

BMP IMPACTS ON SEDIMENT AND NUTRIENT YIELDS FROM AN AGRICULTURAL WATERSHED IN THE COASTAL PLAIN REGION

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ABSTRACT. *The goal of the Nomini Creek watershed monitoring study was to quantify the effectiveness of BMPs at the watershed scale and to determine if the improvements in water quality could be sustained over a long-term period. Information on the long-term effectiveness of BMPs is critical since BMPs are being implemented under the state cost-share program to reduce nonpoint source pollution (NPS) to the Chesapeake Bay. The Nomini Creek project started in 1985 and was completed in 1997. A pre- versus post-BMP design was used. A combination of managerial and structural BMPs was implemented. Major BMPs implemented in the Nomini Creek watershed included no-tillage, filter strips, and nutrient management. The data collected at the 1463 ha Nomini Creek watershed consisted of land use, hydrologic, water quality, soils, and geographical information.*

The BMPs implemented at Nomini Creek reduced average annual loads and flow-weighted concentrations of nitrogen (N) by 26% and 41%, respectively. Average annual total-N loads discharged from the watershed were reduced from 9.57 kg/ha during the pre-BMP period to 7.05 kg/ha for the post-BMP period. Largest reductions were observed for dissolved ammonium-N, soluble organic-N, and particulate-N. In contrast, nitrate-N loads increased after BMP implementation. Increase in nitrate exports was likely due to ammonification and nitrification, and subsequent leaching of particulate-N species that were conserved on the field. In comparison to N, reductions in phosphorus (P) loads and concentrations were not significant. BMP implementation resulted in a mere 4% reduction for total-P with a corresponding 24% reduction in flow-weighted concentration. The average annual total-P loads exported from the watershed were 1.31 and 1.26 kg/ha for the pre- and post-BMP periods, respectively. Reductions in total-P loads were due to decreases in particulate-P. Exports of ortho-P and dissolved organic-P increased after BMP implementation. It is likely that some of this post-BMP increase in dissolved P fractions was associated with dissolution and leaching of particulate-P, and higher rainfall-runoff activity in the watershed during the post-BMP period. In comparison to nutrients, there was no significant change in suspended solids discharged from the watershed. Overall, the findings of this study indicate that the BMPs were effective in reducing the losses of some forms of nutrients, such as ammonium-N and particulate-P, from the Nomini Creek watershed, but additional BMPs are necessary to achieve significant reductions in all forms of N and P.

Keywords. *Monitoring, NPS, BMPs, Agricultural runoff, Nitrogen, Phosphorus, Water quality.*

A study conducted by the U.S. EPA in 1983 concluded that point and nonpoint sources (NPS) of pollution were responsible for the decline of the Chesapeake Bay (USEPA, 1983). NPS pollution was found to be responsible for 67% of the total nitrogen (N) and 39% of the total phosphorous (P) loads that entered the Bay (USEPA, 1983). Agriculture, including cropping and livestock operations, was identified as a major contributor to the NPS pollution of the Bay. Of the total NPS N and P

discharges to the Bay, contributions from cropland were estimated at 60% and 27%, respectively (USEPA, 1983). These nutrient loadings to the Bay have led to excessive plankton production, which has contributed to the loss of submerged aquatic vegetation and an increase in the extent of hypoxic waters.

After the 1983 EPA report documented the decline of the Bay, the governors of Pennsylvania, Maryland, and Virginia; the mayor of the District of Columbia; and the administrator of the EPA drafted a document known as the Chesapeake Bay Agreement. In this agreement, the three states and the District of Columbia agreed to initiate cost-share programs to reduce the amount of NPS pollution entering the Bay and its tributaries. One of the objectives of this agreement was to reduce the N and P loads entering the Bay by at least 40% by the year 2000 (CBP, 1987).

Virginia's cost-share program encourages implementation of BMPs to help reduce NPS pollution to the Bay. Although there is considerable evidence supporting the water quality benefits of BMPs at the field scale, little is known about their impact at the watershed scale, especially, in watersheds with varying topography, soils, and mixed land uses.

Article was submitted for review in November 2000; approved for publication by the Soil & Water Division of ASAE in June 2001.

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Moreover, previous studies on BMP effectiveness have investigated the short-term effectiveness of BMPs and do not indicate whether the water quality improvements from BMP implementation can be sustained over the long term. To answer some of these critical issues, the Nomini Creek watershed project was initiated with the goal of quantifying the effectiveness of BMPs in improving water quality at the watershed scale and over the long term. Specific focus of the study was on cropland BMPs and their impact on surface and ground water quality. The Nomini Creek watershed project was initiated in 1985 and completed in 1997, stretching over a period of nearly 12 years. A pre- versus post-BMP monitoring design was implemented. This article presents the impacts of BMPs on streamflow, sediment, N, and P losses from the watershed.

METHODS

SITE DESCRIPTION

The Nomini Creek watershed is located in Westmoreland County, approximately 80 km northeast of Richmond, Virginia. The watershed is in the Coastal Plain physiographic region. The majority of the watershed lies in the upland area, which is a gently rolling plateau with mild slopes with gradient of less than 6%. The streams in this area are generally found at the bottom of deep ravines that have extreme side-slopes ranging up to 50%. Westmoreland County has a typical humid continental climate with an average annual precipitation of 1000 mm. Of this amount, 560 mm usually falls in April through September, which includes the growing season for most crops. The average air temperature during winter is 3°C, while summer air temperatures average at about 25°C.

Soils of the watershed are generally classified as Ultisols because they are moist soils with argillic horizons. These soils develop in areas that have long frost-free seasons, abundant rainfall, and adequate groundwater supplies. The soils in the watershed consist mainly of the Suffolk and Rumford soil series (approximately 91% of the watershed).

The Nomini Creek watershed was selected for hydrologic and water quality monitoring because it was an agricultural watershed with no point source discharges. The total watershed area is 1463 ha with 49% of the area under cropland, 47% woodland, and 4% under residential and roads. The predominant agricultural activity in this watershed is row cropping, with soybeans, corn, and small grains being the typical crops.

MONITORING SYSTEM

The monitoring system at the Nomini Creek watershed consisted of two runoff monitoring stations, seven raingauge stations, eight groundwater wells, and a weather station. The locations of these stations within the watershed are shown in figure 1.

The main monitoring station (QN1) was located at the watershed outlet. A subwatershed of 214 ha was monitored at station QN2. Water samples from baseflow were collected weekly by a field observer, and storm events were sampled based on changes in stream level using data loggers/microcontrollers and automatic samplers. Discrete water quality samples were collected by water samplers for every 30 mm change in the stream stage during storm events.

ISCO (1979) samplers were installed at both stations and were controlled by 21x microloggers (Mostaghimi et al., 1989). Runoff samples collected at QN1 and QN2 were analyzed for sediment and concentrations of soluble and particulate forms of N and P.

Precipitation was measured at seven sites within and immediately surrounding the watershed (PN1 through PN7, fig. 1). A weather station was installed at site PN5. In addition to precipitation amount and quality, other meteorological variables such as evaporation, wind direction and speed, air temperature, soil temperature, relative humidity, and solar radiation were also monitored at the weather station.

Collection of land use information, including crop rotation and tillage practices began in 1985 through farm operator interviews and Agricultural Soil Conservation and Stabilization (ASCS) records. Starting in 1986, data were also collected on chemical application from the farm operators and commercial chemical suppliers. Geographical information such as elevation, stream networks, and watershed boundaries were digitized from USGS topographic maps.

CROPPING PRACTICES AND BMP IMPLEMENTATION

Row crop agriculture is the primary agricultural activity in the watershed, with corn, soybeans, and small grains (wheat and barley) being the major crops. The typical rotation in the watershed is conventionally tilled corn, followed by small grains, with no-till soybeans planted in small grain residues. Occasionally, full season, conventionally tilled soybeans are also grown (Mostaghimi et al., 1993). Corn is usually planted between late April and early May. Spring N fertilizer application in the Nomini Creek watershed coincides with corn planting in late April. Usually liquid formulations of nitrogen are applied two to four weeks after planting. Additional fertilizer applications occur in October and November when small grains are planted. Typical application rates of fertilizer in the watershed are 112 kg ha⁻¹ of N, 64 kg ha⁻¹ of P, and 84 kg ha⁻¹ of K.

A pre- versus post-BMP monitoring design was implemented in the Nomini Creek watershed. The pre-BMP period extended from January 1986 through June 1989. BMPs were gradually implemented starting July 1989 and

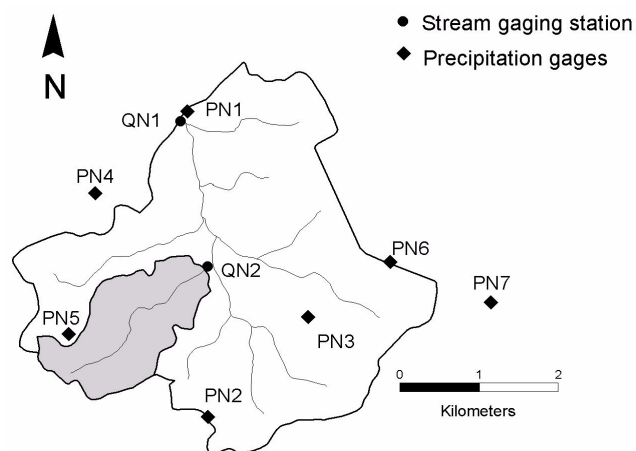


Figure 1. Location of the monitoring stations within the Nomini Creek watershed.

continued until March 1990. The post-BMP period extended from April 1990 to the end of the project in June 1997. The extent of BMP implementation on the 26 farms within the watershed varied from farm to farm (Mostaghimi et al., 1990). Specific agronomic BMPs adopted included: strip cropping, conservation tillage, nutrient management, and integrated pest management (IPM) (Mostaghimi et al., 1999). Structural BMPs implemented in the watershed included: vegetative filter strips, grade stabilization structures, and drop structures (Mostaghimi et al., 1999). Agronomic BMPs were based on NRCS recommendations (NRCS, 1999).

Compared to the pre-BMP period, average cropland area during the post-BMP period increased by 9 %. Increase in fertilizer N application was even greater, at 22% from pre-BMP to the post-BMP period. The area under no-till corn increased by five times compared to the levels reported in 1986. The area under conventionally tilled corn decreased by 50%. Practices such as no-till full-season soybeans, full-season soybeans with strip cropping, and minimum tillage corn were also introduced in 1992–1993. For subwatershed QN2, the increase in no-till corn area was relatively smaller than that for the entire watershed (QN1). In QN2, approximately 22% or 16 ha of additional land was brought under agricultural production (Sims, 1996).

DATA ANALYSES

Average annual streamflow and sediment and nutrient loads and concentrations were computed to perform a pre-versus post-BMP comparison. The annual loads were calculated by accumulating the daily loads. The average annual loads for the pre- or post-BMP period were then computed by dividing the loads by the number of years in each period. Average annual flow-weighted concentrations were computed by dividing the average annual loads by the average annual streamflow.

To obtain an estimate of baseflow and stormflow components of streamflow for possible implications for nutrient delivery, streamflow was separated into baseflow and stormflow components. This separation was performed using the procedure described by Gustard et al. (1992). This procedure has previously been used successfully for Atlantic Piedmont and Coastal Plain watersheds (Jordan et al., 1997b).

Statistical tests were performed on monthly flow-weighted concentrations. Monthly concentrations were computed using a procedure similar to that used for annual values, except that daily loads were accumulated for a month and then divided by the monthly streamflow. Statistical tests were performed on monthly flow-weighted concentrations to exclude the influence of streamflow variability on nutrients.

Since the exploratory tests (Shapiro-Wilk statistic; SAS, 1990) yielded a normal distribution for the log-transformed data and a non-normal distribution for the non-transformed geometric means, the log-transformed data were used for pre- versus post-BMP statistical analysis. The “t” test was the parametric test used, while the Wilcoxon Rank Sum (WRS) test represented the nonparametric alternatives (SAS, 1990).

RESULTS AND DISCUSSION

PRECIPITATION AND STREAMFLOW

The average annual precipitation amounts for the Nomini Creek watershed during the pre- and the post-BMP periods were 967 and 1094 mm, respectively (table 1). Compared to the pre-BMP period, there was a 13% increase in annual precipitation for the post-BMP period. However, the “t” and the WRS tests on monthly precipitations totals did not indicate any significant difference in pre- versus post-BMP values at an α level of 0.10 (table 2).

A plot of monthly averages for pre- and post-BMP periods (results not shown) indicated that most of the 13% increase in rainfall during the post-BMP period was primarily due to high rainfall amounts that occurred during the summer (June–August) and fall (September–November) periods. In the Coastal Plain region, precipitation during summer typically occurs in the form of short, intense, convective storms. A detailed comparison of daily rainfall confirmed that the post-BMP period experienced a larger number of convective storms. With regard to pollutant export from a watershed, an increase in the number of short intense storms would generally suggest a greater possibility for transport of sediment and associated particulate nutrients with surface runoff.

Table 1. Pre- and post-BMP average annual loads and concentrations for QN1 and QN2 (reductions highlighted in bold).

Parameter	QN1						QN2					
	Loads (kg/ha)			Concentration (mg/L)			Loads (kg/ha)			Concentration (mg/L)		
	Pre-BMP	Post-BMP	% change	Pre-BMP	Post-BMP	% change	Pre-BMP	Post-BMP	% change	Pre-BMP	Post-BMP	% change
Precipitation (mm) ^[a]	967	1094	13	—	—	—	967	1094	13	—	—	—
Streamflow (mm) ^[a]	184	231	26	—	—	—	404	469	16	—	—	—
Solids	355	426	20	193.16	183.97	-5	938	1162	24	231.90	247.58	6.7
Dissolved ammonium-N	0.50	0.13	-74	0.27	0.05	-79	1.15	0.34	-70	0.28	0.07	-75
Nitrate-N	1.49	2.02	36	0.81	0.87	8	5.68	8.25	45	1.40	1.76	25
Dissolved-organic-N	2.60	1.21	-53	1.41	0.52	-63	5.56	2.56	-54	1.37	0.54	-60
Particulate-N	4.97	3.68	-26	2.70	1.59	-41	13.20	9.92	-25	3.26	2.11	-35
Total-N	9.57	7.05	-26	5.20	3.04	-41	25.60	21.07	-18	6.32	4.49	-29
Dissolved ortho-P	0.03	0.06	92	0.02	0.03	53	0.07	0.15	123	0.02	0.03	92
Dissolved organic -P	0.26	0.48	83	0.14	0.21	45	0.64	1.0	55	0.16	0.21	34
Particulate-P	1.01	0.70	-30	0.55	0.30	-45	3.0	1.78	-41	0.74	0.38	-49
Total-P	1.31	1.26	-4	0.71	0.54	-24	3.72	2.93	-21	0.92	0.62	-32

^[a] Precipitation and streamflow values are in mm/ha of contributing area: QN1 area = 1463 ha, and QN2 area = 214 ha.

Table 2. "p" values from "t" and Wilcoxon Rank Sum tests.

Parameter	QN1		QN2	
	"t" test	Wilcoxon Rank Sum (WRS) test	"t" test	Wilcoxon Rank Sum (WRS) test
Precipitation	0.3144	0.3076	0.3144	0.3076
Streamflow	0.0001	0.0011	0.0001	0.0001
Solids	0.1798	0.1054	0.3534	0.2897
Ammonium-N	0.0000	0.0001	0.0000	0.0001
Nitrate-N	0.1327	0.0541	0.0001	0.0001
Dissolved-organic-N	0.0001	0.0001	0.0000	0.0001
Particulate-N	0.0204	0.0032	0.0086	0.0017
Total-N	0.0001	0.0001	0.0002	0.0001
Ortho-P	0.8313	0.7832	0.0803	0.5364
Dissolved organic-P	0.3023	0.3774	0.7904	0.1999
Particulate-P	0.0879	0.1953	0.0635	0.0379
Total-P	0.2087	0.3228	0.4141	0.2688

Average annual pre- and post-BMP streamflows from the entire watershed (QN1) were 184 mm and 231 mm, respectively, indicating a 26% increase in streamflow during the post-BMP period (table 1). The corresponding streamflow values for subwatershed QN2 were nearly twice those measured for QN1 (table 1). The pre- and post-BMP average annual streamflows for QN2 were 404 mm and 469 mm, respectively. Both the "t" test and the WRS test for QN1 and QN2 indicated significant differences in the pre- and post-BMP streamflows at an α level of 0.10 (table 2).

For both QN1 and QN2 watersheds, baseflow was the major contributor to streamflow. The pre- and post-BMP baseflows for QN1 and QN2 were 139 mm and 172 mm, respectively. For QN2, the corresponding pre- and post-BMP values were 351 and 405, respectively. The baseflow index values (BFI), defined as the ratio of baseflow to total streamflow, during pre- and post-BMP periods were 0.77 and 0.75 for QN1, respectively. For QN2, the BFI values were even greater at 0.87 and 0.86 for the pre- and post-BMP periods, respectively. The magnitude of BFI values suggests that a large portion of the rainfall falling on upland areas within these watersheds is lost to infiltration, which eventually contributes to streamflow via deep groundwater flow paths. These ratios are similar to those obtained by Shirmohammadi et al. (1987) for the coastal plain physiographic region in the southeast United States.

Streamflows at QN1 and QN2 were generally high during winter and highest during the spring. For both watersheds, streamflows were at their minimum during early fall (August). High spring streamflows were a result of rainfall falling on relatively wet soil surfaces. Compared to the pre-BMP period, post-BMP streamflows during the summer and fall seasons were greater. These higher post-BMP streamflows were primarily a result of greater number of high-intensity rainfall events that occurred during the post-BMP summer and fall seasons and post-BMP periods. However, if such a loss is occurring, it is difficult to estimate the exact amount of nutrient loads exported from the watershed.

BMPs implemented in the Nomini Creek watershed, such as conservation tillage and filter strips, enhance infiltration to deep percolation. Long-term runoff records show that infiltration and potential groundwater recharge could increase by more than 100 mm/year in watersheds farmed

with no-tillage practices, as compared to similar fields with conventional tillage (Edwards et al., 1990). Hence, one would expect that implementation of conservation tillage would tend to increase baseflow contributions for the post-BMP period, but pre- and post-BMP baseflow indices (BFI) estimated for QN1 and QN2 did not vary significantly. On the contrary, average annual BFI values for both QN1 and QN2 decreased slightly during the post-BMP period, indicating a reduction in baseflow contribution. This suggests that BMP implementation did not significantly affect the flow paths taken by water through the watershed.

With respect to the potential nutrient delivery to streams, high baseflow contributions, such as those observed for QN1 and QN2, generally suggest that nutrients that are dissolved and are primarily transported via vertical drainage and groundwater flow have a greater likelihood of being discharged from the watershed, compared to dissolved and sediment-bound nutrients associated with stormflow or episodes of high flow. This has been confirmed by the work of Jordan et al. (1997b), who found that for Atlantic Coastal Plain and Piedmont watersheds, nitrate-N concentrations in streamflow increased with increase in BFI values when cropland constituted more than 10% of the watershed area.

TOTAL SUSPENDED SOLIDS

The pre- and post-BMP average annual suspended solid loads and flow-weighted concentrations for QN1 and QN2 are presented in table 1. Increases in average annual loads were measured at both QN1 and QN2 during the post-BMP period. In contrast, solid concentrations for QN1 decreased slightly (5%) during the post-BMP period, while the concentrations at QN2 increased. Although comparisons of average annual solid concentrations for QN1 indicated a slight decrease, the "t" and the WRS tests did not reveal any significant changes at an α level of 0.10 (table 2).

In general, during the pre-BMP period, discharge of solids from QN1 and QN2 peaked during late winter (February) and early fall (August). Peaks in solids concentration during the fall could have been associated with high-intensity rainfall events during this period. In comparison to the large variations in seasonal values for QN1 solids during the pre-BMP period, post-BMP seasonal solid concentrations were much steadier and did not display any distinct peaks. Despite more high-intensity storms during the post-BMP summer and fall seasons, the solids concentrations during these seasons were lower than their corresponding pre-BMP levels.

Suspended solids and nutrient concentrations discharged from watersheds QN1 and QN2 were determined from streamflow samples collected at the respective watershed outlets. Concentrations measured using this scheme are representative of the combined influence of field and instream sediment transport mechanisms. Studies that have measured solids or sediment at the edge-of-field have found significant reductions due to BMP implementation (Staver and Brinsfield, 1994). Staver and Brinsfield (1994) reported solids concentration of 917 mg/L for a conventionally tilled watershed and 250 mg/L for a no-till watershed. In contrast, watershed scale studies where solids were collected in streams have yielded mixed results (Edwards et al., 1997; Walker and Graczyk, 1993). Edwards et al. (1997) found no significant change in total suspended solids due to BMP

implementation in predominantly pasture watersheds. Suspended solids concentration reported by Edwards et al. (1997) were much lower than those reported by Staver and Brinsfield (1994) and varied between 5 and 30 mg/L. Walker and Graczyk (1993) found reductions in suspended solids from one watershed where no-tillage was implemented, but results from another watershed proved inconclusive.

Instream mechanisms that could influence suspended solids concentration include resuspension and deposition within the stream, and sediment generation via streambank erosion. Instream sediment dynamics are dictated to a large extent by variations in streamflow. Hence, as opposed to edge-of-field measurements, which are more likely to reproduce the influences of BMP implementation, streamflow sampling of suspended solids may also include influences of instream processes that could diminish or mask the effects of BMP implementation. Furthermore, although suspended solids concentration can provide an indication of how much sediment is carried by a stream, it does not include a measure of larger particles that are carried along the streambed as bed load during high flows.

The increase in both suspended solids loads and concentrations from QN2, and the increase in loads and a reduction in concentration from QN1, tend to conform to the mixed trend in results reported in previous watershed-scale BMP impact studies. Because streamflows were higher during the post-BMP period, it is possible that instream contributions via streambank erosion or sediment resuspension were also greater during the post-BMP period compared to the pre-BMP period. A more definitive judgment on the impact of BMPs on solids can only be made by considering both suspended solids measured in the stream and edge-of-field measurements.

NITROGEN

The pre- and post-BMP comparisons for average annual loads and concentrations for QN1 and QN2 are included in table 1. Although QN1 streamflows were greater in the post-BMP period, the average annual total-N load from QN1 decreased by 26%, from 9.57 kgN ha⁻¹ during the pre-BMP period to 7.05 kgN ha⁻¹ for the post-BMP period. The reductions in post-BMP QN1 total-N concentrations were even greater. Reductions in the average annual total-N flow-weighted concentrations were larger compared to corresponding loads, since the percent reduction in loads was offset by the increase in streamflow during the post-BMP period. A considerable portion of the total-N reduction was due to large decreases in dissolved ammonium N, soluble organic-N, and particulate-N components (table 1). In contrast, nitrate-N loads increased by 36%, from 1.49 kg/ha during the pre-BMP period to 2.02 kg/ha for the post-BMP period. Compared to the loads, the increase in annual nitrate-N concentrations was much smaller, at only 8%.

In comparison to the N exports from QN1, N discharges from QN2 were much higher (table 1). QN2 loads were higher because streamflow per unit contributing area from QN2 was nearly twice that from QN1. QN2 total-N concentrations were slightly higher than QN1 and were most likely due to the relatively greater areas of cropland in the QN2 watershed. However, similar to QN1, QN2 displayed decreasing trends for some N species due to BMP implementation and an increasing trend for others. Average

annual total-N load from QN2 decreased by 18%, from 25.60 kgN ha⁻¹ during the pre-BMP period to 21.07 kgN ha⁻¹ for the post-BMP period (table 1). The corresponding average annual total-N concentrations decreased by 29%. Again, similar to the trend observed for QN1, a considerable portion of the QN2 total-N decrease was due to large reductions in dissolved ammonium, soluble organic-N, and particulate-N species. The nitrate-N loads and concentrations, however, increased through the post-BMP period by 45% and 25%, respectively.

For both QN1 and QN2, the "t" test as well as the WRS test indicated a significant difference at an α level of 0.10 between the pre- and post-BMP dissolved ammonium concentrations (table 2). For nitrate, the p value for the "t" test was slightly greater than the α level of 0.10, and hence, the test failed to register a significant difference. On the other hand, the p value for the WRS test was lower than 0.10, indicating that the increase in nitrate concentrations during the post-BMP period was significant. Unlike QN1, both tests indicated significant increases in QN2 nitrate concentrations. The post-BMP reductions for both dissolved organic-N and particulate-N concentrations were significant and were supported by the "t" test and the WRS test (table 2).

To investigate the impact of BMPs on individual N species and their proportions, bar graphs of pre- and post-BMP values for QN1 and QN2 are presented in figures 2 and 3, respectively. Particulate-N constituted nearly half of the total-N discharged from QN1 during the pre-BMP period, and although the absolute amount of particulate-N decreased through the post-BMP period, its proportion of total-N remained the same (fig. 2). The proportion of dissolved organic-N decreased from 27% of total-N during the pre-BMP period to 17% for the post-BMP period. Similarly, the proportion of dissolved ammonium-N dropped from 5% of total-N in the pre-BMP period to 2% during the post-BMP period. In contrast, the proportion of nitrate-N increased from 16% during the pre-BMP period to 29% during the post-BMP period.

Similar to QN1, QN2 particulate-N accounted for 52% of the total-N load but reduced slightly to 47% during the post-BMP period (fig. 3). The proportion of dissolved organic-N reduced from 22% during the pre-BMP period to 12% for the post-BMP period. Similarly, dissolved ammonium-N decreased from 4% to 2% due to BMP implementation. QN2 nitrate-N increased from 22% to 39% for the post-BMP period.

The pre-BMP dissolved ammonium-N exports from both QN1 and QN2 peaked during the spring season, possibly as a consequence of fertilizer application and high rainfall-runoff during this period. BMP implementation resulted in a large reduction in dissolved ammonium exports, not only for spring but across all seasons. Pre-BMP nitrate exports from both QN1 and QN2 were at their maximum during winter, with a continuous decline until early fall, followed by an increase in late fall. For both QN1 and QN2, post-BMP nitrate-N concentrations exceeded the pre-BMP levels. Pre-BMP dissolved organic-N discharges from both QN1 and QN2 were highest during early spring and then decreased toward late summer and early fall. Similar to dissolved ammonium, dissolved organic-N concentrations during the post-BMP period were much lower than their pre-BMP values and did not display the seasonal fluctuations

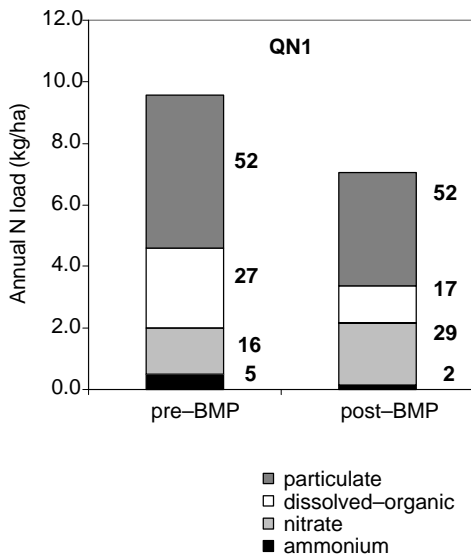


Figure 2. Proportions of various N forms comprising pre- and post-BMP total-N loads for QN1 (percent values listed next to the bars).

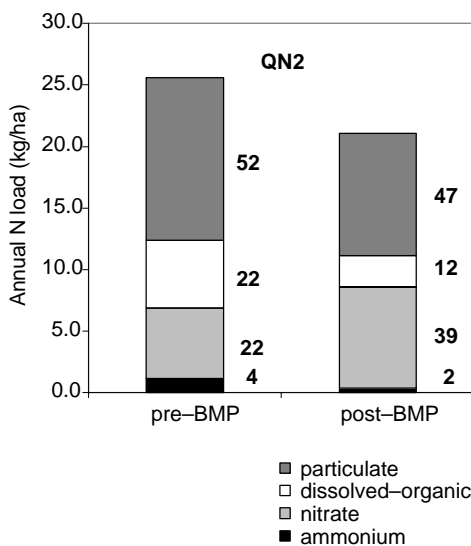


Figure 3. Proportions of various N forms comprising pre- and post-BMP total-N loads for QN2 (percent values listed next to the bars).

observed during the pre-BMP period. In general, pre-BMP concentrations for particulate-N peaked during summer and fall, most likely in association with high-intensity rainfall events and the occurrence of surface runoff during these seasons. In contrast, although the number of summer storms increased during the post-BMP period, the summer and fall particulate-N concentrations were much lower than their pre-BMP values.

For both QN1 and QN2, it appears that although the absolute particulate-N loads decreased due to BMP implementation, the proportion of particulate-N (as that of total-N) remained nearly the same. Dissolved ammonium-N and organic-N species both showed a reduction, but at the cost of increases in nitrate-N. These changes in N species can be attributed to implementation of no-tillage practice within the QN1 and QN2 watersheds. Conservation tillage practices generally result in a reduction of total-N lost via surface runoff (Baker, 1985). This reduction has primarily been attributed to the reduction in the sediment and surface runoff

loss under conservation tillage systems. Although total-N loads in surface runoff are reduced due to conservation tillage, increases in the concentrations of both dissolved N and sediment-bound total-N species have been reported (Lafren and Tabatabai, 1984; Baldwin et al., 1985; Angle et al., 1984). Higher dissolved N concentrations have been attributed to: (a) surface application of fertilizer N (Wells, 1984), (b) higher soil N concentration at the soil surface (Powlson and Jenkinson, 1981), and (c) leaching of N from plant residues remaining on the soil surface (McDowell and McGregor, 1980). Conservation tillage practices have been observed to increase the losses of nitrate-N via leaching (Kanwar et al., 1988; Tyler and Thomas, 1977). One of the reasons for the higher nitrate-N loss is the increased potential for infiltration via macropores under conservation tillage systems (Goss et al., 1978).

With respect to processes that affect the fate and transport of N through the soil profile, conservation tillage seems to have mixed effects. Due to higher levels of residue associated with conservation tillage practices, net N mineralization rates have been observed to be lower than with comparable conventional systems (Gilliam and Hoyt, 1987). Nitrification activity is also reduced under no-till practices compared to conventional tillage (Doran, 1980; Rice and Smith, 1983). For Nomini Creek watersheds, it is very likely then that particulate-N that was retained on the soil surface (due to reduction in surface runoff) leached into the soil profile with infiltrating water. Further, it is likely that the well-drained soil profile of Nomini Creek in the uplands provided a favorable environment for these N species to be converted into nitrate-N through processes such as ammonification and nitrification. This suggests that, although BMPs such as conservation tillage may be effective in reducing the loads and concentrations of organic-N and ammonium-N, a portion of the reductions achieved could be offset by conversion of these species to nitrate-N. Gilliam and Hoyt (1987) and Baker (1985) reached similar conclusions.

The current Bay restoration strategy has a goal of reducing N inputs to the Bay by 40% by the year 2000 (CBP, 1987). BMP implementation on watersheds draining to the Bay is a major component of the efforts to achieve this goal. Although the goal of 40% reduction is not necessarily applicable uniformly across all watersheds within the Chesapeake Bay drainage basin, results from Nomini Creek are evaluated against this target for comparison purposes. BMP implementation for the Nomini Creek watershed (QN1) yielded a decrease in average annual total-N load of 26%. If average annual total-N loads are used as a measure, then BMP implementation for Nomini Creek would not succeed in meeting the required target.

Total-N loads presented above include the influence of streamflow, which in the case of Nomini Creek increased through the post-BMP period, primarily due to an increase in precipitation. To avoid inclusion of variation in rainfall and consequent changes in streamflow (which cannot be controlled), annual flow-weighted total-N concentrations averaged over the pre- and post-BMP periods can also be used as a measure to evaluate the 40% reduction goal. If average annual total-N concentrations exiting QN1 are used, then the BMP implementation yields a reduction of 41%, which meets the targeted reduction goal and suggests that BMPs are effective in reducing N exports.

PHOSPHORUS

In contrast to the 26% drop in average annual total-N loads due to BMP implementation, there was only a 4% reduction in total-P loads at QN1 (table 1). Pre- and post-BMP annual total-P loads discharged from the QN1 watershed changed slightly, from 1.31 to 1.26 kg ha⁻¹. The BMPs were effective in reducing the average annual flow-weighted total-P concentrations by 24%. Total-P reduction was primarily due to large reductions in particulate-P load. Post-BMP particulate-P load was reduced by 30%, from a pre-BMP level of 1.01 to 0.70 kg ha⁻¹. Reductions in particulate-P due to BMP implementation were offset by increased releases of dissolved inorganic and organic P species. Dissolved ortho-P and soluble organic-P loads increased by 92% and 83% from their pre-BMP levels (table 1). Percent increases in concentrations of the dissolved forms were slightly lower than the corresponding load increases. Compared to QN1, BMP implementation appears to have had a greater impact on annual P discharged from QN2. Average annual total-P loads and concentrations discharged from QN2 decreased by 21% and 32%, respectively, due to BMP implementation (table 1). Similar to QN1, although QN2 particulate-P loads and concentrations decreased due to BMP implementation, there was an increase in the discharge of dissolved P forms (table 1).

For QN1, both the “t” test and the WRS test yielded no significant differences between pre- and post-BMP ortho-P values at $\alpha = 0.10$ (table 2). Similar results were observed for dissolved organic-P. For particulate-P, except for the QN1 WRS tests, all other tests indicated significant decreases in P concentrations for the post-BMP period (table 2). Although there was a significant decrease in particulate-P, both the “t” test and the WRS test failed to register any significant difference in total-P for either QN1 or QN2.

The pre- and post-BMP proportions of P for watersheds QN1 and QN2 are reported in figures 4 and 5, respectively. For QN1, although there was only a slight change in the total-P loads for the pre- and post-BMP periods, the shift in the composition of P was apparent (fig. 4). The fraction of particulate-P (as that of total-P) decreased by 21% with an identical increase in dissolved P species. If all of soluble P (dissolved orthophosphorus and organic-P) is assumed as bioavailable, then this would yield a fractional increase in bioavailable P of at least 20%. It is very likely that some increases in the bioavailable fraction of particulate-P can be expected despite the reduction in the total particulate-P load (Sharpley and Halvorson, 1994). Interestingly, although total-P loads discharged from QN2 were nearly three times greater than those discharged from QN1, the reduction in the proportion of particulate-P (and the consequent increase in the proportion of dissolved forms) for QN2 due to BMP implementation (20%) was similar to that observed for QN1 (figs. 4 and 5).

Pre-BMP dissolved ortho-P concentrations displayed multiple peaks during the year. Pre-BMP ortho-P concentrations were high during winter and late spring. During the post-BMP period, ortho-P concentrations increased across all seasons and were especially high during early fall (August). Post-BMP fall ortho-P concentrations were more than five times their pre-BMP values. Again, similar to the pre-BMP ortho-P concentrations, there was no distinct seasonal trend in soluble organic-P concentrations,

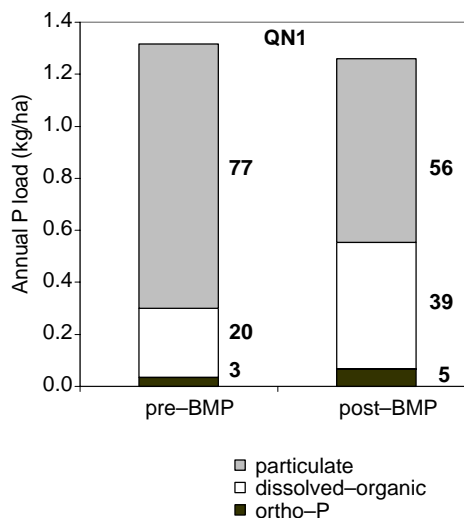


Figure 4. Proportions of various P forms comprising pre- and post-BMP total-P loads for QN1 (percent values listed next to the bars).

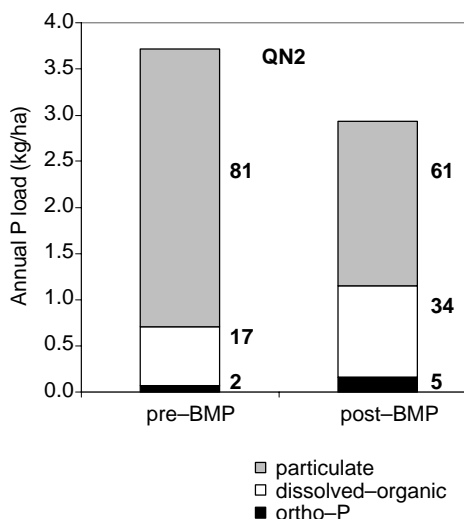


Figure 5. Proportions of various P forms comprising pre- and post-BMP total-P loads for QN2 (percent values listed next to the bars).

although concentrations were high during winter and spring. In contrast, during the post-BMP period, there were distinct peaks in soluble organic-P concentrations during winter and spring that were twice the pre-BMP concentrations. Similar to particulate-N, BMP implementation was effective in reducing fall concentrations of particulate-P.

The amount and form of P discharged from a watershed is influenced by: (a) the processes that affect transport of P on fields and within the soil profile, and (b) those that occur in receiving streams. BMPs such as conservation tillage will typically influence the P transport processes within the fields. Sharpley et al. (1992) and Smith et al. (1991) found that, although the total loss of P from conservation tillage watersheds was lower than from similar conventionally tilled watersheds, the amount of bioavailable P (soluble-P plus bioavailable particulate-P) increased. They found that the mean annual soluble-P concentrations were significantly greater from conservation compared to conventionally tilled watersheds.

Instream processes can significantly alter the amount and bioavailability of P discharged from a watershed compared

to the edge-of-field losses. The processes and transformations include: (a) the selective transport of fine material, which has a greater capacity to sorb or desorb P and hence influences the bioavailability of P; (b) uptake or release of bioavailable P by aquatic biota; (c) deposition or erosion of particulate-P from the streambed with changes in streamflow (Meyer, 1979; Vincent and Downes, 1980); and (d) generation of particulate-P by streambank erosion.

P losses have been observed to increase with watershed discharge (Sharpley et al., 1995). Sharpley et al. (1995) attributed this to the influence of discharge on: (a) the amount of P lost, and (b) the transformations of P that occur during streamflow. Dorioz et al. (1986) found that during wet periods, rainstorms produce a slight increase in river transport of P, while intensive rains during otherwise dry periods increase total-P concentration and load. Sharpley et al. (1995) suggest that the positive relation between discharge and P load indicated in studies by Dorioz et al. (1986) and House and Casey (1988) confirms that even if P concentration increases only slightly or even changes unpredictably, the P load transported during the period of greater discharge increases significantly.

Based on the discussion presented above on P exports from watersheds, two likely explanations can be applied to the post-BMP increase in dissolved P for Nomini Creek watersheds. From results of Sharpley and Halvorson (1994) and Staver and Brinsfield (1994), it is very likely that implementation of BMPs such as no-till practice were responsible for reduction in particulate-P and the consequent increase in the dissolved P concentration. Alternatively, if studies by Dorioz et al. (1986) and House and Casey (1988), and interpretations of Sharpley et al. (1995), are considered, then there is also the possibility that some of the increase in dissolved P concentration during the post-BMP period was associated with increased streamflow. Some of the mechanisms that may be responsible for increases in dissolved P with increased streamflow discharge include: (a) resuspension of stream sediments and consequent desorption, (b) release of P from sediments eroded from streambanks, and (c) reduced opportunity for absorption of dissolved P by phytoplankton.

Similar to N, the current Bay restoration strategy has a goal of reducing P inputs by 40%. As mentioned before, the 40% reduction is not necessarily applicable uniformly across all watersheds within the Chesapeake Bay drainage basin. Based on average annual total-P loads, BMP implementation on the Nomini Creek watershed (QN1) yields only a 4% reduction. If average annual total-P loads are used as a measure, then results from Nomini Creek suggest that the 40% reduction goal for P being sought in the Chesapeake Bay restoration effort cannot be achieved for Nomini Creek watersheds. Alternately, if average annual total-P concentrations (which indicate a 25% reduction) are used, then BMP implementation appears to be much more effective, but the target of 40% reduction is still not achieved.

CONCLUSIONS

The Nomini Creek study indicated that BMPs are effective in improving water quality at the watershed scale, and that these improvements continue to be observed through the 7-year post-BMP period. The reductions in N and P loads

are especially noteworthy considering that rainfall and runoff increased during the post-BMP period, and importantly, that cropland N application also increased during the post-BMP period. Specific observations that can be made from this study include:

- BMPs implemented in the Nomini Creek watershed resulted in reductions of 26% and 41% in average annual N loads and flow-weighted concentrations, respectively. Reductions occurred for dissolved ammonium-N, dissolved organic-N, and particulate-N. However, BMP implementation led to a 36% increase in nitrate-N load. These results suggest that BMPs such as no-tillage, although effective in controlling particulate-N species and reactive (easily adsorbed) soluble N forms, may not be effective in controlling dissolved nitrate-N, which can be lost via leaching and groundwater flow. In contrast, it is likely that these BMPs contributed to nitrate-N exports via conversion and subsequent leaching of particulate-N and soluble N species that were conserved on the field. However, it is important to note here that nitrate-N concentrations leaving the watershed were very low (less than 1 mg L⁻¹). These concentrations are comparable to those reported for forested watersheds (Correll and Weller, 1998) and those observed leaving effective riparian buffers (Correll et al., 1992). It is likely then that nitrate-N concentrations exiting Nomini Creek watersheds have already reached some low background threshold, and it may not be realistic to expect further reductions in these values due to BMP implementation.
- A 26% reduction in N loads was observed for Nomini Creek. Since particulate-N comprised nearly half of the post-BMP load, and since concentrations of dissolved species were already low, it appears that additional efforts need to be directed towards reducing the contribution of particulate-N species.
- In comparison to the 26% reduction in average annual N load, only a 4% reduction in total-P load was achieved due to BMP implementation. Although there was some reduction in particulate-P load (30%), these reductions were offset by increases in soluble ortho-P (92%) and dissolved organic-P loads (83%). The increase in soluble forms indicates that a greater amount of bioavailable P was released into receiving waters. It is likely, though, that some further reductions could be achieved by reducing the particulate-P component, which constituted more than half of the post-BMP load. However, any reduction in particulate-P would also be accompanied by a slight increase in the bioavailable fractions. Despite the possibility of increased proportions of bioavailable P, BMPs can still be considered effective because the release of bioavailable P from particulate material retained on the soil surface affords a greater opportunity for its consumption by crops, as opposed to its release into streams where all of it would be available for phytoplankton.
- This study also seems to confirm an emerging consensus in the literature (Staver and Brinsfield, 1994; Jordan et al., 1997a) that for nutrients like P, despite BMP implementation, it may be impossible to achieve large reductions in watersheds where soils and the hydrogeologic conditions tend to favor its transport and release.

ACKNOWLEDGEMENTS

Funding for this project was provided, in part, by the Department of Conservation and Recreation (DCR), Division of Soil and Water Conservation, Richmond, Virginia. The authors would like to thank the faculty, staff, and students from the department of Biological Systems Engineering who were involved with this project. Assistance of Jan Carr, Anthony Payne, and Dexter Davis in data collection is greatly appreciated. Many thanks to the staff of the Water Quality Laboratory for sample analysis.

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