Allocating Nutrient Load Reduction across a Watershed: Implications of Different Principles

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Abstract

A watershed based model, the Soil and Water Assessment Tool (SWAT), along with transfer coefficients is used to assess alternative principles of allocating nutrient load reduction in the Raccoon River watershed in central Iowa. Four principles are examined for their cost-effectiveness and impacts on water quality: absolute equity, equity based on ability, critical area targeting, and geographic proximity. Based on SWAT simulation results, transfer coefficients are calculated for the effects of nitrogen application reduction. We find both critical area targeting and downstream focus (an example of geographic proximity) can be more expensive than equal allocation, a manifestation of absolute equity. Unless abatement costs are quite heterogeneous across the subwatersheds, the least-cost allocation (an application of the principle of equity based on ability) have a potential of cost savings of about 10% compared to equal allocation. We also find that the gap between nitrogen loading estimated from transfer coefficients and nitrogen loading predicted by SWAT simulation is small (in general less than 5%). This suggests that transfer coefficients can be a useful tool for watershed nutrient planning. Sensitivity analyses suggest that these results are robust with respect to different degrees of nitrogen reduction and how much other conservation practices are used.

Key words: Least-cost allocation, Soil and Water Assessment Tool (SWAT), Transfer coefficients.

1. INTRODUCTION

It is increasingly recognized by the public that nonpoint source pollution is the largest remaining source of water pollution in the US. In agricultural dominated regions, nonpoint source control is primarily aimed at preventing and reducing agricultural pollutants. It is clear that the allocation of loading among nonpoint sources, which has not been a focus of the current implementation of the Total Maximum Daily Load (TMDL) program, will be a critical issue in the search for measures to achieve water quality goals at the lowest cost possible. A variety of criteria can be used to decide where conservation measures should be implemented to control agricultural nonpoint source water pollution.

The first is absolute equity, which often requires that every subwatershed makes the same percentage of load reduction per hectare or per capita. A second often used criterion is equity based on ability. According to this criterion, those with lower marginal costs of abatement are required to make bigger cuts in pollutant load. In addition to being simple, these two criteria also have direct policy relevance—absolute equity can be implemented with a command-and-control type policy, while equity based on ability is consistent with a market-based mechanism such as taxes or permit trading. Geographical proximity is another criteria often used to decide which areas should be considered to share the responsibility of improving water quality in a watershed. For example, conservation measures are sometimes assumed to be implemented in the entire county where the cropland impaired water body lies [*USEPA*, 2006]. Finally, it is not unusual that conservation measures are required only in critical areas responsible for a disproportionate share of loading or having most potential for improvement. In this paper, we examine the consequences of these four criteria in terms of water quality benefit and abatement costs.

There are practical challenges in applying some of the criteria mainly because they require the assessment of the impacts of conservation practices on the fate and transport of pollutants at a watershed scale. Nutrient loads discharged from specific source areas can be further impacted by ongoing in-stream processes including deposition or assimilation along the waterway, the input of additional nutrients via riverbank and/or riverbed erosion, or additional nutrient inputs through atmospheric deposition. Moreover, the degree of such deposition and erosion effects can often be affected by what happens in other subwatersheds across the watershed. Thus, to understand the effects of a particular conservation practice adopted in a subwatershed, it is often necessary that we understand the complex hydrologic process in the whole watershed.

Simulation models that are designed to capture such complex hydrologic and pollutant transport processes can be used to aid our understanding. These models range in complexity and can require extensive input data and a high level of expertise, in order to perform a successful calibration and validation exercise for a given watershed. Besides, while useful in providing insights on the fate and transport of nutrients, it can be difficult to base policy design directly on these models often because of their complexity.

For both theoretical and empirical policy analyses, transfer coefficient models, which use simple parameters to capture the long term impacts of conservation measures, have been used for a long time. There is a broad literature of nutrient transfer coefficients by land-cover type based on decades of field-based research [*Beaulac and Reckhow,* 1982, *Johnes,* 1996]. In this literature, transfer coefficients are usually referred to as export coefficients. As early as the 1970s, economists used transfer coefficients to study how market based mechanisms can be utilized to minimize the cost of pollutant abatement [*Montgomery*, 1972]. Recently, *Hung and Shaw* [2005] showed that a trading-ratio system based on transfer coefficients can achieve the least cost to reach a water quality goal. *Khanna et al*. [2003] found that incorporating endogeneity in transfer coefficients can have a large impact on the costs of abatement, based on an application of the Agricultural Non-Point Sources Pollution (ANGPS) water quality model to a watershed in Illinois. In the context of local versus regional water pollutants, *Kling et al*. [2005] examined the extent to which transfer coefficients based on the Soil and Water Assessment (SWAT) model [*Arnold et al.,* 2000; *Arnold and Forher*, 2005] can be used to assess nutrient trading at a large regional scale. There are other simple biophysical measures that can be used to quantify the potential of pollutant loading from a source. For example, the phosphorous index can be used to characterize the potential of a field or region to load phosphorous into surrounding waters [*Johansson and Randall*, 2003].

In this study we use SWAT and transfer coefficients, similar to those used in *Kling et al* [2005], to assess alternative policy scenarios in a small watershed—the Raccoon River Watershed in Iowa. In the scenarios, the allocation of load reduction responsibility is based on the four principles discussed earlier and on the transfer coefficients derived from the output of SWAT for the Raccoon River Watershed. We then examine how the allocations differ in terms of cost-effectiveness and water quality improvement. By assessing the four principles, we provide some general guidance to researchers and watershed planners in selecting treatment areas. We also contribute to the literature by testing the validity of allocating load reduction responsibility in a small watershed based on transfer coefficients.

2. STUDY REGION

The Raccoon River Watershed drains a total area of about 9397 km^2 in west central Iowa (Figure 1). The land use in the watershed is dominated by agriculture, with about 75.3% in cropland, 16.3% in grassland, and 4.4% in forest. Urban use accounts for the remaining 4.0% of the total area. The Raccoon River and its tributaries drain all or parts of 17 counties before joining the Des Moines River in Des Moines. The watershed is the primary source of drinking water for central Iowa communities including the city of Des Moines which has a population of 200,000.

Intensive agriculture with widespread application of nitrogen fertilizer has been identified as the primary source of high nitrate concentration in the watershed which is a major concern both locally and regionally. Since the late 1980s, the Des Moines Water Works has operated the world's largest nitrate removal facility, due to the high concentration of nitrate. Sections of the Raccoon River are included in Iowa's Federal Clean Water Act 303(d) list of impaired waters, due to the high nitrate levels. Nitrates discharges from the Raccoon and other rivers in the Upper Mississippi River Basin have been implicated as a key source of the Gulf of Mexico seasonal hypoxic zone, that has covered upwards of 20,000 km^2 in recent years [*Rabalais et al.*, 2002]. The Committee on Environment and Natural Resources (CENR) recommended the implementation of several on-farm practices for reducing discharges of nitrogen to streams and rivers. Among these practices is a 20% reduction in nitrogen fertilizer application [*Mitsch et al.,* 1999].

3. MODELING FRAMEWORK

The SWAT model is a conceptual, physically based long-term continuous watershed scale simulation model that operates on a daily time step. In SWAT, a watershed is divided into multiple subwatersheds, which are then further subdivided into Hydrologic Response Units (HRUs) that consist of homogeneous land use, management, and soil characteristics. Key components of SWAT include hydrology, plant growth, erosion, nutrient transport and transformation, pesticide transport, and management practices. Detail theoretical description of the SWAT model and its major components can be found in *Neitsch et al*. [2002]. Outputs provided by SWAT include streamflows and in-stream loading or concentration estimates of sediment, nutrients, and pesticides. Previous applications of SWAT for streamflows and/or pollutant loadings have compared favorably with measured data for a variety of watershed scales [e.g., *Arnold and Allen*, 1996; *Arnold et al.*, 1999; *Arnold et al.*, 2000; *Santhi et al.*, 2001; *Borah and Bera*, 2004].

This study is based on the SWAT modeling framework developed by *Jha et al.* [2006], who calibrated and validated SWAT for streamflow, sediment loads, and nitrogen and phosphorus losses for the Raccoon River Watershed. This framework facilitates analyses of the impacts of potential policy scenarios on flow, sediment and other water quality indicators in the region. Basic input data used to setup the SWAT simulation include topography, weather, land use, soil, and management data. A key source of land use, soil and management data was the National Resources Inventory (NRI) database [*Nusser and Goebel*, 1997]. The NRI is a statistically based survey database that contains information for the entire U.S. such as landscape features, soil type, cropping histories, tile drainage, and conservation practices for the whole nation. The climate data were obtained from the National Climatic Data Center for 10 weather stations located in and around the watershed. In the modeling framework, the watershed is delineated into 26 subwatersheds identical to the 10-digit level of Hydrologic Unit Codes. The outlet of subwatershed 25 is also the outlet of the whole Raccoon River watershed (Figure 1).

4. THEORETICAL POLICY ANALYSIS—AN APPROXIMATE LEAST-COST ALLOCATIONS OF NUTRIENT ABATEMENT

Suppose there is a goal of reducing nutrient loading at the watershed outlet by $\bar{N}$ kilograms for a watershed divided into J subwatersheds. Let the cost of nutrient application reduction be $C_j(N_j A_j)$, where N_j is the nutrient application reduction in kilograms per hectare, and A_j is the total hectares in subwatershed j. The effect of nutrient application reduction at all subwatersheds (i.e., the total nutrient loading reduction at the watershed outlet) is represented by a function $f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w})$, where $\boldsymbol{w}$ represents other land use characteristics and natural elements such as weather. The function $f(\bullet)$ reflects the complex hydrologic process in the watershed, takeing into account the possibility that the effect of nutrient application reduction at one subwatershed can depend on the characteristics of the whole watershed and the action taken at other subwatersheds. We can write the total nutrient standard as

(1) $$f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w}) \le \bar{N}$$

Then the following problem can be set up to find the least-cost allocation of nutrient application reduction to meet the nutrient standard at the watershed outlet:

(2) $$\min \sum_{j=1}^{J} C_j(N_jA_j)$$

subject to (1).

The solutions can be characterized as

(3) $$\frac{\partial C_j(N_j^* A_j)/\partial N_j}{\partial f(N^*_1A_1, N^*_2A_2, ..., N^*_JA_J; \boldsymbol{w})/\partial N_j} = \frac{\partial C_k(N_k^* A_k)/\partial N_k}{\partial f(N^*_1A_1, N^*_2A_2, ..., N^*_JA_J; \boldsymbol{w})/\partial N_k};$$

where $\partial C_j(N_jA_j)/\partial N_j$ represents the marginal cost incurred from an incremental change in N_j and $\partial f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w})/\partial N_j$ represents the marginal benefit, i.e., the extra loading reduction achieved from an incremental change in N_j. Equation (3) requires that the ratio of marginal benefit over the marginal cost be equalized to achieve the least cost allocation.

It is difficult to apply equation (3) to allocate nutrient application reduction in a watershed mainly because $f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w})$ represents a complex hydrological process. The denominator, $\partial f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w}) / \partial N_j$ can vary with N_i for any $i = 1, 2, ..., J$. In this paper, we explore an approximate form of $f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w})$ which is much simpler and thus has the potential of being utilized in reality. We then examine whether allocations based on the simplified version can be used to achieve water quality standards at the least cost. Specifically, we consider a linear approximation of $f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w})$, i.e.,

(4) $$f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w}) = \sum_{j=1}^{J} d_j N_j A_j$$

where transfer coefficient (d_j) is an approximation of the amount of nutrient reduction at the watershed outlet achieved by one unit of nutrient application reduction in subwatershed *j*.

In order to explicitly solve the problem represented by expression (2), we assume that $C_j(N_jA_j) = A_j c_j(N_j)$, with

(5) $$c_j(N_j) = \alpha_0 + \gamma_j \alpha \frac{\theta}{\theta+1} N_j^{\frac{\theta+1}{\theta}}.$$

The parameters α_0, $\alpha > 0$, $\theta > 0$, and $\gamma_j > 0$, control the scale, shape, and heterogeneity of the abatement cost function in the subwatersheds. In the next section, we provide more discussion on the cost function. With (4) and (5), we can derive a closed form solution for the problem in (2) as follows:

(6) $$N_j^* = \frac{\bar{N} d_j^{\theta} \gamma_j^{-\theta}}{\sum d_i^{\theta+1} \gamma_i^{-\theta} A_i}.$$

Thus, the optimal nitrogen application reduction in subwatershed j depends on the transfer coefficients and cost parameters in all subwatersheds. The solution in equation (6) is very fortuitous for our empirical analysis in that we do not need to know the precise size of the abatement cost function in order to allocate nutrient load, because α_0 and α do not appear in equation (6). As far as abatement cost is concerned, we only need to know the shape of the cost function as represented by θ and the relative magnitude of the cost across the subwatersheds as represented by γ_j. This not only facilitates our empirical analysis but also is very important in the real world given that the exact magnitude of cost for nutrient reduction can be hard to obtain.

5. EMPIRICAL POLICY ANALYSIS

In our empirical analysis, we focus on the reduction of nitrogen fertilizer application on cropland, which as mentioned earlier is a recommended practice by the NCER. We examine the effectiveness of four strategies, which are applications of the four principles we discussed in the introduction section, to reduce the application of nitrogen fertilizer. For the first strategy, an example of absolute equity, the fertilizer reduction is reduced by the same percentage across all subwatersheds. In the second strategy, the allocation of nitrogen application reduction for the subwatersheds is based on their marginal benefits and marginal costs and is determined by equation (6). This is an application of the equity based on ability principle in the following sense: those that can reduce nutrient load at relatively low costs are required to have larger nitrogen application reductions. In the other two strategies, nitrogen application reduction is only required for roughly half of the subwatersheds which are: (a) located in the downstream (lower) reaches of the watershed, or (b) found to be the most effective treatment areas in terms of nutrient reductions as measured by higher transfer coefficients. These two strategies are manifestation of principles based on geographic proximity and critical areas. In all strategies, we assume that the total nitrate load reduction is the same as estimated from the transfer coefficients. (We choose nitrate as our nutrient indicator because it is the predominant form of nitrogen pollution in water in the study region.) We then examine the cost-effectiveness of these strategies and the nitrate loading reduction achieved based on SWAT simulations, as opposed to transfer coefficients.

In our empirical analysis, we use the following procedure to derive the transfer coefficients (d_j):

1. Conduct one SWAT run: assuming no reduction at all in the watershed, obtain the baseline nitrate loadings at the watershed outlet.
2. Conduct 26 SWAT runs: assuming x percent nitrogen fertilizer application reduction in subwatershed j and 0% reduction at all other subwatersheds. Denote the amount of nitrate loading reduction obtained at the watershed outlet as y_j.
3. Transfer coefficient d_j is then defined as

$$d_j = \frac{y_j}{x * (\text{Baseline nitrogen fertilizer application at sub-watershed j})}.$$

In addition to the baseline simulation and the simulations performed to derive the coefficients, we also perform four additional SWAT simulations for the four strategies of allocating nitrogen fertilizer application reduction: (i) reduction in each subwatershed by the same percentage, say 20%; (ii) reduction in each subwatershed based on equation (6); (iii) reduction in only 14 downstream subwatersheds; and (iv) reduction in only 13 subwatersheds with the highest transfer coefficients. For the runs in (ii)-(iv), it is assumed that the nitrate loading reduction estimated from the transfer coefficients is the same as that achieved in (i). The issue we want to explore is how much costs differ among the four strategies and whether the same nitrate load reduction will be achieved giving that the transfer coefficients are only based on approximate estimates of nitrate loading impacts.

In order to compare the cost of these scenarios, it is important that we understand the cost of nitrogen application reduction. The estimation of the yield effect is the most important and probably the most controversial and unresolved issue in costing the reduction of nitrogen fertilizer application due to a number of factors. The yield effect of a moderate reduction in nitrogen fertilizer application has been estimated to be almost none, positive, or negative. Some states still recommend more fertilizer for a higher yield goal, while others have discontinued the practice [*Lory and Scharf*, 2003]. It is difficult to estimate the impacts of fertilizer application because the effects may be masked by weather, previous crops, soil condition, etc. Moreover, the reduction of fertilizer may have an insignificant effect in the short run; however, the long run effect may be large. In addition to the issues related to yield effects, *Babcock* [1992] also showed that the seemingly over-application of nitrogen fertilizer is actually consistent with profit maximization, which implies that a payment will be needed for farmers to reduce their nitrogen fertilizer application.

Given the diverse opinions on the cost of nitrogen fertilizer reduction, we adopt a cost function with flexible shape and scale. In $c_j(N_j) = \alpha_0 + \gamma_j \alpha \frac{\theta}{\theta+1} N_j^{\frac{\theta+1}{\theta}}$, all of the parameters can be calibrated to the abatement cost in a particular watershed. Parameters α_0 and α determine the scale of the cost function. Since α_0 and α do not appear in (6), the optimal allocation of nutrient loading reduction does not depend on these two parameters. The parameter θ determines the curvature of the cost function—the smaller the θ, the faster the cost increases as N_j increases. For a very large θ, the cost function is approximately linear in N_j. The heterogeneity of the cost function among the 26 subwatersheds is reflected by γ_j. If $\gamma_j = 1$ for all *j* then the cost function is the same for all of the subwatersheds.

A sensitivity analysis is conducted with respect to different values of the parameters θ and γ_{j}. In addition, the transfer coefficients may also be sensitive to other conservation measures adopted in the watershed. So we also derive transfer coefficients for all of the subwatersheds assuming no-till is adopted in all cropland. To investigate the sensitivity of transfer coefficients to different amounts of nitrogen fertilizer application reduction, three reduction levels are considered: 10, 20, and 30%. In the rest of the paper, we will call these scenarios the 10% scenarios, the 20% scenarios, and the 30% scenarios, respectively. In 10% scenarios, the transfer coefficients are based on a 10% nitrogen fertilizer application reduction and the nitrate reduction goal is the nitrate loading reduction achieved by a 10% nitrogen fertilizer application reduction in all 26 subwatersheds. For the 20% and 30% scenarios, similar logic applies.

6. RESULTS

Baseline average annual nitrate loadings for each subwatershed are presented in Table 1 as a function of the total land area, corn production area, and baseline nitrogen application rates. The results also show that the loading predicted at each subwatershed outlet varies substantially with the relative location of the subwatersheds in the whole Raccoon watershed as shown in Figure 1. The average corn area accounts for about 50% of total area, which is consistent with the fact that corn-soybean is the dominant rotation in the watershed. The fertilizer application rates in the region, which are assumed based on state and county fertilizer use information, are quite homogeneous with a mean of 148 kg/ha and a standard deviation of 4.7 kg/ha.. The 24-year (1981-2003) baseline simulation run of the calibrated SWAT model resulted in an annual average nitrate load of 15,200 tons at the watershed outlet (i.e., subwatershed 25 outlet). As in the baseline, SWAT simulations for all scenarios are performed over the same period (1981-2003) and then annual average output is used for nitrate load.

6.1. The fate and transport of nitrogen in the watershed

We present a schematic diagram of the Raccoon watershed (Figure 2) to highlight the connection and interactions among the 26 subwatersheds. The dark dots and gray circles represent the subwatersheds. The seven subwatersheds represented by the gray circles receive flow and nitrate from two or more upstream subwatersheds. Of the remaining subwatersheds, two have one upstream subwatershed and the others have no upstream subwatersheds.

The transfer coefficients capture the fate and transport of the nitrate losses, which are the foundation of this study for allocating nitrogen fertilizer reduction in the Raccoon River Watershed. Thus, we first present the transfer coefficients for all three levels of nitrogen application reduction in Figure 3. The average of transfer coefficient is about 0.23 for the 10% scenario, which means that for every 1 kg of nitrogen fertilizer application reduction about 0.23 kilograms of nitrate reduction is achieved at the watershed outlet. It is clear from Figure 3 that the transfer coefficients are almost the same for the 20 and 30% scenarios. This is good news for policy design in that watershed planners can use the same set of transfer coefficients to allocate loading reduction responsibilities regardless of the degree of reduction. By examining Figures 2 and 3, we see that there is not a clear pattern as to how the transfer coefficients vary with the location of a subwatershed. Some upstream subwatersheds have relatively high transfer coefficients (e.g., subwatershed 3), whereas some downstream subwatershedss have relatively low transfer coefficients (e.g., subwatershed 23).

To examine whether the transfer coefficients are sensitive to tillage practices, we derive transfer coefficients when no till is adopted on all cropland. The average of the new transfer coefficients is 0.26, which is slightly higher than the transfer coefficients presented in Figure 3. The underlying reason is that when no till is adopted, there is more nitrate runoff. Thus, even though the fertilizer reduction would be about as effective in terms of percentage reduction as in the case with baseline tillage practices, the reduction of loading in terms kilograms increases.

6.2. Comparison of the four principles

For downstream targeting, about half of the subwatersheds were designated as downstream subwatersheds as shown by the subwatersheds inside the gray loop in Figure 2. It is easy to see that there is no clear advantage of targeting these subwatersheds as shown by the distribution of the transfer coefficients (Figure 3). This underscores the fact that reducing nitrogen applications in those subwatersheds that lie in close proximity to the watershed outlet does not necessarily imply that that more effective reductions of nitrate loading will occur at the watershed outlet. To further illustrate this point, consider a downstream targeting scenario where the target is set the same as the corresponding equal percentage reduction scenario. Then, when the abatement cost increases fast (e.g., setting $\theta = 1$), the total cost for downstream targeting can be twice as expensive as the equal percentage reduction scenario. However, when the abatement cost is closer to being linear (e.g., setting $\theta = 5$), the cost difference between the two scenarios would be reduced dramatically to a few percentage points.

For critical area targeting, we assume that those subwatersheds that have transfer coefficients greater than the median should be managed with reduced nitrogen fertilizer applications. How much more effective this criterion is compared to the equal percentage reduction scenario also depends on how fast the cost increases. If the cost function is close to being linear, then the critical area can be more cost effective than equal allocation. Concentrating reduction responsibility in a small set of subwatersheds can be more expensive, when the abatement cost increases fast. This is because this small set of subwatersheds has to make larger fertilizer application reductions which are becoming increasingly more expensive. In our simulation, for $\theta = 5$ and $\theta = 3$, critical targeting is slightly cheaper than the equal percentage reduction scenario. However, at $\theta = 1$, critical targeting is actually 34% more expensive.

By definition, the least cost criterion has the lowest cost among the four criteria. Based on this criterion, subwatersheds with higher transfer coefficients and/or lower marginal cost will require a larger reduction in nitrogen application. Figure 4 is one illustration of the least-cost nitrogen application reduction across all the subwatersheds. Even though the curves are quite flat overall in the figure, the zigzagged pattern is obvious, which is in contrast with the equal percentage reduction scenario. As in the downstream and critical area targeting scenarios, how much cost saving potential there is depends on the characteristics of abatement cost in the watershed: the curvature of the cost function and how heterogeneous cost is between the subwatersheds. The greater the heterogeneity there is and/or the more linear the cost function is, the more potential for cost saving.

In Table 2, there are three panels, each of which presents the cost and nitrate loading reduction under the least cost scenario compared to the equal percentage reduction scenario. We will discuss the numbers in the second row of the panels in the next sub-section. From panel A we see that for $\theta = 1$ (cost increases relatively fast) the cost saving is small; only about 5% for all reduction levels. However, panel B shows that for slower rising costs the potential for cost savings can be as high as about 11.5%. Such cost saving is quite modest compared to the SO2 permit trading program which has a cost saving estimated at about 40% relative to “command

and control" regulations [*Carlson et al.*, 2000]. For panel B as well as panel C and the rest of this section, sometimes only one of the three percentage reduction scenarios is discussed or presented to avoid clutter. The results for other scenarios are similar.

Heterogeneity in cost and benefit is a main reason for cost savings from a least-cost program [*Newell and Stavins*, 2003]. Intuitively, if every subwatershed has the same cost and the same transfer coefficient, equal percentage reduction would achieve the least cost. Thus the heterogeneity of cost functions across the subwatersheds can have a large effect on the potential gains from implementing a least-cost program such as permit trading. We examined three scenarios in order to evaluate the effects of heterogeneity in the cost function. In the first one, there is no heterogeneity, that is every subwatershed has the same cost function (in mathematical terms, $\gamma_j = 1$ for all j). In the second scenario, there is some heterogeneity and γ_j is drawn from a transformed Beta distribution with a sample mean of about 3.5 and a standard deviation of about 0.8. In the last scenario, there is more heterogeneity— γ_j is also drawn from a similarly transformed Beta distribution with a sample mean about the same size but a standard deviation about 75% larger. Consistent with the literature, Panel C of Table 2 illustrates that as heterogeneity increases, the gain from trading also increases.

6.3. The implication of approximate transport function on nutrient load allocation

In the above scenarios, all allocations were designed to achieve the same nitrate loading reduction as the corresponding equal percentage reduction scenario. However, the loading reduction is estimated based on transfer coefficients which are only a simplified representation of a complex hydrologic process. Thus, for all of the scenarios, we use SWAT to simulate the nitrate load reduction resulting from a reduction of applied nitrogen fertilizer. We then compare the simulated loading reduction with the loading reduction based on transfer coefficients.

In the 20% case, an equal percentage reduction in nitrogen application in all subwatersheds reduces nitrate loading at the watershed outlet by 17.13%, based on the SWAT estimates. If only the downstream subwatersheds are required to reduce nitrogen applications, such that the nitrate loading reduction based on the transfer coefficients would be 17.13%, then the SWAT simulated nitrate reduction is 15.42% which is about 10% less than the impacts of the equal percentage reduction scenario. Similarly, if only subwatersheds in the critical areas are required to reduce nitrogen application, such that the nitrate loading reduction based on the transfer coefficients is 17.13%, then the actual simulated nitrate reduction at the outlet is only slightly (5.8%) lower than the achievement in the equal percentage reduction scenario. These differences seem to be relatively small, compared to the gaps among the scenario costs that we discussed in the previous subsection.

For the least cost scenarios, the numbers in the second row of the panels in Table 2 are the percentage differences of nitrate loading reduction at the watershed outlet between the equal percentage reduction scenario and the scenario indicated by the column names. It is clear that all of the least cost scenarios achieve lower nitrate loading reduction than the equal percentage reduction scenario. However, the differences are quite small, especially compared to the corresponding differences in costs.

7. CONCLUSIONS

Allocation of pollutant reduction responsibilities between subwatersheds at reasonable costs is a key factor in achieving water quality goals for TMDL and other watershed-based water quality initiatives. In this study, we report the results of SWAT simulation assessments focused on four common and important principles that can be used to allocate nitrogen application reductions within a watershed. We found that it can be more expensive to obtain similar nitrogen loading goals if only downstream subwatersheds or areas, considered more effective in reducing nitrogen loading, are required to reduce nitrogen fertilizer applications. We also find that, contrary to the popular belief and the experience from the sulfur permit trading program, least cost allocations do not necessarily imply significant cost savings in our study area. Large cost savings (greater than 25%) only occur when the abatement cost is sufficiently heterogeneous. This has important implications in that each watershed may have its own characteristics; what is a cost-saving plan for one watershed may actually increase costs for other watersheds when the cost of planning and implementation is taken into account.

We also tested the idea of using transfer coefficients based on SWAT simulations for allocating nutrient loading in a watershed. In our study region, we find that the loading estimates based on the coefficients are close to the model-simulated loadings with a difference of only a few percentage points in general. This result is encouraging for watershed planners and for the TMDL process. This is because watershed planning can then be based on the transfer coefficients, which indicate the relative effectiveness of a practice implemented in the subwatersheds. However, while important, generalizing this result to other watersheds needs to be carefully evaluated.

Table 1. Baseline description of Raccoon River Watershed at subwatershed level.

Subwatershed	Area (hectares)	Corn (% of total area)	Nitrogen Fertilizer (kg/ha)	Nitrate loading (1000 kg)
1	90000	50.2	148.8	1,600
2	68000	49.9	146.1	1,700
3	22000	50.3	145.6	500
4	54000	49.7	145.6	6,100
5	23000	47.7	161.1	400
6	38000	53	147.2	700
7	33000	48.2	156.3	900
8	19000	54.6	147.9	400
9	39000	50	152.4	10,600
10	42000	50.2	145.6	800
11	44000	51.2	145.6	1,200
12	35000	55.1	137.5	1,200
13	19000	47.3	152.3	600
14	18000	50	153.1	200
15	48000	49.5	147.4	11,700
16	65000	55.1	150.0	300
17	32000	49.3	148.8	300
18	30000	53.1	145.6	800
19	30000	50.9	148.0	600
20	28000	45.3	145.6	900
21	36000	48.9	145.5	300
22	37000	50.7	145.6	400
23	26000	52.1	145.6	2,800
24	17000	54.1	153.2	200
25	26000	54.4	145.6	15,200
26	21000	51.3	141.7	300

Table 2. Comparison between the approximate least-cost scenario and the equal percentage reduction scenario—sensitivity to alternative cost structures.

A. Sensitivity to different degrees of nitrogen reduction
($\theta = 1$, *same cost function for all subwatersheds*)

	10%	20%	30%
Cost difference (%)	-5.00	-5.64	-4.88
Loading difference (%)	0.00	-0.82	-0.77

B. Sensitivity to the curvature of the cost function
(*10% reduction scenario, same cost function for all subwatersheds*)

	$\theta = 1$	$\theta = 3$	$\theta = 5$
Cost difference (%)	-5.00	-8.77	-11.47
Loading difference (%)	0.00	-1.46	-2.96

C. Sensitivity to heterogeneity in the cost function
(*20% reduction scenario,* $\theta = 1$)

	No heterogeneity	Some heterogeneity	More heterogeneity
Cost difference (%)	-5.64	-10.13	-26.12
Loading difference (%)	-0.82	-3.09	-1.56

Figure 1. Location of the Raccoon River Watershed in Iowa and the delineated subwatersheds.

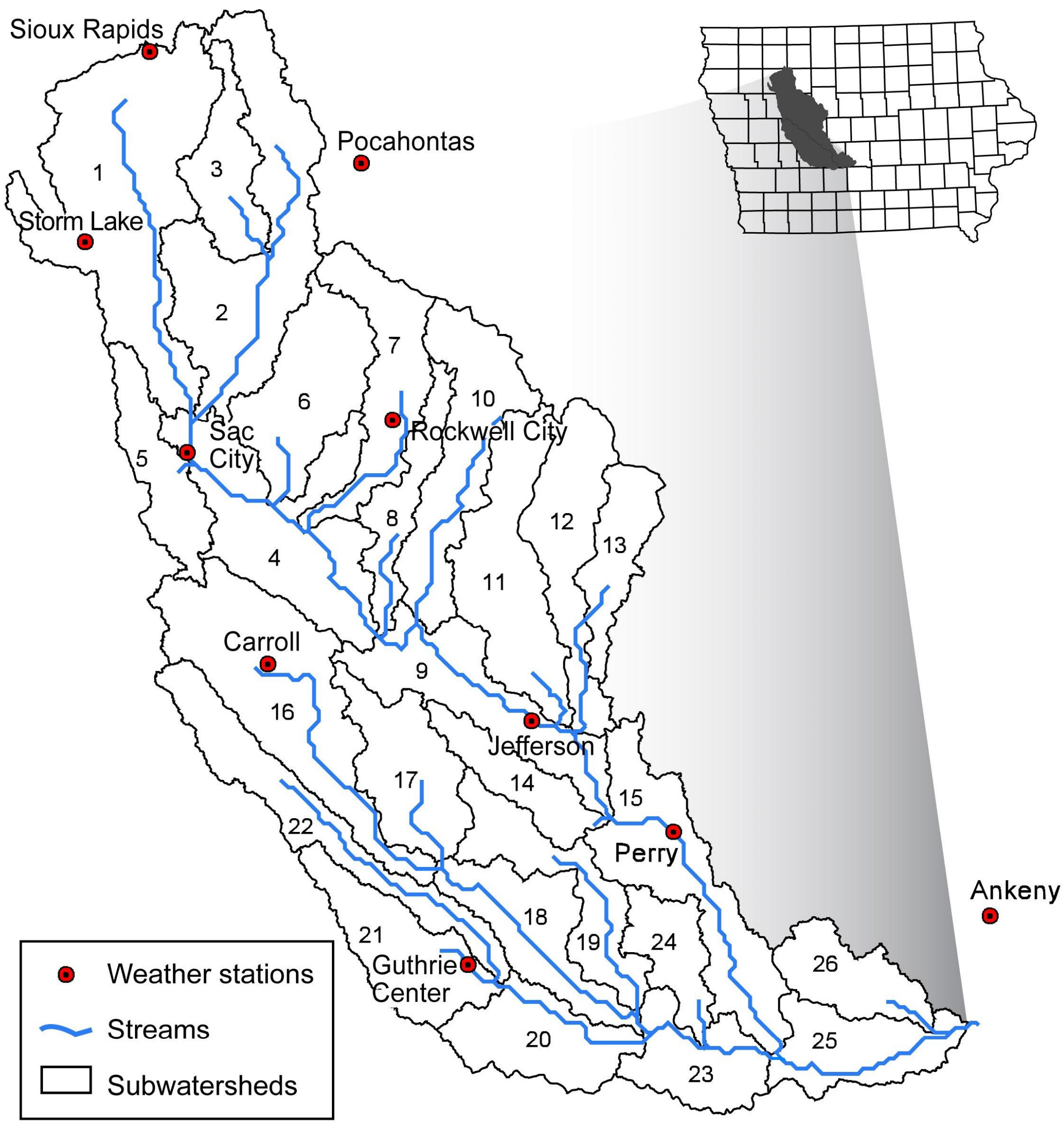


Figure 2. A schematic diagram of the 26 subwatersheds (with designated downstream subwatersheds used in analysis)

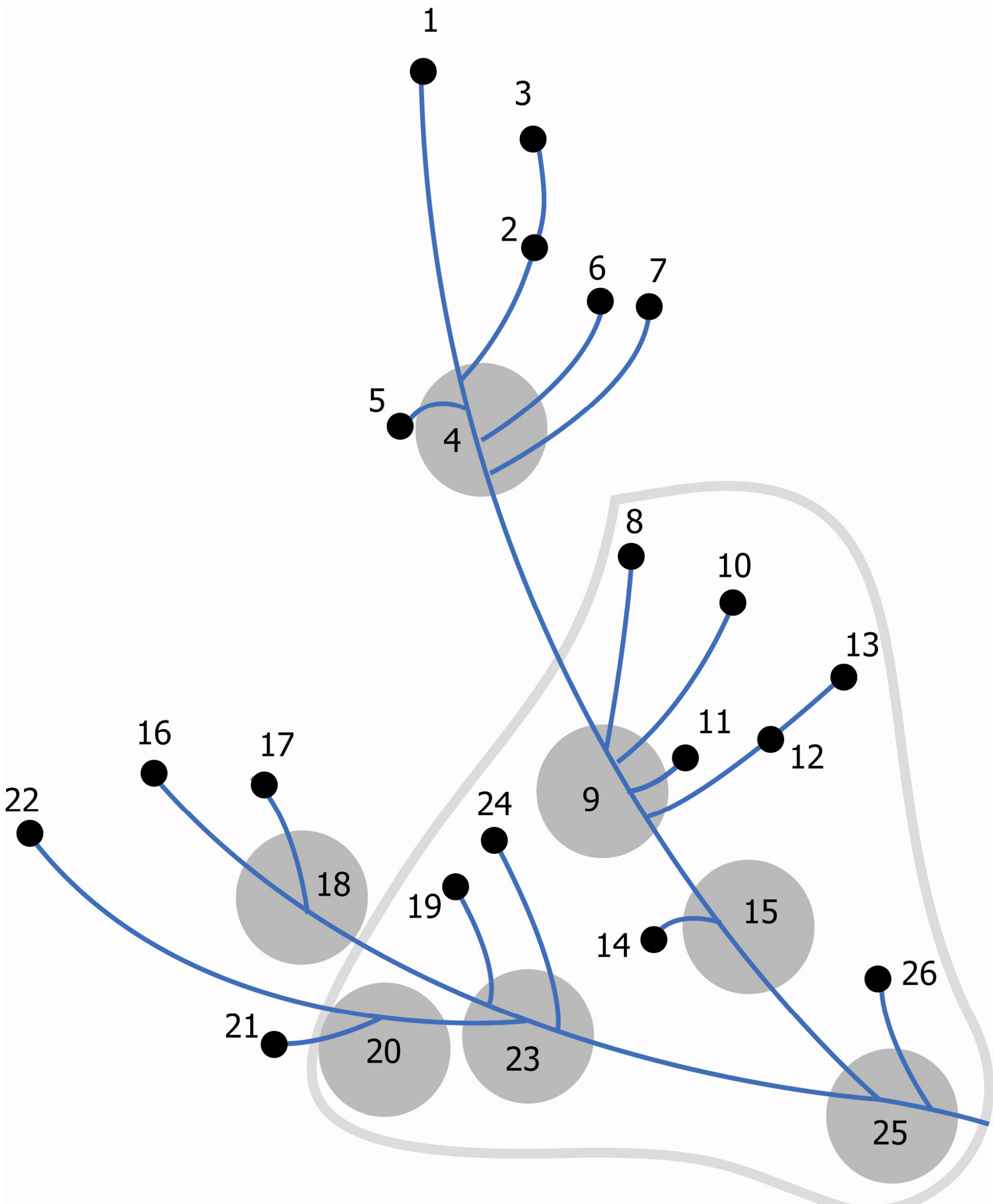


Figure 3. Transfer coefficients by the 26 subwatersheds.

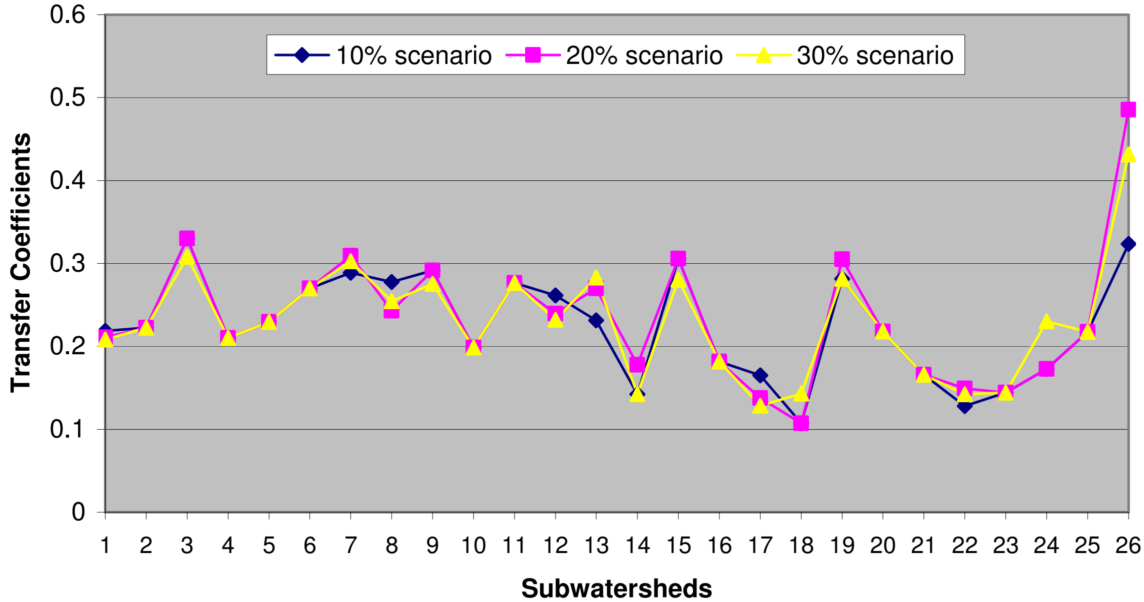
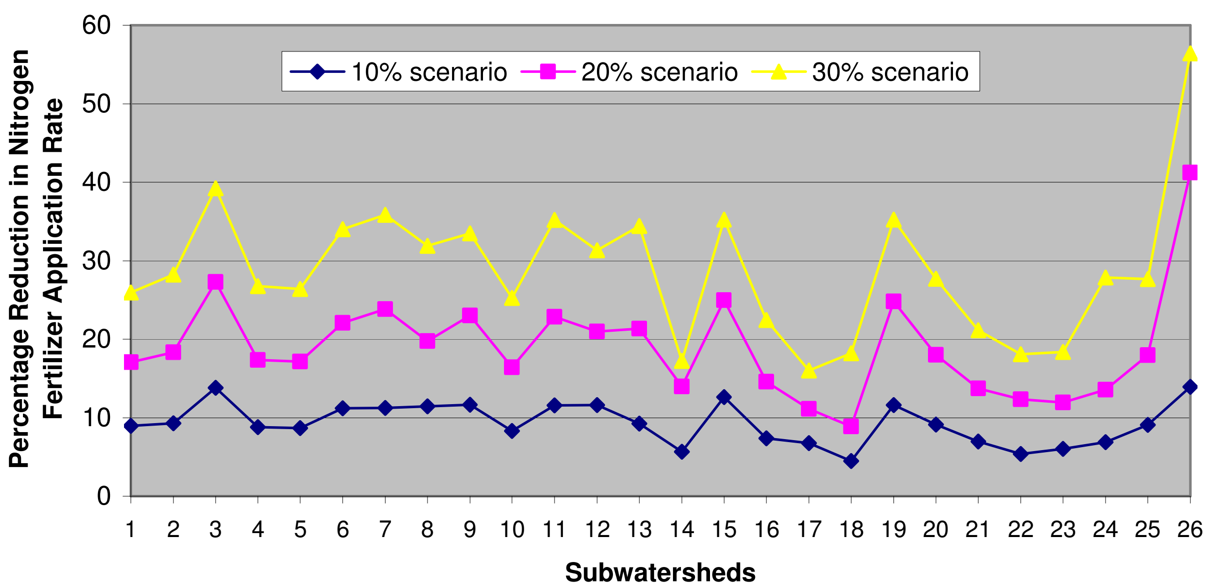


Figure 4. The distribution of nitrogen fertilizer application reduction in an approximate least-cost scenario (For $\theta = 1$ and $\gamma = 1$).



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