CONSTITUENT LOADS AND TRENDS IN TWO NORTHWEST ARKANSAS NONPOINT SOURCE MANAGEMENT PROGRAM PRIORITY WATERSHEDS

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Executive Summary

The Arkansas Department of Agriculture – Natural Resources Division (ANRD) has identified two Northwest Arkansas hydrologic unit code (HUC) 8 watersheds for prioritization by the Nonpoint Source (NPS) Management Program. The Upper Illinois River Watershed (UIRW; 11110103) encompasses the Illinois River from its headwaters to the state line with Oklahoma and has been the subject of interstate disputes over water quality for decades. The Upper White River Basin (UWRB; HUC 11010001) includes Beaver Lake in its borders, the drinking water source for 1 in 6 Arkansans. Nonpoint source pollution concerns in these watersheds are excess nutrients from agriculture and sediment from changes in land use/land cover (LULC).

Local, state, and national groups, including the NPS Source Management Program, have invested in education, best management practices, and streambank restoration in the UIRW and UWRB. These watersheds are also subject to regulation on the application of poultry litter as fertilizer and permitted limits on phosphorus discharge from point sources, such as municipal wastewater treatment plants (WWTP). Long-term water-quality monitoring data are necessary to identify whether these interventions are having an effect on water quality. The lag time before water-quality response can be considerable. Robust data are also needed to guide where additional resources should be targeted, or to identify potential emerging water quality concerns.

The objectives of this project (19-1100) were to collect water samples at 13 sites to estimate constituent loads and understand how water quality changed in these priority watersheds over time. This project was a continuation of a series of NPS projects since 2009. Sampling sites were selected to represent a variety of LULC characteristics in the watershed, as well as important tributaries to the river mainstems. All sites are located at existing U.S. Geological Survey (USGS) stream gaging stations. At each site, ~31 water samples were collected during each project year (October 1 through September 30; 2019 - 2022) at base flow and a range of surface runoff conditions. Water samples were analyzed for concentrations of nitrate-nitrogen nitrogen (NO_3-N), total nitrogen (TN), soluble reactive phosphorus (SRP), total phosphorus (TP), chloride (Cl), sulfate (SO_4), and total suspended solids (TSS).

We combined water quality data from the current and past projects for a period of analysis of 2010 – 2022 at most sites. We integrated USGS average daily streamflow data and estimated annual loads and average concentrations, using the statistical modeling algorithm Weighted Regressions on Discharge, Season, and Time (WRTDS). The WRTDS model also estimates flow-normalized (FN) concentrations and loads, with the influence of random variability in streamflow removed. Trends in FN values were evaluated for statistical significance using the WRTDS Bootstrapping Method.

Annual mean FN concentrations and total FN loads varied through time and between sites in both the UIRW and UWRB. In particular, loads increased across sites as watershed area and, therefore, streamflow increased. The magnitude and temporal patterns in concentrations differed between water quality constituents. For the UIRW, these patterns generally corresponded to gradients in watershed characteristics that suggested greater or less human influence. The UWRB sites had more similar watershed characteristics, however, so potential watershed effects were more complex to decipher.

Trend analysis suggested that phosphorus (TP and SRP) concentrations have decreased over the last 15 years throughout the UIRW and UWRB. In the UIRW, sites with point-source influences also had decreasing phosphorus loads. Phosphorus reductions represent a major water quality gain for both watersheds, but FN concentrations at IR59, the Illinois River site at the border with Oklahoma, remain approximately two times greater than the Oklahoma Scenic River Criteria. Concentrations decreases have also not yet resulted in phosphorus load reductions in the UWRB.

Other potential water quality gains included decreasing nitrogen concentrations, loads, or both in the Illinois River at Savoy, in the Baron Fork, and in the West Fork and White River above Beaver Lake. However, nitrogen levels have not had widespread change over the last 15 years to the same degree as phosphorus. Substantial nitrogen reductions will likely require strategies specifically tailored to addressing the unique sources, sinks, and biogeochemical cycling of nitrogen.

A few instances of potentially degrading water quality were observed in the UIRW. Chloride concentrations were potentially increasing at the most urban sites. Increasing SO_4 concentrations and loads on Osage Creek are likely tied to the use of aluminum sulfate (alum) to remove excess phosphorus from municipal WWTP discharge. But, SO_4 concentrations were also potentially increasing at Mud, suggesting other human activities, such as use of detergents, are also a likely source.

For the majority of site-constituent combinations, trend analysis suggested no change in water quality. Stable water quality is a positive outcome for watershed management activities in the UIRW and UWRB. In particular, the overall limited changes in TSS suggest that watershed-scale erosion is not worsening. It appears that NPS management strategies targeted to mitigating accelerated erosion risks in a rapidly urbanizing watershed have been successful. However, significant investment in NPS pollution reduction strategies for mitigating pasture LULC and deforestation have not yet shown a clear water quality return.

The relative loading intensity for individual sites in each watershed was shown using yields, which were 2022 FN loads divided by the watershed area. Yields show the load produced, on average, for each unit of watershed area. Site-specific yields were compared to the yield of the total watershed area. Depending on the constituent, site-specific yields differed considerably from the total watershed yield. For the UIRW, the largest differences were in Cl, SO_4, and nitrogen compounds and followed the human influence gradient. For the UWRB, yields varied, but were also more similar between sites. At WFWR and WEC, four constituent yields were greater than the total watershed, as well as three constituents at Richland. However, these constituents were not the same across sites in the UWRB.

Spatial patterns in yield variability within the UIRW and UWRB have implications for watershed management. In the UIRW, trend analysis showed that point-source controls have likely contributed substantially to progress on phosphorus over the last 15 years. But, watersheds with municipal WWTPs still yield Cl, SO_4, nitrogen, and, in some cases, phosphorus, at disproportionately large rates. Alternately, phosphorus yields at Savoy and TSS yields at Mud and Spring suggested that continued NPS activites can also make a difference in the overall watershed load. For the UWRB sites, similarities between watershed characteristics make it challenging to differentiate NPS and point-source contributions. But, specific sub-watersheds clearly contribute more intensively to the total watershed load. Most notably, the West Fork remains a hotspot for sediment export, as well as Richland Creek. War Eagle Creek was the only UWRB sub-watershed with a greater yield of nitrogen compounds compared to the total watershed yield. Future non-point source management activities can target these areas, or areas with similar watersheds.

Chapter 1. Upper Illinois River Watershed

Introduction

The Upper Illinois River Watershed (UIRW) is located in Northwest Arkansas and is a priority hydrologic unit (HUC) 8 watershed for the Arkansas Department of Agriculture – Natural Resources Division (ANRD) Nonpoint Source (NPS) Pollution Management Program. The biggest NPS challenges for the UIRW are excess nutrients and sediment (ANRD, 2018). Animal agriculture is the primary NPS for excess nutrients in the watershed, especially phosphorus. Rapid urbanization and other land use changes have led to accelerated soil erosion and sediment export. Because phosphorus tends to associate strongly with soil particles, increased sediment transport in runoff is also a pathway for excess phosphorus to enter and build up in the waterbodies of the UIRW.

The Illinois River flows from its headwaters in Northwest Arkansas into Oklahoma, where it is designated a Scenic River (82 OS §1451-1471). Its major Northwest Arkansas tributaries drain areas with significant pasture and urban land use, as well as point-source dischargers. The magnitude of phosphorus concentrations and loads in the Illinois River has been a source of interstate conflict for decades (Haggard et al., 2017). The State of Oklahoma has promulgated a Scenic Rivers numeric criterion for total phosphorus (TP) of 0.037 mg/L to protect recreational use and prevent nuisance algal growth (King, 2016). While the assessment methodology for the standard is still in development, the ambient TP concentration in the Illinois River at the state line will be expected to meet this requirement in the future.

The State of Arkansas has taken steps to address excess phosphorus and mitigate land use changes in the UIRW over the last decades. The UIRW is designated as a Nutrient Surplus Area (AR Code § 15-20-1104), requiring controls on the application of phosphorus-rich poultry litter as fertilizer for pastures. The NPS Management Program and local watershed groups, such as the Illinois River Watershed Partnership, have invested in education, best management practices (BMPs), and streambank restoration. Additionally, point sources in the watershed, such as wastewater treatment plants (WWTPs), have been required to upgrade and adapt to lower permitted discharge limits for phosphorus.

The Arkansas Water Resources Center (AWRC) has used consistent methodologies to monitor water quality in the UIRW since 2009 through contracts with the NPS Pollution Management Program. Robust data are necessary to establish baseline conditions and detect potential improvements resulting from the implementation of NPS projects, state regulations, and other watershed management activities. Long-term data are essential because the lag time between NPS project activities and the water quality response can be years to decades (Meals et al., 2010). These data are also needed to determine if, when, and where water quality is degrading when land use or other watershed changes have occurred.

The current study (NPS Pollution Management Program project 19-1100) objectives were to:

1. continue water sample collection throughout the UIRW for an additional three years,
2. estimate annual loads for the cumulative period of record,
3. evaluate trends in water quality and loading to allow quantitative assessment of response to mitigation and management in the UIRW.

Methods

Site Information

The AWRC sampled eight locations in the UIWR on the current project (19-1100), which are all located at U.S. Geological Survey (USGS) stream gaging stations (Figure 1.1). Three site are located on the river mainstem (Savoy, IR59, and Watts), four on upper watershed tributaries (Mud on Mud Creek, OC112 and Osage on Osage Creek, and Spring on Spring Creek), and one on Baron Fork, a lower watershed tributary to the Illinois River. Within the greater Illinois River Basin, the sites are positioned from upstream to downstream: Savoy, Mud, OC112, Spring, Osage, IR59, Watts, and Baron. The Baron Fork's confluence with the Illinois is located downstream of Watts, the most downstream river mainstem site.

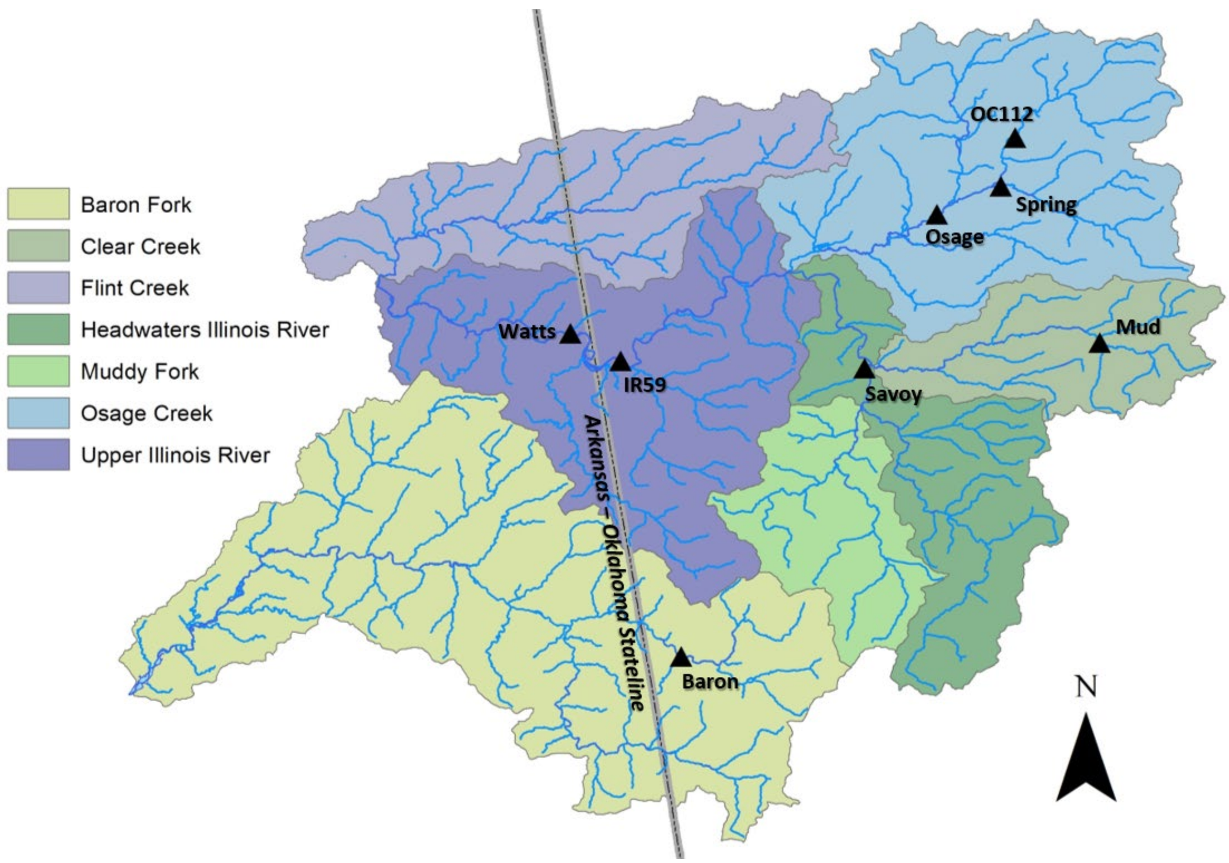


Figure 1.1. Monitoring locations in the Upper Illinois River Watershed.

Sites have a range of watershed sizes and land use-land cover (LULC) profiles (Table 1.1), including gradients of forest-pasture mix (Baron and Savoy) to highly urbanized (Mud, OC112, and Spring). Osage Creek and Spring Creek directly receive discharge from municipal WWTPs. The Rogers, AR WWTP discharge is upstream of OC112 and the Northwest Arkansas Conservation Authority discharges upstream of Osage. Spring Creek receives discharge from the Springdale, AR WWTP before flowing into Osage Creek upstream of Osage. Savoy also receives municipal WWTP discharge from the Fayetteville, AR Westside Facility via Goose Creek. Segments of the Illinois River were listed as impaired for sulfates, turbidity, and E. coli in Arkansas's 2020 draft 303(d) list (ADEQ, 2020), while a segment of the Baron Fork was listed as impaired for critical season dissolved oxygen levels.

Water Sample Collection

Water samples were collected manually from bridge access locations. Samples were collected using either an alpha-style horizontal sampler or a Kemmerer-type vertical sampler from a single representative point in the stream (i.e., near the vertical centroid of flow). The sampling approach was designed to capture both flow-driven and seasonal variation in constituent concentrations. On average, ~31 samples were collected per site each project year during the current (October 2019 – September 2022) project period. Base flow samples were collected at least once monthly. Whenever possible, stormflow was sampled at least monthly with the goal of capturing all the largest storm events each year. All samples were collected according to an approved quality assurance project plan (QAPP; QMP # 21-052). Sample collection intervals, methods, and design were consistent with preceding projects.

Table 1.1. Site information for the eight AWRC monitoring locations in the Upper Illinois River Watershed. The period of analysis is based on water years (i.e. October 1 – September 30), where the water year is identified by the calendar year of the last nine months (January 1 – September 30) of the water year. The watershed land use/land cover information is adapted from the National Land Cover Database (NLCD), 2019 and was obtained using modelmywatershed.com.

Site	Latitude	Longitude	USGS Gage	Period of Analysis	Watershed Area (km^2)	% Urban[1]	% Forest[2]	% Pasture[3]
Baron	35.88	-94.4864	07196900	2010-2022	105.2	5.49	51.31	43
IR59	36.10861	-94.5333	07195430	2010-2022	1489.6	21.35	31.22	46.81
Mud	36.12281	-94.1626	071948095	2016-2022	43	66.86	15.72	17.17
OC112	36.28147	-94.228	07194880	2016-2022	89.9	66.38	6.88	26.6
Osage	36.22194	-94.2883	07195000	2010-2022	336.8	42.28	12.15	45.38
Savoy	36.10306	-94.3444	07194800	2010-2022	432.6	10.38	39.24	49.65
Spring	36.24378	-94.2391	07194933	2013-2022	90.7	49.77	11.29	38.82
Watts	36.13008	-94.5722	07195500	2010-2022	1632.1	20.11	31.72	47.52

[1] % Urban is the sum of all developed land categories, as well as barren land
[2] % Forest is the sum of all forest categories, as well as shrub/scrub
[3] % Pasture is the sum of the pasture/hay and grassland/herbaceous categories

Sample Analysis

All water samples were stored on ice after collection and returned promptly to the Arkansas Water Resources Center Water Quality Lab (WQL). Samples were analyzed for concentrations (mg/L) of nitrate-nitrogen (NO_3-N), total nitrogen (TN), soluble reactive phosphorus (SRP), total phosphorus (TP), total suspended solids (TSS), chloride (Cl), and sulfate (SO_4) using standard analytical procedures for the analysis of water and wastewater and following the approved QAPP. The WQL is certified by the Arkansas Department of Energy and Environment - Environmental Quality Division (ADEQ) for the analysis of all the measured parameters in water. The WQL used standard quality assurance and quality control (QA/QC) practices, such as blanks, duplicates, and spikes.

Streamflow Record

The monitoring sites are located at active USGS stream gaging stations. A high-quality streamflow record is essential for load estimation. Adjusting constituent concentrations and loads for streamflow variability also enhances our understanding of how these values vary through time. Adjusting for flow variability prior to trend analysis makes change over time more readily detectable. Mean daily streamflow (cfs) and gaged watershed area (km^2) data were obtained through the USGS National Water Information Systems (NWIS; USGS, 2022) for all gages at the end of the project period.

Mud's daily streamflow record had several missing dates. We determined by comparing to other sites that base flow conditions applied on these dates, except for one. We made a best estimate of average daily streamflow to fill in all missing dates. In particular, not including the date when a large, region-wide storm even occurred could have underestimated annual load estimates for that year at Mud. For baseflow, we averaged streamflow on the day preceding and following the missing date(s). For the storm event, we estimated that streamflow at Mud was ~65% of streamflow observed that day at OC112. Average daily streamflow at Mud was 60 – 70% of that observed at OC112 during high-flow events in the week before and after the missing date.

Weighted Regressions on Time, Discharge, and Season (WRTDS)

Constituent loads and trends were calculated using the Weighted Regressions on Time, Discharge and Season (WRTDS) statistical modeling algorithm developed by the USGS (Hirsch et al., 2010; Sprague et al., 2011). The method considers the influence of time, discharge, and season in estimating loads and detecting trends in water quality at a site. The method removes the influence of random variations in streamflow that make it difficult to discern patterns in constituent concentrations and loads. We carried out the WRTDS analysis using the statistical software R, version 4.1.3, (R Core Team, 2021) paired with the EGRET package (Hirsch and DeCicco, 2015).

The WRTDS algorithm uses paired water quality and streamflow data as a calibration dataset for describing the water quality-streamflow relationship through time. This relationship is described with the following equation, where c is concentration, q is streamflow, T is time, and ε and β values are the estimates of regression standard error and model coefficients:

$$ln(c) = \beta_0 + \beta_1 q + \beta_2 T + \beta_3 \sin(2\pi T) + \beta_4 \cos(2\pi T) + \varepsilon$$

This underlying equation is well-established for the estimation of loads (Helsel et al., 2020). But, WRTDS is unique from other common load estimation tools because the parameters of the relationship are dynamic through time, with unique estimates of the regression coefficients and standard error each day. The model parameters are not stored and are not useful for global estimation of concentrations or loads. The WRTDS algorithm is a smoothing procedure that should not be used to extrapolate outside the period of record of paired water quality and streamflow data.

Concentrations, streamflow, and loads are often not normally distributed, so WRTDS estimates the daily time series of concentrations and other outputs in log-space. The WRTDS algorithm uses a bias correction factor when transforming the log unit concentration estimates back to standard concentrations (i.e., mg/L).

From each unique daily model, the WRTDS algorithm provides a daily estimate of constituent concentrations (mg/L) for the entire streamflow record. These concentrations are then the basis for estimates of constituent loads (kg/d) after multiplying by mean daily streamflow. Flow-normalized (FN) concentrations and FN loads are also calculated by multiplying by the probability distribution function for streamflow. Standard concentrations and loads are the actual estimated value for a given day, while FN concentrations and FN loads are corrected for the influences of variations in water quality and loads arising from random day-to-day variations in streamflow

These daily time series can be used to determine monthly, annual, or longer time scale values, either by summing loads or averaging concentrations. We based annual values on water years, which run from October 1 – September 30. A water year is denoted by the calendar year of its last nine months (i.e. January 1 – September 30), but begins on October 1 of the preceding calendar year. For example, the 2010 water year began on October 1, 2009 and ended on September 30, 2010.

In order to compare the contribution of sites to watershed loading, we calculated constituent yields for each site by dividing the FN load by watershed area. Site-specific loads are not directly comparable because streams with larger watershed areas are expected to transport greater loads. Conversely, streams with smaller watershed areas carry smaller loads. Watershed yields can be compared between sites, however, and show which areas of the greater watershed contribute most to total constituent export to downstream waters.

We used the WRTDS Bootstrap Test in the EGRETci package (Hirsch et al., 2015) to determine the statistical significance of potential changes in FN concentrations and loads over time. The p-value of the WRTDS Bootstrap Test describes the probability that a pattern over time is random. We considered $p<0.05$ to suggest a highly likely trend (i.e. <5% probability of a random pattern, or a ≥95% probability of a real trend) and $p<0.10$ to suggest a likely trend (i.e. <10% chance of random pattern, or a ≥90% probability of a real trend). Sites and water quality constituents with $p \geq 0.10$ were considered likely not changing.

Results and Discussion

Annual Concentrations and Loads

The annual time series of concentrations, loads, FN concentrations, and FN loads are available in the Appendix to this report for all UIRW monitoring locations. Annual values are provided for each full water year (i.e. October 1 – September 30) in each site's period of analysis. Within this report, we focus on results for FN concentrations and FN loads at select time points (2010, 2016, and 2022) and watershed locations. The select sites communicate a watershed perspective on water quality and loading variability and have all been monitored since calendar year 2009. Baron and Savoy have rural watersheds in a mix of pasture and forest with primarily NPS human influences. But, Savoy has a point-source discharger, and the watershed is somewhat more urban. Osage has a large human footprint, with greater urban LULC and three point-source dischargers. Finally, IR59's location at the Oklahoma state line shows where we stand in efforts to bring ambient water quality in line with Oklahoma's Scenic River criteria.

Mean annual FN concentrations (Figure 1.2) and total annual FN loads (Figure 1.3) in the UIRW varied both through time and spatially within the watershed. Variability in FN concentrations was clearly observed between the time points and between sites. The degree of this variation differed between constituents, but tended to reflect the gradient of human influence in the watershed. Concentrations were typically least for Baron but greater for Savoy, IR59, and, especially, Osage.

In contrast, the dominant source of variability in FN loads was between sites proportional to watershed area, consistent with preceding analysis for these sites (Scott and Haggard, 2018). This watershed loading pattern reflects that streamflow increases with watershed area. Streamflow, the dominant component of load, varies by orders of magnitude as watershed area increases, while concentrations tend to vary less, even in response to major differences in watershed characteristics. Nevertheless, site-specific interannual variability was also observed in FN loads and most often followed similar patterns to FN concentrations.

We observed three patterns in site-specific interannual variability. First, some site and constituent combinations moved toward smaller FN concentrations or loads across all the time points. Most notably, the FN concentrations of phosphorus compounds (SRP and TP) consistently stepped down in magnitude at all sites, as well as FN loads at Osage and IR59. Differences in FN concentrations and loads of phosphorus compounds were as much as 50% less when comparing 2010 and 2022, such as at Osage. The FN concentrations and loads of TSS at Osage also followed this pattern over time, with a >60% difference between 2010 and 2022. The magnitude of interannual variability was less, but this pattern also applies to NO_3-N, TN, and Cl at Savoy, as well as Cl at IR59 and Baron.

Conversely, other site and constituent combinations moved toward larger FN concentrations or loads across all the time points. At Osage, both the FN concentrations and loads of SO_4 consistently stepped up in magnitude, with a ~25% difference in both between 2010 and 2022. Both FN concentrations and loads of Cl at Osage followed a similar pattern to SO_4, though the differences between time points were less.

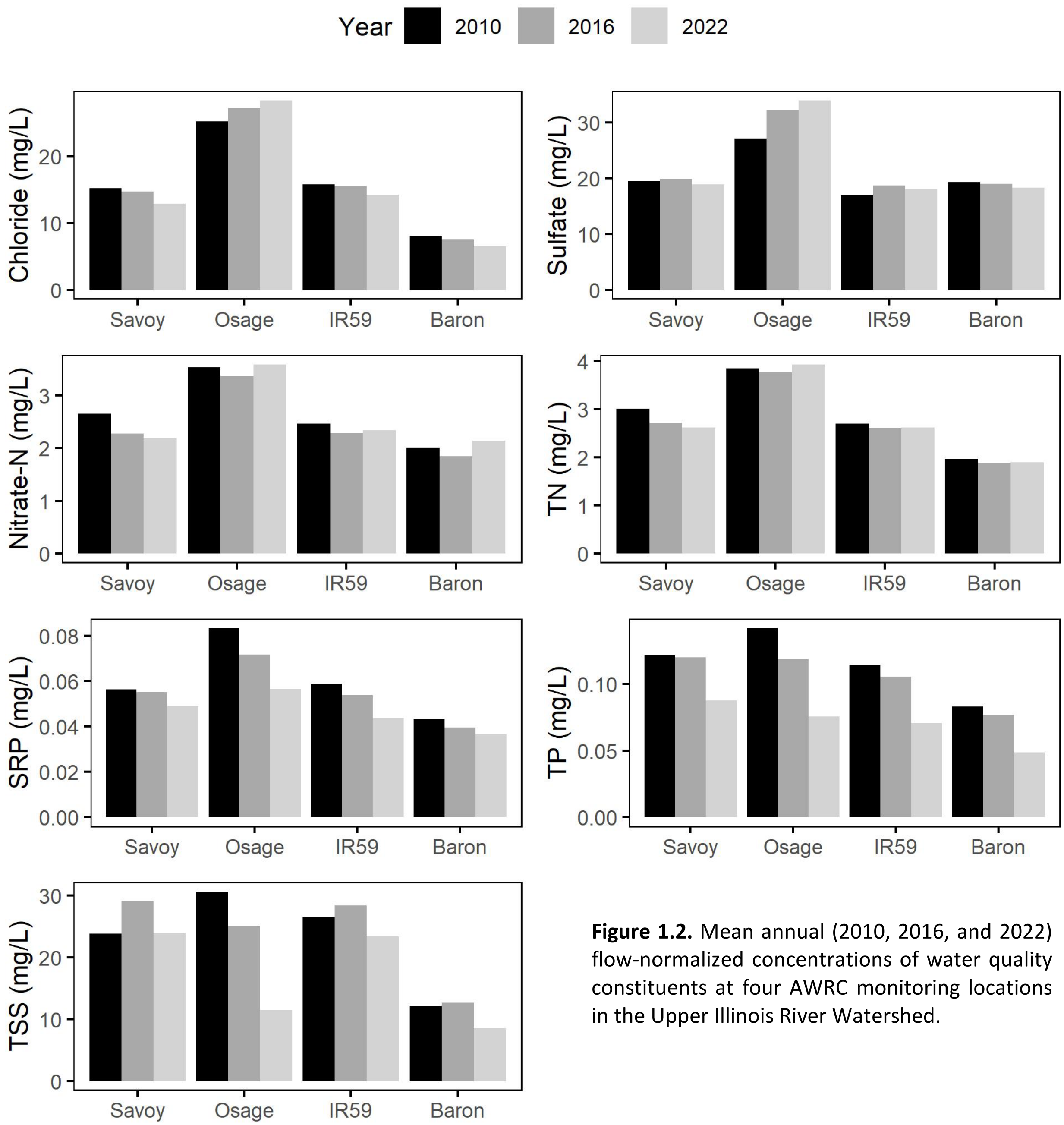


Figure 1.2. Mean annual (2010, 2016, and 2022) flow-normalized concentrations of water quality constituents at four AWRC monitoring locations in the Upper Illinois River Watershed.

Lastly, other site and constituent combinations showed no consistent trajectory in variability from year to year, or variability from year to year stayed within a more narrow range. The FN concentrations and loads of nitrogen compounds (NO_3-N and TN) varied minimally, especially compared to phosphorus compounds. Observed variations in nitrogen compounds tended to not have a consistent direction, or only weak signals of a specific trajectory, such as FN loads at IR59 and Baron. These patterns also describe variation in Cl, SO_4, and TSS, but with some exceptions, such as the patterns described above for Osage.

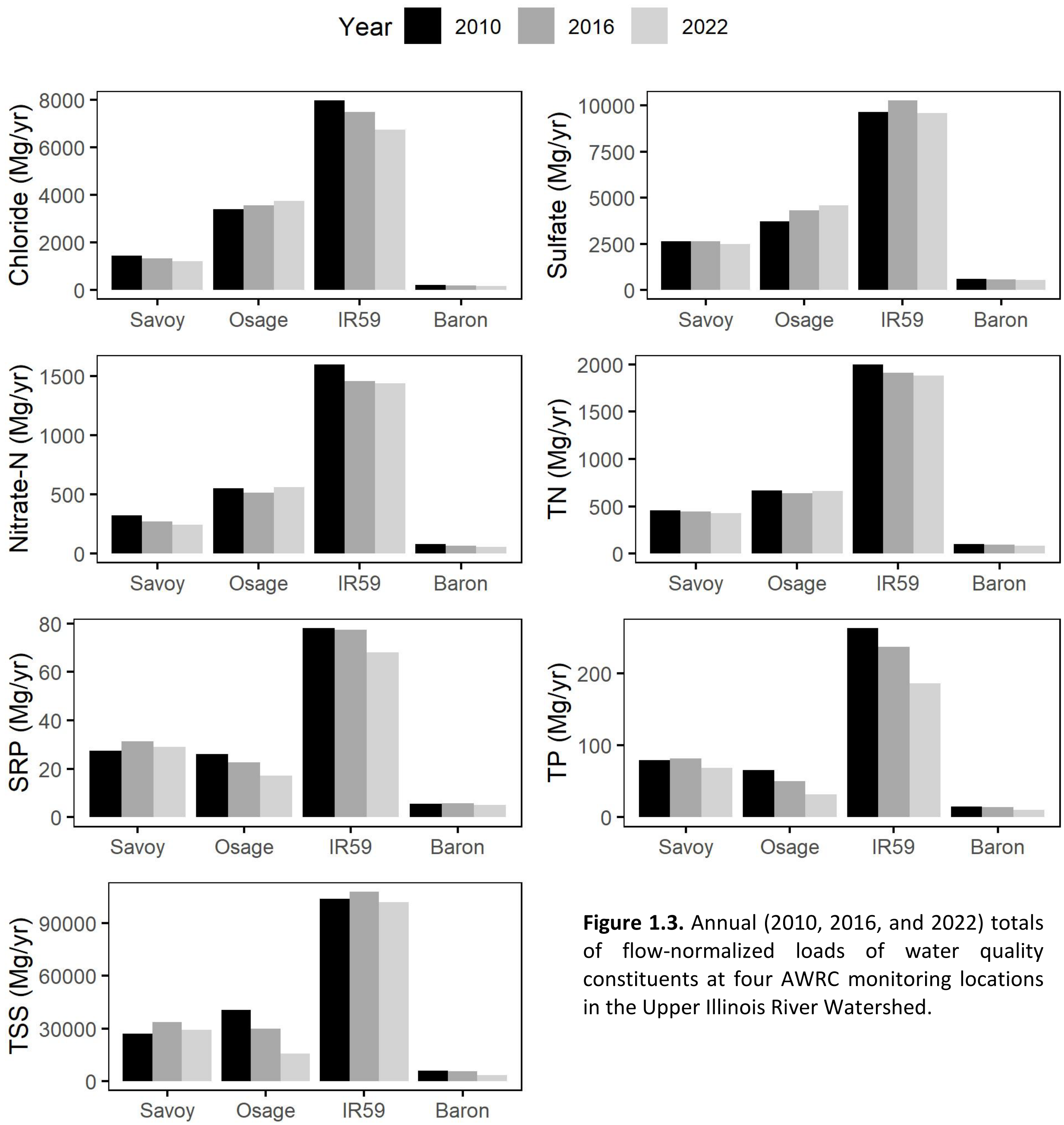


Figure 1.3. Annual (2010, 2016, and 2022) totals of flow-normalized loads of water quality constituents at four AWRC monitoring locations in the Upper Illinois River Watershed.

Water Quality Trends

Interannual variability is a normal characteristic of environmental datasets and was expected in the mean annual FN concentrations and loads. Considerable interannual variability is also consistent with estimates from preceding studies in the UIRW (Scott and Haggard, 2018). The results of trend analysis on FN concentrations (Table 1.2) and FN loads (Table 1.3) over time show whether the observed temporal variability is part of a consistent water quality trend over time, or likely due to random variability.

Table 1.2. Trend analysis results on flow-normalized concentrations for all the UIRW monitoring locations. The period of analysis differs between sites and is given below the site name. For all sites, the period of analysis begins with the first full water year (October 1 – September 30) and ends with water year 2022.

	Mud (2016-2022)	OC112 (2016-2022)	Spring (2013-2022)	Osage (2010-2022)	Savoy (2010-2022)	IR59 (2010-2022)	Watts (2010-2022)	Baron (2010-2022)
Analyte	% change in flow-normalized concentrations							
Cl	3.7*	3.1*	-1.9**	No change	-1.3*	No change	No change	-1.6**
SO_4	1.4*	4.7**	No change	2.1**	No change	No change	0.71**	No change
NO_3-N	No change	No change	No change	No change	-1.4*	No change	No change	No change
TN	No change	No change	No change	No change	No change	No change	No change	No change
SRP	-5.5**	No change	-3.8*	-2.7**	No change	-2.1*	-2.2**	No change
TP	-11**	-6.6*	-4.8**	-3.9**	No change	-3.2*	-3.4**	-3.4*
TSS	No change	No change	No change	No change	No change	No change	No change	No change

* denotes trends that are "likely" (i.e. p<0.10)
** denotes trends that are "very likely" (i.e. p<0.05)

Table 1.3. Trend analysis results on flow-normalized loads for all the UIRW monitoring locations. The period of analysis differs between sites and is given below the site name. For all sites, the period of analysis begins with the first full water year (October 1 – September 30) and ends with water year 2022.

	Mud (2016-2022)	OC112 (2016-2022)	Spring (2013-2022)	Osage (2010-2022)	Savoy (2010-2022)	IR59 (2010-2022)	Watts (2010-2022)	Baron (2010-2022)
Analyte	% change in flow-normalized loads							
Cl	6**	4**	-1.8**	No change	-1.4*	No change	No change	-2**
SO_4	No change	4.4**	No change	1.9**	No change	No change	No change	No change
NO_3-N	8.5**	No change	No change	No change	-2**	-0.88**	No change	-2.5*
TN	No change	No change	No change	No change	No change	No change	No change	-1.6*
SRP	No change	No change	-3.2*	-2.8**	No change	No change	No change	No change
TP	No change	No change	-4.7*	-4.3*	No change	No change	-2.7*	No change
TSS	No change	No change	No change	No change	No change	No change	No change	No change

* denotes trends that are "likely" (i.e. p<0.10)
** denotes trends that are "very likely" (i.e. p<0.05)

Phosphorus

Trend analysis results suggested near watershed-wide decreases in phosphorus in the UIRW over the last 15 years. Decreases in FN concentrations of TP ranged from ~3 – 11% annually and were considered very likely ($p<0.05$) for Mud, Spring, Osage, and Watts and very likely ($p<0.10$) for OC112, IR59, and Baron. No change was detected for Savoy ($p>0.10$). These decreases in concentration have also led to likely declines (~3 – 5% annually) in the FN load of TP at three sites (Spring, Osage, and Watts). The FN concentration of SRP has also potentially been decreasing (~2 – 6% annually) at five of the eight sites. However, the downward trajectory observed in timepoint comparisons of 2010, 2016, and 2022 at Savoy, IR59, and Baron was not a significant trend ($p>0.10$). Downward trends in FN loads of SRP were likely (Spring, ~3% annually) to very likely (Osage, ~3% annually) at two sites.

The near watershed-wide downward trend in phosphorus is a major water quality gain for the UIRW. Watershed patterns suggest that both NPS and point-source reductions have contributed to decreases. With a few exceptions, the rate of decrease in FN concentrations of TP did not vary considerably between UIRW sites, even though very different watershed characteristics were represented (i.e. presence or absence of point sources and gradients of human influence on LULC). The rate of decline in TP concentrations at Baron was within range of decline at Osage, suggesting that watershed management activities targeted to NPS pollution reduction are making a difference.

The downward trends in SRP and FN loads of TP, in turn, suggest that curbing point-source dischargers has also played an important role. Municipal WWTPs discharge phosphorus primarily as SRP, and the greatest SRP rates of decrease were at the sites with the greatest WWTP influence (i.e., Spring and Osage), though not at OC112. Encouragingly, the water quality improvements that most likely tie back to point-source controls are seen as far downstream as the Oklahoma state line at IR59 and Watts. It is also encouraging that the FN concentration reductions in TP appear to be contributing to meaningful load reductions at the state line at Watts (but not at IR59).

Rates of TP concentration decrease were greater at Mud and OC112 compared to other sites, but the larger rate likely reflects that phosphorus levels were already least at these sites throughout the study. Even minor changes in smaller magnitude concentrations equates to a large rate of change when expressed as a percentage. The period of analysis for these two sites is also relatively short, and the rate might even out closer to other sites with continued monitoring.

Nitrogen

Nitrogen compounds were measured at relatively constant levels throughout the UIRW over the last 15 years. No changes were detected in FN concentrations, with the exception of likely decreasing NO_3-N at Savoy. However, a few potential changes in FN loads were detected. At Mud, FN loads of NO_3-N were very likely increasing by ~9% annually, while FN loads of NO_3-N were very likely decreasing by 2% annually at Savoy. The FN loads of both nitrogen compounds were likely (NO_3-N) to very likely (TN) decreasing by 2 – 3% annually at Baron.

The overall limited variability in nitrogen concentrations over the last 15 years suggests that nitrogen pollution is likely not worsening. But, the measures undertaken to address excess phosphorus, with apparent success, will not automatically bring about concurrent nitrogen reductions. Achieving substantial water quality improvements for nitrogen compounds will likely require strategies specifically tailored to address the sources, sinks, and biogeochemical cycling of nitrogen.

In particular, point-source management strategies showed no effect on nitrogen concentrations or loads. Municipal WWTPs face permitted limits on phosphorus in discharge, but are regulated only on the type of nitrogen that is released. While technological upgrades can and do result in more thorough nitrogen removal at WWTPs, water quality returns on any such investments in the UIRW are not showing at the AWRC's monitoring sites.

Implications for NPS management were more mixed. The potential decreases in nitrogen loads at Savoy and, especially, Baron suggest that watershed management strategies targeted to NPS pollution reduction have made a measurable difference on water quality in these mostly rural, pasture-influenced watersheds. However, the large potential increase in the NO_3-N load at Mud suggests that NPS strategies may not be as effective at mitigating the effects of urbanization in the UIRW.

Total Suspended Solids

Trend analysis results suggested that TSS has not changed throughout the UIRW over the last 15 years. Scott and Haggard, (2018) noted previously for these sites that total annual TSS loads from 2009 to 2018 were highly variable, the most variable of any of the analyzed constituents. Though the FN values estimated in this study smooth random interannual variability, the fact that TSS has inherently greater variability may mean that trends can only be detected at a high level of confidence with continued monitoring.

Stable TSS is in itself a positive result for watershed management efforts. The overall limited changes in TSS suggest that watershed-scale erosion is not worsening. It appears that NPS management strategies targeted to accelerated erosion risks in a rapidly urbanizing watershed have been successful. However, the investments by national, state, and local watershed management entities to reduce sediment export from existing pasture and urban lands in the UIRW are not yet showing returns as decreasing TSS concentrations and loads.

Anions

Like phosphorus, chloride has also widely been in transition in the UIRW over the last 15 years. But changes did not occur in a uniform direction. The FN concentrations of Cl have likely (Spring and Savoy ~1 - 2% annually) to very likely (Baron, ~2% annually) decreased at three sites, while increases were likely at two sites (Mud and OC112, ~3 – 4% annually). For all these sites, FN loads of Cl were also changing consistent with concentrations. Notably, no trends in Cl were detected for Osage, where time series comparisons between 2010, 2016, and 2022 suggested a possibility of increases in both FN concentrations and loads.

Sulfate, as FN concentration, was potentially increasing at four UIRW sites, with the increases considered likely at Mud (~1% annually) and very likely at OC112, Osage and Watts (~1 - 5% annually). As FN load, SO_4 was very likely increasing at both OC112 (~5% annually) and Osage (~2% annually). Increases in FN concentration of SO_4 at Mud did not result in a change to the load.

Increases in Cl and SO_4 are among the few potential signs of water quality degradation observed in this study. Chloride is a conservative tracer of human activity in a watershed. Decreases at Spring, Savoy, and Baron, therefore suggest better controls on constituent exports related to human activities, which could include both point-source discharges (Spring and Savoy) and NPS watershed management strategies (Baron). However, Cl increases at Mud and OC112 suggest that urbanization in the region measurably affects, and is potentially degrading, water quality.

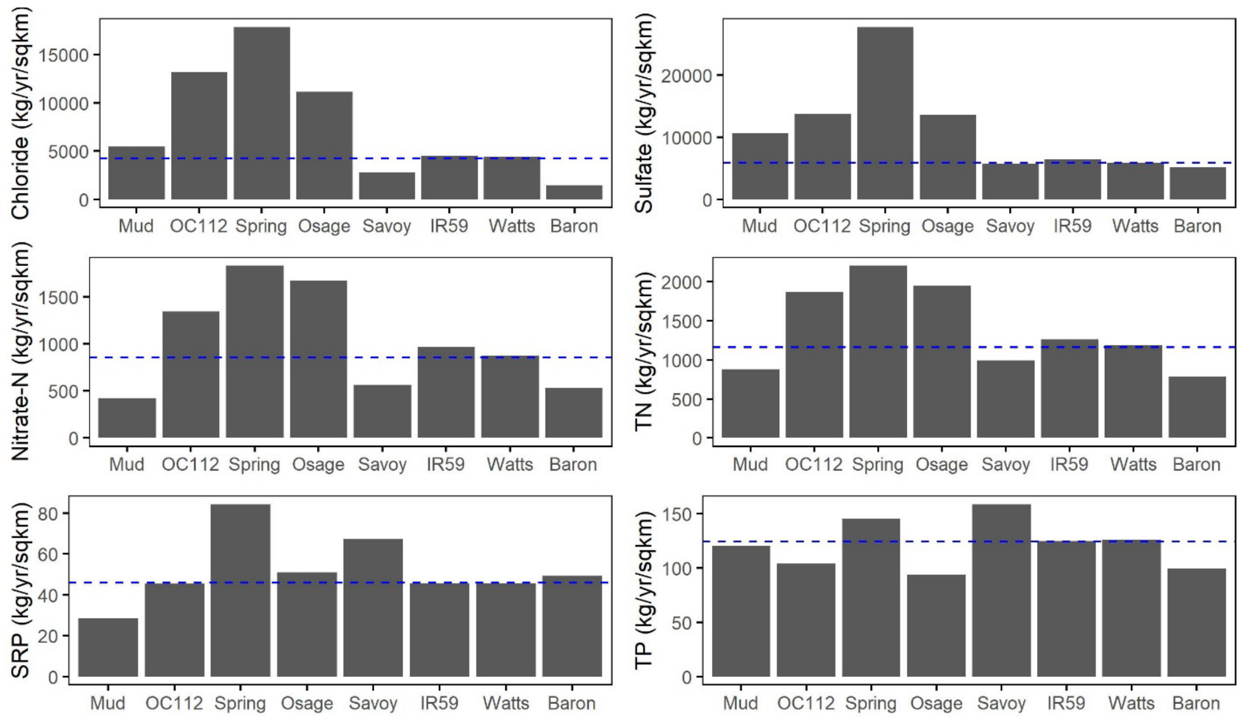
The most likely source of increasing FN concentrations and loads of SO_4 at OC112 and Osage is municipal WWTP use of aluminum sulfate, or alum, addition to remove excess phosphorus from wastewater. Chemical reactions between alum and wastewater result in aluminum binding SRP. But, these same reactions release free sulfate ions. Sulfate is also a common ingredient in detergents, which may also explain these increases, especially at Mud. The reason for the small increase at Watts is unclear. Overall, trend analysis results on SO_4 suggest that resolving both excess phosphorus and SO_4 impairments in the Illinois River will be a challenge.

Watershed perspectives on load and yield

In this section, we examine FN loads at the UIRW sites from a watershed perspective. The 2022 constituent loads were scaled to each site's watershed area and are shown as yields in Figure 1.4 to facilitate comparison between sites. As seen in Figure 1.3, loads are highly influenced by watershed area, but yields are normalized across watershed areas. Yields show the load produced for each unit of watershed area, here square km. Site-specific yields were indexed to the yield of the total gaged area, which is the combined watershed area of Watts and Baron. Constituent yields for the total gaged watershed area are shown as blue dashed lines in Figure 1.4 and represent an average condition.

If a site's yield is greater than the value of the blue dashed line, the site's watershed produces a greater FN load for its size relative to the total watershed area. Conversely, if a site's yield is less than the value of the blue dashed line, the site's watershed produces a smaller FN load for its size relative to the total watershed area. Otherwise stated, sites with yields above the blue line contribute more intensively to the total watershed load than sites with yields below the blue line. This information can be useful for understanding where to target NPS watershed management activities, or how well point-source controls are working. The yields at IR59 and Watts tend to be near the value of the dashed blue lines because their watershed area comprises >90% of the total.

Sites in the UIRW had very different watershed yields, both in magnitude and relative to the total watershed yield, depending on the constituent. These differences were largely in-line with the gradient of human influence on the watershed. The sites with more human influence, either as point-source dischargers, greater urban LULC, greater pasture LULC, or a combination of these characteristics, most often had yields that were greater than the blue dashed line. However, for some constituents, the human influences of urban LULC and municipal WWTPs left a more significant signature on yields compared to pasture LULC.



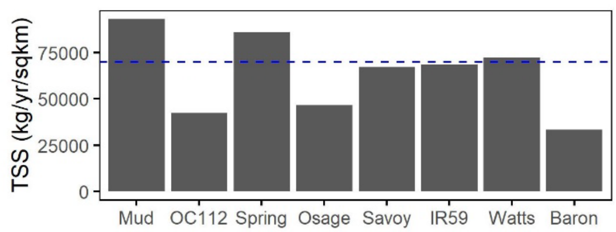


Figure 1.5. Constituent yields, or flow-adjusted loads per square km of watershed, at all UIRW sites in 2022. The dashed blue line shows the yield of the combined watersheds at Watts and Baron. Yield above the line means a relatively greater load for the watershed size, while yield below the line means relatively less.

The largest differences in yields between sites were in Cl and SO_4. Sites with the greatest urban LULC, point-source dischargers, or both (i.e. Mud, OC112, Spring, and Osage) all had Cl and SO_4 yields that were greater than the blue dashed line. Conversely, Baron and Savoy had Cl and SO_4 yields below the blue dashed line. These constituents are both anions, or negatively charged particles, that tend to be repelled by soils. They move easily and rapidly in the environment, particularly with runoff, which is often greater in urban environments due to impervious surfaces. Greater yields of both constituents at these sites is most likely tied to the overall intensity of human activities in the watershed. For SO_4, the use of alum by WWTPs to remove excess phosphorus from discharge likely plays a role, particularly at Osage. Sulfate is also a common ingredient in detergents.

Watershed patterns in NO_3-N and TN yields were similar to Cl and SO_4, but differences between sites were smaller in magnitude, especially for TN. Nitrate-N is an anion like Cl and SO_4, and streams and rivers in the UIRW are often super-saturated with NO_3-N. Nitrate-N tends to dominate TN, meaning that TN patterns will mirror NO_3-N. In contrast to Cl and SO_4, however, NO_3-N yield at Mud was less than the total watershed yield. This difference between sites suggests that the greater yields for nitrogen compounds at OC112, Spring, and Osage are more specifically related to municipal WWTP effects than for Cl and SO_4.

Yields of phosphorus compounds did not vary much between sites, especially compared to Cl, SO_4 and NO_3-N. Only Spring, Savoy, and Mud deviated from the total watershed yield of SRP, with Spring and Savoy having a greater yield and Mud less. For TP yields, OC112 and Baron also deviated from the total watershed yield. Both had a smaller yield than the total watershed area. Sites with yields above and below the blue dashed line did not group clearly by their gradient of human influence. The greatest SRP yield was at Spring, suggesting a likely connection to the Springdale, AR WWTP. But, Savoy had the second greatest SRP yield and the greatest TP yield. Savoy has an upstream municipal WWTP, but did not stand out as highly influenced by point-source discharge in any other results. Thus, greater yields of phosphorus compounds at Savoy may reflect pasture LULC, which is near 50%, as much as point-source discharge.

Differences in TSS yields relative to the total watershed area also did not divide clearly along the human influence gradient. Mud and Spring, two of the most urbanized watersheds, both had TSS yields above the blue dashed line, linking disproportional sediment export downstream to urban LULC. However, TSS yields for both OC112 and Osage were below the blue dashed line, despite having % urban LULC on par with Mud and Spring. Exploring differences between watershed management strategies in use in the Osage Creek watershed compared to Mud and Spring could provide useful information on which interventions are more effective for TSS. The TSS yield at Baron was the smallest overall in the UIWR, which fits with the narrative that urban LULC primarily increases the intensity of TSS export from the sub-watersheds of the UIRW.

Conclusions

A key water quality concern in the UIRW appears to have improved over the last 15 years, with trend analysis suggesting widespread decreases in FN concentrations and loads of phosphorus. These changes represent an important water quality gain for the UIRW and progress toward meeting Oklahoma's Scenic River criteria. However, the mean annual FN concentration of TP at IR59 in 2022 was still almost two times greater than the criteria.

The annual FN concentrations and FN loads of all the water quality constituents varied between sites and years in the UIRW. Other than for phosphorus and chloride, trend analysis showed that the majority of site-constituent combinations were likely not consistently changing over time. Notable exceptions included potential nitrogen decreases at Savoy and Baron, and potential SO_4 increases at OC112 and Osage. Trends in Cl were widespread in the UIRW, but not in a consistent direction. Overall, like trends, or absence of trends, were observed at sites with similar watershed characteristics that can help decipher potential causes and effects.

Watershed yields also varied throughout the UIRW, and spatial patterns in this variability have implications for watershed management. Trend analysis showed that point-source controls have likely contributed substantially to progress on phosphorus over the last 15 years. But, watersheds with municipal WWTPs still yield Cl, SO_4, nitrogen, and, in some cases, phosphorus, at disproportionately large rates. Therefore, significant potential for load reduction remains around better point-source controls, such as technology upgrades and stricter permits on discharge. Alternately, phosphorus yields at Savoy and TSS yields at Mud and Spring suggest that NPS strategies for mitigating effects of pasture and urban LULC can also make a difference in the overall watershed load. Siting projects within these watersheds, or other watershed areas with similar characteristics, has the greatest potential for water quality return on investment in the UIRW.

Chapter 2. Upper White River Basin

Introduction

The Upper White River Basin (UWRB) is located in Northwest Arkansas and is a priority watershed for the Arkansas Department of Agriculture – Natural Resources Division (ANRD) Nonpoint Source Pollution (NPS) Management Program. The biggest NPS challenges for the UWRB are excess nutrients and sediment (ANRD, 2018). Animal agriculture is the primary NPS for excess nutrients in the watershed, particularly phosphorus. Rapid urbanization and other land use changes have led to accelerated soil erosion and sediment export. Because phosphorus tends to associate strongly with soil particles, increased sediment transport in runoff is also a pathway for excess phosphorus to enter and build up in the waterbodies of the UWRB.

The White River's headwaters originate in rural areas of the Boston Mountains. Much of the UWRB remains in a mix of forest and pasture, but areas throughout the basin are also rapidly urbanizing. The White River and its major tributaries are impounded to form Beaver Lake, the drinking water source for approximately 1 in 6 Arkansans. The water quality in Beaver Lake is essential to the health and economic well-being of Arkansans. Maintaining water quality that is compatible with safe and affordable drinking water is a primary goal for watershed conservation in the UWRB as it undergoes rapid land use changes in the coming decades.

The State of Arkansas has taken steps to address excess phosphorus and mitigate land use changes in the UWRB in recent decades. The UWRB is designated as a Nutrient Surplus Area (Ark. Code Ann. § 15-20-1104), requiring controls on the application of phosphorus-rich poultry litter as fertilizer for pastures. The NPS Management Program, Beaver Water District, and local watershed groups, such as Beaver Watershed Alliance, have invested in education, best management practices (BMPs), and streambank restoration. The uppermost 16.5 miles of the West Fork of the White River were removed from the State of Arkansas' 2018 list of impaired waterbodies, a major success story for the NPS Management Program and its watershed management partners.

The Arkansas Water Resources Center (AWRC) has used consistent methodologies to monitor water quality in the UWRB since 2009 through contracts with the NPS Management Program. Robust data are necessary to establish baseline conditions and detect potential improvements resulting from the implementation of NPS projects, state regulations, and other watershed management activities. Long-term data are essential because the lag time between NPS project activities and the water quality response can be years to decades (Meals et al., 2010). These data are also needed to determine if, when, and where water quality is degrading when land use or other watershed changes have occurred.

The current study (NPS Management Program project #19-1100) objectives were to:

1. continue water sample collection throughout the UIRW for an additional three years,
2. estimate annual loads for the cumulative period of record (either 2009 – 2022, 2010 – 2014, or 2016 - 2022, depending on the site),
3. evaluate trends in water quality and loading to allow quantitative assessment of response to mitigation and management in the UWRB.

Methods

Site Information

The AWRC samples five locations in the UWRB under the current project (Figure 2.1), which are all located at U.S. Geological Survey (USGS) stream gaging stations (see Table 2.1). One site is located on the river mainstem (Wyman) and four are on tributaries (WFWR on the West Fork, TB on Town Branch, Richland on Richland Creek, and WEC on War Eagle Creek). An additional site (RC45) that was monitored on preceding projects was included in this analysis to provide a longer data record for Richland Creek. The USGS gage on Richland Creek was relocated in 2015, and RC45 was at the original gage location. The sites are positioned from upstream to downstream: WFWR, TB, Wyman, Richland, RC45, and WEC.

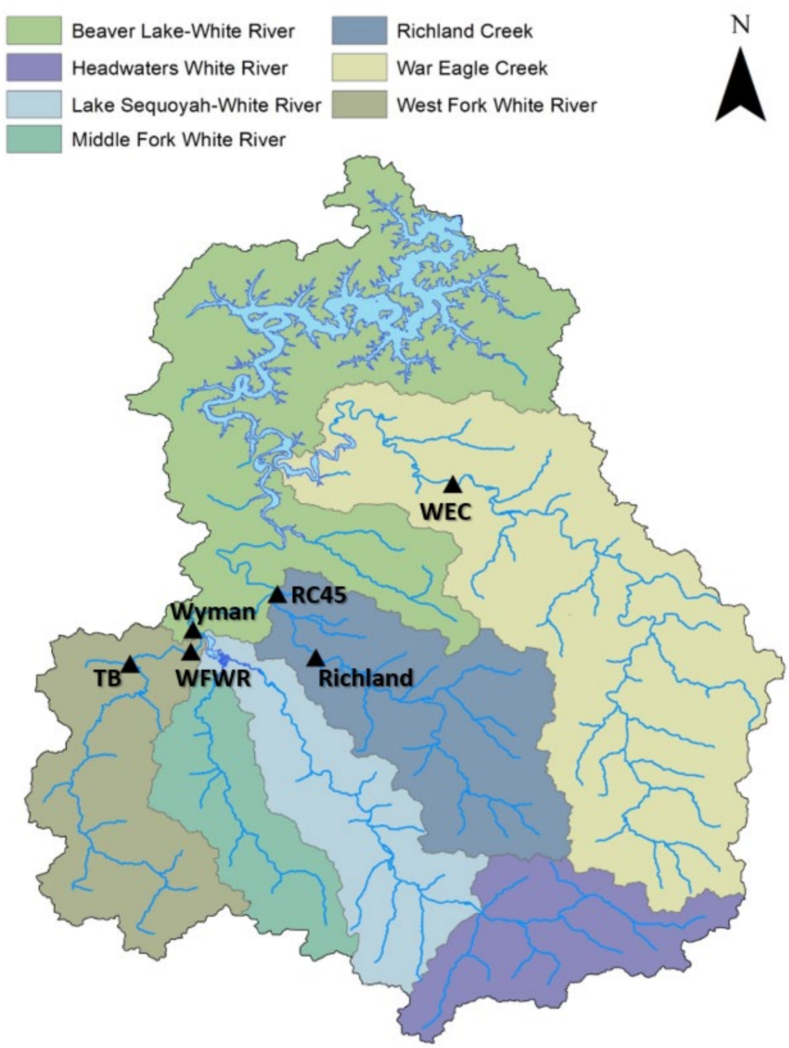


Figure 2.1. Monitoring locations in the Upper White River Basin.

Sites have a range of watershed land use-land cover (LULC) profiles, including gradients of forest-pasture mix to highly urbanized (Table 1.1). Wyman, WFWR, and WEC directly receive discharge from municipal WWTPs. The West Fork, AR WWTP discharges upstream of WFWR, the Fayetteville, AR Nolan WWTP discharges upstream of Wyman, and the Huntsville, AR WWTP discharges upstream of WEC. Segments of the White River were listed as impaired for critical season dissolved oxygen levels and turbidity in Arkansas's 2020 draft 303(d) list (ADEQ, 2020). Segments of War Eagle Creek were also cited for critical season dissolved oxygen. Segments of the West Fork remain on the 303(d) list for critical season dissolved oxygen, long-term continuous water temperature, turbidity, and sulfates. Town Branch was listed as impaired for turbidity and nitrate. Beaver Lake itself was included on the most recent 303(d) list for turbidity and E. coli.

Water Sample Collection

Water samples were collected manually from bridge access locations. Samples were collected using either an alpha-style horizontal sampler or a Kemmerer-type vertical sampler from a single representative point in the stream (i.e., near the vertical centroid of flow). The sampling approach was designed to capture both flow-driven and seasonal variation in constituent concentrations. On average, ~31 samples were collected per site each project year during the current (October 2019 – September 2022) project period. Base flow samples were collected at least once monthly. Whenever possible, stormflow was sampled at least monthly with the goal of capturing all the largest storm events each year. All samples were collected according to an approved quality assurance project plan (QAPP; QMP # 21-052). Sample collection intervals, methods, and design were consistent with preceding projects.

Table 2.1. Site information for the six AWRC monitoring locations in the Upper White River Basin. The period of analysis is based on water years (i.e. October 1 – September 30), where the water year is identified by the calendar year of the last nine months (January 1 – September 30) of the water year. The watershed land use/land cover information is adapted from the National Land Cover Database (NLCD), 2019 and was obtained using modelmywatershed.com.

Site	Latitude	Longitude	USGS Gage	Period of Analysis	Watershed Area (km^2)	% Urban[1]	% Forest[2]	% Pasture[3]
RC45	36.10417	-94.0075	07048800	2010 - 2014	357.5	5.11	63.48	31.28
Richland	36.04856	-93.9742	07048780	2016 - 2022	310.9	4.97	66.43	28.52
TB	36.04326	-94.136	07048495	-	30.6	52.05	33.44	14.11
WEC	36.04326	-94.136	07049000	2010 - 2022	681.3	5.49	59.01	35.29
WFWR	36.05389	-94.0831	07048550	2010 - 2022	318.7	14.53	64.09	20.65
Wyman	36.07306	-94.0811	07048600	2010 - 2022	1036.3	7.57	73.62	18.13

[1] % Urban is the sum of all developed land categories, as well as barren land
[2] % Forest is the sum of all forest categories, as well as shrub/scrub
[3] % Pasture is the sum of the pasture/hay and grassland/herbaceous categories

Sample Analysis

All water samples were stored on ice after collection and returned promptly to the Arkansas Water Resources Center Water Quality Lab (WQL). Samples were analyzed for concentrations (mg/L) of nitrate-nitrogen (NO_3-N), total nitrogen (TN), soluble reactive phosphorus (SRP), total phosphorus (TP), total suspended solids (TSS), chloride (Cl), and sulfate (SO_4) using standard analytical procedures for the analysis of water and wastewater and following the approved QAPP. The WQL is certified by the Arkansas Department of Energy and Environment - Environmental Quality Division (ADEQ) for the analysis of all the measured parameters in water. The WQL used standard quality assurance and quality control (QA/QC) practices, such as blanks, duplicates, and spikes.

Streamflow Record

The monitoring sites are located at active USGS stream gaging stations. A high-quality streamflow record is essential for load estimation. Adjusting constituent concentrations and loads for streamflow variability also enhances our understanding of how these values vary through time. Adjusting for flow variability prior to trend analysis makes change over time more readily detectable. Mean daily streamflow (cfs) and gaged watershed area (km^2) data were obtained through the USGS National Water Information Systems (NWIS; USGS, 2022) for all gages at the end of the project period.

Upon retrieving data from NWIS, we found that these streamflow is no longer estimated at the USGS gage at TB, and that the entire streamflow record is no longer available in the USGS historic database. Therefore, we were unable to carry out analysis of loads and trends for TB. We also found that Richland's daily streamflow record had several missing dates. We determined by comparing to other sites that base flow conditions applied on these dates. We made a best estimate of average daily streamflow to fill in all missing dates by averaging streamflow on the day preceding and following the missing date(s).

Weighted Regressions on Time, Discharge, and Season (WRTDS)

Constituent loads and trends were calculated using the Weighted Regressions on Time, Discharge and Season (WRTDS) statistical modeling algorithm developed by the USGS (Hirsch et al., 2010; Sprague et al., 2011). The method considers the influence of time, discharge, and season in estimating loads and detecting trends in water quality at a site. The method removes the influence of random variations in streamflow that make it difficult to discern patterns in constituent concentrations and loads. We carried out the WRTDS analysis using the statistical software R, version 4.1.3, (R Core Team, 2021) paired with the EGRET package (Hirsch and DeCicco, 2015).

The WRTDS algorithm uses paired water quality and streamflow data as a calibration dataset for describing the water quality-streamflow relationship through time. This relationship is described with the following equation, where c is concentration, q is streamflow, T is time, and ε and β values are the estimates of regression standard error and model coefficients:

$$ln(c) = \beta_0 + \beta_1 q + \beta_2 T + \beta_3 \sin(2\pi T) + \beta_4 \cos(2\pi T) + \varepsilon$$

This underlying equation is well-established for the estimation of loads (Helsel et al., 2020). But, WRTDS is unique from other common load estimation tools because the parameters of the relationship are dynamic through time, with unique estimates of the regression coefficients and standard error each day. The model parameters are not stored and are not useful for global estimation of concentrations or loads. The WRTDS algorithm is a smoothing procedure that should not be used to extrapolate outside the period of record of paired water quality and streamflow data.

Concentrations, streamflow, and loads are often not normally distributed, so WRTDS estimates the daily time series of concentrations and other outputs in log-space. The WRTDS algorithm uses a bias correction factor when transforming the log unit concentration estimates back to standard concentrations (i.e., mg/L).

From each unique daily model, the WRTDS algorithm provides a daily estimate of constituent concentrations (mg/L) for the entire streamflow record. These concentrations are then the basis for estimates of constituent loads (kg/d) after multiplying by mean daily streamflow. Flow-normalized (FN) concentrations and FN loads are also calculated by multiplying by the probability distribution function for streamflow. Standard concentrations and loads are the actual estimated value for a given day, while FN concentrations and FN loads are corrected for the influences of variations in water quality and loads arising from random day-to-day variations in streamflow

These daily time series can be used to determine monthly, annual, or longer time scale values, either by summing loads or averaging concentrations. We based annual values on water years, which run from October 1 – September 30. A water year is denoted by the calendar year of its last nine months (i.e. January 1 – September 30), but begins on October 1 of the preceding calendar year. For example, the 2010 water year began on October 1, 2009 and ended on September 30, 2010.

In order to compare the contribution of sites to watershed loading, we calculated constituent yields for each site by dividing the FN load by watershed area. Site-specific loads are not directly comparable because streams with larger watershed areas are expected to transport greater loads. Conversely, streams with smaller watershed areas carry smaller loads. Watershed yields can be compared between sites, however, and show which areas of the greater watershed contribute most to total constituent export to downstream waters.

We used the WRTDS Bootstrap Test in the EGRETci package (Hirsch et al., 2015) to determine the statistical significance of potential changes in FN concentrations and loads over time. The p-value of the WRTDS Bootstrap Test describes the probability that a pattern over time is random. We considered $p<0.05$ to suggest a highly likely trend (i.e. <5% probability of a random pattern, or a ≥95% probability of a real trend) and $p<0.10$ to suggest a likely trend (i.e. <10% chance of random pattern, or a ≥90% probability of a real trend). Sites and water quality constituents with $p \geq 0.10$ were considered likely not changing.

Results

Annual Concentrations and Loads

The annual time series of concentrations, loads, FN concentrations, and FN loads are available in the Appendix to this report for all analyzed UWRB monitoring locations. Annual values are provided for each full water year (i.e. October 1 – September 30) in each site's period of analysis. Within this report, we focus on results for FN concentrations and FN loads at select time points. The years 2010, 2016, and 2022 are presented, as the first, mid-point, and last water years in the analysis. Note that the results shown for Richland in 2010 are from RC45 and from Richland in 2016 and 2022. Any observed variability between years observed for Richland Creek may be due to the site relocation. Trends were analyzed separately for the two sites however.

Mean annual FN concentrations (Figure 1.2) and total annual FN loads (Figure 1.3) in the UWRB varied both through time and spatially within the watershed. Variability in FN concentrations was observed between the select time points of 2010, 2016, and 2022 and between sites. This variability was different between constituents and watershed locations. In most cases, variability between years followed the same pattern at WFWR and Wyman, and FN concentrations were within similar range, or less at Wyman. The magnitude and temporal patterns in FN concentrations for RC45/Richland and WEC, conversely, tended to be more similar to each other.

The dominant source of variability in FN loads was between sites and proportional to watershed area. In most cases, loading was greatest for Wyman and WEC, while RC45/Richland and WFWR were more similar. However, SO_4 at WFWR and TSS at Richland broke with this pattern. This watershed loading pattern is a function of increasing streamflow with watershed area. Streamflow, the dominant component of load, varies by orders of magnitude as watershed area increases, while concentrations tend to vary less, even in response to major differences in watershed characteristics. Nevertheless, site-specific interannual variability was also observed in FN loads and most often followed similar patterns to FN concentrations.

We observed three patterns in site-specific interannual variability in FN concentrations and loads. First, some site and constituent combinations moved toward smaller FN concentrations or loads across all the time points. The most notable example of this pattern was in both concentrations and loads of TP at all sites, except concentrations at Richland. When comparing 2010 and 2022, the concentration was up to 50% smaller in 2022 and ~20 – 30% less for loads. At WFWR and Wyman (but not RC45/Richland or WEC), concentrations and loads of NO_3-N in 2022 were 40 - 45% less when comparing 2022 to 2010. Chloride loads in 2022 were ~20 – 30% smaller compared to 2010.

Conversely, other site and constituent combinations moved toward larger FN concentrations or loads across all the time points. This pattern was observed rarely and the differences between timepoints were smaller in magnitude. The most notable example was the TSS load at WFWR, which was 25% greater in 2022 than in 2010.

Lastly, other site and constituent combinations showed no consistent trajectory in variability from year to year, or variability stayed within a more narrow range. The FN concentrations and loads of SO_4, NO_3-N, and SRP at Richland and WEC, as well as TSS concentrations and loads at all sites except WFWR varied minimally or without a consistent direction across timepoints.

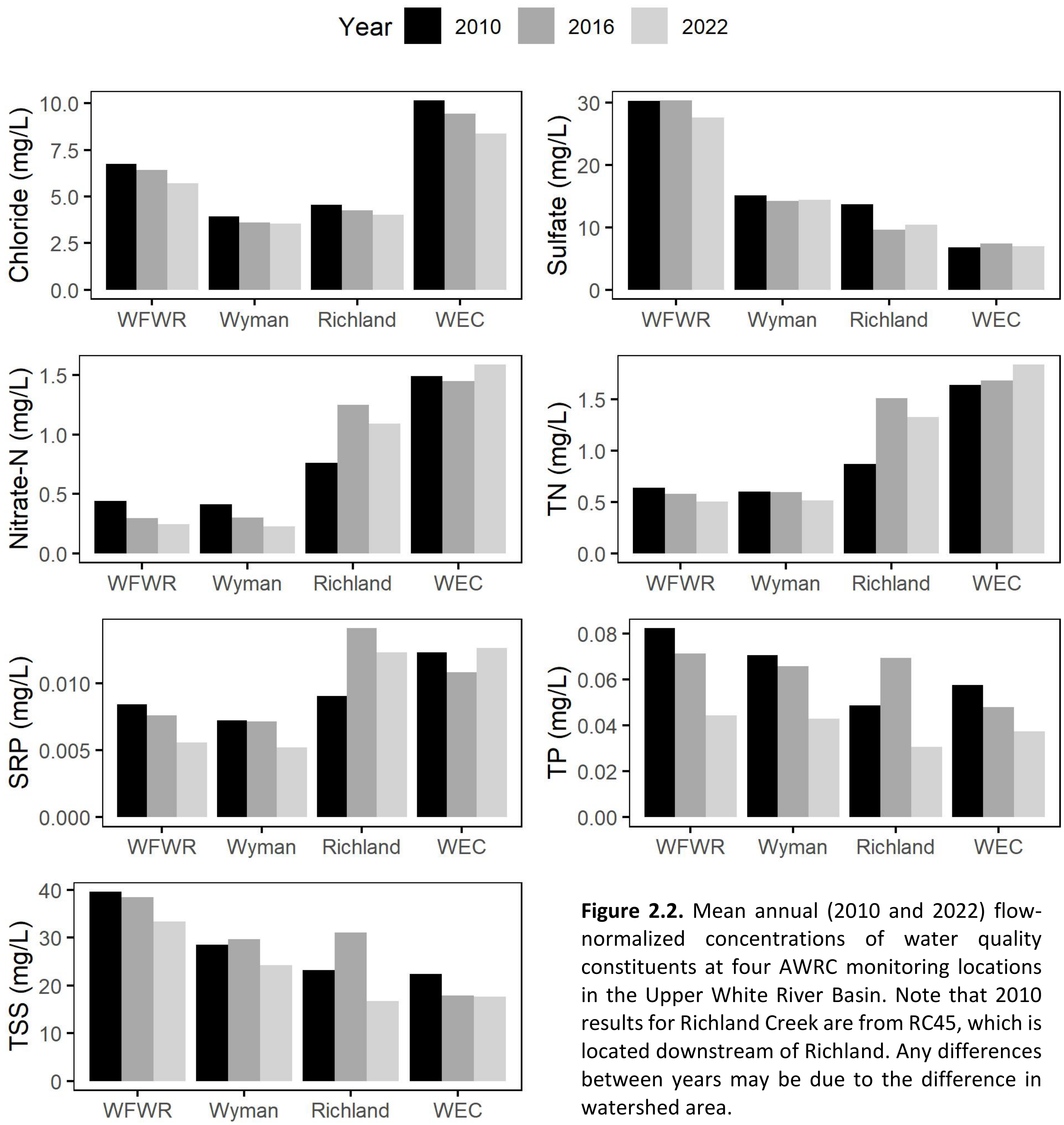


Figure 2.2. Mean annual (2010 and 2022) flow-normalized concentrations of water quality constituents at four AWRC monitoring locations in the Upper White River Basin. Note that 2010 results for Richland Creek are from RC45, which is located downstream of Richland. Any differences between years may be due to the difference in watershed area.

Water Quality Trends

Interannual variability is a normal characteristic of environmental datasets and was expected in the mean annual FN concentrations and loads. Considerable interannual variability is also consistent with estimates from preceding studies in the UWRB (Scott and Haggard, 2018). The results of trend analysis on FN concentrations (Table 2.2) and FN loads (Table 2.3) over time show whether the observed temporal variability is part of a consistent water quality trend over time, or simply due to random variability.

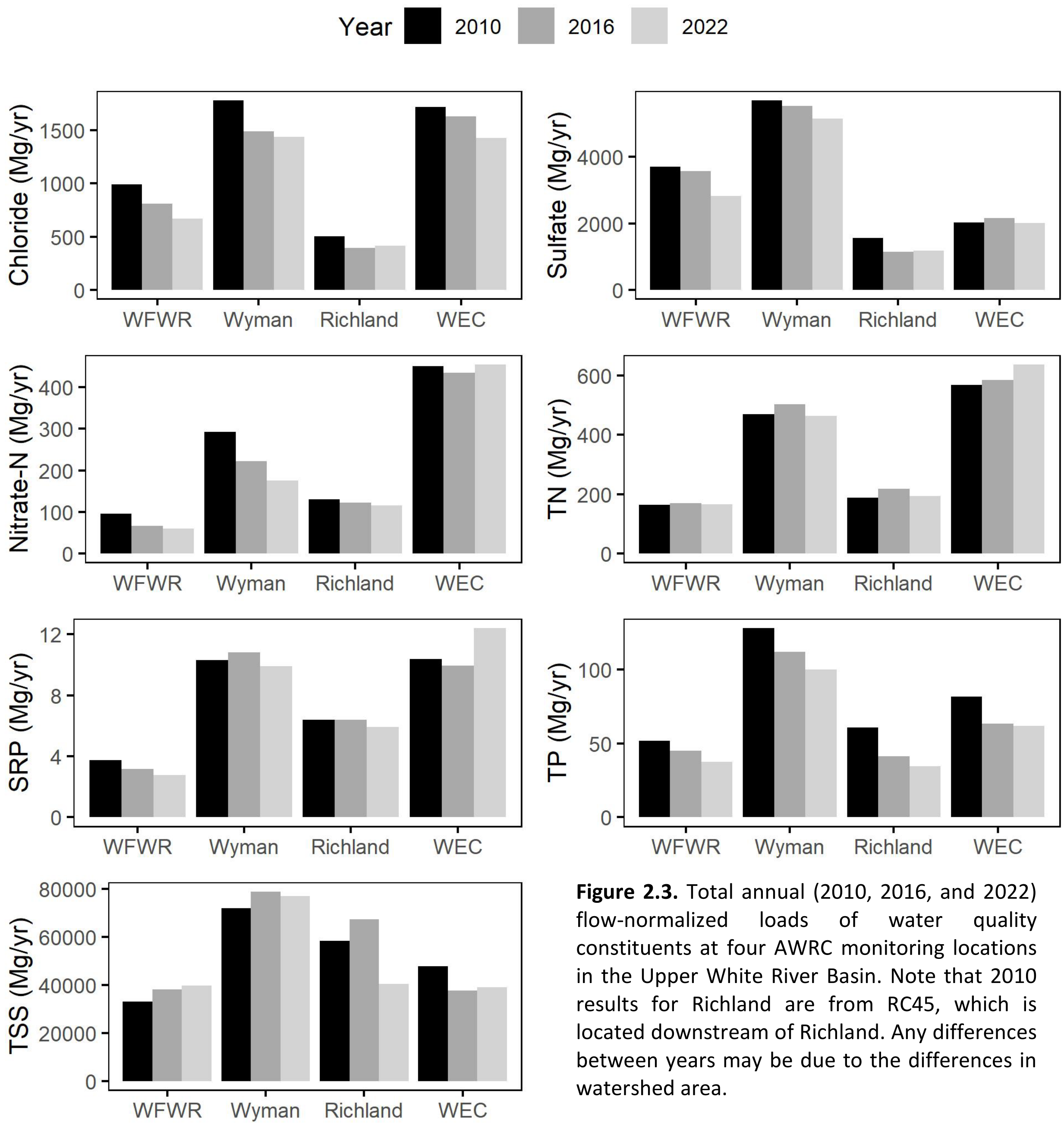


Figure 2.3. Total annual (2010, 2016, and 2022) flow-normalized loads of water quality constituents at four AWRC monitoring locations in the Upper White River Basin. Note that 2010 results for Richland are from RC45, which is located downstream of Richland. Any differences between years may be due to the differences in watershed area.

Phosphorus

Trend analysis results suggested watershed-wide decreases in FN concentrations of phosphorus in the UWRB over the last 15 years, but not in loads. Decreases in FN concentrations of TP ranged from ~3 – 9% annually and were considered very likely for all sites ($p<0.05$), except Richland, where the decrease was considered likely ($p<0.10$). Richland has a shorter period of analysis than the other sites, which introduces greater uncertainty in trend analysis. The FN concentrations of SRP were also likely decreasing at WFWR and Wyman.

Table 2.2. Trend analysis results on flow-normalized concentrations for the UWRB monitoring locations. The period of analysis differs between sites and is given below the site name. Note that the Richland Creek monitoring location moved from RC45 to Richland in 2015, so trends describe a shorter period of analysis for both RC45 and Richland. For all other sites, the period of analysis in 2010, with the first full water year (i.e., October 1 – September 30), and ends with water year 2022.

	WFWR (2010 – 2022)	Wyman (2010 – 2022)	RC45 (2010 – 2014)	Richland (2016 – 2022)	WEC (2010 – 2022)
Analyte	% change in flow-normalized concentrations				
Cl	No change	No change	No change	No change	-1.5*
SO_4	No change	No change	No change	No change	No change
NO_3-N	-3.7**	-3.7*	No change	No change	No change
TN	No change	No change	8.5*	No change	No change
SRP	-2.8*	-2.3**	No change	No change	No change
TP	-3.8**	-3.3**	No change	-9.3*	-2.9**
TSS	No change	No change	No change	No change	No change

* denotes trends that are “likely” (i.e. $p<0.10$)
** denotes trends that are “very likely” (i.e. $p<0.05$)

Table 2.3. Trend analysis results on flow-normalized loads for the UWRB monitoring locations. The period of analysis differs between sites and is given below the site name. Note that the Richland Creek monitoring location was moved from RC45 to Richland in 2015, so trends describe a shorter period of analysis for both RC45 and Richland. For all other sites, the period of analysis in 2010, with the first full water year (i.e., October 1 – September 30), and ends with water year 2022.

	WFWR (2010 – 2022)	Wyman (2010 – 2022)	RC45 (2010 – 2014)	Richland (2016- 2022)	WEC (2010 – 2022)
Analyte	% change in flow-normalized loads				
Cl	No change	No change	No change	No change	No change
SO_4	No change	No change	No change	No change	No change
NO_3-N	-3.1**	-3.3**	No change	No change	No change
TN	No change	No change	No change	No change	No change
SRP	No change	No change	No change	No change	No change
TP	No change	No change	No change	No change	No change
TSS	No change	No change	No change	No change	No change

* denotes trends that are “likely” (i.e. $p<0.10$)
** denotes trends that are “very likely” (i.e. $p<0.05$)

The watershed-wide downward trend in FN concentrations of TP is a water quality gain for the UWRB. Watershed patterns suggest that both NPS and point-source reductions have contributed to decreases. With the exception of Richland, the rate of decrease in FN concentrations of TP was similar between sites, including Wyman with its >70% forested watershed, WFWR with its greater (though still <15%) urban land extent, and WEC with its greater pasture influence. The UWRB has received considerable attention and funding from national, state, and local watershed management entities, and study results suggest that these NPS control efforts are making a difference.

The potential decreases in FN trends of SRP, in turn, suggest that curbing point-source dischargers has also played an important role in phosphorus declines. Municipal WWTPs discharge phosphorus primarily as SRP, and the two sites with a significant decrease in SRP concentrations both have a WWTP influence (i.e., WFWR and Wyman). However, no changes in SRP were detected at WEC, which also receives discharge from the municipal WWTP at Huntsville, AR.

Neither NPS nor point-source strategies appear to have led to phosphorus load reductions in the monitored areas of the UWRB over the last 15 years. But, timepoint comparisons of 2010, 2016, and 2022 showed steps downward in FN loads of TP at each timepoint for all sites. It is possible that decreases in TP concentrations are having an effect on loads that is still too small, or too variable, to detect with a high level of confidence in trend analysis. Additional years of monitoring are needed.

Nitrogen

Nitrogen compounds were measured at relatively constant levels throughout the UIRW over the last 15 years. Trends were partitioned between the upper (WFWR and Wyman) and lower (RC45/Richland) watershed. Most notably, both NO_3-N concentrations and loads were likely (Wyman, FN concentration) to very likely decreasing by ~3 – 4% annually. In contrast, no changes were detected in FN concentrations or loads of TN, with the exception of a likely increase in TN concentration at RC45 from 2010 - 2014. However, the period of analysis at this site ended after just five years, and the increase was not detected subsequently upstream at Richland from 2016 - 2022.

The limited variability in nitrogen relative to phosphorus over the last 15 years suggests that nitrogen pollution is likely not worsening, but that the measures that have been undertaken to address excess phosphorus will not automatically bring about concurrent nitrogen reductions. Signs of progress on nitrogen, as NO_3-N, were at the same sites (WFWR and Wyman) that had progress on SRP concentrations. Therefore, point-source management strategies may be effectively reducing nitrogen concentrations and loads in the UWRB. Municipal WWTPs in Arkansas do not have permitted limits on nitrogen in discharge, just on the nitrogen form, but upgrades at plants in the region in recent years have included better treatment for nitrogen.

Total Suspended Solids

Trend analysis results suggested that TSS has not changed throughout the UWRB over the last 15 years. Though time series comparisons suggested potential both for decreasing TSS concentrations and increasing TSS loads at WFWR, these patterns were not identified as a consistent trend over time. Scott and Haggard, (2018) noted that total annual TSS loads from 2009 to 2018 were highly variable, the most variable of any of the analyzed constituents. Though the FN values estimated in this study smooth random interannual variability, the fact that TSS has inherently greater variability may mean that trends additional monitoring will be required to detect trends at a high level of confidence.

Stable TSS is in itself a positive result for watershed management efforts. The overall limited changes in TSS suggest that watershed-scale erosion is not worsening. It appears that NPS management strategies targeted to accelerated erosion risks in a rapidly urbanizing watershed have been successful. However, the investments by national, state, and local watershed management entities to reduce sediment export from existing pasture and urban lands in the UIRW are not yet showing returns as decreasing TSS concentrations and loads. This finding for WFWR does not align with the recent water quality success story for the West Fork, but WFWR is also located downstream of the delisted stream reaches.

Anions

Chloride and SO_4 were also not changing throughout the watershed, except for Cl at WEC. Trend analysis suggested that Cl was very likely decreasing as both concentration and load by ~1.5% annually at WEC. Sulfate concentration and load was not changing at any site. Chloride is a conservative tracer of human activity in a watershed. Decreases at WEC therefore suggest better controls on constituent exports related to human activities, which could be related to either NPS or point-source activities.

Watershed perspectives on load and yield

In this section, we examine FN loads at the UWRB sites from a watershed perspective. The 2022 constituent loads were scaled to each site's watershed area and are shown as yields in Figure 2.4 to facilitate comparisons between sites. As seen in Figure 2.3, loads are highly influenced by watershed area, but yields are normalized across watershed areas. Yields show the load for each standardized unit of watershed area, here square km. Site-specific yields were indexed to the yield of the total gaged area, which is the combined watershed area of Wyman, Richland, and WEC. Constituent yields for the total gaged watershed are shown as blue dashed lines in Figure 2.4.

If a site's yield is greater than the value of the blue dashed line, the site's watershed produces a greater FN load for its size relative to the total watershed area. Conversely, if a site's yield is less than the value of the blue dashed line, the site's watershed produces a smaller FN load for its size relative to the total watershed area. Otherwise stated, sites with yields above the blue line contribute more intensively to the total watershed load than sites with yields below the blue line. This information can be useful for understanding where to target NPS watershed management activities, or how well point-source controls are working.

Sites in the UWRB had different watershed yields, both in magnitude and in relationship to the total watershed yield, depending on the constituent. None of the sites had consistently greater or smaller yields than the total watershed area across all water quality constituents. Whether sites had a disproportionally larger or smaller yield did not follow consistent patterns with the degree of human influence on the watershed. This is likely because the UWRB sites that we were able to include in analysis (i.e., not TB) have similar watershed characteristics. Any patterns in yields related to differences in watershed characteristics are thus subtle.

Both WEC and WFWR most often had yields over the blue dashed line (four of the seven constituents). Both had relatively greater yields of Cl, suggesting the greatest human footprint. Except for Cl, these four constituents were not the same, however. Yields at Richland were also greater than the total watershed yield for three analytes. Richland and WEC both had greater SRP yields than the total watershed. Both have ~30% pasture LULC, but other analysis suggested a strong point-source influence on SRP at WEC. Richland and WFWR both had greater TP and TSS yields, which suggests TSS and TP export are coupled in the UWRB, but, again, sources are unclear. Other analysis suggested urban LULC is a driver for TSS at WFWR, but Richland's dominant human influence is pasture.

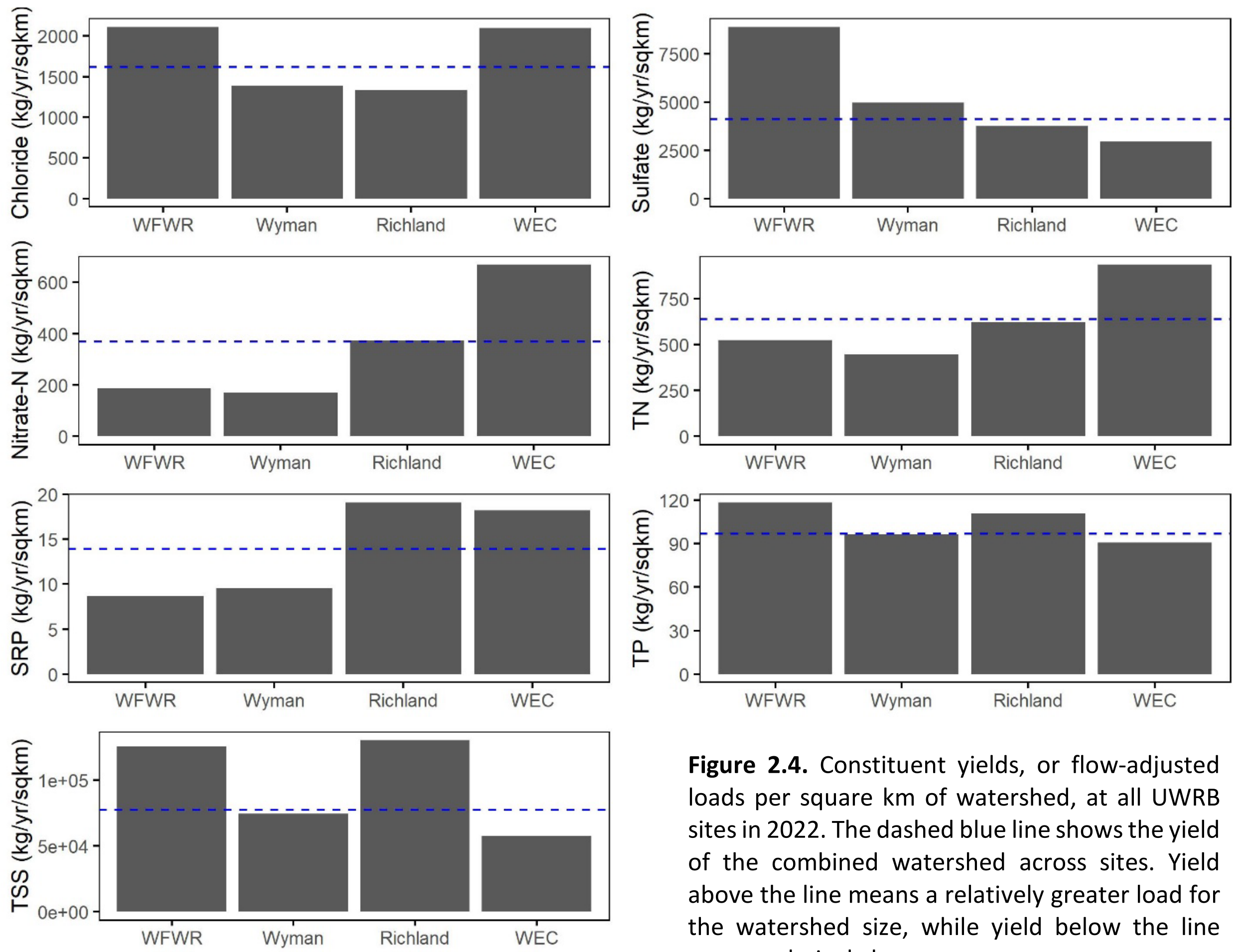


Figure 2.4. Constituent yields, or flow-adjusted loads per square km of watershed, at all UWRB sites in 2022. The dashed blue line shows the yield of the combined watershed across sites. Yield above the line means a relatively greater load for the watershed size, while yield below the line means relatively less.

For SO_4 and nitrogen compounds, yields were substantially greater than watershed average at only one site, WFWR and WEC, respectively. Though, SO_4 can be a signal of human influence, Scott and Haggard, (2021) showed in a previous study with sites throughout the West Fork watershed that SO_4 concentrations follow a natural gradient moving downstream that is likely related to underlying geology. For nitrogen compounds, WEC has both a point-source discharger and the greatest % pasture LULC in the watershed. This watershed profile makes it difficult to determine whether NPS or point-sources are the cause, and it may also be attributable to a combination of these factors.

Yields at Wyman were close in range or less than the dashed blue line, which may reflect that Wyman's watershed comprises the greatest portion of the total watershed area. For SO_4, yields at Wyman were slightly greater than the total watershed yield. This pattern likely reflects that the greater yields at WFWR are absorbed into the White River just upstream of Wyman.

Conclusions

A key water quality concern in the UWRB appears to have improved over the last 15 years, with trend analysis suggesting widespread decreases in FN concentrations of phosphorus. Beaver Lake has the only numeric criteria for phosphorus in the State of Arkansas. Study findings show that phosphorus concentrations are headed in the right direction to make meeting this criteria possible into the future. However, water quality gains on phosphorus concentrations did not extend to loads. Phosphorus loads have the potential to build up as sediment in Beaver Lake and to become an internal source as phosphorus leaches into the water column over time. Sediment loads were also not decreasing, and TP and sediment movement in the UWRB appeared tightly coupled.

The annual FN concentrations and FN loads of all the water quality constituents varied between sites and years in the UWRB. Other than for phosphorus concentrations, trend analysis showed that the majority of site-constituent combinations were likely not consistently changing over time. Notable exceptions included potential NO_3-N decreases in concentrations and loads at WFWR and Wyman and a potential Cl concentration decrease at WEC.

Watershed yields also varied throughout the UWRB, and spatial patterns in this variability have implications for watershed management. Similarities between sites make it challenging to differentiate NPS and point-source contributions. But, specific sub-watersheds clearly contribute more intensively to the total watershed load. Notably, results from WFWR suggest that the West Fork remains a hotspot for sediment export. This sub-watershed is therefore still a reasonable priority area for watershed management activities targeted to erosion control. Richland Creek was also a hotspot for TSS, despite the currently limited urban LULC. Links between TSS and TP suggest that successful interventions in sediment reduction could be necessary to bring about phosphorus load reductions. Finally, the War Eagle Creek watershed is a hotspot for nitrogen export. Strategies specifically targeted to nitrogen reduction would likely be necessary to accomplish reduced nitrogen concentrations and loads.

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Appendix

Savoy

Standard annual mean concentrations at Savoy, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	7.66	12.8	18.2	2.43	2.85	0.0671	0.1452	29.9
2011	7.46	17	20.6	2.74	3.05	0.0454	0.0997	19.4
2012	3.25	17.7	21.2	2.63	2.97	0.0411	0.0871	13.8
2013	3.22	17.2	21	2.63	2.97	0.0454	0.094	17.2
2014	2.5	16.3	20.9	2.51	2.84	0.039	0.0785	11.3
2015	7.4	12.9	19	2.19	2.69	0.068	0.1531	41.3
2016	6.82	14.9	20	2.31	2.72	0.0538	0.1144	26.1
2017	6.09	16.6	20.7	2.4	2.8	0.0465	0.098	24.8
2018	4.3	16.6	20.7	2.39	2.77	0.0419	0.0841	20.3
2019	7.46	11.5	18.1	2.05	2.58	0.0634	0.1278	33.5
2020	10.15	10.7	17.1	1.92	2.54	0.0805	0.1613	46.4
2021	5.83	12.5	18.7	2.16	2.61	0.05	0.0924	24.2
2022	7.48	12	18.4	2.13	2.6	0.0522	0.0945	26.9

Flow-normalized annual mean concentrations at Savoy, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	7.66	15.2	19.5	2.65	3.01	0.0565	0.1214	23.8
2011	7.46	15.2	19.6	2.57	2.95	0.0564	0.1219	24.9
2012	3.25	15.2	19.7	2.5	2.9	0.0564	0.1224	26
2013	3.22	15.1	19.8	2.44	2.85	0.0563	0.1223	27
2014	2.5	15	19.9	2.39	2.81	0.0562	0.1222	28.1
2015	7.4	14.9	20	2.32	2.76	0.0559	0.1219	29.1
2016	6.82	14.7	19.9	2.28	2.72	0.0553	0.1198	29.1
2017	6.09	14.4	19.7	2.25	2.7	0.0542	0.1144	27.8
2018	4.3	14.1	19.5	2.23	2.68	0.0532	0.1089	26.9
2019	7.46	13.8	19.4	2.21	2.66	0.0522	0.1033	26.1
2020	10.15	13.5	19.2	2.2	2.65	0.0513	0.0979	25.4
2021	5.83	13.2	19.1	2.2	2.64	0.0501	0.0925	24.6
2022	7.48	12.9	18.9	2.19	2.63	0.0491	0.0874	23.9

Standard total annual loads at Savoy, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	7.66	1.822	3.4	0.411	0.573	0.0338	0.0895	27.31
2011	7.46	1.232	2.39	0.288	0.467	0.04466	0.136	53.67
2012	3.25	1.015	1.78	0.208	0.271	0.01038	0.0278	9.23
2013	3.22	0.952	1.62	0.18	0.25	0.01002	0.028	9.96
2014	2.5	0.899	1.49	0.169	0.211	0.00576	0.0141	4.23
2015	7.4	1.623	3.12	0.32	0.534	0.03472	0.0961	38.53
2016	6.82	1.309	2.68	0.289	0.495	0.04981	0.13	51.13
2017	6.09	1.07	2.12	0.214	0.395	0.0325	0.0875	39.52
2018	4.3	0.962	1.89	0.18	0.313	0.02076	0.0548	25.22
2019	7.46	1.579	3.29	0.325	0.541	0.02997	0.0742	28.51
2020	10.15	1.948	4.23	0.414	0.749	0.0578	0.1328	51.7
2021	5.83	1.259	2.57	0.258	0.419	0.02098	0.0503	20.39
2022	7.48	1.387	2.99	0.282	0.514	0.03281	0.0791	34.19

Flow-normalized total annual loads at Savoy, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	7.66	1.44	2.65	0.321	0.455	0.0274	0.0794	27
2011	7.46	1.42	2.65	0.311	0.453	0.0282	0.0805	28.3
2012	3.25	1.4	2.65	0.302	0.451	0.029	0.0814	29.7
2013	3.22	1.38	2.65	0.293	0.449	0.0298	0.0821	31.1
2014	2.5	1.36	2.64	0.285	0.448	0.0306	0.0827	32.4
2015	7.4	1.34	2.64	0.277	0.447	0.0312	0.0831	33.5
2016	6.82	1.33	2.63	0.27	0.446	0.0313	0.0821	33.5
2017	6.09	1.3	2.61	0.265	0.442	0.0308	0.0795	32.4
2018	4.3	1.28	2.58	0.259	0.439	0.0306	0.0773	31.7
2019	7.46	1.26	2.56	0.255	0.436	0.0303	0.075	31
2020	10.15	1.24	2.53	0.251	0.434	0.0299	0.0728	30.4
2021	5.83	1.22	2.5	0.248	0.432	0.0295	0.0707	29.7
2022	7.48	1.2	2.48	0.244	0.43	0.0291	0.0687	29.1

Mud

Standard annual mean concentrations at Mud, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2016	0.865	11.3	27.3	0.452	0.744	0.0154	0.0639	16.3
2017	1.037	11.7	27.3	0.432	0.725	0.0141	0.0557	21.2
2018	0.612	12.7	29.9	0.474	0.737	0.0115	0.0349	12.1
2019	0.917	11.4	25.8	0.545	0.852	0.0138	0.044	16.5
2020	1.246	11.3	25.2	0.575	0.9	0.0152	0.0458	18
2021	0.735	12.7	27.1	0.612	0.884	0.0111	0.0273	10.3
2022	0.968	13.3	28	0.633	0.89	0.0107	0.0253	12.2

Flow-normalized annual mean concentrations at Mud, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2016	0.865	10.9	26.2	0.454	0.767	0.0162	0.0701	19.7
2017	1.037	11.2	26.6	0.477	0.785	0.015	0.0575	17.9
2018	0.612	11.5	27	0.501	0.802	0.014	0.0477	16.4
2019	0.917	11.9	27.3	0.526	0.819	0.0131	0.0401	15.1
2020	1.246	12.3	27.7	0.552	0.835	0.0123	0.0342	13.9
2021	0.735	12.8	28	0.58	0.851	0.0115	0.0294	12.8
2022	0.968	13.2	28.4	0.609	0.868	0.0108	0.0257	11.9

Standard total annual loads at Mud, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2016	0.865	0.155	0.404	0.01201	0.0291	0.001613	0.00707	4.71
2017	1.037	0.167	0.412	0.01353	0.0427	0.002066	0.00996	12.92
2018	0.612	0.136	0.308	0.00916	0.0212	7.38E-04	0.00332	2.87
2019	0.917	0.22	0.484	0.0149	0.0331	0.001011	0.00459	3.27
2020	1.246	0.283	0.614	0.02215	0.0463	0.001703	0.00644	4.01
2021	0.735	0.216	0.437	0.01385	0.0263	6.50E-04	0.00261	1.55
2022	0.968	0.254	0.481	0.01892	0.041	0.00131	0.00605	5.21

Flow-normalized total annual loads at Mud, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2016	0.865	0.173	0.438	0.012	0.0313	0.00141	0.00656	6
2017	1.037	0.181	0.442	0.0129	0.0323	0.00137	0.00623	5.57
2018	0.612	0.19	0.445	0.0138	0.0333	0.00133	0.00594	5.19
2019	0.917	0.2	0.448	0.0148	0.0343	0.0013	0.00571	4.85
2020	1.246	0.211	0.451	0.0158	0.0354	0.00127	0.0055	4.54
2021	0.735	0.223	0.454	0.0169	0.0365	0.00124	0.00533	4.26
2022	0.968	0.235	0.457	0.0181	0.0377	0.00122	0.00517	4

OC112

Standard annual mean concentrations at OC112, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2016	2.12	24.9	24.2	2.9	3.25	0.0583	0.1004	10.45
2017	2.27	26.3	26.5	2.85	3.25	0.0552	0.0946	12.53
2018	1.51	30	30.1	2.99	3.42	0.0516	0.0803	8
2019	2.16	23.6	24.4	2.59	3.1	0.0488	0.0808	10.37
2020	3.08	20.5	21.2	2.29	2.86	0.0486	0.0833	14.65
2021	1.82	27.8	29.3	2.75	3.33	0.0431	0.0632	7.56
2022	1.8	29.8	31.6	2.79	3.41	0.0413	0.0583	7.57

Flow-normalized annual mean concentrations at OC112, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2016	2.12	23.9	23.5	2.79	3.16	0.058	0.1022	11.49
2017	2.27	24.8	24.7	2.78	3.19	0.0551	0.0938	10.85
2018	1.51	25.4	25.8	2.75	3.21	0.052	0.0858	10.37
2019	2.16	26.1	26.8	2.73	3.23	0.0492	0.0788	9.96
2020	3.08	26.8	27.9	2.71	3.26	0.0466	0.0725	9.6
2021	1.82	27.5	29	2.69	3.28	0.0442	0.0667	9.32
2022	1.8	28.3	30.1	2.67	3.3	0.0418	0.0614	9.07

Standard total annual loads at OC112, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2016	2.12	0.941	0.978	0.121	0.157	0.00681	0.01763	10.18
2017	2.27	0.98	1.027	0.118	0.161	0.00595	0.01469	10.26
2018	1.51	0.932	0.938	0.102	0.128	0.00299	0.00663	3.17
2019	2.16	1.106	1.148	0.13	0.171	0.00396	0.00827	2.45
2020	3.08	1.302	1.376	0.155	0.218	0.00665	0.01438	5.06
2021	1.82	1.101	1.135	0.116	0.155	0.00325	0.00745	3.17
2022	1.8	1.12	1.174	0.111	0.151	0.0031	0.0068	2.32

Flow-normalized total annual loads at OC112, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2016	2.12	0.959	0.983	0.124	0.159	0.00518	0.01218	6.47
2017	2.27	0.989	1.025	0.123	0.16	0.00508	0.01188	5.93
2018	1.51	1.024	1.066	0.122	0.162	0.00485	0.01129	5.4
2019	2.16	1.06	1.108	0.122	0.163	0.00464	0.01075	4.94
2020	3.08	1.1	1.15	0.121	0.165	0.00444	0.01023	4.5
2021	1.82	1.142	1.194	0.121	0.167	0.00426	0.00979	4.14
2022	1.8	1.187	1.239	0.121	0.168	0.00408	0.00935	3.81

Spring

Standard annual mean concentrations at Spring, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2013	1.58	52.3	71.3	3.37	3.74	0.167	0.209	18.62
2014	1.46	45.3	62.2	3.23	3.64	0.145	0.181	9.63
2015	1.79	41.1	57.3	3.11	3.52	0.133	0.174	16.04
2016	2.37	40.4	57	3.06	3.49	0.127	0.165	18.61
2017	2.35	42.9	61.6	3.15	3.53	0.123	0.158	30.02
2018	1.84	41.6	60.4	3.16	3.54	0.117	0.143	16.11
2019	2.26	34.8	51.3	2.94	3.36	0.113	0.145	18.89
2020	3.56	29.4	44.1	2.67	3.16	0.112	0.156	32.16
2021	2.02	36.7	55.7	3.1	3.47	0.106	0.125	13.94
2022	1.84	36.5	56	3.16	3.5	0.102	0.115	10.4

Flow-normalized annual mean concentrations at Spring, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2013	1.58	44.1	59.9	3.12	3.57	0.157	0.209	25
2014	1.46	43.5	60	3.11	3.55	0.147	0.195	23.6
2015	1.79	42.8	59.9	3.09	3.53	0.138	0.181	22.2
2016	2.37	42.2	59.8	3.09	3.51	0.13	0.168	20.7
2017	2.35	41.4	59.4	3.08	3.5	0.123	0.156	19.2
2018	1.84	40.5	58.8	3.09	3.49	0.118	0.147	17.6
2019	2.26	39.5	58.3	3.1	3.48	0.114	0.139	16.2
2020	3.56	38.6	57.7	3.11	3.48	0.11	0.132	15
2021	2.02	37.7	57.1	3.12	3.48	0.107	0.125	13.9
2022	1.84	36.7	56.4	3.13	3.48	0.103	0.118	12.8

Standard total annual loads at Spring, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2013	1.58	1.74	2.35	0.131	0.164	0.00813	0.01519	7.56
2014	1.46	1.82	2.48	0.138	0.16	0.00649	0.00916	1.65
2015	1.79	1.87	2.61	0.152	0.183	0.00767	0.01214	3.67
2016	2.37	1.91	2.69	0.17	0.224	0.01234	0.03018	22.12
2017	2.35	1.79	2.6	0.157	0.221	0.01211	0.03366	58.74
2018	1.84	1.74	2.53	0.151	0.185	0.00726	0.01284	7.88
2019	2.26	1.84	2.71	0.176	0.216	0.00831	0.01325	4.78
2020	3.56	2.03	3.09	0.238	0.314	0.01544	0.02964	14.5
2021	2.02	1.67	2.53	0.164	0.199	0.00728	0.01226	7.46
2022	1.84	1.61	2.46	0.16	0.186	0.00624	0.00908	3.14

Flow-normalized total annual loads at Spring, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2013	1.58	1.93	2.61	0.157	0.204	0.01075	0.0228	18.88
2014	1.46	1.9	2.61	0.157	0.202	0.01024	0.0214	17.34
2015	1.79	1.87	2.61	0.157	0.201	0.00975	0.0201	15.9
2016	2.37	1.84	2.61	0.157	0.2	0.00929	0.0188	14.48
2017	2.35	1.81	2.6	0.158	0.199	0.00888	0.0176	13.11
2018	1.84	1.77	2.58	0.159	0.199	0.0086	0.0165	11.8
2019	2.26	1.73	2.56	0.16	0.199	0.00834	0.0156	10.64
2020	3.56	1.7	2.54	0.162	0.199	0.00809	0.0147	9.57
2021	2.02	1.66	2.53	0.164	0.199	0.00786	0.0139	8.66
2022	1.84	1.62	2.51	0.166	0.2	0.00764	0.0132	7.8

Osage

Standard annual mean concentrations at Osage, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	5.3	26.2	28.1	3.56	3.88	0.0837	0.1371	28.5
2011	7.21	29.3	32.1	3.7	4	0.0817	0.1318	23.25
2012	3.49	30.6	34.1	3.69	3.97	0.0764	0.1099	10.6
2013	4.72	30.7	34.6	3.66	3.99	0.0751	0.1169	18.04
2014	3.76	28.9	33.3	3.6	3.9	0.0711	0.1037	8.32
2015	5.75	25.1	29.5	3.34	3.74	0.073	0.1217	26.24
2016	6.19	26.6	31.5	3.4	3.76	0.0708	0.1126	22.11
2017	6.42	29.8	35.2	3.53	3.93	0.0674	0.1046	18.01
2018	5.38	30	35.4	3.54	3.91	0.0646	0.0971	16.23
2019	7.29	23.5	28.2	3.2	3.64	0.0671	0.1087	23.19
2020	11.64	19.1	23.2	2.88	3.36	0.0719	0.1263	36.14
2021	7.18	24.9	30	3.41	3.77	0.0606	0.0876	15.61
2022	8.14	23.3	28.3	3.37	3.73	0.0598	0.0856	16.17

Flow-normalized annual mean concentrations at Osage, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	5.3	25.2	27.1	3.53	3.85	0.0835	0.1416	30.6
2011	7.21	25.6	28.1	3.48	3.82	0.0816	0.1383	29.9
2012	3.49	25.9	28.9	3.44	3.8	0.0798	0.1347	29.2
2013	4.72	26.3	29.8	3.4	3.78	0.078	0.1312	28.4
2014	3.76	26.6	30.7	3.37	3.76	0.0763	0.1276	27.8
2015	5.75	27	31.6	3.36	3.76	0.0745	0.124	27.4
2016	6.19	27.2	32.2	3.37	3.78	0.0719	0.1183	25.1
2017	6.42	27.4	32.4	3.4	3.79	0.0689	0.1103	21.5
2018	5.38	27.5	32.7	3.42	3.81	0.0662	0.1027	18.8
2019	7.29	27.7	33	3.46	3.84	0.0636	0.0954	16.6
2020	11.64	27.9	33.3	3.49	3.87	0.0612	0.0885	14.7
2021	7.18	28.1	33.6	3.54	3.9	0.0588	0.0819	13
2022	8.14	28.3	33.9	3.59	3.94	0.0566	0.0756	11.5

Standard total annual loads at Osage, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	5.3	3.22	3.5	0.499	0.584	0.01747	0.0419	29.75
2011	7.21	3.28	3.78	0.535	0.687	0.04339	0.1087	58.7
2012	3.49	2.88	3.19	0.378	0.419	0.00896	0.0168	6.7
2013	4.72	3.13	3.6	0.425	0.506	0.01662	0.037	18.78
2014	3.76	3.14	3.61	0.41	0.452	0.00859	0.0137	1.95
2015	5.75	3.58	4.27	0.51	0.61	0.01635	0.034	17.19
2016	6.19	3.49	4.24	0.524	0.636	0.02805	0.0632	49.04
2017	6.42	3.49	4.26	0.494	0.62	0.02514	0.0539	32.05
2018	5.38	3.44	4.08	0.472	0.572	0.0157	0.0342	18.26
2019	7.29	3.96	4.83	0.61	0.738	0.01888	0.0367	15.32
2020	11.64	4.66	5.94	0.843	1.07	0.04058	0.0847	45.39
2021	7.18	3.95	4.82	0.617	0.731	0.01968	0.0376	19.23
2022	8.14	4.18	5.15	0.68	0.809	0.02271	0.0432	22.35

Flow-normalized total annual loads at Osage, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	5.3	3.4	3.72	0.55	0.665	0.0261	0.0653	40.5
2011	7.21	3.43	3.83	0.541	0.658	0.0256	0.0627	38.7
2012	3.49	3.46	3.93	0.533	0.652	0.025	0.0601	36.9
2013	4.72	3.48	4.03	0.526	0.647	0.0244	0.0576	35.2
2014	3.76	3.5	4.13	0.52	0.642	0.0239	0.0551	33.7
2015	5.75	3.53	4.24	0.515	0.639	0.0234	0.0528	32.3
2016	6.19	3.55	4.31	0.515	0.638	0.0226	0.05	30
2017	6.42	3.58	4.35	0.521	0.64	0.0215	0.0465	26.7
2018	5.38	3.61	4.39	0.527	0.642	0.0206	0.0431	23.9
2019	7.29	3.64	4.44	0.534	0.644	0.0197	0.04	21.5
2020	11.64	3.67	4.48	0.543	0.648	0.0188	0.0369	19.3
2021	7.18	3.71	4.54	0.553	0.653	0.018	0.0342	17.5
2022	8.14	3.75	4.59	0.563	0.658	0.0172	0.0316	15.7

IR59

Standard annual mean concentrations at IR59, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	25.6	14.5	15.9	2.51	2.76	0.0615	0.1218	28.9
2011	27.8	17.5	18.4	2.41	2.64	0.053	0.1036	24.3
2012	13.3	18.4	19.8	2.3	2.53	0.0503	0.0833	13
2013	16.5	18	19.7	2.32	2.59	0.051	0.0935	20.8
2014	12.1	17.3	19.5	2.36	2.61	0.0471	0.0768	11.4
2015	26.6	14.7	17.8	2.31	2.66	0.06	0.1277	40
2016	26.2	15.4	18.4	2.3	2.62	0.0543	0.1021	23.5
2017	25.9	16.4	19.7	2.27	2.59	0.0484	0.0927	27.8
2018	17.6	16.4	19.7	2.31	2.59	0.0439	0.0755	20.8
2019	26.9	13.1	16.7	2.3	2.65	0.0548	0.1035	31.3
2020	41.4	11.4	15.2	2.19	2.62	0.0663	0.1371	51.6
2021	24.6	13.3	16.9	2.36	2.66	0.0472	0.078	23.4
2022	28.5	13.1	16.8	2.35	2.65	0.0464	0.0774	26.6

Flow-normalized annual mean concentrations at IR59, as estimated by WRTDS

Year	Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	25.6	15.8	16.9	2.46	2.7	0.0588	0.1138	26.5
2011	27.8	15.9	17.3	2.42	2.68	0.0582	0.1136	27.1
2012	13.3	15.9	17.7	2.38	2.66	0.0576	0.1131	27.6
2013	16.5	15.9	18	2.35	2.64	0.057	0.112	28
2014	12.1	15.8	18.3	2.32	2.63	0.0564	0.1108	28.4
2015	26.6	15.8	18.6	2.29	2.62	0.0556	0.1092	28.9
2016	26.2	15.5	18.7	2.29	2.61	0.0541	0.1052	28.4
2017	25.9	15.3	18.6	2.29	2.61	0.052	0.0992	27.4
2018	17.6	15.1	18.5	2.29	2.61	0.0502	0.0931	26.6
2019	26.9	14.9	18.4	2.3	2.61	0.0485	0.087	25.7
2020	41.4	14.6	18.2	2.31	2.61	0.0469	0.0812	24.9
2021	24.6	14.4	18.1	2.33	2.62	0.0452	0.0756	24.1
2022	28.5	14.2	18	2.34	2.63	0.0437	0.0704	23.4

Standard total annual loads at IR59, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	25.6	8.77	10.61	1.773	2.15	0.0807	0.2396	88.6
2011	27.8	6.82	8.73	1.409	2.02	0.1193	0.512	217.7
2012	13.3	5.76	6.84	0.997	1.16	0.0307	0.0793	30.4
2013	16.5	6.2	7.6	1.076	1.34	0.0439	0.126	52.3
2014	12.1	5.55	6.64	0.905	1.04	0.0221	0.0486	15.3
2015	26.6	8.45	11.46	1.611	2.11	0.0824	0.2366	104.2
2016	26.2	7.44	10.49	1.557	2.1	0.1091	0.3476	153.9
2017	25.9	6.66	9.32	1.305	1.84	0.0931	0.3375	178.5
2018	17.6	6.05	8.08	1.106	1.42	0.0469	0.1331	71.3
2019	26.9	8.39	11.72	1.734	2.21	0.0678	0.1689	72.2
2020	41.4	10.63	16.16	2.477	3.34	0.1405	0.3589	174.9
2021	24.6	7.41	10.32	1.573	1.98	0.0582	0.1508	76.5
2022	28.5	7.53	10.78	1.654	2.19	0.08	0.2404	140.1

Flow-normalized total annual loads at IR59, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	25.6	7.96	9.64	1.6	2	0.0781	0.263	104
2011	27.8	7.9	9.78	1.57	1.98	0.0783	0.261	106
2012	13.3	7.84	9.91	1.54	1.96	0.0784	0.258	108
2013	16.5	7.77	10.03	1.51	1.94	0.0786	0.254	109
2014	12.1	7.69	10.15	1.49	1.93	0.0787	0.25	110
2015	26.6	7.61	10.27	1.47	1.92	0.0788	0.246	110
2016	26.2	7.48	10.27	1.46	1.91	0.0775	0.237	108
2017	25.9	7.34	10.16	1.45	1.9	0.0756	0.227	107
2018	17.6	7.21	10.04	1.44	1.89	0.074	0.218	106
2019	26.9	7.09	9.93	1.44	1.88	0.0725	0.209	105
2020	41.4	6.97	9.82	1.43	1.88	0.0711	0.201	103
2021	24.6	6.84	9.71	1.43	1.88	0.0696	0.193	103
2022	28.5	6.73	9.6	1.44	1.88	0.0681	0.186	102

Watts

Standard annual mean concentrations at Watts, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	23.1	15.1	15.7	2.37	2.64	0.0617	0.1288	26.6
2011	28.1	17.6	17.7	2.26	2.49	0.0523	0.1118	26
2012	12.9	19.1	19.6	2.15	2.39	0.0472	0.0854	14
2013	15.7	18.9	19.3	2.18	2.44	0.0494	0.0997	22.1
2014	13	17.4	18.6	2.26	2.51	0.0466	0.0827	14.1
2015	25.4	15.6	17.5	2.25	2.57	0.0575	0.1302	36.9
2016	27.6	15.9	17.9	2.23	2.54	0.0526	0.1044	25.5
2017	25.6	18.2	19.8	2.14	2.44	0.0463	0.0959	28.7
2018	19.1	17.4	19.3	2.16	2.46	0.0433	0.0813	23.5
2019	29.3	13.6	16.5	2.2	2.6	0.0559	0.1135	37.8
2020	43.8	11.3	14.7	2.15	2.63	0.0678	0.1441	57.1
2021	26.1	13.5	16.4	2.25	2.62	0.0481	0.0843	27.9
2022	30.7	13.3	16.3	2.23	2.61	0.0479	0.0846	31.4

Flow-normalized annual mean concentrations at Watts, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	23.1	16.1	16.4	2.32	2.57	0.0598	0.1256	27.2
2011	28.1	16.2	16.8	2.29	2.56	0.0588	0.1242	28
2012	12.9	16.2	17.2	2.27	2.55	0.0578	0.1225	28.8
2013	15.7	16.3	17.5	2.25	2.54	0.0568	0.1202	29.4
2014	13	16.3	17.8	2.23	2.53	0.0558	0.1179	30
2015	25.4	16.4	18.1	2.21	2.52	0.0547	0.1153	30.6
2016	27.6	16.2	18.2	2.2	2.52	0.0531	0.1109	30.6
2017	25.6	16	18.2	2.19	2.52	0.0512	0.1045	30
2018	19.1	15.8	18.1	2.19	2.52	0.0496	0.0981	29.3
2019	29.3	15.6	18	2.19	2.53	0.0481	0.0919	28.5
2020	43.8	15.5	18	2.19	2.53	0.0466	0.086	27.8
2021	26.1	15.4	17.9	2.19	2.54	0.0452	0.0802	27
2022	30.7	15.3	17.8	2.19	2.55	0.0438	0.0747	26.2

Standard total annual loads at Watts, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	23.1	8.29	9.58	1.566	1.87	0.0749	0.232	70.1
2011	28.1	6.9	8.63	1.386	1.94	0.1285	0.5529	224.5
2012	12.9	5.92	6.62	0.958	1.11	0.0277	0.0707	22.6
2013	15.7	6.22	7.28	1.006	1.24	0.0407	0.1206	44.2
2014	13	6.11	6.91	0.962	1.1	0.023	0.0506	14.3
2015	25.4	8.32	10.84	1.495	1.96	0.0795	0.2584	102.6
2016	27.6	7.8	10.47	1.599	2.18	0.1269	0.422	191.5
2017	25.6	6.65	8.83	1.219	1.78	0.1003	0.3737	194.7
2018	19.1	6.39	8.22	1.126	1.51	0.058	0.1727	91.1
2019	29.3	9.01	12.29	1.822	2.4	0.0787	0.2102	95.2
2020	43.8	11.42	16.54	2.589	3.59	0.1564	0.4168	211.8
2021	26.1	7.98	10.62	1.61	2.1	0.066	0.1759	93.2
2022	30.7	8.25	11.29	1.733	2.39	0.0921	0.2739	163.6

Flow-normalized total annual loads at Watts, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	23.1	8.01	9.47	1.56	1.94	0.0877	0.307	107
2011	28.1	8	9.64	1.54	1.94	0.0869	0.302	110
2012	12.9	7.98	9.79	1.52	1.93	0.0861	0.296	113
2013	15.7	7.96	9.94	1.51	1.93	0.0854	0.29	115
2014	13	7.95	10.08	1.49	1.93	0.0846	0.283	117
2015	25.4	7.94	10.23	1.48	1.93	0.0837	0.276	119
2016	27.6	7.84	10.26	1.46	1.93	0.0821	0.266	120
2017	25.6	7.68	10.16	1.45	1.93	0.0806	0.254	120
2018	19.1	7.57	10.06	1.44	1.93	0.0793	0.244	119
2019	29.3	7.47	9.96	1.44	1.93	0.0781	0.234	119
2020	43.8	7.38	9.85	1.43	1.93	0.0769	0.224	118
2021	26.1	7.29	9.75	1.43	1.94	0.0758	0.215	118
2022	30.7	7.21	9.64	1.43	1.94	0.0747	0.206	118

Baron

Standard annual mean concentrations at Baron, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	1.569	7.17	17.6	2.07	2.13	0.0499	0.0918	11.93
2011	1.594	8.5	20.7	2.05	1.87	0.0359	0.0708	11.21
2012	0.847	7.86	18.2	1.7	1.75	0.0305	0.0617	7.12
2013	0.746	8.42	21.1	2.02	1.87	0.0321	0.0602	6.65
2014	0.416	8.81	20.7	1.92	1.74	0.0252	0.0479	4.5
2015	1.99	7.05	19	1.96	2.1	0.053	0.1032	19.51
2016	1.66	7.59	19	1.78	1.83	0.0362	0.0692	11.74
2017	1.01	9.18	20.7	1.97	1.71	0.0306	0.0579	9.97
2018	1.087	7.97	19.5	1.94	1.75	0.0286	0.0489	8.58
2019	1.819	6.12	17.7	1.91	2.02	0.0455	0.0795	13.39
2020	2.518	5.45	16	1.68	1.97	0.0613	0.1003	17.07
2021	1.513	6.06	17.9	1.86	1.92	0.0377	0.055	9.31
2022	1.681	6.18	18	1.95	1.85	0.0357	0.0495	9.03

Flow-normalized annual mean concentrations at Baron, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	1.569	8	19.3	2	1.97	0.0432	0.0827	12.15
2011	1.594	7.93	19.3	1.96	1.95	0.0421	0.0814	12.02
2012	0.847	7.85	19.2	1.92	1.93	0.0413	0.0802	11.97
2013	0.746	7.77	19.2	1.9	1.92	0.0405	0.0788	11.97
2014	0.416	7.69	19.1	1.88	1.91	0.0399	0.0776	12.08
2015	1.99	7.62	19.1	1.86	1.9	0.0396	0.077	12.4
2016	1.66	7.5	19	1.85	1.89	0.0397	0.0766	12.67
2017	1.01	7.33	18.9	1.84	1.88	0.0391	0.0729	11.91
2018	1.087	7.17	18.8	1.86	1.87	0.0386	0.0682	11.13
2019	1.819	7	18.6	1.9	1.87	0.0381	0.0631	10.42
2020	2.518	6.83	18.5	1.96	1.88	0.0376	0.0581	9.78
2021	1.513	6.67	18.4	2.05	1.89	0.0371	0.0531	9.13
2022	1.681	6.51	18.3	2.14	1.9	0.0366	0.0485	8.52

Standard total annual loads at Baron, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	1.569	0.2513	0.736	0.1013	0.1199	0.005249	0.01119	2.529
2011	1.594	0.183	0.572	0.0705	0.1022	0.007227	0.02476	13.038
2012	0.847	0.1482	0.436	0.0593	0.0693	0.002349	0.00522	1.323
2013	0.746	0.1288	0.38	0.0396	0.049	0.001776	0.00439	1.196
2014	0.416	0.0794	0.231	0.0253	0.0286	7.39E-04	0.00159	0.312
2015	1.99	0.2351	0.753	0.0805	0.1289	0.008917	0.0231	8.086
2016	1.66	0.1779	0.574	0.0748	0.1097	0.009272	0.02283	11.919
2017	1.01	0.102	0.33	0.0344	0.059	0.004547	0.01229	5.328
2018	1.087	0.1265	0.414	0.0447	0.0658	0.004037	0.01016	4.616
2019	1.819	0.2205	0.739	0.0776	0.1146	0.00597	0.01354	4.208
2020	2.518	0.2862	0.972	0.1088	0.1629	0.01155	0.02212	6.163
2021	1.513	0.1763	0.608	0.0618	0.0897	0.004571	0.00969	2.982
2022	1.681	0.1763	0.625	0.0614	0.0931	0.005187	0.01097	3.501

Flow-normalized total annual loads at Baron, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	1.569	0.203	0.606	0.0805	0.1024	0.00556	0.0152	5.96
2011	1.594	0.199	0.6	0.0777	0.1013	0.00559	0.015	5.87
2012	0.847	0.196	0.595	0.075	0.1002	0.00565	0.0149	5.79
2013	0.746	0.192	0.589	0.0726	0.0992	0.0057	0.0148	5.75
2014	0.416	0.188	0.583	0.0704	0.0984	0.00577	0.0147	5.72
2015	1.99	0.184	0.578	0.0682	0.0977	0.00585	0.0147	5.71
2016	1.66	0.18	0.573	0.0658	0.0963	0.00589	0.0145	5.63
2017	1.01	0.176	0.569	0.0635	0.0936	0.0058	0.014	5.27
2018	1.087	0.171	0.563	0.0615	0.091	0.00568	0.0133	4.86
2019	1.819	0.167	0.557	0.0598	0.0887	0.00556	0.0125	4.5
2020	2.518	0.163	0.552	0.0584	0.0866	0.00544	0.0119	4.15
2021	1.513	0.159	0.547	0.0571	0.0845	0.00531	0.0112	3.83
2022	1.681	0.155	0.543	0.056	0.0825	0.00518	0.0105	3.52

WFWR

Standard annual mean concentrations at WFWR, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	5.4	6.58	28.9	0.466	0.643	0.00852	0.0756	34.5
2011	5.53	7.1	32.2	0.369	0.569	0.00687	0.0694	32.4
2012	3.89	7.01	31.3	0.364	0.569	0.00667	0.0543	22.5
2013	3.49	7.24	33.9	0.301	0.54	0.00635	0.0611	26.8
2014	4.15	6.62	30.3	0.346	0.594	0.00744	0.0626	26.6
2015	8.58	6.29	29.4	0.324	0.633	0.00921	0.0982	57.7
2016	7.8	6.14	29.2	0.306	0.597	0.00828	0.0744	43
2017	5.57	6.61	31.3	0.246	0.526	0.00626	0.0594	31.3
2018	4.43	6.65	31.6	0.23	0.499	0.0056	0.0489	25.4
2019	7.54	5.46	26.2	0.29	0.594	0.00759	0.0689	43.4
2020	10.16	5.15	24.5	0.297	0.63	0.009	0.0823	61.1
2021	5.59	5.69	27.4	0.268	0.512	0.00566	0.0454	31.2
2022	6.8	5.57	26.8	0.25	0.523	0.00562	0.0478	35.6

Flow-normalized annual mean concentrations at WFWR, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	5.4	6.74	30.2	0.443	0.642	0.00841	0.0824	39.6
2011	5.53	6.73	30.3	0.41	0.631	0.00837	0.0815	39.4
2012	3.89	6.7	30.4	0.38	0.62	0.00832	0.0806	39.3
2013	3.49	6.66	30.4	0.355	0.61	0.00823	0.079	39.1
2014	4.15	6.61	30.4	0.333	0.602	0.00811	0.077	38.9
2015	8.58	6.55	30.5	0.314	0.593	0.00792	0.0742	38.7
2016	7.8	6.42	30.3	0.297	0.583	0.00759	0.0712	38.5
2017	5.57	6.28	29.7	0.285	0.567	0.00718	0.0664	37
2018	4.43	6.16	29.3	0.275	0.552	0.00682	0.0618	36
2019	7.54	6.04	28.8	0.266	0.539	0.00649	0.057	35.3
2020	10.16	5.92	28.4	0.258	0.526	0.00617	0.0525	34.6
2021	5.59	5.81	28	0.252	0.515	0.00585	0.0482	33.9
2022	6.8	5.71	27.6	0.248	0.504	0.00556	0.0443	33.4

Standard total annual loads at WFWR, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	5.4	0.926	3.62	0.0859	0.1408	0.00325	0.0387	22.7
2011	5.53	0.769	2.95	0.0719	0.1461	0.00344	0.0608	42.6
2012	3.89	0.727	2.6	0.0648	0.1152	0.00227	0.0259	17.8
2013	3.49	0.547	2.29	0.039	0.0856	0.00146	0.0221	14
2014	4.15	0.75	2.99	0.0545	0.1075	0.00174	0.0207	13.6
2015	8.58	0.998	4.68	0.0849	0.2402	0.00476	0.0786	59.8
2016	7.8	0.881	4.05	0.0908	0.2132	0.00498	0.0503	58.3
2017	5.57	0.638	2.82	0.0515	0.1545	0.00254	0.0457	40
2018	4.43	0.567	2.27	0.0459	0.1277	0.00204	0.0314	27
2019	7.54	0.957	4.07	0.0773	0.2033	0.0033	0.0537	48.4
2020	10.16	1.197	5.18	0.1173	0.2986	0.00611	0.0677	62.1
2021	5.59	0.651	2.88	0.0539	0.1442	0.0023	0.0315	30.6
2022	6.8	0.799	3.1	0.0646	0.1855	0.00256	0.0398	39.4

Flow-normalized total annual loads at WFWR, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	5.4	0.99	3.71	0.0953	0.165	0.00376	0.0516	33.2
2011	5.53	0.964	3.71	0.0892	0.166	0.00367	0.0506	33.9
2012	3.89	0.935	3.7	0.0835	0.166	0.00358	0.0497	34.6
2013	3.49	0.906	3.68	0.0785	0.167	0.00349	0.0486	35.3
2014	4.15	0.875	3.66	0.0741	0.168	0.0034	0.0475	36.2
2015	8.58	0.843	3.64	0.0701	0.169	0.0033	0.0462	37.2
2016	7.8	0.809	3.57	0.0669	0.17	0.00317	0.0452	38.1
2017	5.57	0.783	3.42	0.0653	0.17	0.00306	0.0437	37.6
2018	4.43	0.76	3.29	0.0637	0.169	0.00298	0.0424	37.7
2019	7.54	0.736	3.17	0.0623	0.168	0.00292	0.0411	38.2
2020	10.16	0.713	3.05	0.0611	0.168	0.00286	0.0399	38.7
2021	5.59	0.692	2.93	0.0602	0.167	0.0028	0.0387	39.2
2022	6.8	0.672	2.83	0.0595	0.167	0.00276	0.0377	39.9

Wyman

Standard annual mean concentrations at Wyman, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	19.5	3.7	13.5	0.441	0.619	0.00759	0.0708	28.7
2011	20.3	4.13	16.1	0.357	0.566	0.00656	0.0648	25.9
2012	11.3	4.25	16.4	0.339	0.565	0.00584	0.0483	15.9
2013	11.3	4.15	17	0.305	0.555	0.00591	0.0571	21.6
2014	11.1	3.69	14.2	0.336	0.573	0.00619	0.054	19.7
2015	25	3.34	12.9	0.345	0.647	0.00866	0.0889	44.6
2016	18.3	3.55	13.9	0.306	0.578	0.0067	0.0608	27.1
2017	14.5	3.8	15.4	0.242	0.529	0.00562	0.0547	24
2018	15.8	3.74	15.1	0.243	0.53	0.00574	0.0494	22.6
2019	21.8	3.14	12.4	0.296	0.591	0.0067	0.0622	31.3
2020	29.5	3.11	12.2	0.301	0.629	0.00905	0.0725	41.8
2021	20.4	3.36	13.4	0.26	0.542	0.00591	0.0489	26.7
2022	19.9	3.52	14.3	0.235	0.516	0.00532	0.0423	24.1

Flow-normalized annual mean concentrations at Wyman, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	19.5	3.92	15.1	0.414	0.601	0.00722	0.0706	28.5
2011	20.3	3.87	14.9	0.393	0.602	0.00727	0.0702	28.8
2012	11.3	3.81	14.7	0.372	0.603	0.00731	0.0698	29.2
2013	11.3	3.76	14.5	0.353	0.603	0.00733	0.069	29.4
2014	11.1	3.71	14.4	0.336	0.604	0.00733	0.0682	29.6
2015	25	3.66	14.2	0.32	0.604	0.00729	0.0673	29.9
2016	18.3	3.62	14.2	0.304	0.596	0.00712	0.0657	29.7
2017	14.5	3.61	14.2	0.288	0.581	0.00676	0.0618	28.6
2018	15.8	3.6	14.2	0.274	0.566	0.00643	0.0578	27.8
2019	21.8	3.59	14.2	0.261	0.552	0.00612	0.0538	27
2020	29.5	3.58	14.2	0.249	0.54	0.00581	0.05	26.1
2021	20.4	3.57	14.3	0.239	0.528	0.0055	0.0463	25.1
2022	19.9	3.56	14.4	0.23	0.517	0.00521	0.0429	24.3

Standard total annual loads at Wyman, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	19.5	1.93	6.4	0.31	0.479	0.01088	0.1143	61.7
2011	20.3	1.5	4.95	0.268	0.547	0.01533	0.2346	145.6
2012	11.3	1.2	3.7	0.199	0.32	0.00615	0.0598	34.5
2013	11.3	1.08	3.71	0.145	0.258	0.00431	0.0499	27.7
2014	11.1	1.11	3.79	0.153	0.271	0.00445	0.0425	24.8
2015	25	1.86	7.32	0.268	0.654	0.01493	0.1741	113
2016	18.3	1.38	5.28	0.233	0.525	0.01221	0.1106	108.3
2017	14.5	1.05	3.97	0.139	0.38	0.00801	0.0955	67.6
2018	15.8	1.3	4.47	0.181	0.441	0.00914	0.0883	65.3
2019	21.8	1.83	6.57	0.23	0.528	0.00959	0.109	71.5
2020	29.5	2.48	8.77	0.352	0.804	0.01951	0.1536	108.1
2021	20.4	1.61	5.84	0.199	0.5	0.01078	0.1028	73.9
2022	19.9	1.62	5.53	0.186	0.489	0.00957	0.095	68.8

Flow-normalized total annual loads at Wyman, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	19.5	1.78	5.69	0.292	0.47	0.0103	0.128	72
2011	20.3	1.73	5.67	0.278	0.477	0.0104	0.125	73.5
2012	11.3	1.68	5.64	0.265	0.482	0.0106	0.122	74.8
2013	11.3	1.63	5.61	0.254	0.488	0.0107	0.119	76.1
2014	11.1	1.58	5.59	0.243	0.495	0.0108	0.116	77.2
2015	25	1.53	5.57	0.233	0.502	0.0109	0.113	78.3
2016	18.3	1.49	5.53	0.222	0.503	0.0108	0.112	78.9
2017	14.5	1.48	5.44	0.213	0.497	0.0106	0.11	78.3
2018	15.8	1.47	5.37	0.204	0.49	0.0105	0.108	78.1
2019	21.8	1.46	5.3	0.196	0.484	0.0103	0.106	77.9
2020	29.5	1.45	5.24	0.188	0.477	0.0102	0.104	77.7
2021	20.4	1.44	5.19	0.182	0.471	0.01	0.102	77.3
2022	19.9	1.44	5.14	0.176	0.464	0.0099	0.1	77.1

Richland

Standard annual mean concentrations at Richland, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2016	4.43	4.19	9.45	1.24	1.45	0.0128	0.0574	29.1
2017	3.76	5.09	10.98	1.26	1.53	0.0108	0.0511	33.7
2018	3.13	4.7	10.7	1.24	1.47	0.0105	0.0367	17.1
2019	5.74	3.71	9.73	1.15	1.39	0.0148	0.0508	22.6
2020	7.13	3.72	9.37	1.12	1.41	0.0166	0.055	23.6
2021	4.94	3.95	10.15	1.1	1.32	0.0123	0.0347	17
2022	5.05	3.96	10.54	1.11	1.33	0.0122	0.031	14.7

Flow-normalized annual mean concentrations at Richland, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2016	4.43	4.27	9.65	1.25	1.51	0.0141	0.0694	31.1
2017	3.76	4.23	9.78	1.23	1.48	0.0136	0.0587	27.7
2018	3.13	4.18	9.91	1.2	1.45	0.0133	0.0504	24.7
2019	5.74	4.14	10.05	1.17	1.42	0.013	0.0438	22.2
2020	7.13	4.1	10.19	1.15	1.39	0.0127	0.0385	20.1
2021	4.94	4.06	10.33	1.12	1.36	0.0125	0.0342	18.3
2022	5.05	4.02	10.47	1.09	1.33	0.0123	0.0307	16.7

Standard total annual loads at Richland, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2016	4.43	0.351	1.005	0.1164	0.208	0.00995	0.0613	175.4
2017	3.76	0.253	0.8	0.0668	0.163	0.00559	0.0474	91
2018	3.13	0.255	0.723	0.068	0.127	0.00301	0.0205	22.4
2019	5.74	0.489	1.387	0.1431	0.236	0.00572	0.0348	32.7
2020	7.13	0.637	1.785	0.2036	0.316	0.00871	0.0389	28.3
2021	4.94	0.426	1.209	0.1213	0.195	0.00542	0.0298	28.7
2022	5.05	0.437	1.224	0.1134	0.185	0.00401	0.0224	15.1

Flow-normalized total annual loads at Richland, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2016	4.43	0.397	1.14	0.122	0.218	0.0064	0.0414	67.5
2017	3.76	0.399	1.14	0.122	0.214	0.00629	0.0397	61.8
2018	3.13	0.402	1.15	0.121	0.21	0.0062	0.0383	56.5
2019	5.74	0.405	1.15	0.12	0.207	0.00611	0.0371	51.8
2020	7.13	0.408	1.16	0.119	0.202	0.00604	0.036	47.5
2021	4.94	0.411	1.16	0.118	0.198	0.00599	0.0352	43.8
2022	5.05	0.415	1.17	0.116	0.194	0.00593	0.0345	40.4

RC45

Standard annual mean concentrations at RC45, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	5.42	4.29	13.2	0.856	0.992	0.0113	0.0549	24.1
2011	7.79	4.6	14.1	0.808	0.894	0.00946	0.0582	33
2012	2.88	4.72	12.8	0.76	0.875	0.00756	0.039	12.5
2013	2.9	4.61	13.8	0.85	0.98	0.00996	0.052	16.7
2014	2.66	4.55	12.7	1.132	1.159	0.00941	0.0493	12.6

Flow-normalized annual mean concentrations at RC45, as estimated by WRTDS

Year	Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	5.42	4.55	13.7	0.764	0.871	0.00903	0.0488	23.2
2011	7.79	4.54	13.5	0.828	0.94	0.00969	0.0523	23.4
2012	2.88	4.52	13.3	0.898	1.012	0.01049	0.0565	23.7
2013	2.9	4.51	13.1	0.975	1.088	0.01142	0.0612	24.1
2014	2.66	4.49	12.8	1.061	1.169	0.01254	0.0668	24.6

Standard total annual loads at RC45, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	5.42	0.61	1.95	0.1574	0.209	0.00564	0.035	25.2
2011	7.79	0.613	1.77	0.1763	0.309	0.01376	0.1838	219.1
2012	2.88	0.369	1.06	0.1008	0.132	0.00236	0.0166	11.79
2013	2.9	0.356	1.07	0.0882	0.122	0.00207	0.013	6.12
2014	2.66	0.357	0.98	0.0985	0.125	0.00173	0.0123	6.96

Flow-normalized total annual loads at RC45, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	5.42	0.504	1.56	0.13	0.189	0.00641	0.0607	58.4
2011	7.79	0.507	1.54	0.133	0.198	0.00633	0.0612	59
2012	2.88	0.51	1.52	0.136	0.207	0.00629	0.0618	59.6
2013	2.9	0.514	1.51	0.139	0.216	0.00629	0.0628	60.9
2014	2.66	0.518	1.48	0.143	0.225	0.00634	0.064	62.3

WEC

Standard annual mean concentrations at WEC, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Concentration (mg/L)						
2010	11.36	8.69	6.76	1.5	1.64	0.01418	0.058	20.4
2011	12.48	11.08	7.11	1.49	1.64	0.0107	0.0543	22.4
2012	7.84	11.08	6.91	1.39	1.57	0.00919	0.0428	12.9
2013	7.12	11.24	7.29	1.4	1.59	0.00986	0.0449	14.7
2014	7.57	10.06	7.22	1.4	1.6	0.00935	0.0413	12.7
2015	10.41	9.76	7.51	1.45	1.67	0.01118	0.0519	20.7
2016	12.61	8.93	7.3	1.48	1.71	0.01086	0.0464	16.4
2017	10.51	10.29	7.51	1.44	1.67	0.00935	0.0425	17.6
2018	8.25	10.98	7.41	1.47	1.7	0.00792	0.0338	13.9
2019	13.62	7.46	7.06	1.48	1.74	0.01455	0.0538	22.5
2020	19.49	6.71	6.74	1.52	1.83	0.01921	0.0694	32.5
2021	12.88	7.37	6.86	1.56	1.82	0.014	0.0434	19.5
2022	10.35	7.55	6.95	1.57	1.81	0.01205	0.0345	16.1

Flow-normalized annual mean concentrations at WEC, as estimated by WRTDS

Year	Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized concentration (mg/L)						
2010	11.36	10.14	6.83	1.49	1.64	0.0123	0.0577	22.4
2011	12.48	10.02	6.93	1.47	1.64	0.012	0.0562	21.5
2012	7.84	9.91	7.03	1.46	1.64	0.0116	0.0548	20.7
2013	7.12	9.8	7.13	1.44	1.64	0.0113	0.0533	19.8
2014	7.57	9.7	7.24	1.44	1.65	0.011	0.0516	19
2015	10.41	9.6	7.35	1.44	1.66	0.0108	0.0498	18.3
2016	12.61	9.45	7.38	1.45	1.68	0.0108	0.048	17.9
2017	10.51	9.22	7.3	1.46	1.7	0.0111	0.0465	18
2018	8.25	9.04	7.25	1.48	1.72	0.0115	0.0449	18
2019	13.62	8.87	7.19	1.5	1.74	0.0118	0.0431	17.9
2020	19.49	8.7	7.12	1.52	1.77	0.012	0.0411	17.8
2021	12.88	8.54	7.06	1.55	1.8	0.0123	0.0391	17.7
2022	10.35	8.37	6.98	1.59	1.84	0.0126	0.0373	17.7

Standard total annual loads at WEC, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Load (million kg)						
2010	11.36	1.95	2.19	0.49	0.588	0.01121	0.0716	36.5
2011	12.48	1.49	1.88	0.414	0.575	0.01411	0.145	89.2
2012	7.84	1.31	1.56	0.336	0.425	0.00593	0.0462	26.3
2013	7.12	1.3	1.46	0.291	0.36	0.00478	0.0284	14.6
2014	7.57	1.36	1.6	0.322	0.402	0.00448	0.0295	16
2015	10.41	1.59	2.01	0.39	0.529	0.00884	0.0568	33.5
2016	12.61	1.71	2.33	0.511	0.697	0.01707	0.0913	60.7
2017	10.51	1.3	1.76	0.336	0.501	0.00933	0.0744	47.9
2018	8.25	1.08	1.46	0.288	0.427	0.00723	0.0547	36.1
2019	13.62	1.94	2.62	0.549	0.742	0.01144	0.0616	34.4
2020	19.49	2.54	3.74	0.84	1.158	0.0236	0.1067	61.5
2021	12.88	1.7	2.38	0.533	0.725	0.01313	0.0638	38.7
2022	10.35	1.41	1.92	0.415	0.565	0.00788	0.0435	27.4

Flow-normalized total annual loads at WEC, as estimated by WRTDS

Year	Annual Mean Daily Streamflow (cms)	Cl	SO_4	NO_3	TN	SRP	TP	TSS
		Flow-normalized load (million kg)						
2010	11.36	1.72	2.02	0.451	0.568	0.01037	0.0818	47.9
2011	12.48	1.71	2.04	0.444	0.568	0.01017	0.0785	46
2012	7.84	1.7	2.06	0.439	0.568	0.01002	0.0753	44
2013	7.12	1.69	2.09	0.435	0.57	0.00991	0.0723	42.2
2014	7.57	1.68	2.11	0.433	0.574	0.00985	0.0691	40.4
2015	10.41	1.66	2.14	0.432	0.579	0.00983	0.0656	38.6
2016	12.61	1.63	2.15	0.435	0.586	0.00996	0.0634	37.8
2017	10.51	1.59	2.12	0.435	0.593	0.01037	0.0637	38.4
2018	8.25	1.56	2.1	0.436	0.599	0.01075	0.0633	38.5
2019	13.62	1.53	2.08	0.439	0.607	0.01114	0.0629	38.6
2020	19.49	1.5	2.05	0.443	0.616	0.01153	0.0623	38.6
2021	12.88	1.46	2.03	0.449	0.627	0.01195	0.062	38.9
2022	10.35	1.43	2.01	0.455	0.638	0.01241	0.0618	39.2