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A multivariate analysis of covariance to determine the effects of near-stream best management practices on nitrogen and phosphorus concentrations on a dairy farm in the New York Conservation Effects Assessment Project watershed

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Abstract: Near-stream best management practices (BMPs), such as exclusionary fencing or cattle crossings, are often recommended to improve water quality, but quantification of their impacts is limited. Surface and subsurface processes in these near-stream areas impact the contribution of nitrogen and phosphorus to watercourses and the effectiveness of near-stream BMPs. To test the impact of the near-stream BMPs, groundwater samples from 30 piezometers and streamwater samples along two adjacent creeks (one control and one with BMP treatment) were collected over a three-year period before and after installation of exclusionary fencing with a cattle crossing in the Catskill Mountains of New York State. Samples were analyzed for nitrate nitrogen (NO_3^--N), soluble reactive phosphorus (SRP), dissolved oxygen, and dissolved organic carbon. Analysis results and other ancillary variables (rainfall and groundwater table depth) were incorporated into a multivariate statistical model to evaluate the impact of the BMPs on the concentrations of NO_3^--N and SRP in the treatment creek. Results of the analysis indicate that the installation of the near-stream BMPs resulted in a 27% yearly (34% during the growing season) reduction in treatment creek SRP concentrations, while there was little impact on NO_3^--N concentrations. Incorporating the SRP concentrations measured in a nearby control creek and controlling for the effects of groundwater SRP levels and groundwater hydrology (water table height) had a significant effect on the overall analytical model performance. These results indicate that protecting near-stream areas from potentially pollution-causing practices can be an important means of controlling phosphorus levels in water bodies.

Key words: analysis of covariance—best management practices—catchment management—Catskill Mountains—land-use management—nitrogen—phosphorous—water quality

Agriculture in the United States is responsible for 47% of total phosphorus (P) and 52% of total nitrogen (N) discharged into US streams (Allan 1995). As a result, agricultural producers face pressure to reduce or more efficiently manage nutrients, particularly animal wastes, such as manure, to minimize loss of contaminants such as P and N. This is particularly important in the New York City (NYC) source watersheds in the Catskill Mountains, where local economic development can be curtailed when in-reservoir P levels are above the New York State Department of Environmental Conservation standard of 20 μg L^{-1} (NYSDEC 1993). To reduce P levels in the NYC source watersheds, both point and nonpoint source controls have been installed beginning in the mid-1990s, and although in-stream/reservoir P levels have been consistently declining, dissolved P concentration in runoff from intensively managed pasture and hayfields in the upper reaches of the watersheds are 10 to 30 times the New York State Department of Environmental Conservation standard (Hively et al. 2005). Nitrate nitrogen (NO_3^--N) concentration in the NYC source watershed is, in general, three to four times below the standard for drinking water of 10 mg L^{-1} (USEPA 2003) but may pose a threat to water quality at much lower levels. Indeed, Effler and Bader (1998) documented NO_3^--N additions to the Cannonsville Reservoir well below USEPA standards as problematic, and the reservoir has shown a tendency towards N limitation on algal production during midsummer.

Although the overall concentrations of N and P generally decrease after installation of best management practices (BMPs) (Bishop et al. 2005; Brannan et al. 2000; Lee et al. 2000; Inamdar et al. 2001; Gitau et al. 2004), it is often unclear which BMPs are most effective, and there is a large range in efficiency of individual BMPs on overall water quality (Gitau et al. 2005). For instance, in the NYC source watersheds, milkhouse buffer strips were only effective in reducing P levels for a five- to ten-year period following installation (Kim et al. 2006). Exclusionary fencing reduced manure additions and hence the load of P deposited in the streams by cows (James et al. 2007), but effectiveness depends on where and how it is installed. Precision feeding resulted in less P excreted in manure (Maguire et al. 2005; Rotz et al. 2005; Toor et al. 2005), but it is unclear how this impacts water quality in the short term. Barnyard improvements can reduce runoff losses and are beneficial for animal health but often fail to actually reduce P levels in runoff (Robillard and Walter 1984; Bishop et al. 2005).

One BMP that seems to be successful in reducing the N and P loads to streams and has been installed more than any other practice in the NYC source watershed is exclusionary fencing with cattle crossings (Line et al. 2000; Meals 2000; James et al. 2007). Bishop et al. (2005) speculated that the improvement in water quality on a dairy farm was partially due to stream crossings and exclusionary fencing that prevented direct cow access to the stream. Line et al. (2000) documented reductions of 76% in total P in a stream after

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fencing was installed in North Carolina. Exclusionary fencing has been tested to a limited extent in the northeast United States, yet there are few conclusive reports in the literature relating to the effectiveness of these BMPs on water quality. Bishop et al. (2005) via statistical analysis and Easton et al. (2008) with a distributed model documented significant dissolved P reductions resulting from a suite of BMP installations on a farm in the Catskill Mountains of New York State. Yet determining which specific BMPs were most effective still remains a difficult task and one that cannot reasonably be accomplished when suites of BMPs are installed in a short time frame.

The objective of this research is to quantify the streamwater quality impact of BMPs that exclude livestock from streams. Although most studies simply look at the downstream effect before and after installation of BMPs, we examine in detail the cause of the reduction. Two streams are compared, one that had exclusionary fencing with a cattle crossing installed (treatment) and one stream that did not (control). Because there is evidence that dissolved oxygen (DO) and dissolved organic carbon (DOC) influence dissolved P and NO_3^--N levels in water, we measure DOC and DO in all samples collected and include them in the statistical analysis. We also incorporate the effect of groundwater NO_3^--N, dissolved P, DO, and DOC levels on stream flow concentrations because the site is influenced by groundwater. An analysis of covariance (ANCOVA) model was employed to determine the relative impact of the BMPs on soluble reactive phosphorus (SRP) and NO_3^--N concentrations using data from a control (Creek B) and a treatment (Creek A) creek during the pre- and post-BMP periods. Bishop et al. (2005) state that variation in a response variable (Creek A SRP levels, for instance) that is not explained by the covariates can be attributed to treatment effects (in this case BMPs) as represented by an indicator variable or to unexplained error. Thus, the ANCOVA model serves as a simple methodology that provides a straightforward means of evaluating the impact of related environmental and management factors on BMP effectiveness. The study was conducted at a valley farm in the Catskill Mountains in central New York State from October 2003 to April 2006.

Materials and Methods

Study Site. The study site is located the Cannonsville Reservoir Conservation Effects Assessment Project watershed in the Catskill Mountains in central New York State on one of the many dairy farms. The farm is located along a lowland tributary of the West Branch of the Delaware River. The dairy farm has 19 ha (46.9 ac) of valley bottom land and 119 ha (294 ac) of uphill lands. Small creeks originate from springs, either on the hillslopes surrounding the farm or on the farm itself. The creeks flow from north to south (figure 1). Springs on the hillslopes are formed at several locations where the steep hillslopes flatten and are active for the majority of the year except at times in the summer. The water from the spring, sampling site A1, in the northwest area of the site forms a creek (Creek A in figure 1) and then flows though the farm. Creek A is gaining water from groundwater flow during most of the year but looses water for a period during the summer and fall and may become dry during extended rainless periods. In the southern area of the farm, regional groundwater intersects the surface and forms saturated areas around Creek B (figure 1), especially during the period when precipitation exceeds evapotranspiration (October to May). Creek B is solely fed by groundwater.

Rainfall data were obtained from Northeast Regional Climate Center in Walton, New York, approximately 9 km (5.6 mi) to the southeast of the field site. The annual average precipitation for the study site is 1,120 mm y^{-1} (44.1 in yr^{-1}), approximately one-third of which falls as winter snow (December to April) (National Climatic Data Center 2000). The climate of the Cannonsville Reservoir watershed is humid continental with an average temperature of about 8°C (46°F), and the growing season extends from May to October.

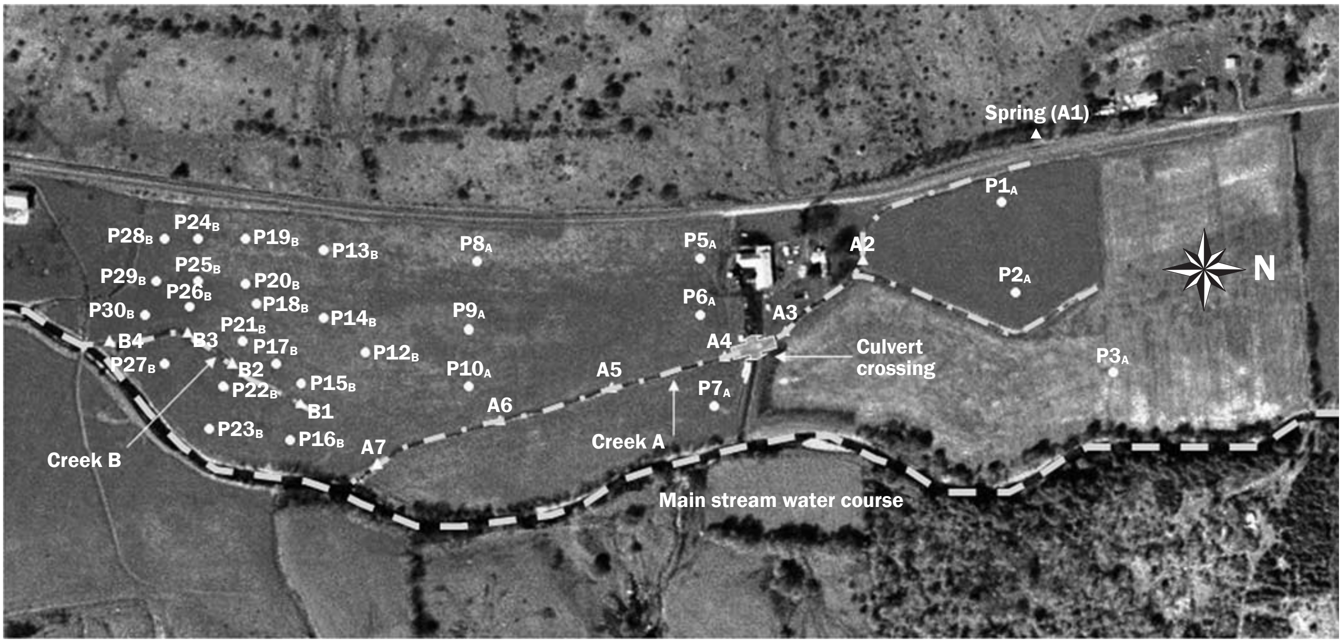
In 2004, the farm had 60 adult dairy and beef cows and 36 heifers, producing 833 Mg (ton) (1,836,450 lb) of manure per year that was spread mainly on the valley bottom lands during the winter, when 80% of the manure was applied (and mainly on the corn land). During the remainder of the year, the herd is pastured. Thus, less manure is produced in the barnyard, but more is deposited in the pasture. Since September 1995, the farm has participated in the Watershed Agricultural Program. This Program is an organization that is managed by the Watershed Agricultural Council and is partly funded by the New York City Department of Environmental Protection and aids farmers with whole farms plans, including BMP implementation. As part of this program, a 5 m (16 ft) long culvert crossing was installed in Creek A during the third week of September 2005 so that the cattle could cross the stream without directly entering it. Exclusionary fencing was installed to delimit the 5 m (16 ft) width of the cattle path from the barnyard to the pasture and to prevent livestock free access to the stream. Before the culvert crossing was installed on Creek A, cattle and farm machinery entered the stream to cross. Creek B has no exclusionary fencing or culvert crossing, allowing direct livestock access and thus serves as a control with which the fenced Creek A may be compared.

Water Sampling. Streamwater was sampled at 11 locations along the course of Creek A and Creek B (figure 1) (Flores-López et al. Forthcoming). The 11 stream sampling locations were divided up as follows: one at the spring site, six along Creek A, and four along Creek B (figure 1). A total of 655 stream flow samples were collected during the study for analysis. An effort was made to sample during a range of events (i.e., base and stormflow) to capture realistically the expected variability in conditions. Following BMP installation in September, cattle were pastured through November 2005, during which 43 creek water samples were collected (i.e., 43 samples collected from September to November 2005). During the nongrowing (December 2005 to March 2006) season following BMP installation, there were 70 creek samples collected.

The spring sampling site, A1, was located on the northwest hillslope (figure 1). Sampling site A2 was located 230 m (755 ft) downstream from A1 on Creek A (figure 1). Sampling sites A3 and A4 were located 330 and 350 m (1,083 and 1,148 ft) downstream from sampling site A1, and between them, the cattle path crosses Creek A (figure 1). Streamwater samples were collected from a sampling site directly upstream of the cattle-crossing path (A3) and one directly downstream (A4). Sampling sites A5 and A6 were located 420 and 540 m (1,378 and 1,772 ft) downstream from A1, and sampling point A7 was 710 m (2,329 ft) downstream from A1 and directly before the confluence with the main stream course (figure 1).

Figure 1

Study site showing the location of Creek A and Creek B, the culvert crossing best management practice, and the thirty piezometers. The groundwater table depth, measured with capacitance probes at each piezometer site was averaged for each creek (the direction of the steady state groundwater flow was derived from the apparent equipotential lines).



Sites B1 to B4 were located in Creek B, which drains groundwater in a low-lying area of the downstream field (figure 1). Sampling points B2, B3, and B4 were located along Creek B at 65, 130, and 200 m (213, 427, and 656 ft) downstream, respectively, from B1. Sampling site B4 was located directly upstream from the confluence with the main watercourse (figure 1).

Thirty subsurface piezometers were installed in the field site at different depths (0.3 to 1.5 m) (1 to 5 ft) (figure 1) and were used to extract groundwater samples to measure soluble reactive P, NO_3^--N, DOC, and DO (Flores-López et al. Forthcoming). A total of 717 groundwater samples were collected for analysis. The piezometers and streams were always sampled at the same time. The groundwater samples were compared against the stream measurements during the analysis. The groundwater table depth measured with capacitance probes at each piezometer site was averaged for each Creek. To determine the direction of the steady state groundwater flow, we used the apparent equipotential lines from the water table levels derived from the capacitance probe data. We present some of the groundwater data in table 1 and discuss it in the manuscript. Greater detail of groundwater sampling and analysis at the site can be found in Flores-López et al. (Forthcoming).

Chemical Water Analysis. Water samples were collected at least two times a month, although occasionally were collected more frequently from October 2003 through April 2006. A volume of 100 mL (6.1 in^3) of water was collected in precleaned plastic bottles. Precleaning entailed rinsing thoroughly with distilled water. Water samples were collected with no headspace and were stored in coolers to prevent temperature increases during transport to the laboratory. Water chemistry analysis is discussed in detail in Flores-López et al. (Forthcoming), while this paper will briefly outline the procedures.

Dissolved oxygen concentration was measured directly in each sample bottle using a Traceable Digital Oxygen Meter (Fisher Scientific) (electrode type) within three to four hours of sampling. The instrument has a resolution of 0.1 mg L^{-1} and is calibrated by exposing the sensor to the ambient atmosphere and correcting to the oxygen content (approximately 21%). The probe was inserted into each bottle to obtain the DO concentration reading while avoiding interaction with ambient air. Before each use, the probe was rinsed with distilled water. Dissolved oxygen concentration readings were started in January 2005. Testing showed that there was no significant difference between DO measurements taken in the field and the laboratory. After measuring the DO concentration, samples were filtered through 0.45 µm (0.0000177 in) membrane filters using a vacuum pump filtering system. Filters were washed with 5 mL (0.305 in^3) of distilled water before filtering.

Water samples were analyzed for SRP within 24 hours of sampling or were filtered and stored in a refrigerator until they could be analyzed. The filtered samples were analyzed using the OI Analytical FlowSystem 3000 Automated Ascorbic Acid Method for SRP, with a detection limit of 0.001 mg L^{-1} following the instructions in the in-house manual for the operation (Method 4500-P G (ortho-P) and Method 4500-Ph (total P) (APHA, AWWA, and WEF 1999). SRP analyses began with the first sample set on October 1, 2003.

Water samples were analyzed for NO_3^--N beginning in March of 2004. The spectrophotometric method and a Spectronic 501 instrument by Bausch & Lomb were used following the instructions in the in-house manual for the operation (Cataldo et al. 1974). Nitrate nitrogen concentrations were below the detection limits (0.05 mg L^{-1}) during the winter.

Dissolved organic carbon analyses were conducted using the OI Analytical Model 1010 Total Organic Carbon Analyzer with

Table 1
Descriptive statistics by season for soluble reactive phosphorus, nitrate-nitrogen, dissolved organic carbon, and dissolved oxygen concentrations and groundwater table depth for the growing season (April to November) and nongrowing season (December to March).

Season	Statistics	SRP ($mg\ L^{-1}$)	NO_3^--N ($mg\ L^{-1}$)	DOC ($mg\ L^{-1}$)	DO ($mg\ L^{-1}$)	Groundwater table depth (m)
Creek A						
Growing season	Mean	0.045	1.042	1.707	5.199	0.795
	Standard deviation	0.05	0.88	1.46	1.44	0.20
	Maximum	0.35	2.31	9.10	9.20	1.23
	n	247	96	86	91	204
Nongrowing season	Mean	0.037	0.090	1.850	6.387	0.684
	Standard deviation	0.04	0.08	1.43	2.33	0.13
	Maximum	0.26	2.83	8.50	11.80	1.01
	n	99	6	60	53	54
Creek B						
Growing season	Mean	0.040	1.035	1.333	5.085	0.558
	Standard deviation	0.06	0.82	1.70	1.58	0.25
	Maximum	0.40	2.53	12.90	10.20	1.47
	n	148	56	58	60	116
Nongrowing season	Mean	0.028	0.203	3.294	5.973	0.582
	Standard deviation	0.04	0.10	4.14	1.96	0.25
	Maximum	0.18	2.26	20.40	10.40	1.31
	n	57	4	35	30	35
Overall						
	Mean	0.041	1.039	1.885	5.538	0.695
	Standard deviation	0.05	0.86	2.19	1.85	0.24

Notes: SRP = soluble reactive phosphorus. NO_3^--N = nitrate-nitrogen. DOC = dissolved organic carbon. DO = dissolved oxygen. *n* = sample size.

a detection limit of 0.1 mg L^{-1}, following the instructions in the operator manual (OI-Analytical 1997). Sample analyses for DOC started in January 2005.

Multivariate Analysis of Covariance Model. A multivariate analysis of covariance (ANCOVA) model (USEPA 1997; Bishop et al. 2005) using matched treatment (Creek A) and control (Creek B) data was used to determine the impact of the near-stream BMPs in the treatment creek (Creek A). An ANCOVA allows comparison of a response variable in two or more groups by considering the variability of other variables, called covariates, or more simply an ANCOVA can be used to control for the effect of a covaraite before making inferences on treatment effects. Variables collected in both the control creek and the treatment creek or in the respective drainage areas of both creeks included in-stream and groundwater SRP, NO_3^--N, DO, and DOC concentrations, groundwater table height (GWTD), and season (growing [April to November] or nongrowing [December to March]). In-stream samples were collected at seven locations in the treatment creek (Creek A, sites A1 to A7) and four locations in the control creek (Creek B, sites B1 to B4) (figure 1). The locations of the sampling sites A1 to A7 were aggregated into two classes, above or below the BMP (note that the culvert BMP was installed on Creek A). Table 1 presents the descriptive statistics collected during the study. Since less than 200 m (656 ft) separates the treatment and control creeks, rainfall (*RF*) was assumed to be evenly distributed between the two creeks and was incorporated as a variable in the analysis.

We constructed three stepwise ANCOVA models to explain the effect of the near-stream BMPs. First, we test whether there was a significant difference between the pre- and post-BMP time periods. If there is a significant difference between the pre- and post-BMP periods, the second ANCOVA tests for a significant difference between Creek A (treatment) and Creek B (control) in the post-BMP period by controlling for the variance associated with measurements in Creek B and groundwater. The third ANCOVA model tests for a significant difference between the location on Creek A above and below the BMP during the post-BMP period. Variables were natural log transformed to remedy increasing error variance and nonnormality of residuals. Note that initial model analysis showed that NO_3^--N was not a significant independent variable to explain SRP concentrations (nor SRP to explain NO_3^--N) and was thus not included in the model. Least squared means were used to estimate BMP effects in all models. For each seasonal model, the complete model with all main and interaction effects was fit, and nonsignificant terms (at $\alpha = 0.05$) were subsequently removed (see table 2 for significant terms). Model variables were selected using all-subsets regression and forward and backward regression methods with the partial *F*-statistic to add or drop a model variable at the $\alpha = 0.05$ level. The ANCOVA model (excluding interaction terms) is (using SRP as an example):

Table 2

Analysis of covariance (ANCOVA) for test of significant differences on in-stream soluble reactive phosphorus concentrations for Creek A.

Parameter ln(SRP_A)	Growing season p-value	Growing season SumSq	Nongrowing season p-value	Nongrowing season SumSq	Full year p-value	Full year SumSq
ANCOVA test for concentrations between pre- and post-BMP periods (equation 2)						
Intercept	0.044	—	0.002	—	0.033	—
ln($GWSRP_A$)	<0.001	79.46	ns	—	0.006	147.61
ln(SRP_B)	0.006	63.46	0.031	43.23	0.009	102.12
ln($GWTD_A$)	0.033	37.14	0.049	26.12	ns	—
ln(RF)	0.019	12.61	ns	—	ns	—
Period	0.007	8.62	ns	—	0.004	12.07
Adjusted r^2 (*p-value*)	0.89 (<0.001)		0.78 (0.181)		0.83 (<0.001)	
ANCOVA test of BMP effectiveness in the post-BMP period (equation 3)						
Intercept	ns	—	0.056	—	0.038	—
ln($GWSRP_A$)	<0.001	162.40	ns	—	0.007	269.06
ln(SRP_B)	<0.001	106.12	<0.001	146.15	<0.001	212.23
ln(RF)	<0.001	149.32	ns	—	0.018	200.29
BMP	0.011	16.06	ns	—	<0.001	34.51
Period	ns	—	ns	—	ns	—
BMP × Period	0.036	7.94	ns	—	0.048	4.43
Adjusted r^2 (*p-value*)	0.75 (<0.001)		0.44 (0.132)		0.56 (<0.001)	
ANCOVA test of location in Creek A (above or below the BMP) (equation 4)						
Intercept	ns	—	0.004	—	0.014	—
ln($GWSRP_A$)	<0.001	49.02	0.039	110.22	<0.001	271.35
ln(SRP_B)	0.015	14.58	ns	—	0.019	11.50
ln($GWTD_A$)	<0.001	16.06	0.048	150.54	<0.001	168.94
Period	ns	—	ns	—	ns	—
BMP	ns	—	ns	—	ns	
Location	<0.001	29.15	ns	—	<0.001	36.82
Period × BMP × Location	<0.001	5.24	ns	—	0.004	3.99
Adjusted r^2 (*p-value*)	0.82 (<0.001)		0.52 (0.028)		0.61 (0.016)	

Notes: *SRP* = soluble reactive phosphorus. SumSq = sum of squares. BMP = best management practice. — = no value, corresponding to the NS (not significant) values, or for the intercepts for which no SumSq is calculated. ns = not significant. *GWSRP* = groundwater soluble reactive phosphorus. SRP_B = soluble reactive phosphorus with slope B. $GWTD_A$ = groundwater table height with slope A. *RF* = rainfall. r^2 = coefficients of determination. *Period* and *BMP* were included in model due to significant interaction effect.

$$ln(SRP_A) = a + b(Period) + c(BMP) + d(Location) + e(lnSRP_B) + f(lnDO_A) + g(lnDO_B)\ h(lnDOC_A) + i(lnDOC_B) + j(lnRF) + k(lnGWSRP_A) + l(lnGWSRP_B) + m(lnGWDO_A) + n(lnGWDO_B) + o(lnGWDOC_A) + p(lnGWDOC_B) + q(lnGWTD_A) + r(lnGWTD_B) + \varepsilon_i, \quad (1)$$

where *ln* represents the natural log function, *a* is the intercept, letters *b* to *r* are the slopes of the individual variables, *Period* is the BMP time period (0 for pre-BMP, and 1 for post-BMP), *BMP* indicates the presence of the BMP (0 for no BMP present [Creek B], and 1 for BMP present [Creek A]), *Location* is an indicator variable in Creek A (0 for sampling locations above the culvert, and 1 for sampling locations below the culvert), ε_i is the model error, *GWDO* is groundwater dissolved oxygen, *GWSRP* is groundwater soluble reactive phosphorus, and all others variables as defined earlier. Subscripts *A* and *B* indicate which creek or contributing area the variable was measured (i.e., *A* indicates the measurement was made in Creek A or the Creek A contributing area).

Three seasonal ANCOVA models are used to analyze the effectiveness of the near-stream BMPs. First, we determine if there is a significant difference in-stream SRP or NO_3^--N concentrations between pre- and post-BMP time periods using the following ANCOVA (using SRP as an example):

$$ln(SRP_A) = a + b(Period) + c(lnSRP_B) + d(lnRF) + e(lnGWSRP_A) + f(lnGWTD_A) + \varepsilon_i. \quad (2)$$

For the seasonal model, the difference between the pre- versus post-BMP time period was tested using a one-sided *t*-test on the *b* coefficient at the $\alpha = 0.05$ level. Next, we test the lumped impact of the culvert on stream SRP or NO_3^--N levels during the post-BMP period (using SRP as an example):

$$ln(SRP_A) = a + b(BMP) + c(Period) + d(lnSRP_B) + e(lnGWSRP_A) + f(lnRF) + g(BMP \times Period) + \varepsilon_i, \quad (3)$$

where *a* is the intercept, *b* to *g* are the slopes of the individual variables, *BMP × Period* is an interaction term that indicates the presence of the BMP during the post-BMP period, and all others are defined earlier. The difference between Creek A and B with and

Table 3

Analysis of the Period (equation 2), *BMP*Period* interaction (equation 3) and *Period* × *BMP* × *Location* interaction (equation 4) for Creek A showing the estimated difference (in mg L^{-1}) in SRP concentrations. Note that the estimates were back transformed to nonlog units for display in the table, *p-values* are from log transformed tests.

	Growing season		Nongrowing season		Full year	
Parameter: *lnSRP*	Estimate	*p-value*	Estimate	*p-value*	Estimate	*p-value*
Pre-BMP versus post-BMP (equation 2)	0.0077	0.003	0.0001	0.614	0.0053	0.032
Post-BMP with BMP xversus Post-BMP no BMP (equation 3)	0.0071	<0.001	0.0003	0.822	0.0085	0.041
Pre-BMP period (equation 4)						
Above BMP versus below BMP	0.0056	<0.001	0.0030	0.081	0.0085	0.003
Post-BMP period (equation 4)						
Above BMP versus below BMP	0.0004	0.105	0.0001	0.194	0.0005	0.137

without the BMP during the post-BMP time period was tested using a one-sided *t*-test on the *g* coefficient at the $\alpha = 0.05$ level. Finally, we tested the direct impact of the culvert in Creek A during the post-BMP period (using SRP as an example):

$$\ln(SRP_A) = a + b(Period) + c(Location) + d(BMP) + e(\ln SRP_B) + f(\ln GWSRP_A) + g(\ln GWTD_A) + h(Period \times BMP \times Location) + \varepsilon_i, \quad (4)$$

where *Period* × *BMP* × *Location* is an interaction term that tests the effect of the BMP location in Creek A (above or below the BMP) during the post-BMP period. Differences between sampling locations in Creek A above and below the BMP location during the post-BMP period were tested using a one sided *t*-test on the *h* coefficient at the $\alpha = 0.05$ level.

Model Assumptions. Model assumptions for an ANCOVA model are similar to standard analysis of variance model assumptions. Three assumptions need to be verified prior to presenting and discussing results. First, ANCOVA models assume linearity, (i.e., the covariate has a linear relationship with the dependent variable), which upon initial testing was not satisfied but was corrected by a natural log transformation of the covariates. Second, ANCOVA models assume homogeneity of variance or the variance of group one is equal to the variance of group two, etc., which was verified on the natural log–transformed data using normal probability plots and performing a chi-square goodness of fit test (Snedecor and Cochran 1989). Third, the homogeneity of the regression slopes, or for each level of the independent variable (SRP, for instance), the slope of the prediction from the covariate must be equal. This assumption was tested by comparing the interaction effects of the *Period, BMP,* and *Location* variables in ANCOVA models against a *p*-value of 0.05. Table 2 shows that there were significant interaction effects for the ANCOVA model testing the effect of *BMP* in equation 3 and *Location* (equation 4). Because the slopes for the *Period*, *BMP*, and *Location* variables are not statistically equal, we use a model that estimates slopes for all *Period, BMP,* or *Location* variables separately. Using the "estimate" statement in SAS (SAS Institute 2008), we constructed statements to test the effect of *Period, BMP,* and *Location* for both the pre- and post-BMP periods. This test allowed us to estimate the impact of the BMP on SRP levels in Creek A while controlling for other variables in the ANCOVA (table 3).

It should be noted that all three variable selection methods (i.e., all subsets, forward, and backward regressions) resulted in the same parameter sets for all models. Identical analysis was performed for in-stream NO_3^--N concentrations, but no significant *Period, BMP,* or *Location* effects were detected, indicating that the fencing did not affect the NO_3^--N concentration in the stream. Therefore, our analysis focused mainly on SRP concentrations, but some discussion of NO_3^--N is included.

Results and Discussion

Descriptive Statistics. The overall average SRP concentrations measured in Creek A and Creek B were 0.043 and 0.037 mg L^{-1}, respectively, similar to groundwater for SRP levels (Flores-López et al. Forthcoming). Average NO_3^--N concentrations were 1.042 and 1.035 mg L^{-1} for Creek A and Creek B, respectively (table 1), but temporal variability was observed. The average DOC concentration in Creek B was significantly higher (2.07 mg L^{-1}) than the concentration in Creek A (1.76 mg L^{-1}). The average DO concentrations were not significantly different, 5.63 and 5.38 mg L^{-1} for Creek A and Creek B, respectively. The seasonal variability of all four chemicals (table 1) indicates that the lowest average stream SRP concentration was observed during the nongrowing season for both creeks (figure 2). Stream NO_3^--N concentrations were highest during the growing season and were below the detection limit during the winter (figure 3). The highest DOC concentrations for Creek A were observed during the growing season, and for Creek B, were observed during the nongrowing season. Higher stream DO concentrations were observed during the nongrowing season in both creeks (table 1).

In general, SRP concentrations in Creek A were higher than in Creek B for the pre-BMP period (before the crossing was installed). In Creek A, before the culvert was installed, the highest mean SRP concentration of 0.11 mg L^{-1} was measured during the growing season, and in Creek B, the highest levels observed were 0.04 mg L^{-1}. In the pre-BMP period, the lowest mean SRP concentrations were observed during the nongrowing season for both Creek A and Creek B (both were 0.03 mg L^{-1}). The mean SRP level for Creek A during the pre-BMP period was 0.06 mg L^{-1}, nearly twice the concentration in Creek B (0.03 mg L^{-1}). After the crossing was installed (post-BMP period), the SRP concentrations in Creek A declined by 30% (average of 0.04 mg L^{-1}), while conversely SRP levels in Creek B increased 41% (average of 0.05 mg L^{-1}) compared to the pre-BMP period (table 4). The increase in SRP concentrations in Creek B post-BMP indicates that exogenous variables, such as precipitation or temperature, might actually have reduced the perceived effectiveness of the crossing in Creek A.

Precipitation (figure 2) in 2004 and 2005 was 23% and 6%, higher than the long-term annual average precipitation (1,120 mm y^{-1} [44.1 in yr^{-1}] based on National Climatic

Figure 2
The control (Creek B) and treatment (Creek A) creek soluble reactive phosphorus (SRP) concentrations and precipitation depths during the pre- and post-best management practice (BMP) periods. The lowest average stream SRP concentration was observed during the nongrowing season, from December to March. Symbols show the average SRP concentrations for each creek (Sites A1 to A7 and B1 to B4, respectively). See figure 1 for site locations.

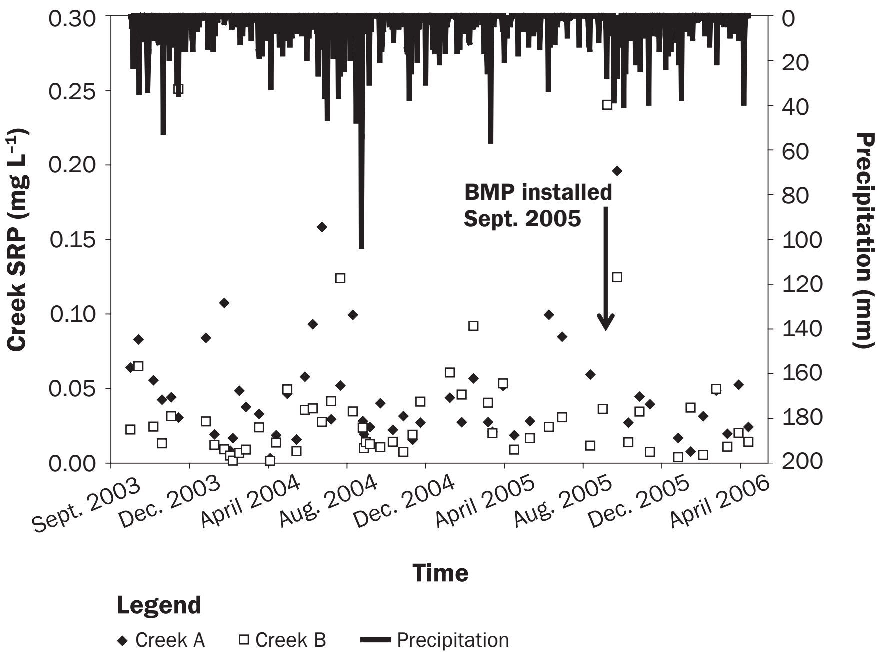
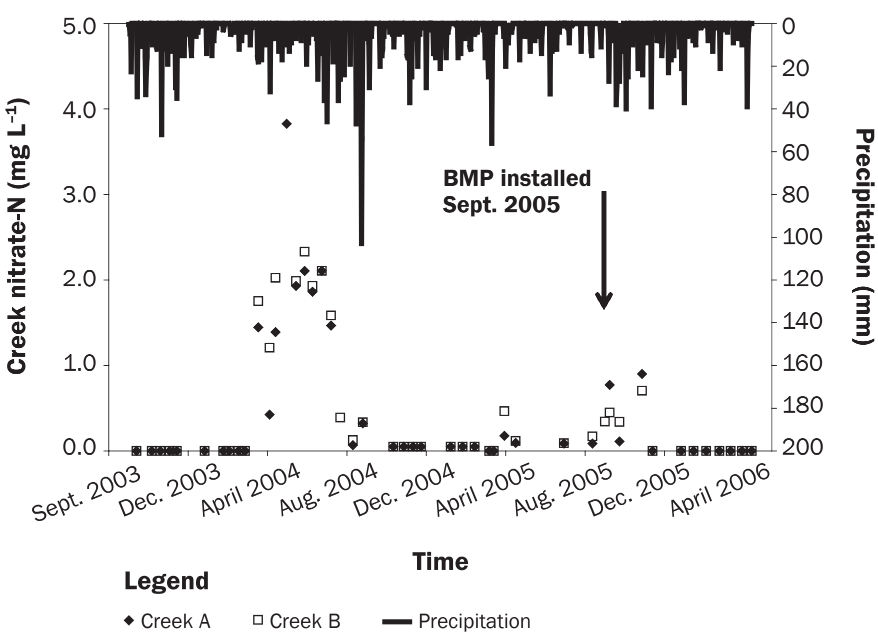


Figure 3
The control (Creek B) and treatment (Creek A) creek nitrate-nitrogen (NO_3^--N) concentrations and precipitation depths during the pre- and post-best management practice (BMP) periods. Stream NO_3^--N concentrations were highest during the growing season, from April to November. Symbols show the average NO_3^--N concentrations for each creek (Sites A1 to A7 and B1 to B4, respectively). See figure 1 for site locations. Note that the NO_3^--N detection limit is 0.05 mg L^{-1}.



Data Center 2000). The partial years sampled during the study, 2003 and 2006, had approximately normal precipitation levels. A significant difference was observed between the 2004 and 2005 growing seasons, where 1,093 mm (32.6 in) of precipitation was measured from April to November of 2004, compared to 796 mm (12.2 in) during the same months in 2005. The average precipitation for the growing season over the study period was 676 mm (23.5 in).

Figures 2 and 3 show time series of the SRP and NO_3^--N concentrations in Creek A and Creek B, respectively, over the study period. The NO_3^--N concentrations in Creek A and Creek B (figure 3) during the pre- and post-BMP period were well below the 10 mg L^{-1} standard for drinking water.

Note in figure 2 that elevated SRP concentrations occur in both the pre- and post-BMP periods. However, it is also apparent that the SRP concentrations in Creek B are elevated during the post-BMP period (figure 2). Simply looking at the time series, one might be tempted to assume that the BMP was effective since Creek A concentrations were significantly lower during the post-BMP period than during the pre-BMP period (paired *t*-test at α = 0.05). Creek B concentrations were, on average, significantly lower during the post-BMP period as well. Despite the two high SRP concentrations in Creek B during the post-BMP period, it is relatively clear that Creek B concentrations have fallen as well (figure 2). However, it is not clear from figure 2 what the source of the SRP reduction is, as other exogenous variables, such as climatic variability, could have influenced the Creek A concentrations. Thus, use of the matched concentration ANCOVA should correct for imbalances in precipitation, groundwater, and other factors between the pre- and post-BMP periods (Bishop et al. 2005) and thus result in detectable BMP effects if significant. Figure 4 shows a relatively strong relationship between the log of the SRP concentrations between Creek A and Creek B during both the pre- and post-BMP time periods, indicating that incorporating parameters controlling Creek B SRP concentrations should help improve model results.

Multivariate Analysis of Covariance. Table 2 presents the results of the three ANCOVA analyses by season and for the whole year. First, we test whether there was a significant difference between the pre- and post-BMP

Table 4

Average soluble reactive phosphorus concentrations by sampling sites for the pre-best management practice (BMP) and post-BMP scenarios in Creek A and Creek B.

Sampling site	Pre-BMP Mean (± se) (mg L^{-1})	Pre-BMP n	Post-BMP Mean (± se) (mg L^{-1})	Post-BMP n	Difference between scenarios (%)
Creek A (above BMP)					
A1	0.013 (±0.002)	32	0.008 (±0.002)	8	–38
A2	0.019 (±0.001)	40	0.013 (±0.003)	9	–32
A3	0.026 (±0.003)	41	0.036 (±0.008)	8	38
Creek B (below BMP)					
A4	0.058 (±0.008)	43	0.036 (±0.007)	10	–38
A5	0.080 (±0.013)	41	0.049 (±0.012)	10	–39
A6	0.053 (±0.009)	41	0.047 (±0.011)	10	–11
A7	0.057 (±0.009)	42	0.040 (±0.017)	11	–30
Creek B (no BMP)					
B1	0.033 (±0.008)	38	0.073 (±0.036)	11	121
B2	0.029 (±0.007)	39	0.040 (±0.016)	12	38
B3	0.027 (±0.006)	43	0.042 (±0.014)	12	56
B4	0.047 (±0.010)	38	0.039 (±0.017)	12	–17

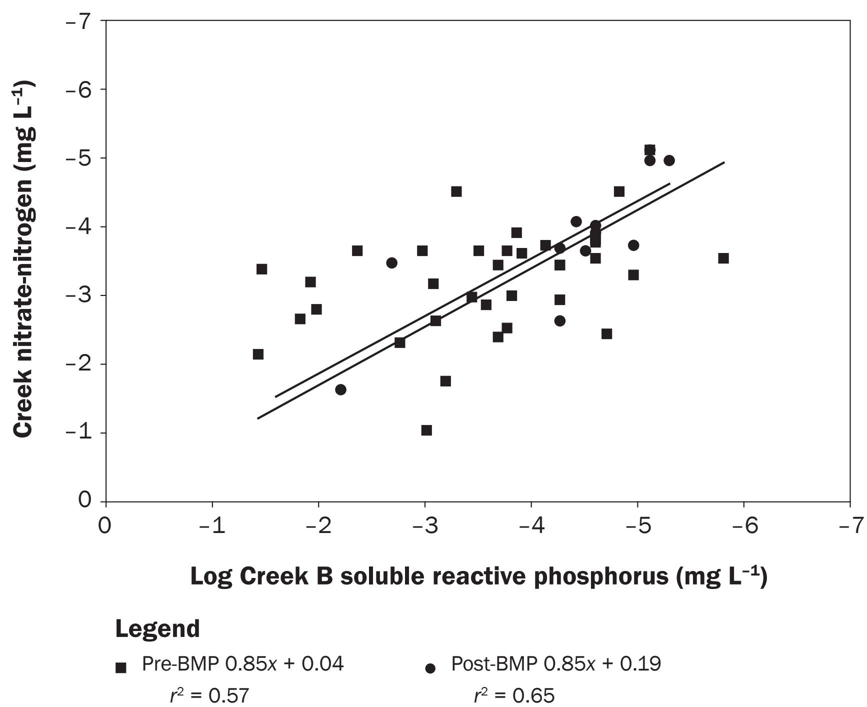
Note: se = standard error. n = sample size.

time periods (equation 2). The second ANCOVA tests for a significant difference between Creek A (treatment) and Creek B (control) in the post-BMP period by controlling for the variance associated with measurements in Creek B and groundwater (equation 3). The third ANCOVA model tests for a significant difference between location on Creek A above and below the BMP during the post-BMP period (equation 4).

The ANCOVA model, which was used to test for a detectable difference in the log of the SRP concentrations between the pre- and post-BMP time periods in Creek A (equation 2, table 2), resulted in high predictive accuracy for both the growing season and whole-year models (adjusted r^2 = 0.89 and 0.83, respectively, table 2), and significant overall effects. The nongrowing-season model was not significant at α = 0.05, yet still had good predictive power (adjusted r^2 = 0.78). The log of Creek A SRP levels during the growing season were highly covariate with the log of Creek B SRP, the log of groundwater SRP levels, and the log of groundwater table depth in the area contributing to Creek A, as indicated by the highly significant p-values and large sums of squares (table 2). More importantly, the model was able to detect a significant difference in the log of the Creek A SRP levels between the pre- and post-BMP periods for both the growing season and whole-year models (e.g., one sided t-test of the *Period* variable, table 2). The reductions in the pre- and post-BMP period SRP concentrations estimated by the

Figure 4

Log-matched concentrations (averaged by creek) for Creek A and Creek B over the pre- and post-best management practice (BMP) periods.

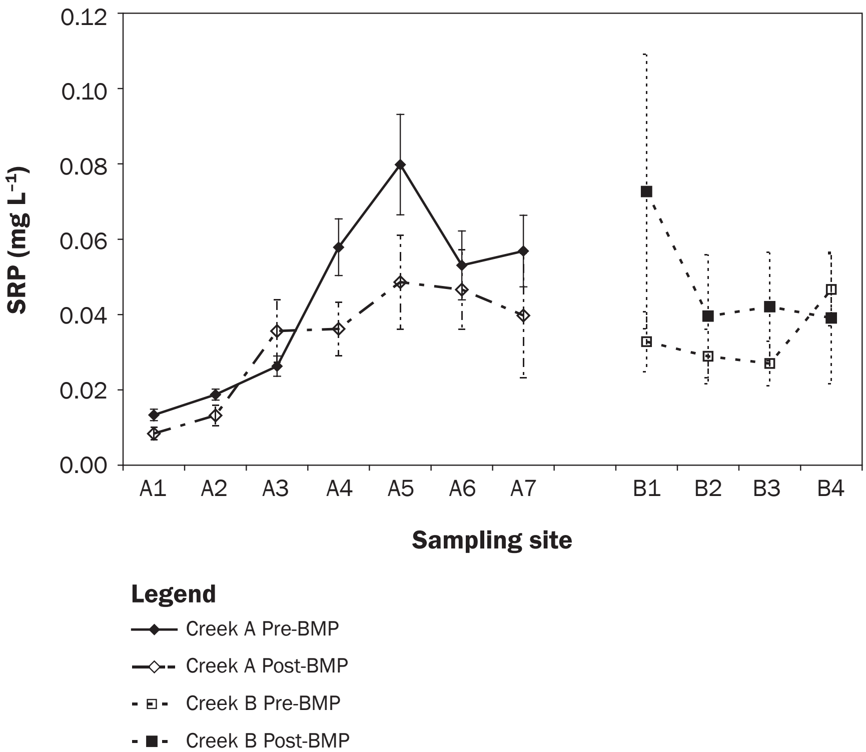


model were 0.0077 mg L^{-1} during the growing season and 0.0053 mg L^{-1} for the whole year (table 3).

To test the effectiveness of the BMP installed on Creek A, we constructed a second model (equation 3 and table 2). This overall model was highly significant for the growing season and whole-year periods but not for the nongrowing season period (table 2). Similar to the model to test the pre- and

Figure 5

Mean soluble reactive phosphorus (SRP) stream concentrations by sampling sites for Creek A and Creek B. Stream sampling site locations are shown in figure 1. Note that the best management practice (BMP) was installed between sampling sites A3 and A4 in Creek A. Sampling sites in Creek A were lumped into above and below the BMP site for analysis.



post-BMP time periods (equation 2), the log of the groundwater SRP levels in the areas contributing to Creek A and the log of the SRP concentrations in Creek B were highly significant covariates (table 2). The test of the effectiveness of the BMP was assessed only during the post-BMP period (e.g., the *BMP* × *Period* interaction in equation 3 and table 2). The ANCOVA model detected a significant *BMP* × *Period* interaction in Creek A for the growing season and whole-year model (table 2). Table 3 shows the magnitude of the reduction attributable to the BMP to be 0.0071 and 0.0085 mg L^{-1} SRP for the growing season and whole-year models, respectively.

To investigate the impact of the BMP in Creek A, we constructed a final test indented to ensure that other factors were not unduly influencing the BMP effect (equation 4 and table 2). As with previous models, the log of the Creek A SRP concentration was covariate with the log of the Creek B SRP levels, the log of the groundwater SRP levels, and groundwater table height in the area contributing to Creek A, (table 2) for both the growing season and whole-year models. Interestingly, the nongrowing-season model was significant at $\alpha = 0.05$ (table 2), but there was no BMP effect, only the log of the groundwater SRP and the log of the groundwater table height in the Creek A area were significant predictors of the log of the SRP concentrations in Creek A (table 2). Thus, since our focus is on BMP effects, we do not explore the impact of these variables on the SRP levels in Creek A. The three-way interaction term in the model, *BMP* × *Period* × *Location,* results in a test of the stream sampling location for each of the pre- and post-BMP periods (e.g., pre-BMP period below the BMP location versus pre-BMP period above the BMP location and post-BMP period below the BMP location versus post-BMP above the BMP location). The results of these tests are shown in table 3, where it is estimated that during the pre-BMP period, there was a significant difference between locations in Creek A. Specifically, sampling points located above the BMP location (figure 1) had SRP concentrations 0.0056 and 0.0085 mg L^{-1} lower than locations below the BMP for the growing season and whole-year models, respectively (table 3). There was no significant difference in the SRP concentrations during the nongrowing season. During the post-BMP period, there was no significant difference between the sampling locations above and below the BMP location for any of the models (table 3). Figure 5 shows the SRP concentrations by sampling sites in both Creek A and Creek B for the pre- and post-BMP periods. Estimated reductions during the post-BMP period resulting from the ANCOVA models for equations 3 and 4 both indicate that the BMP resulted in a SRP reduction of approximately 27% for the yearly model and 34% for the growing-season model.

While culverts and exclusionary fencing are widely accepted BMP practices to reduce N and P pollution of water bodies in agricultural areas, there is little quantitative data regarding their impact on water quality. Intuitively, preventing direct livestock access from water bodies will reduce the chance of fecal deposits and the disturbance of P-laden stream sediments. However, environmental noise can often obscure the effects of BMPs in agricultural systems, making detection of their impacts difficult. For instance, while N reductions were not attributable to BMPs installed on Creek A, it appears from figure 3 that NO_3^--N concentrations in both Creek A and B were lower during the post-BMP period. This difference might be due to BMP impacts (not yet detectable) or from climatic variation and may become significant over time. Thus additional analysis is warranted.

Analysis of SRP results presented in table 3 indicate that during the pre-BMP period, there was a significant (at $\alpha = 0.05$) difference between SRP concentrations from Creek A sampling locations located above and below where cattle had direct stream access (i.e., the in-stream cattle crossing) for the growing season. The full-year model was significant as well but slightly less so than the growing-season model (table 3). During the nongrowing season, there was no significant difference between sampling locations, which is somewhat intuitive, as cattle are not pastured during the winter and thus had no access to the stream. Estimates of the contrast differences likewise show a large difference in the SRP concentrations in Creek A above and below the crossing in the pre-BMP period (table 3). In the pre-BMP period, during the growing season, the SRP concentrations were, on average, 0.0056 mg L^{-1} lower at sampling locations above the crossing (table 3 and figure 5). Expected reductions in the SRP levels during the nongrowing season were lower (0.003) and not significant (p-value = 0.822) (table 3).

During the post-BMP period, both the seasonal and full-year ANCOVA models (equation 4 and table 3) indicate that there is no significant difference between sampling locations above and below the BMP installation (table 3), which is due somewhat to large sample variance in the Creek A samples (figure 5). While the estimated differences remain negative (i.e., estimated reductions in SRP concentrations between locations above and below the BMP were 0.0001 to 0.0005 mg L^{-1}), they are not significant at the α = 0.05 level. The marginal significance of the growing-season model (p-value = 0.105) might simply be a result of limited sampling duration during the post-BMP period. However, it is somewhat remarkable that the estimated differences in SRP concentrations equalized across sampling locations so quickly following installation of the BMP. Table 4 presents the SRP concentrations during the pre- and post-BMP periods for both Creek A and Creek B at all sampling points, and clearly shows that, while SRP concentrations declined significantly below the BMP location (A4 to A7), they were still somewhat higher than the SRP concentrations above the BMP location (A1 to A3). We speculate that a longer sampling duration would have further reduced the differences between sampling locations along Creek A (i.e., above and below the BMP).

The majority of the water in alluvial valley streams in the Catskill Mountains originates from hillsides and flows both overland, from the runoff source areas, and subsurface to the creeks. During the period of the year when precipitation exceeds evapotranspiration, runoff is generally the largest input to the stream from saturated areas. During the remainder of the year, subsurface and return flows are the major sources of creek water. Thus, both surface and groundwater flows are sources of SRP in the stream. From a water quality standpoint, agricultural activity can have a large impact on alluvial streamwater quality. In this study, the SRP concentration measured at the spring sampling site (A1 in figure 1) was 0.011 mg L^{-1}, indicating that Creek A SRP levels are initially controlled by the SRP concentrations at the spring site (A1) (figure 5). This is corroborated by the similar SRP levels measured at site A2 (0.016 mg L^{-1} of SRP). directly downstream from site A1, where little agricultural activity takes place. Further downstream (and prior to BMP installation), the concentration is influenced by P inputs from manure applications to fields (Kleinman et al. 2007), nonfield areas (e.g., barnyard and cattle crossing path) (Hively et al. 2005), and the P concentration in streambed sediment (McDowell et al. 2001; Evans et al. 2004; van der Perk et al. 2007). Re-entrainment of P-rich sediments in streams is speculated to be a large P source in the Cannonsville Reservoir watershed, where livestock often graze freely and have access to streams (James et al. 2007). In the study site, the dominant source areas for SRP are located in the regions downstream of the spring site (A1), specifically the nonfield areas of the cattle crossing, from subsurface flow generated in the agricultural areas surrounding the stream, and directly from the stream channel itself. The concentration in the stream is generally higher than in the groundwater, indicating that surface sources, such as manure application, or the stream sediments and livestock are contributing to the increased stream SRP concentration, particularly during the summer when groundwater contributions are limited. Interestingly, Creek A SRP levels decline somewhat at sites A6 and A7, which might be due to several factors, such as periphyton growth in the stream removing soluble P or increased groundwater contributions diluting Creek SRP concentrations. Conversely, it might be more interesting to ask why the SRP concentration at site A5 was so high. The elevated SRP concentration at site A5 was likely due the area being a common stream entry point for pastured cows.

Conversely, NO_3^--N, which is very soluble and does not become fixed on clays or organic matter and is easily transported with water, was measured at a higher concentration in the groundwater (average of 2.2 ± 0.2 mg L^{-1}) (Flores-López et al. Forthcoming) than in the streams (average of 1.04 ± 0.07 mg L^{-1}). That the NO_3^--N concentrations measured in the stream are substantially lower than those measured in the groundwater indicates that N in groundwater may be undergoing denitrification in the carbon rich, saturated, near-stream areas, thus reducing the ultimate input to the stream. The increased DO and DOC levels from manure applications in coincidence with saturated areas near the stream increase the microbial activity and thus the denitrification. Indeed, several researchers have noted the importance of DO and DOC levels in N and P dynamics (Flores-López et al. Forthcoming; Boyer et al. 1997; Schilling and Jacobson 2008). These effects might explain why there was no detectable change in NO_3^--N concentrations following BMP installation.

Flores-López et al. (Forthcoming) showed that groundwater had an impact on in-stream concentrations of SRP. The average SRP groundwater concentration (0.041 mg L^{-1}) in the piezometers was slightly lower, but not statistically different (p-value = 0.612, sample size [n] = 1,198), from the in-stream concentration (0.043 mg L^{-1}) in Creek A and Creek B over three years of sampling. The groundwater SRP distribution depends not only on agricultural practices but also on factors such as depth to the groundwater table (Flores-López et al. Forthcoming; Weiler and McDonnell 2006), occurrence of subsurface riparian flow (Schilling et al. 2007; Schilling and Jacobson 2008), or exfiltrated water in low-lying areas or near the stream channel (Scott and Weiler 2001). Creek B in figure 5 appears to be influenced more by these factors than Creek A, which is more heavily dominated by agricultural activity, particularly sites A3 to A7. For piezometers P6, P7, and P10, located in the contributing area for Creek A sites (A4 to A7), there were no significant differences in SRP concentrations during the pre- and post-BMP periods. However, at stream sites A4 to A7 (figure 1), the in-stream SRP concentration for the post-BMP period was significantly lower (0.043 mg L^{-1}) than the in-stream SRP concentration for the pre-BMP period (table 4) (0.062 mg L^{-1}) (p-value = 0.017, n = 208). The inclusion of the groundwater SRP levels was an important parameter to consider in the analysis of the BMP effect.

There are two ways that the installation of the cattle crossing might have reduced the SRP concentrations in the stream. Probably the most significant impact that BMPs such as cattle crossings have is that they prevent direct access of the cattle in the stream channel and thus prevent the high P sediment in the streambed from becoming entrained in the water column and desorping to equilibrate in-stream P concentrations. Excluding cattle from the stream also prevents direct fecal inputs to the water bodies. Previous studies have shown that stream crossings and cattle paths that allow direct access to stream channels are a large source of P in streams (Bishop et al. 2005; Hively et al. 2005). Hively et al. (2005) found that hydrologically active nonfield areas, which oftentimes cover a small spatial extent but produce high P concentrations (e.g., cow paths and barnyards), can contribute substantially to soluble P loading in streams. Indeed, prior to the installation of the culvert, SRP levels were excessively high at sampling sites directly adjacent to the crossing. The measured decline in SRP concentrations at sampling points downstream of the culvert crossing indicates that these structures considerably lower SRP contributions to the stream. Interestingly, in Creek B, higher SRP concentrations were measured for the post-BMP period. An average increase of 41% in SRP concentrations were observed at the four sampling sites in Creek B during the post-BMP period (B1 to B4 in figure 1). This gives some perspective to the reduction of SRP levels in Creek A, in that controlling for environmental variability in the ANCOVA, the estimated reduction due to the BMP in Creek A is much more significant.

Finally, the fact that low SRP concentrations were observed in streamwater samples implies that this valley bottom farm is not as large a pollutant source as many of the upland farms that have been monitored in the upland areas of the Catskills. Although high SRP concentrations were observed from samples taken in the saturated areas adjacent to Creek B, they were not as high as the SRP concentrations reported by Hively et al. (2005) and Kim et al. (2006) in nearby upland farms, again indicating limited contribution of the alluvial areas, at least on this farm.

Summary and Conclusions

We collected three years of groundwater and surface water samples to quantify the impacts of groundwater and near-stream BMPs that exclude livestock from streams on SRP and NO_3^--N surface water concentrations by fencing and improved cattle crossings on a valley dairy farm in the Catskill Mountains of New York State.

A multivariate analysis of covariance incorporating groundwater and surface water measurements from a control and treatment creek was developed to determine the impact of near-stream BMPs (fencing and cattle crossing) on in-stream SRP concentrations. The results of the analysis indicate that incorporation of these exogenous variables into the model increased the sensitivity and capability of the model to detect BMP effects. The ANCOVA model showed that the installation of the BMPs resulted in a 27% reduction in Creek A SRP concentrations on a yearly basis and nearly 34% for the growing season. There was no detectable effect of the BMP during the nongrowing season.

The temporal dynamics of processes governing P levels in streams indicate that many factors are involved. The fact that low SRP concentrations were observed in stream samples, irrespective of BMP installation, implies that the valley bottom farms are not contributing as much as many of the upland farms that have been monitored in the West Branch of the Delaware River watershed, although high SRP concentrations were observed in groundwater from saturated areas adjacent to Creek B. These results indicate that near-stream and in-channel processes should be considered when assessing the impact of agricultural activities on water quality.

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