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RESEARCH ARTICLE

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Land cover, stream discharge, and wastewater effluent impacts on baseflow sediment and nutrient concentrations in SW Ohio streams

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Abstract

Elevated nutrient and suspended sediment concentrations often result in negative environmental impacts within freshwater environments. Studies that directly compare suspended sediment and bioavailable nutrients between predominantly agricultural and predominantly urban watersheds during baseflow conditions are largely lacking. The purpose of this study was to determine the impacts of land cover, stream discharge, and wastewater treatment plant (WWTP) discharge on nutrient and sediment concentrations, across a large land cover gradient in Southwest Ohio streams. Weekly baseflow samples were collected from eight streams over 1 year from November, 2016 through November, 2017. Total suspended sediment, nitrate, and phosphate concentrations were measured. Results indicate that agricultural land cover and WWTPs increase nitrate and phosphate concentrations in the study area. Total suspended sediment and nitrate concentrations increased with discharge, and discharge was a relatively weak predictor of phosphate concentrations. Seasonal water quality trends varied by parameter and land use also had unique impacts on seasonal water quality trends. Results suggest that to improve water quality in the study area, efforts should focus on improving WWTP effluent treatment and agricultural land management.

KEYWORDS

agriculture, baseflow, land use, nutrients, suspended sediment, urban, wastewater treatment, water quality

1 | INTRODUCTION

Streams are widely utilized as sources of water for consumption in urban areas (Padowski & Gorelick, 2014), as irrigation in agricultural regions (Hashemi et al., 2016), as habitat for wildlife (Raven et al., 2000), and as corridors for sediment and nutrient transfer from terrestrial environments (Glińska-Lewczuk et al., 2016). Expansion of agricultural and urban land cover in the Midwest of the United States and throughout many areas of the world have significantly increased stream sediment and nutrient concentrations, fundamentally altering the chemical, physical, and biological properties of stream environments (Gallo et al., 2015; Padowski & Gorelick, 2014; Renwick et al., 2008; Riseng et al., 2011). Despite widespread degradation of water quality due to anthropogenic development (Giri et al., 2018), relative impacts between predominantly urban and predominantly agricultural land cover on stream sediment and nutrient concentrations remain poorly understood in the region and globally.

Sediment is considered the primary cause of water quality impairment in U.S. streams (Govenor et al., 2017). Excessive sediment inputs from developed watersheds can have negative impacts on biota by clogging stream bed habitat (Gayraud & Philippe, 2003), increasing abrasion on flora and fauna (Heatherly et al., 2007), and reducing overall biodiversity (Skarbovik et al., 2012). Excessive nutrient concentrations often lead to degradation of aquatic environments by promoting hypoxia and eutrophication (Brion et al., 2015; Dodds & Smith, 2016) and altering food web dynamics (Cashman et al., 2013; Sánchez-Pérez et al., 2009). Streams in the Midwest region of the United States contain some of the highest nutrient concentrations in the country (Bellmore et al., 2018). Most recently, excessive nutrient loading in the region has led to costly algal blooms in the Ohio River and Lake Erie (Daloğlu et al., 2012; Henson et al., 2018).

Agricultural practices such as tilling, crop harvesting, and fertilizer application often promote soil erosion and transport of sediment and nutrient-enriched water to nearby streams (Bates & Arbuckle Jr., 2017; Miller et al., 2011). In streams draining urban watersheds, sediment concentrations are often increased by construction in the watershed and destabilization of streambanks (e.g., through stream incision and widening) (Russell et al., 2017). Additionally, many urban areas contain aging sewage networks, which may lead to increased nutrient concentrations in adjacent streams (Ferreira et al., 2018). In watersheds with urban development and WWTPs, reducing nutrient concentrations in wastewater is an important component of water quality management (Yamashita & Yamamoto-Ikemoto, 2014). Nutrient-enriched effluent is particularly detrimental during baseflow periods (e.g., summer and fall in the U.S. Midwest), when it can account for a majority of stream discharge (Brion et al., 2015).

In addition to anthropogenic disturbances such as land cover modification and the development of WWTPs, sediment and nutrient concentrations are often influenced by streamflow (De Girolamo et al., 2015). During storm events, sediment and nutrient concentrations are often higher relative to baseflow conditions, due to increased runoff and sediment transport capacity (Pizarro et al., 2013). Yet, during baseflow periods, higher discharge may actually decrease suspended sediment concentrations, due to increased groundwater inputs of sediment-free water (Estrany et al., 2008). High discharge can also increase baseflow nutrient concentrations by increasing groundwater transport of nitrate and soluble phosphate through soils and into streams (Calhoun et al., 2002; Gallo et al., 2015). However, higher baseflow may also decrease nutrient concentrations through dilution, particularly in watersheds with high effluent discharge (Rodríguez Benítez et al., 2015).

Many studies have demonstrated that sediment and nutrient concentrations increase with land development (both agricultural and urban; Carpenter et al., 1998; Turner & Rabalais, 2003; Jin et al., 2020). Yet, most studies focus on either predominantly agricultural (Ford et al., 2018), predominantly urban (Hoellein et al., 2011), or mixed land cover watersheds (Richards et al., 2008). Currently, studies that directly compare water quality between highly urbanized watersheds (>50%) and highly agricultural watersheds (>50%) are largely lacking. This makes prioritization of best management practices in mixed land cover watersheds difficult, particularly as seasonal patterns of baseflow water quality may vary across land cover types. Baseflow concentrations of bioavailable nutrients are particularly important in the region and globally as they have significant implications for harmful algal blooms (Ford et al., 2018; Shore et al., 2017). In this study, we aim to increase our understanding of impacts from land cover (agricultural vs. urban), the presence or absence of WWTPs, and stream discharge on stream total suspended sediment (TSS), nitrate (NO_3^-), and phosphate (PO_4^{3-}) concentrations within the Ohio River Valley. We also examine seasonal discharge and water quality dynamics between land cover types. We hypothesize that 1) agricultural land cover, rather than urban land cover, will be the primary driver of sediment and nutrient concentrations in the study streams, due to increased sediment and nutrient availability from exposed and fertilized soils; 2) WWTPs will significantly increase nutrient concentrations in the watersheds in which they are present; 3) sediment and nutrient concentrations will increase with baseflow stream discharge; and 4) sediment and nutrient concentrations will show greater seasonal variability in agricultural watersheds relative to urban watersheds.

2 | STUDY AREA

This study was conducted in Southwest Ohio located in the Eastern Corn Belt Ecoregion, where forested areas have largely been converted to urban or agricultural land cover (Figure 1). The two largest cities near the study area are Cincinnati to the south and Dayton to the north. One stream sampling point across eight watersheds was selected to attain a large land cover gradient (i.e., watershed sampling points are not nested and are thus independent of each other). In the eight study watersheds urban land cover (developed open space, low intensity, medium intensity, and high-intensity development) ranged from 7% to 92%, agricultural land cover (cultivated crops, pasture/hay) ranged from 4% to 71%, and forested land cover (deciduous) ranged from 4% to 29% (Figure 1; Table 1). Land cover within each watershed was determined based on the 2011 National Land Cover Database in ArcGIS (version 10.5). Vegetation in the region primarily consists of crops, mostly corn and soybeans, followed by deciduous forest cover (Rech et al., 2018). Corn is generally planted in May and harvested in mid-November, while soybeans are typically planted in early May and harvested in late October (USDA, 2010). Following cultivation, soils are left bare for about 5–6 months unless cover crops are planted. The average temperature in the study area is lowest in January (−1.1°C) and highest in July (24°C). The average annual precipitation is 1097 mm and average precipitation is lowest in February (68 mm) and highest in May (134 mm). Stream discharge is typically highest in early spring (March–May) and lowest in the fall (September–October). The underlying geology within the study area consists of limestones and shales from the Ordovician Period (Weiss & Sweet, 1964). Soils are primarily composed of silt and silty clay loams (Lerch et al., 1980).

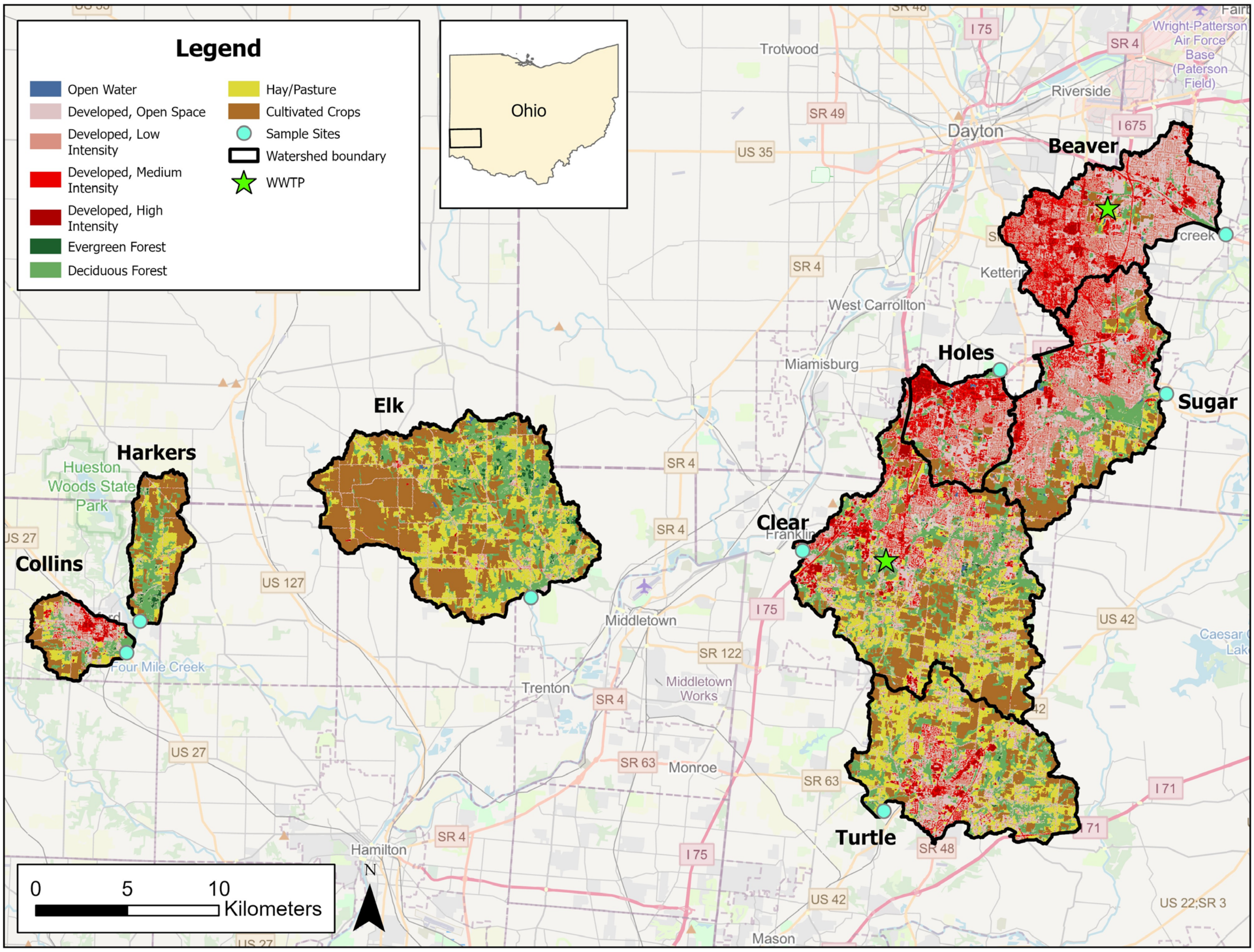

FIGURE 1 Watershed land cover, wastewater treatment plant (WWTP) locations, and sampling sites. [Color figure can be viewed at wileyonlinelibrary.com]

TABLE 1 Land cover and wastewater treatment plant (WWTP) discharge within each study watershed.

Site	Watershed area (km^2)	Agriculture (%)	Urban (%)	Forest (%)	WWTP (m^3/day)
Beaver Creek	62.0	3.8	92.3	3.8	49,200
Holes Creek	29.9	6.6	83.6	9.3	NA
Sugar Creek	78.5	21.5	61.6	16.3	NA
Collins Creek	16.9	35.5	47.4	16.3	NA
Clear Creek	129.0	39.7	36.7	22.2	15,100
Turtle Creek	77.2	45.4	33.2	20.4	NA
Elk Creek	113.1	71.4	7.1	20.9	NA
Harker's Run	16.9	62.5	6.69	29.2	NA

Two of the study watersheds contain large WWTPs with National Pollutant Discharge Elimination System (NPDES) permits that allow for limited effluent discharge. The Springboro WWTP discharges into Clear Creek and has an average design flow of 15,142 m^3/day (4,000,000 gal/day; Ohio EPA, 2017). The Eastern Regional Water Reclamation Facility discharges into Beaver Creek and has an average design flow of 49,210 m^3/day (13,000,000 gal/day). The WWTP is permitted to discharge a maximum TP load of 24.6 kg/day during the months of May through October and must limit TP discharge concentrations to 1.0 mg/L. Phosphorus concentrations during the winter have no maximum limits (Ohio EPA, 2013). Due to its smaller watershed area and greater WWTP discharge, Beaver Creek receives nearly 7 times more effluent per unit watershed area relative to Clear Creek.

3 | METHODS

3.1 | Water sample collection and analysis

Baseflow water samples were collected weekly from each study watershed over a 1-year period from November 19, 2016 through November 11, 2017. During this period, water samples were collected for 50 of 52 weeks. We were unable to sample for 2 weeks due to logistical constraints. When storm events occurred, sampling was postponed typically for 24–48 h to mitigate the collection of water containing overland flow, however, it is possible that large precipitation events may have resulted in samples containing some overland flow. A total of 400 water samples were collected across all watersheds. Between 1% and 5% of collected samples, depending on parameter, were not included in the data analysis due to sample contamination in the field or lab (NO_3^-, $n = 379$; PO_4^{3-}, $n = 379$; TSS, $n = 395$). Nutrient samples from the first two sampling days had to be discarded due to processing errors. These 2 days account for the majority ($n = 16$ of 21) of excluded data. The few remaining samples that were not included in the analyses were not tied to any time period or study site.

During each sampling day, a 4 L pre-washed polyethylene bottle was triple rinsed with stream water and then filled from the thalweg of each stream. Once obtained, water samples were transported on ice in a cooler and subsequently refrigerated at 4°C until processing (within 48 hours). The TSS concentration (mg/L) for each sample was determined by filtering 2–3 L of collected stream water through pre-weighted type A/E glass micro-fiber filters (pore size of 1 μm), dehydrating sediment-laden filters for 48 h at 105°C, weighing dried filters on a microbalance (Mettler Toledo model XP6, Columbus, Ohio, USA), and dividing dried weight by the volume of water filtered. During sediment filtration, a 125 mL subsample of filtered water was collected and preserved with sulfuric acid for NO_3^- and PO_4^{3-} analyses. Nutrient concentrations were measured with a Lachat Quickchem 8500 (series 2) auto-analyzer (Lachat Instruments, Loveland, Colorado, USA) following methods 10-107-04-1-A (for NO_3^-; Wendt, 2000) and 10-115-01-1-Q (for PO_4^{3-}; Diamond, 2007).

3.2 | Discharge

During water sampling, the stream stage was recorded from a staff gauge that was installed at each of the eight study streams. Stream stages were converted to discharge with USGS WinXSPRO software (e.g., Cornwell et al., 2003; Sandercock & Hooke, 2010). To compute discharge, WinXSPRO requires the user to input each stream's cross-section profile, low and high flow slopes, and Manning's *n* value (Hardy et al., 2005). At each stream's sampling location, the channel cross-section was manually surveyed with a surveyor's level and stadia rod following Harrelson et al. (1994). Cross-section surveys were completed at ~30 cm increments with additional survey locations in areas of high topographic variability. Stream slope at low flow was generally too small to accurately measure in the field and was determined by solving for *S*:

$$V = k/n\, R^{2/3} S^{1/2} \tag{1}$$

where *V* is velocity, *k* is a conversion factor, *n* is Manning's roughness coefficient, *R* is the hydraulic radius, and *S* is channel slope. In Equation (1), *V* was determined from manual discharge measurements that were completed in the field with a Flowtracker 2 velocity meter (SonTek, San Diego, California, USA; e.g., Lazar et al., 2019). Since measuring slope under high flow was not possible due to unwadable conditions, high flow slopes for each stream were determined based on slope measurements between geomorphic channel units (e.g., riffles) following Hardy et al. (2005). A Manning's *n* value was determined for each stream based on channel bed substrate, degree of channel irregularity, variation in channel cross-section, the presence–absence of obstructions, and amount of vegetation (Hardy et al., 2005). Following input of stream metrics, WinXSPRO calculated discharge values at stage increments of ~3 cm for each stream. Observed stages during sampling were matched with discharge calculations for each stream.

3.3 | Statistical analyses

Linear mixed-effects models (LMMs) were utilized to determine if watershed land cover (as reported by the percentage of land that is agricultural; see Section 4 for details), stream discharge, and/or WWTP effluent discharge significantly influenced TSS, NO_3^-, and PO_4^{3-} concentrations. Stream discharge and WWTP discharge were scaled by watershed area to allow for cross-watershed comparisons. The structure of the model is

$$Y_{ij} = \beta_{0j} + \beta_{1j} \times Discharge + \gamma \times WWTP + \delta \times LandUse + \varepsilon_{ij},$$

where Y_{ij} is the TSS, NO_3^-, and PO_4^{3-} concentration recorded at time point *i* in stream *j*. The fixed effects in the models were percent land cover and WWTP discharge (modeled by δ and γ, respectively) while stream discharge was considered a random effect (the β_{1j} term), as streamflow can vary by site due to natural watershed variability and heterogeneity of precipitation characteristics between rain events. To account for any underlying different creek dynamics, the intercept (the β_{0j} term) is a random effect, and to model any potential temporal serial correlation in the data, an autoregressive model of order 1 (AR (1)) was fit on the underlying noise terms (the ε_{ij} term). All response variables were log-transformed to satisfy the underlying statistical assumptions. Stream discharge and agricultural land cover percentage were log-transformed to improve the fitted model. Statistical models with various land covers (urban, agricultural, and forested) were compared with Akaike information criterion (AIC) and the model using agricultural land cover provided the best fit. The significance of coefficients (*p*-values) was utilized to determine which independent variables significantly impacted each dependent water quality variable where *p*-values

<0.05 are considered to be significant and p-values <0.10 are considered marginally significant. Pseudo-R^2 (hereafter referred to as partial R^2 or R^2) values were determined following Edwards et al. (2008) with the *r2glmm* package in R (e.g., Borzooei et al., 2019; Jaeger, 2017; Sarremejane et al., 2017). The relative influence of predictor variables within each model was then determined by comparing partial R^2 values for variables that had a significant impact on each water quality parameter. Model fitting and analysis were completed using the *nlme* package (Pinheiro et al., 2018) within R (version 4.3.1, R Core Team, 2023).

4 | RESULTS

4.1 | Temporal discharge and water quality patterns

Over the course of the study period, discharge during baseflow sampling days followed typical patterns for the study area, with the highest baseflow discharge in the spring and lowest baseflow discharge in late summer and early fall (Figure 2). Discharge patterns during

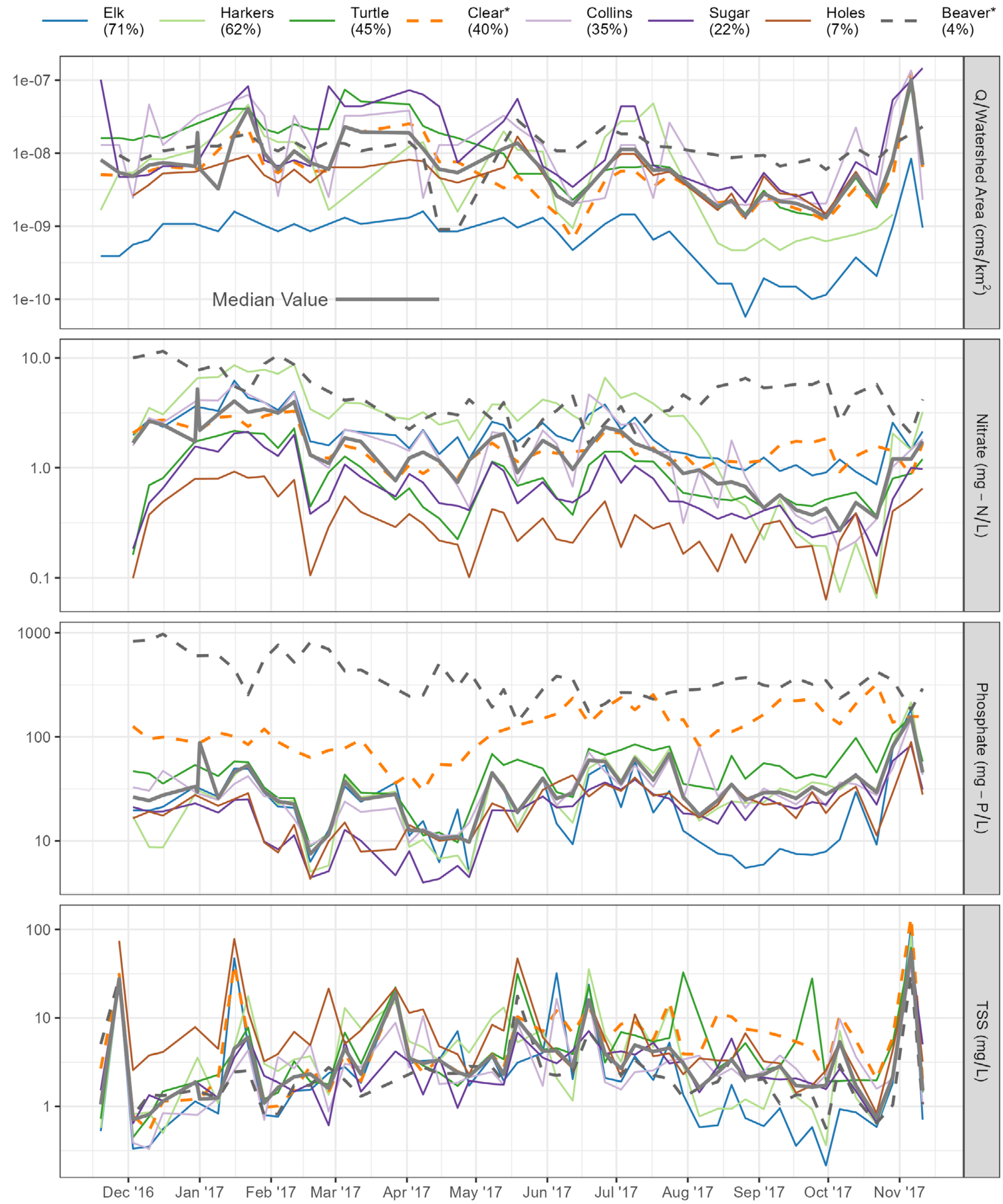

FIGURE 2 Observed discharge scaled by watershed area, nitrate (NO_3^-), phosphate (PO_4^{3-}), and total suspended sediment (TSS) concentrations from each sampling day (approximately $n = 50$) across each study watershed ($n = 8$), with median concentration displayed. Percent agricultural land cover is specified for each watershed at the top of the figure. [Color figure can be viewed at wileyonlinelibrary.com]

baseflow sampling days were generally consistent across sites with some notable exceptions. For example, within the two most agricultural watersheds (Elk Creek and Harker's Run) decreases during the dry season appear to be more drastic relative to less agricultural watersheds. Beaver Creek has the lowest decrease during this period, likely due to consistent effluent discharge from the WWTP which increases overall discharge within the stream. Thus, discharge within Beaver Creek is more consistent throughout the year relative to other sites (Figure 2).

Nitrate concentrations were highest during the winter months and lowest during the fall. Nitrate concentrations increased from November through February, then decreased and varied through July and continued to decrease with more consistency through late October until again increasing through November (Figure 2). Within the two sites that contained WWTPs (Beaver Creek and Clear Creek), NO_3^- concentrations appear to decrease less during the fall and summer months than within the remaining sites. Phosphate concentrations are generally lowest during late winter and early spring months and are highest during summer and fall months. Phosphate concentrations were highly variable from the start of the study period through April, increased but remained variable through July, then decreased and remained relatively steady through mid-October, and increased again in November (Figure 2). Beaver Creek tends to deviate from the normal seasonal patterns of other sites and has the highest PO_4^{3-} concentrations in the winter. Clear Creek which has a lower amount of effluent discharge and a larger watershed area relative to Beaver Creek tends to follow seasonal PO_4^{3-} patterns consistent with other sites, although concentrations are higher throughout the year than in sites without WWTPs (Figure 2). Seasonal patterns in TSS are consistent across sites, with no apparent differences driven by land cover. TSS concentrations peak in the spring and early summer. Total suspended sediment concentrations varied throughout the year, but generally showed an increase from February through June and a decrease from August through October (Figure 2).

4.2 | Land cover

Agricultural land cover, rather than urban or forested land cover, were included in the statistical models for all response variables (TSS, NO_3^-, and PO_4^{3-}), as AIC determined that it explained the greatest proportion of each model's variance (Table S1). Due to the strong inverse relationship between agricultural and urban land cover within the study watersheds (Figure S1), if the concentration of a water quality parameter increased with agricultural land cover, it decreased with urban land cover. This does not indicate that urban land cover improves water quality, but rather, the observed patterns are due to a strong negative correlation between urban and agricultural land cover (i.e., if agricultural land cover increases in a watershed, then urban land cover will decrease). The correlation coefficient between urban and agricultural land cover is −0.9911. To reduce information redundancy (highly correlated variables do not necessarily add new information to a model and can make model interpretation difficult, see Harrison et al., 2018), we report findings with respect to agricultural land cover. However, it is important to note that the significance of independent variables does not change (to or from significance at $p = 0.05$ or 0.10) when substituting urban land cover for agricultural land cover within the statistical model.

4.3 | Nitrate

The LMM model for NO_3^- was highly significant ($p < 0.001$) and demonstrated that NO_3^- concentrations increased with agricultural land cover ($p = 0.0038$), stream discharge ($p = 0.036$), and WWTP discharge ($p = 0.0011$) (Table 2; Figure 3). The model contained an R^2 value of 0.70 and indicated that the strongest predictor of NO_3^- concentration was WWTP discharge ($R^2 = 0.68$), followed by agricultural land cover ($R^2 = 0.53$), and stream discharge ($R^2 = 0.17$).

4.4 | Phosphate

The LMM model for PO_4^{3-} was highly significant ($p < 0.001$) and demonstrated that PO_4^{3-} concentrations increased with agricultural land cover ($p = 0.069$), stream discharge ($p = 0.059$), and WWTP discharge ($p = 0.0037$) (Table 2; Figure 3). The model contained an R^2 value of 0.80 and indicated that the strongest predictor of PO_4^{3-} concentration was WWTP discharge ($R^2 = 0.79$), followed by agricultural land cover ($R^2 = 0.40$), and stream discharge ($R^2 = 0.22$).

4.5 | Total suspended sediment

The LMM model for TSS was significant ($p = 0.039$) and demonstrated that TSS concentrations significantly increased with discharge ($p = 0.0029$). Sediment concentrations were not significantly related to land cover or WWTP discharge ($p > 0.10$) (Table 2; Figure 3). The model contained an R^2 value of 0.19 and stream discharge was a weak predictor of sediment concentrations ($R^2 = 0.16$).

TABLE 2 Y-intercepts and model coefficients for agricultural land cover, stream discharge (stream Q), and wastewater treatment plant discharge (WWTP Q). Standard errors of coefficients are in parenthesis.

Parameter	Y-intercept	Agriculture %	Stream Q $(m^3/s)/km^2$	WWTP Q $(m^3/s)/km^2$
NO_3^- (mg/L)	1.0 (0.750)	0.0139 (0.003)	0.19 (0.090)	5.8e−06 (9e−07)
PO_4^{3-} (μg/L)	2.8 (0.902)	0.0097 (0.004)	0.21 (0.108)	7.0e−06 (1e−07)
TSS (mg/L)	7.4 (2.172)	0.0019 (0.007)	0.75 (0.250)	−2.5e−06 (3e−08)

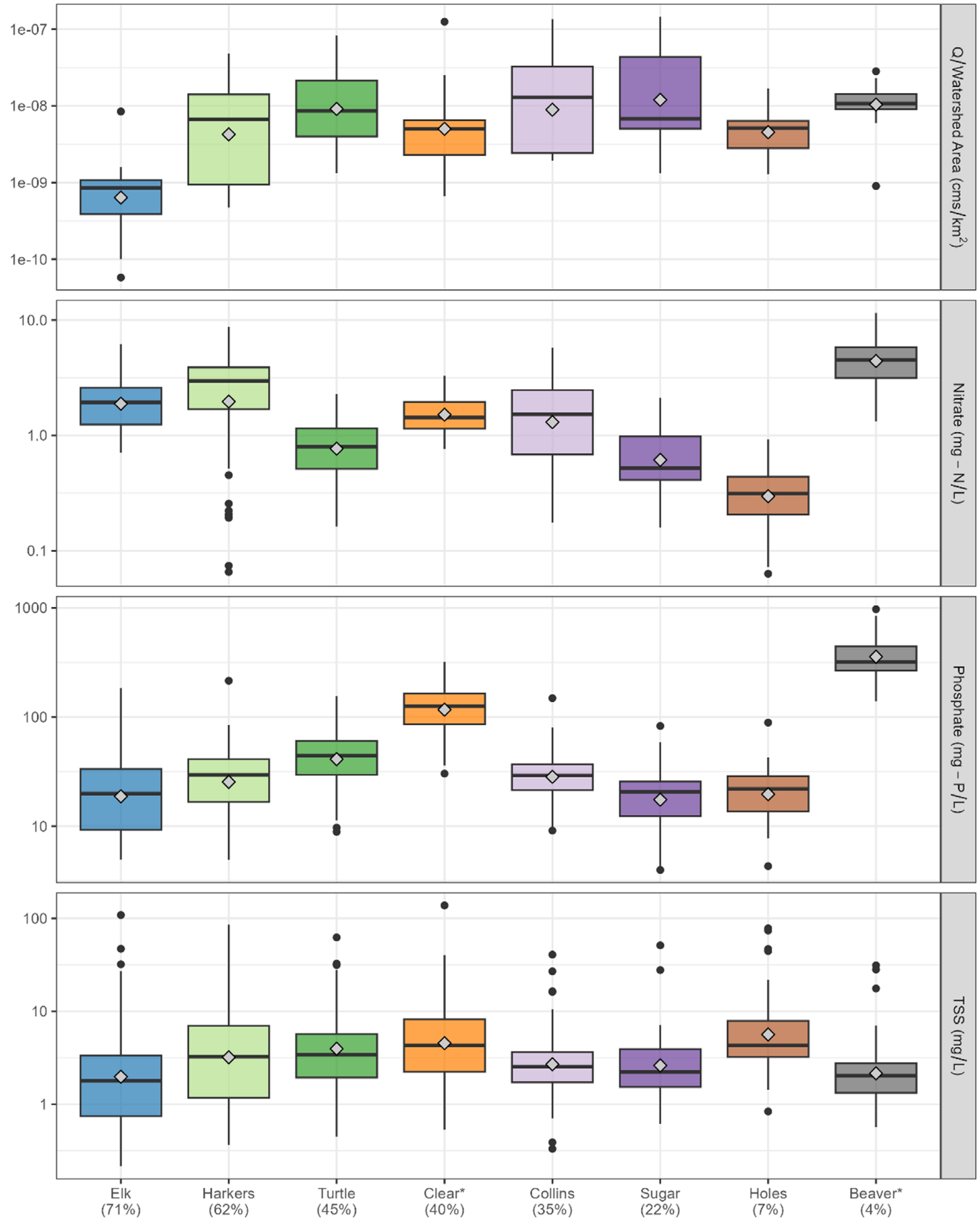

FIGURE 3 Distribution of discharge scaled by watershed area, nitrate (NO_3^-), phosphate (PO_4^{3-}), and total suspended sediment (TSS) concentrations within each stream. Sites (watersheds) are organized from the highest percentage of agricultural land cover (left) to the lowest percentage of agricultural land cover (right). *Clear Creek and Beaver Creek contain WWTPs. Percent agricultural land cover is specified for each watershed at the bottom of the figure. [Color figure can be viewed at wileyonlinelibrary.com]

5 | DISCUSSION

Baseflow nutrient concentrations in the study area were increased by agricultural land cover and WWTP discharge. Stream discharge also significantly increased NO_3^- and PO_4^{3-} concentrations, but explained the least amount of each model's variance. Suspended sediment concentrations increased with streamflow and were not significantly impacted by land cover or WWTP discharge. Although statistically significant, stream discharge was a weak predictor of TSS concentrations. Overall, WWTP discharge followed by agricultural land cover appears to be most detrimental to water quality in the study region. Land cover and the presence of a WWTP in a watershed create distinct seasonal patterns in nutrient concentrations, but not sediment concentrations. Thus, watershed-specific best management practices will likely need to be developed for each watershed and will also depend on which water quality parameter is being prioritized for improvement and during which season. While we are not aware of any studies that have compared seasonal patterns of baseflow NO_3^- or PO_4^{3-} concentrations between various land cover types (e.g., majority agricultural vs. majority urban), Ford et al. (2018) examined total nitrogen and total phosphorus concentrations within an agricultural watershed. Our PO_4^{3-} seasonal patterns generally agree with their seasonal total phosphorus patterns with our high values generally occurring during the summer in agricultural watersheds. Furthermore, our NO_3^- concentrations reflect their total nitrogen concentrations within agricultural concentrations with peaks during winter months which coincide with periods of higher baseflow discharge. Higher NO_3^- concentrations during winter months are also likely influenced by lower microbial activity and denitrification rates during colder periods (Gervasio et al., 2022). Nitrate is highly soluble and easily transported through soils, thus higher baseflow values are likely to lead to increased connectivity with NO_3^- enriched groundwater (Ford et al., 2018). While stormflows are known to rapidly increase sediment concentrations, our results indicate that relatively small increases in baseflow discharge can also increase sediment concentrations within both agricultural and urban streams. Additional studies are needed that compare seasonal nutrient and sediment dynamics between predominantly urbanized watersheds and predominantly agricultural watersheds, particularly during baseflow conditions.

Nitrate and PO_4^{3-} concentrations increased with agricultural land cover within the study area. The use of fertilizers has significantly increased in the United States since the 1940s, particularly in the Midwest U.S. (Cao et al., 2018). Fertilizers, especially nitrogen, have been shown to increase baseflow stream nutrient concentrations, because they are prone to leaching from agricultural soils, into groundwater, and subsequently into streams (e.g., Chand et al., 2011; Collins et al., 2017; Stets et al., 2015). Surprisingly, the percentage of agricultural land cover in a watershed was not significantly related to baseflow TSS concentrations. Agricultural practices such as tilling and harvesting loosen soil making it readily available for transport during runoff events (Jones & Schilling, 2011). In regards to TSS, land cover may be a more important factor during stormflow conditions as sediment is predominantly transported to streams during overland flow (Koskelo et al., 2018; Lazar et al., 2019).

Stream discharge may have only been a relatively weak predictor of PO_4^{3-} as concentrations were largely impacted by effluent inputs (Table 2). Higher streamflow often has a greater capacity to suspend and transport substrate (Lenhart et al., 2010), including sediment that contains bound nutrients (Harrington, 2014). The relationship between streamflow and TSS indicates that within the study area, higher baseflow rates are more likely to increase sediment transport capacity than to drive dilution. However, the overall low predictiveness of suspended sediment concentrations in relationship to discharge indicates that variables not measured in this study may also have a stronger effect on sediment transport during baseflow (e.g., underlying geology and stream geomorphology; Bywater-Reyes et al., 2017, previous land use; Lizaga et al., 2021, and legacy effects; Jiang et al., 2020).

WWTP discharge increased NO_3^- and PO_4^{3-} concentrations. Beaver Creek had the highest nutrient concentrations of all the sampled sites (Figure 3), likely due to the significant effluent discharge from the Eastern Regional WWTP. Beaver Creek is located in the most urban watershed within the study area and subsequently discharges the most wastewater effluent (>49,000 m^3/day). Furthermore, the watershed is relatively small (62 km^2) and only provides limited dilution potential (i.e., drains into a stream with low discharge). During sampling the scent of effluent was evident within Beaver Creek, particularly during summer days. In 2010, the United States Environmental Protection Agency conducted biological sampling and found that a longitudinal pattern of impact and recovery was evident in relation to the Springboro WWTP, and the macroinvertebrate community was rated as marginal downstream of the plant (Ohio EPA, 2012). Numerous studies outside of these watersheds have also found negative biological effects in relation to WWTP effluent (e.g., Berninger et al., 2014; Drury et al., 2013). This is especially concerning during dry periods when WWTP discharge can be the main source of flow in rivers and effluent is not effectively diluted. Without advanced treatment techniques that are effective at removing nutrients from effluent, significant NO_3^- and PO_4^{3-} concentrations are likely to continue to be detrimental to local water bodies (e.g., Edlund et al., 2009; Volf et al., 2013).

There were several limitations that should be considered when interpreting the results of this study. First, only two of the watersheds contained WWTPs upstream of sampling sites. While WWTP discharge had a significant impact on nutrient concentrations, other WWTPs may have variable impacts due to differing treatment practices, discharge rates, and characteristics of receiving streams. Although the impact of the two WWTPs in this study should be cautiously interpreted due to limited replication, upstream-downstream sampling during the summer of 2017 (Figure S2) further supports that the WWTPs in the study area are driving the observed increases in nutrient concentrations. Specifically, the average NO_3^- and PO_4^{3-}concentrations below the WWTPs were, respectively, 5 and 30 times higher downstream of the two WWTPs. Another limitation of this study is that there were no undisturbed, predominantly forested, watersheds to use as controls near the study area, as most of the land in Southwest Ohio is developed. Thus, we were unable to determine natural nutrient and sediment concentrations during the study period. Lastly, other factors such as fertilizer application rates and timing, crop types, use of soil conservation practices (no-till agriculture and cover crops), local variability in geology and stream geomorphology, and land use history may also have some influence on the sediment and nutrient concentrations observed in the study streams, but examining the impact of these factors was outside the scope of this study.

6 | CONCLUSION

Water quality in the study area during baseflow conditions is negatively impacted by agricultural land cover and WWTP effluent discharge. WWTP discharge was the primary driver of nutrient concentrations but showed no impact on TSS concentrations. Agricultural land cover also significantly increased NO_3^- and PO_4^{3-} concentrations. Stream discharge increased all nutrient and suspended sediment concentrations, however, was a generally weaker predictor of nutrient concentrations. Agriculture along with the presence of WWTPs influence seasonal patterns of nutrient concentrations in the study area. Improved management practices are needed in the study area to bolster water quality by reducing nutrient concentrations that are sourced from WWTPs and agricultural land cover.

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DATA AVAILABILITY STATEMENT

All raw data will be published within the "Mendeley Data" data repository https://data.mendeley.com/ upon publication of the manuscript.

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