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# Water Chemistry During Baseflow Helps Inform Watershed Management: A Case Study of the Lake Wister Watershed, Oklahoma

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**Abstract:** Nonpoint source (NPS) pollution from agricultural and urban development is a primary source of nutrients and decreased water quality in aquatic systems. Installation of best management practices (BMPs) within critical source areas of the watershed can be helpful at reducing the transport of nutrients to waterbodies; however, prioritizing these areas may be difficult. The objective of this study was to develop several potential frameworks for prioritizing subwatersheds using baseflow water chemistry data in relation to a simple human development index (HDI; total percent agriculture and urban development). At a monthly interval, samples were collected at 26 sites throughout the Oklahoma portion of the Lake Wister Watershed (LWW) and analyzed for total nitrogen, total phosphorus, total suspended solids, and chlorophyll *a*. Changepoint analysis for each parameter found significant thresholds for each of the parameters ranging from 20 to 30% HDI. Changepoint analysis summary statistics were used to develop prioritization frameworks for the LWW that could be used to target subwatersheds where BMP installation would have the greatest effect at improving water quality. Additionally, regression models developed from the relationships between water quality parameters and HDI values serve as realistic targets for improving water quality, with the modeled line representing the target concentration for a given HDI value. After BMPs have been implemented, baseflow monitoring should continue at the subwatershed scale to track changes in water quality. Focusing monitoring efforts at the subwatershed scale will provide an earlier indication of the effectiveness of BMPs, as it may take several decades to detect improvements in water quality at the larger watershed scale.

**Keywords:** *human development index, nonpoint source pollution, best management practices, changepoint analysis, regression analysis*

Accelerated eutrophication from excess nutrients entering aquatic systems is a global issue. Nutrients from the landscape associated with human activities [i.e., nonpoint sources (NPS)] are one of the leading causes of impairment to water ways in the United States (EPA 2000). Nutrient enrichment decreases water quality and water clarity through increased algal production (Smith et al. 1999). Increased algal production can form nuisance and or harmful algal blooms (HABs) (Heisler et al. 2008; Paerl et al. 2016) and increases prevalence of hypoxic conditions in coastal waters (Rabalais et al. 2002), such as that in the Gulf of Mexico proximal to the inflow of the Mississippi River.

The Mississippi River Basin drains the heartland of agricultural production in the United States, where the nutrient cycle in agriculture, from a systems perspective, is broken. Nutrients (i.e., fertilizers) are input into the Midwest to grow crops (e.g., corn and soybeans) which are then used as feed in animal production (e.g., poultry production) outside the region. The feed grains are exported from row crop production areas (e.g., Midwest) to animal production areas (e.g., Southeast), where food products are then exported globally but the manure remains locally (Sharpley and Withers 1994). The manure left behind is an excellent fertilizer, but it has an imbalance in terms of nitrogen (N) and phosphorus (P) in relation to plant needs (Eck and Stewart 1995). The manure was historically applied locally to pastures, which has led to P buildup in soils and P loss during rainfall and runoff (Sharpley et al. 1996).

The loss of nutrients from fields fertilized with manures is an overwhelming water quality concern, and it is important to understand that only a small fraction (< 10%) of the nutrients applied are lost in runoff annually. For example, plot studies have shown that only 4 and 2% of the N and P applied as manure was lost in surface runoff in Northwest Arkansas (Edwards and Daniel 1993), although these initial rates of loss may vary based on location, soil type, and slope. Interestingly, these percent losses from manure applied to the landscape can be scaled up to the large watershed scale; a mass balance often shows that nutrient loads from a watershed are small percentages of the total amount of manure produced and likely applied within the watershed (e.g., Haggard et al. 2003). The important point is that a large percent of the nutrients applied remain on the landscape within the watershed, i.e., legacy nutrients from past application and management.

Legacy nutrients in soils slowly move with water, either vertically with infiltration (Tesoriero et al. 2009, 2013; Puckett et al. 2011) or laterally with surface runoff (Gburek and Sharpley 1998; Tesoriero et al. 2009), with the rate at which legacy nutrients leave the landscape varying greatly between soil types (Sharpley 1985). The legacy nutrients moving along these surface and subsurface pathways may end up in nearby waterbodies (Basu et al. 2010). This nutrient source and the other sources (e.g., current fertilizer and manure applications) with transport potential result in increases in stream nutrient concentrations. This is why stream nutrient concentrations (from individual samples to annual means) are often positively correlated to the proportion of agricultural lands (sum of % crop, % pasture, and % grassland) and urban development (sum of % developed open-space, and % low, medium, and high intensity development) in the watershed. This relationship has been documented across the nation (Byron and Goldman 1989; Jordan et al. 1997; Jones et al. 2001; Howarth et al. 2002; Haggard et al. 2003; Toland et al. 2012; Cox et al. 2013; Giovannetti et al. 2013).

Best management practices (BMPs) are often used to reduce nutrient and sediment loss from the landscape, which hopefully translates into improved water quality downstream. Buffer strips and riparian buffers can be installed along the edge of fields to slow overland flow and intercept nutrients and sediment in runoff (Schoumans et al. 2014). Conservation tillage practices (e.g., no-till, spring-till, and cover crops) reduce erosion in the field during the non-growing season (Tilman et al. 2002), decreasing the amount of nutrients and sediment lost from the landscape. Implementing these practices throughout the entire watershed would have the greatest effect at reducing NPS of nutrients and sediments. However, implementation of these BMPs [and others; see (Schoumans et al. 2014)] throughout the entire watershed may not be feasible due to low landowner participation, and limited funds and resources. Targeting critical source areas to implement these BMPs would optimize the benefit while reducing the cost (Sharpley et al. 2000; Niraula et al. 2013).

A variety of techniques have been used to identify priority locations for BMP implementation to improve water quality, including qualitative indices [e.g., P Index, (Lemunyon and Gilbert 1993; Sharpley et al. 2001)] and watershed modeling (Pai et al. 2011). Recent work suggests that water quality monitoring during baseflow conditions can be used to prioritize subwatersheds for BMP implementation (McCarty and Haggard 2016). The premise is that stream water quality during baseflow conditions reflects the influence of NPS pollution across the watershed. Thus, stream water quality can be related to human development (i.e., percent urban and agriculture land cover) across a target watershed and this relation can be used to suggest priority areas for BMP installation.

Here, we present a case study focusing on baseflow water quality monitoring within the Lake Wister Watershed (LWW), near Wister, Oklahoma. The primary goal of this monitoring was to assist the Poteau Valley Improvement Authority (PVIA) and other stakeholders in prioritizing subwatersheds for BMP implementation to help reduce sediment and nutrient transport from the landscape. At the end of the case study we provide several potential methods for subwatershed prioritization and also a means for setting realistic targets for water quality improvement.

## Case Study

Lake Wister is on Oklahoma's 303(d) list for impaired water quality, including excessive algal biomass, pH, total phosphorus (TP), and turbidity (ODEQ 2016). To address these water quality issues, the PVIA released its "Strategic Plan to Improve Water Quality and Enhance the Lake Ecosystem" in 2009. The strategic plan divides the restoration efforts into three zones of action to focus on, including the watershed, the full lake, and Quarry Island Cove. The purpose of this project was to focus on the watershed by monitoring stream water quality during baseflow conditions at or near the outlets of the subwatersheds, in the Oklahoma portion of the LWW.

## Methods

### Study Site Description

The LWW covers an area of 2,580 km$^2$ (~640,000 acres) and makes up the southern half (52%) of the entire Poteau River sub-basin (hydrologic unit code (HUC) 11110105; Figure 1). The primary land use and land cover (LULC) across the Oklahoma portion of the LWW is 72% forest (sum of % deciduous, % evergreen, and % mixed forest), 19% agriculture, and 4% urban; the LULC for the 845 km$^2$ (~209,000 acres) portion of the LWW in Arkansas is similar with 71% forest, 20% agriculture, and 5% urban.

Within the Oklahoma portion of the LWW, there are 26 HUC 12 subwatersheds that range in size from 42 to 125 km$^2$ (10,300 to 30,800 acres). Forest is the dominant LULC across the HUC 12s, ranging from 45 to 95% of the watershed. The proportion of human development (i.e., agriculture plus urban) was less than half of the LULC across the stream sites (4 to 48%; Table 1). Additionally, across the LWW there are seven EPA national pollutant discharge elimination system (NPDES) permitted point sources, including wastewater treatment plants (WWTPs), sewage systems, and a poultry processing plant.

For this study, 26 sites were selected at bridge crossings near the outflow of 23 of the HUC 12's in the Oklahoma portion of the LWW shown in Figure 1. The LULC for the catchments upstream of the 26 sample sites ranged from 49 to 95% forest, < 1 to 37% agriculture, and < 1 to 10% urban. LULC data in Table 1 represent the land use for the entire catchment upstream of each sampling location.

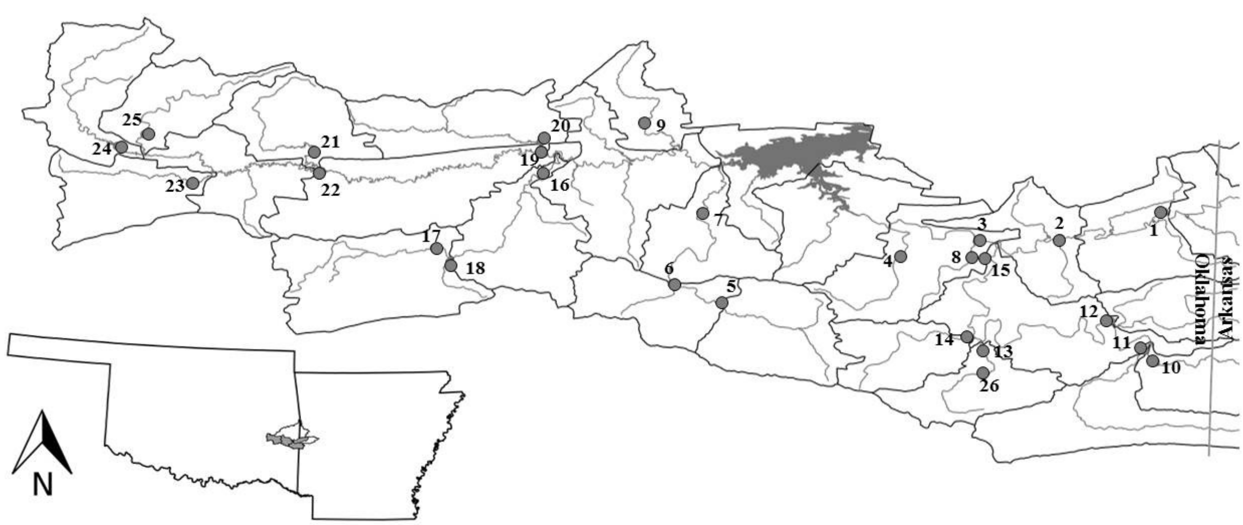


**Figure 1**. Sample sites within the Lake Wister Watershed of Oklahoma. Site numbers on the figure correspond to site numbers in Table 1.

**Table 1**. Sample sites and land cover within the Lake Wister Watershed organized by hydrologic unit code (HUC) 10s. The number in the HUC 12 column is the final two digits associated with the HUC10 number listed at the top of each group of sites. Watershed area and land use and land cover values are representative of the full catchment upstream of the sites.

| Site # | HUC 12 | Stream Name | Area ($Km^2$) | %F[1] | %AG[2] | %U[3] | % HDI[4] | Lat. | Long. |
|---|---|---|---|---|---|---|---|---|---|
| HUC10-1111010503: Upper Poteau River | | | | | | | | | |
| *1 | 03 | Poteau River | 694 | 66 | 25 | 5 | 30 | 34.8798 | -94.4830 |
| *2 | 04 | Poteau River | 768 | 66 | 25 | 5 | 30 | 34.8587 | -94.5657 |
| *3 | 05 | Poteau River | 1335 | 74 | 18 | 5 | 22 | 34.8584 | -94.6292 |
| HUC10-1111010505: Middle Poteau River | | | | | | | | | |
| 4 | 02 | Conser Creek | 34 | 95 | 3 | 2 | 5 | 34.8671 | -94.7039 |
| 5 | 04 | Holson Creek | 73 | 94 | 3 | 2 | 5 | 34.8069 | -94.8376 |
| 6 | 05 | Holson Creek | 132 | 92 | 4 | 3 | 7 | 34.8227 | -94.8765 |
| 7 | 06 | Holson Creek | 182 | 91 | 5 | 3 | 7 | 34.8795 | -94.8533 |
| 8 | 02 | Rock Creek | 11 | 67 | 30 | 2 | 32 | 34.8431 | -94.6357 |
| 9 | 03 | Coal Creek | 27 | 72 | 19 | 2 | 21 | 34.9514 | -94.8900 |
| HUC10-1111010502: Black Fork Poteau River | | | | | | | | | |
| 10 | 02 | Black Fork | 122 | 88 | 6 | 2 | 9 | 34.7600 | -94.4902 |
| 11 | 01 | Big Creek | 112 | 90 | 3 | 5 | 8 | 34.7692 | -94.4987 |
| 12 | 03 | Black Fork | 323 | 89 | 5 | 3 | 8 | 34.7926 | -94.5257 |
| *13 | 04 | Shawnee Creek | 48 | 88 | 1 | 6 | 8 | 34.7679 | -94.6276 |
| 14 | 05 | Cedar Creek | 48 | 95 | 1 | 4 | 4 | 34.7785 | -94.6400 |
| *15 | 06 | Black Fork | 509 | 88 | 6 | 4 | 9 | 34.8432 | -94.6248 |
| *26 | 04 | Shawnee Creek | 23 | 93 | 1 | 5 | 6 | 34.7894 | -94.6279 |
| HUC10-1111010504: Fourche Maline | | | | | | | | | |
| 16 | 08 | Long Creek | 180 | 80 | 13 | 1 | 15 | 34.9084 | -94.9803 |
| 17 | 07 | Long Creek | 77 | 83 | 12 | 1 | 13 | 34.8512 | -95.0662 |
| 18 | 07 | Long Creek tributary | 20 | 87 | 8 | 3 | 12 | 34.8401 | -95.0538 |
| *19 | 09 | Fourche Maline | 417 | 63 | 28 | 4 | 32 | 34.9293 | -94.9813 |
| *20 | 06 | Red Oak Creek | 71 | 54 | 37 | 6 | 43 | 34.9360 | -94.9809 |
| 21 | 04 | Little Fourche Maline | 55 | 70 | 23 | 3 | 26 | 34.9275 | -95.1626 |
| *22 | 05 | Fourche Maline | 313 | 67 | 26 | 4 | 30 | 34.9124 | -95.1561 |
| *23 | 03 | Bandy Creek | 59 | 49 | 37 | 10 | 47 | 34.9023 | -95.2615 |
| 24 | 02 | Fourche Maline | 72 | 81 | 12 | 4 | 16 | 34.9325 | -95.3195 |
| 25 | 01 | Cunneo Creek | 45 | 90 | 7 | >1 | 7 | 34.9419 | -95.2975 |

[1] % Forest, includes deciduous, evergreen, and mixed forest; [2] % Agriculture, includes crops, grassland, and pasture/hay; [3] % Urban, includes developed-open space, low, medium, and high intensity development; [4] % Human Development Index (HDI) is the sum of % agriculture and % urban; * Sites downstream of EPA NPDES permitted point sources.

### Sample Collection and Analysis

Water samples were collected at the 26 sites at monthly intervals from July 2016 through July 2017 during baseflow conditions, as defined by no measurable precipitation seven days prior to sampling. Samples were not collected in October of 2016 due to abnormally dry conditions which resulted in no flow in several of the smaller streams, resulting in a total of 12 samples collected. Samples were collected from the vertical centroid of flow where the water is actively moving, either by hand or with an Alpha style horizontal sampler lowered from the bridge. Water samples were split, filtered, and acidified in the field based on the specific storage needs for each analyte. All samples were stored on ice until delivered to the Arkansas Water Resources Center certified Water Quality Lab (AWRC WQL).

All water samples were analyzed for total nitrogen (TN), TP, total suspended solids (TSS), and sestonic chlorophyll-*a* (chl-*a*) using standard methods that are available at https://arkansas-water-center.uark.edu/water-quality-lab.php (accessed 11/18/2018).

### Data Analysis

All LULC data for the LWW, HUC 12s within the LWW, and catchments upstream of each sampling location were compiled using GeodataCrawler (see http://www.geodatacrawler.com/; accessed 11/18/2018) and Model My Watershed (see https://app.wikiwatershed.org/; accessed 11/18/2018). LULC data were used to calculate a simple human development index (HDI) value as the total percent agriculture and urban land use for the catchment upstream of each sample site and for each subwatershed (Table 1).

All water quality data collected over the course of this study can be found in the data report "DR-WQ-MSC385" available at https://arkansas-water-center.uark.edu/publications/DR-WQ-MSC385_Water-quality-monitoring-Poteau-Valley-Improvement-Authority.xlsx (accessed 11/18/2018). The geometric mean of constituent concentrations at each site was used in the data analysis, because it is less sensitive to extreme low and high values than arithmetic means. The geometric mean is typically a good estimate of the central tendency or middle of the data.

Both seasonal and annual geometric means were calculated for the water quality parameters at each site. The geometric means of all the data from each site were related to HDI using simple linear regression. This statistical analysis shows how geometric mean constituent concentrations change across a gradient of HDI, or agriculture plus urban land use, in the drainage area.

Changepoint analysis is another way to examine how HDI might influence constituent concentrations in streams. Changepoint analysis looks for a threshold in the geometric mean concentration and HDI relation, where the mean and variability in the data changes. This statistical analysis is not dependent on data distributions, and it gives a threshold in HDI where the geometric mean concentrations likely increase.

## Results and Discussion

### Nitrogen

The majority of TN in the flowing waters was in the particulate form, where dissolved inorganic N (DIN: $NH_3$-N plus $NO_3$-N) was typically less than 35% of the total. Annual geometric mean concentrations for TN ranged from 0.10 to 1.50 mg $L^{-1}$. This range in TN is consistent across all four seasons, and there was no real seasonal pattern (Figure 2A). In roughly 60% of the samples, TN was within the range of the nutrient supply threshold needed to promote algal growth and cause shifts in algal community composition [0.27 to 1.50 mg $L^{-1}$; (Evans-White et al. 2013)] potentially creating nuisance algal conditions.

The geometric mean concentrations of the TN species varied across the LWW, reflecting changes in nutrient sources and land uses within the drainage areas. TN geometric means increased with the proportion of agriculture and urban development (Figure 3A), i.e., HDI values, in the watershed, explaining 78% of the variability in TN. This relationship with stream N concentrations and HDI has been observed across the region (e.g., see Haggard et al. 2003; Migliaccio and Srivastava 2007; Giovannetti et al. 2013). The regression lines provide a possible water-quality target to which TN concentrations might be reduced at a given HDI. The sites, or streams, with concentrations well above this line might be of specific interest for management purposes, e.g., Site 23. Additionally, streams with greater HDI that fall below the line may also be of interest to determine why these stream reaches have low constituent concentrations despite having a higher HDI value (i.e., is it due to good riparian, implementation of BMPs, etc.).

The geometric mean concentrations for TN also showed a changepoint response to increasing HDI; that is, the average and deviation of the geometric means increased above a HDI value of 28% (Figure 4A). The average of the data above the changepoint was generally two to three times greater than the data below that HDI value.

**Phosphorus**

Geometric mean concentrations for TP ranged from 0.013 to 0.208 mg $L^{-1}$; much of which was in the particulate form, where the dissolved form (SRP) typically made up less than 33% of the measured TP. This range was consistent across all of the seasons except for summer, when median TP concentration was elevated relative to the other seasons and annual median (Figure 2B). The increase in TP across the streams during summer corresponded with slight increases in sediment and Chl-*a* in the water column (discussed later). In roughly 80% of the samples, TP was within the range of nutrient supply threshold needed to increase algal growth and drive shifts in algal community composition in streams [0.007 to 0.100 mg $L^{-1}$; (Evans-White et al. 2013)] and potentially cause nuisance algal conditions. However, two sites with values much higher than this range were directly downstream of effluent discharges (Bandy Creek and Shawnee Creek at Hwy 59).

Geometric mean P concentrations varied across the streams draining the LWW, showing that 70% of the variability in P concentrations was explained by HDI (Figure 3B). These relationships between stream TP concentrations and HDI, like TN, have been observed across the region (e.g., see Haggard et al. 2003; Cox et al. 2013), reflecting potential TP

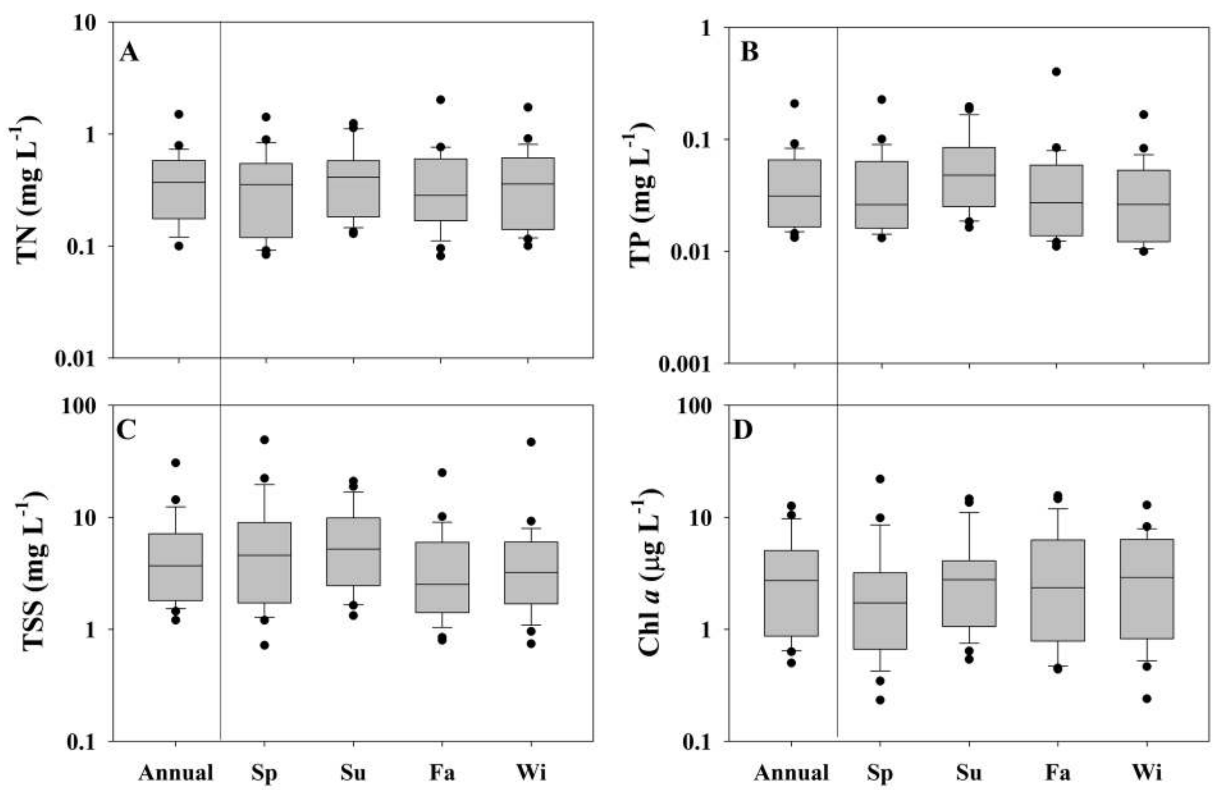


**Figure 2**. Box and whisker plots of constituents showing medians (horizontal line within each box), range (error bars show the 5th and 95th percentiles), and outliers (points above and below error bars) for each of the constituents analyzed at the Oklahoma sites in the Lake Wister Watershed. Annual data are to the left of the vertical line, while seasonal data are to the right. The abbreviations stand for: spring (Sp), summer (Su), fall (Fa), and winter (Wi).

sources such as poultry litter applied to pastures (DeLaune et al. 2004; Cox et al. 2013). The regression lines provide a realistic water quality target to which P concentrations might be reduced (without conversion to forest) and show sites that deviate greatly from concentrations at a given HDI.

The geometric mean concentrations of TP showed changepoint responses to increasing HDI. The changepoint for TP was slightly lower than TN at 21% HDI. As with TN, mean TP values above the threshold were more than two times greater than the mean values below the threshold. Site 23 consistently shows elevated P and N concentrations relative to other sites across the LWW, suggesting nutrient sources upstream might need to be investigated (Figure 4B).

**Suspended Sediments**

Annual geometric means for TSS ranged from 1 to 31 mg $L^{-1}$. Geometric mean TSS concentrations were greater in the spring and summer than the fall and winter (Figure 2C). Low values in the fall may be explained by the drier conditions that began towards the end of summer through early winter 2016. The less frequent rainfall-runoff events reduce erosion from the landscape and within the stream channel, and the lower flows throughout this season have less power to erode the channel and suspend particulates in the water column (Morisawa 1968). The more frequent storms and elevated baseflow during spring and early summer likely kept TSS elevated in the streams (relative to fall) across the LWW. TSS was positively correlated to TP in streams of the LWW ($r = 0.739$; $p < 0.001$), which has been documented elsewhere (Stubblefield et al. 2007).

Many factors influence the amount of particulates in the water column of streams,

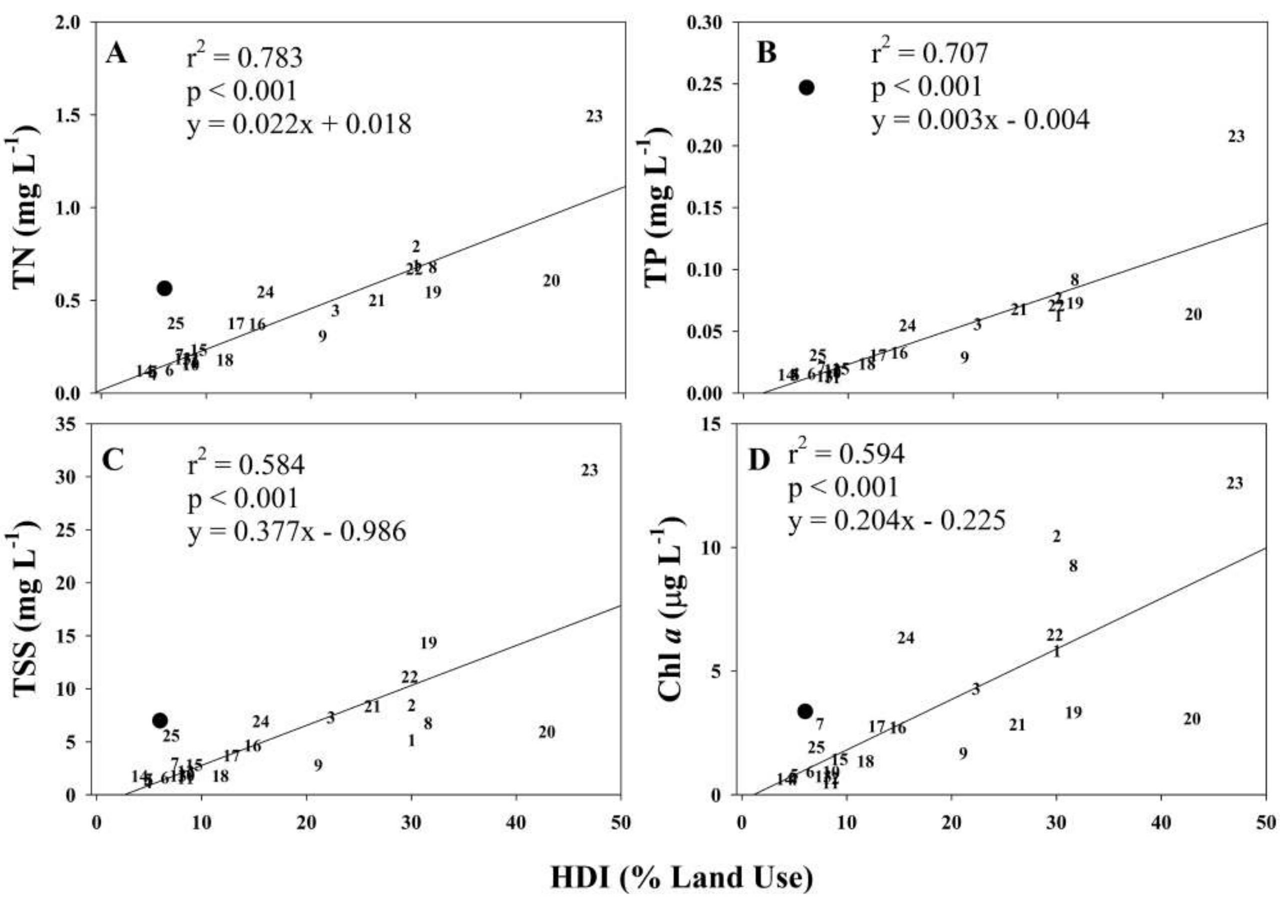


**Figure 3.** Simple linear regression of geometric mean constituent concentrations versus human development index (HDI) values for the Oklahoma portion of the Lake Wister Watershed. The site number in red is Shawnee Creek at highway 59, downstream of effluent discharge, thus it was not used in the statistical analysis.

including rainfall-runoff, discharge, channel erodibility, and even algal growth. The myriad of factors that influence TSS in water are also influenced by human activities, which is likely why HDI explained more than half of the variability ($R^2$=0.584; P<0.001) in the geometric means of TSS across the streams of the LWW (Figure 3C). These relationships are not well defined regionally, but where data are available, similar observations have been made (Price and Leigh 2006). There was also a significant threshold response in TSS at 22% HDI (Figure 4C). It is interesting to note that while samples were collected at baseflow, TSS was still strongly correlated to HDI across these sites.

**Chlorophyll *a***

Annual geometric mean concentrations of sestonic Chl-*a* (algal biomass in the water column) ranged from 0.5 to 12.6 µg $L^{-1}$ across the LWW. Geometric mean Chl-*a* concentrations were consistent throughout the year, without much variability between seasons (Figure 2D). Additionally, Chl-*a* concentrations across these sites were strongly (positively) related to total nutrient concentrations in the water column, where TP explained 78% on Chl-*a* variability ($p < 0.001$), while TN explained 85% ($p < 0.001$). This relationship between sestonic algae and total nutrients has been documented in other systems (Chambers et al. 2012; Haggard et al. 2013).

The geometric mean concentrations of Chl-*a* increased with the proportion of human development in the watershed (i.e., HDI values), where HDI explained 59% of the variability in sestonic Chl-*a* ($p < 0.001$; Figure 3D). This strong relationship was surprising, because many physical,

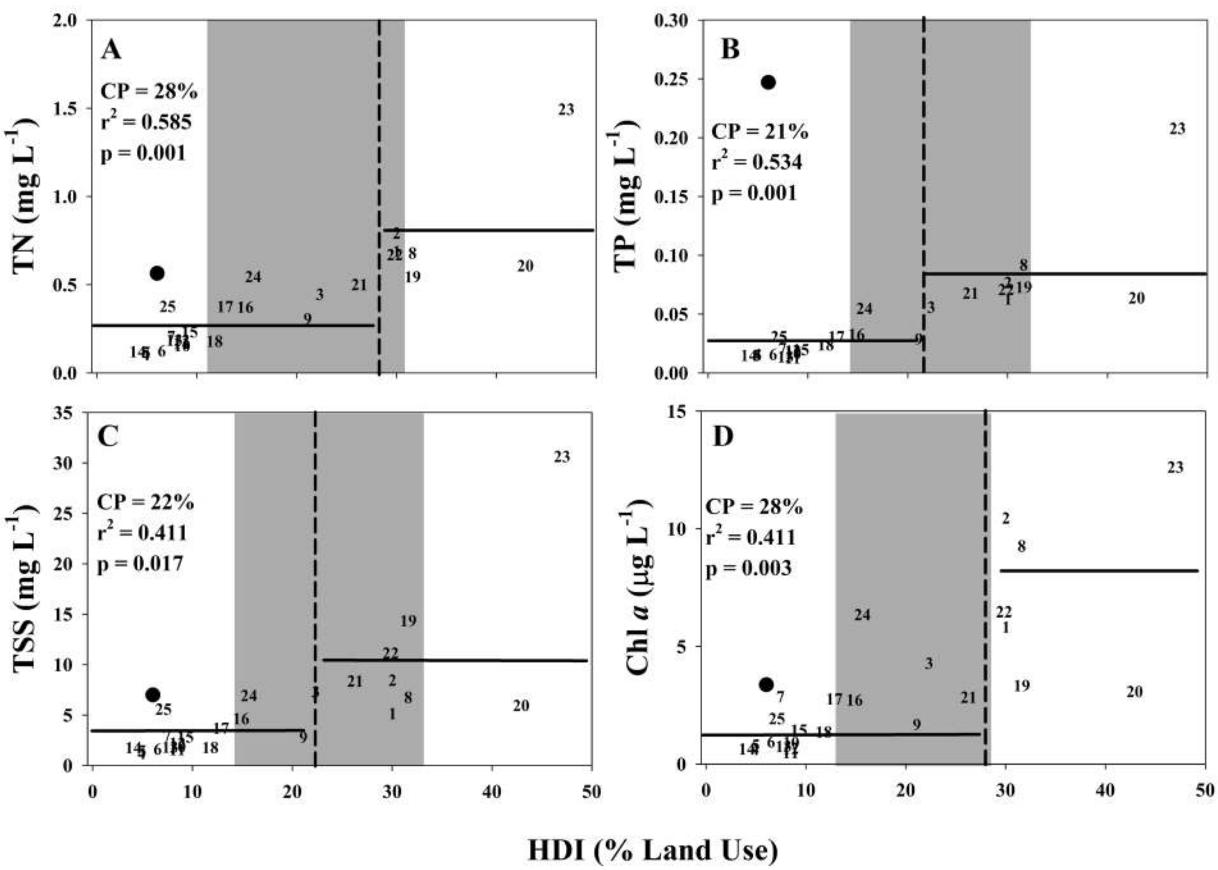


**Figure 4.** Changepoint analysis of geometric mean concentrations versus human development index (HDI) value for sites in the Oklahoma portion of the Lake Wister Watershed. The vertical dashed line represents the changepoint values specific to each constituent. The gray box shows the 90% confidence interval about the changepoint. Horizontal bars represent the mean of the data points to the left and right of the change point. The site number in red is Shawnee Creek at highway 59, downstream of effluent discharge, thus it was not used in the statistical analysis.

chemical, and biological factors influence algal growth in streams (Evans-White et al. 2013). It is likely that this correlation is driven by the increased nutrient concentrations that are found at sites with higher HDI values. Additionally, hydrology [e.g., discharge and velocity (Honti et al. 2010)] may also be an important factor controlling sestonic algal growth, where slower velocities in low gradient streams might allow for greater growth than in high gradient streams, when nutrients are elevated. Interestingly, sestonic Chl-*a* still showed a threshold at a HDI value (28%) similar to that observed with the chemical concentrations (Figure 4D).

## Criteria for Prioritizing HUC 12s

Changepoint analysis is a powerful statistical tool, and one of its most useful aspects is that it gives a threshold, i.e., specific value on the X−axis. In this case, the changepoint is the HDI value where land use begins to have a significant influence on water quality, as seen by increasing constituent concentrations. Thus, this information can be used to help design a process for PVIA and its stakeholders to use in establishing which HUC 12s or smaller subwatersheds are priorities for NPS management. The following sections provide some guidance on how this might be done.

When water quality data at all subwatersheds are absent, constituent specific HDI thresholds can be used. The HUC 12s could be prioritized and separated into categories based on the example (Figure 5A). The hypothetical categories could include:

- Preservation: HDI < 15%; These subwatersheds would be background or reference sites, as established by the lower end of the $90^{th}$ percentile confidence interval about the changepoint.
- Low priority: HDI from 15 to 25%; These subwatersheds would be a low priority for NPS management, as established by the lower end of the $90^{th}$ percentile confidence interval about the changepoint and the changepoint.
- Medium priority: HDI from 25 to 30%; These subwatersheds would be a medium priority for NPS management, as established by the changepoint and the upper end of the $90^{th}$ percentile confidence interval about the changepoint.
- High priority: HDI > 30%; These subwatersheds would be a high priority for NPS management, as established by the upper end of the $90^{th}$ percentile confidence interval about the changepoint.

Based on the LWW stream data, sites with HDI values less than the lower $90^{th}$ percentile confidence interval about the changepoint had low constituent concentrations (Figure 5A). The goal here would be to keep or preserve these HUC 12s to maintain existing water quality conditions. On the opposite end of the spectrum, streams with HDI values greater than the threshold, and even greater than the upper $90^{th}$ percentile confidence interval around the changepoint, generally had greater constituent concentrations. Thus, PVIA and stakeholders might focus efforts on HUC 12s with HDI values above the threshold (i.e., medium and high priority) because these catchments likely have the greatest restoration potential. Using the LULC for each individual HUC 12 (Table 1), this classification scheme shows the HUC 12s along the Fourche Maline River and Poteau River in Oklahoma (Figure 6) as areas of priority. In the absence of water quality data, this option can be a good method for selecting HUC 12s when developing the watershed management plan.

When water quality data are available, thresholds can be used differently to select HUC 12s based on measured constituent concentrations, as opposed to predicted values based on human development (Figure 5B). This method focuses on the average constituent concentrations on either side of the threshold. The HUC 12s could be prioritized and separated into categories based on the example in Figure 5B, where the hypothetical categories would include:

- Low priority: HUC 12s with constituent concentrations less than average constituent concentration below the threshold plus two standard deviations (horizontal dashed line or 0.05 mg $L^{-1}$ for TP; Figure 5B).
- Medium priority: HUC 12s with constituent concentrations greater than the horizontal dashed line but less than the average constituent concentration above the threshold (upper solid line or 0.08 mg $L^{-1}$ for

TP; Figure 5B).

- High Priority: HUC 12s with constituent concentrations greater than upper solid line or 0.08 mg $L^{-1}$ (Figure 5B).

As stated earlier, constituent concentrations below the thresholds were generally low. The horizontal dashed line (i.e., for TP 0.05 mg $L^{-1}$) provides a realistic bench mark for separating low and medium priority watersheds, as it represents the upper limits of baseline conditions for the constituents analyzed in this study. This method could be conducted for each constituent of interest,

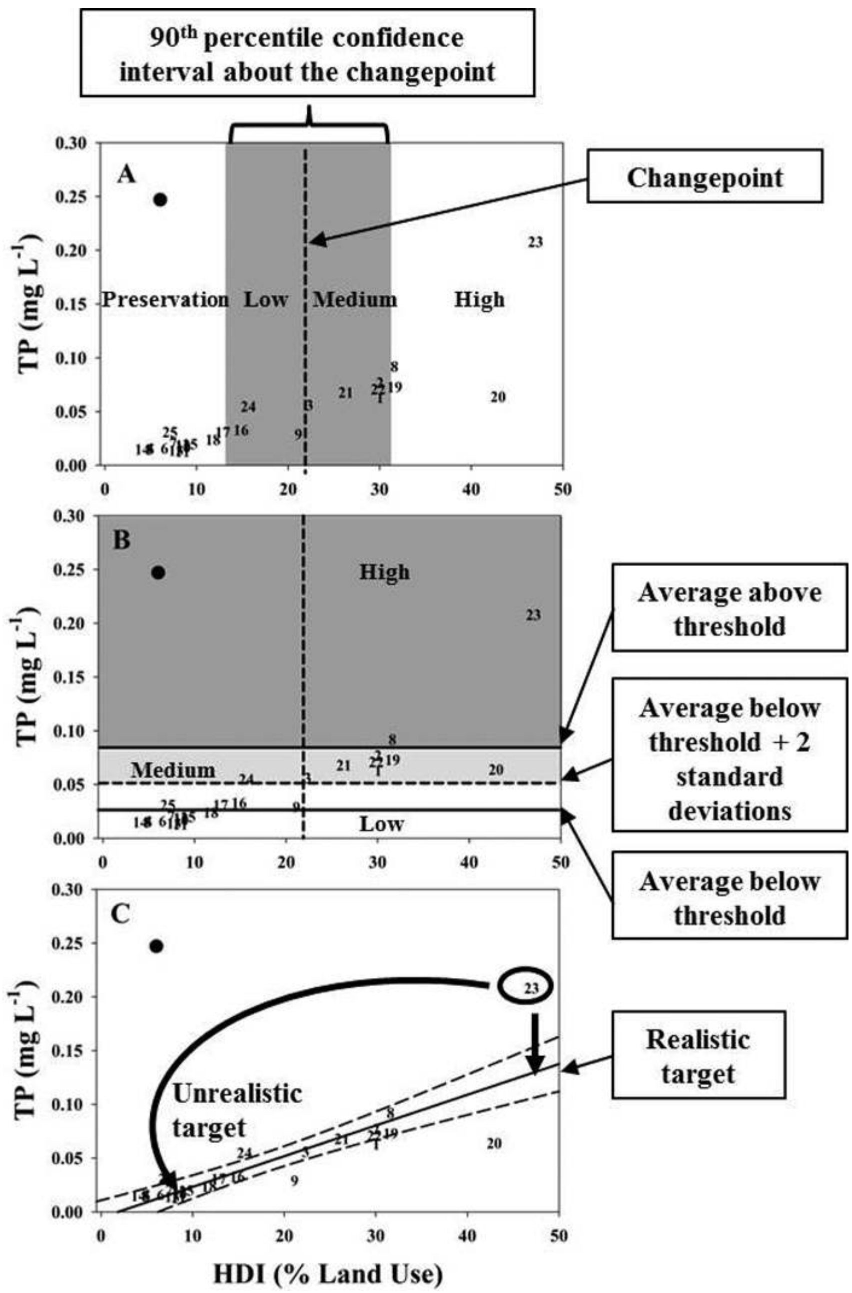


**Figure 5.** Potential methods using changepoints to identify watersheds for nonpoint source (NPS) management. Categorization of hydrologic unit code (HUC) 12s based on their human development index (HDI) value only (A); separation of HUC 12s based on measured water quality data (B). Linear models (regression line) represent realistic targets for improving water quality within a HUC 12 of a given HDI value (C).

resulting in the selection of constituent specific HUC 12s (Figure 7).

A weight of evidence approach may be used to combine HUC 12 priorities developed for individual constituents. Low, medium, and high priorities can be ranked 1, 2, and 3, respectively, for each constituent. Rankings for each constituent can then be added together to form a cumulative rank for each HUC 12. The cumulative ranks across all HUC 12s within the Oklahoma portion of the LWW were divided into five categories, where the subwatersheds shaded the darkest had the highest priority (Figure 7).

With this approach you must be mindful of the nested nature of the watershed, in that several subwatersheds are down river of one or more other subwatersheds. Water quality in an upstream subwatershed may result in higher than expected constituent concentrations, based on the level of human development. In such a case, it may be beneficial to compare subwatershed priorities identified by both methods.

Constituent concentrations change with land use, where the relation can often be described with a simple linear model (Figure 5C). Once subwatersheds have been prioritized, the goal should be to move the higher priority HUC 12s below the linear regression, which represents the average conditions at a given HDI level. Continued routine monitoring methods, such as establishing an annual geometric mean concentration point by collecting and analyzing 12 monthly baseflow samples, can be used to track improvements within the watershed. The geometric mean data point should be plotted against the most current land use information available, to reflect the changing LULC and HDI gradient. Once the data point shifts from above the line to below the line, then this site has reached its target concentration as defined by the original regression. However, it would be wise to make sure the HUC 12s have consistently changed priority categories (e.g., moved from high to low) over multiple years before assuming the target has been met.

## Discussion

In addressing the issue of eutrophication, it is important to focus on both point and NPS of nutrients. Point sources, such as municipal WWTPs, can be a major component of a watershed's overall nutrient load, especially for P (Haggard

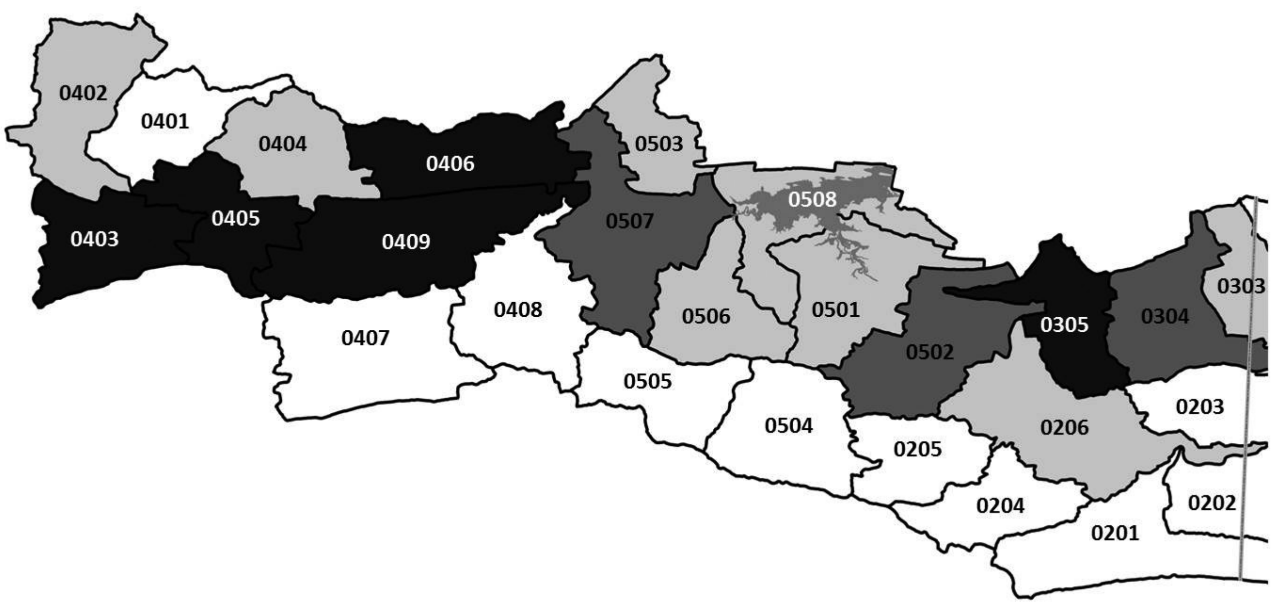


**Figure 6.** Potential prioritization of hydrologic unit code (HUC) 12 subwatersheds based on the threshold response of constituent concentration to the human development index (HDI); the priority for nonpoint source (NPS) management varies from lightest (preservation) to darkest (highest priority). HUC 12 subwatersheds are labeled with the last four digits of their HUC 12 code.

2010). However, improvements to these WWTPs have been successful in reducing the nutrient concentration in the effluent leaving treatment facilities and, as a result, reducing nutrient loads in receiving waters downstream of urban areas (Jaworski 1990; Haggard 2010; Scott et al. 2011). The contribution of nutrients to receiving waters from point sources is likely to continue to decrease as more stringent and widespread controls are put in place (Jarvie et al. 2013). However, decreasing nutrient inputs from point sources is only part of the solution.

Reducing nutrient loads associated with NPS pollution is often much more difficult than for point sources. In fact, over the past four decades, most NPS management plans have reported little to no improvement in surface water quality, even with extensive BMP installation throughout their watersheds (Meals et al. 2010; Jarvie et al. 2013). Low landowner participation resulting in poor distribution of BMPs, poor site selection, and inappropriate BMP selection for NPS pollution type are just a few factors that contribute to the failure of NPS management plans (Meals et al. 2010). Identification of critical source areas in need of BMPs can increase the success of NPS management plans.

Both proposed methods in the case study suggest subwatersheds along the Fourche Maline and Poteau Rivers were priority areas in need of BMPs, which aligned well with target areas previously highlighted in the LWW using the Soil

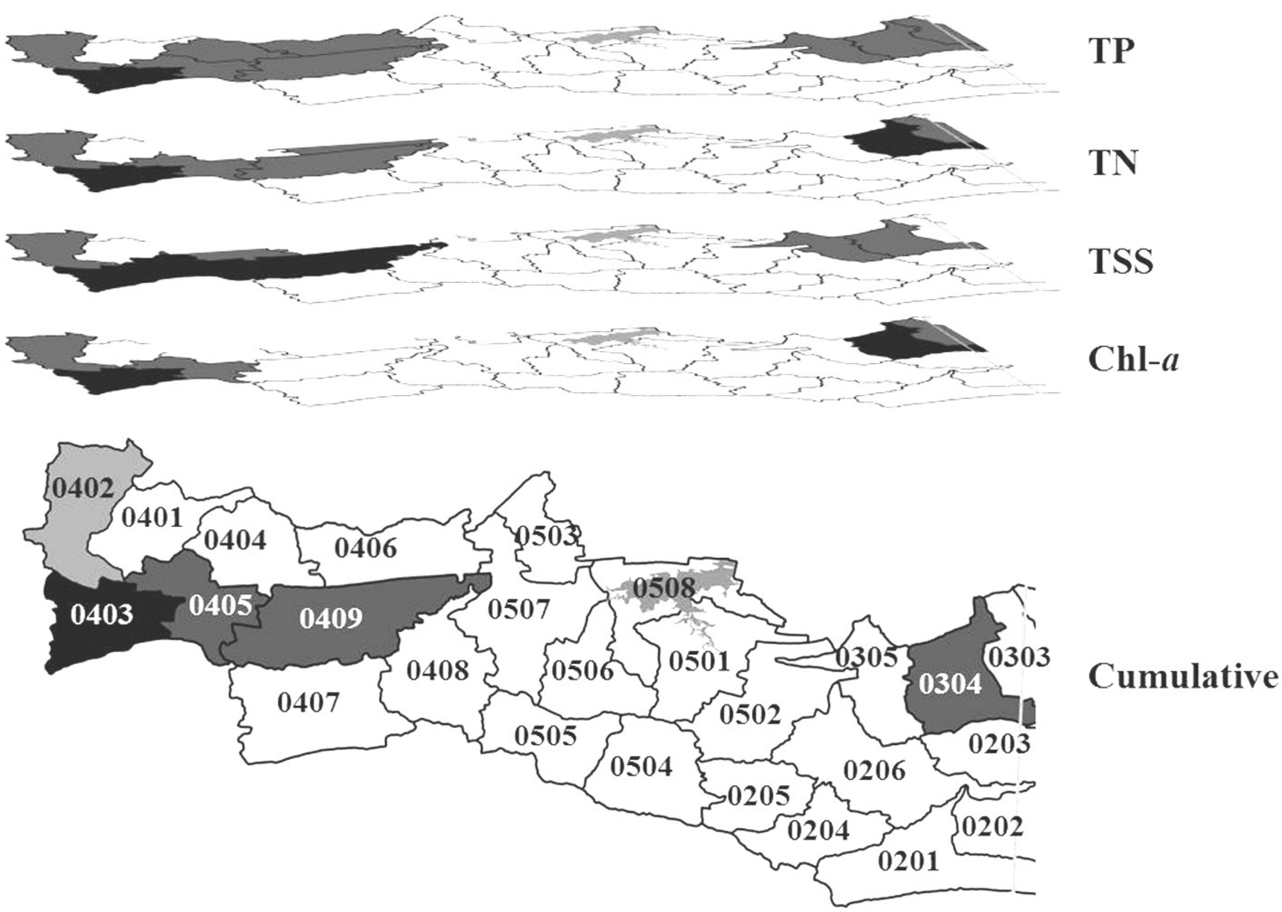


**Figure 7.** Potential prioritization of hydrologic unit code (HUC) 12 subwatersheds when chemical concentrations are available in streams. Priorities for individual constituents can be used to meet specific management needs, or priorities can be added across multiple constituents to prioritize subwatersheds based on a cumulative approach. For each constituent shown and for the cumulative map, the priority for nonpoint source (NPS) management varies from lightest (low priority) to darkest (highest priority). Each subwatershed is labeled with the last four digits of its HUC 12 code.

Water Assessment Tool (SWAT) (Busteed et al. 2009). Although the Oklahoma NPS Management Program Plan suggests that monitoring and assessment at the HUC 12 scale is the most effective means to identify water quality problems associated with NPS pollution (OCC 2014), this scale is coarse when compared to the hydrologic response units (HRU's) used in SWAT models. However, these methods can be applied at a finer scale within the higher priority watersheds to further isolate the specific areas in need of BMPs.

Across the LWW of Oklahoma, there was a significant threshold at roughly 25% human development, with catchments above this threshold having nutrient and sediment concentrations greater than catchments below this threshold. However, in an analysis of Arkansas watersheds, the threshold HDI where nutrients and sediments began to increase was closer to 50% (McCarty et al. 2018), suggesting that these watersheds were more resilient to increasing land use. This suggests that, while there is variability between watersheds, this approach is applicable to other watersheds as long as there is a gradient in human development across the watershed. For instance, this method would likely not work in areas heavily developed for agriculture such as the Mississippi River Delta and areas in the Midwest with greater than 90% agriculture. Additionally, these methods require that baseflow constituent concentrations relate to human development in a predictable way, as seen in this case study and in other areas outside of Arkansas and Oklahoma (e.g., see Jones et al. 2001; Buck et al. 2004). Application of this method in other watersheds also requires that the threshold responses between constituent concentrations and HDI are developed for each specific watershed.

While these methods can assist watershed managers in identifying priority subwatersheds for the development of NPS management plans, determining the success or failure of these plans requires assessment at the appropriate spatial and temporal scales. Most often BMPs are installed at edge-of-field or small watershed scale, but then assessed for effectiveness at the sub-basin or larger watershed scale, resulting in difficulties in detecting BMP effectiveness (Mulla et al. 2008). Nutrient hot spots throughout larger watersheds that are responsible for the storage and eventual release of nutrients from riparian buffers, wetlands, and stream and lake sediments, likely mask the effect of reduced nutrient export from the landscape following the implementation of BMPs (Haggard et al. 2005; Ekka et al. 2006; Jarvie et al. 2013). So, while improvements in water quality may result from BMP implementation, they may not be detected, especially if monitoring is occurring further down in the watershed than where the management practices are occurring.

The issue of eutrophication in streams and lakes arises over decades of intensive agricultural practices and increasing human development, and cannot be solved overnight. Nutrient management plans that reduce or eliminate fertilizer application to fields can take up to 50 years or more to cause reductions in $NO_3^-$, due to the long residence time of $NO_3^-$ in groundwater (Bratton et al. 2004). While P is more likely to stay in the soil, it can take a decade or more to draw down soil P reserves through removal in crop biomass (Zhang et al. 2004; Hamilton 2012). Additionally, many BMPs require time to establish; for instance, it can take up to a decade for trees in riparian buffer strips to become fully established and start removing nutrients from subsurface flow (Newbold et al. 2010). Sediment-bound P in the fluvial channel is not mitigated by edge-of-field BMPs (Dunne et al. 2011), and can be a substantial source of P to the water column (Jarvie et al. 2005). Lag times associated with stream bed sediments are highly variable and depend on flow regime, hydromorphology, and sediment retention (Jarvie et al. 2006), but sediments can take 50 years or more to be flushed from larger watersheds (Clark and Wilcock 2000). These pools of N and P constitute legacy nutrients that can contribute to the system for decades after BMPs have been put in place.

Many of the issues associated with long lag times between BMP implementation and improvements to water quality at the larger watershed scale are reduced in smaller watersheds. In general, improvements to water quality should be detected in smaller watersheds (e.g., < 15 km$^2$) faster than larger watersheds because monitoring efforts are likely closer to the source and the mitigation efforts (Meals et al. 2010). Additionally, water quality in smaller streams tends to respond more quickly and directly to watershed alterations

(Lowe and Likens 2005). Thus, targeting smaller watersheds for water quality monitoring following BMP installation should provide watershed managers a better indication of the effectiveness of implemented BMPs due to a shorter lag period between installation and observed changes in water quality.

## Conclusion

Managing NPS pollution can be difficult, and the results of such efforts may take several decades or longer to be fully realized at the larger watershed or basin scale. The first issue for watershed managers is to identify or prioritize the areas within the watershed in need of mitigation. In the case study of the LWW, we found that in lieu of generating calibration and validation data needed for watershed models, baseflow water quality monitoring at the subwatershed scale provided an effective way of identifying areas in need of BMPs, producing recommendations similar to those generated by SWAT models (Busteed et al. 2009). Once BMPs are implemented, the effects of legacy nutrients that have built up on the landscape and in the fluvial channel can mask the effects of improvements made in the watershed. However, focusing monitoring efforts at the subwatershed scale can provide an early assessment of the effectiveness of BMPs implemented.

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