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

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Poster prepared for presentation at the Agricultural & Applied Economics Association's 2013 AAEA &CAES Joint Annual Meeting, Washington, DC, August 4-6, 2013

Introduction

The year 2012 marked four decades of existence for the US main regulatory framework for water pollution control the Federal Water Pollution Act (Horan and Shortle, 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[1]. The legislation places stringent regulations on the industrial and municipal polluters, *i.e.* Point sources, but does not specify any regulations for the agricultural polluters, *i.e.* nonpoint sources. In spite of the numerous efforts in reducing the water pollution, water quality remains a significant problem as it is 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 indicates that 53 percent of the 28 percent assessed rivers and streams that 67 percent of the 43 percent assessed lakes and 82 percent of the assessed wetlands have the water quality impaired for designated uses. The assessments designates agriculture as being the leading source of river and streams impairments, the third largest source for lakes and ponds impairments, and the fifth contributor for the wetlands impairments.

The National Rivers and Streams Assessment 2008-2009 (NRSA), the first statistically based survey on water quality of the total rivers and streams, reports that 55 percent of the nation's river and stream miles do not support the aquatic life because of the high content of phosphorus and nitrogen, with 23 percent being in fair conditions, and 21 percent being in good

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

condition. Overall, the study finds that the nation's river and streams are "under significant stress. Reducing 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 suggests that in spite the fact that many actions have been taken towards improving the 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 point out the significance of water pollution commonly produced by agriculture 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 (Freeman 1993, Ribaudo, 2009). Further, emphasizing the contributing role of agriculture to water pollution, Ribaudo et al.(2008) note that the complete elimination of nitrogen point sources across the United States would reduce the total nitrogen emissions by only 10 percent. This facts are not surprising given that 71 percent 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 are above the accepted levels for aquatic activities (Ribaudo, 2009)[2].

In this paper, we present a conceptual model to manage the ambient water quality in a watershed impaired by agricultural runoff (nitrogen and phosphorus). Next, by focusing on the abatement actions available at field scale and by incorporating several challenging issues related to the agricultural pollution and assuming that regulations can be imposed on the nonpoint sources we propose different approaches for reducing both nitrogen and phosphorus pollution.

[2] Nitrogen, phosphorus , suspended sediment and faecal coliform bacteria are considered the most common water pollutants.

The proposed approaches require different levels of cost information on the regulator's side and different degrees of flexibility on the individual polluters' side.

We 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 certain 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 the credits associated with abatement actions with other participants in the watershed (Kling 2011). Additionally, we present a method of estimating the credits that links the abatement actions at field scale to their ability to reduce field level emissions as well the overall level of ambient pollution. We compare the efficiency of a point based trading approach assuming first that the only one of the two pollutant markets is in place, and then by having both pollutant markets functioning simultaneously

Our model captures several critical for the agricultural pollution. In addition to the stochastic nature of the nonpoint sources mentioned in the literature review, there are several important issues that have been proved challenging for the programs designed for reducing the agricultural nonpoint source emissions. Some of these issues are related to the idiosyncratic characteristics of nonpoint source pollutions such as: (i) imperfect information on the abatement costs of individual farms, (ii) difficulties in measurement and monitoring abatement activities at the field level, and (iii) imperfect knowledge of the inherent nonlinearities in the transport and fate of emissions from the edge-of-field to the watershed outlet (the water quality production function). Other challenges have a political component, being related to the ownership of the pollution property rights.

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) that are likely to vary by farm characteristics, climate and other farm related characteristics such as the farmer's knowledge and experience. 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 cost, it is difficult to identify ex-ante the least cost solutions that allocates efficiently the reductions across resources. However, incentive based instruments, such as market based instruments, can improve the efficiency of the allocation by transferring the burden of cost minimization from the regulator to farmers.

Next, observing and monitoring the pollution impacts of farming activities on water quality are difficult to conduct and impose significant costs. Focusing on the observable abatement actions or targeting observable inputs represents possible solutions to this problem as suggested by Griffin and Bromley(1982), Shortle and Dunn (1986). A cost efficient outcome is expected if the target inputs are correlated with the field emissions (Shortle and Horan, 2013).

A third challenging issue for the nonpoint source pollution is the emissions movement (the ultimate fate and transport process) from field up to the point they reach the water bodies where the ambient pollution is observed. Earlier theoretical papers assume that the fate and the transport process is linear and separable between emissions originating from different fields. However, water quality scientist and hydrologists note that the impact on the water quality from different fields is non-constant and depends on the field's location, hence the process is more likely to be non-linear and nonseparable. Additionally, the emissions from one field interact with the emissions from the surrounding fields (Horan and Shortle,2013), making even more difficult to observe the individual farm impacts(Braden, 1989, Khanna et al. 2003). In practice, researchers rely on the use of the various biophysical simulations models to capture the key features of the water pollution process, process we refer as the water quality production function.

The current regulatory framework is another knotty issue in addressing agricultural pollution, as the property rights to pollute are assigned to the nonpoint sources. In spite of the lax (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 the agricultural polluters. It is worthwhile to mention the case of the Everglades Agricultural Area (EAA) 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 EAA by implementing a best management permitting program. Over the 17-year history of the program, more than 55 percent of measurable reductions in the ambient pollution have been met.

Defining the pollution problem is another issue generally, raised in the environmental economics. There are two alternative approaches to define the water pollution problem: the economic efficiency and the cost efficiency. The first approach recommends that the allocation of resource is performed according to the principles of welfare economics where the marginal abatement costs are equal to the marginal benefits of reducing pollution, or equivalently the marginal social cost of pollution is equal to the marginal benefits of pollution. This approach is not largely used because it requires information on the social cost of pollution. The alternative approach is the cost effectiveness, where the social cost is minimized with respect to achieving an ambient (physically) goal for water quality (Horan and Shortle, 2013)[3]. We model the water pollution as a cost effectiveness problem.

This paper is organized as it follows. We begin by introducing the conceptual model of pollution as it relates to agricultural pollution. Next, we outline different policies approaches proposed for addressing the water quality under various model's assumptions and the method created for obtaining a system of points that captures the abatement actions ' efficiency in reducing the agricultural pollution. In the second part, we provide a description of the watersheds and of various data used in the empirical evaluation of my model.

Conceptual Model

Consider a simple model of pollution where the water quality in a watershed is impaired by runoff from agricultural fields (for example, nitrogen and phosphorus). There are *N* farms in the watershed. 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, we consider a set of conservation practices or abatement actions, x that can be implemented at the farm level to reduce the edge-of-field run-off emissions [4]. The ambient level of water quality is measured in stream, at the watershed's outlet. Let r_i^e 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 $r_i^e = 0$, e denotes the fact that the emissions can be nitrogen or phosphorus.

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

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

$$r_i^e = r_i^e(x_i, \gamma_i, \xi) \ \forall \ i = 1, \ldots, N, \tag{1}$$

Where x_i represents the $J \times 1$ vector of abatement actions implemented by farm i, γ_i

Represents the farm's physical characteristics such that soil type and topography, and ξ represents the random environmental factors that are influenced by weather or by the pollutant fate and transport through the watershed[5]. The abatement actions are the only farmers' input choices that can be used at farm level to reduce the field level runoff (Horan, Shortle and Abler, 2004).The baseline edge-of-field emissions are the result of the maximization behavior given that no abatement actions is implemented. Farmers are assumed to be rational and perfect informed risk neutral optimizers and price takers in both output and input markets. Let

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

be the abatement costs[6].

Abatement costs are defined as the difference between baseline profits and the profits associated with the adoption of an abatement action. We assume that the costs of adoption vary across locations due to both difference in physical characteristics (soils, slope, etc., γ_i.) and management abilities or farming experience, θ_i. Thus, the abatement costs are farm and

[5] We recognize the role and impact of the weather stochastic elements. Throughout my dissertation we abstract away from the stochastic elements by considering the mean of the edge-of-field abatement values.

[6] Horan, Shortle and Abler (2004) use an input based abatement control cost function.

abatement action specific. The costs are defined per acre basis, hence assuming constant economies of size[7].

The total ambient pollution is given by an expected water quality production function $W^e\ (r)$, represented as a function of the vector of each farm's individual edge-of-field emission reductions , r^e. Many other factors, 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.

The water quality production function is unlikely to be known given the complexity of the biochemical and hydrological process that take place in a watershed and is assumed to be a nonlinear and non differentiable function of individual edge of-field reductions. In practice, though the true form of this function is not likely to be exactly known, there is a range of watershed-based water quality models that approximate these hydrological and biophysical processes, such as the Soil and Water Assessment Tool (SWAT) to name one of them,

Let, $W^e(r) = W^{e,0} - A^e\ (r)$ be the ambient water quality at the watershed outlet, where $W^{e,0}$ is the level of water quality given the current activity, and $A^e(r)$ is the expected ambient pollution reduction associated with r^e, the vector of field emission reductions, or more simply the abatement function[8]. The expected ambient water quality level can be expressed as the difference between the no-control (baseline) expected ambient water quality level and the in-stream expected abatement associated with the edge-of-field emission reductions given that an array of abatement actions is taken.

[7] Economies of size are used to describe the 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] The literature uses the terminology of water quality production function and abatement function interchangeable. From here on we will refer to ambient function as representing the change in the ambient water quality at the watershed outlet.

We consider an environmental authority who seeks to achieve a particular expected abatement pollution levels for both nitrogen (N) and phosphorus (P) denoted as $\bar{A}^{e}$, and sets up the cost minimizing problem of achieving the expected water quality goal by finding the least cost allocation of the available abatement actions to the fields in the watershed. First, we identify the first best solution to the above defined problem assuming that the regulator and farmers have the same cost information, situation identified as the perfect cost case. This solution is contrasted to the solution where the regulator does not know the true abatement costs but can use the first moment (mean) of the costs distributions to solve the least cost solution. This case is identified as asymmetric cost information.

First best

We begin by assuming that the regulator knows : (i) the field level abatement costs, (ii) the relation between abatement actions and reduced emissions $r_i^e(x_i)$, (iii) the true form of the ambient abatement action $A^e(r)$. 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:

$$\min_{x_i} \sum_i C_i(x_i, \gamma_i, \theta_i) \;\; s.t. A^N(r) \geq \; \bar{A}^N \; and \quad A^P(r) \geq \; \bar{A}^P \qquad (3)$$

Where θ_i shows that field level abatement costs are used in solving the cost minimization problem. The solution identifies the least-cost conservation practice assignment for each field x_i^*, and thus implicitly an optimal amount of edge of field pollution $r_i^{e,*}(x_i^*)$, $\forall\, i = 1, \ldots, N$ farms and $\forall\, k = 1, \ldots, J$ available conservation practices. The total cost is given by $TC^* = \sum_i^F C_i(x_i^*, \gamma_i, \theta_i)$. An “*” is used to indicate that this is the least-cost solution.

The first solution is achieved when the regulator has the ability to solve the above problem in the presence of complete cost information and the ability to implement a command and control policy where he can mandate the abatement action x_i^*. this is not the case because it is unlikely for the regulator to have complete cost information. The cost asymmetry can be overcome if incentive based policies such that performance standard and trading that shift the burden of optimization from the regulator to the private farmers are pursed. The implementation of any such of incentive based policies requires a functional form for both the abatement function $A^e(r)$ and for the relation between field level abatement actions (x_i) and the expected edge-of-field abated emissions $(r_i^e(x_i))$. Next, we are considering how different policies performed relative to the first-best by considering a linear approximation of the nonlinear and nonseparable abatement function.

In the next section, we describe an approach to linearize the abatement a function that allows for the estimation of the delivery coefficients. In addition, to using a linear approximation of the water quality production function; the true $r_i^e(x_i)$ functions are imperfectly measured the impact of the abatement actions on reducing the emissions.

Designing a method for approximating to the abatement based on a set of abatement actions

A trading program 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 important barriers to an exact quantification of the emissions (Malik et al. 1994). Estimating a system of points that capture 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) proposes a point based trading system where agricultural producers would be required to implement abatement actions that accrue enough points per acre to meet the standard. The point values assigned to each abatement practice approximate: (i) how effective an abatement practice is in reducing the edge-of-field emissions and (ii) the impact of the edge-of-field reduced emissions on the ambient water quality. Since the abatement function ($A\ (r)$) 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 conservation practices x:

$$A^e(r) = A^e(x) \cong \sum_{i}^{N} d_i^{e,\prime} r_i^e(x)^9 \qquad e = N \text{ or } P \tag{4}$$

Next, assuming that there are nonlinearities at field level, the edge-of-field reductions are approximated as $r_i^e(x) \cong \sum_j^J {w^e}_{ij} x_{ij}$, where w_{ij}^e measure the impact of abatement action j given field i, at location i The impact of the edge-of-field abatement of field i on ambient water quality is $d_i^{e,\prime} r_i^e(x) \cong \sum_j^J d_i^{e,\prime} {w^e}_{ij} x_{ij} = \sum_j a_{ij}^e x_{ij}$, where $a_{ij}^e = d_i^{e,\prime} {w^e}_{ij}$, referred hereon as "point coefficients", 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

[9] Without loss of the information, WE will drop γ_i from notation

contribution to the total abatement of a particular field given i that the j abatement action is taken. Finally, the linear approximation of the abatement function can be re-written as:

$$A^e(x) \cong \sum_i^N d_i^{e,\prime} r_i^e(x) = \sum_i^N \sum_j^J a_{ij}^e x_{ij} = a^e X \tag{5}$$

i.e.; a linear combination of the abatement actions x_{ij} and field specific point values, a_{ij}^e. Next, we describe an approach for estimating the vector $\boldsymbol{a^e}$ **.**

An approach for estimating the point values

We employ a multi-step procedure to estimate the point coefficients for each field and each abatement action using the special features of a watershed based hydrological model, SWAT. As previously specified, a watershed is delineated in sub-basins and further on in smaller fields units called HRU. As a result, a watershed can contain thousands of fields. One way to estimate the point coefficients at such refined scale would be to generate M sets of allocations of abatement actions in the fields in the watershed, where each set of random allocation represents a unique watershed configuration. In this case M should be greater the NxJ, where N is the number of and J the number of abatement actions available at watershed level. The impacts on the ambient level of water quality, in terms of mean annual abatement loadings nitrogen and phosphorus, are obtained by simulating the random watershed configurations with SWAT model. 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**,** $\boldsymbol{a}$, by combining the results of a series ordinary least square estimations $\min_a (A^e - Xa^e)'(A - Xa^e)$.

It is infeasible to generate a sufficient number of watershed configurations to estimate NxJ point coefficients. My approach of estimating point values takes advantage of the outputs displayed by SWAT and helps breaking the above estimation into two steps. *(i)* estimate the point values at sub-basin level using the ambient level measure at the watershed exit, *(ii)* estimate point values at field level using the field provided outputs, and *(iii)* combine the results to obtain field specific point coefficients for each abatement action. Combining the two sets of results allows me not only estimating the field specific point coefficients for each abatement action but also estimating the delivery coefficients.

Once we determine of point values that are credited to a particular abatement action in a specific field, we are 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 this approach, a farmer is free to choose the conservation practices that solve the cost-minimization problem:

$$\min_{x_{ij},b_i\in X} C_i^P(x_i\,,\gamma_i,\theta_i\,) \quad \text{s.}t. \;\; \sum_{j=1}^{J} a_{ij}^e x_{ij} \geq \; b_i^{e,o} \qquad \forall\, e = N,P\,, \tag{6}$$

Where the performance requirement is specified by b_i^o.

Under the trading approach, credits generated by abatement actions are tradable, on a one-to-one basis, across the watershed. As a result, a farmer solves:

$$\min_{x_{ij},b_i\in X} C_i^P(x_i\, ,\gamma_i, \theta_i\,) + p^N b_i^N + p^P b_i^P$$
$$\text{s.}t.\ \ \sum_{j=1}^{J} a_{ij}^e x_{ij} + b_i^e \geq\ b_i^{e,o} \quad \forall\ e = N, P, \tag{7}$$

and the point priced p^e are determined by the market clearing conditions by $\sum_i b_i^e = 0$.

This trading approach can be conceptually viewed as a combination of an emissions permit and ambient permit system (Rabotyagov *et al.* 2012). Under 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 at 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.

Empirical framework

In the next sections, we describe the two agricultural watershed used as support for our empirical estimations, the set of abatement actions together with the corresponding estimates for the abatement cost , and a description of the estimates obtained for the point values assigned to each abatement actions. The point values estimates are watershed specific, and within each watershed field and pollutant specific, with two pollutants being considered: nitrogen and phosphorus.

Watershed Description

We am using the available data for two typical Midwestern watersheds, both located in Iowa: Boone River Watershed (BRW) and Raccoon River Watershed (RRW). The National River and Streams Assesment 2008-2009 include Iowa in the Temparate Plains Ecoregion[10]. The survey finds high levels of nitrogen in 58 percent of the rivers, medium levels of nitrogen in 13 percent of the rivers. At the same time, 31 percent (24 percent) of the rivers have high (medium) levels of phosphorus.

The Boone River Watershed

The Boone River Watershed (BRW) is located in the north central part of Iowa. The watershed covers more than 537,000 acres (2,370km2) in six counties (Hamilton, Hancock, Humboldt, Kossuth, Wright, and Webster).

The watershed area is crop intensive, with corn and soybean representing almost 90% of the agricultural activity. The surface area had been intensively tile drained, as a consequence the wetlands area had been reduced significantly. Moreover, the Boone watershed agricultural area has been found responsible for some of the highest nitrogen loadings among Iowa's watersheds (Libra *et. al 2004).*

The required data for modeling system (i.e. SWAT (2009)) was collected at Common Land Unit level (CLU)[11]. More than 16,300 CLUs have been identified in the BWR. As HRU is the unit required by SWAT model, the CLUs were regrouped in roughly 2,968 HRU. Data

[10] Other states included in the same ecoregion are: the Eastern Dakotas, western Minnesota, portions of Missouri, Kansas and Nebraska, western Ohio, central Indiana, Illinois, and southeastern Wisconsin.

[11] "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)

related to crop rotations, land uses, fertilizer management, tillage and conservation practices were provided by a field level survey conducted by Kiepe (2005).

The Raccoon River Watershed

The Raccoon River Watershed (RRW) is one of the largest watersheds in the state of Iowa. It covers an area over 9,400km2 in the west central Iowa, with the Des Moines River being its major tributary. It flows approximately 300 km from its origin in Buena Vista County to the confluence with the Des Moines River in the Des Moines City.

The landscape of the south part of the watershed is characterized by higher steep relief, with many hills and a well-developed drainage system, while the landscape in the northern part is characterized by low relief and poor surface drainage system (Schilling et al., 2008). With more than 73 % of the planted area being use for corn and soybean, the land use is dominated by agricultural row production. Other land uses include grassland (16.3%), woodland (4.4%), and urban (4.0%) (Gassman and Jha, 20xx card report).

Table 1 summarizes the baseline N and 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 compute using the historical available data for 1995-2001, with the first two years being dropped out.

Table 1   Watershed description

Watershed	Baseline N (kg)	Baseline P (kg)	Subbasins	Fields[12]	Area (km2)
Boone (BRW)	4,725,826	218,828	30	2,968	2,370
Raccoon (RRW)	18,604,642	632,406	112	1,569	9,400

[12] SWAT HRUs

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. The above set is augmented with all feasible combinations of these practices but land retirement (i.e. the combination of no till and cover crops is considered as an independent conservation practice). The baseline is also considered as choice alternative. Table 2 provides a description of the abatement actions used in the empirical applications for Boone River Watershed and Raccoon River Watershed

Table 2 Abatement actions

	Abatement action	Abatement action description
1	Baseline	No action required
2	No till (NT)	No till, no more than 30 % of crop residue is removed
3	Reduced Fertilizer (RF)	Reducing fertilizer application rate by 20 %.
4	Cover Crops (CCr)	Establishment of cover crops between crop rotations.
5	Land retirement (CRP)	Retirement of land from production
6	NT RF	No till and 20 % reduction in nitrogen application rate
7	NT RF	No till, no more than 30 % of crop residue is removed
8	RF CCr	Reduced fertilizer and establishment of cover crops.
9	NT RF CCr	No till,20 % reduction in nitrogen application rate and cover crops

The costs of abatement actions

Costs for each conservation practice were drawn from several sources. All costs are expressed as dollars per acre. Table 3 summarizes the mean and the standard deviations for assumed abatement actions implementation costs for the two watersheds. The 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.

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 the similar to the one used by Rabogtyagov (2007), Sawyer *et al*. (2006) 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[13]. The cost of reducing fertilization is given by multiplying a 20 % percent reduction in the baseline fertilizer rate by the price of corn. The price of corn used is $3.08 per bushel.[14] 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 of fertilizer.

Cash rental rates available online (Edward and Smith, 2009) in conjunction with the corn suitability ratings (CSR) 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 CSR by area and corresponding CSR. 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 conservation practices obtained as a combination of the primary ones (i.e. no till and reduced fertilizer) are obtained by summing per

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[13] The coefficients of nitrogen response yield curve Y=-3.32904824784026E-09*N^4+8.88402E-06*N^3-0.004459448*N^2+0.822128904200617*N-0.374570292118776

[14] Price per bushel and 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

acre cost of each conservation practice considered in the combination. Table 2 summarizes the costs used in carrying the simulation in the two river basins.

Table 3 Abatement actions: assumed costs

	Boone River Watershed		Raccoon River Watershed		
	Cost ($/acre)		Cost ($/acre)		
Conservation practice	Mean	Std.dev	Mean	Std.dev	Cost source
No action	0	0	0	0	
No Till(NT)	5.1	1.91	10.42	7.59	Kling et al. (2007)
Cover Crop	24.09	4.71	19.28	10.5	T. Kaspar
Reduced fertilizer (RF)	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. (2007)

Obtaining the Point Value Estimates

Two sets of point values are obtained for each watershed. The first set of points estimates the effectiveness of the abatement actions in reducing the nitrogen emissions. The second set of points is estimated with respect to phosphorus emissions. Table 4 presents the estimates for point values as an area weighted average of the point estimates across the watershed.

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

	No action	No Till (NT)	Cover Crops (CC)	NT CC	Red. fertilizer	Red.Fert, NT	Red.Fert CC	Red.Fert. NT,CC	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

In general, the results follow the prior expectations. The abatement practices that are known to be highly effective at reducing one pollutant emissions are awarded a higher point value than less effective practices (*i.e*., CRP receives the highest number of points for both pollutants). Also, the points associated with adopting a combination of conservation practices are not equal to the summation of the individual ones: *i.e*.; the abatement action that combines no till and cover crops, receives a lower number of points (4.264) than the sum of the points assigned to each of them (2.347+2.420=4.767). Reduced fertilizer has the lowest number of points as it is the less efficient abatement practice for reducing the nitrogen loss and has virtually no impact on reducing the phosphorus losses.

The difference in the magnitude of the estimates for the two pollutants is explained by the difference in total pollutant levels; the quantity of nitrogen measured at the main outlet is much higher than the quantity of phosphorus measured at the same outlet. Interestingly, the estimates are comparable across different watersheds; *i.e.*, the point values for the same abatement determining the watershed configuration that achieves a water quality goal, $\bar{A}^{e}$

The regulator sets up a water quality goal, expressed as percentage reductions in baseline's level of total nitrogen or total phosphorus. Given no cost information is available; it 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 $\bar{A}^{e}$

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

$$B^e = A^e(X) = \sum_{i}^{N}\sum_{j}^{J} a_{ij}^{e} x_{ij}\ s_i \qquad (9)$$

Where x_{ij}^{*} is a dummy variable that takes value 1 if practice j is assigned to field i and 0 otherwise, a_{ij}^{e} denotes the number of points corresponding to field i for pollutant e, given the abatement action j and the 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 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 which specifies the credits 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 $B^e = A^e(X) = \sum_i \sum_j a_{ij}^{e} x_{ij}\, s_i = \sum_i b_i^{e,0}$. The total number of points can be assigned as initial farm level requirements in several ways: allocate the points according to the initial watershed configuration, or the total number of points could be equally divided 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.

Results

Nitrogen and phosphorus are considered to be the most important agricultural pollutants. First, we consider the case where the proposed policies consider only one pollutant at a time (i.e.. either nitrogen or phosphorus) and later we consider the case where both pollutants are considered jointly. We present the results for three levels of desired water quality improvement: 20 percent, 30 percent and 40 percent desired reductions in mean annual loadings (nitrogen or/ and phosphorus) relative to the baseline.

First best scenario

To be able to evaluate the performance of the three regulatory approaches, we solve for the first best solution: the least cost placement of the abatement actions across to the watershed to achieve any given level of ambient water quality level.

Choosing the on-farm or watershed goals under the proposed policy approaches can be challenging under a nonlinear water quality production function. Under a CAC program, the regulator can mandate the farm level abatement actions. If she is interested in achieving the abatement target $\bar{A}^{e}$, then he needs to find the set of abatement actions (X_{CAC^e}) that satisfies $\{A(X_{CAC^e}) = \bar{A}^e\}$

Setting the point requirements for the three policies approaches

Choosing the on-farm or watershed goals under the proposed policy approaches can be challenging under a nonlinear water quality production function. Under a CAC program, the regulator can mandate the farm level abatement actions. If she is interested in achieving the abatement target $\bar{A}^e$, then he needs to find the set of abatement actions (X_{CAC^e}) that satisfies $\{A^e(X_{CAC^e})\ ) = \bar{A}^e\}$.

One option that does not require any cost information and involves the evaluation of a range of different watershed configurations until the regulator finds one that meets $A^e(X_{CAC^e}\ ) = \bar{A}^e$.

Under a CAC program, $\{X_{CAC^e}\}$ can be implemented directly. However, the on farm performance standard program or the credit trading program, setting field level requirements requires mapping the abatement actions to the on farm point coefficients or total watershed points requirements. For the PS, this implies using field level estimates for the point coefficients to compute the farm level requirement $b_i^{e,0,PS} = \sum_j^J a_{ij}^e x_{ij}$, where x_{ij} is the abatement actions assigned to field i. Next, the on farm requirements can be summed up to determine the total watershed points required for setting up a trading program, $P^e = \sum_i^N b_i^{e,0,PS} = \sum_i^N \sum_j^J a_{ij}^e x_{ij}$. The total number of points, P^e is translated into farm individual point requirements as $P^e = \sum_i^N b_i^{e,0,trading}$. The initial (pre-trading) point allocations $b_i^{e,0,trading}$ may or may not correspond to the point requirements under a performance standard program($b_i^{e,0,PS}$) ,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 points targets (satisficing and optimizing) results in six different policies to simulate for each pollutant and each watershed. The results are obtained for three levels of desired water quality improvements: 20 %, 30 % and 40 % reductions in the mean annual loadings of nitrogen and phosphorus. Next, the results for each water quality target and pollutant are carried using two sets of simulations.

In the first set of simulations, we assume 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 the abatement costs have the following form: $C_i(x_i, \gamma_i, \bar{\theta})$. The outcomes are compared relative to the Pareto frontier optimal solutions.

In the second set of simulation, maintaining the same costs assumptions, we compare the outcomes of a point based trading approach that attempts to regulate both nitrogen and phosphorus with the outcomes of a point-based trading approach that regulates either nitrogen or phosphorus.

Cost-efficiency performance under the same cost information

Boone River Watershed Simulated Policy Performance

Table 5 Boone Watershed Multiple Pollutant Policy Approaches

Boone Watershed Multiple Pollutant Policy Approaches								
Abatement Target/Command and Control			Performance Standard			Point-Based-Trading		
N	P	Total Cost	N	P	Total Cost	N	P	Total Cost
20%	20%	6,652,189	26.3	27.9	5,070,587	22	29.6	1,071,091
30%	30%	17,992,828	34.5	35.3	15,850,030	32.2	37.6	3,044,345
40%	40%	36,075,292	42.9	43.6	35,464,586	41.2	37.8	7,043,165

Table 6 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*	1,071,091	*12.6*	**19.3**	378,465
30%	**32.2**	*37.6*	3,044,345	*19.2*	**27.9**	853,634
40%	**41.2**	*37.8*	7,043,165	*27.4*	**36.7**	1,905,560

Table 5 summarize the simulated outcomes under the three policies approaches when both N and P are targeted. Under the command and control 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 point-based trading, the costs are much lower, being on average about 20 percent of the costs under a command and control program. Both N and P abatement target are over met for 20 and 30 percent target. Interestingly, for 40 percent reductions in both N and P, under a point based trading, the N target is slightly over met, while the P target is not attained.

Table 6 summarize the simulated outcomes for the point-based trading scenarios where only one pollutant is target. 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 percent and the total costs are much lower than the case of a nitrogen only point-based trading. However, the total costs are much lower.

Raccoon River Watershed Simulated Policy Performance

Table 7 Raccoon Watershed Multiple Pollutant Policies Approach

Raccoon Watershed Multiple Pollutant Policies Approach								
Abatement Target/Command and Control			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 8 Raccoon Watershed Single Pollutant Point-Based Trading

Raccoon 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 7 and 8 summarize qualitatively similar results for Raccoon River Watersheds: the trading setting outcomes have the lowest costs and the outcomes of a nitrogen point based trading are the same with the outcomes of a nitrogen and phosphorus point based trading approach.

Figure 1 and Figure 2 depicts the spatial distribution of the abatement actions under a goal of 30 percent reductions in both nitrogen and phosphorus. Overall, a more diversified distribution can be observed under the trading approach

Conclusions

Montgomery (1972) demonstrates 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. 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, efficient approach of water quality requires the consideration of multiple pollutants such as nitrogen and phosphorus.

We propose a design for marketable point systems to regulate two pollutants (phosphorus and nitrogen).We describe the approach for creating a point-based trading system, including a procedure to efficiently choose points to approximate the effectiveness of abatement actions and the watershed-based water quality production function. Then, using a detailed biophysical watershed-based water quality model together with a range of estimates for the abatement costs, we demonstrate the efficiency tradeoffs of using a point-based system.

We use a two-step procedure to estimate field-level point coefficients for each available abatement action. The procedure for estimating the point coefficients uses information from a watershed-based quality model (SWAT).

We study the performance of the trading system using a watershed-based water quality model calibrated for two typical Midwestern watersheds. Our findings show a promising performance of the points-based trading system in terms of both reaching the water quality objectives and cost-efficiency when farmers participate in both pollutant markets. However, the same outcomes can be obtained under a trading approach that targets only nitrogen. Our results also so that phosphorus based trading program has the potential to achieve its environmental goals at much lower cost.

The results of the policies described above are relying on a framework where standards are imposed on the agricultural polluters rather than a framework relying on voluntary actions. We do not suggest that such a framework is foreseeable at the federal level, but at the same time we want to point out that a precedent exists at state level (an initiative implemented to address phosphorus emissions in an agricultural district in Florida). Our approach provides a potentially attractive guide to policy implementation by demonstrating how flexible incentive-based programs can improve cost-effectiveness of multiple nonpoint-source pollutant control efforts

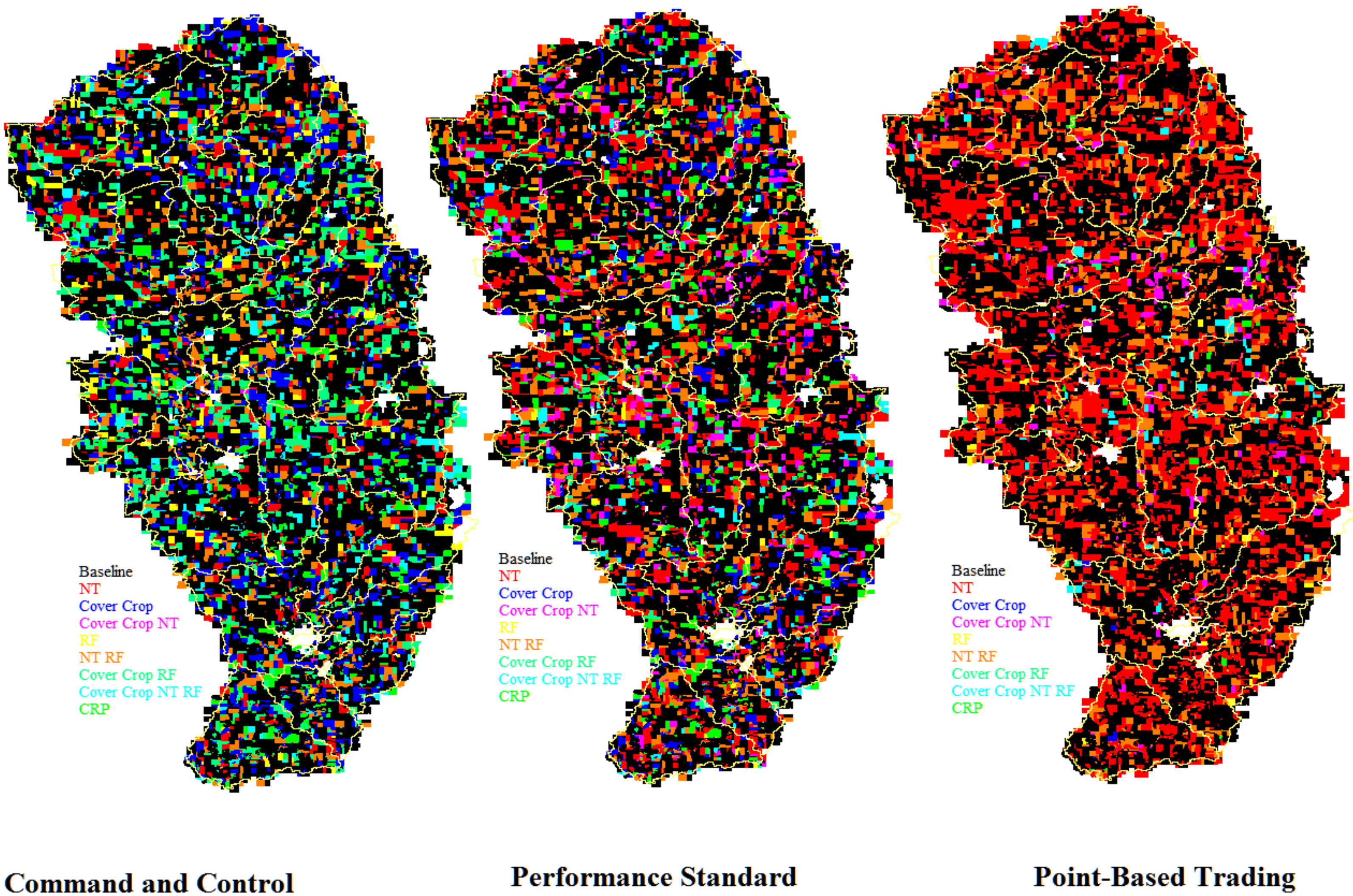
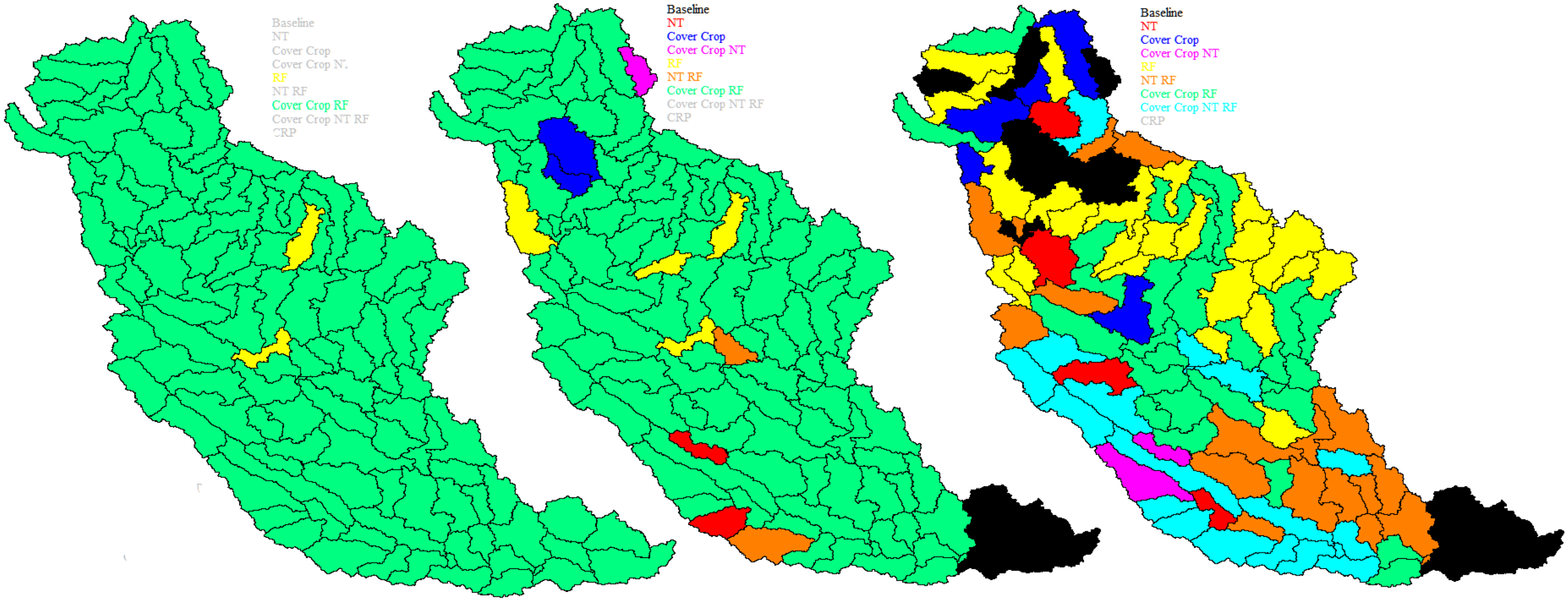


Figure 1 Boone River Watershed: The spatial distribution of abatement practices



Command and Control **Performance Standard** **Point-Based Trading**

Figure 2 Raccoon River Watershed: The spatial distribution of abatement practices

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