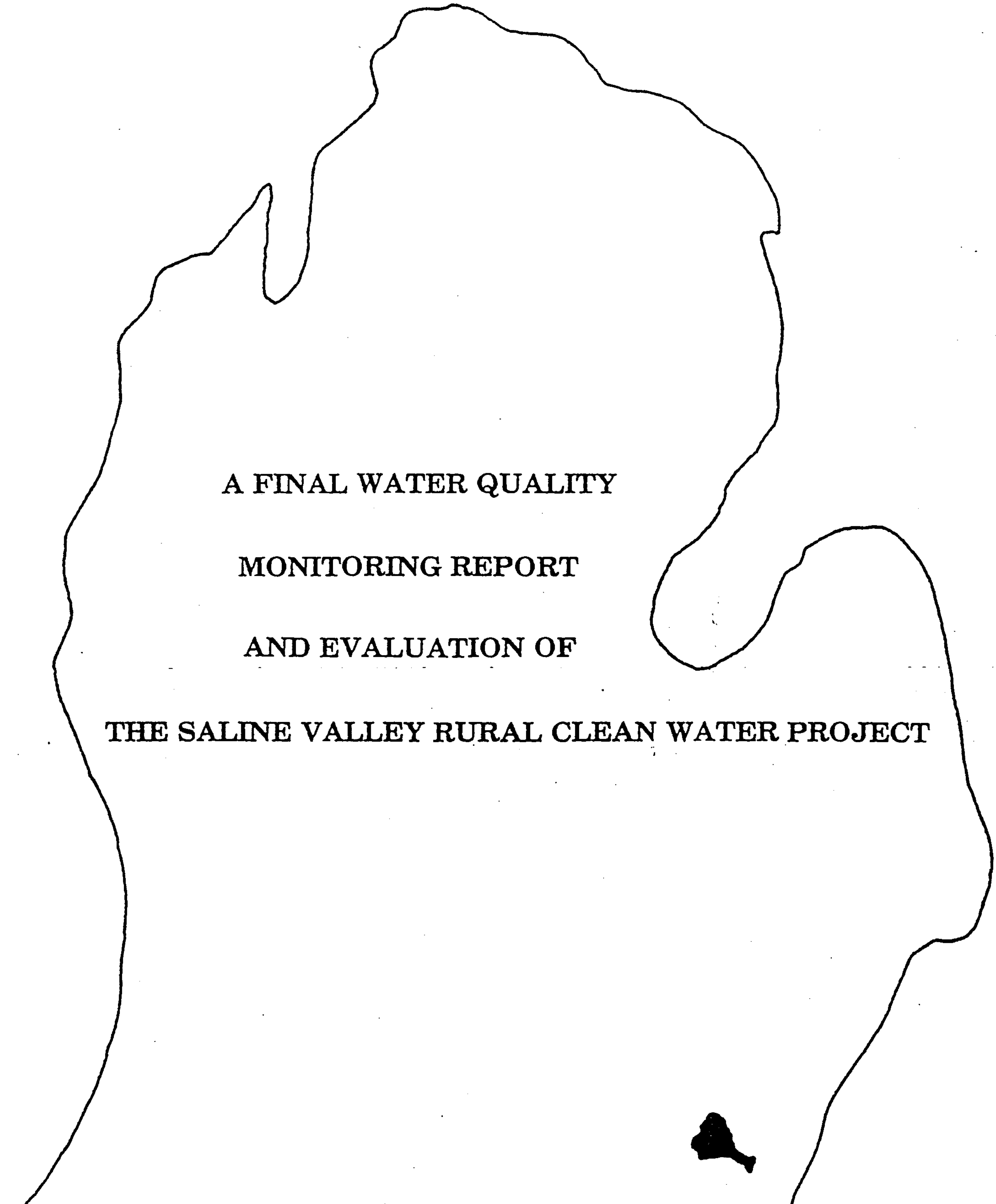
A FINAL WATER QUALITY MONITORING REPORT AND EVALUATION OF THE SALINE VALLEY RURAL CLEAN WATER PROJECT



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A FINAL WATER QUALITY MONITORING REPORT AND EVALUATION OF THE SALINE VALLEY RURAL CLEAN WATER PROJECT

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EXECUTIVE SUMMARY

Saline Valley was one of twenty national sites selected by the U.S. Department of Agriculture for inclusion in the Rural Clean Water Program (RCWP). Goals of the Saline Valley project were (1) to evaluate whether a volunteer-based land treatment approach, using cost-share incentives, would provide adequate participation, (2) to reduce phosphorus loads from the area by 40 percent, and (3) to quantify relationships between land treatment and resulting water quality changes. Water quality monitoring was carried out from 1981 through 1989 at both stream and groundwater sites. Surface sites were sampled approximately weekly and well sites were sampled on a variable schedule ranging from monthly to quarterly.

Loading and concentration data for suspended solids and nutrients were highly variable both within and between years. This variability, along with low levels of participation by landowners, prevented the project from documenting water quality changes resulting from applied best management practices (BMP). Land treatment-water quality relationships were examined by correlating annual trends in, (1) regressions of pollutant concentration versus discharge and (2) discharge-normalized mean concentrations, against areal estimates of BMP implementation.

Monitoring wells revealed that leakage from animal waste storage structures produced elevated nutrient concentrations in groundwater at the down-flow well site. Contamination was not, however, observed in pre-existing wells supplying drinking water and did not appear to pose any potential health threat. These result do, however, indicate a potential

conflict between improving surface water quality at the detriment of groundwater.

Monitoring data at the watershed's downstream terminus (station 8) indicated that a 56 and 71 percent decrease in total and soluble phosphorus loads respectively occurred after 1986. Reductions were obtained through point source control programs which included design upgrades at the wastewater treatment plants of Saline and Milan and the elimination of an industrial-waste discharge. The nonpoint source contribution to phosphorus loads observed at station 8 were calculated to be around 0.5 kg P/ha/yr, which equals the national average for agricultural land.

Monitoring results confirmed the importance of identifying all pollutant sources and their relative magnitudes before attempting to assess the effectiveness of a nonpoint source control program. The project's ability to meet its goals were limited by the facts that no specific water quality impairment was documented within the watershed and that loading rates from nonpoint sources were not as severe as expected.

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INTRODUCTION

Over the past decade there has been increased awareness of the impact of nonpoint source pollution on water resources. Easier identification and better control technologies for point source pollution has led to nonpoint sources accounting for a larger percentage of the total pollutant discharge (Sharpley et al., 1989). The U.S. Environmental Protection Agency reported to Congress that six of ten EPA regions found nonpoint source pollution to be the remaining cause of water quality impairments (U.S. EPA, 1984). It has been estimated that nonpoint sources now account for 99, 63, and 62 percent of the nation's pollutant loadings for suspended solids, phosphorus, and nitrogen respectively (Gianessi et al., 1986). Although sources of nonpoint pollution vary from region to region, agricultural activities have been identified as being the most significant and pervasive. This finding reflects the fact that nearly 63 percent of all non-federal land in the U.S. is used for agricultural purposes, including cropping and livestock production (Meyers et al., 1985). Agricultural activities may contribute up to 53, 67, and 74 percent of the sediment, phosphorus, and nitrogen loadings respectively of all nonpoint source pollution (Gianessi et al., 1986). These pollutants have been identified as a primary cause of increased eutrophication of the nation's lakes and streams (Vollenweider and Kerekes, 1980).

In 1980, the federal government created the Rural Clean Water Program (RCWP) to address this negative impact of agricultural nonpoint pollution on our nation's water resources. The RCWP was an experimental program designed to promote cooperation among existing federal, state, and local agencies and to control agricultural nonpoint pollution at the watershed level by soliciting voluntary participation by landowners. The RCWP had three primary goals; (1) to improve water quality and beneficial uses in the most cost-effective manner possible, (2) to help rural landowners and farmers practice nonpoint source pollution control, and (3) to develop and test programs, policies, and procedures designed to control agricultural nonpoint source pollution (NWQEP, 1989). Control strategies were designed around voluntary implementation of best management practices (BMPs) by local landowners. BMPs refer to a variety of agronomic practices and structural devices that are designed to reduce the transport of sediment, nutrients, and toxics from the watershed into surrounding water resources (Table 1.1). In addition to improving water quality, BMPs are also designed to sustain producer profitability by maximizing the conservation aspect of farming. A voluntary approach to agricultural nonpoint source control will not succeed unless BMPs meet both objectives (NWQEP, 1989). Landowners were encouraged to participate in the project through educational programs offered by the Cooperative Extension Service, cost-share incentives provided by the Agricultural Stabilization and Conservation Service and technical assistance from the Soil Conservation Service.

There were twenty projects nationwide within RCWP. RCWP project sites were selected from state lists of nonpoint pollution priority watersheds developed under section 208 areawide management plans. The Saline Valley watershed was chosen as a study site because it was identified as having one of the highest areal phosphorus loading rates in southeast Michigan, and was contributing excessive phosphorus loading to Lake Erie. The initial project area covered 125,000 acres and contained watersheds of both the Huron and Saline rivers. Over 70 percent of this area is under intensive agricultural production, including cropping and livestock operations. A water quality monitoring program was initiated in July of 1981 and provided continuous sampling on an approximately weekly basis until the termination of the study in December of 1989. The goal of the monitoring program was to characterize patterns of sediment and nutrient loading to area streams and to document any water quality changes which occurred as a result of the application of BMPs within the watershed.

Information regarding the effectiveness of BMPs to control agricultural nonpoint pollution at the watershed level is just beginning to emerge as result of RCWP. Results within the program, however, have not always been consistent. Only a few projects have been able to document significant reduction in pollutant loading as a result of BMP implementation (NWQEP, 1989). Consequently, many of the relationships between land treatment and water quality remain unclear.

CHAPTER I

WATER QUALITY MONITORING RESULTS

Introduction

The Saline Valley watershed has large areas of active cropland and medium sized livestock operations. This intensive agricultural activity within the watershed has resulted in ammonia concentration high enough to be toxic to aquatic life and phosphorus concentrations high enough to accelerate eutrophication of aquatic systems (Beeton, 1982). The watershed was identified in section 208 areawide management plans as having one of the highest per area loading rates of phosphorus in southeast Michigan (Anon., 1985). The Saline Valley project was initiated to alleviate this pollution.

The overall goal of the project was to reduce phosphorus loading from the watershed by 40 percent. Strategies designed to meet this goal called for reducing phosphorus losses from fertilizer application by 50 percent, losses from animal waste sources by 30 percent, and losses from sediment-bound phosphorus derived from soil erosion by 30 percent. These sources of phosphorus loss from the farms often have no agronomic significance, but they do represent significant loadings to lakes and streams and lead to accelerated rates of eutrophication (Sharpley, et al., 1988). This concept is one of the primary lessons about which RCWP wanted to educate landowners.

A water quality monitoring program for Saline Valley RCWP was initiated in July of 1981 and terminated in December 1989. The goal of the monitoring program was threefold: first, to characterize temporal and spatial patterns in pollutant concentrations and loadings from the watershed, second, to evaluate whether a volunteer-based land treatment program could significantly reduce sediment and nutrient loadings to area streams, and third, to quantify the relationship between any observed water quality changes and BMPs.

The design and scope of the monitoring program were limited by both RCWP protocols and by inadequate funding. Saline Valley was not selected as one of the five comprehensive monitoring and evaluation projects for RCWP, and therefore funding for the monitoring program was the responsibility of local sponsors. Funding was obtained from a variety of sources (see Acknowledgements) but was limited to between $15,000 to $23,000 dollars a year throughout the study. These levels did not allow the project to establish permanent gauging stations or to purchase automated samplers for monitoring storms on a detailed basis. RCWP's protocols limited the monitoring design because an initial water quality data base could not be established prior to the beginning of BMP implementation. Additionally, the project was not able to control the amount, timing, or location of the land treatment being applied. RCWP's policy of relying on voluntary participation resulted in BMPs being established on a gradual basis throughout the entire study area and over the duration of the study period. This approach did not allow the monitoring project to use a before/after treatment design or a paired watershed design to evaluate water quality changes. Instead, the project relied on time-trend analyses. Due to the high degree of temporal and spatial variability associated with nonpoint source pollution this approach proved quite difficult, even given an unprecedented long period of record.

This report describes concentration and loading patterns and land treatment implementation at Stations 3 through 9 for water years 1982 through 1989. The water year refers to a time period from December 1 to November 30 and is delineated into four seasons of equal length as Winter (December-February), Spring (March-May), Summer (June-August), and Fall (September-November).

In the need to be concise data presented here focuses primarily on results for station 9. This station had nutrient concentration and loading patterns typical of the other station but had the advantage of being the only sub-basin with levels of land treatment high enough to test for differences between years. When possible results are compared against other sub-basins to examine patterns of spatial variability. Results for the Saline river stations 5 and 8 are also discussed to describe the effects of point source control efforts which occurred within the watershed during the study. Results for stations 1 and 2 are ommitted from this report because monitoring was discontinued after 1984 (see page 9). Additionally, station 3A has been ommitted from most comparative figures because of its unequal record of observation and lack of significant deviation from station 5. Complete sumations of all monitoring data are available from the authors in a series of annual monitoring reports (Beeton, 1982; Beeton et.al., 1984; Holland et.al.; 1985, 1986, 1987, 1988, 1989; Johengen et.al., 1990). Annual loads of all parameters monitored at stations 3 through 9 are summarized in appendix A.

Study Area

The Saline Valley project watershed initially included the Saline River Basin and part of the Huron River Basin (Fig. 1.1). Eight sampling stations were established in 1981 to determine trends in water quality. Several revisions occurred to project boundaries and station locations during the study period. In 1983 the project was scaled down from an initial area of 125,000 acres to 77,000 acres because limited manpower and dollars did not allow for adequate levels of participation. Redefined boundaries eliminated Stations 1 and 2, which were located in the Huron River watershed. Also at this time, station 9 was added on Macon Creek, a tributary of the Raisin River. The final revision to the project occurred in 1985 with the addition of Station 3A on the Saline River, upstream from the tributaries sampled.

The revised watershed has been classified as 95% rural and contains a population of approximately 19,000 people in the two towns of Saline and Milan (Anon, 1989). About 70% of the area is intensively cropped, with a distribution of 50% corn, 25% small grains, 13% pasture, and 12% soybeans. There are 351 farms within the watershed which contain a total of approximately 9,500 livestock.

Of the final eight surface water monitoring stations, five were located on tributaries of the Saline and Raisin rivers to allow evaluation of land treatment at the sub-basin level. The remaining three stations were located on the river with station 3A being upstream of all sub-basins, station 5 being upstream of the city of Saline, and station 8 at the confluence with the Raisin River (Fig. 1.1). With the exception of station 8, sampling sites were selected to avoid impacts from point source loadings. Station 8 represents the downstream terminus of the project watershed and was established at the request of the Southeast Michigan Council of Governments. A brief description of each station and their respective drainage basins is given below.

Sampling Locations

<u>Station 3: Saline-Bridgewater Drain</u>

Station 3 is a short distance upstream from the confluence of the drain and the Saline River at Feldkamp Road. The 3,969 acre watershed is in Saline and Bridgewater Townships in Washtenaw County. There is no known point sources of pollution within the watershed. The drain was dredged in June 1983 and a new hydrograph was established by project personnel with water level referenced from the top of the culvert pipe.

<u>Station 3A: Saline River</u>

Station 3A is immediately upstream of the confluence of the Saline-Bridgewater drain. It was established in October, 1984 to cover the 14,010 acres which were west of the existing sampling stations. This portion of the watershed contains three lakes and 3 major drains. No hydrograph was developed and discharge was calculated by subtracting the sum of flow for stations 3 and 4 from the flow for station 5.

<u>Station 4: Bauer Drain</u>

Station 4 is immediately east of Austin Road. The 4,900 acre watershed is in Saline, Lodi, and Freedom Townships in Washtenaw County. No known point sources of pollution exist within this watershed. The hydrograph was reevaluated by the MDNR in 1989 and found to be seriously under estimating discharge at most water levels. There was no known point where the change occurred so loads were not recomputed.

Station 5: Saline River

Station 5 is located at Dell Road is just downstream of where Bauer Drain enters the River. The initial staff gauge was lost in February of 1985 and thereafter water level was determined from a reference point on the bridge which had also been calibrated. Data from this station reflects conditions including the upstream stations 3, 3A, and 4. The additional watershed area between 3A and 5 is 6, 240 acres.

Station 6: Bear Creek

Station 6 is located where Wells Road crosses Bear Creek. The watershed covers 2,470 acres and is located in Milan and London Townships in Monroe County. Water level was measured from a reference point on the concrete bridge. Flow occasionally ceased in the summer during prolonged dry periods.

Station 7: Wanty Drain

Station 7 is located at Plank Road, above most of the residential area. The 1,920 acre watershed is located in London Township in Monroe County. The watershed is very flat and fields appear to be extensively drained.

Station 8: Saline River

Station 8 is located at Bigelow Road above the confluence of the Saline and Raisin Rivers in Dundee Township, Monroe County. The total watershed is 128 square miles (81,920 acres). Municipal and industrial point sources and urban nonpoint sources are known to affect water quality at this station. Water level is measured from a reference point on the bridge at Bigelow Road. Discharge was determined from a USGS hydrograph.

Station 9: Macon Creek

Station 9 is located at the bridge of Ridge Highway in Section 31 of York Township, Washtenaw County. The station monitors flow from the Macon Creek watershed which covers 9,728 acres. Water level was measured from a reference point on the bridge and from a staff gauge until it was lost in 1984. The hydrograph was developed by project personnel. It was recently reevaluated by staff at DNR and found to be overestimating discharge at low levels. All loading values were calculated with discharge values from the original hydrograph.

Materials and Methods

Sampling

The sampling scheme devised for the project consisted of taking grab samples and instantaneous discharge measurements at the eight stations on a fixed weekly schedule. Initially the sampling scheme was to be adjusted to include additional sampling during periods of storms and snowmelt. However, constraints imposed by time, money, and personnel limited the number of sampling periods to approximately 35 to 40 times per year after the first year. Sampling was still performed on a fixed schedule, i.e. a fixed day of the week, but often times a two week interval occurred between sampling.

Samples were collected with a bucket at midstream immediately below the surface. Field samples were collected by Washtenaw Soil Conservation District personnel and delivered to the laboratory of the Great Lakes Research Division, University of Michigan for analyses, within four hours of collection. Samples were processed according to the procedures outlined in figure 1.2. Instantaneous discharge was determined at the time of sampling by measuring water level readings from a fixed reference point or staff gauge and applying established rating curves. Rating curves were established initially by the U.S. Geological Survey and then later by project personnel when changes to the site occurred.

Groundwater was also monitored at three sites which had adopted the use of animal waste storage systems as part of BMP strategy. The three sites monitored included an earthen lagoon, a two pit solid-liquid system, and an in-ground concrete storage tank. Three wells were drilled at each site, one on the up-flow side of the storage structure and two down-flow to provide a means for determining if leakage from the systems would contaminate the groundwater. Well depths ranged from 25 to 30 feet and were constructed with 2" schedule 40 PVC pipe with a three foot PVC well screen attached, threaded caps, and vent holes. Wells were sampled with a hand bailer made of schedule 80 PVC pipe and line check vales. Bailers were rinsed twice before the actual sample was collected. Wellwater samples were processed according to the same procedures as surface-water samples with the exclusion of suspended solids.

Analytical

Water samples were analyzed for conductivity, pH, suspended solids, orthophosphate (soluble reactive), available phosphorus, total phosphorus, ammonium, nitrate-nitrite, silica, and chloride. As nitrite consistently represented less than one percent of nitrate, nitrate-nitrite is subsequently referred to as only nitrate. All phosphorus and nitrogen concentrations are expressed as P and N. Chemical analyses were performed on an Auto Analyzer II system. Analyses were performed according to standard technicon procedures as modified in the lab manual of Davis and Simmons (1979). Available P was determined by extracting material retained on a 0.45 um nucleopore filter in 0.1N NaOH. Total-P was determined on unfiltered water using a potassium persulfate digestion and then analyzing as orthophosphate. Suspended solids were determined gravimetrically for material retained on a Whatman GF/C filter and dried for 24 hours at 60 ^{o}C. The sequence used for these analytical procedures is shown in figure 1.2.

Loading calculations were performed by multiplying sample concentrations by the instantaneous discharge measurement and extrapolating over the midpoints of the sampling interval. The resultant values were summed to obtain estimates of seasonal and annual loadings by station.

Data for available P has been included in loading summaries in the appendicies but ommitted from figures in the text. Available P concentrations and loading patterns were very similar to those of soluble P and did not alter or enhance interpretations of project results.

Results and Discussion

Concentration Time Series

Time series plots of concentration and discharge values measured at station 9 reveal the extreme variability in pollutant loading both within and between years (Figs. 1.3 - 1.5). Concentration and discharge values varied by two and three orders of magnitude respectively, between base-flow and storm-flow conditions. Characteristic of most nonpoint source pollution, increases in pollutant concentration were caused and/or accompanied by increased discharge. These increases typically occurred after heavy spring melting and after rainfall events which were large enough to generate significant amounts of overland and subsurface runoff. The time series plot for discharge (Fig. 1.3) shows that several large runoff events occurred each year and that particularly large storms occurred in years 1983, 1986, and 1989. All chemical parameters exhibited similar responses to stormwater runoff periods, but suspended solids showed the strongest response and nitrate the least. Ammonia concentrations were more variable but did exhibit several extremely high values in response to storms, occasionally reaching the one mg/l range.

Concentration spikes produced by randomly generated storm events made it difficult to observe seasonal concentration patterns within years, or whether there were any long-term trends. A sliding, four-point boxcar average was applied to the time series to filter out the effect of outliers and elucidate seasonal and long-term trends (Figs 1.6 - 1.8). Filtered time series still contained high variability among years, but a general pattern of increased concentrations in both the spring and fall became more evident.

The filtering technique was not able to remove the extreme concentration spikes observed for phosphorus and ammonia in 1988 and 1989. These results indicate that individual storms may have an extremely large impact on annual concentration and loading values.

Mean and Median Concentration Trends

Annual mean concentrations for suspended solids, total phosphorus, soluble phosphorus, and nitrate at station 9 were plotted to examine general long-term trends (Fig. 1.9). Error bars are omitted because the extreme concentration ranges found within a given year result in standard deviation often greater than mean itself. Means were also highly variable between years. In general, suspended solids and phosphorus closely followed the pattern for total annual discharge (see Fig. 1.14). This relationship simply reflects the fact that means were sensitive to the high concentration values which occurred during storms. The pattern for nitrate, however, was quite different from the other parameters (Fig. 1.9). Mean nitrate values did not follow the discharge pattern, but rather exhibited an increasing trend from 1986 to 1989. One possible explanation for this pattern is that excessive fertilization has led to a build up of nitrogen in the soil which is being leached out to the stream. Examination of soil fertility test performed within the sub-basin could provide a test of this hypothesis.

To avoid the bias of extreme values on mean concentrations a nonparametric approach can be used looking at median values and distribution patterns between years. This procedure was done by creating box plots which give the 25^{th}, 50^{th}, and 75^{th} percentiles of the pollutant concentrations for each years measurements. Box plots for suspended solids, total phosphorus, soluble phosphorus, and nitrate are presented for each station in figures 1.10 - 1.16.

If BMPs significantly reduce sediment and nutrient losses, then both median concentration and the distribution spread above the median would be expected to decrease. With the exception of phosphorus concentration at station 8, there was no consistent decrease in either median concentration or spread for any of the stations. The pattern at station 8 resulted from point source control efforts and will be discussed in detail late. Another interesting trend occurred in the distribution spread for soluble phosphorus. While the spread appeared much lower after 1983 at stations 3, 4, & 5, ironically the median value tended to increase in these following years. So although the frequency and magnitude of high concentrations may have decreased the "average" concentration tended to increase.

In general, the size of the distribution spread between quartiles was highly variable between years at all station. Specific patterns reflect both the number and intensity of storms which occurred during that year as well as the timing of our sampling in relation to those storms. In all plots the spread between 50th and 75th percentiles was consistently greater than that between the 25th and 50th. This pattern can be misleading in that it does not indicate a greater number of high concentrations but rather a greater range in concentrations for this 25 percent of the sample distribution. These higher values are again reflective of the effect of a few large storms. A clearer depiction of the concentration distribution can be seen using frequency histograms. Figures 1.17 & 1.18 show the frequency distribution of observed concentrations at station 9 for all years sampled. These plots clearly reveal that most values are indeed at the low end, reflecting baseline or low flow conditions, with only a few high values representing the more infrequent storm conditions.

Annual Loads

Mean annual loads for suspended solids, nitrogen, phosphorus, and discharge for the eight year study period are listed by station in table 1.2. (Annual loads for all parameters and all stations are tabulated by year in Appendix A). As evidenced by standard deviations equal to 50 to 100 percent of the mean, inter-annual variability was extremely large. This variability was mostly due to a combination of meteorological differences between years and errors in the loading estimates themselves based on sampling methods. The large standard deviations in the mean loading data suggests that even an eight year record might not provide a reliable estimate of loading rates if they are based on weekly grab samples and instantaneous discharge measurements. If a continuous record of discharge had been available, empirical regressions of concentration and discharge could have been constructed to provide reliable estimates of concentration in between sampling events (eg. Johnson et al., 1976; Lathrop, 1986). Using this approach, weekly water quality samples would probably have been adequate to accurately characterize loading patterns.

Most of the inter-annual variation in pollutant loads could be explained by total annual discharge. For example, the magnitude of total and soluble phosphorus loads closely followed variations in annual discharge at each of the stations (Figs. 1.19-1.25). Most monitoring studies have found that differences in discharge have a much greater effect on loading values than do variations in concentration (Sharpley et al., 1976; Stevens and Smith, 1978; Hill, 1987). The importance of discharge suggests that

BMPs must not only be concerned with reducing sources of pollutants but also with how they might alter drainage patterns.

Plots of total annual discharge indicate there were large differences in drainage patterns among each of the sub-basins (Figs. 1.26 & 1.27). Differences exist not only in the amount of discharge for a given year, but also in the patterns of which years had high discharge. These spatial differences resulted from a combination of factors including precipitation patterns, characterisitics of the sub-basins such as size, topography, soil-type, and morphology of the drainage system.

One potential approach for assessing the effects of land treatment on water quality is to look for patterns where annual loads are less than would be predicted by the amount of discharge. This situation might suggest that the practices applied within the basin reduced pollutant losses. Such a pattern occurred at station 9 in 1989 (Fig 1.25). However, since less than 20 percent of the basin had BMPs applied at this time, it is difficult to assume a cause-effect relationship between the reduced load and these BMPs.

Indicative of the differences in discharge, loading rates were highly variable between stations (Table 1.2). This result reflects the fact that discharge values were much more variable between stations than were concentrations. While the amount of runoff was largely determined by watershed size, other physical characteristics such as topography, soil types, drainage system, and land use were also important. This argument is supported by comparing differences in water yields or runoff, i.e. annual discharge normalized by watershed area (Table 1.3). Mean annual runoff ranged from 10 to 61 centimeters within the various sub-basins. The highest value, at station 7, most likely resulted from tile drainage enhancing water transport to the stream. The lowest value, at station 9, probably reflects the limited drainage system within the basin, i.e., predominantly one small stream throughout the basin. Presumably, more water is retained on the ground or in the vadose zone during storms and then lost to the atmosphere via evaporation and transpiration over time.

In addition to variations in runoff amounts, individual sub-basins also showed a wide range of export rates (kg/ha/yr) for suspended solids, total-P, soluble-P, and nitrate (Table 1.3). Differences in these values again suggested a strong relationship to the physical characteristics of the watershed. The most dramatic differences among basins occurred in the amounts of suspended solids exported. Basins 3 and 4 had export rates over six times greater than for basins 6 and 7. Basins 3 and 4 have steeper slopes, finer soils, and less tile drainage, all of which promote increased surface erosion. Station 5 showed the cumulative effect of high erosion rates over this whole northwest portion of the watershed and exhibited rates over 2.5 times the project watershed average. A much different pattern was seen for nitrate. Basin 7 exhibited nitrate losses from two to seven times greater than all other basins. Since nitrate moves easily through soils it can be efficiently transported to the stream by tile drainage (Kladiviko et al., 1991). Export rates for phosphorus were more consistent between basins, but station 8 did show rates about two times higher than the average even after correcting for known point source contributions. This result may have been caused from additional but unquantified point sources or simply from errors caused by trying to extrapolate over so large an area. Basin 9 showed the lowest phosphorus export rates despite typically having the highest concentrations among the basins. Basin 9 also had the lowest runoff rates which suggests that high concentrations may have resulted from a lack of dilution versus greater sources of phosphorus on the land.

Rast and Lee (1983) reported that the national average total-P export rate is 0.5 kg/ha/yr. This value is exactly what was found for the project watershed, i.e. at station 8, after correcting for known point source contributions. Other basins had export rates ranging from 30 to 80 percent of this average.

Seasonal Loadings

Plots of seasonal phosphorus loads for all stations (Figs. 1.28-1.34) reveals that the timing and magnitude of high loading periods varies between both seasons and years (Fig. 1.16). Seasonal patterns for total and soluble phosphorus were nearly identical. Typically, one major peak occurred per year, with loads often four times greater than for the remaining seasons. Although the timing of this peak varied, increased loads at most stations occurred predominantly in either late winter or early spring. Occasionally high loading rates lasted throughout both seasons. Specific loading patterns were of course determined by the timing of the spring thaw and the timing and intensity of storms. In 1983 a single loading peak occurred in the spring. In 1984 through 1988 the loading peak began in late winter and extended into the spring as well. In 1988 there was a second major loading peak in the fall due to several large storms which caused flood conditions for nearly a week. In 1989, winter and spring loading was minimal compared to that which occurred in the summer and fall, again due to extreme storms events. These results again point to how difficult it is to characterize nonpoint pollution and how variable watershed responses can be between years. It should be noted that the amount of variability expressed here may even be greatly underestimated because the sampling scheme did not measure the full effect of individual storms and runoff events.

Temporal Variability in Pollutant Loading

One of the definitive characteristics of nonpoint source pollution is that a significant portion of the annual load can be produced in relatively few days. This relationship was examined graphically by plotting cumulative loading distributions for suspended solids, phosphorus and nitrogen at station 9 (Figs. 1.35 - 1.41). These plots show the percentage of load which occurred between each consecutive sampling date throughout the year. If loading rates were uniform throughout the year, distributions would plot as straight lines. Interesting differences were found among the behavior of the various parameters as well as differences among years.

Suspended solids consistently showed the largest effect of increased loading rates during snowmelt or rainfall events. This effect was characterized in the distribution plots by an extreme jump in loading percentage between two sampling interval. The loading pattern for suspended solids reflects the fact that the highest rates of erosion and overland transport occur during large storm events. Annual phosphorus loads were also dominated by only a few sampling intervals within the year, but not to the same extent as was seen for suspended solids. Total and soluble phosphorus showed similar loading patterns despite their functional differences as particulate versus dissolved forms. The similarity in patterns occurred partly because soluble-P comprised around one-third of total-P at this site. Apparently most phosphorus was transported to the stream via overland flow, but sub-surface contributions and perhaps internal recycling tended to even out the loading distribution throughout the year. Nitrate exemplified this effect well. Loading rates for nitrate were much more consistent throughout the year than for other parameters, as evidenced by greater linearity in its cumulative distribution plots. Nitrate input to the stream appeared to be dominated by sub-surface flow. Peak concentrations typically occurred one to two days after peak discharge, when overland flow had ceased. The fact that peak discharge and peak concentrations did not coincide resulted in less severe loading peaks. Of course, significant increases in loading rate still occurred after storms due to the large increases in discharge volume.

Plots of cumulative loading distributions also revealed interesting differences in the timing and magnitude of loading spikes among years. Despite these variations, for each year, much of the annual load occurred in relatively few days. In 1983 peak loading intervals occurred in April during spring rains. For the years 1984 through 1986 the first loading spike was observed in February as a result of early thaws and rain while the ground was still partially frozen. Frozen ground promotes heavy amounts of runoff due to the lack of infiltration. During these years there were also significant loading spikes during spring-time rains. In 1987 a significant portion of the annual load occurred during the first two sampling intervals in December. These two intervals alone produced around 55 and 45 percent of the suspended solids and phosphorus loads respectively. In 1988, loading was nearly non-existent throughout the spring and summer, and then dominated by the last two sampling intervals in the fall when flooding occurred. Around 70, 50, and 40 percent of the annual suspended solids, total-P, and nitrate loads respectively occurred during this flood.

The opportunity for large losses of soil and particulate phosphorus in the fall is enhanced by the practice of fall plowing. If conventional farming techniques are used the top soil is left completely exposed with no plant material to help hold it in place. In 1989 the largest phosphorus loading interval also occurred during heavy fall rains while the remainder of the year was quite linear.

To quantify the extent to which annual loads were dominated by storm events the percent of loading which occurred during the three highest recorded discharges was calculated (Table 1.4). Annual averages for years 1983 through 1989 revealed that 76, 56, and 51 percent of the loads for suspended solids, total-P, and soluble-P respectively occurred in only eight percent of the time (28 days). It should be noted that these calculations represent a minimal estimate of this "storm effect" since the weekly sampling schedule was not designed to sample storm events specifically.

Seasonality

Plots of monthly mean discharge, concentration, and load, pooled for all years, were constructed to examine seasonal patterns on a finer scale (Figs. 1.42 - 1.49). Comparing monthly patterns of concentration and loading helped distinguish when loads may have been elevated because of farming activity versus being artifacts of periods of high discharge.

The monthly mean plot for discharge (Fig. 1.42) suggests two rather distinct flow regimes exist throughout the year at station 9. A period of high discharge occurred from November through April, and a period of low discharge occurred from May through October. As noted earlier, concentration and loading patterns would be expected to follow these discharge patterns; however, some interesting deviations were found.

All parameters tended to have higher mean loadings in the winter and early spring when discharge was high (Figs. 1.43 - 1.49). Suspended solids, phosphorus, and ammonia also exhibited pronounced concentration and loading spikes during the summer when discharge was low. Elevated concentrations in July were great enough to cause a corresponding spike in loading patterns even though discharge was quite low. The reason for these increased mean concentrations in July is puzzling. Phosphorus and nitrogen are biologically utilized parameters and would not be expected to increase during the growing season. In addition, during low periods of discharge more of the streamflow should have been supplied by subsurface inputs which are typically low in phosphorus and ammonia.

Loading patterns for nitrate, silica, and chloride (Figs. 1.47 - 1.49) did closely follow the discharge pattern. Nitrate loading peaks were greatly enhanced when concentration values were also high, as seen for December and November. Concentration patterns for silica and nitrate appeared to be influenced by both variations in runoff amount and by seasonal variations indicative of biological utilization within the stream. Silica decreased rapidly from its winter maximum to a springtime low in April and then returned to higher levels throughout the summer and fall. As indicated by phytoplankton collections on artificial substrates (unpublished data), this timing may have been caused by the proliferation of a springtime diatom bloom. Nitrate also exhibited a rapid decrease in the spring but then continued to decrease throughout the summer until replenished by high fall discharge.

Similarities in the loading patterns of nitrate, silica, and chloride suggest that their biogeochemical cycles are similar and much different from those for phosphorus and ammonia. Nitrate, silica, and chloride concentrations consistently peaked well after the peak recorded discharge. This timing appears to result from a combination of initial dilution by high volumes of surface runoff followed by increased leaching from the soil as water infiltrates and sub-surface contributions become dominant.

Groundwater Contamination Monitoring

Occasionally a conflict may arise between using BMPs which improve surface-water quality but are detrimental to groundwater quality. For example, terracing and animal waste BMPs applied in the Pennsylvania, RCWP led to significantly higher nitrate concentrations in groundwater (NWQEP, 1989). To avoid such conflict requires a thorough understanding of both surface and groundwater hydrology in the area where BMPs are applied.

Project personnel were concerned that building in-ground storage structures, as part of an animal waste management BMP, could result in groundwater contamination. Monitoring wells were established on three farms, each of which used a differently designed storage structure to test for possible contamination. Wells were established both upflow and downflow of the storage structures, plus an existing deep-water well was also monitored.

The first site, at Brueninger Farm, contained an in-ground concrete tank with a slotted top. The downflow well was located at the edge of a 25 foot grassed filter strip surrounding the structure. Monitoring results revealed elevated levels of both phosphorus and nitrate in the downflow well (Figs. 1.50 & 1.51). Concentrations were somewhat erratic but nitrate levels often exceeded the federal drinking water standard of 10 mg N/l. Ammonia levels were also elevated at the downflow well during the first three years of monitoring but then dropped to the lower values typical of the upflow well for the remaining four years. This result is puzzling since the primary nitrogen species in animal waste is ammonia, not nitrate. One possible explanation is that nitrifying bacteria oxidized the ammonia to nitrate. No measurements were made to evaluate whether this process was occuring and whether it could be sufficient enough to prevent any ammonia build-up.

A second well-monitoring site was established at the Huehl Acres farm where an earthen, clay-lined pit was used to store animal waste. Extremely high levels of ammonia were detected at the downflow well and it was suspected that cracks developed in the clay lining allowing substantial leakage (Fig. 1.52). The leakage problem appeared to became much more severe after 1986 and ammonia concentrations went from around 3 mg/l to 50 mg/l. Phosphorus levels did not follow a similar pattern. It appears that any phosphorus that leaked from the pit was bound up in the soils before reaching the downflow well. Presumably at some point the soil will lose its binding capacity as exchange sites become saturated and then phosphorus will also move through the soil matrix. Beginning in 1986 nitrate levels also became elevated at the downflow well (Fig. 1.53) but results were much more erratic. Some of this variation may reflect rates of leakage and differences in the amounts of ammonia which may have been oxidized to nitrate. Leakage from the pit also appeared to be high enough to dilute chloride concentrations in the downflow well (Fig. 1.53).

A third well-monitoring site was established at Rogers' Farm, where a two earthen pit (solid and liquid separated) structure was used to store waste. Water quality results at this site were highly variable. Soluble phosphorus showed elevated concentrations in the downflow well beginning in 1984 (Fig 1.54). Conversely, nitrate levels were elevated in the downflow well only from 1983 to 1986 (Fig. 1.55). Ammonia and chloride values were both highly variable throughout all years, but were always higher in the upflow well. All parameters tended to decrease after 1987 suggesting the rate of leakage from the pit was subsiding. It is difficult, however, to interpret whether contamination from the animal waste system was responsible for any of the observed patterns without a clearer understanding of hydrologic flow patterns.

In summary, animal waste storage structures appeared to leak and elevate nutrient concentrations in groundwater on the downflow side. Clay-lined pits appear to be the least advisable design because large cracks may develop and leakage can be quite severe. Nutrient species behaved differently both among sites and among years at a given site. A more detailed understanding of flow rates and pathways would be required to quantify the effects of this groundwater contamination. Results suggest that a potential conflict exists between reducing rates of nutrient input to surface water and elevating inputs to groundwater quality by adopting these animal waste BMPs.

Point Source Load Reductions

Beginning in the middle of 1986 a dramatic decrease in total and soluble phosphorus loads was observed at station 8 (Fig. 1.24, Table 1.5). Since similar decreases where not seen at station 5 (Table 1.5) or at any of the other sub-basins it was suspected that reductions were due to point source controls taking place in the towns of Saline and Milan. Information obtained from the managers of the Saline and Milan wastewater treatment plants verified this presumption. Both treatment plants underwent upgrades in capacity and treatment capabilities early in 1986. In addition, the Saline treatment plant required that a phosphorus-laden industrial waste be removed from its inflow because it was not amenable to treatment and was causing the plant to violate its discharge permits. These factors combined to produce a 71 and 56 percent reduction in the post-1986 mean annual soluble and total phosphorus load respectively (Table 1.5). Data on daily total phosphorus discharge from the treatment plants was used to segregate observed loads at station 8 into point and nonpoint source contributions (Fig. 1.56). Nonpoint source contributions were discharge-normalized to allow for comparison between years. Prior to 1986, point and nonpoint sources contributed 12.6 and 11.4 mtons of phosphorus respectively for a total of 24 mtons (Table 1.6). The post 1986 average was 2.7 and 8.7 mtons respectively for a total of 11.3 mtons.

The significance of these point source contributions within the watershed may not have been adequately evaluated at the onset of the project. With the exception of sub-basin 9, phosphorus levels within the study area were not exceptionally high for small rural streams. It is obviously more difficult to document significant reductions in phosphorus losses as a result of land treatment applications when levels are near normal background levels. The RCWP projects which were most successful in demonstrating water quality improvements from BMP implementation had much more severe nonpoint source impacts (NWQEP, 1989).

CHAPTER II

LAND TREATMENT IMPLEMENTATION

Introduction

One of the key challenges facing all RCWP monitoring projects was to develop methods to evaluate and quantify the types and amounts of BMPs implemented within their watersheds. This process was critical for tracking progress in soliciting participation as well as for interpreting water quality monitoring data (Maas et al., 1988). In this study, BMPs were quantified according to the total acreage over which they were applied, or in the case of structural practices over the acreage which the BMP influenced. Acreages for all practices were then summed on a yearly basis to provide totals of the percent of each sub-basin under BMP implementation for any given year (Tables 2.1 - 2.6). Acreage contracted under crop rotation plans were not included in annual totals because their large acreage outweighed all other BMPs, their effects could not be delineated into specific years, and they were fairly standard practices for all producers. Estimates of tons of soil saved, calculated by the universal soil loss equation (Wischmeier and Smith, 1978), indicated that permanent vegetative cover to critical areas, pasture plantings, and conservation tillage had similar areal effects. Although it is difficult to assess whether other BMPs have equivalent "effects" on an areal basis, the areal estimates did provide a way to make qualitative comparisons both over time and between sub-basins.

Ideally future nonpoint source control projects could quantify land treatment directly in terms of the specific pollutants of interest. This would typically involve developing mass balance calculations for quantities of animal waste, commercial fertilizers, pesticides, crops, and livestock which are processed on the farm. This approach would provide much more insight into the source amounts of pollutants which are being lost from the watershed and the potential for BMPs to reduce these losses.

Results and Discussion

A summary of the annual percent of each sub-basin under BMP implementation reveals that overall participation in the project was very low (Fig. 2.1). Annual percentages of BMPs changed over the years as both new contracts were added and previous ones expired. Within the Saline Valley project area only sub-basin 9 accumulated even thirty percent of its area under implementation. The maximum level of implementation reached 56 percent in 1985, but then dropped off sharply from that point on as contracts expired. By the end of the project only 15 percent of this sub-basin was under best management practices.

Since no follow up with the participants was performed after their contracts expired it had to be assumed that the BMPs were not being continued. It was unfortunate RCWP did not mandate that the BMP contracted for had to be implemented for the entire duration of the study. This procedure would have allowed projects to obtain a maximium amount of their watersheds under implementation as new particdipants were brought into the program.

The combination of low participation and small changes in the amount of land treatment between years minimized the chance for detecting any effects on water quality. A further problem was that specific areas could not be targeted for treatment because of the voluntary approach used. Many of the BMPs applied were quite far removed from the drainage channel and probably had minimal effect on water quality (Fig. 2.2). The combination of these factors dictates that unknown changes within nonparticipating areas could easily mask the effects of any land treatment.

The levels of participation within the Saline Valley project contrasted sharply with that for other RCWP projects. According to a summary by Maas et al. (1988), all other projects achieved or exceeded their implementation goals and obtained between 60 to 100 percent of their study area under contract. The low response by area farmers could have resulted from difficult economic conditions in this area, negative attitudes towards the practices involved, or inadequate contact and education by personnel charged with obtaining contracts.

CHAPTER III

RELATING WATER QUALITY CHANGES TO LAND TREATMENT IMPLEMENTATION

Introduction

One of the stated goals of the Saline Valley monitoring program was to quantify the relationship between the types and amounts of BMPs applied and resulting changes in water quality. This process was complicated by the fact that there is no quantifiable unit of measure which is equivalent for all the different types of BMPs applied. It was therefore difficult to quantitatively define the amount, or the effect, of land treatment applied within the watershed for any given time period. Unfortunately the project did not have the means to collect sufficient data which would enable it to quantify BMPs within the framework of the amounts of animal waste, fertilizers, or soil erosion that was being managed or reduced. This later approach may be the key to establishing a meaningful and predictable relationship between BMP's and pollutant loadings.

The process of quantifying the water quality/land treatment relationship was also made difficult by the extreme variability in pollutant concentrations and loadings on both temporal and spatial scales. Ideally the problem of variability can be handled by an appropriately designed monitoring program and use of statistical tests. Limitations in resources and the constraints of RCWP's voluntary approach, however, precluded much of this opportunity.

The final approach arrived at by the monitoring project was to use annual summations of the percent acreage within individual sub-basins under some form of BMP. These percentages were then correlated to annual trends in sediment and nutrient concentrations as described by, (1) concentration/dicharge empirical models, and (2) discharge normalized annual means. Although various BMPs clearly have different effects in terms of pollutants controlled and in the magnitude of their effectiveness, this approach allowed for a qualitative assessment of whether BMPs can reduce pollutant losses from the watershed.

Results and Discussion

Creams Model Predictions

Prior to the development of an adequate data base from the monitoring program the Soil Conservation Service applied the CREAMS model to both estimate phosphorus loads from the watershed and predict reductions in these loads once certain BMPs were applied. Comparison of the model's predicted loads to the nonpoint contribution observed at station 8, the watersheds downstream terminus, revealed serious descrepencies. The model predicted that losses from animal waste, commercial fertilizers, and soil erosion would produce phosphorus loads of 39.6 tons. Conversely the phosphorus loads observed by the monitoring program averaged around 10.4 tons. As stated earlier, however, there are large uncertainties in observed loading rates as well, and without specifically sampling storm events the observed loads were very probably under-estimated.

The more critical problem was in the way the CREAMS model was used to predict expected loading reductions after BMP implementation. The model's predictions for load reductions of greater than 90 percent (Anon, 1985) are clearly untenable and were unsubstantiated. These results point to danger of relying on uncalibrated model predictions to provide information on which future managerial decisions are based.

Concentration/Discharge Empirical Models

In assessing the effects of land treatment on water quality between years it was essential to account for differences in yearly precipitation and runoff. One approach was to use regression models of concentration versus discharge for each year (Spooner et al., 1985). This approach has been useful for describing patterns of nutrient losses from watersheds (eg. Johnson et al., 1976; Lathrop, 1986) and eliminates much of the variation resulting from meteorological conditions. Most BMPs are designed to reduce the amount of of pollutants available for transport in runoff. Assuming stream discharge remains similar after BMP implementation, the effects of BMPs can be examined on the basis of changes in empirical regressions of concentration versus discharge for each year. Empirical relationships were derived from regressions of concentration versus the square root of discharge and stratified for sampling periods with above median discharge (Fig 3.1). Stratification improved the strength of the correlations by removing the high variability in concentrations observed under low-flow conditions. Regressions of log concentration versus discharge and log concentration versus log discharge were also evaluated but found to produce less significant relationships. Correlation coefficients for regressions of conentration versus the square root of dishcarge varied greatly between parameters and years (Table 3.1). As expected, suspended solids and total-P had a stronger correlation with discharge than soluble-P. A similar observation was made by Meyers and Likens (1979) in their study of phosphorus losses from the Hubbard Brook experimental watershed. Nitrate was also not well correlated to discharge, and as discussed earlier concentrations tended to peak well after peak discharge. Concentration-discharge regressions would have been improved had sampling occurred at more frequent intervals and particularly during runoff events. The characteristics of such pollutant transport are such that even during a single runoff event differences in the regression can be seen between the rising versus falling limbs of the hydrograph (eg. McDiffett, 1989). This pattern suggests that storms should be sampled on a time-scale of hours, and that discharge records should be stratified into rising and falling stages to predict accurately concentrations from the relationship.

Predicted suspended solids and total-P concentrations at discharge levels of 0.1 and 0.5 m^3 sec^{-1} were taken from the regressions and correlated against the amounts of BMPs applied within sub-basin 9 for each year (Fig 3.2). There was no significant relationship for either parameter, at either discharge level, which would suggest that BMP implementation decreased suspended solids or phosphorus levels. There are several reasons why this type of analysis may not have demonstrated a significant relationship. First, participation was low and there was no way of assessing the effects of non-participating farms. Additionally, areal estimates of BMPs may not accurately measure their potential to reduce pollutant delivery. The most significant problem, however, is that variations in the timing and intensity of storms for different years are likely to change the concentration/discharge regression regardless of any land treatment applications.

Annual Means

A second approach used for analyzing the water quality land treatment relationship was to look at variations in annual means. The advantage of this approach was both its simplicity and the fact that BMPs are designed to reduce high concentrations which should be reflected in the annual means. When annual mean solids and total-P concentrations were plotted against areal BMP implementation at station 9 (Fig. 3.3) no significant correlation was observed. However, comparing annual means does not account for the effect of differences in precipitation and runoff between years. As seen in figure 3.4, annual mean concentration was positively correlated to annual mean discharge. One way to correct for these differences is to normalize each year to an average discharge. When data were replotted using normalized annual means there was again not a significant correlation, but for the exception of one point it appeared as if a relationship might exist with BMP implementation (Fig. 3.5). Indeed prior to the inclusion of the 1989 data point (the outlier) there was a highly significant relationship (alpha $< .001$, $r^2 = 0.85$) between annual mean total-P and BMPs. The regression predicted a 25 percent reduction in total P when 75 percent of the watershed was under implementation. The data point for 1989 does not fit with other years and completely destroys the regression's statistical significance. Since there was no reason to believe that the 1989 mean value was any less reliable, the relationship can not assume to hold true. Other figures have already shown that the loading pattern for this year was atypical. One possible explanation is that the dominance of a Fall storm events produced concentrations lower than expected in relationship to the amount of discharge. Without further conjecture, it is important to realize that such inconsistency is likely to be an inherent characteristic of nonpoint source control programs and that there may be very little of the predictability or reliability observed in the counterpart point source control efforts. This fact may be of critical importance as limitations in available dollars dictate that the most cost-effective and reliable programs be utilized.

CONCLUSIONS

Saline Valley RCWP was not successful in meeting its participation or phosphorus reduction goals. Consequently the monitoring program was not able to establish relationships between the land treatment applied and resulting water quality changes. There were many factors which contributed to the projects' shortcomings. Two of the most significant were that; (1) no specific water quality impairment existed within the project area, and (2) nonpoint loadings were not that severe when compared to the national average for agricultural lands. These factors both served to reduce landowner participation and limit the chance for detecting water quality changes above the level of natural variability.

The water quality monitoring program was also severely limited by lack of money, manpower, and equipment. Additionally, protocols used by RCWP limited the monitoring design because land treatment was applied on a gradual basis throughout the entire area and contracts were not required to serve the duration of the study. Despite these shortfalls, or perhaps because of them, many lessons were learned pertaining to strategies and policies for managing nonpoint pollution in the agricultural sector.

Obtaining high levels of participation is the key to success when working at the watershed level. Although cost-share and technical assistance helped increase the levels of participation, they did not guarantee it. Farmers had to be convinced of the fact that BMPs could both improve water quality and sustain on-farm profitability. Additionally, there needs to be a locally recognized water quality impairment or water use restriction to induce public concern and facilitate participation. Before the project begins there needs to be an accurate accounting of all contributing sources and quantities of pollution in order to assess the projects' potential for improvement. Then the specific activities and areas that are most critical should be targeted for BMP implementation. Because targeting is difficult with a voluntary approach, regulatory authority may be required in certain circumstances.

The ability to document direct water quality improvements as a result of BMP implementation can be enhanced by monitoring streams within well defined sub-basins with sizes of 10,000 acres or less. When participation is not high enough within the watershed, pollutant loadings from non-participating areas may simply mask any effect. Therefore the closer you can monitor to the actual land treatment the beter your chances for detecing a treatment effect. This approach would also help to minimize the effects of changes in water quality which occur internally within the stream as water is being transported through a drainage system.

Given the extreme variability associated with nonpoint pollution, it is mandatory that continuous discharge records be collected to ensure accurate loading estimates. Monitoring programs should be designed to sample all possible storm events in addition to scheduled baseline monitoring. The most important question may be whether these BMPs are effective during such storm events when levels of runoff are greatly increased.

If the possibility exists monitoring programs should incorporate a paired watershed approach or a before and after approach to maximize the statistical sensitivity of documenting water quality changes (see Spooner et al., 1987). As stated before, howwever, these designs are difficult to establish in voluntary programs.

From a nationwide water quality management perspective the key approach may be to focus on a limited number of important water bodies that have significant water quality impairments due to nonpoint sources and that have a good potential for improvement. This approach would conserve limited resources and increase chances for realizing significant water quality improvements. Early successes would help promote further participation in future of nonpoint source control projects .

Although the twenty national RCWP projects have not met with complete success, they have increased public awareness of the problem and begun to develop a working model for successfully managing agricultural nonpoint pollution. Lessons learned from RCWP will be invaluable to the States as they develop section 319 plans detailing management strategies for nonpoint source pollution control.

REFERENCES

Anonymous, 1985. Saline Valley Rural Clean Water Project, Annual Progress Report.

Beeton, A.M., 1982. Report to SEMCOG on the Saline Valley Rural Clean Water Trend Monitoring Project, July29, 9181 - September 30, 1982.

Beeton, A.M., R.E. Holland, & J.H. Judd, 1984, Saline Valley Interim Report on Monitoring During 1983.

Davis, C.O. and M.S. Simmons, 1979. Water chemistry and phytoplankton field and laboratory procedures. Great Lakes Research Division, Special Report No. 70.

Hill, A.R., 1986. Stream-nitrate loads in relation to variations in annual and seasonal runoff regimes.

Holland, R.E., A.M. Beeton, and D. Conley, 1985. Saline Valley Rural Clean Water Project Interim Report on Monitoring During 1984.

Holland, R.E., A.M. Beeton, and D. Conley, 1986. Saline Valley Rural Clean Water Project Interim Report on Monitoring During 1985.

Holland, R.E., A.M. Beeton, and T. Johengen, 1987. Saline Valley Rural Clean Water Project Interim Report on Monitoring During 1986.

Holland, R.E., A.M. Beeton, and T. Johengen, 1988. Saline Valley Rural Clean Water Project Interim Report on Monitoring During 1987.

Holland, R.E., A.M. Beeton, and T. Johengen, 1989. Saline Valley Rural Clean Water Project Interim Report on Monitoring During 1988.

Johengen, T.H., A.M. Beeton, and R.E. Holland, 1990. Saline Valley Rural Clean Water Project Interim Report on Monitoring During 1989.

Johnson, A.H., D.R. Bouldin, E.A. Goyette, and A.M. Hedge, 1976. Phosphorus loss by stream transport from a rural watershed; Quantities, Processes, and sources. Journal of Environmental Quality, vol. 5:148-157.

Kladivko, E.J., G.E. Van Scoyoc, E.J. Monke, K.M. Oates, and W.Pask, 1991. Pesticide and nutrient movement into subsurface tile drains on a silt loam soil in Indiana. Journal of Environmental Quality, vol. 20: 264-270.

Lathrop, R.C., 1986. A simplified method for obtaining monitored phosphorus loadings. Journal of Lake and Reservoir Management, vol. 2: 20-26.

Maas, R.P., et al., 1988. Agricultural nonpoint source control: Experiences from the Rural Clean Water Program. Journal of Lake and Reservoir Management, vol 4:51-61.

McDiffett, W.F., A.W. Beidler, T.F. Dominick, and D. McCrea, 1989. Nutrient concentration-stream discharge relationships during storm events in a first-order stream. Hydrobiologia, vol. 179: 97-102.

Meyer, J.L. and G.E. Likens, 1979. Transport and transformation of phosphorus in a forest stream ecosystem. Ecology, vol. 60: 1255-1269.

National Water Quality Evaluation Project, 1989. *1988 Annual Report: Status of agricultural nonpoint source projects.* EPA 506/9-89/002.

Rast, W. and G.F. Lee, 1983. Nutrient loading estimates for lakes. Journal of Environmental Engineering, vol. 109: 502-517.

Sharpley, A.N., J.K. Syers, and P.W. O'Conner, 1976. Phosphorus inputs into a stream draining an agricultural watershed. I: *Sampling.* Water, Air, and Soil Pollution, vol. 6: 39-52.

Sharpley, A.N., S.J. Smith, and J.R. Williams, 1988. Nonpoint source pollution impacts of agricultural land use. Journal of Lake and Reservoir Management, vol. 4: 41-49.

Spooner, J., et al., 1985. Appropriate designs for documenting water quality improvements from agricultural nonpoint source control programs. In, *Perspectives on Nonpoint Source Pollution.* U.S. Environmental Protection Agency, Washington, D.C. EPA 440/5-85-001.

Spooner, J., et al., 1987. Determining statistically significant changes in water pollutant concentrations. Journal of Lake and Reservoir Management, vol 3:195-200.

Stevens, R.J. and R.V. Smith, 1978. A comparison of discrete and intensive sampling for measuring the loads of nitrogen and phosphorus in the River Main, County Antrim. Water Research, vol. 12: 823-830.

U.S. Environmental Protection Agency, 1984. *Report to Congress: Nonpoint source pollution in the U.S.* U.S. Government Printing Office, Washington, DC.

Wischmeier, E.H. and D.D. Smith, 1978. *Predicting rainfall erosion losses.* Agricultural Handbook No. 537, U.S. Department of Agriculture, Washington, D.C.