Powder Basin Water Quality Monitoring Report

STATUS AND TRENDS: 2022-2024




POWDER BASIN WATERSHED COUNCIL
2034 AUBURN AVENUE, SUITE B
BAKER CITY, OR 97814

FINAL REPORT PREPARED BY:

Justin Thorson

Water Quality Monitoring Coordinator

Powder Basin Watershed Council

WQ-Monitoring@pbwsc.org

With support from

Oregon Watershed Enhancement Board

Oregon Department of Environmental Quality

Wallowa-Whitman National Forest

Burnt River Irrigation District

And

Powder Basin Watershed Council Volunteers

May 2026

Table of Contents

Table of Figures

Table of Maps

Acknowledgements

The Powder Basin Watershed Councils long-term water quality monitoring program was funded through grants from the Oregon Watershed Enhancement Board and support from the Oregon Department of Environmental Quality, Wallowa Whitman National Forest, and the Burnt River Irrigation District.

The Council would like to thank all of the volunteers who helped with the collection and review of water quality data listed below:

Ben Titus	Kathleen Baggett	Pam Conely	Tonia Humbert
John Runyon	Margaret Durner	Suzanne Fouty	Troy Tubbs
Karen Riener	Mike Beaty	Debi Lorence	

In addition to our volunteers, we would like to thank those who donated funds to replace older temperature loggers listed below:

Karen Reiner	Randy Guyer	Krystal Irby	Amanda Carroll
Frances Vaughan	Michael Beaty	Carolyn Kulog	
Steve Brink	Dmitri Videgar		

The activities and data included in this report would not be possible without the planning and contributions from the following individuals:

Bob Hassmiller, Wallowa-Whitman National Forest
Keegan Dutto, Wallowa-Whitman National Forest
Shawn Klaus, Burnt River Irrigation District
Nick Haxton-Evans, Oregon Department of Environmental Quality
Tim Baily, Powder Basin Watershed Council

Finally, we would like to thank the private landowners who graciously provided access to collect water quality samples, including Charles and Diane Erickson, City of Huntington, Cliff Wilmarth, Eagle Creek Orchard, Idaho Powder, the Powder Valley Water Control District, and Mark Schacht

Executive Summary

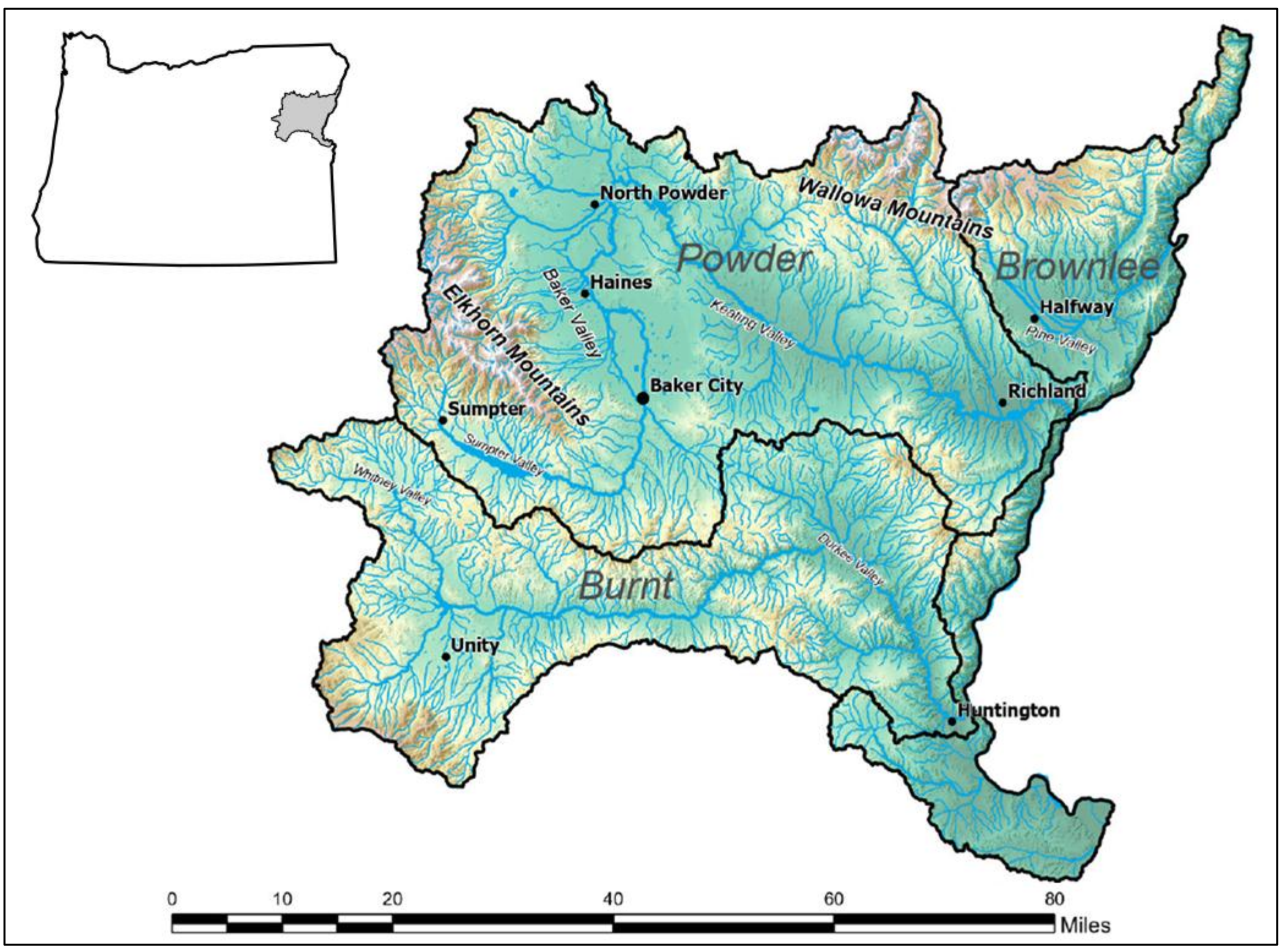
The Powder Basin Watershed Council's Long-Term Water Quality Monitoring Program has been a critical component of efforts to understand the status and trends in water quality throughout the basin and engage the community with the watershed since its inception in 2013. This report contains data and analysis from the Council's most recent iteration of the monitoring program between 2022 and 2024, including the collection of stream temperature, specific conductivity, pH, dissolved oxygen, and turbidity samples at 55 sites. The 2022-2024 monitoring also included several new activities, including the collection of bacteria and nutrient samples in the Burnt River sub-basin to identify issues with *E. coli* contamination and nutrient enrichment, along with the installation of dissolved oxygen loggers to monitor Redband and Bull trout spawning conditions. In addition to tracking the status and trends in individual water quality parameters, this report also details the use of a Water Quality Index (WQI) that combines multiple parameters into one metric to more easily track the status and trends in water quality at sites in the basin for the 2022-2024 period and since 2013.

The results of this monitoring found water quality issues throughout the basin, primarily in the lower reaches of the Powder River, Burnt River, and Pine Creek, along with several of their major tributaries. Warm stream temperatures were the most common form of water quality impairments and were found throughout the basin. High pH, likely related to nutrient enrichment, was also common in most larger streams. Overall water quality was highest in headwater streams and during the fall and was lowest in the summer and in larger streams, primarily due to high pH and stream temperatures. The 2022-24 monitoring was useful in identifying when, where, and which impairments were present among our current monitoring sites. However, important questions remain regarding sources of impairments, the impact of the PBWC's stream restoration projects on water quality, and the patterns of stream temperature and overall water quality for Redband and Bull Trout in the summer in headwater streams.

Introduction

The Powder Basin watershed covers 3,109 mi^2 in northeastern Oregon, primarily in Baker County, along with small sections of Union, Wallowa, and Malheur counties. The basin is composed of three major sub-basins; the Powder Sub-basin, the Burnt Sub-basin, and the Brownlee Sub-basin. The basin is home to approximately 17,000 people, with Baker City (pop. 10,135) being the largest city in the region, along with numerous smaller towns such as Haines, Unity, Richland, Halfway, Sumpter, and Huntington. Elevations range from over 9,000 feet in the upper portions of the Elkhorn and Wallowa mountains, to 1,700 feet at the Snake River/Pine Creek confluence. Primary economic activities include agriculture, mining, tourism, outdoor recreation, and transportation services along the Interstate 84 corridor. Significant portions of the upper basin as well as some of the foothill regions are managed by federal, state, and county agencies, primarily the Wallowa-Whitman National Forest, Bureau of Land Management, and Oregon Department of Fish and Wildlife. Private landownership predominates in the lower basin, particularly in Baker Valley, Keating Valley, Durkee Valley, and Pine Valley, where grazing and agriculture are the dominant land uses.

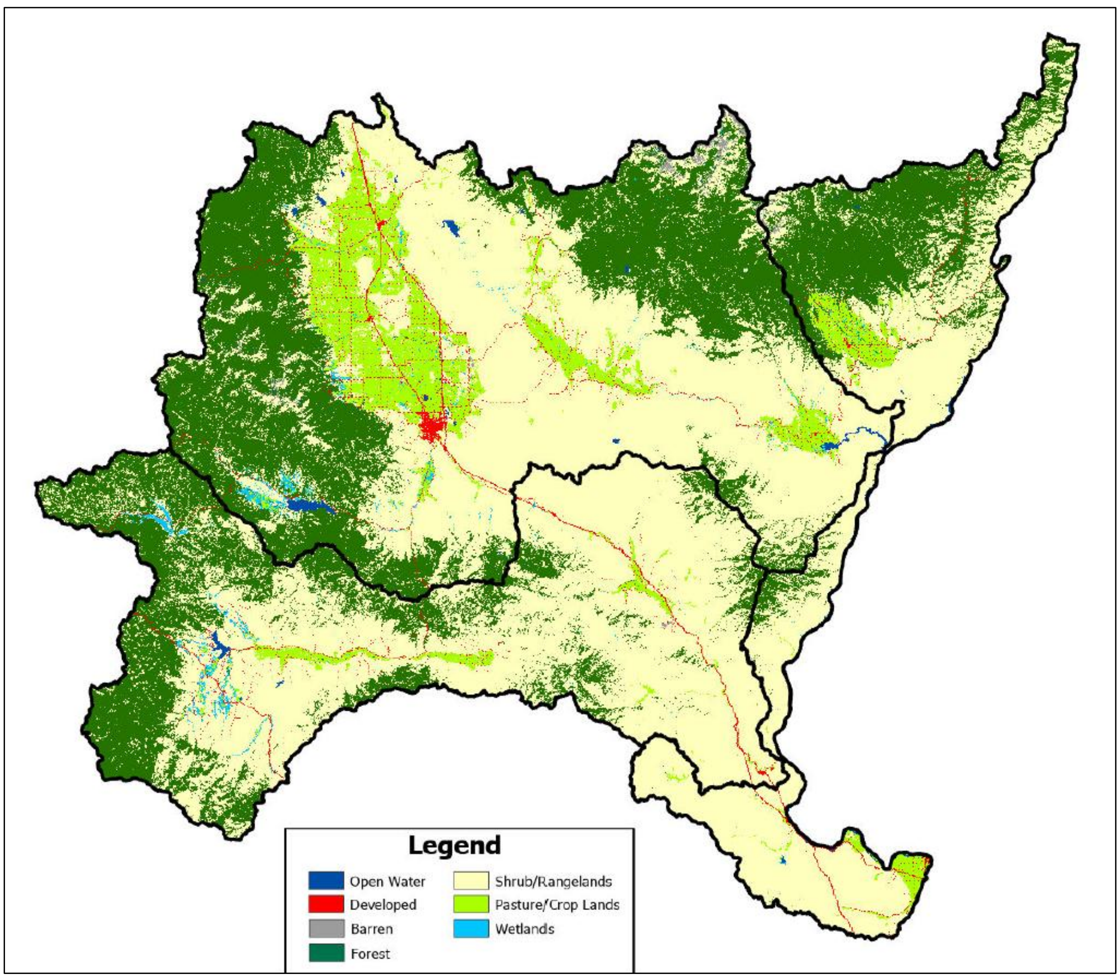
Map 1. The Powder Basin watershed, with major geographical features highlighted. Sub-basin boundaries outlined in black.



Climate in the basin is mainly cold semi-arid or temperate continental with warm dry summers and cool wet winters. Precipitation varies significantly throughout the basin, with over 40” in the higher elevations of the Elkhorn and Wallowa Mountains, to 10” or less per year in lower elevation areas such as the Baker Valley, with most precipitation falling between November and March. There also is significant spatial variation in the types of precipitation, with snow dominating higher elevations in winter, while lower elevations are dominated by rain/snow mix in the winter and rain in the spring (**NRCS 2006c**).

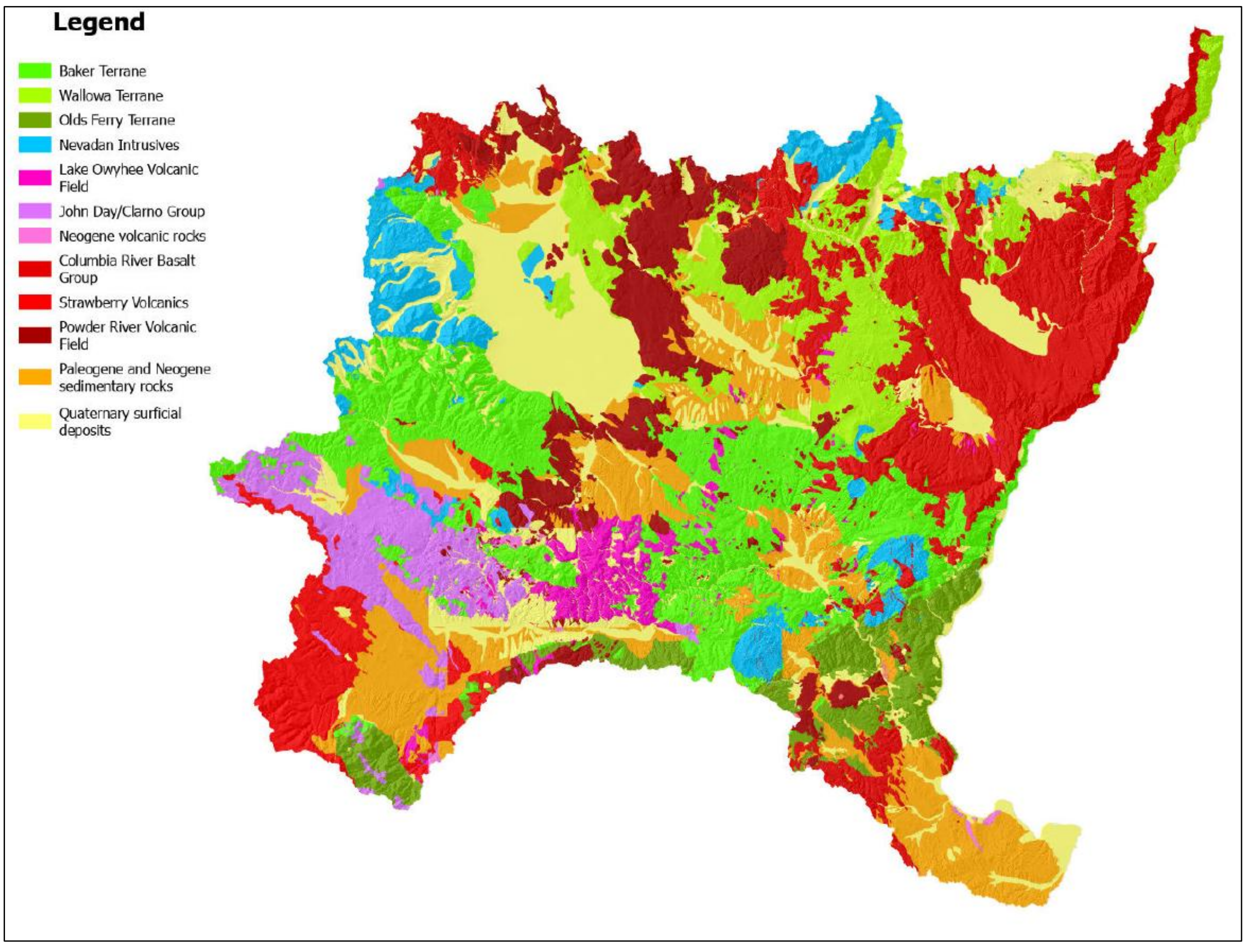
The variation in temperature and precipitation has a large impact on ecosystem types within the basin. In the lower elevation foothills, habitats are primarily composed of bunchgrass and sagebrush steppe with some Western Juniper present. Low elevation forests are mostly dominated by Ponderosa Pine in drier areas, with mixed conifer forests including Grand Fir, Lodgepole Pine, and Douglas Fir in cooler and wetter regions. High elevation forests found above 7000 feet in the Elkhorn and Wallowa Mountains are composed of Subalpine Fir and Englemann Spruce, with the highest peaks containing alpine meadow ecosystems (**NPCC 2004b**). Depositional valleys throughout the basin used to be home to significant amounts of wetland habitat but have been heavily altered from channel alteration and other human activities.

Map 2. Powder Basin watershed land cover. From the National Land Cover Database (USGS 2024).



Underlying geology plays a large role in the geography and hydrology of the region. Underlying rocks found in the Wallowa and Elkhorn Mountains and lower Burnt River watershed are primarily composed of shale, limestone, and metamorphosed rocks that were part of ancient island chains and associated sea floor known as exotic terranes. As these terranes were accreted to North American plate, they resulted in significant volcanism. The granitic batholiths found in portions of the Elkhorn Mountains, Wallowa Mountains, and the Coyote Hills in the central part of Baker Valley are the remnants of lava chambers from the earliest period of this volcanism (**Orr and Orr 1996**). Further volcanism between 50 and 15 million years ago resulted in rhyolitic and basalt formations common throughout the basin. Faulting and uplift associated with the continued subduction of the Juan De Fuca Plate resulted in the creation of the depositional valleys common throughout the basin, with glaciation, fluvial (river), and lake processes filling these valleys with thousands of feet of unconsolidated gravels, sand, and other sediments over the course of millions of years (**Ferns, Streck, and McClaughry 2017**). Groundwater sources in the upper basin tend to be driven by snowmelt and other precipitation, while those in the depositional valleys tend to be driven more by stream flow and groundwater flow from upland sources (**PBWC 2025, Gingerich et al. 2024**).

*Map 3. Bedrock geology of the Powder Basin. Older Terrane/Sedimentary rocks (green), Granitic batholith rocks (blue), rhyolitic volcanic rocks (pink), basalt volcanic rocks (red), recent sedimentary rocks (orange), and recent surficial deposits (yellow). Data from Oregon Geologic Data Compilation, release 7 (OGDC-7) (**Franczyk et al. 2023**)*



Stream hydrology in the Powder Basin is primarily driven by snow accumulation in the higher elevation sections of the Elkhorn and Wallowa Mountains. Streamflow for most streams peaks between March and June when snow melt is highest, and gradually decrease through the summer, with the lowest flows generally found in the late summer and early fall. The impacts of snow melt on stream hydrology differs depending on location, with flows in headwater and middle elevation streams more heavily influenced by spring snow melt than lower elevations streams. These lower elevation streams and rivers tend to show greater sensitivity to rainfall, with higher winter flows and lower summer flows than those influenced by snowmelt. Historically, discharge patterns in many of these streams and rivers were moderated by groundwater interactions with the floodplain, but channel straightening, a reduction in woody vegetation, beaver removal, and erosion have resulted in more “flashy” stream hydrology in these reaches (**Dwire et al. 2018**).

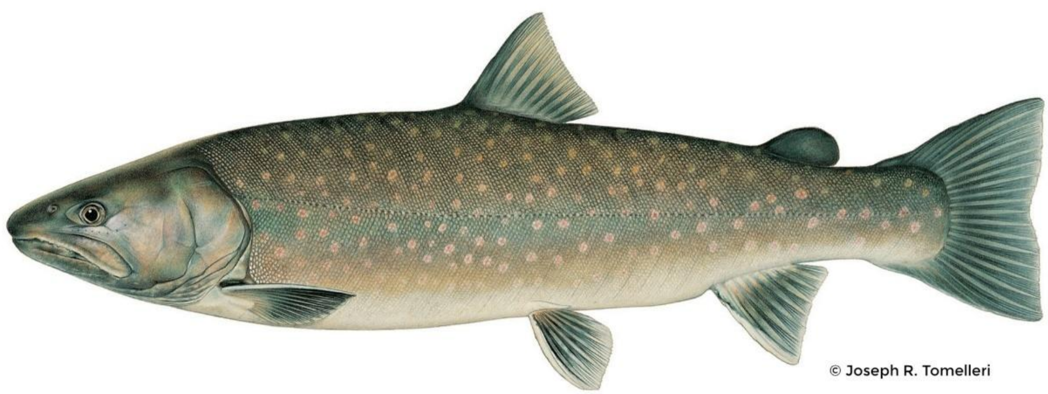
The Powder Basin used to be home to several salmon species, including Chinook Salmon, Coho Salmon, and Steelhead. The construction of the Thief Valley dam in 1937 blocked passage to the upper basin to these species, with the completion of the Snake River Hells Canyon dam complex in 1967 blocking passage to the entire basin for anadromous fish populations (**Buchanan, Hanson, and Hooton 1997, NPCC 2004b**). The basin is currently home to two native salmonid species: Bull Trout (*Salvelinus confluentus*) and Redband Trout (*Oncorhynchus mykiss gairdneri*).

Redband trout in the basin primarily exhibit migratory fluvial and resident life history strategies, with fluvial populations found in larger streams in the fall and winter and migrating to smaller streams for thermal refuge during the summer. Resident populations are found year-round in small and middle sized streams where temperatures and oxygen concentrations are suitable. Redband trout primarily spawn during the spring, with spawning occurring earlier in March in lower elevation streams and going until June in higher elevation and snowmelt dominated systems. Redband trout can tolerate temperatures up to 26 °C, but temperatures above 20 °C generally result in significantly higher mortality and stress, along with lower growth and reproductive success (**NPCC 2004b, NPCC 2004c**). While Redband trout are widely distributed throughout the Powder Basin, populations have seen significant declines in both range and reach utilizations compared to historical conditions due to warm streams temperatures, migration barriers, and competition with non-native species such as Brook Trout (**NPCC 2004b, Miller et al. 2014**).

Figure 1. Redband Trout (O. mykiss gairdneri) are widespread throughout the Powder Basin watershed

Bull Trout populations in the Powder Basin are much more constrained than redband trout and are currently listed as Threatened under the Endangered Species Act. Populations historically had significant fluvial life histories but now are mostly constrained to resident populations within these headwater streams. Bull Trout are also significantly more constrained by warm stream temperatures, preferring streams below 12 °C with temperatures above 15 °C associated with higher mortality and stress (**USFS 2015**). Bull Trout spawn in the fall low flow season in headwater streams. Populations in the Powder Basin are currently threatened by warm stream temperatures, genetic isolation, as well as hybridization and competition with non-native Brook Trout, which often occupy similar stream reaches (**Buchanan, Hanson, and Hooton 1997, NPCC 2004a, NPCC 2004b**).

Figure 2. Bull Trout (Salvelinus confluentus) are found in headwater streams in the Power and Pine Creek watersheds.



The Powder Basin contains over 2,860 miles of perennial streams, with 604 of these stream miles listed as 303(d) impaired by the Oregon Department of Environmental Quality (DEQ). Major water quality impairments include 581.8 miles impaired for temperature, 69 miles impaired for dissolved oxygen, 127.1 miles impaired by fecal coliform bacteria, 24.6 miles impaired for turbidity and sedimentation, and 32.2 miles impaired for chlorophyll a (**NRCS 2006a, NRCS 2006b, NRCS 2006c**). Water quality monitoring by the PBWC from 2014 to 2018 found similar widespread impairments in the basin, particularly high temperatures exceeding 12 °C in Bull Trout streams and 20 °C in redband trout streams throughout all sub-basins, including streams in both the upper and lower watersheds. Issues with pH, dissolved oxygen, and turbidity were also noted, but were less common and mostly occurred in the lower watershed (**PBWC 2018**).

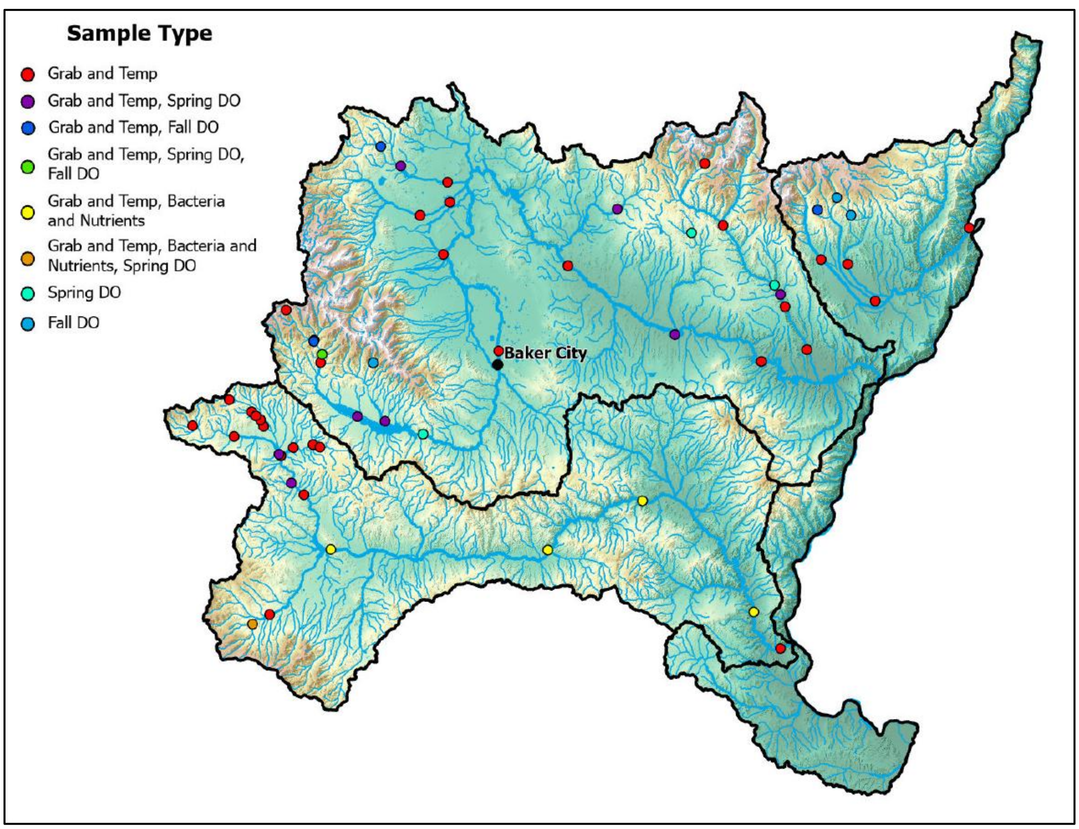
Climate change is expected to alter hydrology of streams throughout the Powder Basin over the next 20-60 years. These changes include greater variability in overall precipitation, shifts from snow dominated to more rain dominated precipitation patterns, lower summer precipitation, and warmer temperatures throughout the summer, resulting in higher winter stream flows, lower summer stream flows, and higher stream temperatures (**Dwire et al. 2018**). Many of these changes have already materialized, with a 1.46 °C increase in observed air temperature between 1977 and 2011, a 36.6% decrease in snowpack volume from SnoTel monitoring stations in or near the basin between 1950 and 2000, a 20.1% decrease in summer stream flows between 1977 and 2011, and a 0.58 °C increase in stream temperatures between 1977 and 2011 (**OWEB 2023, Issak et al.2017, EPA 2023**). Changes in stream temperatures are expected to continue under expected warming trends, with mean August stream temperatures projected to increase by 1.198 °C by 2040, and 1.956 °C by 2080 over the 1993 to 2011 period (**Isaak et al. 2017**). These changes will negatively impact water quality in the basin through greater thermal stress on fish and other aquatic organisms, higher algal productivity, increased sedimentation, and changes to water chemistry.

Background on the Powder Basin Long-Term Water Quality Monitoring Program

The Powder Basin Watershed Council's (PBWC) long-term water quality monitoring program began in 2013 as a collaboration with the Wallowa-Whitman National Forest (WWNF), the Bureau of Land Management's Baker Field Office, and the Oregon Department of Environmental Quality (DEQ), with funding primarily coming from the Oregon Watershed Enhancement Board (OWEB). The primary objective of this monitoring was to identify water quality impairments within the watershed and track trends in water quality over time. Since then, the program has sampled 115 sites located throughout the basin from headwater streams to the mainstems of the Powder and Burnt Rivers.

The most recent iteration of the Long-term Water Quality Monitoring program was funded between 2022 and 2024. After a hiatus in funding during 2021, the program was restarted in summer of 2022, sampling a total of 55 sites, with 49 of these sampled monthly between May and November using a similar grab sample protocol to previous efforts. In anticipation of Total Maximum Daily Load (TMDL) implementation efforts for bacteria and nutrients, the PBWC included the collection of bacteria and nutrient samples at five sites in the Burnt River sub-basin. The most recent monitoring efforts also included dissolved oxygen monitoring using oxygen data loggers to assess spawning conditions at 13 sites in the spring for redband trout and seven sites in the fall for Bull Trout.

Map 4. Locations of PBWC monitoring sites for the 2022-2024 period. Color represents the types of monitoring conducted at the site.



Flow and climate conditions varied significantly throughout the monitoring timeframe. Overall, conditions in 2022 were defined by a mild winter with a pulse of snowfall during the early spring (~April) followed by hot and dry conditions during the summer and into the early fall (Figure 3). Conditions were much wetter during 2023, with drought dissipating during the winter due to high levels of snowpack, followed by warm and wet summer conditions characterized by frequent afternoon thunderstorms (Figure 4). Conditions returned to mostly dry weather in 2024, with moderate snowpack during the winter, while the summer was noted for particularly hot and dry weather (Figure 5).

Figure 3. Drought conditions for Oregon from March, July and October 2022. Data from US Drought Monitor

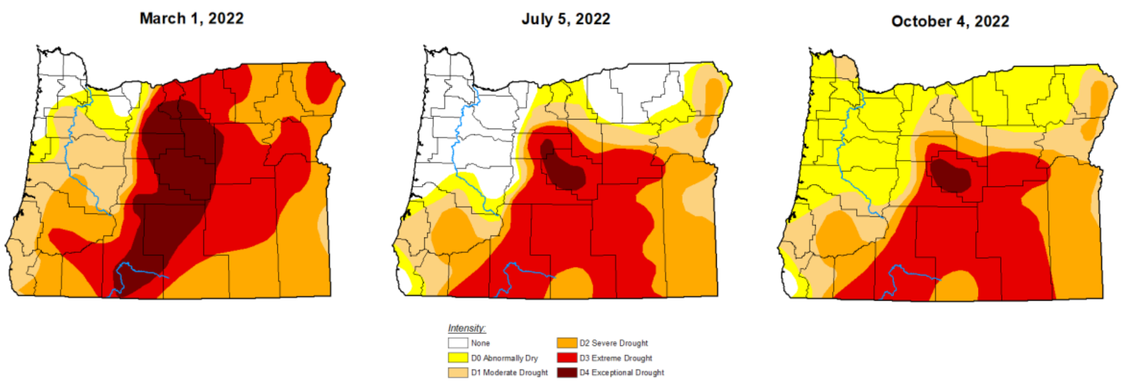


Figure 4. Drought conditions for Oregon from March, July and October 2023. Data from US Drought Monitor

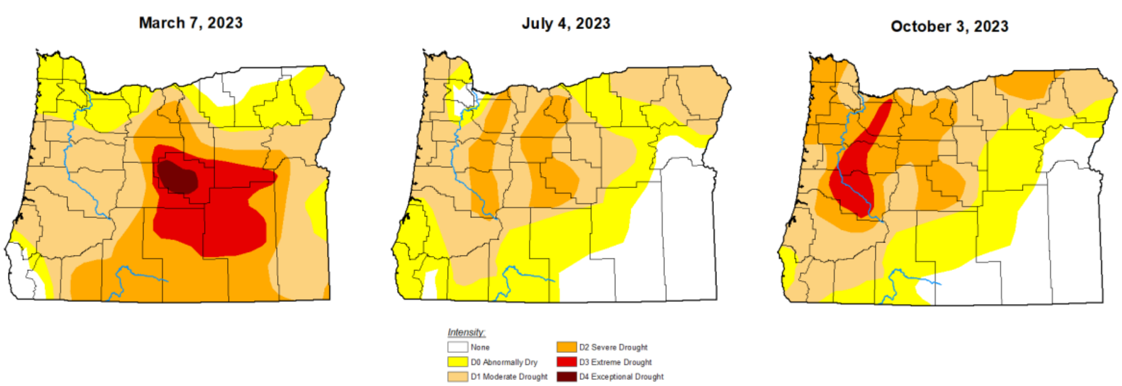
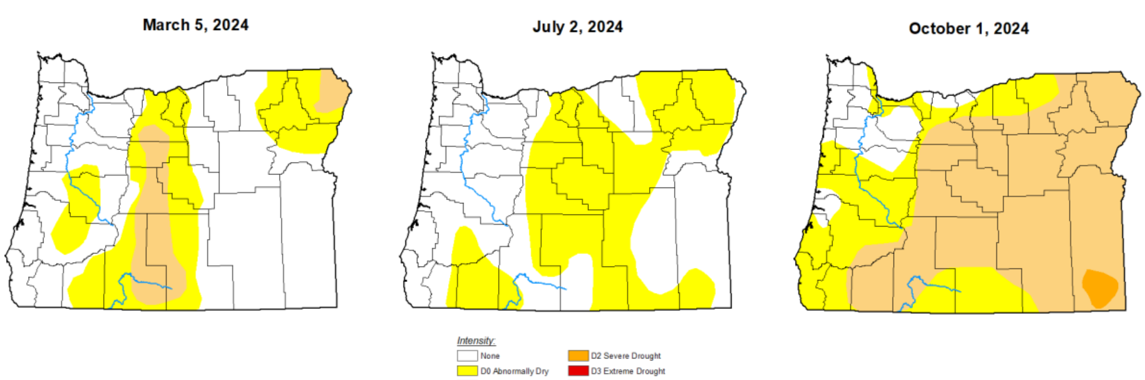


Figure 5. Drought conditions for Oregon from March, July and October 2024. Data from US Drought Monitor



Sampling in 2022 came across several issues related to restarting the program after a year and a half hiatus, with a later start in June reducing data coverage compared to the subsequent years. Equipment issues were also persistent, especially during the first year of sampling. Older equipment, particularly temperature loggers and pH probes, had significant impacts on data quality and coverage. By 2024, these issues had mostly been addressed, with new temperature loggers replacing older ones with dead batteries, while new pH probes dramatically improved the accuracy and precision of sampling.

Volunteers and partnerships with other organizations have always been an important component of the PBWC's monitoring activities in the basin. DEQ provided both equipment and training for PBWC staff and volunteers and was instrumental in developing and approving the required quality assurance procedures. Volunteers were particularly important in restarting the program in 2022, and provided critical support and training, as well improving coverage for monitoring in watersheds far from the PBWC's office in Baker City. As part of the PBWC's bacteria and nutrient monitoring efforts in the Burnt River sub-basin, the Council partnered with the Burnt River Irrigation District (BRID), which greatly expanded the coverage of sampling efforts and assisted with funding for sample collection and analysis. The PBWC also partnered with the WWNF for temperature monitoring at sites in the North Fork Burnt River, including pre-project monitoring at nine sites located within Camp and Trout Creeks where restoration projects were planned and implemented.

Monitoring Parameters

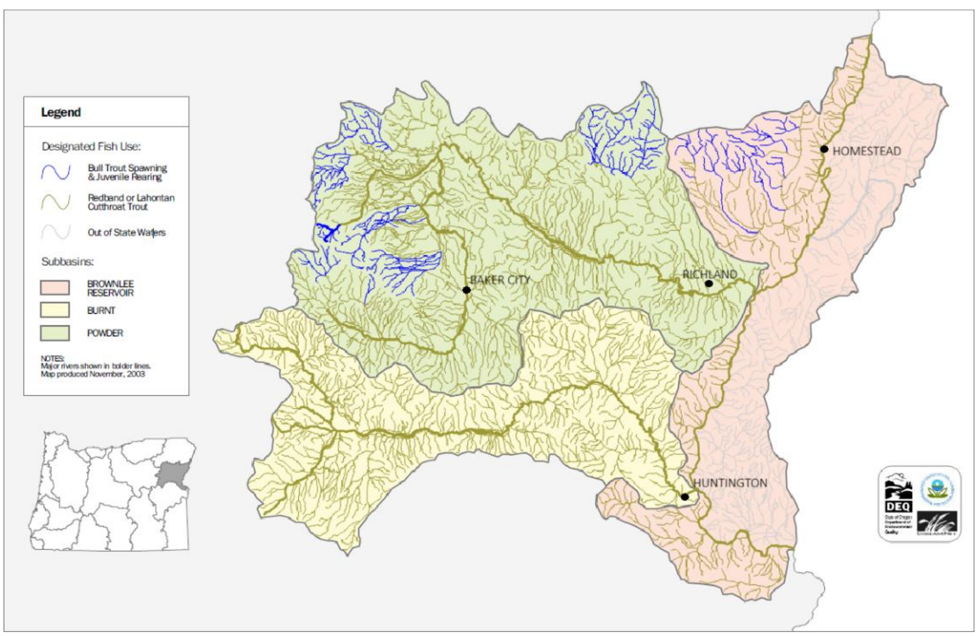
The PBWC's monitoring efforts included four distinct types of monitoring: Grab sample monitoring of dissolved oxygen, pH, conductivity, and turbidity; Continuous temperature monitoring using temperature data loggers; Continuous dissolved oxygen monitoring using oxygen data loggers; and bacteria and nutrient monitoring. The following section details the background of these parameters.

Temperature

Water temperature plays an important role in determining the productivity and diversity of organisms within stream ecosystems, as well as being a primary determinant of other parameters such as dissolved oxygen, pH, conductivity, nutrients, and bacteria (**Poole and Berman 2001**). Salmonids in particular are sensitive to warm stream temperatures, with most species adapted to life in cool and cold-water stream systems. Impacts from warm stream temperatures on salmonids include stress, reduced growth, lower reproductive success, and mortality (**Feldhaus 2006, Lee and Rinne 1980, Sullivan et al. 2000**).

Temperature standards set by DEQ are designed to be protective of salmonids and are separated into two zones based on location within the watershed and historical fish use, with the cool-water standard for streams found in middle to lower elevation reach inhabited by redband trout, while the cold-water standard applies to higher elevation areas and where with current or historical Bull Trout presence (Figure 6). The criteria for the temperature standards are set based on the seven-day average maximum temperature, with the standard for cool-water streams set at 20 °C and the cold-water standard set at 12 °C (**DEQ 2008**).

Figure 6. Oregon DEQ Fish use designations for the Powder Basin, with cool-water (redband trout) stream in yellow and cold-water (Bull Trout) streams in blue. Modified from Oregon Designated Fish Use map 260A



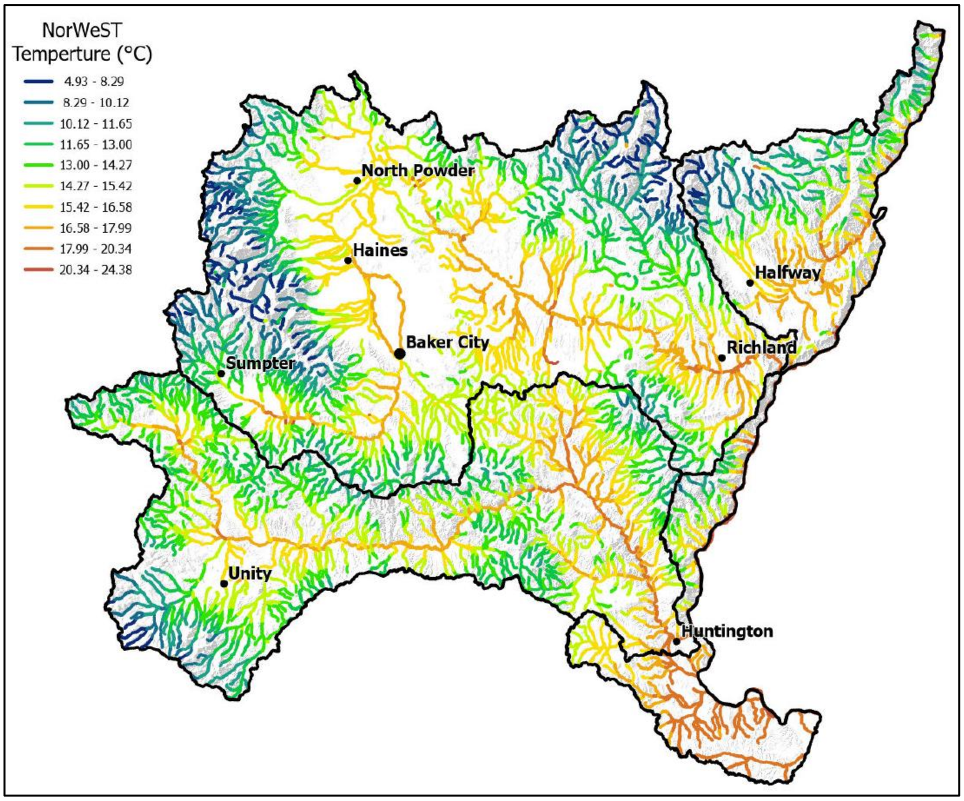
Incoming short-wave solar radiation is the dominant source of energy in streams, particularly in large streams and rivers. Factors like shading from topography and vegetation play an important role in limiting the accumulation of energy by blocking incoming solar radiation. Air temperature also results in energy transfer to streams when the latent heat of the atmosphere is higher than that in stream water, as is the case during the day and in the summer (**Leach et al. 2023**). The source of precipitation impacts stream temperature as well, particularly for streams where snowmelt is a major source of flow. The longer duration of snowmelt, generally from mid-spring through early summer, results in cooler temperatures in streams than those where runoff from rain is the more dominant flow source. The impact of snowmelt is particularly important in mountain stream reaches, where it buffers stream temperatures from air temperature changes.

Groundwater contributions to streams are another factor influencing stream temperatures, with notable shifts in daily and seasonal temperature patterns compared to streams that derive more of their flow from surface water sources. Because groundwater is mostly isolated from solar radiation and atmospheric temperature changes, the temperature patterns for these sources tend to be more muted, with smaller seasonal and daily temperature changes (**Pool and Berman 2001, Rey et al. 2024**). These impacts are most prevalent in the summer, with mean stream temperatures up to 1.4 °C lower in streams where groundwater contributions to surface flow are large (**Mousumi et al. 2025**).

The impact of these factors differs depending on stream size, location within the watershed, and type of dominant precipitation. Headwater streams tend to be more heavily impacted by snowmelt, groundwater, and vegetation shading, and generally have cooler stream temperatures and smaller seasonal temperature variation. In contrast, larger streams and rivers are more heavily influenced by surface flows and have lower levels of vegetation shading, resulting in warmer stream temperatures and greater variability in seasonal stream temperatures (**Isaak et al. 2020, Poole and Berman 2001**).

Several efforts have been made to model stream temperatures on a landscape scale, one of the most well-known of these efforts being the NorWeST Stream temperature model (**Issak et al. 2017**), which uses landscape factors, air temperatures, and precipitation to estimate stream temperatures. The primary output of the NorWeST model are preditions of mean August stream temperatures over the 1993-2011 period (Figure 7). The NorWeST model also includes predictions for stream temperatures for 2040 and 2080 time period based on expected warming and precipitation trends from climate models. Both of these outputs have been useful tools for fisheries and water resource managers to identify potential thermal refuge habitat, assess scenarios for fish conservation efforts, and predict fish size (**John et al. 2026, Winkowski et al. 2024, Al-Chokhachy et al 2022**).

Figure 7. Mean August stream temperatures (in °C) over the 1993-2011 for the Powder Basin from the NorWeST Stream temperature model. (data from Issak et al. 2017).



While the NorWeST model is a useful tool, there are some issues that reduce the utility of its comparisons to observed temperature data. One of the largest issues is that the spatial resolution for model predictions occur over ~1 km stream segments, making it less useful for comparing fine scale stream temperature patterns. The error ranges of the model predictions are another important consideration, with a 1.10 °C Root Mean Squared Percentage Error (a measure of error biased by outliers) and 0.72 °C Mean Absolute Percentage Error (a measure of error that is less biased by outliers). Although these error ranges are adequate when assessing spatial trends in stream temperature, they might obscure smaller temporal scale trends such as variation between years or time periods. Also, because the predictions for NorWeST are based on averages taken over 18-20 year intervals, the smaller 2022-24 timeframe might make inferences between observed and predicted temperatures less comparable. Still, despite its shortcomings, the NorWeST model provides a useful metric to assess trends in observed temperatures for sites where long-term temperature data is lacking.

Dissolved Oxygen

Dissolved oxygen within water plays an important role in determining the suitability of habitats for fish and other aquatic species and is measured in milligrams per liter (mg/L). In addition to concentration, oxygen can also be measured by percent saturation, which is determined by the relative measure of the concentration of oxygen compared to the theoretical maximum given the temperature and air pressure. Water with oxygen saturation above 100% of the theoretical is defined as being super saturated, while water with oxygen saturation under 100% is defined as being under saturated.

For cool water fish species such as redband trout, oxygen concentrations generally need to be above 6.5 mg/L, while more sensitive cold water species such as Bull Trout require higher oxygen concentrations, generally above 8 mg/L. Spawning in both species also requires oxygen concentrations, with DEQ requiring minimum intergravel oxygen concentration of 8 mg/L and concentrations above 11 mg/L seen as ideal to help the developing eggs and fry within spawning gravels (**DEQ 1996a**). Low levels of oxygen can result in higher stress, reduced feeding, lower growth, reduced reproductive success, and mortality in fish (**Wu 2002**). While high oxygen concentrations are generally associated with higher water quality, large changes in oxygen concentrations and saturation related to algal photosynthesis can create issues for fish, especially related gas bubble disease (**EPA 2025**).

Oxygen concentrations in water are heavily related to stream temperature and elevation/air pressure, with higher air pressures and lower temperatures associated with higher oxygen concentrations. Oxygen concentrations are also heavily influenced by stream roughness and gradient, with steeper and more complex streams with larger substrate and higher amounts of woody debris having higher oxygen concentrations, whereas slow or still water systems have lower levels of turbulence and therefore lower overall oxygen concentrations (**Abdul-Aziz and Gebreslase 2023**). While groundwater oxygen concentrations are generally low due to respiration and separation from atmospheric sources, the cooler temperatures can raise oxygen saturation in downstream reaches in streams with higher groundwater contributions (**EPA 2025**).

pH

pH is a measure of the acidity or alkalinity of a solution and is recorded on a scale from 0 to 14, with measurements over 7 indicating alkaline conditions, measurements under 7 indicating acidic conditions, with 7 indicating neutral conditions. Within streams in Eastern Oregon, pH is generally more alkaline due to the presence of magnesium, potassium, calcium, sodium, and other cations within the volcanic rocks common to the region (**Cude 2001**). Low pH, particularly below 5.5, is associated with lower biodiversity and the loss of important sport fish species in streams. Fish exposed to low pH generally see increased gill damage, decreased growth, reduced egg fertilization and development success, respiratory issues, and higher rates of mortality. Low pH is also associated with higher concentrations of heavy metals like arsenic and toxic compounds like cyanide. In contrast, high pH can result in lower growth and damage to skin and other organs in fish, along with indicating issues with eutrophication (See the section on phosphorus for more information) (**US EPA 2026**). DEQ rules for the Powder and Burnt sub-basins recommend that pH should be between 6.5 and 9 (**DEQ 2003**).

Conductivity

Specific conductivity is the measure of how well electrical current passes through water. It serves as a proxy for the amount of dissolved salts, minerals, and other ions in the water, and is measured in micro-Siemens per centimeter (µS/cm). While no standards for conductivity exist, in general, higher conductivity values are associated with more negative impacts to aquatic life (**Clements and Kotalik 2016**). Large variation in conductivity over seasonal timeframes can also indicate stream degradation from runoff or other pollution sources as well.

Upstream drainage basin size is positively correlated to conductivity measurements due to several factors, primarily concentration of upstream minerals from evaporation and erosion. Groundwater contributions to stream surface flow also have significant impacts on conductivity measurements in streams, with groundwater generally having higher conductivity measurements due to higher levels of dissolved minerals (**Brown, Milner, and Hannah 2007, Olson and Cormier 2019**). Similarly, bedrock geology has a large impact on conductivity related to the composition of non-silicate minerals in rocks, with higher conductivity measurements seen in streams draining carbonate rock basins compared to those with a higher percentage of sandstone or shale bedrock. While conductivity varies by large amounts naturally as a result of these factors and from climatic changes such as drought, anthropogenic sources can heavily impact it as well, with higher conductivity measurements in streams associated use of road salt, mining, and fertilizer use (**Kney and Brandes 2007**).

Turbidity

Turbidity is a metric used to assess the cloudiness of water and is measured in Nephelometric Turbidity Units (NTU's). In water quality monitoring, turbidity is often used as a proxy for the levels of suspended sediment in water which can indicate issues with erosion. Along with water quality and stream impacts, high levels of turbidity (>50 NTU's) are also associated with stress and gill damage in fish, particularly in juveniles. High levels of fine sediment can also smother developing fish eggs by filling pores in spawning gravels (**Bash, Berman, and Bolton 2001**). DEQ standards for turbidity do not mention discrete levels implying impairment but rather focus on limiting turbidity impacts from in-stream activities, requiring a less than ten percent increase in turbidity compared to an upstream reference point (**DEQ 1996b**). This report uses measurements above 20 NTU's to indicate moderate levels of turbidity, and above 50 NTU's to indicate high levels of turbidity, thresholds which are supported by studies on turbidity impacts on salmonids (**Bash, Berman, and Bolton 2001**).

Turbidity is generally positively related to drainage basin size, with larger streams having higher levels of turbidity compared to smaller streams (**Klein, Lewis, and Buffleben 2012**). Other factors, such as soil erodibility, particle size, and stream gradient, can influence turbidity measurements. Turbidity is also related to seasonal patterns in precipitation and stream flow, with higher turbidity seen during the spring period during periods of higher flow. Anthropogenic factors that can increase turbidity in streams include road building, timber harvest, tilling of agricultural fields, and runoff from urban areas (**Chen and Chang 2019, Klein, Lewis, and Buffleben 2012**). Other factors such as stream bank erosion happen naturally, but can be intensified by human impacts as well.

E. coli

Escherichia coli (*E. coli*) is a fecal coliform bacteria found in the intestines of warm-blooded organisms and is associated with manure, sewage, and urban runoff. While infections from most *E. coli* are harmless or mild, certain strains, particularly Shiga toxin-producing strains, can cause sever foodborne illness, including diarrhea, vomiting, and dehydration (**Mueller, Rausch-Phung, and Tainter 2025**). *E. coli* are also used as an indicator of broader fecal contamination and are often associated with other pathogens such as norovirus and enterococcus. *E. coli* is measured in colony forming units (cfu) per 100mL. Several standards for *E. coli* exist depending on the timeframe of the sampling, with a higher single sample standard of 406 cfu/100mL and lower 90-day geometric mean standard of 126 cfu/100mL (**DEQ 2016**).

E. coli can be widespread in stream systems, inhabiting both riparian and stream sediments, often after large precipitation events (**Wu, Reese, and Dorner 2011, Edge et al. 2021**). *E. coli* concentrations are generally positively correlated with percent agricultural and urban area. Point sources from wastewater treatment plants can also be a significant source of *E. coli*, although most modern plants have systems in place to significantly reduce *E. coli* concentrations from discharge. Decreases in *E. coli* concentration in streams are associated with sedimentation, with negative relationship between upstream drainage area and fecal bacteria levels seen in some watersheds. Other factors having negative correlations with *E. coli* concentrations include forested area and wetland area, mostly due to filtration and processing of fecal bacteria within soils in these ecosystems (**Brendel and Soupir 2017**).

Phosphorus

Phosphorus, is an important macronutrient, that, along with nitrogen, is responsible for much of the primary productivity within aquatic ecosystems. Differences in how these nutrients are sourced, particularly due to nitrogen fixation by cyanobacteria, along with longer water residence times mean that phosphorus is a more important nutrient in determining productivity within lake and reservoir ecosystems, while the shorter residence time of water in streams and rivers result in both nutrients having roughly equal importance in determining productivity (**Dodds and Smith 2016**). Concentrations of phosphorus vary considerably depending on location and time, but sampling in the Malheur River found concentration ranging from 0.86 to 0.43 mg/L, with mean concentrations in streams between 0.086 mg/L and 0.357 mg/L (**Shock et al. 2001**). Similarly, identifying thresholds for phosphorus concentrations that can create negative impacts is difficult, but studies of other inland northwest streams have found negative impacts at 0.126 mg/L (**Wise et al. 2009**)

While primary production in the form of algal biomass forms an important component of the base of aquatic food chains, excessive amounts of nutrients can create a condition known as eutrophication. Eutrophication is a major source of impairment in many inland waterways, with a variety of negative impacts on fish and other aquatic organisms. In streams, eutrophication results in excessive growth of periphyton (algal mats or rocks) and macrophytes (rooted aquatic vegetation), while algal blooms are more common in lakes and reservoirs (**Wise et al. 2009**). The uptake of carbonate within waters with eutrophic conditions increases water alkalinity and reduces the buffering capacity of streams to pH changes (**Wise et al. 2009, Smith et al 1999**). Excessive algal production also alters the dissolved oxygen profile in streams and lakes, resulting in supersaturation of oxygen during the day from photosynthesis. Metabolic oxygen demand during the night and from decomposition in stratified waters like lakes and reservoirs also can result in hypoxic (low oxygen) conditions within these systems (**Müller et al. 2012**). Algal blooms can also cause health issues in reservoirs due to the production of microcystin, an algal biotoxin associated with blue-green algae.

Results from landscape models indicate that large amounts of phosphorus within streams and rivers come from in-stream weathering processes such as bank and channel erosion. Other important sources, especially in eastern Oregon, come from the weathering of volcanic rock, which contain high concentrations in mineral form. While these natural sources of phosphorus are significant contributors to total phosphorus loads in aquatic ecosystems, large amounts of phosphorus also come from anthropogenic sources. In areas with significant agricultural activity, phosphorus primarily comes from fertilizers and manure that end up in streams and lakes through runoff. While more common to urban watersheds, phosphorus releases from sewage treatment plants are also an issue and can result in high levels of phosphorus in streams (**Wise and Johnson 2011**).

Methods and Study Design

Grab Sample Monitoring

Grab sample monitoring was used to measure levels of dissolved oxygen, pH, conductivity, and turbidity at sites. Orion A329 Multimeters were used to collect data on oxygen concentrations, pH, and conductivity, while Hach 2100Q turbidimeters were used to collect turbidity data. All grab sample equipment was recalibrated at the beginning of the field season using standards provided by DEQ. Dissolved oxygen calibrations were achieved using the saturated air method with temperature and pressure used to identify theoretical dissolved oxygen concentrations, pH probes were calibrated using 4, 7, and 10 pH standards. Conductivity probes were calibrated using 100 μS/cm and 1000 μS/cm standards. Turbidimeters were calibrated using 10, 20, 100, and 800 NTU standards. Accuracy checks were performed before and after each sampling event to assess equipment accuracy. Equipment falling outside of required accuracy standards from DEQ's Data Quality Matrix (see Appendix A) were recalibrated.

Figure 8. Grab sample monitoring near a monitoring site using the Orion A329 and Hach 2100Q Turbidimeter

Grab sample monitoring activities took place once per month between April and November at each site. Two samples were collected within the stream flow using 1 L bottles in a manner to avoid collection of disturbed sediment. Samples were taken to a nearby spot to measure the water quality parameters (Figure 8). Turbidity samples were collected first, gently to overturning the sample bottle to resuspend the sediment in the bottles before being dispensed into a 10 mL cuvette. One turbidity sample per bottle was collected, then measured three times, resuspending sediment within the cuvette before each measurement. The median of the three measurements was used as the record for the sample. Probes from the Orion multimeter were kept submerged in a third 1L bottle to keep them cool during sampling, and were transferred to the sample bottle when ready. DO concentration (in mg/L) and saturation (in %), pH, and conductivity (in µS/cm) were recorded once measurements stabilized to ambient temperatures and conditions. This processes was repeated for each of the two samples at each site. Differences between the samples for each parameter were used to assess precision using the grading methods provided by DEQ. All grab sample data was recorded on the grab sample collection form (Appendix B), along with any notes on stream conditions, weather, or other important information related to sampling.

After sampling was completed, all grab sample data from the collection forms was entered into an Excel spreadsheet. All data was checked by an independent observer to correct entry errors or typos and then entered into the PBWC's grab sample Access database for long-term storage. All grab sample collection data sheets were stored in a file cabinet for future reference.

Temperature Monitoring

Temperature monitoring was conducted using Onset U22 HOBO temperature data loggers. Calibration checks for temperature loggers took place before and after deployment using a NIST certified thermometer to assess logger accuracy. For the calibration check procedure, loggers were set to record at 1-minute intervals and submerged in a warm water bath for ten minutes followed by a cool water bath for another ten minutes. Loggers were required to fall within DEQ's 0.5 °C accuracy threshold of the NIST thermometer to be used in the field.

Figure 9. Example of a temperature logger placement at a site in the Upper Powder watershed. Arrow included to note the logger location.

Loggers were launched in the office using a 30-minute recording interval and were installed at each site during the first field visit of the season, generally April-June. The loggers were attached to sturdy stream bank objects such as tree roots, trunks, and boulders using coated wire and U-bolts. All loggers were placed in a location with continuous flow and covered in rocks to prevent becoming dislodged or removed. A 1½” PVC pipe was used to cover the logger and reduce impacts of solar radiation on stream temperature measurements (Figure 9). Logger serial number, depth (in cm), and temperature using a NIST certified thermometer were recorded alongside a description of the site, photos, and GPS coordinates to aid in future site visits using the PBWC Logger install/removal form (Appendix C).

Loggers were revisited during subsequent grab sample visits, with field audits taken using a NIST certified thermometer. Audits also included logger depth and condition. Loggers were placed back in stream flow when found near the water surface or exposed to dry air, with conditions and new locations noted on the grab sample collection form. At the end of the field season, temperature loggers were removed from the site and brought to the office for download. All logger data was checked using temperature profiles and recorded conditions, with data associated with dry conditions noted in the file. Temperature logger data was also compared to audits data to assess logger accuracy using the sample grade criteria from DEQ’s Data Quality Matrix.

Dissolved Oxygen Monitoring

Continuous dissolved oxygen monitoring was conducted using PME MiniDOT and Onset U26 oxygen data loggers. Calibration checks were completed on all loggers three times each year: Once before spring deployment, once during the summer after spring deployment but before fall deployment, and once after fall deployment. For the calibration check procedure, oxygen loggers were set to record at 1-minute intervals and submerged in a warm water bath for ten minutes followed by a cool water bath for another ten minutes. Oxygen calibration checks took place in the warm water bath with oxygen concentrations near saturation. Oxygen concentration from an Orion A329 multimeter and temperatures from a NIST certified thermometer were used to assess logger accuracy. All oxygen loggers were required to fall within DEQ’s accuracy thresholds of 0.3 °C for temperature to be used in the field. Several oxygen loggers fell outside of DEQ’s required oxygen accuracy thresholds of 0.3 mg/L in 2022 and 2023, but still fell within a broader accuracy threshold of 1 mg/L. The reason for the discrepancy in oxygen measurements by the loggers was identified as resulting from equipment drift, a problem addressed by software updates from PME.

Oxygen loggers were launched in the office on a 15-minute recording interval and were installed using a similar method to those for the temperature loggers. The MiniDOT loggers were placed in the stream without any cover, but the U26 logger were installed in 4” PVC pipe to protect them. Logger serial number, depth (in cm), and temperature using a NIST certified thermometer were recorded alongside a description of the site, photos, and GPS coordinates to aid in future site visits using the PBWC Logger Install/Removal form. Dissolved oxygen concentration, saturation, and atmospheric pressure (in mmHg) from grab sample measurements were also recorded to aid in logger accuracy checks.

Oxygen loggers were generally kept in the stream for 2-3 weeks to continuously record stream oxygen concentrations and temperatures. The same audit procedure was used during logger removals, with temperature, depth, condition, oxygen concentration, percent saturation, and atmospheric pressure recorded upon removal. Due to algal buildup and its impact on oxygen readings, the logger sensor was cleaned using a soft toothbrush after removal and the logger was placed in a bucket with nearby stream water. Another audit was taken within the bucket after 15 minutes to record conditions after sensor cleaning and provide a second record for accuracy checks.

After removal, the oxygen loggers were taken back to the office to download the recorded data. All logger data was checked using the temperature and oxygen profiles and recorded conditions, with unusual data or data associated with dry conditions noted in the file. Oxygen logger data was also compared to audit data to assess logger accuracy using the sample grade criteria from DEQ's Data Quality Matrix for both temperature and oxygen measurements.

Bacteria and Nutrient Monitoring

E. coli and total phosphorus samples were taken alongside grab sample measurements at five sites on the South Fork Burnt River and Burnt River between June and October. Samples were taken in stream, with *E. coli* samples collected in 250 mL widemouth bottles and phosphorous samples collected in 250 mL Boston round containers. One duplicate was taken at a randomly chosen site during each sampling event for both *E. coli* and phosphorus to assess sample precision. All samples were placed in an ice filled cooler after collection and were transported to Ontario, OR for shipment to US Bureau of Reclamation Pacific Northwest Water and Soil Laboratory in Boise, ID for analysis. *E. coli* samples were analyzed within 24 hours using EPA method 9213D with cultures on mTec ager at 44.5 °C for 22 hours to measure colony counts per 100mL. Phosphorus samples were analyzed at a later date with EPA method 365.1 using acid reduction and colorimetry to measure phosphorus concentrations in mg/L. All laboratory data was returned by the USBR laboratory to the PBWC and BRID at the end of the field season, including reporting limits and spike levels from the laboratory analysis. Results from the duplicate samples were used to assess sample accuracy using the sample grade criteria from DEQ's Data Quality Matrix.

Water Quality Index (WQI)

While the measurements provided by the grab sample and temperature monitoring efforts are extremely useful to assess water quality on their own, there was still a need for one score to represent water quality and easily track changes in status and trends over time. A useful metric to achieve this is a water quality index or WQI. A WQI uses multiple sub-index scores representing water quality parameters to describe general water quality at a monitoring site at a point in time.

The WQI the PBWC used in this report was based off the Oregon Water Quality Index, or OWQI, which was first developed by the Oregon DEQ in 1979 (**Cude 2001**). The scores included in the OWQI were based on National Sanitation Foundations' WQI and use logarithmic transformations to convert variables into a score from 10-100, with higher scores representing higher water quality. The OWQI includes seven parameters in its WQI score: Temperature, pH, dissolved oxygen, biological oxygen demand (a measure of oxygen use), total solids, nitrogen, phosphorus, and bacteria. Each parameter within the OWQI is determined by inputting the measured values into mathematical equations to generate the sub-index scores.

While the OWQI provides a useful metric for overall water quality, the broad selection of parameters wasn't a great fit for the PBWC given the smaller number of parameters measured by our monitoring efforts. Still, the sub-index equations provided a useful foundation for the generation of a unique WQI to assess water quality trends in the Powder Basin using PBWC monitoring data. This unique WQI uses dissolved oxygen, pH, temperature, and turbidity to generate a similar index to track changes in water quality at the PBWC's monitoring sites both for the 2022-24 monitoring period and for the longer 2013-2024 monitoring period. WQI scores will also be incorporated into the 2025-2035 Watershed Restoration Action Plan to help prioritize future Watershed Council actions and/or focus geographies.

The sub-index scores for oxygen were based on biological oxygen demand for salmonids and use both oxygen saturation and concentration to generate the sub-index scores. The OWQI uses the equation

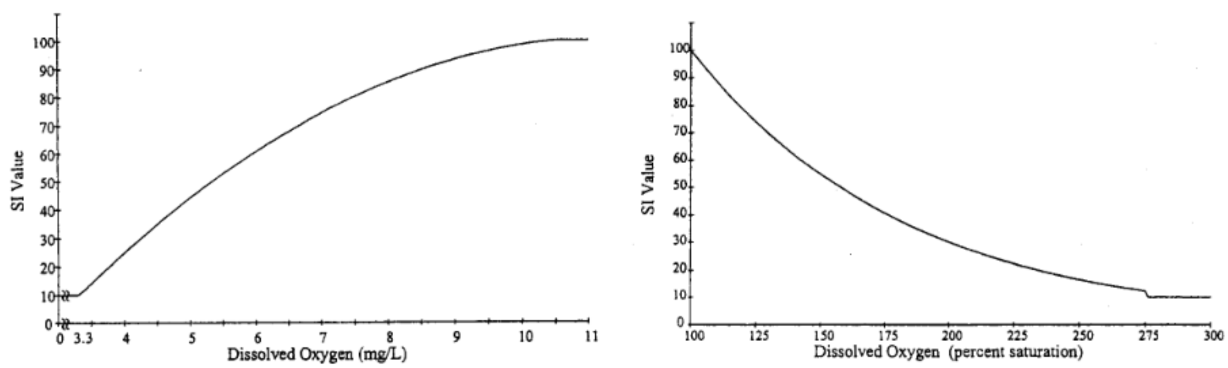
$$SI_{DO} = -80.29 + 31.88 * DO_C - 1.401 * DO_C^{\,2}$$

to generate subindex scores when oxygen saturation is below 100%, with sub-index scores decreasing logarithmically at lower concentrations due to hypoxia causing stress and death with the lowest scores occurring below 3.3 mg/L whereas the highest scores occur above 10.5 mg/L (Figure 10). For saturations above 100%, OWQI uses the equation

$$SI_{DO} = 100 * \exp(DO_S - 100) * -0.01197$$

with sub-index scores decreasing exponentially at saturations over 100% due to issues with gas bubble trauma and respiratory distress and bottoming out at 275% saturation (Figure 10).

Figure 10. Subindex equations for dissolved oxygen, with the equation on the left used when oxygen saturation is below 100% (in mg/L), and the equation on the right used when oxygen saturation is above 100% (in % saturation).

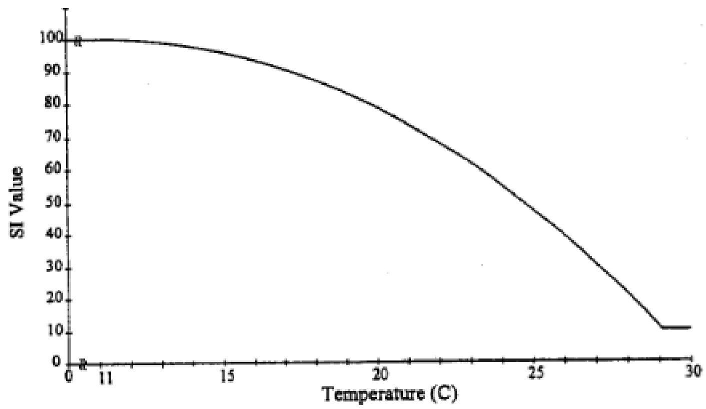


Sub-index scores for temperature are designed to be protective of cold-water fishes and uses mean monthly maximum daily temperature during the month of grab sample collection to find sub-index value. The OWQI uses the equation

$$SI_T = 76.54 + (4.172 * T) - (0.1623 * T^2) - (0.0020557 * T^3)$$

with sub-index scores decreasing logarithmically at temperatures above 11 °C and bottoming out at 29 °C (Figure 11).

Figure 11. Sub-index equation for temperature (in °C)



The pH sub-index scores were designed to protect aquatic life while encompassing the broad variation in natural alkalinity of Oregon's' watersheds. The OWQI uses two equations to assess pH, with the highest sub-index scores between 7 and 8. For pH values less than 7, OWQI uses the equation

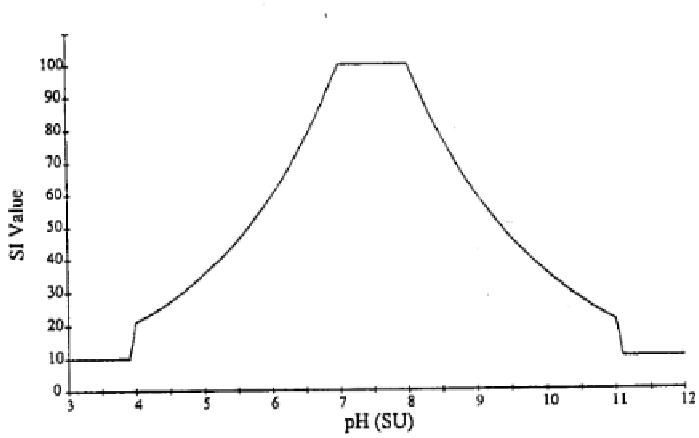
$$SI_{pH} = 2.628 * \exp(pH * 0.5200)$$

with subindex scores decreasing exponentially and bottoming out at pH values below 4. For pH above 8, the OWQI uses the equation

$$SI_{pH} = 100 * \exp\,((pH - 8) * \; -0.5188)$$

with sub-index scores decreasing exponentially and bottoming out at pH values above 11 (Figure 12).

Figure 12. Sub-index equation for pH

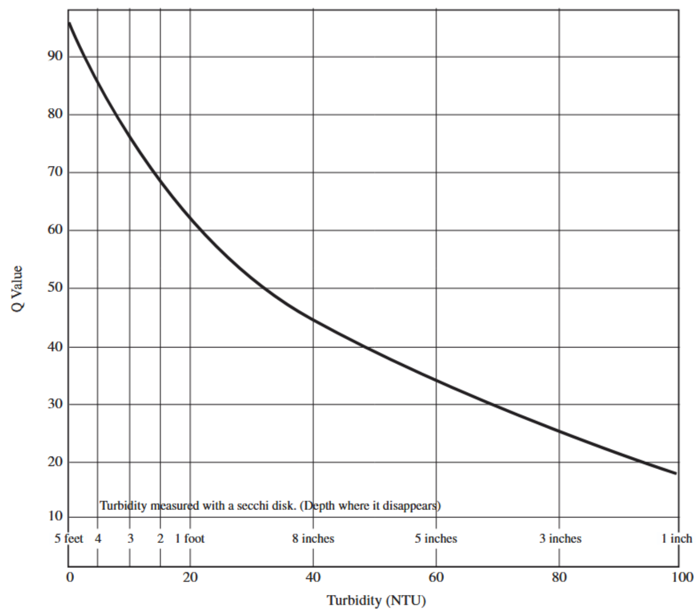


Because the OWQI uses total solids, rather than turbidity, as a metric to assess the impact of suspended sediment on water quality, the National Sanitation Foundations turbidity equation was used instead. Estimates of sub-index scores from this equation were entered into Microsoft Excel, and a quartic equation was fit to calculate to sub-index scores, with the equation

$$SI_{Turb} = (0.000003 * Turb^4) - (0.0007 * Turb^3) + (0.062 * Turb^2) - (2.9187 * Turb) + 100$$

With sub-index scores highest at 0 and decreasing exponentially with the lowest scores at turbidity levels above 100 NTU's (Figure 13).

Figure 13. The sub-index equation for turbidity (in NTU's)



While the OWQI used weighted measures for each sub-index value to generate the final WQI score, the WQI used in this report combined all of the sub-index scores into one value unweighted to better assess the impact of each sub-index value on overall water quality. The same water quality classes were still used from OWQI based on the final WQI score (Table 1).

Table 1. WQI classes based off of WQI scores

WQI Range	WQI Class
90-100	Excellent
85-90	Good
80-85	Fair
60-80	Poor
10-60	Very Poor

Analysis

Understanding the spatial and temporal patterns in each water quality parameter and water quality being a primary focus of the PBWC's monitoring efforts. The primary questions for the 2022-2024 monitoring efforts were as follows:

- How does each water quality parameter differ between sites within watersheds?
- How does each water quality parameter change throughout the year?
- Are there differences in water quality parameters between years, and what patterns are associated with them?
- When and where are water quality issues present?
- What are the patterns in WQI scores between sites, months, and years?
- What are trends in WQI scores over the 2022-2024 period and the 2013-2024 period?

Given the large amount of data collected over the 2022-2024 period (1,657 individual grab samples, 99 *E. coli* and phosphorus samples, 845,991 individual temperature logger measurements, and 109,890 dissolved oxygen logger measurements) methods were required to answer relevant questions and visualize the data, as well as to make useful comparisons between sites and over time.

The primary method used to analyze the data was the use of multiple linear regression models using site, month, and year as explanatory variables. These models were used to identify mean values in each variable, including 95% confidence intervals for the mean which were useful in identifying variability in the estimates for each site, month, and year. Estimates for monthly and yearly differences assumed that all sites change in the same direction and magnitude for each month and year. These models were also used to assess how well site, month, and year explained the variation in data, a metric known as R^2, which was useful in identifying if spatial and temporal patterns were important in estimating mean values. The timeframe for the analysis was set to include data for the June – October period, when data coverage was highest, and for each year from 2022 through 2024. While no transformations were needed for models estimating oxygen, pH, and temperature metrics, a log transformation was applied to parameter estimates for conductivity, turbidity, and WQI to account for non-linearity in responses, meaning that estimates for lower values changed less than estimates for higher values.

The nature of the grab sample collection meant that the utility of the analysis was mostly focused on assessing ambient, daytime conditions. For example, some grab sample parameters, such as dissolved oxygen, differ significantly throughout the day due to changes in temperature, weather, or other conditions. The analysis also hides the high degree of variability that can occur within months that we were unable to assess due to the nature of the once-per-month sample design. Still, the analysis was useful in assessing spatial and temporal patterns for sites with a large enough dataset and methods to account for variation in data. In addition to the multiple linear regression analysis, the percentage of measurements above or below DEQ standards was determined for each parameter.

While the analysis of grab sample data was able to use the once monthly sample design to easily assess monthly and yearly changes in parameters, temperature provided several analytical problems due to the thousands of measurements taken per month by the temperature loggers. One method to smooth the variability in the temperature data was to use the seven-day averages of mean and maximum daily temperatures which also aligned them with the methods DEQ used for their temperature standards. These metrics were then averaged for each calendar month to find mean monthly mean daily temperature (hereafter referred to as mean temperature) and mean monthly maximum daily temperature (hereafter referred to as maximum temperature). Finally, comparisons between observed mean August stream temperatures, model estimate for mean August temperatures, and NorWeST estimates for the 1993-2011 period were found at each site to detect changes in stream temperatures between the 1993-2011 and 2022-2024 periods.

Analysis for data from dissolved oxygen loggers focused on the percentage of records below 8 mg/L minimum intergravel concentration and above 11 mg/L preferred spawning oxygen concentration. Dissolved oxygen logger analysis also looked at the timing and range of oxygen maximum and minimum daily values as well as correlations with stream temperature to assess the sensitivity of oxygen concentrations to temperature changes.

E. coli and phosphorus analysis in the Burnt River followed similar methods as those for the grab sample parameters. Along with this analysis, correlations with grab sample parameters and temperature metrics were used to identify relationships with these parameters. *E. coli* concentrations were log transformed to account for non-linearity in measurements, while no transformations were needed for the phosphorus data. Simple linear regressions were used to determine correlations between the log transformed *E. coli* data or phosphorus data and dissolved oxygen concentrations, pH, mean temperature, and maximum temperature. Log transformations were applied to conductivity and turbidity data for these correlations.

Only sites with one or fewer failed accuracy checks and “A” or “B” precision grades were used in the grab sample analysis. Temperature data during months with failed field audit checks (“C” grade data) was not used in temperature analysis. For the WQI analysis, Accuracy and Precision indices were created to be able to select high quality data from multiple parameters. For the Precision Index, a value of 1 was assigned for samples with “A” grade quality, 0.8 was assigned for data with “B” grade quality, a value of 0.3 was assigned for samples with “C” grade quality, with 0 assigned to missing parameter data. These values were then averaged out to generate the value for the Precision Index. For the Accuracy Index, a value of 1 was assigned to samples without any failed accuracy checks, 0.8 was assigned for samples with one failed accuracy check, 0.5 was assigned to samples with 2 failed accuracy checks, a value of 0.3 was assigned to samples with three total failed accuracy checks or for samples with more than two failed accuracy checks for one parameter, and a value of 0 was applied for samples with more than three failed accuracy checks. Only samples with an Precision Index value over 0.86 and Accuracy Index value over 0.5 were used for the WQI analysis.

Upper Powder

Background

The Upper Powder basin drains 895 km^2 (345 mi^2) and contains nine monitoring sites located primarily in the Cracker Creek and Deer Creek watersheds and on the mainstem of the Powder River (Figure 1.). Most of these sites were established in 2013 or 2014, with one site on Fruit Creek established in 2018 and one site below Mason Dam established in 2023 (Table 2).

Map 5. The Upper Powder watershed with sample sites, major tributaries, and important features highlighted.

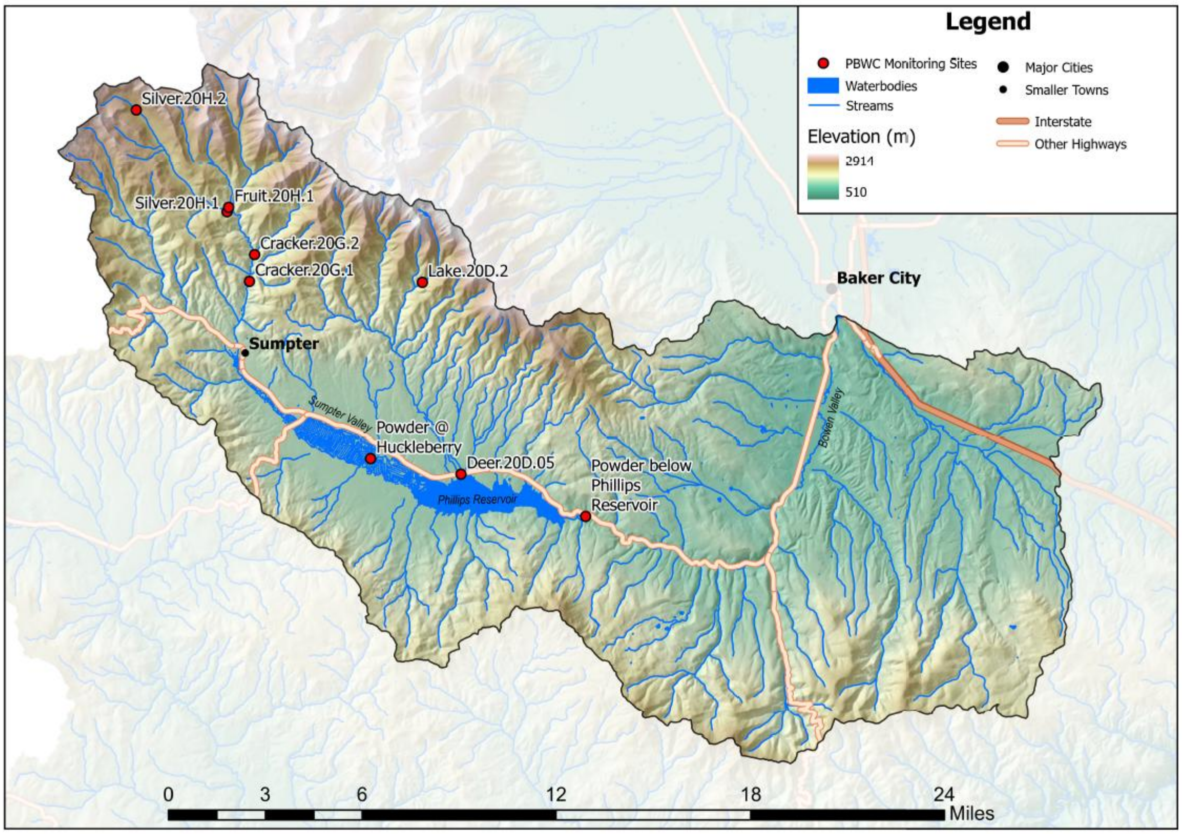


Table 2. Site characteristics of Upper Powder monitoring sites including fish habitat type, Elevation (m), Drainage Area (km^2), Predicted NorWeST 1993-2011 mean August Stream temperature (°C), and established date

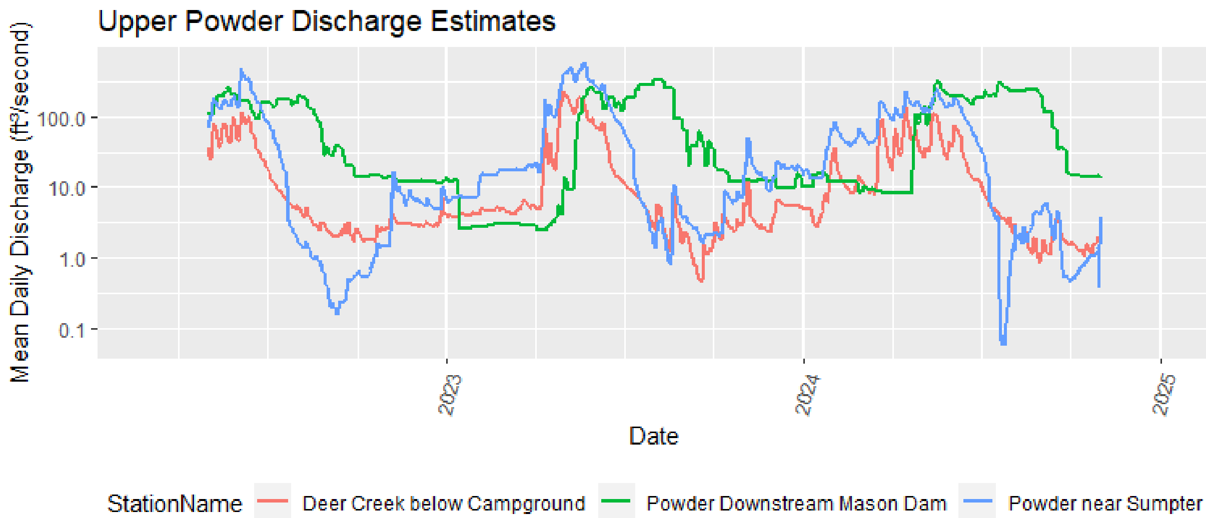
Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
Cracker.20G.1	37229	Cool	1406	78.89	13.4	5/20/2014
Cracker.20G.2	37417	Cool	1441	67.58	12.96	5/20/2014
Deer.20D.05	37326	Cool	1243	74.58	15.25	6/25/2013
Fruit.20H.1	40986	Cold	1561	11.03	11.09	7/25/2018
Lake.20D.2	41726	Cold	1624	7.38	10.07	5/20/2014
Powder @ Huckleberry	37327	Cool	1261	199.63	17.3	6/25/2013
Powder DS Mason Dam	41792	Cool	1185	403.76	16.34	5/31/2023
Silver.20H.1	37728	Cold	1542	32.74	11.37	5/20/2014
Silver.20H.2	37788	Cold	1927	7.79	8.56	6/16/2014

The hydrology of Upper Powder watershed is dominated by the impacts from mining in the Sumpter Valley and from Mason Dam. Dredging for gold occurred in the Sumpter Valley between 1913 and

1953, resulting in 11.75 km^2 of displaced soils and stream substrates known as mining tailings. These tailings exist from the upstream section of Phillips Reservoir to Cracker Creek above Sumpter and alter the hydrology of the area by decreasing alluvial groundwater storage capacity and reducing the stream complexity that drives alluvial groundwater recharge. These changes result in a "flashy" seasonal flow pattern, with higher flows during the spring runoff period and lower flows in summer and fall (Chart 1).

Mason Dam, constructed in 1968, impounds ~114,000 acre-feet of water forming Phillips Reservoir. Dam operations needed to store and distribute water for irrigation users alter flow regimes in the Powder River Downstream, with lower flows during the traditional high flow period (March – June) and higher flows in the summer and early fall (Chart 1). The hypolimnetic discharge from the dam during the summer and early falls also results in lower stream temperatures during the summer through fall period as well. Irrigation withdrawals above Mason Dam also have an impact on the hydrology of the watershed, with three major ditches diverting flow from the Powder River within this reach.

Chart 1. Estimated discharge (on a log scale) for three OWRD gaging stations in the Upper Powder watershed. Note the high degree of variability in discharge for the station on the Powder River upstream of Phillips Reservoir in the Sumpter Valley dredge tailings.

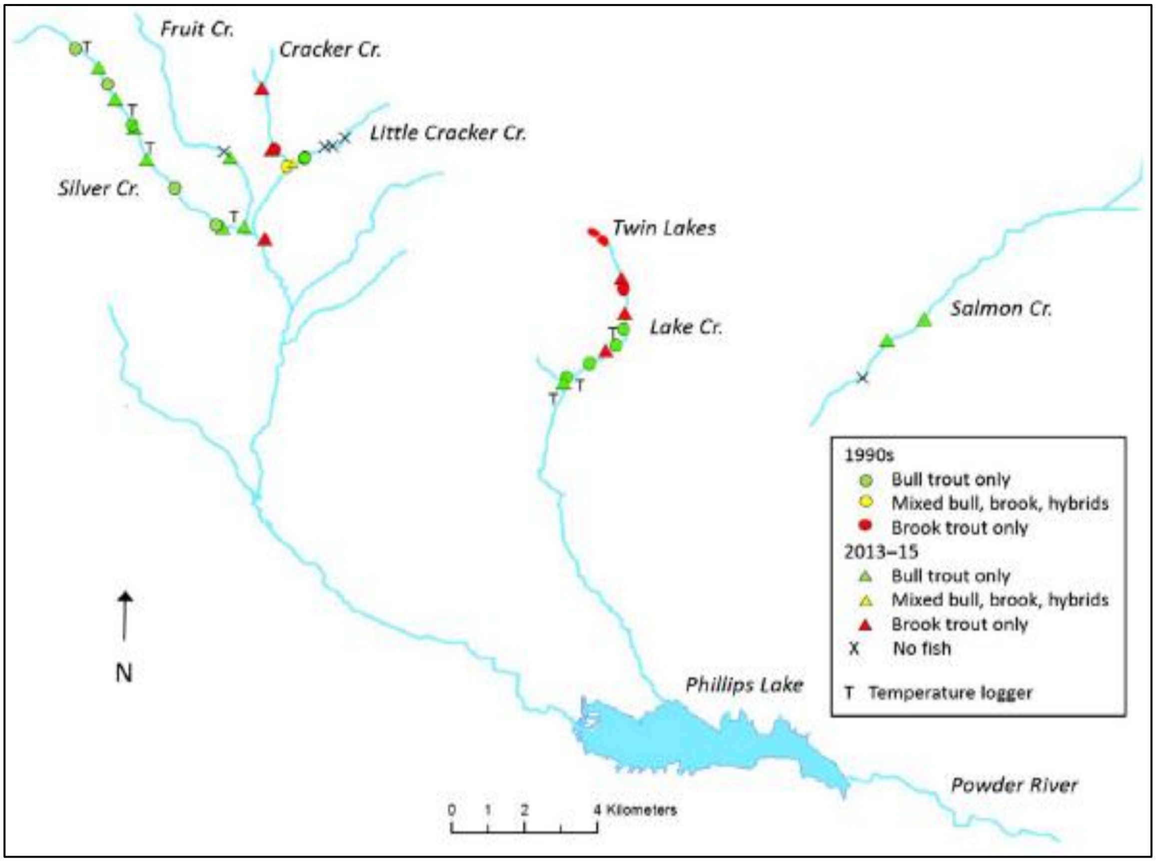


Redband trout are widespread throughout the Upper Powder watershed, but are more limited in the section within Phillips Reservoir and the dredge tailings. Spawning and incubation is primarily found in Deer Creek, Little Dean Creek, Union, Creek, and McCully Fork Creek, along with the lower section of Cracker Creek. Summer rearing is mostly found in the higher part of the watershed, primarily Cracker Creek and McCully Fork Creek, with some usage in the upper part of streams draining the Elkhorn Mountains. Winter rearing is more focused on the lower reaches of the larger tributaries, primarily Deer Creek, Union Creek, and Little Dean Creek, with lower usage within the mainstem of the Powder River within the dredge tailings (**NPCC 2004b**)

The Upper Powder is also a focus area for Bull Trout conservation, with resident populations present in Lake Creek, Cracker Creek, Fruit Creek, and Silver Creek. Populations within the Cracker

Creek watershed are more robust than those found in the Deer Creek watershed, but are still limited by temperature and habitat degradation, along with competition and interbreeding with non-native Brook Trout (**NPCC 2004b**). Brook Trout are present throughout much of the basin, including the upper sections of Lake Creek, and Cracker Creek upstream of the Silver Creek confluence, but are largely absent from Silver Creek (**Howell 2017**, Figure 14).

Figure 14. Distribution of Bull Trout and Bull Trout in the Upper Powder basin. From Howell 2017.



Monitoring in 2022 started later than other years, with grab sampling and temperature loggers installed in late June. While monitoring started earlier in 2023, temperature loggers were not installed at the Fruit and Silver Creek sites until late July due to limited supplies of temperature loggers that year. In 2024, a large pulse of sediment from upstream mining occurred in May on Silver Creek and Fruit Creek, resulting in high turbidity values. While the cause of this turbidity pulse remains unknown, conversations with campers visiting the area during the site visit mentioned small-scale mining by visitors during Labor Day Weekend on upper Fruit Creek as a likely cause.

Continuous dissolved oxygen monitoring took place at four sites including Deer Creek, the Powder River within the dredge tailings, upper Cracker Creek, and on the Powder River downstream of Mason Dam during the spring to monitor Redband Trout spawning conditions, and at three sites on Lake Creek, Silver Creek and lower Cracker Creek to monitor Bull Trout spawning conditions in the fall. A late start to the field season in 2022 limited DO monitoring in the spring, but was able to be conducted in the spring in 2023 and 2024. Fall monitoring was able to take place each year, with coverage highest in late September to mid-October.

Grab Sample Monitoring

Dissolved oxygen concentrations were generally supportive of cold- and cool-water salmonid population at most sites and months, with all DO measurements above the 6.5 mg/L cool-water Standard for Redband Trout. Among all sites, only Deer.20D.05 and Powder @ Huckleberry had dissolved oxygen concentrations fall below the 8 mg/L cold-water standard in June, July, and August. The dissolved oxygen saturation at these sites was at or above 100%, indicating that temperature is likely the primary factor for these lower oxygen levels. Strong patterns related to elevation existed between sites and estimated mean dissolved oxygen concentrations for the 2022-2024 period. DO measurements were highest in Silver Creek, Fruit Creek, and Cracker Creek, with mean Dissolved oxygen concentration above 9 mg/L during the 2022-2024 period. Little noticeable patterns existed among sites regarding dissolved oxygen saturation, although in general sites lower in the basin had higher levels of oxygen saturation during the summer and early Fall.

Chart 2. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the Upper Powder watershed monitoring sites over the 2022-2024 period. Cool-water standard (red line) highlighted.

Table 3a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the Upper Powder watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Huckleberry	8.64	9.04	9.43
Deer.20D.05	8.27	8.65	9.04
Cracker.20G.1	8.87	9.26	9.65
Cracker.20G.2	9.12	9.50	9.89
Silver.20H.1	9.15	9.57	9.98
Fruit.20H.1	9.12	9.55	9.97
Silver.20H.2	8.99	9.41	9.84

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.18	0.25	0.68
July	-1.35	-0.92	-0.49
August	-1.07	-0.73	-0.39
September	-0.12	0.22	0.56
October	0.82	1.17	1.52

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.61	-0.30	0.02
2023	-0.17	0.14	0.45
2024	-0.17	0.15	0.47

Overall, pH conditions throughout the Upper Powder watershed were within DEQ recommend levels, although Deer.20D.05 and Cracker.20G.2 had pH measurements below the 6.5 lower threshold for samples collected in June and July of 2024. Both measurements had accuracy checks and precision checks within recommended levels for pH, meaning these measurements are likely outliers rather than measurement errors. pH measurements in fall 2022 were lower than expected based on measurements in the fall of 2023 and 2024, but these lower levels are likely the result of lower levels of precipitation and flow and higher stream temperatures during 2022. Differences in pH measurements among sites did not follow any spatial patterns, with both the lowest and highest pH measurements found at higher elevation sites (Silver.20H.2 and Fruit.20H.1, respectively), and most sites having similar pH values.

Stronger patterns were observed throughout the season, but these differences were generally not statistically significant except for August, with near significant differences in October. pH was generally lowest in the spring and highest in summer, with pH measurements 0.11 higher in July and 0.25 higher in August than in June. pH measurements differed between years, although these differences were only significant in 2023, where pH was 0.30 higher than 2022. Differences between 2022 and 2024 were less significant but still noticeable, with measurements 0.22 higher in 2024 than 2022.

Chart 3. Observed pH measurements for the Upper Powder watershed monitoring sites over the 2022-2024 period. Upper (dashed line) and lower (solid line) pH recommended standards highlighted.

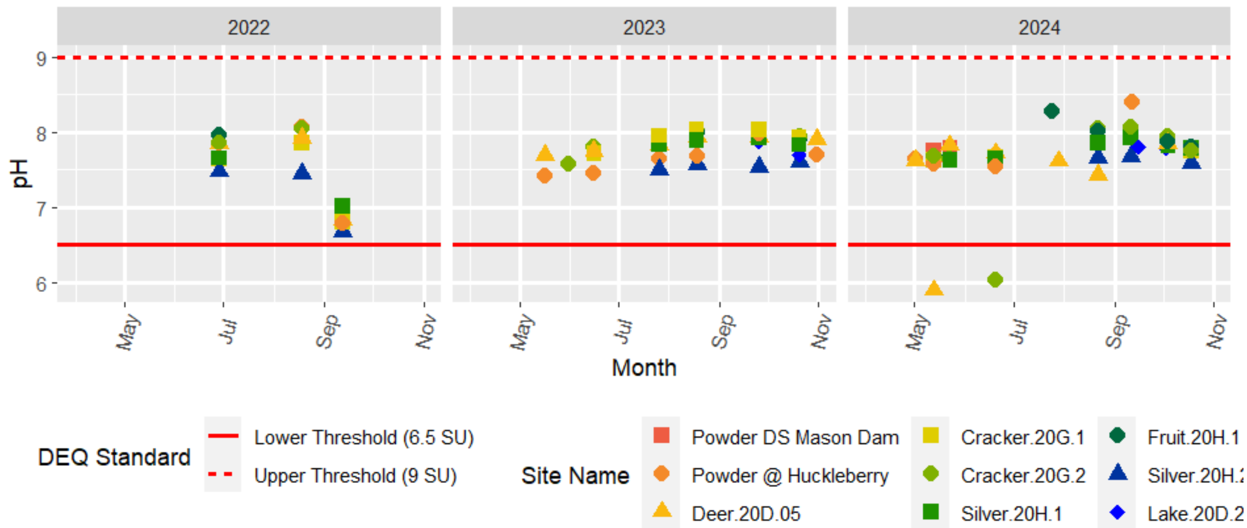


Table 4a. Estimates of mean pH for the Upper Powder watershed sites over the 2022-2024 period

Site Name	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
Powder @ Huckleberry	7.42	7.69	7.96
Deer.20D.05	7.42	7.69	7.95
Cracker.20G.1	7.47	7.73	8.00
Cracker.20G.2	7.44	7.7	7.97
Silver.20H.1	7.44	7.72	8.00
Fruit.20H.1	7.58	7.87	8.17
Silver.20H.2	7.16	7.45	7.73

b. Mean monthly pH differences from mean values over the 2022-2024 period

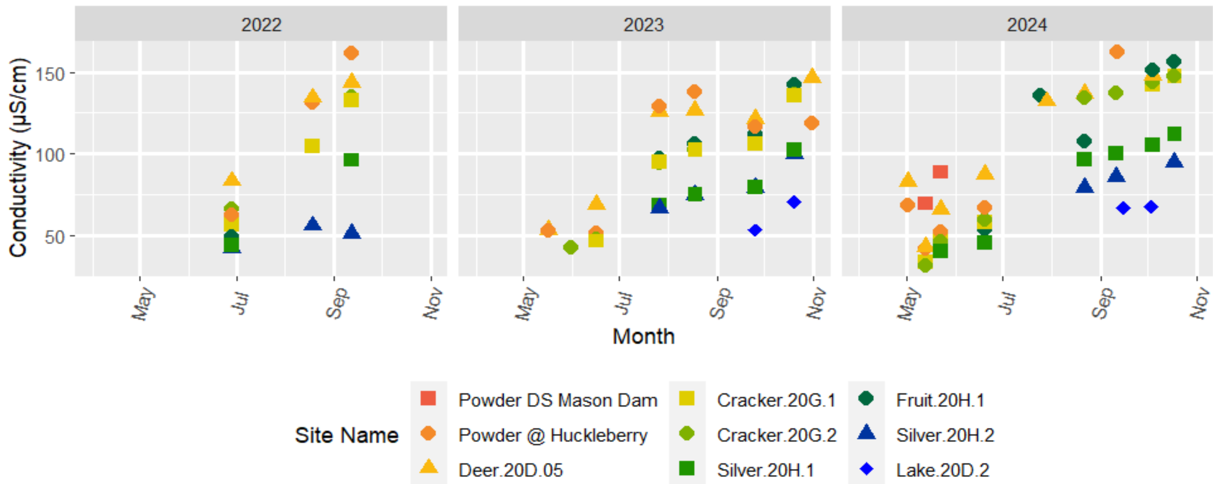
Month	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
June	-0.37	-0.09	0.18
July	-0.28	0.02	0.31
August	-0.07	0.16	0.39
September	-0.33	-0.1	0.13
October	-0.23	0.01	0.25

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
2022	-0.45	-0.17	0.10
2023	-0.08	0.13	0.34
2024	-0.17	0.05	0.26

Differences in conductivity between sites were strongly associated with upstream basin size, with sites higher in the watershed (Silver Creek, Lake Creek) having lower conductivity values compared to sites lower in the Watershed. Among the higher elevation sites, Fruit.20H.1 had higher conductivity values than the Silver Creek sites, likely related to a higher proportion of shale and older sedimentary rocks in the upstream watershed. Lake Creek also had lower measured and estimated conductivity measurements than the other high elevation sites, likely due to the smaller area of the basin upstream of the site. Among the lower elevation sites, Deer Creek was an outlier, with higher conductivity measurements than sites with similar upstream drainage basin area, such as those on Cracker Creek, possibly due to higher groundwater contributions to baseflow.

Chart 4. Observed conductivity measurements (in µS/cm) for the Upper Powder watershed monitoring sites over the 2022-2024 period.



Seasonal patterns in conductivity were noticeable and heavily related to flow, with an increasing trend found from spring through fall. Overall, measurements were lowest in June and highest in October, with higher levels of flow associated with lower conductivity values. A similar relationship with flow was observed for differences in mean conductivity values between years, with conductivity 5.0 µS/cm lower in the comparatively wetter 2023 than 2022, whereas differences were less significant for 2024, with conductivity measurements 4.55 µS/cm higher in 2024 than 2022.

Table 5a. Estimates of mean conductivity (in µS/cm) for the Upper Powder watershed monitoring sites over the 2022-2024 period

Site Name	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
Powder @ Huckleberry	111.84	122.51	134.20
Deer.20D.05	118.49	129.42	141.35
Cracker.20G.1	95.81	104.68	114.38
Cracker.20G.2	99.19	108.28	118.21
Silver.20H.1	73.15	80.39	88.35
Fruit.20H.1	96.44	106.36	117.29
Silver.20H.2	62.40	68.72	75.67

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

Month	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
June	-50.03	-45.57	-40.69
July	-9.54	1.72	14.44
August	-5.59	3.8	14.15
September	2.09	11.77	22.40
October	15.39	28.28	42.65

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Year	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
2022	-5.51	0.16	6.37
2023	-10.11	-4.87	0.87
2024	-1.52	4.71	11.56

Turbidity levels showed a strong relationship to stream size and elevation, with the highest turbidity values for the baseline period found at Powder @ Huckleberry and Deer.20D.05, while the lowest turbidity values were found at the highest elevation sites, particularly Silver.20H.2, where turbidity was 1.19 NTU's lower than at Powder @ Huckleberry. Some outliers to this pattern were present, primarily Fruit Creek, where high turbidity measurements were recorded during the 5/23/2024 sampling event. This pulse of turbidity appeared to have little long-term impacts on subsequent turbidity measurements at the site and locations downstream of it.

Chart 5. Observed turbidity measurements (in NTU's) for the Upper Powder watershed monitoring sites over the 2022-2024 period. Log transformation used for turbidity measurements.

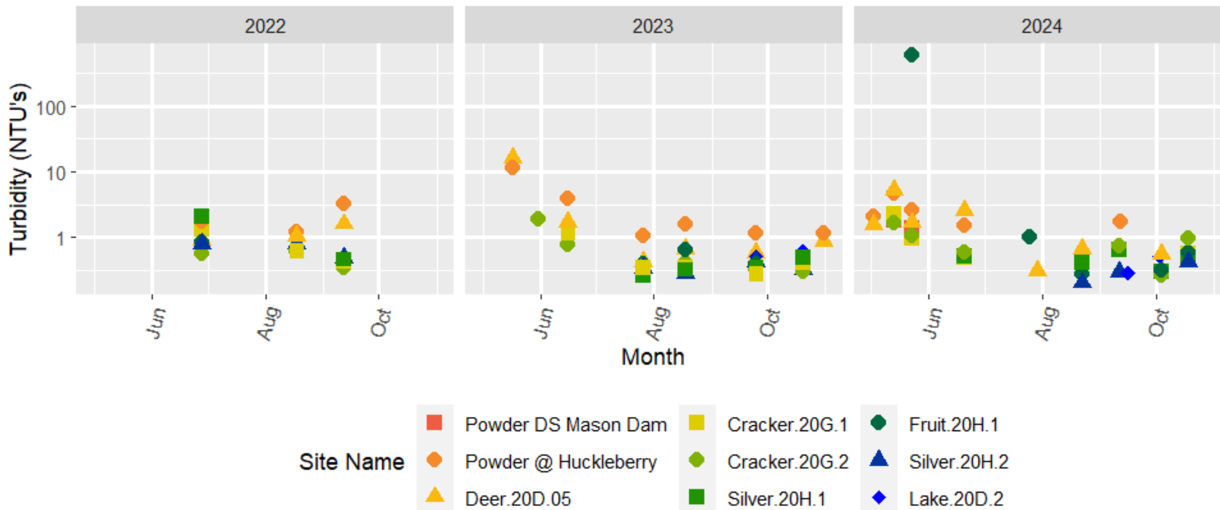


Table 6a. Estimates of mean turbidity (in NTU's) for the Upper Powder watershed monitoring sites over the 2022-2024 period

Site Name	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
Powder @ Huckleberry	1.18	1.62	2.23
Deer.20D.05	0.63	0.86	1.17
Cracker.20G.1	0.36	0.5	0.67
Cracker.20G.2	0.36	0.49	0.67
Silver.20H.1	0.37	0.51	0.71
Fruit.20H.1	0.39	0.55	0.77
Silver.20H.2	0.31	0.44	0.61

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
June	0.26	1.11	2.27
July	-1.01	-0.49	0.31
August	-0.77	-0.27	0.44
September	-0.70	-0.2	0.50
October	-0.74	-0.16	0.70

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Mean Estimate	Lower 95% Estimate	Upper 95% Estimate
2022	-0.28	0.57	1.73
2023	-0.88	-0.26	0.60
2024	-0.93	-0.31	0.55

Flow played a large role in determining seasonal patterns in Turbidity, with measurements highest early in the monitoring season and decreasing throughout the season. Turbidity values in June, near the end of the runoff period, were on average 1.30 NTU's higher than in July, when high flows have generally subsided and baseflow conditions begin. Flow was less important in determining differences in turbidity between years, with turbidity values lower in 2023 and 2024 than in 2022, despite higher flows during these years compared to 2022.

Stream temperature Monitoring

Similar to grab sample monitoring, continuous temperature monitoring experienced a late start to sampling in 2022, limiting the knowledge of spring and early summer conditions this year. Among the biggest issues unique to the temperature monitoring program was shifts in flow resulting in loggers becoming exposed to air temperatures. This was especially the case for the higher elevation sites where access is somewhat limited and for sites on Deer Creek and Powder @ Huckleberry due to rapid changes in flow between spring and summer. Results from periods where loggers were exposed were removed from the modeling analysis and graphs of temperature profiles.

Table 7a. Estimates of mean and Maximum Daily temperatures (in °C) for the Upper Powder watershed monitoring sites over the 2022-2024 period

Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Huckleberry	13.93	14.63	15.32	17.84	18.81	19.78
Deer.20D.05	13.62	14.31	15.01	17.56	18.53	19.50
Cracker.20G.1	10.14	10.84	11.53	12.90	13.87	14.84
Cracker.20G.2	9.47	10.17	10.86	11.18	12.15	13.12
Silver.20H.1	8.99	9.71	10.43	10.35	11.35	12.36
Fruit.20H.1	8.89	9.67	10.46	10.65	11.75	12.84
Silver.20H.2	6.46	7.18	7.91	7.52	8.52	9.53

b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Month	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-2.51	-1.82	-1.12	-3.10	-2.14	-1.17
July	1.91	2.60	3.29	2.22	3.18	4.14
August	2.23	2.93	3.63	2.43	3.41	4.38
September	-1.17	-0.48	0.22	-1.72	-0.75	0.21
October	-3.92	-3.23	-2.54	-4.66	-3.70	-2.74

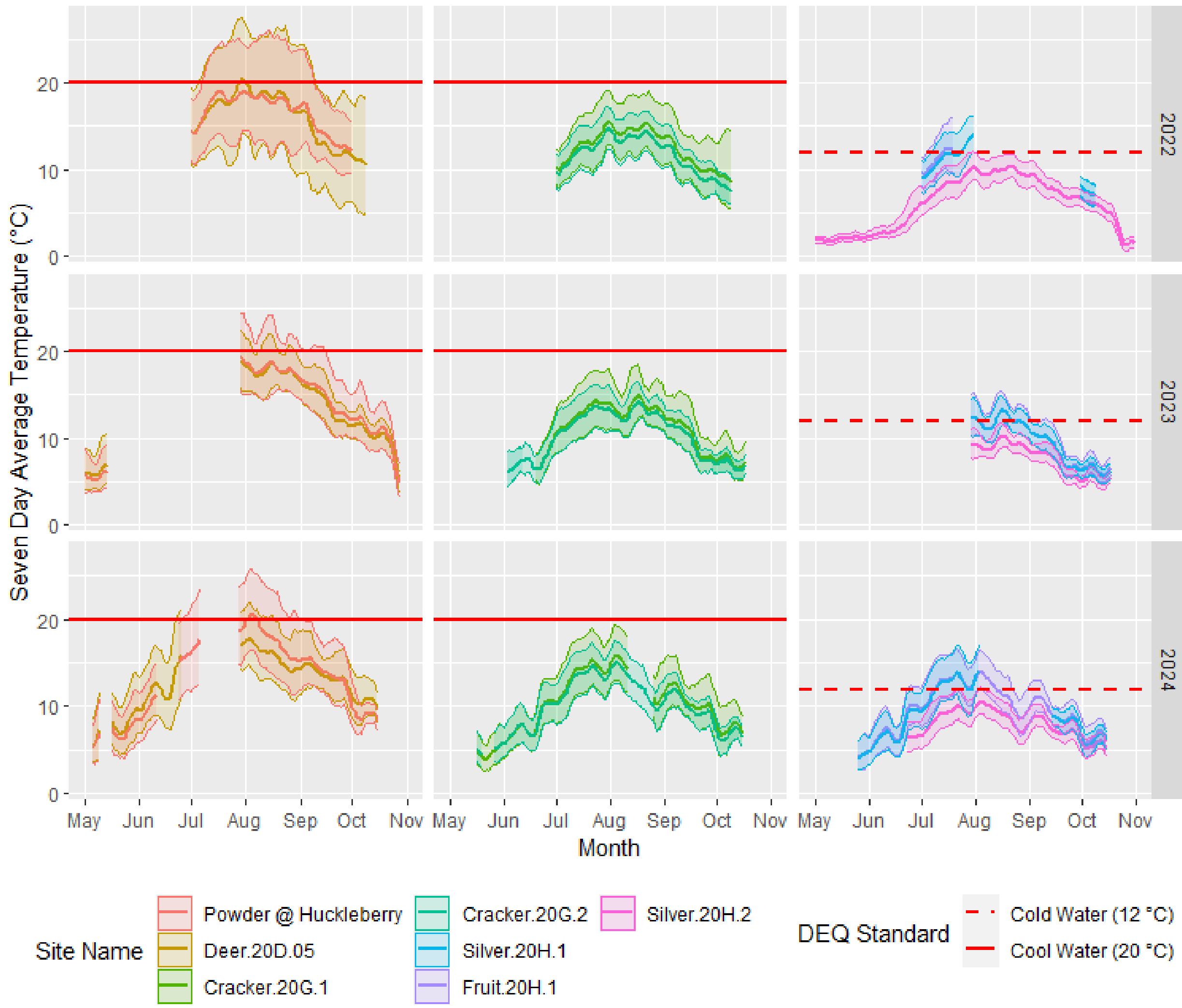
c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Year	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.22	0.48	1.17	0.02	0.99	1.96
2023	-1.03	-0.30	0.42	-1.77	-0.77	0.24
2024	-0.87	-0.17	0.53	-1.20	-0.23	0.75

The late start to temperature monitoring and exposed loggers later in the season limited the utility of the temperature loggers in determining maximum and mean daily temperatures during 2022 for the summer period. Better logger coverage during 2023 and 2024 for Silver.20H.1 and Fruit.20H.1 identified maximum temperatures above the 12 °C standard for most of July and August, with daily maximum temperatures of 14.55 °C at Silver.20H.1 and 14.32 °C at Fruit.20H.1 for these months in

2023 and 2024. Of the upper elevation sample sites, only Silver.20H.2 had maximum temperatures below the 12 °C maximum daily standard for Bull Trout for each year of the monitoring period.

Chart 6. Temperature profiles (in °C) for the Upper Powder watershed monitoring sites over the 2022-2024 period, with lower watershed sites on the left, Cracker Creek sites in the middle, and Fruit and Silver Creek sites on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cold water (dashed line) and cool water (solid line) standards highlighted in red based off relevant fish use at the site.



For monitoring sites on Cracker Creek, stream temperatures were similar early in the season, but diverged from one another during the summer and fall, with mean temperatures 0.71 °C higher and maximum daily temperatures 1.64 °C higher at the lower site than the higher site. Maximum daily temperatures at both sites were below the 20 °C standard for redband trout but were above the 12 °C standard for Bull Trout, with average daily maximum temperatures of 16.81 °C for Cracker.20G.1 and 15.21 °C for Cracker.20G.2 for July and August over the 2022-2024 monitoring period.

Temperature profiles were the warmest at Deer.20D.05 and Powder @ Huckleberry, the lowest monitoring sites in the Upper Powder watershed. There was limited logger coverage in June for all three years. This was particularly noticeable in 2023 due to rapid changes in flow resulting in dry loggers for much of the month. Data coverage was also limited in 2024 due to similar conditions resulting in dry loggers. Overall, there was greater variability in daily temperatures in 2022 compared to other years, with an average daily range of 11.88 °C for 2022 for July and August, versus 6.37 °C in 2023 and 8.29 °C in 2024 for the same months. Maximum temperatures in all three years exceeded the 20 °C standard for Redband Trout between June and September for both sites, with 2022 seeing the highest levels of exceedance. On average, maximum daily temperatures at Powder @ Huckleberry were 10.28 °C warmer and mean daily temperatures were 7.44 °C warmer than those at Silver.20H.2.

Both mean and maximum temperatures were lowest early in the season and highest in the summer, with maximum temperatures 5.54 °C warmer and mean temperatures 4.74 °C warmer in August than in June. Temperatures were coolest in the fall, with mean temperatures 1.42 °C cooler and maximum temperatures 1.56 °C cooler in October than in June. Lower levels of precipitation and warmer air temperatures likely resulted in higher temperatures in 2022 than either 2023 or 2024 for both mean and maximum daily temperatures. On average, maximum temperatures were 1.76 °C lower in 2023 and 1.22 °C lower in 2024 than in 2022. Mean temperatures followed similar trends, with temperatures 0.78 °C lower in 2023 and 0.636 °C lower in 2024 compared to 2022.

Both estimated and observed temperatures showed an increase in 2022-2024 Mean August stream temperatures compared to the 1993-2011 mean August temperature estimates from the NorWeST stream temperature model (Table 9). Observed temperatures for the 2022-2024 period were 0.94 °C warmer (Range 0.048-2.04), with the smallest increases found for Powder @ Huckleberry and Cracker.20G.2, while the largest increases were found at Silver.20H.1 and Deer.20D.05.

Table 8. Estimated and Observed mean August stream temperatures for the Upper Powder watershed monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

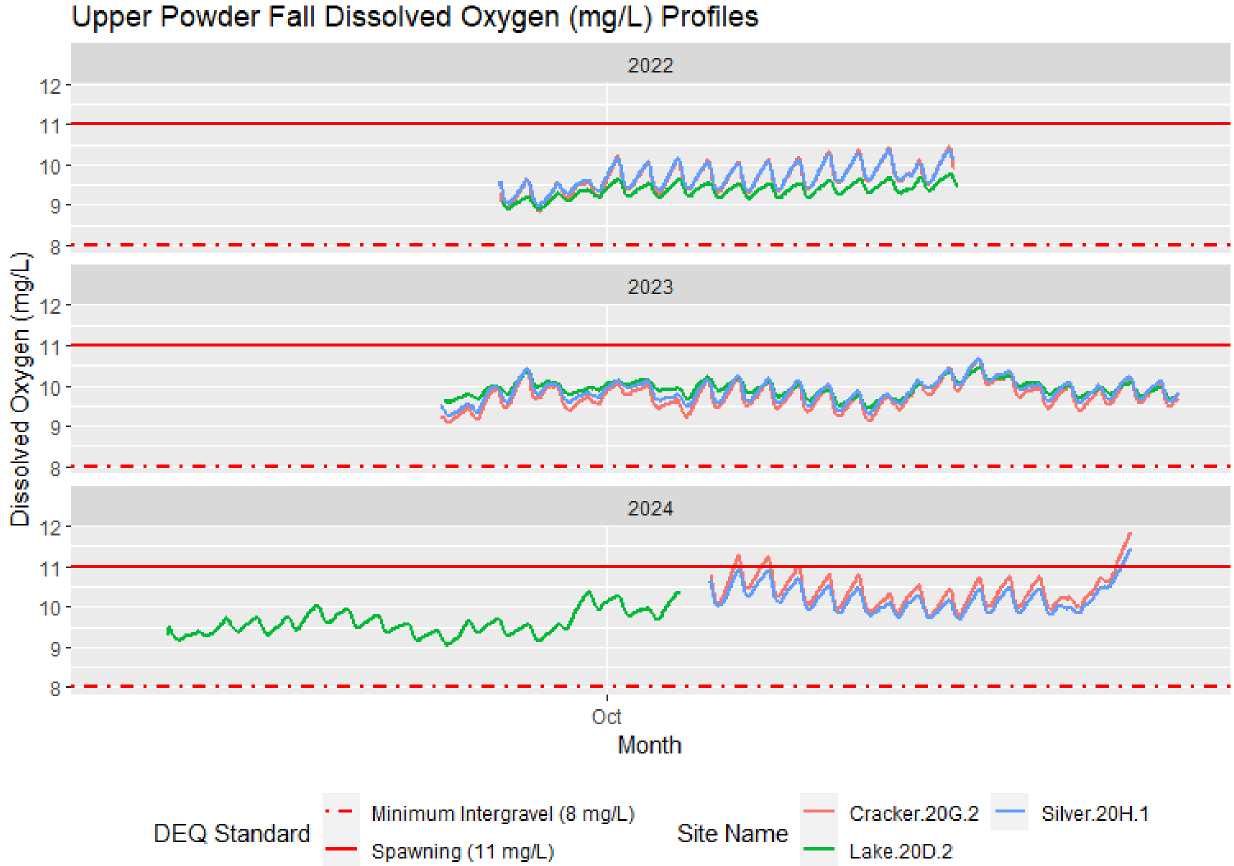
Site	Estimated 2022-2024 Mean August Temp (°C)	Observed 2022-2024 Mean August Temp (°C)	NorWeST 1993-2011 Temp (°C)	NorWeST 2040 Temp (°C)	NorWeST 2080 Temp (°C)
Powder @ Huckleberry	17.33	17.93	17.30	18.57	18.57
Deer.20D.05	17.13	17.28	15.25	16.47	17.24
Cracker.20G.1	13.78	13.81	13.40	14.58	15.32
Cracker.20G.2	13.29	13.01	12.96	14.12	14.86
Silver.20H.1	12.69	13.41	11.37	12.49	13.20
Fruit.20H.1	12.67	11.70	11.09	12.21	12.91
Silver.20H.2	10.14	9.39	8.56	9.61	10.27

Dissolved Oxygen Monitoring

Fall

Dissolved oxygen loggers were installed at three sites in the upper watershed in the fall to monitor Bull Trout spawning locations for each year between 2022 and 2024. Logger installation in the fall period were generally consistent between 2022 and 2023, but differed somewhat in 2024, with logger installations on Lake Creek occurring earlier and separately from those on Silver Creek and Cracker Creek.

Chart 7. Dissolved oxygen profiles (in mg/L) for the Fall Upper Powder watershed oxygen logger monitoring sites over the 2022 to 2024 period. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



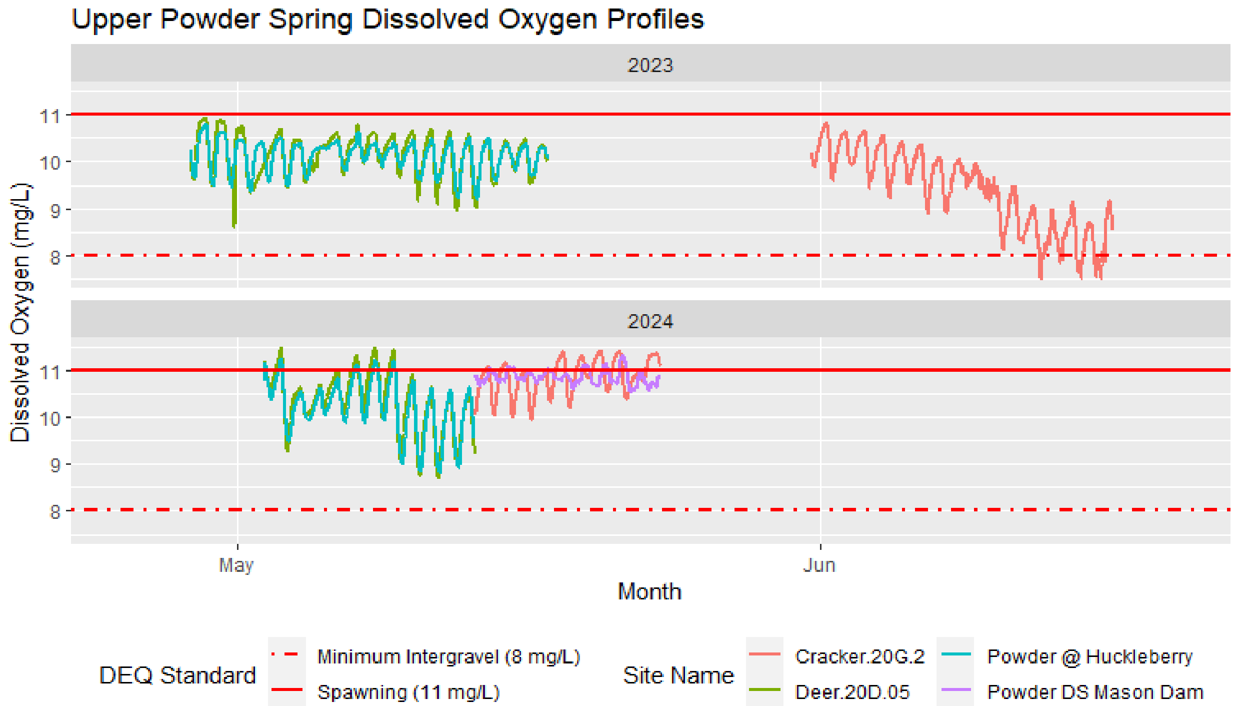
Dissolved oxygen concentrations were above the 8 mg/L minimum intergravel standard but were under the 11 mg/L spawning standard for most of the Fall Bull Trout spawning period at all Upper Powder sites. Overall dissolved oxygen concentrations and daily variation in concentrations were lower at Lake.20D.2 in 2022 than the other monitoring locations but were generally similar for subsequent years. Timing of dissolved oxygen also differed between sites, although only slightly, with the highest oxygen concentrations seen at 8:00 in the morning at Cracker.20G.2 and Silver.20H.1, while the highest concentrations were seen later in the day, around 10:00, for Lake.20D.2. All sites had lower concentrations in the evening, around 17:00.

Correlations between stream temperatures and oxygen concentrations were generally similar between sites, with dissolved oxygen concentrations decreasing by 0.251 mg/L per 1 °C increase in temperatures (Appendix D). These temperature relationships had large impacts on overall patterns of dissolved oxygen over the entire monitoring timeframe, with higher dissolved oxygen concentrations higher later in the season due to cooler temperatures. Differences between years were obscured by logger accuracy issues, particularly for Lake.20D.2, with the logger recording oxygen concentrations 0.69 mg/L lower than audits in 2022, compared to logger measurements 0.50 mg/L lower in 2023 and 0.34 mg/L lower in 2024 than audits.

Spring

For the spring season, loggers were installed at four sites to monitor Redband Trout spawning habitats. Logger installations in 2023 were more spread out temporally, with logger installations occurring later in the season on Cracker Creek than for the lower watershed sites. Installations in 2024 were more condensed, with all logger deployments occurring in May that year. All sites were monitored in May, but only Cracker.20G.2 was monitored in June 2023.

Chart 8. Dissolved oxygen profiles (in mg/L) for the Spring Upper Powder watershed oxygen logger monitoring sites in 2023 and 2024. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



While dissolved oxygen concentrations were consistently above the 8 mg/L minimum intergravel standard, overall concentration were mostly below the 11 mg/L spawning standard. No sites had oxygen concentrations above 11 mg/L in 2024, and while all sites did have concentrations above the spawning standard in 2024, only 19 % of records were above 11 mg/L, most commonly at Cracker.20G.2 and Powder downstream of Mason Dam. Concentrations decreased significantly at Cracker.20G.2 through June 2023, with 6.9% of measurements at this site in 2023 below 8 mg/L, and concentrations more commonly below 8 mg/L from June 12th through the end of monitoring on June 16th. Among the spring sites, Powder DS Mason Dam had the lowest level of variation in dissolved oxygen, likely due to the stable water temperatures coming from the dam outflow. The other sites had greater variability in daily oxygen concentrations, particularly for Deer.20D.05.

The timing of oxygen concentrations differed over the course of the season and between sites for the spring monitoring sites. For Deer.20D.05 and Powder @ Huckleberry, the highest oxygen concentrations were seen in the morning between 8:00 and 10:00 , with an earlier peak in 2024 compared to 2023. At Cracker 20G.2, oxygen concentration peaked earlier than the lower watershed sites, at around 8:00, and were generally consistent over time. The lowest concentrations at all of these sites were seen later in the afternoon, around 15:30, with late low levels seen later in the season, at around 16:00. Oxygen profiles were completely different at the site downstream of Mason Dam, with the highest concentrations observed near 12:00, while the lowest concentrations were seen at night between 21:00 and 5:00.

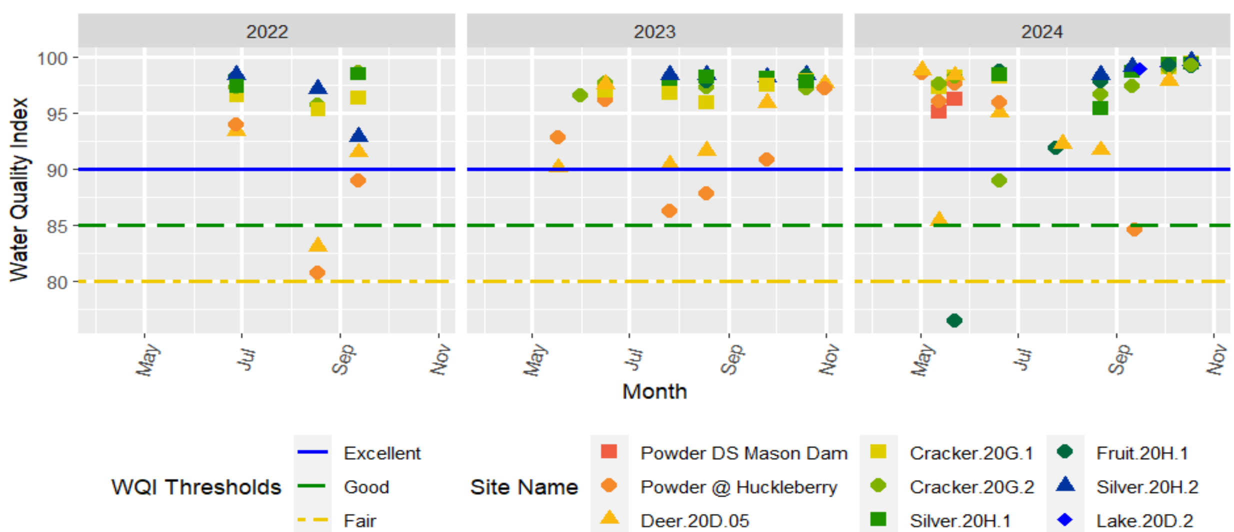
Correlations between dissolved oxygen concentration and temperatures differed significantly between sites. Sensitivity to temperature was highest at Cracker.20G.2, with oxygen concentrations decreasing by 0.36 mg/L per 1 °C increase in temperature, compared to 0.21 mg/L for Deer.20D.05 and 0.20 mg/L for Powder @ Huckleberry. The largest differences were seen at Powder DS of Mason Dam, where oxygen concentration increased by 0.04 mg/L per 1 °C increase in temperature, although this correlation was weak, with an R^2 of 0.0086. This difference is likely related to processes within the reservoir, likely from hypolimnetic discharge and daily patterns from photosynthesis.

Water Quality Index

Trends in WQI for 2022-24 period

WQI scores varied between sites and throughout the season, with the largest source of variation coming from variation between sites. Overall, sites in the Upper part of the watershed had higher WQI scores, with the Silver Creek, Fruit Creek, and Lake Creek sites all having WQI scores over 97, indicating excellent water quality for the 2022-2024 period. WQI scores were lower for the lowest elevation sites in the basin, with Deer.20D.05 and Powder at Huckleberry having mean WQI scores near or less than 90. These values still indicate excellent water quality for these sites, but seasonal and yearly variation did reduce their status to Good or Fair condition.

Chart 9. WQI scores at the Upper Powder watershed monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.



The lowest WQI scores were seen at Fruit.20H.1 on 5/23/2024 where high turbidity values were measured. These high turbidity measurements resulted in extremely low sub-index scores for this parameter, which were by far the lowest among the sites in the Upper Powder and reduced the Total WQI score from an average of 98.72 for the three other sub-indices. Given its outlier status, the index values of this site visit were not included when estimating overall WQI status and trends over the 2022-2024 and 2013-2024 periods.

Differences in WQI between months varied less than for sites but still showed strong seasonal patterns, with the highest WQI scores found in the Spring and Fall, while the lowest values were found in July and August. Plots of WQI scores over time show larger changes in WQI for Powder @ Huckleberry and Deer.20D.05 compared to the other sites, indicating that WQI changes over the course of the season were not equal for all sites and that sites with poor water quality on average saw larger decreases over the summer compared to sites with higher water quality. WQI differences between years were smaller than those seen between months or sites. Overall WQI scores were lower in 2022, while WQI was higher in 2023 and 2024, with mean WQI scores 2.58 and 1.65 points higher, respectively. These differences are likely related to flow, with the higher flows in 2023 associated with higher WQI values.

Table 9a. Estimates of mean WQI scores (from 10-100) for Upper Powder watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Huckleberry	87.76	89.73	91.75
Deer.20D.05	90.94	92.92	94.94
Cracker.20G.1	94.54	96.61	98.72
Cracker.20G.2	94.41	96.45	98.54
Silver.20H.1	95.27	97.49	99.77
Fruit.20H.1	94.77	97.06	99.41
Silver.20H.2	95.66	97.94	100.00

b. Mean monthly WQI differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.34	1.74	3.86
July	-4.70	-2.04	0.69
August	-3.80	-1.65	0.55
September	-1.97	0.18	2.37
October	-0.58	1.78	4.21

c. Mean yearly WQI differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-3.49	-1.41	0.71
2023	-1.01	1.17	3.40
2024	-1.94	0.24	2.48

Among sites visits with Fair (WQI from 80-85) and good (WQI from 85-90) water quality, temperature appeared to have a larger impact on the WQI scores compared to other parameters, with average temperature sub-index scores of 69.06 compared to sub-index scores of 88.37 for dissolved oxygen, 90.07 for pH, and 95.75 for turbidity.

Estimates of WQI for the 2013-2024 period only included sites monitored during the 2022-24 period (9 of 13 sites) and showed similar trends as those found for the 2022-2024 period, with similar patterns in WQI scores seen between sites, months, and years as the 2022-24 estimates. Differences in WQI mean values between years over the 2013-2024 period were found, although none of these changes were significant. The lowest WQI values were found in 2017, with WQI 2.76

points lower than the 2013-2024 mean, while WQI values were highest in 2020, with WQI values 1.15 points higher than the 2013-2024 mean. Estimates of mean yearly WQI for the Upper Powder were much more variable in 2013 and 2020 due to the composition of sites sampled during these years. Low WQI values in 2017 were mostly a result of lower values for the higher elevation sites compared to other years, with lower subindex value for dissolved oxygen at Silver Creek and Lake Creek later in the season (Aug-Sept.) compared to other years (Avg. 90.96 vs. 96.02 for two years prior and after) as well as lower pH subindex values.

Chart 10. WQI scores at the Upper Powder watershed monitoring sites over the 2013-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

Differences in estimates of mean WQI scores between the 2022-24 and 2013-2024 timeframe for the sites showed some patterns related to basin size and elevation, with lower 2013-2024 WQI values seen at the sites on Silver, Lake, and Fruit Creeks when compared to 2022-2024 WQI values. In contrast, the 2013-2024 WQI values for Powder at Huckleberry were higher than those for the more recent monitoring period, indicating that more data might result in slightly better WQI over 2013-2024 period than the most recent period.

Differences in WQI scores between months for the 2013-2024 monitoring period were generally larger than those for the 2022-24 monitoring period. WQI scores were lower on average in August for the 2013-2024 period compared to the more recent monitoring period. Overall, though, differences between site and monthly responses for the 2013-2024 and 2022-2024 monitoring periods were much smaller than the variation in WQI between sites, months, and years within the monitoring periods.

Table 10. Estimates for differences in mean WQI scores for the Upper Powder watershed monitoring sites from mean values over the 2013-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2013	-3.19	0.68	4.71
2014	-1.83	-0.09	1.68
2015	-1.81	-0.47	0.88
2016	-1.27	0.22	1.72
2017	-4.48	-2.76	-1.00
2018	-1.43	0.24	1.93
2019	-1.63	0.60	2.88
2020	-2.90	1.15	5.36
2022	-2.51	-0.92	0.70
2023	-0.32	1.18	2.70
2024	-1.36	0.19	1.76

Like the 2022-2024 monitoring period, temperature was the primary variable resulting in lower WQI among sites with WQI rated as Fair or Good. Average sub-index values for temperature among these sites averaged 71.85, compared to subindex values of 87.82 for Dissolved Oxygen, 89.80 for pH, and 96.05 for turbidity.

Discussion and future plans

Overall water quality in the Upper Powder watershed was mostly excellent, with 6 of the 9 sites having WQI scores over 90 for the 2022-2024 period and 35 out of 42 samples having excellent WQI scores. The biggest factor lowering water quality is primarily from high stream temperatures, with highest number of exceedances and lowest WQI subindex values for this parameter seen in the lower watershed sites. Oxygen is also another important contributor to water quality impairments but is not as much of a driver of water quality impairments in the watershed and is heavily related to temperature impacts.

Dissolved oxygen monitoring found that while oxygen concentrations were consistently above the 8 mg/L minimum for intergravel oxygen concentrations, most sites during both the spring redband trout and fall Bull Trout spawning periods fell below the 11 mg/L spawning standard. Low levels of dissolved oxygen at Cracker.20G.2 in June of 2023 likely indicate broader issues with dissolved oxygen concentrations at the lower elevation sites during this period, likely limiting the suitability of these sites for Redband Trout spawning and rearing. 2025-2027 monitoring will continue to monitor all site monitored during both the spring and fall, with a special focus on conditions in June to better identify reaches where low concentrations of oxygen are present.

Given the importance of the Upper Powder to Bull Trout, continued monitoring at most sites will be needed to ensure conditions are suitable for spawning and rearing. Of the sites monitored in 2022-2024 period for Bull Trout, Silver.20H.1, Fruit.20H.1, and Cracker.20G.2 will be sampled in the 2025-2027 period. The PBWC will also continue to monitor water quality at the Powder @ Huckleberry and Deer.20D.05 to track changes in water quality at these sites and assess how extensive impairments are at these sites. In addition to these sites, 2025-2027 monitoring on Deer Creek above the private in-holding and expanded monitoring at the site downstream of Mason Dam will take place to better identify water quality impairments at sites further downstream.

Looking toward future monitoring efforts, investigations into overall water quality trends within sub-watersheds (HUC-12 basins) are needed to identify how different parameters and overall water quality differ between them. Particularly important focus areas for this type of monitoring include the Deer Creek/Lake Creek watershed due to potential Bull Trout presence as well as the McCully Fork Creek watershed due to its importance as thermal refuge in the summer for redband trout and potential for Bull Trout habitat and reintroduction efforts. Another focus for monitoring should be placed on the Powder River within "Fish the Powder" reach to better track changes in water quality and identify where issues might impact fish suitability. Other locations, such as Powder River tributaries near Bowen Valley, including Beaver Creek, Stices Gulch, and Elk Creek, are important redband trout spawning and migratory corridors, but likely differ from the higher elevation systems located in the foothills of the Elkhorns. Identifying factors related to water quality in these streams might be important for both identifying water quality trends and prioritizing restoration actions in these systems as well.

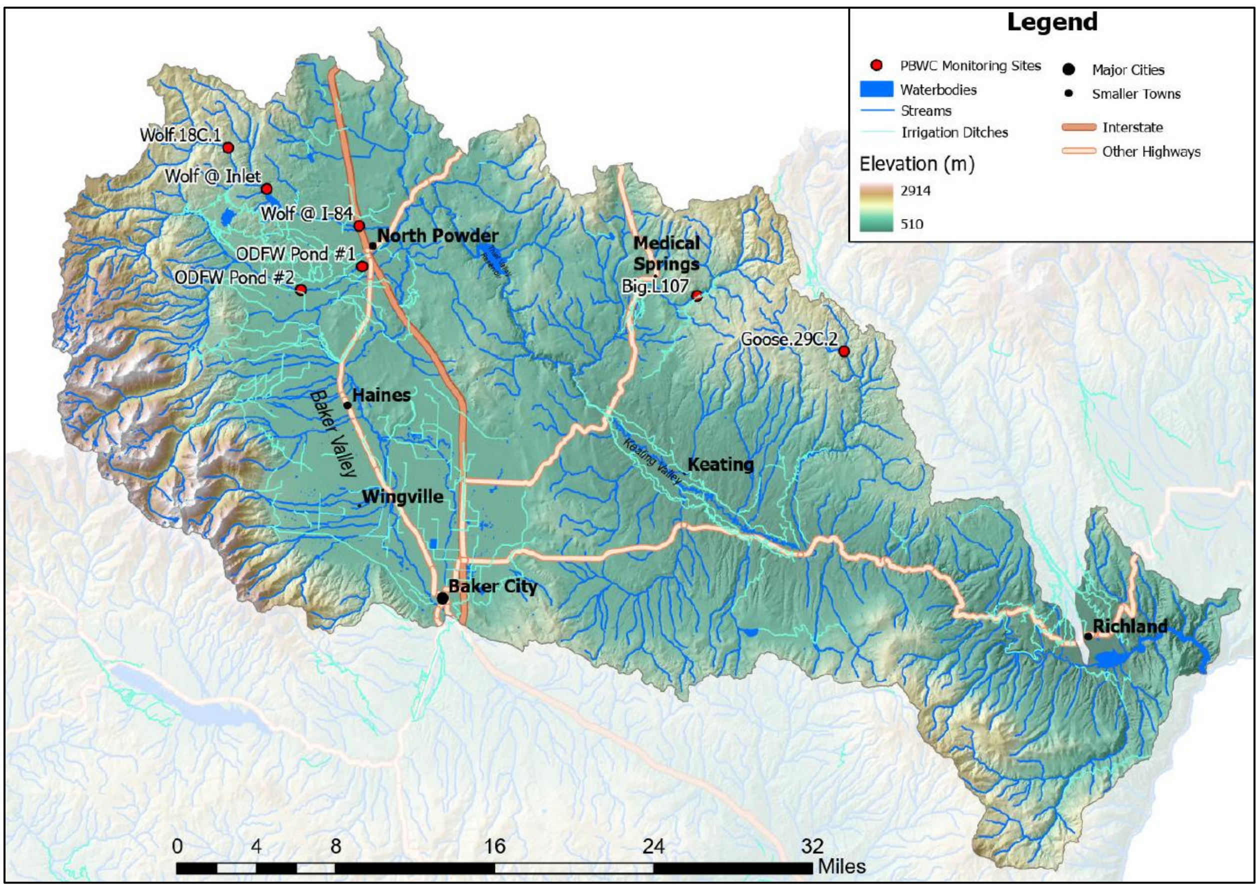
Finally, future monitoring should look into what factors are driving water quality impairments in the watershed, primarily for temperatures. Several factors are potential reasons for temperature impairments, including limited riparian vegetation cover and low summer stream discharge. There are several locations on both public and private land throughout the watershed that may benefit from irrigation improvements and stream restoration. Identifying the causes of impairments in the watershed might be useful in identifying reaches where restoration or improvement actions can improve water quality and suitability for fish while improving flow conditions for downstream reaches.

Middle and Lower Powder River Tributaries

Background

The tributaries of the Powder River in the Middle and Lower Powder River watersheds contain seven monitoring sites (Map 6 and Table 11). These sites are located primarily on the North Powder River and Wolf Creek, with one site each on Big Creek and Goose Creek. Sites on the North Powder River are located at two locations in the lower watershed near quarry ponds managed by ODFW for wildlife habitat and fishing access. Sites on Wolf Creek are located throughout the watershed, with one site located on land managed by WWNF in the Elkhorn foothills, one site above Wolf Creek reservoir, and one site in the lower watershed by Interstate 84. The sites on Big Creek and Goose Creek are located on WWNF in the Wallowa foothills. Most sites were established in either 2013 or 2018, with the site on Goose Creek established in 2023 as part of continuous dissolved oxygen monitoring efforts (Table 11).

Map 6. The Middle and Lower Powder watersheds with tributary sample sites and important features highlighted.



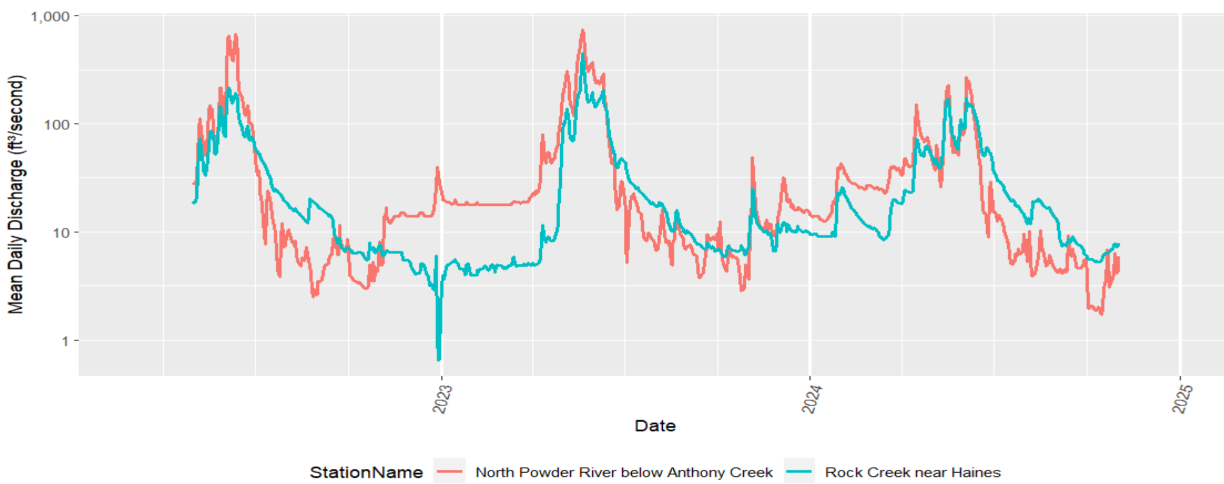
The lower watershed lies within Baker Valley where agriculture is the primary user of land and water. Diversions on tributaries are primarily found on the North Powder River and Anthony Creek, with several associated with Wolf Creek and Pilcher Creek reservoirs. These reservoirs have a total storage capacity of 16,710 acre/ft, are operated by the Powder Valley Water Control District

(PVWCD), with construction completed in 1977 for the Wolf Creek Reservoir and 1984 for Pilcher Creek Reservoir. As part of the construction and implementation of these projects, users were required to update irrigation infrastructure and rotate crops more frequently to improve water efficiency (**Kraynick and Shevener 1981**). Diversion ditches are also found in the foothills of the Wallowa Mountains to transfer water between basins, with one of the largest of these inter-basin transfer ditches found between West Eagle Creek and Goose Creek.

Table 11. Site characteristics of the Middle and Lower Powder River tributary monitoring sites including fish habitat type, Elevation (m), Drainage Area (km^2), Predicted NorWeST 1993-2011 mean August Stream temperature (°C), and established date

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
ODFW Pond #1	37424	Cool	1000	304.68	15.21	5/14/2018
ODFW Pond #2	40718	Cool	1033	294.61	15.10	5/14/2018
Wolf @ I-84	37426	Cool	1006	105.39	16.57	6/1/2013
Wolf @ Inlet	40321	Cool	1136	80.22	14.71	6/1/2013
Wolf.18C.1	37425	Cold	1279	45.91	13.21	6/1/2013
Big.L107	37334	Cool	1106	80.88	14.43	6/24/2013
Goose.29C.2	41791	Cool	1281	24.81	12.95	5/19/2023

Chart 11. Measured discharge for stream gaging stations on Middle and Lower Powder River tributaries over the monitoring period (2022-2024).

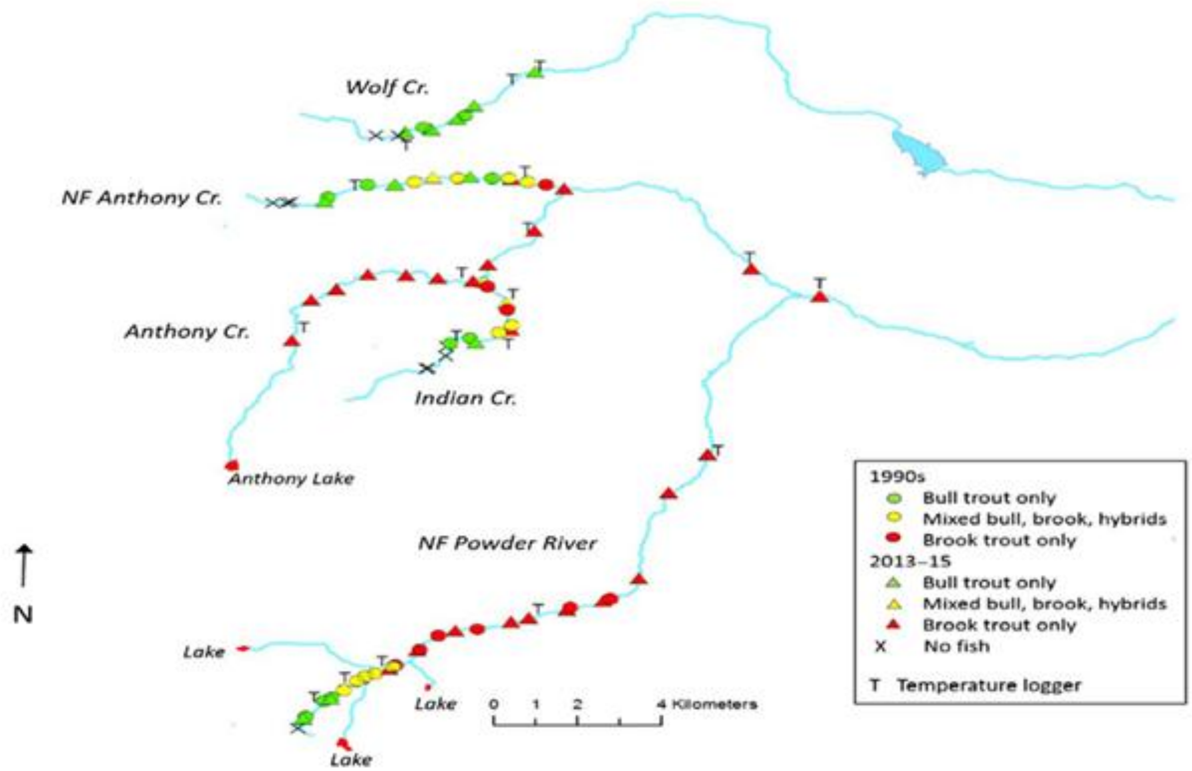


Redband trout habitat exists in numerous cool water tributary systems in the watershed, with resident populations found in cool-water tributaries in the Elkhorn and Wallowa Mountain foothills. Spawning habitat is primarily found in the upper portions of the North Powder River, Rock Creek, Muddy Creek, Pine Creek, Goodrich Creek, Salmon Creek, and Mill Creek. Summer rearing and thermal refuge habitat is also found in many of the same stream reaches, along with the upper sections of Anthony Creek, Wolf Creek, Big Creek, Balm Creek, and Goose Creek. Winter rearing and migratory corridors are found primarily in the lower sections of these tributaries near the Powder River (**NPCC 2004b**).

Bull Trout populations are much more limited and are primarily found in cold water tributary systems higher in the watershed. The most resilient populations are primarily found in Wolf Creek above the Forest boundary where downstream Brook Trout and Bull-Brook Trout hybrids are absent.

Other populations can be found on the North Fork of Anthony Creek, Upper Anthony Creek, and the North Powder River above the confluence with North Fork of the North Powder River. Areas downstream of these reaches contain hybrids and Brook Trout and are at risk of further competition and hybridization from these populations ([Howell 2017](), Figure 15).

Figure 15. Distribution of Bull Trout and Brook Trout in the Middle Powder River tributaries. From Howell 2017.



Due to a late start in 2022, the first samples at the Middle and Lower Powder River tributary monitoring sites were taken in mid-July. Sampling in 2023 and 2024 started earlier in May to identify water quality patterns during spring runoff period. Sites on the North Powder River, Wolf Creek and Big Creek were sampled throughout spring to fall period, while Goose Creek was only sampled alongside dissolved oxygen logger installations and retrievals in the spring of 2023 and 2024.

Grab Sample Monitoring

Dissolved oxygen concentrations (in mg/L) varied between sites, with sites higher in the basin having higher concentrations of dissolved oxygen and levels of saturation closer to 100% throughout the year. Oxygen measurements were more variable in the lower watershed, with these sites containing both high and low oxygen concentrations and saturation compared to the upper and foothill sites.

Estimates found that the lower watershed sites at ODFW Pond #1 and Wolf @ I-84 had the lowest mean oxygen concentrations among the Powder River tributary sites. Contributions from cooler and less oxygenated groundwater at baseflow are likely higher at these sites compared to others located higher in the watershed, with these contributions highest at ODFW Pond #1, based on cooler temperatures, lower pH, lower oxygen concentrations, and high conductivity levels. Dissolved oxygen concentrations also fell below the 6.5 mg/L cool water DEQ standards at ODFW Pond #1 later in the summer into early fall each year, with 4.79 mg/L seen in September 2024.

July and August were the months where low concentrations of oxygen were most likely to occur, with average oxygen levels more than 0.5 mg/L lower than the mean, although this was not the case for all sites, with oxygen concentrations higher during these months at ODFW Pond #2 and at Wolf at I-84. Average measurements also differed between years, although not significantly, with lower oxygen concentrations found in 2022, while the highest concentrations were seen in 2023.

Chart 12. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period. Cool-water standard (red line) highlighted.

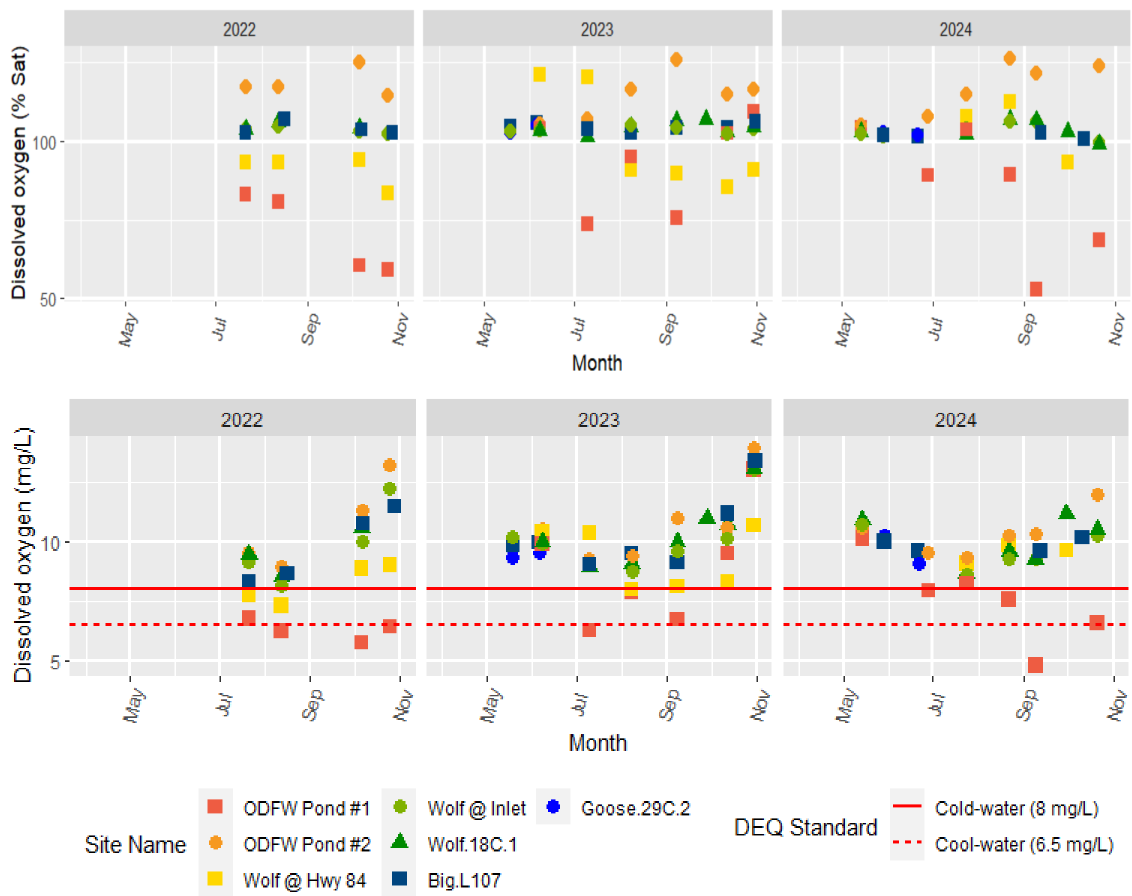


Table 12a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
ODFW Pond #1	6.22	7.31	8.39
ODFW Pond #2	9.26	10.35	11.43
Wolf @ I-84	7.71	8.84	9.97
Wolf @ Inlet	8.43	9.56	10.69
Wolf.18C.1	8.70	9.84	10.98
Big.L107	8.58	9.67	10.75

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.89	0.19	1.28
July	-1.40	-0.55	0.31
August	-1.46	-0.61	0.25
September	-1.37	-0.36	0.66
October	0.55	1.32	2.09

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-1.60	-0.51	0.57
2023	-0.44	0.49	1.42
2024	-0.95	0.03	1.00

pH at all monitoring sites fell within ranges suitable for aquatic life and appeared to be heavily related to concentrations and saturations of dissolved oxygen, with higher pH values observed at ODFW Pond #2, while lower pH values were seen at ODFW Pond #1 and Wolf @ I-84. pH was more stable at the foothill sites but was more variable on Big Creek compared to Upper Wolf Creek. Seasonal pattern in pH were hard to determine, and appeared to be mostly consistent throughout the year, with only measurements in October differing from other monthly values. Yearly differences were also small, with higher pH seen in 2023 and 2024 than 2022.

Chart 13. Observed pH measurements *for the Middle and Lower Powder River tributary monitoring sites* over the 2022-2024 period. Upper (dashed line) and lower (solid line) recommended pH standards highlighted

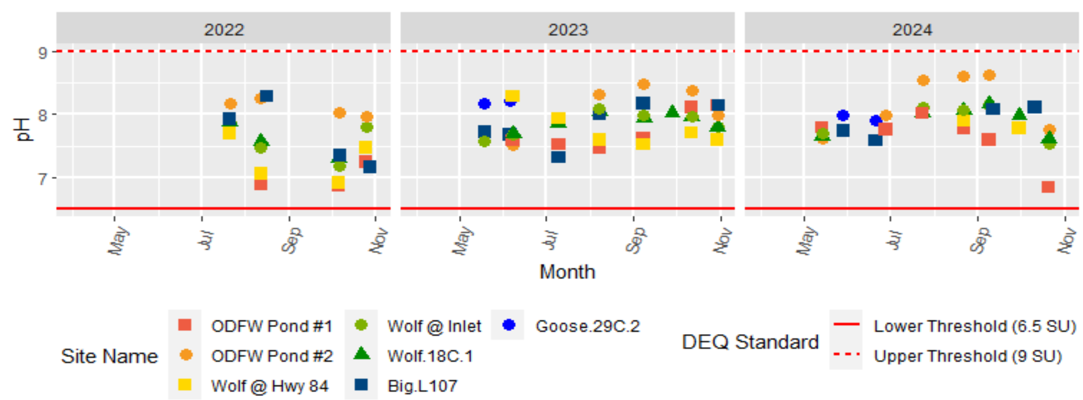


Table 13a. Estimates of mean pH for the Middle and Lower Powder River tributary sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
ODFW Pond #1	7.25	7.53	7.80
ODFW Pond #2	7.88	8.16	8.43
Wolf @ I-84	7.35	7.63	7.92
Wolf @ Inlet	7.53	7.82	8.10
Wolf.18C.1	7.50	7.79	8.08
Big.L107	7.58	7.86	8.13

b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.45	-0.17	0.10
July	-0.09	0.12	0.34
August	-0.15	0.06	0.28
September	-0.16	0.10	0.36
October	-0.30	-0.11	0.08

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.50	-0.23	0.05
2023	-0.12	0.11	0.35
2024	-0.13	0.12	0.36

Conductivity measurements heavily followed basin size and indicated large amounts of groundwater inputs at the lower watershed sites, with particularly large increases in conductivity seen between upper and lower sites on Wolf Creek and between the ODFW Pond sites on the North Powder River. Conductivity was highest at the lower watersheds site in the late summer and fall when surface flow contributions were lowest. Seasonal patterns in conductivity were relatively similar between years, with an increasing trend in conductivity from spring through fall. Seasonal changes in conductivity were lower at the upper watershed sites, with larger variations seen at Wolf @ I-84 and ODFW Pond #1, particularly from earlier in each season. Differences between years were small, with large variation seen within years, but demonstrated some patterns related to flow, with lower conductivity measurements seen in 2023 and higher measurements seen in 2024.

Chart 14. Observed conductivity measurements (in µS/cm) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period.

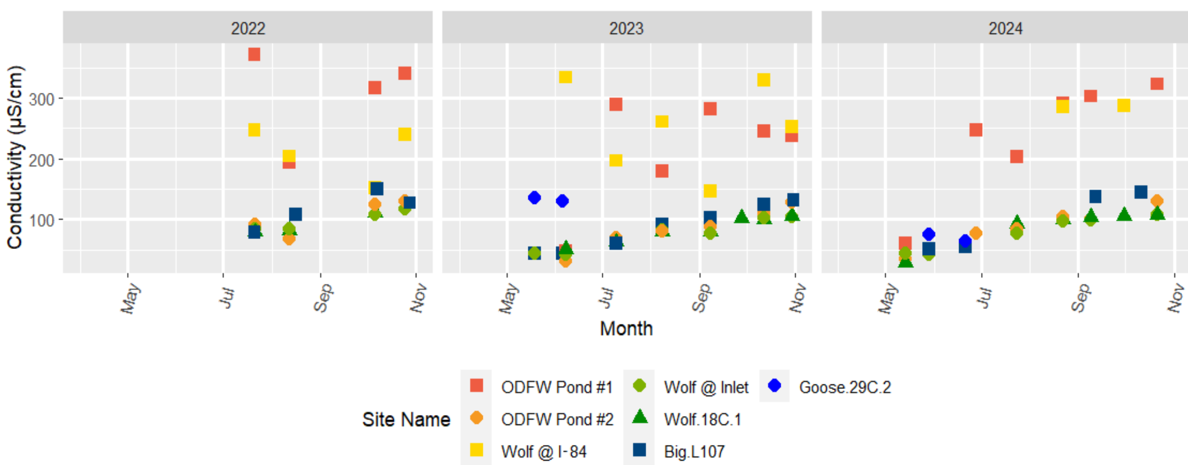


Table 14a. Estimates of mean conductivity (in µS/cm) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
ODFW Pond #1	181.99	232.10	296.00
ODFW Pond #2	68.16	86.92	110.86
Wolf @ I-84	175.79	226.48	291.78
Wolf @ Inlet	63.26	81.46	104.91
Wolf.18C.1	65.37	84.32	108.77
Big.L107	74.89	95.47	121.71

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-121.19	-91.10	-52.73
July	-50.45	-12.44	33.59
August	-36.02	5.03	54.76
September	-26.56	25.56	91.02
October	24.87	72.95	130.09

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-31.52	-1.43	36.94
2023	-36.95	-12.90	16.74
2024	-16.13	14.32	52.21

Turbidity measurements followed basin size for most sites, with lower mean turbidity measurements at Wolf.18C.1 and Big.L107, and the highest mean turbidity seen at ODFW Pond #2 and Wolf @ I-84. Turbidity measurements were highest at Goose Creek, where observed turbidity measurements were exceeded 80 NTU's in both 2023 and 2024, above levels that can significantly impact salmonid health, feeding, and spawning. Turbidity was also above 20 NTU's on lower Wolf Creek in October 2023. While below the high turbidity cutoff, the North Powder River monitoring sites and Wolf Creek above the reservoir had moderate levels of turbidity above 10 NTUs observed during the spring runoff period.

Turbidity was highest in the spring and lowest in the summer and early fall, primarily related to flow and runoff. All sites showed an increase in turbidity in October of 2023, but this pattern wasn't present for this month in 2022 and 2024. Larger seasonal changes in turbidity were seen at the lower watershed sites than those higher in the watershed, particularly for Wolf @ I-84. Differences in turbidity were present between years but didn't follow patterns related to flow like the seasonal patterns, with 2022 having highest mean turbidity and 2024 having the lowest.

Chart 15. Observed turbidity measurements (in NTU's) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period. Log transformation used for turbidity measurements.

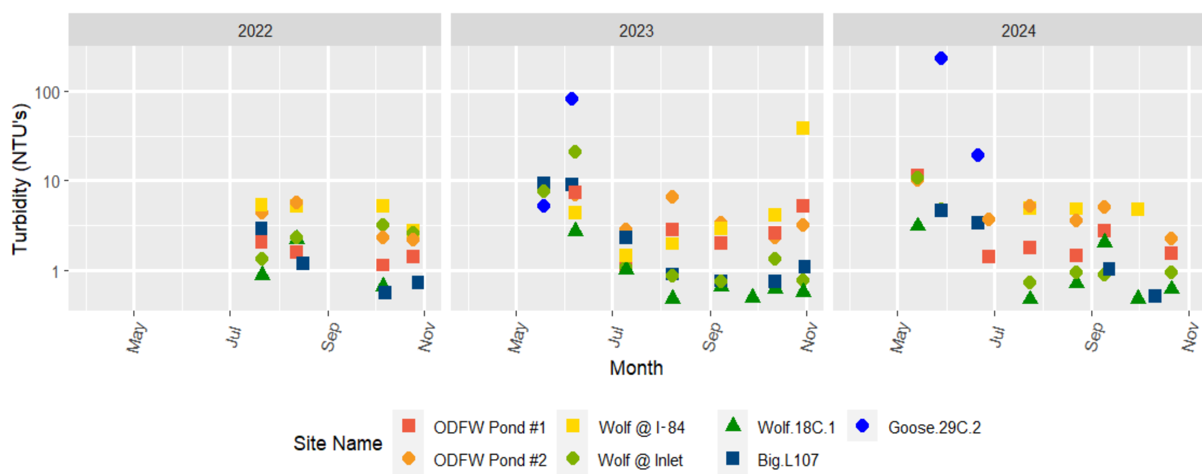


Table 30a. Estimates of mean turbidity (in NTU's) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
ODFW Pond #1	1.41	2.53	4.52
ODFW Pond #2	2.52	4.51	8.08
Wolf @ I-84	3.06	5.61	10.28
Wolf @ Inlet	1.04	1.91	3.50
Wolf.18C.1	0.58	1.07	1.96
Big.L107	0.85	1.52	2.72

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	0.52	3.23	8.09
July	-1.57	-0.79	0.45
August	-1.50	-0.68	0.63
September	-1.67	-0.77	0.79
October	-1.65	-1.00	-0.01

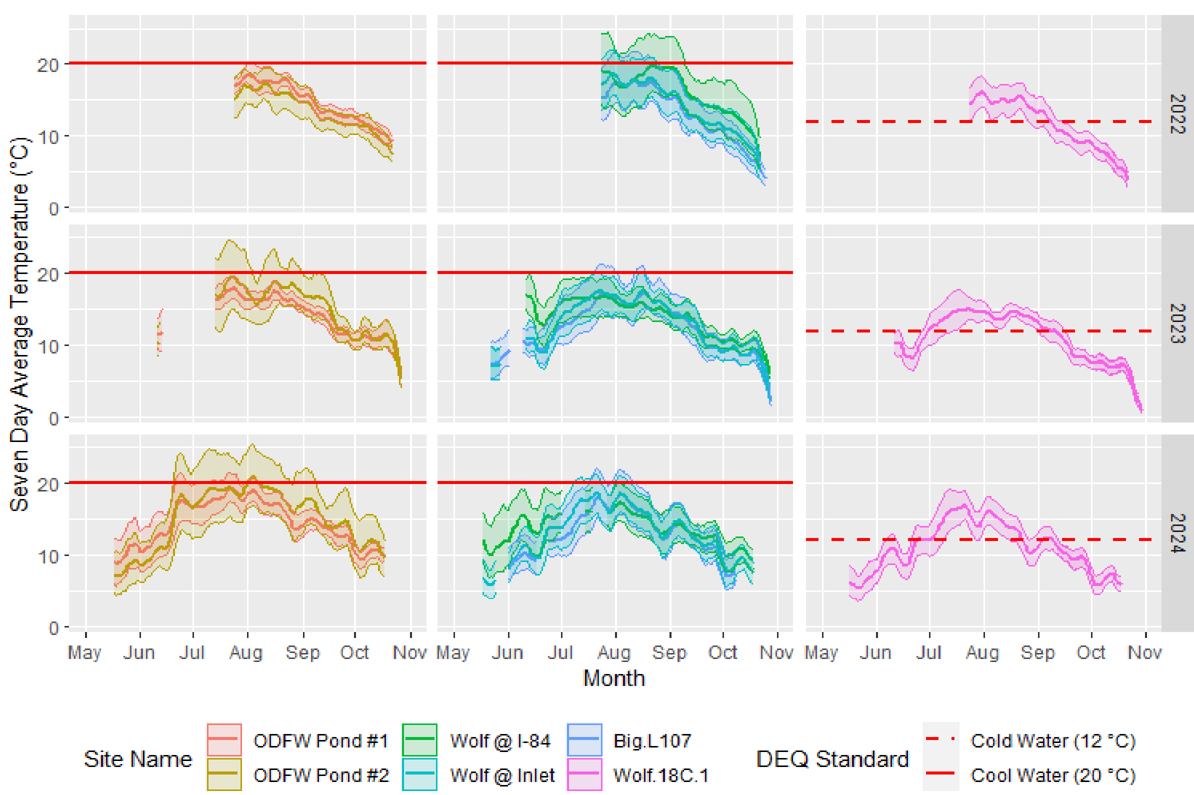
c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-1.93	0.78	5.64
2023	-2.01	0.16	3.73
2024	-2.74	-0.94	2.11

Stream temperature monitoring

Stream temperatures at all sites were highest during July and August. Sites where maximum temperatures were above the 20 °C cool water standard were found in the upper, middle, and lower watershed, particularly at Wolf @ Inlet, Wolf @ I-84, and at ODFW Pond #2, each of which had between 20% and 90% in July and August with maximum temperatures above 20 °C. Other sites with a significant number of days with maximum temperatures above 20 °C included Big.L107 and at ODFW Pond #1. Alongside temperature exceedance in the cool-water streams, Wolf.18C.1 also had every day in July and August with maximum temperatures above the 12 °C cool water standard.

Chart 16. Temperature profiles (in °C) for the Middle and Lower Powder tributary monitoring sites over the 2022-2024 period, with North Powder River sites on the left, lower Wolf Creek and Big Creek sites in the middle, and upper Wolf Creek on the right. Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cold water (dashed line) and cool water (solid line) standards highlighted in red based off relevant fish use at the site.



Cooler temperatures were found in the spring and fall period. Fall temperatures were less suitable for bull trout in September at Wolf.18C.1, with 42% of days having maximum temperatures above12 °C, but were more suitable in October, where every day had temperatures below the cold-water standard. In the spring, both Wolf @ Inlet and Wolf.18C.1 had every day in May below the cold-water standard during the Redband spawning period. Temperatures at these sites were less suitable for spawning in June, with maximum temperatures above 12 °C for over 50% of days during this month.

Warm water conditions were more likely to occur in 2022 compared to 2023 or 2024, with mean temperatures 0.33 °C higher and maximum temperatures 0.50 °C higher than average. In comparison, higher flows in 2023 were a likely contributor to cooler temperatures during this year, with mean temperatures 0.36 °C and maximum temperatures 0.64 °C lower than average.

Table 15a. Estimates of mean and Maximum Daily temperatures (in °C) for the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 period

Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
ODFW Pond #1	13.32	14.15	14.98	14.49	15.79	17.08
ODFW Pond #2	13.44	14.27	15.10	16.15	17.45	18.75
Wolf @ I-84	13.50	14.33	15.16	15.88	17.18	18.48
Wolf @ Inlet	12.28	13.11	13.94	14.27	15.57	16.86
Wolf.18C.1	10.28	11.11	11.94	11.44	12.74	14.03
Big.L107	11.56	12.39	13.22	14.07	15.37	16.67

b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Month	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-2.17	-1.34	-0.51	-2.33	-1.04	0.26
July	2.65	3.34	4.03	2.96	4.04	5.13
August	2.31	3	3.69	2.08	3.16	4.25
September	-1.12	-0.43	0.26	-1.96	-0.88	0.21
October	-5.26	-4.57	-3.88	-6.38	-5.3	-4.21

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Year	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.50	0.33	1.16	-0.80	0.50	1.80
2023	-1.09	-0.36	0.37	-1.78	-0.64	0.50
2024	-0.70	0.03	0.76	-1.00	0.14	1.28

Every site except Wolf @ I-84 had higher observed mean August temperatures than the NorWeST 1993-2011 predicted average. ODFW Pond #2 had the largest increase in temperatures, with observed 2022-2024 mean August temperatures 2.35 °C higher than the 1993-2011 estimate. Big.L107, ODFW Pond #1, and Wolf @ Inlet also showed higher temperatures than the NorWeST model, with each having 2022-2024 mean August temperatures over 1.7 °C higher than the 1993-

2011 estimate. All of these sites also had higher observed 2022-2024 mean August temperatures than the NorWeST estimates for 2040 as well. Only Wolf @ I-84 had lower observed mean August temperatures than the NorWeST estimates, likely due to higher groundwater inputs.

Table 16. Estimated and Observed mean August stream temperatures for the Middle and Lower Powder River tributary monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

Site	Estimated 2022-2024 Mean August Temp (°C)	Observed 2022-2024 Mean August Temp (°C)	NorWeST 1993-2011 Temp (°C)	NorWeST 2040 Temp (°C)	NorWeST 2080 Temp (°C)
ODFW Pond #1	17.15	16.82	15.21	16.43	17.20
ODFW Pond #2	17.27	17.45	15.10	16.31	17.08
Wolf @ I-84	17.33	16.32	16.57	17.82	18.61
Wolf @ Inlet	16.11	16.47	14.71	15.92	16.68
Wolf.18C.1	14.11	14.14	13.21	14.37	15.11
Big.L107	15.39	16.17	14.43	15.62	16.38

Dissolved Oxygen Monitoring

Fall

Dissolved oxygen loggers were installed at Wolf.18C.1 to measure oxygen concentrations during the fall Bull Trout spawning season each year for the monitoring period. Overall oxygen concentrations were consistently above the 8 mg/L minimum intergravel concentrations but were rarely above the 11 mg/L spawning standard. While oxygen concentrations increased throughout the fall, they were only above 11 mg/L for 10.5% of measurements in October.

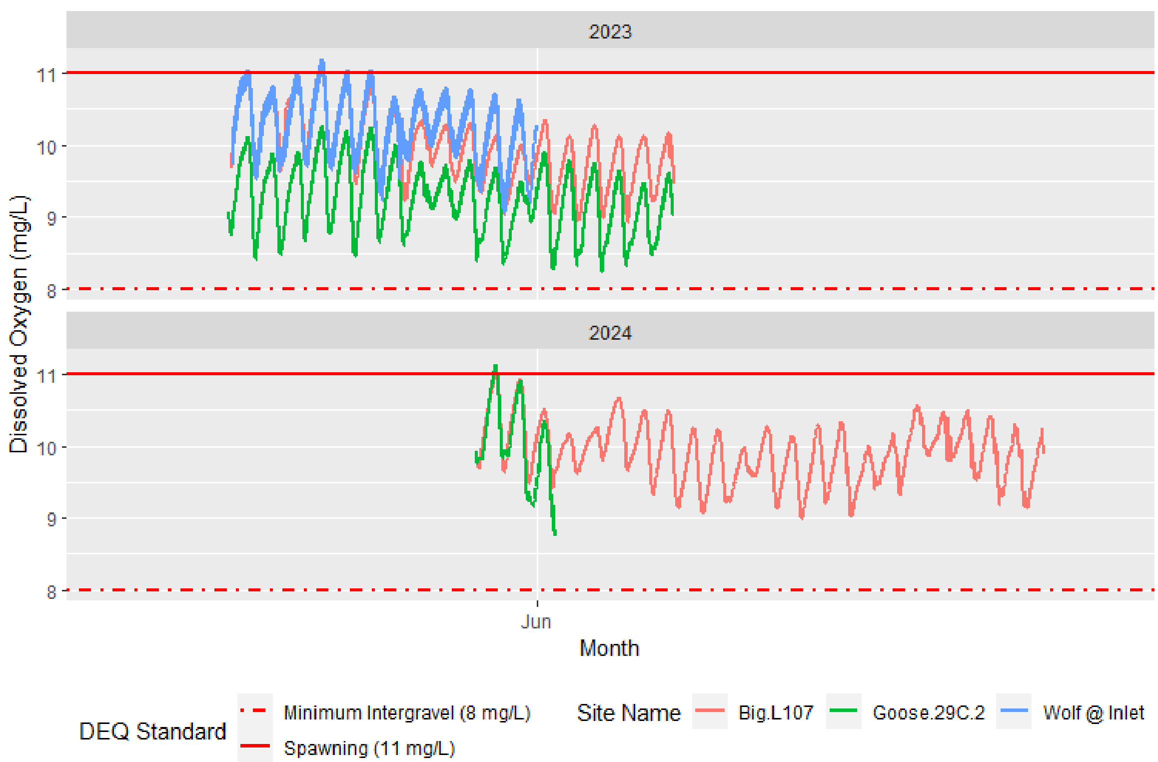
Chart 17. Dissolved oxygen profiles (in mg/L) for the Fall Wolf Creek oxygen logger monitoring site over the 2022 to 2024 period. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.

Daily ranges in oxygen concentration were lower in the fall at Wolf.18C.1 than the sites monitored in the spring, with daily ranges of 1.06 mg/L compared to 1.52 mg/L for the spring sites. Timing of maximum oxygen concentrations differed from many oof the fall sites, with the highest oxygen concentration observed at around 11:00 in the morning, with the lowest oxygen concentrations seen between 6:00 and 7:00 in the evening. Oxygen concentrations were still heavily related to temperature, with concentrations decreasing by 0.269 mg/L per 1 °C increase in temperature (Appendix D).

Spring

Dissolved oxygen loggers installed at Wolf @ Inlet, Goose.29C.2, and Big.L107 were used to measure oxygen concentrations throughout the spring Redband Trout spawning season in 2023 and 2024. All oxygen measurements at the sites were above the 8 mg/L intergravel minimum standard, but rarely exceeded the 11 mg/L spawning standard. Only Wolf @ Inlet in 2023 and Goose.19C.2 in 2024 had oxygen concentrations exceeding 11 mg/L and comprised less than 3% of all measurements taken in May. The highest concentrations were found at Wolf @ Inlet, while lower concentrations were seen at Goose.29C.1, likely related to the high turbidity observed at the site. Overall oxygen concentrations decreased slightly from spring to early summer.

Chart 18. Dissolved oxygen profiles (in mg/L) for the Spring Middle and Lower Powder River tributary oxygen logger monitoring sites in 2023 and 2024. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



Oxygen concentrations at these sites were highest in early morning, around 7:00, when temperatures were coldest, with the lowest concentrations seen between 1:00 to 4:00 in the afternoon, with later minimum levels seen later in the season. Among the sites monitored in the spring, Goose.29C.2 had the largest daily range in oxygen concentrations, 1.65 mg/L, while Big Creek had the lowest daily range at 1.38 mg/L. No differences were noticeable between years or between months within seasons. Oxygen sensitivity to temperature was highest at Wolf @ Inlet, with concentrations decreasing by 0.253 mg/L per 1 °C increase in temperatures, while sensitivity was lowest at Goose.29C.2, where oxygen concentrations decreased by 0.208 mg/L per 1 °C increase in temperatures. Sensitivity was intermediate at Big.L107 (Appendix D).

Water Quality Index

Sites with consistently excellent water quality included Big.L107, Wolf @ Inlet, and Wolf.18C.1. Water quality at these sites only fell below excellent levels twice during high turbidity events in the spring 2023 at Wolf @ Inlet and high temperatures at both sites observed in August 2022.

Water quality impairments were more frequently observed at sites lower in the watershed, with Wolf @ I-84, ODFW Pond #2, and ODFW Pond #1 having water quality fall below the "Fair" threshold eight times throughout the monitoring period. Temperature was an important contributor to these lower WQI values at Wolf @ I-84 and ODFW Pond #2, while dissolved oxygen was an important contributor to low WQI values for ODFW Pond #1. Moderately low dissolved oxygen sub-index values were also found at ODFW Pond #2 and Wolf @ Inlet. High pH at ODFW Pond #2 contributed to low overall WQI values there as well.

Chart 19. WQI scores at the Middle and Lower Powder River tributary monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

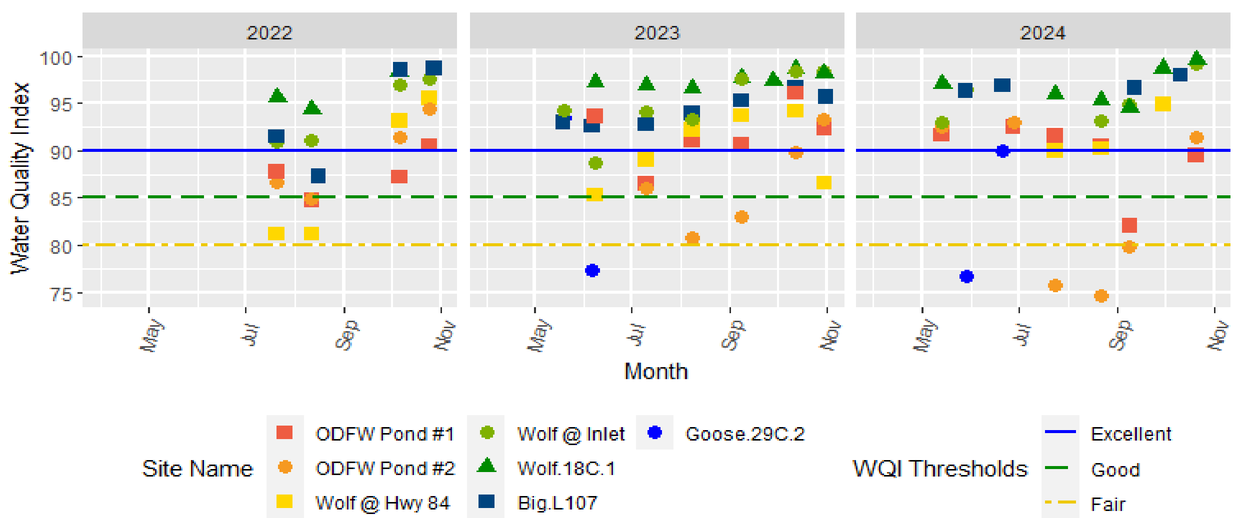


Table 30a. Estimates of mean WQI scores (from 10-100) for the Middle and Lower Powder River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
ODFW Pond #1	85.79	89.28	92.90
ODFW Pond #2	82.59	85.94	89.43
Wolf @ I-84	85.93	89.57	93.36
Wolf @ Inlet	90.47	94.29	98.28
Wolf.18C.1	92.78	96.72	100.84
Big.L107	90.43	94.09	97.90

b. Mean monthly WQI differences from mean values over the 2022-2024 period

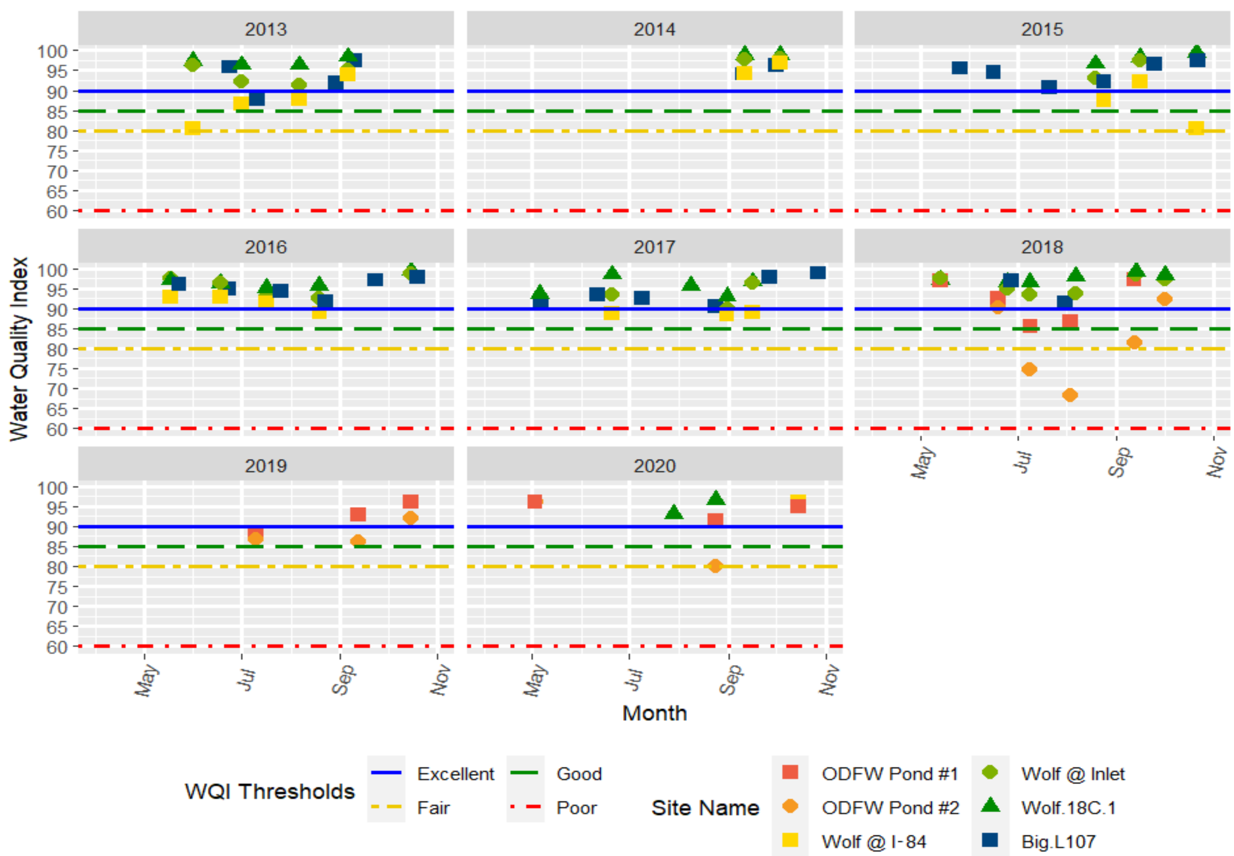
Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-2.30	1.20	4.85
July	-4.55	-1.88	0.88
August	-5.09	-2.43	0.31
September	-3.46	-0.23	3.13
October	0.77	3.33	5.96

c. Mean yearly WQI differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-4.17	-0.67	2.97
2023	-2.69	0.35	3.50
2024	-2.87	0.32	3.62

Water quality was generally lowest in July and August, with average WQI scores 1.88 to 2.43 points lower than average. Summer WQI changes were larger for the lower watershed sites than those higher in the watershed. Temperature was the predominant factor for these lower WQI values, with an average sub-index value of 80.7 for all sites during these months. Dissolved oxygen was also an important contributor, with average subindex values of 86.4 for all site during the same months. Lower temperatures and turbidity values resulted in higher WQI values in the fall, with average values over three points higher than average. WQI was somewhat lower in 2022 compared to 2023 and 2024, although the difference between years were small.

Chart 20. WQI scores at the Middle and Lower Powder River tributary monitoring sites over the 2013-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WQI), and Poor (80-60 WQI) thresholds highlighted.



For WQI trends since 2013 at the six monitored sites, differences between WQI over the years was generally small. WQI scores were higher than average in 2019 and 2020, but this difference was likely an artifact resulting from the lower number of monitored sites during these years. Lower WQI scores were also observed in 2013, 2015, 2017, and 2022, likely due to persistent drought conditions during these years.

Table 17. Estimates for differences in mean WQI scores for the Middle and Lower Powder River tributary monitoring sites from mean values over the 2013-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2013	-3.47	-1.03	1.47
2014	-2.67	0.35	3.47
2015	-3.37	-0.71	2.04
2016	-2.22	0.34	2.97
2017	-3.24	-0.64	2.04
2018	-3.00	-0.80	1.47
2019	-1.35	1.97	5.42
2020	-1.38	1.84	5.16
2022	-3.27	-0.88	1.58
2023	-2.00	0.10	2.25
2024	-2.75	-0.54	1.72

Discussion

Hydrology differs between different locations within the watershed, with monitoring locations in the upper and foothill regions showing more stable flow regimes throughout the year. Lower in the watershed, sites demonstrated more flashy conditions, with higher flows in the spring and stronger shifts to groundwater flow as upstream surface flow contributions declined over the summer (Figure 16). As a result, seasonal and spatial patterns in water quality for Middle and Lower Powder tributaries appear to be more dependent on these changes in water source, particularly for monitoring sites lower in the watershed.

At ODFW Pond #1, lower levels of oxygen saturation and higher conductivity measurements, alongside lower levels of discharge, indicate that groundwater contributions are a higher percentage of surface flow than the other tributary sites. These high groundwater inputs result in cooler temperatures but also lower concentrations of dissolved oxygen. Further upstream at ODFW Pond #2, larger daily fluctuations in daily temperatures and higher concentrations and saturation of oxygen measurements indicate larger surface flow contributions. High pH measurements in the summer indicate that eutrophication might also be an issue at this site. While oxygen and temperature profiles were less indicative of large groundwater contributions at Wolf @ I-84, these parameters were still indicative of significant amounts of groundwater at this site, particularly in the summer.

While less prevalent than the lower watershed sites, warm water conditions were still present at Big Creek and Wolf Creek above the reservoir, with maximum temperatures exceeding 20 °C at these sites in July and August. Temperatures exceeding 12 °C were also common at Wolf.18C.1, including a significant portion of September when Bull Trout spawning is present. While oxygen concentrations were consistently above 8 mg/L at these upper elevation sites, they are generally low during the spring redband trout and fall Bull Trout spawning periods and might limit spawning presence and successful egg incubation in these reaches. Goose.29C.2 had some of the largest impairments among the sites where fish spawning conditions were monitored, primarily due to high levels of turbidity regularly exceeding 80 NTU's, above levels generally considered suitable for salmonids.

Figure 16. Difference in Stream flow conditions on the North Powder River at ODFW Pond #1 (top) and ODFW Pond #2 (bottom) between May and July 2024

Elsewhere in the basin, currently planned monitoring efforts are focused on PBWC priority basins, particularly related to bacteria and nutrients in the lower watershed. Current efforts are monitoring these pollutants at sites on the North Powder River and identifying sources with a focus on prioritizing improvement actions in the broader Powder sub-basin. Elsewhere, monitoring efforts are also currently focused on cataloging turbidity at Goose.29C.2 to identify when turbidity problems are present.

Mesic habitats are important features for the sagebrush ecosystems found in foothill-valley transitional areas and in the tributaries of Keating south of the valley. A better understanding of temperature patterns are needed in these stream reaches for Redband Trout suitability and restoration potential. Higher in the watershed, a better understanding of dissolved oxygen and temperature patterns would be helpful in identifying spawning and thermal refugia for these systems. There are significant gaps regarding the status and suitability of Upper Wolf Creek, North Powder River, and Anthony Creek for Bull Trout and the Wallowa foothill tributaries such as Big Creek, Goose Creek, and Balm Creek for Redband Trout.

Finally, there is a need to identify potential influences of planned restoration on Anthony Creek on water quality. Better coverage of the both the North Powder River and Anthony Creek on the Elkhorn Wildlife Area with temperature loggers is proposed for restoration reaches on both stream reaches as part of effectiveness monitoring of this project. Changes related to these projects are more likely to be found within and immediately downstream of restoration reaches, but some longer-term changes might be noticeable in the downstream sections of the North Powder River.

Powder River downstream of Bowen Valley

Background

The Middle and Lower watersheds of the Powder River drain an area of 3,028 km^2 (1,169 mi^2) and begins downstream of Bowen Valley. River sections near Baker City are still heavily influenced by Mason Dam, with high flows during the irrigation season and lower flows during the period when the dam is refilling from November through April. Downstream of Baker City, the Powder River flows through Baker Valley, a wide, flat alluvial fill plain with productive agricultural areas and a significant number of irrigation diversions and canals. The Powder River then turns to the southeast, flowing through constrained, canyonlike reach before entering Thief Valley Reservoir. Construction of the reservoir finished in 1937 with a storage capacity of 17,400 acre-feet (**USBR 2026**). Downstream of Thief Valley Dam, the Powder River flows through Keating Valley, which, like Baker Valley, contains a significant number of irrigation diversions and canals and flows through significant agricultural and grazing areas. After flowing through another constrained canyon reach, the Powder River joins Eagle Creek near Richland before flowing into Brownlee Reservoir.

Map 7. The Middle and Lower Powder River watersheds with sample sites and important features highlighted. Powder River delineated with the thicker blue line.

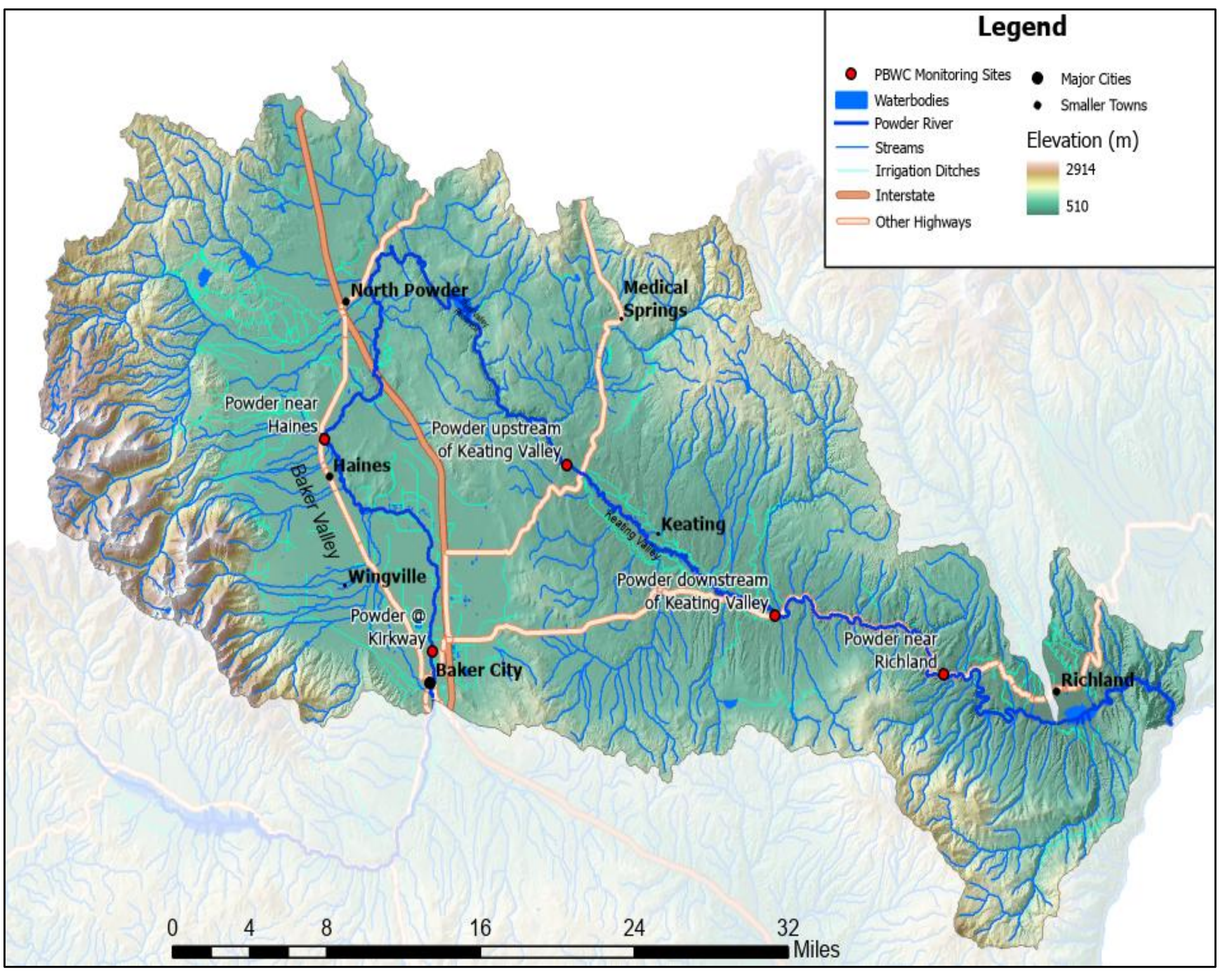


Table 18. Site characteristics of the Middle and Lower Powder River monitoring sites including fish habitat type, Elevation (m), Drainage Area (km²), Predicted NorWeST 1993-2011 mean August Stream temperature (°C), and established date

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
Powder @ Kirkway	37328	Cool	1043	771.02	16.59	6/25/2013
Powder near Haines	37786	Cool	1007	1301.29	16.31	5/19/2014
Powder DS Keating	10724	Cool	804	2916.74	17.49	5/19/2014
Powder near Richland	41727	Cool	697	3240.99	17.69	5/19/2014
Powder US Keating	40377	Cool	844	2474.48	17.46	6/24/2013

Thief Valley Reservoir has a large impact on both flow regimes and water quality downstream, particularly locations above Keating Valley, with higher flows in the summer and lower flows in fall (Chart 21). Alongside these flow changes are algal blooms in the reservoir. During these blooms, chlorophyll a concentrations regularly exceed the 24 µg/L limit accepted as the safe threshold for human and fish use, with estimates as high as 140.8 µg/L observed on August 17th,2024. The blooms were particularly prevalent in the summer and fall, with harmful algal bloom levels occurring for more than 50% of days between June and September (Chart 22). Photosynthesis and decomposition from these blooms likely have impacts on downstream water quality.

Chart 21. Measured stream discharge (in ft^3/sec) for gaging stations on the Powder River over the 2022-2024 monitoring period. Discharge on log scale.

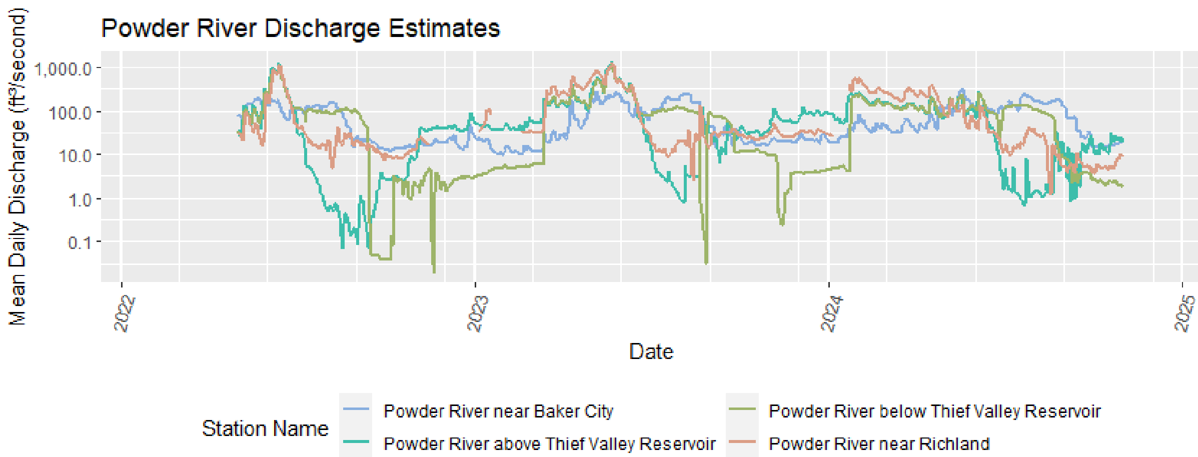
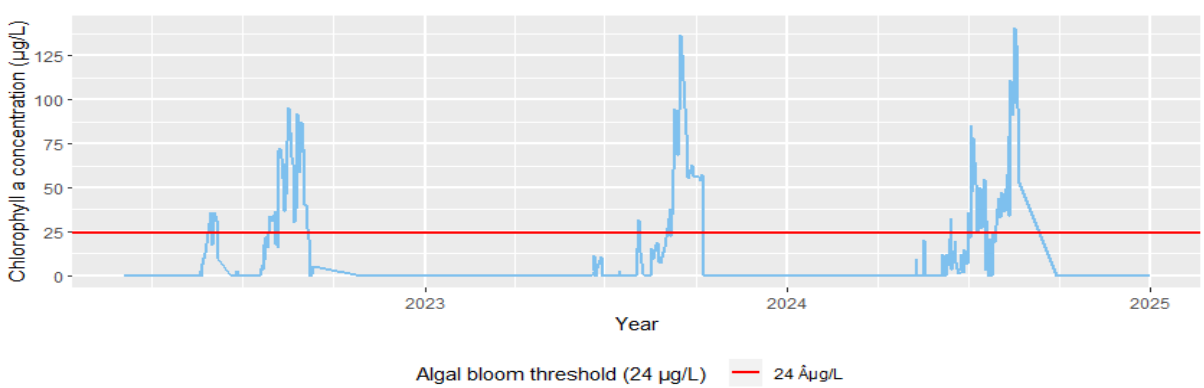


Chart 22. Estimates of algal productivity (chlorophyll a in µg/L) for Thief Valley Reservoir for the 2022-2024 monitoring period. Harmful algal bloom threshold (24 µg/L) highlighted in red. Data from the Cyanobacteria Assessment Network (CyAN).



Redband trout primarily use the Powder River as winter habitat. Important stream reaches with high utilization during the winter include section between the Ritter Creek and Goose Creek confluences, sites in Keating downstream of Big Creek, and the section downstream of Thief Valley Dam, with some usage of Thief Valley Reservoir as habitat. Other sections have seen use as migratory corridors, but lack required habitat conditions for more intensive usage otherwise (**NPCC 2004b**)

Due to access issues and a late start to the season, no temperature loggers were installed at the Powder near Haines site in 2022. Similarly, no grab sample or temperature data was taken at Powder DS Keating in 2022. Further upstream, changes in flow at Powder upstream of Keating Valley made it difficult to keep the temp logger consistently submerged during the late summer and fall period. In addition to these issues at Powder upstream of Keating Valley, the measurements taken at this site in 2024 did not meet the required accuracy and precision standard for analysis.

Grab Sample Monitoring

Oxygen levels were above the 6.5 mg/L cool-water standard for most sites throughout the year, with only one measurement at the site downstream of Keating having dissolved oxygen concentrations below the cool-water standard in October 2024. Lower oxygen saturation and concentrations were more commonly seen in the summer, with higher concentrations seen in the fall, appearing to be primarily driven by cooler temperatures.

Table 19a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the Powder River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Kirkway	7.94	9.76	11.58
Powder near Haines	7.76	9.91	12.05
Powder US Keating	7.74	9.81	11.89
Powder DS Keating	7.66	9.87	12.09
Powder near Richland	7.31	9.18	11.05

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-2.22	-0.35	1.52
July	-2.16	-0.51	1.13
August	-3.51	-1.63	0.26
September	-1.05	0.84	2.73
October	-0.04	1.65	3.33

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-1.76	0.11	1.98
2023	-1.04	0.56	2.16
2024	-2.45	-0.66	1.12

High oxygen saturation was seen downstream of Keating Valley and near Haines, while lower levels of saturation were seen near Richland. Oxygen saturation was generally stable at Kirkway but was more variable at other sites on the Powder River. The site downstream of Keating, in particular, had a high degree of variability, with oxygen saturation above 150% observed in July and September 2023 and near 60% in October 2024. Similar patterns were seen near Haines, although the changes were not as large as those seen downstream of Keating Valley. In general, this high level of variability obscured site, monthly, and yearly patterns in dissolved oxygen concentrations.

Chart 23. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the Powder River monitoring sites over the 2022-2024 period. Cool water standard (dashed red line) highlighted.

pH at the Powder River sites were generally high, particularly for the Powder upstream of Keating, where mean pH over the 2022-2024 period was 8.80. pH was above the upper recommended limit near Haines, upstream of Keating Valley, and near Richland seven times, primarily in the summer. The impact of water releases by Thief Valley Dam could influence the presence of high pH downstream of Keating Valley given the issues with algal blooms and their impact on pH within the reservoir. Other than at Kirkway, pH was generally variable throughout the year for the sites on the Powder River. This resulted in no noticeable seasonal or yearly patterns in pH, although differences between sites were still the strongest overall factor.

Table 20a. Estimates of mean pH for the Powder River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Kirkway	7.77	8.10	8.44
Powder near Haines	7.99	8.38	8.78
Powder US Keating	8.42	8.80	9.18
Powder DS Keating	7.90	8.31	8.72
Powder near Richland	8.04	8.38	8.73

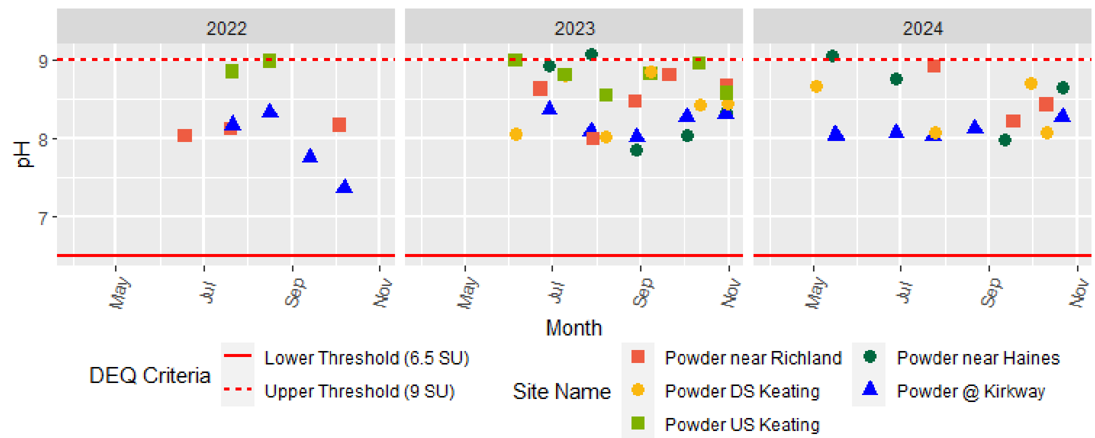
b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.27	0.07	0.42
July	-0.24	0.07	0.37
August	-0.45	-0.11	0.24
September	-0.32	0.03	0.38
October	-0.38	-0.07	0.25

c. Differences between mean yearly pH from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.49	-0.14	0.20
2023	-0.20	0.10	0.39
2024	-0.28	0.05	0.38

Chart 24. Observed pH measurements for the Powder River monitoring sites over the 2022-2024 period. Upper (dashed line) and lower (solid line) recommended pH standards highlighted.



Similar to the other watersheds, conductivity was heavily related to basin size. This increase in mean conductivity was particularly notable between the sites upstream and downstream of Keating Valley, where mean conductivity values increased by 280 µS/cm. While the basin size trends were noticeable among most sites, some outliers existed, with small decreases in conductivity (< 20 µS/cm) seen between the Powder River near Haines and upstream of Keating Valley, as well as between the sites downstream of Keating Valley and near Richland.

Chart 25. Observed conductivity measurements (in µS/cm) for the Powder River monitoring sites over the 2022-2024 period.

An increasing trend in conductivity was also noticeable from spring through fall. Sites with particularly large increases over the season were seen downstream of Keating Valley and near Richland, each increasing over 200% between May and October.

Table 21a. Estimates of mean conductivity (in µS/cm) for the Powder River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Kirkway	86.60	110.05	139.84
Powder near Haines	173.43	229.84	304.60
Powder US Keating	163.72	214.99	282.32
Powder DS Keating	370.94	496.20	663.75
Powder near Richland	375.51	480.02	613.61

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-182.80	-106.63	-9.25
July	-125.10	-45.37	53.53
August	-96.83	4.16	133.49
September	-84.26	20.77	155.45
October	45.89	170.36	325.67

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-87.35	-11.18	86.20
2023	-76.32	-9.71	72.49
2024	-58.18	21.83	122.99

Turbidity was highest on average near Haines, where mean turbidity values were highest and where turbidity exceeded 20 NTU's in the fall of 2023 and 2024. Turbidity was also above 10 NTU's in the spring near Richland, upstream of Keating Valley, and at Kirkway. Turbidity didn't show any strong site or seasonal patterns outside of high turbidity events in the spring and fall periods and low values seen near Richland, where mean turbidity was almost 4 NTU's lower than seen downstream of Keating Valley.

Chart 26. Observed turbidity measurements (in NTU's) for the Powder River monitoring sites over the 2022-2024 period. Log transformation used for turbidity measurements.

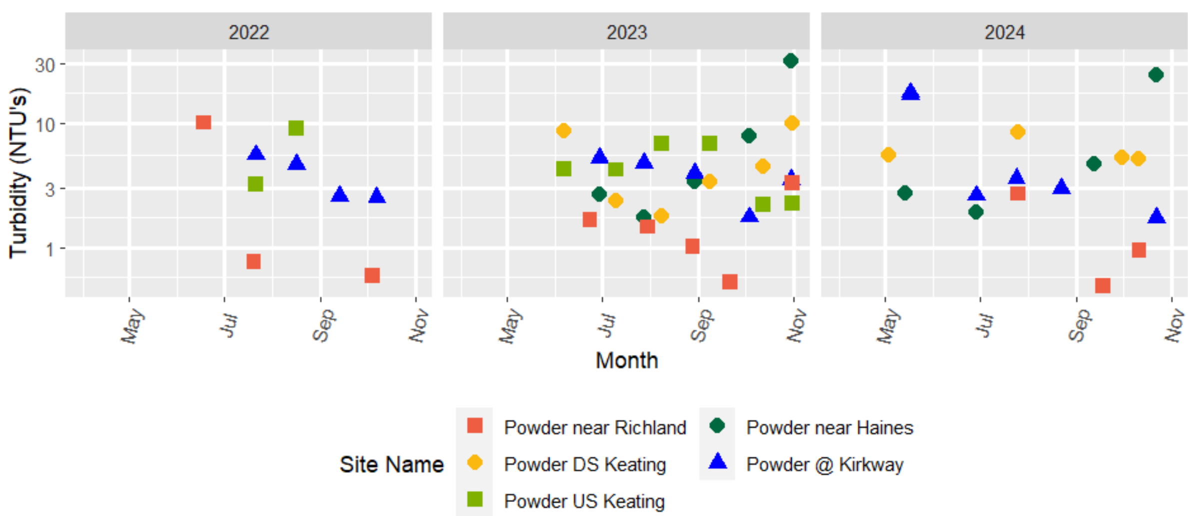


Table 22a. Estimates of mean turbidity (in NTU's) for the Powder River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Kirkway	1.47	3.22	7.05
Powder near Haines	2.29	5.76	14.49
Powder US Keating	1.85	4.51	11.02
Powder DS Keating	2.06	5.35	13.88
Powder near Richland	0.61	1.37	3.07

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.75	0.33	2.72
July	-0.85	-0.07	1.50
August	-0.90	-0.01	2.00
September	-1.09	-0.43	1.06
October	-0.74	0.18	2.09

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.80	0.27	2.67
2023	-0.92	-0.18	1.29
2024	-0.94	-0.09	1.72

Stream Temperature monitoring

In total, the four sites lower in the watershed all had mean estimates for Maximum temperatures above the 20 °C cool water standard. Only the site at Kirkway had temperatures consistently below the cool-water standard throughout most of the year, with 80% of days having temperatures below 20 °C. 100% of days in July and more than 90% of days in August were above 20 °C for sites near Haines, upstream of Keating Valley, downstream of Keating Valley, and near Richland had maximum temperatures above the cool-water standard.

Table 23a. Estimates of Mean and Maximum Daily temperatures (in °C) for the Powder River monitoring sites over the 2022-2024 period

Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Kirkway	13.26	14.66	16.06	14.98	16.61	18.23
Powder near Haines	17.19	18.82	20.45	19.49	21.37	23.26
Powder US Keating	16.85	18.29	19.74	18.95	20.63	22.30
Powder DS Keating	17.50	19.13	20.76	19.83	21.71	23.60
Powder near Richland	18.62	19.95	21.27	22.15	23.68	25.21

b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

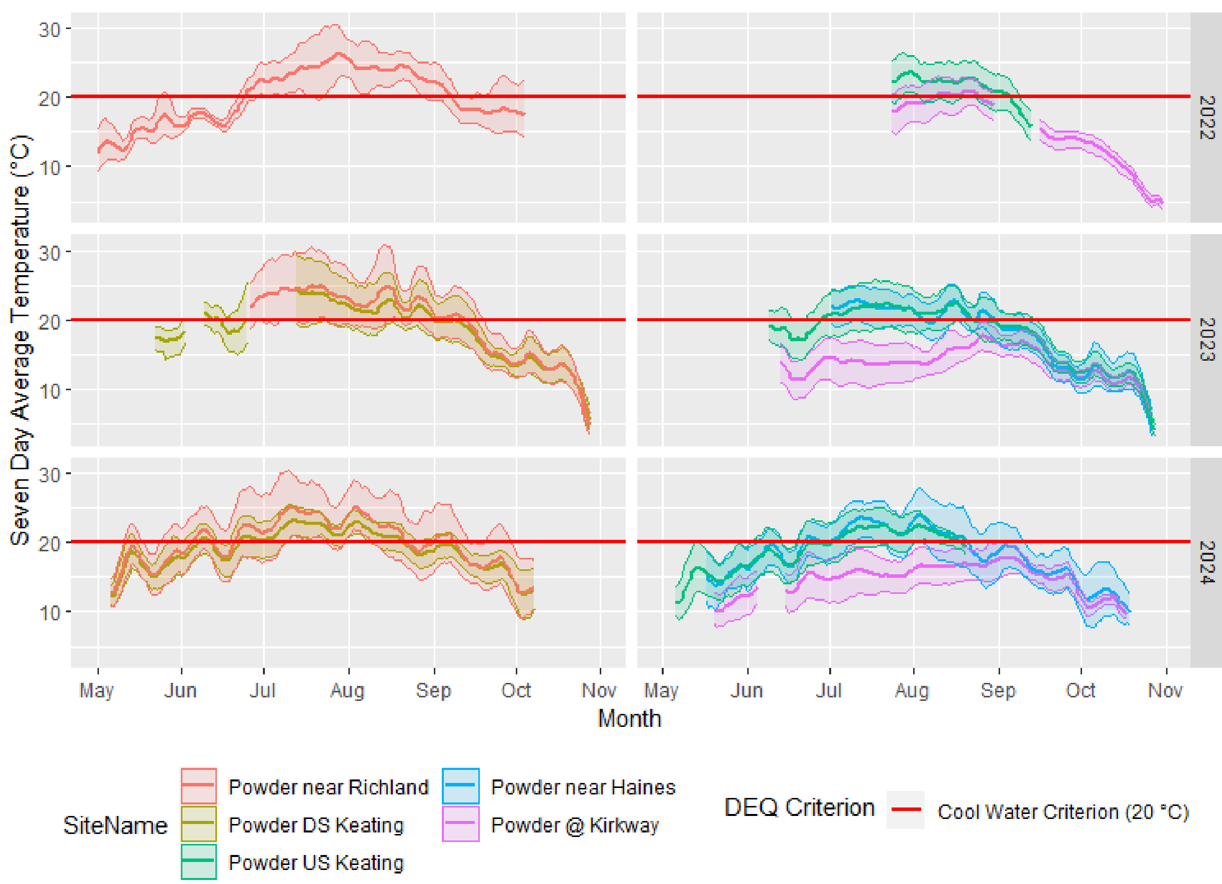
Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.52	0.88	2.28	-0.84	0.78	2.40
July	2.02	3.28	4.54	2.40	3.86	5.32
August	1.52	2.78	4.04	1.45	2.91	4.37
September	-2.20	-0.94	0.32	-2.75	-1.29	0.17
October	-7.30	-5.99	-4.68	-7.77	-6.26	-4.74

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.39	1.01	2.41	-0.85	0.77	2.39
2023	-1.76	-0.54	0.68	-1.88	-0.46	0.95
2024	-1.69	-0.47	0.74	-1.72	-0.31	1.10

Large increases in mean and maximum temperatures were seen on the Powder River between Kirkway and Haines, with 4.16 °C increase in mean temperatures and 4.77 °C increase in maximum temperatures between these sites. A small drop (~0.6 °C) in mean and maximum temperatures were found between the site near Haines and the site upstream of Keating Valley, possibly related to impacts of Thief Valley Reservoir. More modest increases (~1 °C) in mean temperatures were seen between the sites upstream of Keating and downstream of Keating, and from downstream Keating to the site near Richland. Another interesting pattern in temperature was an increase in maximum temperatures from downstream Keating to near Richland of 1.97 °C, indicating more variable daily temperatures here than the upstream sites.

Chart 27. Temperature profiles (in °C) for the Powder River monitoring sites over the 2022-2024 period, with Lower Powder sites on the left and Middle Powder sites on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cool water (solid line) standard in red.



Seasonal patterns in mean and maximum daily temperatures were noticeable for all sites, with July having the highest mean and maximum stream temperatures, while the coolest temperatures were seen in October. Most sites showed similar patterns in maximum and mean temperatures, throughout the year, The main exception was for the Middle Powder sites, where temperatures were warmer earlier in the year near Haines than at Kirkway, but were more similar to each other in the fall. Mean and maximum temperatures were warmer in 2022 than other years, with mean temperatures 1.01 °C and maximum temperatures 0.77 °C warmer than average, while temperatures in both 2023 and 2024 were cooler than average.

All of the Powder River sites showed an increase in 2022-2024 mean August temperatures when compared to the 1993-2011 estimates from the NorWeST model. Lower levels of warming were seen at Kirkway, with average increase of 0.59 °C, while the increases were more pronounced at the other four sites, ranging from 3.86 °C upstream of Keating to 5.20 °C near Richland. These four sites also had observed temperatures above expected temperatures by 2080, with the site near Richland 3.1 °C warmer than the expected 2080 temperatures. The impact of dams at Kirkway and upstream of Keating are likely a factor mitigating temperature increases, but are less impactful on other sites further downstream of these dams.

Table 24. Estimated and Observed mean August stream temperatures for Powder River monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

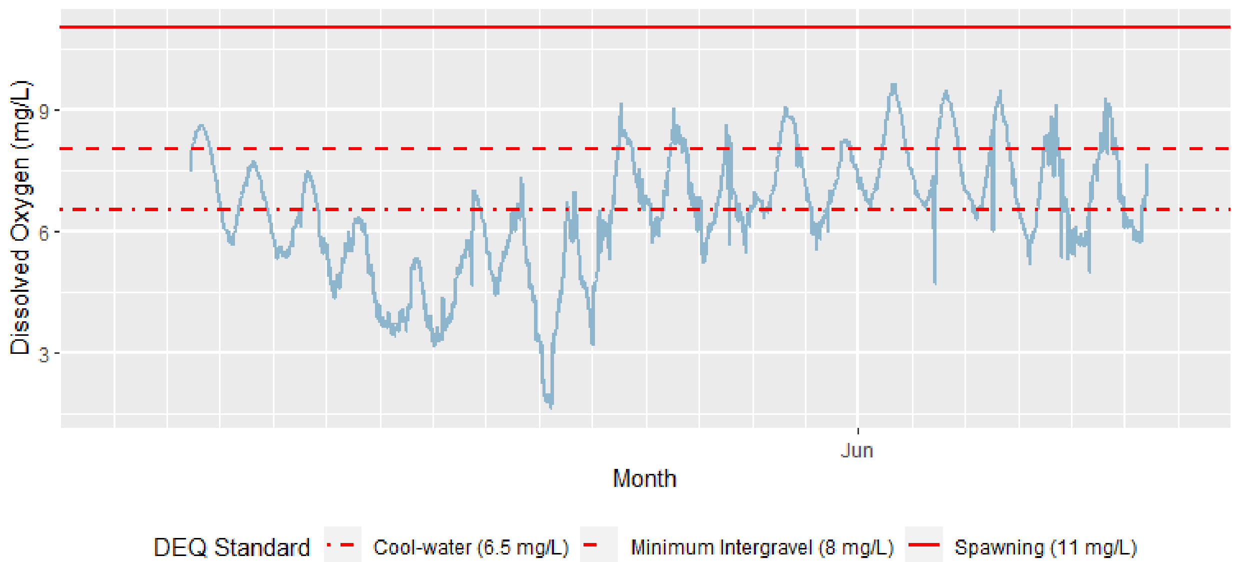
Site	Estimated 2022-2024 Mean August Temp (°C)	Observed 2022-2024 Mean August Temp (°C)	NorWeST 1993-2011 Temp (°C)	NorWeST 2040 Temp (°C)	NorWeST 2080 Temp (°C)
Powder @ Kirkway	17.43	17.44	16.85	18.11	18.91
Powder near Haines	21.60	20.76	16.31	17.56	18.35
Powder US Keating	21.07	21.32	17.46	18.74	19.55
Powder DS Keating	21.91	21.10	17.49	18.77	19.57
Powder near Richland	22.72	22.89	17.69	18.98	19.79

Dissolved Oxygen monitoring

One dissolved oxygen logger was installed at the site downstream of Keating in the spring of 2023 to look at daily oxygen profiles for sites lower in the watershed. While the logger was likely buried between May 22nd and May 27th, higher quality results indicated large swings in DO over the day, with changes of 3.37 mg/L each day. The data also indicated conditions generally unsuitable for fish use and spawning, with 24.5% of records below the 6.5 mg/L cool-water standard and 72.5% of records below 8 mg/L spawning standard for data after May 27th. These patterns were also observed for DO logger data installed at a site further downstream in spring of 2025, where 18.8% of records were below the cool-water standard and 42.8% of records were below the spawning standard.

The concentrations at the site downstream of Keating also showed significant differences in response to temperature than other oxygen profiles at sites higher in the watershed. Concentrations at this site were highest during the daytime, around 3:00 in the afternoon, while concentrations were lowest at 5:00 before dawn. This more closely follows patterns related to photosynthesis and biological oxygen demands, which as a whole suggests that there are issues with eutrophication related to nutrient pollution. Relationships with oxygen and temperature also differed from most other dissolved oxygen logger sites, with oxygen concentrations increasing by 0.345 mg/L per 1 °C increase in temperature, with the mean response for every other site being a 0.231 mg/L decrease per 1 °C increase in temperature (Appendix D). These temperature/oxygen correlations more closely fit the hypothesis that photosynthesis is the primary factor impacting oxygen concentration rather than temperature at this location.

Chart 28. Dissolved oxygen profile (in mg/L) for the Powder River downstream of Keating Valley oxygen logger data in 2023. Cool-water (dotted dashed line), Minimum intergravel (dashed line), and spawning (solid line) oxygen standards highlighted in red.



Water Quality Index

The Powder River at Kirkway was the only site with mean WQI estimates above the “Excellent” water quality threshold, while the other sites lower in the watershed had mean WQI estimates within the “Poor” water quality threshold. For these lower watershed sites, more the 50% of samples had WQI scores within “Poor” or “Very Poor” levels. Among index scores for these “Poor” and “Very Poor” samples, temperature was the primary factor lowering WQI scores, with an average sub-index score of 69.12. Dissolved oxygen and pH were also factors resulting in low WQI scores, although these scores were higher than for temperature, with sub-index scores of 80.15 and 79.64, respectively.

July had the lowest overall WQI scores, with values 6.75 lower than average. Temperature was the primary factor resulting in low WQI scores this month, with an average sub-index score of 46.38 this month. pH and DO had a smaller impact on low WQI score for this month, with average sub-index scores of 80.82 and 79.53, respectively. WQI was highest in October, with scores 6.37 higher than average. Sub-index scores for temperature were much higher this month, with an average sub-index score of 95.51. Overall, turbidity was not an important factor resulting in low WQI scores for sites or between months.

Chart 29. WQI scores at the Powder River monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WQI), and Poor (80-60 WQI) thresholds highlighted.

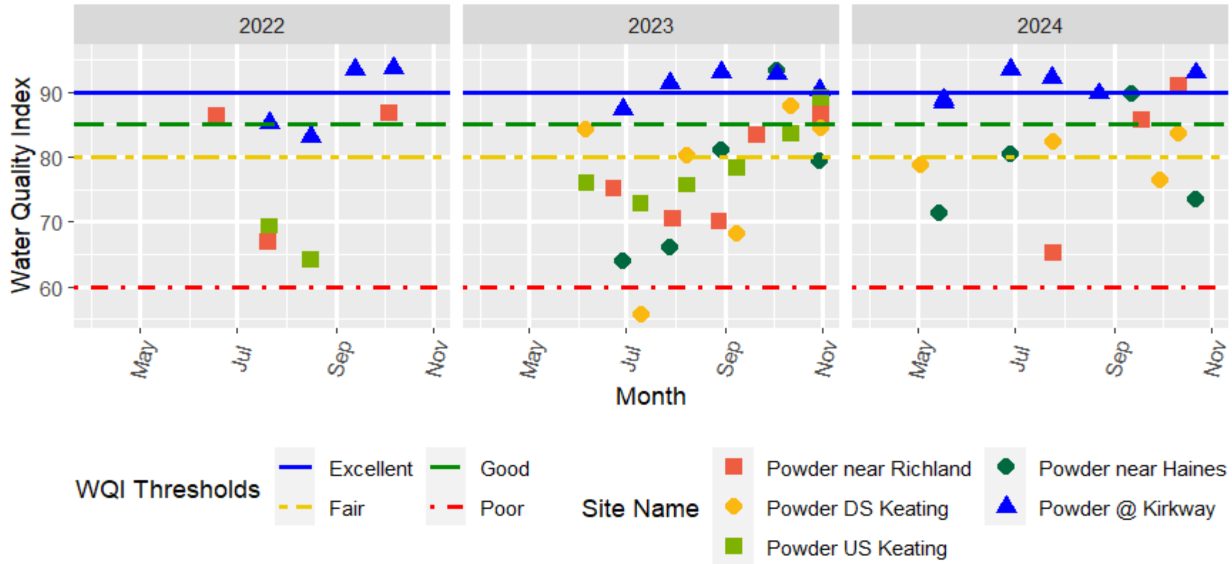


Table 25a. Estimates of mean WQI scores (from 10-100) for the Powder River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Powder @ Kirkway	82.58	90.71	99.63
Powder near Haines	67.97	75.90	84.75
Powder US Keating	69.23	77.03	85.70
Powder DS Keating	68.06	76.28	85.49
Powder near Richland	71.28	78.47	86.39

b. Mean monthly WQI differences from mean values over the 2022-2024 period

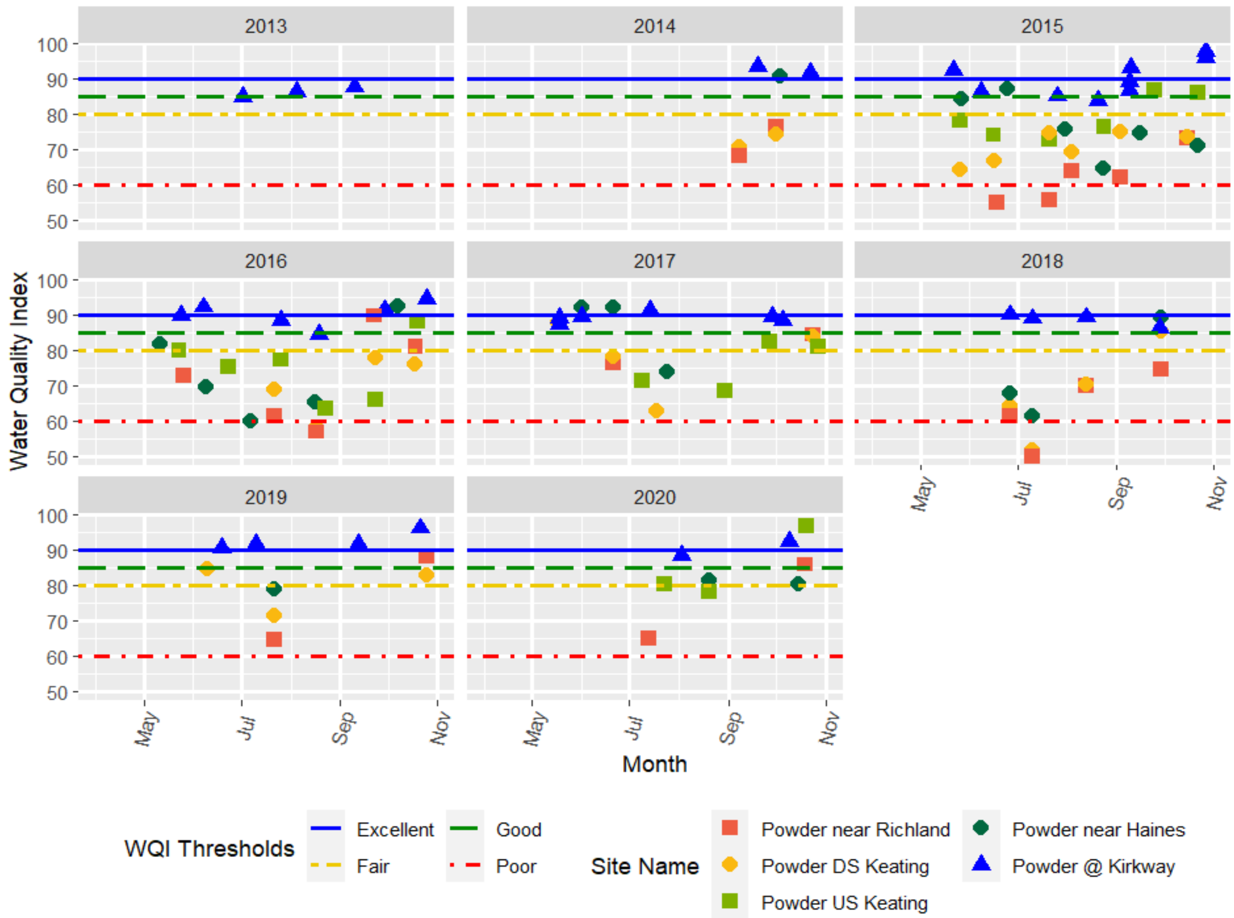
Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-6.90	0.15	7.92
July	-12.42	-6.75	-0.58
August	-8.98	-2.07	5.53
September	-5.03	2.30	10.39
October	-0.54	6.37	13.90

c. Mean yearly WQI differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-8.76	-1.71	6.06
2023	-6.73	-0.55	6.15
2024	-4.84	2.26	10.04

WQI scores were higher in 2024 than either 2022 or 2023, although these differences were not significant. Like scores for sites and months, low temperature sub-index scores were an important factor lowering overall WQI scores in 2022 and in 2023, with sub-index scores of 67.69 and 72.03, respectively, compared to a sub-index score of 81.33 for 2024. pH, dissolved oxygen, and turbidity sub-index scores were mostly similar between years.

Chart 30. WQI scores at the Powder River monitoring sites over the 2013-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WQI), and Poor (80-60 WQI) thresholds highlighted.



Trends in WQI scores since 2013 appeared to be heavily related to drought, with WQI lowest in 2014, 2015, 2016, and 2018 when drought conditions were more intense. These results could be biased by the lower number of sites monitored in 2013, 2014, 2019, and 2020, although trends within the sites seem to support the validity of these differences.

Table 26. Estimates for differences in mean WQI scores for the Powder River monitoring sites from mean values over the 2013-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2013	-9.02	-0.74	8.62
2014	-8.65	-2.85	3.48
2015	-6.82	-2.96	1.14
2016	-6.84	-2.71	1.67
2017	-2.96	1.58	6.42
2018	-8.51	-4.32	0.15
2019	-2.00	3.43	9.28
2020	-2.41	3.47	9.84
2022	-4.75	0.61	6.39
2023	-3.20	0.90	5.24
2024	-1.46	3.59	9.01

Discussion and future plans

Patterns of water quality in the Lower Powder indicate significant issues with high stream temperatures and less severe, but still persistent, issues with pH, dissolved oxygen, and turbidity. Issues with high stream temperatures are by far the biggest concern and are widespread throughout the reaches through Baker Valley, Thief Valley reservoir, and Keating Valley. While some processes, such as discharge from the reservoir and groundwater flow downstream of Keating Valley appear to buffer temperatures somewhat, consistent warming seems to occur throughout these reaches. Alongside these warm temperatures, impacts from eutrophication result in high pH and large fluctuations in oxygen concentrations at these sites, particularly for sites below the reservoir.

While the monitoring has uncovered patterns in water quality, important questions remain regarding other impairments, plausible sources, and potential solutions to water quality issues in the lower Powder River. The PBWC's monitoring from 2025-2027 will look to address some of these questions in identifying when and where bacteria and nutrient problems are present and in focusing restoration and improvement efforts to address them. eDNA monitoring might also be a useful tool in these efforts, both in identifying sources as well as relationships with concentrations over the season.

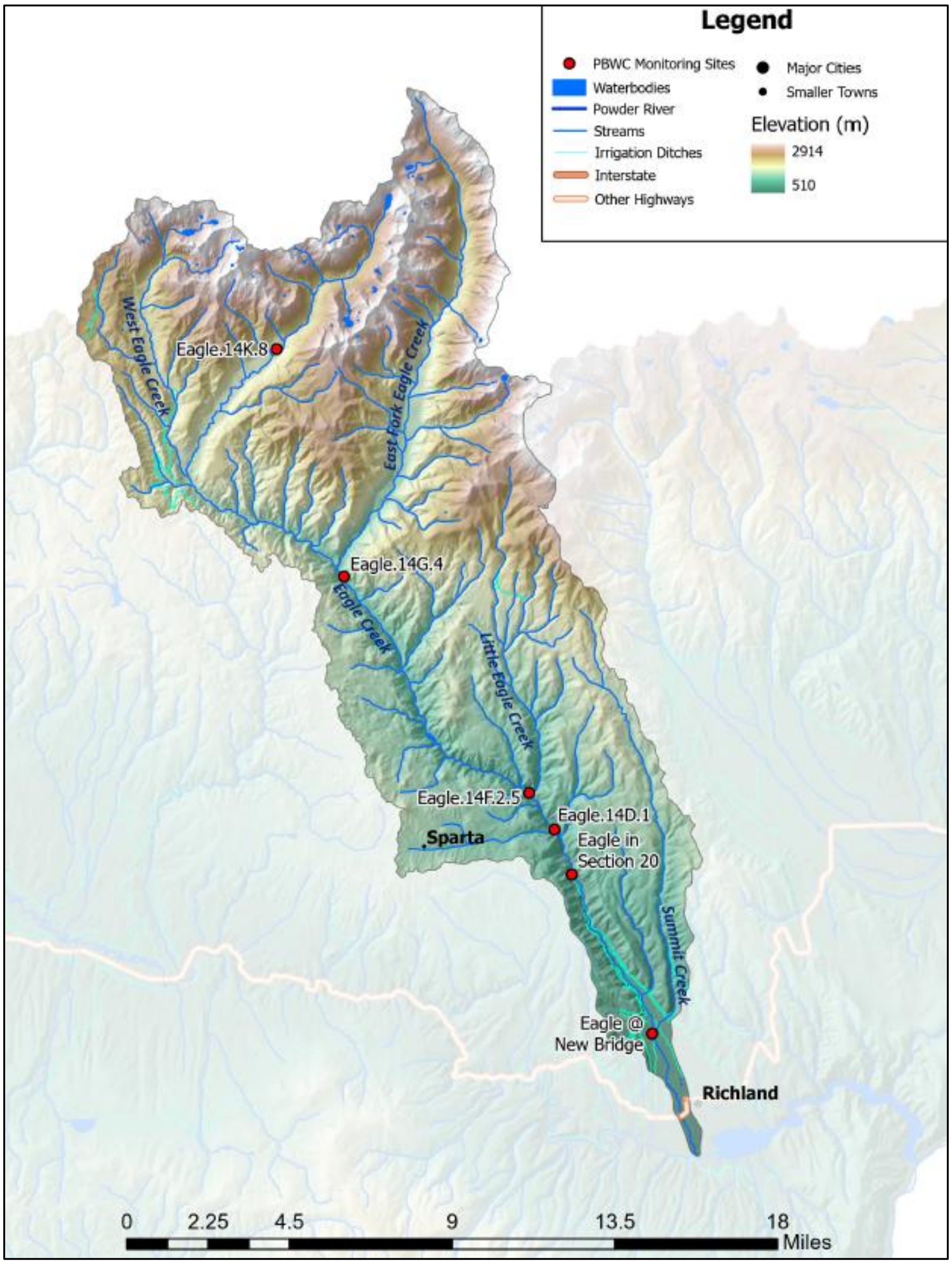
There is also a need to identify which strategies might be useful to improve temperatures and issues associated with eutrophication. The riparian planting efforts conducted by the Keating SWCD are expected to reduce mean and maximum temperatures and filter out bacteria and excess nutrients, but better data is needed on the landscape effect of these projects. Continued monitoring will be useful in tracking water quality and improve attribution to impacts from these projects. In the end, monitoring should be used to help the Council, landowners, communities, and interested partners in identifying sources and developing projects on the Powder to address these concerns in an effective manner.

Eagle Creek

Background

The Eagle Creek watershed drains 500 km^2 (193 mi^2) and covers a diverse array of landscapes, starting in the high country of the Eagle Cap Wilderness and carving deep canyons through the foothills of the Wallowa Mountains, before flowing in the broad alluvial Eagle Valley and joining the Powder River close to its confluence with the Snake River.

Map 8. The Eagle Creek watershed with 2022-2024 sample sites, major tributaries, and important features highlighted.



The headwaters for all three forks of Eagle Creek originate in the glacial valleys and cirques of the Eagle Cap Wilderness, which includes numerous alpine lakes, the largest being Eagle Lake, Looking Glass Lake, Echo Lake, and Bear Lake. After leaving the Wilderness, Eagle Creek flows through high gradient reaches in the basalt foothills of the Wallowa's, large portions of which contain good floodplain connectivity and high vegetation cover. In the Eagle Valley, diversion ditches divert flow to agricultural uses further into Eagle Valley and adjacent Powder River valley while altering downstream flow on Eagle Creek. The valley section is well shaded with cottonwood and other larger riparian vegetation, but appears to be more incised than historical or upstream conditions in these reaches.

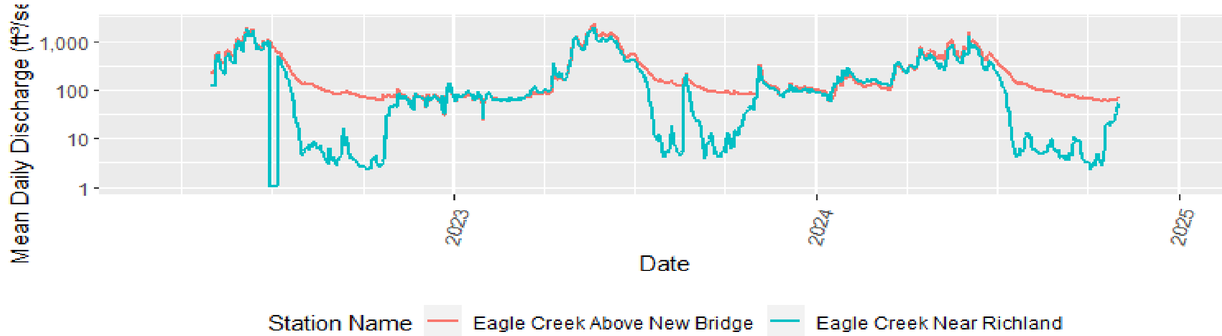
Table 27. Site characteristics for monitoring locations on Eagle Creek including ODFW Fish Habitat Type, Elevation (m) upstream drainage area (km^2), Modeled 1993-2011 NorWeST mean August stream temperature (°C), and established date.

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
Eagle @ New Bridge	37336	Cool	704	466.1	15.81	6/12/2013
Eagle in Section 20	37787	Cool	816	444.4	14.56	6/13/2014
Eagle.14D.1	37737	Cool	855	404.3	14.09	6/13/2014
Eagle.14F.2.5	41789	Cool	888	350.5	14.19	6/23/2023
Eagle.14G.4	37738	Cool	1122	270.2	11.79	6/13/2014
Eagle.14K.8	37739	Cold	1508	34.6	12.02	6/13/2014

The hydrology of Eagle Creek is heavily impacted by the large amount snowmelt during the spring runoff period from March through June. During the winter and early spring, the Wallowa Mountains and its foothill regions accumulate large amounts of snow in the winter, up to 80" on average at the West Eagle Meadows. Large volumes of cold water originating as snow in these upland areas begin in March and often last well into July, buffering stream temperatures during the early summer. Stream flows drop precipitously between June and September, as upland snowmelt declines with warmer air temperatures. These changes are particularly notable at the upper watershed sites where surface sources make up a larger contribution of stream flow (Figure 17). Low flows are more prevalent at the lower watershed sites, particularly at Eagle @ New Bridge, due to irrigation withdrawals for agricultural uses (Chart 31).

Figure 17. Changes in flow at Eagle.14G.4 between June and October 2022. Flow regimes at this site and others within the foothills region are heavily influenced by patterns in upland snowmelt

Chart 31. Measured discharge (in ft^3/sec) at gaging stations near Eagle.14D.1 and Eagle @ New Bridge sites. Note the large decrease in flows from irrigation withdrawals at New Bridge in comparison to the more consistent flows seen upstream.



Bull Trout used to be present throughout much of the Upper Eagle Creek watershed, with the lower reaches primarily used as migratory habitat. Stocking of Brook Trout among lakes in the Eagle Cap Wilderness resulted in the competitive exclusion of the local Bull Trout within the watershed, with surveys conducted in 1991 and 1994 failing to locate any Bull Trout within the Creek (**ODFW 2005**). Redband trout are still present throughout the Eagle Creek watershed, with populations found in East Fork, West Fork, Little Eagle, and Summit Creek, as well as locations on the Main Fork near the confluence with Skull Creek. Uses in these reaches are primarily used for spawning and summer rearing, with some migratory corridors, although historic populations had far more winter and migratory habitat within the watershed (**NPCC 2004b**).

During the week of August 20th, 2023, an extreme precipitation event totaling 1.7 inches of rain fell in the Powder Basin watershed, including sections of Eagle Creek. This large amount of rain (more than 34 times the normal expected daily precipitation) resulted in both unseasonably high stream flows and super saturated soils in upland areas. The debris flows and erosion from the event dislodged enough sediment and large wood to create a full landslide in one section of Hudson Creek, a tributary of East Eagle Creek. This landslide also resulted in a large pulse of sediment from Hudson Creek into East Eagle Creek and Eagle Creek downstream of the slide, which was noticeable for several months afterwards, including the spring runoff period in 2024 (Figure 18).

Figure 18. The debris flow associated with the Hudson Creek landslide (left) moved large of large amounts of rock and sediment into East Fork Eagle Creek. Fine sediment from this event was seen even further downstream, like this site on Eagle Creek downstream of the East Fork confluence (right).

Another major event impacting water quality was the Town Gulch fire, which started on August 5th, 2024, and eventually impacted ~18,000 acres in the middle part of the Eagle Valley. In general, fire severity was low, mainly impacting lowland vegetation while having a limited impact on forest cover, but some areas were more heavily impacted, particularly those downstream of the private/forest boundary (Figure 19). Two monitoring sites, Eagle in Section 20 and Eagle.14D.1, were located within the fire zone, and were heavily impacted by reduced canopy cover and sedimentation, with Section 20 the more heavily impacted location.

Figure 19. Aftermath of the Town Gulch Fire on August 13th, 2024, as seen from the Sparta Cutoff Road

The PBWC has been invested in monitoring throughout this watershed, with five long-term monitoring sites and one dissolved oxygen monitoring site in reaches throughout Eagle Creek. Eagle.14K.8, the highest site in the watershed, is located on the main fork of Eagle Creek where it leaves the alpine lakes and glaciated valleys at Boulder Park. Further downstream, the site at Eagle.14G.4 monitors the creek downstream of where West Fork and East Fork of Eagle Creeks enter. Eagle.14D.1 is on the creek just upstream of the WWNF/Private land boundary, 1.3 miles downstream of a dissolved oxygen monitoring site above the confluence with Little Eagle Creek. The PBWC also collects water quality data at two sites on private land in the gorge below the WWNF boundary and in New Bridge as Eagle Creek enters the broad Eagle Valley near Richland. Sampling at these sites is heavily influenced by snowmelt and access, with no samples in May due to high water levels and difficult access to upper elevation sites.

Grab Sample Monitoring

Oxygen concentrations were consistently above the 8 mg/L cold-water standard throughout the entire year at all sites in the Eagle Basin. Oxygen concentrations were highest at sites near the Forest boundary and were lowest at the highest site, with mean oxygen concentrations over 0.8 mg/L lower than seen in Section 20. The saturation levels at these sites were generally consistent with one another, indicating that the lower oxygen concentrations were likely related to the higher elevation lowering theoretical concentrations at the upper watershed sites.

Chart 32. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the Eagle Creek watershed monitoring sites over the 2022-2024 period. Cool-water standard (red line) and cold-water standard (solid line) highlighted.

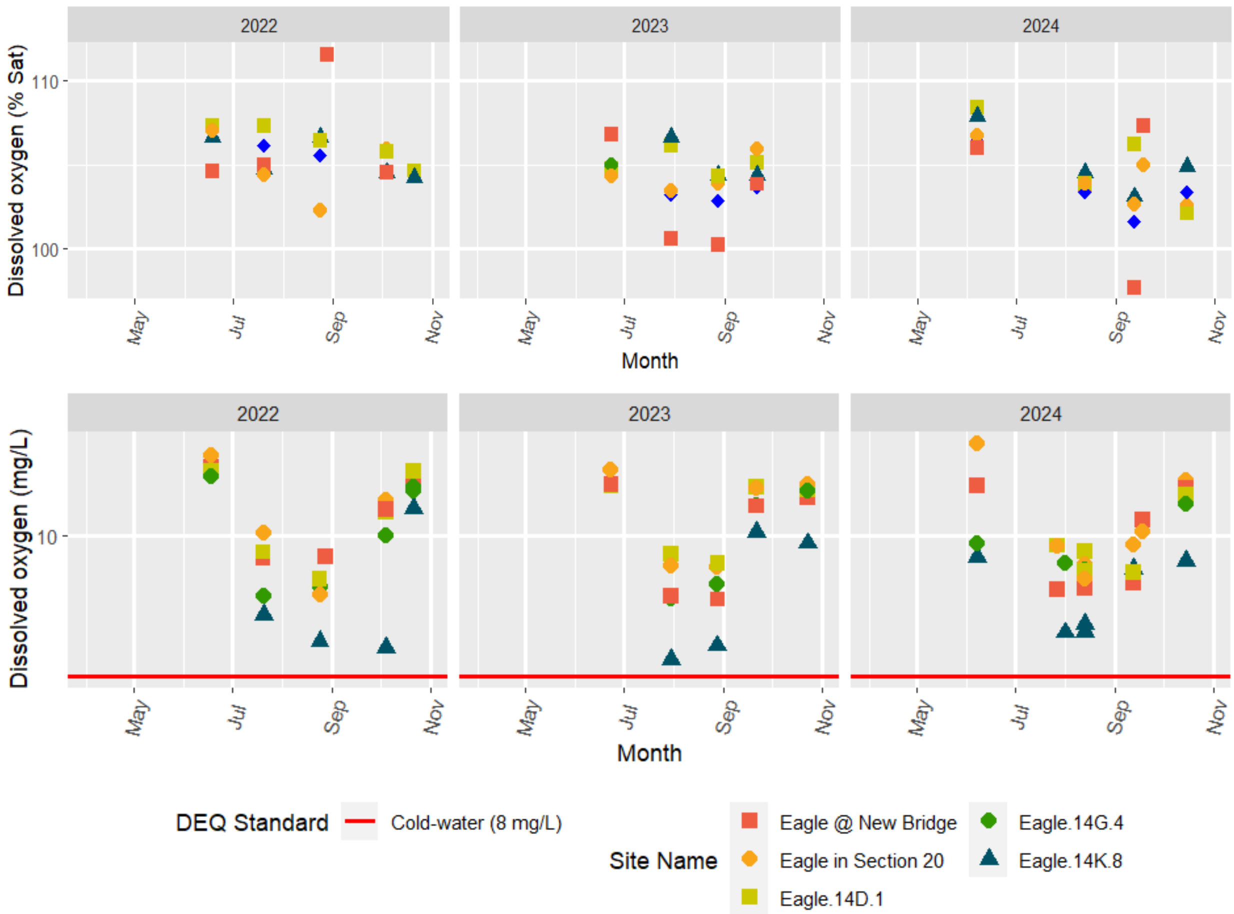


Table 28a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the Eagle Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Eagle @ New Bridge	9.73	9.99	10.25
Eagle in Section 20	9.96	10.22	10.48
Eagle.14D.1	9.87	10.13	10.39
Eagle.14G.4	9.61	9.88	10.15
Eagle.14K.8	8.99	9.29	9.60

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	0.46	0.72	0.98
July	-0.92	-0.65	-0.38
August	-0.94	-0.69	-0.43
September	-0.12	0.17	0.46
October	0.17	0.45	0.73

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.18	0.08	0.34
2023	-0.20	0.07	0.33
2024	-0.40	-0.15	0.11

Oxygen concentrations were highest in the spring and fall and lowest in summer, with temperature being a major factor for the lower oxygen concentrations as indicated by relatively consistent saturation levels throughout the year. Differences between years were small, with higher concentrations seen in 2022 and 2023 and lower concentrations seen in 2024.

pH was within recommended standards at most sites but was below the 6.5 pH lower standard in 2022 at Eagle.14K.8. pH variability was also higher in 2022. Both of these factors are likely related to older pH probes during this year biasing reading somewhat, especially in the colder and less mineral-rich waters higher in the Eagle Creek watershed. Outside of 2022, pH was highest at the sites between the WWNF boundary and the confluence with East Eagle Creek and lowest at Eagle.14K.8. Seasonal patterns in pH were small and were most noticeable in 2022, again likely related to pH probe issues. Overall, pH was lowest in the spring and higher in the fall, with an increasing trend seen through the season. pH probe issues were most noticeable in yearly differences in pH, with lower values recorded in 2022 than 2023 and 2024, but differences were generally small between 2023 and 2024.

Chart 33. Observed pH measurements for the Eagle Creek monitoring sites over the 2022-2024 period. Upper (dashed line) and lower (solid line) recommended pH standards highlighted.

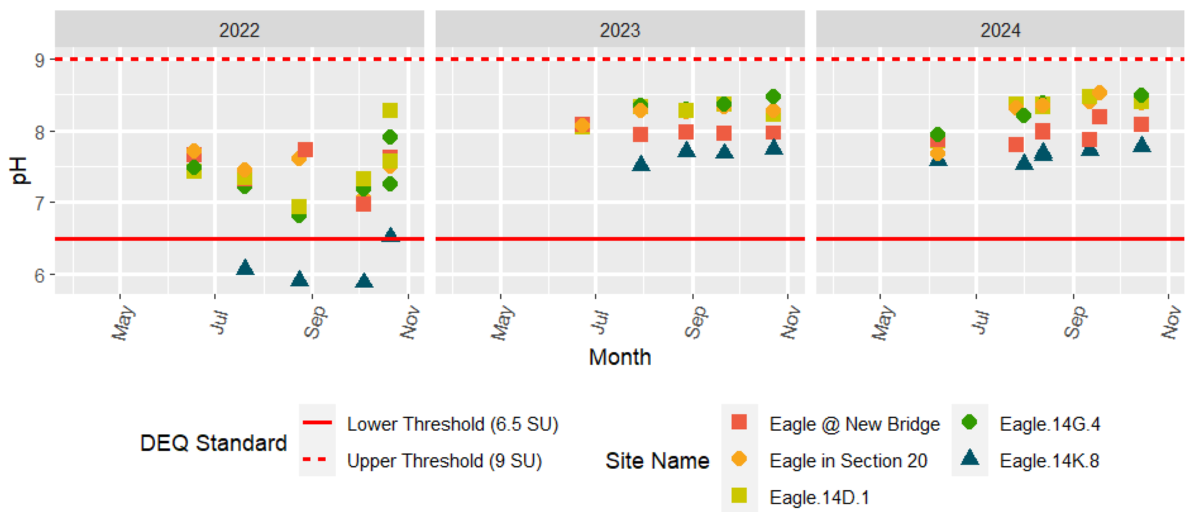


Table 29a. Estimates of mean pH for the Eagle Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Eagle @ New Bridge	7.57	7.79	8.00
Eagle in Section 20	7.78	7.99	8.21
Eagle.14D.1	7.79	8.01	8.22
Eagle.14G.4	7.75	7.98	8.20
Eagle.14K.8	6.92	7.16	7.39

b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.30	-0.09	0.13
July	-0.26	-0.05	0.16
August	-0.21	-0.01	0.19
September	-0.15	0.09	0.33
October	-0.11	0.07	0.25

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.82	-0.61	-0.39
2023	0.08	0.30	0.52
2024	0.10	0.31	0.52

Conductivity differences between sites were quite noticeable for the Eagle Creek watershed, particularly for the higher watershed sites. Overall, conductivity showed strong spatial patterns related to basin size, with the highest overall conductivity measurements seen at Eagle @ New Bridge and the lowest measurements seen at Eagle.14K.8. The low values at Eagle.14K.8 were unique in that they differ both substantially from other sites in the Eagle Creek watershed as well as the rest of the basin. The average conductivity measurements at this site of 21.89 µS/cm were the lowest recorded of any site in the Powder Basin. Although the reasons for these low measurements are still to be determined, they could be related to the granitic bedrock and the resulting low concentrations of non-silicate minerals in these rocks.

Chart 34. Observed conductivity measurements (in µS/cm) for the Eagle Creek monitoring sites over the 2022-2024 period.

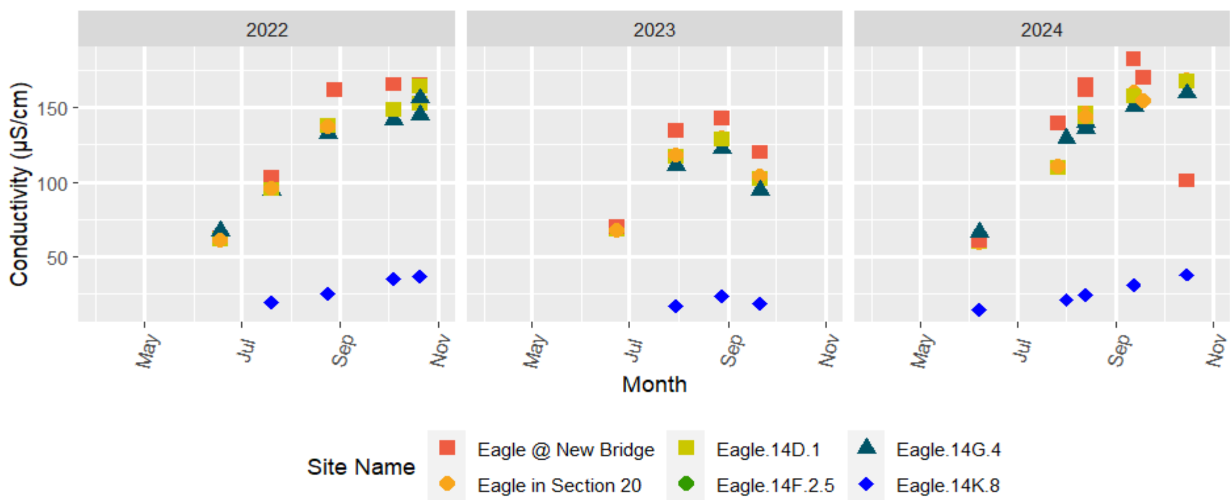


Table 30a. Estimates of mean conductivity (in µS/cm) for the Eagle Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Eagle @ New Bridge	115.06	125.89	137.73
Eagle in Section 20	103.77	113.62	124.41
Eagle.14D.1	103.41	113.17	123.86
Eagle.14G.4	99.28	108.99	119.66
Eagle.14K.8	19.79	21.89	24.22

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-44.87	-40.94	-36.64
July	-14.49	-7.35	0.50
August	2.84	10.77	19.41
September	-2.37	6.57	16.47
October	21.81	30.94	40.84

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-9.51	-2.61	4.94
2023	-12.12	-5.14	2.52
2024	0.11	7.75	16.11

Seasonal changes in conductivity were strongly associated with flow, increasing throughout the year from June through September. Seasonal changes were more noticeable at the middle and lower watersheds, and were generally similar between years. Overall conductivity differences between years were smaller than monthly differences, and were higher in 2024 and lower in 2023, likely related to differences in flow between these years.

Landslide activity related to heavy precipitation in late August 2023 had a significant impact on turbidity patterns between sites, months, and years for the Eagle Creek sites. Overall, turbidity differences between sites were heavily related to basin size, with higher values associated with larger upstream basin sizes. Lower turbidity measurements were seen at Eagle.14K.8, particularly after the landslide, which only affected locations downstream of the East Eagle Creek confluence. Turbidity during this month was 13.92 NTU's higher at the sites downstream of East Eagle when compared to Eagle.14K.8, compared to an average of 1.71 NTU's higher for the rest of the monitoring period. Turbidity during this month and the following June were also the only two periods where turbidity exceeded 10 NTU's.

Chart 35. Observed turbidity measurements (in NTU's) for the Eagle Creek monitoring sites over the 2022-2024 period. Log transformation used for turbidity measurements.

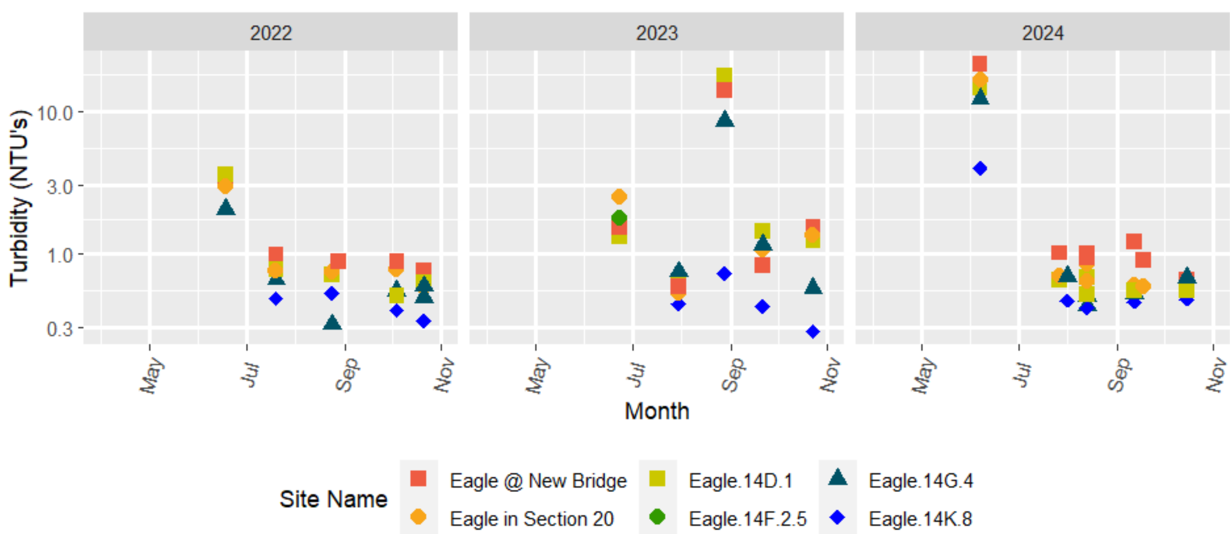


Table 31a. Estimates of mean turbidity (in NTU's) for the Eagle Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Eagle @ New Bridge	0.82	1.47	2.62
Eagle in Section 20	0.71	1.26	2.24
Eagle.14D.1	0.67	1.19	2.11
Eagle.14G.4	0.55	1.00	1.81
Eagle.14K.8	0.33	0.62	1.17

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	0.94	2.85	6.24
July	-1.13	-0.86	-0.39
August	-0.82	-0.35	0.48
September	-1.15	-0.84	-0.26
October	-1.07	-0.80	-0.38

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.65	-0.29	0.35
2023	-0.25	0.42	1.63
2024	-0.54	-0.13	0.59

The landslide also had significant impacts on seasonal turbidity patterns in the Eagle Creek watershed, although to a smaller degree than those seen between sites. Turbidity was highest in the early summer during snowmelt runoff, with a notable increase seen in June 2024 compared to June 2023 and 2022, likely related to lingering impacts from the landslide. Landslide impacts were also notable between months, with average August turbidity 0.48 NTU's higher than July, September, and October averages. Finally, the landslide impacts were notable enough to result in higher turbidity for all of 2023, with mean turbidity 0.63 NTU's higher than either 2022 or 2024.

Stream Temperature monitoring

Most sites and months in the Eagle Creek basin had maximum temperatures below the 20 °C cool-water standard for redband trout. Temperature exceedance above 20 °C was most common at Eagle @ New Bridge, but was also present at Eagle in Section 20 and Eagle.14D.1. Temperatures above 20 °C at Eagle @ New Bridge most commonly occurred in August, where 78.5% of days were above the cool water standard. July and September also had more than 49.5% and 25.5% days above 20 °C as well at Eagle @ New Bridge. Exceedance was far less common at the sites near the WWNF boundary, where less than 13% of days at Eagle in Section 20 and 5% of days at Eagle.14D.1 were above 20 °C in July and August. Maximum daily temperatures were consistently below the cool water standard at the upper watershed sites. For the cold-water standard, maximum temperatures were consistently above 12 °C at Eagle.14K.8, where maximum temperatures were above 12 °C for more than 50% of days July through September. Only October had temperatures below the cool-water standard every day of the month.

Table 32a. Estimates of Mean and Maximum Daily temperatures (in °C) for the Eagle Creek monitoring sites over the 2022-2024 period

Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Eagle @ New Bridge	14.35	14.63	14.91	17.48	17.83	18.17
Eagle in Section 20	12.62	12.90	13.18	14.80	15.15	15.50
Eagle.14D.1	12.13	12.41	12.69	13.97	14.32	14.67
Eagle.14G.4	9.94	10.22	10.51	12.63	12.98	13.34
Eagle.14K.8	9.58	9.89	10.19	11.84	12.22	12.60

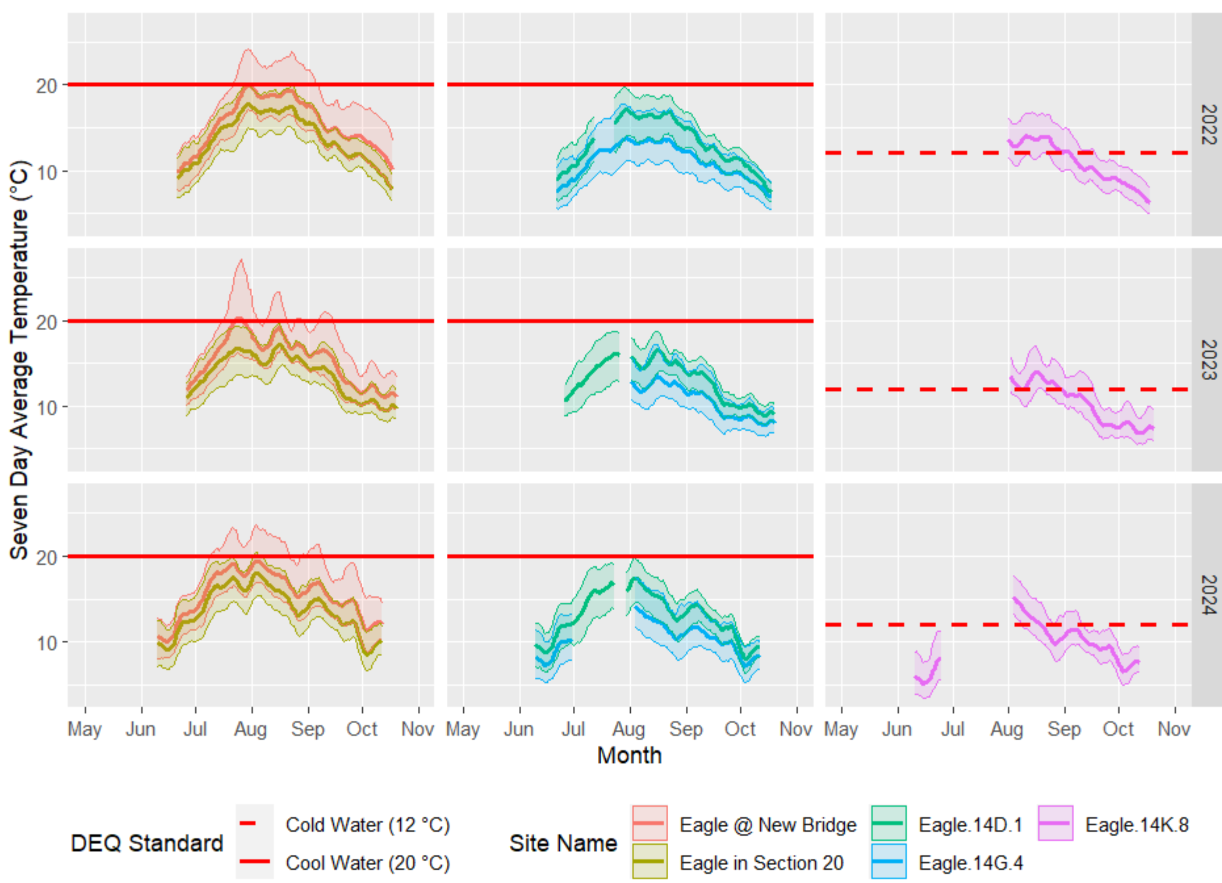
b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Month	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
June	-2.84	-2.56	-2.28	-2.97	-2.62	-2.28
July	1.97	2.20	2.43	2.32	2.61	2.90
August	2.77	3.00	3.22	3.03	3.30	3.58
September	-0.05	0.18	0.40	-0.26	0.02	0.30
October	-3.05	-2.81	-2.57	-3.61	-3.31	-3.00

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Year	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
2022	-0.23	0.05	0.33	-0.29	0.06	0.41
2023	-0.36	-0.07	0.22	-0.51	-0.15	0.21
2024	-0.25	0.01	0.27	-0.24	0.09	0.41

Chart 36. Temperature profiles (in °C) for monitoring sites on Eagle Creek, with lower sites on the left, upper cool-water sites in the middle, and upper sites cold-water sites on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cold water (dashed line) and cool water (solid line) standards highlighted in red based off relevant fish use at the site.



Seasonal patterns in maximum and mean daily temperatures were similar between sites, with the highest temperatures occurring in August during low flow conditions and the lowest temperatures occurring in June during snowmelt runoff and in the fall related to cooler air temperatures. Maximum temperatures increased more from June through August and decreased more in the fall than mean temperatures. Yearly differences were small, with 2023 having lower mean and maximum temperatures than either 2022 or 2024.

Table 33. Estimated and Observed mean August stream temperatures for Eagle Creek monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

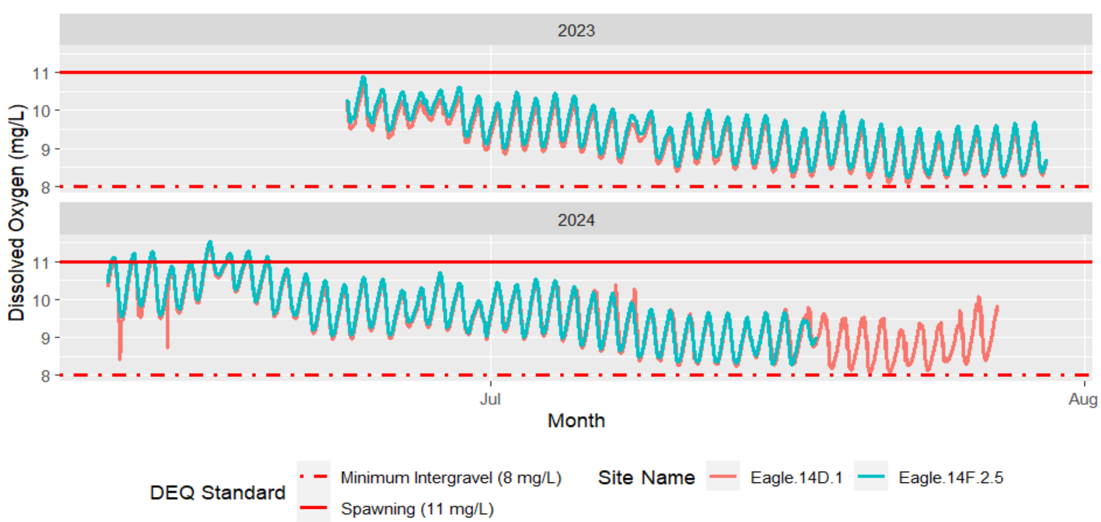
Site	Estimated 2022-2024 Mean August Temp (°C)	Observed 2022-2024 Mean August Temp (°C)	NorWeST 1993-2011 Temp (°C)	NorWeST 2040 Temp (°C)	NorWeST 2080 Temp (°C)
Eagle @ New Bridge	17.89	17.50	15.81	17.04	17.82
Eagle in Section 20	16.08	15.82	14.56	15.76	16.52
Eagle.14D.1	15.52	15.35	14.09	15.28	16.04
Eagle.14G.4	12.67	13.19	11.79	12.92	13.64
Eagle.14K.8	12.88	13.16	12.02	13.16	13.88

Observed mean August Stream temperatures for the Eagle Creek monitoring sites were on average 1.35 °C higher when compared to the 1993-2011 modelled estimates from the NorWeST stream temperature model. Increases were closely related to basin size, with the largest increase seen at Eagle @ New Bridge with at 2.08 °C and lowest at Eagle.14K.8 at 0.86 °C.

Dissolved Oxygen monitoring

Two dissolved oxygen loggers were installed near the Eagle Forks reach upstream and downstream of the Little Eagle Creek confluence to assess oxygen concentrations during the spring redband trout spawning season. Due to high stream flows early in the season, the loggers were installed and deployed later into the year than other dissolved oxygen logger sites, with installation dates ranging from 6/11 to 6/23 and removal dates of 7/27 to 7/29. Over the course of the deployment, flow conditions often changed dramatically, leaving some loggers buried or exposed to the surface and resulting in lower data quality or dry measurements near the end of the data series.

Chart 37. Dissolved oxygen profiles (in mg/L) for the Spring Eagle Creek oxygen logger monitoring sites in 2023 and 2024. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



Overall dissolved oxygen concentrations at both sites were consistently above the 8 mg/L minimum intergravel standard but were mostly below the 11 mg/L spawning standard. Concentrations were more likely to be over 11 mg/L in June, with more than 4.7% of records at Eagle.14D.1 and 8.1% of records for Eagle.14F.2.5 above the spawning standard during this month. Mean concentrations were also higher in June than July at both sites, decreasing by 0.94 mg/L at Eagle.14D.1 and by 0.93 mg/L at Eagle14.F.2.5 between June and July. Dissolved oxygen concentrations were 0.101 mg/L higher in 2024 than 2023, with a larger difference seen for Eagle.14D.1. Some of these differences between years can be attributed to lower logger accuracy in 2023 as well as differences in timing of deployment, but these differences were generally small and within the level of precision for the loggers.

The timing of oxygen throughout the day was similar between sites and between years, with the highest oxygen concentrations observed at 9:00 in the morning and the lowest concentrations observed around 5:00 in the afternoon. This pattern was closely related to temperature, with daily minimum and maximum temperatures occurring during the same time periods. Overall, temperature was a strong predictor of oxygen concentrations, with oxygen concentrations decreasing by 0.218 mg/L per 1 °C increase in temperature at both sites. Patterns in oxygen concentrations and temperatures were generally similar between sites, differing by only 0.015 mg/L per 1 °C increase. Baseline oxygen concentrations at the Eagle Creek sites were also the highest in the Powder Basin for sites monitored with dissolved oxygen loggers, with estimated oxygen concentrations of 12.26 mg/L at 0 °C, 1.35 mg/L higher than average for all sites monitored with oxygen loggers during both spring and fall seasons (Appendix D).

Water Quality Index

Mean estimates for WQI scores in at the Eagle Creek sites were indicative of Excellent water quality but contained a large degree of variability over the course of the 2022-2024 monitoring period. Overall water quality was closely related to basin size, with the highest water quality at Eagle.14K.8 and the lowest water quality at Eagle @ New Bridge. Water quality with “Good” or “Fair” thresholds was common at Eagle.14K.8 in 2022, and at the lower and middle elevation sites in August 2023 and June 2024.

Turbidity was a major factor resulting in lower WQI scores at these sites during the landslide and its after-effects, with mean turbidity subindex scores of 69.90 for August 2023 and 66.65 for June 2024 compared to average subindex scores of 91.02 and 96.84 for the other parameters, respectively. Low pH was the most important factor resulting in low WQI values at Eagle.14K.8 in 2022, with an average subindex score of 60.35 for this year compared to an average of 95.01 for the other parameters this year. In contrast, warm stream temperatures were also the primary reason for low WQI scores in August 2022 at Eagle @ New Bridge, with a subindex score of 63.46 compared to an average of 94.86 for the other parameters this month and year.

Chart 38. WQI scores at the blank watershed monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

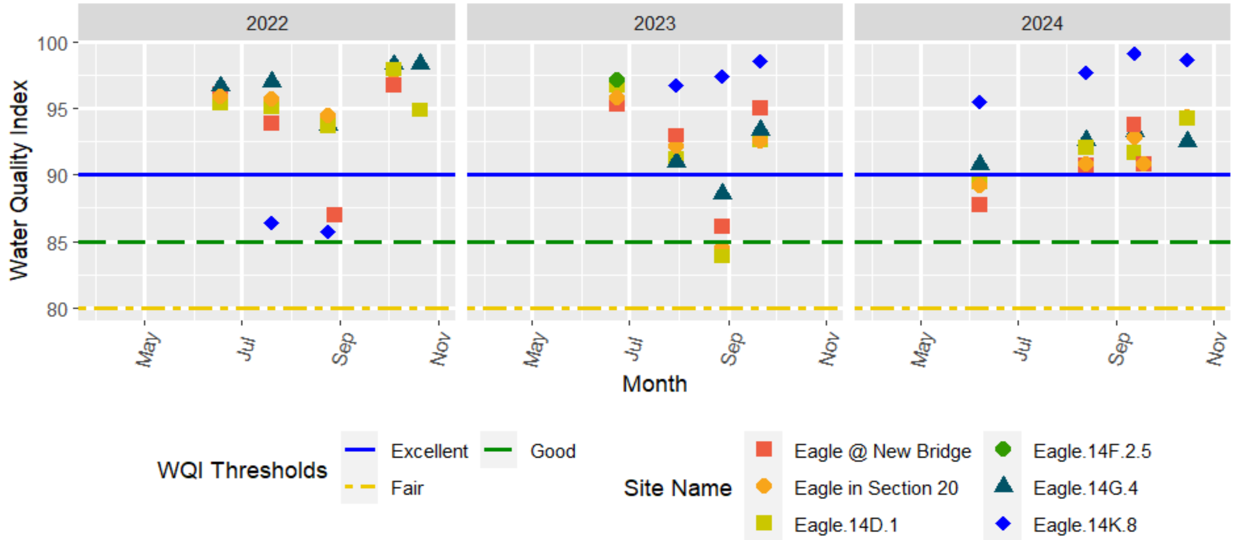


Table 34. Estimates of mean WQI scores (from 10-100) for the Eagle Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Eagle @ New Bridge	89.74	92.47	95.28
Eagle in Section 20	90.20	92.96	95.80
Eagle.14D.1	90.11	92.84	95.64
Eagle.14G.4	90.85	93.71	96.66
Eagle.14K.8	92.59	95.92	99.38

b. Mean monthly WQI differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-2.24	0.53	3.39
July	-3.76	-0.92	2.01
August	-5.70	-3.11	-0.44
September	-2.20	0.93	4.16
October	-0.44	2.56	5.66

b. Mean yearly WQI differences from mean values over the 2022-2024 period

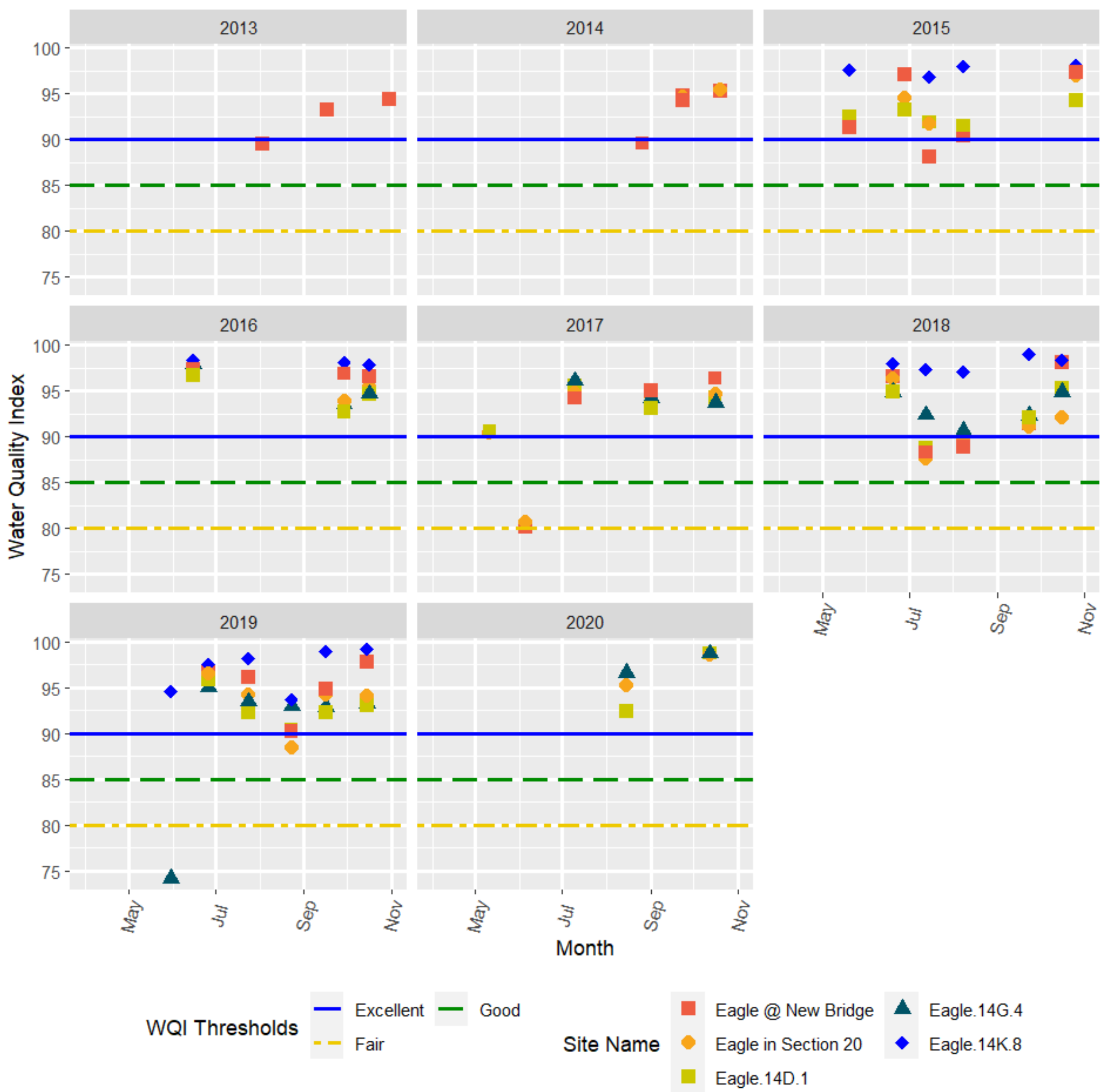
Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-1.51	1.29	4.17
2023	-3.15	-0.30	2.64
2024	-3.67	-0.99	1.78

Seasonal changes in WQI showed higher estimates in June and the fall and the lowest estimates in July and August, with WQI varying by as much as 5.67 between August and October. Temperature was a significant contributor to lower WQI values during August, with an average subindex value of 86.33 compared to an average of 92.01 for the other parameters. This discrepancy was even larger for 2022 and 2024 when turbidity impacts from the landslide were not present, with an average subindex value of 85.23 compared to an average of 95.08 for the other parameters. Lower turbidity and cooler stream temperatures were important contributors for higher WQI values for October, with average subindex values of 98.20 for turbidity and 99.42 for temperature compared to averages of 92.34 and 92.01 for the other months, respectively. Yearly differences in WQI values were small, with higher WQI in 2022, compared to 2023 and 2024, likely related to lingering effects of the landslide on turbidity.

Table 35. Estimates for differences in mean WQI scores for the Eagle Creek monitoring sites from mean values over the 2013-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2013	-4.35	-0.93	2.62
2014	-2.31	0.21	2.81
2015	-1.62	0.30	2.26
2016	-1.16	0.72	2.65
2017	-3.58	-1.64	0.34
2018	-2.54	-0.88	0.81
2019	-1.42	0.27	1.98
2020	0.76	3.70	6.73
2022	-1.05	0.76	2.60
2023	-2.85	-1.01	0.85
2024	-3.22	-1.50	0.26

Chart 39. WQI scores at the Eagle Creek monitoring sites over the 2013-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

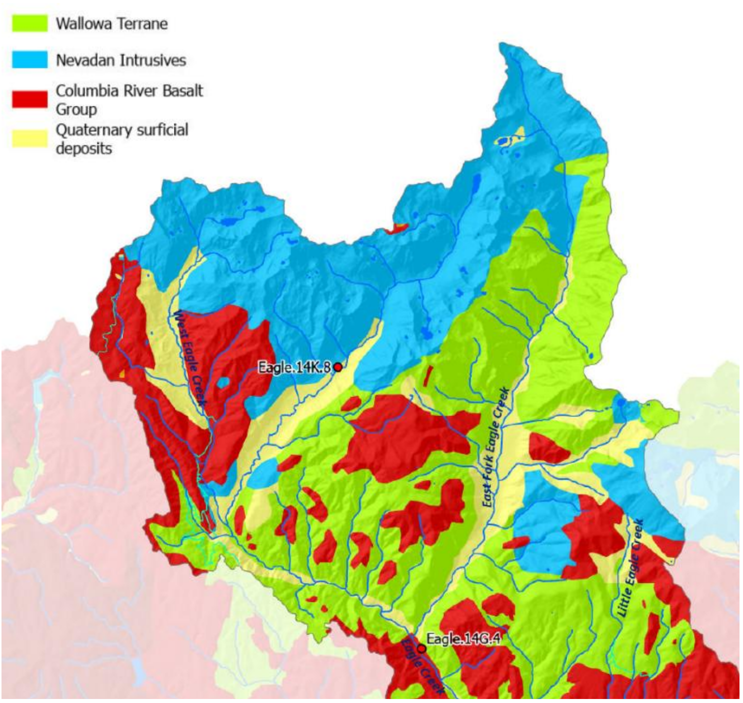


Mean estimates of WQI for sites were 0.60 higher for 2013-2024 period than for the 2022-2024 period, with mean estimates larger for Eagle.14K.8 and smaller at Eagle in Section 20 for the 2013-2024 period. While mean monthly WQI estimates were very similar overall, seasonal differences were more prominent, with lower WQI values for 2013-2024 period in fall and higher WQI estimates for 2022-24 in the spring. Overall water quality was lower in 2013, 2017, 2018, 2023, and 2024, with the lowest water quality seen 2017, with 2024 WQI values only 0.16 higher. Low WQI values in 2013 and 2014 were likely the result of biases from site composition, with sites sampled these years found within the lower watershed. In contrast, overall water quality was higher in 2016, 2020, and 2022, with highest water quality seen in 2020, although this is likely biased by the smaller numbers of months sampled and site selection.

Discussion and future plans

Important water quality impairments in the Eagle Creek watershed were mostly located in the lower watershed, with temperatures above 20 °C throughout much of the summer at Eagle @ New Bridge. Low water levels from irrigation withdrawals, lower levels of vegetation cover, and channelization might contribute to these warm temperatures. Temperatures were also above the cool water standard at Eagle in Section 20 and Eagle.14D.1, although the amount of time above the 20 °C was more limited than at New Bridge. Further upstream, temperatures were consistently below 20 °C but were generally above 12 °C throughout much of the summer. Overall, temperatures were generally suitable for redband trout in the middle and upper watershed but were less suitable for Bull Trout given elevated temperatures in the upper watershed.

Figure 20. Close up view the Upper Eagle Creek watershed showing bedrock geology. Note the large amounts of granitic/intrusive rock in the Main Eagle Creek and West Eagle Creek watersheds when compared to the primarily terrane and basalt volcanic derived rocks of East Eagle Creek and the lower watershed.



Most grab sample parameters were within standards at the Eagle Creek sites. The largest issue was related to low pH at Eagle.14K.8, but these low values were likely an artifact of the older pH probes used in 2022. Overall, oxygen was suitable for trout use at all sites, although oxygen concentrations did decrease significantly throughout the spring into summer. Similar decreases in oxygen from June to July were seen at Eagle.14D.1 and Eagle.14F.2.5 from dissolved oxygen loggers. While overall oxygen concentrations were above the minimum intergravel standard, they were consistently below the 11 mg/L spawning standard in both June and July. Conductivity was generally similar between the downstream and middle elevation sites but was extremely low at Eagle.14K.8. The higher conductivity measured at other sites indicates that this effect is limited to Eagle.14K.8, which is likely the result of low levels of dissolved minerals from the granitic bedrock of the Main Fork Eagle Creek watershed (Figure 20).

Most sites in the watershed were heavily impacted by the Hudson Creek landslide in August 2023, with higher turbidity measured in August and following June at the sites downstream of the East Eagle Creek confluence. The impacts were most notable immediately after the landslide but were more muted afterwards in the fall. Resuspended sediments caused by higher water levels in the spring resulted in high turbidity the following June and were deposited again through summer or moved into downstream reaches (Figure 21).

Figure 21. Fine sediments deposited by the Hudson Creek landslide. Sediments from the landslide both increased turbidity immediately after the landslide and during following spring.

Future monitoring in the Eagle Creek watershed should focus on continued monitoring to look at restoration and fire impacts and identify thermal refugia in the upper watershed. An important priority is assessing the impact of potential restoration on Eagle Creek between Eagle in Section 20 and Eagle @ New Bridge. These monitoring locations might be able to assess the impact of restoration on temperature and grab sample parameters but would require continued sampling and more sophisticated statistical tools to identify impacts. There is also a need to identify the impacts of the Town Gulch Fire on water quality in the affected areas, particularly near Eagle in Section 20, Eagle.14D.1, and Eagle.14F.2.5 where fire severity was highest. Reduction in vegetation cover and increase in erosion should increase stream temperatures and turbidity immediately after the fire, and should also be useful metrics in assessing the recovery in stream function after the fire as well.

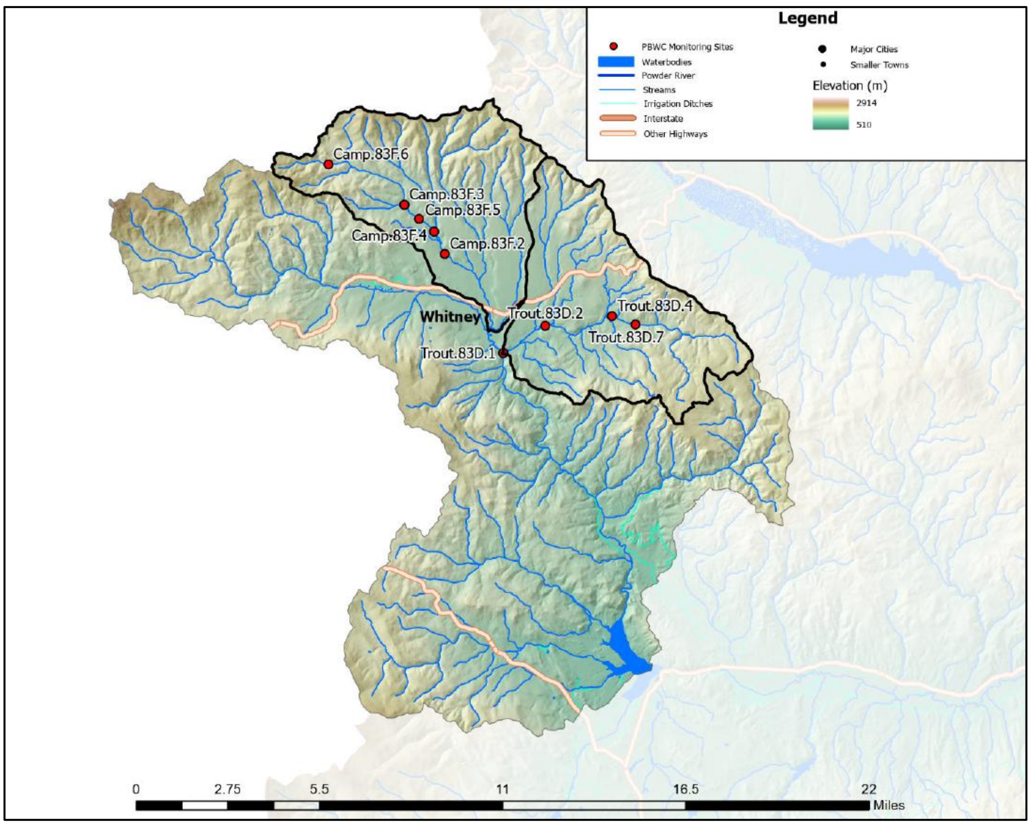
Finally, there is a need to look at other locations to better understand hydrology and water quality elsewhere in basin. Focus should be placed on East Eagle Creek to identify if geology is primary determiner of low conductivity and pH at Eagle.14K.8. There is also a need to identify cold water refuge in Upper Basin and in some smaller creeks. While temperature might be over 12 °C on the mainstem of Eagle Creek, smaller tributaries might contain suitable temperatures for Bull Trout, if reintroduction efforts were to be undertaken. These streams might also be useful for summer rearing and refugia for redband trout.

Camp and Trout Creek

Background

Camp Creek and Trout Creek are two tributaries to the North Fork Burnt River (NFBR). Both drain into Whitney Valley, a broad depositional valley in the middle section of the NFBR watershed. Camp Creek flows into the middle part of the valley near the town of Whitney, while Trout Creek flows into the lower part of the valley before it enters a more constrained section. Redband trout are present within both creeks, with primary usage related to spawning and winter rearing as well as a migratory corridor ([NPCC 2004a]()).

Map 9. The North Fork Burnt River watershed with 2022-2024 sample sites, major tributaries, and important features highlighted. Camp and Trout Creek watersheds outlined in black.



The headwaters of both creeks are primarily composed of mixed conifer forests with flow primarily driven by snowmelt and springs. The mid-elevation reaches are more dominated by ponderosa and lodgepole pine, with significant runoff coming from rain-snow events in the spring. A major feature of Trout Creek within these reaches is the Alder Creek Meadow, a smaller depositional valley located in the middle of the watershed (Figure 22). Further downstream, the creeks enter Whitney Valley where vegetation is more dominated by willow, alder, and grasses. At its confluence, Trout Creek drains a larger watershed than Camp Creek, but conditions are generally drier throughout much of the year (Table 36).

Table 36. Site characteristics for monitoring locations on Camp Creek and Trout Creek including ODFW Fish Habitat Type, Elevation (m) upstream drainage area (km^2), Modeled 1993-2011 NorWeST mean August stream temperature (°C), and established date.

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
Camp.83F.2	37741	Cool	1302	35.33	16.13	6/6/2014
Camp.83F.3	40977	Cool	1350	14.09	14.30	6/16/2018
Camp.83F.4	40979	Cool	1315	19.38	15.53	6/16/2018
Camp.83F.5	40978	Cool	1330	19.38	14.98	6/16/2018
Camp.83F.6	41460	Cool	1513	3.14	12.61	6/2/2022
Trout.83D.1	37124	Cool	1256	79.64	17.41	6/6/2014
Trout.83D.2	40982	Cool	1277	58.08	15.24	6/26/2018
Trout.83D.4	40981	Cool	1326	22.68	16.05	6/26/2018
Trout.83D.7	41459	Cool	1353	21	15.31	6/2/2022

Both streams have substantial downcutting and disconnection from their historical floodplain, particularly in the lower sections where riparian vegetation is sparse compared to historic conditions, particularly in the Alder Creek Meadows. Beaver presence is also limited throughout much of the stream reaches, with only a few dams present in lower Camp Creek and Trout Creek above private property. The largest beaver dam complexes are found in the lower part of Trout Creek downstream of the private inholding near the confluence with the NFBR, which were established in the early 1990's. In this section, 4-5 dams were present during the 2022-2024 monitoring period.

Figure 22. View of Alder Creek Meadows on Trout Creek. The buck and pole fencing was constructed in 1994 as part of efforts to address channel incision.

To address this degradation, the PBWC has been heavily focused on restoration actions in these watersheds, focusing on Low-Tech Process Based restoration (LTPBR) actions to aggrade the stream channel, increase riparian vegetation cover, as well as improving stream channel/ alluvial aquifer connectivity (Figure 23). Given the importance of this area for PBWC restoration activities, both creeks have been monitored more extensively than other locations within the watershed. Initial monitoring began in 2014 at two sites in lower Camp and Trout Creeks consisting of grab sample and temperature monitoring. Further temperature monitoring took place at eight more sites in 2018, including six sites included in the 2022-24 monitoring period.

Figure 23. BDA (Beaver Dam Analog) from restoration efforts on lower Camp Creek in summer 2024.

In total, 12 sites were sampled in the Camp Creek and Trout Creek watersheds during the 2022-2024 period. Of these sites, 9 were sampled over multiple years, with three located on Trout Creek and five located on Camp Creek. These sites on Trout Creek were placed in a variety of geomorphic zones, with one site located above the Forest-Private ownership boundary, one was located within the Alder Creek Meadow, and one site located upstream of the meadow. On Camp Creek, one located in the upper part of the watershed, with three located in the middle section of the stream within the restoration project reach, and one above the Forest-private ownership boundary. Among the sites sampled for only one year, one was located on Gimlet Creek, a tributary of Camp Creek sampled in 2023, one was located on Trout Creek just below Alder Creek Meadow sampled in 2024, and one was located on lower Trout Creek within a beaver dam pool sampled in 2022. The lowest site, Trout.83D.1, was sampled in 2023 and 2024.

The Wallowa Whitman National Forest was involved with the deployment and removal of temperature loggers at the Camp and Trout Creek sites. Due to other commitments, loggers were removed in September, earlier than other watersheds. While the logger deployments were generally successful most years, some issues occurred that reduced the comparability of the data, particularly at the site at Trout.83D.7, which had a temperature logger lost in 2023. Another issue was the placement of loggers in 2022, which differed from previous monitoring efforts, mostly on Trout Creek. Subsequent monitoring in 2023 and 2024 put significant effort into identifying the correct locations.

Grab Sample Monitoring

Dissolved Oxygen concentrations were above the 8 mg/L cold-water standard at all sites on Camp and most sites on Trout Creek. In general, oxygen concentrations were higher on Camp Creek, although there was some variation in both oxygen concentration and saturation were present within each creek. Similar patterns were seen in oxygen concentrations, with the lowest concentration found at the lowest sites in each creek's watershed. Some differences from this trend were seen at Camp.83D.5, where oxygen concentrations were lower than those seen at sites upstream and downstream. This, along greater daily temperature variation and lower conductivity measurements indicate greater surface water contributions at this site than for other locations within the middle Camp Creek reach.

Chart 40. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the Camp Creek monitoring sites over the 2022-2024 period. Cold water oxygen standard (8 mg/L) highlighted with solid red line.

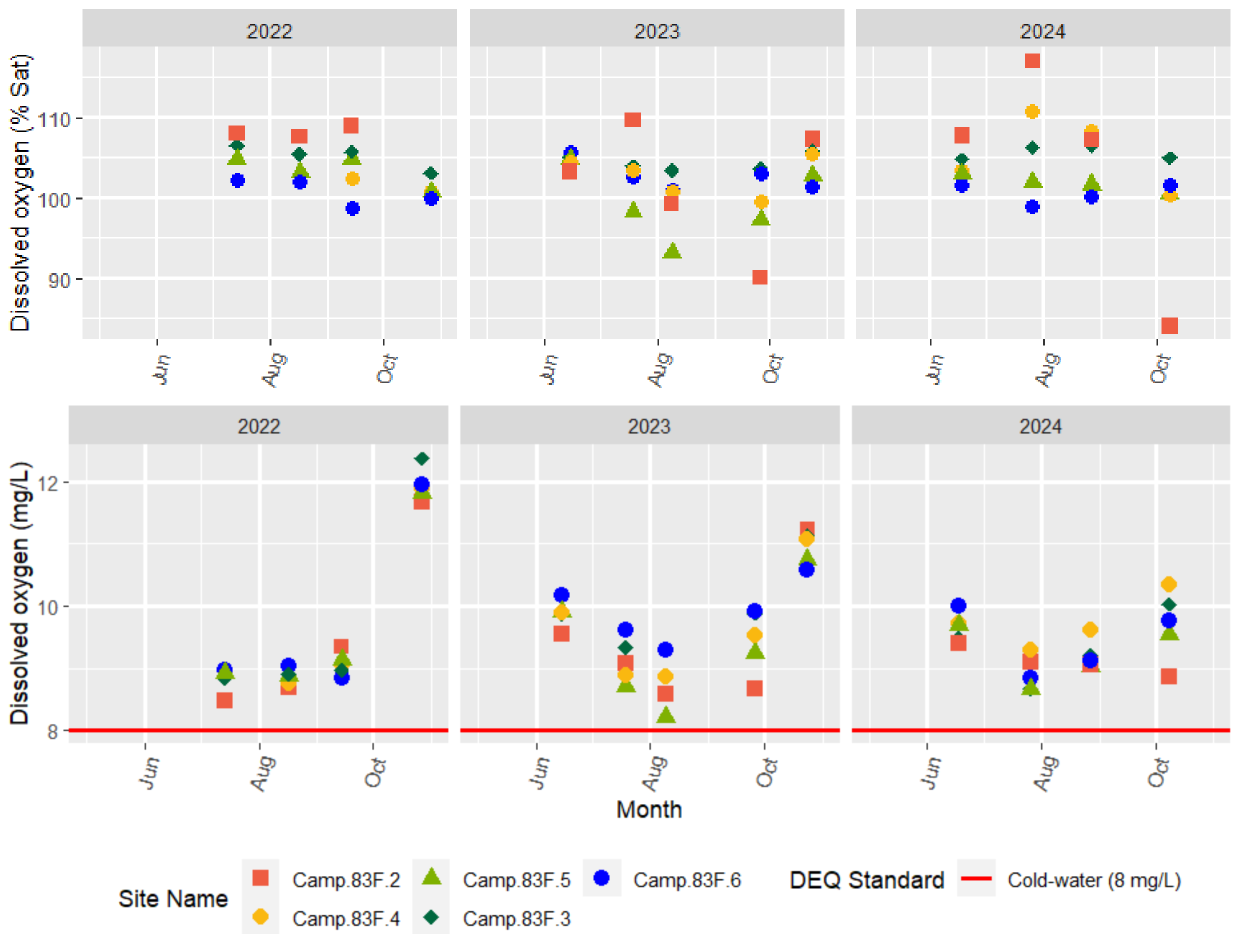


Chart 41. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the Trout Creek monitoring sites over the 2022-2024 period. Cool-water standard (dashed line) and cold-water standard (solid line) highlighted in red.

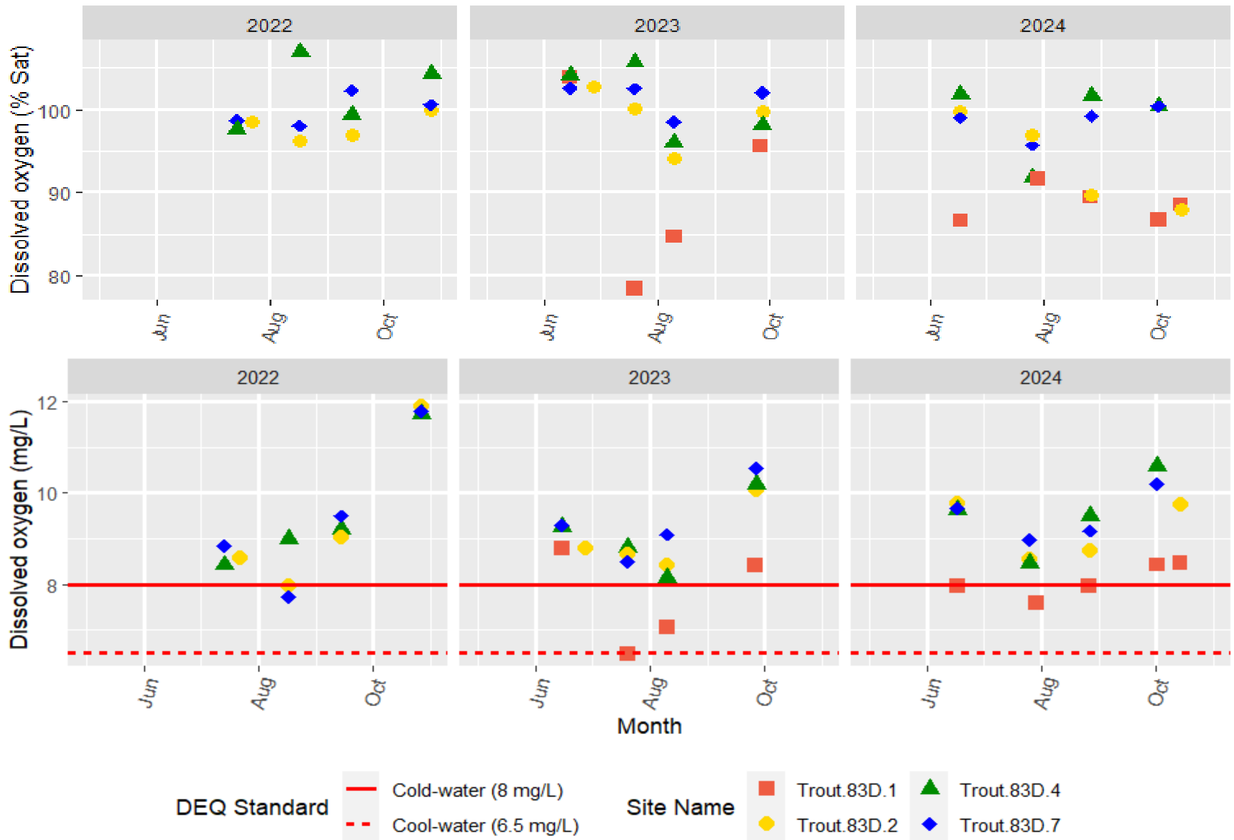


Table 37a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the Camp Creek and Trout Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Camp.83F.2	8.80	9.28	9.77
Camp.83F.4	9.21	9.69	10.18
Camp.83F.5	8.95	9.42	9.89
Camp.83F.3	9.22	9.69	10.16
Camp.83F.6	9.23	9.70	10.17
Trout.83D.1	7.40	7.95	8.50
Trout.83D.2	8.82	9.29	9.76
Trout.83D.4	9.05	9.52	9.99
Trout.83D.7	9.07	9.55	10.02

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	0.68	0.19	-0.29
July	-0.27	-0.69	-1.12
August	-0.23	-0.65	-1.07
September	0.31	-0.12	-0.56
October	1.71	1.27	0.84

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.39	0.10	0.58
2023	-0.36	0.06	0.49
2024	-0.59	-0.16	0.27

Seasonal patterns in oxygen were also present, with concentrations highest in the spring and fall, and lowest in July and were most likely the result of seasonal temperature patterns. These seasonal patterns were most clearly present at Trout.83D.1, downstream of the beaver dam complex. Low summer oxygen concentrations here are likely the result of pool stratification and a lack of mixing, particularly at this site where summer discharge can be quite low. While monthly patterns were present, yearly patterns in oxygen concentrations were less noticeable and were not significant from each other. Greater seasonal variability was also seen at the lowest Camp Creek sites in oxygen saturation, with higher levels of saturation seen during the summer than the middle and upper Camp Creek sites, while saturation was lower in the fall.

pH measurements were consistently within standards for aquatic life throughout the monitoring period on both creeks. Patterns for pH between sites indicated that pH decreased as watershed area increased, although this trend was not significant. pH was lowest at Trout.83D.1, likely due to the lower oxygen concentrations at this site related to beaver dam activity. The highest pH values were found further upstream at Trout.83D.2, although these measurements were not significantly different from the other Trout Creek sites above the Forest property boundary.

Chart 42. Observed pH measurements for the Camp Creek monitoring sites over the 2022-2024 period. Upper (solid line) pH standard highlighted.

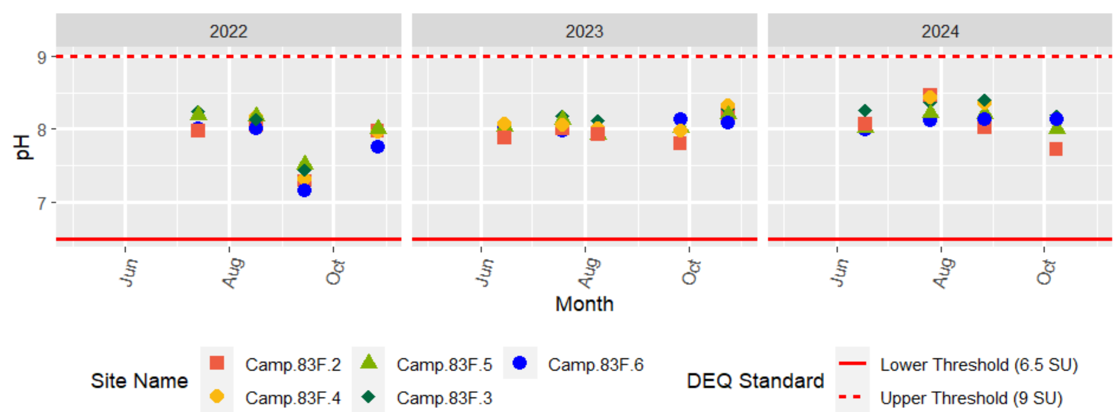
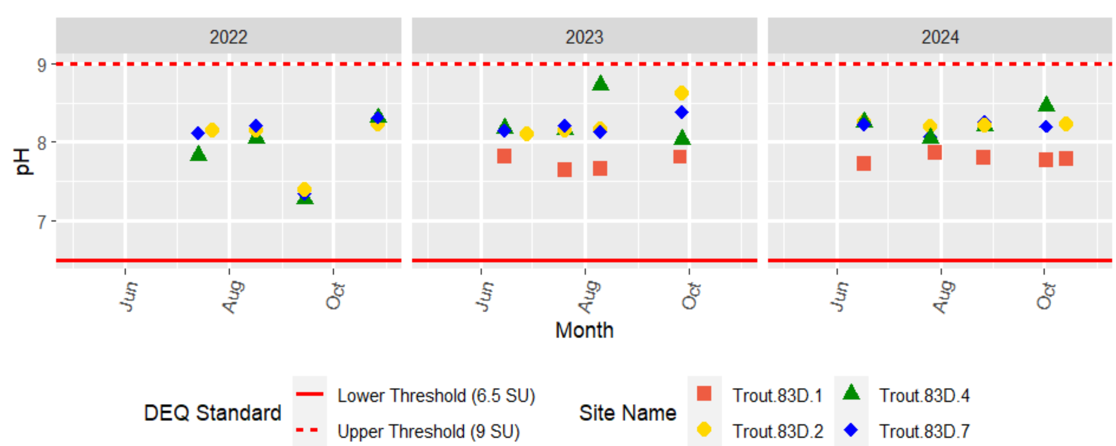


Chart 43. Observed pH measurements for the Trout Creek monitoring sites over the 2022-2024 period. Upper (dashed line) and lower (solid line) pH standards highlighted



Greater variability was also seen in seasonal pH measurements at Trout.83D.4, which is likely related to shifts in flow contribution from surface flow to groundwater flow during the summer period. No overall seasonal or yearly patterns were seen in pH. Regarding low pH values in September 2022, these measurements appear to be an outlier possibly related to equipment issues.

Table 38a. Estimates of mean pH for the Camp Creek and Trout Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Camp.83F.2	7.75	7.92	8.08
Camp.83F.4	7.87	8.03	8.20
Camp.83F.5	7.86	8.02	8.18
Camp.83F.3	7.95	8.11	8.27
Camp.83F.6	7.77	7.93	8.09
Trout.83D.1	7.47	7.66	7.85
Trout.83D.2	7.98	8.14	8.30
Trout.83D.4	7.95	8.11	8.28
Trout.83D.7	7.96	8.12	8.28

b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.19	-0.02	0.15
July	-0.06	0.08	0.23
August	-0.05	0.10	0.24
September	-0.40	-0.25	-0.10
October	-0.06	0.09	0.24

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.33	-0.16	0.01
2023	-0.06	0.09	0.23
2024	-0.07	0.07	0.22

Patterns in conductivity between sites were noticeable in both creeks, with higher conductivity measurements found at sites lower in the watershed. In general, conductivity was higher in Trout Creek after adjusting for basin size, with conductivity values 8% to 22% higher per km^2 at the Trout Creek sites when compared to similarly sized sites on Camp Creek. Similar to oxygen concentration, conductivity measurements at Camp.83D.5 differed somewhat from this trend, with higher conductivity measurements here on average than the two sites downstream. A similar pattern was seen at Trout.83D.2, where conductivity was 24% higher than Trout.83D.1. These patterns were likely the result of higher groundwater contributions at these sites compared other locations within each Creek.

Table 39a. Estimates of mean conductivity (in µS/cm) for the Camp Creek and Trout Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Camp.83F.2	170.68	182.38	194.88
Camp.83F.4	172.25	184.06	196.68
Camp.83F.5	174.00	185.57	197.91
Camp.83F.3	145.34	155.01	165.32
Camp.83F.6	130.73	139.42	148.70
Trout.83D.1	213.57	229.84	247.35
Trout.83D.2	268.23	285.07	302.97
Trout.83D.4	188.57	201.12	214.51
Trout.83D.7	185.16	197.49	210.64

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-54.16	-44.62	-34.42
July	-10.05	-0.43	9.71
August	-7.18	2.80	13.32
September	15.27	26.59	38.51
October	4.91	15.65	26.98

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-16.00	-3.94	8.95
2023	-22.04	-10.53	1.77
2024	3.12	14.47	26.48

Chart 44. Observed conductivity measurements (in µS/cm) for the Camp Creek monitoring sites over the 2022-2024 period.

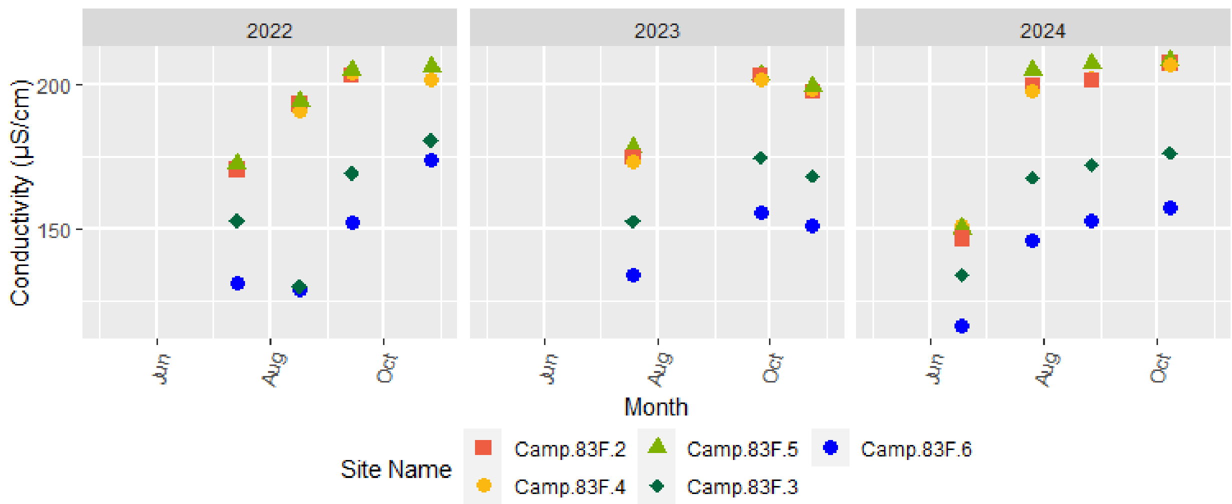
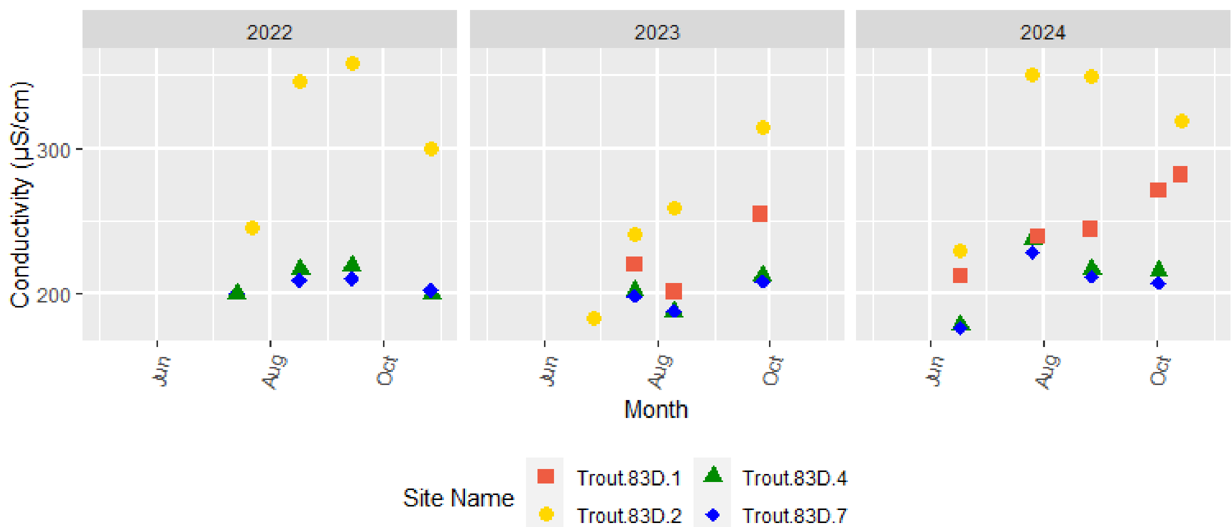


Chart 45. Observed conductivity measurements (in µS/cm) for the Trout Creek monitoring sites over the 2022-2024 period.



Seasonal trends within each creek also showed strong increases in conductivity over the course of the season from spring to early fall. These changes were most noticeable at Trout.83D.2, where larger variations in conductivity were observed over the course of the season, likely indicating shifts in flow contribution from surface flow in the spring to groundwater flow during the summer and fall. Yearly patterns in conductivity were also present, with measurements higher in 2024 and lower in 2023, likely related to differences in precipitation during the summer season.

Compared to the other parameters for the Trout and Camp Creek sites, the turbidity data had numerous samples (43/114) with failed accuracy checks, indicating suspect measurements at these sites. Still, the exceedance of the accuracy thresholds for these samples were generally small and almost all samples with failed accuracy checks only had one exceedance during the post-trip calibration check.

Table 40a. Estimates of mean turbidity (in NTU's) for the Camp Creek and Trout Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Camp.83F.2	0.82	1.39	2.38
Camp.83F.4	0.62	1.05	1.79
Camp.83F.5	0.62	1.03	1.73
Camp.83F.3	0.62	1.04	1.74
Camp.83F.6	0.48	0.81	1.35
Trout.83D.1	2.67	4.74	8.42
Trout.83D.2	1.29	2.04	3.21
Trout.83D.4	1.22	1.99	3.23
Trout.83D.7	1.58	2.57	4.17

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-1.08	-0.67	0.03
July	-0.35	0.27	1.20
August	-0.03	0.65	1.61
September	-0.45	0.11	0.93
October	-0.76	-0.36	0.20

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.60	0.10	1.28
2023	-0.67	-0.05	0.99
2024	-0.64	-0.05	0.92

Chart 46. Observed turbidity measurements (in NTU's) for the Camp Creek monitoring sites over the 2022-2024 period.

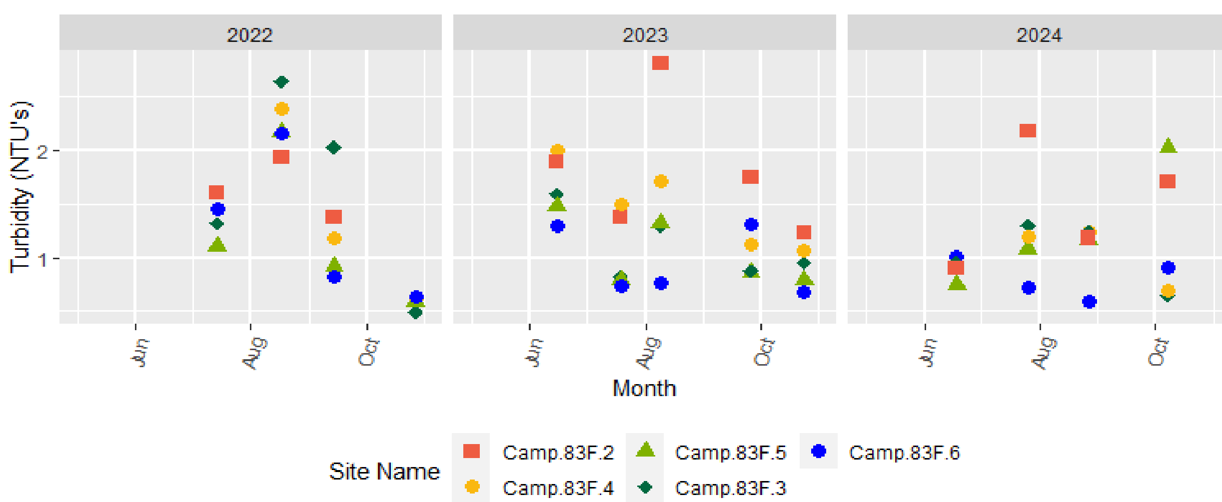
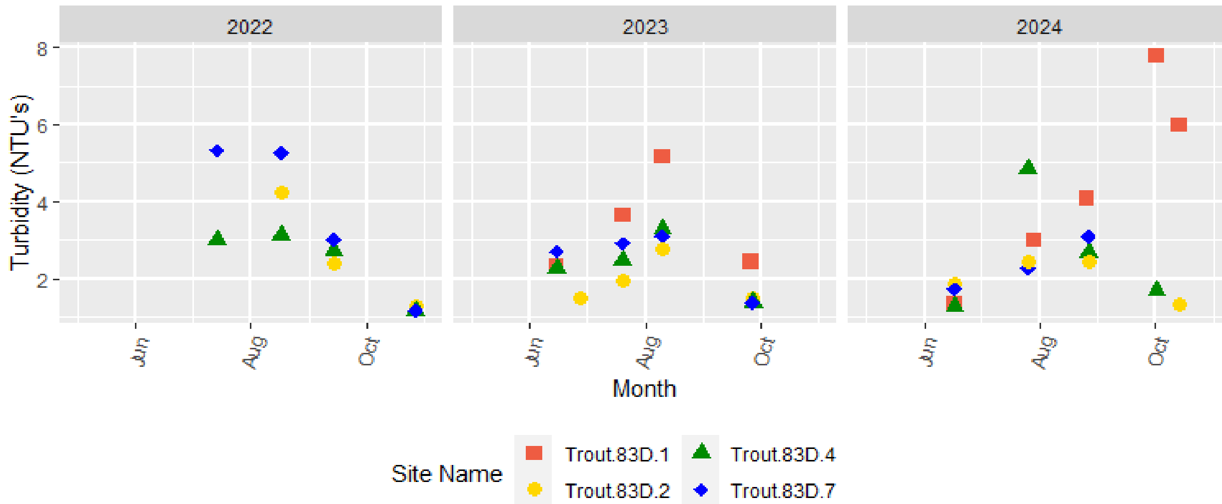


Chart 47. Observed turbidity measurements (in NTU's) for the Trout Creek monitoring sites over the 2022-2024 period.



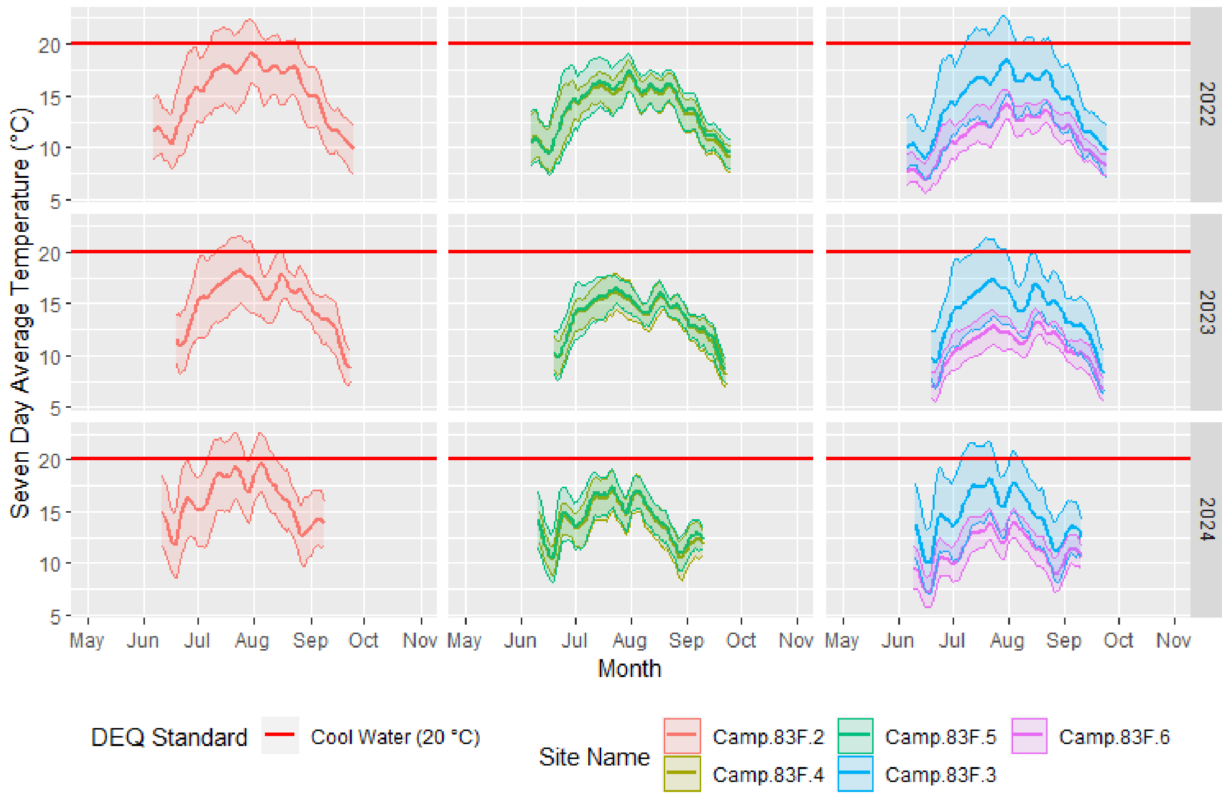
Spatial patterns between sites were mostly determined by creek and upstream drainage basin size. In general, turbidity was higher in Trout Creek than in Camp Creek, with mean turbidity values on Trout Creek ~80% higher compared to sites with similar upstream basin sizes in Camp Creek. The lower elevation sites draining a larger area also had higher turbidity values, although one exception to this pattern was found at Trout.83D.7, likely related to higher gradient and more surface flow at this location. Seasonal patterns differed to most other watersheds in the Powder basin, with turbidity highest during the summer at most sites in both creeks with mean turbidity in August 1.32 NTU's higher than in June.

Stream Temperature monitoring

Maximum daily stream temperatures were above the 20 °C cool water standard for significant periods of the summer at Camp.83F.2, Camp.83F.3, and Trout.83D.1, with each of these sites having more than 70% of days in July above 20 °C. Maximum daily temperatures were also above 20 °C for significant number of days at these sites in August, particularly at Camp.83F.2, where 43% of days were above the cool water standard. Other sites with maximum daily temperatures above the cool water standard include Camp.83F.3, Trout.83D.4, and Trout.83D.7, although no month had more than 6% of days above 20 °C.

Both mean and maximum stream temperatures showed increasing trends by drainage basin size at Camp Creek, although there was some variation from this trend. High mean and maximum temperatures were observed at Camp.83F.3, located above the restoration project reach. Mean temperatures at this site were 0.64 °C and 0.37 °C higher and maximum temperatures 2.55 °C and 2.00 °C higher than those at Camp.83F.5 and Camp.83F.4, respectively. Temperatures were also higher at Camp.83F.3than these two middle sites, including significantly lower maximum temperatures. These temperature patterns, alongside other evidence like higher conductivity measurements and lower dissolved oxygen, suggest that groundwater contributions appear higher at Camp.83D4 and Camp.83D.5 than the other sites on Camp Creek within the restoration reach.

Chart 48. Temperature profiles (in °C) for the Camp Creek monitoring sites over the 2022-2024 period, with lower site on the left , middle sites in the center, and upper sites on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cool water (solid line) standard in red.



On Trout Creek, mean temperatures were also heavily related to drainage basin size, but maximum temperatures showed some variation from this trend. This was particularly noticeable at Trout.83D.2 and Trout.83D.4, where maximum temperatures were 0.22 °C and 0.78 °C lower than the upstream Trout.83D.7 site, respectively. Like Camp Creek, these patterns suggest larger groundwater contributions at these sites, particularly at Trout.83D.2, where the increases in conductivity measurements and decreases in maximum temperatures from the upstream sites were largest.

Table 41a. Estimates of Mean and Maximum Daily temperatures (in °C) for the Camp Creek and Trout Creek monitoring sites over the 2022-2024 period

Site Name	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Camp.83F.2	14.26	14.99	15.72	17.08	17.89	18.70
Camp.83F.4	12.55	13.31	14.07	13.90	14.74	15.59
Camp.83F.5	12.85	13.58	14.31	14.48	15.29	16.09
Camp.83F.3	13.22	13.95	14.68	16.48	17.29	18.10
Camp.83F.6	9.86	10.59	11.32	11.19	12.00	12.80
Trout.83D.1	15.87	16.72	17.57	17.44	18.38	19.31
Trout.83D.2	12.38	13.11	13.84	14.03	14.83	15.64
Trout.83D.4	12.31	13.04	13.77	14.59	15.40	16.20
Trout.83D.7	12.18	13.00	13.82	14.71	15.61	16.52

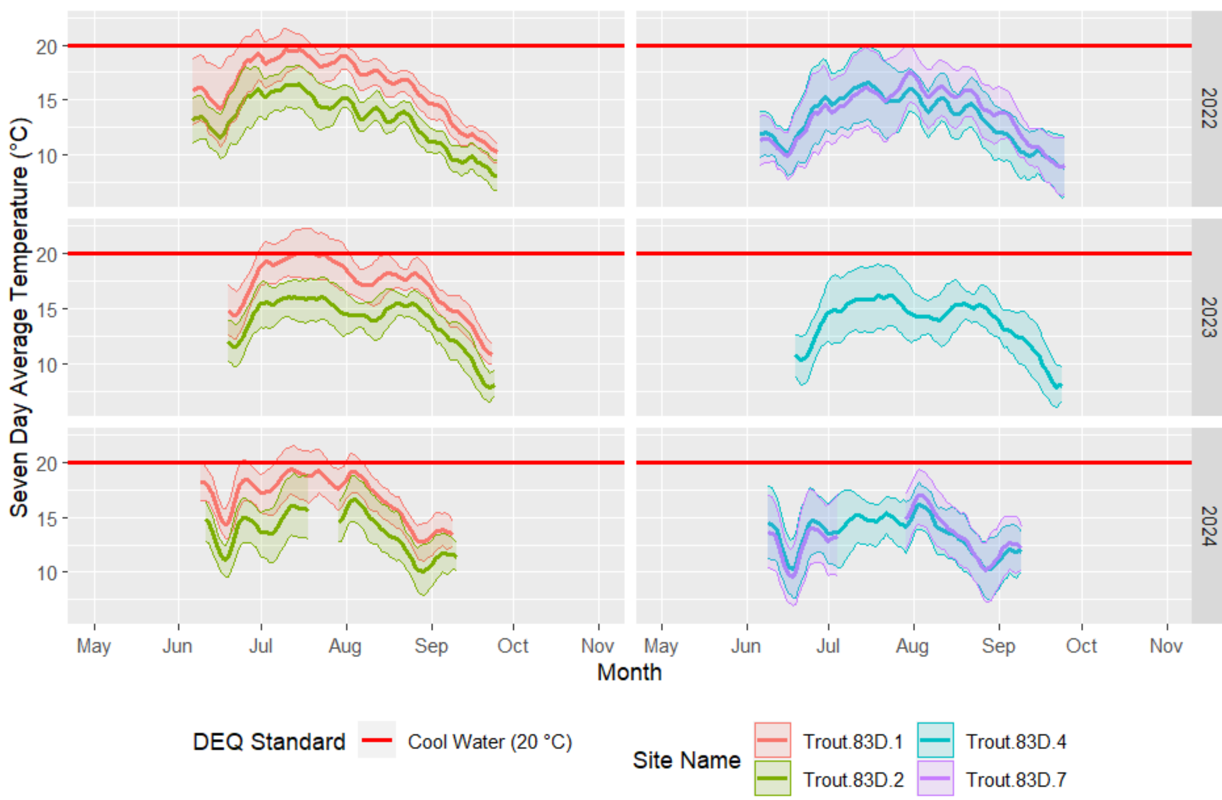
b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Month	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
June	-1.57	-0.84	-0.11	-1.45	-0.64	0.16
July	1.47	2.10	2.74	1.82	2.52	3.23
August	0.61	1.23	1.86	0.34	1.03	1.73
September	-3.14	-2.50	-1.85	-3.63	-2.91	-2.20

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Year	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
2022	-0.40	0.33	1.06	-0.41	0.39	1.20
2023	-0.81	-0.17	0.46	-0.99	-0.29	0.41
2024	-0.79	-0.16	0.46	-0.80	-0.10	0.59

Chart 49. Temperature profiles (in °C) for the Trout Creek monitoring sites over the 2022-2024 period, with lower sites on the left and upper sites on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cool water (solid line) standard in red.



Season patterns showed similar trends to those in other watersheds, with both mean and maximum temperatures highest in July and August and lowest in June and September. Differences in temperatures also existed between sites, although these differences were not significant. On average, mean and maximum temperatures were lower in 2023 and 2024 than in 2022, with patterns stronger for maximum temperatures. These patterns are likely related to higher stream flows and lower air temperatures during these years, particularly for 2023.

Comparisons to NorWeST stream temperatures models showed intriguing trends, particularly compared to other watersheds in the Powder Basin. Increases in mean August stream temperatures were predicted at Camp.83D.2 and Camp.83D.3, with lower temperatures found at the other sites. The most pronounced decreases were seen at Trout.83D.2 and Trout.83D.4, where observed 2022-24 temperatures were 1.37 and 2.06 °C cooler than the predicted 1993-2011 mean August temperatures. There might be several reasons for these lower observed temperatures, the location of Trout and Camp Creek within the large depositional Whitney Valley, where groundwater influences tend to be particularly strong. Another factor for these discrepancies is that the broad spatial resolution of the NorWeST dataset might be less suitable in predicting temperatures at these smaller spatial scales. At these smaller resolutions, localized impacts such as beaver dams and groundwater inputs might have a larger impact than the broader processes used in the NorWeST model such as basin size, elevation, and vegetation cover.

Table 42. Estimated and Observed mean August stream temperatures for Camp Creek and Trout Creek monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

Site	Estimated 2022-2024 Mean August Temp (°C)	Observed 2022-2024 Mean August Temp (°C)	NorWeST 1993-2011 Temp (°C)	NorWeST 2040 Temp (°C)	NorWeST 2080 Temp (°C)
Camp.83F.2	16.22	16.69	16.13	17.38	18.16
Camp.83F.4	14.54	14.58	15.53	16.78	17.54
Camp.83F.5	14.81	14.84	14.98	16.23	16.96
Camp.83F.3	15.18	15.53	14.30	15.55	16.25
Camp.83F.6	11.83	12.18	12.61	13.86	14.49
Trout.83D.1	17.95	16.91	17.41	18.66	19.49
Trout.83D.2	14.35	13.87	15.24	16.49	17.23
Trout.83D.4	14.27	13.99	16.05	17.30	18.08
Trout.83D.7	14.23	14.51	15.31	16.56	17.30

Water Quality Index

The lack of temperature logger data in October meant that no WQI values were available for this month. Still, useful patterns in WQI were identified between sites, months, and years. Overall, the WQI values indicated that water quality was consistently excellent at Camp.83F.6 and Camp.83F.5, while WQI was lowest at the lowest sites on both creeks, primarily at Camp.83F.2 and Trout.83D.1. Temperature was an important factor reducing water quality at Camp.83F.2 and Camp.83F.3, with temperature subindex values 7.15 lower than the other sites on Camp Creek and 6.98 lower than all other sites on Trout and Camp Creek. Oxygen was a primary facture reducing water quality at Trout.83D.1, particularly during the summer months, with mean July and August Oxygen sub-index values 14.68 lower than the other sites during these months. Trout.83D.4. also had low WQI values, primarily due to high pH values in August 2023.

Chart 50. WQI scores at the Camp Creek monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

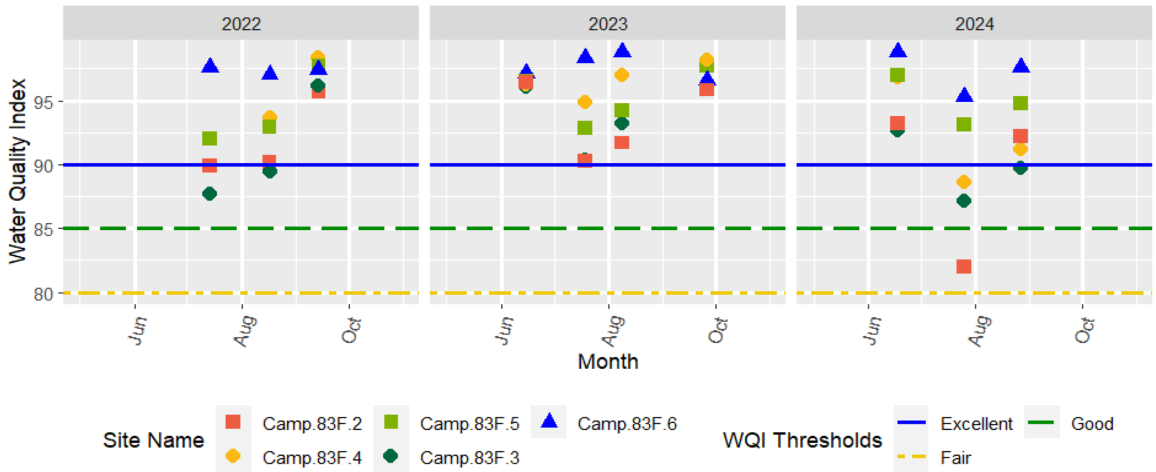


Chart 51. WQI scores at the Trout Creek monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

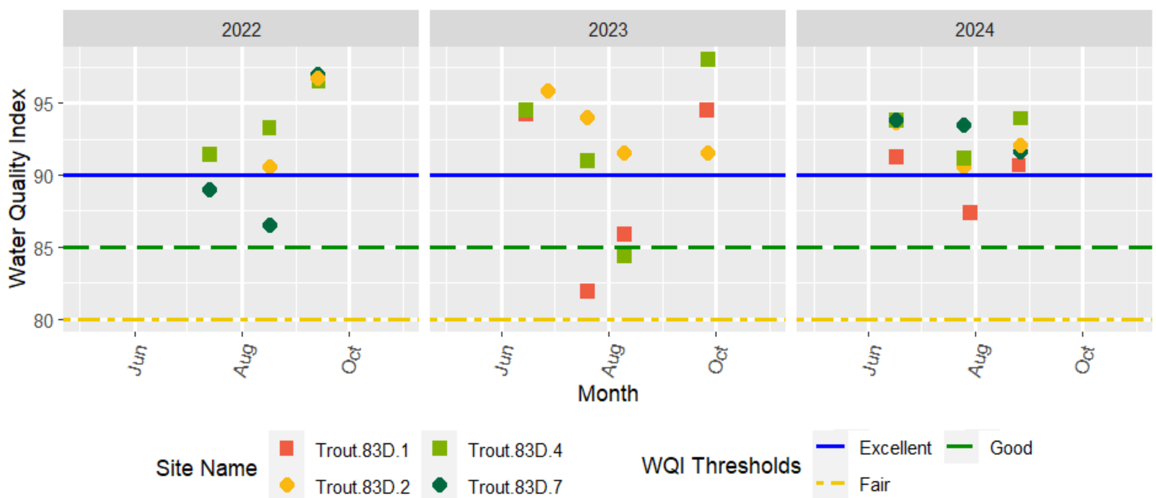


Table 43a. Estimates of mean WQI scores (from 10-100) for the Camp Creek and Trout Creek monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Camp.83F.2	89.82	92.04	94.31
Camp.83F.4	92.65	95.04	97.50
Camp.83F.5	92.98	95.27	97.62
Camp.83F.3	89.96	92.18	94.46
Camp.83F.6	95.50	97.86	100.27
Trout.83D.1	87.04	89.55	92.12
Trout.83D.2	90.65	92.99	95.40
Trout.83D.4	90.87	93.11	95.41
Trout.83D.7	90.21	92.70	95.27

b. Mean monthly WQI differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.65	1.64	3.99
July	-4.71	-2.79	-0.82
August	-3.41	-1.50	0.44
September	0.60	2.65	4.74

b. Mean yearly WQI differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-2.36	-0.11	2.19
2023	-1.63	0.35	2.37
2024	-2.20	-0.24	1.76

Overall water quality was lowest in July and August, primarily due to temperature, with an average subindex value of 87.8 for these months, compared to subindex values over 90 for the other parameters for the same period. WQI was also higher in 2023 than the other months, although this estimate wasn't significant from other years. The result was likely due to lower temperatures during this year, as the temperature subindex value was 1.47 higher than 2022 and 2.06 higher than 2024. Lower WQI values in 2024 were most likely driven by pH, with average pH subindex values were 2.38 lower than 2023 and 4.25 lower than 2022.

Full data for sites monitored over the 2013-2020 was only available for Trout.83D.1 and Camp.83D.2 for the years 2014-2017, limiting the comparability of WQI values between monitoring periods. Similar patterns were seen in WQI by site, with the largest difference being that mean predicted values for WQI were 1.5 index points higher for 2014-2024 period than for 2022-24. This result was likely biased by the lack of October index values for the 2022-2024 period. There were also stronger monthly patterns, also likely biased by lack of October data and smaller selection of sites. Overall, WQI was lower during the 2014-2017 period than 2022-2024 period, although these results were likely biased by number and types of sites monitored during the 2014-2017 period, since the 2014-2017 sites were those that had the lowest predicted WQI values for the 2022-2024 period.

Chart 52. WQI scores at the Camp Creek and Trout Creek monitoring sites over the 2014-2017 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

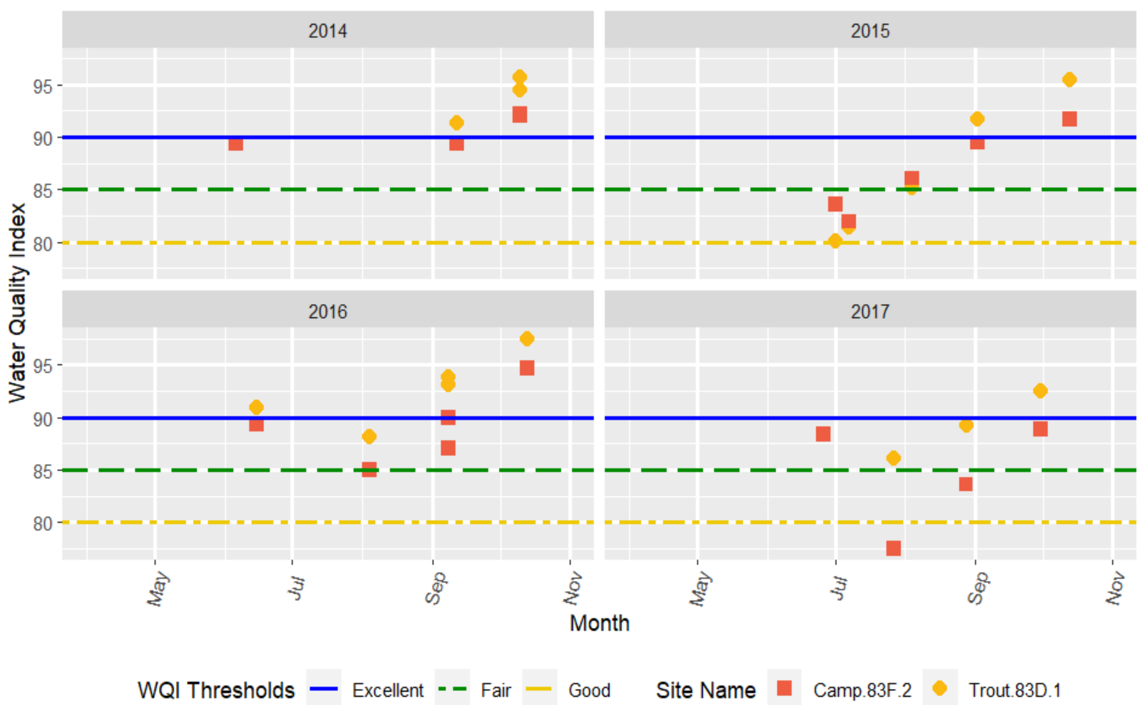


Table 44. Differences in mean yearly WQI values for sites on Camp and Trout Creeks from the overall mean over the 2014-2024 monitoring period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2014	-4.41	-1.89	0.70
2015	-5.00	-2.91	-0.76
2016	-2.65	-0.60	1.49
2017	-4.40	-2.19	0.07
2022	0.39	2.55	4.76
2023	0.94	2.76	4.61
2024	0.45	2.28	4.14

Discussion and future plans

Localized patterns in flow contribution and beaver presence play a large role in water quality within both Trout Creek and Camp Creek. These impacts are particularly notable in the middle section of Camp Creek, where higher groundwater contributions result in lower mean and maximum stream temperatures, as well as higher than expected conductivity measurements. Similar patterns with groundwater are seen on Trout Creek in and below alder Creek meadows, although these influences are less notable than on Camp Creek. Further downstream on Trout Creek, ponding by beaver dams resulted in cooler mean and maximum stream temperatures. Stratification within these dams and lower levels of mixing results in lower levels of dissolved oxygen, that while still above cool water standards, negatively impact overall water quality at this location (Figure 24).

Figure 24. Beaver ponds like those in lower Trout Creek create different patterns in water temperature and oxygen than those unimpacted by beaver dams due to stratification.

Moving forward there are several important questions that monitoring will need to address, the most important of these being the impacts of restoration on water quality on these creeks, especially temperature. Increased alluvial aquifer storage and subsurface flow from dam pools might result in lower mean and maximum temperatures, while increased absorption from solar radiation resulting from a larger wetted area might result in warmer overall stream temperatures. Previous studies have identified both patterns (**Mayer and Chang 2025**). Further identifying what patterns in stream temperature are present prior to and after restoration will require a longer series of data to account for variation in flow and temperatures to isolate the impact of restoration.

Another important issue to address is the impact of restoration on other parameters. As seen in lower Trout Creek, the reduction of mixing and stratification within beaver dams pools, while potentially beneficial to temperature, might reduce levels of oxygen compared to un-impounded reaches. Subsequent monitoring at Trout Creek and the NFBR looking at beaver dams influences on water quality identified similar patterns, with lower temperatures and oxygen levels within beaver dams, but with flow rates also having a strong impact on oxygen, with higher flow sites having higher oxygen levels and less stratification than those with lower flows. While not as important to water quality on Camp and Trout Creek compared to other parameters, turbidity is also another area of interest, especially given the higher turbidity measurements seen in the summer compared to other watersheds. Identifying the sources and processes related to these turbidity patterns would be useful in identifying strategies to mitigate their impact on aquatic organisms.

Alongside restoration actions on Camp and Trout Creek, there is also a need to identify the status of water quality on tributaries, particularly on Gimlet and Alder Creek. Monitoring in 2023 on Gimlet Creek identified low levels of oxygen saturation with moderate concentrations and elevated summertime turbidity. Both these tributaries also have significant amounts of intermittent reaches, where flow is only present during the high flow period in spring and early summer. Identifying the impact that these processes have on water quality in downstream reaches is a high priority for the PBWC, especially given their relationship to the restoration reaches on Camp and Trout Creek. There are also questions around the impact of restoration on these reaches, not just on water quality, but also on vegetation, flow permanence, and stream morphology.

North Fork Burnt River

Background

The North Fork Burnt River (NFBR) watershed covers 503 km^2 (193 mi^2) of mid-elevation habitat to the southwest of the Upper Powder Watershed. The upper sections of NFBR contain headwater streams in the foothills of Vinegar Hill near Greenhorn and mostly contain mixed conifer forests of Grand Fir, Larch, and Lodgepole Pine. Stream flow in these upper reaches is dominated by snowmelt in the spring and early summer with significant spring contributions in some creeks. The middle section is dominated by Whitney Valley, a large depositional valley that historically contained extensive wetlands and beaver dam complexes but is currently highly incised and lacks riparian vegetation such as willow, alder, and cottonwood. Major tributaries in the valley include Camp and Trout Creek, as well as several intermittent tributaries such as Dry Creek and Gimlet Creek. Downstream of the valley, the river enters a more constrained section, with heavier forest cover dominated by Ponderosa Pine and Larch. Several major tributaries enter the NFBR in this section, particularly the West Fork and Middle Forks of the Burnt River and China Creek.

Map 10. The NFBR watershed with 2022-2024 sample sites, major tributaries, and important features highlighted

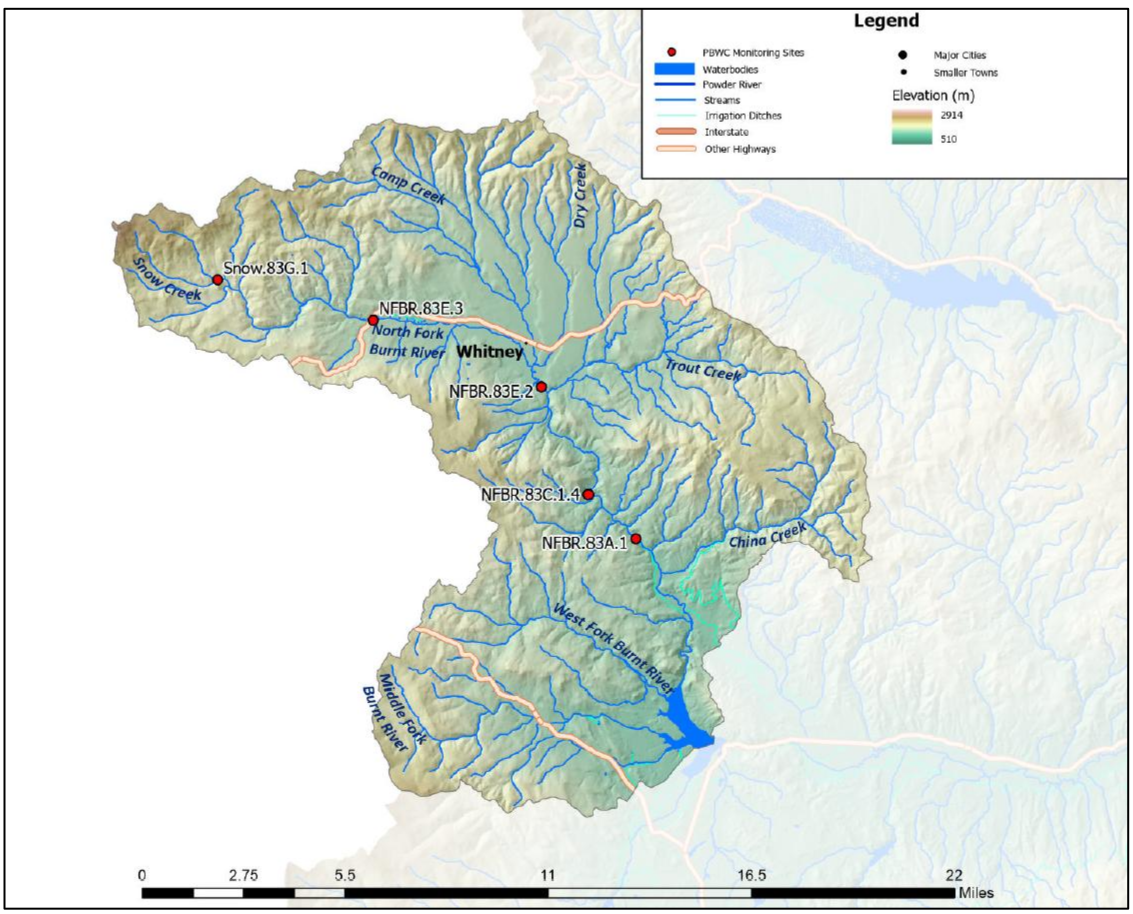


Table 45. Site characteristics for monitoring locations on the NFBR ncluding ODFW Fish Habitat Type, Elevation (m) upstream drainage area (km²), Modeled 1993-2011 NorWeST mean August stream temperature (°C), and established date.

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
NFBR.83A.1	37727	Cool	1216	316.72	17.04	6/6/2014
NFBR.83C.1.4	34253	Cool	1231	286.24	17.11	6/16/2018
NFBR.83E.2	37119	Cool	1256	186.3	18.66	6/6/2014
NFBR.83E.3	37120	Cool	1315	78.16	16.51	6/6/2014
Snow.83G.1	37122	Cool	1515	4.37	13.47	6/6/2014

One of the more interesting features of the NFBR is the large beaver colony in the lower section of Whitney Valley near the confluence with Trout Creek (Figure 25). This colony was established in the early 90's after beavers moved into the section and currently includes at least 10 dams on 0.6 miles of NFBR in both Whitney Valley and the downstream constrained reach. The changes in vegetation and hydrology in this section compared to pre-beaver conditions have been quite noticeable, particularly regarding willow cover and flood extent (Figure 26). Flooding in spring of 2023, for example, deposited large amounts of sediment that have had positive impacts of grass forage and willow growth within the floodplain.

Figure 25. Beaver dam in the Lower Section of the NFBR. This dam is part of a larger complex on the NFBR and Trout Creek that extends for ~0.6 miles.

Redband trout use was historically widespread throughout the NFBR watershed, and the upper section above Whitney Valley continues to have high utilization, particularly for summer and winter rearing. Stream reaches within Whitney Valley have seen significant declines in Redband Trout utilization, particularly for summer rearing, but is still heavily used as a migratory corridor. The stream reaches below Whitney Valley are primarily used as migratory corridor, with some winter rearing and spawning uses. Redband trout utilization is low in the reaches further downstream and on MFBR and WFBR, with little spawning and rearing and limited use as a migratory corridor ([NPCC 2004a]()).

Figure 26. Photos of NFBR in lower Whitney Valley in Fall 1992 (left) and Summer 2024 (right). Beaver activity in the lower valley since the early 1990's has resulted in stream channel aggradation and more hydric conditions for streamside vegetation.

Monitoring in the NFBR outside of Camp and Trout Creek during the 2022-24 period took place at five sites located on NFBR and its tributaries. Most of these sites were established in 2014 and have data for each year since allowing useful comparisons over time. These sites were primarily located to isolate the impacts of land use on water quality, with one site in upper watershed on Snow Creek, one site above Whitney Valley, and one site below. Two sites were located within the lower watershed, with one site in middle of constrained reach near a decommissioned gaging station and one site in the lower reach above the confluence with China Creek and a major diversion.

Monitoring in 2022 was heavily impacted by misplaced sites near the bottom of Whitney Valley and in the lowest parts of the watershed. The site for NFBR.83E.2 was placed within a beaver dam pool, and as a result had different temperature and oxygen profiles than the true site sampled in 2023 and 2024, which was located ~700 feet upstream in a free-flowing section of the creek. Sampling in 2025 continued at this site and at NFBR.83E.2 to determine the impact of beaver dams on the NFBR on temperature and other water quality parameters compared to free flowing conditions. Further downstream, monitoring took place at a location downstream of a major diversion, whereas the true site, NFBR.83A.1, was located upstream of the diversion. The difference in flow between these sites resulted in large changes in flow, pH, oxygen, and conductivity when compared to the true site. As a result of these factors, grab sample and temperature data associated with these sites in 2022 was not included in analysis.

Grab Sample Monitoring

Dissolved Oxygen was above both the 6.5 mg/L cool-water standard and the 8 mg/L cold-water standard at all sites throughout the 2022-24 monitoring period. Mean dissolved oxygen was higher at the lower elevation sites than those in the upper part of the basin and also had higher levels of oxygen saturation, particularly in the summer and fall. Patterns in oxygen concentrations at the sites above Whitney Valley were more closely associated with temperature, with both lower concentrations of and saturation of oxygen in the summer compared to those in early summer and fall. Concentrations of dissolved oxygen were higher later in the year at NFBR.83E.2, with high levels of saturation (>115%) beginning in late summer and peaking in September and October. Similar patterns in dissolved oxygen were seen at the lower sites (NFBR.83C.1.4, NFBR.83A.1), but with an earlier peak in oxygen saturation at these sites in August. The patterns in oxygen concentrations and saturations in these lower sites are indicative of photosynthetic oxygen production from eutrophication and related nutrient enrichment issues.

Chart 53. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the NFBR watershed monitoring sites over the 2022-2024 period. Cool-water standard (red line) highlighted.

Table 46a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the NFBR watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
NFBR.83A.1	9.52	10.29	11.06
NFBR.83C.1.4	9.77	10.45	11.12
NFBR.83E.2	9.50	10.27	11.04
NFBR.83E.3	9.24	9.92	10.59
Snow.83G.1	9.13	9.81	10.49

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-1.36	-0.59	0.18
July	-1.76	-1.07	-0.39
August	-1.38	-0.69	0.00
September	-0.90	-0.21	0.48
October	1.69	2.56	3.43

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-1.03	-0.26	0.51
2023	-0.55	0.02	0.58
2024	-0.33	0.24	0.81

pH Measurements were within biological thresholds at the sites upstream and downstream of Whitney Valley but were above the upper threshold in the lower part of Whitney Valley. Overall, there was an increasing trend in pH with increasing basin size, with mean measurements lowest in Snow Creek and highest at NFBR.83A.1. Geology might contribute to the high pH at the lower sites, with significant amounts volcanic rock throughout the basin.

Seasonal patterns in pH appear to be more consistent at the upper sites, which were generally similar to each other throughout entire year. pH measurements were more similar between the upper and lower watershed sites early in the year, but the lower sites diverged during the summer and fall. Seasonal patterns in pH followed similar patterns as oxygen, with pH elevated at the lower sites in late summer and fall, particularly at NFBR.83E.2. pH was especially elevated in September and October, with measurements during these months exceeding the 9.0 upper threshold in 2024. Like oxygen concentrations, the highest pH was seen in late summer at the sites downstream of Whitney Valey, with pH measurements close to exceedance at NFBR.83A.1in July during 2023 and 2024.

Chart 54. Observed pH measurements for the NFBR watershed monitoring sites over the 2022-2024 period. Upper (solid line) and lower (dashed line) pH recommended standards highlighted.

Table 47a. Estimates of mean pH for the NFBR watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
NFBR.83A.1	8.27	8.56	8.85
NFBR.83C.1.4	8.21	8.47	8.72
NFBR.83E.2	8.13	8.42	8.71
NFBR.83E.3	8.05	8.30	8.56
Snow.83G.1	7.91	8.16	8.42

b. Mean monthly pH differences from mean values over the 2022-2024 period

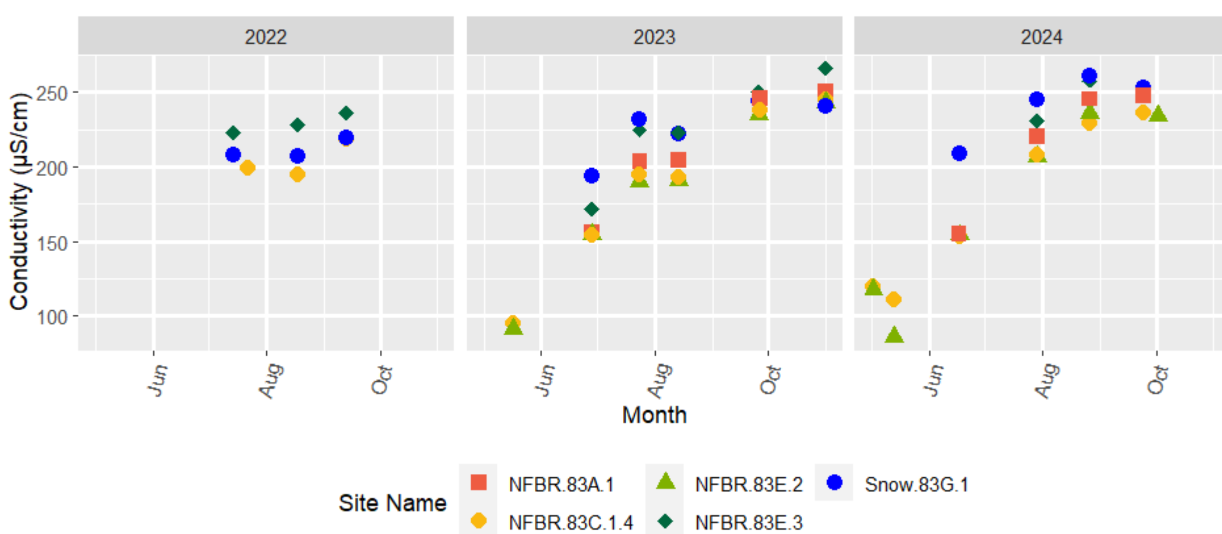
Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.46	-0.16	0.13
July	-0.09	0.17	0.43
August	-0.22	0.04	0.30
September	-0.37	-0.11	0.15
October	-0.27	0.05	0.38

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.63	-0.34	-0.05
2023	-0.12	0.10	0.31
2024	0.03	0.24	0.46

Like oxygen, the patterns in pH, indicate possible issues with eutrophication and nutrients in the lower sites, although the impact differs between sites within and below Whitney Valley. Summer is likely a bigger period for eutrophication at NFBR.83C.1.4 and NFBR.83A.1, while these conditions are more prevalent in the fall at NFBR.83E.2. pH was slightly lower in 2022 and higher in 2024, although the trends in pH in 2022 are likely related to equipment, as this year had issues with pH probes during some trips. Flow might also be a factor in why 2024 had higher mean pH measurements than 2023 due to lower summer flows which might exacerbate issues with eutrophication.

Chart 55. Observed conductivity measurements (in µS/cm) for the NFBR watershed monitoring sites over the 2022-2024 period.



Unlike many other watersheds, conductivity was highest at the upper elevations sites above Whitney Valley, with the highest mean conductivity found at Snow.83G.1, possibly related to groundwater or underlying geology. In the downstream reaches, there was a large decrease in conductivity between NFBR.83E.3 and NFBR.83E.2 through Whitney Valley, while conductivity increases were smaller from the lower Whitney Valley site to locations downstream.

Table 48a. Estimates of mean conductivity (in µS/cm) for the NFBR watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
NFBR.83A.1	194.08	209.33	225.78
NFBR.83C.1.4	189.56	202.53	216.38
NFBR.83E.2	185.06	199.51	215.09
NFBR.83E.3	208.80	223.09	238.35
Snow.83G.1	211.78	226.27	241.75

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

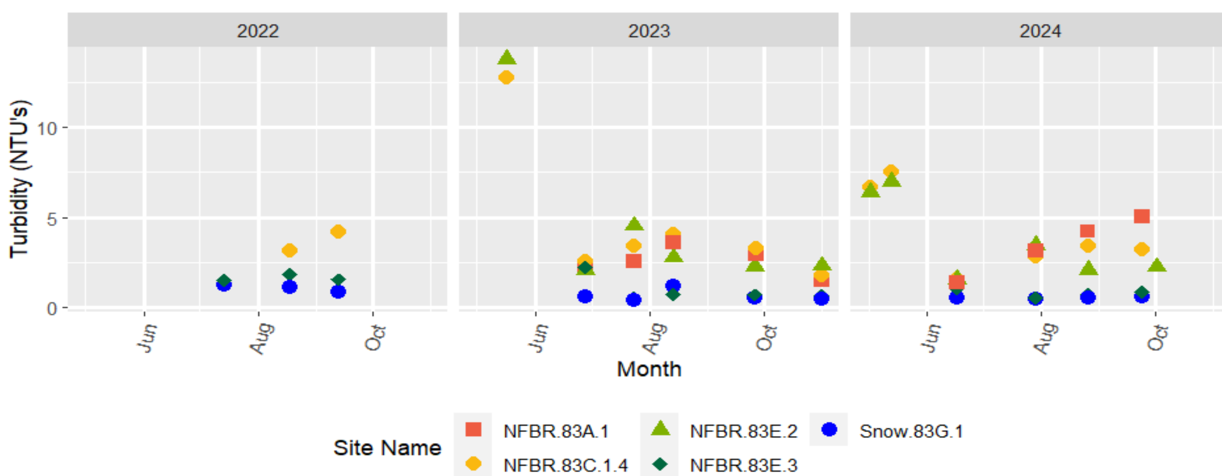
Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-65.61	-53.93	-41.34
July	-17.17	-3.47	11.19
August	-10.23	3.96	19.13
September	5.86	21.23	37.67
October	12.60	32.22	53.54

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-26.23	-11.63	4.13
2023	-11.54	-0.09	12.01
2024	-0.40	11.72	24.53

Seasonal patterns showed an increasing trend in conductivity throughout the summer into the early fall. This pattern was particularly notable in lower sites, with larger differences in conductivity between the upper NFBR in the summer giving way to similarly higher conductivity discharge in the fall. While mean conductivity at the monitoring sites was higher in 2024, there was limited evidence that these differences were significant from the other years. Alongside this, mean 2022 measurements were likely biased by smaller selection of sites.

Chart 56. Observed turbidity measurements (in NTU's) for the NFBR watershed monitoring sites over the 2022-2024 period.



While turbidity was below lower than 20 NTU's at all of the NFBR sites, it remained somewhat elevated overall compared to other similarly sized watersheds during the summer and fall, particularly at the lower watershed sites. Mean turbidity measurements increased in a downstream direction, with particularly large increases in mean turbidity through Whitney Valley between NFBR.83E.3 and NFBR.83E.2. The reason for this large increase is possibly related to primary production from wide shallow conditions and lack of vegetation in Whitney Valley. Residual impacts from the higher erosion rates due to the large amounts of incision within the valley might also be an important contributor to these high turbidity values.

Table 49a. Estimates of mean turbidity (in NTU's) for the NFBR watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
NFBR.83A.1	2.63	4.21	6.74
NFBR.83C.1.4	2.75	4.04	5.95
NFBR.83E.2	2.65	3.98	5.97
NFBR.83E.3	0.85	1.30	1.98
Snow.83G.1	0.59	0.90	1.37

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	1.40	3.39	6.34
July	-1.71	-1.09	-0.09
August	-1.50	-0.87	0.08
September	-1.30	-0.64	0.32
October	-1.39	-0.79	0.06

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.19	0.81	2.23
2023	-1.08	-0.24	1.09
2024	-1.17	-0.56	0.34

Turbidity was highest in the spring and dropped to yearly lows in June at most sites. Turbidity increased each month afterwards from early summer through fall with a peak in August and September, although differences in peaks outside of spring differed between years. Seasonal patterns in turbidity differed between years, with decreases after August in 2023, while increases were seen through October in 2022 and 2024. There was also a noticeable divergence in turbidity between upper and lower sites over course of the season, particularly in 2022 and 2024.

Stream Temperature monitoring

Stream temperatures were above the 20 °C cool-water standard at NFBR.83E.3 and the lower watershed sites but were more suitable for Redband Trout throughout the year at Snow Creek, with no days above 20 °C during the monitoring period. Temperatures increases were notable moving in a downstream direction from the Upper watershed, particularly in mean and maximum temperatures between Snow Creek and NFBR.83E.3. Overall, a 3.83 °C increase in mean temperatures, and a 4.78 °C increase in maximum temperatures were seen between Snow Creek and NFBR.83E.3. While changes in elevation and vegetation might be important factors for these changes, there has been significant alteration of stream morphology due to past dredge mining, which might alter groundwater patterns in the lower site.

There was also a noticeably large increase in temperatures between upper sites above and below Whitney Valley. Overall, there was a 2.96 °C increase in mean temperatures and a 2.87 °C increase in maximum temperatures between NFBR.83E.3 and NFBR.83E.2, likely due to lower groundwater recharge alongside reduced vegetation cover in Whitney Valley. Overall, mean temperatures were 1.02 °C and 1.70 °C and maximum temperatures were 1.81 °C and 2.04 °C lower at NFBR.83C.1.4 and NFBR.83A.1, respectively, when compared to NFBR.83E.2. Forest and topographic shade might also be important factors limiting temperature increases in this reach as well

Chart 57. Temperature profiles (in °C) for the NFBR watershed monitoring sites over the 2022-2024 period, with lower sites on the left and upper sites on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cool water standard in highlighted with red line.

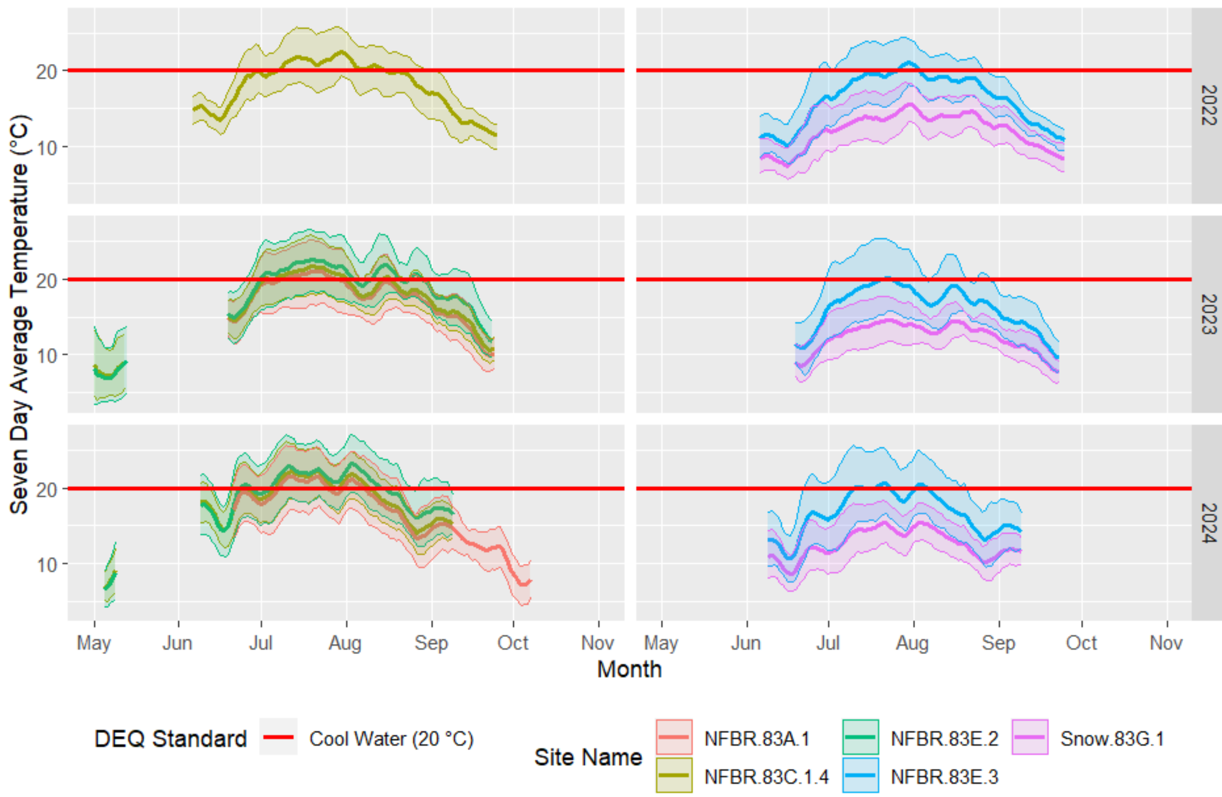


Table 50a. Estimates of Mean and Maximum Daily temperatures (in °C) for the NFBR watershed monitoring sites over the 2022-2024 period

Site Name	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
Snow.83G.1	11.03	11.92	12.81	13.63	14.53	15.43
NFBR.83E.3	14.86	15.75	16.64	18.41	19.31	20.22
NFBR.83E.2	17.59	18.71	19.82	21.06	22.19	23.32
NFBR.83C.1.4	16.79	17.68	18.58	19.47	20.38	21.28
NFBR.83A.1	15.89	17.01	18.12	19.02	20.15	21.28

b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Site Name	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
June	-2.53	-1.64	-0.75	-2.58	-1.68	-0.78
July	1.95	2.84	3.73	2.77	3.68	4.58
August	0.47	1.36	2.25	0.40	1.31	2.21
September	-3.45	-2.56	-1.67	-4.21	-3.30	-2.40

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Site Name	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
2022	-0.97	-0.07	0.82	-1.13	-0.23	0.67
2023	-1.09	-0.20	0.70	-1.16	-0.26	0.65
2024	-0.62	0.27	1.16	-0.41	0.49	1.39

All sites in the NFBR watershed except for Snow Creek showed increases in 2022-2024 mean August stream temperatures when compared to modelled 1993-2011 NorWeST modelled estimates. No detectable change was seen in mean August temperatures at Snow Creek, with the temperature increases likely moderated due to high levels of groundwater at this site. Mean stream temperature increases compared to the NorWeST for the NFBR watershed overall averaged 1.24 °C, with largest increase seen at NFBR.83C.1.4.

Table 51. Estimated and Observed mean August stream temperatures for the NFBR watershed monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

Site	Modelled 2022-2024 Mean August Temp (°C)	Observed 2022-2024 Mean August Temp (°C)	NorWeST 1993-2011 Temp (°C)	NorWeST 2040 Temp (°C)	NorWeST 2080 Temp (°C)
Snow.83G.1	13.28	13.41	13.47	14.64	15.39
NFBR.83E.3	17.11	17.69	16.51	17.76	18.55
NFBR.83E.2	20.07	20.00	18.66	19.96	20.79
NFBR.83C.1.4	19.04	18.84	17.11	18.38	19.18
NFBR.83A.1	18.37	17.74	17.04	18.31	19.11

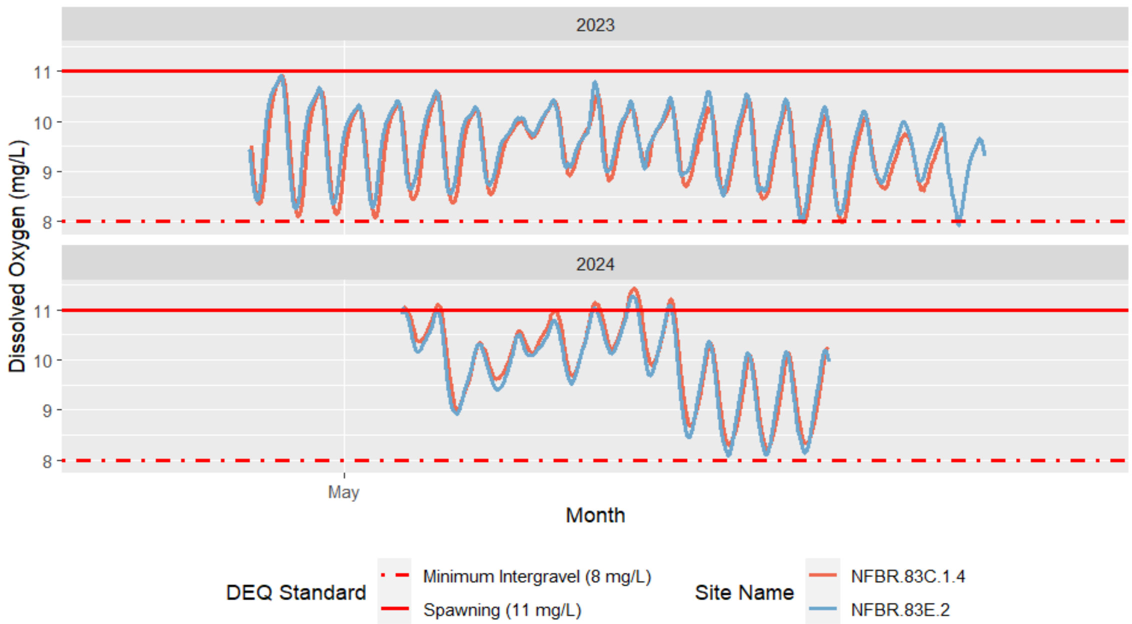
Dissolved Oxygen monitoring

Dissolved oxygen loggers were installed at NFBR.83E.2 and NFBR.83C.1.4 in 2023 and 2024 to monitor oxygen concentration during the spring redband trout spawning season. Logger installations were earlier and longer in 2023. Accuracy checks found that logger readings were 0.51 mg/L lower at NFBR.83C.1.4 and 0.33 mg/L lower at NFBR.83E.2 in 2023 when compared to 2024. This bias reduced the comparability between sites and years for the monitoring period.

Oxygen concentrations at the sites were closely similar to one another, with NFBR.83E.2 0.07 mg/L higher in 2023 and -0.17 mg/L lower in 2024 than NFBR.83C.1.4. Oxygen concentrations were consistently above the 8 mg/L intergravel minimum except for two days in 2023 but were mostly below the 11 mg/L spawning standard. Oxygen concentrations never exceeded 11 mg/L in 2023 and were only above 11 mg/L for 6.8% of measurements, but logger accuracy likely resulted lower recorded oxygen concentrations in 2023. Overall, oxygen concentrations were indicative that oxygen levels might be too low for egg and fry survival, particularly in June when oxygen concentrations are lower and temperatures are higher.

Despite the accuracy concerns, the observed data identified useful relationships in dissolved oxygen over time and between sites in the spring. Oxygen concentrations closely followed temperature (R^2 = 0.82), with concentrations decreasing -0.20 mg/L for every 1 °C increase in temperature (Appendix D). Oxygen concentrations were highest at noon and lowest between 4:00 and 8:00 in the evening, indicating that photosynthesis is a major source for oxygen for these systems even in the spring. Daily oxygen concentrations changed 0.84 mg/L on average, with daily ranges similar between sites and years. The timing of maximum and minimum oxygen concentrations were similar over time, with peaks in concentrations occurring one hour earlier at NFBR.83E.2.

Chart 58. Dissolved oxygen profiles (in mg/L) for the Spring NFBR watershed oxygen logger monitoring sites in 2023 and 2024. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



Water Quality Index

Water quality was variable throughout watershed and over the course of the year, and somewhat related to stream size, with Snow Creek having consistently excellent water quality, while the sites in the lower watershed frequently had conditions indicating Poor water quality. Large decreases in WQI scores were notable between Snow Creek and NFBR.83E.3 and between NFBR.83E.3 and NFBR.83E.2, decreasing 8.11 and 7.77 points, respectively. For NFBR.83E.2, temperature was the parameter most associated with low WQI scores, with an average sub-index score of 62.0. pH also resulted in low WQI scores, with a sub-index score of 74.9. Temperature and pH were also important factors limiting water quality at NFBR.83E.3, with average sub-index values of 77.2 and 81.5, respectively.

WQI estimates indicated that overall water quality was highest in June and lowest in July. Temperature was a major factor lowering WQI scores during this season, particularly on the NFBR mainstem, with mean temperature sub-index scores of 60.7 compared to 79.7 for DO, 73.8 for pH, and 91.7 for turbidity during July and August. pH also had a negative impact on water quality at these sites, although the impacts were more noticeable in the fall, with September and October pH subindex scores of 79.5, compared to overall WQI scores of 86.4 for these months. Oxygen sub-index values were also lower, especially at NFBR.83E.2, averaging 71.2 in September and October. While water quality was consistently excellent at Snow.83G.1, high pH in the summer and fall was still an important factor in lowering WQI, with an average pH subindex value of 86.6, compared to 92.1 for temperature.

Chart 59. WQI scores at the NFBR watershed monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.

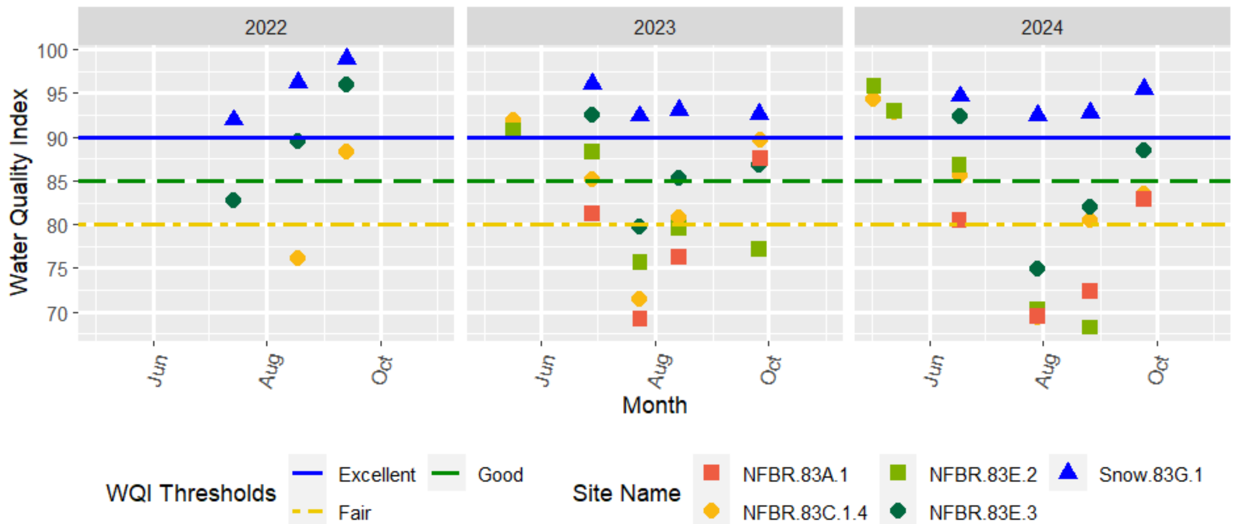


Table 52a. Estimates of mean WQI scores (from 10-100) for the NFBR watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
NFBR.83A.1	73.95	78.29	82.90
NFBR.83C.1.4	77.06	81.11	85.38
NFBR.83E.2	74.76	79.20	83.90
NFBR.83E.3	82.77	86.92	91.28
Snow.83G.1	90.48	95.03	99.79

b. Mean monthly WQI differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	0.49	5.45	10.71
July	-10.67	-6.70	-2.53
August	-6.40	-2.35	1.91
September	-0.75	3.60	8.18
June	0.49	5.45	10.71

c. Mean yearly WQI differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-2.34	2.45	7.53
2023	-3.64	-0.22	3.34
2024	-5.54	-2.23	1.22

Trends in WQI since 2014 were somewhat biased by site composition, particularly in 2019 and 2020, when only NFBR.83E.2 was sampled. Excluding these years, WQI was highest in 2022 and lowest in 2017. For site monitored from 2022-24, only 2024 had WQI values lower than the longer 2013-2024 period. Other years where WQI was lower than average were 2014, 2016, and 2017. Flow likely plays a large role in these patterns, especially considering drought from 2014 to 2016, although the lack of consistent sites and the large range in mean WQI estimates cannot rule out other factors.

Chart 60. WQI scores at the NFBR watershed monitoring sites over the 2014-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WQI), and Poor (80-60 WQI) thresholds highlighted.

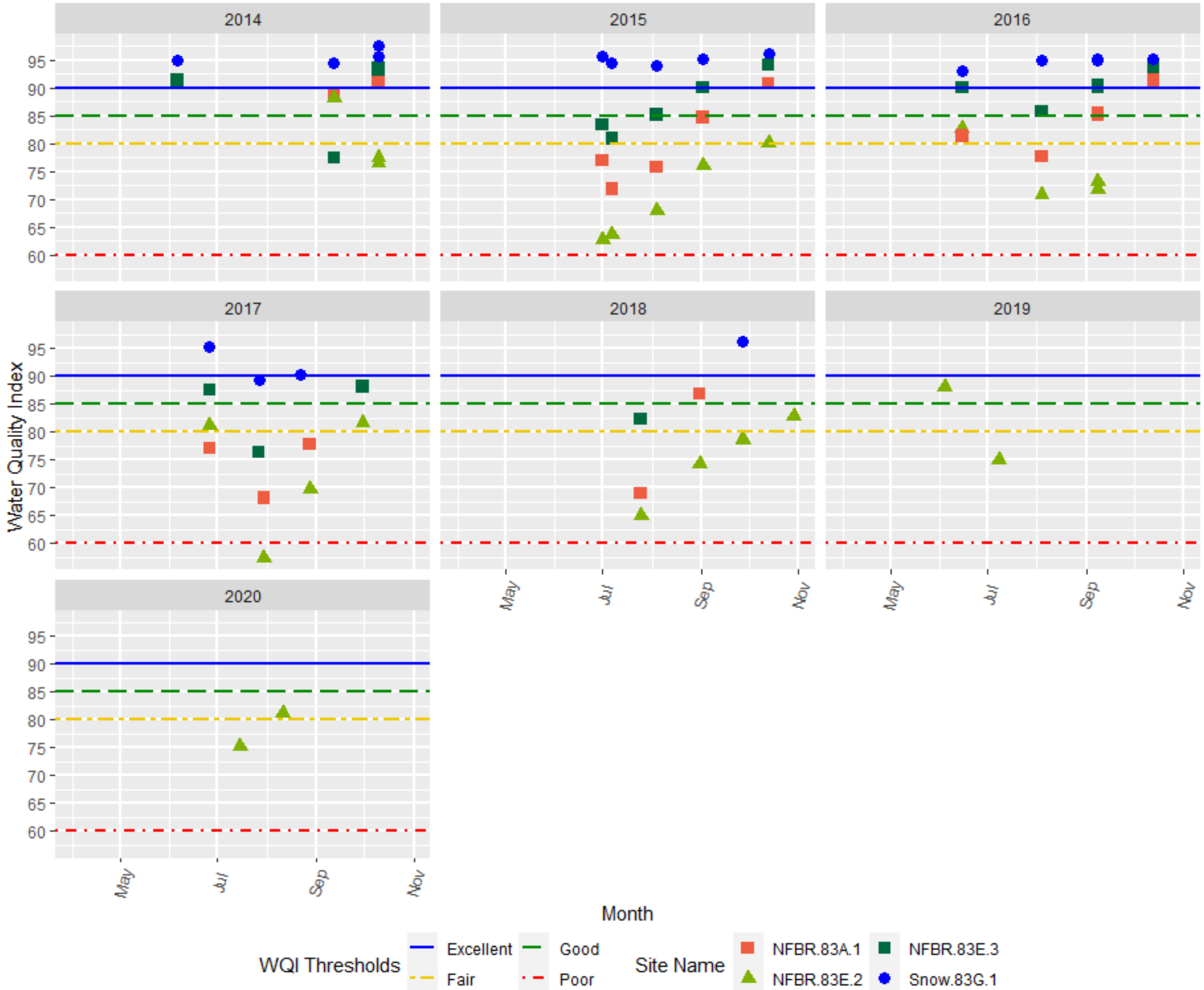


Table 53. Estimates for differences in mean WQI scores for the NFBR watershed monitoring sites from mean values over the 2014-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2014	-6.15	-2.56	1.19
2015	-4.93	-1.62	1.81
2016	-5.78	-2.77	0.35
2017	-7.69	-4.57	-1.33
2018	-5.14	-1.06	3.22
2019	-0.08	7.14	14.96
2020	-0.40	7.10	15.26
2022	-2.77	1.70	6.40
2023	-3.61	-0.55	2.62
2024	-5.79	-2.79	0.32

Discussion and future plans

Geography plays a large role in water quality in the NFBR, particularly changes from upper watershed streams into Whitney Valley and how impairments in the valley might impact locations downstream. Groundwater plays an important role in limiting some impairments by buffering temperature changes and mitigating some issues related to turbidity and eutrophication. In the upper part of the watershed on Snow Creek, groundwater inputs limited temperature impairments in the summer and buffered changes from climate change due to more consistent dry season flows. The large increases in mean and maximum temperatures moving downstream between Snow Creek and NFBR.83E.3 are likely the result of impairments to stream/groundwater connectivity. Similar patterns can be seen in Whitney Valley, where changes to stream temperature, pH, oxygen, and turbidity at the lower Valley site indicate some issues with incision and lower groundwater contributions. Water quality impairments are less severe in the lower watershed, but upstream impairments still heavily impact water quality in these reaches.

Monitoring in the NFBR generally found suitable conditions for redband trout in the spring, but high temperatures, turbidity, and pH wouldn't support year-round utilization in the lower watershed sites. A better understanding of thermal refugia in the upper parts of the watershed is needed if redband trout conservation efforts are to be successful moving forward. Increased monitoring in these reaches will also be useful in identifying the factors driving increasing stream temperatures between the upper watershed and Whitney Valley, as well as assessing seasonal patterns in thermal regimes related to groundwater sources.

There is also a need to understand the impact of beaver activity on water quality given the presence of beaver dam complexes in lower Whitney Valley and the Councils LTPBR efforts. As seen on lower Trout Creek, while beaver dams might reduce mean and maximum temperatures, particularly in the summer and fall, they can also contribute to low dissolved oxygen concentrations that might stress aquatic organisms. Preliminary data from monitoring in 2025 noted similar changes in temperature due to beaver activity, with mean temperatures 1.35 °C warmer and maximum temperatures 3.02 °C warmer upstream of the dam complexes than within them. This monitoring also showed that oxygen measurements were also 2.81 mg/L lower in the dam pools, although concentrations were still well above the 6.5 mg/L cool-water standard. Continued monitoring to identify if patterns in temperature and oxygen in the valley complex are similar to those seen in Trout Creek, or if differences in size and discharge result in different outcomes, will be important moving forward given the PBWC's focus on beaver restoration in the watershed.

Finally, while flows from Camp and Trout Creek are generally small compared to the NFBR, it's possible that restoration impacts might be noticeable during the later season. Continued monitoring at sites above and below Whitney Valley could detect these changes in water quality related to upstream restoration, although these responses would be small. If restoration does occur on private lands in Whitney Valley, continued monitoring will also be useful in detecting these water quality changes as well and if they address issues related to temperature and eutrophication identified during the 2022-24 monitoring efforts.

SFBR and Burnt Mainstem

Background

The Burnt River and South Fork Burnt River (SFBR) are the southernmost of the watersheds within the Powder Basin and drains 2,333 km^2 (913 mi^2). The headwaters of the SFBR begins in the Monument Rock and Table Rock wilderness areas, with elevations ranging between 7,770 ft to 4,300 ft above sea level. Forest cover in the upper section of the watershed is dominated by mixed conifer forest of Ponderosa Pine, Larch, Grand Fir, and Lodgepole Pine. Further downstream, the SFBR enters a broader valley with grazing and hay production as major land uses. Several irrigation ditches divert water from the SFBR within this reach, resulting in low flow during the summer at locations above Unity Reservoir. Large areas of the watershed were heavily impacted by the Rail Fire in August 2016, which burned 41,614 acres of primarily higher elevation forests (**NICC 2016**). The fire had some noticeable impacts on water quality at the nearby water quality monitoring sites and locations downstream, including several large landslides in 2017 and 2018. Fire impacts were also noticeable on macroinvertebrate communities in the SFBR, with 2018 samples on the SFBR collected by the PBWC identifying lower biodiversity than expected given the location and water quality (**PBWC 2019**).

Map 11. The SFBR and Burnt River watershed with 2022-2024 sample sites, major tributaries, and important features highlighted.

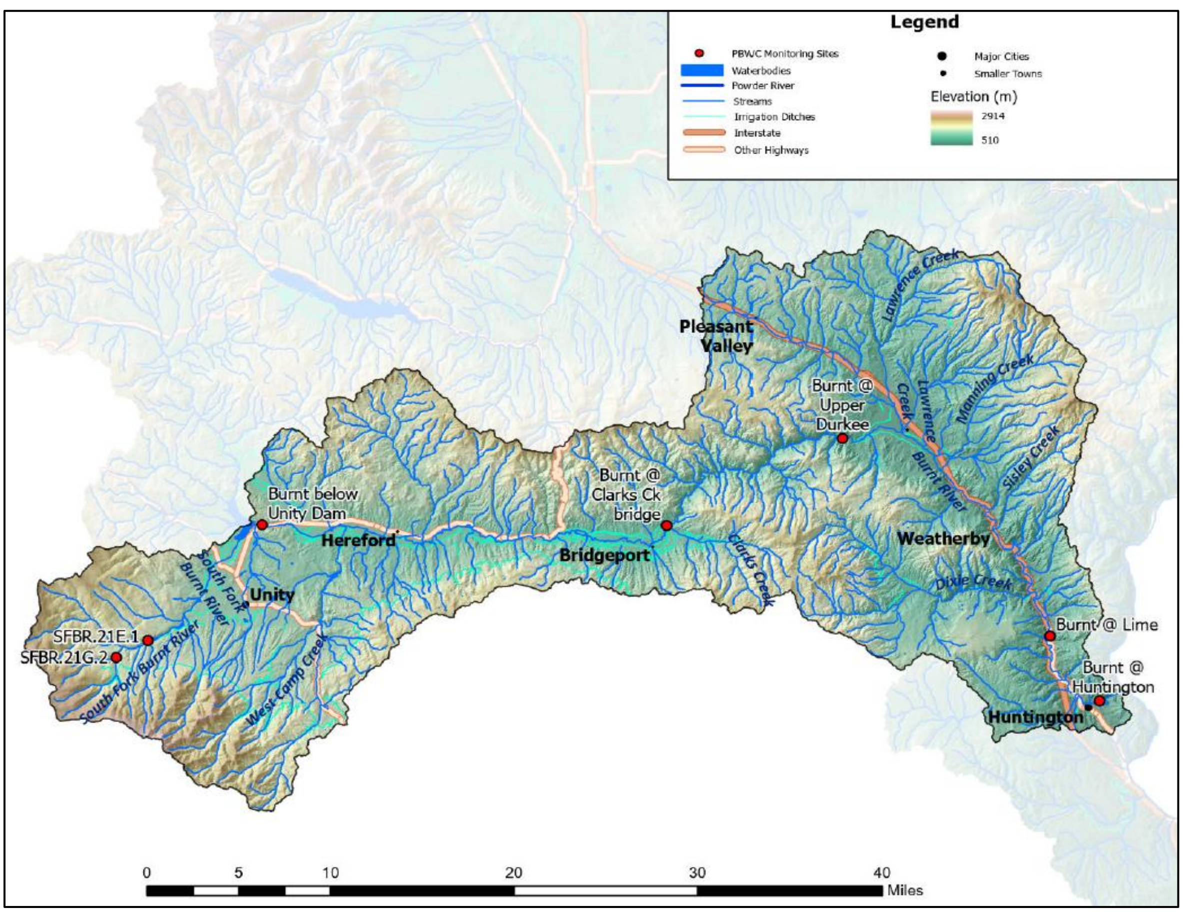
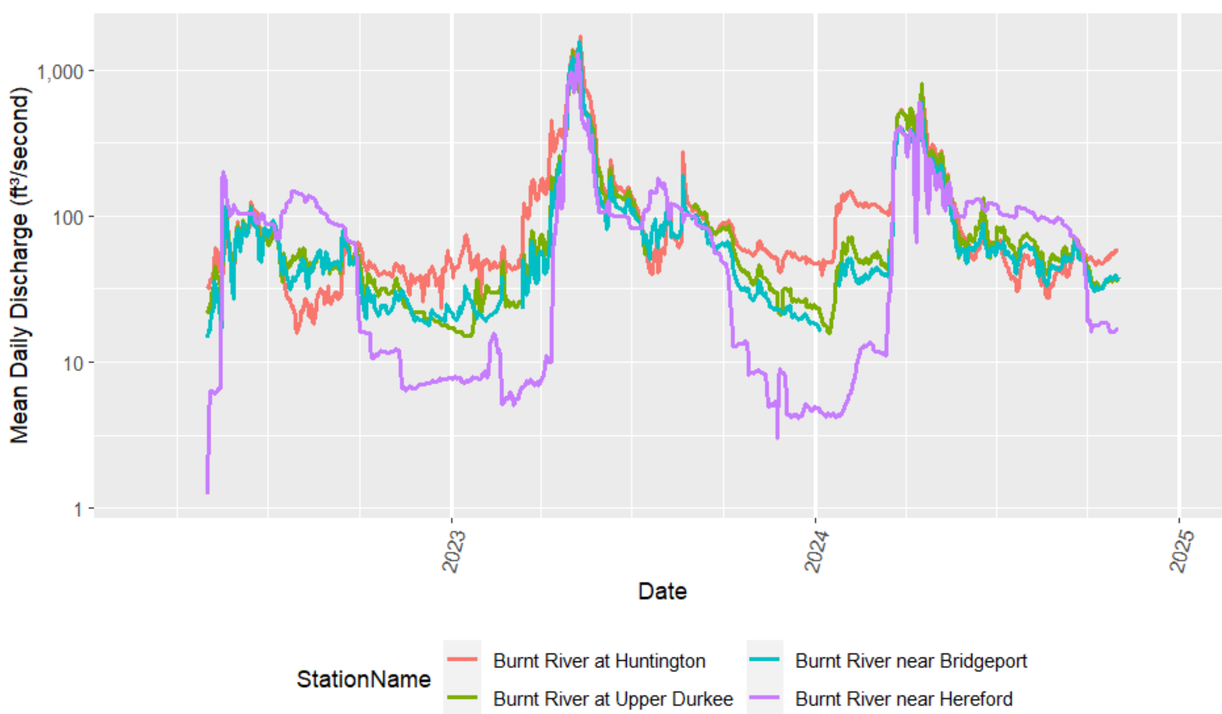


Table 54. Site characteristics for monitoring locations on the SFBR and Burnt River, including ODFW Fish Habitat Type, elevation (m), upstream drainage area (km^2), modeled 1993-2011 NorWeST mean August stream temperature (°C), and established date.

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
Burnt @ Clarks Creek Bridge	34256	Cool	1027	1417.61	17.83	6/18/2013
Burnt @ Huntington	11494	Cool	639	2692.77	19.51	6/18/2013
Burnt @ Lime	37128	Cool	826	2597.48	19.13	6/18/2013
Burnt @ Upper Durkee	41464	Cool	850	1666.96	18.60	6/22/2022
Burnt below Unity Dam	36195	Cool	1151	772.55	16.90	6/5/2014
SFBR.21E.1	37123	Cool	1327	116.54	11.21	6/5/2014
SFBR.21G.2	37740	Cool	1397	23.63	10.00	6/5/2014

The SFBR, along with the NFBR, WFBR, MFBR and Job Creek, flows into Unity Reservoir. The reservoir, constructed in 1939 and operated by the Burnt River Irrigation District (BRID), creates the reservoir and provides 25,502 ac/ft of water for irrigation downstream (**USBR 1992**). Like other dam operations in the basin, the reservoir alters flow regimes compared to the upstream reaches, with higher flows during July and August and lower flows later in the season and through the winter to refill the reservoir. Downstream of Unity Reservoir and past Hereford, the Burnt River flows through relatively broad and productive agricultural areas for approximately 40 miles. Sinuosity and vegetation cover is generally high through this reach, but incision is noticeable in some areas. Significant mining activity is present on several tributaries to the Burnt within this reach as well, particularly on Pine Creek and Clarks Creek.

Chart 61. Measured discharge (in ft^3/second) for stream gaging stations on the Burnt River over the monitoring period (2022-2024).



Further downstream past Bridgeport, the Burnt River flows through the Burnt River Canyon. Vegetation through this reach is dominated by Juniper and Mountain Mahogany, with smaller stands of Grand Fir and Ponderosa Pine in more shaded reaches and Willow, Alder, and cottonwood along the riparian area. Downstream of the canyon, the Burnt River flows through the broad Durkee Valley, another productive agricultural area. Vegetation cover through this valley is variable, with some sections containing significant willow and cottonwood canopy cover, while others have more limited vegetation cover. Further downstream, the Burnt River flows alongside I 84 through another constrained canyon reach, before reaching Huntington and the confluence with the Snake River. Much of the area within and downstream of the Burnt River Canyon were heavily impacted by the Durkee Fire, which burned ~295,000 acres from July 17th to August 5th 2024 (**InciWeb 2024**). While significant areas of upland vegetation were heavily impacted by the fire, riparian vegetation is mostly intact, limiting the impact of the fire on water quality (Figure 27).

Figure 27. Burned hills within the Burnt River Canyon from the Durkee Fire, near the Burnt River at Upper Durkee site in August 2024.

Redband trout are the only native salmonid currently found in the SFBR and Burnt River watersheds. The highest Redband Trout utilization in the watershed is found in the SFBR and Camp Creek, a tributary adjacent to the SFBR that drains similar mountainous habitats. Use in these reaches is primarily for summer rearing and spawning but is also used for winter rearing and as a migratory corridor. Redband trout use in other reaches of the Burnt River are focused on winter utilization, particularly in the mainstem Burnt River and sites above Clarks Creek. Redband Trout use is also high in Manning, Pritchard, Lawrence, and Dixie Creeks, primarily for summer rearing, winter rearing, and spawning. Summer rearing is especially high at Dixie Creek and Sisley Creek, where temperatures are cooler than on the mainstem Burnt River. In the lower Burnt River mainstem, warmer stream temperatures limit Redband Trout use to winter rearing and migration (**NPCC 2004a**).

Monitoring efforts over the 2022-2024 period focused on grab sample and temperature monitoring at five sites on mainstem Burnt and two sites on the SFBR, with most of these sites established in 2013 and 2014. For monitoring of the SFBR, the sites were located upstream and downstream of a popular campground. Sites on the Burnt River were located to take advantage of public property and to isolate the impacts of Unity Dam, land use, and geomorphic features, with one site below Unity Dam, one site above the Burnt River Canyon, one site below Durkee Valley near an old Lime processing plant, and one near Huntington. In addition to these sites, one site near the downstream end of the Burnt River Canyon was added in 2022 as part of the most recent monitoring efforts.

In anticipation of TMDL (Total Maximum Daily Load) requirements by Oregon DEQ, the PBWC, in partnership with BRID, collected *E. coli* and phosphorus samples alongside the grab sample and temperature monitoring sites to establish baseline data and identify spatial and seasonal patterns. Focus was placed on collecting these samples once every two weeks to assess finer temporal scale patterns, although actual sampling efforts only achieved this during the 2024 sampling season. Given the recent addition of the *E. coli* and phosphorus sampling, along with the large area and limited time to collect them, several samples collected in 2022 did not meet both lab and PBWC standards, including late arriving samples in June. These issues only impacted the results of the laboratory *E. coli* analysis for these samples, with the phosphorus analysis still providing accurate results.

Several other events impacted sampling efforts in the SFBR and Burnt River monitoring sites. In 2022, the logger at Huntington was discovered missing losing important data on temperature patterns at this site for this year. Also, sampling on August 24th 2023 was conducted to assess the impact of an unseasonably large rain event on *E. coli* and phosphorus concentrations, but the lack of laboratory personnel upon arrival meant these samples were unable to be analyzed within the required timeframe. Finally, the Durkee fire heavily impacted sampling efforts in July and August of 2024. Samples collected afterwards were heavily impacted by conditions from the fire, including shifts in turbidity and temperature, although most of the impacts were more noticeable in 2025.

Grab Sample Monitoring

Dissolved oxygen concentrations were consistently above the 6.5 mg/L cool-water standard at all sites during the 2022-24 monitoring period. Average oxygen concentrations were highest at the SFBR sites and at Lime, with average concentrations at these sites above 9.8 mg/L. The lowest dissolved oxygen concentrations were found below Unity Dam, where average concentrations were below 9 mg/L. This site was also the only location where oxygen concentrations fell below the 8 mg/L cold-water standard, occurring three times in 2023 and 2024, all in August. Overall, oxygen concentrations were lowest in June and August and Highest in October. Changes in oxygen concentrations over the course of the season were similar between sites.

Chart 62. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for the SFBR and Burnt River monitoring sites over the 2022-2024 period. Cold-water standard (red line) highlighted.

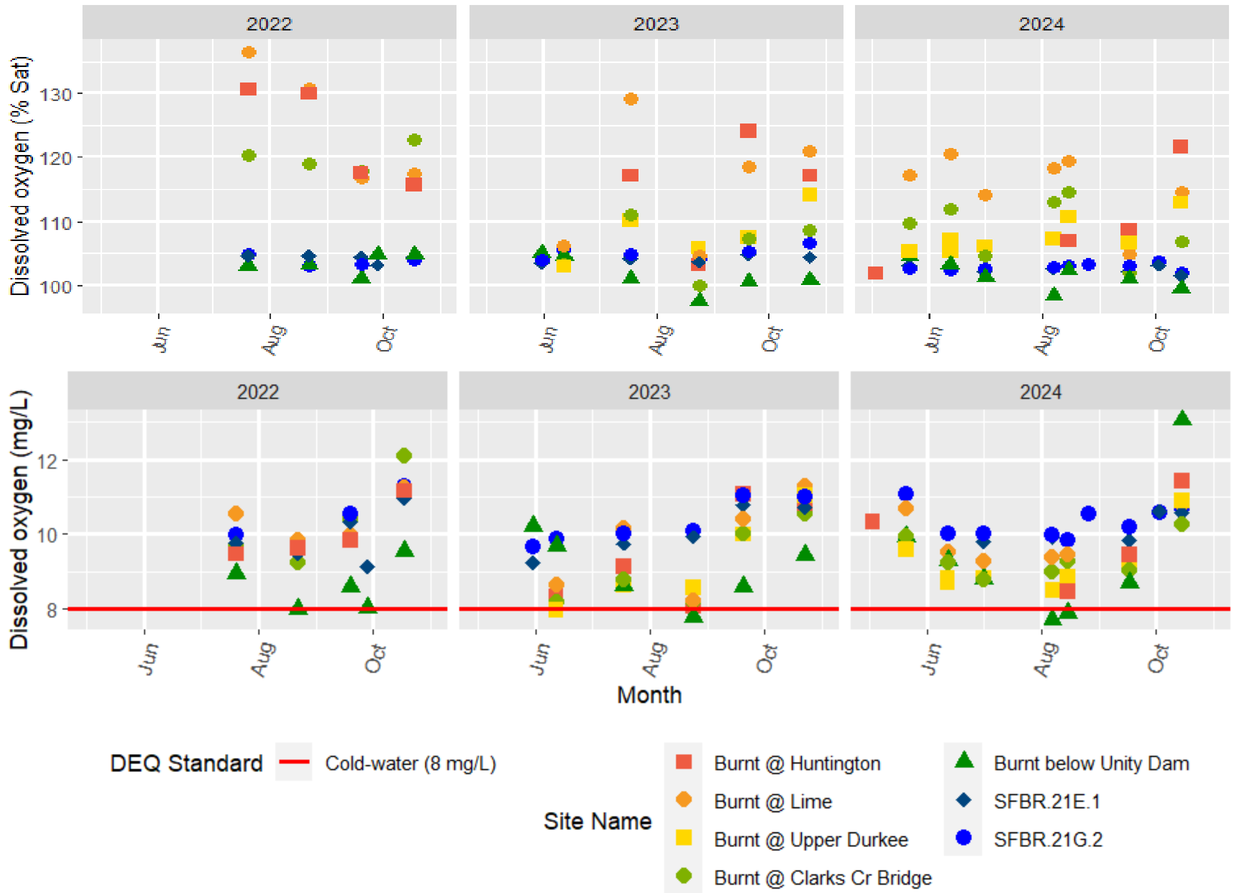


Table 55a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Huntington	9.07	9.65	10.23
Burnt @ Lime	9.33	9.88	10.43
Burnt @ Upper Durkee	8.72	9.31	9.90
Burnt @ Clarks Cr Bridge	8.98	9.53	10.08
Burnt below Unity Dam	8.38	8.92	9.46
SFBR.21E.1	9.33	9.91	10.50
SFBR.21G.2	9.75	10.30	10.85

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-1.07	-0.49	0.09
July	-0.73	-0.24	0.25
August	-1.07	-0.60	-0.12
September	-0.36	0.10	0.57
October	0.74	1.22	1.69

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.47	0.11	0.69
2023	-0.60	-0.08	0.45
2024	-0.58	-0.03	0.52

Oxygen saturation differed among sites and was heavily associated with basin size and the impacts of Unity dam reservoir. Saturation was generally consistent and close to 100% at the SFBR sites, likely related to cooler stream temperatures. Below Unity Dam, saturation was generally below 100%, mostly in late summer and fall. Saturation was above 115% at Lime, Clarks Cr Bridge, and Huntington, mostly in late summer and fall, and most commonly occurring at Lime, with 11/16 measurements having oxygen saturation above 115%. These high levels of oxygen saturation could indicate issues with eutrophication due to nutrient pollution from upstream sources. Overall, oxygen concentrations were lowest in June and August and Highest in October. Changes in oxygen concentrations over the course of the season were similar between sites. There were also no significant differences in oxygen between years for sites.

pH was within DEQ standards at 6 of the 7 sites and generally showed an increasing trend with upstream basin size. Some sites differed from this trend, with a noticeable decrease in pH seen between Unity Dam to Clarks Creek bridge along with a smaller decrease from Upper Durkee to Lime. Greater variability was seen in pH below Unity Dam, with a low pH seen in early summer, but large increase in later in the season, rising above the 9.0 upper threshold four times in August and September. These shifts in pH might indicate issues with eutrophication within the reservoir, particularly later in the season.

Chart 63. Observed pH measurements for the SFBR and Burnt River monitoring sites over the 2022-2024 period. Upper (dashed line) and lower (solid line) recommended pH standard highlighted.

Table 56a. Estimates of mean pH for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Huntington	8.30	8.55	8.80
Burnt @ Lime	8.19	8.44	8.69
Burnt @ Upper Durkee	8.32	8.59	8.86
Burnt @ Clarks Cr Bridge	7.77	8.02	8.27
Burnt below Unity Dam	8.11	8.36	8.60
SFBR.21E.1	7.58	7.85	8.11
SFBR.21G.2	7.59	7.84	8.09

b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.38	-0.13	0.12
July	-0.18	0.04	0.26
August	-0.13	0.08	0.29
September	-0.23	-0.01	0.20
October	-0.19	0.02	0.24

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.42	-0.17	0.08
2023	-0.17	0.06	0.29
2024	-0.12	0.11	0.34

Other than the patterns seen below Unity Dam, pH was generally consistent between months at the SFBR and Burnt River sites, with no significant monthly differences. While pH was lower in 2022 and higher in 2024, these differences were not significant from each other or from 2023.

Conductivity generally increased in a downstream direction, with higher conductivities associated with larger upstream drainage basin areas. A large increase in conductivity was observed between Unity Dam and Clarks Cr Bridge as well as between Upper Durkee and Lime, while decreases in conductivity were seen between Clarks Cr Bridge and Upper Durkee and between Lime. An increasing trend in conductivity was seen over the course of each season from July through October, primarily related to lower stream flows over the summer. Seasonal changes in conductivity were generally smaller at the SFBR sites. Seasonal changes were stronger at Upper Durkee and Clarks Cr Bridge, particularly between August and September of 2024, likely related to impacts of the Durkee Fire. Yearly differences were smaller than those seen between months, but still noticeable, with higher conductivities seen in 2022 and 2024 and lower conductivities in 2023. Higher precipitation in 2023 was the likely reason for lower values in 2023.

Chart 64. Observed conductivity measurements (in µS/cm) for the SFBR and Burnt River monitoring sites over the 2022-2024 period.

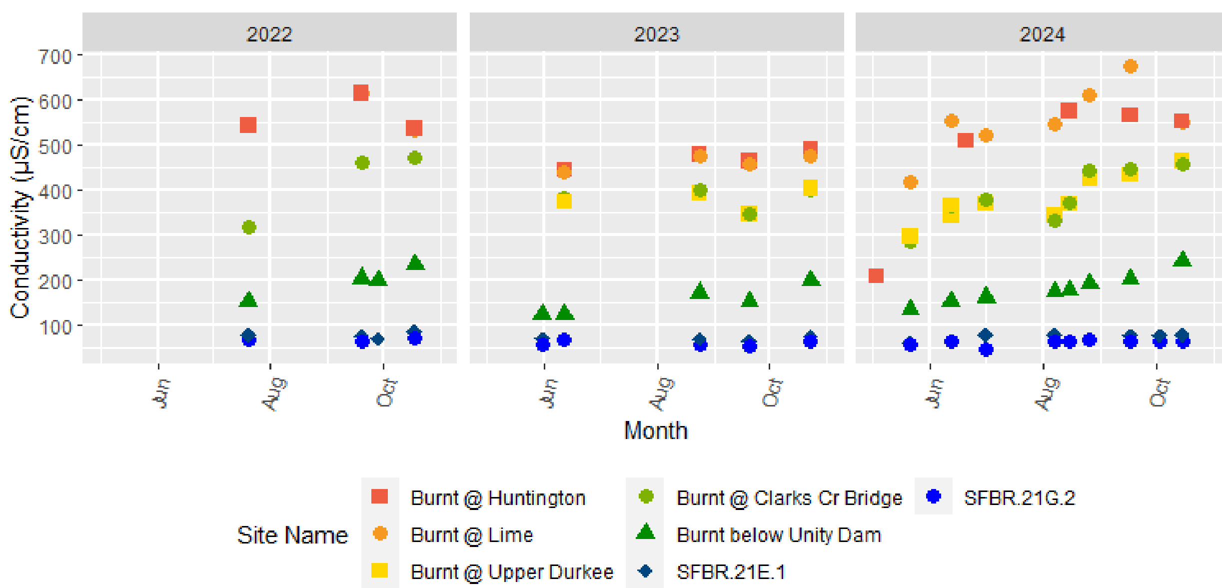


Table 57a. Estimates of mean conductivity (in µS/cm) for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Huntington	472.38	516.35	564.41
Burnt @ Lime	483.46	528.75	578.28
Burnt @ Upper Durkee	352.82	386.89	424.25
Burnt @ Clarks Cr Bridge	357.86	390.57	426.26
Burnt below Unity Dam	163.30	177.86	193.72
SFBR.21E.1	67.29	73.89	81.15
SFBR.21G.2	57.68	62.95	68.70

b. Mean monthly conductivity differences from mean values over the 2022-2024 period

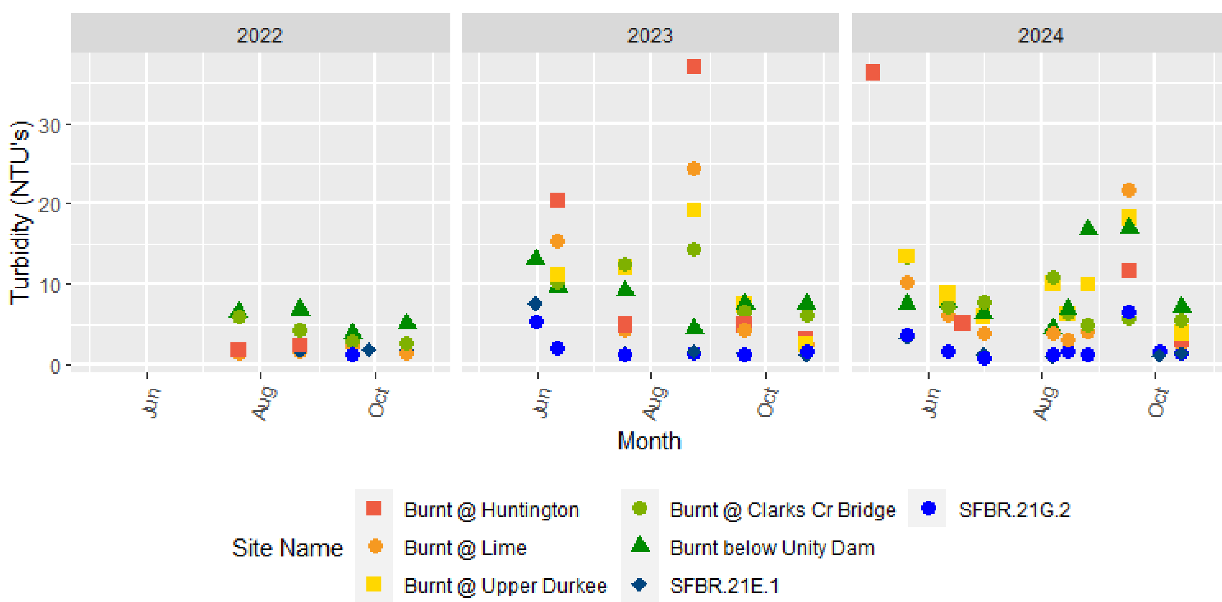
Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-29.67	-10.98	9.46
July	-37.10	-20.29	-2.01
August	-14.72	4.21	24.79
September	-9.77	7.02	25.08
October	1.85	20.04	39.66

c. Mean yearly conductivity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-6.52	14.33	37.12
2023	-35.32	-19.82	-3.09
2024	-12.11	5.49	24.51

Turbidity was above 10 NTU’s at all sites on the Burnt River, with the largest number of measurements above 10 NTU’s seen at Upper Durkee. Turbidity was also above 20 NTU’s at Huntington and Lime, particularly notable during the large rainfall event in August 2023. Overall, turbidity was highest at Below Unity Dam, Clarks Cr Bridge, and Upper Durkee, and lowest at the sites on the SFBR.

Chart 65. Observed turbidity measurements (in NTU’s) for the SFBR and Burnt River monitoring sites over the 2022-2024 period.



Occurrence of high turbidity was found throughout the year, with every month except July having measurements above 20 NTU’s. Every month also had turbidity measurements above 10 NTU’s, most commonly in May but also occurring frequently in the fall. Overall, mean turbidity was highest in May and lowest in October and July. Turbidity was generally lower in 2022 and higher in 2023 and 2024, likely associated with higher spring and summer precipitation and the large turbidity pulse in August 2023. Variability in turbidity was also higher in 2023 and 2024, particularly at the downstream sites.

Table 58a. Estimates of mean turbidity (in NTU's) for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Huntington	3.23	5.31	8.72
Burnt @ Lime	2.58	4.08	6.45
Burnt @ Upper Durkee	4.25	6.94	11.34
Burnt @ Clarks Cr Bridge	3.96	6.26	9.90
Burnt below Unity Dam	4.47	7.07	11.17
SFBR.21E.1	1.08	1.75	2.84
SFBR.21G.2	0.97	1.53	2.42

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-1.02	0.99	4.30
July	-1.85	-0.57	1.43
August	-1.38	0.14	2.50
September	-0.97	0.81	3.57
October	-2.37	-1.37	0.20

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-2.40	-1.27	0.58
2023	-0.84	0.96	3.74
2024	-1.30	0.32	2.84

Stream Temperature monitoring

Temperatures in the SFBR and Burnt River were on average warmest in July and August and Coolest in October. Stream temperatures were above the cool water standard for Redband Tout at all sites on the Burnt River. Warm stream temperatures were especially notable at Clarks Cr Bridge, Upper Durkee, Lime, and Huntington, where 100% of days were above 20 °C in July. October was the only month with 0 days above 20 °C at these sites. September had fewer days above 20 °C than June at all sites except below Unity Dam but still had more than 20% of days where maximum temperatures were above 20 °, particularly at Lime and Huntington. Temperatures were below the 12 °C cold water standard at the SFBR sites in October. Maximum temperatures were also consistently below 12 °C in May at SFBR.21G.2, but were above 12 °C at both sites all of July and most of June and August.

A later peak in temperatures was particularly notable at the site below Unity Dam, with higher average temperatures and days above 20 °C in August compared to July. This pattern is probably related to reservoir levels and conditions. The site also had lower daily temperature ranges than any other site, as well as a lower difference between mean and maximum temperatures, less than 1 °C, compared to average of 2.86 °C for the other SFBR and Burnt Sites.

Chart 66. Temperature profiles (in °C) for the SFBR and Burnt River monitoring sites over the 2022-2024 period, with lower Burnt River sites on the left , Upper Burnt River sites in the middle, and SFBR sites of the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cool water (solid line) standard in highlighted in red.

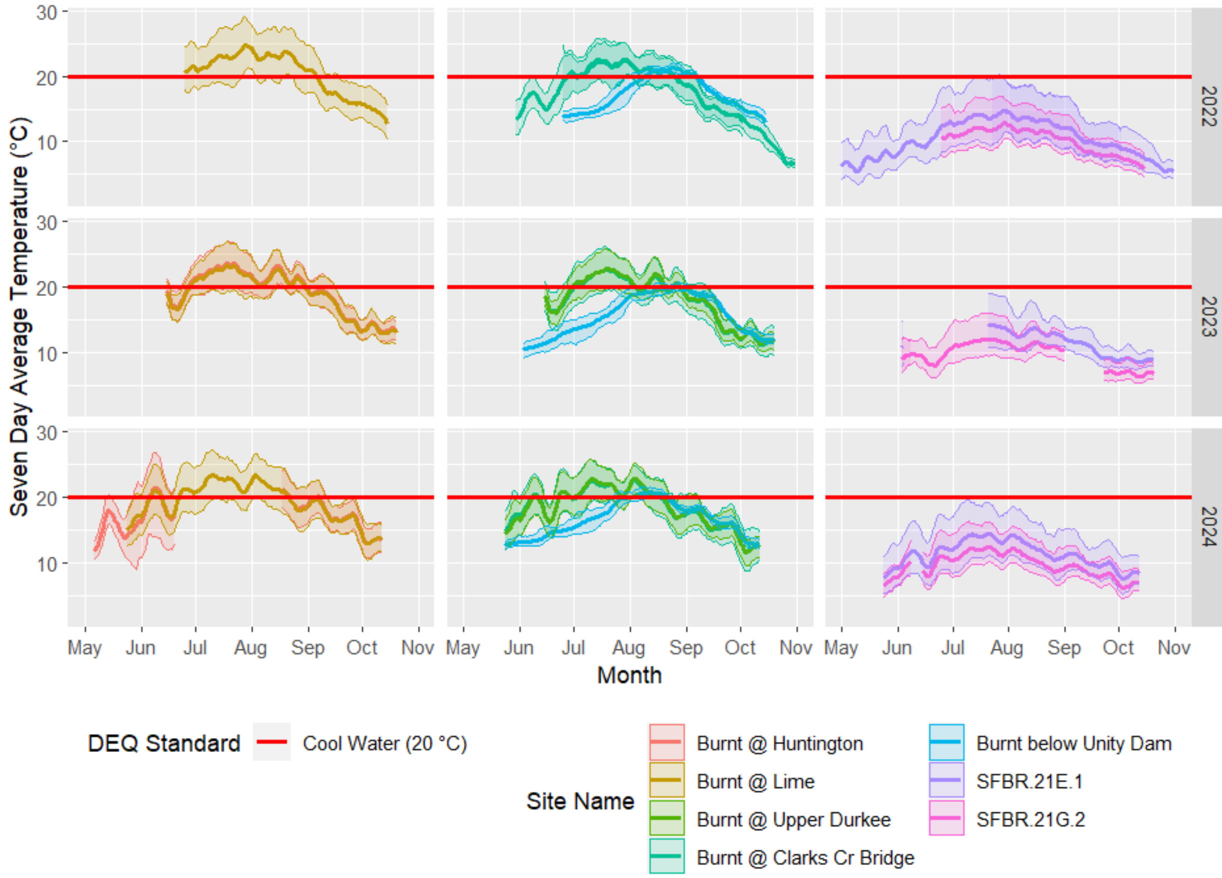


Table 59a. Estimates of Mean and Maximum Daily temperatures (in °C) for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
Burnt @ Huntington	17.31	18.53	19.75	20.07	21.35	22.63
Burnt @ Lime	17.89	18.87	19.85	20.85	21.88	22.90
Burnt @ Upper Durkee	16.55	17.74	18.93	18.66	19.91	21.16
Burnt @ Clarks Cr Bridge	16.40	17.38	18.35	18.91	19.93	20.96
Burnt below Unity Dam	14.98	15.96	16.93	15.86	16.89	17.92
SFBR.21E.1	10.20	11.17	12.15	13.76	14.78	15.81
SFBR.21G.2	8.48	9.46	10.44	11.42	12.44	13.47

b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Month	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
June	-1.30	-0.08	1.14	-1.05	0.23	1.51
July	1.50	2.76	4.02	2.19	3.51	4.84
August	1.21	2.43	3.65	1.22	2.51	3.79
September	-2.05	-0.82	0.40	-2.58	-1.30	-0.02
October	-5.50	-4.28	-3.06	-6.23	-4.95	-3.67

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Year	Mean Temperature Lower 95% Estimate	Mean Temperature Mean Estimate	Mean Temperature Upper 95% Estimate	Max Temperature Lower 95% Estimate	Max Temperature Mean Estimate	Max Temperature Upper 95% Estimate
2022	-0.95	0.27	1.49	-0.78	0.50	1.79
2023	-1.27	-0.20	0.87	-1.68	-0.56	0.56
2024	-1.17	-0.07	1.02	-1.09	0.06	1.20

A large increase in mean and maximum temperatures was seen between SFBR.21G.2 and SFBR.21E.1, where mean temperatures increased by 1.71 °C and maximum temperatures increased by 2.34 °C. There was also a notable increase in mean and maximum temperatures between SFBR.21E.1 and below Unity Dam, where mean temperatures increased by 4.78 °C. Maximum temperature change between these two sites was more muted, increasing by 2.11 °C. Maximum temperature increases were more notable in from Unity Dam to Clarks Cr Bridge, increasing by 3.04 °C. Changes in maximum and minimum temperatures were more muted between the lower sites. Slight decreases in mean and maximum temperatures were seen between Lime and Huntington, although these changes were likely biased by lack of data in 2022 at Huntington. Overall, differences in temperatures between years were small. Both mean and maximum temperatures were lower in 2023 than in 2022 or 2024 related to differences in flow along with cooler air temperatures.

Table 60. Estimated and Observed mean August stream temperatures for SFBR and Burnt River monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

Site	Estimated 2022-2024	Observed 2022-2024	NorWeST 1993-2011	NorWeST 2040	NorWeST 2080
Burnt @ Huntington	20.96	20.44	19.51	20.84	21.67
Burnt @ Lime	21.29	21.48	19.13	20.45	21.28
Burnt @ Upper Durkee	20.17	20.14	18.60	19.91	20.73
Burnt @ Clarks Cr Bridge	19.80	20.08	17.83	19.12	19.93
Burnt below Unity Dam	18.38	19.90	16.90	18.16	18.96
SFBR.21E.1	13.60	12.80	11.21	12.33	13.03
SFBR.21G.2	11.89	10.88	10.00	11.09	11.78

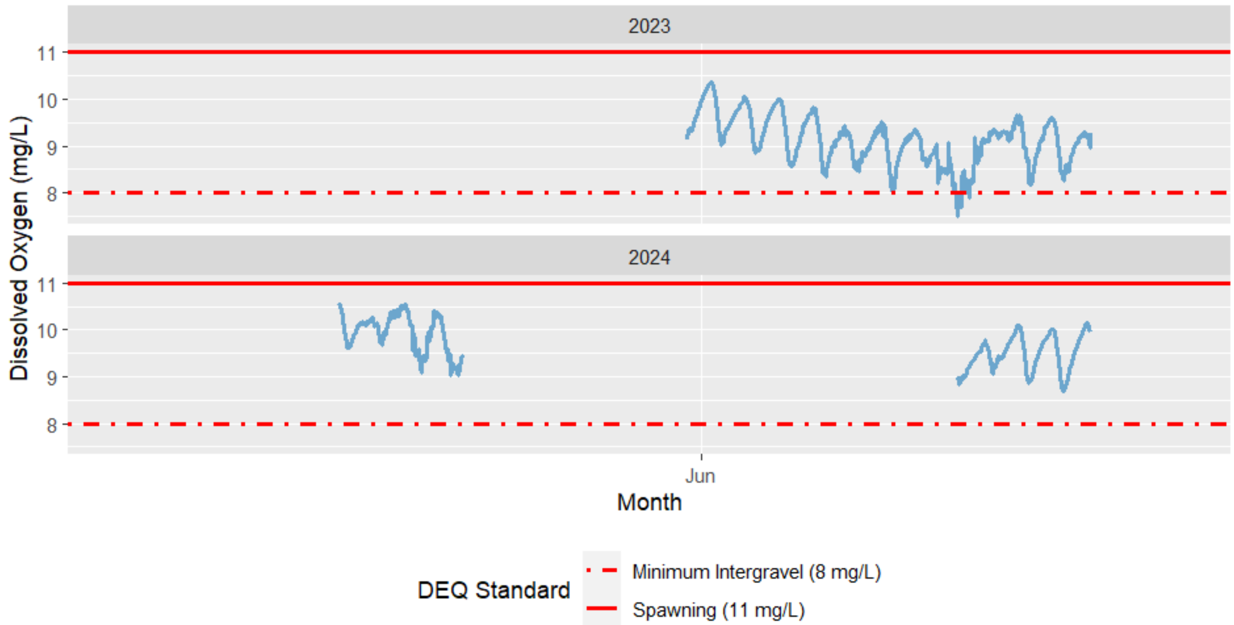
Comparisons to the NorWeST model found all sites showed an increase in mean August temperatures over the 1993-2011 estimates, with a mean increase of 1.79 °C for all sites. Increases were largest below Unity Dam, likely related to a later August peak in temperatures compared to the other sites. Increases in mean August temperatures were also large at Clarks Cr Bridge and Lime. Increases over the 1993-2011 estimates were lowest at SFBR.21G.2, indicating there might be stronger groundwater influences at this site. Increases were larger at SFBR.21E.1, possible due to reduction in forest cover due to the Rail Fire.

Dissolved Oxygen monitoring

Continuous dissolved oxygen loggers were installed at SFBR.21G.2 in spring of 2023 and 2024. Logger installation was heavily biased by burial in sediment both years, limiting the usability for the logger data collected at this site. There were also issues with oxygen accuracy in 2023, with oxygen concentrations 0.74 mg/L lower than measured oxygen concentrations for accuracy checks.

Measurements were more accurate in 2024, with logger measurements 0.25 mg/L lower than those from the multimeter during accuracy checks. The raw data showed that oxygen concentrations were consistently above the 8 mg/L spawning standard except for a period around June 8th 2023, where the logger was likely buried in sediment. 3.9 % of records in 2023 and 30.3% of records in 2024 were above 10 mg/L with the difference likely due to lower logger accuracy in 2023.

Chart 67. Dissolved oxygen profiles (in mg/L) from loggers installed at SFBR.21G.2 in Spring 2023 and 2024. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



The highest oxygen concentrations in both years were found at 8:00 in morning and lowest at 2:00 in the afternoon, while oxygen saturation was lowest at night and highest around noon. Average daily ranges in dissolved oxygen were similar between years, averaging 0.89 mg/L between daily minimum and maximum concentrations. Oxygen concentrations were strongly correlated to temperature (R^2 = .712) where concentrations decreasing by 0.18 mg/L for every 1 °C increase in temperatures, which was similar to those seen in other watersheds (Appendix D).

E. coli and Phosphorus monitoring

High *E. coli* concentrations were more common later in the summer and early fall and at Clarks Creek Bridge and Lime. Mean *E. coli* concentrations were above the 126 c0t/100 mL 90-day standard at Clarks Creek Bridge and above 100 cfu/100mL at Lime. Six samples collected at Clarks Cr Bridge and Lime were above 406 cfu/100mL single sample standard at Clarks Creek Bridge, all from August through October. One unusually large measurement (>86,000 cfu/100mL) was recorded at Lime for the October 2023 sampling event. These values were determined by the laboratory very likely to be accurate and could be related to sediment disturbance from 2023 construction activity on the nearby rail bridge.

Lower concentrations of *E. coli* were seen at SFBR.21G.2, below Unity Dam, and at Upper Durkee. A notable decrease in *E. coli* concentration was observed between Clarks Cr Bridge and Upper

Durkee, with *E. coli* concentrations 85.3% lower at Upper Durkee on average. Concentrations were consistently below both single sample and 90-day average standards at SFBR and below Unity Dam, and less common at Upper Durkee. Differences in *E. coli* concentration between years were small, with mean concentrations higher in 2023 and 2024, likely related to higher flows or erosion from the Durkee Fire.

Chart 68. E. coli concentrations (in colonies per 100mL) for the SFBR and Burnt River monitoring sites over the 2022-2204 period. 90-day average standard (126 cfu/100mL, solid line) and single sample standard (406 cfu/100mL, dashed line) highlighted. Log transformation used for E. coli measurements.

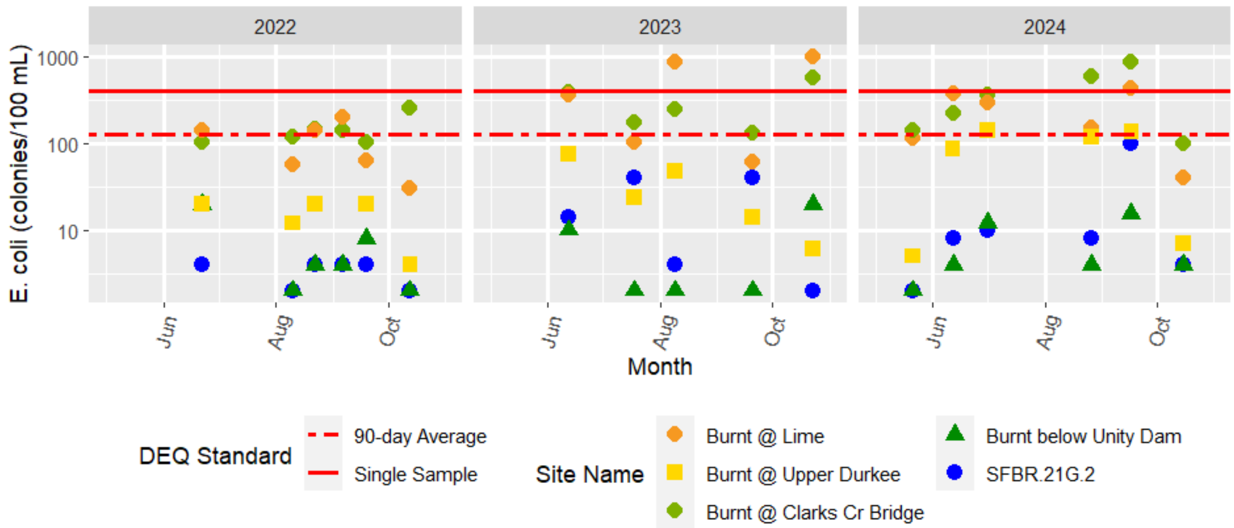


Table 61a. Estimates of mean E. coli concentrations (in colonies/100mL) for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Lime	22.76	103.68	472.36
Burnt @ Upper Durkee	4.10	20.77	105.32
Burnt @ Clarks Cr Bridge	31.03	141.36	644.05
Burnt below Unity Dam	0.99	4.50	20.49
SFBR.21G.2	1.45	6.59	30.05

b. Mean monthly E. coli differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-27.41	-19.69	11.57
July	-18.80	20.83	201.41
August	-20.62	2.96	86.17
September	-25.77	-17.91	4.78
October	-14.95	13.80	97.71

c. Mean yearly E. coli differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-19.05	-7.37	26.31
2023	-19.23	2.14	99.49
2024	-18.12	5.23	105.19

Variation within sites was a strong predictor of *E. coli* concentration. Overall lower oxygen concentrations were associated with higher *E. coli* concentrations, although the correlation was weak and somewhat small, with a 1.50 cfu/100mL increase when oxygen concentrations decreased from 8 mg/L to 7 mg/L and a 1.4 cfu/100mL increase when oxygen concentrations decreased from 9 mg/L to 8 mg/L. Higher turbidity was more strongly associated with higher *E. coli* concentrations with a 58 cfu/100mL increase in *E. coli* when turbidity increased from 0 NTU to 10 NTU's and a 159 cfu/100mL increase when turbidity increased from 10 NTU's to 20 NTU's. Most of the variation in the data was explained by correlations with dissolved oxygen concentration and turbidity and by differences between sites.

Chart 69. Phosphorus concentrations (in mg/L) for the SFBR and Burnt River monitoring sites over the 2022-2204 monitoring period.

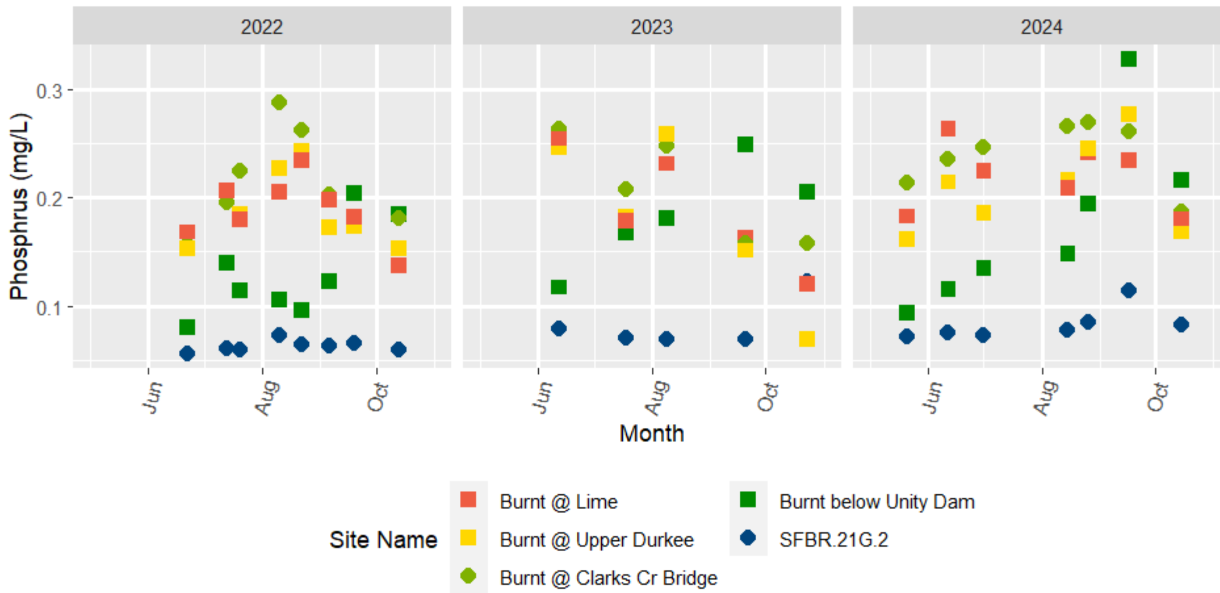


Table 62a. Estimates of mean Phosphorus concentrations (in mg/L) for SFBR and Burnt River sites over the 2022-2024 period.

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Lime	0.143	0.195	0.247
Burnt @ Upper Durkee	0.125	0.181	0.236
Burnt @ Clarks Cr Bridge	0.164	0.216	0.268
Burnt below Unity Dam	0.110	0.162	0.214
SFBR.21G.2	0.020	0.071	0.123

b. Mean monthly phosphorus differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.090	-0.042	0.006
July	-0.039	0.013	0.065
August	-0.048	-0.005	0.038
September	-0.025	0.011	0.047
October	-0.014	0.023	0.060

c. Mean yearly phosphorus differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.052	-0.016	0.021
2023	-0.058	-0.006	0.046
2024	-0.028	0.022	0.071

Phosphorus concentrations generally increased in a downstream direction, with mean concentrations highest at Clarks Cr Bridge and the sites downstream and lowest at SFBR.21G.2. Phosphorus concentrations also generally increased throughout the season, with a notable increase seen below Unity Dam, where average concentrations increased by 0.09 mg/L or 63% between August and September. There were some decreasing trends throughout the season at Upper Durkee and Lime, most notably in 2023. Decreases in phosphorus concentrations between Clarks Cr Bridge and Upper Durkee were smaller than for *E. coli*, with phosphorus concentration at Upper Durkee 12% lower than upstream. Phosphorus concentrations were higher in 2024 than 2022 or 2023.

Like *E. coli*, differences between sites were the most important factor in explaining phosphorus concentrations, with correlations between maximum temperature, conductivity, and turbidity also explaining a large amount of variation. Increasing maximum temperatures were associated with higher phosphorus concentrations, with a 0.023 mg/L increase in phosphorus when maximum temperatures increased from 15 °C to 25 °. Increasing conductivity was also associated with higher phosphorus concentrations, increasing by 0.015 mg/L when conductivity increased from 100 µS/cm to 250 µS/cm and increasing by 0.026 mg/L when conductivity increased from 250 µS/cm to 500 µS/cm. Turbidity was best correlated to phosphorus, with concentrations increasing by 0.017 mg/L when turbidity increased from 0 NTU's to 10 NTU's and increasing by 0.022 mg/L when turbidity increased from 10 NTU's to 20 NTU's. While the effect sizes for individual parameters were smaller than those seen for turbidity and *E. coli*, in combination they could result in larger changes in phosphorus concentrations than in isolation. The strong correlation with turbidity is also an indication that erosion might be a factor in higher concentrations for both *E. coli* and phosphorus concentrations at the sites.

Water Quality Index

Overall water quality decreased with increasing drainage basin size and was highest at SFBR.21G.2 and lowest at Huntington. Mean water quality was highest at the SFBR sites, with overall WQI scores having consistent excellent values. Water quality was much more variable below Unity Dam, with WQI scores showing excellent water quality until later into season followed by large decrease in WQI scores from late summer into the fall. Water quality was also variable at the lower sites and contained several months where WQI scores indicated poor water quality. Water quality at these sites was lowest in July and August and highest in October. Stronger improvements in water quality from summer into fall were seen at Clarks Cr Bridge than the other Burnt River sites, but all sites showed better water quality in October than the summer.

Table 63a. Estimates of mean WQI scores (from 10-100) for the SFBR and Burnt River monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Burnt @ Huntington	72.91	76.94	81.21
Burnt @ Lime	74.81	78.30	81.94
Burnt @ Upper Durkee	75.34	78.87	82.57
Burnt @ Clarks Cr Bridge	81.21	84.75	88.45
Burnt below Unity Dam	83.84	87.50	91.33
SFBR.21E.1	91.59	95.83	100.00
SFBR.21G.2	93.40	97.47	100.00

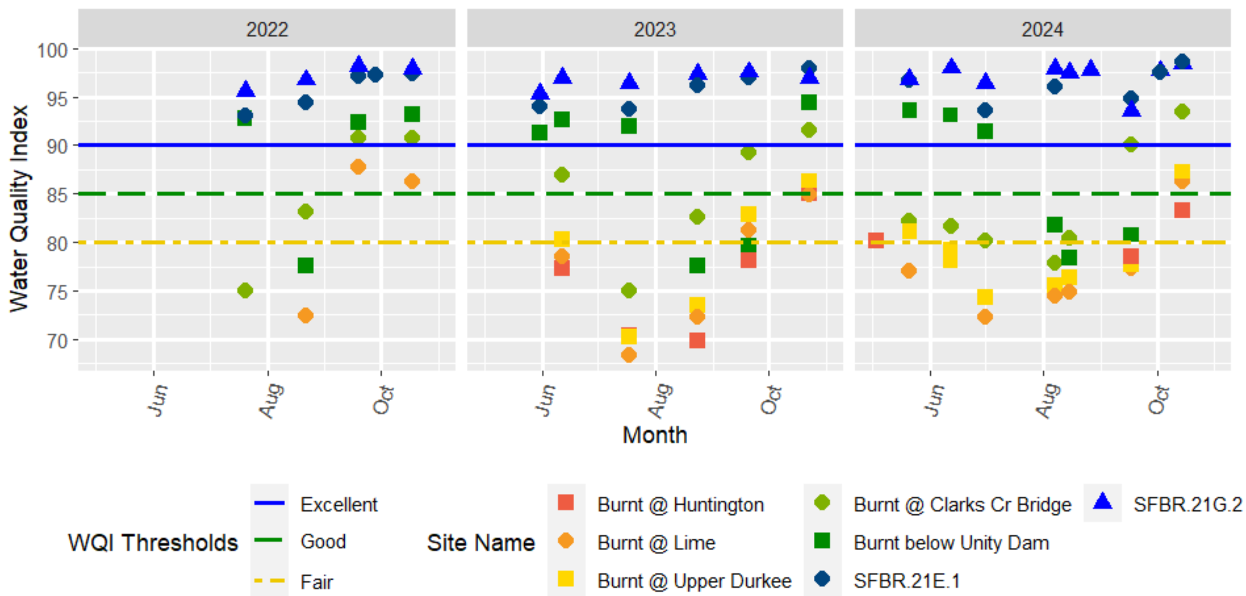
b. Mean monthly WQI differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-3.22	1.33	6.13
July	-7.52	-3.60	0.52
August	-7.62	-3.79	0.23
September	-2.79	1.11	5.20
October	0.82	4.95	9.28

c. Mean yearly WQI differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-3.71	0.82	5.59
2023	-4.24	-0.52	3.37
2024	-4.29	-0.30	3.89

Chart 70. WQI scores the SFBR and Burnt River monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WI) and Poor (80-60 WQI) thresholds highlighted.

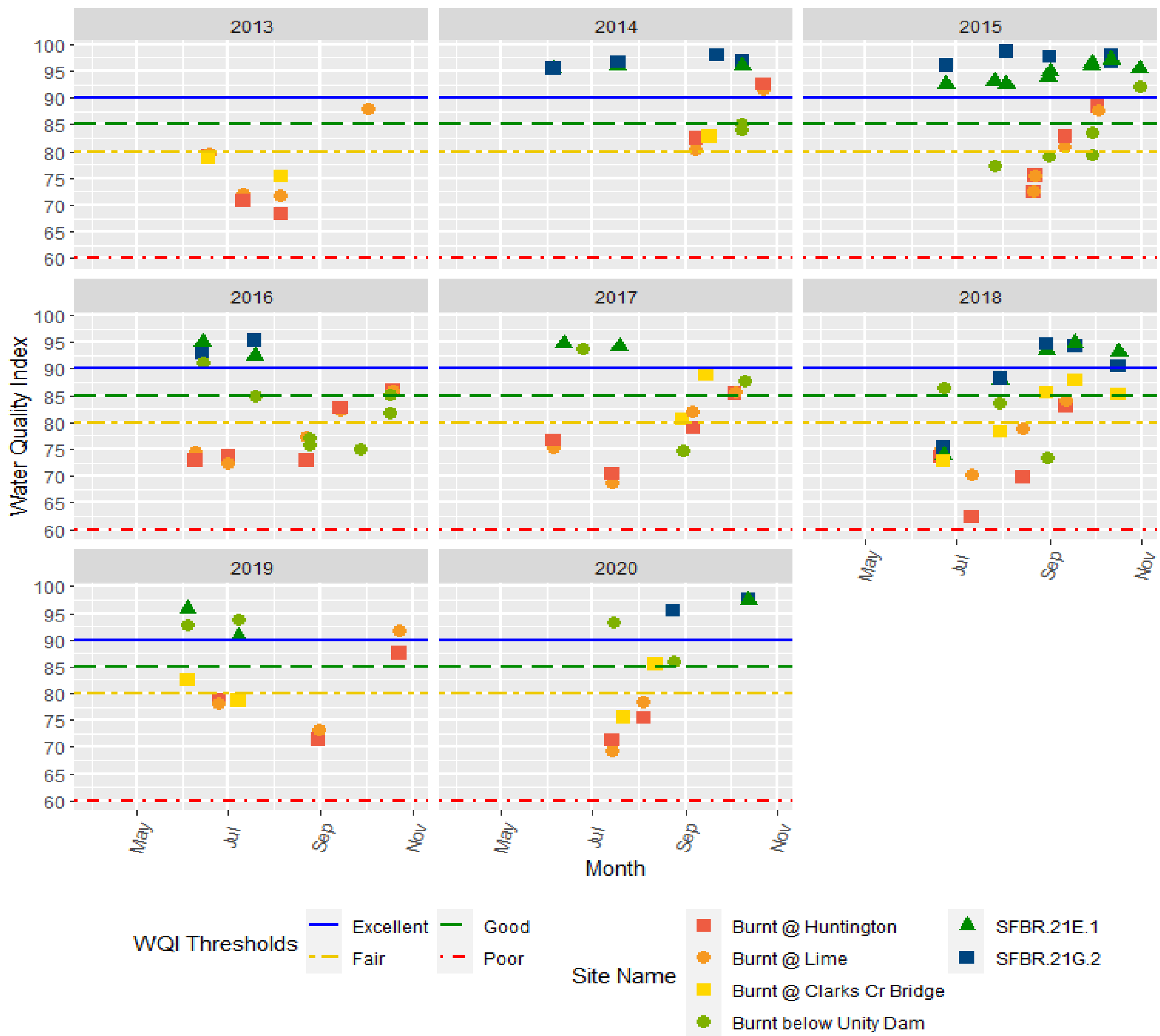


Factors resulting in low WQI scores differed between sites and time of year. Most WQI subindex values were generally similar between the SFBR sites and were all above 94 for the summer months. Temperature subindex scores differed somewhat between the sites, with 98.4 for SFBR.21G.2 and 94.9 for SFBR.21E.1. For the site below Unity Dam, low water quality in August was primarily related to temperature, with subindex score of 75.4, and pH, with subindex score of 64.9. Low WQI scores in September were more related to high pH, with sub-index score of 72.3, and to turbidity, with a subindex score of 78.66. Temperature and oxygen were generally good at this site in September, with subindex values for these parameters above 90.

Temperature was a more important factor for low WQI score at Clarks Cr Bridge, with subindex value of 58.9. Turbidity also lowered WQI scores at this site, with a subindex score79.0, whereas pH and oxygen subindex values were above 85. Temperature was also an important factor lowering WQI scores at the lower watershed sites, temperature was the largest contributor to low WQI scores in the summer, with an average subindex score of 48.3 at Lime and Huntington during July and August. Oxygen and pH were also important in lowering WQI, with subindex scores of 79.6 and

75.4, respectively, for these months. Oxygen was less important for the low WQI scores at Upper Durkee in the summer, with a subindex score of 90.9, compared to 60.9 for temperature, 76.1 for turbidity, and 68.1 for pH. Improvement in turbidity and temperature subindex scores were a major factor in the better WQI scores in fall, increasing to 82.6 for temperature and 87.2 for Turbidity.

Chart 71. WQI scores at the SFBR and Burnt River monitoring sites over the 2014-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WI) and Poor (80-60 WQI) thresholds highlighted.



Patterns in WQI scores between sites were mostly similar between the 2022-2024 and 2013-2024 periods, except for the SFBR sites, where WQI scores were higher over the 2013-2024 period than for 2022-24. Patterns in WQI scores between the monitoring periods was also similar between months. Overall WQI scores were lower in 2013, 2016, 2018 and higher in 2014, 2019, and 2022, although these trends were likely biased by the selection of sites, especially in 2013 and 2014.

Table 64. Estimates for differences in mean WQI scores for SFBR and Burnt River monitoring site from mean values over the 2013-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2013	-5.68	-2.29	1.23
2014	-2.01	1.17	4.48
2015	-3.20	-0.35	2.60
2016	-3.85	-1.14	1.67
2017	-3.09	-0.13	2.93
2018	-6.15	-3.57	-0.90
2019	-0.75	2.44	5.76
2020	-2.82	0.76	4.50
2022	-1.44	1.84	5.26
2023	-2.18	0.47	3.21
2024	-1.99	0.80	3.69

Discussion and future plans

SFBR and Burnt River water quality varied between the sites and over the course of the year. Overall water quality was best at the SFBR sites, where larger groundwater contributions and snow melt hydrology reduced temperature and pH impacts on water quality. Changes in water quality were more noticeable below Unity Dam, particularly between the summer and fall. pH and temperature increases related to reservoir drawdown resulted in very high pH that could indicate eutrophic conditions within the reservoir. Lower water levels in the reservoir during late summer and fall might also exacerbate temperature issues along with turbidity. Total phosphorus concentration also increased in the reservoir over this time period, adding further support for eutrophic conditions during this period.

High temperatures and early season turbidity issues were more noticeable water quality problems at Clarks Cr Bridge. Lower turbidity and cooler temperatures resulted in improvement in water quality in the fall, but overall water quality was still significantly lower than below Unity Dam. *E. coli* issues are more consistently an issue at Clarks Cr Bridge than other sites, along with issues related to phosphorus. Higher turbidity values at this site are likely driven by stream bank erosion in the upstream valley. Temperature is also an issue at this site and locations downstream, with temperatures consistently above 20 °C during July, August, and September. Further downstream, the Burnt River Canyon has several large impacts on water quality, particularly on *E. coli*, with concentrations decreasing significantly compared to Clarks Cr Bridge. Despite positive changes in *E. coli* concentrations through this reach, phosphorus concentrations were still elevated at Upper Durkee, alongside issues related to temperature and high pH measurements.

Water quality issues were also present in the Burnt River below Durkee Valley. While less frequent than at Clarks Cr Bridge, *E. coli* concentrations were still above recommended DEQ standards multiple times, primarily during the summer and fall. Issues with warm temperatures and high pH were also major factors impacting water quality. Turbidity was less of a problem but still impacted water quality during noticeable spikes related to storms in 2023 and fire in 2024.

There is a need to identify sources of *E. coli* and phosphorus at sites and over time to better develop and implement strategies to address these water quality impairments. Grab samples taken alongside bacteria and nutrient samples have been useful in identifying some probable sources, like erosion, but are still limited in assessing underlying causes. For *E. coli*, multiple sources could be potential reasons for high levels of bacteria, particularly for the downstream sites, and include septic system leakage, livestock, and wildlife. eDNA monitoring is currently being undertaken at the Burnt River and other locations to help understand potential sources of *E. coli* and how they compare to concentrations and change over time. For phosphorus, high concentrations might be related to high background levels caused by volcanic rock weathering but could be exacerbated by erosion and other anthropogenic impacts like septic systems, fertilizers, and manure. High levels of turbidity were more strongly associated with high concentrations of bacteria and nutrients at the Burnt River sites, likely related to erosion, but there is still a need to identify the sources of this erosion if efforts to reduce concentrations are to be successful.

Related to *E. coli* and phosphorus is understanding both broader and finer scale concentrations in phosphorus. There is a need to understand the changes in *E. coli* concentrations through the Burnt River Canyon. The decrease in concentrations through the Canyon indicates that *E. coli* might be less mobile than expected. Identifying the change in concentrations alongside eDNA sampling might better elucidate the processes determining *E. coli* and to better manage them. Understanding the status and processes above and within the reservoir are also important in identifying where phosphorus is coming from and its impacts on the reservoir. The PBWC is currently taking *E. coli* and phosphorus samples at two sites on the SFBR and MFBR directly above the reservoir, in conjunction with samples below Unity Dam. Understanding sources and processes should be extremely useful in assisting dam operations to limit water quality impacts, particularly if they are related to eutrophic conditions.

Pine Basin

Background

The Pine Creek watershed is part of the Brownlee sub-basin and drains 783 km^2 (302 mi^2) of the Northeastern-most part of Powder Basin. The watershed is notable for having several large tributaries adjacent to one another before converging further downstream. Pine Creek, the largest of these tributaries, has its headwaters in the highlands of the Wallowa Mountains above Cornucopia and drains several large cirque lakes, the largest being Pine Lake. The other tributary headwaters, including Clear Creek, East Pine Creek, and North Pine Creek, drain the lower elevation foothills and basalt plateaus to the east of the highlands.

Map 12. The Pine Creek watershed with 2022-2024 sample sites, major tributaries, and important features highlighted.

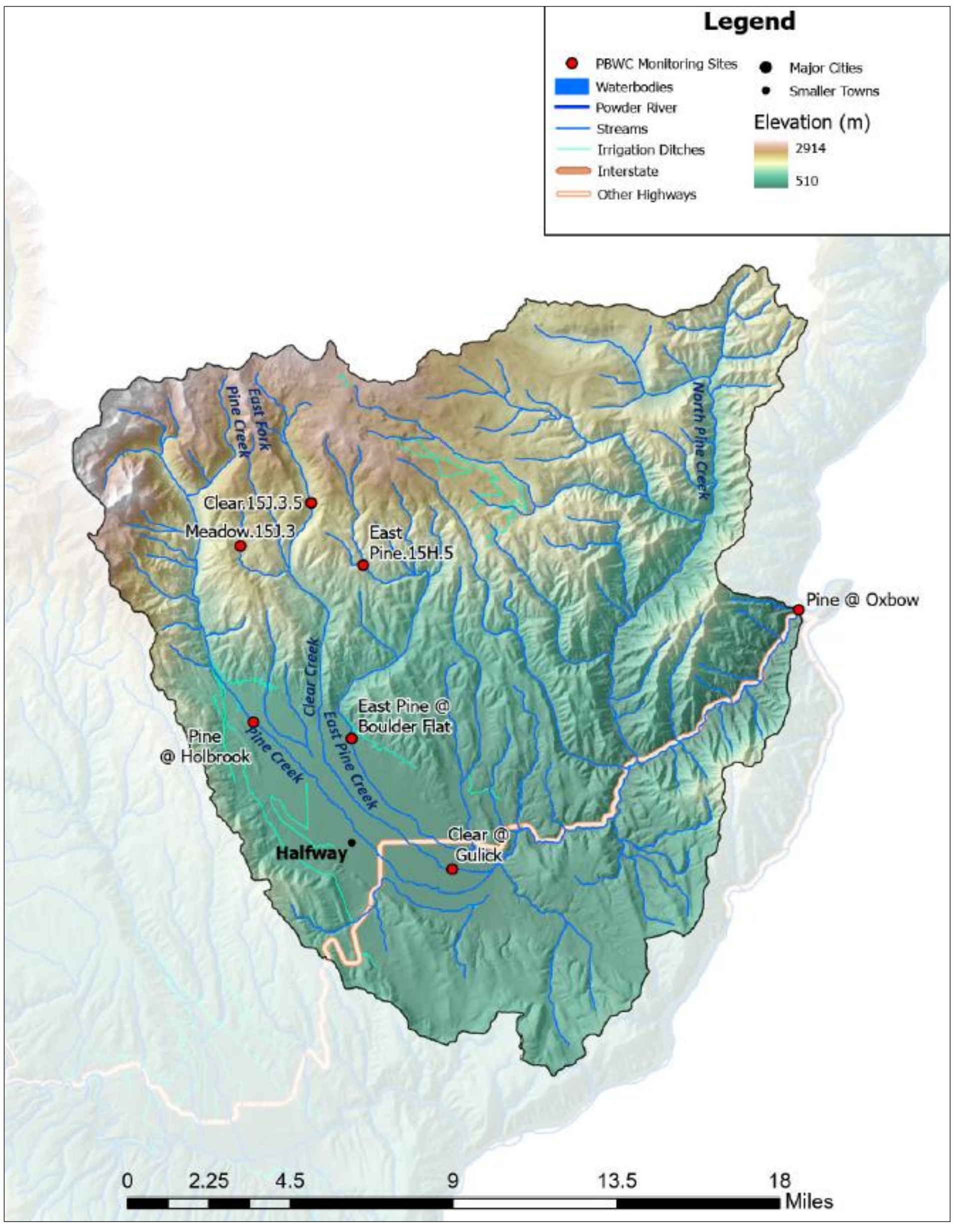


Table 65. Site characteristics for monitoring locations in the Pine Creek watershed, including ODFW Fish Habitat Type, Elevation (m) upstream drainage area (km^2), Modeled 1993-2011 NorWeST mean August stream temperature (°C), and established date.

Site Name	Site ID	Habitat Type	Elevation	Drainage Area	NorWeST Temp	Established Date
Pine @ Oxbow	36382	Cool	522	718.43	19.37	6/12/2013
Clear @ Gulick	37428	Cool	755	67.45	16.73	6/12/2013
Pine @ Holbrook	37337	Cool	970	141.47	14.69	6/12/2013
East Pine @ Boulder Flat	41790	Cool	882	72.52	15.87	6/26/2023
Clear.15J.3.5	41462	Cold	1632	23.47	10.65	9/9/2022
East Pine.15H.5	41463	Cold	1381	15.12	11.64	9/9/2023
Meadow.15J.3	37735	Cold	1640	5.05	9.64	5/22/2014

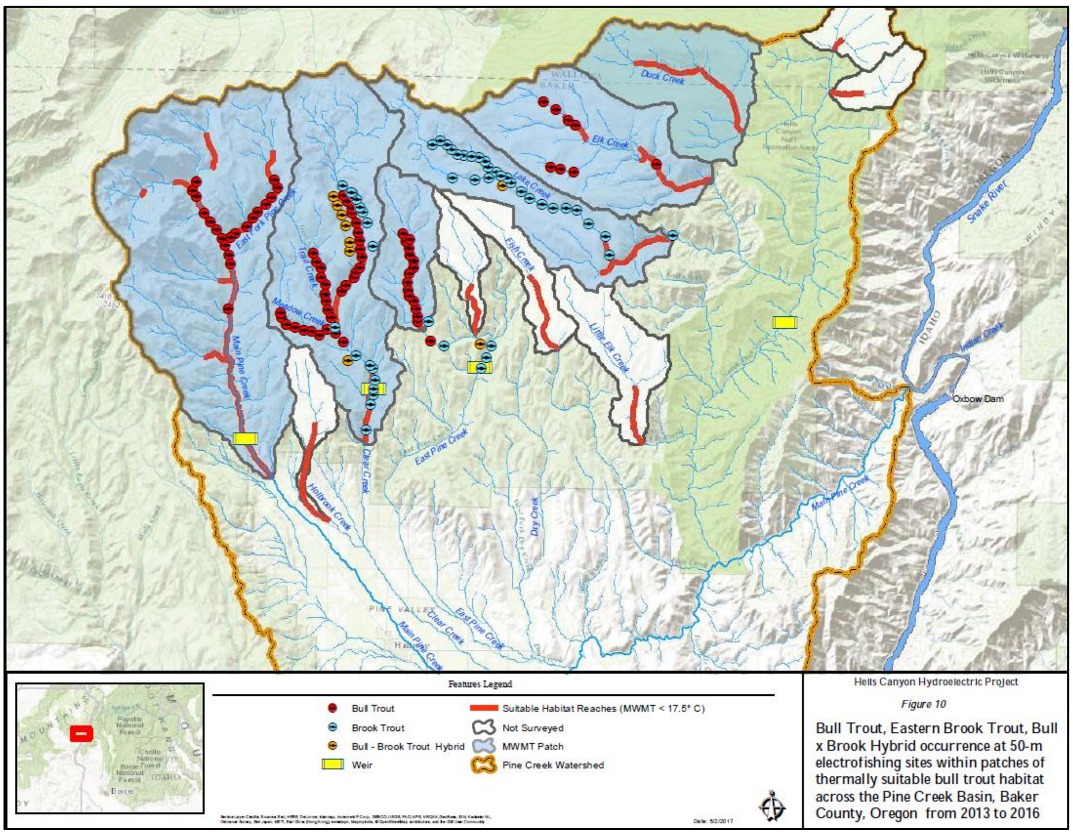
Pine Creek, Clear Creek, and East Pine Creek enter the broad Pine Valley after exiting the mountains (Figure 28). Here the creeks run roughly parallel through the valley, with several historic side channels and distributaries linking them. Overall canopy cover is high and composed of Cottonwood and Ponderosa Pine, but some incision and low vegetation cover can be found in some reaches. Irrigation withdrawals are also numerous through the valley, resulting in low flows in some tributaries during the late summer months. Further downstream the tributaries meet up and then flow through a more constrained reach. After meeting up with North Pine Creek, Pine Creek flows another ~11 km before reaching the confluence with the Snake River downstream of Oxbow Dam.

Figure 28. Several large tributaries of Pine Creek flow parallel in the broad Pine Valley.

Both Redband trout and Bull Trout are present in the Pine Creek watershed. Redband trout are more widespread, with most tributaries having some utilization over the course of the season. The highest Redband trout utilization is found in North Pine Creek and Lake Creek, with spawning and summer rearing particularly high in these reaches. Redband trout somewhat lower in Pine Creek above and below Pine Valley, Fish Creek, and East Pine Creek. Spawning uses can be mostly found in Fish Creek and East Pine Creek, while summer rearing is found primarily in the upper reaches of Pine Creek. The lowest Redband Trout utilization is found in Clear Creek, Pine Creek within Pine Valley, and Pine Creek below the confluence with North Pine, although winter rearing is high in the latter (**NPCC 2004c**).

Bull Trout populations in the Pine Creek watershed are much more constrained and are considered "At Risk", with small populations found in the headwaters of Upper Pine Creek, Clear Creek, East Pine Creek, and Elk Creek. Bull Trout spawning is primarily found in the headwaters of these creeks, while historically, migratory corridors were found in the lower sections of Pine Creek (**NPCC 2004c**).Populations are primarily limited by warm summer stream temperatures and by competition and interbreeding with Brook Trout introduced in high mountain lakes (**Buchanan, Hanson, and Hooton 1997, Wilkison and Trainer 2017**, Figure 29).

Figure 29. Bull Trout (Red circle), Brook Trout (Blue circle), and hybrid (Orange circle) distributions in the Pine Creek watershed from electrofishing surveys. Bull Trout primarily occupy headwater streams of the Pine, Clear, East Pine, and North Pine Creek watersheds, but face competition from Brook Trout in Clear Creek and Lake Creek. Source: Wilkison and Trainer 2017.



The PBWC has been involved in restoration and improvement projects within the Pine Creek watershed, primarily irrigation improvements in Pine Valley as part of water conservation program funded by Idaho Power. Alongside these projects have been stream bank erosion control and fish habitat improvement projects, mostly in partnership with the Pine Valley Soil and Water Conservation District and Idaho Power.

Monitoring during the 2022-2024 period occurred at five sites, including three longer term sites on Clear Creek and Pine Creek, one new site on East Pine, and one site on Meadow Creek within Schnieder Meadows. Dissolved oxygen loggers were installed at three sites to assess oxygen concentrations in Bull Trout spawning areas, including Meadow Creek, along with two new sites on upper Clear Creek and East Pine Creek. Temperature loggers were installed later in the season than other sites in 2023 but were able to capture temperatures during the late summer and early fall low flow period. Temperature logger data was also absent from Pine @ Oxbow for much of the summer due to dry conditions from changing water levels.

Grab Sample Monitoring

Dissolved oxygen concentrations were above the 6.5 mg/L cool water standard at all sites during the 2022-24 monitoring period. Concentrations were also consistently above the 8 mg/L cold-water standard at Pine @ Holbrook, East Pine @ Boulder Flat, and Meadow.15J.3. Overall oxygen concentrations were highest at Pine @ Holbrook and lowest at Pine @ Oxbow. Oxygen saturation was generally low at Meadow.15J.3, where saturation was below 100% for 10 of 17 measurements, most commonly in late summer and fall. Oxygen saturation was more variable at Clear @ Gulick with oxygen saturation below 100% for six out of 16 measurements and above 110% saturation for two measurements.

Chart 72. Observed dissolved oxygen saturation (top, in %) and concentrations (bottom, in mg/L) for sites in the Pine Creek watershed over the 2022-2024 monitoring period. Cool-water (solid line) and sold-water (dashed line) standards highlighted in red.

Overall oxygen concentrations were highest in the fall and lowest in July. Seasonal changes in oxygen were smaller at Meadow.15J.3 but were generally similar for the other sites lower in the basin. Only Meadow.15J.3 showed any seasonal patterns in oxygen saturation, with higher saturation in spring and lower saturation in the fall. Differences in oxygen concentrations between years were generally small and not significant, with higher concentrations seen in 2024 and lower concentrations seen in 2023. This pattern differed somewhat from the other basins, where 2023 had higher concentrations.

Table 66a. Estimates of mean Dissolved oxygen concentrations (in mg/L) for the Pine Creek watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Pine @ Oxbow	8.55	9.08	9.60
Clear @ Gulick	8.75	9.29	9.83
Pine @ Holbrook	9.13	9.66	10.18
East Pine @ Boulder Flat	8.76	9.45	10.14
Meadow.15J.3	8.53	9.09	9.65

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.22	0.31	0.83
July	-1.37	-0.85	-0.32
August	-1.18	-0.66	-0.15
September	-0.54	-0.05	0.44
October	0.75	1.25	1.76

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.51	0.02	0.54
2023	-0.77	-0.25	0.26
2024	-0.27	0.23	0.74

All pH measurements were within the 6.5 to 9 standard at the Pine Creek watershed sites, although were somewhat elevated at Pine @ Oxbow and Clear @ Gulick. pH measurements generally increased with larger basin size, with the lowest measurements found at Meadow.15J.3 and the highest measurements seen at Pine @ Oxbow. Mean pH differences between months were small and not statistically significant, with values higher in July and August and lower in June and in the Fall. Greater variability in pH was seen at Pine @ Oxbow, particularly in 2023, but was more consistent at Pine @ Holbrook, Clear @ Gulick and East Pine @ Boulder Flat. Differences between years were also small, but more noticeable than seasonal differences, with mean pH higher in 2024 and lower in 2022. Similar to the patterns in dissolved oxygen, these yearly patterns differed from other basins, where 2022 had higher pH than other years.

Table 67a. Estimates of mean pH for the Pine Creek watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Pine @ Oxbow	7.81	8.05	8.28
Clear @ Gulick	7.64	7.88	8.11
Pine @ Holbrook	7.63	7.87	8.10
East Pine @ Boulder Flat	7.59	7.88	8.18
Meadow.15J.3	7.18	7.42	7.66

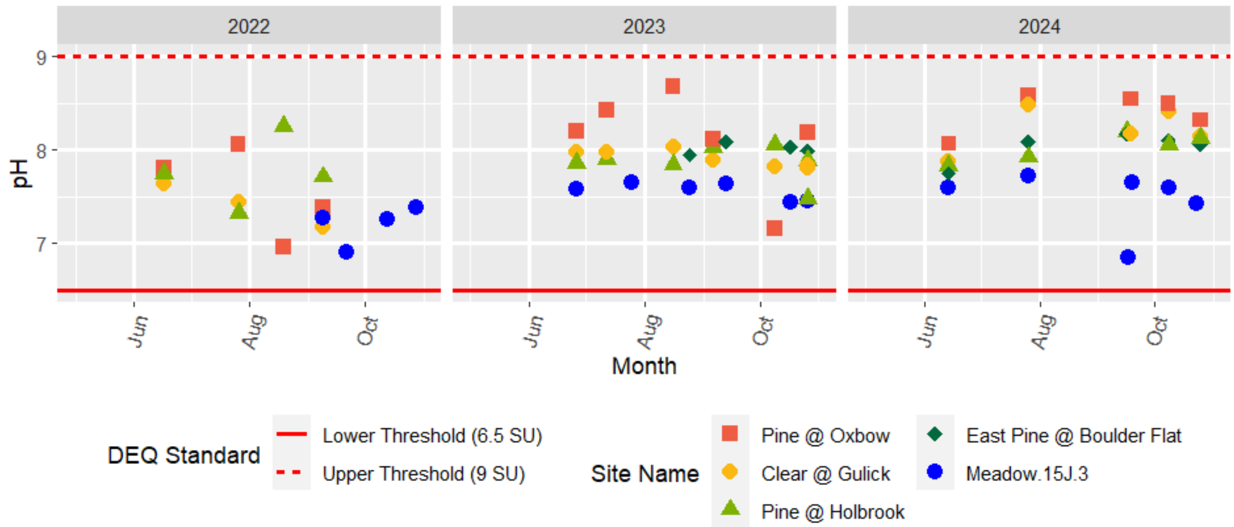
b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.30	-0.06	0.17
July	-0.16	0.08	0.31
August	-0.20	0.07	0.34
September	-0.27	-0.05	0.18
October	-0.27	-0.04	0.19

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.54	-0.30	-0.07
2023	-0.15	0.08	0.30
2024	0.01	0.23	0.45

Chart 73. Observed pH measurements for sites in the Pine Creek watershed over the 2022-2024 monitoring period. Upper (dashed line) and lower (solid line) recommended pH standards highlighted.



Conductivity patterns between sites generally showed an increasing trend related to increasing basin size, with the lowest measurements seen at Meadow.15J.3 and the highest values seen at Clear @ Gulick and Pine at Oxbow. Large increases in conductivity were seen between sites in Upper Pine Valley and those in lower Pine Valley, where conductivity measurements were generally similar between East Pine @ Boulder Flat and Pine @ Holbrook. The increase through Pine Valley was particularly notable given the smaller drainage basin size for Clear Creek above Gulick Lane (23.5 km^2 compared to 141.5 km^2 for Pine @ Holbrook and 75.5 km^2 for East Pine @ Boulder Flat).

Chart 74. Observed conductivity measurements (in µS/cm) for the Pine Creek watershed monitoring sites over the 2022-2024 period.

Table 68a. Estimates of mean conductivity (in µS/cm) for the Pine Creek watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Pine @ Oxbow	121.15	138.10	157.42
Clear @ Gulick	121.00	138.31	158.10
Pine @ Holbrook	86.72	98.83	112.63
East Pine @ Boulder Flat	73.43	87.58	104.45
Meadow.15J.3	69.89	80.31	92.28

b. Mean monthly pH differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-46.44	-37.83	-28.01
July	-16.18	-1.55	15.41
August	-0.90	14.09	31.17
September	-2.11	11.75	27.43
October	-0.17	13.54	28.99

c. Mean yearly pH differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-10.43	2.94	18.19
2023	-20.17	-8.59	4.56
2024	-7.57	5.64	20.63

Changes in conductivity through the season showed an increasing trend from early summer through fall, with the lowest mean values in June and the highest values in August and October. These increasing seasonal trends were most notable at Pine @ Oxbow and Clear at Gulick. Mean conductivity differences between years were small, but followed patterns seen in other basins, with conductivity highest in 2024 and lowest in 2023, likely related to higher precipitation and cooler temperatures during the winter and summer of this year.

Turbidity was generally suitable at all sites monitored in the Pine Creek watershed, with no site having measurements above 10 NTU's and only two sites having measurements above 5 NTU's on three occasions. Mean turbidity was highest at Meadow.15J.3, although some measurements were likely biased due to low flows and disturbance making sampling difficult to take representative samples. Measurements found that turbidity was highest in June, but no discernable pattern was noticeable during the remainer of the year. Yearly patterns in turbidity were generally small, but had noticeable differences between years, with turbidity in 2024 ~ 0.77 NTU's higher than the 2022-23 average.

Table 69a. Estimates of mean turbidity (in NTU's) for the Pine Creek watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Pine @ Oxbow	0.77	1.15	1.72
Clear @ Gulick	0.97	1.46	2.19
Pine @ Holbrook	0.77	1.15	1.72
East Pine @ Boulder Flat	1.07	1.78	2.98
Meadow.15J.3	1.32	2.06	3.21

b. Mean monthly turbidity differences from mean values over the 2022-2024 period

Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	0.26	1.16	2.50
July	-0.67	-0.24	0.41
August	-0.60	-0.13	0.56
September	-0.81	-0.44	0.11
October	-0.72	-0.35	0.18

c. Mean yearly turbidity differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.72	-0.33	0.26
2023	-0.61	-0.18	0.46
2024	-0.15	0.51	1.49

Chart 75. Observed turbidity measurements (in NTU's) for the Pine Creek watershed monitoring sites over the 2022-2024 monitoring period. Log transformation used for turbidity measurements.

Stream Temperature monitoring

Average mean and maximum stream temperatures were warmest in the lower part of Pine Valley and at locations downstream. Stream temperatures were above the 20 °C cool-water standard for Redband trout at Pine @ Oxbow, Clear @ Gulick, and East Pine a@ Boulder Flat for some periods, particularly in July and August. Overall stream temperatures were highest at Pine @ Oxbow, where maximum temperatures exceeded the cool water standard 98.4 % of days in July and 100% of days in August. Temperatures were also above the cool water standard for significant periods of time at Clear @ Gulick, with 98.6% of days in July and 84.2% of days in August above 20 °C. June and September also had a significant number of days with temperatures above the cool water standard, with exceedance more common in September for Pine @ Oxbow and June more common at Clear @ Gulick.

Maximum temperatures were above the 12 °C cold-water standard at Meadow.15J.3, primarily in July and August, where over 50% of days had temperatures above 12 °C. While common, these exceedances were small, with no day having seven-day average maximum temperatures above 12.99 °C and with the average maximum temperature for these months being below 12 °C.

While the sites within and above Pine Valley generally drain separate watersheds from one another, patterns related to mean and maximum temperatures were still noticeable on a landscape scale. Notable increases in mean temperatures were seen between Clear @ Gulick and Pine @ Oxbow, increasing by 3.19 °C, although the increase in maximum temperatures between these sites was smaller at 1.98 °. Differences in temperatures between the Upper Pine Valley sites and Clear @ Gulick were more pronounced for Pine @ Holbrook, where mean temperatures increased by 3.37

°C and maximum temperatures increased by 5.36 °C. Warmer temperatures overall at East Pine @ Boulder Flat resulted in smaller changes compared to Clear @ Gulick, but were still above 1.2 °C for mean temperatures and 2.7 °C for maximum temperatures. Large changes in temperature were also notable between Meadow.15J.3 and the Upper Pine Valley sites, increasing by 4.63 °C on average for mean temperatures and 4.29 °C for maximum temperatures, with larger changes seen between Meadow Creek and Pine @ Holbrook, particularly for maximum temperatures.

Chart 76. Temperature profiles (in °C) for the Pine Creek watershed monitoring sites over the 2022-2024 period, with lower watershed sites on the left, upper Pine Valley sites in the middle, and the Meadow Creek site on the right. Seven-day Mean temperatures highlighted with shaded range for 7-day Average maximum and minimum temperatures. Cold water (dashed line) and cool water (solid line) standards highlighted in red based off relevant fish use at the site.

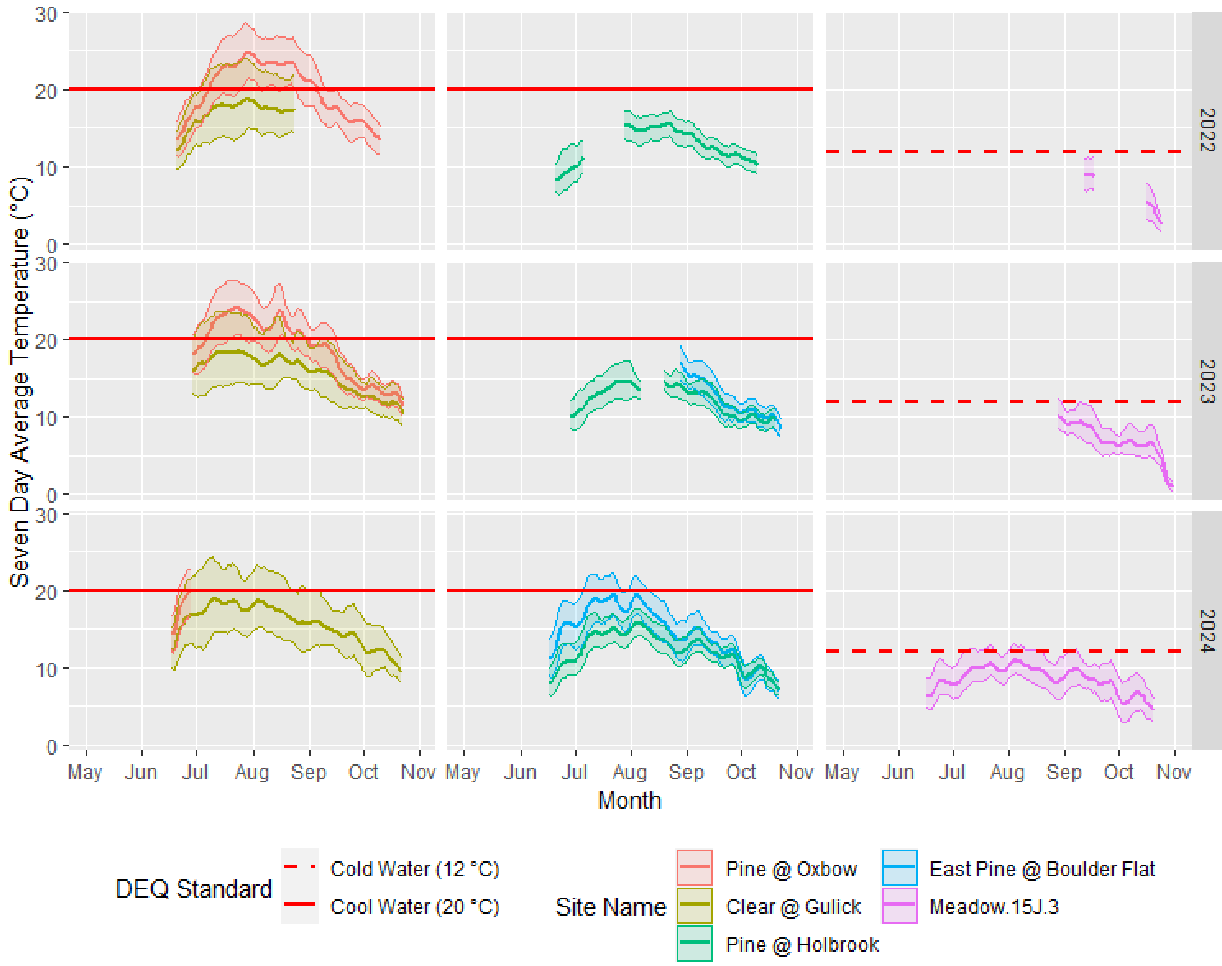


Table 70a. Estimates of Mean and Maximum Daily temperatures (in °C) at the Pine Creek watershed monitoring sites for the 2022-2024 period

	Mean Temperature			Max Temperature		
Site Name	Lower 95%	Mean	Upper 95%	Lower 95%	Mean	Upper 95%
Pine @ Oxbow	17.38	18.30	19.23	19.77	20.83	21.90
Clear @ Gulick	14.16	15.12	16.07	17.75	18.85	19.95
Pine @ Holbrook	10.82	11.75	12.69	12.42	13.50	14.57
East Pine @ Boulder Flat	12.66	13.89	15.13	14.68	16.09	17.51
Meadow.15J.3	7.07	8.20	9.32	9.22	10.51	11.80

b. Average differences in monthly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Month	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-2.22	-1.30	-0.37	-2.33	-1.27	-0.20
July	1.56	2.49	3.41	2.19	3.25	4.31
August	1.66	2.61	3.56	1.76	2.85	3.94
September	-1.16	-0.24	0.69	-1.60	-0.53	0.53
October	-4.51	-3.56	-2.61	-5.39	-4.29	-3.20

c. Average differences in yearly Mean and Maximum temperature compared to mean values over the 2022-2024 period

Year	Mean Temperature			Max Temperature		
	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-0.94	-0.02	0.91	-1.00	0.07	1.13
2023	-0.85	0.07	1.00	-1.09	-0.03	1.03
2024	-1.01	-0.05	0.90	-1.13	-0.04	1.06

Mean and maximum daily temperatures differed somewhat in seasonal patterns for the sites, with mean temperatures highest in August, whereas maximum temperatures were highest in July. The differences between July and August was smaller for mean temperatures than for maximum temperatures, 0.12 °C compared to 0.4 °C. Both mean and maximum temperatures were lowest in October, and no days had maximum temperatures over 20 °C for all sites during this month. Seasonal patterns were generally similar between sites for both maximum and mean temperatures.

Comparison of observed mean August stream temperatures to the 1993-2011 estimates from the NorWeST model found a 0.86 °C increase in temperatures on average. Increases in temperature from the NorWeST model were most notable for Pine @ Oxbow, where observed mean August stream temperatures for the 2022-24 period were 3.05 °C warmer than the NorWeST estimates. Differences from the NorWeST model were more muted for Meadow.15J.3 and Pine @ Holbrook, sites where temperature profiles and grab sample data indicate high snowmelt and groundwater contributions, with observed differences of 0.27 ° for Meadow.15J.3 and -0.39 °C at Pine @ Holbrook seen for the 2022-2024 period.

Table 71. Estimated and Observed mean August stream temperatures for blank watershed monitoring sites alongside 1993-2011 estimates of mean August stream temperatures from NorWeST stream temperature model and temperature projections for 2040 and 2080.

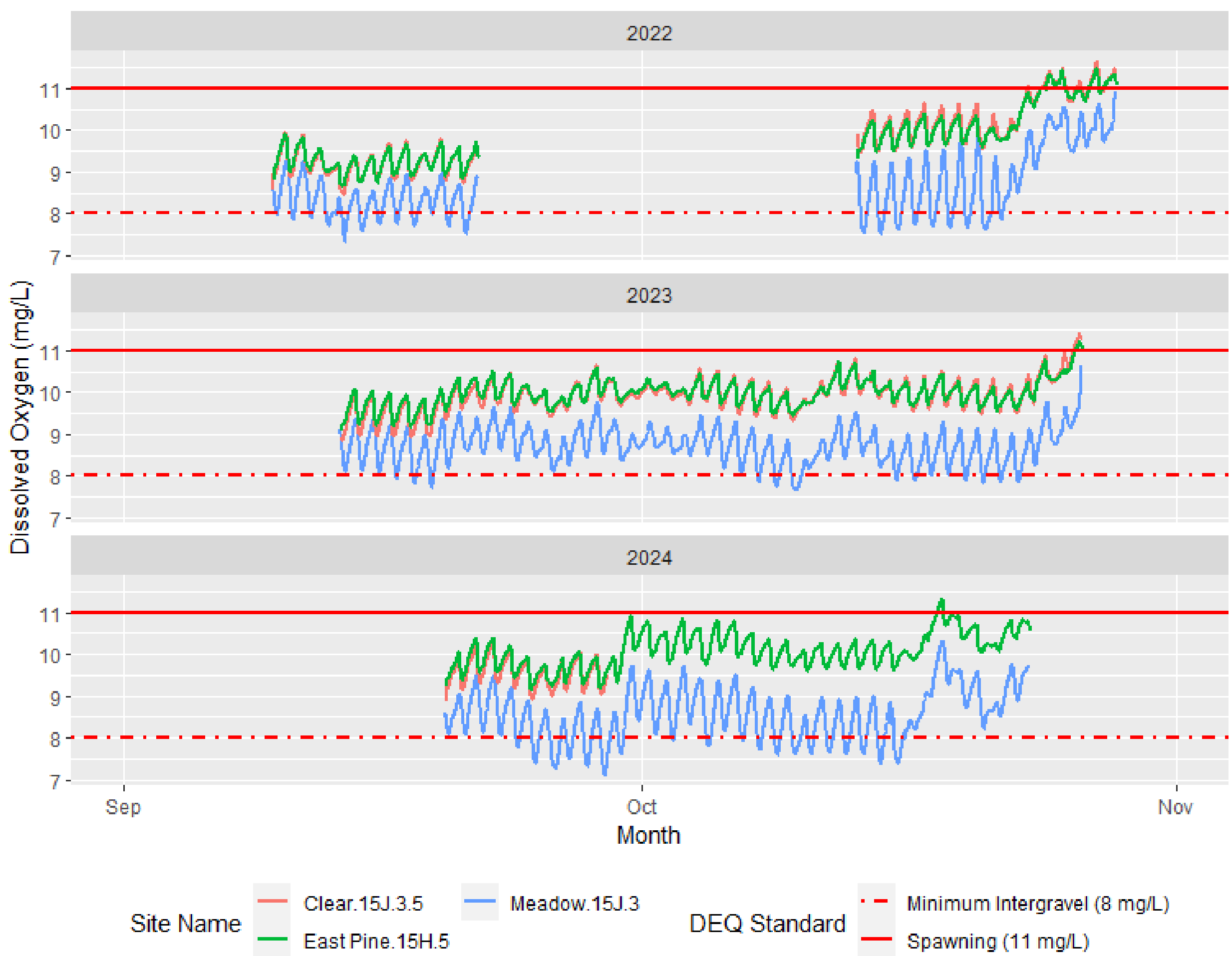
Site	Modelled 2022-2024	Observed 2022-2024	NorWeST 1993-2011	NorWeST 2040	NorWeST 2080
Pine @ Oxbow	20.91	22.42	19.37	20.69	21.53
Clear @ Gulick	17.72	17.27	16.73	17.98	18.78
Pine @ Holbrook	14.36	14.30	14.69	15.89	16.65
East Pine @ Boulder Flat	16.50	16.69	15.87	17.10	17.88
Meadow.15J.3	10.80	9.91	9.64	10.72	11.40

Dissolved Oxygen monitoring

Dissolved Oxygen Loggers were installed at three sites in the Fall to monitor oxygen concentration during the Bull Trout spawning season. These sites were all located within tributaries in the upper watershed, including Meadow.15J.3, and two sites on East Pine Creek and Clear Creek above the

Meadow Creek confluence. Two ~14 day deployments were used in 2022 to improve overall logger coverage in other watersheds, while deployments in 2023 and 2024 were longer and more continuous, averaging ~38 days. The dissolved oxygen logger deployment at Clear Creek in 2024 was heavily biased by algal buildup on the logger sensor, which resulted in poor data quality for measurements taken after September 29th. Overall dissolved oxygen concentrations were consistently above the 8 mg/L intergravel minimum standard on Clear Creek and East Pine Creek but were below this level for 14.7% of records at Meadow.15J.3. Oxygen concentrations were below the 11 mg/L spawning standard at all sites for most of the monitoring period, with concentrations only exceeding this level at Clear.15J.3.5 and East Pine.15H.5 late in the season. Lower oxygen concentrations were more common in 2022 and 2024 at Meadow.15J.3, although patterns in 2022 were likely related to low oxygen accuracy, where logger measurements were 0.44 mg/L lower than measurements taken with the multi-meter.

Chart 77. Dissolved oxygen profiles (in mg/L) for the Fall Pine Creek watershed oxygen logger monitoring sites over 2022 to 2024 period. Minimum intergravel (dashed line) and spawning (solid line) oxygen standards highlighted in red.



Timing of dissolved oxygen differed somewhat between Meadow Creek and the other logger sites. Maximum oxygen concentrations were found around 11:00 at Meadow.15J.3, whereas maximum oxygen levels were found earlier in the day (~9:00) at Clear Creek and East Pine Creek. Sensitivity of oxygen concentrations to temperature was lower at Meadow.15J.3, where oxygen concentration decreased by 0.17 mg/L per 1 °C increase in temperature compared to decreases of 0.24 mg/L per 1 °C increase at Clear and East Pine Creeks (Appendix D). Differences in oxygen concentrations between years differed by sites and were more likely related to logger accuracy issues than to actual conditions. On average, oxygen concentrations recorded by the loggers were 0.16 mg/L higher in 2023 and 0.9 mg/L higher in 2024 than in 2022.

Water Quality Index

Water quality between 2022 and 2024 for Pine Creek watershed sites demonstrated large variation between sites and seasons, particularly for the lower valley and downstream sites. Overall WQI scores were highest at Pine at Holbrook, with slightly lower scores seen at Meadow.15J.3 and East Pine @ Boulder Flat. WQI scores were consistently above Excellent thresholds at these sites throughout the monitoring period except for one measurement in July 2024 at East Pine @ Boulder Flat. WQI scores were more variable at the lower sites, with both Clear @ Gulick and Pine @ Oxbow showing larger changes in WQI scores from early summer through fall.

Table 72a. Estimates of mean WQI scores (from 10-100) for the Pine Creek watershed monitoring sites over the 2022-2024 period

Site Name	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
Pine @ Oxbow	83.19	86.43	89.79
Clear @ Gulick	87.65	91.06	94.61
Pine @ Holbrook	92.49	96.09	99.83
East Pine @ Boulder Flat	88.70	93.58	98.73
Meadow.15J.3	89.69	93.62	97.72

b. Mean monthly Dissolved oxygen differences from mean values over the 2022-2024 period

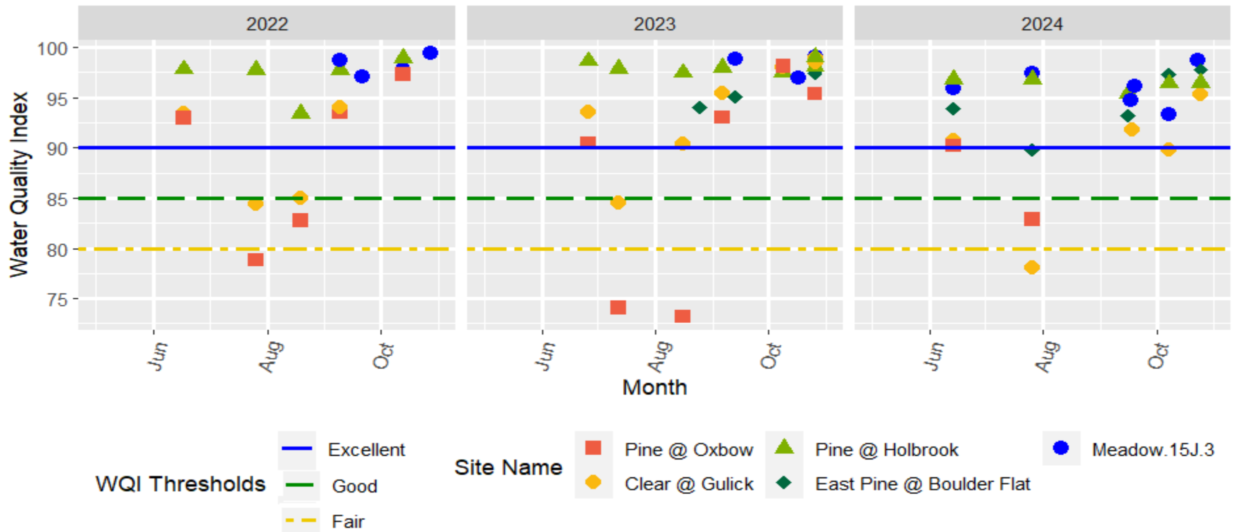
Month	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
June	-0.94	2.61	6.30
July	-9.54	-6.10	-2.53
August	-7.96	-4.55	-0.99
September	-0.29	3.24	6.91
October	0.99	4.80	8.76

c. Mean yearly Dissolved oxygen differences from mean values over the 2022-2024 period

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2022	-2.75	0.73	4.34
2023	-2.70	0.59	4.00
2024	-4.73	-1.32	2.23

Low WQI scores at the lower watershed sites during the summer months were primarily the result of warm maximum temperatures, with an average temperature sub-index score of 65.5 for Clear @ Gulick and 46.8 for Pine @ Oxbow during July and August. Dissolved oxygen was less of a factor lowering WQI scores but was still noticeably lower than the upper watershed sites, averaging 83.3 for Clear @ Gulick and 84.9 for Pine @ Oxbow during the same months. While high pH was not a particularly large factor resulting in low WQI scores at either of these sites, pH sub-index scores were still somewhat low at Pine @ Oxbow, averaging 84.3 for the summer period.

Chart 78. WQI values at the Pine Creek watershed monitoring sites over the 2022-2024 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), and Fair (85-80 WQI) thresholds highlighted.



While overall WQI scores for the upper watershed sites were generally excellent, some factors did lower WQI scores more than others. Like the lower watershed sites, warm maximum temperatures were an important factor lowering WQI scores at East Pine @ Boulder Flat, primarily during the summer, with subindex scores of 74.9 in July and 82.6 in August compared to 92.5 for the other months. In contrast, temperature sub-index scores were consistently above 99 at Meadow.15J.3. Instead, turbidity and dissolved oxygen were the factors most responsible for lower WQI scores at this site, with average sub-index scores of 93.6 and 95.5 respectively. All average sub-index scores were above 96 for Pine at Holbrook, including during the summer months when temperature, dissolved oxygen, and pH issues were more prevalent at other locations.

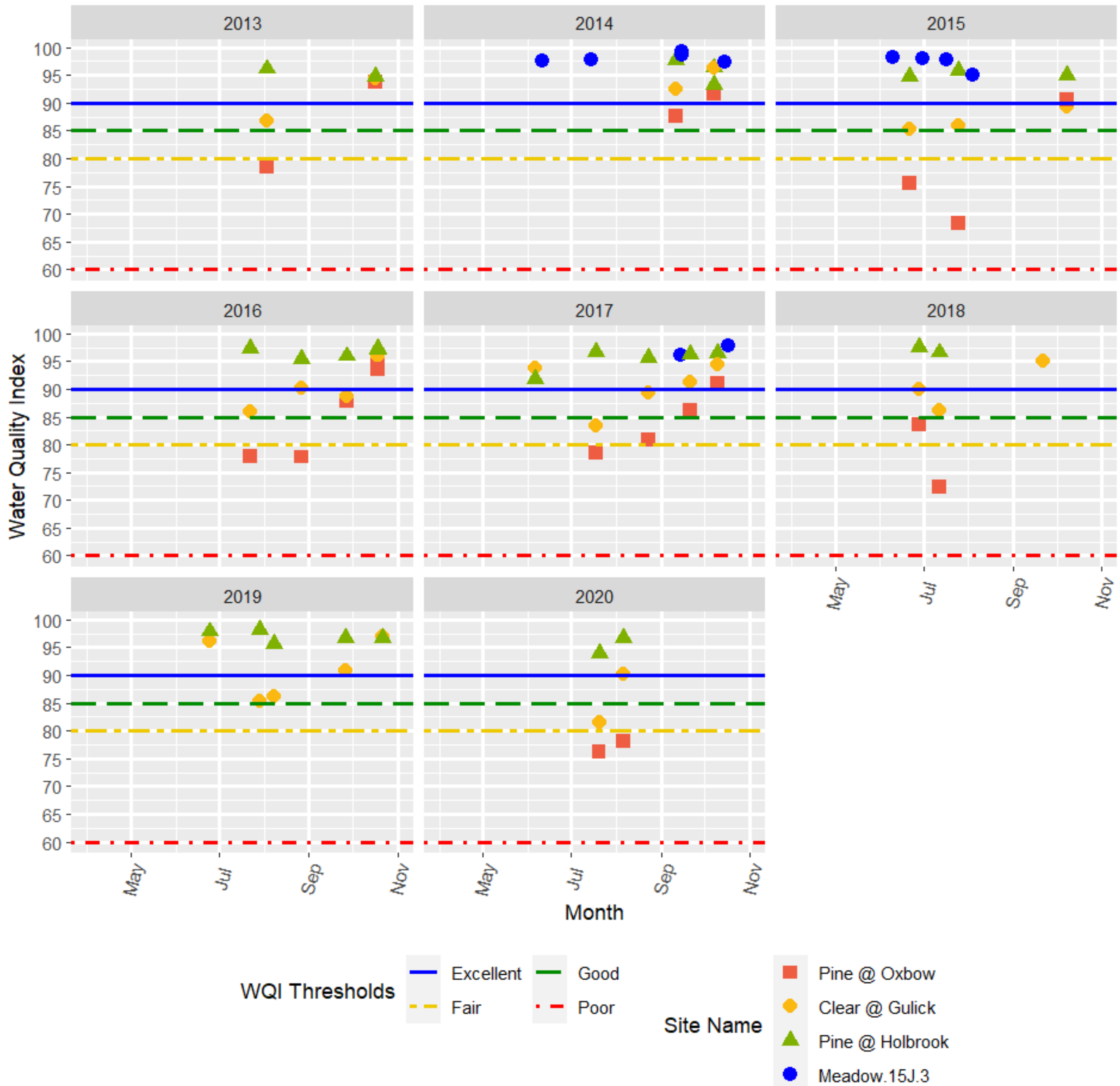
Table 73. Estimates for differences in mean WQI scores for the Pine Creek watershed monitoring sites from mean values over the 2013-2024 period.

Year	Lower 95% Estimate	Mean Estimate	Upper 95% Estimate
2013	-3.89	0.17	4.41
2014	-3.14	0.11	3.48
2015	-5.06	-2.27	0.62
2016	-2.34	0.77	3.99
2017	-3.26	-0.34	2.67
2018	-4.30	-0.86	2.72
2019	-2.59	0.87	4.47
2020	-4.83	-0.79	3.45
2022	-1.87	1.04	4.05
2023	-1.18	1.56	4.39
2024	-3.71	-0.27	3.31

Patterns in WQI scores over the 2013-2024 period for the four sites with data prior to 2022 (Pine @ Oxbow, Clear @ Gulick, Pine @ Holbrook, and Meadow.15J.3) showed some small differences from the 2022-24 period. Mean WQI scores were lower at most sites during the 2013-2024 period and mean monthly differences were smaller, but overall these differences were small (< 2.0 WQI). WQI scores were lower than average in 2015, 2018, 2020, and 2024, with the lowest WQI scores found in

2015, while higher than average WQI scores were seen in 2016, 2019, 2022, and 2023. These patterns appear to be related to drought conditions and snowpack, with lower precipitation years having lower WQI scores.

Chart 79. WQI values at the Pine Creek watershed monitoring sites over the 2013-2020 monitoring period. Excellent (>90 WQI), Good (90-85 WQI), Fair (85-80 WQI), and Poor (80-60 WQI) thresholds highlighted.



Discussion and future plans

Patterns in water quality differed between the headwaters, upper valley, and location downstream of the lower valley. Overall water quality was highest at Meadow.15J.3 in the headwaters of Clear Creek. Temperatures were generally suitable for Bull Trout in this stream, but low oxygen concentrations were more important in lowering water quality, with lower concentrations than needed for successful egg incubation during the fall Bull Trout spawning period. Some turbidity issues were also present at this site, but these might be an artifact of sampling issues related to low flow. Moving forward, more information is needed at this site to identify the impact of planned restoration on temperatures and oxygen concentrations. The hope is that greater floodplain connectivity and channel roughness might increase oxygen concentrations while continuing cold temperature conditions, but data is needed to ensure the project achieves these results (Figure 30).

Figure 30. Channel incision and lack of woody debris at Meadow Creek might be the reason for low concentrations of dissolved oxygen at the monitoring site there.

Water quality in the Upper Pine Valley differed between Pine Creek and East Pine Creek. While temperatures on Pine Creek were suitable for redband trout throughout the summer, temperatures were higher on East Pine Creek, where maximum stream temperatures in July and August exceeded 20 °C routinely. Differences in flow between the sites are likely the reason for the discrepancy in stream temperatures, with greater snowmelt and higher summer flows at Pine @ Holbrook likely moderating temperature increases during the summer compared to East Pine @ Boulder Flat.

Multiple water quality issues exist at the sites in the lower Pine Valley and locations downstream, including warm stream temperatures, high pH, and low concentrations of dissolved oxygen. These issues are more notable at Pine @ Oxbow near the Snake River confluence, where maximum daily temperatures exceeded the 20 °C cool water standard frequently from June to September. Temperature issues were less severe at Clear @ Gulick in the lower Pine Valley, but were still present, particularly in the summer months. Issues related to high pH and low concentrations of dissolved oxygen were common in the lower valley, also primarily in the summer.

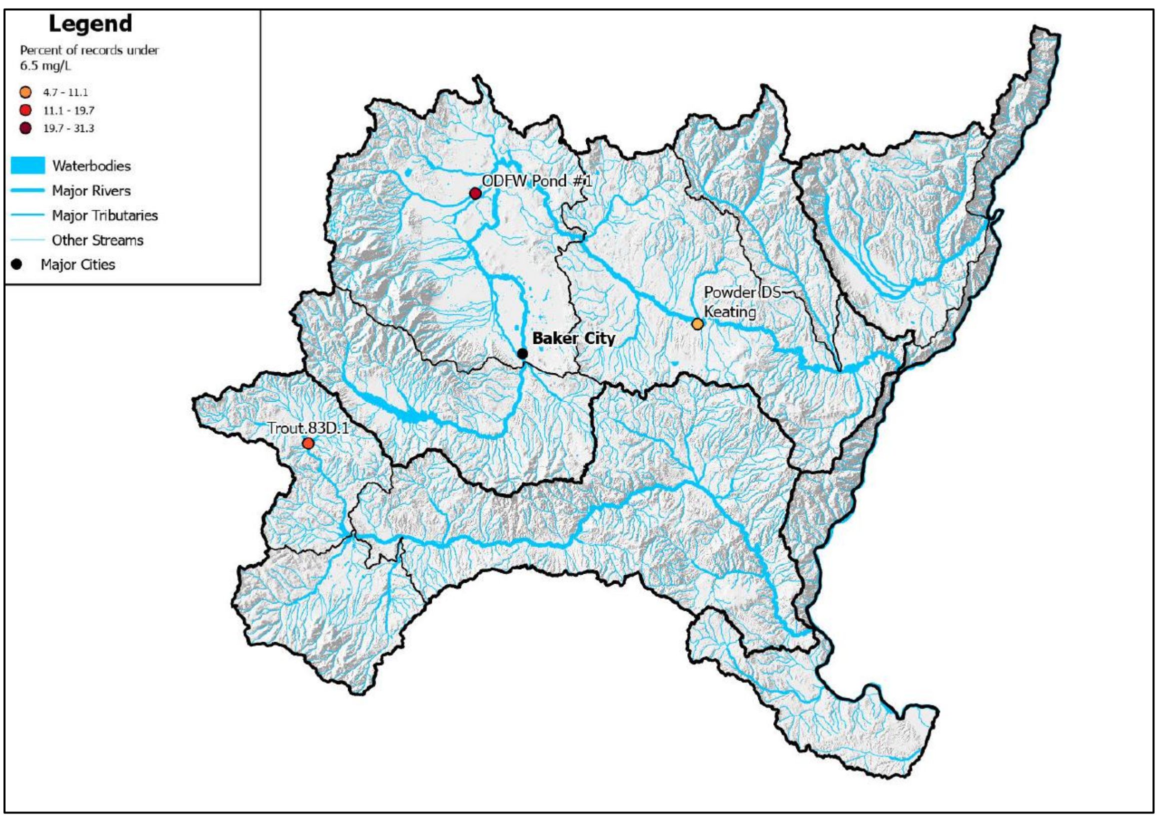
Looking toward future monitoring efforts in the watersheds, information is needed on other Bull Trout streams, particularly Upper North Pine Creek watershed. While past monitoring has taken place at a site on lower North Pine Creek and current monitoring at two sites on Elk and Aspen Creek with dissolved oxygen loggers in the fall, the lack of data in the upper watershed makes it difficult to identify thermal refuge and priority habitats in this region. There is also a need to identify patterns in water quality within Pine Valley to identify locations where decreases in water quality are present and to focus restoration and improvement opportunities. The PBWC will need to look to private landowners for access to these sites given the lack of public land and access points within the Valley.

Results and Discussion

Patterns in Grab sample Parameters

Dissolved oxygen concentrations in cool water streams were below the 6.5 mg/L cool-water standard at three sites: ODFW Pond #1, Trout.83D.1, and Powder DS Keating (Map 13). The large increase in conductivity measurements from the upstream ODFW Pond #2 and cooler maximum stream temperatures indicate that groundwater might be an important factor resulting in lower concentrations of dissolved oxygen at ODFW pond #1. Separation from atmospheric oxygen sources and slow flow through substrates often result in lower levels than surface water sources, particularly for broad valley streams like the North Powder River in this reach (**Greig et al. 2007**). Low levels of dissolved oxygen were also seen downstream of the beaver dam complex in lower Trout Creek. Similar patterns in oxygen concentrations within beaver dam pools and in Beaver-assisted restoration projects have been seen in past studies (**Rozhkova-Timina et al. 2018, Stevenson et al. 2022**) due to stratification and low levels of stream mixing within these systems.

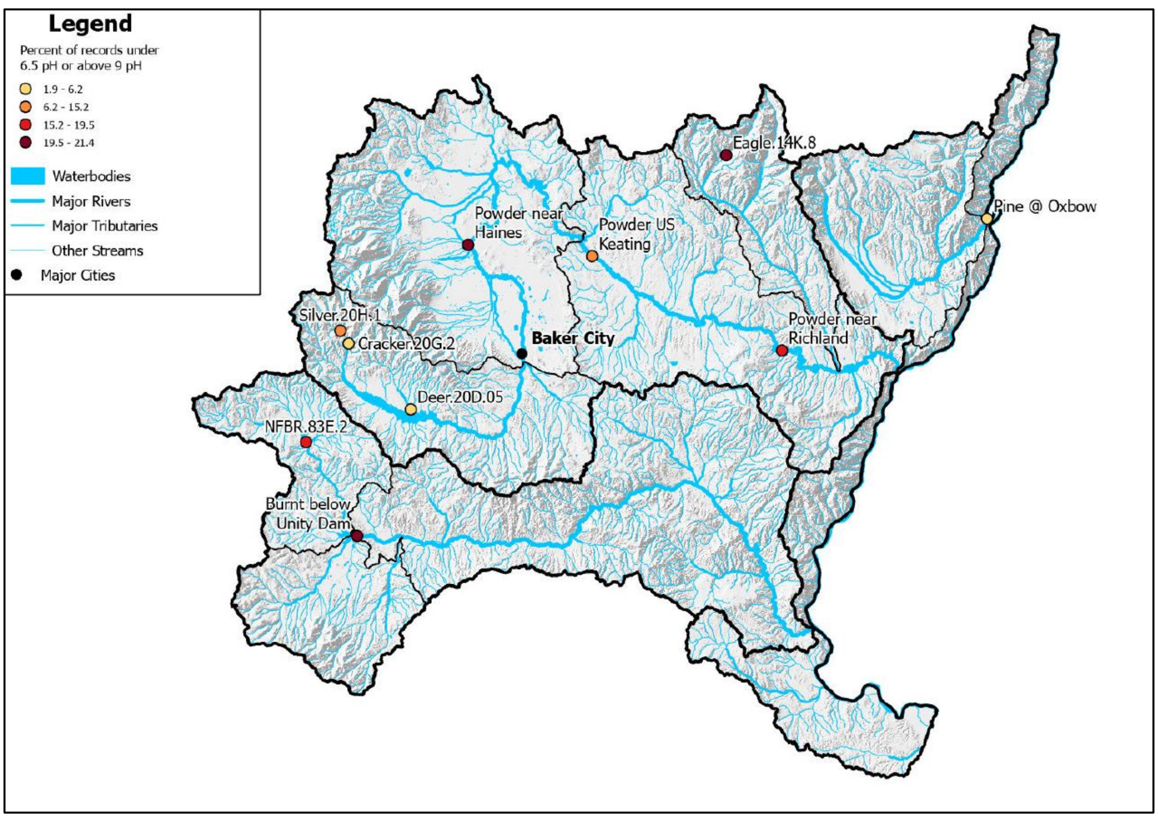
Map 13. Powder Basin cool-water monitoring sites where oxygen concentration measurements were below the 6.5 mg/L cool-water standard. No cold-water sites had grab sample oxygen concentrations below 8 mg/L.



High variability in both concentrations and saturation of oxygen, as well as a weak positive relationship stream temperature from dissolved oxygen loggers, indicate eutrophic conditions at Powder DS Keating are the likely source of low oxygen levels, primarily during the night when decomposition rates are highest. Similar patterns in dissolved oxygen were seen in a study of periphyton and macrophytes on dissolved oxygen concentrations in the Yakima River, particularly in reaches where observed biomass was highest (**Wise et al. 2009**). These patterns were also strongly related to concentrations of nitrogen and phosphorus, indicating these nutrients are a primary driver of the increase in macrophyte and periphyton biomass.

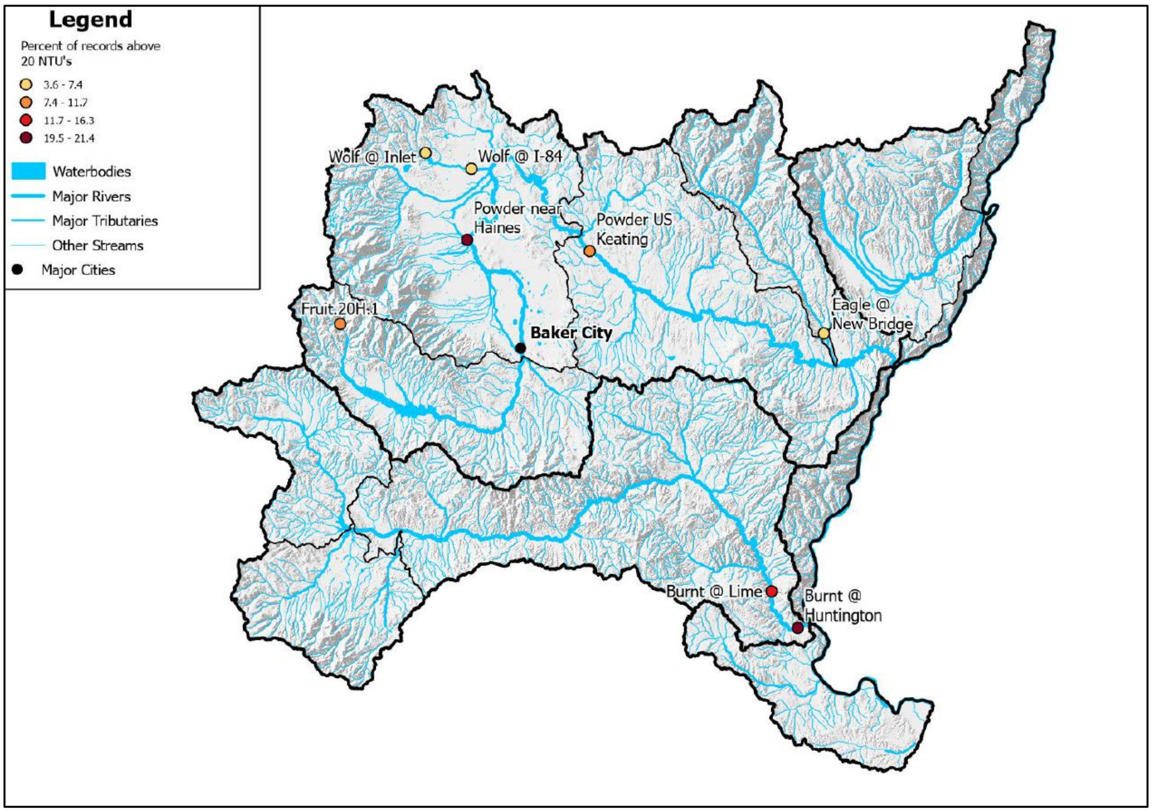
pH measurements were outside recommended DEQ standards at ten sites (Map 14). While some of these measurements were likely impacted by equipment issues, most reflect the true conditions at these sites. While the low pH at Eagle.14K.8 is likely an artifact of low pH probe accuracy in 2022, the high pH seen at other sites in the watershed likely reflect actual conditions. The largest number of samples exceeding the upper pH threshold was seen at the Burent River site below Unity Dam. Here, high levels of dissolved oxygen concentration and saturation, along with the high pH measurements, indicate issues with eutrophication. Significant number of samples exceeding the upper pH threshold were also seen at Powder near Richland and NFBR.83E.2, both of which showed similar patterns in oxygen and pH indicating eutrophic conditions. Similar pH patterns were seen alongside high variability in dissolved oxygen in the Yakima River, where more intensive investigations using phosphorus and nitrogen concentrations affirmed the impact of primary production on water quality and the presence of eutrophic conditions (**Wise et al. 2009**).

Map 14. Powder Basin monitoring sites where pH measurements were below the 6.5 threshold or above 9 threshold recommended by DEQ.



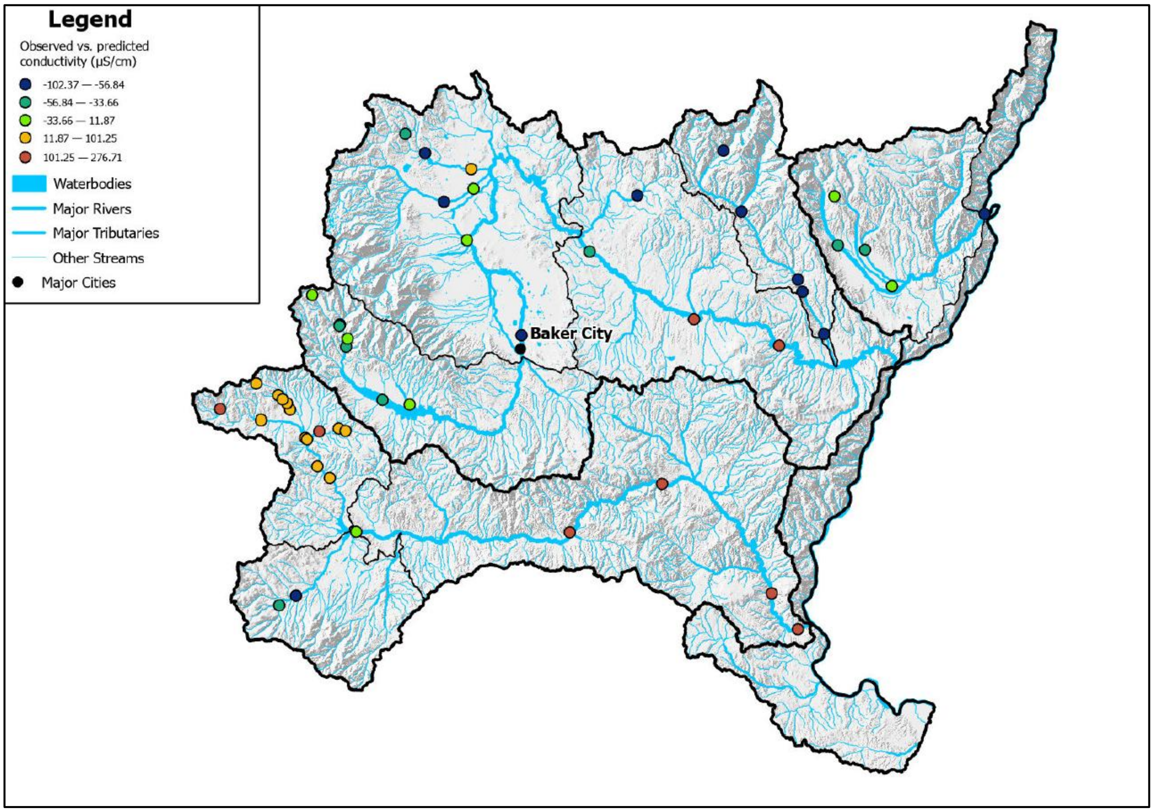
High turbidity levels (>20 NTU's) were seen at 9 sites, mostly found in lower Burnt River, lower Eagle Creek, and Wolf Creek (Map 15). Sites showing a particularly large number of measurements above 20 NTU's were Burnt @ Huntington, Burnt @ Lime, and Powder near Haines. Isolated occurrence of high turbidity, like those at Fruit.20H.1 also occurred but did not have any significant impact on overall water quality. Other sites had elevated turbidity levels (>10 NTU's) occurring mostly in the spring, like at Wolf @ Inlet. The largest turbidity exceedance was seen at Goose.29C.2, where turbidity regularly exceeded 50 NTU's during several site visits, and a high turbidity measurement of 230.5 NTU's recorded on 5/29/2024. Addressing these turbidity issues should be a high priority for the PBWC given the importance of the Goose Creek watershed as spawning and thermal refuge habitat for redband trout.

Map 15. Powder Basin monitoring sites where Turbidity measurements were above 20 NTU's, indicating moderate or higher levels of turbidity.



Conductivity measurements identified several trends throughout the basin, with a weak positive correlation ($R^2 = 0.25$) between larger upstream basin sizes and higher conductivity, as well as an increasing trend in conductivity from spring to early autumn. Locations with lower mean conductivity than expected in relation to basin size were found in the Upper Powder, Eagle Creek, the SFBR, Pine Creek, and Wolf Creek. In contrast, sites with higher than expected conductivity in relation to basin size were found in the NFBR and in the lower mainstem reaches of the Bunt River and Powder River (Map 16). In combination with other parameters such as temperature and oxygen, conductivity measurements identified locations where groundwater contributions likely compose a large percentage of surface flow (**Brown, Milner, and Hannah 2007, Rey et al. 2024**). These patterns were most notable at sites on Camp Creek, Trout Creek, and the lower portions of Powder River tributaries, particularly the North Powder River and Wolf Creek.

Map 16. Observed vs expected estimates of mean conductivity (in µS/cm) over the 2022-24 period from correlations with basin size (in km^2). Sites with negative values indicate that observed mean conductivity was lower than expected based off of basin size (blue) while sites with positive values indicate that observed mean conductivity was higher than expected based off of basin size (red).

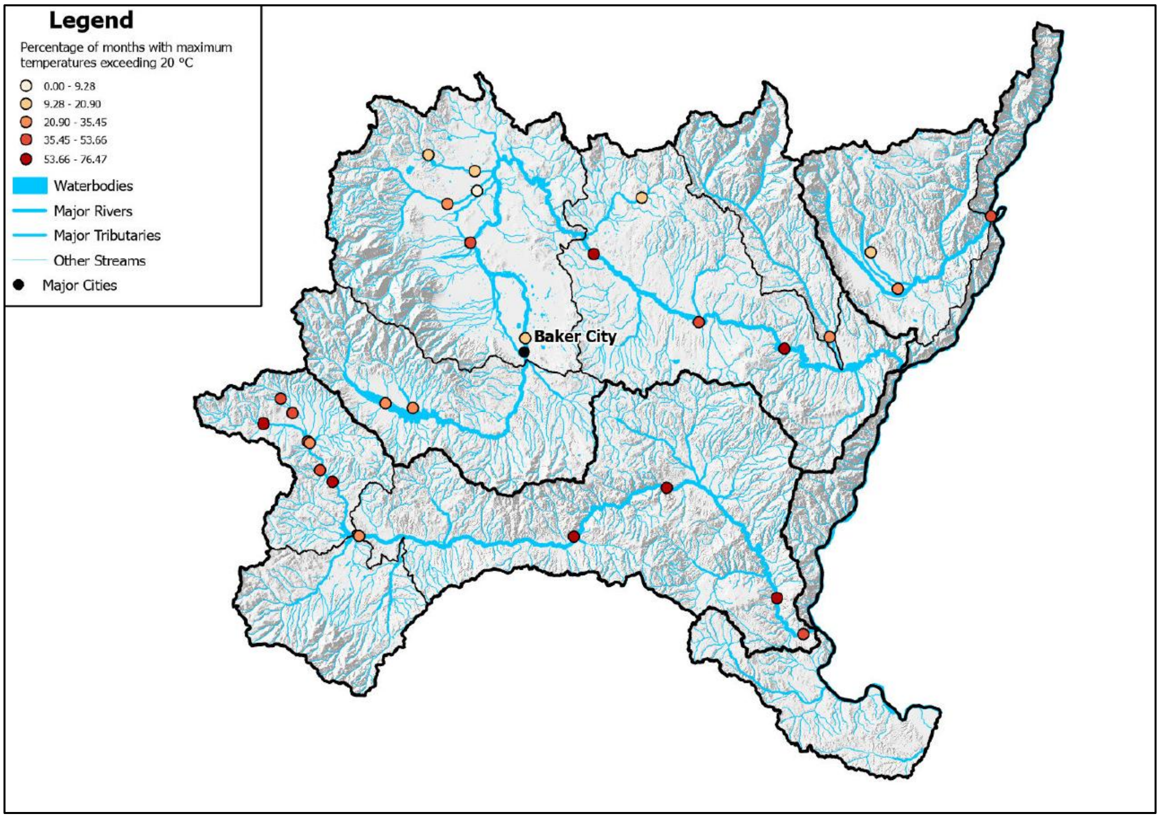


Patterns in Water Temperature

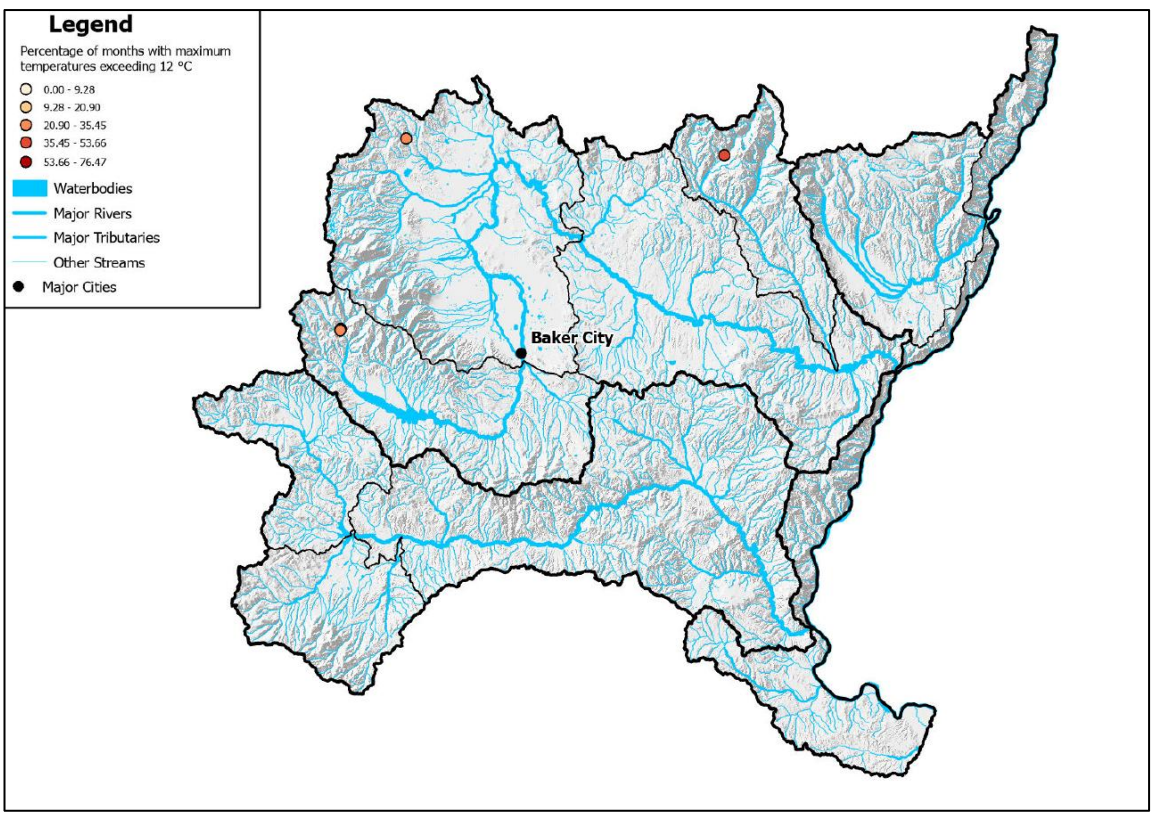
Maximum stream temperatures were above 20° at 29 sites for some part of the monitoring period, but were more common at sites in lower watershed, with >70% of months at Powder near Richland with average monthly temperatures above 20 °C (Map 17). Other sites with more than 50% of months with maximum temperatures above 20 ° included sites in the lower NFBR, lower Burnt, and Lower Powder Rivers. Sites with maximum temperatures below 20 °C were primarily located in the upper watershed, including sites on Cracker Creek, Eagle Creek, Silver Creek, the SFBR, Meadow Creek, Snow Creek, Trout Creek, and sites in middle and upper Camp Creek.

Maximum temperatures exceeded 12 °C at some point of the monitoring period at all cold-water Bull Trout sites, although mean monthly maximum temperatures did not exceed 12 °C at Meadow.15J.3 and Silver.20H.2. The largest number of months exceeding 12 °C were found at Eagle.14K.8 and Fruit.20H.1, with maximum temperatures above 12 °C for more than 30% of months at Wolf.18C.1 and Silver.20H.1(Map 18).

Map 17. Powder Basin cool-water habitat monitoring sites where maximum temperatures exceeded 20 °C.

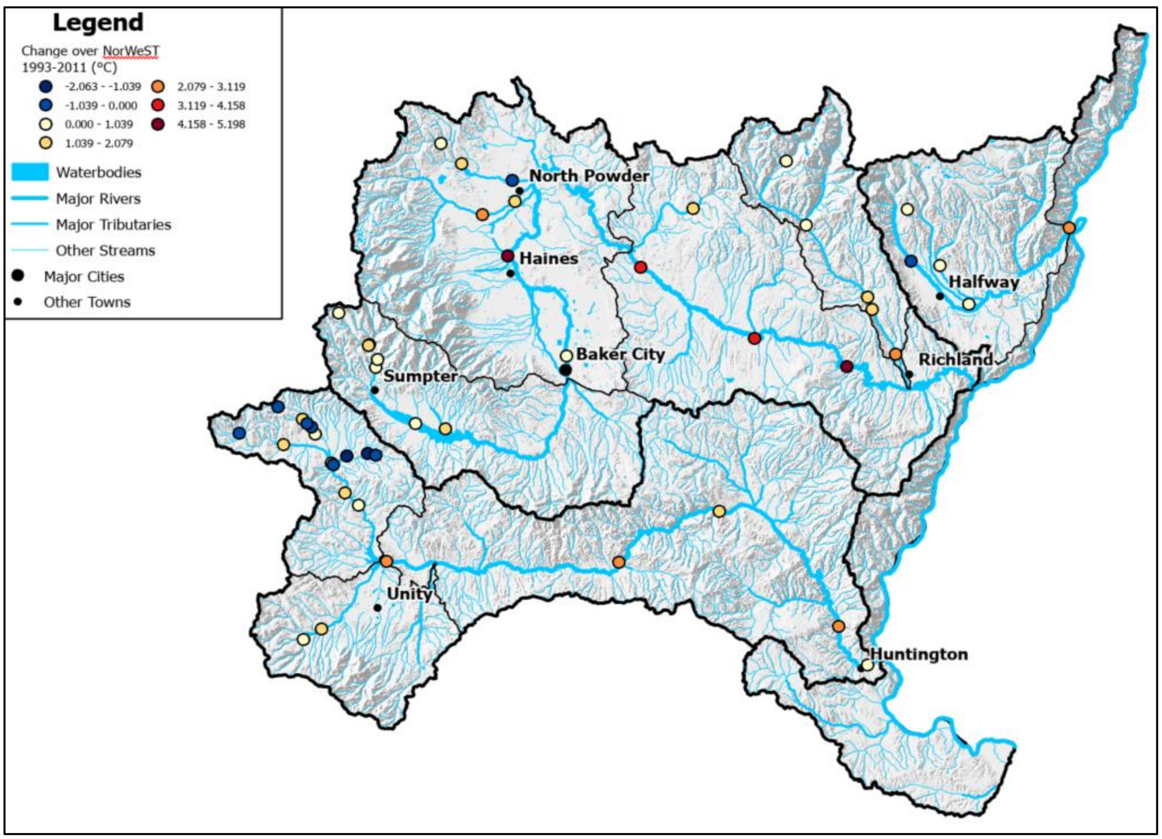


Map 18. Powder Basin cold-water habitat monitoring sites where maximum temperatures exceeded 12 °C.



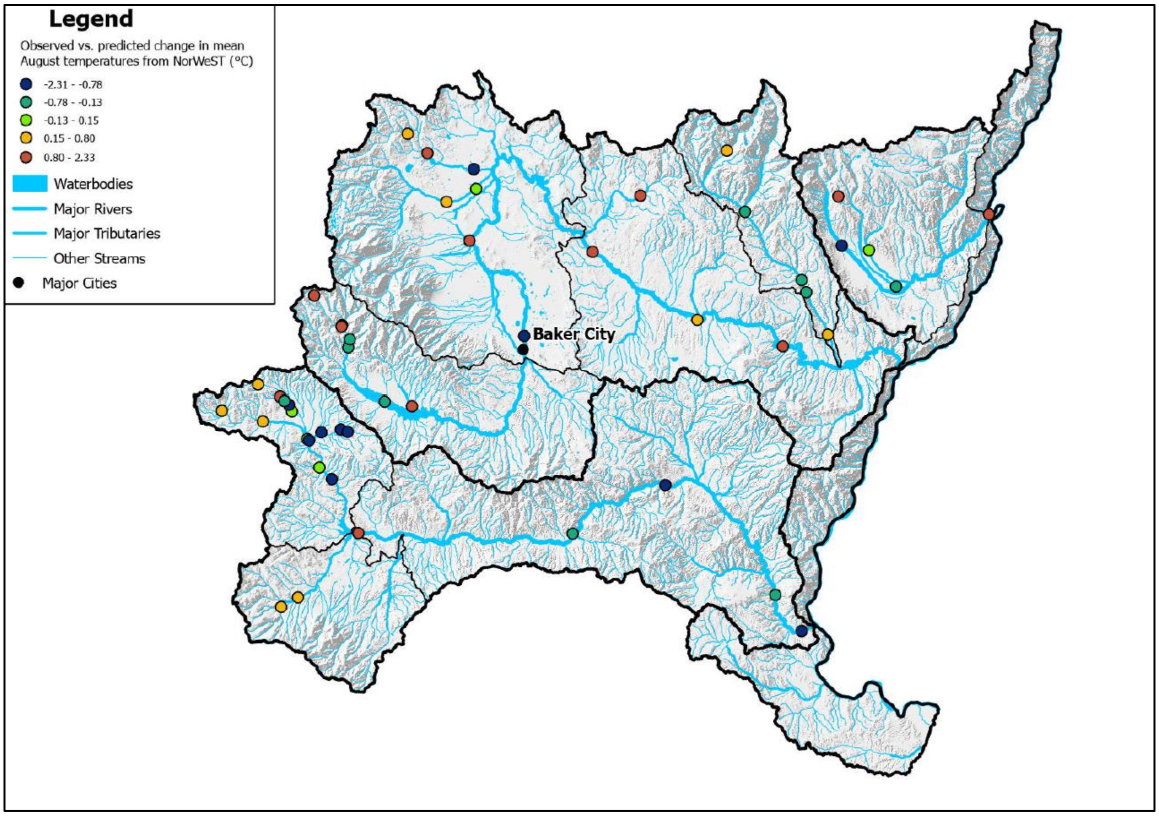
Regarding mean August stream temperature changes over the 1993-2011 NorWeST modelled averages, 39 sites had higher observed 2022-24 mean August stream temperatures than NorWeST average, while 10 had lower temperatures, mostly sites in the NFBR watershed (Map 19). On average, sites had mean August stream temperatures of 1.15 °C higher than the NorWest model predictions. This difference between observed mean August temperatures to those from the NorWeST model was significant, with a 95% confidence interval of the true mean difference ranging from 0.74 to 1.56. The increase in observed temperatures was also larger than the RMSPE of 1.10 °C and MAPE of 0.72 °C from the NorWeST model predictions.

Map 19. Change in observed mean August stream temperatures from the 2022-2024 monitoring period compared to estimates for mean August stream temperatures from the NorWeST stream temperature model. Sites with warmer stream temperatures than the NorWeST model colored in yellow to red, while sites with cooler than expected temperatures than the NorWeST model in blue.



Stream temperature increases were moderately correlated to the log of stream basin size (R^2 = 0.462), with larger stream basin sizes associated with larger temperature increases over the NorWeST estimates. Sites with smaller than expected increases were found on Trout Creek, Camp Creek, and the lower Burnt River, while sites with larger than expected increases were seen at Powder @ Haines, Powder near Richland, Deer Creek, and Silver Creek. The distribution of sites with higher than and lower than expected increases in observed temperatures compared to the NorWeST model varied widely and did not follow any patterns related to elevation, basin size, or watershed.

Map 20. Observed vs expected estimates of difference in observed mean August Stream temperatures over the 2022-24 period to NorWest 1993-2011 predictions from correlations with basin size (in km^2). Sites with negative values indicate that increases in observed temperatures over the NorWeST predictions was smaller than expected based off of basin size (blue) while sites with positive values indicate that increases in observed temperatures over the NorWeST predictions were higher than expected based off of basin size (red).



Moving forward, monitoring should assess the validity of the model outputs and if the increase in observed temperatures from the 1993-2011 NorWeST predictions are the result model artifacts and variability or reflect real increases in stream temperature. There is also a need to understand what the key drivers of these potential increases in stream temperatures might be, and particularly what factors could be resulting in lower temperatures at the NFBR sites. Factors like increased air temperatures and shifts from snowmelt to rain dominated precipitation are a likely source of warming, but larger groundwater contributions could be factor for the lower observed temperatures in the NFBR, particularly the middle reaches of Camp, Trout Creek and Snow Creek.

Patterns in Dissolved Oxygen Logger Monitoring

Oxygen concentrations were above the 8 mg/L intergravel minimum concentration but were generally below the 11 mg/L spawning standard at most sites monitored with dissolved oxygen loggers. One possible reason for why oxygen concentrations were consistently below the spawning standard are the high elevation of the dissolved oxygen logger sites, which were on average 4,262 feet (or 1,299 meters) above sea level. At these elevations, temperatures would need to be below 4.7 °C for the water to hold 11 mg/L of oxygen at saturation, compared to 6.2 °C at Baker City elevation (~3450 feet) and 11.1 ° at sea level. Temperatures like these generally only occur during the early spring snow melt period or very late in the fall, limiting the time when these oxygen concentrations can be present.

Sites with oxygen concentrations below the 8 mg/L intergravel minimum included Cracker.20G.2, Meadow.15J.3, NFBR.83C.1.4, NFBR.83E.2, Powder DS Keating, and SFBR.21G.2, although most of these sites had less than 1% of records below this standard. Among sites with observed Redband Trout or Bull Trout spawning, only Meadow.15J.3 had oxygen concentrations below 8 mg/L for 14% or more of records. Oxygen concentrations were generally similar between seasons (9.70 mg/L for Fall vs. 9.61 for Spring).

Almost all sites showed negative correlations between stream temperatures and oxygen concentrations, with correlations ranging between -0.176 and -0.269 for fall sites and -0.358 and -0.181 for spring sites (Appendix D.). Sites showing greater sensitivity to temperature were Wolf.18C.1, Wolf @ Inlet, and Cracker.10G.2 in the fall, while sites showing less sensitivity to temperatures were Meadow.15J.3, SFBR.21G.2, NFBR.21E.2 and Powder @ Huckleberry. These patterns could be related to higher groundwater contributions at the lower sensitivity sites, particularly for Meadow.15J.3. The only sites showing positive correlations between stream temperature and oxygen concentrations were Powder DS Mason Dam and Powder DS of Keating. With the site below Mason Dam, impacts of the reservoir on oxygen and temperature are the likely reason for the discrepancy, while eutrophication is a more likely factor downstream of Keating Valley.

Patterns in Water Quality Index Scores

Water Quality Index Scores were variable throughout the sampling period and between sites (Map 21). In general, WQI Scores were highest at the higher elevation sites and locations with smaller drainage basins, while lower WQI scores were seen at sites on the mainstem of the Powder River and Burnt River lower in the watershed. Temperature and dissolved oxygen were the most important parameters resulting in low WQI scores among observations with WQI scores within "Poor" or "Very Poor" classes. For these classes, average temperature sub-index scores were 45.03 and average dissolved oxygen sub-index scores were 59.82. Sub-Index scores also differed over time, with turbidity sub-index scores lowest in the spring and highest in the summer and fall, while temperature sub-index scores were highest in the spring and fall and lowest during the summer (Figure 31).

WQI scores from 2022 to 2024 found higher WQI scores in 2022 and lower scores in 2024, although these results patterns varied by watershed, particularly Eagle Creek due to the Hudson Creek landslide and in the NFBR watershed due to differing site selection over the period. Trends in WQI scores since 2013 found lower scores in 2015, 2017, and 2018, years where drought conditions were more widespread, particularly in the summer. Higher WQI scores were found in 2019 and 2020, although a smaller selection of sites likely biased mean scores for these years.

Map 21. WQI Status based on mean yearly WQI scores over the 2022-2024 monitoring period.

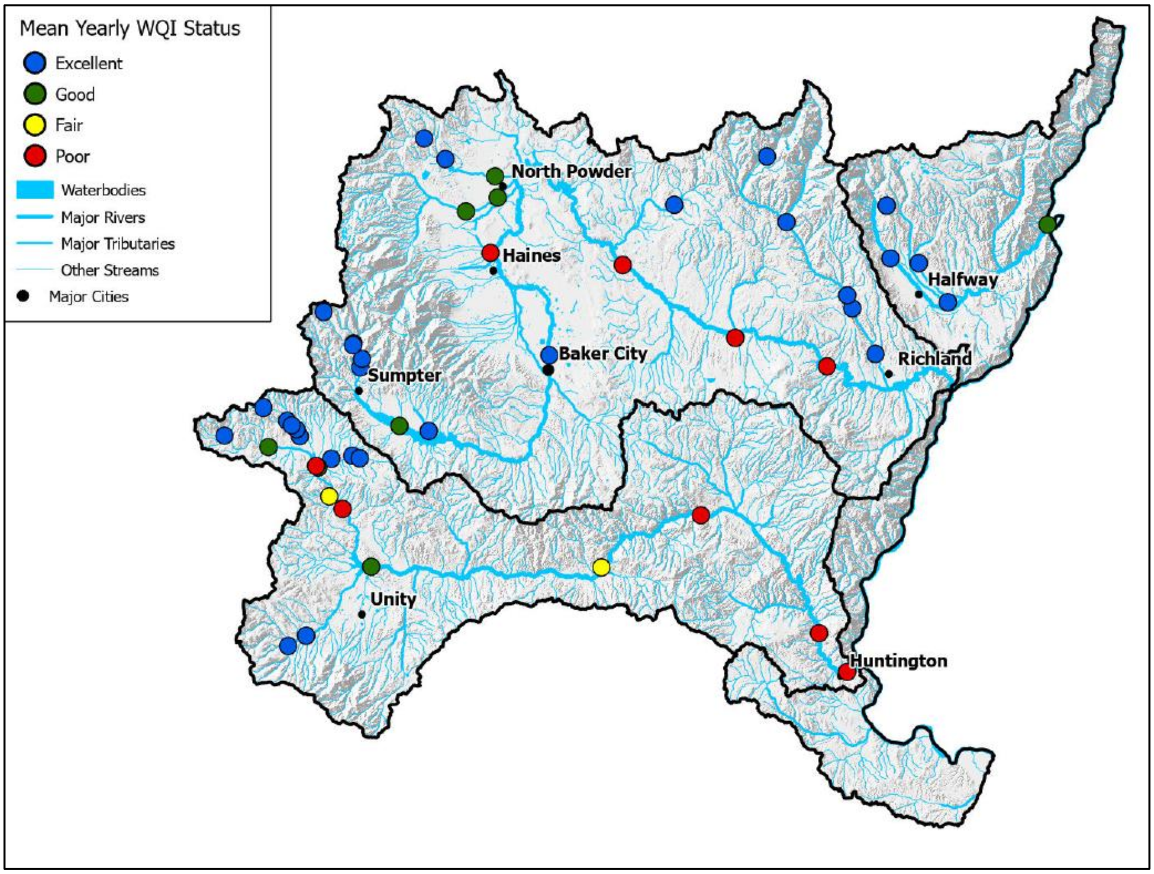
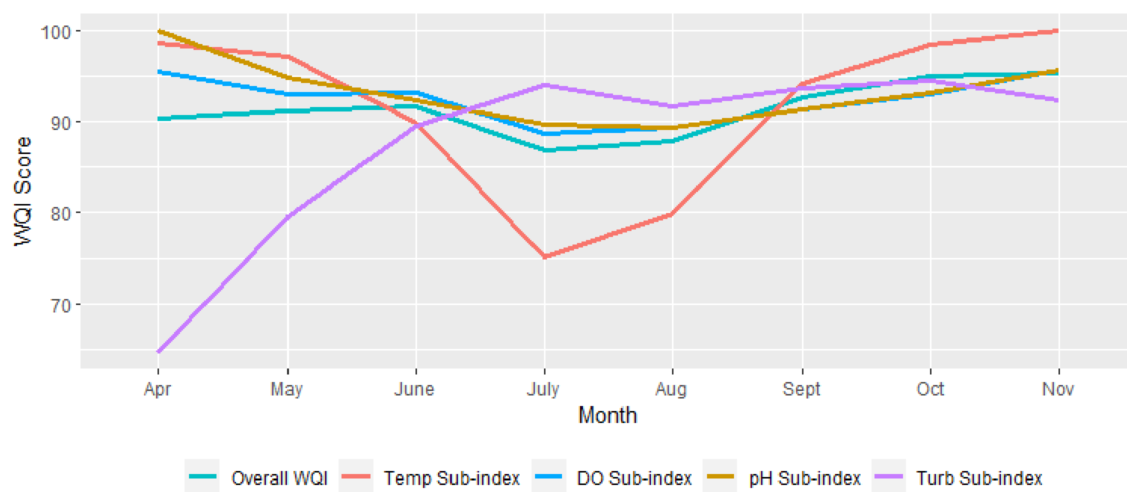
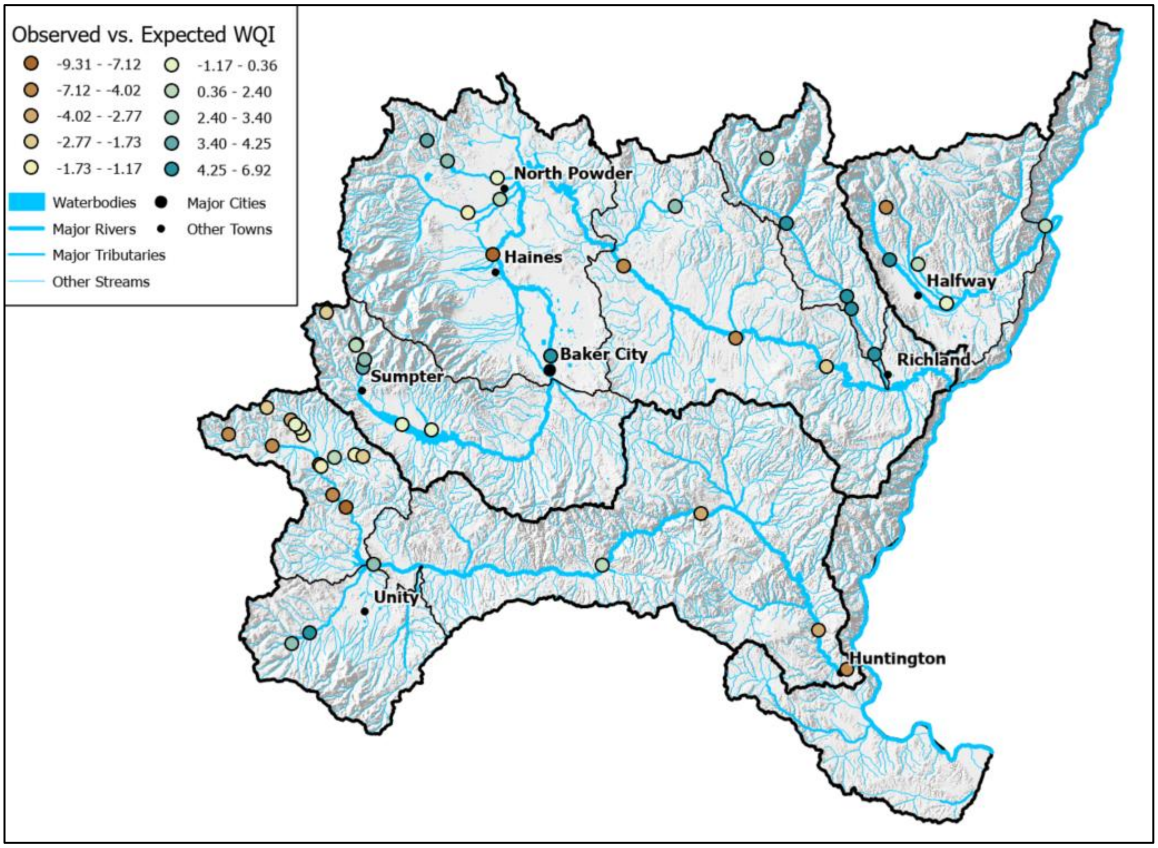


Figure 31. Average monthly WQI scores and Sub-index scores for temperature, dissolved oxygen (DO), pH, and turbidity for the 2022-2024 period.



A negative correlation between WQI scores over the 2022-2024 period and basin size was observed (R^2 = 0.61), with sites having higher WQI scores found higher in the watershed. Sites having higher WQI scores than expected given upstream basin size were Powder @ Kirkway, Pine @ Holbrook, SFBR.21E.1, and sites on Eagle Creek. Other sites with higher than expected WQI scores included Wolf @ Inlet, Big.L107, and the sites on Cracker Creek. In contrast, sites with lower WQI scores than expected based off of basin size were Powder near Haines, Meadow.15J.3, and locations on the NFBR from the lower end of Whitney Valley and points downstream. Other sites with lower-than-expected WQI scores included Snow.83G.1, Camp.83F.3, and sites in the lower Burnt and Powder Rivers (Map 22).

Map 22. Observed vs. Expected WQI values based on correlation with basin size (in km²). Negative values indicate WQI values lower than expected based on upstream basin size (brown) while positive values indicate WQI values higher than expected based on upstream basin size (blue).



Lessons Learned

One of the biggest lessons learned from monitoring over the 2022-2024 period was the importance of high-quality data that is both precise and accurate. Overall, average accuracy index values were higher each year of 2022-24 period than the average over the 2013-2024 period, indicating fewer failed accuracy tests during pre- and post-trip accuracy checks (Figure 33). Comparisons between years showed that overall accuracy was lower in 2022 and 2024 and higher in 2023. Precision was roughly similar to previous years, with higher average precision index values in 2023 and lower values in 2022 and 2024 when compared to the 2013-2024 average (Figure 32). Given the importance of water quality to both the PBWC and its partners, improving quality data should always be a high priority. Increasing precision is a more important factor, with volunteer training and assistance from the monitoring coordinator being useful ways to increase sample precision in the field.

Figure 32. Precision of samples from 2013 to 2024

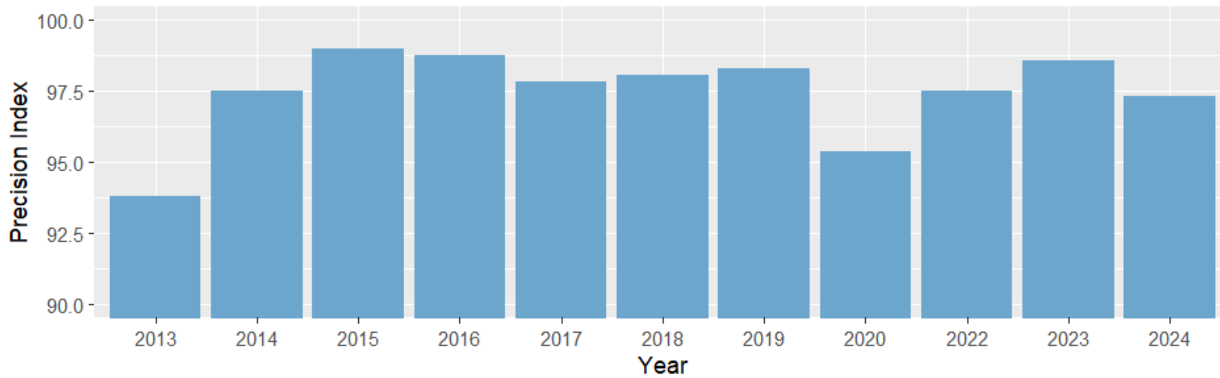
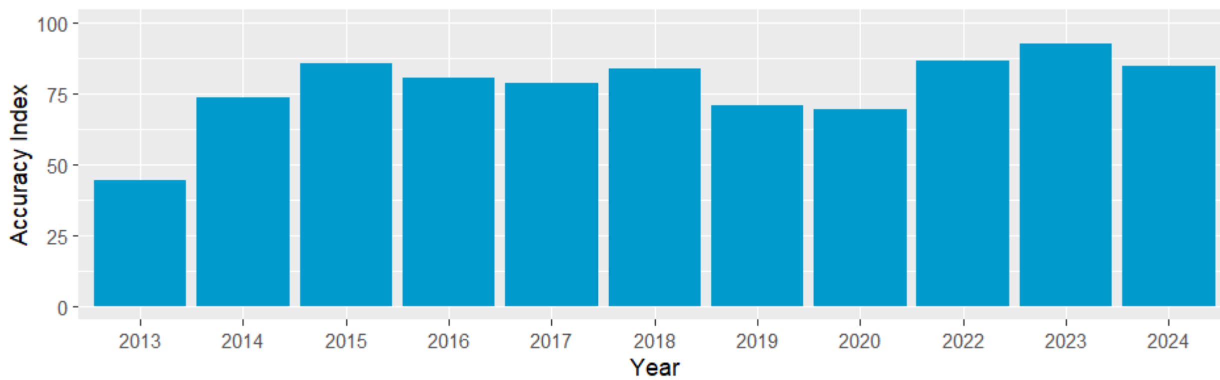


Figure 33. Accuracy of samples from 2013 to 2024



Overall temperature logger data accuracy was good and increased each year of monitoring. Temperature data quality was lowest in 2022, with 67.2% of records having "A" grade audits, while 20.0% of visits had "C" grade audits. Temperature data quality was much higher in 2023 and 2024, with 86.2% and 89.4% of data with "A" grades, compared to 4.5% and 4.2% with "C" grades, respectively. This was primarily due to a focus on improving audit methods, mostly by taking temperature audits closer to the logger location. Audit measurements were also taken closer to the time when the logger recorded temperatures, either on the hour or half hour, to reduce bias related to daily changes in temperature.

Overall dissolved oxygen logger accuracy was low during the first few year of monitoring, with an average oxygen concentration difference of 0.22 mg/L lower than field measurements during installation and 0.52 mg/L lower than the field measurements during removal. Overall Accuracy between the MiniDOT and U26 loggers differed little, -0.37 mg/L for the MiniDOT logger compared to -0.40 mg/L for the U26 loggers, but differences were more noticeable in 2023, where MiniDOT loggers had an accuracy of -0.75 mg/L. Improvements in MiniDOT logger accuracy were made in 2024 thanks to the recalibration feature added to logger software by PME (the MiniDOT manufacturer), dramatically improving accuracy to 0.08 mg/L. Continued use of this feature will be important moving forward with the use of dissolved oxygen loggers for future monitoring efforts.

Increasing data coverage, both spatial and temporal, is another important consideration considering the use of multiple linear regression to identify site, month, and yearly mean values for parameters. The monthly grab sample visits were mostly able to identify overall patterns, but likely hide large variability, especially oxygen and pH. Further sampling throughout different times of the day through randomly selecting sites might better identify variation in parameters at sites. Overall coverage for sites was also generally good, with most sites sampled at least once per month over the May – October period. Earlier sampling might be able to identify patterns in April and May, but results from June show most sites are unlikely to have water quality issues during this period, although some lower sites might show impairments.

Recommendations

One of the largest questions generated from this report is the status of water quality at other sites, primarily in upper watershed where Redband Trout and Bull Trout generally seek thermal refuge in the summer. Future monitoring efforts should be used to identify where water quality impairments occur within these upper watershed reaches to focus habitat improvement and restoration efforts. Important reaches to consider for this monitoring are the upper NFBR watershed, Deer Creek, Pine Creek. There is also a need for a greater number of monitoring sites in lower watersheds, particularly in Baker Valley, Keating Valley, and Burnt River between Hereford and Clarks Cr Bridge. Outreach to interested landowners in these valleys might be useful in both improving coverage within these reaches as well increasing knowledge of and cooperation with the PBWC regarding our monitoring activities.

Another question relates to the sources of impairments within the watershed, particularly for *E. coli* and phosphorus. One method the PBWC is currently undertaking to identify the sources of impairment is the use of eDNA measurements at sites in the Burnt River to use as a comparison to *E. coli* concentrations. Such samples have been useful in tying the number of primers from human sources to bacterial concentrations in urban watersheds (**Edge et al. 2021**) and might be a useful method to identify both short duration pulses in high *E. coli* concentration like those observed at Lime in October 2023, and the more persistent issues observed at Burnt @ Clarks Cr Bridge.

To identify potential sources of nutrients, along with other grab sample and temperature impairments, monitoring can use correlations to land use, geology, air temperature, precipitation, or other factors. Several studies, including those looking at temperature and nutrients on a basin wide scale, have used models known as Spatial Statistical Networks, or SSNs, to account for the autocorrelation among sites regarding water quality parameters (**Issak et al. 2017, Wise et al. 2011, Isaak et al. 2014**). The end goal of any attempt at identifying sources should be used to aid in the mitigation, improvement, and restoration activities and to improve the effectiveness of these actions in addressing water quality impairments.

While monitoring at new sites would be useful in determining water quality impairments and identifying potential causes of impairments, there is still a need to continue monitoring at most sites to assess trends over time. The application of multiple linear regression models was useful in identifying trends over the 2022-2024 period but was less useful in determining longer term trends over the 2013-2024 given the differing composition of sites by sample year and limited coverage of sampling by month. The use of better statistical tools might be able to assess these trends after taking into account the patterns within sites and years to better assess long term trend in water quality. One method could be the use of Linear Mixed Models's, LLMs, which can use this variation to increase the accuracy of mean estimates. Future analysis of monitoring should consider the use of LMMs to better identify trends in parameters and WQI and improve the accuracy of estimates.

Furthermore, there is a need to assess successes and failures of the PBWC's restoration actions, both in the NFBR and in other watersheds. Restoration actions such as the Low-Tech Process Based restoration in the NFBR and more intensive Stage-Zero actions planned have been found to decrease maximum stream temperatures, increase habitat complexity, increase salmonid survival and densities, increase groundwater contributions to streamflow, and improve the amount of vegetation productivity (**Dittenbrenner et al. 2022, Bouwes et al. 2016, Silverman et al. 2019, Bosworth et al. 2025**). Ensuring that PBWC restoration projects achieve these results, and altering project designs if they don't is already the focus of planned project effectiveness monitoring, but including water quality monitoring within these efforts will be an important component of identifying longer-term trends in water quality and applying them to future project areas.

Related to restoration actions are the impact of beaver activity on water quality. Results from Trout Creek indicate that while beaver dams can lower maximum stream temperatures, they also result in lower concentration of dissolved oxygen, often below the DEQ standards. More monitoring within beaver dams is currently taking place at other sites in the NFBR watershed to assess their impacts and benefits and/or impairments. Preliminary results seem to show that the higher flow of the NFBR, as well as greater groundwater contributions, still result in cooler stream temperatures while leading to oxygen concentrations over 6.5 mg/L, but additional data is needed to see if these results continue during other conditions.

Given the degree of water quality impairments within the basin, particularly in lower watershed sites, a comprehensive method of identifying locations where restoration actions might be beneficial is needed. The methods detailed in the report, including the observed/expected WQI method mentioned above, might be useful to identify these reaches. Potential locations where improvement or restoration actions might improve water quality include lower watershed sites and those in the NFBR, where WQI was low compared to expected values. It might also be useful to identify depositional meadow systems where LTPBR restoration is most likely to succeed.

Literature Cited

Abdul-Aziz, O. I., and Gebreslase, A. K. (2023). Emergent scaling of dissolved oxygen (DO) in freshwater streams across contiguous USA. Water Resources Research, 59(2), e2022WR032114.

Al-Chokhachy, R., Letcher, B. H., Muhlfeld, C. C., Dunham, J. B., Cline, T., Hitt, N. P., Roberts, J. J., and Schmetterling, D. (2022). Stream size, temperature, and density explain body sizes of freshwater salmonids across a range of climate conditions. *Canadian Journal of Fisheries and Aquatic Sciences*, *79*(10), 1729-1744.

Bash, J., Berman, C. H., and Bolton, S. (2001). Effects of turbidity and suspended solids on salmonids. University of Washington Water Center.

Bosworth, L., Smith, R. M., Culbertson, A., Jamison, L. R., Burnett, P., and Brooks, P. D. (2025). Streamflow and Groundwater Response to Stream Restoration Using Beaver Dam Analogues in a Semi-Arid Perennial Stream. *JAWRA Journal of the American Water Resources Association*, *61*(1), e13254.

Bouwes, N., Weber, N., Jordan, C. E., Saunders, W. C., Tattam, I. A., Volk, C., Wheaton, J. M. and Pollock, M. M. (2016). Ecosystem experiment reveals benefits of natural and simulated beaver dams to a threatened population of steelhead (Oncorhynchus mykiss). *Scientific reports*, *6*(1), 28581.

Brendel, C., and Soupir, M. L. (2017). Relating watershed characteristics to elevated stream *Escherichia coli* levels in agriculturally dominated landscapes: An Iowa case study. Water, 9(3), 154.

Brown, L. E., Milner, A. M., and Hannah, D. M. (2007). Groundwater influence on alpine stream ecosystems. Freshwater Biology, 52(5), 878-890.

Buchanan, D. V., Hanson, M. L., and Hooton, R. M. (1997). Status of Oregon's Bull Trout. Distribution, Life History, Limiting Factors, Management Considerations, and Status.

Chen, J., and Chang, H. (2019). Dynamics of wet-season turbidity in relation to precipitation, discharge, and land cover in three urbanizing watersheds, Oregon. River Research and Applications, 35(7), 892-904.

Cude, C. G. (2001). Oregon water quality index a tool for evaluating water quality management effectiveness 1. JAWRA Journal of the American Water Resources Association, 37(1), 125-137.

Dittbrenner, B. J., Schilling, J. W., Torgersen, C. E., and Lawler, J. J. (2022). Relocated beaver can increase water storage and decrease stream temperature in headwater streams. *Ecosphere*, *13*(7), e4168.

Dodds, W. K., and Smith, V. H. (2016). Nitrogen, phosphorus, and eutrophication in streams. Inland Waters, 6(2), 155-164.

Clements, W. H., and Kotalik, C. (2016). Effects of major ions on natural benthic communities: an experimental assessment of the US Environmental Protection Agency aquatic life benchmark for conductivity. Freshwater Science, 35(1), 126-138.

Dwire, K. A., Mellmann-Brown, S., and Gurrieri, J. T. (2018). Potential effects of climate change on riparian areas, wetlands, and groundwater-dependent ecosystems in the Blue Mountains, Oregon, USA. Climate Services, 10, 44-52.

Edge, T. A., Boyd, R. J., Shum, P., and Thomas, J. L. (2021). Microbial source tracking to identify fecal sources contaminating the Toronto Harbour and Don River watershed in wet and dry weather. Journal of Great Lakes Research, 47(2), 366-377.

Feldhaus, J. W. (2006). Heat shock proteins and physiological stress in redband rainbow trout (Oncorhynchus mykiss gairdneri) in the South Fork John Day River, Oregon.

Ferns, M.L., H.J. Streck, and J.D. McClaughry (2017). Field-trip guide to Columbia River flood basalts, associated rhyolites, and diverse post-plume volcanism in eastern Oregon: U.S. Geological Survey Scientific Investigations Report 2017–5022–O, 71 p., https://doi.org/10.3133/sir20175022O.

Franczyk, J.J., I.P. Madin, C.J.M. Duda, and J.D. McClaughry (2023). Oregon Geologic Data Compilation, release 7 (OGDC-7)

Gingerich, S.B., D.E. Boschmann, G.H. Grondin, and H.J. Schibel (2024). Groundwater model of the Harney Basin, southeastern Oregon (No. 2024-5017). US Geological Survey.

Howell, P. J. (2018). Changes in native bull trout and non-native brook trout distributions in the upper Powder River basin after 20 years, relationships to water temperature and implications of climate change. *Ecology of Freshwater Fish*, *27*(3), 710-719.

InciWeb (2024). Durkee Fire. Source: https://inciweb.wildfire.gov/incident-information/orvad-durkee-fire

Isaak, D. J., Peterson, E. E., Ver Hoef, J. M., Wenger, S. J., Falke, J. A., Torgersen, C. E., Sowder, C., Steel, E. A., Fortin, M. J., Jordan, C. E and Monestiez, P. (2014). Applications of spatial statistical network models to stream data. *Wiley Interdisciplinary Reviews: Water*, *1*(3), 277-294.

Isaak, D. J., Wenger, S. J., Peterson, E. E., Ver Hoef, J. M., Nagel, D. E., Luce, C. H., ... and Parkes-Payne, S. (2017). The NorWeST summer stream temperature model and scenarios for the western US: A crowd-sourced database and new geospatial tools foster a user community and predict broad climate warming of rivers and streams. Water Resources Research, 53(11), 9181-9205.

Isaak, D. J., Luce, C. H., Horan, D. L., Chandler, G. L., Wollrab, S. P., Dubois, W. B., and Nagel, D. E. (2020). Thermal regimes of perennial rivers and streams in the western United States. JAWRA Journal of the American Water Resources Association, 56(5), 842-867.

John, A., Michalak, J. L., Svancara, L. K., Randels, C., and Lawler, J. J. (2026). Identifying climate-change refugia for species management and conservation in the Pacific Northwest. Conservation Science and Practice, 8(1), e70174.

Klein, R. D., Lewis, J., and Buffleben, M. S. (2012). Logging and turbidity in the coastal watersheds of northern California. Geomorphology, 139, 136-144.

Kney, A. D., and Brandes, D. (2007). A graphical screening method for assessing stream water quality using specific conductivity and alkalinity data. Journal of environmental management, 82(4), 519-528.

Kraynick, R.G., and H. H. Shevener (1981). An Economic Study of Three Small Watershed Projects in Midlife. Department of Agriculture, Oregon State University, Corvallis, OR.

Leach, J. A., Kelleher, C., Kurylyk, B. L., Moore, R. D., and Neilson, B. T. (2023). A primer on stream temperature processes. Wiley Interdisciplinary Reviews: Water, 10(4), e1643.

Lee, R. M., and Rinne, J. N. (1980). Critical thermal maxima of five trout species in the southwestern United States. Transactions of the American Fisheries Society, 109(6), 632-635.

Mayer, M., and Chang, H. (2025). Thermal Regimes of Beaver-Impacted Stream Reaches in the Tualatin River Basin, Oregon, United States. River Research and Applications, 41(9), 1904-1924.

Miller, S. A., Gunckel, S., Jacobs, S., and Warren, D. R. (2014). Sympatric relationship between redband trout and non-native brook trout in the Southeastern Oregon Great Basin. *Environmental biology of fishes*, *97*(4), 357-369.

Mueller M, Rausch-Phung EA, Tainter CR. (2025). Escherichia coli Infection. [Updated 2025 Dec 14]. In: StatPearls [Internet]. Treasure Island (FL): StatPearls Publishing; Available from: https://www.ncbi.nlm.nih.gov/books/NBK564298/

Müller, B., L.D. Bryant, A. Matzinger, and A. Wüest. (2012). “Hypolimnetic Oxygen Depletion in Eutrophic Lakes.” Environmental Science and Technology 46 (18): 9964–71.

Natural Resource Conservation Service (2006a). “Brownlee Reservoir – 17050201 8-Digit Hydrologic Unit Profile”

Natural Resource Conservation Service (2006b). “Burnt River – 17050202 8-Digit Hydrologic Unit Profile”

Natural Resource Conservation Service (2006c). “Powder River – 17050203 8-Digit Hydrologic Unit Profile”

Northwest Interagency Coordination Center (2016). 9/01/2016 Rail Fire Update. Source: https://nwccinfo.blogspot.com/2016/09/9012016-rail-fire-update.html

Northwest Power and Conservation Council (2004a). " Oregon Side of the Lower Middle Snake Subbasin Plan."

Northwest Power and Conservation Council (2004b). “Powder River Subbasin Plan”

Northwest Power and Conservation Council (2004c). “Burnt River Subbasin Plan”

Olson, J. R., and Cormier, S. M. (2019). Modeling spatial and temporal variation in natural background specific conductivity. Environmental science and technology, 53(8), 4316-4325.

Oregon Department of Environmental Quality (2016). Water Quality Standards: Beneficial uses, Policies, and Criteria for Oregon, 340-041-0009: Bacteria, source: https://secure.sos.state.or.us/oard/viewSingleRule.action?ruleVrsnRsn=68695

Oregon Department of Environmental Quality (1996a). Water Quality Standards: Beneficial uses, Policies, and Criteria for Oregon, 340-041-0016 Dissolved Oxygen, source: https://secure.sos.state.or.us/oard/viewSingleRule.action?ruleVrsnRsn=310035

Oregon Department of Environmental Quality (1996b). Water Quality Standards: Beneficial uses, Policies, and Criteria for Oregon, 340-041-0036, source: https://secure.sos.state.or.us/oard/viewSingleRule.action?ruleVrsnRsn=68758

Oregon Department of Environmental Quality (2003). Basin-Specific Criteria (Powder/Burnt): Water Quality Standards: Beneficial uses, Policies, and Criteria for Oregon, 340-041-0265, pH, source: https://secure.sos.state.or.us/oard/viewSingleRule.action?ruleVrsnRsn=68897

Oregon Department of Environmental Quality (2008). Temperature Water Quality Standard Implementation – A DEQ Internal Management Directive.

Oregon Watershed Enhancement Board (2023) Regions 5 and 6 Observed and Projected Climate Changes.

Oregon Department of Fish and Wildlife (2005). Oregon Native Fish Status Report Volume II Assessment Methods & Population Results.

Orr, W.N., and E.L. Orr (1996). Geology of the Pacific Northwest. Second Edition. Waveland Press.

Poole, G. C., and Berman, C. H. (2001). An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental management, 27(6), 787-802.

Powder Basin Watershed Council (2018). Powder Basin Water Quality Report 2013-2017. Baker City, Oregon.

Powder Basin Watershed Council (2019). 2018 Powder Basin Macroinvertebrate Survey. Baker City, Oregon.

Powder Basin Watershed Council (2025). Powder Basin Well Logs Report. Baker City, Oregon.

Rey, D. M., Hare, D. K., Fair, J. H., and Briggs, M. A. (2024). Diel temperature signals track seasonal shifts in localized groundwater contributions to headwater streamflow generation at network scale. Journal of Hydrology, 639, 131528.

Rozhkova-Timina, I. O., Popkov, V. K., Mitchell, P. J., and Kirpotin, S. N. (2018). Beavers as ecosystem engineers–a review of their positive and negative effects. In *IOP Conference Series: Earth and Environmental Science* (Vol. 201, No. 1, p. 012015). IOP Publishing.

Shock, C. C., Pratt, K., and Jones, R. (2001). Phosphorus Content of the Malheur River. Malheur County Soil, and Water Conservation District.

Silverman, N. L., Allred, B. W., Donnelly, J. P., Chapman, T. B., Maestas, J. D., Wheaton, J. M., J. White and Naugle, D. E. (2019). Low-tech riparian and wet meadow restoration increases vegetation productivity and resilience across semiarid rangelands. *Restoration Ecology*, *27*(2), 269-278.

Smith, V. H., Tilman, G. D., and Nekola, J. C. (1999). Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environmental pollution*, *100*(1-3), 179-196.

Stevenson, J. R., Dunham, J. B., Wondzell, S. M., and Taylor, J. (2022). Dammed water quality—Longitudinal stream responses below beaver ponds in the Umpqua River Basin, Oregon. *Ecohydrology*, *15*(4), e2430.

Sullivan, K., Martin, D. J., Cardwell, R. D., Toll, J. E., and Duke, S. (2000). An analysis of the effects of temperature on salmonids of the Pacific Northwest with implications for selecting temperature criteria. Sustainable Ecosystems Institute, Portland, OR.

U.S. Bureau of Reclamation (1992). Unity Reservoir 1991 Sedimentation Survey. Denver, CO.

U.S. Bureau of Reclamation (2026). Unity Reservoir. Source: https://www.usbr.gov/projects/index.php?id=300

U.S. Environmental Protection Agency (2022). Climate Indicator April Snowpack, 1955-2021. data accessed from: https://www.arcgis.com/home/item.html?id=d02cf542655245e2b18480680f9779d9

U.S. Environmental Protection Agency (2025). Dissolved Oxygen. Source: https://www.epa.gov/caddis/dissolved-oxygen

U.S. Environmental Protection Agency (2026). pH. Source: https://www.epa.gov/caddis/ph

U.S. Fish and Wildlife Service. (2015). Recovery plan for the coterminous United States population of bull trout (Salvelinus confluentus). Portland, OR.

U.S. Geological Survey (2024). Annual NLCD Collection 1 Science Products (ver. 1.1, June 2025): U.S. Geological Survey data release, https://doi.org/10.5066/P94UXNTS.

Wilkison, R., and Trainer, J. (2017). ESA Section 10 Federal Fish and Wildlife Permit 2016 Annual Report & Results Summary from 2012-2016 Pine Creek Bull Trout Assessment. Permit #: TE-72143A-2

Winkowski, J. J., Olden, J. D., and Brown, S. (2024). Integrating spatial stream network models and environmental DNA to estimate current and future distributions of nonnative Smallmouth Bass. Transactions of the American Fisheries Society, 153(2), 180-199.

Wise, D. R., and Johnson, H. M. (2011). Surface-Water Nutrient Conditions and Sources in the United States Pacific Northwest 1. JAWRA Journal of the American Water Resources Association, 47(5), 1110-1135.

Wise, D. R., M. L. Zuroske, K. D. Carpenter, and R. L. Kiesling. (2009). Assessment of Eutrophication in the Lower Yakima River Basin, Washington, 2004–07. U.S. Geological Survey Scientific Investigations Report 2009–5078

Wu, R. S. (2002). Hypoxia: from molecular responses to ecosystem responses. Marine pollution bulletin, 45(1-12), 35-45.

Wu, J., Rees, P., and Dorner, S. (2011). Variability of *E. coli* density and sources in an urban watershed. Journal of water and health, 9(1), 94-106.

Appendices

Appendix A. Sample Grade Criteria for water quality parameters measured in the field

Data Quality Matrix (DEQ04-LAB-0003-QAG) v5.0

Data Quality Level	***Quality Assurance Plan***	***Water Temperature Methods***	***pH Methods***	***Dissolved Oxygen Methods***	***Turbidity Methods***	***Conductivity Methods***	***Bacteria Methods***	***Data Uses***
A	Approved QAPP	Thermometer Accuracy checked with NIST standards A < ± 0.5°C *P < ± 0.5°C*	Calibrated pH electrode *A < ± 0.2 S.U.* *P < ± 0.3 S.U.*	Winkler titration A < ± 0.2 mg/L P < ± 0.3 mg/L Calibrated oxygen meter/LDO A < + 0.4 mg/L > -0.3 mg/L *P <± 0.3 mg/L*	Nephelometric Turbidity meter *A < ± 10% Standard value* *P < ± 20% (+ 3 NTU if NTU < 20)*	Meter with temp correction to 25°C *A < ± 7% of standard value* *P < ± 10%*	DEQ Approved Methods *Absolute difference between log-transformed values* *P < 0.6 log*	Regulatory, permitting, compliance (e.g., 303(d) and 305(b) assessments)
B	Minimum Data Acceptance Criteria Met	Thermometer Accuracy checked with NIST standards *A < ± 1.0°C* *P < ± 2.0°C*	Any Method *A <±0.5 S.U.* *P <±0.5 S.U.*	Winkler titration or Calibrated oxygen meter/LDO *A < ± 1 mg/L* *P < ± 1 mg/L*	Any Method *A < ± 30%* *P < ± 30%*	Meter with temp correction to 25°C *A < ± 10% of standard value* *P < ± 15%*	DEQ Approved Methods *Absolute difference between log-transformed values* *P < 0.8 log*	Regulatory, permitting, compliance (e.g., 303(d) and 305(b) assessments) with professional judgment
C		A > ± 1,0°C P > ± 2.0°C	A > ± 0.5 S.U. P > ± 0.5 S.U.	A > ± 2 mg/L P > ± 2 mg/L	A > 30% P > 30%	A > ± 10% P > ± 15%	Absolute difference between log-transformed values P > 0.8 log	Not used for 303(d) and 305(b) assessments *Based on project manager judgment, the data may be Voided with a DQL of D.*
D		Missing or voided data	*Missing or voided data*	*Missing or voided data*	*Missing or voided data*	*Missing or voided data*	*Missing or voided data*	*Missing or voided data*
E	*No QAPP Provided*	No precision or accuracy checks available	Any Method No precision or accuracy checks available	Any Method *No precision or accuracy checks available*	Any Method *No precision or accuracy checks available*	Any Method *No precision or accuracy checks available*	Any Method *No precision or accuracy checks available*	Any Method *No precision or accuracy checks available*
F	*See definitions table*	*See definitions table*	*See definitions table*	*See definitions table*	*See definitions table*	*See definitions table*	*See definitions table*	*See definitions table*

Appendix B. PBWC Grab sample collection form

Field Data Collection QA/QC Sheet

Field Instrument Infromation	Equipment Set:

Range for each Parameter the Equip. must be in for accuracy	
DO	±.3
Turbidity	Below 20 ±1 above 20 10% (ie 5 for 50)
pH	±.2
Conductivity	7% of standard

If the equipment is out of range for a parameter, please call Justin (541) 523-7288

Site Information			
Date			
Site ID			
Site Description			
Sample Collector			
Start Time		Volunteer Hours/Miles	

Dissolved Oxygen:

STD Type	Time	°C	mmhg	mg/L	% Sat	Theor(mg/L)	Diff
Air/Water							
Air/Water							
Air/Water							
Air/Water							
Air/Water							
Air/Water							

Turbidity:

Time	Std ID	Theor.	Obs.	Diff.
	DI water			
	DI water			

pH (Be sure and switch the temp. cable):

Time	Std. ID	°C	Theor.	Obs.	Diff.

Conductivity:

Time	Std. ID	°C	Theor.	Obs.	Diff.

Comments:

Field Data

General Site Description	Site ID number	Bottle number	Sample number	Time	Temp °C (X.X)	Cond µS/cm (X)	pH SU (X.X)	DO mg/L (X.X)	DO %Sat (X)	Turb NTU (X)	Comments

Digital Thermometer Data

Site ID number	Time	Air Temp. °C	Water Temp. °C	Logger Depth cm	Comments

Appendix C. The PBWC Logger Installation and Removal Form for temperature and dissolved oxygen loggers.

Logger Installation/Removal Form

Site Name: ______________________ **Site ID:** ______________________

Lat: ______________________ **Long:** ______________________

Install

Date: ______________________ **Time:** ______________________

Logger Type: Temperature Dissolved Oxygen **Logger Serial #:** ______________________

Bank: L M R

Logger Depth (cm): ______________ **Temperature (˚C):** ______________

Comments: ____

____

____

____

____

____

Removal

Date: ______________________ **Time:** ______________________

Logger Depth (cm): ______________ **Temperature (˚C):** ______________

Comments: ____

____

____

____

____

____

Photos (Description of photo + photo #'s): ______________________________

____

____

____

____

Appendix D. Modeled dissolved oxygen-temperature correlations for data from continuous dissolved oxygen loggers. Season (Fall for Bull Trout, Spring for redband trout), y-intercept (in mg/L), difference in oxygen concentrations from 2022 to 2023 (in mg/L), difference in oxygen concentrations from 2022 to 2024 (in mg/L), mean oxygen response to temperature (in mg/L per 1 °C temperature increase), standard deviation of oxygen response to temperature, and R^2 value for model fit.

Site Name	Season	Intercept	2022-2023 Diff	2022-2024 Diff	Temp Estimate	Temp SD	R^2
East Pine.15H.5	Fall	11.72	0.20	0.26	-0.246	0.000	0.97
Clear.15J.3.5	Fall	11.42	0.14	0.09	-0.240	0.001	0.94
Meadow.15J.3	Fall	9.75	0.14	-0.07	-0.176	0.002	0.47
Wolf.18C.1	Fall	11.77	-0.07	0.50	-0.269	0.001	0.88
Cracker.20G.2	Fall	11.70	-0.20	0.40	-0.250	0.001	0.97
Silver.20H.1	Fall	11.58	-0.13	0.19	-0.251	0.001	0.97
Lake.20D.2	Fall	11.44	0.22	0.25	-0.252	0.002	0.78
SFBR.21G.2	Spring	10.81		0.41	-0.184	0.003	0.71
NFBR.83C.1.4	Spring	11.30		0.59	-0.225	0.002	0.89
NFBR.83E.2	Spring	10.99		0.38	-0.181	0.002	0.77
Eagle.14D.1	Spring	12.09		0.18	-0.210	0.001	0.95
Eagle.14F.2.5	Spring	12.43		0.06	-0.225	0.000	0.98
Big.L107	Spring	11.86		0.14	-0.225	0.001	0.89
Goose.29C.2	Spring	11.27		0.30	-0.208	0.001	0.93
Wolf @ Inlet	Spring	12.15			-0.253	0.001	0.93
Cracker.20G.2	Spring	11.80		0.69	-0.358	0.007	0.77
Deer.20D.05	Spring	11.51		0.29	-0.213	0.001	0.91
Powder @ Huckleberry	Spring	11.26		0.24	-0.197	0.001	0.87
Powder DS Keating	Spring	0.31			0.345	0.020	0.15
Powder DS Mason Dam	Spring	10.53			0.038	0.113	0.01