREMENTS	Φ	R^2
4	7	0.254	5	0.642	0.998
5	9	0.032	9	0.773	0.900
7	7	0.873	7	1.615	0.865
8	9	0.768	6	0.897	0.888
9	7	0.478	7	0.777	0.952
10	7	0.914	3	0.510	0.984

Conclusions

Accurate, continuous baseflow measurements in shallow, intermittent streams are required to meet the annual TMDL pollutant load requirements. The most commonly used "Stage-Discharge" method uses stage data to indirectly determine streamflow, but it is only valid for steady, uniform flows. Acoustical velocity meters are accurate and capable of measuring non-uniform flows, but they require a minimum stream depth to operate properly. This paper presents the mounting of an AVM in a submerged concrete "U-Channel" to overcome the limitations of shallow stream depths while maintaining measurement accuracy. The results of this 3-year study at six sites with varying flow conditions indicate a goodness of fit in the range of 0.87 to 0.99.

All but two of the study sites (Site 08 and Site 09) exhibited baseflow stream depths that were below the measuring limitations of the AVM. In other words, without the "U-Channel", continuous baseflow measurements using the AVM would only have been collected at these two sites. Three of the study sites (Site 04, Site 05 and Site 09) were significantly impacted by tailwater fluctuations which caused their correlations with the rating curves to fall below one-half. Therefore, it would not have been possible to collect accurate, continuous streamflow data during the 3-year study period at Site 04 or Site 05 if the "U-Channel" had not been constructed.

The results provide evidence that the "U-Channel" was able to collect accurate, continuous streamflow data down to even dry channel conditions. This approach was not affected by fluctuating tailwater conditions, floating debris or sedimentation. The uniform, rectangular cross-section was shown to minimize "side lobe" effects and direct the maximum velocity through the center axis of the structure where the AVM was mounted. The concrete "U-Channel" provided stable bathymetry and a constant defined channel roughness. Therefore, the "U-Channel" is an effective approach and can offer good performance in uninterrupted measurement in most intermittent natural streams.

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CHAPTER THREE: DUAL PARAMETER SAMPLING TRIGGER IN AN URBAN WATERSHED

Introduction

Florida is surrounded by water, and its many internal lakes and rivers have long been recognized for their excellent fishing and boating. This notoriety draws land developers to the lake shores to establish residential and commercial infrastructure. This land development brings with it flood plain alteration, water level stabilization, and increased nutrients, which cause adverse impacts to our lakes. The extensive paving of pervious surfaces in conjunction with hydraulic improvements to existing channels from this urbanization has resulted in higher combinations of pollutant concentrations and flows. In addition, these land alterations have also caused large fluctuations in stream baseflows and stages throughout the year. The pollutant waves generated from these developed areas need to be analyzed to adequately assess the health of our lakes.

Accurate water quality measurements during a rainfall event require the triggering of the automatic sampler to occur at the start of the pollutant wave. Extreme variability in the temporal rainfall patterns of Florida can affect the lag time and travel time of runoff hydrographs during storm events. These effects on the runoff hydrographs, combined with urbanization and complex watershed configurations, can delay the response of streamflow from rainfall inputs and create multiple imbedded pollutant waves. Current autosampling triggers require a threshold of rainfall depth, stream stage or flow rate to be established prior to the occurrence of the rainfall event. The resulting erratic behavior in flow levels and discharge rates of these pollutant waves often causes the autosampler to either falsely trigger or entirely miss the pollutant wave.

Florida rainfall patterns can also have extreme spatial variability, which can generate a significant pollutant wave at an upstream station while depositing little to no rainfall at a downstream monitoring station. Unless an effective, non-rainfall autosampler trigger is installed at the downstream monitoring station, the pollutant wave will not be measured. Autosamplers triggered by only rainfall rates will obviously not detect the pollutant wave in these situations. To solve this problem, some autosamplers are triggered by change in stage or flow rate, but large seasonal fluctuations in baseflow rates and tailwater elevations also make these triggers ineffective.

The objective of this paper was to demonstrate an innovative approach to trigger automatic water quality samplers at the onset of pollutant waves using dual parameters in the absence of rainfall and presence of both highly variable flows and stage levels. The data used in this study was collected throughout the City of Kissimmee, Florida within tributaries of Lake Tohopekaliga. The monitoring sites were located to identify inflows from outside the city limits and quantify external pollutant loads. Also, additional internal monitoring sites were established to determine the pollutant loads of intermediate sections of the city.

Background

The current database of pollutant loadings is insufficient to determine the impact of a particular land use, surface cover or management practice for many of our nation’s lakes. Accurate water quality measurements for various land uses are needed to determine the actual pollutant loadings from our local watersheds. Due to the effect of spatial and temporal rainfall distributions in Florida on pollutant wash-off rates, year-round samples must be collected from both baseflow and rainfall events.

For baseflow, a representative grab sample is collected for water quality analysis. For rainfall events, the concentrations for not only a flow-weighted composite sample are measured, but an additional sample for the first flush of pollutants is collected for analysis. The concept of first flush was initially established almost a century ago when horses were the primary mode of urban transport (Metcalf and Eddy, 1916). Referred to as "first foul flush", it described the initial rainfall volume that transported horse fecal solids from roads into the receiving streams.

The first flush phenomenon is typically associated with smaller watersheds with relatively high impervious surfaces. During the higher intensities of initial rainfall, pollutants deposited prior to the storm are washed off in high concentrations. Although first flush can occur in larger watersheds with less impervious surfaces, the longer travel times allow runoff from adjacent areas to comingle and can reduce the effect (Kim et al., 2005). There is also a seasonal first flush effect, which refers to storms that occur at the end of a dry season and wash off a disproportionally larger mass of pollutants that have collected due to the lack of rainfall (Kim et al., 2004).

The time interval for the first flush can vary with each watershed, but a period of 30 minutes is typically used for the initial volume of concentrated pollutant runoff. In a study performed by Kim et al. (2007), it was found that the accumulated pollutants were essentially washed off from the paved surfaces within the initial 15 to 20 minutes of rainfall. In the same study, the final concentration of all pollutants dropped as much as 50% from their initial first flush levels due to wash off effects.

The first flush affect is a mass limited pollutant transport process where concentrations are controlled by the amount of pollutants available for transport according to the following equation (Sheng et al., 2008).

$$\Delta M_T = M_0(1 - e^{-K_1 V_T}) \qquad \text{Eqn. 16}$$

Where: ΔM_T is the mass transported, M_0 is the initial mass available for transport, K_1 is a transport rate constant and V_T is the transport volume.

When the rainfall amount is sufficient to where the wash-off rate is no longer a critical factor in limiting pollutant transport, it becomes a flow limited transport process. Under flow-limited transport, the wash-off rate is unrelated to the amount of mass remaining on the surface and follows Equation 17 (Sheng et al., 2008).

$$\Delta M_T = K_0 V_T \qquad \text{Eqn. 17}$$

Where: K_o is the transport rate constant.

Most watersheds start off as mass limited, but as the rainfall amounts significantly increase, they typically transition into flow-limited transport. For larger and more complex watersheds the first flush effect is short lived to the point of even non-existent (Sansalone and Cristina, 2004). When a cumulative mass versus cumulative volume curves is constructed, mass limited pollutant transport tends to follow a first-order exponential pattern whereas flow limited pollutant transport exhibited a linear pattern (Sheng et al., 2008).

Thirteen separate urban watersheds with distinct types of residential and industrial development were selected for a stormwater runoff monitoring study on first flush effects (Lee et al., 2002). A distinct mass limited transport pattern of first flush was found for pollutants of all 38 storms monitored; however, no correlation was found between the first flush phenomenon and antecedent dry weather period. These studies show that the first flush can be a significant source of pollutants and it is necessary to measure it from the start of the pollutant wave.

Recently, pollutant loads have been determined by using automatic sampling equipment linked with stage measuring recorders to obtain continuous, rating curve based streamflow data. These automatic samplers use either local rainfall depths or stream stage levels as triggers to obtain runoff pollutant concentrations from rainfall events. More advanced pollutant monitoring stations measure in-situ stream velocity using Acoustical Velocity Meters (AVM) mounted on the stream banks or Acoustical Doppler Profilers (ADP) mounted within the streamflow. Permanently mounted ADP provide a continuous velocity measurement along the channel axis by using two acoustical beams pointed upstream and downstream, respectively (Ward et al., 2007). These stations use a stage-area rating curve with continuous stream velocities to provide more accurate streamflow measurements under non-uniform flow conditions (Gurr and Nnadi, 2011).

There is a gap between the availability of accurate first flush pollutant load data for local land uses and the physically based models used to determine the Total Maximum Daily Loads (TMDLs) for receiving waterbodies. Within this gap is a need to improve the accuracy of annual streamflow measurements and to develop a rainfall event trigger to start the automatic samplers at the beginning of the pollutant wave.

Factors that Affect the Pollutant Wave

The pollutant wave can be affected by numerous factors including atmospheric conditions, watershed characteristics and anthropogenic impacts. Regional atmospheric conditions such as the Humid Subtropical climate produce significant amounts of precipitation year-round within Florida. Central Florida has the highest annual number of days with thunderstorms in the United States (Williams et al., 1992). Large storm fronts traveling from

west to east deposit rain from November through April and the migration of the inter-tropical convergence zone creates convective rainfall from May through October (Baigorria et al., 2007). Hurricanes also exert an infrequent but significant influence on seasonal and annual rainfall totals. Another significant influence opposite to annual rainfall totals are periods of dry days that occur during late fall through spring, which increase the pollutant wave buildup within the watershed.

Localized atmospheric conditions can create large variations in spatial and temporal rainfall distributions. Florida regularly experiences seabreeze (SB) fronts and rainfall-induced outflow boundaries (OB) due to its unique geographical and meteorological conditions. The interactions between SB and OB, coupled with their associated convergence, provide lift to initiate convection (Shepherd et al., 2001). Lines of single-cell and multi-celled thunderstorms can form squall lines, containing newer cells on the leading edge of the front and weakening cells on the trailing edge (Yuter et al., 1994). Heavy rain and frequent lightning within these fronts typically lasts anywhere from 1 to 2 hours and can contribute up to 40% of Florida's total annual rainfall (Shepherd et al., 2001). These variable rainfall durations and intensities can make it difficult to detect the actual pollutant wave amongst fluctuating flow rates.

The rainfall in Florida is not only temporally distributed, but also has a strong spatial distribution. Heymsfield et al. (2000) found that more than three-quarters of the storm cells from convective precipitation in Florida have diameters which are less than 3 mi. and Goldhirsh and Musiani (1986) suggest that the median convective cell diameter is just slightly over 1 mi. The intensity of rainfall within these convective cells also varies exponentially in space throughout the width of their fronts due to varying vertical velocities in the form of strong updrafts and

downdrafts. These distributed rainfall deposits can lead to multiple smaller waves occurring prior to the actual pollutant wave.

The pollutant wave can also be affected by various watershed characteristics such as the response of runoff from excessive rainfall called “lag time”. The area, shape, slope and surface cover of the watershed affects the time it takes for the excess rainfall to reach a channelized flow. A related watershed affect called “travel time” depends upon the conveyance between the channelized flow and the sample measuring point. The watersheds lag time and travel time are also a function of the intensity, duration and spatial coverage of the rainfall. As the basin area, land use and physical terrain vary between watersheds, the shape of the runoff hydrograph can be significantly altered. These irregularly shaped runoff hydrographs can make it difficult to identify the actual pollutant wave.

The complexity of a watershed has an effect on the pollutant wave shape. The runoff hydrograph from a simple watershed will respond significantly different than a watershed with multiple subbasins. Lateral inflows into the stream from multiple subbasins can cause a deviation from a simple rainfall driven runoff hydrograph of a single basin. The convolution of runoff from the individual subbasins can lead to multiple peaks in the streamflow hydrograph even when impacted by relatively stable rainfall intensities. Given the spatial and temporal variations of rainfall, the streamflow hydrograph in complex watersheds can be very irregular in comparison to a simple watershed hydrograph. Multiple peaking runoff hydrographs can affect the detection of the pollutant wave.

Study Sites

The study area encompasses the corporate limits of the City of Kissimmee located in Osceola County, Florida, which has a population of approximately 55,000 residents. Adjacent portions of Osceola and Orange counties were also included in this study to define the points where stormwater flows in to and out of the City of Kissimmee. This area was chosen because of its significant contribution of water flow into Lake Tohopekaliga. Lake Tohopekaliga is located in the upstream portion of the Upper Kissimmee Watershed.

The city covers approximately 20 square miles of surface area with relatively flat topography and poorly drained soils. A mixed land use of residential, commercial and agricultural can be found throughout the City of Kissimmee. Stormwater runoff in the city is conveyed to Lake Tohopekaliga by six distinct tributaries, which receive flow from the runoff of their respective watersheds.

Shingle Creek is the largest of these tributaries, which has its headwaters in Orange County and discharges along the western side of the City of Kissimmee into Lake Tohopekaliga. Shingle Creek is mostly rural and the lower portions, which flow through the City of Kissimmee, are undeveloped wetland floodplains. The second largest tributary flowing through the City into the lake is Mills Slough, which is located towards the east side of the city and has its headwaters in southern Orange County. Bass Slough is located at the eastern side of the City of Kissimmee and has its headwaters in northern Osceola County. Both Mills Slough and Bass Slough are mostly residential land uses. East City Ditch, West City Ditch and Downtown Area are the other three tributaries and have their headwaters completely inside the city limits. East City Ditch is a mixture of residential and light commercial land use. West City Ditch is a mixture of residential and light industrial land use. The Downtown Area has mostly a light commercial land use. To

investigate the effect of dual parameter trigger on pollution wave, 19 monitoring stations were constructed throughout the watershed along the six tributaries in the fall of 2005. The watersheds and monitoring station locations are depicted in Figure 10.

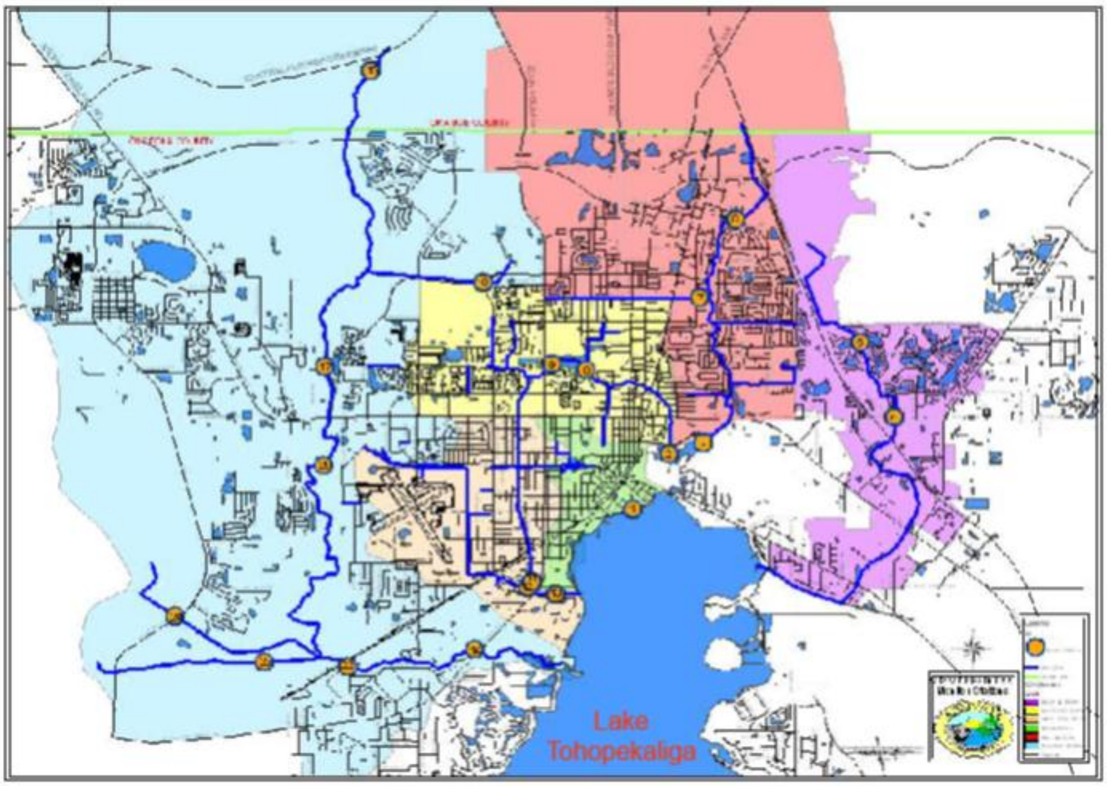

Figure 10: Study Sites

Each site was equipped with a pressure transducer for stage measurement, while either Argonaut-SW (Shallow Water) or Argonaut-SL (Side Looker) by YSI, Inc. were used for acoustic velocity measurement. The pressure transducers were set to measure stage at five-minute increments. The Argonauts (ADP) were set to measure continuously within the water column and report a moving five minute average. This data were collected using YSI Econet units, which transmitted it to a remote electronic data storage site. The analyses in this study

were based on the five- minute data from one of the 19 stations and discrete streamflow measurements during the period from January 2006 through December 2008.

Methodology

A new method for triggering automatic samplers in the absence of rainfall and presence of both highly variable flows and stage levels is needed. Two pollutant waves were selected for monitoring station number 14 to demonstrate this new method. Station 14 is located at the downstream end of Shingle Creek at the confluence to Lake Tohopekaliga. The first pollutant wave (Figure 11) occurred on June 16th, 2006 in the absence of rainfall and exhibits a small localized disturbance prior to the actual pollutant wave. However, a second pollutant wave occurred one month later on July 16th, 2006 following a local rainfall event (Figure 12).

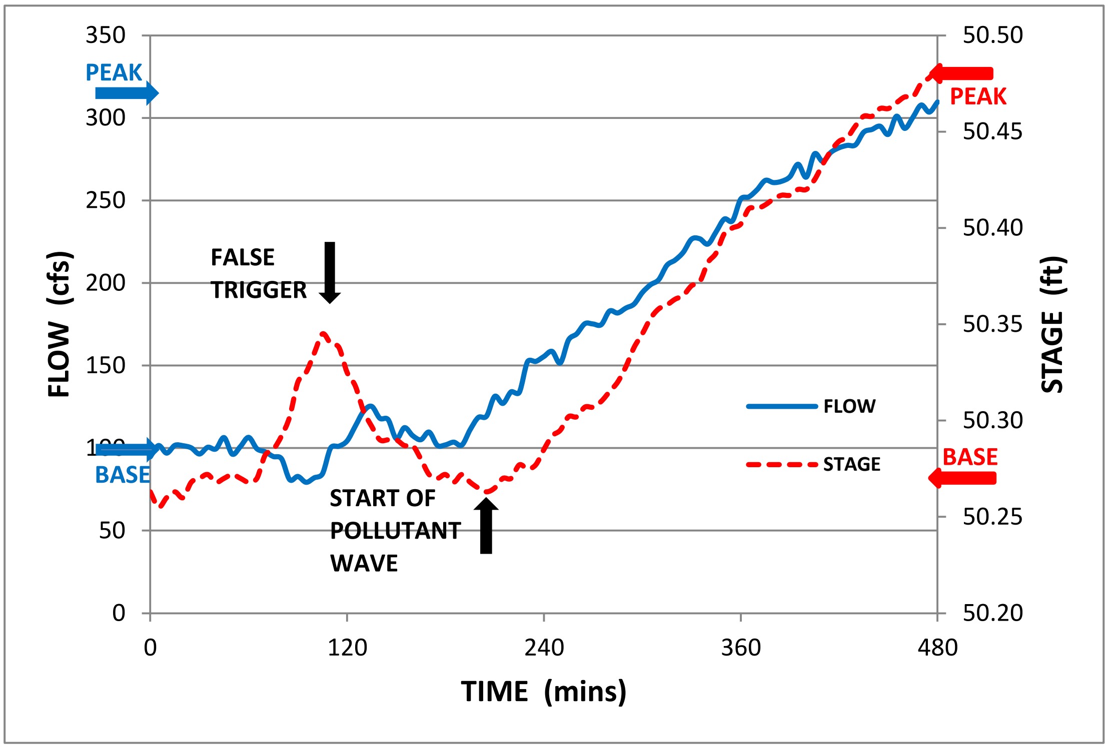

Figure 11: Flow and Stage Threshold (June 16th, 2006)

The automatic sampler could not be triggered on June 16th, 2006 (Figure 11) using only a rainfall gage trigger. The pollutant wave was generated as a result of significant rainfall in upstream regions of Shingle Creek when there was no local rainfall. Capturing the entire pollutant wave from this event in the downstream reaches of Shingle Creek was necessary to determine any changes in pollutant loads during transport and also to know the actual pollutants reaching Lake Tohopekaliga. Hence the need for an alternate method to capture pollutant waves without rainfall triggers.

Flow threshold triggers are often used to detect pollutant waves in the absence of rainfall when the flow rate reaches a predetermined level. The flow levels for the pollutant wave on June16th, 2006 ranged from a base flow of approximately 95 cfs to a peak of 306 cfs. Compare this to the pollutant wave on July 16th, 2006 that ranged from a base flow of approximately 240 cfs to a peak of 557 cfs. If a flow threshold trigger was set to trigger at 280 cfs to capture the pollutant wave on July 16th, 2006, it would have missed the initial pollutant wave only a month earlier on June 16th, 2006. If a flow threshold trigger was set to trigger at 100 cfs to capture the pollutant wave on June 16th, 2006, it would have falsely triggered during the baseflow leading up to the pollutant wave on July 16th, 2006. In addition, it would have most likely false triggered 90 minutes earlier during the small wave on June 16th, 2006 (Figure 11).

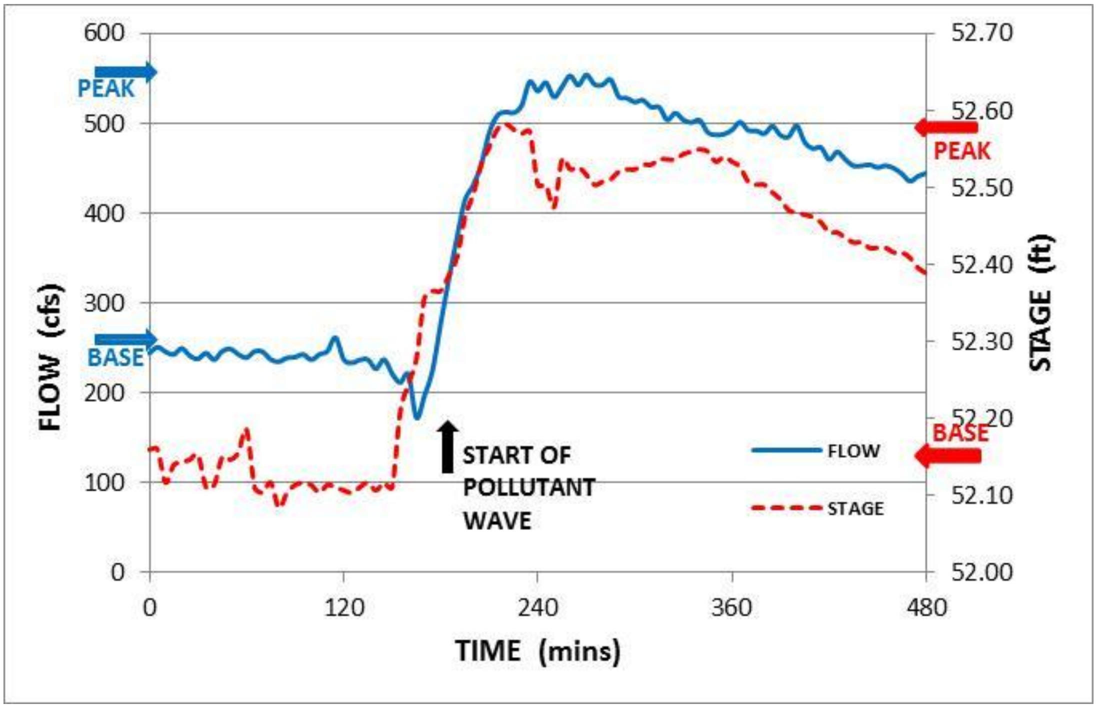

Figure 12: Flow and Stage Threshold (July 16th, 2006)

In a similar manner, stage threshold triggers are also used to detect pollutant waves in the absence of rainfall when the stage reaches a predetermined level. The stage levels for the pollutant wave on June 16th, 2006 ranged from a base level of 50.26 ft. to a peak level of 50.48 ft, while that on July 16th, 2006 ranged from a base stage level of 52.11 ft. to a peak level of 52.59 ft. If a stage threshold trigger was set to trigger at 52.30 ft. to capture the pollutant wave on July 16th, 2006, it would have missed the entire pollutant wave a month earlier on June 16th, 2006. A stage threshold trigger set to trigger at 50.30 ft. to capture the pollutant wave on June 16th, 2006 would have falsely triggered during the baseflow leading up to the pollutant wave on July 16th, 2006. Similar to the flow-based trigger, it would have most likely false triggered 90 minutes earlier during the small wave on June 16th, 2006.

To avoid this problem with the variability in flow rates and stage levels of the automatic sampler trigger, the automatic sampler can be triggered on instantaneous changes in flow rate or stage level. These types of triggers seem to work well in relatively stable streamflow conditions when the magnitude of the hydrographs can be predicted. However, in more complex watersheds, or where large variability in temporal and spatial rainfall distributions exists, these triggers often fail to operate properly. The instantaneous change in flow and stage for station number 14 during the June 16th, 2006 pollutant wave are shown in Figure 13 and Figure 14, respectively.

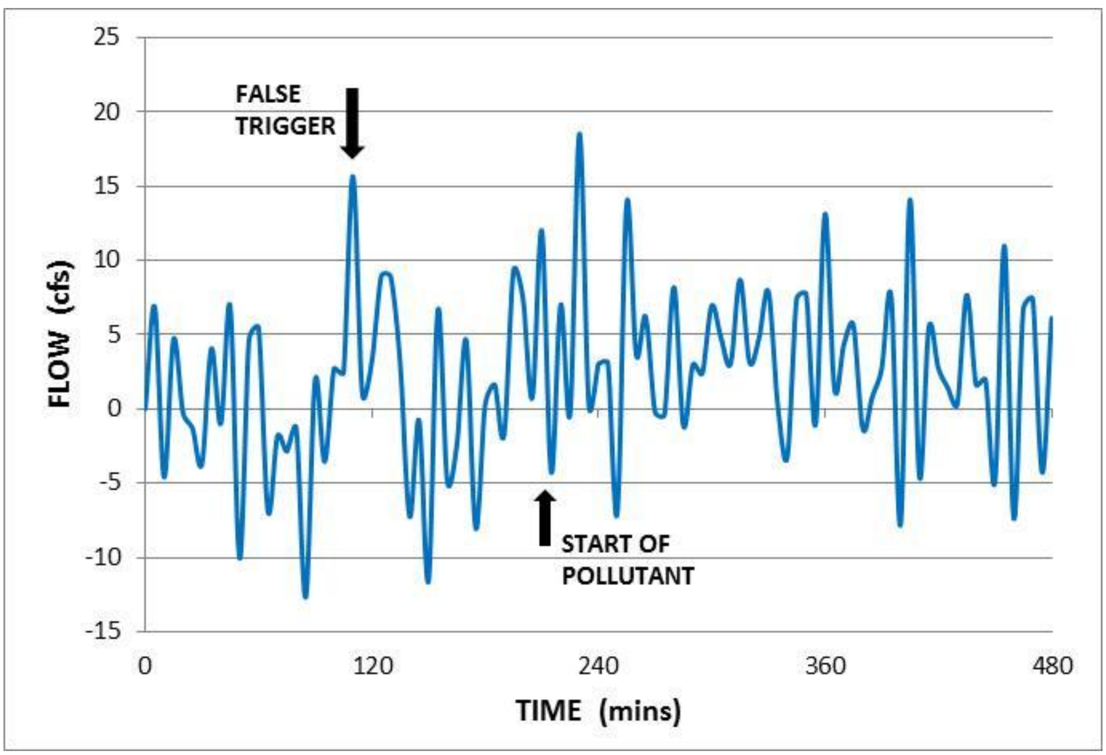

Figure 13: Instantaneous Change in Flow (June 16th, 2006)

Figure 13 shows that the instantaneous change in flow during the small wave (false trigger) was almost as pronounced as the start of the actual pollutant wave. In Figure 14, the

instantaneous change in stage during the small wave (false trigger) was actually larger than at the start of the pollutant wave. Also, during the study, many monitoring devices exhibited erratic behaviors, which recorded false magnitude spikes and caused the sampler to start with these instantaneous change triggers. A new method needed to be implemented to capture the start of a pollutant wave without false triggering on smaller waves or instrument spikes.

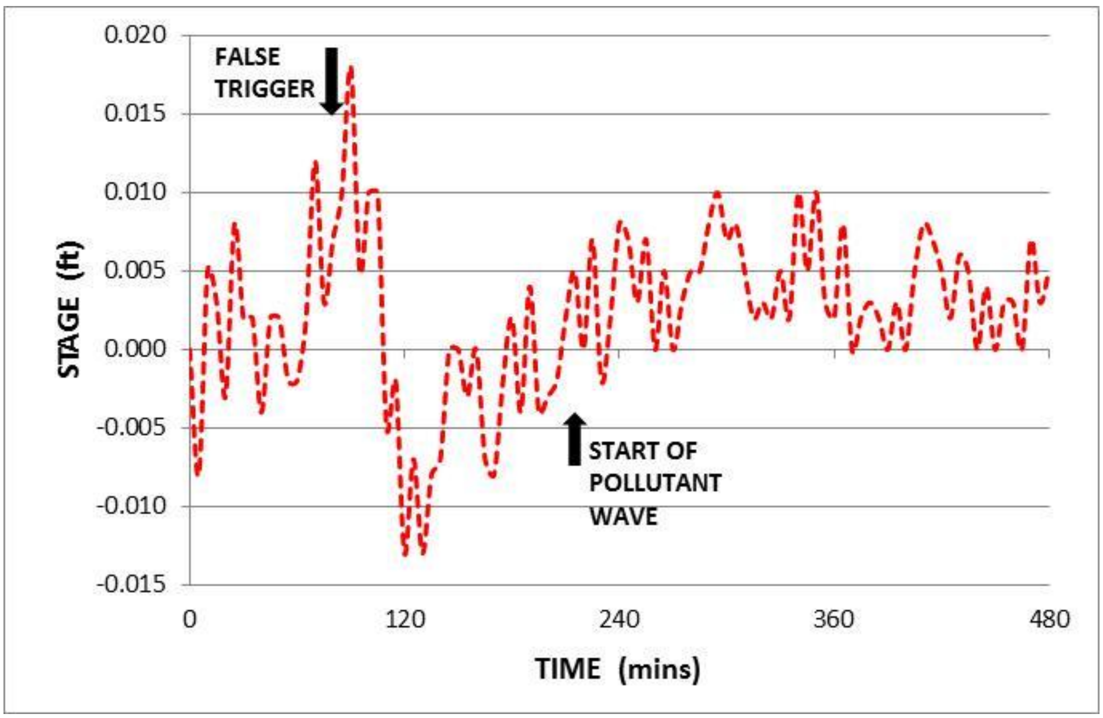

Figure 14: Instantaneous Change in Stage (June 16th, 2006)

As an alternative to these existing autosampler triggering methods, a dual parameter trigger was used to minimize false triggers while still capturing the start of the pollutant wave. In addition to this dual parameter method, the concept of a moving average was also incorporated to stabilize the data from more complex watersheds and to accommodate spatially and temporally variable rainfall distributions.

The dual parameter trigger required two separate conditions to be present to enable the automatic sampler trigger. It was important to select two parameters, which used separate equipment to measure the stream conditions. This was done to minimize the potential of false triggering from the presence of any individual instrument spikes. The measured data were sent to the YSI Econet unit, which checks the data of each device. If both of the two selected parameters met their threshold conditions then a signal was sent to trigger the autosampler. False triggers were avoided when only one of the selected parameters met its threshold condition.

Various parameters were evaluated at varying threshold values based on their operational independence by observing plots of the recorded data. These parameters included rainfall, flow, velocity, stage, temperature, conductivity, total dissolved solids, dissolved oxygen, turbidity and acidity. The greatest weight was given to the two parameters, which showed the largest variance during false triggers and yet yielded strong agreement during the start of the pollutant wave. Once the parameters were selected, their threshold values were established using a spreadsheet to identify the optimum operating levels. Simple logic conditions were used to simulate the function of the YSI Econet unit with the recoded data. A final plot of the selected threshold level was made against 10 separate pollutant waves to verify that the trigger was operating properly.

Historical recorded data was reviewed to determine the number of time steps necessary to depict a stable representation of the stream base flow. Typically, if the streamflow was more stable, it required a smaller window of measurements. Once the window size was determined, it was entered into the YSI Econet unit to establish a moving average of the previous recorded data within that number of time steps. This window and its corresponding average moved with the current instrument measurements. An incremental increase or percent increase over this moving average was used as a trigger parameter.

Results

An evaluation of the measurement parameters showed that rainfall often occurred at varying times prior to the actual pollutant waves or not at all, so it was not a reliable trigger parameter. The large fluctuations and spikes of velocity in the absence of a pollutant wave also proved this parameter to be unacceptable. Although changes in temperature, conductivity, total dissolved solids, dissolved oxygen, turbidity and acidity were all present during the pollutant waves, they typically occurred well after the start of the waves and were unable to be used as trigger parameters. The two parameters which were routinely present and occurred at the start of the pollutant waves were stage and flow. These two parameters also showed the largest variance during false triggers, while yielding strong agreement during the start of the pollutant waves. Since stage was measured by the pressure transducer and flow was measured by the ADP, they also met the requirement of using separate equipment to measure stream conditions. Further evaluation was necessary to determine whether to use an instantaneous change in value, an incremental change in average value, or percent change in average value for the two parameters.

The first attempt to develop the dual parameter trigger used the instantaneous change in flow and the instantaneous change in stage to trigger the autosampler. The plot of the dual parameter trigger for instantaneous change in flow and instantaneous change in stage is shown in Figure 15. The plot shows some variance of the false trigger during the small wave preceding the actual pollutant wave, but there was a significant overlap of the values during the recession limb of the hydrograph. An analysis of 10 pollutant waves occurring at this monitoring site was conducted to determine the optimum instantaneous threshold values of change in flow and stage of 11 cfs and 0.005 ft., respectively. These threshold values resulted in the minimum of false triggers while still capturing the start of the pollutant waves. However, it was observed that if

thresholds of instantaneous change in flow of 11 cfs and that of stage for 0.005 ft. were established to catch the start of the actual pollutant wave on June 16th, 2006, it would result in a false trigger during the small wave.

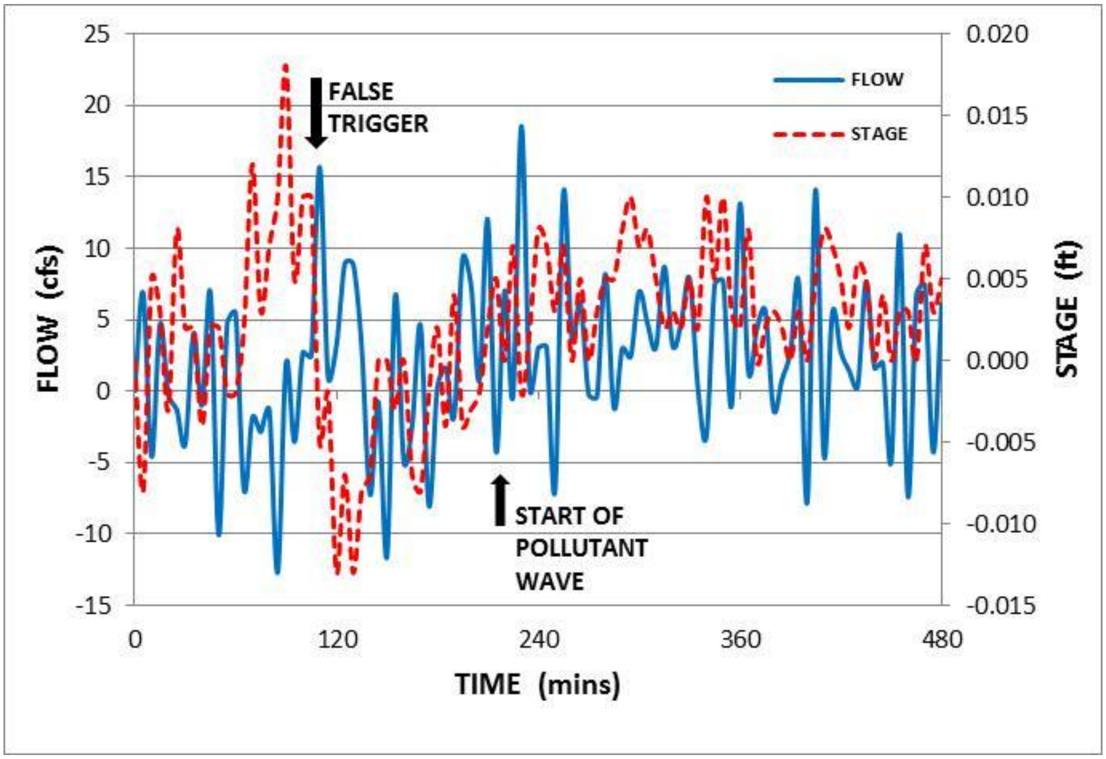

Figure 15: Instantaneous Change in Flow and Stage (June 16th, 2006)

The second attempt used the change from average flow and the change from average stage to trigger the autosampler. The plot of these dual parameters for June 16th, 2006 is shown in Figure 16.

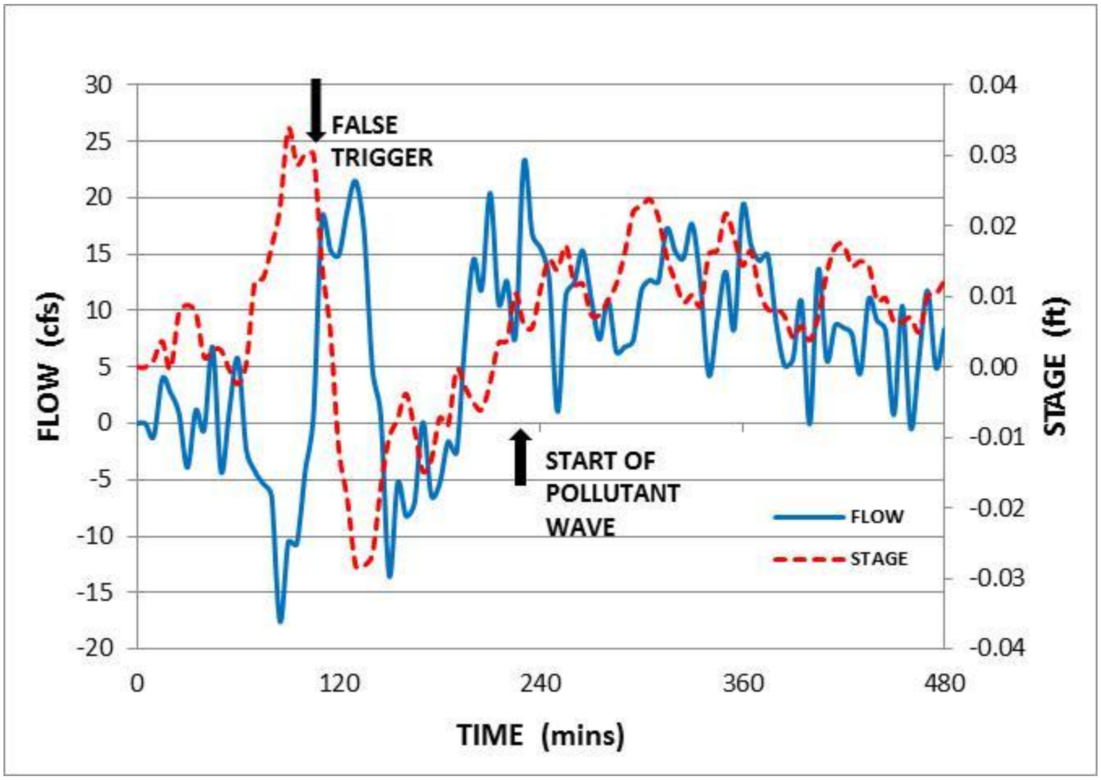

Figure 16: Change from Average Flow and Stage (June 16th, 2006)

This plot provides a variance of the false trigger during the small wave preceding the actual pollutant wave, but the overlap of the values during the recession limb was not as severe. The same 10 pollutant wave analysis for this monitoring site determine the optimum change from average threshold values for flow and stage to be 18 cfs and 0.001 ft., respectively. These threshold values also resulted in the minimum of false triggers while still capturing the start of the pollutant waves. These trigger thresholds were able to catch the start of the actual pollutant wave on June 16th, 2006 without causing a false trigger during the small wave. However, when these trigger parameters and corresponding threshold values were used on the other 9 pollutant waves they did not operate properly on all of the pollutant waves. The relatively small increase

in stage of 0.01 ft from the average stage occurred too often in the fluctuating stages of non-pollutant waves.

The third attempt used the percent change from average flow and the percent change from average stage to trigger the autosampler. Using the percent change from average value as the parameter causes the magnitudes of the plotted values to be attenuated. The plot of the dual parameters of percent change from average flow and average stage for June 16th, 2006 is shown in Figure 17.

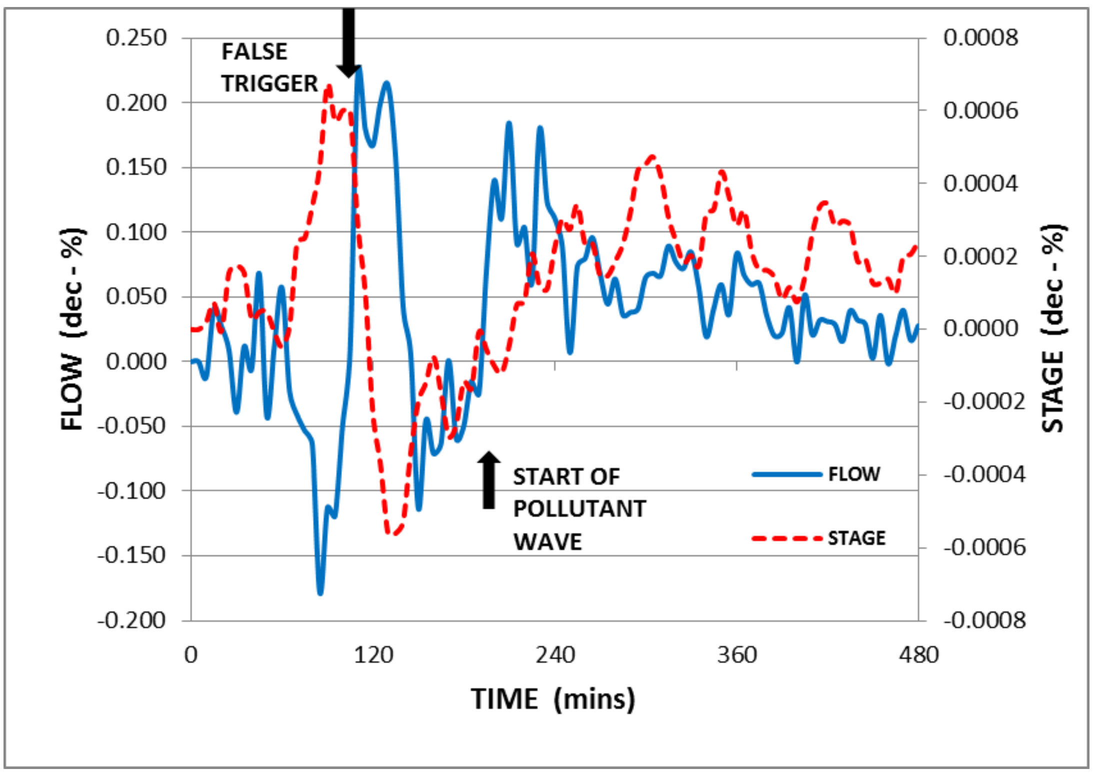

Figure 17: Percent Change from Average Flow and Stage (June 16th, 2006)

Good variance of the false trigger during the small wave preceding the actual pollutant wave is obtained and only a small overlap of the values during the recession limb was observed. The percent change from average flow and stage from the 10 pollutant wave analysis determine

the optimum threshold values to be 17% and 0.01%, respectively. Again, these threshold values represented the minimum false triggers while still capturing the start of the pollutant waves for this trigger parameter. These thresholds were able to catch the start of the actual pollutant wave without causing a false trigger during the small wave. However, a review of subsequent 9 pollutant waves revealed that small fluctuations in flow during low base flow conditions yielded large percent increases from the average flow. These fluctuations occurred too often in non-pollutant waves to use percent change from average flow as one of the trigger parameters.

A review of the first three trigger attempts revealed that under baseflow conditions most of the streams exhibited small incremental changes from their average flow, but their percent changes from their average flows were much larger. Conversely, when examining the average stream stage during baseflow conditions, the incremental changes were much larger than the percent changes. When pollutant waves occurred, these observations of incremental and percent magnitudes for flow and stage were found to be the opposite of baseflow conditions. This relationship held true for even relatively shallow streams, but the difference in magnitudes became less pronounced as the stream depth was reduced.

Noting these observations, it was decided to conduct a fourth trial using the combination of incremental change from average flow and the percent change from average stage to trigger the autosampler for June 16th, 2006 as shown in Figure 18.

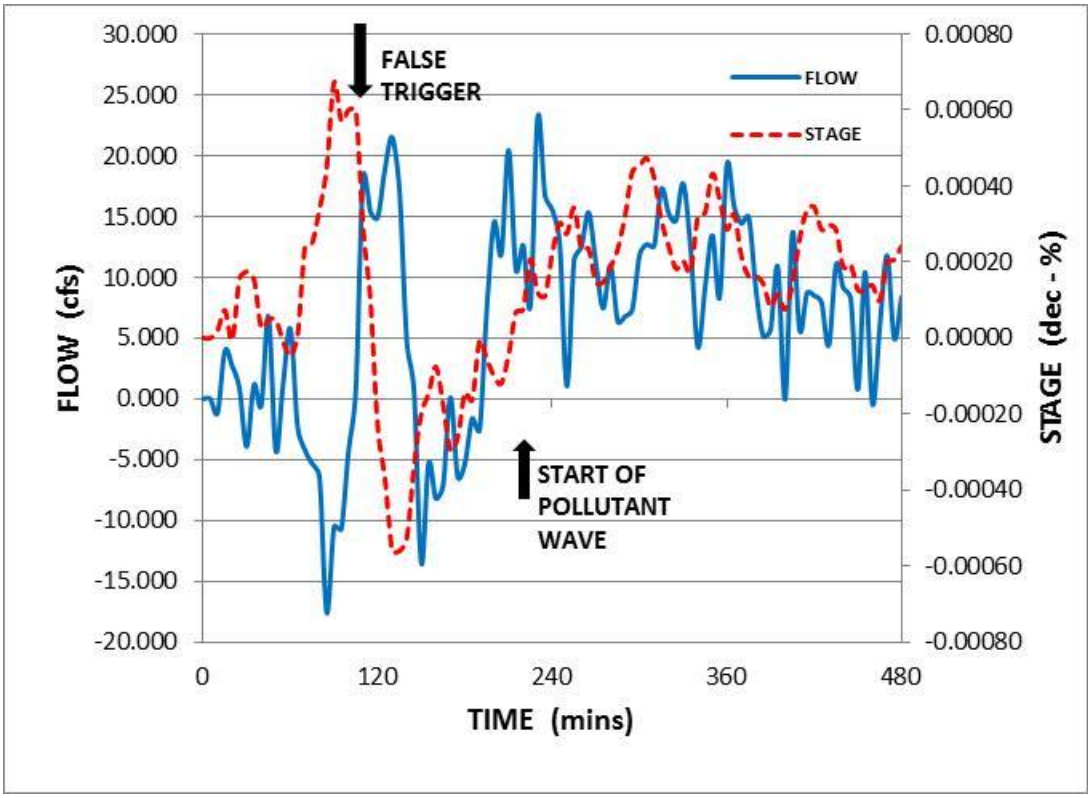

Figure 18: Change Average Flow and Percent Change Average Stage (June 16th, 2006)

As before, good variance of the false trigger during the small wave preceding the actual pollutant wave is obtained and only small overlap of the values during the recession limb was observed. Trigger thresholds of 18 cfs and 0.01% for change from average flow and percent change from average stage, respectively were able to catch the start of the actual pollutant wave without causing a false trigger during the small wave.

A review of non-pollutant waves revealed that the small fluctuations in streamflow during low baseflow conditions did not result in false triggers. The results of this approach were applied to 9 additional pollutant waves at station 14 to verify the results. The plots of the trigger parameter thresholds for flow and stage for all 10 pollutant waves are shown in Figure 19 and Figure 20, respectively.

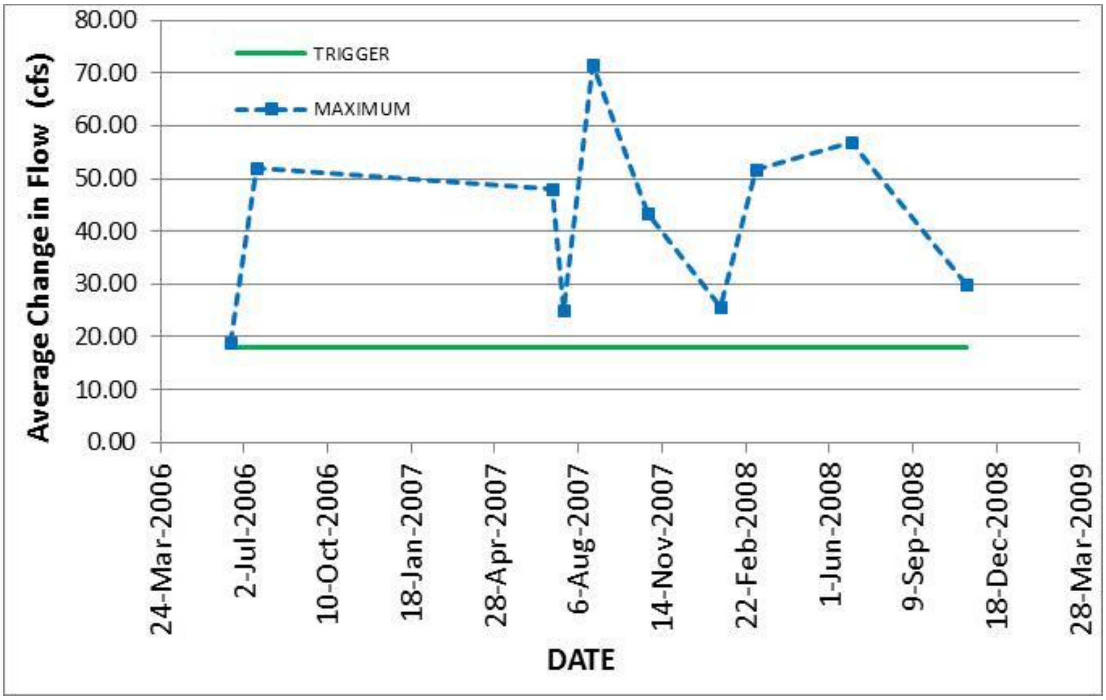

Figure 19: Change from Average Flow Threshold Value

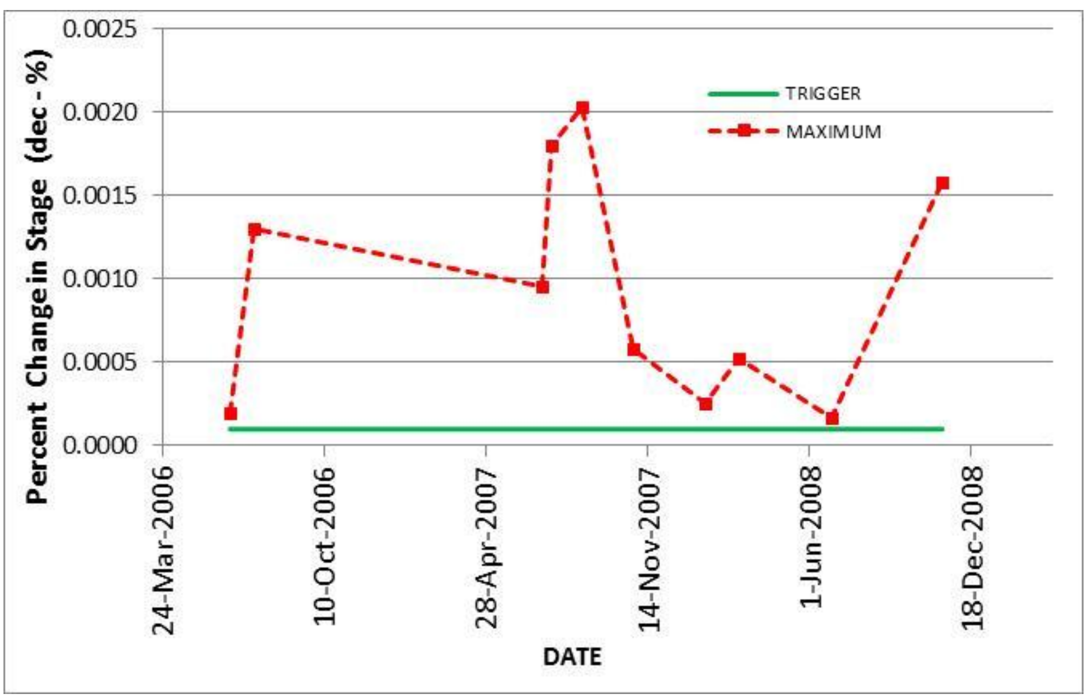

Figure 20: Percent Change in Stage Threshold Value

The selection of the June 16th, 2006 pollutant wave was made to reflect the most severe false trigger condition. Although for the purpose of this analysis it was referred to as a false trigger, it actually was a small pollutant wave. It was assumed that the trigger parameters and corresponding threshold values that successfully ignore this small wave and still catch the start of the larger pollutant wave would be adequate to accommodate most baseflow fluctuations. Figure 19 and Figure 20 show that the trigger thresholds for all 10 pollutant waves were above the June 16th, 2006 threshold level. Therefore, setting the threshold at this level yielded positive triggering of the autosampler where other parameters failed.

The July 16th, 2006 pollutant wave using these two parameters is shown in Figure 21. It can be seen that the dual parameter trigger actually caught the start of the pollutant wave. The abrupt rise in stage, which preceded the rise in flow, was a common occurrence in the data collected during this study. For smaller waves, the increase in stage declined prior to the rise in flow rate. In the case of significant pollutant waves, this increase in stage continued until it overlapped with an increasing flow. Using these dual parameters, the autosamplers were not triggered until the actual flow rate increased.

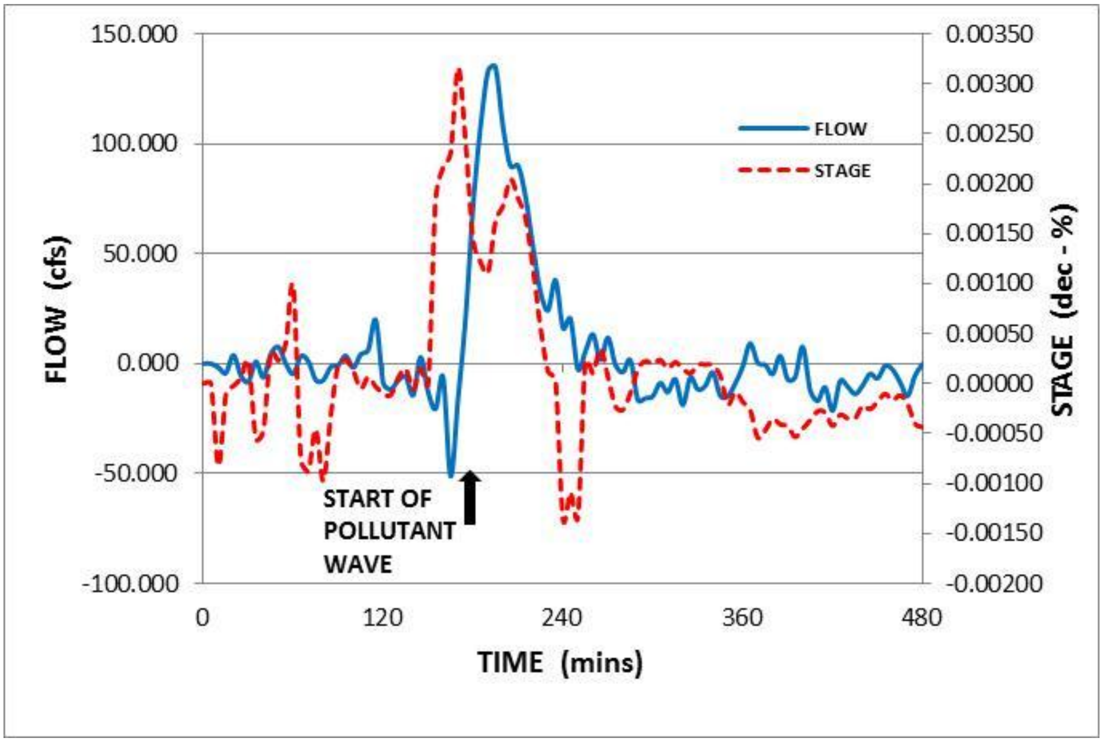

Figure 21: Change Average Flow and Percent Change Average Stage (July 16th, 2006)

This same procedure was repeated for all nineteen monitoring stations and the results are presented in Table 2.

The sampling window sizes for all the stations, except station 1, were set to 5, which yields a 30-minute averaging window. Station 1, which is located at the downstream end of a single watershed, is comprised of mainly urban development and conveys runoff through a close conduit system to Lake Tohopekaliga. As a result, station 1 had a very stable streamflow condition and the averaging window was set to only 20-minutes. A 30-minute averaging window for the remaining sites allowed for successful stabilization of streamflow fluctuations.

The change in average flow triggering threshold for this study ranged from 0.5 cfs to 30 cfs. These values appeared to depend on the size of the watershed and proximity of the monitoring station with respect to lateral inflows. The percent change in average stage triggering

threshold ranged from 0.01% to 0.2%. These values appeared to depend on the width and depth of the streamflow and the degree of water level fluctuation in the stream. Additional information on channel type, instrumentation, land use and watershed location are provided within Table 2.

Table 2: Final Trigger Parameter Threshold Values

SITE	WINDOW SIZE (data points)	CHANNEL TYPE	ADP	LAND USE	WATERSHED POSTION	AVERAGE CHANGE IN FLOW (cfs)	PERCENT CHANGE IN STAGE (dec - %)
1	3	Box Culvert	SW	Urbanized	Outfall	2.0	0.0040
2	5	Pad	SW	Mixed Use	Outfall	10.0	0.0004
3	5	Pad	SW	Residential	Outfall	10.0	0.0004
4	5	U-Channel	SW	Residential	Outfall	2.0	0.0004
5	5	U-Channel	SW	Residential	Inflow	0.5	0.0004
6	5	U-Channel	SW	Residential	Inflow	0.5	0.0004
7	5	U-Channel	SW	Residential	Tributary	2.0	0.0020
8	5	U-Channel	SW	Mixed Use	Midstream	1.0	0.0020
9	5	U-Channel	SW	Mixed Use	Midstream	2.0	0.0020
10	5	U-Channel	SW	Agricultural	Tributary	0.5	0.0010
11	5	Pad	SW	Residential	Tributary	4.0	0.0010
12	5	Pad	SW	Light Industrial	Tributary	3.0	0.0010
13	5	Pad	SW	Mixed Use	Outfall	10.0	0.0010
14	5	Catwalk	SL	Mixed Use	Outfall	18.0	0.0001
15	5	Pad	SW	Mixed Use	Inflow	10.0	0.0010
17	5	Pad	SW	Mixed Use	Midstream	15.0	0.0010
20	5	Catwalk	SL	Mixed Use	Midstream	15.0	0.0002
22	5	Catwalk	SL	Mixed Use	Midstream	30.0	0.0010
23	N/A	N/A	N/A	N/A	N/A	N/A	N/A
24	5	Catwalk	SL	Agricultural	Tributary	4.0	0.0010

Conclusions

Sampling the entire first flush is critical in water quality analysis since large pollutant loads are often associated with the start of the pollutant wave and decline at an exponential rate over time. Untimely triggering can cause the sample bottles to be filled with inaccurate representations of the pollutant plume. In addition, minimizing false triggering of the autosampler is necessary to reduce operational manhours, save battery life and eliminate equipment down time during sample bottle replacement.

This study investigated the concept of dual parameter triggers and moving average parameters to enable the water quality samplers at the start of the pollutant wave while minimizing false triggers. Flow measurements from the ADP Flowmeter and stage measurements from the pressure transducer or shaft encoder were used as the two independent apparatus. The final parameters selected to trigger the autosamplers were the change from average flow and the percent change from average stage. These parameters were shown to catch the start of the pollutant wave with minimal false triggers.

Nineteen autosamplers installed within 6 watersheds of varying land uses within the City of Kissimmee, Florida were evaluated over a 3-year period. Threshold levels for the two parameters were established for all nineteen monitoring sites and each were verified with a minimum of 10 pollutant wave events. The results suggested that using a 20 to 30 minute moving average of 5-minute measurements was sufficient to detect the pollutant wave with minimal false triggers. The percent increase threshold from average stage ranged from 0.01% to 0.20% the incremental increase threshold from average flowrate ranged from 0.5 cfs to 30 cfs. These autosampler trigger parameters were found to vary with land use, streamflow condition and position within the watershed.

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CHAPTER FOUR: EFFECT OF SAMPLING FREQUENCY ON ANNUAL POLLUTANT LOAD ANALYSIS

Introduction

Florida is surrounded by water, and its many internal lakes and rivers have long been recognized for their excellent fishing and boating. This notoriety draws land developers to the lake shores to establish residential and commercial infrastructure. This land development brings with it flood plain alteration, water level stabilization, and increased nutrients, which cause adverse impacts to our lakes. In response, the United States Environmental Protection Agency (EPA) and the Florida Department of Environmental Protection (DEP) have developed Total Maximum Daily Loads (TMDLs) to identify pollutant-impaired waters in Florida. These TMDLs were established based on the best available historical information for flows and pollutant concentration levels in the study area. Since the best available data lacked the necessary detail to directly calculate annual pollutant loads, the DEP resorted to using regression equations for streamflows and empirical Event Mean Concentration (EMC) values for the pollutant concentrations. These methods worked well for the initial planning level analysis, but greater detail is required to validate the annual pollutant load results.

Pollutant loads are a result of a build-up of nutrients and metals over a watershed and then the subsequent washoff of these constituents into streams from excess rainfall runoff. The build-up of pollutants is a direct function of the local watershed characteristics such as land use, surface coverage, flow conveyance and pollutant abatement activities. The washoff of pollutants is also a function of these same watershed characteristics, but it is also directly affected by local climatic conditions. These local climatic conditions can have significantly high spatial and temporal fluctuations throughout the year, which cause large variations in pollutant loading

(Gurr & Nnadi, 2011). To provide accurate pollutant load data necessary to validate impairments to waterbodies, the sampling frequency needs to be often enough to account for the highly variable flows and concentrations of the study area.

Accurate pollutant load data must account for both the background pollutant loads within the stream baseflow and the washoff of pollutants during rainfall events throughout the year. The most intense concentration of pollutants during rainfall events, which occurs within the first 30-minutes of the pollutant wave, must also be analyzed. Therefore, to obtain accurate pollutant load data, an approach must be implemented that uses discrete water quality sampling frequent enough to accurately represent baseflow concentrations while also capturing the pollutant concentrations within rainfall events.

Currently, the minimum DEP water quality sampling frequency to determine the annual pollutant loads to Florida's waterbodies is a single sample collected during each of four consecutive calendar quarters. Although quarterly water quality sampling cannot fully depict the fluctuations in annual pollutant loading, the cost of more detailed sample collection and corresponding laboratory analysis is too high for most state and local governmental budgets. The consequence of these budgetary constraints leads to the absence of more accurate data. Without the investment in accurate pollutant data, severe environmental impacts can go undiscovered and eventually lead to even more costly corrective measures in the future. A balance must be reached between reducing the expense of a pollutant load analysis program and providing the accurate data necessary to identify environmental impacts. This study was conducted to promote efficient stream water quality monitoring by examining the accuracy of current quarterly grab sampling and provide assistance to future planning of more detailed water quality sampling programs.

Background

The altering of the natural environment during the urbanization of watersheds can cause harmful side effects. Building impervious structures reduces the area available for soil infiltration which increases the quantity of stormwater runoff. Altering the ground slope and surface cover of the watershed reduces the time-of-concentration which increases peak runoff rates. These two factors cause accelerated streambank erosion, which is the main transport mechanism for pollutant export to receiving waterbodies (Schueler, 1987). The influx of these pollutants carried by the runoff from developed watersheds can lead to severe water quality impacts. The magnitude of water quality impacts are more pronounced in watersheds with nutrient rich, heavily urbanized surface runoff. However, some studies have shown significant impacts to aquatic life in ponds with less than 10 percent urbanization (Klein et al., 1975).

Nutrients carried by the runoff from urban development can result in large blooms of microscopic algae (phytoplankton) in the lake. As these algae die and settle to the bottom of the lake their decomposition depletes the lake's dissolved oxygen. Not only does the decay of plankton decrease the dissolved oxygen levels but the algae mats that are typically produced in the process allow very little sunlight to reach the plants. This reduced sunlight lowers, or in severe cases even stops the photosynthesis process and thereby prevents the production of dissolved oxygen. When this occurs there is not enough dissolved oxygen produced during the day to compensate for normal daily uses by fish, plants and bacteria. If this condition continues until the dissolved oxygen is depleted the fish will suffocate. In shallow ponds that are heavily vegetated and have high levels of decomposing organic matter this can occur in only a few days.

To protect our nation’s waterbodies, the United States Environmental Protection Agency (EPA) passed the Federal Clean Water Act (CWA) in 1972 which set the framework for the

water quality standards for the entire United States. As a result of the CWA many point sources were eliminated, but in the process it became apparent that nonpoint source loads represented more than 65 percent of pollutants entering our nation's waterbodies (Rushton and Dye, 1993, Livingston, 1985). Research that began prior to the adoption of the CWA documented that a large source of nonpoint pollution is the runoff from urban and industrial areas (Whipple and Hunter, 1977). The Nationwide Urban Runoff Program (NURP) was established in 1978 to collect basic data on the physical and chemical characteristics of urban runoff across the country (EPA, 1983).

Using the results from NURP, a series of management options, named Best Management Practices (BMPs) were developed to control the pollutants transported in urban runoff (Schueler, 1987). These BMPs can be either maintenance or development practices that do not include the construction of a permanent stormwater management structure like street sweeping or Low Impact Development (LID) which are referred to as "non-structural" or they can be actual ponds, swales or physical processes which are referred to as "structural". The effectiveness of each of these BMPs varies according to the targeted pollutant, pollutant concentration, and site conditions. Nitrogen and phosphorus are often selected as targeted pollutants since they are the essential chemical compounds that all plants require to grow and flourish.

In 1999 the Florida legislature adopted the Watershed Restoration Act, which authorized the Department of Environmental Protection (DEP) to assess the quality of Florida's surface waters, identify pollutant-impaired waters and work with other agencies and local stakeholders to finance and implement these BMPs. As a result, DEP has developed Total Maximum Daily Loads (TMDLs) for Florida's impaired waterbodies, but the research is still in its infancy and the

water quality data on the effectiveness of many of these systems to remove nutrients is currently insufficient.

Monitoring the pollutants that enter the tributaries of our receiving waterbodies is essential to understanding the initial pollutant loading and the effectiveness of any nutrient removal systems. Effective pollutant monitoring approaches include the accurate measurement of streamflow and concentrations of each constituent during both baseflow conditions and in response to rainfall events. These individual streamflow and concentration measurements are combined and then summed to obtain the annual pollutant loads to the receiving waterbody. The results of previous annual pollutant loading analysis have been used by EPA to determine Event Mean Concentrations (EMC) for various land uses and to evaluate the effectiveness of nutrient removal systems.

Numerous spreadsheet based studies have suggested that these empirical EMC values can be used in conjunction with streamflow regression equations to determine annual pollutant loads. Other studies rely on empirical EMC values incorporated into computer models to determine annual pollutant loads. These computer models typically use synthetic rainfall distributions in the absence of historical data to determine streamflow rates. Since existing EMC values do not adequately account for regional variations in local weather conditions and watershed characteristics, these empirical EMC methods are usually limited to only planning level analysis. More accurate physically based studies use United States Geological Survey (USGS) stage-discharge rating curves in conjunction with water quality grab samples to depict the streams annual pollutant loading, but it is difficult for grab sampling programs to include the pollutant loads within rainfall events.

Figure 22 and Figure 23 show the quarterly and biweekly volumetric streamflow for one of the monitoring sites in the City of Kissimmee, respectively. Note that the red high-lighted areas in the figures represent the distribution of rainfall events during the sampling periods. Grab sampling can be effective in accurately representing the background pollutant loads within the stream if the sampling is frequent enough to capture the average baseflow concentrations during the sampling period. These stream baseflows rise and fall with not only the seasons, but also with the subsidence of rainfall events and groundwater interactions. The nutrient and metal concentrations during baseflow are typically a residual of the pollutant transport process resulting from rainfall events or subsequent groundwater releases. These baseflow pollutant concentrations vary relatively gradually in comparison to the fluctuation of concentrations during rainfall events. It can be seen in Figure 22 and Figure 23 that a better resolution of baseflow concentrations resulting from rainfall events can be obtained by more frequent grab sampling. However, it is currently not known how significant the variance of average baseflow pollutant concentrations are with respect to sampling frequency when determining annual pollutant loads.

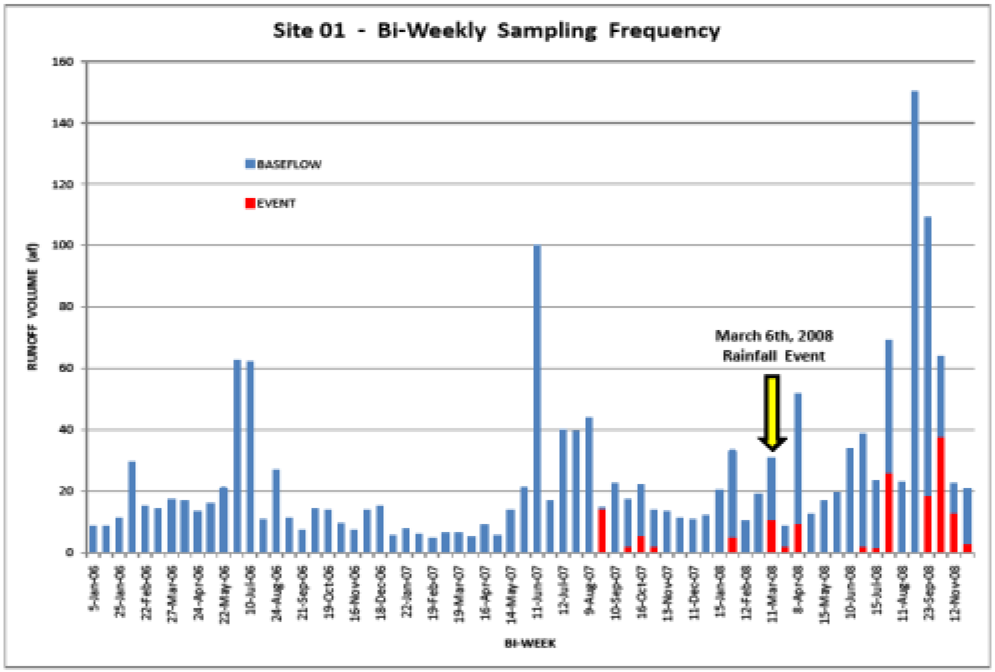

Figure 22: Site 1 - Quarterly Grab Sampling

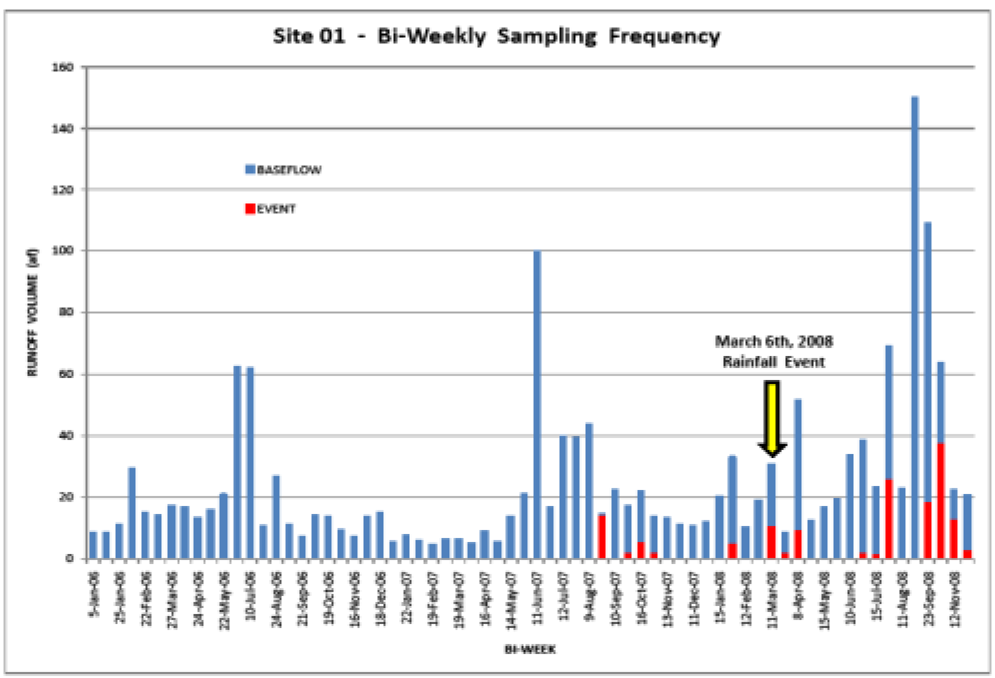

Figure 23: Site 1 - Biweekly Grab Sampling

The runoff from rainfall events typically result in pollutant waves that are associated with greater streamflow rates and higher pollutant concentration levels, which are difficult to capture with grab sampling. Figure 24 shows a rainfall event that occurred at Site 1 on March 6, 2008, which exhibits increased streamflow rates during the pollutant wave.

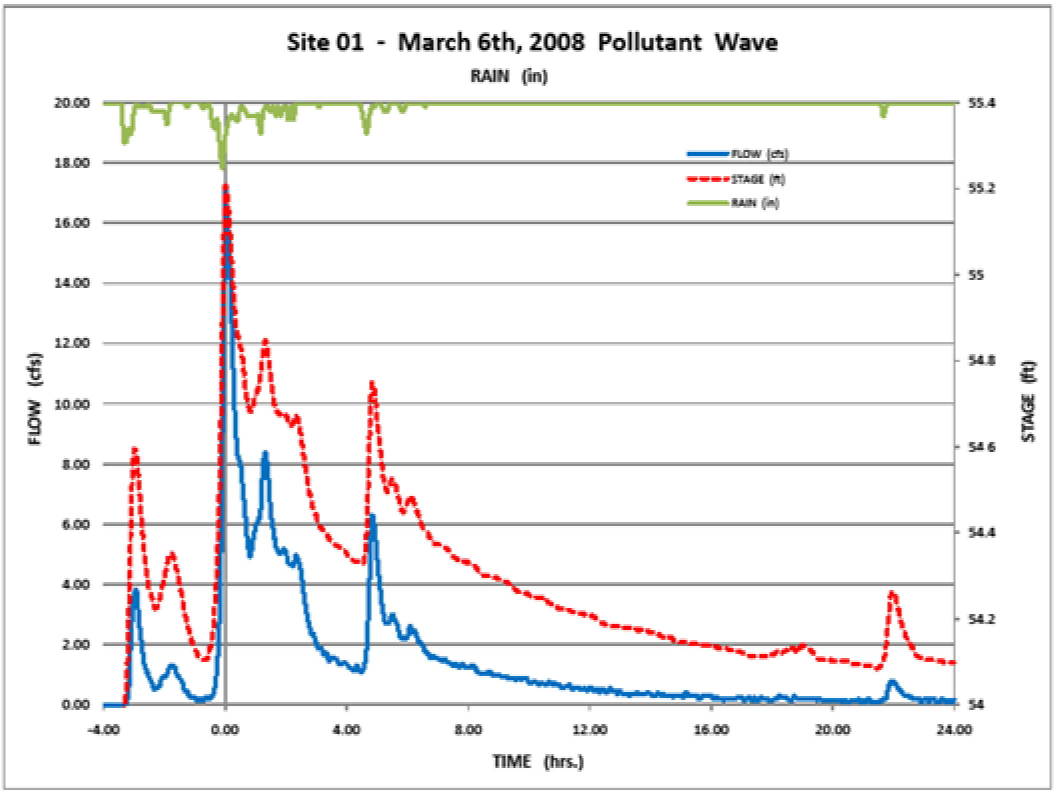

Figure 24: March 6th, 2008 Pollutant Wave

These pollutant waves are caused by significant rainfall intensities and high runoff volumes during storm events that wash off the build-up of nutrients and metals from the watershed surface. Potentially embedded within these pollutant waves is an initial first-flush of pollutants from the watershed, which can convey much higher concentrations of pollutants (Gurr and Nnadi, 2011). Although the streamflow rates and corresponding pollutant concentrations

during these rainfall events are higher than baseflow conditions, baseflow transport usually represents a majority of the total annual pollutant load. How significant of a pollutant increase to annual loadings these rainfall events represent in comparison to the annual loadings from only baseflow concentration is still in question.

The recent rise in the use of automatic water quality samplers has provided much needed rainfall pollutant concentration data to determine annual pollutant loads. Although the data for these systems are much more detailed, the cost of equipment, construction, maintenance, collection and laboratory fees are much higher when compared to traditional grab sampling. Therefore, in recognition of the limited state and local governmental budgets, DEP has mandated that simply collecting a single water quality grab sample during each of four consecutive calendar quarters is sufficient to meet the regulatory requirements for determining annual pollutant loads (DEP, 2006). Stipulations were placed on the location of these grab samples to maintain adequate spatial separation of data points, but there are limited temporal requirements to secure an adequate representation of quarterly average concentrations.

Due to the large fluctuation in pollutant concentrations throughout the year, a Margin of Safety (MOS) is used by DEP in their modeling of impaired watersheds as an attempt to deal with the uncertainties of the limited water quality data. This study was conducted to compare a detailed water quality monitoring approach for the City of Kissimmee with the quarterly DEP water quality grab sampling process. Specifically, this study examines the effects that performing more frequent grab sampling and additional pollutant wave analysis will have on the annual pollutant loads based solely on quarterly grab sampling. The results of this study also provide information for future planning of more detailed water quality sampling programs and offer additional data to assist DEP in establishing future MOS values.

Study Area

The City of Kissimmee monitoring program consists of 19 automatic sampling stations across approximately twenty (20) square miles of surface area with a relatively flat topography and poorly drained soils. This study focuses on four of these stations located within two of the six tributaries of Lake Tohopekaliga as depicted in Figure 25. The first site (Site 1) is located in the downtown area of the city, at the outfall into the lake. It is comprised of mainly urban commercial land use with a closed conduit drainage system. Site 1 is a small single watershed basin with highly impervious surface and very low baseflows. This basin was selected for analysis based on its similarities to most small urban watersheds across Florida. It represents a predictable hydrologic response to rainfall and a relatively stable buildup of nutrients and metals.

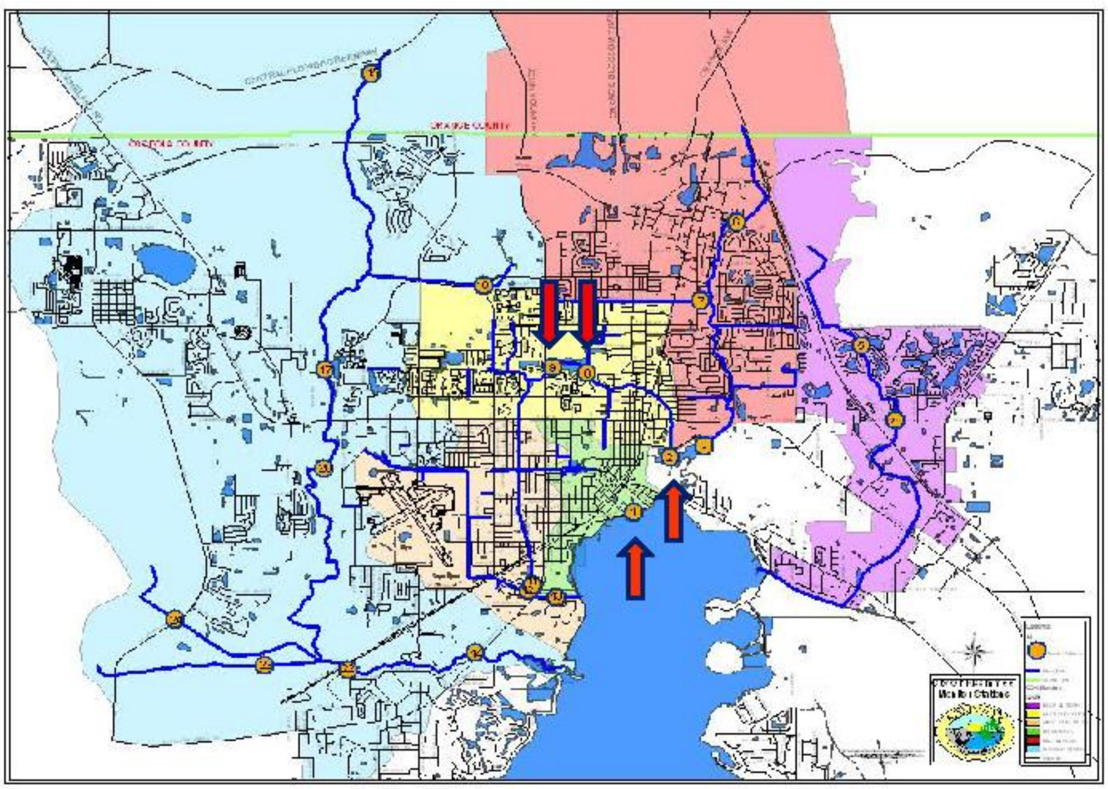

Figure 25: Study Area

The remaining three sites (Sites 2, 8, and 9) are located within the east city ditch basin of the city and are comprised of light commercial and residential land uses. Site 9 is located within a channel, at the inflow to a wet detention pond. This site has a year-round baseflow with a variable tailwater influenced by the stage in the downstream pond. Site 8 is located at the outfall of the wet detention pond and immediately upstream of a weir to control the pond's discharge. This site also has a year-round baseflow from the pond and is usually operating under a free outfall condition. Site 2 is located at the downstream end of east city ditch basin, immediately upstream of the outfall into Lake Tohopekaliga. This site has a year-round baseflow and accepts additional runoff from agricultural land uses. All three of the east city ditch monitoring sites exhibit highly variable hydrologic responses to rainfall and have relatively unstable buildups of nutrients and metals. These sites were selected for analysis based on their similarities to most mixed land use watersheds and to represent the pond inflow, pond outfall and downstream watershed conditions across Florida.

Methodology

The overall goal of this research was to compare pollutant loads based on quarterly grab sampling to a more frequent and detailed sampling schedule. This goal was met in two phases by first conducting a more frequent biweekly grab sampling over a 3 calendar year period from 2006 through 2008 and then performing an intense rainfall event sampling over the single calendar year of 2008. Both phases used continuous 5-minute velocity and stage measurement data collected throughout the 3-year study to determine streamflow rates. The in-situ velocity measurements were made using YSI Sontek Argonaut (SW) acoustical doppler profilers installed near the stream bottoms. In-situ stage measurements were initially made using Sutron SDI-12

Pressure Transducers, but by the end of the first year they were all switched to YSI Level Scout pressure transducers due to maintenance issues.

The first phase of the goal was met by collecting discrete water quality grab samples at all 4 sites, twice per month, spaced approximately two weeks apart (biweekly). Water quality sampling protocols were established to cover field sampling procedures, sample labeling conventions, sample transit and laboratory result verification. Analysis of water quality samples were conducted by laboratories certified in the state of Florida and the continuous field monitoring equipment was maintained on a daily basis by equipment manufacturer licensed personnel.

The results of these biweekly grab samples were extrapolated over the corresponding 5-minute streamflow rate measurements for the approximately two week period between samplings. The sum of these individual biweekly loads were used to determine total loads for ammonia, total kjeldahl nitrogen, nitrate-nitrite, total nitrogen, organic nitrogen, phosphorous, ortho-phosphorous, lead, copper and iron. This procedure was repeated for all 4 sites throughout the 3-year study.

The same total loads procedure was repeated using combinations of only one grab sample result from each month, extrapolated over a one month period, to determine monthly based pollutant loads for the 3-year study. The bi-monthly and quarterly based total pollutant loadings were then determined in a similar procedure by extrapolation of a single grab sample taken every other month and every third month, respectively. The results of the quarterly, bi-monthly and monthly total pollutant loads for all of the constituents throughout the 3-year period were then compared to the total pollutant loads from the more frequent biweekly grab sampling. Since

precise values of the actual pollutant loadings were not known, the degree and direction of deflections were examined numerically and graphically to search for trends in the data.

The second phase of the goal was met by collecting water quality samples at the 4 sites for all significant pollutant waves resulting from rainfall events during 2008. These pollutant waves were detected by using a dual parameter autosampling trigger set to enable water quality sample collection (Gurr and Nnadi, 2011). The two trigger parameters used were an incremental flow increase from the average flow and a percent stage increase from average stage. These rainfall event samples included a separate laboratory analysis of both the initial 30-minutes of the pollutant wave to represent first flush and a flow-weighted composite sample of the entire rainfall event. The composite samples were collected by an ISCO Avalanche refrigerated autosampler programmed to collect 4-bottle samples at a set volume, frequency and duration. The first bottle was filled within the initial 30-minutes by collecting four, 1200 milliliter samples every 10 minutes. Bottles 2, 3, and 4 each collected twenty, 200 milliliter samples spaced every 9, 18 and 45 minutes, which occurred over a period of 3, 6 and 15 hours, respectively. Post-processing of the streamflow data was used to determine the portion of each of the 4 bottles required to create a flow-weighted composite sample of the pollutant wave. The first 30-minute concentrations were determined from an additional laboratory analysis performed on only the first sample bottle.

A very detailed annual pollutant load analysis was performed on all 4 sites using the results of the biweekly grab sampling and the detailed rainfall event samples. First, laboratory results for the initial 30 minutes of each pollutant wave were combined with the measured streamflow rates during their occurrences to represent the first-flush pollutant loads of all of the rainfall events. Second, the laboratory results for the rainfall event composite samples were

combined with the measured streamflow rates from the start until the end of the pollutant wave duration to determine the composite pollutant loads of all of the rainfall events. Finally, the biweekly laboratory results were combined with the measured streamflow rates of all non-rainfall event baseflows to determine a more refined baseflow pollutant load. The sum of these first 30-minute, composite and refined baseflows for the calendar year were used to determine the total annual load at each site and every pollutant as listed in phase one.

To study the initial 30-minute pollutant impact, a slightly less detailed annual pollutant load analysis was performed on all 4 sites by ignoring the first 30-minute sampling, thereby using the results of only the biweekly grab sampling and the composite rainfall event samples. The sums of the composite and refined baseflow loads for the calendar year were used to determine the composite annual load at each site for all of the pollutants. The results of the quarterly, biweekly, composite and total annual pollutant loads for all of the constituents during the 1-year period in 2008 were then compared against the biweekly grab sampling annual loads. This comparison used the same numerical and graphical methods as in phase one to search for trends in the data.

Results

The combined laboratory results for the collected grab samples from all 4 of the sites during the three-year study period are shown in Table 3.

Table 3: Three-Year Pollutant Load Comparison

GRAB SAMPLE POLLUTANT LOAD COMPARISONS FOR YEARS 2006 TO 2008						
POLLUTANT LOAD (%)		SITES 01 - 02 - 08 - 09				
		BI - WEEKLY	MONTHLY	BI - MONTHLY	QUARTERLY
AMMONIA NITROGEN	HIGH	100	111	119	155
	LOW		91	86	73
KJELDAHL NITROGEN	HIGH	100	103	121	134
	LOW		91	78	65
NITRATE - NITRITE	HIGH	100	105	133	132
	LOW		85	74	76
TOTAL NITROGEN	HIGH	100	102	119	130
	LOW		91	82	71
ORGANIC NITROGEN	HIGH	100	106	127	138
	LOW		80	67	57
PHOSPHOROUS	HIGH	100	115	147	187
	LOW		88	51	19
ORTHO - PHOSPHOROUS	HIGH	100	112	151	182
	LOW		92	54	55
LEAD	HIGH	100	145	133	243
	LOW		50	47	47
COPPER	HIGH	100	117	120	163
	LOW		81	68	74
IRON	HIGH	100	106	120	139
	LOW		93	87	81

These results are represented in a percentage deviation from the more frequent biweekly grab sampling. Although the least deviation from biweekly to quarterly grab sampling was noticed in iron (81% to 139%), all of the constituents displayed a significant variation in results as the grab sampling frequency was reduced. The constituent that exhibited the greatest variation when the grab sampling was altered from a biweekly to just a monthly frequency, was lead (50% to 145%). The greatest under measurement of concentrations from quarterly

sampling, when compared to biweekly sampling, was for ortho-phosphorous (55%). Conversely, the greatest over measurement of concentrations from quarterly sampling, when compared to biweekly sampling, was for lead (243%).

The three-year pollutant loads determined from different grab sample frequencies were plotted against each constituent to search for trends in the data. In general, the variations from biweekly grab sampling pollutant loads increased as the sampling frequency decreased for all 4 sites, but not necessarily with the same order of magnitude or direction. Overall, the variations in pollutant loads were relatively symmetric about the 3-year biweekly pollutant loads; however, their deviations varied with constituent and monitoring site.

Figure 26 shows the results of varying the grab sampling frequency at Site 1. This small urban watershed exhibited the most significant variation from biweekly grab sampling with quarterly sampling of ortho-phosphorous. The largest overestimation of biweekly sampling was with quarterly sampling of ortho-phosphorous and the largest underestimation was for lead.

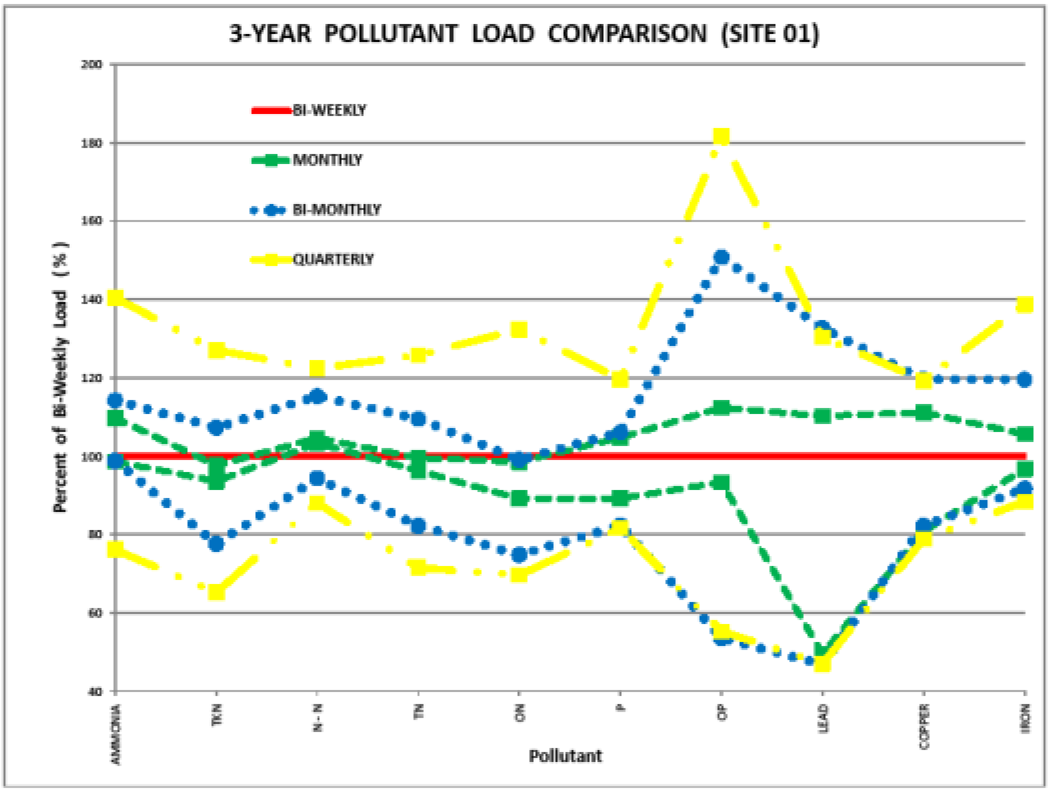

Figure 26: Three-Year Pollutant Load Comparison for Site 1

The results of varying the grab sampling frequency at Site 2 are shown in Figure 27. This site represents the downstream outfall into Lake Tohopekaliga for light commercial, residential and agricultural land use. Pollutant loads from quarterly grab sampling exhibited the greatest variance from the biweekly sampling for organic nitrogen and phosphorous. The most significant overestimation from biweekly grab sampling was for quarterly sampling of phosphorous. The most significant underestimation using quarterly sampling compared to biweekly sampling was for organic nitrogen.

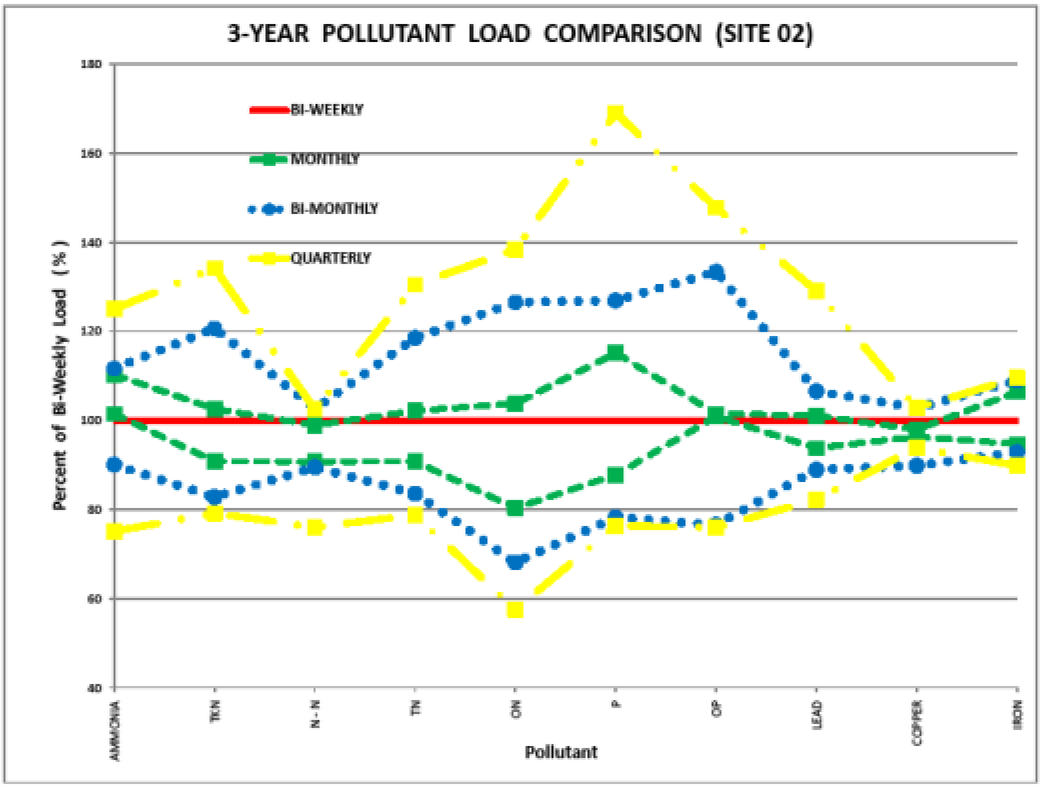

Figure 27: Three-Year Pollutant Load Comparison for Site 2

Site 8 represents the outfall of the wet detention pond. The results of varying the grab sampling frequency at this outfall are shown in Figure 28. Pollutant loads from quarterly grab sampling at site 8 exhibited the greatest variance from the biweekly sampling for phosphorous. The most significant overestimation from biweekly grab sampling was for quarterly sampling of both phosphorous and ammonia. The most significant underestimation from biweekly grab sampling was for quarterly sampling of phosphorous.

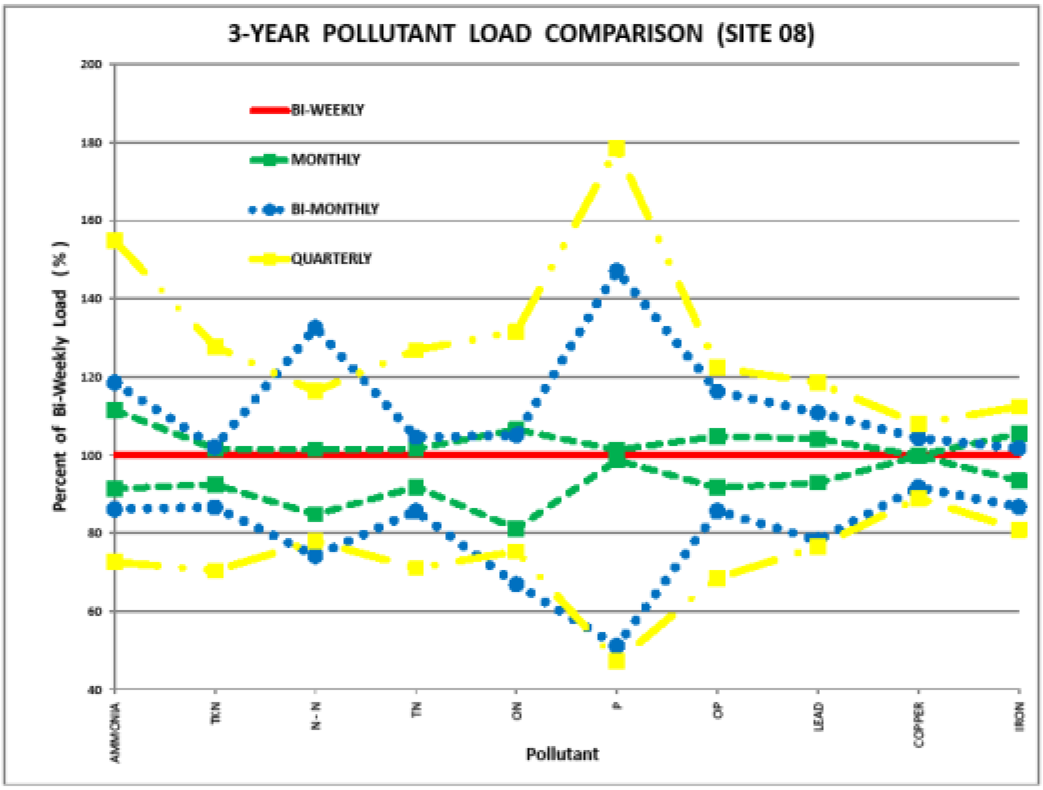

Figure 28: Three-Year Pollutant Load Comparison for Site 8

The results of varying the grab sampling frequency at the inflow to the wet detention pond at site 9 are shown in Figure 29. This site represents the washoff of pollutants from light commercial and residential land uses. Pollutant loads from quarterly grab sampling of all nitrogen constituents at site 9 exhibited the relatively good agreement with biweekly sampling. Less frequent grab sampling at site 9 resulted in relatively large variation of lead loads when compared to the biweekly sampling. The pond inflow exhibited the same quarterly over and underestimations in phosphorous as the pond outfall. Quarterly grab sampling of lead also exhibited a significant overestimation when compared to biweekly grab sampling.

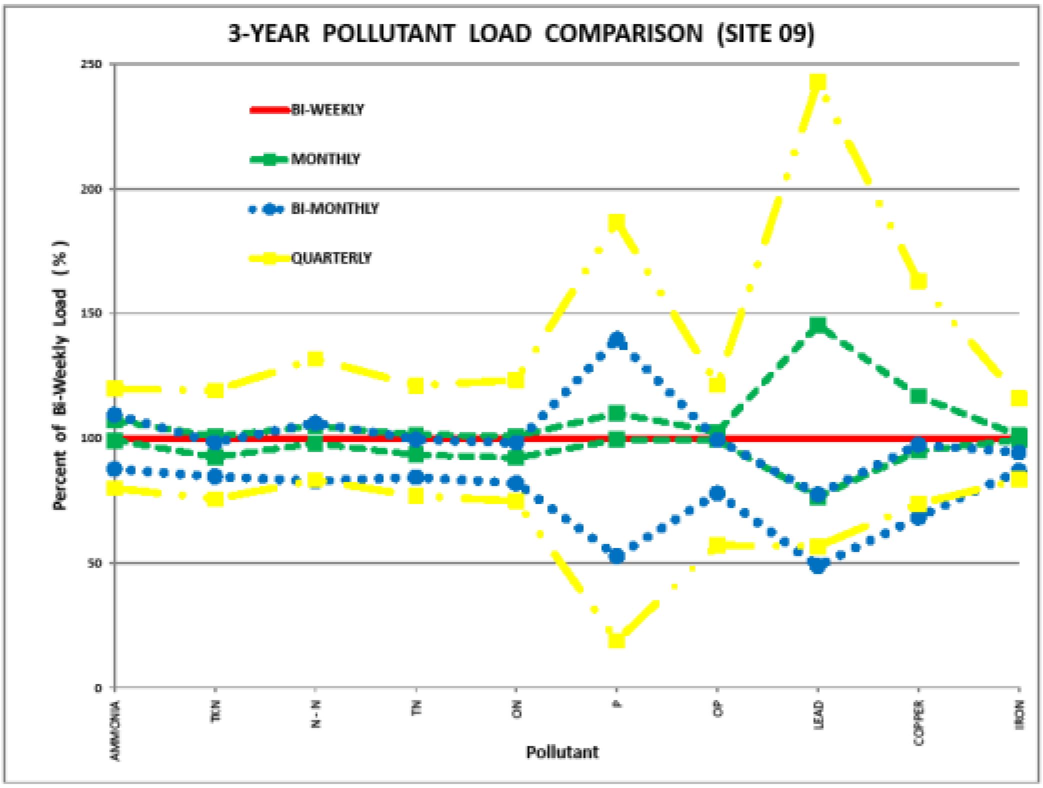

Figure 29: Three-Year Pollutant Load Comparison for Site 9

The combined laboratory results for the 4 quarterly and 24 biweekly collected grab samples from all 4 of the sites during the 2008 study period are shown in Table 4.

Table 4: 2008 Pollutant Load Comparison

GRAB SAMPLE POLLUTANT LOAD COMPARISONS FOR 2008						
POLLUTANT LOAD (%)		SITES 01 - 02 - 08 - 09				
		BI - WEEKLY	TOTAL	COMPOSITE	QUARTERLY
AMMONIA NITROGEN	HIGH	100	144	146	200
	LOW		98	97	22
KJELDAHL NITROGEN	HIGH	100	171	168	139
	LOW		102	102	46
NITRATE - NITRITE	HIGH	100	129	130	153
	LOW		93	93	49
TOTAL NITROGEN	HIGH	100	161	158	139
	LOW		99	99	62
ORGANIC NITROGEN	HIGH	100	122	120	143
	LOW		99	98	55
PHOSPHOROUS	HIGH	100	146	144	284
	LOW		105	105	40
ORTHO - PHOSPHOROUS	HIGH	100	109	108	227
	LOW		99	97	33
LEAD	HIGH	100	260	223	411
	LOW		95	98	32
COPPER	HIGH	100	191	188	272
	LOW		93	95	53
IRON	HIGH	100	165	163	118
	LOW		98	102	73

This table also includes the laboratory results for the composite sampling and total pollutant wave sampling for all of the significant rainfall events during 2008. All of these water quality sampling results are represented in a percentage deviation from the most frequent biweekly grab sampling. Although biweekly grab sampling was used as the norm for this analysis, the more detailed total event sampling is considered to be the most accurate depiction of the annual pollutant loading. The percentages given in Table 4 are presented as a means of comparison to the other sampling frequencies.

Table 4 shows that the deviation from biweekly to quarterly grab sampling was relatively large during the single-year study of 2008. The nutrient values for quarterly sampling varied from 22% to 284%, and the metals varied from 32% to 411% from the biweekly grab sampling. The results of the rainfall event sampling showed the pollutant loads were rarely overestimated by the biweekly grab sampling. The nutrient values for event sampling varied from 93% to 171%, and the metals varied from 93% to 260% from the biweekly grab sampling. The constituent that displayed the least variation from biweekly grab sampling was event sampling of ortho phosphorus (99% to 109%). The constituents that displayed the greatest variation from biweekly grab sampling were event sampling of lead (95% to 260%) and quarterly grab sampling of lead (32% to 411%).

Figure 30 shows the results of varying the sampling frequency at Site 1 during the detailed, 12-month study in 2008. This small urban watershed exhibited relatively good agreement between rainfall event and biweekly grab sampling for nutrient pollutant loads. The rainfall event sampling for lead and copper loading increased significantly from the biweekly grab sampling. The effects of the first 30-minute sampling (total rainfall event sampling) showed only a minor increase in pollutant loadings from the composite rainfall event sampling. The most significant variation of nutrient loading from biweekly grab sampling was with the 4-quarter grab sampling of ortho-phosphorous.

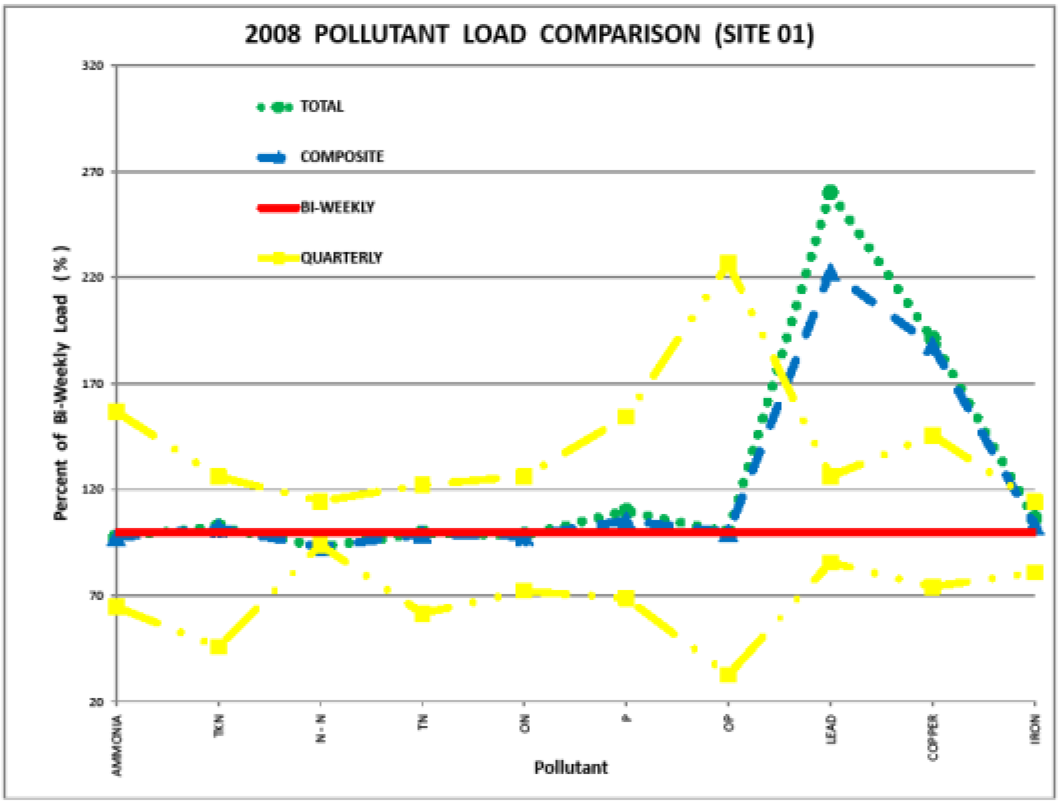

Figure 30: 2008 Pollutant Load Comparison for Site 1

The results of varying the grab sampling frequency during the detailed, 12-month study in 2008 at Site 2 are shown in Figure 31. Pollutant loads from the 4-quarterly grab samples at the downstream outfall into Lake Tohopekaliga exhibited the greatest overall variance from the biweekly sampling. The largest variance of the 4-quarter grab sampling from biweekly sampling occurred for ammonia, phosphorous and ortho-phosphorous. Virtually no difference was observed between the total first flush pollutant loads and the composite rainfall event loads. Except for lead, the results of the rainfall event sampling showed only modest increases in pollutant loads over the biweekly grab sampling. Even though the grab sampling results

displayed strong agreement in annual loads, the spike in lead loads from the rainfall event sampling was relatively large.

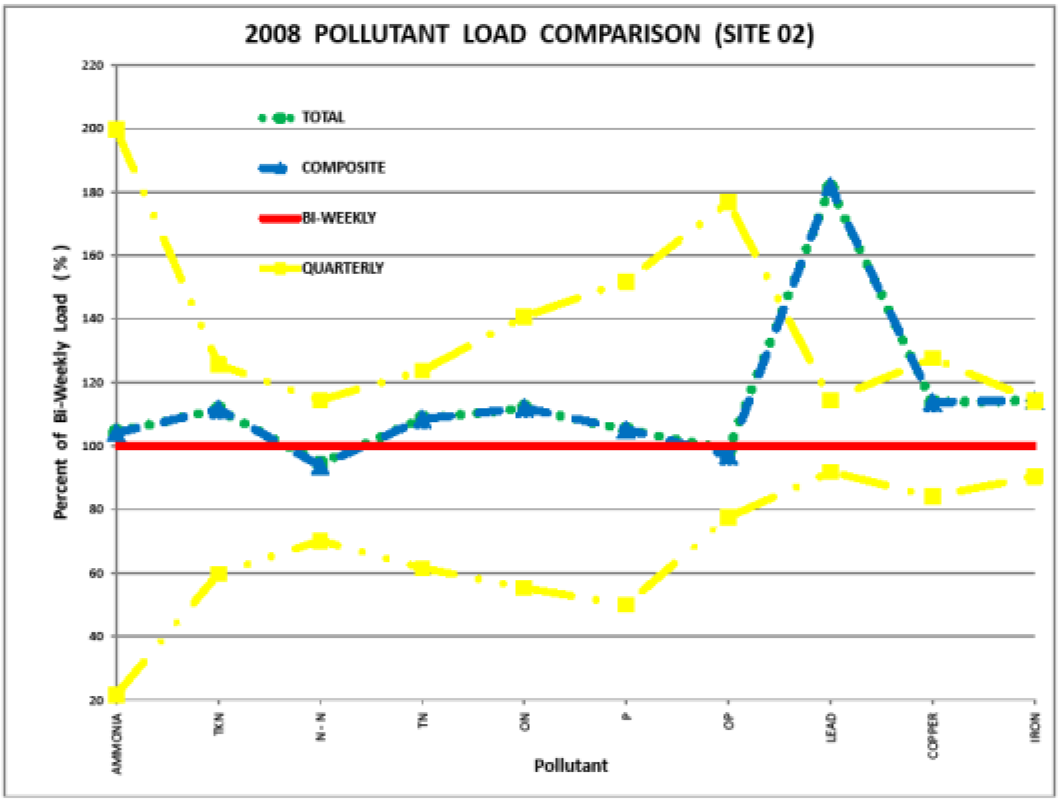

Figure 31: 2008 Pollutant Load Comparison for Site 2

The results for the grab and rainfall event sampling at the wet detention pond outfall (Site 8) during 2008 are shown in Figure 32. Pollutant loads from 4-quarterly grab samples generally resulted in significant variation of annual loads when compared to biweekly grab sampling. The nutrient loads from rainfall event sampling were significantly above the annual loads from biweekly grab sampling. Conversely, the metal loads from rainfall event sampling were slightly below the annual loads from biweekly grab sampling. The largest underestimations of annual

pollutant loads for biweekly sampling when compared to rainfall event sampling were for ammonia and nitrate-nitrite.

The results of the 2008 detailed sampling at the inflow to the wet detention pond at site 9 are shown in Figure 33. In general, the 4-quarter grab samples resulted in large variations from biweekly grab sampling. In particular, significant overestimations of annual pollutant loads were exhibited by the 4-quarterly grab samples for phosphorous, lead and copper. No significant effect was observed for the annual pollutant loads of any constituent at site 9 for the first 30-minute of rainfall event loads. However, the biweekly grab sampling exhibited significant underestimation of annual loads when compared to the rainfall event sampling.

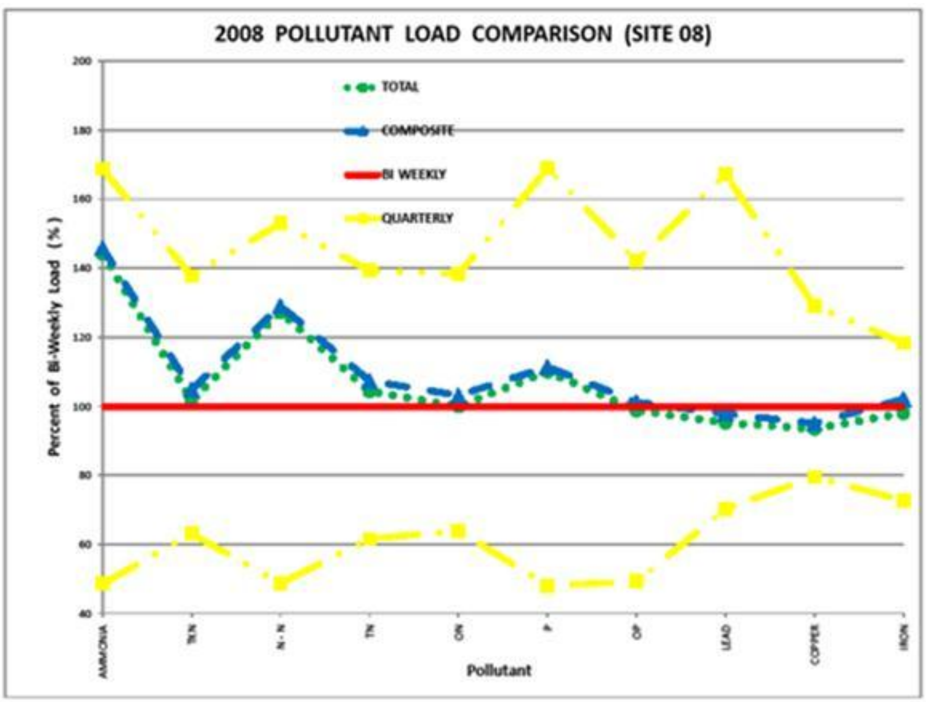

Figure 32: 2008 Pollutant Load Comparison for Site 8

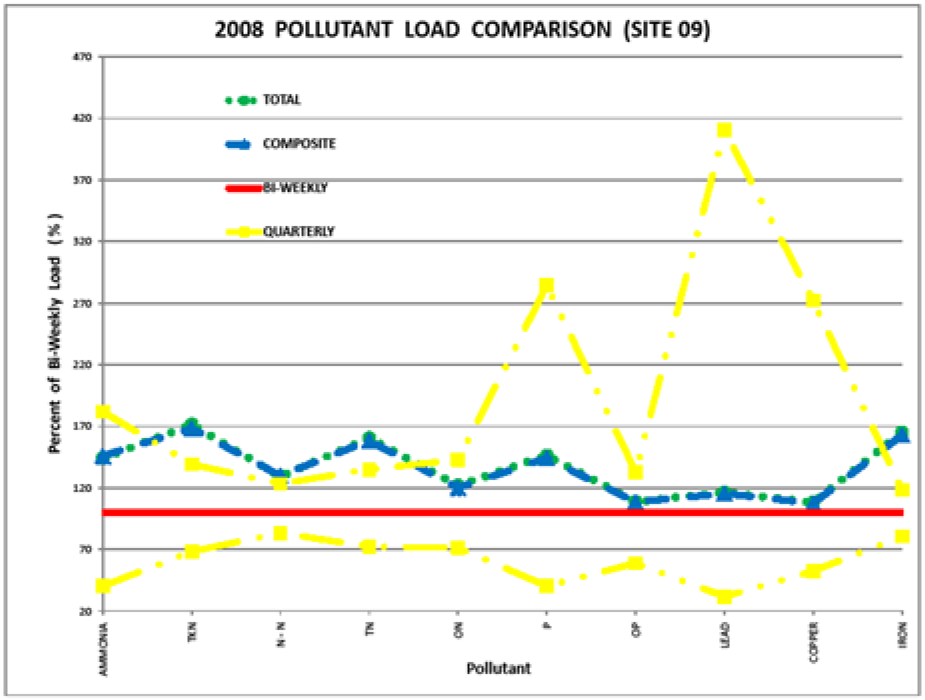

Figure 33: 2008 Pollutant Load Comparison for Site 9

<u>Conclusions</u>

The concentration of pollutants in the streams fluctuated during the 3-year grab sample study. These fluctuations can be caused by many factors, such as, the rate of buildup and subsequent washoff of nutrients from the watershed, dilution from high rainfall volumes, deposition or resuspension of sediments within the stream, and human activities within the watershed. Regardless of the exact cause, for the annual loads from grab sampling to be accurate, the resulting data must be able to represent the average pollutant concentrations throughout these fluctuating values. Although exact values for the annual pollutant loads are not known, this study showed that increasing the sampling frequency not only provided additional data points, but the overall variance between annual pollutant load values was reduced as the sampling frequency was increased. This trend suggests that the most accurate grab sampling

schedule used during this 3-year study to determine annual pollutant loads was biweekly. Further increasing the frequency of grab sampling is projected to result in even more accurate values for annual pollutant loads, but from the observed trend, the effect on annual loadings is expected to be minimal. For the purpose of this study, the biweekly schedule was used as the benchmark for grab sampling.

The DEP currently requires a minimum of 4 water quality samples to be collected during consecutive quarters for establishing annual pollutant loads. If these grab samples fail to collect the average concentrations during these 4 quarters, then resulting annual pollutant loads will not accurately reflect actual stream conditions. Even a single water quality sample, which accurately depicts the stream concentration, but fails to capture the average quarterly concentration is enough to cause extreme error in the annual pollutant load. In order to account for any annual fluctuations in pollutant loading, this study was extended to 12-quarterly grab samples taken over a 3-year period. When these results were compared to the 4 quarterly and 12 monthly grab sampling results for the single-year study of 2008, the 3-year study exhibited much smaller variance from the biweekly annual loads. These results show that if quarterly sampling frequency is extended to a longer duration, it can potentially increase the accuracy of the resulting annual pollutant load values. During this 3-year study, the more frequent, biweekly grab sampling was shown to be even more reliable than the required quarterly schedule at determining annual pollutant loads, but it also should be conducted over a multi-year duration. If the water quality sampling frequency cannot be altered, then the tables and figures within this study can serve as an aid to DEP in selecting appropriate MOS values.

Since part of the fluctuation in stream pollutant concentrations is a result of rainfall events, a more detailed pollutant wave analysis was conducted during the calendar year of 2008.

Although this analysis indicated a presence of pollutants in the first 30 minutes, it revealed that they had little effect on long-term annual loadings. However, the flow-weighted composite sampling of the pollutant waves showed a pronounced increase in annual loadings over biweekly grab sampling for specific constituents at particular monitoring sites. For the small urban watershed, the biweekly grab sampling proved adequate in identifying the annual nutrient loads, but the annual metal loads required composite rainfall event sampling to capture essential concentrations within pollutant waves. Flow into the wet detention pond from the residential and light commercial land uses exhibited significant impact from rainfall event driven pollutant waves. Composite rainfall event samples were necessary at this site to capture the annual pollutant loadings for both nutrients and metals. The outfall of this wet detention pond also required composite rainfall event sampling for all nutrients, but only for certain targeted metals. The final outfall of this system into Lake Tohopekaliga showed only the need to perform rainfall event sampling on certain targeted nutrient and metal pollutants.

Since the pollutant concentrations are typically higher during the washoff from rainfall events, the biweekly grab sampling rarely overestimated the pollutant loads when compared to the more detailed event sampling. Therefore, a long-term, biweekly grab sampling schedule can serve as an indication of the minimum annual pollutant loads. If more accurate annual pollutant loading is required, then composite rainfall event sampling is necessary to target particular constituents at certain monitoring sites. In the absence of these rainfall event monitoring sites, the tables and figures created in this study can aid in estimating the annual loads for similar site conditions. It is recommended that additional analysis be conducted at the remaining 15 monitoring sites within the City of Kissimmee to validate or refine the results of this study.

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CHAPTER FIVE: SUMMARY OF RESEARCH

The current water quality monitoring approaches used in Florida employ techniques that may not accurately depict the actual annual pollutant loading to our receiving waterbodies. Even though many of these current methods use state-of-the-art equipment, their monitoring approaches often fail to measure the entire annual streamflow or capture all of the fluctuations in pollutant concentration. In *Chapter 2*, an innovative approach was used to enable the continuous flow measurement in intermittent streams subjected to frequent dry flow conditions. A concrete U-Channel was introduced which allowed a pressure transducer and ADP to remain submerged and provide direct measurement of stage and velocity in a defined cross-sectional during all flow ranges. This new approach minimized debris blockage and backwater effects in natural, subcritical flow conditions. Seven of these U-Channel structures were designed, constructed, and installed within the City of Kissimmee, Florida. U-Channel measurements were conducted during high and low streamflow conditions over a three year period. The results of this 3-year U-Channel study were compared to detailed velocity indexing measurements and indicate a goodness of fit ranging from 0.87 to 0.99 in contrast to traditional methods, which only range from 0.03 to 0.91.

The continuous, accurate water quantity measurements that the U-Channel provides are needed to determine actual pollutant loading to the tributaries of our nation's lakes. This method also allows for water quality sampling within lower stream depths, which otherwise might be difficult to collect. In addition, the increased runoff rates caused by urbanization have led to increases in erosion and sediment transport. This erosion has affected the morphology of many streams, making it difficult to maintain a constant stream crosssection. This method provides a stable stream crosssection, which is necessary for the stage-area relationship to determine

accurate flow measurements. Finally, a sump is provided to allow suspended sediment to settle away from the in-situ velocity instrument to minimize data loss due to clogging.

In *Chapter 3*, the enabling of the automatic water quality samplers to trigger at the start of a flood wave was investigated. The main focus was on the accurate detection of flood waves while minimizing the potential for false autosampler triggering. Traditional autosampler triggering is performed by measuring rainfall and delaying the start of sampling to catch the pollutant wave. These rainfall triggers are typically limited to small watersheds, where the lag and travel times are both consistent and predictable. In larger, more complex watersheds, stage or flowrate is typically used to trigger the autosampler by either a set threshold level or an incremental increase. These trigger values are difficult to establish due to seasonal fluctuations in streamflow, as well as the spatial and temporal variations in rainfall. In Florida, rainfall patterns can exhibit extreme spatial variability, which can generate a significant pollutant wave at an upstream station while depositing little to no rainfall at a downstream monitoring station. Autosamplers triggered by only rainfall rates will not detect the pollutant wave in many of these situations. This study used a dual parameter trigger to enable the autosampler based on either an incremental increase or a percentage increase over a moving average for flow rate or stage. Nineteen autosamplers installed within 6 watersheds of varying land uses within the City of Kissimmee, Florida were evaluated over a 3-year period. The results indicate using a 20 to 30 minute moving average of 5-minute measurements was sufficient to detect the pollutant wave with minimal false triggers. An increase from average flowrate for the first parameter and a percent increase from average stage for the second parameter were found to yield the best results. A percent increase threshold from average stage ranging from 0.01% to 0.20% yielded the best results for the first parameter. The second parameter threshold was found to be an incremental

increase from average flowrate ranging from 0.5 cfs to 30 cfs. These autosampler trigger parameters varied with land use, streamflow condition and position within the watershed.

Chapter 4 compiled the work of Chapter 2 and Chapter 3 by using the water quantity and water quality data collected by the two new methods in an investigation of sampling frequency. Current requirements of the Florida Department of Environmental Protection (DEP) call for a single water quality sample to be collected during each of four consecutive calendar quarters. Depending on watershed conditions and the targeted pollutant, these quarterly based pollutant loads can vary significantly from a more intense biweekly based sampling schedule. This chapter presents the results of a three-year study in the City of Kissimmee, Florida to collect water quality samples according to varying schedules and determine the optimum sampling frequency for pollutant load analysis. Over this 3-year study it was found that the nutrient loads from 12 consecutive quarterly samples ranged from 19% to 187% of those for a more frequent biweekly based sampling schedule. The 12 consecutive quarterly samples for metal loads ranged from 47% to 243% of those for the biweekly schedule.

A more detailed, 1-year study was also conducted in 2008 to determine the effects of first-flush and rainfall event sampling on annual pollutant loadings. The results of this detailed study were compared to a biweekly grab sampling schedule and were also found to depend on watershed conditions and the targeted pollutant. Rainfall event sampled annual nutrient loads ranged from 93% to 171% of those for a less detailed biweekly based sampling schedule. The annual metal loads for the rainfall event sampling ranged from 93% to 260% of those for the biweekly schedule. The results indicate that in order to obtain accurate annual pollutant load data, a multi-year, biweekly grab sampling frequency should be used at all four of the sites studied. It was also found that rainfall event sampling is needed for certain watershed

conditions, which target particular nutrient or metal pollutants. Furthermore, this study found that although first-flush concentrations were present, they did not represent a significant portion of the annual pollutant loading.

Some of the new methods presented within this research were used in a one-year, nutrient load pilot study provided in *Appendix F*. The focus of this pilot study was to investigate the nutrient removal efficiencies of a man-made, wet detention pond in the City of Kissimmee, Florida. Emphasis was placed on demonstrating the effectiveness of gathering water quality and quantity data in the generation of accurate pollutant loads to Lake Tohopekaliga. The nutrient removal efficiency was performed using both the event mean concentration method and the summation of loads method to check for seasonal variation. Analysis was based on 25 discrete grab samples collected on a bi-monthly basis over a twelve month period. The results indicated that concentration levels of total nitrogen did not seem to vary significantly from its mean value of 0.90 mg/l throughout the year, while there were some relatively lower values in late spring. The study also found that concentration levels of total phosphorus ranged from 0.02 mg/l to 0.48 mg/l, but not in relation to either season or flow volume fluctuations. The wet pond showed a little release of total nitrogen and was actually found to be releasing significant amounts of total phosphorus to the downstream receiving waters.

The new methods for stormwater monitoring presented in this research have provided an alternative to current methods for increasing the level of accuracy and consistency of water quantity and water quality data collected within the City of Kissimmee. New information has also been presented to help EPA predict the annual impacts of pollutant wash-off loads for both baseflows and rainfall events. In addition, the use of methods presented within this research can aid in dealing with the variations in precipitation, land use and other anthropogenic factors to

improve stormwater management, minimize developmental impacts, and prevent environmental hazards.

APPENDIX A: MONITORING STATION LOCATIONS

The first priority for location of the monitoring stations was at the outfalls of each tributary to Lake Tohopekaliga. Refer to the Monitoring Site Location Map in Figure 34 to see a view of how these stations are placed within the City of Kissimmee. These outfall locations were chosen because they were along tracks of land owned by the city and were the closest available land to Lake Tohopekaliga that were still accessible for construction and maintenance personnel.

Shingle Creek outfall is Station Number 14, which is located in a relatively straight portion of the creek, just upstream of a bridge at John Young Parkway. Station Number 3 is the outfall of Mills Slough and it was placed south of US 192 on a long, straight canal section, immediately prior to its discharge to Lake Tohopekaliga. The outfall of Bass Slough does not occur within the corporate limits of the City of Kissimmee, so Station Number 4 was located immediately upstream of the bridge a Boggy Creek Road. This location represents the outfall of water from the City of Kissimmee into the waters of Osceola County. Station Number 4 was placed immediately downstream of a discharge structure for a residential retention pond in a rip rap lined channel. The outfall of East City Ditch is Station Number 2, which is located south of Oak Street along a straight canal section just upstream of Lake Tohopekaliga. Station Number 13 is the outfall of West City Ditch, which is located east of John Young Parkway on a straight canal section just upstream of Lake Tohopekaliga. The final watershed outfall into Lake Tohopekaliga is for the Downtown area. Station Number 1 is the outfall for the Downtown area and it is located along Lakeshore Drive and Dakin Street at the downstream end of a concrete box culvert, which drains into Lake Tohopekaliga.

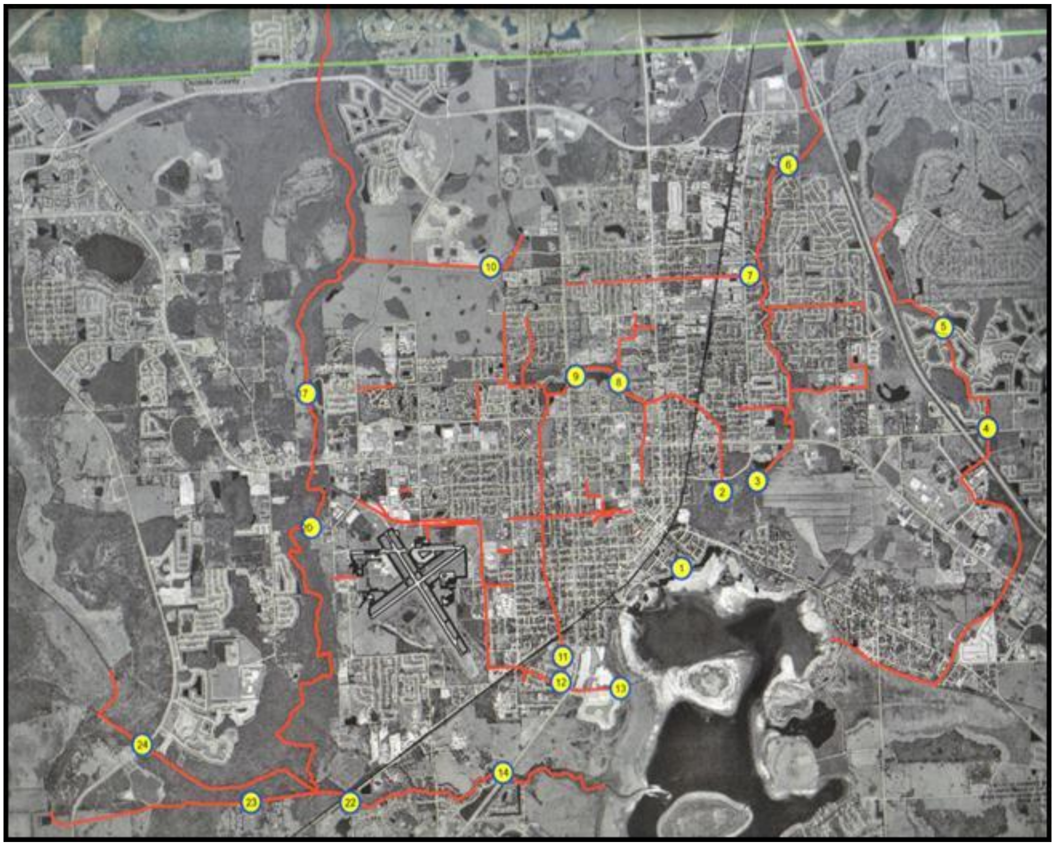


Figure 34: Monitoring Station Locations

The next priority for location of the monitoring stations was to collect data at the inflow points to the study area. Only three of the 6 watersheds have headwaters located outside of the City of Kissimmee. Shingle Creek, Mills Slough and Bass Slough will have monitoring stations placed at these inflow points to determine the pollutant contributions from areas outside of the study area. These monitoring station locations were also chosen because they were on public owned tracks of land and were the closest available land to inflow points that were still accessible for construction and maintenance personnel.

Shingle Creek has 4 points of inflow into the City of Kissimmee limits from Osceola and Orange Counties. Station Number 15 represents the primary channel of Shingle Creek from its headwaters in Orange County. The closest viable location for this monitoring station was in

southern Orange County, on the banks of a straight section of Shingle Creek, just upstream of a bridge at Hunters Creek Boulevard. Station Number 10 is located at the intersection of Thacker Road and Carroll Street, just upstream of a concrete box culvert bridge. This location monitors contributions from Osceola County into Shingle Creek flowing from the east into the City of Kissimmee. Station Number 24 is located on the banks of Browns Canal immediately upstream of a bridge at Poinciana Boulevard. This location monitors contributions from Osceola County into Shingle Creek flowing from the west into the City of Kissimmee. Station Number 23 is located in a drainage ditch east of Poinciana Boulevard which is typically dry. This location monitors contributions from Osceola County into Shingle Creek flowing from the west into the City of Kissimmee when heavy upstream flows cross a weak basin divide.

Mills Slough has two points of inflow into the City of Kissimmee limits from Osceola County. Station Number 6 represents the primary channel of Mills Slough from its headwaters in Orange County. The closest viable location for this monitoring station was downstream of a natural wetland and upstream of a bridge at Mill Run. Station Number 7 is located on a straight section of the drainage ditch, just downstream of a cross drain at Michigan Street. This location monitors contributions from Osceola County into Mills Slough flowing from the west into the City of Kissimmee.

Bass Slough has only one point of inflow into the City of Kissimmee limits from Osceola County. Station Number 5 represents the primary channel of Bass Slough from its headwaters in Osceola County. The closest viable location for this monitoring station was downstream of a natural wetland and adjacent to a cul-de-sac at the northwest side of Lakeshore Subdivision.

The remaining seven monitoring stations are located in the Shingle Creek, East City Ditch and West City Ditch watersheds. Station Numbers 17, 20 and 22 were located on the main

channel of Shingle Creek to provide more information on the distribution of pollutant concentrations and to help identify the flow characteristics of the natural stream. Station Numbers 9 and 8 were placed on the upstream and downstream points, respectively, of a man-made lake, which was constructed for water quality treatment and attenuation. Station Numbers 11 and 12 were placed on two separate contributing sections of the West City Ditch to help isolate light industrial and light commercial pollutant generators. The detailed locations of all of the monitoring stations can be found in Appendix E.

APPENDIX B: MONITORING STATION CONFIGURATIONS

There are two basic configurations of the monitoring stations with slight modifications to accommodate variations in field conditions at each site. The first of these two configurations is the catwalk monitoring station which is depicted in Figure 35. This system has the automatic sampler, telemetry system and measuring equipment mounted at the end of a long, narrow wooden structure. The foundation of the catwalk extends out into the flow of the water and is typically used on wide and deep channels.

Figure 35: Catwalk Monitoring Station

There are 4 catwalk monitoring stations used in this study and all of them are located within the Shingle Creek watershed. These 4 catwalks are located at station numbers 14, 20, 22 and 24, which are all deep flowing channels with wide cross sections.

The second basic monitoring station configuration is the side mounted monitoring station which is depicted in Figure 36. This system has the automatic sampler, telemetry system and measuring equipment mounted on the side of the channel. Pipes come out from the structure, which is mounted on the side bank and extend into the flow of water. This system is typically used on narrow and shallow channels. The flow measuring instrument is anchored to a 3 foot square concrete pad to maintain its orientation and integrity.

Figure 36: Side Mounted Station

There are 7 standard side mounted monitoring stations used in this study and the remaining 9 monitoring stations are modified versions of the side mounted configuration. The 7 standard side mounted sites are located at station numbers 2, 3, 11, 12, 13, 15 and 17, which have flows ranging from 4 to 8 feet deep and shallow cross sections.

The side mounted configuration was modified at station number 1 to accommodate a concrete box culvert. Rather than running a pipe from the equipment structure to the measuring

instruments, a hole was cut into the top of the concrete box culvert and the equipment structure was installed directly over the culvert. The flow measuring instrument was anchored directly to the base of the concrete culvert instead of to a separate concrete pad.

The remaining eight monitoring stations used in this study were located in areas where the depth of flow reaches very shallow levels. In fact, some of these sites even experience dry conditions. Since the measuring and sampling equipment needed wet conditions to operate effectively, a unique concrete channel was designed to maintain a minimum water depth and to direct lower flows across the instruments. With this design, the equipment structure is still located on the side bank, but the concrete pad is replaced with 4 interconnected concrete boxes. Figure 37 shows the installation of 3 concrete boxes. The channel is excavated to suppress the concrete boxes 18 inches lower than the surrounding channel bottom.

Figure 37: Concrete Box Installation

Figure 38 shows the concrete boxes being installed in the excavated portion of the channel. A hole was cut in the side of one of the concrete boxes to allow for pipes to extend from the equipment structure to the measuring instruments. The installation of the measuring equipment into the concrete channel is shown in Figure 39. Once these concrete channels were installed it was important that the water levels remained at a minimum of 12 inches over the instruments so that the flows could be measured and the water quality samples could be collected through the sampling tubes. Figure 40 shows the concrete channel under normal operation.

Figure 38: Suppressed Concrete Channel

Figure 39: Concrete Channel Equipment Installation

Figure 40: Submerged Concrete Channel

The concrete channels were installed at monitoring station numbers 4, 5, 6, 7, 8, 9, 10, and 23. At station number 7 the side slopes of the channel are extremely steep so sheet piling

was driven to provide bank stability. The sheet piling interfered with the hole in the concrete channel so a short wooden structure was constructed to allow for the extension of the pipes to the flow measuring instruments inside the concrete channel.

APPENDIX C: WATER QUALITY SAMPLING

A sampling protocol for the water quality monitoring program of this study was developed. This protocol included procedures during the water quality monitoring phase of the project to assure the samples were properly collected, handled, and transported to the environmental lab for analyses. The main focus of this study was on the grab sampling rather than the automatic sampling of rainfall events.

A training program was held at the City of Kissimmee to demonstrate the sampling protocol to the sampling team. The training included a demonstration of sample collection procedures, sampling equipment, sampler programming, sample container handling, field quality assurance/quality control (QA/QC) procedures, field sampling documentation, equipment decontamination, waste management, sampler maintenance, sample handling, sample documentation, sample labeling, chain-of-custody, and sample shipment.

The grab samples of this study were collected on Mondays, Wednesdays, and Thursdays for 4 weeks per month. This resulted in each site initially being sampled 3 times per month. After a 4 month period, the sample results were analyzed and only critical pollutants were tested from that point on. The number a samples collected was reduced to twice per month on Mondays and Thursdays. This rate of 2 samples per month was maintained throughout the remainder of this study.

The sites were divided into 4 groups designated by A, B, C, and D. Each site group included four to six sites as shown in Table 5. With samples being collected from each site twice a week in four groups, each site was visited twice a month, which also worked well for the equipment maintenance schedule. Duplicate samples were collected at each site during the first, seventh, and fifteenth sampling events of that site.

Table 5: Designated Site Groups

Site Group	Site Number	Basin
A	3, 4, 5, 6, 7	Mill Slough and Bass Slough
B	2, 8, 9, 11, 12, 13	East City Ditch and West City \Ditch
C	1, 10, 15, 17, 20	Downtown and Upper Shingle Creek
D	14, 22, 23, 24	Lower Shingle Creek

These field duplicates were obtained by subsampling the composite samples. Field blank samples were also collected at the same intervals as the duplicate samples for quality control purposes. Filed blanks were used to test the purity of the chemical preservatives, check for contamination of sample containers or equipment that was used in sample collection. These field blanks also helped detect handling, transportation, systemic or random errors.

Sample containers were provided by the certified labs without any information on the labels. Prior to collection of the water quality samples the containers needed to be marked with an identification of where the sample was grabbed, what date it was collected and for which pollutants it needed to be tested. This was accomplished by marking the containers a unique series of letters and numbers that provided the necessary information. The first 3 characters of this alphanumeric series were “COK” to indicate that the sample is for the City of Kissimmee. The next 2 digits indicate at which monitoring station location the sample was collected (i.e., the

site number). The next 6 digits indicate the date the sample which consist of the year (2 digits), the month (2 digits), and the day of month (2 digits), for example “060429” would indicate a sample collected on April 29th, 2006. On occasion a final character was added to the sample identification to indicate by either a letter “D” or “B” if the sample was a duplicate or a blank, respectively.

Certain items were required to be able to collect accurate water quality samples. These items included gloves to keep any contaminants on the hands from getting into the sample containers. The sample containers themselves also needed to be contamination free and in some cases, such as with metals, filled with a stabilizing agent. Sample bottles and composite containers needed to be clean and protected. Ice chests and ice were needed to keep the samples cool during transport. Finally, sampling rods and clean glass jars were needed to actually collect the grab samples from the channel.

Procedures were established for the collection of the water quality samples to maintain their validity. The sampling team used a glass bottle attached to a long sampling pole to collected grab samples manually from the channel. For each site, a different glass bottle was used to avoid any cross contamination between sites. Also, the grab sample was taken from the middle of the channel approximately one-foot below the water surface to avoid any surface or side channel contaminants. The first grab sample from the channel was not used to avoid any potential for residual contaminants in the glass from reaching the sample. Finally, the glass bottle was inverted as it entered the water and then righted once it was fully submerged to avoid the suction of surface water into the sample.

The grab samples from the channel were used to fill a five liter composite container. This composite container was gently rotated 180 degrees (upside down) twice prior to gently

pouring off the sub-sample into the corresponding laboratory container. The laboratory containers were labeled immediately after the sub-samples were collected to avoid any potential notation errors. The unique alphanumeric identification given to each sample bottle was used to separate different samples and avoid later confusion between samples. Once the laboratory containers were filled and labeled they were immediately sealed into plastic bags and placed into ice chests. Ice cubes were then added on top of the sealed laboratory containers as soon as possible to preserve the samples at a temperature near 4 °C. As previously mentioned, powder-free latex gloves were used in handling the samples to avoid any cross contamination between the sites. The Chain-of-Custody (COC) was prepared at each site to document the water quality sample collection and field conditions.

The composite grab samples were placed into containers at each site and transported in ice chests to the state certified laboratories for biological and chemical analysis. The success of the remaining data collection process was based on how well the water quality samples were handled and analyzed. This included the selection of the proper laboratories to analyze the samples, choosing the best means of transporting the containers, maintaining accurate documentation for sample tracking and reviewing the laboratory results to verify any needs for re-testing.

Sixteen different laboratories in the Central Florida area were initially contacted to verify which laboratories could meet the City of Kissimmee project requirements. These 16 laboratories were asked to give their bids for performing the necessary analysis. The final selection of the two laboratories was based on their proximity to the project and the ability to perform the required water quality analysis within the required time frame. The PE LaMoreaux & Associates (PELA) Lab located at 4320 Old Highway 37, Lakeland, Florida was chosen to

perform the nutrient and metal laboratory analysis. These nutrient and metal water quality parameters are listed as items 1 through 26 in Table 6. Test America Lab located at 4310 East Anderson Road, Orlando, Florida performed the analyses for the bacteriological parameters. These bacteriological water quality parameters are listed as items 27 through 29 in Table 6 and the preservative used to maintain the sample is provided on this COC form. The full names of all of the water quality sample collectors and their signatures are required on this COC form, as well as the full names and signatures of who they transferred the water quality samples to for transport to the laboratory. The dates and times of sample transfer from the water quality sample collectors in the field to the transporters and then finally to the laboratory are also included on this COC form. The final step in the process is the signature of the state certified laboratory accepting the successfully transported water quality samples. This process was repeated for every sample collected for this project.

The reported preliminary results of the analyzed samples received from the laboratories were checked for data quality assurance. The laboratories that performed the analyses were asked to verify any doubtful results such as outliers, missing data or syntax issues. In addition, the preliminary results were checked to determine if the laboratory testing methods needed to be revised to better analyze field conditions.

The objective of this study was to determine the water quality condition of the tributaries to Lake Tohopekaliga. This information is used in conjunction with the water quantity data to estimate the corresponding pollutant loadings. To meet these objectives nineteen water quality monitoring stations were constructed at strategic points of the study area. These stations were equipped with instruments to measure the physical, chemical and biological characteristics of the six watersheds contributing flow from the City of Kissimmee to Lake Tohopekaliga. Manual

grab samples were collected and transported to the state certified laboratories to be analyzed and the results were verified.

Table 6: List of Analytical Parameters for Water Analysis

No.	Parameter
1	Ammonia as N
2	Kjeldahl Nitrogen-total
3	Nitrate-Nitrite as N
4	Organic Nitrogen
5	Orthophosphorous
6	Phosphorous, total
7	Residue-filterable (TDS)
8	Residue-nonfilterable (TSS)
9	Biological Oxygen Demand-BOD5
10	Chemical Oxygen Demand (COD)
11	Turbidity
12	pH
13	Chlorophyll a
14	Mercury, total
15	Lead, total
16	Copper, total
17	Zinc, total
18	Iron, total
19	Cadmium, total
20	Chromium, total
21	Nickel, total
22	Arsenic, total
23	Silver, total
24	Barium, total
25	Selenium, total
26	Total Petroleum Hydrocarbons
27	Total Coliforms
28	Fecal Coliforms
29	E. Coli (if Fecal Coliform is positive)

APPENDIX D: MONITORING STATION EQUIPMENT

Support Facilities

Facilities are required at each site to support the operation of the measuring and sampling equipment. These support facilities include such items as a walk-in enclosure, YSI EcoNet data acquisition system, solar panels, three 300 amp 12VDC batteries, wiring junctions, solar regulator, antenna, desiccant, conduit, mounting pipes and a telemetry system. Figure 41 shows a view of these support facilities from the outside of the walk-in enclosure.

Figure 41: Support Facilities

All of the monitoring stations have basically these same support facilities regardless of whether they are the catwalk or side mount configuration. The main difference is the addition of a vault in the side mount configuration. In the catwalk configuration the collection pipes extend directly from the enclosure down the wooden structure into the water. To protect the instruments

from vandalism and to make them more aesthetically pleasing to the eye, a 2’x4’x3’ vault was installed below grade as a conduit junction for the instrument pipes.

Continuous monitoring devices have been installed in all of the monitoring stations except for station number 23. The flows at station number 23 only occur in extreme rainfall events when water levels in Osceola County breech the watershed divide. Since these extreme rainfall events happen too infrequently to maintain a wet condition in the channel, the continuous monitoring equipment could not be permanently installed at this location. Future plans are to construct a mobile sampling unit to be used in this and other similar dry channels.

The remaining 19 monitoring stations have been installed with continuous monitoring equipment which automatically collect water quality samples and gather continuous measurements of the channel parameters. This data will be compiled in the future to determine the pollutant concentrations and estimate the corresponding pollutant loading to Lake Tohopekaliga.

Water Quantity Sampling

The continuous monitoring equipment includes instruments that gather physical data from the channel. One of these is a long, tubular, multi-parameter water quality instrument called the YSI 6600 EDS Component. It is used to measure temperature, dissolved oxygen, pH, Chlorophyll, conductivity, salinity, turbidity and total dissolved solids. Another one of the continuous monitoring equipment is the Sontek Argonaut (SL) which has a shorter, stubbier cylindrical shape used to measure water level, velocity and temperature. Both of these instruments are shown in Figure 42.

Figure 42: Continuous Measurement Devices

The Sontek Argonaut (SL) instrument in which the “SL” stands for “Side Looker” was used on the deeper wider channels in conjunction with the catwalk monitoring station configuration. It is mounted on the side of the channel and measures flow sideways across the channel. For shallow, narrow channel flow conditions a Sontek Argonaut (SW) in which the “SW” stands for “Shallow Water” was used for measuring the same parameters. This unit is mounted at the bottom of the channel and measures in a vertical direction. Figure 43 shows a view of the Sontek Argonaut (SW) unit fastened to a mounting bracket.

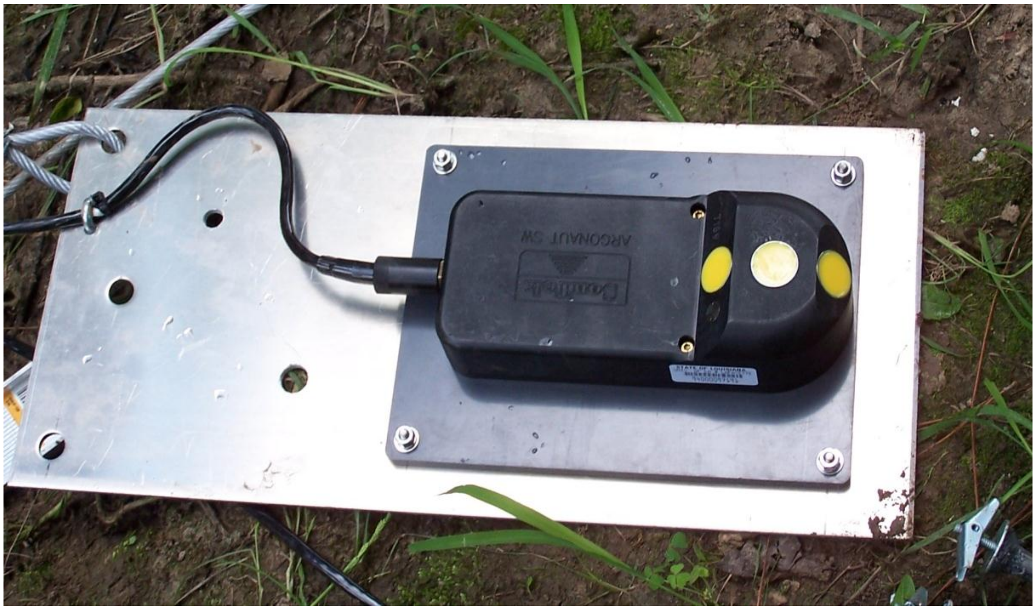


Figure 43: Sontek Argonaut (SW) Flowmeter

Although both Sontek Argonaut units will give a water depth measurement, each station was equipped with a specialized water level measuring instrument for a higher accuracy. The catwalk monitoring stations were equipped with Shaft Encoder instruments and the side mount configurations were equipped with Pressure Transducers for determining the water levels. All stations were outfitted with a Sutron Rain Gauge to measure the rainfall depths and intensities.

Two of the stations were equipped with YSI 9600 Nitrate Analyzers. The limited number of nitrate analyzers was due to budget constraints and the relatively high operation costs. The two sites chosen for these units were Monitoring Station Numbers 9 and 8, which are located on the inflow and outflow points of a man-made lake, respectively. These units provide analysis of nitrate concentrations on a continuous 2-hour interval and are shown in Figure 44.

Figure 44: YSI 9600 Nitrate Analyzer

Water Quality Measurement

The continuous monitoring equipment includes instruments that gather samples of water from the channel. The water quality sampling instrument installed at all sites is called the ISCO Avalanche Refrigerated Autosampler. It is used to drawl specific volumes of water through a tube at selected intervals throughout a duration of time and deposit them into containers. These containers are refrigerated and stored until the samples are ready to be transported to the lab for analysis. The specified volumes and times of sampling are established prior to the time of collection based on the type of pollutants that are expected to be captured for analysis. The ISCO Avalanche Refrigerated Autosampler is shown in Figure 45.

Figure 45: ISCO Avalanche Refrigerated Autosampler

Since this project is focused on collecting the pollutant loading from runoff a 24-hour overall sampling duration was selected with 4 distinct sampling periods. The ISCO automatic sampler was programmed to collect 1200 milliliters of water in the first container 4 times every 10 minutes. This first sampling would last over a 30 minute period and be an indication of the first flush of runoff. Programming was set to continue collecting 200 milliliters of water 20 times every 9 minutes in the second container. The third container was then to collect 200 milliliters of water every 18 minutes, 20 more times. The ISCO automatic sampler was programmed to fill the final container twenty additional times, every 45 minutes with 200 milliliters of water. This programming would last for just over a 24-hour duration.

APPENDIX E: MONITORING STATION MAPS

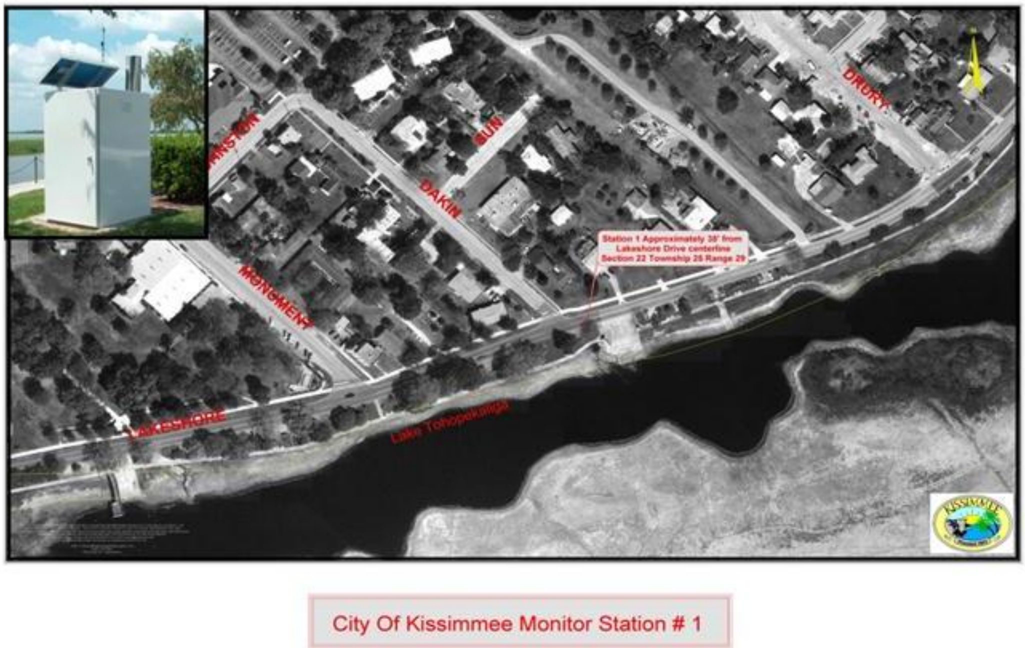
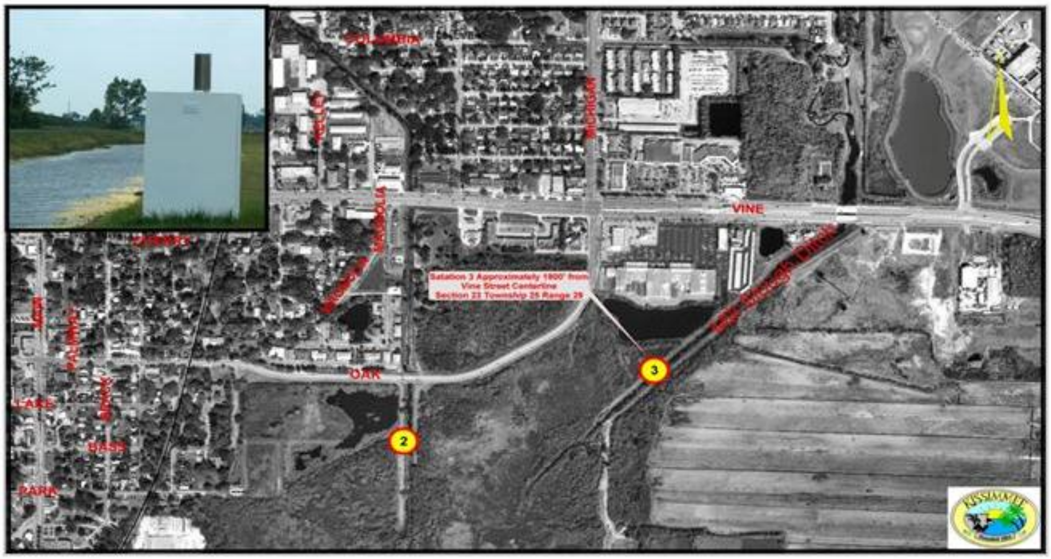


Figure 46: Monitoring Station 01

Figure 47: Monitoring Station 02



City Of Kissimmee Monitor Station # 3

Figure 48: Monitoring Station 03

City Of Kissimmee Monitor Station # 4

Figure 49: Monitoring Station 04

City Of Kissimmee Monitor Station # 5

Figure 50: Monitoring Station 05

City Of Kissimmee Monitor Station # 6

Figure 51: Monitoring Station 06

Figure 52: Monitoring Station 07

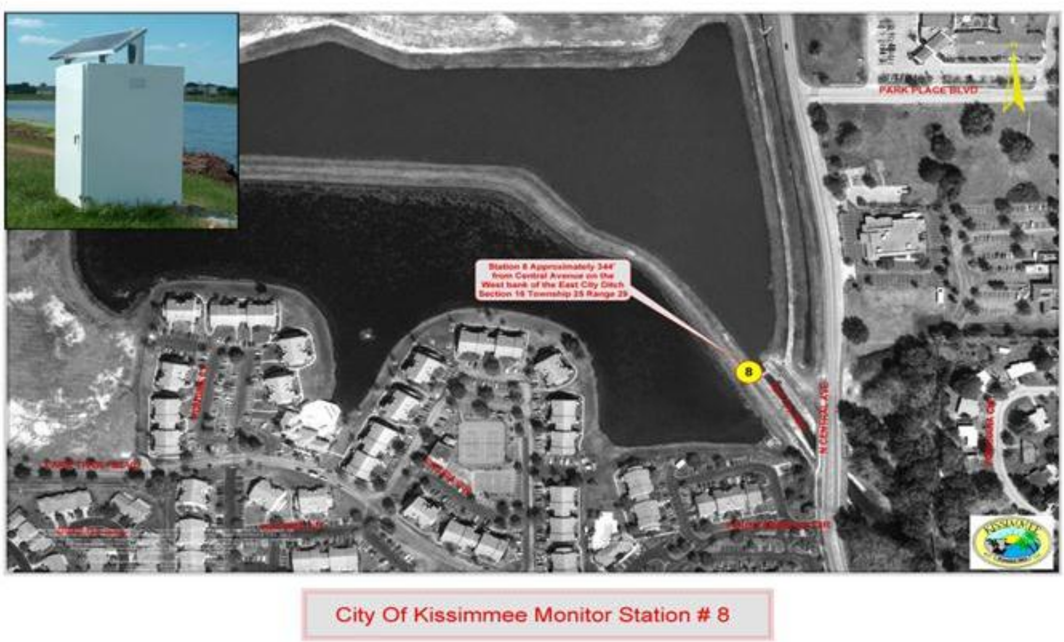


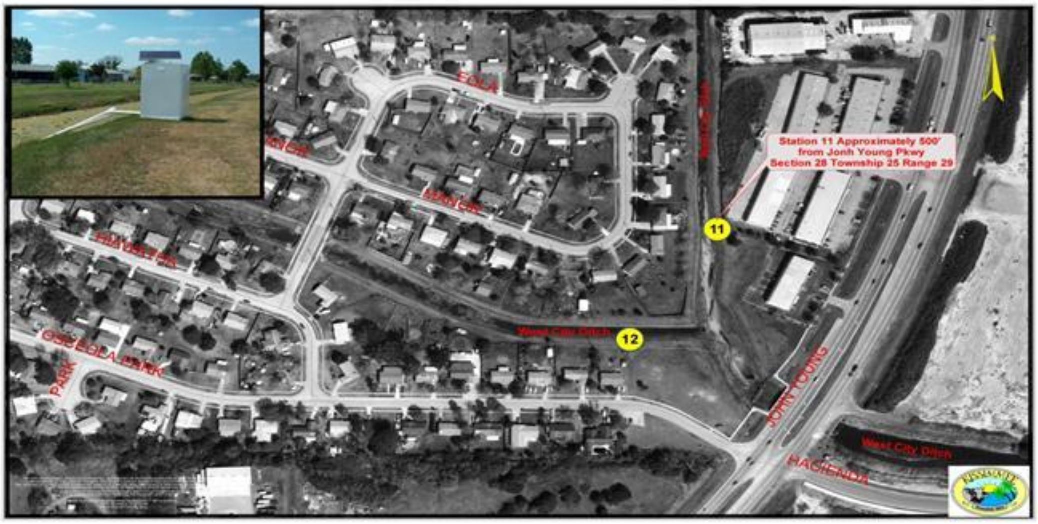
Figure 53: Monitoring Station 08

City Of Kissimmee Monitor Station # 9

Figure 54: Monitoring Station 09

City Of Kissimmee Monitor Station # 10

Figure 55: Monitoring Station 10

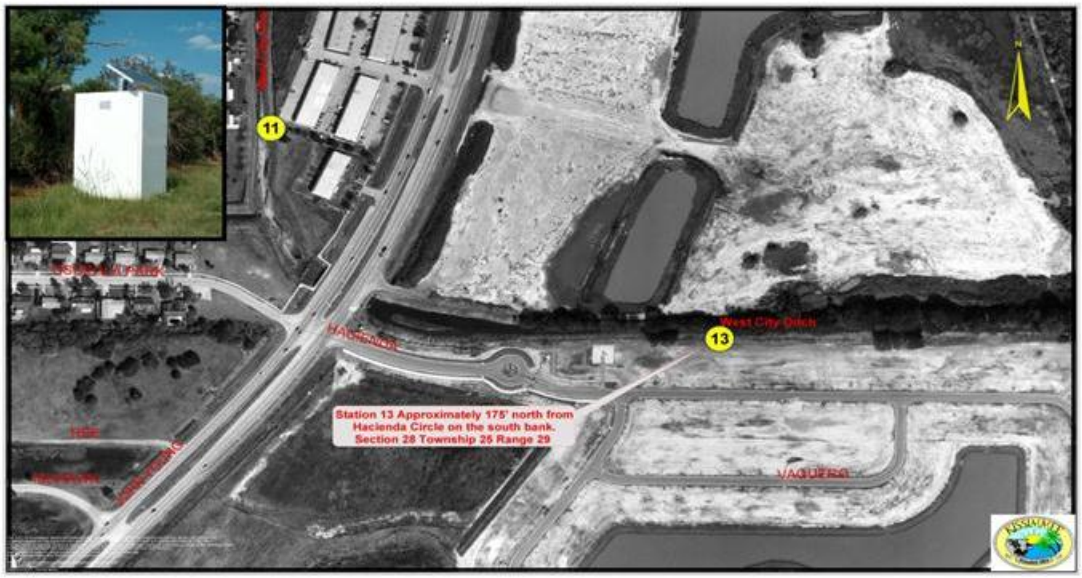


City Of Kissimmee Monitor Station # 11

Figure 56: Monitoring Station 11

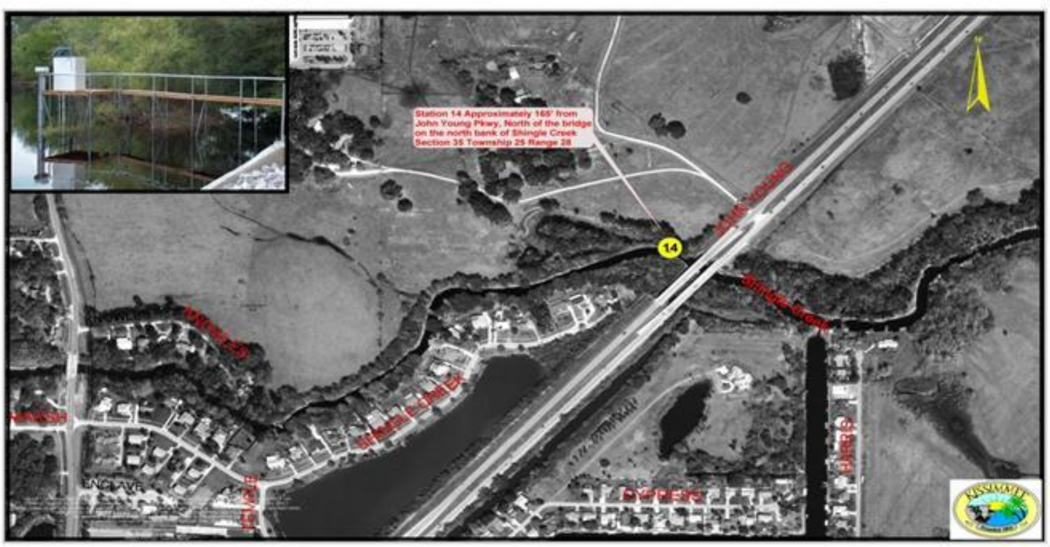
City Of Kissimmee Monitor Station # 12

Figure 57: Monitoring Station 12



City Of Kissimmee Monitor Station # 13

Figure 58: Monitoring Station 13



City Of Kissimmee Monitor Station # 14

Figure 59: Monitoring Station 14

Figure 60: Monitoring Station 15

Figure 61: Monitoring Station 17

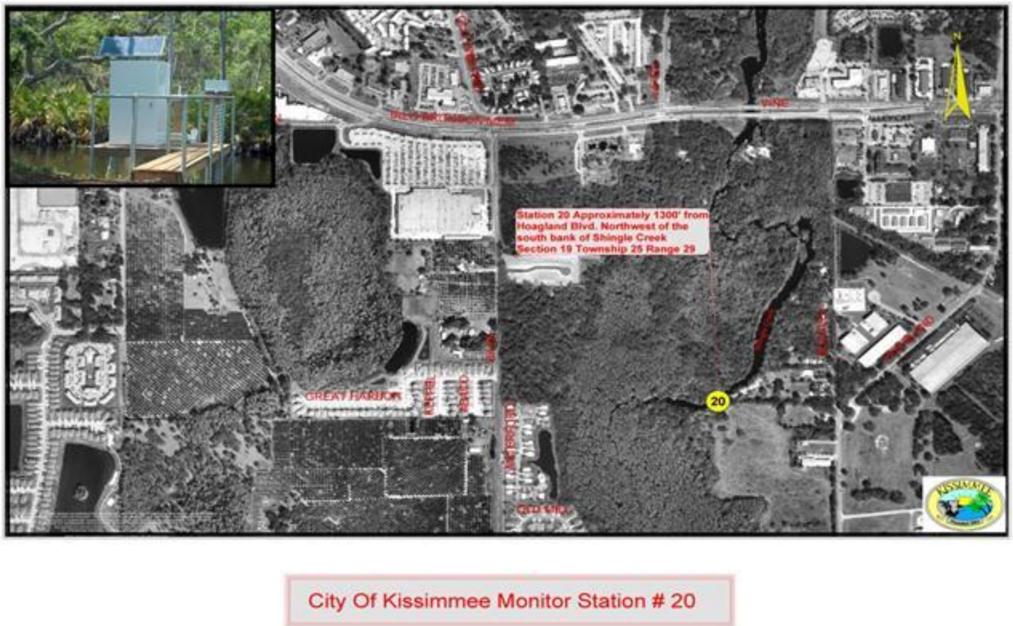
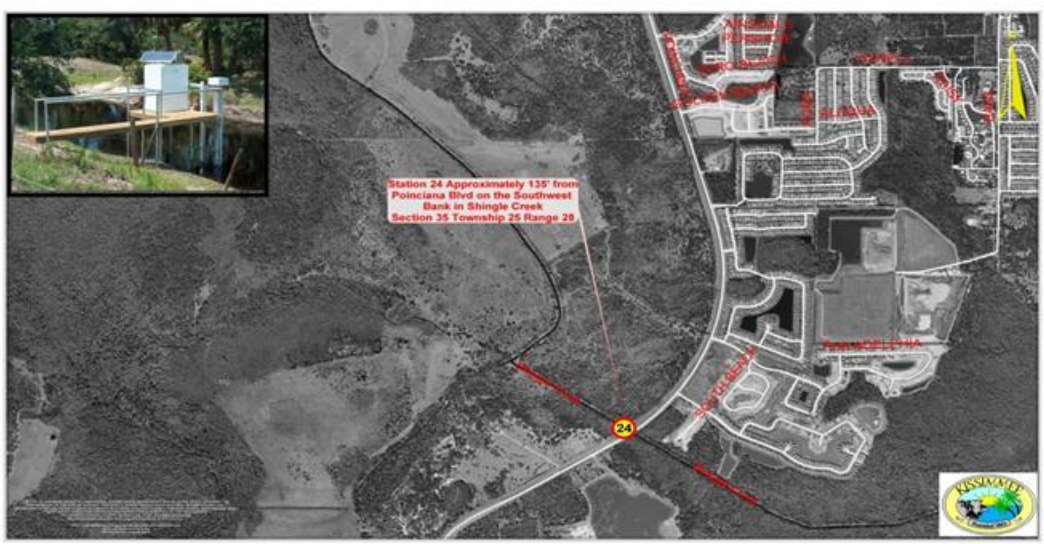


Figure 62: Monitoring Station 20

Figure 63: Monitoring Station 22

City Of Kissimmee Monitor Station # 23

Figure 64: Monitoring Station 23



City Of Kissimmee Monitor Station # 24

Figure 65: Monitoring Station 24

APPENDIX F: PILOT STUDY

Non-Point Source Nutrient Loading in a Wet Treatment Pond

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Running Title: "Pilot Study"

ABSTRACT

This paper presents the results of a one-year nutrient load pilot study in the City of Kissimmee, Florida. The goal of this pilot study was to demonstrate the effectiveness of gathering water quality and quantity data in generating accurate pollutant loads to Lake Tohopekaliga. This paper uses a portion of the study results to focus on the nutrient removal efficiencies of a man-made, wet detention pond. The nutrient removal efficiency was performed using both the event mean concentration method and the summation of loads method to check for seasonal variation. Analysis was based on 25 discrete grab samples collected on a bi-monthly basis over a twelve month period. The results indicated that concentration levels of total nitrogen did not seem to vary significantly from its mean value of 0.90 mg/l throughout the year, while there were some relatively lower values in late spring. The study also found that concentration levels of total phosphorus ranged from 0.02 mg/l to 0.48 mg/l, but not in relation to either season or flow volume fluctuations. The wet pond showed a little release of total nitrogen and was actually found to be releasing significant amounts of total phosphorus to the downstream receiving waters.

Introduction

Florida is surrounded by water, and its many internal lakes and rivers have long been recognized for their excellent fishing and boating. This notoriety draws land developers to the lake shores to establish residential and commercial infrastructure. The altering of the natural environment during the urbanization of watersheds can cause harmful side effects such as decreased infiltration of rainfall, increased runoff volumes and increased occurrences of flooding. These hydrologic factors lead to streambank erosion which is the main transport mechanism for pollutant export to receiving waterbodies (Schueler, 1987). The influx of these nutrients carried by the runoff from developed watersheds can lead to algae blooms which reduce water quality levels.

This paper focuses on a pilot study established to verify the results of a nationwide urban runoff program conducted by the Environmental Protection Agency, which rated permanent wet pool detention basins as very effective in reducing nutrients from urban runoff (EPA 1983). This objective was met by collecting nutrient data at the inflow and outflow points of an in-line, wet detention pond to provide more information on effectiveness of essential nutrient removal.

Background

The United States Environmental Protection Agency (EPA) passed the Federal Clean Water Act (CWA) in 1972 which set the framework for the water quality standards for the entire United States. As a result of the CWA many point sources were eliminated, but in the process it became apparent that nonpoint source loads represented more than 65 percent of pollutants entering our nation's waterbodies (Rushton and Dye, 1993, Livingston, 1985). Research that began prior to the adoption of the CWA documented that a large source of nonpoint pollution is

the runoff from urban and industrial areas (Whipple and Hunter, 1977). The Nationwide Urban Runoff Program (NURP) was established in 1978 to collect basic data on the physical and chemical characteristics of urban runoff across the country (EPA, 1983).

A series of management options, named Best Management Practices (BMPs) were developed to control the pollutants transported in urban runoff (Schueler, 1987). These BMPs can be either maintenance or development practices that do not include the construction of a permanent stormwater management structure like street sweeping or Low Impact Development (LID) which are referred to as "non-structural" or they can be actual ponds, swales or physical processes which are referred to as "structural". The effectiveness of each of these BMPs varies according to the targeted pollutant, pollutant concentration, and site conditions. Nitrogen and phosphorus are often selected as targeted pollutants since they are the essential chemical compounds that all plants require to grow and flourish.

Nitrogen compounds are primary constituents of concern in surface waters due to their limiting role for plant growth. The most important forms of inorganic nitrogen in surface waters are ammonia, nitrite, and nitrate. Organic nitrogen is also an important constituent of surface waters and occurs in both dissolved forms and in particulate organic matter. Nitrogen is the critical element required for protein synthesis and, hence, is critical to life of all plants.

Phosphorus occurs as soluble and insoluble complexes in both organic and inorganic forms in aquatic systems. The principal inorganic form is ortho-phosphate and is the preferred form for plant (macrophyte) growth. Dissolved phosphorus includes both phosphate and dissolved organic phosphorus. Particulate phosphorus includes biological matter such as plankton (microbiota) and phosphorus sorbed on biotic and abiotic suspended particles. Dissolve

organic phosphorus and insoluble forms of organic and inorganic phosphorus are generally not biologically available until they are transformed into soluble inorganic forms.

Phosphorus may be permanently or semi-permanently lost from aquatic ecosystems to the sediments and to a lesser extent as phosphine gas to the atmosphere. Because organic phosphorus can be transformed and used by plants, it is generally sufficient to consider the ambient concentrations of total phosphorus in natural water bodies to anticipate ecological effects. Naturally occurring inputs of phosphorus originate from surface inflows, groundwater inflows, leaching from soils, and atmospheric deposition. Anthropogenic inputs are typically from the use of inorganic phosphorus fertilizers for agriculture and landscaping, the use of animal feeds rich in phosphorus, and from discharges of phosphorus in wastewaters and stormwaters.

Wet detention ponds are a BMP which use a permanent pool of water to remove nitrogen and phosphorus. To maintain a permanent pool it is important to have sufficient surface runoff, fairly impermeable soils, and an adequate base flow to the pond. The effectiveness of these permanent pools at removing nutrients depends on the inflow rate and detention time, which are both functions of the storm intensity, runoff volume, and pond size. These parameters determine the fraction of nutrients captured in the pond for treatment, especially during quiescent periods between events (Woodward-Clyde, 1986)

Sizing of these wet detention ponds typically consider the runoff volume in relation to the water depth and pond length so that settlement of suspended solids is achieved. This pond depth is usually shallow enough so it does not become anoxic and to encourage mixing, which prevents thermal stratification (Schueler, 1987). However, the pond depth should be deep enough so that

wind-generated disturbance of bottom sediments does not cause resuspension of bottom sediments. The recommended permanent pool depths are between one meter and 3 meters.

Study Area

The entire study area encompasses approximately fifty (50) square kilometers of surface area with a relatively flat topography and poorly drained soils. A mixed land use of residential, commercial and agricultural can be found throughout the City of Kissimmee. Stormwater runoff in the city is conveyed to Lake Tohopekaliga by six (6) distinct tributaries which receive flow from the runoff of their respective watersheds.

The pilot study selected a 6 hectare, man-made pond that ranges from 1 to 3 meter deep and has an average depth of approximately 2 meters. This pond receives an average annual flow of 100 liters per second and results in a residence time of 11 days with no littoral zone. Monitoring stations were located at the influent and effluent sections of the pond, which provided data on the hydraulic and hydrologic parameters. The pond is located within the East City Ditch basin, which has a mixture of residential and light commercial land use.

Methodology

The overall goal of this research was to generate accurate and effective water quality and water quantity data to aid in future stormwater management decisions. Specifically, this study aimed to establish automatic monitoring sites throughout the City of Kissimmee, Florida to determine the pollutant loadings into the tributaries of Lake Tohopekaliga. These monitoring sites were located such that inflows from outside the city limits were isolated and external pollutant loads quantified. Also, additional internal monitoring sites were established to

determine the pollutant loads of internal sections of the city. These internal monitoring sites were used to determine the variable pollutant removal efficiencies and hydraulic fluctuations of natural, irregular riverine systems.

Discrete grab samples in tandem with the continuous hydraulic and hydrologic data from monitoring stations were gathered on the man-made pond. The data were analyzed to determine if there are any seasonal variations in pollutant loading or removal efficiencies. For the purpose of this study, only total nitrogen and total phosphorous were examined.

The influent-effluent approach is the most effective method for estimating the pollutant removal efficiency of a structural BMP. This is because pollutant removal efficiencies are based on calculating the difference between influent and effluent loads (Urbonas, 1994). Since the locations of the sampling points are immediately upstream and downstream of the BMP, it makes it possible to isolate the pollutant loads for the mass balance calculations.

The nutrient load transported at any time interval is determined by mass balance. The United States Environmental Protection Agency proposed two different mass balance methods for computing nutrient removal efficiency in a pond. The first method, called the average event mean concentration efficiency ratio (E_{emc}), uses an average of the event mean concentrations from all of the samples distributed over the sum of the sample volumes. The E_{emc} expressed as percentages is computed as follows:

$$E_{emc} = (1 - E_{emc\ (ave)out} / E_{emc\ (ave)in}) \times 100 \qquad \text{Eqn. F-1}$$

Where $E_{emc\ (ave)in}$ is the averaged inflow E_{emc} and $E_{emc\ (ave)out}$ is averaged outflow E_{emc}. The loads are computed as the product of event mean concentrations and the associated volume.

Since the average event mean concentration efficiency method averages all of the event volumes, it gives equal weight to each storm event.

The second method, which is the summation of loads efficiency ratio (E_{sol}), sums the product of each sample volume multiplied by its corresponding event mean concentration. The E_{sol} expressed as percentages is computed as follows:

$$E_{sol} = (1 - SOL_{out} / SOL_{in}) \times 100 \qquad \text{Eqn. F-2}$$

Where SOL_{in} is the summed inflow loads and SOL_{out} is the summed outflow loads. Loads are then computed as the product of event mean concentrations and the associated volume. However, unlike the average event mean concentration method, sample data is required for each events input and output loads.

Both of these methods are independent of the number of samples collected and assume that their results represent the storms that normally occur in the region. The summation of loads method however, assumes the collected samples represent all significant input and output loads (Martin and Smoot, 1986). Even though the average event mean concentration method is capable of providing efficiencies of BMPs, the summation of loads method was found to be a better measure of the overall efficiency of a BMP (Martin and Smoot, 1986). Additional research on BMPs found that where there is a permanent pool, computing pollutant removal effectiveness for individual storms may not be meaningful since the outflow typically has limited relationship to the inflow. For wet detention ponds, it is generally more appropriate to use total loads over the monitored period to compute removal efficiencies (Strecker et al., 1992).

The physical data to fulfill the goal of the pilot study were gathered by continuous monitoring equipment installed at two of the nineteen permanent sites located within the six tributaries of Lake Tohopekaliga. These two monitoring sites were located at the upstream (Site 9) and downstream (Site 8) channels of the pilot study pond (Figure 66). The multi-parameter water quality instrument used to measure temperature, dissolved oxygen, pH, Chlorophyll, conductivity, salinity, turbidity and total dissolved solids was the YSI 6600 EDS Component. While Sontek Argonaut (SW) instrument was installed along the bottom of the channel to measure water level, velocity and temperature. The automatic samplers were not in place during some of the initial water quality sampling so the data used in this study were limited to a twelve-month data period.

Frequency of sample collection and the level of detail for the water quality analysis were constrained by budgetary limits. Water quality sampling protocols were established to cover field sampling procedures, sample labeling conventions, sample transit and laboratory result verification. All analysis of water quality samples were required to be conducted by laboratories certified in the state of Florida and the continuous field monitoring equipment needed to be maintained on a daily basis by personnel licensed by the equipment manufacturers.

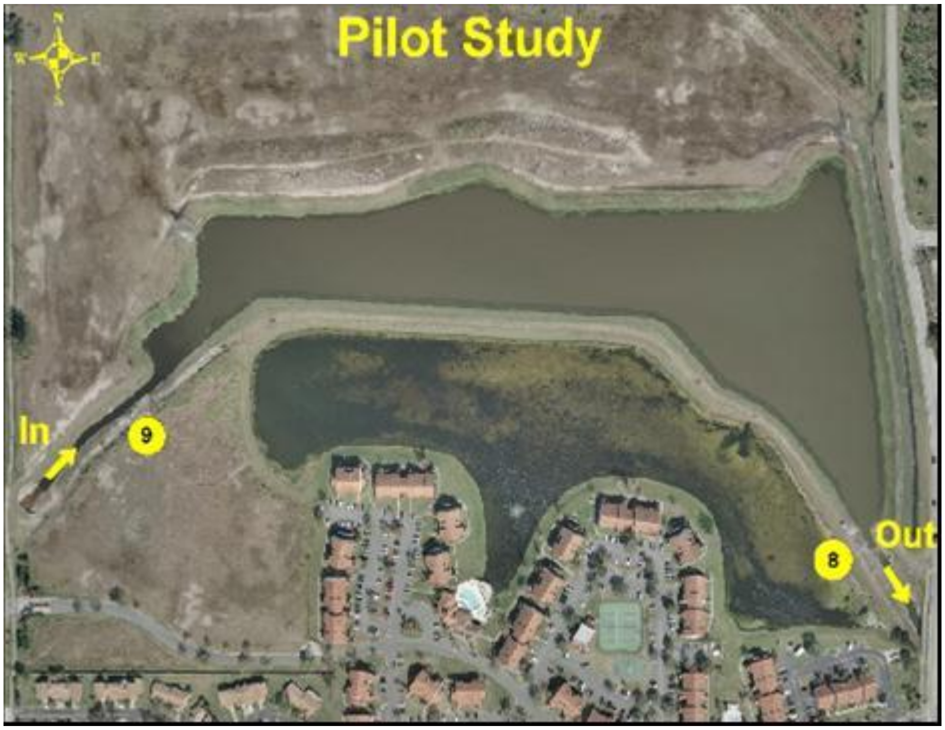

Figure 66: Pilot Study Site (28° 18’ 48”N - 81° 24’ 42”W)

Results

The laboratory results from the collected grab samples in the pilot study pond are shown provided in Figure 67 and Figure 68. These results show little variation of total nitrogen inflow concentration levels from their mean value of 0.90 mg/l whereas the concentration levels of total phosphorus ranged from 0.02 mg/l to 0.48 mg/l during the year-long study period. The total nitrogen outflow concentrations exhibited more variation, but were centered on the mean value of 0.90 mg/l. The levels of total nitrogen and total phosphorus in the pond were within the range for algae bloom generation of 1.5 mg/l and 0.15 mg/l for total nitrogen and total phosphorus, respectively. During the study period, only two minor algae blooms were observed and they only lasted a few days each.

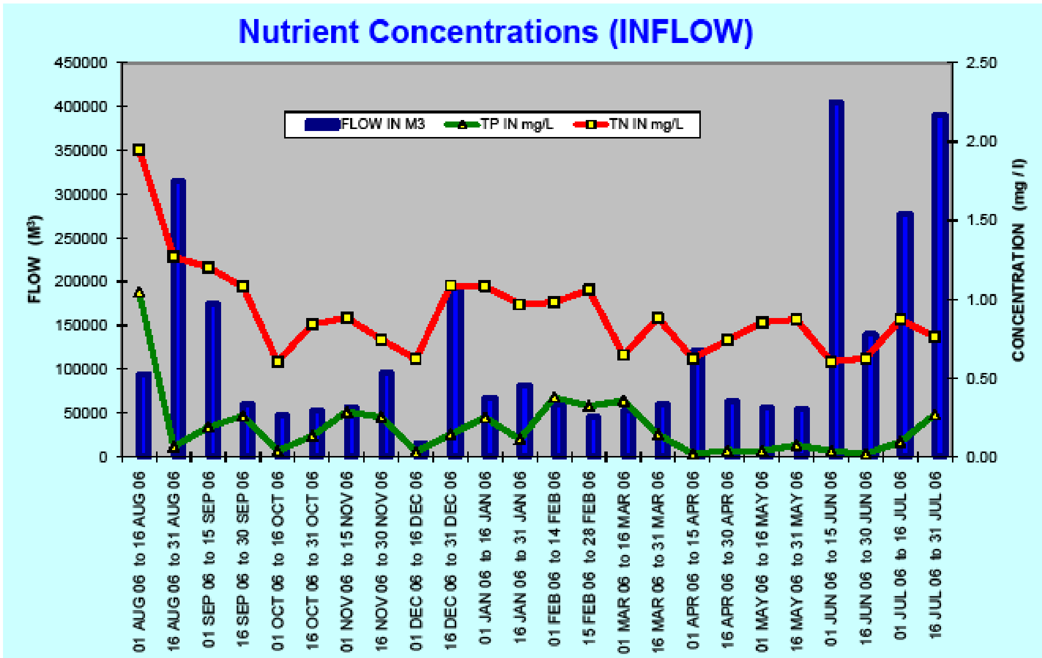

Figure 67: Inflow Nutrient Concentrations

Both E_{emc} and E_{sol} approaches were used to compute the mass balance for nutrient loads. The E_{sol} approach was performed on a bi-monthly and monthly basis whereas the E_{emc} approach was performed for the entire year. Table 7 and Table 8 provide the results of Summation of Loads analysis. Figure 69 and Figure 70 represent the seasonal nutrient loads flowing into and out from the pond for total nitrogen and total phosphorus, respectively.

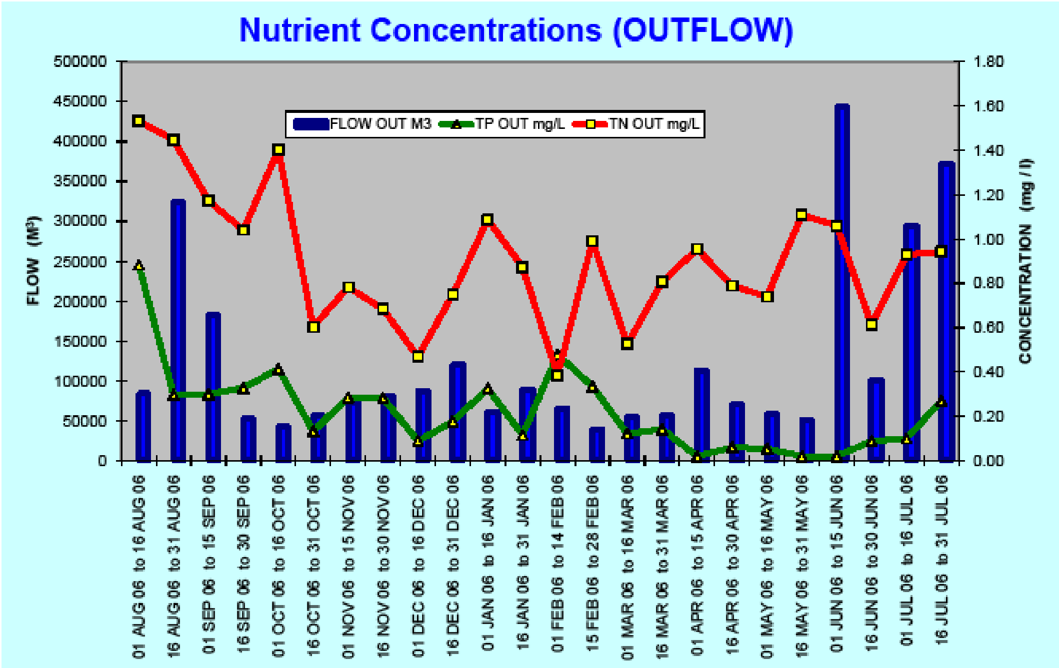

Figure 68: Outflow Nutrient Concentrations

Table 7: Summation of Total Nitrogen Loads

DATES	FLOW VOLUME		TN CONCENTRATION		TN LOAD		REMOVAL EFF.	RAIN
	IN	OUT	IN	OUT	IN	OUT		
	M^3	M^3	mg/L	mg/L	KGS.	KGS.	%	mm
01 AUG 06 to 16 AUG 06	95,089	84,117	1.95	1.53	185	129	30	27
16 AUG 06 to 31 AUG 06	314,801	324,080	1.27	1.45	400	469	-17	144
AUG-06 TOTALS	204,945	204,099	1.61	1.49	292	299	-2	171
01 SEP 06 to 15 SEP 06	174,911	182,054	1.20	1.17	210	213	-1	43
16 SEP 06 to 30 SEP 06	60,285	52,698	1.08	1.04	65	55	16	9
SEP-06 TOTALS	117,598	117,376	1.14	1.11	138	134	3	52
01 OCT 06 to 16 OCT 06	46,752	43,356	0.60	1.40	28	61	-116	4
16 OCT 06 to 31 OCT 06	52,945	56,288	0.84	0.60	44	34	24	9
OCT-06 TOTALS	49,848	49,822	0.72	1.00	36	47	-30	13
01 NOV 06 to 15 NOV 06	57,475	72,563	0.88	0.78	51	57	-12	14
16 NOV 06 to 30 NOV 06	95,453	80,336	0.75	0.68	71	55	23	21
NOV-06 TOTALS	76,464	76,449	0.81	0.73	61	56	8	36
01 DEC 06 to 16 DEC 06	15,462	87,582	0.62	0.47	10	41	-328	60
16 DEC 06 to 31 DEC 06	192,552	120,432	1.09	0.75	209	90	57	62
DEC-06 TOTALS	104,007	104,007	0.85	0.61	109	66	40	122
01 JAN 06 to 16 JAN 06	67,893	61,674	1.08	1.09	73	67	9	3
16 JAN 06 to 31 JAN 06	81,980	88,811	0.97	0.87	79	77	2	27
JAN-07 TOTALS	74,936	75,243	1.03	0.98	76	72	5	30
01 FEB 06 to 14 FEB 06	59,253	65,375	0.98	0.39	58	25	57	11
15 FEB 06 to 28 FEB 06	45,337	39,472	1.06	0.99	48	39	19	3
FEB-07 TOTALS	52,295	52,423	1.02	0.69	53	32	39	15
01 MAR 06 to 16 MAR 06	53,251	55,008	0.65	0.53	34	29	16	13
16 MAR 06 to 31 MAR 06	59,957	57,123	0.88	0.81	53	46	12	2
MAR-07 TOTALS	56,604	56,065	0.76	0.67	44	38	14	15
01 APR 06 to 15 APR 06	121,301	113,481	0.63	0.95	76	108	-43	54
16 APR 06 to 30 APR 06	63,267	71,085	0.74	0.79	47	56	-20	2
APR-07 TOTALS	92,284	92,283	0.68	0.87	61	82	-34	56
01 MAY 06 to 16 MAY 06	55,767	59,090	0.85	0.74	48	44	8	18
16 MAY 06 to 31 MAY 06	54,955	51,522	0.87	1.11	48	57	-19	1
MAY-07 TOTALS	55,361	55,306	0.86	0.92	48	50	-5	19
01 JUN 06 to 15 JUN 06	405,472	443,811	0.60	1.06	245	471	-92	219
16 JUN 06 to 30 JUN 06	140,309	101,543	0.63	0.61	88	62	29	39
JUN-07 TOTALS	272,890	272,677	0.62	0.84	166	267	-60	258
01 JUL 06 to 16 JUL 06	277,160	294,991	0.87	0.93	242	274	-13	90
16 JUL 06 to 31 JUL 06	390,881	372,411	0.76	0.94	298	351	-18	143
JUL-07 TOTALS	334,020	333,701	0.82	0.94	270	313	-16	233
TOTALS	1,491,254	1,489,451	NA	NA	1,355	1,456	-7	1,019
AVERAGE EVENT MEAN CONCENTRATION			0.91	0.98				

Table 8: Summation of Total Phosphorus Loads

DATES	FLOW VOLUME		TP CONCENTRATION		TP LOAD		REMOVAL EFF.	RAIN
	IN	OUT	IN	OUT	IN	OUT		
	M^3	M^3	mg/L	mg/L	KGS.	KGS.	%	mm
01 AUG 06 to 16 AUG 06	95,089	84,117	1.047	0.882	99.6	74.2	25	27
16 AUG 06 to 31 AUG 06	314,801	324,080	0.065	0.296	20.5	96.1	-369	144
AUG-06 TOTALS	204,945	204,099	0.556	0.589	60.0	85.1	-42	171
01 SEP 06 to 15 SEP 06	174,911	182,054	0.189	0.300	33.0	54.6	-66	43
16 SEP 06 to 30 SEP 06	60,285	52,698	0.260	0.329	15.7	17.3	-10	9
SEP-06 TOTALS	117,598	117,376	0.224	0.314	24.3	36.0	-48	52
01 OCT 06 to 16 OCT 06	46,752	43,356	0.037	0.414	1.7	18.0	-934	4
16 OCT 06 to 31 OCT 06	52,945	56,288	0.132	0.132	7.0	7.4	-6	9
OCT-06 TOTALS	49,848	49,822	0.085	0.273	4.4	12.7	-191	13
01 NOV 06 to 15 NOV 06	57,475	72,563	0.286	0.286	16.4	20.7	-26	14
16 NOV 06 to 30 NOV 06	95,453	80,336	0.252	0.283	24.1	22.8	5	21
NOV-06 TOTALS	76,464	76,449	0.269	0.285	20.2	21.7	-7	36
01 DEC 06 to 16 DEC 06	15,462	87,582	0.031	0.089	0.5	7.8	-1496	60
16 DEC 06 to 31 DEC 06	192,552	120,432	0.143	0.177	27.4	21.3	22	62
DEC-06 TOTALS	104,007	104,007	0.087	0.133	14.0	14.5	-4	122
01 JAN 06 to 16 JAN 06	67,893	61,674	0.250	0.325	17.0	20.0	-18	3
16 JAN 06 to 31 JAN 06	81,980	88,811	0.112	0.115	9.1	10.2	-12	27
JAN-07 TOTALS	74,936	75,243	0.181	0.220	13.1	15.1	-16	30
01 FEB 06 to 14 FEB 06	59,253	65,375	0.379	0.479	22.4	31.3	-39	11
15 FEB 06 to 28 FEB 06	45,337	39,472	0.321	0.336	14.6	13.3	9	3
FEB-07 TOTALS	52,295	52,423	0.350	0.407	18.5	22.3	-20	15
01 MAR 06 to 16 MAR 06	53,251	55,008	0.357	0.121	19.0	6.7	65	13
16 MAR 06 to 31 MAR 06	59,957	57,123	0.142	0.142	8.5	8.1	5	2
MAR-07 TOTALS	56,604	56,065	0.250	0.132	13.8	7.4	46	15
01 APR 06 to 15 APR 06	121,301	113,481	0.020	0.020	2.4	2.3	6	54
16 APR 06 to 30 APR 06	63,267	71,085	0.040	0.060	2.5	4.3	-69	2
APR-07 TOTALS	92,284	92,283	0.030	0.040	2.5	3.3	-32	56
01 MAY 06 to 16 MAY 06	55,767	59,090	0.037	0.054	2.1	3.2	-55	18
16 MAY 06 to 31 MAY 06	54,955	51,522	0.074	0.020	4.1	1.0	75	1
MAY-07 TOTALS	55,361	55,306	0.056	0.037	3.1	2.1	31	19
01 JUN 06 to 15 JUN 06	405,472	443,811	0.037	0.020	15.1	8.9	41	219
16 JUN 06 to 30 JUN 06	140,309	101,543	0.020	0.089	2.8	9.0	-221	39
JUN-07 TOTALS	272,890	272,677	0.029	0.054	8.9	8.9	0	258
01 JUL 06 to 16 JUL 06	277,160	294,991	0.094	0.100	26.1	29.5	-13	90
16 JUL 06 to 31 JUL 06	390,881	372,411	0.271	0.271	105.8	100.8	5	143
JUL-07 TOTALS	334,020	333,701	0.182	0.185	65.9	65.1	1	233
TOTALS	1,491,254	1,489,451	NA	NA	249	294	-18	1,019
AVERAGE EVENT MEAN CONCENTRATION			0.167	0.198				

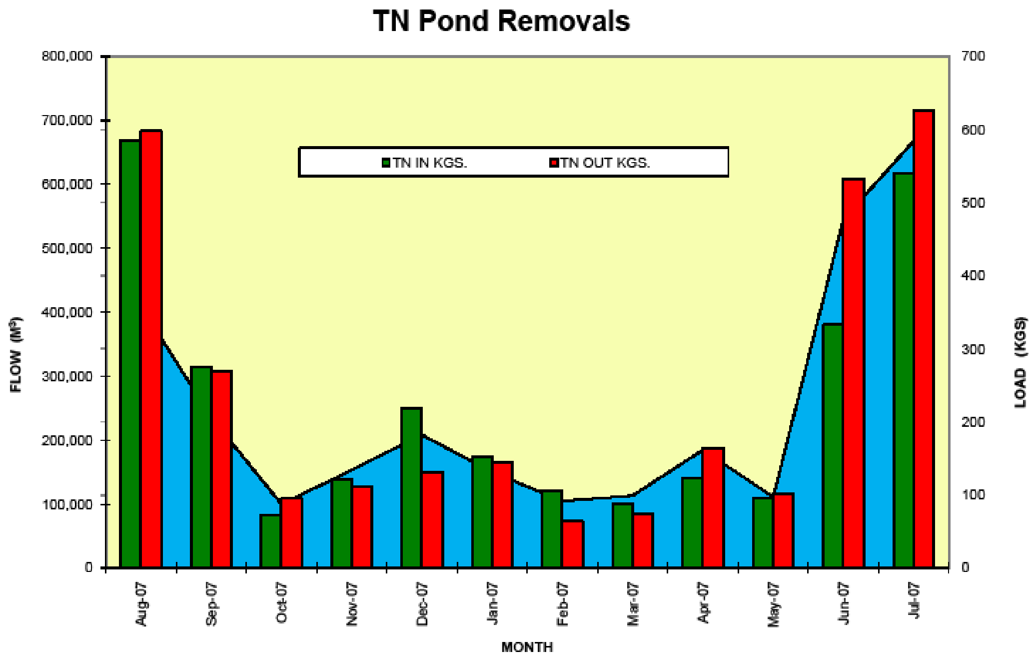

Figure 69: Total Nitrogen Loads

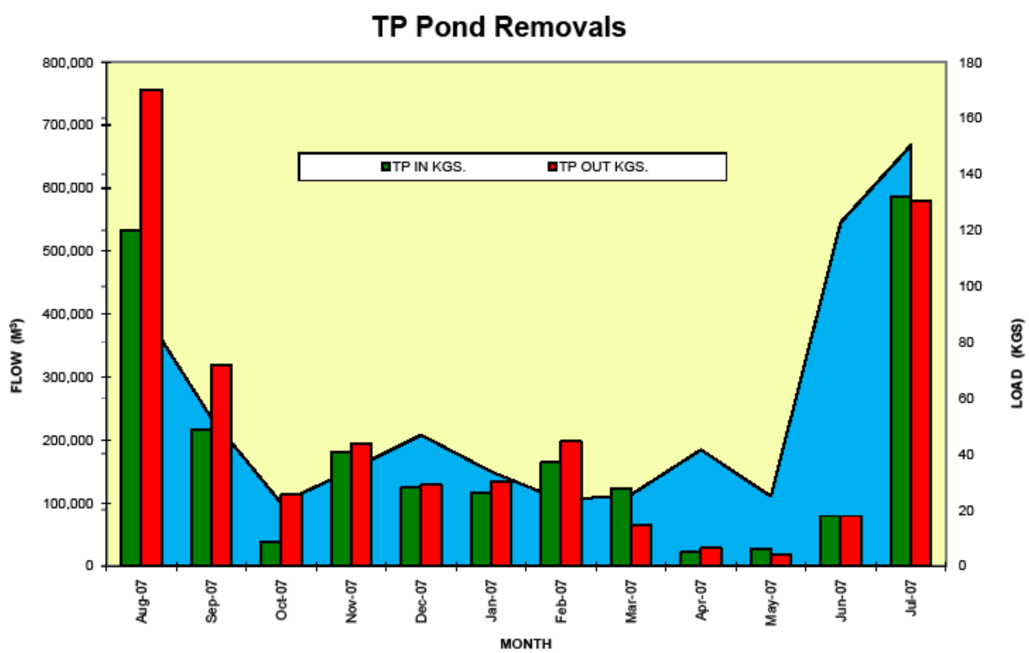

Figure 70: Total Phosphorus Loads

Conclusions

The laboratory results from the collected grab samples show slight variation of total nitrogen concentrations and moderate variation of total phosphorus concentrations throughout the year. The concentration levels of total phosphorus ranged from 0.02 mg/l to 0.48 mg/l, but not in relation to either season or flow volume variations.

A review of the data shows that the E_{emc} and E_{sol} methods yielded approximately the same results. This is mostly because the study period was short duration of only one year. It is interesting to note that the magnitude of total nitrogen loads coming into the pond were basically identical for either pollutant load analysis method. In contrast, the total phosphorus levels increased at the outlet of the pond, which suggests that the study pond was actually releasing significant amounts of total phosphorus into the downstream receiving waters.

The nutrient loads flowing into and out from the pond were evaluated for wet and dry cycles, but no significant variation of nutrient concentration levels nor removal efficiencies were found with respect to season. The only nutrient that showed any seasonal variation was nitrogen and it only showed slightly lower values towards the later part of spring and early summer.

Limitations

Resuspended pollutants from the pond floor were not accounted for in the analysis since sediment data was not collected during this study. Atmospheric contributions of pollutants to the pond were expected to be insignificant, but the data was not available to verify this assumption. The initial and final pollutant loads within the pond were not measured, so an assumption was made that it remained unchanged from year to year. Also, any contribution from waterfowl was

not accounted for even though there is a rather large roust of birds along the maintenance berm of the pond.

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