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THE ECONOMIC AND POLITICAL FEASIBILITY OF INCORPORATING WETLANDS INTO A WATER QUALITY TRADING PROGRAM AT THE WATERSHED SCALE

Gregory Mikota ii
Clemson University, mikota@clemson.edu

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Gregory Michael Mikota II
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Accepted by:
Dr. James B. London, Committee Chair
Dr. Robert H. Becker
Dr. M. Grant Cunningham
Dr. Stephen J. Klaine
Dr. Alan R. Johnson

ABSTRACT

Incorporating the use of wetlands as a technology to enhance water quality trading programs requires the consideration of many political, economic, ecological, and legal issues. It is desirable to include wetlands as a nutrient reducing practice in certain water quality trading programs because these wetlands can provide additional benefits, beyond those of other technologies, such as carbon dioxide sequestration and increased habitat area and biodiversity. There is a great deal of interest from policymakers in extending the implementation of the United States EPA's *Water Quality Trading Policy* to include the use of constructed or restored wetlands. However, is the incorporation of wetlands technology to enhance water quality trading programs economically and politically feasible at the watershed scale?

This study evaluated the economic and political feasibility of establishing water quality trading programs that incorporate constructed or restored wetlands as a type of nutrient abatement technology. A review of the current literature concerning water quality trading and wetlands combined with selected case studies was used to identify critical knowledge gaps that could encumber the implementation of trading programs for both point and non-point sources as stakeholders. Four case studies provided various examples from a national perspective that illustrate the feasibility of implementing a wetlands and water quality trading program based on current practices managed by existing programs.

DEDICATION

I dedicate this work to: my parents, Gregory and Cathie Mikota; my brother, Geoffrey Mikota; my sister, Christie Wright; and my girlfriend, Sarah Woods. Their encouraging love, support, and patience have enabled me to accomplish this goal.

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CHAPTER 1

INTRODUCTION

Statement of the Problem

Modern industrial societies like the United States face dual challenges in coordinating efforts to protect vital elements of the natural environment that are essential for life. Current growth pressures and complexities of common-access resources have generated creative new approaches to address environmental management problems and the notion of market failure in effective and equitable ways.

The first ongoing environmental challenge deals with the magnitude of some emerging threats to environmental quality on a global scale with specific emphasis on atmospheric changes and loss of biological diversity. This challenge cannot be easily handled by independent nations throughout the world. These global problems need global solutions.

The second challenge deals with the direct regulatory approaches that implement uniform control methods to decrease pollution. Regulatory approaches are utilized to combat the issues that arise from common pool resource problems. In many cases common pool natural resources like air and water tend to be over-utilized and public goods tend to be undersupplied by the marketplace because individual free-riders within an institutional setting are hard to limit or exclude. In the past, ambient air or water quality legal standards have been set to allow for maximum concentration levels of pollutants. These standards are then enforced by making specific technological choices or certain types of behaviors mandatory. These conventional methods to regulating the environment have been referred to as "command and control" regulations because they provide minimal flexibility in the means of achieving pollution emission targets. These regulations have been established in order to contend with market failures that do not capture the expensive effects of pollution discharge through the prices and costs generated during the market process. The inability to account for pollution discharge through production leads to increased negative external pollution effects that are not internalized by producers or consumers. A vast number of negative environmental externalities attributable to market failure have been limited by some extent through the use of command and control methods, but increased reliance on this type of approach could prove to be expensive, disruptive, and counter productive to overall economic well-being (Buchanan & Tullock, 1975). These regulatory approaches have been used in an attempt to counteract market and policy failures that produce negative external

To combat this second challenge many policy analysts have in recent years advocated the use of incentives or market-based approaches to improve environmental protection policy tools. Market-based theory and practical environmental policy are currently separated by information and transaction cost barriers, but the continual incremental approach to environmental policymaking over the past twenty years has resulted is various types of market-type innovations within traditional regulatory frameworks at all levels of government. The most well-known example of this type of approach is the sulfur dioxide trading program established under the 1990 Clean Air Act Amendments.

After learning from successful experiences with emission trading programs focused on reducing acid rain, the United States Environmental Protection Agency (USEPA) now actively supports the application of emission trading to water quality. This approach seems quite attractive because it could provide financial incentives for increased pollution control activities in unregulated sectors. Agriculture and urban runoff are major contributors to effluent load levels in many watersheds, but with a few exceptions these sources do not fall under regulatory guidelines because of political sensitivities and perceived monitoring difficulties (Crutchfield, 1994).

Vast improvements in water quality throughout the United States over the past three decades can be traced primarily to conventional regulation approaches and financial support given to point sources, municipalities, and traditional large pipe dischargers. Regulatory initiatives concerning surface water policy in the United States have been guided by the 1972 Federal Water Pollution Control Act, also known as the Clean Water Act (CWA). This, act and amendments in 1977 and 1987, set the overarching principles and implementation mechanisms that direct the efforts to prevent water pollution. The USEPA has been granted a primary tool to regulate water pollution through the National Pollutant Discharge Elimination System (NPDES). The NPDES requires point sources to obtain state and USEPA approved permits of defined number of pollutant discharges. The USEPA and other federal agencies use a variety of subsidies and grants to mitigate point source (point source) and non-point source (non-point source) pollution that does not fall under the NPDES criteria.

The combination of regulations and subsidies has met with considerable success in controlling and reducing point source pollution during the past thirty years. By 1990 87% of the major municipal facilities and 93% of the major industrial plants were in compliance with NPDES standards (King, 2003). However, many waterbodies within the United States are not in good condition. In 2000 the USEPA's *National Water Quality Inventory Report* claimed that approximately 40% of assessed rivers, 45% of assessed streams, and 50% of assessed lakes in the United States did not meet water quality standards. The report lists sediments, bacteria, metals, and nutrients as the major causes of impairment. Agricultural non-point source pollution and urban runoff are the two single greatest sources of pollution according to the USEPA's report. For example, one of the most polluted areas in the United States, the Chesapeake Bay watershed, receives most of its effluent from agriculture: 40.8% of nitrogen inputs and 47% of phosphorus inputs. Point sources contribute 22.1% of nitrogen inputs and 22.3% of phosphorus inputs (King, 2003).

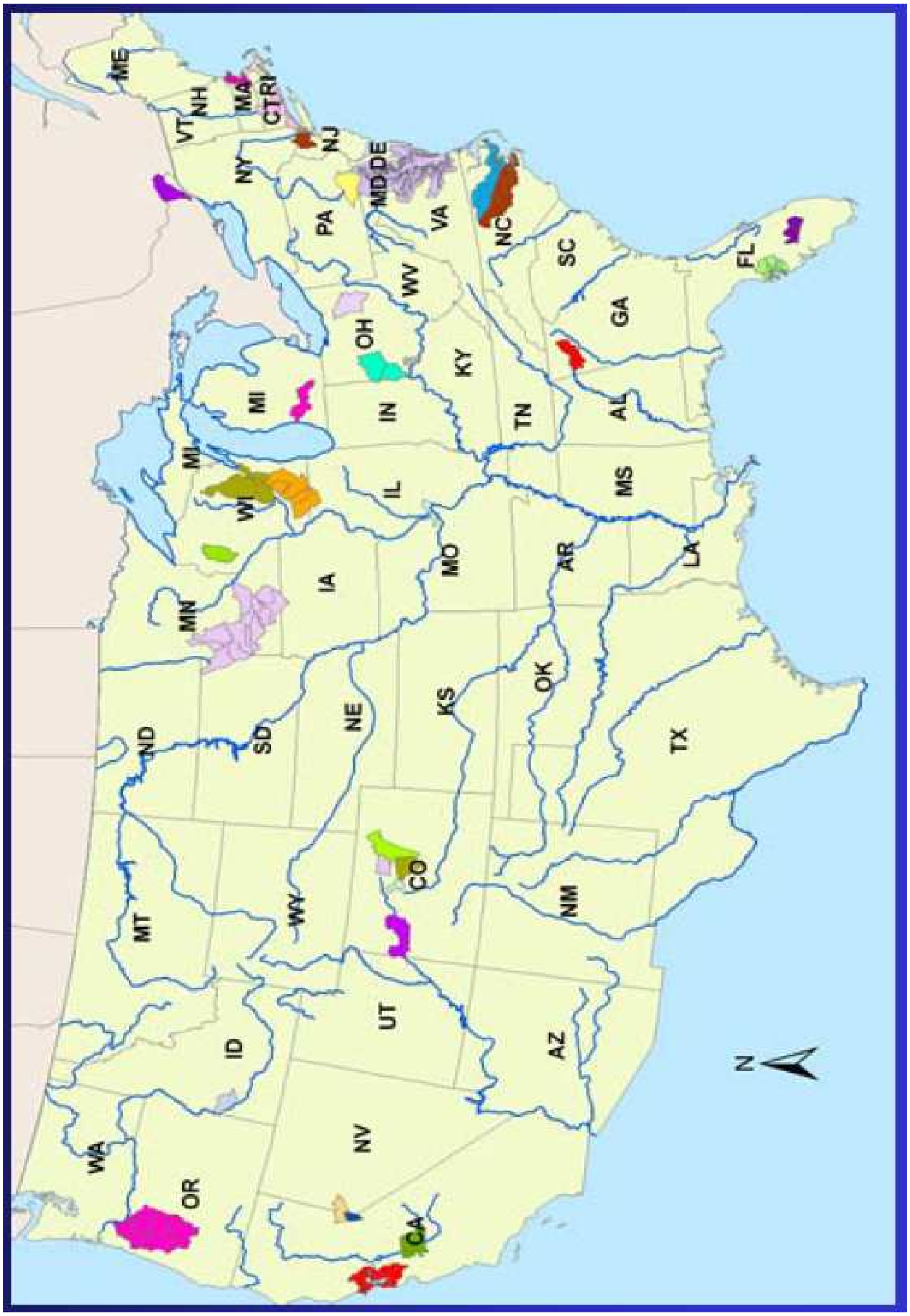
Various pollutants that are associated with non-point source pollution include fecal bacteria, toxic organic compounds, heavy metals, suspended solids and sediments, phosphorus, nitrogen, and other oxygen-demanding organic material. Unlike point source pollution, which tends to be a steady discharge, non-point source pollution occurs during different times based on periods of rainfall or the melting of snow. If unchecked, these non-point source pollutants eventually reach our lakes, rivers, oceans, and even underground sources of drinking water as they seep into the ground. "Despite the expenditure of hundreds of billions of dollars over the last 30 years, the 1972 Clean Water Act goals of fishable and swimmable waters have not been achieved, largely because contaminants from diffuse [non-point] sources have not been controlled successfully" (National Research Council, 2001).

Recent water quality challenges in the United States have stimulated sustained interest from policymakers to incorporate the use of non-traditional market mechanisms to lower costs of compliance and improve aquatic environments. This type of approach allows facilities with high pollution control costs to purchase lower cost pollution reduction from another source to meet their effluent reduction requirements (USEPA, 2004). Water quality trading is conceptually similar to air emissions trading, but effluent trading has lagged in development. A few effluent trading programs were developed in the early 1980s and during the 1990s, but in recent years new interest has sparked conversations about policy changes that would improve the capabilities for local and state authorities to implement water quality trading programs.

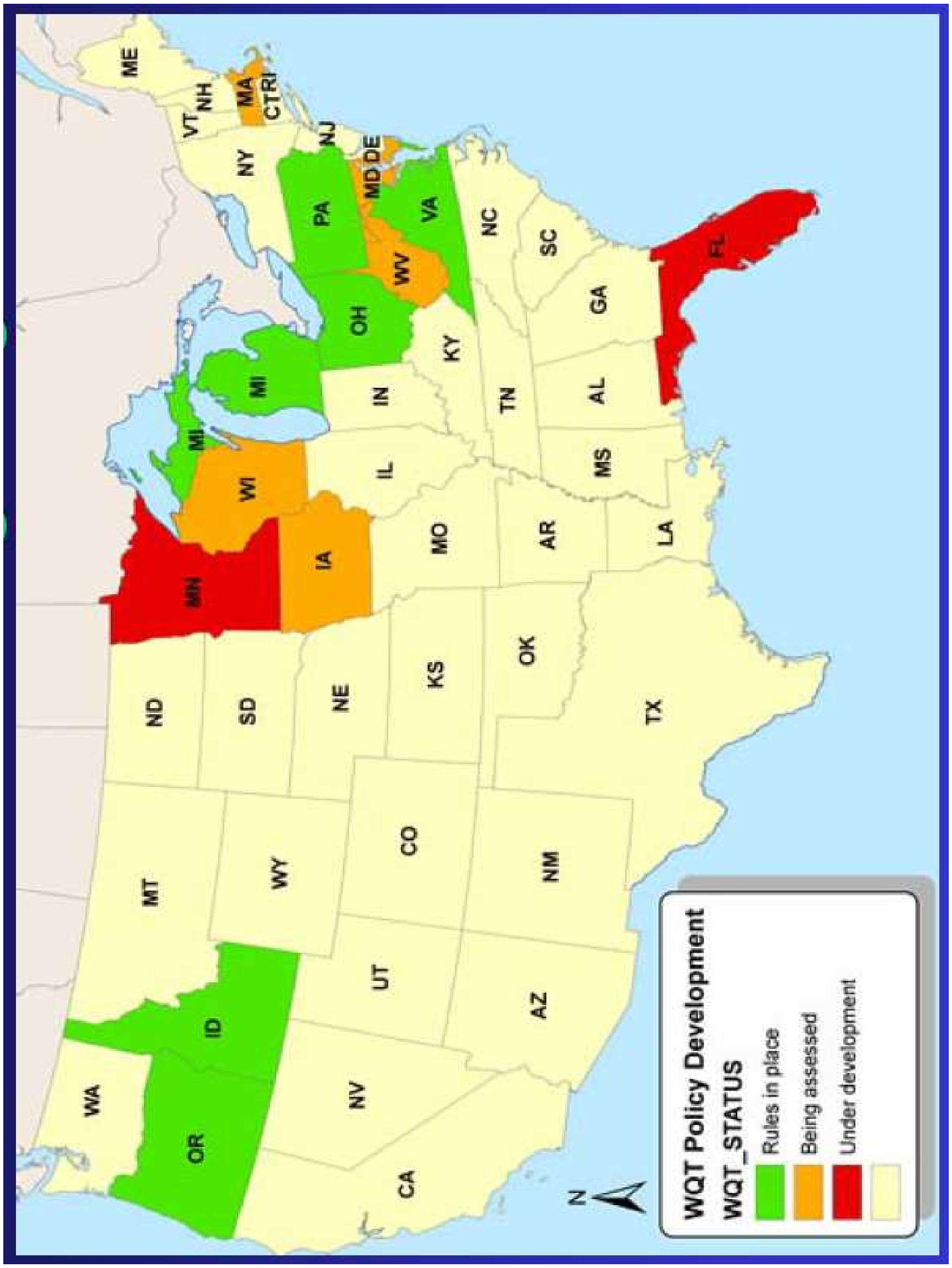
In January of 2003 the United States Environmental Protection Agency issued rules through its water quality policy statement to encourage the trading of nutrients and sediments among point and non-point sources. The policy states that its purpose is "to encourage states, interstates agencies and tribes to develop and implement water quality trading programs for nutrients, sediments, and other pollutants where opportunities exist to achieve water quality improvements at reduced costs" (USEPA, 2005). In that document the USEPA states that market-based approaches like water quality trading provide greater flexibility and have the potential to achieve levels of environmental benefits that would not otherwise be attained under a traditional command and control approach. The policy focuses on the idea that different sources within a watershed may face significantly different costs to control the same pollutant. When working with non-point source pollution problems, the USEPA is required to work with individual states and local agencies because of the provisions defined in the revision of the Clean Water Act in 1987. The law leaves non-point source control planning to the states and local agencies because of local environmental and economic considerations. "The actual site-specific selection of particular management practices to control non-point source pollution (called "Best Management Practices") will involve local environmental and economic considerations, as well as considerations of effectiveness and acceptability of the practice (USEPA 1984a)" (Portney, 2000).

In order to provide information to state and local policymakers the USEPA published a *Water Quality Trading Assessment Handbook* in 2004 (USEPA, 2004). This guidance is very general, and it focuses on setting the institutional arrangements of the necessary processes that take place during trading. But, the handbook does not discuss the changes in policy that are necessary in order to create supply and demand for these types of markets. Despite the guidance and support by USEPA, nutrient trades have been relatively scarce to date. Scarce pollution credit supply from non-point sources and lackluster credit demand from point sources are primarily responsible for this weak market performance (King, 2005; King & Kuch, 2004). To a limited extent point source to point source trading has been effective, and programs have shown signs of further development. In 2004 there were more than 70 water quality trading initiatives in the United States (Breetz, 2004). (Map 1.1 and Map 1.2) This rise in interest has been noted, but there are still significant obstacles that stand in the way of many water quality trading approaches.

King and Kuch (2003) found in an analysis of thirty-seven approved trading sites in the United States that only three programs had engaged in water quality trading. At the time, six nutrient trades had occurred, and only one involved a non-point source. But, even with this lackluster performance, there has also been a considerable amount of interest given to extending the implementation of the USEPA's *Water Quality Trading Policy* to include point source to non-point source trades. Over the past few years particular interest has been focused on the use of constructed or restored wetlands, in addition to traditional abatement technologies, as non-point sediment and nutrient credit producers within a water quality trading program (Raffini & Robertson, 2005). This type of approach seems attractive because it could inevitably assist in enhancing, restoring, and protecting wetlands as well as improve water quality. In certain cases the approach could provide additional benefits like carbon dioxide sequestration or habitat creation which is specifically supported by the USEPA's *Water Quality Trading Policy*.



Map 1.1. Watershed Scale Water Quality Trading Programs in the United States (2006)
(Source: U.S. EPA Office of Water: Office of Wetlands, Oceans, and Watersheds, (2006)



Map 1.2. U.S. States that have Developed Water Quality Trading Program Rules (2006)
(Source: U.S. EPA Office of Water: Office of Wetlands, Oceans, and Watersheds, (2006)

Purpose of the Study

Incorporating the use of wetlands as a technique to enhance water quality trading programs requires the consideration of many political, economic, ecological, and legal issues. Despite theoretical arguments for the implementation of water quality trading programs, actual trades have hardly taken place. What economic and institutional arrangements are necessary in order to allow the programs to operate efficiently and effectively? In relation to water quality trading, policy analysts understand the importance of wetlands in natural systems as nutrient transformers or immobilizers, but there is much debate concerning how water quality aspects of wetlands function at the watershed scale. Therefore, is the incorporation of wetlands technology to enhance water quality trading programs economically and politically feasible at the watershed scale?

To assess this question this study evaluates the economic and political feasibility of establishing water quality trading programs that incorporate constructed or restored wetlands as a type of nutrient abatement technology. A review of the current literature concerning water quality trading and wetlands combined with selected case studies are used to identify critical knowledge gaps that could encumber the implementation of trading programs that include both point and non-point sources as stakeholders. Four case studies provide various examples from a national perspective that illustrate the feasibility of implementing a wetlands and water quality trading program based on current practices managed by existing programs. Economic and institutional impediments that are brought forth from the literature provide outline for analysis of these cases and a comparison synthesis, conclusions, and policy recommendations provide a better understanding of the knowledge obtained from the literature and cases.

CHAPTER 2

REVIEW OF THE LITERATURE

History of U.S. Surface Water Quality Policy

Protecting water quality was not a major concern to the federal government until the beginning of the 20^{th} century. Three early acts defined federal government involvement with water quality issues. The first federal act that dealt with controlling water pollution was the Rivers and Harbors Act of 1899. This act was established in order to prevent impediments to navigation and serves as a reminder that at one time the physical volume of waste in waterways threatened to block some rivers and channels. The Public Health Service Act of 1912 allowed federal investigation of water pollution that affected public health, and the Oil Pollution Act of 1924 disallowed the discharge of oil into Coastal Waters (Portney, 2000). These early acts set precedent, but none of them were broadly interpreted or widely enforced in a way that greatly influenced water quality policy.

As the U.S. population grew and waterborne disease rates increased the federal government began to become more involved with water quality issues. The Water Pollution Control Act of 1948 was the first federal legislation to deal explicitly with conventional forms of water pollution. This act authorized the federal government to make loans to municipalities in order to construct sewage treatment facilities, and it also allowed the federal government to engage in investigation, research, and surveys dealing with water pollution problems. Prior to 1948 there was no direct federal role in water pollution control policy, and the new law opened the door for future federal organizational responses to state actions concerning water quality policy (Kneese, 1975).

The 1956 amendments to this act strengthened the role of the federal government, but primary responsibility for dealing with water quality issues was left to the states. The Water

Pollution Control Act Amendments of 1956 established a mechanism for joint state/federal enforcement directed by the Public Health Service. This act also established a grant program that allowed municipalities to share costs to construct sewage treatment facilities (Kneese, 1975).

The first federal act that mandated state actions with respect to water pollution policy was the Water Quality Act of 1965. The Water Quality Act of 1965 directed states to set minimum water quality standards for portions of interstate waters within individual borders. Individual state monitoring, standard-setting, and enforcement became significant problems in relation to this act, and many policymakers viewed this approach as ineffective and unworkable (Zwick, 1971). There were three main reasons why this particular arrangement did not work. First of all, there was a great deal of difficulty experienced by states in determining how much each individual source would have to cut back in order to meet quality standards. Secondly, state agencies did not have the organizational structures in place to handle the enforcement issues associated with discharge. Implementation plans were needed in each state to do this. Lastly, states varied enormously in their commitment to pollution control objectives; the expertise of their personnel; the monetary resources that allocated for implementation monitoring, and enforcement; and their willingness to compete for new industry by offering a lax regulatory environment (Portney, 2000).

In 1972 Congress passed the Federal Water Pollution Control Act (FWPCA). This act established new federal goals and standards for water quality, set deadlines for cleanup actions, and provided new means in which new tools could be used for regulation and enforcement. All discharges of pollutants into navigable waters were to be eliminated by 1985 (Portney, 2000). This legislation specifically focused on point source pollution while establishing a national goal for water pollution policy. One of the major departures of the FWPCA from past policies was that the major responsibility for issuing permits to dischargers was shifted to a federal agency: the United State Environmental Protection Agency (USEPA) (Portney, 2000).

In 1977 and 1987 Congress adopted revisions to the FWPCA. The amended act, now known as the Clean Water Act (CWA), expanded USEPA's powers to address non-point pollution through voluntary programs. The 1977 revisions further defined the distinction between conventional pollutants (suspended solids and organic matter) and toxic water pollutants. The revisions also established new technology procedures and deadlines for determining toxic effluent limitations. The 1987 act established funds to continue support of sewage treatment plant construction and set new requirements for states to establish non-point source pollution control programs.

Additionally, Congress passed legislation creating the Safe Drinking Water Act in 1974 and revised it in 1986. Both of these mandates focus on mitigating existing pollution and constructing new wastewater and drinking water treatment plants, but these laws can also potentially fund initiatives focused on protecting source waters through land conservation initiatives.

The USEPA funds three water quality programs under the CWA: the clean water state revolving fund, non-point source program, and national estuary program. In 2003, states were awarded over one billion dollars and provided almost five billion dollars in assistance for wastewater, non-point source, and estuary projects. Five percent of the clean water state revolving funds are used for mitigating non-point source pollution while the other ninety-five percent goes toward wastewater treatment infrastructure (USEPA, 2007). The non-point source program receives less than twenty percent of the clean water funding despite the fact that this type of effluent now accounts for over sixty percent of all effluent in the U.S. (USEPA, 2007).

Water quality throughout the United States has improved since the passage of the Clean Water Act in 1972 largely because of point source reductions of toxic and organic chemical loadings. Toxic pollution discharges have been reduced by an estimated billion pounds per year since the early 1970s (Adler, 1994). Between 1970 and 1992 sewage treatment plants show a reduction in ammonia (Mueller & Helsel, 1996). A widely applied national surface-water monitoring network has provided evidence that fecal bacteria and phosphorus concentrations have declined (Knopman & Smith, 1993; Mueller & Helsel, 1996). Additional information provided by case studies and anecdotal information suggest that reductions in pollutants since the beginning of the 1970s have improved the health of aquatic ecosystems (Knopman & Smith, 1993).

Great strides have been made concerning water quality during the past few decades, but significant challenges still remain. In fact, the FWPCA set a national goal of achieving zero discharge by 1985, but over the past decade significant evidence suggests that we are moving in the opposite direction away from this goal. More than one-third of all surface waters are deemed impaired in the United States, but larger regional problems continue to experience significant environmental degradation. The Chesapeake Bay is the largest estuary in the world and it has experienced significant water quality problems linked to nutrients including phosphorus and nitrogen. Excessive fish kills in 1997 were attributed to inflated nutrient levels that provided a catalyst for the outbreak of a micro-organism, pfiesteria (Mlot, 1997). Poor water quality is also the suspected cause of declining shellfish harvests (Chesapeake Bay Commission, 2002). The Gulf of Mexico has also experienced dramatic water quality problems caused by increased levels of nitrates. The oxygen-deficient “dead” zone is now larger than 7,000 square miles (Rabalais, Turner, & Wiseman, 1997), and is growing in size from year to year. The major source of increased nitrate levels has been identified as fertilizers from the Upper Mississippi Basin

(Antweiler, Goolsby, & Taylor, 1995). Since 1954 the amount of nitrate that is discharged into the gulf has tripled (Goolsby & Battaglin, 1997).

One of the reasons why the Clean Water Act has failed to achieve proposed goals is that water quality problems have gotten bigger with increased population pressures, an expanding economy, and more demand placed on watersheds as receptors of pollution while also providing drinking water, recreation, and other services. The second major reason for failure concerns fractured policy formulation and implementation. At the federal level both the USEPA and the U.S. Department of Agriculture (USDA) address water quality problems in splintered and inconsistent ways. Both agencies use a variety of policy tools, including grants, regulations, and subsidies, to decrease point and non-point source discharges, but most agency efforts have been individually undertaken and disjointed from other agency initiatives. This gap between the two agencies concerning water quality may be shrinking because of increasing concerns over environmental degradation to our watersheds and the potential opportunities to make significant positive impacts linked to contemporary dynamic polices like water quality trading. On October 13, 2006 the Assistant Administrator of the Office of Water at the USEPA, Benjamin Grumbles, and the Under Secretary of Natural Resources and the Environment at the USDA, Mark Rey, signed a partnership agreement to encourage water quality trading nationwide (USEPA, 2007). Policymakers are extremely interested in moving forward with water quality trading initiatives. Positive outcomes from trading, as an adjunct to regulation, could lead to lower compliance costs while meeting or exceeding water quality goals.

The evolving nature of water quality challenges in the United States over the past two decades has spurred a reassessment of conventional approaches to water pollution control. Traditional regulatory approaches like the National Pollutant Discharge Elimination System (NPDES) cannot easily address non-point source pollution without the development of new statutory authority, and federal grants and subsidies have proved to be insufficient in mitigating non-point source pollution. One prominent policy solution to regulatory constraints and subsidy limitations is water quality trading. Water quality trading has been promoted by economists for years, and during the past few years many political leaders and upper-level government officials have endorsed the idea as well. Theoretically, a water quality trading program allows regulated point sources with high pollution costs to buy pollution reduction credits from unregulated non-point sources with low control costs. From a conceptual standpoint water quality trading achieves pollution control targets at lower total cost and reduces non-point source pollution. But, in practice, the future of water quality trading depends on the resolution of many political, economic, ecological, and legal issues.

Economic Rationale for Water Quality Trading

In order to better understand this increased interest in water quality trading policy it is beneficial to review how market-based strategies evolved. Incentive based solutions to environmental externalities have been around for quite some time. In the early part of the 20^{th} century A. C. Pigou (1920) proposed a policy of using corrective taxes to counteract externalities. He suggested that a specific externality could be internalized if taxes were set at a rate that would discourage the output of a pollutant through additional cost. One of the major problems with this type of approach is the concern that in many cases it is extremely difficult to isolate and calculate a tax that is equal to the damage incurred by an externality. Other researchers like Baumol (1972) expanded upon Pigou's work by demonstrating that a per-unit emission tax on polluting firms does not have to be set at the optimum level provided that the government continuously adjusts that tax. Researchers like Collinge and Oates (1982) have contributed to the Pigouvian tax policy instruments by presenting the notion of rental emission permits that theoretically follow the marginal social damages curves. Therefore, in theory, marginal abatement costs would not need to be calculated, and permit charges would be set equally to the external costs to society. Pigovian taxation has greatly influenced the economic analysis of public policy since the early part of the 20th century. Today, there are many types of emission and pollution taxes set around the world from emission taxes on industry in California to carbon taxes in Scandinavia. Other notable work is more critical of Pigovian taxes emphasizing the distortion effects that occur in the long-run (Gould, 1977; Rose-Ackerman, 1973). This research claims that per-unit taxes raise a firm's average cost curve.

There are also other methods of reducing pollution through incentives other than imposing a tax or developing a charge for the purchase of permits. Midway through the 20th century economists began to seek out other ways of internalizing the cost of environmental externalities. Ronald Coase (1960) arrived at a very different way of looking at pollution problems after studying the Pigovian solution. Coase argued that if property rights are clearly defined, the costs of negotiating are low, and the number of affected parties is small, then externalities can be eliminated through private bargaining solutions. John Dales (1968) expanded the idea of market-based approaches to the environment by suggesting that transferable property rights could be used to promote environmental protection at a lower cost than conventional standards. Tietenburg (1985) quantitatively compared the creation of environmental markets with traditional command and control regulation to reveal that substantial economic and environmental gains could be realized with a market trading system.

Many of these early market based studies focused on economic cost savings and efficiency, but they failed to consider the underlying cost estimation concerning uncertainties that pertain to the environmental modeling of a natural process. In most respects these studies did not attempt to identify the political constraints that may be impressed upon a market-based institutional design. These studies also failed in many ways to accurately show how the promotion of social goals such as equity can be achieved without destroying the cost saving efficiencies or the incentives that help to improve and expand abatement practices. Policymakers are interested in incorporating institutional structures and stakeholder participant meetings into the development of incentive based policy applications in order to evaluate the total private and public good that is being generated from exchanges within the marketplace. In many cases, cost is the determining factor for implementation. Minimizing the cost of abatement is a very powerful driver when developing plans for environmental protection, but the effects of a proposed program must also be viewed from a political and equitable viewpoint as well.

For a trading market to reach the optimal allocation of pollution control, it will need well-defined property rights, a large number of buyers and sellers, good information about consumption and production, low or no transactions costs, and rational behavior by the stakeholders in the marketplace. Market failure is avoided if most of these conditions are met, but this is not usually the case for markets that trade pollution control practices, environmental goods, or services (Fullerton & Stavins, 2000). Many problems may arise because the assessment of benefits and costs may be hindered because the environmental goods at hand are extremely complex in nature. In many cases concerning trading programs there is typically not enough reliable information to estimate the optimal allocation of pollution control, and in some cases monitoring and verification expenses may be too great for a market to even exist. In spite of these problems a trading market may prove to be more cost effective than traditional command and control programs. In fact, the air quality trading programs have experienced very encouraging results from both a cost perspective and a pollution reduction perspective.

Transition From Air Quality Trading to Water Quality Trading

Pollution permit trading began in the United States in the mid-1970s as air quality offset programs. Regional areas that did not meet air quality standards viewed offset programs as a way to move toward improving their environment while allowing for economic development. Offset programs have been localized to specific areas. Over 90% of the 10,000 offset trades during the past 30 years have occurred in California (USEPA, 2001). In 1977 Congress provided its formal approval to offsets in the 1977 amendments to the Clean Air Act. These amendments allowed emission reduction activities to be banked for the storage, consolidation, and future sale of credits.

In 1978 the USEPA introduced its bubble policy. The bubble policy allowed polluting firms with multiple emissions points in the same facility to face a single aggregate emission limit instead of separate limits for each source. The process was hampered by a cumbersome approval process, but some estimates revealed an estimated $435 million cost savings by 1986 (Hahn, 1989).

In 1986 USEPA presented the Air Emissions Trading Policy Statement, and air trading programs began to be implemented around the country. Two of these programs that were successful were the lead credit trading program and the acid rain program. The original lead trading program limited averaging of lead concentrations to individual refineries. USEPA would later expand the lead credit trading program to allow averaging among refineries. This change in policy in many respects was very similar to the bubble policy. The averaging of concentrations transformed certain areas into active lead credit markets. This development was enhanced by the permission by the USEPA to bank credits in 1987. In the next couple of years over 10 billion grams of lead were banked and these provisions saved $226 million in abatement costs (USEPA 2001).

The acid rain program was also successful in the late 1990s. Sulfur dioxide credits were traded and banked by polluters, and pollution rates and costs of abatement declined rapidly. The difference between the lead credit trading program and the acid rain program can be distinguished by observing the process of how the pollution credits were traded. The acid rain program was a cap-and-trade program, and the lead credit trading program was a rate-based program. The acid rain program had an absolute cap, and trades had to take place within that capped amount. The lead trading program was set in terms of concentrations. A refinery could produce as much lead pollution as it desire as long as the refinery's average lead concentration met USEPA's limit.

The acid rain program set a cap of 8.95 million tons of sulfur dioxide per year. Firms over-achieved this reduction target from 1995 through 1997 with less than half the cost of a command and control structure. Initial estimates of abatement costs determined by USEPA and the industry placed reductions per ton between $750 and $1000. But, allowance prices have remained below $200 and prices reached a low of $66 per ton in 1995. Estimates suggest that allowance trading could save from $700 to $800 million a year compared to a uniform emission standard regulatory approach (Stavins, 1998).

Air emissions trading is conceptually similar to water quality trading, but effluent trading has lagged in development and has not produced any success stories similar to the acid rain program. In 1983 the bubble policy debuted in USEPA's water programs after several years of application to air quality. The effluent bubble policy was the result of a settlement agreement between the USEPA, the Natural Resources Defense Council, and the American Iron and Steel Institute. The bubble policy only applied to facilities within the iron and steel industry, and consequently, the program saved approximately $122 million versus the $435 million saved with the initial air program bubbles (Kashmanian, 1995).

Besides the iron and steel bubble policy a few effluent trading programs were developed in the early 1980s. The state of Wisconsin created one of the earliest programs for point sources along the industrialized Fox River. During this early stage of trading the Wisconsin program hampered buyers and sellers by placing restrictions on the suppliers of credits. Suppliers could only sell credits to three types of firms: new entrants, those that could not meet required discharge limits after complying with government required technological controls, and those that intended to increase production (USEPA, 2007).

In the air quality trading programs point sources trade with other point sources similarly to the Fox River example above, but many water quality trading programs would likely take place between a point source credit purchaser and a non-point source credit producer. Several water quality trading programs have been set up between point and non-point sources, but over its twenty year history effluent trading has not had the same type of robust development that has occurred from air emissions trading. Despite this lack of success, in the last several years water quality trading has experienced increased interest by many political leaders and upper-level government officials.

Institutional Barriers to Water Quality Trading

Economists and more recently, policy makers, have advocated water quality trading programs as a way to improve efficiencies of environmental protection initiatives, minimize or postpone costs for abatement activities, offer flexible paths toward compliance, while continuing to allow population growth and economic development to occur. Unfortunately, at the present time water quality trading markets fall far short from the ideal benefits that may be gained through various implementation strategies. The literature has revealed that uncertainties and risk that arise from non-point source nutrient reduction activities, the complex interactions between nutrients and individual waterbodies, and the temporal discharge differences between point sources and non-point sources on a yearly or multi-year timescale have presented obstacles for water quality trading programs. Issues of liability between point sources and non-point sources also play a role in preventing water quality trading programs from developing fully. Water quality trading programs around the country are immature. There are a number of programs that have been developed, but at the present time only a few trades have taken place because of the obstacles that have been created.

Many water quality trading programs are not able to realize the potential benefits that could be gained from trading because of institutional, political, or informational barriers that exist. Unlike markets for most commodities, many water quality trading programs do not have the public or private institutional structures to provide market information, to locate potential trading partners, or to enforce rules for exchange. The lack of these types of institutional, political, and informational structures cause potential trading parties to expend considerable amounts of resources in order to engage in water quality trading transactions.

The analysis of the incorporation of wetlands technology into emerging water quality trading programs can be guided by the transaction costs economics literature originated by Oliver Williamson (1985, 1996). Transaction costs theory focuses on why various governance structures emerge and how they adapt in response to the challenge of mitigating transaction costs. The costs that are considered by this type of theory are ex-ante costs including: negotiating, proposing, and safeguarding agreements. Ex-post costs are also considered. These may include costs associated with contractual breakdowns, enforcement costs, and rent seeking behavior.

Transaction cost theory assumes that agents are subject to bounded rationality. Bounded rationality stems from Simon's (1957) principal-agent theory. Principal-agent theory is an economic concept that refers to the idea of motivating one party to act on behalf of another. Many types of incentives may be used to align the goals of the agent with those of the principal. A very common example of this type of relationship is that of stockholders and corporate executives. Chief stockholders sit on a board of advisors that hire and dismiss corporate executives. These individuals are considered to be the principals. They hire an individual, the agent, to perform in such a way as to increase the value of their respective stock. The agent has certain expertise, knowledge, or other considerations that make him valuable to the principals. The principals lack the time or resources to operate the company. The principals in this relationship may entice the executive with forms of profit sharing, commissions, efficiency wages, or even through the risk of being fired. Simon (1957) emphasized that agents face information costs in the present and uncertainty about the future. This status forces agents to make decisions by seeking the first satisfactory solution rather than optimizing a rational decision based on perfect knowledge. Simon suggested that classical "economic man" is familiar with all alternative courses of action and their consequences and make decisions based on maximization of welfare. Simon claimed that agents face cost and uncertainty issues in acquiring information in the future. Thus, he proposed the idea that agents would “satisfice” themselves by not maximizing their behavior through fully rational decisions. Simon claimed that people are only rational to a certain extent. He argued that their rationality would relax when it was no longer required. This process is known as “bounded rationality.” From a water quality trading standpoint the principal-agent relationship is similar to the relationship between point sources and non-point sources. Non-point sources are employed to reduce their nutrient output levels in order to meet the permit limits of the non-point sources.

Williamson’s transaction costs theory contends that transactions will be organized so as to economize on bounded rationality while continuously protecting against the hazards of opportunism. Through an evolutionary process competition will occur between firms and the most efficient forms should prevail (Williamson, 1996). If transaction costs are relatively high concerning water quality trading transactions between point sources and non-point sources then there becomes a need for an institutional structure or mechanism to manage the transactions. Some form of initial regulation is necessary in order to set discharge limits for water quality trading, but other institutional structures may also be necessary in order to assist in the actual trading process.

Colby (1995) suggested that the conflict that exists between market-oriented and regulatory approaches to resource allocation stems from disputes over the appropriate balance between market forces and laws promulgated to protect the broader social values of a particular resource. Colby argued that transaction costs reflect the costs associated with the collection of information concerning the legal, scientific, and economic needed in order to address externalities in an efficient manner.

Anderson and Snyder (1997) suggested that only minimal government intervention is necessary in order to set up a market for water resources. They suggested that there are three justifications for government involvement in water resource markets: imperfect capital markets, monopoly, or externalities. They acknowledged that the most legitimate concern deals with externalities, but they warned that rent seeking behavior can stem from too much government regulation. They argued that much of the governmental regulation during the 20th century dealing with reclamation projects in the western United States resulted from rent seeking behavior instead of an interest to protect public water supplies.

From a comparative approach Williamson (1985) argued that the transactions being conducted and the people involved in the transactions should be analyzed in a way to determine the costs of contractual governance. The governance structure with the least transaction costs will be chosen from the best available options. For instance, if a transaction involves a high level of uncertainty or transaction-specific assets, and the participants are subject to opportunism, then the transaction should be organized in a hierarchical structure rather than in a market. This can be interpreted from transaction cost theory because in this case the authority is believed to be a stronger mechanism than price to curb opportunism.

Reasoning for Including Wetlands into a Water Quality Trading Program

An analysis of U.S. water impairment patterns based on section 305 (b) of the Clean Water Act report was submitted by the states through the 1998 National Water Quality Report to Congress (USEPA, 2007). This analysis supports the notion that was discussed earlier concerning the position that non-point sources of pollution are the leading sources of pollutants responsible for the majority of the impairments to waters in the United States.

Managing nutrients like nitrogen pollution is a significant and growing problem for U.S. waters. There is ever-increasing evidence that suggests that human intervention in the nitrogen cycle has dramatically changed the distribution and movement of nitrogen in the landscape, and that these alterations pose risks to human health and the environment (Vitousek, 1997). More recent scientific reports link nitrogen pollution to lake acidification, soil degradation, and eutrophication. Recent reports have established that nitrogen pollution from air and water sources can pose a significant threat to lakes, streams, rivers, and coastal waters leading to toxic algal blooms (Smayda 1989) and creating dissolved oxygen problems (Rabalais & Nixon 2002). These concerns have been extended beyond water resources to include impacts on climate change (Vitousek, 1997), forest declines (Aber, 1998), and reductions in grassland biodiversity (Tilman, 1996). Nitrogen inputs to the nation's aquatic and terrestrial ecosystems are likely to increase in the future, but there is positive recent analysis that suggests that natural systems recover from nitrogen pollution through natural processes over time. Thus, natural systems can absorb nutrients like nitrogen, but many of our current natural systems are overloaded or too few to affect the large portions of nutrients that flow through the waterways of the United States.

Wetlands have recently been targeted for their ecological function of absorption and transformation of nutrients like sediments and nitrogen. In a 2005 *National Wetlands Newsletter* article, Eric Raffini and Morgan Robertson speculated on what a water quality trading market could learn from wetland mitigation banking and observed that "a market could emerge that would provide non-point source credits to point source dischargers, helping them meet permit obligations in much the same way that the wetland credit market emerged to help section 404 permittees meet Clean Water Act mitigation requirements." This notion generated a great deal of interest in incorporating wetlands into a water quality trading program. Over the past two years, federal agencies have sponsored workshops on the use of wetlands in generating water quality trading credits (USEPA, ORD, 2006), a workshop concerning the similarities between wetlands mitigation banking and water quality trading (National Forum, 2005), and a national water quality trading conference (ETI, 2006). Robertson and Mikota (2007) followed the experiences shared at these venues, and gave the assessment in the March-April 2007 edition of the *National Wetlands Newsletter* that "it has become increasingly apparent that point-to-non-point water quality trading faces very different challenges than wetlands mitigation banking did in its formative stage and, thus, must be designed to solve these different problems." There is a great deal of interest in developing an institutional structure that will incorporate wetlands into a water quality trading program design, and there are many reasons why this particular concept shows promise.

Currently, constructed wetlands are being effectively used to remove excess nutrients from wastewater treatment plant effluents. (Kadlec & Knight, 1996) Some experts, (Hey & Philippi, 1999), have suggested that a strategically planned arrangement of constructed, restored, enhanced, and natural wetlands could be used to reduce the nutrient and sediment impacts from surplus non-point source loadings. The USEPA is currently discussing this approach as a means of reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin in order to reduce the annual hypoxia problem that has been occurring for a number of years. Hey (2002) coined the term "nitrogen farming" as a way of describing the process of removing nitrogen from watersheds through wetland economical functions. Technical issues related to how wetland functions operate still must be investigated, and economic values must be assigned to these functions in order to create credits. These investigations must be attended to before wetlands can be incorporated into a water quality trading program.

Designs for watershed trading for control of nitrogen loadings have been proposed from various researchers. However, there is a tremendous amount of research that remains concerning the policy design and implementation utilization of wetlands in a water quality trading program. Under certain conditions markets can lead to efficient and effective outcomes, but the policy structure that creates overarching institutional structure must be firmly in place for this to occur. In some cases trading may not be feasible at the present time because the desired ecological outcomes may not be match accordingly to the transaction costs of implementation. Wetlands, however, in some situations may prove to be the most applicable and most effective nutrient abatement technology that could be implemented into a water quality trading program.

If wetlands were similar to other nutrient abatement technology, there would not be any debate concerning wetland types for use in water quality trading. Producers would choose from a suite of various abatement options based on minimizing their costs. If certain types of wetlands represented the least cost method of creating nutrient credits then those wetlands would be employed.

Wetlands have other functions that benefit humans directly or indirectly unlike other abatement techniques or technology. These benefits are refered to wetland services. These services include habitat creation, flood control, and carbon sequestration. Wetlands that are restored for water quality purposes also would provide additional services. These services are called positive externalities by economists and may not simply accrue to those involved in the restoration of a specific wetland area. USEPA considers these positive externalities to be ancillary benefits of water quality trading which could accrue to the general public or just to the private landowner. Regulators will need to determine whether the ancillary benefits actually should contribute to the generation of credits for water quality. Accurate assessment is vital in order to achieve the most optimal levels of increasing net wetland acres throughout the nation. Increases in wetland acres are important, but it is also very important to maintain wetland diversity within a watershed and across eco-regions. Constructed monotype wetlands may not be recognized as restored wetlands that count towards the increase in wetland acres, and certain wetlands within a watershed may contribute more to nutrient removal based on their spatial setting.

Byström et al. (2000) provide three criteria that must be met in order to use wetlands as a method for nitrogen reduction:

- Wetland abatement capacity must increase as the number of acres of wetlands increase,
- Wetlands must reduce the uncertainty (or variance) of total nitrogen load or keep it the same, and
- If the first two criteria are met, then wetlands must be the least cost technology.

The first two criteria require a knowledge of the wetland abatement function in a particular location and may have to be answered by wetland experts. The last criterion creates part of the incentive for wetlands to be used in the water quality trading program.

Byström estimates the abatement costs of wetlands in Sweden. Costs are based on acres of wetlands and construction costs. Construction costs include the opportunity costs of land or the value of its next best use. Byström calculates that, in the region of Kattegatt of Sweden, wetlands have a lower marginal control cost compared to using cover crops or catch crops to take up the nutrients, but a higher marginal control cost than reducing fertilizer. Hey et al. compare wastewater treatment technology and wetland abatement in the District of Greater Chicago. Depending on the season, Hey et al. (2005) find that marginal control costs for wetlands ranged from $960 to $3000 to remove a ton of nitrogen. The common technology ranged from $2810 to $10,100 per ton. However, both Byström and Hey et al. used private costs only and did not consider the potential benefits of wetlands.

Opportunities to trade credits exist, and some trading programs have included wetlands in their approved lists of best management practices (BMPs). Very few farmers have taken advantage of wetlands for conservation purposes, and there have been no official trades between point and non-point polluters that included wetlands. Demand for credits that are generated from wetlands from agricultural sources may be low because of uncertainty over the credits it can produce. Much of agricultural pollution is considered to be non-point and may also be contributed to through runoff, groundwater leaching, or the atmosphere. Therefore, it is difficult to predict with certainty the amount of discharge reduction (or production of credits) the implementation of management practices will produce at the point in the watershed where credits are measured. This may discourage demand for agricultural credits by regulated firms that are legally responsible for meeting discharge limits. Uncertainty could be reduced by more intensive monitoring, but that may be challenging and possibly even expensive. Such transaction costs could negate the benefits of trading. Future research is needed in order to better estimate the costs of monitoring for certain types of nutrients within watersheds, and the applicability of monitoring nutrient uptake efficiencies of wetlands in a watershed context. One reason why the Acid Rain Trading Program has been so successful is that the cost of measuring emissions is low. The commodity, SO_2, is easily measured and tracking is relatively inexpensive.

Uncertainty over the production of credits affects the supply side as well. Because of the nature of pollution from agriculture, and the need to assess credits at the point where regulated sources actually discharge, land owners may be unaware of the number of credits they can actually produce. There is also the problem of price. If there is not a general understanding of what a credit is then there are not many ways a land owner can assess the price that he may charge for the service he is providing.

Land owners also may be reluctant to participate in a program that is partly regulatory, even with compensation. Some have suggested that land owners are fearful that information about their contributions to water quality and costs of pollution abatement could eventually be used to develop regulations for agricultural pollution. Risk and uncertainty are the leading reasons why trades have not occurred, and this uncertainty is increased in many aspects when discussing the role of wetlands in water quality trading.

Another supply-side issue is the treatment of credits generated on farms through publicly funded conservation programs such as the Conservation Reserve Program (CRP) and Environmental Quality Incentives Program (EQIP). Since credits from conservation programs are already partly or fully funded, some trading programs do not allow them to be traded. A farmer participating in a conservation program would have to implement additional conservation measures to participate in a trading program. This would raise the cost of credits, making them less attractive to those wishing to purchase credits.

The coordination of policy guidelines between trading programs and the farm subsidy programs could be helpful in creating supply of credits. But, monitoring, verification, and enforcement are three factors that are essential for trading to function in an efficient and effective way. The incorporation of wetlands into water quality trading shows promise, but there must be verification standards and analysis that better distinguishes how wetlands operate within a watershed. As well, water quality standards must be desirable and enforceable for a trading program to have a demand and supply driver. Lastly, performance measures must be derived in order to provide more precise estimates of what is actually being traded.

Trading Ratios for Wetlands

Trading ratios are used to translate the ecological effects of pollution from different sources into a common unit. Ideally, they must capture the impacts, uncertainty, and transaction costs related to the individual sources and trade (Horan, 2005). These ratios affect the trades accepted for allowing point sources to purchase pollution control from non-point sources. A 2:1 ratio means that point sourcess would be required to purchase two units of non-point pollution control for every unit of pollution released. Non-point sources are encouraged to reduce more pollution, but the ratios do not necessarily encourage non-point sources to construct or restore wetlands that will produce the ancillary benefits previously discussed. In fact, in most cases current ratios do not express any attempt to assess performance levels or spatial considerations.

Nutrient trading ratios can be designed to account for the location of a wetland or other BMP within a watershed. This can be referred to as a delivery ratio. Currently, delivery ratios are based on distance from the BMP project to the location of the pollution source that is purchasing the credit. Delivery ratios vary greatly in current trading programs from 100% in riparian areas to 10% in areas greater than ¼ mile from the receiving water body. Ratio discounts range from 1:1 to 3:1 (Breetz et al., 2004). The general reason why trading ratios are currently applied to trading programs is to ensure that water quality in a watershed is protected and trades between sources distributed within the watershed positively affect the overall condition of the water body. Ratios in most programs do not distinguish between BMPs and they do not provide accurate accounts of nutrient removal. Future delivery trading ratios will need to provide for the transmission loss of nutrients within the river system, and further research will be required in order to understand how distance and other watershed parameters affect nutrient uptake within wetlands at the watershed scale.

Ratios also may be constructed to better ascertain the actual nutrient reduction of the applied abatement technology. A higher level of certainty of nutrient reduction would yield a lower ratio that would approach a 1:1 scale. Increased levels of uncertainty would yield higher ratios. In this respect, wetlands may outperform other abatement technologies like buffer strips because of the ability to measure direct nutrient discharge. Further study is needed in order to provide information regarding the types of wetlands that would provide more certainty concerning direct nutrient reduction as compared to other abatement technology.

The performance of wetlands concerning nutrient uptake must be assessed and verified to ensure that a trading program will be successful. In many trading programs BMPs are certified when they are installed, and some programs recommend annual spot checks for assessment purposes (Breetz, 2004). Monitoring does not occur in most trading programs because it has been suggested that it would be prohibitively expensive and a long monitoring period is required to provide conclusive results. The Great Miami River Watershed Water Quality Trading Program is currently active in monitoring and verifying reductions within the watershed at points that are located downstream from non-point source BMPs. This monitoring is currently being funded by National Resource Conservation Service Grants. This program also uses site-specific inspection at 5-10% of the BMPs. (Kieser & Associates, 2003) This type of monitoring and verification is not the most accurate method for assessing the performance of BMPs, and further research is needed in order to better ascertain the correct level of monitoring that is necessary for specific types of BMPs, especially wetlands. Non-point sources within the Miami Watershed are not being monitored, and therefore there is not an assessment of how individual BMPs are affecting the nutrient loading that is taking place within the water body. Within the field of monitoring and verification, there is a need to also further analyze how different wetland types reduce nutrient loads at the watershed scale.

Wetland Location: Determining Nutrient Load Reduction Credits

Monitoring and verification of nutrient removal performance of a wetland within a watershed context has advantages and disadvantages. Increased monitoring can lessen uncertainties in terms of modeling nutrient loss and the extent to which nutrients are found in downstream water bodies. The major disadvantage of monitoring the effectiveness of nutrient reduction is that it can be difficult in natural and restored wetlands to measure nutrient inputs because inlets often extend over relatively broad areas. Cost considerations have been raised as obstacle to monitoring, but there are not any current cost models that have been developed to assess nutrient reduction capabilities of wetlands within a watershed context.

Constructed wetlands are much easier to monitor because they can be designed with limited inlets and outlets. Specific measurements can be made at the inlet and outlet areas and differences can be compared to quantify nutrient removal efficiency. However, monitoring approaches should also account for seasonal and spatial variability in nutrient uptake efficiency (Wetzel, 2001).

Temporal factors are very important in analyzing nutrient removal efficiency, but spatial factors are very important as well. Removal efficiency levels can vary significantly depending on the position of the wetland in relation to the surrounding landscape and watershed. Wetlands located at the headwaters of a watershed may have limited opportunity in some cases to intercept nutrients. On the other hand, wetlands located in lower areas of the watershed may experience higher flow rates that limit efficiency. A study of wetlands used for 40 years to treat wastewater in Florida found that total phosphorus in wetlands sediments was significantly correlated with depth and distance from the point of surface water inflow (White, 2003). Variability in flow patterns and velocity are the two greatest limitations to maximizing retention capacities of nutrients in wetlands. Tracer experimentation can be used in order to better understand the water flow patterns or the hydraulic residence time distributions for specific wetland areas (Wetzel, 2001).

Water Quality Trading and Wetland Policy Considerations

An essential factor of incorporating wetlands into water quality trading is the need for verification of various wetland type capacities for nutrient reduction. Monitoring data will be necessary in order to provide information on the efficiency levels of wetlands to reduce nutrient loads in a watershed. This information is necessary in order to differentiate wetlands from other BMPs. If wetlands are not somehow differentiated from other BMPs then least cost applications will prevail.

In the future, multiple markets may exist to better account for the ancillary benefits of wetlands. If well-functioning markets were to exist for the different services provided by wetlands, the ancillary benefits would be accounted and the externalities would be internalized. Currently, there are not markets for some of the services provided by wetlands. Therefore, the focus of differentiation for wetlands would not be more services, but a higher level of service for nutrient reduction. The higher level of efficiency that might be attained with the implementation of wetlands for water quality trading could prove to be the differentiation that distinguishes wetlands from other BMPs.

A word of caution should also be acknowledged with the focus of wetlands as nutrient reducers. Trade-offs could exist in most cases between wetland functions that supported the highest nutrient storage capacity versus increased ancillary benefits. Wildlife habitat or flood mitigation benefits may not be realized if specific types of wetlands are restored or created. It may be necessary for regulators or wetland experts to determine the combination of plant species and hydrologic functions in order to improve water quality and increase wetland acres. However, consider how the regulator or wetland expert could direct the revenues and costs of a particular producer (e.g., location, mix of plants, etc.).

Ecosystems generate societal values and benefits supplied by their resources that resemble ecological stock and flow designs. Value identification and establishment for these resources is important to policymakers. The total economic value of an ecosystem is the amount of money that all people who benefit from the watershed would be willing to pay to see it protected (Whitehead, 1992). This total economic value is the amount society would be willing to pay for the services and attributes of the ecosystem if they were not provided free of charge. This value comprises: market economic value, which is established by transactional precedent, and non-market economic value, which is approximated by public opinion or alternate strategy.

The value that society places on wetlands and watersheds is based on their existence and services provided. As well, values are attributed to these resources because of current and future consumptive and non-consumptive uses. For example, consumptive uses of wetland outputs include fish harvests from fisheries dependent upon wetlands. Consumptive uses of wetlands may also include conversions of acreage to cropland. Non-consumptive benefits usually have a more long-term aspect assigned to them. These types of benefits could be attributed to aesthetics or flood control. Each of these values is multi-dimensional and could be assessed from several perspectives including: individual owners, individual users, regional perspectives, and societal perspectives (Leitch & Frigden, 1998). Some analysts have determined that market values are lower than non-market values for watersheds and wetlands (Stedman & Hanson, 2005).

Public policy makers face short-term and long-term strategic decisions that have great impacts on the productivity and health of natural resource systems like wetlands and watersheds. Various alternatives are always available for managing theses systems, and economic valuation provides a metric for comparing the performance of strategic alternatives over time. The cost of establishing the market economic value of a natural system is directly proportional to the complexity of the system being analyzed. The diversity of preferences given by the valuing stakeholders, and the volatility of defining markets both contribute to multifaceted interpretation of environmental public and private goods. Interpretations of how to value resources like wetlands and watersheds can be vastly different. Some of the causes of these differences include: lack of well defined scientific or economic information, site-specific natural variability, communication barriers among various research disciplines, unclear context of why valuation is needed, and economic principles are not followed (Leitch & Frigden, 1998). Many of these challenges are readily overcome when adequate time is made available and sufficient resources are invested for analysis.

Non-market valuations are essential in assessing the economic value of a policy to direct a water quality trading program that focuses on wetlands. Water quality controls, stream flow controls, and habitat management are all examples of non-marketable natural services that are considered public goods. These values are hard to quantify, but it is essential to include these values in the economic assessments of the value of wetlands in a water quality trading program. Wetlands and watersheds perform multiple biological and hydrological functions that produce both public and private goods and services. The functions and values that they elicit are intricately intertwined and ecologically linked. Non-market economic valuation techniques are widely used in the valuation of policy planning, and these techniques are very important in the valuation of natural resources policy because the non-market component of natural resources economic value typically outweighs the market component of economic value (Fausold & Lilieholm, 1999). However, in many cases it is difficult to complete a non-market economic valuation rapidly enough and with enough detail to inform decision makers. Focus should be place on the effective interpretation of the results and not just the numeric results themselves.

Economic considerations must support the idea of wetland's incorporation into a water quality trading program, and the constructed market must be able to consider the true valuation of and exchange between point and non-point sources. But, the scientific understanding of how wetlands remove nutrients from a watershed must be thoroughly analyzed as well. The establishment of sophisticated methods of decision making and evaluating risk are negligible if information gaps understanding the science of wetland functions are not filled. Better assessments and valuations of wetland functions will enable policy makers to quantify the value of investing in a water quality trading program that incorporates wetland technology as a pollution management strategy of choice.

CHAPTER 3

REVIEW OF THE LITERATURE

Wetland Nutrient Removal Capabilities

Water quality trading policy is still in its infancy, but wetland resources, on the other hand, have been studied, regulated, and politically interpreted in the United States for well over a hundred years. Yet, with all of this attention there is still a great deal of mystery that surrounds the natural services that they provide. The word, wetland, is a generic term that represents a vast universe of wet habitats that include marshes, swamps, fens, bogs, and similar areas. The definition of wetland informally refers the interface between truly terrestrial ecosystems and aquatic systems making them inherently different from each type of system while at the same time dependent on both (Mitsch & Gosselink, 2000). Wetlands are known as ecotones and serve as transition areas for two different types of ecological communities, ecosystems. The word, ecosystem, was coined by A.G. Tansley (1935), a plant ecologist almost fifty years after the concept was defined by Stephen Forbes, a zoologist with interests in aquatic environments. In 1887 Forbes wrote an essay describing the architecture of the environment by specifically discussing how the organisms of a lake constantly affect each other through a complex interaction of predation and competition. This holistic scientific and policy viewpoint lay dormant for many years because of the demand for scientists and engineers to produce information and analysis on the basis of specifics. Over the past couple of decades federal agencies have started to create policy guidelines based the ideas of holistic ecosystem management. Yet, there is still much private confusion over the meaning of this approach (Christensen et al., 1996). In fact, wetland definitions over the past century have been created based on the specific functions being studied at one particular time or another. Nathaniel Shaler's *General Account of the Freshwater Morasses of the United States* was one of the earliest reports on the nation's wetlands published in 1890. At the time, Shaler's report focused on the relationships between wetlands and farming. His definition of a "swamp" was:

> all areas in which the natural declivity is insufficient, when the forest cover is removed, to reduce the soil to the measure of dryness necessary for agriculture. Wherever any form of engineering is necessary to secure this desiccation, the area is classified as swamp. (Shaler, 1890)

By the end of the 20th century scientists were beginning to classify wetlands by focusing on particular attribute types as noted by Lefor and Kennard (1977). These attribute types spanned a large environmental gradient. There are no overarching truths that apply to all wetlands. Wetland types are very diverse and wetland scientists are becoming aware that in their pursuit to restore and create wetlands many unknowns provide obstacles to defining guidelines for successful wetland assessment and design for policy applications. In some instances wetland scientists have an understanding of how certain functions operate within a specific area. It is very important to note all the specific functions that wetlands provide, but it is also important to understand how these various functions interact and correspond with each other to provide benefits at the watershed scale.

Wetland Policy in the United States

Wetland science and wetland regulation policy are both relatively new and rapidly evolving branches of ecosystem and sociopolitical science. Both wetland science and regulation policy develop in a highly charged political atmosphere. The two areas have an intense relationship that "in many ways is reminiscent of the relationship between nuclear physics and national defense 50 years ago" (Lewis, 2001). In many cases regulatory initiatives and policy applications raise unanswered scientific questions.

For over 150 years the U.S. Government has been involved with wetland issues. From the 1850s to the 1970s the Federal Government promoted the elimination of wetland areas. Over the past 30 years the importance of wetlands to water quality, marine fisheries, flood abatement, and biodiversity has been acknowledged and valued to some extent by federal, state, and local policies. Section 404 of the Clean Water Act deals with wetlands, and this regulation has required the replacement of wetlands lost to development under a policy now known as “no net loss of wetlands” (USEPA and USACE 1990; NWPF 1987). The U.S. Army Corps of Engineers (USACE) issues permits to those individuals who wish to dredge or fill wetlands, and these permits often require “compensatory mitigation.” This type of mitigation usually refers to the restoration of former wetlands to balance the effects of wetland loss. Wetlands mitigation banking represents the oldest existing market for ecosystems services. In 2005 the USEPA sponsored a workshop on the lessons for water quality trading to be drawn from wetland mitigation banking (ELI, 2005). From the proceedings it became quite clear that water quality trading faces very different challenges than wetland mitigation banking did in its formative stage. Thus, the policy features and regulatory influences must be designed differently in order to solve different problem sets. Water quality trading and wetlands mitigation banking do share two very important aspects concerning wetlands: wetland condition and functionality. Water quality trading programs incorporate wetlands to improve water quality, and mitigation banking programs assess and offset wetland areas in order to provide “no net loss of wetlands.” Robertson and Mikota (2007) provide an analysis of why the two types of policy programs are different, but from a wetland science perspective there are very similar functions that take place within wetland systems regardless of whether they are incorporated into a water quality trading program or a wetland mitigation banking program. The nutrient removal function of water quality improvement is very necessary to understand in order to set policy guidelines for water quality trading. At the present time mitigation banking policies focus on the condition of the wetland and water quality trading policies focus on the nutrient reduction or transformation abilities of a type of wetland, but condition and functionality are intertwined. A review of what is known about wetland functions for nutrient reduction is very necessary in order to guide water quality trading program policies that incorporate wetland technology.

For a long period of time environmental economists have advocated the implementation of market mechanisms to promote ecosystem services on the grounds that the prices that customers pay for these services should fully consider the financial costs of degradation (Costanza et al., 1997; Daily & Ellison, 2002). Some wetland scientists have also contended that the scientific evidence explaining wetland conditions and functions should guide public policy concerning wetland resources (Lewis, 2001). In order for market mechanisms to promote ecosystem services, the services themselves must be identified and understood. The condition of a wetland may be quite variable over time, and certain functions of wetlands may depend upon climate, temporal, regional, or hydrological conditions. All of these factors must be reviewed to analyze the possibility of incorporating wetlands in to a water quality trading program at the watershed scale.

Wetland Nutrient Removal for Water Quality Trading

The function of specific wetlands areas as nutrient transformers depends on the wetland type, hydrologic condition, and the length of time the wetland is subjected to nutrient loading. Wetlands have been shown to be sinks or storage places for nitrogen and phosphorus, but many wetlands do not exhibit this trait. The function of wetlands is closely related to adjoining land and water bodies, and this is why it is important to research wetland nutrient uptake capacities at the watershed scale. Also, the capacity of natural wetlands to store and transform nutrients is directly related to the amount of nutrients available within an ecosystem. Vast changes in the amount, increases or decreases, could have an effect on the performance of the wetland to provide water quality improvement (Mitsch and Gosselink, 2000). Constructed wetlands may require significant maintenance in order to sustain a specific level of nutrient removal, and it may be economically and ecologically challenging to maintain this or a greater level of uptake in perpetuity without the proper investment.

The results of many wetland nutrient removal studies have been inconsistent. A compilation of data from over sixty studies of fifty-seven wetland ecosystems throughout several countries suggests that the mean percentage change in nutrient load was between 67% and 27% for nitrogen and between 58% and 23% for phosphorus. Estimates suggest that 80% of these wetlands removed some level of nitrogen and 84% removed some level of phosphorus (Fisher & Acreman, 2004). Braskerud (2002) contends that his study of surface flow wetlands in Norway only provide nitrogen removal efficiencies that ranged between 3% and 15% because of high hydraulic load and low temperatures. Luederitz et al. (2001) reported a total nitrogen and phosphorus removal rate of greater than 90% in constructed reed bed wetlands in Germany. Seasonality also plays a very important role in nutrient uptake efficiencies as discussed in some of the studies mentioned. Nitrogen removal performances varied by up to 40% between summer and winter months in Hungary according to Szabo et al. (2001).

Many generalizations can be made from a policy perspective regarding the function of wetlands as transformers, sources, or sinks for nutrients, but the complex and unique situation that involves individual wetlands limits these types of generalizations to actual applied policy procedures. Wetlands can be a sink for a form of nitrogen at one moment in time and a source for the same nitrogen element at another time depending on a range of factors. Inconsistent results from studies also generate problems for generalizations because of the imprecise approaches to measuring nutrient changes over time within various wetlands. There is little consensus in the literature about nitrogen and phosphorus fate in wetlands (Mitsch & Gosselink, 2000). There also seems to be a bias toward researching the vegetation and productivity that limits the understanding of soil and microbial processes specific to wetlands that enable some forms of nutrient transformation (Johnston, 1991; Wetzel, 2001).

Regardless of the inconsistencies concerning wetland nutrient transformation mentioned, constructed wetlands have shown in some cases to be effective at removing nutrients and sediment from wastewater treatment plant effluent when the wetlands are co-located at a discharging facility. Constructed wetlands at the watershed scale may be an effective method to lessen the harmful effects of nutrients on waterbodies (Cooper & Findlater, 1990; Hammer, 1989 and 1996; Kadlec & Knight, 2004), but there are other issues that deal with the sustainability, the implementation, and the long-term objectives of watershed plans that must be analyzed when considering the use of incorporating wetlands as a potential water quality improvement technique in order to improve water quality and increase restored wetland acres. Further research is needed concerning the life-cycle of wetlands that retain nutrients and the long-term capacity of restored wetlands to function as water filters.

At the present time there have not been any successful large scale projects that have monitored, managed, or described the hydrological functions of natural or constructed wetlands at the watershed level. As noted earlier, there have been small scale studies that have shown the effectiveness of wetlands to absorb nutrients. But, what types of changes would be encountered within a watershed based on the scale of wetland restoration activities or construction activities?

The formation, size, functions, and persistence of wetlands are controlled by hydrologic processes. Other differences in wetlands depend upon variables like water quality, the movement of water through the wetland, and the degree of natural or anthropogenic disturbance. Therefore, wetlands are distributed unevenly throughout the United States because of differences in geology, climate, source of water, and other natural and man-made factors.

Scientists commonly investigate hydrologic questions by determining how much water is moving through a system. This knowledge is then used to characterize how the system functions. The hydrologic cycle is a dynamic process that includes the components of precipitation, surface-water flow, ground-water flow, and evapotranspiration. Various wetland types receive different water budgets throughout the year. The water budget that can be measured for a specific wetland area would be derived from subtracting the total outflows from the total inflows. But, as mentioned earlier, this is a dynamic process that depends upon seasonality, climate, and other factors. The accuracy of individual components of a water budget for a wetland depends on how well they can be measured and the magnitude of the associated errors. This measurement is vital in understanding how natural or constructed wetlands would fit into a water quality trading program. Natural wetlands depend on a certain level of moisture in order to thrive, and variations in hydrologic regimes during times of drought or abundance of rain may interfere with large-scale constructed wetlands' ability to treat nutrient loads (Mitsch & Gosselink, 2000).

The hydrologic cycle is also the most significant determinant of vegetation within the wetland. This, in turn, affects water quality and biological diversity. Water chemistry also plays a role in what types of plant species exist within a wetland. Wetland types in many cases can be identified by their hydrologic functions and hydrochemistry. For instance, bogs and fens are two major types of peat land that are identified by the USEPA as being specialized wetland types. Bogs and fens are mostly found in the in the northeast and Great Lakes regions, but some southern bogs and pocosins can be found in the southeast. These types of wetlands may be hard to distinguish by a simple glance, but further observation would conclude that the main difference between the two is their water supply.

Various studies of the effect of hydrologic conditions have revealed inconsistent results. Hydrologic conditions in a wetland can influence the efficiency of processes that remove nutrients from water (Sakadevan & Bavor, 1999). Residence time was negatively correlated with total nitrogen and phosphorus removal in constructed subsurface flow wetlands (Schulz, 2003). Dierberg et al. (2002) found that as residence time increased, nutrient reductions also increased. Lin et al. (2002) found that nutrient removal efficiencies were unaffected by variation in loading rates, and Knight et al. (2000) found that removal of nutrients was a function of inlet concentrations and loading rates within a temporal scale.

In order for wetlands to be nutrient reducers within a water quality trading program the outflows of water from a wetland area must have nutrient levels that are lower than the inflows, and the reductions must be measurable. One of the key elements for nutrient reduction in wetlands involves inundation. Inundation greatly affects the oxygen content of the soil and produces anaerobic conditions. Though, in many wetland areas the surface soil tends to retain an oxidized layer due to the proximity to the water column, microbial activity, and oxygen translocation within rooted plants (Tanner, 2001). Inundation also affects nutrient transformation because it helps to determine pH levels. Water within a natural wetland may be fresh, saline, acidic, or basic depending upon the source and inundation periods. For instance, fens are a type of wetland that depends heavily on ground water for their hydrologic cycle. Fens can be described as a stage in the succession from shallow lakes to bogs. In most cases fens also can be distinguished from bogs in being alkaline of calcareous wetlands with pH levels that stabilize above 6.0. These areas support calcicolous plant species that act as good nutrient uptake agents. Nutrient uptake is highest in a neutral pH range of 5.5 to 8.2. Bogs, on the other hand, are primarily supplied by rainwater. These ombrotrophic areas reside in poorly drained geographic locations where precipitation exceeds evaporation. Growth of higher plants is curbed by peat mosses in bogs because the mosses bind available nutrients and render the bog water acidic. Also, because the water surface is trapped among a tight network of peat stems and leaves, water movement is almost completely lacking, and temperature exchange between air and water is restricted. Some wetland scientists have contended that the amount of oxygen available to the soil is the greatest limiting factor for nitrification (White & Reddy, 2003). Subsurface water systems have been found to display marginal or negative nitrogen removal efficiencies because of the lack of oxygen (USEPA, 1993).

Inundation in many ways also may affect storage times for nutrients that flow through wetlands. Nutrients can be removed from inflow waters through storage within the soil, organic matter, or biota. Phosphorus is a nutrient that is stored through soil by absorption through sediment particles, through plant and animal biomass, and through peat and other aquatic plants. Sediment that is rich with organic matter tends to have the capability of absorbing higher rates of nitrogen and phosphorus (Tanner, 2001). On the other hand, some wetland scientists have shown that aquatic vegetation can sequester twice as much phosphorus as sediment can, but this phosphorus uptake could be mobilized again as absorbing plants decay (Dierberg et al., 2002). The reduction of downstream export of soluble inorganic phosphorus can decrease the nutrient load of downstream waters associated with eutrophication. Insoluble forms of organic and inorganic phosphorus are generally not biologically available until they are transformed into soluble inorganic forms (Mitsch & Gosselink, 2000).

Phosphorus and nitrogen removal rates have been shown to be larger in wetlands that contain aquatic vegetation. Denitrification and plant uptake are two occurrences that assist in removing nutrients (Lin et al., 2002; Stein et al., 2003; Tanner, 2001). Nutrient removal in many cases is enhanced by macrophytes that assist in creating solid sedimentation, reducing algae production, improving nutrient uptake, and releasing oxygen (Bavor et al., 2001; Jing et al., 2002). Wetland scientists in Australia and Taiwan have conducted studies that revealed 96% reduction efficiencies for both nitrogen and phosphorus for wetlands with plant life and approximately 16% for nitrogen and 45% for phosphorus without plant life (Huett et al., 2001; Jing et al., 2002). Floating aquatic macrophytes have been used in constructed wetlands in Brazil to improve drinking water supplies (Elias et al., 2001).

Different species of plants also perform better than others at nutrient removal. In Thailand a recent study suggests that aquatic plants increased phosphorus removal in wetlands constructed to treat saline wastewater, and cattails were most efficient at nitrogen removal (Klomjek & Nitisoravut, 2005). Other species that have proven to be good nitrogen transformers include: Phragmites (Mayo & Bigambo, 2005), Typha angustifolia (Belmont et al., 2004), and Schoenoplectus (Poach et al., 2003). On the other hand, there have been some studies that have found that some wetland plants do not contribute at all to nutrient reduction processes (Huang et al., 2000). Dierberg et al. (2002) found that species differed in their uptake and accumulation in plant tissue, but did not contribute to nutrient reductions in a study in the Florida Everglades.

Variations within Wetland Types

Hydrology is the key component to understanding how a wetland will function within a particular landscape. It has been noted by Mitsch and Gosselink (2000) that the notion of hydrology as the fundamental forcing function of wetlands is well understood from a conceptual level. But, this understanding is not always put into practice by many who are involved in restoring or constructing wetlands. Many problems that have occurred in relation to the restoration or construction of wetlands have involved the problem of placing project sites in areas that are characterize by human-altered landscapes, which cause changes in hydrologic conditions. Unpredictable and rapidly fluctuating hydrology can lead to planting failure, washouts, decreased biodiversity, and loss of water quality functions. Therefore, wetland restoration and construction projects must consider watershed scale hydrology concerns when developing plans for integrating wetlands into a water quality trading program.

Nutrient input, transformation, and concentration within wetlands are highly correlated with climatic influences that affect hydrology and temperature (Mitsch & Gosselink, 2000). Temperature affects growth, productivity, and oxygen levels of wetland biota. Woodwell and Whitney (1977) found that in some cases salt marshes immobilize phosphate in cold months and export phosphate in warm months. Increased levels of precipitation can dilute or increase nutrient concentrations depending on the hydrologic surface or subsurface flow before entering the wetland. Erosion may also play a role in increased nutrient loads through sediment.

Climate also has an affect on the types of vegetation and microorganisms that can grow in a particular wetland ecosystem. The quantity and variety of these organisms in some cases may affect the nutrient transformation capabilities of a particular wetland area. Temperate wetlands typically retain more nutrients in the growing season primarily because of the higher microbial and macrophyte productivity. This productivity allows wetlands to function as nitrogen and phosphorus sinks in summer months. This time of productivity corresponds favorably with the need to reduce summer algae blooms in downstream waters as a result of elevated nutrients (Klopatek, 1978; Lee et al., 1975).

There are four categories of wetland types listed by the U.S. Environmental Protection Agency: bogs, fens, tidal and non-tidal marshes, and swamps (USEPA, 2005). These broad categories are used to delineate wetland areas and to describe boundaries for the U.S. Army Corps of Engineers concerning "no net loss goals." All four types have different characterizations that would be helpful to understand when focusing on incorporating wetland technology into a water quality trading program.

Bogs are the first type of wetland described by the USEPA. The hydrologic cycle of bogs does not lend itself well to being monitored for water quality improvement. In fact, in most respects, bogs may not even remove nutrients from water because of the lack of water flow and the acidic nature of their environment. Therefore, it is unlikely that this type of wetland could be applied to a wetlands and water quality trading program. But, bogs provide many benefits that should not be overlooked, and some of these include water retention, habitat creation, and carbon storage. Naturally existing bogs develop through a very slow process that may take hundreds of years to complete (USEPA, 2005).

Fens, like bogs, provide many benefits in a watershed that include flood mitigation, habitat creation, and the improvement of water quality. Fens are transitional wetlands located between bogs and open waters, and therefore nutrient uptake is a part of a fen's functions. In most respects, fens absorb nutrients from groundwater, and therefore, monitoring must focus on flow from the ground to fen, and finally to surface water, in order to understand the dynamics of water quality improvement (USEPA, 2005).

Tidal and non-tidal marshes are types of wetlands that are also recognized by the USEPA as being distinct types of wetlands. Tidal marshes function as coastline buffers and they also act as nutrient uptake agents absorbing loads before they reach estuaries and oceans. Tidal marshes also act as nurseries for clams, crabs, and juvenile fish, and several species of migratory waterfowl. Within the category of non-tidal marshes there are four sub-categories that include: wet meadows, prairie potholes, vernal pools, and playa lakes. Non-tidal marshes recharge groundwater supplies and moderate streamflow by providing steady supplies of water to streams. This is an especially important function during periods of drought. These marshes also help to mitigate floods by storing excess capacities of water during times of heavy precipitation. The vegetation and microorganisms that thrive in marshlands are active in recycling excess nutrients like nitrogen and phosphorus that can otherwise pollute surface water. This wetland type is very important to preserving the quality of surface waters (USEPA, 2005).

Swamps are other types of wetlands that are recognized by the USEPA as being distinct. A swamp is any wetland dominated by woody plants. Swamps can be very different in appearance and may have different functions when compared to one another. Swamps are usually characterized by saturated soils during the growing season. During other times of the year standing water is the normal hydrology for many swamps. In general terms, swamps are dominated by either shrubs or trees. Swamps are good nutrient recyclers and they also provide a significant amount of habitat for many types of animal species. The ability for swamps to absorb nutrients is highly dependent upon seasonality and also the life-cycle of the swamp itself. Some swamps can be degraded or impaired by nutrient overloads (USEPA, 2005).

Marshes and swamps are types of wetlands that can have a direct impact on surface water quality. But, in order to implement a water quality trading program that uses restored or constructed wetlands as nutrient reducers three wetland type issues must be addressed: the understanding of individual wetland hydrology within specific regions, the understanding of natural design, and the consideration of time.

Geographic Position and Temporal Issues of Wetlands within a Watershed

Wetland restoration and construction projects must also consider the placement of wetland types within a watershed. Wetlands are typically found in low-lying areas where the hydrological regime is consistent with their demands for water. Hunt, Krabbenhoft, and Anderson (1996) investigated how wetland restoration compares to wetland creation. Their definition of wetland restoration centered on the idea of restoring a wetland that was once drained or filled so that it would again function as a wetland. Their definition of a created wetland focused on those areas that had not previously been considered wetlands. They concluded that the construction costs for the restored site was one-fifth the cost of the created wetland, and the restoration implementation time was much shorter (two weeks) than the created wetlands (six months). The authors also identified the fact that in a 1993 delineation of the wetland restoration and creation sites, 100% of the restored wetland had been delineated as a wetland and only 60% of the created site was delineated as a wetland.

Upland and lowland positions of restored or constructed wetlands are just one factor of many geographic considerations of wetland placement. Position and land use surrounding a wetland site directly affect the nutrients flow (Mitsch & Gosselink, 2000). The size of the watershed, soil texture, topography, and landscape features influence nutrient inputs. Land uses can affect nutrient inputs by altering buffer features, affecting erosion rates, and disturbing natural hydrologic flows. A wetland and its ability to transform nutrients may be altered by adjacent land use practices over a short period of time (Gathumbi et al., 2005). Direct input from urban runoff, industry, or wastewater can have dramatic impacts on nutrient loads within a particular wetlands site.

If a water quality trading program also wants to contribute to increasing the number of wetland acres then it is necessary to understand the capacity of nature to both recruit species for water quality improvement and to make choices from those species introduced by humans (Odum, 1989). The human design approach may serve to be successful in creating engineered wetlands that will mechanistically extract nutrients from the watershed, but will the process be sustainable over time? Reinartz and Warne (1993) analyzed the differences between 11 created wetlands in southwestern Wisconsin that were naturally colonized with 5 wetlands in the same region where 22 species were introduced through seeding. They found that the diversity and richness of the natural colonized site were greater after a two year period of time. McKnight (1992) has suggested that many studies have over-predicted the survivability of transplanted species in designed wetlands. This is a very important detail to focus on when implementing a water quality trading program that incorporates wetlands. Restored wetlands may experience the loss of species diversity, and constructed wetlands may be more costly to operate if plant species have to be systematically replanted and maintained on a yearly or seasonal basis.

The notion of time should be another issue that has to be addressed when analyzing wetland types for a water quality trading program. In many cases dealing with policy creation, the legal and economic necessities seem to dictate the ecological patterns of nature, but there are not always quick solutions to handling some of the environmental issues at hand. Many engineering plans for buildings and other types of infrastructure rely on five-year plans. But, in some cases wetlands develop over longer periods of time, and it is important to understand what types of functions may be added or subtracted from a watershed area based on the addition of restored or constructed wetland areas.

Constructed wetlands function similarly to natural wetlands to buffer downstream nutrients by storing and transforming nutrients (DeBusk, 1999). Hydrologic inputs generally dominate elemental inputs into wetlands and determine the various geologic, chemical, and biologic nutrient pathways. On a small scale the cycling of nutrients in constructed wetlands has been extensively researched (Mitsch & Gosselink, 2000). Natural wetlands and their ability to store or transform nutrients have also been researched to some extent. Natural wetlands exist where water inundates land or groundwater is shallow enough to create hydric soils near the surface. This inundation supports hydrophytic plants adapted to living in water or saturated soils. Constructed wetlands have been developed to improve water quality by utilizing natural processes involving wetland vegetation, soils, and their associated microbial assemblages to assist in the treating of effluent or other water sources (USEPA 2000). Constructed wetlands can be designed in a way that replicates the condition and function of a naturally occurring wetland like mitigation banks strive to do, or they can be designed specifically to maximize a wetland function to provide for water quality treatment or stormwater management (Hammer, 1992). Restored and enhanced wetlands are other types of wetlands that were historical naturally occurring wetlands that have been disturbed through filling, dredging, water elevation changes, plant community alterations, or land use modifications to surrounding areas. Disturbed wetlands can be restored through the rehabilitation of hydrologic conditions and reestablishment of vegetation (Mitsch & Gosselink, 2000).

Wetlands that have been degraded may be restored or enhanced through the careful application and operation for water quality treatment. This approach is encouraged if water quality of the wetlands would not be degraded, there was a net benefit to the wetland, and it would promote a return of historic or natural conditions to the wetland (USEPA, 2000). The addition of nutrient rich waters to wetlands with low productivity can increase productivity (Ewel & Odum, 1984). Restored wetlands can be very effective in reducing agricultural non-point source pollution. These systems in some circumstances can remove up to 90% of suspended solids, between 85% and 100% of total phosphorus, and between 80% and 90% of total nitrogen (DeLaney, 1995). A compilation of data from 60 studies of 57 natural occurring wetlands in 16 countries showed that 80% of the wetlands reduced nitrogen loading, and 84% reduced phosphorus loading. The mean percent change in nutrient load between water entering and exiting the wetlands was 67% for nitrogen and 58 % for phosphorus (Fisher & Acreman, 2004).

Constructed wetlands have been incorporated into plans to treat water for some time now, and planning and design considerations for building constructed wetlands have been developed by USEPA (1999). Natural and constructed wetlands exhibit plant and microbial metabolism involved in nutrient transformation and retention that is highly dynamic on daily, seasonal, and long-term annual scales (Ewel & Odum, 1984; Kadlec & Knight, 1996; Wetzel, 2001). The amount and concentration of nutrient loading also influences these processes at all scales.

Nitrogen and phosphorus were reduced by over 90% according to a 50 year study of discharged wastewater into an existing forested wetlands in the Mississippi Delta (Day et al., 2004).

Despite the similarities in natural and constructed wetland, there are several differences between the two wetlands types. Constructed wetlands often vary in the shape and structure from natural wetlands. In many cases constructed wetlands are shaped to fit into the landscape with other features such as roads, buildings, or other man-made structures. This type of structural fit can limit the ability to create a natural looking and functioning wetland. Most research studies of constructed wetlands use conveniently sized plots, mesocosms, that provide straightforward control of soils, plants, and water levels as well as inflow and outflow controls. These highly controlled areas are used to precisely measure water quality parameters affected by the wetland (Dierberg et al., 2002; Jing et al., 2002). In many cases as well constructed wetlands have engineered substrates composed of gravels or artificial liners. This aspect affects the sub-surface nutrient removal processes as well as other interactions with groundwater.

From a broad perspective many natural wetlands reveal larger amounts of biodiversity while constructed wetlands are typically planted with certain highly productive macrophytes and are occasionally inoculated with microorganisms (Wetzel, 2001). The increased species diversity and productivity maximizes nutrients retention, recycling, and storage in the long run. This greater diversity often allows more light to penetrate deeper into the water, and this increases the capability of photosynthesis and survival of microorganism assemblages (Wetzel, 2001).

Externalities of Constructed Wetlands

In 2000 the USEPA provided guidelines for constructing wetlands. This report described more than 600 active projects using constructed wetlands to treat municipal, industrial, agricultural, and storm wastewater sources (USEPA 2000). These projects were analyzed to develop wetlands that improve water quality as well as provide wildlife habitat (USEPA, 2000).

U.S. EPA also has provided two technical assessments of different types of constructed wetlands: Free Water Surface Wetlands for Wastewater Treatment: A Technology Assessment (USEPA, 1993) and Subsurface Flow Constructed Wetlands for Wastewater Treatment: A Technology Assessment (USEPA, 1999). These research analyses are helpful in determining the selection and design of a constructed wetland.

Although natural and constructed wetlands have been used for water quality treatment purposes for many years, there is still a great deal of knowledge that has yet to be gained concerning the performance and design factors of wetlands. Further research is still needed to better understand the chemical and physical characteristics of various nutrient fractions in runoff as well as the nature of nutrients that remain after passage through wetlands (Dierberg et al., 2002). Further monitoring and verification at the regional and national level have been suggested by some wetland scientists in order to provide a more detailed evaluation of wastewater treatment systems to identify variability and factors contributing to variability (Szabó et al., 2001). The wide spectrum of wetland types, positions, climates, geology, hydrology, nutrient inputs, and geographic landscapes needs further analysis in order to identify and compare nutrient removal capabilities in an attempt to better inform policy decisions that incorporate wetland technology for water quality trading purposes.

Constructed wetlands in many circumstances can provide extra benefits in addition to water quality treatment (Kadlec & Knight, 1996). These benefits include: increased habitat, biological export to adjacent systems, aesthetics, hunting, recreation, and research (Knight 1992). Constructed wetlands provide habitat for plants and animals, and many periodically use wetlands as foraging areas, breeding sites, or drinking sources. One summary of 17 case studies conducted by the USEPA assessed a series of sites throughout 10 states and found that constructed wetlands can provide valuable wetland habitat for waterfowl and other wildlife (USEPA, 1993).

Additional water resource benefits have been demonstrated by using wetlands for nutrient treatment purposes. Day et al, (2004) has shown significant increases in sedimentation and increased accretion rates in the Mississippi Delta related to the application of nutrient rich wastewater over a 50 year period of time. The results of the study suggest that the application nutrient enriched water can increase wetland elevations and counteract some of the negative effects of sea level rise on coastal wetlands. Adding nutrient rich water into natural wetlands has also been demonstrated to increase productivity of woody vegetation and to also increase the growth of herbaceous emergent and aquatic vegetation (Day et al., 2004).

Constructed wetlands play a role in providing additional benefits by buffering against flood waters, storing water for multiple uses, and recharging groundwater (Knight, 1992). Wetlands can be very valuable in watershed management strategies in areas where wetlands have been lost. Watersheds that are comprised of 5% to 10% wetlands are capable of providing a 50% reduction in peak flood period compared to those watersheds that have none (DeLaney, 1995).

There are many benefits that can be received from incorporating wetlands technology into a watershed, but there are some negative externalities associated with them. For example, the optional use of farmland to construct a wetland results in land being taken out of production for another land use. Constructed wetlands located in other water bodies or adjacent to water bodies may have a negative nutrient affect on the natural water quality or quantity of these waters (USEPA, 2000). The design of the constructed wetland and the quality of the natural waterbody both affect the positive and negative externalities in this situation.

Wetlands may present other negative externalities like unpleasant odors. Other potential impacts to air from constructed wetlands include the emissions of nitrous oxide through the denitrification process that is released into the atmosphere as microbes convert nitrates. This release of gas has negative effects on local ground level ozone (DeBusk, 1999). The release of methane gas is also a potential concern of the denitrification process (Wetzel, 2001).

Some studies provide guidance on optimizing the ancillary benefits created by constructed wetlands while avoiding the undesirable negative externalities (Knight, 1992). Planning and design factors of future wetland areas are key essentials to assessing the benefits and problems associated with these resources. Monitoring and maintenance activities are also important after construction of the wetland. These factors are necessary in order to prolong and enhance the nitrogen and phosphorus removal efficiencies while maintaining ancillary benefits.

The monitoring and maintenance requirements for wetlands are needed in order to compare performance of constructed wetlands and impacts of external factors on wetlands. Monitoring implementation strategies should include the surrounding area as well as the constructed wetland site to assess watershed scale nutrient transport factors. Further research is needed concerning the temporal nature of constructed wetlands. An example of one case demonstrated that nitrogen removal efficiency dropped from 79% to 21% over a one year period of time (Tanner et al., 2005). Constructed wetlands technology could benefit from additional monitoring, maintenance, and planning performance evaluation studies that would provide information concerning the long term life cycle, dynamic conditions, and nutrient reduction functions of multiple types of wetlands at the given watershed scale.

Wetland Nutrient Removal Policy Considerations

Various studies have investigated the function of wetlands in the removal of pollutants, including high levels of nutrients (Cooper & Findlater, 1990; Fisher & Acreman, 2004; Hunt & Poach, 2001; Mitsch & Gosselink, 2000;). These studies have been used to guide the development of constructed wetlands to treat water high in nitrogen and phosphorus (Mitsch & Wilson, 1996). The natural chemical processes that take place within wetlands have been mimicked by constructed treatment wetlands in an effort to improve water quality. Most of these studies have taken place in a confined experimental setting where the hydrology and flora have been specifically defined and managed. These efforts have encouraged scientist and policy makers to optimize the water quality functions of wetlands within a watershed framework. However, the undertaking of a watershed scale research initiative in order to study how wetlands improve water quality has not been applied at the present time.

Wetland characteristics are almost unique from wetland to wetland, and, more importantly, from the social and economic context. For example, calculations can be developed to better interpret the different values that riparian wetlands, coastal marshes, or prairie potholes may elicit based on the variation in functions and services they provide. It is more difficult to identify the range of contributions that are provided by wetlands based on their surrounding area in relation to land use, population, income, education, and other characteristics. Hypothetically, identical wetlands that provide similar services could be valued differently in isolated and remote areas compared to those that are found in densely populated areas. Likewise, the social or political landscape may define the values that are attributed to wetlands compared to their actual services. Increasing population and development in many places throughout the U.S. have amplified the pressure on current wetlands to provide their natural services. This increased pressure will continue to occur in the near future, and new policies will need to provide guidelines that improve the success levels of restored and created wetlands.

Aside from these social, political, economic variables, basic wetland characteristic diversity must be better understood within the context of a watershed framework in order to provide insights into how wetlands may play a role in improving water quality. There are various gaps in the scientific knowledge associated with the understanding of the connections between structural measures (plant diversity, productivity, vegetation density) and the various functions that wetlands provide (nutrient retention, organic sediment accretion, or wildlife use). There is a need to quantitatively account for these connections through designed experiments located in various eco-regions. Simple lists of flora or fauna species are inadequate determinants for regulators or wetland managers to set wetland typologies or estimate ecosystem function. This research is needed in order to better ascertain natural and constructed wetlands function within watershed types in different eco-regions with various climates and hydrological regimes. This scientific understanding would better inform policymakers considering the opportunities to use specific wetland types as an option for reducing nutrient loads at a watershed scale.

Economic Considerations of Point to Non-Point Water Quality Trading

Point source discharges have traditionally held three choices for managing their discharger liability: meet standards by investing in technology or addition control measures, meet standards by trading for credits, or try to evade regulations by using the political or legal process to minimize enforcement penalties (Kydland & Prescott, 1977). Direct enforcement action from a federal or state level over the course of the past thirty years concerning water quality has been expensive and increasingly ineffective. Many point and non-point pollution dischargers have recently incorporated strategies involving avoidance or liability transfer in an attempt to delay, evade, or reduce penalties they cannot avoid (King, 2005).

Water quality trading is a voluntary regulatory option for achieving compliance. This policy application is a relatively new process that includes various stakeholders that may meet their discharge standards through trading. From a theoretical standpoint discharges with high costs of pollution control buy credits from other dischargers with lower costs. Savings can be realized from this type of exchange, and an example of this type of cost savings was a simulated trade incurred in the Lower Boise River Idaho Trading Program. Instead of installing point source controls a constructed wetland and sediment basin estimated reduced costs to range from $10 to $158 per pound of phosphorus (Breetz et al., 2004). The credits may be generated by various means of pollution control, and overall cost of pollution management for the various stakeholders should be reduced. The Tar-Pamlico Basin Association estimated potential costs of pollution reduction controls at $1 million for non-point controls compared to $7 million to achieve a comparable level of nutrient reduction through point source controls (DeAlessi, 2003).

Water quality trading policies presume that point source dischargers will prefer to meet pollution standards by purchasing nutrient reduction credits from non-point source providers if the is transaction cost is less expensive than installing and operating new controls. Likewise, if risks and returns on investments are favorable then non-point dischargers will elect to generate and sell credits to point sources. Despite these presumptions, established market infrastructure, and political support, water quality trades have been scarce. Programs, on the other hand, have continually developed since the 1980s, and by 2004 more than 70 water quality initiatives have been set up in the United States (Breetz, 2004). Some trades have resulted from these initiatives, but a lack of supply of credits from non-point source producers and a lack of demand from point sources are the two primary reasons for this scarcity of trading. One of the hesitations that hinder participation in a water quality trading program is the assessment that there is an incomplete economic valuation of the trading commodity, the reduction of nutrients from the non-point sources. In order to better place a value on options for trading, stakeholders will need a stronger approach to credit pricing (King, 2005; King & Kuch, 2003). As well, there is a need to better understand the costs of creating a market that links point sources with non-point sources.

Transaction Costs

Stakeholders have various types of transaction costs that they must consider before they become involved in a water quality trading program. Stavins (1995) points out that there are two potential types of transactions costs in permit markets. Information costs and bargaining costs are both impediments to the implementation of property rights. Stakeholders are faced with many decisions during the implementation stages of a water quality trading program. These decisions are driven in most cases by the available information that is easily accessible and the potential ease of bargaining for trades. Transaction costs in a water quality trading program are generated by the regulator and by the traders. In theory, costs should be internalized in the price of the credits to provide an estimate of the cost of trading. An excess amount of transaction costs have been identified as a primary reason for limited trading within markets that have been established, and most scholars suggest that transaction costs should be minimized in order to promote trading (Andersson, 1997; Gangadharan, 2000; Stavins, 1995).

Information and bargaining costs can be described in two broad categories when discussing water quality trading. Transaction costs can be incurred by either the agency or institution overseeing the program and they can be incurred by the traders. Agency or institutional development, execution, and oversight of trades are all part of the regulatory expenditures for operation and maintenance of the program. Other operational costs may include monitoring and verification, trade oversight, and enforcement. Monitoring and verification would consist of the costs for regulatory site inspection, discharge tracking, confirmation reports, and data management. Trade oversight would refer to costs for regulatory review of and approval of trade applications and external reporting for agency review. Enforcement refers to the costs of identifying and restricting point and non-point sources from discharging excess loads beyond their individual allocations.

Some of these agency or institutional trade support activities overlap other normal agency duties, and in some cases this would distort details of agency costs to support trading. In some cases documentation presents the overall costs that are separate for water quality trading participants. An example of this type of process occurs in the Cherry Creek Water Quality Trading Program just outside of Denver, Colorado. For this program applications cost $100 and a discharger must pay an additional $500 to cover costs incurred by the managing institutional authority to evaluate the request for credit withdrawal from the phosphorus bank. The cost to apply for credits from the reserve pool is $2,500, and this is regardless of the number of credits that may be involved (Breetz, 2004).

Transaction costs also may accrue directly to the traders, and these costs may include consulting costs, legal costs, or possible broker costs. Consulting costs refer to fees that would be paid to a scientist or engineer that would advise a trader of the technical risks and returns based on using a BMP in order to reach compliance or generate a credit. Legal costs refer to those fees that would be paid by credit generators and credit buyers to provide contracts for trades or to ensure that trades are executed with adequate legal protections. In some cases fees may also be paid to a broker or agent to gather credits and sell them to potential buyers. All of these trade-specific transaction costs are incurred by the producer or buyer of credits and these costs are proportional to their activities in the trade (Fang & Easter, 2003).

Credit producers would expect to generate a profit from the implementation of a water quality trading program. These non-point source reducing credit providers in some cases commit capital to water quality trading programs in order to create value for their organizations. At the present time participation in a water quality trading program may present several risks to value creation. Some risks may include the potential loss of government subsidies, negative cash flow because of upfront costs of construction, or possible assumption of regulatory discharge restrictions. Likewise, credit producers may also gain from opportunities associated with the implementation of certain types of BMPs. These benefits may include reduced operating costs, improved land value, or increased revenue generated from credit sales or payments for enhanced land usage like hunting permits, eco-tourism, or other types of recreation. Credit producers could possibly use consultants and analyses of BMP costs and effectiveness to lessen the risks and increase the opportunities of trading pollution credits. Producers must also assess program risk before committing to develop BMPs for water quality trading purposes.

It has been suggested that point sources and non-point sources do not compete equally on a cost basis because non-point sources receive subsidies and green payments to implement voluntary programs. Some arguments have demanded actions that would make the compliance standards for both entities more equal in nature by: reducing subsidies and regulating point sources and non-point sources equally (USEPA, ORD, 2006; ETN, 2006; National Forum, 2005). The non-point source discharger may place more emphasis on the risks of a water quality trading program compared with the benefits. These risks may include non-compliance risks, subsidy risks, discharger risks, performance risks, and production risks.

Non-compliance risks refer to those risks that are incurred through the acceptance of regulatory audits and inspections of ongoing BMP operations. If the BMP is not constructed or maintained properly then the credit producer could be faced with non-compliance penalties. Subsidy risks are those risks that focus on additional regulatory oversight that might lead to the loss of subsidy compensation for perceived control BMPs installed under a different green payment system. Likewise, most non-point sources are unregulated, and discharger risk refers to those risks that increase regulatory involvement in their current operations. Performance risk refers to those risks that deal with the expectation that the BMP will perform as a nutrient reduction technology. There is no guarantee that all BMPs will perform at the levels that they are expected to. Production risks also may be of concern. This risk is the risk that non-point sources produce credits for pollution reduction, but they are unable to sell their credits because the market structure may have collapsed, there were no buyers, or the price for the credits was too high for available buyers. These perceived risks could be allocated as costs into the price of the potential credits to be sold. However, in most water quality trading programs contingencies reflecting risks to private interests were not accounted for (King, 2005). Agencies could reduce many of these risks by correctly structuring the water quality trading market. By providing legal protections and removing regulatory uncertainty agencies could increase the benefits of trading by encouraging participation and influencing credit supply.

Costs of Creating a Water Quality Trading Market

The selection of the best market structure for water quality trading involves professional collaboration, research, consultation, and careful assessment of stakeholder perspectives. Open trading, closed trading, and full closed trading market alternatives are available. Initial decisions determine the types of trades that are allowed and the geographic limit of completed trades. Market structures must balance the needs for low-cost trades that ensure environmental protection with minimal oversight (Woodward, 2002). Clearly defined rules, responsibilities, and conveyance of liability are crucial initial considerations.

Market structures can be categorized as exchanges, clearinghouses, bilateral negotiations, and sole-source offsets based on several criteria. These criteria may include: the commodity traded, the market size, the market structure, the purpose of the program, and the governing authority for water quality (King & Kuch, 2003). Regulators have the important role of structuring exchanges so that credit pricing is attractive based on the criteria mentioned. Regulators also have a great deal of influence over the trading organization responsible for approving trades, protecting the environment, and administering the data generated by trading.

Market designers create demand in a water quality trading market by assigning source responsibility for effluent control and setting discharge limits. Allowances must be measurable and easily quantified by all parties. Demand for credits increases as the spread increases between internal cost of compliance and trading costs. When regulations are weak and the cost of enforcement or penalties are low then demand will not increase (King, 2005). Market designers and regulators must also define the watershed water quality objectives during the initial phases of program formation. Total maximum daily loads (TMDLs), or other types of limits must be set and enforced at early stages of the program implementation. Regional watershed water quality objectives, such as TMDLs, provide the primary driver for water quality trading (Stephenson & Shabman, 2001). These types of water quality objectives can be expressed as pollution caps, step-down caps, fractional rate reductions, or other metrics that are clearly measurable in space, time, and mass. These objectives must be set early on in the trading development process, and they must be enforced. This was not the case in a specific example concerning water quality trading markets in a Maryland jurisdiction. TMDLs in that location according to King (2005) were not enforced, and buyers and sellers did not appear because they lacked a tradable commodity. In the meantime, regulators used traditional command-and-control methods to direct point source investment in traditional wastewater treatment technologies while at the same time providing subsidies to non-point sources to install BMPs. These actions led to an overall pollution reduction at a much higher cost than what would be expected in a trading program (King, 2005). These extra funds could have been spent elsewhere in the watershed to create additional ecosystem value without additional pollution management cost or risk.

Increased supply of BMPs and interactions between point sources and non-point sources can be effective in producing increased ecological benefits to a watershed at a lower cost than traditional methods. Supply increases occur when non-point source polluters implement BMPs. This reduces their discharges and creates allowance credits that can be sold to point sources. Many factors may influence supply including: financial risk, BMP cost, and other transaction costs. In the Cherry Creek Basin market some restrictions on BMPs limited supply. Credits in the basin had to be generated where previous BMPs had not existed. Modifications to existing structures to decrease phosphorus loadings were not counted as credit generators in the program (Breetz, 2004).

The four fundamental criteria of an effective trading program involving non-point sources include: equivalency, additionality, accountability, and efficiency. (Fang & Easter, 2003) Equivalency is a measure of how pollutant loads from various sources relate to the pollutant of concern to be offset. This measure is vital to understanding what pollutant is being traded. In many cases conversion ratios account for temporal, spatial, and/or chemical differences in the sources. Additionality refers to any non-point source offset that would have occurred regardless of the trading program. These actions cannot count towards credits in a market. This prevents double counting by ensuring that a nutrient control activity counts toward only one objective if multiple objectives are met. For example, phosphorus reduction from a BMP that is already necessary for land development activities is not eligible for trading as was the case in the Cherry Creek Basin restrictions (Breetz, 2004). Accountability mandates appropriate monitoring and oversight to ensure proper implementation of all program requirements. Monitoring and verification directly affect the first three criteria that address technical and administrative issues necessary to evaluate efficiency. The question as to what level of monitoring is needed on the non-point side of trading has not been analyzed to date. Most of the time trading ratios are incorporated instead of implementing a monitoring plan. In most circumstances the last criterion of Fang and Easter's fundamentals of an effective trading program is the one that receives most attention. Efficiency implies that the trade will proceed only when one source is able to more cost-effectively reduce its discharges as compared to another source. This condition is critical to making the program financially attractive and cost effective (Fang & Easter, 2003).

Another crucial element of a trading program is the link between point sources and non-point sources. This element is vital to the creation of a water quality trading market that seeks to decrease nutrient loads on a watershed. From a rational standpoint it seems clear that point source discharge sources would initially search for inexpensive ways to improve their internal pollution reduction technology in order to avoid paying another source to make reductions for them. In many cases small internal adaptations are implemented before long-term compliance strategies are adopted. But, when costs for pollution control become incrementally high for a particular firm, then the firm may decide to implement their next best alternative. King (2005) has suggested that in many cases the next best alternative for regulated firms is to game the system. According to King (2005), the expected marginal cost of gaming will approach zero based on weak trading rules, non-enforcement, and trifle penalties for non-compliance. He has also suggested that state and/or federal limits on individual discharges will be required before their will be any credits to trade despite well-designed exchanges for trading (King, 2005).

These monitoring and enforcement activities will have to addressed in order for water quality trading programs to thrive, but they do not affect the concept development and market development costs of setting up a water quality trading program. Several one-time setup costs will occur when a market is being formed. Once the market is operation then certain administration and maintenance transaction costs occur. Initial one-time costs may include, but are not limited to: program review and approval costs, baseline assessment costs, credit allocation costs, market development and stakeholder buy-in costs, and BMP development and credit pricing costs. In order to cut costs and improve internal efficiencies some lead agencies hire dedicated staff for water quality trading market development (Jaksch, 2000).

Program review and approval costs can be considerably variable based on watershed-specific physical conditions, stakeholder views, and other agency factors. Agencies that are interested in conducting water quality trading in a specific area consider watershed-specific criteria such as geology, hydrology, ecology, biology, and point and non-point source pollution distribution.

Baseline assessment costs refer to the assessments that take place before the market begins to operate. Agencies oversee field studies that focus on the current state of the ecosystems, hydrology, biota, and other natural systems in the watershed. Approximate discharge histories for point sources and non-point sources are studied as well. An example of this type of baseline assessment took place before the Tar-Pamlico Trading Program was implemented. An association of prospective point source traders paid $300,000 to develop a special estuary model to track and predict the behavior of pollutant behavior in the river (Gannon, 2005). Grants, subsidies, or other types of contributions might be available to offset most baseline assessment costs, and these costs would not become a transactional burden for trades to occur.

Credit allocation costs refer to the assessment of the distribution of the total point source and non-point source pollution loads on a watershed. The total load for a water body is determined by the sum all pollution loads accounting for seasonality, projected growth, and a margin of safety. Watershed management action plans usually help to determine allocations with caps being determined by total maximum daily loads (TMDL) or a total maximum annual load (TMAL). Modeling and monitoring techniques are used to calculate the distribution among individual discharges (Michigan DEQ, 2002). The likelihood and value of an exchange is based on the urgency for stakeholders to trade. Regulators can create a marketable commodity by allocating allowances (pound per day). The tools used and costs incurred to set allowances depends upon the complexity of the pollution problem within a given watershed. Site specific characteristics determine the allocations based on program review and baseline assessments.

Lead agencies are responsible for identifying and engaging stakeholders at the water quality trading program level and at individual project levels. Governments, private parties, institutions, and other stakeholders create trading structures that fit their stakeholder needs based on the ecological situation, regulatory jurisdiction, local economy, and impacted natural resources. Some obligations may include leading public hearings, addressing stakeholder concerns, developing and maintaining communication channels, and arranging education and public outreach. Specific costs for these services vary depending on stakeholder sensitivities, special interests, and the regulatory structure proposed (Fang & Easter, 2003). Agencies also are able to develop the market by establishing supply and demand by creating a marketable commodity, proposing an attractive price structure, and retaining control of pollution discharge risk. For example, 40% of the Cherry Creek Basin Water Quality Authority's (Authority) budget is assigned to monitoring, special studies, planning documents, technical reports or memoranda, and administrative costs (Cherry Creek Basin Water Quality Authority, 2006). Some of these costs overlap into other transaction cost categories, but most of these costs fall into the market development category.

Costs of Generating a Water Quality Trading Credit

One very important factor concerning information and bargaining costs in the type of pollution technology that will be implemented at a non-point source location. This kind of technology to reduce non-point source pollution is most commonly referred to as best management practices (BMPs). In order to generate credits to create the supply for a market potential credit producers are faced with costs of producing an effective BMP. There are three sub-costs that must be considered by the party seeking to generate credits. These costs include the cost to create the opportunity to trade, the cost to implement a best management practice

(BMP), and the cost to manage the BMP. At the watershed level the sum of all three of these costs must be compared with other costs like point-source controls, zero-growth, or gaming the system in order to better estimate the cost-effectiveness of a credit generation decision (King & Kuch, 2003).

Private and public benefits can be gained from the implementation of certain types of BMPs. More efficient farming techniques and increased property values can be realized by private landowners, and these benefits can be assessed by traditional market-use valuation techniques. Public benefits also can be realized through the incorporation of some types of BMPs. Raffini and Robertson (2005) have suggested that the incorporation of restored or constructed wetlands as BMPs could reduce nutrient loadings under a water quality trading program. Heberling, Thurston, and Mikota (2007) further develop this point by suggesting that public benefits like habitat creation and carbon sequestration could be realized through the implementation of this type of BMP. A thorough net benefit valuation would be appropriate in some cases when involving stakeholders, selecting a BMP to implement, or valuing credits in the marketplace.

The cost of implementing a BMP is established from the investments made to design, permit, and install the technology that is potentially tradable. Certain BMPs are not tradable and these include practices that are required by law, funded by subsidies, or government programs that do not involve water quality trading. In some cases these competing BMP incentives or regulations reduce a producer's potential to generate credits (King, 2005).

The cost of implementing a BMP is the responsibility of either the point source without compensation, the non-point source with compensation, or by a third-party. This cost includes expenses that resulted from the design, installation, and management of the BMP during construction or implementation. Some BMPs are quite simple and involve changes in processes like a change in tillage or a relocation of livestock. On the other hand, some BMPs involve constructed vegetated filter areas or even wetland restoration initiatives. The cost of BMP implementation can range widely. For example, in the Tar-Pamlico program values for agricultural BMP ranged from $1 to $80 per pound of nitrogen. Similar values for wetland restoration ranged from $11 to $20. Values for stormwater BMPs ranged from $57 to $86 per pound of nitrogen removed from urban runoff (Gannon, 2005).

Private landowners and other non-point dischargers would typically account their cost of BMP implementation in terms of dollars. These costs can be referred to as sunk costs once a BMP is constructed or implemented regardless of whether a credit is generated or a trade is enacted. On the other hand, the regulatory agency that oversees the trading program values the BMP investment from a public perspective. Their value is determined by the public benefits that are created from the BMP. Therefore, long-term measures like easements and re-vegetation efforts (buffers, wetlands, etc.) are the most effective of BMPs because they provide pollution reduction with minimal change in practice. Likewise, the lifetime of these types of controls is more similarly comparable to the lifetime of the pollution reduction estimation. This minimizes risks and provides more certainty for prospective trades (Fang & Easter, 2005).

At the present time, many state regulatory agencies are charged with identifying and listing BMPs that non-point sources could implement to generate credits. The USEPA and US Army Corps of Engineers do not identify or maintain a list of BMPs at the national or local level. In many cases, the list is created from a larger and more complex list of effluent reduction technologies and strategies that have been used in site-specific instructions to dischargers (Breetz, 2004). The cost for acquiring these assessments is minimal, but there are no standard lists, and individual programs include various types of BMPs with various cost and performance valuations.

The performance of a BMP is a very important component of ascertaining the credit value. Ineffective or inconsistent BMPs are worth proportionally less than more effective ones, and this value is reflected in both the trading ratio and the trading price. The trading ratio is a figure that represents the number of credits that a buyer must purchase in order to receive one pollution credit. The trading ratio usually includes the BMP effectiveness, safety factors, and administrative factors. USEPA's water quality trading policy suggests methods to address uncertainty in non-point source pollution control. USEPA suggest the implementation of ratios greater that 1:1. Trading ratios express the quantity of expected non-point source pollution reduction that is necessary to generate a given quantity of pollution credits for use by a point source. Ratios can fall into a range from 2:1, 3:1, or even 4:1. Trading ratios conceptually serve as tools to address uncertainty, but are unnecessary. Risk-adjusted pollution reduction of certain non-point source pollution abatement activity is what is important. However, USEPA's policy does not indicate that trading ratios depend on the type of abatement activity (USEPA, 2004). Trading sites may not discriminate between types of BMPs. For instance, a reduction in fertilizer application, a buffer strip, and a restoration of a wetland may all be grouped together as BMPs, but they definitely have different levels of uncertainty based on their abatement performance over time. By grouping abatement activities together this way, trading ratios make sense as an averaging technique. In order to maximize gains from trade and encourage active pollution credit markets, ratios should approach 1:1. In order to achieve a trading ratio of unity, the uncertainty of non-point source pollution abatement activities must be reduced. The verification of load reductions by non-point sources through increased monitoring could move trading ratios closer to unity. At the present time, almost all of these trading ratios in any given program will exceed one because safety factors and uncertainty usually exceed one (Breetz, 2004). Ratios are usually determined near the beginning of program development, and some programs like the Great Miami

River Watershed Water Quality Credit Trading Pilot Program have used initial ratio offerings to encourage trades between point sources and non-point sources (Kieser & Associates, 2003). As trading ratios increase then the price differential between buyers and sellers decreases. In the Great Miami River Watershed Water Quality Trading Pilot Program initial offerings were set at 1:1 ratios. This initial offering was not based on monitoring information concerning the uncertainties and risks of various BMPs. It was initially arranged to increase the involvement of stakeholders in the program. After a TMDL is set in place during the next couple of years, the ratios will change to 2:1 or 3:1 (Kieser & Associates, 2003). BMP credits in certain programs should also reflect their specific lifetimes. Tar-Pamlico credits for non-structural BMPs (activities like no-till farm techniques) were assigned a credit life of three years. Structural BMPs were assigned a credit life ten years (Breetz, 2004).

When a BMP is chosen and implemented, it is the responsibility of the installer to meet credit requirements. Appropriate maintenance, monitoring techniques, organized data management, and compliance reporting are all duties that should be performed. Failure to meet credit requirements may result in penalties paid by either the BMP discharger or the point-source discharger depending on the institutional structure of the program and types of contractual agreements. Monitoring criteria may be judged by performance or by activity (King & Kuch, 2003). Costs will be negligible for simple practices, such as rearranging agricultural field sites. Performance costs for network monitoring will be low to moderately expensive depending on: the applied technology, the size and density of the monitoring network, and the frequency of monitoring events. In some cases capital costs for fixed monitoring devices can add to the costs significantly. Activity costs may be negligible for simple practices like rearranging ranch grazing, but other activity costs like restoring wetland acres may be significant.

Impediments to Incorporating Wetlands into a Water Quality Trading Program

The literature suggests that water quality trading markets that incorporate wetlands may prove to be successful if they are created with well defined property rights, include a large number of buyers and sellers, acquire and maintain good information concerning nutrient reducing capabilities of wetlands, minimize transaction costs, and exhibit rational behavior by stakeholders. These conditions are necessary for an effective water quality trading market to exist, and the literature provides background information of the applicability of these key components to a water quality trading program that incorporates wetlands at the watershed scale.

Property rights in water quality trading programs are defined with the use of a watershed cap on nutrient loads (Stephenson & Shabman, 2001). These caps are referred to in the Clean Water Act as total maximum daily loads (TMDL). Discharge restrictions that enforce this cap must be binding and set at an optimal level to reflect scarcity (Coase, 1960; Dales, 1968; Tietenburg, 1985). Therefore, an derived value is given to those entities that can reduce nutrient loads in the most efficient and least costly way. Supply and demand for water quality markets is dictated by this cap (Faeth, 2000; King, 2005).

A certain number of buyers and sellers must be accounted for when assessing the applicability of water quality trading programs. Unlike the air quality trading programs, water quality trading programs are confined to watershed basins, and only a certain number of point sources and non-point sources may be located within a particular area. There is no reference in the literature that suggests how large or small a water quality trading program can be, and this question of scale will be better determined as more markets are defined and become fully functioning (Faeth, 2000).

Good information is necessary for water quality trading markets to work, but at the present time there is little consensus concerning the nitrogen and phosphorus fate in wetlands

(Mitsch & Gosselink, 2000). Site specific examples have provided a great deal of information concerning the applicability of wetlands to absorb or immobilize nutrients, but at present, no successful watershed scale wetland assessment projects have been monitored, managed, or described. Increased information concerning the ability for wetlands to reduce nutrient loads within a watershed context would be very helpful in defining the commodity (pounds of nutrient) that could be traded between point source and non-point source.

Transaction costs could be a big impediment to water quality trading programs that incorporate wetlands (Andersson, 1997; Gangadharan, 2000; Stavins, 1995). The literature supports the idea that monitoring and verification of credits are substantial transaction costs. There are also transaction costs that take place between point and non-point sources. These costs of exchange of credits may be lessened with government taking on the responsibility for non-point source reductions, but there may also be large governmental transactions costs because of inefficient transaction processes or cumbersome application and assessment of credit processes (King & Kuch, 2003; Williamson, 1985).

Lastly, from an economic impediment standpoint, stakeholders must exhibit rational behavior (Williamson, 1986). This idea of rational behavior reflects the choice by a non-point source to install a wetland BMP versus some other type of BMP. The stakeholder may choose to do this if he were to receive a better credit price because of the efficiency of wetland to reduce nutrients, or a particular program may ascribe higher credit value for wetlands versus other best management practices.

As the literature suggests, environmental trading markets are not natural. In fact, these markets are contrived to approximate market conditions with environmental conditions imposed. Therefore, proper institutional arrangements are vital to the performance of these types of markets.

From an institutional standpoint the initial setting and allocation of a cap must be arranged. There must be an institutional enforcement mechanisms in place in order to enforce this type of cap. The literature suggests that one of the reasons that water quality trading markets might not work depends upon the costs of noncompliance. If the costs of noncompliance are not high then there is no demand to trade (King, 2005).

Secondly, the literature suggests that environmental trading markets are inherently risky. If these risks are not assigned to the buyer or seller of water quality credits, then by default, they fall on the public (Fang & Easter, 2003; King, 2005). Information about the nutrient reducing capabilities of wetlands may decrease the level of risk that buyers, sellers, or the public faces, but other institutional mechanisms may spread out risk and more directly focus liability.

One of the more challenging issues described in the literature that affects the supply side of water quality trading is how credits are generated, or how baselines for trading are defined. Credits are usually determined by ratios, and in most cases there is not any direct monitoring application for a type of wetland. In accordance with additionality (Fang & Easter, 2003), trading programs usually prohibit farmers from selling credits for undertaking land use/land management changes that the farmer has received green payments for. Although farmers are being paid through green payments to undertake best management practices (BMPs), the real intention of these programs is to provide subsidies to agriculture.

Monitoring and verification of nutrient reductions could be an institutional impediment as the literature describes. Water quality trading markets are never self-regulating, and trades can be visualized in a three way fashion that entails the point and non-point source and the regulator. Monitoring and verification is necessary in order to define what amount of a nutrient is being reduced (Wetzel, 2001). Water quality trading transaction costs may not allow for monitoring of every BMP, but some of the risk needs to be removed from the trades without increasing transaction costs to the extent that it stifles the programs. The right amount of monitoring and verification must be assigned through an institutional setting.

Lastly, from an institutional standpoint, it is necessary that government entities account for the positive and negative environmental externalities that may be associated with wetlands that are incorporated into a water quality trading program (Leitch & Frigden, 1998; Whitehead, 1992). The literature has revealed that, at present, markets fail to account for the externalities that are associated with wetlands, and institutional mechanisms must be clearly defined to account for and distinguish these externalities that are associated with wetland BMPs versus other BMPs. By identifying and focusing on the positive externalities associated with wetlands an institutional arrangement would provide the public goods that are generated from wetlands while at the same time minimizing the negative externalities that may be associated with these types of BMPs.

CHAPTER 4

METHODOLOGY

Research Design

The outline for the research design was developed based on the question: Is the incorporation of wetlands technology to enhance water quality trading programs economically and politically feasible at the watershed scale? The research outline was arranged in a way that focused on economic and institutional failures. Economic failure issues that were identified in the literature included: the setting of an optimal nutrient cap, property rights issues (initial distribution of credits), current available and readily accessible information, the positive and negative externalities generated from incorporating wetlands into a trading program, and the transaction costs of creating a trading program. Institutional failure issues many of which relate to the establishment and maintenance of a market framework were identified included: the setting of a cap (TMDL), the assigning of property rights concerning current transactions that focused on risk and liability issues, information that dealt with the starting point (baseline) for trades between point and non-point sources, and the role of monitoring and verification to identify positive and negative externalities and minimize transactions costs.

Information for this study was collected from four water quality trading programs: Cherry Creek WQT Program in Colorado, Tar-Pamlico River WQT Program in North Carolina, Lower Boise River WQT Program in Idaho, and Neuse River WQT Program in North Carolina. Face-to-face interviews took place with state and federal environmental officials, professionals and private technical consultants contracted by trading programs, representatives of trading program organizations, and other stakeholder group representatives during the month of April 2007 (see Appendix A.1). Two local stakeholder meetings were observed: the Lower Boise River Watershed Council (April 12, 2007) and the Tar-Pamlico Association Meeting (April 25, 2007). Both of these meetings included representatives from point sources, non-point sources, government, and environmental organizations.

The case study selection process began with the study of approximately 70 water quality trading programs in the U.S. and 10 other programs throughout the rest of the world. Information for these programs was acquired mainly through resources provided by Virginia Kibler. She is an economist with the Office of Water at the USEPA Headquarters in Washington, D.C. Supplementary information was gathered for programs in the U.S. from compiled reports on the status of water quality trading programs (Breetz, 2004; Evironomics, 1999). A matrix was constructed to compare the 37 major water quality trading programs in the U.S. as of 2006 (See Table 4.1). Selection criteria incorporated into the matrix to help identify the potential case study candidates for the research. Criteria included: location defined by state, name of the project, the year trading applications were defined, the type of pollutant being traded, the types of trades (point source (point source) to non-point source (non-point source), non-point source to non-point source, etc.), if wetlands were incorporated into the trading program, and an assessment of whether the program was a candidate for the research study based on the six other factors. From the matrix nine programs were identified as being candidates for the research study based on the six stated selection criteria. Each of these nine programs were further reviewed and assessed for the applicability of their inclusion into the study. The nine programs that met the candidate criteria were Boulder Creek, Cherry Creek, the Lower Boise River, the Massachusetts Estuary Project, the Minnesota River, Rahr Malting Co., Southern Minnesota Sugar Beet Cooperative, the Neuse River, and the Tar-Pamlico River.

After further review of these nine programs more in-depth information revealed that several of the trading programs were labeled as trading programs, but they were structured as offset programs that were inclusive within one particular entity. For instance, the Boulder Creek trading Program was not a formal trading program. The City of Boulder, CO simply created an offset framework to purchase land and implement stream restoration projects. This was also the case for the Rahr Malting Co. Trading Program, and the Southern Minnesota Sugar Beet Cooperative. The Massachusetts Estuary Project and the Minnesota River Water Quality Trading Program did not have trading structures in place, and they did not have regulatory drivers to initiate the trading process as of 2006. Therefore, the four case studies that met the criteria for research concerning the economic and political feasibility of incorporating wetlands into a water quality trading program at the watershed scale were: the Colorado Cherry Creek Water Quality Trading Program; the North Carolina Tar-Pamlico River Water Quality Trading Program; the Idaho Lower Boise River Water Quality Trading Program; and the North Carolina Neuse River Water Quality Trading Program.

Table 4.1. Decision Matrix of Current Effluent Offset or Trading Programs in the United States

Ref. #	Loca-tion	Project	Year Trading Adopted	Pollutant	Types of Trades	Wetlands Used in Trading?	Candidate for Study
1	CA	Grassland Area Farmers	1998	Selenium	NPS-NPS	no	no
2	CA	San Francisco Bay	2003	Mercury	not determined	no	No
3	CA	Sacramento Regional County Offset Program	2002	Mercury	PS-NPS	no	No
4	CO	Bear Creek	1990	phosphorus	PS-PS	no	No
5	CO	Boulder Creek	1990	Nitrogen	PS-NPS	yes	Yes
6	CO	Cherry Creek	1998	phosphorus	PS-NPS	yes	Yes
7	CO	Clear Creek	1994	Heavy metals	PS-NPS	no	No
8	CO	Lake Dillion	1984	phosphorus	PS-NPS and NPS-NPS	no	no
9	CO	Lower Colorado	2002	Selenium	PS-PS, PS-NPS, NPS-NPS	no	No
10	CT	Long Island Sound	1997	Nitrogen	PS-PS	no	No
11	ID	Lower Boise River	2001	phosphorus	PS-NPS	yes	Yes
12	IL	Piasa Creek	2001	Sediment	PS-NPS	no	No
13	MA	Town of Action	1998	phosphorus	PS-NPS	no	No
14	MA	Charles River	2003	water flow	PS-NPS	no	No
15	MA	Edgarton	1999	Nitrogen	PS-NPS	no	No
16	MA	Falmouth	2003	Nitrogen	PS-NPS	no	No
17	MA	Massachusetts Estuaries Project	2001	Nitrogen	PS-NPS	yes	Yes
18	MA	Specialty Minerals	2002	temperature	PS-NPS	no	No
19	MA	Wayland Business Center	1998	phosphorus	PS-NPS	no	No
20	MA	Nashua River	2005	phosphorus	PS-NPS	no	No
21	MI	Kalamazoo River	1996	phosphorus	PS-NPS	no	No
22	MN	Minnesota River	1998	phosphorus	PS-PS	yes	Yes

Table 4.1. Decision Matrix of Current Effluent Offset or Trading Programs in the United States (continued)

Ref. #	Loca-tion	Project	Year Trading Adopted	Pollutant	Types of Trades	Wetlands Used in Trading?	Candidate for Study
23	MN	Rahr Malting Co.	1997	phosphorus, nitrogen, Sediment	PS-NPS	yes	Yes
24	MN	Southern Minnesota Sugar Beet Coop.	1999	phosphorus	PS-NPS	yes	Yes
25	NC	Cape Fear River	2005	undefined	PS-NPS	yes	Initial planning stages
26	NC	Neuse River	1997	nitrogen	PS-NPS	yes	Yes
27	NC	Tar-Pamlico River	1992	Nitrogen and phosphorus	PS-NPS	yes	Yes
28	NV	Truckee River	2001	nitrogen, phosphorus, Sediments	PS-PS, NPS-NPS	no	No
29	OH	Clermont County	1994	Nitrogen and phosphorus	PS-NPS	no	No
30	OH	Greater Miami River	2004	nitrogen and phosphorus	PS-NPS	no	No
31	PA	Conestoga River	2003	nitrogen and phosphorus	PS-NPS	no	No
32	VA	Colonial Soil and Water	2005	nitrogen and phosphorus	PS-NPS	no	Initial planning stages
33	VA	Henry County	1997	Total Dissolved Solids	PS-PS	no	No
34	WI	Fox-Wolf Basin	1997	phosphorus	PS-PS and PS-NPS	no	No
35	WI	Red Cedar River	1997	phosphorus	PS-NPS	no	No
36	WI	Rock River	1997	phosphorus	PS-PS and PS-NPS	no	No
37	WV	Cheat River	2005	heavy metals	not defined	no	Initial planning stages

(Compiled from EPA documents provided by Virginia Kibler (Kibler, 2007), Hanna Breetz et. al., 2004, and Environomics 1999.)

The outline for the interviews was constructed in a way that would uncover information related to the background of the program, the track record of the program, and the market and institutional impediments that were associated with the particular program being studied. Specific information under these headings was identified, and the interviews followed the format listed:

Background

- History
- Issues
- Driving force for the establishment of trading system
- Administrative Unit – oversight
- Stakeholders
- Structure of trading system

Track Record

- # of permit trades (point to point and point to non-point)
- Water quality impact
- Costs of program

Impediments

- Market failure issues
    - Info – current available info – is info readily accessible
    - Externalities
    - Initial distribution of credits
    - Transaction costs of creating a trading program

- Institutional failure (arrangements/impediments)
    - Current transactions
    - Setting of a cap (TMDL, TMAL, etc.)
    - Starting point for trading between point sources and non-point sources (baselines)

Additional market, institutional, and wetland specific questions were encapsulated into this format. Market association focused on assessing the market structures of the programs, the standards used to make the programs efficient, and the transaction costs and alternatives to incorporating a trading program that includes wetlands technology. The institutional questions centered on the discussion of the regulatory drivers for the program, the administration of the programs, and the interactions between point and non-point source entities. The wetland questions focused on the reasons for incorporating wetlands into the specific program being studied, the methods of monitoring and verifying wetland credits, and the desire to incorporate wetlands as the preferred type of restored BMP for the program versus other types of BMPs (See Appendix B.1).

A semi-structured interview protocol was constructed to gather descriptive data regarding the experiences of the various stakeholders concerning the adoption of the water quality trading program, the political debates that took place concerning effluent allocation loads, the structural factors influencing actual trades, the performance record of the program, and the economic and political feasibility of incorporating wetlands technology into the program. The semi-structured interview process used a core set of structured emphasis areas and specific questions to branch off into less structured questions in order to explore responses in greater depth. The interview protocol was directed by a list of primary questions that were followed by a set of secondary questions.

The interview protocol was designed to allow open and flowing conversation. Through open-ended questions respondents were able to discuss their understanding of the motivations for the programs, the existing obstacles for the programs, and the general strengths and weaknesses of their existing programs. The type of semi-structured arrangement of the interview protocol allowed the individuals being interviewed to provide contextual and specific undocumented information. Water quality trading programs across the U.S. are in their infancy, and the best way to interpret what is occurring within these programs is to collect in-depth face-to-face interview data from various stakeholders, program administrators, state officials, and federal officials. This type of methodological approach provides insight into organizational processes, contextual situations, factors that inhibit trades from occurring, issues of actual trades concerning wetlands technology, and an understanding of what information and policy gaps need to be filled in order for these types of programs to work efficiently and effectively.

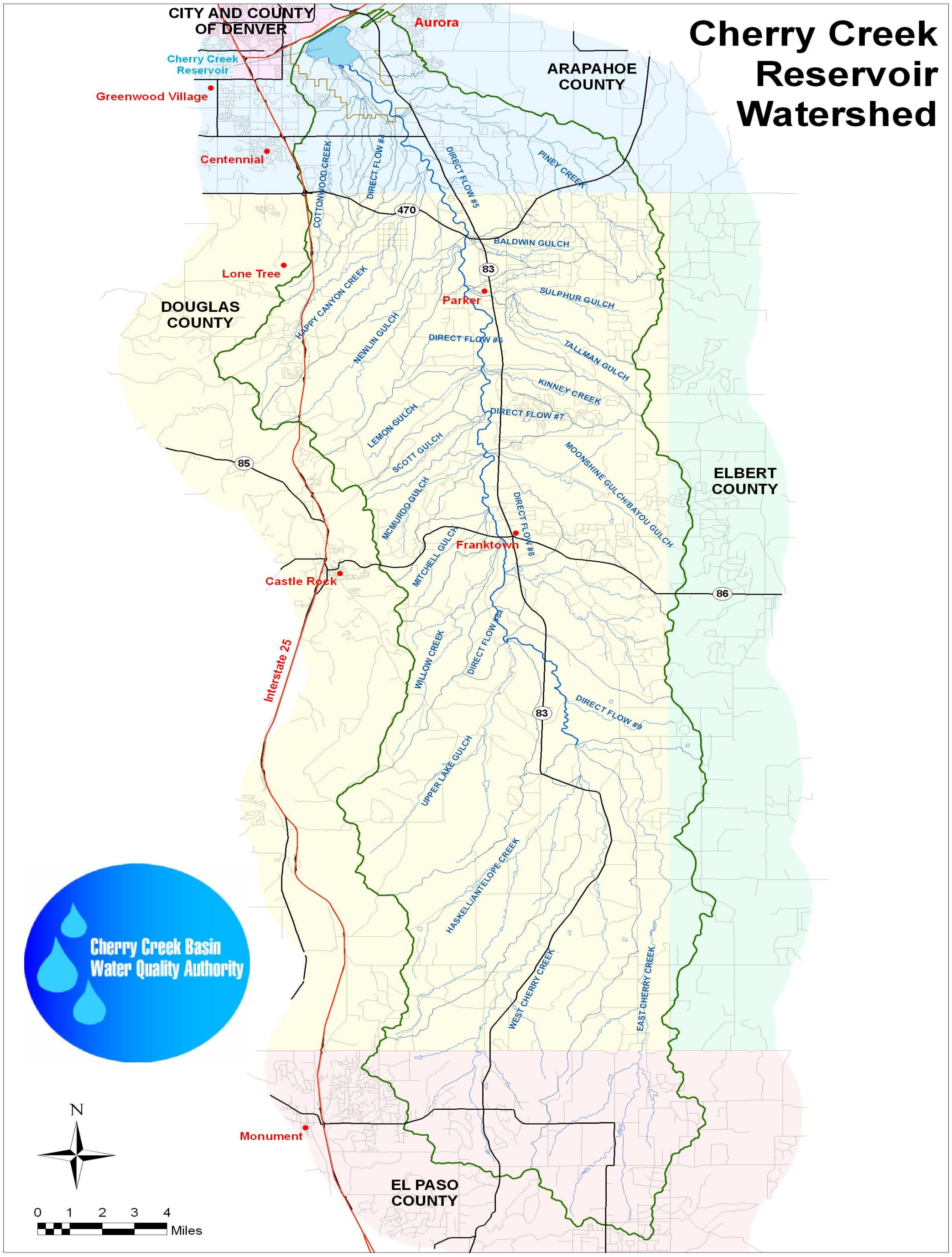
The case studies presented were chosen to represent water quality trading programs in different regions of the country, and they illustrate specific regional issues that may limit the feasibility of incorporating wetlands technology into a water quality trading program. The research design was structured strictly to address the research question to define the economic and political feasibility of incorporating wetlands technology into a water quality trading program at the watershed scale. The data collected from the cases are presented in a geographic order. The two western cases are discussed first, and then the two North Carolina cases are presented.

CHAPTER 5

CHERRY CREEK WATER QUALITY TRADING CASE STUDY

Background

Cherry Creek Reservoir was created in 1950 by the U.S. Army Corps of Engineers (USACE) to protect Denver, Colorado from flooding. Several devastating floods had inundated the young city during the early part of the 20th century, and the USACE built an earthen dam just to the southeast of the city to control the flow of Cherry Creek and the tributaries that fed into it. The damming of the creek created a reservoir for the City of Denver. Cherry Creek Reservoir is approximately 850 acres and is owned by the USACE. The USACE leases the reservoir and 3,915 acres of land surrounding it to the state of Colorado. The state of Colorado developed this land into a state park and created the Cherry Creek State Recreation Area. The Cherry Creek Watershed is located within one of the fastest growing areas in the country (R. Parachini, personal interview, April 10, 2007). As shown on the Cherry Creek Basin Watershed Map 5.1 (Annual Report, 2007), the watershed includes approximately 245,500 acres and 32 sub-watershed. The northern portion of the watershed has been experiencing extensive urban growth during the past 20 years. Only a small portion of the watershed, in the southern upstream area, still has agricultural influences. The Cherry Creek Reservoir and State Recreation Area serve as an important urban recreational site that receives extensive use including hiking, horseback riding, bird watching, bicycling, boating, swimming, and sport fishing. The recreation area is currently the most visited state park in Colorado with over one million visitors in the 2005-06 fiscal year (R. Parachini, personal interview, April 10, 2007). Development pressures in the watershed have led to increases in urban runoff, in-stream erosion, and higher levels of phosphorus discharges by point sources.



Map 5.1. Cherry Creek Reservoir Watershed Map (Wind, 2007)

In the late 1970s and early 1980s a series of fish kills were recorded in the reservoir. The reservoir was closed several times for recreational purposes during the 1980s because of extreme fish kills and the contamination due to e-coli. The CDPHE reported that "body contact" activities should not take place within the reservoir because of the extreme outbreaks of cyanobacteria in the reservoir (R. Parachini, personal interview, April 10, 2007).

Because of these harsh environmental episodes the state of Colorado undertook a massive study with financial support from the U.S. Environmental Protection Agency (USEPA) to determine the causes of the fish kills. The Water Quality Control Division (WQCD) of the Colorado Department of Public Health and the Environment (CDPHE) collected data between November 1981 and October 1982 on the reservoir as part of their Clean Lakes Studies (J. Minter, personal interview, April 11, 2007; R. Parachini, personal interview, April 10, 2007). Phosphorus was identified as the nutrient that had acted as a catalyst to increase the levels of the toxic blue-green algae in the reservoir that led to the kills.

Based on information gathered from the studies the Colorado Water Quality Control Commission (CWQCC) established a reservoir total phosphorus standard of 35 mg/L. This standard maintained that chlorophyll concentration levels in the reservoir would not exceed 15 mg/L as an average for the time period from July through September. An in-lake phosphorus model, the Jones-Bachmann chlorophyll-phosphorus model, was incorporated in 1984 to define the maximum allowable annual load of phosphorus from all combined sources that would maintain the standard of 35 mg/L.

Table 5.1. Cherry Creek Water Quality Trading Program Overview

Year Watershed Organization Formed	1985
Year Trading Guidelines Adopted	1998
Administrative Unit	Special Purpose District
Program Structure	Brokerage/Clearinghouse
Pollutant Traded	Phosphorus

Administration of Cherry Creek WQT Program

The mission of The Cherry Creek Basin Authority (Authority) is to "maintain beneficial uses in the Cherry Creek Reservoir by preserving its water quality" (Annual Report, 2007). The entity was formed by an intergovernmental agreement between point source wastewater discharges in 1985, but the Authority became a recognized statutory body in 1988 when the Colorado General Assembly enacted the Cherry Creek Basin Water Quality Authority Act (R. Parachini, personal interview, April 10, 2007). This act established the authority as a political subdivision of the state. The act also empowered the authority to develop and implement watershed plans in the basin, recommend and allocate wasteloads among sources in the watershed, raise revenue through taxes and fees, and develop and implement programs for credits or incentives for water quality control projects in the watershed. The Authority is comprised of two counties, four cities, and seven special districts committed to promote and maintain water quality in the watershed. The Authority's members include: Arapahoe County and Douglas County; the City of Aurora, City of Greenwood Village; Town of Castle Rock and Town of Parker; Arapahoe County Water and Wastewater Authority, Cottonwood Water and Sanitation District, Iverness Water and Sanitation District, Meridian Metropolitan District, Parker Water and Sanitation District, Pinery Water and Sanitation District, and Lincoln Park Metropolitan District

(Stonegate) (W. Ruzzo, personal interview, April 10, 2007). The Authority is legally mandated by Regulation #72 to administer the basin (R. Parachini, personal interview, April 10, 2007). Other entities that work closely with the Authority include the Colorado Department of State Parks, the WQCD, the WQCC, the Natural Resources and Conservation Service, and the U.S. Army Corps of Engineers.

The Authority originally consisted of board members that represented the special wastewater treatment districts. In 1996 the board encountered an issue with one wastewater treatment plant seeking expansion within the watershed, but the plant did not have the sufficient wasteload allocation for expansion. At this time other wastewater treatment plants recognized that their next increment of expansion would be at or near their particular wasteload allocation. The Authority members at the time incorporated the use of trading to obtain additional phosphorus wasteload allocations and to provide additional means for the authority to obtain credit for the substantial investment that it had made in non-point source control projects within the watershed (W. Ruzzo, personal interview, April 10, 2007). The Authority adopted trading guidelines, but the entity spent most of its funding and time during the late 1990s lobbying to gain significantly more flexibility in point source allocations for phosphorus. In the meantime the water quality diminished substantially in the Cherry Creek Reservoir. Thousands of handwritten letters were sent to Governor Bill Owens to persuade the state of Colorado to reject the Authority's plan to increase its allocation limits for phosphorus. The very affluent City of Greenwood Village became very irate at this time with the Authority's self-interested actions, and several members of the community lobbied the Colorado General Assembly to take the appropriate measures to focus the Authority on its mission to protect the watershed. Therefore, to curb the political interests of the Authority and to increase the equitable water quality rights of stakeholders in the basin the Colorado General Assembly passed the Colorado State Statute 25-

8.5-111(3) in 2001 (R. Parachini, personal interview, April 10, 2007). The statute restructured the representation of the board for the Authority and dictated that the Authority must spend at least 60% of revenues on construction and maintenance of pollution reduction facilities (PRFs). The remaining 40% can be allocated towards monitoring, planning, technical reports, or administration. The Authority had been spending only 22% of its budget on water quality improvements through PRFs.

The Authority receives funding for its activities from property taxes, Cherry Creek State Park user fees, wastewater bill surcharges, and building permit fees. The 2006 budget for the Authority was approximately $1,692,000 in revenue and approximately $2,896,000 in expenditures (Annual Report, 2007). Approximately 66% of the Authority's budgeted revenue was tax based, 29% was from fees and surcharges on wastewater, and 5% was from miscellaneous sources including grants.

Table 5.2 shows the makeup of the Authority after restructuring in 2001. It now consists of a Board and Technical Advisory Committee (TAC).

Table 5.2. Cherry Creek Basin Authority Organizational Structure

Member Entities	Number of 1988 Board Members	Number of 2001 Board & TAC Members
County (Arapahoe, Douglas)	2	2
Municipality (Aurora, Castle Rock, Centennial, Foxfield, Greenwood Village, Lone Tree, Parker)	7	5
Special Districts	1	1
Appointed by the Governor	7	N/A
Board Appointed	N/A	4
Other (Cherry Creek State Park, COE, DRCOG)	N/A	3
Total Members of Authority Board	17	15

(Authority Annual Report, 2007)

The Authority conducts annual water quality monitoring in the Cherry Creek Reservoir and basin. Reservoir water quality, reservoir inflow and loading, surface and groundwater quality in the watershed, and effectiveness of Authority pollution reduction facilities (PRFs) are all measured. Continued allocation of traded credits relies on both point and non-point source compliance with Regulation #72 and abiding by their revised allocations. The Authority is responsible for producing an annual report on watershed activities, and every three years the CWQCC must update Regulation #72 as necessary. This triennial review provides flexibility for improving the water quality of the watershed, and this review is critical to achieving current needs of the dynamic basin (W. Ruzzo, personal interview, April 10, 2007). The Authority was awarded two Colorado Non-point Source Program grants on October 27, 2005. The two grant projects are the "Cherry Creek State Park Wetlands" involving the design and construction of Phase 1 of the multi-phase wetlands construction project, and "TMAL Actions" to conduct three studies monitoring the outcomes specified in Regulation #72 (M. Wind, personal interview, April 11, 2007).

Water quality credits are not traded outside of the Authority. The Authority acts as a brokerage service or clearinghouse, and the Authority maintains the liability and responsibility that is incurred with trades that take place within the basin. The Authority must approve any withdrawal from the Phosphorus Bank or Reserve Pool. For each potential trade the Authority must consider the type of trade, trade ratios, monitoring and reporting needs, and other watershed dynamics associated with phosphorus loading into the reservoir. In order for a point source to purchase, lease, or create credits an application must be submitted to the Authority that justifies the need to trade, describe the project's design if applicable, provide a schedule for construction, and deliver a plan for operation, maintenance, monitoring, and reporting. The Authority considers comments from interested parties and holds a hearing before finalizing its decision over a proposed sale, lease, or transfer of credits (W. Ruzzo, personal interview, April 10, 2007).

Structure of the Cherry Creek Water Quality Trading System

The amount of discharged phosphorus per year standard was initially set at 14,270 pounds in 1983 for the Cherry Creek Reservoir, and by 1985 the annual load was allocated among wastewater treatment facilities (point sources), non-point sources, industrial, and septic loadings (M. Wind, personal interview, April 11, 2007). Approximately 74% (10,300 pounds per year [lb/yr]) was allocated to non-point sources and regulated stormwater sources, 15 % (2,000 lb/yr) to municipal and industrial point sources, 8 % (1,170 lb/yr) to background sources, and 3 % (450 lb/yr) to individual septic systems. Initially, an additional 3 percent was allocated to reductions achieved by the Reserve Pool and Phosphorus Bank, but that additional allocation was reallocated as part of the 2004 Triennial Review Hearing for Control Regulation #72 (Annual Report, 2007).

Table 5.3. Cherry Creek Basin Load Allocations

Allocation Type	Total Phosphorus Pounds/Year
Non-point and Regulated Stormwater Sources	10,506
Background Source	1,170
Wastewater Facility Sources Including Reserve Pool and Phosphorus Bank)	2,094
Industrial Process Wastewater Sources	50
Individual Sewage Disposal Systems	450
Total Maximum Annual Phosphorus Load	14,270

(Authority Annual Report, 2007)

In 1985 the CWQCC approved the master plan for allocation and, concurrently, adopted the Cherry Creek Reservoir Control Regulation which established the annual Total Maximum Daily Load (TMDL). Although the 14,270 pound limit is most often referred to as a TMAL or Total Maximum Annual Load. (R. Parachini, personal interview, April 10, 2007) After the adoption of this phosphorus standard the governmental entities in the watershed at the time came together to develop the Cherry Creek Basin Water Quality Management Master Plan. This plan was developed in order to prescribe a long-term focus for improving water quality in the Cherry Creek Watershed Basin. The Cherry Creek Basin Authority (Authority) was formed by local governments in the watershed in 1985 through an intergovernmental agreement. The authority was developed in order to implement the master plan developed to manage phosphorus loads on the reservoir through the TMAL (W. Ruzzo, personal interview, April 10, 2007).

In 1989 the Colorado Department of Public Health and Environment (CDPHE), through its Water Quality Control Commission, adopted the Cherry Creek Reservoir Control Regulation: Regulation #72. The regulation approved water quality trades for phosphorus reductions between point and non-point source dischargers and provided the Authority with regulatory oversight control of trading procedures. The CDPHE accepted trades with non-point sources despite the fact that these sources were unregulated. At the time the CDPHE estimated that approximately 80% of the phosphorus load into the basin could be attributed to non-point sources (R. Parachini, personal interview, April 10, 2007). Regulation #72 legally set forth policy guidelines for water quality trading, and satisfied the state's desire to incorporate a watershed program that would allow growth while at the same time preserving the aquatic ecosystem of the basin.

In November 1997 the CWQCC approved the Cherry Creek Water Quality Trading Program. The trading program is administered by the Authority and allows point source dischargers to receive credit from reductions of phosphorus from pollution reduction facilities

(PRFs) (W. Ruzzo, personal interview, April 10, 2007). In March of 1998 the Authority adopted specific procedures, policies, and standards for trading in the watershed. These water quality trading guidelines are much more detailed and comprehensive than the original control regulation. The guidelines authorize two types of trades: authority phosphorus bank trades and reserve pool trades. During Phase I of the trading program the Authority had only encouraged trades in conjunction with the four existing pollution reduction facilities that were designed, constructed, and maintained by the Authority. In Phase II of the trading program the Authority will continue to include additional PRF projects that it has funded, and the authority will approve PRFs developed by point sources as water quality credit generators (M. Wind, personal interview, April 11, 2007).

Revisions to Regulation #72 in 2001 established the TMAL allocating phosphorus loads into the basin to both point and non-point sources. In 2003 the Authority adopted the Cherry Creek Basin Watershed Plan. This strategy put forth revised trading program guidelines that comply with the 2001 modifications to the Regulation #72, and it provides a more detailed framework for trades. The plan set the surface water standard for TP at 40 mg/L. According to the guidelines, point sources have the opportunity to purchase or lease a total of 432 lbs of phosphorus for new or increased phosphorus wasteload allocations. They also have the option under Phase II of developing a PRF in order to generate phosphorus reduction credits.

The Cherry Creek Water Quality Trading Program provides water quality credits for phosphorus reductions for non-point source projects involving existing developed areas that originally lacked best management practices (PRFs), retrofits to PRFs in order to achieve a greater level of phosphorus detention, or PRFs in a new development that reduce a greater amount of phosphorus than what is required to comply with the TMAL. In the trading program one credit is equivalent to 1 pound of phosphorus per year. Trading of credits functions through a clearinghouse structure, and the Authority is the broker that can sell credits to dischargers needing to increase their allocation. A point source discharger may also trade directly with another point source discharger if the buyer has made a valid attempt to minimize phosphorus loadings (Breetz, 2004). Point sources must remove, prior to discharge, as much phosphorus as possible through primary or secondary treatment applications. For point sources the 30 day average concentration of phosphorus in effluent must not exceed 0.05 mg/L. Dischargers using land application must achieve a 30-day average concentration of phosphorus less than 0.05 mg/L divided by the return flow rate, unless lysimeters are used, in which case the effluent concentration limit is 1.0 mg/L (W. Ruzzo, personal interview, April 10, 2007). Such restrictions aim to control the release of phosphorus in the solid phase into the watershed through stormwater runoff.

The Cherry Creek Water Quality Trading Program has incorporated a number of safety factors to provide for accountability. These factors are included in the program to account for PRF project uncertainties. Fate and transport considerations of non-point source PRFs are individually taken into consideration in order to insure the decreased levels of phosphorus into the reservoir. In the program a trade ratio is applied to the pounds of phosphorus removed by a PRF to determine the pounds of credit that can be generated. The trade ratios are specifically determined for each new project, and they can be subject to change depending upon monitoring assessments and changes in the effectiveness of the PRF. The minimum trade ratio used in calculating credits exchanged concerning new PRFs is 2:1. Therefore, 2 pounds removed from a non-point source project can be traded for a maximum of 1 pound of credit toward point source discharges. The ratio can be adjusted up to a value of 3 on a project-specific basis. The trading ratio is initially determined through best available scientific evidence involving similar types of projects that are researched through literature reviews, and then credits are adjusted through site-specific monitoring. Initial ratios may be increased or decreased based on performance monitoring after the PRF is installed and is fully operational. There is not a fate and transport model that has been developed to assess PRF site location ratio factors for the entire basin, but trading ratios may be individually adjusted based on the location of the point source discharger in relation to the PRF and the Cherry Creek Reservoir (M. Wind, personal interview, April 11, 2007).

Performance Record

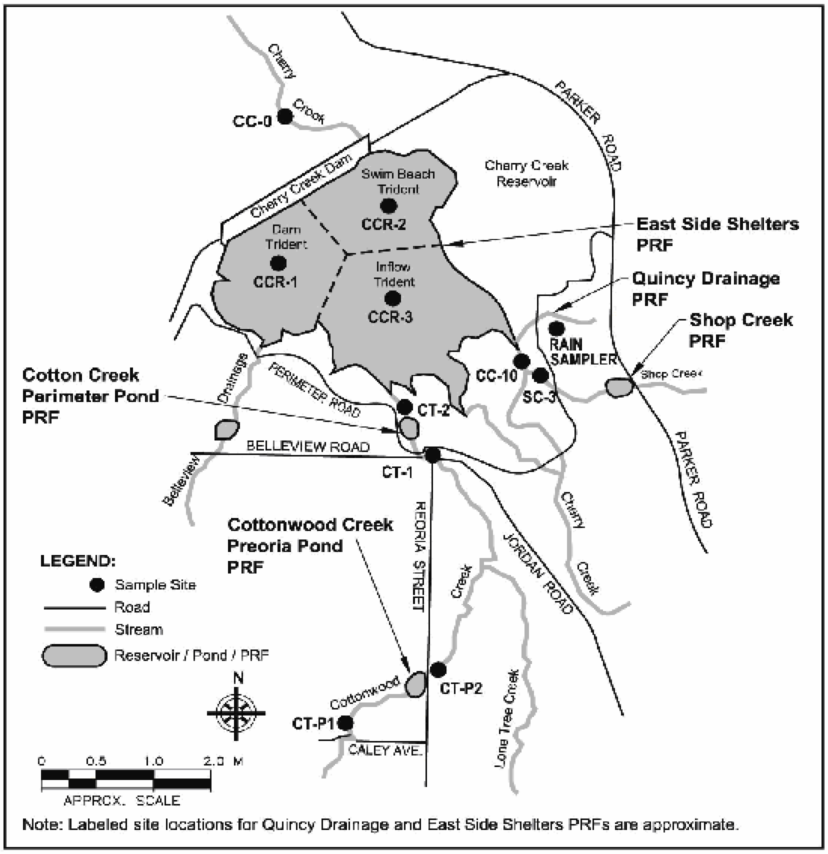
The Cherry Creek Water Quality Trading Program functions as a clearinghouse. Trading can occur if a point or non-point source in the basin implements measures to reduce its phosphorus load below the reduced levels mandated by the TMAL. The Authority either approves or denies the efficacy and efficiency of the proposed reduction, and if the reduction is approved then it becomes a credit that can be sold. The credit can then be purchased as an offset by a discharger seeking to increase an allocated phosphorus load. The program has be developed in a way that allows both trades that are either bought through the Authority or trades that are transferred from credit generator to credit user. Non-point source credits that are generated from Authority projects are placed into a Phosphorus Bank, and other basin projects created by individual non-point sources are applied to a Reserve Pool. Either the Phosphorus Bank or Reserve Pool converts non-point source reductions into available credits for point sources to purchase.

The Authority set up these two types of exchanges to provide flexibility for dischargers and maintain a level of phosphorus reduction control. Each exchange was originally worth up to 216 pounds of phosphorus per year. The Phosphorus Bank obtained its 216 pounds of phosphorus per year through four projects the Authority initiated during the 1990s and has been maintaining them since then. The Reserve Pool could earn its 216 pounds of phosphorus per year through new non-point source control projects constructed within the basin. A point source discharger had the opportunity to buy or lease up to 432 pounds of phosphorus per year of new or increased allocations. In 2004 amendments were made to Regulation #72 that removed the upper limit of 216 pounds of phosphorus per year that the Reserve Pool could achieve. These changes were implemented to encourage more interest in trading by eliminating ceilings on a potential trade (W. Ruzzo, personal interview, April 10, 2007).

The Authority retains the right to purchase phosphorus reduction credits from non-point source project owners and sell them to other dischargers who may need them to meet their allocation. From a Phosphorus Bank perspective the Authority has been constructing, maintaining, and monitoring PRFs since the early 1990s. There are currently four projects in operation that supply the Authority Phosphorus Bank with 216 credits. These PRFs include the Shop Creek detention pond and wetlands established in 1991, Cottonwood Perimeter Road Pond established in 1997, Quincy Drainage detention pond established in 1995, and improvements to the East Shade Shelter streambank established in 1995 (W. Ruzzo, personal interview, April 10, 2007). A point source discharger could purchase or lease credits from the Authority for a price set by the authority. For example, a point source discharger needing an additional 40 credits of phosphorus reduction credits, worth 116 credits with a 2.9:1 equivalence, could purchase twice that, 232 credits, from the Authority's Phosphorus Bank, which is now part of the non-point source allocation. A base price for credits purchased from the Phosphorus Bank is set by the Authority. An application to apply for credits costs $100, and an additional $500 must be provided by the discharger to cover costs incurred by the Authority to evaluate the request for credit withdrawal from the Phosphorus Bank. At the present time no discharger has requested a withdrawal from the Phosphorus Bank, but Parker Water Sanitation District has shown interest this year in leasing credits from the bank in order to retrofit wastewater treatment facilities. Once the retrofits are in place then the district will not need to lease the phosphorus credits from the Authority. This would be a temporary transaction needed in order to comply with the Parker Water Sanitation District phosphorus allocation (W. Ruzzo, personal interview, April 10, 2007).

The four historic PRFs in the basin earned their credits primarily through erosion control and wetland restoration. All four PRFs continue to reduce phosphorus loads into the Cherry Creek Reservoir to date. The Authority monitors each PRF upstream and downstream of each site to measure and compare phosphorus load reduction performance on an annual basis.

In the 1980s significant land conversion and development had significantly eroded one of the major tributaries to the Cherry Creek Reservoir, Shop Creek (See Map 5.2). The Shop Creek Water Quality Improvement Project created wetlands to stabilize channel erosion and reduce phosphorus load to the Cherry Creek Reservoir. The project entailed the establishment of a 9 acre foot detention pond upstream of five wetland channels in a series. The detention pond usually fills with water during a storm event, and then it drains slowly into the wetland areas. This slow water movement allows time for the particulates with phosphorus to settle. All five wetland channels add biological, chemical and physical treatment as well as settling time. From 1990 to 2000 phosphorus leaving the Shop Creek wetlands to enter the Cherry Creek



Map 5.2. Cherry Creek Basin with Selected PRFs Identified (M. Wind, April 11, 2007)

Reservoir averaged 261 pounds less than that entering the detention pond. This decrease represented an average of 63% reduction of phosphorus during that timeframe. The total cost of construction for the Shop Creek wetlands include the capital costs of $668,286, the annual operation and maintenance costs of $38,824, and an annualized cost at a 20 year lifespan of $72,238. The Authority has seen a decline in the productive capability of the Shop Creek wetlands to immobilize phosphorus at the same rates that it has in the past.. There is also concern over the future costs of retrieving and removal of the phosphorus that is immobilized in the sediment within the Shop Creek wetlands (R. Parachini, personal interview, April 10, 2007).

The Cottonwood Creek PRF involved both the improvements to roadways in the area and the restoration of wetland vegetation through a channel area thereby reducing phosphorus loadings. In 2004 the Authority measured an approximate reduction of about 742 pounds: 3,334 pounds before area and 2,592 pounds after. This decrease indicated an average annual load reduction of approximately 22% (Authority, 2007). On average it has been estimated that the Cottonwood Creek PRF reduces about 517 pounds per year.

Two other non-wetland PRFs implemented initial credit generators for the Phosphorus Bank. Phosphorus loads that flowed into the Quincy Drainage detention pond were reduced by restoring a vegetated infiltration basin. From 1996 to 1999 measurements indicated that average load reductions of approximately 158 pounds. The East Side Shade Shelters area shoreline was reconstructed. It had suffered from severe erosion and was stabilized through the incorporation of vegetation and gravel benching along the shoreline. Actual monitoring data does not exist for this PRF, but the Authority has estimated that approximately 18 pounds of phosphorus loadings are reduced into the Cherry Creek Reservoir per year (M. Wind, personal interview, April 11, 2007).

All four historic PRFs reduce phosphorus loads by approximately 1,000 pounds annually, and they contribute greatly to the scenic beauty and wildlife habitat of the Cherry Creek State Recreation area. In fact, wetland restoration and construction projects were chosen by the Authority because of the ancillary benefits that they provide. The Cherry Creek State Recreation area uses all four of the PRFs for educational, recreational, and preservation areas. From the initial starting points of the Cherry Creek Water Quality Trading Program the Authority envisioned the practice of restoring wetland areas and improving habitat areas for the rapidly growing counties southeast of Denver (W. Ruzzo, personal interview, April 10, 2007).

Table 5.4. Cherry Creek Basin Net Credit Costs for PRFs Constructed to Year 2000

	Pollution Reduction Facilities (PFRs)					
	Shop Creek	Cottonwood Creek	Quincy Drainage	East Side Shade Shelter	Total	Average
Capital Cost	$668,286	$342,978	$218,672	$125,759	$1,355,695	$338,924
Annual OM&M Cost	$38,824	$20,512	$13,144	$7520	$80,000	$20,000
Annualized TC 20YR PRF Life	$72,238	$37,661	$24,078	$13,808	$147,785	$36,946
Avg Annual Pounds Reduced	261	517	158	18	954	238.5
Annual Cost Per Pounds	$276.77	$72.85	$152.39	$767.11	N/A	$317.28
Net Credits Generated (Pounds/YR)	186	172	93	10	461	115.25
Annual Cost Per Credit	$388.38	$218.96	$258.90	$1380.80	N/A	$561.76

The Authority recognizes the life span changes of these four particular PRFs, and the organization continues to pursue other PRFs intended to improve the water quality. In 2002 the Authority contributed almost 17%, $118,000 of the funds needed for the Piney Creek Reclamation project that was completed in 2004. This project established riparian vegetation and soil erosion controls along 5,100 feet of shoreline to reduce approximately 90 pounds of phosphorus annually from entering the Cherry Creek Reservoir. In 2003 the Authority expanded upon the Cottonwood Creek Reclamation project to restore the natural wetlands capabilities in the area just outside the Cherry Creek State Recreational area. The wetland areas along an 11,600 feet stretch of a stream were restored. Phosphorus loadings were estimated to have been decreased by approximately 730 pounds per year. The Cottonwood Creek Reclamation will cost

$2,100,000 when it is completed with a long-term average annual cost of $330 per pound of phosphorus per year. Within the past two years the Authority has developed the The Cherry Creek State Park Wetlands Project representing a capital cost of $1,928,000 with a long-term average cost of $280 per pound phosphorus per year. The project will be phased-in to minimize impacts on the heavily used recreation area and the wetlands themselves. The project will restore approximately 60 acres of wetlands and will immobilize 600 pounds of phosphorus per year. The Authority has also been conducting several other feasibility studies to further restore, reclaim, and construct wetlands in the Cherry Creek State Park. Credits from Authority these funded projects, aside from the original four Phosphorus Bank PRFs, are not eligible for trading. The intent of these projects has not been to compete against point source controls. Instead, the Authority wishes to supplement the reductions that are being achieved through point source controls and water quality trading activities. The Authority is focused on the entire watershed and plans to continue to restore wetland areas regardless of credit production for trading. The Authority envisions its role as something larger than a trading clearinghouse, and the organization assumes the overarching lead role of improving the quality of water throughout the Cherry Creek Basin (W. Ruzzo, personal interview, April 10, 2007).

The Reserve Pool supplies credits to the Authority from non-point source projects to allow for growth and expansion. The Authority may purchase the Reserve Pool credits from various point and non-point source groups within the basin. Any entity constructing or planning a PRF may apply to the Authority for credits with assessments of how many credits will be generated. When the Authority approves the credits then that entity may then buy those credits to offset its own discharge, sell them to another discharger, or retire them. The Reserve Pool is no longer capped at 216 credits, and prospective entities may achieve however many credits an innovative approach may offer (W. Ruzzo, personal interview, April 10, 2007). Ratios are assessed by the Authority and must fall within a 2:1 to 3:1 range. These ratios are set to assure equivalence. From an additionality standpoint, only new PRFs installed to reduce phosphorus loadings may be used to generate Reserve Pool credits (W. Ruzzo, personal interview, April 10, 2007).

From a Reserve Pool perspective the Authority received and reviewed three trade project applications in 2003. Two applications were presented by Parker Water Sanitation District (PWSD), and one application was presented by the Arapahoe County Water and Wastewater Authority (ACWWA) (W. Ruzzo, personal interview, April 10, 2007). The applications that PWSD had presented included two non-point source projects that involved the use of wetlands technology to reduce phosphorus. PWSD withdrew both applications after the Authority found them problematic in the initial review. The Authority did not conclude that the plans for the restoration and construction of the two wetland sites were thorough enough to comply with guidelines, and proper post-monitoring plans were not set to be in place after construction (W. Ruzzo, personal interview, April 10, 2007).

In January of 2004 the Authority did grant ACWWA a conditional allocation of 57 pounds of phosphorus for the restored wetland area and detention pond two miles upstream from Cherry Creek Reservoir along the Lone Tree Creek. This restoration initiative was speculated to reduce 165 pounds of phosphorus per year at a trade ratio of approximately 2.9:1. ACWWA refers to this site as L-3, and construction for this site was completed in 2005 (W. Koger, personal interview, April 10, 2007). Pond L-3 was an existing wetland pond that was upgraded by adding a concrete bottom forebay for sediment removal and a micropool (M. Trujillo, personal interview, April 10, 2007). Monitoring equipment installation was incorporated in 2006 at the inflow and outflow points of the wetland area, and annual reporting will be submitted to the Authority in 2007. The allocation of 57 pounds may be increased or decreased by the Authority based on the monitoring outputs of L-3. Table 5.5 is a project cost summary (M. Trujillo, personal interview, April 10, 2007).

Table 5.5. Cherry Creek Basin Pond L-3 Project Cost Summary

ITEMIZED COSTS	DOLLARS ($)
Design	99,000
ACWWA Staff	36,000
Construction	232,000
Geotechnical Inspection	4,000
Legal/Permit/Misc	5,000
Monitoring (Estimate)	50,000
TOTAL	426,000
	$7473/pound

From ACWWA's perspective the ability to increase its phosphorus discharge allocation through non-point source credits represents a cost effective response to demographic pressures. In 2003 ACWWA was only using approximately 90% of its phosphorus wasteload allocation when it applied for the L-3 trade, but the organization anticipated the need for the credits. ACWWA further rationalized that since the organization had already achieved the .05 mg/L phosphorus discharge concentration using advanced technology the cost of upgrading its treatment facilities would far exceed that of implementing non-point source projects. Therefore, a $426,000 project that yields 57 pounds of credit (worth $456,000 at $8,000/lb) appears to be financially favorable (W. Koger, personal interview, April 10, 2007).

In the spring of 2006 ACWWA applied for additional Reserve Pool credits through the modification of two stormwater dry-detention ponds. These ponds are referred to as W-6 and W-

7. Ponds W6 and W7 are currently non-wetland ponds that were upgraded to include detention and water quality. The ponds are located in the Windmill Creek basin that is roughly 45% developed and is anticipated to have buildout flows that could establish and support a wetland in the micropool (Trujillo, personal interview, April 10, 2007).

Table 5.6. Cherry Creek Basin Pond W-6 & W-7 Project Cost Summary
(Water Quality Only)

ITEMIZED COSTS	DOLLARS ($)
Design	54,000
ACWWA Staff	23,000
Legal/Permit/Misc	5,000
Construction	220,000
Construction Services	26,000
Monitoring and Equipment (Estimate)	115,000
TOTAL	426,000
	$15,821/pound

The ACWWA Lone Tree Creek Wastewater Treatment Facility will receive 28 pounds of trade credits from this project at a 2.5:1 ratio. ACWWA is also in the process of building a new wastewater treatment facility within the next 5 years. The organization will continue to monitor and maintain there two credit sites and they are interested in developing new sites as well. Once there new treatment facility is operational they will be able to trade excess credits that they have invested in to other point source dischargers within the basin through the Authority (W. Koger, personal interview, April 10, 2007). Table 5.6 shows the project cost summary for Cheery Creek Basin Pond W-6 & W-7 for water quality only (M. Trujillo, personal interview, April 10, 2007).

Despite the progress that has been made concerning the Phosphorus Bank and the Reserve Pool the Cherry Creek Reservoir chlorophyll standard of 15 mg/L has only been met in 3 of the past 15 years, and the phosphorus goal of 40 mg/L has never been achieved in the past 15 years. However, the phosphorus load totals have been lower than the TMAL of 14,270 pounds in 14 of the past 15 years. In 2006 the chlorophyll level reached 14.7 mg/L and phosphorus was 87 mg/L. Table 5.7 lists the Cherry Creek Reservoir water quality (July-September average concentration) and total phosphorus loads from 1992 to 2006 (Annual Report, 2007). The phosphorus load total for 2006 was 6,185 pounds. Further limnological assessments in recent years have determined that approximately 4,000 pounds of phosphorus may annually accumulate in the Cherry Creek Reservoir based on the phosphorus sediments already accumulated. These accumulated amounts of phosphorus act as an internal load for which the TMAL allocations do not account for (W. Ruzzo, personal interview, April 10, 2007).

Table 5.7. Cherry Creek Reservoir Water Quality (July-Sept. Average Concentration) and Total Phosphorus Loads 1992-2006

Year	Chlorophyll a (mg/L)	Total Phosphorus (mg/L)	Total Nitrogen (mg/L)	Annual Phosphrous Load (lbs/yr)	Annual Inflow (ac-ft)	Standardized Phosphorus Load (lbs/ac-ft)	Net Phosphorus Load (lbs/yr)
1992	17.0	66	970	5,857	7,474	0.78	4,543
1993	14.4	62	826	4,110	5,905	0.70	3,399
1994	10.0	59	1,144	4,049	7,001	0.58	3,056
1995	9.4	48	913	7,972	11,781	0.68	5,923
1996	20.5	62	944	4,715	7,644	0.62	3,723
1997	22.3	96	1,120	5,761	10,362	0.56	4,765
1998	26.5	89	880	13,577	20,903	0.65	9,370
1999	28.9	81	753	17,471	27,739	0.63	7,821
2000	25.2	81	802	12,593	18,610	0.68	8,905
2001	26.1	87	757	9,837	17,250	0.57	4,995
2002	18.8	74	858	4,246	7,498	0.57	2,745
2003	25.8	90	1,121	8,568	14,929	0.57	3,590
2004	18.4	102	977	12,512	17,177	0.73	7,007
2005	17.1	116	990	10,047	18,534	0.54	6,378
2006	14.7	87	914	6,185	12,009	0.51	3,376
Mean	19.7	80.0	931	8,500	13,654	0.62	5,306
Median	18.8	81	914	7,972	12,009	0.62	4,765

(Authority Annual Report, 2007)

Although the described developments in the Cherry Creek Watershed have not resulted in immediate measurable improvements to the Cherry Creek Reservoir's water quality, the watershed management strategies implemented thus far should provide a flexible and operational structure for future water quality trading initiatives. These initiatives should reveal improvements to the water quality of the reservoir in the future. From a broad perspective the Cherry Creek Trading Program structure does meet the four criteria of an effective trading program involving non-point sources include: equivalency, additionality, accountability, and efficiency (Fang & Easter, 2003). The Authority provides equivalency through trading ratios that qualitatively account for spatial differences in loads. Additionality is recognized in the program and precludes a credit from counting towards a trade if it already existed or was required. The Authority focuses on monitoring and verification of PRFs within the basin, and these essential components provide accountability. Lastly, a point source discharger could increase its phosphorus allocation through trading more cost-effectively than through implementing its own controls.

Impediments to Incorporating Wetlands into the
Cherry Creek WQT Program

Economic and institutional impediments to water quality trading in the Cherry Creek Basin have stagnated the performance of the program, and the Cherry Creek Reservoir continues to experience high chlorophyll levels that are environmentally hazardous. At the present time only two point to non-point trades involving wetlands have been executed in the Cherry Creek Basin program to date through the Reserve Pool program, and there have been no credit exchanges through the Phosphorus Bank program. Both of these PRFs in these trades were developed and maintained by a wastewater discharger (ACWWA). The credits were allocated by the Authority, but the point to non-point trade actually occurred within the same organization. From an entire basin perspective water quality standards for phosphorus remain in violation for the Cherry Creek Reservoir. One of the major economic and institutional impediments of this case that leads to the lack of demand for credits is the readily-affordable TMAL and the fact that the cost of command-control compliance has been low. The initial allocations that were set at 14,270 pounds were set in a way as to allow for growth. As population continues to grow and regulation increases, point source wastewater dischargers may advance as an integral element of the trading program. In many cases the financial incentive that exists for point sources is that PRF implementation to gain credits is typically more cost-effective than point source controls to abide by their allocated credits. This incentive is lost when the point source already easily complies with its load allocation. From an institutional standpoint the Cherry Creek TMAL allocations were distributed with growth in mind and trading will only achieve efficiency as this growth is realized or the TMAL is lowered with redistributions of phosphorus allocations. In the summer of 2007 the Authority will be reviewing the initial allocation of 14,270 pounds, and it has been suggested that this allocation will be sliced in half (R. Parachini, personal interview, April 10, 2007; W. Ruzzo, personal interview, April 10, 2007). The procedure to reduce the phosphorus cap will be very political, and opponents to the reductions will claim that the cap has been in place since 1985. The agencies that make up the Authority share many common interests, but sometimes there are competing interests. The Authority has taken a significant amount of time to mature, and it has taken the group some time to set the proper water quality policies in conjunction with the understanding of the ecology of the Cherry Creek Watershed as a whole. The future focus of the Authority will be to increase ways of coordinating with local governments to decrease non-point phosphorus loads to the reservoir (W. Ruzzo, personal interview, April 10, 2007). The focus of the Authority is to provide structure to the trading program by allocating property rights to the various dischargers within the basin.

From an economic standpoint the Cherry Creek Basin is just now beginning to gather valued information concerning the ability for wetlands to reduce nutrient loads. The PRFs that the Authority constructed and maintains have proven to be effective immobilizers of phosphorus. Approximate reductions of phosphorus include: 42% reduction at Cottonwood-Peoria Pond, a 22% reduction at Cottonwood Perimeter Pond, a 63% reduction at Shop Creek, and over a 90% reduction at Quincy Drainage (M. Wind, personal interview, April 11, 2007). These successes have not translated into meeting the compliance goal of 40 mg/L of total phosphorus in the Cherry Creek Reservoir, but this development may emerge over time. The Authority recently discovered the problem of the internal loadings in the reservoir. From an economic informational standpoint this occurrence indicates that the TMAL may be too lenient for the water body to achieve the target of 40 mg/L of total phosphorus. The TMAL will be reassessed in the summer of 2007, and the demand for credits should increase with a reduction in the total allocation. In the beginning, point source allocations were typically large enough to preclude the need for credit purchases. These types of purchases should be more attractive as population growth demands and a smaller allocation pool encourages dischargers to purchase credits.

Economic and institutional impediments are found in this case concerning the transactions costs of monitoring and verifying credits and identifying the appropriate scientific information to assess wetland externalities. The wetlands that have been constructed or restored in the Phosphorus Bank have been effective in reducing nutrient loads on the reservoir, and extensive monitoring has taken place concerning these projects. To a lesser extent verification of phosphorus reductions from the Reserve Pool (ACWWA) have been monitored. In 2007 and early 2008 ACWWA should provide the Authority with monitoring information concerning their wetland PRFs. This information should be very helpful to the Authority in assessing future PRF credit projects. In the future the Authority should focus on more accurately accounting for fate and transport issues. Equivalency factors are compromised in the determination of conversion and trading ratios, and assessment of these factors should account for temporal dynamics and be more quantitative in structure. More sound research must go into the development of delivery ratios to better assess the dynamic nature of the ecosystem. The establishment of this type of equivalency with more certainty must be achieved without burdening the program with added transaction costs. At the present time financial incentives are not established enough, but in the long run these incentives are critical to perpetuating the program. Monitoring and financial information from the ACWWA sites should help improve Authority models, and future transactions with the Authority should also increase the overall efficiency and effectiveness of the program.

Other institutional transaction costs were cited by stakeholders that have participated in producing PRFs for water quality credits. Once the information is gained concerning the monitoring and financial aspects of creating wetland PRFs in the basin the Authority should focus on streamlining the processes for application and assessment of PRFs within the Reserve Pool. ACWWA officials claimed that the transaction process of applying for Reserve Pool credits and acquiring was quite cumbersome (W. Koger, personal interview, April 10, 2007; M. Trujillo, personal interview, April 10, 2007). The transaction costs of developing a proposal and presenting it before the Authority were great, and then ACWWA had to work with the Authority over the course of year to receive an appropriate initial trading ratio for credits that were proposed to be generated. The two credit approvals for ACWWA through the Authority were the first applications to be approved and pass through the Reserve Pool process. The process between the Authority and ACWWA was more of a slow and cumbersome phased movement through a process that had never occurred before (M. Trujillo, personal interview, April 10, 2007). The Authority believes that this process can be managed more efficiently as more trade applications are presented. The Authority plans on streamlining the application, approval/denial, and confirmation process based on what it learned through the ACWWA credit approval process (W. Ruzzo, personal interview, April 10, 2007). The Authority has worked well with the CDPHE and USEPA, but both of these agencies have not been provided the resources concerning non-point discharges to assist the Authority directly. The Authority has received grants from both entities, but expertise concerning the trading program has come from within the Authority or outside private consulting firms (J. Minter, personal interview, April 11, 2007; R. Parachini, personal interview, April 10, 2007).

From an institutional standpoint, the Cherry Creek program also faces impediments because of the fact that point sources are regulated and non-point sources are not. Currently, non-point sources face a total load allocation, and regulations do not apply to individual non-point sources. In the future, local government entities will be looked upon to better monitor and enforce non-point source PRFs in the basin as they relate to current unregulated entities. At the present time the Authority places liability for the implementation of PRFs onto the point source project owner. The focus of responsibility for nutrient reductions placed on point sources provides incentive for non-point sources to engage in trading where they otherwise may not have (Breetz, 2004). This policy aspect follows the law stipulated by the Clean Water Act, Section 402, specifying that national pollutant discharge elimination system permits do not allow liability to be transferred away from the NPDES permit holder who purchases water quality credits as a compliance measure to the individual who produces and sells the credits, although liability can be shared. The U.S. USEPA's 1996 Water Quality Trading Framework provides two options for this dilemma. The Authority chose the option to make point sources responsible for non-point source reductions. The other option would place responsibility on the state to make sure that non-point source reductions are being attained (B. Crowder, personal interview, April 11, 2007). Unfortunately, though, in the Cherry Creek Basin the incentive for the non-point sources forces a liability issue to the point sources, and this shifts the focus of the point source dischargers to retain control of their individual PRFs. Therefore, organizations like ACWWA will be more inclined to construct their own PRF projects, and organizations like the Parker Water and Sanitation District will purchase credits from the Authority through the Phosphorus Bank rather than purchasing credits from other non-regulated non-point source dischargers.

The structure of the Cherry Creek Trading Program is in place after many years of institutional development statutory directives. The Authority is committed to continuing the process of incorporating wetlands into their water quality trading program. The Authority has been unstable in past years because of political influences or because of inequitable representation by stakeholders in the basin, but the Authority has provided an institutional mechanism for focusing on the water quality problems of the Cherry Creek Reservoir. The Authority is guided by Regulation #72, and this rule is flexible enough to adapt to future developments in the dynamic watershed. The flexibility of this rule is important because of the anticipated future population growth in the area. Officials at ACWWA are pursuing trade opportunities at present because they expect to increase the treatment capacity of their facilities from the current 3 million gallons per day to over 6 million gallons per day in the next 20 to 30 years (W. Koger, personal interview, April 10, 2007). Population growth in the region will drive the demand for trades, but at the present time the TMAL allocations are set at a level easily met by all sources. The ease at which the phosphorous levels are met can be explained by observing that the overall discharge loads have been lower than the TMAL for 14 out of the past 15 years. However, the chlorophyll standard has only been met in 3 of those 15 years. Therefore, the market allocation of total phosphorus load is not appropriately set with the water quality indicator of chlorophyll a. A reduction in the total phosphorus load allocation would better represent the correlation of phosphorus to chlorophyll in the reservoir.

CHAPTER 6

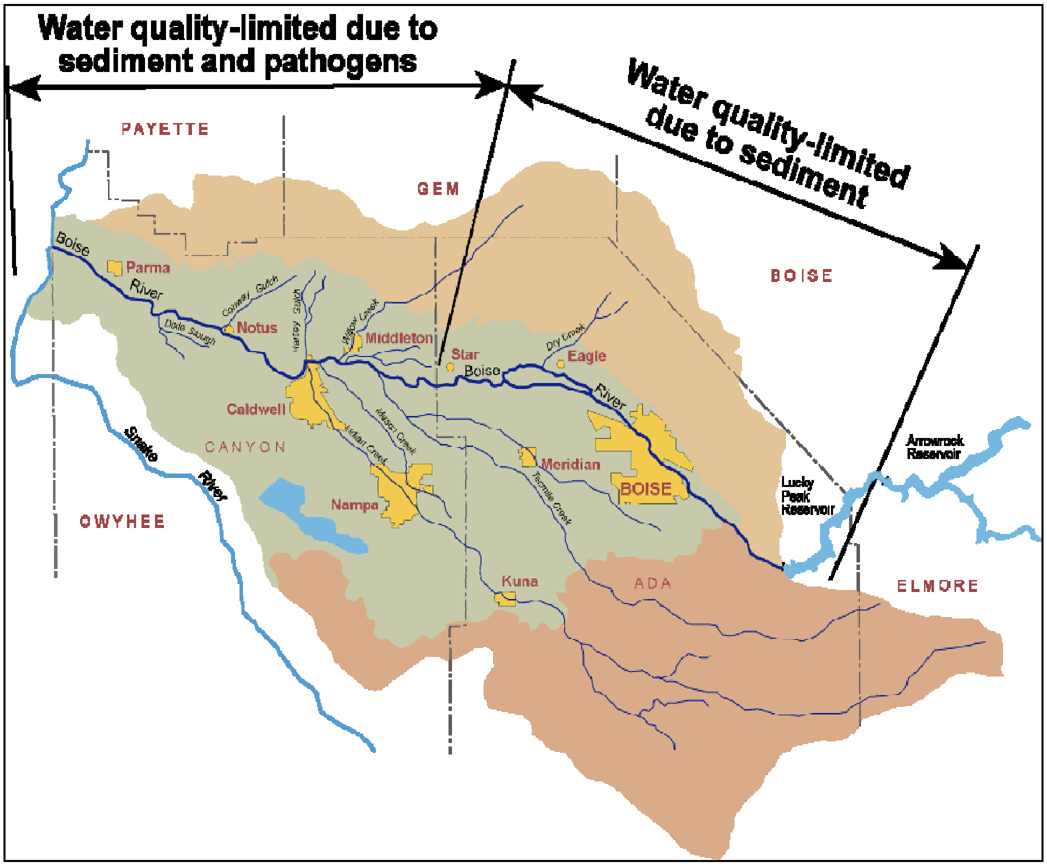
LOWER BOISE RIVER WATER QUALITY TRADING

CASE STUDY

Background

The Lower Boise River (LBR) Watershed encompasses approximately 1,290 square miles and is located in the southwestern part of Idaho. The main channel of the river flows toward the northwest for approximately 64 miles from Lucky Peak Dam to its confluence with the Snake River near Parma, Idaho. The LBR crosses through Ada County, Canyon County, and the City of Boise (Map.6.1.). Eight other cities are located within the watershed and are adjacent to the river. One-third of Idaho's population is contained within the watershed, and this area is growing rapidly (R. Finch, personal interview, April 12, 2007). 15 sub-watersheds are located within the basin, and 4 stream segments are listed on the 303(d) list for impaired waters. Nutrients, bacteria, temperature, dissolved oxygen and sediment are all listed as problems in these 4 segments. Land use within the basin is quite diverse and includes: forestry, agriculture, grazing, and urban development. The water budget for the LBR depends upon snow accumulation and melting events in the spring, and majority of the water that flows through the basin is diverted in some capacity for irrigation or other demands (R. Finch, personal interview, April 12, 2007).

In 1997 the USEPA began working with individual states in the Pacific Northwest to assess and coordinate methods for handling the implementation of Total Maximum Daily Loads (TMDLs) throughout the region. Idaho, Oregon, and Washington in the late 1990s were all faced with the considerable challenge of implementing these TMDLs based on a strict court-ordered schedule.



Map 6.1. Lower Boise River Watershed Map (IDEQ, 2007)

USEPA Region 10 worked with Idaho, Oregon, and Washington to examine how water quality trading could reduce the cost of meeting Total Maximum Daily Load (TMDL) requirements. The USEPA and the Idaho Department of Environmental Quality (IDEQ) contracted with Ross & Associates in 1997 to examine the market feasibility of water quality trading within the basin. In August of 1998 the initial trading structure and protocols were developed on two trading simulations. Over the course of the next two years Ross & Associates worked with a group of 40 to 50 stakeholders to fine tune the trading structure. Stakeholder groups included agricultural interests, stormwater interests, wastewater interests, municipal interests, and development interests. The initial grant and facilitation costs of creating the trading framework with the inclusion of the stakeholder groups were estimated to be approximately $120,000 (S. Burke, personal interview, April 12, 2007).

According to previous monitoring data and information provided by the USEPA and the IDEQ the group of stakeholders focused the trading framework around one of the nutrient pollutants of concern: phosphorus. The LBR in Idaho is highly enriched with phosphorus. Monitoring sites at cities throughout the basin show that the highest concentrations of phosphorus can be found at the mouth of the river near Parma (see Map above) (R. Finch, personal interview, April 12, 2007). In 2005 the IDEQ conducted a study and found that the phosphorus level is not currently high enough in LBR to cause algal blooms, but the study did reinforce previous assessments that indicated that the loads in the LBR contribute to the high phosphorus concentrations downstream in the Snake River (S. Burke, personal interview, April 12, 2007). From the late 1990s to date the LBR has not been visibly impaired by algae blooms or other aquatic growth, but the river is the largest contributor of phosphorus to the Brownlee Reservoir in conjunction with the Snake River (M. Bridges, personal interview, April 12, 2007). Idaho law requires that surface waters of the state cannot maintain visible slime growths or other nuisance aquatic growths that impair designated beneficial uses (Idaho Administrative Procedures Act [IDAPA] 16.01.02.200.06). The Brownlee Reservoir does suffer from excess nutrient loading, dissolved oxygen problems, and prolonged algae blooms. In September 2004 the Snake River-Hell's Canyon TMDL was issued. This TMDL presented allocations to the various tributaries that flow into Snake River. Before the Snake River-Hell's Canyon TMDL a sediment and bacteria TMDL had mandated a "no net increase" of phosphorus along the LBR as an interim measure for the phosphorus allocation until a well defined LBR phosphorus TMDL could be put into place. The Snake River-Hell's Canyon TMDL allocates a less than or equal to 0.07 mg/L phosphorus concentration level from each tributary as measured at the mouth of the tributary between May through September (M. Bridges, personal interview, April 12, 2007). The 2005 IDEQ study in the LBR watershed found that phosphorus concentrations along river increase by more than 10 times from Boise to the confluence with the Snake River. At a monitoring site in Boise a concentration level of phosphorus was measured at 0.02 mg/L, and at a monitoring site near the confluence with the Snake River a reading of 0.26 mg/L of phosphorus was registered (IDEQ, 2005). It has been suggested that the TMDL for the Snake River-Hell's Canyon area could require the LBR TMDL to reduce phosphorus loads by up to 80% (M. Bridges, personal interview, April 12, 2007). Now that the phosphorus TMDL is completed for the Sank River-Hell's Canyon area, the phosphorus allocations for the LBR TMDL will be adjusted and finalized.

Table 6.1. Lower Boise River Water Quality Trading Program Overview

Year Watershed Organization Formed	2000
Year Trading Guidelines Adopted	2001
Administrative Unit	Idaho Clean Wager Cooperative (Non-profit NGO)
Program Structure	Contractual Market
Pollutant Traded	Phosphorus

The trading program has been developed to help comply with the current policy of no net increase in total phosphorus established in the sediment and bacteria TMDL for LBR completed in 1998 and approved by USEPA in 2000 (R. Finch, personal interview, April 12, 2007) (Map 6.1). It is anticipated that the new LBR TMDL will be issued in late 2007 or early 2008 in relation to the nutrient reduction goals accounted for in the Snake River-Hell's Canyon TMDL

(B. Stewart, personal interview, April 13, 2007). The allocations for the TMDL will be based on the seasonal phosphorus loads during the irrigation months from April to October.

Administration of the Lower Boise River WQT Program

The regulatory driver for the water quality trading program is the TMDL that will be set for the Lower Boise River in the near future. Currently, the Lower Boise River Watershed Council is in the process of allocating discharge limits to stormwater districts, municipalities, and irrigation (agricultural) districts. At the monthly meeting on April 12, 2007 the council met to discuss the finalization of comments to the IDEQ and the USEPA concerning the TMDL for the LBR. Debates occurred during the meeting considering the allocations for each group. (J. Bell, personal interview, April 12, 2007). However, the group is close to finalizing the allocations to present to the USEPA by July 2007 (B. Stewart, personal interview, April 13, 2007). The Lower Boise River Watershed Council will continue to act as a advisory board to the water quality trading program, but the actual trades that will take place within the basin will be tracked and administered by the Idaho Clean Water Cooperative (ICWC).

The ICWC was formed in 2000 as a non-profit association to act as an information gathering body to relay information to the USEPA, IDEQ, and the general public concerning the trades within the basin. The ICWC is not a clearing house for trades, and it does maintain any governmental or oversight authority for water quality trading transactions. The ICWC simply acts as an entity that records and relays information concerning trades within the basin. The ICWC is responsible for tracking trading activity and maintaining a trade tracking database (R. Finch, personal interview, April 12, 2007). The ICWC sets submittal timeframes for trade notification forms and reduction credit certificates. The entity reviews trade contracts and accepts or denies them based on completeness and consistency with the trading program requirements. The ICWC tracks all trades in a central database that is transparent to regulatory agencies and the general public. The entity shows trading impacts on phosphorus allocations, and the non-profit organization also reconciles all trades in the market area to account for buyer and seller transaction balances and to ensure that credits are not used more than once. The ICWC also produces trade summary reports and provides them to point sources that require them for NPDES permit compliance (R. Finch, personal interview, April 12, 2007).

The ICWC plans on recording trade activities through an online database program that is easily accessible by federal agencies, state agencies, and other stakeholders within the watershed. The concept for creating a non-governmental group to track the trading program was generated by initial stakeholders to reduce the fears of trading partners of government intervention (R. Finch, personal interview, April 12, 2007). In the beginning the agricultural community did not want to be involved in a regulatory process concerning water quality trading, but a compromise was struck with this group to discuss water quality issues as long as water quantity issues remained a separate issue (R. Finch, personal interview, April 12, 2007). At the present time the watershed is being greatly impacted by land conversion from agriculture to suburban or urban land. Other stormwater and wastewater concerns are associated with these growth problems, but agriculture still plays a primary role in the proposed trading program (R. Finch, personal interview, April 12, 2007).

The ICWC provides transparency for water quality trading activity, but the formal rules of the trading program are governed by a Memorandum of Understanding (MOU) between the USEPA, IDEQ, and Idaho Soil Conservation Commission (ISCC). This MOU was signed on April 27, 2001, and it defines the roles of the agencies in verifying credits purchased and used by NPDES-permitted sources that choose to participate in the water quality trading program (L. Woodruff, personal interview, April 13, 2007).

There are also other regulatory drivers for the LBR water quality trading program. From a statutory standpoint, there are rules that call for "no net increase" in phosphorus for the LBR (IDAPA 16.01.02.054). These rules also specifically identify water quality trading as a tool for meeting phosphorous mitigation requirements. These rules set forth a point source allocated phosphorus reduction caps to dischargers along the LBR. Idaho has not assumed the authority from the USEPA to distribute NPDES permits. Therefore, permits are issued by USEPA Region 10. non-point sources may also be subject to load allocations for phosphorus in the near future with the implementation of the new LBR TMDL (L. Woodruff, personal interview, April 13, 2007).

Structure of the Lower Boise River Water Quality Trading Program

The LBR Effluent Trading Demonstration Project was launched in 1998 as the first effluent trading project in the Pacific Northwest (S. Burke, personal interview, April 12, 2007). The project is designed to be a start-up program for phosphorus trading in the LBR watershed in Idaho, and the USEPA and the Idaho Department of Environmental Quality (IDEQ) have invested a large amount of time and resources formatting the project so that it can possibly be a guide for similar programs in other areas of the region and throughout the country. IDEQ signed and interagency agreement with USEPA in 2000 to assume responsibility for the water quality trading project. The interagency agreement provided guidance for continued support of the program by allocating various responsibilities to several different agencies other than IDEQ including: the U.S. Bureau of Reclamation, the National Resource Conservation Service (NRCS), the Idaho Soil Conservation Commission (ISCC), the Ada Soil and Water Conservation District (ASWCD), and the Canyon Soil and Water Conservation District (CSCD). The focusing goal of the project is to create a business-like trading framework that can be implemented to help achieve the nutrient reduction goals set by CWA Section 303(d). The guidelines of the program include the requirements for parties to trade in a market-based manner, for the trades to be environmentally and legally sound, and for the parties to interact with existing regulatory programs to optimize pollution reduction initiatives.

State water quality programs in Idaho, Oregon, and Washington all supported the exploration of water quality trading programs during the latter portion of the 1990s. Considerable challenges faced by the states to develop and implement TMDLs based on a court order generated a great deal of motivation to proactive search for ways of controlling pollutant loads within watersheds (S. Burke, personal interview, April 12, 2007). Water quality trading was considered by Idaho to be a flexible and cost-effective option to meet the policy of “no net increase” in phosphorus established by the LBR sediment and bacteria TMDL.

The LBR Effluent Trading Demonstration Project has been developed through interagency collaboration to produce a trading framework that can be used for nutrient trading within the LBR watershed. During the initial phases of the development of the trading framework the stakeholders involved in the preliminary meetings held by Ross & Associates agreed upon six objectives of the program. The stakeholder group concluded that they required the program to be legally defensible and enforceable. They required that the ultimate goal of the program would be to protect water quality. The stakeholders wanted all transactions to be transparent to the general public. The group required that the program would maximize market flexibility to minimize transaction costs. The participants wanted a program that did not create other environmental problems, and they wanted the program to support active participation (M. Bridges, personal interview, April 12, 2007).

The stakeholder group that was brought together by Ross & Associates also developed a set of design principles that would guide trades and present cost-effective solutions to TMDL reduction regulations. These principles included the concepts of avoiding trade-by-trade changes to the TMDL, avoiding multiple changes depending on trades to NPDES permits, providing well defined compliance and enforcement efforts, creating environmentally equivalent reductions, and minimizing trade transaction costs through individual private contracts.

The framework that was created by the initial stakeholder group established a step-by-step process for point source to non-point source trades. Step 1 consisted of identifying possible trading partners. The trading framework does not define how buyers and sellers are to identify possible trades. point sources may be able to contact sellers directly or through a third party broker. The ISCC should play an important role in identifying potential trading opportunities through education, outreach, and the cost share programs it already manages (S. Koberg, personal interview, April 12, 2007). The second step of the trading process includes that calculation and measurement of the water quality contribution for the non-point source participant. This analysis is assessed through monitoring or assessing BMP design and performance. The third step includes the pricing negotiations and the signing of the contract between the point source and the non point source dischargers. Then, based on inspection of the BMP, the buyer signs and submits the first "Reduction Credit Certificate." The buyer and seller then sign a "Trade Notification Form" and submit the document to the ICWC. Trade information is updated by the ICWC, and the USEPA audits trades through NPDES permits (R. Finch, personal interview, April 12, 2007).

The USEPA and the IDEQ review and verify BMP implementation procedures through reports that are generated by the ISCC. The ISCC inspects BMPs installed by non-point sources in the basin to document and monitor performance. Each agency may also visit BMP sites to confirm their performance. The ultimate responsibility for ensuring the proper implementation of the BMP lies with the NPDES permit holder. The point source wastewater dischargers that hold the NPDES permits have the responsibility of inspecting the BMP's performance, and these entities also receive copies of the ISCC's inspection reports. Performance guidance and monitoring agreements can be incorporated into individual contracts. Compliance matters or enforcement actions delivered by the USEPA or the IDEQ must deal with the NPDES permit holder. The credit generator, BMP installer, does not carry any liability for the actual phosphorus reductions (M. Bridges, personal interview, April 12, 2007).

IDEQ in collaboration with the ISCC developed a list of eligible BMPs for the LBR Water Quality Trading Program (S. Burke, personal interview, April 12, 2007). The effectiveness of each BMP that was listed was assessed to be different depending upon the particular region of the state that it may be applied. The non-point source BMPs that are eligible to be traded range from crop sequencing and nutrient management techniques to sediment basins and constructed wetlands as shown in Table 6.2.

These BMPs represent the current eligible practices or implementations in order to generate credits for the LBR water quality trading program. This list was finalized in 2002, and life spans for individual BMPs are estimated based on dependability and effectiveness assessments provided for in the general literature of nutrient management. Additional BMPs may be incorporated over time or can be newly proposed by point or non-point sources (S. Burke, personal interview, April 12, 2007). From the initial meetings that took place concerning the trading programs it was decided that wetlands would be added to the initial BMP list generated by ISCC and IDEQ (IDEQ, 2001).

Table 6.2. Lower Boise River Water Quality Program Eligible BMPs for Trading

BMP	Effectiveness Discount (%)	Effectiveness Uncertainty (%)	Life Span
Polyacrylamide	95	10	1 Irrigation
Nutrient Management	N/A	N/A	1 Year
Sediment Basins	65-85	15	20 Years
Filter Strips	55	15	1 Season
Crop Sequencing	90	10	1 Season
Underground Outlet	85-95	15-25	20 Years
Straw in Furrows	Not listed	Not listed	1 Season
Microirrigation	100	2	10 Years
Tailwater Recovery	100	5	15 Years
Surge Irrigation	50	5	15 Years
Sprinkler Irrigation	100	10	15 Years
Constructed Wetland	90	5	15 Years

The BMP list that was developed by the IDEQ and the ISCC also includes procedures for generating credits. A reduction in phosphorus discharge amounts beyond the regulatory requirement of the point source must be generated by a non-point in order to create credits that can be traded in the water quality market. The reduction is calculated in reduced pounds of phosphorus by one of two methods. The reduced pound of phosphorus is then converted to water quality credits for trading purposes. The selection of the method used to analyze the amount of phosphorus that is being reduced depends on data availability. The phosphorus reduction can be monitored on-site and a specific assessment of reduction can be allocated for the credit, or the estimated average reduction of a specific BMP can be calculated based on the nutrient literature. The estimated average reduction method also incorporates a discount rate because of the potential uncertainty in the effectiveness of the BMP and other installation and maintenance factors. The on-site monitored reduction methods incorporate inflow and outflow comparisons that quantify grab samples taken during the implementation of the BMP (J. Bell, personal interview, April 12, 2007).

The estimated method of calculating the reduction of phosphorus from eligible BMPs must be discounted based on the effectiveness of the BMP and uncertainties in the effectiveness determination. These discounts are provided at the in the literature from the field, farm, and watershed scale (S. Koberg, personal interview, April 12, 2007). At the present time constructed wetlands literature lacks the sufficient data to determine efficiency or uncertainties of phosphorus reductions at the watershed scale, and the ISCC does not recommend that wetlands be used through the estimated method to generate credits. Therefore, the use of constructed wetlands in the LBR watershed requires actual monitoring assessments of phosphorus reduction to determine credits (S. Koberg, personal interview, April 12, 2007).

The LBR program has created three ways of factoring in location determinants for providing better assessments of phosphorus reductions at the watershed scale. River location ratios, delivery ratios, and site location figures have all been incorporated to address the dynamic transitions of the phosphorus concentrations in the watershed as the concentrations apply to the net impacts at Parma (the mouth of the LBR). Various types of trades within the watershed could have the potential to cause local water quality impacts in the areas where trading occurs because water diversions in the watershed are considerable. Therefore, irrigated diversions may be returned to the river at a later point that may ultimately affect the phosphorus concentration at Parma. The localized impacts on water quality are smallest when the non-point source implements a BMP upstream of the point source discharger. But, because of the diversions in the LBR certain factors were incorporated to better assess phosphorus transport and fate issues (R. Finch, personal interview, April 12, 2007).

The river location ratio factors in the influence of diversions that prevent phosphorus from reaching the LBR mouth at Parma. This ratio provides a means to determine equivalent loads between sources along the LBR (R. Finch, personal interview, April 12, 2007). Each municipality or tributary drain is assigned a ratio that is calculated and provides for diversions along the river. As the river approaches Parma, phosphorus levels increase and the value of one pound of reduction of phosphorus also increases compared to the lower value of phosphorus reduction that would be found at the head of the river at Lucky Peak Dam.

In addition to diversions transmission loss can also occur with the water body itself, but at the present time a water quality trading model for transmission of nutrients does not exist. Therefore, the LBR trading program has incorporated a linear calculation that represents the transmission loss along the river. This delivery ratio is calculated by subtracting the distance in miles to the mouth of river from the BMP's point of discharge by 100. Then the figure is divided by 100. This figure is not scientifically sound based on phosphorus transmission in the river, but it does provide some type of delivery ratio to weigh the possible outcomes of a properly constructed BMP (R. Finch, personal interview, April 12, 2007).

Site location factors were also included because of the transmission loss that may occur between the BMP location where the phosphorus reduction takes place and the location of the point source discharge to a river. Three site location factors were developed to account for this transmission loss. A site factor of 0.6 was incorporated when land runoff flows into a canal that is likely to be reused by a party downstream. A site factor of 0.8 was instituted for land runoff that flows through or around other fields before it flows into a canal or drain. A site factor of 1.0 was decided upon when land runoff flows directly to a drain or stream through a ditch or washbasin (R. Finch, personal interview, April 12, 2007).

The calculation of a credit depends on the different location and delivery factors mentioned, but the determination of a credit in the LBR trading program begins with the assessment of the amount of phosphorus produced at a given location. The ISCC estimates the current phosphorus loads with the Surface Irrigation Soil Loss (SISL) tool. According to the USDA this tool is currently the most accurate and simple method to estimate soil loss from surface-irrigated croplands. The SISL is used to calculate the number of tons of lost soil that can be assessed in a given irrigation season. The trading program has used baseline information concerning phosphorus loading from 1996 as recommended by the ISCC (IDEQ, 2001). The 1996 baseline was also used for the original TMDL for sediment, and therefore, phosphorus reductions can be compared to phosphorus loads from 1996 (S. Koberg, personal interview, April 12, 2007).

In most cases phosphorus discharges are higher during the beginning of the irrigation season from April through October because there is more erosion from rain events and less uptake by crop plants. The phosphorus reduction calculation is a culmination the effectiveness of the BMP selected. Then an uncertainty factor is subtracted, multiplied by the river location ratio, the delivery ration, and the site location factor. The final calculation is referred to as a "Parma Pound" (S. Burke, personal interview, April 12, 2007; R. Finch, personal interview, April 12, 2007). The concept of the "Parma Pound" explains the fact that not all phosphorus pounds are equal in the watershed because of diversions and water reuse in the basin. The "Parma Pounds" are only allocated during the months of the irrigation season to reflect the phosphorus load variability over the season. This irrigation season coincides with the seasonal TMDL reduction requirements.

In order to produce "Parma Pounds" agricultural non-point sources are encouraged to work with either the Ada Soil and Water Conservation District (ASWCD) or the Canyon Soil

Conservation District (CSCD). Farmer can develop a conservation plan with these entities in cooperation with NRCS (USDA) and the ISCC. BMPs are designed as part of these conservation plans to address water quality concerns. The BMPs can be certified and installed according to NRCS and participate in cost-share programs. Once the BMP is operational it can begin generating phosphorus reduction credits. Within the LBR watershed the BMPs will only operate to reduce phosphorus during the irrigation season, and credits can only be made available during this time. Fully functioning and certified BMPs must be inspected prior to their seasonal operation. Inspections can also take place at any time during the lifetime of the specific BMP (S. Burke, personal interview, April 12, 2007).

Performance Record

The LBR Effluent Trading Demonstration Project set up a framework for trading pollutant discharges among sources. The framework is a bilateral market structure that allows trades to take place between point and non point dischargers. Buyers and sellers are expected to agree upon individual contracts for credit delivery. Credits are generated and used on a monthly basis during the irrigation season. Non point source credits are generated in given month, and point sources must use those credits to offset nutrient loads within the same month.

Elements of a trading process were developed by an initial group of stakeholders facilitated by Ross & Associates. Key elements of the initial trading drafts included the types of permit conditions, necessary forms, agencies' roles, and generation of credits. At the present time no trades have occurred. The main reason for the lack of trades deals with the delays in finalizing the LBR phosphorus TMDL. The politically influenced allocation of phosphorus loads to various stakeholder groups has taken over seven years to implement, and there has been a large amount of transaction costs that have been incurred in the process (R. Finch, personal interview, April 12, 2007). The expected value of phosphorus in the basin looks promising for a trading scenario because the stringent target set by the Snake River-Hell's Canyon phosphorus TMDL will be difficult to meet without point sources seeking trades. The cost to reduce phosphorus loads will continue to increase as the final portion of the phosphorus reduction allocations are realized, and trading will become a more viable option (S. Burke, personal interview, April 12, 2007). Once the regulatory driver is in place the structure for trading should be able to help to direct and promote trades within the basin.

The structure of the trading program was developed in a way that should minimize administrative and governmental transaction costs. Administrative costs should be low because the responsibility of finding trading opportunities is on the point sources. The regulatory focus has been on defining the trading conditions instead of evaluating or brokering trades (R. Finch, personal interview, April 12, 2007). Transaction costs were high initially during the beginning phases of the program development, but future transaction costs could vary to a large extent based on the mechanisms that arise to identify trades, communicate with trading partners, and negotiate trade contracts.

Other cost savings of the program are directly associated with non-point source BMP technology implementation versus point source technology implementation. Municipalities and other point sources the LBR watershed estimated in 1999 that it would cost between $12 and $178 to reduce one pound of phosphorus. Agriculture and stormwater stakeholders anticipated that it would cost between $2 and $20 to reduce one pound of phosphorus for non-point sources. Both the point source and the non-point source reduction estimates were based on 80% phosphorus reduction levels. Therefore, it has been estimated that there could be a potential cost savings of $10 to $158 per pound of phosphorus reduced within the LBR watershed if point sources and non-point sources are able to cooperate through individual contracts (Environomics, 1999).

In 2000 the LBR Effluent Trading Demonstration Project conducted a trading simulation for a point source to non point trading scenario. Two eligible BMPs, constructed wetland and sediment basin, were installed in a sequence for the simulation. The actual process included a concept design of the two eligible BMPs, a sample permit, documentation, cost estimates, and monitoring evaluation techniques (Ross & Associates, 2000).

The design of the BMPs entailed a system of running water through a series of drainage areas through the constructed wetland and the sediment basin to mimic the continuous agricultural runoff that is present during the irrigation season. The design of these BMPs took into account the maintenance requirements for each system. The wetland was designed to provide for the accumulation of biomass by setting a target depth. The sediment basin was also designed to store up to six years of sediment at a depth of 2 feet. Both systems were designed to function with minimal flows through the operation of control gates to keep plants alive and minimize decay. The systems also had inflow and outflow monitoring sites to measure phosphorus concentrations and flows.

The efficiency of constructed wetlands in removing phosphorus depends on the design, maintenance, and flow rate of water through a designated area. Similar engineering constraints can be identified for sediment basins as well. The constructed wetland and sediment basin removed phosphorus at different rates using a flow rate of 15 cubic feet per second (cfs) and a concentration of 0.366 mg/L of phosphorus. It was estimated that over thirty year life span the sediment basin could remove up to 1,040 pounds of total phosphorus per irrigation season. The constructed wetland was estimated to remove approximately 980 pounds of total phosphorus per season. The combined total of these two BMPs if used in conjunction would equal a reduction of approximately 2,020 pounds of total phosphorus per season. This would equal approximately 60,600 pounds over 30 years (Ross & Associates, 2007). This estimate could fluctuate over that period of time depending upon hydrologic conditions, climate change, or maintenance and upkeep factors.

Probable costs for the design of the systems were also calculated in the simulation. A public bid process helped to determine the cost estimate that included equipment, materials, and labor in year 2000 dollars. The estimate of cost was initial investment capital that included engineering, construction, and contingency. Also assessed in this calculation was land acquisition set at $10,000 per acre. Capital and operation and maintenance were estimated at $3,004,000 and $145,800 respectively for a 54 acre construction site as can be seen in Table 6.3. The operation and maintenance cost was composed of $71,800 for annual upkeep and $74,000 for harvesting wetlands plants every five years. These costs with the inclusion of a 3% inflation rate set an annualized cost for removal of phosphorus at $118 per pound. An average cost for constructing wetland systems for treating stormwater has been estimated at $10,000 to $30,000 per acre (Reed, 1991). This simulation for the LBR watershed estimated that the cost to construct this type of wetland structure would be approximately $67,000 per acre. In order to justify the high cost of constructing this type of BMP the value of a "Parma Pound" will need to be high in order to balance the cost with the value of implementing the BMP.

Stringent TMDL standards may make the option of constructing a type of wetland system like this a more economically feasible option, but will the market arrangements of the LBR trading program drive other types of more cost effective BMPs? point source stakeholders in the basin are mainly focused on the least cost option for phosphorus reduction allocations, and they are presently not inclined to choose a more costly BMP in order to receive reduced phosphorus levels (D. Keil, personal interview, April 12, 2007).

Table 6.3. Lower Boise River Constructed Wetland Simulation Cost Summary

Simulation Feature	Quantity
Amount of Wetlands	54 acres
Life Span	30 years
Flow Rate	15 cfs
Effluent concentration	0.366 mg/L
Capital Cost	#3,004,000
O&M Cost	$145,800
TP Removed by the Wetlands per Irrigation Season	980 lbs
TP Removed per Irrigation Season	2,020 lbs
TP Removed per Life Span	60,600 lbs
Annualized Cost per Pound of TP Removed	$118

(Ross & Associates, 2007)

Impediments to Incorporating Wetlands into the LBR WQT Program

The LBR Water Quality Trading program suffers from various economic and institutional impediments. The LBR Effluent Trading Demonstration Project has been successful in developing a trading framework, but at the present time no trades have occurred. From an economic standpoint, the USEPA and the IDEQ as well as other agencies spent several years developing and formatting the trading structure, but the setting of the cap (TMDL) has not occurred. The lack of a trade driving cap (TMDL) that is enforceable disallows the program from functioning at any level.

Secondly, property rights within the basin have not been assigned because a cap (TMDL) has not been set. The LBR Effluent Trading Demonstration Project did incur high expenses and intensive use of resources to develop the trading framework, but the property rights to discharge pollutants have not been assigned. Throughout the initial phases of the program the irrigation districts and farmers in the watershed were skeptical about losing water rights by participating in a program. As the literature indicated, this case reveals that non-point sources have been concerned that their participation in generating credits by reducing phosphorus loads might encourage or facilitate their being subjected to increased regulations (King, 2005). This political apprehension could account for some of the politically influenced delay of the LBR TMDL. From another perspective, public comments by environmental interest groups within the watershed initially expressed concerns about the ability of point sources to account for trades. These groups felt that the program necessitated a trade-by-trade regulatory approval process (M. Bridges, personal interview, April 12, 2007). The stakeholders that participate in the trading program at the present are pleased that the LBR framework has established a highly effective and locally tailored solution under the umbrella the Clean Water Act (CWA) and Idaho statutory guidance to help resolve the phosphorus problems in the LBR watershed (R. Finch, personal interview, April 12, 2007).

From an economic information standpoint, the LBR Effluent Trading Demonstration Project has provided a list of BMPs that could prove to be practical and possibly cost-effective to implement within the watershed. There was much time spent in calculating the estimations for assessing a credit within the watershed. The spatial considerations that were accounted for concerning the irrigated and diverted nature of the watershed were thoroughly analyzed. The location ratios, delivery ratios, and site specific ratios all were developed to minimize localized areas with high level of pollution in a watershed. The background research for support concerning the discounts developed to generate credits is incomplete. Transmission losses and uptake capacity between the trading partners need greater study to refine discounts. Another factor of the program that should be further researched concerns the issue of equivalency. Point sources within the basin discharge at a fairly constant rate over a twelve month period of time, but irrigation during the earlier set portion of a given year produces more phosphorus through erosion and less uptake by crops. Equivalency, in this situation, can be difficult to demonstrate or calculate because of the dynamic fluctuations of phosphorus generated within a given timeframe.

From a monitoring and verification standpoint of the assessment of wetland externalities, the program has witnessed a simulation using a constructed wetland model that could be potentially duplicated in the basin. The estimated 2,020 pounds of reduced phosphorus could be converted to “Parma Pounds” based on the location of the site within the watershed in relation to the point source that has a contract for the credits the BMP is providing. The case study confirmed the assessment of the literature that there is not enough consistent information concerning the applicability for wetlands to reduce nutrients at the watershed scale. At the present time the ISCC does not maintain enough data to determine efficiency or uncertainties involved with constructed wetlands technology implemented to reduce phosphorus (M. Bridges, personal interview, April 12, 2007). Therefore, if a constructed wetland were to be implemented as a non-point source credit producer then phosphorus reductions must be measured on site through inflow and outflow structures. Constructed wetlands within the basin could prove to be as effective as the simulation or more effective at removing phosphorus based on their designs and maintenance. But, at the present time there is not any interest from stakeholders within the basin to certify wetlands as the BMP most favorable to be implemented. In fact, the current political mindset of the basin focuses on reducing the phosphorus loads through the least possible cost methods. Therefore, if some other BMP can effectively decrease phosphorus loads at a lower cost than wetlands can, then the market will drive the demand for that type of BMP (R. Finch, personal interview, April 12, 2007)

From an economic transaction costs standpoint, the durability and return on investment on BMP structures are also concerns of stakeholders in the watershed (R. Finch, personal interview, April 12, 2007). In the LBR Effluent Trading Demonstration Project the life span assigned to BMPs reflected the professional judgments of scientists, regulators, and the most current literature. Constructed wetlands were originally assigned a 5-year life span, but the lifespan for a constructed wetland in the LBR BMP list is now assigned a 15 year limit based on a technical focus group decision among participants during the early development of the water quality trading program (S. Koberg, personal interview, April 12, 2007). However, the simulation that was managed in 2000 by Ross & Associates used a 30 year BMP lifespan. It is most cost effective to use a constructed wetland for as long as they are functional because of the high investment costs incurred to build them. Further research should focus on life span factors of certain types of BMPs versus constructed wetlands to better assess cost of phosphorus reduction in the watershed in the long-term. When dealing with the long-term effects and lifespans of BMPs it should also be noted that intensive planning should take place when analyzing the long-term fate of the phosphorus that is removed using BMPs such as constructed wetlands or sediment basins. At some point in time the phosphorus must be harvested through the removal of vegetation and sediment. The phosphorus that remains in this harvested material must then be moved or disposed of in a fashion that does not lead to further environmental problems in other locations.

From an institutional standpoint, the politically influenced processes of establishing allocations for the LBR TMDL have delayed any regulatory enforcement criteria that may generate needs for trading. There must be a driver for a water quality trading program to exist. If there is not a cap set for the amount of a type of discharge then there is not any need to seek reduction credits. As the literature has stated, drivers include a regulatory requirement or some other standard that allocates the maximum amount of a pollutant that can be discharged into a watershed. A driver could be derived from a non-point sources' ability to reduce pollutants most cost-effectively that certain point sources. It is very important that a trading program educates potential stakeholders that will be involved. From an institutional standpoint, the LBR Effluent Trading Demonstration Project did a good initial job of including a vast number of stakeholders during the beginning phases of development. Trading programs can be hampered by the lack of an established or known trading framework. Additionally, from a transaction costs standpoint, trading would fail to be effective if it is viewed as being too cumbersome for traders to use or regulators to evaluate.

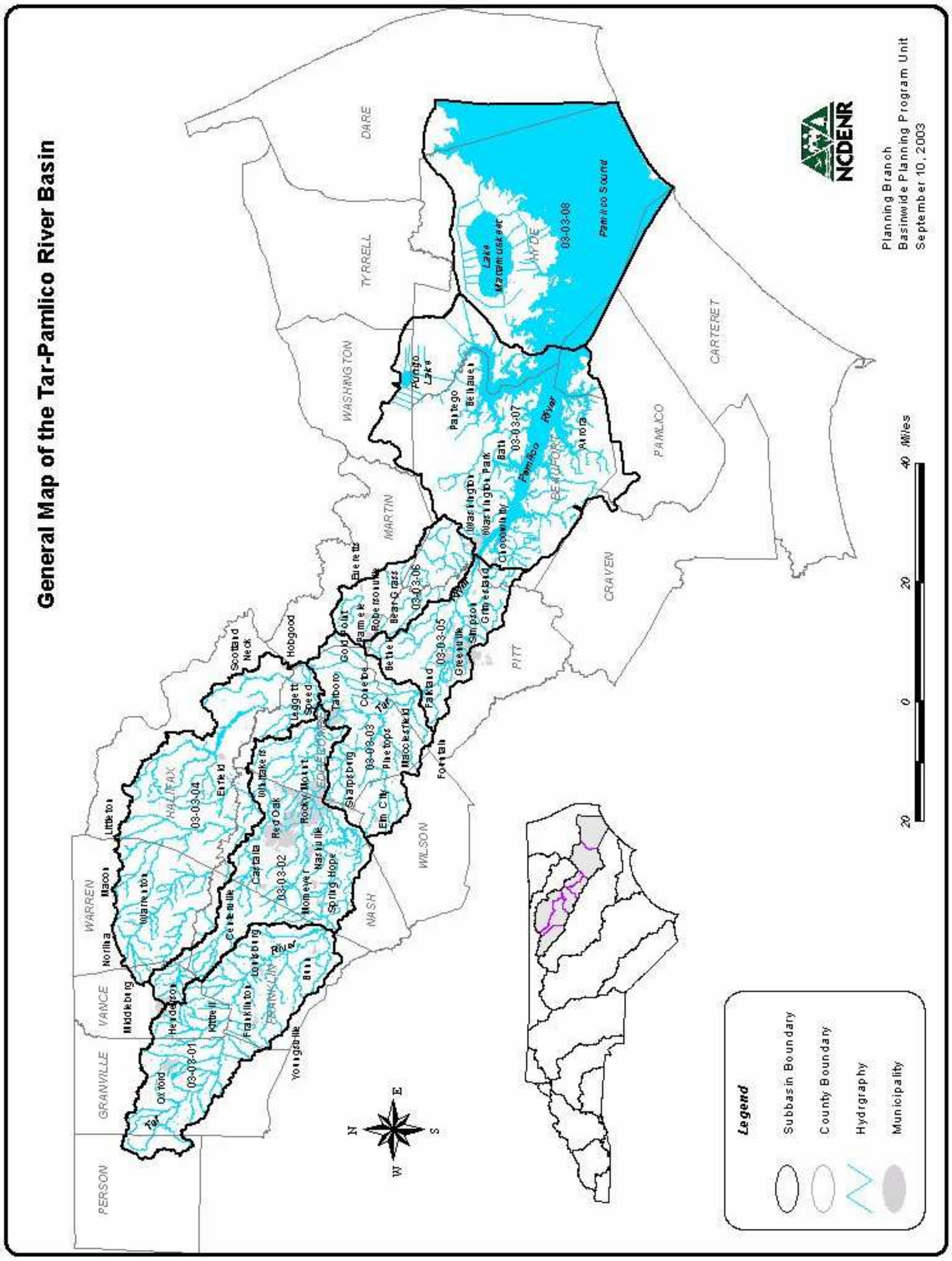
The growth in population within the LBR has been consistently rising in recent years, and this strain has placed a great deal of pressure on point sources and non-point sources to minimize phosphorus discharge increases into the watershed. At the present time the LBR does play a large role in the dissolved oxygen problems that face the Brownlee Reservoir. The LBR is the major contributor of phosphorus to the Brownlee Reservoir and the water quality along the LBR must improve in order to prevent future environmental hazards within the watershed. At the present time economic and institutional impediments hinder the operation of the trading market. There is an institutional structure for the trading program that is becoming more viable politically as TMDL allocations are distributed through the basin and more pressure is placed on point sources to reduce phosphorus discharges, but property rights for pollution discharge have not been assigned. Non-point sources in the basin seem wary that their participation in the program may encourage the future subjection to regulations requiring load reductions. The future of the program depends on the optimal setting, maintenance, and enforcement of the TMDL. Without this trade driver then all of the other efforts of the program to reduce nutrient loads will not be realized through this type of trading structure.

CHAPTER 7

TAR-PAMLICO WATER QUALITY TRADING CASE STUDY

Background

The Tar-Pamlico River Basin extends approximately 180 miles from the Piedmont of North Carolina through the Coastal Plain to Pamlico Sound. The Tar River combines flows from approximately 2,300 miles of freshwater streams before it enters the Pamlico River near Washington, North Carolina. The entire basin covers approximately 5,400 square miles that encompasses portions of 17 counties. The Tar-Pamlico River Basin includes the cities of Greenville, Rocky Mount, and Tarboro. Agriculture and forest are the main land uses in the basin. Five of North Carolina's 10 leading hog-producing counties and the leading chicken producing county all reside within the basin. As of 1989 there were approximately 875 hog, chicken, and dairy operations in the basin (Harding, 1990). The increasing population growth, intensive agricultural practices, and expanding livestock operations have placed significant environmental pressures on the Tar-Pamlico Basin over the past thirty years. In the mid-1980's the Pamlico River Estuary saw an increase in harmful algae blooms, low oxygen levels, increased number of fish kills, and other stressful symptoms in the aquatic biota. In 1986 10% of all chlorophyll-a samples taken in the Pamlico Estuary exceeded the state standard of 40 g/L (Steel, 1991).



Map 7.1. Tar-Pamlico River Basin

Following a severe record-setting fish kill in the Pamlico Sound in 1989 the North Carolina Environmental Management Commission (NCEMC) designated the Tar-Pamlico Watershed as nutrient sensitive waters. That same year the North Carolina Department of Environment and Natural Resources (NCDENR) developed an initial management strategy that focused on reducing nutrient loads from point source dischargers.

Initially, the NCEMC proposed to reduce nutrient loads upon the Pamlico Sound by simply setting technology-based nutrient concentration limits on point sources. These regulations were to be incorporated over the course of several years. In response to these actions the point source dischargers in the basin formed the Tar-Pamlico Basin Association (Association). This group along with the Environmental Defense Fund and the Pamlico Tar River Foundation proposed a collective nutrient trading program to the state of North Carolina. By 1990 the state, the Association, and the two environmental groups had signed an agreement marking the initial phase of the Tar-Pamlico Water Quality Trading Program. The agreement stipulated that the Association would reduce their phosphorus and nitrogen loads to the estuary as a group by setting a loading cap. If the Association exceeded this cap then point sources involved agreed to fund agricultural Best Management Practices (BMPs) through the North Carolina Agricultural Cost Share Program.

The initial agreement between the state of North Carolina, the Association, and the two environmental groups allowed the point sources within the Association to find more cost-effective ways to collectively meet their loading cap by allowing those facilities that were more capable of removing nutrients to do so within the group framework. At the time of the initial agreement economic models had documented that payments for BMPs were a more cost effective way of reducing nutrients as compared to retrofits or treatment modifications implemented during expansion (J. Huisman, personal interview, April 24, 2007).

Table 7.1. Tar-Pamlico Water Quality Trading Program Overview

Year Watershed Organization Formed	1989
Year Trading Guidelines Adopted	1992
Administrative Unit	NC Dept. of Water Quality/ Basin Association/ Agricultural Cost Share Program
Program Structure	In-Lieu Fee/Exceedance Tax
Pollutant Traded	Nitrogen & Phosphorus

Phase I of the Tar-Pamlico Water Quality Trading Program began with the adoption of the initial agreement between the Association, environmental groups, and the state of North Carolina. The parties involved in the initial agreement began the process of implementing the trading program by developing trading rules in 1992. In 1991 the Association hired an engineering firm to estimate nutrient reduction measures and costs at individual discharge locations throughout the basin. The Association also hired consultants to help develop an estuarine water quality model to help determine an in-stream nutrient reduction target goal. A target was set for nitrogen and phosphorus reductions was developed with the help of the estuarine modeling initiative, and the nutrient discharge cap was approved in 1995 (Environomics, 1999). During Phase I the Association approved a model that predicted that a 45% total reduction from point sources and non-point sources would be necessary to meet in-stream water quality goals (J. Huisman, personal interview, April 24, 2007).

Phase II of the Tar-Pamlico Water Quality Trading Program began in 1995, and during this phase the focus of nutrient reduction goals centered on non-point sources. The estuary model that was designed during Phase I was implemented at this time to set a 30% reduction in nitrogen and a no net increase of phosphorus loading for all sources to the estuary from 1991 conditions. The parties that came together for the Phase II agreement include the Association, NCDWQ, and the North Carolina Division of Soil and Water Conservation (NCDSWC). The environmental organizations Environmental Defense and Pamlico-Tar River Foundation (PTRF) were participants in the Phase I agreement, but these groups opted out of the Phase II agreements because they were not satisfied with the 30% reduction goal (M. Templeton, personal interview, April 25, 2007). The Phase II strategy provided technical information that suggested that the original 45% reduction in nitrogen loading would be infeasible given the limitations of point source and non-point source treatment technologies and BMP effectiveness. It was recognized in Phase II that there was some model error and uncertainty in the initial predictions confirmed during Phase I.

Phase III of the program was developed and agreed upon in 2005. The third phase continues the structure established in Phase II. Phase III rules were developed to improve and refine the program by updating the Association membership, defining temporal water quality issues within the basin, and revisiting alternate offset options. Better monitoring, modeling, and documentation strategies have improved the knowledge of nitrogen and phosphorus fate to the Pamlico Sound. The Environmental Defense Fund and the Pamlico Tar River Foundation returned as parties to the Phase III agreement. These organizations want to continue to work cooperatively with other parties in order to ensure the protection of the estuary (J. Rudek, personal interview, April 25, 2007).

Administration of the Tar-Pamlico River WQT Program

The initial partners involved in the effort to create the Tar-Pamlico Water Quality Trading Strategy were the NCDWQ, the NCDSWC, the North Carolina Farm Bureau, North Carolina Department of Agriculture, North Carolina State University, and the Association. The initial group that formed the Association consisted of fourteen dischargers that equaled approximately 90% of all point source discharges to the river (J. Huisman, personal interview, April 24, 2007). During the initial phase of development the NCEMC brought together stakeholder groups of affected parties throughout the basin and provided the participants with a chance to express differing viewpoints. Stakeholders involved in the process included environmental groups, municipalities, developers, businesses, and the general public (Husiman, 2007). The Association evolved from these initial stakeholder meetings and the organization is now made up of

The initial agreement signed by the Association, NCEMC, NCDWQ, and Soil and Water is the primary mechanism used to provide structure and assure accountability for the water quality trading program. The NPDES permits, administered by the NCDWQ, provided to the parties in the Association do not contain limits for nitrogen. Therefore, the parties in the Association are not required by the CWA to readjust their permit limits if the members show they could meet more stringent requirements. This type of regulatory requirement to meet more stringent requirements could penalize economic incentives to perform at a higher level of efficiency to decrease nutrient discharges as a collective body. The NPDES permits contain a clause that allows the NCDWQ to revise new discharge limits to individual members or the Association as a whole if conditions in the agreement are violated (J. Huisman, personal interview, April 24, 2007).

During the initial phase of the program it was estimated that to meet water quality standards concerning nutrients it would cost point sources in the basin between $50 and 100 million in capital costs for technology upgrades. Instead of investing the totality of that amount of funding to technology upgrades the Association has been instrumental in providing the financial support for the water quality trading program (Breetz, 2004) During the initial two years of the program the Association contributed $150,000 to fund additional NCDSWC personnel to assist in BMP review and identification. These funds were necessary in order to design and establish the trading system. The Association also provided additional initial funding of almost $1 million earmarked for agricultural BMPs. (J. Huisman, personal interview, April 24, 2007). This funding was acquired in large part through a USEPA grant. The Association has been able to bank the credits that were generated from some of this money toward future cap exceedances (Templeton, 2007).

The Association operates as an advisory board to the state of North Carolina and the group also coordinates actions of the trading program with point source members, agricultural representatives, and environmental group representatives. The Association members include: Belhaven, Bunn, Enfield, Franklin Water & Sewer Authority, Greenville Utilities, Louisburg, Oxford, Pinetops, Robersonville, Rocky Mount, Scotland Neck, Spring Hope, Tarboro, Warrenton, and Washington (Tar-Pamlico Association Meeting, 2007).

The NCEMC, NCDWQ, and the NCDSWC are the key administrative bodies for the Tar-Pamlico Water Quality Trading Program. These government agencies coordinate with the Association and the trading program participants, but each of these agencies retain the ability to take enforcement actions against point sources and non-point sources in the event that they are not able to demonstrate compliance.

Structure of the Tar-Pamlico Water Quality Trading Program

The Tar-Pamlico Water Quality Trading Program was not created to allow individual trades between point sources and non-point sources to take place. The point source dischargers pay an offset fee for each mass unit of pollutant by which as a group they exceed each year. The funds that are collected are used directly by the voluntary agricultural cost share program managed by the NCDSWC to implement BMPs. The cost share program pays farmers up to 75% of the cost of installing nutrient reducing BMPs on farms that are located within the basin.

The trading model for this program can best be described as a group cap-and-trade program with an exceedance tax. point sources are assigned individual baseline maximum nutrient loads and nutrient reduction goals. These allocations that are set by the Association in accordance with the NCDWQ set the overall nutrient loading goals for the water body. The trading program was designed specifically to target agricultural non-point source nutrient loads. Offset credits from point sources are paid directly to the cost share program and are targeted geographically for the most cost-effective nutrient reductions to the estuary. When the respective point sources in the Association have purchased credits they are no longer liable for ensuring that non-point source BMPs are installed and operating effectively. The state of North Carolina assumes responsibility for the monitoring and verification of BMPs within the basin. The NCDSWC inspects local soil and water conservation districts after every five years to maintain that the local districts are inspecting at least 5% of the contracts it allows into the program each year. non-point sources that are found to be in noncompliance must return the cost share funds to the state. The primary focus of the cost share program is to provide farmers with assistance implementing agricultural BMPs aimed at reducing nutrients loads on the estuary.

The NCDWQ maintains authority over nutrient tradeoffs and allocations within the program (Breetz, 2004). An agreed upon cap was assigned to the Association for combined discharges in 1990 during the initial phase. The cap required a 44,000 pound per year reduction in total nitrogen and phosphorus over five years (Kerr, 2000). The Association was also required to: perform an optimization study for capital improvements to point source discharge facilities in the basin; develop an estuarine model for nutrient loading; fund the initial design and administration of the water quality trading program ($150,000 was provided by the Association); make minimum payments into the offset fund for future non-point source BMPs even if cap was not exceeded; and perform water quality monitoring to document compliance with the cap (Breetz, 2004).

Non-point source credits are purchased by the point sources at a fixed price. The price considers the BMP life expectancy, area affected, farmers' capital cost, maintenance costs, and BMP effectiveness (McCarthy, 1996). BMP values were estimated based on literature reviews that included empirical studies of conservation tillage, buffer strip, and terracing in the Chesapeake Bay (J. Huisman, personal interview, April 24, 2007). Structural BMPs have a credit life of 10 years in the program, and non-structural BMPs have a credit life of 3 years.

The Tar-Pamlico Water Quality Trading Program established a fixed fee per pound of total nitrogen. The standard trading ratio for the program is 2:1, and then there is a 10% cost differential set to absorb administrative costs. The effective trading ratio is set at 2.1:1. The fixed fee charged to point source dischargers that represented the 2.1:1 ratio was initially set at $25.40 per pound. The credits expire after 10 years after the funds have been allocated by the NCDSWC. The cost share program that is used as a tool to fund BMPs was a pre-existing program throughout the state funded by the USDA's NRCS. The Tar-Pamlico Water Quality Trading Program funds 75% of the capital costs associated with voluntary implementation of agricultural BMPs on farmland in the basin.

In 1986 a nutrient source budget was prepared for the Tar-Pamlico basin, and this budget was revised in 1988 to reflect dynamic changes in the watershed. In 1989 the NCDWQ projected a 30.55 million gallons a day flow for all the Association members by 1994. The NCDWQ estimated that total nutrient loading in 1994 would reach 1,278,000 pounds per year based on trends during the late 1980s. The original agreement required mandatory phosphorus and nitrogen limits for point sources to decrease by 1994 to an estimated 936,965 pounds per year. Therefore, the NCDWQ, the Association, NCEDF, and the PTRF together established

440,924 per year as the reduction goal for Phase I of the water quality trading program. 396,832 pounds per year were allocated for nitrogen reduction and 44,092 pounds per year were allocated for phosphorus (J. Huisman, personal interview, April 24, 2007).

A hydrodynamic model was incorporated during the initial phase of the program to predict the impacts of nutrient loading in the estuary. The model focused on the basin area from Greenville to Pamlico Point. This distance between the two sites covered approximately 60 miles. The calibration year for the model was 1991. This year was chosen because it was set as the baseline year when point sources in the Association were required to perform nutrient monitoring (Templeton, 2007). A critical portion of the river, near the town of Washington, was chosen as the point where management strategies would be evaluated. This segment of the river revealed the highest level of chlorophyll a and dissolved oxygen violations in the late 1980s (Templeton, 2007). There were plans to recalibrate the model to lower nutrient loading conditions after the 30% nitrogen reductions were achieved in Phase II. This proposal was suggested to better determine additional reductions that may be necessary in order to improve the water quality of the estuary. The recalibration of the model has been postponed pending the results of other estuary evaluations (Templeton, 2007).

During Phase II of the program, 1995 through 2004, the focus on improving water quality shifted to include non-point sources based on the recognition that these contribute the majority of nutrient loading to the watershed. The modeling completed by the Association in Phase I estimated that non-point sources accounted for 92 percent of the nutrient loads (Templeton, 2007). Target reductions were set in Phase II at 2,778,000 pounds per year of total nitrogen and 397,000 pounds per year of total phosphorus based on the low flow year of 1991. Total nitrogen loads were calculated to be 4.28 million pounds a year at the Washington site. During this phase the 30% total nitrogen reductions goal for all sources was set at 1,285,000 pounds per year. point sources were allocated 8% of this reduction and non-point sources were allocated 92% (Templeton, 2007). An interim target of 60% progress towards the reduction was set in 1995 to be achieved by 1999. The NCDWQ and the NCEMC stipulated that they would determine if additional regulatory requirements were necessary if progress was at the 60% progression state by 1999 (J. Huisman, personal interview, April 24, 2007). The goal of reaching 60% progress toward the reduction goal was not met, and mandated rules on riparian buffers, fertilizer application, stormwater, and agriculture were adopted by the NCEMC and went into effect in 2000 and 2001 (Templeton, 2007). The Phase II agreement reduced the fixed price of non-point source credits to from approximately $25.40 per pound to $13 per pound. During Phase II the Association has easily maintained discharges well below the caps assigned without needing to purchase credits from non-point sources (Breetz, 2004).

The Phase III agreement is currently in progress and spans an additional 10 years from 2005 to 2015. Amendments may be added after every two years to address potential needs for improvements (J. Huisman, personal interview, April 24, 2007). During Phase III load reductions were set using an agreed upon cap for point sources of 891,271 pounds per year for total nitrogen and 161,070 pounds per year for total phosphorus. Total nitrogen and phosphorus caps for non-point sources were set at 2,109,220 pounds per year for total nitrogen and approximately 1,851,883 pounds per year for total phosphorus (Templeton, 2007). The agreement for Phase III proposes to resolve temporal issues related to the lifespan of non-point source credits (currently set for 10 years). The participants in Phase III are currently working to resolve issues concerning life span and previously banked credits. The agreement also sets a 10 year estuary performance goal with alternatives established to manage the water quality trading program. A process will be determined to reassess and re-model the loads on the estuary if reports suggest that the water quality is deteriorating (Templeton, 2007).

Point sources within the Tar-Pamlico Basin meet trading program compliance measures through weekly effluent monitoring for total nitrogen, total phosphorus, and flow. Association facilities have been performing effluent monitoring reports for nutrients since 1991. The Association reports monitoring data to NCDWQ annually. Water quality monitoring is performed at the standard defined in the NPDES permits for the individual point sources and as a collective group. Guidelines have been developed by the NCDWQ for estimating flow and concentration if this information is not provided by the point sources (J. Huisman, personal interview, April 24, 2007).

From a non-point source perspective wetland technology has not been incorporated fully as a primary method for reducing nutrient loads within the basin, but the methodologies developed for assessing the progress of nutrient reduction goals for non-point sources are applicable to assessing the efficiency and effectiveness of constructed and restored wetlands as BMPs. The NCDWQ recognized early on in the Tar-Pamlico Water Quality Trading Program that measuring compliance with instream loading targets would have required a very costly and complex model to estimate the edge of water non-point source nutrient discharge and nutrient levels within the instream water column. A significant amount of quantitative water quality monitoring over a number of years would be necessary to support that type of modeling (J. Huisman, personal interview, April 24, 2007). Instead of that method of monitoring and modeling the NCDWQ has developed procedures to assess compliance based on land use modification accounting methods that provide estimates of nitrogen and phosphorus reductions based on the type of BMP that is implemented.

The latest estimates of nutrient removal efficiencies are available for point sources and non-point sources to interpret. The Association is currently involved in estimating the BMPs that are most cost effective and efficient in the Tar-Pamlico Basin (Tar-Pamlico Association Meeting, 2007). The NCDWQ has developed estimates of nutrient removal efficiencies based on monitored stormwater projects developed under the Tar-Pamlico and Neuse stormwater rules and agency research.

Table 7.2. Tar-Pamlico Nutrient Removal Efficiencies for Stormwater BMPs

Practice	TN Efficiency (%)	TP Efficiency (%)
Wet Pond	25	40
Stormwater Wetland	40	35
Sand Filter	35	45
Bioretention	35	45
Grass Swale	20	20
Vegetated Filter Strip with Level Spreader	20	35
50-Foot Restored Riparian Buffer with Level Spreader	30	30
Dry Detention	10	10

(Bennett & Gannon, 2004)

Table 7.2 is the latest assessment developed by NCDWQ of approximate nutrient removal efficiencies. This table has been used by the Tar-Pamlico Water Quality Trading Program to calculate estimates for non-point source pollution reductions (Bennett & Gannon, 2004).

Other types of tools have been developed to help assess non-point source reductions in order to solidify the exchange value within the trading program. The Nitrogen Loss Evaluation Worksheet (NLEW) and the Phosphorus Loss Assessment Tool (PLAT) were developed for nitrogen and phosphorus accounting under the Tar-Pamlico agriculture rule. The NLEW was developed as a field-based procedure to estimate changes in nitrogen losses from agricultural management units. PLAT is a similar software program that determines the relative phosphorus losses from agricultural fields based on site specific information. Both models use data like fertilization rates, crop and soil acreages, and areas of BMP implementation to estimate nutrient exports to the Tar-Pamlico Watershed from agricultural land. The models were developed and designed by groups including: North Carolina State University, USDA's NRCS, and the NCDENR. Before these models were developed trade estimates for non-point source credits relied on previous local cost share record and best professional judgment based on nutrient reduction literature and documented projects (Gannon, 2003).

Performance Record

From the initial point in time of the creation of the Tar-Pamlico Water Quality Trading Program the Association has been able to meet nutrient reduction goals set by the nutrient cap collectively through improvements in operational efficiencies. At present, no trades have occurred between point and non-point sources, but the Association did allocate nearly $1 million to fund agricultural BMPs in anticipation of need non-point source credits in the future. The Tar-Pamlico Water Quality Trading Program has allowed the Association to incur substantial financial savings. Estimates for potential costs for technology upgrades were assessed at $7 million compared to the $1 million investment the association made in non-point source control (DeAlessi, 2003). The flexibility of the collective discharge goals has provided allocation maneuverability for the Association. The members within of the Association have been able to improve treatment efficiencies and devise future upgrade plans for technology so that improvements in treatment efficiency are cost-effective (Allen & Taylor, 2000). As opportunities for cost-effective technology upgrades are exhausted, trading will likely occur in the future. Though, environmentalists have criticized the Tar-Pamlico Water Quality Trading Program from the initial start-up phases because many objectors contend that the caps were initially set too high and the nutrient reduction target was set to low (J. Rudek, personal interview, April 25, 2007).

The Association successfully met all the nutrient reduction goals that had been set for Phase II, and by 2003 the group had decreased nitrogen and phosphorus discharges by 45% and 60% respectively (J. Huisman, personal interview, April 24, 2007). After the agriculture rules were implemented the non-point sources were successful in meeting their nutrient reduction goals by collectively decreasing nitrogen discharges by 45% as of 2003 (J. Huisman, personal interview, April 24, 2007). The watershed wide efforts to reduce nutrients in the basin have resulted in the reduction of impaired acreage in the estuary by approximately 90% (J. Huisman, personal interview, April 24, 2007). In fact, one segment of the Pamlico estuary has been removed from the 303(d) list for chlorophyll a (USEPA, 2005).

The Tar-Pamlico Water Quality Trading Program has provided flexibility and financial investment relief for the Association, but there have been significant public investments made in order to establish the program. The USEPA's Office of Water (2005) compiled a listing of overall costs of the Tar-Pamlico Water Quality Trading Program. The NCDSWC contribute $12.5 million between 1992 and 2003 through the North Carolina Agriculture Cost Share Program. The Conservation Reserve Enhancement Program, administered by the NCDSWC, has provided approximately $33 million to the Tar-Pamlico River Basin since 1998. Over $2.5 million in CWA section 319 expenditures were allocated to the program between 1995 and 2003. This funding from all sources supported a variety of non-point source projects in the Tar-Pamlico Basin. BMP implementations, monitoring and model tools, technical assistance and education, and program creation and structure development were all supported by these funding vehicles.

Impediments to Incorporating Wetlands into the Tar-Pamlico WQT Program

The Tar-Pamlico Water Quality Trading Program has suffered from economic and institutional impediments. The Tar-Pamlico Water Quality Trading Program has been assessed as being successful at reducing nutrient loads, but point to non-point trades have not occurred at present. By 2003, nitrogen had been reduced in the Tar-Pamlico by 34 percent over 10 years (Gannon, 2003). An institutional structure is set in place so that this option is available if needed in the future. Some stakeholders claim that the success of a trading program should not be measured by the number of trades taking place (A. Coan, personal interview, April 24, 2007). Others have claimed that there have been no trades because the trading cap has been set too high (J. Rudek, personal interview, April 25, 2007). In fact, from an economic standpoint the cap must be set at an optimal level in order to reflect scarcity. If this does not occur then the correct allocation of property rights for pollution cannot be assigned.

Population growth, intensive agricultural practices, and expanding livestock operations are engulfing the basin at a rampant pace. Some improvements have been realized concerning the ecological quality of the Pamlico Sound, but it is imperative that monitoring assessments and political factors enable future reduction caps to be set in the most optimal way to continue to improve water quality within the Tar-Pamlico Basin. From an economic standpoint further information is lacking in the program concerning the reduction capabilities of non-point sources and the positive and negative externalities that are associated with them. At the present time, the Association is in the process of funding basin-wide studies to assess the performance capabilities of various types of BMPs (J. Rudek, personal interview, April 25, 2007).

From an institutional standpoint, the NCDWQ has set plans to develop a more robust offset rate for exceedences of the phosphorus cap, and the agency also plans on creating a better estimation of the lifespan of current BMPs in the basin that includes the assessment of the geographic distribution of the projects. The NCDWQ also plans on negotiating an agreement with the Association concerning the payment longevity and credit life initiation of BMPs implemented within the basin (J. Huisman, personal interview, April 24, 2007). There are not any plans by the NCDWQ or the Association to emphasize the importance of incorporating wetlands technology into the Tar-Pamlico Water Quality Trading Program. The program relies upon the cost share program of the state to deliver non-point source credits through agricultural BMPs, and at the present time financial estimates for wetland BMPs are estimated to be greater than other BMP choices. Farmers choose what types of BMPs they want to install or practice, and in most cases they will choose the least costly or most productive reducer of nitrogen or phosphorus. The choice not to incorporate wetlands by farmers may be a rational choice, but there is a level of institutional failure that is revealed from this standpoint at the state level. There are not any financial incentives or other market driven incentives for farmers to choose wetlands over other BMPs. Funding sources for the farmers play a key role in what type of BMPs they choose. Funding sources for non-point source reductions also influence the trading program as a whole. One key factor that hampered the progress of non-point source nutrient reduction activities from an economic standpoint during early part of Phase II was limited funding, or lack of, resources to facilitate monitoring and verification of non-point source BMP implementation (A. Coan, personal interview, April 24, 2007). During Phase I the Association paid for an initial banking of non-point source credits, but by Phase II that money had already been absorbed. The Association was not required to pay for credits in Phase II unless the group exceeded its nutrient cap.

From an institutional standpoint the program has targeted agricultural BMPs as its focus for non-point source credits. The type of BMP eligible for generating nutrient reduction credits was intentionally left broad so that farmers would be able to choose any BMP listed with the cost share program to generate credits (J. Huisman, personal interview, April 24, 2007). The only limitation to the use of the cost share program for water quality trading purposes is that with the agricultural rules for the basin the nutrient reductions from BMP projects designed to satisfy the

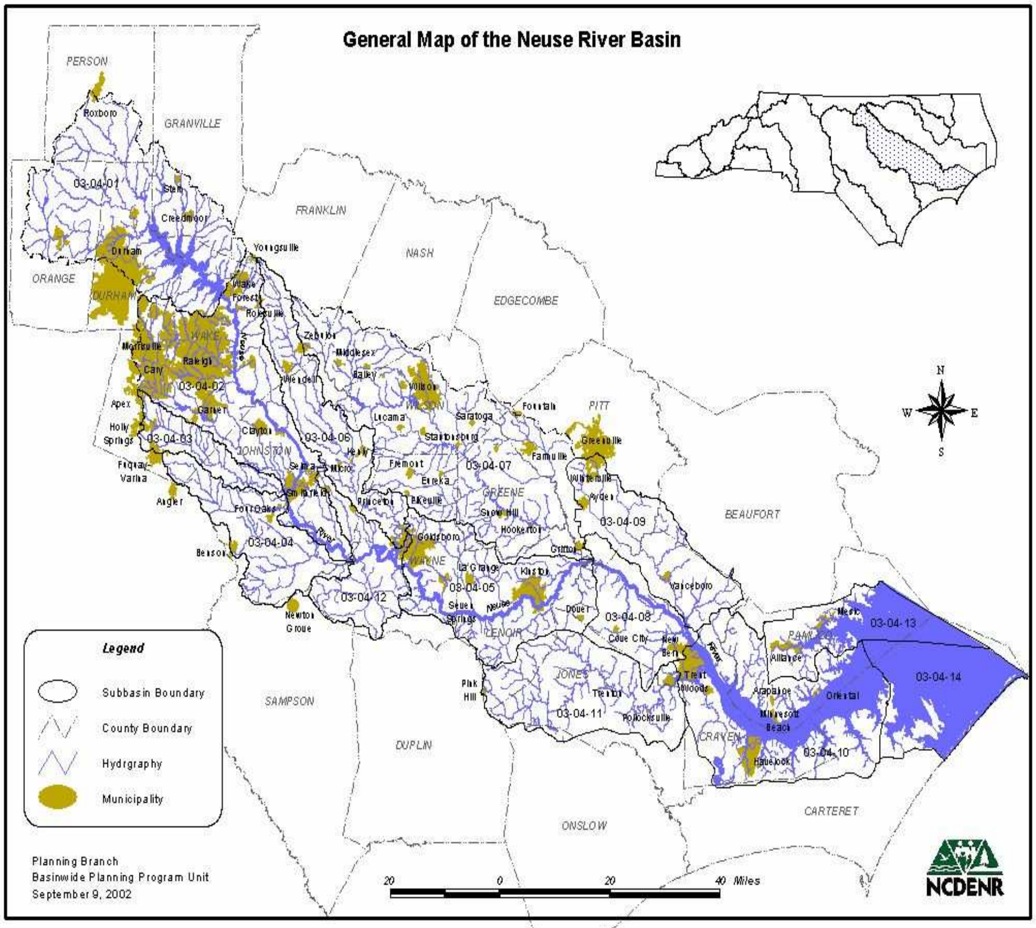
30% total nitrogen reduction required of all agricultural operations cannot also be used to generate nutrient offset credits. But, there has not been a consistent level of monitoring and assessment of BMPs to verify that the agricultural offsets are improving water quality.

CHAPTER 8

NEUSE RIVER WATER QUALITY TRADING CASE STUDY

Background

The Neuse River Basin is located in the eastern plain of North Carolina that lies directly to the south of the Tar-Pamlico River Basin and to the north of the Cape Fear River Basin. The river flows from Raleigh, North Carolina toward the east-southeast and empties into the Pamlico Sound. The Neuse River Basin covers approximately 6,192 square miles, and portions of the watershed lie in some of the most densely populated areas of North Carolina. One sixth of the state's population lives within 19 counties that lie within the basin. Approximately 1.5 million get their drinking water from and/or discharge wastewater in the Neuse River (Neuse River Foundation, 2007). Of the 9.2 million acres of land area within the basin, approximately 490,000 acres are forested wetlands, and about 24,000 acres are non-forested wetlands that include salt and freshwater marshes (Neuse River Foundation, 2007).



Map 8.1. Neuse River Watershed Map (NCDENR,2007)

In many communities along the Neuse River, population growth has increased dramatically as well as an increase in infrastructure needs. Since the 1950s wastewater discharges have increased by 650%, and approximately 100 million gallons of partially treated wastewater enters the Neuse River each day. At the present time more than 400 point source discharge permits exist for the basin, and approximately 2 million hogs reside in intensive livestock operation sites along the river (Neuse River Foundation, 2007). The rapid growth of population and hog farms within the basin has led to an overloading within the watershed of nutrients, especially nitrogen. Nitrogen levels within the basin have caused smaller tributaries to clog up from algae and vegetation growth, and the state of North Carolina has previously instituted fishing and shellfishing closures because of the monitored high levels of pollution. Recurring massive fish kills have taken place in the Neuse River Basin since 1991. In 1991 dissolved oxygen problems and Pfiesteria caused more that 1 billion fish kills in the watershed (NCRWA, 2007). By 1995, 1996, and 1997 the American Rivers environmental group classified the Neuse River as on of North America's most threatened watersheds (Neuse River Foundation, 2007).

Table 8.1. Neuse River Water Quality Trading Program Overview

Year Watershed Organization Formed	2002
Year Trading Guidelines Adopted	1997
Administrative Unit	NC Dept. of Water Quality/ Ecosystem Enhancement Program/ Basin Association
Program Structure	In-Lieu-Fee/Exceedance Tax
Pollutant Traded	Nitrogen

In 1988 the state of North Carolina developed the original Nutrient Management Strategy (Neuse NSW Strategy). At this time most of the nutrient problems in the watershed were occurring in the lower freshwater segments of the river near the mouth of the basin, and phosphorus was determined to be the limiting nutrient (NCDENR, 1998). For many years the main focus of nutrient reduction was placed on reducing phosphorus. During this period of time specific goals were not established for the reduction of total nitrogen. It was not until 1997 that a nitrogen trading program was included in the Neuse River Basin Nutrient Sensitive Waters Management Strategy. At this point in time the focus shifted from phosphorus to nitrogen, and the Agricultural Cost Share Program was identified as the primary mechanism for reducing nitrogen from non-point sources within the basin. This idea was based on the previously implemented Tar-Pamlico Water Quality Trading Program created just to the north of the Neuse River Basin where monitoring and modeling during the early part of the 1990s indicated that nitrogen appeared to be the more important nutrient compared to phosphorus for the brackish estuarine waters that formed the river basin.

The first watershed plan for the Neuse River Basin was developed in 1993. By this time it was becoming more and more evident through monitoring techniques and grab samples that nitrogen was becoming a major concern along the river. The new plan recommended that the Neuse Nutrient Sensitive Waters Strategy be reassessed before it was updated in 1998 (NCDENR, 1998). In 1995 between 20 and 100 million fish were killed in the same areas as the 1991 fish kill event, and this episode provided further impetus to revise and update nutrient controls (Neuse River Foundation, 2007). The Neuse Nutrient Sensitive Waters Strategy was revised by the North Carolina Department of Water Quality in 1997, and the new version focused on nitrogen and established the Neuse Nutrient Sensitive Waters Rules. These new rules were implemented to meet and maintain a 30% nitrogen reduction goal by 2003 while at the same time retaining the technology-based concentration limits for total phosphorus. The impacts of the fish kills in the watershed also led to the basin being listed on the 303(d) impaired waters list by the USEPA. In 2001 the USEPA Region 4 approved a Total Maximum Daily Load (TMDL) for the Neuse River Basin (Environomics, 1999).

The Neuse Nutrient Sensitive Waters Rules and the TMDL were developed by North Carolina in an effort to address the major known sources of nutrients in proactive manner (NCDENR, 1998). At the time the TMDL was established it was estimated that point sources contributed approximately 24 % of the nitrogen and phosphorus loading to the estuary. In 1995 it was estimated that there were over 111 discharges in the basin, but 32 discharges accounted for over 95% of the total phosphorus loading to the estuary (M. Templeton, personal interview, April 25, 2007).

In 1997 more than 600 people participated in the public hearing process for the development of the Neuse River Basin Nutrient Sensitive Management Strategy. Point sources during public hearings expressed concerns that strigent nutrient allocations would be burdensomely expensive, and they were interested in more cost-effective and flexible regulatory structures (Breetz, 2004). Water quality trading opportunities were discussed and the Tar-Pamlico Nutrient Trading Program, which had entered into Phase II at that point, was used as a pre-program model for the Neuse Trading Program. The draft rules were finally brought to the public for comment before being adopted in December 1997.

Administration of the Neuse River WQT Program

By 1997 the Neuse River Basin Nutrient Sensitive Waters Management Strategy had established nitrogen allocations and control options to improve water quality in the Neuse River Basin. The strategy included elements of point source to non-point source trading for nitrogen allocations and point to non-point source offsets for nitrogen loading (Breetz, 2004). The strategy established a group compliance option that included those point source dischargers that released over 5 million gallons per day. The Neuse River Compliance Association (NRCA) was established in 2002 from this group of dischargers, and in 2003 the group was issued a combined discharge NPDES permit. The 22 member group, primarily containing large municipalities, was allotted a collective NPDES permit for nitrogen based on the sum of the members' individual nitrogen allocations established in the TMDL (M. Templeton, personal interview, April 25, 2007).

The North Carolina Department of Water Quality (NCDWQ) and the North Carolina Ecosystem Enhancement Program (EEP) serve as administrative entities funded by the state to oversee and assist in the operations of the Neuse Nutrient Sensitive Waters Strategy. The Neuse River Basin Rules serve as a regulatory guide for compliance, and it is the responsibility of the point source or non-point source to demonstrate nutrient reductions at the individual or group level. The total nitrogen limit for the NRCA is specified in the group NPDES permit, but the NRCA manages the individual discharges of members through an internal fee structure (M. Templeton, personal interview, April 25, 2007).

Each individual member within the NRCA has been assigned an allocation that is subject to change based on the ongoing transactions and shifts in costs of compliance among the other NRCA members. The individual parties involved with the NRCA monitor discharges and report individual discharges to the NCDWQ on a monthly basis. This type of reporting is done in accordance with their individual NPDES permits. The individual entities also provide discharge information to the NRCA, and the NRCA compiles the individual reports and provides a bi-annual report to the NCDWQ (M. Templeton, personal interview, May 25, 2007).

The NCDWQ oversees compliance of the NRCA group nitrogen cap, but if new or expanding dischargers cannot secure nitrogen allocations from other point sources within the NRCA, they can purchase non-point source offsets by paying into a North Carolina Wetlands Restoration Fund. Offset payments are paid to the Ecosystem Enhancement Program (EEP) that administers the wetlands fund, and the transactions are tracked by the In-Lieu Fee Coordinator based in Raleigh, NC. The EEP works with local governments to identifying potential restoration projects, and the EEP establishes contracts with private companies to restore or construct wetland areas. The offset BMP projects are required to be located no farther from the Pamlico Estuary than the point source discharging entity (S. Klimek, personal interview, April 25, 2007). Wetland restoration projects are awarded to private contractors that have demonstrated partnerships with EEP. The contractors provide design and construction expertise, and they are responsible for up to one year of performance monitoring of the BMP (S. Klimek, personal interview, April 25, 2007). After one year of performance monitoring by the be private contractor the state of North Carolina takes on the responsibility that the installed BMP will perform at a certain estimated level. EEP is responsible for monitoring and verifying performance of offsets that have been created through the in-lieu fee program.

The EEP was created in 2003 by the state of North Carolina for the purpose of restoring and protecting wetland areas and waterways. The EEP combines ongoing wetland restoration initiatives with the NCDENR and efforts made by the North Carolina Department of Transportation. The U.S. Army Corps of Engineers (USACE) also has signed off on the agreement to coordinate with the EEP. All of these agencies work with the EEP because the entity is the main operational body for the state of North Carolina that deals with wetlands mitigation banking, in-lieu fee programs for environmental offsets, and other North Carolina Department of Transportation offset procedures (S. Klimek, personal interview, April 25, 2007). EEP is funded through the in-lieu fees that are paid to them for non-point source offsets, through general tax revenue allocation provided by the state of North Carolina, and a direct transfer of tax revenue per year from the North Carolina Department of Transportation (K. Williams, personal interview, April 25, 2007).

Structure of the Neuse River Water Quality Trading Program

In 1998 point sources were discharging 4.1 million pounds of nitrogen per year into the Neuse River Estuary. In order to achieve a 30% reduction set by the Neuse Rules point sources needed to reduce their nitrogen contribution by 2.8 million pounds per year. Nitrogen allocated to individual dischargers was based on the ratio of their permitted flow to the total permitted flow of all point sources within the basin (Templeton, 2007). Before the TMDL process was concluded the 30% nitrogen reduction goal was established through a series of modeling analyses developed to evaluate the effects of various nutrient reduction scenarios. The results from the models confirmed that the 30% reduction in nitrogen from the 1995 baseline for total nitrogen would be a reasonable initial target for the watershed (Templeton, 2007).

The 30% reduction target determined the initial standard to set individual nitrogen export reduction allocations throughout the basin. Local governments were assigned a nitrogen export standard of 3.6 pounds of nitrogen per acre per year to meet the overall load reduction on the watershed. Non-point source loading for the Neuse River Watershed was originally estimated using export coefficients for different land cover types. Export coefficients use land cover information to determine the amount of nitrogen, or some other substance, expected to be transported from land to water through runoff. LANDSAT imagery was used to interpret land cover classification through GIS programs for the time period ranging from 1993 to 1995 (NCDENR, 1998). The modeling revealed that nutrient loads from agricultural operations within the Neuse River Basin account for more than 50% of the nitrogen load. Point sources were identified as contributing approximately 24% of the nitrogen load, and the inputs came from urban areas, forested land, and influxes from interactions with air (Templeton, 2007).

The system established for point source to non-point source trades in the Neuse River Watershed can best be described as an exceedance tax and in-lieu fee program. Trading parties within the basin include members of the NRCA, any other discharger holding an allocation, and other landowners. Landowners may voluntarily participate in the trading program, but agricultural BMPs are not eligible for trading within this program (M. Templeton, personal interview, April 25, 2007; J. Huisman, personal interview, April 25, 2007). Non-point source trades are conducted indirectly through the North Carolina Wetlands Restoration Fund at a fixed price of $11 per pound of nitrogen per year. Landowners who apply for grants from the Wetlands Restoration Fund in order to establish non-point source BMPs are indirect trading partners, but the responsibility rests with the state for ensuring nutrient offset projects are implemented and successful (M. Templeton, personal interview, April 25, 2007).

The original unit offset payment accepted for the Neuse Rules, $11 per pound, was set to include operation and maintenance of restoring wetland areas for 30 years. New or expanding point sources that belong to the NRCA have their offset rates multiplied by 200% to account for uncertainty (K. Williams, personal interview, April 25, 2007). Therefore, a point source discharger in the NRCA that needs to purchase 1 pound of nitrogen through the EEP would an effective fee of $660 per pound. The $11 per pound rate was initially assessed by the NCDWQ with help from environmental economists at North Carolina State University (M. Templeton, personal interview, April 25, 2007). The figure was based on the cost of restoring degraded wetlands along the Neuse River. Most recently, however, there have been revisions made to the offset rate that raises the figure to $57 per pound of nitrogen reduced (S. Klimek, personal interview, April 25, 2007). A shift in focus of the EEP from site specific wetland restoration to stormwater BMPs has effectively changed the price. The $57 per pound offset rate reflects the higher price of this sort of BMP. At the present time the change from $11 per pound to $57 per pound has not taken place, and political debates are ongoing concerning this issue of increasing the price per pound of nitrogen in both houses of the North Carolina Legislature.

The Neuse Rules also created a mechanism for non-point source to non-point source trades. The Neuse Stormwater Requirements (15A NCAC 2B.0235) set a nitrogen discharge standard for local governments based on population and growth rate. These local governments are required to develop stormwater management plans and have them approved by the North Carolina Environmental Management Commission (NCEMC). If local governments do not comply with the regulation then they will be subject to NPDES permitting requirements enforced by the NCDWQ. The regulation is designed to help the local governments ensure that nutrient reduction goals are met. Developers have the option through these local government entities to install stormwater BMPs to satisfy standards. These developers may choose to purchase stormwater BMPs credits that will meet the allowable level, 3.6 pounds of nitrogen per acre per year, through the EEP program. New developments within the watershed are required to implement on-site stormwater controls at least to assure that nitrogen export does not exceed 6 to 10 pounds per acre per year from residential and commercial properties. Additional purchases of reduction credit may be purchased through the EEP for $11 per pound of nitrogen reduced in order to meet the 3.6 pounds of nitrogen per acre per year requirement (K. Williams, personal interview, April 25, 2007).

The trade structure of the Neuse Water Quality Trading Program has been constructed within the water quality protection programs developed by the state of North Carolina. There are no trading ratios for point to non-point trades, but there is a target price per pound of nitrogen discharged into the watershed. The state provides the linkage between point and non-point sources and the state of North Carolina assumes the responsibility for ensuring that the payments made into the Wetlands Restoration Fund result in non-point source nitrogen reductions. Point sources within the NRCA provide reporting data specified within their NPDES permits, but the EEP is responsible for verifying reductions from specific non-point source site nitrogen reduction BMPs.

Performance Record

The goal of the Neuse River Trading Program was to provide another option for achieving regulatory compliance concerning nitrogen allocations (Breetz, 2004). The 30% reduction goal has been met in recent years by the NRCA, and further reductions have been discussed as possible goal initiatives throughout the basin. (M. Templeton, personal interview, April 25, 2007) Non-point source loads from agriculture have been reduced by considerably and almost 200 acres of riparian buffers have been preserved along the watershed (M. Templeton, personal interview, April 25, 2007).

The Neuse Rules that have been enforced by the NCDWQ have provided the regulatory authority to decrease nitrogen levels that have been discharged by point sources within the basin. This reduction has been measured through NPDES reports. But, it is too early to tell if the EEP has been efficient and effective in its role to decrease nitrogen loads from non-point sources (M. Templeton, personal interview, April 25, 2007).

The EEP manages the North Carolina Wetland Restoration Fund. The EEP uses the fund to restore and construct wetlands and other BMPs throughout the watershed. The construction costs for wetlands can fall into three main categories: land acquisition, construction, and maintenance. Cost estimates compiled by Wossink and Hunt (2003) (Table 8.2) have been used by the EEP to develop the following cost estimates for various components of wetland construction. Table 8.2 provides a cost comparison for four stormwater BMPs for a 10-acre watershed and the nutrient removal efficiencies of each BMP.

Table 8.2. Neuse River Cost Comparison of Four BMPs for 10 Acre Watershed

Practice	Wet Pond ($)	Wetland ($)	Bioretention Clay Soils ($)	Bioretention Sandy Soils ($)
Construction Cost	65,357	11,740	124,445	7,843
Annual Maintenance Cost	4,411	752	583	583
Opportunity Cost of Land ($217,800/acre)	43,560	65,340	65,340	65,340
Present Value of Total Cost	146,474	83,486	194,751	78,137
Annualized Cost Per Acre Watershed	1,721	981	2,288	918
Annualized Cost Per Pound Nitrogen Removed	61	45	51	20

Impediments to Incorporating Wetlands into the
Neuse River WQT Program

Economic and institutional impediments have been revealed throughout the Neuse River Water Quality Trading program. The state of North Carolina has recognized that wetlands play a valuable role in the removal of nutrients from stormwater runoff. The state has specifically targeted wetland restoration and construction initiatives as key components of reducing nitrogen levels within the watershed for this trading program. The NCDWQ has provided the enforcement of the Neuse Rules to set the standard total nitrogen removal efficiencies, and the EEP receives payments for nitrogen discharges that exceed allocated amounts. The EEP then proceeds to pay for wetlands restoration and stormwater BMP construction within the watershed.

The Neuse River Water Quality Trading Program has been successful at reducing nutrient loads, but there is still much to be accomplished. In a seven year period of time from the implementation of the Neuse Rules the basin has experienced a 37% load decrease of nitrogen (M. Templeton, personal interview, April 25, 2007). At the present time no point source to non-point source trades have occurred because the state of North Carolina acts as an intermediary body between the two groups. The state assumes the transaction burden of trading by collecting payments from point sources that do not meet their allocations. The transactions that take place through the state as an in-lieu fee clearinghouse are politically influenced by the setting of the fixed price for nitrogen reduction. At the present time there is a highly contested debate taking place in both houses of the North Carolina as to what the true cost of a pound of nitrogen should be within the Neuse River. This institutional failure to identify the appropriate price for nitrogen reduction greatly encumbers the program from operating efficiently.

The Neuse River Water Quality Trading Program assigned legal requirements at the beginning stages of implementation to require certain discharge levels from point sources and non-point sources, but these two groups were not integrated as independent trading partners. Under the Neuse Rules for agriculture, farmers can implement BMPs voluntarily or participate in their individual county plans, but they cannot trade directly with point sources. Trading between agricultural non-point sources and other point sources was not initially authorized because there was concern in the beginning stages of the program that farmers would not be able to meet their own 30% reduction goals for nitrogen. Therefore, it was believed that these agricultural entities would not be able to easily generate excess reductions beyond their 30% reduction allocations.

The Neuse Strategy has benefited from a significant amount of new resources funded by the state of North Carolina. The additional funding has decreased transaction costs for point sources and non-point sources in the basin, but new assignments and responsibilities have been assumed by state agencies like the NCDWQ and the EEP. The state has become the intermediary between point sources and non-point sources. However, the significant investments that have been made into the EEP have made it less likely that point sources and non-point sources will directly work together to decrease nitrogen loads. There is no incentive for non-point sources to decrease loads as long as the EEP is responsible for providing offsets for point source discharges.

Therefore, there is no incentive for agricultural operations to reduce allocations beyond their 30% reduction standards.

Population continues to rapidly expand in the Neuse River Basin and water quality indicators within the basin still indicate that there is a nitrogen overload problem that has not been solved completely. An in-lieu fee structure has been arranged by the state of North Carolina to facilitate trading between point and non-point sources, but there are economic and institutional issues that have arisen concerning the appropriate fixed price for a pound of nitrogen reduction. A full cost analysis of this fixed price must be assessed in order to fully compensate nitrogen point discharges with appropriate nitrogen non-point offsets. This price should be determined by measuring the amount of nitrogen that is reduced by several types of wetland BMPs that have been created by the EEP. Then, costs to maintain these sites and monitor these sites should be calculated. A price for the number of pounds of nitrogen removed from the system could be determined based on these measurements.

At the present time economic information is severely lacking, there are no precise measurements of the amount of nitrogen reduction that is taking place within the basin. The additional state funding used to develop the EEP should also be used to assess and verify the performance of the restored and created wetland sites within the basin. Precise monitoring and verification would greatly improve the ability for policy makers to assess the performance of the EEP and the performance of the Neuse River Water Quality Trading Program.

CHAPTER 9

CASE STUDY SYNTHESIS

General Overview

The four case studies in this research analysis identified some of the key economic and institutional impediments constraining water quality trading programs that promote point source to non-point source water quality trades including those programs listing wetlands as a BMP option. Each of the four case studies represents a different way of incorporating a water quality trading program that includes wetlands nutrient reduction techniques. The four case studies analyzed revealed distinctive environmental water quality stressors and human settlement pressures with market and institutional constraints. The watersheds ranged in size from the smallest, Cherry Creek, at 245,500 acres to almost 4 million acres, the Neuse River Basin. All four watersheds have experienced some extreme form of nutrient related environmental degradation episode or series of hazardous events over the course of several years. Vast fish kills within each basin have been the most dramatic and profound focusing events within the basins to date. Two programs identified phosphorus as the watershed limiting nutrient to be traded: Cherry Creek and Lower Boise. The Tar-Pamlico program focused both on phosphorus and nitrogen, and the Neuse program dealt exclusively with nitrogen (Table 9.1).

Table 9.1. Pollutant Traded Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
Pollutant Traded	phosphorus	phosphorus	nitrogen & phosphorus	nitrogen

Three of the four programs engaged extensively with agricultural interests within their particular trading programs: the Lower Boise, the Tar-Pamlico, and the Neuse (Table 9.2). The Lower Boise Water Quality Trading Program incorporated the issues of the various irrigation districts within the basin while the Tar-Pamlico and Neuse programs accounted for the intensive hog, poultry, and other livestock operations within their basins. The Cherry Creek Basin was predominantly an urban and suburban environment, and the other three basins continue to experience increased population growth pressures.

Table 9.2. Eligible Parties Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
Eligible Parties	Authority members and other approved authority groups	Any entity in the basin; entry into the market not restricted	Association members and agriculture	Association members; agriculture BMPs not allowed

Three of the four watersheds flow through large population growth centers with more than 200,000 residents: Cherry Creek flows through Denver, Colorado; the Lower Boise River flows through Boise, Idaho; and the Neuse River flows through Raleigh, North Carolina. In these three areas urban and suburban population growth is increasing at explosive rates. The Tar-Pamlico River Basin also has seen increased population growth in cities of Greenville, Rocky Mount, and Tarboro.

Three out of the four programs had formally formed watershed organizational groups before they had adopted trading guidelines. The Neuse Water Quality Trading Program instituted trading guidelines in 1997, and then the Neuse River Compliance Association (NRCA) was formed after the state of North Carolina allotted a collective NPDES permit to the 22 largest dischargers within the basin (Table 9.2).

The federal government, USEPA, only played a direct role in developing one of the programs, the Lower Boise River Water Quality Trading Program. The Cherry Creek Water Quality Trading Program was initially formed through the Cherry Creek Basin Authority. That entity was created by an intergovernmental agreement and a state statute later provided the organization with certain water quality protection powers. The Tar-Pamlico Association formed from initial stakeholder group meetings held by the state of North Carolina concerning the water quality issues within the basin. The North Carolina Division of Water Quality (NCDWQ) played the key role in the developing the Neuse River Water Quality Trading Program because they set the regulatory discharge constraint (total maximum daily load TMDL) for the basin which directed the formation of the trading program (Table 9.3).

Table 9.3. Stakeholders Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
Stakeholders	Authority, Colorado State Water Quality Regulators, Parks, Municipalities	EPA, IDEQ, ICWC, ISCC, irrigation districts	Wastewater Association, NCDENR, NCDWQ, NCDSWC, Environmental Defense	NCDWQ, EEP, Association, local governments

The Neuse River Water Quality Trading Program is the only program of the four cases that has a fully operational TMDL as a regulatory discharge limiting constraint for point sources (Table 9.4). The Lower Boise River Water Quality Trading Program is expected to phase-in an operational TMDL sometime within the next year. The Cherry Creek Program uses a total maximum annual load (TMAL) criterion. The Tar-Pamlico Program has a cap set for nutrients, but this constraint is not officially recognized as a regulatory TMDL.

Table 9.4. TMDL Status Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
TMDL Status	TMAL implemented; being reassessed	TMDL currently under review	None: loading caps established for nitrogen and phosphorus	TMDL established

Differences between the case studies were revealed in the many methods of program administration, structure for trading, and performance of different trading strategies. These differences can be explained through a distinct set of factors that include trading drivers like regulatory TMDLs or other caps on nutrient loads, local culture and politics, investment and development influence from federal, state, and local government agencies, and the creation and development of water quality trading organizational bodies. Further interpretation of the differences between the four case studies can be explained through a discussion and analysis of the specific economic and institutional impediments that hinder the trading programs from operating effectively or prevent the incorporation of wetlands into the individual water quality trading program.

Distinctive Institutional Arrangements

Each of the four water quality trading programs has its own distinct type of administration. The initial program structure and stakeholder groups helped to define the administrative units. The Neuse program was developed and is maintained through the NCDWQ and the EEP. Funding for the program is generated through nitrogen offset fees and other state revenues. The state of North Carolina and the Association split responsibilities in the Tar-Pamlico program. The Association provides funding for the North Carolina Agricultural Cost Share Program, and the state of North Carolina assumes the responsibility for providing and managing non-point source credits. The Authority in the Cherry Creek program levies taxes, charges development fees, and issues recreation fees in order to financially support the trading program, but the state does have some control concerning the Authority because seven members of the 17 member Authority Board are appointed by the Governor of Colorado. The Lower Boise program initially was funded through EPA and IDEQ grants. These agencies, however, do not maintain any amount of control over the program. The ICWC, a non-profit entity, tracks trades and presents the trade contracts to the corresponding agencies and general public in a transparent way, but contracts between the independent point sources and non-point sources direct the trades along this watershed (Table 9.5).

Table 9.5. Administrative Roles Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
Administrative Roles	Authority responsible for trade interactions (tracking, enforcement, credit verification)	ICWC tracks trades; binding contracts define point/non-point interactions	State of North Carolina and Association split responsibilities (liability for credits rest on cost share program)	State of North Carolina through the NCDWQ and the EEP

Dissimilar Trading Structures

Multiple models exist for program structure used to guide and regulate trading programs, and the four case studies present the various ways this can be accomplished. The Tar-Pamlico program in North Carolina established an association of point source dischargers who were collectively regulated and allowed to trade among themselves to achieve group compliance. The non-point source entities attached to the established association of point sources were arranged through the state agricultural cost share program. The Neuse Program developed a TMDL and then an association of point sources was formed in order to meet nutrient reduction standards. The non-point controls for this program were left to the EEP to provide for offsetting nutrient reductions (Table 9.6). Neither of these North Carolina trading programs have experienced any trades to date. The flexibility afforded by the group compliance option has allowed members within the Tar-Pamlico and Neuse compliance associations to trade amongst themselves. Trading may occur within these programs as population growth pressures or agricultural pressures increase and opportunities for cost-effective technology upgrades are exhausted.

The Cherry Creek program has established a Phosphorous Bank and a Reserve Pool to accomplish non-point source reductions in the basin, and the TMAL is set to be reduced by half sometime in the near future. The LBR program in Idaho allows for trades to occur freely between any potential trading partners. The ICWC is required to report trades to the regulatory authority and the general public for review (Table 9.6).

Table 9.6. Program Structure & Market Mechanisms Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
Program Structure	Brokerage/ Clearinghouse	Contractual Market	In-Lieu Fee/Exceedance Tax	In-Lieu Fee/Exceedance Tax
Market Mechanisms	Phosphorus Bank & Reserve Pool	estimated average reduction or on-site monitoring assessment through contracts	credits generated on-going basis; point sources purchase as needed	point sources pay fee to EEP ($11 per pound of nitrogen)

A strong determinant for the feasibility of trades can be the understanding of the nutrient cap limitations in comparison to the type and number of exchanges that take place within a given program. The Cherry Creek trading program illustrates this point clearly. The load allocations were initially assigned to point sources in 1985 allowing for projected growth capacity. The point sources have been able to easily operate within their compliance limit since the time of allocation, and that is why there is no demand for trades. As point sources continue to grow, it will become more difficult for them to operate within the same load allocation limits. It should become feasible at some future point with the reduction of the TMAL and the increase in point source capacity that water quality trades will become economically preferable in comparison to facility upgrades. In contrast to this program there have been no direct trades in either North Carolina programs, and the state continues to be responsible for offsetting point source discharges. In the Lower Boise program it is necessary that the TMDL be set at the appropriate level to induce trading between point sources and non-point sources. If the level of nutrient loading allowed is set too high then trading will not occur. There is an absolute need for an appropriate TMDL to be set in order to reflect scarcity. If a limitation is not constructed then there will not be a market for nutrient reducing credits. The enforcement activities that surround the set allocations of nutrient discharge will also affect participation in trading. If there is sufficient enforcement of the discharge limits then there could be an increased necessity to trade, but if the likelihood of enforcement is remote then dischargers may decide to game the system instead of participating in the program. The issue is one of in-stream capacity, and if the trading system does not reflect scarcity then market forces will not operate effectively as a resource allocation medium.

In each of the trading programs studied it was revealed that non-point source nutrient loads exceed point source loads in these basins. The focus of all four programs was to create an incentive for non-point sources to control their discharges through trading the load reductions for a price that was less than the cost of nutrient reducing technology upgrades for the point sources within the basin. Non-point sources have several disincentives for participating in a water quality trading program. They may be able to acquire financial gains from the programs, but these gains also may include the new compliance requirements through contracts (in the Lower Boise) or through state mandated rules (Neuse and Tar-Pamlico). Regulation of non-point sources may be increasingly applicable if reliable methods of monitoring and verification are developed to isolate and quantify non-point source load reductions (Table 9.7). For a water quality trading program to work efficiently and effectively a thorough understanding of nutrient loading on a watershed scale is necessary to align the right incentives for non-point source dischargers to generate credits.

Table 9.7. Regulatory Context Analysis Matrix

Program Factor	Cherry Creek, CO	Lower Boise, ID	Tar-Pamlico, NC	Neuse, NC
Regulatory Context	Colorado State regulations specific to basin; trades managed by Authority	State of Idaho has statutory trading rules; LBR is water quality limited by sediment and pathogens	Association has bubble NPDES permit; State of North Carolina has point source, storm water, and agriculture discharge rules	State of North Carolina has point source, storm water, and agriculture discharge rules for the basin

Motivation to generate water quality credits with the incorporation of wetland may be driven by ancillary benefits to property owners. Individual non-point source dischargers may increase the value of their property or profit from other multiple benefits like habitat creation for hunting or fishing, flood mitigation, or carbon reduction. At the present time these multiple benefits are undervalued or not accounted for in the water quality trading programs referenced. Though, the Cherry Creek program and the Neuse program have focused their programs heavily on providing the extensive ancillary benefits that can be provided by wetland areas.

Performance Records

Only one of the four programs has experienced at least one point to non-point source trade. The Cherry Creek Program has processed two trades that have included the incorporation of a restored wetland site and a constructed wetland site. Through this process the stakeholders involved in the process quite vividly described the laborious process of applying for, negotiating, assessing, and monitoring credits. The application, negotiation, and assessment of the credits took place over the course of a year, and the monitoring for the trades is still taking place to date. These types of transaction costs are a major deterrent for trading programs.

The most frequently cited problem discussed by individuals that were interviewed for the case studies pertained to high transaction costs. These stakeholders and agency officials discussed the time consuming and difficult process of gaining political acceptance for trading. Cultural barriers and mistrust from environmental groups and non-point source stakeholders made initial participation in the programs difficult. Educational efforts were used in all four programs to instruct stakeholder groups of the particular trading process. The administrative cost to the regulatory agencies that managed the programs was mentioned by stakeholders in both North Carolina programs and Cherry Creek. The lengthy and expensive process of creating TMDLs or setting appropriate caps on discharges was mentioned as a high transaction cost, and the scientific uncertainties in non-point source credit evaluation procedures relating to wetlands nutrient reducing techniques was cited as a source for high transactions cost for all four programs. Specifically, in the Cherry Creek Trading Program the TMAL was originally set to allow for growth and increased capacity, but trades are not in high demand because point sources can easily maintain discharges within their allocations. From an institutional standpoint the cap (TMAL, TMDL, or other standard) should be set at a point that reflects scarcity. Without this standard there cannot be an effective market for water quality credits. In the Tar-Pamlico and Neuse Programs the state of North Carolina accepts the transfer of liability and the majority of transaction costs. In the Lower Boise Program the initial transactions costs of allocating the nutrient discharge limits for the TMDL have slowed the process of actual trading.

Transaction costs occur at every stage of the trading process, but the four trading programs handle these costs differently. The time spent on permit negotiation, the search for trading partners, administrative expenditures, transparent communication between permittees and government agencies, regulatory staff time, and monitoring and verification initiatives are all handled according to the structure of the program. In both the Neuse program and the Tar-Pamlico program the state of North Carolina shouldered most of the transaction costs through the North Carolina Agricultural Cost Share Program (for the Tar-Pamlico) and through the North Carolina Ecosystem Enhancement Program (for the Neuse). The control and oversight for environmental accountability in North Carolina for these two programs is attained at the expense of higher staff costs for agency staff. The Association in the Tar-Pamlico program helped to provide funding for additional staff resources for the North Carolina Agricultural Cost Share Program, but the Neuse program costs were solely based on the figure of $11 per pound of nitrogen removed. In the Cherry Creek and Lower Boise programs the transaction costs were mainly shouldered by the point sources that were required to meet certain discharge allocations based on their NPDES permits.

Water quality programs have taken different approaches in dealing with issues of property rights and transfer of liability. The question of who would be liable if a BMP project fails is addressed slightly differently in each of the programs included in the four case studies. In the Cherry Creek Reserve Pool and Lower Boise programs the credit purchaser is not offered a release from liability if the non-point source reduction technology implementation is ineffective. The point sources in these cases may be required to continuously monitor and maintain the non-point source credit site to reduce nutrient loads. In the Tar-Pamlico, Neuse, and Cherry Creek Phosphorus Bank programs the liability is taken on by a third party. The transfer of liability from the credit purchaser to the third party (the State of North Carolina or the Authority) assists in allowing the point source purchasers to rapidly avoid verification needs and transaction costs. The Cherry Creek Reserve Pool and the Lower Boise programs expose the point sources to risk. The risk that these entities take on concerns the notion that purchasing water quality credits does not eliminate the possibility that the same discharge issue could arise again some time in the future. The purchase does not eliminate their liability. The additional costs associated with monitoring and verifying BMPs included with this type of risk makes the notion of water quality trading less appealing to point sources.

From a non-point source perspective water quality trading programs do not tend to be structured in a way that compensates credit generators for the risks taken to implement BMPs. In the Lower Boise program non-point sources were not driven by regulation, but these stakeholders realized the opportunity to improve their property with outside funds without having to monitor or assess reduction loads (R. Finch, personal interview, April 12, 2007). In properly functioning markets investors build their cost of risk and uncertainty into the price of their goods. For water quality trading programs, increased understanding of the ancillary benefits of BMPs like wetlands may generate understanding of value beyond water quality credit prices. This type of representation of the implementation may increase the attractiveness of participating in a trading program to non-point sources.

The key economic issue that makes water quality trading appealing is the efficiency that is created when one discharge source is able to more cost-effectively reduce its outputs compared to another source. Without this guideline then the program is not financially viable. It is necessary that economic considerations are incorporated into performance assessments of trading programs in order for the programs to be considered viable tools to achieve water quality standards. The economic trading barriers that were revealed in the case studies presented were highlighted the economic impediments that were discovered in the literature. The inability to set appropriate caps, the inappropriate allocation of property rights (nutrient allocations set through TMDLs), the lack of information concerning non-point source reductions, inaccurate assessments of positive and negative externalities, and high transaction costs were all economic impediments that were brought forth by the case studies. These types of economic barriers prevent equitable and efficient negotiations from occurring. Negotiations do not occur because the lack of economic information increases the risk in relation to the return on investment to the point source, non-point source, or to both parties.

There are also several political constraints that were observed in the case studies presented that dealt with the institutional settings affecting the performance of the programs. Institutional impediments that were identified by the literature were also revealed in the cases, and are directly related to the economic impediments. The inability for government agencies to set caps, the inability to assign property rights, the lack of good information, the inability for the government entity to account for positive and negative externalities, and the inability to efficiently manage transaction costs were all institutional failures highlighted in the case studies.

One of the institutional impediment examples within the cases dealt with the effective implementation of the programs. The Cherry Creek Authority was given a statutory mandate by the State of Colorado to provide for the use of trading within the basin. The North Carolina programs were provided with state regulations and nutrient limiting caps in order to direct trading initiatives. Therefore, from an institutional setting everything is in place to direct trades based on water quality regulation. But, as the case studies revealed, effective implementation of the programs did not occur immediately because of insufficient funding, undeveloped property rights, lack of political will, or stakeholder inexperience with the understanding of the water quality trading concept.

Impediments to Incorporating Wetlands into a WQT Program

The literature and cases have revealed several important economic and institutional impediments that prevent water quality trading programs from operating effectively. The case studies revealed that a water quality trading market cannot exist without the creation of a market framework that reflects scarcity. Scarcity can be imposed through the implementation of a cap (TMDL) that limits the amount of nutrients that can be discharged into a watershed. This cap that mirrors resource constraints is absolutely necessary in order for a market to function.

The case studies revealed that only two point to non-point trades that have occurred within the four research sites, and both of these took place within the Cherry Creek Basin. The lack of trades is directly related to the lack of a cap that is set at a level that reflects artificial scarcity. If there is no enforceable cap, then there is no real market for water quality credits.

The case studies also revealed that transaction costs play a large role in the operation of these programs. If a cap is set and enforced, then transaction costs are another economic impediment to trading. At the present time the case studies revealed that there is a lack of good information that assesses the ability of wetlands to remove nutrients at the watershed scale. Monitoring, in order to gather information to lessen transaction costs, is essentially necessary for wetlands to perform as nutrient reducing non-point source credit generators. Most water quality trading programs bypass performance monitoring for quantifying nutrient load reductions from non-point sources. Instead of monitoring, these programs use conservative estimates derived from scientific literature of effectiveness of a type of BMP to reduce nutrient loads. For instance, a larger amount of wetland acreage may be required to achieve the desired nutrient load reduction through the conservative estimate compared to actual monitoring information targets. Safety factors are used to increase confidence in performance, but this type of conservative estimation is flawed because there is never a true assessment of the amount of nutrients that the non-point source credit is reducing. The Cherry Creek program is a notable exception to this type of estimation. The Authority requires direct measurement of nutrient load reduction. This type of information can be gathered in the basin by creating inflow and outflow points for a restored or constructed wetland. The Lower Boise program also proposed performance monitoring for constructed wetlands, but this proposal is only an option. The alternative choice for non-point sources would be to earn credits through a ratio derived by estimation rather than direct monitoring. The other two water quality trading program case studies did not record actual performance in reducing nutrient loads. The assessment of nutrient load reductions from non-point sources was presumed based on estimates and safety factors.

An institutional impediment that was discovered through the case studies, revealed that the rationale for not monitoring non-point reduction sites is based on the idea that monitoring is not feasible because accurate measurements cannot be attained from various types of non-point source BMPs. It also has been assumed that it is more cost effective to overestimate the size of a particular wetland site (or BMP area) to overcome uncertainty rather than to directly monitor. This institutional assumption may stem from the fact that there is a wealth of scientific information concerning the nutrient removing capabilities of various types of wetlands, but the available information has not been compiled in a way that would be useful in determining the possible performance of restored or constructed wetlands in reducing nutrient loads. In the Lower Boise program case study, the ISCC presented the opinion that constructed wetlands should not be used for creating non-point source credits because at the present time there is not any watershed scale data on the effectiveness of wetlands to reduced nutrient loads. Most information concerning the uncertainties of wetland nutrient reduction capabilities have been measured at single sites. Many interrelated parameters need further study in order to better understand the watershed effects that can altered through the incorporation of restored and constructed wetland technology. Some of the parameters include: specific drainage patterns over time, relative location of a wetland within the watershed, type of wetland, seasonality dynamics, and temporal changes in nutrient reduction performance rates with varying loads over time.

Another institutional impediment that was identified by the case studies dealt with the assessment of risk and uncertainty. The Cherry Creek program case study revealed an opportunity to reduce uncertainty and increase water quality trading program potential by establishing objective and reliable means of determining performance of restored and constructed wetlands. The Authority's approach is to develop cost-effective guidelines for collecting monitoring guidelines. The original restored wetlands in the Phosphorus Bank have provided valuable monitoring assessments of wetland capacities to reduce and immobilize phosphorus loads. The EEP that handles the non-point source reductions for the Neuse program uses a combination of existing information and new research to develop general performance data to inform the creation of generalized calculation guidelines for estimating non-point source reductions. The EEP acknowledges the dynamic nature of wetlands in the Neuse Basin, and the agency is involved in assessing nutrient retention rates within the context of the larger geographic scale (S. Klimek, personal interview, April 25, 2007). Additional applications like establishing baseline nutrient levels and mapping wetland sites within the watershed should provide better quantifications of nutrient fate and transport. The Cherry Creek and Neuse program officials indicated that wetlands were the preferred BMP for those particular watersheds. Both programs are interested in continuing to promote the incorporation of wetland technology into their watershed water quality trading programs (Table 9.8). From an institutional standpoint, the Cherry Creek and Neuse programs have identified the positive externalities associated with incorporating wetlands within their watersheds, but the ancillary benefits that wetlands provide are either undervalued or non-existent in the Lower Boise and Tar-Pamlico programs. The case studies have revealed that is it imperative that institutional arrangements be set in order to incorporate wetlands into a water quality trading program. Without institutional identification of the value of these resources the market may fail to distinguish these BMPs versus other types of pollution reduction technologies.

Table 9.8. Future Wetland Incorporation Analysis Matrix

Program Factor	**Cherry Creek, CO**	**Lower Boise, ID**	**Tar-Pamlico, NC**	**Neuse, NC**
Future Plans to Incorporate Wetlands into their Program	Yes--Authority is focused on continued use of wetlands for nutrient reduction and other ancillary benefits	No--Market will determine least cost solution to reducing phosphorous loads	No-focus is on agriculture BMPs financed by the cost share program;	Yes--EEP is focused on wetland restoration initiatives throughout the basin

All four case studies revealed the importance of scientific information in determining how wetlands reduce nutrient loads within a given watershed. The cases brought forth the need for water quality trading programs to better assess time limits concerning the useful life of water quality credits that have been generated by wetlands. Various life spans were discussed in the trading programs presented, but there may be additional regulatory implications associated with the wetland areas when credits expire. The potential option value of the land may be diminished if the wetland could become regulated under the Clean Water Act after the useful water quality credit life is used up. This potential aspect could be a deterrent of incorporating constructed wetlands into a water quality trading programs. Further policy implications will need to be addressed by the federal and state governments concerning the incorporation of wetlands into a water quality trading program. If the policy guidelines continue to support and encourage the use of restored or constructed wetlands in water quality trading programs then the long-term regulatory implications of this type of inclusion will need to be more clearly validated.

Lastly, the case studies did reveal that even without an effective market for trading, there was evidence that suggested that institutional benefits that can be gained through cooperation among various stakeholder groups. Some stakeholders claimed that the success of trading programs should be measured in more general terms instead of the number of trades that have occurred. All four of the cases provided evidence supporting the idea that very diverse groups of stakeholders had been brought together to identify, assess, and try to solve a severe water quality problem within a basin. In fact, this benefit of coordination and cooperation was suggested as being the most successful aspect of the Tar-Pamlico program by a couple of individuals that were interviewed (A. Coan, personal interview, April 24, 2007; J. Huisman, personal interview, April 25, 2007).

CHAPTER 10

SUMMARY AND CONCLUSIONS

Summary

The common assumption based on economic theory that suggests that market-based approaches can be directly substituted for outdated or inefficient traditional regulatory procedures has not been supported by evidence in the case studies presented. Market based environmental trading programs are often touted as alternatives to market regulation, but the markets are only successful to the degree that there are binding caps and allowances that are well defined. Water quality trading programs require supportive legislation, strong institutions, and effective monitoring and enforcement procedures to be viable. At the present time an increased level of administrative intensity for water quality trading is necessary because of the high level of uncertainty of non-point source reductions. This conclusion follows the transaction costs theory provided by Williamson (1985; 1996) discussed in the literature review.

As the literature suggests and the case studies reveal, all aspects of markets for water quality trading are determined by regulatory decisions. These markets are not an alternative to regulation, but rather, a supplement. Public involvement, statutory requirements, monitoring assessments, and enforcement actions on the surface may not be substantially different from traditional regulatory approaches in the broad assessment of water quality trading programs. Therefore, the determination of water quality trading would not be considered a direct replacement for regulatory requirements. In fact, the application could be looked upon as an additional tool to provide watersheds with an option to provide structure in order to meet or exceed regulatory standards. A strong institutional base is required in order for a water quality trading program to be implemented within a watershed. It is absolutely necessary that resource scarcity be reflected through institutional guidelines. The literature and the case studies show the importance of defining the enforceable cap that is necessary in order to create and maintain a market. Once that base is defined then various types of trading structures may form in accordance with stakeholder demands, desires, and political influences.

Water quality trading programs are quite complex, and the incorporation of wetlands within the programs increases the complexity in many ways. Two of the case studies, Cherry Creek and the Neuse River, illustrated the feasibility of incorporating wetland technology into a water quality trading program, but there are several aspects of these types of programs that may increase uncertainty and risk factors. The Cherry Creek program requires monitoring and verification of non-point source reduction credits, but the Neuse River program assesses an $11 per pound nitrogen reduction fee for non-point source mitigation efforts managed by the EEP. If this price is inclusive of all the costs of restoring or constructing wetland acres then the fee is compensating for the nitrogen discharges by point sources. If this price is too low then there is a gap in the funding for wetland restoration and construction initiatives, and funds must come from other sources.

There are two vague areas concerning the ability for wetlands to reduce non-point source nutrient loads. The information gaps that exist in these areas involve the ability to quantify the performance of multiple wetland areas in reducing nutrient loads and the ability to interpret nutrient load reductions by wetlands spatially throughout a particular watershed. The dynamic nature of many types of wetlands increases the complexity of these ambiguous areas of study. Many factors influence nutrient removal efficiency within wetlands and further research needs to establish wetland monitoring strategies that allow for quality assurance mechanisms. There is also a need to conduct research on the long-term fate of nutrients removed or immobilized using constructed wetlands and to compile scientific information that analyzes the effectiveness of certain types of wetlands in removing nutrients and the long-term removal capacity over time.

The literature review and case studies in this report illustrate the need for additional policy research for water quality trading programs to successfully integrate non-point source nutrient load reduction through the use of constructed wetlands.

Economic Impediments

The setting of a nutrient cap and the allocation of property rights is the most important part of making a water quality trading program effectively viable. The literature provides information discussing the importance of setting a cap in order to reflect scarcity, but the case studies truly reveal that the cap must be set at the optimal level in order to create demand and supply within a water quality trading market. At the present time, risk, is entirely assumed by either point sources or a state. Total maximum daily loads (TMDLs) allocate individual nutrient loads to point sources, but non-point sources receive one lump allocation. Therefore, point sources are required to decrease their loads based on individual allocations while non-point sources are not responsible for individual loads. The state of North Carolina in the Tar-Pamlico and Neuse programs accepted the property rights responsibility for decreasing non-point sources loads. In the Cherry Creek program the responsibility for reducing nutrient loads rests with the Authority. In the Lower Boise program the responsibility is shared between the point source and the non-point source through a contractual agreement. Although in this program, the property rights have not been defined because the TMDL has not been adopted. Property rights must be assigned in order to distribute equitable pollution rights along with liability for failure to decrease loads. Property rights must be clearly documented in reports based on a temporal scale. If these allocations are not arranged then the relationship between a point source's impairment and the actual site providing abatement may be unclear. Liability assessments must be distributed to point or non-point sources as in the Lower Boise program, or a third party will have to accept the responsibility. At present, according to the Clean Water Act (CWA), the third party must be the state in which a program is managed or a statutory body like the Authority in the Cherry Creek program because point sources cannot transfer liability.

A significant driver that may entice point sources to purchase water quality credits is the precision with which the permittee can meet nutrient allocations through direct purchases. The traditional option for point sources is to purchase expensive pollution abatement technology. These costs are often “lumpy.” For instance, the point source may be required to purchase 50 units of reduction when perhaps only 10 units are needed. Therefore, water quality trading would be a viable option because a point source could purchase 10 units from a point source in order to smooth out his cost curve. However, the problem that arises concerns the lack of readily available information to both credit buyers and sellers and the assessment of positive externalities generated by wetlands.

The literature and the case studies revealed that at the present time there is a lack of information concerning nutrient reductions by non-point sources. The information of how wetlands perform in nutrient reduction is varied. Abundant performance monitoring data for site specific areas exists, but the results show wide variations in the ability for wetlands to reduce nutrient loads. Also, there has not been any watershed scale analysis of how wetlands function to reduce nutrient loads. Effects of seasonality, geography, hydrology, wetland type, and scale are not documented in most watersheds. There is a great deal of data on specific functions of wetland sites, but there are many gaps that are left to be filled concerning how wetlands remove or immobilize nutrients. There is also a lack of information concerning the nutrient removal capabilities of wetlands over time. It has been discovered that nutrient removal capabilities may degrade over time (i.e. Shop Creek at Cherry Creek), but the variations in this process are not fully understood or explained. Uncertainty in understanding how wetlands remove nutrients creates risk that is difficult to quantify. In some instances wetlands may be more easily monitored that other BMPs, but design specifications for wetlands do not guarantee that nutrient loads will be reduced by an exact amount in all situations. Without good information directing the water quality credit exchanges, then there is a lack of understanding of what is taking place. In fact, buyers and sellers cannot assess the commodity (pounds of nutrient removed) that is being developed for trading. In order to improve the understanding of wetlands ability to reduce nutrient loads, there is a need to monitor and verify nutrient reduction performance at the watershed scale.

Nutrient input information for point sources and non-point sources is also important to understand as well as nutrient output. Further information is needed to understand the nutrient output from both point sources and non-point sources. Credits should only be constructed in a equivalent manner in order to offset point source discharges. From a regulatory standpoint there is a lack of information concerning new technologies that might be mandated by USEPA in order to decrease the influx of hormone inhibitors in the watersheds throughout the country. Regulations may cause point sources to upgrade technology that would decrease overall nutrient loads discharged from these entities. Most point source dischargers have not exhausted their preferable alternatives for nutrient reduction, and there is additional regulatory uncertainty in the knowledge that watershed nutrient criteria are now being developed in some states. With this regulatory uncertainty both point sources and non-point sources could perhaps be taking a risk by investing in a practice that may change based on nutrient rules that are adopted in the near future.

Non-point source reduction information is important in order to define the commodity being traded, but other information is also necessary in order to allow a trading program to operate efficiently and effectively. At the present time wetland performance is typically estimated or modeled, and ratios are applied in order to mitigate uncertainty. Methods for monitoring and verification vary widely throughout the country, and there is no standard approach to assess credits. This lack of information drastically increases transactions costs. In the Cherry Creek program there was no standard way to account for the restored and constructed wetland areas that the Arapahoe County Water and Wastewater Authority (ACWWA) implemented. The process of individually applying for credits from the Authority took a great deal of time and expense. The Authority did not have a process that allowed for a standard approach to monitoring and verification, and this caused inefficiency. The Lower Boise and Tar-Pamlico programs have lists of BMPs that can be incorporated into their programs. These lists also provide nutrient reduction levels that are based on current literature, but these lists are only estimates. These estimates incorporate safety factors, but the commodity is not defined enough in order to minimize risk and uncertainty. Therefore, more information must be provided to credit generators and purchasers in order to fully assess what is being traded.

The final major economic impediment for incorporating wetlands into a water quality trading program at the watershed scale deals the choice by non-point sources to install wetland BMPs. As mentioned, if wetlands were similar to other nutrient abatement technology then credit producers would choose from all available BMPs within a trading program in a way that minimized their cost. These producers would choose wetlands if they represented the least cost method of creating credits. In some situations wetlands may present the least cost option, but in the case studies presented the wetlands that were discussed were not the least cost option. The wetlands that were restored and constructed in the Cherry Creek program were expensive compared to the installation of buffer strips or other BMPs. In the Lower Boise and the Tar-Pamlico programs other agricultural BMPs like no-tillage practices were less expensive. In the Neuse program the focus was on wetland restoration, but at the present time the charge of $11/per pound of reduced nitrogen does not necessarily encapsulate the total cost of restoring wetlands within the basin.

From an economic standpoint wetlands could be emphasized over other types of BMPs. For instance, the Neuse program specifically funds wetland restoration projects through the state EEP program. There may be other ways of emphasizing the importance of wetland technology in the reduction of nutrients through trading ratios and multiple markets. A higher level of certainty of nutrient reduction that can be gained through an inflow/outflow monitored wetland (like Cherry Creek) would yield a lower ratio and a better estimation of the commodity being traded to the point source. As well, if credit producers were able to sell different types of functions then wetlands may be the preferred method of choice when reducing nutrients in a water quality trading program. If pollution targets are set optimally (TMDLs) and information costs are low then non-point sources should be able to trade credits for flood mitigation, habitat, carbon sequestration, etc. By providing a market for some of these positive externalities, private landowners may begin to at least consider using wetlands as nutrient reducing BMPs.

The case studies from the research presented extends the current literature to emphasize the absolute importance of reflecting scarcity within a water quality trading program. To do this there must be a nutrient cap that is set and enforced for a watershed. Secondly, transaction costs must be minimized with increased monitoring and verification that improves information about wetland nutrient reduction capabilities. Lastly, the case studies revealed that wetlands may not be chosen as nutrient reducing BMPs based simply on their performance. Other BMPs may be substituted for wetlands. Therefore, at the present time institutional arrangements must be defined in order to focus attention on wetlands as nutrient reducers as well as producers of other ancillary benefits. In the future, multiple markets for environmental goods (habitat, flood control, carbon sequestration) may help to define wetlands as the preferred BMP choice for land use change within watersheds.

Institutional Impediments

The concept of water quality trading is rooted in section 402 of the Clean Water Act (CWA). TMDLs were an original part of the FWPCA of 1972, but the application and enforcement of these standards have been almost non-existent to date. As the case studies verified, water quality trading ultimately depends upon a capped nutrient limitation. This cap in most cases could be a regulatory issued TMDL. Without this cap, property rights cannot be allocated and there is a lack of scarcity that is generated by the manufactured market. Ultimately, as the case studies have revealed, the institutional failure of not being able to correctly set a cap limitation is the ruin of water quality trading programs. Other economic and institutional issues become secondary when discussing the importance of setting a cap. It is important to set and allocate nutrient load caps at the level that is most equitable to all stakeholders while also reducing total effluent loads. Demand for water quality trading credits is absolutely determined by the initial distribution of rights to pollute under the initial cap that is set or through changes in the technology based or water quality based effluent limitations (TBELs or WQBELs) set by USEPA or states. Under USEPA all NPDES permittees must only purchase credits to meet WQBELs. All NPDES permittees must meet the TBELs without the purchase of credits. Therefore, credit providers (non-point sources) must be able to provide for the level of demand that exists between these two effluent limitations. If the WQBELs are not set at a stringent enough level in order to meet a TMDL then point sources can meet their allocation limits without searching for non-point source credits. This notion of meeting allocation limits without the need for non-point source credits was made very apparent in three of the four case studies presented. The Cherry Creek program's total maximum annual load (TMAL) that was set in 1985 was set at a level that has been consistently attainable, and all point source dischargers have met their allocation load permits since the cap was set. There have not been large demands for trades in the

Tar-Pamlico or Neuse programs because point sources have been able to stay well below their permit allocation loads. The TMDL has not been officially set for the Lower Boise River, but allocation discussions have taken place since 2000. The lack of properly set limit that is equitable and firm can greatly inhibit a trading program from operating efficiently and effectively. A water quality credit provider and purchaser have a difficult time planning a production or purchasing strategy in sizing up the watershed market for nutrient reductions when an exact and enforceable measure is not set. In order to assess demand in water quality trading a credit producer or purchaser must have an intimate knowledge of the regulatory forces that drive discharge limitations. If a cap is not correctly allocated and maintained or if the cap is not enforceable then there is no demand to purchase nutrient reduction credits.

A second institutional impediment that was highlighted by the case study research dealt with the allocation of property rights and focused on the transfer of liability. The literature suggested that environmental trading markets are inherently risky, but the case studies further clarified the relationship between risk and liability concerning water quality trading markets. The National Pollution Discharge Elimination (NPDES) Permits held by point sources does not allow liability to be transferred to a non-permit holder. Those point sources that purchase water quality credits do not relieve themselves from the responsibility of meeting their permit requirements. The inability to transfer this NPDES liability is one of the issues that face point sources when dealing with non-point sources. This inability is also the reason why most water quality trading programs can be described as offset programs instead of true trading programs. Institutional arrangements like the state of North Carolina in the Tar-Pamlico and Neuse cases and the Authority in the Cherry Creek case take on the liability for the transactions that take place between point and non-point sources. The liability is never transferred in these programs from the point source to the non-point source. In the Lower Boise program the liability is shared through a contractual arrangement between the point source and the non-point source. In this scenario the contract provides a safety mechanism for the point source place some responsibility on the non-point source to reduce nutrient loads. But, the point source does not entirely transfer its liability and it is ultimately responsible for its discharge allocations including non-point source credit purchases.

The literature also points to the importance of baseline information in order to direct how credits are generated from a supply side after a cap is implemented. In theory, institutional forces are responsible for setting baselines for the allocation of nutrient loads. Baseline allocations are important in order to determine if the water quality trading program is actually improving the environment by decreasing nutrient loads on a watershed. The case studies revealed that some water quality trading programs do not assess baseline criterion before they implement trading procedures, and in some cases it is not apparent that baseline assessments and allocations match TMDL limits. For instance, the Tar-Pamlico program was the only program out of the four case studies that initially set baseline conditions for the entire watershed. But, the nutrient cap was set too high and trading did not become feasible. The program, however, incorporated the use of a computer model to assess baseline allocations, but the Association members did not demand water quality credits from non-point sources because they easily were able to meet their allocation loads as a group. The state of North Carolina had to develop agricultural rules in order to decrease nutrient loads from non-point sources instead of using market forces to drive the demand for non-point source credits. The agricultural community now has to meet the rules stipulated by the state while they still may receive subsidies for implementing BMPs. The Tar-Pamlico example serves as a reminder of institutional failure, and it demonstrates that if the correct baselines are not implemented then a trading program will be inefficient. On the other hand, in the Cherry Creek program the Authority set individual baseline assessments for the wetland credit generating projects installed by the Arapahoe County Water and Wastewater Authority (ACWWA). Both baseline assessments were measured before installation, and trading ratios for projects were determined from these baselines. The accounting for credits in the Cherry Creek program only assessed individual areas, and it did not assess the overall change in the quality of the watershed based on changes incorporated through the additional BMP offsets. Although in relation to the specific projects, the restored wetland area did receive a better credit production ratio as compared to the constructed wetland. This information serves as an example of why there is a need for further research to determine nutrient reduction capabilities of wetland areas within specific watersheds to distinguish restored credits from constructed credits during baseline analyses. The case studies reveal that baselines are absolutely necessary in order to determine that trading is improving the quality of the water at the watershed scale.

Another institutional impediment that was revealed in the Lower Boise River and the Tar-Pamlico case studies is fact that point sources are regulated through the Clean Water Act (CWA) and the majority of non-point sources are not. Potential water quality trading credit providers as revealed in three of the four case studies presented are largely members of the agricultural community. This group waged a long and successful battle to exempt themselves from the regulations of the CWA, and therefore there is some apprehension about the regulatory control that may arrive with the acceptance of a water quality trading program. The current political reality is that the U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS) is charged with supporting conservation through green payments. Policymakers at the NRCS view water quality trading markets as a way to distribute green payments to benefit farmers. Point sources may hesitate to participate in program with non-point sources that receive subsidies to install BMPs. As well, the subsidies that are distributed to potential credit producers may not be delivered to potential trading parties in an equitable way. Therefore, the subsidies would decrease the costs for some credit producers and allow them to sell their credits to point sources at a lower price compared to other credit producers that did not receive subsidy payments.

The literature also provides insight as to the importance of monitoring and verification in order to assess positive and negative externalities. But, the case studies reveal that monitoring in most instances does not take place. Some water quality trading programs do not actively monitor non-point source effluent reductions at all. Some programs like the Tar-Pamlico program spot-check BMP sites to make sure that they are installed properly. The Cherry Creek program directly monitors inflow and outflow of its wetland sites. This method of inflow/outflow monitoring is applicable because of the wetland BMPs, and this method may not work for other types of BMPs like buffer strips, conservation tillage, or other practices. At the present time it is unclear what level of monitoring is necessary concerning water quality trading in order to provide a level of confidence to buyers, sellers, and regulators while at the same time decreasing nutrient loads in a specified waterbody. Although monitoring and verification are crucial aspects of a water quality trading program, monitoring and verification activities do increase transaction costs, but they reassure the purchaser of credits and the regulatory body that the commodity being traded actually exists. Therefore, some level of monitoring has to exist in order to provide assurance that the water quality trading program is actually improving water quality.

The last institutional impediment that was identified through this research dealt with the focus of institutional structure of the trading program on wetlands. The literature suggested that, at present, markets fail to account for the positive externalities that are associated with wetlands. The case studies highlighted this market failure and also provided examples of the institutional failure to account for these public benefits that are generated by wetlands. Wetlands have other functions that directly or indirectly benefit humans, but economic theory suggests that credit producers will not consider these functions because they do not affect profit. Therefore, “bounded rationality” takes place in programs like the Lower Boise River. The principal (point source) contracts with the agent (non-point source) to decrease nutrient loads by a certain amount. The agent (non-point source) chooses the first satisfactory solution to decrease the nutrient level in accordance with the point-source. Stakeholders in the Lower Boise do not take into account the positive externalities that are created by wetland areas, and thus this type of approach to nutrient reduction may be overlooked. The Tar-Pamlico program does not have a preference for wetland BMPs, and therefore, this type of technology may be overlooked by stakeholders within the basin. The Cherry Creek and Neuse programs specifically focus on wetland areas, but both of these programs are managed directly by government agencies. The government agencies in these two cases have focused their attention on wetlands because of the ancillary benefits that are provided in conjunction with water quality enhancement. At the present time analysis does not provide information that suggests that wetlands would always provide a higher level of certainty compared to other BMPs when reducing nutrient loads. Similarly, the many functions that wetlands provide cannot be captured in multiple markets. There are not markets in the U.S., at present, for carbon sequestration, habitat, or flood mitigation. Therefore, with this lack of information it is important for policymakers to prescribe wetland techniques as a preferred method of nutrient abatement in order for them to be incorporated within a water quality trading program.

Conclusions

Water quality trading in the United States is still in its infancy, and the following conclusions provide insight concerning the major challenges that face this innovative approach to improving water quality.

Economic Impediments

The central economic issue concerning water quality trading programs is that there are not effective market conditions within the case studies analyzed to allow for effective trading to take place. For a water quality trading market to exist their must be a cap that is defined and enforceable that reflects effective resource scarcity. The main reason that trading within water quality markets has not been robust is the fact that the costs associated with noncompliance are not high.

The following discussion of economic impediments focus on determining value and risk associated with strategies that use wetlands to reduce nutrient loads within a water quality trading context. These recommendations are incorporated to address the economic gaps and barriers that complicate the uncertainties and risks associated with water quality trading. Economic recommendations include setting a nutrient cap to reflect scarcity, a better method of assigning and enforcing property rights in relation to the set nutrient cap, the institutional acts of providing information and improving regulatory practices, the assessment of the economic functions of wetlands, and the further research of adopting multiple markets for wetland functions to promote a credit producer's ability to sell different credits in different markets.

One of the major issues discovered by this research was the inability set an optimal trading cap and to assign property rights from this cap. From an economic perspective a better method for assigning property rights and assessing liability should be adopted. Increased enforcement activities and stronger penalties for point source discharge violations could increase demands for water quality trading credits, and the allocation of property rights at the beginning a program's development would help credit producers and purchasers forecast future production strategies and help to reduce the applicability of market failure. The setting of a cap and the allocation of property rights would also allow regulators to better assess reduction capabilities and the overall performance record of the program.

The improvement of regulatory practices could decrease economic transaction costs at present. Improved monitoring and verification techniques, improved credit assessments, and improved credit process negotiation would greatly enhance trade potential if the initial caps are properly set. The improvements to regulatory practices would require investment by government agencies to increase staff levels and expertise, simplify and standardize policies and practices, and upgrade equipment. Also, increased focus should be placed on applying current knowledge of monitoring and verification of wetlands into standardized procedures at the watershed scale within water quality trading programs. Therefore, monitoring and verification could have a more direct impact on the trading ratios that are assigned to particular non-point source BMPs.

Assessing the economic value of the functions of wetlands would provide a more comprehensive understanding to stakeholders of the monetary worth of the resource compared to other option values for land use or development. A better understanding of how wetlands remove nutrients is needed. From the literature it is apparent that there is a great deal of information that has already been generated concerning the functions of wetlands. Information concerning nutrient uptake capabilities should be compiled and analyzed. Information gaps should be identified from this compilation in order to direct the next steps that should be taken to better define the performance of various types of wetlands in removing nutrients within specific venues of the country. This information is vital to defining the applicability of incorporating wetlands into a specific water quality trading program. Water quality trading programs will also require site specific assessment work in order to incorporate wetlands in a watershed scale program.

Valuable economic information also could be gained through the investigation of the feasibility of making trading credits available for multiple environmental amenities. At present, markets either undervalue or disregard many environmental amenities provided by wetlands. These amenities could be assessed as marketable commodities and may include water quality, flood control, habitat, or carbon sequestration. These ancillary benefits could be provided by implementing restored or constructed wetlands, and then the additional wetland benefits could be better accounted for. This type of research would require the assessment of market valuations of specific functions over time. Multiple ecological values could increase the opportunity to improve the return on investment that non-point sources are able to retrieve based on the construction of a type of wetland on their property. This notion of stacking credits could offset the lost opportunity costs associated with the non-use of the designated land for other purposes.

Institutional Impediments

Politics determines many of the regulatory factors and policies that guide the administration, structure, and performance of water quality trading programs. Ultimately, it is institutional failure that causes water quality markets not to succeed. Water quality trading markets are artificial, and the institutional structure that directs them is responsible for their achievements or failures.

Additional focus must be set on defining the correct cap limits while allowing these limits to be adjusted readily in order to adapt to the dynamic changes that take place within a particular watershed. For instance, the TMAL for the Cherry Creek program was set in 1985. It may be revised within the next year, but the program has still not met its chlorophyll a standards. The political process of setting a TMDL is extremely complex, and arrangements should be made so that these cap structures should be incrementally modified based on relevant information that is derived from ongoing monitoring that takes place within a particular watershed.

It was revealed through the case studies that at the present time point sources must share liability with non-point sources or the state may assume liability for non-point source discharge reductions. In order to decrease the institutional problem of liability for water quality trading programs further policy analysis is recommended to estimate the benefits and costs of modifying section 402 of the Clean Water Act (CWA) to allow the transfer of liability from NPDES permittees to non-point source credit generators.

Baselines should be set in order to ascertain the performance of a particular water quality trading program. Consistent and standardized monitoring and verification of the water quality within a watershed as well and individual non-point source sites should be a focus of a trading program. Rules of engagement should be identified early in the program, and these rules should be agreed upon by all stakeholders. Audit plans must be made transparent and implicit in nature to prevent confusion or misrepresentation. This type of monitoring is necessary in order to determine the level of water quality before a program is developed, and it is also necessary in order to define the water quality trading credit that is created from the reduction of nutrients by a specific BMP. Wetlands should be incorporated in order to assess inflow/outflow BMPs because of the way in which they can be designed or restored. Other types of BMPs may not be as easily monitored or verified.

Point sources are regulated and non-point sources are not. Therefore, there is a need for a binding constraint that ties these two types of stakeholders together. A contractual agreement, like the one proposed in the Lower Boise, is a mechanism to achieve this type of constraint. There are transactions costs associated with these types of contracts, but without the transfer of liability from one entity to the other the contract for services is the only mechanism to legally bind the two parties.

If point sources and non-point sources are not able to arrange contracts with each other then a statutory body or the state must take on the responsibility for reducing non-point source nutrient loads. Through an open market contractual agreement, like the Lower Boise, the non-point source is free to choose the BMP that will be most effective at the lowest cost. At present, there are not markets to assess ancillary benefits of wetlands, and in some cases these BMPs may not be the lowest cost technologies to reduce nutrient loads. Therefore it is recommended that institutions (government agencies) better define the ancillary benefits that wetlands provide in order to prescribe or promote their use in water quality trading programs. This is the case in both the Cherry Creek program and the Neuse program. Institutionally, beneficial ratios may be created or other mechanisms could be developed by agencies in order to differentiate the public benefits that wetlands provide versus other BMPs. If wetland assessments cannot be accomplished through the institutional venue then the current lack of information about the ability for wetlands to reduce nutrient loads (as well as other ancillary benefits) will be overlooked by those landowners. These stakeholders may look to "satifice" their rational decisions by minimizing their costs in choosing the abatement technology that reduces nutrient loads, but may not provide all of the positive externalities that are associated with wetlands.

Conditions under which Water Quality Trading is a Viable Option

As previously discussed, certain economic and institutional components are necessary for water quality trading market to operate efficiently and effectively. Within a particular watershed there are conditions that must be met before water quality trading can be determined to be a viable option for nutrient reduction. First, there must be an in-depth understanding of the nutrient pollution problem within the particular basin. It is critical that a water quality assessment be taken throughout the watershed at various points of emphasis. This type of monitoring and verification is absolutely necessary in order to define the baseline conditions and the nutrient reduction cap for a trading program. In fact, this type of broad based monitoring throughout the watershed is necessary in order to verify the effectiveness of any type of nutrient reduction program, with or without markets.

Secondly, there is a need to accurately measure in-stream capacity. If the monitoring and assessment of the watershed determines that there is a significant nutrient problem, then the amount of nutrient input to the basin from point sources and non-point sources must be identified. For instance, there needs to be an understanding of the nutrient load contribution levels from point sources and non-point sources. Unfortunately, nutrient problems are almost always discovered through tragic environmental focusing events like fish kills. The understanding of the specific nutrient problem within a basin coupled with knowledge of where the loads are coming from would provide a great deal of baseline information necessary in order to evaluate water quality trading as a policy option. An optimal cap must be set for critical constituents in relation to baseline and in-stream capacity information, and then ongoing monitoring of the watershed must take place in order to verify changes to water quality.

The next condition necessary for the assessing the option to incorporate water quality trading programs focuses on the number and type of potential buyers and sellers of water quality credits. There must be a certain number of interested point and non-point sources within a basin for a trading program to develop. Point sources are regulated, and may be required to participate in a program to meet certain reductions, but non-point sources must participate in a program in order to decrease significant nutrient load amounts within specific watersheds. A condition that would greatly increase the option to implement a water quality trading program would be the incorporation of some type of binding constraint that linked point sources to non-point sources. As previously discussed, transfer of liability is a significant issue for point sources, and more linkages between these two groups would greatly improve the cooperation of participants within a trading program.

Another condition that is absolutely necessary for an effective water quality trading program to become a viable option is the retrieval and incorporation of better information concerning non-point source nutrient reductions. At the present time, as revealed by the case studies, monitoring and verification is not necessarily a condition of incorporating a trading program. The knowledge of what levels of nutrients are being reduced at non-point source credit sites is essential if the effectiveness of a water quality trading program is to be measured. Better information for all BMPs is necessary, and wetlands should be specifically targeted for further monitoring research because of possibility of measuring direct inflows and outflows.

Lastly, both positive and negative externalities must be considered when assessing the conditions under which water quality trading is a viable option. When a cap is set and enforced, and monitoring takes place throughout a particular basin, there is still the need to assess the benefits and costs that are being generated by the program. For instance, if the nutrient levels for a particular basin are being monitored and the assessments show that the water quality trading is reducing nutrient loads then measurements will suggest that water quality has improved. But, there must also be consideration for the land use, ecological, and hydrological changes that have taken place within the basin to improve the water quality. Planning decisions must be arranged before a water quality trading program is adopted in order to estimate the positive and negative externalities that would evolve from this type of policy option.

APPENDICES

Appendix A

List of Individuals Interviewed

Name	Affiliated Trading Organization/Program/Department
Richard Parachini	Colorado Department of Public Health and Environment
Wilbur Koger	Authority Engineer, Arapahoe County Water and Wastewater Authority
Molly Trujillo	Project Manager, Arapahoe County Water and Wastewater Authority
William Ruzzo	Representative for the Cherry Creek Basin Water Quality Authority
Michelle Wind	Representative for the Cherry Creek Basin Water Quality Authority
Brad Crowder	Economist, U.S. EPA Region 8
Jill Minter	Water Quality Trading Coordinator, U.S. EPA Region 8
Susan Burke	State Water Quality Programs, Idaho Dept. of Environmental Quality
Marti Bridges	TMDL Program Manager, Idaho Dept. of Environmental Quality
Robbin Finch	Water Quality Manager, City of Boise, Idaho
Johanna Bell	Stormwater Program Coordinator, City of Boise, Idaho
Bill Stewart	Environmental Protection Specialist, U.S. EPA Region 10
Leigh Woodruff	TMDL Coordinator, U.S. EPA Region 10
Scott Koberg	Idaho Division of Soil and Water Conservation
David Keil	Representative for the Lower Boise River Watershed Council
Ann Coan	Natural Resources Director, North Carolina Farm Bureau Federation
John Husiman	Environmental Specialist, North Carolina Division of Water Quality
Joseph Rudek	Senior Scientist, Environmental Defense
Steve Coffey	Tar-Pamlico River Coordinator, NCDSWC
Suzanne Klimek	Director of Operations, North Carolina Ecosystem Enhancement Program
Kelly Williams	In-lieu Fee Coordinator, North Carolina Ecosystem Enhancement Program
Natalie Jones	Conservation Reserve Enhancement Program Manager, NCDSWC
Michael Templeton	Environmental Engineer, North Carolina Division of Water Quality

Appendix B

Research Questions for Case Study Interviews
(G. Michael Mikota, 2007)

Research Question:

Is the incorporation of wetlands technology to enhance water quality trading programs economically and politically feasible at the watershed scale?

Specific Information Needed:

Background

- History
- Issues
- Driving force for the establishment of trading system
- Administrative Unit – oversight
- Stakeholders
- Structure of trading system

Track Record

- # of permit trades (point to point and point to non-point)
- Water quality impact
- Costs of program

Impediments

- Market failure issues
    - Info – current available info – is info readily accessible
    - Externalities
    - Initial distribution of credits
    - Transaction costs of creating a trading program

- Institutional failure (arrangements/impediments)
    - Current transactions
    - Setting of a cap (TMDL, TMAL, etc.)
    - Starting point for trading between point sources and non-point sources (baselines)

Case Specific Questions:

Program Background:

(1) What was the motivation for creating this WQT program?

(2) What ecological, cost savings, cost postponement or flexible paths to compliance goals were set during the initial WQT program creation discussions?

Research Questions for Case Study Interviews
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(3) Who are the major stakeholders (participants)? Did these groups belong to certain associations or other cooperative organizations before the WQT program was implemented?

(4) What are the regulatory drivers for the WQT program, and how are they enforced? (TMAL, TMDL, etc.)

(5) Why were wetlands included on the list of BMPs that could be implemented to generate WQT credits? Were wetlands targeted as a primary BMP, or were they included as another option for landowners?

Trade Structure Information with Focus on Wetland Credits (Technical and Economic Performance):

(1) How is a credit determined?

> What types of information do credit buyers need to know from credit producers, and how can watershed groups or wetland owners provide this information? Are credits weighted by performance? Are there any incentives to choose one BMP over another? (cost, performance, other benefits?)

(2) What types of trading ratios or other mechanisms are implemented in order to deal with uncertainty?

> How are credits verified? Are credits needed year round? How do we account for seasonality (dependent upon region or rainfall)? Are spikes allowed in nutrient loads during winter months when BMPs (wetlands) are not functioning as sinks?

(3) How are Equivalency, Additionality, and Accountability (Fang and Easter, 2003) standards used to make the program efficient?

> Equivalency: Temporal, spatial, or chemical differences?
> Additionality: Prevention of double counting by ensuring that a nutrient control activity counts toward only one objective if multiple objectives are met.
> Accountability: What types of monitoring and oversight techniques are used to ensure proper implementation of all program requirements?

Appendix B (continued)

Research Questions for Case Study Interviews
(G. Michael Mikota, 2007)

(4) What is the market structure of the program and why was this structure chosen? (bilateral, clearinghouse, third party brokers, etc.)

> Does the type of market structure play a role in bringing together point sources and non-point sources? What are some of the mechanisms used for trade identification and communication? (education, outreach, third party facilitation, incorporation of evolved watershed groups)

(5) Is there a WQT program watershed administration or governing body? (an authority or local agency?)

> Does this (or would this) type of administrative agency improve the efficiency and effectiveness of the program? Would this type of entity be needed if trade levels increased and more activity took place?

Outcomes with Focus on Wetland Incorporation (Administrative Performance):

(1) What types and volume of trades have occurred, and how many trades have included wetlands?

(2) What have been the administrative costs for implementing and maintaining this program?

(3) Have there been any realized cost savings?

(4) Has the program achieved the goals that were set in the beginning?

Organization (Federal, State, or Local Government) Specific Questions:

(1) Are there better alternatives for improving water quality at lower costs then WQT? (If so, what are they? Do they allow for less uncertainty and risk?)

(2) What types of binding constraints will be necessary to link point sources to non-point sources?

(3) At the present time there is no standard approach to performance monitoring for wetland nutrient reduction. Methods and metrics vary widely. Would a type of standardization assessment technique be beneficial in assessing program requirements?

(4) How are monitoring and verification costs (and other administrative costs) paid for?

Appendix B (continued)

Research Questions for Case Study Interviews
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(5) Please describe any program obstacles that have been significant to date?

(6) What level of non-point source involvement has been realized, and what types of incentives are in place to engage those individuals in trading?

(7) Are wetlands a preferred type of restored BMP for your organization versus others (buffer strip, no-till, etc.)? Based on your knowledge of this watershed, do you think that stakeholders would be inclined to implement a restored wetland BMP versus some other type?

LITERATURE CITED

Adler, R. W. (1994). Reauthorizing the Clean Water Act: Looking to tangible values. *Water Resources Bulletin, 30*(5), 799-807.

Anderson, T. L., & Snyder, P. (1997). *Water Markets: Priming the invisible pump*. Washington, DC: The Cato Institute.

Andersson, F. (1997). Small pollution markets: Tradable permits versus revelation mechanisms. *Journal of Environmental Economics and Management*, *32*(1), 38-50.

Antweiler, R. C., Goolsby, D. A., & Taylor, H. E. (1995). *Nutrients in the Mississippi River*. Contaminants in the Mississippi River, (U.S. Geological Survey Circular 1133). U.S. Department of the Interior.

Baumol, W. (1972). On taxation and control of externalities. *American Economics Review*, *62*, 307-322.

Bavor, H. J., Davies, C. M., & Sakadevan, K. (2001). Stormwater treatment: Do constructed wetlands yield improved pollution management performance over adetention pond system? *Water Science and Technology, 44,* 11-12.

Belmont, M. A., Cantellano, E., Thompson, S., Williamson, M., Sánchez, A., & Metcalfe, C. D. (2004). Treatment of domestic wastewater in a pilot-scale natural treatment system in Central Mexico. *Ecological Engineering, 23*(4-5), 299-311.

Bennett, B., & Gannon, R. (2004). *Updates to stormwater BMP efficiencies, memorandum to local programs, Neuse and Tar-Pamlico stormwater rules*. North Carolina Division of Water Quality. Retrieved from http://h2o.enr.state.nc. us/nps/documents/ BMPNutrientRemovalEfficiencis_001.pdf.

Bowden, W. B. (1987). The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry, 4,* 313-348.

Braskerud, B. C. (2002). Factors affecting nitrogen retention in small constructed wetlands treating agricultural non-point source pollution. *Ecological Engineering, 18*(3), 351-370.

Breetz, H. L., Fisher-Vanden, K., Garzon, L., Jacobs, H., Kroetz, K., & Terry, R. (2004). *Water quality trading and offset initiatives in the U.S.: A comprehensive survey*. Dartmouth College Hanover, NH. Retrieved from hppt://www.dartmouth.edu/ ~kfv/waterqualitytradingdatabase.pdf

Buchanan, J. M., & Tullock, G. (1975). Polluters' profit and political response. *American Economic Review, 65,* 139-47.

Bystrom, O., Andersson, H., & Gren, I. (2000). Economic criteria for using wetlands as nitrogen sinks under uncertainty. *Ecological Economics, 35*(1), 35-45.

Canfield, T., Burden, D., Schubauer-Beigan, J., Sumner, R., Heberling, M., Franz, W., Hurld, K., Minamyer, S., & Mikota, M. (2006, February). *The role of wetlands in water quality trading*. Research Planning Conference of the Environmental Protection Agency, ORD, Chicago, IL.

Cherry Creek Basin Water Quality Authority. (2007). *2006 Annual Report on Activities*. Retrieved from hppt//www.cherrycreekbasin.org/pdf/CherryCk2005 %20AnnRpt.pdf

Chesapeake Bay Commission. (2002). *2002 Annual Report*. Retrieved from hppt://www.chesbay.state.va.us/CBC%20annual%20report%202002.pdf

Coase, R. (1960). The problem of social cost. *Journal of Law and Economics, 3,* 1-44.

Colby, B. (1995). Regulation, imperfect markets, and transaction costs: The elusive quest for efficiency in water allocation. In D. W. Bromley (Ed.), *The handbook of environmental economics*. Cambridge, MA: Blackwell Publishers.

Collinge, R., & Oates, W. (1982). Efficiency in pollution control in the short and long runs: A system of rental emission permits. *The Canadian Journal of Economics, 15,* 346-354.

Cooper, P. F., & Findlater, B. C. (1990, September). Constructed wetlands in water pollution control. *Proceedings of the International Conference on the Use of Constructed Wetlands in Water Pollution Control*. Cambridge, UK.

Costanza, R., d'Arge, R., de Groot, R., et al. (1997). The value of the world's ecosystems and natural capital. *Science, 387*, 253-60.

Christensen, N. L., Bartuska, A. M., Brown, N.J., Carpenter, S., D'Antonio, C., Francis, R., et al. (1996). Report of the Ecological Society of America Committee on Scientific Basis for Ecosystem Management. *Ecological Applications, 65*, 665-691.

Crutchfield, S. (1994). Water quality. *Agricultural Resources and Environmental Indicators,* (Agricultural Handbook Number 705). Washington, DC: USDA, Economic Research Service.

Dales, J. H. (1968). Land, water and ownership. *Canadian Journal of Economics, 1,7*91-804.

Daily, G. C., & Ellison, K. (2002). *The new economy of nature: The quest to make conservation profitable*. Washington, DC: Island Press.

Day, J. W., Ko, J. Y., Rybczyk, J., Sabins, D., Bean, R., Berthelot, G., et al. (2004). The use of wetlands in the Mississippi Delta for wastewater assimilation: A review. *Ocean & Coastal Management, 47*, 671–691.

DeAlessi, M. (2003). Removing muck with markets: A case study on pollutant trading for cleaner water. *Policy Brief to Reason Foundation*. Retrieved from http:// rppi.org/pb24.pdf

DeBusk, W. F. (1999). *Nitrogen cycling in wetlands*. Gainesville, FL: University of Florida, Institute of Food and Agricultural Science.

DeLaney, T. A. (1995). *Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes*. American Farmland Trust.

Dierberg, F. E., DeBusk, T. A., Jackson, S. D., Chimney, M. J., & Pietro, K. (2002). Submerged aquatic vegetation-based treatment wetlands for removing phosphorus from agricultural runoff: Response to hydraulic and nutrient loading. *Water Research, 36*, 1409-1422.

Elias, J. M., Filho, E. S., & Salati, E. (2001). Performance of constructed wetland system for public water supply. *Water Science and Technology, 44*, 11-12.

Environmental Law Institute. (2005). *National forum on synergies between water quality trading and wetland mitigation marking*. Retrieved from http://www2.eli.org/research/ wqt_forum.htm

Environmental Trading Network. (2006). *Notes from the 2nd National Water Quality Trading Conference*. Retrieved from http://www.envtn.org/WQTconf_main.htm

Environomics. (1999). A summary of U.S. effluent trading and offset projects. [A report prepared for Dr. Mahesh Podar, U.S. Environmental Protection Agency. Office of Water]. Retrieved from http://www.epa.gov/owow/watershed/trading/ traenvrn.pdf

Ewel, K. C., & Odum, H. T. (1984). *Cypress swamps*. Gainesville, FL: University of Florida Press.

Faeth, P. (2000). *Fertile ground: Nutrient trading's potential to cost effectively improve water quality*. Washington, DC: World Resources Institute.

Fang, F., & Easter, K. W. (2003, July). *Pollution trading to offset new pollutant loadings: A case study in the Minnesota River Basin*. Presented at American Agricultural Economics Association Annual Meeting, Montreal, Canada.

Fang, F., & K.W. Easter. (2005, June). Pollution trading to offset new pollutant loadings: A case study in the Minnesota River Basin. *Journal of the American Water Resources Association*.

Faulkner, S. P., & Richardson, C. J. (1989). Physical and chemical characteristics of freshwater wetland soils. In *Constructed Wetlands for Wastewater Treatment-Municiple, Industrial and Agricultural*. Chelsea, MI: Lewis Publishers.

Fausold, C. J., & Lilieholm, R. J. (1999). The economic value of open space: A review and synthesis. *Environmental Management, 23*(3), 307-320.

Fisher, J., & Acreman. M. C. (2004). Wetland nutrien removal: A review of the evidence. *Hydrology and Earth System Sciences, 8*(4), 673-685.

Forbes, S. A. (1887). The lake as a microcosm. *Bull. Sci. Assoc.*, Peoria, Illinois. Pp. 77–87. Reprinted in *Illinois Nat. Hist. Survey Bulletin*, *15*(9), 537–550.

Fullerton, D., & Stavins, R. (2000). How economists see the environment. In R. Stavins (Ed.), *Economics of the Environment*. New York: Norton.

Gangadharan, L. (2000). Transactions costs in pollution markets: An empirical study. *Land Economic, 76*(4), 601-614.

Gannon, R. (2003). *Request for approval of local nitrogen strategies Tar-Pamlico agriculture rule: A report to the NC Environmental Management Commission from the Tar Pamlico Basin Oversight Committee*, [WQC Item no. 3 NCEMC Item no. 03-38]. Retrieved from http://h2o.enr.state.nc.us/nps/.

Gannon, R. (2005). *Tar-Pamlico nutrient sensitive waters implementation strategy: Phase III*, [NCEMC Agenda Item No. 0511]. North Carolina Division of Water Quality. Retrieved from http://h2o.enr.state.nc.us/nps/documents/ PhIIIAgreementFinal4-05.pdf

Gathumbi, S. M., Bohlen, P. J., & Graetz, D. A. (2005). Nutrient enrichment of wetland vegetation and sediments in subtropical pastures. *Soil Science Society of America Journal, 69*, 539-548.

Goolsby, D. A., & Battaglin, W. A. (1997). Effects of episodic events on the transport of nutrients to the Gulf of Mexico. *Proceedings of the First Gulf of Mexico Hypoxia Management Conference* (EPA-55-R-97-001, pp. 144-145). December 5-6, 1995. Kenner, LA: Gulf of Mexico Program Office.

Gould, J. R. (1977). Total conditions in the analysis of external effects. *Economic Journal, 87*, 558-563.

Hahn, R., & Hester, G. (1989). Where did all the markets go? An analysis of EPA's emissions trading program. *Yale Journal on Regulation, 16*, 109-153.

Hammer, D. A. (1989). *Constructed wetlands for wastewater treatment municipal, industrial and agricultural*. Chelsea, MI: Lewis Publishers.

Hammer, D. A. (1992). *Creating freshwater wetlands*. Boca Raton, FL: Lewis Publishers.

Harding, D. (1990). Tar Pamlico nutrient sensitive waters management strategy (NWQEP Notes, Number 44). National Water Quality Evaluation Project. North Carolina Agricultural Extension Service.

Heberling, M., Thurston, H., & Mikota, M. (2007, Jan.-Feb.). Incorporating wetlands into water quality trading: Economic considerations. *National Wetlands Newsletter, 29*(1).

Hey, D., Kostel, J., Hurter, A., & Kadlec, R. (2005). *Nutrient farming and traditional removal: An economic comparison*. (Final Report. 03 WSM-6CO). Alexandria,VA: Water Environment Research Foundation.

Hey, D. (2002). Nitrogen farming: Harvesting a different crop. Restoration Ecology, *10*, 1-10.

Hey, D., & Philippi, N. (1999). *A case for wetland restoration*. New York: *John Wiley*.

Horan, R., & Shortle, J. (2005). When two wrongs make a right: Second best point-nonpoint trading ratios. *American Journal of Agricultural Economics, 87*(2), 340-352.

Huang, J., Reneau, R. B., Jr., & Hagedorn, C. (2000). Nitrogen removal in constructed wetlands employed to treat domestic wastewater. *Water Research, 34*(9), 2582-2588.

Huett, D. O., Morris, S. G., Smith, G., & Hunt, N. (2005). Nitrogen and phosphorus removal from plant nursery runoff in vegetated and unvegetated subsurface flow wetlands. *Water Resources, 39*(14), 3259-3272.

Hunt, P. G., & Poach, M. E. (2001). State of the art for animal wastewater treatment in constructed wetlands. *Water Science and Technology, 44*(11-12), 19-25.

Hunt, R. J., Krabbenhoft, D. P., & Anderson, M. P. (1996). Groundwater inflow measurments in wetland systems. *Water Resources Research, 32*(3), 495-507.

Idaho Department of Environmental Quality. (2001). Lower Boise effluent trading demonstration project, (1st Annual Status Report). Retrieved from http://www.deq. state.id.us/water/data_reports/surface_water/tmdls/bois river_lower/boise_ river_lower_effluent_status01.pdf

Idaho Department of Environmental Quality. (2005). *Phosphorus total daily maximum load for Lower Boise River*, (Lower Boise River Water Quality Plan). Retrieved from http://www.lbrwqp.boise.id.us/phos_tmdl.htm

Jaksch, J. (2000). Appendix D, Rahr Malting. In R. L. Kerr, S. J. Anderson, & J. Jaksch *Nine Case Studies, Appendices (A-I. Kerr, Greiner, Anderson & April, & Battelle)*. Pacific Northwest Division. Retrieved from http://www.napawash.org/ pc_economy_environment/learning_texts.html

Jing, S., Lin, Y., Wang, T., & Lee, D. (2002). Microcosm wetlands for wastewater treatment with different hydraulic loading rate and macrophytes. *Journal of Environmental Quality, 31*, 690-696.

Johnston, C. A. (1991). Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Environmental Control, 12*, 491-565.

Kadlec, R. H., & Knight, R. L. (1996). *Treatment wetlands*. Boca Raton, FL: Lewis Publishers.

Kashmanian, R. M., Podar, M. K., et al. (1995). The use and impact of intraplant trading in the iron and steel industry to reduce water pollution. *The Environmental Professional*, *17*(4), 309-315.

Kerr, R. L., Anderson, S. J., & Jaksch, J. (2000). *Cross cutting analysis of trading programs: Nine case studies*. Prepared for the National Academy of Public Administration. Retrieved from http://www.napawash.org/ pc_economy_environment/epa0601.pdf.

Kieser, M. S. (2003, September). *Market approaches for environmental improvements: Opportunities in the Great Lakes Region and beyond*. Presented at the Forum on Environmental Markets in the Potomac Basin hosted by Friends of the Potomac and the Environmental Finance Center. Washington, DC.

King, D. M. (2005). Crunch time for water quality trading. *Choices Magazine*, *20*(1), 71-75.

King, D. M., & Kuch, P. J. (2003). Will nutrient trading ever work? An assessment of supply problems, demand problems and institutional obstacles. In *The Environmental Law Reporter*. Washington, DC: Environmental Law Institute.

Klomjek, P., & Nitisoravut, S. (2005). Constructed treatment wetland: A study of eight plant species under saline conditions. *Chemosphere, 58*(5), 585-593.

Klopatek, J. M. (1978). Nutrient dynamics of freshwater riverine marshes and the role of emergent macrophytes. In *Freshwater Wetlands: Ecological Processes and Management Potential*. New York: Academic Press.

Kneese, A. V., & Schultze, C. L. (1975). *Pollution, prices and public policy*. Washington, DC: Brookings Institute.

Knight, R. L., Payne, V. W. E., Borer, R. E., Clarke, R. A., & Pries, J. H. (2000). Constructed wetlands for livestock wastewater management. *Ecological Engineering*, *15*(1), 41-55.

Knopman, D. S., & Smith, R. A. (1993). 20 Years of the Clean Water Act. *Environment, 35*(1), 17-20, 35-51.

Kydland, F. E., & Prescott, E. C. (1977). Rules rather than discretion: The inconsistency of optimal plans. *Journal of Political Economy, 85*(3), 473-491.

Lee, G. F., Bentley, E., & Amundson, R. (1975). Effects of marshes on water quality. In *Coupling of Land and Water Systems*. New York: Springer-Verlag.

Lefor, M. W., & Kennard, W. C. (1977). *Inland wetland definitions* (Report No. 28). Storrs, CT: University of Connecticut, Institute of Water Resources.

Leitch, J. A., & Fridgen, P. (1998). Functions and values of prairie wetlands: Economic realities. *Great Plains Research, 8*(1), 157-168.

Lewis, W. M. (2001). *Wetlands Explained: Wetland science, policy and politics in America*. New York: Oxford University Press.

Lin, Y-F., Jing, S-R., Wang, T-W., & Lee, D-Y. (2002). Effects of macrophytes and external carbon sources on nitrate removal from groundwater in constructed wetlands. *Environmental Pollution, 119*(3), 413-420.

Luederitz, V., Eckert, E., Lange-Weber, M., Lange, A., & Gersberg, R. M. (2001). Nutrient removal efficiency and resource economics of vertical flow and horizontal flow constructed wetlands. *Ecological Engineering, 18*(2), 157-171.

Mayo, A. W., & Bigambo, T. (2005). Nitrogen transformation in horizontal subsurface flow constructed wetlands I: Model development. *Physics andChemistry of the Earth, 30*(11), 658-667.

McCarthy, M., Dodd, R., Tippett, J. M., & Harding, D. (1996, June). Cost-effectiveness and targeting of agricultural BMPs for the Tar-Pamlico nutrient trading program. In the *Proceedings of the Watershed 1996 Moving Ahead Together Technical Conference and Exposition*. Baltimore, MD. Retrieved from http://www.epa. gov/owowwtr1/watershed/ Proceed/mccarthy.html.

McKnight, S. K. (1992). Transplanted seed bank response to drawdown time in a created wetland in East Texas. *Wetlands, 14*, 159-165.

Michigan Department of Environmental Quality (DEQ). (2002). *Water quality trading rules executive summary*. Retrieved from http://www.michigan.gov/ printerFriendly/0,1687,7135-3313_3682_3719-14329--,00.html .

Mitsch, W. J., & Gosselink, J. G. (2000). *Wetlands*. (3rd ed.). New York: Wiley & Sons.

Mitsch, W. J., & Wilson, R. F. (1996). Improving the success of wetland creation and restoration with know-how, time and self-design. *Ecological Applications, 6*(1), 77-83.

Mlot, C. (1997, September 27). The rise in toxic tides. *Science News*, 202-204.

Mueller, D. K., & Helsel, D. R. (1996). *Nutrients in the Nation's Water – Too Much of a Good Thing?* (U.S. Geological Survey Circular 1136). U.S. Department of Interior.

Neuse River Foundation. (2007). Retrieved from http://www.neuseriver.org/.

North Carolina Department of Environment and Natural Resources (NCDENR). (1998). *Neuse River Basinwide water quality plan*. Retrieved from http://h2o.enr.state.nc.us/su/Neuse_NSW_Management_Strategy.htm

North Carolina Riverkeepers Watershed Alliance (NCRWA). (2007). Retrieved from http://www.riverlaw.us/fishkills.html

National Wetlands Policy Forum (NWPF). (1987). Protecting America's wetlands: an action agenda. Washington, DC: The Conservation Foundation.

Odum, W. E. (1989). Predicting ecosystem development following creation and restoration of wetlands. In J. Zelazny & J. S. Feierabend (Eds.), *Wetlands: Increasing Our Wetland Resources* (pp. 67-70). Washington, DC: National Wildlife Federation.

Pigou, A. C. (1920). *The Economics of Welfare*. London: Macmillan.

Poach, M. E., Hunt, P. G., Vanotti, M. B., Stone, K. C., Matheny, T. A., Johnson, M. H., & Sadler, E. J. (2003). Improved nitrogen treatment by constructed wetlands receiving partially nitrified liquid swine manure. *Ecological Engineering, 20*(2), 183-197.

Portney, P. R., & Stavins, R. N. (2000). *Public policies for environmental protection*. Washington: RFF Press.

Rabalais, N. N., & Nixon, S.W. (2002). Nutrient over-enrichment of the coastal zone. *Estuaries, 25*, 497.

Rabalais, N. N., Turner, R. E., & Wiseman, W. J., Jr. (1997). Hypoxia in the Northern Gulf of Mexico: Past, present and future (EPA-55-R-97-001, pp. 25-36). *Proceedings of First Gulf of Mexico Hypoxia Management Conference* December 5-6, 1995. Kenner, LA. Gulf of Mexico Program Office.

Raffini, E., Hall, L., Hurld, K., Hough, P., Robertson, M., & Mikota, M. (2005, July). *National forum on synergies between water quality trading and wetlands mitigation banking*. Washington, DC: Carnegie Endowment for International Peace Conference Center.

Raffini, E., & Robertson, M. (2005). Water quality trading: What can we learn from 10 years of wetland mitigation banking? *National Wetlands Newsletter, 27*(4), 3.

Reed, S., & Brown, R. (1991). Constructed *wetland design the second generation*. 64th WPCF Annual Conference & Exposition.

Reinartz, J. A., & Warne, E.L. (1993). Development of vegetation in small created wetlands in southeastern Wisconsin. *Wetlands, 13*(3), 153–164.

Robertson, M., & Mikota, M. (2007). Water quality trading and wetland mitigation banking: Solving different problems; Taking different paths? *National Wetlands Newsletter, 29*(2).

Rose-Ackerman, S. (1973) Effluent Charges: A Critique. *The Canadian Journal of Economics, 6*, 412-528.

Ross and Associates Environmental Consulting, Ltd. (2000). *Summary of participant recommendations for trading framework.* Lower Boise River Effluent Trading Demonstration Project. Retrieved from http://www.deq.state.id.us/water/data_reports/surface_water/tmdls/bois_river_lower/boise_river_lower_effluent_report.pdf

Sakadevan, K., & Bavor, H. J. (1999). Nutrient removal mechanisms in constructed wetlands and sustainable water management. *Water Science Technology, 40*, 121-128.

Schulz, C., Gelbrecht, J., & Rennert, B. (2003). Treatment of rainbow trout farm effluents in constructed wetland with emergent plants and subsurface horizontal water flow. *Aquaculture, 217*(1), 207-221.

Shaler, N. S. (1890). *General Account of the freshwater morasses of the UnitedStates, with a description of the Dismal Swamp District of Virginia and North Carolina* (U.S. Geological Survey, 10th Annual Report 1888-1889, pp. 255-339). Washington, DC.

Simon, H. A. (1957). *Models of Man*. New York: Wiley.

Smayda, T. J. (1989). Primary production and the global epidemic of phytoplankton blooms in the sea: A linkage? *Coastal and Estuarine Studies*, 35.

Stavins, R. (1995). Transaction costs and tradable permits. *Journal of Environmental Economics and Management, 29*, 133-148.

Stavins, R. (1998). What can we learn from the grand policy experiment? Lessons from SO2 allowance trading. *The Journal of Economic Perspectives*, *12*(3), 69-88.

Stedman, S. M., & Hanson, J. (2007). *Wetlands, fisheries & economics in the South Atlantic coastal states*. NOAA Wetlands Web Site: http://www.nmfs.noaa.gov/ abitat/habitatconservation/publications/bitatconections/num2.htm

Steel, J. (1990). *Albemarle-Pamlico estuarine system technical analysis of status and trends* (Report 90-01). Albemarle-Pamlico Estuarine Study.

Stephenson, K., & Shabman, L. (2001). The trouble with implementing TMDLs. *Regulation, 24*(1), 28-32.

Szabo, A., Osztoics, A., & Szilagyi, F. (2001). Natural wastewater treatment in Hungary. *Water Sci Technology, 44*(11-12), 331-338.

Tar-Pamlico Association Meeting. (2007, April). *Meeting Minutes.*

Templeton, M. (2007). *Pollutant trading in North Carolina's river basins: Tar-Pamlico and Neuse river basins*. Presentation to the USDA-CSREES National Water Conference.

Tietenberg, T. (1985). Emissions Trading: An Exercise in Reforming Pollution Policy. Washington, DC: *Resources for the Future.*

Savannah, G. A., Tillman, D., Wedline, D., & Knops, J. (1996). Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature, 379*, 718-720.

U.S. Environmental Protection Agency. (2001). *The U.S. experience with economic incentives for protecting the environment*. Washington, DC: Office of Policy, Economics and Innovation.

U.S. Environmental Protection Agency. (2003). *U.S. water quality trading policy*. Washington, DC: Office of Water. Web site: http://www.epa.gov/ owow/watershed/trading/tradingpolicy.html

U.S. Environmental Protection Agency. (2004). *Water quality trading assessment handbook* (EPA-841-B-04-001). Washington, DC: Office of Water.

U.S. Environmental Protection Agency. (2007). *EPA trading information*. Web site: http://www.epa.gov/ owow/watershed/trading/tradelinks.html

U.S. Environmental Protection Agency and U.S. Army Corps of Engineers. (1990). *Memorandum of agreement between the Environmental Protection Agency and the Department of the Army concerning the determination of mitigation under the Clean Water Act Section 404(b)(1) Guidelines*. (Federal Register 55: 9210-13). Washington, DC: Author.

Vitousek, P. M., et al. (1997). Human alterations of the global nitrogen cycle: Sources and consequences. *Ecological Applications*.

Wetzel, R. G. (2001). Fundamental processes within natural and constructed wetland ecosystems: Short-term versus long-term objectives. *Water Science and Technology*, *44*, 11-12.

White, J. R., & Reddy, K. R. (2003). Nitrification and denitrification rates of Everglades wetland soil along a phosphorus-impacted gradient. *Journal of Environmental Quality*, *32*, 2436-2443.

Whitehead, J. C. (1992). Ex ante willingness to pay with supply and demand uncertainty: Implications for valuing a marine turtle protection programme. *Applied Economics*, *24*, 981–988.

Williamson, O. E. (1985). *The economic institutions of capitalism*. New York: Free Press.

Williamson, O. E. (1996). *The mechanisms of governance*. New York: Oxford.

Woodward, R. T., et al. (2002). The structure and practice of water quality trading markets. *Journal of the American Water Resources Association*, *38*(4), 967-979.

Woodwell, G. M., & Whitney, D. E. (1977). Flax Pond ecosystem study:Exchange of phosphorus between a salt marsh and the coastal waters of Long Island Sound. *Marine Biology*, *41*, 1-6.

Wossink, A, & Hunt, B. (2003). The economics of structural stormwater BMPs in North Carolina, (WRRI Research Report Number 344). Web site: http://www.ag-econ.ncsu.edu/faculty/wossink/outreach.html

Zwick, D., & Benstock, M. (1971). *Water Wasteland: Ralph Nader's study group report on water pollution*. New York: Grossman, 264-84.