Agricultural Issues: Midwestern Experience

IDENTIFYING CRITICAL NPS CONTRIBUTING WATERSHED AREAS

KENT W. THORNTON
DENNIS E. FORD
FTN Associates
Little Rock, Arkansas

ABSTRACT

By general agreement, nonpoint sources contribute significantly to receiving system water quality problems. Sediment is the greatest pollutant both by weight and volume. Nutrients, pesticides, and other constituents may be adsorbed to sediment particles and transported to the receiving system through erosion. Best management practices (BMP's) are usually expensive to implement; however, they do not have to be implemented throughout the entire watershed to control NPS. In many watersheds, certain critical areas contribute the majority of the pollutants. Identifying these areas permits the most efficient implementation of BMP's and the most economical approach for controlling NPS. Eleven major steps in identifying these critical areas are discussed. These range from identifying the physiographic characteristics through cost/benefit analysis and implementation of BMP's. This approach forms the base for nutrient, organic, or other mass loading. An example from an rural watershed is given.

INTRODUCTION

Waste treatment facilities have been significantly upgraded and improved over the past decade. Reduced organic and nutrient point source loadings from these waste treatment facilities have improved water quality in lakes, streams, and reservoirs but not always as dramatically as anticipated.

Water quality integrates all sources of pollutants. Aquatic systems receive and process both point and nonpoint source loads from their watersheds. In many aquatic systems, nonpoint sources may contribute significantly to receiving system water quality problems. Sediment, for example, is the greatest pollutant in aquatic systems both by weight and volume (Vanoni, 1977). Nutrients, pesticides, and other constituents may be adsorbed to sediment particles and transported to the receiving system through runoff and erosion (Green et al. 1978; Johnson et al. 1976; Karickhoff and Brown, 1978). Achieving "fishable–swimmable" objectives in aquatic systems, then, requires control and regulation of both point and nonpoint sources.

Point sources are relatively easy to identify and control. Treatment technology is available and affordable and effluent regulations generally have been promulgated. Nonpoint sources, by definition, are diffuse and not easily identified or quantified. The control technologies and best management practices (BMP) available are generally expensive to implement. In addition, not all areas of the watershed contribute equally to the nonpoint source loads because of the heterogeneities in watershed slopes, soils, and vegetative cover. Certain critical combinations of these and other factors result in greater pollutant loadings.

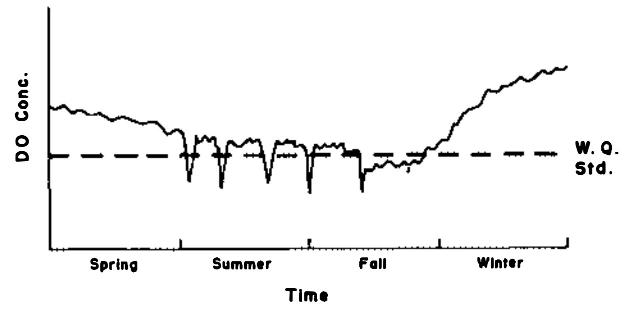

Figure 1.—Temporal variability in stream dissolved oxygen showing the nonpoint impact in summer and point source impact in summer and fall.

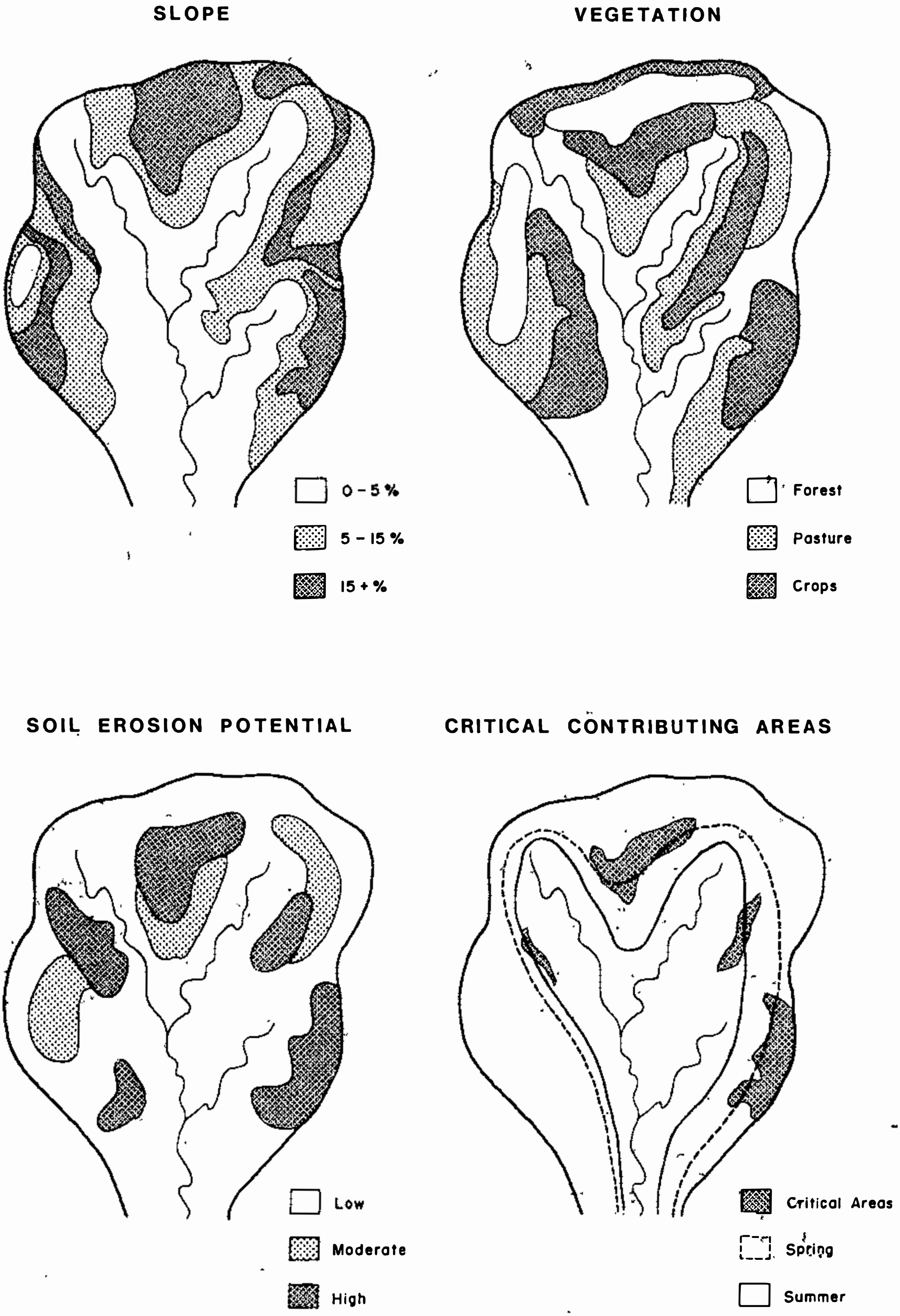

Figure 2.—Location of watershed factors that influence water quality and intersection of factors potentially contributing the greatest loads and runoff areas in the spring (inside dashed line) and summer (inside solid line).

Cost-effective nonpoint source pollution control can be achieved if these critical areas (the areas of significant nonpoint source loadings) can be isolated and subjected to BMP's. This paper discusses steps for a procedure to identify and evaluate critical watershed areas contributing nonpoint source pollution.

ASSUMPTIONS

All techniques and approaches incorporate a set of underlying assumptions. These assumptions are important for proper application and interpretation of any technique. The major givens underlying this procedure include:

(1) Water quality impairment has occurred. The technique is also applicable for identifying land resources impairment but this application of the procedure is not discussed here.

(2) Surface water quality: Ground water quality or the coupled surface–ground water system is not considered. The approach of the following procedure may be applicable to evaluating the impact of nonpoint sources on ground water recharge areas but this paper considers only surface water.

(3) Existing information: Data deficiencies or missing data may be identified through this approach but the procedure itself uses existing available data in delineating critical areas.

PROCEDURE

Several different approaches exist to identify and evaluate critical areas and then select and implement BMP's (Maas et al. 1985; Monteith et al. 1981). Eleven general steps in this procedure are:

1. Delineate the watershed and subcatchment boundaries. This defines the management units. Subcatchments may be further subdivided or aggregated depending on objectives and available resources.

2. Document the water quality impairment. Plotting the water quality data can indicate the location, temporal variability, and relative magnitude of the water quality impairment (Fig. 1). Water quality problems that generally occur during or following storm events imply nonpoint source contributions, while problems that occur during low flow conditions imply point source contributions.

3. Determine natural background. The key to this step is to find a similar but relatively undisturbed area where natural background concentrations have been measured. Several states currently use this approach, sometimes referred to as an ecoregion or physiographic approach to water quality (Omernik and Hughes, 1983; Jarman, 1984).

4. Identify point sources. Compared to nonpoint sources, point sources typically are relatively constant contributors of pollutants. The location and magnitude of these contributions must be identified and quantified to assess the point source impact on receiving system water quality. The relative point source contribution to the pollutant budget and the natural background must be known to evaluate nonpoint source contributions. If point sources account for the major water quality impacts, expenditures for nonpoint source controls may not be warranted.

5. Compute "back of the envelope" nonpoint source contributions. Annual runoff estimates, available from the U.S. Geological Survey, Soil Conservation Service, and State or local agencies, can be multiplied by export coefficients for various constituents to estimate nonpoint source constituent loads to the receiving system for comparison with point source loadings in assessing water quality impairment (Reckhow and Simpson, 1980; Reckhow and Chapra, 1983). These calculations can be refined based on soil series, vegetative species, hydrologic class, and other considerations. If these preliminary calculations indicate nonpoint sources may be contributing to water quality impairment, then the watershed areas contributing the greatest percentage of the load need to be identified and BMP's evaluated to reduce these loads.

6. Visit Site. An obvious, and too often ignored, step is to visit the watershed and inspect the point sources, septic systems, drainage patterns, vegetative cover, and other watershed characteristics. Septic system drainage may not be evident from topographic maps or aerial photographs but may contribute to water quality impairment. Interpreting water quality patterns and reaching informed conclusions on appropriate BMP's can only be achieved by physically visiting the area and observing the watershed conditions and characteristics.

7. Identify critical areas. Considerations include:

a. *Controlling factors:* Factors known or suspected to affect erosion, runoff, and water quality—for example, slope, physiography, geology, soils, vegetative cover, land use, and cultural resources—might be considered.

b. *Class intervals:* Class intervals for each controlling factor should be established to represent similar impacts or effects on water quality. Slopes between 0-3 percent, for example, may similarly affect water quality. Forest cover may represent a generic vegetation class that has minimal impact on water quality compared with other vegetative classes such as row crops. Although the specific class intervals are based on the specific objectives and available data, the number of intervals for each factor must be restricted to three or four.

c. *Class interval mapping.* The specific watershed areas exhibiting similar class intervals should be mapped (Fig. 2). Potential problem areas become apparent as this exercise continues.

d. *Controlling factor overlap:* The intersection among multiple controlling factors indicates potential problem areas. Transparent overlays can be used to identify the areas of overlap when a consistent map scale is used for all class intervals on the watershed. Intersecting areas with class intervals indicate a high potential for water quality impacts (Fig. 2d). These areas also can be digitized and the individual files merged to delineate intersecting areas.

e. *Contributing areas:* After the potential problem areas are identified, runoff or transport processes need to be evaluated to determine the contributing flow areas to the receiving waters. Variable source contributing area relations can be determined (Beven and Kirkby, 1979; Beven and Wood, 1983) or simpler runoff formulations can be used to determine potential transport to the aquatic system (Chow, 1964; Soil Conserv. Serv. 1974). By overlaying this information on the potential problem areas, the critical areas contributing pollutants can be defined (Fig. 2d). Constituent loading from these critical areas can be calculated using runoff estimates and export coefficients.

8. Evaluate candidate BMP's. Determining the location and contribution of critical areas in the watershed provides the necessary data to evaluate appropriate BMP's to reduce constituent loads. Slope, soil series, vegetative cover, land use, and other factors associated with each critical area can be used to screen BMP's and determine candidate BMP's for further evaluation. The percent reduction in constituent loads should be calculated for each candidate BMP.

9. Prepare benefit/cost analysis. The costs associated with various BMP's generally are available from the Agricultural Stabilization and Conservation Service (ASCS), State and local agencies, or private contractors. Associating benefits with various BMP's, however, is more difficult since water quality improvements also have intangible benefits. The initial analyses may be performed using a unit load reduction/dollar cost ratio. This ratio provides a comparative basis for alternative BMP's.

10. Rank alternatives. Some critical areas may be of greater priority because of designated stream or lake uses in that area, desired uses, or other considerations. This objective or subjective priority can be predicted on factors such as the ease of implementation, longevity, maintenance, land owner willingness to participate, or budgetary constraints. Regardless of the criteria, the rationale for establishing priority order alternatives should be documented to increase the likelihood of objectivity.

11. Seek funding sources and implement BMP's. The final step in the procedure is to locate sources of funds to implement the BMP's. Various cost-sharing programs are available through both Federal and State agencies for some BMP's. Some local watershed districts also provide cost-sharing funds.

DISCUSSION

Specific steps in identifying critical watershed areas vary among investigators (Maas et al. 1985; Monteith, 1981). However, the perception and acknowledgement of critical watershed areas is more important than the specific steps in identification. Funds do, and probably will always, limit the possibilities so the objective should be to maximize the benefits derived from each dollar spent. If 10 to 15 percent of the watershed areas are contributing 80 to 90 percent of the water quality problem, these areas should receive the priority for BMP's.

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GROSS EROSION RATES, SEDIMENT YIELDS, AND NUTRIENT YIELDS FOR LAKE ERIE TRIBUTARIES: IMPLICATIONS FOR TARGETING

DAVID B. BAKER
KENNETH A. KRIEGER
R. PETER RICHARDS
JACK W. KRAMER
Heidelberg College
Tiffin, Ohio

ABSTRACT

Studies of agricultural nonpoint pollution in selected watersheds within the Lake Erie Basin have included estimates of gross erosion rates, as well as measurements of sediment, phosphorus and nitrogen yields. These studies incidate that under conventional management practices average gross erosion rates for watersheds are not reliable indicators of the export of either soluble or sediment associated pollutants. Average gross erosion rates are not even good indicators of sediment yields. Consequently, factors other than gross erosion need to be considered in watershed level targeting for agricultural pollution abatement programs.

INTRODUCTION

In the early 1970's, measurements of phosphorus transport during runoff events in the Sandusky River suggested that phosphorus loading to Lake Erie from agricultural sources was being underestimated (Baker and Kramer, 1973). Extension of event monitoring to the lake's other major tributaries confirmed that agricultural sources were indeed larger than previously estimated. (U.S. Army Corps Eng. 1975). Phosphorus modeling studies for the lake indicated that phosphorus reductions from nonpoint sources would be necessary to achieve desired water quality in the lake. Consequently, a major focus of the U.S. Army Corps' Lake Erie Wastewater Management Study was to evaluate options for reducing phosphorus loading from agricultural sources (U.S. Army Corps Eng. 1979).

Part of the evaluation established water quality monitoring stations for a number of smaller watersheds, so that the relationships between watershed characteristics (e.g. land use, soils, slopes, and gross erosion) and pollutant export could be incorporated into the management plans. This paper summarizes some of the relationships between watershed characteristics and pollutant yields we have observed during these studies. In a companion paper in this volume, we have described some of the characteristics of sediment, nutrient, and pesticide transport in area streams and rivers (Baker et al. 1985). Our sampling methods and analytical procedures are summarized in that paper.

WATER QUALITY PROBLEMS ASSOCIATED WITH RURAL NONPOINT SOURCES IN THE LAKE ERIE BASIN

Phosphorus Loading to Lake Erie

Table 1 shows an estimate of the sources and amounts of total phosphorus and bioavailable phosphorus loading to Lake Erie. This estimate draws upon several sources of information. The rural nonpoint phosphorus loads represent an average load for the 1970–80 period (Yaksich, 1983; Yaksich et al. 1985). The urban nonpoint load is based on the Lake Erie study (U.S. Army Corps Eng. 1982).

Point source phosphorus loading to the lake has been greatly reduced by phosphorus removal programs at municipal sewage treatment plants. The point source loads shown on Table 1 represent 1982 loading data. Since by 1982 most municipal dischargers were meeting the phosphorus removal requirements, little further reduction in municipal phosphorus loading can be expected.

The data for atmospheric inputs and Lake Huron inputs, as well as the municipal and industrial point source inputs, are taken from International Joint Commission sources (1983a, b). Direct point sources empty into river mouth areas, bays, or the nearshore zone of the lake while indirect point sources empty into streams and rivers in tributary watersheds. The estimates of bioavailable phosphorus loading are based on direct measurements by our laboratory (Baker, 1983b) and other bioavailability studies summarized by Sonzogni et al. (1982).

Rural nonpoint sources account for 60 percent of the total phosphorus loading and 56 percent of the bioavailable phosphorus loading to the lake. Soluble phosphorus derived from rural sources is a significant part of the rural bioavailable load.

The phosphorus target load for Lake Erie is 11,000 metric tons per year (Int. Joint Comm. 1984). The estimated load of 13,996, based on long-term nonpoint loads and 1982 point source load, exceeds the target load by 3,000 metric tons. Programs to achieve the additional 3,000 metric ton per year reduction are focusing on nonpoint sources, since the most cost-effective portions of municipal phosphorus removal programs have already been implemented (U.S. Army Corps Eng. 1982).

Other Water Quality Problems

The stream monitoring programs have also measured sediments, nitrates, and pesticides. Suspended sediments in area streams and rivers result in the usual problems associated with erosion (Clark et al. 1985). In the Sandusky River, nitrate concentrations exceed drinking water standards about 16 percent of the time in May, June, and July (Baker, 1985). Rivers are frequently used as public water supplies in northwestern Ohio, and nitrates are not removed in treatment.

Several soluble herbicides are present in the rivers in May and June and pass directly through conventional water treatment plants (Baker, 1983a). Until specific maximum contaminant levels or health advisory levels are established by the U.S. Environmental Protection Agency for the currently used pesticides that frequently occur in drinking water, assessing the significance of these exposures will be difficult.

STUDY WATERSHEDS

From 3 to 10 years of chemical sampling have been completed for the 15 watersheds listed in Table 2. The table includes the U.S. Geological Survey stream gauge station number, the drainage area, the mean discharge and period of hydrological sampling, the period of chemical sampling, and the number of samples analyzed for sediments and nutrients. The average annual runoff ranged from 23.2 cm for the Raisin River to 39.8 cm for the Cuyahoga River. Watershed sizes range from 44 to 16,359 km².

The land use within each watershed is summarized in Table 3. Except for the Cuyahoga River basin, cropland dominates land use for all of the watersheds. The geographical data system used for developing land use statis-

Table 1.—Total and bioavailable phosphorus loads for Lake Erie, based on 1982 point source loads, long-term nonpoint source loads, and recent studies of bioavailable phosphorus loads.

		Total Phosphorus		Bioavailable Phosphorus		
		Load Metric Tons	Percent of Total	Fraction Bioavailable	Load Tons	Percent of Total
I. NONPOINT SOURCES						
Rural[1]						
Particulate	80%	6720	48.0	.30	2016	31.2
Soluble	20%	1680	12.0	.95	1596	24.7
Subtotal		8400	60.0	(.43)	3612	55.9
Urban[1]		900	6.4	.43	387	6.0
Atmospheric[2]		660	4.7	.38	251	3.9
Total Nonpoint Source		9960	71.1		4250	65.8
II. POINT SOURCE						
Municipal						
Flows > 1 MGD						
Direct[2]		1388	9.9	.70	972	15.0
Indirect[3]		1061	7.6	.35	371	5.7
Flows < 1 MGD						
Direct		110	0.8	.70	77	1.2
Indirect		330	2.4	.35	116	1.8
Subtotal		2889	20.6		1536	23.7
Industrial[2]		67	0.5	.50	34	0.5
Total Point Sources		2956	21.1		1570	
III. LAKE HURON INPUT[2]		1080	7.7	.60	648	10.0
IV. TOTAL LOADS		13,996	100		6468	100

[1] Yaksich, 1983.
[2] I.J.C. 1983a.
[3] I.J.C. 1983b.
[4] onzogni et al. 1982; Baker 1983b.

Table 2.—Watershed areas, mean annual discharges, period of chemical sampling, and number of samples analyzed for the Lake Erie tributary monitoring program.

Transport Stations	U.S. Geological survey station number	Drainage Area Km²	Mean Annual Discharge			Chemical Sampling Period	Number of Samples Analyzed
			Years of Record	m³/s	cm		
Sandusky River Stations							
1. Fremont	04198000	3,240	57	27.75	27.0	1974–84	4590[2]
2. Mexico	04197000	2,005	55	16.62	26.2	1974–81	2178
3. Upper Sandusky	04196500	722	57	6.967	28.5	1974–81	2973
4. Bucyrus	04196000	230	40	2.461	33.8	1974–81	2998
Sandusky River Tributaries							
5. Wolf Creek, East	04192450	213	5	1.82	27.0	1976–81	2425
6. Wolf Creek, West	04197300	171.5	5	1.34	24.6	1976–81	2419
7. Honey Creek, Melmore	04197100	386	7	3.908	32.0	1976–84	4595[2]
8. Honey Creek, New Wash.	04197020	44.0	3.908	(0.445)[1]	(32.0)[1]	1979–81; 83–84	2271[2]
9. Tymochtee Creek	04196800	593	19	4.956	26.3	1974–81	2471
10. Broken Sword Cr.	04196200	217	5	2.45	35.5	1976–81	2512
Other Lake Erie Tributaries							
11. Maumee River	04193500	16,359	58	139.5	26.8	1975–80; 82–84	3154[2]
12. Raisin	04176500	2,699	43	19.85	23.2	1982–84	805[2]
13. Cuyahoga	04208000	1,831	52	23.14	39.8	1981–84	1380[2]
14. Portage	04195500	1,109	51	9.091	25.9	1974–78	1856
15. Huron	04199000	961	31	8.496	27.9	1974–79	2027

[1] Extrapolated from Honey Creek at Melmore.
[2] Sampling continued on 1985 program.

tics for Lake Erie watersheds has been summarized by Adams et al. (1982).

WATERSHED POLLUTANT EXPORT

The flux-weighted mean concentrations of sediments and nutrients at each of the transport stations are summarized in Table 4. These represent the flux-weighted averages for the entire period of chemical sampling and for the number of samples indicated in Table 2. For all of the stations, sampling is conducted throughout the year and consequently the averages reflect the seasonal distribution of storms that occurred during the sampling period.

One method of estimating mean annual pollutant export is to multiply the flux-weighted pollutant concentrations by the mean annual discharge. This procedure will provide an accurate mean annual load estimate so long as: (1) the flux-weighted average concentration accurately reflects current watershed responses to current weather and rainfall regimes in the region; and (2) the mean annual discharge for the period of record reflects contemporary watershed responses to current weather and rainfall regimes. The latter condition does not appear to be met since the average discharges for the Maumee, Sandusky, and Cuyahoga rivers during the 1973–83 period are higher than long-term average discharges (as of 1983), by 17 percent, 26 percent and 30 percent respectively. Rather than attempt to adjust each watershed to current "average' discharges, the discharge data in Table 2 have been adjusted to long-term stations (Baker, 1983c). Other methods of using these data to estimate mean annual pollutant loads have been discussed by Baker (1984).

In Table 5, the mean annual loads of sediments and nutrients are expressed on a unit area yield basis. For each watershed the mean annual load was divided by the total watershed area to determine unit area yields. The gross erosion rates, as calculated in the Lake Erie study (Logan et al. 1982), are also listed for each watershed. The unit area total phosphorus yield of 1.26 kg/ha/year for the Sandusky Basin calculated by this method is significantly lower than the 10-year average export of 1.55 kg/ha/year actually measured at the station (Baker et al. 1985).

The unit area yields shown in Table 5 reflect the inputs of both point and nonpoint sources. For the parameters shown in Table 5, point sources constitute significant inputs only for phosphorus and for certain watersheds. Table 6 illustrates the typical procedure of correcting for

Table 3.—Watershed land use for the Lake Erie tributary nutrient and sediment transport stations.

Transport Stations	Cropland %	Pasture %	Forest %	Water %	Other %
Sandusky River Stations					
1. Fremont	79.9	2.3	8.9	2.0	6.8
2. Mexico	80.3	2.3	8.7	2.1	6.6
3. Upper Sandusky	78.0	3.4	9.1	2.0	7.5
4. Bucyrus	73.3	4.9	9.4	2.1	10.2
Sandusky River Tributaries					
5. Wolf Creek, East	81.9	2.7	6.3	2.0	7.0
6. Wolf Creek, West	83.3	1.4	4.7	3.1	7.6
7. Honey Creek, Melmore	82.6	0.6	10.0	0.5	6.3
8. Honey Creek, New Wash.	89.1	—	7.5	—	3.4
9. Tymochtee Creek	84.0	1.2	7.6	2.3	4.8
10. Broken Sword Creek	84.7	1.4	8.5	1.3	4.1
Other Lake Erie Tributaries					
11. Maumee River	75.6	3.2	8.4	3.5	9.4
12. Raisin	67.1	6.8	9.0	3.0	14.1
13. Cuyahoga	4.2	43.1	29.1	3.0	20.6
14. Portage	85.5	3.6	5.6	0.9	4.3
15. Huron	75.3	3.5	12.5	2.2	6.4

Table 4.—Flux weighted mean concentrations of sediments and nutrients at the transport stations for the period of chemical sampling.

Transport Stations	Suspended Solids mg/L	Total Phosphorus mg/L	Soluble Reactive Phosphorus mg/L	Nitrate + Nitrite-N mg/L	Total Kjeldahl Nitrogen mg/L
Sandusky River Stations					
1. Fremont	249	0.468	0.084	4.61	1.73
2. Mexico	230	0.409	0.061	4.32	—
3. Upper Sandusky	312	0.583	0.126	4.60	—
4. Bucyrus	273	0.633	0.199	3.71	—
Sandusky River Tributaries					
5. Wolf Creek, East	246	0.479	0.109	5.32	—
6. Wolf Creek, West	251	0.461	0.089	6.95	—
7. Honey Creek, Melmore	198	0.413	0.074	4.79	1.79
8. Honey Creek, New Wash.	254	0.458	0.088	5.05	1.81
9. Tymochtee Creek	231	0.424	0.065	5.60	—
10. Broken Sword Creek	312	0.479	0.064	5.08	—
Other Lake Erie Tributaries					
11. Maumee River	218	0.474	0.090	5.13	1.85
12. Raisin	81.1	0.238	0.046	3.51	1.23
13. Cuyahoga	188	0.428	0.105	1.82	1.36
14. Portage	164	0.402	0.119	5.89	—
15. Huron	220	0.362	0.104	3.61	—

point source inputs. Point source inputs, expressed on a unit area basis, are subtracted from the total unit area yield to determine a unit area nonpoint yield. This procedure assumes 100 percent delivery of point source phosphorus through the stream system.

The point source inputs for the Maumee River, Sandusky River–Fremont, Honey Creek–Melmore, and Sandusky River–Bucyrus reflect inputs from all municipal sewage treatment plants within the watershed. For the Raisin River and the Cuyahoga River, only sewage treatment plants with flows greater than 1 million gallons per day were included. Consequently, the nonpoint yields for the Raisin River and Cuyahoga River are probably being overestimated.

Table 6.—Calculation of nonpoint phosphorus yields at representative transport stations.

Watershed	Total-P Export kg/ha/yr	Point Source-P Input kg/ha/yr	Non Pt. P-Export kg/ha/yr	Percent Nonpoint
Maumee R.	1.27	0.20	1.07	84
Sandusky R., Fremont	1.26	0.14	1.14	89
Cuyahoga R.	1.71	1.11	0.60	35
Raisin R.	0.55	0.11	0.44	80
Honey Cr.	1.32	0.09	1.23	93
Sandusky R., Bucyrus	2.14	1.17	0.97	45

DISCUSSION

In general, the unit area yields of total phosphorus and total nitrogen (nitrate-N + total Kjeldahl nitrogen) in Lake Erie tributaries are much larger than the average values cited for agricultural lands. Based on a thorough review of the literature on loading studies, Rast and Lee (1983) recommended the use of 0.5 kg/ha/year and 5 kg/ha/year, respectively, for estimating total phosphorus and total nitrogen inputs into lakes from agricultural watersheds. The total phosphorus yields from the Maumee and Sandusky river basins are 2.5 times higher than these national averages. Total nitrogen yields in northwestern Ohio rivers are 3.5 times higher than the average values. These high rates of nutrient export occur even though the average gross erosion rates of 4.1 to 9.8 tons/ha/year (1.9 to 4.4 short tons/acre/year) for these watersheds are lower than the average for cropland in the United States.

Within the study watersheds, average gross erosion is not a consistent indicator of pollutant yields. In Table 7 the ratios of average gross erosion rates and yields of suspended sediments, nonpoint total phosphorus, soluble phosphorus, and nitrates are shown for three pairs of watersheds. The gross erosion rate is 2.2 times higher for the Broken Sword watershed than for the Wolf Creek West watershed, while the suspended solids and total phosphorus yields are 1.79 and 1.50 times higher, respectively. The gross erosion rate in the Honey Creek watershed is 1.6 times higher than Wolf Creek West, yet the suspended solids and nonpoint phosphorus yields are essentially the same.

The Raisin River watershed has the highest gross erosion rate—1.18 times higher than that of the Sandusky River watershed. The sediment and nonpoint total phosphorus yields for the Raisin are only 0.27 and 0.38 of those for the Sandusky. In part, the higher sediment and phosphorus deliveries from the Sandusky Basin result from finer textured soils.

Land use effects are evident in the data of Tables 5 and 6. The Cuyahoga Basin, which has a small percentage of cropland, has relatively low nitrate and nonpoint phosphorus yields. Stations impacted by large proportions of point source inputs (e.g. the Sandusky River Bucyrus station and the Cuyahoga stations) have high soluble reactive phosphorus yields.

Watershed size has little effect on suspended sediment and nutrient yields. The Maumee, Sandusky, and Honey Creek watersheds all have similar unit area exports. This suggests that instream delivery losses do not increase with watershed size for this region.

These studies do not support the use of the universal soil loss equation to select specific watersheds for targeted nonpoint phosphorus control programs. Instead, an areawide program to support best management practice (BMP) implementation on critical areas, regardless of subwatershed boundaries, would likely result in the most cost efficient reductions in sediment and particulate phosphorus loads to Lake Erie.

The source areas for high nitrate concentrations in this region are the tile-drained fields. Fertilizer BMP's need to be implemented throughout this region. Since much of the soluble phosphorus export occurs in the winter season (Baker et al. 1985), the contributing areas are probably much larger than those for sediment and particulate phosphorus.

Table 5.—Gross erosion rates and unit area yields of sediments and nutrients at the transport stations.

Transport Stations	Gross Erosion Rate kg/ha/yr	Suspended Solids kg/ha/yr	Total Phosphorus kg/ha/yr	Soluble Reactive Phosphorus kg/ha/yr	Nitrate + Nitrate-N kg/ha/yr	Total Kjeldahl Nitrogen kg/ha/yr
Sandusky River Stations						
1. Fremont	8,250	673	1.26	0.22	12.5	4.68
2. Mexico	9,370	601	1.07	0.16	11.3	—
3. Upper Sandusky	9,350	950	1.78	0.38	14.0	—
4. Bucyrus	7,850	922	2.14	0.67	12.5	—
Sandusky River Tributaries						
5. Wolf Creek, East	5,110	663	1.29	0.29	14.3	—
6. Wolf Creek, West	4,190	619	1.14	0.22	17.1	—
7. Honey Creek, Melmore	6,860	633	1.32	0.24	15.3	5.72
8. Honey Creek, New Wash.	7,060	811	1.46	0.28	16.1	5.78
9. Tymochtee Creek	8,410	609	1.12	0.17	14.8	—
10. Broken Sword Creek	9,390	1,110	1.71	0.23	18.1	—
Other Lake Erie Tributaries						
11. Maumee River	6,840	585	1.27	0.24	13.8	4.97
12. Raisin	9,750	188	0.55	0.11	8.1	2.85
13. Cuyahoga	—	749	1.71	0.42	7.3	5.42
14. Portage	5,000	424	1.04	0.31	15.2	—
15. Huron	7,510	614	1.01	0.29	10.1	—

Table 7.—Comparison of erosion rate and nonpoint yields for representative watersheds.

Watershed	Gross Erosion kg/ha/yr	Nonpoint			
		SS kg/ha/yr	TP kg/ha/yr	SRP kg/ha/yr	NO_3 kg/ha/yr
Broken Sword	9,390	1,110	1.71	0.23	18.1
Wolf, East	4,190	619	1.14	0.22	17.1
Ratio—Broken Sword: Wolf East	2.24	1.79	1.50	1.04	1.06
Honey Creek	6,860	63	1.23	0.24	15.3
Wolf, East	4,190	619	1.14	0.22	17.1
Ratio—Honey: Wolf East	1.63	1.02	1.08	1.09	0.89
Raisin River	9,750	188	0.44	0.11	8.1
Sandusky River	8,250	673	1.14	0.22	12.5
Ratio—Raisin R: Sandusky R.	1.18	0.27	0.38	0.50	0.65

The timing of pesticide export during storm events suggests that the contributing areas are those from which surface runoff water reaches stream systems. This will vary considerably from year to year depending on rainfall amounts and intensities.

CONCLUSIONS

1. The nonpoint source nutrient exports from Lake Erie tributaries are very high relative to average agricultural export rates even though gross erosion rates in these watersheds are low.

2. Within this region, average gross erosion rates for watersheds are not reliable indicators of sediment and nutrient yields.

3. Contributing areas for nitrate and soluble phosphorus probably encompass a larger portion of the land surface than contributing areas for sediment and pesticides.

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WATERSHED WATER QUALITY PROGRAMS: LESSONS LEARNED IN ILLINOIS

TOM DAVENPORT
JOHN LOWREY
U.S. Environmental Protection Agency
Chicago, Illinois

INTRODUCTION

Today every segment of our society is looking for ways to improve efficiency. Governmental agencies as well as the business community are increasingly aware that financial times have changed. No matter how important a cause, Federal, State and local agencies realize that except for defense spending, major new Federal funding initiatives are unlikely in the near future. For agencies to fulfill their legislative mandates and responsibilities, they must look for new ways to improve the delivery and efficiency of existing approaches and programs.

Several nonpoint source control projects—Sec. 108 Great Lakes Demonstration Projects, Clean Lakes Projects, Sec. 314 Agricultural Conservation Program Projects, and Rural Clean Water Projects—have been implemented in watersheds critical for agricultural pollution. Evaluation of these ongoing nonpoint source control projects is necessary for facilitating future NPS control programs. Presently in the State of Illinois, two major watershed nonpoint source evaluation projects exist. Recommendations on project selection, development, and implementation will be discussed based on evaluation of these projects.

BACKGROUND

The State of Illinois began seriously to reevaluate its soil and water conservation programs in 1977, during the development of its Water Quality Management (WQM) Plan, completed in 1979. The WQM Plan identifies the scope and seriousness of nonpoint source pollutants and assigns agency management responsibility. This initial WQM planning effort documented that agricultural activities are a major source of pollution and mandated the development of plans to control NPS pollution from agriculture. The most severe problem identified was soil erosion resulting in lake sedimentation. Through the initial WQM planning process, 11 pirority watersheds were selected in 1979 (Table 1). This study discusses the Lake Pittsfield (Blue Creek) and Silver Lake (Highland) Watershed projects.

STUDY AREAS AND PROGRAMS

The Blue Creek Watershed encompasses slightly more than 2,833 hectares in east central Pike County, Illinois (Fig. 1). The terrain is hillier than most areas of Illinois, and the area has a high soil loss potential because of its steep slopes, fine-textured soils, and agricultural land use practices. The Blue Creek Watershed drains into Pittsfield City Lake, which was constructed with P.L. 566 assistance from the Soil Conservation Service in 1961 as a multiple purpose flood control and water supply reservoir. Average soil loss was estimated at 18.1 tons (M)/ha/year. Erosion from livestock operations, primarily hog enterprises, significantly contributes to the total basin sediment load. Soil Erosion and Sediment Transport in the Blue Creek Watershed (Davenport, 1983) describes the Blue Creek Watershed Project in detail. The Blue Creek project started in 1979 and ended in 1982.

The Highland Silver Lake Watershed encompasses 12,524 hectares in the eastern portion of Madison County, Illinois (Fig. 1). It drains into Highland Silver Lake through Silver Creek and numerous tributaries. The City of Highland built Highland Silver Lake, an artificial impoundment, in 1962 as a public water supply. Agriculture is the dominant land use with 88 percent of the land devoted to row crops (Table 2). The terrain is relatively flat, and the soils have a high detachment potential because of their fine texture, the influence of sodium, and the agricultural land use practices. Average soil loss was estimated to be 15.2 tons (M)/ha/years (Ill. Environ. Prot. Agency, 1979). The Highland Silver Lake project started in 1980, with signup lasting to June 1985 and implementation completed in 1990.

Land treatment activities were funded in the Blue Creek Watershed through the ACP Special Water Quality Pro-

Table 1.—Priority lakes for agricultural nonpoint source water quality problem abatement in order of priority (Ill. Environ. Prot. Agency, 1979).

Ranking	Watershed	Watershed size (ha)	Lake size (ha)	Average erosion rate (ton (M)/ha)
(1)	Lake Pittsfield (Blue Creek)[1]	2,838	96	18.1
(2)	Lake Carlinville	6,755	68	15.7
(3)	Lake Canton	3,940	101	14.3
(4)	Silver Lake (Highland)[1]	12,399	223	15.2
(5)	Spring Lake	5,236	112	13.9
(6)	Lake Springfield	66,973	1,630	13.4
(7)	Lake Taylorville	34,029	465	13.0
(8)	Lake Lou Yeager	29,808	514	14.6
(9)	Lake Bloomington	17,621	257	12.3
(10)	Paris Lake	5,184	89	11.4
(11)	Lake Paradise	4,690	71	11.2

[1] Projects evaluated

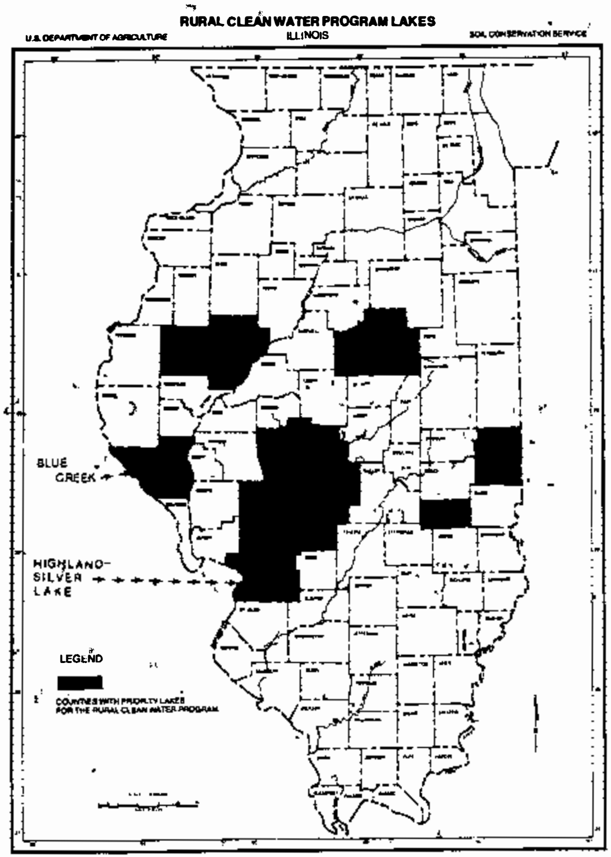

Figure 1.—Rural Clean Water Program Lakes in Illinois.

gram and in Highland Silver Lake Watershed through the RCWP. Both the RCWP and ACP Special Water Quality Program endorse the concept "that the condition of the lake is a reflection of the condition and management of its watershed." Both programs assume that resource management systems in critical water quality areas will improve the quality of the downstream water resources. Both programs rely on voluntary landowner/operator participation to address water quality problems.

DISCUSSION

For both the Blue Creek and Highland Silver Lake projects, the overall goal is to improve water quality by reducing soil loss. The Blue Creek Project reached its soil reduction goal; Highland Silver Lake probably will not achieve its goal. Two years after most of the Blue Creek Watershed project was completed, signs of water quality improvement were noted. In Highland Silver Lake, water quality has not improved. In both project areas, however, landowners/operators believe water quality has improved.

Five program elements form the basis for examining the effectiveness of both projects.

1. **Problem Identification/Definition:** This is the most crucial step in the NPS control project implementation process, because this step determines the type of program. The problem identified was excessive sedimentation in both Pittsfield City and Highland Silver Lakes. In both cases the cause of the sedimentation was excessive sheet and rill erosion from agricultural lands. In the case of Pittsfield City Lake the project sponsors documented the lake sedimentation problem with a lake sedimentation survey before applying for funding. In Highland Silver Lake the project sponsors stated in the application that lake sedimentation was a problem, although it was not documented. (Mad. Co. Soil Water Conserv. Dist. 1979). As part of the Highland Silver Lake monitoring program, a lake sedimentation survey was completed. It showed that turbidity, not lake sedimentation, was the water quality problem (Davenport and Kelly, 1984). In both projects, programs were implemented to control excessive soil erosion as the cause of lake sedimentation problems. In the case of Highland Silver Lake, the program should have addressed the turbidity problem.

Table 2.—Land use/cover in the Highland Silver Lake (1981) and Blue Creek (1980)[1] watersheds in hectares and percent.

	Highland Silver Lake Watershed (Ill. State Coord. Comm. 1982)	Blue Creek Watershed (Davenport, 1983)
Cropland	10,200.5	1,602.9
(Percent)	(83.3)	(56.4)
Pasture/hayland	672.6	614.9
(Percent)	(5.4)	(21.7)
Woodland	505.9	327.8
(Percent)	(4.1)	(11.6)
Urban	85.0	—
(Percent)	(0.7)	—
Feedlots	46.9	26.5
(Percent)	(0.4)	(0.9)
Interstate	19.8	—
(Percent)	(0.2)	—
Wildlife	132.3	126.1
(Percent)	(1.1)	(4.4)
Farmsteads	250.5	44.2
(Percent)	(2.0)	(1.6)
Gravel	6.1	—
(Percent)	(0.0)	—
Residential	144.1	—
(Percent)	(1.2)	—
Water	335.9	95.7
(Percent)	(2.7)	(3.4)
Total	12,399.6	2,838.1

[1] Year Reported

2. Critical Area Determinations: Both projects required that local sponsors determine critical areas. Critical areas are those areas or sources that contribute most to impairment of downstream water resources.

In Blue Creek Watershed, a local coordinating committee defined critical area locations within the watershed based on three factors. The first factor considered was whether or not the area was located in the high erosion hazard area. (High erosion hazard areas were considered as those with a soil erodibility factor of at least 0.37 and a slope greater than 5 percent.) Second, they considered the site's close proximity to the lake. The third factor was whether the site had a feedlot with soil losses exceeding the allowable limits. These guidelines assumed that excessive soil erosion was causing the water quality problem. In Highland Silver Lake Watershed Project the committee established that the critical area consisted of all natric soils with 2 percent or greater slope, fine particle size, and high erodibility, and all non-natric soils with 5 percent or greater slope, high erodibility, and close proximity to the stream system. Since the lake sedimentation survey documented turbidity rather than sedimentation as the problem, the critical areas should have been changed.

3. Local Institutional Arrangements: Both projects used similar working arrangements and methods to ensure participation. In both projects three primary methods—information/education, technical, and financial assistance—were used to encourage adoption of Resource Management Systems (RMS) in the critical areas. The "one-to-one" technical assistance and education was a combined effort of all the involved agencies' personnel. The major differences were that existing resources financed the intensive information/educational effort spearheaded by the Cooperative Extension Service in the Blue Creek Watershed, and the effort began 2 years before the special technical and financial assistance was approved; but in the Highland Silver Lake Watershed Project, the National RCWP funded the Cooperative Extension Service's activities; thus, Extension's intensive activities started after Highland Silver Lake had been approved for special technical and financial assistance.

4. Effectiveness of Practices/Programs: In the Blue Creek Watershed, practices cost-shared under the ACP Special Water Quality Project were feasible and acceptable to landowners/operators and could correct the identified soil erosion problem. In the Highland Silver Lake Watershed, 100 percent of the farms had to be treated to receive cost-sharing, and the high sodium concentration in the soils made establishing vegetation difficult.

5. Timeframe for Implementation: Both projects had adequate timeframes for implementation. The Blue Creek Watershed Project achieved its land treatment goals, but it is doubtful that the Highland Silver Lake Watershed Project can. A major difference is that in the Blue Creek Watershed the pre-project information/education efforts obtained early project participation, which paved the way for the technical assistance personnel. In contrast, the Highland Silver Lake Watershed technical and educational efforts began simultaneously.

RECOMMENDATIONS

1. Require documentation of the nature and extent of the water resource impairment. Special physical characteristics that could preclude achieving water quality goals (whether documented or perceived) must be recognized.
2. The type of water quality problem determines critical area delineation.
3. The program must be able to correct the identified water quality problem.
4. The information and education program must precede the technical assistance and cost-sharing components.
5. The use of interagency coordinating committees at the local, State and Federal levels in RCWP projects has proven to be effective in providing leadership and management for water quality projects.

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PRAIRIE ROSE LAKE RURAL CLEAN WATER PROGRAM PROJECT

UBBO AGENA
MONICA WNUK
Iowa Department of Water, Air and Waste Management
Des Moines, Iowa

C. MERLE LAWYER
Soil Conservation Service
U.S. Department of Agriculture
Harlan, Iowa

ABSTRACT

Through the cooperative efforts of private landowners and a number of Federal, State, and local agencies, the Prairie Rose Lake Rural Clean Water Program (RCWP) project substantially reduced sediment and nutrient delivery to the lake from its agricultural watershed. Landowner participation has been high, with over 75 percent of the eligible watershed area included in RCWP contracts. Best management practices installed to date have reduced annual sediment delivery to the lake by 55 percent, from an estimated 23,670 tonnes (26,300 tons) per year in 1980 to 10,627 tonnes (11,808 tons) per year in 1984. Similar reductions have been projected for phosphorus. Lake water quality monitoring has shown improvements resulting from the project. Water clarity has increased during most seasons of the year, and lake turbidity no longer routinely increases following runoff events. However, algal growths during late summer periods appear to be increasing because of a combination of high in-lake nutrient levels and decreased sediment-related turbidity levels.

BACKGROUND

Prairie Rose Lake is an 86 ha (215 acre) State-owned lake constructed in 1962 near Harlan in Shelby County, Iowa. The lake and associated Prairie Rose State Park are used extensively for camping, picnicking, swimming, boating, and fishing.

The lake's watershed is 1,844 ha (4,610 acres): 259 ha (648 acres) is lake and park; 1,459 ha (3,648 acres) is cropland; 59 ha (148 acres) pasture; and 66 ha (166 acres) farmsteads, roads, and woodland. Prior to the RCWP project, the watershed had severe erosion problems, with average annual soil loss of the watershed area (excluding the park and lake) exceeding 44.8 tonnes per hectare (20 tons/acre), and erosion on 62 percent of the nonpark land exceeding 67.3 tonnes per hectare (30 tons/acre).

This excessive soil erosion was the major source of the lake's water quality problems. Since its construction in 1962, sediment had significantly reduced both the lake's surface area and its water volume. Between 1968 and 1977, 4.8 ha (12 acres) of lake surface area were lost to sediment, and an additional 3.2 ha (8 acres) became inaccessible to boats with outboard motors. From 1971 to 1980, lake volume was reduced 19 percent, from 250 ha-m (2,031 acre-feet) to 203 ha-m (1,650 acre-feet) because of sediment deposition.

Excessive sediment loads also caused the lake to become extremely turbid following runoff. The high lake turbidity levels reduced the lake's aesthetic acceptability and its use, prompting numerous public complaints.

Sedimentation and high turbidity levels also affected the lake's fish populations. Sediment reduced the amount of aquatic habitat suitable for fish reproduction and growth, and turbid conditions reduced the ability of sight-feeding fish to catch their prey. Since these conditions affected sport fish more than rough fish, rough fish predominated in the lake. Rough fish predominated so greatly that in 1981 when all fish in the lake were killed to carry out a restocking program, the rough fish (mainly carp and gizzard shad) totalled 689 k per ha (615 pounds per acre); other fish totalled only 100 k per ha (90 pounds per acre).

Other agricultural pollutants of concern in Prairie Rose Lake are nutrients and pesticides. Nutrients stimulate algal growths and thus increase lake eutrophication. Pesticides are of concern mainly from a human health perspective since the lake is the source of drinking water for the park and a major fishing resource in that area of Iowa.

RURAL CLEAN WATER PROGRAM

Funds to implement agricultural nonpoint pollution control practices in the lake watershed became available in August 1980 when the project was approved for funding under the U.S. Department of Agriculture's (USDA) Experimental Rural Clean Water Program (RCWP).

Under the RCWP, Federal funds pay part of the costs of installing Best Management Practices (BMP's) on farmlands, provided the practices reduce agricultural nonpoint pollution of a receiving stream or lake. The practices used may include temporary and permanent soil conservation practices, animal waste controls, and fertilizer and pesticide management practices.

In RCWP projects, a water quality plan is first developed for cooperating farms, identifying the practices needed to control agricultural pollutants and scheduling BMP installation. The plan serves as the basis for a contract between the farmer and the USDA; the farmer agrees to install the needed practices in exchange for RCWP funds. The contracts may cover a 3 to 10 year period.

PRAIRIE ROSE PROJECT

Because the water quality problems in Prairie Rose Lake were directly related to the large quantities of sediment entering it, improving the lake's water quality required substantially reducing that entry. In the RCWP project, this reduction is being accomplished mainly through soil conservation practices on the watershed's agricultural lands. This approach will also decrease the levels of nutrients and pesticides reaching the lake since these materials are frequently attached to eroded soil particles.

The soil conservation practices being used include conservation tillage, contour farming, terraces, grassed waterways, grade stabilization structures, and pasture management. Under the project, RCWP funds pay up to 75 percent of the installation costs of structural practices such as terraces and grade stabilization structures, while a per acre payment is made for management practices such as conservation tillage. A project goal is controlling excessive soil erosion on 80 percent of the watershed's nonpark lands.

The Prairie Rose project also uses nutrient and pesticide management programs. Under the nutrient management program, Iowa State University Soil Testing Laboratory analyzes soil samples collected from farm fields. The soil test results are used to recommend fertilizer application rates and methods which meet crop nutrient needs while minimizing potential nutrient runoff into the lake.

The pesticide management program involves scouting the fields to determine whether weed or insect infestations are sufficient to justify the use of pesticides. If so, the field scouting results help determine the pesticides, application rates, and methods to use. The pesticide use recommendations are designed to assure that weed and insect pests are adequately controlled while the potential for pesticide runoff is minimized.

PROJECT ADMINISTRATION

A number of Federal, State, and local agencies are cooperating in the Prairie Rose RCWP project. These include

- Agricultural Stabilization and Conservation Service, USDA
- Soil Conservation Service, USDA
- Shelby County Soil Conservation District
- Cooperative Extension Service, Iowa State University
- Iowa Department of Water, Air and Waste Management
- Iowa Conservation Commission
- University Hygienic Laboratory, University of Iowa
- U.S. Environmental Protection Agency
- Iowa Department of Soil Conservation

The Shelby County Agricultural Stabilization and Conservation Service (ASCS) office has major responsibility for the day-to-day administration of the Prairie Rose project. Its duties include entering into contracts with participating farmers and administering the RCWP cost share funds. The State ASCS office assists the Shelby County office as needed and works with other Federal and State agencies to assure that needed coordination of agency activities occurs.

Soil Conservation Service (SCS) personnel assigned to the Shelby County Soil Conservation District develop water quality plans for cooperating farms and assist farmers in selecting the soil conservation practices to be used on their farms. SCS staff are also responsible for the design and construction layout of the soil conservation practices used.

The Cooperative Extension Service (CES) of Iowa State University, working through the Shelby County extension office, is responsible for conducting the project's nutrient and pesticide management programs. In addition, the CES conducts the project's public information and education programs.

RCWP regulations require that a general water quality monitoring program be carried out, but prohibit the use of RCWP funds for monitoring. As a result, the Iowa Department of Water, Air and Waste Management (DWAWM) has obtained water quality funds from the U.S. Environmental Protection Agency to support the monitoring program. DWAWM also develops the scope of the project's monitoring program and prepares annual water quality monitoring reports.

The major sample collection activities are conducted by Iowa Conservation Commission (ICC) field staff located at Prairie Rose Lake. ICC staff also collect and maintain records on lake conditions and lake use.

Under contract with DWAWM, the University Hygienic Laboratory (UHL) analyzes the water samples and provides training to ICC staff on sample collection procedures.

The Iowa Department of Soil Conservation (DSC) had a major role in developing the initial RCWP application for the Prairie Rose project. DSC also serves on the State and local project coordination committees.

BMP IMPLEMENTATION PROGRESS

Prior to the RCWP project, soil conservation practices were being used on only a small percentage of the Prairie Rose watershed. Conservation plans had been developed for 14 of the 47 farms in the watershed, contour farming was used on about 400 ha (1,000 acres), and 24.2 km (15.1 miles) of grassed backslope terraces and 2 erosion/sediment control structures had been constructed (Lawyer, 1983). Even with these practices, average erosion rates in the watershed were extremely high, particularly on the cropland areas (see Table 1).

Since being approved for funding in August 1980, the Prairie Rose RCWP project has made considerable progress. The project has been well accepted by farmers in the lake watershed: as of October 1984, 35 of the 47 landowners in the lake watershed had applied for RCWP contracts, 33 had been signed. The 33 signed contracts cover 1,236 ha (3,089 acres), or 79 percent of the eligible area, and represent a commitment of $351,000 in RCWP funds, out of a total project cost share allocation of $446,000. Lands under contract are shown in Figure 1.

Considerable progress has also been made in implementing BMP's in the lake watershed. Table 2 shows the BMP's implemented as part of the project by December 1984 and includes only those practices using RCWP funds.

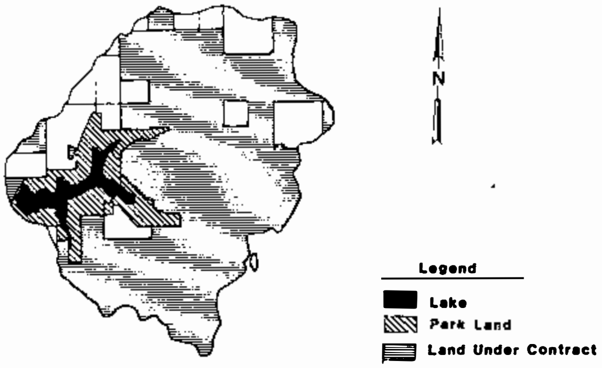

Figure 1.—Land under contract, Prairie Rose Lake Watershed (ASCS, 1984).

Table 1.—Cropland erosion rates (Shelby County ASCS, 1980).

Location	Annual Soil Loss Tonnes/ hectare	(tons/ acre)	Area Hectares	(acres)
Sidehills	67.3	(30)	975	(2,438)
Hilltops	11.2	(5)	197	(492)
Bottomland	11.2	(5)	396	(990)

Three landowners not under RCWP contract have also installed soil conservation practices on their lands. Two of these landowners established a total of 14.4 ha (36 acres) of permanent pasture seeding and one constructed 4.5 km (2.8 miles) of terraces.

The RCWP practices and non-cost-shared practices such as crop residue management and contouring together are adequately protecting 55 percent of the water-

Table 2.—BMP Implementation (ASCS, 1984).

BMP	Amount Implemented
Pasture seeding	12.8 hectares (32 acres)
Terrace system	70.7 kilometers (43.9 miles)
Waterway systems	4.0 hectares (10.1 acres)
Conservation tillage system	224 hectares (560 acres)
Sediment/water control structure	8 structures
Nutrient management	952 hectares (2,379 acres) (27 farms)[1]
Pesticide management	952 hectares (2,379 acres) (27 farms)[1]

[1] Total contracted as of December 1984—includes some contracted after 1984 growing season.

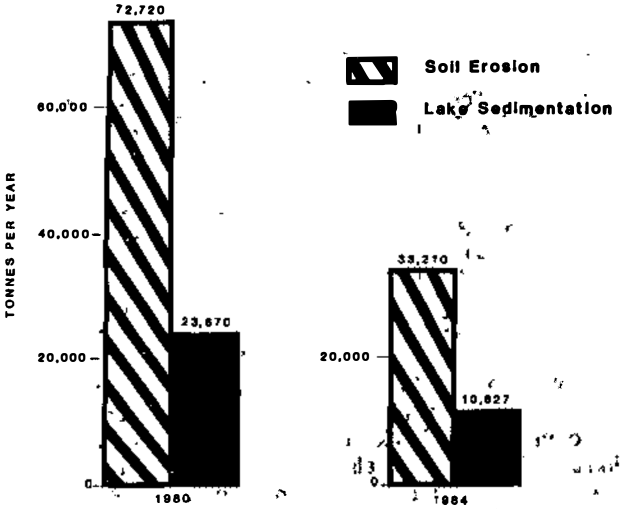

Figure 2.—Reduction in soil erosion and lake sedimentation, 1980–1984.

shed from excessive soil erosion—substantial progress toward the project goal of erosion control on 80 percent of the eligible watershed area.

Soil losses in the lake watershed have been reduced from a pre-project level of 72,720 tonnes (80,800 tons) per year to a current level of 33,210 tonnes (36,900 tons) per year. Assuming a sediment delivery ratio of 32 percent, the annual sediment delivery to the lake has been reduced 55 percent, from a pre-project level of 23,670 tonnes (26,300 tons) to a current level of 10,627 tonnes (11,808 tons). The reductions in soil erosion and sediment delivery are shown graphically in Figure 2.

The reduction in sediment delivery has created a parallel reduction in the delivery of sediment-borne nutrients and pesticides. Implementation of the nutrient and pesticide management program is reducing these pollutant loads even further. For example, data from participating farmers indicate that the nutrient management program has influenced their use of fertilizers, with average application rates of phosphorus (as P_2O_5) declining from 49.3 and 61.7 kg/ha (44 and 55 lbs/acre) for corn and soybeans, respectively, in 1982 to 22.4 and 6.7 kg/ha (20 and 6 lbs/acre), respectively, in 1984.

Several activities besides the RCWP project have been undertaken to improve the lake. During 1982, the Shelby County Board of Supervisors reconstructed a road adjacent to the lake. As part of the reconstruction, a bridge spanning the upper arm of the lake was replaced by a box-inlet culvert that temporarily impounds watershed runoff above the road, thereby allowing soil particles to settle out before the runoff enters the lake.

In the fall of 1981 the ICC initiated a complete fish renovation project for the lake. At that time lake levels were reduced and all fish were killed. The ICC has since conducted a fish restocking program, and has completed several projects to improve fish habitat and public access.

WATER QUALITY MONITORING PROGRAM

A monitoring program tracks both water quality and water-quality-related data. These data include records of lake attendance, major use activities, fish population inventories, lake bottom profiles, and lake physical conditions. Water-quality information is obtained from water samples collected at five in-lake locations from May through September. Fish and sediment analyses are being performed on an annual basis.

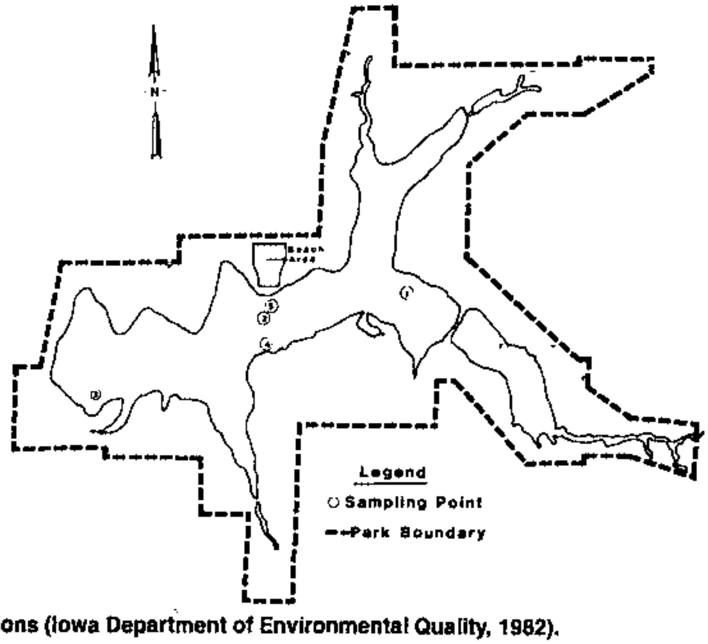

Figure 3.—Sampling locations (Iowa Department of Environmental Quality, 1982).

Figure 3 shows the five in-lake sampling locations (Iowa Dep. Environ. Quality, 1982):

Site	Description	Maximum Depth	Type of Sampling
1	Upper Arm of Lake	2.4 meters (8 feet)	fixed schedule
2	Mid-Lake South of Swimming Beach	3.3 meters (11 feet)	fixed scheduled
3	Deepest Part of Lake—Near Dam	7.2 meters (24 feet)	fixed schedule
4	Drinking Water Intake	4.5 meters (15 feet)	after runoff
5	Swimming Beach	3.3 meters (11 feet)	after runoff

Samples are collected from sites 1, 2, and 3 biweekly and analyzed for a variety of water quality parameters. Site 4, the drinking water intake, is sampled once annually following a 5-cm (2-inch) rainfall and analyzed for pesticides and heavy metals. Site 5, at the swimming beach, is sampled following all rainfall events greater than 2.5 cm (1 inch) from June through August and analyzed for fecal coliform.

The DWAWM presents monitoring results in annual water quality monitoring reports. These reports discuss both the current monitoring results and long-term water quality trends. The first of these reports, Prairie Rose Lake Monitoring RCWP Project—Year 1 (1981), also reviewed all data available from previous monitoring activities. Since insufficient data existed on pre-1981 lake conditions, it has been necessary to assume that 1981 monitoring results reflect preproject conditions.

IN-LAKE WATER QUALITY CHANGES

Since its start in 1981, the monitoring program has yielded a large amount of water-quality and water-quality-related data. Data analysis indicates that several significant changes have occurred in the lake since the beginning of the RCWP project.

Observations by Prairie Rose Park personnel indicate that the BMP's installed in the lake watershed have significantly reduced the impact of watershed runoff on the lake's water quality. Their observations indicate that, in contrast to the preproject situation where lake turbidities routinely increased immediately following runoff events, lake turbidity levels now do not change appreciably following runoff events.

Monitoring results for turbidity, Secchi depth, and chlorophyll a confirm that lake water quality conditions have changed appreciably. As compared to 1981 levels, lake turbidities decreased dramatically in 1982, with surface and bottom turbidity levels dropping 33 and 50 percent, respectively. At the same time, lake clarity increased significantly, with 1982 Secchi depths being nearly three times those measured in 1981. Since 1982, turbidity levels have been increasing, with the 1984 mean surface turbidity values being about the same as those measured in 1981. Table 3 presents the turbidity data for 1981 through 1984.

While the trend of increasing turbidity values in 1983 and 1984 is of some concern, analysis of other monitoring data indicates these turbidity levels are due mainly to increased algal growths rather than being sediment related.

Chlorophyll a data collected in 1981 showed relatively low levels of algal growth, indicating that the high turbidities found at that time were due mainly to the high sediment loads entering the lake in runoff, as well as by sediments being stirred up by carp and other bottom feeding fish. The 1982 monitoring results were somewhat of an anomaly, since chlorophyll a levels decreased, even though the low turbidity and high water clarity levels measured that year would be expected to result in increased algal growth. Chlorophyll a values did increase significantly in 1983 and 1984, with mean surface values at sites 2 and 3 over twice the values measured in 1981. These increases indicate that lake turbidity is now related primarily to algal growth, rather than sediment loads. Table 4 presents the chlorophyll a values measured for 1981–1984.

Algal assay results have generally shown phosphorus to be the limiting nutrient in Prairie Rose Lake, indicating that further BMP implementation efforts should emphasize the use of practices effective in reducing phosphorus delivery to the lake. However, since high levels of phosphorus currently exist in the lake and its sediments, reducing the annual phosphorus load to the lake is unlikely to reduce algal growth levels in the short term. Other lake restoration measures, such as precipitation of phosphorus from the water column or removal of bottom sediments by dredging, may be required if significant reductions in algal growths are to be achieved more quickly.

Water samples taken at the swimming beach have gen-

Table 3.—Mean turbidity (NTU's) and ranges (Wnuk, 1984).

	Site #1		Site #2		Site #3	
	Surface	Bottom	Surface	Bottom	Surface	Bottom
1981 Mean	20.7	31.1	10.7	102	8.8	84.3
1981 Range	4.9-75	12.0-85	5.1-23	17.2-540	1.9-15	13.0-340
1982 Mean	7.1	14.7	3.0	12.2	2.7	10.3
1982 Range	2.8-32	4.4-44	1.5-4.8	2.1-22	1.2-4.2	4.7-16.0
1983 Mean	12	15	8.2	19	7.5	18
1983 Range	2.3-28	2.7-24	1.6-24	1.9-60	1.6-20	2.0-55
1984 Mean	15.8	17.5	11.5	19.2	10.9	14.8
1984 Range	3.5-31	5.3-32	2.5-20	4.6-50	2.3-18	5.5-28

Table 4.—Mean chlorophyll-a (μg/l) and ranges observed (Wnuk, 1984).

	Site #1		Site #2		Site #3	
	Surface	Bottom	Surface	Bottom	Surface	Bottom
1981 Mean	3.7	33	21	24.4	17.3	24.1
1981 Range	16.0-85.0	14.0-68.8	12.0-38.0	16.0-38.0	9.0-33.0	7.0-87.0
1982 Mean	12	14	13	16	12	15
1982 Range	3-29	7-30	3-27	3-43	4-24	4-28
1983 Mean	40	41	43	24	39	21
1983 Range	4-98	3-73	3-145	3-67	3-120	3-65
1984 Mean	60	54	51	29	46	29
1984 Range	21-116	16-105	16-94	8-92	7-102	6-97

erally been below the 200 organisms per 100 ml fecal coliform standard established for primary contact waters in Iowa's Water Quality Standards, indicating the water is safe for swimming. Similarly, although several pesticides were detected in samples collected at the drinking water intake, the levels found are well below the levels considered harmful to human health.

ICC records indicate that lake use has fluctuated considerably from year to year, largely because of climatic factors. Total use levels are generally increasing, however, with total use in 1984 reported at 129,147 user days. Use of the swimming beach has increased annually, going from a 1981 total of 55,279 user days to a 1984 level of 75,500 user days.

Fishing at the lake declined in 1982 and 1983. This decline was expected, since the fish stocked in the lake following the 1981 fish renovation had generally not reached a catchable size. Fishing increased in 1984 and is expected to continue increasing as the lake's fishery develops.

OBSERVATIONS ON PROJECT

At this time several general observations can be made regarding the Prairie Rose RCWP project:

1. Considerable progress has been made toward accomplishing the project's goals. Complete achievement of these goals is expected during the project's lifetime.
2. A major factor in the project's success has been the willingness of local farmers to participate in it. Their participation can be attributed to a number of factors, including: their awareness of the lake's water quality problems; extensive efforts by county SCS, ASCS, and CES staffs to inform farmers about the project and solicit their participation; and the higher cost share rates available under the RCWP than under other State or Federal cost share programs.
3. A large number of local, State, and Federal agencies are cooperating in the project. This cooperation has not only helped to ensure the project's success, but has also resulted in completion of several related projects which increase the lake's public value. For example, the reconstruction of the county road through the upper arm of the lake has provided a secondary sedimentation basin for runoff entering the lake, and the fish renovation project will greatly improve the lake's recreational value.
4. The water quality changes observed thus far point out that implementing BMP's in a lake's watershed will not necessarily correct all of the lake's water quality problems, and additional measures may be needed to deal with remaining problems. In the case of Prairie Rose Lake, the reduction of sediment loads appears to have resulted in increased algal growths. Although these algal growths may eventually decline as a result of lower annual phosphorus loadings entering the lake, the high phosphorus levels currently found in the lake and its sediments make short-term reductions unlikely.
5. Although it may appear that the Prairie Rose RCWP project has simply changed the lake's water quality problem from one of excessive sedimentation to one of high algal growth, it is important to recognize that the sedimentation problem was threatening the very existence of the lake itself. While high algal growths may be aesthetically objectionable during certain periods of the year, the problems associated with high algal growth are minor by comparison.

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AGRICULTURAL SOURCES OF NITRATE CONTAMINATION IN A SHALLOW SAND AND GRAVEL AQUIFER OF EASTERN SOUTH DAKOTA

JEANNE GOODMAN
South Dakota Department of Water and Natural Resources
Pierre, South Dakota

ABSTRACT

The South Dakota Department of Water and Natural Resources initiated a water quality study of the Big Sioux aquifer in eastern South Dakota because of growing concerns over potential water quality degradation in the aquifer. Reports of unacceptable levels of nitrates (>10 mg/L No_3-N) in 25 percent of available domestic well samples and in several community water systems were of particular concern. A random sampling network was established to characterize the general water quality of the aquifer. Results of the random network sampling indicated nitrate contamination was generally confined to numerous localized rather than widespread areas. Specifically, 37 percent of 27 domestic well samples exceeded the United States Environmental Protection Agency (EPA) drinking water limit of 10 mg/L No_3-N. The major agricultural sources of nitrates were animal wastes from feedlots and accidental releases of fertilizers, with the contamination problems compounded by improper well construction and poor location. Two special studies were conducted near rural community water supplies contaminated with excessive concentrations of nitrates. Monitoring in the Egan, South Dakota, area indicated contamination with concentrations reaching 240 mg/L NO_3. The source of contamination was determined to be leaking liquid fertilizer tanks near the city well. Monitoring in Elkton, South Dakota, also indicated the most likely source of contamination was a fertilizer distributor. A plume of contamination was defined with a high concentration of 67 mg/L NO_3-N in a well near the distributor. Because of these findings, a public education program was initiated as part of the Big Sioux Aquifer Water Quality Study. This program focused on nitrate contamination sources, means of contamination prevention, and recommendations for well construction and location.

INTRODUCTION

Agriculture is the predominant land use in eastern South Dakota and is characterized by small diverse farms averaging 175 ha (Census of Agriculture, 1982). Corn, soybeans, and small grains are the major cash crops raised with some feed crops grown for small livestock operations. Draining the eastern counties, the Big Sioux River drainage basin supports the typical agricultural activities of eastern South Dakota (Fig. 1).

The Big Sioux aquifer is a shallow glacial aquifer associated with, and underlying the floodplains of the Big Sioux River and its tributaries. The aquifer, the major water resource in the basin, consists of sand and gravel outwash and alluvial material averaging 9 m (30 feet) in thickness in most areas. Because these permeable aquifer materials are at or near the land surface, the aquifer is susceptible to contamination by land surface activities.

The susceptibility of surficial aquifers to surface contamination is evidenced by the South Dakota Department of Water and Natural Resources' (DWNR) water quality data base for shallow wells (less than 50 feet in depth). In 1979, 25 percent of the available samples exceeded the United States Environmental Protection Agency's (EPA) Maximum Contaminant Level for public drinking water supplies of 10 mg/L nitrate-nitrogen (NO_3-N). Since the Big Sioux aquifer is the primary source of drinking water for approximately one-third of the State's population and is susceptible to contamination, a possibility that the aquifer was undergoing widespread water quality degradation prompted the Big Sioux Aquifer Water Quality Study. This study was initiated to determine the general water quality of the aquifer and define the areal extent and sources of nitrate contamination.

A random network of observation and domestic wells was established and sampled to characterize the general water quality of the Big Sioux aquifer. Also, several special studies were conducted in areas of known nitrate contamination. Results of the random network sampling indicated nitrate contamination was not widespread but consisted of numerous localized problems. Several sources of nitrates were identified including landfills, sewage logoons, feedlots, septic tanks, and accidental releases of nitrogen-based fertilizers. The following discussion, however, focuses on agricultural sources of nitrates causing contamination of domestic and public drinking water supplies.

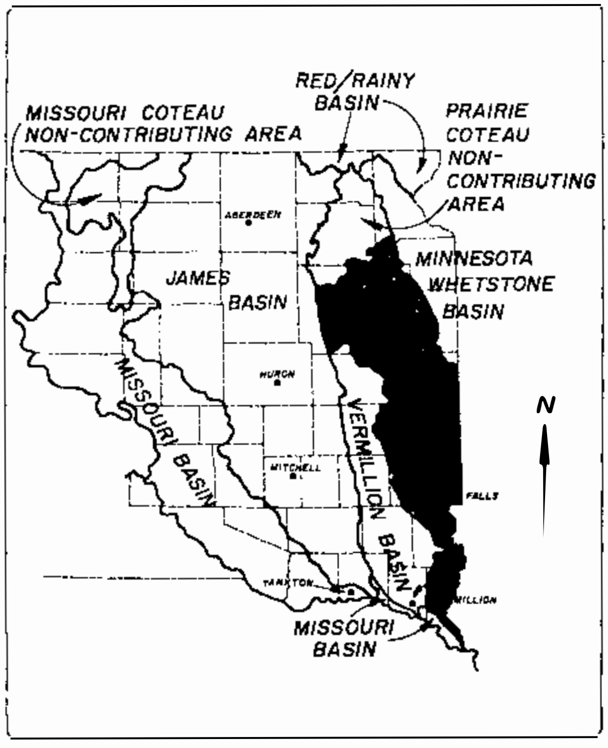
DISCUSSION

Domestic Wells

The predominantly rural population in the basin relies primarily on shallow wells constructed into the Big Sioux aquifer for domestic use and stock watering. Past sampling has revealed that elevated nitrate levels in domestic wells are common in the aquifer with nearly 25 percent of 136 domestic wells exceeding the EPA drinking water limit. The severity of the problem was illustrated during the random sampling conducted for the Big Sioux Aquifer Water Quality Study when an alarming 37 percent of the domestic wells sampled exceeded 10 mg/L NO_3-N, with nitrate concentrations ranging from <0.1 to 120 mg/L. In comparison, less than 9 percent of the sampled observation wells exceeded 10 mg/L NO_3-N with a range of <0.1 to 50 mg/L (South Dakota Dep. Water Nat. Resour. 1985).

Several problem areas were identified as contributing to the contamination of domestic wells. First, the leaching of nitrogen from animal wastes in feedlots was identified as the main source of nitrogen. The number of livestock per farm is usually less than 100. However, over 80 percent of the farms in the Big Sioux River Basin have some type of livestock operation (South Dakota Dep. Water Nat. Resour. 1985). Therefore, feedlots are a common source of nitrogen.

Second, the location of the domestic well was a problem. Frequently, the contaminated wells were located close to (and in some cases within) a livestock containment area and therefore, close to a contamination source. In addition, if the shallow water table mimics the topography, many of the wells sampled were downgradient from the contamination source. Unfortunately, the wells used



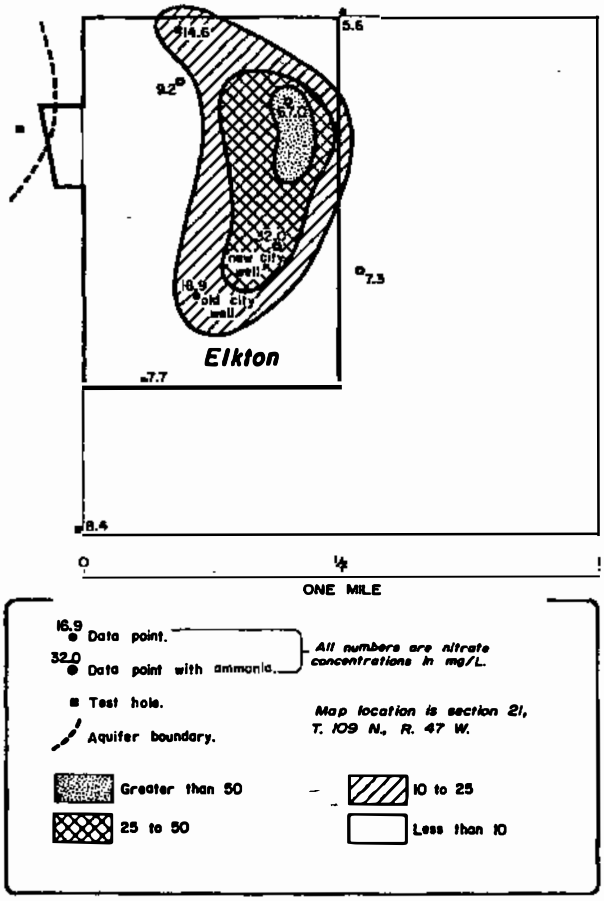
From Stach,et.al., 1984

Figure 1.—Location map of Big Sioux Basin.

for stock watering were often also used for domestic supplies, thereby supplying contaminated drinking water for humans.

Third, improper well construction allowed nitrate contamination of domestic wells. Domestic wells often lacked a sufficient surface seal to prevent contaminated surface runoff from entering the well annulus (the area between the well casing and the sides of the bore hole). Additionally, since 40 percent of the domestic wells constructed after 1976 were bored, many wells were cased with large-diameter porous concrete, wood curbing, or similar materials which allow the infiltration of any surface runoff entering the well bore into the well. Since many bored wells were completed to a depth just below the water table they were likely to have higher nitrate concentrations than if constructed deeper into the aquifer. The water quality data from nested observation wells (more than one well at a site) in the Big Sioux aquifer indicated higher nitrate values in wells constructed in the upper portion of the aquifer than in wells constructed into the base of the saturated material.

Case Study 1. A contaminated domestic well that illustrates the three problems discussed above was investigated when a case of infant methemoglobinemia (blue baby) was reported in 1979 to DWNR. The field investigation revealed that the well, used for drinking water and stock water, was a shallow, large diameter, bored well. It was cased with porous concrete and was located in a depression down-slope from two hog confinement areas and from the septic tank drainfield. The feedlots were active and had been used for several years. It appeared that feedlot runoff and leachate as well as septic leachate were the sources of nitrates. The poor location and improper well construction compounded and magnified the contamination potential. Well samples had nitrate concentrations varying from 120 mg/L to 210 mg/L NO_3-N (Busch and Meyer, 1982). This situation was typical of domestic wells sampled where nitrate concentrations exceeded the EPA drinking water limit of 10 mg/L NO_3-N.



From Stach, et.al., 1984

Figure 2.—Nitrate concentrations in the Elkton, South Dakota, area.

Public Supply Wells

In 1979, 33 public supply wells, providing water from the Big Sioux aquifer, exceeded the EPA drinking water limit of 10 mg/L NO_3-N. As part of the Big Sioux Aquifer Water Quality Study, several special studies were conducted in areas near selected contaminated wells. Although there were several probable sources of nitrogen, mishandling of commercial fertilizers was the primary agricultural source. As in the contaminated domestic well situation, the location of the public wells close to the fertilizer distributors magnified the potential for nitrate contamination.

Case Study 1. Elkton, South Dakota—Elkton had consistently exceeded the EPA drinking water limit for nitrates of 10 mg/L NO_3-N. The town derived water from two wells constructed into the Big Sioux aquifer. A special investigation was conducted in and near Elkton to delineate the extent of nitrate contamination and to attempt to identify the source of the contamination.

Several observation wells were constructed from which water level elevations were measured and water samples were taken. Nitrate values within the town limits ranged from 7.3 mg/L to 67 mg/L NO_3-N (Stach, et al. 1984). As illustrated in Figure 2, a plume of nitrate contamination was centered near the north end of Elkton and obviously affected the public drinking water supply. Field investigations indicated the location of the highest nitrate concentration was near a commercial fertilizer dealer. Spilled fertilizers or equipment rinsate had reached the ground water in the area causing a contamination problem for Elkton.

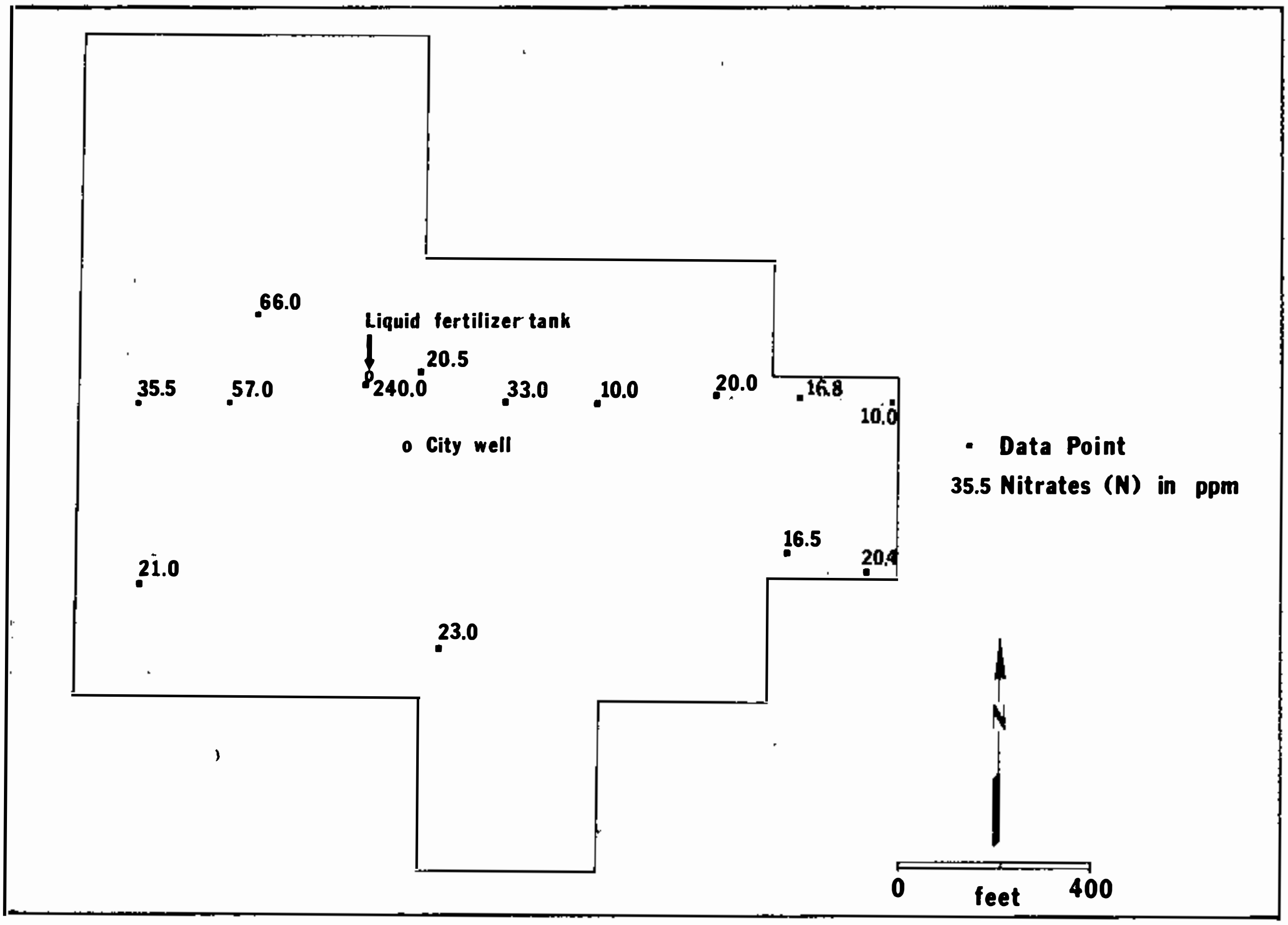


Figure 3.—Egan, South Dakota, nitrate study.

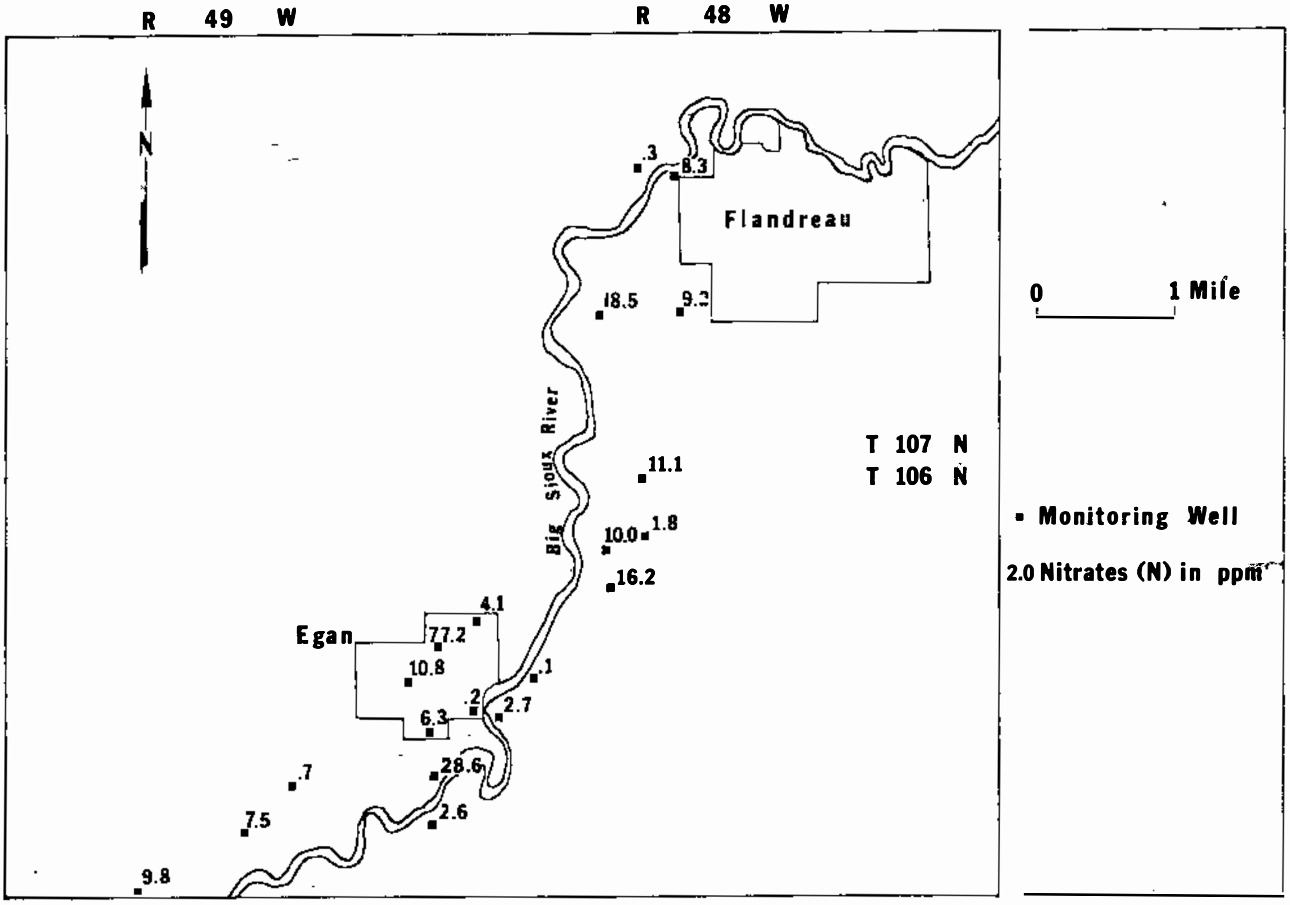


Figure 4.—Nitrates at Flandreau-Egan, South Dakota.

Case Study 2. Egan, South Dakota—Egan also consistently exceeded the EPA drinking water limit of 10 mg/L NO_3-N. The concentration in the public well drastically increased from 25 mg/L to 70 mg/L NO_3-N in 1979. Egan initiated a study which revealed nitrate contamination of ground water with a high value of 240 mg/L NO_3-N in the immediate vicinity of several liquid fertilizer storage tanks (Fig. 3; Stach et al. 1984). The public well was approximately 1 block from the tanks.

As a special study for the Big Sioux Aquifer Water Quality Study, an area including Egan and Flandreau, South Dakota, was investigated in 1982 to determine the extent and sources of nitrogen contamination. In an attempt to define the areal extent of the 1979 fertilizer spill, subsequent sampling of ground water in the contaminated area of Egan indicated that even though no further fertilizer releases had occurred, nitrate concentrations remained elevated with 77.2 mg/L NO_3-N detected near the fertilizer tanks (Fig. 4; Stach et al. 1984).

CONCLUSIONS

The results of the Big Sioux Aquifer Water Quality Study in eastern South Dakota indicate that the nitrate contamination occurs in numerous localized areas rather than being widespread throughout the aquifer. Several sources of contamination were identified. Among these were agricultural sources of nitrates causing contamination of domestic wells and public supply wells.

Generally, the major problem areas resulting in high nitrate concentrations in domestic wells were (1) feedlots or livestock containment areas as sources of nitrogen, (2) location of the well relative to the source providing contaminated drinking water, and (3) improper well construction allowing contamination. Public supply wells with nitrate concentrations exceeding the EPA drinking water limit had the following common problems: (1) improper handling or accidental releases of commercial fertilizer as a source of nitrogen and (2) poor location of the public well relative to the source providing contaminated drinking water.

To disseminate information gained from the study, a public education program was initiated by the East Dakota Conservancy Sub-District (a locally elected and State-funded water district). In addition to making public presentations, the agency prepared three pamphlets presenting the basic data and information gathered from the Big Sioux aquifer study stressing the importance of proper well construction and location. It is the intention of DWNR to use the Big Sioux aquifer study to further educate the general public to prevent contamination of the aquifer and provide a safe source of drinking water.

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