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The Designation of Co-benefits and Its Implication for Policy: Water Quality versus Carbon Sequestration in Agricultural Soils

Silvia Secchi, Manoj Jha, Lyubov A. Kurkalova, Hongli Feng,
Philip W. Gassman, and Catherine L. Kling

Working Paper 05-WP 389
March 2005

**Center for Agricultural and Rural Development
Iowa State University
Ames, Iowa 50011-1070
www.card.iastate.edu**

Silvia Secchi, Lyubov Kurkalova, and Hongli Feng are associate scientists, Philip Gassman is an assistant scientist, and Manoj Jha is a postdoctoral research assistant, all in the Resource and Environmental Policy (REP) Division at the Center for Agricultural and Rural Development (CARD) at Iowa State University. Catherine Kling is a professor of economics and head of the REP Division at CARD.

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For questions or comments about the contents of this paper, please contact Silvia Secchi, 560C Heady Hall, Iowa State University, Ames, IA 50011-1070; Ph: 515-294-6173; Fax: 515-294-6336; E-mail: ssecchi@iastate.edu.

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Abstract

This study investigates the implications of treating different environmental benefits as the primary target of policy design. We focus on two scenarios, estimating for both of them in-stream sediment, nutrient loadings, and carbon sequestration. In the first, we assess the impact of a program designed to improve water quality in Iowa on carbon sequestration, and in the second, we calculate the water quality impact of a program aimed at maximizing carbon sequestration. In both cases, the policy instrument is the retirement of land from agricultural production.

Our results, limited to the state of Iowa, and to the case of set-aside for water quality or carbon sequestration purposes, indicate that the amount of co-benefits depends on what indicators are used to measure water quality. In general, this study shows that improving “water quality” in the sense of reducing nutrient or sediment loadings is too vague. Even if it is taken to refer to in-stream nutrients, because the responses of nitrogen and phosphorus to conservation efforts are not well correlated, this terminology may not provide much guidance.

Keywords: carbon sequestration, co-benefits, environmental benefits targeting, Iowa, land set-aside, water quality.

THE DESIGNATION OF CO-BENEFITS AND ITS IMPLICATION FOR POLICY: WATER QUALITY VERSUS CARBON SEQUESTRATION IN AGRICULTURAL SOILS

The design of policies to induce the adoption or maintenance of conservation practices is often complicated by the fact that many conservation practices produce multiple benefits. For example, conservation tillage has the potential to sequester carbon as well as to reduce soil erosion. Likewise, land set aside and planted with grass or trees can improve wildlife habitat and stop or reduce nutrient runoff in addition to storing carbon and reducing erosion. If the social value of each benefit were known, then the total value from the multiple benefits could be summarized in a comprehensive index, and policy design would be straightforward: sound policy would aim to maximize the value of this index for any given budget.

However, it is difficult to assess the value of benefits from conservation practices. Most of these goods are non-market and so monetary values are not always available, even if we know the environmental improvements in physical quantities. Water quality provides an apt example. There are many studies that estimate the value of improved water quality through contingent valuation methods or travel cost models; nonetheless, it is a challenging task to connect the “water quality” used in these studies (which is often represented by criteria such as whether a lake or river is swimmable, fishable, or boatable, etc.) to the “water quality” represented by the reduction of nutrient or sediment loading, the criterion typically used in actual measurement or water quality simulation models. Moreover, even the size of environmental improvements in physical quantities is a major task to assess, whether through costly direct measurement or complex and uncertain model simulations.

There is a large body of literature that focuses on a single benefit from conservation practices. For example, there are many studies that investigate the cost of carbon sequestration without considering other benefits. Likewise, the least-cost way of improving water quality or nutrient loading in waterways has also been examined extensively (e.g.,

Ribaudo et al. 2001), again largely without consideration for the other benefits of the practices that would implement the policy such as carbon sequestration. More recently there has been some attention paid to the multiple benefits of conservation practices. Often in this literature one benefit is treated as the primary benefit and others as co-benefits (Kurkalova, Kling, and Zhao 2004), although the benefits have also been considered as a bundle.

The views in the literature are mirrored in the historical development of at least some conservation programs. For example, in the early days of the Conservation Reserve Program (CRP), the primary land set-aside program in the United States, erosion reduction was the main focus. However, the current CRP takes into account a number of benefits through an index that gives weights to a variety of environmental factors. As reflected in the index, water quality receives more weight than carbon sequestration, but carbon is also included.

In this paper, we investigate the implications of treating different environmental benefits as the primary target of policy design for the entire bundle of environmental benefits that society values. That is, when we consider other environmental goods as co-benefits resulting from policies that sequester carbon rather than the focus of the programs, it implies that programs will target carbon sequestration more heavily. In contrast, if water quality is the focus of the program, it implies that the program will not heavily target carbon. We undertake an empirical study to demonstrate the importance of this issue in the context of a vital policy debate occurring in much of the midwestern United States related to the policies and conservation practices that will be necessary to improve water quality in local rivers and streams. In particular, we focus on two issues: (a) the carbon sequestration co-benefits of currently proposed and/or implemented programs designed to improve water quality and (b) the value of various bundles of water quality and carbon benefits associated with the programs and alternatives to the programs that focus more on carbon than on water quality.

Understanding the carbon sequestration co-benefits of current programs and the degree to which changes in program design could increase carbon storage with relatively small trade-offs in other environmental benefits is a vital public policy issue. Further, we perceive something of a mismatch between the focus that the academic literature has placed on

carbon sequestration and its co-benefits and the policy arena in which programs are more often being considered with a focus on water quality (making carbon the co-benefit). A large amount of money is expended on conservation in agriculture, much of which is meant for water quality, and many evaluation studies have shown that people are willing to pay for water quality improvement (for example, Lipton 2004 and Desvousges, Smith, and Fisher 1987). According to Ribaudo (1989), water quality benefits for the Corn Belt were greater than \$80.00 per acre; the number was twice as high in the Lake States. In addition to the potential benefits, the National Needs Survey indicates that a large amount of funding may be needed to meet water quality needs. This implies that the nation might be poised to devote a large amount of resources to improving water quality. Undoubtedly, a co-benefit of these activities will be an increase in carbon sequestration and in this paper we will examine empirically the potential magnitude of carbon that can be achieved through programs mainly intended for water quality.

Specifically, we study a set of policies designed to support the Clean Watersheds Needs Survey for the state of Iowa. This survey is required as part of the Clean Water Act and it requests that states identify the financial resources necessary to meet water quality goals from non-point source reductions (primarily in agriculture). As part of this effort we are performing an analysis of the costs of improving Iowa's in-stream water quality by linking an economic model of farmers' choices to a watershed-based hydrologic model, the Soil and Water Assessment Tool (SWAT).

In the next section, we describe the water conservation "policy" developed to support the needs assessment for the state of Iowa and two additional policies focused on carbon sequestration. These form the basis for our comparisons and assessment of the magnitude of co-benefits. In section 2, we describe the estimation procedures and data we employ to estimate the costs and instream water quality benefits associated with these three policies. In section 3, basic results of the simulations are presented and interpreted. In section 4, we undertake a simple break-even type of analysis to examine the implicit "price" of nutrient and erosion reductions (measured at the edge of the field) that would be necessary to make these policies pass a simple cost-benefit test. Lastly, we give our conclusions and final thoughts.

1. Conservation Policies for Water Quality and Carbon Sequestration

The U.S. Environmental Protection Agency (EPA) is required to perform a periodic national Clean Watersheds Needs Survey in response to directives that were established in the 1972 U.S. Clean Water Act. The purpose of the survey is to identify all existing water quality or public health problems. As interpreted by the 2000 EPA National Needs Assessment (USEPA 2003), the process of determining the needs of states to address nonpoint source water quality problems consists of two steps. The first is the identification of the set of conservation practices and land use changes that should be placed on the landscape, and the second is the estimation of the costs of those practices. As part of this process for the State of Iowa, in consultation with the Iowa Department of Natural Resources (IDNR), we determined the location of the practices based on the potential environmental impact as captured by several indicators, such as proximity to a stream, an erodibility index, and slope. Several practices were included in this exercise. Here, we consider a key one: land set-aside. We choose this practice to demonstrate the importance of the issues we raise because land set-aside is a major change in land use: it is both costly and land well suited for the sequestration of carbon is not necessarily well suited to the reduction of sediment and nutrient loadings to waterways. Thus, there is the possibility of very different efficient configurations of a policy designed to focus on carbon sequestration relative to water quality.

The water quality improvement scenario studied here assumes that all land within 100 feet of a waterway will be placed out of production and planted with perennials. This area amounts to about 251,600 acres. Additional land is retired based on the Erodibility Index (EI), until 10 percent of total cropland is retired from production. This amounts to an additional 2,172,100 acres. The EI index indicates the potential of a soil to erode, based on both climatic factors and the properties of the soil. A higher index indicates higher erodibility potential. The rationale for this choice is that choosing land closer to waterways to retire means that this land can filter sediment and pollutants from upstream farms. The highly erodible land, on the other hand, is highly correlated with elevated levels of sediment loss. The 10 percent cropland cap was chosen in concert with the IDNR, taking into account both the high level of impairment of Iowa's waters and the issues linked with taking large areas of farmland out of production.¹

We also consider carbon-focused policies by ranking each piece of land based on its ability to sequester carbon when retired from production. We then “enroll” land into the program based on the highest carbon benefit. To maintain comparability with the water quality scenario, we consider two cases. In the first, we keep constant the amount of acres enrolled in the set-aside program (i.e., we enroll acreage until a total of 10 percent of the cropland in the state is retired). In the second, we keep constant the cost of the program. That is, we enroll acreage into the program until the expenditure total is the same as the amount spent under the hypothetical water quality scenario.

2. Data and Models

To develop our models, we draw heavily from the National Resource Inventory (NRI) (USDA-NRCS 2003) to provide data on the land use, cropping history, and farming practices in the state of Iowa. The NRI is the most comprehensive data set on land use in the United States, and we use data on the 14,472 physical points in Iowa that represent cropland (Nusser and Goebel 1997). Conceptually, our data and models are based on individual producer and farm-level behavior, and we treat an NRI point as a producer with a farm size equal to the number of acres represented by the point (the expansion factor provided by the survey). Figure 1 illustrates the 35 watersheds corresponding to the eight-digit Hydrologic Cataloging Units of the United States Geological Survey that are largely contained in the state and that are modeled in this study.

The costs of enrolling into the land set-aside program are estimated using the model developed in Feng et al. (2004), in which the opportunity cost of land retirement is determined by using the cropland cash rental rate. The average cost of land retirement in the sample is \$123 per acre with a standard deviation of \$21 per acre. The rental rates are the highest in the central and northern parts of Iowa, averaging as high as almost \$140 per acre in subwatersheds of the Iowa and Wapsipinicon rivers (Figure 1). The least expensive land to retire from crop production is in the southern portion of the Des Moines River as well as in the Nishnabotna and Nodaway river watersheds where the average land retirement cost is as low as \$105 per acre.

To compute the amount of carbon sequestered when an NRI point is retired from cropland, we rely on estimates from the Environmental Policy Integrated Climate (EPIC)

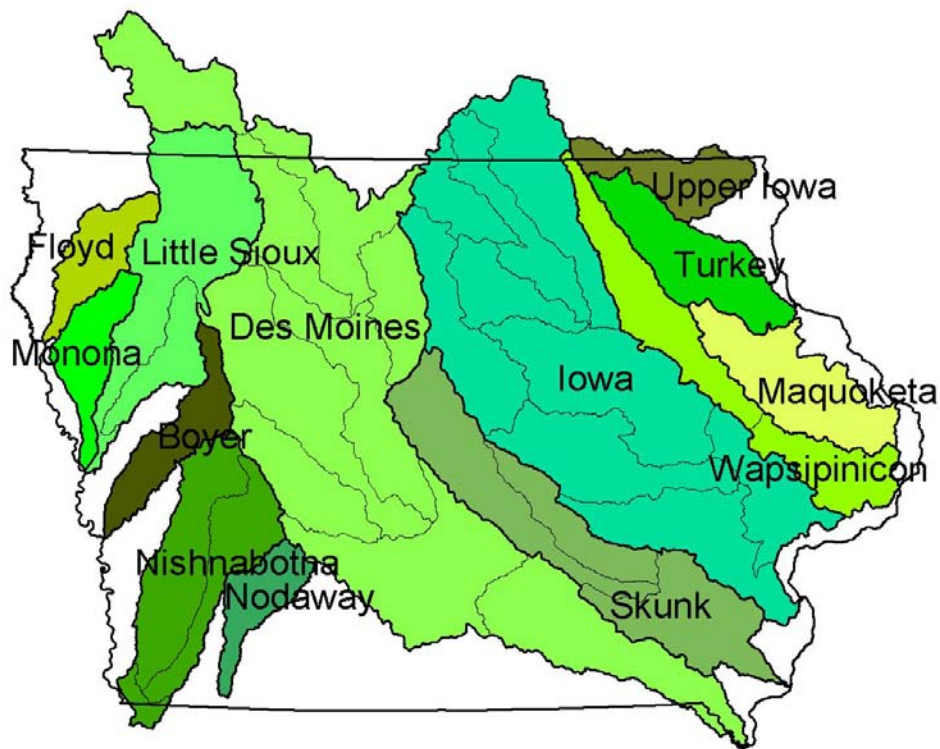


FIGURE 1. Study area and watershed delineations

model version 3060 (Izaurrealde et al. 2002). The current version of the EPIC model features enhanced carbon cycling routines (Izaurrealde et al. 2002) that are based on the approach used in the Century model (<http://www.nrel.colostate.edu/projects/century5/reference/index.htm>; Parton et al. 1994). More generally, EPIC is a model that estimates the changes in erosion, carbon sequestration, and nutrient runoff measured at the edge of the field from changing farming practices. Inputs into the model include weather, soil, landscape, crop rotation, and management system parameters. When land is retired from crop production, we assume that annual grasses are planted and maintained on the land and we run a 30-year simulation with EPIC to predict the carbon sequestration level associated with this change. For each NRI point, we also calculate the soil erosion reduction at the edge of the field.

In addition to EPIC, we also rely on estimates from a watershed-based model to assess the conservation policies. Unlike carbon sequestration, a key concern in the design and implementation of policies to improve water quality is to recognize that there are critical interactions between land uses within a watershed that affect the total water

quality level obtained. Thus, for otherwise identical tracts of land, more water quality improvement may occur from retiring a piece of land from production that is located downstream from numerous other cropped points. The potential filtering effect is just one example of the physical processes that a fully integrated watershed hydrological model should capture. So that we can study in-stream water quality changes, we employ SWAT to estimate changes in nitrogen, phosphorous, and sediment loads from retiring a particular set of parcels from production within a watershed. The model represents hydrology, plant growth, erosion, fate, and transport and various management practices (Arnold et al. 1998 and Neitsch et al. 2002). The SWAT model achieves a high level of spatial differentiation by dividing a large watershed into a number of subwatersheds and then dividing further into Hydrologic Response Units (HRUs). Each HRU represents homogeneous land use, management, and soil characteristics. To estimate the in-stream water quality consequences of the increase in land set-aside, we have calibrated the SWAT model for each of the watersheds identified in Figure 1 to baseline levels.² By running the model at the set-aside levels “after” the policy, we can compute the changes in water quality attributable to the increase in land set-aside.

Given that political boundaries and watershed boundaries do not perfectly correspond in Iowa, there is something of a geographic mismatch between our study regions on the cost side and those on the water quality side.

The watersheds studied correspond to 13 outlets, at which the in-stream water quality is measured. The water quality measures of interest are sediment, nitrogen, and phosphorus. For the cost analysis we consider placing the identified set of practices all across the state and exclude the pieces of the watersheds that fall outside of the state boundaries (for example, the section of the Des Moines River watershed that falls in Minnesota). Thus, the costs and water quality benefits we report are not quite consummate: one represents a political boundary (the statewide costs); the other represents a natural system boundary. Direct comparisons between the aggregate cost and water quality benefit may be misleading, although the unit costs and benefits (per acre costs and/or per outlet of the watershed measures) can still be appropriately compared.

3. Results

The three policies considered are quite similar in terms of the acreage enrolled, as shown in Table 1. However, Figures 2, 3, and 4 show that the location of those acres is very different. The water quality policy would enroll more acres along the Mississippi and Missouri rivers, in the more hilly parts of the state. The carbon policies, on the other hand, would focus on the central part of Iowa, in the ecoregion known as the Des Moines Lobe, a flat area with very productive agriculture, which is particularly suited for carbon sequestration. Note the similarities between Figures 3 and 4. Because the criterion used to enroll land is the same, the only difference is the additional amount of land allowed in the equal area scenario.

The policies are extremely different in the levels and locations of environmental benefits. The carbon-based policies sequester about 10 times as much carbon as the policy based on water quality. Similarly, the water quality policy is around four times more effective than the carbon-based policies in reducing soil erosion at the edge of the field as calculated with EPIC.

Figures 5 and 6 illustrate the location of the carbon sequestered across the state in the water quality policy and in the carbon policy that keeps costs constant,³ and Figures 7 and 8 show the location of the edge-of-field soil erosion reduction across the watersheds studied. As with the levels of the benefits, we find the location of the benefits to be

TABLE 1. Simulation results

Policy	Budget, (mil. \$)	Area Enrolled (mil. acres)	Carbon Sequestered (mmt)	Erosion Reduction (Edge of Field) (mmt)	Sediment Load Reduction (In Stream) (mmt)	Nitrates load reduction (In Stream) (tmt)	Total P Load Reduction (In Stream) (tmt)
Water quality	242.1	2.2	0.3	16.0	2.2	2.47	1.49
Carbon, equal area	287.0	2.2	3.1	4.0	0.0	10.05	0.81
Carbon, equal cost	242.1	1.9	2.7	3.5	0.0	8.96	0.57

Note: The carbon and edge of field reductions are calculated for the state, while the in-stream water quality measures are the sum of the reductions across the 13 watersheds of Figure 1. Therefore, the numbers are not exactly commensurate.

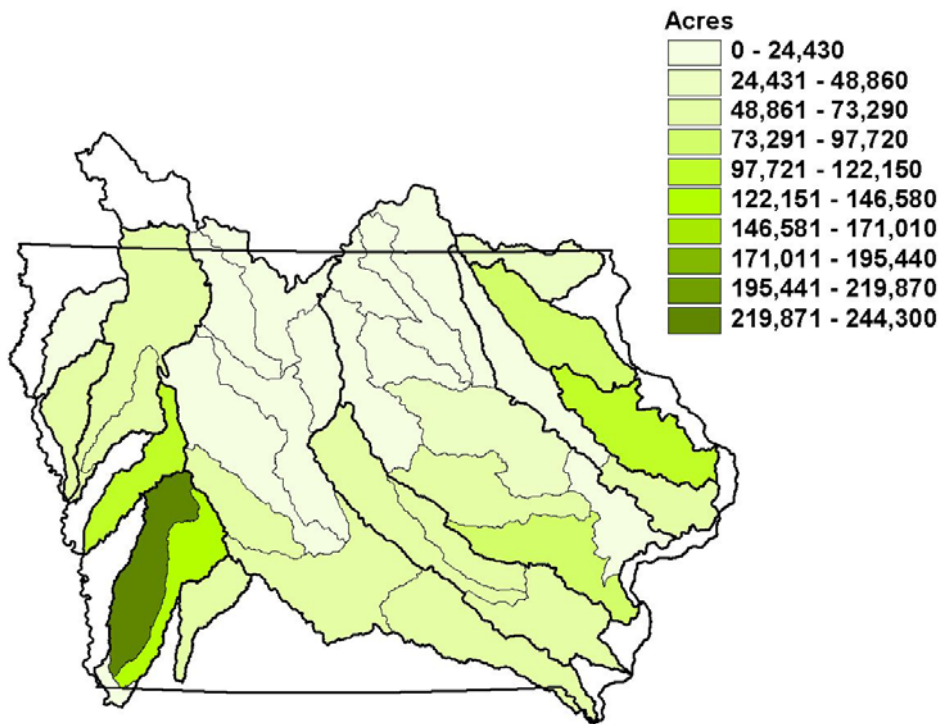


FIGURE 2. Set-aside acres—water quality policy

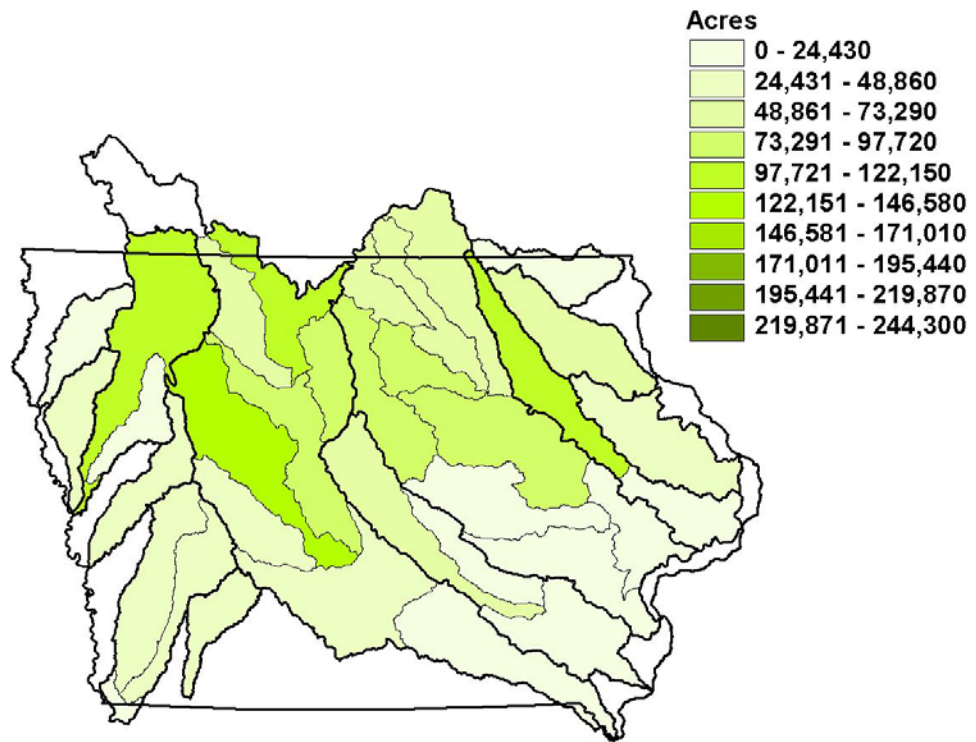


FIGURE 3. Set-aside acres—carbon policy with equal cost

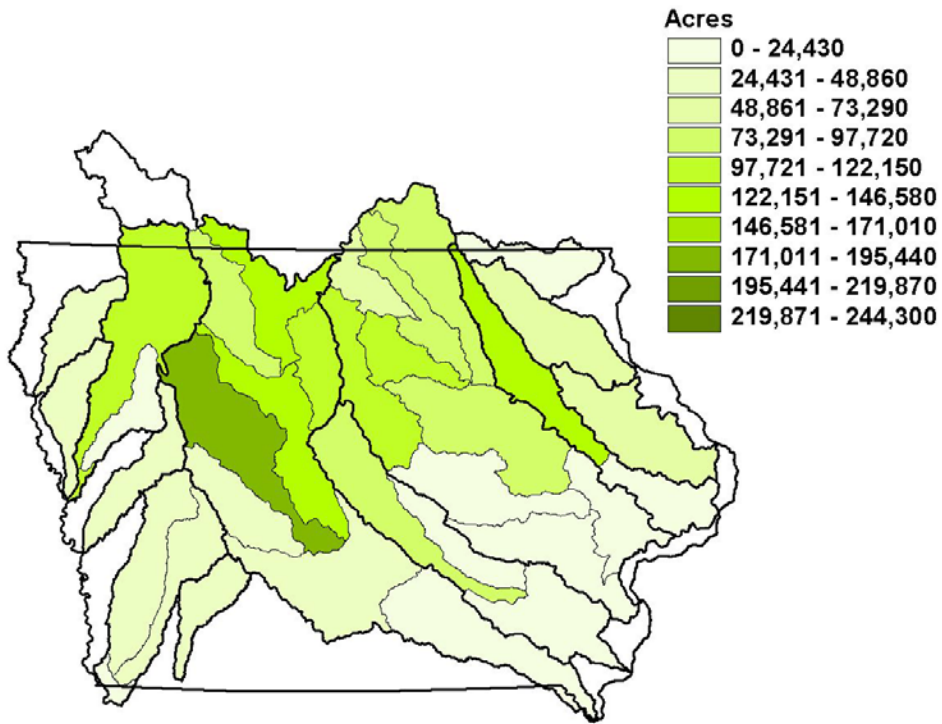


FIGURE 4. Set-aside acres—carbon policy with equal acres

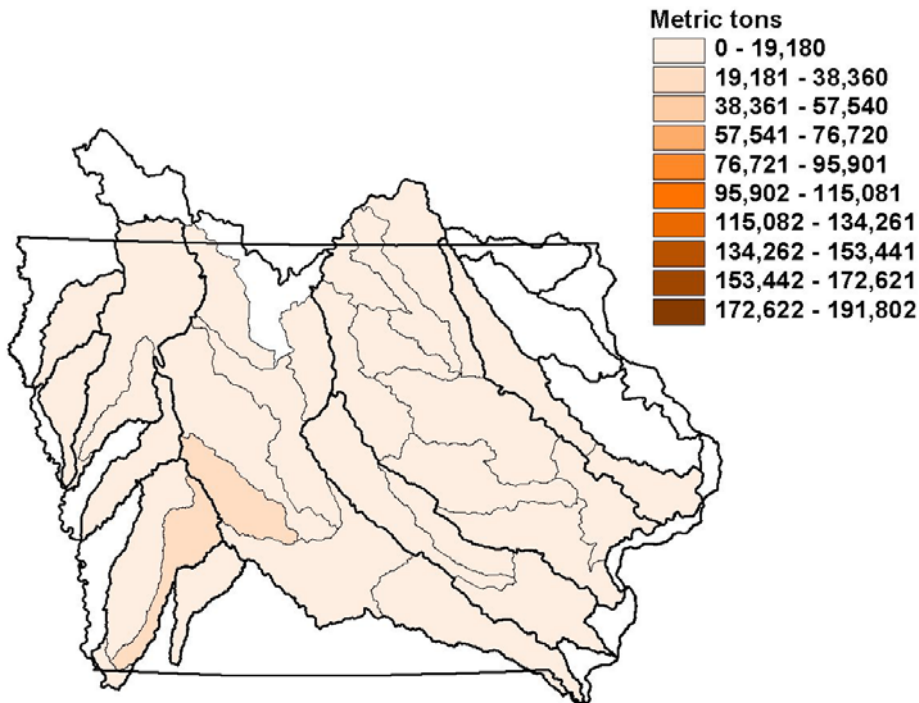


FIGURE 5. Carbon sequestration from water quality policy

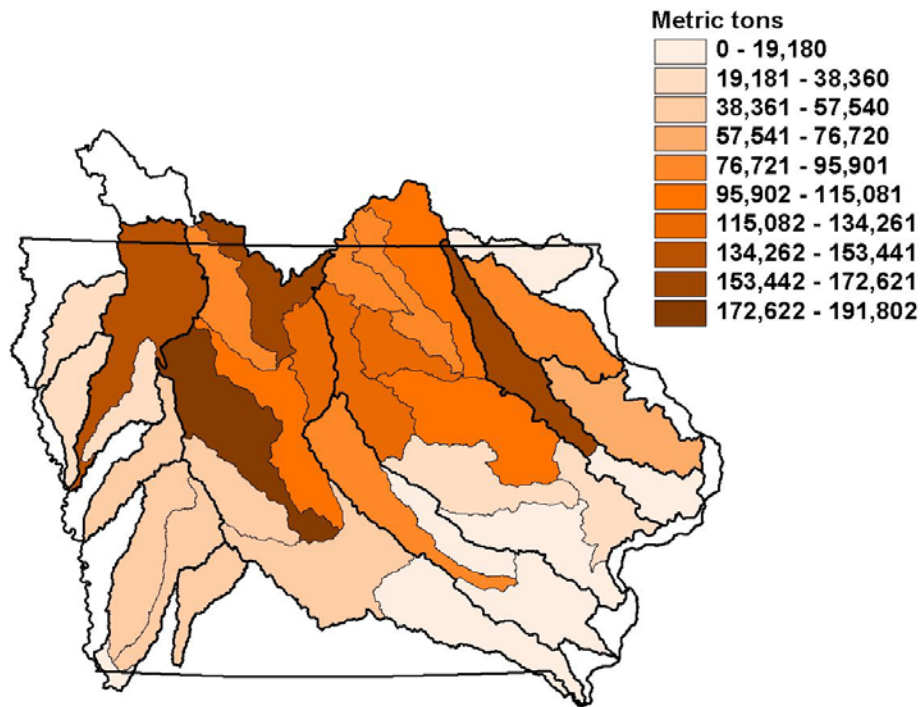


FIGURE 6. Carbon sequestration from carbon policy with equal cost

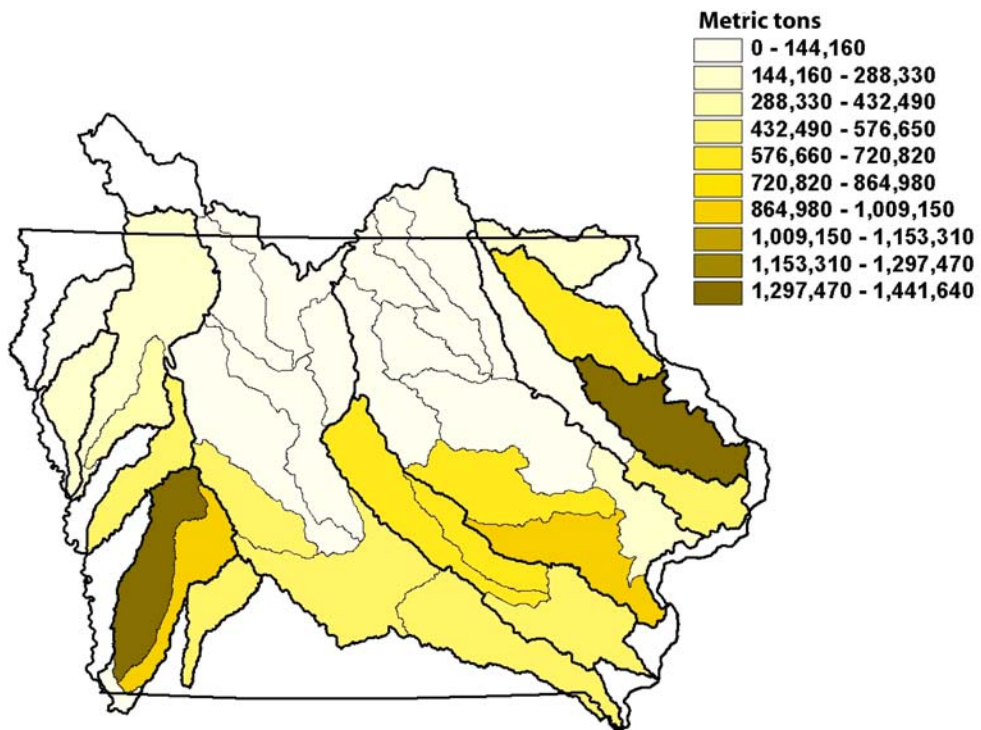


FIGURE 7. Edge-of-field erosion reduction from water quality policy

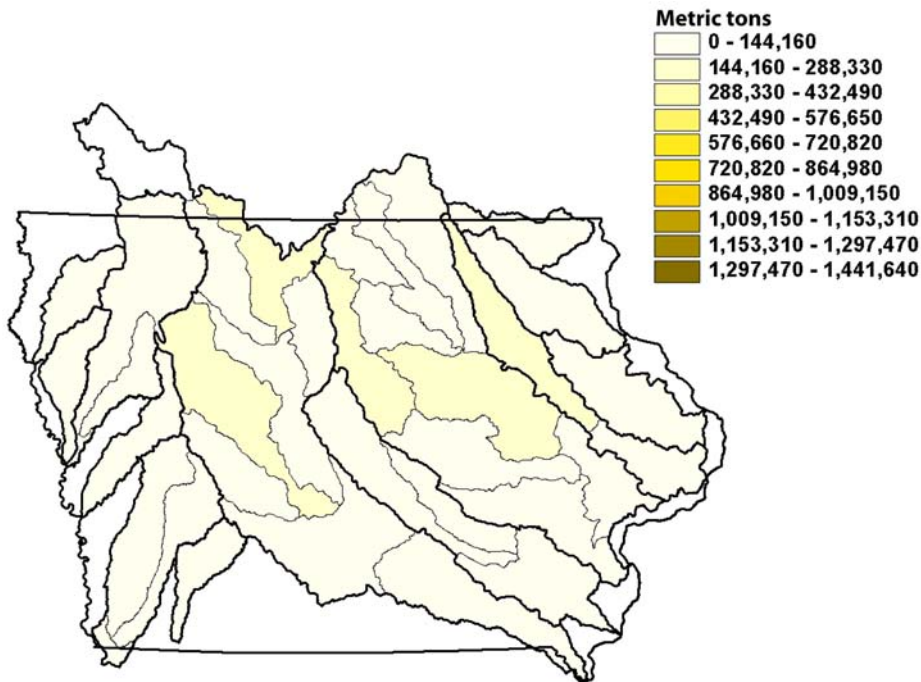


FIGURE 8. Edge-of-field erosion reduction from carbon policy with equal cost

quite different across the policy designs. The extreme differences are at least partly driven by the fact that both policies are affecting only a relatively small percentage of the cropland. More substantial overlap of benefits could occur in policies applying to more extensive areas, as was found in the case of conservation tillage adoption in Iowa (Kurkalova, Kling, and Zhao 2004).

The policy design also substantially alters the in-stream water quality impacts, as reported in Tables 2 and 3. For a water quality policy (Table 2), with only one exception, sediment is reduced across all the watersheds from the baseline.⁴ Since the enrollment criterion was the EI, this is an expected result. If we sum the sediment loads across the watersheds, the total reduction is 11 percent. The higher reductions are along the Mississippi River and in Southwest Iowa, reflecting the watersheds where more land is put out of production. Reductions in phosphorus loads are somewhat smaller, but still significant, as phosphorus loads are linked to sediment. The policy, however, is not effective in reducing nitrates, and, since nitrates are the most important form of nitrogen in surface water, total nitrogen is not reduced by much.

In contrast, for a carbon policy (equal cost) (Table 3), we find that in several watersheds, particularly in eastern Iowa, the sediment loadings actually increase. Further

TABLE 2. Percentage reduction in sediment, N and P in stream for CRP water quality policy

		Sediment	Nitrate	Org N	Total N	Org P	Min P	Total P
1	Floyd	4	-2	3	-1	1	-4	-3
2	Monona	6	-2	10	0	11	7	8
3	Little Sioux	-2	-9	8	-6	13	6	7
4	Boyer	17	0	19	5	19	17	17
5	Nishnabotna	25	0	15	3	15	18	18
6	Nodaway	21	6	20	10	20	19	19
7	Des Moines	3	3	8	5	9	6	7
8	Skunk	20	7	15	8	14	15	15
9	Iowa	1	4	0	2	0	5	1
10	Wapsipinicon	4	-2	-4	-2	-2	-11	-9
11	Maquoketa	11	-2	11	0	9	9	9
12	Turkey	19	1	14	3	14	13	13
13	Upper Iowa	13	2	6	3	0	3	2
	Total	11	1	5	2	4	7	6

TABLE 3. Percentage reduction in sediment, N and P in stream for CRP carbon (equal cost) policy

		Sediment	Nitrate	Org N	Total N	Org P	Min P	Total P
1	Floyd	0	2	-1	1	-3	-8	-7
2	Monona	0	1	-1	1	0	-3	-2
3	Little Sioux	0	-1	1	0	0	2	2
4	Boyer	2	2	1	2	1	1	1
5	Nishnabotna	-1	-3	0	-2	1	-1	0
6	Nodaway	12	3	12	6	12	12	12
7	Des Moines	0	8	9	9	8	9	9
8	Skunk	5	4	4	4	4	4	4
9	Iowa	1	9	3	6	2	2	2
10	Wapsipinicon	-7	-1	-15	-3	-15	-17	-16
11	Maquoketa	-3	1	-12	-1	-11	-11	-11
12	Turkey	-8	1	-4	0	-4	-4	-4
13	Upper Iowa	1	-1	-1	-1	0	-3	-2
	Total	0	4	3	4	3	2	2

investigations on this result are needed, but the likely reason for this finding is the fact that the NRI points selected for carbon sequestration purposes in these areas were farmed with effective conservation practices such as filter strips and grassed waterways that actually were more effective at capturing sediment in our modeling system. On the other hand, it is interesting to note that the carbon-based policies are better than the water quality policy in reducing nitrates. The reason is that the land taken out of production in the carbon-based

policies is prime agricultural land, heavily fertilized. The higher reductions are in the central watersheds of the Des Moines Lobe, where most of the acres of set-aside in the carbon-based scenarios are located. These results suggest that even a water quality–based policy may have to deal with trade-offs, depending on which measure of water quality is used. Implicit in our set up was a heavier weight on the importance of sediment and phosphorus loadings reduction, since the NRI points were selected based on land erodibility.

4. Indirect Monetization and Lessons for Policy Design

As noted in the introduction, it is not straightforward to assign monetary values to the nonmarket goods of reduced nutrients or soil erosion. It is also not straightforward to assign a monetary value associated with sequestered carbon, although a number of analyses have been undertaken that suggest likely values of the price of carbon in an efficient and fully functioning trading program. In this section, we undertake a simple break-even type of analysis to consider the marginal value of erosion and nutrient reductions that would have to hold under a range of likely carbon prices to make the alternative policies socially efficient, that is, to be assured that they would pass a simple cost-benefit test. We are implicitly assuming in this exercise that the hypothetical carbon prices would reflect the social value of carbon reductions.

First, we estimate the minimum value of erosion reduction that would have to be held by society to justify the policy outlay. This is calculated assuming that the total cost of the policy must be equal to the monetary valuation of the benefits from the policy. Two benefits are considered at a time: carbon sequestration and erosion reduction. The monetary valuation of the benefits is equal to the monetary valuation of carbon sequestration, C , multiplied by its price, p^C , plus the monetary valuation of erosion reduction, E , multiplied by its price, p^E . Thus, for the policy to “break even,” we need

$C \cdot p^C + E \cdot p^E = Budget$. For the scenarios considered, we have the *Budget*, C , and E .

The literature provides several estimates of p^C . We use these pieces of information to find p^E from the previous equation. Specifically, we consider three prices for carbon, $p^C = 10, 50, \text{ and } 100$, and get three prices for erosion, p^E , for each of the scenarios considered. The results are reported in Table 4. At high enough carbon prices, carbon

TABLE 4. Simulation results: Break-even price of erosion (\$ per metric ton)

Policy	Carbon Price of \$10 per mt	Carbon price of \$50 per mt	Carbon price of \$100 per mt
Water quality	14.9	14.2	13.3
Carbon, equal area	64.5	33.6	-5.0
Carbon, equal cost	61.4	31.1	-6.8

benefits are more than enough to justify these carbon-based policies. On the other hand, if the goal of the policy is to improve water quality/reduce erosion, increasing the price of carbon will not affect the value of erosion reduction. The reason is that the water quality policy reduces erosion by 16 million metric tons, versus 0.3 million tons of carbon sequestered. Carbon prices would have to rise much higher than \$100/metric ton to affect the break-even price of erosion. Pimentel et al. (1995) estimate an off-site⁵ cost of erosion of about \$3 a ton in 1992 dollars. This estimate assumes that more than half of the costs are caused by wind erosion and does not consider biological impacts such as the effect on biodiversity. As the authors of the study note, this is likely to underestimate the impact of soil erosion. However, even tripling the number calculated by Pimentel et al. would produce prices lower than the lowest implicit break-even prices from our scenarios. This suggests that a land set-aside policy is unlikely to be justified on the grounds of soil erosion alone.

The calculation of break-even prices for in-stream pollutant load reductions is more complex to interpret for several reasons. First, there is less precedent on how the reduction should be measured, and therefore there is no consensus on how the benefits should be compared. We are using reductions in metric tons of the loads at the outlet of each watershed, but, as we noted earlier, this does not directly correlate with measures such as swimmability and transparency. The units of analysis are not immediately apparent either. For example, Table 5 reports the break-even prices for total phosphorus load reductions, using metric tons of reduction as the common unit. Because the edge-of-field values are much higher than the reductions in-stream, the price of phosphorus load reductions is extremely high compared to the price of carbon.

An alternative way to assess the trade-offs is to present the budget outlays for each benefit, either in dollar terms or in percentages, as illustrated in Table 6. This allows a comparison of the trade-offs involved without having to decide the appropriate units of

TABLE 5. Simulation results: Break-even price of total P load reduction (\$ per metric ton)

Policy	Carbon Price of \$10 per mt	Carbon price of \$50 per mt	Carbon price of \$100 per mt
Water quality	51,540.3	152,540.4	142,800.2
Carbon, equal area	60,373.0	164,781.7	-24,736.1
Carbon, equal cost	70,488.1	190,620.3	-41,870.7

TABLE 6. Simulation results: Budget break-up (percentage values in parentheses)

Policy		Carbon Price of \$10 per mt	Carbon price of \$50 per mt	Carbon price of \$100 per mt
Water quality	Carbon	2,905,800 (1.2%)	14,529,000 (6.0%)	29,058,000 (12.0%)
	Co-benefit	239,162,090 (98.8.0%)	227,538,890 (94.0%)	213,009,890 (88.0%)
Carbon, equal area	Carbon	30,698,440 (10.7%)	153,492,200 (53.5%)	306,984,400 (107.0%)
	Co-benefit	256,252,000 (89.3%)	133,458,240 (46.5%)	-20,033,960 (-7.0%)
Carbon, equal cost	Carbon	26,602,270 (11.0%)	133,011,350 (54.9%)	266,022,700 (109.9%)
	Co-benefit	215,465,620 (89.0%)	109,056,540 (45.1%)	-23,954,810 (-9.9%)

comparison. Clearly, this is an issue that needs further investigation and additional empirical analysis of various categories of co-benefits.

Additional analysis is also needed because, as our results show, viewing different benefit indicators as the primary benefits and other indicators as co-benefits in policy making would produce substantially different policy scenarios. In particular, there should be careful consideration of the environmental goals of conservation policies implemented on only a fraction of cropland because the trade-offs for such programs are likely to be more extreme. Policies included in this category, besides land set-aside, are likely to be wetland conversion and adoption of no till, as opposed to reduced/mulch till, which is likely to be more widespread.

Though our results are limited to the state of Iowa, and to the case of set-aside for water quality or carbon sequestration purposes, it is apparent that at our scale of analysis, the amount of co-benefits depends on what indicators are used to measure water quality. Local

water quality issues are usually linked with phosphorus, since phosphorus tends to be the limiting factor in freshwater bodies. However, at least until recently, there was a consensus that nitrogen was the main cause of hypoxia in the Gulf of Mexico. In general, this study shows that improving “water quality” in the sense of reducing nutrient or sediment loadings is too vague. Even if this is taken to refer to in-stream nutrients, because the responses of nitrogen and phosphorus to conservation efforts are not well correlated, this terminology may not provide much guidance. In the case of the CRP’s environmental benefits index, for example, the water quality benefits refer both to a generic impairment that includes nutrients and specifically to sediment (USDA-FSA 2004).

We have not considered co-benefits such as wildlife habitat or biodiversity, which are harder to assess because they have only indirect links to “hard” environmental indicators such as water quality and depend on a large number of factors. However, the scenario results provide some indication of the issues involved with including these co-benefits. For example, the GAP (Gap Analysis Program) for Iowa suggests that species richness for mammals, birds, amphibians, and reptiles is higher along the Mississippi and Missouri rivers, and in southern Iowa (Kane et al. 2003). The Des Moines Lobe area, where it would be more efficient to retire land for carbon sequestration purposes, has lower biodiversity, since the land is intensely cropped, and there are few forests, grasslands, and rivers. This indicates that a land set-aside program similar in size to those analyzed here, and designed to preserve biodiversity, would focus largely on the same areas as our water quality scenario.

In terms of policy design, our results support the use of different instruments (programs) to achieve different environmental goals. A weighted index could also be used. In such a case, because the overlap of benefits can be modest, the weights would essentially reflect the relative priority of each environmental improvement category and its share of the budget.

Endnotes

1. Note that this area is in addition to the already existing CRP area.
2. The details of the calibration and validation process for SWAT can be found in Jha et al. 2005 and Gassman et al. 2005.
3. The two carbon policies are very similar; therefore, only the results for the equal cost case are shown.
4. See the appendix for the baseline loads.
5. We do not consider the on-site impacts of soil erosion because they directly affect productivity and should be accounted for in the farmer's profit-maximizing problem.

Appendix

Baseline Loads

		Baseline Loadings						
		Annual Average Values in Metric Tons (1980-1997)						
	Baseline	Sediment	Nitrate	Org N	Total N	Org P	Min P	Total P
1	Floyd	241,423	7,125	1,281	8,406	104	301	405
2	Monona	198,589	4,847	757	5,605	58	269	327
3	Little Sioux	632,456	23,851	4,730	28,569	381	1,687	2,067
4	Boyer	777,245	4,947	2,044	6,991	161	548	709
5	Nishnabotna	1,968,399	8,257	2,399	10,656	207	794	1,001
6	Nodaway	520,045	3,312	1,344	4,656	113	405	518
7	Des Moines	2,174,303	38,252	25,098	63,349	2,411	4,664	7,075
8	Skunk	3,800,345	28,122	3,965	32,087	343	1,678	2,021
9	Iowa	3,423,237	54,050	49,688	103,738	7,000	786	7,786
10	Wapsipinicon	2,238,966	27,533	3,950	31,484	342	1,228	1,571
11	Maquoketa	1,218,739	14,781	2,965	17,746	245	674	919
12	Turkey	1,297,814	12,436	2,406	14,842	208	642	850
13	Upper Iowa	730,155	3,675	1,008	4,683	112	267	379
	Total	19,221,717	231,188	101,635	332,811	11,686	13,943	25,629

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