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Land Use and Water Quality Correlations in Miami-Dade, Florida

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FLORIDA INTERNATIONAL UNIVERSITY

Miami, FL

LAND USE AND WATER QUALITY CORRELATIONS
IN MIAMI-DADE, FLORIDA

A thesis submitted in partial fulfillment of the

requirements for the degree of

MASTER OF SCIENCE

in

ENVIRONMENTAL STUDIES

by

Audrey Rukhmin Becnel

2014

To: Interim Dean Michael R. Heithaus
College of Arts and Sciences

This thesis, written by Audrey Rukhmin Becnel, and entitled Land Use and Water Quality Correlations in Miami-Dade, Florida, having been approved in respect to style and intellectual content, is referred to you for judgment.

We have read this thesis and recommend that it be approved.

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Mahadev Bhat, Co-Major Professor

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Date of Defense: June 27, 2014

The thesis of Audrey Rukhmin Becnel is approved.

Interim Dean Michael R. Heithaus
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University Graduate School

Florida International University, 2014

DEDICATION

I dedicate this to my granny, the original Audrey Angelette Becnel. Her strength and devotion to family inspire me every day.

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I would like to thank the members of my committee for being a part of this project. Dr. Mahadev Bhat and Dr. Assefa Melesse, your direction and guidance have been invaluable. I am so grateful for your help through my proposal, research, and finally the writing process.

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I have been very lucky to grow up with Krishna Patel and Daniel Coward. Thank you for keeping me sane throughout my academic career. I hope that our group chat will continue to keep us close in the next stages of our lives. May the odds be ever in our favor.

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ABSTRACT OF THE THESIS
LAND USE AND WATER QUALITY CORRELATIONS
IN MIAMI-DADE, FLORIDA

by

Audrey Rukhmin Becnel

Florida International University, 2014

Miami, Florida

Professor Mahadev Bhat and Professor Assefa Melessa, Co-Major Professors

South Florida continues to become increasingly developed and urbanized. My exploratory study examines connections between land use and water quality. The main objectives of the project were to develop an understanding of how land use has affected water quality in Miami-Dade canals, and an economic optimization model to estimate the costs of best management practices necessary to improve water quality. Results indicate Miami-Dade County land use and water quality are correlated. Through statistical factor and cluster analysis, it is apparent that agricultural areas are associated with higher concentrations of nitrogen, while urban areas commonly have higher levels of phosphorous than agricultural areas.

The economic optimization model shows that urban areas can improve water quality by lowering fertilizer inputs. Agricultural areas can also implement methods to improve water quality although it may be more expensive than urban areas. It is important to keep solutions in mind when looking towards future water quality improvements in South Florida.

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Chapter 1

1. Introduction

Florida has one of the most managed hydraulic systems in the country. South Florida water bodies, canals and rivers are being altered to facilitate development. Surface water pollution is a persistent problem with the development of agricultural, industrial and urban areas (Graves et al., 2004). South Florida continues to become increasingly developed and urbanized.

Figure 1. South Florida hydrologic flow (www.evergladesplan.org)

Since the 1900's, South Florida has been dredged and drained to create navigable and economically productive environment (Briceno et al., 2011). By 1917, four major canals were completed: Miami, North New River, Hillsboro and West Palm Beach canals flowed from Lake Okeechobee to the coast (Carter, 2011). These canals run through extremely urbanized as well as agricultural areas. Intense land use development can put immense runoff pressures on surrounding water bodies (Osborne and Wiley, 1988). The

main nutrients found in runoff water in South Florida are nitrogen and phosphorus (Briceno et al., 2011).

Figure 2. Miami-Dade county land use map (FLUCCS, 2008)

Nutrients can come from many different sources, which may be either natural or anthropogenic (Harper, 1994). Phosphorous can be found in runoff from fertilizers for lawns and croplands, septic systems, drained wetlands, waste water treatment, soil and rocks (Graves et al., 2004). Once these levels are increased, it can accelerate plant growth and cause algae blooms (Arias, 2011). This is eutrophication. Nitrogen is abundant in the natural environments, occurring in the form of nitrite, nitrate, and ammonium (Graves et al., 2004). Nitrogen can also be introduced to the natural environment through sewage and chemical fertilizers (Dauer et al., 2000). Farmers generally apply nitrogen inputs to their crops in order to obtain higher yields (Basu, 2008). The runoff following heavy rains can carry chemicals applied to crops to nearby lakes, streams, and canals. Excess levels of nitrogen can also lead to eutrophication.

Turbidity measures how cloudy or murky the water is. High turbidity may not necessarily be harmful, but it can be an indication of the amount of suspended and total dissolved particles (TDS) (Briceno, 2011). An increase in turbidity can reflect human activities like construction, mining, and agriculture (Graves et al., 2004).

Miami-Dade agricultural land decreased by 26% from 2002 to 2007 (Census of Agriculture, 2007). Although there has been a steady decrease in acreage devoted to agriculture, the number of farms has increased and the total agricultural area remains is fairly significant. Miami-Dade is trending towards smaller, but more numerous, individual nurseries. The land cover transitions in South Florida may have considerable implications to water quality in local canals. Nitrogen and phosphorous represent the largest fertilizer cost to farmers, as well as the biggest threat to water quality (FDACS, 2005).

Nutrient concentrations in agricultural canals are the results of non-point source pollution from the offsite crop lands (FDACS, 2005) and other land use practices. The water quality improvements in the subject canals would require appropriate changes in the land management practices and fertilizer use. These changes would involve considerable economic costs to implement best management practices (BMPs). This begs a policy relevant question what are the overall costs of implementing changes in land use practices, and in turn, reducing the nutrient loading to study canals.

1.1 The Problem

South Florida has faced rapid land use changes in the past three decades. There has been extensive urban development along Biscayne Bay and intense agricultural development in South Miami-Dade (Brown, 2011). The main goal of the current research was to understand the association between land use changes in the highly developed areas of South Florida and quality of the surrounded water bodies. This information is useful for understanding the implications future development will have on South Florida natural resources.

1.2 Research Questions

There are many unknowns when anticipating the future of water quality in South Florida. Between the uncertainty of impending climate change to the unpredictability of development, it is important to identify the fundamental relationships between land use and nutrient loading. The present study will address the following questions:

- What is the relationship between land use types and water quality parameters (nitrogen, phosphorous, turbidity)?
- What are the most suitable and cost-effective best management practices for specific land use types (urban and agricultural)?
 - How much nutrient loading can potentially be reduced?
 - How much will it cost?

1.3 Objectives

The main objectives of the project were to develop an understanding of how land use has affected water quality in Miami-Dade canals. The first step was to quantify spatiotemporal land use changes for areas contributing to the selected canals. Once the local land use was determined, then the effect of land use changes and field management on water quality was estimated. Finally, the study developed an economic optimization model to estimate the costs of BMPs necessary to improve water quality. The project shows how land use has affected local water quality historically and looks for possible techniques to improve hydrologic health in the future.

1.4 Significance of Study

Healthy water resources are extremely important to human development and Miami-Dade County is not an exception. Water quality is not only a public health issue but is also the basis of recreation and tourism in South Florida. The focus of this study was to determine how water quality has historically been affected by land use

development. By looking conducting a basin wide study for each canal sampling site, a better understanding of land use and local water quality interactions can be obtained.

Previous literature had produced broad connections between land use and water quality, but generally have not addressed specific and cost effective solutions to improving local storm water runoff. There remain gaps in region-wide costs estimates of pollution reduction through land use management. A specific objective of the present study is to fill those gaps. With the help of an economic optimization model, the research develops a least cost combination of three separate BMPs for all land use categories of two selected watersheds in the study area, in order to improve water quality.

The results from the present study could be valuable for water quality management and policy formulation in that it can be used to predict how nutrient concentrations may fluctuate with future land use development. It also shows cost effective solutions to reducing local pollution based on different target reduction rates.

Chapter 2

2. Literature Review

2.1 Water Quality

There have been many studies looking at water quality in Biscayne Bay and hydrology of South Florida, but there is a gap when looking at trends over time in relation to land use. Briceno (2011) analyzed how Biscayne National Park has been affected by land use changes and population growth. Biscayne Bay is a very productive ecosystem and is extremely biodiverse. It houses 800 species of animals and is a major stopover for migrating shorebirds (Alleman et al, 1995). Biscayne National Park (BNP) was established by the U.S. Congress in 1982 to protect the 172,000 acres of Biscayne Bay. Briceno concluded that in the past, nutrients were effectively fixed through the vegetation and soil. Today, the canal waters are practically surface runoff from surrounding agricultural areas and therefore receive significant amounts of nutrients (Ongley, 1996).

Figure 3. Biscayne National Park Sampling Sites and Dendrogram (Briceno, 2011)

Biscayne Bay has varying quality and composition across the bay. As seen in Figure 3, the sampling sites in Biscayne Bay National Park were grouped by a dendrogram using factor and cluster analysis. There was shown a relationship between upstream land use type and sampling site water quality. The North Bay is mostly affected by urban development and has the highest phosphorous, chlorophyll-*a* and silica concentrations. The central bay has the highest inorganic nitrogen concentrations, lowest chlorophyll-*a* and turbidity. It is mostly impacted by agricultural activities in its watershed. South Biscayne Bay is surrounded by wetlands and is the least developed surrounding area (Briceno, 2011).

South Florida is very responsive to small levels of nutrients. Biscayne Bay is oligotrophic and limited by phosphorous. The watershed is a mosaic of agricultural, urban and wetland areas. A study in Biscayne National Park by Briceno, (2011) determined that concentrations of nitrogen were higher in agricultural areas, while urban areas seem to have higher total phosphorous

In another study looking at Biscayne Bay water quality, a landscape development intensity (LDI) index was developed to describe the implications on water quality samples (Carey, 2009). Water quality and land use were most related on the sub-basin level. Parameters like nitrate/nitrite-nitrogen and phosphorous loads were most indicative of watershed processes. The efficiency of pollutant control strategies were also measured to see which techniques were best fitted for zoning regulations in South Florida. Carey (2009) addressed the difficulties associated with analyzing changes water quality with specific land use. There is uncertainty associated with large errors in load estimates because of South Florida's unusually high levels of flow through canals (Carey, 2009).

While this can be less important when looking at South Florida specific information, it can be misleading when trying to make conclusions about water quality trends. When constructing water quality models to make policy and management decisions, it is imperative to keep these uncertainties in mind.

The relationship between land use and water quality can also be used to predict fluctuations in freshwater resources nationally. In a 2014 report, it was predicted the urban expansion will be the biggest threat to freshwater resources in the United States (Januchowski, 2014). An econometric model used four different potential land use development strategies in 2111 watersheds to predict how the quality and quantity of local water bodies would be affected. The business as usual strategy portrays 1990s land use trends. This resulted in only a few of the watersheds having 10% natural vegetative land cover (Januchowski, 2014). One land use strategy was urban containment; this restricted urban expansion to solely metropolitan counties. This is an example of a smart growth regulation and resulted in a drastic increase in biodiversity. This study found that the only way to adequately protect these resources is to establish alternative policy and market driven scenarios that can increase both water quality and biodiversity (Januchowski, 2014). While urban expansion looks like a common and pervasive threat, the implementation of conservation policies could alleviate some of the pressures on natural resources.

2.2 Land Use Impact

South Florida's history of hydrologic management has been influenced by a combination of development and remediation. The continual struggle between growth and conservation stresses land and water quality.

The Indian River Lagoon (IRL) and St. Lucie Estuary (SLE) are some of the most diverse ecosystems in the United States (Swain et al., 1995). The development of these watersheds has been dramatic and has had direct implications on the water quality. Excess nutrients, especially nitrogen and phosphorous, have caused algal blooms in this area that have disturbed the entire ecosystem and food web. Graves et al. (2004) characterized storm water quality of various coastal watershed land uses: citrus, pasture, urban, natural wetland, row crop, dairy, and golf courses. The major objective of this study was to analyze water quality corresponding to major land uses. The Graves et al. (2004) study found that runoff from most land use types had a low dissolved oxygen concentration and that nutrient concentrations were highly correlated to particular land use.

Figure 4. Study area and Pair-wise comparison chart (Graves et. al., 2004)

Each water quality parameter (e.g. phosphorous, nitrogen, inorganic nitrogen, organic nitrogen, ammonia nitrogen, and nitrate/nitrite) was directly compared between different land uses in a pair-wise fashion (Figure 4). The pair-wise comparison chart displays which land use has higher levels of each water quality parameter in the study area. For nitrogen, the concentration levels were relatively low except for crop runoff at 1.88 mg/L. Phosphorous levels were higher in urban and wetland areas compared to every other land use type. The study recommends designing treatment wetlands to decrease the damage to St. Lucie ecosystems. Reducing suspended sediment in storm water runoff may improve water quality in local environments (Graves et al, 2004).

Phosphorous nutrient loading has drastically increased since the establishment of the Everglades Agricultural Area (Scheidt, 2007). South Florida canals move water with increased phosphorous concentrations through the Everglades ecosystem. Phosphorous runoff from these agricultural areas has impaired the water quality in South Florida. The stormwater treatment areas (STAs) have attempted to mitigate this excessive nutrient flow (National Research Council, 2010). This practice has not been sufficient in treating existing flows and phosphorous loading.

In previous water quality and land use studies, principal component analysis (PCA) and factor analysis have been used to show correlations (Carey, 2009). These analysis methods were able to identify which factors were the most responsible for surrounding water quality. Since urban areas are concentrated in north Miami-Dade County, it is not surprising that phosphorous concentrations are higher in the northern bay (Carey, 2009).

A similar land use impact analysis was conducted in the Densu River basin in Ghana (Attua et al., 2014). This area is one of the most reliable freshwater sources and greatly stressed by surrounding land use activities. Spatial differences in water quality were observed reflecting variability in land use, geology, lithology, and soil properties (Attua et. al., 2014). It also found that by using land cover information, water quality parameters like pH, temperature, turbidity, dissolved oxygen (DO), phosphate-phosphorous, and nitrate-nitrogen could be estimated. Urban expansion directly correlated to nitrate concentrations, while agricultural cover influenced variable such as phosphorous and nitrogen. While Ghana often lacks adequate resources to monitor human impacts on water resources, it is important to incorporate studies like this towards management and sustainable development decisions (Water Resources Commission, WRC, 2008).

2.3 Estimating Nutrient Loading, Reduction, and Costs

Tegtmeier and Duffy (2004) estimated the external costs of agricultural production in the United States. They identified specific methods to place a monetary cost of water pollution. However, they recognize the lack of geographic region specificity in their external cost estimates. Agricultural production has an estimated environmental cost of \$5.7-16.9 billion per year in the United States (Tegtmeier and Duffy, 2004). The environmental cost comes from negative impacts to natural resources like soil, water, air, wildlife and human health. Further, not much research has been done on estimating the actual costs of achieving greater water quality (Ongley, 1996). This

research conducts a site specific evaluation to develop costs of water quality improvement.

Several studies have investigated the region-wide costs of reducing agricultural and other land use related nutrient runoffs. Claassen et al. 2001 nutrient loading reduction was quantified in terms of economic effects as well as impacts on the environment. This study focused on the “Dead Zone” in the Gulf of Mexico by reducing nutrient inputs from the Mississippi River Basin.

Claassen et al. 2001 used an empirical model to directly compare two types of nutrient reduction techniques: changing farming practices to use less nitrogen and creating riparian buffers to treat the runoff from farms before it enters rivers and streams. This Mississippi River Basin study found that it is more cost effective to initially reduce fertilizer. Although, at increased target nutrient reductions more expensive techniques were required (Figure 5).

Figure 5. Summary of Nitrogen-reduction strategies (Claassen et al., 2001)

Estimates for nutrient loading were developed by the FDEP by taking averages of nutrient loadings from a variety of areas around Florida (Harper and Baker, 2007).

Additionally FDEP in conjunction with SFWMD identified expected runoff nutrient concentrations, potential reduction factors, and expected costs for a variety of BMPS by averaging measures taken across the Caloosahatchee and St. Lucie River watersheds (SWET, 2008).

Best management practice effectiveness was measured by the review of modeling studies and numerous projects across Florida (SWET, 2003). The study incorporated crop responses to management techniques concerning water and fertility to measure the ecological response. The SWET study resulted in a reduction potential percent for implementation of different BMPs.

The costs associated with the listed BMPs took into account many factors including money saved on fertilizer, equipment and construction, operations, maintenance, energy/fuel, crop revenue loss, crop displacement, and additional land purchases (SWET, 2008). In agricultural areas, BMPs can be incorporated into existing land. While urban areas need to purchase additional land to incorporate storm water treatment areas (SWET, 2008).

Chapter 3

3. Methodology

I used a combination of geographical information system (GIS) tools and statistical analysis to determine relationships between land use and water quality. The GIS tools were used to quantify land use data and depict the spatial relationship between land development and water quality sampling sites. Statistical tools such as factor analysis and principal component cluster analysis were used to identify historical correlations between nutrient parameters and land use types. The least cost abatement model was developed to determine the most cost effective solutions to increasing future water quality in South Florida.

3.1 Land Use Dynamics

To identify the spatiotemporal changes in land use in Miami-Dade and initial data collection was made from the South Florida Water Management District for 1988, 1995, 1999, 2004, and 2008. The datasets included areas under different land uses for Miami-Dade watersheds, and water quality data for SFWMD canals at various sampling locations.

Once the data were collected, individual watersheds were demarcated for each canal. The intersect tool in ArcGIS 10.2 was used to identify which watersheds are directly surrounded each canal (Figure 6). Chosen watersheds were then merged to create consolidated canal basins (Figure 7).

Figure 6. Watersheds of Miami-Dade

Figure 7. Little River, Miami River, Princeton Canal, and Mowry Canal Watersheds

The next step identifies the percent land use composition for each of the canal basins. The SFWMD created the FLUCCS (Florida Land Use, Cover and Forms Classification System) to categorize land cover types (FLUCCS, 1985). The FLUCCS code partitions land use into eight different categories: Urban and Built-up land, Agricultural, Upland non-forested, Upland forested, Water, Wetlands, Barren lands, and Transportation/Communication/ Utilities. Table 1 displays how FLUCCS organizes land use types into eight main categories and further classifies more specific types within that category. Through use of the intersect tool, the land use data were extracted for each of the basins. Each basin was then summarized to show the percent composition by land use type.

Table 1. Florida Land Use, Cover and Forms Classification System Code

1000	Urban and Built-up Land	<ul style="list-style-type: none"> ■ 2000 Agricultural <ul style="list-style-type: none"> ■ 2100 Cropland and Pastureland <ul style="list-style-type: none"> ■ 2110 Improved Pasture ■ 2120 Unimproved Pasture ■ 2130 Woodland Pasture ■ 2140 Row Crops ■ 2150 Field Crops <ul style="list-style-type: none"> ■ 2156 Sugar Cane ■ 2200 Tree Crops <ul style="list-style-type: none"> ■ 2210 Citrus Groves ■ 2220 Fruit Orchards ■ 2230 Other Groves ■ 2300 Feeding Operations <ul style="list-style-type: none"> ■ 2310 Cattle Feeding Operations ■ 2320 Poultry Feeding Operations
2000	Agricultural	
3000	Upland Non-Forested	
4000	Upland Forested	
5000	Water	
6000	Wetlands	
7000	Barren Lands	
8000	Transportation, Communication, and Utilities	

3.2 Water Quality

Sampling sites were selected on the basis of increasing the diversity of land use types associated with the analysis. The eight different land use types will give a diverse representation of the relationship between land use and nutrients in surface water runoff. At least two sampling sites were chosen for each canal basin. The SFWMD samples canal water monthly throughout the year. Some sampling sites are sampled for a specific purpose at a particular point in time and do not have the complete data. Sampling sites were chosen on the basis of completeness of historical sampling by SFWMD. The selected sampling sites were:

Table 2. South Florida Water Management District (SFWMD) Sampling Sites

MDC Canals	Sampling Sites	Latitude (ddmmss.sss)	Longitude (ddmmss.sss)
Little River	LR01	255042.711	801024.083
	LR10	255211.445	802019.043
Miami River	MR01	254615.294	801112.978
	MR03	254643.952	801225.16
	MR05	254728.26	801410.786
	MR06	254745.202	801438.664
	MR07	254822.634	801529.492
Princeton Canal	PR01	253110.518	801955.2
	PR03	253110.254	802149.975
	PR08	253514.854	803038.453
Mowry Canal	MW01	252814.687	802026.531
	MW04	252816.027	802246.258
	MW13	253102.85	803233.249

Once the water quality data were compiled, they were subjected to simple data quality assessment and data quality control (Briceno, 2011). Water quality data includes high variability as a result of environmental changes and seasonality. It is therefore important to identify data gaps and outliers. Since the present study was based on historical data spanning more than twenty years, it is important to only consider long term trends rather than spikes in seasonality over a given year.

Statistical analysis were used to describe the link between land use and water quality. The following continuous factors were used in the analysis: Date, Nitrate levels (mg/L), Phosphorous levels (mg/L), Turbidity levels (NTU), Percent Urban, Percent Agricultural, Percent Upland non-forested, Percent Upland forested, Percent Water, Percent Wetlands, Percent Barren lands, Percent Transportation/Communication/Utilities. All statistical analyses were performed using Minitab 16 statistical software.

3.3 Statistical Analysis

Factor analysis was used to reduce the complexity of the data since there were a large number of variables (Bhat et al., 2014). This tool reduces the parameters to a new set of principal component scores. This technique shows how many variables are important to describing the variance in the data (Bhat et al., 2014). Once the principal component scores were collected, descriptive statistics were performed for each sampling site (Briceno, 2011). The mean and standard deviation for each site's scores were then used as the input data to cluster the sampling sites.

The cluster analysis uses the derivative principal components to group sampling sites on the basis of their similarity. Sampling sites within a closer proximity are more related than those in distant groups. These results are illustrated through a dendrogram, or tree diagram (Dodge, 2008). A dendrogram lists variables according to which are most similarly related. This visual interpretation of the data was run twice: once with 2-clusters, and once with 4-clusters. This shows how sampling sites are further categorized on the basis of different levels of similarity.

3.4 Economic Model of Basin-wide Pollution Reduction

The purpose of modeling was to estimate the basin wide costs of reducing offsite nutrient loading and improving water quality. The standard approach would be to require all land owners to implement certain BMP's to achieve a desired water quality improvement. However, the marginal costs of BMPs can vary across land use types. The marginal costs of different BMP's vary as a function of land use. The conventional

economic wisdom dictates that applying a uniform pollution reduction measures across all land use types when there exists a degree of variation in costs is not economical. Therefore, a least-cost economic model was developed to optimally assign a BMP to each land use category taking into consideration the unit costs and pollution reduction potential of each BMP.

3.4.1 Description of the Least Cost Abatement Model

A simple accounting framework was developed to clearly identify the total area of each land use type within the watershed basin of each sample site, total nutrient loadings of each study watershed, and list of potential BMPs and possible reductions in loadings. In South Miami-Dade, BMPs implementation divided between non-irrigated and irrigated cropland (FDACS, 2005). For instance, irrigated croplands can affect local canals from nutrient enriched surface water runoff. It is assumed that a social planner would like to minimize the basin wide cost of implementing various BMPs on different land use to achieve certain loading reduction target. Formally, the planner minimizes the following objective function:

$$\text{Min } Z = \sum_{i=1}^n \sum_{j=1}^3 A_i C_{ij} I_{ij} \quad (1)$$

Subject to:

$$\sum_{i=1}^n \sum_{j=1}^3 A_i l_i^{nt} \left(1 - \frac{r_{ij}^{nt}}{100} \right) I_{ij} \leq N \quad (2)$$

$$\sum_{i=1}^n \sum_{j=1}^3 A_i l_i^{po} \left(1 - \frac{r_{ij}^{po}}{100} \right) I_{ij} \leq P \quad (3)$$

$$\sum_{j=1}^3 I_{ij} = 1 \text{ for all } i = 1, \dots, n \quad (4)$$

$$I_{ij} = 0 \text{ or } 1 \quad (5)$$

Model variables and parameters are explained in Table 3. The objective function (equation 1) is the basin wide costs of implementing a set of BMPs (I_{ij}). The watershed land use acreage represented in the objective function was classified using FLUCCS to two significant figures. The FLUCCS code allowed for more specific loading and cost information to be implemented. For example, loading was categorized not simply as “Agriculture” but further split up between row crops, pasture, citrus, etc. (seen in Table 1).

The model was run at incremental target percent loading reductions to determine the maximum amount that can be reduced for each watershed. Constraint equations (2) and (3) require that the actual total nitrogen and phosphorous loadings (left-hand sides of the equations), respectively, from adapting different BMPs (I_{ij}) be less than or equal to a set loading target, N and P , respectively. The above constraint equations combine the area, land use type, percent reduction, and implementation variables to determine the actual loading after the BMP implementation. This valuation also depicts which BMP is more appropriate on each land use type based on the target nutrient percent reduction.

The constraint in equation (4) forces that only one BMP (j) be chosen for a given land use type (i). Equation (5) forces the BMP variables to be integers and have a value of either 0 or 1.

Table 3. Linear Programming Model Parameters and Variables

i	Land Use type (1, 2, ...n)
j	Type of BMP (1,2,3)
A_i	Area of specific land use type (ha)
C_{ij}	Cost of implementation of BMP j on land use i (\$/ha)
I_{ij}	BMP implementation of type j on land use type j (a binary variable with a value of 0 or 1)
l_i^{nt} and l_i^{po}	Nitrogen and phosphorous loadings from land use type i (kg/ha)
r_{ij}^{nt} and r_{ij}^{po}	Potential nitrogen and phosphorous reduction percentage after specific BMP j is implemented on a particular land use type i
N	Nitrogen target loading level (kg/ha)
P	Phosphorous target loading level (kg/ha)

BMPs were divided into three categories:

1. Owner Implemented Fertilizer Reduction: This practice would entail simply reducing the amount of fertilizer which can decrease the amount of nutrients in storm water runoff. This is generally cost effective because owners can spend less on inputs. This may also reduce profits because areas may be less productive due to less fertility inputs. This method has a slow response rate compared to other techniques.
2. Cost share Retention Ponds: More expensive practices that can be funded through an incentive based program. This is based on water management and creating retention ponds near developed areas. This is the most expensive scenario

involved in the model. A dry retention pond is made to temporarily collect storm water (Harper et al., 2007). The retention pond method treats the storm water runoff through physical, chemical, and biological methods. This natural filtering system catches and filters runoff before it can discharge directly into surrounding water bodies. Nutrient concentrations in the groundwater beneath retention basins are generally lower than the runoff that enters the retention system (Harper et al., 2003).

Figure 8. BMP 2 Dry Retention Diagram (Harper and Baker, 2007)

3. Alternative BMP: Additional practices like chemical treatment. Chemical treatment areas can be established at the edge of major urban and agricultural land use types (SWET, 2008). The alternative BMP strategy is a very site specific method to reduce the amount of nutrients.

3.4.2 Input Data

Little River, Miami River, Princeton Canal and Mowry Canal were chosen for this study to represent a variety and juxtaposition of land use types in Miami-Dade. These are not only some of the longest waterways in Miami-Dade, but they are also surrounded by a mosaic of different land use types. Little River and Miami River are found in north Miami-Dade and are primarily surrounded by urban and built-up land use types. Princeton Canal and Mowry Canal are in south Miami-Dade and mostly influenced by agricultural areas. The sampling sites found on these waterways will provide for a direct comparison of water quality parameters.

The parameters for objective and constraint functions were derived through literature review and GIS land use analysis. The nutrient loading for each land use and BMP reduction percentage was determined by taking the average of the Caloosahatchee River and St. Lucie watershed from the SWET 2008 study. See Table 4 for nutrient loading parameters used in this study. Similarly, the total area under each land use category was computed using the GIS summary tool (see Table 4 for the land use area). The cost parameters were calculated from the information available from SWET (2008) study, which provides initial and yearly costs of alternative BMPs for different land use types. I added the yearly cost to 10% of the initial cost to arrive at the annual total unit cost (\$/ha) for each BMP (see Tables 5-7 and 9-11 for cost parameters). The initial cost is under the assumption that the BMP will be effective and paid off over the course of 10 years.

Table 4. Miami River Land Use Area (ha) and Loading (kg/ha) (SWET, 2008)

Miami River Land Use Area (ha) and Loading (kg/ha)				
Canal Watershed	LUCODE	Area (ha)	Nutrient Load	
			N (kg/ha)	P (kg/ha)
Miami River	1180	13	1.80	0.21
Miami River	1210	8017	2.32	0.37
Miami River	1290	5	2.32	0.37
Miami River	1310	319	2.60	0.58
Miami River	1320	87	2.60	0.58
Miami River	1330	1632	2.60	0.58
Miami River	1340	292	2.60	0.58
Miami River	1390	31	2.60	0.58
Miami River	1400	4270	1.32	0.20
Miami River	1411	196	1.32	0.20
Miami River	1423	24	1.32	0.20
Miami River	1480	97	2.69	0.39
Miami River	1490	175	2.69	0.39
Miami River	1550	2045	1.35	0.29
Miami River	1560	59	1.35	0.29
Miami River	1630	253	1.35	0.29
Miami River	1650	77	1.35	0.29
Miami River	1660	709	1.35	0.29
Miami River	1700	321	1.35	0.29
Miami River	1710	316	1.35	0.29
Miami River	1730	7	1.35	0.29
Miami River	1820	456	1.35	0.29
Miami River	1830	112	1.35	0.29
Miami River	1840	2	1.35	0.29
Miami River	1850	198	1.35	0.29
Miami River	1870	14	1.35	0.29
Miami River	1900	342	1.29	0.06
Miami River	1920	3	1.29	0.06
Miami River	2110	178	3.89	0.69
Miami River	2140	26	2.97	0.66
Miami River	2410	244	3.13	0.48
Miami River	2430	13	3.13	0.48
Miami River	2610	2	3.13	0.48

Table 5. Miami River BMP 1 (SWET2008)

Miami River BMP 1 (SWET2008)							
	N Reduction%	P Reduction%	N Initial Cost \$/ac/yr	N Cost \$/ac/yr	P Initial Cost \$/ac/yr	P Cost \$/ac/yr	Average Cost (\$/ha)
1110	15	5	800	5.30	800	0.00	6.54
1180	15	5	800	5.30	800	0.00	6.54
1210	25	5	800	5.30	800	0.00	6.54
1290	25	5	800	5.30	800	0.00	6.54
1310	30	5	800	5.30	800	0.00	6.54
1320	30	5	800	5.30	800	0.00	6.54
1330	30	5	800	5.30	800	0.00	6.54
1340	30	5	800	5.30	800	0.00	6.54
1390	30	5	800	5.30	800	0.00	6.54
1400	25	5	800	5.30	800	0.00	6.54
1411	25	5	800	5.30	800	0.00	6.54
1423	25	5	800	5.30	800	0.00	6.54
1480	25	5	800	5.30	800	0.00	6.54
1490	25	5	800	5.30	800	0.00	6.54
1550	25	5	800	5.30	800	0.00	6.54
1560	25	5	800	5.30	800	0.00	6.54
1630	25	5	800	5.30	800	0.00	6.54
1650	25	5	800	5.30	800	0.00	6.54
1660	25	5	800	5.30	800	0.00	6.54
1700	25	5	800	5.30	800	0.00	6.54
1710	25	5	800	5.30	800	0.00	6.54
1730	25	5	800	5.30	800	0.00	6.54
1820	25	5	800	5.30	800	0.00	6.54
1830	25	5	800	5.30	800	0.00	6.54
1840	25	5	800	5.30	800	0.00	6.54
1850	25	5	800	5.30	800	0.00	6.54
1870	25	5	800	5.30	800	0.00	6.54
1900	25	5	800	5.30	800	0.00	6.54
1920	25	5	800	5.30	800	0.00	6.54
2110	15	11	4.4	2.20	3.85	4.00	7.65
2140	30	30	20.9	18.00	20.9	3.52	26.58
2410	25	32	47	11.00	20.9	4.00	18.53
2430	25	32	47	11.00	20.9	4.00	18.53
2610	25	32	47	11.00	20.9	4.00	18.53

Table 6. Miami River BMP 2 (SWET2008)

Miami River BMP 2 - Cost Share (SWET2008)							
	N Reduction %	P Reduction %	N Initial Cost (\$/ac/yr)	N Cost (\$/ac/yr)	P Initial Cost (\$/ac/yr)	P Cost (\$/ac/yr)	Average Cost (\$/ha)
1110	20	80	800	2560	800	2560	6523.35
1180	20	80	800	2560	800	2560	7314.06
1210	20	80	800	2560	800	2560	7314.06
1290	20	80	800	2560	800	2560	7314.06
1310	20	80	800	2560	800	2560	7314.06
1320	20	80	800	2560	800	2560	7314.06
1330	20	80	800	2560	800	2560	7314.06
1340	20	80	800	2560	800	2560	7314.06
1390	20	80	800	2560	800	2560	7314.06
1400	20	80	800	2560	800	2560	7314.06
1411	20	80	800	2560	800	2560	7314.06
1423	20	80	800	2560	800	2560	7314.06
1480	20	80	800	2560	800	2560	7314.06
1490	20	80	800	2560	800	2560	7314.06
1550	20	80	800	2560	800	2560	7314.06
1560	20	80	800	2560	800	2560	7314.06
1630	20	80	800	2560	800	2560	7314.06
1650	20	80	800	2560	800	2560	7314.06
1660	20	80	800	2560	800	2560	7314.06
1700	20	80	800	2560	800	2560	7314.06
1710	20	80	800	2560	800	2560	7314.06
1730	20	80	800	2560	800	2560	7314.06
1820	20	80	800	2560	800	2560	7314.06
1830	20	80	800	2560	800	2560	7314.06
1840	20	80	800	2560	800	2560	7314.06
1850	20	80	800	2560	800	2560	7314.06
1870	20	80	800	2560	800	2560	7314.06
1900	20	80	800	2560	800	2560	7314.06
1920	20	80	800	2560	800	2560	7314.06
2110	15	19	4.4	14.08	3.85	12	42.41
2140	30	30	20.9	66.88	20.9	66.88	216.90
2410	5	35	47	150.5	20.9	67	352.61
2430	5	35	47	150.5	20.9	67	352.61
2610	5	35	47	150.5	20.9	67	352.61

Table 7. Miami River BMP 3 (SWET2008)

Miami River BMP 3 (SWET2008)							
	N Reducti on%	P Reducti on%	N Initial Cost (\$/ac/yr)	N Annual Cost (\$/ac/yr)	P Initial Cost (\$/ac/yr)	P Annual Cost (\$/ac/yr)	Average Cost (\$/ha)
1110	15	70	320	1024	320	1024	3320.98
1180	15	70	320	1024	320	1024	3320.98
1210	15	70	320	1024	320	1024	3320.98
1290	15	70	320	1024	320	1024	3320.98
1310	15	70	320	1024	320	1024	3320.98
1320	15	70	320	1024	320	1024	3320.98
1330	15	70	320	1024	320	1024	3320.98
1340	15	70	320	1024	320	1024	3320.98
1390	15	70	320	1024	320	1024	3320.98
1400	15	70	320	1024	320	1024	3320.98
1411	15	70	320	1024	320	1024	3320.98
1423	15	70	320	1024	320	1024	3320.98
1480	15	70	320	1024	320	1024	3320.98
1490	15	70	320	1024	320	1024	3320.98
1550	15	70	320	1024	320	1024	3320.98
1560	15	70	320	1024	320	1024	3320.98
1630	15	70	320	1024	320	1024	3320.98
1650	15	70	320	1024	320	1024	3320.98
1660	15	70	320	1024	320	1024	3320.98
1700	15	70	320	1024	320	1024	3320.98
1710	15	70	320	1024	320	1024	3320.98
1730	15	70	320	1024	320	1024	3320.98
1820	15	70	320	1024	320	1024	3320.98
1830	15	70	320	1024	320	1024	3320.98
1840	15	70	320	1024	320	1024	3320.98
1850	15	70	320	1024	320	1024	3320.98
1870	15	70	320	1024	320	1024	3320.98
1900	15	70	320	1024	320	1024	3320.98
1920	15	70	320	1024	320	1024	3320.98
2110	50	49	22	70.4	11	35	170.99
2140	50	50	44	140.8	44	140.8	456.63
2410	50	50	55	176	44	141	513.96
2430	50	50	55	176	44	141	513.96
2610	50	50	55	176	44	141	513.96

Table 8. Princeton Canal Land Use Area (ha) and Loading (kg/ha)

Princeton Canal Land Use Area (ha) and Loading (kg/ha)				
Canal Watershed	LUCODE	Area (ha)	Load	
			N (kg/ha)	P (kg/ha)
Princeton Canal	1180	1815	1.80	0.21
Princeton Canal	1210	1561	2.32	0.37
Princeton Canal	1290	221	2.32	0.37
Princeton Canal	1310	24	2.60	0.58
Princeton Canal	1320	115	2.60	0.58
Princeton Canal	1330	152	2.60	0.58
Princeton Canal	1390	31	2.60	0.58
Princeton Canal	1400	160	1.32	0.20
Princeton Canal	1490	6	2.69	0.39
Princeton Canal	1550	2	1.35	0.29
Princeton Canal	1650	23	1.35	0.29
Princeton Canal	1700	13	1.35	0.29
Princeton Canal	1710	25	1.35	0.29
Princeton Canal	1850	42	1.35	0.29
Princeton Canal	1900	155	1.29	0.06
Princeton Canal	1920	114	1.29	0.06
Princeton Canal	2110	29	3.89	0.69
Princeton Canal	2120	2	3.89	0.69
Princeton Canal	2140	4142	2.97	0.66
Princeton Canal	2150	879	2.97	0.66
Princeton Canal	2160	9	2.97	0.66
Princeton Canal	2210	736	2.51	0.21
Princeton Canal	2230	1537	2.51	0.21
Princeton Canal	2240	38	2.51	0.21
Princeton Canal	2410	1877	3.13	0.48
Princeton Canal	2420	94	3.13	0.48
Princeton Canal	2430	4432	3.13	0.48
Princeton Canal	2500	4	3.13	0.48
Princeton Canal	2510	40	3.13	0.48
Princeton Canal	2610	1749	3.13	0.48

Table 9. Princeton Canal BMP 1 (SWET2008)

Princeton Canal BMP 1 (SWET2008)							
	N Reducti on%	P Reduction%	N Initial Cost (\$/ac/yr)	N Cost (\$/ac/yr)	P Initial Cost (\$/ac/yr)	P Cost (\$/ac/yr)	Average Cost (\$/ha)
1110	15	5	800	0	800	0	0
1180	15	5	800	0	800	0	0
1210	25	5	800	0	800	0	0
1290	25	5	800	0	800	0	0
1310	30	5	800	0	800	0	0
1320	30	5	800	0	800	0	0
1330	30	5	800	0	800	0	0
1390	30	5	800	0	800	0	0
1400	25	5	800	0	800	0	0
1490	25	5	800	0	800	0	0
1550	25	5	800	0	800	0	0
1650	25	5	800	0	800	0	0
1700	25	5	800	0	800	0	0
1710	25	5	800	0	800	0	0
1850	25	5	800	0	800	0	0
1900	25	5	800	0	800	0	0
1920	25	5	800	0	800	0	0
2110	15	11	4.4	2.2	3.85	4	7.65
2120	9	7	2.2	1.2	1.1	1	2.71
2140	30	30	20.9	18	20.9	3.52	26.58
2150	30	30	20.9	18	20.9	3.52	26.58
2160	30	30	20.9	18	20.9	3.52	26.58
2210	10	12	20.9	6.4	7.7	0	7.90
2230	10	12	20.9	6.4	7.7	0	7.90
2240	10	12	20.9	6.4	7.7	0	7.90
2410	25	32	47	11	20.9	4	18.53
2420	25	32	47	11	20.9	4	18.53
2430	25	32	47	11	20.9	4	18.53
2500	25	32	47	11	20.9	4	18.53
2510	30	20	3.85	4	3.85	4	9.88
2610	25	32	47	11	20.9	4	18.53

Table 10. Princeton Canal BMP 2 (SWET2008)

Princeton Canal BMP 2 (SWET2008)							
	N Reduction%	P Reduction%	N Initial Cost (\$/ac/yr)	N Cost (\$/ac/yr)	P Initial Cost (\$/ac/yr)	P Cost (\$/ac/yr)	Average Cost (\$/ha)
1110	20	80	800	2560	800	2560	8302.45
1180	20	80	800	2560	800	2560	8302.45
1210	20	80	800	2560	800	2560	8302.45
1290	20	80	800	2560	800	2560	8302.45
1310	20	80	800	2560	800	2560	8302.45
1320	20	80	800	2560	800	2560	8302.45
1330	20	80	800	2560	800	2560	8302.45
1390	20	80	800	2560	800	2560	8302.45
1400	20	80	800	2560	800	2560	8302.45
1490	20	80	800	2560	800	2560	8302.45
1550	20	80	800	2560	800	2560	8302.45
1650	20	80	800	2560	800	2560	8302.45
1700	20	80	800	2560	800	2560	8302.45
1710	20	80	800	2560	800	2560	8302.45
1850	20	80	800	2560	800	2560	8302.45
1900	20	80	800	2560	800	2560	8302.45
1920	20	80	800	2560	800	2560	8302.45
2110	15	19	4.4	14.08	3.85	12	42.41
2120	7	13	2.2	7.04	1.1	4	17.72
2140	30	30	20.9	66.88	20.9	66.88	216.90
2150	30	30	20.9	66.88	20.9	66.88	216.90
2160	30	30	20.9	66.88	20.9	66.88	216.90
2210	30	5	20.9	66.88	7.7	24.64	148.41
2230	30	5	20.9	66.88	7.7	24.64	148.41
2240	30	5	20.9	66.88	7.7	24.64	148.41
2410	5	35	47	150.5	20.9	67	352.61
2420	5	35	47	150.5	20.9	67	352.61
2430	5	35	47	150.5	20.9	67	352.61
2500	5	35	47	150.5	20.9	67	352.61
2510	22	22	3.85	12	3.85	12	39.16
2610	5	35	47	150.5	20.9	67	352.61

Table 11. Princeton Canal BMP 3 (SWET2008)

Princeton Canal BMP 3 (SWET2008)							
	N Reduction%	P Reduction%	N Initial Cost (\$/ac/yr)	N Annual Cost (\$/ac/yr)	P Initial Cost (\$/ac/yr)	P Annual Cost (\$/ac/yr)	Average Cost (\$/ha)
1110	15	70	320	1024	320	1024	3320.97
1180	15	70	320	1024	320	1024	3320.97
1210	15	70	320	1024	320	1024	3320.97
1290	15	70	320	1024	320	1024	3320.97
1310	15	70	320	1024	320	1024	3320.97
1320	15	70	320	1024	320	1024	3320.97
1330	15	70	320	1024	320	1024	3320.97
1390	15	70	320	1024	320	1024	3320.97
1400	15	70	320	1024	320	1024	3320.97
1490	15	70	320	1024	320	1024	3320.97
1550	15	70	320	1024	320	1024	3320.97
1650	15	70	320	1024	320	1024	3320.97
1700	15	70	320	1024	320	1024	3320.97
1710	15	70	320	1024	320	1024	3320.97
1850	15	70	320	1024	320	1024	3320.97
1900	15	70	320	1024	320	1024	3320.97
1920	15	70	320	1024	320	1024	3320.97
2110	50	49	22	70.4	11	35	170.99
2120	25	44	11	35.2	5.5	18	86.11
2140	50	50	44	140.8	44	140.8	456.63
2150	50	50	44	140.8	44	140.8	456.63

2160	50	50	44	140.8	44	140.8	456.63
2210	50	52	22	70.4	24.2	77	239.18
2230	50	52	22	70.4	24.2	77	239.18
2240	50	52	22	70.4	24.2	77	239.18
2410	50	50	55	176	44	141	513.96
2420	50	50	55	176	44	141	513.96
2430	50	50	55	176	44	141	513.96
2500	50	50	55	176	44	141	513.96
2510	30	49	11	35	11	35	113.66
2610	50	50	55	176	44	141	513.96

To determine the appropriate nitrogen and phosphorous reduction targets, it was necessary to first compute the maximum nutrient loadings (N^{max} and P^{max}) if no model BMPs were implemented. These levels were calculated by taking the sum of total area under each land use multiplied by the respective nutrient loadings without a BMP.

Formally,

(6)

$$\sum_{i=1}^n A_i l_i^{nt} = N^{max}$$

$$\sum_{i=1}^n A_i l_i^{po} = P^{max} \quad (7)$$

The model was run by setting the nitrogen and phosphorous reduction targets N and P in equation (2) and (3) at different percent levels lower than N^{max} and P^{max} .

The optimization model in equations (1) to (5) is a standard linear programming (LP) model. However, the main decision variables I_{ij} are binary integer variables (0 or 1) in that the model will select or reject a particular BMP (j) for a given land use type (i). Therefore, the model becomes a Binary Integer Programming (BIP) Model. I used the Excel Solver package to solve the above BIP model.

3.4.3 Output Data

First, the baseline model was run by setting the nitrogen and phosphorus reduction targets at 95% of the current nutrient loading level with no BMP adaption. The optimization routine assigns the appropriate BMP to each land use type and estimates the basin wide total costs of adapting the BMPs and the actual nutrient loading associated with those BMPs. Then the model was re-run by setting nitrogen and phosphorous levels at different reduction targets using increments of 5% (i.e., up to 45% of the current loading levels for Princeton Canal and 25% of the current loadings for Miami River Canal). These multiple optimization runs allow us to establish a relationship between the total costs of BMP adaption at different levels of nutrient reduction targets.

Chapter 4

4. Results and Discussion

Through GIS and statistical analysis, the relationship between land use and water quality was determined. The following sections describe the land use composition of Miami-Dade over time and what nutrients are associated with each area. The binary linear programming model establishes the most appropriate BMP combinations at incremental percent reduction rates.

4.1 Land Use

The study watersheds show a great juxtaposition to have a direct contrast between land use types (Figures 10, 11, and 12). This is also the nature of Miami-Dade county land use. Areas are heavily developed and there is not a lot of diversity. Little River watershed is primarily urban and is built-up with urban infrastructure (roads, buildings and other physical structures), at 90 percent (Figures 13 and 14). Miami River watershed is also primarily urban and built up at 91 percent (Figures 15 and 16). Princeton Canal watershed is primarily agricultural with 82% of its land covered with crops (Figures 17 and 18). Mowry Canal watershed is also primarily agricultural with 72% land area under crops (Figures 19 and 20).

Figure 9. Miami-Dade County Land Use Percent Composition (2008)

Table 12. Canal Watershed Land Use (FLUCCS) Area (ha) (2008)

Canal Basin Land Use (FLUCCS) Area (ha)									
Water Canal	Urban and Built-Up	Agriculture	Upland Non-Forested	Upland Forested	Water	Wetlands	Barren Lands	Trans., Comm., & Utilities	Total Area (ha)
Little River	9386	0	7	36	440	3	0	437	10310
Miami River	20081	463	0	11	463	0	0	1064	22083
Princeton Canal	5168	15572	3378	143	476	17	0	1032	25786
Mowry Canal	10952	15495	2544	1337	976	976	390	390	33062

Figure 11. Miami-Dade County Canal Watershed Land Use (FLUCCS) (2008)

Figure 12. Little River Watershed Land Use Comparison, 1988-2008

Figure 13. Little River Watershed Land Use Map (FLUCCS) (2008)

Figure 14. Miami River Watershed Land Use Comparison (1988-2008)

Figure 15. Miami River Watershed Land Use Map (FLUCCS) (2008)

Figure 16. Princeton Canal Watershed Land Use Comparison (1988-2008)

Figure 17. Princeton Canal Watershed Land Use Map (FLUCCS) (2008)

Figure 18. Mowry Canal Watershed Land Use Comparison (1988-2008)

Figure 19. Mowry Canal Watershed Land Use Map (FLUCCS) (2008)

4.2 Water Quality Analysis

4.2.1 Spatiotemporal Water Quality Analysis

Once the land use types in Miami-Dade County watersheds were quantified, they were compared with water quality parameters through statistical analysis. To incorporate water quality, data was collected from SFWMD DBHYDRO. DBHYDRO is an online database that compiles SFWMD collected information. Nitrogen (mg/L), phosphorous (mg/L), and turbidity (NTU) data were collected for each of the 13 Miami-Dade sampling sites since 1988. These parameters are most indicative of specific land use and storm water runoff (Briceno, 2011). Tables 13 through 25 give a brief description of each sampling site in terms of water quality. This is not meant to be an in depth trend analysis. These tables and Figures 20 through 22 simply set historical bench marks for each parameter.

Table 13. Sampling Site LR01 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
LR01	1988	Nitrate (mg/L)	0.05	0.1	0.02
LR01	1995	Nitrate (mg/L)	0.04	0.1	0.01
LR01	1999	Nitrate (mg/L)	0.08	0.11	0.02
LR01	2004	Nitrate (mg/L)	0.05	0.13	0.01
LR01	2008	Nitrate (mg/L)	0.14	0.29	0.02
LR01	1988	Phosphorous (mg/L)	0.02	0.07	0.003
LR01	1995	Phosphorous (mg/L)	0.01	0.019	0.005
LR01	1999	Phosphorous (mg/L)	0.009	0.016	0.003
LR01	2004	Phosphorous (mg/L)	0.012	0.026	0.005
LR01	2008	Phosphorous (mg/L)	0.01	0.029	0.01
LR01	1988	Turbidity (NTU)	6.32	12.3	2.5
LR01	1995	Turbidity (NTU)	0.77	1.23	0.08
LR01	1999	Turbidity (NTU)	2.7	3.2	1.9
LR01	2004	Turbidity (NTU)	6.5	23.8	2.9
LR01	2008	Turbidity (NTU)	1.36	2.8	0.7

Table 14. Sampling Site LR10 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
LR10	1995	Nitrate (mg/L)	0.02	0.03	0.01
LR10	1999	Nitrate (mg/L)	0.01	0.02	0.01
LR10	2004	Nitrate (mg/L)	0.02	0.05	0.01
LR10	2008	Nitrate (mg/L)	0.04	0.11	0.01
LR10	1995	Phosphorous (mg/L)	0.005	0.008	0.003
LR10	1999	Phosphorous (mg/L)	0.01	0.038	0.002
LR10	2004	Phosphorous (mg/L)	0.01	0.023	0.004
LR10	2008	Phosphorous (mg/L)	0.02	0.028	0.007
LR10	1995	Turbidity (NTU)	1.3	8.63	0.17
LR10	1999	Turbidity (NTU)	1.1	1.3	0.8
LR10	2004	Turbidity (NTU)	1.7	3.98	0.7
LR10	2008	Turbidity (NTU)	0.07	1.7	0.4

Table 15. Sampling Site MR01 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MR01	1988	Nitrate (mg/L)	0.037	0.09	0.01
MR01	1995	Nitrate (mg/L)	0.05	0.11	0.02
MR01	1999	Nitrate (mg/L)	0.06	0.11	0.03
MR01	2003	Nitrate (mg/L)	0.01	0.02	0.01
MR01	2008	Nitrate (mg/L)	0.056	0.1	0.02
MR01	1988	Phosphorous (mg/L)	0.02	0.09	0.003
MR01	1995	Phosphorous (mg/L)	0.008	0.013	0.005
MR01	1999	Phosphorous (mg/L)	0.005	0.014	0.002
MR01	2003	Phosphorous (mg/L)	0.006	0.008	0.005
MR01	2008	Phosphorous (mg/L)	0.004	0.005	0.002
MR01	1988	Turbidity (NTU)	5.69	15	2.1
MR01	1995	Turbidity (NTU)	0.5	1.5	0.14
MR01	1999	Turbidity (NTU)	2.8	2.9	1.9
MR01	2008	Turbidity (NTU)	0.5	0.5	0.5

Table 16. Sampling Site MR03 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MR03	1988	Nitrate (mg/L)	0.093	0.19	0.04
MR03	1995	Nitrate (mg/L)	0.19	0.28	0.14
MR03	1999	Nitrate (mg/L)	0.18	0.25	0.14
MR03	2003	Nitrate (mg/L)	1.9	0.28	0.1
MR03	2008	Nitrate (mg/L)	0.2	0.23	0.17
MR03	1988	Phosphorous (mg/L)	0.03	0.02	0.05
MR03	1995	Phosphorous (mg/L)	0.017	0.03	0.012
MR03	1999	Phosphorous (mg/L)	0.01	0.012	0.005
MR03	2003	Phosphorous (mg/L)	0.012	0.016	0.009
MR03	2008	Phosphorous (mg/L)	0.012	0.014	0.009
MR03	1988	Turbidity (NTU)	3.46	5.8	1.3
MR03	1995	Turbidity (NTU)	0.5	0.844	0.19
MR03	1999	Turbidity (NTU)	1.4	1.6	1.3
MR03	2003	Turbidity (NTU)	3.2	4.2	1.9
MR03	2003	Turbidity (NTU)	1.5	1.9	1
MR03	2008	Turbidity (NTU)	0.7	0.7	0.7

Table 17. Sampling Site MR05 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MR05	1988	Nitrate (mg/L)	0.1	0.19	0.03
MR05	1995	Nitrate (mg/L)	0.2	0.27	0.14
MR05	2008	Nitrate (mg/L)	0.21	0.27	0.16
MR05	1988	Phosphorous (mg/L)	0.03	0.05	0.02
MR05	1995	Phosphorous (mg/L)	0.016	0.024	0.009
MR05	2008	Phosphorous (mg/L)	0.01	0.014	0.009
MR05	1988	Turbidity (NTU)	3.35	11.5	1.4
MR05	1995	Turbidity (NTU)	0.525	0.94	0.18
MR05	2008	Turbidity (NTU)	0.76	0.9	0.6

Table 18. Sampling Site MR06 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MR06	1988	Nitrate (mg/L)	0.113	0.29	0.03
MR06	1995	Nitrate (mg/L)	0.14	0.26	0.01
MR06	1999	Nitrate (mg/L)	0.08	0.18	0.02
MR06	2003	Nitrate (mg/L)	0.024	0.37	0.08
MR06	1988	Phosphorous (mg/L)	0.03	0.06	0.02
MR06	1995	Phosphorous (mg/L)	0.016	0.026	0.008
MR06	1999	Phosphorous (mg/L)	0.007	0.01	0.004
MR06	2003	Phosphorous (mg/L)	0.012	0.018	0.008
MR06	1988	Turbidity (NTU)	3.51	10	1.5
MR06	1995	Turbidity (NTU)	0.625	0.999	0.29
MR06	1999	Turbidity (NTU)	2.6	5.9	0.9
MR06	2003	Turbidity (NTU)	1.76	2.7	1

Table 19. Sampling Site MR07 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MR07	1988	Nitrate (mg/L)	0.063	0.21	0.02
MR07	1995	Nitrate (mg/L)	0.04	0.11	0.01
MR07	1999	Nitrate (mg/L)	0.08	0.11	0.04
MR07	2003	Nitrate (mg/L)	0.15	0.26	0.07
MR07	1988	Phosphorous (mg/L)	0.04	0.07	0.02
MR07	1995	Phosphorous (mg/L)	0.02	0.028	0.005
MR07	1999	Phosphorous (mg/L)	0.01	0.041	0.004
MR07	2003	Phosphorous (mg/L)	0.014	0.019	0.009
MR07	1988	Turbidity (NTU)	4.23	16	1
MR07	1995	Turbidity (NTU)	0.479	0.798	0.14
MR07	1999	Turbidity (NTU)	2.35	3.8	0.9
MR07	2003	Turbidity (NTU)	1.5	2.5	0.8

Table 20. Sampling Site MW01 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MW01	1988	Nitrate (mg/L)	0.32	0.01	0.76
MW01	1995	Nitrate (mg/L)	0.585	2.09	0.02
MW01	1999	Nitrate (mg/L)	0.76	1.75	0.01
MW01	2004	Nitrate (mg/L)	0.66	2.72	0.03
MW01	2008	Nitrate (mg/L)	0.585	1.64	0.1
MW01	1988	Phosphorous (mg/L)	0.02	0.044	0.008
MW01	1995	Phosphorous (mg/L)	0.01	0.029	0.004
MW01	1999	Phosphorous (mg/L)	0.006	0.011	0.005
MW01	2004	Phosphorous (mg/L)	0.011	0.017	0.005
MW01	2008	Phosphorous (mg/L)	0.006	0.01	0.005
MW01	1988	Turbidity (NTU)	1.67	2.3	0.7
MW01	1995	Turbidity (NTU)	0.47	1.33	0.2
MW01	1999	Turbidity (NTU)	1.65	2.7	1
MW01	2004	Turbidity (NTU)	1.19	2.2	0.7
MW01	2008	Turbidity (NTU)	0.68	1	0.4

Table 21. Sampling Site MW04 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MW04	1995	Nitrate (mg/L)	2.4	3.2	1.68
MW04	1999	Nitrate (mg/L)	2.43	3.12	1.79
MW04	2004	Nitrate (mg/L)	2.2	3.23	1.69
MW04	2008	Nitrate (mg/L)	1.8	3.26	1.1
MW04	1995	Phosphorous (mg/L)	0.003	0.005	0.002
MW04	1999	Phosphorous (mg/L)	0.0037	0.008	0.003
MW04	2004	Phosphorous (mg/L)	0.02	0.102	0.001
MW04	2008	Phosphorous (mg/L)	0.004	0.003	0.005
MW04	1995	Turbidity (NTU)	0.27	0.59	0.11
MW04	1999	Turbidity (NTU)	0.8	1.1	0.7
MW04	2004	Turbidity (NTU)	2.3	17.4	0.5
MW04	2008	Turbidity (NTU)	0.55	1.8	0.3

Table 22. Sampling Site MW13 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
MW13	1995	Nitrate (mg/L)	0.013	0.01	0.03
MW13	1999	Nitrate (mg/L)	0.035	0.06	0.03
MW13	2004	Nitrate (mg/L)	0.15	1.44	0.11
MW13	1995	Phosphorous (mg/L)	0.003	0.005	0.001
MW13	1999	Phosphorous (mg/L)	0.002	0.005	0.001
MW13	2004	Phosphorous (mg/L)	0.022	0.19	0.004
MW13	1995	Turbidity (NTU)	0.6	1.27	0.003
MW13	1999	Turbidity (NTU)	1.95	3.8	0.8
MW13	2004	Turbidity (NTU)	2.45	3.69	0.9

Table 23. Sampling Site PR01 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
PR01	1988	Nitrate (mg/L)	1.31	2.66	0.02
PR01	1995	Nitrate (mg/L)	1.5	3.06	0.36
PR01	1999	Nitrate (mg/L)	2.23	3.48	0.81
PR01	2004	Nitrate (mg/L)	1.32	3.74	0.06
PR01	2008	Nitrate (mg/L)	2.1	3.92	0.07
PR01	1988	Phosphorous (mg/L)	0.02	0.03	0.002
PR01	1995	Phosphorous (mg/L)	0.009	0.015	0.004
PR01	1999	Phosphorous (mg/L)	0.004	0.008	0.001
PR01	2004	Phosphorous (mg/L)	0.017	0.021	0.007
PR01	2008	Phosphorous (mg/L)	0.006	0.011	0.003
PR01	1988	Turbidity (NTU)	2.6	7.2	1.3
PR01	1995	Turbidity (NTU)	0.6	2.33	0.11
PR01	1999	Turbidity (NTU)	1.05	1.6	0.5
PR01	2004	Turbidity (NTU)	2.7	13.3	1.2
PR01	2008	Turbidity (NTU)	0.93	1.6	0.4

Table 24. Sampling Site PR03 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
PR03	1988	Nitrate (mg/L)	3.71	4.17	2.66
PR03	1995	Nitrate (mg/L)	3.73	4.86	1.89
PR03	1999	Nitrate (mg/L)	4.06	4.45	3.29
PR03	2004	Nitrate (mg/L)	4.6	3.69	5.07
PR03	2008	Nitrate (mg/L)	4.5	5.27	2.12
PR03	1988	Phosphorous (mg/L)	0.01	0.02	0.001
PR03	1995	Phosphorous (mg/L)	0.002	0.006	0.001
PR03	1999	Phosphorous (mg/L)	0.002	0.003	0.001
PR03	2004	Phosphorous (mg/L)	0.005	0.013	0.004
PR03	2008	Phosphorous (mg/L)	0.005	0.002	0.006
PR03	1988	Turbidity (NTU)	2.67	19.9	0.5
PR03	1995	Turbidity (NTU)	0.28	0.97	0.1
PR03	1999	Turbidity (NTU)	0.38	0.5	0.2
PR03	2004	Turbidity (NTU)	2.53	7.5	0.379
PR03	2008	Turbidity (NTU)	0.78	0.3	3.8

Table 25. Sampling Site PR08 Historical Water Quality Data (SFWMD)

Site	Year	Parameter	Average	Max	Min
PR08	1988	Nitrate (mg/L)	0.4	2.81	0.04
PR08	1995	Nitrate (mg/L)	0.02	0.05	0.01
PR08	1999	Nitrate (mg/L)	0.08	0.37	0.02
PR08	2004	Nitrate (mg/L)	0.029	0.05	0.01
PR08	1988	Phosphorous (mg/L)	0.03	0.06	0.015
PR08	1995	Phosphorous (mg/L)	0.003	0.004	0.001
PR08	1999	Phosphorous (mg/L)	0.003	0.007	0.001
PR08	2004	Phosphorous (mg/L)	0.016	0.127	0.004
PR08	1988	Turbidity (NTU)	2.17	2.7	0.4
PR08	1995	Turbidity (NTU)	0.77	1.36	0.3
PR08	1999	Turbidity (NTU)	12.9	33.7	3.9
PR08	2004	Turbidity (NTU)	3	5.8	0.8

Figure 20. Nitrate/Nitrite Historical Water Quality Data

Figure 21. Phosphorous Historical Water Quality Data

Figure 22. Turbidity Historical Water Quality Data

4.2.2 Effect of land use type on water quality: Factor and Cluster Analysis

The 13 water quality and land use variables were run through a factor analysis. These parameters were reduced to four principal components. They had a 68.2% variability which shows that while the complexity has been reduced, there is still a large amount of variance in the variables. Many researchers only accept greater than 75% variability. Water quality is irregular and unpredictable by nature. This is due to the influences of seasonality and rainfall implications. This could be the reason that the variability falls just below the acceptable range at 68.2%. The proximity of the following variables to one another show which are more related over time and across Miami-Dade County.

Figure 23. Sampling Site Factor Analysis

The next step in the statistical analysis was to use those remaining four factors in a cluster analysis. This process groups the sampling sites so that each cluster contains similar land use sites. For the first dendrogram (Figure 21), the sampling sites were only divided into two clusters. This shows a distinct separation between land use percent compositions found in different watersheds. Cluster 1 contains sites PR01, PR03, PR08, MW01, MW04, and MW13. All these sampling sites are all greater than 90% Urban and Built-Up land use. Cluster 2 contains sites MR01, MR13, MR05, MR06, MR07, LR01, and LR10. These sampling sites are greater than 70% agricultural land use, as seen in Figure 23.

Figure 24. Sampling Site Dendrogram (2 clusters)

Figure 25. Sampling Site Watershed Land Use Comparison

In the next dendrogram (Figure 23), the sampling sites were further split into four separate clusters. This also displayed more detailed distinctions in land use percent composition. While PR08, MW01, MW04, and MW13 were initially grouped with PR01 and PR03, in the 4 cluster analysis they create a separate group. This may be due to the fact that they have a higher urban percent composition (Figure 24). Although it is not the

dominating land use type, the dendrogram shows that an increased amount of urban storm water runoff may change overall water quality over time.

Figure 26. Sampling Site Dendrogram (4 clusters)

Figure 27. Sampling Site Watershed Land Use Comparison

This statistical analysis shows that there is a relationship between land use and water quality in Miami-Dade County. It could potentially be used to predict how water quality will be affected by further development in South Florida. Miami-Dade County is an intensely populated area and is predicted to continue growing. This will put immense

pressures on the land and surrounding water bodies. It is important to understand how land use will affect South Florida's surface water runoff before it becomes a serious issue.

4.3 Least Cost Model of Nutrient Abatement

As seen in Figures 15 through 18, Miami River basin and Princeton canal basin represent a juxtaposition of South Florida land use compositions. Miami River basin is over 90% Urban and Built-Up land while Princeton Canal basin is over 80% dominated by Agricultural land. This has extreme implications on the type of runoff received in local water ways.

The Princeton Canal watershed has a current nitrogen loading of 56,746 kg/year and phosphorous loading of 9,263 kg/year (Table 4). This is directly correlated with the high concentration of agricultural areas reflected by the high nitrogen loading. The factor analysis presented earlier shows that nitrate/nitrite (NOX) are more closely related to agricultural areas, while phosphorous is on the other side of the spectrum with urban land use types. For the Princeton Canal watershed model, the two nutrient loading levels could be only be reduced by a maximum of 45% with the three model BMPs. Nitrogen and phosphorous can be reduced up to 25% without any significant changes in the total costs while utilizing BMP1 alone. The least-cost model assigned BMP 3 to more than 90% of the Agricultural areas in this watershed when the nutrient reductions are set at 45% lower than the current level. This shows that more expensive techniques are necessary to improve agricultural surface water runoff reduction targets were increased. Conversely,

in urban areas nutrient reductions can be accomplished without incurring significant costs. In urban areas, the least cost model shows that it is more cost effective to simply lower the amount of fertilizer inputs in the area. BMP 2 and BMP 3 are excessively expensive and do not have an adequately high reduction percentage to prove useful in urban areas.

Until the target 30% reduction, Princeton Canal Basin's 20,739 hectares are solely treated with BMP 1 (fertilizer reduction). At the 45% maximum potential reduction, the more expensive BMP 3 (chemical treatment) must be applied to over 15,000 ha to reach the target. BMP 2 (water management and detention) was rarely implemented due to its high investment costs (\$320/ha/year) and additional yearly maintenance fees (\$1,024/ha/year).

Table 26. Princeton Canal Least Cost Abatement Model Results

Percent Reduction (%)	Total Cost (\$)	N Loading (kg)	P Loading (kg)
0	0	56,746	9,263
5	122,966	43,102	6,863
10	122,966	43,102	6,863
15	122,966	43,102	6,863
20	122,966	43,102	6,863
25	174,892	42,362	6,802
30	599,941	39,191	6,484
35	1,101,811	36,877	5,958
40	1,788,050	33,342	5,558
45	6,819,824	31,210	4,996

Figure 28. Princeton Canal Percent Reduction and Total Cost: the total abatement cost increases steeply as the target reduction exceeds 40 percent of the current level.

The Miami River watershed has a current nitrogen loading of 39,630 kg/year and phosphorous loading of 6,945 kg/year (Table 5). The Miami River watershed nutrient loading levels could only be reduced a maximum of 25% with a combination of BMP techniques (Figure 27). Nearly 90% of the BMPs used for this 25% reduction was found through the inexpensive BMP 1 (Figure 30) and agricultural areas were solely allotted to the moderately expensive BMP 3. This is similar to the Princeton canal agricultural areas where BMP 3 was also applied to maximize the reduction.

According to the least cost abatement model, BMP 2 and BMP 3 are not cost effective in the Miami River watershed until higher nutrient loading reduction targets are set. At the maximum reduction scenario, it is extremely costly, at 25 million dollars, to reduce nutrient loading in Miami River basin. This area is dominated by urban land use

development and requires the most expensive BMP 2 to reach its target. It is significantly more expensive to reduce Miami River loading by 25% than it is to reduce Princeton Canal loading by 45% (Figure 28).

This may be different if a time restraint was added to the equation. While BMP 1 is significantly less expensive than BMP 2 and BMP 3, it also has a much slower response time. If a nutrient reduction project required a quicker response, it may be optimal to incorporate the more expensive BMPs.

Table 27. Miami River Least Cost Abatement Model Results

Miami River Least Cost Abatement Model Results			
Percent Reduction (%)	Total Cost (\$)	N Loading (kg)	P Loading (kg)
0	0	39,630	6,945
5	138,359	29,484	6,552
10	2,324,122	29,275	6,250
15	5,575,002	29,879	5,888
20	8,537,290	29,977	5,556
25	24,930,588	29,722	5,208

Figure 29. Miami River Percent Reduction and Total Cost: the total abatement cost increases steeply as the target reduction exceeds 20 percent of the current level.

Figure 30. Princeton Canal and Miami River Percent Reduction and Cost Comparison

Figure 31. Princeton Canal Watershed BMP Percent Implementation at Maximum Reduction

Figure 32. Miami River Watershed BMP Percent Implementation at Maximum Reduction

Figure 33. Princeton Canal Watershed BMP Acreage

Figure 34. Miami River Watershed BMP Acreage

4.4 Land Use and Water Quality Management Implications

The management implications from the results of this study are very different for urban and agricultural areas. There are fundamental economic, technical, and political challenges to implement these strategies in South Florida. Agricultural areas have a multitude of strategies to reduce the amount of nutrient loading in Miami-Dade canals (Figure 33). This study shows that it is extremely costly to reduce nutrient loading in residential areas due to urban design issues. It would be difficult to implement these best management practices in Miami-Dade due to restrictions on land parcels in urban areas. In the current study, creating detention ponds (BMP #2) would be almost impossible to incorporate into the current urban landscape in areas like downtown Miami. There are few properties available to create substantial storm water collection areas.

Local management base most water quality decisions based on the Clean Water Act (CWA) and command and control structured policies. Command and control approaches are usually inefficient and result in high financial costs for polluting land owners. The EPA reports that in 1997 annual private and public point source costs associated with achieving water quality standards were about \$48 billion (EPA, 2000). More market based approaches can be used to create incentives for technological development and more voluntary reductions in nutrient loading.

Water quality trading programs could prove to be the optimal solution for Miami-Dade County. Trading programs are designed to allow one source to maintain its nutrient loading rate while reducing another source with lower costs associated with reduction (King and Kuch, 2003). This helps protect the overall water quality in an area without using an excessive amount of funds. This effort can be more effective since they are

flexible in terms of monetary allocation and overall reduction goals. Water quality trading credits would be distributed amongst different land owners. Since it is so expensive to implement nutrient reduction in Miami-Dade's urban areas, water quality trading programs would be a feasible solution in South Florida. Instead of spending 25 million dollars to remove a fraction of the nutrient loading in urban areas, Miami-Dade management and policy makers could focus on agricultural areas.

Urban developers would buy water quality credits from agricultural land owners to reduce the total amount of reduction they need to reach. Not only is this an additional source of income for farmers, but it would also ensure that the overall water quality in Miami-Dade County is protected. In a similar study on the S-191 sub-basin in Lake Okeechobee the estimated potential cost savings was of 27% (\$1.3 million per year), based on a water quality trading credit price of \$179 (Corrales, et al., 2014). Dairy farmers were the major buyers of the trading credits whereas improved pastures sold most of the credits. More flexibility in policy and management strategies can lead to better overall results. Similar results could possibly be reached by implementing creative policies like a trading program in South Florida.

It is important to remember that implementing one best management practice or environmental policy to an entire area is not the most efficient or cost effective solution to reduce nutrient loading. Specific land use types in Miami-Dade, and across the globe, need the appropriate strategies to have the best results.

Chapter 5

5. Conclusion and Recommendation

This study shows that there is a relationship between land use and water quality in Miami-Dade County. It could potentially be used to predict how water quality will be affected by further development in South Florida. Miami-Dade County is an intensely populated area and is predicted to continue growing. This will put immense pressures on the land and surrounding water bodies. It is important to understand how land use will affect South Florida's surface water runoff before it becomes a serious issue. It is also important to consider how to mitigate potential increases in nutrient loading to these waterways. The linear programming model shows that there are multiple strategies available to increase water quality in South Florida.

Local land use type has huge implications on the concentrations of nutrients in surrounding water bodies. Through GIS and statistical analysis, it was found that there is a correlation between nutrient parameter concentrations and surrounding land use types. Sampling sites were directly clustered using a dendrogram according to what land use type the particular basin had.

Miami-Dade County projects almost a 15% population increase by 2020. Rapid increase in population could lead to further development and contaminated runoff potential. Increased pressure on existing resources may have direct implications on Miami-Dade's surrounding ecosystems and hydraulic health. Watersheds in areas near the Little River and Miami River will have increasing nutrients and potential pollutants in its surface water runoff.

According to the Binary Integer Model of least cost abatement, Princeton canal loading could be potentially reduced up to 45% using a combination of BMP techniques. Miami River could only be reduced up to 25% due to its higher concentration of urban areas within the basin. It is also practically three times more expensive to reduce nutrient loading in Miami River than Princeton canal.

5.1 Limitations

One of the biggest challenges of this project was collecting complete and comprehensive historical water quality data. Future studies should focus on areas with more complete collection of information. Some sampling sites in this project were taken out due to incomplete historical water quality records. The more sampling sites included and data considered can only strengthen the conclusions reached in further studies.

The practicality of implementation is another limitation found in the current study. In the Miami River watershed basin, to achieve maximum percent nutrient reduction, the model suggests to create retention ponds in urban areas. This model does not take into account the amount of revenue lost on development in downtown Miami along the Miami River. To create detention ponds, buildings would have to be destroyed to make enough room to be effective storm water filtration areas. This is a study area specific issue that is important to consider in Miami-Dade County. The model only considers the overall costs associated with creating detention ponds averaged over the entire state of Florida. Property values would have to be incorporated to calculate the true costs associated with implementation in Miami-Dade County,

5.2 Recommendations

While the future of water quality in South Florida is a huge concern, there are many different techniques to combat excess nutrient concentrations. There is a range of BMP scenarios that can be altered for any target nutrient reduction and specific funding range. Agricultural areas can easily and inexpensively lower fertilizer inputs and increase water quality. Miami-Dade urban areas can also implement methods to better water quality although it may be more expensive.

These findings can be used to shape future policy and management decisions in Miami-Dade County. By looking at the historical correlations between land use and water quality, it can be the foundation for more strict legislation regarding development opportunities near vulnerable waterways. It is important to keep these solutions in mind when looking towards future environmental health in South Florida.

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