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Cost-Effective Targeting for Reducing Soil Erosion in a Large Agricultural Watershed

Craig M. Smith, Jeffrey R. Williams, Amirpouyan Nejadhashemi, Sean A. Woznicki, and John C. Leatherman

Erosion of agricultural croplands is a significant contributor of sedimentation to reservoirs. Here, physiographic and economic models for a large agricultural watershed (2377 square miles with 27 subwatersheds) are integrated for the reduction of sedimentation of one Midwestern reservoir. Sediment reduction and the cost-effectiveness of three agricultural best management practices (no-till, filter strip, and permanent vegetation) implementation were considered under three modeling scenarios: random assignment; the globally most cost-effective approach; and a cost-effective targeting approach. This study demonstrates how physiographic and economic data can be harnessed to yield readily comprehendible cost-effective targeting maps. Cost-effective targeting may be preferable to watershed managers for its "user-friendliness" without too great a sacrifice of the globally most cost-efficient solution.

Key Words: cost-effective targeting, cropland best management practices, reservoir sedimentation, watershed model

JEL Classifications: Q15, Q25, Q53

Many water reservoirs built in the United States from 1930 to 1960 were designed to operate for 50 or more years before the designated uses of flood control, electricity generation, water supply, and recreation would be negatively affected by sediment accumulation. In some cases, sedimentation rates have greatly exceeded original estimates (Hargrove et al., 2010; Juracek, 2007).

Erosion of cropland is a major source of sediment accumulation in reservoirs (Devlin and Barnes, 2008). A study in northeast Kansas, the geographic focus of this research, found that unprotected croplands contributed the majority of sediment load in the Kansas River basin (Natural Resources Conservation Service [NRCS], 1992). Sedimentation reduces water storage capacity and negatively affects water supply, flood control capability, river barge navigation, viability of aquatic life, and the recreational value of reservoirs, because public funds for best management practices (BMPs) to reduce sedimentation are increasingly limited, and federal, state, and local governments are placing more emphasis on achieving economically efficient sediment reduction.

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There have been previous studies of the effects of BMPs in agricultural watersheds (Azzaino, Conrad, and Ferraro, 2002; Khanna et al., 2003; Rodriguez et al., 2009; Veith, Wolfe, and Heatwole, 2004; Yang et al., 2003; Yang, Khanna, and Farnsworth, 2005; Yuan, Dabney, and Bingner, 2002). Most of these analyses have been conducted on smaller watersheds than the one considered here and focused mainly on cropland retirement programs such as the Conservation Reserve Program (CRP). Alternatively, there have been other studies focusing on larger watersheds, but these have focused solely on nutrients such as nitrogen and phosphorus (e.g., Doering et al., 1999; Petrolia and Gowda, 2006). This study considers a large-scale agricultural watershed that is causing sediment to accumulate in a large lake and identifies the subwatershed priority areas for the implementation of three cropland BMPs.

The objective of this research was to determine the economically efficient combination of three land management practices that minimize sediment flow from the Kansas portion (2377 square miles) of a large agricultural watershed into Tuttle Creek Lake (TCL) reservoir while considering an annual budget constraint (Figure 1).[1] The TCL watershed is located in northeastern Kansas with approximately 41% of the total land area classified as cropland, the vast majority nonirrigated. The major crops for this area are corn, soybeans, wheat, and grain sorghum.

TCL has thus far lost approximately 77% of its designed sediment storage capacity and 42% of its total storage (multipurpose and sediment) capacity as a result of sediment accumulation (Kansas Water Office, 2010). State and federal officials are increasingly concerned over the prospects of losing the multipurpose benefits of the facility or dealing with the prohibitive costs of dredging (Smith et al., 2013).

Farm managers will adopt BMPs that are clearly profit-enhancing (e.g., increased use of no-tillage systems over that past two decades). However, managers may receive little to no benefit from the use of some BMPs, because the benefits accrue mostly to society at large and farm managers are not compensated through markets for these external benefits. Government agencies and private organizations seek to provide incentives for environmental protection where markets have failed to do so (Claassen, 2009). Therefore, this analysis considers the financial costs that would have to be expended (e.g., from a governing authority) to entice producers to adopt a given set of BMPs across the watershed to achieve the most sediment reduction given an annual budget constraint.

In large watersheds such as TCL, there is a need to focus BMP implementation efforts in smaller geographic regions that are critical (source of pollutants) within the watershed. It is frequently infeasible to implement BMPs throughout the entire watershed as a result of communication, coordination, transportation complexities, and budget constraints across different organizations, counties, and states. Rather, it may be more feasible to focus on one or several priority areas within a large watershed. This research was intended to help clarify options and costs associated with land management alternatives in Kansas and to illustrate a practical implementation approach for large watersheds.

Our research proceeds to compare three approaches to implementing BMPs to reduce sedimentation in a large agricultural watershed. We first show the inefficiency of a "random" implementation approach, similar to purely voluntary programs. We then show the most globally cost-efficient solution that mixes and matches BMPs throughout the watershed, recognizing the relative impracticality of large-scale watershed management in this fashion. Finally, we present a cost-effective targeting approach that permits the clustering of BMPs within the highest priority subwatersheds in a cost-effective way. Previous research (e.g., Arabi, Govindaraju, and Hantush, 2006; Hsieh and Yang, 2007; Rodriguez et al., 2009; Veith, Wolfe, and Heatwole, 2004) has focused on identifying theoretically optimal solutions. Here we develop and lay out a procedure to carry out a more practical targeting approach

[1] The entire TCL watershed is approximately 9600 square miles with 75% located in Nebraska and 25% in Kansas. However, the Nebraska portion of the watershed contributes only 30.5% of the total sediment load. This analysis proceeds under the assumption that investments and BMPs will only occur within Kansas.

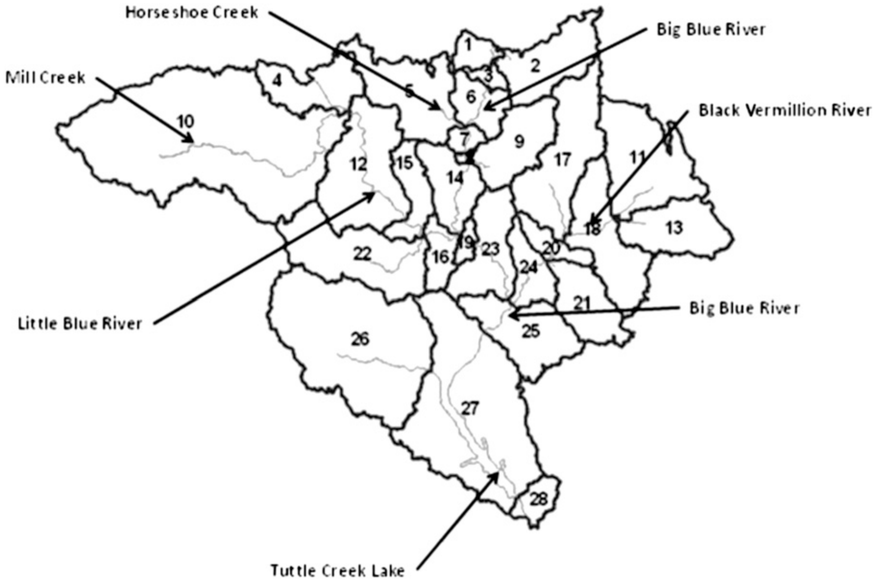

Figure 1. Major Watercourses and Subwatershed Delineation for the Kansas Portion of Tuttle Creek Lake (TCL) Watershed

not heretofore presented in the nutrient/sediment-watershed management literature.

Methods and Data

The Soil and Water Assessment Tool (SWAT) model was applied to the TCL watershed located in Kansas and Nebraska to predict the changes in sediment entering TCL in response to the adoption of the three cropland BMPs (no-till, filter strip, and permanent vegetation). The problem is considered from the perspective of a watershed manager who seeks to achieve maximum sedimentation reduction subject to an annual budget constraint. The amount of annual sediment reduction that can be achieved by implementing the three alternative BMPs on the most cost-effective crop acres under various budget constraints are determined.

The SWAT model predicts the sediment loading from each land parcel and BMP implemented (Gassman et al., 2007). The model estimates edge-of-field loading and also factors in a delivery ratio to predict the average annual amount of sediment entering the reservoir based on the application of BMPs. The load reduction and associated cost for each BMP implemented on each cropland parcel are used to estimate individual cost-effectiveness values (e.g., dollars per ton of sediment reduction). The individual costs of implementing a given BMP on a given cropland parcel are equal to the sum of the lost revenues and the additional costs incurred (both one-time and annual over a 15-year time horizon) for that parcel. Nonmarket benefits from BMP adoption may also be a consideration for some land managers and society, but these were not the focus of this analysis and therefore not considered. The annual aggregate cost of sediment reduction is represented by the sum of the annualized individual costs of the BMPs implemented.

Three types of BMP implementation approaches were modeled: "random" BMP management (random), globally most cost-effective, and a "cost-effective targeting" (targeted) approach. The random approach models the case in which BMPs are implemented randomly across the watershed. Although targeting is used in some conservation programs, the random approach is similar to conservation programs that rely largely on voluntary land manager interest and adoption (Nelson et al., 2011). The targeted approach implements BMPs on cropland that have the most attractive cost-effectiveness values (i.e., the lowest dollar cost per ton of sediment reduction). Implementation (in both approaches) continues until a budget constraint is reached. The annual total and marginal costs of BMP implementation are determined and cost-effective targeting maps are presented. Although sediment is the pollutant of concern in this study, it is noted that Total Maximum Daily Loads address nutrients as well. The approach presented here can just as easily be applied to the case of nutrients. Here, we simply note that BMPs that reduce sedimentation also will likely reduce nutrient runoff.[2]

Watershed Model Setup and Calibration

SWAT is a physically based, spatially distributed hydrological/water quality model developed by the USDA Agricultural Research Service (Gassman et al., 2007). SWAT is capable of simulating the impact of management practices and land use change on water, sediment, and nutrient yields in large watersheds over long time periods at a daily temporal resolution. A watershed in SWAT is delineated into several subwatersheds based on topography, which allows for a more accurate use of site-specific hydrology. Subwatersheds are further divided into hydrologic response units (HRUs), which are areas of unique land use, soil, topography, and management practice combinations (Giri, Nejadhashemi, and Woznicki, 2012). Sediment loads are calculated for each HRU, combined at the subbasin level and routed through the river network to the watershed outlet.

Based on previous research and reports (Langemeier and Nelson, 2006; O'Brien and Duncan, 2008a–d; Williams et al., 2009), data for cropping rotations and their typical field operations were determined. Corn, grain sorghum, soybeans, and wheat were produced under six different cropping rotations in the Kansas portion of the watershed. The proportions of each cropping rotation were estimated from USDA National Agricultural Statistics Service (NASS) data and were randomly applied in the baseline SWAT model consistent with the existing proportions of crops and rotations in the watershed.

Before application of the SWAT model, it must be calibrated and validated to ensure its performance is capable of emulating watershed behavior. Calibration is performed by adjusting model parameters and comparing simulation results with observed streamflow and sediment data. Model validation demonstrates that the model is capable of making accurate simulations following calibration without further adjustment of model parameters (Refsgaard, 1997). Model calibration (time period 1998–2000) and validation (time period 2001–2002) were performed on a daily time step to ensure that the SWAT model accurately predicted streamflow and sediment in the watershed river network. Goodness-of-fit between simulated and observed data were evaluated using the Nash-Sutcliffe Efficiency (E_{NS}) and coefficient of determination (R^2). An optimal E_{NS} value is one, although greater than 0.5 is considered satisfactory for calibration and validation purposes on a monthly time step (Moriasi et al., 2007).

The SWAT model was calibrated for flow and sediment. The SWAT model was set up based on 31 years (1978–2008) of climatological data from nine streamflow stations and 24 weather stations in the watershed. Observed streamflow discharge into TCL was obtained from the U.S. Army Corps of Engineering station, whereas total suspended solids concentration was obtained from Kansas Department of Health and Environment. Streamflow calibration resulted in an E_{NS} of 0.65 and R^2 of 0.68, whereas validation had an E_{NS} of 0.58 and R^2 of 0.59. Sediment calibration resulted in an E_{NS} of 0.57 and R^2 of 0.86, whereas validation had an E_{NS} of 0.55 and R^2 of 0.99. For more information regarding model calibration and validation, refer to Woznicki, Nejadhashemi, and Smith (2011). Additional information about the results of observed versus uncalibrated and calibrated SWAT model output as well as statistical analyses and model performance before and after calibration can be found in Nejadhashemi et al. (2011).

[2] Although nutrient reduction values were not a focus of this article, the physiographic and economic models used here did calculate and tabulate these results. Briefly, it was found that if the priority focus was on phosphorus, the results are not drastically different than if sediment was priority. This is because sediment and phosphorus tend to "move" together through erosion processes. If nitrogen were the priority, the results are a little more different.

Flows, sediment, and nutrient contributions from the Nebraska portion of the watershed were included in the SWAT model, but the BMP implementation scenarios are focused on the portion of the TCL watershed located in Kansas (Figure 1). This is because it is unlikely that Kansas would make expenditures for implementing BMPs in Nebraska or have significant influence in determining Nebraska BMP implementation priorities. The average annual amount of sediment coming from Nebraska streams and rivers into the Kansas portion of the TCL watershed is 817,394 tons, which is 30.5% of the total sediment loading in TCL.[3]

The entire TCL watershed area is 6,144,000 acres with 25% of the watershed area located in Kansas. There are 2752 HRUs, which are unique combinations of land use and soil that occur within the Kansas portion. Within these 2752 HRUs, 1858 were categorized as cropland (1015 mi^2). The average size of the 1858 HRUs was 350 acres with the smallest being five acres and the largest being approximately 8175 acres. The average annual sediment flow into TCL is 1,861,030 tons per year from the 1858 cropland HRUs in the Kansas portion of the watershed.

Economic Model

The SWAT watershed model described in the previous section generated sediment loading for each of the 1858 HRUs. Using these data, the economic model determines the optimal combination of BMPs to implement using the procedure described in the following equation.

$$(1) \qquad Max \sum_{i=1}^{I} \sum_{j=1}^{J} S_{ij} \times A_{ij}$$

subject to:

$$\sum_{i=1}^{I} \sum_{j=1}^{J} C_{ij} \times A_{ij} \leq B, \text{ and } S_{ij} > 0$$

where:

S_{ij} = annual sediment reduction in tons/acre with BMP_i in HRU_j

A_{ij} = acres of BMP_i in HRU_j

C_{ij} = annualized cost in \$/acre for BMP_i in HRU_j

B = annual budget constraint

i = one to three BMPs

j = one to 1858 HRUs

It is assumed that a governing authority or watershed manager has set a goal of maximizing the amount of sediment reduction, S, while operating under an annual budget of B dollars per year. Each HRU can generate up to S_i units of sediment reduction at a total annualized cost of C_i. The ratios of C_i/S_i are the costs of sediment reduction in dollars per ton are calculated for each BMP in each HRU. Annualized costs are allowed to vary across HRUs but are constant for a given HRU. Each HRU can potentially adopt one of i BMPs, i = one to three. The three BMPs are filter strips, no-till, and the conversion of cropland to permanent vegetation. This is based on local experts' opinions and observations. Each BMP implemented must result in a positive amount of sediment reduction. The model selects BMPs and HRUs to apply them in a cost-efficient order by searching from lowest to highest for the value of C_{ij}/S_{ij} associated with each BMP in each HRU. The BMP implementation process occurs by iterating through all potential HRU-BMP implementation projects from most cost-effective to least cost-effective projects until: 1) no additional sediment reduction exists; or 2) no other BMPs can be implemented without violating the budget constraint. The annual budget constraint was varied from \$50,000 to \$450,000 in increments of \$100,000. These values were consistent with the estimated minimum and maximum funding amounts that could be available through the state of Kansas for purposes of reducing sedimentation in the TCL watershed (Smith et al., 2013).

[3] The benefits of TCL accrue primarily to the state of Kansas and not Nebraska. There may in fact be value (e.g., increased cost-efficiencies) related to an interstate cooperative approach to address these issues, but this study only considers the case of BMPs being implemented in Kansas.

Pre-Existing Best Management Practices

Although no BMPs (other than permanent vegetation, i.e., the "Grassland" land use category) were applied in the baseline SWAT model, it was calibrated to actual flow and known sedimentation in the watershed. Because BMPs do exist in the watershed, the calibrated loading values incorporate the fact that there are BMPs in place. However, the types and locations of the BMPs in 1858 HRUs are not known. Therefore, the challenge is to determine where BMPs exist so as not to implement additional BMPs to those locations in the modeling process. Determining these amounts and location with any precision and incorporating this into the SWAT model would have been prohibitively difficult and expensive and was considered beyond the scope of this research. To account for pre-existing BMPs, simplifying assumptions were used.

Specifically, Smith, Peterson, and Leatherman (2007) found that approximately 20% of Kansas farms have already adopted filter strips and 30% of farms have already adopted no-till. To account for this in the analysis, it was first assumed that 25% of the HRUs had already adopted BMPs. This 25% was removed from the choice set before initiating the BMP implementation procedure described in equation (1). The problem was determining which 465 of the 1858 HRUs should be eliminated from further consideration.

The approach used assumed that BMPs were already adopted in HRUs that have the greatest potential for visible soil erosion. In cases where soil erosion is severe, farmers often install BMPs in an effort to save significant losses of top soil and/or prevent gullies from forming in their fields. Therefore, 25% of HRUs that exhibit the greatest amount of baseline soil erosion per acre were eliminated before the cost-effective implementation process proceeded in the manner as previously described. For sensitivity purposes, the cost-effective implementation process also was run with 15% of the most erosive HRUs being eliminated.

To ensure that the final results were not sensitive to a particular set of random draws, all simulations were repeated 3000 times with a "new" set of eligible HRUs picked each time. The model was tested and it was found that 3000 iterations were sufficient to ensure that the mean performance measures computed across the 3000 iterations were a stable statistic. This was done in both the cost-effective and random BMP implementation schemes.

Best Management Practices Data

There are two main types of strategies for reducing the amount of sediment that enters a reservoir: in-field and in-stream strategies. Three in-field strategies (filter strips [FS], no-till [NT], and permanent vegetation [PV]) were considered. Unprotected croplands likely contribute the majority of sediment loads in the TCL watershed (NRCS, 1992). On a per-acre basis, cultivated cropland contributes three to five times the amount of soil erosion as pastureland and CRP land in the TCL watershed (NRCS, 1992). Although streambank erosion also contributes sediment to TCL, the SWAT model does not have the ability to accurately analyze streambank erosion unless site-specific data are available, which was not the case for the TCL watershed.

The economic model requires an estimate of BMP costs. The annualized costs (2012 dollars) of a filter strip were estimated with the KSU Vegetative Buffer Decision-Making Tool (Smith and Williams, 2010). Managers that have not already adopted BMPs will incur lost income from removing land from production as well as labor and material costs for establishment of a filter strip with a design life of 15 years. A discount rate of 4.625% was used for annualizing this cost (NRCS, 2009). A FS is a grass strip at the edge of each cropland HRU. The edge of each HRU does not necessarily border surface water; hence, each FS does not necessarily border surface water. As a result of the multitude of field shapes and sizes throughout the watershed along with varying types of filter strips, we made the simplifying assumption that each acre of FS treats runoff from 25 acres of cropland. The estimated annualized costs ranged from \$7.66 to \$11.34 per cropland acre treated or \$191.50 to \$283.50 per acre of actual FS installed depending on in which HRU they are installed.

No-till is a form of conservation tillage in which herbicides are used in place of tillage for weed control and seedbed preparation. Costs (2012 dollars) for converting to a NT management system are based on NRCS Environmental Quality Incentives Program incentive payments that range from $13.00/acre to $20.00/acre paid out over a three-year period (Kansas Department of Health and Environment [KDHE], 2009). Annualizing this income stream over a 15-year time period gives a range of approximately $3.25/acre to $5.00/acre per year. Practitioners involved in BMP implementation verified this was a good estimate for farmers that have not already adopted NT, because NT is seen by some farmers as increasing costs.

Establishment of permanent vegetation has the potential to significantly reduce soil erosion. The Kansas State University (KSU) Vegetative Buffer Decision-Making Tool (Smith and Williams, 2010) was used to calculate the annualized costs (2012 dollars) of converting cropland to permanent native grass vegetation over a 15-year time horizon with a discount rate of 4.625% (NRCS, 2009). The values used represent the annualized cost of establishing and maintaining the vegetation along with the lost value of production range from $162.10/acre to $216.30/acre depending on the HRUs in which they are applied.

Cost-Effective Targeting for Sediment Reduction

Considering the underlying economic costs of BMPs and the physiographic characteristics of each HRU, cost-effective targeting can be prescribed. Here, the budget constraint and sediment reduction goals are both set high enough so that all possible BMPs that reduce sediment are implemented in the HRUs after first eliminating the 25% (or 15%) most erosive HRUs from consideration. These results provide information on the costs and pollution reduction achieved by implementing a given BMP on a HRU (referred to here as a project). Ideally, implementation priority can be assigned to each of the 1858 HRU projects. Practically, this can be difficult to accomplish. Watershed managers prefer a more practical approach.

Each HRU is located in one of the 27 subwatersheds. Using the cost, sediment reduction, and the acreage being treated by the BMP in each HRU within a subwatershed, averages were calculated for each subwatershed. The average costs were categorized into four groups as a function of the BMP cost per acre and cost-effective targeting maps were created. The maps are based on an unlimited budget constraint. Maps under a limiting budget constraint could contain some anomalies. For example, a subwatershed may have just one BMP that should be implemented in one HRU, which would be a low priority from a subwatershed average perspective compared with all subwatersheds when other subwatersheds may have numerous BMP projects that should be implemented. However, the resulting map with a limited budget constraint might show that the subwatershed with one BMP is high priority if that one BMP is very cost-effective. In reality, this subwatershed would not have a high priority to implement BMPs, but the map would indicate otherwise.

Results

The total costs for random and targeted implementation of BMPs with 25% of the most erosive HRUs eliminated are graphed in Figure 2. The random implementation of BMPs is considerably more expensive than the targeted approach and less sediment reduction is achieved per dollar spent. The difference in cost between the methods where 25% of the most erosive HRUs are eliminated and 15% of the most erosive HRU projects initially eliminated is shown (Figure 2). Additional detail concerning the effects of the number of pre-existing BMPs is provided in Table 1. At an annual budget constraint of $450,000, the average costs are $0.69/ton higher for sediment reduction when 25% versus 15% most erosive HRUs are removed before implementation of BMPs. There is a 44.1% increase in sediment reduction achieved if we assume 15% (instead of 25%) of the most erosive HRUs are removed.

Table 1 reports the results as the annual budget constraint is varied from $50,000 to $450,000 in increments of $100,000. The second column reports the average sediment reduction costs per unit. As the budget increases

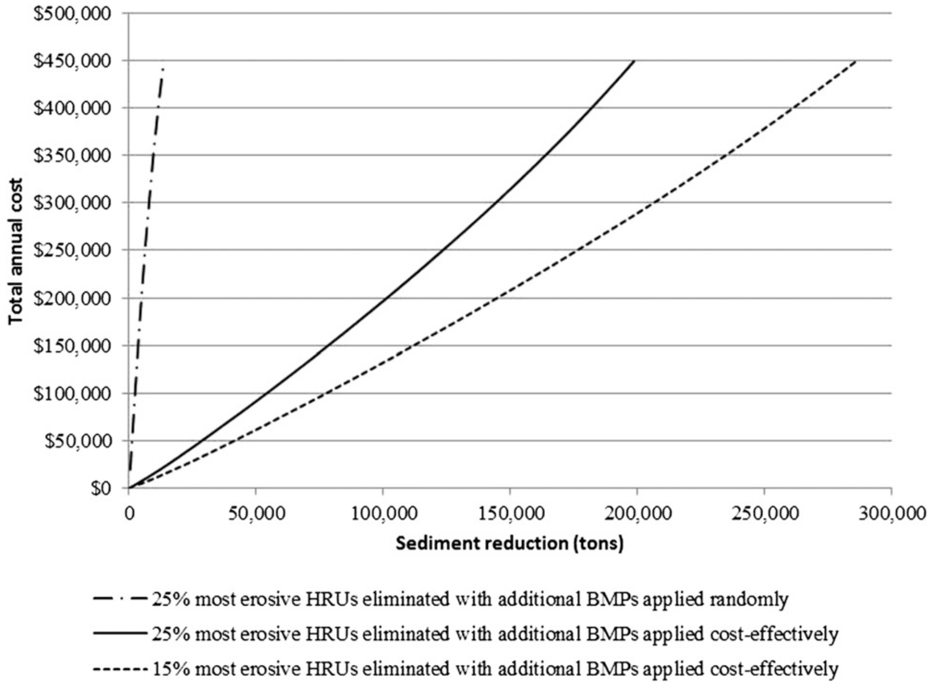

Figure 2. Total Cost Curves for Sediment Reduction with a Maximum $450,000 Budget

from $50,000 to $450,000, with the random approach, the average costs ($/ton) for reducing sediment increases from $14.44 to $32.79/ton. When BMPs are applied in a cost-effective manner assuming the 25% most erosive HRUs have been previously treated, the average costs increase from $1.75 to $2.26/ton. The costs are moderately smaller when we assume that 15% of the most erosive HRU projects are already in place ($1.21 to $1.57/ton). With a $450,000 annual budget, the predominant projects are FS projects and are followed by NT projects.

Commodity prices and overall farm profitability have trended upward since 2007 and increased net returns are being capitalized into land values and rents. Thus, opportunity costs associated with converting cropland to filter strips and permanent vegetation are increasing. Increasing fuel prices also result in higher establishment costs for FS and PV BMPs while NT becomes relatively more cost-effective. Sensitivity analyses, not provided in tabular form, show that as net incomes and fuel cost increase, more NT projects are selected as opposed to FS and PV.

Figure 3 shows the proportion of the total BMP projects that are cost-effectively implemented in each subwatershed by type of BMP for sediment reduction after first eliminating the 25% most erosive HRUs from consideration. FS are the most frequently applied and subwatershed 10 has the greatest number of projects mainly as a result of its relatively large area (i.e., consisting of 259,609 acres occupying 17.2% of the total land area). Fifty-three percent of all projects are applied in seven of the 27 subwatersheds. These are subwatersheds 10, 22, 14, 12, 27, 15, and two listed in order by number of projects ranging from 34 in subwatershed 10–14 in subwatershed two. Fifty-five percent of the FS projects are applied in subwatersheds 22, 10, 14, two, 12, 27, and 11 with a range in the number of projects from 20 in subwatersheds 22 and nine in subwatershed 11. Fifty-two percent of NT projects are implemented in subwatersheds 10, 12, 14, 22, 26, 25, and five with a range from 15 in subwatershed 10 to seven in subwatershed five.

In further examining the physical (e.g., soil types, baseline sediment loading, etc.) and cost characteristics of each subwatershed and

Table 1. Results with Alternative Methods of Selecting Hydrologic Response Units (HRUs) Already Having Adopted Best Management Practices (BMPs) and Implementation Approaches

Scenario	Average Sediment Reduction Cost for All Land Treated by BMPs (/ton)	Total No. of BMP Projects	No. of Filter Strip Projects	No. of No-till Projects	No. of Permanent Vegetation Projects	Total Area of Land Treated by BMPs (acres)	Total Amount of Sediment Reduction (tons)
25% Most erosive HRUs eliminated with additional BMPs applied randomly							
$50,000	$14.44	20	7	10	3	3,690	3,425
$150,000	$21.85	31	11	14	6	6,938	6,836
$250,000	$27.16	37	13	16	8	8,916	9,138
$350,000	$30.14	43	15	18	10	10,938	11,538
$450,000	$32.79	49	17	20	12	12,729	13,640
25% Most erosive HRUs eliminated with additional BMPs applied cost-effectively							
$50,000	$1.75	59	16	43	0	9,484	28,215
$150,000	$1.91	107	44	63	0	23,172	78,022
$250,000	$2.03	168	82	87	0	36,957	122,876
$350,000	$2.14	236	123	113	0	50,915	163,063
$450,000	$2.26	318	175	142	0	63,790	198,542
15% Most erosive HRUs eliminated with additional BMPs applied cost-effectively							
$50,000	$1.21	58	22	36	0	11,121	40,468
$150,000	$1.34	107	58	49	0	23,941	111,645
$250,000	$1.42	152	88	64	0	36,326	175,984
$350,000	$1.49	215	125	90	0	49,395	234,233
$450,000	$1.57	273	160	113	0	62,228	286,020

comparing these with the number of BMPs implemented, we find there is no consistent correlation present other than subwatershed size. That is, larger subwatersheds tend to have a larger percentage of BMP projects placed on them. This result is intuitive and expected. Outside of subwatershed size, neither physical characteristics nor costs by themselves are consistently sufficient for identifying cost-effective targeting areas. Rather, both physical and cost factors need to be considered in tandem when identifying areas for BMP implementation.

Cost-Effective Targeting Results

Implementing BMPs in areas that exhibit the most potential for erosion is a good first step in efficient targeting. This approach is commonly used in watershed management projects (Giri, Nejadhashemi, and Woznicki, 2012). However, this may not be the most cost-effective technique because costs are not considered. Cost-effective targeting includes the economics of sediment reduction and focuses BMPs in areas of the watershed that deliver the greatest benefits (sediment reduction) for the cost.

Although our cost-effective implementation approach described in equation (1) is the optimal approach under ideal conditions to implement BMPs throughout the watershed, it may not be feasible from a practical standpoint. For example, with a cost-effective modeling approach such as the one used here, a FS may be identified as having the highest priority for an HRU in one part of the watershed, whereas the next highest priority BMP may be NT implemented in a portion of the watershed far away from the first. Watershed restoration and protection stakeholder groups are likely to focus on implementing projects in smaller geographic regions in which they work and reside rather than scattering projects across a large watershed for practical reasons. Jumping from one part of the watershed to another may not be

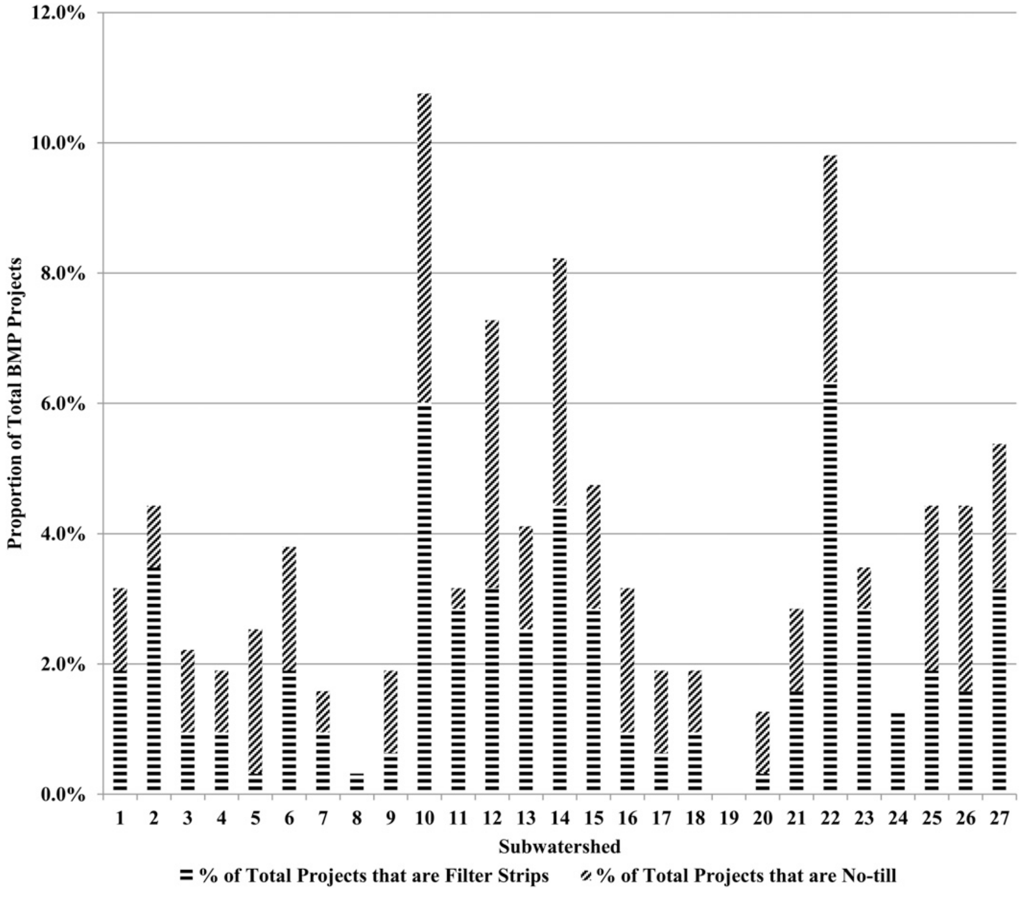

Figure 3. Distribution of the Two Sediment Reduction Best Management Practices (BMPs) across Subwatersheds with the Most Erosive 25% of Hydrologic Response Units (HRUs) Removed Given a $450,000 Budget (note permanent vegetation was never selected)

an issue in smaller watersheds, but in larger watersheds like TCL, this may create additional difficulties in administering the implementation of BMPs (e.g., dealing with many different organizations in a variety of counties and regions poses communication, coordination, and transportation complexities). Rather, it may be more feasible to focus on one or two areas of the TCL watershed. The data in Table 2 and the maps presented in Figures 4–6 discussed next are useful for this approach. They offer information on where to focus watershed BMP implementation to get the most cost-effective sediment reduction "on average."

"User-friendly" prescriptions for cost-effective targeting can be derived using the approach discussed in previous sections. The cost-effective targeting approach described here answers the following question. On average, where in the watershed will a given BMP (i.e., FS, NT, or PV) provide the most cost-effective sediment reduction? Two sets of results are generated. The first assumes that 15% of the most erosive HRUs have already adopted BMPs and are removed from the choice set and the second is with 25% most erosive HRUs eliminated before additional BMPs are applied in cost-effective order following equation (1). The only differences from the initial analysis, including all three BMPs with a budget constraint, are that the budget constraint is set infinitely high and only one BMP is considered at a time. For example, NT would be applied to every HRU in the watershed and the costs of doing so would be recorded. For each subwatershed, the total amount of sediment reduction achieved and the

Table 2. Average Sediment Reduction Costs for each Best Management Practice (BMP)

	15% Most Erosive HRUs Removed			25% Most Erosive HRUs Removed		
Subwatershed	Sediment Reduction with FS ($/ton)	Sediment Reduction with NT ($/ton)	Sediment Reduction with PV ($/ton)	Sediment Reduction with FS ($/ton)	Sediment Reduction with NT ($/ton)	Sediment Reduction with PV ($/ton)
1	$13.19	$19.01	$193.20	$14.01	$20.66	$205.46
2	$6.88	$8.27	$101.01	$7.31	$9.30	$107.68
3	$7.27	$8.49	$107.08	$8.09	$9.42	$119.04
4	$7.13	$8.54	$98.98	$7.88	$9.49	$109.35
5	$7.71	$9.74	$112.09	$8.30	$9.85	$120.34
6	$6.83	$8.03	$100.28	$7.28	$8.30	$106.84
7	$7.80	$8.96	$113.91	$8.18	$9.32	$119.53
8	$33.37	$39.52	$477.90	$33.37	$39.52	$477.90
9	$5.67	$6.57	$83.88	$7.51	$8.98	$111.40
10	$5.29	$6.40	$75.37	$6.55	$7.62	$92.87
11	$4.55	$4.87	$67.68	$7.25	$7.55	$108.13
12	$5.19	$5.79	$72.92	$6.54	$7.03	$91.77
13	$5.42	$5.36	$79.22	$5.87	$5.76	$85.75
14	$6.28	$7.10	$91.96	$7.29	$8.33	$106.55
15	$4.66	$5.36	$67.85	$6.21	$6.75	$90.56
16	$6.19	$7.08	$90.28	$7.30	$8.41	$106.59
17	$4.07	$4.05	$59.68	$6.09	$5.99	$89.53
18	$4.33	$4.96	$63.17	$8.15	$8.75	$119.58
19	$8.95	$10.35	$130.38	$8.95	$10.35	$130.38
20	$4.34	$4.68	$62.53	$7.66	$7.86	$110.85
21	$4.07	$4.96	$61.38	$7.44	$8.40	$110.48
22	$5.69	$6.45	$80.08	$6.24	$7.09	$87.58
23	$6.79	$7.67	$100.33	$7.49	$8.40	$110.85
24	$6.25	$6.72	$90.14	$12.20	$12.38	$176.36
25	$6.88	$7.25	$102.52	$9.46	$9.94	$141.47
26	$5.30	$6.05	$74.58	$6.08	$6.97	$86.89
27	$5.88	$5.48	$84.69	$7.51	$7.02	$107.44

Note: HRUs, hydrologic response units; FS, filter strip; NT, no-till; PV, permanent vegetation.

total BMP cost is calculated. Dividing the total cost by the total amount of sediment reduction gives the average sediment reduction costs for each subwatershed. This was performed separately for FS and PV as well.

The results provide information on the average sediment reduction costs resulting from implementing a given BMP in a subwatershed. Using the cost, pollution reduction, and the acreage being treated by the BMP, average sediment reduction costs are reported for each subwatershed in Table 2. For example, a $13.19/ton sediment average reduction cost for subwatershed one reported in the first cell of Table 2 indicates that for an average acre in this subwatershed, sediment can be reduced for $13.19/ton using FS. This is more cost-effective than implementing FS in subwatershed eight, which exhibits $33.37/ton sediment reduction costs, but not nearly as cost-effective as investing in FS in subwatershed 17, which has $4.07/ton sediment reduction costs. The sediment reduction costs reported in Table 2 and the associated maps in Figures 4–6 are averages and some are significantly higher than the results in Table 1. This is because BMPs are implemented on each HRU in the TCL watershed, even the least cost-effective HRUs.

The data in Table 2 may be better represented in a map form. Dividing each of the scenario's results into "quartiles," cost-effective targeting

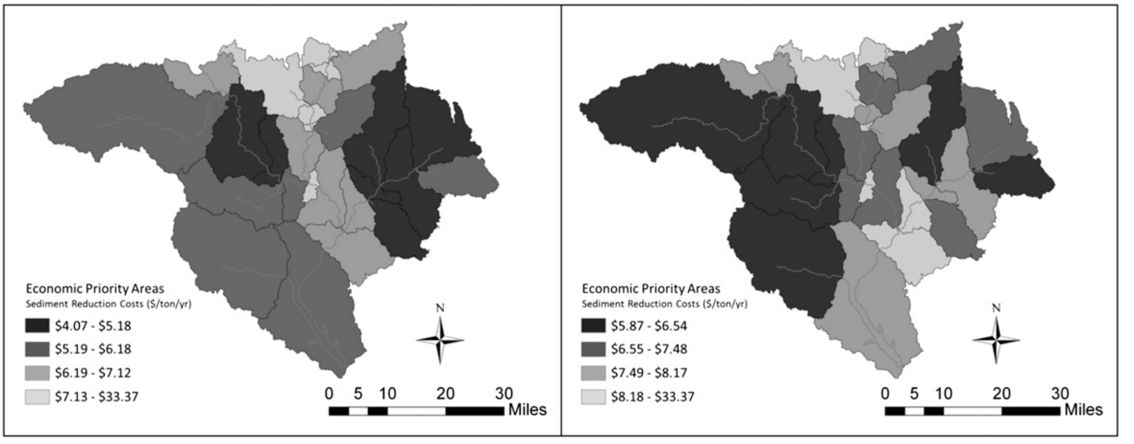

Figure 4. Average Sediment Reduction Costs with Filter Strips with 15% (left) and 25% of the Most Erosive Hydrologic Response Units (HRUs) Removed (right)

maps are created.[4] In other words, sorting the average costs for a given scenario in ascending order and then dividing the data into four groups of seven subwatersheds each is a useful way of presenting the results cartographically. Figures 4–6 show the priority areas for implementing FS, NT, and PV, respectively, using these cost-effective quartiles. Figure 4 shows the priority areas in the TCL watershed for reducing sediment with the elimination of 15% and 25% of the most highly erodible HRUs with FS, respectively. According to the figure with 25% of the most highly erodible HRUs removed (Figure 4 map on the right side), the most cost-effective sediment-reducing locations for placing additional BMPs in the form of FS projects are in the western and northeastern parts of the watershed, including subwatersheds 10, 12, 13, 15, 17, 22, and 26. In these subwatersheds, sediment can be reduced with FS projects for $5.87 to $6.54/ton annually. Only three of these subwatersheds, 12, 15 and 17, are in the highest cost-effective quartile when 15% HRUs are eliminated (Figure 4 map on left side). This demonstrates the importance of having knowledge of pre-existing BMPs in the watershed. It should be noted that the subwatershed surrounding TCL, number 27, does not have the highest cost-effective priority. The least cost-effective places for sediment reducing FS projects to be implemented are subwatersheds one, five, eight, and 19.

Figure 5 displays the average annual sediment reduction costs when NT is applied in each of the subwatersheds. The most cost-effective sediment reducing locations for placing NT is the same as those for FS with the exception that subwatershed 10 is not included, but the subwatershed directly surrounding TCL, number 27, is included (Figure 5 map on the right side). Sediment can be reduced in subwatersheds 12, 13, 15, 17, 22, 26, and 27 for $5.76 to $7.08/ton annually. Only three of these subwatersheds, 13, 15, and 17, are in the highest cost-effective quartile when 15% most erosive HRUs are eliminated (Figure 5 map on the left side). The least cost-effective places for sediment reduction using NT are the same as for FS. These are subwatersheds one, five, eight, and 19.

Although the model never selected PV as a BMP in the analysis, Figure 6 displays the annual average sediment reduction costs with PV. The subwatersheds selected as high priority if PV projects are used are the same as those for FS. The cost, however, is substantially higher.

Assuming 25% (instead of 15%) of the most erosive HRUs in TCL have already adopted

[4] The word "quartiles" is in quotes because the number 27 is not perfectly divisible by four. So, the quartiles used here contain seven, seven, seven, and six subwatersheds in the high, medium-high, medium-low, and low categories, respectively.

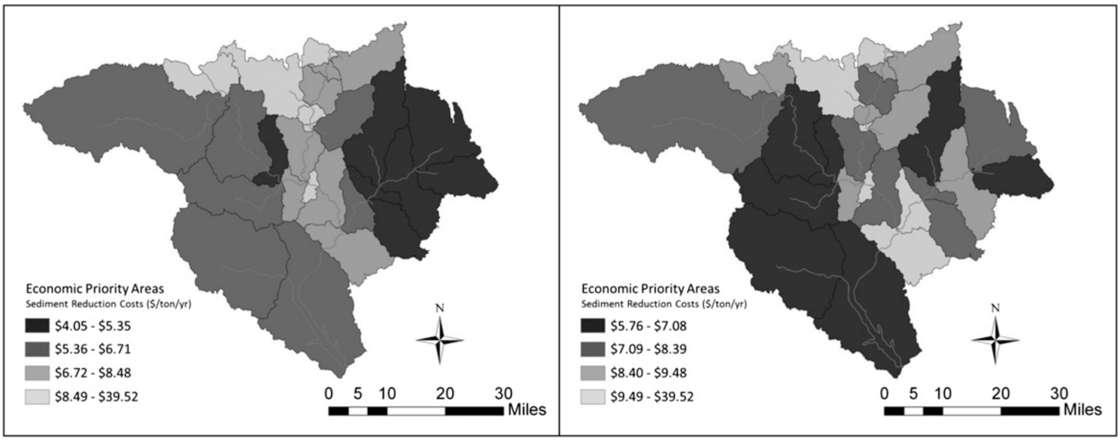

Figure 5. Average Sediment Reduction Costs with No-Till with 15% (left) and 25% of the Most Erosive Hydrologic Response Units (HRUs) Removed (right)

BMPs results in fairly significant changes in terms of cost-effective targeting prescriptions. By comparing the two maps in Figure 4, we can see that several of the economic priority areas for filter strip move from the eastern portion of TCL watershed to the western portion. Only subwatersheds 12, 15, and 17 remain as a high priority in each case. For NT (Figure 5), we see a similar shift of economic priority from the eastern part of TCL. Subwatersheds 13, 15, and 17 are the only two that remain classified as "high" in each case. Another significant finding is that subwatershed 24 moved from being "medium-high" priority to "low" priority. The reason for this is that there are a few highly erosive HRUs in this subwatershed. If these few HRUs have already adopted BMPs, there is not much relative value in funding additional BMPs such as NT in these two subwatersheds.

The east-to-west change area of emphasis for PV is again evident in Figure 6. Again, subwatersheds 12, 15, and 17 are the only subwatersheds that remain a "high" economic priority in each case.

In summary, under the assumption that 25% most erosive HRUs have already been treated, implementation of additional BMPs should be in the western and northeastern subwatersheds

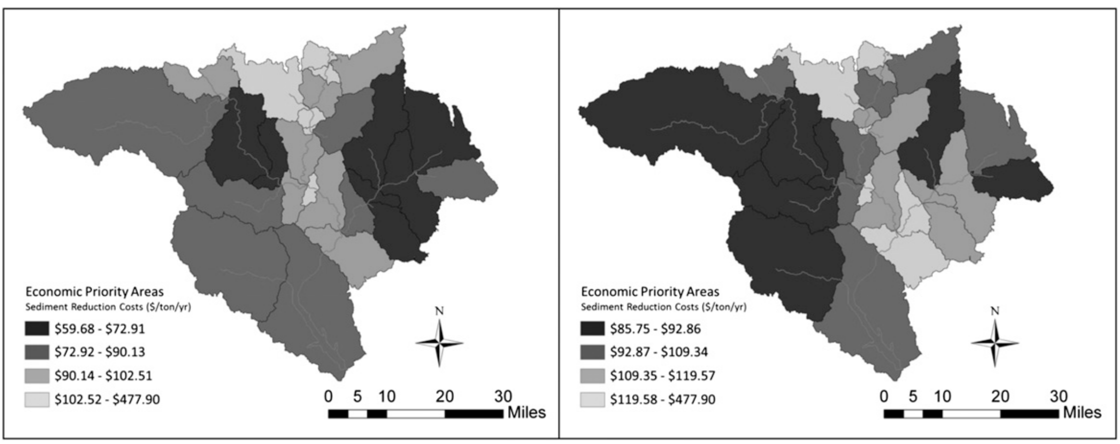

Figure 6. Average Sediment Reduction Costs with Permanent Vegetation with 15% (left) and 25% of the Most Erosive Hydrologic Response Units (HRUs) Removed (right)

of the TCL watershed. If there have been less BMPs implemented or if previously existing BMPs have been implemented randomly, then additional BMPs should generally be implemented first in the eastern subwatersheds of the TCL watershed and then in the western subwatersheds. This highlights the importance of accurately tracking where BMPs have been implemented.

Cost-Efficiencies of Using Targeting Maps to Guide Implementation

Although the maps provide a "user-friendly" alternative approach for targeting, how does the cost-efficiency of this approach compare with either the random or highly targeted approaches presented earlier (Table 1)? A few examples are presented next.

Suppose a watershed group had \$50,000 to spend in their watershed. If one were to spend \$50,000 for filter strips using the information from the map, they would need to focus only on subwatershed 13. Sediment would be reduced at an average cost of \$5.87 per ton resulting in 8515 tons of annual sediment reduction.[5] This compares to an average cost of \$14.44 per ton and \$1.75 per ton for random and cost-effective implementation scenarios, respectively (Table 1).

If, on the other hand, a watershed group had \$450,000 to spend in their watershed, the results are as follows. Spending \$450,000 for filter strips using the information from the map, they would need to focus on subwatersheds 13 and 26. Sediment would be reduced at an average cost of \$5.99 per ton resulting in 75,127 tons of annual sediment reduction. [6] This compares to an average cost of \$32.79 per ton and \$2.26 per ton for random and cost-effective implementation scenarios, respectively (Table 1). For comparison purposes, sediment reduction costs in existing literature range from \$7.00 to \$256.00 per ton (Khanna et al., 2003; Yang et al., 2003; Yang, Khanna, and Farnsworth, 2005; Yuan, Dabney, and Bingner, 2002). Costs are highly variable across BMPs, regions, and time.

Using the maps to guide implementation would result in an average sediment reduction cost somewhere in between the random and cost-effective implementation scenarios, but much closer to the "globally most cost-efficient" scenario. Thus, using this more practical approach gives up a little in terms of cost-efficiency but makes great strides in terms of ease of implementation/practicality.

Characteristics of Cost-Effective Targeted Areas

As described previously, the targeted areas take into account both the physiographic and the economic characteristics of the HRU and the BMP. In general, the three primary physiographic factors affecting sediment contribution to TCL for a given HRU and BMP are land slope, hydrologic soil group, and delivery ratio.

Subwatersheds 15 and 17 are always in the highest priority quartile, whereas subwatershed 19 is always in the lowest priority quartile (Figures 4–6). Based on the underlying physiographic data, subwatershed 19 actually has a much greater percentage of land with slopes greater than 6% than either subwatershed 15 or 17.[7] However, only 49.6% of the land in subwatershed 19 is classified as being in hydrologic soil group D, which represents areas with higher risks for runoff-generating potential. This compares to 14.7% and 92.3% of the land in subwatersheds 15 and 17. In terms of delivery ratios for sediment, subwatersheds 15 and 19 have the highest at 1.00, whereas subwatershed 17 has a sediment delivery ratio of 0.72.[8] From this, it is not clear which physiographic characteristic is the primary cause of reservoir sedimentation. Certainly, the physiographic characteristics that make

[5] The values increase slightly for NT and increase dramatically for permanent vegetation (results available from the authors on request).

[6] The values increase slightly for NT and increase dramatically for permanent vegetation (results available from the authors on request).

[7] Complete physiographical data are available on request from the authors.

[8] Delivery ratio for a subwatershed is defined as the ratio of pollutant load from the subwatershed that reaches the watershed outlet (i.e., TCL) to the total pollutant load generated by the subwatershed (Woznicki and Nejadhashemi, 2013).

Table 3. Subwatershed Priority Rankings Resulting from Consideration of Alternative Physiographic and Economic Factors

Subwatershed Priority Ranking for FS Implementation (1 = highest; 27 = lowest)	Subwatershed Rank Based on SWAT Model Results Alone	Subwatershed Rank Based on Economic Cost Results Alone	Subwatershed Rank Based on Integrated Economic/Physiographic Model Results
1	22	10	13
2	17	26	26
3	12	25	17
4	13	27	15
5	11	21	22
6	15	12	12
7	26	22	10
8	16	15	11
9	7	4	6
10	10	18	14
11	23	6	16
12	6	7	2
13	2	8	21
14	14	9	23
15	9	14	27
16	18	16	9
17	4	19	20
18	21	20	4
19	20	23	3
20	25	24	18
21	19	17	7
22	27	3	5
23	5	11	19
24	3	13	25
25	1	5	24
26	24	2	1
27	8	1	8

Note: FS, filter strip; SWAT, Soil and Water Assessment Tool.

up each subwatershed are only part of the story. The economic characteristics help to explain the other part.

To illustrate this point, we show how integrating the physiographic and economic information affects our results in Table 3. Consider the case of only having the SWAT results to use for targeting. Furthermore, assume that we are focusing on installing filter strips in areas that result in the most erosion reduction in terms of tons/acre/year. The ranking priority in this case is shown in column two of Table 3. Next, consider the case in which we ignore the soil erosion baseline and reduction potential of each subwatershed (i.e., ignore the SWAT modeling results) and only focus on the costs of the filter strips. Here, we give highest priority to those subwatersheds that have the lowest annual costs for filter strip implementation. The ranking priority for this case is shown in column three of Table 3.

Finally, consider the case of combining the economics information and the SWAT model results. If we focus on installing filter strips in areas that result in the most cost-effective erosion reduction, we have the ranking priority shown in the in the last column of Table 3.

One can see that the rankings are quite different. For example, subwatershed 25 should have the highest ranking if we only go by the SWAT model. However, when economics are considered as well, subwatershed 25 is among the lowest priority subwatersheds. The exact opposite is true for subwatershed 26. The integrated results suggest it should be the highest priority, whereas the SWAT model alone ranks it as among the lowest.

Neither the SWAT model nor the economic model is the primary driver of the integrated model results. Note the results for subwatersheds 18 and 19. Both the SWAT model and the economic model alone rank these areas relatively low. However,, when the information from the two is combined, the subwatersheds are highly ranked. This clearly illustrates why it is important to consider the two type of information in tandem and how the integrated approach is clearly different than alternatives that simply rely on one or the other.

Conclusions

Increases in global demand for food, feed, and fuel over the past five years have given agricultural producers sufficient market signals (i.e., higher prices) to increase commodity production. Increased production requires additional land, nutrients, water, and other cropping inputs to be used to produce greater overall yields. If not done carefully, these changes in agricultural production can create the potential for greater sediment and nutrient runoff resulting in poorer surface water quality. Thus, the potential tradeoff between agricultural production and environmental quality has renewed urgency (Claassen, 2009).

This study examined physiographic and economic relationships within a large Midwestern watershed to provide insights into the selection of cost-effective sediment reduction strategies. A geographically based watershed model and economic model were used to determine cost-effective cropland management sedimentation reduction strategies.

This study demonstrated how large amounts of physiographic and economic data can be harnessed to yield readily understood cost-effective targeting maps. Equipped with this information, watershed managers can devise more effective implementation strategies for their available resources. Likewise, state and federal officials can better anticipate the costs and effort needed to achieve objectives of effective investments and resource preservation.

It needs to be understood, however, that our analysis can only go so far in providing definitive targeting answers. Much depends on the nature and location of the existing BMPs or any unaddressed problems that cannot be accurately modeled (e.g., streambank erosion). From here, subwatershed information can be broken out and evaluated by local stakeholder groups who have direct knowledge of local conditions. Superior local knowledge coupled with analysis results will yield the most useful outcomes.

Clearly, significant sediment reduction is an expensive proposition from a cost perspective (i.e., not considering the value of any resulting benefits). Furthermore, if net returns from crop production continue to increase, the costs of reducing sediment with practices that remove cropland from production will increase and make NT more cost-effective relative to other strategies.

Our study also shows that the random BMP implementation is not cost-effective. Random BMP implementation is somewhat representative of a policy where conservation funds are issued to any interested and willing landowner in a watershed. Although this approach achieves equity, conservation dollars are being spent in areas that do not deliver a good return in terms of sediment reduction for the dollars expended. Specifically, a highly targeted approach can reduce more sediment for a given budget than a random approach. It should be noted, however, that a highly targeted approach can be costly from an administrative standpoint. Thus, we presented an "alternative" cost-effective targeting approach that may be preferable to watershed managers for its "user-friendliness." This increased user-friendliness may be worth the loss in terms of cost-efficiency relative to the most cost-efficient solution. Other results also indicate that it is important to know where BMPs have already been established when modeling cost-effective implementation approaches.

Our results provide clear evidence that conservation programs can and probably should explicitly emphasize cost-effectiveness in the allocation of scarce resources. We recognize, however, that states have variable capacity to perform the modeling and conduct the analyses necessary. Federal agencies such as U.S. Department of Agriculture and U.S. Environmental Protection Agency, the source of much of the conservation support, can implement programmatic rules to motivate development of such state-level capacity and/or the establishment of partnerships with public and private entities where such capacity currently exists. Our belief is that such a directive will itself be a cost-effective investment leading to better environmental outcomes.

Consideration of only physical data or economic costs in isolation do not result in an optimal sediment reduction strategy from a cost-effectiveness standpoint. Targeting areas that produce the most pollution per acre is more cost-effective than a random approach but may miss the mark if those areas also exhibit high BMP costs (e.g., as a result of high opportunity costs). Likewise, focusing only on areas where BMP costs are low may be an improvement over a random approach but may not achieve cost-effective pollution reduction if the areas do not exhibit high levels of sediment reduction. Both physiographic and economic factors must be considered for cost-effective conservation to occur.

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