THE ECONOMICS OF FLOW ENHANCEMENTS VERSUS NUTRIENT CONTROLS IN

MEETING WATER QUALITY STANDARDS

by

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(Under the Direction of Andrew G. Keeler)

ABSTRACT

Many of the rivers and streams in Georgia are still not supporting their designated uses. Poor water quality reduces the productivity of water downstream. Water quality can be improved either by reducing loads from point and nonpoint sources or by increasing water for dilution. Reduction of loads from point sources has been the primary strategy for improving water quality, but this approach has not led the rivers and streams of Georgia to meeting the water quality standards. The Total Maximum Daily Load program is a key policy tool to address this problem. While the TMDLs written for impaired segments in Georgia address load reduction from point and nonpoint sources, the role of instream flow is not addressed as a management option. The purpose of this paper is to explore the economics of using enhanced flow as part of strategies to meet water quality targets.

INDEX WORDS: Water Quality, TMDLs, Cost-Effective, Instream Flow

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CHAPTER I

INTRODUCTION

I.1 BACKGROUND

Since the passage of the Clean Water Act (CWA) in 1972, there has been considerable

progress towards the goal of achieving 'fishable' and 'swimmable' surface water quality for all

our streams. Despite these improvements, in 1999, over half of the surface waters tested in

Georgia were found to be too polluted to be used for fishing, swimming, or drinking – much

higher than the national average. Forty percent of the US waters currently meet the standards to

support their designated uses.

Section 303(d) is one directive of the CWA that has generated considerable attention.

The basic process behind the program is assigning the States' the responsibilities of (1)

identifying and listing polluted water bodies according to the severity of pollution; (2)

establishing Total Maximum Daily Loads (TMDL) for each pollutant for all polluted water

bodies; and (3) implementing TMDLs as part of States' Water Quality Management Plans

(USEPA, 1999a). The TMDL process involves estimating the maximum allowable pollutant

loadings for a polluted water body and then determining the allocation of maximum allowable

loadings for point and non-point sources separately. Additionally, it requires an allocation for a

'margin of safety' to account for uncertainties or lack of knowledge in the relationship between

pollutant loads and the quality of receiving streams. The allowable pollutant loads must ensure

that the water body will attain and maintain water quality standards throughout the year.

Until recently, the emphasis has been on point source pollution reduction through the National Pollution Discharge Elimination System (NPDES) program. Improvements in our water quality have been largely due to a reduction in toxic and organic chemical loadings from point sources, such as industries and municipal wastewater treatment plants. Due to the complex nature of nonpoint source pollution, traditional policy instruments based on emissions (e.g. emissions taxes or regulations) have been virtually nonexistent. As a result, the objectives of the TMDL process proved to be unattainable without adequate control of the nonpoint source discharges.

The TMDL program has taken on a new initiative for water quality management, called the watershed approach, in order to integrate nonpoint sources into the equation. It is a collaborative approach to environmental management composed of state, tribal, federal and local governments and the private sector. The new approach is an overall water quality approach that aims to build a greater sense of participation and collaboration among all agents in the watershed rather than focusing on technology based point source control (USEPA, 1998). Since watershed level planning allows pollution control by the least-cost sources and provides opportunities for pollution trading among sources, it bears the promise of cost-savings and innovative approaches for TMDL implementation. This paper evaluates an approach for addressing water quality problems that involves the possibility of increasing flows through increased water use efficiency or reduced withdrawals. Increased river flow could increase the assimilative capacity of the river while benefiting aquatic habitat, increasing supply downstream, and reducing other problems associated with low flows.

I.2 PROBLEM STATEMENT

Despite the separation in water quantity and water quality law and policy, the two are interrelated. The TMDL program requires that the water bodies of Georgia be evaluated for their capacity to assimilate pollutants without adversely affecting its quality – usually an "acceptable" concentration. Increases in both surface and groundwater withdrawals can reduce the assimilative capacity of the water body and exacerbate the problem of nonpoint source pollution, resulting in loss of aquatic habitat and a greater intermittency of streams and river. In a drought year (for example, half the flow), the same load will have twice the effect on instream concentration, placing a greater burden on reduction of pollutants in the presence of low flow levels. Despite this Georgia does relatively little to maintain instream flows. Permits are issued for surface and ground water withdrawals over 100,000 gallons per day, but there is no requirement for reporting actual water use in agriculture.

Water quality is not only determined by the amount of waste discharged into a stream, but also by the quantity and location of water depletion. Upstream water depletion reduces the amount of water available to the downstream system. Consequently, there is a reduction of the stream's capacity to transport and assimilate wastes. Less water for dilution of waste substances results in a higher concentration of the pollutant downstream. While the focus of the TMDL program has been controlling waste discharge, the real pollution damage to the environment and human-beings is a function of the waste concentration rather than of the waste discharge per se. Still, there has been largely no discussion about reducing both water usage and waste discharge simultaneously.

Improving water quality by means of controlling waste discharge without considering the effects of the water withdrawals is unlikely to result in an efficient solution. Similarly, allocation of water resources between all users based on the equimarginal principle will not be effective if

damages to downstream users from upstream water depletion are not taken into consideration. The problem of water quality as well as quantity management is a problem of resource allocation. Policies should aim to allocate water resources to achieve an effective solution.

I.3 OBJECTIVES

The TMDL program has focused on a reduction in pollutants from point and nonpoint sources. In this study we will investigate the feasibility of using instream flow as a management option for reaching water quality targets. Increased river flow through irrigation reductions could increase the assimilative capacity of the river, achieve reductions in nonpoint loadings, enhance and restore riverine and wetland habitats, and improve water quality. In any case, farmers may have the option of reducing irrigation levels, which may be cost-effective in comparison to TMDL-mandated pollutant removal alternatives.

The introduction of flow augmentation as a management option may provide the opportunity to help decrease river pollution and reverse the trend of diminishing flows in the Flint River Basin. We will explore the feasibility of inducing increased river flow as a means to offset or complement required pollutant reductions. Possible methods of increasing flow include improving irrigation efficiency or reducing the amount of surface irrigation withdrawals through voluntary, incentive-based methods. We use the spreadsheet water quality model (SWQM) to simulate the effects of best management practice (BMP) implementation and flow augmentation on instream water quality. This model will be applied to two stream segments in the Flint River Basin. The costs of augmenting flows and using agricultural BMPs will be estimated and compared. Finally, an economic model will be used to determine how to reach water quality targets at the least cost. For a given level of water quality, the costs of achieving water quality at

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that level will be compared under three options: reducing pollutant loads; augmenting flow; and a combination of the two.

I.4 CASE STUDY

The southern part of the Flint River Basin provides the case study for this thesis. Southwest Georgia has experienced a drastic increase in agricultural land use, especially row crops, since the 1960's. Since the beginning of 1994, agriculture lands have increased by about 20% in the region. This has led to associated increases in center pivot irrigation (Hicks et al., 1987). In 1999, approximately 85% of agricultural lands were irrigated (Litts et al., 2001). Increases in irrigated agriculture often create water quality problems associated with nonpoint source runoff into streams and rivers. The rise in acres of irrigated land and increase in population, along with the recent drought, has brought water supply concerns to the area. In the two segments that we examine, cropland and forestry comprise the majority of the watersheds. There are several point sources in the watersheds, all of which are all sewage plants.

The impairment this study addresses is low concentration levels of dissolved oxygen (DO). DO is the amount of molecular oxygen dissolved in water and is a primary indicator of overall water quality and the viability of the aquatic habitat (Melching and Flores 1999). A number of abiotic and biotic factors, such as reaeration, stream respiration, nutrient and organic loading affect DO concentration in streams. Low base-flow can also aggravate DO problems. Various point and nonpoint sources of pollution such as waste water treatment plant (WWTP) discharges and agricultural runoff affect DO concentration in streams as well. The water's carrying capacity for DO, also known as the DO saturation level, depends on the temperature of the water and is significantly related to stream flow. DO water quality standards have been established ranging from 4.0 to 6.0 milligrams per liter (mg/L) depending on the stream

classification and type of measurement. As the DO concentration decreases, the stress placed on aquatic life increases.

Possible remedies for low dissolved oxygen include implementing agricultural BMPs, reducing loadings from point sources, or augmenting flow. Since agriculture is the principal source of surface water diversions in the area, any efforts to augment stream flow will necessarily concentrate on reducing irrigation diversions. This, in turn, will impact farm enterprises in the region of reducing irrigation water use. Likewise, implementing agricultural BMPs will incur costs. A central public and policy consideration will be the cost of actions to improve water quality. The costs of management options may be prohibitive in some cases due to site specifics including physical characteristics and social acceptance. A goal in the design of pollution reduction programs is to achieve the greatest possible improvement for a given cost (Heatwole et al., 1987). Understanding the impact of various sources on the DO concentration in water bodies as well as the cost of management practices which will improve the concentration is important to figuring out the best way to improve water quality.

I.4 CONTRIBUTIONS OF THIS RESEARCH

This study contributes to the existing literature on water quality management in two aspects:

1. It provides a framework for integrating water quality and quantity in watershed management.

2. It provides an innovative approach to reaching water quality goals at the least cost.

I.4.1 The Integration of Water Quality and Quantity

Historically, the first concern of water management has been quantity. Individually or collectively, people have always sought to assure themselves enough water to meet their requirements. However, during the twentieth century, water quality considerations have also

become an important part of water management. Maturing concerns about water quality began to result in a dualism in the water management world. One set of policies, laws and institutions was created and given authority over developing and maintaining an adequate quantitative supply of water, while another set of policies, laws and institutions was generated and given responsibility for assuring that the quality of the nation's water was guarded. While water quantity issues remain primarily in the hands of the States', a number of institutions at local, state, and federal levels deal with water quality issues. Jurisdiction over water resources policy is fragmented among at least thirteen Congressional committees, twenty-three Congressional subcommittees, eight Cabinet level departments, six independent agencies and two White House offices (Deason et al. 2001). To further complicate water resource policy issues, those federal entities with authority over water resource planning are not the same entities that have jurisdiction over the funding for water-related projects. It has been argued that states are often left out of the federal planning process, but the reports that the states are required to make every two years concerning the state of their water resources are the basis for national water quality policy (Deason et al. 2001).

The allocation and quality of water are interrelated in a number of ways. A critical part of the water supply problem is the continuing, spreading, and intensifying degradation of water quality at many locations. This adds to the problem of managing supply and demand, as it means that water available in a quantitative sense is not always available to water users in a qualitative sense. Thus water quality directly affects the quantity of water that can be employed for various purposes. Water quantity also affects water quality in various ways but primarily through flow (i.e. scour) and dilution associated with volume changes both on a geographically

distributed and a seasonal basis. Additional water quality impacts can be associated with water reallocation, particularly from agriculture to municipal and industrial uses.

The quality and quantity of water in our rivers and streams will invariably be affected by land use by humans in the form of agriculture, construction, manufacturing, and forestry. This includes increased nutrients loads from point and non-point sources (NPS), toxic loading from point sources, and increases or decreases in the stream flows – all of which might lead to a decline in the overall water quality of the streams and make them unfit for recreation or human and wildlife consumption. As an increase in irrigated agriculture and population continues to place more stress on the waters of Georgia, water policy is becoming an increasingly important topic of debate.   For example, in 1996 the Georgia Chapter of the Sierra Club and other nongovernmental organizations filed a lawsuit against the EPA. The charge stated that for 20 years the EPA had failed to force Georgia to develop and implement TMDLs. The environmental organizations won the case and the EPA has since signed a series of Consent Decrees. These state that TMDLs must be completed in a timely fashion for all impaired waters in Georgia. Also, the EPA must take responsibility for developing TMDLs if the EPD fails to do so. The consent decrees gave specific dates for the completion of TMDLs for all impaired waters in Georgia. This had forced the EPD and the EPA to develop TMDLs with limited resources and time.

While public awareness about water quality policy is growing, concerns over water supply for industry and municipalities in the Atlanta region and agriculture in the south are also on the rise. These concerns were exacerbated by the recent drought and are reflected in recent policy statement and actions. Georgia has specified the need for water conservation and, in O.G.C.A. Section 12-5-474(c), requires preparation of a water conservation plan for use of any

water supply facilities. The EPD has also established a water conservation rule that amends the allowance of Surface Water Withdrawals (GA DNR 2002). In addition, the ongoing debate with Alabama and Florida over water rights is forcing attention to the issue of water allocation. The costs of developing and implementing policies to address all of these problems are a central issue.

I.4.2 The Costs of TMDLs

The EPA (2001a) conducted a study on the national costs to develop and implement TMDLs. Public and private costs associated with the TMDL efforts are estimated to be $1.035 billion for development of TMDL plans, $255 million for additional monitoring to support TMDLs and $13.5 to $64.5 billion for the implementation of TMDL plans over the next fifteen years (USEPA 2001). Though these are estimates, one conclusion that can be drawn from EPA's cost study is that implementing TMDLs will undoubtedly cost billions of dollars. It is therefore necessary to consider all options when attempting to reach water quality goals at the least cost.

I.5 OVERVIEW OF THESIS

The thesis is divided into five chapters: introduction; literature review; methods; results and discussion; and summary and conclusions. The literature review begins with a discussion of the history of efforts to model water quality, and more specifically, dissolved oxygen. A brief description of the TMDL program follows. The economics of water quality and instream flow augmentation are then covered. In the methodology chapter, data sources are cited, the modeling approach is explained, and the methodology behind the choice of different management scenarios and their costs are presented. Chapter four reports the results of each step: water quality predictions under different management scenarios, and the associated costs of their

implementation. It includes a discussion of the results and their implications. The concluding

chapter summarizes major findings and discusses some future applications.

CHAPTER II

LITERATURE REVIEW

A substantial amount of work has been done on estimating the costs of water quality improvement. These range from engineering and econometric studies of how costs vary with the degree of waste removal in treatment plants, through studies of the costs of implementing agricultural best management practices, to estimates of the costs of water quality improvement through flow augmentation for a particular region. Studies have differed greatly in complexity, used different types of water quality targets, and different constraints.

For the purpose of our study, both water quality modeling and economic theory related to water quality and quantity are employed. In this chapter an overview of the existing relevant literature on water quality modeling, water quality management, and the economics of instream flow are presented.   Because the study takes place in the context of the TMDL program, a background of the program and modeling efforts for its implementation is included in the water quality modeling section. The main objective is to draw methodological insights for the present study from the existing literature.

II.1 WATER QUALITY/ DISSOLVED OXYGEN MODELLING

II.1.1 Introduction

Assessing the impact on receiving waters was the first water-quality modeling endeavor taken up by engineers. The immediate impact of raw sewage discharge on water bodies such as streams and rivers is the depletion of dissolved oxygen, since the aquatic microorganisms utilize

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dissolved oxygen for decomposing the degradable part of sewage. In addition, a sediment oxygen demand supplements the decay in the water. As oxygen levels drop, atmospheric oxygen enters the water to compensate for the imbalance. Initially, oxygen consumption in the water and to the sediments is more than reaeration, but after some time, the reaeration rate becomes equal to the depletion rate. At this point the critical level of oxygen in the water is reached. This is the lowest level of oxygen in the water. Beyond the point, reaeration rate overcomes the depletion rate and the oxygen level in the water starts increasing. At some point downstream of the "oxygen sag", the level of oxygen returns to the initial values. This is the zone of recovery and is characterized by the growth of plants on the nutrients released from the decomposition process. Reaeration is the most important natural means of DO recovery for polluted streams (Melching and Flores, 1999). Besides reparation, aquatic plant photosynthesis and respiration are a major source and sink of oxygen in water bodies respectively.

Nutrient inputs from both agricultural nonpoint source pollution and municipal and industrial discharges may stimulate autotrophic growth in streams. The presence of autotrophic organisms in streams affects the DO balance in streams and primary production. This forms the link between nutrient loads and degradation of water quality (Hajda and Novotny, 1996). When nutrients enter a water body, it leads to a depressed DO. Low flow seems to aggravate the DO depression (Hajda and Novotny, 1996). The reasons may include limited nutrient dilutions, decreased turbidities and increased residence times. The process of DO dynamics is, then, not only a balance of biochemical oxygen demand (BOD) decay and reaeration, but also involves other anthropogenic factors such as nutrient loading and the subsequent eutrophication, as well as demands from sediment decays and benthic respiration. Table 2.1 presents a summary of the major sources and sinks for DO.

Table 2.1 Summary of Sources and Sinks of Dissolved Oxygen.

Sources of DO	Sinks of DO
1. Reaeration from the atmosphere 2. Photosynthetic oxygen production. 3. DO incoming tributaries or effluents.	1. Oxidation of carbonaceous waste material. 2. Oxidation of nitrogenous waste material. 3. Oxygen demand of sediments of water body. 4. Use of oxygen for respiration by aquatic plants.

II.1.2 Models

Dissolved oxygen is a surrogate variable for the general health of the aquatic ecosystem. The impact of low DO concentrations or of anaerobic conditions is reflected in an unbalanced ecosystem, fish mortality, odors, and other aesthetic nuisances. The problem of dissolved oxygen has been recognized for over a century, and attempts to model DO have been occurring for almost as long.

Streeter and Phelps did the pioneering work in the field of dissolved oxygen modeling in 1925. They developed the relationship between BOD and DO resources of the river, producing the classical DO sag model (Thomann and Mueller, 1987). Theriault in 1931 summarized the methods for estimating the model's parameters and Thomas accounted for settleable BOD in the DO sag equation. In 1956, Odum comprehensively described the interaction of photosynthesis, respiration and reaeration that cause diurnal DO variations in streams. O'Connor and DiToro incorporated these factors into a computer model for calculating DO concentrations in surface waters in 1970 (Ansa-Asare et al., 2000). Hann was one of the first researchers to apply digital computers to calculate waste assimilation capacity of a stream in terms of BOD.

The classic Streeter-Phelps equation can be modified to account for both distributed BOD loadings and distributed effects of SOD. Finally, the combined Streeter-Phelps model for point

and nonpoint (distributed) sources is given by Chapra (1997). Equation (1) shows the Streeter-

Phelps relationship with components to account for nitrification and SOD:

$$D = \frac{K_1 L_0}{K_2 - K_1}\left(e^{-K_1 t} - e^{-K_2 t}\right) + \frac{K_3 N_0}{K_2 - K_3}\left(e^{-K_3 t} - e^{-K_2 t}\right) + \frac{SOD}{K_2 H}(1 - e^{-K_2 t}) + D_0 e^{-K_2 t}$$

(1)

Where: D = dissolved oxygen deficit at time t, mg/l

L_0 = initial CBOD, mg/l

N_0 = initial NBOD, mg/l (NBOD = NH_3-N x 4.57)

D_0 = initial dissolved oxygen deficit, mg/l

K_1 = CBOD decay rate, 1/day

K_2 = reaeration rate, 1/day

K_3 = nitrification rate, 1/day

SOD=sediment oxygen demand, g O_2/ft^2/day

H=average stream depth, ft

t = time, days

This equation can be used for a realistic representation of the steady-state processes

taking place in the system as it incorporates both point and nonpoint sources of loading to a

stream. Recent concern over nonpoint source pollution has directed attention to distributed

sources that contribute flow. Both analytical and computer-based numerical approaches are being

applied to the issue of nonpoint sources pollution. Engineers use computerized models that

incorporate numerical methods for analyzing and providing alternative solutions involving

arbitrary geometries and flow conditions (Bravo, 1998). Moreover, some of the recent studies go

beyond describing the traditional processes of describing advection, dispersion and basic kinetics

to include the effects of other factors such as nutrient loading, nitrification, eutrophication,

sediment oxygen demand, and benthic oxygen demand. Such models give a more detailed and

realistic assessment of water quality in terms of DO, as they incorporate most of the sources and sinks of oxygen in streams.

II.1.3.1 Total Maximum Daily Loads

A Total Maximum Daily Load (TMDL) is defined as the maximum amount of a pollutant that a water body can receive without violating water quality standards, and an allocation of that amount to the pollutant's source (USEPA 1999a). It is based on the relationship between polluting sources and instream water quality conditions. A TMDL must take into account considerations of seasonal variability and provide for a margin of safety (MOS) that accounts for uncertainties in the way the pollutants are loaded into the system and future increase in pollutant loadings. To put it simplistically:

TMDL = WLAs + Las + Background + MOS

Where:

TMDL = Total Maximum Daily Load

WLAs = Waste Load Allocations for point sources

LAs = Load Allocations for nonpoint sources

Background = background concentration of pollutant in the system

MOS = Margin of Safety

Hann (1962) devised computer methods to determine the ultimate BOD loading which a stream can take without violating water quality standards. His work is an early precursor to the TMDL concept that took shape ten years later.

The history of TMDLs begins with the passage of the Clean Water Act in 1972. The CWA has focused on technology-based solutions for reducing the pollution of waters by prescribing Best Available Treatment (BAT) or Best Practicable Treatment (BPT) methods to

municipal and industrial dischargers. The CWA has led to improvements in water quality largely due to reduction in toxic and organic chemical loadings from point sources. The impetus of this approach was in the creation of permits under the National Pollutant Discharge Elimination System (NPDES).

Though the nation has enjoyed improvement in the quality of its rivers and streams, water quality goals have not been achieved. More than 40% of assessed waters do not meet the water quality standards the states, territories and tribes have established for them (Shortle 2001). The majority of the waters are affected by nonpoint sources – 60% of which is from agriculture. Impaired waters include 300,000 miles of river and shorelines, and approximately five million acres of lakes. The contamination, which is caused mostly by sediments, excess nutrients and microorganisms, is reducing the ability of the water resources to provide services to water users and inflicting large costs upon society. This is bringing the TMDL program into the center stage of the water policy debate.

The TMDL program in section 303(d) of the CWA requires states to:

1. Identify waters that do not meet water quality standards adopted by that state;

2. Prioritize these waters depending upon the severity of their pollution; and

3. Establish 'total maximum daily loads' for these waters taking into account the seasonal variability of water quality and account for a margin of safety to reflect the uncertainty involved in assessing discharges and water quality.

The basis for establishing the TMDL process was to provide for more stringent water quality based controls when water quality goals cannot be met using technology based control (Novotny, 1996). The States are required to submit the 303(d) list and subsequently the TMDL document. Failure to comply or gain approval will result in the EPA taking over the state

TMDL. The failure of states, territories, and authorized tribes to implement the TMDL program according to the CWA deadlines has led to citizen organizations bringing lawsuits against the EPA requiring the listing of impaired waters and the development of TMDLs. A string of successful citizen's suits forcing EPA and the states to fulfill their duties under 303 (d) has brought the program to life, shifting focus from cleaning up pipe discharges to addressing both point and non-point source pollution on a watershed basis (Birkeland 2000).

Though the TMDL process is primarily concerned with water quality, the process will continue to highlight the interrelationship between water quality and quantity issues. In a Resources for the Future study by Boyd (2002), several reasons for this are stated, including the following:

1. Changes in stream flow affect the transport of pollutants through a waterbody.

2. The amount of water taken or returned to a waterbody may affect the dilution of pollutants in that system.

3. The suitability of a waterbody as habitat for fish or other species is often determined by water supply.

In 1994 the Supreme Court ruled that because water withdrawals can completely eliminate a water use, and because maintaining the use of water is the whole point of water quality standards, any distinction between water quality and quantity is purely artificial. The assimilative capacity of a waterbody is the first consideration of a TMDL. Water quality standards are designed to protect the designated uses of water, and insofar as they cannot be attained because there is not sufficient water to support those uses, the CWA requires their restoration (Jefferson County PUD v. Washington Dept. of Ecology, 1994).

EPA's Guidance for Water Quality-Based Decisions: The TMDL Process (USEPA 1999a) points out that by identifying threatened good quality waters, states take a more proactive, 'pollution prevention' approach to water quality management for the following reasons:

1. Consistent with 40 CFR 130.7 (c) (1) (ii) which requires that TMDLs be established for all pollutants that prevent or are expected to prevent water quality standards from being achieved.

2. Encourages States to maintain and protect existing water quality

3. Easier and less costly in the long term to prevent impairments than retrofit controls to clean up pollution problems.

4. Meets EPA objectives to support the State's collection of data on impacted or threatened waters.

The reasons for taking this 'pollution prevention' approach to water quality management are consistent with reasons for considering the altering instream flows in order to maintain water quality.

II.1.3.2 TMDL strategies for DO

Lasting solutions to water quality problems are best achieved by considering all activities in a watershed. This means that both point and nonpoint sources of pollutant generation should be assessed. Specifically for DO, this means that all oxygen demanding sources from point and nonpoint sources should be assessed. Point sources include BOD, ammonia, nitrogen and high temperature discharges from municipal and industrial wastewater treatment plants. Nonpoint sources include nutrient and sediment runoff from agricultural lands, sediment wash-offs during storm events and bacteria loadings. Once the sources are assessed and quantified, a TMDL for

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each pollutant can be established based on the standard DO criteria of 5.0mg/L. Addition control

of both point and nonpoint sources can effect the desired pollutant load reduction (Novotny,

1996).

The EPA Use Attainability regulations suggest that water body improvements that could

remedy the cause of impairment should be considered (Novotny, 1996). As can be seen by

equation (1) there are several points at which engineering control can be utilized to improve the

DO. These points can be grouped as follows:

1. Point and non-point reduction source of CBOD and NBOD through reduction of
    effluent concentration and/or effluent flow.

2. Aeration of the effluent of a point source to improve initial value of DO.

3. Increase in river flow through low flow augmentation.

4. Instream reaeration by turbines and aerators.

5. Control of nutrients to reduce aquatic plants and resulting DO variations.

The majority of the strategies aim to enhance waste assimilation of a stream. Waste assimilative

capacity can be increased by flow augmentation (Hann, 1962). However, water quality

restoration continues to focus on two major management facets. One is the implementation of

nest available technology for point source and the other is the implementation of BMPs for

nonpoint sources. An alternative management practice could be low flow augmentation to dilute

the concentration of oxygen demanding pollutants and to increase the assimilative capacity of

DO for the water body. An objective of this thesis is to examine the economics of flow

augmentation versus load reductions in reaching water quality standards.

II.2 ECONOMICS OF INSTREAM FLOW

Water is used for many purposes, amongst which we may distinguish two major categories: (a) *in-stream uses,* including power generation, water transport, commercial fishing, recreational and ecological uses, and (b) *withdrawal uses,* which include irrigated agriculture, rural domestic and stock, mining, industrial and commercial, municipal and domestic uses. Increasingly, the importance of the instream value of water is being appreciated. Water resources yield important ecological values, the existence of which do not require human contact with the resource. This section will briefly discuss the concepts of economic efficiency, opportunity cost, and externalities in the context of instream flows. We will then review the relevant literature related to the valuation of instream flow, and different ways of conceptualizing the costs of augmenting flow.

II.2.1 Economic Efficiency

The economic efficiency occurs when marginal benefits are equal across uses. When applied to instream flows, this means an efficient outcome will result if the last acre foot of water produces a benefit exactly equal to its opportunity cost. In reality, the efficiency test is not typically used in evaluating instream flow. Among the reasons for this is the difficulty associated with realizing the economic value of water and quantifying the benefits.

II.2.2 Opportunity Cost

Water withdrawal has an opportunity cost elsewhere in the basin. The opportunity cost of allocating water to one sector is the value of the best alternative use given up. This cost addresses the fact that by consuming water, the user is depriving another user of the water. If that other user has a higher value for the water, then there are some opportunity costs experienced by society due to this misallocation of resources. The opportunity cost of water is zero if there is no alternative use. Instream flow debates raise concerns about opportunity costs,

or the potentially lost value of water for other use.   Restricting the use of water supplies in a

basin to ensure that certain volumes remain in the stream may require potential water users to

pursue other, more expensive supply options. Under certain circumstances, instream flow rights

may even preclude certain types of magnitudes of future economic development in the area.

II.2.3 Benefits and Valuation

The amount of water in a stream or river affects the water body's ability to support its

designated use and to provide the goods and services humans and animals depend on. If an

aquatic ecosystem is healthy and functions, humans can enjoy an array of "ecosystem goods and

services", including the dilution of wastewater, natural purification of water, erosion control,

habitat for fish and wildlife, and recreation (Loomis et al 2000). Each of these services has an

associated value. These values, which are often difficult to quantify, have been the focus of many

studies. In this section, we will briefly review a few of the studies that focus on people's

willingness to pay to protect instream flows for habitat and recreation.

Some uses of water, particularly instream uses, are un-priced. It is theoretically possible

to quantify the economic benefit of preserving species, habitats and ecosystems through various

types of non-market valuation studies. In cases where there is no direct economic value of the

species or habitat, this type of economic value is known as existence value. The primary tool for

placing an economic or monetary value on existence is the survey-based technique of contingent

valuation, which is still debated in both economic and policy circles (Jeavons and Harvey 1999).

There have been numerous studies documenting people's willingness to pay for the protection of

instream flows and the activities associated with those flows, several of which are summarized

by Walsh Johnson and Mckean (1988) and presented in Table 2.2

Table 2.2.Willingness to Pay for Activities Associated with Instream Flows

Activity	Average Value per activity day		Number of studies evaluated
	1989 dollars	2002 dollars	
Camping	19.50	30.95	18
Picknicking	17.33	27.51	7
Swimming	22.97	36.46	11
Cold water fishing	30.62	48.60	35
Non-consumptive wildlife	22.20	35.24	7

Shelby et al. (1992) list many studies that report on the relationship of streamflow quantity to recreation quality or value. The values of instream flow may be higher than the values listed because the values of different activities are additive where participants in more than one activity can concurrently utilize increased flows. Almost all of these studies indicate that flow contributes positively to the recreation level up to some maximum flow level and then has diminishing returns.

Included in the study by Shelby et al. (1992) are studies by Daubert and Young (1981), Hansen and Hallam (1991), and Duffield et al. (1991). They all compared the value of instream flow with the values of withdrawal for irrigation. They found that, during low flow periods, the value of instream flow was often greater than the marginal value of withdrawal for irrigation. For example, Hansen and Hallam (1991) found that the average September instream flow, 105cfs, results in an aggregate marginal fishing return equal to $10.54 per acre foot and a marginal crop return equal to $7.22 per acre foot. Their study also highlights the fact the recreational value derived from flow augmentation varies greatly throughout the year. The marginal crop return per acre foot varies greatly from month to month.

II.2.4 Cost of Increasing Flows

Since increased streamflows amounts to reducing the amount of water consumed in irrigated agriculture, the cost of increasing streamflow will equal the value of using water in agriculture. If competitive water markets existed, then the value of water could be easily inferred from the prices at which farmers buy and sell it. However, water rights transfers between farmers do not exist in this area.

Enhanced flows during critical periods may be achieved by physical measures (allowing storage of surplus flows during certain months to be released when needed for instream flow objective); economic, incentive-based measures, or a combination. Potential incentive-based measures include programs to purchase water rights from farmers or retire agricultural lands. Reducing agricultural water use inflicts costs on the farmers in terms of lost crop production. Apart from this direct opportunity cost, such a program could inflict other opportunity costs that are less obvious and more difficult to quantify. Agricultural related businesses, such as machinery companies and processing companies can be affected, as can the sustainability of agricultural communities.

Especially when streams are not fully appropriated, opportunity costs include the potential precluded future offstream uses. In this case, opportunity costs of foregone economic development can be thought of as equal the discounted present value of the foregone economic benefits multiplied by the probability of the economic development taking place. When farmers are faced with having to reduce the amount of surface water they withdraw they can either use alternative sources or increase irrigation efficiency and conservation.

There are additional costs and benefits associated with augmenting flow during low flow periods or increasing the required minimum instream flow. The benefits of increasing instream flow include all reduced losses to downstream users. These losses include availability of water

for irrigation, fish habitat, and recreation. Approaches to measuring the costs of increasing the instream flows include the forgone benefits of water use, forgone revenues, cost of purchasing or leasing water, or alternative supplies.

The methods estimates presented in this section include direct evidence of the value of water in agriculture, as well as estimates based on economic studies. Information about market purchases of water right, or ones which involve water, will represent direct evidence of the value of water. Though estimating the agricultural value of water in a precise way is a complicated task given the site-specific nature, economic analyses can supply a number of different techniques to estimate the value of water. There is, then, some basis on which to make some broad estimates of the cost of augmenting stream flows by reducing water consumption in agriculture.

II.2.5.1 Measuring direct and foregone benefits of water

Most withdrawal and consumption of water occurs for use by profit making entities – in irrigated agriculture, and to a much lesser extent in industry. One method that can be used to estimate the agricultural value of water in its current use involves determining the contribution of irrigation water to net revenue from agriculture production. Economic benefits attributable to water are calculated as the net income remaining after all non water production costs are deducted from estimated revenues. The approach is well-suited for measuring the foregone agriculture revenues resulting from production losses due to reduction in available water supply. Physical characteristics of the land, irrigation application, delivery system, and crop yields under irrigated and non-irrigated conditions can be incorporated into the analysis to reflect on-farm conditions as accurately as possible. Available information about physical factors such as irrigation efficiency, crop requirements, return flows, and variation in local climate conditions

are used to determine the irrigation application requirements for a specific area. This approach ignores the various substitutions among crops and other inputs that farmers would likely make in anticipation of a water shortage. Particularly in the long run, farmers can adapt not only by changing the crop mix or reducing the area irrigated but also by substituting other inputs for water, such as capital (e.g., more efficient irrigation systems) or improved management (e.g., variations in the number of irrigation applications) (Young, 1986b; Gibbons, 1986).

A large, more detailed model, which represented farm-level adjustments among crops when irrigation is restricted, was developed by USDA (1996) to estimate the costs augmenting flows by reducing irrigation diversions. The estimates of the value of water for farming in these regions indicate that the costs of augmenting stream flow rises with increases in the size of the reductions in irrigation diversions. The study found that the costs per acre-foot of water were $20/af for a 14% reduction in baseline water diversions, $24/af for a 20% reduction, and $29 for a 29% reduction. This study took place on the Upper Snake River Basin.

II.2.5.2 Leasing and Selling of Water Rights

Many studies have examined the transfer of water from agricultural to municipal use. The interest in acquisition of water rights for instream uses has also emerged, with more than 2.3 million acre-feet of water being acquired from 1990-1998 for instream uses (Landry 1998). Environmental organizations such as the Nature Conservancy have participated in acquiring water for instream uses, and other non-profits have started water trusts in order to purchase water rights to improve flows for fish habitat or recreation (Landry 1998).

These activities involve actual transactions where those holding the right to water sell the right. The amount they are paid, then, is the direct cost. Indirect costs would include loss to the farmer and the surrounding economy. These prices may be equal to or greater than the value of

the water to the farmer. However, the price paid may exceed the value of the water, especially if

the transaction is noncompetitive. Leasing provides an avenue for realizing streamflow

restoration without permanently severing the farmer's right to use the water. Table 2.3

summarizes transactions in several different states.

Table 2.3. Purchases of Water for Instream Flow, Wetlands, and Lake Levels

Year	Purchaser	Quantity (acre-foor/year)	Price ($/acre-foot/year)	Type of transaction	Original Use	Proposed use
1987	Lander County, Nevada	3,000	9	Right	Irrigation	Maintain lake level for recreation
1987	Nature Conservancy	2,000		One-time	Irrigation option	Fish on N Poudre River in Colorado
1989	California Department of Fish and Game	30,000	5	One-time	Miscellaneous	Waterflow and fish on San Jouquin
1989	Califonia DF&G	1,500	10	25 years	Effluent	Duck ponds and riparian vegetation
1989	Nature Conservancy	400	14	Rights	Irrigation	Stillwater National Wildlife Refuge
1990	U.S. F&WS	67,442	7	One-time	Unused storage	Wildlife Refuge

II.2.5.3 Improving efficiency

Apart from reducing the amount of water being used for irrigation, farmers can increase

irrigation efficiency and conservation or use alternative sources of water. Application efficiency

is defined as the ratio of water being withdrawn from a particular resource to the water available

for target plant production. Irrigation conservation is then the process of improving the

application efficiency. There are many methods for making irrigation more efficient. Leveling

fields, using sewage effluent for irrigation, switching to lower water crops and appropriate

irrigation management can result in substantial water savings. Howe et al. (1980) found that

farmers used various conservation strategies to reduce water use during times of drought. These

included eliminating winter applications, consolidating water in efficient reservoirs, lining canals

and using more efficient water application.

Another analysis of the economics of alternative irrigation systems takes place in Kittitas

Valley, Washington (Hoffman and Willett 1999). Comparing the costs of different technologies

with the improved irrigation efficiencies, the found the cost per acre-foot of "saved" water

ranges from $40/af to $61/af.

Although irrigation efficiency is an important factor affecting instream flow, it should be

noted that when improved efficiency results in less water being diverted, it will also reduce

return flows. Also, improved irrigation efficiency does not necessarily mean more economic

efficiency or higher net revenues. It may be that the cost of improved irrigation technology is

high compared to the increased revenues that it will generate for the farmer.

The cost of increasing instream flow through improved irrigation efficiency will

primarily be the capital costs of the new irrigation technology and their associated maintenance

costs. The benefits to the farmer may include labor and energy savings, and the elimination of

costs associated with previous irrigation technology. A side benefit may be a reduction in

nonpoint pollution to the extent that larger return flows are responsible for agricultural runoff.

II.2.5.4 Replacement cost

The replacement cost method involves examining the potential supply side of the market.

Constraints on existing water resources in a basin as well as current uses of water with the study

area have important value impacts that must be considered. The value of a water right in an area is limited by the costs of obtaining water from an alternative source. For example, the cost of switching to ground water or building new reservoirs could be estimated. Studies done by Bash and Young (1994) assessed how farmers in the South Platte Basin might use alternative sources such as leased water and groundwater in times of water shortage. They found half the farmers in the sample increased their use of groundwater to some extent.

II.2.6 Additional Studies

Several studies have been completed that examine economic consequences of allocating water for environmental needs. Keplinger, et al. (1998) examined payments required to reduce agricultural diversions from the Edwards Aquifer in Texas to promote environmental needs. The tradeoffs between endangered fish species and irrigated agriculture for the western states were analyzed by Moore, et al. (1996). Turner and Perry (1997) examined least cost strategies for increasing instream flows for environmental benefits in the Deschutes River basin. Willies, et al. (1998) examined various ways to minimize economic damages to irrigated agriculture associated with setting up a contingent water contract to protect three species of endangered fish during critical low flow periods.

In 1996, Resources for the Future reported that water's value is highest in the Western US for its withdrawal uses. Their report includes nearly 500 water values in the continental U.S. from 41 studies performed under a wide range of economic conditions over the last several decades. In this report, they examined water's value for four in-stream uses; hydroelectric power generation, recreation and fish and wildlife habitat, navigation, and waste disposal and four withdrawal uses for irrigation, industrial processing, thermoelectric power generation, and domestic uses (Frederick et al., 1996). Using 1994 dollars/ acre-foot, they found the value of

water nationally to be highest in the drier, more water-scarce Rio Grande ($191) and lower

Colorado ($122) regions; and lowest in the Great Lakes ($7) and New England ($4) regions. By

use, water's value is highest for industrial processing ($282) and domestic uses ($194); and

lowest for recreation and fish wildlife habitat ($5) and waste disposal ($3).

In 1996, Briscoe examined the interaction of three critical values; the value of water, the

use cost, and the opportunity cost. He found that irrigation water for food grain has the lowest

value while urban water supply and irrigation of high value crops have the highest values

(Briscoe, 1996).

In 1997, Rogers and others estimated the value –in-use as the summation of values to

users of water, net benefits from return flows, net benefits from indirect uses, adjustment for

social objectives, and intrinsic value. Full cost of water should include the opportunity cost of

water as well as the environmental externalities. The full cost should present incentives for

pollution control. They concluded that raising water tariffs, levying effluent charges and

encouraging water markets can play significant roles in improving economic efficiency and

environmental sustainability of water use. For economic equilibrium the value of water, which

can be estimated from the value-in-use should be equal to the full cost of water. At that point,

social welfare is maximized (Rogers et al., 1997).

II.3 ECONOMICS OF WATER QUALITY

Water pollution is an externality produced by various types of economic activity that

causes welfare losses to groups of users. From another perspective, clean water is a public good

giving benefits to users but requiring costs for its attainment. A fundamental goal of

environmental policy is to get polluters to treat the external costs of pollution as a cost of

production, or internalize the costs. This goal can be accomplished by inducing or mandating

that polluters internalize the external costs that they impose on society through their pollution-related activities. Ideally, the resulting level of pollution control is an efficient solution, or one where the expected net economic benefits to society are maximized. Expected net economic benefits are defined as the private net benefits of production minus the expected economic damage costs of pollution.   In the case of water quality policy, Pareto efficiency is difficult to achieve because the damage functions are unknown or poorly understand and are difficult to quantify. The unique nature of nonpoint source pollution makes the efficiency goal particularly difficult to attain.

The economics of pollution control began with theory of externalities by Pigou (1932). According to Pigou, if a charge is imposed on an economic activity which causes an externality and the change in net at a rate of the marginal social cost of the externality, the level of the activity becomes Pareto efficient. Charges of this kind are now called Pigouvian or corrective taxes. Although Pigouvian taxes are the first best solution to pollution as an externality, it is sometimes difficult to measure the marginal social cost of pollution. Baumol and Oates (1971) proved that taxes on externality-generating activities could be the least cost method to achieve a predetermined standard for environmental quality if the taxes are set appropriately. Montgomery (1972) showed that a system of marketable permits could also be a least cost solution to attain a set of environmental standards. This is the theoretical foundation of the proposition that policies which use market mechanisms or economic incentives have a comparative advantage over uniform standards and command and control approaches to controlling pollution.

Griffin and Bromley (1982) characterized the nonpoint externality as one resulting from the difficulty of monitoring pollution sources and concluded that the following four strategies are allocatively efficient in a deterministic setting: economic incentives for controlling emission

discharges (e.g., taxes on emission); standards on emission discharges, economic incentives for controlling management practices (e.g. taxes on fertilizer); and standards for management practices. They emphasized the importance of the nonpoint production function, which relates production activities to nonpoint source pollution, and argued that if the nonpoint production function can be identified and the aforementioned incentives and standards are properly specified, then those four strategies are cost effective.

II.3.1 Cost – effective Analysis

Considering all the difficulties in assessing pollution and economic damages associated with it, efficiency may not be attainable in practice. Baumol and Oates (1988) suggested designing pollution control policy to meet an emissions or ambient pollution target when damages are unknown, as is the case with non-point water pollution. The goal of policy could then be to achieve the water quality target at the least cost. Pollution in a watershed comes from many different sources, many of which have different marginal costs of abatement. The most cost-effective solution is reached when those with a lower cost of abatement are targeted first. A policy can be considered cost effective if, on the basis life cycle costs analysis of completing alternatives, it is determined to have the lowest costs for a given amount of benefit, however that benefit may be defined. It is assumed that all emitters are profit maximizers and seek to minimize the costs of their pollution reduction. The analysis provides a tool for looking at the tradeoffs of using alternative strategies. It is important to note that there are legal, political and fairness issues that cannot be disregarded when making final decisions about which strategies to use to achieve water quality targets.

In order to perform a cost effectiveness analysis, it is necessary to identify pollutant sources, their respective potential pollutant reductions in a stream segment, and their costs for

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additional pollutant abatement. The next step would be to identify the combination of pollutant sources and respective pollutant reductions that minimize the total cost of achieving the reduction necessary at any determined critical stream segment in order to avoid exceeding the established TMDL for these regulated pollutants.

Water quality and hydrology models are used to derive transfer functions, which are found by solving the equations in the water quality model that relate the change in DO in a particular segment to an input of a pollutant in segment (Kneese 1975). In some cases, they simplify to a set of linear relationships. In matrix notation the system can be written as follows:

$$Ax = r$$

Where A is a matrix of transfer coefficients, x is a vector of pollutant discharges in mass per day, and r is a vector of DO concentrations in milligrams per liter at various specified locations. This is convenient as it lends itself to incorporation within linear economic models.

Assume that a stream segment consists of m segments and c_i represents the improvement in water quality required to meet a DO target in segment i. The water quality target can be met by changes of inputs to the watercourse from various combinations in management practices. Define $x = (x1, x2, ...xn)$ as the changes in mass of discharges (or concentration runoff) at the different locations (n). If A (m by n) is the matrix of transfer coefficients, then Ax yields the vector of DO changes corresponding to X.

Given S (the target improvement or standard), we have two restrictions, namely Ax>S and x>0. These are sets of linear constraints. Let d′ be a row vector where dj = unit cost of xj, j=1,…,n. This assumes linear cost functions. The problem then becomes:

$$\text{Min } d'x$$

$$\text{S.T. } Ax \geq S$$

$$x \geq 0$$

In other words, determine the optimal level of pollutant reduction among sources and the cost at each sources' employed increment of additional pollutant abatement technology, and solve for the constrained minimum total cost of achieving the target.

There is a substantial amount of empirical work on agricultural water pollution. Cost-effectiveness analyses for achieving TMDLs have been performed by Keplinger and Santhi (2002). Schleich (1996) and others also examine least-cost solutions for achieving water quality targets. An array of studies examines the cost effectiveness of BMPs. Those studies include Crowder and Young (1985). In some cases the marginal cost for all pollutant sources is not available. One way to estimate costs was demonstrated by Schleich, White and Stephenson (1996). For each of the five categories of pollutants they considered (municipal wastewater treatment plants, industry, urban runoff, construction runoff, and agriculture runoff), they estimated an annual per kilogram treatment. For example, for municipal treatment plants they identified the level of treatment technology at each plant and then estimated the cost and load reduction resulting from the upgrade to the next most effective treatment technology. For construction and urban runoff, they estimated the costs by comparing the cost and effectiveness of different techniques used to reduce certain pollutants in other reports and studies.

Wu et al. (1995) developed and applied an analytic framework for evaluating policy options for controlling water quality. The study area is the southern high plains overlying the Ogallala aquifer in Texas. Policy alternatives for controlling nitrate water pollution consisted of restrictions on per-acre nitrogen use, taxes on nitrogen use and irrigation water use, and incentives to convert conventional gravity irrigation systems to modern irrigation systems. They constructed a mathematical programming model for agricultural production combined with a

crop-growth/chemical-transport simulation model. They concluded that producers clearly preferred restrictions on per-acre nitrogen use to nitrogen use taxes or irrigations water taxes, because farm income is reduced less under the regulation than under the two tax policies. They also concluded that from society's point of view; however, nitrogen use taxes may be more desirable than nitrogen use restriction.

II.3.2 Studies Related to Dissolved Oxygen

There are several studies directly related to dissolved oxygen. Sobel (1965) proposed a linear programming model for dissolved oxygen problems. This study focuses on characterizing the problem of improving water quality under four different programs: (1) improvement at minimum cost; (2) uniform treatment policy; (3) maximizing the benefit-cost ratio, and (4) improvement in a stochastic environment. Different models were used for each of the programs. This study benefited from the suggestions for the objectives of improvement at minimum cost, which used a linear cost function with linear constraints plus an upper bound on the DO improvement.

Loucks (1967) proposed two deterministic LP models to determine the least-cost combination of wastewater treatment required to meet any level of stream dissolved oxygen standards. They used an approximation of the non-linear oxygen-sag equation to define the DO concentration constraints in their models.

In a study that attempted to minimize total operating costs of BOD removal by determining the degree of removal required at each treatment facility without violating DO standards, Lohani and Thanh (1978) adopted a chance-constrained programming framework to analyze the DO concentration problem. The cost function was piecewise linear.

Other approaches for addressing water quality problems include dynamic programming, utility maximization framework, and simulation analysis. Liebman and Lynn (1966) used a discrete dynamic programming approach to investigate DO problems. The main objective of this study was to determine thee amount of BOD removal for each waste discharger such that DO concentration standards would be met at minimum total cost of waste treatment. The empirical analysis was conducted with deterministic stream flow. The model was estimated several times with different stream flows in attempt to deal with variable flows.

Upton (1970) uses an expected utility maximization framework for water quality management. The model assumes there is a central authority managing stream quality for the benefit of the society. The authority uses both flow augmentation and adequate treatment as tools to ensure that the water quality standard is not violated when the flow drops below a critical minimum. Society derives utility from the usage of good quality water from the stream but it incurs costs for waste treatment in the stream and for maintaining adequate water in the reservoir for lean season flow augmentation. The objective function takes the form of maximization of expected utility. The study shows that under certain assumption this problem could be transformed to an equivalent non-stochastic problem. This requires selecting a critical level of stream flow different that mean low flow and managing water quality as if the actual flow were non-stochastic and equal to that critical level.

Studies that examine using upstream water to improve downstream water quality include one performed by Brown et al. (1990). The study addressed river flow uncertainty by estimating mean water use under four different levels of flow and estimated the economic value of increased streamflow and improved water quality resulting from increased runoff from timber harvest to water uses in the Basin. Scherer (1977) developed a stream model that incorporated

upstream water uses and allowed dilution as a mitigation alternative. Oamek (1998) studied the

economics of interbasin water transfers and the effects of the transfers on erosion and salinity in

the Colorado River Basin. Booker (1990) developed a model to study the economics of both

interstate and intrastate water transfers within California. Using mass balance constraints to

account for changes in salinity, the effect of water transfer on salinity was examined. Historic

flows between 1960 and 1969 were selected as base flows to represent average conditions.

CHAPTER III

MATERIALS AND METHODOLOGY

This section is divided into two parts. The first part consists of data and methodology used to model the hydrology and water quality of the Spring Creek Watershed, both Spring Creek and Dry Creek sub-watersheds. The second part consists of the data and methods used to estimate the costs of different management practices and to solve the cost minimization problem.

III. 1. SPRING AND DRY CREEK WATERSHED ANALYSIS

III.1.1 Watershed Description

The models used in this study are tested using data obtained from Spring Creek and Dry Creek sub-watersheds located in the Spring Creek watershed in the lower part of the Flint River Basin (FRB). The eight-digit hydrologic unit identification number for Spring Creek watershed is 0310010. The sub-watersheds lie within Early, Miller, Mitchell, and Calhoun counties, all in southwest Georgia. Dry Creek flows approximately 12 miles, all of which are listed as impaired. It has a drainage area of 104.5 square miles. Spring Creek is 34 miles in length, 22 of which are listed as impaired, and has a drainage area of 366.23 square miles.

Historical climate data are available from weather station data (National Oceanic and Atmospheric Administration). Southwest Georgia has a humid climate. The average rainfall in this area is 125 cm, and is evenly distributed. The average annual temperature is 20 degrees C with approximately 260 freeze-free days.

The forestland is made of overstory species round in several community types. These include southern hardwood forest, oak hickory, and Appalachian cove. Dominant overstory species include sweetgum, oaks, tulip polar, black gum, and American Basswood.

III.1.2 Water Quality Standards for the Spring Creek Watershed

TMDLs are developed to meet applicable water quality standards. These standards may include numeric water quality standards, narrative standards for the support of designated uses, and other associated indicators of support of beneficial uses. The numeric target identifies the specific goals or endpoints for the TMDL that equate to attainment of the water quality standard. General narrative standards and numerical criteria for dissolved oxygen (DO) have been defined for the waters of the State of Georgia. The applicable numeric water quality standards are a daily average of 5.0 mg/L and no less than 4.0 mg/l at all times for waters supporting warm water species of fish (GADNR 2000). For the purposes of our study, a minimum DO level 5.0 mg/L will be implemented, allowing for an implicit margin of safety resulting from the conservative assumptions used in modeling. The reductions in the ultimate carbonaceous biochemical oxygen demand (CBODu) and nutrient runoff concentrations are targeted to runoff concentrations that, in concert with the nitrification of ammonia, will not deplete the DO concentration below this level as a result of the decaying process.

III.1.3 Data

This thesis uses historical and current water quality and quantity data from the USGS monitoring stations in the Spring Creek Watershed. In July 2000, a study done by Johnson et al (2001) sampled two places on Spring Creek. This data was also used for the analysis. Another source of data is EPA's Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) website. BASINS is a multi-purpose environmental analysis system that integrates a geographical information system, national watershed data, and environmental assessment and modeling tools. Data is available for downloads for each 8-digit watershed. The reach file within this data was created by EPA and provides information about stream length, slope,

background water quality conditions, and other stream characteristics. The BASINS data also provides information about the location of point sources, water quality stations, soil types, and other relevant information.

Listed below are previous water quality data and studies in the watersheds.

1. Data collected by USGS for the Flint River DO TMDL (GA EPD 2002).

2. Data collected by the USGS from the Iron City station from 5/17/95 to 11/30/95 and 5/10/2000 to 10/25/2000 (USGS 2003).

3. Flow data collected by USGS from the Iron City station for January 1995 to November 2000 (daily) (USGS 2003).

4. Data collected for the Johnson study from two stations along Spring Creek for July to October 2000 (Johnson et al 2001).

5. Data from EPA's BASINS (USEPA 2003).

The model requires inputs for flow in the headwaters, incremental inflow, and tributaries. Incremental inflow refers to all natural stream flow not considered by the other two sources of natural flow - headwaters and tributaries. It encompasses flows from groundwater recharge, small tributaries not considered in the model, and nonpoint source runoff.

The original flow input used was the 7Q10 value. The 7Q10 refers to the lowest consecutive seven day stream flow that is likely to occur in a ten year period under natural conditions. The 7Q10 is typically assumed to be the critical condition flow for the summer season. A low-flow analysis, which was performed by EPD, examined the adjacent long-term USGS gauges to develop a representative flow during June 2000 (GA EPD 2002). The USGS gauge that was selected to represent a low flow for modeling purposes for both of the modeled segments was 02357000 near Iron City. The Bingham equation was used to determine the 7Q10

for the modeled watersheds. The Bingham Equation can be found on page 3 of Low-Flow

Characteristics of Alabama Streams, Bulletin 117 (Alabama GS 1998). The method uses

drainage area of the watershed, drainage area of the USGS station, and the 7Q10 of the USGS

station to determine a 7Q10 for the modeled watersheds. Flow levels for Dry Creek, which

were taken during the USGS survey for the Flint River DO TMDL, were available for the critical

conditions period. The low-flow analysis summary is listed in table 3.1 below:

Table 3.1. Low Flow Analysis for Spring Creek and Dry Creek

	Spring Creek	Dry Creek
Drainage Area to bottom of segment	366.23	104.5
Nearest USGS gage	0235700	0235700
Drainage Area of Gage (sq miles)	485	485
7Q10 of USGS Gage	11.84	11.84
Productivity Factor of 7Q10 (cfs/sq mile)	0.0244	0.0244
Weighted 7Q10 for (cfs)	8.941	2.550

As can be seen from the flow data (appendix A), the actual flow rates in the year 2000

fell far below the 7Q10. The model was also run using a median value occurring in the late

summer to early fall in both the year 2000 and 1995. The actual flow data for the year of the

impairment was used to calibrate the model.

III.1.4 Developing the Model

In order to evaluate the linkage between pollutant sources and water quality, a water

quality model was used. Since the observed DO concentrations were driven by low flows and

high temperatures, which occurred over several summer months, a steady state modeling

approach was adopted as appropriate for the analysis. The model is referred to as the

Spreadsheet Water Quality Model (SWQM). It is an adaptation of the Streeter-Phelps dissolved

oxygen deficit equation with modifications to account for the oxygen demand resulting from nitrification of ammonia and oxygen demand found in water body sediment. It lends itself to being developed with limited data, which is the present situation for this waterbody. It also has the ability to handle tributary inputs and both point and non-point source inputs. A more detailed description of the model can be found in Appendix C.

In order to apply the model to the stream segments in this study, a few adaptations were necessary. The SWQM categorized all crop land as one land use category with the same runoff concentrations. Because we had access to more data about specific land uses and current practices, we further subdivided these subcategories. The second adaptation we made was to create a parameter that allows us to test the effects of whether or not an agricultural BMP was implemented in the cropland in the acres along each of the segments. There is one of these parameters for each segment, expressed as a percentage of the row crops being implemented with the TMDL. Furthermore, this allows us to account for spatial variability, and allows us to account for the implementation of the BMP on all types of runoff included in the model. The third adaptation was the addition of a cost worksheet. This allows us to transfer the amount of acres undergoing BMP implementation, or the amount of flow augmentation, and multiply it by cost per unit. Finally, adding Optquest to the excel program allows us to vary one or more factor at the same time while keeping most of the factors constant.

In summary, several excel spreadsheets are used to calculate runoff volumes and loads from identified point and nonpoint sources. These calculations rely on both land use data and watershed information such as depth, drainage area, and velocity. The advantage of the spreadsheet is that a mixture of land uses (with varying concentrations) may easily be simulated, and overall load and flow-weighted concentration obtained from the study area (Walker et al,

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1989). Equation (1) shows the Streeter-Phelps relationship with the additional components to account for nitrification and sediment oxygen demand (SOD):

$$D = \frac{K_1 L_0}{K_2 - K_1}\left(e^{-K_1 t} - e^{-K_2 t}\right) + \frac{K_3 N_0}{K_2 - K_3}\left(e^{-K_3 t} - e^{-K_2 t}\right) + \frac{SOD}{K_2 H}(1 - e^{-K_2 t}) + D_0 e^{-K_2 t}$$

(1)

Where: D = dissolved oxygen deficit at time t, mg/l

L_0 = initial CBOD, mg/l

N_0 = initial NBOD, mg/l (NBOD = NH_3- N x 4.57)

D_0 = initial dissolved oxygen deficit, mg/l

K_1 = CBOD decay rate, 1/day

K_2 = reaeration rate, 1/day

K_3 = nitrification rate, 1/day

SOD = sediment oxygen demand, g O2/ft2/day

H = average stream depth, ft

T = time, days

This equation shows the net effect on dissolved oxygen concentration of the simultaneous processes of deoxygenating through the decay of carbonaceous organic matter, nitrification of ammonia, SOD and reaeration.

III.1.4.2 Assumptions

The following assumptions were used in the model:

- DO concentrations for incremental flow were assumed to be 70% of the saturated value at the given temperature. This is 15% lower than what is

normally assumed and creates an additional implicit margin of safety in the TMDL model.

- The land use percentages of the watersheds were assumed to be the same for the headwaters, tributaries, and incremental inflow.

- Background conditions, which are in the range of 2-3 mg/L CBODu, 0.2-1 mg/l NH3, and 1-2 mg/l TON, were assumed for forest runoff. This is because forests typically have a good filtering mechanism with respect to runoff.

- Ratios for CBODu/NH3ODu and CBODu/TONODu were calculated using water quality data for the waterbody. The CBODu/BOD5 ratio used was 2.5.

- NH3ODu is equal to 4.57 times the ammonia nitrogen concentration. This is because the stoichiometric requirement for oxygen in the oxidation of ammonia is 4.57 mg of O_2 per mg of NH_4^+-N oxidized.

III.1.4.3 SOD representation

SOD can be an important part of the oxygen demand in streams. While no field measurements were found for the Flint River Basin, there were several SOD measurements from the South 4 Basins, ranging from 0.9 to 1.9 g/m/day (EPD, 2000). The EPD found the average value to be 1.35 g/m/day for models that had mixed land uses and point sources activity. When the models were run with no point sources and all forested lands, SOD averaged 1.25g/m/day. The anthropogenic nonpoint source contributions are then accounted for in the 0.1 g/m/day.

III.1.4.4 Calibration Conditions

By calibration, we mean that some parameters are specified *a priori* while others are adjusted so that the model replicates available data for the baseline scenario. For the most part, the specified parameters are drawn from a literature that reports a range of values. In consequence, many parameters of the model are not known with certainty.

The calibration period used was defined by the lowest, steady flow period, the lowest DO concentrations, and the highest BOD concentrations. These conditions occurred during June - July of 2000. The stream conditions (i.e., D.O., temperature) during this period were incorporated into the calibrated model. Following the methodology used by EPD and EPA to develop DO TMDLs in Georgia (EPD 2002), SOD was set at 1.35 g/m/day. For the point sources, the NPDES permit levels for the critical period were used for calibration. Concentration runoff from the various land uses and reaeration rates were adjusted in order to calibrate the model.

III.1.4.5 Model inputs

The model is comprised of seventeen worksheets, five of which require the input of parameters – WQ MODEL, Land Use, Chronic NH3 TOX, Acute NH3 TOX, and the Cost worksheet. The input requirements of the WQ MODEL can be further divided into the following sections: reaction rates, section characteristics, headwaters conditions, tributary conditions, incremental inflow conditions, and effluent conditions. The equations used to derive the reaction rates are covered in Appendix C.

III.1.4.5.1 Section Characteristics:

The model reaches for Spring and Dry Creeks consist of 6 and 4 segments, respectively. Dividing the reaches allows the model to account for changing physical features of the streams. SWQM requires reach physiography to be defined in order to do the in-stream mass balance and

routing calculations. The data required includes depth and elevations upstream and downstream. This information was obtained from the National Hydrography Set, which was displayed in Arcview.

III.1.4.5.2 Headwater Conditions

The headwater conditions are the physical and chemical stream characteristics immediately upstream of the reach to be modeled. Inflow rates for the headwaters were based on the USGS daily flow data for Spring Creek at the station near Iron City. Based on the flow per square mile of drainage area for the gage, flows at the upstream and downstream ends of the sub segments were calculated using published drainage areas for the watersheds.

Temperature input is based on historical weather data, and is assumed to be the 90th percentile seasonal temperature for the reach being modeled. Long term temperature data from the gage at Iron City was used to calculate a 90th percentile summer temperature of 25 degrees. Because the segments have a year round standard for DO, and DO violations have only occurred in summer months, a winter projection simulation was not performed. Headwaters water quality parameters include CBODu, NH3-N, and total organic nitrogen, which are calculated by the model.

III.1.4.5.3 Incremental Inflow:

This is the total flow at the downstream end of the sub segment minus the sum of all the headwater and tributary inflows. This total amount of incremental inflow was assumed to be uniformly distributed along the entire length of the channels being modeled; therefore, the inflow for each reach was proportional to its length. The DO concentration for incremental flow was set at 70% of the saturation concentration at the given temperature.

III.1.4.5.4 Tributary conditions:

The methodology behind and parameters for the tributaries are the same as headwaters and incremental inflow. While Dry Creek did not have any significant tributaries, Spring Creek has four. However, the critical conditions for the model are when the flows are so low that the tributaries do not contribute a significant amount. Dry Creek was modeled as a tributary to Spring Creek. This is partly due to the fact that Blakely Pond A and Blakely WPCP are connected to Spring Creek by Dry Creek.

III.1.4.5.5 Point Sources

There are several NPDES permits that discharge into or upstream from Spring Creek and Dry Creek. Table 3.2 contains the June 2000 permit limits that were used for model development. They include Arlington Pond #1, Colquitt WPCP, Blakely WPCP, Blakely Pond A, and Blakely Pond B. All of these are identified as sewerage systems. Through conversations with the plant engineers, we obtained information on the concentrations and amount or the actual monthly average discharge, which were both lower than the permit limits allow. The actual permit levels are used in order to allow for conservative assumptions for the margin of safety and to allow for growth.

Table 3.2. Point Sources for Spring Creek Watershed

NPDES permit	Facility Name	County	Flow (mgd)	DO (mg/L)	BOD	NH$_3$ (mg/L)
GA0031968	Blakely Pond A	Early	0.12	NP	30	NP
GA0031976	Blakely Pond B	Early	0.12	NP	30	NP
GA0026204	Arlington Pond #1	Early	0.10	NP	30	NP
GA0050075	Arlington on Pond #2	Calhoun	0.06	NP	30	NP
GA0047252	Colquitt WPCP	Miller	0.40	5.0 (min)	30	15

The locations of these point sources were determined using GIS software and can be seen in the map at Appendix B. In the case of Dry Creek, all three of the relevant point sources are located upstream of the source. Because the model will only allow one point source per stream section, all three of these were combined and a loading factor was calculated for them. In order to include the Blakely ponds in the Spring Creek model, a branch was added above the Dry Creek tributary through which the effluent flows to get to Spring Creek. The CBODu concentration was set to the BOD5 limit times 2.5 to convert to CBODu. The organic nitrogen was assumed to be half of the ammonia nitrogen based on typical values for mechanical treatment systems.

III.1.4.5.6 Nonpoint Sources

Land use percentages and mean runoff concentrations are used to calculate the CBODu, TON, and NH3-N concentration throughout the streams. The model also allows the user to assign organic loadings on the basis of land use. The model assigns differing pollutant concentrations to flow from headwaters, tributaries, and incremental inflow (all natural stream flow not considered by the other two sources of natural flow) according to the major land use percentages in the watershed. The selected stream reach is then divided into individual segments in order to account for changing physical features of the stream. These would include the addition of flow and pollutants from tributaries, incremental inflow, and point sources, changes in stream slope, velocity, and any of the reaction rates.

The land use characteristics of Spring Creek were determined using data from Georgia's Multiple Resolution Land Coverage (MRLC), which was produced from Land sat Thematic Mapper digital images developed in 1995. Table 3.3 lists the land use distribution of the watersheds.

Table 3.3. Land Use Distribution of Spring Creek Watershed

Stream	Total Contributing Area (Acres)	Row Crops (%)	Pasture (%)	Forest (%)	Wetland (%)	Built-Up Impervious (%)	Built-Up Pervious (%)
Spring Creek	234,388	38.3%	12.4%	29.6%	15.2%	0.6%	3.9%
Dry Creek	66,861	33.2%	13.5%	29.0%	18.6%	1.3%	4.4%

Both Spring Creek and Dry Creek watersheds have mixed land use, as can be seen on the map in Appendix B. The two major land uses are forest and cropland, which together account for almost 70% of watersheds. Compared to other land uses organic enrichment from forested land is normally considered to be small. This is because forested land tends to serve as a filter of pollution originating within its drainage areas. However, organic loading can originate from forested areas due to the presence of wild animals such as deer, raccoons, turkeys, waterfowl, etc. Control of these sources is usually limited to land management best management practices (BMPs) and may be impracticable in most cases. In contrast to forested land, agricultural land can be a major source of organic loading.   To better understand the potential loadings from the cropland, data was obtained from the United States Department of Agriculture's National Agriculture Statistics Service (AGSTATS).

Since AGSTATS gives the crop acreage in the counties, not subwatersheds, the percentages of the various row crops for all the counties in the watershed was calculated. Those percentages were used to determine the number of acres planted in each row crop, and the percent irrigated. Spring and Dry Creek, which encompass the same counties, were assumed to have the same percent distribution. Table 3.4 shows the distribution of row crops.

Table 3.4 Distribution of Row Crops in the Spring Creek Basin

Crop	% of total acres	Acres Planted		% irrigated	Acres Irrigated	
		Spring	Dry		Spring	Dry
Peanuts	32%	28,726	9,192	41	11,777.66	3,768.72
Cotton	50%	44,885	11,099	37	16,607.45	4,106.63
Corn	17%	15,261	3,773	55	8,393.55	2,075.15
Soybeans	1%	897	222	10	90	22

III.1.4.5.7 Low-flow augmentation to rectify the problems:

In addition to leaving more water in rivers, streams, and wetlands for fish and wildlife, as well as reducing the number of fish killed directly by water diversions, water conservation also improves water quality by reducing the volume of surface runoff and limiting irrigation-induced erosion and sediment loads. For the purposes of our study, we do not consider the value of these added benefits in the models.

One of the objectives of this project was to evaluate the effects of low flow augmentation on the DO concentration of impaired streams. As stated earlier, most of the DO problems in streams occur during low flow conditions when the water is stagnant, the concentration of nutrients, algae and BOD is high and reaeration rate is low. The June 2000 observed DO is an example of such conditions. It shows that the DO criterion was violated at several instance with concentrations as low as 0.1mg/l. Such low flow conditions usually occur during dry periods when there is no rainfall and the temperatures are hot. It should be noted that Georgia experienced a record drought during this time.

For the present study, water augmentation was modeled as a decrease in surface water irrigation withdrawals compared to the baseline scenario. The amount of water used for irrigation was estimated and derived from data available from EPD concerning the location and type of

irrigation. The agriculture demands were computed by multiplying estimated acreage of each

type of irrigated crop with the applicable application rate during the critical time of year. The

GIS-based Arcview program was used to determine where this water would be available for flow

augmentation. Flow augmentation can be included in the model by adding a 'point source' to the

stream of interest or by adding incremental inflow. The exact location of the source can be

specified by calculating the number of miles from the most downstream point on the stream of

interest and adding the source to that segment. The quality of the water was assumed to be the

same as the headwater. After calibrating the model with calibration conditions discussed earlier,

different scenarios were run using the calibrated model as a base case.

Flow augmentation scenario analysis

1. Control – Simulated flow and DO without any external flow source. This is the 'as is'
   scenario.

2. Scenario 1 – Flow augmented by adding 1cfs.

3. Scenario 2 – Flow augmented by adding 5 cfs.

4. Scenario 3 – Flow augmented by adding 10 cfs.

While this analysis showed that increasing flow did increase DO concentrations, the results

varied greatly in terms of where and what combination or flow was added. Therefore, we

adapted the model to have the capacity to change more than one variable at a time. Running the

model with OptQuest allows us to determine the amount of flow augmentation necessary within

each segment to achieve the water quality targets while minimizing the cost of doing so.

III. 2 COST AND EFFECTIVENESS ESTIMATES OF MANAGEMENT PRACTICES

Cost estimates for management practices will vary from site to site and year to year. It is

therefore not the intention of this study to provide exact cost figures, but to give a simple

methodology used to arrive at reasonable estimates. Both of the study sites are largely agricultural. Therefore, the BMPs analyzed are agricultural. Both the cost and efficiency figures are based on values from the literature, and, to the extent possible, are applied to the site-specific conditions.

III.2.1 Agricultural BMPS

Managing agricultural nonpoint sources of pollution presents a series of complex control problems. The localized nature of crop production conditions, for example, makes adoption of a given BMP over a wide area impractical in many cases. Slight changes in field conditions can have substantial effects on both cost of implementation and relative effectiveness of various agricultural BMPs (Schleich 1996). A summary of the range of cost and effectiveness of various agricultural BMPs is presented in table 3.5. Because Georgia is encouraging filter strips through several programs, as well as conservation tillage, they will be the focus of this analysis.

Table 3.5. Agricultural BMP Summary

BMP	Nutrient Reduction	Sediment Reduction	Cost
Conservation Tillage	15-85%	15-90%	$9-$26[1] per acre per year
Contour Farming	Similar to sediment	30-50%	$4[1] per acre per year
Filter Strips	62%	74%	$20-25 per acre per year[2]
Grassed Waterways	30%[3]	80%[3]	$447 per acre per year[4]
Streamed Fencing	Medium to High	40%	$0.15 per linear foot per year[4]
Nutrient Management Strategies	Potentially high	Potentially high	$4-13 per acre per year

1 Producer costs, rounded to nearest dollar, as calculated in Natural Resources Conservation Service, 1995.
2This value is estimated using cost estimate from Baun (1997) and amortizing to an annual value. An 8% interest rate and the practice life expectancy were used. Life expectancies were according to NRCS guidelines.
3Estimated from urban grassed waterway data.
 4This is an amortized value from cost estimates from the Dane County Land Conservation Department Rate Sheet. Amortization used 8% interest and the practice life expectancy. Life expectancies were according to NRCS guidelines.

Cost estimates are comprised of investment cost, annualized over the expected life of the BMP, annual (operation and maintenance) costs, and any other changes in farm net income resulting from changes in yields, cropped acreage, etc. Effectiveness range estimates, based on published literature, are given for reductions in nutrients and sediment. The effectiveness ranges are presented as a percentage reduction from the original loadings or mean runoff concentrations. These are presented in table 3.6. Of course, losses will deviate from these ranges in some instances.

Table 3.6. Estimated Loads from Agriculture Lands by Counties in Spring Creek Basin

County	Acres with nutrient application	Sediment (tons)	Sediment (ppm)	Nitrogen (tons)	Nitrogen (ppm)	P (tons)	P (ppm)
Early	123,292	146,0088	32.6	391	0.13	153	0.051
Miller	94,148	58,928	22.0	180	0.08	66	0.029
Seminole	74,143	51,918	24.1	148	0.08	56	0.031

Note. Mass estimates are based on whole county. Concentration estimates are average event runoff.

III.2.1.2 Conservation tillage

The common element in various definitions of conservation tillage is the presence of crop residues on the soil surface to reduce water and wind erosion. The plant residue also serves to increase retention of soil moisture. Conservation tillage systems disturb or invert the soil less than conventional tillage. Switching from conventional to conservation tillage has several implications for water quality in an agriculture watershed. It reduces the amount of sediment eroded from a given field and allows for the interception or filtering of sediment-bound nutrients carried on to the field from neighboring locations through surface runoff.

Costs

Many factors influence the machinery equipment investment costs for alternative tillage systems. It has been argued (see for example, Defiance Soil and Water Conservation District, 1984; Griffith et al., 1977) that the annualized net investment cost stemming from the purchase of new machinery (i.e. chisel plow, drill) is effectively zero since the machinery needed for each type of tillage is comparable in cost. In fact, net long-run investment costs may be negative for conservation tillage. However, with the conservation tillage, the moldboard plow, multiple diskings, and multiple chisel plowings are replaced with field cultivators, single diskings, and chisel plowings. There will be costs associated with the change in equipment.

Operating costs and yields are affected by changes in tillage systems. Machinery labor and expenses for fuel, oil, and repairs decrease as tillage decreases. The inverse may be true with herbicide costs if additional herbicides are required as tillage decreases. In general, operation and maintenance costs decrease from conventional tillage by about 3% (or $3 to $7.50 per acre) for the various forms of 'reduced tillage' and about 7% ($5-15 per acre) with no-till. Yields can increase, decrease, or remain constant depending on soil series, climatic conditions, management techniques and timing. Yield variances of 10% for both corn and soybeans have been reported in the literature. Nine years of data from a Tennessee experiment shows an average yield of 36 bushels of soybeans per conventionally tilled acre compared to 32 bushels of a no-till (Hayes 1999).

There are also learning costs associated with changing tillage systems. The adoption of a new system can add costs in the form of mistakes made or precautionary measures taken, but time and experience with new tillage systems that reduce soil erosion should improve the long-term yield potential and profitability of the farm. There are many studies examining the cost of conservation tillage, as can be seen in table 3.7.

Table 3.7. Summary of Cost Studies for Conservation Tillage

Location	Year	Constant Costs ($/Acre)	Reference
Maryland	1987	$21.99	Smolen and Humenik (1989)
Michigan	1897	$8.25	Smolen and Humenik (1989)
Wisconsin	1981	$42.65	Smolen and Humenik (1989)
Minnesota	1987	$35.79	Smolen and Humenik (1989)
Virginia	1987	$35.79	Smolen and Humenik (1989)
North Carolina	1987	$17.12	NCAES, 1982
Alabama	1982	$26.84	Russell and Christensen, 1984
Florida	1982	$55.09	Russell and Christensen, 1984
Georgia	1982	$46.61	Russell and Christensen, 1984
North Carolina	1982	$16.95	Russell and Christensen, 1984
South Carolina	1982	$38.14	Russell and Christensen, 1984
Virginia	1982	$22.60	Russell and Christensen, 1984

III.2.2.2 Filter Strips

Organizations such as the Georgia Natural Resources Conservation Service and the Georgia Department of Natural Resources are encouraging landowners to establish riparian buffers in order to improve Georgia's surface waters. This can be seen in a variety of programs, including the Conservation Reserve Program, the Georgia Buffer Initiative, the Georgia Greenspace Program, and the Conservation Reserve Enhancement Program. A vegetative filter strip is an area along a ditch, gully, stream, pond, lake, or sink hole that is covered permanently by vegetation such as grass, hay, or timber. The vegetation reduces or removes sediments,

chemicals, nutrients, and organic materials carried in runoff (Chesapeake Bay Program 1995). While filter strips can be costly to install, in some circumstances they may provide economic benefits to offset the lost profits from growing other crops. We calculated the total cost of vegetative filter strips, including installation and maintenance costs, using a spreadsheet template adapted from a template developed by Yeh and Sohngen (1999). Because the costs of filter strips occur at different times, costs need to be expressed as present values. The spreadsheet template does the discounting automatically.

The benefits depend on the type of filter strip. Grass filter strips are the focus of this part of the analysis. A grass filter consists of the land along a stream or ditch is taken completely out of conventional agricultural production and converted to a mixture of grass and legume. Grass filter strips do not provide future profits to landowners, although they may provide other environmental benefits on or off the farm. These benefits are not included in the template used in this analysis, although they may be important in many situations.

Costs

Costs can be broken down into installation, operation and maintenance, and opportunity costs. Cost categories, which are presented in more detail in table 3.8, included in the spreadsheet are:

1. Seedling Purchase Costs – Seedling purchase that occurs at the beginning of the project.

2. Fertilizer Costs – Application of fertilizer is assumed to occur at the start of the project.

3. Labor and Equipment – These costs depend on the type of tillage system.

4. Mowing Cost – Mowing is assumed to occur twice per year.

5. Land Rent – This is the opportunity cost of taking land out of some other productive use and occurs for the entire life of the project.

Table 3.8. Inputs used to Calculate Costs for Filter Strips

Items	Units	Notes
Interest Rate	%	
Time Period	Years	Length of time they plan to maintain the strip.
Number of Acres	Acres	Size of planted filter strip in acres.
Costs		
Annual Land Rent	$/acre/year	Value of the land you are using for filter strips in crops.
Cost of Seed	$/acre	This will vary with chosen type of vegetation
Mowing Costs	$/acre/year	This varies by landowner. A default value of $25 is used.
Conventional Tillage	(0 or 1)	

The average area that could be protected by each acre of buffer was chosen to be 15 acres based on field and soil map observations reported in the relevant literature (Davis 1998). It was assumed that in most cases beyond this 15:1 field acres to buffer acres ratio, the buffer would not be effective. Fifteen acres was used as the average area protected by each buffer to determine how many acres would be occupied by the buffers. We calculated the present value of the net costs or impact of the filter strip on farm profits over the life of the project. Because the standard is one acre of filter strips for each fifteen, the final values are divided by fifteen. The baseline number of acres with filter strips was assumed to be zero.

Effectiveness

A number of studies on trapping of sediment and nutrients in natural and constructed grass filter strips have been conducted. Studies have been done on sites with various slopes, plot lengths, crop type, and other varying factors. A summary of the results can be found below.

Table 3.9 Summary of Previous studies on Nutrient and Sediment Trapping

Study Location	Filter Description	Plot Length	Impairment	Trapping Efficiency (%)	Reference
Virginia	Orchard grass	4.6 9.1	N	61 61	Dillaha and others, 1989
Maryland	Fescue	9.2	N	35	Magette and others, 1989
North Carolina	Bermuda-crab grass	4-5	N	50	Parsons and others, 1991

Given the above assumptions, and the cost and effectiveness ranges, it is possible to estimate a cost range per unit of sediment and nutrients saved, and, from that, per unit of improvement in DO concentration. This cost-effectiveness is based upon the midpoint of the cost range estimated for implementing filter strips.

III.2.3 Cost of Flow Augmentation

The literature review provides some background on several methods used to measure the costs of flow augmentation. The economic analysis of flow augmentation in this paper considers three alternate ways of estimating the costs of flow augmentation. Because the only permitted withdrawals are from agriculture, the necessary reductions in withdrawals will mean a reduction in irrigation. The first perspective identifies the potential reduction to farmer's net income due to having less water available for irrigation. The second approach uses recent payments to farmers

in southwest Georgia to forgo irrigating a certain number of acres to arrive at an estimate for the cost of leasing the water. Finally, the cost of changing to more efficient irrigation technology is examined. It should be noted that while there have been numerous studies attempting to generalize a water use measure for southwest Georgia agriculture, there are no agreed upon measures. This adds to the uncertainty of this analysis. The intention of this analysis is not to arrive at an exact estimate for the cost of flow augmentation but to provide several methodologies that can be used to determine a range of costs.

III.2.3.1 Value of Irrigation Water

The value of irrigated water represents the annual economic returns to irrigation water. Any specific value depends on the profitability of irrigation as influenced by crop prices, irrigation costs, other production costs, and on dry land versus irrigated yields. Irrigation costs in turn depend primarily on the type of irrigation system and on the source of water.

The Comprehensive Study's Agricultural Report estimates that the percentage reduction in maximum yield associated with a 10% reduction in water use (from 18 inches) would be 8% for corn, and 3% for cotton and peanuts. Following the methodology used by Cummings et al (2002), we used the average yield and price of each crop, along with percentage of crop land planted in cotton, corn, and peanuts, to arrive at an estimate for the decline in gross revenue due to a 10% decrease in irrigation. Using data from the Mckissick et al study (2000), we also estimated the loss associated with not irrigating at all. These values are assumed to be the values of the water taken out of use for irrigation.

III.2.3.2 Flint River Drought Protection Act

In 2001, the Flint River Drought Protection Act was invoked after a drought was declared. The state accepts bids from farmers who sign an agreement not to irrigate qualifying

farmlands. Participation is limited to surface water users due to the uncertainty of the relationship between groundwater withdrawals and stream flow. Out of the 575 potential permits, farmers holding 347 permits registered to participate in the auction. The 209 permit offers that were accepted were estimated to withdraw 33,006 acres from irrigation. The Georgia EPD estimated that the Act assured an additional 200 cubic feet of water per second, or 130 million gallons per day, in the Flint River during critical periods.

The average bid price per acre to remove land from irrigation was $135. The total direct amount spent on the program was $4.5 million. We estimate the amount of water applied to the crop land in the area and use this to determine how much the state paid per acre inch of water.

III.2.3.3 Improving Efficiency

In order to arrive at an estimate for the cost of flow augmentation that could be achieved by improving irrigation efficiency, we used data from a paper by Evans et al (1998) investigating the irrigation system efficiency and water conservation characteristics for the primary irrigation systems present in Georgia. The study on irrigation conservation practices provides estimates for costs and water saved from improving irrigation efficiency. They found many systems have been in place for 20 years and are in need of upgrades and improvements. They (1) classify the different irrigation systems and assess the effectiveness of the irrigation practices; (2) present alternatives based on the type of irrigation system, and present the estimated water savings range and average; (3) estimate how many irrigation systems are potentially affected and the average system size; (4) estimate how much water will be saved if all systems suitable for improvement were to implement the proposed system changes (total potential gallons saved) in both an average and a dry year. The estimated cost for implementation on an average system is given, along with a cost per unit of water saved in an average year. This value is calculated from the

estimates for water saved and total statewide cost figures. This provides a value to represent the cost/benefit ratio. By knowing what types of irrigation systems are used in the study area, we can estimate the cost of improvements, how much water will be saved, and the cost per unit of water saved. We convert this into cubic feet per second, which is the unit required by the model.

III.2.4 Economic Model

The economic model aims to minimize the cost of improving DO levels subject to Georgia EPA water standards. The results of the water quality modeling and cost estimates are used to determine the cost of reaching the water quality target.

When evaluating a problem with many different combinations of management practices and locations, it can be difficult to execute all possible combinations and identify all of the possible trade offs. The model developed for this thesis included an optimization routine to find the least cost solutions. Although we were able to test each of the different scenarios by changing the appropriate input data and simulations, the optimization system helps to eliminate the need for random trial and error. OptQuest searches for the best set of input parameters by evaluating carefully selected simulation runs. This technology greatly enhances the utility of the simulation models. OptQuest, which is based on the principles of tabu search and scatter search (Glover and Laguna 1997), allows us to run thousands of simulations, varying both the amount and location of the implementation of the management practices.

The economic model described by Kneese (1975) describes a cost minimization problem that uses transfer functions derived from water quality modeling. If the transfer coefficients are the same for each variable (pollutant or flow) throughout the stream (location does not matter), marginal cost equalization is a necessary condition. When coefficients are not equal, unit of discharge reduction (or flow augmentation) should be shifted to location where the overall cost is

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the lowest. We adapt the model so that it will include two management practices: (1) agricultural BMPs that reduce nutrient and sediment runoff and (2) flow augmentation. Recall the equation described in Chapter two:

$$\text{Min } d'x$$

$$\text{S.T. } Ax \geq S$$

$$x \geq 0$$

Increasing water flow by limiting use or increasing conservation measures comes at a cost of $C(w)$ where w is the increase in the quantity of water relative to the status quo level Q. The actual flow is then denoted as $F = Q + w$. F is constrained on the lower end by the amount of flow in the status quo plus the amount of flow if no irrigation were to take place. It is constrained on the upper end by the base flow plus the maximum amount of water that could by conserved by reducing irrigation at the critical time, or the amount that each scenario is testing.

The cost of reducing x is $R(y)$, where y is amount that pollutant is reduced relative to x. The actual level of the pollutant load is then $P = x - y$. P is constrained on the lower bound by natural conditions and on the upper bound by amount the load could be reduced if the BMP was implemented on all feasible acres of cropland. The cost functions are then as follows:

(1) $C = C(w)$

(2) $R = R(y)$

A watershed approach aims to reach an ambient standard. To do this, a quantitative connection may be established between what occurs at the points where the residuals are discharged and the quality of water at various points along the watercourse. The mathematical functions making this connection are known as transfer functions, a_{ij}, $i=1, \ldots, m$; $j=1, \ldots n$, for pollutant reductions, and b_{ij}, for flow augmentation. The transfer functions are automatically

taken from the water quality modeling. Let S denote the required concentration of dissolved oxygen in the stream reach during the critical period. The goal of the TMDL can be stated follows:

$$(a_{11}y_1 + a_{12}y_2 + \ldots + a_{1n}y_n) + (b_{11}w_1 + b_{12}w_2 + \ldots + b_{1n}w_{n)} \geq S$$

or in matrix notation, $A'Y + B'W \geq S$

Suppose that stakeholders wish to achieve S at the lowest possible cost. This objective can be written as a constrained cost minimization problem with a target environmental constraint. The least cost way of achieving this as a Lagrangian is:

$$L = Min\ R(y) + C(w) + \lambda\ (S + A'Y + B'W)$$

Where,

R = cost of reducing load

C = cost of increasing water flow

S = ambient standard

y = reduction in load compared to status quo

w= quantity of water increased relative to status quo

A = matrix of transfer coefficients for load reductions

B = matrix of transfer coefficients for flow augmentation

Additional constraints can be imposed on the variables for the different scenarios. For example, a certain amount of water must remain in the stream. For this analysis, the cost minimization problem was solved allowing the flow and load parameters to change independently as well as simultaneously. The model was run using the actual conditions of June 2000, the 7Q10, and the conditions of a more typical 'wet' year.

CHAPTER IV

RESULTS

This chapter presents the results of both the water quality and economic components of

the thesis and a discussion of their implications. The results of the baseline, flow augmentation,

and load reduction scenarios coming from the water quality model described in chapter three are

presented. The cost estimates of the different management practices and the cost minimization

problem, which was solved while allowing the amount and location of changes in flow and loads

to vary, are then presented. The results of Spring and Dry Creek sub-watersheds were so similar

in nature the only the results of Spring Creek will be reported in detail. Within each section is an

explanation of and discussion of the results.

IV.1 WATER QUALITY MODELING

The SWQM was used to simulate the loading of NH3, CBOD, NBOD and the resulting

DO from the various land uses, flow levels, and point sources. Projection models were run based

upon the flows, stream characteristics, kinetic coefficients, temperatures, DO levels, and

reaeration rates as discussed in chapter three. For the calibrated model simulation, nonpoint

source loadings and reaeration rates were adjusted so that model predictions simulate the

measured DO value as closely as possible at sampling locations in June 2000 while still

providing a reasonable representation of water quality in the stream at the time of the sampling

event. Shown in Table 4.1 below are the results of the calibrated model simulation and a

comparison of the calibrated model predictions versus actual field data obtained from the same

location. The calibrated model uses actual flow rates rather than the 7Q10 flows.

Table 4.1 Model Flow Parameters for Spring Creek Sub-basin

Description	Flow	DO	CBODu	NH3N	TON	Temp
Headwaters	1.25	4.80	6.8	0.4507	1.7245	25.6
Conditions@ calibration point	3.07	4.5016	3.4346	0.6852	1.4219	25.4
Flow @ End of Model	3.4867	3.92	3.06	0.5947	1.4537	25.52

Model Incremental Flow Parameters

Sections	CBODu(mg/L)	NH3N (mg/L)	TON(mg/L)	Total Flow (cfs)	Length (miles)
1	9.7	.56	2.155	0.45	8.9
2	9.7	.56	2.155	0.38	7.5
3	9.7	.56	2.155	0.18	3.5
4	9.7	.56	2.155	0.32	6.3
5	9.7	.56	2.155	0.13	2.5
6	9.7	.56	2.155	0.32	6.3

Comparison of Calibrated Model Flow Parameters to Actual Data for Spring Creek

Description	CBODu(mg/l)	NH3N (mg/l)	TON(mg/l)	DO(mg/l)	Total Flow (cfs)	Temp(degrees C)
Calibrated	3.4346	0.6852	1.4219	4.5016	3.07	25.4
Actual 6/7	3.2	0.69	1.3	4.72	3.6	24.5

The agriculture BMPs discussed in chapter 3 have an effectiveness range between 20 and

75%. We base our simulation on a near-midpoint value of 50%. In this case, the mean DO

concentration increases in the range of 0.14 and 0.33, depending on the flow level. All of the

load reduction was allocated to the agricultural sector.

When considering different agricultural BMPs for load reduction scenarios, we first

assumed that load reductions would be required from all acres in the watershed. We then

adapted the model so that the amount of acres requiring BMP implementation could be varied

over all the cropland, or BMPs could be required for the land associated with each of the sections

of the stream (6 or 4 for Spring Creek and Dry Creek, respectively). The results of the

simulation with the load reduction can be seen in Table4.2.

Table 4.2 Load Reduction Model Flow Parameters

Description	Flow	DO	CBODu	NH3N	TON	Temp
Conditions@ calibration point	3.07	4.755	2.58	0.556	1.019	25.42
Conditions @ End of Model	3.4867	4.533	2.56	0.4639	1.02	25.43

Load Reduction Incremental Inflow Parameters

Sections	CBODu	NH3	TON	Length (miles)
1	6.7879	0.4714	1.4655	8.9
2	6.7879	0.4714	1.4655	7.5
3	6.7879	0.4714	1.4655	3.5
4	6.7879	0.4714	1.4655	6.3
5	6.7879	0.4714	1.4655	2.5
6	6.7879	0.4714	1.4655	6.3

Flow Augmentation

Plots of the four scenarios depicting the quantity of flow augmentation as the predictor

variable (X) and daily mean DO (mg/l) as the response variable (Y) for the June 2000 flow rate

can be seen can be seen in figure 4.1. The average DO concentration does increase while flow

increases. However, the results vary when augmentation takes place at different points along the

stream and when different combinations are attempted.
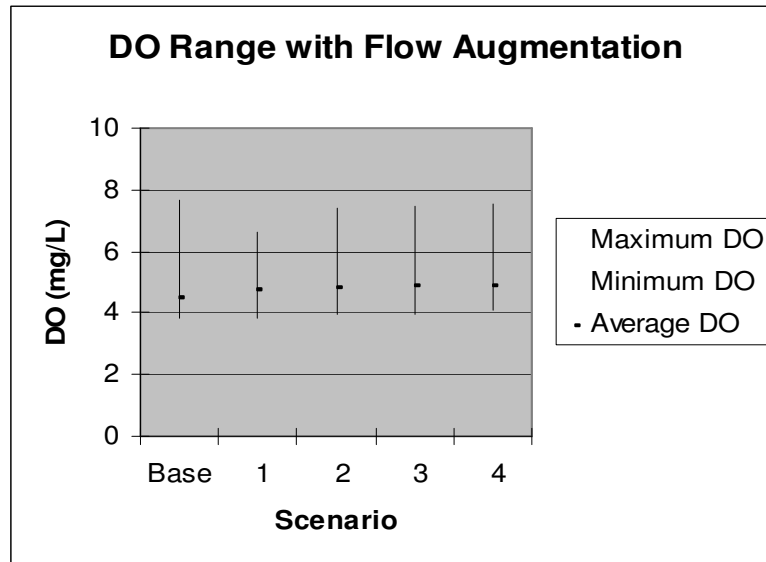
Figure 4.1 DO Range with Flow Augmentation Scenarios for June 2000

Simulation Results

After evaluating each of these scenarios independently, a simulation program was used to

allow us to view the results of varying several parameters at once. We chose average DO (mg/l)

that occurs along the stream and minimum DO (mg/l) that occurs along the entire stream as

forecast variables. The variables tested for this analysis were the flow parameters on all segments

of the stream, the amount of acres the BMPs are implemented on for each of the segments, the

reaeration rate, temperature, and point source concentration. This part of the analysis was does

while minimizing cost, and the results will be presented with the results of the economic

analysis.

Sensitivity analysis

A sensitivity analysis was performed to allow us to judge the influence each assumption

cell has on a particular forecast cell. Crystal Ball software, an excel add-in, was used to calculate

how important each assumption is to each forecast while the simulation is running.

During a simulation, Crystal Ball saves all the calculated assumption and forecast values. After a sample size, N, number of trials, a list of ranks is generated. The smallest number is 1, and the largest number is the N. Each assumption rank list is paired up with each corresponding forecast rank list and then a Pearson correlation coefficient is calculated for each pair. The running sums of the Pearson correlation coefficients for each pairing in separate matrices are stored. After the simulation, the computed correlation coefficients are averaged. The sensitivity is measured by rank correlation. The assumption variables tested for this analysis were the flow parameters on all segments of the stream, the amount of acres the BMPs are implemented on for each of the segments, the reaeration rate, temperature, and point source concentration.

The parameter with the strongest correlation is temperature. The amount of headwater flow and the reaeration rate are next, followed by the concentration of runoff from the stream segment at the first stream segment.

IV.2 RESULTS OF ECONOMIC ANALYSIS

This section presents the cost analysis for the different management options and the results of the cost minimization problem. Because the marginal water quality impact of a pollutant load varies from place to place, allocations and the TMDL must be developed simultaneously. For this study, the objective was to minimize the total costs of meeting the water quality requirement. We used a stochastic optimization modeling framework in order to consider all available options and identify the combination of management practices that will meet DO criteria. The allocation decisions should reflect tradeoffs between the various combinations of management practices.

IV.2.1 Cost Estimates of Different Management Options

IV.2.1.1Filter Strips

The costs of installing and maintaining filter strips on cropland include: (1) land rental costs, (2) seed and fertilizer costs, and (3) equipment and labor cots. Seed and fertilizer costs will only occur when the strip is installed; however, the others may occur throughout the life of the filter strip.

The rental value of farmland varies throughout the state. The rental value of cash cropland in 1999 was reported to be $90.00 if the land had a water use permit and $37.00 with not water use permit (Cummings et al 2002). Cummings also reports that in Southwest Georgia, farmers report land rental value between $100 and $150 per acre for irrigated lands. Though crops and land productivity classes are variable throughout the state, for the purposes of our study we assume that the land rental value is $125. The remaining cost estimates are entered into the chart below. Given those cost estimates, the present value per acre is then calculated.

Table 4.3 Cost of Filter Strips

INPUTS	Items	Numbers	Units	Notes
1	Interest Rate	5	%	*expected discount rate
2	Time Period	15	Years	*expected project duration
3	Number of Acres	1	Acres	*project region in acre
8	Annual Land Rent	$125	Dollars/acre/year	*actual annual land rental rate
9	Cost of Seed	$30	Dollars/acre	*see table for seed cost at right
10	Mowing Cost	$10	Dollars/acre/year	*expected mowing cost
11	Conventional Tillage	0	(0 or 1)	*put "1" if yes;" 0" if no

	Calculations	A	B ($/acre) per	C (PV) per	D (PV) total
Items		(year)	item	acre	acres
1	Costs				
1.1	Seed	1	($30.00)	($30.00)	($30.00)
1.2	Fertilizer	1	($37.00)	($37.00)	($37.00)
1.3	Labor & Equipment (a or				

				PV	PV
	b)				
	a) Conventional Tillage or	0	($32.50)	$0.00	$0.00
	b) No Tillage	1	($21.50)	($21.50)	($21.50)
1.4	Mowing (annual)	15	($10.00)	($108.99)	($108.99)
1.5	Land Rents (annual)	15	($125.00)	($1,297.46)	($1,297.46)
1.6	Total Costs	15		**($1,494.94)**	**($1,494.94)**
4.2	Annual Net Benefits (Costs)	15		($144.03)	($144.03)

*numbers in **(Red)** are negative
PV stands for present value

If the present value of the cost per acre for a filter strip is $144.03 and for every fifteen acres

planted in cropland, one acre should be converted to a grass filter strip, then the cost per acre for

using filter strips as a BMP would be $9.60.

IV.2.1.2 Flow Augmentation

We used GIS software in the study areas to identify the surface withdrawal points within

the sub-basins. These are the potential augmentation points. Using existing literature on water

demand along with data available from EPD the USDA, the amount of water that could be

conserved by reducing irrigation was determined. The net irrigation requirements in both normal

and dry years are reported in table 4.4.   If a "typical" acre consists of 25% corn, 50% cotton and

25% peanuts, then the net irrigation requirements per acre would be 10.30 and 11.875 acre

inches for a normal and dry year, respectively.

There have been several attempts to arrive at a general water use measure in for

southwest Georgia. Estimates of agricultural water use are provided as part of the Georgia,

Florida, and Alabama's Comprehensive Study report on water use in the ACF basin (Cummings

et al 2002).   One estimate provided by this study suggests that 7.2 inches is an estimate for agricultural water use in an 'average' year when the rainfall is distributed over the March-October period in an 'average' way (Cummings et al 2002). A study sponsored by USGS estimates irrigation application for corn, cotton, peanuts, turf, and other crops to average 8.8 inches for the year 1996, which does not appear to differ from an average year by a large amount. Finally, preliminary results from an AES study estimated irrigation applications for the 12 months period from August 1, 1999 through July 31, 2000. Their rough estimate for irrigation applications in the Flint River Basin is almost 12 inches per acre.

Table 4.4 Irrigation Requirements for Wet and Dry Years.

Crop		Lower Flint Estimates (Acre Inch)
Corn	Normal Year	11.14
	Dry Year	12.17
Cotton	Normal Year	11.74
	Dry Year	13.68
Peanut	Normal Year	6.58
	Dry Year	7.97
Soybean	Normal Year	7.58
	Dry Year	9.04

The results of two different methods of measuring the costs of augmenting flow through irrigation reduction are presented.

Cost Estimate of Flow Augmentation Based on the Flint River Drought Protection Act

The results of the second attempt to value the cost of flow augmentation, which used the Flint River Drought Protection Act to arrive at an estimate, are summarized in this section. There are several different ways of arriving at an actual cost figure. The results reported in a paper by Doherty (2000) are reported first. She used a 7 acre inch irrigation estimate for an average year and determined that the amount of water conserved would be 6.27 billion gallons (Doherty

2002). Following this logic, the direct cost per gallon for 2001 was roughly $0.0007, or $228.09 per acre foot.

Using the estimate of 11.875 acre inches, or .98 acre feet, arrived at earlier in this section may by more appropriate for a dry year, such as 2000. If 33,006 acres were taken out of irrigation, then multiplying these two figures together gives you 391,946.25 acre inches, or 32,662.1875 acre feet. The state spent $4.5 million total on the FRDPA that year, or $137.77 per acre foot. A simpler way of thinking about this is to consider the average price of $135.70 per acre foot and divide this by 0.98958 acre feet, which gives us approximately $137 per acre foot. This figure does not include effects on the local economy. Furthermore, this is only for one year.

Irrigation Efficiency

In Georgia, there are more than 9,000 center pivot irrigation systems used to irrigate farmland. Many of these center pivots make full circles and/or apply water to non-cropped areas. One way to conserve water is to make irrigation systems more efficient in an effort to improve the accuracy of these systems. The study done by Evans et al (1998) examines the economics of improving efficiency. The results are summarized in Table 4.5.

Table 4.5 Cost of Water Conserved by Improving Irrigation Efficiency

Irrigation System		Average year –total potential gallons saved (million gall)	Estimated cost for full implementation on an average system (1998$)	Cost/ unit water saved, average year $/mill gallon (1998$)	Cost/unit water saved, average year $/AF
Center pivot	End gun shut off	2851	$500	$351	$114
	Replacing the sprinkler package with new package	4181	$250	$521	$169
	Reducing the angle impacts and using medium pressure	1616	$1,200	$631	$205
	Putting low pressure sprinklers on drops	570	$4,000 $1,500 ($1.50/ft)+$2,500 booster pump	$421	$137
Traveling gun	Repairing water delivery system	2585	$3,500 $500 to $6,000	$2,166	$706
	Using speed compensation	2100	$5,000 $3,500 to $6,500	$3,905	$1,272
	Changing angle and trajectory	969	$1,500	$1,857	$605

1 ac-ft = 325,851 gall

When individual costs were weighted by the amount of water conserved, a weighted estimate of $0.0005 per gallon, or $162.93 per acre foot, was arrived at when only using center pivot improvements. The amount of water that can be conserved in this manner is limited by the amount of improvements that can take place. In this case, only 9 billion gallons of water could be saved if these improvements were implemented. An estimate of $0.002 per gallon, or $651.70 per acre foot, was arrived at for the traveling gun improvements with a restriction of 5.2 billion gallons of water savings possible. If we weight the costs of each type of system, we

arrive at a total cost of $0.001 per gallon, or $325.85 per acre foot. These estimates are for the first year only. If you assume that the improvements will last for 15 years, the cost would be $31.39 per AF per year. Although the data on augmentation costs used here are incomplete, these data are thought to be reasonable estimates.

Economic Model

After determining the cost of the various management practices, a spreadsheet modeling framework was developed to identify cost effective allocations under physical and legal requirements. The framework allows us to identify the management practices and decisions that meet the water quality criteria while achieving cost minimization objectives. Furthermore, the model allows us to identify the amount and location of BMP implementation and flow augmentation necessary to meet the requirements.

This section defines the equations that model water quality as a function of streamflow and pollutant loading as well as the cost minimization problem used to achieve water quality targets at the least cost. The decision variables in the model are the flow variables for each stream segment (which are a function of the amount of water saved by decreasing agricultural water withdrawals) and the degree of BMP implementation throughout the watershed. Upper bounds, derived from the amount flow could be augmented from irrigation reductions or efficiency improvements, are placed on the flow augmentation variables. There are two requirements for the model. The first is that the average DO should not be below 5mg/l. The second is that DO concentration should not go below 4mg/l at any point. The objective is to minimize the cost of meeting these requirements.

For each iteration in the optimization process, the following steps occur. First, the model selects new values for the decision variables; (2) decisions variables are used to estimate costs

and loads; (3) loads and impact coefficients assumptions are used to forecast concentration

distribution; and (4) the model compares concentrations to criteria requirements and determines

the degree to which the objective is satisfied.

Our analysis found filter strips to be more cost effective than conservation tillage for

improving DO concentration. Filter strips are therefore the only agricultural BMP considered

here. Using the lagrangian from chapter 3, the mathematical equation to be solved is:

$$\text{Min } R(y) + C(w) + \lambda \ (S + A*Y + B*W)$$

Where,

R = cost of reducing load

C = cost of increasing water flow

S = ambient standard

y = reduction in load compared to status quo

w = quantity of water increased relative to status quo

A = matrix of transfer coefficients for load reductions

B = matrix of transfer coefficients for flow augmentation

The first order conditions for a minimum to equation () are that

$$\frac{\partial L}{\partial Y} = \frac{dR}{dY} + (-A) = 0$$

$$\frac{\partial L}{\partial W} = \frac{dR}{dW} + (-B) = 0$$

$$\frac{\partial L}{\partial \lambda} = R + AY + BW$$

$$\lambda = \frac{dR}{dY} * \frac{1}{A}$$

$$\lambda = \frac{dC}{dW} * \frac{1}{B}$$

75

The predicted concentrations associated with minimum cost allocation satisfy environmental goals. The model is run with both observed conditions in June 2000 (the calibrated model), the 7Q10 flow (both low flow conditions), and with observed conditions in 1995. In 1995, which was a 'wet' year, the DO concentration met the water quality criteria both in actuality and in the model results. For each of these flow scenarios, the model was run allowing both flow and load parameters to change simultaneously, allowing only flow to change, and allowing only load reductions to change.

These results show that the least cost way to reach the water quality targets in the year 2000 would be to implement BMPs on 12% of the row crops in the watershed and to increase flow by 5,158 AF per year, at a cost of $465,528. This is based on the assumption that 23% of all irrigation for the year would occur during the study period. The reason for high amount of flow augmentation is that the flows in this year were so low that in order to have an average concentration of 5mg/l throughout the stream, flow would have to be increased throughout the stream. The results suggest that locating the filter strips in more critical areas is more effective than evenly distributed throughout. They also suggest that augmenting flow in the same area as the filter strips is successful at improving DO concentration. Because 2000 experienced such low flows, it is impossible to reach the concentration goal of 5mg/l using only BMP implementation. Applying filter strips on all feasible acres of row crops gives us an average DO concentration of 4.9mg/l, with portions of the stream falling below 4mg/l. This would cost $1,091,520.

Using the 7Q10 flow rate, the least cost way to reach the concentration goal is by applying BMPs to 10% of the row crops and augmenting flow by 1,984 AF/year. This would

cost $291,648. Even with the 7Q10 flow rates, the minimum DO concentration cannot be met by load reductions alone.

Table 4.6 Cost Summary

Flow rate	Cost with both load reductions and flow augmention	Cost with only BMP
June 2000	$465,528	Infeasible ($1,091,520)
7Q10	$291,648	Infeasible ($1,091,520)
June-July 1995	$0	$0

Policy Implications

Water management as well as nonpoint source control has been proven as an important control technique in improving DO concentration levels. If water management for dilution purposes is ignored, then methods to reduce loading, or increase DO concentration, result in an unnecessarily high social cost of control. An effective combination of management options should involve both BMPs to reduce runoff concentration and increasing the assimilative capacity of the waterbody. The results of this research can be used as a springboard to discuss the important link between water quality and water quantity issues. The recognition of this link is becoming increasingly critical as state policies are beginning to address the quality and scarcity of our water resources.

The three year drought in Georgia along with the Tri-State water wars has brought more attention to the issue of water supply, especially in the heavily irrigated lower part of the Flint. This has brought a necessity for new policies to deal with water supply issues. There has been increasing concern over the amount of water available not only for withdrawal purposes but also for the protection of fish and wildlife habitat and for dilution of loads discharged into our waterbodies. At the community level, water use restrictions have been implemented throughout the state as supplies of clean fresh water fail to meet the demand. At the municipal and industrial

level, any new permit is already required to consider the withdrawal's effect on the water quality of that portion of the water body. There have been additional holds on assigning new permits and the changing of existing ones. Other attempts to protect instream flows have focused on irrigation withdrawals. Farmers will reduce their water consumptive use when the marginal benefits of selling one acre-foot of water is equal to or greater than the marginal benefits of using that acre-foot of water in growing crops. Farmers will decide whether to abandon irrigated lands, substitute crops requiring less water, etc. Cost-sharing programs to improve irrigation efficiency and the Flint River Drought Protection Act's attempt to take land out of irrigation recognize this and subsidize farmers, in many cases at a high cost. All of these efforts aim to change the amount of water withdrawals, which will in turn affect the assimilative capacity of the waterbody.

At the same time, TMDLs must account for variation in flow, or seasonal variation, but few actually do. The first step in developing a TMDL is determining whether or not the there are impairments that cause a waterbody not to support its designated use. The most basic issue is whether there is sufficient water in our streams and rivers to maintain the designated uses. Concern over this may lead to increased reliance on programs to maintain instream flow. The amount of water withdrawn and returned to a waterbody will affect the dilution of pollutants in that system. Therefore, reducing withdrawals or using water more efficiently may be a management option worth examining when developing a TMDL. Especially in dry conditions, water supply will have to be balanced against ecological considerations. The TMDLs in GA often include flows in the modeling, but they are most often simulated. The TMDLs do not mention that the flows are necessary for diluting the loads. While dilution is certainly not a solution, an understanding of instream flow is critical to assessing the impacts of diffuse

pollutants. For a more complete picture, TMDLs could establish locations within the basin where there are withdrawals of water. The TMDL could also address water diversions and ground water withdrawals. Without this, the TMDL will not be able to address all the actions necessary to reach water quality goals. The model used for this paper is able to indicate where and at which level alternative methods of improving DO levels should be put into action. Models such as this are capable of aiding the policy makers in making policy decisions on water resource allocation and dissolved oxygen management within an integrated framework for the benefit of society.

CHAPTER V

SUMMARY AND CONCLUSIONS

This chapter provides a brief summary of the major aspects of this thesis. This is followed by a discussion of the implications that can be drawn from the research for water quality management in general. Finally, this chapter concludes with a discussion of the limitations of the methodology of the present research and areas for future study.

V.1 SUMMARY

This thesis examined the economics of flow augmentation relative to BMPs in meeting water quality targets as determined by TMDLs.   The SWQM and an economic model were used to meet the objectives of this research. Management practices that contributed to improving water quality standards at the watershed level were identified. Each practice, or scenario, was first evaluated for its effect on instream water quality. Secondly, an evaluation of the costs of different scenarios was performed using economic theory. The results were used to determine cost-effective management practices in an agricultural watershed in Southwest Georgia.

The main reasons for undertaking this study were: (1) the interrelationship between water quantity and water quality is a challenging issue that the TMDL policy should consider; (2) to demonstrate the use of a watershed approach to address the economics of all feasible management options, including flow augmentation, under the assumption of that water quality targets should be met at the least cost.

The main objectives of the literature review in Chapter 2 were to draw from past studies concerning the (1) water quality modeling and management; (2) cost of management options and the cost of flow augmentation in order to improve water quality, and (3) reaching water quality

targets at the least cost. One lesson learned from this review was that very few studies address the problem of minimizing the costs of meeting water quality standards using a combination of flow augmentation and load reductions, especially in the context of the TMDL program.

The methodology developed in Chapter 3 is generic enough to be applicable for addressing general water quality management problems. Though the general models do not have to be applied to any specific regulatory policies, they were related to the TMDL process because; (1) they estimate pollution control costs by capitalizing on the cost-minimizing features inherent in the watershed approach of the TMDL process – control by least-cost agents; (2) pollution control costs are estimated for attaining the same water quality standards as required by the TMDL process; and (3) the assimilative capacity of a water body is the first consideration of a TMDL, and flow is a specific consideration of temperature concerns under Section 303(d) of the CWA.

Chapter 4 presents the results and a discussion of what they imply. The water quality modeling component of this study tested the hypothesis that increasing dry period base flow in streams by decreasing withdrawals (with pre-defined water quality parameters) enhances water quality in terms of DO. This has been demonstrated to be the case. An increase in daily mean DO was observed for all scenarios, a significant change was observed during low-flow periods. As mentioned in the previous chapter, the various flow augmentation alternatives are well within the capacity of the area. The agricultural BMPs are also predicted to reduce pollutant concentrations in cropland runoff. Both flow augmentation and BMPs are expected to improve aquatic integrity and contribute to the TMDL goal of compliance with State water quality standards. Flow augmentation has additional benefits of improving aquatic habitat and an increased amount of water instream to dilute other pollutants. Though the stream segments that were the focus of this

case study are not used largely for recreational purposes, the same methods could be applied to streams and rivers that would experience not only an increase in recreational values but also in property values. While several of these benefits are also associated with improvements in solely water quality, the study highlights the fact that the two cannot be considered completely separately.

V.2 IMPLICATIONS OF REASEARCH

Water quality degradation is due to the concentration of waste substances in the water. The concentration level depends on the volume of wastes as well as the volume of water in a stream. Water quality degradation has generally been identified as being only the result of increase the amount of waste discharges, but large water withdrawals can also cause high concentrations, other things being equal. When a portion of the water is withdrawn, the stream has less capacity to assimilate and dilute wastes. Therefore, the concentration of the wastes in the stream increases. Historically, water pollution control policies have emphasized reducing waste loads rather than flow augmentation or a combination of both. This study has demonstrated that, in some cases, goals can be reached at significantly lower cost if both approaches are considered.

For the empirical work, the model is formulated and applied to two impaired segments of the Spring Creek watershed in the Lower Flint River Basin. By minimizing the total cost of TMDL implementation under resource and institutional constraints, the model provides a framework that allows us to determine cost effective solutions to the water resource problem. Both pollutant loads from agricultural and other activities and decisions made about water allocation contribute to dissolved oxygen concentrations. However, the current TMDL program is focusing solely on pollutant load reductions. The results of different load reduction scenarios

without flow augmentation considerations are presented in chapter 4, along with the results of flow augmentation scenarios without load reductions. The model result indicates that DO concentration will be increased by with the load reduction scenario and by with the flow augmentation scenario.

Limitations and Directions of Future Research

One contribution of this research was the development of a methodology for evaluating the costs and effects of different management practices in order to solve a cost minimization problem. The methodology may be used to estimate watershed-based TMDL implementation costs such that water quality standards are maintained. While the methodology is intended to produce realistic results, the reliability of these results does depend on the data available and the assumptions used.

Limitations of the results of the present study are partly due to the limitation in the data. The results were then based in part on assumptions such as the fact that all land parcels were identical in physical characteristics such as soil type. The transport coefficients derived from the water quality model were based on the assumption that the same BMP could be adopted over the entire watershed for the same cost.

The water quality model is currently loosely calibrated for CBOD, ammonia, and DO. A very simplistic process was simulated for BOD and nutrient loading from different land uses which may not represent true conditions. There are also limitations associated with using a steady-state model. As such, the model is good for predicting general trends but cannot accurately simulate diurnal swings or changes throughout the year. It is recommended that a more detailed analysis be carried out using a steady-state in stream water quality model such as

QUAL2E. DO conditions can then be studied in detailed segment by segment there by giving a longitudinal resolution to water quality.

Finally, there is uncertainty involved in any modeling process. This uncertainty gets translated in model predictions. Therefore, it is critical for decision-makers to be cognizant of this uncertainty before making decisions based on model performance.

In conclusion, there are many complexities associated with managing both issues of water supply and non point source pollution. The solutions to these problems will be costly both to develop and to implement. With the growth in population and irrigated agriculture, coupled with issues such as the tri-state water wars, drought years, and water quality policies such as the TMDL program, there is a need to identify management options that will improve watersheds at a least cost. We suggest that, at the very least, flow augmentation is a cost effective management option when low flow is aggravating the water quality problem and it is feasible to augment flow through water conservation measures.

For a more complete picture, TMDLs could establish locations within the basin where there are withdrawals of water. Ideally, it would account for changes in return flow from nonpoint sources. The TMDL could also address water diversions and ground water withdrawals. Water quality goals can be reached more effectively if the full range of management practices and flow augmentation alternatives are part of TMDL implementation.

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Appendix A

Water Quality and Flow Data

Date	CFS	Date	CFS	Date	CFS	Date	CFS
6/1/2000	3.6	7/1/2000	3	6/1/1995	101	7/1/2000	204
6/2/2000	3.2	7/2/2000	3	6/2/1995	98	7/2/2000	197
6/3/2000	2.6	7/3/2000	2.8	6/3/1995	97	7/3/2000	207
6/4/2000	2	7/4/2000	2.3	6/4/1995	96	7/4/2000	215
6/5/2000	1.5	7/5/2000	1.8	6/5/1995	218	7/5/2000	255
6/6/2000	1.9	7/6/2000	1.3	6/6/1995	801	7/6/2000	258
6/7/2000	2.4	7/7/2000	0.9	6/7/2000	1320	7/7/2000	253
6/8/2000	2.2	7/8/2000	0.8	6/8/2000	1600	7/8/2000	235
6/9/2000	1.4	7/9/2000	0.6	6/9/2000	1510	7/9/2000	206
6/10/2000	1.4	7/10/2000	0.5	6/10/2000	1260	7/10/2000	181
6/11/2000	1.4	7/11/2000	0.48	6/11/2000	885	7/11/2000	163
6/12/2000	1.4	7/12/2000	0.49	6/12/2000	678	7/12/2000	145
6/13/2000	1.4	7/13/2000	0.48	6/13/2000	721	7/13/2000	127
6/14/2000	1.4	7/14/2000	0.61	6/14/2000	872	7/14/2000	115
6/15/2000	1.4	7/15/2000	0.61	6/15/2000	884	7/15/2000	107
6/16/2000	1.4	7/16/2000	0.6	6/16/2000	670	7/16/2000	97
6/17/2000	1.5	7/17/2000	0.6	6/17/2000	496	7/17/2000	90
6/18/2000	1.5	7/18/2000	0.5	6/18/2000	410	7/18/2000	83
6/19/2000	1.3	7/19/2000	0.39	6/19/2000	359	7/19/2000	76
6/20/2000	1	7/20/2000	0.36	6/20/2000	340	7/20/2000	70
6/21/2000	0.9	7/21/2000	0.32	6/21/2000	370	7/21/2000	65
6/22/2000	0.9	7/22/2000	0.27	6/22/2000	341	7/22/2000	61
6/23/2000	1.1	7/23/2000	0.26	6/23/2000	318	7/23/2000	58
6/24/2000	1.2	7/24/2000	0.26	6/24/2000	301	7/24/2000	54
6/25/2000	1.6	7/25/2000	0.25	6/25/2000	295	7/25/2000	53
6/26/2000	2	7/26/2000	0.23	6/26/2000	329	7/26/2000	68
6/27/2000	2.2	7/27/2000	0.22	6/27/2000	316	7/27/2000	65
6/28/2000	2.6	7/28/2000	0.21	6/28/2000	265	7/28/2000	69
6/29/2000	2.8	7/29/2000	0.21	6/29/2000	240	7/29/2000	80
6/30/2000	2.8	7/30/2000	0.18	6/30/2000	222	7/30/2000	83
		7/31/2000	0.24			7/31/2000	85

Year 2000 Water Quality Data from USGS Station Near Iron City

Date Time	Temp (DEG C)	CFS	DO (mg/L)	BOD5 (mg/L)	NH3-N	NO2+NO3 Total (mg/L)
5/10/2000 9:20	25.5	30	6.5	1	0.06	0.8
5/17/2000 9:05	28.0	17	7.4			
5/24/2000 9:20	29.0	9.4	6.1			
6/7/2000 9:15	23.5	4.3	5.9	0.8	0.08	1.5
7/19/2000 8:15	30.0	0.38	5.5	1.1	0.08	1
7/26/2000 8:50	26.0	0.22	5			
8/2/2000 8:55	28.5	0.19	5.8			
8/16/2000 9:40	30.0	0.06	4.9	1.2	0.19	.
9/13/2000 9:25	28.0	0.04	8.1	0.9	0.19	M
10/25/2000 9:15	18.5	1.1	5.3	1.2	0.1	1.5

Year 1995 Water Quality Data from USGS Station Near Iron City

Date Time	Temp (Deg C)	CFS	DO (mg/L)	BOD5 (mg/L)	NH3-N	NO2+NO3 Total (mg/L)
5/17/1995 8:40	23.5	84	6.5	0.4	0.07	0.7
6/21/1995 14:25	22.5	160	7.6		0.05	0.7
7/19/1995 6:45	26	14	6.5	1.5	0.23	0.8
8/23/1995 13:45	26	18	7.6	0.7	0.12	0.8
9/20/1995 14:00	24.5	15	8.3	0.6	0.13	0.7
10/25/1995 10:00	20	20	7.6	1.3	0.12	0.6
11/30/1995 13:00	14	59	9	< 1.0	0.04	0.3

2000 Water Quality Data Collected at Spring Creek for the Flint River DO TMDL

Date	BOD5 (mg/L)	DO (mg/L)	% Saturation	NH3 (mg/L)	NO2-NO3 (mg/L)	Temp (deg C)
1/20/2000	0.9	7.04	69	0.02	0.31	14.3
2/2/2000		11.6	94			7
2/9/2000		10.36	88			9
2/16	1.1	8.26	81	0.04	0.11	15.1
3/29/2000	1.2	7.31	75	0.04	0.23	16.8
4/26/2000	0.8	8.57	87	0.08	0.56	16.1
5/10/2000	1.0	7.13	81	0.08	0.62	21.7
5/17/2000		6.94	79			22.2
5/24/2000		6.0	72			24.6
6/7/2000	3.2	4.72	56	0.69	1.3	24.5
7/19/2000	4.6	2.2	28	1.2	0.43	27.5
7/26/2000		1.87	23			25.2
8/2/2000		7.45	89			24.5

8/16/2000	4.9	3.71	46	0.56	0.32	26.6
9/13/2000	1.1	5.69	68	0.12	0.52	24.4
10/25/2000	2.2	7.6	79	0.14	0.83	17.7
11/7/2000	2.5	6.7	75	0.06	0.71	20.8
11/14/2000		7.8	79			16.3
11/20/2000		8.1	74			12.2
12/6/2000	0.8	9.6	82	0.03	0.42	8.9

2000 Water Quality Data Collected in Dry Creek for Flint River DO TMDL

Date	BOD5 (mg/L)	CFS	DO (mg/L)	% Saturation	NH3 (mg/L)	NO2-NO3 (mg/L)	Temp (deg C)
1/25	1.7	124	10.09	81	0.05	0.08	5.9
2/22	0.6	47	9.58	83	0.04	0.21	9.6
3/7		33	7.91	75			13.4
3/14		73	8.85	78			10.6
3/21	0.8	81	8.31	80	0.07	0.11	14
4/4		50	6.05	63			17.9
4/11		25	8.91	84			13.1
4/19	1	25	7.51	76	0.09	0.46	16
5/2	1.1	9.819	6.97	73	0.09	0.49	18
9/19	1.1	1.1	2.47	28	0.1	0.06	21.8
9/26		3.8	4	47			23.4
10/4		1.8	3.3	36			19.6
10/17	0.7	2	6	59	0.07	0.05	14.9
11/28	2.4	35	8.2	71	0.03	0.03	9.6
12/12	0.8	25	7.1	67	0.03	0.05	13.3

Appendix B Maps

Map 1

NPDES Permits for Spring Creek Watershed

N P D E S   P e r m i t s   f o r   S p r i n g   C r e e k   W a t e r s h e d

Permit Compliance System
Reach File, V1
Cataloging Unit Boundaries
State Boundaries
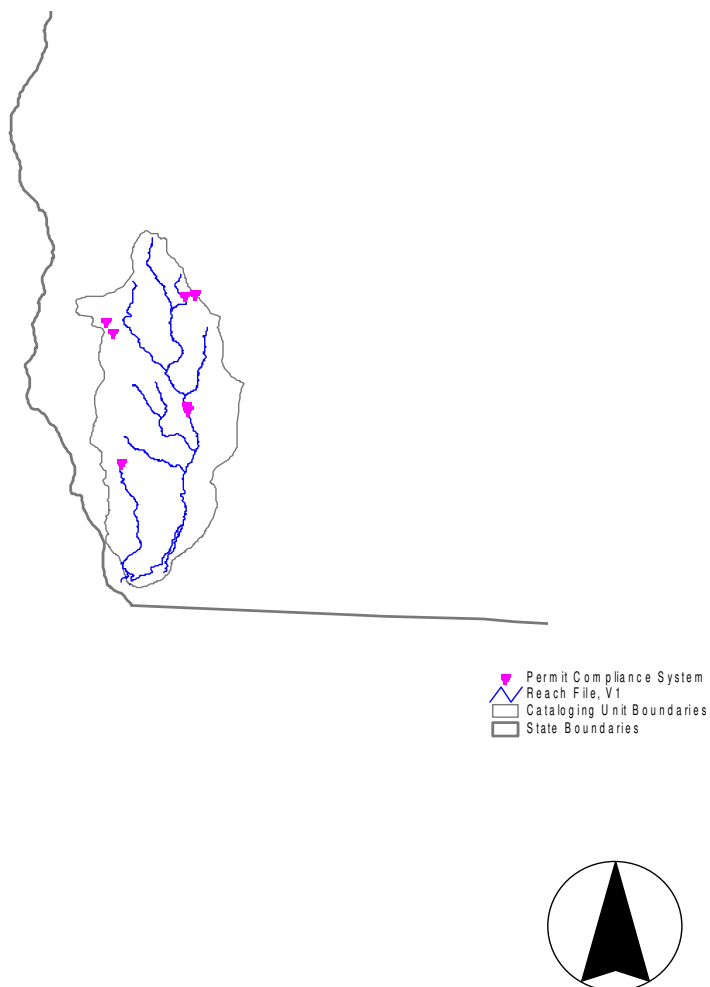
Map 2
Spring Creek Land Use Distribution

Clip3
- ⬛ Urban or Built-up Land
- 🟩 Agricultural Land
- 🟨 Rangeland
- 🟫 Forest Land
- 🟦 Water
- 🟦 Wetland
- 🟨 Barren Land
- 🟩 Tundra
- ⬜ Perennial Snow or Ice

N

W — E

S
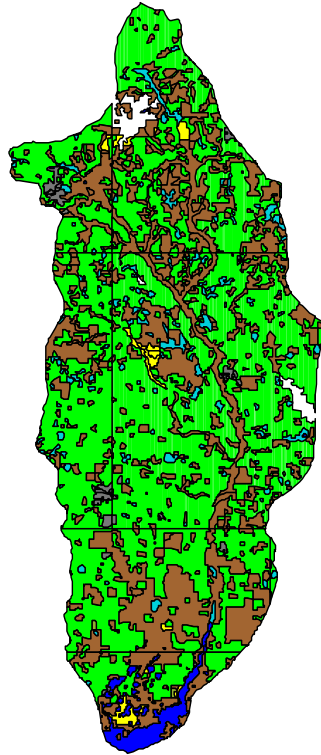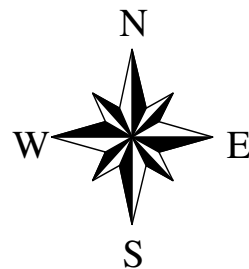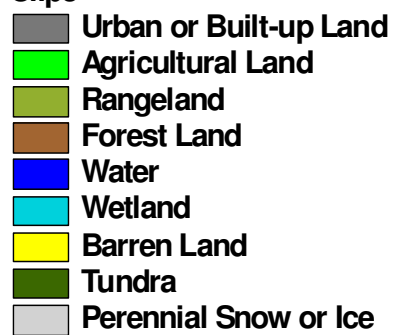
Appendix C

Spreadsheet Water Quality Model

The Spreadsheet Water Quality Model (SWQM) is used by the Alabama Department of Environmental Management to develop waste load allocations (WLAs) and TMDLs for oxygen demanding wastes. As mentioned in Chapter 3, it is based on an adapatation of the Streeter-Phelps equation. This appendix provides an overview of the model.

An excel spreadsheet is used to calculate runoff volumes and loads from identified point and nonpoint sources using land use data and watershed information such as depth, drainage area, and velocity. The advantage of the spreadsheet is that a mixture of land uses (with varying concentrations) may easily be simulated, and overall load and flow-weighted concentration obtained from the study area (Walker et al, 1989). Equation (1) shows the Streeter-Phelps relationship with the additional components to account for nitrification and sediment oxygen demand (SOD)

$$D = \frac{K_1 L_0}{K_2 - K_1}\left(e^{-K_1 t} - e^{-K_2 t}\right) + \frac{K_3 N_0}{K_2 - K_3}\left(e^{-K_3 t} - e^{-K_2 t}\right) + \frac{SOD}{K_2 H}(1 - e^{-K_2 t}) + D_0 e^{-K_2 t}$$

(1)

Where: D = dissolved oxygen deficit at time t, mg/l

L_0= initial CBOD, mg/l

N_0= initial NBOD, mg/l (NBOD = NH_3- N x 4.57)

D_0= initial dissolved oxygen deficit, mg/l

K_1= CBOD decay rate, 1/day

K_2= reaeration rate, 1/day

K_3= nitrification rate, 1/day

SOD = sediment oxygen demand, g O2/ft2/day

H= average stream depth, ft

T = time, days

This equation shows the net effect on dissolved oxygen concentration of the simultaneous processes of deoxygenating through the decay of carbonaceous organic matter, nitrification of ammonia, SOD and reaeration. The resulting pattern in DO concentration versus distance downstream from a waste source is known as the DO sag curve.

In terms of DO, the equation is the DO sag equation of Streeter and Phelps:

$$c = c_s - \left\{ \frac{K_d}{K_a - K_r} \left[\exp\left(- K_r \frac{x}{U} \right) - \exp\left(- K_a \frac{x}{U} \right) \right] \right\} L_0 - (c_s - c_0) \exp\left(- K_a \frac{x}{U} \right).$$

The maximum concentration of dissolved oxygen that can be found in a body of water after the water equilibrates with the atmosphere is known as the saturation concentration and is called DO sat. DO sat is dependent on the temperature of the water. Lower temperatures have higher DO sat's then higher temperatures. For example the DO sat at 10 C is 11.28 mg/L and the DO sat at 25 C is 9.1 mg/L. If the actual DO level goes above DO sat, then oxygen will diffuse out of the water into the atmosphere. This can happen if the DO is near DO sat and the water temperature rises quickly.

Assumptions

The following assumptions were used in the model:

- DO concentrations for incremental flow were assumed to be 70% of the saturated value at the given temperature.

- The land usage of the watersheds was assumed to be proportioned the same for the headwaters, tributaries, and incremental flow.

- Background conditions, which are in the range of 2-3 mg/L CBODu, 0.2-1 mg/l NH3, and 1-2 mg/l TON, were assumed for forest runoff.

- Ratios for $CBOD_u/NH_3OD_u$ and $CBOD_u/TONOD_u$ were calculated using water quality data for the waterbody. $CBOD_u/BOD_5$ ratio used was 2.5.

- NH_3OD_u is equal to 4.57 times the ammonia nitrogen concentration. This is because the stoichiometric requirement for oxygen in the oxidation of ammonia is 4.57 mg of O_2 per mg of NH_4^+-N oxidized.

- $TONOD_u$ is equal to 4.57 times the organic nitrogen concentration.

SOD representation

Sediment Oxygen Demand can be an important part of the oxygen demand in streams. While no field measurements were found for the Flint River Basin, there were several SOD measurements from the South 4 Basins, ranging from 0.9 to 1.9 g/m/day (EPD, 2000). The EPD found the average value to be 1.35 g/m/day for models that had mixed land uses and point sources activity. When the models were run with no point sources and all forested lands, SOD averaged 1.25g/m/day. The anthropogenic nonpoint source contributions are then accounted for in the 0.1 g/m/day.

Calibration Data

By calibration, we mean that some parameters are specified *a priori* while others are adjusted so that the model replicates available data for the baseline scenario. For the most part, the specified parameters are drawn from a literature that reports a range of values. In consequence, many parameters of the model are not known with certainty.

The calibration period used was defined by the lowest, steady flow period, the lowest DO concentrations, and the highest BOD concentrations. These conditions occurred during June - July of 2000. The stream conditions (i.e., D.O., temperature) during this period were incorporated into the calibrated model. Using an average of the measurements reported by the EPA of the South 4 Basins, SOD was set at 1.35 g/m/day.   For the point sources, the NPDES permit levels for the critical period were used for calibration.

Model inputs

The model is comprised of twelve worksheets, four of which require the input of parameters – WQ MODEL, Land Use, Chronic NH3 TOX, and Acute NH3 TOX. The input requirements of the WQ MODEL can be further divided into the following sections: reaction rates, section characteristics, headwaters conditions, tributary conditions, incremental inflow conditions, and effluent conditions.

Rates

Reaeration is the process by which oxygen enters a stream (). The Streeter Phelps equation above (equation) shows the net effect of the processes of deoxygenation through the decay of carbonaceous organic matter, nitrification of ammonia, SOD and reaeration on dissolved oxygen concentration. The result of the processes on DO concentration versus distance downstream of a waste source is known as the DO sag curve. The shape of the curve is

dependent upon the magnitude of the reareation rate relative to the concentration of oxygen demanding materials and the magnitude of their decay rates.

There are many different ways to estimate a stream's reaeration rate. Rearation rates can be entered into the spreadsheet model directly or computed using the formula developed by E.C. Tsivoglou and shown in equation (2).

(2) $K_2 = C$ (slope)(velocity)

Where: K_2 = rearation rate at 20 C, 1/day

C = Tsivolglou Coefficient

This formula, which is thought to be appropriate for smaller streams where the underlying mechanics are not well known, was used for the calibration.

The rates for CBOD decay were based on EPA region 4 guidelines. For streams with CBOD greater than 7 and less than 15, the CBOD decay is 0.4 a day.

The nitrification rate is entered for each section. Ammonia is oxidized first to nitrate, then to nitrate. The effect of the reaction rate on this oxidation is shown in equation s. The user's guide for the SWQM suggests a value of 0.3/day for streams with a slope of less than 20 feet/mile.

$$N = N_0\, e^{-k_3 t}$$

where: N = NBOD remaining at any time, t, mg/l

N_0 = initial NBOD, mg/l

K_3 = nitrification rate, 1/day

t = time, days

Though organic nitrogen does not exert a direct oxygen demand, as proteins are hydrolyzed and ammonium ions are released, an indirect demand is created. The conversion of

organic nitrogen to ammonia is represented by following equation. The TON hydrolysis rate is assumed to be 0.05/day.

$$NH3 - N = ORG \ (1-e^{-k4t})$$

where: NH_3-N = ammonia nitrogen produced by hydrolysis of organic nitrogen, mg/l

ORG = initial organic nitrogen concentration, mg/l

K_4 = organic nitrogen hydrolysis rate, 1/day

t = time, days

Temperature affects reaction rates, which are generally expressed with units of per day at 20 degrees C. If the ambient temperature is not 20 degrees C, the reaction rates must be corrected for. The model uses a modified Arrhenius equation (equation w) to adjust reaction rates according to the input temperatures.

where: K_{T2} = reaction rate at the new temperature, 1/day

$K_{20°C}$ = reaction rate at 20°C, 1/day

The temperature correction factors used in the model were consistent with those used by the QUAL2E model in BASINS. The correction factors used were listed below:

Table . Temperature Correction Factors Used in the SWQM

Rate, 1/day	*Temperature Correction Factor,*
CBOD decay, K_1	1.047
Reaeration, K_2	1.024
Nitrification, K_3	1.080
Organic Nitrogen Hydrolysis, K_4	1.047
Sediment Oxygen Demand, SOD	1.060

Section Characteristics:

The model reach for Spring and Dry Creeks consisted of 6 and 4 segments, respectively. Dividing the reaches allows the model to account for changing physical features of the streams. SWQM requires reach physiography to be defined in order to do the in-stream mass balance and routing calculations. The data required includes depth and elevations upstream and downstream. This information was obtained from the National Hydrography Set, which was displayed in Arcview.

The velocity at which a stream is flowing is another important factor affecting the DO sag curve. Generally, higher velocities result in higher reaeration rates and a less pronounced "sag" in the DO sag curve. Velocity at any given point on a stream can be computed using the continuity equation shown in equation (3).

(3) $V = Q/A$

Where: V= velocity, f/s

Q = stream flow, cfs

A = cross sectional area. square feet

Velocity through any given stream reach is usually a function of stream flow and can be written in the form of Equation (4).

(4) $V = aQ^b$

Where: V= velocity, f/s

Q = stream flow, cfs

a = coefficient of velocity versus flow relationship

B = exponent of velocity versus flow relationship

The model has an option to compute stream velocity using an empirical relationship developed by EPA for streams in the Southeast. This is shown in equation (5).

(5) $V = 0.144Q^{0.4} (Slope)^{0.2} - 0.2$

Headwater Conditions

The headwater conditions are the physical and chemical stream characteristics immediately upstream of the reach to be modeled. Inflow rates for the headwaters were based on the USGS daily flow data for Spring Creek at station x. Based on the flow per square mile of drainage area for the gage, flows at the upstream and downstream ends of the subsegments were calculated using published drainage areas for the watersheds. The headwater flow used for calibration was the 1Q10 value.

Temperature input is based on historical weather data, and is assumed to be the 90th percentile seasonal temperature for the reach being modeled. Long term temperature data from the gage at Iron City was used to calculate a 90the percentile summer temperature of 25 degrees. Because the segments have a year round standard for DO, a winter projection simulation was not performed. As discussed earlier, the most critical time of year for meeting a DO standard is the period of high temperatures.

Headwaters water quality parameters include CBODu, NH3-N, and total organic nitrogen, which are calculated by the model.

Incremental Inflow:

This is the total flow at the downstream end of the subsegment minus the sum of all the headwater and tributary inflows. This total amount of incremental inflow was assumed to be uniformly distributed along the entire length of the channels being modeled; therefore, the inflow for each reach was proportional to its length. The D.O. concentration for incremental flow was

set at 70% of the saturation concentration at the given temperature, which is a conservative estimate.

Tributary conditions:

The methodology behind and parameters for the tributaries are the same as headwaters and incremental inflow. While Dry Creek did not have any significant tributaries, Spring Creek has four. However, the critical conditions for the model are when the flows are so low that the tributaries do not contribute a significant amount. Dry Creek was modeled as a tributary to Spring Creek. This is partly due to the fact that Blakely Pond A and Blakely WPCP are connected to Spring Creek by Dry Creek.

Point Sources

There are several NPDES permits that discharge into or upstream from Spring Creek and Dry Creek. Table 2 includes contains the June 2000 permit limits that were used for model development. They include Arlington Pond #1, Colquitt WPCP, Blakely WPCP, Blakely Pond A, and Blakely Pond B. All of these are identified as sewerage systems. Through conversations with the plant engineers, we obtained information on the actual monthly average discharge. This is also included in Table 2.

NPDES permit	Facility Name	County	Flow (mgd)		DO (mg/L)		BOD (mg/L)		NH$_3$ (mg/L)	
			Permit leves	Monthly average	Permit levels	Monthly average	Permit Levels	Monthly average	Permit levels	Monthly average
GA0031968	Blakely Pond A	Early	0.12		NP		30		NP	
GA0031976	Blakely	Early	0.12		NP		30		NP	

	Pond B							
GA0026204	Arlington Pond #1	Early	0.10		NP		30		NP	
GA0050075	Arlingt on Pond #2	Calhoun	0.06		NP		30		NP	
GA0047252	Colquitt WPCP	Miller	0.40		5.0 (min)		30		15	

The locations of these point sources were determined using GIS software and can be seen in figure y. In the case of Dry Creek, all three of the relevant point sources are located upstream of the source. Because the model will only allow one point source per stream section, all three of these were combined and a loading factor was calculated for them. In order to include the Blakely ponds in the Spring Creek model, a branch was added above the Dry Creek tributary through which the effluent flows to get to Spring Creek. The CBODu concentration was set to the BOD 5 limit times 2.5 to convert to CBODu. The organic nitrogen was assumed to be half of the ammonia nitrogen based on typical values for mechanical treatment systems.

Nonpoint Sources

Land use percentages and mean runoff concentrations are used to calculate the CBODu, TON, and NH3-N concentration throughout the streams. The model also allows the user to assign organic loadings on the basis of land use. The model assigns differing pollutant concentrations to flow from headwaters, tributaries, and incremental inflow (all natural stream flow not considered by the other two sources of natural flow) according to the major land use

percentages in the watershed. The selected stream reach is then divided into individual segments in order to account for changing physical features of the stream. These would include the addition of flow and pollutants from tributaries, incremental inflow, and point sources, changes in stream slope, velocity, and any of the reaction rates.

The land use characteristics of Spring Creek were determined using data from Georgia's Multiple Resolution Land Coverage (MRLC), which was produced from Land sat Thematic Mapper digital images developed in 1995. Table 1 lists the land use distribution of the watersheds. A more detailed distribution of land use can be found in appendix X.

Table 1. Land Uses

Stream	**Total Contributing Area (Acres)**	**Row Crops (%)**	**Pasture (%)**	**Forest (%)**	**Wetland (%)**	**Built-Up Impervious (%)**	**Built-Up Pervious (%)**
Spring Creek	234,388	38.3%	12.4%	29.6%	15.2%	0.6%	3.9%
Dry Creek	66,861	33.2%	13.5%	29.0%	18.6%	1.3%	4.4%

Both Spring Creek and Dry Creek watersheds have mixed land use, as can be seen on map x. The two major land uses are forest and cropland, which accounts for about 40% of watersheds. Compared to other land uses organic enrichment from forested land is normally considered to be small. This is because forested land tends to serve as a filter of pollution originating within its drainage areas. However, organic loading can originate from forested areas due to the presence of wild animals such as deer, raccoons, turkeys, waterfowl, etc. Control of these sources is usually limited to land management best management practices (BMPs) and may

be impracticable in most cases. In contrast to forested land, agricultural land can be a major source of organic loading.   To better understand the potential loadings from the cropland, data was obtained from AGSTATS.

Since AGSTATS gives the crop acreage in the counties, not subwatersheds, the percentages of the various row crops for all the counties in the watershed was calculated. Those percentages were used to determine the number of acres planted in each row crop, and the percent irrigated. Spring and Dry Creek, which encompass the same counties, were assumed to have the same percent distribution. Table Y shows the distribution of row crops.

Crop	% of total acres	Acres Planted		% irrigated	Acres Irrigated	
		Spring	Dry		Spring	Dry
Peanuts	32%	28,726	9,192	41	11,777.66	3,768.72
Cotton	50%	44,885	11,099	37	16,607.45	4,106.63
Corn	17%	15,261	3,773	55	8,393.55	2,075.15
Soybeans	1%	897	222	10	90	22

Sensitivity analysis

All modeling studies necessarily involve uncertainty and some degree of approximation. It is therefore of value to consider the sensitivity of the model output to changes in the model coefficients, and in the hypothetical relationships among the parameters of the model. The sensitivity analyses were performed by allowing the model to vary one input parameter at a time

while holding all other parameters to their original value. The calibration simulation was used as the baseline for the sensitivity analysis. The percent change of the model's minimum DO projections to each parameter is presented in Table w. Each parameter was varied by 30% except for temperature, which was varied 2%. Values reported in Table w are sorted by percentage variation of average DO from smallest percentage variation to largest.