Targeting and the Economics of Cumulative Watershed Effects

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Paper prepared for presentation at the American Economics Association Annual Meeting, Denver, Colorado, August 4, 2004

1. Introduction

The action plan to reduce hypoxia in the Gulf of Mexico and the large number of lawsuits related to the Total Maximum Daily Loads regulation illustrate the heightened concerns over water quality issues, both at a local and regional level. Non-point sources, in particular agricultural ones, are a significant source of pollutants. This explains in part the increased availability of conservation funds for agriculture in the last decade. The latest Farm Bill in particular, through the Conservation Security Program (CSP), has introduced several innovations in conservation policy, including payments for adopting conservation practices on land in prodiction and watershed-level targeting.

The institution of watershed-level targeting presents interesting policy design questions because environmental watershed-level benefits from conservation practices have not, so far, been the subject of many study. As far as we are aware, there is no study that incorporates the analysis of both the environmental benefits and the economic costs. Such an analysis requires extensive micro-level data, river basin scale models, and a simultaneous economic and hydrologic analysis.

The institution of watershed-based conservation also means that edge-of-field benefit analysis, which has been the standard so far, does not provide adequate information to assess the effectiveness of policy. The impact of conservation practices on water quality, as measured at an outlet, is typically not simply the sum of the reduction of sediment at the edge of each field. This also implies that water quality benefits from adopting a conservation practice downstream may be dependent on the level of conservation practice adoption upstream. More in general, the water quality benefits will depend on the hydrology of the watershed at hand.

Clerly, the point at which the water quality benefits are measured is likely to be very important. For example, in a large watershed, assessing benefits only at the main outlet may give disproportionate representation to the lower part of the watershed. Assessing benefits at several points may be more representative of the water quality impact of the policy, but it may mean a substantial increase in expenditure (most monitoring data would have to be daily, and include the measurement of several pollutants). Furthermore, historical data with which to compare the *ex post* measurement are limited.

The goal of the current analysis is two-fold. First, we discuss what watershed-based conservation policy effectively mean when assessing policy performance. In particular, we focus on the parallel between watershed-based conservation policy and ambient-based pollution control.

Secondly, we analyze the issue of targeting within a watershed and its link to the point of measurement, or the outlet at which water quality assessments are made.

We start by developing a theoretical model that illustrates the mechanisms of the discharge and loadings of a pollutant at the watershed level, using sediment as an example, and the role of conservation practices in reducting discharge. We focus on conservation budgets for adopting conservation tillage, one of the most common best management practices available to farmers in the Midwest and one of the practices included in the CSP. We then develop an empirical analysis of the issue by focusing on the conservation practice of reduced tillage. To do so, we link a tillage adoption model based on the National Resource Inventory (NRI) data (Nusser and goebel, 1997) to the Soil and Water Assessment Tool (SWAT) model. The SWAT model is a watershed-level water quality model that calculates loading and concentrations of sediment and nutrients at the overall watershed outlet and for each subwatershed within the area of analysis. We can therefore examine the effectiveness of various configurations and targeting policies aimed at reducing sediment loads in the water at the watershed and subwatershed level through the adoption of conservation tillage. This allows us to determine empirically the extent to which non-linearities are present in the system and across watersheds and what are the efficiency losses from ignoring these non-linearities.

We apply our analysis to the Des Moines River watershed, a large, mostly agricultural, watershed in Iowa and Minnesota that drains into the Mississippi river, and is therefore a contributor to the hypoxic zone in the Gulf of Mexico (CENR, 2000). The scale of the water quality degradation problem in the area is demonstrated by the inclusion of 43 stream segments and lakes within the watershed on the current U.S. Environmental Protection Agency (USEPA) listing of impaired waterways. In this paper, we focus on sediment reduction. High sediment loads are a water quality problem because sediment can fill reservoirs and cause channel siltation, thereby raising the costs of water treatment and channel dredging. Moreover, high levels of sediment can have a negative impact on fish and wildlife, greatly reducing the economic and recreational value of streams and lakes.

We use the 1997 levels of conservation tillage adoption as our baseline, and examine three policy scenarios. In the first, we assume that conservation funds are allocated exclusively either upstream or downstream in the watershed, and in the second, we assume funds are targeted to the sub-watersheds with the highest initial sediment loads. Finally, we study various levels of conservation tillage payments not spatially targeted. This is a policy payment scheme after those discussed in the proposed CSP implementation rules (USDA/NRCS, 2004). Specifically, we consider the schemes under which farmers are offered the maximums of $10, $20, or $30 per acre for adopting conservation tillage. In each case, we use the econometrically fit tillage adoption model to predict which acreage will convert to conservation tillage under subsidy payments. The SWAT model is then used to estimate the changes in sediment loading resulting from the policy.

The study provides a numerical evaluation of ambient-based pollution reduction, and how the optimal allocation of funds across the landscape may be affected be the level of funds available. The quantification of the effects is also expected to contribute to the discussions of pollutant trading involving non-point sources. While the trading idea is being actively pursued by regulators, because of cumulative watershed effects, there is substantial uncertainty over the permit-to-emission ratios to use. This study will help identify the permit-to-emission ratios.

2. The Theoretical Model

The model developed below is discussed in terms of sediment loads. However, the model could also be applied to other pollutants such as nitrogen and phosphorous, or bacteria. In all these cases, non-linearities are allowed to be present. Our analysis could also be extended to a combination of pollutants. In that case, it would be necessary to specify weights for the relative importance of each pollutant to create an index[1]. We simplify by abstracting from the impact of flow on sediment discharge and delivery, and

[1] This would be similar to the construction of the Environmental Benefit Index that is used to enroll land in the Conservation Reserve Program (CRP).

concentrate on the processes going on across watersheds rather than within each watershed.

We consider a river which contains *I* watersheds. We order the watersheds from upstream to downstream (note that this will not necessarily be a line): 1, 2, 3…*I*, and use the subscript *ui* to denote the quantities associated with the watersheds upstream of watershed *i*. Define q_i as the amount of sediment in the water at the outlet of watershed *i*. We define d_i as the amount of sediment discharged in the water in watershed *i*. Thus, $q_1 = d_1$ and $q_i = f(d_{ui}) + d_i \Rightarrow q_I = f(d_1, d_2, d_3 ... d_{I-1}) + d_I$. The sediment discharged in the watershed *i* can be affected by a scalar conservation effort e_i. The amount of sediment discharged is a non-increasing function of the conservation effort in the watershed, i.e. $\frac{\partial d_i}{\partial e_i} \leq 0$. c_i. Then $d_i = g\left(e_i\right)$ where $g' \leq 0$. Therefore $q_i = f\left[d_{ui}\left(e_{ui}\right)\right] + d_i\left(e_i\right)$.

Finally, we define a conservation policy as a collection of conservation efforts $\overline{e} = \left\{e_1, e_2, ..., e_I\right\}$. The water quality benefits of the policy can be defined in several ways. First, they could be the sum of the reduction in sediment discharge at each outlet.

<u>Definition</u> Emission-based evaluation of water quality benefits of a policy $\overline{e}$ at the j-th watershed outlet, $b_j^{emission}\left(\overline{e}\right)$, is defined as the sum of the discharge reductions at the watershed j and all the watersheds upstream of it. That is,

$$b_j^{emission}\left(\overline{e}\right) \equiv \left[\sum_{uj}\left(q_k\left(0\right) - q_k\left(e_k\right)\right)\right] + \left(q_j\left(0\right) - q_j\left(e_j\right)\right).$$

<u>Definition</u> Ambient-based evaluation of water quality benefits of a policy $\overline{e}$ at the *j*-th watershed outlet, $b_j\left(\overline{e}\right)$, are defined as the difference between the sediment in the water in the absence of the policy and the sediment in the water in the presence of the policy, at the *j*-th watershed outlet. That is,

$$b_j\left(\overline{e}\right) \equiv q_j\left(\overline{0}\right) - q_j\left(\overline{e}\right)$$

Thus, the emission-based accounting of water quality benefits would double-count benefits and therefore tend to overestimate impacts compared with the ambient-based accounting of water quality benefits.

Note that the benefits could also be defined in terms of the level of conservation effort, as $b_j^{cons.effort}\left(\overline{e}\right) \equiv \sum_{i=1}^{j}\left(e_i - 0\right)$. Most of the current policies are practice-based, and so are concerned primarily with the effort level rather than its effectiveness in reducing pollution.

The shown difference in the emission-based and the ambient-based evaluation of benefits has important implications for targeting conservation policy. A policy that is designed to minimize the discharges from the watersheds comprising the river basin would not, in general, minimize the amount of sediment at the bottom of the basin. In the empirical application below we demonstrate this difference.

Therefore, while, the total costs of the conservation practice, $\sum_{I}^{1} c_j\left(e_j\right)$, are additive across the watersheds, the benefits generally are not. Note that additive costs across space are not necessarily linear in the level of adoption of conservation practice, since the marginal cost of enrolling land in the program depends on the cost of giving up conventional tillage for the marginal parcel of land, which itself is a function of land characteristics and profitability.

As we noted, the system is non-linear at the spatial level. Clearly, these non-linearities have implications for an efficient allocation of conservation funds across the landscape. For example, in general, the gains from conservation practices at the outlet for a given level of conservation practices will be higher if the practices are adopted near the outlet, because part of the sediment from upstream gets deposited, and this decreases the downstream gains from reducing the sediment. However, since the cost of adoption is not spatially homogeneous, if enrolling land upstream is cheaper than enrolling land downstream, it may not make sense from an economic standpoint to limit payments to the downstream portions of the watershed. The determination of the efficient allocation of funds is an empirical issue that has to take into account the hydrology of the watershed.

As the empirical results below demonstrate, however, optimal targeting would have to take into account – indirectly, through their effects on the marginal cost-benefit curves, other watershed characteristics as well. For example, if funding is limited to certain sub-watersheds, as the latest CSP rule (USDA/NRCS, 2004) suggests, the level of forest and pasture in those watersheds may put a ceiling to the potential gains from conservation.

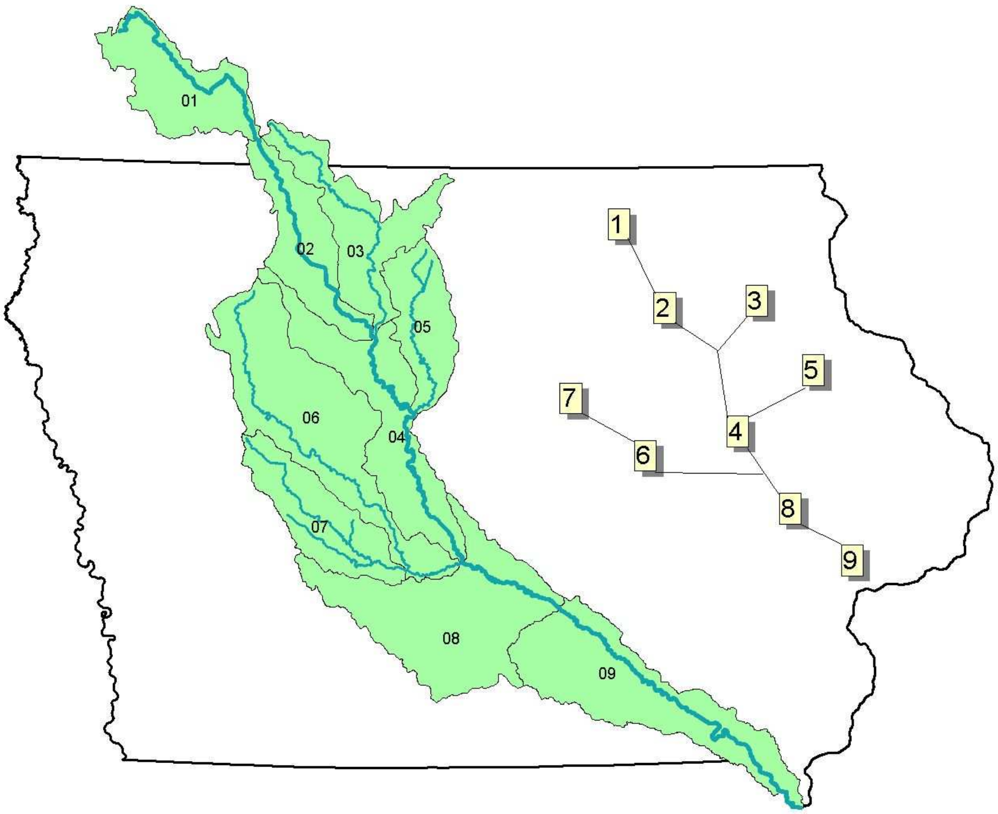
The Empirical Model

The empirical model has to main components: the conservation tillage adoption model and the Soil and Water Assessment Tool (SWAT) model. The economic model presented in detail in Kurkalova, Kling, and Zhao (2003) is used to predict the level of subsidy required for adoption of conservation tillage at every crop production point in NRI database in Iowa.

The SWAT model is a conceptual, physically based long-term continuous watershed scale simulation model that operates on a daily time step. Flow generation, sediment yield, and non-point-source loadings from each sub-watershed are routed through channels, ponds, and/or reservoirs to the watershed outlet (Arnold et al. 1998). The model is capable of simulating a high level of spatial detail by allowing the division of a watershed into a large number of subwatersheds. In this study, the subwatersheds correspond to the U.S. Geological Survey (USGS) Hydrologic Cataloging Units that are commonly referred to as "8-digit" watersheds (see Fig. 1). This is the same level of analysis of the Hydrologic Unit Model for the United States (HUMUS) modeling framework (Arnold et al. 1999).

Within each subbasin there are subwatersheds with unique combinations of soils and land-uses, called hydrologic response units (HRU). HRUs are virtual units of analysis. We know in which subbasin they are, but we do not know where they are located within that subbasin. The NRI database is the basis for the construction of the HRUs. NRI points are aggregated on the basis of soil characteristics, crop and rotation, and tillage practice as predicted from the tillage adoption model.

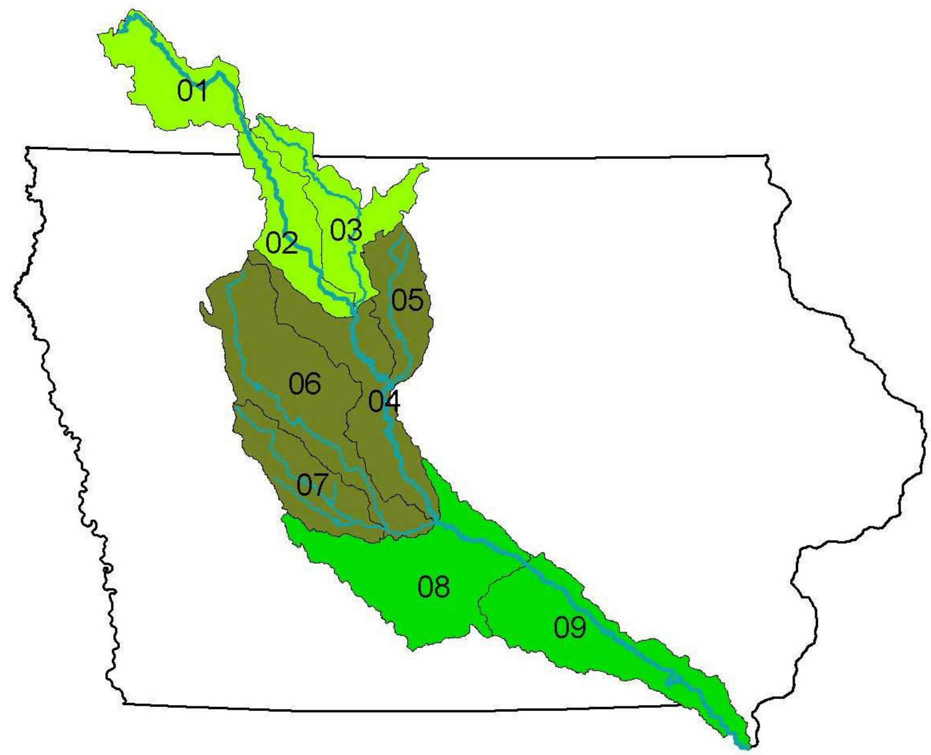
Figure 1 – The structure of the Des Moines River watershed.



The Des Moines River is in darker blue, its major tributaries in lighter blue. The tree to the right shows the hydrological structure of the watershed.

The integrated models are used to examine policy scenarios, where conservation funds are allocated spatially. Specifically, we divide the watershed in three areas, up-, mid- and downstream (Fig. 2). This allows us to examine spatial targeting issues without having to consider a very large number of alternative targeting policies. The second scenario focuses on targeting the worst watersheds in terms of the highest loads per hectares. The final scenario examined concentrates on various levels of conservation tillage payments not spatially targeted: any producer in the watershed can participate. Specifically, we look at payment levels of \$10, \$20, and \$30 per acre. Throughout the analysis we assume an existence of a true-cost-revealing mechanism that allows the policymaker enroll the producers in the program by paying them exactly the minimum per acre subsidy needed.

Figure 2 – The division of the Des Moines River watershed.



Results

We start by presenting some of the characteristics of the watersheds at the baseline, that is, with the level of conservation tillage and the lend uses prevalent in 1997, and in the absence of a conservation tillage subsidy. The watershed is divided in nine sub-basins of varying areas (Table 1). The area of the upstream watersheds sums up to about 25% of the total, and that of the midstream ones at 44%, so that 31% is left to downstream sub-basins. The number of HRUs in which each of the subbasin is divided at the baseline ranges from 82 to 215. In general, there is one HRU for land in CRP, one each for pasture, forest and "other" land, a residual category that includes urban land. All the other HRUs are devoted to cropland.

Table 2 shows the land use by sub-basin. In the whole watershed, about 68% of the watershed is cropped (excluding CRP – the CRP total area is about 4.66% of the total watershed area). However, there are substantial differences among the sub-watersheds. The northern watersheds are more heavily cropped, and, correspondingly, have less land in CRP, forest and pasture. This reflects the fact that the land in the northern part of the watershed is more productive. In the South, the less productive land is used for pasture based livestock production or is left not cropped, as forest.

Table 1. The sub-basins'.

Reach	Reach area as % watershed area	# of HRU in the baseline
1	8.538	191
2	7.636	138
3	9.216	128
4	12.042	131
5	6.774	82
6	16.911	154
7	7.930	167
8	16.783	215
9	14.171	158
Total		1364

Table 2. The sub-basins' land uses.

Land uses as % of each reach area	Reach								
	1	2	3	4	5	6	7	8	9
Cropland	77.69	79.65	89.46	77.39	91.29	81.77	64.72	47.06	30.24
CRP	4.96	5.25	1.46	1.41	0.75	1.04	6.75	7.70	10.45
Other	8.57	8.75	5.92	10.46	6.18	9.16	9.97	13.77	15.64
Forest	1.42	1.93	0.57	5.78	0.54	1.48	4.66	9.31	21.54
Pasture	7.36	4.42	2.60	4.97	1.25	6.55	13.90	22.15	22.13

The great majority of the agricultural land, over 80%, is in corn soybean rotations. The second rotation in terms of acreage, at less than 8% of the cropped area, is corn-corn-soybean. Corn followed by alfalfa accounts for a little over 5% of agricultural land, and soybean-soybean-corn for about 4%. Across the watersheds there are some differences, as illustrated by Table 3. The higher level of corn-alfalfa in the Southern portion of the watershed is another consequence of the presence of less fertile land and of the higher level of pasture-based livestock production.

The baseline levels of adoption of conservation tillage by sub-watershed are provided in Table 4. Overall adoption for the watershed is 46%. Historically, there is a certain degree of fluctuation in the levels of adoption of conservation tillage. The geographical heterogeneity of adoption in the watershed reflects historical differences and dissimilarities in land characteristics.

Table 3. Percent of agricultural area by rotation

Rotation	Reach								
	1	2	3	4	5	6	7	8	9
CC	0.00	2.19	3.35	2.12	1.42	0.68	3.58	2.67	1.36
SS	1.19	0.63	0.48	0.00	0.00	0.00	0.00	0.92	2.33
CS	81.55	80.44	82.77	86.42	87.75	87.15	78.05	67.82	54.22
CCS	5.40	7.43	11.31	6.32	7.77	5.66	6.68	10.35	13.21
SSC	6.30	6.67	0.96	3.68	2.01	3.54	2.08	7.28	6.08
CA	5.55	2.64	1.13	1.45	1.05	2.97	9.62	10.95	22.80

Notation: C- corn, S – soybeans, A – alfalfa.

The SWAT model is calibrated for the first 10 years of the simulation and validated using the second decade of data. Figure 3 shows the model's predictions together with the measured data for flow (monthly data), while Figure 4 has the same information for sediment loads. In the case of flow, the R^2 for the first decade is 0.6687, for the second it is 0.7608. For sediment, the R^2 for the first decade is 0.6687, for the second it is 0.7608.

Table 4 – Baseline levels of adoption of conservation tillage by watershed

Reach	% cropland in conservation tillage in the baseline (1997)
1	38.33
2	41.88
3	43.39
4	58.49
5	43.33
6	47.64
7	39.07
8	49.80
9	41.50

Figure 3 – SWAT calibration - flow

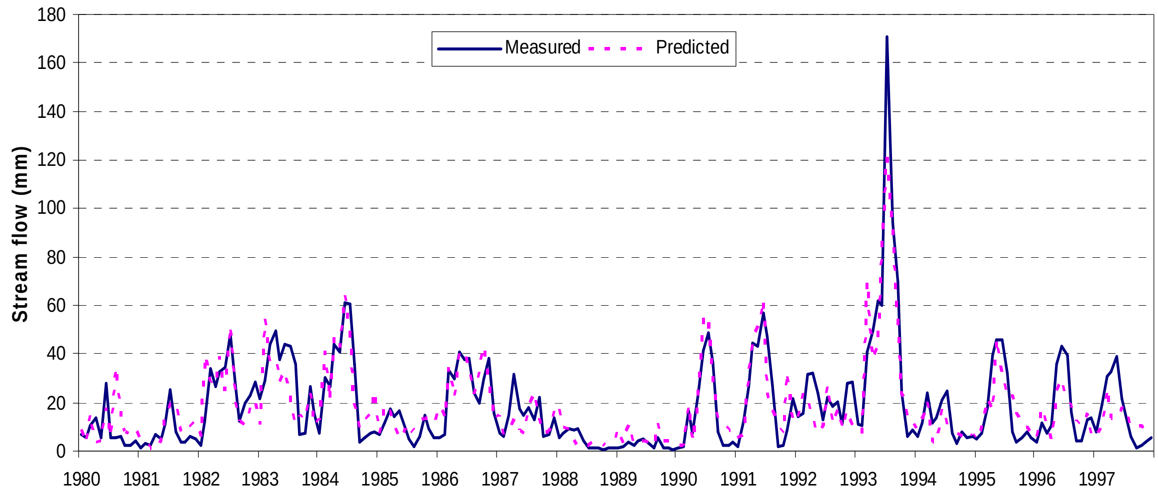


Figure 4 – SWAT calibration - sediment

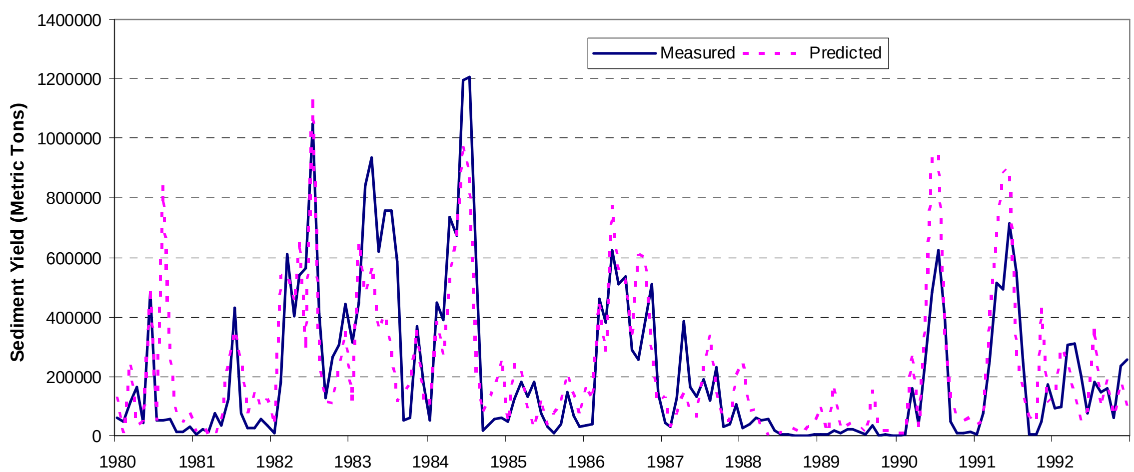


Table 5 reports what happens to the sediment loads at each watershed's outlet at the baseline (BL), and when the upper, middle, and lower part of the Des Moines watershed are completely converted into conservation tillage (UC, MC, and DC respectively). The last column illustrates what happens when the entire watershed is cropped using conservation tillage. The table shows that conservation tillage upstream greatly reduces the sediment loads at the outlets of reach 1, 2 and 3. A great part of that reduction is maintained downstream, at the outlet of sub-basin 4. However, there is almost no effect further down, at the outlet 8 and then 9. The reason is that two large dams, Saylorville and Red Rock, act as barriers and effectively create isolated watersheds. According to the Army Core of Engineers, "Both Saylorville and Red Rock are effective sediment traps, often capturing more 80 to 99% of the suspended solids load" (p.114, Lutz and Cummings).

Table 5 – Sediment loads in case of total conversion to CT by watershed (metric tons/year) and associated costs

Reach	BL	UC	MC	DC	AC
1	195,358	114,298	195,358	195,358	114,219
2	590,683	339,331	590,683	590,683	339,104
3	545,711	314,754	545,711	545,711	314,563
4	2,200,478	1,729,067	1,850,406	2,200,478	1,377,617
5	311,848	311,848	170,273	311,848	170,172
6	873,144	873,144	589,756	873,144	589,283
7	997,356	997,356	649,778	997,356	649,367
8	3,678,722	3,647,833	3,380,067	3,315,117	2,926,539
9	3,556,283	3,562,622	3,559,156	3,335,722	3,280,956
Cost	$ 0	$30,741,300	$41,736,360	$10,009,310	$82,487,000

Figure 5 illustrates the location of the reservoirs. These two large dams are also responsible for the ineffectiveness (in terms of sediment reduction at the overall outlet, 9) of converting the middle section of the watershed into conservation tillage. The watershed is effectively split into 3. This illustrates some of the problems an effective targeting policy has to solve. If a whole watershed is targeted and the only monitoring data is available at the overall outlet, the presence of dams will mask the upstream situation. More data would be needed to gather information on the whole watershed, and data is very costly to obtain, and relatively lengthy time series are necessary. Targeting very small watersheds will likely not solve the problem, since there is no monitoring information for most of them. The lack of monitoring information means there is no benchmark to compare the policy impacts with. Note that a relatively long time series would be needed to adequately assess long-term water quality.

Figure 5 – The reservoirs on the Des Moines River.

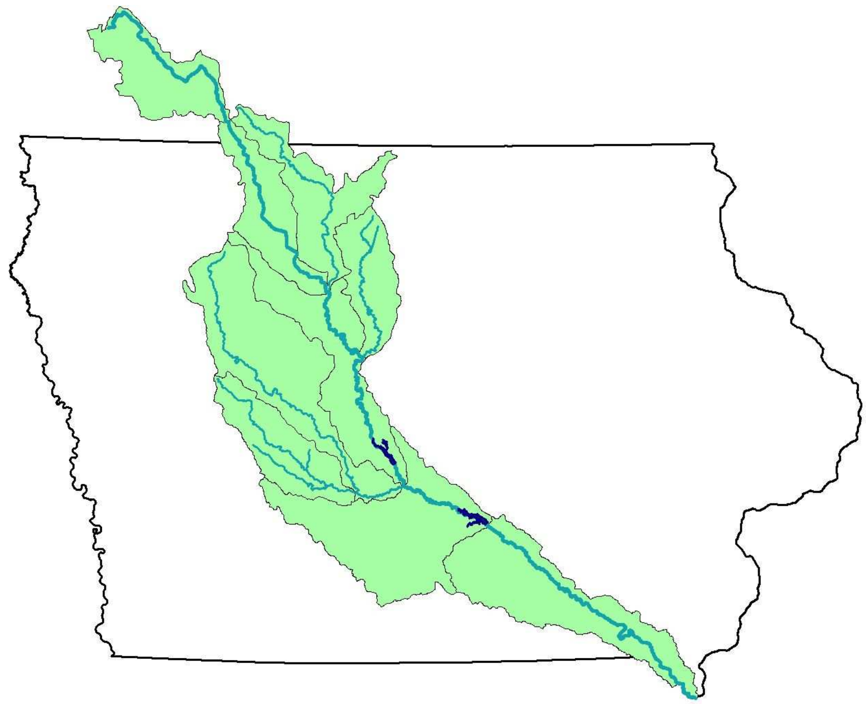


Table 5 shows that the most cost effective policy in terms of sediment reduction where the reduction is likely to be measured would be to implement conservation tillage downstream. Clearly, this policy would not be the most beneficial across the watershed. This raises the question of what the aim of the targeting is. If local water quality is the concern, the fact that this watershed is effectively split into three parts would require to target all of them. If the concern is regional water quality, or, say, the Gulf of Mexico, then focusing on the downstream portion of the watershed may be optimal. In the case of a water quality index comprising several pollutants, determining the optimal policy may be harder, since some pollutants have a local nature (phosphorus and partly sediment) while others are a concern for estuaries and sea water (nitrates). This also illustrates the usefulness of *ex ante* policy scenarios. Simulations are a faster and flexible tool in evaluating policy, and they can be very useful in complementing measured data.

The second scenario assumes funds are targeted to the sub-watersheds with the highest initial sediment loads. Since the watersheds have different size, we look at sediment per hectare. Such a focus would be closer in spirit to emission-based pollution control.

Table 6 shows that targeting the worst watersheds in terms highest loads per hectares would focus the policy to the upstream portion of the watershed. Again, this would imply that there is no water quality improvement at the main outlet. This clearly illustrates the ineffectiveness of using emissions as the basis for policy if the concern is the overall outlet.

Table 6 - Target on the basis of the highest load

Reach	Sediment Transport to Main Channel (metric tons/ha)
1	51.39
2	52.61
3	57.78
4	48.83
5	47.89
6	46.67
7	41.72
8	44.89
9	48.06

The final policy we consider is several levels of untargeted payments. Table 7 shows that even the lowest per acre payment, $10, would substantially increase the level of adoption of conservation tillage. It is interesting to note that the total costs of all these policies, even the $30/acre payment, are lower than most of the costs of complete conversion. This is likely due to the high marginal costs of converting some land that would have to be paid to achieve complete conversion. Table 8 shows that, in terms of the overall outlet, the marginal benefits of increasing payments from $20/acre to $30 would be insignificant, while the costs would be substantial. To assess the benefit per dollar of this policy, we calculated the sediment reduction at each outlet when the $10 payment is offered in that watershed only. So, to obtain the sediment reduction per dollar at reach 4, we simulate what would happen if the $10 payment were to be offered only to farmers in subbasin 4, and divide it to the cost of the subsidy ($10 $\times$ the number of acres enrolled). This measure is the relevant one if the basis for targeting is the benefit per dollar at each outlet.

Table 7 - Untargeted payments – CT adoption levels

Reach	BL	$10	$20	$30
1	38.33	46.45	59.12	65.48
2	41.88	59.14	73.76	76.83
3	43.39	58.97	72.08	79.76
4	58.49	66.14	77.03	85.00
5	43.33	52.19	73.91	79.33
6	47.64	59.20	68.97	77.94
7	39.07	58.68	65.17	77.34
8	49.80	58.07	66.43	73.65
9	41.50	62.17	69.86	77.46
CT adoption whole watershed	46.00	58.25	69.85	77.36
Total cost	$ 0	$4,023,120	$14,895,010	$26,380,430

Table 8 - Untargeted payments - sediment (metric tons/year)

	BL	$10	$20	$30
Reach	Sediment	Sediment	Sediment	Sediment
1	195,358	184,689	167,315	166,045
2	590,683	540,861	476,937	459,945
3	545,711	541,894	453,100	453,444
4	2,200,478	2,056,656	1,824,817	1,751,672
5	311,848	277,502	225,779	221,182
6	873,144	819,644	772,800	735,589
7	997,356	845,106	830,739	738,350
8	3,678,722	3,612,944	3,391,667	3,269,550
Total cost	$ 0	$4,023,120	$14,895,010	$26,380,430

Table 9 shows that the highest benefit per dollar would be obtained at the subbasin 9. The range of values is quite large, indicating that there is ample room for inefficiency. Because of the hydrology of this watershed, the highest benefit per dollar as measured at the outlet 9 would also be obtained by offering the $10 payment to farmers in subbasin 9.

Table 9 – Benefit per dollar in sediment reduction (metric tons/year) for a $10 subsidy

Reach	benefit per dollar
1	0.04
2	0.08
3	0.01
4	0.20
5	0.10
6	0.07
7	0.31
8	0.03
9	0.36

Conclusions

The empirical analysis of one pollutant in one watershed illustrates the complexity of watershed conservation policy and targeting. In general, it is evident that "targeting" is too vague to be an operative concept. It is necessary to identify at least two aspects: The first is the spatial dimension. The determination of where the objective of the policy is to be measured is crucial in determining which areas are the optimal ones to target. The second factor to be considered is the objective of targeting, be it an initial pollutant load, in absolute terms or per acre, the potential pollutant reduction or the potential benefit per dollar. Simultaneously targeting several pollutants would be even more complex, because, as we mentioned before, they might be relevant at different spatial scales, and the creation of an index would be necessary. However, conservation policies for agriculture have started to shift their focus from income support to being result-oriented. This is partly because the Clean Water Act was effective in curbing pollution from point sources but did not tackle non point sources, and agriculture in particular. Today, water quality problems are mostly due to non-point sources. The high number of impaired waters, and the elevated levels of impairment are pushing towards a more performance-based conservation policy. This will require adequate information on the magnitude of the problem, and reasonable assessments of the efficacy and costs of the various policies available.

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