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Analysis

Nutrient prices and concentrations in Midwestern agricultural watersheds

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ABSTRACT

This paper assesses the impact of nutrient prices on nutrient concentrations in agricultural watersheds. Specifically, we find that the price elasticity of nutrient emissions from agricultural watersheds is −0.17 to −0.34, suggesting that a 10% increase in nitrogen or phosphorus prices faced by farmers would lead to up to a 3.4% reduction in nitrogen or phosphorus emissions from a watershed. While this sounds modest, it is about the same size as estimates of the price elasticity of nutrient demand by farmers, a relationship which also is very inelastic. Our results suggest that when prices for nutrients rise, there is a direct effect on nutrient emissions from watersheds. Given recent concerns about phosphorus in Lake Erie, we assess the potential implications of applying a phosphorus usage fee to reduce phosphorus emissions there. We find that a 25% increase in phosphorus prices would reduce nutrient outputs from the three Lake Erie watersheds we modelled by 6.5%, or 210 t phosphorus per year, and cost about \$6 ha^{-1} yr^{-1}. These costs are similar to estimates of the costs of reducing phosphorus through waste water treatment plants, and less than the costs of other widely used agricultural best management practices like cover crops.

1. Introduction

Despite years of effort at reducing nutrient inputs into streams, the concentration of nutrients in Midwestern American waterways continues to rise (Sprague et al., 2011). While industrial and urban sources contribute significantly to loading, modeling studies estimate that agriculture contributes over 75% of the nitrogen (N) and phosphorus (P) in the Mississippi River Basin (Alexander et al., 2008) and the Great Lakes (Robertson and Saad, 2011). Given the large area of land devoted to growing crops in the United States this is no surprise. These trends are likely to continue; with record high crop prices in recent years, more land has been devoted to the most nutrient intensive crop, corn.

Within agriculture, substantial resources have been devoted to reducing nutrient and soil run-off. Payments to farmers for conservation through the US Farm Bill have increased by 3.2% per year over the past two decades and currently amount to approximately \$5 billion per year, or nearly \$14 per hectare of US farmland per year (Pavelis et al., 2011). In Ohio, the level of payment per hectare of cropland is higher, amounting to around \$20 per hectare of cropland per year.[1] Some of these payments have been devoted to removing land from production, but in recent years, the largest share has been used to fund conservation practices on working farmlands. Programs such as the Natural Resource Conservation Service (NRCS) Environmental Quality Incentive Program (EQIP), for example, fund nutrient management plans and best management practices that are designed to reduce nutrient runoff. Although these programs have distributed significant resources to landowners, society has not yet achieved a decisive payoff in terms of reduced nutrient exports from agricultural watersheds.

The rationale for current agricultural conservation policy is that society funds the installation of best management practices that will, in theory at least, reduce nutrient emissions (also referred to as outputs or concentrations) in the watershed through avoidance, trapping, or controlling. For example, current conservation programs assume that nutrient loss can be avoided or controlled through the use of residue and reduced tillage management, nutrient management, cover crops, or changes in crop rotations, among other practices. It is recognized that agricultural systems are leaky, and practices touted to trap nutrients prior to their entry into a waterway include buffer strips, filter strips, and constructed wetlands. All of these practices allow farmers to continue applying nutrients for optimal economic performance on their fields, with the belief that the component that becomes an externality will be captured. While these programs and practices do a reasonable job of addressing sediment related water quality issues, their ability

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[1] From 2008 to 2012, the federal government spent an average of \$85 million on conservation programs in Ohio. With around 4 million ha in crops, this amounts to \$20 ha^{-1} yr^{-1}.

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to reduce soluble nutrient loss, and in particular soluble phosphorus, has not been proven. Current losses of soluble phosphorus in the Lake Erie Basin of Ohio (approximately 0.6 kg ha^{-1}) are less than 5% of that which is applied on an annual basis and has a market value between \$2.50 and \$5 ha^{-1}.

Best management practices have limited potential because they put the focus on implementing technology, but not reducing nutrient input. Additionally, many of the practices that are recommended to avoid, control, and trap nutrients are based on theory and have generally only been shown to be effective at a small plot scale. Studies aimed at demonstrating their effectiveness at the watershed level have yielded mixed results (Walker and Graczyk, 1993; Makarewicz et al., 2009; Inamdar et al., 2001). One study, Richards et al. (2009), illustrates a reduction in suspended sediments and particulate phosphorus over several decades in two Lake Erie watersheds in Ohio. They suggest that the likely explanation for this reduction is the widespread adoption of conservation tillage. The argument by Richards et al. (2009) is plausible for suspended sediments, particularly given the strong economic incentives farmers have had to reduce labor costs by reducing the amount of tilling they do.

There has been less research investigating the link between farm nutrient inputs and agricultural watershed nutrient outputs, particularly in large watersheds (e.g., >100,000 ha). This is not surprising because it is very difficult to determine how much nutrient is applied by farmers in specific watersheds. The USDA National Agricultural Statistics Service (USDA-NASS, 2012) surveys farmers and determines their input use based on their responses, but this survey no longer occurs every year, and, in any event, the results are not specific to watersheds. Data on sales of nutrients can help, but nutrient sales in an area need not correspond to nutrient application since they can be stored from year to year and can be purchased and applied in different regions. For example, Bruuselma et al. (2011) use data from nutrient sales to show that phosphorus inputs in the Midwest have declined as phosphorus exports have risen with crop yields. While these results are telling, they do not allow us to determine a relationship between inputs and outputs in farmed watersheds.

We argue in this paper that nutrient prices can be used as a proxy for nutrient inputs and we establish a relationship between nutrient prices and nutrient outputs in agricultural watersheds. Using historical data from five Midwestern watersheds, we show this relationship both for phosphorus and for nitrogen, the two main nutrients emitted from agricultural watersheds. In addition to linking nutrient input prices to nutrient emissions from watersheds, we also show how other factors such as crop prices and, in the case of phosphorus, historical emissions influence nutrient emissions.

To our knowledge, this is the first study to establish an empirical link between nutrient input prices and nutrient outputs from agricultural watersheds. Given that the farmer demand function for nutrients is downward sloping, higher nutrient prices should lead to lower nutrient inputs and consequently emissions from farm fields. Economic studies have shown that the demand function for nutrients does indeed slope downwards, with input price elasticity for nutrients in the US around −0.25 (Gunjal et al., 1980; Denbaly and Vroomen, 1993). Because much of the fertilizer used by farmers ultimately is taken up by crops and harvested, attached to soil, or taken up by other plants in the ecosystem, it is not clear whether this same elasticity estimate would apply to nutrient outputs from agricultural watersheds. The results in this paper show that the relationship between nutrient prices and nutrient outputs from watersheds is indeed negative, with an elasticity parameter of similar size as the input demand function.

Using our estimated model, we develop a policy analysis that examines the costs of a phosphorus usage fee to reduce nutrients in the Lake Erie Basin. In recent years, policy makers have focused on phosphorus as a key ingredient causing recent harmful algal blooms in the lake (see Ohio Phosphorus Task Force, 2013). We present estimates showing that a 25% phosphorus usage fee could reduce phosphorus loadings to Lake Erie by 6.5%. The costs of the policy are \$6 per hectare to reduce phosphorus (P). Given the relationship between nutrient inputs and nutrient outputs, we estimate that each 10.6 ton reduction in phosphorus inputs leads to a 1 ton reduction in phosphorus outputs.

2. Model and Data

This analysis develops a model of nutrient concentrations in agricultural watersheds. We model nutrient concentrations in agricultural watersheds as a function of water flow, nutrient prices, crop prices and crop nutrient uptake. The relationship between nutrient concentrations and water flow is well established (see for example, Cohn et al., 1992; Richards et al., 2009; Sprague et al., 2011), although the relationship differs depending on a number of different factors, including the nutrient type. Nutrient inputs used by farmers and nutrient uptake also should play an important role in nutrient concentrations in rivers. Unfortunately, the annual use of nitrogen and phosphorus by farmers is not well known. Survey data from the National Agricultural Statistics Service provides estimates of nutrient uptake by major crops for each state, but these surveys are state-wide and they are no longer collected annually.

To address this shortcoming, we utilize nutrient prices as a proxy for nutrient inputs in farming. Within a given watershed, the quantity of nutrients used by farmers will have an inverse relationship with nutrient prices due to the farmer's demand for nutrients. With higher nutrient prices, farmers will demand fewer nutrients (and vice-versa). We suspect that this same relationship should hold for nutrient outputs from watersheds, and empirically test whether it does.

Other ecological, hydrological, and management factors will affect nutrient outputs as well. The first variable we include is water flow (e.g., Cohn et al., 1992; Sprague et al., 2011). While it is widely accepted that including flow is important when predicting nutrient concentrations, Cohn et al. (1992) found that flow could be positively or negatively related to nutrient concentration, depending on the watershed and nutrient. A second component that affects nutrient outputs from watersheds is nutrient uptake and export via crop harvesting. In good years, crops will use more nutrients, which ultimately will be exported from the watershed in what is harvested. Higher nutrient uptake and export via crops should result in lower nutrient exports from the watershed through streams.

The types of crops grown in a watershed will also influence nutrient exports through streams by the different types of management undertaken when various crops are grown. For instance, many landowners and managers employ conservation tillage techniques when planting soybeans, but they often till their fields when planting corn. For phosphorus, which is emitted in both a soluble form and a form attached to soil sediments, we should see an increase in total phosphorus emissions when more corn is planted, due to an increase in phosphorus applications and an increase in tillage. Higher corn prices should also increase nitrogen emissions because nitrogen applications are highest with corn.

A final issue to consider in the phosphorus model is historical phosphorus applications. Because phosphorus attaches to soil particles and can remain in soils for long time periods, it is important to account for historical phosphorus emissions when measuring current emissions. One way to do this in a regression model is to include a lagged dependent variable. Thus, for the phosphorus model we include the one year lag of phosphorus concentrations.

To test our model empirically, we pool data for five Midwestern watersheds (Table 1) and regress nutrient concentrations for each watershed against a set of explanatory variables. The watersheds in this analysis cover 3.9 million ha in Ohio, Indiana and Michigan. Over 85% of land in the watersheds is used for agriculture. The Maumee, Sandusky and Raisin watersheds flow into Lake Erie while the Scioto and Great Miami watersheds flow into the Ohio River. For the analysis, a fixed effects panel data model is estimated, whereby the data for all

Table 1
Total area above stream gaging stations, proportion agricultural area, years in sample, flow characteristics and average concentrations for watersheds in our sample.

	Total area	Agricultural area	Years	Average annual flow	Average annual total N	Average annual total P
	ha	ha		$m^3 s^{-1}$	$mg L^{-1}$	$mg L^{-1}$
Maumee[a]	1,640,162	1,474,506	1976–2011	156	7.54	0.40
Sandusky[a]	324,664	273,042	1976–2011	34	7.19	0.41
Raisin	269,992	213,294	1982–2011	21	5.33	0.18
Scioto[a]	998,607	800,883	1996–2011	114	4.77	0.32
Great Miami	695,709	571,177	1996–2011	86	5.38	0.38

[a] For the Maumee River watershed data for 1979 and 1980 are missing; for the Sandusky River watershed data for 1980 are missing; and for the Scioto River watershed data for 2010 are missing.

of the watersheds are pooled together, and dummy variables (fixed effects) are used to identify separate means for each watershed. The fixed effects take on the value of 1 for the given watershed, or 0 otherwise. The fixed effects control for factors in each watershed that differ across the watersheds, but that are constant over time, such as the overall size of each watershed, baseline levels of nutrients, and other factors.

The two models we estimate are:

$$\ln(TNW_t) = f(\ln(CFS_t), \ln(PN_t), \ln(PC_t), \ln(NUSE_t), Maumee_t, Sandusky_t, Raisin_t, Scioto_t) \quad (1)$$

$$\ln(TPW_t) = f(\ln(CFS_t), \ln(TPW_{t-1}), \ln(PP_t), \ln(PC_t), \ln(PUSE_t), Maumee_t, Sandusky_t, Raisin_t, Scioto_t). \quad (2)$$

In Eq. (1), annual nitrogen concentration in the watersheds (TNW_t) is a function of flow rate (CFS_t), nutrient or crop prices (PN_t for nitrogen; PC_t for corn), the amount of nitrogen used by crops in the watersheds ($NUSE_t$), and a set of watershed specific fixed effects ($Maumee_t$, $Sandusky_t$, $Raisin_t$, $Scioto_t$). Price variables will be the same in each watershed, while the other variables differ by watershed. The phosphorus model is presented in Eq. (2), with annual phosphorus concentration (TPW_t) as a function of flow rates (CFS_t), lagged phosphorus concentrations (TPW_{t-1}), prices (PP_t for phosphorus; PC_t for corn), phosphorus used by crops ($PUSE_t$), and watershed specific fixed effects. The model for phosphorus includes lagged phosphorus concentrations (TPW_{t-1}) because phosphorus attaches to soil and is maintained in the soil profile for longer time periods. Within the soil profile, phosphorus acts like a stock. It receives annual additions from fertilizer use; stores phosphorus while it is attached to soil; is used by plants in a soluble form; is emitted into the environment as either a soluble or attached form; and can be carried over to the next year. There is also evidence that widely adopted measures like conservation tillage have reduced sediment exports and the phosphorus attached to soil in these watersheds (Richards et al., 2009).

The left hand side in both equations is the time-weighted annual concentration of total nitrogen (TNW_t) and total phosphorus (TPW_t). These variables are calculated from sub-daily observations obtained from the National Center for Water Quality Research (Heidelberg College, 2012). Data for the Maumee and Sandusky have been collected since 1976, while data for the Raisin, Scioto and Great Miami have been collected since the 1982, 1996 and 1996 respectively (Table 1). In addition, we lack data for the Maumee in 1979 and 1980, the Sandusky in 1980, and the Scioto in 2010. This leads to an unbalanced panel both because of missing data and different initial data collection years. We argue that the missing data, which is driven by funding issues, is not due to a selection process that is correlated with the error term in our model, so it does not affect our results. The second reason for our unbalanced panel, different data series lengths for different watersheds, may be correlated with our errors and may lead to biased estimates. As discussed below, we conduct regressions with balanced subsets of the data to test the robustness of our findings with the full dataset.

One data collection process that could affect the results is that data are collected on a daily basis for most days, but multiple observations are collected on days with rain events. This means that our data is more intensively sampled on rainy days. To handle this, we calculate time weighted annual values for concentration and flow measurements. In order to calculate the time weighted average annual concentrations of phosphorus (TPW) we use the sample time interval in days (S), the flow measurement in cubic feet per second (CFS) and the total phosphorus of the sample in $mg L^{-1}$ (TP). The annual weighted average phosphorus concentration for each watershed is given by:

$$TPW_{t,i} = \frac{\sum_{t=1}^{T} S_{t,i} CFS_{t,i} TP_{t,i}}{\sum_{t=1}^{T} S_{t,i} CFS_{t,i}}. \quad (3)$$

Similarly, our flow rate variable (CFS) is calculated as the denominator of Eq. (3). Total nitrogen (TN) of the sample is the sum of the concentration of oxidized nitrogen compounds in the sample (NO_{23}), and the total Kjeldahl nitrogen in the sample (TKN). To determine weighted average concentrations of total nitrogen (TNW) we use the sample time interval in days (S), the flow measurement in cubic feet per second (CFS) and the total nitrogen of the sample in $mg L^{-1}$ (TN). The weighted average nitrogen level for watershed i is given by:

$$TNW_{t,i} = \frac{\sum_{t=1}^{T} S_{t,i} CFS_{t,i} TN_{t,i}}{\sum_{t=1}^{T} S_{t,i} CFS_{t,i}}. \quad (4)$$

Each observation of the dependent variable is thus constructed using numerous measurements. Within the data, the number of measurements available to calculate the dependent variable varies. For example, TKN was missing for many samples in the 1970s for the Maumee and Sandusky Rivers. When TKN was missing from a sample, we did not include the sample in the calculation of annual concentration. Because this reduces the number of measurements available for calculating the average concentration variables of interest, we include regression models that are weighted by the time period of sampled observation. Thus, years with large numbers of missing measurements and are constructed with fewer samples were weighted less heavily in the analysis. For the analysis, we used the United States Geological Survey (USGS) water years to calculate average annual concentrations. The USGS water years run from October 1 to September 30. We used the year in which October 1 fell to denote the particular year.

As discussed above, we hypothesize that the relationship between nutrient prices and nutrient concentrations in watersheds should be negative. This relationship follows demand theory, whereby rising prices for nutrients will induce farmers to purchase and use less phosphorus or nitrogen. Our estimations are conducted in several watersheds in Ohio, and the farmers in these watersheds are assumed to be price takers for nutrient inputs, such that nutrient input prices are exogenous to the farmer's decisions. This makes sense, as it is highly unlikely that large changes in nutrient inputs in these watersheds would dramatically alter nutrient input prices.

The prices used in the analysis are April prices, while the nutrient concentration variables and the flow variables are for the following water year (i.e., starting in October). We assume that fertilizer added in the growing season will not be used completely by crops during the year, and a portion not used is emitted during the fall, winter, and following spring. As well, decaying plant material will contribute phosphorus and nitrogen to the watershed. Fertilizer prices were obtained from the US Department of Agriculture Economic Research Service (USDA-ERS, 2012). For nitrogen, we used the price of anhydrous ammonia for spring of each calendar year. For phosphorus, we calculated the price of P_2O_5 based on the average of the price of P_2O_5 contained in diammonium phosphate and triple-superphosphate.

Corn prices are included in the models to account for the impact on nutrient outputs due to changes in crop rotations driven by output prices. Conservation tillage has been widely adopted in the watershed, but is mainly used when shifting from corn to soybeans. When land is converting into corn, farmers typically till the soil, which will release phosphorus attached to sediment during later precipitation events. Higher corn prices will encourage more corn planting, and more tillage, and they should be positively related to phosphorus output. Corn prices were obtained from Farmdoc website at the University of Illinois (Farmdoc, 2012). All prices were deflated to a base year of 1982 using the all commodity producer price index.

The nitrogen and phosphorus used by corn, soybeans, and wheat in each watershed controls for broad changes in crop types, yields, and other factors that influence uptake of nitrogen and phosphorus by crops over time. Their predicted sign is uncertain. Higher nutrient uptake by crops could reduce nutrient exports by river systems if the nutrients are exported from the system in marketed products. Higher nutrient uptake, however, could also increase nutrient exports by river systems due to the decay of remaining plant material. Nitrogen and phosphorus use by crops was calculated using data from the USDA-NASS (2012) "Quickstats" database. We use county level data on crop area and crop yields to calculate nutrient uptake by country. These estimates are then linked to watersheds by using the information from counties that overlay the watersheds, but that are upstream from the sampling location.[2] We count uptake only for the proportion of each county that is within each watershed, and we consider only corn, soybean and wheat, since these are the primary crops in these watersheds.

The formula used to calculate crop use is:

$$\text{Crop use}_t = \sum_c \sum_i \alpha^c Y_i^c A_i^c \qquad (5)$$

where Y_i^c is the yield for crop "c" in county "i", and A_i^c is the area for crop "c" in county "i". The crop use coefficient, α^c, is the amount of the nutrient in the harvested proportion of the crop. For corn, the crop use coefficient is 2.9 kg t^{-1} for phosphorus, and 17.9 kg t^{-1} for nitrogen. For soybeans, it is 4.6 kg t^{-1} for phosphorus, and 17.9 kg t^{-1} for nitrogen; and for wheat it is 5.8 kg t^{-1} for phosphorus, and 75.0 kg t^{-1} for nitrogen. Crop use coefficients for phosphorus were derived from Bruuselma et al. (2011), while nitrogen crop use coefficients were derived from Vitosh et al. (1995). Crop yields and crop areas change annually although the time subscript is suppressed.

The fixed effects account for watershed specific factors that are unobservable, but that do not change over time. These include things like soil type, which stays constant over time but will differ by watershed. The fixed effects take on the value 1 for observations in the watershed and 0 otherwise. The base watershed is the Great Miami, so a dummy variable for that watershed is excluded from our regression.

3. Results

For the nitrogen model, the parameter on nitrogen price is negative and significant in both the un-weighted and weighted models (Table 2a). The parameter, however, is smaller in the weighted model. Because the model is estimated in log–log form, the parameter estimate can be interpreted as an elasticity, such that the price elasticity of nitrogen export in these watersheds ranges from −0.17 to −0.28. This implies that each 10% increase in prices will reduce nitrogen outputs in the watershed by 1.7 to 2.8%. The elasticity estimate of nitrogen output from a watershed is consistent with the price elasticity of demand as an input into production found in the literature. Put another way, our negative sign is consistent with the negative relationship between nutrient prices and the quantity of nutrients used by farmers found in Denbaly and Vroomen (1993). The size of the elasticity is about the same as their estimated elasticity.

The parameter on the flow variable is negative indicating that higher flows lead to lower average flow-weighted concentrations of nitrogen. Corn prices have little effect on nitrogen outputs. Crop nitrogen use also has minimal effect. This is perhaps surprising, but as suggested above, the parameter on crop nitrogen could have either sign.

In the phosphorus model (Table 2b), the parameter estimate for the price of phosphorus is negative and significant in both the un-weighted and weighted models. The price elasticity of phosphorus ranges from −0.17 to −0.24, indicating that a 10% increase in phosphorus prices will reduce phosphorus outputs in the watershed by 1.7% to 2.4%. This

Table 2a
Nitrogen regression model results. All variables have been transformed by the natural logarithm. The dependent variable is LN(TNW).

Variable	N model		N model	
	Unweighted		Weighted by time	
	(n = 128)		(n = 128)	
	Parameter	t-Stat	Parameter	t-Stat
Constant	3.682***	2.781	4.518***	3.750
LN(CFS)	−0.113*	−1.930	−0.243***	−4.190
LN(PN)	−0.277**	−3.064	−0.170**	−2.120
LN(PC)	−0.080	−0.841	−0.060	−0.690
LN(NUSE)	0.044	0.340	0.010	0.080
Maumee (1,0)	0.357**	3.054	0.481***	4.530
Sandusky (1,0)	0.192*	1.791	0.079	0.810
Raisin (1,0)	−0.117	−1.479	−0.325	−1.510
Scioto (1,0)	−0.138	−0.592	−0.054	−0.760
R2	0.470		0.530	

*** Significant at 0.001 level.
** Significant at 0.05 level.
* Significant at 0.10 level.

Table 2b
Phosphorus regression model results. All variables have been transformed by the natural logarithm. The dependent variable is LN(TPW).

Variable	P		P	
	Unweighted		Weighted by time	
	(n = 125)		(n = 125)	
	Parameter	t-Stat	Parameter	t-Stat
Constant	−1.961**	−2.275	−1.903**	−2.330
LN(CFS)	0.200***	5.662	0.177***	3.710
$LN(TPW_{t-1})$	0.270***	3.784	0.258***	3.760
LN(PP)	−0.235**	−2.777	−0.167**	−2.090
LN(PC)	0.419***	4.785	0.363***	4.260
LN(PUSE)	0.089	0.940	0.061	0.670
Maumee (1,0)	−0.198**	−2.262	−0.159*	−1.940
Sandusky (1,0)	0.197**	2.453	0.179**	2.300
Rasin (1,0)	−0.193	−1.087	−0.250	−1.500
Scioto (1,0)	−0.178**	−2.677	−0.168**	−2.700
R2	0.830		0.830	

*** Significant at 0.001 level.
** Significant at 0.05 level.
* Significant at 0.10 level.

[2] This yields a necessarily rough approximation of crop area, as county boundaries do not perfectly mirror watershed boundaries.

is about the same size as the estimate in the nitrogen model and it is similarly consistent with estimates for input demand by Denbaly and Vroomen (1993).

Corn prices have a strong positive effect on phosphorus outputs. This likely occurs because higher corn prices induce more acres into corn production. As a result, there is increased tillage and more phosphorus input into the watershed, both of which will increase phosphorus outputs. Some authors suggest that more tillage can reduce soluble phosphorus (Zhao et al., 2001; Gilley et al., 2007a, 2007b), but the overall effect of higher corn prices on total phosphorus in our model is positive. The parameter on the flow variable is positive suggesting that higher flows increase phosphorus concentrations, which is explained by the phosphorus attached to eroded sediments. As with the nitrogen model, the amount of phosphorus used by crops does not significantly influence phosphorus outputs from these watersheds.

The sign on lagged phosphorus concentration is positive and significant, so that higher concentrations of phosphorus in the previous year will lead to increased concentrations in the current year, and vice-versa. Given that phosphorus builds up in soils and is released over time both on soil particles and as soluble P, a positive lagged process makes sense. For example, higher phosphorus concentrations in soils likely contribute to higher phosphorus concentrations in rivers. Since it takes long time periods to increase or decrease soil stocks of phosphorus (see Bruuselma et al., 2011), higher (lower) emissions in any given year will likely be correlated with higher (lower) emissions the next year. This relationship would be strengthened with an increase in conservation tillage and other practices that would tend to hold soil particles and phosphorus in soils for longer periods of time.

One of the more important questions about this analysis is whether the effect of prices on nutrient concentrations is identified. Economically, the relationship between prices and quantities can be positive or negative depending on whether one is estimating a demand or supply relationship. In our proposed models of nitrogen and P, a demand relationship is estimated. It is assumed that nutrient supplies in these watersheds in the short-run are insensitive to price changes. Farmers in these watersheds are a small part of the overall nutrient market, and cannot influence the price of nutrients through their purchases. In addition, we are using prices that are lagged by at least 6 months relative to the starting time period for our water year measurements. While it might be feasible for good or bad farming weather (which would likely be correlated with river flows and nutrient concentrations) to influence future prices of nutrients through market mechanisms, current weather does not influence past prices of nutrients.

As noted in Table 1 and above, our dataset is an unbalanced panel, with several years missing from the datasets for the Maumee, Sandusky and Scioto watersheds, and shorter observation periods for the Raisin, Scioto, and Great Miami Rivers. To test whether the unbalanced nature of the data has an effect on our results, we conduct several different regressions and present them in Appendix A. These alternative regressions consider datasets with observations for only the period 1996–2009 (a fully balanced panel for all of the watersheds), as well as models only for Lake Erie over the period 1982–2011 (also a balanced panel) and models for specific watersheds.

The additional regressions in Appendix A illustrate that the basic model is robust, although the parameter on the fertilizer price becomes insignificantly different from 0 in some models, due to fewer observations. The model for nitrogen suggests that watersheds in the Ohio River Basin are more sensitive to changes in nitrogen fertilizer applications and watersheds in the Lake Erie Basin are more sensitive to changes in phosphorus fertilizer applications. The parameter on fertilizer price has the least significance in the regression for the Great Miami River Basin. As discussed in Appendix A, this results from the buffering impact of the large aquifer system in that region.

Clearly weather and annual water flow influence nutrient emissions in agricultural watersheds, but these results illustrate that variation in nutrient outflows in agricultural watersheds is also directly related to annual variation in nutrient prices and nutrient inputs on farm fields. The effect may be mediated by other intervening influences, such as groundwater, but the results hold over a wide range of watersheds that are predominately agricultural. Higher prices for nutrients invite nutrient conservation and less nutrient runoff, while lower prices invite additional use and additional emissions into watersheds. Based on the full set of results in Appendix A, the price elasticity of nutrient outputs from these watersheds ranges from -0.17 to -0.34. The estimates for nitrogen prices are larger in general, suggesting that nitrogen outputs from watersheds are more sensitive to nitrogen inputs by farmers. When comparing our results across watersheds, the results for nitrogen prices indicate a larger effect in the Ohio River Basin than in the Lake Erie Basin.

4. Policy Analysis

The results of this model can be used to assess the implications of a policy mechanism aimed at reducing nutrient exports from watersheds. The Ohio Phosphorus Task Force (2013) suggests that a 40% reduction in total phosphorus would significantly reduce the likelihood of harmful algal blooms. For the policy analysis, we focus on the Lake Erie Basin and use the phosphorus price elasticity estimate for just the Lake Erie watersheds, which ranges from -0.23 to -0.29 across the weighted and un-weighted models in Appendix A. Given an average elasticity of -0.26, it would take a usage fee of over 150% to achieve a 40% reduction in total phosphorus ($-0.40/-0.26 = +1.5$ or 150%). We do not suspect that this level of usage fee is politically feasible, particularly given that Ohio's Phosphorus Task Force report did not even consider a usage fee. However, a more modest 25% usage fee could be considered to encourage more widespread adoption of the 4-Rs (right rate, right time, right place, right source). By increasing fertilizer prices with a usage fee, farmers would have stronger incentives to use the “right” rate at the right time and in the right place.

Based on our estimates, a 25% usage fee would achieve a 6.5% reduction in phosphorus entering Lake Erie ($0.25 * 0.26 = 0.065$). To calculate the predicted reduction in phosphorus emissions, we use the elasticity estimates from our model to determine the change in phosphorus concentrations for the three watersheds in our data that drain to Lake Erie. We focus only on phosphorus, given the importance of this pollutant, although the same methods could be used to estimate the costs of a reduction in N. Using our elasticity estimates, we calculate change in annual nutrient loading that would occur with the nutrient fee by comparing phosphorus concentrations with and without the fee, and multiplying those times average flows over a given time period.

To estimate the potential costs of this proposed usage fee, we calculate the reduction in nutrient inputs under the usage fee for the three Lake Erie watersheds considered in our study (the Raisin, Maumee and Sandusky Rivers). The analysis is conducted using data for the period 2007–2011. When farmers face a higher price for P, they will consume less of it. Based on Denbaly and Vroomen (1993), the farmers' price elasticity of phosphorus demand is -0.25, suggesting that a 25% increase in the phosphorus price will reduce phosphorus use by 6.25%. We cannot know the change in phosphorus inputs exactly because we do not know the farmer's initial inputs exactly, but we can estimate them based on estimates of the area of various crops planted and survey data that provides information on current phosphorus application rates (see USDA-ERS, 2012).

To determine the area of the three main crops (corn, soybeans and wheat) in each watershed, we start by calculating the area of the three main crops for the counties that overlay each watershed, using data from USDA-NASS (2012). We then apply this estimate of the proportion of land in each crop to the total area of land in major crops in the three

Table 3
Implications of a 25% increase in nutrient prices on nutrient inputs and outputs in three Lake Erie watersheds, Maumee, Sandusky and Raisin.

	Corn	Soybeans	Wheat	Total
Crop Area (ha)[a]	686,295	980,421	294,126	1,960,842
Baseline				
Annual phosphorus application (kg P ha^{-1})[b]	34.1	27.2	27.9	18.2
Hectares receiving phosphorus (%)[b]	100%	22%	78%	58%
Annual application (1000 kg)	23,385	5,869	6,411	35,665
Total cost of phosphorus (millions $)[c]	$45.8	$11.5	$12.6	$69.9
25% phosphorus usage fee				
Annual phosphorus application (kg P ha^{-1})[d]	31.9	25.5	26.2	17.1
Hectares receiving phosphorus (%)	100%	22%	78%	58%
Annual application (1000 kg)	21,923	5,502	6,011	33,436
Total cost of phosphorus (millions $)	$53.7	$13.5	$14.7	$81.9
Reduction in phosphorus application (1000 kg)	1,462	367	401	2,229
Increase in phosphorus cost (millions $)	$7.87	$1.98	$2.16	$12.01
Increase in phosphorus cost ($ ha^{-1})	$11.47	$2.02	$7.34	$6.12
Baseline phosphorus emission from watersheds (1000 kg)				3,236
Reduction in phosphorus output from watersheds with 25% usage fee[e]				210
Reduction in phosphorus application in tons to achieve a 1 ton reduction in outputs				10.6

[a] Total farmland hectares is above the gaging stations where water quality measurements are taken; area of each crop is proportional to area in counties that overlay the watersheds, using data from USDA-NASS (2012) and USDA-ERS (2012).
[b] Nutrient inputs differ by crop and by nutrient and are obtained from USDA-ERS (2012).
[c] Initial phosphorus price is $1.96 per kg elemental P, or $350 per ton triple superphosphate (45% P_2O_5).
[d] Change in nutrient input is total input times −0.0625.
[e] Change in output is calculated by model.

watersheds (Table 3). Baseline phosphorus nutrient uptake is estimated with data from USDA-ERS (2012) on input levels for the three main crops, and the proportion of cropland hectares that receive nutrients (Table 3). The costs of nutrient applications in the baseline are estimated, assuming a price of $350 per ton of triple superphosphate.[3]

When the 25% usage fee is implemented, two adjustments will occur: Farmers will pay higher prices for phosphorus, and they will apply less. Based on the elasticity estimates we are using, phosphorus inputs will fall 2229 t yr^{-1} (1.1 kg ha^{-1} yr^{-1}). With a 25% higher price for phosphorus, but 6.25% less phosphorus used, the net cost to farmers is $12 million, or $6 per ha. The reduction in nutrient output from the three watersheds is estimated to be 210 t. Given the reduction in inputs we have calculated, this suggests that each 10.6 kg reduction in phosphorus use translates into 1 kg reduction in phosphorus export.

The proposed 25% usage fee provides a 5% reduction in phosphorus loadings, although it would cost farmers around $6 per hectare per year. This is a relatively small amount, however, compared to the current payments for federally funded conservation programs, which as discussed above cost society $20 ha^{-1} yr^{-1} in Ohio. One of the limitations of our analysis thus far is that we have not accounted for potential reductions in crop yields. A recent study by Elobeid et al. (2011) suggests that the yield and price effects of nutrient taxes will be quite modest, although they focus on nitrogen rather than phosphorus. Webb et al. (1992) directly examine the effects of phosphorus applications on corn and soybean yields. Averaged across a range of initial soil phosphorus tests, they find that reducing phosphorus inputs from 33 kg ha^{-1} to 22 kg ha^{-1} reduces corn yields by 0.3% and it does not reduce soybean yields. Reducing phosphorus inputs to 11 kg ha^{-1} reduces corn yields by only 2.2% and it does not reduce soybean yields. Further reducing phosphorus inputs to 0 reduces corn yields by 5.6% and soybean yields by 5.3%. Our study examines a reduction in phosphorus used by farmers of only 6.25%, which would appear to have a negligible effect on corn and soybean yields.

It is challenging to quantify the gains one would expect from this relatively modest reduction in P. Anecdotally, we would note that a 210 t yr^{-1} reduction in total phosphorus would likely lead to substantial decreases in the size of harmful algal blooms (HABs) in Lake Erie. Using observations from Stumpf et al. (2012), 210 tons is approximately the difference in spring phosphorus loading between the years 2008 (which experienced an HAB of 1047 km^2) and 2007 (which experienced an HAB of 288 km^2). This is likely an overestimation of the benefits from a 6.5% reduction in phosphorus emissions, since the Stumpf et al. (2012) calculations focus on reducing phosphorus in spring. While it is unlikely that all nutrient reductions will occur in a single season, the fact that as the 4Rs achieves more widespread use, an increasing share of fertilizers will be applied to fields during the spring suggests that a usage fee on fertilizer will disproportionately reduce spring runoff.

These results can be translated directly into meaningful actions that individual landowners can undertake. For instance, on a representative 500 hectare farm that uses an average of 9000 kg of phosphorus every year, reducing inputs by 1000 kg (2 kg ha^{-1}) will reduce emissions to Lake Erie by 94 kg. Clearly, society needs to undertake additional actions beyond simply reducing phosphorus emissions to cut the amount that ultimately enters our streams and lakes. These additional actions include more widespread adoption of the 4-Rs, controlled drainage and other techniques that are emerging.

Another issue to consider is how these estimates compare to cost estimates for removing phosphorus from waste water treatment plants. Using a recent study conducted for the Ohio Environmental Protection Agency (2013), the costs of reducing phosphorus emissions from waste water treatment plants range from $17 to $90 kg^{-1}, depending on the technology. Alternatively, estimates from Sano et al. (2005) suggest that costs range from $24 to over $1000 per kg. It is difficult to compare this directly to our estimates for nonpoint sources, given that the point source reductions are estimated at the pipe, rather than at the outlet of the watershed. Our reduction in phosphorus is estimated at the outlet of the watershed into the lake. Based on our calculations, the average cost of reducing phosphorus at the outlet from the watershed with nonpoint sources is $57 per kg. This suggests that the costs

[3] Triple superphosphate prices averaged around $600 per ton from 2007 to 2011, although they have declined since then to around $350 per ton over the past year. To make a more relevant comparison to today's costs of farming, we use the $350 per ton as our base cost.

of reducing phosphorus with nonpoint sources may be similar to the costs of reducing phosphorus with point sources.

A final comparison is to consider the potential for agricultural best management practices to be used. One of the most popular approaches is to use cover crops. In Ohio, the Natural Resources Conservation Service pays landowners around \$100 ha^{-1} to plant cover crops. Current phosphorus loads based on our calculations amount to around 1.65 kg ha^{-1} yr^{-1}. If we assume that cover crops are on the surface during the fall, winter, and spring months when 80% of this is emitted, they could potentially reduce 1.3 kg ha^{-1} yr^{-1}. Ohio's Phosphorus Task Force (2013) assumes that these will be 39% effective on average, implying they could reduce phosphorus by 0.51 kg ha^{-1} yr^{-1}. This implies an average cost of \$196 kg^{-1} to reduce phosphorus via cover crops.

5. Conclusion and Discussion

This study examines the relationship between key economic variables and nutrient outputs in agricultural watersheds. We use data from the National Center for Water Quality Research (Heidelberg College, 2012) to test whether the price of nitrogen and the price of phosphorus affect the annual flow-weighted concentration of nitrogen and phosphorus in five agricultural watersheds in Ohio and Michigan. We propose that the flow weighted concentrations of nitrogen and phosphorus are a function of flow and nutrient input prices, as well as crop prices and nitrogen or phosphorus use by crops. Our models show that the elasticity of nutrient concentrations in agricultural watersheds ranges from −0.17 to −0.34 for nitrogen and phosphorus. Our results suggest that the effects for nitrogen are slightly larger, although the results are watershed dependent. The estimates, however, are similar to estimates of the price elasticity of demand for nutrients used in farming obtained from the literature. Our results imply that a 10% increase in nitrogen or phosphorus prices would lead to a 2 to 3.5% reduction in nitrogen or phosphorus applications and a similar 2 to 3.5% reduction in nutrient outflows from the watershed.

Given recent concerns over harmful algal blooms and their link to phosphorus emissions (Ohio Phosphorus Task Force, 2013), we use the model to assess the implications of a 25% usage fee on phosphorus imposed on the Lake Erie watersheds in our analysis. If the usage fee had been imposed in 2007, we calculate that phosphorus delivery would have declined by 210 t yr^{-1} from 2007 to 2011. The projected reductions in phosphorus delivery (210 t yr^{-1}) are equivalent to 20% of the total reduction (1030 t yr^{-1}) recommended by the Ohio Phosphorus Task Force (2013) to address harmful algal blooms in Western Lake Erie Basin. When compared to an estimate of the impacts on phosphorus inputs in the watersheds, these results suggest that each 10.6 kg reduction in phosphorus use by farmers translates into 1 kg reduction in phosphorus export. The costs of this policy would have been \$12 million yr^{-1}, or around \$6 per ha yr^{-1}.

Aside from the effects of nutrient prices, we find that corn prices have no significant effect on nitrogen concentrations, while they have a strong positive effect on phosphorus concentrations. The explanation in the phosphorus model is that the higher corn prices increase the area of land in corn and the intensity of tilling. More corn area increases the amount of phosphorus input in the watershed, and more tilling leads to more erosion and hence more phosphorus output. The results for nitrogen and phosphorus use are somewhat surprising, given that one might expect higher nitrogen and phosphorus export through crops would lower outputs into the watershed, but higher nitrogen and phosphorus use by crops also likely means larger emissions in the future through decomposition from non-exported plant material.

The results of this analysis imply that traditional economic instruments for environmental pollution control, like a nitrogen or phosphorus usage fee, would be effective for reducing nitrogen or phosphorus outputs from watersheds. Society can continue the practice of subsidizing pollution reductions on farms. These results suggest, however, that these payments may be made cost effective by focusing payments directly on nutrient input reductions in combination with avoiding, trapping, or controlling nutrient loss. Of course, to avoid nutrient leakage across watersheds, these policies must be implemented regionally or even nationally. Lastly, this study focuses on the cost side of nutrient reduction or control policies. A complete benefit cost analysis, one that estimates the benefits of nutrient reductions in addition to the cost of achieving said reductions, is necessary to identify the efficient level of nutrient outputs in the watershed. This is beyond the scope of this paper, but is an important area for future work.

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Appendix A

This appendix presents additional estimates of the weighted and unweighted models. These results are provided to test the robustness of the relationship between fertilizer prices and fertilizer outputs from watersheds. The results presented in the paper are based on an unbalanced panel dataset. As noted in Table 1, data for the Maumee and Sandusky watersheds are available from 1976 to 2011, although some years are missing. Data for the Raisin are available for 1982–2011. Data for the Great Miami and Scioto are available for 1996–2011, although 2010 is missing in the Scioto watershed.

The first column in each table shows the results presented in the paper. The second column presents the results for the periods 1996–2009, a period over which we have a full set of observations for each watershed and can develop a fully balanced panel. The third column presents the results for just the Lake Erie watersheds for the period 1982–2011, a period over which we can also form a balanced panel with just the three watersheds (Maumee, Sandusky, and Raisin).

The fertilizer price parameter is negative and significantly different from 0 in each regression. The base models perform best overall, but in general the parameters have the same signs in each model. Interestingly, our models suggest that phosphorus emissions are more sensitive to fertilizer prices in the Lake Erie Basin than in the Ohio River Basin, and vice-versa for nitrogen.

The fourth and fifth columns present the results for the Scioto and Great Miami watersheds separately. These watersheds are considered separately because we have less data for them, and the Great Miami watershed appears to behave differently. While the sign on fertilizer price is negative for the Scioto watershed for both nitrogen and phosphorus, it is not significantly different from 0. For the Great Miami, the fertilizer price parameter is positive in the case of phosphorus, although the standard error is quite large. The final column presents the results for the period 1996–2009 excluding the Great Miami River Basin.

These results illustrate that the Great Miami watershed behaves differently than the other watersheds. We attribute this to the strong influence of the aquifer system that lies under the Great Miami River. This aquifer system is one of the largest in the United States (Miami Conservancy District, 2012). Given that annual contributions to stream flow from the aquifer are about equal to contributions from surface runoff (Miami Conservancy District, 2012), the aquifer provides an important buffer between changes in farm management and changes in water quality. The presence of this large aquifer decreases the effect that a change in nutrient inputs on the farms will have on nutrient outputs from the watershed.

(See Tables A1–A4.)

Table A1
Unweighted nitrogen regressions. Standard errors in parentheses.

	1975–2011[a]	1996–2009[b]	1982–2011[b]	1996–2011	1996–2011	1996–2009[b]
	Base	All watersheds	Lake Erie watersheds only	Scioto only	Great Miami only	All watersheds, except Great Miami
Constant	3.682 (1.324)**	4.834 (1.793)**	3.785 (1.354)**	3.029 (4.16)	5.480 (3.666)	5.392 (2.162)**
LN(CFS)	−0.113 (0.059)*	−0.110 (0.071)	−0.114 (0.079)	−0.073 (0.151)	−0.192 (0.145)	−0.095 (0.085)
LN(PN)	−0.277 (0.09)**	−0.324 (0.094)***	−0.265 (0.111)**	−0.355 (0.216)	−0.095 (0.211)	−0.360 (0.106)**
LN(PC)	−0.080 (0.095)	−0.061 (0.119)	−0.068 (0.125)	−0.245 (0.211)	−0.036 (0.189)	−0.091 (0.138)
LN(NUSE)	0.044 (0.13)	−0.037 (0.162)	0.013 (0.168)	0.108 (0.359)	−0.149 (0.316)	−0.084 (0.196)
Maumee (1,0)	0.356 (0.117)**	0.417 (0.149)**	0.573 (0.394)	–	–	0.521 (0.134)***
Sandusky (1,0)	0.192 (0.107)*	0.171 (0.124)	0.359 (0.193)*	–	–	0.238 (0.171)
Raisin (1,0)	−0.138 (0.233)	−0.188 (0.294)	–	–	–	−0.168 (0.375)
Scioto (1,0)	−0.117 (0.079)	−0.084 (0.075)	–	–	–	–

* Significant at 0.10 level.
** Significant at 0.05 level.
*** Significant at 0.001 level.
[a] Unbalanced panel.
[b] Balanced panel.

Table A2
Weighted nitrogen regressions. Standard errors in parentheses.

	1975–2011[a]	1996–2009[b]	1982–2011[b]	1996–2011	1996–2011	1996–2009[b]
	Base	All watersheds	Lake Erie watersheds only	Scioto only	Great Miami only	All watersheds, except Great Miami
Constant	4.518 (1.206)***	6.561 (1.71)***	4.124 (1.252)**	3.390 (3.555)	5.719 (3.611)	7.157 (2.022)***
LN(CFS)	−0.243 (0.058)***	−0.251 (0.076)**	−0.261 (0.078)**	−0.192 (0.16)	−0.219 (0.141)	−0.264 (0.093)**
LN(PN)	−0.170 (0.08)**	−0.217 (0.088)**	−0.173 (0.101)*	−0.292 (0.196)	−0.047 (0.207)	−0.254 (0.099)**
LN(PC)	−0.060 (0.087)	−0.064 (0.112)	−0.057 (0.112)	−0.132 (0.243)	−0.080 (0.191)	−0.079 (0.13)
LN(NUSE)	0.010 (0.117)	−0.141 (0.151)	0.030 (0.153)	0.128 (0.311)	−0.172 (0.311)	−0.167 (0.18)
Maumee (1,0)	0.481 (0.106)***	0.604 (0.142)	0.788 (0.368)**	–	–	0.656 (0.127)
Sandusky (1,0)	0.079 (0.097)	0.028 (0.119)	0.386 (0.179)**	–	–	0.027 (0.164)
Raisin (1,0)	−0.325 (0.215)	−0.525 (0.283)*	–	–	–	−0.566 (0.355)
Scioto (1,0)	−0.054 (0.071)	−0.029 (0.071)	–	–	–	–

* Significant at 0.10 level.
** Significant at 0.05 level.
*** Significant at 0.001 level.
[a] Unbalanced panel.
[b] Balanced panel.

Table A3
Unweighted phosphorus regressions. Standard errors in parentheses.

	1975–2011[a]	1996–2009[b]	1982–2011[b]	1996–2011	1996–2011	1996–2009[b]
	Base	All watersheds	Lake Erie watersheds only	Scioto Only	Great Miami Only	All watersheds, except Great Miami
Constant	−1.961 (0.862)**	−2.611 (1.297)**	−2.099 (0.793)**	−1.831 (2.236)	0.865 (2.655)	−3.226 (1.456)**
LN(CFS)	0.200 (0.035)***	0.271 (0.06)	0.361 (0.055)***	−0.052 (0.106)	−0.030 (0.124)	0.338 (0.065)
LN(TPW_{t-1})	0.270 (0.071)***	0.210 (0.101)**	0.259 (0.074)***	−0.112 (0.271)	−0.222 (0.303)	0.207 (0.102)**
LN(PP)	−0.235 (0.085)**	−0.186 (0.096)*	−0.290 (0.095)**	−0.285 (0.155)	0.157 (0.201)	−0.264 (0.101)**
LN(PC)	0.419 (0.088)***	0.298 (0.14)**	0.341 (0.103)**	0.641 (0.279)*	0.205 (0.299)	0.324 (0.145)**
LN(PUSE)	0.089 (0.095)	0.069 (0.139)	−0.011 (0.106)	0.251 (0.178)	−0.310 (0.22)	0.100 (0.16)
Maumee (1,0)	−0.198 (0.088)**	−0.273 (0.126)**	−0.079 (0.249)	–	–	−0.109 (0.112)
Sandusky (1,0)	0.197 (0.08)**	0.242 (0.101)**	0.428 (0.13)**	–	–	0.551 (0.132)***
Rasin (1,0)	−0.193 (0.178)	−0.114 (0.24)	–	–	–	0.262 (0.284)
Scioto (1,0)	−0.178 (0.066)**	−0.213 (0.066)**	–	–	–	–

* Significant at 0.10 level.
** Significant at 0.05 level.
*** Significant at 0.001 level.
[a] Unbalanced panel.
[b] Balanced panel.

Table A4
Weighted phosphorus regressions. Standard errors in parentheses.

	1975–2011[a]	1996–2009[b]	1982–2011[b]	1996–2011	1996–2011	1996–2009[b]
	Base	All watersheds	Lake Erie watersheds only	Scioto only	Great Miami only	All Watersheds, except Great Miami
Constant	−1.903 (0.818)**	−1.824 (1.206)	−2.101 (0.776)**	−2.082 (2.17)	0.893 (2.618)	−2.519 (1.37)*
LN(CFS)	0.177 (0.048)***	0.18 (0.062)**	0.284 (0.058)***	−0.067 (0.111)	−0.031 (0.123)	0.25 (0.071)***
LN(TPW_{t-1})	0.258 (0.069)***	0.192 (0.092)**	0.254 (0.073)***	−0.141 (0.276)	−0.201 (0.297)	0.2 (0.095)**
LN(PP)	−0.167 (0.08)**	−0.148 (0.088)*	−0.231 (0.093)**	−0.264 (0.155)	0.169 (0.194)	−0.226 (0.095)**
LN(PC)	0.363 (0.085)***	0.33 (0.129)**	0.342 (0.098)***	0.638 (0.289)*	0.185 (0.292)	0.346 (0.136)**
LN(PUSE)	0.061 (0.091)	0.033 (0.126)	0.011 (0.103)	0.275 (0.172)	−0.316 (0.216)	0.076 (0.149)
Maumee (1,0)	−0.159 (0.082)*	−0.183 (0.117)	0.004 (0.244)	–	–	−0.052 (0.106)
Sandusky (1,0)	0.179 (0.078)**	0.165 (0.095)*	0.433 (0.126)***	–	–	0.455 (0.126)***
Rasin (1,0)	−0.25 (0.167)	−0.286 (0.226)	–	–	–	0.093 (0.27)
Scioto (1,0)	−0.168 (0.062)**	−0.187 (0.061)**	–	–	–	–

* = significant at 0.10 level.
** = significant at 0.05 level.
*** = significant at 0.001 level.
[a] Unbalanced panel.
[b] Balanced panel.

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