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I am submitting herewith a thesis written by David Carlos Roberts entitled "Identifying and Analyzing Potential Watershed-Based Water Quality Trading Programs." I have examined the final paper copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Agricultural Economics.

Christopher Clark, Major Professor

We have read this thesis
and recommend its acceptance:

Accepted for the Council:

Vice Chancellor and
Dean of Graduate Studies

IDENTIFYING AND ANALYZING POTENTIAL WATERSHED-BASED WATER QUALITY TRADING PROGRAMS

A Thesis Presented
For the Master of
Science Degree
The University of Tennessee, Knoxville

David Carlos Roberts
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Dedication

I dedicate this work to all my kith and kin, and to great rivers and small streams.

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Abstract

Over thirty years since the passage of the Clean Water Act, much of the nation's rivers and streams fail to meet water quality standards. Pollution from nonpoint sources is increasingly responsible for these failures. Existing regulatory approaches may not be capable of meaningfully reducing water quality impairments due to their inability to control emissions from nonpoint sources. Water quality trading may provide a cost-effective solution to many of the persistent water quality impairments caused by nutrients and other oxygen demanding pollutants. However, trading will require the presence of market participants – both buyers and sellers of pollution reduction credits – in order to be successful. Thus, the first paper in this thesis analyzes Tennessee's watersheds to determine which have the conditions necessary to support a successful water quality trading market.

The second paper focuses on the reduction of pollution from nonpoint sources. Specifically, it estimates the social costs associated with a mandatory riparian grassed buffer strip for agricultural lands in the Harpeth River watershed in Middle Tennessee. In addition, this paper approximates a supply curve for buffer strips in this watershed. This latter result constitutes an important step towards understanding the supply of nutrient emissions reduction credits. The next step would be to estimate the reduction in nutrient runoff associated with these buffer strips.

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Chapter I

A Brief Introduction to Water Quality Problems and Solutions

Some History & Background

A fire on the surface of the Cuyahoga River on June 22, 1969 caught the attention of American politicians and citizens alike. Fires began to plague the Cuyahoga River as early as 1939, due to high levels of pollution from oil and grease (Greenhalgh and Faeth); however, the fire of 1969 was the first to draw National attention, thanks to the initial article published in *Time Magazine*'s brand new Environment section, which graphically described the Cuyahoga as "a river that oozes rather than flows." Being contaminated by substances such as oil, grease, untreated sewage and various industrial wastes, this river was an extreme and very visible indicator of the types of water quality problems that were pervasive during the time period, and its burning spurred legislators to action. Shortly after the Cuyahoga came to national attention, the United States passed the Federal Water Pollution Control Act Amendments of 1972, commonly known as the Clean Water Act (CWA), to curb the effects of human activities on water quality.

The CWA is still in force in 2005. Among the goals listed in the CWA are the "restor[ation] and maint[enance of] the chemical, physical, and biological integrity of the Nation's waters," the elimination of "the discharge of pollutants into navigable water . . . by 1985," and the prohibition of the discharge of any "toxic pollutants in toxic amounts."[1] The means specified in the act to aid in the attainment of the goals include a

[1] 33 U.S.C. § 1251(a), (a)1, & (a)3

permitting system, which came to be known as the National Pollutant Discharge Elimination System (NPDES), to be administered by state agencies to regulate point source (PS) emissions (e.g., discharges through pipes or ditches) to the nation's waters (US Environmental Protection Agency Office of Water, 2001). The CWA also states that the "national policy" is that measures be designed and implemented in a timely manner to curb the nonpoint source (NPS) contribution (including agricultural land runoff and other such diffuse sources) to pollutant loadings in waterways, though no specific means of achieving this goal are laid out in the legislation.[2] This "national policy" established by Congress, though not explicit in methodology, recognizes the need for a combination of PS and NPS pollutant load reductions to meet the ambient water quality goals proposed by the CWA.

Point Source Issues

As required by the CWA, all PSs must have an NPDES permit to continue legally discharging into U.S. waterways (US Environmental Protection Agency Office of Water, 2001). These permits place strict limits on the quantity of pollutants or the allowable concentration in polluters' effluents, and are normally based on available technology, except in instances where technology-based standards are insufficient to attain applicable water quality standards (Ryan). Technology-based standards have not been fruitless in bringing about decreases in the level of pollutant discharges. Many case studies, such as that of the Blue Plains Treatment Plant discharging into the Potomac Estuary, have shown improved water quality as a direct result of implementation of treatment plant

[2] 33 U.S.C. § 1251(a)7

upgrades, and monitoring data at several hundred measurement stations throughout the nation have shown estimated 23% reductions in pollutants such as phosphorous and fecal bacteria between 1974 and 1981 (Knopman and Smith). However, such improvements have not been without cost; between 1974 and 1994, federal and state/local governments invested \$96 billion and \$117 billion, respectively, in new and upgraded municipal wastewater treatment plants to meet standards imposed by NPDES permitting (Association of Metropolitan Sewerage Agencies and the Water Environment Federation). Yet, implementation of the water treatment standards for PSs has not completely achieved the goals of the CWA. It has, on the other hand, substantially depleted the range of cost-effective options remaining for the control of PS discharges. Estimates from US EPA and AMSA put the funding needs of municipal treatment works over the next 20 years at approximately \$139.5 billion and \$330 billion, respectively, to fill currently unmet needs for water infrastructure nationwide (Association of Metropolitan Sewerage Agencies and the Water Environment Federation). Though these cost estimates vary widely, they are indicative of the great expense of improving water quality through regulation of PS polluters alone.

Nonpoint Source Reductions: A More Cost-Effective Way?

The good news is that it is not necessary to attain ambient-based water quality standards solely through increasingly strict PS regulation. Frequently, PS and NPS polluters emit the same or similar pollutants. NPS pollution from agriculture, in contrast to PS pollution, does not require permitting and is not regulated by law, except in a few instances.

Several trends have caused NPS pollution of the nation's waterways to become a quantitatively greater contributing factor to water quality impairments than point source emissions (US Environmental Protection Agency Office of Water, 2002). Population increase, for example, has promoted expansion of urban development as well as more intensive agricultural practices to meet the increasing food needs of a growing American public. NPS runoff is a national issue. In fact, when the states and territories performed a recent survey of the nation's waters, they found that 39% of the waters assessed were impaired for one or more uses, and EPA posits that the largest single contributor to water quality impairment is NPS runoff from agricultural lands (US Environmental Protection Agency Office of Water, 2002).

Since agricultural runoff is typically not regulated or monitored, adoption of best management practices (BMPs), which reduce soil erosion and runoff of wastes, among NPS polluters has, for the most part, been voluntary. Because of the voluntary nature of pollution abatement in the agricultural sector and 30 years of increasingly strict PS regulation, there is reason to suppose that a cost differential may exist between PSs and NPSs with regard to abating pollutants common to both groups, with NPS reductions being the less costly route for society as a whole. Because reduction of agricultural runoff has not traditionally been mandated by law, many relatively inexpensive options for pollution control may still remain for most agricultural producers; however, enforcement and monitoring of any pollution control standards for agriculture could be a costly administrative problem due to the sheer number of individuals involved in the creation of NPS pollution (Russell and Clark). The question, then, is whether a policy instrument exists that can assist in achieving scientifically predetermined pollutant reductions in a cost-effective manner.

Water Quality Trading: Is this the Policy Instrument We Seek?

One method that could potentially reduce total abatement costs by taking advantage of the cost differential between PS and NPS polluters who emit the same contaminants is known as water quality trading (WQT), which is a market-based approach to pollution control in which one polluter can purchase a marketable emissions permit from another. The purchaser can increase emissions by the amount specified in the permit, while the seller must decrease emissions by an equivalent amount. Absent transactions costs or other market imperfections, trading would theoretically occur until the marginal abatement costs for all polluters in the market were equalized, yet the total emissions of the pollutant traded would not increase above limits designed to preserve or restore ambient water quality. Emissions could be limited to levels consistent with the goals of the CWA by restricting the number of government-issued permits.

As with any tradable permit market, a WQT market would need a market driver to encourage economic activity. The CWA empowers EPA to establish a system of Total Maximum Daily Loads (TMDLs), which could function as such a driver by making additional pollution reduction necessary in areas where waters have been identified as not meeting ambient water quality standards even after technology-based controls.[3] The CWA sates that "[s]uch load shall be established at a level necessary to implement the applicable water quality standards with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality;"[4] thus, the TMDL system is an ambient-based standard, protecting water quality based on the amount of the pollutant the receiving water body can assimilate, rather than the level to which PSs could "reasonably" be expected to lower their emissions. EPA describes TMDLs as "pollutant budgets" in which the total allowable discharge is allocated across PSs and NPSs in the area (US Environmental Protection Agency Region 10). Assuming no transactions costs, once TMDLs are created for areas and waste loads are allocated between sources, allowing entities to trade those allocations amongst themselves would permit the sources to achieve the necessary abatement at a lower cost.

Although many economists and policy-makers extol the savings potential of marketable permit systems, acceptance has been slow in regard to WQT, and only limited pilot programs are in place in the United States. EPA has expressed interest in and support for WQT. EPA has formally endorsed WQT since 1996 in the *Draft Framework for Watershed-Based Trading* (US Environmental Protection Agency Office of Water, 1996). This document supports "offset" trading, rather than a watershed-based "cap and trade" program. Under an offset program, a polluter under regulatory pressure to further reduce emissions (such as a PS whose emissions cap has been lowered) has the option of

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[3] 33 U.S.C. §1313(1)(A)
[4] 33 U.S.C. §1313(1)(C)

paying another polluter (a PS or NPS) to meet abatement requirements. In contrast, a cap and trade system would require that tradable effluent permits be issued to all polluters in a watershed, thus allocating specific abatement responsibilities to individual PSs and NPSs, in accordance with the abatement required by applicable TMDLs, and then allowing sources to buy and sell permits.

Overview of Subsequent Chapters

This chapter has presented a brief introduction to water quality issues in general, and WQT as a potential part of the solution.

Chapter II is a review of the literature regarding tradable emissions rights and the economic theory on which these systems are based. Also included in chapter II is a discussion of policy concerns, as well as operational programs that allow trading of discharge responsibilities between polluters.

Chapter III develops a methodology for comparing distinct watershed areas based on their suitability for WQT as determined by various factors, including location of PS and NPS polluters with respect to water quality impairments, and the extent of the impairments to which they contribute.

Chapter IV describes a methodology for plotting some approximation of a supply curve for agricultural riparian buffers, which is a BMP that aids in the reduction of nutrient pollution in agricultural runoff. With some extensive water quality modeling, this process could be expanded to approximate a supply curve for NPS nutrient reductions.

Chapter V is a summary that includes some overall conclusory remarks with regard to the aforementioned chapters.

Chapter II

A Review of the Literature

Introduction

This chapter will provide a review of literature related to tradable emissions rights and WQT specifically. Herein, some of the theory behind trading of discharges is reviewed, a few issues of program design are discussed, and some notable WQT programs and their successes are summarized.

The Conceptual Basis for Trading Pollution Allowances

According to Coase (1960), property rights (not just property itself) may be appropriately viewed as factors of production. He effectively shows that market transactions can (absent transactions costs) optimally allocate the use of resources between two economic agents that reciprocally harm one another (i.e., one operator must incur some cost in order to avoid imposing some cost on the other). In such a case the owners of the two enterprises may enter into bilateral negotiations to achieve socially optimal use of their combined resources. Such negotiations are relatively cost free when few economic agents are involved; however, in the instance of a factory that produces smoke, Coase hypothesizes that transactions costs would be prohibitively high due to the vast number and diversity of individuals and organizations adversely affected. High transactions costs would, in turn, bring about a suboptimal allocation of resources since they would prevent some negotiations from proceeding. In such cases, Coase suggests government regulation may be preferable to negotiations because it has the potential to effectuate reductions in discharges even when socially beneficial trades would be precluded by transactions costs; however, such regulation may come at high administrative costs to government as well as great control costs to polluters. Furthermore, since allowable pollutant discharges would be determined by the government to protect environmental quality based on imperfect data, rather than through negotiations between the polluter and all adversely affected groups, there is no reason to believe that social welfare would be optimized. Thus, due to the existence of transactions costs (or administrative costs) and imperfect information, socially optimal solutions may be unattainable in regional and global pollution control problems.

Hung and Shaw note that, in the case of pollution, social efficiency (optimality) is an impractical standard because it poses a heavy information burden for the regulatory agency. They state that in the case of pollution control, the standard of cost-effectiveness, or meeting a regulatory environmental standard at least cost, is the generally accepted policy goal for economists. WQT is a hybrid system. It includes elements of government regulation, which causes polluters to reciprocally harm each other by limiting aggregate use of water for the discharge of residuals. The system also includes bilateral negotiations between these parties that are reciprocally harmful to one another, thus reducing transactions costs by reducing the number of entities involved in the negotiation process.

Issues of Program Design for WQT

Hung and Shaw's study proposes a trading system based on predetermined trading ratios to minimize transactions costs. Under this type of system, a watershed would be divided into zones in which the discharge of one unit of a pollutant would have approximately the same environmental impact. Each zone would have a maximum discharge cap imposed by quotas that could be traded at a ratio of 1:1 within the zone. The inter-zone trading ratios would be determined by the rate at which the pollutant is transferred from one zone to another, and would be published by the regulating agency to improve information and reduce transactions costs. As an added benefit, the zonal system would eliminate incentives for an upstream polluter to purchase reductions from a downstream polluter, as downstream reduction would not affect upstream loadings. The elimination of this incentive could reduce the potential for inter-zone pollution hotspots; however, intra-zone trading could cause the occurrence of hot spots within individual zones, unless zones were very small so that the environmental impacts within the zone were approximately equal. This begs the question: How equal is "approximately equal"? Or, how small would the zones have to be? If the regulatory agency bears the cost of information provision to reduce transactions costs, a trade-off between administrative and transactions costs becomes evident as the number of zones multiplies.

While Hung and Shaw's design is specific to PSs, the authors state that it could be expanded to include NPS polluters as well. Allowing trading between NPS and PS polluters, many economists agree, may provide the greatest potential for abatement cost savings (Horan and Shortle, Faeth); however, this introduces the issue of NPS:PS trading ratios. There is an extensive literature discussing such ratios, as well as their purposes and effects. Horan and Shortle, for example, note that risk is often the major focus of debate concerning trading ratios, and that the uncertain effectiveness of NPS control methods (i.e. NPS control is less dependable, or more risky) is often used as a justification for setting ratios at levels greater than unity in existing programs. However, they posit that this logic only accounts for part of the risk, as NPS pollution, which is dependent on weather events and input use, is extremely stochastic in the first place; thus, controlling NPS pollution may create a greater stability in pollution levels, reducing the risk of high pollutant loadings during adverse weather events. They propose that the real reason for trading ratios above unity is the current regulatory system, under which agricultural NPS polluters have been effectively granted the right to pollute. In existing trading programs of the offset type endorsed by EPA, only PS polluters are monitored and regulated by the NPDES permitting program. If trading program authorities could allocate permits to all sources (including agricultural NPS polluters), then NPS:PS trading ratios less than one would be acceptable because this would encourage greater abatement of risky, stochastic NPS pollution.

Woodward points out that, while the most obvious purpose of trading ratios is that of equating the impacts of different polluters to insure that an increase in emissions by one is at least offset by the other, there are other reasons for employing trading ratios. Many trading programs seek to use trading ratios to provide a net environmental benefit, in addition to an offset credit and a margin of safety to account for uncertain reduction methods (McGinnis). While economists tend to attack the pollution reduction problem as a cost minimization problem (minimizing the cost of meeting an environmental quality standard), Woodward suggests politicians and other policy makers tend to view the problem from the dual perspective – maximizing environmental quality subject to some cost constraint that is politically acceptable. Thus, regulators and environmentalists may favor use of trading ratios for their potential to create environmental gains beyond the requirements of the ambient water quality standards. However, Woodward points out that while the cost minimization perspective completely discounts the value of pollution reductions below the regulatory standard (which does not necessarily reflect public opinion about such reductions), the environmental quality maximization problem ignores the market inefficiencies introduced by non-unitary trading ratios.

Stephenson and Shabman present a system similar to WQT known as a discharger association. This type of program design has been implemented in the Tar-Pamlico and Neuse River Basins in North Carolina. The discharger association is a group of firms which have a collective abatement responsibility assigned by a regulatory agency, but the member firms are allowed to allocate the total responsibility among themselves to achieve the allowable discharge level at the lowest possible cost. Using this particular organizational structure to achieve environmental standards may reduce transactions costs, thus allowing a more cost-effective final distribution of abatement responsibilities among the member firms. The concept of discharger associations harkens back to Coase's "The Nature of the Firm," (1937) wherein he posits that firms exist because it is sometimes more cost-effective to coordinate economic activities within one organization than it is to deal with transactions costs in the market place. The problem with this particular organizational form, at least for pollution control, is that it does not seem to account for the geospatially differential impacts of member firms through any mechanism such as a trading ratio.

Another issue for WQT program design is how to trade NPS loadings for PS emissions, given NPS loadings cannot be regularly or accurately measured. Since NPS reduction credits for most BMPs can only be estimated based on modeling, a large amount of information about land use and agricultural practices surrounding proposed BMPs will be required to accurately predict the reduction produced by the implementation of a given BMP. For example, if an agricultural producer is using a responsible nutrient management plan, installation of a riparian buffer strip on the same land would not create the same load reduction that it would if the producer were to liberally apply fertilizer up to the very edge of the buffer strip. A given BMP may be worth more or less, in terms of nutrient reduction credits, depending on its interaction with surrounding land uses and previously implemented BMPs.

A separate but related issue is the law of unintended consequences; installation of one conservation practice may lead to unpredictable changes in production practices elsewhere on the farm or on other farms, which may be either beneficial or deleterious to environmental quality. Allowing farmers to make money by creating pollution reduction credits may provide perverse incentives for farmers who have already implemented best management practices. Some might even go so far as to farm land that they otherwise would not, just so they could later offer to take the land out of production to create credits for which they would be paid by a PS polluter. Establishing historical base loads for NPS polluters based on historical land use patterns is one way to assuage the potential effects of perverse incentives, so that runoff-reducing practices implemented prior to a base year cannot create tradable credits, and must be maintained if the NPS polluter whishes to create tradable credits after the base year. Another means that might decrease the effect of perverse incentives for agricultural producers is to require that NPS polluters contribute some water quality benefit before they can create any salable discharge credits. Versions of both of these policies are included in the trading rules for the Lower Boise River Effluent Trading Demonstration Project (Ross & Associates Environmental Consulting, Ltd.).

Best Management Practices

Best management practices (BMPs) are practices designed to reduce NPS loadings. This section will review the literature regarding the effectiveness and cost-effectiveness of a few BMPs that are often considered viable means of reducing the nutrient content in agricultural runoff.

Lee, Isenhart and Schultz performed a study regarding the effectiveness of riparian buffer strips in reducing runoff and nutrient content from corn and soybeans in Iowa. They measured nutrient load reductions effectuated by established multi-species riparian buffers composed of switchgrass and switchgrass with woody plants. Such a buffer can dam runoff up to 10 cm high. This slowing of the water permits greater time for runoff to infiltrate the soil and for some nutrients and sediments to settle out of the remaining runoff. Their results showed that the 23 foot switchgrass buffer removed 95%

of the sediment, 80% of the total nitrogen, and 78% of the total Phosphorous. Thus, the authors show that an established, well-planned riparian buffer can effectively remove much of the nutrient content of agricultural runoff.

Rein found that by establishing perennial vegetative buffer strips, strawberry farmers in the Elkhorn Slough watershed of California could derive an estimated net benefit of $1,488 in the first year and $6,171 over a five year period. Benefits accrue from reduced pesticide and fertilizer use and reduced gully erosion, as well as other sources. These results suggest that farmers of certain crops on certain soils would be better off establishing buffer strips than not; however, these results will not hold for all watersheds and all crops.

Norwood and Chvosta found that BMPs do not always have their intended consequences; in fact, they may sometimes have adverse effects on water quality. Their findings are relevant to swine farms in two North Carolina counties, and are based on the results of a survey of 85 producers regarding their likely manure-management practices in the face of phosphorous-based manure application regulations. Manure contains a higher phosphorous to nitrogen ratio than is required for plant growth, but because of the cost and difficulty of transporting it, swine farmers tend to over-apply manure on cropland or forage surrounding their operations. This leads to phosphorous buildup in the soil and water quality problems caused by the associated runoff. The study found that many hog producers would clear land adjacent to their existing operations so that they would be able to apply based on phosphorous requirements of the soil and crop. This being the case, more chemical nitrogen fertilizer would be required for the expanded area, possibly leading to greater nitrogen loadings to surface waters, which could offset the reductions in phosphorous loadings. Some respondents to the survey indicated they would reduce the size of their herd or leave swine production altogether. The results of the study indicate that water quality would be improved unambiguously only if hog production decreased by 25% or more. Thus, the environmental improvements would be due to decreased production, rather than the BMP itself. These results are not necessarily applicable in all areas, but they show that regulation of farming practices can have unintended and adverse consequences.

Some Notable WQT Programs

Several WQT programs have been established in the United States. Some are considered successful, while others have not produced the expected results, at least in terms of trading activity. Appropriate program design is crucial to the success of tradable emissions permit systems; thus, in developing new projects, policy-makers should consider what trading rules have previously proved to be conducive to trade. Two exemplary WQT projects currently in some stage of implementation are the Tar-Pamlico Nutrient Reduction Trading Program (TPNRT) and the Lower Boise River Effluent Trading Demonstration Project (LBRET).

The Tar-Pamlico Nutrient Reduction Trading Program

The TPNRT Program in North Carolina was established in response to the 1989 designation of The Tar-Pamlico River Basin as nutrient sensitive waters, with the specific purpose of reducing the over-nitrification of the Pamlico River Estuary (Environomics). PSs in the market function as an association, and have the option of cooperative abatement within the association or "trading" with NPSs (by contributing $56/kg/yr to a government-operated BMP fund) to meet caps on nitrogen and phosphorous emissions (McGinnis). The use of average NPS abatement costs in place of marginal costs to set the "price" of NPS credits likely eliminates some potential cost savings. However, because the BMP funds are distributed to NPS polluters through a government agency, the burden of transactions costs is at least partially lifted from both PSs and NPSs. The PSs are thus disassociated from any uncertainty inherent in dealing with NPSs; if a PS pays for its emissions above the cap it has no further responsibility. Such disassociation between buyers and sellers could possibly lead to a situation in which PSs pay for NPS emissions reductions that are simply not available (Environomics). As of 1999, however, the PS association had been able to stay below its collective nutrient cap through "minor capital improvements," "the addition of nutrient removal processes at two of the larger plants," and what might be termed PS-PS "trading" (Environomics). Because the PS association has been able to comply with its aggregate load allocation, there has been no need for payments to the BMP fund to date; however, some very limited PS-NPS "trading" has occurred through the purchase and banking for future use of NPS pollutant reduction credits (Woodward and Kaiser, Environomics).

The Lower Boise River Effluent Trading Demonstration Project

The LBRET Demonstration Project, a program trading in total phosphorous emissions, was initiated in 1997 by EPA, Idaho, Oregon and Washington as a pilot program to examine "how trading can help improve water quality and lower the overall cost of meeting pollutant reduction objectives required by [TMDL] processes" (Ross & Associates Environmental Consulting, Ltd.).

Both PS and NPS polluters in the Lower Boise River can create marketable reduction credits; however, one might argue that the market rules discourage some of the most potentially beneficial trades, namely PS-NPS trades. For example, trades involving NPS polluters must provide a water quality contribution, which is to say that "only part of the [NPS] reduction can be available for sale . . . since the sold portion could be offset by a point source increase" (Ross & Associates Environmental Consulting, Ltd.). Thus, agricultural producers must abate significantly more than they can sell as certifiable credits. The required water quality contribution may be enough to discourage PS-NPS trades, especially along with the transaction costs associated with trading. A further stipulation of the program guidelines is that "[n]onpoint credits will be transferable only after the project is installed, installation has been inspected . . . and the reductions have been verified through monitoring and established by submitting the *Reduction Credit Certificate*" (Ross & Associates Environmental Consulting, Ltd.). Potential participants would not likely want to pay in advance for the installation of a BMP only to find that the practice did not create the estimated reduction credit along with the required water quality contribution. On the other hand, the year for calculating the baseline pollutant loadings for NPSs in the lower Boise is 1996, so any BMP installed between 1996 and the present may generate salable credits (Idaho Department of Environmental Quality, 2004). Yet, NPS reductions that are calculated based on models, rather than directly measured, are reduced in value by an uncertainty discount multiplier to account for the variability in effectiveness of the BMP (Ross & Associates Environmental Consulting, Ltd.). Such program rules regarding NPSs may be the primary reason for the lack of trading activity in the Lower Boise River, as they reduce the cost-efficiency that could be gained through PS-NPS trading. The value of each of these restrictions could be argued based on criteria such as equity (i.e., PSs should not be responsible for all reduction, hence the required NPS water quality contribution) and water quality protection; however, their combinatory effect may be prohibitive to PS-NPS trading. According to Craig Shepard, of the Idaho Department of Environmental Quality, no trades have been completed in the program area to date (Personal communication December 9, 2005), despite the development of a TMDL for Snake River-Hells Canyon in 2002, which had been anticipated as the primary market driver for this WQT market (Idaho Department of Environmental Quality).

Chapter III

An Assessment of Potential Water Quality Trading Markets in Tennessee

Introduction

Working within the current regulatory structure of the CWA, offset trading, wherein PSs under regulatory pressure to reduce discharges purchase offset credits from agricultural NPSs, could significantly reduce the costs of attaining a predetermined level of ambient quality. However, such a PS-NPS WQT program would not be suitable to all areas, nor to all water quality issues. Trading the abatement responsibility for toxic substances, for example, has the potential to concentrate toxic substances in the environment around high cost abaters, and substantially increase health and environmental risk. Distinct areas with differing water quality issues will require individualized solutions. The purpose of this study is to rank different watershed areas in Tennessee according to their suitability for PS-NPS WQT as a low-cost solution to the attainment of environmental goals established by federal and state regulatory agencies. Hypothetically, a ranking system that incorporates an index of physical and theoretical economic relationships between NPS and PS polluters and the water quality impairments to which they contribute should identify some areas as being particularly suitable for the offset type WQT proposed by EPA. This study will identify and rank potentially viable trading areas based on a series of hurdles and criteria designed to eliminate areas that cannot support trading, and predict the suitability of trading as a solution to the water quality problems in the remaining areas.

Defining the Potential Market Areas

The market areas under study are commonly called watersheds, which are land areas that drain all water that falls within them to a common point, at which the water flows out of the watershed. Watershed boundaries follow topographical boundaries, such as ridge lines. By this definition, each individual stream or portion thereof might be said to have it own watershed, or drainage area. A hydrologic unit code (HUC) is a drainage basin of one size or another, as indicated by the number of digits in its numeric code. Successively smaller HUCs (which have successively longer numeric codes) are nested within larger a HUC, so that the smaller HUCs flow to a central point within the larger HUC. The largest drainage areas analyzed in this research are the eight-digit HUCs, which are often simply called watersheds. Nested within these watersheds are their subcomponent 12-digit HUCs, which all eventually contribute to the outflow from the watershed in which they lie.

The Ranking Criteria

The number of river and stream miles not meeting the ambient water quality standards to support their designated uses, hereafter referred to simply as "impaired meters," is used both as a filter and as a ranking criterion for the purposes of WQT suitability assessment. In a watershed where no impaired meters are attributed to a given pollutant, there is no need to further reduce discharges of that pollutant. Since the EPA's guidelines and the CWA do not allow increased PS discharges above the permitted level in exchange for the purchase of credits, PSs in a watershed devoid of tradable impairments would have no economic incentive to purchase credits. As impaired meters become more extensive, the need to reduce emissions of the contributory pollutant becomes spatially (and probably quantitatively) more extensive, thus making WQT a more viable solution.

Market participants are a must for WQT to work, and the geospatially weighted emissions of polluters of different types are critical variables in ranking the suitability of a given watershed as a trading area. If no regulated PSs exist in an impaired watershed, there will be no increased regulatory pressure to spur the purchase of credits from agricultural NPSs. The solution to such problems will lie entirely in the hands of NPS polluters. On the other hand, in watersheds where very little agricultural activity takes place and PS emissions are almost entirely responsible for the impairment, demand for NPS reductions may exceed supply. Furthermore, polluters more distant from an impairment have increasingly negligible impacts on the impairment. Thus, it is desirable to have some range of balance between the geospatially weighted PS and NPS emissions affecting impaired waterways. This balance need not be exactly one to one, but it should be such that the full amount of required abatement could be accomplished (at least in theory) by either group of polluters. This condition not only increases the likelihood that supply will meet demand (and vice versa) but also improves the political palatability for PSs by limiting their liability to an amount no greater than their discharge. However, data indicating the necessary pollutant load reductions to meet ambient standards in specific watersheds is not available, so, as a proxy for this measure, we use the ratio of the emissions of one polluter group to the emissions of the other group. The closer the ratio is to one, the more likely PSs could purchase all necessary reductions from agricultural NPSs.

Six hurdles, based on the above discussion, will be used to narrow the target of study by eliminating areas that are definitely unsuitable for WQT. The first three are applied on the eight-digit HUC level to eliminate all watersheds that are unsuitable for WQT. Then hurdles four through six (which are consecutively analogous to hurdles one through three) are applied on the 12-digit HUC level to eliminate all unsuitable sub-watershed areas within the watershed areas that passed the first three hurdles. Hurdles five and six are applied to aggregations of remaining 12-digit HUCs all within the same eight-digit HUC and are linked to each other by the directional flow of water. A list of the hurdles follows.

1. Presence of tradable impaired meters in eight-digit HUC (tradable impaired meters > 0)
2. Presence of PS polluters in eight-digit HUC (tradable PS discharge > 0)
3. Presence of Agricultural NPS polluters in eight-digit HUC (acres of agricultural land use > 0)
4. Contribution to tradable impairments from the 12-digit HUC (tradable impairments affected by a 12-digit HUC within its eight-digit HUC > 0)
5. Presence of PS polluters in flow-linked aggregations of remaining 12-digit HUCs within the same 8-digit HUC (tradable PS discharge > 0)

6. Presence of Agricultural NPS polluters in flow-linked aggregations of remaining 12-digit HUCs within the same 8-digit HUC (acres of agricultural land use > 0)

The above hurdles restrict the areas under consideration to flow-linked areas containing impairments and potential PS and agricultural NPS trading partners. Having thus restricted the set of watersheds and portions of watersheds under study, the next step is to rank the remaining areas that could potentially support a WQT program based on thefollowing criteria, which are analogous to the hurdles used to find the set of potential trading areas, with the exception of criteria 4.

1. Extent of tradable impaired meters
2. Distance-discounted PS contribution to tradable impairments
3. Distance-discounted NPS contribution to tradable impairments
4. Balance between criteria 2 and 3

While this list is not a comprehensive list of variables that might affect the suitability of an area for WQT, it does provide insights into the basic physical criteria that would predict a strong market. For example, established, active watershed organizations may also have an effect on the political feasibility of a WQT program; however, their potential effect is somewhat unpredictable, since they might choose either to legally oppose the trading system or to facilitate transactions in the new market. To assess the potential impacts of such human variables as watershed organizations and public opinion would necessitate an extensive information-gathering effort through surveys of organizations and individuals. Such an effort would be costly and premature without first ranking potential WQT markets based solely on physical properties as affected by the current legal framework.

Data & Methodology

The most general set of watersheds under consideration was comprised of all eight-digit HUCs that are either partially or completely contained by the Tennessee state boundary. Locations of these watersheds relative to the state border were determined by overlaying eight-digit HUC boundaries and state borders in ArcMap to begin the creation of a geospatial information system (GIS). Only the portions of these watersheds that are within the Tennessee boundary were considered, since the effect of involving regulatory agencies and other stakeholders from multiple states is likely to drastically increase the administrative and transactions costs of a WQT program. The geospatial data used in this step may be viewed at and extracted from the National Hydrography Dataset (NHD) Geodatabase[5], which incorporates information of various types from the US Geological Survey and US EPA.

Application of the first hurdle to the eight-digit HUCs eliminates those that contain no impairments attributed to nutrients, organic enrichment, or low dissolved oxygen (hereafter referred to as "tradable impairments"). The presence of such impairments was determined through the use of the final version of Tennessee's 2002 303(d) List (Tennessee Department of Environment and Conservation), which lists by eight-digit HUC all impaired water bodies in the state, along with their respective causes.

[5] Available: http://nhdgeo.usgs.gov/viewer.htm

A geospatial shape file representing the 303(d) listed impairments was then obtained from EPA's Watershed Assessment, Tracking & Results (WATERS) data downloads[6] and added to the GIS to find the exact locations of the impairments within the eight-digit HUCs.

Next, eight-digit HUCs containing no PS polluters with nitrogen limits were eliminated through the initial application of the second hurdle. This hurdle eliminates those watersheds that have no potential buyers of nitrogen credits. PS data were obtained using EPA's Permit Compliance System (PCS)[7] customized query engine, which provides downloads of locational and discharge information for PSs in Tennessee that emit nitrogen. The download provided coordinates for most sources that had discharge limits for nitrogen; however, the coordinates provided were often in direct conflict with other locational information, such as the eight-digit HUC listed in the PCS data. In instances where such a conflict existed (or when no coordinates were reported), the coordinates were either verified or corrected using street addresses to locate each potentially contributing PS in its proper 12-digit HUC.

Use of the third hurdle eliminates eight-digit HUCs that contain no agricultural NPS polluters, which would be the potential sellers of nitrogen credits. The land use data for this study came from the 1992 National Land Cover Dataset (NLCD)[8], and can be viewed and downloaded from the USGS Seamless Data Distribution page. This dataset was used to locate agricultural activities within their eight- and 12-digit HUCs. Areas

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[6] Available: http://www.epa.gov/waters/data/downloads.html
[7] Available: http://www.epa.gov/enviro/html/pcs/adhoc.html
[8] Available: http://seamless.usgs.gov/website/seamless/viewer.php

that do not pass this hurdle could not, in theory, host viable PS-NPS WQT markets involving agricultural NPSs.

The remaining eight-digit HUCs were then subdivided into their sub-component 12-digit HUCs. Doing so allows us to account for the directional flow of water and the approximate distance between every source and the impairment(s) to which it contributes. The 12-digit HUCs, for the purposes of this study, are afforded a status similar to that of the zones described by Hung and Shaw. Also, this step seemed appropriate since US EPA and the state environmental agencies break eight-digit HUCs into their sub-component 12-digit HUCs for the purposes of creating TMDLs. 12-digit HUC boundaries are available for download through the U.S. Department of Agriculture's Natural Resources Conservation Service Geospatial Data Gateway[9]. Next, the fourth hurdle, based on contribution to downstream tradable impairments within the same eight-digit HUC, was applied to the sub-component 12-digit HUCs. Where downstream impairments did not exist, those 12-digit HUCs were considered non-contributors to water quality problems, and were eliminated from consideration. This simply means that polluters that do not contribute to impairments within their eight-digit HUCs will not be considered viable trading partners. The remaining 12-digit HUCs were then combined to create sets of 12-digit HUCs, or "impairment zones," that contribute to the same impairment(s). Contributor status was determined for each 12-digit HUC based on directional flow information from the NHD Geodatabase in conjunction with the boundaries of eight- and 12-digit HUCs. There was sometimes more than one "impairment zone" within an eight-digit HUC.

[9] Available: http://datagateway.nrcs.usda.gov/NextPage.asp

These “impairment zones” were then subjected to the fifth and sixth hurdles. Doing so eliminates unsuitable portions of the eight-digit HUCs from consideration. Those zones that contain no PS polluters or no NPS polluters cannot be considered viable markets, since no trading partners would be present.

PS discharge limits from the PCS data were used as a proxy for actual emissions because these limits are based on the design flow and technological regulatory requirements at the facilities; therefore, the limits should be nearly the same as the discharges. Furthermore, PSs with discharge limits are likely to be the only purchasers of pollution credits. The discharge limits, given in pounds per day, were annualized and summed for each 12-digit HUC, using the formula:

$$D_i = \sum_{j=1}^{n} d_j * 365$$

Where:

D_i is total annual PS discharge within 12-digit HUC i,

d_j is the average daily discharge from PS polluter j, and

n is the set of all PS polluters in12-digit HUC i.

Approximate NPS nitrogen loads were derived from a combination of land use data and nitrogen emissions factors for different land use types as located by the NLCD, which divides Tennessee’s agricultural lands into two major types: pasture/hay and row crops. The acreages of different land use types were multiplied by their emissions factors (23.2 lb/ac/yr and 5.53 lb/ac/yr for row crops and pasture, respectively), as determined by Bhaduri, et al., for each 12-digit HUC. Use of the formula below returns the NPS loadings for a 12-digit HUC:

$$L_i = \sum_{j=1}^{n} \left(H_j * 5.53\right) + \sum_{k=1}^{m} \left(Z_k * 23.2\right)$$

Where:

L_i is total annual NPS nitrogen loading for 12-digit HUC i,

H_j is the acreage of pasture feature j,

n is all pasture features in12-digit HUC i,

Z_k is the acreage of row crop feature k, and

m is all row crop features in 12-digit HUC i.

To obtain raw, discounted NPS and PS scores for each of the 12-digit areas, the downstream impaired meters were multiplied by a discount factor that is a function of the number of 12-digit HUCs between the source HUC and the impairments to which it contributes. These scores were calculated by multiplying the NPS and PS discharges by the sum of the discounted downstream impaired meters to which they contribute as follows:

$$N_i = \sum_{j=1}^{n} M_j * \frac{1}{x_{ij}} * L_i \text{ and}$$

$$P_i = \sum_{j=1}^{n} M_j * \frac{1}{x_{ij}} * D_i$$

Where:

N_i, P_i, L_i and D_i are respectively the NPS score, PS score, NPS loadings, and PS discharges for 12-digit HUC i,

M_j is the length (in meters) of impairment j,

x_{ij} is two plus the number of 12-digit HUCs intervening between 12-digit HUC i and impairment j, when impairment j is not within 12-digit HUC i, and is equal to one when impairment j is within HUC i, and

n is the number of impairments to which the discharges 12-digit HUC i contribute.

Discounting discharges by distance is appropriate for pollutants such as nitrogen because it dissipates during in-stream transport, making more distant impacts relatively smaller than impacts close to the source (Hung and Shaw). The result of the above process is a distance-weighted PS and NPS emissions contribution score for each individual 12-digit HUC.

After developing NPS and PS emissions scores for each 12-digit HUC using the above methodology, the scores were aggregated up to the "impairment zone" level. Each zone was assigned an NPS and a PS score, which are equal to the sums of the respective scores for each of its sub-component, 12-digit HUCs

Thus, criteria five through six listed in the previous section were subsumed into two factors (PS and NPS scores) to be used in developing a comparison between different potential trading areas.

Balance between PS and NPS emissions is the third factor to be used in developing the ranking system. To obtain a measure of this balance, the smaller of the NPS and PS scores for each potential trading area was divided by the larger, which returns a polluter ratio (R) between zero and one. This ratio was used to discount for the possibility that emissions from one type of polluter might not be substantial enough to make PS-NPS trading feasible.

NPS and PS scores, as well as polluter ratios for each potential trading area were then combined in a Cobb-Douglas function designed to predict an area's suitability for WQT. The function follows:

$$\hat{S}_j = N_j^a P_j^b R_j^c$$

Where:

$\hat{S}_j$ is the potential trading area *j*'s predicted WQT suitability score,

N_j is the NPS score trading area *j*,

P_j is the PS score for trading area *j*,

R_j is the ratio of the smaller score to the larger for trading area *j*, and

a, *b*, and *c* are weights that sum to 1.

The output of the function was a composite WQT suitability score (which was then normalized between zero and 1) for each potential trading area. These scores can be compared across the potential areas to identify areas most likely to find that PS-NPS WQT could be part of an appropriate solution for their impairment problems.

The Cobb-Douglas functional form was chosen because of its constant returns to scale assumption and its flexibility. The exponents can be altered to reflect various policy choices. For example, if one desired to rank these watersheds with an NPS:PS trading ratio of 1:1, the exponents a and b should be equal to each other, and equal to the exponent *c* by default. However, if the trading ratio is to be 2:1, the exponent of the NPS score should be twice that of the PS score (*a*=2*b*), making the NPS score worth more in the final ranking. Accounting for trading ratios in the function will change the meaning and importance of the ranking criterion R_j. Specifically, R_j might more appropriately equal the ratio $N_j^a : P_j^b$ when the trading ratio is not equal to one. The exponent *c* would then be forced to equal the lesser of the exponents *a* and *b* if retention of constant returns to scale is desirable. Furthermore, the ratio would no longer be constrained between zero and one, since importance would be placed on which sources were discharging more of the pollutant. It is possible that N^a:P^b ratios greater than or equal to one would all be equally desirable. Note that as the proposed trading ratio increases, the weight of R decreases.

Results

Tennessee includes portions of 56 eight-digit HUCs. Only portions of these watersheds within the Tennessee boundary were considered (Figure 3.1). After applying the first hurdle based on the presence of tradable impairments, 42 eight-digit HUCs or portions thereof remain for consideration (Figure 3.2). Application of the PS polluter hurdle to the 42 eight-digit HUCs eliminates another five watersheds from consideration, leaving 37 for further analysis (Figure 3.3). The NPS hurdle is not binding, since all remaining eight-digit HUCs contain some level of agricultural production, as shown by the land use data.

For the next stage of analysis, the remaining eight digit HUCs were disaggregated to their 934 subcomponent 12-digit HUCs. Application of the fourth hurdle eliminates those 12-digit HUCs neither containing nor contributing to a downstream tradable

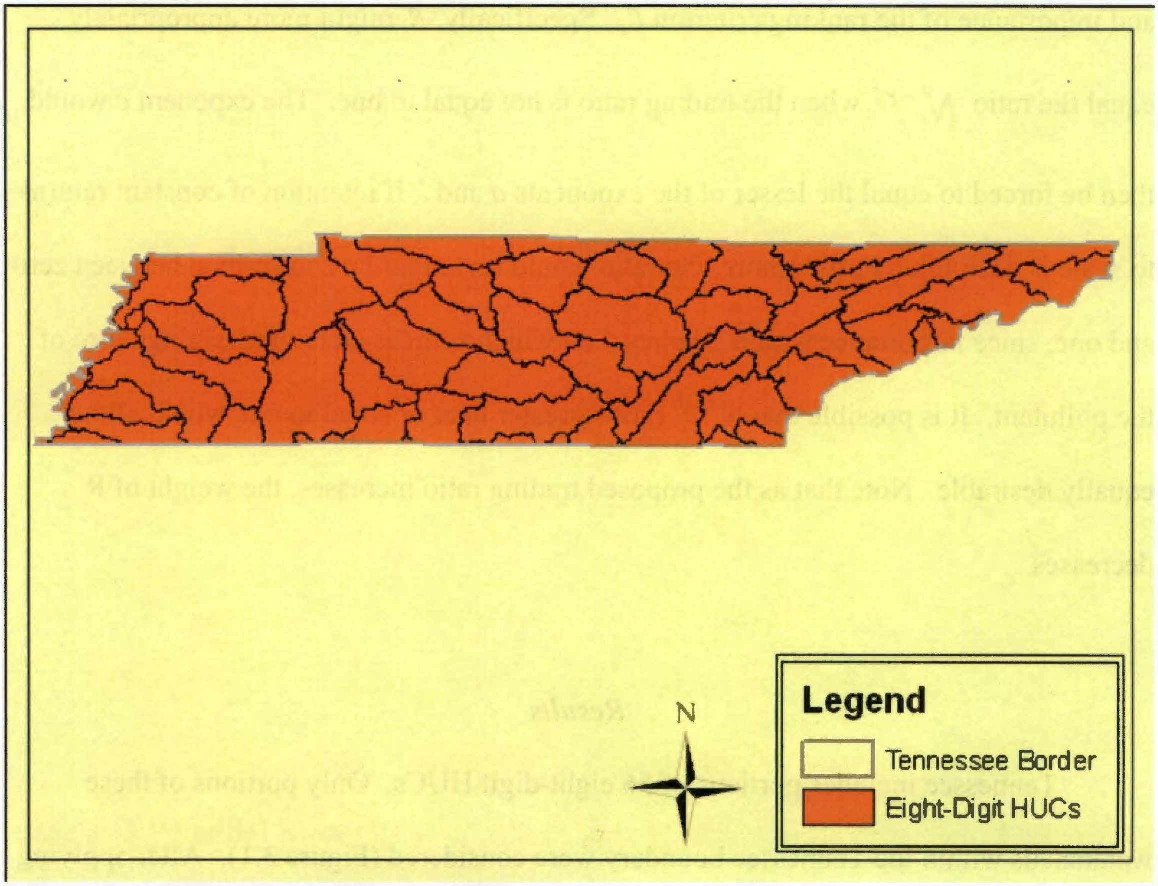

Figure 3.1: Tennessee's 56 Watersheds

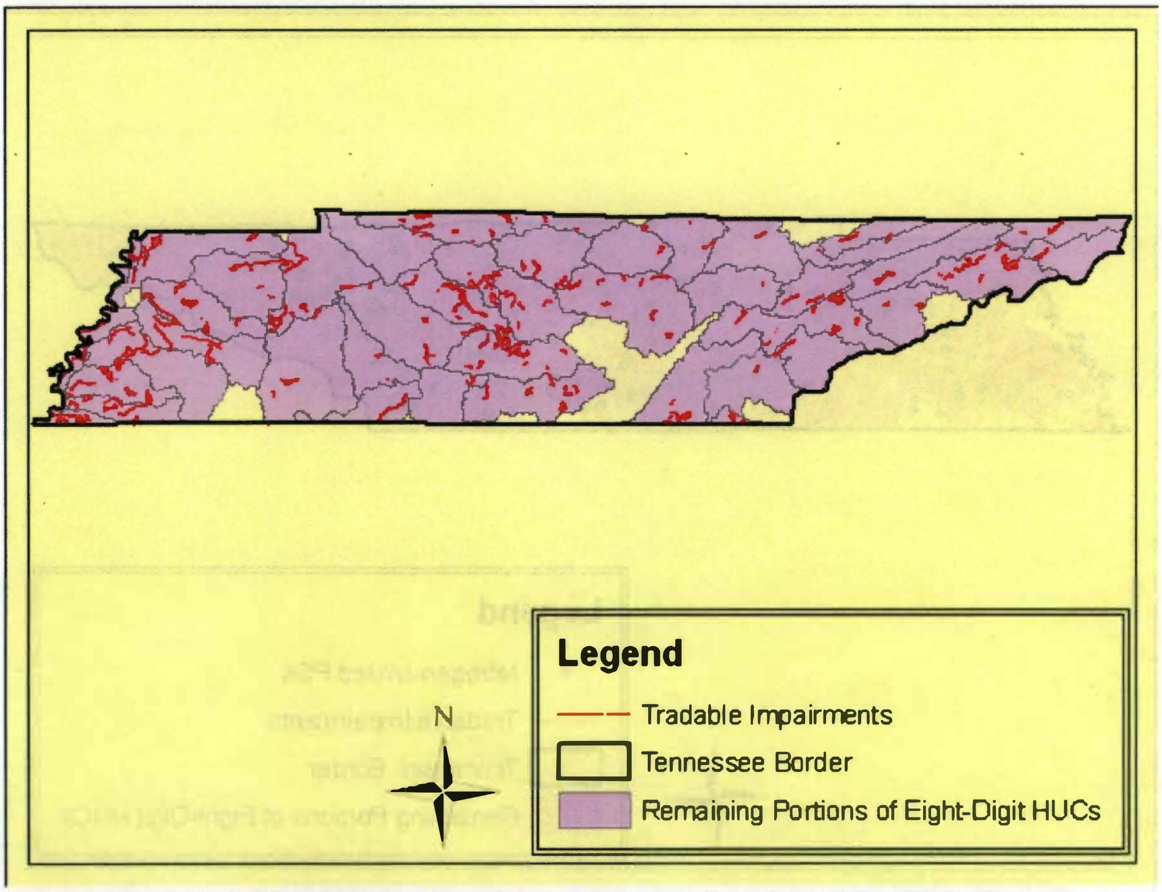

Figure 3.2: Eight-Digit HUCs Remaining after Application of the First Hurdle

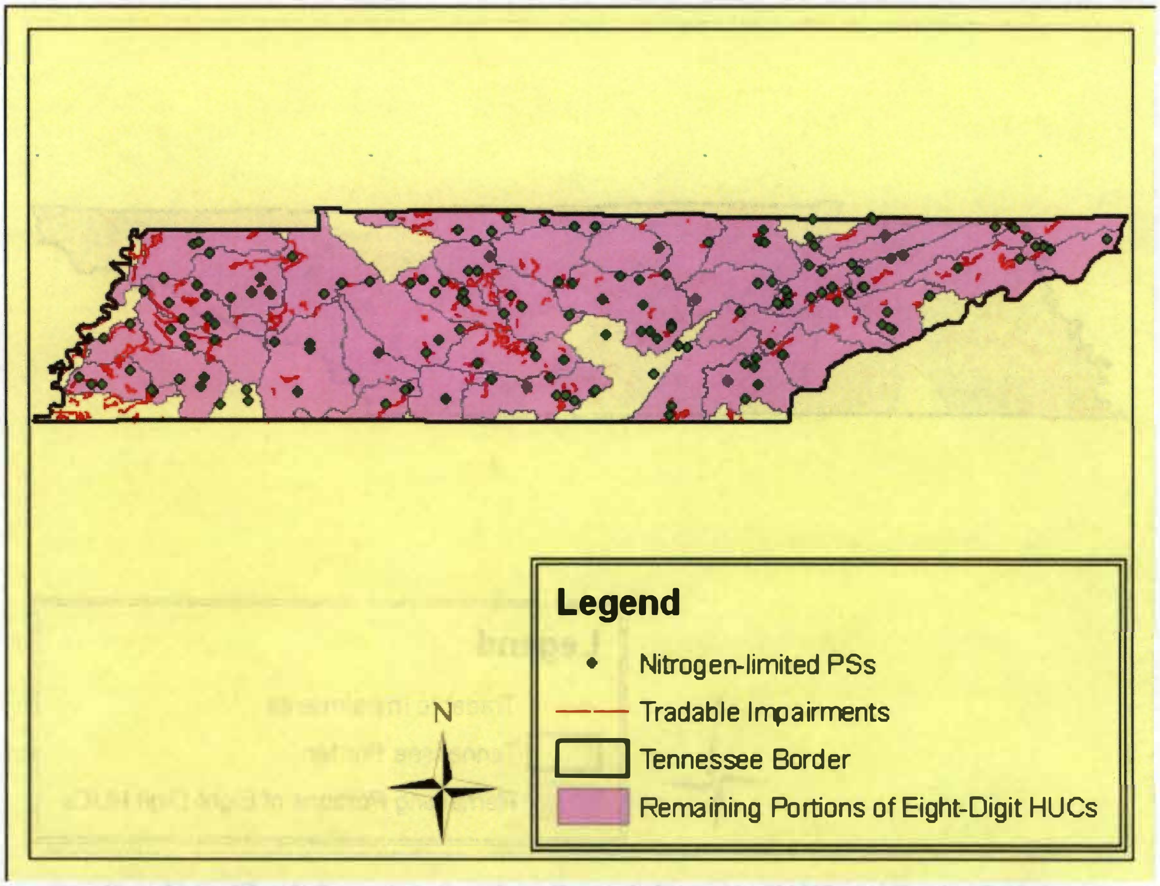

Figure 3.3: Eight-Digit HUCs Remaining after Application of the Second Hurdle

impairment, resulting in a set of 318 12-digit HUCs for further analysis. These 318 12-digit HUCs were then grouped into 103 “impairment zones” within the 37 remaining eight-digit HUCs (Figure 3.4). Note that several of the “impairment zones” share boundaries, and it is possible that there may be flow linkages between some of them; thus, the eight-digit HUC boundary may not be the most logical cutoff point for the boundary of an impairment zone. However, eight-digit boundaries were used because of US EPA’s guidance. At this point, the fifth hurdle was applied to these zones to eliminate those that contain no PS presence. The sixth hurdle was not binding, as all remaining 12-digit HUCs had some agricultural production. The impairment zones were thus whittled down to 45 potential trading areas, consisting of 239 12-digit HUCs collectively (Figure 3.5). The Red River watershed example in Figure 3.6 provides an example of what an eight-digit HUC that crosses the state border might look like after application of hurdles one through six. Note that only 12-digit HUCs within Tennessee are included in the analysis, and that this watershed has two potential trading areas.

Following the steps outlined in the methodology section produces PS and NPS scores, as well as polluter ratios for all 239 12-digit HUCs under consideration. The three ranking factors were then summed across each potential trading area to produce aggregate scores for all 45 of them. Potential trading area PS and NPS scores were then entered into the Cobb-Douglas function to produce WQT suitability scores.

The output of the process provides normalized scores for 45 potential trading areas in Tennessee, which are plotted by score and rank in Figure 3.7. Scores for the top three ranked potential WQT areas are between 1.00 and 0.88, and represent the entirety or a

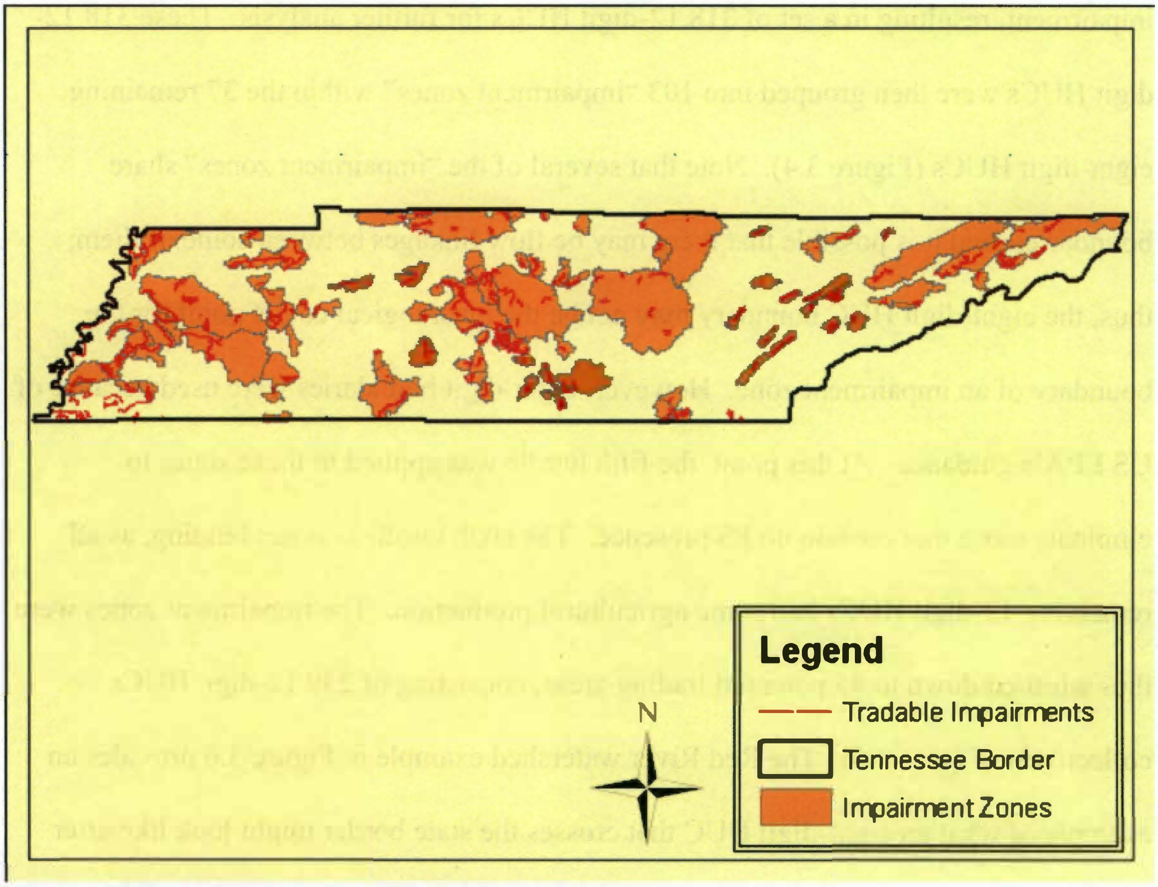

Figure 3.4: Tennessee's 103 Impairment Zones

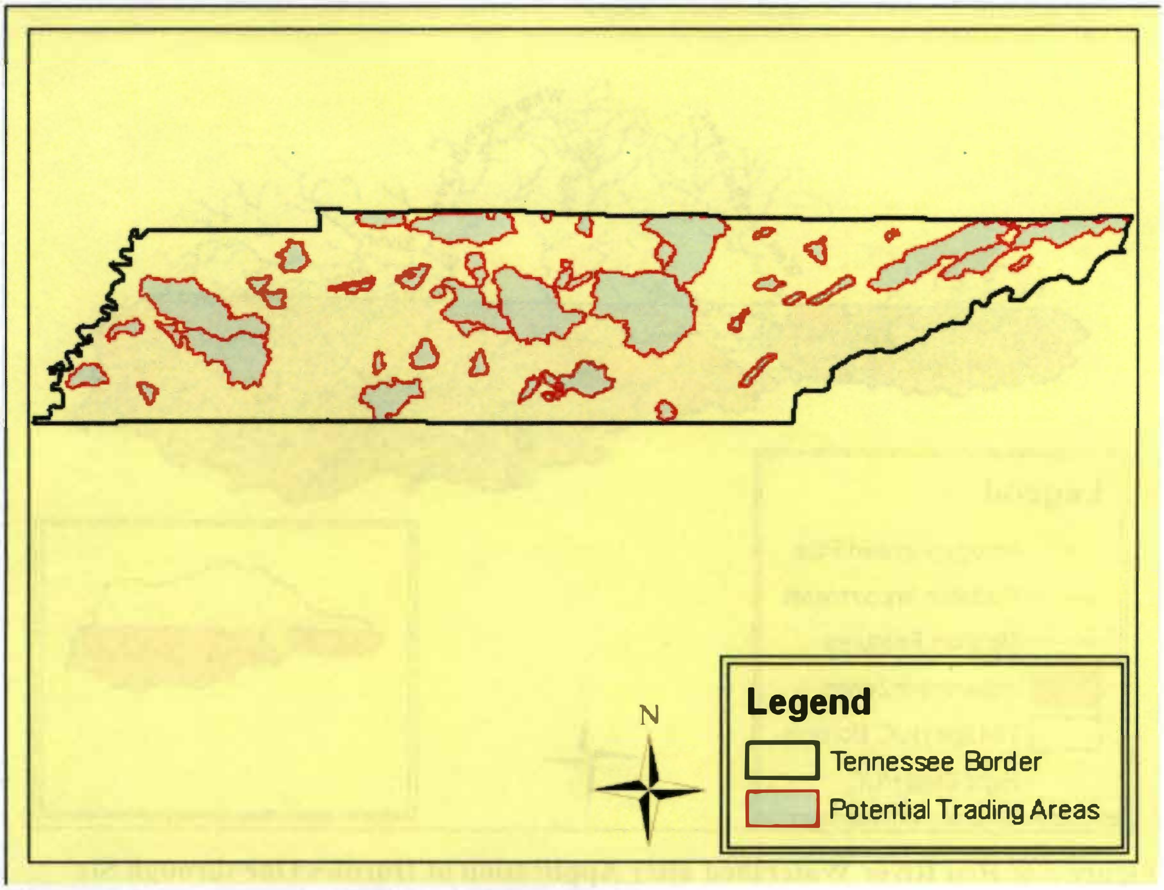

Figure 3.5: Tennessee's 45 Potential Trading Areas

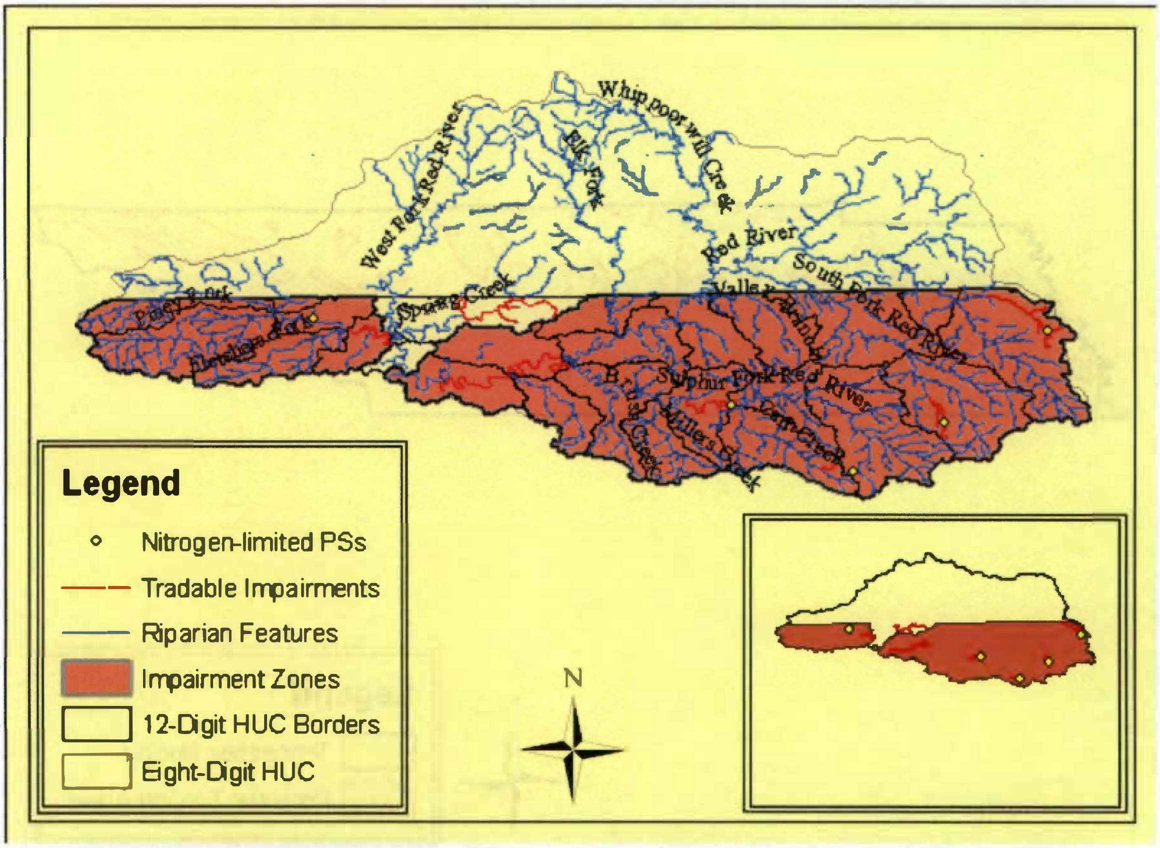

Figure 3.6: Red River Watershed after Application of Hurdles One through Six

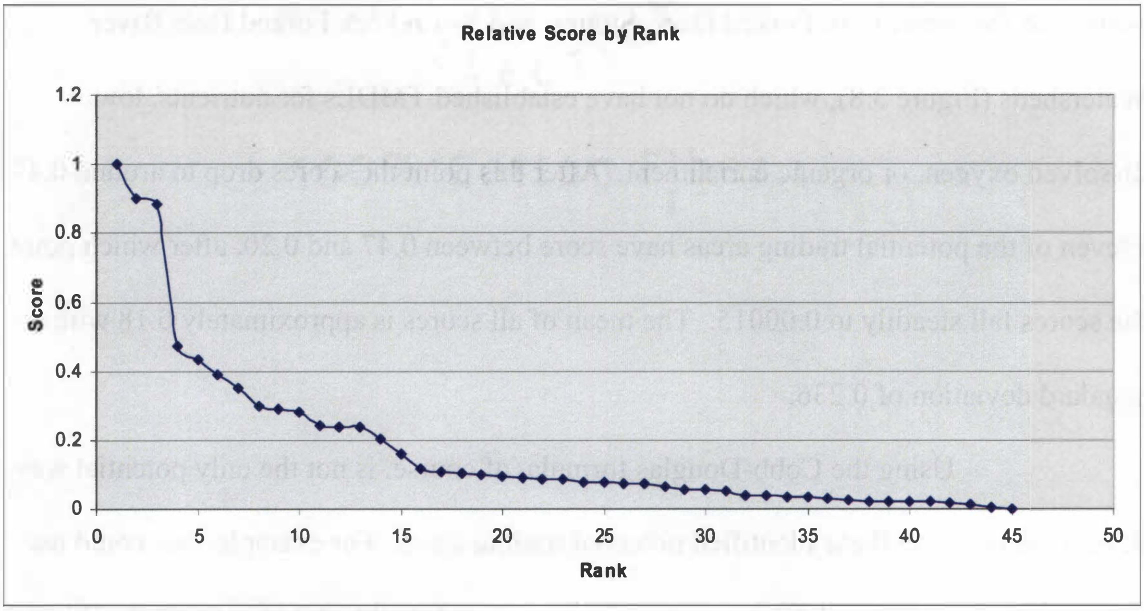

Figure 3.7: Relative Score by Rank

portion of the South Fork Forked Deer, Stones, and North Fork Forked Deer River Watersheds (Figure 3.8), which do not have established TMDLs for nutrients, low dissolved oxygen, or organic enrichment. After this point the scores drop to around 0.47. Eleven of the potential trading areas have score between 0.47 and 0.20, after which point the scores fall steadily to 0.00015. The mean of all scores is approximately 0.18 with a standard deviation of 0.236.

Using the Cobb-Douglas formula, of course, is not the only potential way of ranking or rating these identified potential trading areas. For example, one could use some multi-objective ranking or rating scheme to compare the areas based on the PS and NPS contribution to the tradable impairments. There are also other factors that are correlated with the WQT suitability score. For example, the acreage of the potential trading area and its suitability score have a positive correlation of 0.722, which makes sense because larger areas are likely to contain more tradable impairments, more contributing PS polluters, and more contributing NPS polluters. Table 3.1 presents each trading area, listed by some portion of its respective eight-digit HUC, with its associated raw scores for each of the ranking criteria, as well as the areas' associated acreages. Table 3.2 presents the same scores converted into rankings for each watershed. Both tables are included because 3.1 shows the relative differences between the scores of different trading areas, while Table 3.2 more effectively illustrates the relationship between the ordinal rankings of the potential trading areas.

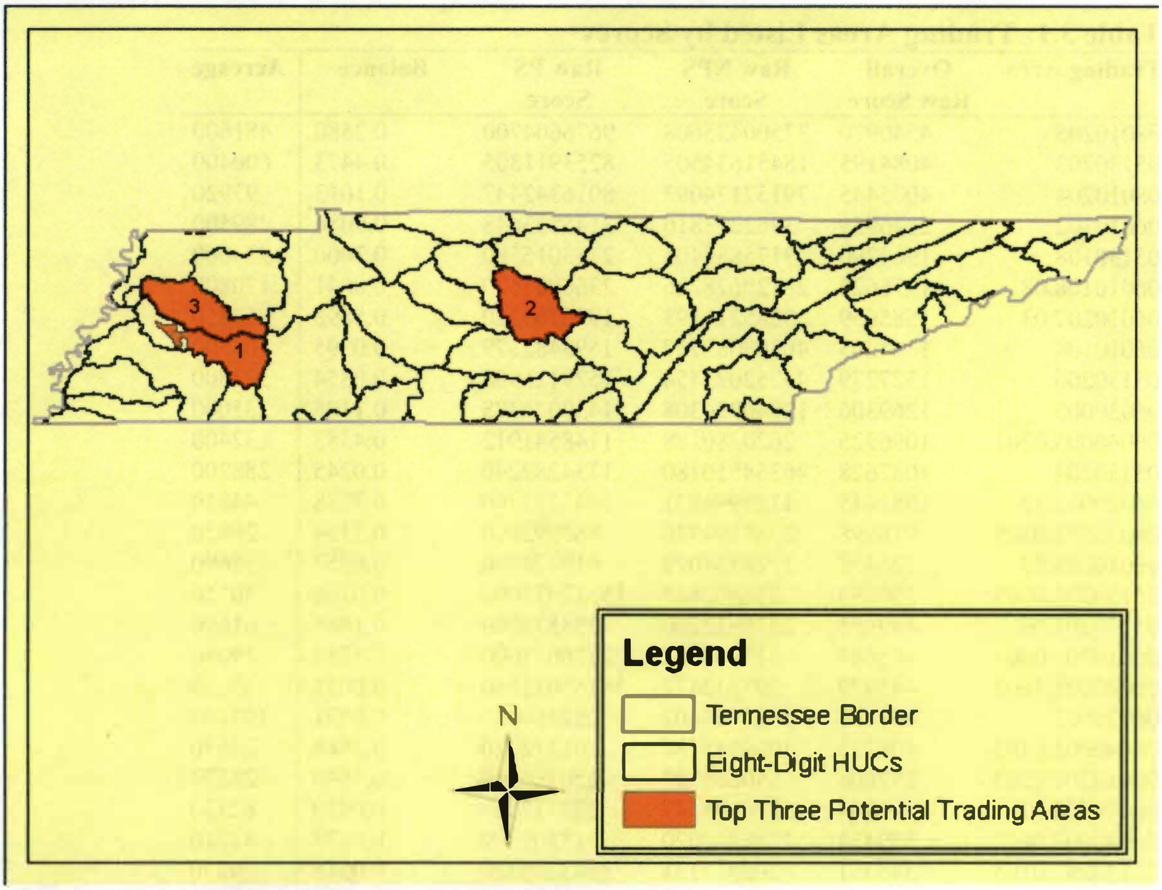

Figure 3.8: Top Ranked Potential Trading Areas

Table 3.1: Trading Areas Listed by Scores

Trading Area	Overall Raw Score	Raw NPS Score	Raw PS Score	Balance	Acreage
08010205	4540970	37500435608	9676604700	0.2580	481600
05130203	4084195	18451634505	8253911305	0.4473	600400
08010204	4005445	79157174093	8016342447	0.1013	97920
06010102	2146959	3462213816	3145833648	0.9086	289500
05130108	1964290	9175887405	2753015580	0.3000	934000
06010108.08	1775624	28129628226	2366065620	0.0841	170800
06010207.03	1585659	3385211393	1996709220	0.5898	57850
06010104	1364263	40309082767	1593482379	0.0395	503500
05130206	1327279	43252023454	1529126908	0.0354	372500
06030005	1269306	12940934308	1430044488	0.1105	231000
06040003.0201	1096725	2620280188	1148541912	0.4383	32400
05130204	1087628	46354516180	1134282240	0.0245	288200
06020002.12	1081045	1123998871	1443173760	0.7788	44810
06030003.0305	918998	2789194746	880992900	0.3159	26820
08010208.07	726455	7228134072	619174800	0.0857	55690
05130202.0103	530693	386602885	36512937000	0.0106	30550
05130204.06	499985	2375932200	353538000	0.1488	61690
06010207.0503	465644	317747418	2576673000	0.1233	19050
06020001.1102	445979	297832477	58155631560	0.0051	39620
06030003	435463	6821564602	287360640	0.0421	197100
06040005.0403	408733	1009585786	261312480	0.2588	24650
06010201.0503	397608	540285327	250716960	0.4640	23570
06040003.03	385264	5210698347	239132565	0.0459	82330
06040002.0601	359454	2587842020	215509500	0.0833	42710
05130202.0105	345351	202951131	3943246320	0.0515	40950
06040005.06	337698	11761136609	196242750	0.0167	93100
05130106	328570	2148746562	188339940	0.0877	45340
08010203.0101	288794	1313177273	155197080	0.1182	47930
05110002.0105	262797	726359409	134720208	0.1855	11680
06030003.0701	258993	3170155524	131805216	0.0416	35100
06010108.0508	239945	1669811969	117535410	0.0704	21010
05130105	183730	1863507747	78753870	0.0423	504200
05130108.0807	182091	2713360649	77702328	0.0286	28790
06040004.0104	153497	547961485	60138468	0.1097	23710
05130201.0201	152724	5229288499	59684526	0.0114	42730
05130104	141098	173009161	53001000	0.3063	17390
08010203.0102	120145	1325252629	41644800	0.0314	33590
06010205.1104	105439	103559366	34237800	0.3306	45190
06010208.0405	93836	219184011	28744686	0.1311	39700
06010206.0404	93401	269815476	28544886	0.1058	11680
05130204.0303	90964	114238891	27435240	0.2402	17610
06040003.0507	69223	513021723	18213120	0.0355	26960
05110002.0503	58483	584255578	14143410	0.0242	10610
08010209.01	10646	5419980370	1098504	0.0002	33580
05130206.07	666	1036409754	17203	0.0000	82610

Table 3.2: Trading Areas Listed by Rankings

Trading Area	Overall Rank	NPS Rank	PS Rank	Balance Rank	Acreage Rank
08010205	1	5	3	12	5
05130203	2	7	4	5	2
08010204	3	1	5	22	12
06010102	4	16	7	1	7
05130108	5	10	8	10	1
06010108.08	6	6	10	25	11
06010207.03	7	17	11	3	17
06010104	8	4	12	33	4
05130206	9	3	13	35	6
06030005	10	8	15	19	9
06040003.0201	11	21	16	6	31
05130204	12	2	17	38	8
06020002.12	13	29	14	2	22
06030003.0305	14	19	18	8	35
08010208.07	15	11	19	24	18
05130202.0103	16	37	2	42	32
05130204.06	17	23	20	15	16
06010207.0503	18	38	9	17	40
06020001.1102	19	39	1	43	27
06030003	20	12	21	31	10
06040005.0403	21	31	22	11	36
06010201.0503	22	35	23	4	38
06040003.03	23	15	24	29	15
06040002.0601	24	22	25	26	24
05130202.0105	25	42	6	28	25
06040005.06	26	9	26	40	13
05130106	27	24	27	23	20
08010203.0101	28	28	28	18	19
05110002.0105	29	32	29	14	43
06030003.0701	30	18	30	32	28
06010108.0508	31	26	31	27	39
05130105	32	25	32	30	3
05130108.0807	33	20	33	37	33
06040004.0104	34	34	34	20	37
05130201.0201	35	14	35	41	23
05130104	36	43	36	9	42
08010203.0102	37	27	37	36	29
06010205.1104	38	45	38	7	21
06010208.0405	39	41	39	16	26
06010206.0404	40	40	40	21	44
05130204.0303	41	44	41	13	41
06040003.0507	42	36	42	34	34
05110002.0503	43	33	43	39	45
08010209.01	44	13	44	44	30
05130206.07	45	30	45	45	14

Conclusions

Since the scores only provide a relative ranking, the potential of any of the WQT areas under study to actually support a viable WQT program is unknown. However, it may be asserted that areas with higher scores are more suitable for WQT than areas with lower scores. Ideally, a threshold score would be identified that would signify the viability of a potential market area. Such a threshold value could, in theory, be identified by comparing the scores of Tennessee's watersheds with scores for other areas that are known supporters of viable markets, such as the various WQT pilot programs that have met with differing degrees of success. Another potential way to discover a feasibility threshold might be to calculate the potential abatement cost savings in each ranked Tennessee watershed. Much could be learned about the threshold by performing a cost savings calculation for a strategically chosen watershed, somewhere in the middle of the ranking. Such an analysis could be performed where the required reductions to meet ambient environmental standards are known (or have been estimated by some regulatory agency), as would be the case in a watershed with a TMDL for nitrogen.

Theoretically, these scores should also be positively correlated with the potential abatement cost savings for each of the possible trading areas, since the ranking process was designed to rank the watershed areas based on their relative potential benefits from tradable abatement responsibilities. Intuitively, then, watershed areas with higher scores are likely to produce greater trading activity because they will likely contain more polluters with varying cost structures, so that their marginal costs of abatement may be equalized at positive discharge levels.

Chapter IV

Assessing the Cost of NPS Technology-Based Regulation

Introduction

Since the passage of the CWA, largely unregulated NPS pollution contributes to an increasing share of the nation's water quality problems. While the amount of agricultural land in the United States has remained relatively constant over the past 30 years, agricultural production has increased as a result of technological advances and increasingly fertilizer-intensive agricultural practices, which have have been a major contributor to a three-fold increase nitrate load from the Mississippi River to the Gulf of Mexico (Greenhalgh and Sauer). When the states and territories performed a recent survey of the nation's waters, they found that 39% of the waters assessed were impaired for one or more uses, and EPA posits that the largest single contributor to water quality impairments is NPS runoff from agricultural lands (US Environmental Protection Agency Office of Water, 2002). One potential means of ameliorating the adverse effects of modern agricultural practices would be to impose a technology-based regulation on all agricultural producers by requiring implementation of some basic BMP to protect water quality.

This paper is an attempt to quantify the costs of implementing a regulation that requires that all agricultural producers create grass buffer areas of approximately 150 feet between agricultural production activities and riparian features in the Harpeth River Watershed (Figure 4.1). Potential financial impacts of such a regulation are of interest

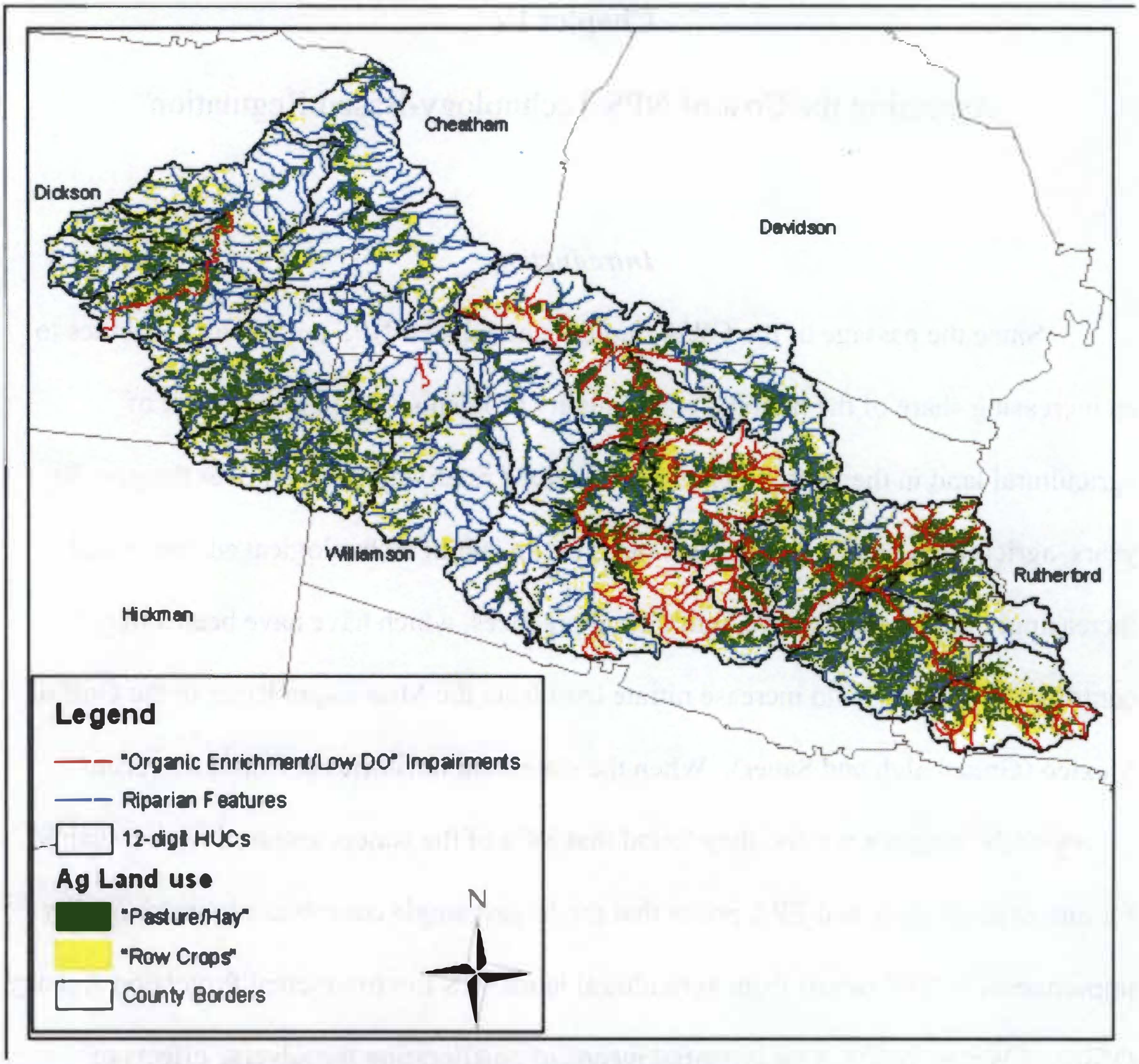

Figure 4.1: The Harpeth River Watershed in Tennessee

with regard to WQT because the supply of riparian buffer strips will bear some proportional relationship to the supply of agricultural NPS nutrient emissions reduction credits. Of course, individual buffer strips will have differential environmental impacts determined by, among other things, the rate of soil loss, the fertility of the soil, their relative proximities to water quality impairments, and the land uses and practices surrounding the buffer area.

Overview of Project Area

The Harpeth River watershed is located to the west and southwest of Nashville in Middle Tennessee, encompassing parts of Cheatham, Davidson, Dickson, and Williamson Counties (Figure 4.1). The watershed drains 867 square miles through approximately 1,364 miles of streams before eventually emptying into the Cumberland River (US Environmental Protection Agency Region 4). The most recent land use data for the watershed is from the 1990-1993 Multi-Resolution Land Characteristic Consortium, in the form of the 1992 National Land Cover Database (NLCD). Although land use changes have certainly occurred in this rapidly-developing watershed over the last decade, all land use data in this paper will be derived from the 1992 NLCD. At that time, approximately 24.2%, or179,335 acres, were in agricultural use. Pasture/hay accounted for 130,294 of these acres, while row crops covered the remaining 49,041. The Harpeth River watershed was chosen for this project because it not only has streams that are impaired due to "Organic Enrichment/Low DO", but more importantly, because it is the only watershed in the state of Tennessee with a finalized TMDL for this nutrient-related impairment (US Environmental Protection Agency Region 4). The importance of the TMDL lies in the detailed information on sources of nutrients within the watershed and the effects of these nutrients on water quality.

Data & Methodology

To start, the 1992 NLCD was clipped in ArcMap to the area of the Harpeth River Watershed. The next step was to isolate agricultural land uses from other land uses. In the watershed, the NLCD only shows two classifications of agricultural land use, namely "row crops" and "pasture/hay." Once these lands were located and isolated, they were clipped based on proximity to riparian features shown in the NHD for the watershed. This process was accomplished by creating a 150 foot buffer in ArcMap around all riparian features, and then clipping the land use features to this buffer. ArcMap was then used to calculate the acreage of each of these patches of agricultural land within the buffer area. These patches were deemed to be the land put to use in buffer activities under a potential regulatory program.

Cropland was valued at its opportunity cost, which was determined based on both cropping practices and soil fertility. The most recent county crop acreage data from the U.S. Department of Agriculture National Agricultural Statistics Service[10] were used to estimate the crop composition in each county. The most current data available were generally for the year 2004; however, the most recent soybean data available for Davidson and Dickson counties were from 2001 and 2002, respectively. Corn acreage for the purposes of this study was the sum of the corn for grain acreage from the

[10]Available: http://www.nass.usda.gov/nass/pubs/histdata

aforementioned datasets and the corn for silage acreage listed for each county in the 2002 census of agriculture, since no more recent data for silage was available.

The State Soil Geographic (STATSGO) Database[11], a product of the US Department of Agriculture Natural Resources Conservation Service, serves as the measure of soil fertility for this study. The dataset divides the area into mapping units, or areas with similar soil patterns and compositions, and also provides the proportions of the mapping units' constituent soils. Further, the data provide estimated per acre yields for each crop and soil type combination. A weighted average yield was obtained for each crop and mapping unit by multiplying the yield for a given soil type by the soil's proportion within the mapping unit divided by the sum of the proportions of all soil types where the crop was grown, then summing these products for each crop. The above process is summarized by the following equation:

$$\bar{Y}_{ij} = \sum_{k=1}^{m}\left(Y_{ik} * P_{jk} \div \sum_{j=1}^{m} P_{jk}\right)$$

Where:

$\bar{Y}_{ij}$ is the weighted average yield for crop i in a mapping unit j,

Y_{ik} is the STATSGO yield for crop i on soil type k,

P_{jk} is the proportion of mapping unit j covered by soil type k, and

m is the total number of soil types on which crop i is grown in soil mapping unit j.

Each crop then had a weighted average yield for each mapping unit. These weighted averages are assumed to be 1994 yields, as the metadata for the STATSGO database indicate that the data are current as of their publication date (1994). As a result, the yields

[11] Available: http://www.ncgc.nrcs.usda.gov/products/datasets/statsgo/index.html

needed to be inflated to current yields to reflect advances in production technology, as well as the interaction of these advances with the changing resource base in each county, over the last decade.

An appropriate yield inflator would be the expected 2004 yield divided by the expected 1994 yield for each county and crop. Using expected yields is more appropriate than using actual yields because actual yields may not reveal the same yield response to technological change, since other factors, such as drought, may affect yields in any given year. Furthermore, producers make decisions based on expectations, rather than actual yields. County level data (as opposed to national data) was used for the creation of the yield inflators in order to account for some of the regional variations and changing land use patterns unique to the area of study. To find predicted yields, time periods were first regressed on actual yields from the NASS downloads for each crop in each county to estimate the values of the parameters of the equation:

$$y_{ift} = \alpha + \beta t$$

Where:

y_{ift} is the actual yield for crop i in county f in time period t,

α & β are parameters to be estimated by an ordinary least squares regression relating time period to yield, and

t is the time period ($t_{1987} = 1, \ldots, t_{2004} = 18$).

The estimated parameters (α^* and β^*) for each crop in each county were then used to predict yields for t_{2004} and t_{1994} using the equation:

$$\hat{y}_{ift} = \alpha^* + \beta^* t_i$$

Where:

$\hat{y}_{ift}$ is the predicted yield for crop i in county f in time period t,

α^* & β^* are parameters estimated by the preceding ordinary least squares regressions for all crops and counties, and

t is the time period for which the yield is predicted.

The resulting regression equations and yield inflators are presented in Table 4.1. Note that the estimated β coefficients for some crops in some counties are negative, which may be attributed to the changing resource base in those counties that are rapidly urbanizing. The yield inflators ($I_{if} = \hat{y}_{if2004}/\hat{y}_{if1994}$) were multiplied by the weighted average 1994 yields from the STATSGO data that appertained to the respective counties and crop types for each potential buffer feature, producing a weighted average expected current yield for each crop on each buffer by mapping unit and county. These weighted average current expected yields for each crop on each buffer feature by county and soil mapping unit were produced using the formula below:

$$\widetilde{Y}_{ifj} = I_{if} * ¥_{ij}$$

Where:

$\tilde{Y}_{ifj}$ is the weighted average current expected yield for crop i in mapping unit j in county f,

I_{if} is the yield inflator for crop i in county f, and

$¥_{ij}$ is the 1994 weighted average expected yield for crop i in soil mapping unit j.

Table 4.1: Estimated Regressions and Yield Inflators

County	Crop	Equation	Yield Inflator (I_{if})
Cheatham	Corn	$\hat{y}_{ift} = 62.52 + 3.428t$	1.381
Davidson	Corn	$\hat{y}_{ift} = 66.19 + 2.502t$	1.290
Dickson	Corn	$\hat{y}_{ift} = 62.739 + 2.811t$	1.330
Hickman	Corn	$\hat{y}_{ift} = 81.33 + 1.697t$	1.179
Rutherford	Corn	$\hat{y}_{ift} = 61.425 + 3.107t$	1.360
Williamson	Corn	$\hat{y}_{ift} = 73.98 + 2.897t$	1.298
Cheatham	Soybeans	$\hat{y}_{fti} = 28.948 + 0.423t$	1.131
Davidson	Soybeans	$\hat{y}_{ift} = 27.431 - 0.141t$	0.947
Dickson	Soybeans	$\hat{y}_{ift} = 23.562 + 0.394t$	1.148
Hickman	Soybeans	$\hat{y}_{ift} = 23.614 + 0.696t$	1.238
Rutherford	Soybeans	$\hat{y}_{ift} = 21.340 + 0.777t$	1.282
Williamson	Soybeans	$\hat{y}_{ift} = 25.353 + 0.577t$	1.193
Rutherford	Cotton	$\hat{y}_{ift} = 367.29 + 18.566t$	1.360

Crop budgets developed by University of Tennessee Extension were consulted to determine the return to land, management, and risk for varying crop yields. Returns were assumed to increase (decrease) linearly with yield. For crops that had budgets for more than one yield goal, the return predicting formulas were piecewise linear functions that apply over distinct ranges of yield. In the case of corn, for example, budgets were available for yield goals of 120 and 150 bushels per acre. The returns are estimated by straight lines between the expected returns at each yield goal for which a budget was available, assuming also that the expected returns for a yield goal of zero bushels per acre would be zero dollars. Figures 4.2 and 4.3 exemplify these functions with graphs for corn and soybeans. After estimating linear or linear piecewise functions to predict the returns for each crop according to expected yield, each of the weighted average expected current yields was plugged into its respective formula to obtain a weighted, per acre return to land, management and risk for each crop on each crop buffer feature by county and soil mapping unit. The formulas below show how returns respond to expected yield for each crop:

$$\Phi_{ifj} = \begin{cases} \widetilde{Y}_{ifj} * 0.4523 \\ \widetilde{Y}_{ifj} * 1.21 - 80.24 \end{cases} for \begin{cases} \widetilde{Y}_{ifj} < 120 \\ \widetilde{Y}_{ifj} > 120 \end{cases} \text{ for corn,}$$

$$\Phi_{ifj} = \widetilde{Y}_{ifj} * 1.5945 \text{ for soybeans, and}$$

$$\Phi_{ifj} = \widetilde{Y}_{ifj} * 0.034621 \text{ for cotton}$$

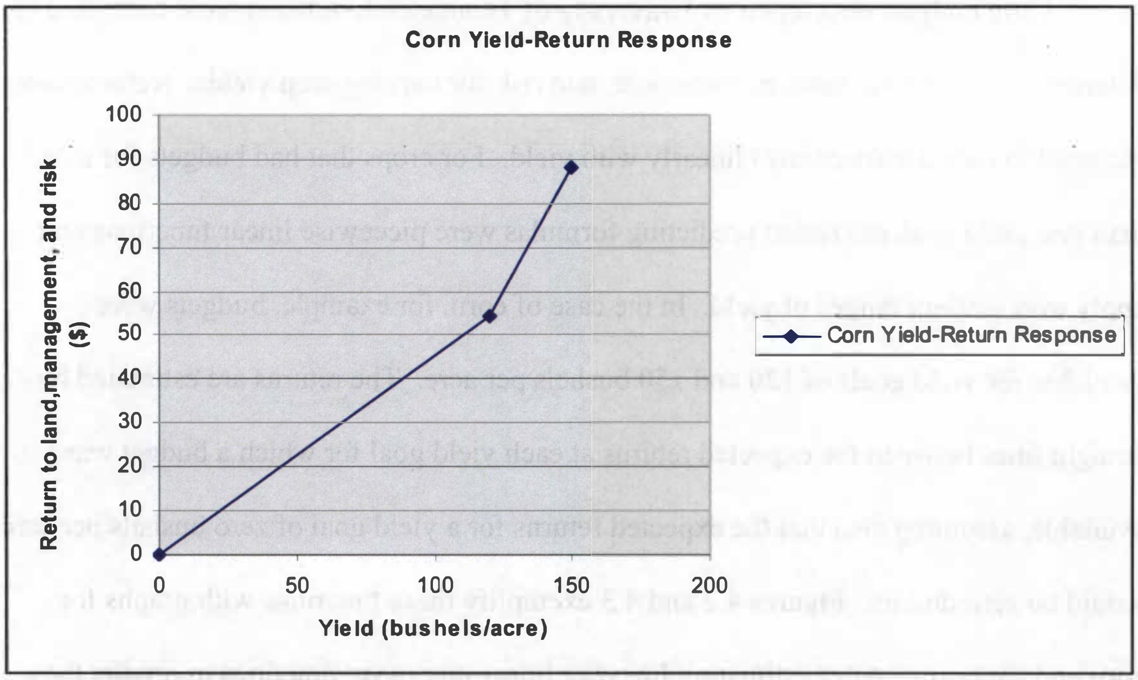

Figure 4.2: Corn Yield-Return Response

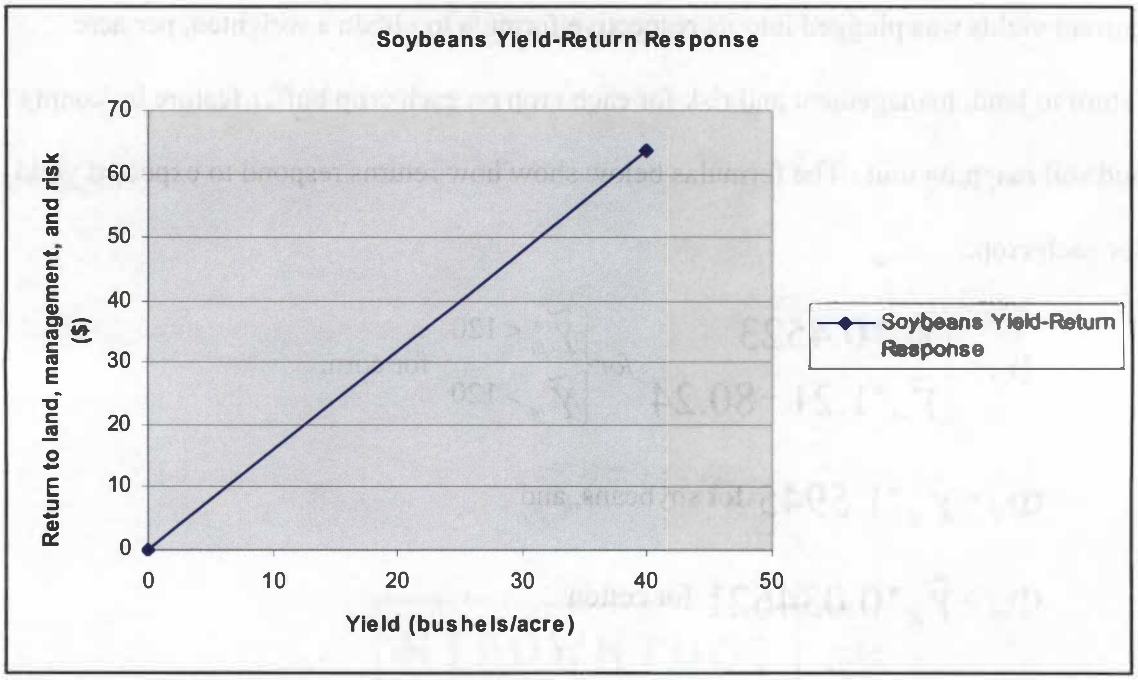

Figure 4.3: Soybeans Yield-Return Response

Where:

Φ_{ifj} is the weighted average current expected return to land, management, and risk for crop i on mapping unit j in county f,

$\tilde{Y}_{ifj}$ is the weighted average current expected yield for crop i in county f in mapping unit j, and

the numeric parameters are derived from UT Extension crop budgets for corn, soybeans, and cotton.

A weighted average return per acre was then calculated for each buffer feature by summing the products of the crop returns multiplied by their crop-mix proportions according to the county data in the following way:

$$\Pi_{fj} = \sum_{i=1}^{n} E_{if} * \Phi_{ifj}$$

Where:

Π_{fj} is the weighted average return per acre on mapping unit j in county f,

E_{if} is the proportion of cropland in crop i in county f,

Φ_{ifj} is the weighted average current expected return to land, management and risk from crop i in county f on mapping unit j, and

n is the total number of crops grown in county f.

County crop proportions used to develop these weighted average returns per acre are presented in Table 4.2. The per acre returns are the opportunity cost of the cropland, or the expected returns that producers would forego by installing a buffer strip on the land.

Table 4.2: Crop Proportions by County

County	Corn (%)	Soybeans (%)	Cotton (%)
Cheatham	29	71	0
Davidson	52	48	0
Dickson	67	33	0
Hickman	41	59	0
Rutherford	34	51	15
Williamson	36	64	0

Agricultural land classified by the 1992 NLCD as "pasture/hay" was all assumed to be pasture land for the purposes of this study, since about 70% of hay in Tennessee is grown for on-farm use in livestock operations, implying that many hayfields may occasionally be used rotationally as pasture by livestock producers (US Department of Agriculture Economic Research Service). Such being the case, fencing for stream exclusion would be required for all land designated "pasture/hay," which may cause an overestimation of the costs of the requirement of a vegetative buffer strip on such land, since not all hay fields are likely to enter a grazing rotation. Costs of livestock exclusion for grazing management are reported in US EPA's *National Management Measures for the Control of Nonpoint Pollution from Agriculture* (US Environmental Protection Agency Office of Water 2003).

The average reported cost of exclusion fencing, adjusted for inflation using the consumer price index, was $590.09 per mile. Each potential pasture buffer polygon has a perimeter in the ArcMap attribute table, which was used to calculate the required exclusion fencing for each polygon. Fencing needs were calculated according to the formulae:

$F_i = B_i - 4 * 0.02841$ *miles* if the polygon straddled the riparian feature,

and

$F_i = (B_i - 2 * 0.02841 \textit{ miles}) \div 2$ if the polygon lay on one side of the water body,

Where:

F_i is the number of miles of fence required for buffer feature *i*,

B_i is the perimeter of buffer feature i in miles, and

0.02841 mi. is the width of the buffer strip (150 ft.).

The reason for subtracting four times (two times) the buffer width from the perimeter for fields spanning water bodies (not spanning . . .) was the assumption that fencing already existed between pasture and other land uses adjacent to it, so the owner need not install fencing perpendicular to the riparian feature to achieve livestock exclusion. For pasture polygons on only one side of a water body, the owner need only install fencing on the pasture-side of the buffer, and not on the stream-side; therefore, the formula for buffers not straddling a water feature includes division by two.

The number of miles of fencing needed for each potential pasture buffer was then multiplied by the per mile cost of permanent exclusion fencing ($590.09) to obtain a total exclusion cost for each potential buffer strip. Exclusion costs for some pasture features within the buffer area were negative because their perimeters were less than 300 feet for features on one side of a stream or less than 600 feet for features straddling a stream. Features with negative exclusion costs were assumed to be land use classification errors in the 1992 NLCD, and were ignored in calculating costs. Eliminating these records should have little effect on the results because their areas were all less than one tenth of an acre, and those that were not errors are likely to be installed stream crossings for cattle or controlled stream access for watering, which could appropriately be permitted by the buffer regulation. The total exclusion cost for each buffer was divided by the acreage of the buffer to obtain a per acre exclusion cost for each potential buffer according to the following formula:

$$E_i = \frac{F_i * \$590.09}{A_i}$$

Where:

E_i is the per acre exclusion cost for buffer feature i,

F_i is the miles of fencing needed for feature i,

\$590.09 is the per mile cost of permanent exclusion fencing, and

A_i is the acreage of buffer feature i.

The opportunity cost of an acre of pasture was assumed to be approximately equivalent to the average rental rate for pasture, since the average rental rate should represent the average expected returns to land, management and risk for land owners. The USDA lists an average 2004 per acre cash rental rate for pasture in Tennessee as \$19.00 (US Department of Agriculture National Agricultural Statistics Service). No attempt was made to account for varying soil fertilities among pasture lands because the proportion of the land in different forage crops was unknown, and the average rental rate was deemed sufficient to represent the opportunity cost of pasture.

Of course, the opportunity cost (and exclusion costs for pasture) incurred by taking land out of production are not the only costs of vegetated riparian buffer strips. There will also be a cost of establishment, as well as annual maintenance costs. According to Bonham, Bosch and Pease, the average cost of an herbaceous buffer strip is \$32.79 per acre, including both the annualized establishment cost and the annual maintenance cost. This figure applies both to acreage that is currently in pasture, as well as land currently in row crops. Thus, the total per acre cost of each buffer strip can be calculated as:

$C_i = \Pi_{fj} + \$32.79$ for cropland buffers, and

$C_i = \$19.00 + E_i + \$32.79$ for pasture buffers,

Where:

C_i is the total per acre cost of buffer strip i,

Π_{fj} is the current weighted average expected per acre return from cropping in county f in mapping unit j, where buffer feature i lies,

\$19.00 is the expected return from the pasture land (the pasture rental rate),

E_i is the per acre exclusion cost for pasture buffer i, and

\$32.79 is the annual cost of maintenance plus the annualized establishment cost.

Having estimated a per acre total cost, as well as acreage, for each potential buffer strip, it was then possible to calculate the estimated total cost of a mandatory regulation requiring the installation of 150 foot buffer strips on all agricultural land bordering streams in the Harpeth River Watershed.

The total cost of this regulatory approach could be calculated at this point by multiplying the acreage of each buffer by its total per acre cost and summing all these products as follows:

$$T = \sum_{i=1}^{n} C_i * A_i$$

Where:

T is the total cost of mandatory 150 ft riparian buffer strips on all agricultural land,

C_i is the per acre cost of buffer feature i,

A_i is the acreage of buffer feature i, and

n is the total number of potential buffer strips in the Watershed.

The next step, which should be more interesting to advocates of tradable emissions rights, is to plot out this supply curve. Doing so entails the summation of the acreage of all potential buffers that would be supplied at various "prices" (using $5.00 price intervals) assuming that price equals marginal cost, or the cost of one additional acre of buffer. This process is described mathematically by the following formula:

$$S_p = \sum_{i=1}^{n} A_i$$

Where:

p is a price level that is a multiple of $5.00,

S_p is the acreage of all buffers supplied at price p,

A_i is the acreage of buffer feature i, and

n is the total number of pasture and row crop features for which $C_i \leq p$.

This calculation was performed between the lowest and the highest per acre prices of all features to obtain a series of points on the supply schedule. Linearity is assumed between consecutive points to calculate integrals under various points on the curve.

Results

Clipping the 1992 NLCD agricultural land use features to the 150 foot buffer around all NLCD riparian features in ArcMap and calculating the areas of all potential buffer strips yielded the result that in the Harpeth River Watershed, approximately 12,605 acres, or eight percent, of agricultural land would be consumed by a regulation requiring 150 foot riparian buffer strips on all agricultural land throughout the watershed. Approximately 8,663 acres of this potential agricultural buffer land is classified as "pasture/hay" by the land use data. The balance of the agricultural land use (3,942 acres) is classified as "row crops."

The STATSGO Database shows portions of 14 different soil mapping units within the watershed, and provides the proportions of each of these mapping units covered by each of its soil types. Yields for each crop are also provided for all soil types upon which the crop was grown in 1994. Using these data, and the process described in the methodology section to create weighted average yields for each crop in each soil mapping unit provides corn yields between 71 and 114 bushel per acre, soybean yields between 31 and 41 bushels per acre, and cotton yields between 471 and 750 pounds per acre.

Multiplying the yield inflators (I_{if}) by their respective 1994 soil mapping unit weighted average yields according to the county in which the potential buffer features lie returns corn yields ranging between 83 and 158 bushels per acre, soybean yields ranging from 31 to 52 bushels per acre, and cotton yields ranging from 642 to 1039 pounds per acre. Each potential buffer strip's weighted, inflated yield was then plugged into its respective formula to predict the expected returns from the crop on the buffer strip. Per acre returns range from $37.80 to $110.44 for corn, $51.97 to $83.06 for soybeans, and $22.22 to $35.98 for cotton, depending on the soil mapping unit and county in which the potential buffer strip lies.

After having estimated per acre returns for each crop on each buffer strip, the returns for each buffer strip were weighted by crop proportions in their respective counties so that a weighted average return to land, management and risk (an opportunity cost) might be obtained for each cropland riparian buffer (Table 4.3). This number is assumed to be the per acre opportunity cost of each potential buffer. Per acre annualized establishment and annual maintenance costs of riparian buffer strips were then added to the opportunity costs of each buffer strip. The results are the total per acre costs of cropland riparian buffers, and they range from $77.92 to $117.87, with a mean of $98.50. The opportunity cost of pasture was assumed to be equivalent to the average rental rate for pasture in the state of Tennessee, or $19.00 per acre. Fencing costs are the other major component of buffer costs unique to pasture land. The formulas for calculating fencing needs and the per mile fencing costs reported in the methodology section were used to calculate fencing costs for each potential buffer strip. After eliminating all potential buffer strips with negative cattle exclusion costs, and dividing the fencing costs of each buffer by its acreage, it was determined that the per acre exclusion costs range from $0.02 to $299.78 for the remaining 8,576 acres of pasture that lie within the buffer area. It should be noted here that eliminating buffer strips with negative cattle exclusion

Table 4.3: Weighted Average Return to Land, Management, and Risk for Cropland

County	Mapping Unit	Acres	Opportunity Cost ($/ac)
Cheatham	TN054	72	60.05
Cheatham	TN064	34	84.05
Cheatham	TN073	173	55.50
Davidson	TN054	4	51.31
Davidson	TN064	101	80.34
Davidson	TN069	7	53.28
Davidson	TN071	6	71.75
Davidson	TN072	5	51.93
Dickson	TN048	26	57.40
Dickson	TN054	193	55.71
Dickson	TN060	649	53.69
Dickson	TN073	36	52.55
Hickman	TN060	41	89.31
Rutherford	TN054	1	59.61
Rutherford	TN062	84	56.90
Rutherford	TN064	44	84.22
Rutherford	TN067	25	55.32
Rutherford	TN076	231	52.46
Williamson	TN054	93	60.46
Williamson	TN060	217	57.61
Williamson	TN062	119	57.51
Williamson	TN064	877	85.08
Williamson	TN066	44	64.89
Williamson	TN067	328	55.73
Williamson	TN069	170	66.68
Williamson	TN071	362	75.91

costs eliminated virtually all pasture features with areas less than 0.1 acre. In fact, it seems that buffer features with areas very close to 0.1 acre are likely to be near the ends of the range of per acre exclusion costs, suggesting the possibility of some systematic error, the effect of which is reduced for larger buffer areas. For each buffer feature, opportunity cost (the rental rate) plus the exclusion fencing price were considered the total cost of an acre of pasture. Annualized establishment costs and annual maintenance costs for the vegetative buffer strips were then added to the exclusion and opportunity costs, resulting in total per acre costs of pasture buffers ranging from \$51.81 to \$351, with a mean of \$111.97.

The cost of taking all the buffer land (including pasture and cropland) out of production was then calculated using the formula for *T* mentioned under the previous subheading, yielding the result that the total cost of a regulation requiring 150 foot grassed riparian buffer strips on the stream edges of all agricultural fields in the Harpeth River watershed would be approximately \$1,318,695 annually to remove about 12,245 acres from production and establish and maintain riparian vegetation thereon. This cost burden would clearly be distributed disproportionately among agricultural producers, as some might even remain unaffected because their land does not border riparian features. The supply curve was approximated by summing all the agricultural acreage that would be offered as riparian buffers at various prices, moving through the range of per acre costs of buffer strips at intervals of \$5.00, and the supply curve is shown in Figure 4.4. If society wanted to purchase full implementation of buffer strips, the total payment to agricultural producers would be approximately \$4,347,195, and the buffer suppliers

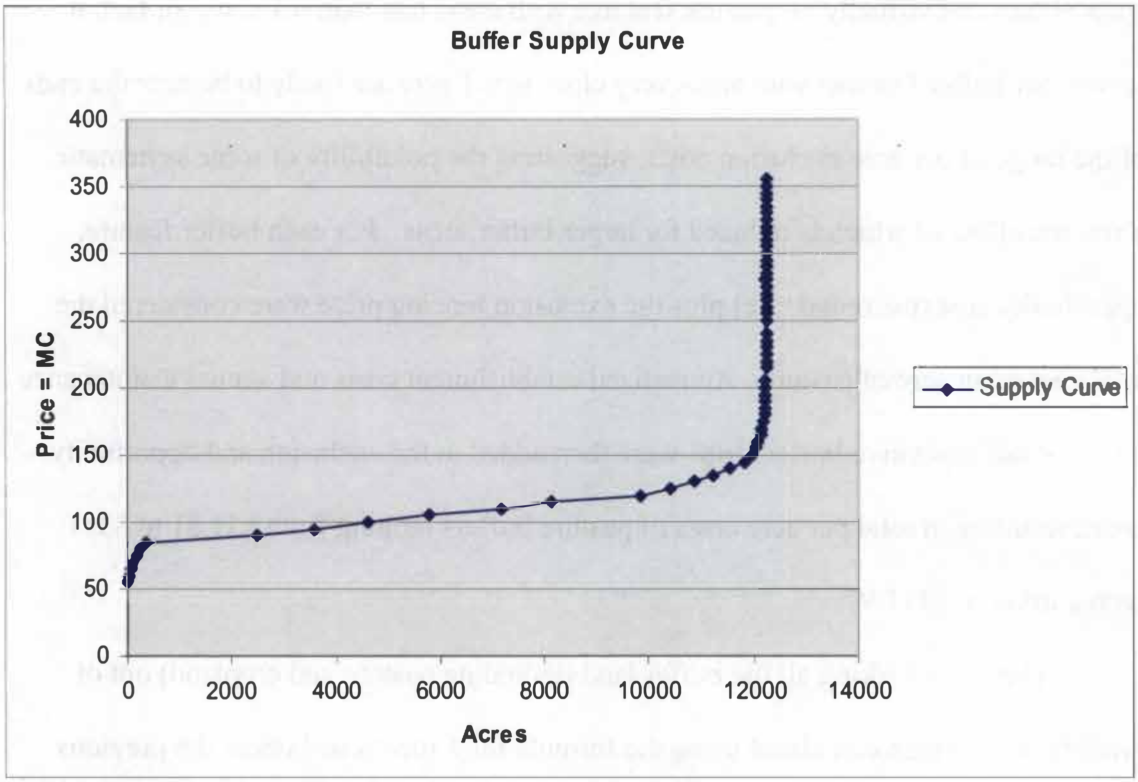

Figure 4.4: Buffer Supply Curve for the Harpeth River Watershed

would collectively receive $3,028,500 worth of producer surplus, assuming the law of one price. On the other hand, because 97.9% of the buffer acreage is available at a cost below $150.00, society could achieve almost the same level of implementation at a much lower total cost, since the supply curve becomes nearly vertical when about 12,000 acres of buffer strips have been purchased. The total cost to agricultural producers of installing buffer strips on the least-cost 11,990 acres (97.9% implementation) would be approximately $1,276,200. This level of voluntary implementation would require a payment of $1,798,850 to agricultural buffer producers, providing them a producer surplus of $522,650.

Conclusions

This study has presented a methodology for developing a supply curve for agricultural riparian buffer strips, which have the potential to greatly reduce nutrient content in agricultural runoff. This supply curve has the additional benefit of establishing a fairly narrow price range for a riparian buffer strip, between $85 and $150 for about 94.9% of the potential agricultural buffer acreage. Though well-functioning markets generally establish their own equilibrium prices and quantities, such methodology and the predictive information it provides may be valuable in seeking to establish market-based tradable discharge permit systems. The next step in predicting market behavior is to use some in-depth water quality modeling to attempt to convert this buffer supply curve into an NPS nutrient reduction credit supply curve. Further research could quantify PSs' marginal willingness to pay, or demand, for NPS nitrogen reduction credits in the watershed.

There are obvious limitations to this study, many of which result from data limitations. For example, the riparian shape files from the NHD are line features, as opposed to polygon features, meaning that the actual interfaces between agriculturallands and waterways are sometimes not found within a 150 foot buffer around the riparian line features. This means that available buffer acreage may be under represented in some areas, especially on the main stem of the Harpeth River, where the water is wide, as they say. Yet, agricultural activities are not nearly as abundant near the main stem as they are in the upper reaches, or eastern portion, of the Harpeth River Watershed. Thus, this data limitation may not be as pronounced as one might expect if agricultural activities were distributed homogenously throughout the watershed. The issue of error in stream, land use interface identification also contributes to error in the calculation of exclusion fencing costs for pasture land, which will depend on the shape and size of the potential buffer feature, which are partially determined by the proximity to the riparian line features. Ideally, data that shows the shapes and areas of all riparian features in the watershed should be used for a study like this; however, such data are not currently available for a sufficiently large region.

This methodology does not account for the cost of alternate watering systems for cattle that are to be excluded from stream access. These costs ought to be included, as they would be incurred as a direct result of complete cattle exclusion; however, in order to calculate such costs, we would need to know how many cattle are on how much pasture, as well as which elements of the "pasture/hay" land use classification are actually pasture, as opposed to hay.

Additionally, farmers who establish buffer strips are also likely to modify practices elsewhere in their operations. These modifications may have adverse results or provide unexpected benefits for society as a whole. This study does not account for the private benefits to agricultural producers brought about as a result of reduced erosion and increased stream bank stability provided by buffer strips. These private benefits, inasmuch as agricultural producers believe they exist, will reduce their perceived opportunity costs, thus reducing the total cost of any given level of buffer strip implementation below those described by the supply schedule. To assess these potential private benefits, extensive in situ work and modeling would need to be performed so that the reduced effects of erosion could be estimated and monetized for each potential buffer strip. Such modeling could also be used estimate the differential environmental impacts of buffer strip implementation on distinctive potential buffer strips. This information could be used to develop a supply curve for nitrogen reduction credits from NPSs.

However, the supply of agricultural riparian buffer strip will change substantially with the passage of time, especially in a rapidly developing area like the Harpeth River Watershed, which is home to Nashville and its various extensions, which are rapidly expanding. Thus, the potential for agricultural NPS reductions changes as land use changes. In this particular watershed, as in most, NPS remains the major source of nutrient loadings, however, an increasing proportion of these loadings is coming from urban and suburban land, as agricultural land is being converted to new uses. These situations are not static, and equilibrium for a market in buffer strips will be constantly changing in watersheds that are rapidly developing.

Chapter V

Summary & Conclusion

WQT may be an important part of a cost-effective solution to persistent water quality impairments caused by nutrients and other oxygen demanding pollutants in some watersheds, though PSs-PSs trading may not be viable at all in other areas. The Question to answer prior to attempting the establishment of such a program is whether balanced supply and demand for nutrient credits will exist in a potential market area. Chapter III of this thesis lays out a methodology that results in PS-NPS WQT scores for the various potential trading regions of Tennessee, which are defined within eight-digit HUC boundaries by the flow of water across 12-digit HUC boundaries. These potential trading areas are ranked based on the relative nitrogen contributions from PSs and NPSs to their water quality impairments. These relative loads are based on measures of the extent of loadings at the source and the distance of the source from the impairments. Three potential trading areas (South Fork Forked Deer, Stones, and North Fork Forked Deer River Watersheds) scored well above the rest, suggesting that these three watersheds merit particular attention to the potential for WQT to assist in reducing the cost of compliance with ambient environmental water quality standards. It would be prudent to assess these areas in greater detail with regard to their suitability for WQT in conjunction with the development of nutrient, organic enrichment, and low dissolved oxygen TMDLs for their water bodies.

Chapter IV presents the methodology and results of a process designed to approximate a supply curve for agricultural riparian grassed buffer strips for the Harpeth River Watershed in Middle Tennessee. Unfortunately, the process does not directly assess the supply of NPS nitrogen credit reductions from agriculture, but the derived supply curve bears some relationship to that for nitrogen reduction credits.

The papers that comprise this thesis have the potential to improve the prospects for cost-effectively achieving the environmental quality standards that government agencies have set. If WQT has a place in the solution, this work has identified some targets for future study.

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Vita

David Carlos Roberts was born in Boone, IA on 28 March 1979. After growing up in such far flung places as Hawaii, Indonesia, Tennessee and Texas, David came to Knoxville to attend The University of Tennessee in June 2000. He graduated Summa Cum Laude in December 2003 with a B.A. in Spanish and International Agricultural Economics through the University's Language and World Business Program. During his undergraduate study, David was named Outstanding Sophomore of the Year in the College of Arts and Sciences (2001) and Outstanding Junior of the Year (2002) by Phi Kappa Phi. David was named Top Graduate of the University of Tennessee's College of Arts and Sciences (Humanities Division). David is a member of various scholarly organizations, including Phi Kappa Phi, Phi Beta Kappa, Sigma Delta Pi, Gamma Sigma Delta, and the National Society of Collegiate Scholars.

David began work on his M.S. in Agricultural Economics at the University of Tennessee in January 2004, and has had the pleasure of presenting a poster at the Annual Meetings of the Southern Agricultural Economics Association (2004), and a paper at the Fifteenth Tennessee Water Resources Symposium (2005). David is currently finishing his M.S., and has plans pursue a Ph.D. in Agricultural Economics at Oklahoma State University beginning in spring 2006.

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