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A Comparison of Alternative Strategies for Cost-Effective Water Quality Management in Lakes

DANIEL BOYD KRAMER*
James Madison College and the
Department of Fisheries and Wildlife
Michigan State University
Case Hall
East Lansing, Michigan 48825-1205, USA

STEPHEN POLASKY
Department of Applied Economics
University of Minnesota
St. Paul, Minnesota 55108, USA

ANTHONY STARFIELD
Department of Ecology, Evolution, and Behavior
University of Minnesota
St. Paul, Minnesota 55108, USA

BRIAN PALIK
North Central Research Station
USDA Forest Service
Grand Rapids, Minnesota 55744
USA

LYNNE WESTPHAL
North Central Research Station
USDA Forest Service
Evanston, Illinois 60201, USA

STEPHANIE SNYDER
PAMELA JAKES
RACHEL HUDSON
North Central Research Station
USDA Forest Service
1992 Folwell Avenue
St. Paul, Minnesota 55108, USA

ERIC GUSTAFSON
North Central Research Station
USDA Forest Service
5985 Highway K
Rhinelander, Wisconsin 54501, USA

ABSTRACT / Roughly 45% of the assessed lakes in the United States are impaired for one or more reasons. Eutrophication due to excess phosphorus loading is common in many impaired lakes. Various strategies are available to lake residents for addressing declining lake water quality, including septic system upgrades and establishing riparian buffers. This study examines 25 lakes to determine whether septic upgrades or riparian buffers are a more cost-effective strategy to meet a phosphorus reduction target. We find that riparian buffers are the more cost-effective strategy in every case but one. Large transaction costs associated with the negotiation and monitoring of riparian buffers, however, may be prohibiting lake residents from implementing the most cost-effective strategy.

Thirty years after the passage of the Clean Water Act (CWA), many of its highly ambitious goals remain unmet. In 2000, 45% of assessed lakes were classified as impaired for one or more uses including swimming, drinking, and aquatic wildlife (USEPA 2000). It is generally agreed that although much progress has been made in addressing point sources of water pollution, mitigating nonpoint pollution sources remains problematic and is today the greatest threat to surface water quality.

KEY WORDS: Water quality; Cost effectiveness; Septic systems; Riparian buffers

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*Author to whom correspondence should be addressed; *email:* dbk@msu.edu

Many impaired lakes are eutrophic. Eutrophication, due to excessive nutrient inputs, is a condition characterized by increased phytoplankton, algal blooms, reduced water transparency, anoxia, the proliferation of emergent aquatic vegetation, and the loss of submerged aquatic vegetation. Biologically, eutrophication has been associated with the loss of aquatic invertebrates, reductions in desirable fish populations (Harper 1992), and the establishment of exotic species (Torke 2001). Economically, impaired waters have been associated with a reduction in property values (Gibbs 2002), tourism, and recreational values (Baylis 2002).

Excessive phosphorus inputs are a primary cause of eutrophication (Dillon and Rigler 1974). Agricultural and urban land uses are the leading contributors of phosphorus to surface waters (Daniel 1998). Failing septic systems, residential development (Eilers and others 2001), household cleaners and detergents, fertilized lawns (Dillon and others 1994; Ellis and Childs 1973), and precipitation (Shannon and Brezonik 1972) are also important sources.

Management actions to address eutrophication of lakes include fixing failing septic systems, reducing the use of phosphorus-based lawn fertilizers and household detergents, increasing shore-land vegetation buffers, and managing agricultural, urban, and forestry land use practices in lake watersheds. A compendium of enforceable state mechanisms to control nonpoint pollution shows that nearly all states have general prohibitions on nonpoint pollution discharges. Ellefson and others (1996) identified 38 states with forest management regulations pertaining to water quality. Four states have enacted legislation encouraging the establishment of riparian buffers (Cutter and others 1999). Ten states have laws regulating sedimentation by agricultural operations. In many states, septic systems are locally regulated by building codes, zoning ordinances, and licensing requirements. A smaller number of the states explicitly require the owners of septic systems to maintain them (Environmental Law Institute 1998). Finally, as of 1999, 27 states and the District of Columbia have enacted complete or partial phosphate detergent bans (Litke 1999), which is thought to reduce phosphorus loads to septic systems by 40–50% (USEPA 1993).

Despite the attention given to water quality and management activities taken to date, many lakes still do not support their designated uses. The lack of progress in addressing lake water quality problems can be attributed to at least two factors. First, and perhaps most important, phosphorus inputs that lead to eutrophication of lakes come primarily from nonpoint sources (USEPA 2000). Nonpoint sources are more difficult to monitor and regulate compared to large point sources. Second, addressing water quality problems is costly.

Much of the effort expended by regulatory agencies and academics has been to identify the relative contributions of total phosphorus to surface waters from various sources (Soranno and others 1996; Reed-Anderson and others 2000). It is clear, however, that this addresses only half of the problem because knowing the source of phosphorus inputs still does not tell us what we should do to reduce phosphorus inputs. There are often many alternative ways to achieve a given reduction in phosphorus inputs. Faced with a long list of strategies to mitigate phosphorus inputs, what are the most cost-effective strategies? Cost-effectiveness analysis can be used to determine which water quality mitigation strategies can achieve a water quality standard for the least cost. This study examines the cost effectiveness of two different strategies in addressing lake water quality: septic system upgrades and riparian stream buffers along agricultural land in the watershed. We couple a physical and an economic simulation model and illustrate this approach by applying it to 25 lakes. We also discuss various obstacles and additional costs that might prevent lake residents from implementing management strategies or might cause one strategy to be favored over the other. Finally, we provide some suggestions for actions that state and local government can take to reduce these obstacles.

Methods

The Simulation Model

In this study, we develop a mass-balance, empirical water quality model using the MATLAB programming language (Mathworks, Inc.). We use an empirical model rather than a dynamic model to incorporate assumed nonlinearities and stochasticity. Furthermore, empirical models can be solved very quickly, allowing analysis of many alternative scenarios. Finally, empirical models are more intuitive to managers and policy-makers.

Eutrophication models are often used to diagnose lake water quality problems by identifying the primary sources of nutrient inputs. Mass-balance models use phosphorus export coefficients for various input categories and outflow of phosphorus from the lake to determine the phosphorus concentration in the lake's water column. Some common assumptions, also adopted in this article, are that the lake is well mixed, that water inflow equals water outflow, and that the phosphorus concentration of outflowing water is the same as that of the lake. This basic modeling approach has been used extensively to study water quality in lakes (e.g., Vollenweider 1968; Dillon and Rigler 1975) and relies on the observation that the delivery of phosphorus to surface waters is primarily via sediment particles. In this model, we use phosphorus export coefficients to determine the contribution of land in various use categories to phosphorus inputs to the lake. This approach requires less data than other water quality models, is easy to apply, and is reasonably accurate. The approach, however, has several drawbacks. Most important, phosphorus export coefficients do not consider proximity to the water body, soil type, or topography, which are all associated with phosphorus attenuation. As a result, these models assume that phosphorus export increases linearly with the land area of the watershed (Reckhow and others 1980). It is

Table 1. Phosphorus export coefficients: low, medium, and high estimates

Watershed[a] (kg/km^2/year)	Low	Med	High
Forest (E_{FOREST})	10.0	20.0	30.0
Agriculture ($E_{AGRICULTURE}$)	50.0	175.0	300.0
Prairie ($E_{PRAIRIE}$)	10.0	15.0	20.0
Wetlands ($E_{WETLANDS}$)	5.0	6.5	8.0
Pasture ($E_{PASTURE}$)	10.0	60.0	110.0
Urban (E_{URBAN})	70.0	155.0	240.0
Lakeshore residences			
Septic systems[b] (kg/capita/year) (E_{SEPTIC})	0.5	1.2	1.8
Lawn runoff[c] (kg/km^2/year) (E_{LAWN})	10.0	50.0	90.0
Precipitation[d] (mg/liter) (E_{RAIN})		0.05	

[a]Reckhow and others (1980), Athadye and others (1983), Frink (1991), Panuska and Lillie (1995), Soranno and others (1996).
[b]Dillon and Rigler (1975), Reckhow and others (1980).
[c]Panuska and Lillie (1995), Brezonik and Stadelmann (2001), Line and others (2002).
[d]Dillon and Rigler (1975).

straightforward to modify the model to incorporate these details, but implementing these changes requires much more data than are typically available.

The model developed in this study tracks four main sources of phosphorus inputs: a) land use in the watershed, b) rainfall, c) lakeshore residents' septic systems, and d) lake lot lawns.

Land Use in the Watershed. Land use in the watershed is divided into five categories: agriculture, forest, prairie, wetlands and marsh, and urban. Each land use is associated with a phosphorus export coefficient. An export coefficient indicates the kilograms of phosphorus exported from 1 km^2 of a particular land use over a period of 1 year. The range of export coefficients we use for each of the five land use categories along with sources are included in Table 1. To calculate the phosphorus load, $\mathbf{J_i}$ from land use *i*, **the area of the particular land use, $\mathbf{A_i}$**, is multiplied by the appropriate export coefficient, E_j

$$J_i = A_i * E_i \qquad (1)$$

Total phosphorus load from land use in the watershed, $\mathbf{J_w}$, is the sum of all loads from each land use type.

$$J_W = J_{FOREST} + J_{AGRICULTURE} + J_{PRAIRIE} + J_{WETLANDS} + J_{URBAN} \qquad (2)$$

Rainfall. The phosphorus load from rainfall, $\mathbf{J_{RAIN}}$, is calculated by multiplying the export coefficient for rain, $\mathbf{E_{RAIN}}$ (Table 1), by the average annual rainfall for a particular lake, **RAIN**. Thus, we consider only rain falling directly on the lake and not storm water and runoff. However, the effects of storm water and runoff are included in the watershed land use export coefficients:

$$J_{RAIN} = RAIN * E_{RAIN} \qquad (3)$$

Septic Systems. The phosphorus load from lakeshore residents is a function of the number of permanent and seasonal lakeshore dwellings, $\mathbf{H_P}$ and $\mathbf{H_S}$, respectively, within a 200-m band around each lake. To calculate the total load from septic systems, we first determine the total number of per capita days spent on the lake for all dwellings. For seasonal residents, this is found by multiplying the number of seasonal dwellings, $\mathbf{H_S}$, by the number of per capita days spent at the lake, $\mathbf{C_S}$, which we assumed to be 205 days. This is then divided by the number of days in the year. For permanent residents, we multiplied the average number of residents per dwelling, **N**, which we assumed to be three, by the number of permanent dwellings, $\mathbf{H_P}$. The sum is the total number of per capita days spent at the lake. The phosphorus load from septic systems, $\mathbf{J_{SEPTIC}}$, is then found by multiplying this figure by the per capita phosphorus load from septic systems, $\mathbf{E_{SEPTIC}}$ (Table 1) as well as one minus the soil retention coefficient, **SR**, of soil for the lake. The soil retention coefficient is a measure of the soil's ability to trap phosphorus. A large soil retention coefficient reflects a high degree of trapping. Most soils are a combination of sand, silt, and clay, and the percentages of these various particle sizes determine the ability of soils to trap phosphorus. Using GIS data on soil composition and assuming a maximum **SR** value of 0.80, we derived **SR** for each lake using a simple linear estimation. Soils with high clay content have larger **SR** values than soils with low clay content.

$$J_{SEPTIC} = E_{SEPTIC} * (H_S * \frac{C_S}{365} + H_P * N) * (1 - SR) \qquad (4)$$

Lake Lot Lawns. The phosphorus load from lakeshore lawns, $\mathbf{J_{LAWN}}$, is found by multiplying the total number of lakeshore dwellings by the average lot size,

Table 2. Model parameters

	Variable	Description	Value
Lakeshore homes and cabins			
	LOT[a]		
		Natural lake lot size (m)	60
		Recreational development lake lot size (m)	45
		General development lake lot size (m)	30
	C_s[b]	Cabin capita days per year	205
	N[b]	Average number of residents per dwelling	3
Riparian buffer[c]			
	Y	Buffer width at 50% reduction (m)	23
	MAX	Maximum buffering reduction (%)	80
	X	Steepness of hill function at inflection	3

[a]MN Department of Natural Resources.
[b]Dillon and Rigler (1975).
[c]USEPA (2002), Karr and Schlosser (1978), McColl (1978), Omemik and others (1981), Schlosser and Karr (1981), Ogg and others (1983), Lowrance and others (1984, 2000), Peterjohn and Correll (1984), Azzaino and others (2002).

LOT (Table 2), and by the export coefficient for lawns, $\mathbf{E_{LAWN}}$ (Table 1). For lot size, we use the recommended lot width and setback for the zoning class of each lake (MNDNR). We assume that for each lake lot, the entire lot is maintained as lawn. Consequently, it is likely that the phosphorus load from lawns is overstated in this model. However, even considering this, the relative contribution from lawns is small compared to watershed land use and septic systems.

$$J_{LAWN} = (H_S + H_P) * LOT * E_{LAWN} \tag{5}$$

Total Phosphorus Input. The total phosphorus load into the lake, $\mathbf{J_{TOTAL}}$, is the sum of the load from the four input categories.

$$J_{TOTAL} = J_W + J_{RAIN} + J_{SEPTIC} + J_{LAWN} \tag{6}$$

Lake Phosphorus Concentration. To calculate the lake phosphorus concentration, we use the following equation:

$$P_{t+1} = P_t * \left(1 - SED - \frac{1}{R}\right) + \frac{J_{TOTAL}}{V} \tag{7}$$

where $\mathbf{P_t}$ is the phosphorus concentration at time **t**. The starting phosphorus concentration used is based on an average of recent measurements from the U.S. EPA's Storet database. **SED** is the proportion of phosphorus lost to the lake sediments each year. We derive values for **SED** using an estimation from Canfield and others (1982) for natural lakes using Equation 8, where **L** is the annual areal phosphorus loading, **z** is the mean lake depth, **TP** is the concentration of total phosphorus in the lake using current measurements, and **R** is the water residence time.

$$SED = \frac{L}{z * TP} - \frac{1}{R} \tag{8}$$

The Minnesota Pollution Control Agency (MPCA) provided values for **R** and **V**, the volume of the lake basin. Parameter values are given in Tables 2 and 3. By setting $\mathbf{P_{t+1}}$ equal to $\mathbf{P_t}$ in Equation 7, it is straightforward to derive Vollenweider's equation (1968) for the steady-state phosphorus concentration, $\mathbf{P_{SS}}$, of each lake (Equation 9).

$$P_{SS} = \frac{J_{TOTAL}}{V(SED + \frac{1}{R})} \tag{9}$$

Although internal loading from lake sediments can be a significant source of phosphorus, internal loadings are ultimately due to external loading in the past. We do not consider internal phosphorus loadings in this model. Neglecting internal loading will not favor either of the strategies considered in this article. Rather, the recovery time from eutrophication would likely be much longer if there is significant internal loading.

Stochasticity. There are two sources of stochasticity that we incorporate in the model First, there is parameter uncertainty; a wide range of phosphorus export coefficients are cited in the existing literature (Table 1). Second, there is natural variability due to weather. To incorporate stochasticity, we adopt a Monte Carlo approach. For each run of the model, the export coefficients for each input class are randomly drawn from a uniform distribution defined over a range of low to high estimates as indicated in Table 1. Annual rainfall amounts are drawn from a normal distribution based on precipitation averages for the county where the lake is located (Minnesota DNR). We ran the model for a period of 50 years. These 50-year runs were then replicated 100 times.

Table 3. Lake characteristics and physical parameters

Lake name	Ecoregion	Watershed area (km^2)	Lake area (km^2)	Watershed to lake area ratio	Mean depth (m)	Water residence time (yr)	Permanent homes	Seasonal Homes	Agricultural riparian stream length (m)	Soil retention coefficient	Water land use proportions: Agriculture	Forest	Prairie	Wetlands and marsh	Urban
Fleming	NLF	12.4	1.2	10.4	2.7	0.5	54	125	624.3	0.30	0.49	0.07	0.32	0.12	0.00
Little Rock	NCHF	55.6	5.2	10.7	2.4	0.8	325	7	7544.6	0.24	0.21	0.42	0.23	0.16	0.00
Smith	NCHF	48.3	2.7	18.1	4.6	1.0	48	27	16817.7	0.60	0.03	0.52	0.36	0.08	0.00
Redrock	NGP	23.3	3.0	7.8	1.6	1.4	40	62	1929.1	0.10	0.02	0.47	0.28	0.23	0.00
Pelican	NCHF	46.7	15.0	3.1	3.0	4.1	135	177	892.5	0.34	0.06	0.32	0.23	0.39	0.00
Third Crow Wing	NLF	6.2	2.7	2.3	3.7	4.5	66	219	0.0	0.26	0.41	0.05	0.10	0.45	0.00
Norway	NCHF	24.0	9.1	2.6	3.0	3.3	133	157	1408.4	0.33	0.06	0.30	0.20	0.43	0.00
Jennie	NCHF	19.8	4.3	4.6	2.2	1.4	103	36	284.2	0.35	0.04	0.61	0.06	0.29	0.00
Spring	NCHF	5.1	0.8	6.4	3.6	1.8	243	3	0.0	0.35	0.01	0.39	0.20	0.17	0.23
Washington	NCHF	25.0	9.8	2.5	2.2	2.3	322	175	329.8	0.60	0.01	0.40	0.17	0.42	0.01
Dunns	NCHF	2.0	0.6	3.2	2.8	2,4	72	16	20.1	0.35	0.09	0.41	0.16	0.34	0.00
Ripley	NCHF	12.5	2.3	5.3	3.0	1.4	25	1	4472.9	0.27	0.04	0.50	0.06	0.32	0.09
Grove	NCHF	23.9	1.5	15.8	2.7	1.1	42	42	4464.2	0.70	0.02	0.48	0.40	0.10	0.00
Leven	NCHF	3.4	1.1	3.0	3.8	5.0	40	11	0.0	0.32	0.03	0.22	0.33	0.42	0.00
Gilchrist	NCHF	48.6	1.4	35.7	3.0	0.4	22	68	3572.1	0.70	0.05	0.29	0.54	0.12	0.00
Reno	NCHF	36.5	11.6	3.1	2.7	3.4	104	46	3038.6	0.45	0.00	0.46	0.11	0.42	0.00
Pelican	NCHF	46.2	2.0	23.0	3.9	0.8	107	28	6762.7	0.80	0.05	0.49	0.36	0.10	0.00
Emily	NCHF	52.1	9.1	5.7	0.9	0.7	22	3	10344.7	0.36	0.00	0.63	0.16	0.20	0.00
Wild Rice	NLF	61.3	9.0	6.8	1.8	0.6	138	1	340.2	0.21	0.49	0.01	0.25	0.17	0.08
Horseshoe	NCHF	6.7	2.5	2.7	4.8	3.5	229	188	62.5	0.25	0.07	0.23	0.28	0.42	0.00
Perkins	NGP	10.3	2.1	5.0	2.1	2.2	23	17	6159,4	0.32	0.01	0.61	0.14	0.23	0.00
Mary	NCHF	5.0	0.5	10.3	7.6	2.0	19	6	247.3	0.60	0.07	0.46	0.33	0.13	0.00
Moose	NCHF	4.0	0.5	7.9	4.0	1.5	17	22	84.1	0.35	0.12	0.44	0.29	0.16	0.00
Long	NCHF	21.0	1.6	13.3	9.8	1.7	135	135	1944.7	0.60	0.13	0.38	0.34	0.15	0.00
Ramsey	NCHF	12.5	1.2	10.5	5.8	2.3	104	11	2872.8	0.35	0.03	0.53	0.22	0.15	0.07
Averages		24.5	4.0	8.8	3.5	2.0	102.7	63.3	2968.7	0.40	0.10	0.39	0.24	0.25	0.02

We use the results of the Monte Carlo approach to calculate the probability of lake phosphorus concentrations surpassing targeted thresholds. Heiskary and Walker (1988) developed a phosphorus criterion in Minnesota based on lakes' natural susceptibility to eutrophication and lake users' perceptions of water quality. They found that lake users in different ecoregions of the state had different expectations for lake water clarity. Users in the northern ecoregions had higher expectations than those in the southern ecoregions. Based on these results, we set a threshold of 30 μg/l phosphorus concentrations for lakes in the Northern Lakes and Forests (NLF) ecoregion of Northern Minnesota, 40 μg/l in the North Central Hardwood Forests (NCHF) ecoregion, and 90 μg/l in the Northern Glaciated Plains (NGP) ecoregion of southern Minnesota. Lake users are typically more concerned about water clarity than phosphorus concentrations, but the two are related. Generally, halving the phosphorus concentration will double water clarity. For example, a eutrophic lake with a phosphorus concentration of 48 μg/L has a secchi depth of approximately 1 m. Reducing the phosphorus concentration to 24 μg/L will improve water clarity by 1 m (Carlson 1997).

Policy

The model incorporates two policy variables to allow evaluation of the cost effectiveness of two strategies aimed at improving lake water quality. The first policy variable, **SEPTIC**, is the percentage of lakeshore homes and cabins upgrading their septic systems. We assume that no phosphorus is exported from an upgraded septic system. Studies of well-maintained and newly installed septic systems have documented a range of 30–90% removal of phosphorus (USEPA 1993, 2002;). The assumption of complete removal makes the strategy of septic upgrades more effective than might be observed. It is also important to note that this study looked solely at the problem of phosphorus loading. Septic systems might be the source of other problems including nitrogen, chlorides, bacteria and viruses, and organic chemicals. Thus, there may be other reasons to consider septic system improvements.

The second policy variable is the riparian buffer width, **B**. Riparian buffers are a recommended best management practice (BMP) and have been shown to reduce nutrient inputs to streams (Lowrance and others 1997). The riparian buffers temper the phosphorus load to lakes according to the following equation:

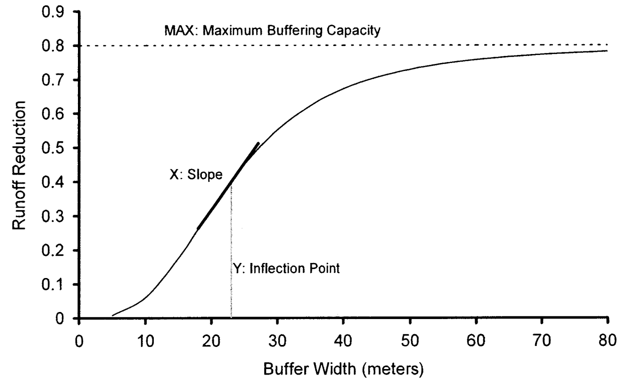

Figure 1. The relationship between riparian buffer width and the proportion of phosphorus runoff reduction.

$$E_{BUFFER} = E_{LU} * \left(1 - \frac{MAX * B^X}{Y^X + B^X}\right) \quad (10)$$

where $\mathbf{E_{BUFFER}}$ is the new export coefficient for the buffered land use in the watershed, $\mathbf{E_{LU}}$ is the original export coefficient for the land use **LU, MAX** is a parameter that sets the maximum buffering capacity regardless of buffer width, **B** is the buffer width, **Y** is the buffer width at which the export coefficient is reduced by half, and **X** is a parameter governing the steepness of the function, $\frac{MAX*B^X}{Y^X+B^X}$ at the inflection point (Figure 1). The use of this function assumes that as the width of the riparian buffer increases from zero, phosphorus attenuation changes nonlinearly. At small widths, buffering capacity is small. At intermediate widths, however, the effectiveness of the buffer increases rapidly. Finally, at greater riparian buffer widths, little attenuation capacity is added by increasing the buffer width. Studies examining the effect of varying riparian buffer widths on nutrient retention suggest that the relationship is nonlinear (Dillaha and others 1989; Desbonnet 1995). Empirical work has documented a range of phosphorus load reductions between 27% and 89% among a variety of sites and buffer types (Peterjohn and Correll 1984; USEPA 2002). These studies provide no definitive answer as to whether grass or forest provides greater buffering capacity due to the confounding influences of study design and site characteristics. Based on the existing literature and assuming no particular buffer type, we use a maximum buffering capacity of 80%. We also assume that at a buffer width of 23 m, there is a reduction of 50% of phosphorus exports, although there is little empirical work to support any precise value. We consider buffer widths of up to 61 m because there is little additional attenuation beyond this width. The response function was chosen because of its malleability. By changing the parameter values, we can alter the function's shape, which we do in our sensitivity analyses.

We choose the policy variables, **SEPTIC** and **B**, to meet phosphorus thresholds at a 10% risk level, meaning that lake phosphorus concentrations would not exceed the threshold value more than 5 years in the 50-year period on average.

Lake Selection

Lake water quality is affected by many physical factors including fetch, area, volume, depth, and shore length (Meeuwig and Peters 1996). In this article, we study 25 lakes that span a broad continuum for several physical parameters (Table 3). Among these lakes, watershed area ranges from 2 to 61 km^2, lake area from 0.5 to 15 km^2, lake residence time (i.e., the time it would take to completely flush the lake) from 0.4 year to 5 years, and housing density from 1 to 46 residences per kilometer of shoreline. Land use in the watershed also varies greatly among the lakes. The percentage of forested land cover ranges from 0 to 49%, agriculture from 1 to 74%, and urban land use from 0 to 23%. We limit the selection of lakes to those in Minnesota in order to assure similar standards of data definition and data gathering. Each lake currently has high phosphorus concentrations compared to other lakes in the same ecoregion. Finally, sufficient data were available for each lake selected to parameterize the simulation model.

The representative lakes come from three of Minnesota's seven ecoregions: NLF, NCHF, and NGP. These three ecoregions contain 96% of all lakes in Minnesota. Ecoregions are areas that share similar attributes in terms of land uses, soil type, geology, vegetation, and biological community structure. These attributes can greatly affect lake water quality. Lakes in the northern portion of the state tend to be smaller and deeper than those in the central and south portions. Soils in the south and central portions of the state are deeper and more nutrient rich than those in the north and therefore are more likely to cause sedimentation problems in lakes. Soils in the north are usually thin and rocky and unable to buffer acidic inputs but are less likely to cause sedimentation problems. Finally, land use in the north is dominated by forest management, whereas agriculture is more common in the central and southern ecoregions.

The Economics of Water Quality Improvement

Degradation of water quality has its costs, but so do projects to improve water quality. Several studies have examined the cost effectiveness of different strategies in addressing water quality problems from an agricultural land-owner's perspective including various BMPs (Ogg and others 1983) and riparian buffers (Stonehouse 1999; Azzaino and others 2002).

In this article, we compare the cost effectiveness of septic system upgrades and riparian buffers because these two strategies seem to be the most commonly considered strategies from the perspective of lake residents (Kramer 2005). Many of the agricultural-based strategies considered in the existing literature are simply not feasible from the perspective of lake residents, primarily because of problems associated with the costs of monitoring and enforcement.

Costs for implementing septic system upgrades and riparian buffers vary widely (Gustafson and others 2002). Depending on type, the installation costs for a new septic system range from $2,000 to $15,000. Annual maintenance costs range from $30 to $3,000 (Table 4). We use a low-cost estimate of $3,000 for installation and ignored the annual maintenance costs because it is assumed that lake residents currently face some annual maintenance costs.

We estimate the opportunity cost of setting aside land for riparian buffers by determining the annual returns for land in agriculture. Data on agricultural productivity, production costs, and crop prices allow the estimation of the average value of 1 acre of farmland used for corn, soybeans, or wheat, the predominant crops grown in Minnesota (Table 5). For the simulations, we use an annual opportunity cost of $110 per acre. We also assume that the initial costs for installing a riparian buffer (e.g., site preparation, planting, etc.) are $400 per acre and annual maintenance costs are $20 per acre. This compares with the U.S. Environmental Protection Agency's (EPA's) estimated costs of between $358 to $396 per acre for establishment and $20 per acre for annual maintenance costs (USEPA 1996). Using a discount rate of 4%, we calculate a net present value to reflect the ongoing annual payments to farmers.

Using the Geographic Information System software ArcMap (ESRI), we identify the five land uses in lake watersheds, determine the locations of privately held land in agriculture, determine soil types, and estimate the stream-side lengths of private land adjacent to farmed lands. We use land-use data from the Upper Midwest Gap Analysis Program of the U.S. Geological Survey. Lake watershed delineations (Minnesota DNR) are overlaid to determine the acreage and percentage of each land use in the lake watershed. Using these data, we determine the acreage of the riparian buffers and thus the costs of foregoing agricultural operations on private land.

Table 4. Estimated costs *of* septic system upgrade

Type of upgrade	Installations costs ($)	Maintenance costs ($)/yr	25-year cost ($)
Tank with trench	3,000–6,000	30–200	6,300
Tank with mound	4,000–12,000	80–500	12,900
Constructed wetland	5,000–15,000	50–550	13,500
Sand or peat filter	5,000–15,000	500–1,000	22,000
Drip dispersal	7,000–10,000	600–1,700	31,500
Aerobic tank	4,000–7,500	600–1,700	28,750
Holding tank	2,000–3,000	2,000–3,000	70,000
Municipal sewer	4,000–10,000	200–400	13,000

Source: University of Minnesota Extension Service.

Table 5. Agricultural land values

	Productivity (bushels/acre)	Cost/acre	Dollars/bushel	Value/acre/year
Corn[a]	151	$152.00	$1.80	$119.80
Soybeans[a]	46	$78.34	$4.16	$113.02
Wheat[a]	55.8	$83.10	$2.45	$53.61
Conservation Reserve Program[b]				$111.11

[a]USDA ERS Heartland Agricultural Region 2001.
[b]USDA/FSA/EPAS/current enrollment.

Model Verification and Validation

Model verification refers to testing the proper functioning of the software programming. Model validation refers to investigating the correspondence between model outcomes and empirical data. We verified the operation of the model by two means. First, we progressively debugged the model throughout the model building process as we added more complexity. Second, we compared model outputs under a wide range of parameter values to the output of a simple spreadsheet model to assure the model functioned as intended. To validate the model, we ran the model for a 50-year time period without modifying the policy variables. Comparing final phosphorus concentrations predicted by the model with the most recent data from STORET (Table 6), we then calibrated the model by making adjustments to the lake water residence time so that the predicted and actual phosphorus concentrations were similar. Adjusting the water residence time has the benefit of not favoring any one policy strategy. Table 6 indicates that on average, the model underestimated the phosphorus concentration of the 25 lakes by 28% before calibration. Removing the one large outlier of Horseshoe Lake, the difference falls to 16%.

Three separate simulations were run. The first used the Monte Carlo approach explained previously. The second and third set of simulations differed by fixing the export coefficient for septic systems to the high and low estimates, respectively, while using the Monte Carlo approach for all other export coefficients.

Results

Simulation results are shown in Table 7. Results from the first simulation run show that of the 25 lakes examined, 19 met their phosphorus criterion within a risk level of 10% using either one or both of the phosphorus reduction strategies. Of these 19, 16 lakes met their phosphorus criteria by agricultural buffers alone at a total cost of between $8,476 and $1,418,027 using riparian buffer widths between 15 and 61 m. On the other hand, 7 lakes met their phosphorus criterion using only septic system upgrades to 50% to 90% of the homes in the lake basin at a cost of between $62,500 and $1,128,400. Four lakes (Norway, Washington, Mary, and Long) met their phosphorus target by using either of the strategies. Comparing strategy costs for these four lakes, in each case it is far less expensive to employ riparian buffers. The cost savings from using riparian buffers versus septic system upgrades are substantial. The cost ratios, the total cost of septic system upgrades divided by the cost of riparian buffers, range from 7.0 to 47.3. For six lakes, it was impossible to meet their phosphorus criteria using either of the two strategies alone. Although not considered here, it is likely that a combination of the two strategies would be effective for many of these lakes.

For the second set of simulations, we fixed the phosphorus export coefficient for septic systems at the highest level found in the literature, meaning we assumed that more phosphorus from septic systems enters the lake. In so doing, we observed that 19 of the 25 lakes met their phosphorus criterion using one of the

Table 6. Phosphorus thresholds and a comparison of empirical and modeling phosphorus estimates

Lake name	Phosphorus input proportions: Watershed	Septic	Lawn	Rain	Phosphorus threshold (μg/l)	Phosphorus empirical value (μg/l)	Phosphorus model estimate (μg/l)	Percentage difference between empirical and estimate values
Fleming	0.49	0.38	0.04	0.08	30	50	49	2%
Little Rock	0.76	0.19	0.01	0.04	40	179	94	90%
Smith	0.92	0.04	0.00	0.04	40	69	68	1%
Redrock	0.88	0.08	0.01	0.04	90	113	97	17%
Pelican	0.70	0.14	0.01	0.15	40	57	55	4%
Third Crow Wing	0.19	0.56	0.07	0.18	30	43	40	7%
Norway	0.57	0.22	0.02	0.19	40	48	42	13%
Jennie	0.78	0.14	0.01	0.08	40	80	78	3%
Spring	0.38	0.58	0.02	0.03	40	207	116	79%
Washington	0.47	0.30	0.01	0.21	40	45	46	−3%
Dunns	0.34	0.56	0.03	0.07	40	128	82	57%
Ripley	0.87	0.06	0.00	0.07	40	78	76	3%
Grove	0.86	0.07	0.01	0.06	40	52	56	−8%
Leven	0.43	0.39	0.03	0.14	40	65	47	39%
Gilchrist	0.91	0.04	0.00	0.04	40	60	55	9%
Reno	0.74	0.09	0.01	0.16	40	46	51	−9%
Pelican	0.85	0.08	0.00	0.06	40	47	48	−2%
Emily	0.92	0.01	0.00	0.07	40	122	121	1%
Wild Rice	0.66	0.18	0.01	0.14	30	66	40	65%
Horseshoe	0.23	0.64	0.05	0.08	40	282	70	304%
Perkins	0.86	0.07	0.01	0.07	90	99	96	3%
Mary	0.79	0.12	0.01	0.07	40	42	46	−8%
Moose	0.76	0.16	0.02	0.06	40	75	69	8%
Long	0.68	0.24	0.02	0.06	40	47	49	−4%
Ramsey	0.75	0.20	0.01	0.04	40	79	67	19%

two strategies, 15 using riparian buffers, 10 using septic improvements, and 6 using either strategy. Among these last six, the cost ratios ranged from 0.5 to 33.1. Thus, the cost ratios fell for all lakes, but again riparian buffers are more cost effective. Perkins Lake was the only lake where septic system improvements are a more cost-effective approach to solving phosphorus loading problems. The result for Perkins Lake is probably due to the large network of streams in this lake's watershed, which makes riparian buffers extremely expensive.

We also fixed the phosphorus export coefficient for septic systems to the lowest value found in the literature for the third round of simulations. Riparian buffers proved effective for 17 lakes and septic improvements for only 5 lakes. The range of cost ratios to meet the phosphorus threshold for the 2 lakes where either option was available, Washington and Mary, are 88.1 and 11.4, respectively. Riparian buffers are also much more cost effective in this scenario.

Sensitivity Analyses

Lastly, using the first simulation scenario, we ran simulations on each of the four lakes where both septic upgrades and riparian buffers are viable options to determine the values for various economic parameters (i.e., agricultural opportunity cost, discount rate, buffer installation costs, and septic upgrade costs) at which there would be cost parity (Table 8). For example, for Norway Lake, the annual opportunity costs of cropland would have to be $1,810 per acre in order for the costs of meeting the phosphorus threshold to be equivalent for riparian buffers and septic upgrades. Conversely, the cost of a septic system upgrade needs to be $280 per household for cost parity. Each entry in Table 8 falls far outside the bounds of reasonable estimates for each of the four economic parameters.

Similarly, we tested the sensitivity of the model's output on the parameters governing the shape of the phosphorus attenuation function for riparian buffers (Equation 10). As **MAX** decreases, **Y** increases and **X** decreases, and riparian buffers become less effective. However, even significant changes to each of these parameters do not make septic upgrades more cost effective on the four lakes where both strategies are available in the first set of simulations. In fact, only by changing these parameters to make buffers ineffective altogether do septic upgrades become the desired policy choice. For example, for the four lakes, only if

Table 7. Simulation results

	Simulation 1					Simulation 2					Simulation 3				
	Riparian buffers		Septic			Riparian buffers		Septic			Riparian buffers		Septic		
Lake name	m	$	%	$	Cost ratio	m	$	%	$	Cost ratio	M	$	%	$	Cost ratio
Fleming								90%	485,800						
Little Rock															
Smith	27	1,037,403				30	1,152,670				27	1,037,403			
Redrock	18	79,332				21	92,553				18	79,332			
Pelican	27	55,052				30	61,169				27	55.052			
Third Crow Wing			60%	515,500				50%	430,000				80%	686,500	
Norway	21	67,573	80%	698,500	10.3	24	77,227	60%	524,500	6.8	21	67,573			
Jennie	43	27,273				52	33,117				37	23,377			
Spring															
Washington	21	15,824	50%	748,000	47.3	24	18,084	40%	598,900	33.1	18	13,563	80%	1,195,300	88.1
Dunns															
Ripley	43	429,192				46	459,848				40	398,535			
Grove	21	214,179				21	214,179				21	214,179			
Leven			80%	124,900				70%	109,600				90%	140,200	
Gilchrist	27	220,345				27	220,345				27	220,345			
Reno	18	124,959				18	124,959	100%	452,500	3.6	18	124,959			
Pelican	18	278,104				18	278,104				18	278,104			
Emily	61	1,418,027									61	1,418,027			
Wild Rice															
Horseshoe			90%	1,128,400				80%	1,003,300				90%	1,128,400	
Perkins	15	211,081				15	211,081	90%	110,500	0.5	15	211,081			
Mary	15	8,476	80%	62,500	7.4	15	8,476	50%	40,000	4.7	12	6,781	100%	77,500	11.4
Moose	46	8,644				58	10,949				37	6.915			
Long	21	93,302	80%	650,500	7.0	24	106,631	60%	488,500	4.6	18	79,973			
Ramsey											42	275,655			

Simulation 1 uses a Monte Carlo sampling method across a uniform distribution of export coefficients. Simulation 2 fixes the export coefficient for septic systems to the highest value in the range of estimates while using a Monte Carlo sampling method for the other export coefficients. Simulation 3 fixes the export coefficient for septic systems to the lowest value in the range of estimates.

Table 8. Parameter values necessary to meet cost parity

Lake	Agriculture opportunity cost per acre per year	Discount rate	Riparian buffer installation costs per acre	Septic upgrade costs
Norway	$1,810	0.00365	$42,800	$280
Washington	$8,540	0.00079	$211,000	$53
Mary	$1,270	0.00510	$29,500	$299
Long	$1,197	0.00540	$27,400	$421

MAX, the maximum buffering capacity, is changed from 80% to less than 32% do buffers become ineffective on these lakes. Similarly, **Y**, the buffer width corresponding to a 50% reduction, must be increased from 23 m to greater than 76 m to make buffers ineffectual.

Mixed Strategies

Thus far, we have discussed only whether the strategy of riparian buffers or septic upgrades, considered separately, are more cost effective. It is possible, however, that a mixture of these two strategies is optimal because the marginal costs of reducing phosphorus loads with increasing buffer widths vary nonlinearly with buffer width. Marginal costs, in this case, are defined as the cost of improving lake phosphorus concentrations by 1 µg per liter. The marginal costs are relatively high at first because of the ineffectiveness of narrow buffers. Costs drop, however, at an intermediate buffer width corresponding to the increasing buffering capacity at these widths. Finally, marginal costs rise again as the maximum buffering capacity is approached. The marginal costs of septic system upgrades for each lake, however, are assumed constant because we have assumed an average upgrade cost per dwelling. In reality, upgrade costs might vary among different properties.

Figure 2 graphs the marginal costs of each strategy for the only two lakes where a mixed strategy is feasible, Emily and Ripley. The solid line represents the marginal costs for riparian buffers and the horizontal dashed line the marginal costs for septic upgrades. The vertical line designates the width of the riparian buffer necessary to achieve the lake's phosphorus criterion. Total costs are the area under each marginal cost curve. Thus, a mixed strategy is advisable if the marginal cost curves for buffers and septics cross before reaching the effective buffer width. Interpreting the graph for Emily Lake, riparian buffers should be used up to a width of 49 m. Then, to meet the phosphorus threshold, we should use septic upgrades because it is less costly to do so. For Ripley Lake, riparian buffers should be used up to a width of 37 m and then septic upgrades thereafter.

Discussion

The policy choice of riparian buffers is a more viable option for a greater number of lakes than septic system upgrades. In each set of simulations, the number of lakes where it was possible to meet a phosphorus threshold using only riparian buffers was greater than the number of lakes using only septic upgrades. In fact, of the lakes meeting their threshold by only septic upgrades (i.e., 5 to 12 lakes depending on the simulation model), 4 of these had very little agricultural land in their watershed (Table 3). Of the remaining lakes, it is clear that greater phosphorus loading reductions are possible by employing riparian buffers.

In the case of lakes where both policy options can meet phosphorus concentration thresholds, upgrading septic systems was generally more expensive, in some cases much more expensive, than establishing riparian buffers. In only one lake was it less expensive to use septic system upgrades. We believe these results are quite robust. First, sensitivity analyses done with reasonable parameter values did not change these results. Second, many of the parameter values used tended to favor making septic upgrades. For example, the assumed cost of septic upgrades was $3,000 but could range as high as $30,000 per system. Lift stations and mound systems are more likely to be used on lake lots because of high water tables, and these systems tend to be more expensive. The opportunity costs for agricultural buffer acreage were in the mid to high range of estimates (Table 5). Third, the model did not identify particular stream segments and adjacent land areas that might be most effective in mitigating phosphorus runoff. If instead we included more heterogeneity in soil types and topography, for example, targeting particular land parcels might make buffers more cost effective compared to requiring buffers on all of the private agricultural lands bordering every stream in the watershed (Gburek and others 2000). Fourth, we assumed that septic upgrades effectively reduced phosphorus leakage by 100%. Not assuming a complete reduction would make septic upgrades less cost effective. Finally, no benefits were assumed for riparian

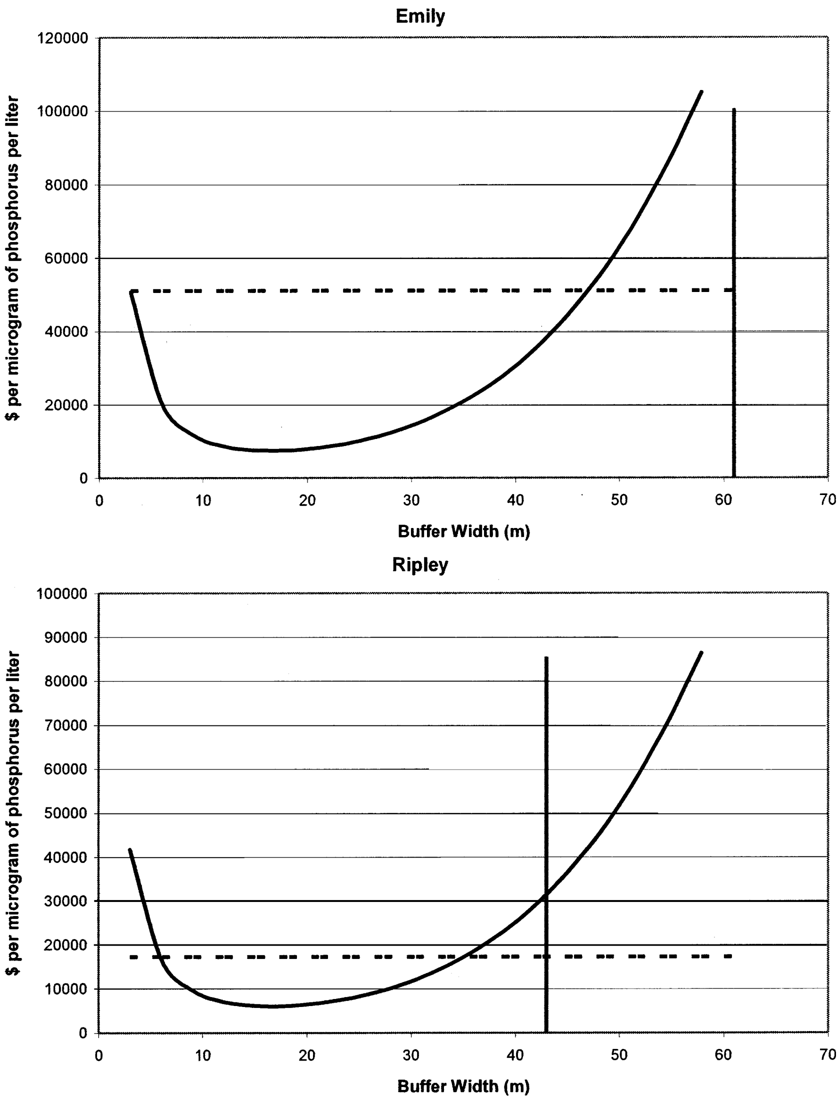

Figure 2. Marginal cost curves for riparian buffers (solid line) and septic upgrades (dashed line). The vertical line designates the width of the riparian buffer necessary to achieve the lake's phosphorous criterion. A mixed strategy is appropriate for each lake.

buffers when in actuality the harvest of grass, hay and trees, erosion and flood control, habitat, shade, decreased water temperatures, and recreational opportunities might be additional benefits of establishing riparian buffers (Schultz and others 1995).

The simple water quality and economic model used here also demonstrates that meeting the phosphorus threshold at minimum cost requires lake-specific details. For example, although riparian buffers are usually more cost effective than septic upgrades, both policy options are not always available. Also, Perkins Lake demonstrated that it is possible for septic upgrades to be more cost effective. Even for lakes amenable to using either policy option to meet the phosphorus threshold, there were orders of magnitude differences between the cost ratios. Finally, a mixed strategy is most cost effective for two lakes.

The model presented here could be expanded to include other strategies to improve water quality. Other agricultural practices that could be considered include no-till, conservation tilling, contour farming, better nutrient management, crop rotations, strip cropping, cover cropping, and constructed wetlands (Boody and others 2005). Active forestry operations can adversely affect lake water quality by increasing sedimentation and therefore alternative forestry BMPs might be included in a more comprehensive cost-effectiveness analysis. Finally, programs aimed at the reduction of phosphorus detergents, which eventually enter through septic systems, might also be included.

The model would also be improved by input of better primary data. More detail on soil conditions, slopes, and topography would improve our cost estimates for riparian buffers. The targeting of some of the more problematic riparian areas might increase the cost effectiveness of buffers. Second, many of the model parameters were extracted from the literature and better estimates might change the results. For example, increasing the per capita days of lake residents or the number of residents per dwelling would increase the load from septic systems and make septic upgrades more cost effective. Third, some costs were omitted. The main obstacle faced by lake residents to implementing a cost-effective, riparian buffer strategy is the likely presence of transaction costs in addition to the engineering costs discussed thus far. Transaction costs can sometimes be substantial. Transaction costs include the costs associated with group organization, negotiation, monitoring, and enforcement of collective action activities (Ostrom 1990). It is likely that lake collective action groups face lower transaction costs with septic upgrades than with riparian buffers. Transaction costs associated with septic upgrades are probably lower because group homogeneity is likely quite high among lake residents, allowing negotiation costs to be low. Furthermore, monitoring costs are very low because permits would be required to make septic investments, and enforcement is not an issue due to a lengthy bureaucratic paper trail. On the other hand, transaction costs associated with the creation of riparian buffers are probably much higher. Negotiation costs are perhaps the most significant transaction cost because farmers and lake residents have infrequent contact and are collectively a more heterogeneous group than lake residents alone. Also, we examine only the strategy of establishing riparian buffers on private agricultural land. Negotiations regarding land that is collectively owned might be more costly still. Furthermore, there would be some monitoring costs to ensure that farmers were complying with the wishes of lake residents. Transaction costs may explain why septic system upgrades are chosen even in cases where engineering cost estimates indicate that riparian buffers appear to be more cost effective.

Conclusions and Policy Implications

A policy of uniform standards is rarely the most cost-effective approach to pollution control. Historically, however, United States policy to control water pollution has been to set a standard and mandate a technology to meet that standard. Although it is doubtful that this approach minimizes costs, it has been effective at controlling point sources of pollution. Progress on cleaning up our nation's surface waters, however, has been stymied by the difficulties of regulating nonpoint sources. Recent policy proposals from the U.S. EPA such as tradable pollution permits (Dressing 2003) are potentially a move in the right direction but are beset with technical difficulties in applications to nonpoint source water pollution.

The goals of the CWA present state regulators with the problem of attaining ambitious targets with limited funding. The approach illustrated in this article suggests that cost-effectiveness analysis may help regulators or lake groups identify the least costly strategies to address lake water quality problems. Furthermore, allowing lake stakeholders to formulate and identify strategies not only may ease the regulatory burden of government but allows tailor-made solutions to eutrophic lakes, given that no two lakes and therefore no two strategies are the same.

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