Relationships between water quality, stream metabolism, and water stargrass growth in the lower Yakima River, 2018 to 2020

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Abstract

Since the early 2000s, water clarity on the lower Yakima River has improved. Changes in best management practices combined with a total maximum daily load for suspended sediment led to these improved conditions. As water clarity improved, so did conditions for aquatic plants; the clearer the water, the better the light penetration, and dramatic increases in plant biomass were observed. In the lower Yakima River, beds of native water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) are prolific and can extend bank to bank in some locations. Increased primary productivity can alter local water quality by increasing daily swings of dissolved oxygen (DO) and pH from photosynthesis. In this study, we collected continuous water quality data for 2.5 years at three sites on the lower Yakima River to provide a detailed examination of water quality conditions. These sites were located just below the Prosser Dam (Prosser site, USGS station 12509489), at a long-term USGS streamgage in Benton County (Kiona site, USGS station 12510500), and in West Richland, WA (Van Giesen site; USGS station 12511800). In addition to the continuous water quality data collected, estimates of water stargrass biomass were made through the growing season (June through September) during water years 2018–2020. The main objectives of this study were to document water quality conditions on the lower Yakima River and to analyze if there was a statistical relation between the amount of water stargrass biomass and the observed daily cycles of water quality.

During summer, frequent exceedances of established water quality criteria were documented each year during this study. Maximum daily temperatures exceeded 21° C, minimum DO concentrations were below 8 milligrams per liter (mg/L), and maximum pH surpassed 8.5 almost every day from June through August each water year across all three monitoring locations. Water stargrass biomass tended to increase from June through August and

September but was ‘reset’ by the following summer likely from high winter and spring streamflows and natural die-off. Results from this study suggest that spring peak discharge and average spring discharge affects late-season water stargrass biomass. In 2018, the highest peak discharge of the study took place, and the August water stargrass biomass values were lower in 2018 than in 2019 and 2020.

Seven different water quality metrics were computed for a 7-day and 28-day period prior to each water stargrass sample to examine possible correlations between the plant biomass and water quality. We examined daily maximum temperature, DO minimum, DO range, pH maximum, pH range, mean nitrate, and nitrate range. While there were some statistically significant correlations among the seven water quality metrics and median water stargrass biomass, the correlations were not consistent across all three sites. At the Prosser site, the 7-day average daily maximum pH and average daily pH range showed significant correlations with median water stargrass biomass. At the Kiona site, both the 7-day and 28-day mean nitrate values showed a significant relationship to median water stargrass biomass. At the Van Giesen site, there were no significant correlations between the seven water quality metrics and median water stargrass biomass. However, whole-stream estimates of gross primary productivity at the Kiona site, which incorporate the entire river community, were related to temperature, DO, and pH indicating the whole river community is influencing surface water quality to some extent.

Additional data on water stargrass biomass and continuous water quality could help elucidate the complex interactions between growth and water quality. At a minimum, collection of water stargrass biomass data near the end of the growing season (mid to late August) could be added to locations where continuous water quality and streamflow discharge measurements are also being collected. In addition, experimental removal of water stargrass and its effects on local water quality could provide insight into the complex relationships between water stargrass growth and water quality. Finally, further investigations into streamflow and its effects on water stargrass could be improved. Our data showed a qualitative relationship between spring peak discharge, average spring discharge, and August water stargrass biomass, but more data are needed to confirm this. If spring high streamflows are important for late-season biomass, then targeted flow releases from reservoirs in the upper watershed could be used to slow down water stargrass growth during summer months.

Introduction

The Yakima River drains a 6,155 mi^2 basin on the east side of the Cascade Range in south-central Washington (**Figure 1**). The area is one of the most intensively irrigated areas in the United States because only 20 to 40 percent of the annual precipitation occurs during the agricultural growing season between March and October; therefore, most crops need to be irrigated (Rinella and others, 1992; Morace and others, 1999). Surface-water diversions for irrigation are equivalent to about 60 percent of the annual streamflow for the basin (Morace and others, 1999). A large Bureau of Reclamation irrigation project includes six large storage reservoirs in the northwestern part of the basin that were constructed between 1908 and 1933 and 14 major diversions from the main-stem Yakima River that feed six major irrigation-district projects and numerous small irrigation systems (Rinella and others, 1992).

Along with significant water diversions in the basin, irrigated agriculture historically caused high levels of nutrients and suspended sediment to the lower Yakima River (Ebbert and others, 2003; Fuhrer and others, 2004) by increasing direct connection between fields, the river, and its tributaries (McCarthy and Johnson, 2009). Historical concentrations of phosphorus and nitrogen measured in the lower Yakima River have been high enough to support abundant growth of phytoplankton, periphyton, and macrophytes (Rinella and others, 1992), but macrophyte growth was not widespread or problematic. In the 1980s and 1990s, several studies noted patches of dense macrophytes located sporadically within the lower basin (Rinella and others, 1992; Morace and others, 1999), whereas other presence-absence surveys, which included the Yakima River at Kiona, did not note any macrophytes (Cuffney and others, 1997).

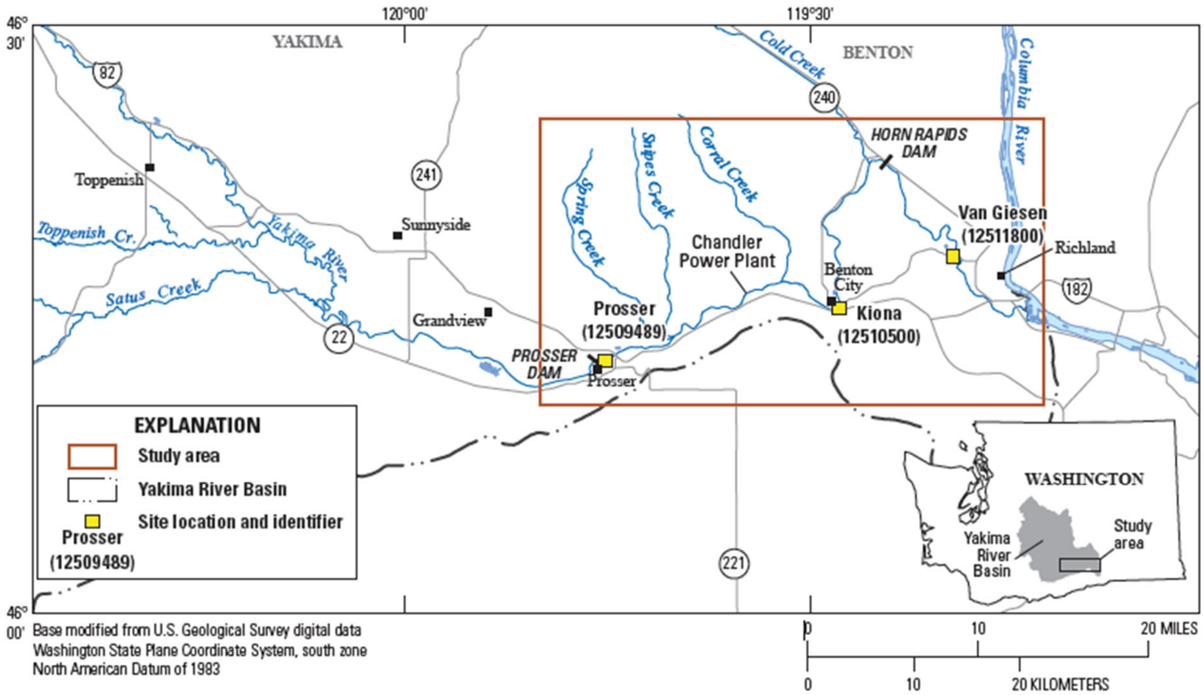

Figure 1. Location of the lower Yakima River reach within the Yakima River Basin and the three U.S. Geological Survey (USGS) monitoring stations used in this study. Information and data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

However, as a result of water quality improvements within the Yakima Basin, including a Total Maximum Daily Load (TMDL) to reduce suspended sediment loads (Joy and Patterson, 1997), nutrient loads and suspended sediment concentration in the lower Yakima River are much lower after 2000 compared to prior years (Ebbert and others, 2003; Fuhrer and others, 2004; Wise and others, 2009). As a result, light availability combined with altered streamflow from irrigation have been favorable for macrophyte growth, with abundant macrophyte growth now observed throughout the lower Yakima Basin. Wise and others (2009) reported observations of USGS hydrographers noting an increase in 'river grass' around the Kiona streamgage in June 2001 following many years of decreasing turbidity and spring streamflow.

Many parts of the lower 43 miles of the Yakima River, below Prosser Dam (referred to herein as the lower Yakima River), are frequently dominated during the summer by the rooted aquatic macrophyte *Heteranthera dubia*, or grass-leaf mud-plantain (hereafter referred to by the locally used name water stargrass; Wise and others, 2009). Water stargrass thrives in the lower Yakima River in a variety of habitats, ranging from finely silted slack water to higher velocity cobble substrates (Hamel and Parsons, 2001). Despite its classification as a native aquatic plant, water stargrass in the lower Yakima River can be invasive in certain locations, exploiting river conditions (Appel and others, 2011). Water stargrass can form bank-to-bank monocultures (**Figure 2**) with magnified effects in low-water years. Water stargrass has been observed in moderate densities in sections between the City of Prosser and Benton City where faster riffles and large rocky substrates dominate (Appel and others, 2011).

Figure 2. Water stargrass (*Heteranthera dubia*) beds at the Yakima River near Kiona, WA (USGS station 12510500; U.S. Geological Survey, 2022); adopted from Wise and others, 2009; photograph by Kurt Carpenter, U.S. Geological Survey, July 2005.

Macrophytes in streams and rivers have been shown to alter the physical and chemical conditions in streams and rivers including increased sedimentation, reduction of streamflow, increased water temperatures, alteration of groundwater-surface water interactions, and influences on water quality through photosynthesis (Hendricks and White, 1998; Duff and others, 2002; Wise and others, 2009). Fall Chinook salmon (*Oncorhynchus tshawytscha*) spawning, once prevalent in the lower Yakima River, has shifted above Prosser Dam because of decreased spawning gravel quality (Appel and others, 2011). In addition, Wise and others (2009) reported that water stargrass greatly influenced river pH and dissolved oxygen (DO) levels during the irrigation season causing degraded water quality conditions. McMichael (2017) noted that high summer river temperatures, lower DO levels, increased pH levels, and dense water stargrass strands might provide favorable conditions for the recruitment and refuge of non-native predator species to the detriment of native salmon species. In addition, a recent study demonstrated that water stargrass beds in the lower Yakima River had greater primary productivity compared to areas without stargrass and the stargrass beds were able to produce localized low DO areas within their habitats (Pelly, 2020).

Dense water stargrass stands also degrade side-channel habitat, block irrigation ladders at the lower Yakima River dams, impact irrigation function, and impede recreational benefits of the lower Yakima River (Appel and others, 2011). More recently, it was discovered that water stargrass may impact local public health. Flowing river water is slowed in areas of dense water stargrass growth resulting in ponded areas that provide breeding grounds for disease-carrying mosquitos (Pelly and others, 2024).

In response to problematic water stargrass growth, South Yakima Conservation District and the U.S. Geological Survey (USGS) monitored the river and its aquatic plants from 2004 to 2007 along the lower 116 miles of the Yakima River (Wise and others 2009). This study was designed as a first step in understanding large-scale relationships between water quality (temperature, pH, DO, conductivity, and turbidity), nutrients, and abundant aquatic plant growth. Wise and others (2009) concluded that light availability as influenced by turbidity, phytoplankton abundance, and water depth, was more likely to limit macrophyte growth than nutrient availability. The limited influence of nutrients on macrophyte growth might be related to nutrient saturation as well. For example, Wise and others (2009) studied the influence of nutrient additions on periphyton growth and showed that only in a few instances did added nutrients result in increased growth. Furthermore, Mebane and others (2021) observed growth limitation in the Lower Snake River at N and P levels similar to what is observed in the lower Yakima River.

Wise and others (2009) concluded that the development of management actions to mitigate water stargrass would require further detailed research into the complex relations between aquatic plant growth, nutrients, DO, and pH, and by utilizing continuous monitoring on a more refined reach-scale. As a result, Benton Conservation District, in cooperation with the USGS, initiated a more comprehensive and detailed water-quality monitoring effort on the lower Yakima River to increase our understanding of the complex relations and drivers between water quality and growth of water stargrass at a smaller scale than conducted by Wise and others (2009).

This report details the water quality and water stargrass biomass collection in the lower Yakima River from spring 2018 through fall 2020 and addresses the recommendation from Wise and others (2009) for more detailed water quality data collection in areas of high macrophyte abundance. In addition, this report expands on the work of Wise and others (2009) to include multiple measures of water stargrass over the growing season to better examine relationships between water quality and macrophyte growth. The Yakima River Basin is considered one of the most vulnerable watersheds in Washington State for climate change impacts, with summers predicted to become warmer and dryer (Pickett, 2016). As water demands in the Yakima River Basin increase, it is imperative to address how to improve lower Yakima River water quality, support thermal refuge locations for fish, and mitigate impacts from water stargrass growth.

Purpose and scope

This study expands on the previous work of USGS by focusing monitoring on the lower Yakima River (referred to as the Kiona reach in Wise and others, 2009) at three locations for a duration of 2.5 years. Continuous water quality parameters included temperature, pH, specific conductance, dissolved oxygen (DO), turbidity, and nitrate. To date, there are few continuous turbidity and nitrate data available for the lower Yakima River, and results from this study represent the most detailed and current data available for those variables. In addition to the water quality data collection, the biomass of water stargrass was measured through the growing season during water years 2018–2020. A water year is a 12-month period from October 1st through September 30th that is named for the calendar year in which it ends. The simultaneous collection of water quality data and in-river macrophyte cover and biomass provided an opportunity to examine relationships between these two variables and to support the development of management strategies to improving river conditions in the future.

Description of lower Yakima River

The lower Yakima River, located in south-central Washington State, flows through arid Benton County (**Figure 1**). The Yakima River passes through the cities of Prosser and Benton and loosely forms the dividing line between Richland and West Richland in the Tri-Cities before joining the Columbia River in Richland (Appel and others, 2011). Agriculture is the primary land use in Benton County, supported by irrigation from the Yakima River. Benton County irrigation use for agriculture and growing urban/residential development heavily influences Yakima River water quality and seasonal streamflow. The Yakima River is a highly managed system with regulated yearly streamflow regimes. The spring freshet typically occurs between April and May, with low streamflows and high temperatures occurring between June and August. Water temperatures rapidly cool with the onset of fall sometime between late August and early September. The irrigation season runs from mid-March to mid-October, during which agricultural users draw water from the Yakima River (Wise and others, 2009). Basalt rock from the Columbia River Basalts Group dominate between the City of Prosser and Benton City, confining the river channel throughout this reach with minimal area for braiding and meandering. Alluvial deposits present between Horn Rapids and West Richland were formed by historical floods and are dispersed throughout this reach. There are few floodplains and island side-channels on the lower Yakima River that mainly exist within Benton City and around Richland, Washington (Appel and others 2011).

Previous USGS studies indicate the lower Yakima River to be a gaining reach from City of Prosser to below the Chandler Power Plant (Vaccaro, 2011), meaning streamflow increases as you move downstream. Above Benton City, the river transitions from a gaining reach to a predominantly losing reach, indicating a decrease in streamflow from the river. Possible sources of nutrients to the lower Yakima River include several overland irrigation return flows and irrigation wasteways on the lower Yakima River. Irrigation-fed subsurface groundwater also contributes cooler water and either returns through subsurface pathways in the floodplain or as tributaries from irrigation wasteways to the Yakima River.

The lower Yakima River hosts anadromous runs of steelhead (*Oncorhynchus mykiss*); spring-, summer-, and fall-run Chinook salmon (*Oncorhynchus tshawytscha*); coho salmon (*Oncorhynchus kisutch*); and sockeye salmon (*Oncorhynchus nerka*). Juvenile salmon out-migrate through the lower Yakima River to the Columbia River, and adult fish migrate from the Columbia River up into the lower Yakima River (Appel and others, 2011). Historically, the lower Yakima River hosted fall Chinook salmon spawning habitat. However, abundant water stargrass growth and excessive temperatures in the lower Yakima River has caused a shift of fall Chinook salmon spawning to above Prosser Dam beginning around 2005 (Appel and others, 2022; Mueller, 2010). As a result, adult and juveniles must migrate further, which conceptually could contribute to decreasing their chances of survival.

Methods

Continuous water quality and estimates of water stargrass biomass were collected at three locations for a period of 2.5 years during 2018–2020. In addition, the continuous water quality data were used to estimate whole stream metabolism to determine the state of macrophyte production in at these locations. A quality assurance project plan (QAPP) is available for this project and details the sampling plan, collection methods, study objectives, and quality assurance/quality control objectives for the study (Appel and Sheibley 2018).

Continuous Water Quality Collection

Continuous water quality data were collected at three locations on the lower Yakima River (**Figure 1**, **Table 1**) from April 2018 through October 2020: Yakima River at Prosser, Washington (Station 12509489), Yakima River at Kiona, Washington (Station 12510500), and Yakima River at Van Giesen Bridge near Richland, Washington (Station 12511800; U.S. Geological Survey, 2022). In this report, these three monitoring locations will typically be referred to simply as “Prosser”, “Kiona”, and “Van Giesen”. Water quality parameters measured at these three locations included temperature, pH, specific conductance, dissolved oxygen, and turbidity. EXO2 water quality sondes (Yellow Springs Instruments Inc.; YSI; Yellow Springs, Ohio) were deployed at Prosser and Kiona, and a YSI 6920 sonde was deployed at Van Giesen.

In addition, continuous nitrate data were collected at both Kiona and Van Giesen using a SUNAv2 (Satlantic, Inc.; Halifax, Nova Scotia, Canada). All continuous data were telemetered and available to the public on the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022). The Yakima River at Kiona, WA, is a long-term streamgage, and continuous streamflow is also available at this location. Water quality equipment was installed on June 26, 2018, at Prosser and Kiona and on August 9, 2018, at Van Giesen. For this project, data were collected at each monitoring station through October 31, 2020.

Table 1. Monitoring stations used during this study for continuous water quality and water stargrass (*Heteranthera dubia*) biomass assessments from April 2018 through October 2020 (U.S. Geological Survey, 2022).

[USGS, U.S. Geological Survey; No., number]

USGS station No.	USGS station Name	Short name	Latitude	Longitude
12509489	Yakima River at Prosser, WA	Prosser	46.2125	-119.7651
12510500	Yakima River at Kiona, WA	Kiona	46.2530	-119.4780
12511800	Yakima River at Van Giesen Br Near Richland, WA	Van Giesen	46.2972	-119.3334

All continuous water quality data collection followed standard USGS protocols for the installation, calibration, maintenance, and data reporting outlined in Wagner and others (2006) and Pellerin and others (2013). This includes periodic field visits to check the deployed water quality sensors with a laboratory-calibrated meter to determine accuracy and apply fouling and drift corrections as needed. In addition, annual cross-section measurements were used to assess how well-mixed the cross-section is and how representative the sensor location is of the entire cross-section. Prior to any data analysis, all continuous records were reviewed and approved according to USGS procedures for continuous records processing (Conn and others, 2017;

Wagner and others, 2006). All continuous water quality data are available through NWIS (U.S. Geological Survey, 2022).

Discrete Water Quality Data Collection

For continuous nitrate monitoring, physical samples for nitrate plus nitrite were collected to check nitrate sensor values, and corrections were applied as needed during the field deployment (Pellerin and others, 2013). Monthly water quality samples for field parameters (temperature, DO, pH, specific conductance) and total and dissolved nutrients were collected at both Prosser and Van Giesen throughout the duration of the project. Kiona (USGS station 12510500) is also part of the USGS National Water Quality Network. Several water quality parameters, suspended sediment, and nutrients are sampled at Kiona 18 times per year, and those data were used for this study.

Water quality data collection and data management followed USGS protocols (Conn and others, 2017; U.S. Geological Survey, 2018), to ensure appropriate equipment selection and cleaning, procedures for surface water sample collection, and field processing of samples for nutrients. Field replicates and blanks for total and dissolved nutrients were collected approximately quarterly during the 2-year project and were spread out among each monitoring station. An equipment blank sample was also collected to ensure that equipment used for data collection was clean and free from contamination. All temperature sensors were checked annually against a NIST (National Institute of Standards and Technology)-certified thermistor at five temperature points ranging from 0 to 40°C, and quarterly at two points (at 10 and 20° C) using a temperature-controlled water bath. All discrete water quality data are available in NWIS (U.S. Geological Survey, 2022).

Turbidity-Suspended Sediment relationship

As mentioned previously, Kiona is a long-term streamgage (USGS Station 12510500; Yakima River at Kiona, WA) for the USGS National Water Quality Network and is sampled for suspended sediment approximately 18 times per year. These data were combined with the continuous turbidity data from the Kiona station to develop a statistical relation between turbidity and suspended sediment. Following the procedures in Rasmussen and others (2009), continuous turbidity values during the time of suspended-sediment concentration (SSC) sample collection were paired, and a regression model was developed (Anderson and others, 2023). This model was used to compute monthly suspended sediment loads throughout the period of study.

Water Stargrass estimates

To assess the amount and distribution of water stargrass at each USGS station, biomass and percent cover were estimated up to three times during each growing season (June through August) in 2018 through 2020. At each monitoring station, up to 10 transects across the channel were established, and the presence-absence of water stargrass was determined at a spacing of about 2 meters across the transect. At Prosser and Van Giesen, water depth and/or river velocities were too high to safely wade across the channel, so personnel used a tagline at each transect to move across the channel while making presence-absence measurements with a mask and snorkel. At Kiona, personnel were able to wade across the channel and make presence-absence measurements. The presence-absence data were used to estimate the percent cover of the stream reach by dividing the total points where water stargrass was present, by the total number of locations surveyed multiplied by 100.

To estimate water stargrass biomass, 10 locations within each sampling reach were randomly chosen, and all above-ground biomass was harvested from a known unit area using a 0.0625 m^2 polyvinyl chloride (PVC) quadrat (**Figure 3**a). Once collected, each macrophyte sample was put into a plastic bag and sealed until returning to the laboratory (**Figure 3**b). If dry weights could not be determined within a few days, samples were frozen until further analysis. Prior to drying the samples, each plant was rinsed in a wash basin to remove non-plant materials, including any remaining roots and sediment. Each sample was dried in a laboratory oven at 60 °C for at least 48 hours. Samples were periodically removed, weighed, and reweighed a day later to ensure final dry weights were reached (± 0.5 grams). Dry weights were divided by the area of the quadrat to calculate a biomass value in units of grams of dry mass per meter squared [g/m^2].

A total of six sample events for water stargrass biomass and percent cover were completed during this study. In 2018, only one sample was collected (August); in 2019, samples were collected in June, August, and September; and in 2020, samples were collected in June and August. During September 2019, presence-absence measurements at Van Giesen and Kiona were not possible due to unsafe streamflow conditions. However, biomass samples were collected at all three sites for all six sample events.

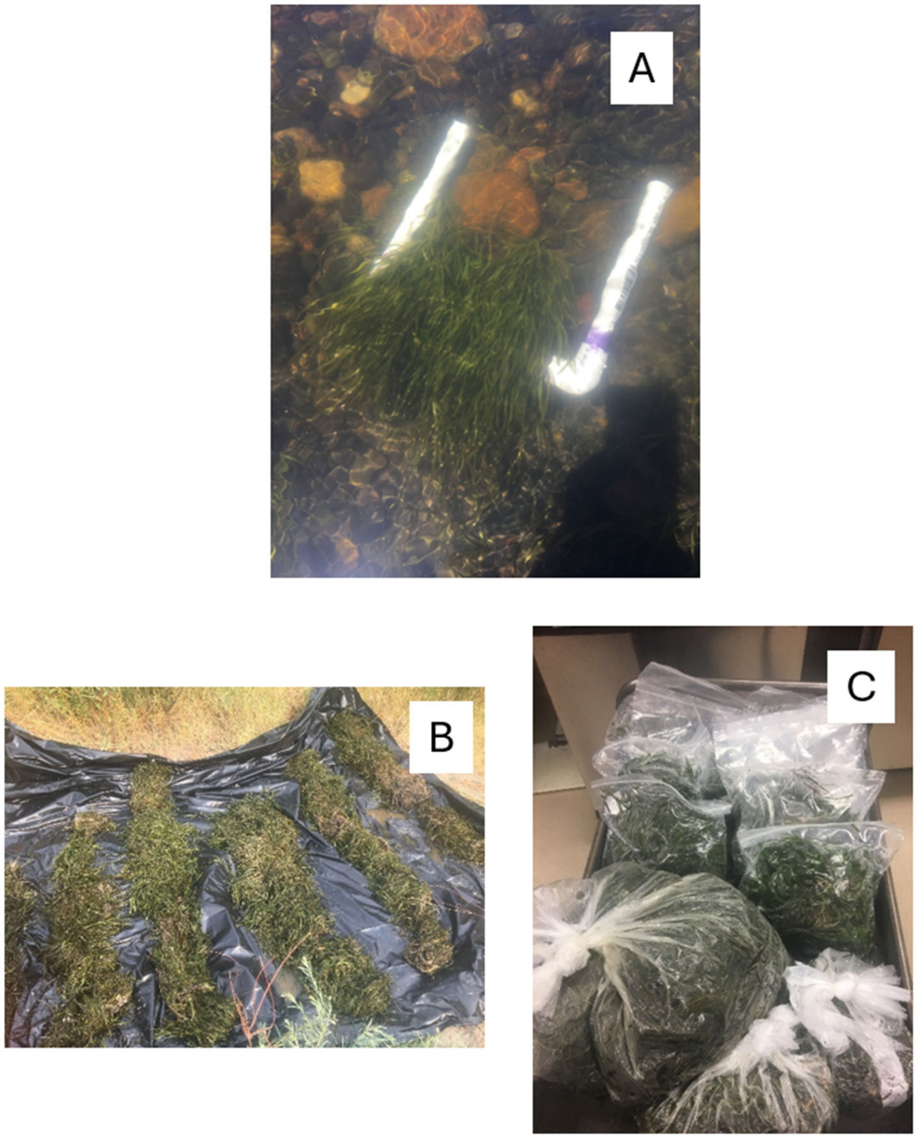

Figure 3. (a) Quadrat used to sample above ground biomass of water stargrass (*Heteranthera dubia*), (b and c) samples of water stargrass to be dried to determine dry weight per unit area (photographs by Rich Sheibley, U.S. Geological Survey, August 2018).

Estimates of Stream Metabolism

One objective of this study was to assess the relations among water quality and aquatic plant characteristics. Besides physical measurements of plant biomass, estimates of whole stream gross primary productivity (GPP) and ecosystem respiration (ER) were estimated at each sample station. The balance of GPP and ER of a river is a measure of the overall metabolism of the reach and growth rate of plants in a stream. During the day, primary productivity increases DO, and during nighttime, respiration reduces DO. Therefore, the balance of GPP and ER can impact DO levels in the river. When GPP is greater than ER, the system is considered autotrophic, when GPP is less than ER, it is considered heterotrophic. A stream that is autotrophic is dominated by primary productivity and can produce enough energy (carbon) that outside energy sources are not needed to support the system. Heterotrophic systems are dominated by respiration and require external inputs of organic matter for energy (such as leafy debris from riparian areas).

Stream metabolism was calculated using the streamMetabolizer package (version 0.12.0; Appling and others 2018) in R (version 4.2.0; R Core Team, 2022). This package uses the single-station method (Odum 1956, Hall and Hotchkiss 2017) to estimate stream metabolism represented by GPP and ER with inverse modeling. We used a three-parameter maximum-likelihood estimate approach from time-series data (15-minute intervals) of DO, stream depth, light, and water temperature. The maximum-likelihood-estimate model also estimates reaeration K600 values (Holtgrieve and others 2010, Grace and others 2015) needed for the model. The K600 value represents the air-water exchange constant that indicates how quickly oxygen moves between the water and atmosphere. Details on all stream-metabolism model runs, including model inputs and outputs, are provided in a companion data release (Sheibley and Foreman, 2024).

The continuous DO and temperature data for the metabolism models were from the deployed water quality sondes and used the final quality-controlled and cleaned-up data. The light data for the models was measured using Hobo S-LIA-M003 light sensors (Onset Computer Corp., Bourne, Massachusetts) established along the stream margin to collect continuous photosynthetically active radiation (PAR) at each station. Reach average channel depth was estimated by taking detailed measurements of water depth along an approximate 100-meter reach at each station at least once each summer. At Kiona, the water was shallow enough to allow personnel to take physical measurements of depth using a wading rod across established transects along the reach. At Prosser and Van Giesen, conditions were too deep or the current was too swift for physical depth measurements. At these stations, an inflatable Kayak was used to tow a Teledyne RD Instruments (Poway, California) StreamPro acoustic Doppler current profiler (ADCP) back and forth across the reach while continuously measuring river depths. After at least 10 passes across the river, the measurement was stopped and reach average channel depth was calculated using the instrument software. This information was only used to obtain the reach average channel depth measurements; velocity and discharge measurements were not recorded.

These reach-averaged channel measurements were related to a continuous measure of stream stage at each station to calculate a time series of reach-averaged channel depth for the metabolism models (Sheibley and Foreman, 2024). At Kiona, where streamflow was measured continuously, our summer reach-averaged depths were related to gage height at the streamgage. For Prosser and Van Giesen, paired non-vented pressure transducers (Hobo model U20-001-01, Onset Computer Corp.) deployed in the river and on land to collect barometric pressure allowed for calculation of sensor depth. The gage height and sensor depths were related to the reach average channel depth using an offset.

Daily estimates for GPP and ER were first plotted and examined for erroneous values such as negative values for GPP, positive values for ER, and obvious outliers. Occasionally, the model produces a large value for GPP or ER on a single day, immediately followed by a value which is similar to the surrounding daily estimates. Those points were considered outliers and were removed prior to any further analysis.

Data Analysis

Daily statistics were determined from the 15-minute continuous water quality data to examine daily minimum and maximum values as well as daily ranges and means over time and across locations. These daily statistics were compared to water quality reference conditions and state-established criteria.

The State of Washington has not established standards for nutrients and algal biomass. However, the U.S. Environmental Protection Agency (EPA), has published suggested reference conditions on a regional scale to protect water bodies from the negative effects of nutrient enrichment that were summarized in Wise and others (2009). The lower Yakima River is in the EPA Nutrient Ecoregion III (Xeric West), subgroup 10 (Columbia Plateau Ecoregion) (U.S. Environmental Protection Agency, 2000) and suggested nutrient reference conditions for Yakima River Basin streams in the Columbia Plateau Ecoregion are given in **Table 2**. The suggested reference conditions for streams in the Yakima River Basin were based on the 25th percentile of available nutrient concentrations, which were presumed to represent the least impacted conditions and intended to be protective of designated uses (U.S. Environmental Protection Agency, 2000).

Table 2. Reference conditions for nutrients suggested by the U.S. Environmental Protection Agency (2000) for streams in the Yakima River basin, Washington.

[mg/L, milligrams per liter; N, nitrogen; P, phosphorus; FNU, formazin nephelometric units; NTU nephelometric turbidity units]

Constituent	Units	Number of streams	Suggested reference conditions
Nitrate plus Nitrite as N	mg/L	71	0.072
Total N as N	mg/L	24	0.221
Total P as P	mg/L	127	0.030
Turbidity as FNU	FNU	41	2.03
Turbidity as NTU	NTU	69	1.45

Water Quality Standards for Surface Waters of the State of Washington, Chapter 173-201A WAC (Ecology, 2019) established beneficial uses of waters and incorporated specific numeric and narrative criteria for parameters such as water temperature, DO, pH, and turbidity. For the lower Yakima River, water temperature shall not exceed 21°C due to human activities except when natural conditions exceed 21°C, in which case temperature increases that will raise the receiving water temperature by more than 0.3 °C are not allowed. Under the Washington State water-quality standards, the lower Yakima River is in the category of salmonid spawning, rearing, and migration habitat. Based on this designation, the water-quality standards specify a lowest 1-day minimum dissolved oxygen concentration of 8.0 mg/L and pH within the range of 6.5 to 8.5 units, with a human-caused variation within this range of less than 0.5 pH units. Lastly, standards specify that turbidity should not exceed 5 nephelometric turbidity units (NTU) above background when the background is 50 NTU or less; if background is above 50 NTU, then increases shall not exceed 10 percent of the background value (Ecology, 2019).

Comparisons of water quality and biomass data were compared across sample events and monitoring stations using a non-parametric Kruskal-Wallis test for the analysis of variance across each case. These comparisons were combined with a *post hoc* Friedman's least significant difference test to test what categories were statistically different from each other (at a p-value of 0.05). Non-parametric linear correlations between water quality metrics, GPP, and water stargrass biomass were analyzed using Spearman's rho. We evaluated all these correlations using a 90 percent significance level ($p < 0.10$) due to the complex interactions of the system and the small dataset in this study. All statistical analysis was completed using R (version 4.2.0, R Core Team, 2022) RStudio (version 2022.02.3), and the "Hmisc" (Harrell Jr., 2022) and "Agricolae" (de Mendiburu, 2021) packages.

Analysis of Data Quality

When evaluating the data quality of a project, there are three aspects to consider: (1) quality assurance (QA) elements, (2) quality control (QC) data, and (3) overall quality assessment (Mueller and others, 2015). The QA elements refer to the procedures used to sample; for example, sampling the correct time and place, using established collection and analysis methods, and using the proper equipment for the samples being collected. The QC data are those generated from the collection and analysis of QC samples (blanks, replicates) used to assess the error and variability of collected samples. Lastly, quality assessment is the overall evaluation of data quality based on the QA and QC elements of the project.

Quality assurance was achieved for this project by following established USGS protocols for the preparation, collection, and processing of samples for water quality investigations published in the National Field Manual (U.S. Geological Survey, 2018). In addition, procedures in the QA plan of the Washington Water Science Center were followed (Conn and others, 2017). Conn and others (2017) outline details for project management, periodic project reviews, data management and archiving, and methods for collecting and analyzing water quality samples. Lastly, all field-collected samples were analyzed by the USGS National Water Quality Laboratory (NWQL) using standard methods for the analysis of nutrients (Fishman 1993; Patton and Kryskalla, 2011).

Quality control samples to assess possible contamination and bias and the variability during collection of field samples were measured using equipment blanks, field blanks, and field replicates. Blanks are samples prepared with water that are intended to be free of measurable concentrations of the analytes that will be analyzed by the laboratory. They are used to measure bias caused by contamination, the unintentional introduction of target analyte into the sample. Equipment blanks are prepared in a controlled laboratory situation to test just the equipment being used in the field for sample bias. Field blanks are prepared in the field and are used to assess possible contamination from sources through the whole process of sample collection, processing, shipping, and laboratory analysis. Both equipment and field blanks for this project used a laboratory-certified inorganic blank water sourced from the USGS NWQL. A total of 5 field blanks and 1 equipment blank for nutrients were collected for this project (**Table 3**), and not a single detection was recorded. The lack of detections in blanks provides a high degree of confidence that contamination is less than the reporting levels and that all parameter detections in environmental samples are free of contamination bias.

Table 3. Summary of field blank data for sampled constituents from June 2018 through September 2020 from the lower Yakima River, Washington. Data available from Sheibley and Foreman (2024).

[mg/L, milligrams per liter; N, nitrogen; P, phosphorus]

Constituent	Units	Reporting level	Number of blank samples	Number of quantified values	Percent of detections
Ammonia as N	mg/L	0.01	6	0	0
Nitrate plus Nitrite as N	mg/L	0.04	6	0	0
Nitrite as N	mg/L	0.001	6	0	0
Orthophosphate as P	mg/L	0.004	6	0	0
Total N as N	mg/L	0.05	4	0	0
Total P as P	mg/L	0.004	6	0	0

Six field replicates were also collected and analyzed for each parameter throughout the project. A field replicate is a set of two samples that are collected, processed, and analyzed such that they are considered to be the same sample and measure the variability of the whole sample collection and analysis life cycle. Field replicates collected for this project were sequential replicates which are collected one after another and therefore include sources of variability from sample collection and processing as well as any possible temporal change in the surface water environment between the samples (Mueller and others, 2015). A statistical evaluation of replicate variability was determined based on the standard deviation of replicate pairs. For each replicate pair, the standard deviation was determined and an average value for each parameter over the duration of the project was calculated. This average standard deviation was then used to determine an upper and lower confidence limit based on the methods described by Mueller and others (2015). The 90th percentile confidence interval for all parameters was low and much less than the replicate concentrations (**Table 4**).

Table 4. Summary of field replicate data for sampled constituents from June 2018 through September 2020 from the lower Yakima River, Washington. Data available in Sheibley and Foreman (2024).

[mg/L, milligrams per liter; ug/L, micrograms per liter; %, percent]

	Units	Number of replicates	Range of replicate concertation	90th percentile confidence interval	Range of relative percent difference
Ammonia as N	mg/L	6	<0.010 to 0.050	± 0.009	0.0 to 95%
Nitrate plus Nitrite as N	mg/L	6	0.455 to 1.200	± 0.011	0.0 to 2.3%
Nitrite as N	mg/L	6	0.007 to 0.020	± 0.0001	0.0 to 15.0%
Orthophosphate as P	mg/L	6	0.064 to 0.119	± 0.001	0.0 to 3.4%
Total N	mg/L	3	1.0 to 1.34	± 0.020	0.7 to 3.0%
Total P	mg/L	6	0.079 to 0.215	± 0.001	0.0 to 3.4%

In addition, the relative percent difference between all replicate pairs was less than 5 percent, except for a single replicate pair for ammonia and nitrite. In this case, the concentrations were close to the detection limits, so small changes in replicate values can lead to high relative percent differences. For that replicate pair, the absolute difference between the replicates was 0.006 mg-N/L for both ammonia and nitrite. Taken together, the relative percent difference and 90th percentile confidence intervals indicate an overall low amount of variability in the field data collected for this study.

Laboratory QC was assessed internally at the NWQL using laboratory duplicates, reference materials, and matrix spike duplicates. All NWQL QC results were within the measurement quality objectives defined in the project QAPP (Appel and Sheibley, 2018), indicating that laboratory data were accurate and that measurement bias was low.

Quality control data for continuous data collection (DO, pH, temperature, specific conductance, and turbidity) were described previously. This included frequent calibration and verification checks of all sensors, periodic cross-section checks, and frequent cleaning of instruments. PAR and water level sensors used to calculate stream metabolism were verified prior to use in the field. The PAR sensor was calibrated by the manufacturer prior to use. Level loggers were tested at 3 different depths in the USGS laboratory in Tacoma, WA, using a calibrated water tank. Any level logger that was not within 0.05 feet of the expected water tank depth was not used in this study.

Results

Hydrologic Conditions from 2018 to 2020

Spring through summer streamflow conditions on the lower Yakima River varied across the three water years during which sampling occurred (**Figure 4**). Spring runoff (March through May) was much greater in water year 2018 compared to 2019 and 2020 and the average of recent historical spring streamflows (2000-2017). Streamflow in water year 2019 was similar to streamflow in 2018 from March to early May. Streamflows from May through June 2019 were much lower than in 2018 and were similar to conditions in 2020. The decline in spring streamflow in 2019 was quicker and sustained at lower levels longer than in 2018 and 2020. Water year 2020 showed the lowest spring runoff across the whole study period. Across all three years, streamflow in summer (June through September), was lower compared to average of near recent historical conditions in 2000 to 2017.

To quantify the differences in streamflow during the 2018 through 2020 water years, two simple streamflow metrics were calculated from the streamflow record at the Kiona streamgage (**Table 5**). These metrics included the instantaneous peak discharge and average discharge during the spring runoff period (March 1 to May 31) for each water year.

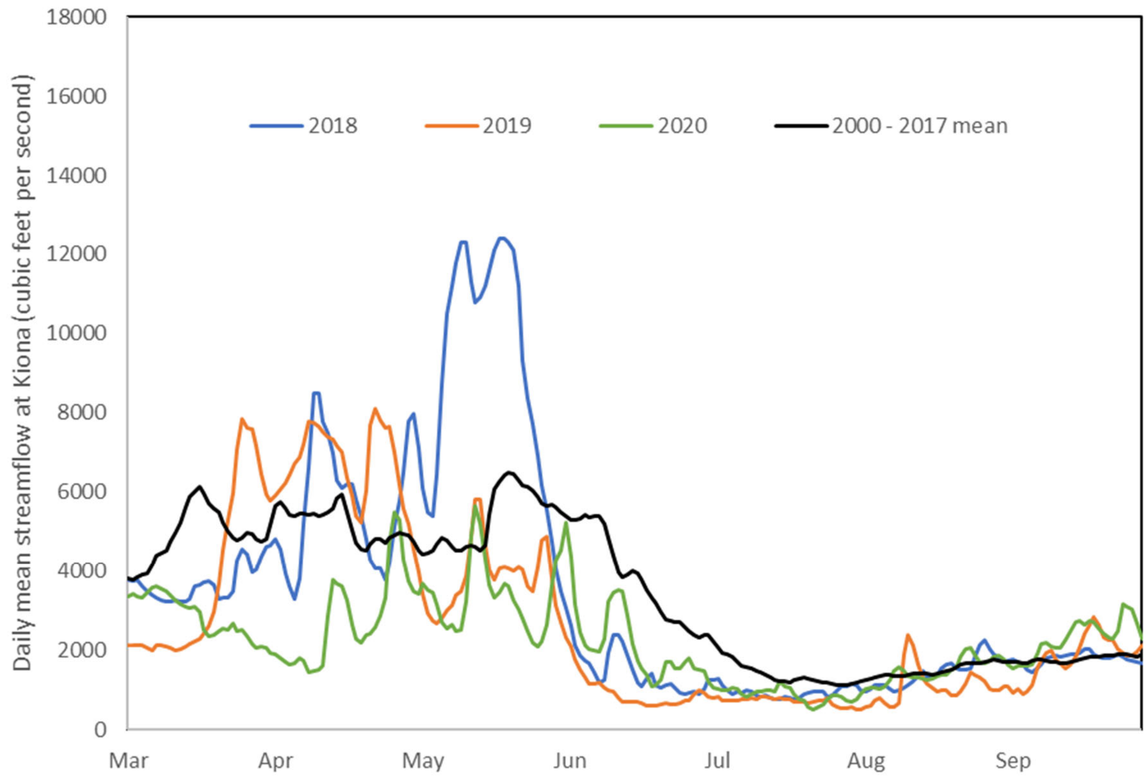

Figure 4. Streamflow from March through September for water years 2018 through 2020 compared to mean streamflow for water years 2000–2017 for the Yakima River at Kiona, Washington (USGS station 12510500; U.S. Geological Survey, 2022).

Table 5. Streamflow metrics at the Yakima River at Kiona, Washington streamgage (USGS Station 12510500; U.S. Geological Survey, 2022) computed from streamflow records from March 1 to May 31 for water years 2018-2020.

[ft^3/s, cubic feet per second; WY, water year]

Streamflow metric	Units	WY2018	WY2019	WY2020
Instantaneous spring peak discharge	ft^3/s	12,500	8,490	5,730
Average spring discharge	ft^3/s	6,150	4,750	2,995

The focus on the spring runoff period was chosen because it was different for each water year and because we hypothesized that spring streamflow has an impact on how much water stargrass will grow in the summer. Both peak spring discharge and average spring discharge decrease each water year from 2018 to 2020. These streamflow metrics illustrate the difference in spring streamflow regime despite each summer having a comparable streamflow record.

Continuous Water Quality Data

Water quality data collection began on June 26, 2018, at Prosser and Kiona and on August 9, 2018, at Van Giesen. Data collection at all three sites ended on October 31, 2020. Unfortunately, there were several data gaps due to equipment malfunctions, faulty sensors, and during high streamflows when sensors were buried by sediment (**Table 6**). In addition, fieldwork was substantially more difficult in 2020 due to the COVID-19 pandemic and extreme heat and fire conditions in the summer on the lower Yakima River. However, every effort was made to minimize the periods of data loss by conducting frequent site visits. The most substantial data losses occurred at Van Giesen, when DO data in both summer 2019 and 2020 were lost. This was a key part of the year for the DO data, the loss of which impacted our ability to determine relationships between DO and water stargrass.

Table 6. Summary of data gaps in continuous water quality records for all stations from June 2018 to October 2020 (U.S. Geological Survey, 2022).

[--, no data collected; start date for data collection at Prosser and Kiona was 6/26/2018 and for Van Giesen was 8/9/2018; dates provided in format of MM/DD/YY: month/day/year]

Parameter	Prosser	Kiona	Van Giesen
Temperature	6/19/20 to 7/21/20	10/14/19 to 11/5/19	11/27/18 to 12/10/18 12/29/18 to 1/16/19 2/10/19 to 2/11/19
Dissolved Oxygen	6/19/20 to 7/21/20	10/14/19 to 11/5/19 2/10/20 to 3/5/20	11/27/18 to 12/10/18 12/27/18 to 1/16/19 6/11/19 to 9/18/19 3/18/20 to 10/31/20
Specific Conductance	6/19/20 to 7/21/20	10/14/19 to 11/5/19	11/28/18 to 12/10/18 12/31/18 to 1/16/19
pH	4/13/19 to 6/26/19 6/19/20 to 7/21/20	10/14/19 to 11/5/19 3/3/20 to 4/9/20	11/16/18 to 4/3/19 5/14/19 to 5/16/19 6/19/19 to 6/24/19 12/16/19 1/8/20
Turbidity	6/19/20 to 7/21/20	7/10/19 to 8/8/19 10/14/19 to 11/5/19 2/15/20 to 2/16/20 3/6/20 7/2/20 to 8/22/20	11/27/18 to 12/20/18 12/26/18 to 1/16/19 12/16/19 1/8/20 to 1/9/20
Nitrate	--	--	11/27/18 to 12/10/18 1/1/19 to 1/16/19 12/20/19 to 10/31/20

Water Temperature

Observed water temperature data at all three locations for 2018 – 2020 are typical of current river conditions for the lower Yakima River. Maximum daily temperatures in summer were high and exceeded the 21 °C criteria almost every day from June through August each year; winter temperatures were close to 0 °C (**Figure 5**). The daily maximum measured temperatures at Prosser are slightly cooler during the summer months compared to daily maximum temperatures at Kiona and Van Giesen (**Figure 5**). Differences in maximum daily temperatures between monitoring stations show downstream warming from Prosser to Kiona throughout the study period except for a few months in winter (**Figure 6**). Downstream temperature increases in summer between Kiona and Prosser ranged from about 1 º to 3 º C and were highest in 2018 and lowest in 2020 (**Figure 6**). Conversely, the differences between maximum daily temperatures between Van Giesen and Kiona were low throughout the study, often less than 1 ºC (**Figure 6**). There is evidence of warming between Kiona and Van Giesen during summer months but cooling during other times of year. Interestingly, there were a few months in summer 2019 where the river cooled between Kiona and Van Giesen (**Figure 6**). Taken together, summer warming in the lower Yakima River was greater from Prosser-to-Kiona than from Kiona-to-Van Giesen.

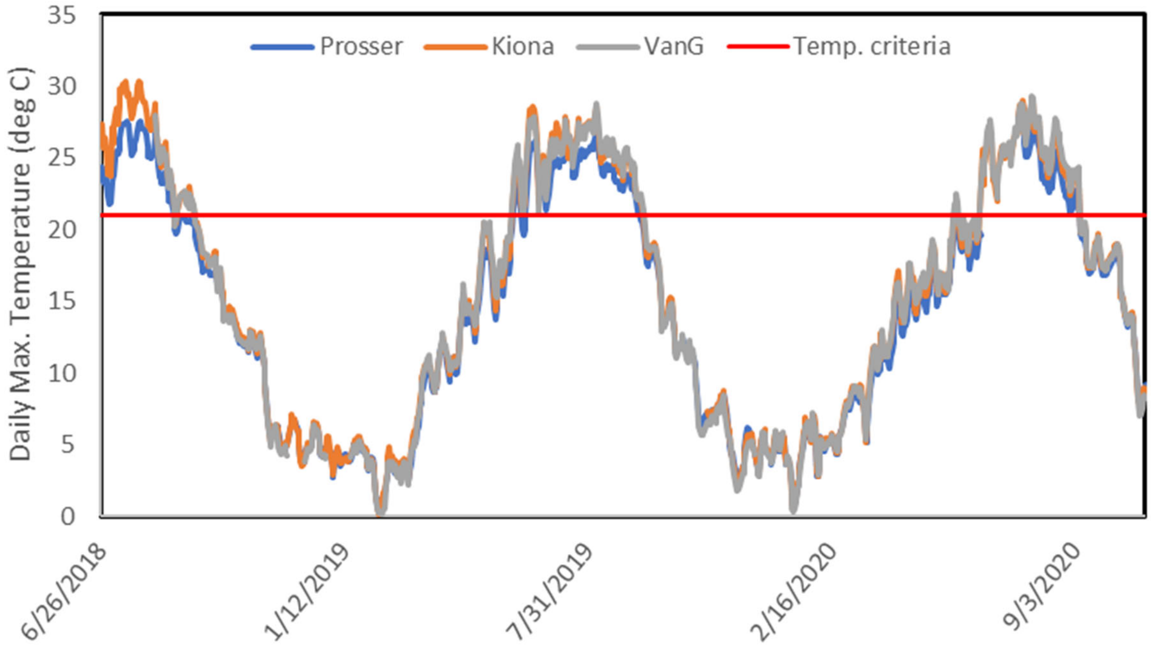

Figure 5. Daily maximum temperatures at Prosser, Kiona, and Van Giesen in the lower Yakima River. The red line indicates the water quality criteria for the lower Yakima River that maximum daily temperatures should not exceed 21° C (Ecology, 2019). Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

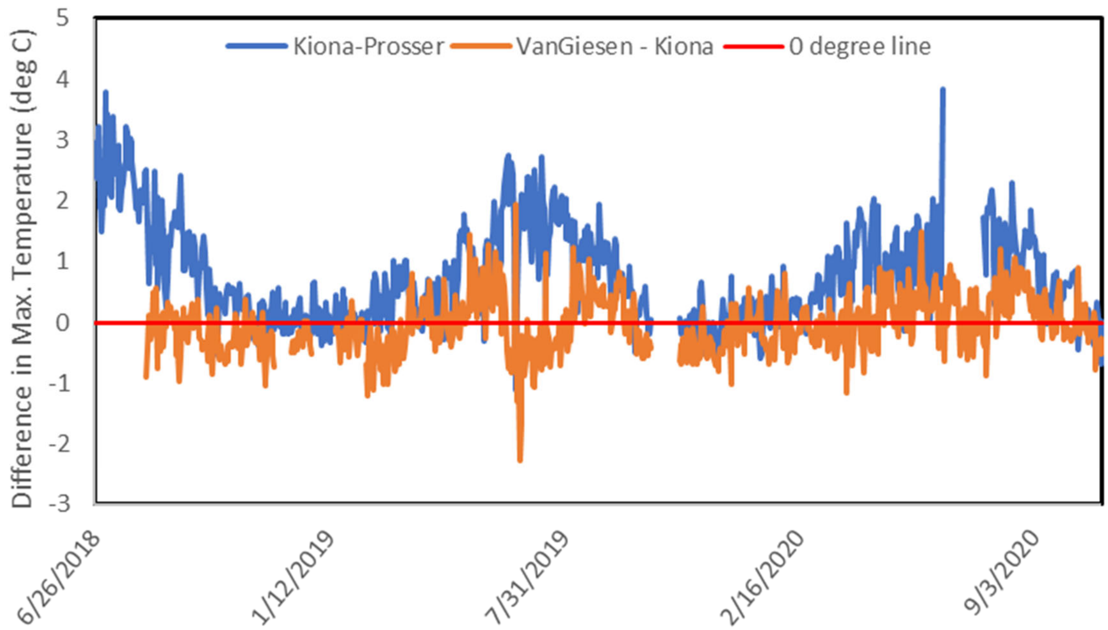

Figure 6. Difference in daily maximum water temperatures between Prosser and Kiona monitoring locations, and between Kiona and Van Giesen monitoring locations. Positive values indicate downstream warming, negative values downstream cooling. Red line represents a 0-degree difference between the stations. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

The number of days when the maximum daily water temperature exceeded the 21°C criteria were similar across all sites for water years 2018 through 2020 (**Table 7**). The drought year in 2019 had the most days in exceedance of the three summers monitored. Even though average ambient air temperatures were higher in 2020, 2019 had lower sustained river baseflows than observed in 2020 (**Figure 4**).

Table 7. Summary of maximum daily temperature exceedances from 2018 through 2020 at the three monitoring station locations in the lower Yakima River: Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

[WY, water year; No., number]

Site	WY 2018*		WY 2019		WY2020	
	No. days exceeding 21 °C	No. of days monitored	No. days exceeding 21 °C	No. of days monitored	No. days exceeding 21 °C	No. of days monitored
Prosser	60	97	101	365	48	333
Kiona	75	97	108	365	86	343
Van Giesen	31	53	109	330	87	363

*WY2018 did not include a full year of sampling and Van Giesen. Sampling at this site began about seven weeks after the other two sites

Dissolved Oxygen

The daily DO minimum at all three monitoring sites was below the 1-day minimum of 8 mg/L at various times during the 2018–2020 monitoring period (**Figure 7**a, **Table 8**); annual minimums occurred during the summer baseflow conditions. During summer, daily DO minimums were higher at Prosser compared to Kiona. The daily range of DO, the difference between daily maximum and minimum values, is greatest in the late spring and summertime and smallest in winter (**Figure 7**b). Prosser had the smallest daily DO range during spring and summer throughout the study, and daily DO ranges at Kiona and Van Giesen were similar in 2018 when both stations were collecting DO data (**Figure 7**b). Much of the summer DO data was lost at Van Giesen in 2019 and 2020 due to sensor malfunctions that limited our ability to compare water stargrass biomass to DO at that location.

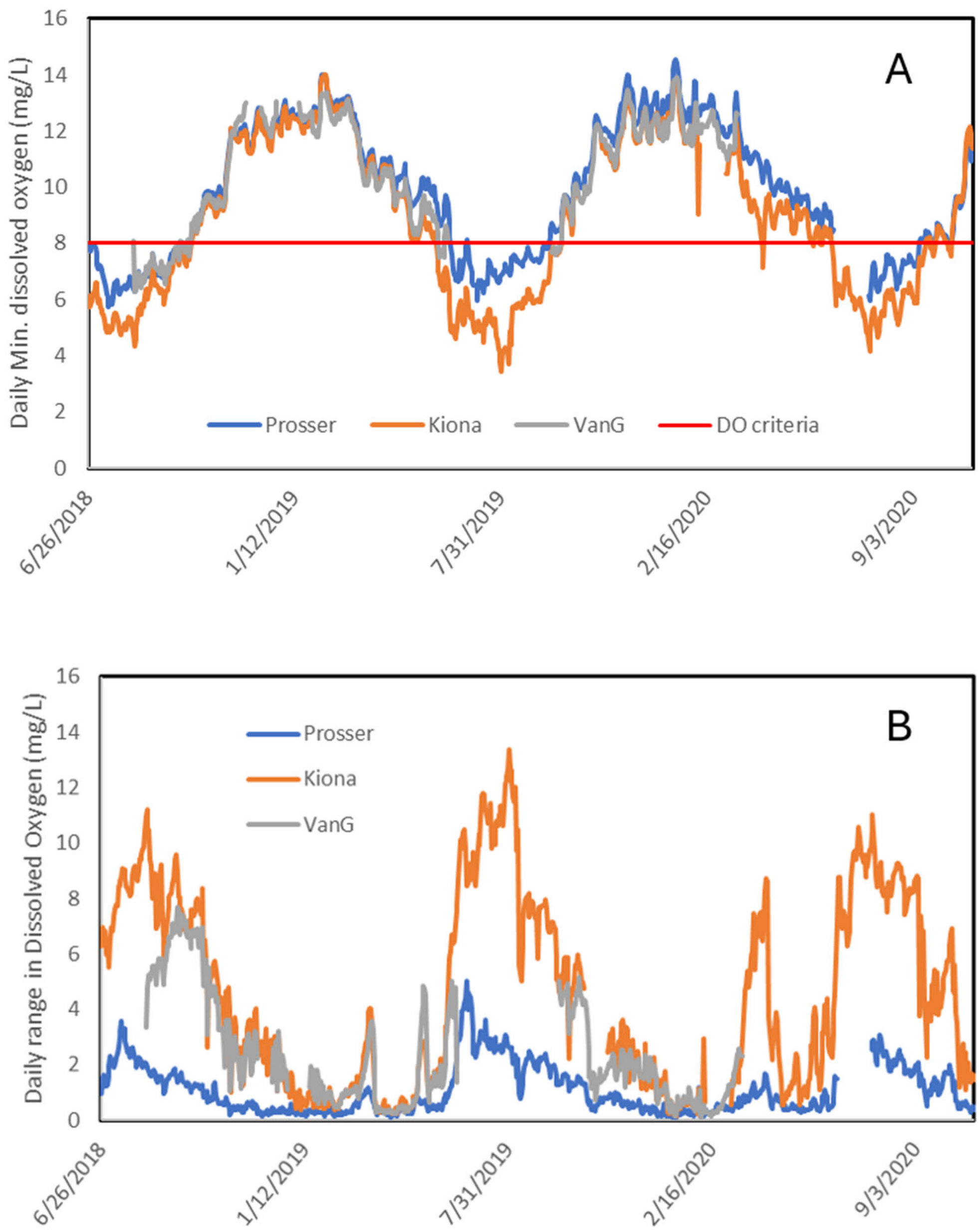

Figure 7. Dissolved oxygen daily minimum (A) and daily range (B), calculated as the daily maximum minus daily minimum at Prosser, Kiona, and Van Giesen from 2018 to 2020. The red line in the upper panel represents the minimum dissolved oxygen criteria for the lower Yakima River (8 mg/L) (Ecology, 2019). Sustained DO levels below that threshold may be fatal to salmon. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

Table 8. Summary of minimum daily dissolved oxygen exceedances during 2018 through 2020. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

[WY, water year; No., number; ND, no data, sensor failed]

Site	WY 2018		WY 2019		WY2020	
	No. days less than 8 mg/L	No. of days monitored	No. days less than 8 mg/L	No. of days monitored	No. days less than 8 mg/L	No. of days monitored
Prosser	97	97	97	365	52	333
Kiona	97	97	124	365	101	318
Van Giesen	49*	53*	16*	230*	0*	169*

*Van Giesen was started about 7 weeks later in WY2018 and no data was collected in summer in WY2019 or WY2020

pH

The lower Yakima River aquatic use designation pH levels must fall within the daily range of 6.5 to 8.5 pH units (Ecology, 2019). Daily maximum pH frequently exceeded 8.5 at Kiona and Van Giesen throughout the summer months, with additional short-term exceedances in winter months (**Figure 8**a, **Table 9**). Prosser only exceeded the maximum pH criteria of 8.5 a few times throughout the study (**Figure 8**a, **Table 9**). In contrast, at no time during the study did pH values fall below the established criteria for daily minimum pH of 6.5 units at any of the monitoring locations (**Figure 8**b, **Table 9**).

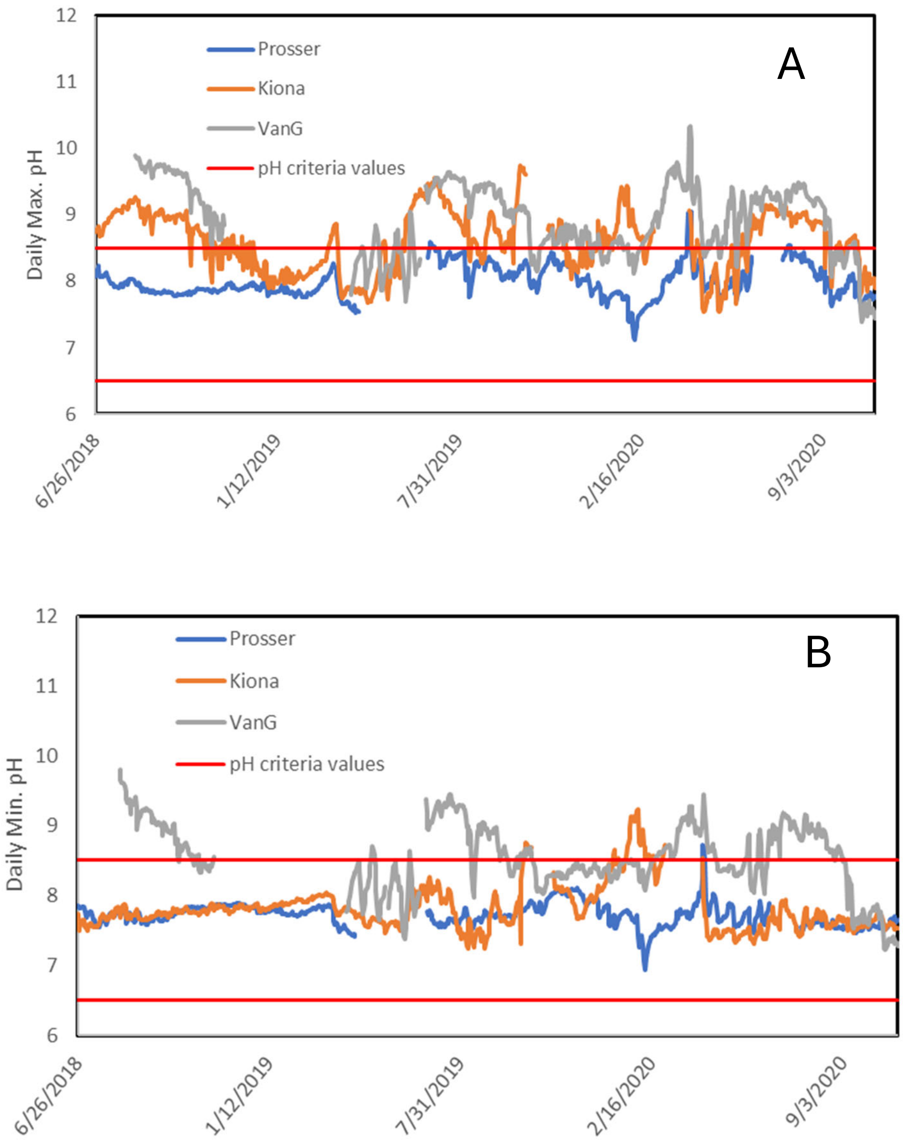

Figure 8. Daily maximum (a) and daily minimum (b) for pH measured at Prosser, Kiona, and Van Giesen from 2018 to 2020. The red lines represent the pH criteria for the lower Yakima River, 6.5 and 8.5, respectively (ecology, 2019). Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

Table 9. Summary of minimum and maximum daily pH exceedances of pH criteria for the lower Yakima River (Ecology, 2019) during 2018 through 2020. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

[WY, water year; No., number; ND, no data]

Site	WY 2018*			WY 2019			WY2020		
	No. days exceeding 8.5	No. days less than 6.5	No. of days monitored	No. days exceeding 8.5	No. days less than 6.5	No. of days monitored	No. days exceeding 8.5	No. days less than 6.5	No. of days monitored
Prosser	0	0	97	7	0	290	8	0	333
Kiona	97	0	97	168	0	365	202	0	305
Van Giesen	53	0	53	164	0	218	293	0	364

*Van Giesen was started about 7 weeks later in WY2018

Similar to daily DO ranges, daily pH ranges were greatest in summer and smallest in winter months (**Figure 9**). In general, of all three sites, the greatest daily pH ranges were observed at Kiona (**Figure 9**).

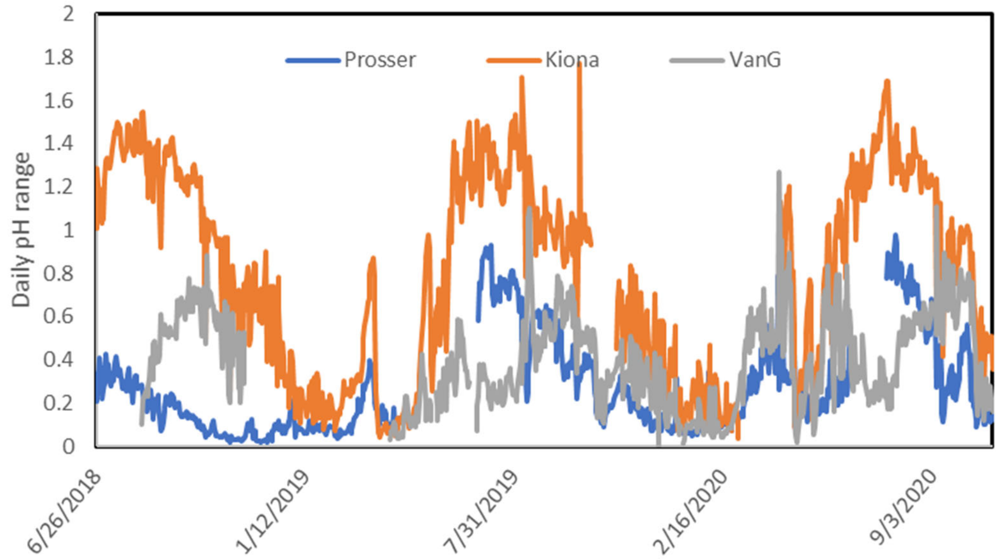

Figure 9. Daily range for pH measured at Prosser, Kiona, and Van Giesen from 2018 to 2020. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

Specific Conductance

Daily mean specific conductance was less than 350 microsiemens per centimeter at 25 ºC (µS/cm), and these values were similar throughout the study across all three monitoring locations (**Figure 10**). Specific conductance showed an inverse relationship with streamflow measured at the Kiona streamgage. When streamflow increased at Kiona, specific conductance decreased, and as streamflow decreased, specific conductance increased (**Figure 10**). The observed patterns indicate that increased streamflows tend to dilute the surface water with respect to major ions.

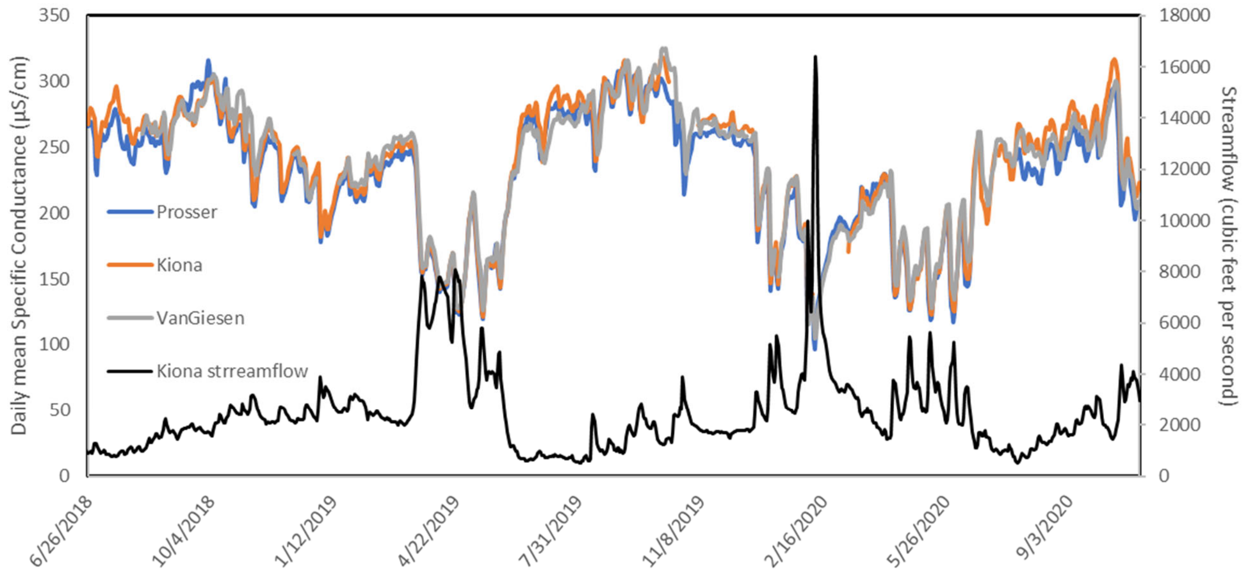

Figure 10. Daily mean specific conductance measured at Prosser, Kiona, and Van Giesen from 2018 to 2020. Streamflow discharge at Kiona is shown by the black line. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

Turbidity

Turbidity was relatively low across all three monitoring locations with average turbidity during the whole study period of 5 Formazin Nephelometric Units (FNU) at Prosser, 9 FNU at Kiona, and 1 FNU at Van Giesen (**Table 10**). Maximum daily turbidity values followed seasonal patterns with highest values during storm events from fall through spring and reaching background levels in summer low streamflow months (**Figure 11**). Background turbidities were determined by taking the average turbidity during summer baseflow conditions and were 1 FNU for Prosser, 3 FNU for Kiona, and 1 FNU for Van Giesen (**Table 10**). The background turbidity values at Prosser and Van Giesen were lower than the recommended value for reference conditions established by the EPA (**Table 2**), whereas background turbidity at Kiona was only slightly greater than this value.

The water quality criteria for turbidity in the lower Yakima River is no more than 5 NTU over background when background is 50 NTU or less (Ecology 2019). However, for this study, we measured turbidity in Formazin Nephelometric Units (FNU) instead of NTU; FNU is the most common approach used by the USGS. Turbidity measurements in ambient waters using the FNU designation generally indicate the measurements followed ISO 7027, which is an international standard for measuring turbidity (Anderson, 2005). However, there is no way to directly convert between NTU and FNU because these units use different methods and are the result of inherent properties of the wavelength of the light source used to measure the light scatter. However, both units are nephelometric measures because both use light scattered at 90 degrees from the incident light beam. While there is no simple conversion between NTUs and FNUs that holds across different water types, these different turbidity measures are correlated (Davies-Colley and Smith, 2001). For this study, the assumption that the measured turbidity in FNU is comparable to NTU is reasonable because of how low turbidity was during the study period.

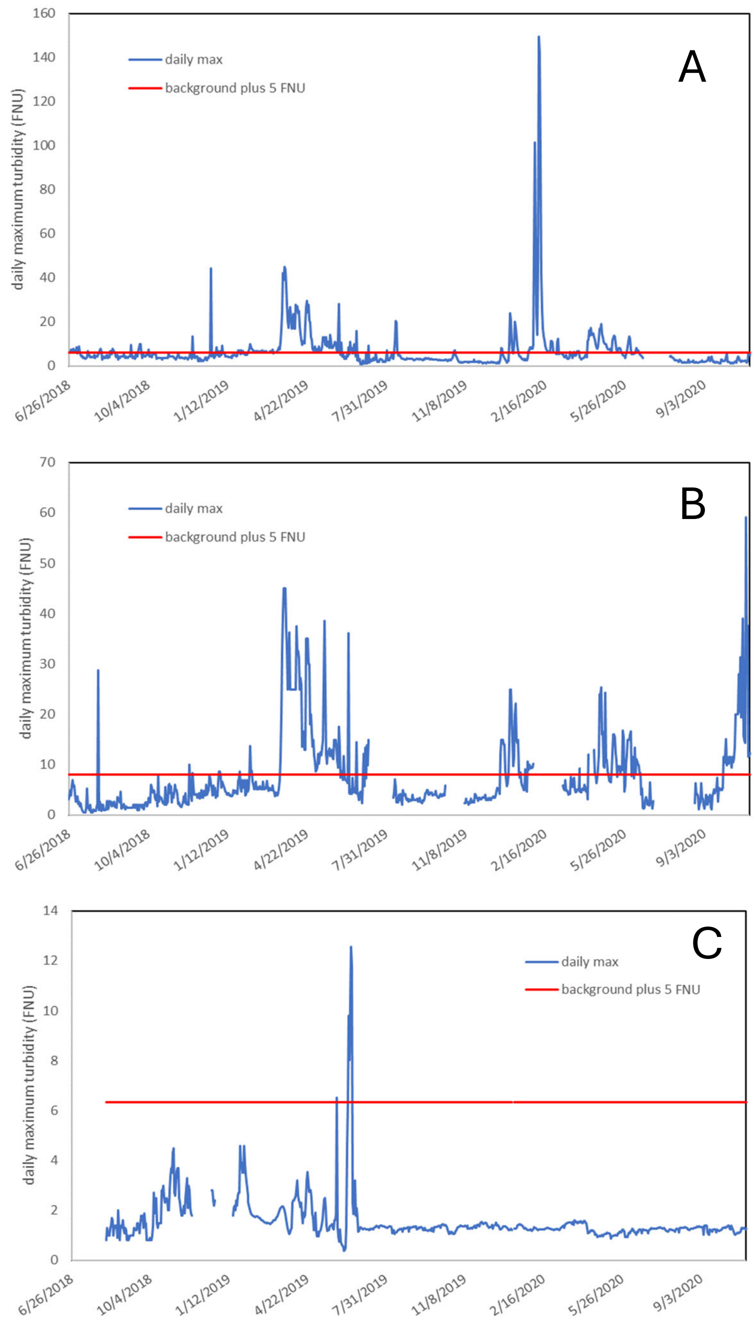

Figure 11. Maximum daily turbidity at (a) Prosser, (b) Kiona, and (c) Van Giesen. The red lines refer to the background turbidity plus 5 FNU which defines the site-specific water quality criteria (Ecology, 2019. Note: the range in y-axis is not the same across the three panels. Additional site information is provided in table 1.

Table 10. Summary of turbidity conditions in the lower Yakima River, from 2018 to 2020. Water Quality criteria in the lower Yakima River are based on background turbidity plus 5 FNU (Ecology, 2019). Additional site information is provided in table 1.

[FNU, Formazin Nephelometric Units; >, less than]

Site	Overall mean (FNU)	Background (FNU)	Background plus 5 FNU	Percent of days the daily mean was above background plus 5 FNU
Prosser	5	1	6	33
Kiona	9	3	8	28
Van Giesen	1	1	6	< 1

Results from turbidity monitoring in the lower Yakima River indicated that there were days when maximum turbidity exceeded 5 FNU over background at all three sites, but predominantly, the FNU values were well below this criteria during the monitoring period (**Figure 11**). Temporary exceedances of the '5 FNU over background' threshold used in this report were observed; however, they were usually associated with high streamflows, and increases were transient with levels dropping quickly back to background levels. The percent of days the daily mean was above background plus 5 FNU during the entire study period was 33 for Prosser, 28 for Kiona, and less than 1 for Van Giesen (**Table 10**).

Turbidity-suspended sediment relationship

From June 2018 through October 2020, 31 discrete suspended sediment concentration (SSC) samplers were collected at Kiona as part of the USGS National Water Quality Program. Anderson and others (2023) used these data combined with additional SSC samples collected in 2021 and 2022 to develop models to estimate SSC concentrations at 15-minute intervals throughout the study period. In total, 68 SSC samples were collected from June 2018 through

September 2022, of which 47 were paired with concurrent turbidity measurements. Although these models included data from 2021 and 2022, which were outside the window of the study period in this report, they still represent the best available information and helped improve the model accuracy. Two models were used to create the SSC record: one based on turbidity and the other based on discharge when turbidity data were not available. The best fit regression for the SSC-turbidity model used a segmented linear approach on untransformed data and had an adjusted R^2 value of the fit of 0.88 (Anderson and others, 2023). The best regression model for the SSC-discharge model used a three-part segmented linear approach on transformed data, with an adjusted R^2 value of 0.71 (Anderson and others, 2023). Both models are summarized using the following equations:

SSC-Turbidity

$$SSC = \begin{cases} -0.645 \times Turb + 3.6, & Turb < 2.8 \\ 3.37 \times Turb - 7.8, & Turb > 2.8 \end{cases} \tag{1}$$

SSC-Discharge

$$SSC = \begin{cases} 0.198 \times Q^{0.42}, & Q > 2570 \\ 6.98 \times 10^{-11} \times Q^{3.19}, & 2570 < Q < 3960 \\ 2.31 \times 10^{-5} \times Q^{1.66}, & 3960 < Q \end{cases} \tag{2}$$

where

SSC is the estimated suspended sediment concentration in mg/L

Turb is the sensor turbidity in units of FNU

Q is the discharge in cubic feet per second, ft^3/s

Monthly suspended sediment loads for the study period were variable and followed seasonal patterns (**Table 11**). During summer months, suspended sediment loads were the lowest, corresponding to low summer streamflow. During winter through spring, when Kiona streamflow was elevated, the sediment loads were greater.

Table 11. Monthly suspended sediment loads in the lower Yakima River at Kiona (USGS Station 12510500; U.S. Geological Survey, 2022), June 2018 to September 2020. Data are available from Anderson and others (2023).

[WY, water year; --, no data available]

	Monthly total (tons)		
Month	**WY2018**	**WY2019**	**WY2020**
October	--	540	1500
November	--	710	470
December	--	800	2100
January	--	1800	7900
February	--	1500	79600
March	--	16700	2600
April	--	31000	6600
May	--	9600	6500
June	780	860	2500
July	270	310	360
August	380	390	550
September	450	530	930

Nitrate

The seasonal patterns of continuous nitrate measured at Kiona and Van Giesen were similar. In general, nitrate concentrations followed patterns in streamflow with a few distinctive periods observed in the data (**Figure 12**). First, during fall and winter months, nitrate concentrations were the highest and stayed high ranging from 1.2 to 1.8 milligrams of nitrogen per liter (mg-N/L). Second, as spring streamflows increased, nitrate concentrations decreased. Third, as summer months progress, there was a gradual increase in concentrations until fall storms began. In addition to these seasonal patterns, nitrate concentrations at Van Giesen tended to be slightly higher than Kiona during fall and winter and slightly lower from spring through summer.

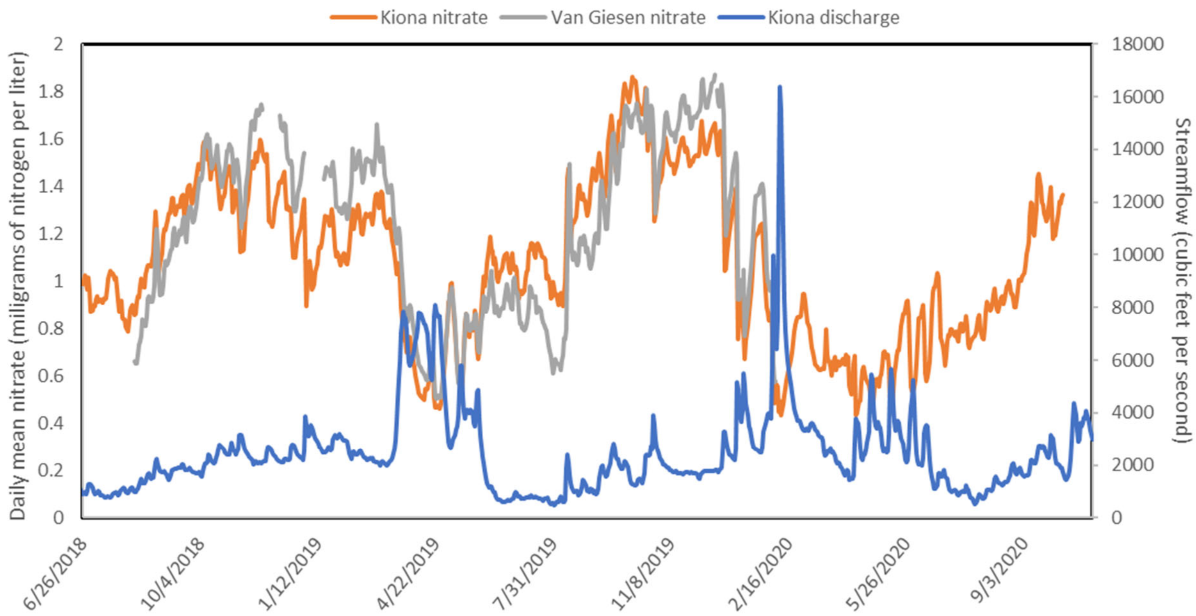

Figure 12. Daily mean nitrate concentration from Kiona and Van Giesen during the study. The blue line shows streamflow at Kiona for reference (U.S. Geological Survey, 2022). Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

Streamflow at Kiona was used to examine the relationship between nitrate concentration and streamflow by computing the mean daily nitrate load at Kiona for the study period. The mean daily load was determined by multiplying the mean streamflow and the mean nitrate concentration for each day. In water year 2019, nitrate load was relatively constant, showing peaks during fall and winter with increases in streamflow (**Figure 13**a). Nitrate load was highest in spring during peaks in streamflow and decreased to a baseline nitrate load in summer when streamflows were more stable (**Figure 13**a). This period of lowest nitrate load lasts until around the end of July when the nitrate load increases through the rest of the water year despite the lack of concomitant streamflow increases. Nitrate load in water year 2020 followed similar patterns to loads in 2019. The fall and winter loads showed increases in nitrate loads with storm-related peak streamflows but was more variable in 2020 than in 2019 (**Figure 13**b). However, as in 2019, the nitrate load in 2020 increased from the end of July through the end of the water year, despite sustained lower streamflow conditions (**Figure 13**b).

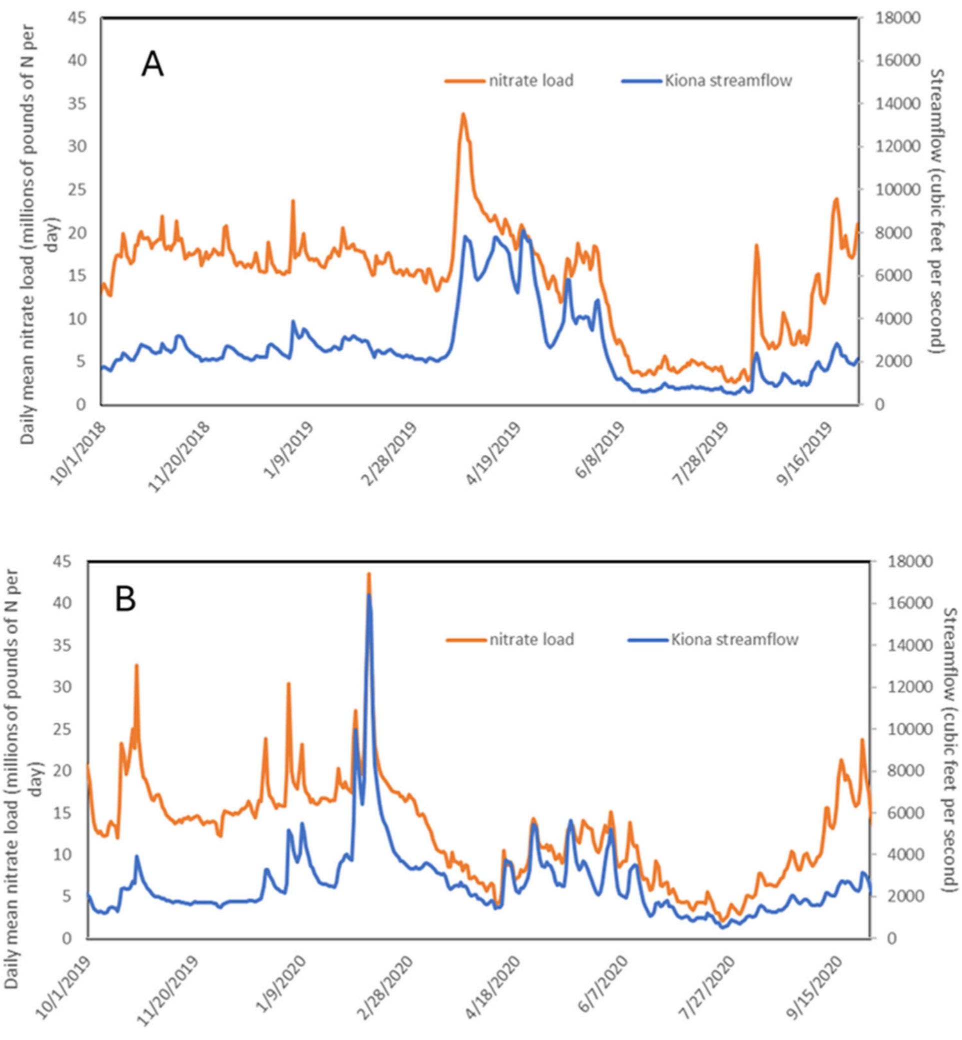

Figure 13. Daily mean nitrate load at Kiona for (a) water year 2019 and (b) water year 2020 (U.S. Geological Survey, 2022). Additional site information is provided in table 1. Data for USGS stations are

available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

Discrete water quality data

Discrete water quality samples were collected for nutrients approximately monthly at Prosser and Van Giesen, and more frequently at Kiona. Kiona is a long-term site for the USGS National Water Quality Program and is sampled 18 times per year, and samples are analyzed for several parameters. Overall, differences in key nutrient parameters were minimal across the three monitoring locations (**Table 12**). Ammonia was low and not detected in most samples analyzed. Nitrate plus nitrite ranged from 0.26 to 1.87 mg-N/L across all samples. This range was much higher than the EPA suggested reference conditions for nitrate plus nitrite (0.072 mg-N/L, **Table 2**). Orthophosphate ranged from 0.03 to 0.12 milligrams of phosphorus per liter (mg-P/L) across all samples. Total phosphorus ranged from 0.06 to 0.22 mg-P/L and was higher than the EPA suggested reference condition of 0.03 mg-P/L (**Table 2**). Total nitrogen ranged from 0.55 to 2.43 mg-N/L across all samples and was higher than the EPA suggested reference conditions of 0.221 mg-N/L (**Table 2**).

Table 12. Summary of nutrient data from the lower Yakima River, 2018 to 2020. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

[N, nitrogen; P, phosphorus; mg/L, milligrams per liter; No., number; >, less than]

Parameter	Units	Prosser			Kiona			Van Giesen		
		No. of samples	Mean	Range	No. of samples	Mean	Range	No. of samples	Mean	Range
Ammonia as N	mg/L	24	0.01	<0.01 to 0.04	45	0.01	<0.01 to 0.04	24	0.01	<0.01 to 0.04
Nitrate + nitrite as N	mg/L	24	1.17	0.43 to 1.86	45	0.95	0.26 to 1.79	24	1.07	0.45 to 1.87
Orthophosphate as P	mg/L	24	0.07	0.05 to 0.12	45	0.06	0.03 to 0.10	24	0.07	0.03 to 0.12
Total Phosphorus as P	mg/L	24	0.12	0.06 to 0.18	45	0.10	0.06 to 0.22	24	0.10	0.06 to 0.14
Total Nitrogen as N	mg/L	24	1.40	0.70 to 2.43	45	1.21	0.55 to 2.05	24	1.32	0.73 to 2.07

Water stargrass estimates

Estimates of water stargrass biomass were made throughout the growing season during water years 2018, 2019, and 2020. Unforeseen delays in the first year of the project resulted in the collection of water stargrass biomass on only a single sampling event in water year 2018. During water year 2019, three sampling events for water stargrass biomass were completed. In water year 2020, the COVID-19 pandemic caused complications in field work, so only two sampling events were completed. In total, six unique sampling events for water stargrass biomass were completed: August 2018, June 2019, August 2019, September 2019, June 2020, and August 2020.

Estimates of water stargrass biomass were highly variable within and across sites during the study period. Median water stargrass biomass ranged from 157 to 1,090 grams of dry mass per meter squared (g-dry mass/m^2) (**Table 13**). Overall, Prosser had the highest water stargrass biomass and Van Giesen had the lowest throughout the study except for September 2019 when biomass data across sites were statistically similar (refer to letter groups in **Table 13**). At Kiona, the biomass values were statistically similar to Prosser in some months and Van Giesen in others (refer to letter groups in **Table 13**). Percent cover estimates based on presence/absence of water stargrass also varied across sites and during the growing seasons. Kiona consistently had the highest percent cover estimates across sites and Van Giesen had the lowest, except for June 2019, when Kiona and Prosser were similar (41 percent compared to 47 percent, respectively; **Table 13**). When multiple field visits occurred within a single growing season, percent cover estimates always increased, indicating the spread of water stargrass within each sample reach during summer months. During September 2019, streamflows were too high to allow presence/absence measures at Kiona and Van Giesen, and data were not collected (**Table 13**).

Table 13. Summary of water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass data from the lower Yakima River, August 2018 to August 2020. Additional site information is provided in table 1. Data are available in Sheibley and Foreman (2024).

[No., number; g, grams, dry mass; m^2, square meters; nd, no data collected; letters in parentheses across each row represent the statistical groups from a post hoc Friedman's least significant difference test on the Kruskal-Wallis non-parametric analysis of variance procedure to test what categories were statistically different from each other; groups across rows that share a letter signify that there is no statistical difference between those groups; groups with the same letter across each row are statistically similar.]

Sample date(s)	Sample ID	Prosser			Kiona			Van Giesen		
		No. of samples	Median (g/m^2)	Percent cover	No. of samples	Median (g/m^2)	Percent cover	No. of samples	Median (g/m^2)	Percent cover
8/14/18 to 8/16/18	Aug 2018	10	1090(a)	48	10	289(b)	62	10	379(b)	27
6/25/19 to 6/27/19	June 2019	10	816(a)	47	10	494(a)	41	10	159(b)	17
7/31/19 to 8/1/19	Aug 2019	10	767(a)	73	11	533(ab)	81	10	420(b)	31
9/18/19	Sept 2019	6	948(a)	81	6	850(a)	nd	7	836(a)	nd
6/27/20	June 2020	10	323(a)	33	10	179(ab)	59	10	179(b)	19
8/10/20 to 8/11/20	Aug 2020	10	705(a)	73	10	481(a)	83	10	157(b)	24

Comparison of water stargrass biomass during the growing season (June through September) showed differing patterns within and across stations (**Figure 14**a-c). At Prosser, water stargrass biomass during the growing season in 2019 was statistically similar from June through September (**Figure 14**a). However, in 2020, water stargrass showed a statistical increase from June through August indicating an accumulation of water stargrass within the reach (**Figure 14**a). Similarly, at Kiona, water stargrass biomass was not significantly different from June through September in 2019, but in 2020 there was a significant increase in water stargrass biomass from June to August (**Figure 14**b). At Van Giesen, water stargrass growth behaved differently. In 2019, water stargrass biomass showed statistically significant increases in biomass from June through September (**Figure 14**c). In contrast, water stargrass biomass at Van Giesen in 2020 was not significantly different in June compared to August (**Figure 14**c).

A more common feature of the water stargrass biomass data across sites and water years was a statistically significant decrease in biomass between the final sample in one water year compared to the first sample in the next water year. For example, water stargrass biomass in September 2019 was significantly greater than June 2020 biomass values at all three locations (**Figure 14**a-c). A similar pattern was observed between August 2018 and June 2019 at Van Giesen (**Figure 14**c). These decreases in water stargrass biomass from one water year to the next indicates that a ‘resetting’ of biomass from the previous growing season is occurring across all sites.

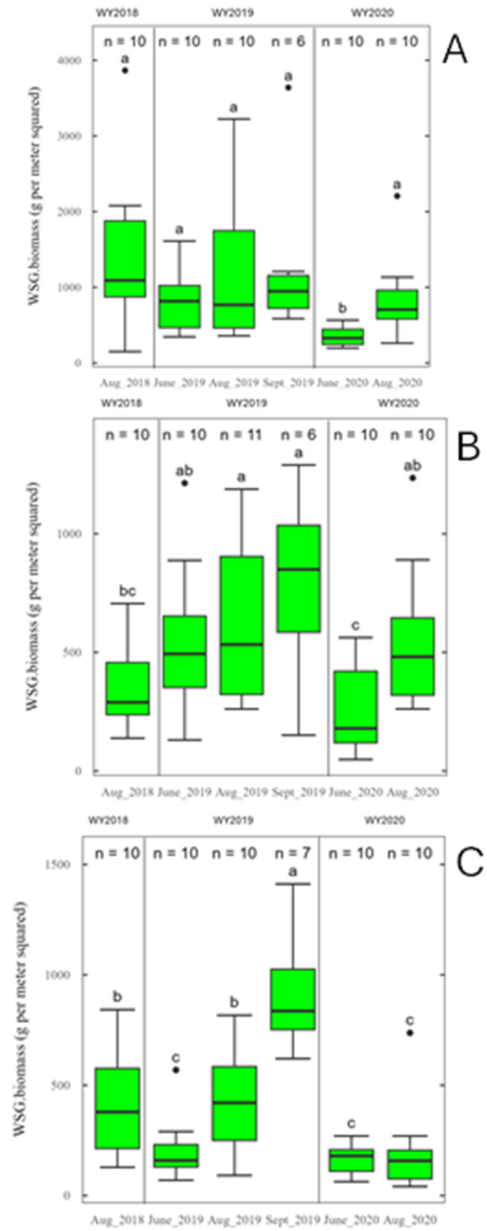

Figure 14. Comparison of water stargrass (WSG; grass-leaf mud-plantain, *Heteranthera dubia)* biomass estimates across sample dates for (a) Prosser, (b) Kiona, and (c) Van Giesen. Vertical lines indicate boundaries of each water year (WY). Boxes with the same letter(s) are not statistically different from each other based on a Kruskal-Wallis non-parametric analysis of variance procedure. Additional site information is provided in table 1. Data are available from Sheibley and Foreman (2024).

Stream metabolism estimates

Estimates of stream metabolism as gross primary productivity (GPP) and ecosystem respiration (ER) were determined using the streamMetabolizer package (version 0.12.0; Appling and others 2018) at each monitoring station. All model inputs and outputs are provided in a companion data release (Sheibley and Foreman, 2024). Model fits at Prosser were poor, and daily estimates of GPP, ER and the reaeration coefficient (K600) were highly variable. Large spikes in these parameters were frequent and unrealistic throughout the modeling period. Because of the erratic estimates, data from Prosser are not presented here. It is likely that the location of the site just below a dam combined with a wide, deep channel (often over 6 feet deep in the middle of the channel) contributed to these poor fits. For example, water flow over a dam can result in oversaturated DO conditions, and deep waters impact how quickly oxygen exchange takes place between the water surface and atmosphere. The streamMetabolizer approach is not optimized for these kinds of conditions.

Metabolism models at Kiona and Van Giesen were more stable than for Prosser, and values for GPP followed expected seasonal patterns. For Kiona and Van Giesen, GPP was lowest in the winter months, and highest in summer months when days were longer, and temperatures are much warmer (**Table 14**). Monthly average GPP at Kiona ranged from 1.2 to 26.0 grams of oxygen per meter squared per day (g-O_2/m^2-day). At Van Giesen, dissolved oxygen sensor problems resulted in erroneous data from June 2019 through September 2019, and from March 2020 until the end of the study (October 2020); as a result, summer GPP was not estimated for Van Giesen in 2019 and 2020. The monthly GPP values at Van Giesen ranged from 1.8 to 22.1 grams of oxygen per square meter per day (g-O_2/m^2-day), and data were only primarily available for times outside of the water stargrass growing season.

Table 14. Monthly average gross primary productivity in the lower Yakima River, August 2018 to September 2020. Additional site information is provided in table 1. Data are available from Sheibley and Foreman (2024).

[WY, water year; --, no data available; all data given in grams of oxygen per square meter per day (g-O_2/m^2-day)]

	Kiona			Van Giesen		
Month	**WY2018**	**WY2019**	**WY2020**	**WY2018**	**WY2019**	**WY2020**
October	--	13.2	12.1	--	14.6	10.8
November	--	6.9	6.8	--	7.1	8.5
December	--	5.1	3.7	--	6.8	5.6
January	--	2.0	1.7	--	4.1	2.1
February	--	2.4	0.6	--	3.0	--
March	--	4.7	4.1	--	5.1	--
April	--	1.2	7.4	--	1.8	--
May	--	7.1	6.0	--	10.4	--
June	13.8	17.6	13.4	--	17.8	--
July	20.0	26.0	21.2	--	--	--
August	24.6	24.2	24.2	17.4	--	--
September	20.4	17.7	17.1	22.1	16.6	--

Daily ecosystem respiration (ER) and GPP were plotted against each other for Kiona and Van Giesen (**Figure 15**). The 1:1 line indicates when GPP equals ER and points above the line indicate heterotrophy, and points below the line indicate autotrophy. At both sites, a similar pattern was observed each water year: streams were heterotrophic most days from late October until early June (fall, winter, and spring) and autotrophic from June through early October (summer).

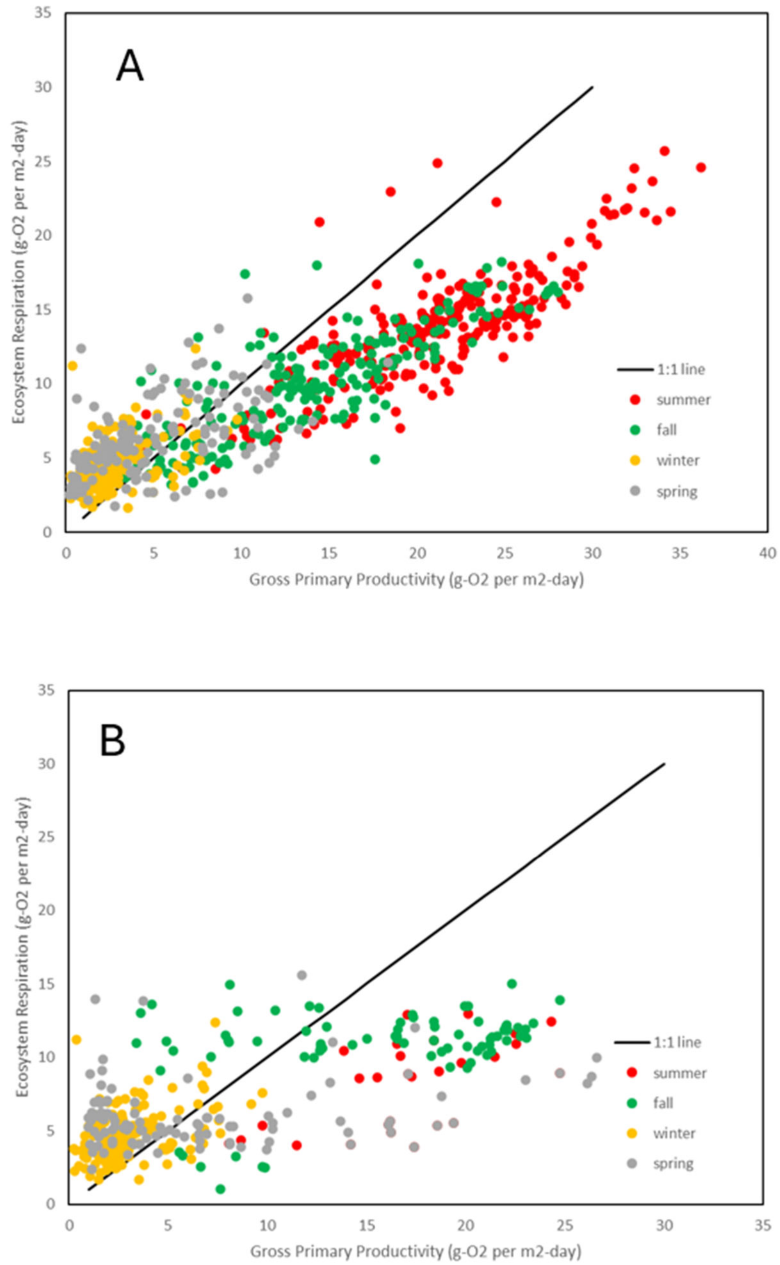

Figure 15. Daily estimates of gross primary productivity (GPP) and ecosystem respiration (ER) for (a) Kiona and (b) Van Giesen. The solid line represents GPP = ER and points above the line represent heterotrophic conditions and values below the line represent autotrophic conditions. Data are available from Sheibley and Foreman (2024). [g-O_2 per m^2-day, grams of oxygen per square meter per day]

Relationships between streamflow, water quality, stream metabolism, and water stargrass biomass

Through the process of photosynthesis, aquatic plants can influence the diel (daily) patterns in DO and pH of surface waters. Typical patterns in DO and pH, increases during daylight hours and decreases during nighttime, were observed at all three monitoring stations throughout this study and were most noticeable in summer months. In addition, nutrients are used by plants to assist growth and may also show diel patterns. A plot of DO, pH, and nitrate during a typical summer week in August illustrates this pattern at Kiona (**Figure 16**). We observed increases in DO and pH during the day and decreases in both parameters during evening hours. Nitrate showed an opposite pattern because nutrient uptake is greater during daylight, therefore nutrients tend to decrease during daylight and increase during nighttime hours (**Figure 16**).

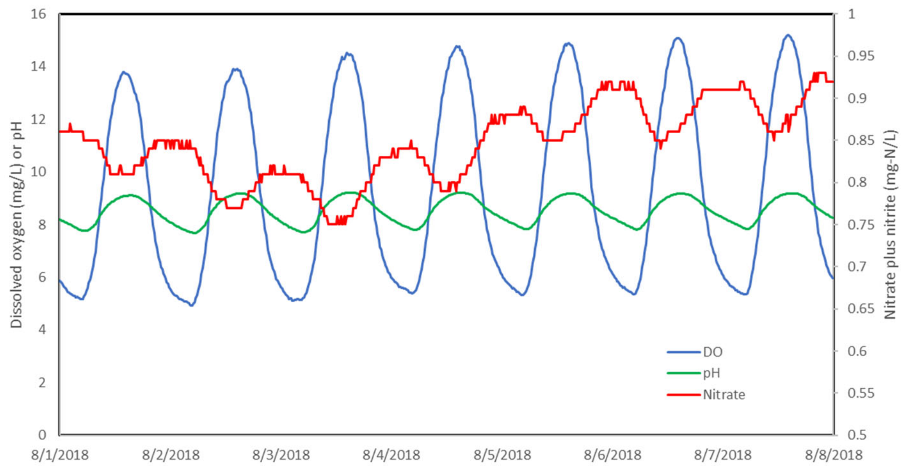

Figure 16. Daily variation in dissolved oxygen, pH and nitrate at Kiona from August 1 to August 8, 2018. Where mg/L is milligrams per liter and mg-N/L is milligrams of nitrogen per liter. Dates labels on the x-axis indicate midnight and the start of the day. Additional site information is provided in table 1. Data for USGS stations are available through the USGS National Water Information System (NWIS; U.S. Geological Survey, 2022; https://doi.org/10.5066/F7P55KJN).

To test the potential influence of water stargrass growth on surface water quality, we conducted a detailed analysis of continuous water quality data on measures of water stargrass and stream metabolism. In addition, because high streamflow can physically affect plant growth through scour and reduction in water clarity, we compared some simple streamflow metrics to water stargrass biomass. These comparisons are presented in the section called “Streamflow conditions and water stargrass biomass.”

Comparisons between water quality and water stargrass biomass

We focused on seven daily water quality metrics and their relationships to water stargrass biomass: maximum temperature, DO minimum, DO range, pH maximum, pH range, mean nitrate concentration, and nitrate concentration range. These metrics capture the important parameters influenced by photosynthesis (DO, pH, nitrate) and frequently observed exceedances to water quality criteria (maximum temperature, DO minimum, and pH maximum) on the lower Yakima River. Daily maximum temperature was also included because we hypothesized that dense beds of aquatic macrophytes can slow down stream velocity and potentially increase the heating of surface water from the summer sun. However, even though each monitoring station was in an open reach with minimal canopy, shading from bank vegetation might be important as well.

The first step in this analysis was to determine what time frame to use for calculating these water quality metrics. To determine this, a non-parametric analysis of variance was completed on the seven water quality metrics for four different periods prior to each water stargrass sample: a 7-day, 14-day, 21-day, and 28-day subset of the daily data. Each period was a factor in the analysis of variance, and the water quality metric data were from the same set of daily data, from 7 to 28 days, making the data temporally autocorrelated. These analyses were done for each sample location (n=3) and for each biomass sample date (n=6) for each of the seven water quality metrics for a possible 126 different combinations of water quality and period across the three monitoring locations. However, due to missing data in summer, particularly for DO at Van Giesen, only 103 of the possible 126 analyses of water quality and period were examined. Of these comparisons, about half showed there was no significant difference between the water quality metric and the period. The rest of these analyses showed there was a significant difference between the 7-day average and 28-day average. As a result, we examined both the 7- and 28-day average for the seven metrics to assess short-term (7-day) and long-term (28-day) water quality and water stargrass biomass. The analysis of variance tests were not used to establish final statistical significance values but rather were used to identify the periods most appropriate for further consideration.

A non-parametric correlation analysis (Spearman's rho) between median water stargrass biomass estimates and the 7-day and 28-day metrics was done for each site individually and then a second time combining the data from all three sites into a single 7-day and 28-day analysis. The combined analysis was done to increase the sample size and variability by incorporating all the potential data. The site-specific analyses only had at most six data points for each correlation corresponding to the number of times biomass was determined throughout the study. At Van Giesen, several comparisons could not be made because of missing summer DO data. Statistical significance was evaluated with a 90 percent confidence interval ($p<0.10$) based on the small sample sizes being evaluated.

Statistically significant results from the correlation analysis between the seven water quality metrics and median water stargrass biomass for the individual sites were not consistent. At Prosser, only the 7-day average daily maximum pH ($p=0.037$) and average daily pH range ($p=0.037$) showed significant correlations with median water stargrass biomass (**Table 15**). At Kiona, both the 7-day and 28-day mean nitrate values showed a significant relationship to median water stargrass biomass (**Table 16**). At Van Giesen, there were no significant correlations between the seven water quality metrics and median water stargrass biomass (**Table 17**).

Table 15. Summary of non-parametric correlation analysis between the 7-day and 28-day water quality metrics and median water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass at Prosser, June 2018 to August 2020. Additional site information is provided in table 1.

[Max., maximum; DO, dissolved oxygen; N, sample size; **bold** rows represent significant correlations between the metric and water stargrass biomass; --, no data available]

Metric	Time interval	N	Spearman rho	p-value
Max. Temperature	7-day	5	-0.600	0.285
DO minimum	7-day	5	-0.100	0.873
DO range	7-day	5	-0.600	0.285
Max. pH	**7-day**	**5**	**-0.900**	**0.037**
pH range	**7-day**	**5**	**-0.900**	**0.037**
Mean nitrate	7-day	--	--	--
Nitrate range	7-day	--	--	--
Max. Temperature	28-day	6	0.200	0.704
DO minimum	28-day	6	-0.543	0.266
DO range	28-day	6	0.143	0.787
Max. pH	28-day	6	-0.143	0.787
pH range	28-day	6	-0.429	0.397
Mean nitrate	28-day	--	--	--
Nitrate range	28-day	--	--	--

Table 16. Summary of non-parametric correlation analysis between the 7-day and 28-day water quality metrics and median water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass at Kiona, June 2018 to August 2020. Additional site information is provided in table 1.

[Max., maximum; DO, dissolved oxygen; GPP, gross primary productivity; N, sample size; **bold** rows represent significant correlations between the metric and water stargrass biomass]

Metric	Time interval	N	Spearman rho	p-value
Max. Temperature	7-day	6	-0.429	0.397
DO minimum	7-day	6	0.029	0.957
DO range	7-day	6	-0.029	0.957
Max. pH	7-day	6	-0.257	0.623
pH range	7-day	6	-0.029	0.957
Mean nitrate	**7-day**	**6**	**0.886**	**0.019**
Nitrate range	7-day	6	-0.086	0.872
GPP	7-day	6	-0.371	0.468
Max. Temperature	28-day	6	-0.086	0.872
DO minimum	28-day	6	-0.200	0.704
DO range	28-day	6	0.200	0.704
Max. pH	28-day	6	0.314	0.544
pH range	28-day	6	-0.314	0.544
Mean nitrate	**28-day**	**6**	**0.943**	**0.005**
Nitrate range	28-day	6	-0.314	0.544
GPP	28-day	6	0.486	0.329

Table 17. Summary of non-parametric correlation analysis between the 7-day and 28-day water quality metrics and median water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass at Van Giesen, June 2018 to August 2020. Additional site information is provided in table 1.

[Max., maximum; DO, dissolved oxygen; N, sample size; **bold** rows represent significant correlations between the metric and water stargrass biomass; --, no data available; na, not applicable sample size was too small]

Metric	Time interval	N	Spearman rho	p-value
Max. Temperature	7-day	6	0.029	0.957
DO minimum	7-day	1	--	--
DO range	7-day	1	--	--
Max. pH	7-day	6	0.086	0.872
pH range	7-day	6	0.257	0.623
Mean nitrate	7-day	4	0.400	0.600
Nitrate range	7-day	4	0.200	0.800
Max. Temperature	28-day	6	-0.314	0.544
DO minimum	28-day	2	na	na
DO range	28-day	2	na	na
Max. pH	28-day	6	0.600	0.208
pH range	28-day	6	0.143	0.787
Mean nitrate	28-day	4	0.400	0.600
Nitrate range	28-day	4	-0.200	0.800

When the correlation analysis was repeated with the dataset combining all three sites together, several water quality metrics showed significant relationships with median water stargrass biomass (**Figure 8**). For the combined 7-day metrics, there was a significant correlation between the median water stargrass biomass and maximum temperature, DO minimum, DO range, pH maximum, and mean nitrate. For the combined 28-day metrics, only nitrate range showed a significant correlation with median water stargrass biomass. These results indicate that water stargrass might influence short-term water quality more than long-term water quality. However, the direction of these correlations are opposite of what was expected. For example, if greater water stargrass biomass resulted in larger daily swings in DO and pH, the expectation would be a positive correlation with DO range, and maximum pH; however, the correlation results showed that water stargrass biomass was inversely correlated with DO and pH. The relation between median water stargrass biomass and mean nitrate showed a significant positive correlation for both the 7-day and 28-day metrics, indicating that nitrate in the water column may be important to water stargrass growth.

The correlation analysis was also completed between the 7-day and 28-day median GPP and median water stargrass biomass; however, only data from Kiona was used because of the missing DO data in summer 2019 and 2020 at Van Giesen. Results showed there was no significant correlation between the 7-day ($p = 0.468$) and 28-day ($p = 0.329$) median GPP and median water stargrass biomass (**Table 18**).

Table 18. Summary of non-parametric correlation analysis between the 7-day and 28-day water quality metrics and median water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass from a combination of three lower Yakima River monitoring sites, June 2018 to August 2020. Additional site information is provided in table 1.

[Max., maximum; DO, dissolved oxygen; GPP, gross primary productivity; N, sample size; **bold** rows represent significant correlations between the metric and water stargrass biomass]

Metric	Time interval	N	Spearman rho	p-value
Max. Temperature	**7-day**	**17**	**-0.620**	**0.008**
DO minimum	**7-day**	**12**	**0.524**	**0.080**
DO range	**7-day**	**12**	**-0.692**	**0.013**
Max. pH	**7-day**	**17**	**-0.699**	**0.002**
pH range	7-day	17	0.039	0.881
Mean nitrate	**7-day**	**10**	**0.697**	**0.025**
Nitrate range	7-day	10	0.200	0.580
GPP	7-day	6	-0.371	0.468
Max. Temperature	28-day	18	-0.150	0.553
DO minimum	28-day	14	-0.213	0.464
DO range	28-day	14	-0.332	0.246
Max. pH	28-day	18	-0.253	0.311
pH range	28-day	18	0.092	0.717
Mean nitrate	**28-day**	**10**	**0.733**	**0.016**
Nitrate range	28-day	10	-0.248	0.489
GPP	28-day	6	0.485	0.329

Comparisons between water quality and GPP

A correlation analysis between the 7-day and 28-day water quality metrics and GPP estimates was completed for Kiona. These comparisons were not possible at Van Giesen due to insufficient DO data during the periods when water stargrass was measured (June through September).

Significant correlations ($p < 0.10$) between GPP and maximum temperature, minimum DO, DO range, and pH range for the 7-day metrics were observed (Table 19). For the 28-day metrics, both minimum DO and DO range showed significant correlations with GPP. The direction of these correlations were as expected (**Table 9**). For example, as GPP increases, the higher productivity is driving larger changes in pH and DO causing the DO and pH ranges to increase and DO minimum to decrease.

Table 19. Summary of non-parametric correlation analysis between daily water quality metrics and gross primary productivity, June 2018 to August 2020.

[Max., maximum; DO, dissolved oxygen; N, sample size; **bold** rows represent significant correlations between the metric and water stargrass biomass]

Metric	Time interval	N	Spearman rho	p-value
Max. Temperature	**7-day**	**6**	**0.943**	**0.005**
DO minimum	**7-day**	**6**	**-0.771**	**0.072**
DO range	**7-day**	**6**	**0.771**	**0.072**
Max. pH	7-day	6	0.143	0.787
pH range	**7-day**	**6**	**0.771**	**0.072**
Mean nitrate	7-day	6	-0.486	0.329
Nitrate range	7-day	6	0.600	0.208
Max. Temperature	28-day	6	0.600	0.208
DO minimum	**28-day**	**6**	**-0.829**	**0.042**
DO range	**28-day**	**6**	**0.829**	**0.042**
Max. pH	28-day	6	0.371	0.468
pH range	28-day	6	0.314	0.544
Mean nitrate	28-day	6	0.600	0.208
Nitrate range	28-day	6	0.429	0.397

Streamflow conditions and water stargrass biomass

Three simple indicators of streamflow were used to examine the relation between streamflow and water stargrass biomass: the reach average channel depth at each station, the spring peak (March through May) discharge at Kiona, and the average of spring (March through May) discharge at Kiona. The reach average channel depth was an input to the streamMetabolizer model (Sheibley and Foreman, 2024) and was used as an indicator of physical conditions at each station. Similar to the water quality metrics, a 7-day and 28-day reach average channel depth was determined and compared to median water stargrass biomass at each location. This average channel depth was used because observations at the stations showed that in deeper and slower moving water, typical of conditions at Prosser, water stargrass tended to be much taller when compared to water stargrass in shallow and fast-moving waters typical of conditions at Van Giesen. A correlation analysis between reach average channel depth and median water stargrass biomass was completed (**Table 20**). However, there were no significant relationships between the reach average channel depth and median water stargrass biomass across individual sites or when the data were combined across all sites (**Table 20**).

Two streamflow metrics, the spring peak discharge value and the average spring discharge were calculated at Kiona and compared to late-season (August) median water stargrass biomass values. The peak spring discharge and the average spring discharge followed a similar pattern across water years, showing the largest values in 2018 and the smallest values in 2020 (**Table 5**, **Figure 17)**. The spring peak discharge at Kiona was 12,500, 8,490, and 5,730 ft^3/s in 2018, 2019, and 2020, respectively (**Table 5**). The average spring discharge was 6,150, 4,750, and 2,995 ft^3/s in 2018, 2019, and 2020, respectively (**Table 5**).

Table 20. Summary of non-parametric correlation analysis between reach average channel depth and median water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass, June 2018 to August 2020.

[N, sample size; **bold** rows represent significant correlations]

Location	Time interval	N	Spearman rho	p-value
Prosser	7-day	6	-0.257	0.623
Prosser	28-day	6	-0.257	0.623
Kiona	7-day	6	0.257	0.623
Kiona	28-day	6	0.143	0.787
Van Giesen	7-day	6	0.086	0.872
Van Giesen	28-day	6	-0.029	0.957
Combined sites	7-day	18	0.307	0.216
Combined sites	28-day	18	0.222	0.376

Spring peak and average spring discharges were compared to August water stargrass biomass data at each location. August biomass across each year was selected to compare a similar duration into the growing season across the three water years. At Kiona, the August median water stargrass biomass decreased as both spring peak discharge and average spring discharge increased (**Figure 17**). The results from Kiona indicate a possible relation between August biomass and spring streamflows; as spring streamflows increase in magnitude and duration, water stargrass growth may be hindered in the lower Yakima River. The data at Prosser and Van Giesen did not follow this pattern. August biomass increased with increasing spring streamflows at Prosser and Van Giesen (**Figure 17**). Interestingly, the median August water stargrass biomass at both Kiona and Van Giesen was the highest in 2019. This results indicates that the relation between spring streamflows and water stargrass growth may not be linear and that an optimal set of streamflow conditions may exist between high and low streamflow years.

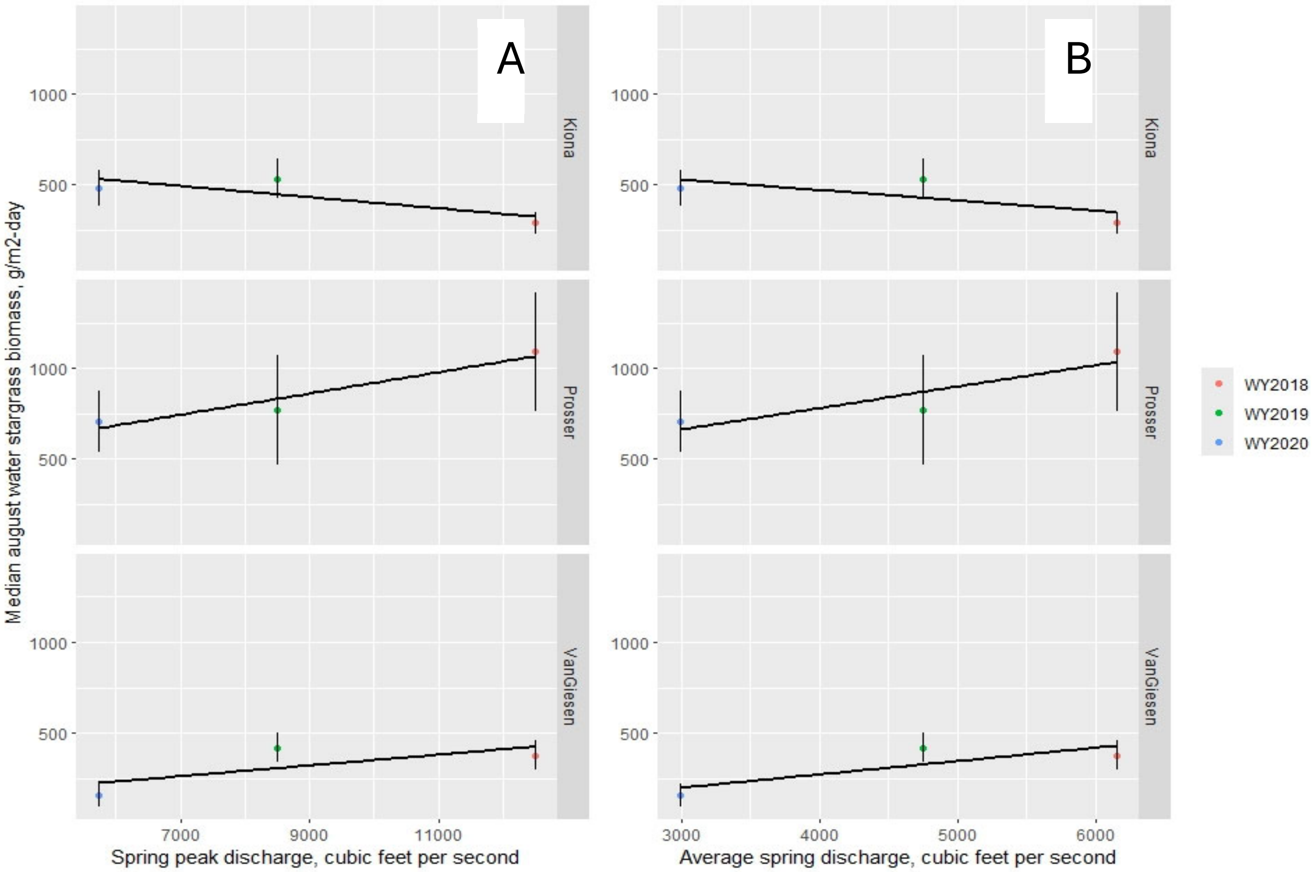

Figure 17. Spring peak discharge (A) and average spring discharge (B) at Kiona compared to median water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass in August for water years 2018, 2019, and 2020. Additional site information is provided in table 1.

Discussion

Streamflow on the lower Yakima River follows a typical seasonal pattern with highest streamflows in during winter storms and spring runoff, and relatively steady low streamflow in summer. During summer, there are large irrigation demands placed on the river and river stage decreases rapidly in late spring as ambient air temperatures begin to rise and irrigation demands increase (Vaccaro, 2011). Hydrologic conditions on the lower Yakima River varied during water years 2018 through 2020, particularly with respect to spring streamflows. During water year 2018, the highest spring peak discharge and the largest sum of spring discharge were recorded for this study. In subsequent years, these streamflow metrics were lower.

The three monitoring sites had variable channel conditions such as river width and average water depth which likely influenced the water quality and water stargrass biomass at these sites. At Prosser, the river was an intermediate width but was very deep in the center of the channel, often exceeding 6-7 feet in depth. In addition, water clarity in the middle of the channel was lowest of all three monitoring sites. The streamflow at Prosser could be categorized as a glide habitat, with turbulent streamflow features rarely observed. Substrate at Prosser was sandy along the margins of the channel with large boulders, sand, and cobble throughout the center of the reach. At Kiona, the river was the widest and shallowest of all three sites, with depths rarely exceeding 2 feet in the study reach. Water at Kiona was swift, turbulent, and clear. The substrate consisted mainly of medium to large cobbles and contained a large amount of filamentous algae relative to the other two sites. The most downstream site at Van Giesen had the smallest width, and the reach had a range of depths, from over 6 feet on the right back, to less than 1 foot on the left bank. The channel geometry at Van Giesen was complex, with turbulent rapids in some parts, and deep glides in others. The variability in both channel characteristics and spring streamflows provide an opportunity to examine differences in water quality, water stargrass biomass, and GPP.

Daily maximum temperatures exceeded the Washington State criteria (21 °C) almost every day at all three monitoring stations from June through August for all water years in this study. This was consistent with results from several other temperature studies of the lower Yakima River including multiple studies to document thermal conditions during longitudinal floats of the lower Yakima River (Vaccaro and others, 2008; Vaccaro, 2011; Appel and others, 2011; Gendaszek and Appel, 2021). These conditions are detrimental to migrating salmonid populations; however, areas of cooler water from cool-water tributaries, groundwater inputs, and localized channel features of the river may provide refuge areas for adult migratory fish species and help lessen thermal stress during the early summer through early fall migration periods (Appel and others, 2011; Gendaszek and Appel, 2021; Sheibley and others, 2024).

Diel (daily) fluctuations in DO are common in rivers due to in-river photosynthesis processes where photosynthesis occurs during the day (increasing DO levels) and respiration occurs at night (decreasing DO levels). While normal, the large fluctuations resulting from this daily cycle can cause drops in DO levels at night that are less than the state minimum water quality criterion of 8 mg/L (Ecology, 2019). All three stations in this study showed that daily minimum DO concentrations during summer can frequently fall below this criterion. Of the three monitoring stations, Kiona had the lowest minimum DO concentrations and demonstrated the most obvious, substantial, and consistent pattern in diel DO fluctuations during the growing season (**Figure 7**). Large daily fluctuations in DO are indicative of aquatic plant photosynthesis and respiration, which seem to be controlling dissolved oxygen dynamics on the lower Yakima River.

Daily pH values are also influenced by photosynthesis, with increases in pH during the day and decreases during the nighttime. The amplitude of the daily range of pH was largest during baseflow conditions (June through August), which coincides with the growing season for aquatic primary producers. Similar to patterns observed with the daily DO data, the largest swings in pH were observed at Kiona. Exceedances of the state water quality standards (Ecology, 2019) were observed in the summer with maximum pH exceeding 8.5 almost every day at Kiona and Van Giesen and for only a few days each summer at Prosser. At no time did the pH fall below the state's minimum criteria of 6.5 during the study.

Understanding how the relation between respiration and photosynthesis of the lower Yakima River community affect DO and pH levels is vital to basin-wide efforts in restoring summer salmon stocks (Appel and others, 2011). Respiration and photosynthesis in the lower Yakima River from the dense macrophyte beds are causing pH and DO values in ranges that are likely detrimental to salmonids (Appel and others, 2011). Continuous monitoring of the lower Yakima River highlights that during baseflow conditions and peak biomass growing season, there are locations that fail to meet the maximum daily temperature, minimum daily DO, and maximum daily pH criteria of the lower Yakima River. Late spring and summer adult salmonid migrants that are exposed to thermal stress during the day may encounter added stress from low DO levels during their movement at night. In fact, studies of juvenile salmon movements in the lower Yakima River show that populations 'hold' in the Columbia River at the mouth of the Yakima River until temperatures start to decline in September (Kock and others, 2020). Elevated river temperatures in combination with lower DO levels and increased pH also create more favorable conditions for non-native predator species over native salmonids (McMichael, 2017), further compounding challenges for the more sensitive migratory populations.

Turbidity of surface water can influence both light availability and macrophyte growth. Continuous turbidity data on the lower Yakima River showed that background turbidity is generally very low across all sites (5 FNU or less), and peaks during high streamflows are short-lived. A suspended sediment load model was derived from the turbidity data (Anderson and others, 2023) and showed that sediment loads at Kiona are greatest during winter storms and spring runoff. In summer months, sediment loads are lowest. Removal of high levels of suspended sediments in the early 2000s improved light clarity and water quality in the lower Yakima River as suspended solids, predominantly originated from Yakima River Basin agricultural lands, carried pesticides, chemicals (such as polychlorinated biphenyls, or PCBs), and nutrients (nitrogen and phosphorus). While the concerted basin-wide efforts decreased levels of suspended sediments and toxics to the lower Yakima River, they also improved water clarity that likely contributes to the abundant plant growth that is observed each summer (Wise and others, 2009; Appel and others 2011). This is consistent with the data collected for this study, summer baseflow tends to have low turbidity and low sediment loads indicating relatively clear water throughout the aquatic plant growing season.

Continuous nitrate at Kiona and Van Giesen showed similar seasonal patterns. There were three distinct 'nitrate periods.' During fall and winter, nitrate concentrations remain high until spring high streamflows, when nitrate starts to decline. During spring, peaks in both nitrate concentration and load correspond to increases in streamflow until they reach an annual minimum in the early summer. During summer, both the nitrate concentration and load increase from June through September. This increase in nitrate load without a significant increase in streamflow indicates there is a sustained input of nutrients in the summer. Possible sources of this summer nitrate are from agricultural return flows and transport from groundwater discharge into the river. Previous studies have shown that legacy nitrate from agricultural fertilization can take decades to reach nearby rivers and streams (Tesoriero and others, 2013). In the lower Yakima River, groundwater inputs might contribute a sustained input of nutrients to the river during summer months. This would explain the patterns in nitrate load at Kiona, specifically dilution of nitrate during spring high streamflows and increases during summer low streamflows.

Diel nitrate signals (**Figure 16**) follow patterns of nutrient uptake by the river communities (algae, macrophytes, fish, macroinvertebrates) with decreases in nitrate during the day when uptake is most active and increases during nighttime when uptake rate goes down. Nitrate can be utilized by aquatic plants, filamentous algae, biofilms on hard substrate, and through microbial transformations, such as nitrification and denitrification (Duff and Triska, 2000). At Kiona, both the 7-day and 28-day mean nitrate values showed a significant relation to median water stargrass biomass. This indicates that water stargrass may use some water column nitrate for growth; however water stargrass may also be utilizing nutrients from the sediment and hyporheic flow. The nutrients within the sediment bed and hyporheic zone were not examined as part of this study, and future research could investigate the cycling of nutrients between the sediment bed, water column, and uptake by water stargrass. Overall, the diurnal patterns in nitrate indicate uptake by the river community, which is influenced by water stargrass to some extent.

Discrete water quality data collected for nutrients were collected at each site over the 2-year study period. Overall, ammonia was low and rarely detected in surface water samples during the study. Nitrate plus nitrite, total phosphorus, and total nitrogen concentrations all exceeded suggested EPA reference conditions for the Columbia Plateau Ecoregion (U.S. Environmental Protection Agency, 2000). Nutrient concentrations were compared to data collected at similar sites in 2004 to 2007 (Wise and others, 2009). The data from Wise and others (2009) were presented as averages for the "Kiona Reach" in that study, which represented river miles 4 through 47 of the Yakima River. In comparison, Kiona and Van Giesen from this study are located within the 'Kiona Reach' of Wise and others (2009). Data from this study showed that nitrate plus nitrite, and total nitrogen concentrations increased between 2004–2007 and 2018-2020 (**Table 21**). In contrast, both orthophosphate and total phosphate between these periods (**Table 21**).

Table 21. Comparison of nutrient data from this study (water years 2018–2020) to data collected from 2004-2007. Additional site information is provided in table 1.

[N, nitrogen; P, phosphorus; mg/L, milligrams per liter]

Parameter	Units	Prosser	Kiona	Van Giesen	Wise and others (2009)
Nitrate + nitrite as N	mg/L	1.17	0.95	1.07	0.875
Orthophosphate as P	mg/L	0.07	0.06	0.07	0.126
Total Phosphorus as P	mg/L	0.12	0.10	0.10	0.140
Total Nitrogen as N	mg/L	1.40	1.21	1.32	1.11

Water stargrass biomass estimates were variable across sites and sampling seasons. The factors controlling water stargrass growth are complex and are likely related to seasonal streamflow patterns of the river and local channel conditions. For example, we observed that August water stargrass biomass estimates were related to spring peak streamflows, particularly at Kiona (**Figure 17**). At Kiona, median water stargrass biomass was lower during a high-streamflow year (2018) compared to a low streamflow year (2020). In addition, median water stargrass biomass in 2019 was higher than in 2020 at Kiona. Water year 2019 had lower and more sustained baseflow than observed in water year 2020. These observations provide evidence that yearly variations in basin hydrology impact the total yearly water stargrass biomass production. For example, during the spring period (March through May), streamflows at Kiona tended to be high due to a combination of snowmelt in the upper basin and changes in water releases in upstream dams. These increased streamflows may lead to decreased water stargrass biomass in the lower Yakima River. Therefore, the implementation of prescribed streamflow pulses from upper basin reservoirs could be explored further as a management strategy in the future. However, many of the observations between streamflow and water stargrass biomass at Kiona were not consistent with observations at Prosser and Van Giesen. This departure in the relation between streamflow and water stargrass biomass at Prosser and Van Giesen is likely related to (1) comparing biomass data to streamflow data at a different location, and (2) the influence of dams and diversions in the lower Yakima River. Data at the Prosser site were collected just downstream of the Prosser Dam and a diversion for a fish hatchery, and Horn Rapids Dam and an associated water diversion is located between Kiona and Van Giesen. The location of these streamflow modifications will influence the streamflow dynamics that are observed at Kiona where the streamgage is located in a part of the lower Yakima River that is fairly unaltered.

Local channel conditions also contribute to water stargrass growth. At Prosser, where the channel is deeper and streamflow velocity is slower, the largest values for water stargrass biomass were observed. The channel conditions at Prosser allow for much larger water stargrass plants, some often exceeding 4 feet or more in length. In contrast, Van Giesen has swifter streamflows, a shallower channel than Prosser, and lacks deep pools. As a result, measured biomass at Van Giesen was much less than at Prosser and characterized by smaller plants and a percent cover that never exceeded 31 percent (Sheibley and Foreman, 2024). In contrast, at Prosser, the percent cover of water stargrass often exceeded 50 percent. Kiona represents a middle condition to the other two sites with respect to channel conditions. The channel is fairly uniform and lacks any substantially shallow or deep areas, with water depths on average 1 – 2 feet deep. Visual observations of velocity indicate streamflow is moderately swift but likely not fast enough to hinder or scour plants once they become established. Therefore, the percent cover at Kiona (41 to 83 percent) was greater than the other two sites likely because of the channel uniformity and lower depths and because the water stargrass plants were bigger than at Van Giesen, but smaller than at Prosser.

When multiple estimates of water stargrass biomass were made within the same growing season, similar patterns were observed across all sites. First, during the growing season, biomass and percent cover tended to increase, indicating that plants were getting bigger as the season continued. Second, the first sample of the subsequent water year showed a decline in biomass relative to the value at the end of the previous growing season. This second characteristic indicates that water stargrass biomass was 'reset' to some extent during winter and spring, likely from a combination of high streamflows and natural die-off from cooler and shorter days during these seasons.

An objective of this project was to examine whether the growth of water stargrass influenced surface water quality. If water stargrass is influencing surface water quality and contributing to exceedances in water quality criteria, then management approaches to control water stargrass growth could help improve river water quality and conditions favorable to migrating salmonids. In this study, seven different water quality metrics over two different periods were examined for relations with water stargrass biomass. Statistically significant results from the correlation analysis between the seven water quality metrics and median water stargrass biomass for the individual sites were not consistent. At Prosser, only the 7-day average daily maximum pH and average daily pH range showed significant correlations with median water stargrass biomass (**Table 15**). At Kiona, both the 7-day and 28-day mean nitrate values showed a significant relation to median water stargrass biomass (**Table 16**). At Van Giesen, there were no significant correlations between the seven water quality metrics and median water stargrass biomass (**Table 17**). These inconsistencies between water stargrass biomass and water quality metrics do not necessarily indicate that water stargrass growth is not important but do identify some limitations of this study. For example, we are only able to make six comparisons between biomass and water quality metrics because that is how many times biomass was sampled. In addition, lost data for DO in summer 2019 and 2020 at Van Giesen did not allow a complete analysis at this site. Finally, there is substantial variability in the water stargrass biomass data, so reducing the 10 samples to a single value (mean or median) may not capture the complexity of the relations between water quality and biomass. Water stargrass beds can influence DO on a local scale (Pelly, 2020; Pelly and others, 2024); however, similar to Pelly (2020), when we examine water quality of the river, the effects from water stargrass are difficult to pinpoint because the water in the river is well mixed.

Stream metabolism estimates at Kiona and Van Giesen showed that GPP and ER varied throughout the year: in winter and fall, stream reaches were heterotrophic (GPP < ER) and in summer, reaches were autotrophic (GPP > ER). A stream that is autotrophic is dominated by primary productivity and can produce enough energy (carbon) that outside carbon sources, such as leaf litter, are not needed to support the system. Heterotrophic systems are dominated by respiration and require external inputs of organic matter for energy. The GPP values at Kiona were high compared to other studies (Bernot and others, 2010; Hoellein and others, 2013; Hall and others, 2016; Munn and others, 2020), but comparable to some urban impacted streams in

New England (Izbicki and Morrison, 2021). Overall, the frequency of autotrophy in the lower Yakima is unusual because most streams are heterotrophic (Hoellein and others, 2013); however, most studies do not include metabolism estimates beyond a few weeks at a time (Munn and others, 2020), whereas here, we provide daily estimates for the duration of the summer season at Kiona.

There were no significant relations between the 7-day and 28-day average GPP and median water stargrass biomass. In contrast, GPP at Kiona showed significant relations with several water quality metrics including maximum temperature, DO minimum and range, and pH range. The fact that GPP was correlated to several water quality metrics but not related to water stargrass biomass is understandable because GPP is a more inclusive measure of whole stream metabolism that incorporates the metabolic activity of all plants, algae, biofilms, fish, macroinvertebrates, and microbes. For example, Munn and others (2020) showed that GPP and minimum DO were correlated with several fish metrics in small streams throughout the United States. In another national synthesis, Bernot and others (2010) showed that GPP was correlated with open channels with high light, nutrient inputs, and proximal urban and agricultural land use. Second, water stargrass may be indirectly affecting water quality dynamics. Macrophytes can provide increased surface area for the growth of biofilms and epiphytic algae (Ray and others, 2014; Mebane and others 2021). Therefore, the whole community, which includes water stargrass, is likely contributing to changes observed in water quality; however, teasing out water stargrass and its contribution is complex; additional studies would be needed to further describe the contributions of water stargrass.

Summary

This study provided a detailed examination of water quality and water stargrass (grass-leaf mud-plantain, *Heteranthera dubia*) biomass on the lower Yakima River from June 2018 through October 2020 at three monitoring locations. Frequent instances of not meeting established water quality criteria are observed in the summer for temperature, dissolved oxygen (DO), and pH across all monitoring stations. The main purpose of this study was to document if water stargrass was contributing to these detrimental water quality conditions. Comparisons between water stargrass biomass measurements and the continuous water quality data were made to investigate this question. However, when comparing several water quality metrics specifically to water stargrass biomass, the results were inconsistent. At Prosser (U.S. Geological Survey station 12509489), daily maximum pH and daily pH range were correlated with water stargrass biomass. At Kiona (U.S. Geological Survey station 12510500), water stargrass biomass was correlated to mean daily nitrate concentrations. Finally, at Van Giesen (U.S. Geological Survey station 12511800), there were no significant relations between water quality and water stargrass biomass.

There were several limitations in our correlation analysis. For example, the sample size was small, with at most six water stargrass biomass estimates at each monitoring location. Lost data, especially in summer at Van Giesen, reduced the power of our analyses. Finally, the water stargrass biomass estimates were highly spatially variable, and reducing the data from 10 samples to a single mean or median might be masking some of the observations in the continuous water quality data. However, when we examined the whole biological community by calculating whole-stream metabolism, we observed clearer relationships with water stargrass biomass. For example, estimates of GPP were correlated to metrics of temperature, DO, and pH,

indicating the whole river community is influencing surface water quality to some extent. The continuous data clearly show the lower Yakima River experiences large daily swings in DO and pH in summer, which is indicative of photosynthesis and respiration within the river. However, identifying the contribution from water stargrass is challenging, and additional studies would be needed to further understand the contribution of water stargrass.

Additional water stargrass biomass sampling (both spatially and temporally) may help to better understand the complex interactions between water stargrass growth and water quality. At a minimum, sampling water stargrass biomass near the end of the growing season (mid to late August) at the continuous water quality monitoring sites could help estimate the total seasonal growth of water stargrass. In addition, monitoring before and after targeted water stargrass removal and its effects on local water quality could provide insight on the complex relationships between growth and water quality. Finally, additional studies could help improve understanding of the effects of streamflow on water stargrass. For example, if spring high streamflows impair or delay the growth of water stargrass, then targeted streamflow releases from reservoirs in the upper watershed could be used to help impede water stargrass growth during the summer growing season.

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