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Constituent Loads and Trends in the Upper White River Basin: A Nonpoint Source Management Program Priority Watersheds

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Executive Summary

The Arkansas Department of Agriculture – Natural Resources Division (ANRD) has identified the Upper White River Basin (UWRB; HUC 11010001) a hydrologic unit code (HUC) 8 watersheds, located in Northwest Arkansas, for prioritization by the Nonpoint Source (NPS) Management Program. UWRB 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 UWRB. This watershed is 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 is necessary to identify whether these interventions are influencing 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 this priority watershed 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 the 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. UWRB sites had similar watershed characteristics making potential watershed effects complex to decipher.

Trend analysis suggested that phosphorus (TP and SRP) concentrations have decreased over the last 15 years throughout the UWRB. Although, concentrations decreases have not yet resulted in phosphorus load reductions in the UWRB.

Other potential water quality gains included decreasing nitrogen concentrations, loads 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.

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 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 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 UWRB have implications for watershed management. 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 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 1), which are all located at U.S. Geological Survey (USGS) stream gaging stations (see Table 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 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). 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 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. By adjusting all the constituent concentrations and loads for streamflow variability our understanding of how these values vary through time will be enhanced. 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 this 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 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 2) and total annual FN loads (Figure 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 narrower 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.

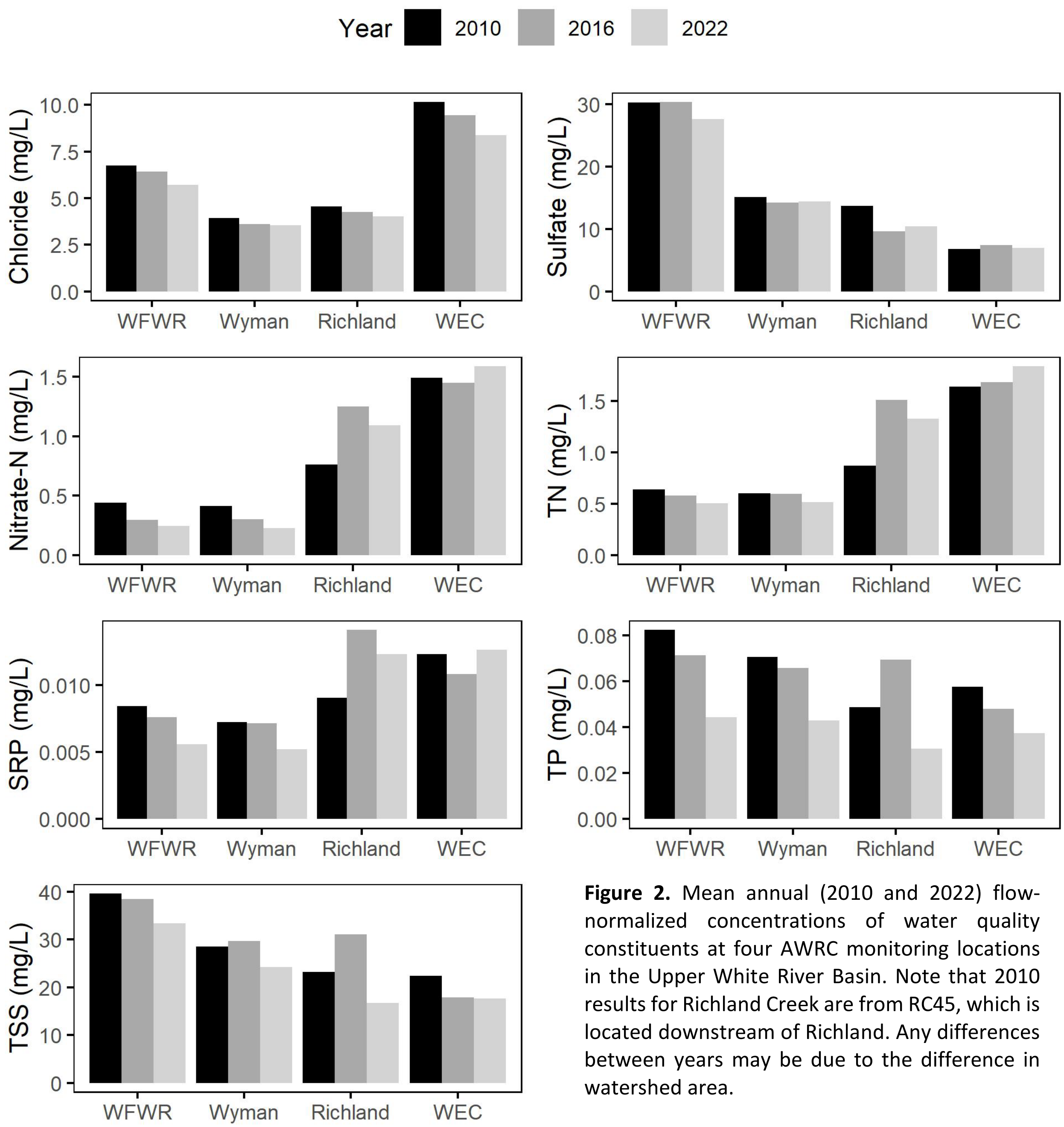


Figure 2. Mean annual (2010 and 2022) flow-normalized concentrations of water quality constituents at four AWRC monitoring locations in the Upper White River Basin. Note that 2010 results for Richland Creek are from RC45, which is located downstream of Richland. Any differences between years may be due to the difference in watershed area.

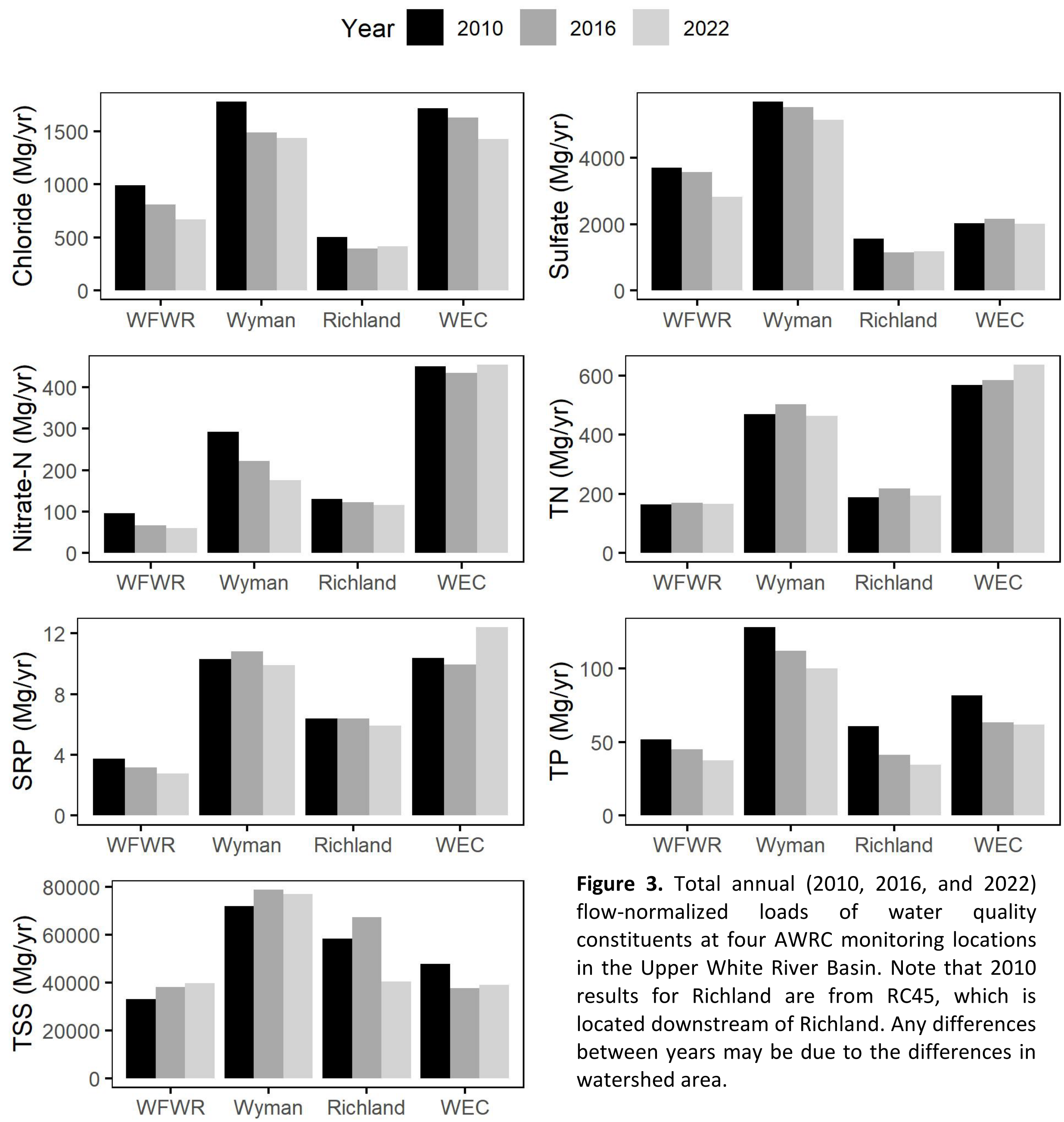


Figure 3. Total annual (2010, 2016, and 2022) flow-normalized loads of water quality constituents at four AWRC monitoring locations in the Upper White River Basin. Note that 2010 results for Richland are from RC45, which is located downstream of Richland. Any differences between years may be due to the differences in watershed area.

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) and FN loads (Table 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.

Table 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 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)

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.

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 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 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.

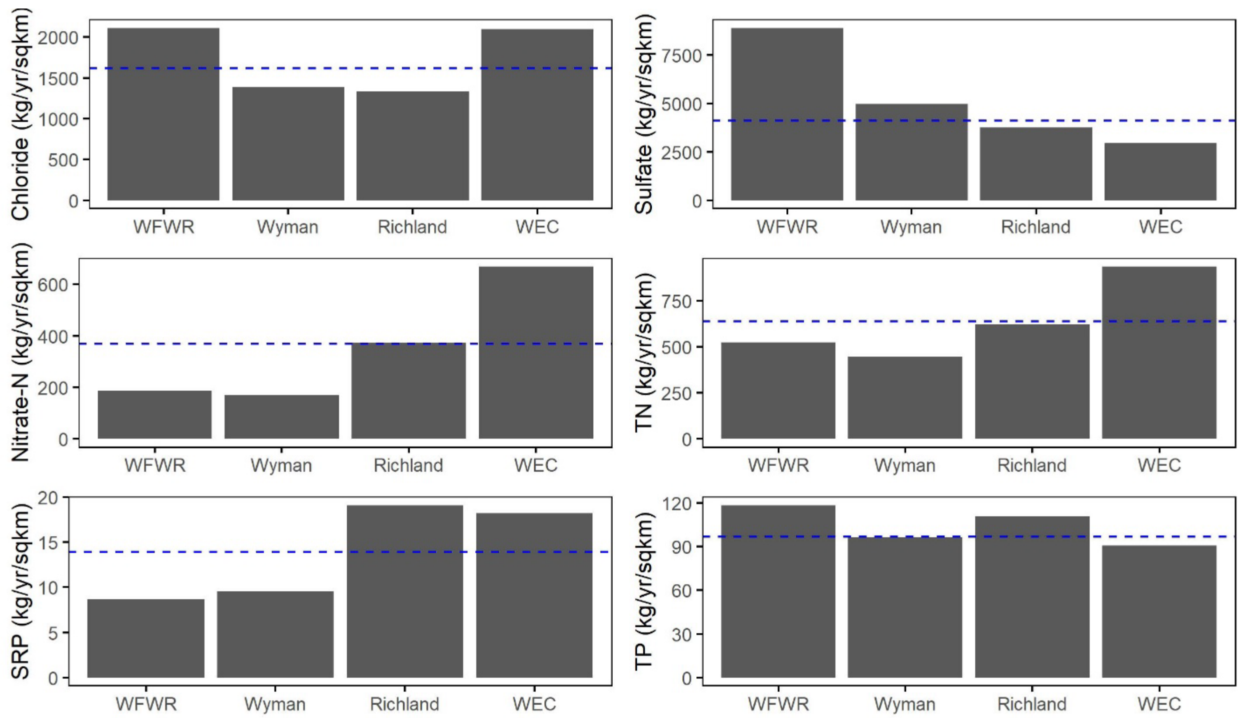
Watershed perspectives on load and yield

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 4 to facilitate comparisons between sites. As seen in Figure 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 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. If sites had a disproportionally larger or smaller yield they 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.



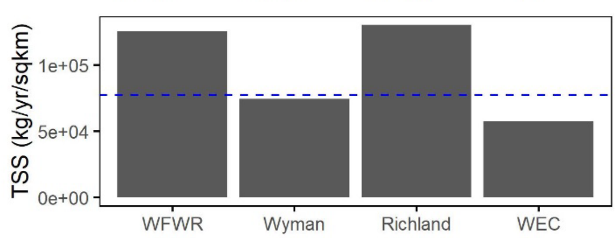


Figure 4. Constituent yields, or flow-adjusted loads per square km of watershed, at all UWRB sites in 2022. The dashed blue line shows the yield of the combined watershed across sites. Yield above the line means a relatively greater load for the watershed size, while yield below the line means relatively 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. This variability in spatial patterns have implications for watershed management practices. Similarities between sites make it challenging to differentiate NPS and point-source contributions. Although, 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

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