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Agricultural nonpoint source pollution and water quality trading: empirical analysis under imperfect cost information and measurement error

Adriana Mihaela Valcu
Iowa State University

Agricultural nonpoint source pollution and water quality trading: empirical analysis under imperfect cost information and measurement error

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Adriana M. Valcu

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Program of Study Committee:
Catherine Kling, Major Professor
Joseph Herriges
Tanya Rosenblat
John Schroeter
Quinn Weninger

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

DEDICATION

To my family,

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Change, Mitigation, and Adaptation in Corn-Based Cropping Systems.", and the USDA's Economic Research Service "The Supply of Greenhouse Gas Offsets from Agriculture and Their Water Quality Effects in the Upper Mississippi River Basin."

ABSTRACT

Water quality problems associated with agricultural nonpoint-source pollution remain significant in the majority of US watersheds. In this dissertation, I present a theoretical model of water quality that captures the main characteristics of agricultural pollution (the unobservability and the interactions between the field-level emissions, the imperfect knowledge of the abatement costs), propose and empirically estimate a simplified proxy model for the complex process that characterizes the fate and the transport of agricultural pollutants, and apply this model in a variety of empirical studies to evaluate alternative policy programs designed to improve water quality. Under a linear approximation of the abatement function, more flexible policies like the performance standard or trading program may outperform a command-and-control program in terms of abatement costs, but they may also result in the non-attainment of the abatement goal. However, the incentive-based policies can overcome, partially or totally, the issue of cost asymmetries, since the regulator does not need to know the farm-level abatement costs.

I propose and estimate an approach for linearizing the abatement function using a system of point coefficients that measure the impact of an abatement action on the overall abatement level. The point coefficients are estimated for nitrogen and phosphorus with consideration that the two pollutants have separate abatement functions.

The empirical assessments of the proposed policies for two agricultural watersheds in Iowa show an overall good performance of the incentives based programs: the deviations from the abatement goals are not significant and sizable cost savings relative to the command-and-control programs are realized. A robustness analysis shows that the results are consistent across different: (a) pollutants (nitrogen and phosphorus), (b) sets of point coefficients (field-specific level, subbasin-specific, or watershed-specific), and (c) the distribution of historical weather. The

point approximation procedure is extended to two pollutant markets, where each market uses a separate set of point coefficients. Given that the same abatement actions that have the potential to increase the amount of carbon sequestration in soil, the point-based trading program is extended to allow trading participants to enter a market for carbon, including selling the carbon offsets associated with the abatement actions.

CHAPTER 1. GENERAL INTRODUCTION

The year 2012 has marked four decades since the main US regulatory act for improving water quality was enacted. In spite of the numerous nascent programs that followed, water quality pollution from agricultural activity remain a significant problem, particularly in watersheds dominated by row crop production[1].

The goal of my dissertation is to propose and evaluate policies that address agricultural nonpoint sources. Specifically, my objectives are to:

1. Present a theoretical model of water quality that captures the main characteristics of the pollution within an agricultural watershed.
2. Propose and empirically estimate a simplified proxy model for the complex process that characterizes the fate and transport of agricultural pollutants such as nitrogen and phosphorus across the watershed.
3. Apply this model in a variety of empirical studies to evaluate alternative policy programs designed to improve water quality.

In the second chapter, I provide a literature review of the economics of nonpoint source pollution associated with agricultural activity. First, I introduce the nonpoint source characteristics and review the different policies approaches discussed in the literature on water quality trading involving single or multiple pollutants. Next, I explore the literature that links the

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[1] "A watershed is the area of land where all of the water that is under it or drains off of it goes into the same place". Watersheds come in all sizes and shapes. In the continental US, there are more than 2, 100 watersheds. http://water.epa.gov/type/watersheds/whatis.cfm.

nonpoint source programs and the programs for carbon offsets. Finally, I provide a brief description of the simulation models and optimization tools used in my empirical analysis.

In the third chapter, I propose a simple model of pollution related to agricultural activity that captures three types of fundamental characteristics of agricultural nonpoint sources: imperfect information on the abatement costs of individual farms, difficulties in observing pollution or abatement activities at the farm level, and difficulties in measuring the emissions leaving the field. Specifically, I consider a watershed where agriculture is the main source of pollution. The regulator or an environmental authority decides to reduce the total level of pollution by requiring each field to adopt a specific set of conservation practices or abatement actions.

I begin by assuming that the regulator and the farmers have the same cost information, perfect information on the emissions leaving the field and on the water quality production function as well. Next, I relax the assumption that the regulator has perfect cost information, while keeping the other assumptions constant. In the third case, I assume that for the ease of implementation of an incentive-based policy, the water quality production function is approximated as a linear combinations of known field level emission reductions. For the last case which represents the focus of my dissertation, I propose a method for efficiently identifying a system of points to approximate both the edge-of-field reductions and the impact on the total ambient level of pollution associated with the abatement actions implemented at the field scale.

In the fourth chapter, I empirically evaluate three different abatement action based policies for improving the water quality where the policies are implemented using the system of point coefficients proposed and estimated in Chapter 2. Next, using a detailed biophysical watershed based water quality model in conjunction with a range of estimates for the abatement costs, I demonstrate the efficiency tradeoffs implied by the use of a point based system by comparing its outcomes to the least cost allocation of conservation practices.

To establish and identify a baseline of comparison, I first identify the least cost solution under the unrealistic assumptions that the amount of emissions leaving a source can be measurable and observable under alternative conservation practices, and that the full fate and transport of these emissions (i.e., the water quality production function) is fully specified and known. Next, I address the design and performance of three practice-based policy approaches, ranging from the command-and-control approach mandating practices, to the more flexible performance standard approach where farmers are free to select the optimal mix of on-farm abatement or conservation practices, to a fully flexible approach where credits or points for conservation practices are freely tradable. Under a points-based trading system farmers are required to undertake abatement actions that accrue a sufficient number of points per field or per acre basis. If, by undertaking a conservation practice they generate more points than their minimum requirement, they can sell the extra credits to other farmers in the watershed who do not meet their requirements (Kling, 2011). I evaluate the performance of the three policy approaches first by considering that the regulator and the farmers have the same cost information, and next by considering that the regulator does not know the field-level abatement costs, but does not know the distribution of these costs and uses the moments of this costs distribution (i.e.; mean) to find the least cost allocation of the abatement actions in the watershed.

Finding the best solution is not trivial because the underlying water quality production function is highly non-linear and non-separable. To overcome this difficulty, I use evolutionary algorithms to approximate the solutions. To solve for the trading outcome of the point-based trading system, I use the mixed integer algorithms that incorporate the discrete nature of the choices (conservation practices) and the continuous nature of the trading system.

The proposed point-based trading system has the potential to be implemented for the case of a single pollutant (e.g., nitrogen or phosphorus) or multiple pollutants (e.g., nitrogen and phosphorus). I compare the efficiency of the three practice-based approaches assuming first that the policy approaches are designed only for one of the two pollutants, and next by considering the case where both pollutants are simultaneously targeted. I empirically evaluate the performance of the above approaches using watershed-based water quality model calibrated for two typical Midwestern watersheds.

Water quality and improved soil are necessary qualities of healthy watersheds, which provide local ecosystem services such as improved fishing and wildlife habitat. At the same time, soil carbon sequestration is a global ecosystem service and plays an important role in reducing greenhouse gases (GHGs). In the fifth chapter, I analyze the impact of a carbon offset market on the efficiency of an already established water quality program. My analysis departs from previous research by considering the participation in a carbon offset market, a global environmental good, as a co-benefit of a water quality trading program with local effects. This chapter highlights the changes in the total cost of achieving an ambient level for water quality when farmers are allowed to participate in two parallel markets: a water quality trading program, and a carbon market where they can sell carbon offsets associated with their abatement actions. The water quality trading program is a local trading program (i.e., at the watershed or state level), while the carbon market is a wider market (i.e., nationwide) with no specific cap requirements at farm level.

CHAPTER 2. LITERATURE REVIEW

2.1. Introduction

The Federal Water Pollution Act—the main US main regulatory framework for water pollution control—has been in use for four decades (Shortle and Horan 2013). The act, also known as the Clean Water Act, emerged as a consequence of the rising concerns related to the water quality in the late 1960s. The legislation places stringent regulations on industrial and municipal polluters (i.e., point sources), but does not specify any regulations for agricultural polluters (i.e., nonpoint sources). In spite of the numerous efforts in reducing water pollution, water quality remains a significant problem, as underlined by several studies conducted by the US Environmental Protection Agency (EPA), such as the National Summary of Assessed Water Report and the National Rivers and Streams Assessment 2008–2009.

The latest National Summary of Assessed Water Report assessed 28% of rivers and streams, and 43% of the lakes in the United States. Of the assessed rivers and streams, 53% were found to be impaired; and of the assessed lakes, 82% were found to be impaired for their designated uses. The report designates agriculture as being the leading source of river and stream impairments, the third largest source for lake and pond impairments, and the fifth largest contributor of wetland impairments.

The National Rivers and Streams Assessment 2008–2009 (NRSA) is the first statistically based survey on water quality of all rivers and streams. The survey reports that 55% of the nation's river and stream miles do not support aquatic life because of the high content of phosphorus and nitrogen, with 23% being in fair condition, and 21% being in good condition. Overall, the study found that the nation's river and streams are under "significant stress." The study also stated “[r]educing nutrient pollution and improving habitat will significantly improve the biological health of the rivers and streams and support important uses as swimming and fishing.” The study also suggested that in spite of the fact that many actions have been taken towards improving water quality, “…we need to address the many sources of pollution—including runoff from urban areas, agricultural practices, and wastewater—in order to ensure healthier water for future generations.”

Both studies pointed out the significance of water pollution commonly produced by agricultural pollutants such as nitrogen and phosphorus, and restate the fact that achieving the desired standards of water quality cannot be done through controlling the point source only (Ribaudo 2009). Emphasizing the contributing role of agriculture to water pollution, Ribaudo et al. (2008) noted that the complete elimination of nitrogen point sources across the United States would reduce the total nitrogen emissions by only 10%. This fact is not surprising, given that 71% of the US crop land (more than 300 million acres) is located in watersheds where at least one of the most common surface water pollutants is above the accepted levels for aquatic activities (Ribaudo 2009).

As water quality issues became a stringent problem with social and environmental implications, they started receiving attention from the environmental economists. In the next section, I present a short review of the economics of water quality focusing on the relevant issues stemming from the agricultural activity.

2.2. The Economics of Nonpoint Source

Traditionally, urban and industrial polluters are identified as point sources, while agricultural polluters are identified as nonpoint sources. Over time, the industrialized countries have shifted their attention from water pollution created by point sources towards the water pollution created by agricultural runoff (Olmstead 2010).

As defined by EPA, point sources are “any discernible, confined, and discrete conveyance, including but not limited to, any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, landfill leachate collection system, or vessel or other floating craft from which pollutants are or may be discharged.” Thus, point sources can be identified and their emissions can be properly measured at a relatively low cost. It is assumed that there are no stochastic elements or errors in emissions measurement. This makes the polluters easily to be identified and thus made accountable for their emissions. Examples of point sources are industrial facilities and sewage treatment plants—they emit pollutants from a fixed and identifiable point such as a pipe or outfall.

Nonpoint sources result from “…precipitation, land runoff, infiltration, drainage, seepage, hydrologic modification, or atmospheric depositions. As runoff from rainfall or snowmelt moves, it picks up and transports natural pollutants and pollutants resulting from human activity, ultimately depositing them into rivers, lakes, wetlands, coastal waters, and groundwater.” (EPA 2003). As the definition suggests, nonpoint source pollution or runoff is stochastic in nature because: (*a*) it is a weather driven process, and (*b*) it involves a complex transportation process from the production’s site to the location where the ambient pollution can be measured. Nonpoint sources do not discharge at particular receptors, their emissions or loadings being diffuse. Thus, nonpoint source emissions’ diffuse nature makes the loadings more difficult and more costly to measure. Nonpoint sources can be identified as emissions coming from mobile sources, leaching, and runoff from farm fields. There is uncertainty about the contribution of each polluter to the total amount of pollution.

Agriculture is the primary contributor of runoff creating nonpoint source pollution and the main cause of water pollution in the United States (EPA 2007). Three main forms of agricultural nonpoint sources have been identified: sediments, nutrients such as nitrogen and phosphorus, and pesticides.

To sum up, point and nonpoint emissions differ in the following: (*a*) point loadings are deterministic while the nonpoint ones are stochastic; (*b*) the effectiveness of control efforts is certain for point sources but uncertain for nonpoint sources; and (*c*) loadings from point sources can be measured directly, whereas nonpoint sources loadings can be estimated (Malik et al. 1993).

As mentioned earlier, the Federal Water Pollution Control Act of 1972 emerged as a consequence of the rising concerns related to the water quality in the late 1960s.[2] The Act establishes the National Pollution Discharged Elimination System Permit (NPDES), requiring each point source to comply with quantitative effluent limits established for each pollutant. To date, the point source compliance with these standards has been successful, but there is evidence that the gains from controlling them are constantly diminishing (Olmstead 2010). In spite of the increasing evidence regarding nonpoint sources as the main contributors to water pollution, the Clean Water Act does not directly address them. There are two sections within the Act that provide recommendations for nonpoint sources: section 319 of the 1987 Clean Water Act, and

[2] The fire on the Cuyahoga River, Ohio, in 1969 was the worst fire since the mid-1800s (Fisher and Olmstead 2013)

section 303 (d) also known as Total Maximum Daily Loads (TMDL). Section 319 provides the legal framework for funding voluntary or monitoring programs, while the TMDL section requires each state to establish pollution budgets for each water body that does not meet the ambient water quality standards for designated uses (i.e., recreational public and industrial water supply use).

2.3. Policies for Mitigating the Nonpoint-source Pollution

While the regulatory framework does not allow for direct enforceable caps on the agricultural nonpoint sources, environmental and agricultural economists have been studying the design of efficient programs to address nonpoint sources water pollution from agriculture for decades. Over time, different policy options have been proposed to mitigate the nonpoint source pollution. These policies can be characterized as voluntary programs, command-and-control programs, and economic instruments such as input and ambient taxes and tradable permit systems.

The voluntary programs can be characterized as: (*a*) voluntary self-regulating actions undertaken by the polluters; (*b*) negotiated contracts between environmental regulators and polluters where the participation is determined by both parties; and (*c*) voluntary government programs where a federal or state authority establishes eligibility and the rewards criteria (B.M. Dowd et al. 2008).

Command-and-control programs have been successfully used to regulate point sources, but since there is a lack of a regulatory framework, there have been few attempts for regulating nonpoint sources. Command-and-control programs for nonpoint source can be implemented by requiring farmers to adopt different measures to reduce the emissions, such as the implementation of conservation or best management practices. Imposing performance standards by which the polluters must comply is another way to impose a command-and-control policy (B.M. Dowd et al. 2008).

Price instruments such as input and ambient taxes have also been pointed out as possible policy instruments for nonpoint sources. An input tax can be imposed on a farm input such as the use of chemical fertilizers. It has been shown that input taxes, set at the point where they equal the abatement costs are an efficient way to achieve environmental standards (Shortle and Horan 2001). Griffin and Bromley (1982) showed, using an input-based model for nonpoint sources, that taxing the inputs that increase pollution and subsidizing the inputs that decrease it replicates the tax on pollution. Shortle et al. (1998) showed that a targeted tax, where polluters who contribute more are taxed more heavily, is more cost effective than a uniform tax.

Effluent taxes for points sources are a common way for controlling water pollution being implemented in several countries (Olmestead 2010).Segerson (1988) was the first to advocate ambient or subsidies taxes for nonpoint source. An ambient-based tax scheme is based on the group performance rather than an individual one. It penalizes the polluters when the ambient pollution level goes beyond a given level and rewards them when the ambient pollution level is lower than the standard. Shortle and Horan (2001) pointed out that an ambient tax might not be efficient in the case of nonpoint source, since the emissions are influenced by the weather and stochastic elements, and it could not recognize the actions taken by the farmers located downstream the watershed.

Additionally, Vanden-Fisher and Olmstead (2013) and Shortle (2013) are two of the most recent surveys that provide a comprehensive review of the policy instruments for water quality pollution with particular focus on nonpoint source pollution. Besides assessing the current status of the water quality programs in the US and worldwide, the surveys provide useful insights about lessons learned so far as well about the research needed for improving the efficiency of the policy.

Market-based instruments, such as trading programs, have been regarded as successful instruments for a broad range of environmental problems such as air pollution control, habitat protection and resource compensation. Based on the historical achievements of the Acid Rain Program, which used tradable permits for sulfur dioxide emissions, market-based instruments have been proposed to address water pollution control, as they can create a more cost effective approach for achieving the environmental goals. Next, I provide a short review of the evolution of permit-based trading systems—how they evolved, and how they relate to the water quality.

2.4. Overview of Trading Systems to for Pollution Externalities

The idea of using market mechanisms to correct economic externalities goes back to Coase (1960). A few years later, Dales (1968) applied the idea of market mechanism to the water pollution problem. Montgomery (1972) was the first to provide the theoretical foundations of a trading market based on pollution permits.

An ambient pollution system (Montgomery 1972) assumes that permits are issued for each receptor. There is a market for each receptor, and a polluter needs to have a portfolio of permits to cover all receptors. If a polluter changes his behavior, he needs to find a trading partner for each receptor. Trading ratios are determined by an exogenous transfer coefficient matrix. Transaction costs are high because there is a market for each receptor point. In addition, Krupnick et al. (1983) showed that in order to have an equivalence between the least-cost and market solutions, the initial allocation of permits must make the pollution constraint binding at all receptor points, otherwise the solutions diverge if the actual water quality is higher than the environmental standard (i.e., the water quality standard is not binding). Only when the environmental constraints are binding at all receptor points are all trade possibilities exhausted. Hung and Shaw (2005) showed that it is unusual to make all standards binding, which means that the most cost effective way is not always achieved.

Krupnick et al. (1983) proposed a solution to fix this inconvenience: a permit offset system. In this framework polluters are free to trade as long as environmental standards are not violated at any receptor point. If this is the case, then the trade takes place with trading ratio given by the ratio of the two sources' transfer coefficients, thus the trading ratios are determined endogenously. Polluters are required to have permits just for the receptor points where quality will be impaired as a result of an increase in emissions, thus transaction costs are lower than in an ambient permit system. Other caveats associated with this system are free riders and high transaction costs.

An exchange rate trading system was proposed as an alternative to endogenous trading ratios. Trading ratios are set up exogenously as being equal to the ratios' of the polluters marginal abatement cost in the least-cost solution. The burden of the cost is transferred to the environmental authority that needs information about polluters' marginal costs. There is the risk that some initial environmental constraints will be violated after the trade takes place.

Most research to date has been focused on modeling trading systems that included either only point source or both point and nonpoint sources (Montgomery 1972; Krupnik et al. 1983; Shortle and Abler 1997; Huang and Show 2005).Trading systems between point and nonpoint sources are based on the fact that, in general, the abatement costs for nonpoint sources is lower than the abatement costs for point sources. There are two main questions related to these trading systems: what to trade, and the ratio at which one can trade. Regarding what to trade, two designs have been proposed. In the first design, increments in point sources emissions are traded for reductions in nonpoint-sources estimated loadings; this system has been regarded as a trading-emissions-for-loadings system. In the second design—a trading-emissions-for-inputs system—point sources emission permits are traded for nonpoint-source (NPS) permits, which in turn restrict the use of polluting inputs (i.e., fertilizer), or influence the adoption of a conservation practice (Horan et al. 2002). Since the point and nonpoint sources might have different contributions to total pollution, a trading ratio must be determined in order to achieve water quality goals.

The trading ratio reflects the rate at which nonpoint-source emission reductions are traded for point-source emission increases. Because estimated loadings are imperfect substitutes for nonpoint-sources emissions, the ratio should be different than one. Existing literature provides little guidance, but suggests that factors such as the relative marginal contributions of point and nonpoint sources, the degree of environmental risk impacts, correlations between environmental and cost relationships, and the overall level of heterogeneity associated with point and nonpoint sources influence the magnitude of the optimal trading ratio (Horan et al. 2002). An optimal ratio should encourage more control for the source whose emission generates the most risk and is most costly to be controlled.

The trading ratio can be equal to, greater than, or less than one. A ratio equal to one implies indifference at the margin between the sources of control. Ratios less than one imply a low abatement cost of nonpoint control relative to point sources control, and thus preference for nonpoint-sources reductions, and the opposite is true for a ratio greater than one (Shortle and Abler 2005).

Horan and Shortle (2005) explained why the observed trading ratio should be greater than one, and not less than one as theory predicts. By setting up a model for point and nonpoint pollution trading, they found that the ratio is higher because the number of point permits is decided by a federal authority, whereas the state authority has to determine the number of nonpoint sources permits and trading ratios in such a way as to achieve environmental goals. Their results assume away the uncertainty regarding the effectiveness in the reductions of nonpoint-source emissions, instead accounting only for uncertainty related to weather and other environmental drivers.

Hung and Shaw (2005) proposed a system for trading pollution discharge permits in a river area. Their model proposed an exogenous trading ratio. Dividing the river into many zones, and accounting for the unidirectional characteristic of river flow, the trading ratio defines the amount by which a polluter can increase emissions if he buys *t* permits from a polluter situated in another zone. Their model is cost effective in achieving environmental goals, and given the assumptions that are made, it can get rid of issues like transaction costs and hotspots. One major critique to this model is the fact that it considers just point source emissions. The trading ratio system model proposed by Huang and Shaw (2005) brought significant improvements regarding transaction costs, least-cost effectiveness, and free rider or hotspot issues to previous existing trading systems such as the ambient permit system, the permit offset system, or the exchange ratio system; however, it does not incorporate uncertainty. Uncertainty is strongly related to nonpoint pollution sources and nonpoint aspects of the loadings are being ignored in this model. Incorporating nonpoint sources would change the model fundamentally, because the trading ratio has to be defined in term of trading point sources permits with nonpoint permits. The trading ratio should reflect the relative expected marginal damage impacts from each source and the relative uncertainty created by each source (Horan 2005).

An extension to the above models that incorporates a zonal approach and nonpoint source was brought by Lankoskyet et al. (2008). They derived an optimal point-nonpoint effluent trading ratio that considered heterogeneity of the emissions and heterogeneity of the environmental impacts of those emissions. They showed that spatial heterogeneity can significantly affect the political attractiveness of effluent trading.

In the context of the point-nonpoint trade, the EPA recommendations are limited to offset programs rather than to cap-and-trade programs. Under an offset program, only the polluters facing regulations (i.e., point sources) have incentive to purchase offsets or reductions. Under a cap-and-trade system, a maximum emission for a particular pollutant is decided and distributed across polluters in the watershed as permits or polluting rights; however, in order to create incentives for the nonpoint source then they should face similar regulations as the point sources.

2.5. Trading Systems for Nonpoint Source Pollution

Several types of permit trading systems for nonpoint-source discharges have been proposed: an ambient permit system, a zonal permit system, and a pollution offset system.

Morgan et al. (2000) proposed a marketable permit trading scheme to manage the nitrate pollution of groundwater supplies for rural communities with intensive agricultural activity with the level of nitrates monitored at the level of drinking water. The authors used a soil and groundwater transportation model to predict the nitrate leaching rates from a particular crop area. According to their model, the contribution of each farm is weighted by a delivery coefficient determined exogenously. The permit trading system is defined as an ambient permit system, with the permits being denominated in terms of nitrate emissions measured at a receptor point (e.g., a well).

Their proposed ambient permit system integrates three models: a production model, a soil model, and a groundwater model. The production model defines the profits as a function of yield, with yields being defined as a function of fertilizer, agricultural practices, and crop rotations. Next, the abatement costs are defined as the difference in profits before any regulation is imposed and the profits after the regulations are adopted. The farm minimizes its abatement costs, where the costs are defined as the sum of the loss of profits and the expenditure on permits. The soil model estimates the water and the nitrogen emissions associated with each practice, and the groundwater model simulates the nitrate's fate to the water through groundwater.

In addition to estimating the delivery coefficients, the marginal abatement costs, and the initial permit allocation, the authors also underlined the importance of the baseline and the timing. The authors simulated the trading outcomes over the span of several years and assumed that the permit price is determined in a repeated auction. Farmers make the trading decision by comparing the equilibrium price with the marginal abatement cost. The abatement cost is higher for the farms whose emissions have the greatest impact on the water quality measured at a specific well. Their model is one of the first papers that showed how using different tools (i.e., a soil and a groundwater model) can be used to transform the nonpoint-source problem into point source, and how point source policies can be applied to the nonpoint.

Ermoliev et al. (2000) discussed the trading mechanisms in pollution permit markets. Their normative findings showed that, in the case of an ambient permit system with a single receptor, the market cost minimization solution is also the least-cost solution regardless of the transaction type: bilateral or sequential. In the case of multiple receptors, the convergence of the solutions is assured if the transactions are sequential and multilateral, and hence a source supplying permits needs at least two trading partners.

Lock and Kerr (2007), in a background paper, analyzed the decisions that need to be made for setting up a nutrient trading system, such as identifying a target, allocating the allowances, and setting up a monitoring system. According to them, the water quality goal needs to take into account both cost benefit analysis and political aspects. Next, a nutrient trading system should be made available only to the nonpoint sources. Furthermore, the trading cap needs to be expressed in units that can be easily allocated across the polluting sources. They also emphasize the importance of mapping the nutrient losses to the allowances needed.

In a follow-up paper, Kerr et al. (2007) introduced a permit trading system for nonpoint sources for a watershed that drains in a lake (a single receptor point). Zonal permits are created to account for the time it takes emissions leaving the field to reach the lake. Hence, zones are distinguished by years, rather than distance, and the permits depend on the year in which the emissions reach the lake. In their setting there is a market for each type of permit, where the permit type is given each year. The trading cap (the maximum acceptable emissions in a given year) associated with a market determines the total number of permits of a given type. Hence, permits across different markets can be traded at a ratio of one to one.

Prabodanie et al. (2009) proposed a nitrate-emissions-based pollution offset trading system applicable to a small-scale watershed. They used a leaching loss model to estimate the nitrate emissions from different land uses and the size of the permit required at field level, a transport model to estimate the a matrix of delivery coefficients, and a linear programing model that used the information based on the demand and supply of permits to determine the optimal trades. The size of a nitrate permit is equal to the estimated nitrate emissions from a land-use option at field level, where the land-use options are defined as a combination of factors such as type of crop or stock, timing, method and rate of fertilizer applications, and other land management practices. The authors used a nitrate transport model to estimate a matrix of delivery coefficients. They simulated the impact of field emissions at a given receptor at different points in time. A linear relationship is assumed between the emissions leaving the field and the impact at the receptor. Finally, they used a linear program to determine the price and permit allocations that maximize the total surplus in the permit market subject to water quality standards and initial permit allocations. The authors simulated the outcomes of their proposed trading system using a hypothetical groundwater watershed draining into a lake, with six farms and five land options. An environmental authority decides environmental standards for two receptors and allocates the nitrate permits among the farms to satisfy the standards. Within the trading system, every farm estimates the profits from each land-use option and submits five bids/offers. The prices depend on the farm size and are equal to shadow prices of the individual environmental constraints. Their findings showed that the pollution offset trading, while incurring small transactions costs, can achieve environmental constraints in a cost efficient way.

2.6. Solving for the Least-cost Allocation

To improve the water quality, best management practices or conservation practices that involve either the retirement of land from production, or practices that can be implemented parallel with the agricultural activity, have been viewed as potential ways to reduce the adverse effects of agriculture on water quality.[3] There are many studies that have modeled and

[3] Examples of conservation practices that can be used together with the agricultural activity are no till, cover crops, and reducing the fertilizer application rate.

researched the impact of different conservation practices on reducing the nitrate, phosphorus, and sediment loading both at field level and at the watershed scale as well (Vache et al. 2002; Maringanti et al. 2009; Panagopoulus et al. 2011; Inamdar et al. 2001). The effectiveness of a conservation practice in reducing the field emissions depends on a various number of factors: the field characteristics such as soil, slope, location, and mostly importantly the conservation practices on the adjacent fields and elsewhere in the watershed. This implies that the off field impacts on the total pollution cannot be accurately determined as a proportion of the emissions generated at field level (Rabotyagov et al. 2010).

There are a number of federal and state programs that provide either cost-sharing or full financial support for the implementation of conservation practices in agricultural areas. A primary concern in administering the limited public funds is finding the most cost effective way to allocate the conservation practices (Schleich and White 1997; Rabotyagov et al. 2010). This is equivalent to solving the least-cost placement of the conservation practices under a limited budget. Solving for the least-cost allocation is also relevant for the instance when a cap is set on the total level of pollution, and a regulator is interested in finding the least-cost method to achieve that cap or pollution target.

The water pollution in a watershed is a spatially complex process that involves many nonpoint sources. Additionally, there is an imperfect relation between adopting a conservation practice and its efficiency in reducing field emissions (Malik et al. 1994; Crutchfield et al. 1994). Many of these problems have been overcome by the development of the physically based, spatially distributed models that are able to simulate the impact of different conservation practices at field level and the field emissions’ fate and transport to the main watershed receptor where the ambient pollution level is measured.

Finding the optimal placement of conservation practices in a watershed is a discrete optimization problem with the search space defined by the possible combinations of the fields located in the watershed and the available conservation practices.

Schleich and White (1997) were among the first to show how linear programming models can be used to identify the least-cost solution to reach the predetermined targets for a watershed in Wisconsin. Their model included both point and agricultural sources, with the latter being aggregated in subwatersheds. By aggregating costs and phosphorus data for each source, the model identified what source should be the target. The model primarily selected the subwatersheds as areas to be policy targets for achieving the desired reductions.

Khanna et al. (2003), using an integrated framework that combines the spatial and biophysical attributes with a hydrological and economic model, developed an analytical framework to determine cost effective cropland enrollment in the Conservation Reserve Enhancement Program, a program designed for reducing off-site sediment. Their study highlights the fact that the amount of sediment transported from a field to a water body also depends on the land-use decisions on the upslope and downslope fields. Therefore, the contribution of each field to the total amount of sediment (the transport or delivery coefficient) needs to be determined jointly or endogenously with the land-use decisions of all the other fields. In order to cope with the complexity of the water pollution process, the authors focused on a narrow strip of land up the stream and only two alternatives for each field: crop production and land retirement. Additionally, they show that the payments per acre offered for the land enrollment should take into account the field’s location and specific characteristics.

2.7. Linking the Water Quality Programs to a Carbon Offset Market

The last chapter of my dissertation explores the conceptual links between a nonpoint-source program for water quality and a carbon offset market. Since many of the same land management actions that improve water quality also may store carbon in soils, it is natural to study the possible links between the two trading markets. In this section, I review the literature that examines the links developed between the two markets.

Carbon has been found in all living organisms, and can be found in many forms such as plant biomass, and soil organic matter. Carbon sequestration can be defined as the long-term storage of carbon that can be found in oceans, soils, and geologic formation, with soils containing more than 75% of the carbon pool. Carbon sequestrated in the soil, also known as soil carbon matter, is the result of the life cycle of a plant. During the process of photosynthesis, plants assimilate carbon—some of it is released into the atmosphere as carbon respiration, some of it remains as plant tissue. The latter one is added to the soil as the plant decays and decomposes. Many factors determine how long the carbon remains in soils, such as climatic conditions, natural vegetation, soil texture, etc.

The amount of carbon in the atmosphere that was once stored in the top soils has increased by more than 30% in the last 150 years. Furthermore, there is increasing evidence that the higher level of atmospheric carbon dioxide is contributing to the rising levels of global temperature. It has been established that soils have a great potential to store up to five times more carbon than currently, given that the current land management is changed (Lal et al. 1998). For example, conservation tillage can enhance the carbon sequestration in soils by minimizing or even eliminating the manipulation of the soil before a new crop production. Cover crops offer another example of conservation practice that can enhance the soil structure by adding organic matter to the soil. Given that many of the same land management practices that store carbon in soils also may improve water quality, it is natural to study the possible links between the two markets.

Although trading programs involving nonpoint sources and carbon sequestration offsets have many similar characteristics, the two literatures have developed separately and have used different language to describe the policies related to credit trading. Stephenson and Bosch (2003) summarized the lessons learned from nonpoint-source and carbon sequestration credit trading and identified some possible areas where the two programs can overlap and improve the policy design for trading programs for environmental protection. They found that areas such as measurement uncertainty, baselines, leakage and trading flexibility are treated differently in the two settings, but the lessons learned can be used to improve the cross design of these programs (nonpoint sources and carbon).

The conservation payments, such as the Conservation Security Program and the Environmental Incentive Quality Program, also known as green payments, have been considered as possible policy instruments to induce the adoption of conservation practices that can enhance the amount of carbon sequestrations.

A larger number of previous studies have investigated the environmental benefits associated with land retirement programs. Antle et al. (2001) compared the relative cost efficiency of two alternative policies. Under the first policy, farmers receive payments for land retirement. Under the second policy, farmers receive payments for changing their crop rotations. The authors linked an econometric cost model with carbon simulation model to obtain estimates for the marginal cost of sequestration that accounts for both spatial heterogeneity in land use and the rates of carbon sequestrations. Their empirical findings for the agricultural area in the

Northern Plains show that the first policy is a relatively inefficient way to increase carbon in soil, and the second policy has the potential to increase the carbon sequestration at a much lower cost.

Feng et al. (2006) investigated the carbon sequestration potential and the corresponding co-benefits associated with land retirement policies, where farmers are paid to take land out of production, and a working land program where farmers are paid to adopt certain conservation practices by considering an index of multiple environmental benefits such as soil erosion reduction, carbon sequestration, and nutrient discharge reductions. They found that conservation payment policies that maximize the land enrollment provide higher carbon benefits than the payments designed to maximize other carbon benefits. Another finding of the same research shows that a working land program is more cost effective for low targets of environmental improvement, while a land retirement program provides better benefits for higher target levels.

The concept of participating in multiple markets, where the participation is driven by the same abatement action, is known as credit stacking or double dipping. Woodward (2001) attempted to answer the question of whether or not it is socially optimal to allow for double dipping. He considered a multiple pollutant abatement cost technology where the pollutants are complements, meaning that the abatement actions that reduce one pollutant also reduce the other pollutant. Two cases were considered: a multiple or double dipping policy market and a single market policy. Under the multiple market policy, firms can sell credits generated by the same abatement action in multiple markets, while under a single market policy firms can participate only in a single market. To find which policy, the single or double market, yields higher benefits, the authors analyzed the degree of complementarities between the two pollutants, the degree of heterogeneity among the pollution abating firms, and the slopes of the marginal benefit curve. They also pointed out that the caps in the two markers have to take into account the possible complementarities. The authors found that a multiple market policy leads to the cost effective outcomes when the caps are set up correctly (i.e., the caps are set by taking into account the interactions between pollutants) when there is evidence for substantial complementarities, and when the marginal benefit curves a single market policy is preferable.

Reeling and Gramig (2012) investigated the possible cost and environmental implications of using the carbon offsets to fund the conservation practices targeted for improving the water quality. The authors used a novel approach that combined the outputs provided by a GHG model and a hydrological model with a genetic algorithm optimization to determine the optimal placement of different conservation practices. Their findings showed that the emissions trading markets that are proposed under the Kyoto protocol have the potential to improve the outcomes of water quality programs.

Yeo et al. (2012) recognized that many abatement practices adopted by farmers can reduce both the GHG emissions and the nutrient runoff, and that there is a potential cost savings from having two pollution permit trading schemes running simultaneously. The authors modeled the abatement costs, the potential level of total cost savings, and the environmental impacts under three scenarios farmers are allowed to participate in: the nitrogen trading market only, the GHG emission trading scheme, and two markets simultaneously. Their model was calibrated for a watershed in New Zealand, New Zealand being the first country worldwide to implement a trading scheme for trading GHG. Several findings emerge from their study: (*a*) the total level of GHG is lower when the two markets function simultaneously; (*b*) there is an inverse relationship between the permit price in the nutrient markets and the price of carbon offsets; and, (*c*) the amount of abated nutrient is does not change in the presence of the two markets, but decreases with the permit price for nitrogen.

2.8. Modeling and Optimization Techniques

In addition to building on previous literature in nonpoint emission trading, I use a number of models and tools to evaluate the empirical part of my dissertation. In this section, I provide a short description of these tools. Difficulties in establishing a direct link between the agriculture activity and ambient pollution level measured at receptors, where each source is made liable for its discharges, has been one of the main impediments in the development of a trading system that approaches agricultural nonpoint sources. Better understanding of the interactions between agriculture activity and water quality has been achieved with development of a hydrological model.

2.8.1. Soil and Water Assessment Tool (SWAT)

The Soil and Water Assessment Tool (SWAT) is a complex water quality watershed-based hydrological model developed by the US Department of Agriculture to simulate the impact of point and nonpoint-source emissions (Arnold et al. 1998; Arnold and Fohrer 2005; Gasmann et al. 2008). The model is designed to run watershed simulations based on a wide range of inputs such as climate data (precipitation, temperature, etc.), soils characteristics information (slope, soil quality, topography, erosion, etc.), plant growth and crop rotations, nutrient management, nutrient transport and transformation, and land use and management practices. Using the above data as input, SWAT outputs consist of in stream concentration estimates for nitrogen, phosphorous and sediment loadings. The output provides overall measures of the concentrations, as well as detailed information about the components of each type of discharge. For example, for nitrogen loadings, I can retrieve detailed information about components of nitrogen loadings.

The obtained information can be used to predict watershed loadings on a wide time range, from daily to annual estimates.

The SWAT model is used to estimate the changes in nutrient loadings as a response to alternative conservation practices under different crop choices and rotation alternatives. In order to run simulations, the watershed, a well-defined geographical entity, is divided into several subwatersheds or subbasins. In SWAT, each subwatershed is delineated further into small hydrological response units (HRU). An HRU is a conceptual entity, with no precise spatial location within the subwatershed, and identified as a percentage of the area in the subwatershed with homogenous soil, land use, and management practices. The primary water and loading simulations are made at the HRU level. The estimated loadings can be interpreted as edge-of-field runoff emissions. The nutrient loadings measured at the final outlet located at the base of the subwatershed are obtained by adding the loadings corresponding to each HRU that previously have been routed through a network of channels and reservoirs.

The development of hydrological models like SWAT, calibrated with watershed specific data, makes the simulation of impact on water quality of different watershed scenarios possible. A watershed is divided into hundreds of fields, and each field may have multiple agricultural practices that are suitable for its type of soils. For example, for a set of 9 agricultural practices and 2,900 fields, the total number of possible watershed scenarios is $9^{2,900}$ possible scenarios. Using SWAT, a water quality level can be estimated for each watershed configuration. With appropriate economic data the cost of agricultural activity associated with a particular scenario can be assessed. The question arises: which of those scenarios is most desirable from a cost and/or pollution reduction perspective? Unfortunately, the high dimensionality of the problem makes finding a solution through traditional optimization tools practically impossible.

2.8.2. Evolutionary Algorithms

One way to deal with the combinatorial nature of the watershed simulation-optimization model is the implementation of an evolutionary algorithm. Evolutionary or genetic algorithms are designed to mimic biological evolution, considered by Mitchel (1996) to be "in effect….a method of searching for solutions among an enormous set of possibilities." Genetic algorithms are heuristic global search algorithms that are able to find the nearly optimal solution by using principles like "natural selection" and "survival of the fittest." The first studies that use evolutionary algorithms for finding the "nearly optimal" solution for conservation practices best placement were published only in the early 2000s, even though the theoretical background for evolutionary computation started at the beginning of the 1950s.

The main terminology used in defining evolutionary algorithms s is similar to that used in biology, and consists of terms such as: population, genome, individual, allele set, offspring, recombination, mutation, etc. In a broad sense, a population is defined by the individuals that share the same defining elements or characteristics known as an allele set (allele set or genome). Within a population, each individual has a unique combination of genes from the allele set. The evolution process is an iterative, continuous, and dynamic process that allows the formation of new generations from an original population. The evolutionary process assumes that only the fittest individuals (the ones that have the best characteristics or genes) can generate offspring (known as crossover) by combining their genes. However, with a given probability, an offspring can suffer mutations. Following the crossover process a new generation or population is created. This process can span over an unlimited number of generations.

Genetic or evolutionary algorithms can successfully handle optimizations problems: (a) that have a large space whose characteristics are not well known or have complex properties such as nonconvexities and discontinuities; (b) are complex; and (c) when a solution near global is acceptable.

In the context of agricultural pollution, the goal is to find the watershed configuration that achieves a predefined level of water quality in the least-cost way. If the number of fields and conservation practices are small enough, given a set of costs, one could evaluate each possible combination of fields and conservation practice to find the corresponding costs and water quality levels and rank order the solutions. As mentioned earlier, in a watershed there are at least a few hundred fields, and considering only two conservation practices, the number of possible combinations increases exponentially. However, the least-cost allocation problem can be emulated as an evolutionary process where: (a) a watershed represents an individual, (b) a field in the watershed represents a gene, and (c) the set of agricultural practices represents the allele set (the properties a field can take). Hence, an individual (a watershed) is defined by a particular combination of fields and conservation practices, whereas a population is a set of watershed configurations that have the same set of conservation practices in common. Therefore, an individual represents a possible candidate solution to the pollution cost minimization problem, whereas a population (a set of watershed configurations) represents the set of all potential solutions to the same problem.

The goal of the evolution process is to find the watershed configuration that achieves a given level of ambient standard at lowest cost, or alternatively given a budget achieves the lowest level of ambient pollution. Moreover, since there are measurement errors in quantifying the effectiveness of different conservation practices in reducing emissions, an average realization of the water quality target is considered sufficient. This is another reason the watershed pollution problem is a suitable area for the use of the evolutionary algorithms.

When water quality optimization is considered with respect to a single pollutant, the optimization problem can be defined as a single objective optimization problem. In most cases, water quality impairment is not limited to a single pollutant; hence, the optimal solution requires solving a multi-objective optimization problem. In some cases, evolutionary algorithms can accommodate this by combining competing multi-objectives into a single known objective function—the solution yielding a single optimal solution. In the cases where a single objective function cannot be determined, the solution to a multi-objective optimization will consist of a set of solutions, or a Pareto frontier optimal set. This is a "near-optimal" set that reveals the tradeoffs between the different objectives. The near optimal characteristics come with the “temporal” aspect of the algorithm. More precisely, since there are no clear stopping criteria, the solutions can always improve if more generations are allowed to survive. The solutions are optimal given the number of generations that survived.

Earlier applications of evolutionary algorithms to watershed management focus on a single objective function, either pollution reduction effectiveness or cost (Srivastava et al. 2002), or on sequential optimization of effectiveness and cost (Gitau et al. 2004; Veith et al. 2003), where optimization is made in stages. Bekele and Nicklown (2005) used a multi-objective function, but the set of agricultural practices is limited to crop management practices. Maringanti et al. (2009) provided a recent application of an evolutionary algorithm for a watershed-scale optimization problem with two conflicting objectives simultaneously: a cost increase and pollution reduction. The set of conservation practices consisted of 54 different combinations. Three different nonpoint pollutants were considered: phosphorus, nitrogen, and sediment; but only one pollutant was considered at a time, hence three different optimization models were estimated. For each model, a Pareto frontier depicts the tradeoff between the two objectives. The allocations of conservation practice in the watershed according to the optimization results were shown to be superior to random allocation, resulting in reductions of 33% in sediment loading, 32% in nitrogen loadings, and 13% in phosphorus loadings.

In the empirical part of my dissertation, I use an evolutionary algorithm application to solve for the cost effectiveness—first for a single pollutant case (nitrogen or phosphorus), and second for multiple pollutant case (both nitrogen and phosphorus). The fact that one conservation practice often has the potential to reduce more than one pollutant provides a solid reason for this joint approach. Next, I provide a brief description of the particular multi-objective evolutionary algorithm (MOEA) that was used to obtain one set of the empirical results (Rabotyagov 2007).

The MOEA is a modification of the Strength Pareto Evolutionary Algorithm2 (SPEA2), proposed by Zitzler and Thiele (1999). The basic logic of the algorithms is as follows: (a) generate an initial population and a storing (temporary) population; (b) create offspring that, together with the parent population, are saved in the storing population; (c) define the objective by creating a metric function given the characteristics of the optimization problem; (d) for each individual compute the corresponding metric value; (e) rank individuals in order according to metric value, (f) decide a cutoff point, individuals whose metric values are below the cutoff point are disregarded, while the individuals that are above the cutoff point define the next generation (population); and finally, (g) repeat the above steps for a sufficient number of iterations.

Two types of populations are required at any given iteration of the evolutionary algorithm: the current generation and a temporary, or storing, population. At the starting iteration, the current generation is generated by randomly assigning individuals with different values from the allele set and the temporary population is empty. The random population plays the same role as starting values do in ordinary optimization routines. The temporary population is populated with offspring created by the mean of crossover and mutation using the individuals from the current population.

Next, a fitness value is calculated for each individual form the current and temporary population sets. In order to determine the fitness value, each individual is compared with all other individuals. For example, take two individuals A and B, individual A is said to dominate, in a Pareto sense, individual B, if all its genes have better values. In this case, individual B is said to be dominated. Strength value is a metric measure that defines the number of individuals that individual i dominates. An individual i can dominate some individuals but can be dominated by other individuals. Another metric measure, the raw fitness, is obtained by summing the strength values of individuals j that dominate. The raw measure is also a metric of the likelihood of individual i to generate offspring that pass his characteristics to next generations. If its raw measure is high, this means that it is dominated by many individuals and less likely to survive, hence it is desirable that the raw measure to be as close to zero as possible. A nondominated individual is an individual with a zero raw value.

The searching process can create individuals that do not spread uniformly over the search space. Some of them tend to cluster into certain areas, leaving some others areas sparse. The metric measures defined above do not provide any information about the degree of clustering. Another two metric measures are defined to incorporate information about clustering around certain areas of the search space. One of the measures differentiates among individuals that are too close one to another, and the second measure preserves diversity in the search space by rewarding the individuals that are further away on the frontier. The resulting individuals are spread more uniformly over the search space.

The selection is made based on a fitness score. The fitness score of each individual takes into account all the metric defined above. A lower fitness score implies the individual is closer to the Pareto frontier, and a zero value implies that the individual is on the frontier.

2.8.3. Environmental Policy Integrated Climate Model (EPIC)

SWAT is a model that operates on a large-scale level and is able to simulate the levels of nitrogen, phosphorus, and sediment losses under a wide range of scenarios of the agricultural activities that take place in a watershed. Yet, SWAT cannot capture the impact of agricultural activity on the total levels of GHGs or the carbon sequestration potential. In the case of water quality, the discharges of one field can be influenced, among other factors, by the distance to the outlet where water quality is measured, and agricultural activity of the neighboring fields. Carbon sequestration of a given field, however, is independent of location and what happens on surrounding fields. Thus, a field-scale model can be employed to measure the potential of carbon sequestration that is associated with an agricultural activity on a given field.

Environmental Policy Integrated Climate (EPIC) is a field-scale biophysical model of crop productivity originally developed to assess the effect of soil erosion on agricultural productivity (William et al. 1984). The field unit is assumed to be homogeneous with respect to soil, crop management, and topography. EPIC can simulate the effects of agricultural practices on crop yields, measure environmental indicators such as edge-of-field losses from fertilizer and pesticide applications, and soil-carbon sequestration potential.

CHAPTER 3. A MODEL OF WATER QUALITY POLLUTION

3.1. Introduction

In this chapter, I present a conceptual model to manage the ambient water quality in a watershed impaired by agricultural runoff (nitrogen and phosphorus). I examine a command-and-control (CAC) approach where the regulator has the ability to mandate specific abatement actions to each field in the watershed. The second approach is a performance standard (PS) where each farm has to meet predetermined farm-level performance requirements by choosing the relevant abatement actions. The last approach is a trading setting, where farmers, conditional on meeting their farm-level performance requirement, can trade credits or points assigned to the abatement actions with other participants in the watershed (Kling 2011). Additionally, I present a method of estimating the credits or points, where a point measures the ability of an abatement action to reduce the field-level emissions and the overall ambient pollution level.

My model captures several critical aspects of agricultural pollution such as: (*a*) imperfect information on the abatement costs of individual farms; (*b*) difficulties in measuring and monitoring the effectiveness of the abatement action at the field level; and (*c*) inherent nonlinearities in the transport and fate of emissions from the edge-of-field to the watershed outlet (the water quality production function).

Agricultural producers or farmers have a variety of abatement actions from which to choose for reducing farm-level emissions. Adopting an abatement action imposes both direct and implicit costs (e.g., lost yield, additional risk, etc.) that are likely to vary by farm characteristics: location, climate, and other farm-related characteristics, such as the farmer's knowledge and experience; thus, the abatement costs are heterogeneous across farmers. In this context, farmers are more likely to be better informed about their cost of adopting the abatement actions than a potential regulator. Given that the regulator has incomplete information on the costs, in general, it is not possible to identify ex ante the least-cost solutions that efficiently allocates the reductions (the abatement actions) across the sources (fields or farms). However, incentive-based instruments can improve the cost efficiency of this allocation by transferring the burden of cost minimization from the regulator to farmers. Moreover, the incentive based instruments can offer cost savings relative to command-and-control regulations, with the heterogeneity being a fundamental factor in determining the size of the potential cost savings (Newell and Stavins 1999).

Next, observing and monitoring the pollution impacts of farming activities on water quality is difficult to conduct and imposes significant costs, thus there is imperfect knowledge of the true relation between the abatement actions and the edge-of-field reductions. Focusing on the observable abatement actions or targeting observable inputs represents a possible solution to this problem as suggested by Griffin and Bromley (1982), and Shortle and Dunn (1986). A cost efficient outcome is expected if the targeted inputs are correlated with the field emissions (Shortle and Horan 2013), but this is generally an empirical question.

A third challenging issue of nonpoint-source pollution is the emissions movement (the ultimate fate and transport process) from a field to the point they reach the water bodies where the ambient pollution is observed. Earlier theoretical papers assumed that the fate and the transport process is linear and separable between emissions originating from different fields (Carpentier, Bosch, and Batie 1998; Ribaudo 1989). However, water quality scientists and hydrologists note that the impact of emissions from different fields on the overall level of water quality is non-constant and depends on the field’s location; hence, the process is likely to be nonlinear and nonseparable. Additionally, the emissions from one field interact with the emissions from the surrounding fields (Horan and Shortle 2013), making it even more difficult to observe the individual farm impacts (Braden 1989; Lintner and Weersink 1999, Khanna et al. 2003). In practice, researchers rely on the use of the various biophysical simulation models to capture the key features of the water pollution process. This process is referred to as the water quality production function. In the section introducing the conceptual model, I assume that this function is differentiable, although for the empirical results I employ a biophysical model to capture the key characteristics of the water quality process.

As mentioned earlier, the current regulatory framework is another difficult issue in addressing agricultural pollution, as the property rights to pollute are assigned to nonpoint sources. In spite of the missing regulations at the federal level, there are cases where states have opted to apply the 'polluter pays' principle and to reverse the property rights for agricultural polluters. It is worthwhile to mention the case of the Everglades Agricultural Area in Florida, where as part of the Everglades Forever Act (1996), the South Florida Agricultural Management District has established mandatory source controls to lower the phosphorus level in the Everglades Agricultural Area by implementing a best-management permitting program. The program includes performance metrics for each best management practice, on site verification, and monitoring to ensure that the conservation practices are implemented consistently, and recommends adjustments if the water quality goals are not achieved. Each landowner in the Everglades Agricultural Area needs to hold a permit that includes an approval for a best management practice for each crop or land use, and an approval for a monitoring discharge plan. Over the 17-year history of the program, a measurable reduction in the ambient pollution of more than 55% has occurred (Daroub et al. 2011).

In this dissertation, I model the water pollution as a cost effectiveness problem, where the ambient pollution target is given to the environmental authority and his goal is to achieve the target in a least cost way (Horan and Shortle 2013).[4] Moreover, cost effectiveness became the preferred conceptual approach as the issue of achieving increasing levels of abatement became more stringent (Newell and Stavins 1999).

This chapter is organized into several sections. In the first section, I introduce a conceptual model of pollution as it relates to agricultural pollution, outline the different policy approaches proposed for addressing water quality, and predict their theoretical outcomes under the different sets of assumptions. In the second section, I outline a multistep employed for obtaining the credits or points to be assigned to each abatement action. In the last section, I present the two watersheds together with various data inputs used as support for the empirical evaluation of my model—results that will be introduced in the next chapter. Within the same section, I also summarize and discuss the results from the estimation of the point values.

3.2. Conceptual Model

I consider a simple model of pollution where the water quality in a watershed is impaired by runoff from agricultural fields (for example, nitrogen or phosphorus). There are N farms in the watershed indexed by $i = 1, \ldots, N$. The farms are heterogeneous with respect to physical characteristics such as soil, slope, rainfall, etc. The ambient water quality level is monitored in-stream at the outlet of the watershed. Next, I consider a set of conservation practices or

[4] Shortle and Horan (2013) show that the two problems are equivalent only under special conditions.

abatement actions, $x = (x_1, x_2, \dots, x_J)$, which can be implemented at the farm level to reduce the edge-of-field runoff emissions, with *J* representing the number of abatement actions available at the watershed level.[5] Let r_i be the i^{th} farm's reduction in pollution measured at the edge-of-field (that is, farm-level pollution abatement). If no abatement action is taken, then $r_i = 0$. The relation between the vector of abatement actions and the farm-level abatement is represented by an abatement function, with the abatement levels denoted by:

$$r_i = r_i(x_i, \gamma_i, \xi) \quad \forall\, i = 1, \dots, N \tag{1}$$

where x_i represents the $J \times 1$ vector of abatement actions implemented by farm *i*, where the *jth* element x_{ij} takes a value of 1 or 0, γ_i represents the farm's physical characteristics such as soil type and topography, and ξ represents the random factors that are influenced by weather or by the pollutant fate and transport through the watershed.[6] The abatement actions are the farmers' input choices that can be used at farm level to reduce the field level runoff (Horan, Shortle and Abler, 2002). The distribution of ξ is assumed to be known and it could be given by the historical distribution of the stochastic weather.

The baseline edge-of-field emissions are the result of the farmers' profit maximization behavior given that no abatement actions are implemented. Farmers are assumed to be rational, perfectly informed, and risk neutral optimizers and price takers in both output and input markets.

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[5] Conceptually, conservation practices and abatement actions can be used interchangeable without any loss of meaning. However, an abatement action can be defined as a combination of two or more conservation practices that can be implemented simultaneously.

[6] As equation (1) indicates, I recognize the role and impact of the weather stochastic elements. However, addressing the stochastic elements is not a focus of my current work. Therefore, for the remainder of my dissertation, I abstract away from the stochastic elements by considering the mean of the edge-of-field abatement value and suppress the ξ from the notation. However, I reserve one section of the empirical part to assess the robustness of my results under the historical weather distribution.

Additionally, the abatement actions are mutually exclusive; only one abatement action can be chosen at a time, but some abatement actions can be defined as a combination of different conservation practices. Abatement costs are defined as the difference between baseline profits when no abatement action is taken and the profits associated with the adoption of an abatement action (Freeman, 1993). The abatement costs are farm and abatement action specific. The costs are defined on a per acre basis; hence I am assuming constant economies of size.[7] Let the abatement cost function be defined as a function of the vector abatement actions :[8,9]

$$C_i(x_i, \gamma_i, \theta_i) \quad \forall\, i = 1, \ldots, N \tag{2}$$

I assume that the costs of adoption vary across locations due to both differences in physical characteristics (soils, slope, etc. γ_i.) and management abilities or farming experience, where the management abilities are reflected by θ_i, a scalar index of farmer's profitability (the farmer's type). Hence, equation (2) defines the abatement costs for a farmer type θ_i if he or she uses the abatement actions vector x_i. The regulator does not know the farmer's type, however, I assume that the regulator knows that each θ_i follows a certain distribution.[10] By allowing farmers to have different types that are not known to the regulator, I assume that certain information such as the abatement costs is known only by farmers and not by the regulator. Given that the

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[7] Economies of size are used to describe a situation where as a farm expands its output, the cost per unit of output decreases. By analogy, under constant economies of size, the farm abatement costs increases by a factor equal to the number of its acres.

[8] Conventionally, costs are modeled as an increasing function of the abatement level, however in the case of nonpoint source, the abatement cost functions are defined as a function of the input chosen to reduce the emissions (Horan, Shortle and Abler, 2002). In this case, the abatement actions represent the input choices.

[9] Cost function could also include the weather stochastic factor, ξ

[10] The distribution of the farmers' type is not essential for the presented model. Standard assumptions include a uniform distribution with support on unit interval, or on $[\theta_L \ \ \theta_U]$ (Smith and Tomasi, 1999).

regulator knows the farmers' type distribution, he can identify a cost function for the average type farmer, $\bar{\theta}$, $C_i(x_i, \gamma_i, \bar{\theta}),\ \forall\, i = 1, \ldots, N$. In terms of abatement costs, this means that the regulator can infer an average cost estimate for each abatement action.

The total ambient pollution is given by an expected water-quality production function $W(\mathbf{r})$, represented as a function of the vector of each farm's individual edge-of-field emission reductions r_i, $\mathbf{r} = (r_1, r_2, \ldots r_N)$, and the expectation is taken with respect to the distribution of ξ. Recall that other factors such as the location in the watershed, the agricultural activities on the surrounding field, and hydrology elements enter into the ambient production function in addition to the edge-of-field emissions. However, addressing the stochastic elements is not the focus of my dissertation, hence for the remainder of my dissertation I suppress the stochastic elements for notational simplicity.[11]

The exact water-quality production function is unlikely to be known given the complexity of the biochemical and hydrological process that takes place in a watershed; but there is a range of watershed-based water quality models that approximate these hydrological and biophysical processes, such as the SWAT model.

Let, $W(\mathbf{r}) = W^0 - A(\mathbf{r})$ be the ambient water quality at the watershed outlet, where W^0 is the level of water quality given the current activity, and $A(\mathbf{r})$ is the expected ambient pollution reduction associated with $\mathbf{r}$, the vector of field individual emission reductions, or more simply the abatement function.[12] The expected ambient water quality level can be expressed as the

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[11] In the empirical work described later in the dissertation, I use the five-year average of the edge-of-field reduced emissions.

[12] The literature uses the terminology of water-quality production function and abatement function interchangeably. For the reminder of my work, I will refer to ambient function as representing the change in the ambient water quality at the watershed outlet.

difference between the no-control (baseline) expected ambient water quality level, W^0, and the in-stream expected abatement associated with the edge-of-field emission reductions given that an array of abatement actions is implemented, $A(\mathbf{r})$.

I consider an environmental authority or a regulator who seeks to achieve a particular expected abatement pollution level, denoted as $\bar{A}$, by finding a least cost allocation of the available abatement actions to the fields in the watershed. First, I identify the first-best solution to this problem assuming that the regulator and farmers both have complete cost information (hence, the regulator knows the farmers' types). This "perfect cost information" solution is contrasted to the solution where the regulator does not know the true abatement costs. Instead, only the average costs of each abatement action is known (hence, the regulator does not know the farmers' types), and used to solve for the least-cost solution, by imposing the same per acre cost for each farmer.

3.2.1. First-best

I begin by assuming that the regulator knows: (*a*) the field level abatement costs, (*b*) the relation between abatement actions and reduced emissions$r_i(x_i)$, and (*c*) the true form of the ambient abatement action $A(\mathbf{r})$.[13] The cost minimization problem faced by a regulator seeking to minimize the overall abatement costs to meet the expected ambient reductions by choosing field-level abatement actions is:

$$\min_{x_i} \sum_i^N C_i(x_i, \gamma_i, \theta_i) \quad s.t. A(\boldsymbol{r}(x)) \geq \overline{A} \tag{3}$$

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[13] For simplicity of notation, I drop out γ_i from the notation $r_i(x_i, \gamma_i)$

where θ_i shows that true field level abatement costs are used in solving the cost minimization problem. Next, consider the discrete change in the total abatement,$\partial A(r(x))/\partial x_{ij}$ given that abatement action j is adopted by the field i defined as:

$$\frac{\partial A(r(x))}{\partial x_{ij}} = \frac{\partial A(r_i(x_i), r_{-i}(x))}{\partial r_i} \Delta_{x_{ij}}(r_i(x_i)) \tag{4}$$

where, Δ is used to show the discrete nature of the set of abatement actions, and $r_{-i}(x)$ accounts for the fact that the abatement actions on other farms affect the impact of farm i (Braden et. al. 1989;Lintner and Weersink 1999; Khanna et al. 2003). The first term of the right-hand side of equation (2) , $\frac{\partial A(r_i(x_i), r_{-i}(x))}{\partial r_i}$, captures the nonseparability and is associated with endogenous transfer or delivery coefficients (Khanna et al., 2003). The presence of nonseparability is what makes a trading program difficult to implement. Next, nonlinearity refers to the fact that impact depends on the abatement action, $\Delta_{x_{ij}}(r_i(x_i))$ (i.e. not being constant in x_{ij}) (Shortle and Horan 2013).

The solution vector $\mathbf{x}^*$ to the problem defined by equation (3) identifies for each field x_i^*, the least-cost abatement action assignment and thus implies an optimal amount of edge of field pollution $r_i^*(x_i^*)$, $\forall\ i = 1, \dots, N$ farms and $\forall\ k = 1, \dots, J$ available abatement actions. The total cost is given by $TC^* = \sum_i^N C_i(x_i^*, \gamma_i, \theta_i)$. An "*" is used to indicate that this is the least-cost solution.

The first-best solution is achieved when the regulator has the ability to solve the problem defined by equation (3) in the presence of complete cost information. Additionally, I assume he has the ability to implement a command and control policy where he can mandate the abatement action x_i^* ; however, it is unlikely for the regulator to have complete cost information.

The cost asymmetry can be overcome by pursuing incentive based policies that shift the burden of optimization from the regulator to private farmers such as PS and trading. The implementation of any of the incentive based policies requires a functional form for the abatement function $A(\mathbf{r})$ and for the relation between field level abatement actions (x_i) and the expected edge-of-field abated emissions $(r_i(x_i))$.

Next, I consider how these different policies perform relative to the first-best by considering two different assumptions for the abatement function. First, I assume that the abatement function is defined as an exact combination of edge-of-field reduced emissions and a set of fixed and exogenously determined delivery coefficients. Next, I assume the abatement function is non-linear and non-separable in the individual field-level reduced emissions, but policies are implemented using an approximation that is a linear combination of individual edge-of-field reduced, and delivery coefficients where the delivery coefficients are fixed.

3.2.2. A linear and separable water quality production function ($\mathbf{A(r) = \sum_i d_i r_i(x_i)}$)

First best, complete cost information, CAC and incentive-based policies

Suppose that a regulator seeks to achieve a given level of total ambient emissions reductions, $\bar{A}$, and ambient function is exactly a linear combination of the individual edge-of-field reductions and a set of delivery coefficients. The delivery coefficients determine how much of the edge-of-field reductions contribute to the total abatement level. Moreover, it is assumed that the delivery coefficients are exogenously determined. According to earlier studies on air and water pollution, the abatement function can be expressed as an exact combination of delivery coefficients and site specific emissions (Montgomery, 1972). Assuming perfect cost information (the regulator and the farmers have the same cost information), this solution can be replicated with the same outcomes in any number of ways: a command-and-control, a performance standard, and a permit trading setting.

Under command-and-control, each farm is mandated to adopt, x_i^*. Alternatively, the environmental agency could require that each farm meets an individualized performance standard $(r_i^* = r_i^*(x_i^*))$. In this case, the farmer can choose the abatement action that minimizes the abatement cost at field level:

$$\min_{x_i} C_i(x_i, \gamma_i, \theta_i) \; s.t \;\; r_i(x_i) \geq r_i^* \tag{5}$$

Another alternative is to rely on private optimizing behavior and to allow trading among farmers such that a total ambient emissions cap is met. As Montgomery (1972) demonstrated, an "ambient based permit system" where each firm is faced with an ambient cap such that the total ambient emissions reduction target is met can achieve the least-cost allocation.

In short, under perfect information on costs and farm-level emissions, and a linear and separable water quality production function, the three above mentioned regulatory approaches can be employed to achieve the least-cost solution. Another alternative is to rely on private optimizing behavior and to allow trading among farmers such that a total ambient emissions cap is met.

First-best, cost asymmetries, CAC and incentive-based policies

In reality, it is likely that while the farmer knows the true cost of their abatement actions, the environmental authority does not. Thus, the environmental authority is unable to identify the abatement actions cost efficient allocation. However, the regulator is likely to have some limited information on the distribution of costs, such as the mean of the abatement costs. I assume that the regulator knows the vector of average costs for each abatement action. In this case, the regulator solves the following problem:[14]

$$\min_{x_i} \sum_i^N C_i\left(x_i, \gamma_i, \bar{\theta}\right) s.t.\ \sum_i^N d_i r_i(x_i) \geq \bar{A}, \tag{6}$$

where, $\bar{\theta}$ denotes that the regulator uses his best estimates of costs (i.e. average estimates of the costs) and the total cap is set at the $\sum_i^N d_i r_i(x_i) = \bar{A}$. The solution to this problem, denoted by "$\hat{\mathbf{x}}$", will generally differ from that obtained in solving equation (4), and the assignment of abatement practices, $\hat{x}_i$, will not necessarily coincide with the least-cost solution, x_i^*. Likewise, the edge-of-field emissions reductions, $\hat{r}_i = \hat{r}_i(\hat{x}_i)$, will be different from the first-best, $r_i^* = r_i^*(x^*)$. The total estimated cost of the regulator is given by: $\widehat{TC} = \sum_i^N C_i(\hat{x}_i, \gamma_i, \bar{\theta})$.

Under a command-and-control policy approach, the solution imposed by the authority,$\hat{\mathbf{x}}$", may not reflect the least cost allocation of abatement actions since individual farmers may have much lower or higher costs than the average cost estimates, which, if known by the regulator, could be used to more cost-effectively assign practices to fields. Nonetheless, the overall abatement target, $\bar{A}$, is met. The total cost of a command and control can be lower or higher than the regulator estimated costs:

$$\hat{T}^{CAC} = \sum_i^N C_i(\hat{x}_i, \gamma_i, \theta_i) \quad <> \quad \widehat{TC} = \sum_i^N C_i(\hat{x}_i, \gamma_i, \bar{\theta}) \tag{7}$$

In this case, the authority can potentially increase social welfare relative to a command-and-control assignment of conservation actions, $\hat{x}_i$ by allowing firms to meet a performance standard, $\hat{r}_i(\hat{x}_i)$, set at a similar level as in CAC. Since farmers know their true costs, they may

[14] In the case of abatement costs being nonlinear in θ_i, a smart regulator would want to minimize the expected costs by taking into account the distribution of the farmers' type. However, given that I consider that θ_i enters in a linear way, minimizing the sum of total costs evaluated at the average farm leads to the same interpretation.

be able to meet the standard allocated to them with less total costs by choosing a different abatement action. In the presence of a performance standard program, farmers face the following optimization problem:

$$\min_{x_i} C_i(x_i, \gamma_i, \theta_i) \; s.t \;\; r_i(x_i) \geq \hat{r}_i \tag{8}$$

Farmers minimize the abatement costs given the field level abatement costs, subject to a performance standard based on average estimates of true costs. The solution is given by $x_i^{PS}, r_i^{PS}(x_i^{PS})$ and the corresponding costs, $TC^{PS} = \sum_i^N C_i\left(x_i^{PS}, \gamma_i, \theta_i\right)$. Again, a clear comparison with: $\widehat{TC} = \sum_i^N C_i(\hat{x}_i, \gamma_i, \bar{\theta})$, the regulator estimated costs, cannot be made, however, there are cost savings relative to $\hat{T}^{CAC} = \sum_i^N C_i(\hat{x}_i, \gamma_i, \theta_i)$, the total costs under a command-and-control.

Additional cost savings are potentially achievable if the environmental authority makes the performance standard tradable. The farmer minimizes the abatement costs by choosing an abatement practice and the number of permits to trade, such that the total reductions measured at the edge-of-field level are less than the amount allowed by the number of permits held after trading. Let $\bar{l}_i^0$ be the i^{th} farm's abatement permit requirement. Then, a farmer solves:

$$\min_{x_j, l_i} C_i(x_i, \gamma_i, \theta_i) + pl_i \;\;\; s.t \;\;\; d_i \, r_i(x_i) + l_i \geq \bar{l}_i^0 \tag{9}$$

and the permit price is determined in a market equilibrium where $\sum_i^N l_i = 0$. Indeed, when the performance standard is fully tradable, the least-cost solution would be achievable, as this would be equivalent to implementing Montgomery's (1972) ambient-based permit system. Since by construction, $\sum_i^N \overline{l_i^0} = \bar{A}$, unfettered trading between firms who each know their own true costs will achieve the least-cost solution and the ambient environmental goal is satisfied.

When the regulator has limited information on the abatement costs and the water quality production function is characterized linear and separable, his optimal solution, $\hat{\mathbf{x}} = (\hat{x}_1, \hat{x}_2, \dots, \hat{x}_N)$, does not coincide with the solution under performance standard or permit trading. However, the water quality goal will be achieved under any of the regulatory approaches. Total costs across the three policy approaches will be lowest under a trading setting. The total costs of the regulator's solution evaluated at the true abatement costs can be higher or lower than the total costs of the three approaches. The magnitude of the divergences is an empirical question.

First-best, no cost information, CAC and incentive based policies

In this case the regulator has no cost information but he has the ability to identify the combinations of abatement actions that achieves the water quality goal. Let the vector of abatement actions by identified as a satisficing solution x^{Sat}; with $A\left(\mathbf{r}(x^{Sat})\right) \geq \bar{A}$. Obviously the cost of implementing this solution via a command-and-control is likely to be very high. However, under a linear and separable production function with fixed and exogenously determined delivery coefficients, trading has the ability to achieve the first-best solution, with both the water quality target met and the abatement costs minimized.

3.2.3. A nonlinear and non separable water quality production function

A linear and separable form for the water quality production (abatement) function, while offering very attractive characteristics for the incentive based policies does not give an accurate description of the reality, where a more complex pollution fate and transport function is required to describe the how the on-field abatement actions reduce the overall ambient pollution level. In this case, the mapping between the edge-of-field reduced emissions and the ambient water quality is not linear.

Assuming differentiability of $A(\mathbf{r}(x))$ with respect to $\mathbf{r}$ and taking a second order Taylor approximation around the an initial vector of pollution reductions (baseline, $\mathbf{r^o}$)

$$\mathrm{A}(\mathbf{r}) \cong \mathrm{A}(\mathbf{r^o}) + \nabla(\mathbf{r^0})(\mathbf{r}-\mathbf{r^o}) + (\mathbf{r}-\mathbf{r^o})\nabla^2(\mathbf{r^o})(\mathrm{r}-\mathbf{r^o})$$

$$= \mathbf{0} + \boldsymbol{d}(\boldsymbol{r^0})(\boldsymbol{r}-\boldsymbol{r^o}) + (\boldsymbol{r}-\boldsymbol{r^o})\boldsymbol{\nabla^2}(\boldsymbol{r^o})(\boldsymbol{r}-\boldsymbol{r^o}) \quad (10)$$

Given equation (10), delivery coefficients can be approximated by $\boldsymbol{\nabla}(\mathbf{r^0})$, the vector of marginal impacts of edge-of-field abatement on the ambient quality. However, as shown by equation (4), the delivery coefficients vector is determined endogenously, being a function of the abatement actions on the other farms. Moreover, in order to determine the delivery coefficients a set of initial abatement actions is needed. The choice of this initial vector will affect the quality of the ex post trading outcomes, because the approximation can be accurate in the vicinity of the initial abatement action but be poor for the post-trading vector of abatement actions. Next, given the curvature of the abatement function around the initial vector, $(\mathbf{r}-\mathbf{r^o})\boldsymbol{\nabla^2}(\mathbf{r^o})(\mathbf{r}-\mathbf{r^o})$, the linear approximation may, on average, overstate (understate) the abatement if the ambient function is convex (concave) in abatement. Given the above approximation, the abatement function can be written as:

$$A(\mathbf{r}) \cong \boldsymbol{\nabla}(\mathbf{r^0})(\mathbf{r}) = \mathbf{d}(\mathbf{r^0})(\mathbf{r}) = \sum_i^N d_{Ai}\, r_i(x_i) \quad (11)$$

where d_{Ai} are fixed delivery coefficients obtained from equation (10). The abatement outcome of a trading program based on trading ratios defined by d_{Ai}, and depending on the curvature of the abatement function around the abatement cap might be lower or higher than the initial target. In this case, the above linearization could be empirically adjusted by relaxing the abatement cap in the case of over attainment (convex curvature) or tightening the cap under attainment (concave curvature). Nevertheless, the magnitude and the direction of the corrections are determined empirically.

In the situation where a linear approximation to the water-quality production function is used, the farmer's problem in a market-based system can be written as:

$$\min_{x_j, l_i \in X} C_i(x_i, \gamma_i, \theta_i) + pl_i \quad s.t. \quad d_{Ai} r_i(x_i) + l_i \geq \bar{l}_i^0 \tag{12}$$

and the market clearing condition is $\sum_i^N l_i = 0$. The only difference from the linear cases described in equation (9) is that, instead of d_i, d_{Ai} being used, the "A" indicates that this is a set of derived delivery coefficients obtained from some form of linearization of the nonlinear water-quality production function.

Another important issue is the selections $\bar{l}_i^0$ (i.e., the vector of on-farm ambient reduction requirements). In the previous case, under a linear and separable ambient function with known delivery functions, any combination of delivery coefficients (d_i) and on-farm requirements ($\bar{r}_i$) that satisfies $\sum_i^N \bar{l}_i^0 = \sum_i^N d_i \bar{r}_i = \bar{A}$ also achieves the water quality target. This means that a cap can be defined by choosing the right number of permits to be distributed when designing a cap-and-trade program. Furthermore, a decrease in the abatement of one farm, when weighted by the appropriate delivery coefficient ratio, does not necessarily result in an increase in the abatement level on other farms.

Once the $d_{Ai}'s$ are determined, a trading program could use the true monitored edge-of-field reductions. However, monitoring imposes costs, $r_i(x_i)$, and cannot be measured exactly, hence a simplification of $r_i(x_i)$ based on observable abatement actions has the potential to make the program easier to implement. Next, I consider that the regulator assigns weights for each abatement action to approximate the effectiveness of these abatement actions in reducing edge-of-field emissions, $r_i(x_i) = \sum_j^J w_{ij} x_{ij}$.

Using a linear approximation of the abatement function, more flexible systems like the performance standard or trading program may outperform CAC in terms of abatement costs. However, they may also lead to non-attainment of the abatement target. The magnitude of the inefficiency or the extent of non-attainment is an empirical question and likely to be directly related to the accuracy of the approximation.

However, the ability of a standard performance or trading program to achieve cost savings by placing the burden of the optimization on farmers makes them appealing for consideration and evaluation by imposing fixed and constant delivery coefficients for ambient impact of edge-of-field abatement, and an additional linearization of the edge-of-field reduced emissions function:

$$A(\boldsymbol{r}(x)) \cong \sum_i^N d_{Ai} r_i(x_i) \cong \sum_i^N \sum_{j=1}^J d_{Ai} w_{ij} x_{ij}. \quad (13)$$

In the next section, I describe an approach to linearize the abatement function and estimate the delivery coefficients. In addition to using a linear approximation of the water-quality production function, the true $r_i(x_i)$ functions are approximated as a linear combination of weights that measure the impact of the abatement actions on reducing the emissions. The combination of edge-of-field points and delivery coefficients results in a system of point coefficients, where a point can be interpreted as the impact of an abatement action on total abatement level when adopted by a particular field. The efficiency of a performance standard and a trading program in the context of the point coefficients is empirically assessed in Chapter 4.

3.3. Generating a Linear Approximation to the Abatement Function

A trading program for pollution involves the existence of a tradable commodity that is able to measure the emissions or the discharges (Stephenson, Norris and Shabman 1998). In the context of water quality trading, it has been argued that the characteristics of nonpoint source represent barriers to the quantification of the emissions (Malik et al. 1994). Therefore the development of a tradable commodity by estimating a system of points that captures the abatement actions' efficiency in reducing ambient pollution offers a possible solution to this problem. In the context of watershed pollution, different abatement actions have different impacts on edge-of-field abated emissions, and identical reductions in the edge-of-field emissions might have different impact on the ambient pollution level.

A well designed system of points needs to account for all these characteristics. In this context, Kling (2011) proposed a point based trading system where agricultural producers would be required to implement abatement actions that accrue enough points per acre to meet a predetermined standard. The point values assigned to each abatement practice approximate:
(a) how effective an abatement practice is in reducing the edge-of-field emissions and (b) the impact of the edge-of-field reduced emissions on the ambient water quality. Since the abatement function ($A\ (\mathbf{r}(x))$ is approximated as a linear combination of the abatement actions impact measured at edge-of-field level and delivery coefficients, and the field level reduced emissions depend on the abatement action, without any loss, the abatement function can be written as a function of the vector of abatement actions x:

$$A(\boldsymbol{r}(x)) = A\ (x) \cong \sum_i^N d_{Ai} r_i(x_i) \tag{14}$$

Next, assuming that there are nonlinearities at the field level, the edge-of-field reductions are approximated as $r_i(x) \cong \sum_j^J w_{ij} x_{ij}$, where w_{ij} measure the impact of abatement action j given field i. The impact of field $i\,'s$ edge-of field reductions on ambient water quality is $d_{Ai} r_i(x_i) \cong \sum_j^J d_{Ai} w_{ij} x_{ij} = \sum_j^J a_{ij} x_{ij}$, where $a_{ij} = d_{Ai} w_{ij}$, referred hereon as "point coefficient", gives the number of points assigned to the abatement action j given field *i*. Since the point values are defined in terms of abatement, they can be interpreted as the marginal contribution to the total abatement of a particular field i given that the j^{th} abatement action is taken. Finally, the linear approximation of the abatement function can be re-written as:

$$A(x) \cong \sum_i^N d_{Ai} r_i(x_i) = \sum_i^N \sum_j^J a_{ij} x_{ij} = \boldsymbol{a} * \boldsymbol{X} \qquad (15)$$

where $\boldsymbol{a}$ is a JXN column vector of point values or coefficients to be estimated, and $\boldsymbol{X}$ is a JXN row vector of abatement actions.

The above linear approximation of the abatement function is made around the baseline emissions. Alternatively, the linear approximation can be made around the optimal solution (i.e. the optimal vector of abatement actions that achieves the desired abatement goal). In this case, the point coefficients can be interpreted as abatement impact relative to the optimal solution.

3.3.1. An empirical approach for estimating the vector of point coefficients a

To estimate the point coefficients for each abatement action and each field, I employ a multistep procedure using the special features of a watershed-based hydrological model, SWAT. In SWAT, a watershed is delineated into subbasins and further on into smaller fields units called hydrological response units (HRU). As a result, a watershed can contain thousands of fields. My method to estimate the point coefficients is to generate M sets of random allocations of abatement actions to the fields in the watershed, where each random allocation represents a unique watershed configuration. The impacts on the ambient level of water quality, in terms of mean annual abatement loadings of nitrogen or phosphorus, are obtained by running the SWAT model for each configuration.[15] The water quality outcomes measured in abatement levels ($A(X)$) are then combined with the vectors of abatement actions' assignments (X) to estimate the vector of point coefficients**, a**, by combining the results of a series ordinary least square estimations $\min_{a}(A - Xa)'(A - Xa)$.[16]

Often cases the number of fields (HRU) in a watershed is large, it is challenging to generate a sufficient number of watershed configurations to estimate NxJ point coefficients.[17] My approach to estimating point values takes advantage of the outputs generated by SWAT to break the above estimation into several steps. (a) estimate the point values at the subbasin level using the ambient levels measure at the watershed exit, (b) estimate point values at the field level using the field provided outputs, and (b) combine the results to obtain field specific point coefficients for each abatement action. Combining the two sets of results allows me to estimate the field specific point coefficients for each abatement action but also to estimate the delivery coefficients.

SWAT computes the ambient emissions at each subbasin outlet as a function of the component fields' emissions. Next, the ambient emissions from each subbasin are routed into a nonlinear and nonseparable way to determine the ambient water quality at the main outlet. Hence, SWAT provides emissions outputs at the field, subbasin, and main watershed level.

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[15] The abatement levels are obtained by subtracting the impacts on the ambient levels from the baseline emissions.
[16] A is a $MX1$ column vector, X is a $MX(JXN)$ matrix a is a JXN column vector
[17] In this case M should be greater than NxJ, where N is the number of fields and J is the number of abatement actions available at watershed level.

Three different set of point coefficients are estimated using a multistep procedure described in Appendix A. The sets of point coefficients differ in the degree of approximation. More specifically, the first set of point coefficients are field specific (i.e., for each field, I estimate a Jx1 point coefficients), the second set of point coefficients are subbasin specific (i.e. a given abatement actions has the same number of points for any field in a subbasin, and finally, the last set of point coefficients is watershed specific.

The obtained point values implicitly contain information on the trading ratios across different locations within the watershed as well as the trading ratios between different abatement actions, hence any trading based on the point coefficients will be made on a one-to-one base.

Once the environmental agency determines the point values that are credited to a particular abatement action in a specific field, he is able to compute the total point values associated with any water quality target. While the command and control policy is not affected by the total number of points, in the case of a performance standard and of a tradable credit program, the total point value chosen by the regulator will directly affect the total abatement level achieved at the watershed level.

For the performance standard policy, the regulator needs to choose the appropriate farm-level point requirements. Under the trading approach, credits or the point coefficients generated by abatement actions are tradable, on a one-to-one basis, across the watershed. As a result, a farmer solves:

$$\min_{x_{ij},b_i} C_i(x_i,\gamma_i,\theta_i) + pb_i \qquad s.t.\ \sum_{j=1}^{J} a_{ij}x_{ij} + b_i \geq\ b_i^o \tag{16}$$

where x_{ij} is a binary variable that takes a value of 1 if the abatement action j is chosen, the abatement actions, b_i is the number of points to be traded, a_{ij} the point values assigned to abatement action j given field i, and b_i^o is the field level constraint assigned by the regulator to field *i*. The point price p is determined in a points market equilibrium by $\sum_i^N b_i = 0$

This trading approach can be conceptually viewed as a combination of an emissions permit and ambient permit system (Rabotyagov et al. 2012). Under an emissions permit system rights are defined in term of what firms emit. Under an ambient permit system, right are defined in terms of pollution contribution to a receptor (Montgomery 1972; Baumol and Oates 1988). In this case point credits are specified at farm (field) level allowing the trade to occur on a one-to-one basis. Next, a point value approximates the impact of an abatement action on the total level of abated pollution measured at a single pollution receptor (watershed outlet). Trading ratios that account both for location and the abatement actions tradeoffs are embedded into the point coefficients.

The point-credit approximation procedure can also be adapted (a) for a single pollutant market; (b) for multiple pollutant markets where a separate system of points is estimated for each pollutant, and (c) to extend the single pollutant market by including the participation in a carbon market.

3.4. Empirical Framework

In the next section, I describe the two agricultural watersheds used as support for my empirical estimations, the set of abatement actions with the corresponding estimates for the abatement costs, and provide a description of the estimates obtained for the point values assigned to each abatement action. The point values estimates are watershed specific, and within each watershed field pollutant-specific, with two pollutants being considered: nitrogen and phosphorus.

3.4.1. Watershed description

I use the available data for two typical Midwestern watersheds, both located in Iowa: the Boone River Watershed (BRW) and the Raccoon River Watershed (RRW). The National River and Streams Assessment 2008–2009 includes Iowa in the Temperate Plains Ecoregion.[18] The survey finds high levels of nitrogen in 58% of the rivers, and medium levels of nitrogen in 13% of the rivers. At the same time, 31% (24%) of the rivers have high (medium) levels of phosphorus.

The Boone River Watershed

The BRW is located in the north central part of Iowa. The watershed covers more than 537,000 acres (2,370km^2) in six counties (Hamilton, Hancock, Humboldt, Kossuth, Wright, and Webster) as shown in Figure 3-1. The watershed area is crop intensive, with the surface being intensively tile drained; consequently, the wetlands area has been reduced significantly. Moreover, the Boone River Watershed agricultural area has been found to be responsible for some of the highest nitrogen loadings among Iowa's watersheds (Libra et. al 2004).

[18] Other states included in the same ecoregion are eastern North and South Dakota, western Minnesota, portions of Missouri, Kansas, Nebraska, western Ohio, central Indiana, Illionois, and southeastern Wisconsin.

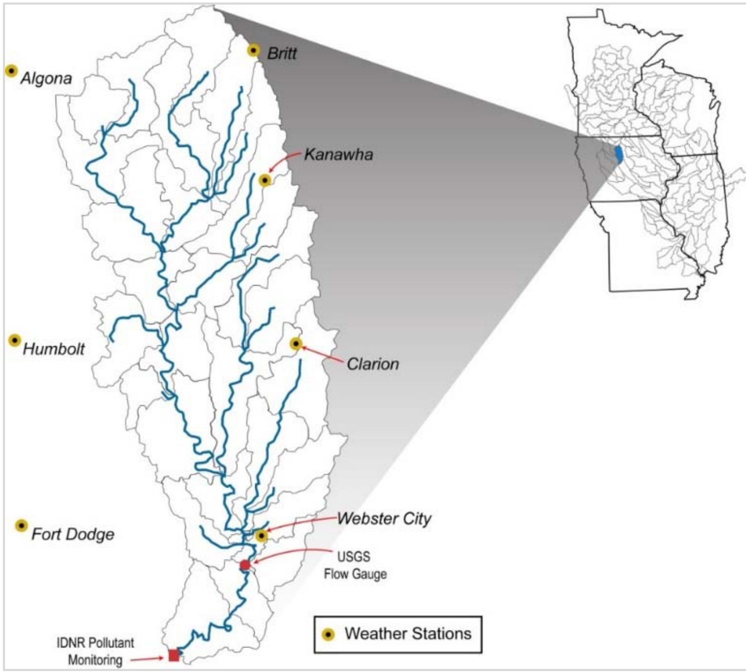

Figure 3-1. Boone River Watershed.

Land use in the watershed is dominated by agriculture: cropland represents 89.7% of total area, retired land represents 5.6% of total area, forestry represents 2.6% of total area, and urban areas and water surfaces account for the remaining 2.1% of total area. Most of the land is a flat, characterized by soils with low slopes (i.e., 73% of the areas have a slope less than 0.01 inches). The corn suitability rating (CSR) is another characteristic that defines potential yield.[19] It is an index that ranges between 0 and 100, where high values are associated with high quality soils. A soil with a high corn suitability index value is less likely to have high rates of fertilization, and at the same time is less likely to be considered for land retirement as a solution for reducing nitrate

[19] CSR is a ranking that rates different kinds of soils for their potential row crop productivity. It was developed for Iowa soils. Detailed information can be found at: (link)

loadings. In the BRW more than 50% of the soil has been rated with corn suitability index values ranging from 50 to 79, and 40% of the soil has corn suitability rating values higher than 80.

The required data for our modeling system (i.e., SWAT 2009) was collected at Common Land Unit (CLU) level.[20] More than 16,300 unit levels have been identified in the BRW. As an HRU is the unit required by SWAT model, the common land unit levels were regrouped into roughly 2,968 HRUs. Data related to crop rotation, land uses, fertilizer management, tillage, and conservation practices were provided by a field-level survey conducted by Kiepe (2005). Figure 3-1 shows the subbasin boundaries together with the location of the weather stations that provided the historical weather data to calibrate the model. The approach for simulating the water quality impact of conservation practices as well as weather, soils, and management characteristics are described in detail in Gassman (2008).

The Raccoon River Watershed (RRW)

The Raccoon River Watershed (RRW), as shown in Figure 3-2, is one of the largest watersheds in the state of Iowa. It covers an area over 9,400km^2 in west-central Iowa, being the Des Moines River major tributary. The RRW flows approximately 300 km from its origin in Buena Vista County to the confluence with the Des Moines River in the city of Des Moines.

The landscape in the south part of the watershed is characterized by higher slopes with many hills and a well-developed drainage system, while the landscape in the northern part is

[20] "A Common Land Unit (CLU) is the smallest unit of land that has a permanent, contiguous boundary, a common land cover and land management, a common owner and a common producer in agricultural land associated with USDA farm programs. CLU boundaries are delineated from relatively permanent features such as fence lines, roads, and/or waterways"(http://www.fsa.usda.gov).

characterized by lower slopes and poor surface drainage system (Schilling et al. 2008). With more than 73% of the planted area being use for corn and soybeans, the land use is dominated by

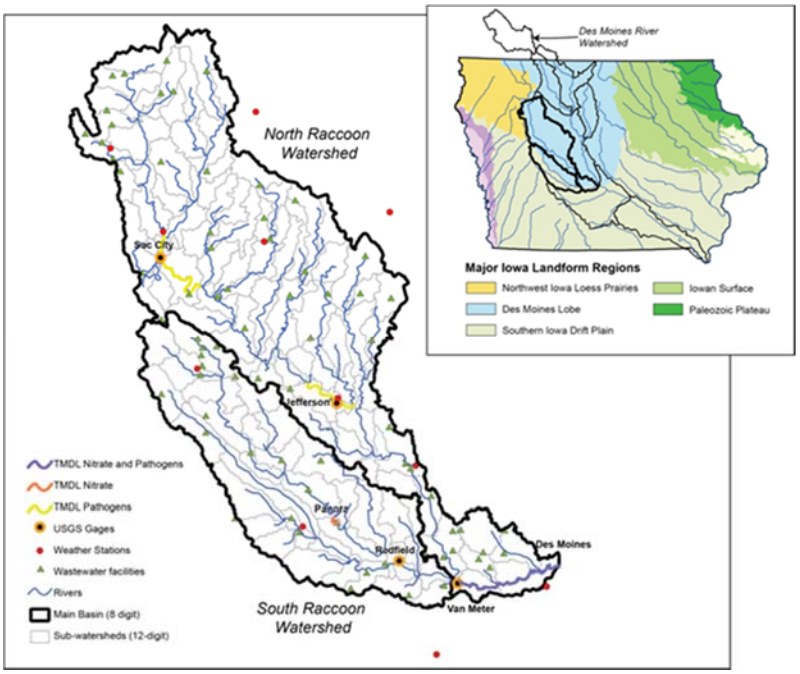

Figure 3-2. Raccoon River Watershed.

agricultural row production. Other land uses include grassland (16.3%), woodland (4.4%), and urban (4.0%) (Gassman and Jha, 2011). The applied fertilizer for corn is one of the main sources of nitrogen and phosphorus in the RRW. There are a significant number of cattle feedlots (135) and confinement operations (424), but there is little impact from cattle grazing on pasture (Schilling et al. 2008). Additionally, 77 waste facilities operate under the National Pollution Discharge Elimination System permit, which contributes a small amount of nitrate. During recent decades, nonpoint sources have been identified as the main contributing source to the high levels of nitrogen, phosphorus, and sediment (Jha et al. 2010; Schilling et al. 2008).

As stated earlier in SWAT, a watershed is divided into multiple subbasins or subwatersheds, further delineated into HRUs. A SWAT (2005) version is used for the baseline calibration and for the results calibration (Schilling et al. 2008; Jha et al. 2010). In this framework, the RRW is divided into 112 subbasins and 3,640 HRUs, with 1,569 being agricultural HRUs.

Figure 3-2 shows the subwatershed boundaries, the watershed stream network, and the location of climate stations used for establishing baseline stream flows and model testing, and impaired stream segments requiring the establishment of Total Maximum Daily Loads (as described in Schilling et al. 2008, and Jha et al. 2010).

Table 3-1. Watersheds: Summary Information

Watershed	Baseline N (kg)	Baseline P (kg)	Subbasins	Fields21	Area (km2)
Boone	4,725,826	218,828	30	2,968	2,370
Raccoon (RRW)	18,604,642	632,406	112	1,569	9,400

Table 3-1 summarizes the baseline nitrogen (N) and phosphorus (P) emissions as well as some of the characteristics for the two watersheds. The baseline values for both nitrogen and phosphorus represent the annual mean values computed using the available historical data for 1995–2001, and 1994–2000 for RRW with the first two years being dropped out for both watersheds.

3.4.2. Abatement actions (Conservation practices)

The set of conservation practices selected as abatement actions for achieving the nutrient loading standards includes reducing the rate of fertilizer application, conservation tillage (i.e., no till), cover crops, and land retirement. Since land retirement is often associated with the federal Conservation Reserve program, the acronym CRP is used. The above set is augmented with all

[21] SWAT HRUs

feasible combinations of these practices but land retirement (e.g.,. the combination of no till and cover crops is considered as an independent conservation practice). The baseline is also considered as a choice alternative. Table 3-2 provides a description of the abatement actions used in the empirical applications for BRW and RRW.

Table 3-2. The set of abatement actions

Abatement action	Abatement action description
Baseline	No action required
No Till (NT)	No till and no more than 30% of crop residue removed
Reduced Fertilizer (RF)	Reduce fertilizer application rate by 20%
Cover Crops (CC)	Establish cover crops between crop rotations
Land Retirement (CRP)	Retire land from production
NT, RF	No till and 20% reduction in nitrogen application rate
NT, RF	No till and no more than 30% of crop residue is removed
RF, CC	Reduce fertilizer and establish cover crops
NT, RF, CC	No till, 20% reduction in nitrogen application rate, and establish cover crops

3.4.3. The costs of abatement actions

Costs for each conservation practice are drawn from several sources, and all costs are expressed in dollars per acre. Table 3-3 summarizes the mean and standard deviations for assumed abatement actions implementation costs for the two watersheds. Per acre average cost for “No Till” and “Reduced Fertilizer” is lower for BRW, while per acre average cost for “CRP” is lower for RRW. The per acre adoption cost of “Cover Crops” is assumed to be the same for both watersheds

Table 3-3. Abatement actions: assumed costs

	Boone River Watershed Cost ($/acre)		Raccoon River Watershed Cost ($/acre)		
Conservation practice	Mean	Std.dev	Mean	Std.dev	Cost source
No action	0	0	0	0	
No Till	5.1	1.91	10.42	7.59	Kling et al. (2005)
Cover Crop	24.09	4.71	19.28	10.5	T. Kaspar[22]
Reduced fertilizer	7.25	5.22	2.52	1.37	Sawyer et al.(2006); Libra et al.(2004)
Land retirement	196.42	33.58	185.56	10.78	Kling et al. (2005)

An implied yield curve for corn-soybean rotation, where yield is estimated as a function of fertilizer applied, was used to derive the cost of reducing the fertilizer application rate. The procedure is similar to the procedure used by Rabotyagov (2007), Sawyer et al. (2006), and Libra, Wolter, and Langel (2004). Data from Iowa field experiments, available through ISU Extension was used to estimate an implicit nitrogen-based yield curve. The cost of nitrogen fertilizer reduction varies across fields based on the fertilizer application rate reported for the baseline scenario. The implied yield curve is a four-degree function of fertilizer rate.[23] The cost of reducing fertilization is given by multiplying a 20% reduction in the baseline fertilizer rate by the price of corn, set at $3.08 per bushel.[24] The cost of reducing the fertilizer application rate is reduced by the cost saving from applying less fertilizer. The cost of fertilizer is assumed to be $0.63 per pound.

[22] Personal communication

[23] The coefficients of nitrogen response yield curve Y=-3.3-e9*N^4+8.8-e6*N^3-0.005*N^2+0.83*N-0.37

[24] Price per bushel represents the average corn price for Iowa for 2004–2009. Source of corn price is: http://www.extension.iastate.edu/agdm/crops/pdf/a2-11.pdf

Cash rental rates available online (Edward and Smith 2009) in conjunction with the corn suitability rate available were used to compute the cost of retiring land out of production. The cost of land retirement for each field is obtained by multiplying the cash rental rate per unit of corn suitability rate by area and corresponding corn suitability rate. The cash rental rates are used as proxies for the opportunity cost of land retirement (Secchi and Babcock 2007). A zero cost is considered for no change from the baseline practices. The cost of the abatement actions obtained as a combination of the primary ones (i.e. no till and reduced fertilizer) are obtained by summing per acre cost of each conservation practice considered in the combination.

While watershed-level per acre abatement costs are assumed to be the same, the field abatement costs are a function of its characteristics. In the case of no till, the abatement costs depends on whether the baseline has conventional till or mulch till[25]. The full abatement costs applies if no till is adopted given that there is conventional till in the baseline while the cost for mulch till is half the full cost. Another example is the costs of land retirement. I assume the same costs per corn suitability index per acre, but the index differs across fields. The costs of reducing fertilizer varies across fields by construction, given that the fertilizer rate differs across field.

3.4.4. Obtaining the Point Value Estimates

Following the described procedure for obtaining the point coefficients, two sets of point coefficients are estimated for each watershed. The first set of points estimates the effectiveness of the abatement actions in reducing nitrogen emissions. The second set of points is estimated with respect to phosphorus emissions. A total of 2968 (fields) x9 (abatement actions) point coefficients are estimated for BRW and a total number of 1569 x 9 point coefficients are

[25] Mulch till is an intermediate type of no till.

estimated for RRW. Table 3-7 presents the estimates for point values as an area weighted average of the point estimates across the watershed. The point coefficients are expressed as per acre kilogram of abatement.

Table 3-4. Abatement point practices (area weighted average across watershed)

	No action	NT	CC	NT CC	RF	RF NT	RF CC	RF,CC NT	CRP
Boone River Watershed									
Nitrogen	0.00	2.35	2.42	4.26	0.62	2.98	2.95	4.79	7.32
Phosphorus	0.00	0.17	0.11	0.16	0.00	0.17	0.11	0.17	0.29
Raccoon River Watershed									
Nitrogen	0.00	1.50	2.66	3.33	0.79	2.28	3.31	4.02	7.97
Phosphorus	0.00	0.14	0.12	0.18	0.00	0.14	0.12	0.18	0.25

Next, I turn to discussing the point coefficients' results. In general, the results follow prior expectations. The abatement practices that are known to be highly effective at reducing one pollutant emissions are awarded higher point values than less effective practices (*i.e.*, land retirement receives the highest number of points for both pollutants). No till for N reductions in BRW receives a higher number of points than in RRW, but cover crops for P reductions receives more points in RRW than in BRW. Reduced fertilizer has the lowest number of points as it is the less efficient abatement practice for reducing nitrogen loss and has virtually no impact on reducing phosphorus loss.

Interesting sub-additivity patterns are realized in the points' estimation—the points associated with adopting a combination of conservation practices are not equal to the summation of the individual points. For example, the abatement action that combines no till and cover crops receives a lower number of points (4.26) than the sum of the points assigned to each of them individually (2.34 +2.42 = 4.76 BRW, nitrogen).

The difference in the magnitude of estimates for the two pollutants is explained by the difference in baseline overall emission levels, where the quantity of nitrogen measured at the main outlet is much higher than the quantity of phosphorus measured at the same outlet (the N baseline emissions are on average 200 times higher than the P baseline emissions—see Table 3-1 columns 2 and 3). Interestingly, the point estimates are comparable across the two watersheds (i.e., the point values for the same abatement practices are within comparable ranges).

Table 3-5. Efficiency of the abatement actions under uniform implementation (same abatement action is implemented by each field in the watershed)

Watershed		NT	CC	NT, CC	RF	RF,NT	RF,CC	RF, NT, CC	CRP
BRW	N red., %	28.8	25.1	48.1	6.3	35.2	30.5	53.5	81.0
	P red., %	37.7	27.5	33.8	0.2	38.0	27.8	34.4	77.4
RRW	N red., %	10.9	26.4	31.5	8.9	19.8	33.9	39.3	84.2
	P red., %	37.5	34.1	48.5	0.8	37.7	34.5	48.9	72.7

Prior expectations on the point coefficients' performance can be inferred from analyzing the obtained nutrient reductions assuming that the same abatement action is taken by each field in the watershed. Table 3-5 summarizes the overall reduction expressed both in relative and percentage terms that would be realized under this assumption. Among the abatement actions that represent a single conservation practice, land retirement offers the highest level of abatement for both N and P. Land retirement is followed by no till. Interestingly, more than double the overall N reductions are obtained under no till in BRW (28.10%) relative to the RRW (10.88 %). At the same time, similar P reductions across the two watersheds are obtained under the no till option. The N reductions obtained under cover crops are similar across the two watersheds (25%). However, more overall P reductions are obtained in RRW (34%) than in BRW (27.5%).

Reduced fertilizer is the conservation practice that has the least impact on N reductions and almost no impact on the P reduction, an outcome that it is expected.

The same pattern of sub-additivity is observed for the abatement actions that represent combination of two or more abatement actions, in the sense that less nutrient reductions are realized under the combination of conservation practice than the sum of the reductions obtained under the individual conservation practices. For example , in the case of BRW the combination no till and cover crops result in 48.1% N reductions which is less than the sum of individual reductions obtained under no till (28.8%) and cover crops (25.1%). This pattern is consistent across the two watersheds.

By using data provided by SWAT, I estimate a different set of point coefficients at different levels of aggregation: field, subbasin and watershed specific. A priori, the fields specific point coefficients should give a better approximation of the abatement function. However, a less specific set of point coefficients might be more appealing for the implementation of a trading program.

Figures 3-3 and 3-4 compare the point coefficients under three levels of specificity (field, subbasin and watershed) for BRW. The field and subbasin specific coefficients are obtained as an area weighted average. The nitrogen point coefficients are similar across the three types of estimation, with the exception of no till at field level, which has the average slightly below the average of no till subbasin and watershed point coefficients. More variation can be found across the phosphorus point coefficients, especially for no till, and the abatement actions that include no till, and land retirement

Figures 3-5 and 3-6 provide the same comparison for RRW. In the case of nitrogen, the no till average field level coefficient is higher than the other two types, while the land retirement field level average point coefficient is higher. Similar patterns as in BRW are observed for phosphorus in RRW.

Detailed results of the point value estimation can be found in the Appendix B. The summary statistics for the field specific points for each pollutant are presented in Table A-1 (nitrogen) and A-2 (phosphorus) for BRW and Tables A-3 (nitrogen) and A-4 (phosphorus) for RRW. Next, the subbasin specific point coefficients are presented in Tables A-5 and A-6. For BRW and Tables A-7 and A-8 for RRW. Finally, Figures A-1 to A-4 describe the distribution of point coefficients, where the point coefficients are field specific.

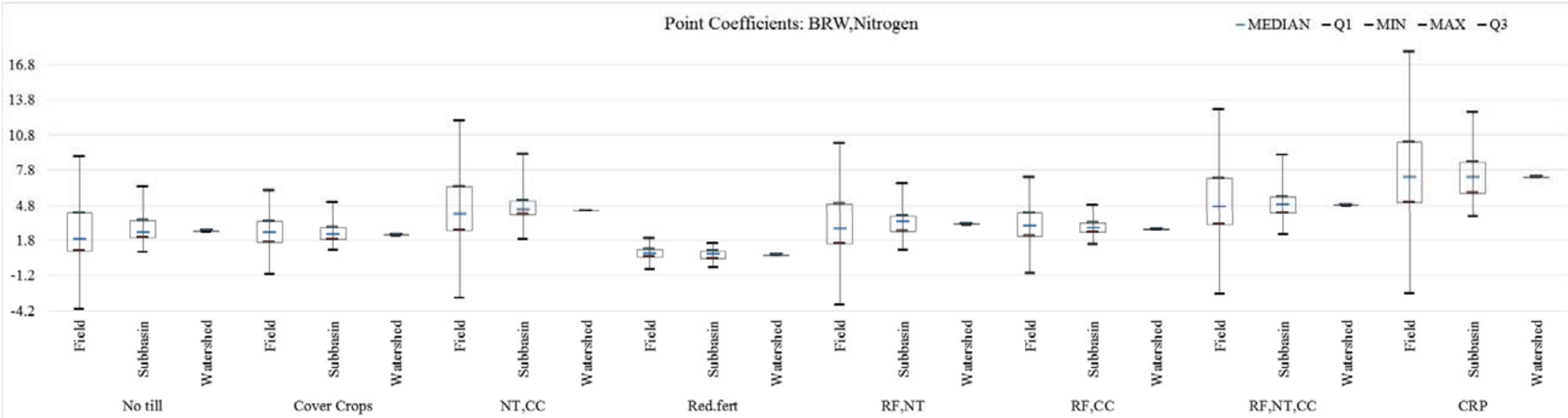

Figure 3-3. Boone River Watershed: different sets of point coefficients, nitrogen

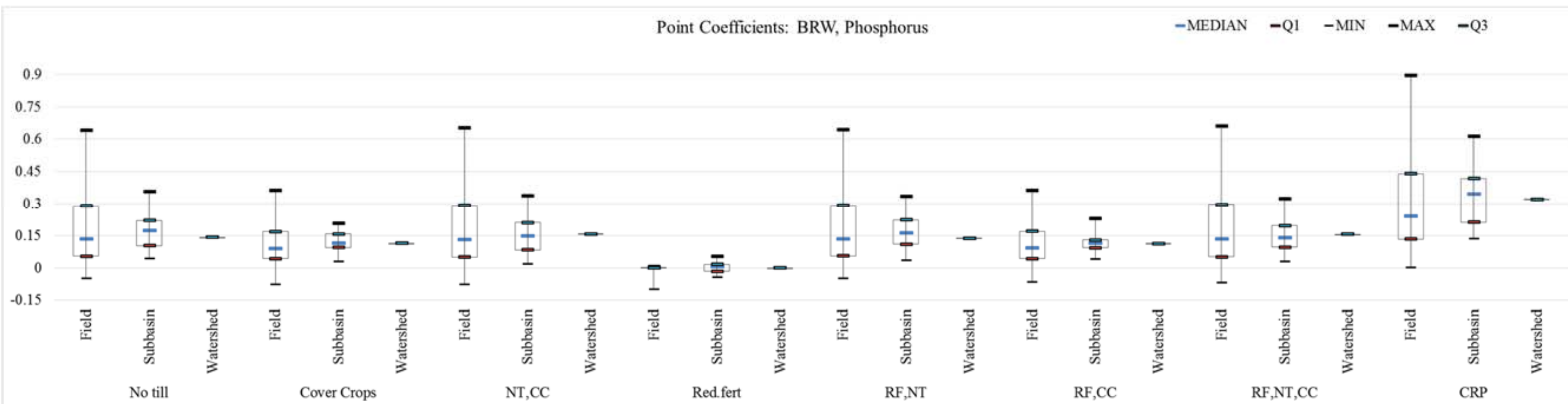

Figure 3-4. Boone River Watershed: different sets of point coefficients, phosphorus

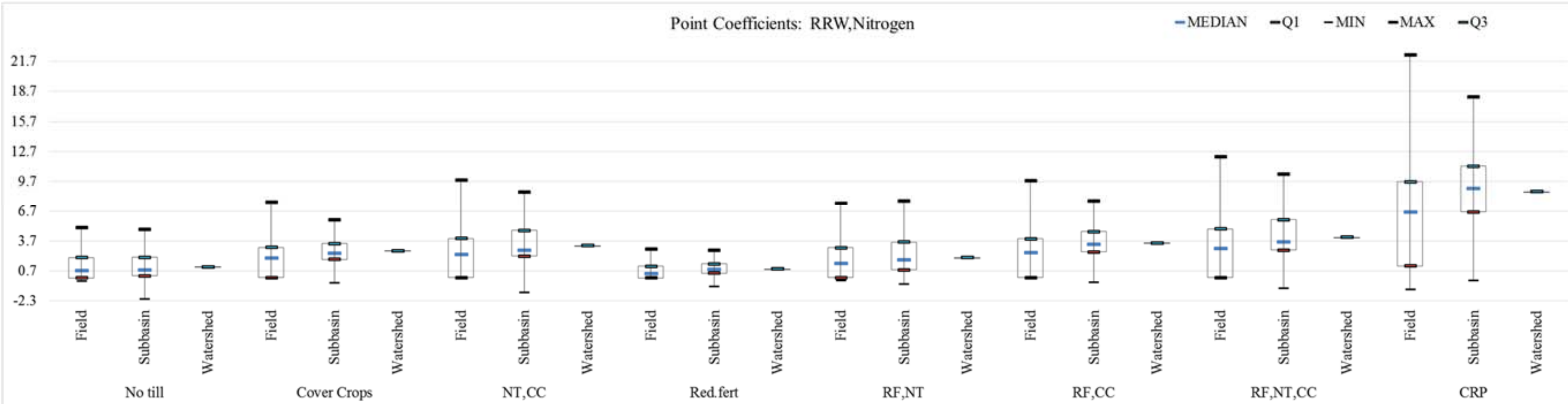

Figure 3-5. Raccoon River Watershed: different sets of point coefficients, nitrogen

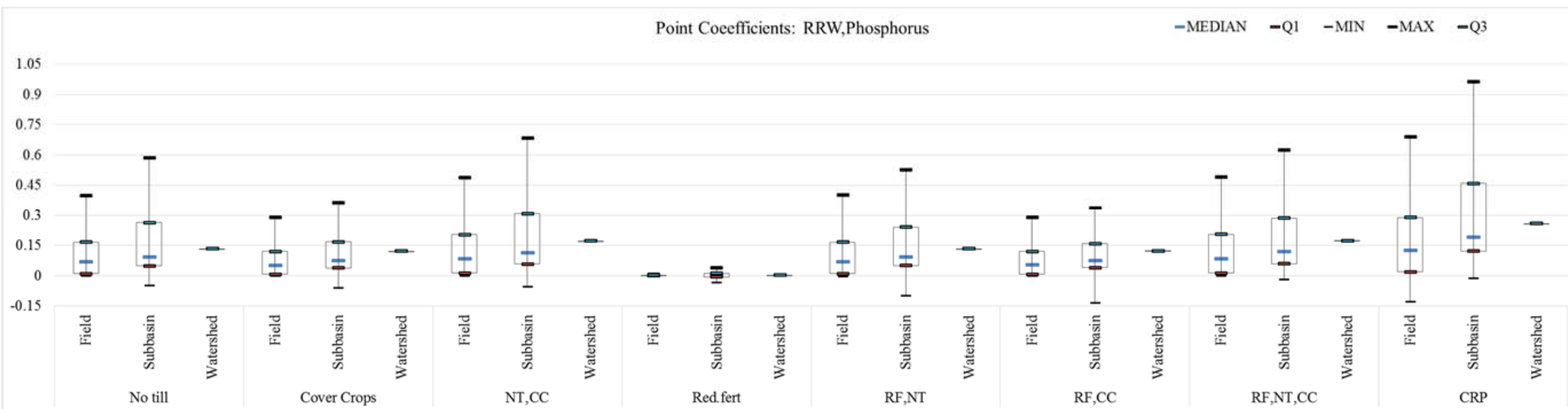

Figure 3-6. Raccoon River Watershed: different sets of point coefficients, phosphorus

Delivery coefficients

Estimating the delivery coefficients was an intermediate step in obtaining the field level point value estimates. The two additional approaches for estimating the point coefficients offer another two alternatives. The results for the delivery coefficients are presented in the Appendix B, Tables A-9 and A -10 for BRW, and respectively RRW.

Figure 3-7 compares the BRW distribution of the nitrogen delivery coefficients obtained under the three approaches. The delivery coefficients obtained under the three approach are labeled as Field, the ones obtained under the second approach are labeled as Subbasin, and finally the last ones are labeled as Watershed.

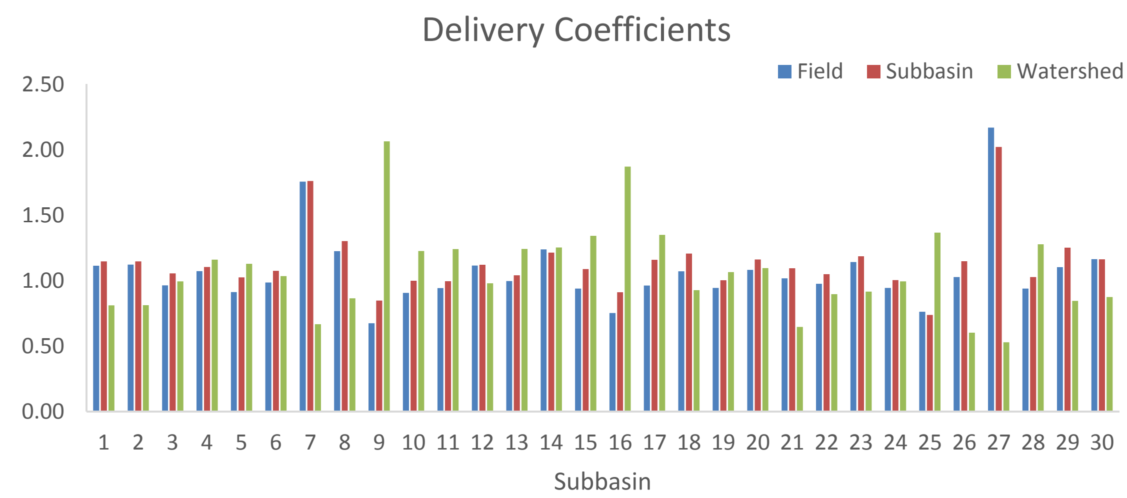

Figure 3-7. Boone River Watershed: nitrogen delivery coefficients

The “Field” and “Subbasin” delivery coefficients are very similar. Moreover, the t-test for equal means does not reject the equal mean hypothesis. However, the distribution of the Watershed delivery coefficients is different from the previous two (see subbasins 9 and 16 where the delivery coefficients are larger and subbasin number 26 and 27 where the coefficients are lower).

The literature related to delivery coefficients assumes that the delivery coefficients should between zero and one, or constrained to be between zero and one. In my empirical application, since my goal is to find a good linear approximation for the abatement function, I do not impose any constraint on the values the delivery coefficients can take. In the case of BRW, the delivery coefficients tend to be lower than one, however there are a few subbasins when the coefficients are higher than one. The average values across the watershed are 1.07 (Field), 1.14 (Subbasin) and 1.07 (Watershed).

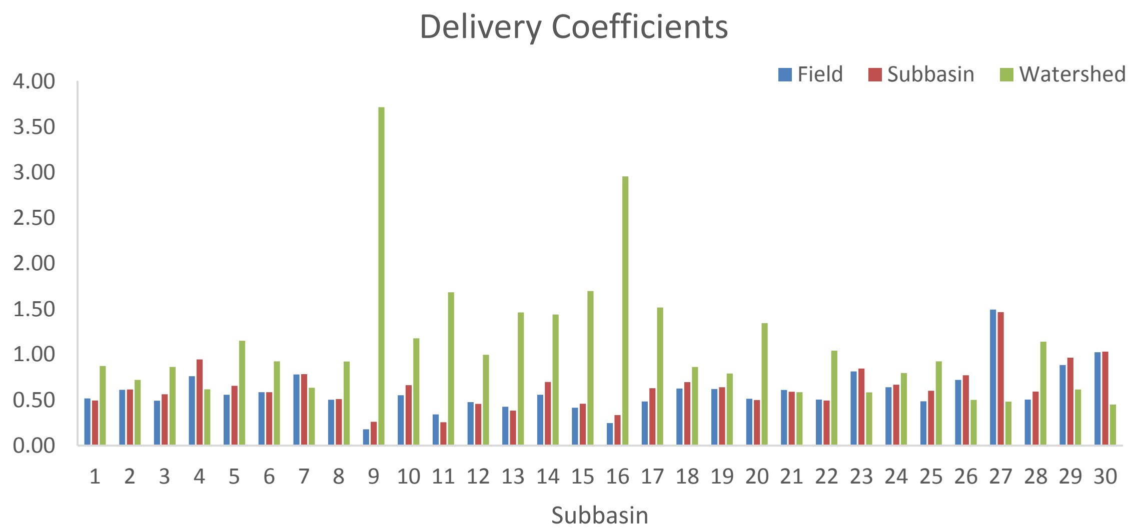

Figure 3-8. Boone River Watershed: phosphorus delivery coefficients.

Table 3-8 summarizes the distribution of the delivery coefficients for the transport of phosphorus in BRW. The average values across the watershed are 0.59 (Field), 0.63 (Subbasin) and 1.12 (Watershed). Hence, in this phosphorus case there is higher variations across the three types. As in the nitrogen case, the same subbasins (9 and 16) have the largest “Watershed” delivery coefficients with values much higher than one. There are no negative delivery coefficient

Figure 3-9. Raccoon River Watershed: nitrogen delivery coefficients.

Figure 3-9 summarizes the delivery coefficients distribution for nitrogen in RRW. As in previous cases, the Watershed delivery coefficients present the highest variations, with a few subbasins having negative delivery coefficients (subbasins 85 and 112). The average value of the delivery coefficients is smaller than in the BRW case: 0.67 (Points), 0.68 (Subbasin), and 0.98 (Watershed).

The phosphorus delivery coefficients follow similar patterns as the nitrogen in RRW, in the sense that the “Watershed” delivery coefficients have higher variability and the highest values (their overall average is 1.70, compared to 0.43 and 0.44—the averages for “Point” and “Subbasin”). As in BRW, the phosphorus delivery coefficients have lower values than the nitrogen coefficients.

The flow of trading

Once the point coefficients and delivery coefficients are estimated, the regulator can set up an abatement-action-based trading program where farmers can trade points associated with the abatement actions, or alternatively they can trade emissions based on the trading ratios defined by the true delivery coefficients. Next, I present the steps that would be followed in setting up a trading program based on the estimated point coefficients.

Determining the watershed configuration that achieves a water quality goal, $\bar{A}$

The regulator sets up a water quality goal, expressed as percentage reduction in the baseline level of total nitrogen or total phosphorus. Given no cost information is available, the regulator can identify a random placement of abatement actions such that the water quality goal is achieved.

Computing total number of points associated with a particular water quality goal $\bar{A}$

Let $\bar{A}$ be the water quality target and X be the vector of abatement actions that is determined by a random watershed configuration that achieves the desired water quality target, then the number of points corresponding to that water quality target, B, is equivalent to:

$$B = A(x) = \sum_i^N \sum_j^J a_{ij} x_{ij} \ s_i, \tag{17}$$

where x_{ij}^* is a dummy variable that takes a value of 1 if practice j is assigned to field i, and 0 otherwise, a_{ij} denotes the number of points corresponding to field i, given the abatement action j, and s_i is the area of field i.

Allocating a number of points to each field (this represent the field level constraints)

Next, the regulator has to decide how he is going to set the field- or farm-level constraints. In terms of practical implementation, farmers are provided with a set of point value estimates that specifies the points earned from the adoption of each abatement action. Given a watershed configuration that achieves a particular level of abatement, the corresponding total level of points is $B = A\ (x) = \sum_i^N \sum_j^j a_{ij} s_{ij} = \ \sum_i^N b_i^0$. The total number of points can be assigned as initial farm-level requirements in two ways: allocate the points according to the initial watershed configuration, or equally divide the total number of points among farms. The initial allocation of points will affect the final outcome of a performance-based program, but will not affect the final outcome in the case of a trading program.

The realization of the trading outcomes

Given the farm-level constraint, farmers choose the abatement actions and the number of points to trade. The final costs and abatement outcomes are realized.

Conclusion

In this chapter, I outlined the properties of three polices under different assumptions for the abatement function and proposed an approach for linearizing the abatement function using a system of point coefficients that measures the impact of an abatement function on the overall abatement level. I presented the results of estimating the point and delivery coefficients under different degrees of specificity.

In the next chapter, I evaluate these policies in a real watershed framework, where I anticipate potential tradeoffs between the cost efficiency and effectiveness of different policy programs, given that the complex water pollution process is simplified by using the proposed linearization.

CHAPTER 4. FLEXIBLE PRACTICE-BASED APPROACHES FOR CONTROLLING MULTIPLE AGRICULTURAL NONPOINT-SOURCE WATER POLLUTION

4.1. Introduction

In this chapter, I empirically evaluate the proposed policy approaches—focused on the set of abatement actions introduced earlier—for regulating emissions from nonpoint sources within two typical agricultural watersheds: the Boone River Watershed (BRW) and the Raccoon River Watershed (RRW). Specifically, I evaluate the potential tradeoffs between cost efficiency and effectiveness given that the different policies are implemented using a system of point coefficients that approximate the abatement actions' efficiency in reducing both field-level emissions and overall abatement.

Three different policies approaches are proposed: (*a*) a command-and-control program (CAC), where a regulator can mandate field-level abatement actions; (*b*) a performance standard program (PS), where the regulator provides only the field-level requirements (expressed as point values), but the farmers, given their private cost information, have the ability to choose the abatement action; and (*c*) a trading program where farmers can additionally choose the abatement action and trade point values as long as they meet the field requirements.

I begin by considering a case in which the proposed policies focus on either nitrogen (N) or phosphorus (P) abatement. Next, I consider the case where the proposed policies target both N and P simultaneously. I present the results for three levels of desired water quality improvement: 20%, 30%, and 40% desired reductions in mean annual loadings (N and/or P) relative to the baseline. A second set of simulations is obtained for the policies that target either nitrogen or phosphorus abatement under the assumption of cost heterogeneity.

The empirical assessment of the proposed policies for the single pollutant case is extended to include risk analysis under different time periods using the available historical weather data and a comparison of the trading outcomes when subbasin or watershed-specific points are used instead of field-specific coefficients.

4.2. Single Pollutant Policies Assessment

4.2.1. Evaluation of alternative policies using the points coefficients

To evaluate the performance of the three regulatory approaches, I solve for the least cost placement of the abatement actions across the watershed to achieve any given level of ambient water quality level using the full abatement function, instead of a linear approximation of it. As shown, in Chapter 2, Section 3.2.2, this problem has a high combinatorial nature, with a large search space and evolutionary algorithm that have been used to find the nearly optimal solutions, known as Pareto frontiers. I use a simulation-optimization system using SWAT and a modification of the Strength Pareto Evolutionary Algorithms 2 (Zitzler et al. 2002, as modified by Rabotyagov et al. 2010) to approximate the solution to a two-objective minimization problem, which simultaneously minimizes the five-year mean annual N (P) loadings and the costs of the abatement practices for each of the watersheds. The solution consists of a set of specific watershed configurations (placement of abatement action in the watershed) that achieves a particular level of N (P) loadings in the least cost way. The set of all least cost solutions obtained by using the evolutionary algorithm can be interpreted as an approximation to the first best solution, given a set of costs for the abatement actions.

4.2.2. Setting the goals for the three policies approaches

Under a CAC program, the regulator can mandate the farm-level abatement actions. If he is interested in achieving the abatement target, $\bar{A}$, then he needs to find the set of abatement actions corresponding to a watershed configuration (X_{CAC}) that satisfies $\{A(X_{CAC}) = \bar{A}\}$. He has at least two options to find the desired watershed configuration X_{CAC}.

The first option does not require any cost information and involves the evaluation of a range of different watershed configurations until the regulator finds one that meets his criteria $\{A(X_{CAC}) = \bar{A}\}$. This configuration is referred to as "satisficing" X_{CAC}^{Sat}, $\{A(X_{CAC}^{Sat}) = \bar{A}\}$. Alternatively, the regulator can use the abatement cost information available to him and solve for the least cost solution to achieve $\bar{A}$. Since this solution is the result of an optimization, it is referred to as "optimizing" X_{CAC}^{Opt}, $\{A\left(X_{CAC}^{Opt}\right) = \bar{A}\}$. In terms of my empirical applications, finding the optimizing watershed configuration implies selecting an individual (a solution) from the Pareto-frontier set that achieves the abatement level $\bar{A}$.

Under a CAC program, both X_{CAC}^{Sat} and X_{CAC}^{Opt} can be implemented directly, by mandating field level implementation. However, under the on-farm PS program or the point (credit) trading program, setting field-level requirements implies mapping the abatement actions to the on farm point coefficients or total watershed points requirements. For the performance standard, the farm-level requirements, $b_i^{0,PS,m}$ $m = Sat, Opt$, are computed using the field-level point estimates as $b_i^{0,PS,m} = \sum_j^J a_{ij} x_{ij}^m$, where x_{ij}^m is the abatement actions assigned to field i under X_{CAC}^{Sat} (X_{CAC}^{Opt} , respectively), and a_{ij} is the corresponding number of points assigned.

Next, the on-farm requirements can be summed up to determine the total watershed points, P^m, required for setting up a trading program, $P^m = \sum_i^N b_i^{0,PS,m} = \sum_i^N \sum_j^J a_{ij} x_{ij}^m$. The total number of points under a trading program, P^m is translated into farm individual-point requirements, $b_i^{0,trading,m}$, as $P^m = \sum_i^N b_i^{0,trading,m}$.

The initial (pre-trading) point allocations $b_i^{0,trading,m}$ may or may not correspond to the point requirements under a PS program ($b_i^{0,PS,m}$), as it can be translated into farm-level allocation of point requirements in any number of ways, such as using the same initial allocation used under a PS program, or alternatively to divide the total number of points equally across all the fields in the watershed.

Evaluating the three policies under the two options available for defining the field requirements (satisficing and optimizing) results in six different policies to simulate for each pollutant and each watershed. The results are obtained for three levels of water quality improvements: 20%, 30%, and 40% abatement in the mean expected annual loadings of N (P). Next, the results for each water quality target and pollutant are obtained using two sets of simulations.

In the first set of simulations, I assume that the farmers and the regulator have the same information on the costs of the abatement actions. The goal of this set of simulation is to assess the empirical performance of the three policies given a linear approximation of the abatement function is used. For each watershed and for each pollutant, using the evolutionary algorithm together with SWAT, I generate the set of least cost solutions for an entire range of abatement levels. Next, I am able to select the watershed configurations that achieve the desired abatement level X_{CAC}^{Opt}, $\{A\left(X_{CAC}^{Opt}\right) = \bar{A}\}$, where $\bar{A} = 20, 30,$ or 40% abatement target. Keeping the above costs assumption, I simulate the outcomes of the three policies with the targets being obtained both as X_{CAC}^{Opt}, or X_{CAC}^{Sat}. The outcomes of different policies are compared to achieving the abatement target (effectiveness) and relative to the cost of achieving the target given X_{CAC}^{Opt}

In the second set of simulations, I assume that the farmers and the regulator have different cost information. Hence, I explore how programs behave in the presence of information asymmetry, where the information asymmetry is simulated as cost heterogeneity. In this case, I assume that the regulator knows how the costs vary by field characteristics, but the costs also vary across the farms in the watershed due to the farmers farming abilities, θ_i.

4.2.3. The set of least-cost solutions

Figures 4-1 and 4-2 depict the set of all least cost solutions (Pareto solutions set) for the single pollutant optimization cases, for BRW, and RRW respectively.[26] The abatement levels as percentage reductions are depicted on the horizontal axis while the total costs expressed as thousands of dollars are depicted on the vertical axis. The Pareto solutions sets can also be interpreted as total abatement cost curves. Generally, the nitrogen and phosphorus frontiers follow similar patterns for the two watersheds. For lower abatement levels, the slope of the phosphorus frontier is lower than the slope of the nitrogen frontier. This implies that, generally, the phosphorus abatement levels can be obtained at lower costs than similar levels of nitrogen reductions. Only above a certain level of abatement is reducing phosphorus more expensive than reducing nitrogen. This threshold level is higher for RRW (about 60% abatement level) than for BRW (about 40% abatement level). The steeper curves should imply higher shadow prices for higher levels of abatement. The shape and the curvature of the Pareto sets suggest that although the same set of abatement actions is used in addition to having different set of abatement costs, the two watersheds also have different soil and agricultural characteristics.

[26] Table B1 in Appendix B summarizes the main parameters used by the evolutionary algorithms.

The presented least solution sets serve as a benchmark for comparing the performance of all three regulatory approaches, assuming that the cost of abatement actions are known both by the farmers and the regulators. Next, I summarize the cost-efficiency performance of the proposed policies assuming that the regulator and the farmers have the same cost information.

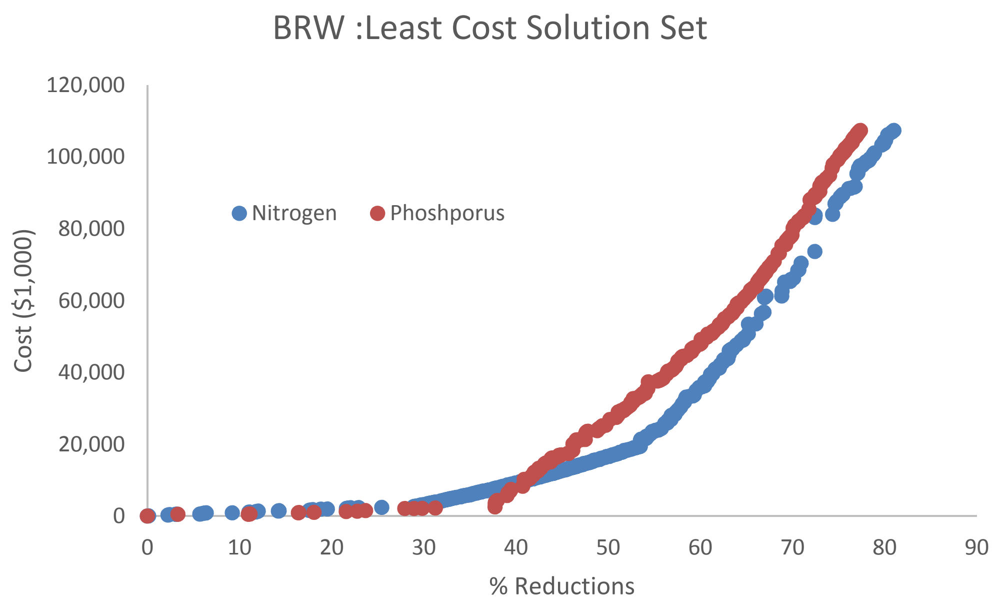

Figure 4-1. Boone River Watershed: Nitrogen and Phosphorus Pareto frontiers.

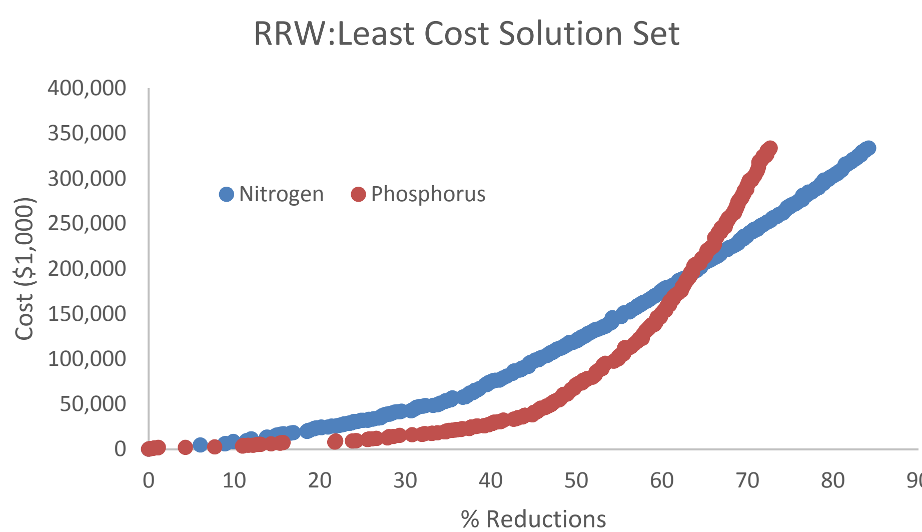

Figure 4-2. Raccoon River Watershed: Nitrogen and Phosphorus Pareto frontiers.

4.2.4. Cost-efficiency performance under the same cost information

In this set of simulations, I assume that the costs of abatement actions are known to both the farmers and the regulator, where the per acre abatement costs vary with the field characteristics. In the context of the empirical applications, the farm-level abatement cost functions are given by

$$C_i(x_i, \gamma_i, \bar{\theta}) = \sum_j^J c_j(\gamma_i) * x_{ij} * s_i \quad \forall\, i = 1, \ldots N \tag{18}$$

where $c_j(\gamma_i)$ is the per acre abatement costs for the abatement action j –per acre costs vary with field characteristics, x_{ij} takes a value of 1 for abatement j and 0 otherwise (i.e. the abatement actions are mutually exclusive), and s_i represent the area of field i.

The regulator finds the set of least-cost allocation solutions by using the cost functions detailed above. He chooses the solutions under which the desired abatement levels are met. In order to assess the cost efficiency and effectiveness of the three policies under a linear approximation of the abatement function (i.e., using the point coefficients) , I assume that the farmers have the same cost functions when they minimize their abatement costs under the PS or trading program.

The optimal solution under the PS program is obtained by solving the following linear programming problem:

$$\min_{x_{ij}} \sum_i^N \sum_j^J c_j(\gamma_i) * x_{ij} * s_i$$

$$s.t. \quad \sum_j a_{ij} x_{ij} \geq b_i^{0,PS,m} \quad \forall i = 1, \ldots, N \quad, m = Sat, Opt$$

$$\sum_j^J x_{ij} = 1 \;\; \forall i = 1, \ldots, \mathrm{N} \tag{19}$$

where a_{ij} is the number of point coefficients assigned to abatement action j, and $b_i^{0,PS,m}$ is the field standard given by the regulator. The field constraint can be obtained based on the least cost solution or based on a random allocation. The first set of constraints specify the field specific constraints, and the last set of equality constraints reinforce the fact that the choice is a binary variable.

The optimal solution under the trading program is obtained in a similar fashion by solving:

$$\min_{x_{ij}} \sum_i^N \{\sum_j^J c_j(\gamma_i) * x_{ij} * s_i \ + p b_i s_i\}$$
$$s.t. \quad \sum_{\mathrm{j}}^{\mathrm{J}} a_{ij} x_{ij} + b_i \geq b_i^{0,trading,m} \forall i \quad , m = Sat, Opt$$
$$\sum_j^J x_{ij} = 1 \quad \forall i = 1, \dots, N \qquad (20)$$

where b_i represents the number of per acre points a farmer will trade. The constraints for the PS program are adjusted to take into account the traded point values. Additionally, the market clearing condition is given by $\sum_i^N b_i s_i = 0$. I assume that all gains from trade are realized; hence the trading solution coincides with the solution of an omniscient social planner that solves the cost minimization problem defined by equation (18).

$$\min_{x_{ij}} \sum_i^N \{\sum_j^J c_j(\gamma_i) * x_{ij} * s_i \} \quad s.t. \ \sum_i^N \sum_{\mathrm{j}}^{\mathrm{J}} a_{ij} x_{ij} s_i \ \geq \ \sum_i^N b_i^{0,trading,m} s_i \ , \ \sum_{\mathrm{j}}^{\mathrm{J}} x_{ij} = 1 \qquad (21)$$

It can be shown that the shadow price of the constraint to the problem defined by equation (19) is equal to the equilibrium price for the point trading. Next, I discuss the cost efficiency and the effectiveness of the three different policy programs.

A priori, the realized abatement goal is expected to be achieved under a CAC policy, since the abatement actions are mandated. However, the outcomes of the PS and the trading policy are mandated. Nevertheless, the outcomes of the PS and the trading policy are the results of reallocating the points associated with the CAC set of abatement actions. Since the points are an approximation of both the abatement actions' effectiveness and of the fate and transport of the nutrients, these reallocations may result in the non- or over-attainment of the abatement goals. The extent to which the abatement goal is not met depends on the quality of the point approximation. The direction of the deviation depends on the curvature of the abatement function around the abatement goal. If the abatement function is concave in the edge-of-field reduced emissions, then a linear approximation results in an over achievement of the abatement goal. The reverse holds for a convex curvature of the abatement function.

Another conjecture that can be empirically tested is whether the realized abatement levels under the PS satisficing are higher than the corresponding abatement levels realized under the PS optimizing approach. This conjecture would hold if the satisficing approach, X_{PS}^{Sat}, selects more effective abatement actions, which in turn are more cost effective.[27]

The PS program allows only for within farm trading where farmers choose the abatement action based on the cost and the farm-level imposed constraint. The trading program allows for both within farm and across farms trading. This implies that both PS and trading outcomes should result in cost savings relative to the CAC costs. The magnitude of the cost savings should be higher for the satisficing approach and lower for the optimizing one, since the CAC-optimizing approach already represents the first best.

Given that the same costs information is used (i.e., the costs are assumed to be known by both the farmers and the regulator), the cost effective performance of the simulated outcomes can be compared with the corresponding CAC outcomes.

The PS optimizing and trading outcomes are expected to perform well relative to the CAC optimizing outcomes. Alternatively, the least flexible approach, CAC-satisficing, is

[27] The correlation between the point coefficients and per abatement costs is as it follows: 0.82 for N points and 0.7 for P point in BRW, 0.82 for N points and 0.39 for P points in the RRW. The correlation values are determined as average of the field-level correlation values.

expected to have a lower performance, and more flexible approaches gradually increase their performance as more freedom in choosing the abatement actions is allowed.

The emergence of hotspots, fields where the environmental outcomes get worse than under the baseline conditions, is a concern that may arise in the context of trading outcomes. To check if hotspots emerge as a result of the points based trading, I determine the number of fields that have negative abatement (i.e., an increase in the N or P emissions) relative to the baseline.

The PS and the trading optimal solutions are the result of the reallocations of the initial points (abatement actions) prescribed by the CAC. To measure the dynamic of the points' reallocation across the two programs, I determine the percentage area of a subbasin that switches to a different abatement action than the one prescribed by the CAC policy.

In the next section, I present the assessment of alternative policies using the point coefficients estimated at field level first using data available for BRW and RRW. I start with the policy programs focusing on the abatement of nitrogen in BRW, and continue with the program targeting the phosphorus abatement in the same watershed. The corresponding policy assessment for RRW is summarized in the second part of this section.

Table 4-1. Boone River Watershed: Simulated policy performance under varying nitrogen abatement targets[28,29,30] .

Target N reduction, % from baseline	CAC, optimizing		CAC, satisficing		PS, optimizing		PS, satisficing		Trading, optimizing		Trading, satisficing	
	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
20	20.73	1.79	20.86	7.23	26.34	1.67	31.52	5.80	19.81	0.85	22.21	1.10
30	30.12	3.23	30.12	19.78	30.05	3.10	40.24	18.59	29.47	2.27	31.98	2.99
40	40.00	9.01	40.00	29.60	39.37	8.94	45.30	28.55	39.35	6.06	41.40	7.16

[28] The placement of optimal abatement actions as well the cost values are obtained by solving equations (19)(PS) and (20)(trading).
[29] The abatement values are obtained by running the watershed configuration (the placement of optimal abatement actions) in SWAT for a period of seven years, disregarding the first two years and taking the average of the remaining five years.

[30] The satisficing allocations represent single random realizations of the abatement goals. The results obtained under this approach need to be interpreted with caution as these random realizations can results in higher or lower total costs.

Boone River Watershed nitrogen simulated policy performance

Table 4-1 summarizes the results for the reductions in expected mean annual loadings for nitrogen in the BRW under the three policy approaches. The table rows summarize the results for different levels of abatement expressed as percent reductions relative to the baseline mean annual loadings. The policy outcomes under the “Optimizing” and the “Satisficing” approaches are presented as ex-post mean annual percent reductions to the baseline and the total costs are expressed in millions of dollars.

Under the CAC policy, abatement actions are mandated, so the attainment of the abatement target is assured (see column 2). Notice that the cost of a CAC-satisficing policy is very large relative to the cost of a CAC-optimizing policy, being from four to seven times higher (see columns 3 and 5). With the exception of the 40% optimizing PS program, the PS reallocation of points result in the over-attainment of the abatement goals (see columns 6 and 8). The over-attainment is much higher for the satisficing PS approaches, being on average 5% higher than the original targets (e.g., the 30% satisficing PS results in a 40.24% abatement level, as shown column 6 vs. column 2). This finding supports the conjecture that the realized satisficing abatement levels are higher than the optimizing levels. While the costs of the PS policies are lower than the costs of CAC policies for both satisficing (columns 9 vs. 5) and optimizing approaches (column 7 vs. 3), the PS-satisficing costs are still higher than the PS-optimizing costs (column 9 vs. 7), although the magnitude of the differences is lower, ranging from three to five times higher.

In contrast, mixed results are obtained under the trading approaches. Under the satisficing approach the abatement targets are, on average, slightly over achieved (column 12), while they are slightly under achieved under the optimizing approach (column 10), although the magnitude of the non-attainment is fairly small (less than 1%). Further cost reductions relative to the CAC polices are observed under both optimizing and satisficing approaches. The magnitude of the cost reductions under the satisficing approach (column 13) is higher than 80% of the CAC satisficing total costs (column 5). Although the optimizing and satisficing trading costs are within similar ranges, a direct comparison cannot be made since they do no achieve the same level of N abatement.

The outcomes of the two trading approaches do not coincide because the total number of points is different (see Table 4-2 column “Total point values”). The total number of points is higher under the satisficing approach suggesting that abatement actions that accrue more points, and hence are more effective, are chosen under this approach. The equilibrium point prices that represent the shadow price of the environmental constraint are summarized in Table 4-2. As expected, the price per point increases with the abatement target. The obtained prices could be also interpreted as a per acre N reduction subsidy that should be offered as an alternative to a trading program. The prices under the optimizing approach are smaller than under the satisficing approach, as expected. This suggests that it is useful for a regulator to acquire some information on abatement costs and use that information to find the least cost solution

Table 4-2. Boone River Watershed: Total point values and point prices[31]

	Trading Optimizing		Trading Satisficing	
Target	Total point values	Price ($)	Total point values	Price ($)
20%	862,241	2.14	967,653	2.58
30%	1,288,380	5.13	1,412,248	6.38
40%	1,791,383	9.8	1,897,278	11.18

[31] Equilibrium prices are determined by solving equation (21)

As stated earlier, the PS and trading outcomes are the results of optimally reallocating the points initially assigned according to the CAC solutions. The results of these reallocations can also be summarized by: (a) the percentage of watershed's area allocated to a particular abatement action; and (b) the percentage of a subbasin's area that switches to different abatement actions other than the one prescribed by the CAC optimizing or CAC satisficing policy.

Table 4-3. Boone River Watershed:The distribution of abatement actions, optimizing policies, nitrogen (% area)

	Abatement goal 20%			**Abatement goal 30%**			**Abatement goal 40%**		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
No Action	31.9	34.1	64.0	0.1	2.4	25.6	0.1	0.9	9.1
No till (NT)	67.4	64.0	31.5	86.4	84.8	62.1	35.8	39.1	52.0
Cover Crop (CC)	0.1	0.1	0.0	0.2	0.1	0.1	0.1	0.2	0.4
NT,CC	0.1	0.1	0.2	0.8	0.9	0.9	0.2	8.3	12.4
Red.Fert.	0.2	0.4	1.2	0.0	0.7	0.6	0.1	0.7	0.9
Red.Fert., NT	0.2	1.1	3.0	12.1	10.7	9.9	32.6	28.1	17.1
Red.Fert.,CC	0.1	0.0	0.0	0.1	0.0	0.0	0.3	1.1	0.3
Red.Fert.,NT,CC	0.1	0.1	0.0	0.3	0.3	0.7	30.8	21.5	7.3
LandRetirement	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.6

Table 4-3 summarizes the distribution of abatement actions expressed as percentage of total watershed area allocated to an abatement action under the optimizing approach. The CAC and PS optimizing distributions for 20 % and 30 % N abatement tend to concentrate around "no till", while the distribution for 40% N abatement is more evenly distributed between no till, reduced fertilizer and no till, and reduced fertilizer, no till and cover crop. Additionally, the CAC and PS distributions are very similar (see column 1 vs. 2, 4 vs. 5, and 7 vs. 8). The trading distribution is more heterogeneous, including more abatement actions, but still similar to the CAC or PS distributions (column 3 vs. 1 , 6 vs. 4, and 9 vs.7).

Table 4-4. Boone River Watershed:The distribution of abatement actions, Satisficing policies, nitrogen (% area)

	Abatement goal 20%			Abatement goal 30%			Abatement goal 40%		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
No Action	19.9	23.5	56.7	20.5	23.1	17.7	17.1	18.7	7.6
No till (NT)	17.6	39.0	37.3	1.0	21.7	64.2	1.1	13.1	45.5
Cover Crop (CC)	13.5	5.6	0.0	8.2	3.2	0.0	10.1	5.3	0.4
NT,CC	2.4	3.8	0.3	1.4	6.4	2.6	13.8	17.1	15.9
Red.Fert.	19.6	4.3	1.3	21.9	5.0	0.8	9.1	2.5	0.9
Red.Fert., NT	16.9	18.3	4.1	10.1	11.6	13.4	8.2	9.8	18.5
Red.Fert.,CC	4.4	0.7	0.0	3.9	0.9	0.1	4.4	0.8	0.4
Red.Fert.,NT,CC	4.5	3.8	0.2	21.7	16.7	1.0	16.3	12.8	10.2
LandRetirement	1.1	1.1	0.0	11.4	11.4	0.2	19.8	19.8	0.7

Table 4-4 summarizes the distribution of abatement actions expressed as a percentage of total watershed area allocated to an abatement action under the satisficing approach. Relative to the CAC optimizing, the distributions under CAC present more heterogeneity. For example, under 30% N optimizing abatement, 86% of the area is allocated to no till and 12% is allocated to the combination of no till and reduced fertilizer, while the distribution under 30% satisficing has the following structure: 20% no action, 8% cover crop, 21% reduced fertilizer, 10 % reduced fertilizer and no till, 21% reduced fertilizer, no till and cover crops, and 11% land retirement. The PS distributions tend to have a different structure than the corresponding CAC (see columns 1 vs 2, 3 vs 5, and 7 vs 8).

Interestingly, the distributions under trading tend to have the same structures as the distributions under trading optimizing outcomes. This result highlights the fact that, for a trading program, the quality of cost information known by the regulator does not have a big impact on the trading outcome.

The change in the relative distribution of the abatement actions relative to the CAC distribution is measured at subbasin level as the percentage area that switches to a different abatement action. Table 4-5 and 4-6 summarize the above change for both optimizing and satisficing as the distribution of the subbasins across different levels of change. The first column in these tables gives the different levels of change (e.g., the first entry,"<=10%", the change in the area is less than 10% of the subbasin area). The rest of the columns counts the number of subbasins within a given level of change across different abatement targets (e.g., PS 20%, 24 subbasins out of 30 switch less than 10% of their total area to a different abatement action.

Table 4-5. Boone River Watershed: The change in the relative distribution of the abatement actions relative to CAC, Optimizing

	Optimizing Approach					
% of subbasin	Performance Standard			Trading		
Area/Abatement	20%	30%	40%	20%	30%	40%
<=10%	24	22	12	1	0	0
(10%- 20%]	6	6	8	0	3	0
(20%- 30%]	0	1	3	1	5	1
(30%- 40%]	0	1	6	3	5	2
(40%- 50%]	0	0	1	2	4	5
(50%- 60%]	0	0	0	6	5	2
(60%- 70%]	0	0	0	9	4	0
(70%- 80%]	0	0	0	7	1	7
(80%- 90%]	0	0	0	1	2	8
(90%- 100%]	0	0	0	0	1	5
Counts of subbasin	30	30	30	30	30	30

Under the optimizing policies, the intensity of change in the distribution increases with the level of abatement. Additionally, the change following the trading program presents more heterogeneity than under the change following the PS program. For example, for a 30% N abatement goal under PS optimizing 22 subbasins, out of 30, have less than 10% of their area switching to a different abatement action, while under the trading 4, 14, and 8 subbasins have 30%, between 30% and 60%, and more than 60%, respectively, of their total areas allocated to different abatement actions(Table 4-5).

Same patterns are observed for the change in the distribution of the abatement actions under the PS optimizing. However, relative to the optimizing case, the change following the trading program is more intense, in the sense that a larger number of subbasins switch a higher percentage of their area to a different abatement action. For example, under 30% N satisficing 25 subbasins have more than 90% of their area allocated to a different abatement action (Table 4-6).

Table 4-6. BRW: The change in the relative distribution of abatement actions relative to CAC, satisficing

	Satisficing approach					
% of subbasin	Performance Standard			Trading		
Area/Abatement	20%	30%	40%	20%	30%	40%
<=10%	0	1	0	0	0	0
(10%- 20%]	3	3	9	0	0	0
(20%- 30%]	10	11	14	0	0	0
(30%- 40%]	8	5	6	0	0	0
(40%- 50%]	7	8	1	0	0	0
(50%- 60%]	2	2	0	0	0	0
(60%- 70%]	0	0	0	1	0	0
(70%- 80%]	0	0	0	11	0	1
(80%- 90%]	0	0	0	12	5	5
(90%- 100%]	0	0	0	6	25	24
Counts of subbasin	30	30	30	30	30	30

The above results obtained for both the distribution of the abatement actions as well as the change in the distribution relative to the CAC outcomes are as expected. The CAC optimizing is an approximation of the least cost solution, hence the distribution of the abatement actions is more homogeneous, and less change is expected under the PS and trading program. Moreover, since PS is less flexible, less change in the distribution is expected. The CAC satisficing is a random allocation; hence significant changes are expected under the PS and trading when the abatement cost information is used.

The fact that the PS-optimizing and trading outcomes are comparable to the CAC optimizing solutions indicates that the overall mix of the abatement actions and their spatial distribution is similar to the solutions discovered by the evolutionary algorithms. Figures 4-3 and 4-4 represent the spatial distribution of the abatement action for a 30% N reductions abatement goal for both satisficing and optimizing approaches. Figure 4-4 also suggests similarities between the CAC optimization solution discovered via evolutionary algorithm and the PS and trading outcomes, the solutions discovered via linear programing, while Figure 4-5 depicts the spatial heterogeneity under the satisficing approaches for CAC, and similarities between the two trading outcomes (optimizing and satisficing).

As shown above, although the distribution of the abatement actions within watersheds for the satisficing and optimizing trading outcomes are similar, the changes at subbasin level are different. Figure 4-3 depicts the spatial representation of the change in the distribution relative to CAC for 30% N abatement by subbasins. The overall change at watershed level measured as a weighted average of the within subbasin change is summarized in Table B-4. In the case of 30% N abatement goal, on average 93% (47%) of the area is allocated to a different abatement action under the trading satisficing (optimizing), hence the change in the distribution is more intense under the satisficing as expected.

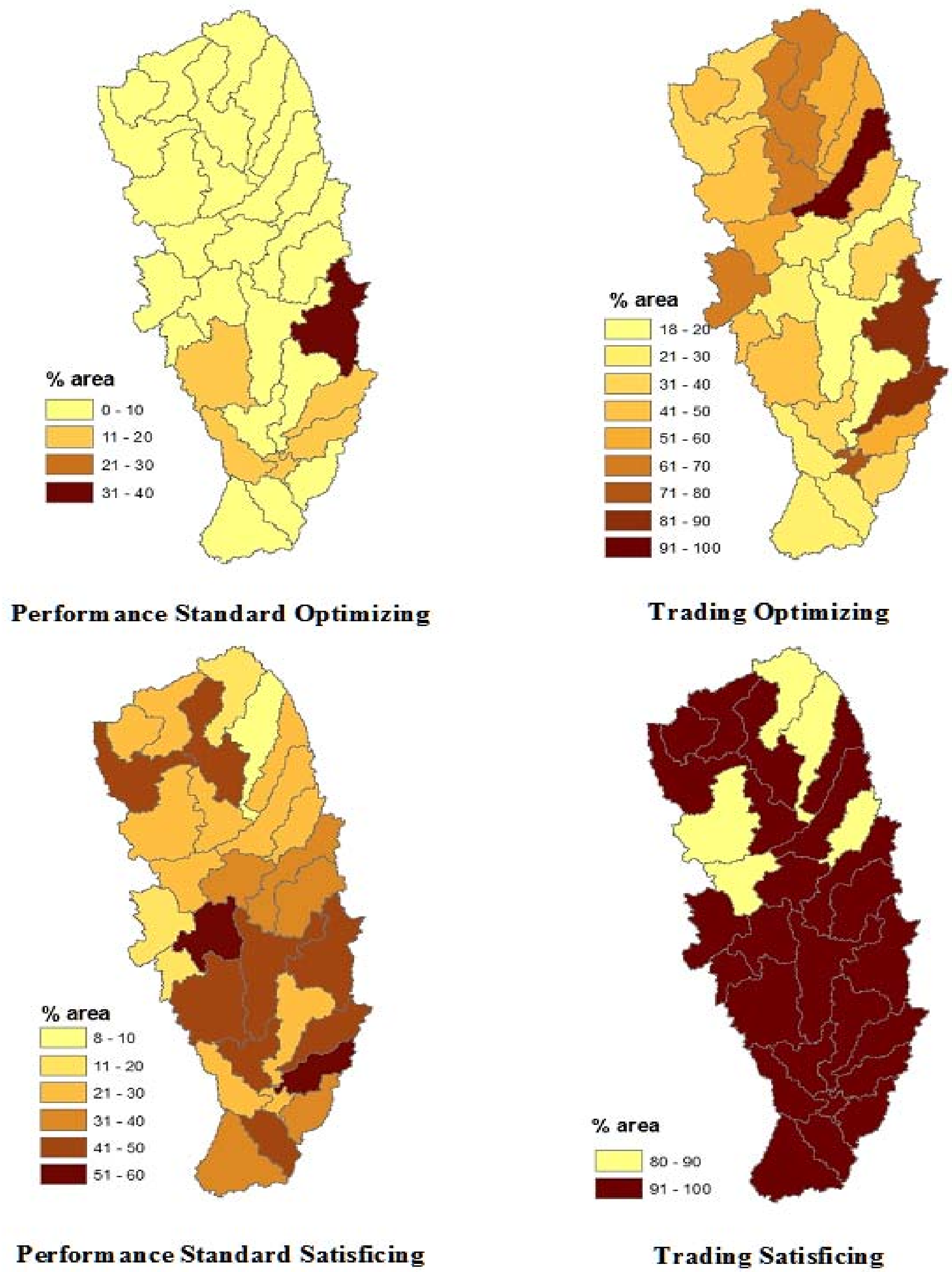

Figure 4-3. Boone River Watershed: Spatial representation of the change in the distribution (percent of subbasin area), 30% N abatement goal.

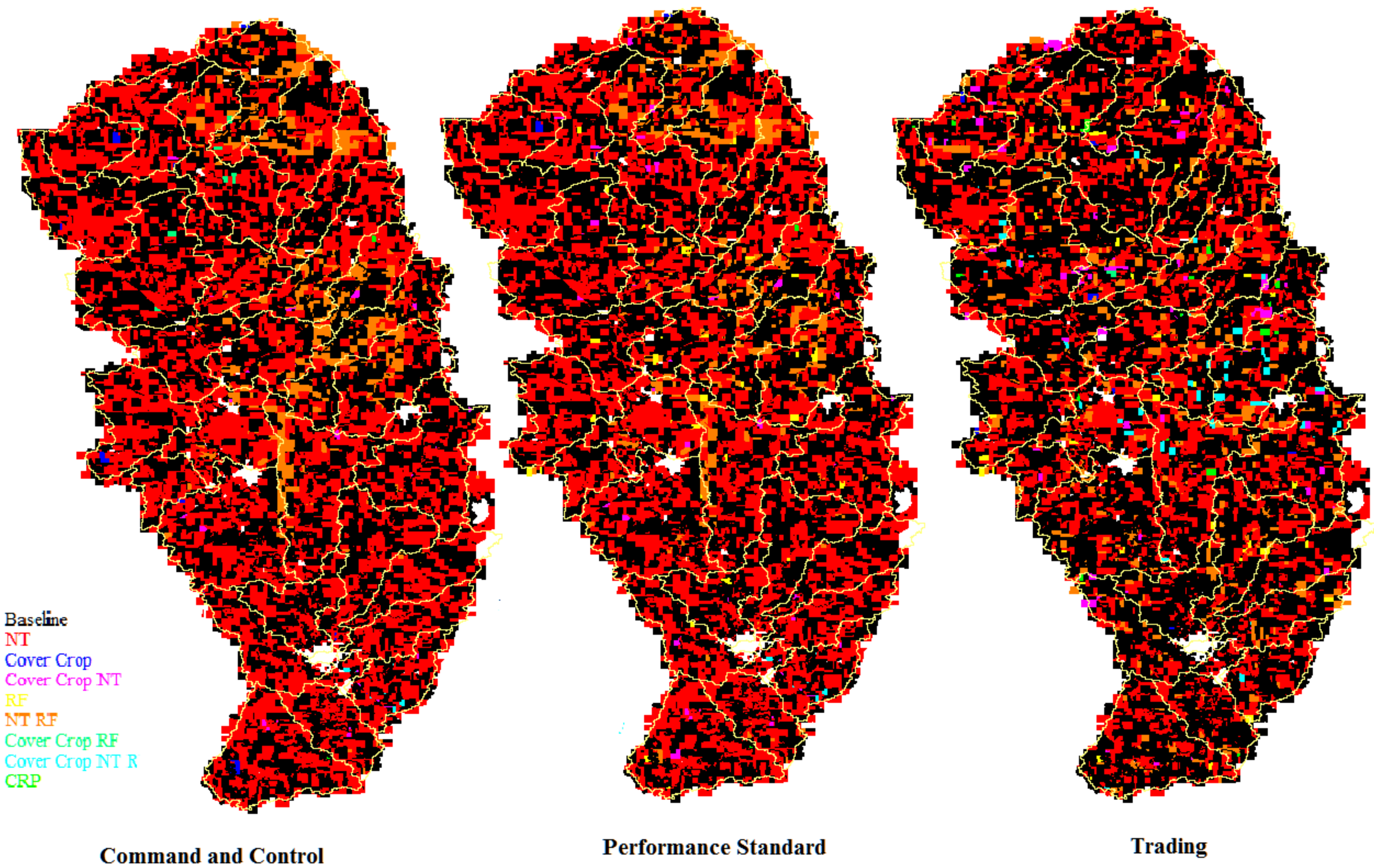

Figure 4-4. Boone River Watershed: The spatial distribution of abatement action, 30% N abatement goal, optimizing,

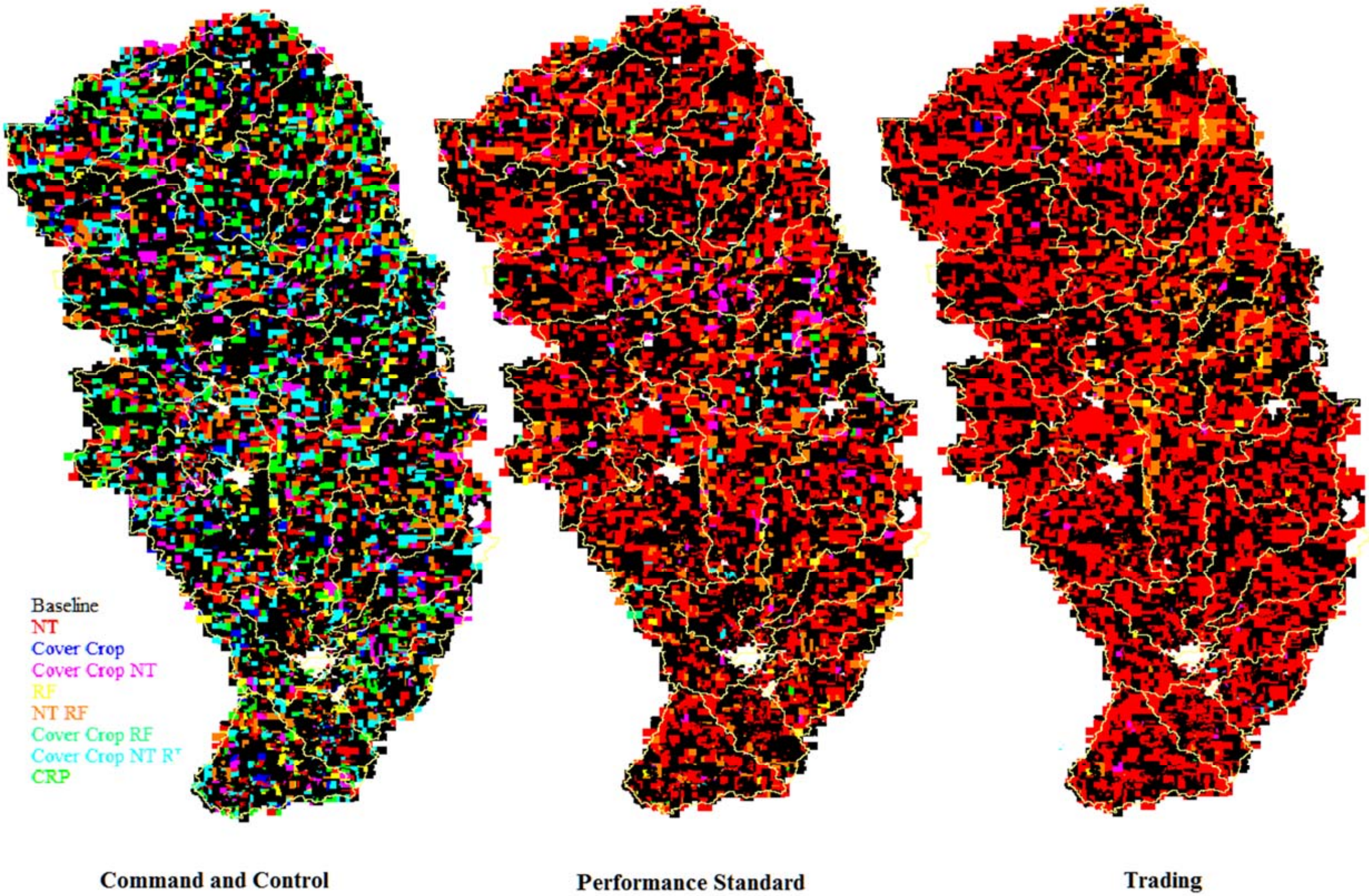

Figure 4-5. Boone River Watershed: The spatial distribution of abatement action, 30% N abatement goal, satisficing

Figure 4-3 depicts the spatial representation of the changes in the distribution of the abatement actions under CAC for 30% N abatement. The lighter colored areas show that in that subbasin a smaller percentage of the area switch to a different abatement actions.[32]The figure depicts the changes across PS and trading, for both satisficing and optimizing.The emergence of hot spots is a common concern for the trading settings. By evaluating the trading outcomes on the field (HRU) level, no evidence is found to support the existence of hotspots. Figure 4-6 introduces the histogram of abatement efforts corresponding to satisficing and optimizing trading outcomes for a 30% N abatement goal, where the abatement effort is measured as a percentage reduction relative to the baseline. Notice again, the similarities of the distributions across the two trading approaches. Similar distributions for 20% and 30 % can be found in Appendix B, Figures B1 and B2.

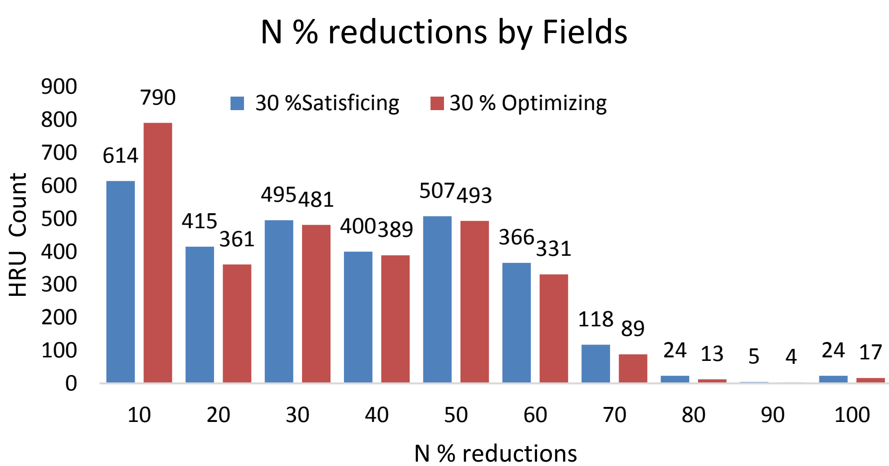

Figure 4-6. Boone River Watershed: Distribution of abatement effort corresponding to trading outcomes, N 30% abatement goal

[32] The lighter contour lines delimitate the subbasins

Table 4-7. Boone River Watershed: Simulated policy performance under varying phosphorus abatement targets

Target N reduction, % from baseline	CAC, optimizing		CAC, satisficing		PS, optimizing		PS, satisficing		Trading, optimizing		Trading, satisficing	
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million
20	21.57	1.24	20.04	9.45	21.66	1.07	26.63	3.72	21.92	0.49	20.50	0.42
30	29.82	2.15	30.12	16.35	29.86	2.00	34.90	11.25	29.98	1.03	29.35	0.97
40	40.00	8.14	40.01	35.53	40.08	7.28	41.75	32.39	40.30	4.60	37.07	2.02

Boone River Watershed phosphorus simulated policy performance

Table 4-7 summarizes the results for reductions in the phosphorus mean annual loadings in the BRW under three different policy approaches. As in the case of nitrogen, the CAC-satisficing outcomes have much higher costs than the CAC-optimizing outcomes, again the magnitude being between four and seven times higher (column 3 vs. 5). The realized abatement levels under the PS-optimizing almost replicate the corresponding levels under the CAC (column 6 vs. 2). Again, the PS-satisficing results in higher realized levels of abatement than the corresponding PS-optimizing. The magnitude of the overachievement in the abatement PS-satisficing outcomes is large in the case of 20% and 30% P abatement (see column 8). However, mixed results are obtained under the trading policies. For example, the abatement goal is not achieved under the 40% abatement trading-satisficing approach, but it is achieved under the trading-optimizing (see columns 10 and 12).

An interesting result emerges for the PS and trading-optimizing results: for all three abatement goals, the CAC-optimizing solutions are outperformed by the PS outcomes, implying that more reductions are obtained at lower cost (e.g., the CAC-optimizing cost for achieving 30% P abatement is $2.15 million, while the costs under PS is $2.0 million) . This is explained by the fact that the solutions provided evolutionary algorithms are themselves an approximation, and further improving can be obtained through linear programming (Whittaker et al. 2009).

The optimizing and satisficing trading outcomes cannot be compared directly since the total point values are different. Compared to the nitrogen case, the total points under the satisficing policies is higher than the satisficing one. The trading equilibrium prices reflect the difference in the total point values being higher for the optimizing outcomes. The shadow prices also reflect a sharp slope of the least-cost solution set around 40% P. The price per point for 40%

P optimizing is much higher than the price for 30% ($593 vs. $38.18), as shown in Table 4-8. The point prices for the P abatement are much higher than the corresponding prices for N abatement.

Table 4-8. Boone River Watershed: Total point values and point prices, phosphorus[33]

	Trading Optimizing		Trading Satisficing	
Target	Total point values	Price ($)	Total point values	Price ($)
20%	50,199	21.47	46,915	18.65
30%	69,026	38.18	67,487	35.90
40%	92,559	593.79	85,390	109.34

The same patterns as in the nitrogen case are obtained for the spatial distribution of the abatement actions: the distributions under CAC and PS optimizing are more skewed towards no till, while the distributions under CAC and PS satisficing are more diverse. Again, the trading distributions across the approaches are similar. The detailed distributions are summarized in Appendix B, Table B-2 (optimizing) and B-3 (satisficing).

Table 4-9. Boone River Watershed: The change in the relative distribution of the abatement actions relative to CAC, satisficing

	Satisificing approach					
% of subbasin	Performance Standard			Trading		
area	20%	30%	40%	20%	30%	40%
<=10%	0	0	0	0	0	0
(10%- 20%]	0	0	0	0	0	0
(20%- 30%]	0	1	9	0	0	0
(30%- 40%]	0	2	11	0	0	0
(40%- 50%]	4	7	5	0	0	0
(50%- 60%]	6	9	3	0	0	0
(60%- 70%]	9	8	1	0	1	0
(70%- 80%]	10	3	1	8	3	1
(80%- 90%]	1	0	0	11	17	8
(90%- 100%]	0	0	0	11	9	21
Counts of subbasin	30	30	30	30	30	30

[33] Equilibrium prices are determined by solving equation (19).

The patterns in the change in the CAC abatement actions' distributions follows the same patterns described for nitrogen (Table B-4) . However, in the case of phosphorus, the change in distribution is more intense under the PS satisficing, with more subbasin having more area switching to a different abatement action (Table 4-9). For example, under 30% N satisficing, in 20 subbasin, less the 50% of area is switching, compared to 5 subbasins under 30% P satisficing. The same patterns are obtained under the 20% and 30% PS satisficing. Figure B-3 in Appendix B depicts the spatial representation of the change in the distribution relative to CAC for 30% P abatement.

Several hotspots emerge under the phosphorus trading, although the number of fields is small relative to the total number of fields. For the 30% abatement level, there are 14 of 2,968 fields that have worse outcomes than under the baseline. A complete summary of the number of hotspots that emerge can be found in Table B-6. The distribution of the abatement effort for a 30% P abatement is provided below in Figure 4-7. Relative to the corresponding distribution for 30% N abatement, the distribution under P have less variations, with a higher number of fields having similar values for the abatement effort. The distribution of the abatement efforts for the 20% and 30 % P abatement are provided in Appendix B (Figure B-4 and B-5).

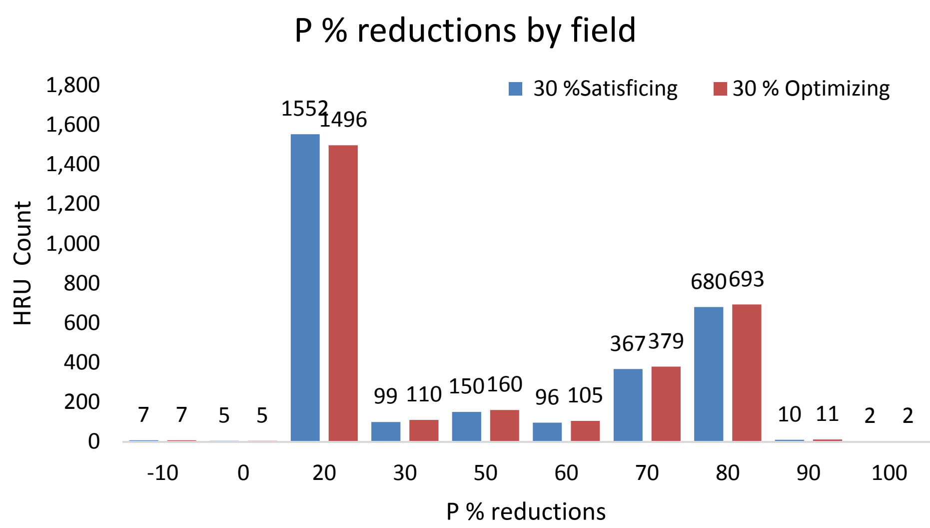

Figure 4-7. Boone River Watershed: Distribution of abatement effort corresponding to trading outcomes, P 30% abatement goal.

Raccon River Watershed Nitrogen and Phosphorus Simulated Policy Performance

Table 4-10 and 4-11 summarize the results for the reductions in mean annual loadings, for RRW under the three policy approaches, for N and P, respectively. The costs under CAC nitrogen satisficing approach are higher than under the CAC nitrogen optimizing approach, although the magnitude is much lower, up to 1.9 times higher compared to 4 to 7 times higher as it is the BRW case (Table 4-10 column 3 and 5). The magnitude in the cost difference is higher for the phophorus outcomes, up to five times higher. As in the BRW, significant cost savings relative to the CAC outcomes are observed across PS and trading policies.

The PS realized abatement levels are almost identical to the abatement levels and the trading outcomes are slightly below the abatement targets. However, the optimizing PS realized abatement levels are slightly superior to the satisficing corresponding outcomes. As before, the outcomes under PS optimizing outperform the CAC optimizing outcomes both under N and P.

Table 4-10. Raccoon River Watershed: Simulated program performance under varying nitrogen abatement targets, nitrogen

Target N reduction, % from baseline	CAC, optimizing		CAC, satisficing		PS, optimizing		PS, satisficing		Trading, optimizing		Trading, satisficing	
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
	N red.	$,million	N red.	$,million	N red.	$,million	N red.	$,million	N red.	$,million	N red.	$,million
20	20.20	23.77	20.01	36.03	20.43	21.61	20.31	33.26	19.97	14.36	20.47	15.22
30	30.67	42.45	30.00	61.96	30.83	39.59	30.31	58.74	29.22	31.41	29.42	31.81
40	40.23	75.25	40.00	130.64	40.34	70.91	40.23	126.75	38.94	56.39	39.18	57.54

Table 4-11. Raccoon River Watershed: Simulated program performance under varying nitrogen abatement targets, phosphorus

Target N reduction, % from baseline	CAC, optimizing		CAC, satisficing		PS, optimizing		PS, satisficing		Trading, optimizing		Trading, satisficing	
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million	N red.	$, million
20	21.77	7.70	20.00	36.60	21.81	7.53	21.89	30.61	21.62	4.30	18.65	3.17
30	30.82	15.84	30.00	36.29	30.84	15.81	30.66	31.45	30.90	9.97	29.46	8.80
40	40.02	28.14	40.01	102.13	40.02	28.07	42.24	92.86	40.07	20.15	37.57	16.76

Table 4-12 presents the total point values and the point prices for RRW for both nitrogen and phosphorus. The figures follow similar patterns to BRW: the total point values and the point prices are higher under the satisficing approach, and the total point values are lower for P abatement but the point prices for P are higher relative to the corresponding N values.

Table 4-12. Raccoon River Watershed: Total point values and point prices, nitrogen and phosphorus

	Nitrogen				Phosphorus			
	Trading optimizing		Trading satisficing		Trading optimizing		Trading satisficing	
Target	Total point values	Price ($)	Total point values	Price ($)	Total point values	Price ($)	Total point values	Price ($)
20%	3,748,119	8.38	3,848,234	8.72	125,827	48.85	145,007	68.24
30%	5,372,838	12.65	5,404,148	12.74	194,944	116.42	204,420	130.92
40%	6,979,735	19.08	7,039,798	19.27	247,321	189.39	263,615	229.01

Table 4-13. Raccoon River Watershed: The distribution of abatement actions, optimizing policies, nitrogen (% area)

	Abatement goal 20%			**Abatement goal 30%**			**Abatement goal 40%**		
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
No Action	0.6	5.6	20.8	0.0	3.6	9.7	0.1	3.0	3.8
No till (NT)	0.2	7.3	9.1	0.0	2.5	7.2	0.2	1.7	3.1
Cover Crop (CC)	0.3	2.9	1.2	0.1	12.9	7.4	0.3	13.5	11.3
NT,CC	0.1	0.5	1.6	0.1	1.1	3.0	0.2	1.5	6.5
Red.Fert.	48.7	44.7	40.2	15.5	14.4	21.4	0.2	0.1	4.7
Red.Fert., NT	22.6	18.5	18.8	0.4	3.5	15.1	0.4	2.9	6.7
Red.Fert.,CC	22.9	17.5	5.3	83.8	62.0	24.6	83.6	64.3	35.0
Red.Fert.,NT,CC	4.5	3.0	3.0	0.1	0.1	11.6	5.7	4.3	26.1
LandRetirement	0.1	0.1	0.0	0.0	0.0	0.0	9.4	8.8	2.9

Table 4-13 summarizes the distribution of the abatement actions under the N optimizing abatement. Similar patterns to BRW are obtained in this case too: the CAC and PS distributions have similar structure (see columns 1 vs. 2, 4 vs. 5, and 7 vs. 8). However, relative to the BRW, the distributions have a different structure - the abatement actions reduced fertilizer and cover crop being predominant (see row Reduced Fertilizer). The distributions for N satisficing and P optimizing and satisficing have similar characteristics to the BRW, therefore the results can be found in Appendix B Tables B-7 (N satisficing), B-8 (P optimizing), B-9 (P satisficing).

The changes in the CAC distributions of the abatement actions also follow similar patterns to the ones described in BRW: less area is switching to a different abatement action than the one described by CAC under the optimizing relative to the optimizing and more area is switching under the trading than under the PS program. The underlying results are summarized in Tables B-9 to B-12 in Appendix B.

Table 4-14. Overall change in distribution of CAC abatement actions

		Nitrogen		Phosphorus	
Approach	Target	PS	Trading	PS	Trading
Satisficing	20%	20.46	78.76	41.08	73.91
	30%	19.77	86.89	58.84	99.13
	40%	14.90	93.66	42.61	96.51
Optimizing	20%	17.99	73.70	3.04	31.05
	30%	23.29	71.46	1.07	44.93
	40%	21.87	64.90	1.61	44.46

Table 4-14 summarized the overall change in the CAC distribution as an area weighted average of the changes at the subbasins level. Overall, there is less change under the optimizing approaches. Notice that the overall change under phosphorus PS outcomes is less than 3%. Also, there is no clear relation between the magnitude of the change and the abatement target.

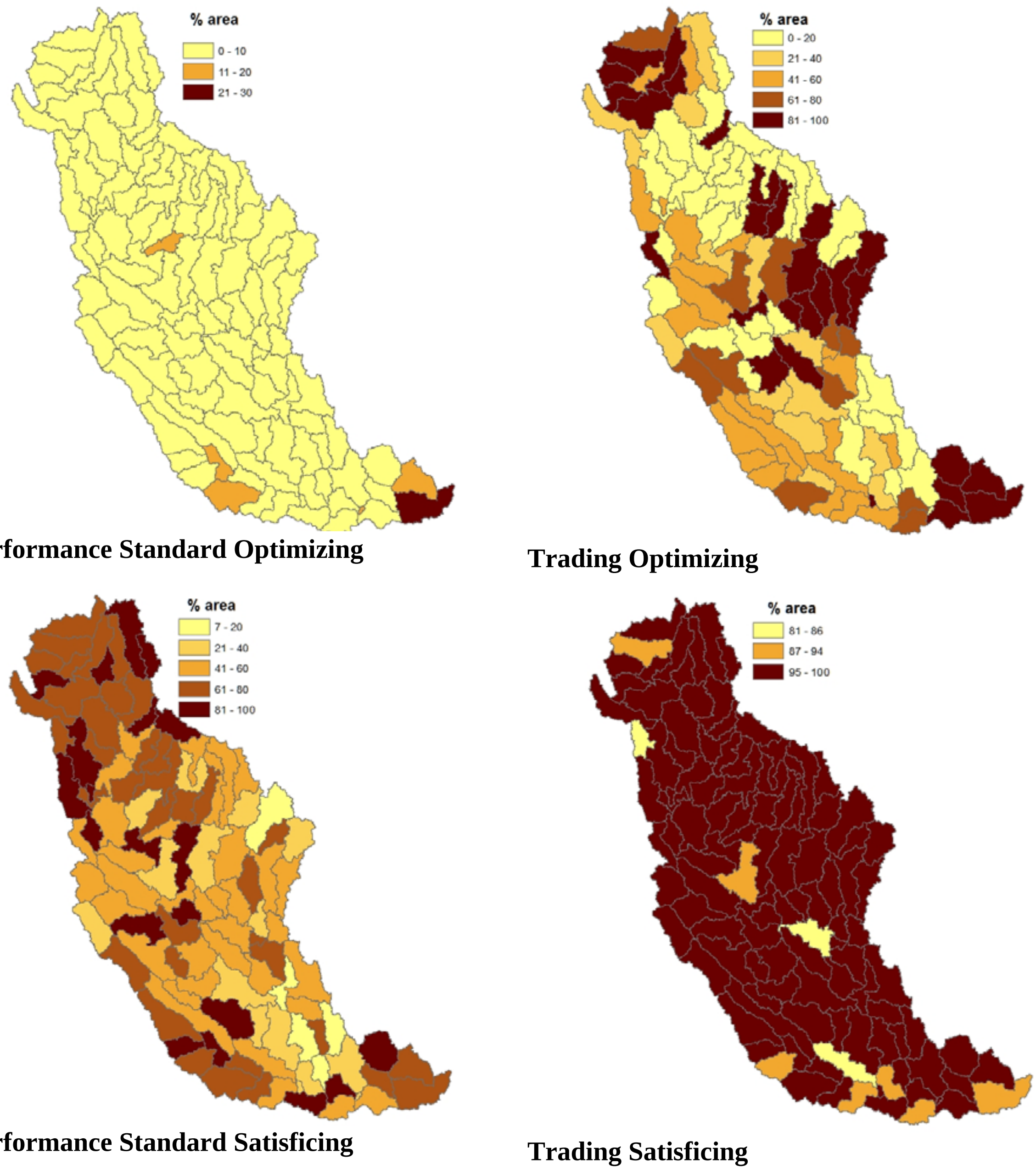

Figure 4-8. Spatial representation of changes in the CAC distribution (% of subbasin area), 30% N abatement goal

Figure 4-8 depicts the spatial representation of the changes in the distribution of the abatement actions under CAC for 30% N abatement. The lighter colored areas show that in that subbasin less area switches to a different abatement actions. The figure depicts the changes across PS and trading, for both satisficing and optimizing.

There are no hotspots under the PS and trading outcomes in RRW. Moreover, the abatement effort is distributed in a similar manner to BRW. Figure 4-9 depicts the abatement effort across the fields for RRW, 30% N abatement goal. The shape of the distribution is comparable to the corresponding BRW distribution, shown in Figure 4-6. Similar figures are provided in Appendix B for the N satisficing and P optimizing and satisficing outcomes.

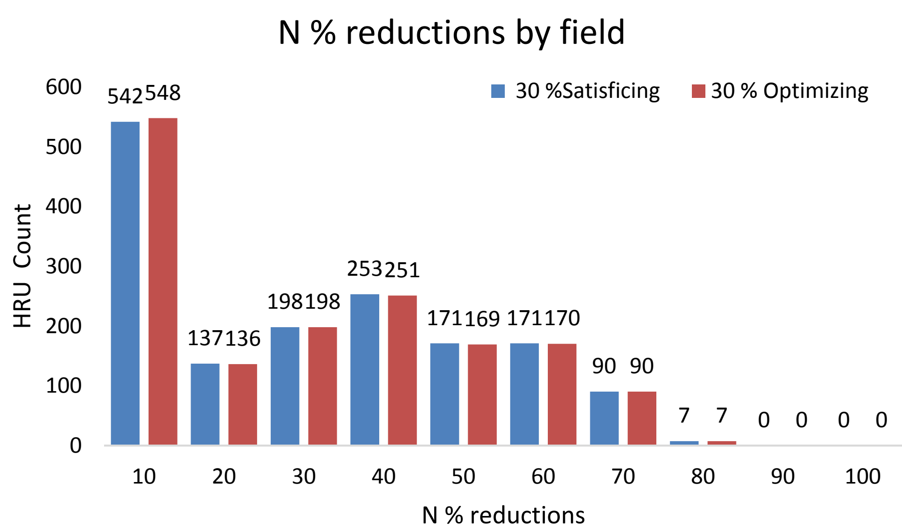

Figure 4-9. Raccoon River Watershed: Distribution of abatement effort corresponding to trading outcomes, N 30% abatement goal

Discussions

I evaluated each of the three policy approaches to regulate nonpoint-source water pollution emissions, where emissions are defined in terms of either nitrogen or phosphorus for two different watersheds located in Iowa. The empirical assessment of the proposed policies shows an overall good performance of the trading programs based on field-specific points measuring the marginal impacts of abatement actions on the total abatement level. Trading and

PS optimizing outcomes are good approximations of the corresponding first-best outcomes, while the CAC and PS satisficing outcomes are generally cost ineffective.

The policies focusing on N abatement are more costly than the policies focusing on P abatement, and this result holds for both watersheds. At the same time, the equilibrium point prices for N trading programs are lower than the corresponding prices for P trading programs, given similar level of abatement targets. Hence, the marginal cost of having an additional unit of P abatement is higher than the marginal cost for having an additional unit of N abatement. For example the equilibrium price for 30% N optimizing trading is $5.13 while the price for 30% P optimizing is $38.13. The corresponding total costs are $2.27 million for N trading and $1.03 million for P trading.

Given the same abatement targets, the spatial distribution of the abatement actions differs across the two watersheds. For example, for 30% N abatement goals in BRW, the distribution of trading-optimizing abatement actions is focused mainly around "no action" and "no till," while in RRW the distribution is more evenly spread across the entire set of abatement options. The overall watershed activity is somewhat similar across the watersheds too, and the same patterns are observed in both of them: trading results in more activity relative to the PS approach, and the activity is more intense under the satisficing approach than under optimizing.

The two watersheds, while located in the same state, differ considerably in size, with RRW being three times larger than BRW. Given that the estimated point coefficients are similar across the two watersheds, and that the per acre baseline N (P) emissions are similar across the two watersheds (BRW 9.1 kg N per acre, RRW 10.5 kg N per acre, BRW 0.42 kg P per acre, RRW 0.36 kg P per acre), the RRW total costs would be expected to be approximately three times higher than total costs for BRW, given the same set abatement targets. However, the per acre abatement costs are larger for RRW than with similar abatement actions for BRW; and consequently, the total costs for RRW are much higher. For example, achieving 30% N abatement in BRW costs about $2.27 million, while reaching a similar level of N abatement in RRW costs $31.41 million. The area and the cost differences across the two watersheds are also reflected in the equilibrium prices, with RRW prices being, again, larger than those of BRW.

4.2.5. The Assessment of Trading Outcomes with Less Refined Point Coefficients

The assessment of the proposed policies shows that the point coefficients estimated by the multistep procedure offer a good approximation of the water quality function at a very fine scale (i.e., field scale). Using such a detailed system of point coefficients eventually represents a potential burden for the implementation of a trading program at a watershed scale. A natural question that arises is how much different would the trading outcomes be if subbasin-specific, or even watershed-specific point coefficients are used?

A priori, given the same level of abatement, total trading costs are expected to increase when less refined sets of points are used, since there is less heterogeneity in the per abatement costs. Having less heterogeneity at the watershed scale implies that there is less gains form trade, and hence less cost savings relative to the command-and-control outcomes being realized. The total level of realized abatement can go up or down depending on the direction of the approximations. A trading program based on subbasin-specific points implies the estimation of a number of set of points equal to the number of subbasins, while a trading program based on watershed-specific points implies the estimation of a single set of points.

Boone River Watershed

Table 4-15 summarizes and compares the trading outcomes under the two approaches (optimizing and satisficing) when subbasin- or watershed-specific point coefficients are used instead of field-level specific points for BRW. Using subbasin-specific points implies the use of 30 sets of points, while the watershed-specific points implies the use of only one set of points.

Table 4-15. Boone River Watershed: Trading outcomes based on watershed and subbasin specific points

		Nitrogen				Phosphorus			
		optimizing		satisficing		optimizing		satisficing	
	Points Level	N red.	Cost (mil $)	N red.	Cost (mil $)	P red.	Cost (mil $)	P red.	Cost (mil $)
20 %	Field	19.81	0.85	22.21	1.10	21.92	0.49	20.50	0.42
	Subbasin	*16.88*	*1.14*	*17.57*	*1.22*	*17.00*	*0.71*	*18.89*	*0.83*
	Watershed	*16.77*	*1.42*	*18.05*	*1.59*	*18.21*	*1.21*	*13.94*	*0.93*
30 %	Field	29.47	2.27	31.98	2.99	29.98	1.03	29.35	0.97
	Subbasin	*28.53*	*2.41*	28.84	2.45	*27.15*	*1.33*	*27.51*	*1.39*
	Watershed	*28.60*	*2.40*	29.04	2.44	*26.66*	*1.81*	*26.86*	*1.82*
40 %	Field	39.35	6.06	41.40	7.16	40.30	4.60	37.07	2.02
	Subbasin	*36.22*	*6.74*	36.34	6.84	38.40	4.12	*38.52*	*4.43*
	Watershed	*37.60*	*8.10*	*38.19*	*8.48*	*39.65*	*6.12*	*39.06*	*4.85*

A clear pattern of non-attainment is observed under both approaches and for all abatement levels. Moreover, the trading outcomes based on subbasin- and watershed-specific points are outperformed by the trading outcomes based on field-specific points (more reductions, lower costs; the outperformed outcomes are italicized in Table 4-15). With the exception of 40% satisficing P, less reductions are realized under both subbasin and watershed points scale. Overall, the outcomes based on watershed-specific points perform slightly better than the subbasin points in terms of deviations from the abatement target, but at the same time, they are more costly.

A solution to increasing the realized abatement is to inflate the total points value corresponding to a given abatement target and the points requirement for each field by a coefficient based on the cost information available to the regulator. While the point coefficients do not change, the field requirements are increased by a factor k, thus constraining the farmer to choose an abatement action that accrues a higher number of points. The factor k can be determined through trial and error and by simulating the trading outcomes under different values. Using an inflation coefficient is similar to tightening the overall cap for the case where the abatement function is concave and a linear approximation of the abatement function results in the under-achievement of the abatement function.

The emergence of hotspots is not a significant problem for the field-specific points, the number of fields with worse water quality outcomes is relatively small (zero for nitrogen abatement and less than 20 for phosphorus abatement). More hotspots emerge under both subbasin- and watershed-specific points. For example, for 30% abatement levels, there are approximately 60 hotspots for trading nitrogen and 100 for trading phosphorus. Table B-6 summarizes the total number of fields where hotspots emerge.

The impact on the overall change in the CAC abatement actions' distribution, measured as the percentage of the watershed area that switches to an abatement action other than the one assigned by the command-and-control policy, is summarized in Figure 4-10 for 30% abatement goal, and Figures B-11 and B-12 in Appendix B for the 20% and 40% targets.

Figure 4-10 shows that for a 30% abatement goal, with the exception of N satisficing, less change in the CAC distribution is realized under the watershed- and subbasin-specific points than under the field-specific points. For example, for N optimizing, the change in the CAC distribution under the watershed-specific points is about one third less than the change under the field-specific points (29.9% versus 47.1%). A similar magnitude is realized for P optimizing (59.5% versus 38.1%). Additionally, Figure 4-11 compares how the change in the CAC distributions varies under the three types of points under 30 % N abatement trading program (optimizing). The same figure also depicts the subbasins where the most change takes place (the darker colored-areas). The corresponding description for 30 % P abatement can be found in Appendix B (Figure B-13)

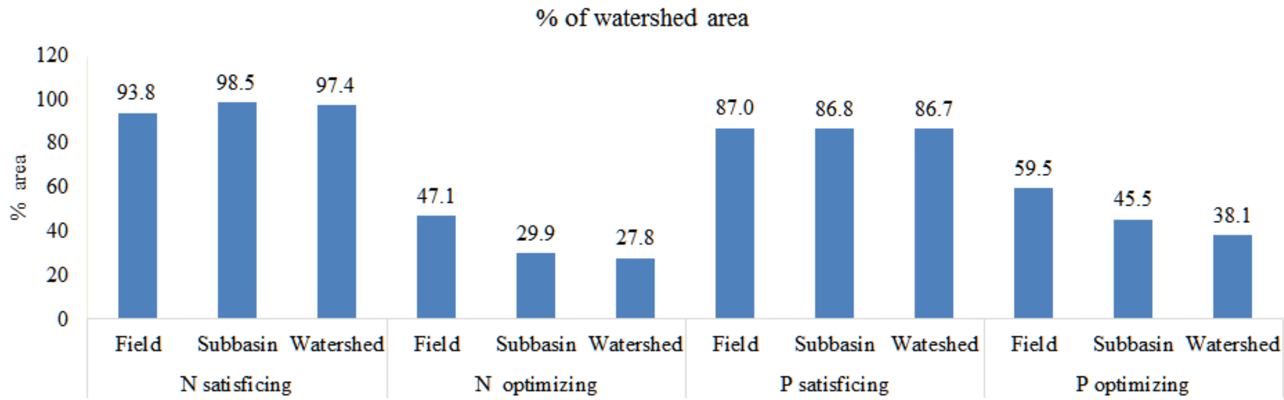

Figure 4-10. Boone River Watershed: The overall change in the distribution of abatement actions under CAC, 30% abatement goal (the height of the bar represents the percent of the total area that changes the abatement actions from CAC).

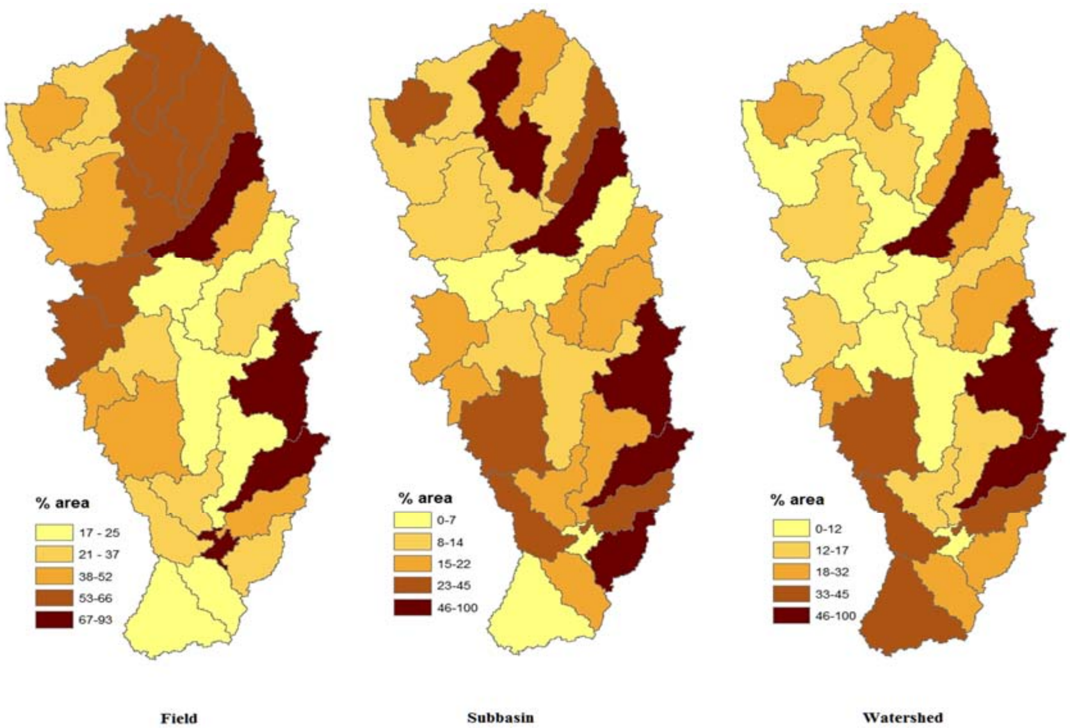

Figure 4-11. Boone River Watershed: The spatial change in the CAC distribution under the three types of points, 30 % N abatement, optimizing.

Raccoon River Watershed

Similar patterns are observed for RRW. The subbasin-specific point trading implies the estimation of 112 sets of point coefficients. The outcomes of trading simulations are summarized in Table 4-16. Generally, lower abatement outcomes are realized under both watershed- and subbasin-specific points for both N and P optimizing. Mixed results are obtained for the satisficing approaches, especially for a high level of abatement (e.g., 40%), when more abatement is realized under the watershed- and subbasin-specific points. Most of the trading outcomes are outperformed by the corresponding outcomes under field specific points. For example, for 30 % N optimizing goal, the realized abatement under the field specific points is 29.2% N with a total costs of $31.4 million. Under the subbasin specific points, the realized abatement is 28.9% N but costs $34.9 million, while under watershed specific points the realized abatement is 29.2 % N andcosts increase to $40.5 million.

Table 4-16. Raccoon River Watershed:Trading outcomes based on watershed and subbasin specific points

		Nitrogen				Phosphorus			
		Optimizing		Satisficing		Optimizing		Satisficing	
	Points Level	N red.	Cost (mil $)	N red.	Cost (mil $)	P red.	Cost (mil $)	P red.	Cost (mil $)
20%	Field	*19.97*	*14.36*	*20.47*	*15.22*	*21.62*	*4.30*	*18.65*	*3.17*
	Subbasin	*18.88*	*15.58*	18.15	14.22	20.15	3.95	*17.42*	*2.93*
	Watershed	*18.42*	*20.22*	*19.79*	*22.93*	*16.18*	*4.71*	*19.44*	*8.95*
30%	Field	*29.22*	*31.41*	*29.42*	*31.81*	*30.90*	*9.97*	*29.46*	*8.80*
	Subbasin	*28.94*	*34.90*	*28.28*	*33.29*	*29.07*	*10.05*	*27.32*	*8.60*
	Watershed	*29.19*	*40.46*	*29.36*	*40.64*	21.84	12.16	*30.12*	*14.83*
40%	Field	*38.94*	*56.39*	*39.18*	*57.54*	*40.07*	*20.15*	*37.57*	*16.76*
	Subbasin	39.08	61.89	*38.99*	*61.42*	37.73	19.30	37.95	19.60
	Watershed	*38.23*	*66.24*	39.44	73.19	*38.08*	*26.98*	41.08	36.82

An additional number of hotspots emerge as a result of a trading based on less specific points; however, the number is not significant. The impact on the overall change in the CAC abatement actions' distribution, measured as the percentage of the watershed area that switches to an abatement action other than the one assigned by the command-and-control policy, is summarized in Figure 4-12 for 30% abatement goal, and Figures B-14 and B-15 in Appendix B for the 20% and 40% targets. The direction and the magnitude of the change across the three types of outcomes does not follow a particular trend. However, as in the case of BRW, the change has the largest magnitude under N optimizing trading (71.5 % Field, 63% Subbasin and 41.7% Watershed). Notice that the magnitude of these changes is higher than the corresponding outcomes for BRW (47.1% Field, 29.9 %, and Subbasin, 27.8% Watershed). The magnitude in the change under the P optimizing is slightly lower for the field level point outcomes. This is the opposite of the corresponding P optimizing outcomes

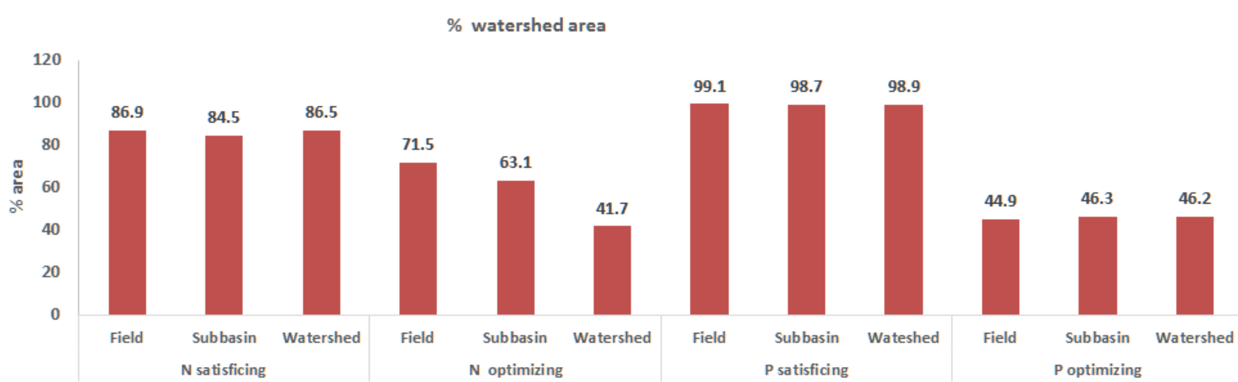

Figure 4-12. Raccoon River Watershed: The overall change in the distribution of abatement actions under CAC, 30% abatement goal (the height of the bar represents the percent of the total area that changes the abatement actions from CAC).

Figure 4-13 compares how the change in the CAC distributions varies under the three types of points under 30 % N abatement trading program (optimizing). The same figure also depicts the subbasins where the most change takes place (the darker colored-areas).

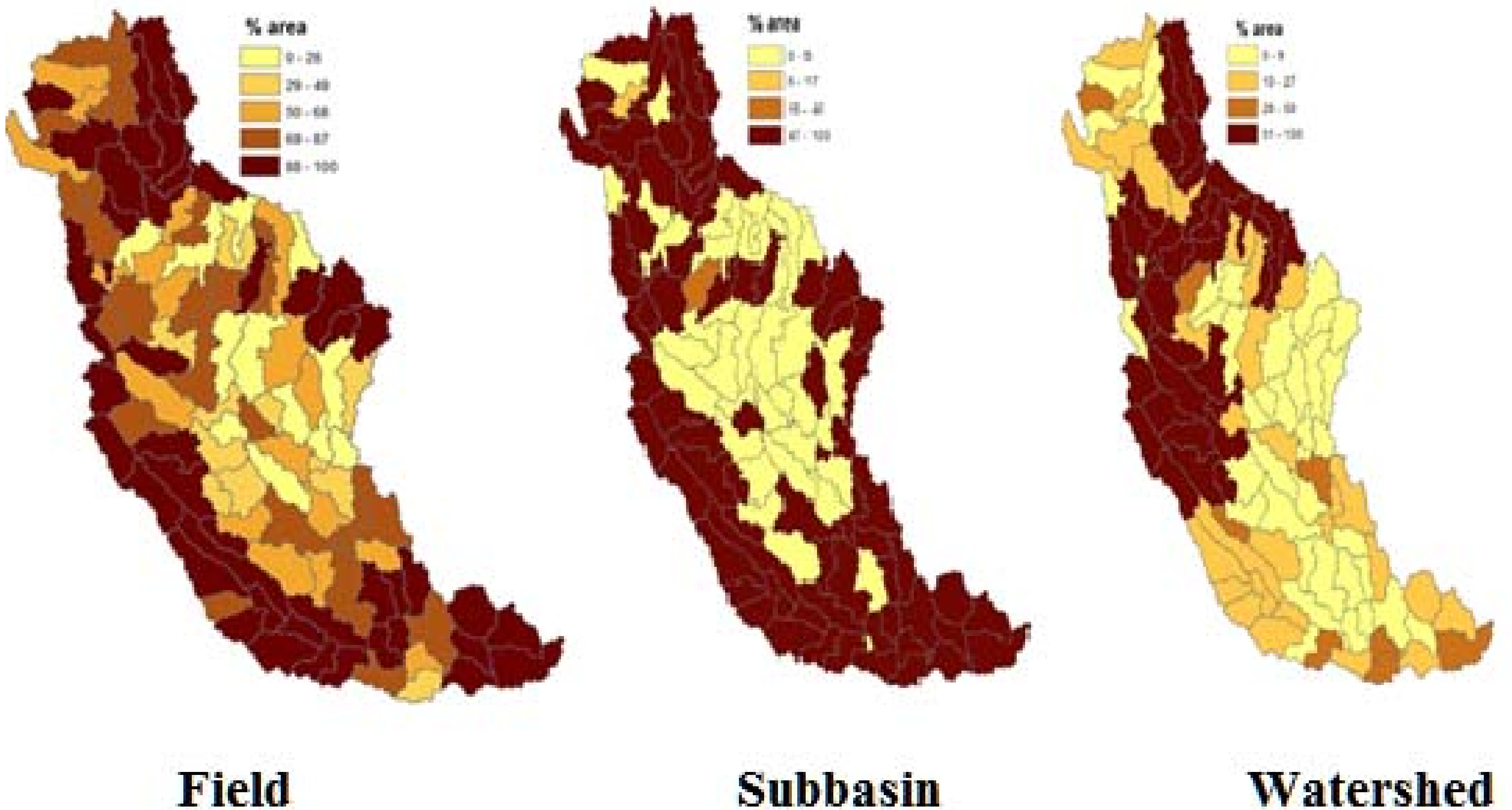

Figure 4-13. BRW: The spatial change in the CAC distribution under the three types of points, 30 % N abatement, optimizing.

4.3. Ex Post Assessment of Policies with Respect to Abatement Risk

The abatement realizations for either pollutant under different policies is subject to the stochastic influence of the weather. Given the stochastic weather elements, some of the proposed policies can be riskier than others, since the cost minimization is solved to achieve an expected ambient pollution under risk neutral behavior.

Using a historical climate data for a longer period of time than was used to generate the policy outcomes, I am able to provide an ex post empirical assessment of variability in attaining the five-year mean nitrogen (phosphorus) abatement targets. Given that CAC and PS satisficing policies select more expensive, and hence a more effective abatement action, a reasonable assumption is that abatement levels realized under these policies have a lower variance than the optimizing policies. Hence, to verify if the ambient outcomes depend on the historically observed weather variability, and if there are any policies that might have a lower risk, I simulate the outcomes for a longer time period, based on water quality and weather data availability for the watershed. Specifically, for each watershed, I run the SWAT model using the optimal placement of the abatement actions obtained as solutions for CAC, PS and trading for 22 years. Next, I disregard the first two years and I compute the new abatement values as the five-year moving average.

The period used for Boone River Watershed spans the 22 years from 1988 to 2009 (the first two years are disregarded). By computing the five-year mean annual N (P) values for each policy, I obtain 16 additional mean annual values for each policy.

Table 4-17 offers an example of a five-year moving average distribution for BRW, 30% N abatement optimizing policies. The below table summarizes per period annual average for N loadings as well as the corresponding level of abatement. Similar distributions for all abatement levels and policy approaches are presented at the end of Appendix B. For each distribution, I compute the mean and the standard deviations. Next, to check whether or not some policies are more risky than others, I apply various F-tests for equal variance.

Table 4-17. Boone River Watershed: Per-period annual average distribution of N loadings, 30% N optimizing policies

	Annual average N loadings (kg)				N abatement (%)		
Year	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990 - 1994	6,127,114	4,351,228	4,255,128	3,995,086	28.98	30.55	34.8
1991 - 1995	6,169,094	4,373,346	4,271,720	4,011,852	29.11	30.76	34.97
1992 - 1996	4,897,034	3,499,172	3,330,636	3,183,968	28.55	31.99	34.98
1993 - 1997	4,751,094	3,375,132	3,210,366	3,070,394	28.96	32.43	35.38
1994 - 1998	3,990,314	2,835,610	2,790,908	2,614,768	28.94	30.06	34.47
1995 - 1999	4,592,024	3,216,136	3,326,134	3,020,930	29.96	27.57	34.21
1996 - 2000	4,207,464	2,924,096	3,011,556	2,735,186	30.5	28.42	34.99
1997 - 2001	4,795,664	3,316,182	3,380,982	3,085,434	30.85	29.5	35.66
1998 - 2002	4,695,550	3,238,834	3,300,300	3,004,002	31.02	29.71	36.02
1999 - 2003	4,129,190	2,848,230	2,921,210	2,650,526	31.02	29.25	35.81
2000 - 2004	4,336,090	3,045,304	3,019,632	2,801,086	29.77	30.36	35.4
2001 - 2005	4,702,210	3,293,698	3,258,116	3,030,406	29.95	30.71	35.55
2002 - 2006	3,907,158	2,753,932	2,739,746	2,543,292	29.52	29.88	34.91
2003 - 2007	4,893,212	3,459,562	3,416,934	3,199,554	29.3	30.17	34.61
2004 - 2008	6,163,132	4,388,648	4,281,888	4,038,490	28.79	30.52	34.47
2005 - 2009	5,448,292	3,878,852	3,803,942	3,586,424	28.81	30.18	34.17

Table 4-18 summarizes the mean values and the standard deviations for the BRW nitrogen-based policies under both optimizing and satisficing approach. The standard deviations are relatively small—representing on average 15% of the mean values for nitrogen. These values are consistent across abatement targets and policies under both the satisficing and optimizing approaches. Next, testing for the difference in variances within satisficing (optimizing) policies and keeping the same abatement targets shows that, given the historical data, policies are equally risky in terms of abatement (i.e., the abatement outcomes under CAC, PS, and trading have the same variance). Testing for the difference in variances *across* satisficing and optimizing policies further shows that there is no difference in terms of risk (i.e., given a policy approach, let say trading, the variance of the realized abatement is the same under both satisficing and optimizing).[34]

Table 4-18. Boone River Watershed: The five-year moving average 1990-2009 N loadings distribution

Abatement goal	**Satisficing Policies**			**Optimizing Policies**		
20% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (mil kg,N)	3.87	3.87	3.61	3.87	3.76	3.82
Std.dev. (mil kg,N)	0.61	0.59	0.55	0.61	0.57	0.57
Average N reduction (%N)	20.48	20.33	25.78	20.48	22.76	21.51
30% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (mil kg,N)	3.42	3.39	3.16	3.34	3.28	3.33
Std.dev. (mil kg,N)	0.55	0.51	0.50	0.49	0.49	0.49
Average N reduction (%N)	29.57	30.19	35	31.37	32.52	31.52
40% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (mil kg,N)	2.99	2.98	2.82	3.82	2.90	2.91
Std.dev. (mil kg,N)	0.49	0.47	0.45	0.58	0.47	0.46
Average N reduction (%N)	38.56	38.62	42.05	21.35	40.26	40.06

Qualitatively similar results are obtained from phosphorus-based policies: there is no difference in risk when testing for differences in variance, either within or across the satisficing and optimizing policies. Table B-16 presents the summary of the phosphorus five year moving average distribution for BRW.

[34] The variances are equal in statistical sense.

The period used for the Raccoon River Watershed spans 21 years from 1984 to 2004. By computing the five-year mean annual N (P) values for each policy, I obtain 15 additional mean annual values for each policy. Table 4-19 summarizes the mean and the standard deviation for nitrogen policies. The results are qualitatively similar to the ones obtained from BRW, in the sense no policy can be assessed as being riskier than other policies. The standard deviations are relatively small, being less than 15% of the annual mean loadings. However, the percentage for nitrogen is, on average, 12, while for phosphorus the average is 15. The summary of the P distributions is presented in Table B-17.

Table 4-19. Raccoon River Watershed. The five-year moving average 1990-2009 N loadings distribution

	Satisficing Policies			**Optimizing Policies**		
20% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (kg,N)	16.60	16.49	16.41	16.34	16.31	16.50
Std.dev. (kg,N)	1.82	1.83	1.86	1.85	1.84	1.87
Average N reduction (%N)	16.83	17.38	17.78	18.13	18.28	17.33
30% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (kg,N)	14.08	14.04	14.44	14.13	14.18	14.48
Std.dev. (kg,N)	1.66	1.66	1.72	1.90	1.87	1.73
Average N reduction (%N)	29.47	29.67	27.67	29.20	28.96	27.46
40% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (kg,N)	12.31	12.29	12.28	12.10	12.13	12.33
Std.dev. (kg,N)	1.32	1.32	1.55	1.68	1.66	1.56
Average N reduction (%N)	38.31	38.45	38.46	39.36	39.24	38.22

Table 4-20 summarizes the five years moving average distributions for the baseline loadings for nitrogen and phosphorus for BRW and RRW. A direct test for equal variances is not meaningful since the baseline values for N and P have different magnitude (i.e, the P loadings are on average 4%–5% of the N total loadings). After scaling down the N loadings testing for equal variances shows that the N and P baselines have the same variance. The null hypothesis of equal variances is not rejected. Figure 4-14 and 4-15 summarize the five-year moving average for baseline P loadings and rescaled N loadings[35]. Again, the figures suggest that there is no difference in the variability across pollutants for either of the two watersheds.

Table 4-20. Five-year moving average baseline loadings distribution

	BRW (1990-2009)		RRW (1986-2004)	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
Mean (kg,P)	4,862,790	240,091	19,960,449	818,846
St.dev. (kg,P)	748,522	33,977	2,276,340	117,106
St.dev/Mean	15.39	14.15	11.40	14.30

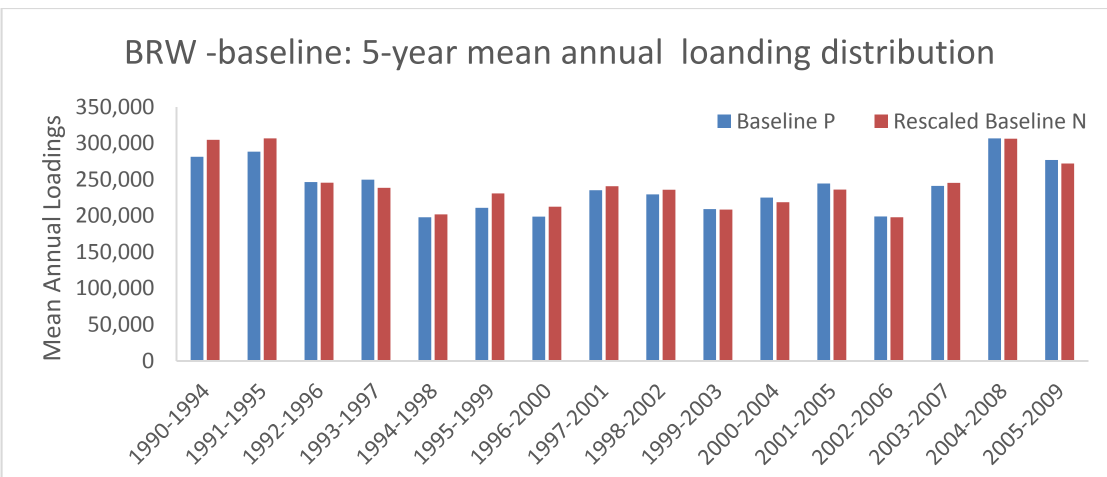

Figure 4-14. Boone River Watershed -baseline: five-year mean annual loadings distribution

[35] Rescaling of N loadings is made with respect to the P loadings

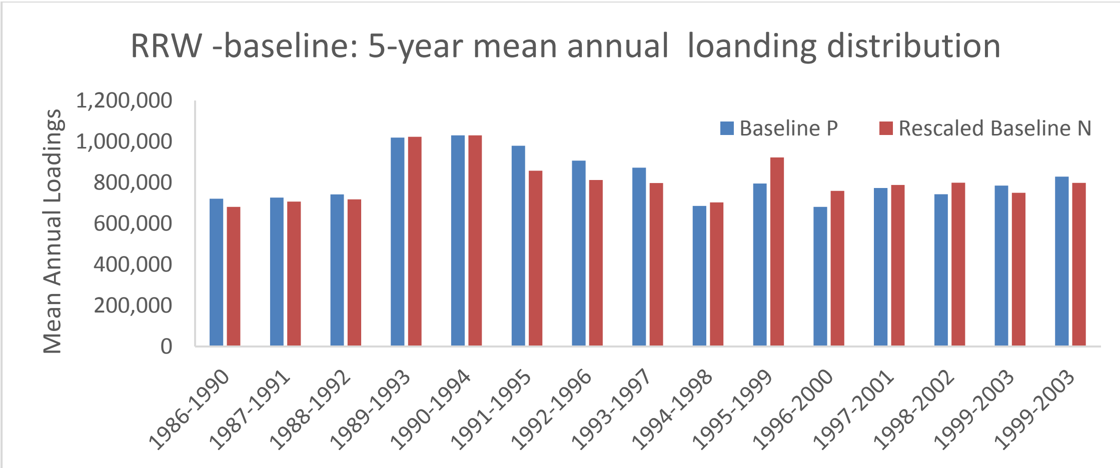

Figure 4-15. Raccoon River Watershed -baseline: five-year mean annual loadings distribution (kg/year)

4.4. An Emission Based Trading Approach

In Chapter 3, along with estimating the point coefficients for each pollutant and watershed, I estimated a set of delivery coefficients, where a delivery coefficient measures how much of the abatement leaving a field contributes to the total ambient level. In the context of my empirical application, the delivery coefficients are subbasin-specific. These delivery coefficients can be used in designing a trading program based on the edge-of-field reduced emissions, where trading ratios are defined as the ratio of the delivery coefficients.

In this section, I compare the efficiency of the point-based trading with the efficiency of a trading program based on the reduced edge-of-field emissions. Under a point-based trading, a system of points is used as an approximation to the edge-of-field abated emissions, thus bringing a potential source of inefficiency. The goal of this comparison is to empirically measure the extent of these inefficiencies.

I assume that the regulator is able to find a watershed configuration that achieves a given abatement level (let $\bar{\boldsymbol{r}}$ be the vector of corresponding emissions). Next, the field-level constraints are set based on the simulated edge-of-field reduced emissions rather than on the estimated points:

$$\bar{A}(\bar{\mathbf{r}}) \approx \sum_i^N d_i\, \bar{r}_i(x_i) \qquad (22)$$

where $\bar{A}$ is the desired level of abatement, d_i is the delivery coefficient for field i, and $\bar{r}_i(x_i)$ is the simulated edge-of-field reduced emissions for field i under the solution identified by the regulator. $\bar{r}_i(x_i)$ also represents the field level constraints.[36] Delivery coefficients are obtained based on a linear approximation of the water-quality production function $\sum_i^N d_i\, \bar{r}_i(x_i) <> \bar{A}(\bar{\mathbf{r}})$, implying that there is no exact mapping between the field-specific constraints distributed as permits and the abatement cap.

Under a trading program based on edge-of-field emissions, a farmer minimizes the abatement costs by choosing the abatement actions and the level of abated emissions e_i to be traded as long as the edge-of-field constraint is satisfied

$$\min_{x_i, e_i} C_i(x_i, \gamma_i, \theta_i) + d_i e_i \qquad s.t \quad r_i(x_i) + e_i \geq \bar{r}_i \qquad (23)$$

where $r_i(x_i)$ are the edge-of-field reductions associated with the chosen abatement actions. The market clearing conditions is given by $\sum_i^N d_i e_i = 0$

Tables 4-21 and 4-22 summarize the results for the simulated outcomes of an emission-based trading program for the two watersheds for both nitrogen and phosphorus considering the optimizing approach (columns 5, and 6 for N, and 8, and 9 for P). The outcomes are compared with the corresponding point based trading outcomes (columns 2, 3, 7, and 8). In the, case of BRW (Table 4-21), slightly more reductions are obtained under the emission based trading (see

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[36] In the empirical applications all fields in a subbasin have the same delivery coefficient,

column 2 vs. 4). However, the additional reduction are reflected in slightly higher prices (see column 3 vs. 5).

Table 4-21. Boone River Watershed: Simulated abated emission based trading program performance under varying abatement target vs points based trading performance

Target	Trading, points		Trading, emissions		Trading, points		Trading, emissions	
	Nitrogen				Phosphorus			
	N red, %	$, million	N red. %	$, million	P red, %.	$, million	P red, %	$, million
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
20%	19.81	0.85	20.41	0.89	19.97	14.36	20.48	0.42
30%	29.47	2.27	29.77	2.31	29.22	31.41	30.5	1.07
40%	39.35	6.06	39.64	6.13	38.94	56.39	37.34	2.84

The RRW nitrogen emissions based trading program has a similar performance to BRW, however, the phosphorus emission based trading program has a worse performance (see column 8, Table 4-22.), with the realized abatement being much lower than the abatement targets (i.e. the realized abatement for 40% P abatement is 26.54%)..

Table 4-22. Raccoon River Watershed: Simulated abated emission based trading program performance under varying abatement target vs points based trading performance

Target	Trading, points		Trading, emissions		Trading, points		Trading, emissions	
	Nitrogen				Phosphorus			
	N red, %	$, million	N red. %	$, million	P red, %.	$, million	P red, %	$, million
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
20%	19.97	14.36	19.16	14.66	21.62	4.30	16.17	3.48
30%	29.22	31.41	30.23	35.60	30.90	9.97	21.22	6.35
40%	38.94	56.39	38.74	59.35	40.07	20.15	26.54	10.44

The above outcomes are simulated using the delivery coefficients corresponding to the field specific point coefficients. Furthermore, the delivery coefficients are constrained to be less than one. Another set of outcomes are simulated by using the unconstrained set of delivery coefficients. Unsurprisingly, the results under the emission trading are very similar to the trading outcomes observed under the point-based trading.

The above results show that on average the outcomes of a point-based trading program, where points are used as a proxy for the reduced emissions, and the outcomes of an emission-based trading program, where delivery coefficients are used as trading ratios, are similar. These results are expected since the delivery coefficients are also estimated based on a linear approximation of the true abatement function. By using a linear approximation, there is no exact matching between the total number of permits and the abatement target. This is reflected in the ex post outcomes of N and P reductions that vary above or below the initial abatement targets. However, an emission-trading program assumes that the regulator and farmers agree that the emissions simulated by SWAT represent the true emissions which often is not the case. Alternatively, the points found by measuring the estimated impact on an abatement action on the total level of abatement are good substitute for measuring the corresponding reduced emissions.

4.5. Simulated Policy Programs under Cost Heterogeneity and Significant Cost Asymmetry

In this section, I explore how the proposed policies behave in the presence of significant cost heterogeneity and under simulated information asymmetry. In the previous sections, I assumed that the regulator and the farmers have the same cost information—implying that the abatement costs vary by field characteristics but do not vary by farmers' management abilities. In this section, I consider a more realistic situation where the regulator has some information about the costs of abatement actions (the portion of the cost information that depends on the physical field characteristics, $c_j(\gamma_i)$), but I allow the abatement costs vary to across farmers' abilities. This cost heterogeneity scenario is simulated by generating random draws of

$u \sim U[-0.8\ \ 0.8]$ and multiplying, the part of the costs that is known by the regulator, $c_j(\gamma_i)$, by $(1+u)$. Additionally, I assume that for a given farm the cost of each abatement action receives the same shock u.[37]

Thus, under cost asymmetry and heterogeneity, the regulator uses the costs functions given by equation (18) and uses this costs structure to: identify the least cost allocation to achieve a given level of abatement and to implement the incentive-based policies. However, the farmers use the following cost information when choosing the abatement action under either the PS or trading program.

$$C_i(x_i, \gamma_i, \theta_i) = \sum_j^J c_j(\gamma_i) * (1 + u_i) * x_{ij} * s_j \quad \forall\, i = 1, \dots N \tag{24}$$

where the notation is similar to Equation 18. Additionally, the random shock u_i simulates the cost asymmetry. In order to create a distribution of outcomes under cost heterogeneity, I solve for the PS (equation (19)) and trading (equation (20)) solutions for a set of 1,000 uniform random draws.

Table 4-23 and 4-24 summarize the mean and standard deviation for the simulation results under cost heterogeneity 30% N and 30 % P abatement for BRW and RRW, respectively. As expected, both CAC optimizing and satisficing have the highest variation in costs, but no variation in the realized abatement (columns 1 and 4). The lowest realization of CAC satisficing is higher than the highest cost realization of CAC-optimizing. Additionally, since the CAC satisficing selects more expensive abatement actions, the cost variability of the corresponding outcomes is much higher than the costs variability of the CAC-optimizing outcomes. For

[37] The values of the random shock u $u \sim U[-0.8\ \ 0.8]$ are chosen at random. The goal is to create some heterogeneity across the field abatement costs.

example, consider the standard deviations under the CAC program for a 30% N (P) abatement goal for both watersheds. For these cases, the standard deviation for CAC satisficing is up to 10 times higher than the CAC optimizing cost variation in the case of BRW. However, the variation in the case of RRW is lower, only up to a factor of 2. For both watersheds, the cost variation under phosphorus CAC policies is lower than the variation for nitrogen CAC policies.

Next, I move to the outcomes obtained under the PS program. As expected, the total costs are lower than under the CAC policies (column 1 vs 3, and 4 vs 6). Given that the farmers have more flexibility in choosing the abatement actions, different cost draws result in some variation in the realized abatement levels. Two interesting outcomes emerge for the 30% P abatement for BRW (Table 4-24 column 2) and the 30% N and 30% P abatement for RRW (Table 4-24 column 2), when the PS optimizing does not result in any variation in the realized abatement.

Table 4-23. Boone River Watershed: Simulated outcomes under cost heterogeneity and asymmetric cost information, 30% abatement goal

	CAC, optimizing		PS, optimizing	CAC, satisficing		PS, satisficing		Trading, optimizing			Trading, satisficing	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
	BRW, Nitrogen											
	Cost, $mil	N red. %	Cost, $ mil	Cost, $ mil	N red. %	Cost, $ mil	N red. %	Cost, $ mil	Price, $	N red. %	Cost, $ mil	Price, $
Mean	3.23	30.3	3.09	19.80	35.5	18.58	29.1	1.95	4.1	31.7	2.52	5.1
StdDev.	0.06	0.1	0.06	0.65	0.1	0.64	0.1	0.05	0.1	0.1	0.06	0.2
	BRW, Phosphorus											
	Cost, $	P Red. %	Cost, $ mil	Cost, $ mil	P Red. %	Cost, $ mil	P Red. %	Cost, $ mil	Price, $	P Red. %	Cost, $ mil	Price, $
Mean	2.16	29.9	2.00	16.35	34.9	11.24	30	0.91	38.3	29.4	0.86	35.9
StdDev.	0.05	0	0.04	0.57	0	0.55	0.1	0.02	1.3	0.1	0.02	1.2

Table 4-24. Raccoon River Watershed: Simulated outcomes under cost heterogeneity and asymmetric cost information, 30% abatement goal

	CAC, optimizing		PS, optimizing	CAC, satisficing		PS, satisficing		Trading, optimizing			Trading, satisficing	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
	RRW, Nitrogen											
	Cost, $mil	N red. %	Cost, $ mil	Cost, $ mil	N red. %	Cost, $ mil	N red. %	Cost, $ mil	Price, $	N red. %	Cost, $ mil	Price, $
Mean	42.43	30.9	39.56	62.26	30.4	58.65	29.7	24.25	10.1	29.9	24.56	10.1
Std.Dev.	0.68	0	0.67	1.55	0	1.52	0.1	0.63	0.3	0.2	0.63	0.3
	RRW, Phosphorus											
	Cost, $	P Red. %	Cost, $ mil	Cost, $ mil	P Red. %	Cost, $ mil	P Red. %	Cost, $ mil	Price, $	P Red. %	Cost, $ mil	Price, $
Mean	15.81	30.8	15.79	36.28	31	58.64	30	8.41	113.1	29.4	7.39	101.3
Std.Dev.	0.36	0	0.36	0.97	0.2	1.52	0.1	0.25	3.8	0	0.23	3.5

The overall level of abatement is higher for the satisficing approaches, offering more evidence for the conjecture made earlier that the higher cost abatement actions tend to be more effective (column 4 vs 2). The PS optimizing again outperforms the PS satisficing approaches in terms of costs, being up to six times less expensive in the case of BRW and up to three times less expensive for RRW (column 5 vs 3).

Finally, the trading outcomes have a similar performance in terms of both cost efficiency and simulated cost variability, with the realized cost abatement outcomes being within similar ranges across the satisficing and optimizing approach (column 11 vs 8). On average, the abatement targets are met, with the exception of 40% phosphorus where the average realized abatement is 37%, for both watersheds (column 7 for optimizing and 8 for satisficing). The equilibrium point prices follow the same patterns as in the case of cost symmetry: the nitrogen prices are lower than the phosphorus prices (column 9 and 12)

Figure 4-16 and 4-17 describe the distributions of the 1,000 simulated outcomes for 30 % N abatement for BRW (Figure 4-16) and RRW (4-17). Each cluster of points represents the simulated outcomes for a particular policy. It can be shown that the CAC satisficing policies have the highest variation followed by the PS satisficing policies. The optimizing policies have lower cost variance but have a larger abatement variance. The summary of the mean and standard deviations for the simulation of outcomes under cost heterogeneity for 20% and 40% abatement goals as well as the figures depicting the distributions are provided in Appendix B, Tables B-18 to B-21 and Figures B-16 to B-20.

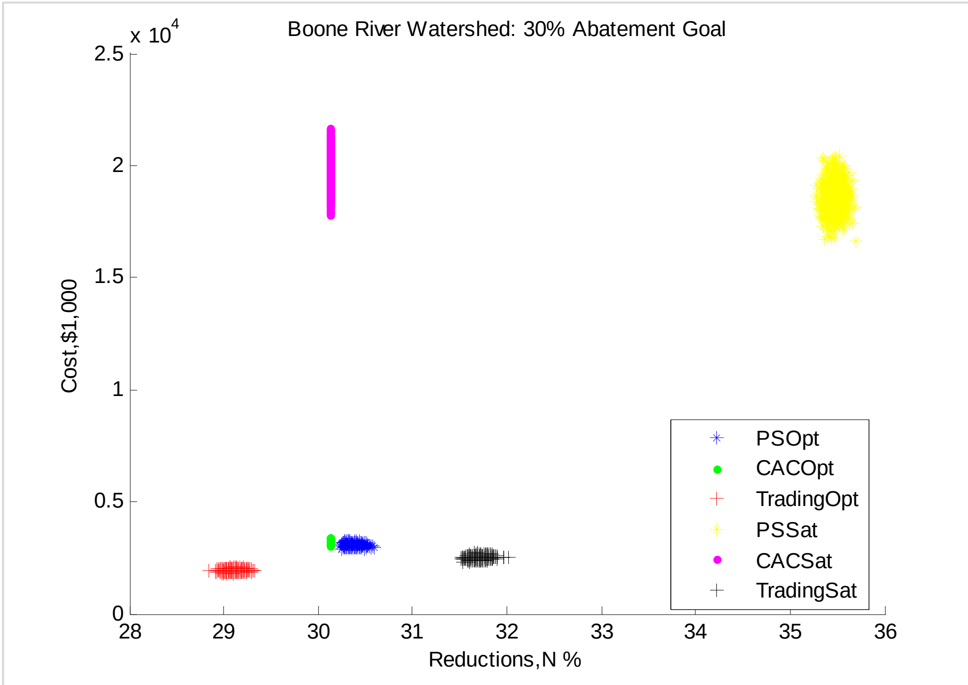

Figure 4-16. Boone River Watershed Simulated program outcomes under cost heterogeneity, 30% N abatement goal

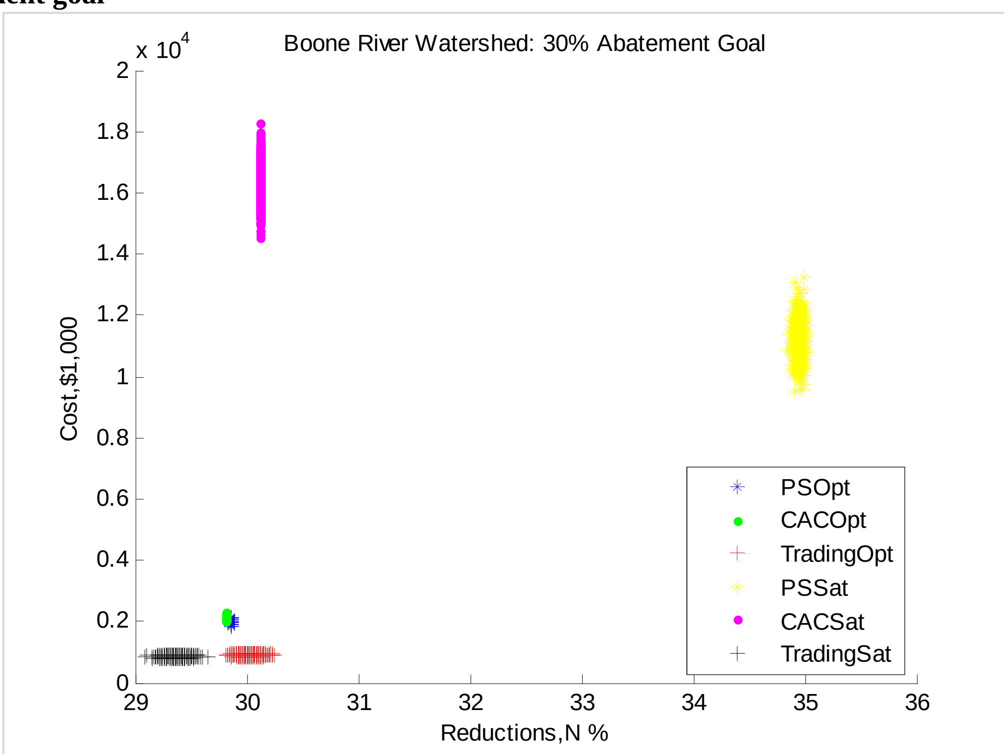

Figure 4-17. Raccoon River Watershed: Simulated program outcomes under cost heterogeneity, 30% N abatement goal

Discussions

The simulated outcomes under cost variability support earlier findings: the CAC policies are cost inefficient. CAC satisficing offers a solution when abatement costs are not known. However, the magnitude of cost savings under the CAC optimizing suggests that a regulator would be better off if he obtains estimates on the abatement costs and uses them in finding a least-cost way to achieve the desired level of abatement, then uses this solution for designing his policies.

Additional cost savings are realized under a performance standard program. The findings obtained for the 30% N and P optimizing target offer more evidence that if the regulator has good cost information, the PS program can be an attractive policy approach. Finally, under trading approach, the burden of the optimization is passed to the market and no optimization is required on the regulator’s side. However, using a linear approximation of the non-linear abatement function might result in the nonattainment or the over attainment of the abatement goals.

4.6. Multiple Pollutant Policy Approaches Assessment

In his seminal work, Montgomery (1972) demonstrated that a trading system for point sources, where the emissions leaving a source are measurable and the contribution of each source to the downstream concentrations are linear, can achieve the economically efficient allocations of abatement to achieve a given ambient water quality level. He developed his mathematical models for the case of markets for a single pollutant. In addition, he showed that if multiple noninteractive pollutants are to be regulated in separate markets, his mathematical models can be extended to include multiple pollutants by adding the corresponding constraints.

Since the problem of nonpoint-source water quality pollution is not easily measurable, and the ambient water quality effects are often thought to be nonlinear, water quality trading programs where agricultural nonpoint sources are required to hold permits to cover their contributions to pollution have generally been considered difficult or impossible to implement. Moreover, an efficient approach of water quality requires the consideration of multiple pollutants with potential synergistic and/or additive effects, such as nitrogen and phosphorus or sediments.

In the previous sections, I empirically evaluated a set of policy approaches for reducing nonpoint-source pollution, where a set of point coefficients was used to estimate efficiency of different abatement actions assuming that only one pollutant, either nitrogen or phosphorus, was targeted at a time. When estimating the set of point coefficients for each pollutant, I assumed that each pollutant has a separate abatement function, and hence, the point coefficients were obtained independently. However, the same set of abatement actions is used for the abatement of both pollutants; thus, a policy targeting one pollutant will reduce the other pollutant also. This implies that there are complementarities in the abatement functions. Given the nature of these complementarities, one question that arises in the context of multiple pollutants is whether it is more efficient to reduce the emissions for both pollutants (N and P in this case) or to focus the abatement efforts in reducing only one of them.

In this section, I simulate and evaluate the three policy approaches (a command-and-control, a performance standard and a point-based trading program) assuming that both nitrogen and phosphorus emissions are regulated. Next, to answer the questions of whether it is more efficient to reduce the emissions for both pollutants or to focus only on one of them, I compare the outcomes of the point-based trading when both markets for N and P function simultaneously with the outcomes of the point-based trading program with only one market available.

4.6.1. Setting the on-farm and watershed goals

Choosing the on-farm or watershed goals under the proposed policy approaches can be challenging under a nonlinear water-quality production function when multiple pollutants are targeted. Under a CAC program, the regulator can mandate the farm-level abatement actions. If he is interested in achieving the abatement target $\bar{A}^{e}$, for both N and P, then he needs to find the set of abatement actions (X_{CAC^e}) that satisfies $\{A(X_{CAC^e}) = \bar{A}^N \; and \; A(X_{CAC^e}) = \bar{A}^P \}$. One way to find the watershed configuration that simultaneously achieves the same level of reductions in both N and P is to randomly generate and simulate watershed configurations until the desired configuration is met. This approach does not require any cost information. Under a CAC program, the solution $\{X_{CAC^e}\}$ can be implemented directly. For the on-farm performance standard program and the point trading program, the field level, as well as the watershed requirements, is determined in a similar way as to the single pollutant case, but now two sets of constraints are set instead of one.

4.6.2. Cost-Efficiency Performance under the Same Cost Information

For generating the results presented below, I consider that the farmers and the regulator have the same information on the costs of the abatement actions. In terms of the model presented in the previous chapter, this implies that I solve for the PS and trading solutions the following cost function: $C_i(x_i, \gamma_i, \bar{\theta}) = \sum_j^J c_j(\gamma_i) * x_{ij} * s_i$.The results are obtained for three levels of desired water quality improvements: 20%, 30%, and 40% reductions in the mean expected annual loadings of nitrogen and phosphorus. The PS and trading cost minimization problems are similar to the ones described by equation (19) to (21). However, since a second constraint is added, the solution is likely to be different if the new constraints is binding.

Next, I present the empirical assessment of the three policies with the assumption of cost symmetry. The set of point coefficients have been introduced previously. The field-level requirements have been set according to random watershed configurations that achieve the same level of abatement for both pollutants. The outcomes for the performance standard and the trading program are obtained by using linear programming methods.

Table 4-25. Boone River Watershed: Multiple Pollutant Policy Approaches

Abatement Target/CAC			Performance Standard			Point-Based-Trading		
N	P	Total Cost	N	P	Total Cost	N	P	Total Cost
20%	20%	6.65	26.3	27.9	5.07	22	29.6	1.07
30%	30%	17.99	34.5	35.3	15.85	32.2	37.6	3.04
40%	40%	36.075	42.9	43.6	35.46	41.2	37.8	7.04

Table 4-25 summarizes the simulated outcomes under the three policies approach when both N and P are targeted for the Boone River Watershed. Under the CAC approach, while the abatement targets are met, the total costs are very high. Under a performance standard program, more reductions are obtained while the costs are lower than in the case of a command and control program. Under a point-based trading, the costs are much lower, being on average about 20% of the costs under a command-and-control program. Both N and P abatement targets are over attained for both 20% and 30% targets. Interestingly, for 40% reductions in both N and P, under point-based trading, the N target is slightly over attained, while the P target is not attained.

Table 4-26. Boone Watershed Single Pollutant Point-Based Trading

Boone Watershed	Single Pollutant Point-Based Trading					
Abatement	Nitrogen only Point-Based Trading			Phosphorus only Point-Based-Trading		
N/P	N	*P*	Total Cost	*N*	P	Total Costs
20%	22.0	*29.6*	1.07	*12.6*	19.3	0.37
30%	32.2	*37.6*	3.04	*19.2*	27.9	0.85
40%	41.2	*37.8*	7.04	*27.4*	36.7	1.90

Table 4-26 presents the simulated outcomes for the point-based trading scenarios where only one pollutant is targeted. Interestingly, the outcomes of a nitrogen point-based trading are similar to the outcomes of the trading policy that targets both N and P. Under phosphorus only point-based trading approach, the P abatement targets are on average underachieved by 2.5%, and the total costs are much lower than for nitrogen only point-based trading. However, the total costs are much lower. Qualitatively similar results are obtained for the RRW: the outcomes of the trading setting have the lowest costs and the outcomes of a nitrogen point-based trading approach are the same as the outcomes of a nitrogen and phosphorus point-based trading approach. (see Tables B-22 and B-23 in Appendix B)

Discussions

In this section, I extend the point-credit approximation procedure to multiple pollutant markets where the regulator seeks to achieve simultaneous reductions for multiple pollutants by using a system with a separate point market for each pollutant. The findings show that abatement outcomes of a trading program that considers separate markets for both pollutants are achieved at lower costs relative the CAC or PS policies. However, there are no additional gains relative to the case where there is only a market for nitrogen, since by targeting N reductions significant higher reductions for P are obtained. A trading program for phosphorus only has the potential to achieve its phosphorus abatement goal at much lower costs, but the associated nitrogen abatement levels are not met. The present findings show that there are no additional gains from focusing on both pollutants and the policy programs should be designed by focusing on the pollutant that raises the most interests.

Conclusions

In Chapter 3, I introduced a simple model of pollution and outlined the properties of three different policies under a linear approximation of the abatement function. In this chapter, I provided an empirical assessment of the tradeoffs between the cost efficiency and effectiveness across these policies given that a system of field-level point coefficients is used as a linear proxy for the abatement function. The outcomes of the policies were simulated under different set of assumptions: cost symmetry and cost heterogeneity, and single or multiple pollutants case. Three different levels of abatement were considered: 20%, 30%, and 40 % reductions in the baseline emissions. A robustness analysis for the single pollutant scenario was conducted by assuming a less precise point coefficients are used. The same single policy outcomes were compared to the outcomes of an emission based trading program where trading takes place according to the trading ratios defined by the delivery coefficients. Additionally, I tested whether the abatement outcomes are consistent under the historical weather distribution. Finally, the same set of policies was assessed assuming that two pollutants are simultaneously targeted.

Since I am interested in assessing how different policies perform under a linear approximation tot the abatement (water quality production) function, I assume that the biophysical model is the “exact” representation of the complex water quality model. However, this is clearly not accurate. In reality, using a water quality model like SWAT introduces another level of approximation which is not the focus of my work.

Another note of caution arises from the assumption that the PS and trading outcomes achieves all gains from trade. However, as many authors have noted, when factors like the type of trading (sequential or bilateral), transaction costs or nonmonetary preferences are taken into account some of the gains of an efficient trade may not occurr (Atkinson and Tietenberg 1991; Stavins 1995; Shortle 2013).

CHAPTER 5. CREDIT STACKING IN AGRI-ENVIRONMENTAL PROGRAMS: WATER QUALITY TRADING PROGRAM AND CARBON MARKETS

5.1. Introduction

Several environmental markets that trade a single ecosystem service have been established in the United States: markets for wetland mitigation, water quality trading, and permit trading markets for SO2 allowances, to name a few. Although the number of individual ecosystem markets has increased in the United States and around the world, credit markets that incorporate more than one ecosystem service are almost nonexistent. One example comes from Australia, where under an auction setting, water quality, greenhouse gases, and habitat are bundled together in a single auctioned commodity (Greenhalgh 2008). Agricultural activity as a provider of multiple ecosystem services will play a major role in the future development of these markets around the world.

The poor development of bundling markets where multiple ecosystem services can be traded as a single commodity does not imply that there is little awareness about the multiple benefits of ecosystem markets. Farmers, as profit maximizing agents, are likely to maximize the economic returns associated with the entire range of ecosystem services provided by their actions. One question that arises is whether or not it is socially optimal to allow farmers to produce credits for multiple markets.

In Chapter 4, I empirically evaluated the effectiveness of three policies for controlling nonpoint-source pollution based on a set of abatement actions, where a system of points coefficients are used as a proxy to the effectiveness of the abatement actions. The obtained results show that relative to a command-and-control or performance standard program, significant cost savings can be obtained under a trading program based on the proposed point coefficients. Next, given that the same abatement actions have the potential to increase the amount of carbon sequestration in soil, the trading program can be extended by allowing the trading participants to enter a market for carbon and sell the carbon offsets associated with the abatement actions.

The previous literature has been focused on the efficient design of carbon credit markets, where the abatement actions primarily directed towards the reduction of carbon emissions have indirect environmental co-benefits such as improving water quality or wildlife habitat. In this chapter, I investigate the implications of allowing the farmers to participate in two programs: a water quality trading program and a carbon offset market. My analysis departs from the previous research by considering the participation in the carbon offset market, a global environmental good, as a co-benefit of a water quality trading program, which has localized effects.

The model introduced in Chapter 3 that captures the key attributes of the nonpoint source water quality problem as they relate to the agricultural emissions from farm fields is extended to include the soil carbon benefits associated with the conservation practices. Next, I maintain the same assumptions for the water quality model and I focus on simulating the outcomes of a point-based trading model that includes the participation in the carbon offset market.

This chapter is organized as follows: first, I extend the point trading model for water quality to incorporate the carbon offset component, and second I simulate and compare the trading outcomes in the presence of a carbon offset market with the outcomes obtained in Chapter 4. I consider the cases when the trading program focuses on: (a) the abatement of nitrogen, and (b) the abatement phosphorus. The simulations are made using the costs and environmental data for the Boone River Watershed.

5.2. Point-Based Trading in the Presence of a Carbon Market.

As written in Chapter 4, in the case of a single pollutant market, under a point based trading a farmer solves the cost minimization problem at farm level:

$$\min_{x_i, b_i} C_i(x_i, \gamma_i, \theta_i) + p * b_i \quad s.t \quad \sum_i^J a_{ij} x_{ij} + b_i \geq b_i^o \tag{25}$$

where b_i^o is the number of points requirement for a farmer. The equilibrium condition is: $\sum_i^N b_i = 0$. Hence, a farmer chooses the abatement actions and the number of points to meet a field level requirement given by b_i^o.The trading outcome is determined by:

$$\min_{x_i, b_i} \sum_i^N C_i(x_i, \gamma_i, \theta_i) \ s.t.$$
$$\sum_i^N \sum_i^J a_{ij} x_{ij} + b_i \geq \sum_i^N b_i^o \quad \sum_i^N b_i = 0 \tag{26}$$

where x_i^* represents the optimal abatement action chosen by field i. The total cost of the trading program is $TC^P = \sum_i^N C_i(x_i^*, \gamma_i, \theta_i)$.

In the presence of a carbon offset market, where there are no constraints on the minimum or maximum amount of carbon offset that can be sold, and the farmers minimization problem becomes:

$$\min_{x_{ij}, b_i} C_i(x_i, \gamma_i, \theta_i) - p_c g(x_i) + p b_i \quad s.t \quad \sum_{ij}^J a_{ij} x_{ij} + b_i \geq b_i^o \tag{27}$$

where, p_c is the price of a carbon offset takes as given to the farmer and $g(x_i)$ represents the amount of soil carbon sequestration given the vector of abatement actions x_i. While the field constraint remains unchanged, the objective function is adjusted to account for the revenue that can be obtained from selling carbon offsets. The carbon offsets are the amount of soil carbon sequestration resulting from adopting an abatement action. The total costs of the trading program to farmers in the presence of a carbon market are given by:

$$TC^{PC} = \sum_i^N \{C_i(x_i^{**}, \gamma_i, \theta_i) - p_c g(x_i^{**})\} \tag{28}$$

where x_i^{**} represents the optimal abatement action chosen by field i when they can sell carbon offsets. The total cost of a trading program, C^{PC}, has two components: the cost of implementing the optimal abatement practices,$\sum_i^N C_i(x_i^{**}, \gamma_i, \theta_i)$, and the revenues from the sale of carbon offsets $\sum_i^N p_c g(x_i^{**})$.

Depending on the price of carbon, the carbon market creates incentives that can alter the choice of abatement action. Thus, farmers are able to reduce the cost of participation in the trading program or even obtain an additional income. At the same time, the carbon market introduces competition for land use. Farmers move towards conservations practices that might have more potential for carbon sequestration but are less effective in reducing the nitrogen or phosphorus emissions. The post trading water quality outcomes can be higher or lower compared to the case where farmers cannot sell carbon offsets, but the cost to the farmers will be lower ($TC^{PC} < \mathrm{TC}^{\mathrm{P}}$) At the same time,the cost of implementing the abatement actions may increase, because the abatement actions that are more effective for carbon sequestration may be more expensive ($\sum_i^N C_i(x_i^{**}, \gamma_i, \theta_i) <> \sum_i^N C_i\ (x_i^{*}, \gamma_i, \theta_i)$. The direction of the realized abatement and the magnitude of the cost savings in the presence of a carbon market are empirical questions that I will evaluate in the next sections.

5.3. The Assessment of Point-Based Trading Market in the Presence of a Carbon Offset Market

The empirical assessment of the trading program when farmers are allowed to participate in a carbon market is made under the assumption of cost heterogeneity. The cost heterogeneity is simulated using the same random draws used in Chapter 4, section 4.4. Hence, the costs are given by:

$$C_i(x_i, \gamma_i, \theta_i) = \sum_j^J (c_j(\gamma_j) - p_c g_j)(1 + u_i) x_{ij} s_i \quad \forall i = 1, \dots, N \tag{29}$$

where $c_j(\gamma_i)$ is the per acre cost for the j^{th} abatement action, p_c is the price per carbon, g_j the amount of carbon associated with the j^{th} abatement action, u_i is the i farmer's random draw, $u_i \sim U[-0.8, 0.8]$, x_{ij} is a dummy variable that takes value “1” if the j^{th} abatement action is adopted and “0” otherwise, and s_i is the area of field *i*. The simulations are run for 1,000 realizations of the random variable u.

5.3.1. Obtaining the soil carbon values associated with each abatement action

The field level soil carbon estimates associated with each abatement actions are needed in addition to the already estimated point coefficients for the empirical assessment of the extended trading policy. The field levels for soil carbon sequestration are simulated by running EPIC (Erosion Productivity Impact Calculator) for each abatement action and each field. The carbon benefit values for each field and each abatement action are equal to the difference between the annual carbon yield under the abatement action and the carbon yield under the baseline scenario.[38]The original EPIC values, originally expressed in kilograms of carbon per hectare, are

[38] The annual carbon yields are measured as the average over 30 years.

multiplied by a factor of 3.67 for conversion to metric tons carbon dioxide equivalent (MtCO2e).[39]

Table 5-1 summarizes the descriptive statistics for the simulated soil carbon sequestration values obtained for the fields in the Boone River Watershed. As expected, the “Reduce fertilizer” is the abatement action that has the least impact on carbon sequestrations. Interestingly, the same abatement action seems to slightly decrease the effectiveness of the “No till ” and “Cover crops” when combined together. This is similar to the sub-additivity patterns observed under the point coefficients assigned to the same abatement actions. By contrast to the point coefficients, a supper-additivity pattern is observed when no till and cover crops are combined together, in the sense that more soil carbon is sequestered under the combination of the two than the sum of the two taken individually. The combination of “No till and Cover crops” with an average of 0.816 MtCO2e per acre is the most efficient abatement action for soil carbon sequestration. Compared to the point values estimated, “Land retirement” is not the most efficient abatement action for carbon sequestration. Likewise, the combination of “No till and Cover crops” is more efficient than the combination “Reduced fertilizer, no till and cover crops”. Similar to the phosphorus point value estimates, “No till” is more efficient in carbon sequestration than “Cover crops”.

For the purpose of the trading simulations, I assume that a given abatement action, in terms of the amount of soil carbon sequestration, has the same per acre impact. This implies that for a given abatement practice j , the per acre carbon sequestration is the same, $g(x_{ij}) = \bar{g}_j , \forall\, i$.

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[39] The EPIC values should be interpreted with caution, since the model was not fully calibrated at the simulation moment. A zero value is accounted for the fields where the EPIC failed to provide accurate estimates.

While this assumption is not likely to be perfectly accurate, it is likely that an actual carbon offset program would credit all fields in a given location that adopt the same practice the same amount of offsets. An example for this was the aggregation from no till sold as the Chicago Climate Exchange.

The column labeled as “Trading” in Table 5-1 shows the per acre equivalent of carbon assigned to each field given an abatement action. Figure C-4 in Appendix B describes the field level distributions of the soil carbon values for each abatement action. Since a relatively high number of fields have zero values for the soil carbon sequestration, the distributions have a high density at zero.

Table 5-1. Soil carbon sequestration (MtCO2e)

Abatement action	Mean	Std.dev.	Min	Max	Median	Trading*
No Action	0.00	0.00	0.00	0.00	0.00	0.00
No till	0.27	0.29	0.00	2.16	0.20	0.28
Cover Crops	0.25	0.23	0.00	2.22	0.22	0.25
No till, Cover Crops	0.82	0.42	0.00	3.78	0.74	0.79
Red.Fert	0.00	0.02	0.00	0.17	0.00	0.00
Red.Fert,No till	0.23	0.27	0.00	2.09	0.14	0.22
Red.Fert, Cover Crops	0.21	0.21	0.00	2.03	0.16	0.19
Red.Fert,No till, Cover Crops	0.76	0.39	0.00	3.60	0.69	0.72
Land retirement	0.55	0.57	0.00	5.42	0.45	0.52

*Represents the MtCO2e assigned to each field within the trading simulations.

5.3.2. Empirical findings

In Chapter 4, I simulated the trading outcomes of a point based-trading market for water quality. Next, I simulate the trading outcomes for the same abatement targets, but in the presence of a carbon offsets market. The two sets of outcomes are compared with respect to different aspects: the post trading abatement levels, the final costs of achieving the water quality target, the costs of implementing the abatement actions in the presence of a carbon market, the total and the additional levels of soil carbon sequestration, and the levels of equilibrium price in the points-based market. I also analyze the extent of land use competition induced by the participation in a carbon market.

I analyze two scenarios: in the first scenario, the trading market is focused on nitrogen abatement and in the second scenario, the trading market is focused on the phosphorus abatement. For each of these scenarios, I use three different abatement targets for the water quality trading program and three different price levels for the carbon offsets[40]. Given the possible combinations of abatement targets and carbon pricing levels, for each pollutant point-based trading scenario, there are nine additional sub-scenarios. For each sub-scenario, the trading outcomes are simulated by assuming the cost heterogeneity at the watershed scale.

For the convenience of notation, the nitrogen (phosphorus) point-based trading baseline is denoted as $\mathrm{PBT}_{\mathrm{N(P)}}$, and the case with a carbon market as $\mathrm{PBTC}_{\mathrm{N(P)}}$. The nitrogen (phosphorus) point-based trading market only can be interpreted as the situation where the carbon offset price is equal to zero, hence the results obtained in the previous chapter can be interpreted as the outcomes when the price in the carbon market are zero. These outcomes will be used as a benchmark comparison for the results obtained in the presence of positive carbon prices[41].

In the presence of a carbon offset market that functions parallel with a trading market designed to improve the water quality, farmers will face a different set of incentives in choosing the abatement actions. This has further implications on the post-trading level of abatement, total

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[40] Water quality abatement levels: 20%, 30%, 40% reductions relative to the baseline;
Price level for carbon offsets: \$5, \$15, \$25 per metric tone equivalent of carbon dioxide.

[41] The trading outcomes are corresponding to the "satisficing" approach. Similar results are expected under the optimizing approach.

costs of trading program, the price of a point permit, and the total level of realized soil carbon sequestrations. Next, I present how these results change in the presence of a carbon market relative to the case where there is no such a market.[42]

The attainment of the abatement target

The first aspect I am interested in in the presence of the carbon offset market is the impact on the post trading level of abatement. Is the abatement target achieved? How does it compare relative to the corresponding outcomes with no carbon market?Let ΔWQ be the difference between the realized abatement levels in a PBTC market (WQ^{PBTC}) and the trading outcomes in a PBT market (WQ^{PBT})

$$\Delta WQ_{N(P)} = WQ_{N(P)}^{PBTC} - \; {WQ_{N(P)}}^{PBT} \tag{30}$$

Table 5-2. Realized abatement levels: nitrogen

Nitrogen Point-Based Trading Market							
Price	\$0	\$5		\$15		\$25	
Target	PBT_N ,%	$PBTC_N$,%	ΔWQ_N ,%	$PBTC_N$,%	ΔWQ_N ,%	$PBTC_N$,%	ΔWQ_N ,%
20%	22.1	22.1	0.04	22.0	0.0	24.9	2.8
30%	31.7	31.8	0.1	32.0	0.3	32.0	0.3
40%	41.4	41.5	0.1	41.6	0.1	41.6	0.2

Table 5-2 presents the post-trading abatement levels for a nitrogen PBTC market ($PBTC_N$) . On average, the post-trading abatement levels for $PBTC_N$ are over attained and

[42]The results are presented both in absolute and relative level.

similar to the post-trading abatements level for a single market trading only (presented in the first column). The difference between the achieved abatement levels (ΔWQ_N ,%) in the case of PBT-nitrogen are small, less than 0.05% However, the differences are statistically significant different than zero.[43]

Table 5-3. Realized abatement levels: phosphorus

	Phosphorus Point-Based Trading Market						
Price	\$0	\$5		\$15		\$25	
Target	PBT_P,%	$PBTC_P$,%	ΔWQ_P ,%	$PBTC_P$,%	ΔWQ_P ,%	$PBTC_P$,%	ΔWQ_P ,%
20%	21.7	20.3	0.0	20.0	-0.3	26.5	6.2
30%	29.4	29.3	0.0	29.0	-0.4	28.4	-0.9
40%	37.0	37.0	0.0	36.7	-0.3	36.2	-0.8

$PBTC_P$ phosphorus point-based trading market and carbon market

Slightly different results are obtained for the trading market scenario that focuses on phosphorus abatement, as shown Table 5-3. On average, the post-trading abatement levels for PBTC-phosphorus are not attained but at the same time they are not very different from the PBT-phosphorus outcomes. For a given abatement target and a given price for carbon, the difference between the post trading abatement outcomes is less than 0.1 %.

Overall, comparing across the two types of pollutants, the post trading abatement levels in the presence of a carbon market are similar to the results observed in the presence of a nutrient trading market only. The post trading abatement levels are higher for a nitrogen point-based market and lower for a phosphorus point based market.

[43] p_values for t_test for equal means is less than 0.05

Soil carbon sequestration gains in the presence of a carbon market

The process of soil carbon sequestration takes place regardless the existence of the carbon offset market hence, carbon soil sequestration will be an accounted outcome of a trading program based on a set of abatement actions which in addition to improving water quality have the potential for soil carbon sequestration. The total carbon sequestration gains are determined relative to the levels of soil carbon sequestration associated with the nitrogen (phosphorus) point-based trading market, since soil carbon levels are realized even in the absence of a carbon market. The soil carbon sequestration gains (ΔCG^C) are determined as:

$$\Delta CG^C = \frac{Csoil^{PBTC} - Csoil^{PBT}}{Csoil^{PBT}} * 100 \quad (31)$$

where $Csoil^{PBT}$ represents the total levels of soil carbon sequestration associated with PBT case and the $Csoil^{PBTC}$ associated with the PBTC case. Both $Csoil^{PBT}$ and $Csoil^{PBTC}$ levels are additional to the soil carbon values associated with the baseline.[44,45]

Table 5-4. Soil carbon gains: nitrogen

	Nitrogen Point-Based Trading Market						
Price	\$0	\$5		\$15		\$25	
Target	$Csoil^{PBT}$	$Csoil^{PBTC}$	ΔCG^C, %	$Csoil^{PBTC}$	ΔCG^C, %	$Csoil^{PBTC}$	ΔCG^C, %
20%	71,429	80,057	12.1	108,003	35.6	156,234	118.8
30%	135,833	148,069	9.0	171,017	25.9	190,630	40.4
40%	200,251	215,537	7.6	242,933	21.3	267,776	33.8

[44] The total amount of post trading carbon sequestration is computed by using the field specific values (the ones presented in Figure C-4) rather than the per acre average values used in obtaining the trading simulations.
[45] The carbon sequestration literature defines the baseline as existing levels of carbon had a program not been implemented. While, for each abatement action, the field soil carbon levels are determined similarly, the trading simulations in the presence of carbon do not take into account the soil carbon levels that would have been realized in the absence of the carbon offset market. The carbon gains presented in this section are determined ex post.

$Csoil^{PBT}$, $Csoil^{PBTC}$, expressed as MtCO2e,

Table 5-4 summarizes the soil carbon levels associated with the nitrogen point based trading market both in absolute and relative size. For a given abatement target, the relative size of the carbon gains increases as the carbon price increases. For example, for a 20 % N abatement target, under a PBTC the level of carbon sequestration increases by 12.10 % ($5), 35.61% ($15), and 144.56% ($25). For a given price for carbon, the relative size of the carbon gains decreases as the abatement targets increase. Given a price level of $5 for carbon, the relative carbon gains are 12.10 % (20%), 9.03 % (30%), and 7.64% (40%), respectively.

Table 5-5. Soil carbon gains: phosphorus

Phosphorus Point-Based Trading*							
Price	$0	$5		$15		$25	
Target	$Csoil^{PBT}$	$Csoil^{PBTC}$	ΔCG^{C}, %	$Csoil^{PBTC}$	ΔCG^{C}, %	$Csoil^{PBTC}$	ΔCG^{C},%
20%	36,512	45,290	24.1	85,031	133.2	156,234	328.6
30%	68,905	74,417	8.0	105,856	53.7	159,922	132.2
40%	123,222	126,031	2.3	145,676	18.2	187,091	51.8

*$Csoil^{PBT}$, $Csoil^{PBTC}$, expressed as MtCO2e,

Similar trends are obtained for the phosphorus point-based trading market. For a given water quality abatement level, the relative size of the carbon gains increases as the carbon price increases (Table 5-5). For example, for a 20% phosphorus abatement, the average relative carbon gains are 6.9 % ($5), 133.16 % ($15), and 328.6% ($25). For a given price for carbon, the relative size of the carbon gains decreases as the abatement levels increase. Given a price level of $5 for carbon, the average relative carbon gains are 24.12 % (20%), 8.01 % (30%), and 2.28 % (40%).

A cross comparison across the two points-based trading market settings shows that in the absence of the carbon market, the soil carbon levels are higher for the nitrogen point-based trading market, being on average, two times higher than those observed in the phosphorus trading market. These patterns are consistent with the correlation values between the points coefficients and the carbon values summarized in Table 5-1(the correlation coefficient between N points and carbon values is 0.74 while the correlation coefficient between P points and carbon is 0.53). Although these results hold in absolute terms, in relative terms the carbon gains are, on average, higher in the latter case. This implies, that the marginal rate of carbon sequestration is higher when the trading program is designed for the phosphorus abatement.

Cost savings to the trading program

In the presence of a carbon market, the potential revenue for selling carbon offsets reduces the initial costs of the abatement actions to farmers. The relative size of the cost savings (ΔCS^C) is:

$$\Delta CS^C = \frac{TC^{PBT} - TC^{PBTC}}{TC^{PBT}} * 100 \qquad (32)$$

where TC^{PBT} is the total trading costs for a nitrogen (phosphorus) trading market and TC^{PBTC} are the total trading costs in the presence of the carbon market. Negative values for TC^{PBTC} imply that by being able to sell carbon offsets, the abatement costs are more than offset by the revenues from the carbon market. Negative values for TC^{PBTC} translate into values higher than 100% for ΔCS^C, the relative size of the cost savings.

Table 5-6. Cost savings to the trading program: nitrogen

	Nitrogen Point-Based Trading						
Prices	$0	$5		$15		$25	
Target	TC^{PBT}	TC^{PBTC}	ΔCS^C, %	TC^{PBTC}	ΔCS^C, %	TC^{PBTC}	ΔCS^C, %
20%	939,440	550,550	41.4	-412,493	144.0	-1,740,452	285.5
30%	2,515,932	1,798,621	28.5	191,169	92.5	-1,648,471	165.6
40%	6,020,182	4,943,189	17.9	2,572,622	57.3	-46,975	100.8

Table 5-6 summarizes the gains in total trading costs for a nitrogen point-based trading market both in absolute and relative size. For a given level of abatement for water quality, the relative size of the cost savings increases as the carbon prices increase. For example, for a 20% nitrogen abatement level , the relative cost savings are, on average, 41.4 % ($5), 144.03% ($15), and 285.54 %($25). For a given price for carbon, the relative size of the cost savings decreases as the abatement levels increase. Given a price level of $5 for carbon, the relative cost savings are 41.4% (20%), 28.53 % (30%), and 17% (40%).

Table 5-7. Cost savings to the trading program: phosphorus

Phosphorus Point-Based Trading							
Price	$0	$5		$15		$25	
Target	TC^{PBT}	TC^{PBTC}	ΔCS^{C}, %	TC^{PBTC}	ΔCS^{C}, %	TC^{PBTC}	ΔCS^{C}, %
20%	337,180	122,121	63.9	-516,067	253.5	-1,740,452	617.5
30%	857,851	499,232	41.8	-373,236	143.6	-1,737,535	302.8
40%	1,953,064	1,343,156	31.2	41,712	97.9	-1,624,690	183.3

Similar trends can be observed for the phosphorus point based trading market. For example, for a 20% phosphorus abatement, the relative costs savings are, on average, 63.9 % ($5), 253 %($15), and 582.15 %($25). For a given price for carbon, the relative size of the cost savings decreases as the abatement levels increase. Given a price level of $5 for carbon, the means of relative costs savings are 63.9 % (20%), 41.8 % (30%), and 31.2 (40%). When the relative size of the cost savings is higher than 100 %, the total costs of PBTC are negative, implying that the revenues from the carbon market offset the abatement costs.

Overall, the relative cost savings are higher in the case where the carbon markets are available parallel with a phosphorus point-based trading market rather than the nitrogen based scenario. Previously, results show that the relative carbon gains are higher in the case of phosphorus scenario. These results can also explain why the relative cost savings from trading are higher for phosphorus scenario.

Relative equilibrium price in the point-based trading market

The point equilibrium price will be lower in the presence of a carbon market, since the carbon offset price can be interpreted as a subsidy for implementing an abatement action. Let $\Delta ReqP^{C}$ be the relative reduction in the equilibrium point price:

$$\Delta ReqP^{C} = \frac{ReqP^{PBT} - ReqP^{PBTC}}{ReqP^{PBT}} * 100 \qquad (33)$$

where $\mathrm{ReqP}^{\mathrm{PBTC}}$ is the equilibrium price in the presence of positive carbon prices and $ReqP^{PBTC}$ is the equilibrium point price otherwise.

Overall, the equilibrium price in either point-based trading market decreases with abatement target levels and price levels of carbon. A similar trend is observed for the relative equilibrium price levels, $ReqP^{C}$. By fixing the price of carbon, $ReqP^{C}$ decreases as the abatement targets increase, while by fixing the abatement target, $ReqP^{C}$ increases as the carbon price increases. This relationship is not surprising as the price of carbon offsets decreases the abatement costs. Hence, if a subsidy were to be paid for adopting abatement actions at field scale, the subsidy's size would be reduced by the value of the carbon offsets that could be sold in a carbon market. The results are summarized in Table C-1 (nitrogen) and C-2 (phosphorus).

Cost Savings for the implementation of the abatement actions

As mentioned earlier, the carbon market induces competition for land use, the benefits for water quality are traded off with the benefits for carbon sequestrations. Facing more options and positive carbon offset prices, farmers may choose abatement practices that are more efficient for soil carbon sequestration, even though otherwise they are more expensive. As shown earlier, the total trading costs in the case of a carbon offset market have two components: the implementing costs for the adopted abatement actions and the revenues from the carbon market. Next, I determine the first component of the total costs. (i.e. how much does it cost to implement the new optimal abatement actions)-CI^{PBTC}. Next, I compute the changes in the total abatement costs relative to the case when carbon offsets are not available,

$$\Delta CI^{C} = \frac{CI^{PBTC} - TC^{PBT}}{TC^{PBT}} * 100 \tag{34}$$

where the implementation costs for a point based trading program only are equivalent to the total cost of the trading program, TC^{PBT}.

Table 5-8. Additional costs to implementing the abatement actions: nitrogen

		Nitrogen Point-Based Trading					
Price		$5		$15		$15	
Target	TC^{PBT}	CI^{PBTC}	ΔCI^{C}, %	CI^{PBTC}	ΔCI^{C}, %	CI^{PBTC}	ΔCI^{C}, %
20%	939,440	962,971	2.5	1,273,419	35.6	2,294,890	144.6
30%	2,515,932	2,545,813	1.2	2,782,293	10.6	3,218,989	28.0
40%	6,020,182	6,324,207	0.6	6,324,207	5.1	6,797,785	12.9

Table 5-8 presents the results for the abatement costs comparison. Overall, more expensive abatement practices are chosen in the presence of a carbon offset market. For a given price level, ΔCI^{C} decreases as the abatement level increases (i.e. for a carbon price of $5: 2.51 (20%), 1.19 (30%), and 0.65 (40%)). Alternatively, for a given level of abatement, ΔCI^{C} increases as carbon price increases (i.e. 20% N reduction: 2.51 ($5), 51.26 ($15), and 118.84 ($25)).

Table 5-9. Additional costs to implementing the abatement actions: phosphorus

Phosphorus Point-Based Trading							
Price		$5		$15		$15	
Target	TC^{PBT}	CI^{PBTC}	ΔCI^{C}	CI^{PBTC}	ΔCI^{C}	CI^{PBTC}	ΔCI^{C}
20%	337,180	360,699	7.0	823,727	144.7	2,294,890	582.1
30%	857,851	873,578	1.8	1,263,399	47.4	2,386,515	178.4
40%	1,953,064	1,960,597	0.4	2,984,105	52.9	3,099,433	58.8

A similar trend is observed in the case of phosphorus. For a given price level, ΔCI^{C} decreases as the abatement level increases (i.e. a carbon price of $5: 6.9 (20%), 1.83 (30%), and 0.83 (40%)). Alternatively, for a given level of abatement, ΔCI^{C} increases as carbon price increases (i.e.20% P reduction: 6.99 ($5), 253.5 ($15), and 328.6 ($25)) . The relative increase in abatement costs is higher in the phosphorus trading market.

Land use competition

In this section, I present an analysis of the land use competition in the presence of a carbon market. Table 5-10 presents number of fields (as both a percentage of total area and as a percentage of total number of fields) that switch to a different abatement action in the presence of the carbon market. Both nitrogen and phosphorus outcomes exhibit similar trends: keeping the abatement target fixed, more land or more fields switch to a different abatement action as the price of carbon increases. At the same time, keeping the price of carbon fixed, less total land and a fewer number of fields change the land use as the abatement target increases.

Table 5-10. Cost savings to the trading program, nitrogen

	Nitrogen			Phosphorus		
			Carbon price($)			
			Area change , %			
Target	$5	$15	$25	$5	$15	$25
20%	21.10	40.66	62.07	21.05	44.84	69.83
30%	18.58	31.44	43.41	18.87	38.35	62.69
40%	18.56	33.36	44.31	14.56	24.32	40.83
			Field switching, %			
20%	23.09	41.40	60.44	21.77	43.57	66.09
30%	21.55	34.36	46.29	19.76	37.24	59.70
40%	19.97	33.10	43.88	15.64	25.03	41.00

Figure 5-1 summarizes the distribution of the abatement actions across different levels for carbon prices and 30% N abatement targets; as the carbon price increases more and more fields choose “No till” and the combination of “No till” and “Cover crops.” A similar pattern is observed for a 30% P abatement target. Figure 5-2 show compares the spatial distribution of the abatement actions across different prices and 30% N abatement goal for Boone River Watershed.

Overall, when the price for carbon offsets is positive, more fields adopt an abatement action different from the baseline. These changes translate in the adoption of “No till” and “No till and Cover crops.” At the same time, fewer fields choose “Land Retirement” and “Reduced fertilizer.” These changes are as expected given that “No till and Cover crops” are more efficient abatement actions for soil carbon sequestration. Similar results for phosphorus-based trading can be found in Appendix B.

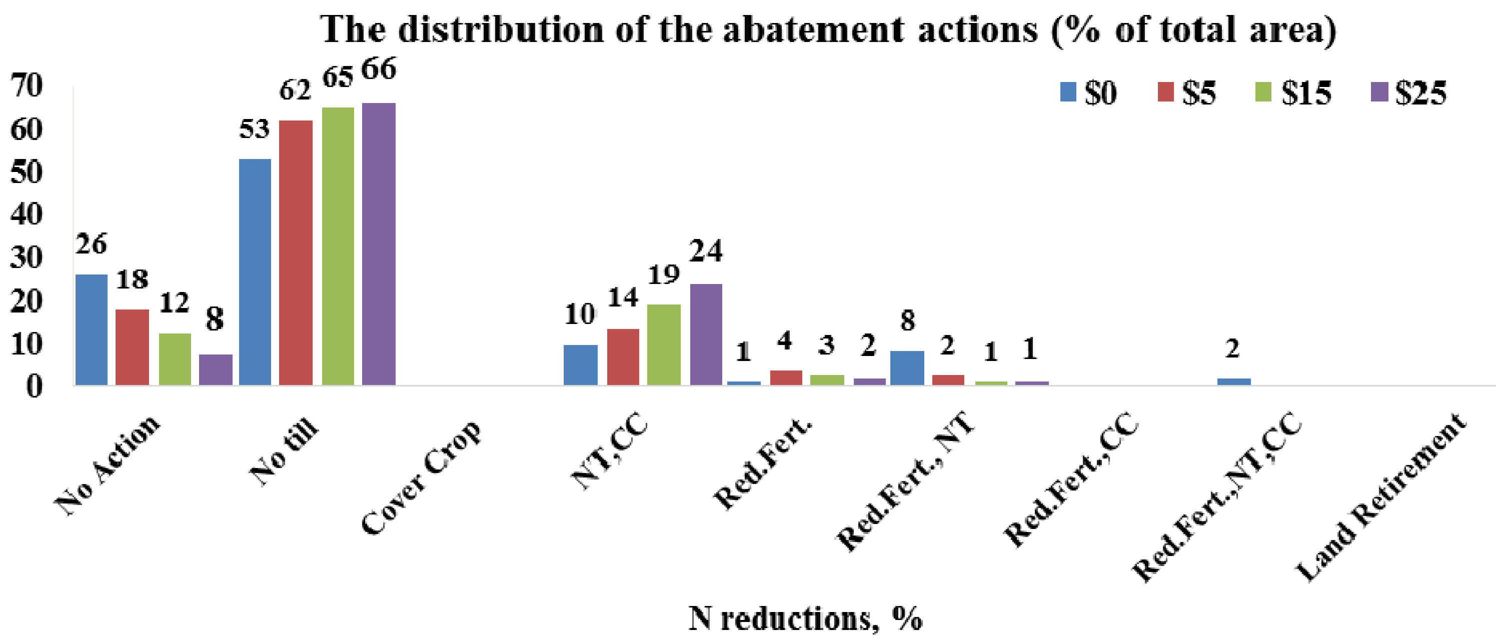
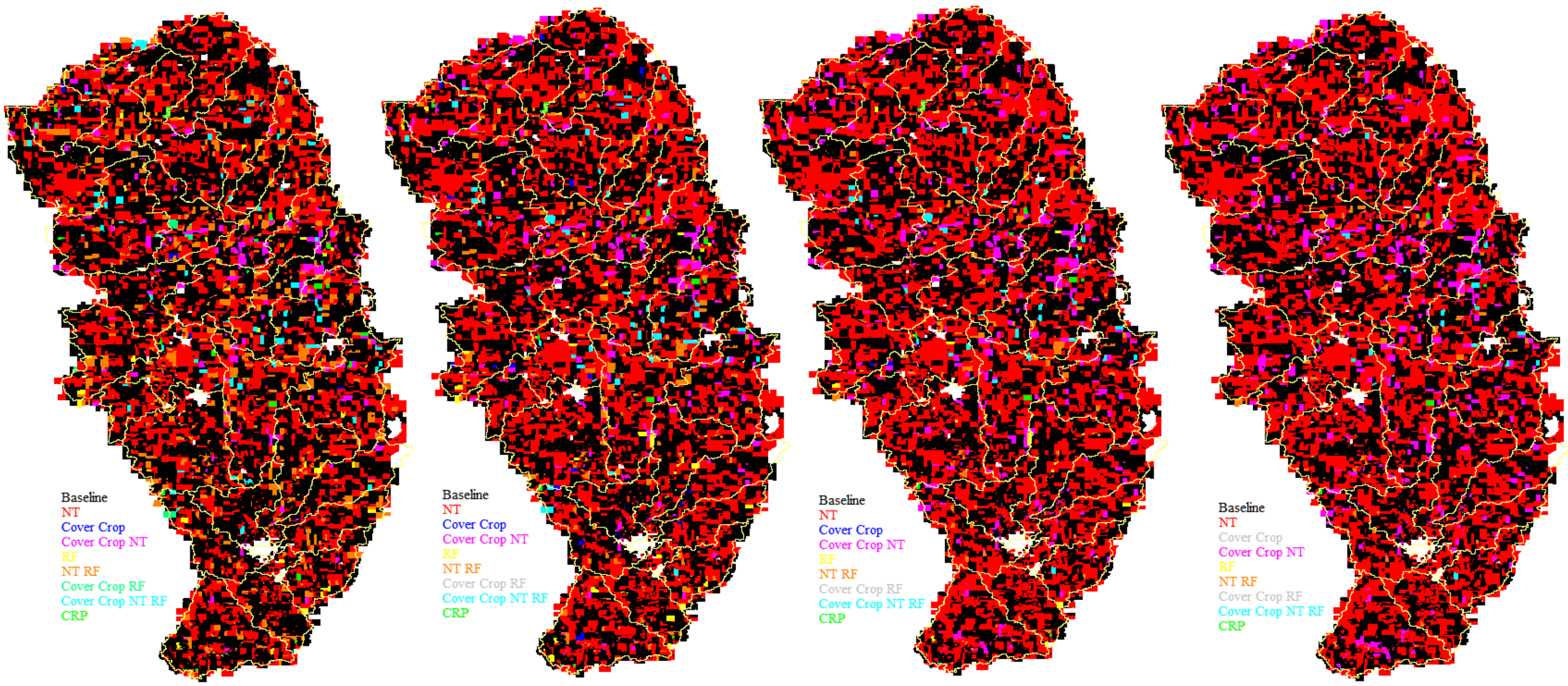


Figure 5-1. The distribution of the abatement actions 30% N abatement goal.



No carbon market **$5 MtC02/tonne** **$15 MtC02/tonne** **$25 MtC02/tonne**

Figure 5-2. Spatial distribution of abatement actions , 30 % nitrogen.

A few hotspots emerge in the presence of a carbon market. However, the number is small relative to the total number of fields in the watershed. Table C-3 summarizes the distribution of the hotspots under both abatement programs. Another aspect to be considered is the overall trading activity across the watershed, where the trading activity is measured as the weighted average area in a subbasin that chooses a different abatement action than the one given by the field level constraint. As Figure 5-3 shows for different levels of N abatement, the trading activity slightly increases as the price of carbon increase. However, the distributions are very similar. The results summarized in Figure 5-3 together with the results summarized in Table 5-8 suggests that even though the trading activity does not change in a significant manner the trading quality is changed, in the sense that different abatement actions are traded as the price of carbon offset increases.

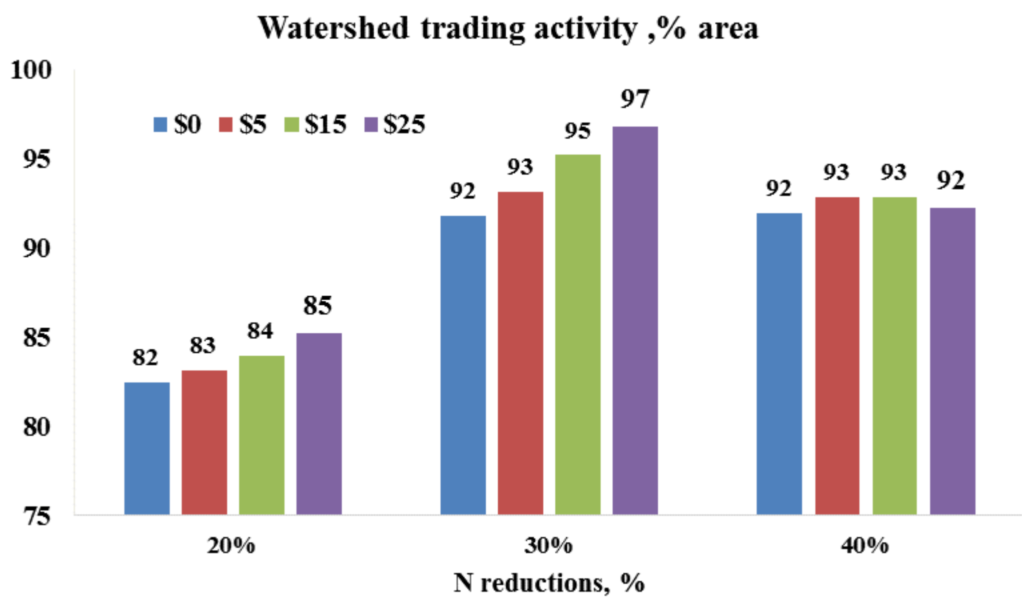


Figure 5-3. Overall trading activity, nitrogen abatement.

5.4. Conclusions

Water quality and improved soil can provide additional ecological services such as improved fishing and wildlife habitat and are among the most important qualities of a healthy watershed which. At the same time, the carbon sequestrated in soil plays one of the most important roles in reducing the greenhouse gases.

This chapter explores the impact of participation in a carbon offset market on the cost efficiency of a water quality program designed for a typical agricultural watershed. The analysis considers participation in a carbon market, a global environmental good, as a co-benefit of a water quality trading program, which has localized effects. The water quality program is a cap-and-trade type of program whereas there is no cap for the carbon market.

This chapter highlights the changes in the social costs of achieving a water quality target when farmers are allowed to participate in two parallel programs: a water quality trading program and a carbon offsets market. My analysis considers three different levels of carbon pricing, \$5, \$15, and \$25 per MtC, and three levels of water quality goal, 20%, 30%, and 40% N reductions. The realized abatement levels are not very different in the cases when a carbon market is available. The farmers' total cost of a point-based trading program decreases as the price of carbon increases. As the price of carbon increases, for lower water quality abatement targets, the total costs become negative, meaning that farmers obtain extra revenues by selling the carbon offsets which offset the cost of implementing the abatement practices. While the costs of the program are reduced, the costs of implementing the abatement actions increases. This relative increase in the implementation costs is higher for lower water quality abatement targets. The equilibrium price of a credit in a water quality trading market is reduced when farmers are allowed to participate in the carbon offset market. Additionally, the results quantifying the extent of land competition between two programs shows that, for the current application, the distribution of the abatement actions being similar across different levels of carbon pricing.

This chapter offers a better understanding on how the agriculture can influence multiple environmental outcomes: water quality and carbon sequestration. It combines the outcomes of a two different watershed based models (EPIC and SWAT) together with an optimization algorithm to evaluate the implications on a hypothetical trading program designed for the non-point sources.

Several caveats must be taken into account. The EPIC estimates for soil carbon have not been fully calibrated for the watershed studied in this chapter. A caution, as in previous chapters in interpreting the results, applies in this case: the performance of the trading programs is based on the theoretical abilities of the optimization problem defined at the beginning of the chapter. In reality, the transaction costs and the nature of the trading may understate the efficiency gains. Moreover, I assume that the carbon offsets can be sold in the current units. In reality, carbon offsets are standardized and require minimum amounts of $MtCO_2$, requirements that cannot be met at farm level. Additionally, leakage, the uncertainty related to the carbon emission, issues broadly discussed in previous literature, have been assumed away.

CHAPTER 6. CONCLUSIONS

Despite large public expenditures on programs such as the Conservation Reserve Program and state-based payment programs to encourage farmers to adopt conservation practices, water quality problems associated with agricultural nonpoint-source pollution remain significant in the majority of US watersheds. The observability of field-level emissions and the complex fate and transport relationship linking them to the ambient water quality, together with the imperfect knowledge of the abatement costs, have been critical aspects of the policy design of agricultural pollution.

In my dissertation, I present a theoretical model of water quality that captures the main characteristics of pollution within an agricultural watershed. Next, I propose and empirically estimate a simplified proxy model for the complex process that characterizes the fate and the transport of agricultural pollutants, a process known also as the water production function. Finally, I apply this model in a variety of empirical studies to evaluate alternative policy programs (command-and-control, a performance standard and trading) designed to improve water quality. In my empirical evaluation, I use a data-rich, spatially detailed model of land use and water quality for two agricultural watersheds in Iowa: the Boone River Watershed and the Raccoon River Watershed. To my knowledge, this is the first time when a simplified trading program based on points that measure the impact of abatement actions has been carefully examined in a simulation environment, where the simulations are calibrated to real word watersheds.

I begin by providing a literature review of the economics of nonpoint-source pollution associated with agricultural activity. I review the different policy approaches related to water quality trading that involves agricultural nonpoint sources. I also provide a brief description of the biophysical models (SWAT, EPIC) and optimization tools used in my empirical analysis.

In Chapter 3, I present a conceptual model to manage the ambient water quality in a watershed impaired by agricultural runoff. My model captures critical aspects of the agricultural pollution. Next, I outline the properties of three policies under two assumptions for the abatement function. The first approach is a command-and-control where the regulator has the ability to mandate specific abatement actions to each farm in the watershed. The second approach is a performance standard where each farm has to meet predetermined farm-level performance requirements by choosing relevant abatement actions. The third approach is a trading program where farmers, conditional on meeting their farm-level performance requirement, can trade points assigned to the abatement actions.

The main message that echoes from the conceptual model is that under a linear approximation of the abatement function, more flexible policies like the performance standard or trading program may outperform a command-and-control program in terms of abatement costs, but they may also result in the non-attainment of the abatement goal. However, the incentive-based policies can overcome, partially or totally, the issue of cost asymmetries, since the regulator does not need to know the farm-level abatement costs. My modelling framework allows me to estimate the magnitude of these efficiency tradeoffs for the first time. Furthermore, the close calibration to two real watersheds offers valuable insights for the design of the actual policy.

Next, I propose and estimate an approach for linearizing the abatement function using a system of point coefficients that measure the impact of an abatement action on the overall abatement level. Three levels of specificity are used to estimate the point coefficients: field, subbasin, and watershed level. The point coefficients are estimated for nitrogen and phosphorus with consideration that the two pollutants have separate abatement functions. The estimated values for the point coefficients follow a priori expectations. Moreover, the point coefficients have similar patterns across the two watersheds.

In Chapter 4, I empirically assess the tradeoffs between the cost efficiency and effectiveness of the three policies when a system of field-level point coefficients is used as a linear proxy for the abatement function. For the first set of simulations, I consider that the regulator and the farmers have the same cost information. Under this assumption, the empirical assessment of the proposed policies shows an overall good performance of the incentives based programs: the deviations from the abatement goals are not significant and sizable cost savings relative to the command-and-control programs are realized. A robustness analysis shows that the results are consistent across different: (a) pollutants (nitrogen and phosphorus), (b) sets of point coefficients (field-specific level, subbasin specific, or watershed specific), and (c) the distribution of historical weather. The point approximation procedure is extended to two pollutant markets, where each market uses a separate set of point coefficients. The findings show that there are no additional gains from focusing on both pollutants. At the same time, it is important to acknowledge that these results may be a special case and do not have to be generalized.

For the second set of simulations, I consider that the regulator and the farmers have different cost information, where the cost asymmetry is simulated as costs heterogeneity. The regulator uses some limited information (average of the abatement costs) when designing his policies. The simulated outcomes support the findings obtained earlier.

Water quality and improved soil are among the most important qualities of a healthy watershed. At the same time, the carbon sequestrated in soil plays one of the most important roles in reducing greenhouse gases. Given that the same abatement actions that have the potential to increase the amount of carbon sequestration in soil, the point-based trading program is extended to allow trading participants to enter a market for carbon, including selling the carbon offsets associated with the abatement actions.

Chapter 5 explores the impact of participation in a carbon offset market on the cost efficiency of a water quality program designed for the Boone River Watershed. The water quality program is a cap-and-trade program whereas there is no cap for the carbon market. The empirical findings show that: (a) the realized abatement levels are not very different in the cases where a carbon market is available, (b) the farmers' total cost of a point-based trading program decreases as the price of carbon increases, (c) as the price of carbon increases the total costs become negative, meaning that farmers obtain extra revenues by selling the carbon offsets, which offset the cost of implementing the abatement practices, (d) the costs of implementing the abatement actions increase, and (e) there are not significant changes in the distribution of the abatement actions. This is also one of the first empirical assessments of a functioning trading program for water quality in the presence of an outside carbon market.

Many caveats regarding the assumption for the conceptual model, the water quality process and data availability underlie my empirical estimation. Moreover, the approaches presented here are simplified versions of any actual water quality program.

First, for the conceptual model, I consider that the farmers are risk neutral and minimize their costs (maximize their profit). Future possible work can incorporate a more elaborate and complete behavior response that considers the farmers risk behavior in adopting different abatement actions. Another note of caution arises from the assumption that the performance standard and trading outcomes achieves all gains from trade. However, as many authors have noted, when factors like the type of trading (sequential or bilateral), transaction costs or nonmonetary preferences are taken into account some of the gains of an efficient trade may not occur. Second, the efficiency results for the proposed policies are for two specific watersheds, two specific pollutants, and a given set of conservation practices. Differences in any of these aspects have the potential to generate quite different efficiency findings. A third caveat relates to the ability of SWAT and EPIC to mimic the environmental processes occurring in the watersheds. My empirical findings are all conditional on the calibration of the two watersheds within these two models.

One of the possible future extensions of my work include adapting the point procedure by bringing cropping choices into the point coefficients system. In this case, crop rotations such as corn-corn would receive negative points while crop rotations as corn-soybean would be rewarded with positive points. Another extension is considering the case of multiple pollutant markets, but considering a single set of point coefficients instead of the two sets. It has been shown that there is a time lag between the moment the emissions leave the field and reach the water, especially for pollutants such as phosphorus. In this case, the point system can be extended by creating temporal markets, with each market having its own time-dependent point coefficients.

In spite of these caveats, I believe that these should not hamper the consideration of proxies, such as the point coefficients, as efficient tools in implementing incentive-based programs designed for improving water quality in the agricultural watershed.

APPENDIX A. AN EMPIRICAL APPROACH FOR ESTIMATING THE VECTOR OF POINT COEFFICIENTS: A TECHNICAL APPROACH

Before presenting the approach used in obtaining the point coefficients, a brief discussion of the SWAT outputs offers a better understanding on the logic behind the estimation of point coefficients. In SWAT, a watershed is delineated into several subbasins, with each subbasin being delineated further into several hydrological units which can be interpreted as fields. SWAT computes the ambient emissions at the each subbasin outlet as a function of the component fields' emissions. Next, the ambient emissions from each subbasin are routed into a nonlinear and nonseparable way to determine the ambient water quality at the main outlet. Hence, SWAT provides emissions outputs at the field, the subbasin, and the main watershed level.

Step 1: Estimate a set of point coefficients that vary by subbasin using the watershed level output.

In this step, I estimate a set of point coefficients that is subbasin specific, implying that: (a) all the edge-of field reductions from a subbasin can be weighted by the same delivery coefficient, (b) a given abatement actions has the same edge-of-field impact in that subbasin (i.e., there is not nonseparability in $r_i(x)$ at the subbasin level). Assuming that in a watershed there are S subbasins and J abatement actions, $A\ (x)$ can be rewritten as:

$$A(x) \cong \sum_i^N d_{Ai} r_i(x_i) = \sum_s^S \sum_{i \in s}^{Ns} d_s r_{si}(x_{si}) = \sum_s^S d_S A_s(x), \qquad (1)$$

where A is a vector $Mx1$ of abatement levels measured at the watershed's outlet, s is an index for the subbasins, $\mathrm{d_s}$ is the delivery coefficient of subbasin s, Ns the total number of fields in the s^{th} subbasin, and $A_s(x) = \sum_{i \in s}^{Ns} r_{si}(x_{si})$ sums up the total abatement realized at the level of subbasin s. The term sum $\sum_s d_S A_s(x)$ represents the total emission reductions corresponding to subbasin s^{th} measured at the watershed exit. Next, the subbasin reductions are approximated as the sum of the total area allocated to the abatement action j, X_{sj} ,weighted by a factor w_{js},

$$A_s(x) = \sum_j^J w_{sj} \sum_{i \in s}^{Ns} x_{sij} = \sum_j^J w_{sj} X_{sj}. \qquad (2)$$

It should be emphasized that w_{js} measures the efficiency of practice j at the s^{th} subbasin and has the same value for all fields in that subbasin. By combining equations (2) and (3), and by defining the product $a_{sj} = d_S w_{sj}$, the abatement function can be written as:

$$A(x) \cong \sum_s^S \sum_{i \in s}^{Ns} d_S \sum_j w_{sj} X_{sj} = \sum_s^S \sum_j^J \underbrace{d_S w_{sj}}_{a_{sj}} X_{sj} = \sum_s^{NS} \sum_j^J a_{sj} * X_{sj} = \boldsymbol{X} * \boldsymbol{a}, \qquad (3)$$

where X is $Mx(SXJ)$ matrix, with each element $X(i,j)$ representing the area allocated to abatement action j, in the subbasin s. Finally, the vector of point coefficients, a, is obtained via regression:

$$A(x) = \sum_s^{NS} \sum_j^J a_{sj} * X_{sj} + \varepsilon \quad \varepsilon \sim N(0,1) . \qquad (4)$$

The point coefficients estimates, $\hat{a}_{sj}$, obtained above includes information on the delivery coefficients by definition. They can be interpreted as the marginal impact of abatement action j on the total level of abatement given that it is implemented by a field located in the subbasin s.

The above estimation considers that, within a given subbasin, an abatement action has the same impact regardless the location in a subbasin. For an accurate representation of the true abatement function, field-specific point coefficients are indicated. Next, using SWAT data obtained at field level, I estimate a set of point coefficients for each field.

Step 2: Estimate a set of point-coefficients for each field.

Using the output available for each field, the reduced emissions for each field, $r_i\,(x)$ can be written as

$$r_i\,(x_i) = \sum_{j=1}^{J} w_{ij}x_{ij} + \varepsilon_i \quad \varepsilon_i \sim N(0{,}1) \quad \forall i = 1, \ldots N \ . \tag{5}$$

This step implies running a regression and obtaining a set of J coefficients for each field. By running a regression for each field, the field characteristic are taken into account.

Let, $\widehat{w}_{sf}$ be the point coefficient estimate for abatement action j, given field i. By aggregating at the s^{th} subbasin level and assuming that the individual edge-of-field reduced emission within the same subbasin has the same impact on the abatement level at subbasin level, I retrieve the total reduced emissions at the subbasin level:

$$A_s(r) = \sum_{si}^{Ns} r_i\,(x) = \sum_{si}^{Ns} \sum_{j}^{J} \widehat{w}_{sij}\, x_{sij} \ . \tag{6}$$

Next, for a given subbasin and each abatement action, I compute a weighted average of the estimated point coefficients, $\overline{w}_{sj}$, where the weight is given by the field's area, ar_{sij},

$$\overline{w}_{sj} \ = \frac{\sum_{si} \widehat{w}_{sij}\, ar_{sij}}{ar_s} \qquad \forall\, j \in \ s \ , \tag{7}$$

where ar_{si} are of field i in subbasin s; and ar_s total area for subbasin s. Given the above notation, equation (7) is becomes

$$A_s(x) = \sum_j \overline{w}_{sj} \ \sum_{si} x_{sij} = \sum_j \overline{w}_{sj}\, X_{sj} \ , \tag{8}$$

where $X_{sj} = \sum_{si} x_{sij}$.

Equations (8) and (3) are approximations of the same total reduced emissions in a given subbasin. Thus, for a given abatement action j and a given subbasin s, the following should hold on average : $\overline{w}_{sj} \cong w_{sj}$ (i.e., the subbasin-level point coefficients should be an average measure of the field level point coefficients).

Step 3: Obtain the delivery coefficients.

The abatement function can be retrieved by aggregating over all subbasins:

$$A(X) = \sum_s^S d_{As}\, A_s \;= \sum_s^S d_{As} \sum_j^J \overline{w}_{sj}\, X_{sj} = \sum_s^S \sum_j^J d_{As}\overline{w}_{sj}\, X_{sj} = \sum_s^S \sum_j^J a_{sj}\, X_{sj}\ . \tag{9}$$

Equation (4) is equivalent to equation (3). Thus, by comparing the results with the results obtained in step 1, the following relation should hold: $d_{As}\overline{w}_{sj} \cong \hat{a}_{sj}$ (i.e., the impact of the abatement action on the overall abatement should be equal to its field level impact weighted by the delivery coefficient). All the elements but the d_{As}'s are known in equation (4) . In this case, delivery coefficients can be obtained as $d_{As} \cong \frac{\hat{a}_{sj}}{\overline{w}_{sj}}$ (i.e., the ratio of the subbasin point-specific coefficients to the weighted average of field-specific point coefficients).

Finally, obtaining the delivery coefficients requires one more level of aggregation because there are j abatement actions and only one delivery coefficient for each subbasin. Furthermore, since both $\hat{a}_{sj}$ and $\overline{w}_{sj}$ being obtained via ordinary least square have normal distributions, averaging over the J ratios $\hat{a}_{sj}/\overline{w}_{sj}$ has a Cauchy distribution. As the Cauchy distribution does not have finite moments of any order, the average of J's ratio has no meaning. Instead, I instead rely on the median measure

$$\bar{d}_{As} = median \left\{\frac{\hat{a}_{sj}}{\overline{w}_{sj}}\right\}_j^J . \tag{10}$$

Step 4: Using the delivery coefficients to obtain the final set of point coefficients.

For a given field and a given abatement action, a more refined set of point coefficients can be obtained by multiplying the point coefficients obtained in the second step, $\widehat{w}_{sij}$, with the delivery coefficients estimated above:

$$\tilde{a}_{sij} = \bar{d}_{As}\widehat{w}_{sij}, \tag{11}$$

where $\tilde{a}_{sij}$ is the point coefficient for the abatement action j, field i, subbasin s .

Alternative Approach to Computing the Delivery Coefficients

In the previous section, I show how to obtain the delivery coefficients and the field-specific point coefficients by using HRU and watershed-level SWAT outputs. This approach requires using data at a very fine scale. Alternatively, both the delivery coefficients and the point coefficient can be obtained using data at a less refined scale such as watershed and subbasin, or watershed only.

Obtaining the delivery coefficients using SWAT watershed and subbasin level outputs

This is also a multistep approach, with the first step being identical with the one described in section 3.2.1, where the estimated point coefficients are subbasin specific and include the delivery coefficients.

Step 1: $A(x) \cong \sum_s d_s A_s(x) = \sum_s d_s \sum_j w_{js} X_{js} = \sum_j \sum_s a_{js} X_{js} = Xa$ (12)

where $a_{js} = d_s w_{js}$.

The first step implies the estimation of a set of point coefficients for each subbasin using a single regression. Let $\hat{a}$ be the set of point coefficients, and $\hat{a}^s$ be the subset of point coefficients for subbasin s.

Step 2: In the second step, using the subbasin SWAT outputs, I estimate a set of point coefficients for each subbasin. The point coefficients can be interpreted as the marginal impact of an abatement level on the total abatement measured at subbasin level. The abatement function for subbasin s, $A_s(x)$, can be written as a linear combination of subbasin specific weights w_j^s , and the area allocated to a particular abatement action:

$$A_s(x) \cong \sum_j w_{js} X_{js} = Xw \qquad \forall s = 1, \dots, S . \qquad (13)$$

The second step implies the estimation of a number of regressions equal to the number of subbasins in a watershed, S. Let $\hat{w}^s$, be the set of point coefficients obtained for the s subbasin.

Step 3: By combining equations (7) and (8), for a given subbasin s, the delivery coefficients can be obtained as[46]: $\hat{d}_s = median\{\hat{a}^s / \widehat{w}^s\}$

Step 4: Next, the delivery coefficients are applied to let $\widehat{w}^s$ obtain the final subbasin specific point coefficients: $\tilde{a}_{js} = \hat{d}_s \widehat{w}_j{}^s \quad \forall j, \forall s$.

Obtaining the delivery coefficients using SWAT watershed only outputs

This approach is also a multistep procedure, with the first step being described before.

Step 1: Obtain via regression subbasin-specific point coefficients. Let the $\widehat{a}$ be the set of point coefficients, and $\widehat{a}^s$ be the subset of point coefficients for subbasin s:

$$\mathrm{A(x)} \cong \sum_{\mathrm{j}} \sum_{\mathrm{s}} \mathrm{a_{js}} \mathrm{X_{js}} = \mathrm{Xa} . \quad (14)$$

Step 2: A unique set of point coefficients is obtained for the entire watershed. The abatement function is approximated as a linear combination of weights, a^W, assigned to each abatement action, with the weights being the same for each field, and the area allocated to that abatement action:

$$A(x) \cong \sum_j a^W X_j . \quad (15)$$

[46] The median is used instead of the average because $\hat{a}^s / \widehat{w}^s$ is the ratio of two normal distributions.

APPENDIX B. TABLES AND FIGURES

Tables and Figures for Chapter 3

Table A-1. Boone River Watershed: Point value estimates descriptive statistics, nitrogen

Abatement practice	Mean	Min	Max	Std.dev.
Baseline	0.00	-4.15	0.00	0.08
No till	2.65	-11.67	14.88	2.41
Cover crops	2.72	-0.25	22.21	1.77
No till, Cover Crops	4.86	-0.15	24.09	3.23
Red.fertilizer	0.77	-12.09	11.60	0.81
Red.Fert, No till	3.44	-2.53	20.89	2.63
Red.Fert., Cover Crops	3.38	-20.94	27.81	2.18
Red.Fert., No till, CC	5.52	-0.04	36.12	3.51
CRP	8.40	0.18	85.29	5.51
Rsquare	0.99	0.00	1.00	0.10

Table A-2. Boone River Watershed: Point value estimates descriptive statistics, phosphorus

Abatement practice	Mean	Min	Max	Std.dev.
Baseline	0.00	-0.05	0.01	0.00
No till	0.21	-0.05	1.74	0.21
Cover crops	0.12	-0.08	0.75	0.11
No till, Cover Crops	0.20	-0.08	1.73	0.22
Red.fertilizer	0.00	-0.10	0.07	0.00
Red.Fert, No till	0.21	-0.05	1.75	0.21
Red.Fert., Cover Crops	0.12	-0.07	0.75	0.11
Red.Fert., No till, CC	0.21	-0.07	1.75	0.22
CRP	0.33	0.00	2.49	0.28
Rsquare	0.98	0.09	1.00	0.08

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Table A-3. Raccoon River Watershed: Point values estimates descriptive statistics, nitrogen

Abatement practice	**Mean**	**Min**	**Max**	**Std.dev**
Baseline	0.00	-0.06	0.12	0.01
No till	1.40	-0.31	11.42	1.85
Cover crops	2.22	-0.08	13.30	2.16
No till, Cover Crops	2.83	-0.04	17.17	2.87
Red.fertilizer	0.60	-0.05	2.59	0.67
Red.Fert, No till	1.97	-0.29	11.81	2.15
Red.Fert., Cover Crops	2.70	-0.04	13.70	2.45
Red.Fert., No till, CC	3.33	-0.04	17.17	3.15
CRP	6.41	-1.17	25.96	5.17
Rsquare	0.92	0.00	1.00	0.23

Table A-4. Raccoon River Watershed:Point value Estimates descriptive statistics, phosphorus

Abatement practice	Mean	Min	Max	Std.dev
Baseline	0.00	-0.01	0.01	0.00
No till	0.13	0.00	1.16	0.18
Cover crops	0.12	0.00	1.32	0.19
No till, Cover Crops	0.17	0.00	1.66	0.25
Red.fertilizer	0.00	-0.08	0.08	0.01
Red.Fert, No till	0.13	0.00	1.16	0.18
Red.Fert., Cover Crops	0.12	0.00	1.32	0.19
Red.Fert., No till, CC	0.17	0.00	1.66	0.25
CRP	0.23	-0.13	2.30	0.33
Rsquare	0.91	0.07	1.00	0.00

Table A-5. Boone River Watershed: Subbasin specific point coefficients, nitrogen

Location	Abatement practices: No till	Cover crops	No till, Cover Crops	Red. fert	Red.Fert, No till	Red.Fert Cover Crops	Red.FertNo till, CC	CRP
Subbasin 1	3.45*	1.87*	5.35*	0.22	4.51*	2.7*	6.15*	10.26*
Subbasin 2	3.96*	2.42*	5.34*	0.74***	4.49*	2.49*	5.92*	9.55*
Subbasin 3	3.55*	1.9*	4.59*	0.32	3.87*	2.56*	4.84*	7.49*
Subbasin 4	2.5**	2.82**	4.53*	0.34	3.82*	2.09**	4.15*	5.86*
Subbasin 5	2.03*	1.98*	3.97*	0.62**	2.61*	2.49*	4.7*	6.42*
Subbasin 6	2.24*	2.15*	4.44*	0.74**	2.45*	2.9*	5.05*	7*
Subbasin 7	6.33*	3.35**	6.51*	1.32	6.61*	3.26**	7.81*	10.13*
Subbasin 8	2.83*	3.14*	5.19*	0.78	3.87*	3.17*	5.16*	7.44*
Subbasin 9	0.86**	1.03*	2*	0.34	1.06*	1.77*	2.33*	4.33*
Subbasin 10	1.67*	2.31*	2.99*	0.51	2.23*	2.7*	4.47*	5.91*
Subbasin 11	2.08*	1.8*	3.39*	0.13	3.21*	2.83*	5.32*	7.29*
Subbasin 12	2.94*	2.33*	4.04*	0.06	3.24*	2.97*	5.13*	6.98*
Subbasin 13	2.17*	2.47*	3.53*	0.29	2.33*	2.46*	3.41*	5.83*
Subbasin 14	2.04*	2.04*	3.23*	1.02**	2.53*	3.19*	3.98*	5.66*
Subbasin 15	2.36*	1.68*	3.96*	0.09	2.67*	1.72*	3.84*	5.09*
Subbasin 16	1.05**	1.12*	2.45*	0.23	1.55*	1.52*	2.76*	3.87*
Subbasin 17	1.77*	1.9*	3.24*	1.03**	2.06*	2.45*	3.57*	4.43*
Subbasin 18	2.88*	3.09*	4.09*	1.5**	3.42*	3.43*	4.82*	7.13*
Subbasin 19	3.1*	1.71**	4.57*	-0.5	2.76*	2.52*	4.52*	7.2*
Subbasin 20	1.91*	2.58*	4.27*	0.85**	2.72*	2.5*	4.48*	6.59*
Subbasin 21	4.13*	2.83*	6.71*	0.92**	4.92*	3.58*	7*	12.18*
Subbasin 22	2.33*	3.11*	4.39*	0.62	3.69*	3.51*	5.37*	8.5*
Subbasin 23	2.58*	3.05*	4.43*	0.43	3.64*	3.48*	5.76*	7.9*
Subbasin24	2.14*	2.06*	4.37*	0.85***	3.47*	3.08*	4.75*	7.77*
Subbasin25	1.52**	1.65**	4.51*	0.15	2.49*	1.85**	4.18*	6.64*
Subbasin 26	3.34*	3.75*	8.14*	0.69	5.57*	4.79*	7.35*	12.7*
Subbasin 27	5.57**	5.02**	9.17*	1.47	3.91***	3.22	9.09*	12.2*
Subbasin 28	1.92*	2*	4.26*	-0.2	2.54*	1.78*	3.77*	5.41*
Subbasin 29	3.49*	2.7*	4.55*	0.52	3.75*	4.11*	5.25*	9.28*
Subbasin 30	3.76*	2.2*	5.37*	1.26**	3.62*	3.4*	5.51*	8.28*

Note: Asterisk (*), double asterisk (**), and triple asterisk (***) denote significance at 1%, 5%, and 10%,

Table A-6. Boone River Watershed: Subbasin specific point coefficients, phosphorus

	Abatement practices							
Location	**No till**	**Cover crops**	**No till, Cover Crops**	**Red.fert**	**Red.Fert. No till**	**Red.Fert Cover Crops**	**Red.Fert No till, CC**	**CRP**
Subbasin 1	0.18*	0.11*	0.15*	0	0.19*	0.09*	0.15*	0.42*
Subbasin 2	0.22*	0.12*	0.19*	0.01	0.22*	0.12*	0.2*	0.43*
Subbasin 3	0.19*	0.11*	0.16*	0	0.19*	0.13*	0.16*	0.37*
Subbasin 4	0.23*	0.19*	0.24*	0	0.26*	0.15**	0.17*	0.31*
Subbasin 5	0.13*	0.09*	0.11*	0.02	0.16*	0.1*	0.12*	0.33*
Subbasin 6	0.18*	0.09*	0.14*	0	0.15*	0.12*	0.15*	0.34*
Subbasin 7	0.33*	0.17**	0.22*	0.05	0.25*	0.12**	0.2*	0.36*
Subbasin 8	0.1**	0.13*	0.15*	-0.02	0.15*	0.13*	0.13*	0.25*
Subbasin 9	0.04**	0.03**	0.02***	-0.01	0.03**	0.05*	0.03**	0.15*
Subbasin 10	0.1*	0.12*	0.07*	0.02	0.1*	0.11*	0.12*	0.23*
Subbasin 11	0.05**	0.05***	0.02	-0.05***	0.08**	0.1*	0.1*	0.19*
Subbasin 12	0.17*	0.1*	0.12*	-0.04**	0.15*	0.12*	0.15*	0.26*
Subbasin 13	0.11*	0.09*	0.08*	-0.04**	0.08*	0.1*	0.06*	0.19*
Subbasin 14	0.11*	0.07**	0.06**	0.01	0.11*	0.13*	0.09*	0.2*
Subbasin 15	0.09*	0.08*	0.08*	0.01	0.08*	0.07*	0.08*	0.15*
Subbasin 16	0.05**	0.05**	0.03**	-0.02	0.05**	0.04**	0.03***	0.14*
Subbasin 17	0.1*	0.1*	0.08*	0.02	0.08*	0.1*	0.08*	0.18*
Subbasin 18	0.19*	0.13*	0.15*	0.02	0.18*	0.13*	0.13*	0.34*
Subbasin 19	0.19*	0.14*	0.16*	0.01	0.18*	0.15*	0.18*	0.36*
Subbasin 20	0.08*	0.11*	0.09*	0.01	0.12*	0.09*	0.08*	0.21*
Subbasin 21	0.23*	0.17*	0.25*	0.01	0.23*	0.16*	0.24*	0.58*
Subbasin 22	0.11*	0.14*	0.12*	-0.02	0.15*	0.1*	0.14*	0.37*
Subbasin 23	0.22*	0.19*	0.21*	0.02	0.25*	0.2*	0.27*	0.41*
Subbasin 24	0.19*	0.14*	0.17*	-0.01	0.21*	0.12*	0.17*	0.44*
Subbasin 25	0.16*	0.1*	0.17*	-0.02	0.17*	0.09**	0.13*	0.35*
Subbasin 26	0.28*	0.21*	0.33*	-0.01	0.3*	0.23*	0.3*	0.61*
Subbasin 27	0.32**	0.16	0.34**	0.02	0.22***	0.09	0.32**	0.52*
Subbasin 28	0.13*	0.11*	0.14*	-0.04	0.13*	0.07**	0.1*	0.28*
Subbasin 29	0.25*	0.18*	0.25*	0.01	0.23*	0.18*	0.25*	0.51*
Subbasin 30	0.36*	0.17*	0.31*	0.05***	0.33*	0.19*	0.32*	0.58*

Note: Asterisk (*), double asterisk (**), and triple asterisk (***) denote significance at 1%, 5%, and 10%,

Table A-7. Raccoon River Watershed: Subbasin specific point coefficients, nitrogen

	Abatement practices							
Location	**No till**	**Cover crops**	**No till, Cover Crops**	**Red.fert ilizer**	**Red.Fer t, No till**	**Red.Fer t., Cover Crops**	**Red.Fer t., No till, CC**	**CRP**
Subbasin 1	0.29*	1.42**	1.46**	-0.16*	0.83**	2.19**	2.11**	5.36**
Subbasin 2	0.6**	3.05**	3.15**	0.44**	0.77**	3.35**	3.44**	10.16**
Subbasin 3	0.33*	1.78**	1.97**	0.49**	0.35*	1.95**	2.01**	5.65**
Subbasin 4	0.14*	2.33**	2.24**	1.12**	0.96**	2.93**	3.1**	5.63**
Subbasin 5	0.17*	1.6**	1.49**	-0.03*	0.26*	1.1**	1.21**	3.02**
Subbasin 6	0.32*	1.67**	2.43**	1**	1.37**	2.86**	2.75**	9.11**
Subbasin 7	0.08*	1.76**	2.2**	0.34*	-0.02*	2.67**	2.87**	9.84**
Subbasin 8	0.82*	2.45**	2.44**	0.65*	0.8*	3.15**	2.73**	7.86**
Subbasin 9	-0.94*	1.86**	1.87**	0.22*	0.52*	2.01**	2.56**	4.24**
Subbasin 10	0.11*	1.52**	1.91**	0.24*	0.22*	1.92**	2.54**	4.15**
Subbasin 11	0.16*	2.56**	2.61**	0.12*	0.38**	2.64**	2.88**	8.36**
Subbasin 12	0.81**	3.17**	3.3**	0.81**	1.91**	3.38**	3.79**	10.07**
Subbasin 13	-0.13*	2.18**	2.18**	0.74**	1.18**	2.93**	3.15**	5.31**
Subbasin 14	-0.2*	0.97*	1.45**	0.64*	0.83*	2.07**	2.27**	3.85**
Subbasin 15	0.8*	2.74**	2.88**	0.68*	0.82*	3.33**	3.3**	8.81**
Subbasin 16	0.3*	1.87**	2.04**	-0.1*	0.26*	2.15**	2.31**	7.04**
Subbasin 17	0.39*	1.21**	1.79**	0.2*	1.03**	2.14**	2.35**	7.73**
Subbasin 18	0.32*	2.02**	2.25**	0.29*	0.49*	2.31**	2.33**	6.27**
Subbasin 19	0.1*	1.72**	2.1**	0.06*	0.36*	1.75**	2.26**	5.37**
Subbasin 20	0.43*	2.1**	2.36**	0.91**	0.85**	2.29**	2.15**	5.65**
Subbasin 21	-0.02*	1.12**	1.65**	0.4*	0.75**	2.35**	1.93**	5.2**
Subbasin 22	0.37**	2.26**	2.62**	0.91**	1.72**	3.14**	3.46**	8.92**
Subbasin 23	0.04*	1.88**	2.09**	0.71**	0.69**	2.43**	2.71**	8.97**
Subbasin 24	0.76**	1.47**	2.24**	-0.03*	0.8**	1.74**	2.36**	5.73**
Subbasin 25	0.24*	1.75**	1.88**	0.63**	0.66**	2.27**	2.57**	6.21**
Subbasin 26	-0.45*	1.05**	0.77**	0.62*	0.31*	1.38**	1.78**	4.62**
Subbasin 27	1.27*	2.26**	2.33**	0.73*	1.41*	3.35**	3.83**	7.22**
Subbasin 28	1.55**	2.93**	3.49**	0.56**	1.73**	3.4**	3.94**	9.05**
Subbasin 29	0.84*	2.18**	2.55**	1**	1.16**	3.18**	3.4**	8.76**
Subbasin 30	0.36*	2.76**	2.68**	1.6**	1.76**	3.45**	3.63**	6.35**
Subbasin 31	0.12*	2.43**	2.5**	1.43**	1.6**	3.28**	3.36**	6.69**
Subbasin 32	0.99**	2.33**	2.91**	0.25*	1.33**	2.52**	3.14**	9.83**
Subbasin 33	0.29*	2.36**	2.3**	0.48*	0.95**	2.69**	3.38**	6.94**
Subbasin 34	0.36*	2.31**	2.25**	0.45*	0.53*	2.58**	2.82**	6.67**
Subbasin 35	0.24*	2.7**	2.71**	0.74*	1.05*	3.58**	3.81**	8.7**
Subbasin 36	0.48**	1.75**	2.25**	1.76**	2.09**	3.32**	3.7**	8.29**
Subbasin 37	0.68*	1.14**	1.17**	1.25**	2.57**	2.85**	2.77**	6.59**
Subbasin 38	0.32*	2.25**	1.82**	-0.04*	0.87**	2.75**	2.78**	8.79**
Subbasin 39	0.36*	2.94**	2.77**	0.63*	1.09**	3.35**	3**	10.09**

Subbasin 40	0.01*	2.42**	2.92**	0.53*	0.95*	3.25**	3.33**	11.12**
Subbasin 41	0.9**	2.79**	3.3**	0.69**	1.45**	3.61**	3.97**	8.73**
Subbasin 42	0.79**	2.79**	2.63**	0.58**	1.23**	3.11**	3.41**	9.32**
Subbasin 43	0.66**	2.68**	2.93**	0.53**	1.2**	3.09**	3.57**	6.79**
Subbasin 44	-0.02*	2.19**	2.64**	2.01**	2.09**	3.64**	4.56**	9.44**
Subbasin 45	3.7**	4.93**	6.14**	1.3**	4.98**	6.28**	7.74**	13.87**
Subbasin 46	0.49**	2.49**	2.57**	0.43**	1.15**	2.8**	2.8**	8.53**
Subbasin 47	1.21**	3.26**	3.81**	1.77**	2.57**	4.14**	5.25**	9.78**
Subbasin 48	1.05**	2.39**	2.79**	1.12**	1.95**	3.18**	3.35**	6.29**
Subbasin 49	1.06**	2.66**	3.19**	1.71**	2.51**	4.2**	4.28**	9.03**
Subbasin 50	1.23**	2.57**	3.43**	1.57**	3.12**	3.57**	4.83**	8.77**
Subbasin 51	1.78**	2.46**	3.52**	1.22**	2.55**	3.55**	3.79**	7.56**
Subbasin 52	4.55**	3.1**	6.79**	1.21*	5.02**	5.37**	7.26**	13.02**
Subbasin 53	2.09*	-0.91*	1.97*	0.44*	1.97*	1.36*	3.26*	10.05**
Subbasin 54	1.56**	3.51**	3.8**	1.24**	3.13**	4.93**	5.21**	10.48**
Subbasin 55	1.76**	2.25**	3.64**	1.35**	2.09**	3.42**	4.62**	10.33**
Subbasin 56	1.04**	2.51**	3.52**	1.01**	1.95**	3.69**	4.2**	9**
Subbasin 57	1.41**	3.39**	3.26**	1.27**	1.69**	3.26**	3.27**	8.97**
Subbasin 58	0.72**	2.15**	3**	1.22**	2.46**	3.95**	4.5**	8.84**
Subbasin 59	2.1**	3.58**	5.12**	2.35**	3.62**	4.73**	5.86**	11.36**
Subbasin 60	1.83**	3.38**	3.74**	1.14**	2.55**	4.06**	3.93**	9.17**
Subbasin 61	5.35**	4.41**	7.32**	1.51*	5.57**	4.88**	8.22**	12.74**
Subbasin 62	1.56**	3.04**	3.88**	1.92**	3.58**	4.54**	5.73**	10.99**
Subbasin 63	1.01**	2.68**	2.87**	1.57**	2.2**	3.5**	4.61**	10.14**
Subbasin 64	4.39**	4.8**	7.14**	0.47*	4.48**	5.34**	7.76**	15.15**
Subbasin 65	4.43**	4.5**	6.47**	1.63*	5.62**	5.2**	6.48**	12.01**
Subbasin 66	1.46**	2.92**	3.32**	2.32**	2.65**	4.19**	4.48**	8.08**
Subbasin 67	4.64**	4.32**	8.56**	3.18**	6.89**	5.68**	9.43**	11.72**
Subbasin 68	0.38*	2.3**	2.48**	1.27**	2.28**	3.78**	4.2**	9.94**
Subbasin 69	0.93*	2.07**	2.52**	1.34**	2.27**	3.09**	3.78**	9**
Subbasin 70	3.7**	5.6**	7.18**	0.59**	4.61**	6.51**	7.92**	13.74**
Subbasin 71	2.81**	2.5*	4.37**	-0.78*	3.33**	2.85**	6.37**	9.28**
Subbasin 72	1.72**	2.6**	3.35**	1.87**	3.8**	4.54**	5.15**	11.68**
Subbasin 73	5.47**	5**	8.09**	2.83**	5.7**	6.2**	8.49**	11.93**
Subbasin 74	1.85**	4.2**	5.19**	0.51*	4.72**	5.51**	5.82**	11.04**
Subbasin 75	3.98**	6.19**	8.48**	0.4*	4.8**	7.33**	8.94**	15.43**
Subbasin 76	5**	7.19**	8.79**	2.59**	6.09**	7.33**	8.25**	14.9**
Subbasin 77	-0.07*	0.24*	-0.81*	0.58*	-0.61*	1.37*	0.3*	-0.81*
Subbasin 78	1.87**	3.49**	4.73**	1.57**	3.44**	5.03**	6.32**	10.99**
Subbasin 79	3.66**	5.23**	6.51**	3.32**	5.31**	6.82**	8.71**	15.47**
Subbasin 80	4.63**	8.03**	8.49**	1.35*	5.87**	8.45**	9.68**	14.91**
Subbasin 81	2.49**	3.41**	4.82**	2.03**	4.52**	5.07**	6.94**	13.88**
Subbasin 82	3.75**	7.33**	8.48**	1.88*	5.36**	8.03**	9.83**	13**
Subbasin 83	4.91**	9.05**	9.22**	0.86*	6.8**	8.45**	9.95**	14.17**

Subbasin 84	0.3*	0.18*	-0.47*	0.46*	0.53*	-0.47*	0.66*	-1.16*
Subbasin 85	-0.85*	-1.09*	-1.49*	-1.57*	0.05*	-0.73*	-1.05*	-1.65*
Subbasin 86	0.18*	2.72**	2.64**	0.15*	0.78**	2.83**	2.96**	9.35**
Subbasin 87	-2.15*	-1.12*	-0.41*	0.74*	1.02*	4.44*	3.39*	7.68**
Subbasin 88	-0.13*	1.77**	1.96**	0.53*	0.55*	2.24**	2.47**	3.88**
Subbasin 89	0.18*	2.36**	2.38**	0.92**	1.31**	3.09**	3.36**	6.12**
Subbasin 90	-0.27*	1.63**	1.64**	0.71**	0.83**	2.28**	2.42**	4.67**
Subbasin 91	0.03*	2.09**	1.74**	1.11**	0.73**	2.42**	3.02**	6.74**
Subbasin 92	0.34*	2.02**	2.52**	1.96**	2.08**	3.38**	3.61**	8.5**
Subbasin 93	-0.61*	1.72**	1.71**	0.46*	0.61**	2.25**	2.63**	5.54**
Subbasin 94	0.33**	2.6**	2.45**	0.83**	1.36**	3.44**	3.4**	6.16**
Subbasin 95	3.51**	4.55**	5.99**	0.76**	4.42**	5.28**	6.93**	12.93**
Subbasin 96	3.16**	3.82**	5.4**	0.47*	3.36**	4.56**	5.56**	11.55**
Subbasin 97	3.57**	5.13**	6.13**	0.9**	4.25**	5.43**	6.78**	13.07**
Subbasin 98	4.44**	6.55**	8.06**	1.15**	5.78**	7.4**	9.5**	15.89**
Subbasin 99	4.6**	5.93**	8.15**	0.64**	4.88**	6.49**	8.28**	14.69**
Subbasin 100	0.21*	2.82**	2.68**	1.06**	1.49**	3.4**	3.32**	9.6**
Subbasin 101	2.39**	4.34**	4.97**	1.75**	4.21**	5.54**	6.58**	11.59**
Subbasin 102	1.47**	2.55**	3.35**	1.55**	2.79**	3.68**	4.57**	9.3**
Subbasin 103	1**	2.44**	3.15**	1.44**	2.47**	3.99**	4.44**	8.94**
Subbasin 104	0.93**	1.55**	2.05**	1.39**	2.51**	2.85**	3.36**	7.71**
Subbasin 105	1.48**	2.23**	3.92**	1.32**	3.29**	4.11**	4.71**	10.9**
Subbasin 106	4.44**	5.85**	7.68**	2.46**	5**	6.47**	8.23**	12.89**
Subbasin 107	13.37*	13.67*	11.3*	1.13*	9.59*	14.85**	10.42*	17.65**
Subbasin 108	7.03*	28.68*	10.24*	7.58*	23.26*	8.77*	26.58*	18.34*
Subbasin 109	3.26**	4.01**	5.63**	1.34**	5.03**	5.62**	7.37**	14.52**
Subbasin 110	0.14*	-0.03*	0.2*	0.13*	0.33*	-0.06*	0.06*	0.08*
Subbasin 111	-0.01*	0.92*	-0.42*	0.59*	-0.27*	0.06*	-0.29*	-1.15*
Subbasin 112	2.77**	3.71**	5.46**	1.62**	4.34**	4.6**	6.49**	11.94**

Note: Asterisk (*), double asterisk (**), and triple asterisk (***) denote significance at 1%, 5%, and 10%,

Table A-8. Raccoon River Watershed: Subbasin specific point coefficients, phosphorus

	Abatement practices							
Location	**No till**	**Cover crops**	**No till, Cover Crops**	**Red.fertilizer**	**Red.Fert , No till**	**Red.Fert Cover Crops**	**Red.Fert .No till, CC**	**CRP**
Subbasin 1	0.06*	0.04**	0.05*	0	0.05*	0.05*	0.06*	0.14*
Subbasin 2	0.06*	0.06*	0.07*	0	0.06*	0.05*	0.07*	0.14*
Subbasin 3	0.02**	0.02**	0.03*	0	0.02**	0.03**	0.03*	0.11*
Subbasin 4	0.03*	0.02**	0.03*	0.01	0.02**	0.03*	0.03*	0.06*
Subbasin 5	0.04**	0.05**	0.04**	0.03***	0.06*	0.03**	0.04**	0.1*
Subbasin 6	0.03***	0.01	0.05**	-0.01	0.02	0.02***	0.03**	0.1*
Subbasin 7	0.08*	0.06**	0.06**	0.02	0.05**	0.05**	0.08*	0.15*
Subbasin 8	0.05**	0.04**	0.06**	0.01	0.05**	0.06*	0.05**	0.12*
Subbasin 9	0	0.02***	0.02	-0.01	0.02	0.01	0.02	0.05*
Subbasin 10	0.02	0.02	0.03**	-0.02	0.01	0	0.03**	0.06*
Subbasin 11	0.05*	0.05*	0.06*	-0.01	0.05*	0.05*	0.06*	0.12*
Subbasin 12	0.08*	0.1*	0.13*	0.01	0.07*	0.09*	0.11*	0.17*
Subbasin 13	0.02**	0.02**	0.03*	0	0.02**	0.02**	0.04*	0.06*
Subbasin 14	0	-0.02	0	0	0.03	0.01	0.01	0.04***
Subbasin 15	0.07*	0.06**	0.07**	0	0.08*	0.03***	0.09*	0.16*
Subbasin 16	0.05*	0.03*	0.05*	0	0.05*	0.04*	0.06*	0.14*
Subbasin 17	0.05*	0.03**	0.06*	-0.01	0.05*	0.04**	0.06*	0.15*
Subbasin 18	0.04**	0.03**	0.04**	0	0.05**	0.05*	0.06*	0.15*
Subbasin 19	0.03***	0.05**	0.06*	0	0.03***	0.03**	0.06*	0.15*
Subbasin 20	0.04**	0.05*	0.07*	0.02	0.05**	0.05*	0.04**	0.12*
Subbasin 21	0.01	0.02***	0.03**	0	0.01	0.03**	0.02***	0.07*
Subbasin 22	0.06*	0.05*	0.08*	0	0.07*	0.05*	0.08*	0.17*
Subbasin 23	0.05*	0.06*	0.06*	0.01	0.05*	0.04**	0.07*	0.15*
Subbasin 24	0.06*	0.03	0.07*	-0.01	0.05**	0.02	0.04**	0.13*
Subbasin 25	0.04*	0.03*	0.05*	0	0.04*	0.03*	0.05*	0.1*
Subbasin 26	0.03**	0.02	0.04**	0.01	0.04**	0.02	0.02	0.07*
Subbasin 27	0.12*	0.06***	0.1**	0.04	0.06**	0.07**	0.16*	0.16*
Subbasin 28	0.12*	0.09*	0.16*	0	0.12*	0.1*	0.15*	0.27*
Subbasin 29	0.08*	0.03***	0.11*	0	0.05**	0.04**	0.09*	0.2*
Subbasin 30	0.05*	0.06*	0.07*	0.01	0.07*	0.06*	0.06*	0.1*
Subbasin 31	0.05*	0.05*	0.06*	0	0.06*	0.05*	0.08*	0.12*
Subbasin 32	0.12*	0.08*	0.13*	0	0.11*	0.07*	0.13*	0.26*
Subbasin 33	0.06*	0.06*	0.07*	0	0.07*	0.06*	0.1*	0.15*
Subbasin 34	0.04*	0.05*	0.05*	-0.02	0.04*	0.05*	0.07*	0.12*
Subbasin 35	0.08**	0.11*	0.1*	0.01	0.09*	0.1*	0.13*	0.2*
Subbasin 36	0.07*	0.03*	0.07*	0	0.07*	0.03*	0.06*	0.12*
Subbasin 37	0.06*	0.01	0.03***	-0.01	0.06**	0.02	0.06*	0.12*
Subbasin 38	0.07*	0.1*	0.11*	-0.01	0.06*	0.09*	0.1*	0.17*
Subbasin 40	0.1*	0.13*	0.16*	-0.01	0.1*	0.13*	0.18*	0.24*

Subbasin 41	0.1*	0.1*	0.14*	-0.01	0.1*	0.1*	0.13*	0.22*
Subbasin 42	0.1*	0.11*	0.14*	0	0.1*	0.11*	0.13*	0.19*
Subbasin 43	0.09*	0.07*	0.1*	0	0.08*	0.08*	0.11*	0.17*
Subbasin 44	0.06**	0.09*	0.1*	0.03	0.11*	0.08*	0.09*	0.15*
Subbasin 45	0.34*	0.39*	0.5*	0.01	0.35*	0.41*	0.49*	0.63*
Subbasin 46	0.08*	0.1*	0.12*	0	0.07*	0.1*	0.11*	0.18*
Subbasin 47	0.11*	0.08*	0.15*	0	0.14*	0.09*	0.15*	0.22*
Subbasin 48	0.09*	0.07*	0.1*	0.02***	0.09*	0.07*	0.1*	0.17*
Subbasin 49	0.09*	0.06*	0.1*	0	0.1*	0.07*	0.09*	0.16*
Subbasin 50	0.12*	0.07*	0.13*	0.01	0.12*	0.08*	0.14*	0.2*
Subbasin 51	0.1*	0.07*	0.1*	-0.02	0.1*	0.06**	0.1*	0.2*
Subbasin 52	0.31*	0.2*	0.38*	0.04***	0.28*	0.24*	0.33*	0.52*
Subbasin 53	0.28**	0.17	0.17	0.09	0.18***	0.16	0.2***	0.45*
Subbasin 54	0.15*	0.12*	0.15*	0.02	0.16*	0.09*	0.15*	0.25*
Subbasin 55	0.15*	0.1*	0.16*	0.01	0.14*	0.08*	0.18*	0.3*
Subbasin 56	0.13*	0.08*	0.15*	0	0.12*	0.09*	0.14*	0.22*
Subbasin 57	0.13*	0.16*	0.18*	-0.01	0.12*	0.16*	0.17*	0.2*
Subbasin 58	0.14*	0.09*	0.17*	0	0.15*	0.09*	0.17*	0.24*
Subbasin 59	0.19*	0.11*	0.23*	0.01	0.19*	0.11*	0.21*	0.31*
Subbasin 60	0.16*	0.16*	0.22*	0.01	0.17*	0.15*	0.19*	0.3*
Subbasin 61	0.62*	0.46*	0.71*	0.04	0.62*	0.49*	0.76*	1.05*
Subbasin 62	0.17*	0.12*	0.21*	0	0.18*	0.11*	0.21*	0.28*
Subbasin 63	0.14*	0.09*	0.14*	0	0.13*	0.07*	0.16*	0.22*
Subbasin 64	0.45*	0.4*	0.6*	0	0.45*	0.42*	0.61*	0.85*
Subbasin 65	0.37*	0.31*	0.44*	0.05	0.34*	0.32*	0.42*	0.57*
Subbasin 66	0.11*	0.09*	0.16*	0.01	0.12*	0.12*	0.18*	0.21*
Subbasin 67	0.4*	0.35*	0.64*	0.12**	0.44*	0.38*	0.59*	0.81*
Subbasin 68	0.11*	0.08*	0.13*	-0.01	0.12*	0.08*	0.15*	0.22*
Subbasin 69	0.11*	0.07*	0.15*	0	0.13*	0.07*	0.14*	0.18*
Subbasin 70	0.45*	0.47*	0.63*	0.01	0.46*	0.5*	0.64*	0.85*
Subbasin 71	0.33*	0.17**	0.39*	0.04	0.34*	0.16**	0.38*	0.53*
Subbasin 72	0.16*	0.12*	0.19*	0	0.17*	0.12*	0.2*	0.28*
Subbasin 73	0.53*	0.48*	0.66*	0.08**	0.46*	0.49*	0.66*	0.91*
Subbasin 74	0.34*	0.39*	0.51*	-0.02	0.4*	0.35*	0.46*	0.62*
Subbasin 75	0.57*	0.54*	0.74*	0.01	0.57*	0.54*	0.76*	0.97*
Subbasin 76	0.44*	0.49*	0.67*	0.05	0.4*	0.47*	0.64*	0.82*
Subbasin 77	-0.01	0	0.01	-0.01	-0.02	0.01	0.02	0.05
Subbasin 78	0.26*	0.22*	0.39*	0.01	0.27*	0.25*	0.35*	0.53*
Subbasin 79	0.42*	0.33*	0.5*	0.04	0.42*	0.29*	0.61*	0.77*
Subbasin 80	0.56*	0.71*	0.82*	0.03	0.55*	0.72*	0.81*	1.02*
Subbasin 81	0.33*	0.21*	0.4*	0.01	0.34*	0.21*	0.42*	0.58*
Subbasin 82	0.49*	0.61*	0.78*	0.01	0.55*	0.62*	0.73*	0.9*
Subbasin 83	0.59*	0.76*	0.9*	0.02	0.62*	0.74*	0.92*	1.1*
Subbasin 84	0.03	0.03	-0.01	0.01	0	-0.02	0.05***	-0.01

Subbasin 85	-0.01	0.01	-0.02	0.01	0.04	0.06	0.01	0
Subbasin 86	0.05*	0.05*	0.07*	-0.01	0.06*	0.04*	0.07*	0.14*
Subbasin 87	-0.05	-0.05	-0.06	-0.1	-0.1	-0.02	-0.02	0.1
Subbasin 88	0.03**	0.03**	0.03**	-0.01	0.02***	0.02	0.04**	0.05*
Subbasin 89	0.03**	0.04**	0.05*	-0.01	0.03**	0.04*	0.05*	0.07*
Subbasin 90	0.03**	0.02**	0.03**	-0.02**	0.03**	0.02***	0.03**	0.06*
Subbasin 91	0.06*	0.06*	0.06*	0.03**	0.06*	0.03**	0.08*	0.13*
Subbasin 92	0.02	0.03***	0.06**	-0.01	0.05**	0.03**	0.04**	0.11*
Subbasin 93	0.05*	0.05*	0.07*	-0.02	0.06*	0.06*	0.06*	0.11*
Subbasin 94	0.06*	0.06*	0.07*	-0.01	0.06*	0.07*	0.09*	0.14*
Subbasin 95	0.37*	0.41*	0.51*	0.01	0.35*	0.41*	0.51*	0.66*
Subbasin 96	0.32*	0.3*	0.42*	-0.01	0.32*	0.32*	0.61	0.58*
Subbasin 97	0.4*	0.47*	0.56*	0.02***	0.38*	0.46*	0.56*	0.73*
Subbasin 98	0.5*	0.56*	0.71*	0	0.49*	0.55*	0.72*	0.94*
Subbasin 99	0.49*	0.53*	0.69*	0.01	0.48*	0.55*	0.69*	0.9*
Subbasin 100	0.09*	0.12*	0.14*	0.01	0.09*	0.13*	0.13*	0.2*
Subbasin 101	0.26*	0.26*	0.37*	0.01	0.27*	0.26*	0.35*	0.47
Subbasin 102	0.13*	0.08*	0.15*	0.01	0.13*	0.08*	0.16*	0.23*
Subbasin 103	0.11*	0.07*	0.12*	-0.01	0.1*	0.07*	0.14*	0.21*
Subbasin 104	0.07*	0.02**	0.07*	0	0.07*	0.02**	0.07*	0.11*
Subbasin 105	0.16*	0.09*	0.22*	0.01	0.17*	0.11*	0.18*	0.28*
Subbasin 106	0.38*	0.36*	0.48*	-0.05	0.37*	0.36*	0.49*	0.67*
Subbasin 107	0.86**	0.75**	0.93*	-0.07	0.89**	1.19*	1.06*	1.27*
Subbasin 108	1.18***	0.29	0.54	-0.58	0.94	-0.58	0.59	0.89
Subbasin 109	0.41*	0.28*	0.51*	-0.01	0.42*	0.29*	0.52*	0.67*
Subbasin 110	0.01	0.01	0.01	0	0	-0.01	-0.01	0
Subbasin 111	-0.04	-0.06	-0.03	0.02	0.05	0.03	0.01	-0.01
Subbasin 112	0.22*	0.13*	0.25*	0	0.21*	0.13*	0.24*	0.37*

Note: Asterisk (*), double asterisk (**), and triple asterisk (***) denote significance at 1%, 5%, and 10%,

Table A-9. Boone River Watershed: Delivery coefficients by subbasin

	Nitrogen			Phosphorus		
Location	**Field**	**Subbasin**	**Watershed**	**Field**	**Subbasin**	**Watershed**
Sub basin 1	1.11	1.15	0.81	0.52	0.49	0.87
Sub basin 2	1.12	1.15	0.81	0.61	0.62	0.72
Sub basin 3	0.96	1.06	1.00	0.49	0.56	0.86
Sub basin 4	1.07	1.11	1.16	0.76	0.95	0.62
Sub basin 5	0.91	1.02	1.13	0.56	0.66	1.15
Sub basin 6	0.99	1.08	1.03	0.59	0.59	0.93
Sub basin 7	1.76	1.76	0.67	0.78	0.78	0.64
Sub basin 8	1.23	1.30	0.87	0.50	0.51	0.92
Sub basin 9	0.68	0.85	2.06	0.18	0.26	3.72
Sub basin 10	0.91	1.00	1.23	0.55	0.66	1.18
Sub basin 11	0.94	1.00	1.24	0.34	0.26	1.68
Sub basin 12	1.12	1.12	0.98	0.48	0.46	1.00
Sub basin 13	1.00	1.04	1.24	0.43	0.38	1.46
Sub basin 14	1.24	1.21	1.25	0.56	0.70	1.44
Sub basin 15	0.94	1.09	1.34	0.42	0.46	1.70
Sub basin 16	0.75	0.91	1.87	0.25	0.34	2.96
Sub basin 17	0.96	1.16	1.35	0.48	0.63	1.52
Sub basin 18	1.07	1.21	0.93	0.63	0.70	0.86
Sub basin 19	0.95	1.00	1.07	0.62	0.64	0.79
Sub basin 20	1.08	1.16	1.10	0.51	0.50	1.34
Sub basin 21	1.02	1.10	0.65	0.61	0.59	0.59
Sub basin 22	0.98	1.05	0.90	0.51	0.50	1.04
Sub basin 23	1.14	1.19	0.92	0.82	0.85	0.59
Sub basin 24	0.95	1.01	0.99	0.64	0.67	0.80
Sub basin 25	0.76	0.74	1.37	0.49	0.60	0.93
Sub basin 26	1.03	1.15	0.60	0.72	0.77	0.50
Sub basin 27	2.17	2.02	0.53	1.49	1.47	0.48
Sub basin 28	0.94	1.03	1.28	0.51	0.59	1.14
Sub basin 29	1.10	1.25	0.85	0.88	0.96	0.62
Sub basin 30	1.16	1.16	0.87	1.03	1.03	0.45

Table A-10. Raccoon River Watershed: Delivery coefficients by subbasin,

	Nitrogen			Phosphorus				Nitrogen			Phosphorus		
Location	**Field**	**Subbasin**	**Watershed**	**Field**	**Subbasin**	**Watershed**		**Field**	**Subbasin**	**Watershed**	**Field**	**Subbasin**	**Watershed**
Subbasin1	0.40	0.36	1.92	0.26	0.27	2.58	Subbasin29	0.71	0.71	1.20	0.33	0.34	1.61
Subbasin2	0.87	0.79	1.05	0.35	0.34	2.23	Subbasin30	0.92	0.91	1.12	0.54	0.54	2.10
Subbasin3	0.52	0.52	1.79	0.19	0.19	5.19	Subbasin31	0.87	0.87	1.21	0.46	0.46	2.27
Subbasin4	0.79	0.79	1.32	0.30	0.31	5.19	Subbasin32	0.62	0.64	1.17	0.36	0.38	1.17
Subbasin5	0.40	0.42	2.88	0.52	0.50	2.62	Subbasin33	0.78	0.77	1.30	0.40	0.40	2.03
Subbasin6	0.66	0.58	1.34	0.16	0.16	4.95	Subbasin34	0.71	0.68	1.45	0.35	0.29	2.59
Subbasin7	0.53	0.57	1.42	0.41	0.41	2.14	Subbasin35	0.55	0.67	1.07	0.35	0.35	1.28
Subbasin8	0.73	0.71	1.34	0.35	0.35	2.57	Subbasin36	0.79	0.80	1.06	0.46	0.47	2.20
Subbasin9	0.61	0.59	1.75	0.00	0.15	7.56	Subbasin37	0.64	0.64	1.32	0.46	0.31	2.15
Subbasin10	0.62	0.62	1.83	0.00	0.18	6.33	Subbasin38	0.50	0.50	1.28	0.40	0.37	1.55
Subbasin11	0.66	0.67	1.33	0.28	0.28	2.61	Subbasin39	0.71	0.71	1.18	0.39	0.37	1.34
Subbasin12	0.67	0.67	1.04	0.37	0.37	1.43	Subbasin40	0.62	0.68	1.13	0.47	0.47	1.01
Subbasin13	0.78	0.74	1.25	0.23	0.31	5.13	Subbasin41	0.77	0.77	1.00	0.48	0.45	1.24
Subbasin14	0.54	0.52	1.79	0.00	0.09	6.86	Subbasin42	0.66	0.66	1.20	0.45	0.45	1.28
Subbasin15	0.57	0.64	1.13	0.26	0.27	1.99	Subbasin43	0.74	0.72	1.14	0.43	0.41	1.53
Subbasin16	0.66	0.65	1.60	0.30	0.29	2.95	Subbasin44	0.76	0.75	0.92	0.44	0.44	1.51
Subbasin17	0.52	0.52	1.82	0.32	0.31	2.77	Subbasin45	0.73	0.70	0.53	0.50	0.49	0.39
Subbasin18	0.75	0.75	1.52	0.35	0.35	2.60	Subbasin46	0.64	0.63	1.27	0.43	0.43	1.47
Subbasin19	0.62	0.55	1.81	0.36	0.36	2.89	Subbasin47	0.90	0.88	0.84	0.57	0.56	1.15
Subbasin20	0.70	0.72	1.53	0.37	0.46	2.36	Subbasin48	0.71	0.69	1.10	0.60	0.56	1.55
Subbasin21	0.49	0.48	1.98	0.23	0.16	5.32	Subbasin49	0.88	0.87	0.95	0.53	0.51	1.62
Subbasin22	0.68	0.69	1.18	0.32	0.33	2.05	Subbasin50	0.79	0.77	0.93	0.48	0.49	1.26
Subbasin23	0.64	0.65	1.45	0.35	0.35	2.46	Subbasin51	0.70	0.69	0.93	0.39	0.38	1.40
Subbasin24	0.58	0.58	1.52	0.29	0.27	2.31	Subbasin52	0.87	0.86	0.56	0.57	0.54	0.50
Subbasin25	0.60	0.61	1.56	0.31	0.31	3.24	Subbasin53	0.00	0.34	1.05	0.49	0.78	0.76
Subbasin26	0.35	0.34	2.30	0.24	0.26	3.88	Subbasin54	0.82	0.81	0.74	0.55	0.55	1.05
Subbasin27	0.67	0.74	1.20	0.39	0.40	1.62	Subbasin55	0.81	0.80	0.88	0.59	0.54	0.98
Subbasin28	0.65	0.67	0.96	0.36	0.37	1.08	Subbasin56	0.77	0.77	0.96	0.52	0.52	1.19

Location	Field	Subbasin	Watershed	Field	Subbasin	Watershed		Field	Subbasin	Watershed	Field	Subbasin	Watershed
Subbasin57	0.80	0.79	0.97	0.55	0.56	1.01	Subbasin85	0.00	0.00	-2.18	0.00	0.01	1.01
Subbasin58	0.72	0.67	0.91	0.57	0.58	1.02	Subbasin86	0.66	0.67	1.24	0.31	0.31	2.27
Subbasin59	0.96	0.94	0.64	0.61	0.59	0.83	Subbasin87	0.00	0.21	0.79	0.00	0.00	-2.55
Subbasin60	0.86	0.86	0.81	0.64	0.64	0.81	Subbasin88	0.59	0.58	1.65	0.28	0.28	4.98
Subbasin61	0.72	0.70	0.50	0.59	0.58	0.24	Subbasin89	0.78	0.78	1.21	0.34	0.29	3.29
Subbasin62	1.00	0.98	0.73	0.74	0.73	0.85	Subbasin90	0.60	0.56	1.68	0.23	0.23	4.67
Subbasin63	0.93	0.83	0.94	0.66	0.66	1.10	Subbasin91	0.64	0.63	1.35	0.43	0.38	2.16
Subbasin64	0.61	0.60	0.53	0.43	0.43	0.30	Subbasin92	0.69	0.69	1.04	0.25	0.25	3.12
Subbasin65	0.70	0.63	0.57	0.40	0.40	0.39	Subbasin93	0.61	0.60	1.57	0.46	0.46	2.39
Subbasin66	0.86	0.85	0.84	0.59	0.62	1.10	Subbasin94	0.75	0.76	1.20	0.51	0.47	2.03
Subbasin67	0.79	0.65	0.38	0.47	0.47	0.32	Subbasin95	0.66	0.64	0.59	0.49	0.48	0.34
Subbasin68	0.76	0.73	0.93	0.59	0.60	1.16	Subbasin96	0.65	0.65	0.72	0.50	0.49	0.42
Subbasin69	0.74	0.71	1.08	0.52	0.52	1.19	Subbasin97	0.72	0.71	0.53	0.57	0.55	0.31
Subbasin70	0.80	0.74	0.49	0.60	0.59	0.29	Subbasin98	0.76	0.73	0.42	0.57	0.55	0.24
Subbasin71	0.49	0.48	0.64	0.39	0.38	0.46	Subbasin99	0.77	0.75	0.46	0.58	0.56	0.27
Subbasin72	0.90	0.82	0.74	0.56	0.56	0.92	Subbasin100	0.79	0.79	1.03	0.56	0.56	1.28
Subbasin73	0.78	0.75	0.40	0.61	0.60	0.26	Subbasin101	0.77	0.79	0.62	0.55	0.58	0.49
Subbasin74	0.63	0.62	0.64	0.58	0.56	0.35	Subbasin102	0.83	0.83	0.89	0.55	0.56	1.11
Subbasin75	0.77	0.76	0.44	0.71	0.71	0.23	Subbasin103	0.81	0.80	0.92	0.52	0.51	1.26
Subbasin76	0.98	0.95	0.37	0.74	0.72	0.27	Subbasin104	0.70	0.70	1.21	0.42	0.42	2.35
Subbasin77	0.00	0.04	0.00	0.00	0.00	5.62	Subbasin105	0.91	0.86	0.80	0.73	0.71	0.92
Subbasin78	0.85	0.81	0.65	0.76	0.77	0.50	Subbasin106	0.85	0.83	0.42	0.61	0.59	0.35
Subbasin79	0.96	0.94	0.47	0.75	0.77	0.33	Subbasin107	1.02	0.98	0.24	0.84	0.78	0.16
Subbasin80	0.93	0.89	0.38	0.84	0.82	0.21	Subbasin108	0.00	2.08	0.15	0.00	0.97	0.29
Subbasin81	0.93	0.88	0.59	0.82	0.81	0.43	Subbasin109	0.80	0.76	0.58	0.70	0.70	0.34
Subbasin82	1.03	0.99	0.38	0.95	0.93	0.24	Subbasin110	0.00	0.01	6.91	0.00	0.01	1.01
Subbasin83	0.87	0.84	0.35	0.88	0.85	0.19	Subbasin111	0.00	0.00	-7.48	0.00	0.00	0.12
Subbasin84	0.00	0.08	2.02	0.00	0.00	0.23	Subbasin112	0.87	0.86	0.60	0.52	0.52	0.72

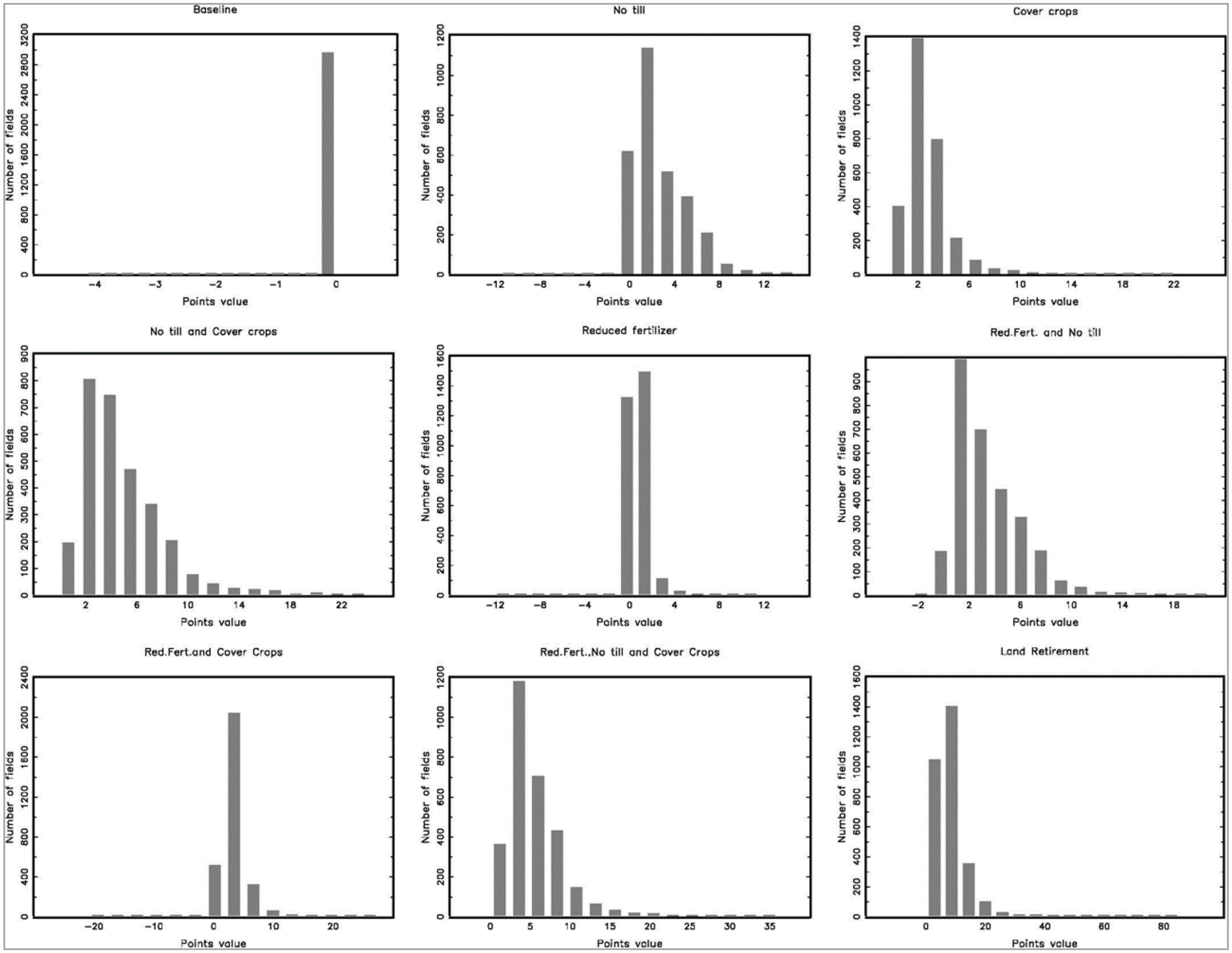


Figure A-1 Boone River Watershed: Distribution of point coefficients across watershed, nitrogen.

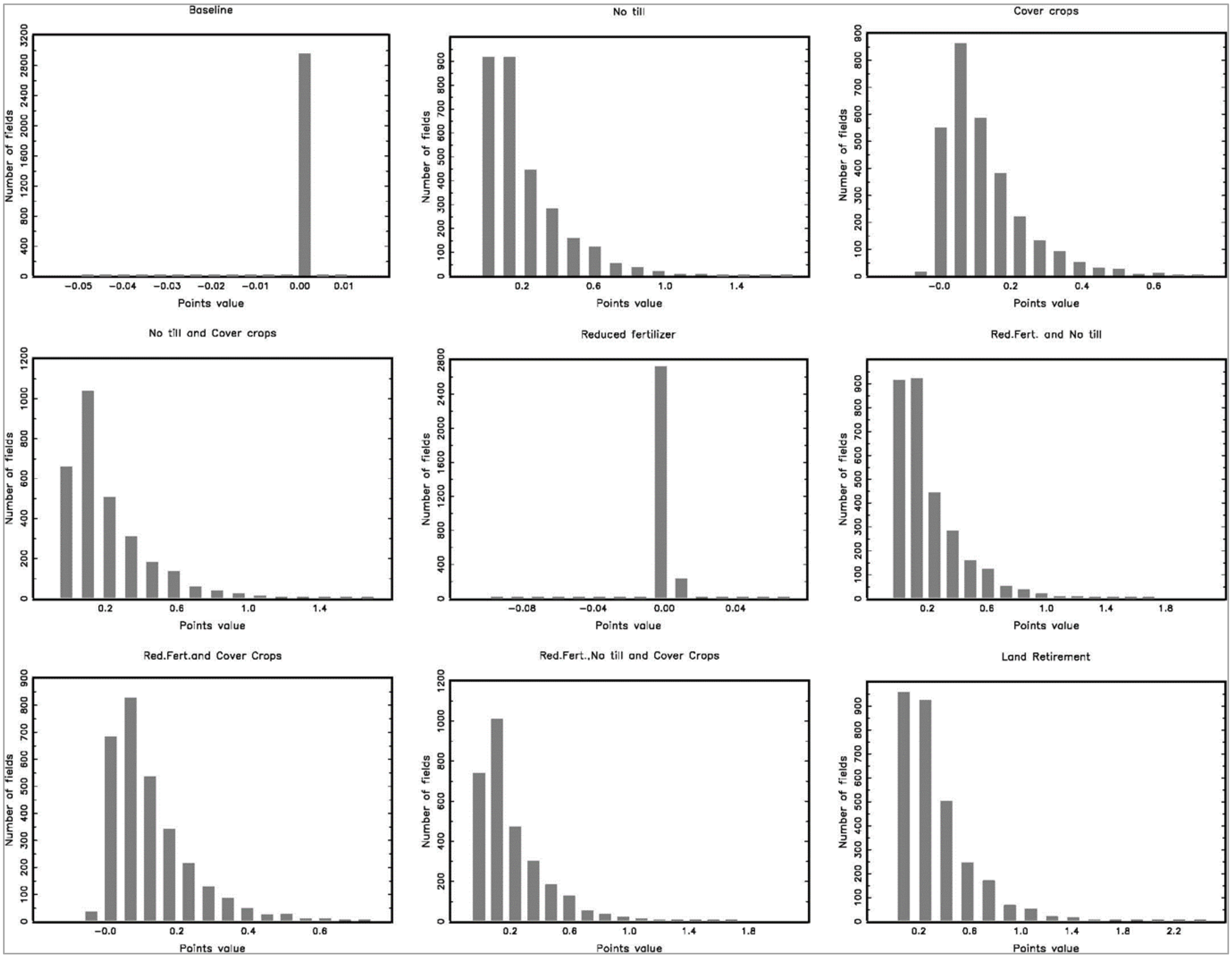


Figure A-2. Boone River Watershed: Distribution of point coefficients across watershed, phosphorus.

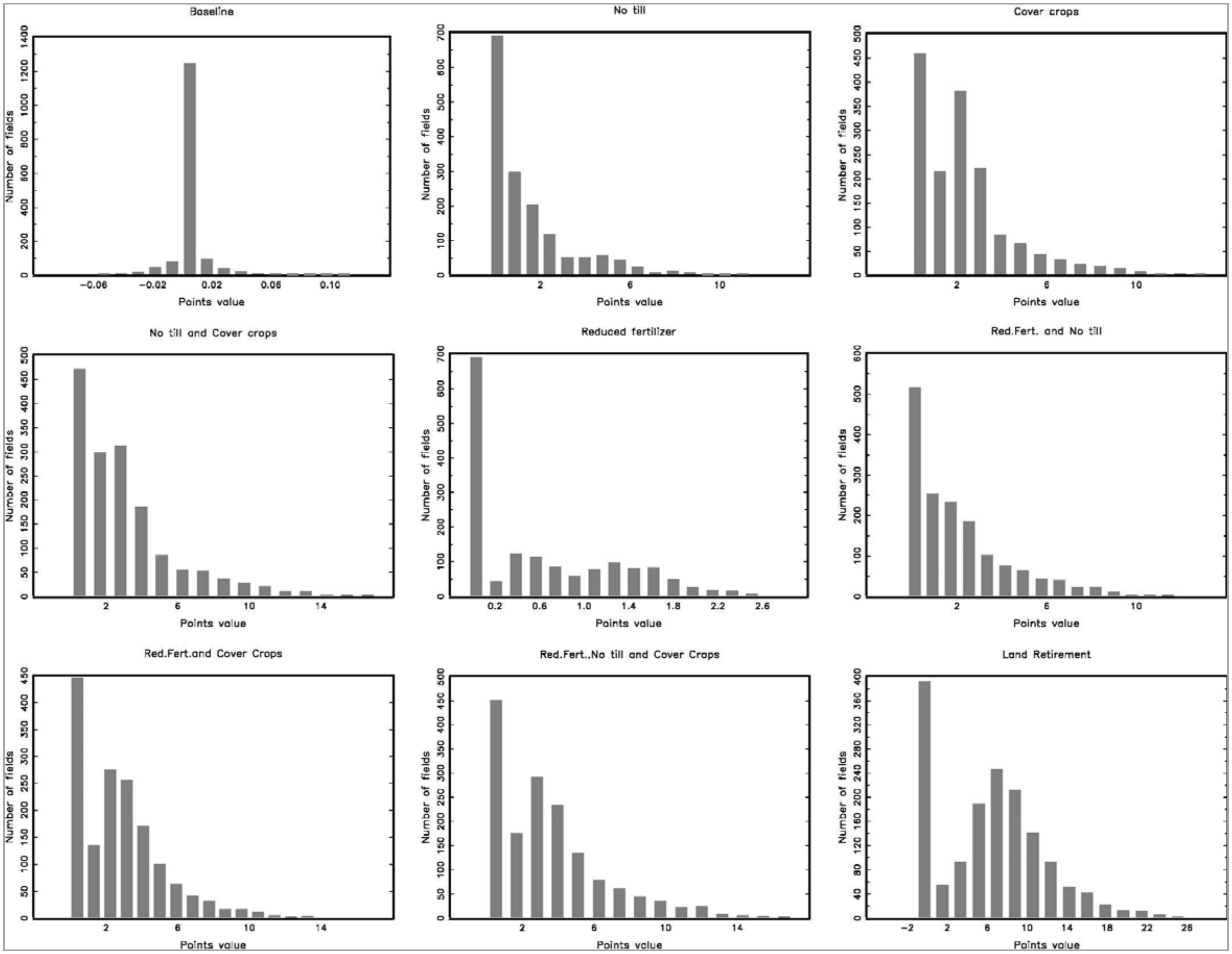


Figure A-3 Raccoon River Watershed: Distribution of Point coefficients across watershed, nitrogen.

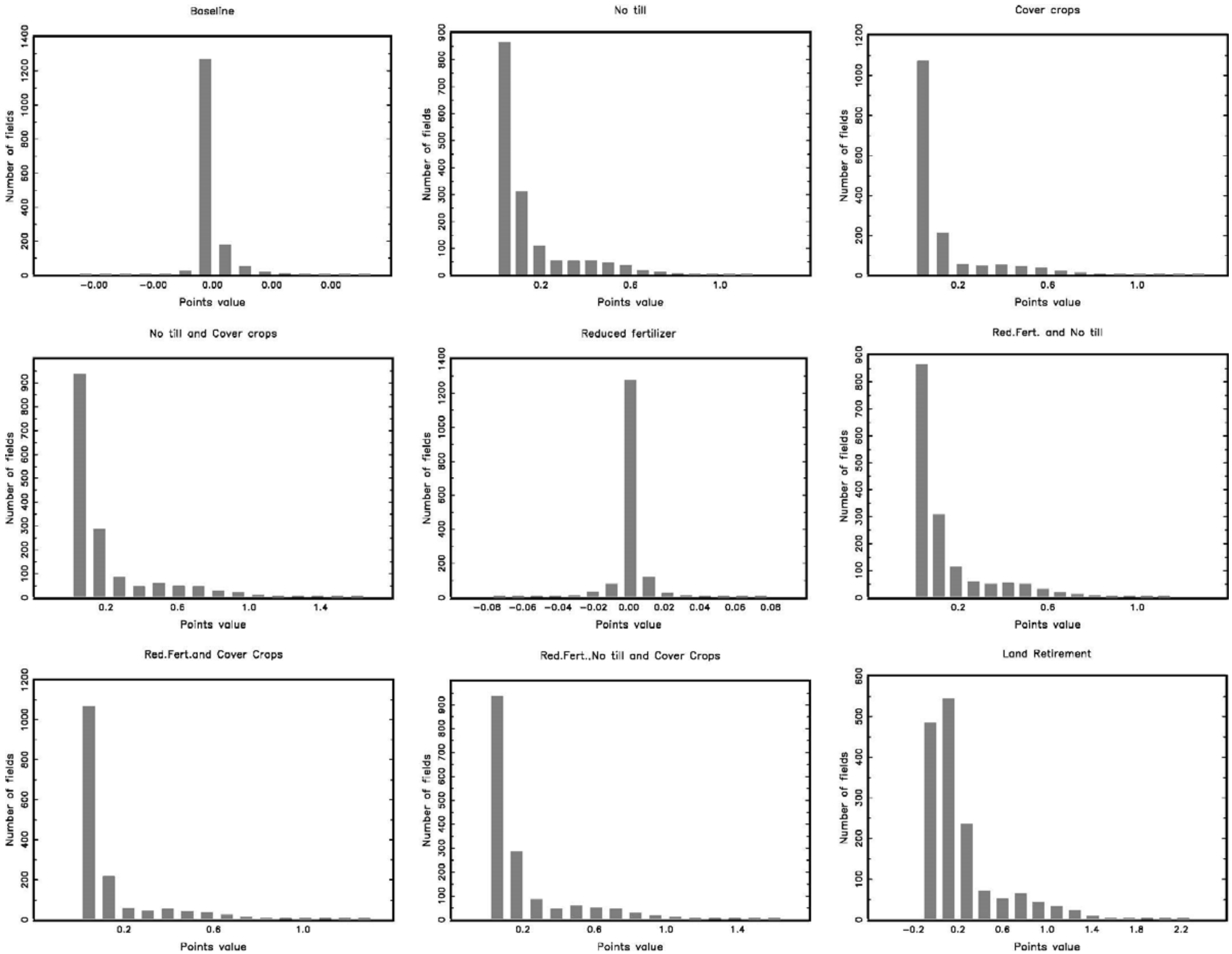


Figure A-4 Raccoon River Watershed: Distribution of point coefficients across watershed, phosphorus.

Tables and Figures for Chapter 4

Table B-1. Evolutionary Algorithm Parameters

Parameter description	Value
Size of temporary population	16
Crossover probability	1
Mutation Probability	0.003
Total Number of Generations	300

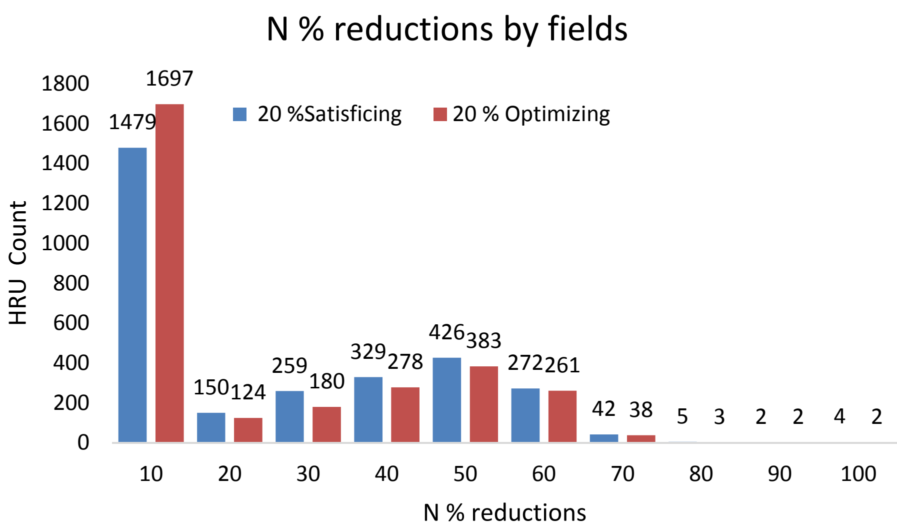


Figure B-1. Boone River Watershed: Distribution of abatement effort corresponding to trading outcomes, N 20% goal.

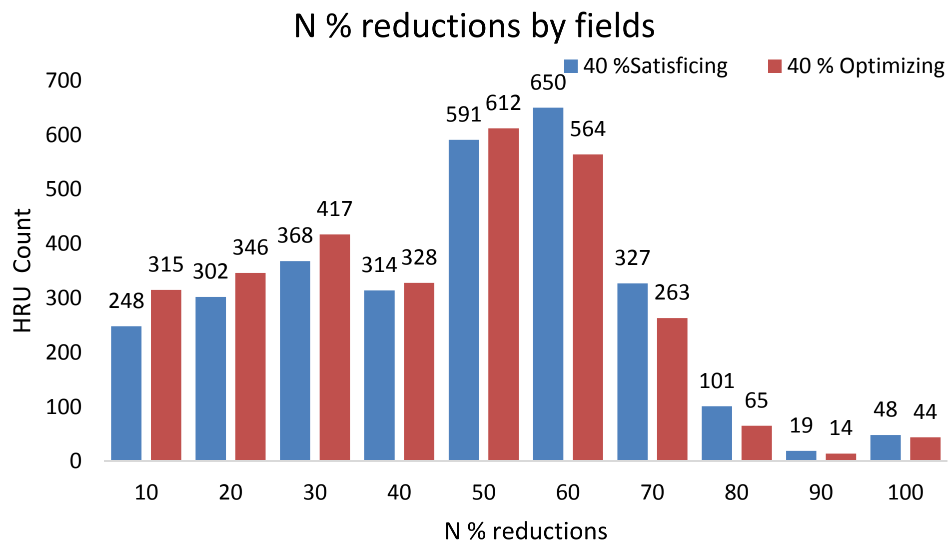


Figure B-2. Boone River Watershed: Distribution of abatement effort corresponding to trading outcomes, N 40% goal.

Table B-2. Boone River Watershed: Spatial distribution of abatement actions (% of total area), optimizing policies, phosphorus

	Abatement goal 20%			**Abatement goal 30%**			**Abatement goal 40%**		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
No Action	55.42	56.8	74.6	19.97	22.3	55.0	0.13	2.2	4.4
No till (NT)	43.67	41.8	18.8	79.40	76.3	35.9	66.66	83.0	75.6
Cover Crop (CC)	0.33	0.0	0.0	0.04	0.0	0.0	0.20	0.2	0.1
NT,CC	0.04	0.0	0.0	0.13	0.0	0.0	0.18	0.0	1.6
Red.Fert.	0.04	0.3	2.4	0.06	0.0	1.9	0.24	0.3	0.1
Red.Fert., NT	0.24	1.0	4.1	0.02	1.3	7.1	28.05	10.2	15.5
Red.Fert.,CC	0.00	0.0	0.0	0.02	0.0	0.0	0.39	0.0	0.0
Red.Fert.,NT,CC	0.21	0.0	0.0	0.33	0.0	0.0	0.16	0.0	0.4
LandRetirement	0.05	0.1	0.0	0.03	0.0	0.0	3.98	4.0	2.3

Table B-3. Boone River Watershed: Spatial distribution of abatement actions (% of total area), satisficing policies, phosphorus

	Abatement goal 20%			Abatement goal 30%			Abatement goal 40%		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
No Action	17.97	30.01	76.84	9.93	16.62	57.10	18.62	20.85	19.04
No till (NT)	2.89	48.33	16.77	15.90	56.88	33.94	5.31	36.75	66.57
Cover Crop (CC)	7.76	5.35	0.04	4.35	4.61	0.03	3.03	1.72	0.01
NT,CC	10.44	3.59	0.00	19.00	5.70	0.01	0.70	0.97	0.05
Red.Fert.	18.72	1.27	2.50	10.03	0.75	1.91	2.81	0.69	0.20
Red.Fert., NT	14.73	7.53	3.78	12.61	6.08	6.95	25.15	8.78	14.07
Red.Fert.,CC	11.95	0.69	0.02	17.03	1.19	0.00	2.19	0.24	0.00
Red.Fert.,NT,CC	15.55	3.23	0.03	4.08	1.10	0.03	14.26	2.07	0.03
LandRetirement	0.00	0.00	0.02	7.07	7.07	0.02	27.93	27.93	0.03

Table B-4. Boone River Watershed: Change in the CAC abatement actions distribution (% of total area)

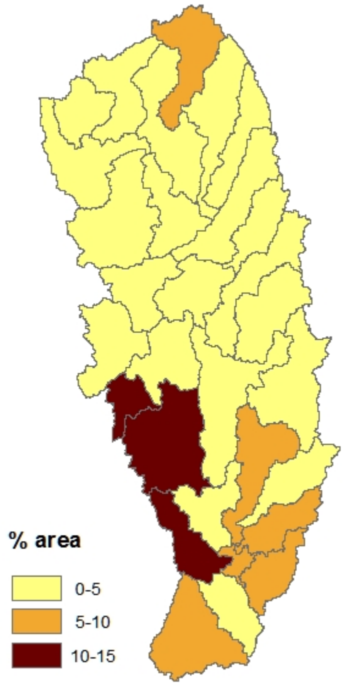
		Nitrogen		Phosphorus	
Approach	Target	PS	Trading	PS	Trading
	20%	33.94	81.94	65.46	85.90
	30%	34.00	93.79	55.78	87.00
Satisificing	40%	23.89	93.75	37.66	91.32
	20%	4.14	56.45	3.51	39.32
Optimizing	30%	6.74	47.08	3.85	59.47
	40%	15.47	68.49	22.12	38.93

Table B-5. Boone River Watershed: The change in the relative distribution of the abatement actions relative to CAC, optimizing, phosphorus

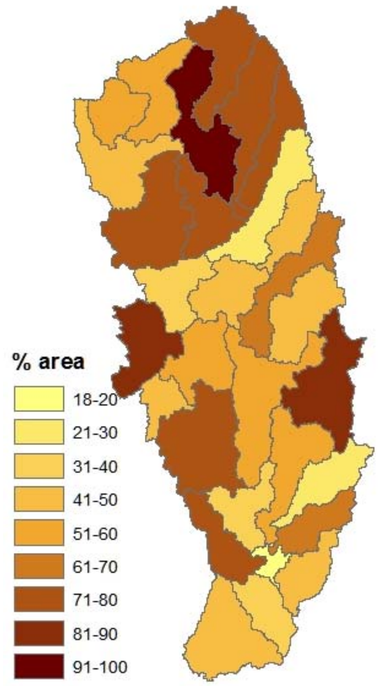
	Optimizing Approach					
% of subbasin	Performance Standard			Trading		
Area	20%	30%	40%	20%	30%	40%
<=10%	26	27	17	6	0	6
(10%- 20%]	4	3	2	4	1	8
(20%- 30%]	0	0	1	2	2	1
(30%- 40%]	0	0	0	3	3	3
(40%- 50%]	0	0	2	0	7	0
(50%- 60%]	0	0	3	3	5	1
(60%- 70%]	0	0	3	6	2	1
(70%- 80%]	0	0	1	5	7	2
(80%- 90%]	0	0	1	1	2	2
(90%- 100%]	0	0	0	0	1	6
	30	30	30	30	30	30

Table B-6. Counts of HRU hotspots

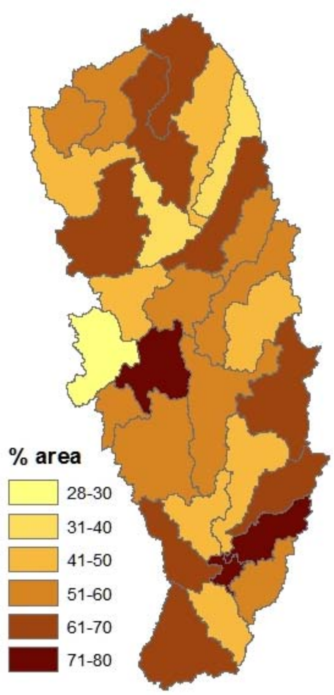
		Boone River Watershed			Raccoon River Watershed		
	Point level	20%	30%	40%	20%	30%	40%
	Field	0	0	0	0	0	0
	Subbasin	30	61	51	1	3	0
N satisficing	Watershed	43	63	10	6	5	0
	Field	0	0	0	0	0	0
	Subbasin	29	59	51	1	3	0
N optimizing	Watershed	38	63	12	6	5	0
	Field	17	12	8	0	0	0
	Subbasin	86	89	81	6	10	13
P satisficing	Wateshed	86	108	57	0	0	0
	Field	16	12	2	0	0	0
	Subbasin	88	89	80	6	10	13
P optimizing	Watershed	80	108	59	0	0	0



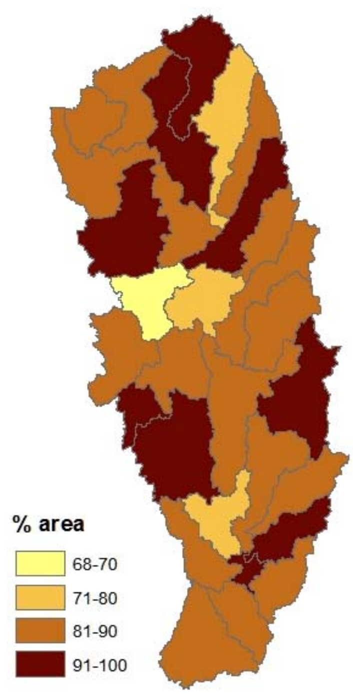
Performance standard Optimizing



Trading Optimizing



Performance Standard Satisficing



Trading Satisficing

Figure B-3. Boone River Watershed: Spatial representation of change in distribution, 30% P abatement goal.

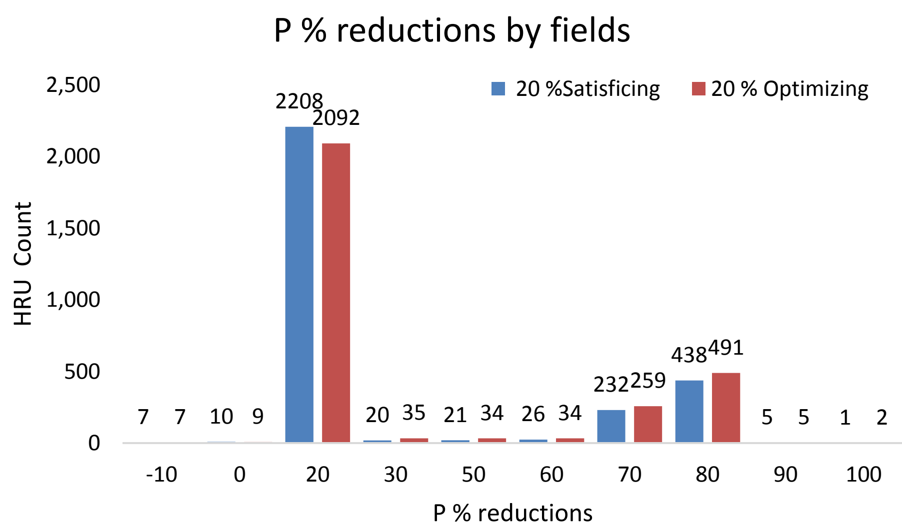


Figure B-4. Boone River Watershed: Distribution of abatement effort corresponding to trading outcomes, 20% P abatement goal.

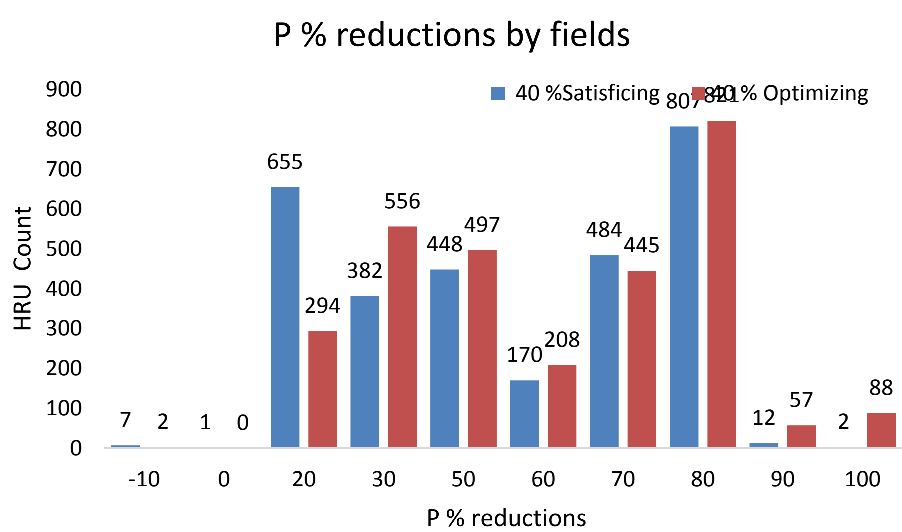


Figure B-5. Boone River Watershed: Distribution of abatement effort corresponding to trading outcomes, 40% P abatement goal.

Table B-7. Raccoon River Watershed: The distribution of abatement actions (% of total area), satisficing policies, nitrogen,

	Abatement goal 20%			Abatement goal 30%			Abatement goal 40%		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
No Action	1.86	5.89	19.56	6.21	9.28	9.27	3.67	6.90	3.83
No till (NT)	1.49	11.60	9.46	1.99	4.64	7.56	0.23	7.06	3.04
Cover Crop (CC)	0.33	3.23	1.38	0.28	3.45	7.59	5.10	4.95	11.28
NT,CC	2.38	1.13	1.69	8.50	9.52	3.03	1.67	0.89	6.48
Red.Fert.	23.15	22.97	38.68	17.97	16.54	20.83	20.94	19.94	4.22
Red.Fert., NT	63.40	47.96	20.29	9.93	8.98	14.99	31.26	24.16	6.96
Red.Fert.,CC	1.58	1.87	5.74	13.07	13.26	25.13	2.14	1.98	34.86
Red.Fert.,NT,CC	2.17	1.91	3.21	36.36	28.72	11.60	0.15	0.08	26.22
LandRetirement	3.64	3.45	0.00	5.69	5.62	0.00	34.84	34.05	3.10

Table B-8. Raccoon River Watershed: The distribution of abatement actions (% of total area), optimizing policies, phosphorus

	Abatement goal 20%			Abatement goal 30%			Abatement goal 40%		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
No Action	58.60	61.31	76.18	30.77	31.60	58.02	0.00	1.07	27.56
No till (NT)	35.03	34.69	22.29	68.92	68.29	36.70	88.10	87.88	61.28
Cover Crop (CC)	0.24	0.02	0.76	0.10	0.02	0.38	0.00	0.03	0.54
NT,CC	0.03	0.07	0.32	0.01	0.00	4.34	11.02	10.46	9.90
Red.Fert.	6.00	3.85	0.08	0.00	0.00	0.08	0.20	0.14	0.22
Red.Fert., NT	0.00	0.00	0.10	0.05	0.00	0.20	0.24	0.12	0.21
Red.Fert.,CC	0.00	0.00	0.12	0.07	0.00	0.14	0.14	0.00	0.14
Red.Fert.,NT,CC	0.11	0.07	0.14	0.01	0.00	0.14	0.08	0.08	0.14
LandRetirement	0.00	0.00	0.00	0.08	0.08	0.00	0.22	0.22	0.00

Table B-9. Raccoon River Watershed:Spatial distribution of abatement actions (% of total area), satisficing policies, phosphorus

	Abatement goal 20%			**Abatement goal 30%**			**Abatement goal 40%**		
Abatement Action	CAC	PS	Trading	CAC	PS	Trading	CAC	PS	Trading
No Action	17.97	30.01	76.84	9.93	16.62	57.10	18.62	20.85	19.04
No till (NT)	2.89	48.33	16.77	15.90	56.88	33.94	5.31	36.75	66.57
Cover Crop (CC)	7.76	5.35	0.04	4.35	4.61	0.03	3.03	1.72	0.01
NT,CC	10.44	3.59	0.00	19.00	5.70	0.01	0.70	0.97	0.05
Red.Fert.	18.72	1.27	2.50	10.03	0.75	1.91	2.81	0.69	0.20
Red.Fert., NT	14.73	7.53	3.78	12.61	6.08	6.95	25.15	8.78	14.07
Red.Fert.,CC	11.95	0.69	0.02	17.03	1.19	0.00	2.19	0.24	0.00
Red.Fert.,NT,CC	15.55	3.23	0.03	4.08	1.10	0.03	14.26	2.07	0.03
LandRetirement	0.00	0.00	0.02	7.07	7.07	0.02	27.93	27.93	0.03

Table B-10.Raccoon River Watershed: Change in the relative distribution of the abatement actions relative to CAC, optimizing, nitrogen

% of subbasin	Performance Standard			Trading		
Area	20%	30%	40%	20%	30%	40%
<=10%	47	34	41	2	2	4
(10%- 20%]	20	21	24	6	2	3
(20%- 30%]	17	19	19	4	8	10
(30%- 40%]	11	12	7	0	4	7
(40%- 50%]	4	12	9	6	7	8
(50%- 60%]	2	2	2	10	10	10
(60%- 70%]	3	4	4	9	9	10
(70%- 80%]	3	3	1	12	15	13
(80%- 90%]	0	0	0	20	12	15
(90%- 100%]	5	5	5	43	43	32
Counts of subbasin	112	112	112	112	112	112

Table B-11. Raccoon River Watershed: Change in the relative distribution of the abatement actions relative to CAC, satisficing, nitrogen

% of subbasin	Performance Standard			Trading		
Area	20%	30%	40%	20%	30%	40%
<=10%	37	32	48	0	0	0
(10%- 20%]	32	38	30	0	0	0
(20%- 30%]	20	23	21	0	0	0
(30%- 40%]	9	3	4	1	0	0
(40%- 50%]	5	7	3	6	0	0
(50%- 60%]	2	3	2	11	4	1
(60%- 70%]	2	0	0	12	9	2
(70%- 80%]	0	2	0	20	20	5
(80%- 90%]	0	0	2	26	26	14
(90%- 100%]	5	4	2	36	53	90
Counts of subbasin	112	112	112	112	112	112

Table B-12. Raccoon River Watershed: Change in the relative distribution of the abatement actions relative to CAC, optimizing, phosphorus

% of subbasin	Performance Standard			Trading		
Area	20%	30%	40%	20%	30%	40%
<=10%	70	57	47	56	28	23
(10%- 20%]	28	48	55	5	7	8
(20%- 30%]	5	6	8	7	10	7
(30%- 40%]	3	1	2	7	5	10
(40%- 50%]	2	0	0	1	17	19
(50%- 60%]	1	0	0	5	6	10
(60%- 70%]	2	0	0	1	6	3
(70%- 80%]	0	0	0	3	5	2
(80%- 90%]	1	0	0	1	1	7
(90%- 100%]	0	0	0	26	27	23
Counts of subbasin	112	112	112	112	112	112

Table B-13. Raccoon River Watershed: Change in the relative distribution of the abatement actions relative to CAC, satisficing, phosphorus

% of subbasin	Performance Standard			Trading		
Area	20%	30%	40%	20%	30%	40%
<=10%	7	1	3	0	0	0
(10%- 20%]	7	4	6	0	0	0
(20%- 30%]	12	3	19	2	0	0
(30%- 40%]	28	11	28	7	0	0
(40%- 50%]	25	20	18	2	0	0
(50%- 60%]	16	15	16	14	0	0
(60%- 70%]	10	21	14	8	0	1
(70%- 80%]	5	17	4	24	0	7
(80%- 90%]	2	8	4	22	6	12
(90%- 100%]	0	12	0	33	106	92
Counts of subbasin	112	112	112	112	112	112

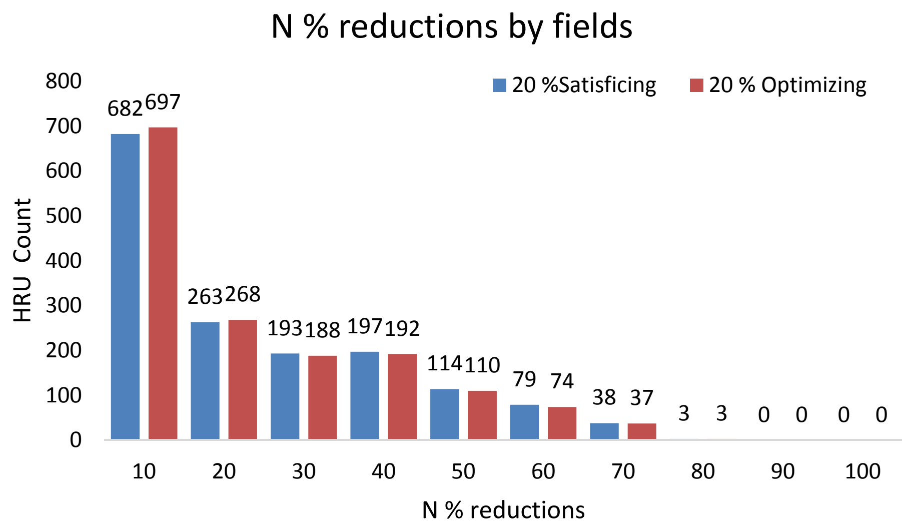


Figure B-6. Raccoon River Watershed: Distribution of abatement effort corresponding to trading outcomes, 20%N abatement goal.

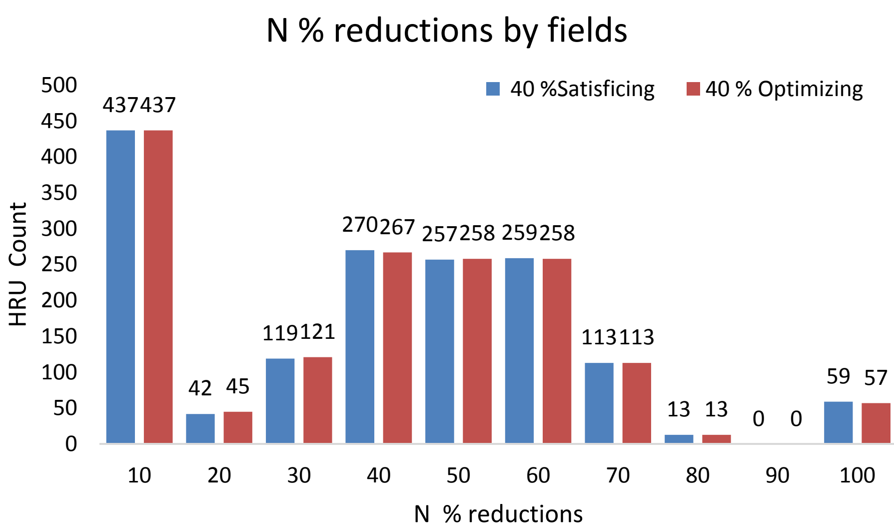


Figure B-7. Raccoon River Watershed: Distribution of abatement effort corresponding to trading outcomes, 40% N abatement goal.

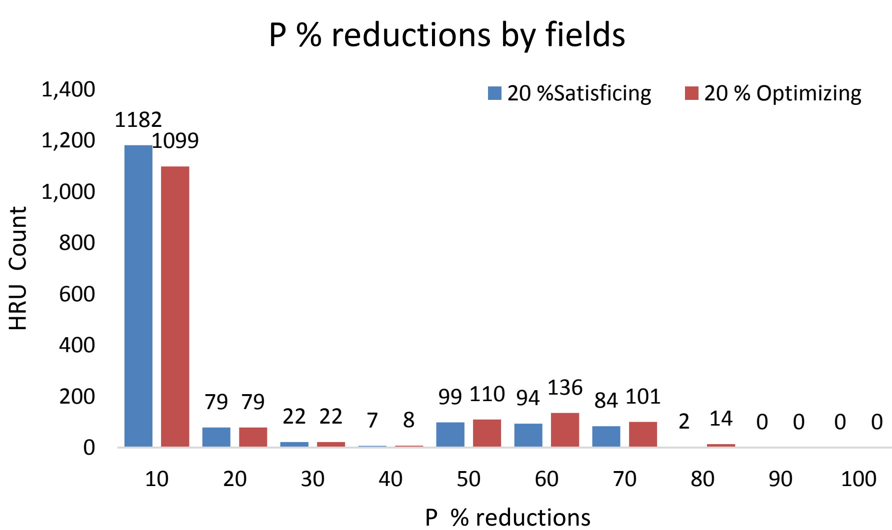


Figure B-8. Raccoon River Watershed: Distribution of abatement effort corresponding to trading outcomes, 20% P abatement goal

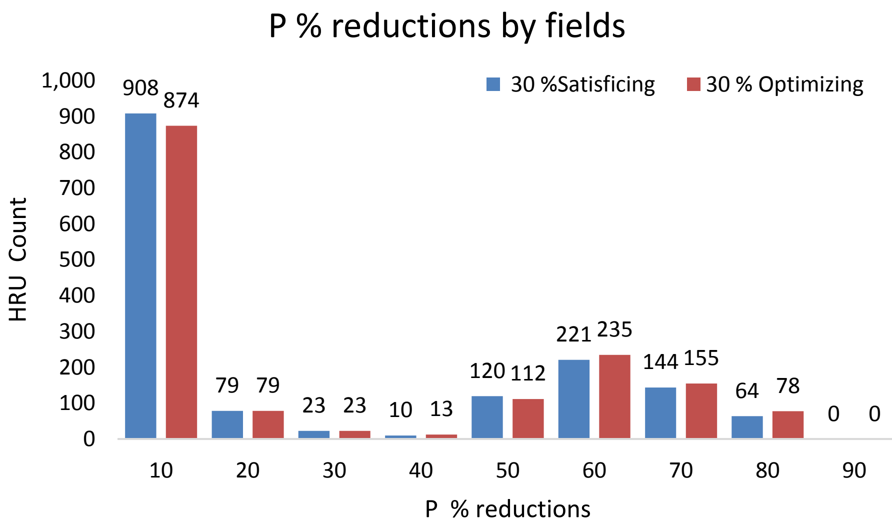


Figure B-9. Raccoon River Watershed: Distribution of abatement effort corresponding to trading outcomes, 30% P abatement goal.

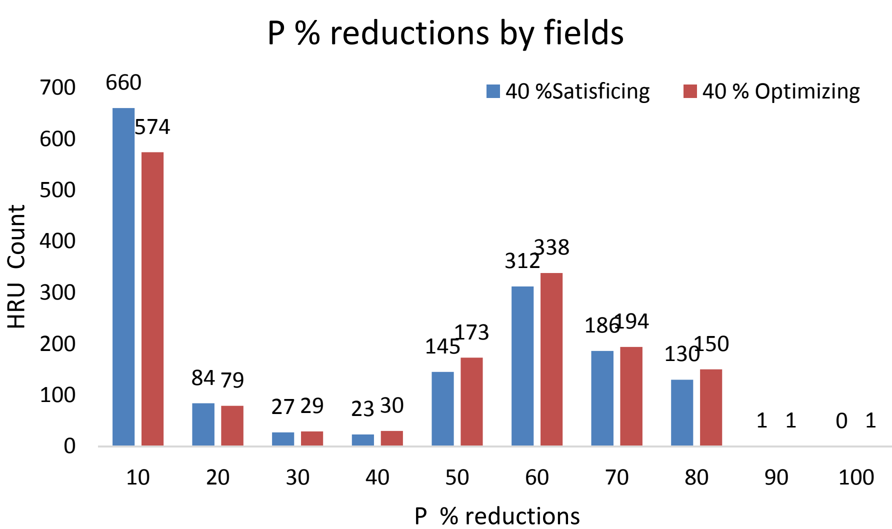


Figure B-10. Raccoon River Watershed: Distribution of abatement effort corresponding to trading outcomes, 40% P abatement goal.

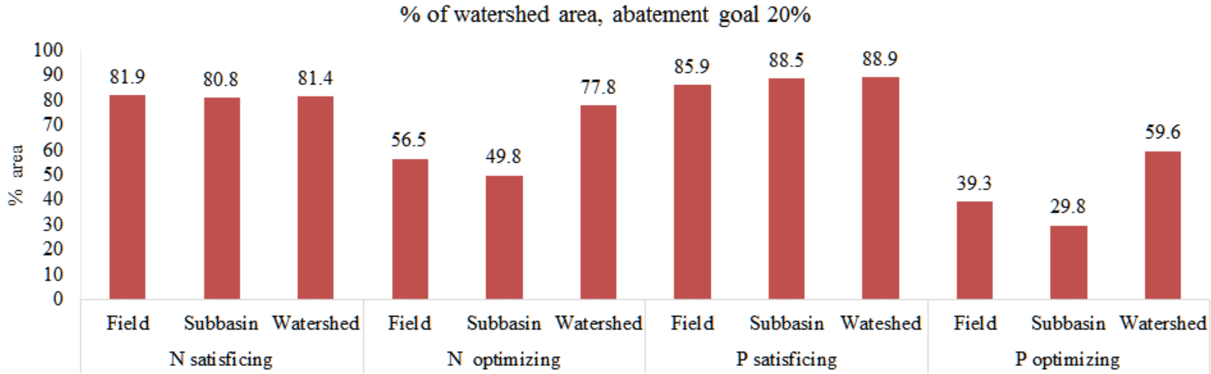


Figure B-11. Raccoon River Watershed: The overall change in the distribution of abatement actions under CAC, 20% abatement goal (the height of the bar represents the percent of the total area that changes the abatement actions from CAC).

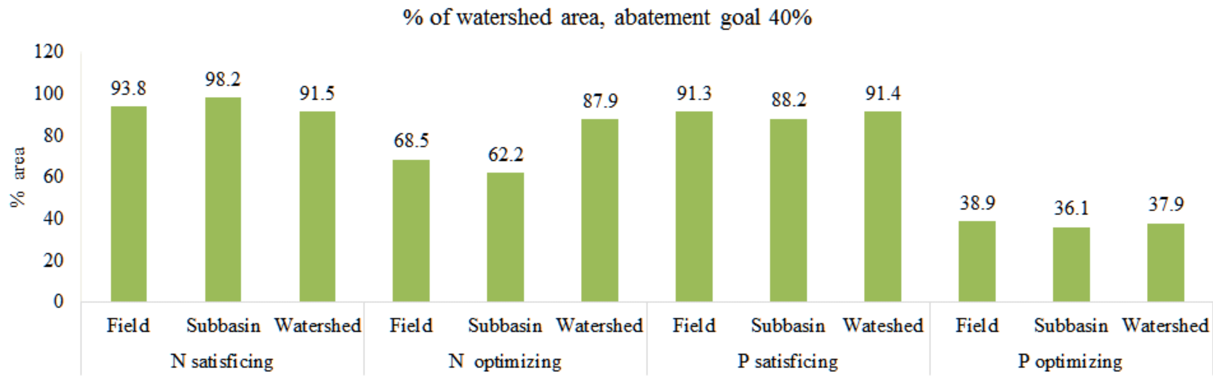


Figure B-12. Raccoon River Watershed: The overall change in the distribution of abatement actions under CAC, 40% abatement goal (the height of the bar represents the percent of the total area that changes the abatement actions from CAC).

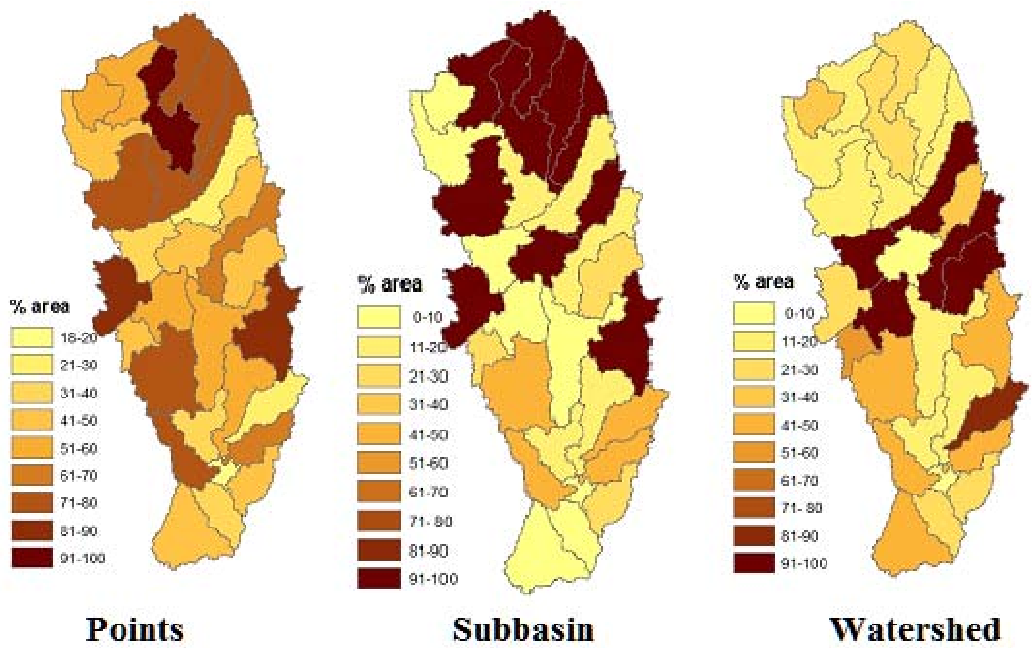


Figure B-13. Boone River Watershed: Spatial change in the CAC distribution under the three types of points, 30 % P abatement, optimizing.

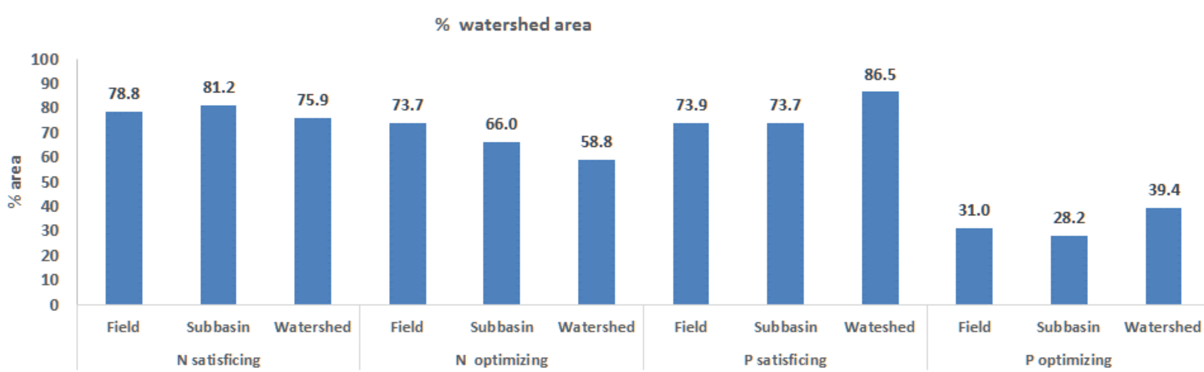


Figure B-14. Raccoon River Watershed: The overall change in the distribution of abatement actions under CAC, 20% abatement goal (the height of the bar represents the percent of the total area that changes the abatement actions from CAC).

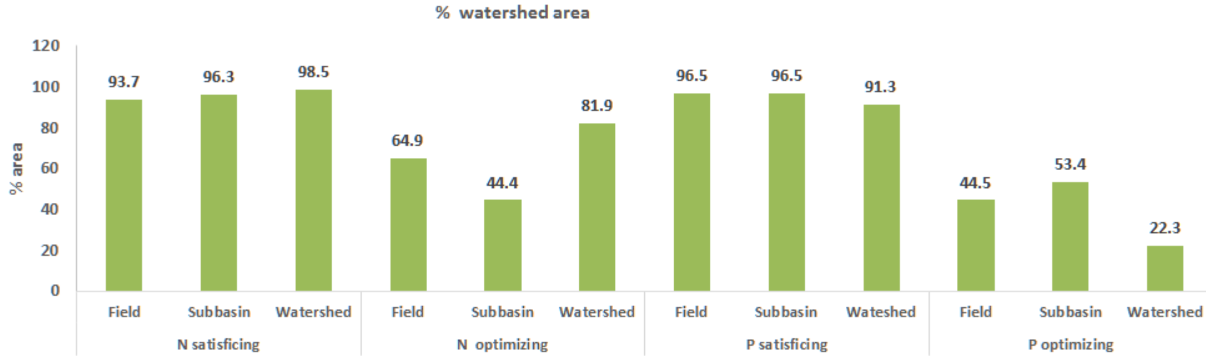


Figure B-15. Raccoon River Watershed: The overall change in the distribution of abatement actions under CAC, 40% abatement goal (the height of the bar represents the percent of the total area that changes the abatement actions from CAC).

Table B-14. Boone River Watershed: The five-year moving average 1999-2009 P loadings distribution

Abatement goal	Satisficing Policies			Optimizing Policies		
20% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (mil kg,P)	0.20	0.18	0.19	0.19	0.19	0.19
Std.dev. (mil kg,P)	0.03	0.02	0.03	0.03	0.03	0.03
Average N reduction (% P)	20.48	20.33	25.78	20.48	22.76	21.51
30% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (mil kg,P)	0.18	0.17	0.16	0.18	0.18	0.17
Std.dev. (mil kg,P)	0.02	0.02	0.02	0.02	0.02	0.02
Average N reduction (% P)	29.57	30.19	35	31.37	32.52	31.52
40% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (mil kg,P)	0.15	0.14	0.17	2.92	2.94	3.11
Std.dev. (mil kg,P)	0.02	0.02	0.02	0.46	0.46	0.48
Average N reduction (% P)	38.56	38.62	42.05	21.35	40.26	40.06

Table B-15. Raccoon River Watershed: The five-year moving average 1986-2003 P loadings distribution

20% goal	**Satisficing Policies**			**Optimizing Policies**		
	CAC	PS	Trading	CAC	PS	Trading
Mean (kg,P)	0.67	0.66	0.67	0.65	0.65	0.65
Std.dev. (kg,P)	0.10	0.10	0.10	0.09	0.09	0.09
Average N reduction (%P)	20.48	20.33	25.78	20.48	22.76	21.51
30% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (kg,P)	0.60	0.59	0.59	0.59	0.59	0.52
Std.dev. (kg,P)	0.08	0.08	0.09	0.08	0.08	0.08
Average N reduction (% P)	29.57	30.19	35	31.37	32.52	31.52
40% goal	CAC	PS	Trading	CAC	PS	Trading
Mean (kg,P)	0.51	0.50	0.54	0.52	0.52	0.52
Std.dev. (kg,P)	0.08	0.07	0.08	0.07	0.07	0.07
Average N reduction (% P)	38.56	38.62	42.05	21.35	40.26	40.06

Table B-16. Boone River Watershed: Simulated outcomes under cost heterogeneity and asymmetric information, nitrogen

	Command and Control and Performance Standard Outcomes,						Trading Outcomes, 20% goal					
	CAC, optimizing	PS,optimizing		CAC, satisficing	PS, satisficing		Optimizing Points			Satisficing Points		
	Cost, $	N red. %	Cost, $	Cost, $	N red. %	Cost	N red. %	Cost, $	Price, $	N red. %	Cost, $	Price, $
BRW, Nitrogen 20% goal												
Mean	1,792,679	20.9	1,667,155	7,231,175	25.7	5,760,977	19.6	715,220	1.9	22.1	939,440	2.34
Std.dev.	38,944	0.0	38,935	200,556	0.1	187,421	0.1	21,631	0.1	0.1	25,934	0.10
BRW, Nitrogen 40% goal												
Mean	9,010,815	40.4	8,906,493	29,573,330	43.1	28,510,856	39.3	5,054,855	8.6	41.4	6,020,182	9.7
Std.dev.	162,446	0.1	163,253	900,772	0.0	896,056.5	0.1	119,496	0.3	0.1	142,750	0.3

Table B-17. Boone River Watershed: Simulated outcomes under cost heterogeneity and asymmetric information, phosphorus

	Command and Control and Performance Standard Outcomes,						Trading Outcomes, 20% goal					
	CAC, optimizing	PS,optimizing		CAC, satisficing	PS, satisficing		Optimizing Points			Satisficing Points		
	Cost, $	P red. %	Cost, $	Cost, $	P red. %	Cost	P red. %	Cost, $	Price, $	P red. %	Cost, $	Price, $
BRW, Phosphorus, 20% goal												
Mean	1,238,579	21.7	1,070,494	9,448,262	26.7	3,705,462	21.8	399,318	19.8	20.3	337,180	18.05
Std.dev.	35,329	0.0	33,030	168,954	0.0	95,389	0.1	15,302	0.6	0.8	14,159	0.60
BRW, Phosphorus, 40% goal												
Mean	8,141,578	40.1	7,277,636	35,555,008	41.8	32,401,774	40.5	3,572,479	346.6	37.0	1,953,064	117.7
Std.dev.	301,178	1.9	300,986	1,100,507	0.0	1,091,362.6	0.1	124,408	27.9	0.1	38,755	4.3

Table B-18. Raccoon River Watershed: Simulated outcomes under cost heterogeneity and asymmetric information, nitrogen

	CAC, optimizing	PS,optimizing		CAC, satisficing	PS, satisficing		Optimizing Points			Satisficing Points		
	Cost, $	N red. %	Cost, $	Cost, $	N red. %	Cost	N red. %	Cost, $	Price, $	N red. %	Cost, $	Price, $
					RRW, Nitrogen 20% goal							
Mean	23,750,320	20.5	21,571,788	36,086,082	20.5	33,251,571	20.4	11,266,110	6.0	20.9	11,882,764	6.27
StdDev	427,516	0.0	413,365	1,091,005	0.2	1,071,269	0.1	265,843	0.2	0.2	282,784	0.21
					RRW, Nitrogen 40% goal							
Mean	75,161,975	40.4	70,818,633	130,796,757	40.4	126,862,504	39.1	44,119,054	14.8	39.4	45,015,774	15.0
StdDev	1,559,094	0.0	1,547,155	3,193,068	0.1	3,177,111.1	0.1	1,151,808	0.4	0.1	1,174,301	0.4

Table B-14. Raccoon River Watershed: Simulated outcomes under cost heterogeneity and asymmetric information, phosphorus

	CAC, optimizing	PS,optimizing		CAC, satisficing	PS, satisficing		Optimizing Points			Satisficing Points		
	Cost, $	P red. %	Cost, $	Cost, $	P red. %	Cost	P red. %	Cost, $	Price, $	P red. %	Cost, $	Price, $
					RRW, Phosphorus 20% goal							
Mean	7,688,685	21.9	7,520,398	36,562,597	22.3	33,242,238	21.6	3,655,285	52.4	18.7	2,759,776	41.69
StdDev	209,004	0.1	207,509	992,526	2.5	1,071,177	0.0	130,025	2.0	0.1	110,773	1.49
					RRW, Phosphorus 40% goal							
Mean	28,102,180	40.0	28,034,179	102,146,184	42.4	92,881,025	40.1	18,025,760	226.6	37.6	14,677,121	186.0
StdDev	550,132	0.0	550,373	2,521,243	0.1	2,512,351	0.0	456,049	6.7	0.0	386,730	5.6

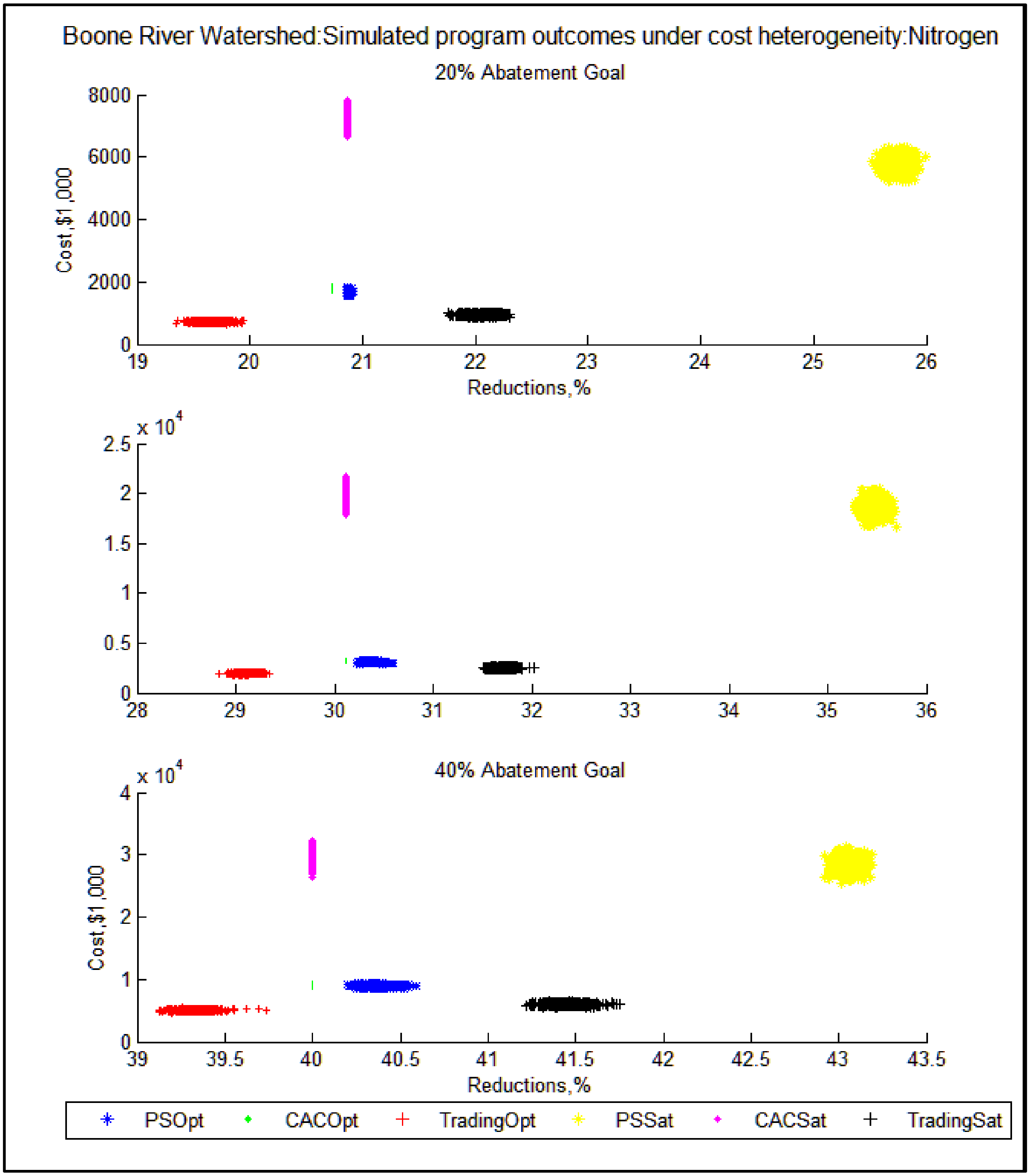


Figure B-16. Boone River Watershed: Simulated outcomes under cost heterogeneity, N abatement goals.

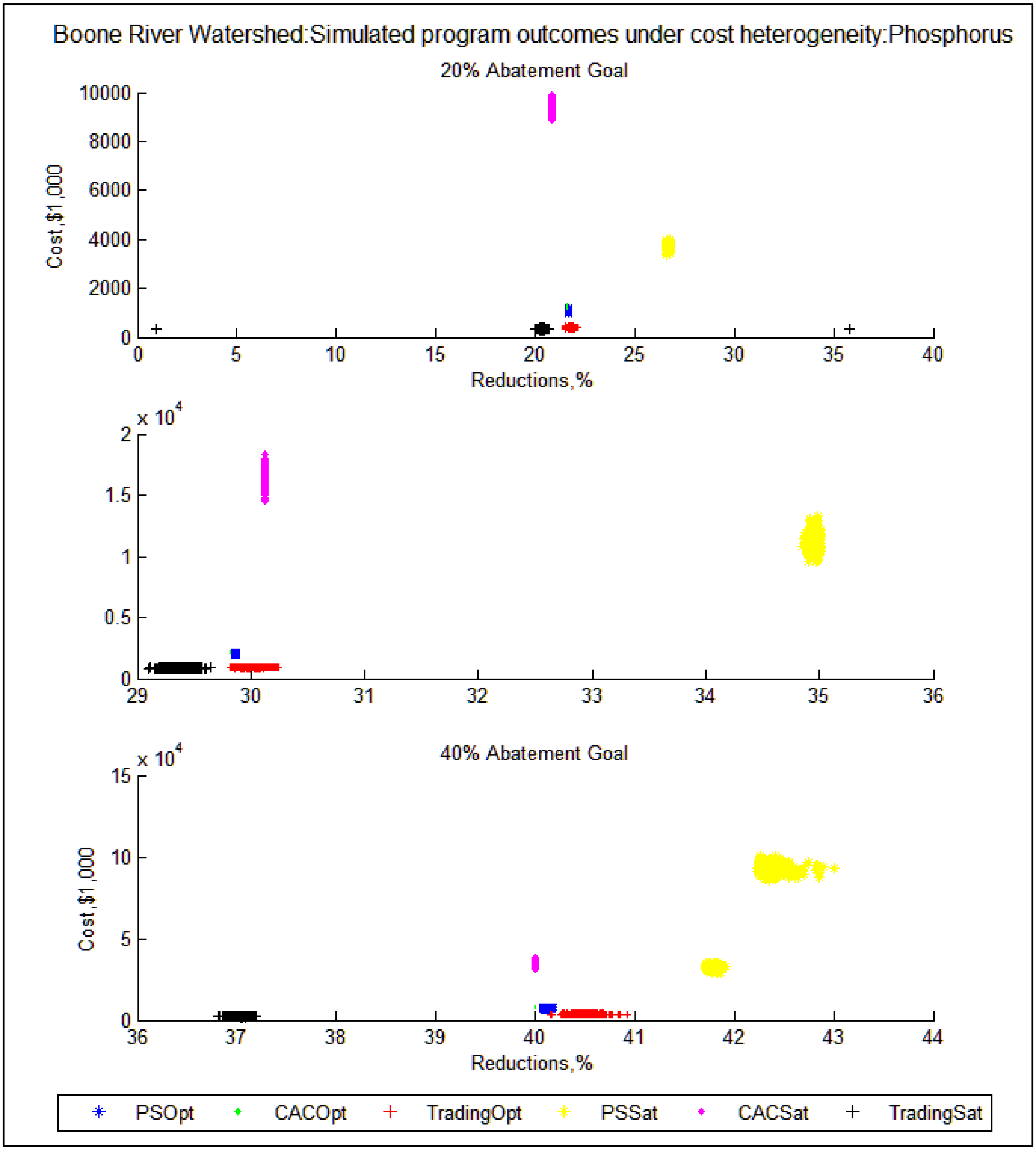


Figure B-17. Boone River Watershed: Simulated program outcomes under cost heterogeneity, P abatement goals.

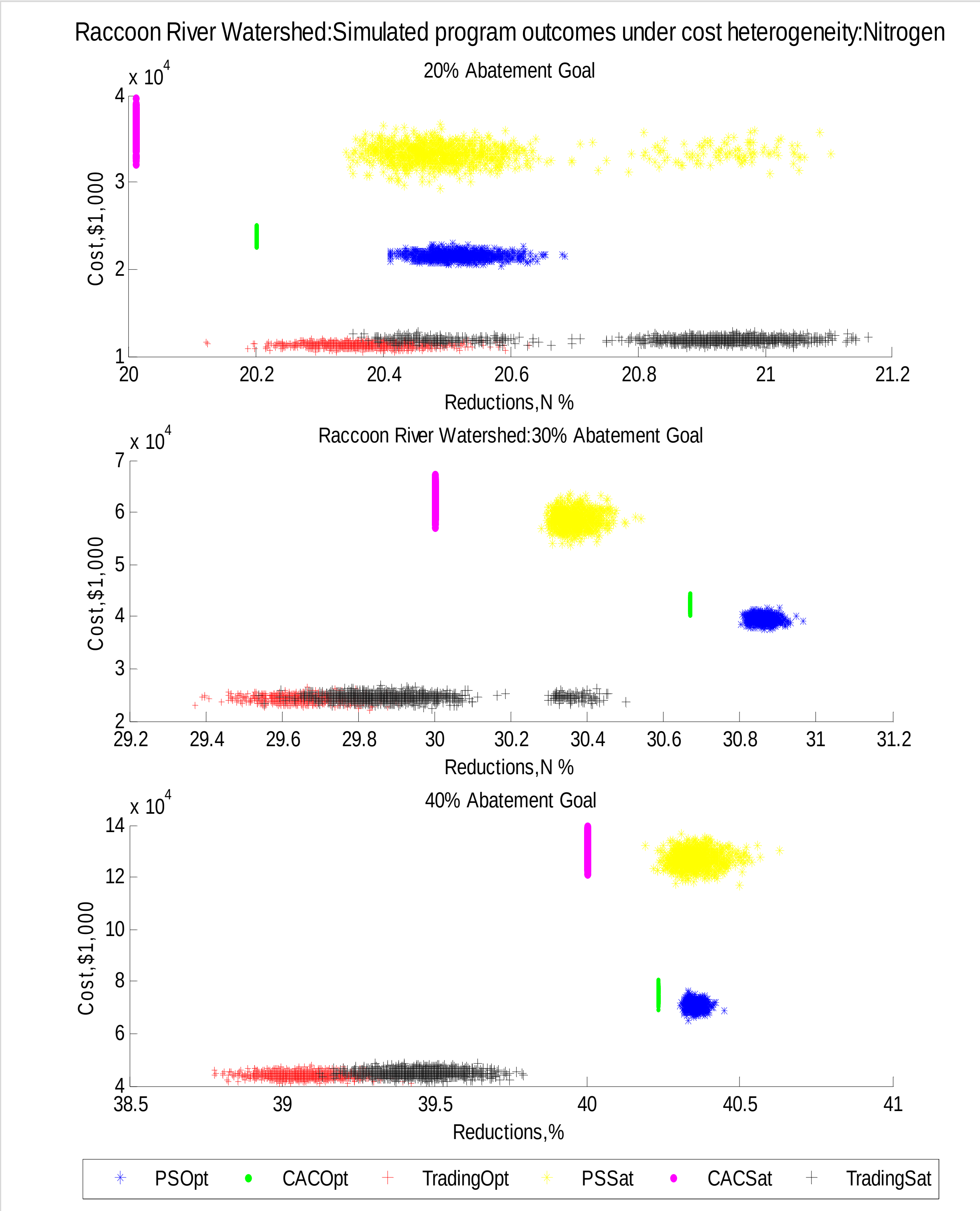


Figure B-18. Raccoon River Watershed Simulated program outcomes under cost heterogeneity, N abatement goals.

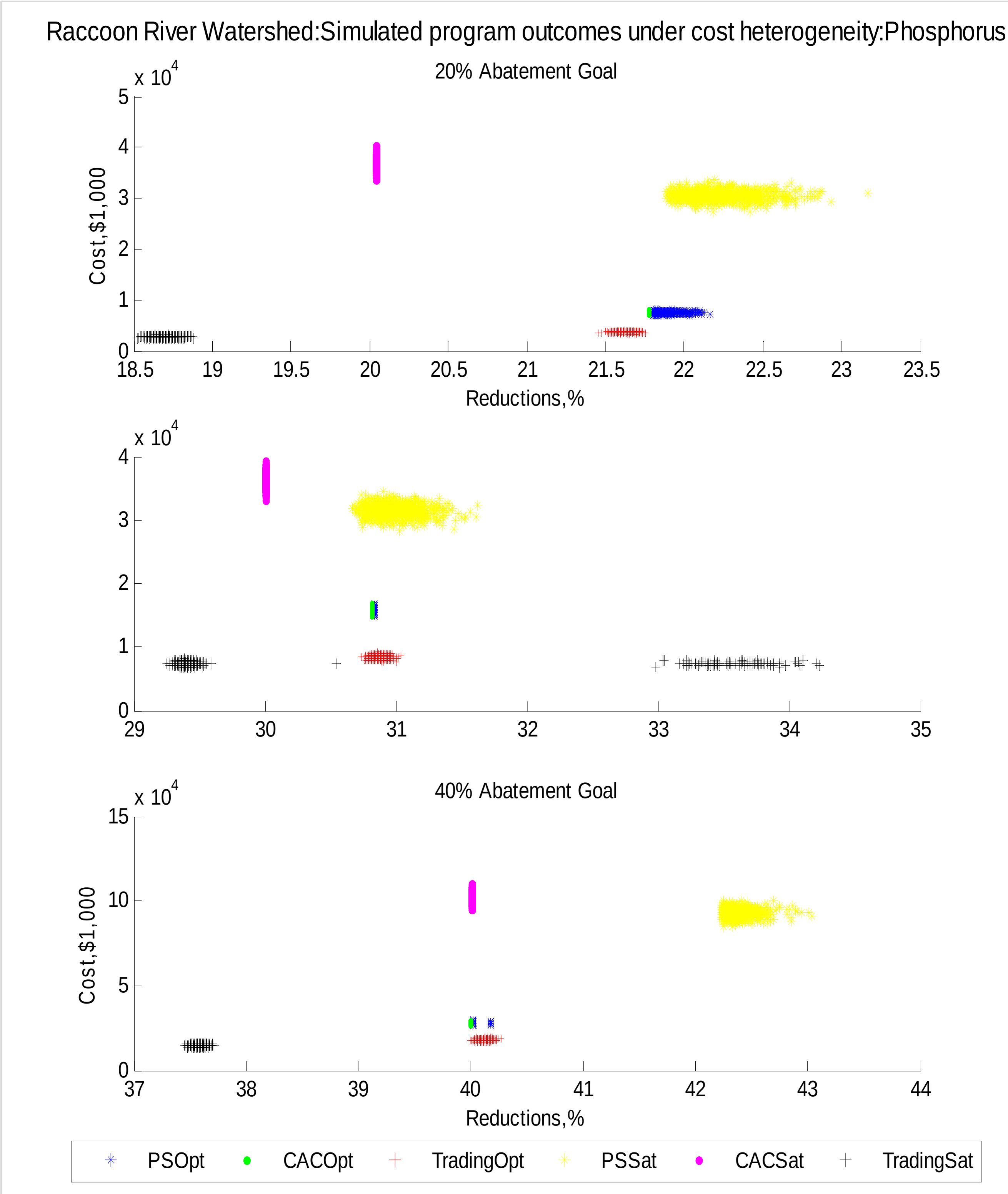


Figure B-19. Raccoon River Watershed: Simulated program outcomes under cost heterogeneity, phosphorus abatement goals.

Table B-20. Raccoon River Watershed: Multiple Pollutant Policies Approach

Abatement Target/CAC			Performance Standard			Point-Based Trading		
N	P	Total Cost	N	P	Total Cost	N	P	Total Cost
20%	20%	34,798,819	20.5	30.5	31,497,076	19.1	28.7	13,083,876
30%	30%	45,878,021	29.4	38.7	42,414,634	28.5	38.0	29,972,066
40%	40%	133,378,501	39.2	46.5	127,983,306	38.9	46.3	56,411,315

Table B-21. Raccoon River Watershed: Single Pollutant Point-Based Trading

Abatement	Nitrogen only Point-Based Trading			Phosphorus only Point-Based Trading		
N/P	N	*P*	Total Cost	*N*	P	Total Costs
20%	19.1	*28.7*	13,083,876	*5.9*	18.7	3,185,214
30%	28.5	*38.0*	29,972,066	*9.3*	28.2	7,905,358
40%	38.9	*46.3*	56,411,315	*12.9*	38.0	17,250,914

Table B-22. Boone River Watershed: Per period annual average distribution of nitrogen loadings, satisficing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
Abatement goal 20%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	6,127,114	4,789,830	4,859,812	4,557,974	21.83	20.68	25.61
1991–1995	6,169,094	4,817,656	4,877,292	4,570,282	21.91	20.94	25.92
1992–1996	4,897,034	3,772,430	3,813,088	3,605,312	22.97	22.13	26.38
1993–1997	4,751,094	3,621,582	3,692,788	3,481,006	23.77	22.27	26.73
1994–1998	3,990,314	3,148,892	3,166,504	2,991,100	21.09	20.65	25.04
1995–1999	4,592,024	3,706,796	3,737,448	3,484,086	19.28	18.61	24.13
1996–2000	4,207,464	3,361,406	3,397,586	3,168,310	20.11	19.25	24.70
1997–2001	4,795,664	3,786,320	3,846,404	3,564,974	21.05	19.79	25.66
1998–2002	4,695,550	3,703,142	3,757,870	3,481,396	21.14	19.97	25.86
1999–2003	4,129,190	3,273,026	3,315,480	3,065,136	20.73	19.71	25.77
2000–2004	4,336,090	3,397,726	3,462,660	3,195,806	21.64	20.14	26.30
2001–2005	4,702,210	3,684,604	3,744,528	3,440,588	21.64	20.37	26.83
2002–2006	3,907,158	3,105,060	3,130,632	2,887,734	20.53	19.87	26.09
2003–2007	4,893,212	3,873,232	3,917,558	3,624,202	20.84	19.94	25.93
2004–2008	6,163,132	4,847,158	4,921,012	4,565,968	21.35	20.15	25.91
2005–2009	5,448,292	4,301,074	4,349,266	4,060,960	21.06	20.17	25.46
Abatement goal 30%	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1990–1994	6,127,114	4,351,228	4,255,128	3,995,086	28.98	30.55	34.80
1991–1995	6,169,094	4,373,346	4,271,720	4,011,852	29.11	30.76	34.97
1992–1996	4,897,034	3,499,172	3,330,636	3,183,968	28.55	31.99	34.98
1993–1997	4,751,094	3,375,132	3,210,366	3,070,394	28.96	32.43	35.38
1994–1998	3,990,314	2,835,610	2,790,908	2,614,768	28.94	30.06	34.47
1995–1999	4,592,024	3,216,136	3,326,134	3,020,930	29.96	27.57	34.21
1996–2000	4,207,464	2,924,096	3,011,556	2,735,186	30.50	28.42	34.99
1997–2001	4,795,664	3,316,182	3,380,982	3,085,434	30.85	29.50	35.66
1998–2002	4,695,550	3,238,834	3,300,300	3,004,002	31.02	29.71	36.02
1999–2003	4,129,190	2,848,230	2,921,210	2,650,526	31.02	29.25	35.81
2000–2004	4,336,090	3,045,304	3,019,632	2,801,086	29.77	30.36	35.40
2001–2005	4,702,210	3,293,698	3,258,116	3,030,406	29.95	30.71	35.55
2002–2006	3,907,158	2,753,932	2,739,746	2,543,292	29.52	29.88	34.91
2003–2007	4,893,212	3,459,562	3,416,934	3,199,554	29.30	30.17	34.61
2004–2008	6,163,132	4,388,648	4,281,888	4,038,490	28.79	30.52	34.47
2005–2009	5,448,292	3,878,852	3,803,942	3,586,424	28.81	30.18	34.17
Abatement goal 40%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	6,127,114	3,830,846	3,804,134	3,601,614	37.48	37.91	41.22
1991–1995	6,169,094	3,840,244	3,824,694	3,610,076	37.75	38.00	41.48
1992–1996	4,897,034	3,075,842	3,001,530	2,868,304	37.19	38.71	41.43
1993–1997	4,751,094	2,959,510	2,874,264	2,764,274	37.71	39.50	41.82
1994–1998	3,990,314	2,490,050	2,461,086	2,348,906	37.60	38.32	41.13
1995–1999	4,592,024	2,810,490	2,898,008	2,694,724	38.80	36.89	41.32
1996–2000	4,207,464	2,543,956	2,614,896	2,434,454	39.54	37.85	42.14
1997–2001	4,795,664	2,880,882	2,937,342	2,747,890	39.93	38.75	42.70
1998–2002	4,695,550	2,807,214	2,857,508	2,670,162	40.22	39.14	43.13
1999–2003	4,129,190	2,462,804	2,533,542	2,349,856	40.36	38.64	43.09
2000–2004	4,336,090	2,638,188	2,621,130	2,475,712	39.16	39.55	42.90
2001–2005	4,702,210	2,854,296	2,832,146	2,676,284	39.30	39.77	43.08
2002–2006	3,907,158	2,387,586	2,368,434	2,237,808	38.89	39.38	42.73
2003–2007	4,893,212	3,019,582	2,984,614	2,833,736	38.29	39.01	42.09
2004–2008	6,163,132	3,816,804	3,770,646	3,575,940	38.07	38.82	41.98
2005–2009	5,448,292	3,388,088	3,375,646	3,196,228	37.81	38.04	41.34

Table B-23. Boone River Watershed: Per period annual average distribution of nitrogen loadings, optimizing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
Abatement goal 20%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	6,127,114	4,789,830	4,713,598	4,779,994	21.83	23.07	21.99
1991–1995	6,169,094	4,817,656	4,877,292	4,808,708	21.91	20.94	22.05
1992–1996	4,897,034	3,772,430	3,813,088	3,765,384	22.97	22.13	23.11
1993–1997	4,751,094	3,621,582	3,692,788	3,614,932	23.77	22.27	23.91
1994–1998	3,990,314	3,148,892	3,166,504	3,142,944	21.09	20.65	21.24
1995–1999	4,592,024	3,706,796	3,737,448	3,699,844	19.28	18.61	19.43
1996–2000	4,207,464	3,361,406	3,397,586	3,354,952	20.11	19.25	20.26
1997–2001	4,795,664	3,786,320	3,846,404	3,779,526	21.05	19.79	21.19
1998–2002	4,695,550	3,703,142	3,757,870	3,695,998	21.14	19.97	21.29
1999–2003	4,129,190	3,273,026	3,315,480	3,266,724	20.73	19.71	20.89
2000–2004	4,336,090	3,397,726	3,462,660	3,390,282	21.64	20.14	21.81
2001–2005	4,702,210	3,684,604	3,744,528	3,677,472	21.64	20.37	21.79
2002–2006	3,907,158	3,105,060	3,130,632	3,098,294	20.53	19.87	20.70
2003–2007	4,893,212	3,873,232	3,917,558	3,864,518	20.84	19.94	21.02
2004–2008	6,163,132	4,847,158	4,921,012	4,837,596	21.35	20.15	21.51
2005–2009	5,448,292	4,301,074	4,349,266	4,293,428	21.06	20.17	21.20
Abatement goal 30%	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1990–1994	6,127,114	4,789,830	4,713,598	4,779,994	21.83	23.07	21.99
1991–1995	6,169,094	4,817,656	4,877,292	4,808,708	21.91	20.94	22.05
1992–1996	4,897,034	3,772,430	3,813,088	3,765,384	22.97	22.13	23.11
1993–1997	4,751,094	3,621,582	3,692,788	3,614,932	23.77	22.27	23.91
1994–1998	3,990,314	3,148,892	3,166,504	3,142,944	21.09	20.65	21.24
1995–1999	4,592,024	3,706,796	3,737,448	3,699,844	19.28	18.61	19.43
1996–2000	4,207,464	3,361,406	3,397,586	3,354,952	20.11	19.25	20.26
1997–2001	4,795,664	3,786,320	3,846,404	3,779,526	21.05	19.79	21.19
1998–2002	4,695,550	3,703,142	3,757,870	3,695,998	21.14	19.97	21.29
1999–2003	4,129,190	3,273,026	3,315,480	3,266,724	20.73	19.71	20.89
2000–2004	4,336,090	3,397,726	3,462,660	3,390,282	21.64	20.14	21.81
2001–2005	4,702,210	3,684,604	3,744,528	3,677,472	21.64	20.37	21.79
2002–2006	3,907,158	3,105,060	3,130,632	3,098,294	20.53	19.87	20.70
2003–2007	4,893,212	3,873,232	3,917,558	3,864,518	20.84	19.94	21.02
2004–2008	6,163,132	4,847,158	4,921,012	4,837,596	21.35	20.15	21.51
2005–2009	5,448,292	4,301,074	4,349,266	4,293,428	21.06	20.17	21.20
Abatement goal 40%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	6,127,114	3,712,472	3,715,339	3,700,865	39.41	39.36	39.60
1991–1995	6,169,094	3,741,080	3,737,280	3,730,302	39.36	39.42	39.53
1992–1996	4,897,034	2,936,242	2,941,900	2,927,510	40.04	39.92	40.22
1993–1997	4,751,094	2,814,376	2,813,612	2,806,920	40.76	40.78	40.92
1994–1998	3,990,314	2,410,366	2,399,370	2,404,024	39.59	39.87	39.75
1995–1999	4,592,024	2,821,724	2,809,066	2,815,552	38.55	38.83	38.69
1996–2000	4,207,464	2,547,911	2,532,842	2,542,055	39.44	39.80	39.58
1997–2001	4,795,664	2,872,227	2,845,110	2,865,533	40.11	40.67	40.25
1998–2002	4,695,550	2,792,261	2,765,408	2,785,065	40.53	41.11	40.69
1999–2003	4,129,190	2,478,187	2,453,456	2,472,313	39.98	40.58	40.13
2000–2004	4,336,090	2,567,917	2,544,812	2,560,457	40.78	41.31	40.95
2001–2005	4,702,210	2,784,520	2,749,876	2,776,538	40.78	41.52	40.95
2002–2006	3,907,158	2,329,822	2,297,662	2,322,936	40.37	41.19	40.55
2003–2007	4,893,212	2,923,328	2,903,100	2,914,786	40.26	40.67	40.43
2004–2008	6,163,132	3,707,904	3,675,404	3,696,602	39.84	40.36	40.02
2005–2009	5,448,292	3,324,382	3,295,412	3,315,372	38.98	39.51	39.15

Table B-24. Boone River Watershed: Per period annual average distribution of phosphorus loadings, satisficing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
Abatement goal 20%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	281,336	230,078	206,926	221,654	18.22	26.45	21.21
1991–1995	288,470	237,302	214,312	228,752	17.74	25.71	20.70
1992–1996	246,506	207,258	188,036	197,516	15.92	23.72	19.87
1993–1997	249,838	209,594	191,234	199,684	16.11	23.46	20.07
1994–1998	197,998	166,674	154,238	159,218	15.82	22.10	19.59
1995–1999	210,966	173,746	162,956	169,748	17.64	22.76	19.54
1996–2000	198,948	164,118	154,464	160,422	17.51	22.36	19.36
1997–2001	235,192	193,282	180,960	189,398	17.82	23.06	19.47
1998–2002	229,538	191,892	178,020	185,224	16.40	22.44	19.31
1999–2003	209,122	175,314	162,588	169,700	16.17	22.25	18.85
2000–2004	225,102	192,280	176,014	183,580	14.58	21.81	18.45
2001–2005	244,472	208,554	190,766	199,760	14.69	21.97	18.29
2002–2006	198,988	172,922	157,958	163,734	13.10	20.62	17.72
2003–2007	241,300	206,000	188,894	196,582	14.63	21.72	18.53
2004–2008	306,696	260,098	236,902	248,412	15.19	22.76	19.00
2005–2009	276,976	234,032	214,788	224,188	15.50	22.45	19.06
Abatement goal 30%	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1990–1994	281,336	200,700	181,960	196,276	28.66	35.32	30.23
1991–1995	288,470	207,384	188,942	202,932	28.11	34.50	29.65
1992–1996	246,506	182,720	167,304	177,100	25.88	32.13	28.16
1993–1997	249,838	185,120	170,632	179,828	25.90	31.70	28.02
1994–1998	197,998	148,376	138,836	144,874	25.06	29.88	26.83
1995–1999	210,966	154,142	146,282	153,632	26.94	30.66	27.18
1996–2000	198,948	146,058	138,976	145,926	26.58	30.14	26.65
1997–2001	235,192	171,182	162,040	171,338	27.22	31.10	27.15
1998–2002	229,538	170,504	159,808	168,102	25.72	30.38	26.77
1999–2003	209,122	156,010	146,268	153,898	25.40	30.06	26.41
2000–2004	225,102	172,050	158,612	166,880	23.57	29.54	25.86
2001–2005	244,472	185,842	171,320	180,748	23.98	29.92	26.07
2002–2006	198,988	155,202	142,676	149,234	22.00	28.30	25.00
2003–2007	241,300	183,040	169,302	178,228	24.14	29.84	26.14
2004–2008	306,696	229,618	211,218	224,126	25.13	31.13	26.92
2005–2009	276,976	206,104	191,690	202,550	25.59	30.79	26.87
Abatement goal 40%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	281,336	166,764	160,206	173,276	40.72	43.06	38.41
1991–1995	288,470	172,196	165,508	180,248	40.31	42.63	37.52
1992–1996	246,506	151,870	146,340	160,168	38.39	40.63	35.02
1993–1997	249,838	153,640	148,488	163,358	38.50	40.57	34.61
1994–1998	197,998	125,744	122,236	134,202	36.49	38.26	32.22
1995–1999	210,966	131,412	128,654	141,494	37.71	39.02	32.93
1996–2000	198,948	124,864	122,548	135,126	37.24	38.40	32.08
1997–2001	235,192	145,248	142,142	157,154	38.24	39.56	33.18
1998–2002	229,538	143,734	139,980	155,070	37.38	39.02	32.44
1999–2003	209,122	131,996	128,410	141,664	36.88	38.60	32.26
2000–2004	225,102	143,054	138,378	153,244	36.45	38.53	31.92
2001–2005	244,472	154,518	149,070	165,330	36.80	39.02	32.37
2002–2006	198,988	129,570	124,840	137,976	34.89	37.26	30.66
2003–2007	241,300	153,004	147,668	163,416	36.59	38.80	32.28
2004–2008	306,696	190,618	183,610	204,078	37.85	40.13	33.46
2005–2009	276,976	172,926	167,008	185,676	37.57	39.70	32.96

Table B-25. Boone River Watershed: Per period annual average distribution of phosphorus loadings, optimizing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
Abatement goal 20%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	281,336	216,618	216,374	217,566	23.00	23.09	22.67
1991–1995	288,470	225,636	225,382	224,556	21.78	21.87	22.16
1992–1996	246,506	196,072	195,864	194,188	20.46	20.54	21.22
1993–1997	249,838	196,870	196,666	196,480	21.20	21.28	21.36
1994–1998	197,998	157,824	157,662	156,784	20.29	20.37	20.82
1995–1999	210,966	168,734	168,594	167,150	20.02	20.08	20.77
1996–2000	198,948	159,314	159,178	158,040	19.92	19.99	20.56
1997–2001	235,192	188,738	188,556	186,468	19.75	19.83	20.72
1998–2002	229,538	185,114	184,902	182,424	19.35	19.45	20.53
1999–2003	209,122	170,492	170,294	167,124	18.47	18.57	20.08
2000–2004	225,102	183,870	183,652	180,946	18.32	18.41	19.62
2001–2005	244,472	201,384	201,140	196,806	17.62	17.72	19.50
2002–2006	198,988	165,026	164,830	161,458	17.07	17.17	18.86
2003–2007	241,300	197,356	197,110	193,656	18.21	18.31	19.74
2004–2008	306,696	248,064	247,742	244,612	19.12	19.22	20.24
2005–2009	276,976	225,110	224,814	220,710	18.73	18.83	20.31
Abatement goal 30%	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1990–1994	281,336	192,726	192,588	194,414	31.50	31.55	30.90
1991–1995	288,470	200,450	200,304	201,084	30.51	30.56	30.29
1992–1996	246,506	176,304	176,184	175,696	28.48	28.53	28.73
1993–1997	249,838	178,826	178,718	178,470	28.42	28.47	28.57
1994–1998	197,998	145,798	145,726	143,952	26.36	26.40	27.30
1995–1999	210,966	154,750	154,682	152,600	26.65	26.68	27.67
1996–2000	198,948	147,332	147,282	144,988	25.94	25.97	27.12
1997–2001	235,192	173,018	172,962	170,154	26.44	26.46	27.65
1998–2002	229,538	170,624	170,544	167,006	25.67	25.70	27.24
1999–2003	209,122	156,726	156,646	152,872	25.06	25.09	26.90
2000–2004	225,102	168,320	168,206	165,756	25.23	25.28	26.36
2001–2005	244,472	181,702	181,552	179,472	25.68	25.74	26.59
2002–2006	198,988	149,736	149,592	148,252	24.75	24.82	25.50
2003–2007	241,300	178,394	178,238	176,922	26.07	26.13	26.68
2004–2008	306,696	224,440	224,248	222,414	26.82	26.88	27.48
2005–2009	276,976	204,992	204,836	201,054	25.99	26.05	27.41
Abatement goal 40%	Baseline	CAC	PS	Trading	CAC	PS	Trading
1990–1994	281,336	164,206	163,800	165,028	41.63	41.78	41.34
1991–1995	288,470	171,448	170,996	172,092	40.57	40.72	40.34
1992–1996	246,506	152,778	152,438	153,724	38.02	38.16	37.64
1993–1997	249,838	155,334	155,062	156,716	37.83	37.93	37.27
1994–1998	197,998	128,328	128,200	129,488	35.19	35.25	34.60
1995–1999	210,966	134,882	134,770	136,300	36.06	36.12	35.39
1996–2000	198,948	128,992	128,942	130,142	35.16	35.19	34.58
1997–2001	235,192	149,996	149,880	151,082	36.22	36.27	35.76
1998–2002	229,538	148,278	148,096	149,274	35.40	35.48	34.97
1999–2003	209,122	136,238	136,006	136,464	34.85	34.96	34.74
2000–2004	225,102	146,668	146,378	147,076	34.84	34.97	34.66
2001–2005	244,472	157,962	157,598	158,536	35.39	35.54	35.15
2002–2006	198,988	131,992	131,678	132,484	33.67	33.83	33.42
2003–2007	241,300	155,914	155,614	156,606	35.39	35.51	35.10
2004–2008	306,696	194,348	194,012	194,928	36.63	36.74	36.44
2005–2009	276,976	177,826	177,570	178,084	35.80	35.89	35.70

Table B-26. Raccoon River Watershed: Per period annual average distribution of nitrogen loadings, satisficing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	17,244,174	13,849,844	12,334,732	13,777,676	19.68	28.47	20.10
1987–1991	17,795,874	14,596,664	12,603,988	14,475,556	17.98	29.17	18.66
1988–1992	18,028,676	14,911,240	12,736,352	14,704,710	17.29	29.36	18.44
1989–1993	24,457,146	19,874,124	17,497,898	19,893,040	18.74	28.45	18.66
1990–1994	24,611,696	20,010,434	17,576,928	20,073,484	18.70	28.58	18.44
1991–1995	20,968,048	16,967,726	14,555,486	17,101,808	19.08	30.58	18.44
1992–1996	20,019,328	16,340,018	14,212,436	16,303,548	18.38	29.01	18.56
1993–1997	19,696,322	16,201,910	13,887,264	16,125,052	17.74	29.49	18.13
1994–1998	17,703,074	14,971,026	12,186,198	14,628,482	15.43	31.16	17.37
1995–1999	22,339,262	18,969,342	15,333,068	18,536,892	15.09	31.36	17.02
1996–2000	18,902,572	16,055,141	13,200,789	15,699,483	15.06	30.16	16.95
1997–2001	19,498,832	16,497,943	13,532,381	16,174,935	15.39	30.60	17.05
1998–2002	19,738,886	17,079,815	13,824,499	16,633,115	13.47	29.96	15.73
1999–2003	18,692,534	15,946,429	13,076,435	15,607,009	14.69	30.04	16.51
1999–2003	19,710,314	16,736,269	14,003,181	16,422,907	15.09	28.96	16.68
Abatement goal 20%							
	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1986–1990	17,244,174	12,357,120	12,334,732	12,458,452	28.34	28.47	27.75
1987–1991	17,795,874	12,629,362	12,603,988	12,844,970	29.03	29.17	27.82
1988–1992	18,028,676	12,759,152	12,736,352	12,960,410	29.23	29.36	28.11
1989–1993	24,457,146	17,546,220	17,497,898	17,918,348	28.26	28.45	26.74
1990–1994	24,611,696	17,630,360	17,576,928	18,032,214	28.37	28.58	26.73
1991–1995	20,968,048	14,581,136	14,555,486	15,145,768	30.46	30.58	27.77
1992–1996	20,019,328	14,261,614	14,212,436	14,611,614	28.76	29.01	27.01
1993–1997	19,696,322	13,930,360	13,887,264	14,328,280	29.27	29.49	27.25
1994–1998	17,703,074	12,247,080	12,186,198	12,577,466	30.82	31.16	28.95
1995–1999	22,339,262	15,393,508	15,333,068	15,902,044	31.09	31.36	28.82
1996–2000	18,902,572	13,259,823	13,200,789	13,586,105	29.85	30.16	28.13
1997–2001	19,498,832	13,572,661	13,532,381	13,989,765	30.39	30.60	28.25
1998–2002	19,738,886	13,873,489	13,824,499	14,347,193	29.71	29.96	27.32
1999–2003	18,692,534	13,087,113	13,076,435	13,499,435	29.99	30.04	27.78
1999–2003	19,710,314	14,044,295	14,003,181	14,360,257	28.75	28.96	27.14
Abatement goal 40%							
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	17,244,174	10,623,488	10,919,790	10,871,738	38.39	36.68	36.95
1987–1991	17,795,874	10,993,204	11,131,086	11,090,088	38.23	37.45	37.68
1988–1992	18,028,676	11,247,356	11,163,668	11,127,010	37.61	38.08	38.28
1989–1993	24,457,146	14,865,246	15,651,546	15,583,958	39.22	36.00	36.28
1990–1994	24,611,696	14,965,832	15,699,474	15,635,096	39.19	36.21	36.47
1991–1995	20,968,048	12,809,666	12,948,212	12,895,086	38.91	38.25	38.50
1992–1996	20,019,328	12,222,590	12,653,326	12,596,672	38.95	36.79	37.08
1993–1997	19,696,322	11,958,660	12,300,768	12,241,008	39.28	37.55	37.85
1994–1998	17,703,074	10,939,178	10,534,222	10,492,188	38.21	40.49	40.73
1995–1999	22,339,262	13,712,928	13,220,346	13,171,682	38.62	40.82	41.04
1996–2000	18,902,572	11,700,009	11,458,669	11,418,432	38.10	39.38	39.59
1997–2001	19,498,832	12,160,595	11,746,289	11,709,484	37.63	39.76	39.95
1998–2002	19,738,886	12,497,439	11,999,501	11,962,564	36.69	39.21	39.40
1999–2003	18,692,534	11,733,771	11,333,823	11,297,276	37.23	39.37	39.56
1999–2003	19,710,314	12,279,533	12,219,611	12,174,106	37.70	38.00	38.23

Table B-27. Raccoon River Watershed: Per period annual average distribution of nitrogen loadings, optimizing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
	Abatement goal 20%						
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	17,244,174	14,116,902	14,017,700	13,861,980	18.14	18.71	19.61
1987–1991	17,795,874	14,510,322	14,478,648	14,561,100	18.46	18.64	18.18
1988–1992	18,028,676	14,636,940	14,631,580	14,793,512	18.81	18.84	17.94
1989–1993	24,457,146	19,954,844	19,887,876	20,007,850	18.41	18.68	18.19
1990–1994	24,611,696	20,103,308	20,031,982	20,190,922	18.32	18.61	17.96
1991–1995	20,968,048	17,064,310	17,016,948	17,210,078	18.62	18.84	17.92
1992–1996	20,019,328	16,433,888	16,355,516	16,396,188	17.91	18.30	18.10
1993–1997	19,696,322	16,175,290	16,112,194	16,213,512	17.88	18.20	17.68
1994–1998	17,703,074	14,453,718	14,435,746	14,704,696	18.35	18.46	16.94
1995–1999	22,339,262	18,178,998	18,183,438	18,628,542	18.62	18.60	16.61
1996–2000	18,902,572	15,454,422	15,442,989	15,776,623	18.24	18.30	16.54
1997–2001	19,498,832	15,866,104	15,891,393	16,255,965	18.63	18.50	16.63
1998–2002	19,738,886	16,301,496	16,323,957	16,705,099	17.41	17.30	15.37
1999–2003	18,692,534	15,448,476	15,463,313	15,689,963	17.35	17.28	16.06
1999–2003	19,710,314	16,431,252	16,410,113	16,514,389	16.64	16.74	16.21
	Abatement goal 30%						
	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1986–1990	17,244,174	13,153,446	13,040,808	12,489,308	23.72	24.38	27.57
1987–1991	17,795,874	12,937,454	12,942,392	12,879,816	27.30	27.27	27.62
1988–1992	18,028,676	12,789,324	12,849,622	12,997,656	29.06	28.73	27.91
1989–1993	24,457,146	18,251,726	18,225,930	17,957,028	25.37	25.48	26.58
1990–1994	24,611,696	18,298,526	18,287,016	18,073,616	25.65	25.70	26.56
1991–1995	20,968,048	14,963,538	15,009,420	15,191,024	28.64	28.42	27.55
1992–1996	20,019,328	14,684,682	14,678,860	14,651,832	26.65	26.68	26.81
1993–1997	19,696,322	14,191,274	14,202,180	14,372,420	27.95	27.89	27.03
1994–1998	17,703,074	11,771,980	11,870,920	12,628,106	33.50	32.94	28.67
1995–1999	22,339,262	14,738,396	14,877,078	15,963,172	34.02	33.40	28.54
1996–2000	18,902,572	12,801,729	12,881,545	13,630,562	32.28	31.85	27.89
1997–2001	19,498,832	13,051,717	13,187,693	14,035,438	33.06	32.37	28.02
1998–2002	19,738,886	13,403,367	13,547,591	14,390,082	32.10	31.37	27.10
1999–2003	18,692,534	12,836,977	12,952,065	13,537,376	31.33	30.71	27.58
1999–2003	19,710,314	14,099,093	14,145,015	14,393,114	28.47	28.24	26.98
	Abatement goal 40%						
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	17,244,174	11,248,238	11,171,568	10,919,790	34.77	35.22	36.68
1987–1991	17,795,874	11,238,692	11,222,944	11,131,086	36.85	36.94	37.45
1988–1992	18,028,676	11,222,552	11,231,380	11,163,668	37.75	37.70	38.08
1989–1993	24,457,146	15,830,840	15,782,404	15,651,546	35.27	35.47	36.00
1990–1994	24,611,696	15,814,590	15,776,818	15,699,474	35.74	35.90	36.21
1991–1995	20,968,048	12,843,692	12,866,058	12,948,212	38.75	38.64	38.25
1992–1996	20,019,328	12,655,824	12,642,726	12,653,326	36.78	36.85	36.79
1993–1997	19,696,322	12,163,306	12,173,888	12,300,768	38.25	38.19	37.55
1994–1998	17,703,074	10,103,164	10,167,608	10,534,222	42.93	42.57	40.49
1995–1999	22,339,262	12,508,072	12,596,162	13,220,346	44.01	43.61	40.82
1996–2000	18,902,572	10,949,788	10,996,029	11,458,669	42.07	41.83	39.38
1997–2001	19,498,832	11,152,630	11,230,935	11,746,289	42.80	42.40	39.76
1998–2002	19,738,886	11,389,622	11,472,381	11,999,501	42.30	41.88	39.21
1999–2003	18,692,534	10,732,466	10,819,707	11,333,823	42.58	42.12	39.37
1999–2003	19,710,314	11,710,010	11,755,429	12,219,611	40.59	40.36	38.00

Table B-28. Raccoon River Watershed: Per period annual average distribution of phosphorus loadings, satisficing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
	Abatement goal 20%						
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	720,320	612,204	562,160	562,160	15.01	21.96	21.96
1987–1991	725,760	600,664	587,420	587,420	17.24	19.06	19.06
1988–1992	741,640	603,624	602,960	602,960	18.61	18.70	18.70
1989–1993	1,018,780	842,444	824,240	824,240	17.31	19.10	19.10
1990–1994	1,029,920	848,230	842,300	842,300	17.64	18.22	18.22
1991–1995	979,440	795,390	812,460	812,460	18.79	17.05	17.05
1992–1996	906,560	744,790	748,480	748,480	17.84	17.44	17.44
1993–1997	872,080	713,710	722,120	722,120	18.16	17.20	17.20
1994–1998	685,140	547,770	581,020	581,020	20.05	15.20	15.20
1995–1999	794,900	632,660	660,480	660,480	20.41	16.91	16.91
1996–2000	680,570	545,896	566,602	566,602	19.79	16.75	16.75
1997–2001	772,830	616,616	625,782	625,782	20.21	19.03	19.03
1998–2002	742,370	599,796	603,142	603,142	19.21	18.75	18.75
1999–2003	784,310	634,636	625,442	625,442	19.08	20.26	20.26
1999–2003	828,070	681,396	668,282	668,282	17.71	19.30	19.30
	Abatement goal 30%						
	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1986–1990	720,320	515,866	508,068	500,598	28.38	29.47	30.50
1987–1991	725,760	528,446	524,028	523,538	27.19	27.80	27.86
1988–1992	741,640	542,326	538,948	536,438	26.87	27.33	27.67
1989–1993	1,018,780	735,680	728,036	729,766	27.79	28.54	28.37
1990–1994	1,029,920	746,384	739,040	745,934	27.53	28.24	27.57
1991–1995	979,440	707,324	702,740	717,194	27.78	28.25	26.78
1992–1996	906,560	662,364	656,460	661,574	26.94	27.59	27.02
1993–1997	872,080	636,324	631,180	635,794	27.03	27.62	27.09
1994–1998	685,140	513,084	510,200	514,294	25.11	25.53	24.94
1995–1999	794,900	584,680	580,460	578,760	26.45	26.98	27.19
1996–2000	680,570	506,806	501,866	498,680	25.53	26.26	26.73
1997–2001	772,830	565,826	560,966	548,760	26.79	27.41	28.99
1998–2002	742,370	551,526	545,346	533,420	25.71	26.54	28.15
1999–2003	784,310	574,266	567,206	551,000	26.78	27.68	29.75
1999–2003	828,070	614,386	609,266	593,600	25.81	26.42	28.32
	Abatement goal 40%						
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	720,320	488,134	455,870	463,836	32.23	36.71	35.61
1987–1991	725,760	466,094	452,870	479,776	35.78	37.60	33.89
1988–1992	741,640	466,294	460,990	489,116	37.13	37.84	34.05
1989–1993	1,018,780	652,354	627,184	667,198	35.97	38.44	34.51
1990–1994	1,029,920	656,148	633,722	680,706	36.29	38.47	33.91
1991–1995	979,440	606,548	594,762	649,526	38.07	39.28	33.68
1992–1996	906,560	573,068	557,542	602,526	36.79	38.50	33.54
1993–1997	872,080	541,928	530,182	577,426	37.86	39.20	33.79
1994–1998	685,140	409,688	414,902	462,166	40.20	39.44	32.54
1995–1999	794,900	470,800	471,640	518,620	40.77	40.67	34.76
1996–2000	680,570	409,054	408,564	447,656	39.90	39.97	34.22
1997–2001	772,830	459,714	462,064	493,316	40.52	40.21	36.17
1998–2002	742,370	455,274	454,842	483,016	38.67	38.73	34.94
1999–2003	784,310	483,294	479,162	500,916	38.38	38.91	36.13
1999–2003	828,070	527,094	518,082	544,156	36.35	37.43	34.29

Table B-29. Raccoon River Watershed: Per period annual average distribution of phosphorus loadings, optimizing policies

	Average Annual N loadings (kg)				Annual Realized Reductions (%)		
	Abatement goal 20%						
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	720,320	549,722	549,380	543,732	23.68	23.73	24.52
1987–1991	725,760	575,902	575,500	569,612	20.65	20.70	21.52
1988–1992	741,640	591,502	591,340	584,712	20.24	20.27	21.16
1989–1993	1,018,780	803,262	802,780	798,492	21.15	21.20	21.62
1990–1994	1,029,920	818,980	818,540	816,080	20.48	20.52	20.76
1991–1995	979,440	788,040	787,560	786,960	19.54	19.59	19.65
1992–1996	906,560	727,680	727,320	724,440	19.73	19.77	20.09
1993–1997	872,080	703,000	702,580	697,480	19.39	19.44	20.02
1994–1998	685,140	567,560	567,460	562,160	17.16	17.18	17.95
1995–1999	794,900	643,340	643,100	637,420	19.07	19.10	19.81
1996–2000	680,570	552,558	552,338	548,006	18.81	18.84	19.48
1997–2001	772,830	613,218	612,798	604,826	20.65	20.71	21.74
1998–2002	742,370	590,818	590,338	584,726	20.41	20.48	21.24
1999–2003	784,310	610,358	609,738	606,546	22.18	22.26	22.67
1999–2003	828,070	652,398	651,758	650,686	21.21	21.29	21.42
				Abatement goal 30%			
	Optimizing	CAC	PS	Trading	CAC	PS	Trading
1986–1990	720,320	489,834	489,618	495,912	32.00	32.03	31.15
1987–1991	725,760	515,614	515,538	516,272	28.96	28.97	28.86
1988–1992	741,640	531,914	531,838	528,452	28.28	28.29	28.75
1989–1993	1,018,780	711,530	711,312	718,584	30.16	30.18	29.47
1990–1994	1,029,920	725,184	724,994	734,514	29.59	29.61	28.68
1991–1995	979,440	698,604	698,514	704,534	28.67	28.68	28.07
1992–1996	906,560	651,204	651,094	650,974	28.17	28.18	28.19
1993–1997	872,080	629,844	629,774	624,694	27.78	27.78	28.37
1994–1998	685,140	519,884	519,894	505,354	24.12	24.12	26.24
1995–1999	794,900	585,940	585,920	567,780	26.29	26.29	28.57
1996–2000	680,570	503,924	503,884	489,430	25.96	25.96	28.09
1997–2001	772,830	562,204	562,184	537,930	27.25	27.26	30.39
1998–2002	742,370	542,884	542,784	524,050	26.87	26.88	29.41
1999–2003	784,310	558,524	558,444	540,910	28.79	28.80	31.03
1999–2003	828,070	599,464	599,384	584,850	27.61	27.62	29.37
				Abatement goal 40%			
	Baseline	CAC	PS	Trading	CAC	PS	Trading
1986–1990	720,320	435,630	435,634	450,678	39.52	39.52	37.43
1987–1991	725,760	456,970	456,974	465,198	37.04	37.04	35.90
1988–1992	741,640	471,790	471,794	473,558	36.39	36.39	36.15
1989–1993	1,018,780	635,282	635,284	646,676	37.64	37.64	36.52
1990–1994	1,029,920	646,576	646,576	659,070	37.22	37.22	36.01
1991–1995	979,440	620,476	620,476	627,930	36.65	36.65	35.89
1992–1996	906,560	581,676	581,696	583,910	35.84	35.83	35.59
1993–1997	872,080	559,156	559,176	559,750	35.88	35.88	35.81
1994–1998	685,140	454,216	454,236	446,150	33.70	33.70	34.88
1995–1999	794,900	509,480	509,500	500,420	35.91	35.90	37.05
1996–2000	680,570	442,968	442,988	432,494	34.91	34.91	36.45
1997–2001	772,830	487,788	487,808	477,514	36.88	36.88	38.21
1998–2002	742,370	475,418	475,438	468,242	35.96	35.96	36.93
1999–2003	784,310	493,378	493,398	487,282	37.09	37.09	37.87
1999–2003	828,070	530,078	530,098	529,922	35.99	35.98	36.01

Tables and Figures for Chapter 5

Table C-1. Point permit price in the presence of a carbon market, nitrogen

Nitrogen							
		$0		$5		$15	
Target	$ReqP^{PBT}$, $	$ReqP^{PBTC}$, $	$\Delta ReqP^{C}$, %	$ReqP^{PBTC}$, $	$\Delta ReqP^{C}$, %	$ReqP^{PBTC}$, $	$\Delta ReqP^{C}$, %
20%	2.3	1.7	25.6	0.6	73.2	0	100.0
30%	5.1	4.2	16.6	2.5	50.5	1	83.9
40%	9.7	9.0	7.1	7.7	21.1	6	34.8

Table C-2. Point permit price in the presence of a carbon market, phosphorus

Phosphorus								
	$0	$5		$15		$25	
Target	$ReqP^{PBT}$, $	$ReqP^{PBTC}$, $	$\Delta ReqP^{C}$, %	$ReqP^{PBTC}$, $	$\Delta ReqP^{C}$, %	$ReqP^{PBTC}$, $	$\Delta ReqP^{C}$, %
20%	18.1	12.6	30.1	3.6	79.9	0.0	100.0
30%	35.9	26.5	26.2	11.4	68.3	1.2	96.7
40%	117.7	94.7	19.5	52.2	55.7	21.6	81.6

Table C-3. Hotspots in the presence of a carbon market

Price		$0	$5	$15	$25
	20%	0	15	7	6
N abatement	30%	0	6	4	6
	40%	0	4	4	4
	20%	9	102	159	160
P abatement	30%	8	79	158	160
	40%	8	74	153	158

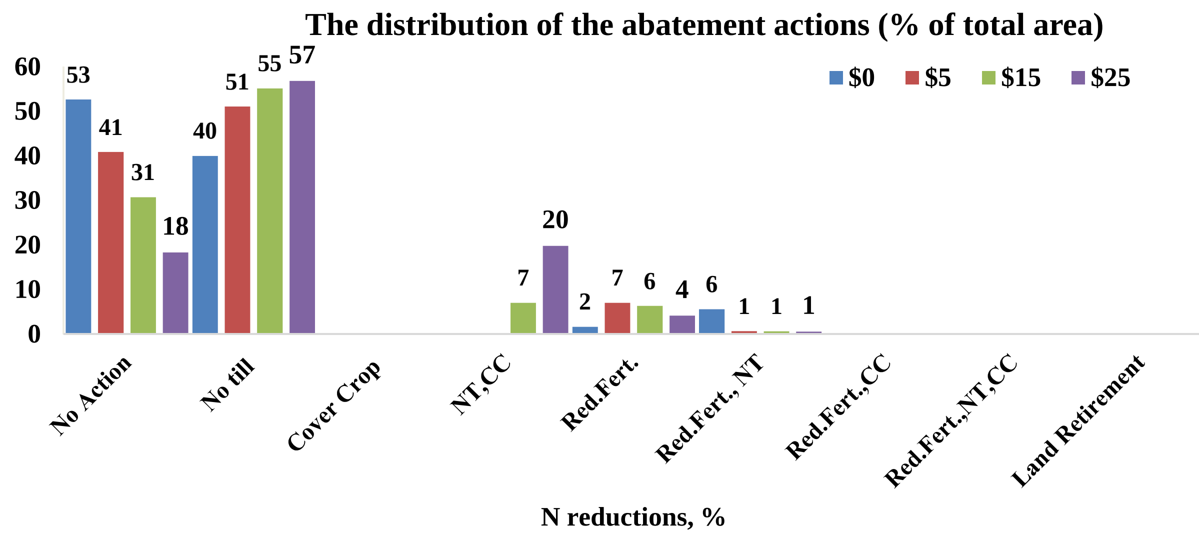


Figure C-1. The distribution of the abatement actions 30% P abatement goal.

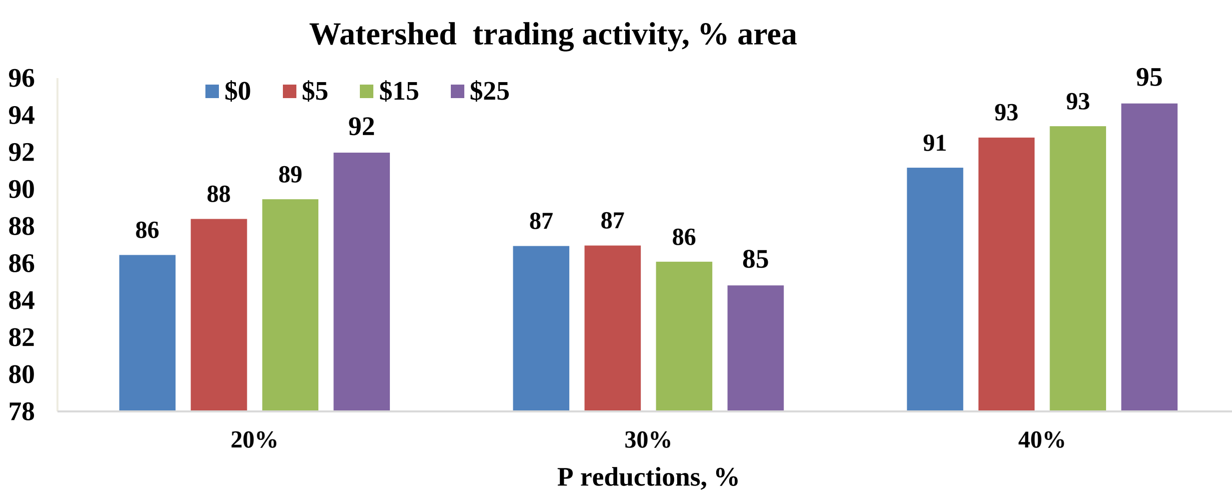


Figure C-2. The overall trading activity, phosphorus abatement

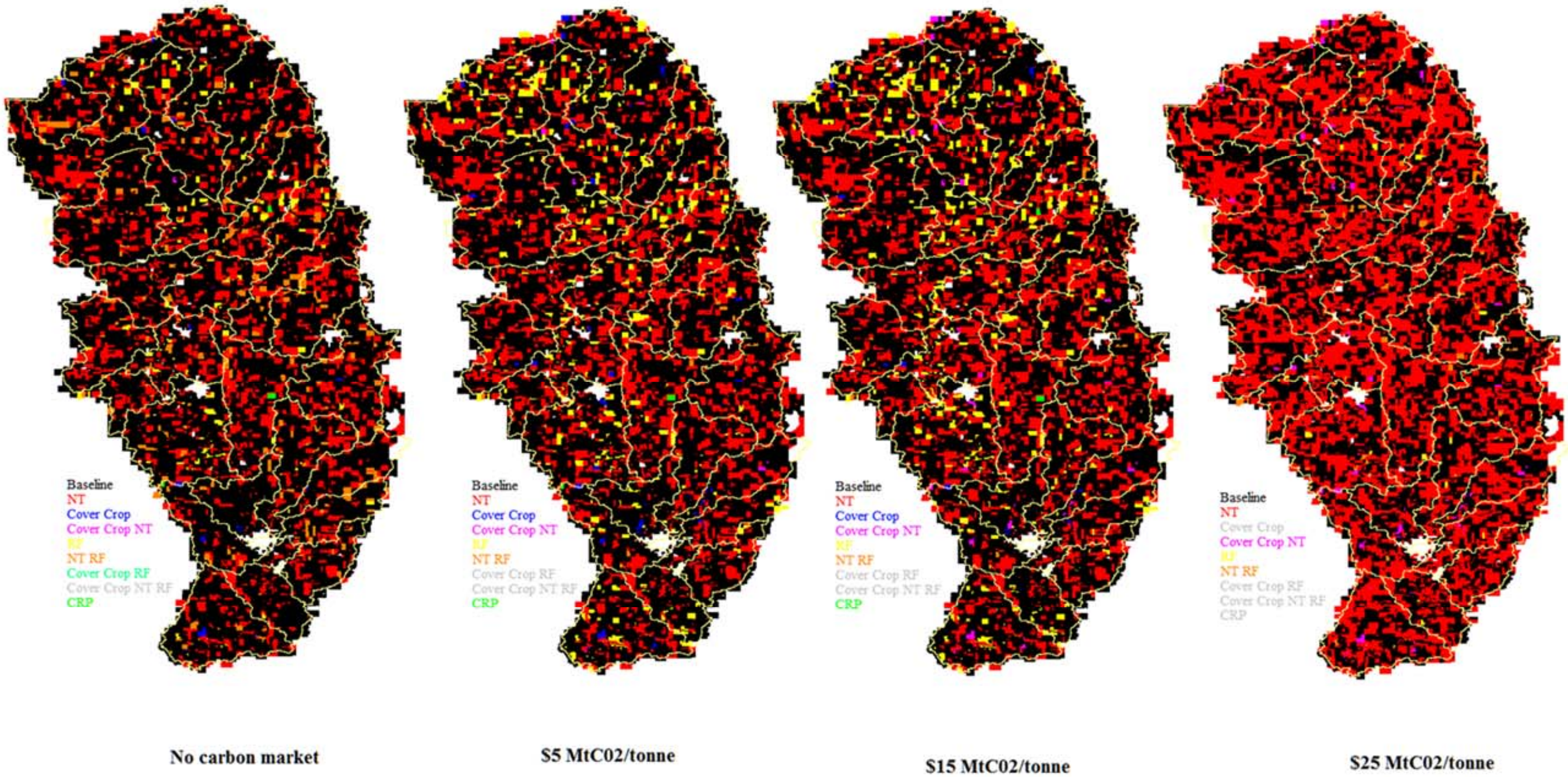
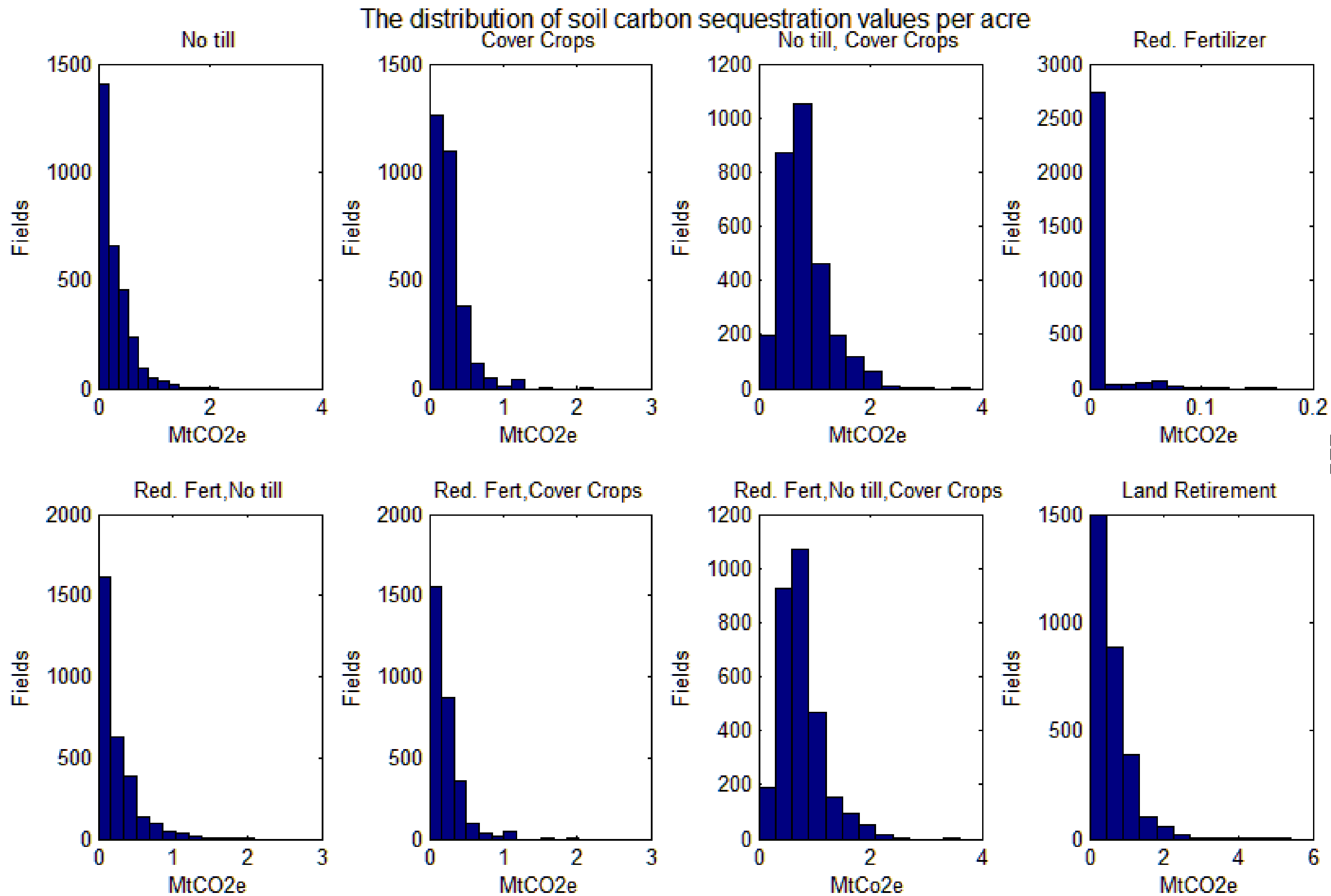


Figure C-3. The spatial distribution of abatement actions, 30% nitrogen.



221

Figure C-4. The distribution of soil carbon sequestration, MtCO2e/acre.

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