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STREAM QUALITY IMPACTS OF BEST MANAGEMENT PRACTICES IN A NORTHWESTERN ARKANSAS BASIN[1]

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ABSTRACT: A variety of management options are used to minimize losses of nitrogen (N), phosphorus (P), and other potential pollutants from agricultural source areas. There is little information available, however, to indicate the effectiveness of these options (sometimes referred to as Best Management Practices, or BMPs) on basin scales. The objective of this study was to assess the water quality effectiveness of BMPs implemented in the 3240 ha Lincoln Lake basin in Northwest Arkansas. Land use in the basin was primarily forest (34 percent) and pasture (56 percent), with much of the pasture being regularly treated with animal manures. The BMPs were oriented toward minimizing the impact of confined animal operations in the basin and included nutrient management, dead bird composter construction, and other practices. Stream flow samples (representing primarily base flow conditions) were collected bi-weekly from five sites within the basin from September 1991 through April 1994 and analyzed for nitrate N (NO_3-N), ammonia N (NH_3-N), total Kjeldahl N (TKN), ortho-P (PO_4-P), total P (TP), chemical oxygen demand (COD), and total suspended solids (TSS). Mean concentrations of PO_4-P, TP, and TSS were highest for sub-basins with the highest proportions of pasture land use. Concentrations of NH_3-N, TKN, and COD decreased significantly with time (35-75 percent/year) for all sub-basins, while concentrations of other parameters were generally stable. The declines in analysis parameter concentrations are attributed to the implementation of BMPs in the basin since (a) the results are consistent with what would be expected for the particular BMPs implemented and (b) no other known activities in the basin would have caused the declines in analysis parameter concentrations.
(KEY TERMS: water quality; Best Management Practices; agriculture.)

INTRODUCTION

Water quality impacts of agricultural production practices have been a matter of public concern in the United States for decades. There is ample evidence to indicate that in general terms, practices such as row crop production (e.g., Baker and Laflen, 1982) and animal manure application (e.g., McLeod and Hegg, 1984; Pote *et al.*, 1994) can lead to increased concentrations of nitrogen (N), phosphorus (P), solids, microorganisms, and other substances in surface waters that receive runoff from agricultural source areas. Both ground and surface water quality are vulnerable to agricultural production practices through leaching of pollutants such as nitrate N (NO_3-N) and pesticides (e.g., Adams *et al.*, 1994). The potential impacts of excessive concentrations of pollutants such as those just mentioned are well known and include accelerated eutrophication (see, for example, Sharpley *et al.*, 1994) and, in extreme cases, health hazards to humans and/or animals.

There is general agreement that pollutant losses should be minimized consistent with practical and economic constraints. To this end, many scientists have developed and tested management options such as no-till (Mueller *et al.*, 1984) and grassed buffer zones (e.g., Dillaha *et al.*, 1989) that minimize transport of pollutants off agricultural source areas. Management options that are proven effective and meet certain other criteria may be labeled "Best Management Practices" (BMPs). A BMP is specifically defined as "a practice or combination of practices that is determined by a state (or designated area-wide planning agency), after problem assessment, examination of alternative practices, and appropriate public participation, to be the most effective practicable (including technological, economic, and institutional considerations) means of preventing or reducing the amount of

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pollution generated by nonpoint sources to a level compatible with water quality goals" (Bailey and Waddell, 1979). Cost sharing is sometimes available from government agencies to agricultural producers who voluntarily implement BMPs.

By their definition, BMPs have the potential for reducing water quality impacts of agricultural production systems. It can be quite difficult, however, to estimate *a priori* the effectiveness of a particular BMP when that BMP is applied under different conditions (e.g., different soil, cover, weather, etc.) than those under which it was developed and tested. It is more challenging still to estimate the integrated water quality impact of implementing possibly dozens of BMPs at various locations within a basin of thousands of hectares. Notwithstanding considerable efforts in data collection and mathematical simulation modeling, there is a great deal of uncertainty regarding the water quality impact of implementing BMPs under untested conditions and particularly on larger (basin) scales, as noted by Park *et al.* (1994) and Walker (1994).

Information regarding water quality impacts of basin-scale BMP implementation is imperative for a number of reasons. From a pragmatic point of view, water quality impact data are necessary to determine whether public resources used to cost-share BMP implementation are providing a measurable benefit. Along the same line, such information could (and ultimately should) be used to assess whether the water quality benefits of BMP implementation justify the costs (whether directly to private citizens or to the public). Data on water quality response to BMP implementation are also needed to improve mathematical simulation models so that they are more reliable tools for accurately assessing the water quality impacts of BMP implementation. Accurate simulation model data could then be used as a surrogate for relatively expensive observed data in various economic and other analyses.

A limited number of basin-scale studies on BMP effectiveness have been reported. Park *et al.* (1994) monitored a 1464 ha basin in eastern Virginia in which the primary agricultural activities were row crop production. Water quality monitoring was conducted before and after implementing no-till, critical area treatment and some structural practices to assess the impacts of these BMPs. The authors concluded that the BMPs reduced N and P concentrations in stream samples by 20 and 40 percent, respectively. Walker and Graczyk (1993) monitored two basins draining 14.0 and 27.2 km^2 in southern Wisconsin. The BMPs that were implemented in the basins (conservation reserve, contour stripcropping, minimum tillage, changing crop rotation, and barnyard treatment) were oriented toward reducing pollution from cropland and livestock barnyards. The authors reported that the BMPs led to reduced mass transport of NH_3-N and suspended sediment for one of the basins, and that significant reductions on the other basin might not have been detected due to an insufficient data set.

The objective of this study was to assess the impact of BMP implementation in a Northwest Arkansas basin on stream base flow quality. This paper contributes to a much needed but quite limited body of information on large-scale BMP implementation impacts on water quality. This study differs from recent similar work by examining different land uses and BMPs than have been reported. The major land use in this study (aside from forest, on which no BMPs were applied) was pasture, and the key BMP was nutrient management (defined in more detail later). The difference in land uses is significant in view of differences in the hydrology and water quality dynamics of the two types of agricultural production systems (e.g., Edwards *et al.*, 1995).

METHODS AND MATERIALS

Study Area

The study area was the Lincoln Lake basin, which is located in Northwest Arkansas, north of the City of Lincoln (36°00′ N, 94°25′ W). The climate is humid with a mean annual temperature of 14.1°C and annual rainfall of 1120 mm (National Climatic Data Center, 1994). The total basin consists of approximately 3240 ha. Elevations within the basin range from approximately 365 to 487 m with a mean elevation of 429 m. Steep slopes exist at the northernmost and southernmost regions of the basin as well as near streams in the northern one-third of the basin.

Fifteen soil series are represented in the basin, with the Captina, Enders, Enders-Allegheny, Hector-Mountainburg, and Linker series covering nearly 70 percent of the total area (Harper *et al.*, 1969). The Captina and Enders series are characterized as having moderately good drainage; the Allegheny and Linker series have good drainage; and the Hector-Mountainburg complex has good to somewhat excessive drainage (Harper *et al.*, 1969).

There is a diversity of land uses within the basin. The major land uses are pasture (56 percent overall) and deciduous forest (34 percent overall). The primary agricultural enterprises in the basin include beef cattle and confined animal (predominately poultry) production. Apple orchards, dairy facilities, and other agricultural operations also exist within the basin, but the quantity and areal extent of these operations are insignificant in comparison to grazing and confined animal production. The confined animal production leads to a large quantity of manure available for land application (Soil Conservation Service and University of Arkansas Cooperative Extension Service, 1990).

BMP Implementation

As reported by Soil Conservation Service and University of Arkansas Cooperative Extension Service (1990), the water quality in Lincoln Lake decreased over a period of years to the point that the lake was assessed as eutrophic with production limited by N. Profuse algal blooms were common as were complaints regarding the palatability of water after treatment for drinking purposes. These problems and a concern over the role of agricultural production practices (primarily land application of manures from confined animal facilities) within the Lake's basin prompted the Natural Resources Conservation Service (NRCS; formerly named the Soil Conservation Service, or SCS) and the University of Arkansas Cooperative Extension Service (CES) to initiate a program to help producers implement BMPs. The NRCS provided direct technical assistance to producers who wished to implement BMPs, while the CES assumed responsibility for public educational activities. The program began in 1990, but producer participation in the program was relatively limited until 1991.

The major BMPs that were implemented included nutrient management, waste utilization, pasture and hayland management, dead poultry composting, and waste storage structure construction (pond/lagoon for liquid manure or stacking shed for dry manure), with the first three BMPs nearly always being simultaneously implemented on a particular land area. The NRCS maintained detailed records on the land areas (in the case of the first three BMPs) and sites (for the last two BMPs) on which BMPs were implemented.

Nutrient management is an areal BMP defined (SCS, 1992) as "managing the amount, form, source, placement, and timing of applications of plant nutrients." The major water quality benefits that could be expected with nutrient management in the context of animal manure application include reduced concentrations of N and unoxidized organic matter. No reductions in concentrations of phosphorus (P) would be expected, because application rates for animal manure are based on meeting plant N requirements, which generally leads to over-fertilization in terms of plant P requirements. In this case, P would thus continue to accumulate in the soil, even if at a slower rate, for N-based application rates. These higher soil P concentrations would then have the potential to cause even higher P concentrations in runoff and ground water.

Waste utilization is an areal BMP that involves "using agricultural waste or other waste on land in an environmentally acceptable manner while maintaining or improving soil and plant resources" (SCS, 1987a). From the standpoint of potential water quality impacts, waste utilization is similar to nutrient management in that nutrient management principles are involved in determining waste application parameters (e.g., amount and timing).

Pasture and hayland management, another areal BMP, consists of "proper treatment and use of pastureland or hayland" (SCS, 1987b) and includes guidelines for beginning and ending grazing, harvesting the forage, and controlling weeds. The potential water quality benefits of pasture and hayland management are related to maintaining desirable soil cover and structure and may include reduced losses of nutrients, solids, and organic matter.

Dead poultry composting is "a process in which the normal daily accumulation of dead birds from a poultry facility is mixed with other organic ingredients and converted through biological activity to a stable and useful end product (compost)" (SCS, 1990). In comparison to dead poultry disposal pits (a formerly typical means of handling dead poultry), implementation of dead poultry composting would be expected to influence water quality by reducing N and organic matter loadings to subsurface water, which could be evidenced by an improvement in the quality of flow (particularly base flow) in nearby streams.

A waste storage structure is "a fabricated structure for temporary storage of animal or other agricultural waste" (SCS, 1977). A waste storage structure is closely related to, and would be expected to produce the same water quality benefits as, nutrient management, because the structure can give the manure user the flexibility to time manure applications appropriately. If the structure alleviates a prior condition in which manure was moving more or less directly into a stream, relatively dramatic improvements in water quality could result, including reductions in nutrients, organic matter, microbes, and other manure constituents.

The key practice of all those just described is nutrient management, because perhaps the best use of most agricultural by-products (whether manure or dead animals) is land application. Nutrient management principles lead to identification of the best land application parameters.

Water Quality Monitoring

Even though the BMPs were implemented in response to the quality of Lincoln Lake, BMP effectiveness was not assessed through direct lake sampling. The rationale was that nutrients and other substances stored in the lake could delay any measurable response to changing base flow inputs, even if the BMPs were effective in reducing those inputs. Lake inputs occurring during storm runoff could further delay any response. Monitoring the contributing streams, on the other hand, could enable a relatively rapid assessment of BMP effectiveness with respect to base flow pollutant load reduction. Using stream monitoring to assess BMP effectiveness would have the added benefit of allowing BMP effectiveness to be measured independently of the pre-existing status of the lake, thereby making the results of wider general use. For these reasons, the effectiveness of the BMPs with respect to base flow quality was assessed using water samples collected from the two main tributaries of the lake: Moores Creek and Beatty Branch.

Moores Creek was monitored at three sites (referred to as MA, MB, and MC), and two monitoring sites (BA and BB) were established for Beatty Branch. The total drainage areas of these two tributaries are 2120 and 1120 ha for Moores Creek and Beatty Branch, respectively. One site per tributary (MA and BA) was located as close to the lake as possible. The remaining three were located further upstream on the tributaries. The locations of the monitoring sites and their corresponding sub-basins are shown in Figure 1. The sub-basin area associated with the MA site was approximately 1800 ha, or 85 percent of the total area drained by Moores Creek. Approximately 800 ha, or 71 percent of the total BA drainage area, drained past site BA. The upstream sites MB, MC, and BB were associated with sub-basin areas of approximately 370, 90, and 150 ha, respectively. Land use was determined for the sub-basin associated with each monitoring site as given in Table 1.

Stream flow samples (1 L sample size) were collected at each monitoring site on a two-week sampling interval beginning September 1991 and continuing until April 1994. The samples generally represented base flow conditions, although the timing of sampling occasionally (< 10 percent of the time) coincided with storm runoff. Samples were transported to the Arkansas Water Resources Center Water Quality Laboratory, prepared for analysis, and analyzed for nitrate N (NO_3-N), ammonia N (NH_3-N), total Kjeldahl N (TKN), ortho-P (PO_4-P), total P (TP), chemical oxygen demand (COD), and total suspended solids (TSS). Standard methods of analysis (Greenberg *et al.*, 1992) were used in all analyses. Ion chromatography was used in analyses of NO_3-N and PO_4-P. The ammonia-selective electrode method was used to determine NH_3-N. The macro-Kjeldahl method was used in TKN analyses. Total P was determined by the ascorbic acid colorimetric method following sulfuric acid-nitric acid digestion. The closed-reflux, colorimetric method was used for COD determinations.

Two tipping bucket rain gages were installed and used to record occurrences and amounts of rainfall. One gage was located in the extreme northern portion of the Lincoln Lake basin, while the other was in the extreme southern portion.

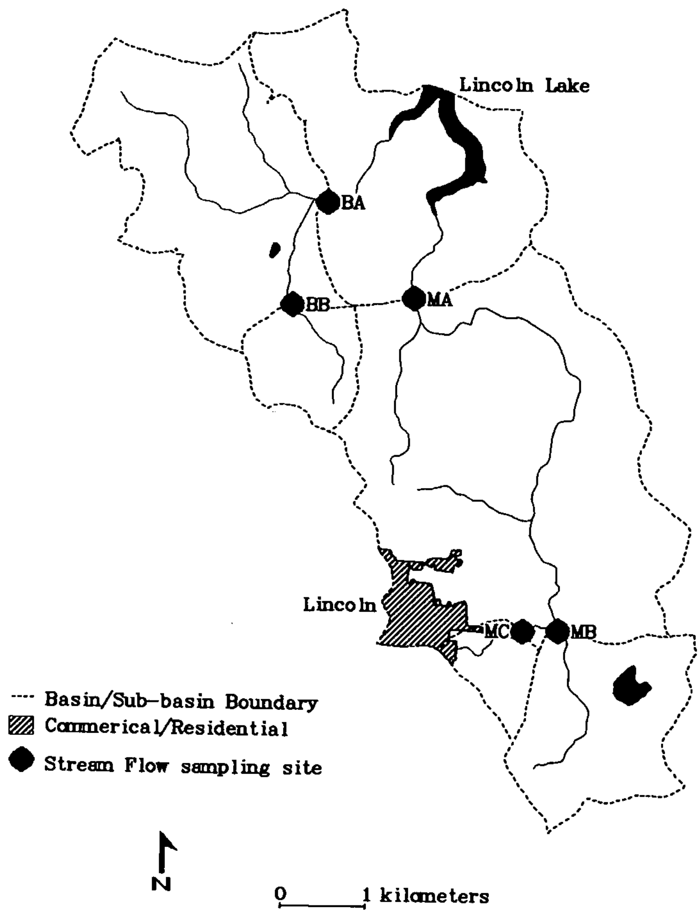

Figure 1. Stream Sampling Sites and Corresponding Sub-Basins.

Analysis of Water Quality Data

Non-parametric analysis of variance (ANOVA) (Kruskal and Wallis, 1952) was used to determine whether there was an overall (p=0.05) significant

TABLE 1. Land Uses Within Monitoring Site Basins.

Category	Monitoring Site				
	MA	MB	MC	BA	BB
	(areal coverage, percent)				
Pasture	61.8	73.2	81.0	56.5	90.2
Orchards	2.5	7.4	0.0	0.5	2.1
Poultry Houses	1.6	0.5	0.0	0.4	2.0
Other Agricultural	0.1	0.0	0.0	1.4	0.0
Residential	5.4	0.0	4.2	0.0	0.1
Commercial	1.7	0.0	0.0	0.0	0.0
Water	0.0	3.7	0.0	0.0	0.0
Forest	25.9	15.2	14.8	39.5	5.6
Other	1.0	0.0	0.0	1.7	0.0

monitoring site effect on analysis parameter concentrations. Non-parametric ANOVA was used in preference to parametric ANOVA, because initial analyses indicated that the data were not normally distributed. When the ANOVA indicated a significant site effect, median concentrations were separated using Dunn's test. This analysis was independent of the regression analysis described in the following paragraph and was performed only to assess variation among sites.

Unlike previous studies on BMP effectiveness assessment (e.g., Park *et al.*, 1994), there were no distinct pre- and post-implementation data sets on stream quality that could be directly compared. Rather, the data in this study were collected concurrently with BMP implementation. In this situation, BMP effectiveness can be assessed by analyzing for the presence of trends in the data set, provided nothing else with the potential for causing the trends occurred. The approach to assessing BMP effectiveness was thus to assess trends in the data with time, considering time as a surrogate for other measures of BMP implementation. Multiple linear regression on the natural logarithms of concentrations (censored data were taken as two-thirds of the respective detection limit) was used to test for the presence of significant trends. Multiple regression was used instead of simple regression because inspection of the data suggested seasonality in the data and that the amplitude of the seasonality function might vary with time. The data were thus fitted to the following model:

$$Y = a + b_1 T + b_2 \sin(t) + b_3 \cos(t) + b_4 t \sin(t) + b_5 t \cos(t) \tag{1}$$

where Y is the natural logarithm of concentration of a particular analysis parameter; a, $b_1, \ldots, b_5$ are regression coefficients; T is time since monitoring began (in days); and

$$t = \frac{2\pi T}{365} \tag{2}$$

The model specification procedure began with stepwise regression of Y against all independent variables shown in Equation (1). As indicated in Equation (1), the coefficient b_1 is the key coefficient in terms of testing the significance of trend. Predicted values of Y and residuals were calculated from the regression results. If the coefficient of serial correlation among the residuals was insignificant ($p \geq (0.05)$, then no further analysis was performed for that particular parameter. If the serial correlation coefficient was significant, then the values of the independent and dependent variables were corrected for serial correlation according to methods described by Ostrom (1978) through the transformations

$$X'_{i,1} = \sqrt{1-r^2}\ X_{i,1} \tag{3}$$

$$Y'_1 = \sqrt{1-r^2}\ Y_1 \tag{4}$$

$$X'_{i,j} = X_{i,j} - rX_{i,j-1}, j = 2,\ldots n \tag{5}$$

$$Y'_j = Y_j - rY_{j-1}, j = 2,\ldots n \tag{6}$$

where the Y is the dependent variable, X_i is the i^{th} independent variable, the subscript j indicates the j^{th} observation, n is the number of observations, r is the serial correlation coefficient for the residuals, and the apostrophe denotes the transformed value. The transformed logarithms of observed concentrations were then regressed against the transformed independent variables identified previously as significant. In isolated cases, independent variables identified as significant before correcting for serial correlation were insignificant after the correction. In these cases, the insignificant variable was dropped from the set of significant independent variables, and the regression was repeated.

RESULTS AND DISCUSSION

Daily rainfall recorded by the two rain gages installed at two locations within the basin was higher by an average of 21 percent than normal for Fayetteville, Arkansas (the nearest weather station with available daily rainfall data). Mean (arithmetic mean of the two rain gages within the basin) rainfall observed during monitoring is given in Figure 2.

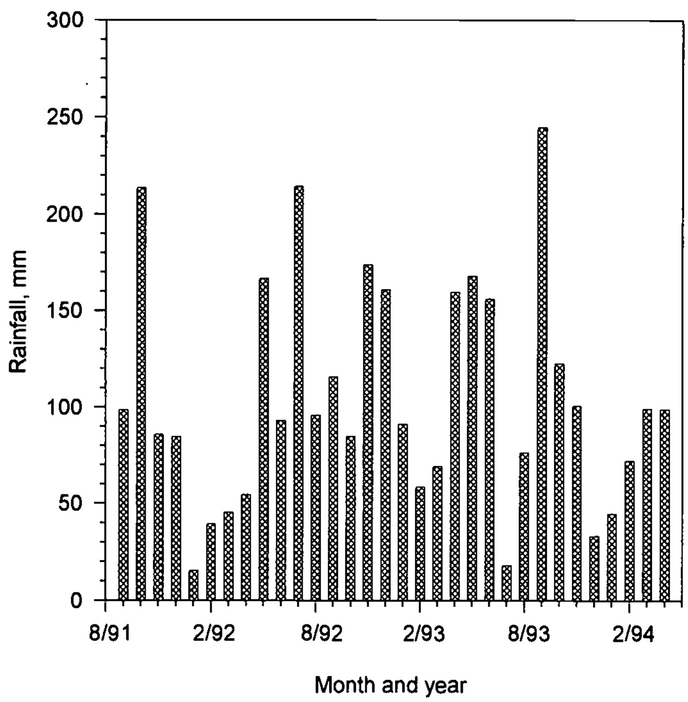

Figure 2. Mean Monthly Rainfall.

BMP Implementation Tracking

The running proportions of available land (pasture land use) having BMPs implemented are given in Figure 3. These data address only land on which the areal BMPs (nutrient management, waste utilization, and pasture and hayland management) were implemented, since the point BMPs (dead bird composting and waste storage structure) are not readily associated with a land area. The proportions of available land under BMP implementation ranged from 33 percent (site MB) to 94 percent (site BB) at the end of the monitoring period (Figure 3).

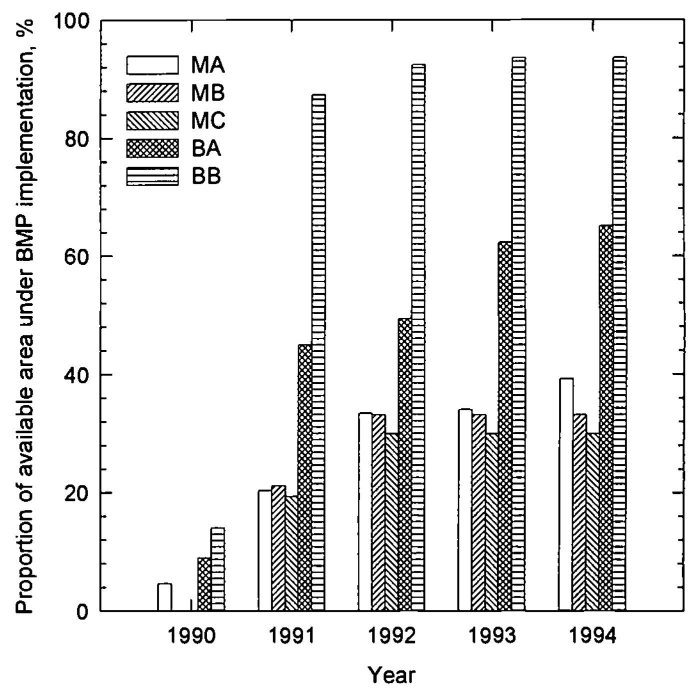

Figure 3. Proportions of Potential Land Area with BMPs Implemented.

Several dead poultry composters and waste storage structures had been constructed in the basin by the end of the monitoring period. Eight dead bird composters as well as six waste storage structures were constructed within site MA's monitored area; of these, one dead poultry composter and two waste storage structures had been constructed within site MB's monitored area. There was only one dead poultry composter and no waste storage facilities constructed within site BA's monitored area, and the composter was not located within site BB's monitored area.

Mean and Median Concentrations of Analysis Parameters

Tables 2 and 3 list arithmetic mean and median concentrations, respectively, of analysis parameters. Comparing Table 1 to Tables 2 and 3 points out that the highest mean and median concentrations of PO_4-P, TP, and TSS were associated with the highest proportions of pasture land use. The reasons for the relatively high NH_3-N ammonia concentrations observed at the MB and (particularly) MC sites are unclear. The outstanding differences in land use appear to be the orchard land use present in the MB sub-basin and the residential land uses in the MC sub-basin, but it is not possible to say whether the high NH_3-N concentrations are attributable in part or in whole to these land uses.

TABLE 2. Arithmetic Mean Concentrations of Analysis Parameters.

Parameter	Monitoring Site MA	MB	MC	BA	BB
	(mg/L)				
NO_3-N	0.90* (0.87)**	1.24 (1.22)	0.89 (1.05)	0.92 (1.34)	1.24 (1.20)
NH_3-N	0.39 (1.94)	0.85 (1.74)	1.21 (5.87)	0.04 (0.08)	0.29 (0.66)
TKN	3.56 (10.59)	2.73 (4.08)	3.32 (9.81)	1.77 (5.70)	2.81 (4.71)
PO_4-P	0.06 (0.08)	0.13 (0.21)	0.11 (0.10)	0.05 (0.09)	0.20 (0.24)
TP	0.16 (0.16)	0.28 (0.23)	0.22 (0.16)	0.11 (0.12)	0.32 (0.26)
COD	29.95 (43.59)	27.76 (29.48)	22.48 (30.95)	21.92 (33.46)	29.39 (33.13)
TSS	8.78 (21.93)	27.76 (61.36)	14.09 (27.15)	5.24 (10.81)	28.01 (57.38)

*Mean.
**Standard deviation.

TABLE 3. Median Concentrations of Analysis Parameters.

Parameter	Monitoring Site MA	MB	MC	BA	BB
	(mg/L)				
NO_3-N	0.67 ab*	0.84 a	0.60 ab	0.51 b	0.86 a
NH_3-N	0.02 bc	0.12 a	0.04 b	0.01 c	0.05 ab
TKN	0.77 ab	1.30 a	0.85 ab	0.50 b	1.04 a
PO_4-P	0.04 c	0.04 bc	0.10 ab	0.02 c	0.13 a
TP	0.09 b	0.19 a	0.19 a	0.06 b	0.25 a
COD	15.2 ab	23.4 a	11.7 ab	12.0 b	18.5 ab
TSS	3.4 bc	11.2 a	5.9 b	2.1 c	11.2 ab

*Within-row medians followed by the same letter are not significantly different by Dunn's test at the p=0.05 level.

Time Trends in Analysis Parameter Concentrations

Table 4 lists the fitted regression equations and coefficients of determination for the seven analysis parameters. Beginning and ending concentration values (computed from equations in Table 4) and annual changes in analysis parameter concentrations are summarized in Table 5. The relationships between observed parameter concentrations and those predicted from the regression relationships are demonstrated in Figure 4 for NH_3-N at site MA and Figure 5 for COD at site BB.

The data of Table 4 indicate that except for NO_3-N at the MC site, all N species at all monitoring sites exhibited significant, periodic behavior. Using the b_2 and b_3 coefficients to determine the phase angle indicates that peak NO_3-N concentrations generally occurred during the winter months (December and January) while both NH_3-N and TKN concentrations generally peaked during the spring months (March through May). The timing of peak N concentrations (particularly NH_3-N and TKN) supports a hypothesis that animal manure application is a significant contributor to stream base flow N concentrations. Animal manures are typically applied in the basin in the spring months, near the beginning of the growing season for pasture grasses. The potential source of unoxidized N is thus generally greatest during the spring months, which coincides with the timing of peak stream flow concentrations of unoxidized N.

Concentrations of NH_3-N, TKN, and COD exhibited significant decreases with time for all monitoring sites. Significant decreases in concentration also occurred for NO_3-N and TSS at site BB and for TP at the BA site. The trends in concentrations of N species and COD are consistent with changes in animal manure management activities that occurred with BMP implementation. The point BMPs such as dead poultry composter installation appear not to have played a large role in the changes in N and COD concentrations. As noted earlier, there was no record of a dead bird composter being installed in the BB drainage basin, and only one was installed within the BA drainage basin. Still, stream flow concentrations of NH_3-N, TKN, and COD at these sites exhibited significant decreasing trends. It thus appears that the areal BMPs alone were capable of causing the observed trends in NH_3-N, TKN and COD concentrations. This inference should not be interpreted as suggesting no benefit or a marginal benefit of dead bird composter installation. It is possible that the duration of monitoring was so short that the effects of abandoned dead bird disposal pits on stream quality did not significantly diminish during the study, in which case the benefits of alternative dead bird disposal

TABLE 4. Equations* Relating Analysis Parameter Concentrations (C) to Time.

Parameter	Site	Equation	r^2
NO_3-N	MA	ln(C) = –0.60 + 0.47 sin(t) – 0.34 cos(t)	0.16
	MB	ln(C) = –0.17 + 0.75 sin(t)	0.35
	MC	NS[2]	NS
	BA	ln(C) = –0.55 + 0.70 sin(t)	0.11
	BB	ln(C) = 0.17 – 0.0016 t + 0.69 sin(t) – 1.00 cos(t)	0.47
NH_3-N	MA	ln(C) = –1.84 – 0.0029 t –1.31 sin(t) – 0.64 cos(t)	0.47
	MB	ln(C) = –0.088 – 0.0022 t –1.39 sin(t)	0.36
	MC	ln(C) = –0.74 – 0.0037 t – 1.22 sin(t)	0.37
	BA	ln(C) = –3.09 – 0.0022 t – 0.49 sin(t) – 1.65 cos(t) + 0.002 t cos(t)	0.52
	BB	ln(C) = –0.64 – 0.0039 t – 0.72 sin(t)	0.45
TKN	MA	ln(C) = 0.86 – 0.0031 t – 0.56 sin(t)	0.26
	MB	ln(C) = 1.13 – 0.0024 t – 0.58 sin(t)	0.39
	MC	ln(C) = 0.77 – 0.0018 t – 0.79 sin(t) – 1.62 cos(t) + 0.002 t cos(t)	0.49
	BA	ln(C) = 0.95 – 0.003 t – 1.46 sin(t) – 0.77 cos(t) + 0.002 t sin(t)	0.55
	BB	ln(C) = 0.90 – 0.0022 t – 0.52 sin(t)	0.27
PO_4-P	MA	NS	NS
	MB	NS	NS
	MC	ln(C) = –1.98 – 0.0014 t	0.12
	BA	NS	NS
	BB	ln(C) = –2.03 + 0.44 cos(t)	0.08
TP	MA	NS	NS
	MB	ln(C) = –1.14 – 0.0007 t – 0.0008 t sin(t) + 0.0005 t cos(t)	0.38
	MC	ln(C) = –1.33 – 0.0007 t	0.09
	BA	ln(C) = –2.68 – 0.46 sin(t)	0.10
	BB	ln(C) = –1.30 + 0.0005 t cos(t)	0.12
COD	MA	ln(C) = 3.26 – 0.0016 t sin(t) – 0.0014 t sin(t)	0.17
	MB	ln(C) = 3.63 – 0.0012 t – 0.0008 t sin(t)	0.28
	MC	ln(C) = 3.68 – 0.0032 t – 0.0020 t sin(t)	0.24
	BA	ln(C) = 3.52 – 0.0030 t – 0.0012 t cos(t)	0.22
	BB	ln(C) = 3.96 – 0.0019 t – 0.0009 t sin(t)	0.33
TSS	MA	ln(C) = 1.10 – 0.0012 t sin(t)	0.09
	MB	ln(C) = 2.57 – 0.0013 t sin(t)	0.26
	MC	ln(C) = 1.81 – 0.0016 t sin(t)	0.17
	BA	ln(C) = 0.73 + 0.0010 t sin(t)	0.07
	BB	ln(C) = 3.39 – 0.0017 t	0.37

*$t=2\pi T/365$ where T is days after the beginning of monitoring; r^2 is the coefficient of multiple determination.

would not be apparent without further monitoring. Similarly, the number of dead bird composters installed might have been insufficient to cause a measurable water quality impact.

No significantly decreasing trends in stream flow P concentrations were expected as a result of implementing the three areal BMPs. As discussed earlier, nutrient management and waste utilization can lead to P accumulation in the soil when these practices are based on meeting plant N requirements. The expected trends, if any were present and detectable given the relatively short monitoring duration, would thus be increases in stream flow P concentrations. As demonstrated in Table 5, stream P concentrations generally did not change during the monitored period. The reason for decreasing TP concentrations observed at the MB site is unknown.

No direct relationship between the proportion of monitored area under BMP implementation and water quality improvement should be inferred. One reason for not assuming a direct correlation is that it is possible for the activities on a relatively small proportion of the total monitored area to have a disproportionately large impact on water quality, depending on what was being done prior to BMP implementation, proximity to the monitoring site, and other such factors. Another reason for exercising caution in interpreting the data is that educational activities of the

TABLE 5. Summarized Changes in Analysis Parameter Concentrations.

Parameter	Site	Beginning* Concentration (mg/L)	Ending* Concentration (mg/L)	Change (percent/ year)
NO_3-N	MA	0.55	0.55	No Change
	MB	0.84	0.84	No Change
	MC	0.56	0.56	No Change
	BA	0.58	0.58	No Change
	BB	1.18	0.25	44.2
NH_3-N	MA	0.16	0.01	65.3
	MB	0.92	0.11	55.2
	MC	0.48	0.01	74.1
	BA	0.05	0.01	55.2
	BB	0.53	0.01	75.9
TKN	MA	2.37	0.12	67.7
	MB	3.11	0.30	58.4
	MC	2.15	0.37	48.2
	BA	2.58	0.14	66.5
	BB	2.45	0.29	55.2
PO_4-P	MA	0.04	0.04	No Change
	MB	0.06	0.06	No Change
	MC	0.14	0.04	40.0
	BA	0.03	0.03	No Change
	BB	0.13	0.13	No Change
TP	MA	0.11	0.11	No Change
	MB	0.32	0.16	22.5
	MC	0.26	0.13	22.5
	BA	0.07	0.07	No Change
	BB	0.27	0.27	No Change
COD	MA	26.0	5.5	44.2
	MB	37.7	11.7	35.5
	MC	39.6	1.8	68.9
	BA	33.7	1.8	66.5
	BB	52.3	8.2	50.0
TSS	MA	3.0	3.0	No Change
	MB	13.1	13.1	No Change
	MC	6.1	6.1	No Change
	BA	2.1	2.1	No Change
	BB	29.6	5.7	46.2

*Calculated from equations in Table 4 without periodicity components.

CES are not directly reflected in the data regarding BMP implementation. While many who were contacted by CES might subsequently have received NRCS assistance in implementing BMPs (and thus have been included in the BMP tracking data), there might have been a significant number of persons who, as a result of CES activities, changed their management practices without benefit of NRCS assistance. Such persons could have had a positive impact on water quality without having been accounted for in the information given in Figure 3.

The results of this study generally complement those reported for other basin-scale BMP effectiveness assessments in terms of reduced N concentrations. Walker and Graczyk (1993) found that BMP implementation decreased NH_3-N (as found for this study) and suspended sediment concentrations on one of two monitored streams. Park *et al.* (1994) noted reductions in TKN (as in this study) and sediment, TP concentrations, and runoff reductions and attributed these findings to BMP implementation. The differences between this and the other studies are most likely due to land use and the specific BMPs

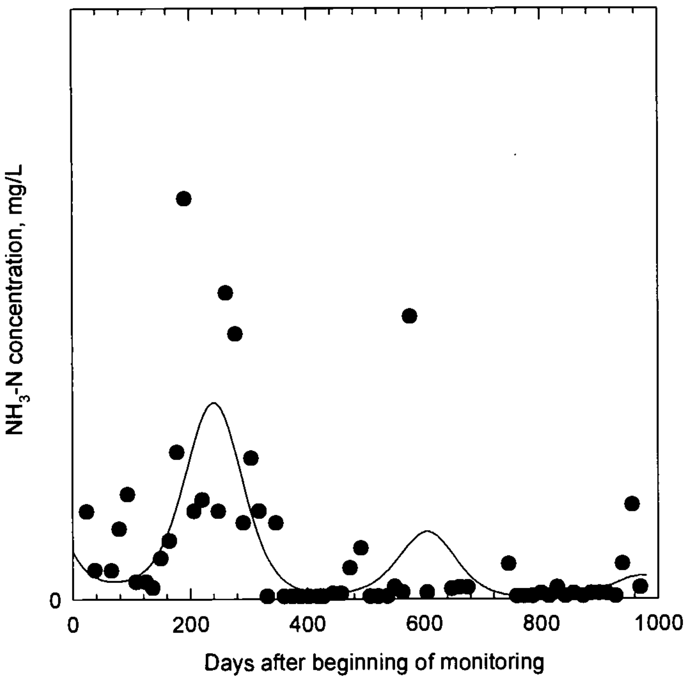

Figure 4. Observed and Modeled Stream Flow NH_3-N Concentrations at the MA Site.

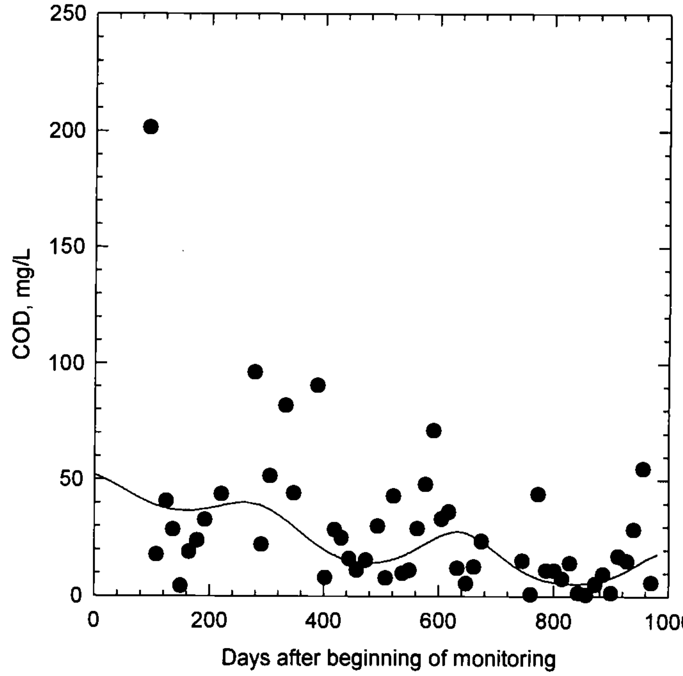

Figure 5. Observed and Modeled Stream Flow COD at the BB Site.

implemented. Reductions in TSS were generally not observed in this study; however, TSS concentrations even at the beginning of monitoring were quite low relative to values typically observed for row-cropped lands. In addition, the BMPs implemented in this study were not oriented toward reducing erosion to the same degree as the other studies. The different findings regarding P concentrations between this study and that of Park *et al.* (1994) might be due to the role of sediment with respect to P transport. It is well-recognized (e.g., Sharpley *et al.*, 1993) that for land areas with high sediment losses (e.g., row-cropped land), more P transport via sediment occurs than for land areas with low sediment losses (e.g., pasture land). The TP reductions reported by Park *et al.* (1994) could thus have been closely related to sediment reductions. The lack of measurable TP reductions in this study might have been due in part to a low proportion of sediment-bound P and no general reduction in TSS concentrations. As discussed earlier, increasing soil P could have been occurring, which would have worked against TP concentration reductions.

SUMMARY AND CONCLUSIONS

Water quality at four stream sites in the Lincoln Lake basin was monitored from September 1991 to April 1994, concurrent with agricultural BMP implementation in the basin. Stream flow concentrations of PO_4-P, TP, and TSS were significantly higher for the two sites having the largest proportions of pasture land use. Regression analyses of the stream flow concentration data indicated significant decreasing trends in concentrations of NH_3-N, TKN, and COD at all sites, with concentrations decreasing from 35-75 percent/year. Stream flow concentrations of NO_3-N, TP, and TSS exhibited decreasing trends only in isolated instances.

The land uses and specific BMPs involved in this study are different from those reported in other studies (Park *et al.*, 1994; Walker and Graczyk, 1993). However, the results of this and the earlier studies are consistent in their findings of water quality improvements associated with BMP implementation. The improving trends in the quality of the basin's tributaries are attributed to BMP implementation within the basin since (a) no other reported activities should have caused the observed water quality changes, and (b) the water quality changes that were observed are consistent with the those that BMP implementation would be expected to produce.

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