Historical Changes in Water Quality at German Branch in the Choptank River Basin

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Abstract Many management strategies to improve
the health of Chesapeake Bay focus on reducing
losses of sediments and nutrients from agricultural
land. Plot-scale studies have suggested that Best
Management Practices (BMPs) reduce these losses,
and natural resource managers have since supported
implementation of a variety of BMPs on farms in the
Chesapeake Bay watershed over the last two decades.
As a test of the efficiency of these BMPs at the
watershed scale, all farms within German Branch
watershed had BMPs implemented in the early 1990s.
Using water quality from two past monitoring
programs (i.e., in 1986 and 1991–1995) and current
water quality monitoring (i.e., collected 2003–2006),
we detected a 28% decrease in baseflow P concen-
trations a decade after BMP implementation. There
were no significant changes in nitrate or total nitrogen
concentrations between BMP implementation and the
most recent sampling. However, the significant rate of
increase ($\sim$0.08 mg N L^{-1} year^{-1}) from 1986 to the

1990s did not continue to 2003–2006 baseflow
conditions, which suggests that BMPs may have
suppressed the rate of increase in nitrogen observed
earlier in German. These data suggest that other
management practices that increase agricultural N
losses and natural processes that attenuate N losses at
the watershed scale may obscure significant N
reductions by current BMPs in the watershed.

Keywords Agricultural management · Water quality ·
Nutrients · Chesapeake Bay

Abbreviation

BMP   Best Management Practice
CBP   Chesapeake Bay Program
CREP Conservation Reserve Enhancement
      Program
USDA United States Department of Agriculture

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1 Introduction

Nutrient enrichment and eutrophication of estuarine
and coastal ecosystems is occurring worldwide. The
amount of nutrients in coastal rivers increases with
human populations and the intensive agriculture
supporting these populations (Peierls et al. 1991;
Jordan and Weller 1996; Vitousek et al. 1997; Beman
et al. 2005). Nutrient enrichment and eutrophication
have received much attention in the Chesapeake Bay

on the east coast of the USA due to the extensive algal blooms, oxygen depletion in bottom waters, increased turbidity, and loss of submerged aquatic vegetation (Carpenter et al. 1969; Orth and Moore 1983; Officer et al. 1984; Seliger et al. 1985; Kemp et al. 2005; Fisher et al. 2006). Concerns for the bay's biological health and protection of natural resources led to the formation of the Chesapeake Bay Program (CBP) and its current large-scale restoration effort (CBP 2000).

Chesapeake Bay is the largest estuary in the USA, and its 167,000 km^2 watershed is home to nearly 16 million people. This shallow aquatic system is weakly flushed by tides and has a relatively high ratio of watershed area to water volume; therefore, water quality in Chesapeake Bay is particularly susceptible to intensive land uses which leak nitrogen and phosphorus into waterways. Agriculture covers nearly a third of Chesapeake Bay watershed and is a dominant source of nitrogen (N) and phosphorus (P) to the bay (Magnien et al. 1995). The Chesapeake Bay Program considers reducing N and P loads to be the most critical element in improving water quality and restoration of natural resources, and substantial reduction goals were described in the Chesapeake Bay Agreement 2000. Because of the importance of agriculture as a source of nutrients, much of the resulting management strategies have focused on reducing agricultural nutrient losses.

The bay's natural resource management community has embraced many different Best Management Practices (BMPs) as tools to reduce agricultural nutrient loads. In this paper, we focus on the BMPs determined to be the most cost effective and widely applicable for the Chesapeake Bay region (Chesapeake Bay Commission 2004). These practices include nutrient management plans, conservation tillage, riparian buffers, and cover crops.

Many farmers in the Maryland portion of the Chesapeake Bay region are now required to file nutrient management plans with local Maryland Department of Agriculture offices. The plans involve managing the amount, timing, and placement of fertilizer to minimize nutrient loss to surface and groundwater while maintaining desired crop yields. The concept behind nutrient management plans is to encourage producers to minimize environmental effects while maximizing crop yields; however, there is no enforcement of the plans by the local Maryland Department of Agriculture offices.

Throughout the watershed, most farmers also use conservation tillage. This includes a broad range of soil tillage practices that leave at least 30% of soil surface covered with plant residue after planting to reduce erosion and increase soil organic matter. Continuous no-till, in which crop residue is maintained on the soil surface year round, is the most common in the Chesapeake Bay watershed (Chesapeake Bay Commission 2004).

On many farms, land is under production up to the edge of streams and ditches. However, farmers and natural resource managers are restoring vegetated streamsides of grass and forest, and these riparian buffers create shade and lower stream water temperature, provide large woody debris essential for healthy aquatic habitats, intercept nutrients and sediment in overland flow and subsurface groundwater, and provide wildlife habitat.

Finally, unfertilized winter cover crops have been shown to significantly reduce leaching of agricultural nitrate from the root zone by infiltrating rain water during dormant periods (Staver and Brinsfield 1998). Instead of fertilizing and harvesting winter grain crops, the US Department of Agriculture (USDA) and local county offices have programs which encourage farmers to plant an unfertilized winter cover crop, which is then left on the soil in the spring after herbicide use or ploughing, to immobilize soil N and P and stabilize the soil surface (Staver and Brinsfield 1998).

Scientists in the bay region have evaluated the effects of several conservation practices at plot scales, a term used in this paper to describe research on a single agricultural field. Among these practices are conservation tillage (Staver and Brinsfield 1994; Butler and Coale 2005), grass and forest riparian buffers (Phillips et al. 1993; Jordan et al. 1997; Lowrance et al. 1997), and cover crops (Clark et al. 1997; Staver and Brinsfield 1998). These practices have been shown to reduce losses of agricultural nutrients from farm fields at the plot scale, and these results have been used to make assumptions concerning the effectiveness of BMPs in models that extrapolate the predicted nutrient reductions to the watershed scale (Boesch et al. 2001). Best Management Practices are currently being recommended by the natural resource management community and implemented by farmers throughout the bay watershed. However, past plot scale studies may not have

included important processes such as stream nutrient uptake and groundwater denitrification, which may obscure the effects of BMPs at the watershed scale (e.g., Sutton et al. unpublished).

In 1989, German Branch (also referred to as Jarmin Branch) was selected by the state of Maryland to test the watershed-scale effect of implementation of BMPs on all farms in the subbasin. Stream water quality was monitored as part of a Targeted Watershed Project from 1991 through 1995. Previous monitoring in this watershed indicated relatively high nutrient loads compared to other subbasins in the upper Choptank watershed on the Delmarva Peninsula and was therefore targeted for large-scale restoration (Primrose et al. 1997). The goal of the Targeted Watershed Project was to implement nutrient management plans and BMPs throughout the entire watershed and monitor the effect on nutrient loading to German Branch.

Because German Branch has a rich monitoring history and has been the focus of intensive BMP implementation through the Targeted Watershed Project, it is an ideal case study to evaluate the long-term effects of management actions at the watershed scale. Our objective was to collect various water quality data sets from German Branch watershed over the past two decades and evaluate any changes in nitrogen and phosphorus in relation to implementation of BMPs throughout this time period.

2 Materials and Methods

2.1 Study Site

German Branch is a third order stream located in the Choptank River watershed on the Delmarva Peninsula (Fig. 1).Two associations of soil groups dominate cropland in this coastal plain region: well-drained Sassafras–Woodstown soils and poorly drained Elkton–Othello soils. Hydrologic soil classes C and D with slow infiltration rates dominate the watershed (A=0.6%, B=33.0%, C=13.3%, D=53.1%) and a large proportion (45.2%) is hydric (Norton and Fisher 2000).

With a total watershed area of 52 km^2 and a human population in year 2000 of approximately 680, land use in German Branch watershed is 72% agriculture, 27% forest, and 1% low-density development and animal feeding operations. Almost 50% of the streams have riparian forests that have regrown since channelization in the 1930s and 1940s (Primrose et al. 1997; Norton and Fisher 2000).Of the agricultural land, intensive row crops of grains, soybeans, and corn predominate, which, like much of the row crops in the area, support the large poultry industry centered in the lower region of the peninsula (Staver and Brinsfield 2001).

2.2 Monitoring

Many investigators were involved in measuring water quality in German Branch during three time periods of the past 20 years (1986, 1991–1995, 2003–2006). Data collected during these three time periods were obtained using somewhat different approaches, and below, we characterize the methods of each time period in order to provide information that enables us to separate the effects of different methods from true temporal changes.

In 1986, Norton and Fisher (2000) obtained grab samples from German Branch five–six times per month, largely under baseflow conditions. Temperature and electrical conductivity were measured in the field with portable meters, and samples were kept cold until nutrient analyses were completed. In the lab, unfiltered samples were autoclaved with the persulfate reagents of Valderrama (1981) and subsequently analyzed for dissolved phosphate (PO_4) and nitrate (NO_3) in a Technicon AutoAnalyzer II to determine total P (TP) and total N (TN). Aliquots of the original samples were also filtered with gain flattening filters for automated colorimetric analysis of NO_3 in a Technicon AutoAnalyzer II.

During the Targeted Watershed Project of 1991 through 1995, Jordan et al. (1997) collected weekly composited samples. Automated samplers continuously monitored stream flow and pumped a fixed volume of sample from the stream after a specified volume of flow had passed (i.e., samples were pumped more frequently at higher flow rates). Samples were composited in a single sample bottle on a weekly basis and include both base and storm flows. Jordan et al. (1997) describe the flow-weighting method in more detail. Sample bottles initially contained sulfuric acid as a preservative and were collected approximately monthly for nutrient analyses. Jordan et al. (1997) used perchloric acid

Fig. 1 Schematic of Chesapeake Bay watershed in the Mid-Atlantic region of the USA and location of the Choptank River. Located in the Choptank watershed is the study site, German Branch, and the US Geological Survey gauging station (#01491000) at Greensboro, MD. The *enlarged version* of German Branch shows the sampling site and land use in the watershed, including agriculture, low-density development, animal feeding operations, and forests
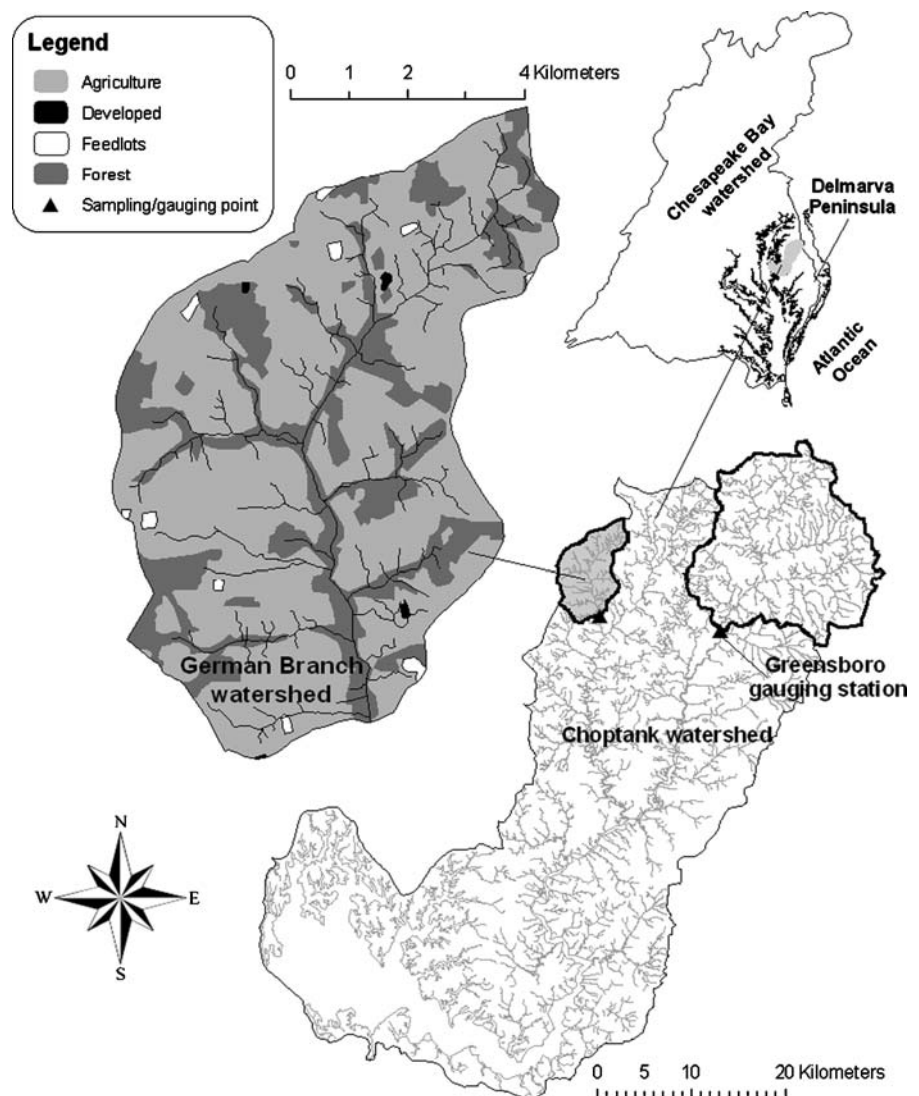
digestion and colorimetric analysis of PO_4 to measure TP. Total N was measured by Kjeldahl N digestion followed by Nesslerization of the NH_4 in the digestate, and NO_3 was reduced to nitrite and measured by colorimetric analysis with sulfanilamide (see Jordan et al. 1997 for details). Previous comparisons of TN measured using Kjeldahl N digestion + nitrate (US Geological Survey, USGS) and TN measured using persulfate digestion (Fisher et al. 1998) revealed similar TN concentrations using these different methods. Samples from Greensboro, MD, a USGS gauging station (#01491000), taken on the same day by the USGS and Fisher et al. (1998) resulted in TN values with differences <10%.

From January 2003 through December 2006, we collected monthly baseflow grab samples from German Branch. These samples were largely processed as described above for 1986. Samples were exclusively collected at baseflow (3 days without rain). In the lab, TN, TP, and $NO_2 + NO_3$ were processed as described above. Nitrate was >80% of the $NO_2 + NO_3$, and we present the analysis of $NO_2 + NO_3$ as NO_3. The three analytical services labs involved in water chemistry analyses between 1986 and 2006 (Horn Point Laboratory, Smithsonian Environmental Research Center, and the USDA Environmental Quality Laboratory) follow strict QA/QC procedures and have repeatedly analyzed split samples to correct interlab bias.

2.3 Data Calculations

Since sampling techniques during the 1991–1995 monitoring period were different from the techniques during the other two monitoring periods, we transformed the data to enable comparisons between all monitoring periods. The water samples analyzed in 1986 and 2003–2006 were collected at baseflow, but data collected in 1991–1995 by Jordan et al. (1997) included both baseflow and stormflow sampling. We separated the weekly baseflow and stormflow measurements in the 1991 to 1995 data of Jordan et al. (1997) using their reported discharge data (Fig. 2). High storm flow weeks are obvious in Fig. 2, but to distinguish between weeks dominated by baseflow from those with small amounts of stormflow, we fit a sine function to the lowest weekly baseflow discharges to estimate seasonal baseflow variations using the following equation:

$$y = y_0 + a \sin\left(\frac{2\pi x}{b} + c\right) \tag{1}$$

where x is the mid-week monitoring date, y is the flow in m^3 week^{-1}, y_0 is the weekly flow at the initial starting date in July 1990, and a, b, and c are constants
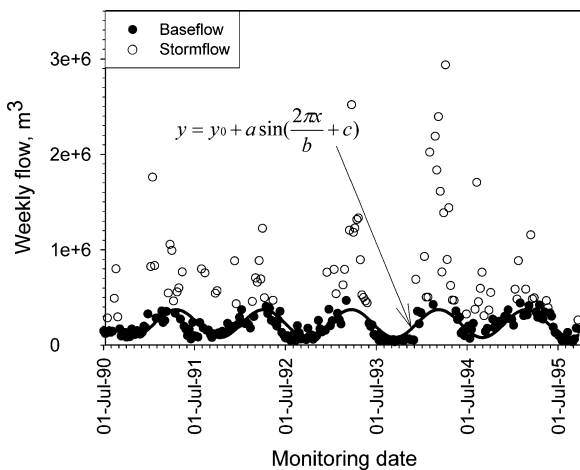
Fig. 2 Weekly streamflow in German Branch during the Targeted Watershed Project from July 1990 to July 1995. The equation $y = y_0 + a \sin\left(\frac{2\pi x}{b} + c\right)$ represents the annual baseflow fluctuations and was fit to the baseflow data with $r^2 = 0.52$***, where y is the weekly streamflow, x is the monitoring date, $y_0 = 2.2 \times 10^5$, $a = 1.5 \times 10^5$, $b = 3.7 \times 10^2$, and $c = -6.3$. Baseflow data ($\pm 1 \times 10^5$ m^3 of predicted weekly baseflow based on the sinusoidal equation) are represented by *closed circles*. Stormflow data ($> 1 \times 10^5$ m^3 of predicted weekly baseflow based on the sinusoidal equation) are *open circles*

($r^2 = 0.52$***). We classified the weekly composited samples of Jordan et al. (1997) as baseflow if the discharge was within 1×10^5 m^3 of the predicted sinusoidal line (closed circles in Fig 2). Using these well-described hydrologic patterns, weekly discharges greater than this departure from the predicted value were assumed to be influenced by rain events during those weeks (open circles in Fig. 2).

We are overestimating stormflow with this approach. One brief storm during a week is probably sufficient to classify an entire week of baseflow in the Jordan et al. (1997) dataset as "stormflow." However, the goal was not to do an accurate baseflow and stormflow separation but rather to exclude stormflow chemistry from the weekly composited data of Jordan et al. (1997) in order to compare baseflow chemistry data during 1991–1995 with the 1986 baseflow chemistry data of Norton and Fisher (2000) and the 2003–2006 baseflow chemistry data reported here.

Nutrient concentrations are often influenced by variations in seasonal and interannual stream flow, and the effects of these must be accounted for when comparing nutrient data over time. German Branch is not a continuously gauged stream, but flow was measured during the Targeted Watershed Project from 1991–1995 by Jordan et al. (1997). In order to calculate volume-weighted nutrient concentrations in baseflow for the other monitoring periods, we compared monthly discharge measured from 1991 to 1995 in German Branch (Q_{GB}, m^3 month^{-1}) to monthly discharge at Greensboro, MD (Q_{GR}, m^3 month^{-1}), a USGS station gauging station (#01491000, Fig. 1) in the Choptank watershed that has been monitored continuously for flow since 1948.

Adjusting for the watershed areas of German Branch (A_{GB}) and Greensboro (A_{GR}), German Branch monthly water yields (Q_{GB}/A_{GB}, cm^3 cm^{-2} month^{-1} = cm month^{-1}) during 1991–1995 were equivalent to those of Greensboro (Q_{GR}/A_{GR}; Fig. 3) for the 1991–1995 period. We used this relationship between the two subbasins to estimate monthly German Branch discharge (Q_{GB}) for 1986 and 2003–2006 using the Greensboro record from those time periods:

$$Q_{GB} = Q_{GR} \times A_{GB}/A_{GR} = Q_{GR} \times 0.19. \tag{2}$$

See Sutton et al. (unpublished) for more details on this approach to regional hydrology.
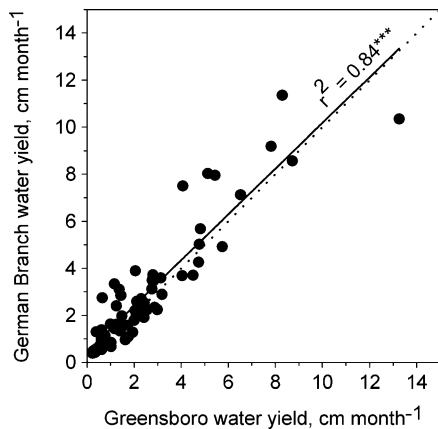
Fig. 3 Monthly water yield from 1991 to 1995 in German Branch and Greensboro USGS gauging station (#01491000) in the Choptank River. The 1:1 line is *dotted* and the regression line for the data is *solid*

Using the estimates of measured monthly discharges, we calculated annual volume-weighted mean concentrations and standard errors for measured chemical parameters at German Branch. The nutrient concentrations reported in the three studies described above were arithmetically averaged by month and combined with the estimated monthly discharges (Eq. 2) in the following formulas:

$$C_{vw} = \frac{\sum\limits_{i=1}^{n} C_i * Q_i}{\sum\limits_{i=1}^{n} Q_i} \qquad (3)$$

$$SE_{vw} = \frac{\sqrt{\sum\limits_{i=1}^{n} (C_i - C_{vw})^2 \Big/ (n-1)}}{\sqrt{n}} \qquad (4)$$

where C_{VW}=annual volume-weighted mean concentration, C_i=average monthly nutrient concentration in month i, Q_i=monthly discharge in month i, SE_{vw}=volume-weighted standard error, and n=sample size (number of months).

2.4 Nutrient Budget

To assess nutrient retention in the watershed, we developed a nutrient budget for German Branch from 1991 to 1995. We used estimates of N and P inputs (atmospheric deposition, fertilizer, soybean N fixa-

tion, and human waste) and compared these inputs to the amount of nutrients exported in the stream and as grain harvests during this monitoring period. Atmospheric deposition was measured by Rochelle-Newall et al., in preparation (8 kg N ha^{-1} year^{-1} and 0.1 kg P ha^{-1} year^{-1}). Total fertilizer application rates (i.e., inorganic and organic sources) for soybean, corn, and grains were estimated by Maryland Cooperative Extension Service (Jim Newcomb, Natural Resources Conservation Service, personal communication), and land area in each crop was obtained from Primrose et al. (1997). Soybean N fixation was derived by multiplying 1.11 kg N ha^{-1} year^{-1} by harvested area (Meisinger and Randall 1991). Human population was based on 2000 census data, and waste production was estimated at 4.0 kg N person^{-1} year^{-1} and 1.2 kg P person^{-1} year^{-1} using regional data (Lee et al. 2001). We calculated baseflow and stormflow exports using nutrient data and discharge measurements from Jordan et al. (1997). Removal of N and P in grain harvest was obtained from Primrose et al. (1997).

2.5 Statistics

Statistical tests were performed using SigmaPlot v9 with SigmaStat v3.2 integration. Statistics for the trends in nutrient concentrations over time were determined by multiple regression analysis. The symbols *, **, and *** indicate statistical significance at the $p<0.05$, 0.01, and 0.001 probability levels, respectively; "NS" is used for $p>0.05$.

3 Results and Discussion

3.1 Agricultural Management Changes

Over the past two decades, German Branch has undergone many changes in nutrient management practices (Table 1). Before the Targeted Watershed Project began in 1990, conservation tillage was the only widely implemented BMP and was applied on approximately 50% of the cropland in the watershed (Mark Waggoner, Natural Resources Conservation Service, personal communication). During the Targeted Watershed Project between 1991 and 1995, various federal and state agencies supported implementation of several BMPs for all of the farms in the watershed.

Table 1 Best Management Practices implemented in German Branch watershed during the three water quality monitoring periods

Year	Soil conservation and water quality plans/nutrient management plans	Conservation tillage	Riparian buffers	Cover crops	Data source
	% watershed	% agricultural land	% streamside buffered	% agricultural land	
1985–1989	25[a]	50	62	0	M. Waggoner personal communication
1990–1995	99	65	62	4.2	USDA 1996
1996–2005	100	60–90[b]	72[c]	2[d]	Various, see footnotes

Time frames in this table are slightly different because BMP records were not completely aligned with the timeframes of the monitoring periods. However, this table illustrates the extent of BMPs prior, during, and after the Targeted Watershed Project of 1991–1995

[a] USDA (1996)

[b] Estimate from Natural Resources Conservation Service

[c] USDA Environmental Quality Lab data

[d] MD Department of Natural Resources data

Soil conservation and water quality plans were implemented on 99% of the watershed (Table 1). These plans contained various combinations of the following BMPs: conservation crop rotation, grassed and lined waterways, roof runoff management, grade stabilization structures, various animal waste management practices, and pest management (USDA 1996). Most of the conservation efforts focused on soil erosion and include the following BMPs implemented within the entire watershed from 1991 through 1995: a total of eight stabilization structures (installed where structures were needed for stabilizing the grade and preventing gullies and erosion), 1.4 ha of grassed waterways (perennial grasses established in concentrated runoff areas), and 378 m of lined waterways (concrete or riprap waterway where a grass waterway is not sufficient or cannot be established; USDA 1996). Based on plot-scale studies, natural resource managers estimated that these erosion control practices prevented the loss of 1.4×10^7 kg of soil in German Branch during the Targeted Watershed Project (USDA 1996).

All farms had a nutrient management plan included in the more substantial soil conservation and water quality plan during the Targeted Watershed Project. Also included were two other BMPs that we discuss in this paper: conservation tillage and winter cover crops. Farmers used conservation tillage on the majority of agricultural land (65%); however, winter cover crops were implemented only on small portions of the watershed (4% of agricultural land; Table 1).

After 1995, all farmers in the basin continued to report the amount, timing, and placement of fertilizer in their nutrient management plans. In 1998, a new program began funding the restoration of grass and forest buffers: the Conservation Reserve Enhancement Program (CREP). This program has supported farmers to restore 102 ha and 175 km of streamsides (10% of all streamsides in the subbasin) with riparian buffers, bringing the total streamside buffered in the watershed to 72% (including established forest buffers, 62%; Table 1). In calculating the percentage of restored streamside vegetation in each subbasin, the length of CREP sites that buffered each side of a stream were used to account for the streamside on both sides (streamside length=2×stream length, see Sutton et al. unpublished for details). Implementation of cover crops has not been as extensive as for other BMPs. Cover crop acreage peaked at 155 ha (4% of basin) during the Targeted Watershed Project (Table 1). However, new funding authorized in the Chesapeake Bay Recovery Act of 2005 may support more cover crops in the Chesapeake Bay watershed.

3.2 Water Quality Data

The goal of this study was to measure the effect of the implementation of these BMPs on water quality in German Branch over the last 20 years. However, flow was not continuously gauged in the stream and the sampling techniques during the 1991–1995 monitoring period differed from the other two monitoring

periods. German Branch streamflow was estimated by using data from Greensboro, a nearby stream that has been continuously gauged since 1948. The magnitude of flow at Greensboro was about five times larger than at German Branch due to the five-fold difference in basin sizes (293 vs. 52 km^2, respectively), and there was a strong relationship between monthly water yields (i.e., flow normalized to basin area) measured by Jordan et al. (1997) at German Branch and monthly water yields at Greensboro measured by USGS (Fig. 3; $r^2=0.84***$). However, the slope of the line is significantly <1 (0.97, $p<0.001$) and the intercept is significantly >0 (0.46, $p=0.02$), indicating a slight bias at the Greensboro gauge compared to German Branch during low flow months. However, the strong relationship between water yields allowed us to estimate the area-normalized monthly water yield for German Branch during 1986 and 2003–2006 based on the monthly water yields measured at Greensboro by the USGS.

Rainfall and discharge variations make it essential to calculate volume-weighted nutrient concentrations when comparing data over two decades (Table 2). Comparison of the annual average concentrations and the annual volume-weighted concentrations of these baseflow samples suggests that differences in general were small, <20%, even though the range of annual rainfall between years was 67 to 166 cm (Table 2). Only annual volume-weighted concentrations are used in the remainder of Section 3.

In addition to annual rainfall and discharge variations, comparison of nutrient concentrations between the three monitoring periods is also affected by sampling during baseflow or stormflow conditions. The weekly flow-composited data collected from 1991 to 1995 by Jordan et al. (1997) includes analysis of stream water during rain events, whereas baseflow data collected in 1986 and 2003–2006 does not. In agriculturally dominated watersheds, total N concentrations in baseflow are high and dominated by nitrate. During storm events, TN may decrease, remain relatively unchanged, or decrease, and concentrations of nitrate, the dominant component of TN, typically decrease and are inversely correlated with discharge because groundwater in agriculturally dominated watersheds is highly enriched in nitrate. In contrast, organic N (particulate and/or dissolved) often increases during a storm event. The total N increases or decreases during storms based on the balance of decreasing nitrate and increasing organic N (e.g., Fisher et al. 1998, 2006). Phosphorus concentrations are usually positively correlated with discharge during rain events because a large fraction of P is supplied to streams during storms as particulate P and leached PO_4 in overland flow during rain events. Figure 4 illustrates the behavior of N and P during a storm event in a Choptank subbasin dominated by agriculture close to German Branch. Both total P and total N increased (~1,000% and 25%, respectively) as the stream responded to a 3-cm rain event, on 28 to

**Table 2** Annual average concentrations of NO3, TN, and TP in baseflow at German Branch

Year	Rainfall	Annual average concentration (mg L^{-1})						Annual volume-weighted concentration (mg L^{-1})					
	cm	[NO3-N]	s.e.	[TN]	s.e.	[TP]	s.e.	[NO3-N]	s.e.	[TN]	s.e.	[TP]	s.e.
1986	97	3.4	0.6	4.4	0.2	0.14	0.03	3.1	0.2	3.9	0.4	0.07	0.05
1991	120	3.5	0.1	4.4	0.1	0.20	0.02	3.5	0.1	4.4	0.1	0.20	0.02
1992	81	4.1	0.1	5.1	0.1	0.24	0.02	3.8	0.2	4.7	0.3	0.20	0.05
1993	67	4.2	0.2	5.0	0.2	0.20	0.02	3.5	0.3	4.5	0.3	0.19	0.05
1994	72	3.9	0.1	4.7	0.1	0.19	0.02	3.7	0.2	4.5	0.3	0.18	0.05
1995	85	4.3	0.2	5.2	0.2	0.20	0.02	3.9	0.4	4.7	0.4	0.17	0.04
2003	151	3.6	0.3	5.4	0.3	0.12	0.02	3.4	0.3	4.7	0.4	0.12	0.02
2004	112	4.4	0.4	5.7	0.4	0.08	0.01	3.7	0.4	5.9	0.3	0.06	0.02
2005	103	4.5	0.3	4.6	0.4	0.11	0.02	4.3	0.3	4.4	0.5	0.09	0.02
2006	166	4.4	0.3	5.2	0.2	0.11	0.02	4.2	0.3	5.0	0.2	0.11	0.02

Standard errors are monthly fluctuations from annual means. Annual volume-weighted concentrations are adjusted for variations in discharge as described by Eq. 2. Rainfall is an average of data from three surrounding stations: Royal Oak, MD, Horn Point Lab in Cambridge, MD, and Dover, DE, except 2003 to 2006 which are only from Horn Point

s.e. standard errors
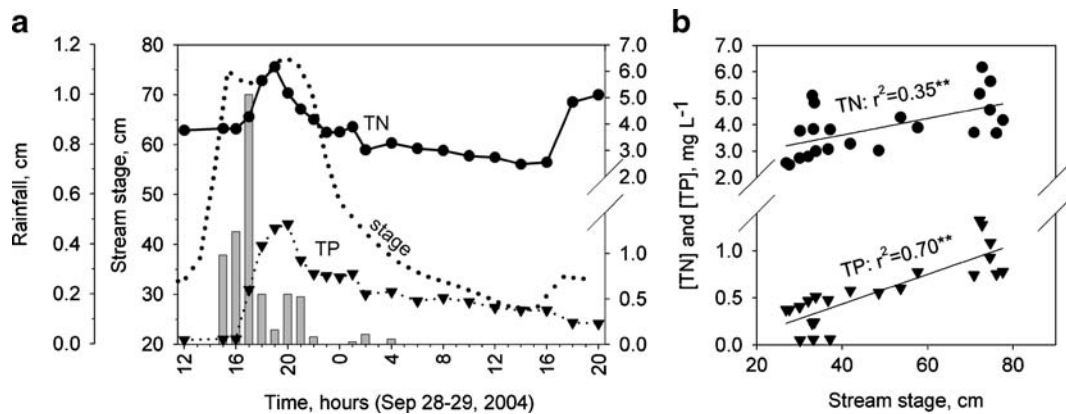
Fig. 4 Stream stage and nutrient concentrations in Blockston Branch in the Choptank watershed during a rain event in September 2004 (rainfall=*bars*; stage=*dotted line*; [TN]=*circles*; [TP]=*triangles*). **a** Stream and nutrient response over time and (**b**) the relationship between stream stage and total N and total P

concentrations (see Sutton 2006 for details). Blockston Branch is a smaller watershed (17 km^2) approximately 10 km south of German Branch and has similar land use (71% agriculture, 28% forest, and 1% low-density development and animal feeding operations)

29 September 2004 (r^2=0.70**; and r^2=0.35**). This behavior of N and P demonstrates changes in stream chemistry between baseflow and stormflow and highlights the importance of comparing data between periods of similar sampling regimes. The data of Jordan et al. (1997) composited at weekly intervals includes these effects but do not enable ready separation of stormflow effects on concentrations because of the composited sampling.

In order to compare data between the three monitoring periods, we analyzed discharge measurements collected by Jordan et al. (1997). Discharge varied during the 1991–1995 monitoring period from 3.4×10^4 to a maximum of 2.9×10^6 m^3 week^{-1} (Fig. 2). This annual sinusoidal pattern in baseflow results from the seasonal variations in groundwater levels. Although long-term average rainfall is relatively constant throughout the year in the Mid-Atlantic region, there are large seasonal variations in evapotranspiration caused by temperature and vegetative growth (Fisher et al. 1998). This results in low groundwater and baseflow at the end of summer and high groundwater, baseflow, and stormflow at the end of winter compounded by random variations in rainfall due to weather patterns. High evapotranspiration rates in summer limit groundwater recharge, but low temperatures and plant harvest or estivation result in high infiltration in fall through spring (Staver 2001a). During the 1991–1995 period, baseflow represented 36% of the total discharge, and stormflow was 64%.

The weekly classification of base and storm flows in Fig. 2 was used to estimate volume-weighted baseflow and stormflow nutrient concentrations during 1991–1995. Volume-weighted average total N concentration over the 5-year monitoring period of Jordan et al. (1997) was 5.1 mg L^{-1} in baseflow and 4.3 mg L^{-1} in stormflow, and average total P concentration was 0.13 mg L^{-1} in baseflow and 0.28 mg L^{-1} in stormflow. Combining the flows and concentrations during base and storm flows enabled us to estimate that during 1991–1995 60% of N and 73% of P was exported during weeks of high flows associated with storm events representing 64% of the total flow. We did not have comparable integrated stormflow data for 1986 and 2003–2006. Therefore, we included only baseflow nutrient concentrations from 1991 to 1995 to compute annual volume-weighted baseflow averages for comparison with baseflow sampling during the other two monitoring periods.

3.3 Water Quality Decadal Trends

Using data from all these monitoring periods (Table 2), the annual volume-weighted nutrient concentrations revealed significant interannual trends in baseflow over the last two decades in German Branch (Fig. 5). Nitrate and total N increased significantly from 1986 through 1995 by 0.08 and 0.09 mg N L^{-1} year^{-1}, respectively (r^2=0.86** for NO$_3$ and r^2=0.83* for TN; Table 3). However, there was no significant change between
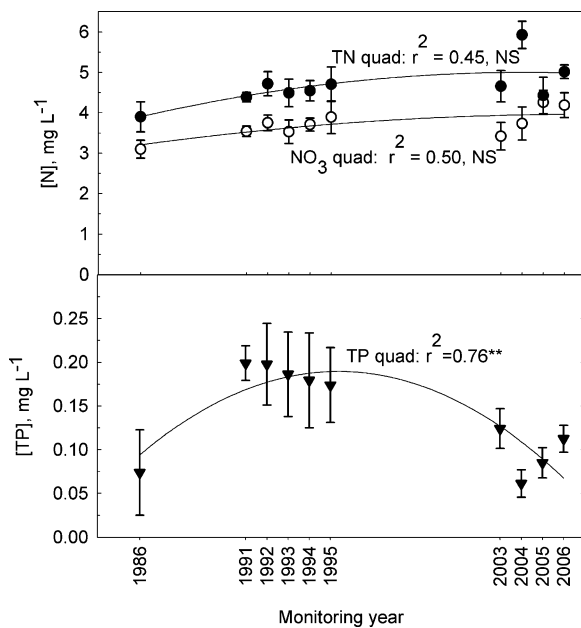
Fig. 5 Annual volume-weighted nitrogen and phosphorus concentrations in German Branch during monitoring years ([TN]=*closed circles*; [NO₃]=*open circles*; [TP]=*triangles*). All data have standard *error bars*. *Solid lines* are quadratic regressions fit to the TN, NO₃, and TP data from the 1986 through 2006 sampling periods

1991–1995 and 2003–2006, suggesting that current N concentrations have stabilized at concentrations observed in the 1990s. The increasing trend in N observed in the earlier time period at German Branch has also been observed in the longer water quality records at Greensboro, the USGS gauging station in the Choptank watershed (Fig. 1). Greensboro is a larger watershed (293 km^2) with less agriculture (48%) than German Branch, and NO$_3$ concentrations are lower and increasing at a lower rate of 0.01 mg NO$_3$-N L^{-1} year^{-1} from 1964 through 2006 (r^2=0.43***; Fig. 6). The smaller size of German Branch (17% of the size of Greensboro watershed) and the larger agricultural

component of land use (72% vs. 48%) explains the three times higher N concentration and ten times faster rate of increase in N concentrations (Table 3). This trend of increasing N observed continuously through four decades at Greensboro and from 1986 through 1995 in German Branch contrasts with the most recent monitoring data at German Branch, which shows no significant changes after the 1990s.

The German Branch data (Fig. 5) showed a significant increase in N through the 1980s and 1990s and an apparent leveling off of concentrations recently. Therefore, we attempted to fit a quadratic equation to the nitrogen data. However, a quadratic relationship did not significantly describe the total N (r^2=0.45, p=0.12) or NO$_3$ (r^2=0.50, p=0.09) concentrations over time (Fig. 5), probably because the changes were small relative to interannual variability.

However, we did detect a significant change in total P concentrations between the sampling periods in German Branch (Fig. 5; Table 3). Although there appeared to be an increase in P concentrations between 1986 and the 1990s, there was no significant trend between 1986 and the 1990s sampling due to interannual variability and low sample size. However, there was a significant decrease in P after the Targeted Watershed Project in the 1990s as expected (Fig. 5; Table 3). Natural resource managers had estimated that the sediment control BMPs would reduce soil erosion in the watershed by a third and nitrogen and phosphorus concentrations by 30–40% (Primrose et al. 1997). A decade after BMP implementation, we detected a decrease of baseflow P concentrations of 28% from an average of 0.134 mg L^{-1} in the 1990s to 0.096 mg L^{-1} in 2003–2006. This change is equivalent to a significant decreasing trend of 0.008 mg P L^{-1} year^{-1} (r^2=0.88***; Table 3). This contrasts with the nearby Greensboro basin where total P has continued to increase significantly by 0.001 mg P L^{-1} year^{-1} from 1970 through 2006 (r^2=0.19*; Fig. 6). Furthermore, unlike nitrogen, total phosphorus concentrations from 1986 through 2006 are described by a quadratic curve with a significant fit (r^2=0.76**; Fig. 5):

$$[TP] = 4.323(year) - 0.001(year)^2 - 4.313 \times 10^3. \quad (5)$$

3.4 Evaluation of Decadal Trends

Since baseflow grab samples were collected in 1986 and 2003–2006, we excluded the composited samples

Table 3 Rate of change of annual volume-weighted nutrient concentrations in German Branch between monitoring periods

	Rate of change (mg L^{-1} year^{-1})		
Sampling periods	TN	NO3-N	TP
1986 to 1990s	0.09*	0.08**	0.01
1990s to 2000s	0.04	0.02	−0.008***

If the symbol representing level of significance (*) is absent, rate of change is not significant

Fig. 6 Annual volume-weighted N and P concentrations in the Choptank River at Greensboro from 1964 to 2006 ([TN]=*closed circles*; [NO₃-N]=*open circles*; [TP]=*closed triangles*; [PO₄-P]=*open triangles*). Data sources: USGS, Fisher et al. 1998, 2006
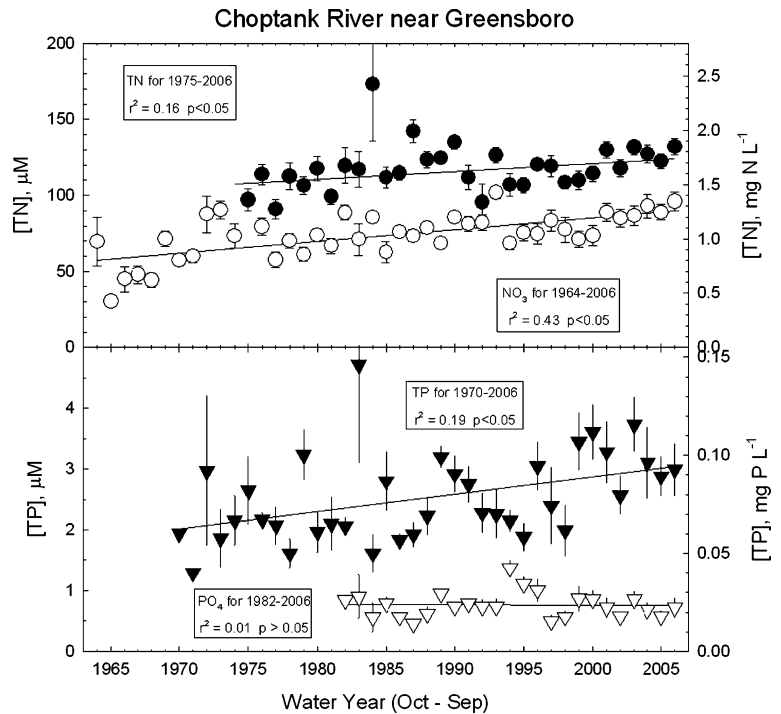
of Jordan et al. (1997) from the trend analysis if collected during weeks of high flow in 1991 to 1995 (Fig. 2). This removes data from large rain events which are likely to contribute high P concentrations in rainwater runoff to the weekly composited samples. Yet, smaller storms may have occurred during weeks of low flow in the 1990s and may be included in the comparison with data from 1986 and 2003–2006, which does not include any storm events. It is possible that the potential inclusion of smaller storm events in the 1990s data of Jordan et al. (1997) contributed to the elevated TP values observed during the 1990s (Fig. 5); however, we have attempted to exclude this as much as possible using the approach shown in Fig. 2. Note that we have focused only on trends in baseflow conditions because there are insufficient historical data available to estimate the impacts of BMPs on nutrient concentrations in stormflow. Focusing on baseflow measurements also ignores most of the particulate-bound P that dominates P transport through the watershed and is not the ideal approach to monitoring P loads to the stream. However, baseflow typically represents 60–80% of total annual streamflow on Delmarva (Lee et al. 2000); our lower value of 36% baseflow at German Branch reported above is an underestimate resulting

from our conservative baseflow classification of the weekly composited data of Jordan et al. (1997), as described above.

Even after considering the sampling differences between monitoring periods, the erosion and sediment control BMPs implemented in German Branch from 1991 to 1995 may explain the apparent decrease in P concentration in 2003–2006. The Targeted Watershed Project estimated that the sediment BMPs reduced soil erosion in the watershed by a third (USDA 1996), and the analysis of the P data in Fig. 5 shows a 28% reduction in stream P concentrations, from 0.134 mg L^{-1} in the 1990s to 0.096 mg L^{-1} during the most recent monitoring period. Although there is a possibility that sampling differences between the two time periods may be driving some of the trend, reductions in baseflow P concentrations have not been observed at the USGS gauging station at Greensboro where BMPs have not been implemented as extensively and total P has continued to increase at a rate of 0.001 mg P L^{-1} year^{-1} since 1970 (Fig. 6). Note that baseflow P concentrations are comparable (0.05–0.20 mg P L^{-1}) in the two Choptank subbasins (Figs. 5–6), but they differ in their temporal trends. Even if we eliminate the 1991–1995 P data of Jordan et al. (1997), it is clear that there has been no significant increase in

baseflow P concentrations at German Branch (Fig. 5), unlike what occurred at Greensboro (Fig. 6).

In addition to erosion control practices, organic nutrient sources applied to cropland in the watershed changed during the monitoring period. Sewage sludge was introduced in German Branch as an organic nutrient source to cropland in the 1980s and peaked in 1990 at applications on 12% of the cropland (Primrose et al. 1997). Poultry manure also increased during this time period from 4% of the P imported into the watershed in 1986 to 13% in 1995 (Primrose et al. 1997). These agricultural and management actions may explain the observed increase in stream P concentrations between 1986 and the 1990s, and the implementation of erosion and sediment control BMPs may have contributed to the reduction in stream concentrations observed by the 2003–2006 sampling. Phosphorus concentrations in streams are likely to respond faster to BMP implementation than nitrogen since a large fraction of P is supplied to the stream by overland flow events as leached, dissolved P or as particulate P. Some of this P mobilized by stormflow is trapped in stream sediments and may contribute to baseflow P. Furthermore, extensive use of no-till agricultural practices tends to concentrate P-rich plant material at the soil surface, increasing the leaching of soluble PO_4 from plant tissues (Staver and Brinsfield 1994), especially in the fall after plant harvest (e.g., fall storm event in Fig. 4). The many erosion and sediment control practices applied in German Branch (Table 1) may also have reduced P losses during rain events and may be a success of the Targeted Watershed Project; however, we suggest that this hypothesis should be tested by analyzing storm-flow in future research for comparison with the stormflow influenced samples of Jordan et al. (1997).

Unlike P, our assessment of monitoring data in German Branch approximately a decade after extensive BMP implementation did not detect significant decreases in nitrogen. However, the data suggest that concentrations in the stream may be beginning to respond to agricultural nutrient management in the watershed (Fig. 5). There was a significant increase in baseflow N at German Branch from 1986 to the 1990s at a rate of 0.08 mg NO_3-N L^{-1} $year^{-1}$ and 0.09 mg TN L^{-1} $year^{-1}$, followed by no significant changes after 1995 (Table 3). In contrast, both TN and nitrate have been steadily increasing by 0.01 mg L^{-1} $year^{-1}$ at the Greensboro gauging station since 1964 (Fig. 6).

This increasing N trend observed continuously at Greensboro and early in the monitoring period in German Branch did not continue at German Branch into 2003–2006 (Fig. 5; Table 3). This suggests that the trend of increasing concentrations of nitrate and total N in the stream has slowed and that concentrations may be maintaining 1990s levels. Continued monitoring could potentially reveal future decreases in N concentrations if the management practices implemented in the 1990s continue.

3.5 Reasons for Lack of Large Responses

Even though N concentrations did not continue increasing after BMP implementation in the 1990s, the goal of the Targeted Watershed Project was to decrease N concentrations. There are three possible explanations for no significant decreases in N from 1991 to 2006: (1) long retention times for groundwater in the surface unconfined aquifer may delay observation of reduced N in baseflow, (2) there may have been changes in farming practices which counterbalanced impacts of the BMPs, and (3) the BMPs may have been ineffective in significantly reducing N concentrations.

With regard to the first possible explanation, Staver (2001b) has estimated groundwater retention time in German Branch watershed by calculating groundwater volume and recharge rate using a digital elevation model, field measurements of groundwater volumes in the unconfined surface aquifer, and groundwater recharge. We have compiled his estimates of groundwater residence time as a cumulative frequency distribution in Fig. 7, which is approximately hyperbolic in form ($r^2=0.99$). The oldest groundwater is less than 80 years old, and the median groundwater residence time is 8 years. Using these data, we estimate that during the Targeted Watershed Project from 1991–1995, <25% of groundwater was replaced under BMPs in the watershed, and it was not likely that decreases in baseflow N would be observed during this short time period. However, if BMPs have been effective, we would expect to observe decreases in N by the 2003–2006 monitoring, when almost 65% of the groundwater had been replaced in the watershed following the extensive BMP implementation in the 1990s (Fig. 7). The response that we observed in Fig. 5, an undetectable rate of increase, suggests that the BMPs listed in Table 1 were only sufficient to
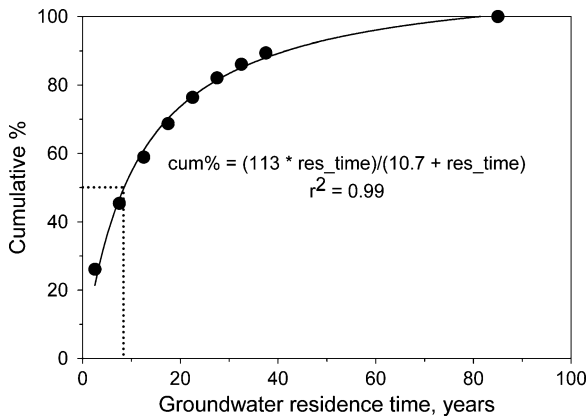
Fig. 7 Cumulative frequency distribution of groundwater residence time in German Branch. A grid of groundwater residence times for German Branch watershed obtained from Staver (2001b) was used to generate the cumulative frequency distribution shown here. Groundwater residence time was estimated using recharge rates collected in the field and groundwater volume, which was calculated using a digital elevation model and field measurements of depth to aquiclude and water table. *Dotted lines* are median values

stop the rate of increase in N concentrations, given other important processes discussed below which can influence the observed concentrations at the watershed scale.

Changes in farming practices may be a second possible explanation for the lack of a decreasing trend in N concentrations. For example, while land areas in agricultural production stayed the same, the total area of harvested crops steadily increased from 4,250 to over 4,860 ha during the Targeted Watershed Project (Fig. 8). This increasing agricultural intensity was due to an increase in wheat and barley production and
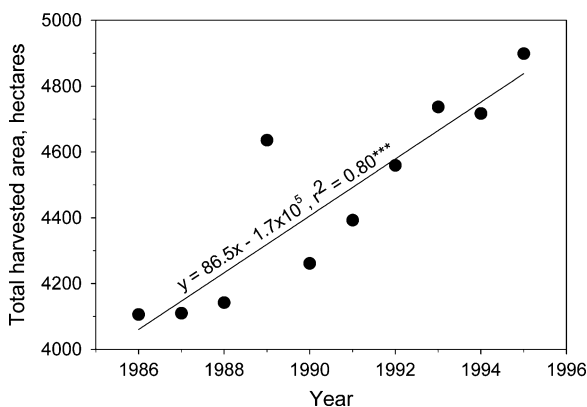
Fig. 8 Amount of harvested cropland in German Branch from 1986 through 1995. Data source: Primrose et al. (1997)

more soybean crop cycles on the same cropland per season (Primrose et al. 1997). Fertilizer applications are likely to have increased as well, as it was applied to more crops during this time period. Nutrient management on farms is voluntary, and changes in farming practices driven by economic and weather-related pressures may overwhelm any current nutrient reductions from BMPs. Changes in federal subsidies, crop prices, and technology (e.g., in this case, double-crop soybean production) can all lead to changes in amounts of fertilizer applied (Primrose et al. 1997).

Other external forces can also affect BMP implementation. For instance, the goal for area of farmland in winter cover crops was not met during the Targeted Watershed Project due to a national shortage of seed in the early 1990s (USDA 1996). This variability in farming practices makes long-term monitoring essential to assessing the water quality effect of BMPs functioning in realistic farming scenarios. These changes in farming practices are likely to have diminished the effects of the BMPs on N and P concentrations.

The third and final reason for the lack of N reductions may be that at the current level of implementation; the chosen BMPs may be ineffective at a large scale. The effects of conservation tillage, riparian buffers, and cover crops on nutrient concentrations have been measured primarily at the plot scale, but not at the watershed scale. Watershed-scale processes such as denitrification and in-stream nutrient processing may dominate and obscure smaller nutrient reductions by BMPs, even when the practices are widely implemented (Sutton et al. unpublished). For example, Lee et al. (2001) have shown that the low oxygen conditions in hydric soils of the Choptank watershed result in <20% transfer of groundwater nitrate to baseflow of streams, presumably due to denitrification.

To assess these watershed-scale nutrient retention processes, we developed an annual nutrient budget for German Branch watershed for the period 1991 to 1995 (Table 4). Fertilizer applications were the largest nutrient input to German Branch (87% of N inputs and 99% of P inputs; Table 4). However, crop removal accounted for only 27% of fertilizer N inputs and 7% of P inputs, and net nutrient export in stream flow was similar to crop removal. Streams exported only 26% of the N inputs and 3% of the P inputs to the watershed (Table 4). Taken together, stream export

Table 4 Annual nutrient budget for German Branch watershed during 1991–1995

Parameters	Data source	Nitrogen		Phosphorus	
		kg N year^{-1}	% of total input	kg P year^{-1}	% of total input
Inputs					
Atmospheric deposition	Rochelle-Newall et al. in preparation	4.2E+04	11.5	5.2E+02	0.3
Fertilizer application	J. Newcomb personal communication	3.1E+05	87.3	1.7E+05	99.2
Soybean nitrogen fixation	Meisinger and Randall 1991	1.6E+03	0.4	–	–
Human waste production	Lee et al. 2001	2.7E+03	0.8	8.2E+02	0.5
Total input		3.6E+05	100.0	1.7E+05	100.0
Outputs					
Export in baseflow	This paper and Jordan et al. 1997	3.8E+04	10.5	1.3E+03	0.8
Export in stormflow	This paper and Jordan et al. 1997	5.6E+04	15.5	3.7E+03	2.2
Crop removal/harvest	Primrose et al. 1997	8.6E+04	23.8	1.3E+04	7.4
Total output		1.8E+05	49.8	1.8E+04	10.4

and crop removal accounted for ~50% of N inputs and ~10% of P inputs, and half of the N inputs and 90% of the P inputs were either stored in the basin or denitrified. The inability to balance watershed inputs and outputs is common (Jordan and Weller 1996; et al. 2001; Lee et al. 2001; Boyer et al. 2002) and represents a major impediment to assessing the effects of attempts to manage watershed nutrient export.

There is a large difference between the amount of fertilizer applied in the watershed and the total amount exported from the watershed as crops or in stream discharge. Stream export and crop removal only accounted for ~50% of N inputs and 10% of P inputs. Therefore, we assume that 50% of N and 90% of P inputs were denitrified or stored within German Branch watershed during this five year period. The unused fertilizer may have remained in the root zone of agricultural fields (primarily P) or it may have been flushed to the groundwater during infiltration events (primarily N). Although N may be stored in groundwater for several decades (Fig. 7), it is likely that denitrification and in-stream processing accounted for a large portion of the remaining N retention.

The percentage of N and P exported as stream flow from German Branch (i.e., 26%) is similar to stream export from other watersheds. Boyer et al. (2002) reported that 10% to 40% of N inputs were exported in stream flow in 16 watersheds in the northeast US Much of the P in German Branch was likely sorbed to soil particles and contributed to reported increases in soil P levels on Delmarva (Sims et al. 1998). However, erosion of P-enriched soil from German Branch may have been trapped by sediment control

BMPs. The nitrogen that was not transformed into crops exported from the watershed or removed in streamflow was likely consumed by noncrop vegetation, denitrified, or transformed within the stream corridors. These large natural sinks for N and P at the watershed scale (i.e., 50% of N inputs, 90% of P inputs) could easily obscure the effects of improvements due to applications of agricultural BMPs.

The nutrient budget revealed that stream nutrient export from German Branch was dominated by stormflow (60% of N and 73% of P). Only using baseflow concentrations in assessing the historical changes in nutrients at German Branch clearly neglects the contributions during stormflow, and monitoring of stormflow in German Branch can reveal other effects of BMP implementation. However, baseflow concentrations capture the changes in nutrient-rich groundwater and baseflow movement of nitrogen and phosphorus over long periods of time. Regardless, BMPs may reduce nutrients at a field plot scale (Staver and Brinsfield 1994, 1998; Butler and Coale 2005; Phillips et al. 1993; Jordan et al. 1997; Lowrance et al. 1997; Clark et al. 1997), but a nutrient budget reveals that BMPs may not always cause detectable changes in nutrient concentrations downstream when applied in watersheds with a natural ability to retain nutrients efficiently (i.e., retention of 50% of N and 90% of P).

3.6 Measuring Effectiveness of BMPs

Many challenges exist for evaluation and assessment of BMP implementation and meeting nutrient reduc-

tion goals in Chesapeake Bay. Conservation tillage is the most widely implemented BMP in German Branch (Table 1). It reduces erosion rates (Staver and Brinsfield 1994) and is potentially cost efficient for farmers by requiring fewer passes over fields to plant crops, which saves time, fuel, and the use of equipment. However, there is evidence that the effect of leaching of plant residue left on the field during conservation tillage can increase P loading to streams (Staver and Brinsfield 1994). Plant residue remaining on the soil surface potentially provides a large amount of P to streams during fall rain events, in some cases, several orders of magnitude above background P levels (e.g., Fig. 4; Fisher et al. 2006).

The management community embraced riparian buffers early in the Chesapeake Bay restoration effort as an effective tool to reduce agricultural nutrient loading (Lowrance et al. 1997). Managers expect restored riparian buffers, mostly through CREP, to be responsible for approximately one third of the total nitrogen and phosphorus reduction goals in Maryland waters (USDA 2004). In the German Branch watershed, CREP buffers added only 10% buffered stream lengths, and funding for new CREP contracts is coming to an end. Almost all buffers that can be restored under this widely used program have been implemented until the program is reauthorized in the next Farm Bill. The challenge now is for scientists to evaluate the actual water quality effects that young buffers have made and the effects as they mature, as long as farmers do not return CREP sites to cropland after initial contracts expire.

Finally, cover crops have been shown to be successful at reducing nitrate leaching to groundwater (Staver and Brinsfield 1998) but have not been widely implemented (e.g., Table 1). As a result of recent Maryland legislation, new monetary sources became available to fund more farmers to use cover crops, but this effort must be continued in order to evaluate potential nutrient reductions. In watersheds similar to German Branch, the effects of management actions on nitrate concentrations in streams are unlikely to be observed for 5 to 10 years (Figs. 5 and 7; Bohlke and Denver 1995) after groundwater nitrate reductions occur under cropland with consecutive plantings of cover crops. It is likely that creating a significant reduction in agricultural nutrient loads to Chesapeake Bay will require long-term funding, more extensive BMP implementation, continuous water quality mon-

itoring, and adaptive management as the success of BMPs is evaluated.

4 Summary and Conclusions

In this paper, we measured the effect of Best Management Practices on water quality in German Branch over the last 20 years. The wide application of BMPs and a long history of water quality monitoring make German Branch a good example of a managed agricultural watershed. This agricultural management may have contributed to the significant reduction in P concentrations in the stream after the 1990s; however, the lack of significant N reductions was not the outcome predicted by the natural resource management community involved in the restoration of German Branch watershed. This evaluation of historical data in German Branch suggests that at a watershed scale, other factors such as storage, denitrification, and in-stream processing may compete with detecting measurable nitrogen reductions. Furthermore, the current level of BMP applications such as CREP and winter cover crops in the watershed is likely to be insufficient to reduce nitrogen concentration. Best Management Practices will affect water quality only if sufficiently and continuously implemented to be at least equivalent to other watershed processes influencing nutrient reductions.

However, we believe that the experience in German Branch has been a valuable exercise in guiding future scientific research and management options in Chesapeake Bay. The outcome of only 6% of river and stream restoration projects are monitored or assessed in Chesapeake Bay watershed (Bernhardt et al. 2005). This makes past studies as well as ongoing research in watersheds such as German Branch critical to the understanding of the effectiveness of BMPs. Research should include studies at the watershed scale in order to assess BMPs embedded with other processes in the environment to determine the level of implementation needed to improve Chesapeake Bay water quality. The health of the bay depends on a more complete understanding of ecological interactions in agricultural landscapes, nutrient management programs, and BMPs that also include consideration of variabilities in socioeconomic factors affecting farming practices.

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