

# Reduced additions to river phosphorus for three years following implementation of a lawn fertilizer ordinance

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## Abstract

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Water quality data collected weekly from May to September 2008–2010 show statistically significant reductions in total phosphorus (TP), dissolved phosphorus (DP), and soluble molybdate-reactive phosphorus (SRP) following implementation of a municipal ordinance limiting the application of lawn fertilizers containing phosphorus (P). No reductions were seen at an upstream control river site not affected by the ordinance. Nontarget analytes including nitrate, silica, colored dissolved organic matter, specific conductance, and pH did not change systematically at experimental sites as did P. The data were compared with a multiyear historical dataset at weekly and subweekly resolution that preceded the ordinance. Mean concentrations of the 3 P analytes were lower than those of the reference period in 43 of 45 comparisons, and statistically significant reductions were observed in one or more of these analytes from June to September, but not in May. A statistical model that incorporated temporal effects, stream flow, and variations at the upstream control site detected highly significant effects of time period (pre- or postordinance) for P, but typically no significant effects for nontarget analytes. The average reductions compared to reference conditions from June to September at 3 sites affected by the ordinance ranged from 24 to 52% for SRP, 23 to 35% for DP, and 11 to 23% for TP.

Key words: eutrophication, fertilizer ordinance, nonpoint source loading

In an earlier report (Lehman et al. 2009) we pointed out that many municipalities and state governments have adopted or are considering adoption of legislation that restricts the residential use of phosphorus-containing lawn fertilizers. The actions are typically based on environmental awareness that phosphorus (P) is not a limiting element in many terrestrial soils and that overapplication of P can contribute to eutrophication of surface waters. Yet in many if not most cases, the restrictions are applied in the absence of any framework to assess the efficacy of the regulatory actions in terms of reducing P runoff to rivers and lakes. Any available evidence on the subject could therefore help inform ongoing management decisions about surface water environmental protection. This report addresses this important issue through a specific case study.

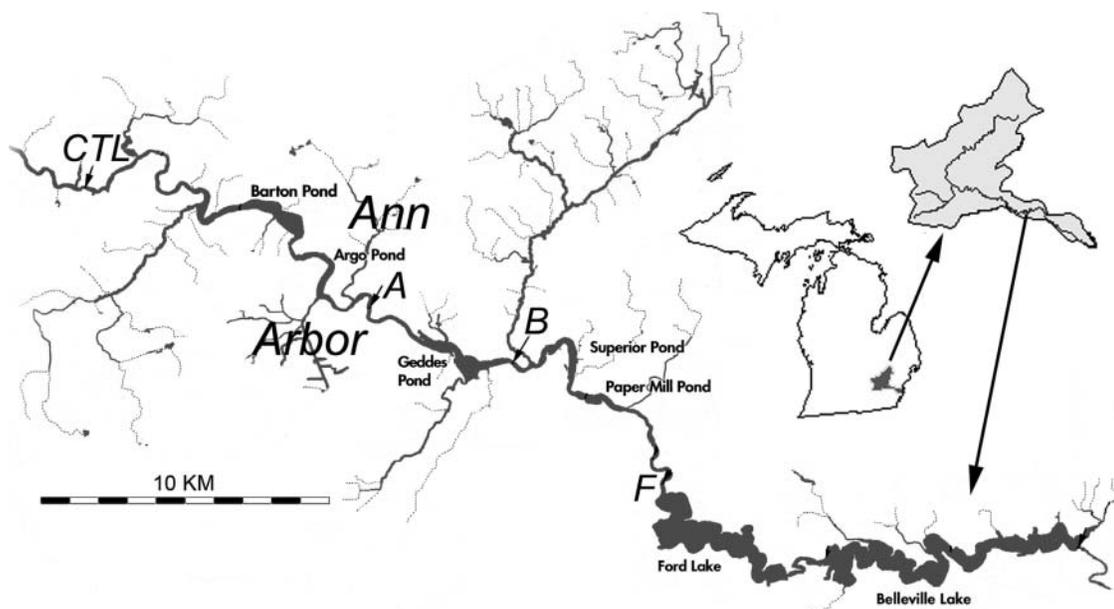
In response to a state-imposed P total maximum daily load (TMDL) that called for a 50% reduction in P discharges to the Huron River, the city of Ann Arbor, Michigan, en-

acted an ordinance that went into effect in 2007 (Ann Arbor 2006) to limit P application to lawns. The projected effect of full compliance was a 22% reduction in P in the river. The prediction was made by estimating the lawn fertilizer runoff from one creek-shed within the city and extrapolating that result to all other city creek-sheds. Statistical modeling (Ferris and Lehman 2008) using historical Huron River water quality data indicated that reductions of that magnitude for total P (TP) could be detected within 2 years by sampling 4 times per month. Detection at similar percentage reductions in dissolved P (DP) was predicted to take 2–3 years, and for soluble molybdate-reactive P (SRP) as long as 8 years. However, after just 1 year (2008), Lehman et al. (2009) were able to report a statistically significant reduction in TP and a trend of decreasing concentration for DP. This report summarizes the results after 3 sampling seasons and concludes the experimental study.

## Study site

Our field site is part of the Huron River catchment in southeastern Michigan (US Geological Survey, USGS Cataloging

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**Figure 1.**—Study site, with sampling stations CTL (Control), A, B, and F identified, from Lehman et al. (2009).

Unit 04090005). Four sampling stations were established for this study (Fig. 1). The station designated Control (CTL) corresponds to station 1 of Ferris and Lehman (Figure 1 in Ferris and Lehman 2008). It is upstream from Ann Arbor and outside the jurisdiction affected by the city ordinance. Stations A and B correspond to Ferris and Lehman's stations 5 and 6. Station A represents about 29 km<sup>2</sup> of catchment attributable to Ann Arbor and station B about 94 km<sup>2</sup>. A fourth station, designated F, corresponds to station 9 of Ferris and Lehman (2008) and is downstream of Ann Arbor at the site where the Huron River discharges into Ford Lake. Station F is also 9.5 km downstream from the outfall of the Ann Arbor wastewater treatment facility (AAWWTP); stations A and B are upstream of the outfall. Water quality data at station F have been reported by Ferris and Lehman (2007) and include 4 years (2003–2006) prior to implementation of Ann Arbor's fertilizer ordinance. Daily discharge of TP to the Huron River from the AAWWTP was obtained from the operator's logs.

By unforeseen coincidence, the start of this postordinance study coincided with removal of a dam in Dexter, Michigan, 5.6 km upstream from the CTL site, which had impounded a significant tributary of the Huron River named Mill Creek. Removal began in early June 2008 and the impoundment was drained over a period of several weeks.

### **Field sampling and analyses**

Water samples were collected at weekly intervals from May to September 2008–2010. Sample analyses were identical to

those described by Lehman et al. (2009) and included SRP, DP, TP, soluble molybdate-reactive Si (SRSi), pH, and nitrate (NO<sub>3</sub>). The SRP was measured as molybdate-reactive phosphate in filtrate; DP and TP were measured as SRP after first oxidizing filtrate (DP) or unfiltered water (TP) with potassium persulfate at 105 C for 1 h. Also measured were specific conductance at 25 C ( $K_{25}$ ,  $\mu$ S) and colored dissolved organic matter (CDOM) as UV absorbance at 254 nm. Ferris and Lehman (2008) showed that CDOM correlates strongly with both dissolved organic carbon (DOC) and dissolved organic nitrogen (DON) in the Huron River. All nutrient analyses were performed according to Ferris and Lehman (2007). For SRP and TP, 3 replicates were measured at each site. For DP, 2 replicates were measured. Laboratory analyses were typically completed the same day as sample collection. Sample means and standard error of the mean (SE) among replicates were calculated for each determination, and additional replicates were added if the ratio of SE to mean exceeded 0.05, which occurred only 10 times during the entire study.

### **Statistical methods**

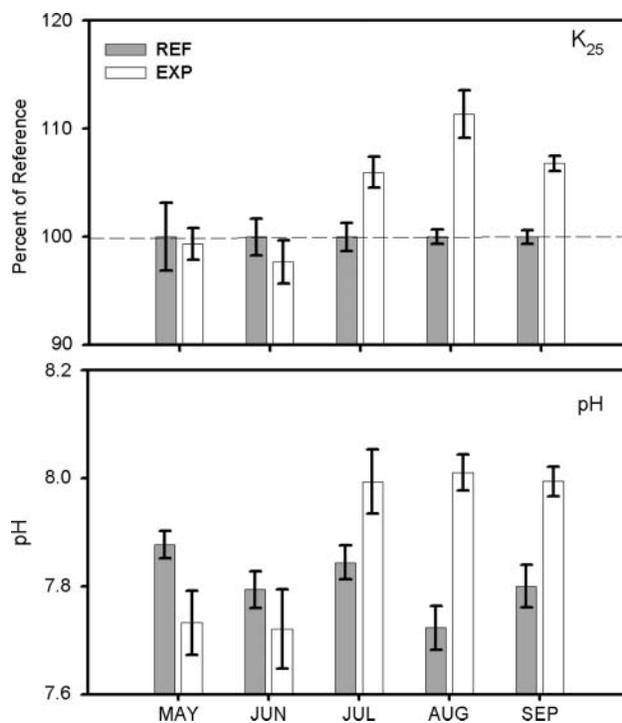
Following Lehman et al. (2009), the primary response variables of interest were SRP, DP, and TP. The analytes NO<sub>3</sub>, CDOM, SRSi, pH, and  $K_{25}$  were included as nontarget or quasicontrol variables. They were meant to represent a sampling of variables for which there was no *a priori* expectation of change and that were easy to measure with high precision. We adopted the statistical model by Ferris and

Lehman (2008) for balancing type I error against type II error such that  $\alpha = 0.1$  and  $\beta = 0.75$ . The object was to hold Type I error reasonably low while seeking a credible level of power to detect environmental changes if they occurred. Statistical power calculations using this model (Ferris and Lehman 2008) indicated that weekly sampling should have a reasonable expectation of detecting 25% changes in DP and TP after 2–3 years. Our *a priori* expectation was that P concentrations would decrease, so we applied one-tailed tests to the P data. Absent *a priori* expectations for the nontarget variables, we applied 2-tailed tests and set  $\alpha = 0.1$  to mimic the threshold probability applied to the P variables.

Phosphorus concentrations entering the experimental unit from upstream were an important determinant of local concentrations. For example, over the 6 years of study, SRP concentration at sites A and B exhibited correlations with coincident concentrations at upstream site CTL with coefficients ( $R$ ) of 0.737 and 0.704, respectively, but the correlations fell to 0.362 and 0.303, respectively, if CTL was compared with A and B 1 week later. Correlations fell even further, to 0.174 and 0.153, respectively, if CTL was compared with A and B 2 weeks later. Consequently, we focused on contemporaneous differences between CTL and experimental sites as the key indication of treatment effect.

Most of our water chemistry variables were known to exhibit lognormal distributions (Ferris and Lehman 2008), and so SRP, DP, TP,  $\text{NO}_3$ , SRSi, and CDOM were log-transformed prior to statistical comparison. The additional parameters  $K_{25}$  and pH exhibited normal distributions and were used in statistical tests without transformation. Because of the spatial and temporal correlations cited above, and because P concentrations at the CTL site were found to have increased postordinance (see *Control site*), we analyzed changes at the experimental sites A, B, and F in terms of their deviations from CTL (i.e., as  $A - \text{CTL}$ ,  $B - \text{CTL}$ ,  $F - \text{CTL}$ ). For ease of presentation we summarized the results in graphical form. For each month at each sampling site, the mean analyte deviations from CTL during the reference period (2003–2005) were regarded as benchmark values of 100%. The experimental mean monthly concentration deviations measured from 2008 to 2010 were then portrayed as percent of reference using vertical bar graphs with error bars indicating the 90% confidence interval for each mean. The exception was pH, for which the deviations were portrayed in pH units directly rather than as percent of reference (see Figs. 2 and 3). Confidence intervals were calculated with respect to the combined standard error of the reference and postordinance datasets:

$$SE_{\text{joint}} = [(\text{var}_{\text{ref}}(n_{\text{ref}} - 1) + \text{var}_{\text{post}}(n_{\text{post}} - 1)) \times (n_{\text{ref}} + n_{\text{post}} - 2) \cdot (1/n_{\text{ref}} + 1/n_{\text{post}})]^{1/2}, \quad (1)$$

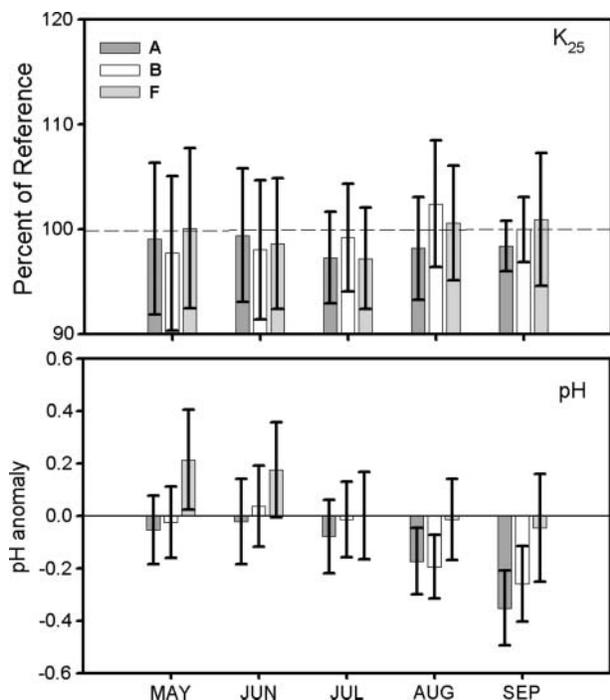


**Figure 2.**—Concentration differences at station CTL for  $K_{25}$  and pH from May to September 2008–2010 (EXP) compared to the reference period (REF: 2003–2005). Error bars represent 2-tailed 90% confidence intervals.

where  $\text{var}_x$  and  $n_x$  represent the variances and sample sizes of the reference and postordinance datasets, respectively.

Significant reductions in P concentration at experimental sites A, B, and F were recognized as cases where both the postordinance sample mean and its one-tailed confidence bar were  $\leq 100\%$  of the corresponding reference values (Fig. 4). Mean values for nontarget variables were likewise portrayed as bar graphs, but the error bars represent 2-tailed 90% confidence intervals. Significant differences from reference conditions were recognized as cases where the postordinance sample mean plus or minus the confidence interval differed from 100%.

Additionally, an analysis of variance (AOV) model was constructed using SYSTAT 10 for each target and nontarget variable at sites A, B, and F. Month and Period were entered as factors; river discharge and analyte values at the CTL site were treated as covariates. Statistical tests had revealed strong same-day correlations among analytes measured at different stations ( $P < 0.000001$ ), and this approach effectively isolated changes within the experimental unit from upstream variations of uncertain cause.



**Figure 3.**—Differences from CTL for specific conductance ( $K_{25}$ ) and pH in 2008 to 2010 compared to reference conditions, with 2-tailed 90% confidence intervals.

Our statistical model was predicated on a potential step change in river P concentrations between pre- and postordinance conditions, but we also tested for evidence of temporal trends postordinance by conducting 2-way AOV using Year and Month as factors, and with SRP, DP, and TP as response variables. This analysis was designed to detect both seasonal changes within the years and progressive changes from year to year.

All original data used in these analyses are archived for public access (<http://www.umich.edu/~hrstudy/dataarchive.htm>), but we report mean and SE for P analytes at the CTL station during preordinance conditions (Table 1) for comparison with other river systems.

**Table 1.**—Mean concentrations (and SE) of the P analytes ( $\mu\text{M}$ ) at the CTL station during the preordinance period (2003–2005).

	SRP	DP	TP	N
May	0.17 (0.06)	0.43 (0.09)	0.80 (0.14)	10
Jun	0.15 (0.02)	0.45 (0.03)	0.91 (0.05)	15
Jul	0.22 (0.02)	0.52 (0.03)	1.13 (0.07)	17
Aug	0.19 (0.02)	0.48 (0.03)	0.93 (0.06)	14
Sep	0.16 (0.01)	0.39 (0.03)	0.64 (0.03)	12

### Hydrology

Monthly mean fluvial discharge of the Huron River at Ann Arbor (USGS 04174500) for May to September was compared between the reference years (2003–2005) and the postordinance period (2008–2010). Data were subjected to logarithmic transformation and 2-way AOV with Period (reference vs. postordinance) and Month as factors. The reference period had lower flow on average than the postordinance period ( $P = 0.015$ ), with strong monthly variation ( $P = 0.005$ ), but no Period  $\times$  Month interaction ( $P = 0.86$ ).

### Control site

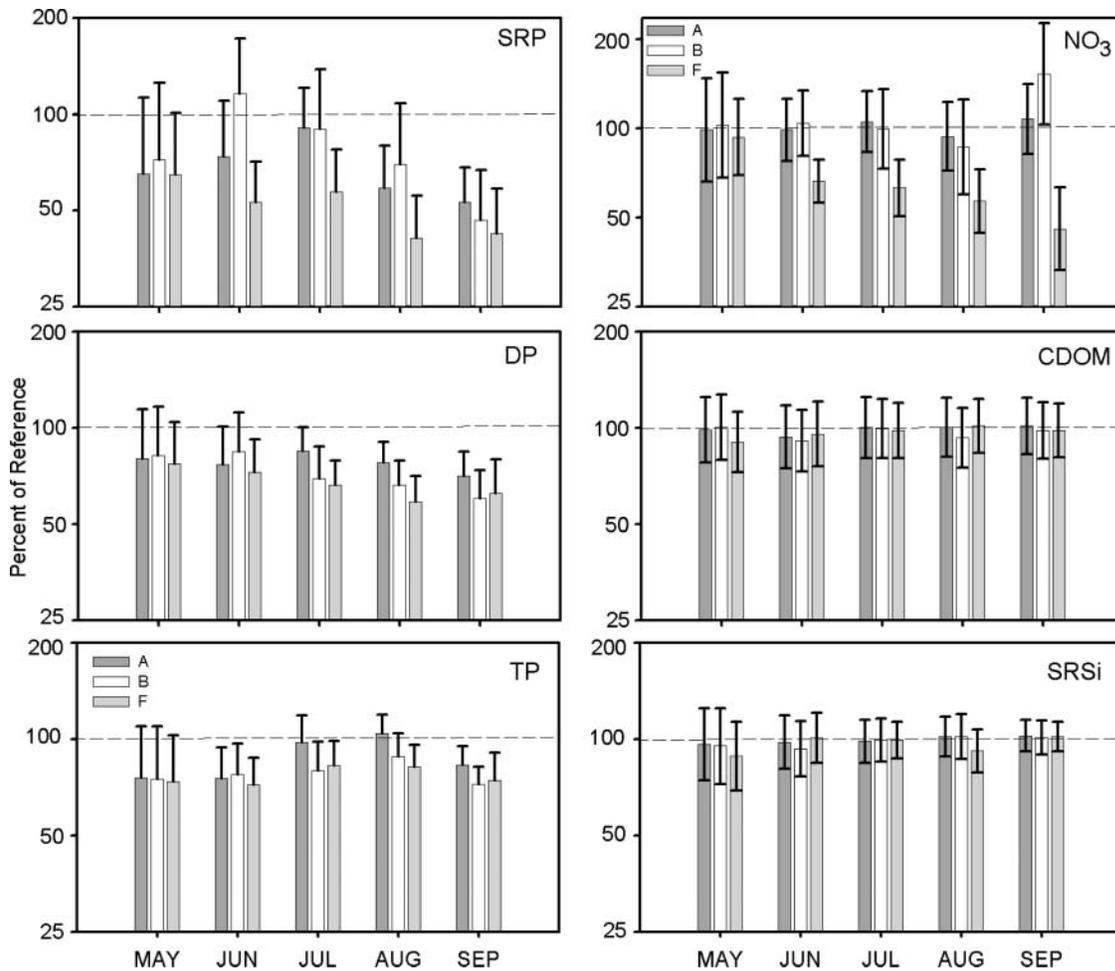
Comparison of pre- and postordinance conditions at the CTL site revealed that P concentrations increased from the reference period (Fig. 5), with statistically significant effects ( $\alpha = 0.1$ ) for SRP in all months, for DP from June to September, and for TP in June and September. Among nontarget variables (Fig. 2 and 5), there were statistically significant differences at the CTL site in 3 of 5 months for  $\text{NO}_3$  and  $K_{25}$ , 4 of 5 months for CDOM and pH, and 2 months for SRSi. To help evaluate whether the differences might be ascribed to hydrologic conditions, we subjected the P variables to 2-way AOV with log-transformed monthly mean fluvial discharge as covariate. For SRP, there was only a Period effect ( $P = 0.015$ ) and no effect of Month ( $P = 0.36$ ) or stream discharge ( $P = 0.43$ ). For DP and TP, however, there was no Period effect ( $P = 0.48$  and  $P = 0.58$ , respectively), but there were effects of Month ( $P = 0.08$  and  $P = 0.005$ , respectively) and stream flow ( $P = 0.05$  in both cases).

### Nontarget variables

Nontarget variables  $\text{NO}_3$ , CDOM, and SRSi (Fig. 4) did not exhibit different deviations with respect to the CTL site from 2008 to 2010 compared to the reference period with the exception of  $\text{NO}_3$  at station F (downstream of AAWWTP). Nitrate is a major component of the effluent from the AAWWTP, but it is not monitored by the facility. Similarly, there were no differences in the deviations for  $K_{25}$  at any of the downstream sites, but there was evidence of lower pH at sites A and B in August and September (Fig. 3). Thus, after accounting for temporal variation at CTL, there was little systematic change in nontarget variables at our downstream experimental sites.

### Phosphorus variables

Consistent with past sampling experience, SRP was more variable than DP or TP (Fig. 4), and although mean values were lower than reference in all cases but one, the confidence intervals overlapped with the reference values in 8



**Figure 4.**—Concentration differences from CTL from May to September 2008–2010 compared to the reference period (= 100%). Left: SRP, DP, and TP with one-tailed 90% confidence intervals. Right:  $\text{NO}_3$ , CDOM, and SRSi with two-tailed 90% confidence intervals. Error bars incorporate the combined variance of pre- and postordinance data.

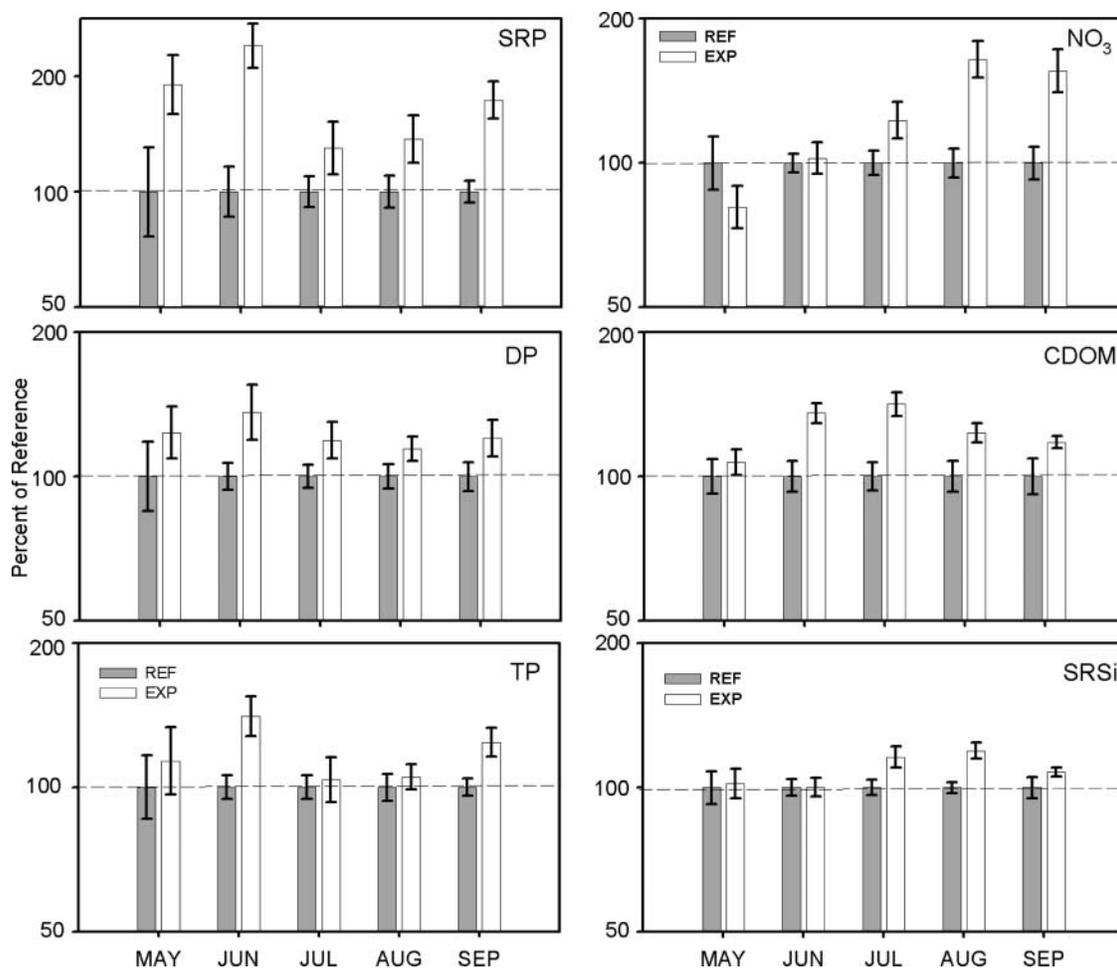
of 15 comparisons. For DP, mean concentration deviations were lower than reference at all stations and for all months. These differences were statistically significant in 9 of 15 cases, mostly July to September. TP concentration deviation means were consistently below reference levels except at station A in August, and like DP, the differences were statistically significant in 9 of 15 comparisons. As presented in our original report (Lehman et al. 2009), the statistically significant reductions occurred from June to September, and seemed more secure at stations B and F than at A, corresponding to larger influence of jurisdiction-influenced catchment area. The mean reductions recorded at A, B, and F from June to September for DP and TP were 23 and 11%, 30 and 21%, and 35 and 23%, respectively. Two-way AOV applied to measured discharge of TP from the AAWWTP detected no statistically significant effects of either Period ( $P = 0.37$ ) or Month ( $P = 0.19$ ), suggesting that reductions observed

at station F could not be ascribed to reduced point-source loading.

### AOV model

AOV detected significant effect of Period for all P variables (Table 2). Period effects for  $\text{NO}_3$  were confined to site F, consistent with observations (Fig. 4). CDOM and SRSi displayed no effect of Period;  $K_{25}$  had a significant Period effect at site F; and pH had a significant Period effect at site A. Comparison with Fig. 3, however, indicates that the statistical effects on  $K_{25}$  and pH were relatively subtle.

Two-way AOV did not detect overall temporal trends in P concentrations postordinance, suggesting that a step change was a reasonable approximation of measured responses in most cases. There were no statistically significant effects of



**Figure 5.**—Concentration differences at station CTL from May to Sep 2008–2010 (EXP) compared to the reference period (REF: 2003–2005). Left: SRP, DP, and TP. Right: NO<sub>3</sub>, CDOM, and SRSi. Error bars represent 2-tailed 90% confidence intervals (no *a priori* expectation of directional change at that site).

Month or Year  $\times$  Month interaction on concentration differences from CTL at any station for any P analyte. Similarly, there were no differences among the 3 years for concentration differences from CTL for TP at any station. However, there was a significant progressive decline in DP (but not SRP or TP) at station A from 2008 to 2010 ( $P = 0.01$ ), such that DP was  $0.04 \mu\text{M}$  greater than CTL in 2008,  $0.04 \mu\text{M}$  less than CTL in 2009, and  $0.09 \mu\text{M}$  less than CTL in 2010 ( $SE = 0.03 \mu\text{M}$  in all cases). To put these declines in context, DP had been  $0.11 (SE = 0.01) \mu\text{M}$  greater than CTL at station A during the reference period. There were also significant progressive declines in both SRP and DP at station B from 2008 to 2010 ( $P < 0.0005$ ). SRP at Station B was equal to CTL in 2008,  $0.07 \mu\text{M}$  less than CTL in 2009, and  $0.14 \mu\text{M}$  less than CTL in 2010 ( $SE = 0.02$  in all cases). DP at Station B was  $0.03 \mu\text{M}$  greater than CTL in 2008,  $0.06 \mu\text{M}$  less than CTL in 2009, and  $0.16 \mu\text{M}$  less

than CTL in 2010 ( $SE = 0.03$  in all cases). SRP and DP had been  $0.01 (SE = 0.01)$  and  $0.14 (SE = 0.03) \mu\text{M}$  greater than CTL at Station B from 2003 to 2005. There were no statistically significant interannual differences from CTL at station F for any P analyte from 2008 to 2010.

## Discussion

After 3 years of postfertilizer ordinance data collection, concentrations of DP and TP in the Huron River, measured at stations within the jurisdiction of the ordinance, are from 11 to 35% lower than expected from preordinance reference levels. The *a priori* prediction for full compliance with the ordinance was 22% reduction (Ferris and Lehman 2008). A summary of key findings from 3 years of investigation follows:

**Table 2.**—Probability levels (*P*) of Month, Period (pre- vs. postordinance), and Month × Period interaction associated with the AOV model for target and nontarget variables. Model included river discharge and CTL site analyte values as covariates to control for upstream watershed-scale variations. NS = not significant. Period effects are emphasized in bold.

Analyte	Site	N	Model <i>r</i> <sup>2</sup>	Month	Period	Month × Period
Target Variables						
SRP	A	134	0.659	0.0002	<b>0.0001</b>	0.06
SRP	B	130	0.568	0.08	<b>0.001</b>	0.02
SRP	F	134	0.286	0.002	<b>0.001</b>	0.04
DP	A	134	0.696	0.0007	<b>0.000001</b>	0.56
DP	B	130	0.703	0.007	<b>0.000001</b>	0.03
DP	F	134	0.402	0.01	<b>0.00001</b>	0.23
TP	A	134	0.667	0.42	<b>0.00009</b>	0.03
TP	B	130	0.610	0.50	<b>0.00001</b>	0.50
TP	F	134	0.337	0.008	<b>0.00002</b>	0.98
Non-Target Variables						
NO <sub>3</sub>	A	134	0.782	0.000004	<b>0.95 NS</b>	0.01
NO <sub>3</sub>	B	130	0.758	0.0001	<b>0.60 NS</b>	0.0002
NO <sub>3</sub>	F	134	0.773	0.00001	<b>0.0005</b>	0.06
CDOM	A	131	0.916	0.04	<b>0.91 NS</b>	0.27
CDOM	B	127	0.857	0.003	<b>0.94 NS</b>	0.43
CDOM	F	131	0.896	0.08	<b>0.75 NS</b>	0.17
SRSi	A	108	0.897	0.03	<b>0.80 NS</b>	0.72
SRSi	B	104	0.877	0.14	<b>0.70 NS</b>	0.09
SRSi	F	108	0.866	0.13	<b>0.72 NS</b>	0.60
K <sub>25</sub>	A	130	0.885	0.64	<b>0.56 NS</b>	0.85
K <sub>25</sub>	B	126	0.793	0.09	<b>0.11 NS</b>	0.11
K <sub>25</sub>	F	130	0.875	0.10	<b>0.002</b>	0.14
pH	A	128	0.599	0.08	<b>0.002</b>	0.22
pH	B	124	0.477	0.17	<b>0.60 NS</b>	0.06
pH	F	128	0.332	0.08	<b>0.12 NS</b>	0.31

- Statistically significant decreased concentrations of SRP, DP, or TP were evident at one or more experimental sites (A, B, and F) from June to September, though not in May (Fig. 4). Mean concentrations of the 3 P analytes were lower than those of the reference period in 43 of 45 comparisons.
- SRP displayed some of the largest mean percentage reductions in concentration, but high variability in SRP concentration limited the power of statistical tests in most cases.
- Significant reductions in DP and TP concentration above and below the AAWWTP outfall, and unchanged point-source loading compared to the reference period, suggest that the statistically detectable effects on river P trace to nonpoint source loading.
- Although the upstream site CTL experienced increased P concentrations, downstream experimental sites exhibited significant reductions from the reference period compared with CTL.
- The nontarget variables showed no evidence of the station-specific response seen for P variables. In most cases they remained unchanged with respect to condi-

tions at CTL. Consistent changes in nutrient concentrations at experimental sites were confined to P. An exception seems to be NO<sub>3</sub> at station F. The AAWWTP is known to be a significant NO<sub>3</sub> source to the river, but NO<sub>3</sub> discharge is not one of the variables measured by facility staff, so we cannot compare pre- and postordinance loading as for P.

## Conclusion

The jurisdiction of Ann Arbor, Michigan, seems to be contributing less nonpoint source P to the Huron River than it did before implementation of its lawn fertilizer ordinance. One may question whether the change is a result of the ordinance or of unrelated human behaviors or climatic trends. Use of our CTL station deviation technique provided some control for these concerns and attempted to account for the effects of the dam removal, but the degree of compliance with the ordinance is unknown. Lawn care services within the jurisdiction no longer include P in their fertilizer preparations, and P-containing lawn products are not available

from area stores without demonstration of specific need. What we do know is that P additions to the river locally have declined, and there is nothing in the evidence to refute the hypothesis that implementation of a fertilizer ordinance has been followed by reduced P in a river that is tributary to Lake Erie, or that similar efforts elsewhere may lead to similar results. We therefore encourage studies of this nature in other watersheds.

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