

Quantifying the Impact of Water Management on Streamflow and Sediment in the Donner und Blitzen River within Malheur National Wildlife Refuge

by
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NATIONAL WILDLIFE REFUGE

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Malheur National Wildlife Refuge (MNWR), located in southeastern Oregon on the edge of the Basin and Range Province, spans 187,000 acres of high desert wetlands, riparian areas, wet meadows, and sage-steppe uplands. Managed for native wildlife and as a key stop on the Pacific Flyway, much of the refuge's habitat depends on water diverted from the Blitzen River through a series of dams used for flood irrigation of the Blitzen floodplain. The river is the largest tributary to Malheur Lake, which has been in a degraded and turbid state since the 1990s, primarily driven by invasive carp, wind-wave action, and lack of vegetation. Recent studies indicate that sediment delivered by the Blitzen River also contributes to lake turbidity, complicating restoration efforts.

Limited monitoring of the river has constrained the understanding of current water quality conditions of the river and the potential impacts of water management on sediment loading to the lake. To address this gap, we monitored five sites along the Blitzen River from March to September 2024, segmenting the river into reaches and assessing water quality changes from upstream to downstream. Using in-situ sondes, streamflow measurements, and water sampling, we collected data on turbidity, water temperature, suspended sediment, and streamflow. Our results indicate that sediment concentrations increase in specific reaches during summer baseflow, influenced by water management and potentially other environmental factors. These findings will support refuge managers by providing baseline water quality data, guiding future rehabilitation efforts, and contributing to a broader understanding of watershed-scale water quality dynamics.

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INTRODUCTION

The Malheur National Wildlife Refuge (MNWR) is located in the Harney Basin of southeastern Oregon. It covers 187,000 acres of diverse habitat, including open water, wetlands, riparian areas, irrigated meadows, grain fields, and uplands (Mayer et al., 2007). The refuge was established by an executive order in 1908, which created protections for Malheur, Harney, and Mud Lakes. Additionally, protections for the Blitzen Valley were implemented in 1935, bringing a large portion of the Donner und Blitzen River (hereafter referred to as the Blitzen River) under Fish and Wildlife protection. The refuge is managed to protect native plants and wildlife, particularly as an important stop for migratory birds flying the Pacific Flyway, and the value of much of the habitat on the refuge is highly dependent on the availability and management of water resources diverted from the Blitzen River (Mayer et al., 2007; U.S. Fish and Wildlife Service, 2013).

The Blitzen River is the primary tributary to Malheur Lake, and since the 1990s, the lake has been recognized as being in a degraded and turbid state, characterized by a lack of submergent and emergent vegetation (Smith and Wood, 2023). Recent studies by the U.S. Geological Survey (USGS) have indicated that, in 2019, 69% of turbidity was attributed to resuspension primarily driven by invasive carp, wind fetch, and lack of vegetation, while the remaining 31% was due to external loading (Smith and Wood, 2023). Smith and Wood (2023) indicated that tributary inputs, particularly from the Blitzen, are reinforcing the high turbidity found in the lake by increasing the amount of fine sediment available for resuspension. Additionally, previous studies in the early 2000s that utilized discrete water sampling found that turbidity was generally low in the river but increased with water control, particularly in relation to dam operations during the summer baseflow period (Mayer et al., 2007). Thus, water management through irrigation practices on the Blitzen may impact water quality in the river by decreasing flows, increasing turbidity, and raising water temperature, which can potentially affect aquatic species and contribute to the degraded conditions found in Malheur Lake.

Study Aims

The objectives of this study were to capture current conditions of the Blitzen River, assess the effects of water management on water quality along the Blitzen River, and identify reaches with the highest impacts on water quality. We achieved this by characterizing the spatial and temporal variability of key water quality parameters, including turbidity, suspended sediment, water temperature, and streamflow, in the river's mainstem. Due to limited monitoring of the Blitzen River within MNWR since the early 2000s, the extent to which water management may be contributing to river degradation—and, possibly to the water quality of Malheur Lake—remains unclear without the acquisition of new, comprehensive data. This study aimed to quantify the effects of water management practices within the MNWR on our chosen parameters, and the findings will inform resource management on water quality conditions and help prioritize possible future rehabilitation initiatives.

We collected high-quality data on suspended sediment, turbidity, and streamflow to quantify sediment and streamflow budgets at five sites along the mainstem Blitzen River. This was accomplished through the use of in-situ water quality sondes that collect continuous turbidity and temperature readings at 30-minute intervals, as well as discrete water sampling and discharge measurements at each site. The five site locations were chosen along the Blitzen River to segment the river into reaches, delineating turbidity sources, sinks, and storage, as well as streamflow inputs and outputs. Water budget equations were created for each site to calculate the mass balance of water flowing into and out of the river between sites. This reach approach allowed us to assess changes from upstream to downstream along approximately 45 river miles within the MNWR. Locations of each site were chosen based on their adjacency to significant inputs and outputs to the river, as well as their proximity to a bridge location for discharge measurements. Data collection was conducted from March 1st through September 5th of 2024 to capture the entirety of the irrigation season as well as differences between spring stormflow (March-June) and summer baseflow (July-September) periods. This report presents findings from the first year of our research conducted in 2024; a second phase is planned for 2025.

SITE DESCRIPTION

The Blitzen River is the primary drainage system for Steens Mountain in southeastern Oregon. The basin is in a hydrographically closed watershed on the northeastern corner of the Basin and Range region and is predominantly situated within the High Lava Plains physiographic province, characterized by alluvial lowlands and surrounding volcanic rock uplands (U.S. Fish and Wildlife Service, 2013). The crest of Steens Mountain is 9,670 feet in elevation, while most of the valley averages 4,150 feet, with an exceptionally shallow gradient for the Blitzen River within the refuge (Harney County Watershed Council, 2003; U.S. Fish and Wildlife Service, 2013). The Blitzen River is primarily driven by snowmelt from Steens Mountain, which typically peaks in late winter and spring. The watershed receives an average annual precipitation of 29.1 inches, mainly in the form of snow between October and June (Mayer et al., 2007; Oregon Climate Assessment). The average annual precipitation for the refuge is 11.0 inches for the water year (October-September), most of which falls from October-June, resulting in cold winters and relatively dry summers for the refuge (U.S. Fish and Wildlife Service, 2013). At the base of the mountain, the river flows into the Blitzen Valley and enters the southernmost point of MNWR at Page Springs Dam (Figure 1). From Page Springs Dam, the Blitzen River flows for approximately 45 river miles through the refuge before it discharges into Malheur Lake. The tributaries of Malheur Lake are the Silvies River from the north and the Blitzen River from the south, although input from the Silvies is often minor due to water allocations to the north (Smith and Wood, 2023). All water in the basin terminates at Malheur Lake, and in high water years, this water can flow into Mud and Harney Lakes, which are the lowest points in the basin.

A series of water control structures were built in the Blitzen Valley along the Blitzen River in the early 1900s to facilitate grazing and farming. During this time, approximately 17

miles of the river were straightened and channelized from the confluence of Bridge Creek to Busse Dam (Mayer et al., 2007; U.S. Fish and Wildlife Service, 2013). The Civilian Conservation Corps (CCC) assisted the refuge in expanding water control in the 1930s, and many of these structures have been maintained by the refuge.

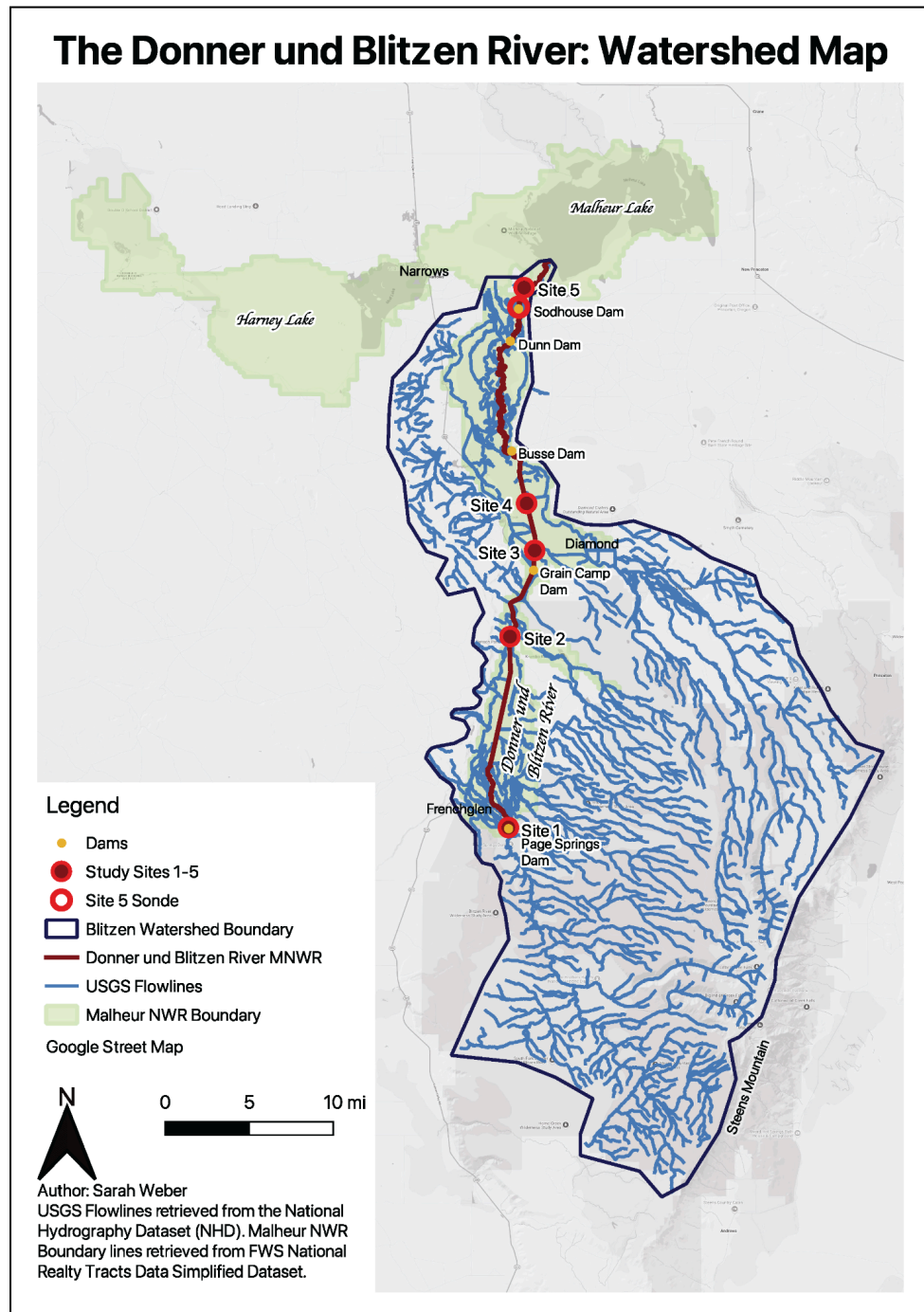


Figure 1: The Donner und Blitzen watershed, with flowlines, dams, and monitoring site locations.

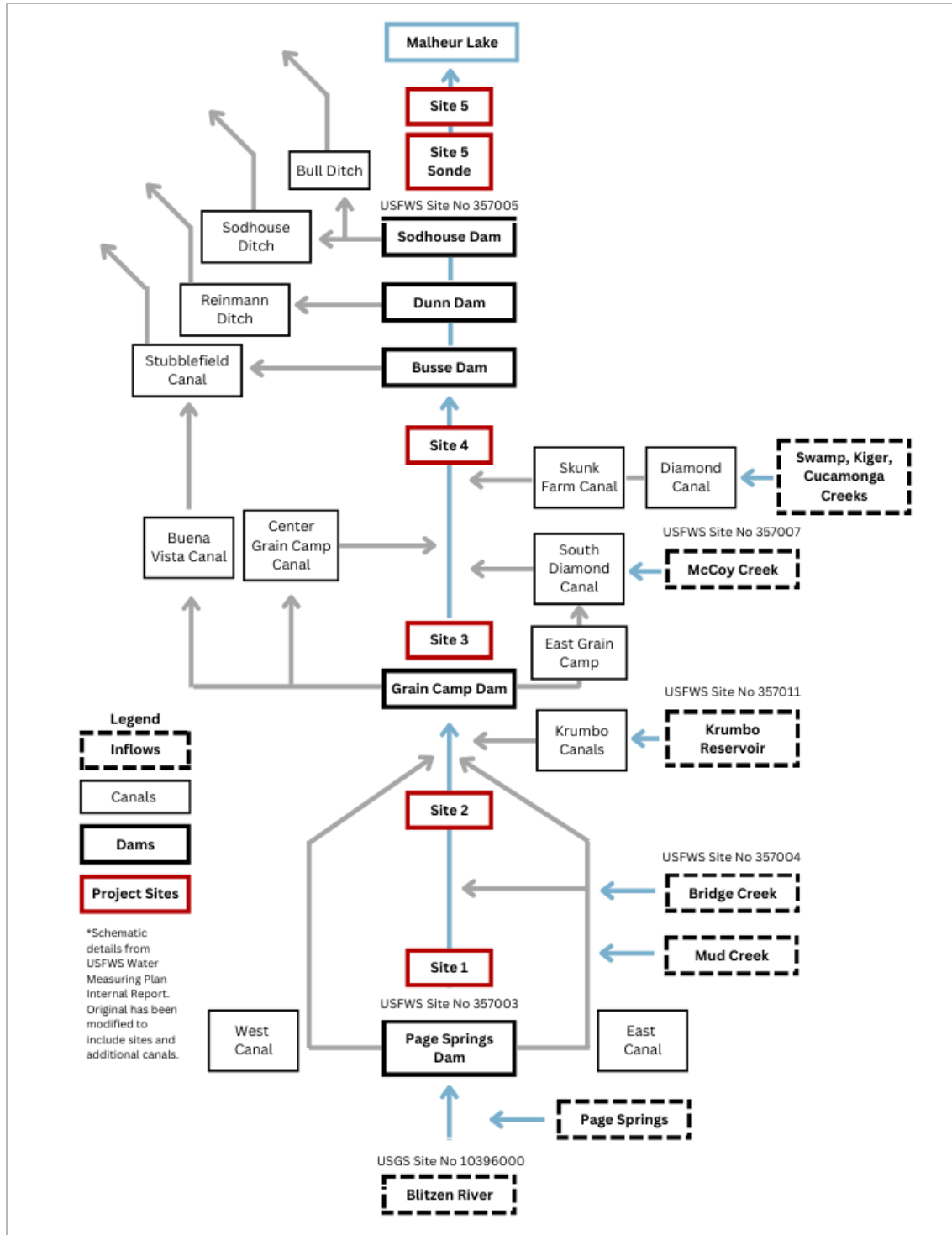


Figure 2: Schematic diagram of the Blitzen River through MNWR, showing major canal diversions and return flows, dams, and study sites. The original diagram from the USFWS Water Measurement Plan was modified to include study sites and additional canals (U.S. Fish and Wildlife Service, 2019).

Today, more than 900 water control structures are present within the refuge, and through this system of dams, canals, and drains, they provide critical habitat and feeding grounds for

migrating birds and native wildlife (U.S. Fish and Wildlife Service, 2013). Much of the riparian habitat is channelized and incised, and it is considered extremely poor due to its shallow and wide channel, limited willow cover, steep banks, and lack of complexity (Mayer et al., 2007). Tributaries that enter the refuge boundaries are also managed through the irrigation system before their confluence with the Blitzen River.

The irrigation season begins in early spring, typically between March and April, when water is diverted from the river into the surrounding floodplain for habitat management by lowering dams and backing up water into the flood-irrigation canal system. Large canals typically flow along the higher edges of the valley and flow north while delivering water to smaller canals through screw gates. These smaller canals commonly run along the southern edge of the field with additional screw gates or flashboard risers that allow water into the field. Sub-irrigation of water through canals is also common throughout the system. The irrigation season ends in July, prompting the opening of dams and the drainage of return flows to the river. Return flows are managed by opening or partially opening flashboard risers along the river that hold back this diverted water within impoundments and canals, or by closing the screw gates to major canals that allow water into fields. This enables the dewatering of fields for haying and grazing in August, which supports the refuge in providing nesting and feeding habitats for birds, as well as facilitating vegetation management. The Blitzen River functions primarily as a drain for the irrigation system. The system is highly intricate, consisting of an extensive network of canals, and the description provided here is not exhaustive. The schematic diagram provides a simplified visual representation of the system as a whole (Figure 2).

Individual site maps (Appendix: Site Maps, Figures 11-15) depict the area captured by each site, with added flowlines obtained from the National Hydrography Dataset, which represent most major canals to help illustrate water movement across the landscape. However, water is not always present where the flowlines indicate. Significant inputs and outputs to the river are noted on the site maps; however, numerous small-scale inputs were outside the scope of this project to account for and map due to their abundance and inconsistent flow over different periods of time. Figure 3 displays all study sites as well as the boundaries of the appended site-specific maps.

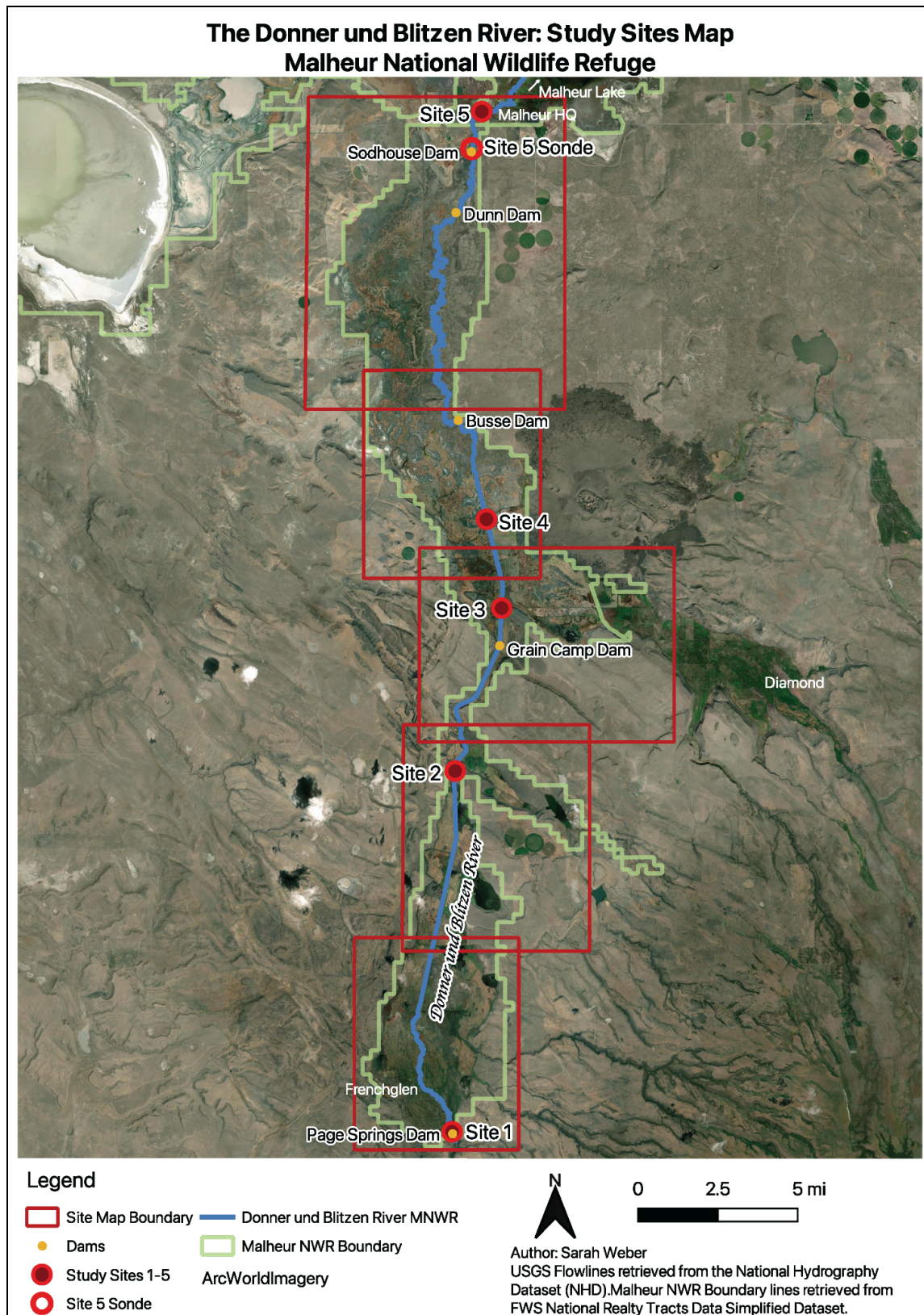


Figure 3: Study sites map of all sites along the Blitzen River. Site-specific map boundaries are indicated by red boxes, and the corresponding site maps are provided in the Appendix (Appendix: Site Maps, Figures 11-15).

METHODS

Site Selection

Site 1 Page Springs (USGS ID 424833118521200 Donner Und Blitzen Below Page Springs):

Site 1 is located at the base of Steens Mountain, where the river enters Malheur National Wildlife Refuge after it has passed over Page Springs Dam. This is the most upstream location of sites 1-5 that were monitored in 2024, referred to as Site 1 or Page Springs. Given its position, Site 1 serves as a reference baseline for the quality of water as it arrives at the refuge and as a reference point for the other sites downstream. Two major canal outputs, East and West Canals, are diverted just above the dam and provide a majority of the water to the irrigation system in the southern area of the refuge. Just below the dam and approximately 50 ft upstream of the sonde is a USFWS discharge gauge, Site No. 357003, that was used to compare discrete discharge measurements and turbidity correlation analysis in this study. The Frenchglen USGS gauge, Site No. 10396000, is located approximately 1.5 miles upstream and was utilized to determine diversions at Page Springs Dam. The sonde was deployed approximately 15 ft upstream of a bridge, discharge measurements were collected approximately 40 ft upstream of the sonde, and the bridge was used to collect discrete SSC samples. The site is located at 42°48'33" N, 118°52'11.5" W.

- Water Budget Equation: Page Springs Dam = Frenchglen Gauge + Page Springs (Spring upstream of refuge) - East Canal - West Canal

Site 2 Krumbo Lane Bridge (USGS ID 425818118520500 Donner Und Blitzen At Krumbo

Reservoir Rd): Site 2 is located in valley lowlands within Malheur National Wildlife Refuge, about 13 river miles downstream of Page Springs site. This is the second most upstream location of sites 1-5 that were monitored in 2024, and it is referred to as Site 2 or Krumbo. Site 2 captures inputs within the southern area of the refuge, notably, the Bridge Creek confluence and return flows from Knox Ponds, 5-Mile Canal, and Benson Ponds. The site sonde was deployed approximately 100 ft upstream of a bridge, and discharge measurements and SSC samples were collected from the bridge. The site is located at 42°58'18.4" N, 118°52'4.6" W.

- Water Budget Equation: Krumbo Lane Bridge = Page Springs Dam + Mud Creek + Fish Creek + Bridge Creek

Site 3 Diamond Lane Bridge (USGS ID 430242118502100 Donner Und Blitzen River at

Diamond Ln): Site 3 is located in valley lowlands within Malheur National Wildlife Refuge, about 6 river miles downstream of the Krumbo site. This is the third most upstream location of sites 1-5 that were monitored in 2024, and it is referred to as Site 3 or Diamond. Site 3 captures return flow inputs from East and West Canals, the Krumbo Creek and reservoir area, and output diversions Center Ditch, East Grain Camp, and Buena Vista Canal from Grain Camp Dam about a mile upstream. The sonde was deployed approximately 150 ft downstream of a bridge, and the

bridge was used to collect discharge measurements and SSC samples. The site is located at 43°2'42.2" N, 118°50'21.4" W.

- Water Budget Equation: Diamond Lane Bridge = Krumbo Lane Bridge + Krumbo Creek + East Canal + West Canal - East Grain Camp - Center Grain Camp Canal - Buena Vista Canal

Site 4 Remauli Bridge (USGS ID 430506118505600 Donner Und Blitzen River at Skunk

Farm Canal Rd): Site 4 is located in valley lowlands within Malheur National Wildlife Refuge, about 3 river miles downstream of the Diamond site. This is the fourth most upstream location of sites 1-5 that were monitored in 2024, and it is referred to as Site 4 or Remauli. Site 4 captures flows from the central refuge area, notably, inputs from McCoy, Kiger, and Cucamonga Creeks, as well as return flows from Center Ditch, partial return of Buena Vista Canal, and Skunk Farm Canal. The sonde was deployed directly below the bridge, and the bridge was used to collect discharge measurements and SSC samples. The site is located at 43°05'06.0" N, 118°50'56.0" W.

- Water Budget Equation: Remauli Bridge = Diamond Lane Bridge + McCoy Creek + Kiger Creek + Cucamonga Creek + Swamp Creek

Site 5 Sodhouse Lane Bridge (USGS ID 431605118510600 Donner Und Blitzen River at

Sodhouse Ln): Site 5 is located in valley lowlands within Malheur National Wildlife Refuge, about 21 river miles downstream of the Remauli site. This is the most downstream location of sites 1-5 that were monitored in 2024, and it is referred to as Site 5 or Sodhouse. This site captures diversion outputs for Stubblefield Canal, Reinmann Ditch, and Sodhouse Canal. The sonde was deployed approximately 90 ft downstream of an established discharge gauge, USFWS Site No 357005, located at 43°15'4.8" N, 118°51'27.9" W. A bridge, approximately 1.5 river miles downstream of the site sonde located at 43°16'4.7" N, 118°51'5.5" W, was used to collect discharge measurements, SSC samples, and reference measurements apart from those taken during cleaning visits for the sonde. The separation between the site sonde and other measurements was due to our discharge method requiring a bridge crossing, but both locations are channelized with no major inputs between them. The site identification number and coordinates are for the bridge location.

- Water Budget Equation: Sodhouse Dam Bridge = Remauli Bridge - Stubblefield Canal - Reinmann Ditch - Sodhouse Canal

Site Instrumentation

In the spring of 2024, USGS and MNWR personnel constructed pipe infrastructure to house the site water quality sondes for all sites. Infrastructure is a 20-ft 4-in-diameter PVC pipe secured somewhat orthogonal to the bank, positioned so that the end of the pipe is mid-water column. The pipe was perforated with multiple 1-in-diameter holes in the last 10 in of the pipe to allow free-flowing water past the sensors. All sites utilized YSI 6-Series water quality sondes; Sites 1-4 used YSI 6920 sondes and Site 5 used a YSI 6600 sonde, and all measured continuous

water temperature and turbidity at 30-minute intervals. All sondes were equipped with a YSI 6136 turbidity sensor that was calibrated and recorded data in FNU, and a thermistor checked with a NIST reference temperature probe that recorded data in degrees Celsius. Thermistors are not calibrated; however, they are compared with a NIST temperature probe at the beginning and end of the data collection period, and all thermistors were within the required ± 0.2 Celsius criteria at both checks. No thermistor exceeded this requirement during our study.

Water quality sondes are subject to sensor fouling by debris or other obstructions. To minimize this, sondes were cleaned once to twice a month from 03/01/2024-06/27/2024 and every two weeks from 06/27/2024-09/05/2024. Data from each sonde were uploaded during the cleaning visits and transferred by text file for processing. Site sonde cleanings were conducted in a bucket on the riverbank. A bucket of native water was collected and the site sonde and reference sonde were placed in the bucket to compare readings. Since the turbidity probe has a wiper, most fouling was prevented, and fouling corrections were applied on a case-by-case basis. Due to the nature of the site, the sonde and pipe were subject to fouling from debris, aquatic organisms, and algae growth, as well as random unknown events. These fouling events were observed on-site and by data examination once back in the office. Any data points exceptionally divergent from the dominant baseline trend, including readings that were double or more than double the previous reading, were removed from the dataset. Interpolation was used to fill in the missing turbidity data.

The water quality sondes are also subject to calibration drifts that are accounted for and corrected periodically. Approximately every six weeks, sondes were returned to the lab for a turbidity calibration check and recalibration, as needed. The sonde was redeployed either the same day or the following. Calibration corrections were applied to the dataset during processing to correct for calibration drift. The polymer standard for calibration is used instead of formazin because 1) polymer is stable without agitation, which eliminates technique differences and uncertainty, 2) polymer has a longer shelf life, and 3) YSI recommends polymer standards for their instruments. In addition, all data collected at MNWR since 2017 were obtained by using YSI 6136 turbidity sensors and the polymer standard, keeping all data comparable and consistent. The turbidity calibration is a two-point calibration to 0.0 and 1000 FNU; the calibration is then checked at 126 or 121 FNU. Standard 126 was used for checking calibrations 03/01/2024 -05/14/2024, and standard 121 was used for checking calibrations 07/12/2024-09/05/2024. The sensor is calibrated with deionized water for the 0 FNU standard and the YSI polymer-based standard of 1000 FNU. YSI polymer-based standard of 126 or 121 FNU is used to check the calibration. Calibration criteria for turbidity require that the sensor must read standards within ± 2 FNU or 5%, otherwise recalibration is required.

Streamflow Measurements

Continuous and discrete discharge data were collected and utilized within the dataset. Continuous discharge data were sourced from two gauges within MNWR and one USGS gauge just outside the refuge boundary. Discharge data sourced from gauges within the refuge include

Page Springs gauge USFWS Site No 357003, located near Site 1, and Sodhouse gauge USFWS Site No 357005, located near Site 5 sonde. Discharge data were sourced from the USGS gauge, Donner und Blitzen River NR Frenchglen OR - 10396000, approximately 1.5 miles upstream of Site 1. All gauges are telemetered and report discharge at 15-minute intervals. Continuous discharge data were used to compare discrete discharge measurements and correlate streamflow with turbidity at appropriate sites, determine diversion outputs at Page Springs, and calculate sediment load estimates at Sites 1 and 5.

Discrete discharge measurements were collected by acoustic instruments, including a Teledyne StreamPro Acoustic Doppler Current Profiler (ADCP) and YSI Flowtracker 2 Acoustic Doppler Velocimeter (ADV). The ADCP was the primary method of discharge collection. Measurements were collected by lowering the instrument from the bridge into the water using a rope and moving the ADCP perpendicularly across the river channel multiple times to gather transects. The ADCP is linked via Bluetooth to a field laptop where live data are projected and stored for each measurement. Training on USGS standard protocol for ADCP discharge collection was provided by USGS hydrologic technicians for the first set of measurements in March, and this protocol was used throughout the duration of the data collection period. The ADCP has a maximum depth range of 7 meters and enables the safe measurement of unwadable river reaches where a bridge is present. Discharge measurements were collected using the ADCP WinRiver II software and then quality-assured through Qrev 4.37 USGS Hydroacoustics software. The Flowtracker 2 was utilized at Site 1 starting in July when the water level became too low to measure with an ADCP (<0.5 feet depth). The Flowtracker has internal quality assurance software, and thus, no additional quality check was required for this method. However, ADCP and Flowtracker measurements were compared to the nearby gauge at Sites 1 and 5, and no significant deviations were identified.

Water Sample Collection

Water samples were collected intermittently from each site over the project's duration, with the aim of gathering samples from a range of high and low streamflows. Prioritizing sample collection in this way allowed us to capture the range of suspended sediment concentrations in the river over the entirety of the sample collection period. Samples were collected with a Van Dorn Water Sampler lowered from the mid-channel at the site bridge and sampled from the mid-water column. Samples were then transferred into a 2.0-liter bottle and labeled for processing. Samples were processed by the USGS Sediment Lab at the Cascade Volcano Observatory in Vancouver, WA, and analyzed for suspended sediment concentration, loss on ignition concentration, and sand-fine break. Suspended sediment concentration includes the inorganic fraction of the sample, while the loss on ignition concentration allows for the separation of inorganic and organic fractions by weighing before and after burning the organic fraction from the sample. Sand-fine break separates the sediment from the sample into sand-sized particles (>0.063 mm) and fine-sized particles (<0.063 mm) to characterize sediment sizes within samples. A total of 60 water samples were collected, comprising 54 unique samples and 6 quality

assurance and quality control (QAQC) samples, in accordance with the USGS protocol, which requires 10% of the samples to be collected as QAQC.

Suspended Sediment Load Calculation

Suspended sediment concentrations were linearly correlated with corresponding turbidity measurements and multiplied by continuous streamflow to quantify sediment loads (Rasmussen et al., 2009). Suspended sediment load allows inquiry into the sediment flux of each section between sites as well as overall net erosion across the system. Suspended sediment concentrations and sediment loads were calculated as:

$$SSC_i = b_1 * (TB) + b_0 \Rightarrow SSL_i = SSC_i \times Q_i \times c_i \Rightarrow SSL_n = \Sigma SSL_i$$

Where

- SSC_i is suspended sediment concentration, in milligrams per liter;
- b_1 is the slope;
- TB is turbidity, in FNU, 30-minute intervals;
- b_0 is the y-intercept;
- SSL_i is the computed suspended sediment load, in tons per 30-minute intervals;
- Q_i is the streamflow, in cubic feet per second;
- c_i are constants for converting units to tons per 30 minutes;
- SSL_n is the computed suspended sediment load for the total time period, in tons

The linear correlation equation between turbidity and suspended sediment concentration was used to convert continuous 30-minute turbidity measurements into continuous 30-minute suspended sediment concentrations. These continuous suspended sediment concentrations were then multiplied by continuous 30-minute discharge measurements for sediment mass per time in seconds. These measurements were converted into tons of sediment per 30 minutes and then summed for the entire time period to compute tons of sediment per total time recorded for each site. The total time unit is the number of half-hours within the total time frame listed. When sites had a strong linear correlation ($R^2 \geq 0.68$) between turbidity and suspended sediment concentration, we utilized the linear equation derived from the site sample set to convert turbidity to suspended sediment concentration. This method was used for Sites 1, 2, and 4. Sites 3 and 5 had poor linear correlations ($R^2 < 0.25$) between turbidity and suspended sediment, which prevented using turbidity as a surrogate for suspended sediment. For Sites 3 and 5, an average suspended sediment concentration was computed from the total samples at the respective sites, and this average concentration was used to compute the suspended sediment loads for Sites 3 and 5. Continuous discharge for Sites 1 and 5 was obtained from the gauges. For Sites 2-4, continuous discharge was obtained by interpolating a 30-minute time series from ADCP discharge measurements.

RESULTS

Site Results: Turbidity, Water Temperature, and Streamflow

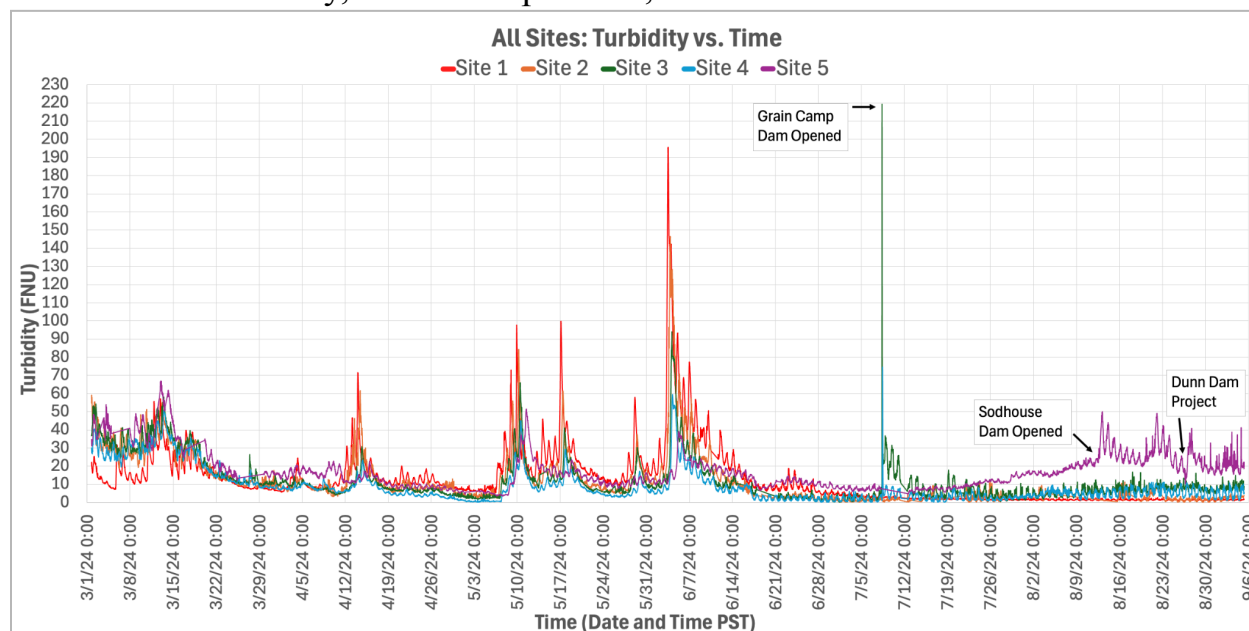


Figure 4: Turbidity (FNU) of all sites overlaid in a time series for the total study period.

The turbidity time series graph shows continuous turbidity of each site over the duration of the study period (Figure 4). Statistical summaries for turbidity at each site over the total, stormflow, and baseflow periods are in Table 1. Detailed turbidity time series graphs, including points for ADCP measurements, water sample collection, and gauge discharge measurements for Sites 1 and 5, are appended, along with daily averaged turbidity graphs (Appendix: Site Graphs, Figures 17-31). The overall trend across sites during stormflows in March through June was generally flashy with increased turbidity, where Sites 1 through 5 had means of 19.3, 16.4, 14.9, 11.7, and 17.9 FNU, respectively. This was followed by a lower and more consistent trend during summer baseflows in July through September, in which Sites 1 through 5 had means of 1.9, 2.5, 7.0, 4.2, and 16.3 FNU, respectively.

Each site deviated from the general trends at certain time periods. Site 1 typically experienced the highest turbidity events during stormflow; however, in March, it had the lowest turbidity of all the sites. During baseflow, turbidity at Site 1 remained consistently low and stable, whereas the downstream sites exhibited greater variability in daily fluctuations, with an overall increasing trend. Site 2 followed a dampened version of Site 1's stormflow pattern, and during baseflow remained relatively low but showed a more irregular and elevated turbidity pattern in comparison to Site 1. The sonde at Site 2 experienced significant algal fouling during July and August, unlike other sites, resulting in substantial loss of baseflow data. Consequently, data interpolation was required, introducing uncertainty and complicating assessments for this period at this site.

At Sites 3 and 4, stormflow turbidity declined similarly to Site 2, with increasing distance from Site 1. At certain peaks, turbidity at Sites 3 and 4 was notably lower than at Site 2, which may be due to sediment retention behind Grain Camp Dam. This trapping effect was observed in July, when the dam released accumulated sediment, resulting in increased turbidity at Site 3 (Figure 4). The release resulted in an elevated daily pattern of turbidity that persisted for over a week. The subsequent release of the Krumbo Canals on July 19th increased turbidity for approximately another week. Then, in August, turbidity began rising again in an irregular trend not associated with known management actions. Site 4 also responded to the dam opening, but with a lower peak and faster return to previous lower levels than Site 3. This difference is likely due to sediment deposition with increasing distance from the dam and the dilution by tributaries between the sites. After the dam opening, Site 4 showed greater daily fluctuations in turbidity, and values increased in August, similar to those observed at Site 3. The pattern at Site 4 also differed with a pronounced daily fluctuation during low flows, a potential indication of bioattenuation effects from the large mussel beds between Sites 3 and 4.

Turbidity at Site 5 was highest from March through early April, until peak flows in mid-April. While Site 5 showed the most attenuated stormflow turbidity spikes, it consistently maintained higher turbidity levels than the other sites when returning to pre-spike levels. In early June, Site 5 continued to deviate from the other sites, consistently having higher turbidity that increased during July and August. When Sodhouse Dam was opened in August, daily fluctuations in turbidity greatly increased (Figure 4). To investigate rising turbidity at Site 5 during baseflow, a canoe survey was conducted on August 14, and continuous 1-minute turbidity and temperature readings were collected along a substantial portion of the reach upstream of Site 5, between Rockaway Bridge and Canoe Take Out Rd. The results showed a gradual increase in turbidity and temperature midway through the reach (Appendix: Site Maps, Figure 16), without an observed surface water input during the survey. Periodic coordinates recorded during the survey are noted in the Site 5 Map, with results appended (Appendix: Site Maps, Figure 15, Table 5). In late August, turbidity at Site 5 increasingly fluctuated following the Dunn Dam removal project (Figure 4), which rerouted the river and increased turbidity through the study's conclusion.

TURBIDITY: TOTAL (FNU)						TURBIDITY: STORMFLOW						TURBIDITY: BASEFLOW					
Site #:	1	2	3	4	5	Site #:	1	2	3	4	5	Site #:	1	2	3	4	5
Min	0.9	0.0	0.1	0.0	3.6	Min	4.0	0.3	0.1	0.0	3.6	Min	0.9	0.0	0.1	0.0	4.5
25th %	2.0	2.7	5.2	3.1	9.6	25th %	9.0	6.9	6.3	4.2	10.2	25th %	1.4	1.2	3.9	2.3	8.1
Median	8.6	6.8	8.1	5.9	14.9	Median	13.9	11.1	10.3	8.1	14.6	Median	1.7	2.2	6.4	3.6	15.6
Mean	13.1	11.5	12.1	9.0	17.3	Mean	19.3	16.4	14.9	11.7	17.9	Mean	1.9	2.5	7.0	4.2	16.3
75th %	16.9	14.3	13.6	10.2	21.0	75th %	22.8	22.4	19.2	15.0	20.4	75th %	2.1	3.2	8.3	5.8	21.7
95th %	40.6	37.4	36.9	31.3	39.8	95th %	48.5	43.9	39.9	33.5	41.6	95th %	3.7	5.6	14.1	8.5	32.2
Max	195.4	146.3	213.5	73.6	66.9	Max	195.4	146.3	94.0	59.5	66.9	Max	6.4	11.3	213.5	73.6	50.0

Table 1: Statistical summaries for turbidity across sites during the total study period, stormflow, and baseflow. All units are in FNU.

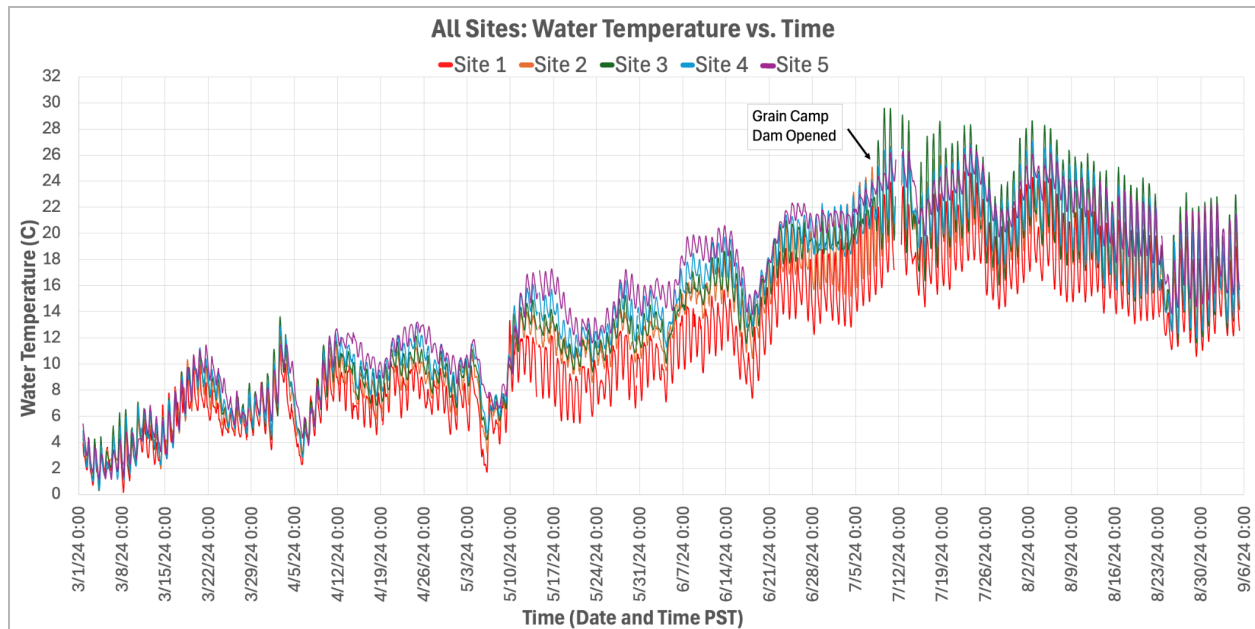


Figure 5: Water temperature (Celsius) of all sites overlaid in a time series for the total study period.

The temperature time series graph displays the continuous water temperature at each site throughout the study period (Figure 5). Statistical summaries of water temperature at each site for the total, stormflow, and baseflow time periods are presented in Table 2. Site-specific temperature time series graphs are provided in the appendix, along with daily averages (Appendix: Site Graphs, Figures 17-31). A seven-day-average temperature graph for all sites is also appended (Appendix: Site Graphs, Figure 32). Overall, water temperature trends showed that all sites initially overlapped during the early stormflow period before transitioning into a pattern of increasing temperature from upstream to downstream for the remainder of the study. The mean water temperatures of Sites 1 through 5 during stormflow were 8.38, 9.94, 10.39, 10.94, and 11.83°C, respectively. The mean water temperatures of Sites 1 through 5 during baseflow were 17.45, 19.94, 20.59, 20.55, and 21.69°C, respectively.

During baseflow, water temperature continued to increase from upstream to downstream, however, temperature ranges varied across sites. The middle sites showed the most variable temperatures during this period, while Site 1 maintained the lowest temperature with moderate fluctuations, and Site 5 maintained the highest temperature with a narrow range. Sites 2 and 4 showed daily temperature fluctuations similar to Site 1 but at consistently higher magnitudes. The most pronounced temperature anomaly occurred at Site 3 following the opening of Grain Camp Dam. Prior to this event, temperature followed a consistent upward trend with a narrow range of daily fluctuations. Immediately afterward, the range in daily fluctuations increased substantially and remained elevated for much of the baseflow period. During baseflow, Site 3 recorded the highest maximum temperature among all sites (29.60°C), followed by Sites 4 (27.07°C), 2 (26.91°C), 5 (26.56°C), and Site 1 (24.71°C) with the lowest maximum

temperature. The temperature increases observed at Site 4 are likely largely influenced by conditions at Site 3, but were diluted by tributary inputs between the sites.

WATER TEMP: TOTAL (C)						WATER TEMP: STORMFLOW						WATER TEMP: BASEFLOW					
Site #:	1	2	3	4	5	Site #:	1	2	3	4	5	Site #:	1	2	3	4	5
Min	0.15	1.26	0.28	0.34	1.14	Min	0.15	1.26	0.28	0.34	1.14	Min	10.57	12.08	11.62	11.99	13.82
25th %	7.02	8.42	8.94	9.37	10.17	25th %	5.85	6.74	7.10	7.28	7.86	25th %	15.35	17.81	18.31	18.71	20.30
Median	10.87	12.83	13.33	14.27	15.73	Median	7.98	9.67	10.03	10.68	11.59	Median	17.42	19.99	20.59	20.87	22.03
Mean	11.55	13.44	13.96	14.31	15.30	Mean	8.38	9.94	10.39	10.94	11.83	Mean	17.45	19.94	20.59	20.55	21.69
75th %	16.16	18.67	19.31	19.73	21.27	75th %	10.58	12.66	13.23	14.36	15.72	75th %	19.49	22.15	22.84	22.82	23.74
Max	24.71	26.91	29.60	27.07	26.56	Max	20.21	21.67	21.00	22.30	22.35	Max	24.71	26.91	29.60	27.07	26.56

Table 2: Statistical summaries for water temperature across sites during the total study period, stormflow, and baseflow. All units are in degrees Celsius.

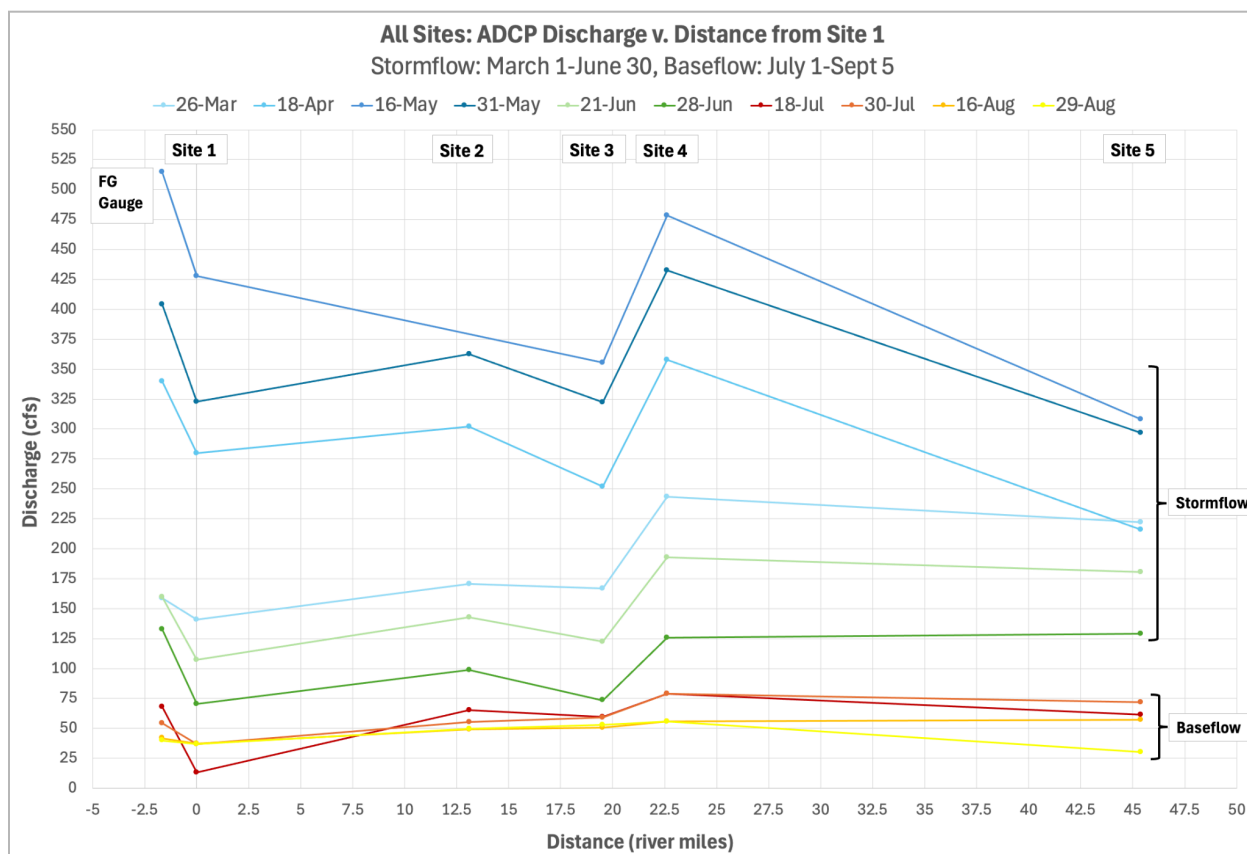


Figure 6: ADCP measurements at each site as a function of river distance downstream, using Site 1 as a zero point. Line colors are associated with the ADCP measurement date in the legend. Discharge recorded at the Frenchglen gauge upstream of the refuge and Site 1 is included at a distance of -1.7 miles. Labels at the top of the graph indicate the positions of Sites 1 through 5. Measurements of all sites on individual dates are appended (Appendix: ADCP Measurements, Figures 43-52).

River discharge as a function of distance illustrates the longitudinal variation in streamflow across study sites for all 50 ADCP measurements collected during the study period (Figure 6). There are four missing site measurements: one at Site 1 on 4/18/24 that has discharge from the Page Springs gauge, one at Site 3 on 4/18/24 that was interpolated, one at Site 2 on 5/16/24 that is left blank, and one at Site 5 on 5/16/24 that has discharge from the Sodhouse Dam gauge. ADCP measurements, occasionally collected over two days, have been consolidated into a single day for clarity in this figure. The Frenchglen gauge readings, located upstream of the refuge, are included to account for withdrawals from Page Springs Dam above Site 1. River distances are represented in river miles, measured by tracing flowlines from the National Hydrography Dataset using QGIS mapping software. All ADCP measurements for individual dates listed in the legend are appended (Appendix: ADCP Measurements, Figures 43-52).

Throughout the study period, discharge patterns show consistent trends between sites, particularly during stormflow. Discharge decreases from the Frenchglen Gauge upstream of the refuge and Site 1, which is attributed to the diversion of the East and West Canal at Page Springs Dam. Discharge increases between Sites 1 and 2, primarily due to the confluence of Bridge Creek. Site 2 lacks a measurement on May 16, however, discharge at this time likely aligns with the overall increasing trend observed between Sites 1 and 2. Between Sites 2 and 3, discharge declined, attributed to East Grain Camp, Center Grain Camp Canal, and Buena Vista Canal diversions at Grain Camp Dam. A sharp increase was observed between Sites 3 and 4, driven mainly by the input from McCoy Creek, as well as contributions from Swamp, Kiger, and Cucamonga Creeks through return flows. Between Sites 4 and 5, discharge again decreased, attributed to the Stubblefield diversion at Busse Dam, and the Reinmann Ditch and Sodhouse Canal diversions upstream of Sodhouse Dam. In general, discharge was the highest and most variable during stormflow periods, particularly during snowmelt events in April and May. Intermediate flows occurred during March and June, while lower and relatively stable flows dominated during July and August, under baseflow conditions.

Some deviations from these general trends occurred during baseflow. The measurement at Site 1 on July 18 showed a notable decrease in discharge compared to other baseflow measurements. This anomaly occurred because the West Canal remained open when it should have been closed. The canal was immediately closed following the site visit to maintain the refuge established with a minimum flow of 25 cubic feet per second in the river. During July, discharge measurements at the three middle sites were slightly elevated compared to other baseflow measurements, likely due to return flows that occurred throughout the month. The final measurement at Site 5 on August 29 deviated from previous baseflow values, coinciding with the Dunn Dam removal project, which temporarily backed up the river and discharged an estimated 25 cubic feet per second into Reinmann Ditch at the time of measurement.

Water Sample Results

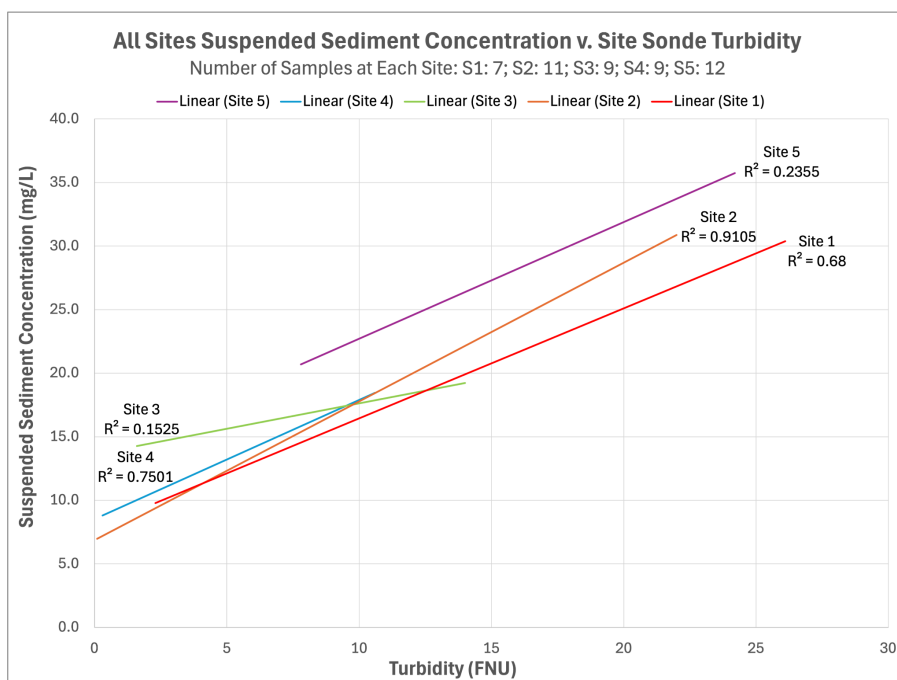


Figure 7: All sites' linear correlations between sampled suspended sediment concentration and site turbidity. Individual data points used for correlations are provided in the appendix (Appendix: Water Samples, Figures 33-37).

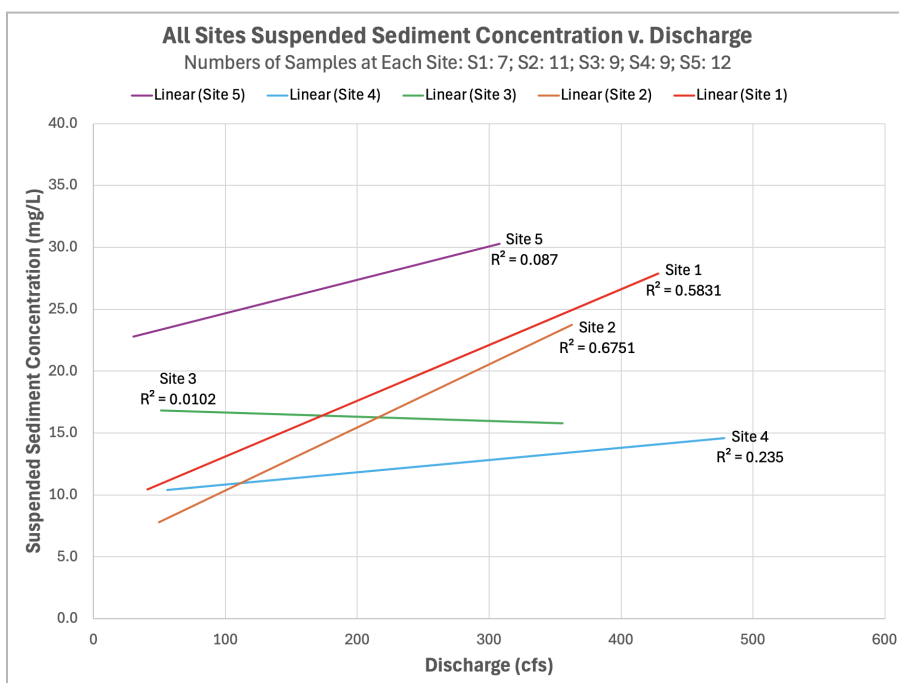


Figure 8: All sites' linear correlations between sampled suspended sediment concentration and discharge. Individual data points used for correlations are provided in the appendix (Appendix: Water Samples, Figures 38-42).

Linear regression analyses were conducted to examine the relationships between suspended sediment and turbidity (Figure 7) and between suspended sediment and streamflow (Figure 8) across all sites. Significant events, such as dam openings, were excluded from the analyses to prevent bias in the correlations. The appendix provides a table detailing all samples and measurements, including those removed (Appendix: Water Samples, Table 6).

Most of the site-by-site correlations show positive relationships, though the strength of the correlation varies by site. For the suspended sediment and turbidity correlations (Figure 7), Sites 1, 2, and 4 strongly correlate turbidity with suspended sediment ($R^2 \geq 0.68$), indicating its reliability as a proxy. Accordingly, the regression equations for these sites were used to estimate suspended sediment loads. In contrast, the turbidity correlations for Sites 3 and 5 were particularly weak ($R^2 < 0.25$), necessitating the use of computed site-specific average suspended sediment concentrations for load calculations. The sampling regime of Site 3 did not capture high suspended sediment flows and may be influencing this poor correlation.

The relationship between suspended sediment and streamflow showed much greater variability across sites (Figure 8). Sites 1 and 2 displayed somewhat strong correlations ($R^2 \geq 0.58$), whereas Site 3 showed a particularly poor and negative correlation ($R^2 = 0.01$), and Sites 4 and 5 showed weak but positive correlations ($R^2 < 0.25$). As with the suspended sediment and turbidity correlation, the poor correlation between suspended sediment and discharge (Figure 8) at Site 3 is likely related to the sampling bias.

SSC				LOI				% Fines				Status
2024 S5 06/28				2024 S5 06/28				2024 S5 06/28				
Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	
Replicate	18	19	5	Replicate	5	5	0	Replicate	94.3	94.4	0	Approved
2024 S4 07/16				2024 S4 07/16				2024 S4 07/16				
Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	
Replicate	114	30	280	Replicate	21.4	7.8	174	Replicate	58.3	75.1	22	Rejected
2024 S3 07/19				2024 S3 07/19				2024 S3 07/19				
Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	
Replicate	22	21	5	Replicate	7.1	3.9	82	Replicate	76.4	91.7	17	Approved
Split	22	26	15	Split	7.1	6.7	6	Split	76.4	57.1	34	Approved
2024 S5 08/12				2024 S5 08/12				2024 S5 08/12				
Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	Type	WS	WSQ	%Diff	
Replicate	55	57	4	Replicate	7	8	13	Replicate	95.3	95.2	0	Approved
Split	55	53	4	Split	7	5	40	Split	95.3	94.4	1	Approved

Table 3: QAQC sample results, including site and date for each QAQC sample (WSQ) alongside its corresponding primary sample (WS). The table presents values for suspended sediment concentration (SSC), loss on ignition (LOI), and the percentage of fine particles (% Fines). The status column indicates whether the QAQC samples were approved or rejected based on their variability or consistency with primary samples.

The QAQC sample results (Table 3) include the site and date for each QAQC sample (WSQ), alongside its corresponding primary sample (WS), and the percent differences. The table presents values for suspended sediment concentration (SSC), loss on ignition (LOI), and the percentage of fine particles (% Fines). The status column indicates whether the QAQC samples were approved or rejected based on their variability or consistency with primary samples. When interpreting percent differences, caution is advised for small values, as minor variations can result in disproportionately large percent differences. The purpose of these QAQC samples is to evaluate the accuracy and precision of both field sampling and laboratory processing. Field precision was assessed by collecting a primary sample followed by a replicate sample, while laboratory precision was evaluated by splitting the primary into two subsamples, with the second labeled as the split. The majority of QAQC samples were within acceptable limits, with the exception of one sample from Site 4. The Site 4 samples were collected after the opening of the Skunk Farm Canal, where dynamic flow conditions contributed to substantial differences between the primary and replicate samples and thus were rejected as an acceptable QAQC sample. The initial high magnitude of the primary sample indicated that this return flow introduced suspended sediment and organic matter into the river, but this effect quickly diminished in magnitude in the subsequent replicate sample.

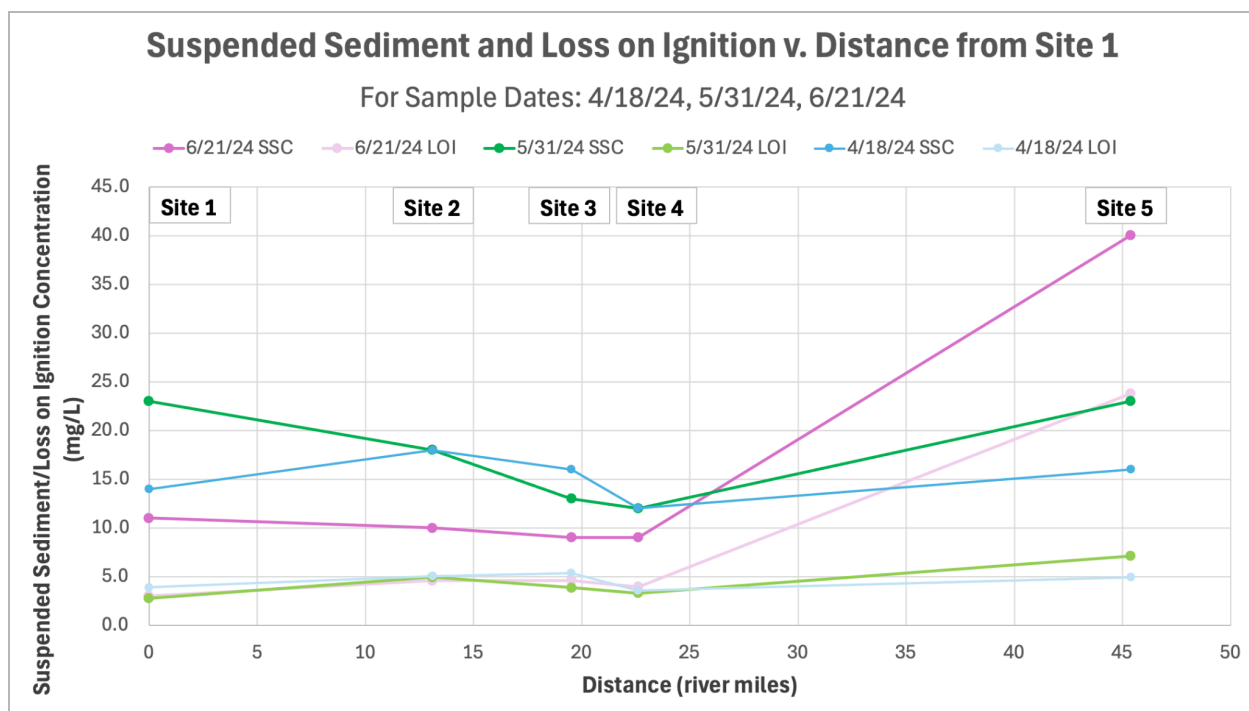


Figure 9: Suspended sediment (SSC) and loss on ignition (LOI) concentrations as a function of river distance downstream, using Site 1 as a zero point. Line colors are associated with the sample type and date of collection in the legend. Labels at the top of the graph indicate the positions of Sites 1 through 5.

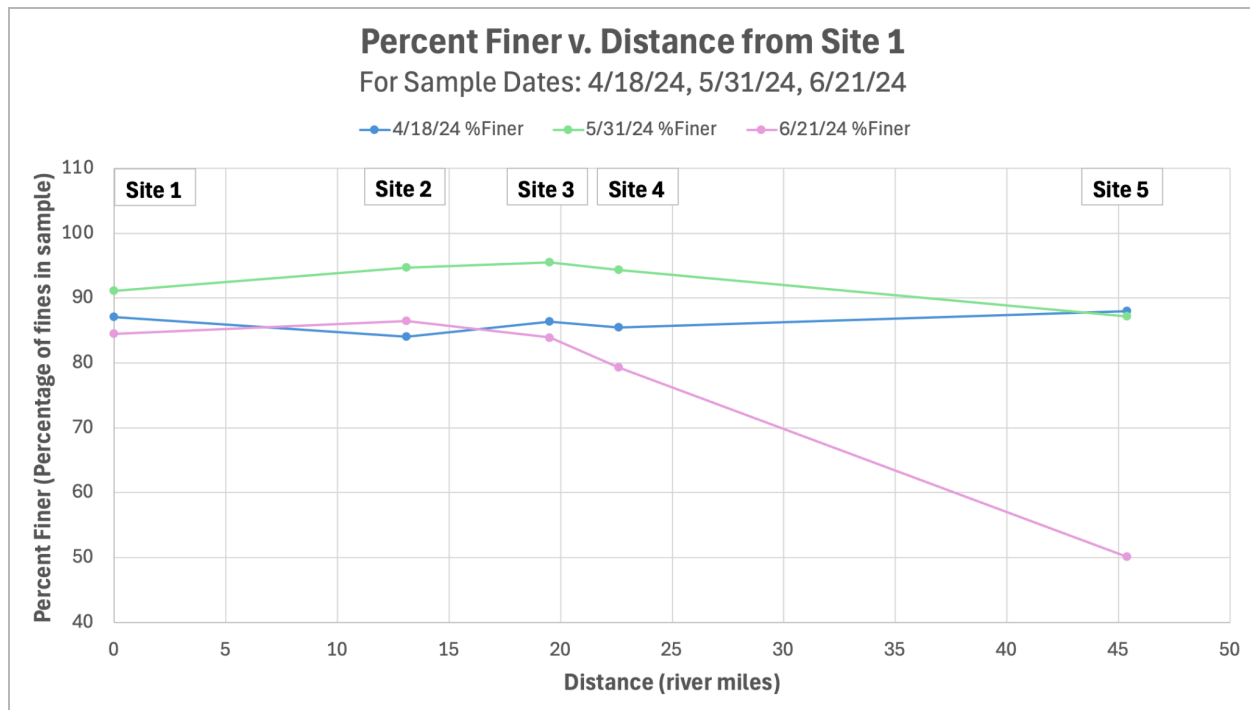


Figure 10: Percent finer (percentage of sample sediments <0.063 mm) as a function of river distance downstream, using Site 1 as a zero point. Line colors are associated with the sample date in the legend. Labels at the top of the graph indicate the positions of Sites 1 through 5. Note that the y-axis range is 40% to 110%.

The plot of suspended sediment and loss on ignition as a function of distance from Site 1 (Figure 9) and percent finer as a function of distance from Site 1 (Figure 10) displays sample results for three sampling dates across all sites. Percent finer represents the proportion of sediments <0.063 mm in size (e.g. clays), with the remaining fraction being >0.063 mm (e.g. sands). The three sampling dates were selected because all sites were sampled on the same day, without any data points removed for known events (e.g. dam or canal openings). Details of all samples and removed data points are provided in the appended water sample table (Appendix: Water Samples, Table 6).

The suspended sediment concentrations (Figure 9) showed varying downstream trends across sampling dates. The overall mean concentrations of suspended sediment samples for all sites during the study period were 17.6 mg/L (Site 1), 11.2 mg/L (Site 2), 16.4 mg/L (Site 3), 11.9 mg/L (Site 4), and 25.7 mg/L (Site 5). In April, suspended sediment increased between Sites 1 and 2, decreased between Sites 2 and 4, and increased again between Sites 4 and 5. In May, concentrations decreased between Sites 1 through 4, with an increase between Sites 4 and 5, where Sites 1 and 5 had the same concentration of 23.0 mg/L. June samples showed similar trends to those in May, but with a significantly larger increase between Sites 4 and 5. This variability in suspended sediment transport appears somewhat independent of streamflow, as indicated by the following streamflow measurements for this sampling date: 107.57 cfs (Site 1), 142.98 cfs (Site 2), 122.36 cfs (Site 3), 193.07 cfs (Site 4), and 180.41 cfs (Site 5).

Loss on ignition concentrations (Figure 9) showed similar downstream trends across sampling dates. The overall mean concentrations of loss on ignition samples for all sites during the study period were 4.5 mg/L (Site 1), 3.8 mg/L (Site 2), 4.7 mg/L (Site 3), 3.5 mg/L (Site 4), and 4.5 mg/L (Site 5). In April, concentrations increased between Sites 1 and 3, decreased between Sites 3 and 4, and increased again between Sites 4 and 5. The samples from May and June showed similar trends across sites, with concentrations increasing between Sites 1 and 2, decreasing between Sites 2 and 4, and increasing again between Sites 4 and 5. The June sample showed a larger increase between Sites 4 and 5, paralleling the suspended sediment concentration results and highlighting variability in transport among reaches.

The percent finer (Figure 10) exhibited varying downstream trends across sampling dates. On average, all sites had fines exceeding 80% during the study period. In April, fines decreased between Sites 1 and 2, increased between Sites 2 and 3, decreased between Sites 3 and 4, and slightly increased between Sites 4 and 5. In May, fines increased between Sites 1 and 3 and decreased between Sites 3 and 5. In June, fines increased between Sites 1 and 2, and decreased between Sites 2 and 5, with the most dramatic decrease between Sites 4 and 5.

Suspended Sediment Load Estimates

Suspended Sediment Load Estimates					
Site-Specific Sediment Loads - Total Time Duration of Each Site					
Site #	1	2	3	4	5
Total Time Duration	3/1/24 17:30-9/5 6:30	3/26 13:30-8/29 12:30	3/26 15:30-8/29 14:30	3/27 10:00-8/30 8:00	3/1/24 17:30-9/5 6:30
Number of 30-minute intervals in duration	9002	7486	7486	7484	9002
Suspended Sediment Load (Tons/Total time)	2440	1720	1210	1610	2300
All Sites Sediment Load Comparison - Fixed Time Duration for All Sites					
Site #	1	2	3	4	5
Time Duration	3/27 10:00-8/29 12:30	3/27 10:00-8/29 12:30	3/27 10:00-8/29 12:30	3/27 10:00-8/29 12:30	3/27 10:00-8/29 12:30
Number of 30-minute intervals in duration	7445	7445	7445	7445	7445
Suspended Sediment Load (Tons/Total time)	2160	1720	1200	1610	1910

Table 4: Suspended sediment load estimates for each site. The site-specific loads were calculated with the total available discharge data for each site. The site load comparisons were calculated within a fixed timeframe when all sites had the same available discharge data. Sediment loads are in tons per total time, where the total time duration for the site-specific loads are noted for each site, and the comparison loads have a fixed time duration (3/27/24 10:00 - 8/29/24 12:30) when discharge data were available for all sites.

Table 4 shows site-specific and comparative suspended sediment loads across all sites. Site-specific sediment loads were calculated by summing the total sediment load over the total time duration of available continuous discharge measurements, which differ by site, with time

represented in half-hour units to account for the 30-minute recording intervals. Sites 1 and 5 have the longest study durations due to the availability of gauge data, whereas discharge measurements at Sites 2 through 4 were limited by the timing of ADCP measurements. To facilitate comparisons among sites, the comparative sediment loads were calculated using discharge data from a shared time period (3/27/24 10:00 - 8/29/24 12:30) when all sites had the same timeframe of available continuous discharge data. Given the weak turbidity-suspended sediment correlations at Sites 3 and 5, sediment loads were estimated using the computed total average suspended sediment concentration for the respective site's samples.

The comparative sediment load for the fixed period is initially high at Site 1. This is primarily due to stormflow-driven sediment transport introduced from higher gradient landscapes upstream of the study area. The lower load at Site 2 suggests deposition and/or a reduction in the river's sediment transport capacity, which aligns with the drop in gradient downstream of Site 1. The further decrease at Site 3 may be attributed to additional sediment deposition within the low-gradient river setting; however, a sampling bias at Site 3 missed high suspended sediment concentrations, influencing the lower load estimate. Additionally, the turbidity-suspended sediment and discharge-suspended sediment correlations indicate dynamic sediment transport in this reach, as described further in the Discussion. Nevertheless, Site 4, with better high-concentration sampling, provides a more representative estimate of sediment transport. The overall decline in sediment loads from Site 2 to Site 4 suggests net deposition and/or a decrease in transport capacity between these reaches. In contrast, the sediment load at Site 5 is higher than at Site 4, indicating increased sediment transport in this reach. Similar to Site 3, Site 5's turbidity-suspended sediment and discharge-suspended sediment correlations indicate dynamic sediment transport in this reach, which is further described in the Discussion. These trends suggest alternating zones of sediment deposition and resuspension, with significant sediment contribution at Site 1 and resuspension at Site 5. Further investigation is necessary to identify the key drivers that influence these variations in sediment transport.

DISCUSSION

Site Monitoring

The objectives of this study were to capture current conditions of the Blitzen River, assess the effects of water management on water quality along the Blitzen River, and identify reaches with the highest impacts on water quality. We did this by characterizing spatial and temporal variability in our selected water quality parameters, including turbidity, suspended sediment, water temperature, and streamflow. The findings highlight the influences of seasonal trends, water management, and sediment transport processes on water quality.

Influences on turbidity and sediment transport

Turbidity followed distinct seasonal patterns (Figure 4), decreasing from upstream to downstream during stormflows but increasing during baseflows. During stormflow events,

sediment was introduced into the system, with downstream attenuation suggesting deposition due to reduced flow energy and sediment transport capacity. In contrast, baseflow conditions showed increasing turbidity downstream, indicating sediment resuspension, particularly following dam operations that released previously deposited sediment. The extent of return flow contributions to overall water quality in the river remains uncertain, particularly given the increased sediment availability in the channel due to dam-induced resuspension.

The impact of return flows on water quality remains uncertain due to overlapping influences and limited coverage during key periods. Return flows released between Sites 3 and 4 from July 8th through July 19th coincided with the opening of Grain Camp Dam, obscuring any distinct turbidity changes at Site 4. The release of Krumbo Canals, between Sites 2 and 3, on July 19th, was the only canal return flow that could be delineated as impacting the turbidity record. The loss of baseflow data at Site 2 limited coverage of southern return flows during July. Additionally, the increase in turbidity observed at Sites 3 and 4 in August did not align with the timing of return flows, suggesting that other factors contributed to this increase. This is evident, as when data for Site 2 were available, this site showed low turbidity, whereas Sites 3 and 4 exhibited high turbidity. While direct sampling during the Skunk Farm Canal opening at Site 4 confirmed that return flows introduce turbidity, evidenced by the rejected QAQC sample (Table 3) with initially high but rapidly diminishing concentrations, their overall influence on river water quality remains difficult to quantify. The next phase should measure specific return flows to better quantify their influence in reaches with increased turbidity.

Dam operations had a pronounced influence at Sites 3 and 5, where sharp increases in turbidity coincided with the opening of the Grain Camp and Sodhouse Dams (Figure 4). Following the Grain Camp Dam release in early July, Site 3 (and subsequently Site 4) displayed increased turbidity until levels subsided by the end of the month. Site 5 also shows a dynamic increase in turbidity after the Sodhouse Dam release in August; however, it is the prior increase in turbidity midway through baseflow that is most notable, but challenging to characterize. The increase in turbidity coincided with the highest recorded water temperatures in July for this site. The August 14 canoe survey also recorded concurrent increases in turbidity and temperature along the reach (Figure 16), despite no surface water inputs at the time. This pattern suggests algal growth as a contributing factor to the increase in turbidity in this reach. However, consistently high suspended sediment concentrations and nonlinear relationships with streamflow indicate the presence of additional, complex sediment transport and deposition processes in this reach.

Inconsistencies in the correlations between turbidity, suspended sediment, and discharge at specific sites require further examination. The weak turbidity-sediment correlations (Figure 7) at Sites 3 and 5 are somewhat anomalous. At Site 3, this discrepancy is likely due to a sampling bias since we did not capture high sediment flows. However, the additional weak correlation at Site 5 suggests factors beyond sampling limitations, such as variations in particle size distributions and organic matter contributions (Schenk et al., 2014), and should be examined further in the next phase of the study. Furthermore, future studies should prioritize same-day

water sampling across sites to enhance the comparability of longitudinal changes in water quality between sites.

Several dam-related factors also likely contribute to the weak correlations between suspended sediment and discharge (Figure 8) at Sites 3 through 5, including flow alterations that reduce sediment transport capacity (Keith et al., 2024), dam trapping efficiencies (Keith et al., 2024; Grant, 2003; Schenk et al., 2014), the timing of dam drawdowns (Keith et al., 2024; Schenk et al., 2014), and the geographical settings of the dams (Grant, 2003). These factors interact and produce dynamic sediment-discharge responses. Flow modifications by dams can reduce transport capacity, promoting sediment accumulation, while drawdowns during low flows may further influence sediment transport and deposition patterns when transport capacity is low. Predicting sediment transport dynamics in the refuge is challenging, as most existing literature focuses on large-scale hydropower dams rather than smaller irrigation dams, which can exhibit trapping efficiencies ranging from 10% to 90%, depending on sediment size (Grant, 2003). The geographical setting of the dams also influences sediment dynamics. Sites 3 and 5, situated in the valley lowlands, show different responses compared to Site 1, where the river exits the high-gradient terrain of Steens Mountain.

Beyond dam influences, legacy effects and the geomorphological responses, such as lateral and vertical incision, further complicate sediment supply and availability, affecting transport rates across different reaches for varying streamflows (Lane and Richards, 1996; Simon and Rinaldi, 2006; O'Connor et al., 2015). The U.S. Environmental Protection Agency (EPA) identifies channelization and damming as activities that can lead to resuspension and bank erosion, thereby altering the in-channel sediment load and resulting in excess turbidity (EPA, 2006). Given the extensive disturbance history of the Blitzen River, particularly due to channelization and the presence of dams, it remains uncertain whether ongoing channel morphological adjustments continue to impact sediment transport. These larger-scale uncertainties are beyond the scope of the present study but highlight the complexity of sediment dynamics in this system that may be affecting discharge-sediment relationships.

Influences on water temperature

Water management influenced water temperature trends during baseflows by altering longitudinal temperature variations at specific sites (Figure 5). As an important consideration for the refuge, the Oregon Department of Environmental Quality (ODEQ) established a temperature criterion for streams supporting redband trout, specifying that the seven-day-average maximum temperature should not exceed 20.0 degrees Celsius (Oregon Department of Environmental Quality, 2008). Sites 2 through 5 exceeded this threshold throughout the baseflow period, as shown in the appended seven-day-average temperature graph (Appendix: Site Graphs, Figure 32). This temperature exceedance occurred before most return flows, which suggests additional environmental factors influenced water temperature, such as riparian shading, channel morphology, or other reach-specific conditions. The distinct temperature deviation of Site 3 following the opening of Grain Camp Dam contrasts with the temperature at Site 2 during this

time, suggesting that return flows or other channel conditions between these sites are a primary driver for temperature increases at Site 3. The temperature increases observed at Site 4 are largely influenced by conditions at Site 3 but are diluted by tributary inputs between the sites. The data also suggest dams at Sites 3 and 5 buffer temperature fluctuations, though this effect is not observed with the dam at Site 1. This is likely due to stratification behind the dams at Sites 3 and 5, where water flows only beneath the dam. In contrast, at Site 1, the ability of water to flow both over and under the dam likely promotes mixing throughout the water column, reducing the effects of stratification.

Influences on streamflow

Numerous efforts have attempted to estimate the volume of water diverted by the refuge for irrigating wetland and wet meadow habitats on the floodplain of the Blitzen River in MNWR, but these efforts have had limited success. Currently, flow measurements are collected at only three locations on the southernmost part of the refuge near Page Springs Dam, and then again nearly 45 river miles downstream below the Sodhouse Dam. The limited monitoring results in a poor ability to quantify water usage and streamflow dynamics along the extent of the Blitzen River in MNWR.

Direct measurement of diversions is challenging due to several factors. Many of the diversion channels consist of standing water, are densely vegetated, have unsafe water depths, or experience changing water levels, making the use of velocimeters or transducers to measure streamflow ineffective. To overcome these limitations, we employed an alternative approach, indirectly estimating inputs and outputs using an ADCP to measure streamflow at selected mainstem site locations and applying water budget equations to quantify net gains and losses between sites. This mass balance approach provided valuable insights into streamflow dynamics that have historically been difficult or impossible to quantify.

Streamflow patterns followed expected seasonal trends, with peak flows during spring snowmelt and reduced flows during summer baseflows (Figure 6). Discharge to diversions was highest during stormflows and lowest or absent during baseflows. Observed discharge reductions at Sites 1, 3, and 5 corresponded with known diversions at Page Springs, Grain Camp, Busse, and Sodhouse Dams, demonstrating the impact of dams and diversions on flow regimes. In July, discharge at Sites 2 through 4 was slightly elevated compared to other baseflow measurements, coinciding with irrigation return flows that occurred throughout the month. Throughout the study, water diversions remained within the diversion capacities specified in the MNWR Comprehensive Conservation Plan (U.S. Fish and Wildlife Service, 2013).

Implications

Overall, these findings indicate that seasonal patterns, water management, and sediment transport processes collectively drive site-specific water quality trends. The reaches of Sites 3 and 5 appeared to be particularly impacted because of temperature and turbidity patterns that deviated from those observed at other sites. The data suggest that baseflow conditions should be

a primary focus of water quality management, while stormflow conditions provide insight into broader sediment dynamics and their role in introducing sediment into the system. It's also important to acknowledge the temporal restriction of our data collection, which excludes the fall and winter seasons, and may limit our understanding of sediment transport processes across the system throughout the year.

Water Quality Assessment and Management

The U.S. Environmental Protection Agency (EPA) designates fluvial sediment as the most common pollutant in the nation's surface waters (EPA, 2006). Sediment distributions vary by watershed, making it critical to understand sediment and streamflow budgets for maintaining water quality, habitat conservation, and effective land management. Given these challenges, this study plays an important role in establishing high-quality turbidity and streamflow data for the Blitzen River, thereby providing a foundation for improved monitoring and management. Turbidity is widely used as a proxy for suspended sediment concentration and as an indicator of the environmental health of water bodies (Schenk et al., 2014; Rasmussen et al., 2009; Anderson, 2005). Turbidity itself is not an inherent water property but rather a measure of light attenuation by suspended and dissolved particles, such as clay, silt, organic matter, algae, and other microscopic organisms (Schenk et al., 2014; Anderson, 2005). Elevated turbidity can degrade water quality and aquatic health by increasing temperature, reducing dissolved oxygen, increasing nutrient concentrations if caused by algal growth, clogging gills and disrupting the feeding habits of fish or filter-feeders, and decreasing light penetration and therefore the ability for submerged plants to photosynthesize (EPA, 2006).

While turbidity and suspended sediment are essential indicators of environmental health, no single criterion is universally applicable due to watershed-specific variability. Consequently, turbidity criteria often vary by state, country, water body, species, and water body use (EPA, 2006). The EPA recognizes Oregon's turbidity criterion, which limits increases to 10% above natural stream levels relative to an upstream control point (EPA, 2006). Although this criterion primarily addresses point sources, applying it to percent differences in turbidity between sites during baseflow provides a helpful context for site comparisons. During baseflow, the percent differences in mean turbidity between consecutive sites were 24% (Sites 1-2), 65% (Sites 2-3), -66% (Sites 3-4), and 74% (Sites 4-5), highlighting the variability in turbidity along the river.

Further complicating turbidity assessment is that different measurement methodologies affect data comparability. The EPA and other outlets for criteria often report suspended sediment criteria using total suspended solids, also called non-filterable residue, which cannot be directly converted to suspended sediment concentration unless both are measured concurrently at the same site (Glysson and Gray, 2000). The EPA also reports turbidity criteria in nephelometric turbidity units (NTU) using EPA Method 180.1 (EPA, 2006), whereas the USGS generally reports turbidity in formazin nephelometric units (FNU; Anderson, 2005), as reported in this study. Due to differences in instrument design and calibration, NTU and FNU values are not

interchangeable, and measurements from different turbidimeters may yield inconsistent results (Gray and Glysson, 2003).

Turbidity criteria provide a relative framework for assessing the impacts of observed turbidity in the MNWR, although defining specific thresholds remains challenging due to variations in NTU and FNU measurements. Regional references and regulatory guidelines offer valuable benchmarks for evaluating turbidity levels. For example, the EPA defines regional reference conditions for Ecoregion III (Xeric West), which includes the Northern Great Basin (Level III Ecoregion ID 80), reporting a 25th percentile turbidity of 2.325 NTU based on decadal data (EPA, 2000). This value is relative to the 25th percentile turbidity levels at Sites 1, 2, and 4 over the study period, which were 2.0, 2.7, and 3.1 FNU, respectively. In contrast, Sites 3 and 5 had higher 25th percentile values of 5.2 and 9.6 FNU, respectively. The EPA also published water quality guidelines for British Columbia, which include aquatic life concentration-duration frequency thresholds. These guidelines state that during clear flows (defined as the time in which streams are typically clear, opposed to turbid, similar to this study's baseflow), turbidity increases should not exceed 8 NTU over 24 hours or 2 NTU over 30 days, while during high flows or turbid conditions, increases should not exceed 5 NTU when background levels are 8-50 NTU or 10% when background levels exceed 50 NTU (B.C. Ministry of Environment and Climate Change Strategy, 2021). Although we were unable to calculate concentration-duration frequencies in our present study, this method should be considered for baseflow assessments in the next phase of research.

The Oregon Department of Environmental Quality (ODEQ) further refines EPA turbidity values for macroinvertebrate health during low flows, classifying turbidity <4 NTU in the 75th percentile as "good" and turbidity >13 NTU in the 95th percentile as "poor" (Oregon Department of Environmental Quality, 2014). Although ODEQ does not specify turbidity units, consistency with referenced EPA data is assumed. The 95th percentile turbidity values at Sites 1 through 5 during baseflow were 3.7, 5.6, 14.1, 8.5, and 32.2 FNU, respectively. However, direct comparisons across datasets are again complicated by differences in measurement methodologies, as NTU and FNU are not directly interchangeable.

Despite the limitations in measurement methodologies and watershed-specific benchmarks, turbidity criteria provide a valuable relative metric for assessing water quality and evaluating deviations among site conditions. While direct comparisons across methodologies are not applicable, regional reference values and criteria guidelines help contextualize turbidity trends and highlight potential impacts. These guidelines helped identify turbidity trends at Sites 3 and 5 as differing from those at other sites, suggesting greater impacts of turbidity at these sites. Additionally, the present study establishes an initial baseline for turbidity in the Blitzen River, providing critical insights into sediment dynamics and potential impacts. These baseline data are crucial for tracking long-term trends, refining water quality assessments, and informing future management strategies aimed at mitigating sediment-related impacts.

CONCLUSION

This study highlights the spatial and temporal variability in turbidity, suspended sediment, water temperature, and streamflow in the Blitzen River of the Malheur National Wildlife Refuge. This report presents findings from the first year of our research conducted in 2024, with a second phase planned for 2025. Throughout the river system, complex interactions between seasonal variability, water management, and sediment transport determined water quality conditions. Stormflow conditions generally contributed to sediment deposition downstream due to reduced flow energy and sediment transport capacity. In contrast, baseflow conditions showed increased turbidity downstream, primarily due to resuspension, particularly from dam operations and possibly return flows. The degree of return flow's impact on the overall turbidity in Blitzen's mainstem was not readily apparent, and future studies should attempt to measure specific return flows to assess their influence in impacted reaches.

The turbidity trends at Sites 3 and 5 showed deviations from those at other sites, which may be indicative of enhanced impacts from water management, based on available turbidity criteria. Additionally, the seven-day average temperatures at Sites 2 through 5 exceeded 20°C throughout baseflow, which may have ecological implications for aquatic habitats. Return flows or channel conditions in Site 3's reach during July may have further increased the temperature variations observed at Sites 3 and 4. These influences highlight the broader challenges of balancing floodplain habitat conservation with maintaining mainstem hydrologic and water quality stability, where baseflow conditions appear particularly critical for effective water quality management. In addition, given the dynamic nature of sediment transport and deposition, the study's temporal scope, which focuses on the irrigation season, may miss key trends during the fall and winter when sediment dynamics may shift.

The broader implications of this study align with national concerns over sediment as a leading pollutant in surface waters, with turbidity serving as a valuable yet complex proxy for assessing suspended sediment and overall water quality. The difficulty in applying standardized turbidity criteria to the MNWR system highlights the need for site-specific evaluation and continued refinement of watershed baselines, as this study has sought to do. Weak correlations between turbidity, suspended sediment, and streamflow at specific sites suggest that sediment transport processes are influenced by multiple interacting factors, including dam operations, particle type distributions, and geomorphological considerations. Future efforts should prioritize water quality management during baseflow conditions and consider rehabilitation in impacted reaches. Additionally, further investigation into the effects of specific irrigation return flows and sediment dynamics within impacted reaches may help clarify their role in water quality trends observed across sites, particularly for Sites 3 and 5. These findings are meant to provide insights for resource managers seeking to mitigate water quality impacts while maintaining essential floodplain habitat functions within the refuge. Continued research and adaptive management strategies will be essential in pursuing water quality goals and balancing habitat management objectives for the Blitzen River system.

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APPENDIX

Site Maps:

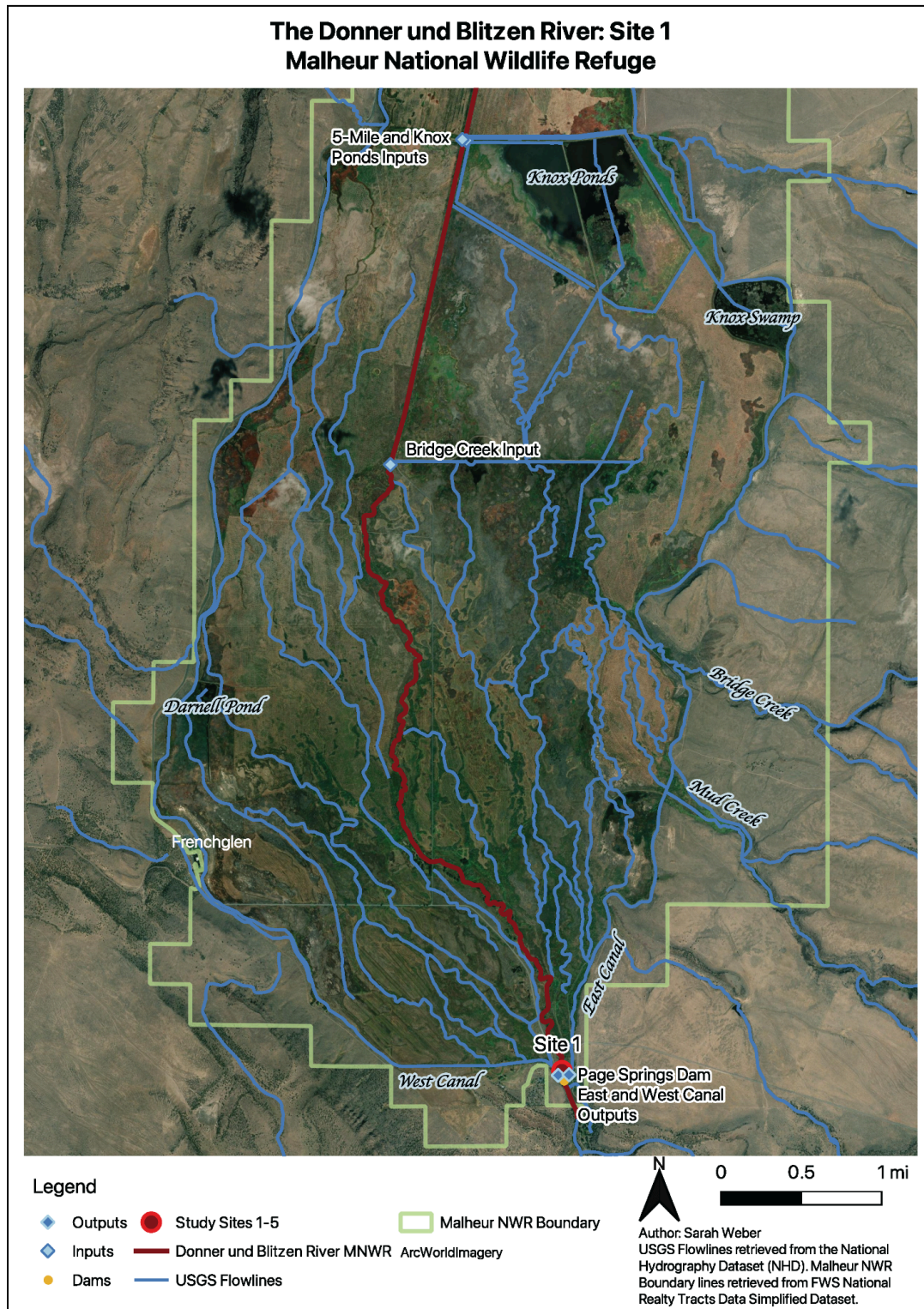
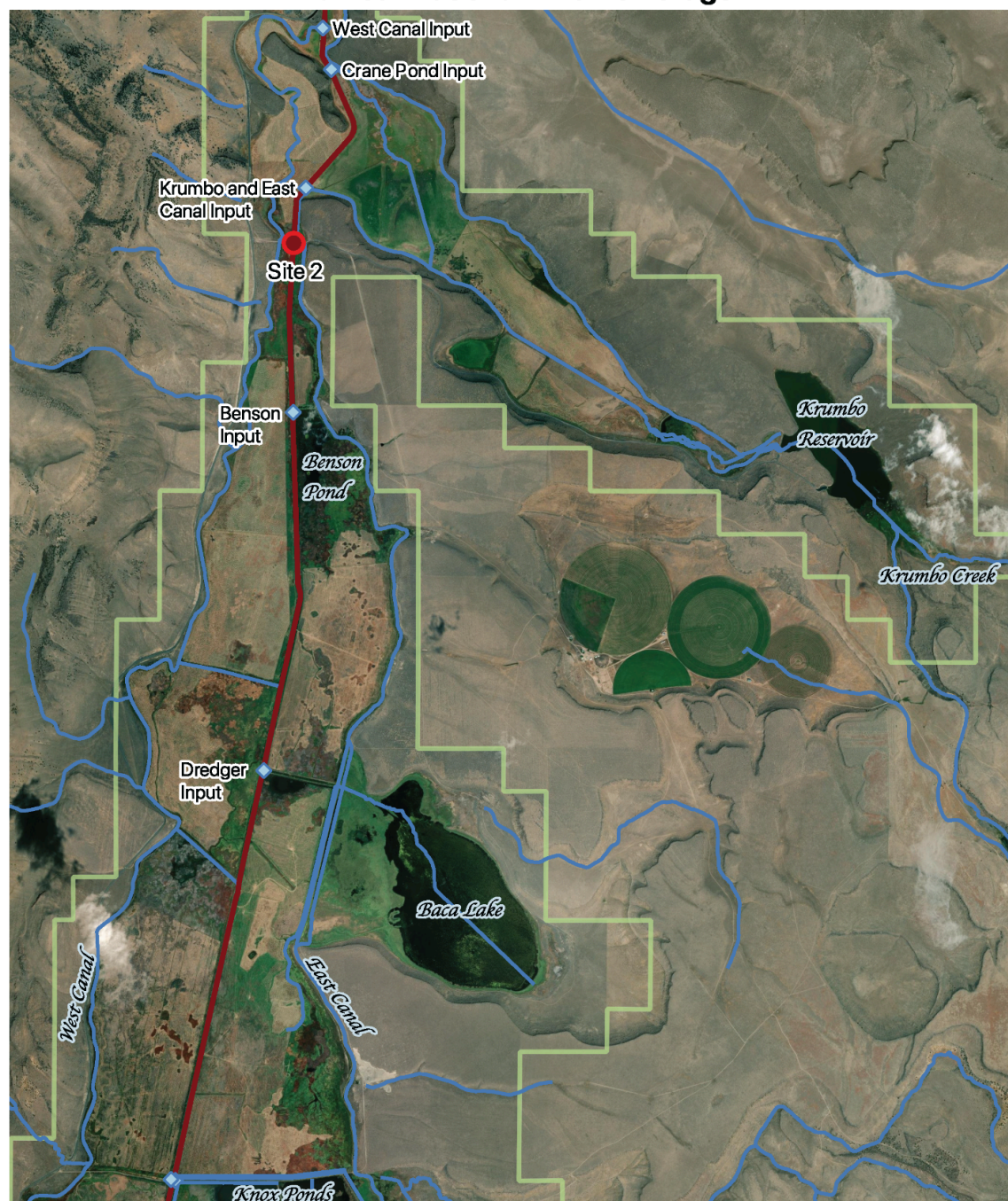


Figure 11: Site 1 map.

The Donner und Blitzen River: Site 2 Malheur National Wildlife Refuge



Legend

- ◆ Inputs
- Study Sites 1-5
- Donner und Blitzen River MNWR
- USGS Flowlines
- Malheur NWR Boundary
- ArcWorldImagery



0 0.5 1 mi

Author: Sarah Weber
USGS Flowlines retrieved from the National Hydrography Dataset (NHD). Malheur NWR Boundary lines retrieved from FWS National Realty Tracts Data Simplified Dataset.

Figure 12: Site 2 map.

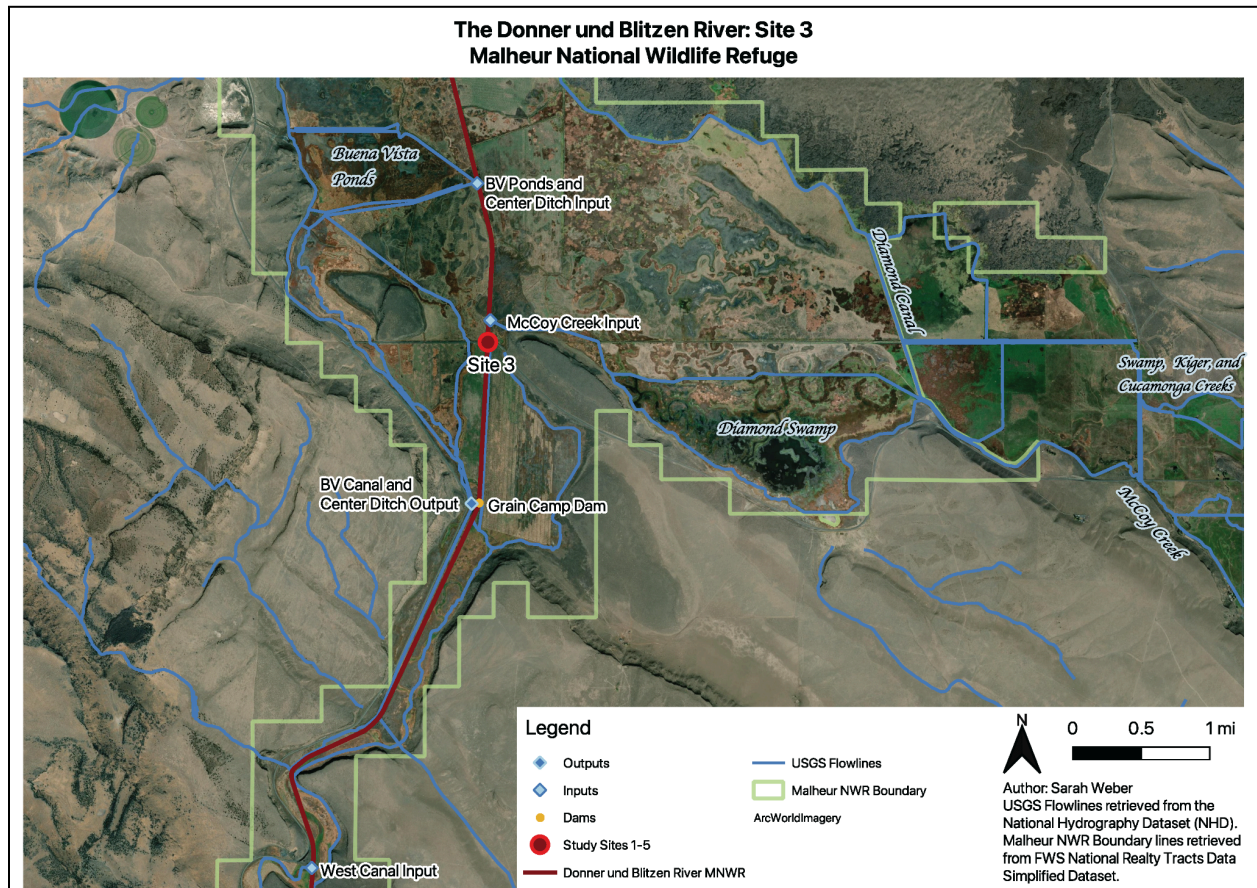


Figure 13: Site 3 map.

The Donner und Blitzen River: Site 4 Malheur National Wildlife Refuge

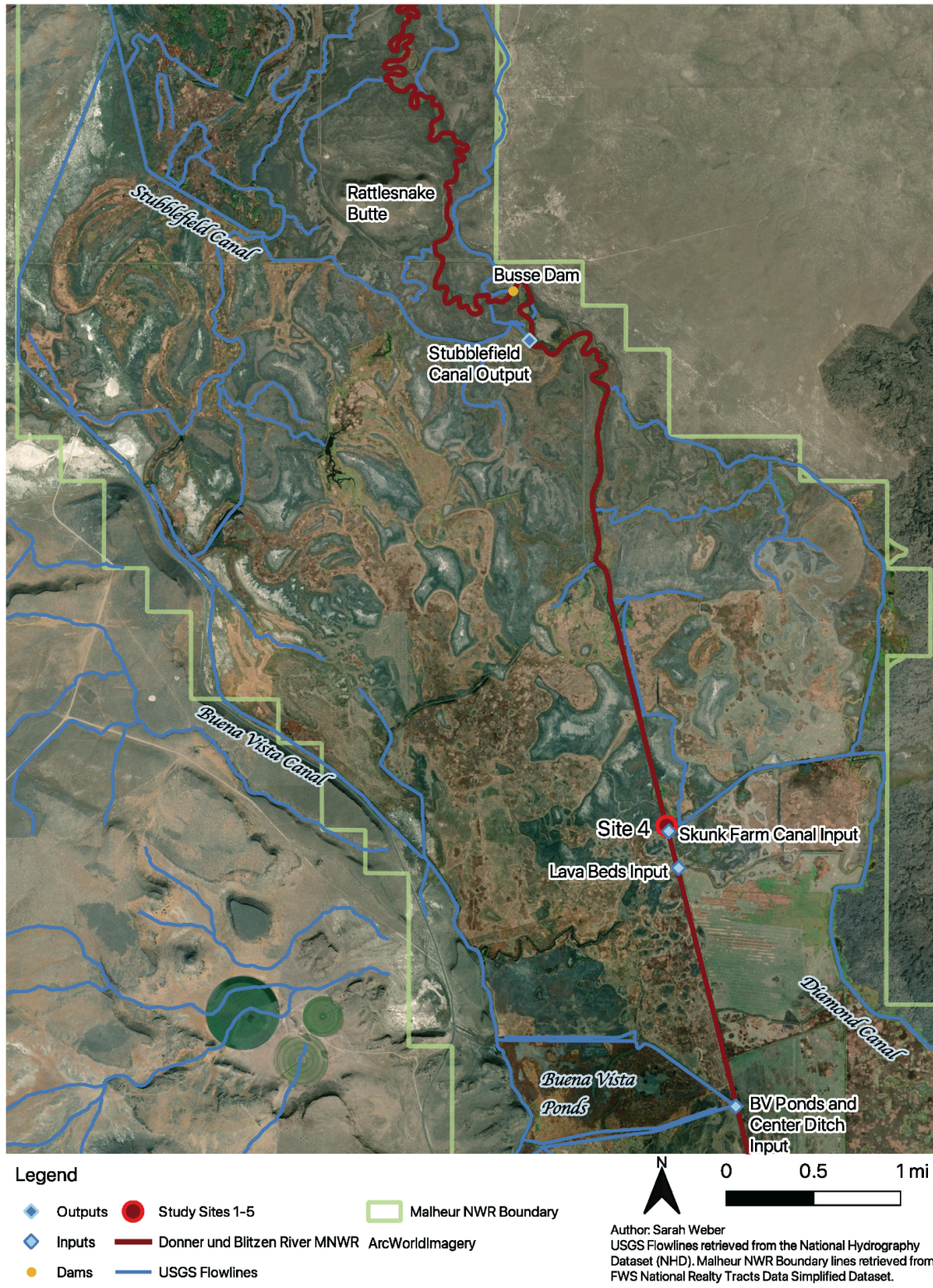


Figure 14: Site 4 map.

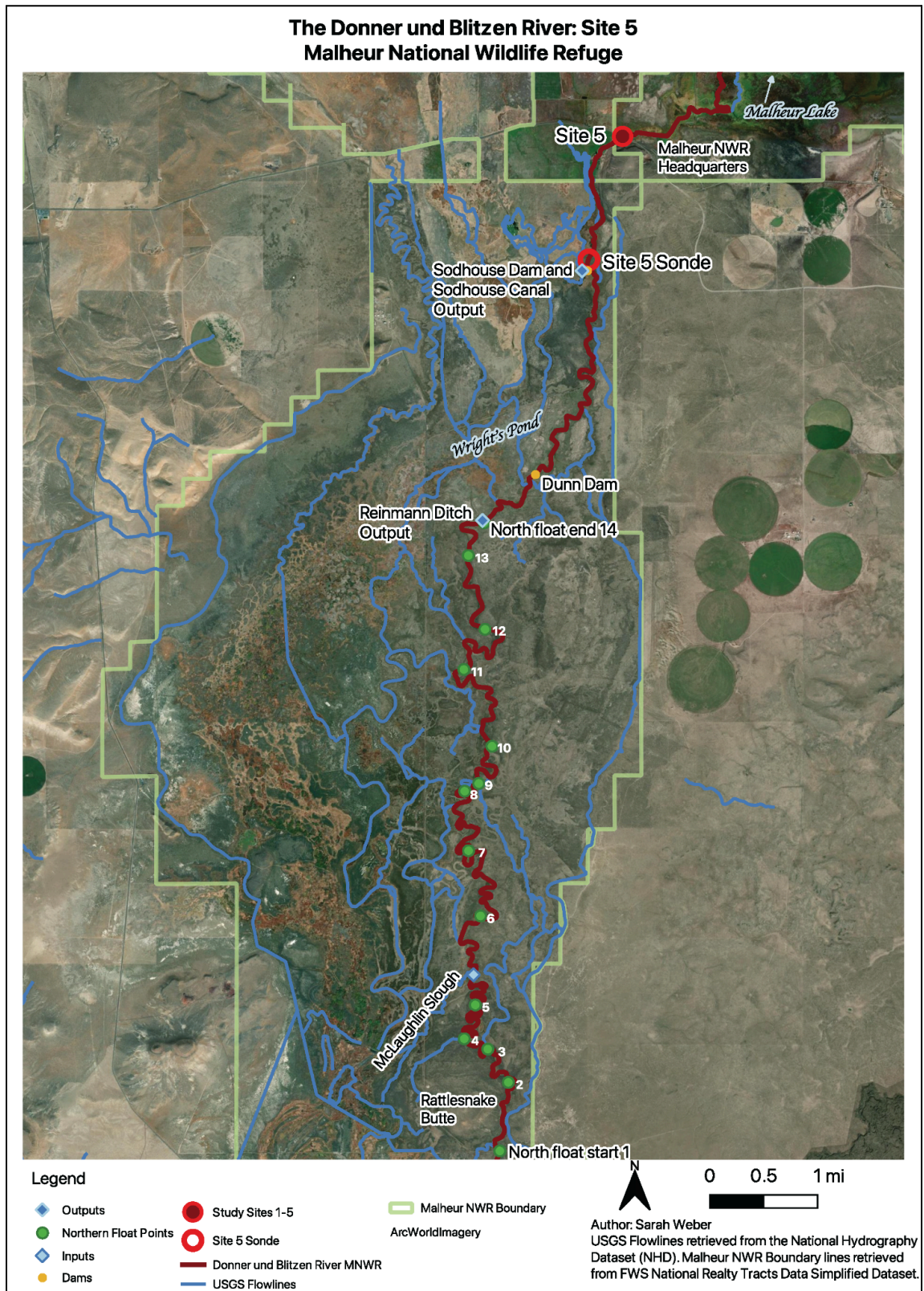


Figure 15: Site 5 map. Numbered points represent recorded GPS locations for the canoe trip point results on August 14, 2024. (Data provided in Figure 16 and Table 5).

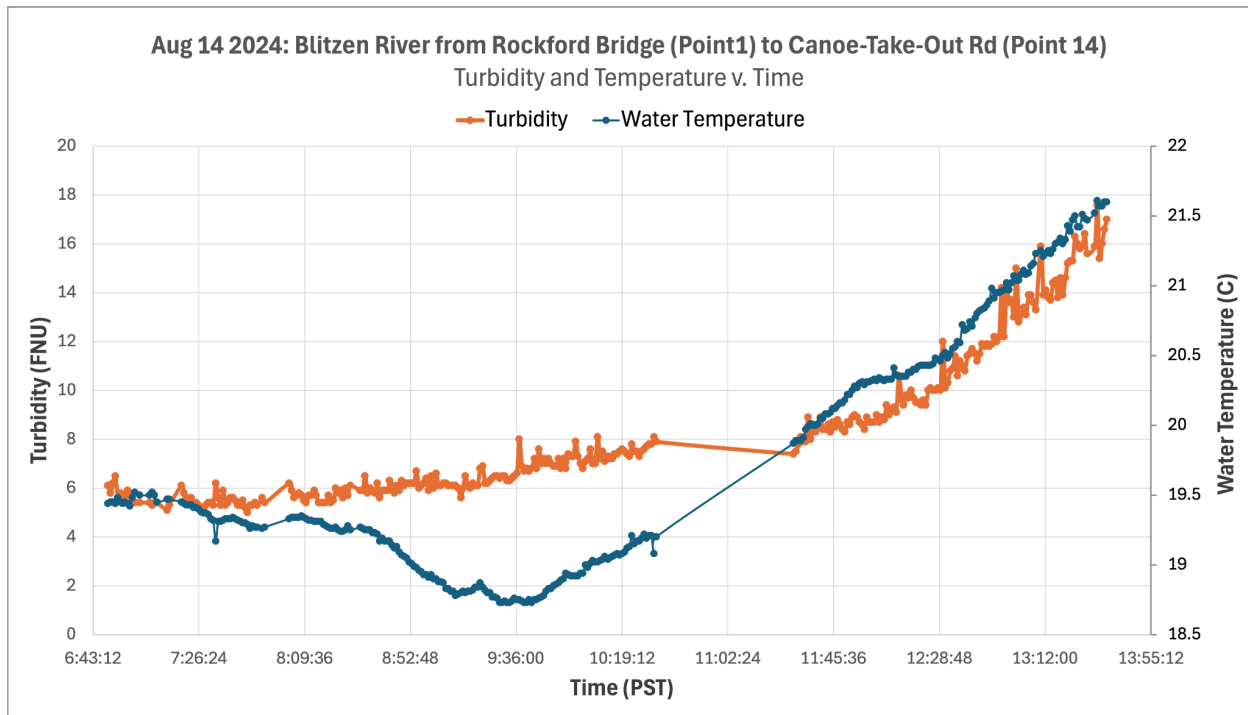


Figure 16: Turbidity and water temperature over time for the canoe trip from Rockford Bridge to Canoe-Take-Out Road on August 14, 2024. Missing values from 10:30-11:30 for a lunch break.

Point	Time (PST)	Latitude	Longitude	Turbidity (FNU)	Temperature (°C)	Miles from Start
1 (Start)	6:50	43.1319	-118.873933	5.8	19.5	0
2	7:13	43.1412	-118.872333	5.1	19.5	0.8
3	7:34	43.145483	-118.87615	5.6	19.3	1.5
4	8:03	43.14705	-118.880567	6.2	19.3	1.97
5	8:26	43.15155	-118.8784	6	19.3	2.89
6	9:11	43.163333	-118.877267	6.1	18.8	4.72
7	9:51	43.172217	-118.879767	6.9	18.9	6.4
8	10:25	43.180217	-118.880433	7.5	19.2	7.8
9	11:29	43.181067	-118.877783	7.4	19.9	8.04
10	11:41	43.186283	-118.87535	8.4	20.1	8.78
11	12:13	43.196433	-118.8805	9.6	20.4	10.1
12	12:45	43.201833	-118.87645	11.5	20.8	11.24
13	13:17	43.21165	-118.879883	13.8	21.3	12.17
14 (End)	13:37	43.216183	-118.876983	17	21.6	12.77

Table 5: Point results from the August 14 canoe trip. Point numbers are correlated to point numbers on the Site 5 map (Figure 15).

Site Graphs:

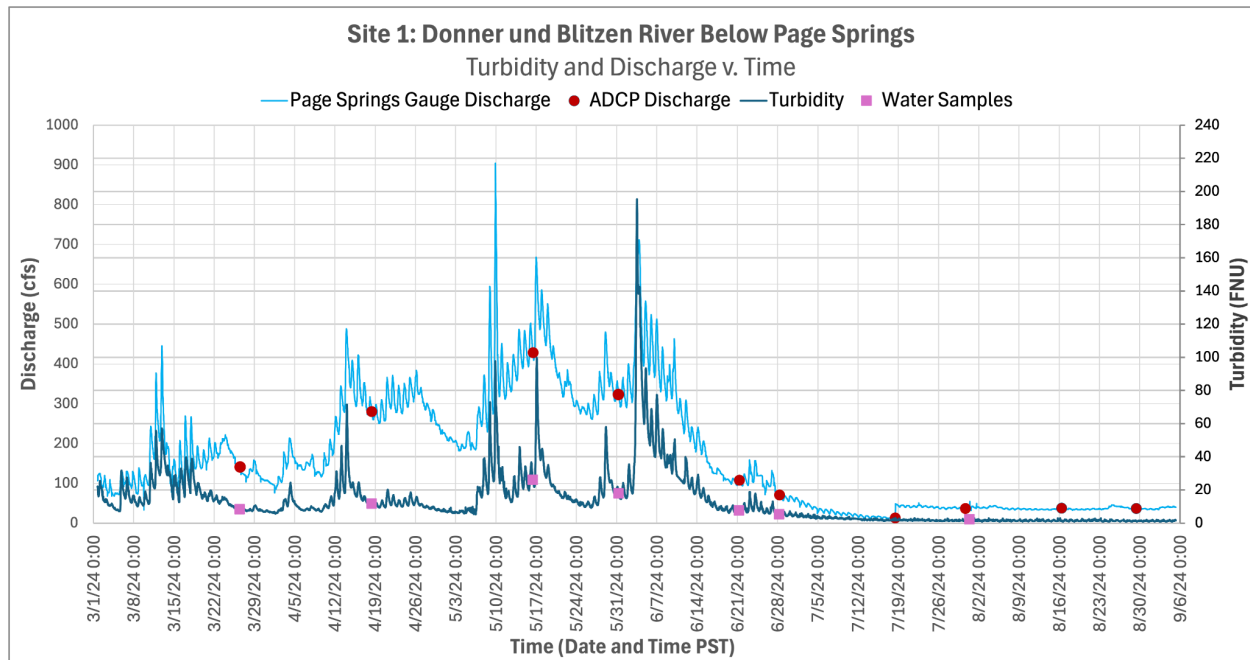


Figure 17: Site 1 turbidity and discharge versus time, with ADCP measurements and water samples.

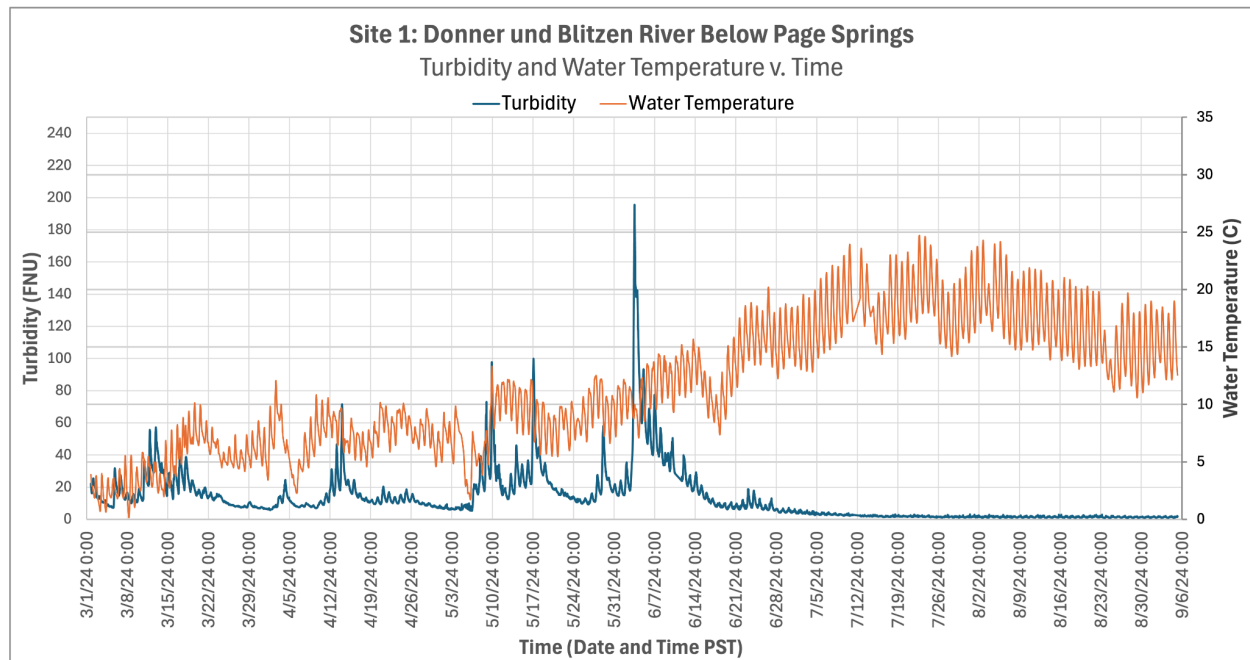


Figure 18: Site 1 turbidity and water temperature versus time.

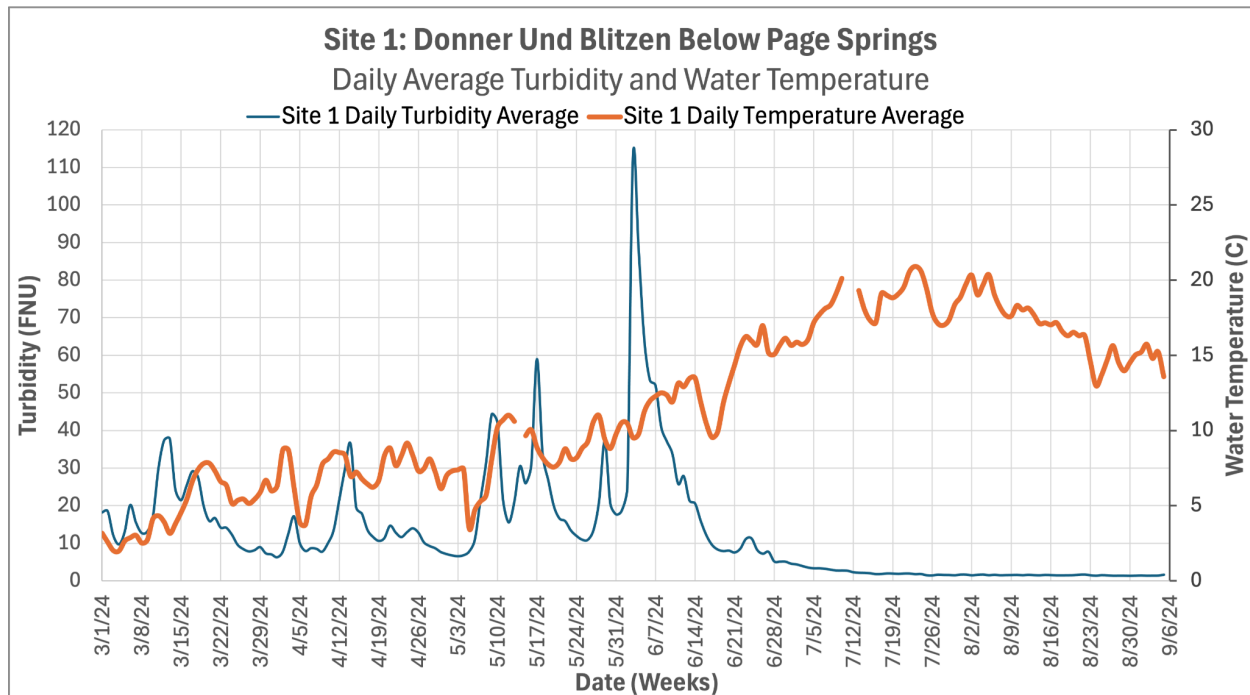


Figure 19: Site 1 daily average turbidity and water temperature versus time, with two gaps in water temperature data on days when sondes were taken to lab for calibration.

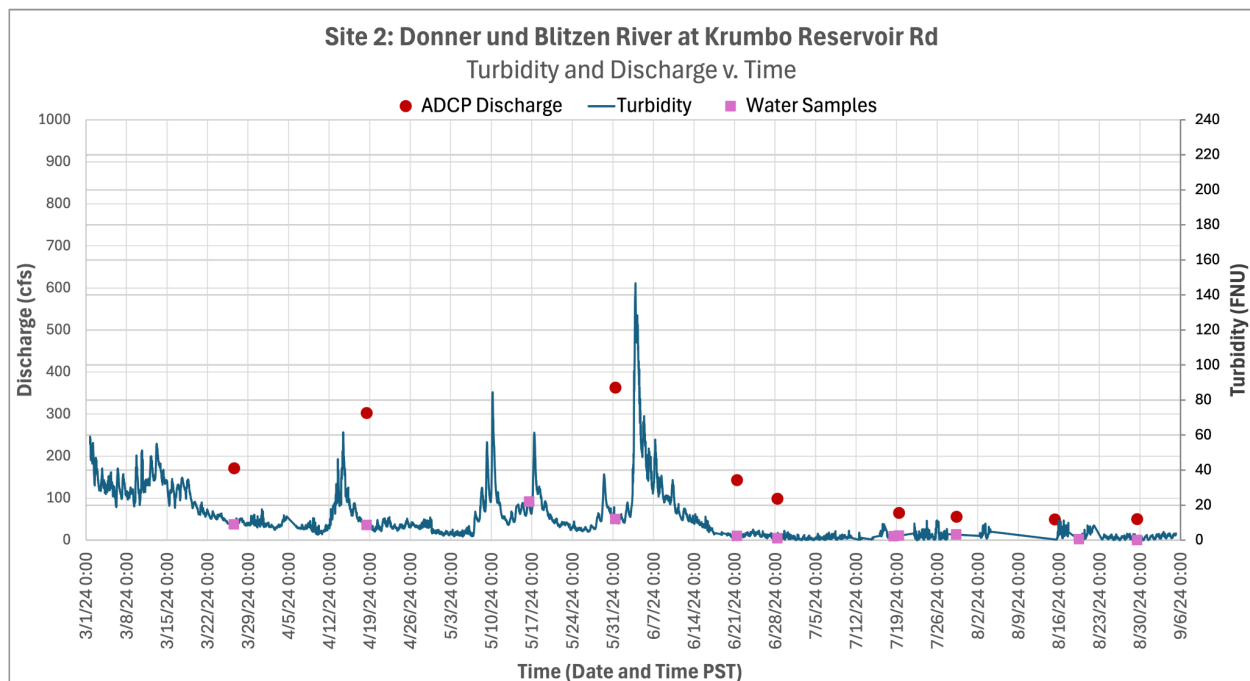


Figure 20: Site 2 turbidity and discharge versus time, with ADCP measurements and water samples.

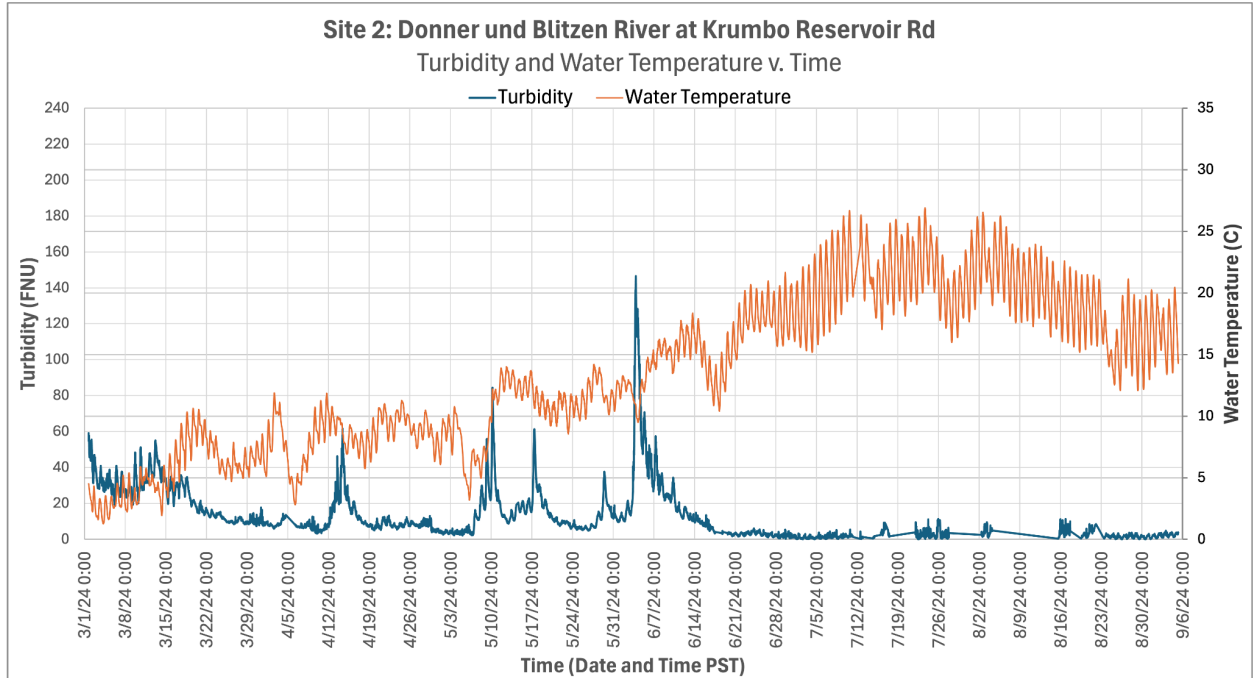


Figure 21: Site 2 turbidity and water temperature versus time.

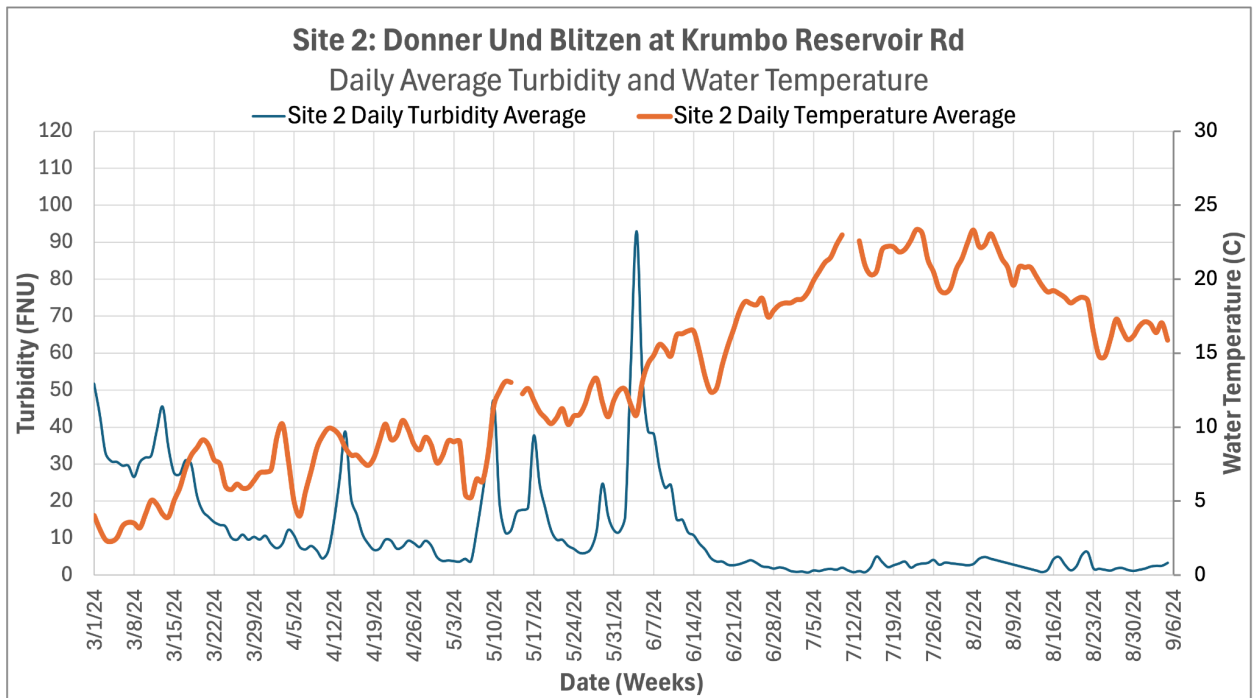


Figure 22: Site 2 daily average turbidity and water temperature versus time, with two gaps in water temperature data on days when sondes were taken to lab for calibration.

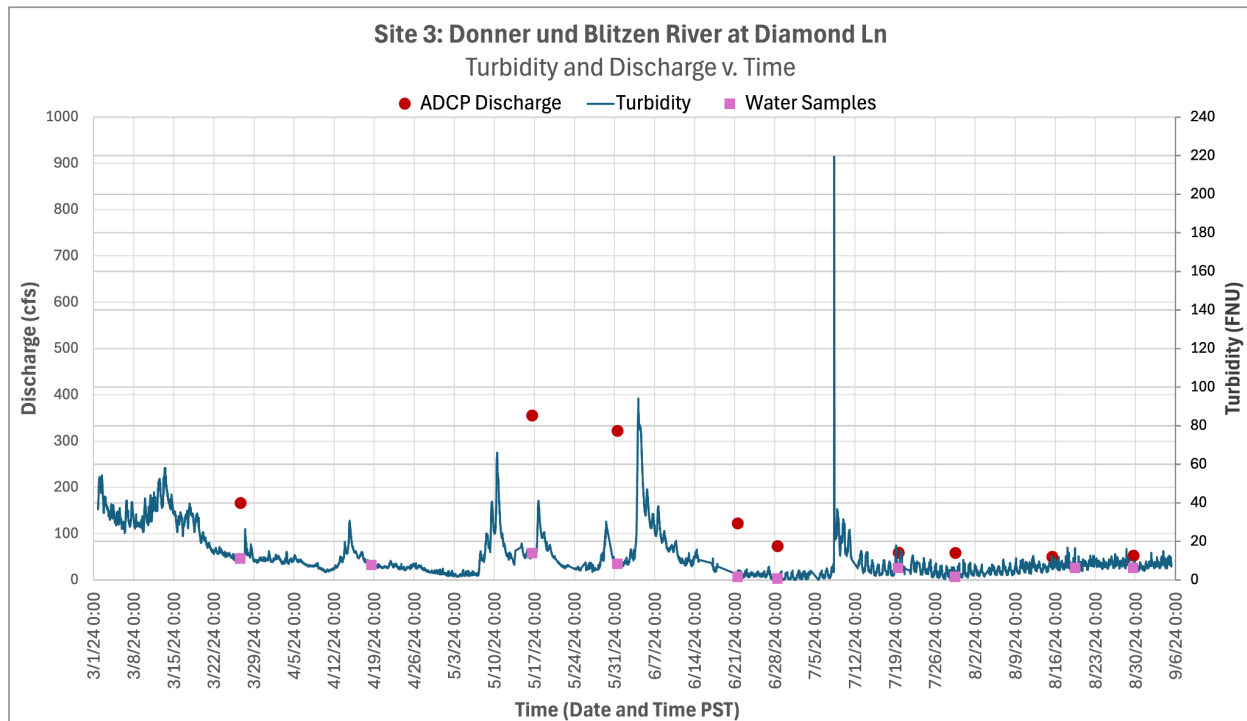


Figure 23: Site 3 turbidity and discharge versus time, with ADCP measurements and water samples.

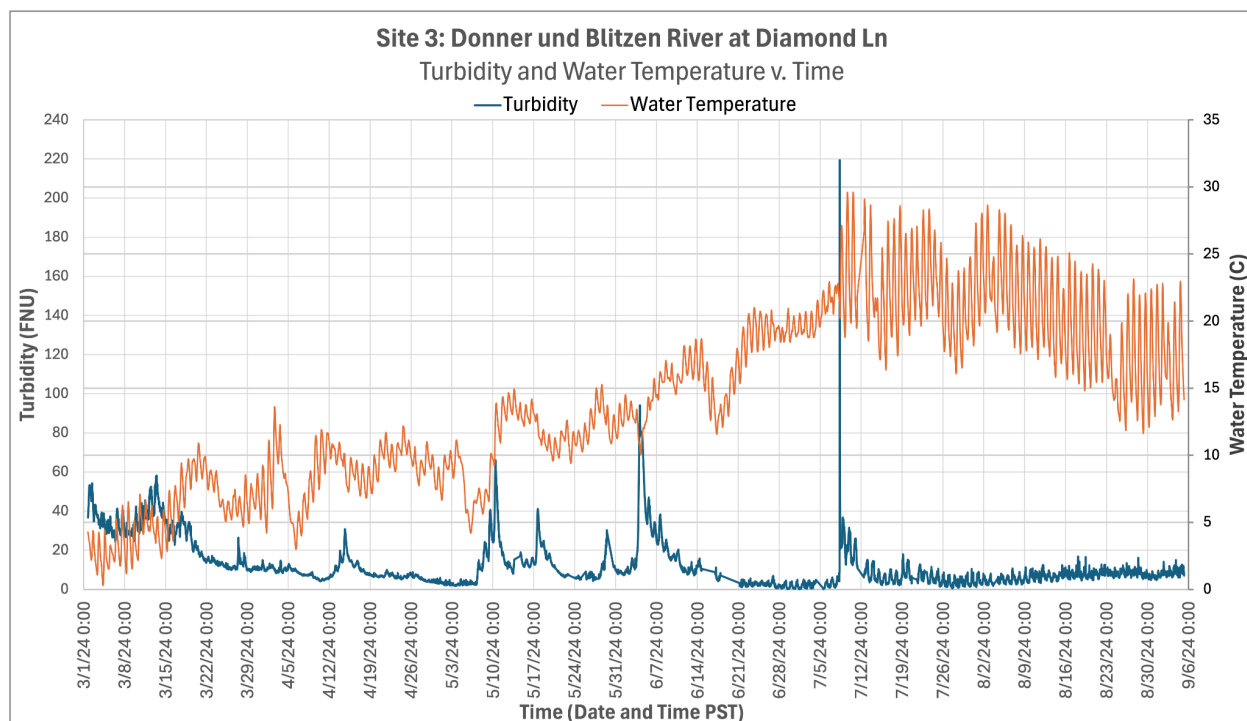


Figure 24: Site 3 turbidity and water temperature versus time.

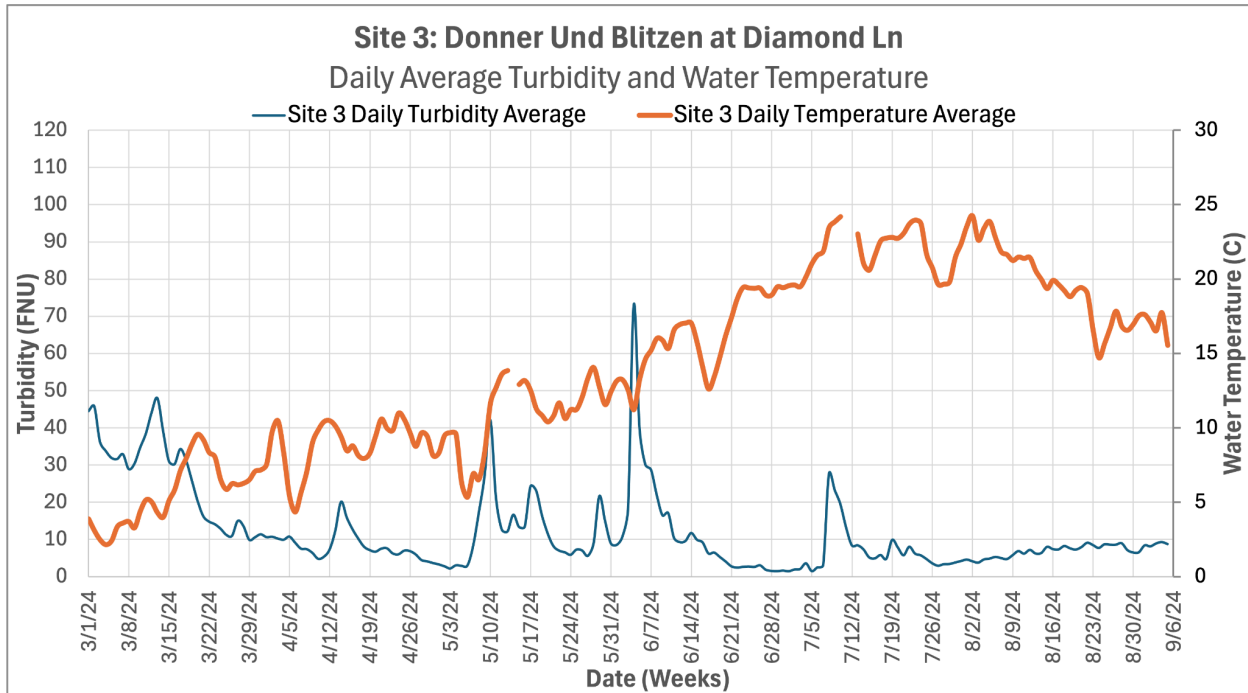


Figure 25: Site 3 daily average turbidity and water temperature versus time, with two gaps in water temperature data on days when sondes were taken to lab for calibration.

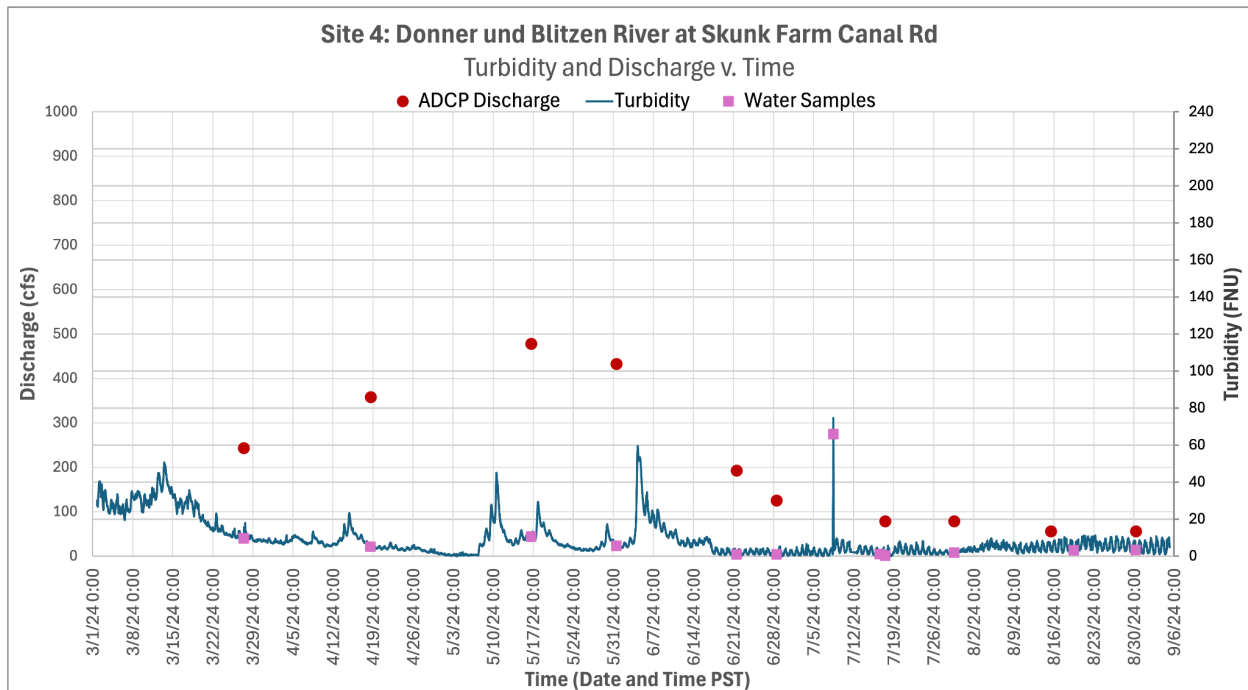


Figure 26: Site 4 turbidity and discharge versus time, with ADCP measurements and water samples.

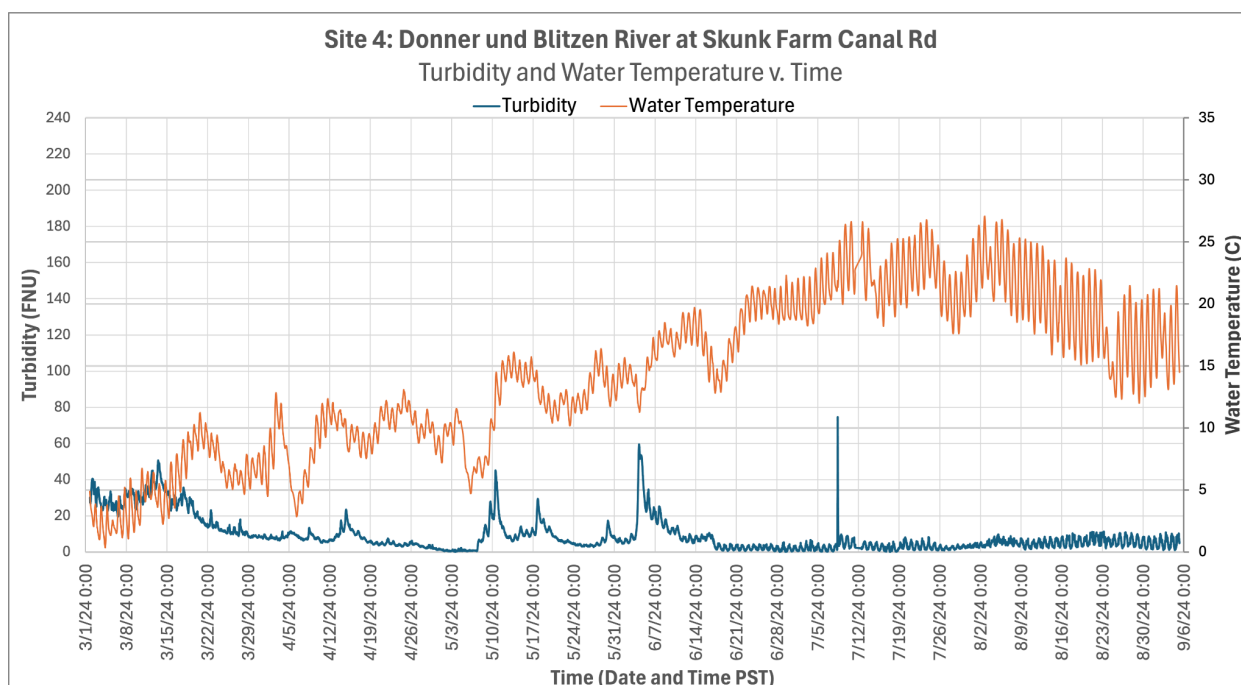


Figure 27: Site 4 turbidity and water temperature versus time.

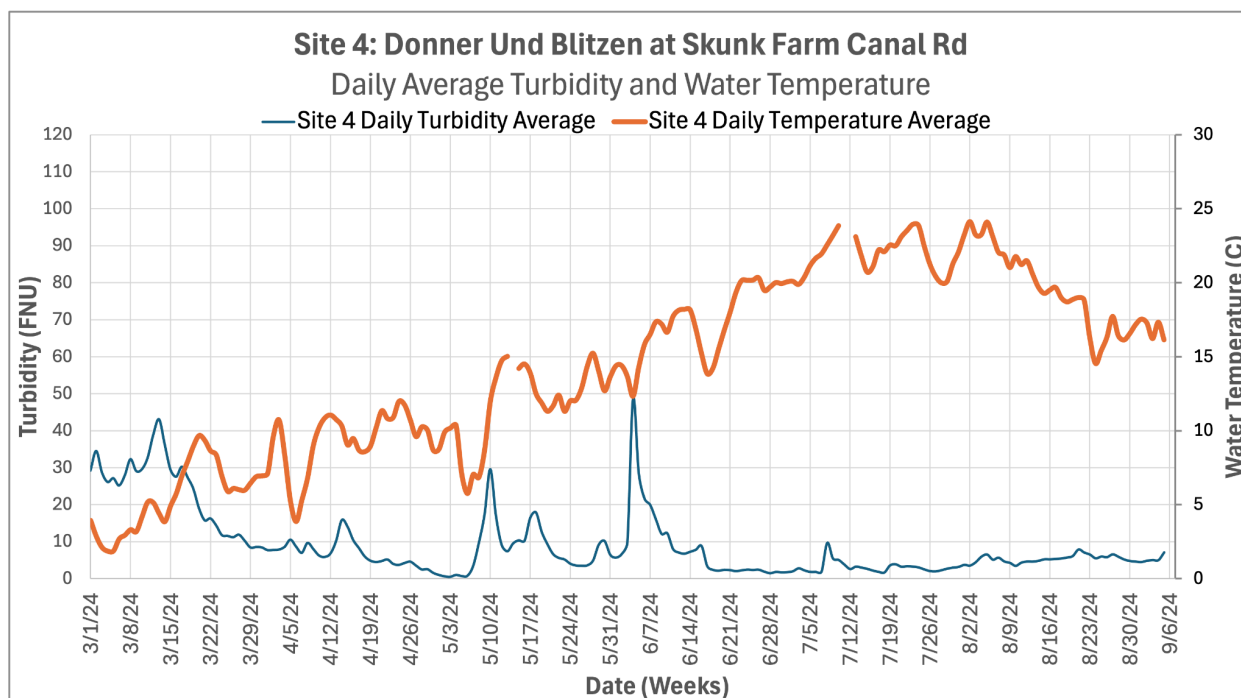


Figure 28: Site 4 daily average turbidity and water temperature versus time, with two gaps in water temperature data on days when sondes were taken to lab for calibration.

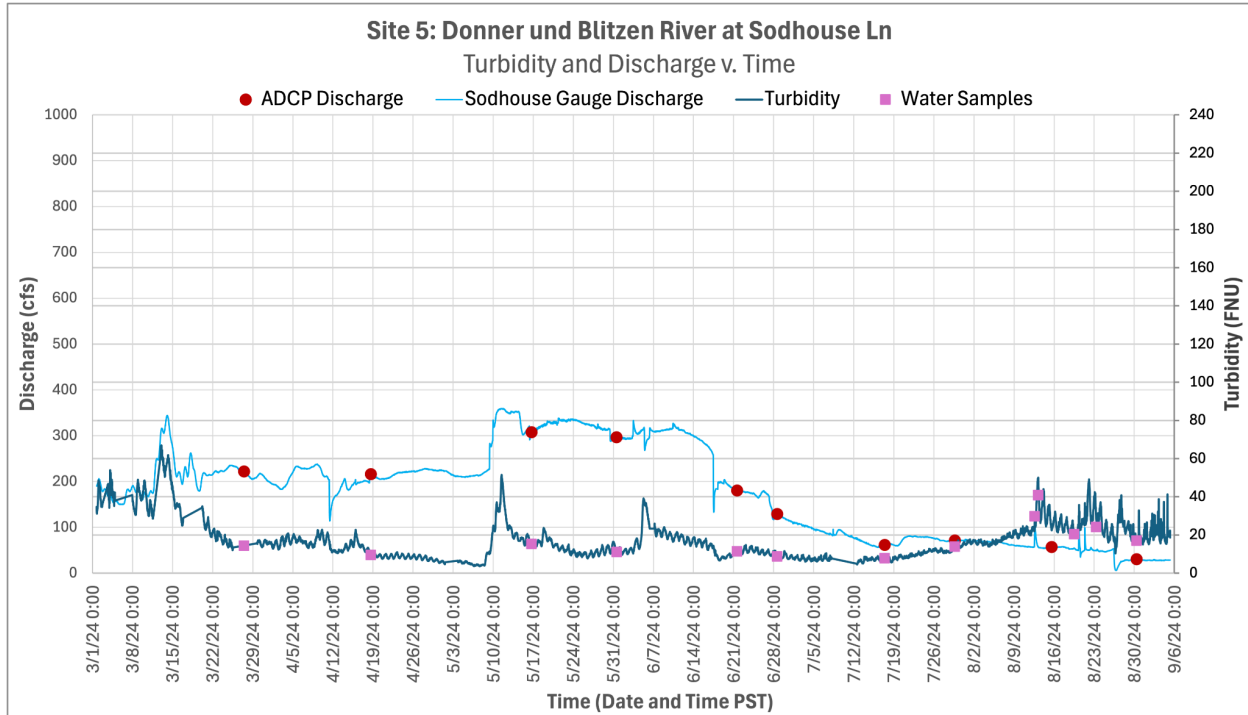


Figure 29: Site 5 turbidity and discharge versus time, with ADCP measurements and water samples.

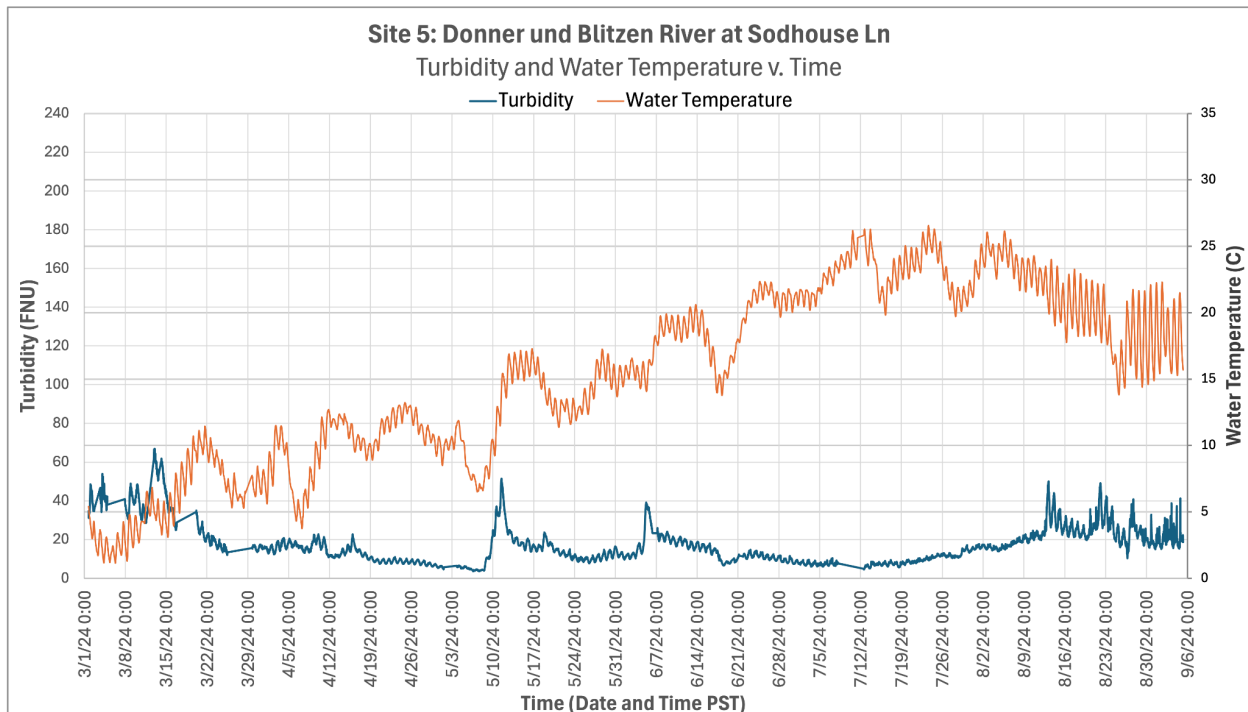


Figure 30: Site 5 turbidity and water temperature versus time.

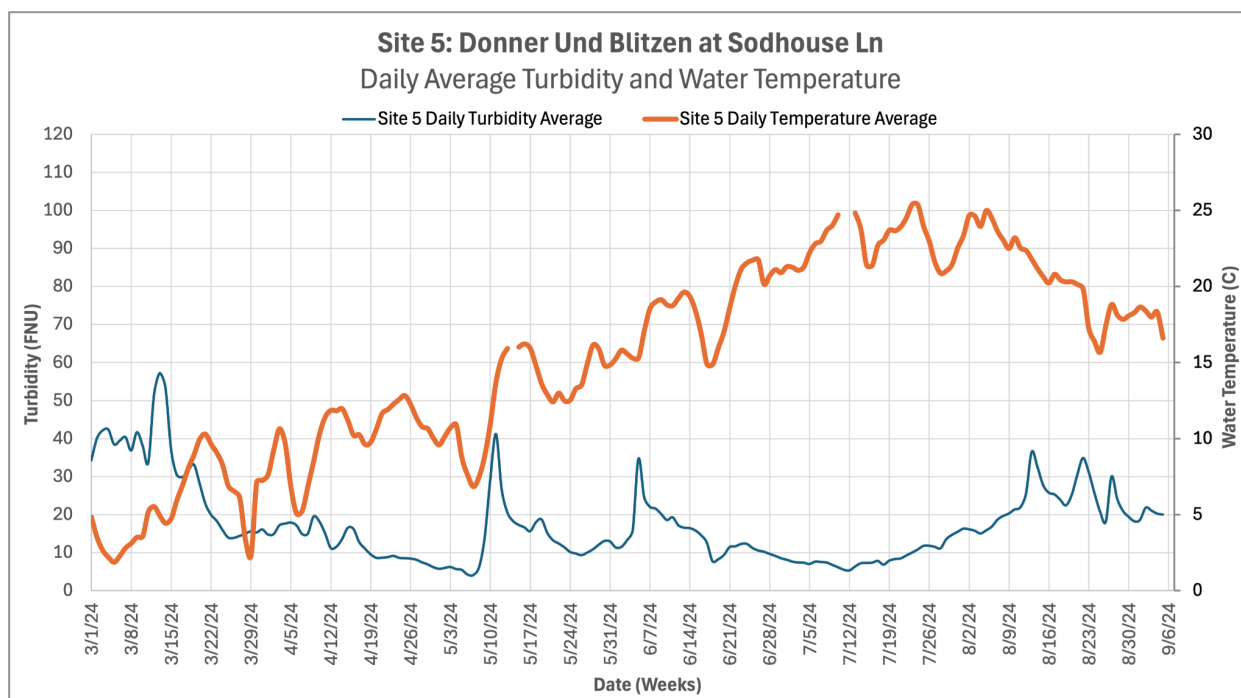


Figure 31: Site 5 daily average turbidity and water temperature versus time, with two gaps in water temperature data on days when sondes were taken to lab for calibration.

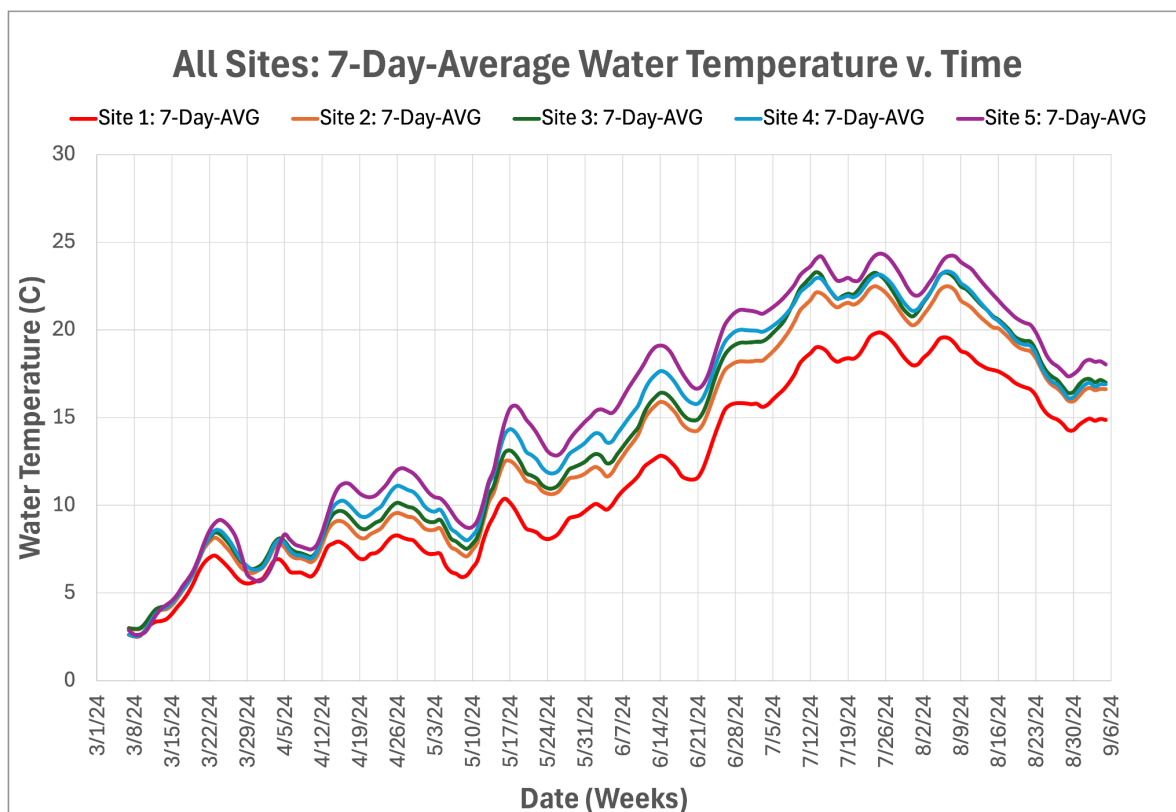
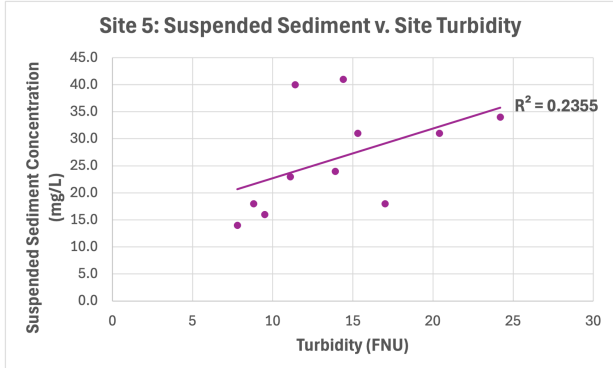
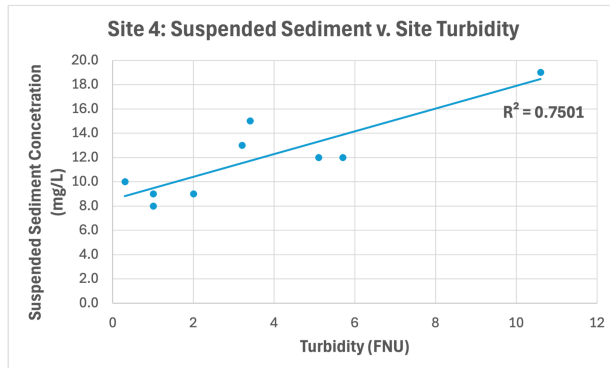
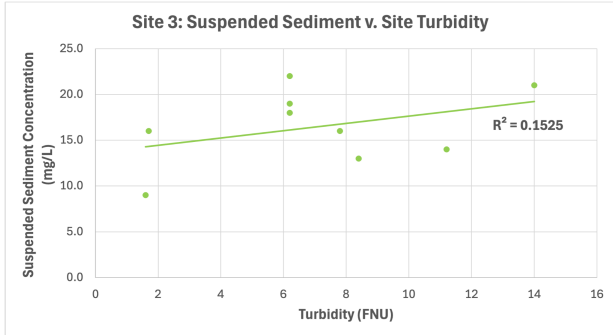
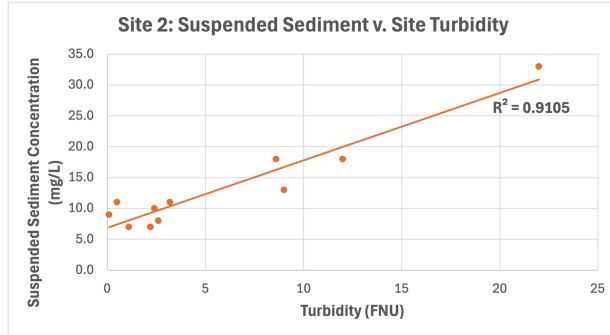
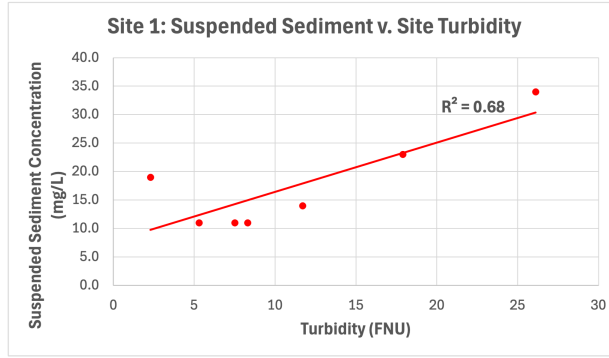


Figure 32: Seven-day-average temperatures for all sites as a function of time.

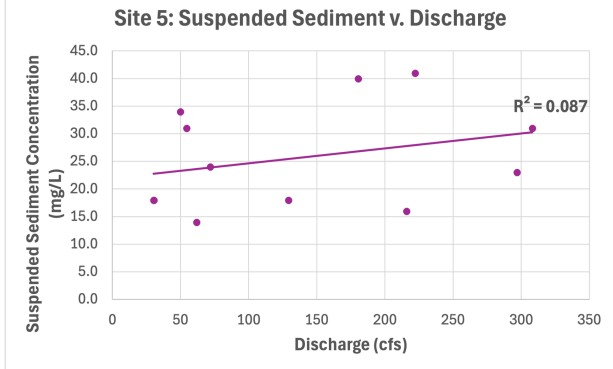
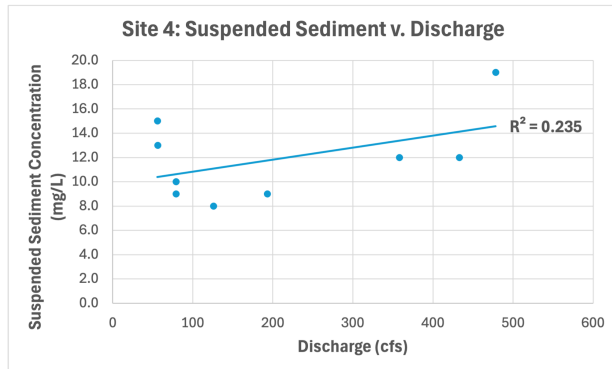
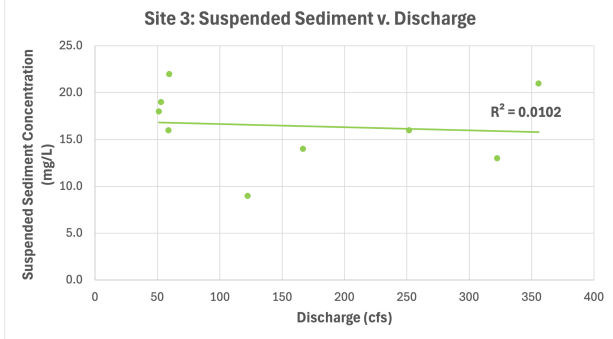
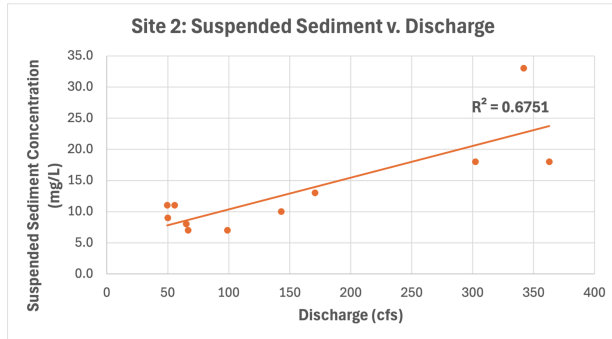
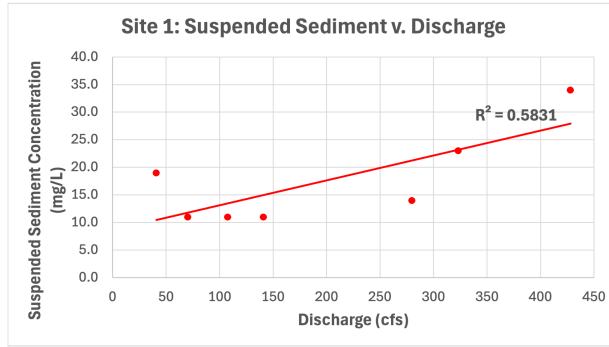
Water Samples

Site Number	Date_Time (PST)	SSC (mg/L)	LOI (mg/L)	Discharge (cfs)	Site Sonde TB (FNU)	Percent Finer	Used in Correlations (Y/N)
1	03/26/2024 1121	11.0	5	141.03	8.3	80.7	Y
1	04/18/2024 0951	14.0	4	280	11.7	87.1	Y
1	05/16/2024 1025	34.0	6	428.05	26.1	89.8	Y
1	05/31/2024 0824	23.0	3	323.09	17.9	91.1	Y
1	06/21/2024 0813	11.0	3	107.57	7.5	84.5	Y
1	06/28/2024 0734	11.0	5	70.47	5.3	71.6	Y
1	07/31/2024 0958	19.0	6.4	40.8	2.3	51	Y
2	03/26/2024 1344	13.0	4	170.73	9	87.1	Y
2	04/18/2024 1109	18.0	5	302.1	8.6	84	Y
2	05/16/2024 1248	33.0	8	341.67	22	82.4	Y
2	05/31/2024 1031	18.0	5	362.66	12	94.7	Y
2	06/21/2024 1017	10.0	5	142.98	2.4	86.5	Y
2	06/28/2024 0929	7.0	3	98.91	1.1	97.2	Y
2	07/18/2024 1200	7.0	2.6	66.6	2.2	86.1	Y
2	07/19/2024 0931	8.0	4.1	65.16	2.6	84.6	Y
2	07/29/2024 0716	11.0	5.9	55.5	3.2	85.8	Y
2	08/19/2024 1130	11.0	2.3	49.41	0.5	93.8	Y
2	08/29/2024 1224	9.0	2.0	49.88	0.1	81.4	Y
3	03/26/2024 1519	14.0	5	166.76	11.2	85.1	Y
3	04/18/2024 1253	16.0	5	251.61	7.8	86.4	Y
3	05/16/2024 1441	21.0	5	355.6	14	90.7	Y
3	05/31/2024 1158	13.0	4	322.48	8.4	95.5	Y
3	06/21/2024 1203	9.0	5	122.36	1.6	83.9	Y
3	06/28/2024 1116	63.0	58	73.56	0.7	10.4	N
3	07/19/2024 1254	22.0	7.1	59.44	6.2	76.4	Y
3	07/29/2024 1204	16.0	5.1	58.98	1.7	83.3	Y
3	08/19/2024 1157	18.0	3.2	51.32	6.2	84.8	Y
3	08/29/2024 1420	19.0	5.2	53	6.2	86.7	Y
4	03/27/2024 0942	107.0	95	243.57	9.7	14.1	N
4	04/18/2024 1406	12.0	4	358.07	5.1	85.5	Y
4	05/16/2024 1610	19.0	5	478.2	10.6	88.3	Y
4	05/31/2024 1308	12.0	3	432.73	5.7	94.3	Y
4	06/21/2024 1329	9.0	4	193.07	1	79.3	Y
4	06/28/2024 1320	8.0	4	125.63	1	82.1	Y
4	07/08/2024 1112	150.0	16.8	101.26	65.9	93.2	N - Event
4	07/16/2024 1313	114.0	21.4	81.36	1.1	58.3	N - Event
4	07/17/2024 1253	10.0	2.8	78.9	0.3	89.7	Y
4	07/29/2024 1411	9.0	2.6	78.82	2	92	Y
4	08/19/2024 1218	13.0	2.3	55.98	3.2	89.6	Y
4	08/30/2024 0749	15.0	4.1	55.84	3.4	89.4	Y
5	03/27/2024 1152	41.0	15	222.05	14.4	70.5	Y
5	04/18/2024 1555	16.0	5	215.86	9.5	88	Y
5	05/16/2024 1737	31.0	5	308	15.3	93.8	Y
5	05/31/2024 1444	23.0	7	296.84	11.1	87.2	Y
5	06/21/2024 1519	40.0	24	180.41	11.4	50.1	Y
5	06/28/2024 1526	18.0	5	129.18	8.8	94.3	Y
5	07/17/2024 1013	14.0	5.3	61.65	7.8	78.7	Y
5	07/29/2024 1649	24.0	3.4	71.76	13.9	92.3	Y
5	08/12/2024 1612	55.0	7.0	101	26	95.3	N - Event
5	08/13/2024 0654	64.0	7.2	56.8	44.5	95.6	N - Event
5	08/19/2024 1309	31.0	4.5	54.5	20.4	94.7	Y
5	08/23/2024 0951	34.0	4.2	49.8	24.2	93.9	Y
5	08/30/2024 1011	18.0	3.8	30.28	17	97.4	Y
5	09/06/2024 1106	18.0	2.8	30.4		92.3	Y

Table 6: Total water samples taken over the study period. The last column indicates whether each sample was included in the correlations. Underlined discharges are from the closest gauge, bold discharges are interpolated, and bold sonde turbidity values are interpolated.



Figures 33-37: Suspended sediment and site turbidity correlations with associated data points for each site.



Figures 38-42: Suspended sediment and discharge correlations with associated data points for each site.

Figures 43-52 (BELOW): All ADCP measurements for all sites, organized by dates. Note that the discharge axis changes from one graph to the next. The Frenchglen gauge (FG Gauge) upstream of the refuge is included to quantify diversions at Page Springs Dam. There are four missing site measurements: one at Site 1 on 4/18/24 that has discharge from the Page Springs gauge, one at Site 3 on 4/18/24 that was interpolated, one at Site 2 on 5/16/24 that is left blank, and one at Site 5 on 5/16/24 that has discharge from the Sodhouse Dam gauge.

