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The impact of agricultural best management practices on downstream systems: Soil loss and nutrient chemistry and flux to Conesus Lake, New York, USA

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ABSTRACT

Six small, predominantly agricultural (>70%) watersheds in the Conesus Lake catchment of New York State, USA, were selected to test the impact of Best Management Practices (BMPs) on mitigation of nonpoint nutrient sources and soil loss from farms to downstream aquatic systems. Over a 5-year period, intensive stream water monitoring and analysis of covariance provided estimates of marginal means of concentration and loading for each year weighted by covariate discharge. Significant reductions in total phosphorus, soluble reactive phosphorus, nitrate, total Kjeldahl nitrogen, and total suspended solids concentration and flux occurred by the second year and third year of implementation. At Graywood Gully, where Whole Farm Planning was practiced and a myriad of structural and cultural BMPs were introduced, we observed the greatest percent reduction (average = 55.8%) and the largest number of significant reductions in analytes (4 out of 5). Both structural and cultural BMPs were observed to have profound effects on nutrient and soil losses. Where fields were left fallow or planted in a vegetative type crop, reductions, especially in nitrate, were observed. Where structural implementation occurred, reductions in total fractions were particularly evident. Where both were applied, major reductions in nutrients and soil occurred. After 5 years of management, nonevent and event concentrations of total suspended solids in streams draining agricultural watersheds were not significantly different from those in a relatively "pristine/reference" watershed. This was not the case for nutrients.

Introduction

Best Management Practices (BMPs) are structural or cultural operational and maintenance practices intended to prevent or reduce the movement of sediment, nutrients, pesticides, and other pollutants from land to surface or groundwater: that is, to prevent nonpoint source pollution. Watershed management programs often cover several levels of geographic scales ranging from the large regional (e.g., Chesapeake Bay, Mississippi River basin) to small local projects representing a field or a single farm (Bishop et al., 2005). In general, BMPs are compatible with the traditional, voluntary approach to resource management practiced in the United States. Nevertheless, it was recognized in the 1990s that this management approach had failed to produce significant national reductions in nonpoint source pollution (Logan, 1990) and had not been widely accepted by the agricultural community, especially in the absence of cost sharing or a clear economic advantage or benefit of the practice (Logan, 1990; Napier et al., 1986). Even though nutrient export to downstream systems from agriculture crop production is believed to be high (e.g., Dillon and Kirchner, 1975; Hill, 1978; Neill, 1989; Correll et al., 1992), implementation of management plans has not been as effective as desired for several reasons (Beegle et al., 2000). Livestock systems, which utilize pastureland for grazing animals and cropland for disposal of manure waste, represent an agricultural activity in which effectiveness of BMPs was not adequately demonstrated (Brannan et al., 2000). Part of the problem was that local implementation of BMPs did not always translate to observable ecosystem or watershed-wide reductions in nutrients and soil loss (Meals, 1996). In fact, BMPs introduced in small portions of watersheds in the Finger Lakes/Lake Ontario region have not effectively demonstrated a connection between nutrient and erosion reduction and visual or measurable lake-wide reductions of nutrients, metaphyton, or aquatic plant populations (Bosch et al., 2001).

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Assessment of BMPs has ranged from subfield experiments (e.g., plots) to large basin-scale monitoring with various sampling and statistical designs. On the subfield level, monitoring studies have demonstrated that agricultural BMPs are effective at decreasing sediment and phosphorus (P) loss (e.g., Murray, 2001; Robillard and Walter, 1984; Udawatta et al., 2002), but relating these results to improvements in water quality has been difficult (Bishop et al., 2005; Meals, 1996; Schindler, 1998). In basin-wide studies, assessments of BMP impacts include monitoring of stream water quality before and after implementation of management practices (e.g., Inamdar et al., 2001; Udawatta et al., 2002) and the use of watershed models based on empirical data (e.g., Benaman, 2002; Gitau et al., 2003). The inability to demonstrate mitigation of stress in downstream systems, however, is the result of confounding factors inherent in nonexperimental evaluations and/or issues of areal size of the BMP relative to the watershed size (Makarewicz, 2009). For example, a single manipulation (i.e., a BMP) in a small area of a large watershed may not provide a large enough reduction in nutrients to affect demonstrable change in downstream nutrient concentration, nutrient loading, and metaphyton and macrophyte population size. Research directed at evaluating BMP impacts on a small watershed scale (e.g., Bishop et al., 2005; Gallichand et al., 1998; Galeone, 1999; Shirmohamadi et al., 1997) is minimal despite the watershed approach being recognized as a method of accounting for environmental variability (Bishop et al., 2005; Carpenter, 1998; USEPA, 1993). The small watershed approach provides an area that is large enough to capture nutrient and soil transport processes and dilution effects (Gburek et al., 2000) yet small enough to focus on nutrient loading from a single farm and BMPs adopted at that scale (Bishop et al., 2005).

Conesus Lake is one of the smaller Finger Lakes in western New York, is used for recreation and fishing, and is a source of municipal water for five local communities. The shoreline area is densely populated with residences, primarily year-round homes. The upstream area is a mixture of agricultural land and mixed deciduous hardwood forests encompassing an area of 16,714 ha. In 1999 about half of the entire land use within the Conesus Lake watershed was and continues to be in agriculture. Much of the agriculture (>70%) is concentrated in the western sub-watersheds of the lake (Fig. 1; SOCL, 2001). The deep, well-drained, glacially derived limestone soils that dominate the watershed are productive and support field crops (field corn, forages, winter wheat, soybeans), vegetable crops (dry beans, sweet corn), and a small acreage of vineyards. Dairy farms are the major animal agriculture with a few livestock and horse farms. More information on Conesus Lake and its surrounding watershed may be found in this issue (Makarewicz, 2009).

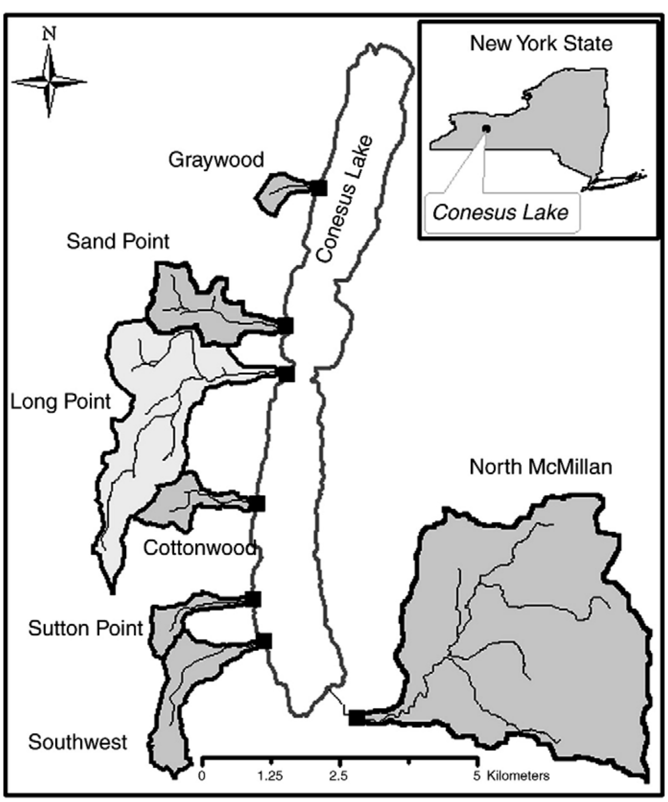

Fig. 1. Conesus Lake and sub-watersheds monitored over the study period. Squares represent sampling sites at the base of the watershed.

The Conesus Lake Watershed Project (Makarewicz, 2009) describes research that quantitatively evaluates and characterizes the effectiveness of various agricultural BMPs in minimizing the local impact of stream loading on water quality. The overall goal of this project was to implement a series of BMPs on individual farms in selected experimental sub-watersheds with an objective of retaining soil and nutrients within the watershed and simultaneously reducing the loss of nutrients and soil to downstream systems. Here we evaluate the effectiveness of agricultural management practices and strategies in improving the water quality of streams by evaluating nutrient and soil loads and their associated concentrations on six watersheds. Specifically, we hypothesize not only reductions in nutrient and soil concentrations and loading from streams draining managed agricultural sites but also reductions in nutrients in soil and in groundwater leaving the watershed.

Methods

Management practices

Within the Conesus Lake watershed, dairy and row crop farms were the focus of Best Management Practices (BMPs) designed to retain soil, nitrate, organic nitrogen (N), and P. Structural BMPs include construction of manure lagoons, terraces, buffer strips, and sediment control basins, while nonstructural or cultural BMPs include such practices that minimize site disturbance through sound planning and design and include cropping sequence, soil testing, fertilization rates, and tillage practices. Structural BMPs (e.g., filter strips) modify the transport of the pollutant to waterways while cultural BMPs (e.g., timing of manure spreading, fertilization rates) are designed to minimize pollutant inputs to waterways through land management practices (Herendeen and Glazier, 2009; Agouridis et al., 2005, Logan, 1990). Structural and cultural BMPs implemented in this study included nutrient management (manure and fertilizer), construction of water and sediment control basins (gully plugs), crop rotation and rotational grazing, removal of acreage from crop production, and in general, improved infrastructure and practices associated with Whole Farm Planning (Table 1).

BMP implementation generally began in the spring of the first year of the study, 9 months after initiation of stream sampling (Herendeen and Glazier, 2009). Structural BMPs were generally in place by the end of the first year, while other practices were introduced in the second year of the study. No management practices were introduced to the North McMillan watershed, our reference watershed. Details on selection of watersheds and implementation of BMPs may be found in Herendeen and Glazier (2009). Acreage of land in various crops was provided by the N.Y. Farm Bureau Office in Livingston County, NY.

Stream discharge

A Water Year (WY) was defined as the period from 1 September to 31 August of the following year. For example, Water Year 1 (WY 1) extended from 1 Sep 2002 to 31 Aug 2003; Water Year 2 (WY 2) extended from 1 Sep 2003 to 31 Aug 2004, etc. A total of 5 water years of daily discharge data was collected starting on 1 Sep 2002 and

Table 1
Watershed area, mean annual daily discharge and daily weighted discharge for seven sub-watersheds of Conesus Lake.

	Cottonwood Gully	Graywood Gully	Sand Point Gully	Sutton Point Gully	Long Point Gully	Southwest Creek[b]	North McMillan Creek	Annual Total for 7 watersheds	Rainfall (cm)
Major management practice implemented	Gully plugs, 28% reduction in cropland, 80% conversion to alfalfa	Whole farm planning (# of BMPs)	Cattle fenced from streams, rotational grazing pens (9.5% of land)	Conversion of 60% cropland to alfalfa	Removal of 37% of crops and cows	Manure pit installed	Reference watershed no BMPs		
Watershed area (ha)	98.8	38.1	188.0	67.5	587.9	176.4	1778.2	2985	
Mean annual discharge (m^3/d)									
(Water Year 1)	1487	629	2324	566	4673	4223	28,790	42,692	87.3
(Water Year 2)	2274	2878	1739	2865	8631	2940	38,796	60,123	110.3
(Water Year 3)	2857	2316	4320	2451	12,708	5214	34,330	64,196	117.8
(Water Year 4)	1556	767	1370	1157	6987	4309	17,434	33,580	90.1
(Water Year 5)	2897	1261	1192	1749	5695	2621	30,430	45,845	95.8
Mean daily discharge (m^3)	2214	1571	2189	1758	7739	3861	29,969	49,287	101.4
Mean annual discharge (m^3/ha/d)	22	41 (29)[a]	12	26	13	22	17	16.5	

Precipitation data were measured in the Graywood Gully watershed. A "Water Year" extends from 1 September to 31 August.
[a] Discharge weighted by extended watershed of Graywood Gully.
[b] Water year is displaced 3 months later compared to other watersheds. Chemistry data for this watershed are presented elsewhere.

ending on 31 Aug 2007. During this period, the stage of six Conesus Lake tributaries (Fig. 1) (Graywood Gully, Long Point Gully, Sand Point Gully, Cottonwood Gully, Sutton Point Gully, and North McMillan Creek) was monitored continuously with a differential pressure transducer (Isco 720) attached to an ISCO recording flow meter (Model 4220 or 6712) equipped with automatic samplers. Southwest Gully was not monitored continuously and is not reported on here. Submerged probe sensors were attached to the cement or metal culverts or bridges at the base of each watershed. In open-channel discharge monitoring, fixed culverts/bases substantially reduce error associated with shifts in stream basin morphometry (Rantz 1982; Chow 1964). Each monitoring station was maintained and calibrated weekly. Streambed movement was verified monthly but was not observed. In addition, discharge measurements were verified annually by remeasuring cross-sectional areas of streams beds, pipes, or cement bridge beds at different stage heights. The velocity–area method (Rantz, 1982) and a calibrated Gurley current meter (Chow 1964) were used to construct rating curves for all streams. During time periods where stage was not available due to probe damage, extreme hydrometeorological events, vandalism, power failure, etc., we estimated stage by using manual readings taken during calibration visits or by using regressions developed from other creek monitoring stations in the Conesus Lake watershed (Rantz, 1982). For example, during a portion of the winter of 2005 when ice dams affected stream depth and discharge at Graywood Gully, we determined discharge here by using a discharge regression from the nearest study tributary, Sand Point Gully ($r^2=0.67$). Similarly, North McMillan Creek and Long Point Gully had abnormally high discharge during the winter of 2004 due to a cracked probe and vandalism. Consequently, hourly discharge here was determined by linear regression ($r^2=0.74$ and 0.98, respectively) based on hourly discharge from Cottonwood Gully.

Soil analysis

Cornell Cooperative Extension personnel collected soils from several fields in the Graywood Gully and Cottonwood watersheds in 2002, 2005, and 2007 (Herendeen and Glazier, 2009). At least 10 replicate soil samples were taken at plow depth from each field using a soil probe. Soil analyses were conducted by the Cornell Nutrient Analysis Laboratory (CNAL, 2007). Bio-available soil nutrients were extracted from soils with Morgan's solution (sodium acetate and acetic acid) buffered at pH 4.8. Activated carbon was added during extraction to aid in organic matter removal and to help decolorize the extraction solution. After 15 min of vigorous shaking at 180 rpm, the extraction slurry was filtered through a fine-porosity filter paper (Whatman #2 or equivalent). Bio-available NO_3–N and PO_4–P were measured using an Alpkem Automated rapid flow analyzer (CNAL Method 1030; Morgan, 1941).

Water sampling and chemistry

Weekly grab stream water samples were taken and analyzed for total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate + nitrite (NO_3+NO_2), total Kjeldahl nitrogen (TKN), and total

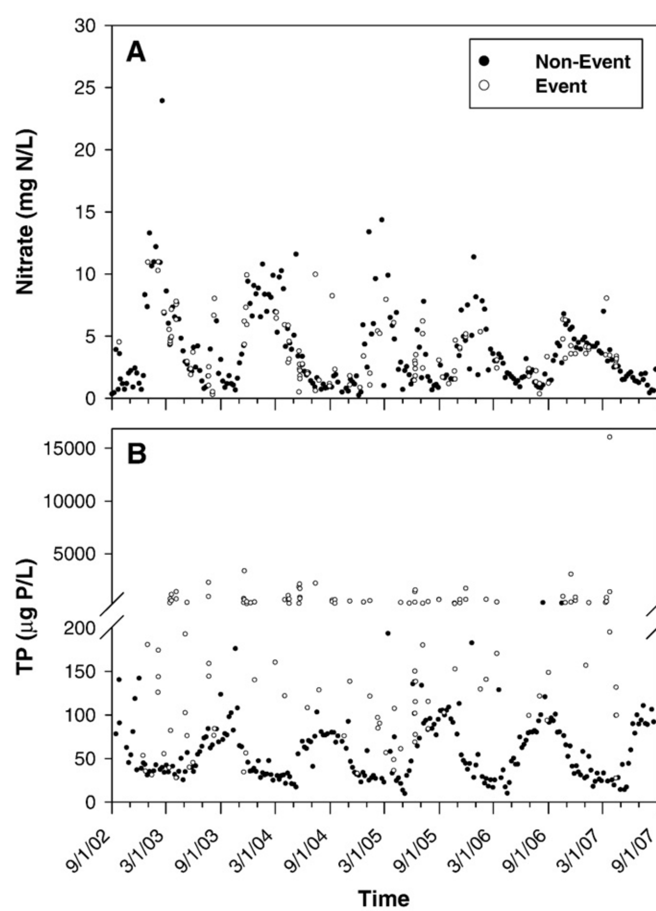

Fig. 2. (a) Time trend in nitrate (Cottonwood Gully) and (b) in total phosphorus concentration (Sand Point Gully), WY 1 to WY 5. A Water Year (WY) is defined as the period from 1 Sep to 31 Aug of the following year.

Table 2
Rating curves of study streams draining into Conesus Lake.

Location	Rating curve equation (Y = discharge in m^3/s, x = stream level in mm)	
Graywood Gully	$Y = 0.000007x^2 - 0.0001756x$	$r^2 = 0.85$
Sand Point Gully	$Y = 0.000007779x^2 + 0.000111669x + 0.000126538$	$r^2 = 0.82$
Long Point Gully	$Y = 0.000030x^2 + 0.00019456x$	$r^2 = 0.86$
Cottonwood Gully	$Y = 0.000006985x^2 - 0.000141730x + 0.000033697$	$r^2 = 0.85$
Sutton Point Gully	$Y = 0.00000000094x^4 - 0.00000010790x^3 + 0.00000947831x^2 - 0.00012311640x$	$r^2 = 0.96$
Southwest Creek	$Y = 0.0088171x^2 + 0.6207933x - 0.0043616$	$r^2 = 0.94$
North McMillan Creek	$Y = 0.000019492x^2 - 0.001387173x + 0.005397970$	$r^2 = 0.82$

suspended solids (TSS) for all watersheds. A hydrometeorological event sample was defined as a rise in the creek level of at least 2.54 cm in 30 min. If this occurred, an ISCO refrigerated sampler (4 to 5 °C) began taking hourly samples until the event had ended. Hourly samples representing rising and falling limbs of the event hydrograph were composited into two water samples. In addition, grab samples of water from two tiles draining a managed (Site 6) and an unmanaged field (Site 7) were taken (Fig. 2, in Herendeen and Glazier, 2009) in and near the Graywood Gully sub-watershed from April through July 2006 (WY 4).

Water samples were taken, preserved, and analyzed using standard methodologies (USEPA, 1979; APHA, 1999). Samples were analyzed for TP (APHA Method 4500-P-F), TKN (USEPA Method 351.2), $NO_3 + NO_2$ (APHA Method 4500-NO_3-F), and TSS (APHA Method 2540D). Except for TSS, all analyses were performed on a Technicon AutoAnalyser II. Method Detection limits were as follows: SRP (0.48 μg P/L), TP (0.38 μg P/L), $NO_3 + NO_2$ (0.005 mg N/L), TKN (0.15 μg N/L), and TSS (0.2 mg/L). Sample water for dissolved nutrient analysis (SRP, $NO_2 + NO_3$) was filtered immediately on site with 0.45-μm MCI Magna Nylon 66 membrane filters and held at 4 °C until analysis the following day.

All water samples were analyzed at the Water Chemistry Laboratory at The College at Brockport, State University of New York (NELAC — EPA Lab Code # NY01449) within 24 h of collection. In general, this program includes biannual proficiency audits, annual inspections and documentation of all samples, reagents and equipment under good laboratory practices. All quality control (QC) measures are assessed and evaluated on an on-going basis. As required by NELAC and New York's ELAP certification process, method blanks, duplicate samples, laboratory control samples, and matrix spikes are performed at a frequency of one per batch of 20 or fewer samples. Field blanks (events and nonevents) are routinely collected and analyzed. Analytical data generated with QC samples that fall within prescribed acceptance limits indicate that the test method was in control. For example, QC limits for laboratory control samples and matrix spikes are based on the historical mean recovery plus or minus three standard deviations. QC limits for duplicate samples are based on the historical mean relative percent difference plus or minus three standard deviations. Data generated with QC samples that fall outside QC limits indicate that the test method was out of control. These data are considered suspect and the corresponding samples are reanalyzed. As part of the NELAC certification, the lab participates semi-annually in proficiency testing program (blind audits) for each category of ELAP approval. If the lab fails the proficiency audit for an analyte, the lab director is required to identify the source and correct the problem to the certification agency.

Calculation of nutrient load

Daily, weekly, monthly, and annual losses of nutrient and soil from each watershed were calculated from continuous discharge measurements and from water chemistry samples collected for each watershed during events and nonevents. During nonevent periods, hourly discharge was summarized into a daily discharge and multiplied by that period's analyte concentration. For non-detect samples, a zero was entered into the calculation; for samples with a recognizable absorbance peak but below the level of quantitation, a concentration of 1/2 the detection limit was used to calculate loss from the watershed. Only eight non-detect samples were measured in the entire study. By summing hourly discharge independently for rising and falling limbs and multiplying by the respective analyte concentration of the composite rising and falling limb of the hydrograph, we calculated the event loss of an analyte via stream drainage from a watershed. The end of an event was defined as the point in time where the descending limb leveled off.

Statistics

A Kolmogorov–Smirnov *D* test revealed that even with natural log (ln) transformations, the data were generally not normally distributed. Moderate violations of parametric assumptions have little or no effect on substantive conclusions from Analysis of Variance (Zar, 1999). Even so, the ln transformed data were used to ensure near normal distribution (Zar, 1999). Natural log transformation is a common practice in hydrologic and watershed studies (Cohn, 1995; USEPA, 1997). Analysis of Covariance (ANCOVA) was used to test for temporal trends in nutrient loading or concentration with stream discharge as the covariate and loading/concentration × sampling year as the interaction term. A significant interaction term indicated that the slope of the loading/nutrient concentration-stream discharge

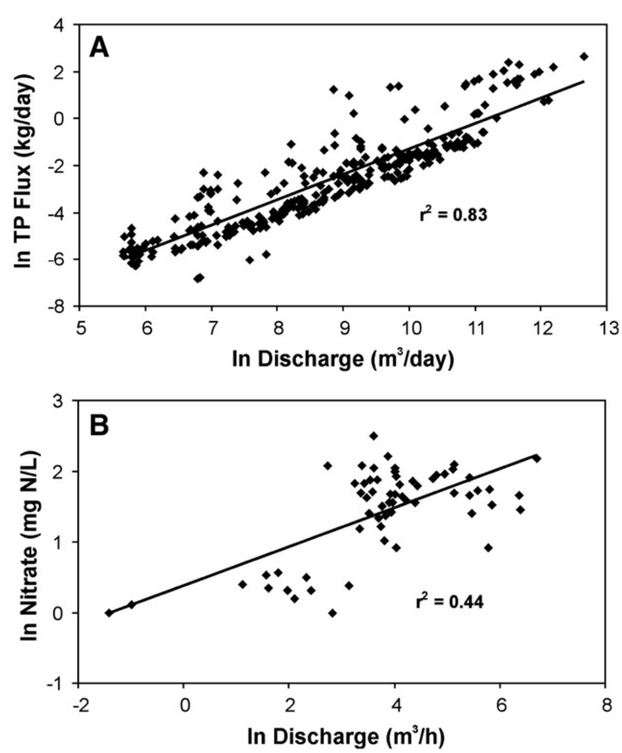

Fig. 3. (a) Total phosphorus (North McMillan Creek) and (b) nitrate (Graywood Gully) concentration versus discharge in WY 5.

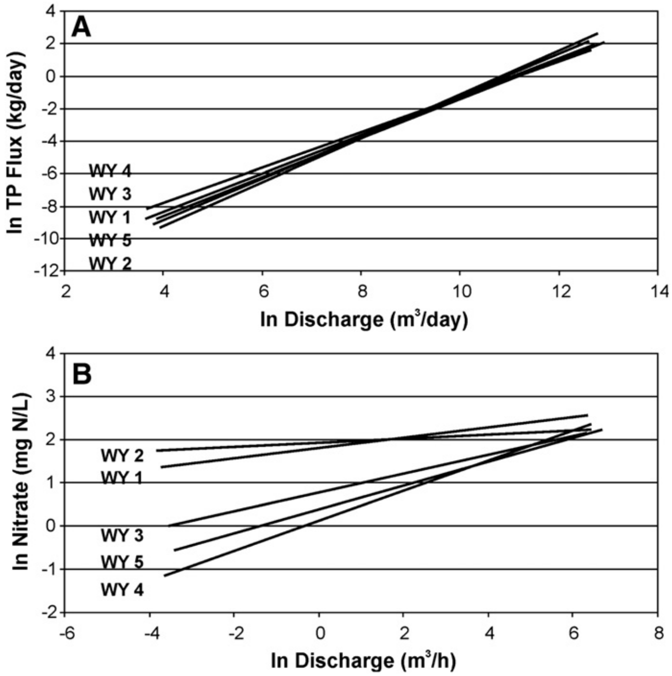

Fig. 4. (a) Total phosphorus flux (North McMillan Creek) and (b) nitrate concentration (Graywood Gully) versus discharge from WY 1 to WY 5. A Water Year (WY) is defined as the period from 1 Sep to 31 Aug of the following year.

regression line was dependent on sampling year. Regression slopes were compared using a pair-wise *t*-test in which significance levels were corrected using the Bonferroni procedure. Regression line elevations were also analyzed for significant differences using a Bonferroni test of estimated marginal means for each sampling year. The Bonferroni procedure offers an adjustment for multiple comparisons and is considered a conservative procedure for post-hoc analysis (Norleans, 2001). Marginal means are analyte concentration means after adjustment for the discharge covariate. The statistical analysis assumed that watersheds were similar in physical, hydrological, and soil properties and that meteorological variations among the closely spaced watersheds were negligible during the study.

Results

Discharge

Correlation between stream stage height and discharge for experimental and reference watersheds was generally very good with coefficient of determination values (r^2) exceeding 0.82 (range 0.82–0.96; Table 2). As expected, considerable variability in precipitation occurred over the 5-year study period. Annual discharge reflected annual precipitation and was highest in WY 3 (Sep 2004–Aug 2005) and lowest in WY 4 (Sep 2005–Aug 2006, Table 1). Overall, greatest discharge occurred in the largest watershed (North McMillan Creek, mean daily = 29,969 m^3) and the lowest in the smallest watershed (Graywood Gully, 1571 m^3). However, areal weighted discharge (m^3/ha) was within the same order of magnitude (13 to 43 m^3/ha), with Graywood Gully having the highest discharge. If weighted by external sources outside the traditional topographical watershed definition, the weighted average dropped to 31 m^3/ha (discussed further in Noll and Magee, 2009).

Analyte concentration and flux

A time-trend analysis indicated that some seasonal patterns existed. For example, in all watersheds, stream $NO_3 + NO_2$ concentration decreased from spring to summer and then increased in early winter (e.g., Fig. 2a). A similar annual cycle of TP and SRP concentrations was observed with a peak in late summer (e.g., Sand Point Gully; Fig. 2b). Because of seasonal cycles, simple plots of time trends do not offer a rigorous statistical approach for analysis. Also, nutrient and TSS concentrations and fluxes are controlled by both analyte mass and water volume (e.g., Figs. 3a and b), two factors which can vary independently. Thus concentrations and fluxes can increase or decrease in response to mass loading and/or dilution. To account for differences in annual discharge in each watershed (Table 1), we utilized the marginal means of the discharge adjusted analyte (TP, SRP, TSS, TKN, and $NO_3 + NO_2$) concentration and flux from the ANCOVA analysis to compare the slopes and elevations of each yearly regression line. We provide two complete examples of this analysis including TP flux of North McMillan Creek and changes in $NO_3 + NO_2$ concentration of Graywood Gully. For all other parameters and creeks, only statistical data are provided for evaluation of elevations of marginal means of flux and concentration.

North McMillan Creek

Temporal trends in TP flux in North McMillan Creek were evaluated by considering regression slope of TP flux versus stream discharge for each collection year using ANCOVA with discharge as the covariate (Fig. 4a). Pairwise *t*-test comparisons of ANCOVA regression slopes for each collection year indicated that the WY 5 regression slope was not significantly different ($df = 1{,}4$; $P > 0.05$; Table 3) from those of all previous water years (1, 2, 3, and 4) (Fig. 4a).

Utilizing marginal discharge means of adjusted TP flux (from ANCOVA analysis), we compared the difference in elevations of each regression line (Table 4). Discharge adjusted marginal mean TP flux remained approximately the same from WY 1 to WY 5 (0.12, 0.15, 0.14, 0.16, 0.15 kg P/D, respectively). A post-hoc Bonferroni test (Table 4)

Table 3
ANCOVA table and slopes of regression lines for respective years for the relationship between discharge and total phosphorus flux, North McMillan Creek.

	df	*F*	*P*
Water Year	4	7.308	0.000
Discharge	1	5019.680	0.000
Year * Discharge	4	6.981	0.000

WY		Pairwise comparisons 1 ($n = 365$)	2 ($n = 366$)	3 ($n = 365$)	4 ($n = 365$)	5 ($n = 334$)
	Slope	1.20	1.35	1.19	1.08	1.29
1			0.277	0.225	0.123	0.132
2				1.000	1.000	1.000
3					1.000	1.000
4						1.000
5						

P values of pair-wise comparisons of the slopes of the regression for each water year (WY) are provided from a Bonferroni multiple comparison test. Probability values <0.05 indicate a significant difference. General linear model in SPSS version 14.0 (SPSS Inc.).

Table 4
Probability values from a comparison (Bonferroni Test) of inter-year elevations of regression lines from Water Years (WYs) 1 to 5 utilizing marginal means of ANCOVA for the relationship between discharge and total phosphorus flux, North McMillan Creek.

WY	1	2	3	4	5
1		0.277	0.225	0.123	0.132
2			1.000	1.000	1.000
3				1.000	1.000
4					1.000
5					

revealed that WY 5 regression line elevation was not significantly different from all other years ($P > 0.05$). The marginal means of the discharge versus TP flux regression lines did not appear to change over time (Table 7).

Graywood Gully

Similar to North McMillan Creek, temporal trends in $NO_3 + NO_2$ concentration of Graywood Gully were evaluated by considering the slope of the regression line of $NO_3 + NO_2$ concentration versus stream discharge for each collection year using ANCOVA with discharge as the covariate. Pairwise *t*-test comparisons of ANCOVA regression slopes for each collection year indicated that WY 5 regression slope was not significantly different ($df = 1{,}4$; $P > 0.05$; Table 5) from WYs 3 and 4 but was significantly different from WYs 1 and 2 ($df = 1{,}4$; $P < 0.001$). Similarly, WYs 3 and 4 were significantly higher than WYs 1 and 2 (Table 5), while WYs 1 and 2 were not significantly different. Over the study, $NO_3 + NO_2$ concentrations decreased faster at lower discharge rates than at higher ones (Fig. 4b).

According to the statistical analyses, discharge adjusted mean $NO_3 + NO_2$ concentration decreased from WY 1 to WY 5 (10.02, 7.77, 5.25, 3.71, and 4.20). A Bonferroni test (Table 6) revealed that WY 5 regression line elevation was significantly lower than in WYs 1 and 2 ($df = 1{,}4$; $P < 0.001$) but not in WYs 3 and 4 ($df = 1{,}4$; $P > 0.72$). Similarly, marginal means were significantly lower in WYs 3 and 4 than in WY 2 (Table 6). The marginal means of $NO_3 + NO_2$ concentration decreased over time (Table 8). In general, the overall results from the six different watersheds are presented as the difference in the elevations of each regression line utilizing the marginal means of the discharge-adjusted analyte concentration and flux from the ANCOVA analysis (post-hoc Bonferroni) (Tables 7–12).

North McMillan Creek

The post-hoc analyses indicated that elevation of the marginal means of concentration and generally the flux of TP, TKN, $NO_3 + NO_2$, SRP, and TSS in WY 1 were not significantly lower ($P > 0.05$) than in WYs 2, 3, 4 and 5 (Table 7). There was no significant difference in the marginal means of flux for TKN and SRP among WYs 1, 2, 3 and 4. Only in WY 5 were marginal means of TKN and SRP significantly lower than in all previous years (Table 7). In general, the marginal means of analyte concentration and flux at North McMillan Creek did not decrease during the study (Table 7).

Graywood Gully

Unlike North McMillan Creek, significant decreases in the marginal means of the flux and concentration of TKN, $NO_3 + NO_2$, TP, and TSS were generally observed from WY 1 to WY 5 (Table 8). There was some analyte to analyte variation in the time from initiation of a BMP to a significant response and in the level or intensity of a response. Significant decreases in $NO_3 + NO_2$ and TKN concentration were observed by WY 3 while significant decreases in TP and TSS were observed a year later in WY 4. Significant decreases in flux mimicked changes observed in concentration but often preceded significant changes in concentration (e.g., TP, TSS, $NO_3 + NO_2$) by a year (Table 8). This difference in the year of significant reduction between concentration and flux was a result of different degrees of statistical freedom between flux and concentration data. For concentration, the number of chemistry samples ranged from ~50 to 90 annually, while flux data were based on daily discharge (~300–400 measurements). No significant decrease was observed in SRP concentration or flux over the study period. Assuming that TP was dominated by the particulate fraction, the general trend from WYs 1 to 5 was a reduction in dissolved and particulate fraction concentration and flux from the watershed. Reductions in analyte concentration ranged from 47% to 65% from 2003 to 2007 (TP: 47%, TKN: 54%, $NO_3 + NO_2$: 58%, TSS: 65%) (Table 13).

Long Point Gully

Inspections of the data revealed that significant ($P < 0.05$) reductions in $NO_3 + NO_2$, TP, SRP, and perhaps TKN concentration and flux were generally observed when compared to the initial WY 1 (Table 9). These decreases corresponded with increases in land taken out of crop production (e.g., Fig. 5). Although there was a significant decrease in TKN concentration and flux from WY 1 to WY 3 and from WY 1 to WY 5, the slight increase in TKN from 527 μg/L in WY 3 to 602 μg/L in WY 4 was large enough to make the 2006 value not significantly different from WY 1. Significant reductions in TSS concentration or TSS flux were not observed. Reductions of 42% and 53% in $NO_3 + NO_2$ and SRP concentration were observed while the total fractions, TP and TKN, decreased 36% and 24%, respectively, from WY 1 to WY 5 (Table 13).

Cottonwood Gully

A general reduction in $NO_3 + NO_2$, TSS, and perhaps TKN was observed over the study period. The post-hoc analysis indicated that the elevation of the marginal means of concentration and the flux was

Table 5
ANCOVA table and slopes of regression lines for respective years for the relationship between discharge and nitrate concentration, Graywood Gully.

		df		*F*	*P*
Water Year		4		9.703	0.000
Discharge		1		34.680	0.000
Water Year*Discharge		4		3.085	0.016

WY		Pairwise comparisons				
		1 ($n=74$)	2 ($n=93$)	3 ($n=87$)	4 ($n=67$)	5 ($n=62$)
	Slope	0.119	0.049	0.218	0.348	0.437
1			0.962	0.000	0.000	0.000
2				0.005	0.000	0.000
3					0.940	0.907
4						1.000
5						

P values of pair-wise comparisons of the slopes of the regression for each Water Year (WY) are provided from a Bonferroni multiple comparison test. Probability values <0.05 indicate a significant difference. General linear model in SPSS version 14.0 (SPSS Inc.).

Table 6
Probability values from a comparison (Bonferroni Test) of inter-year elevations of regression lines from WYs 1 to 5 utilizing marginal means of ANCOVA for the relationship between discharge and nitrate concentration, Graywood Gully.

WY	1	2	3	4	5
1		0.521	0.000	0.000	0.000
2			0.023	0.000	0.000
3				0.154	0.726
4					0.995
5					

significantly lower ($P<0.05$) compared to WY 1 for NO_3 flux (WYs 2 to 5), NO_3+NO_2 concentration (WYs 2 to 5), TKN flux (WYs 4 and 5), TKN concentration (WY 5), and TSS flux and concentration (WYs 4 and 5) (Table 10). There were no significant changes ($P>0.05$) in concentration and flux of SRP and TP over the study period (Table 10). The significant reduction (71%) in TSS concentration was one of the largest observed of any watershed studied (Table 13). Starting 2 years prior to the study, large portions of the cropland were rotated into a long-term vegetative type crop (alfalfa–grass hay). By WY 1, over 80% of the cropland was in alfalfa (Fig. 6). By WY 5, this land was still high in alfalfa (over 50%).

Sand Point Gully

The marginal mean concentrations of TKN, TP, SRP and TSS, and flux of TP and SRP were not significantly different over WYs 1 to 5 (Table 11). Marginal means of NO_3+NO_2 concentration and flux in WYs 2, 3, 4, however, were significantly lower than in WY 1. Unlike other watersheds where significant or insignificant reductions in concentration were generally mirrored in the results for the flux analysis, this was not true at Sand Point. Here, TSS and TKN flux were significantly lower in WY 1 even though TSS and TKN concentration showed no significant change. Reductions in TSS and TKN concentration were observed, but variability was high, thus reducing significance.

Sutton Point Gully

The post-hoc analysis indicated that the elevation of the marginal means of TP and SRP concentration and flux was not significantly lower ($P>0.05$) from WY 2 to WY 5 compared to WY 1. Marginal means of TSS concentration and flux in WYs 4 and 5, however, were significantly lower ($P<0.05$) than in WYs 1, 2 and 3 (Table 12). Nitrate concentration and flux were significantly lower than in WYs 2 to 5 when compared to WY 1. Also, TKN concentration and flux in WY 5 were significantly lower than in all other study years (Table 12).

Soils and tile drains

Soil analysis from Graywood Gully indicated that major decreases in available NO_3 and SRP were observed in some fields during this study. In particular, soil NO_3 levels decreased by 80% to 93% within 2 years in fields H-7A through H-7E, only by 17% in field H-6 in 2 years, but by 84% in 5 years (Fig. 7a). Other fields (H8–10) did not show such dramatic drops. Small declines in SRP were observed in fields H-7A through H-7E (mean = 52%), no decreases in H-8 to H-10, and major drops in fields H-5 and H-6 (72 to 96%) (Fig. 7b). Soluble reactive phosphorus concentrations of groundwater draining field tiles in managed areas were lower (mean ± S.E.: 5.81 ± 3.4 µg P/L) than those draining unmanaged areas (219 ± 162 µg P/L, Fig. 8).

Discussion

Mitigation of soil and nutrient losses from agricultural land continues to be a concern within watersheds of the United States and indeed worldwide. There are a number of reasons for this concern. First, depletion of agricultural soil is counterproductive to good farming practices and crop productivity. And perhaps more importantly, over fertilization and concomitant nutrient loss to downstream aquatic ecosystems may produce undesirable effects including increased numbers of bacteria, algae, and macrophytes (Somarelli et al., 2007; Makarewicz et al., 2007; D'Aiuto et al., 2006; Jamieson et al., 2003; Inamdar et al., 2002), increased siltation, decreased aesthetics — in general, a deterioration in both surface and groundwater quality downstream (Gallichand et al., 1998) resulting in cultural eutrophication of lakes and streams (McDowell et al., 2004, Carpenter et al., 1998). In large lakes (e.g., Great Lakes), cultural eutrophication like this is often manifested in pollution of coastal zones, river mouths, and embayments (Makarewicz and Howell, 2007; Makarewicz, 2000).

Since the 1970s, environmental management of watersheds, particularly those in agricultural settings, has emerged as a promising tool to deal with the water quality problems noted above (Hawkings and Geering 1989; Staver et al., 1989; Whitelaw and Solbe, 1989). Water pollution from intense row crop production is related to erosion of soils, to nutrient fertilization as an insurance to maintain

Table 7
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the North McMillan Creek watershed, Conesus Lake, New York.

North McMillan Creek	Water Years				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	0.31 ± 0.07	0.18 ± 0.04	0.23 ± 0.05	0.27 ± 0.07	0.20 ± 0.05
Total Kjeldahl nitrogen (µg N/L)	331 ± 61	343 ± 71	367 ± 81	410 ± 93	330 ± 72
Total phosphorus (µg P/L)	12.9 ± 3.7	18.2 ± 5.3	14.9 ± 4.7	27.8 ± 9.1	20.7 ± 6.5
Soluble reactive phosphorus (µg P/L)	4.33 ± 0.75	4.11 ± 0.73	4.88 ± 0.92	4.60 ± 0.90	3.69 ± 0.69
Total suspended solids (mg/L)	2.06 ± 1.05	3.41 ± 1.79	2.78 ± 1.57	8.14 ± 4.86	4.02 ± 2.26
Number of chemistry samples (*n*)	83	82	69	72	70
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	3.21 ± 0.34	2.07 ± 0.21[a]	2.44 ± 0.24[a]	2.74 ± 0.29	2.18 ± 0.23[a]
Total Kjeldahl nitrogen (kg N/d)	3.36 ± 0.27[b]	3.48 ± 0.28[b]	4.29 ± 0.34[b]	3.47 ± 0.29[b]	2.82 ± 0.24[a]
Total phosphorus (kg P/d)	0.12 ± 0.013	0.15 ± 0.017	0.14 ± 0.015	0.16 ± 0.018	0.15 ± 0.018
Soluble reactive phosphorus (kg P/d)	0.048 ± 0.004[b]	0.047 ± 0.004[b]	0.054 ± 0.004[b]	0.043 ± 0.003[b]	0.037 ± 0.003[a]
Total suspended solids (kg/d)	15.00 ± 3.01	26.00 ± 5.31	28.65 ± 5.77	28.62 ± 6.11	21.54 ± 4.56
Number of flux estimates (*n*)	365	365	365	365	365

Values are the marginal means (± S.E.) of the concentration (mg/L or µg/L) or flux (kg/d) calculated from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included. Non-detectable values for chemical analytes and TSS and days with no flow are not included.

[a] The mean difference is significantly lower at the 0.05 level (Post hoc Bonferroni) from WY 1.

[b] The mean difference among WY 1, WY 2, WY 3 and WY 4 is not significantly different ($P>0.05$).

Table 8
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the Graywood Gully watershed, Conesus Lake, New York.

Graywood Gully	Water Year				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	10.02 ± 2.03	7.77 ± 1.39	5.25 ± 0.97[a]	3.71 ± 0.78[a]	4.20 ± 0.90[a]
Total Kjeldahl nitrogen (µg N/L)	1124 ± 233	873 ± 160	706 ± 133[a]	531 ± 114[a]	514 ± 113[a]
Total phosphorus (µg P/L)	352.1 ± 73.9	343.7 ± 53.5	240.0 ± 46.1	219.8 ± 48.1[a]	186.6 ± 41.8[a]
Soluble reactive phosphorus (µg P/L)	137 ± 28	103 ± 18	109 ± 20	97 ± 20	104 ± 22
Total suspended solids (mg/L)	21.9 ± 10.8	30.3 ± 13.1	15.4 ± 6.9	5.9 ± 3.1[a]	7.7 ± 4.0[a]
Number of chemistry samples (n)	74	93	87	67	62
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	9.32 ± 0.86	6.99 ± 0.56[a]	3.91 ± 0.34[a]	2.68 ± 0.22[a]	2.94 ± 0.24[a]
Total Kjeldahl nitrogen (kg N/d)	0.57 ± 0.05	0.48 ± 0.04[a]	0.42 ± 0.03[a]	0.32 ± 0.02[a]	0.29 ± 0.02[a]
Total phosphorus (kg P/d)	0.19 ± 0.015	0.17 ± 0.013	0.13 ± 0.010[a]	0.12 ± 0.009[a]	0.10 ± 0.007[a]
Soluble reactive phosphorus (kg P/d)	0.097 ± 0.009	0.073 ± 0.006	0.076 ± 0.007	0.061 ± 0.005	0.061 ± 0.005
Total suspended solids (kg/d)	6.92 ± 1.30	10.29 ± 1.75	4.79 ± 0.84	2.04 ± 0.34[a]	3.25 ± 0.54[a]
Number of flux estimates (n)	300	343	340	360	356

Values are the marginal means (± S.E.) of the concentration (mg/L or µg/L) or flux (kg/d) calculated from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included. Non-detectable values for analytes and TSS and days with no flow are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post-hoc Bonferroni) from WY 1.

production yields rather than nutrient deficiencies of the soil, to untimely tillage and fertilization practices, to the lack of buffer or filter strips along water passages, etc. In New York, water pollution from dairy farming is often associated with high P levels in soils receiving long-term manure application, while recently applied manure can produce high concentrations of total dissolved P in overland flow (Kleinman et al., 2002; Hively et al., 2005); that is, "Critical Source Areas" do exist (McDowell et al., 2004). Soil, P, and N losses from a watershed are variable and may be largely limited to small areas that affect downstream systems. For example, intense livestock production in a limited area may have several problems (e.g., lack of appropriate manure storage facilities, no treatment of feedlot runoff, release of untreated milk house wastewater, and excessive or untimely field applications of manure) (Cooper and Lipe, 1992; Clausen et al., 1992). "Critical Source Areas" can be recognized and allow the focusing of management efforts. This can be accomplished through a technical evaluation of soil types, of production rates and P and N contents of manure, of farming practices, of hydrological sensitive areas and slope of the land, etc. This was the approach taken in implementing management practices in the Conesus Lake Study.

In general, nutrient export from cropland to downstream systems is several times higher than from grass and forest (McDowell et al., 2004; Beaulac and Reckhow, 1982; Frink, 1991; Jones et al., 2004). For example, overland flow of TDP from a manured barnyard in New York was 11.6 mg/L compared to ~0.1 mg/L from a deciduous forest (Hively et al., 2005). Recent research has shown that within the Conesus Lake watershed, agriculture dominated sub-watersheds lose larger amounts of soil and nutrients than those with a lesser percentage of land in farming (D'Aiuto et al., 2006; Makarewicz et al., 2007; Herendeen and Glazier, 2009; SOCL, 2001). This was confirmed by our study. For example, at North McMillan Creek where 12% of the watershed is in farmland, marginal mean concentrations of all analytes were low compared to those in Graywood Gully where agriculture dominated land use (>70%) (Tables 7 and 8).

In the Graywood Gully watershed where row crops and dairy farming were present, application of a full spectrum of management practices [fertilizer reduction, cover crops, contour strips, reduction in fall and winter manure spreading, various grass filters for runoff from bunker storage of silage and milk house wastes, cows and heifers fenced from the creek and pond (Herendeen and Glazier, 2009; Jacobs, 2006)] resulted in significant reductions in the flux and concentration ranging from 47 to 65% from WY 1 to WY 5 in four out of five analytes monitored (TP, $NO_3 + NO_2$, TKN, TSS); that is, significantly more soil and nutrients were maintained on the watershed (Table 13). The time from BMP implementation to a significant impact varied with the analyte: $NO_3 + NO_2$ in 1 year, TKN in 2 years, TSS and TP in 3 years. For

Table 9
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for Long Point Gully watershed, Conesus Lake, New York.

Long Point Gully	Water Year				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	5.82 ± 1.33	4.73 ± 1.07[a]	2.83 ± 0.60[a]	2.96 ± 0.70[a]	3.35 ± 0.83[a]
Total Kjeldahl nitrogen (µg N/L)	737 ± 115	632 ± 97	527 ± 76[a]	602 ± 97	558 ± 93
Total phosphorus (µg P/L)	97.7 ± 24.1	82.4 ± 20.1	58.5 ± 13.3[a]	50.0 ± 12.8[a]	62.1 ± 16.5[a]
Soluble reactive phosphorus (µg P/L)	47.7 ± 13.8	26.1 ± 7.5[a]	25.8 ± 6.9[a]	20.7 ± 6.2[a]	21.6 ± 6.7[a]
Total suspended solids (mg/L)	4.8 ± 2.3	9.8 ± 4.7	4.3 ± 1.9	3.3 ± 1.7	3.3 ± 1.7
Number of chemistry samples (n)	52	53	62	48	45
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	19.79 ± 2.08	15.50 ± 1.47[a]	8.31 ± 0.76[a]	8.84 ± 0.81[a]	9.81 ± 0.98[a]
Total Kjeldahl nitrogen (kg N/d)	1.95 ± 0.13	1.69 ± 0.10	1.55 ± 0.09[a]	1.80 ± 0.11	1.63 ± 0.10[a]
Total phosphorus (kg P/d)	0.25 ± 0.03	0.20 ± 0.02[a]	0.17 ± 0.01[a]	0.14 ± 0.01[a]	0.16 ± 0.02[a]
Soluble reactive phosphorus (kg P/d)	0.131 ± 0.017	0.077 ± 0.009[a]	0.078 ± 0.009[a]	0.057 ± 0.006[a]	0.058 ± 0.007[a]
Total suspended solids (kg/d)	6.89 ± 1.34	19.71 ± 3.50	12.57 ± 2.17	9.43 ± 1.66	7.67 ± 1.45
Number of flux estimates (n)	246	302	320	310	269

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post hoc Bonferroni) from WY 1.

Table 10
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the Cottonwood Gully watershed, Conesus Lake, New York.

Cottonwood Gully	Water Years				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	3.70 ± 0.63	3.01 ± 0.49	2.16 ± 0.38[a]	2.51 ± 0.44[a]	2.51 ± 0.44[a]
Total Kjeldahl nitrogen (μg N/L)	687 ± 114	699 ± 112	587 ± 101	556 ± 95	469 ± 81[a]
Total phosphorus (μg P/L)	90.2 ± 16.5	104.0 ± 18.5	94.8 ± 18.2	79.7 ± 15.0	79.8 ± 15.3
Soluble reactive phosphorus (μg P/L)	34.5 ± 5.5	25.2 ± 3.9	41.0 ± 6.8	27.7 ± 4.5	35.8 ± 6.0
Total suspended solids (mg/L)	9.5 ± 4.7	12.2 ± 5.9	8.1 ± 4.3	3.0 ± 1.5[a]	2.8 ± 1.5[a]
Number of chemistry samples (n)	79	80	69	71	70
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	4.44 ± 0.34	3.48 ± 0.26[a]	2.25 ± 0.17[a]	2.81 ± 0.21	2.56 ± 0.19
Total Kjeldahl nitrogen (kg N/d)	0.61 ± 0.04	0.55 ± 0.04	0.58 ± 0.04	0.50 ± 0.03[a]	0.45 ± 0.03
Total phosphorus (kg P/d)	0.08 ± 0.005	0.08 ± 0.005	0.09 ± 0.006	0.07 ± 0.004	0.08 ± 0.005
Soluble reactive phosphorus (kg P/d)	0.039 ± 0.002	0.037 ± 0.002	0.046 ± 0.003	0.035 ± 0.002	0.045 ± 0.003
Total suspended solids (kg/d)	4.64 ± 0.92	5.03 ± 0.96	5.72 ± 1.09	1.53 ± 0.29[a]	1.78 ± 0.34[a]
Number of flux estimates (n)	365	366	365	358	365

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post-hoc Bonferroni) from WY 1.

SRP, however, there was no impact even after 4 years. This variability was due, in part, to the differing times at which BMPs were implemented during the study.

Although reductions of 24% in stream SRP concentration (137 ± 28 to 104 ± 22 μg P/L) were observed from WY 1 to WY 5, the difference was not statistically significant (Table 13). Yet soil analysis indicated that available SRP in soils decreased in ~4 years, especially in areas where rates of P fertilization were decreased (Fig. 7b, Herendeen and Glazier, 2009). Similarly, SRP concentrations in groundwater from tiles draining managed Graywood Gully fields had lower concentrations of SRP than in adjacent non-managed fields (Fig. 8). Using a completely different analytical approach, the Thornthwaite–Mather soil moisture status model, Zollweg and Makarewicz (2009) demonstrated a 55% reduction in SRP concentration from events in the Graywood watershed.

The lack of statistical significance in the reduction of SRP is related to at least two factors: unexpected application of manure to fields and external inputs from outside the topographic watershed boundary. Lewis and Makarewicz (2009) demonstrated that in the winter of WY 3, unexpected manure operations caused significantly elevated levels of P in stream water draining Graywood — likely affecting our annual analysis. Similarly, Noll et al. (2009) demonstrated that during storm events, flow high in P from outside of the topographic watershed boundary of Graywood Gully affected our estimates of stream P, perhaps prohibiting a statistical significant decrease in our annual calculation. Work in other small watersheds including this study (see Long Point Gully) has demonstrated that significant reductions in SRP loss are possible from the Conesus Lake watershed. Also, Bishop et al. (2005) reported reductions in TP and SRP fluxes of 43 and 29%, respectively, from dairy farms in the watershed of the Cannonsville Reservoir in New York State, while BMPs were not effective in reducing SRP in three watersheds in Owl Run in Virginia (Brannan et al., 2000).

Dairy cattle were removed from the Long Point Gully watershed in WY 1, and a 37% reduction (76.7 ha) in crop acreage occurred by WY 2. Here major reductions in $NO_3 + NO_2$ (42%), TP (36%), and SRP (53%) concentrations were observed within a year (WY 2) of removal of cropland from production (Table 13, Fig. 5). As expected, removing land from crop production reduced nonpoint nutrient sources and led to major reductions of nutrients from the watershed.

In Cottonwood Gully where row crops predominate, BMPs were limited to two: construction of three water and sediment control

Table 11
Descriptive statistical data collected from 1 Sep to 31 Aug 2007 for the Sand Point Gully watershed, Conesus Lake, New York.

Sand Point Gully	Water Years				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	2.09 ± 0.44	1.53 ± 0.32[a,b]	1.21 ± 0.24[a,b]	1.25 ± 0.28[a,b]	1.18 ± 0.26[a,b]
Total Kjeldahl nitrogen (μg N/L)	718 ± 120	765 ± 126	545 ± 86	640 ± 115	631 ± 114
Total phosphorus (μg P/L)	79.9 ± 19.9	103.1 ± 25.5	75.7 ± 17.8	94.0 ± 25.2	95.3 ± 25.6
Soluble reactive phosphorus (μg P/L)	28.07 ± 5.8	28.0 ± 5.6	33.8 ± 6.5	25.3 ± 5.5	25.7 ± 5.6
Total suspended solids (mg/L)	8.8 ± 4.1	14.7 ± 6.7	6.6 ± 2.9	8.6 ± 4.3	7.9 ± 4.0
Number of chemistry samples (n)	76	77	90	66	66
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	2.03 ± 0.19	1.53 ± 0.13[a]	0.99 ± 0.09[a,c]	1.31 ± 0.11[a,c]	1.15 ± 0.10[a,c]
Total Kjeldahl nitrogen (kg N/d)	0.66 ± 0.04	0.54 ± 0.03[a]	0.45 ± 0.03[a,c]	0.51 ± 0.03[a,c]	0.46 ± 0.03[a,c]
Total phosphorus (kg P/d)	0.07 ± 0.006	0.07 ± 0.006	0.06 ± 0.005	0.06 ± 0.005	0.06 ± 0.005
Soluble reactive phosphorus (kg P/d)	0.026 ± 0.002	0.026 ± 0.002	0.035 ± 0.003	0.023 ± 0.002	0.024 ± 0.002
Total suspended solids (kg/d)	6.63 ± 1.19	6.39 ± 1.06	3.11 ± 0.54[a,c]	3.58 ± 0.60[a,c]	2.64 ± 0.44[a,c]
Number of flux estimates (n)	328	365	365	365	365

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post hoc Bonferroni) from WY 1.
[b] The mean difference among years WY 2, WY 3, WY 4 and WY 5 is not significantly different ($P > 0.05$).
[c] The mean difference among years WY 3, WY 4 and WY 5 is not significantly different ($P > 0.05$).

Table 12
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the Sutton Point Gully watershed, Conesus Lake, New York.

Sutton Point Gully	Water Year				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	2.8 ± 0.5	1.9 ± 0.3[a]	1.3 ± 0.2[a]	1.3 ± 0.2[a]	1.7 ± 0.3[a]
Total Kjeldahl nitrogen (μg N/L)	428 ± 84	371 ± 61	381 ± 64	332 ± 59	286 ± 52[a]
Total phosphorus (μg P/L)	39.2 ± 7.6	40.2 ± 6.6	45.5 ± 7.5	38.4 ± 6.8	39.7 ± 7.1
Soluble reactive phosphorus (μg P/L)	20.2 ± 3.7	21.0 ± 3.2	23.8 ± 3.7	21.5 ± 3.6	22.6 ± 3.8
Total suspended solids (mg/L)	5.0 ± 2.3	3.0 ± 1.2	2.9 ± 1.2	1.8 ± 0.8[a]	1.4 ± 0.6[a]
Number of chemistry samples (*n*)	50	63	62	54	53
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	1.43 ± 0.12	1.12 ± 0.08[a]	0.70 ± 0.05[a]	0.75 ± 0.05[a]	0.97 ± 0.07[a]
Total Kjeldahl nitrogen (kg N/d)	0.20 ± 0.02	0.20 ± 0.01	0.20 ± 0.01	0.18 ± 0.01	0.15 ± 0.01[a]
Total phosphorus (kg P/d)	0.018 ± 0.001	0.022 ± 0.001	0.025 ± 0.002	0.021 ± 0.001	0.021 + 0.001
Soluble reactive phosphorus (kg P/d)	0.011 ± 0.001	0.012 ± 0.001	0.014 ± 0.0001	0.012 ± 0.001	0.013 ± 0.001
Total suspended solids (kg/d)	1.75 ± 0.20	1.45 ± 0.21	1.70 ± 0.25	0.94 ± 0.14[a]	0.61 ± 0.09[a]
Number of flux estimates (*n*)	299	366	337	357	332

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post-hoc Bonferroni) from WY 1.

basins (gully plugs) and strip cropping designed to retain soils. Previous to BMP introduction in this small watershed (98.8 ha), soil loss was high and conservatively estimated in the 1990s at 130 tons (metric) per year (Makarewicz et al., 2001). As in Graywood Gully, significant impacts from management practices were observed in the second year (WY 3) after introduction of BMPs (WY 2). Unlike Graywood Gully, retention of soil and nutrients was recorded for only three of five analytes (TKN, TSS and NO_3+NO_2). With the exception of TSS (71% reduction), the magnitude of reduction was low relative to Graywood Gully [e.g., NO_3+NO_2 concentration: 32% (Cottonwood) versus 58% (Graywood)]. With regard to TKN, concentration reduction was not significant until WY 5 (4 years post BMP) when a 28% reduction of acreage in crop production occurred. While construction of gully plugs undoubtedly reduced soil loss, conversion of portions (up to 80% the watershed) to a long-term vegetative type crop (alfalfa–grass hay) likely contributed to a reduction in stream NO_3+NO_2 levels. Lysimeter studies indicated that NO_3+NO_2 concentrations were much less in groundwater from fields cropped to a forage legume, such as alfalfa, than from fields with corn (Owens 1990). Also, N fertilizers are not generally used for alfalfa. Therefore, the reduction in N fertilizer use and establishment of alfalfa likely contributed to the reduction in stream NO_3+NO_2 observed.

Sheffield et al. (1997) and Owens et al. (1996) observed major reductions in total fractions (TSS: 90%, total N: 54%, TP: 81%) but not in dissolved fractions when water troughs and cattle were fenced out of streams. At Sand Point Gully rotational grazing pens and water troughs were installed, and cattle were fenced out of the creek starting in May of WY 2. Two gully plugs and tiles were also installed in a small portion of the watershed in November 2002 prior to the beginning of this project. We did not expect a large impact of management practices here, especially since the major management area, rotational grazing and the “gully plugs” accounted for less than 9.5% of the entire watershed. Also, manure-spreading operations continued in large portions of the watershed throughout the study (P. Kanouse, Personal Communication, Livingston County Soil and Water Conservation District), which theoretically could cause elevated levels of NO_3+NO_2 and TP. In spite of these expectations, a significant 44% reduction in NO_3+NO_2 concentration was observed (Table 13) by WY 3 with no further significant changes over the study period. A reduction in other analytes was not observed.

The reduction in Sand Point Gully water concentrations of NO_3+NO_2 was unexpected because no structural management practices were implemented that would account for the decrease in NO_3+NO_2. If anything, the continued manure spreading would suggest the potential for elevated levels of NO_3+NO_2. Since only NO_3+NO_2 decreased and not TP, SRP, TKN or TSS, cropland rotation was considered a possible cause. Fallow land, wheat, and the alfalfa grass mix were converted to soybean production acreage starting in WY 1 at Sand Point Gully. Production of soybeans increased from zero, 1 year prior to this study beginning, to 33.0 ha in WY 2 to 83.7 ha by WY 5, almost 45% of the watershed. However, soybeans are not suited to high uptake of NO_3 (K. Czymmek, Personal Communication, Cornell University), and a fairly strong negative correlation existed between acres of soybeans planted and NO_3+NO_2 ($r^2=0.55$). Thus soybean uptake of NO_3 should be fairly low and not the likely cause of the reduction observed in stream water.

However, the plow down of sod (spring of WY 1) would be expected to release a significant amount of NO_3. Estimates of 224 to 336 kg of N/ha per acre are typically lost from sod plow down in New York State (K. Czymmek, Personal Communication, Cornell University). Also, wheat planted in fall of WY 1 would have been fertilized in spring of WY 1, and this fertilizer could be a source of NO_3 to groundwater resulting in the elevated levels observed in WY 1. What may have happened is that WY 1 stream NO_3+NO_2 concentrations were elevated due to a practice that occurred only in the first year of our study. After this ended, a slow but steady decrease in NO_3+NO_2

Table 13
Percent change in marginal mean concentration from Water Years 1 to 5.

Watershed	Management practice implemented	NO_3	TKN	TP	SRP	TSS
North McMillan	No management practices	−35	0	+60	−15	+95
Sutton Point	60% Conversion of cropland to alfalfa	−39*	−33*	+1	+12	−72*
Graywood	Whole farm planning (myriad of BMPs)	−58*	−54*	−47*	−24	−65*
Cottonwood	Gully plugs, 28% reduction in croplands	−32[a]*	−32*	−12	+4	−71*
Sand Point	9.5% Converted to rotational grazing, Fenced cattle from streams	−44*	−12	+19	−8	−10
Long Point	37% Reduction in croplands, removal of cows	−42*	−24*	−36*	−53*	−31

A star (*) indicates a significant decrease.
[a] Significant decrease from Water Years 2 to 5.

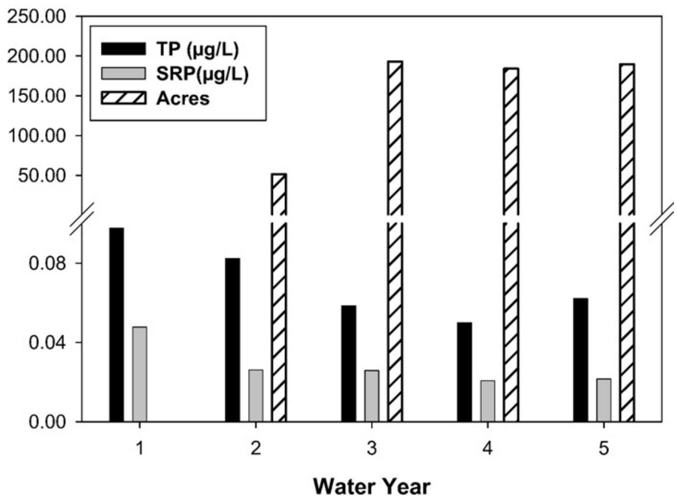

Fig. 5. Time trend in soluble reactive phosphorus (SRP), total phosphorus (TP) and acres of land taken out of crop production in the Long Point Gully watershed. 1 acre = 0.405 ha.

occurred with time. Since no other analyte (TP, SRP, TKN, and TSS) was affected, the response of $NO_3 + NO_2$ was likely due to crop land rotation — a cultural management practice. Typically, when fallow or alfalfa fields are plowed and moved into production, corn is the recommended rotation crop to utilize NO_3 lost to groundwater as a result of plowing.

Significant reductions in TKN and TSS flux, but not concentration, occurred by the second year and third year of the study in the Sand Point Gully watershed. Since there was general agreement in statistical significance between data that considered changes in stream concentration and stream flux in the other five watersheds examined, we were surprised at this difference. Other researchers have discussed such inconsistencies and have noted that expectations of results can be confounded by catastrophic loading events (Meals 2001), unexpected and unknown changes in cropping practices (Boesch et al., 2001), and natural interannual variability (Longabucco and Rafferty, 1998). Any of these factors could explain the differences observed.

Significant reductions in $NO_3 + NO_2$ (39%), TSS (72%), and TKN (33%) occurred at Sutton Point (Table 13) within 1, 3, and 4 years, respectively, after WY 1. No physical infrastructure improvements were implemented in this watershed until WY 5 when gully plugs were added. However, a significant and increasing portion of the watershed has been in alfalfa/grass production since WY 1 (37% in WY 2 to 60.3% in WY 5). As in Cottonwood Gully, the conversion of portions of this watershed to a long-term vegetative type crop (alfalfa–grass hay), a cultural BMP, would indicate that no N fertilizer was added to these fields (N. Herendeen, Personal Communication, Cornell Cooperative Extension). Also during this period, manure slurry was not added to fields (P. Kanouse, Personal Communication,

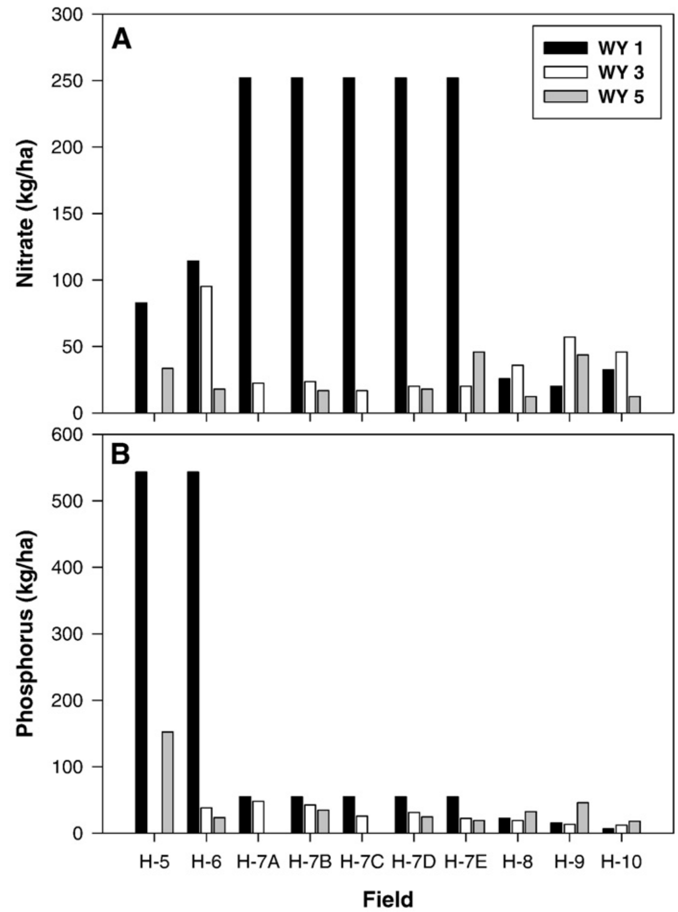

Fig. 7. (a) Nitrate and (b) phosphorus in soils from Graywood Gully. Field names refer to locations within the Graywood Gully watershed in Fig. 2 of Herendeen and Glazier (2009).

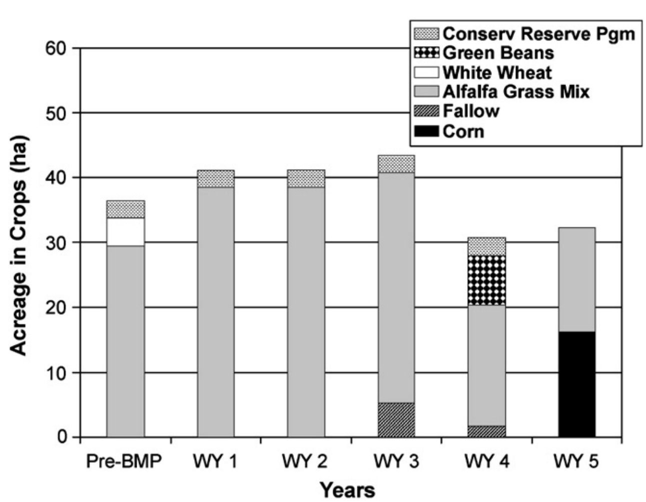

Fig. 6. Acreage of various crops in the Cottonwood Gully watershed over time.

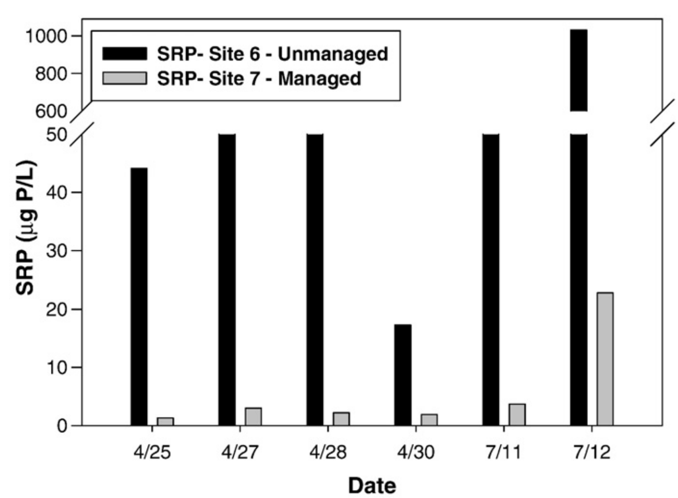

Fig. 8. Concentration of soluble reactive phosphorus in water draining from tiles at Sites 6 and 7 in and near the Graywood Gully watershed (see Fig. 2 of Herendeen and Glazier 2009).

Table 14
Annual average nonevent and event (in parentheses) concentrations of nitrate and total suspended solids.

	North McMillan	Graywood Gully	Cottonwood Gully	Long Point	Sutton Point	Sand Point
Nitrate (mg/L)						
WY 1	0.38 (0.62)	15.53 (11.34)	4.43 (5.26)	8.01 (6.75)	2.83 (3.98)	3.02 (4.56)
WY 2	0.26 (0.29)	11.83 (9.61)	4.71 (3.68)	5.19 (6.02)	2.31 (2.05)	1.87 (1.96)
WY 3	0.27 (0.40)	8.41 (6.47)	3.34 (3.32)	4.28 (4.46)	1.68 (1.99)	1.45 (3.31)
WY 4	0.25 (0.43)	4.53 (4.92)	3.05 (2.68)	3.47 (3.68)	1.47 (1.96)	1.37 (3.02)
WY 5	0.26 (0.30)	4.72 (5.60)	3.17 (3.91)	3.72 (4.50)	1.87 (2.04)	1.31 (1.33)
Total suspended solids (mg/L)						
WY 1	2.9 (267)	16.7 (282)	9.7 (183)	2.9 (54)	7.4 (154)	6.7 (247)
WY 2	11.5 (268)	11.6 (1,110)	7.8 (297)	11.5 (49)	4.4 (49)	6.5 (369)
WY 3	7.9 (113)	6.7 (499)	5.7 (257)	7.9 (26)	4.1 (26)	5.1 (154)
WY 4	3.8 (123)	3.4 (289)	1.5 (125)	3.8 (23)	2.0 (23)	7.1 (161)
WY 5	2.6 (265)	5.3 (318)	2.0 (200)	2.6 (24)	2.7 (25)	2.1 (511)

WY = Water Year.

Livingston County Soil and Water District). Both practices, reduction in manure spreading and the establishment of increasing acreage of a vegetative crop, likely led to the observed decrease in NO_3+NO_2 and TKN to the downstream system.

Widely variable lengths in response times to BMPs are reported in the literature. Generally, smaller watersheds show water quality improvements in less time than larger ones (Gallichand et al., 1998). In the St. Albans watershed (1384 ha), Vermont, USA, no significant change in concentration or flux was observed in tributary streams 10 years after implementation of BMPs (Clausen et al. 1992). Coffey et al. (1992) suggested response times of 6 to 15 years depending on the size of the catchment, but quicker responses to BMPs have been reported. In the Belair River watershed of Quebec Province, Canada (529.4 ha), for example, where river pollution was due to intense livestock production, reductions in TP and SRP were observed 2 years post BMPs (Gallichand et al., 1998). Within the Conesus Lake watershed, significant changes were generally observed within 1 to 2 years after BMP implementation and were still observable 3 to 4 years later. Since data were evaluated on an annual basis, responses in less than 1 year were not observed. However, one exception was the Graywood Gully watershed where positive effects were evident a few weeks after cessation of certain manure practices (Lewis and Makarewicz, 2009).

Within the Conesus Lake study, both event and nonevent decreases were generally observed after BMPs were introduced. For example, both event and nonevent NO_3+NO_2 concentrations decreased as a result of management practices in Cottonwood and Graywood Gullies (Figs. 2a and 4b). In event and nonevents, dissolved fractions such as NO_3+NO_2 and SRP decreased in Cottonwood, Graywood, Sutton, Sand, and Long Point Gullies. At best, a weak trend was observed for total fractions. For example, during nonevents TSS decreased in both Long Point and Cottonwood Gullies while no trend was observed during events. This was somewhat surprising especially at Cottonwood Gully and suggested that gully plugs were more effective during small rain events than large ones. A similar result was observed at Garfoot and Brewery Creeks in Wisconsin (Graczyk et al. 2003). Here, after providing stream bank protection and fencing cattle from streams, a significant decrease in suspended solids occurred during base-flows (i.e., nonevents) but not during events. In the Belair River watershed, Gallichand et al. (1998) observed a 10-fold decrease in TP concentration during peak flow snowmelt runoff in March and April but no significant changes in median annual concentrations after implementation of BMPs.

As noted above, interpretation of field results may be confounded by catastrophic loading events. Nevertheless, by using a small watershed approach, experimental signal to noise (other stresses on the watershed) ratio is likely reduced (Makarewicz, 2009), thereby facilitating evaluation of BMPs on downstream water quality. In general, where implementation impacted downstream water quality, significant reductions in TP, SRP, NO_3+NO_2, TKN and TSS concentration and flux occurred by the second year and third year of implementation. The implementation of structural (e.g., gully plugs) and cultural (e.g., modification of manure practices) BMPs had significant effects in preventing soil loss. In Graywood Gully, where Whole Farm Planning was practiced and a myriad of structural and cultural BMPs were introduced on a dairy farm, the greatest percent reduction (average = 55.8%, range 47% to 65%) and the largest number of significant reductions in analytes (4 out of 5) were observed. At Long Point Gully where 37% of the cropland was taken out of production and dairy cows were removed, concentration reductions were significant, averaging 39% in 4 out of 5 analytes. One of the largest impacts was a 71% reduction in soil loss to downstream systems that resulted from the implementation of both a structural (installation of three gully plugs) and a cultural [long-term vegetative type (crop, alfalfa–grass hay) as associated buffers and strip crops] management practice in the Cottonwood watershed. Removal of nutrient sources also had a major impact on losses from these watersheds.

The importance of proper rotation of crops, a cultural BMP, is also evident. Maintenance of cover crops, such as alfalfa grass mix, can also lead to major reductions in nutrient loss from these managed systems. At Sutton Point no physical infrastructure improvements were implemented. However, the progressive conversion of large portions of cropland in this watershed to a long-term vegetative type crop (60.3% as alfalfa–grass hay by WY 5) was related to major reductions in TSS (72%), TKN (33%), and NO_3+NO_2 (39%) concentrations. Similarly, the proper choice of a production crop after long-term maintenance of a vegetative-type plant is important. For example,

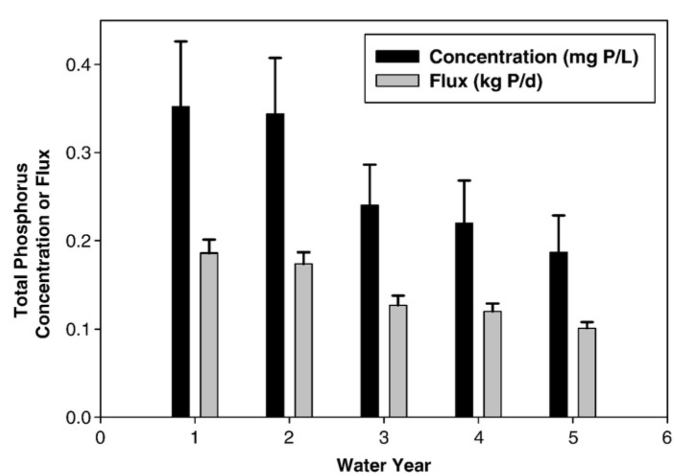

Fig. 9. Mean annual concentration and flux of total phosphorus in Graywood Gully. A Water Year (WY) is defined as the period from 1 Sep to 31 Aug of the following year. Values are the marginal mean ± S.E.

after plow down of sod/alfalfa in Sand Point Gully, a crop such as corn, which readily takes up NO_3, would have been a better choice to rotate over alfalfa and sod and reduce losses to downstream systems. Without this rotation, NO_3+NO_2 levels appeared to be elevated in the first year of this study at Sand Point Gully. Where rotational grazing pens and fencing out of cattle from streams were implemented (Sand Point Gully), no significant effect on downstream systems occurred, as only a small amount of acreage was affected by the BMP.

Both structural and cultural BMPs were observed to have profound effects on nutrient and soil loss. Where fields were left fallow or planted in a vegetative type crop (alfalfa), reductions especially in NO_3+NO_2 were observed. Where structural implementation occurred, reductions in total fractions, most likely particulate fractions, were particularly evident. Where both were applied, major reductions in nutrients and soil occurred. Taking significant portions of the watersheds out of crop production or by removing dairy cows had a similar effect; nutrients and soil were maintained on the watershed and significant reductions in nutrient and soil loads and concentrations to downstream systems were evident. In fact, significant decreases of TP and SRP concentrations occurred only in the two watersheds where considerable effort went into managing manure (Graywood Gully) and where dairy cows were removed (Long Point Gully) (Table 13).

Although significant reductions in nutrient and soil loss to streams were observed in the managed watersheds, the question does arise as to how much more soil and nutrients can be maintained on the watershed and not be lost to the downstream system. In order to evaluate the effectiveness of BMP implementation, reductions in nutrient and soil loss in managed sites in the watershed were compared to the relatively 'pristine' North McMillan Creek watershed. This watershed was deemed pristine because with only 12% of the land in agriculture, nutrients and soil concentrations in stream water draining the watershed were low (Table 14). After 5 years of management, nonevent and event concentrations of TSS in WY 5 in streams draining watersheds dominated by agriculture were not significantly different (ANOVA, $df=6{,}100$; $P>0.05$) from the "pristine" North McMillan Creek (Table 14). This result suggests that management for soil loss can be very effective in a relatively short period of time and that the reductions are comparable to our "pristine/reference" watershed of North McMillan Creek.

This was not the case for nutrients. Although significant reductions in nutrient levels in managed agricultural systems were noted 5 years post BMP implementation, event and nonevent nutrient concentrations in agricultural watersheds were still significantly different (ANOVA, $df=1{,}6{,}100$; $P<0.05$) than those in North McMillan Creek (Table 14). Post hoc analysis (Bonferroni) indicated that during events all watersheds, except Sand Point Gully, were significantly higher in NO_3+NO_2 than North McMillan. Hysteresis is the concept that there is a lag effect between an action and an effect. We are likely observing such an effect with nutrients. Annual marginal mean concentrations in streams still appear to be decreasing 5 years after implementation of management practices (e.g., Fig. 9). Just how long it could take before a new equilibrium would be reached between groundwater, soil, and stream chemistry is not known or suggested by our data. Clearly, it would take longer than 5 years.

Interestingly, reductions in nutrients and soil delivered to downstream systems have had an effect on metaphyton, macrophytes, and microbial communities in the nearshore area of Conesus Lake. Comparisons of Pre-BMP (2–3 years) to the Post-BMP (4 years) periods at Cottonwood Gully, Graywood Gully, and Sand Point Gully (sites receiving the most extensive BMPs) revealed that algal cover was statistically lower than baseline in 8 of 11 sample years (72.7%) (Bosch et al., 2009a). In sites downstream from sub-watersheds that were not extensively managed, percent cover of filamentous algae was lower than Pre-BMP levels in only 3 of 12 sample years (25%). Where major reductions in percent cover occurred, strong positive relationships existed with summer flux of NO_3+NO_2 and SRP to the nearshore of Conesus Lake; that is, reductions in N and P resulted in reductions of metaphyton populations.

Similarly, in macrophyte beds downstream from managed sub-watersheds, quadrat biomass decreased by 30–50% within 1 or 2 years of BMPs implementation and was statistically lower than Pre-BMP values in 7 of 11 sample years (Bosch et al., 2009b). In the three macrophyte beds where minimal or no BMPs were introduced, biomass was statistically indistinguishable from Pre-BMP values in 12 experimental sample years. Lastly, microbial populations declined in the nearshore below managed watersheds. For example, over the 5-year study period, a major decrease in bacterial levels in nonevent Graywood Gully stream water was evident after management practices were implemented. *Escherichia coli* levels dropped more than 10 fold to levels significantly below the 235 cfu/100 mL EPA bathing beach standard while the yearly maximum for *Enterococcus* dropped by a factor 2.5 (Simon and Makarewicz, 2009a,b).

Clearly, BMPs implemented in the Conesus Lake watersheds have had major impacts within the nearshore of Conesus Lake. The utility and effectiveness of the implemented BMPs should allow regional policy makers and managers to develop strategies for improving watershed land usage while improving downstream water quality in the embayments, nearshore, and open waters of large lakes. But, as Moran and Woods (2009) suggest, effective watershed management requires far more than a narrow focus on water quality. The importance of a strong science foundation coupled with an ongoing effort to build political consensus cannot be overstated. It is essential that watershed residents be engaged in a dialog where they can communicate to politicians how to restore and protect the resources they value.

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