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Landscape Planning for Agricultural Non–Point Source Pollution Reduction. II. Balancing Watershed Size, Number of Watersheds, and Implementation Effort

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Abstract Agricultural non–point source (NPS) pollution poses a severe threat to water quality and aquatic ecosystems. In response, tremendous efforts have been directed toward reducing these pollution inputs by implementing agricultural conservation practices. Although conservation practices reduce pollution inputs from individual fields, scaling pollution control benefits up to the watershed level (i.e., improvements in stream water quality) has been a difficult challenge. This difficulty highlights the need for NPS reduction programs that focus efforts within target watersheds and at specific locations within target watersheds, with the ultimate goal of improving stream water quality. Fundamental program design features for NPS control programs—i.e., number of watersheds in the program, total watershed area, and level of effort expended within watersheds—have not been considered in any sort of formal analysis. Here, we present an optimization model that explores the programmatic and environmental tradeoffs between these design choices. Across a series of annual program budgets ranging from \$2 to \$200 million, the optimal number of watersheds ranged from 3 to 27; optimal watershed area ranged from 29 to 214 km^2; and optimal expenditure ranged from \$21,000 to \$35,000/km^2. The optimal program configuration was highly dependent on total program budget. Based on our general findings, we delineated hydrologically complete and spatially independent watersheds ranging in area from 20 to 100 km^2. These watersheds are designed to serve as implementation units for a targeted NPS pollution control program currently being developed in Wisconsin.

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Introduction

Excess loading of nutrients to streams as a result of agricultural runoff is a leading cause of eutrophication of lakes, rivers, and estuaries as well as the loss of aquatic ecosystem services (United States Environmental Protection Agency [USEPA] 1990; National Research Council 1993; Turner and Rabalais 1994; USEPA 1996; Carpenter and others 1998; Burkart and James 1999). These impacts have led many state and federal agencies to take action to improve surface water quality through reductions in agricultural non–point source (NPS) pollution (USEPA 2002). The magnitude of these programs has been tremendous. For example, since 1987, the United States Department of Agriculture (USDA) Conservation Reserve Program has distributed \$29.7 billion to owners of agricultural land to implement conservation practices that reduce soil loss, restore wetlands, and conserve forested areas (USDA 2006). Despite these expenditures, there is little evidence that application of conservation practices has produced measurable improvements in stream water quality at broad spatial scales (Wolf 1995; Meals 1996; Boesch and others 2001).

In response, state and federal agencies are recognizing the need to improve the design of conservation programs (Hansen and Hellerstein 2006). At the forefront of these improvements has been the emergence of adaptive watershed management (Freedman and Nemura 2004; Allan and others 2008). In active adaptive management, existing knowledge is used to design experiments that test hypotheses about ecosystem responses to management (Walter 1986; Lee 1993; Blumenthal and Jannink 2000; Pahl-Wostl 2007). In the context of watersheds, adaptive management serves as an experimental framework in which responses in water quality are used to assess the effectiveness of pollution-reduction approaches. Although watersheds represent important venues for the application and development of new knowledge (Allan and others 2008), resource management agencies have lacked the guidance needed to make adaptive watershed management effective. In Wisconsin, recent efforts to revise NPS regulations have provided an opportunity to evaluate current approaches to addressing agricultural runoff, identify reasons for the lack of broad-scale water-quality improvements, and revisit critical program design features. A multistakeholder group, called the Wisconsin Buffer Initiative (WBI), provided a forum for consideration of these issues as well as an opportunity to design a NPS pollution-control program based on adaptive management principles (WBI 2005).

This is the second in a series of three articles deriving from our work with the WBI. In the first article, Diebel and others (this issue "a") used a simulation approach to examine alternative scenarios of allocating NPS pollution-reduction efforts at two spatial scales: among and within watersheds. Several different allocation strategies were assessed based on their potential for achieving phosphorus (P) load reductions as well as generate measurable water-quality improvements. The preferred strategy involved aggregating efforts in the subset of watersheds with the greatest P-reduction potential, and, within these watersheds, targeting fields with the greatest P loading.

Implementation of this targeting strategy requires an initial comparison of many watersheds. This being the case, how might we designate these watersheds in a way that maximizes the overall utility of a targeted program and facilitates implementation of adaptive watershed management? Solutions to this issue must consider the obvious trade-offs between watershed size and the number of watersheds: For a given budget, if watersheds are larger, then fewer watersheds can be targeted. In addition, consideration must be given to the amount of effort expended within each targeted watershed.

This article introduces a framework for exploring how trade-offs between individual watershed size, the total number of watersheds, and the amount of effort expended within watersheds can affect the overall utility of an adaptive watershed management program. We used this framework to identify optimal designs for P- and sediment-reduction programs across a range of budget levels. Based on our results, we delineated a series of 1,598 watersheds to be used for implementing this program in the state of Wisconsin. The third and final article of the series (Diebel and others, this issue "b") uses these watersheds for modeling broad-scale patterns of NPS P and sediment loading to streams and forms the basis for ranking watersheds according to water-quality restoration potential.

Methods

Any agricultural pollution-reduction program will face the inevitable design issues of how to designate and select management areas (in this case, watersheds) and how to allocate implementation efforts among the selected watersheds. To examine the trade-offs between program design options, we constructed a simple model of a watershed management program. Total program cost (dollars), C, of a watershed-based NPS pollution-reduction program is calculated as: $C = N \times A \times E$, where N is the number of watersheds in the program, A is the mean watershed area (km^2), and E is the conservation expenditure per unit area within the watersheds (\$/km^2). The preceding equation can also be expressed in terms of program utility, such that an increase in N, A, or E increases total program utility accordingly. Total program utility, U_P, can be estimated as: $U_P = U_N \times U_A \times U_E$, where U_N, U_E, and U_A are scaled to range from 0 to 1 and represent the utility gained from the number of watersheds in the program, the mean watershed area (km^2), and the level of effort per unit area within targeted watersheds (\$/km^2), respectively. If utility is linearly related with N, A, and E, then watershed size, watershed area, and implementation effort are interchangeable, and there exists no optimal combination among these three program features. Alternatively, if there are nonlinear relations among utility and N, A, and E, then the combination of program features producing the highest total program utility (U_P) can be approximated and thus considered to be the optimal program design. In the following sections, we generate nonlinear utility functions for N, A, and E, which are subsequently used to solve for optimal combinations of number of watersheds, watershed size, and level of implementation effort across a range of program costs. To facilitate solution of optima, utility functions were specified as continuous functions.

Number of Watersheds

The number of watersheds included in a program is the first NPS pollution program design feature we consider. Determining an appropriate number of watersheds is of particular relevance within the context of an adaptive management approach (Walter 1986; Lee 1993). Adaptive management treats policies as scientific hypotheses that can be tested and used to gain new knowledge (Walter

1986; Lee 1993). In an adaptive NPS pollution–control program, strategies designed to reduce pollution (for example, best management practices or regulatory policies) would be applied across all fields within a set of independent watersheds. The benefits of each strategy would be evaluated and compared with alternatives implemented in other watersheds to assess relative effectiveness. Thus, within this framework each watershed represents a single observation that can be compared statistically.

In light of uncertainty associated with water-quality responses to conservation practices and complexity across the broader landscape, it is particularly important to implement an adaptive NPS pollution–control program in multiple watersheds. Implementing an NPS control program in $\leq$1 or a small number of watersheds provides individual case studies but does not provide the statistical power required for a rigorous assessment. In contrast, adding one additional watershed to an already large set of watersheds provides little additional benefit in terms of study design and generality. Although treating more watersheds would clearly yield greater total benefits, the *marginal* knowledge gained from adding an additional watershed in an adaptive management–based program decreases as the total number of watersheds increases. Based on this rationale, we constructed a utility function based on the marginal gains in statistical power resulting from additional watersheds.

Given an expected difference between population means, an expected population variance, and a chosen significance level, the statistical power of any difference test is related to the number of observations recorded (Zar 1999). This relation is approximated by a log-sigmoid curve. Because the parameters of this function depend on the experimental design, which would vary among programs, we defined three utility functions that span a range of designs. Each of these is a log-sigmoid utility function for N, with midpoints (M; where $U_N = 0.5$) of 5, 10, and 20 watersheds (Fig. 1A). The M = 10 function is the baseline scenario, and is used in further evaluations. The equation is as follows:

$$U_N = \frac{1}{1 + e^{\frac{\log_e(M) - \log_e(N)}{0.5}}} \qquad (1)$$

Watershed Area

Watershed area is the second program design feature considered in our model. Some benefits derived from watershed-scale management accumulate only as watershed area increases. For example, water-quality improvement in a large watershed can potentially benefit a broader range of aquatic species than improvements in many small watersheds because some species do not occur in small streams.

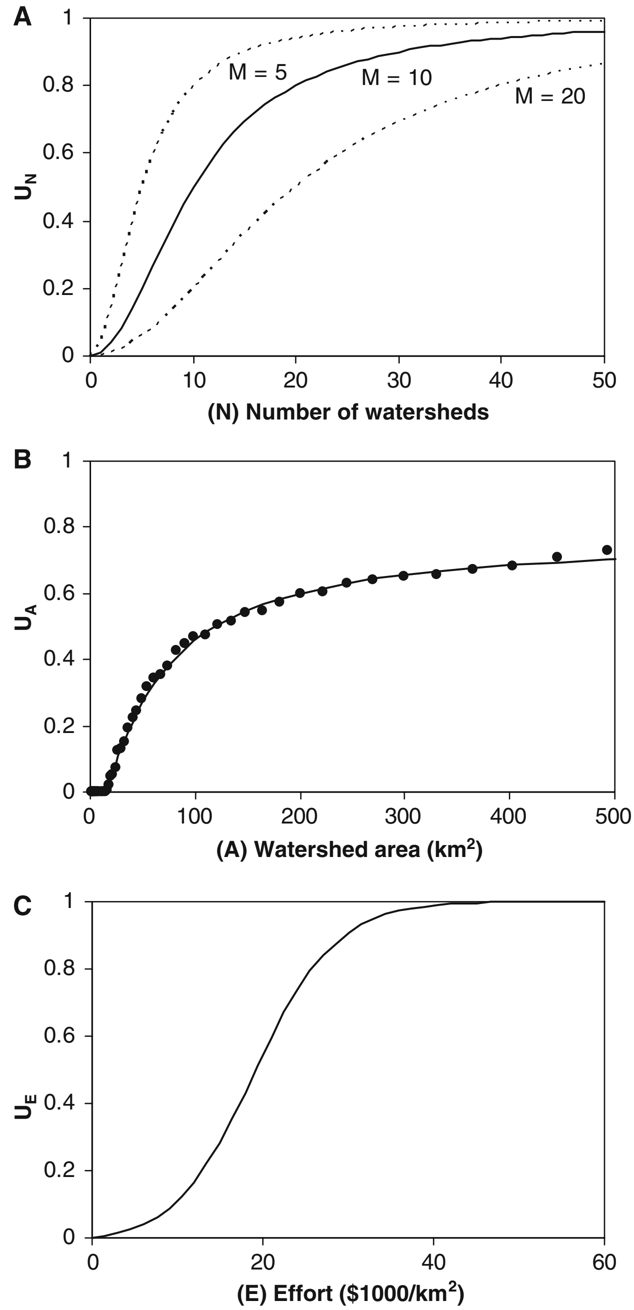

Fig. 1 Utility functions for three basic design parameters of a watershed-based NPS pollution-reduction program. (**A**) Log-sigmoid utility functions with midpoints (M) of 5, 10 (baseline scenario), and 20. U_N = utility gained from the given number of watersheds in the program. (**B**) U_A = utility gained from watersheds with the given mean watershed area. (**C**) U_E = utility gained from the given level of effort per unit area within the targeted watersheds

Although the links between NPS pollution reduction and the diversity and abundance of aquatic organisms are complex (Wang and others 2006), management efforts have the best chance of benefiting aquatic organisms in a given water body if they are implemented in the entire upstream contributing area of that water body (Wang and others 2002).

Therefore, to design the watershed area utility function, we started with the scenario in which pollution-control efforts are initiated in headwater systems. From there, we move downstream through the watershed network, thereby encompassing an increasingly large contiguous area. Given that watershed area is the main determinant of stream size (Leopold and Maddock 1953), and stream size is a major habitat component that can influence species' occurrence (e.g., Lyons 1996), then as we proceed downstream, the total range of habitat types and number of species encompassed by the watershed expands. Because the species–area relation is nonlinear, we can identify an optimal watershed size that maximizes the species-to-area ratio. Although this ratio probably varies according to the size of regional species pools and the influence of other habitat characteristics (e.g., coldwater streams have lower species richness, Lyons and others 1996), it can serve as a general guide for management programs whose jurisdictions encompass diverse landscapes.

To estimate the watershed area utility function, we used fish survey records for Wisconsin streams from the United States Geological Survey (USGS Great Lakes Aquatic GAP Project, Middleton, WI, personal communication, 2005) to quantify how fish species accumulate as watershed size increases. Fish communities were sampled using electrofishing and/or netting at a total of 7,413 streams in Wisconsin between 1970 and 2003. For each sample location, the upstream drainage area was calculated. For each of the 152 fish species in Wisconsin, we calculated the median drainage area for all point occurrences of that species and used this median value as a measure of a species' preferred stream size. We then created a species accumulation curve by cumulatively plotting species' preferred stream size versus watershed area (A). The watershed area utility function (U_A) was generated by expressing these values as a percentage of the total number of fish species in the state (Fig. 1B) and then fitting a modified exponential function to the points. The equation is a follows:

$$U_A = 0.78(e^{-53/A}) \tag{2}$$

Implementation Effort

Implementation of conservation practices within target watersheds is the third and final program design feature considered in our model. All else being equal, a greater investment in conservation practices that reduce NPS pollution should result in greater pollution reduction. However, the benefit derived from each additional unit of pollution reduction effort may not be a linear function. To specify utility (U_E) as a nonlinear function of implementation effort (E), we estimated relations between (1) program expenditure per unit area (E) and P pollution reduction (R_w), and (2) R_w and the probability of detecting a statistically significant water-quality improvement (U_E).

Step 1: Between 1979 and 2006, the Wisconsin Priority Watershed Program (PWP) paid for the implementation of conservation practices by landowners in 87 "priority watersheds" (mean area = 369 km^2) where the need for NPS water-pollution abatement was deemed most critical (Wisconsin Legislative Fiscal Bureau 2007). Funded practices included tillage and nutrient-management changes, fencing to restrict animal access to streams, stream bank shaping and reseeding, and structural barnyard improvements to reduce manure runoff. In 8 of the priority watersheds, total agricultural P loads were estimated before and after implementation of conservation practices (Wisconsin Department of Natural Resources, personal communication, 2007) using a barnyard P model and a cropland P model. The barnyard model, BARNY (Wisconsin Department of Natural Resources [WDNR] 1994a), calculates average annual P runoff from individual barnyards based on animal numbers, feedlot area, slope, and drainage area. WINHUSLE (WDNR 1994b) estimates cropland P runoff from individual fields by multiplying sediment loss estimates from the Universal Soil Loss Equation by the average regional soil P content. Both models can be used to simulate the effect of conservation practices on P runoff to streams. We regressed these modeled P load reductions (R_w, as a proportion of the preimplementation level) on the cost of program implementation per unit watershed area (E) in each watershed (Fig. 2). The resulting equation is as follows:

$$R_w = 0.017E \tag{3}$$

Step 2: An effective adaptive management program should seek to make changes that are likely to be detectable. Because P concentrations in streams vary through

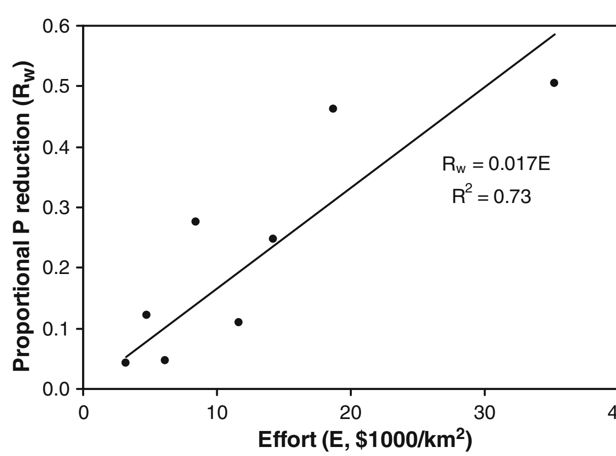

Fig. 2 The percentage of annual P loss reduced by best management practices (BMPs) in eight Wisconsin watersheds is related to the amount of effort expended per unit of watershed area

time as a result of natural factors, small values of R_w are difficult to detect statistically. In the first study reported in this series (Diebel and others, this issue "a"), we used statistical simulations to describe how the probability (p) of detecting a statistically significant P reduction at the outlet of a watershed depends on R_w from land in the watershed. This is a sigmoid function as follows:

$$p = \frac{1.017}{1 + 59(e^{-12.8 \cdot R_w})} - 0.017 \tag{4}$$

Substituting Equations 3 into 4 and setting $p = \mathrm{U_E}$ gives the utility function for E as follows:

$$U_E = \frac{1.017}{1 + 59(e^{-0.21 \cdot E_w})} - 0.017. \tag{5}$$

Thus, the utility of a unit expenditure depends on how much it contributes to the probability of causing detectable water-quality improvements at the watershed outlet (Fig. 1C).

Estimating Optimal Program Design

Using the previously described functions describing how utility varies with the number of watersheds, the watershed area, and the implementation effort, we used the Generalized Reduced Gradient (GRG2) Algorithm (Lasdon and others 1978) in Microsoft Solver to estimate the optimal combination of these three program features (i.e., the combination producing the highest total program utility $\mathrm{U_P}$). The calculation of optimal program design features was conducted across a range of annual program budgets, ranging from \$2 to \$200 million. We estimated program efficiency as total program utility ($\mathrm{U_P}$) divided by total program cost (C, millions of dollars). Program efficiency was estimated across the range of annual program budgets and used to identify the most efficient program budget (the budget that produces the highest value of $\mathrm{U_P}$/C).

Results

The optimal combination of number of watersheds, watershed size, and effort varied strongly as a function of total program budget (Fig. 3). For example, the optimal program design for a project with a \$2 million budget was 3 watersheds of 29 km^2 each, with an expenditure of \$21,000/km^2. The optimal design for a \$200 million budget was 27 watersheds of 214 km^2 each, with an expenditure of \$35,000/km^2. Across the range of budgets considered here, effort (E) was relatively high and varied less than two-fold. In contrast, the number of watersheds (N) and watershed area (A) varied strongly as a function of program budgets.

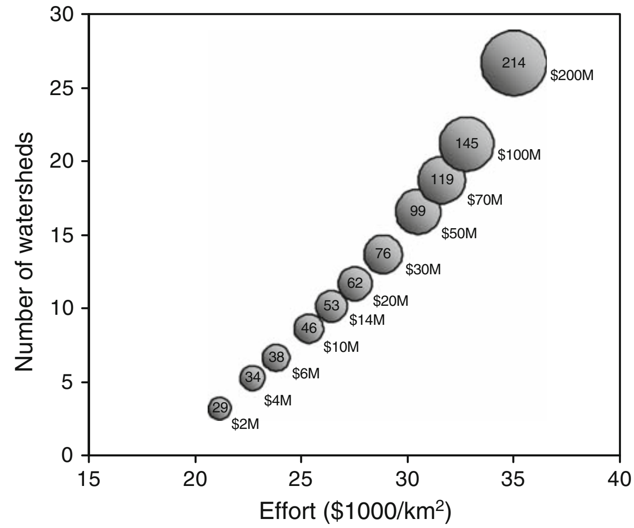

Fig. 3 Optimal program characteristics across a range of program budgets. The size of each circle is proportional to the optimal watershed size (noted in km^2 in each circle) at that budget level. M = million

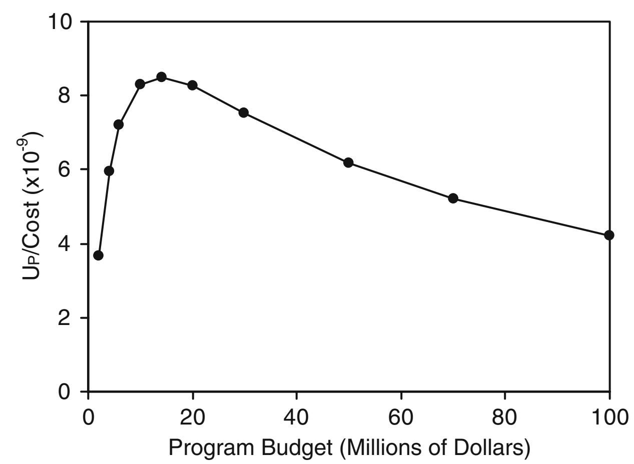

Fig. 4 Total program utility ($\mathrm{U_P}$) per unit cost across a range of program budgets. Efficiency is maximized with a \$14 million budget

The number of watersheds increased faster than watershed area until a budget of \$100 million, after which watershed area increased more rapidly. Our estimate of program efficiency ($\mathrm{U_P}$/C) was highest at a program budget of \$14 million (Fig. 4: N = 10 watersheds, A = 53 km^2, and E = \$26,000/km^2). Using alternative functions to calculate $\mathrm{U_N}$ affected the optimal values for N and C when program efficiency ($\mathrm{U_P}$/C) is maximized (for M = 5, N = 5, and C = \$7 million; for M = 20, N = 20, and C = \$28 million). Optimal values of E and A in the most efficient program were not affected by changes in the $\mathrm{U_N}$ function because each utility function has a single point at which the ratio between utility and the parameter value is maximized.

Watershed Delineation for Wisconsin

We used the model described previously to help guide our general approach for designating watersheds to serve as implementation units for the program being developed by the Wisconsin Buffer Initiative (WBI 2005). For watershed-based NPS pollution-reduction programs, it is important to use hydrologically complete watersheds that account for the entire upslope contributing area, instead of downstream "hydrologic units" (Griffith and others 1999). Our objective was to delineate a set of spatially independent, hydrologically complete watersheds with a target size of 53 km^2, the value for A when U_P/C is maximized. A synoptic assessment (USEPA 1992) of these watersheds would lead to prioritization (Diebel and others, this issue "b"), selection of an optimal number of watersheds, and subsequent implementation of conservation efforts.

Before delineating watersheds, we used the ArcHydro (Maidment 2002) extension to ArcGIS version 8.3 (Environmental Systems Research Institute, Redlands, CA 2006) to integrate mapped streams (1:24,000 scale) with a 7.5 minute (30-meter) digital elevation model (DEM) of Wisconsin (USGS 1997). This process minimizes watershed delineation errors that stem from errors in the DEM, which can lead to inaccurate calculations of land cover proportions and subsequent loading estimates (Baker and others 2006). Watershed outlet points were placed at the downstream-most stream confluences where flow accumulation was within a range of 20 to 100 km^2 (a range that approximates the target mean size of 53 km^2). Finally, watersheds were delineated for each outlet point using the DEM and automated routines within the ArcHydro GIS software. A total of 1,598 watersheds with a mean watershed area of 54 km^2 were delineated for Wisconsin (Fig. 5).

Discussion

In light of the importance of agricultural NPS pollution as a global environmental management issue, natural resource

Fig. 5 Hydrologically complete watersheds delineated for the WBI (mean size 54 km^2, range 20–100 km^2)

management agencies are increasingly adopting a targeted approach for the implementation of NPS pollution-reduction programs (Hansen and Hellerstein 2006). Furthermore, increased attention to adaptive watershed management has led agencies to consider program designs with a deliberate focus on learning (Allan and others 2008). These trends stem from the need to demonstrate environmental benefits, such as stream water–quality improvements, as a result of major agricultural NPS pollution-control expenditures. Program design features of targeted NPS programs—particularly those relating to the trade-offs between watershed size, number of watersheds, and intensity of effort within targeted watersheds—have not been subject to a rigorous analysis. These essential design features of a NPS pollution-control program should be subject to careful consideration before designing and implementing a costly program.

States that use a watershed approach to manage water quality have used spatial frameworks with mean watershed sizes ranging from 153 (New Jersey) to 8,200 km^2 (North Carolina) (USEPA 2002). In Wisconsin, the $201 million PWP designated 87 priority watersheds with a mean size of 369 km^2 (Wisconsin Legislative Fiscal Bureau 2007). These programs have not consistently demonstrated that significant water-quality improvements can be achieved at the watershed scale. Studies have suggested that this is due to insufficient effort within targeted watersheds (Wolf 1995; McNitt and Kepford 1999) or by insufficient monitoring to detect improvements (Bernhardt and others 2005).

The framework presented here provides practical guidance for the development of programs that target conservation practices within watersheds. The optimal watershed sizes identified here are small relative to current state programs, suggesting that current programs may be able to improve the balance between watershed size and other program design features. According to our model, as program budgets increase, the number of watersheds should increase most quickly, followed by watershed size, and then by management effort. Even with a small budget, watersheds should not be smaller than approximately 30 km^2, or few fish species will benefit from the water-quality improvements that occur. Management effort should be greater than approximately $20,000/$km^2$ or water-quality improvements will be difficult to detect. This quantitative effort guideline is probably the least transferable to other programs because the relationship between management cost and environmental outcome is dependent on the specific management practices used as well as on institutional arrangements for funding those practices, both of which are highly variable. Similarly, although fish distributions served well in our model development because they are valued culturally and extensive data sets exist, alternative utility functions could be used to describe the relationship between watershed size and program utility. For example, nonlinear relations between pollutant reduction potential and watershed size may exist. In addition, societal benefits from fishing, boating, and other recreational benefits may increase with increases in watershed size. Finally, the utility function for the number of watersheds may also vary depending on the number of treatment variations imposed and the expected effect size of those treatments. Therefore, the specific utility functions we defined should not be regarded as universally applicable but rather should be viewed as illustrative of the trade-offs among program features of a watershed-based NPS pollution-control program.

Even within a program that explicitly considers the balance between watershed size, number of watersheds, and conservation effort, the issue of reducing diffuse agricultural pollution remains complex. Additional social factors, including the difficulties of establishing cooperation among land owners and entrenched characteristics of natural resource management agencies (Allan and others 2008), may result in deviations from the optimal program. Adjustments that result in a practical and realistic implementation strategy may involve a reasonable trade-off between cost-effectiveness and the likelihood of widespread program adoption.

Variations in landscape characteristics make the existence of a universal solution to NPS pollution unlikely. In the absence of a panacea, a framework that targets comparably sized and spatially independent watersheds provides the opportunity to test hypotheses about the management of agricultural runoff across landscapes. These hypotheses can be formed around questions within and among watersheds to account for variations in soil type, topographic relief, climate, geology, and other variables. For example, fields *within* a watershed could be monitored and adaptively managed to implement the crop rotations, tillage practices, and nutrient management strategies that most effectively reduce nutrient and sediment delivery to nearby streams. *Among* watersheds, regulatory variations can be tested to determine which program design options (such as development of economic incentives or imposition of a regulatory structure) result in improved water quality at the watershed scale. As investments in watershed conservation and subsequent understanding of the linkages between land use and water-quality increase, we may be able identify the most effective methods of decreasing NPS pollution. With time, the new knowledge that is applied locally may yield additional environmental benefits to downstream water bodies (e.g., Gulf of Mexico, Chesapeake Bay) that integrate diverse landscapes.

Conclusion

Although conservation practices are effective at decreasing NPS pollution inputs from individual fields, a central resource management challenge is to “scale up” to yield watershed-level benefits in the form of improved stream water quality. Targeting and aggregation of conservation efforts are increasingly recognized as important ways of producing such broader benefits. Although such programs often adopt a watershed approach, studies have not considered the trade-offs among fundamental program design features, such as watershed size, number of watersheds, and level of implementation effort. Our efforts to explore optimal program design configurations provide a framework for designing NPS pollution-control programs that also incorporate principles of adaptive watershed management. This approach is designed to produce improvements in stream water quality at targeted locations and to maximize learning about how water quality responds to conservation practices at the watershed scale.

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