Little Shasta River 2017-2019:
Pre-Project Assessment of the Proposition 1 Ecosystem Restoration Grant Activities

Authors:

Amber Lukk, Priscilla Vasquez-Housley, Robert Lusardi, Ann Willis

Contents

Little Shasta River: Proposition 1 Pre-Project Assessment

Introduction

The Little Shasta River is a tributary in the Shasta River watershed in Siskiyou County, northern California, where substantial investments have been made to support the on-going recovery and conservation of anadromous fishes. These investments have primarily targeted the extent of and access to cold-water habitat for steelhead and coho salmon (*Oncorhynchus kisutch*). Preliminary work has shown that upper reaches of this tributary contain high-quality, cold-water habitat that could support juvenile coho salmon (Nichols et al. 2016, 2017). However, access to these upstream reaches is limited when low flows disconnect existing habitat from downstream reaches and the mainstem Shasta River.

To address this limitation, the California Department of Fish and Wildlife (CDFW) awarded a Proposition 1 Watershed Restoration Grant to California Trout, the Hart Ranch, and partners to improve channel function and dedicate a portion of streamflow at the Hart-Haight diversion to instream flows. The project includes two primary objectives:

1. replace the existing Hart-Haight diversion, comprised of a concrete weir, fish screen, and fish ladder walls, with a roughened channel to provide fish passage, and
2. dedicate a portion of diverted water to instream flow, creating a foundation for fuller connectivity flows to existing cold-water habitat.

The University of California, Davis Center for Watershed Sciences (CWS) has implemented a monitoring program to assess pre-project conditions of the Little Shasta River and its ecological functions to provide a baseline to which post-project conditions can be compared. This report documents the purpose, methods, data, and analysis of the pre-project assessment. First, a general description is provided of the monitoring program, including parameters, locations, and frequency of monitoring. Next, the methods used to quantify each element of the monitoring program is described. Then, the data gathered during this pre-assessment period is presented, organized by the three distinct sub-reaches identified in the Little Shasta River. Finally, implications of the findings relative to the project objectives and post-project monitoring are discussed. The findings of this assessment can be used to evaluate the effectiveness of local conservation activities in the Little Shasta River, as well as to understand how the Little Shasta River fits in the broader context of cold-water ecosystem function in the Shasta River watershed.

Study Area

The Little Shasta River extends approximately 41.7 km (25.9 mi) west from the Cascade Mountains of northern California until its confluence with the Shasta River within the lower Klamath Basin (see Figure 1, inset map). Monitoring efforts began in July 2017 and were organized by previously identified subreaches within the Little Shasta River: headwaters, foothills, and bottomlands (SVRCD, McBain & Trush 2013; Figure 1, Table 1). In the

headwaters reach, monitoring consisted of publicly accessible snow-water content data from the California Data Exchange Center (CDEC) to determine the hydrologic year-type for each monitored water year. Monitoring within the foothills reach focused on existing cold-water habitat and the planned footprint of the roughened channel, located at the downstream boundary of the foothills reach. Parameters included discharge, water temperature, and water quality, as well as opportunistic macroinvertebrate sampling, snorkel surveys, and carcass surveys. In the bottomlands reach, monitoring focused on quantifying periods of connectivity with the foothills reach. Streamflow, water temperature, water quality, and snorkel surveys were all conducted opportunistically given sufficient flow. The monitoring program was adapted each year as on-going data collection improved understanding of the watershed.
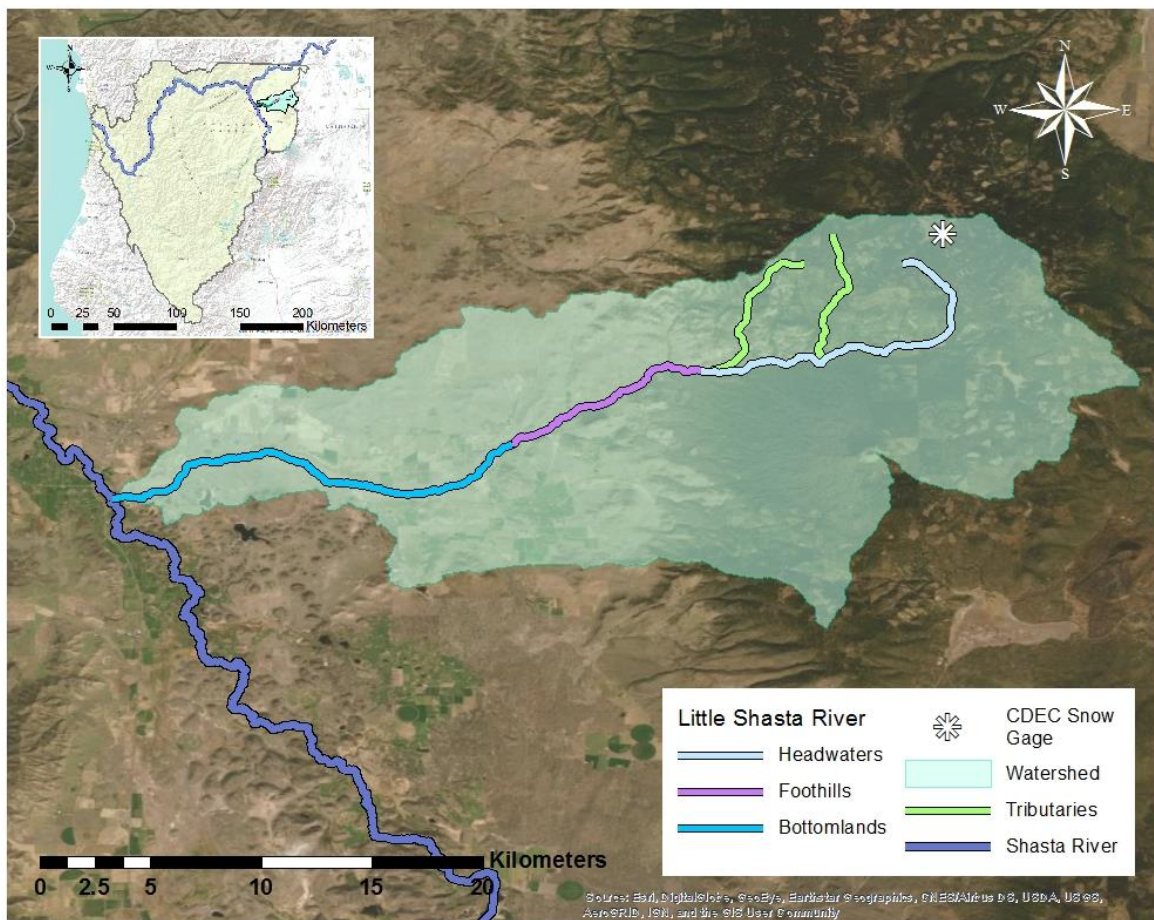

Figure 1. The boundary of the Little Shasta River watershed, including the full course of the river divided by study reach. Inset map shows the Lower Klamath watershed and the flow of water from the Little Shasta River to the Shasta River, which leads to the Klamath River, and finally to the Pacific Ocean.

Table 1. A summary of monitoring sites, coordinates, objectives, and parameters for the pre-project monitoring program. CEDEN site codes are those used to label data submitted to the California Environmental Data Exchange Network (CEDEN), if different from the monitoring site code.

Monitoring site code	CEDEN site code	Site name	Coordinates (UTM – Zone 10T)	River kilometer (rkm)	Sub-reach (objective)	Monitoring elements
LSH-CDEC		Little Shasta	566868.47 m E, 566868.47 m N	NA	Headwater (year type)	snow-water content
LSR	LSR	Little Shasta, Soda Springs boundary	554172.00 m E, 4620390.00 m N	20.9	Foothill (habitat)	discharge, water temperature, water quality
SR1		Snorkel Reach 2	552881.04 m E, 4619457.51 m N	19.1	Foothill (habitat)	snorkel surveys
LSM	MUS	Little Shasta, above Musgrave diversion	552716.00 m E, 4619490.00 m N	18.9	Foothill (habitat)	discharge, water temperature, water quality
CS1		Carcass Survey 1	552477.28 m E, 4619198.17 m N –553035.51 m E, 4619568.02 m N	18.5-19.3	Foothill (habitat)	carcass surveys
H1		Upper Hart	552526.15 m N, 4619314.24 m E	18.8	Foothill (habitat)	macro-invertebrate sampling
SR2		Snorkel Reach 1	552498.69 m E, 4619296.37 m N	18.6	Foothill (habitat)	snorkel surveys
LSH	HCR	Little Shasta, above Hart-Haight diversion	552480.00 m E, 4619190.00 m N	18.5	Foothill (habitat)	discharge, water temperature, water quality
CWL		Little Shasta, Cowley Ranch	550994.00 m E, 4617665.00 m N	16.2	Bottomlands (connectivity)	water quality
SR3		Snorkel Reach 3	550990.3 m E, 4617661.55 m N	16.2	Bottomlands (habitat)	snorkel surveys
C1		Cowley Ranch 1	550958.71 m N, 4617552.24 m E	16.1	Bottomlands (habitat)	macro-invertebrate sampling
CS2		Carcass Survey 2	550732.56 m E, 4617366.49 m N –551028.53 m E, 4617672.47 m N	15.75-16.2	Bottomlands (habitat)	carcass surveys
C2		Cowley Ranch 2	550690.16 m N, 4617379.6 m E	15.5	Bottomlands (habitat)	macro-invertebrate sampling

Monitoring site code	CEDEN site code	Site name	Coordinates (UTM – Zone 10T)	River kilometer (rkm)	Sub-reach (objective)	Monitoring elements
PMK		Little Shasta, Peacemaker Ranch	540784.00 m E, 4617018.00 m N	2.3	Bottomlands (connectivity)	water quality

Methods

Hydrologic Year Type

Snow-water content (SWE) assesses the water stored in snowpack. This calculated measurement uses both the depth and density of snow to estimate the amount of water contained within a given level of snowpack (Sturm et al 2010). The observed snowpack for a given year can then be compared to an average for the area to identify the percentage of the average that the snowpack has reached, and subsequently used to identify whether hydrologic conditions indicate a dry, normal, or wet year.

SWE is monitored at the Little Shasta station (LSH-CDEC) by the U.S. Forest Service Goosenest Ranger District. This meadow station is located at an elevation of 1,890 m (6,200 ft) where the snowmelt runoff to the Little Shasta River begins its descent from the western slopes of the Cascade Range toward the Shasta Valley (see Figure 1). This station has been actively monitored since 1946, with recordings of snow-water content taken annually in April of each year. SWE data were used to determine hydrologic year types based on wet-normal-dry percentiles established in Nichols et al. (2016).

Discharge

Periodic discharge measurements were performed across a range of observed streamflows at each monitoring site (see Figure 2Figure 2) following standard measurement and computational methods (Rantz 1982a, b). Depths and velocities were measured using a top-set wading rod and Hach FH950 flow meter, respectively, at established cross-sections. Hach FH950 flow meters measure velocity over the range of 0.00 to 6.09 m/s (0 to 20 ft/s) in a minimum water depth of 0.0313 m (1.25 in). Accuracy ranges from 2% for velocities up to 3.04 m/s (10 ft/s), and 4% for velocities between 3.04 and 4.87 m/s (10 to 16 ft/s). Where stage was not provided by automated, remote monitoring stations, it was recorded using Solinst Levelogger Edge M10 pressure transducers in 15-minute intervals, and compensated for barometric effects using data recorded by a Solinst Barologger. M10 Leveloggers have an accuracy of 0.5 cm (0.016 ft), and function over the range -20°C to 80°C; Barologgers have an accuracy of 0.05 kPa.

Figure 2. The monitoring sites on the Little Shasta River where discharge, water temperature, and/or water quality samples are collected.

River stage-discharge relationships were quantified using standard rating methodologies (Rantz 1982a), from which continuous streamflow time-series were calculated. Measured river stages greater than those observed during periodic discharge measurements (and corresponding discharges) were excluded from the calculated streamflow time series. Discharge was calculated from the manual measurements of depth, velocity, and wetted cross-sectional distances that were taken at various times during the study.

Water Temperature

Water temperature measurements were concurrent with stage monitoring. Where water temperature data was not provided by automated, remotely accessible monitoring stations, it was measured using Solinst Levelogger Edge M10 data loggers. Solinst Levelogger Edge M10 temperature sensors have an accuracy of 0.05°C and operate over the range -20°C to 80°C.

Water Quality

Water quality was monitored using monthly grab samples that were analyzed for their physiochemical content. Samples were collected in 125 mL bottles, which were previously triple-rinsed with environmental water. Spot measurements of water temperature were concurrently taken using an Oakton Temp 5 Acorn Series thermometer (±0.2°C, -40°C to 125°C). A summary of the water quality parameters included in the sample analysis are presented in Table 2.

Table 2. A summary of parameters included in the water quality analysis.

Parameter	Description (unit)
Tw	Water temperature (°C)
EC	electroconductivity (µS/cm)
pH	Acidity/alkalinity
Turbidity	(ntu)
DOC	dissolved organic carbon (mg/L)
NH_4	Ammonium (mg/L)
NO_3	Nitrate (mg/L)
TN	Total Nitrogen (mg/L)
PO_4	Orthophosphate (mg/L)
TP	Total Phosphorus (mg/L)

Aquatic Macroinvertebrates

Stream macroinvertebrates were sampled from the Little Shasta River at three locations during June 2019 (Figure 3, Appendix C). Two of the locations were located downstream of the current point of diversion at the Cowley Ranch property. A third site was located upstream on the Hart property immediately above the current point of diversion. Sampling followed standard operating procedures for the collection of benthic macroinvertebrate samples using the reach-wide, multi-habitat procedure, following the Surface Water Ambient Monitoring Program Bioassessment Procedures for wadable streams (SWAMP 2007). A 150-meter reach of stream was delineated at each sampling location using a reel tape. After reach delineation, macroinvertebrates were systematically every 15 meters, alternating left, right, and center of the wetted channel. The first transect sample was always located at the most downstream position at each sample site. Stream macroinvertebrates were collected using a 500 µ mesh D-frame net by disturbing 0.09 m^2 of substratum to a depth of 6 cm and capturing entrained invertebrates. Transect samples were composited into one sample to provide a quantitative measure of stream macroinvertebrate density and diversity at the reach scale. All samples were preserved in 95% ethanol and returned to the laboratory for processing. In the laboratory, a folsom plankton splitter was used to subsample each sample to reach a minimum count of 550 organisms. Stream macroinvertebrates were identified to the lowest practical order (usually genus or species) using Merrit et al. (2008), Thorp and Covich (2001), as well as various taxonomic-specific references. Oligochaetes were identified to class; non-biting midges (Chironomidae) and fingernail clams (Sphaeriidae) were identified to family.

For baseline comparison and to gauge project effectiveness, seven common macroinvertebrate metrics were selected that include various measures of taxonomic richness and organism tolerance values (see results). Tolerance values are a measure of an organism's ability to survive and reproduce in the presence of known levels of stressors. Tolerance values range from zero (highly intolerant) to 10 (highly tolerant). A description of the specific metrics examined in this
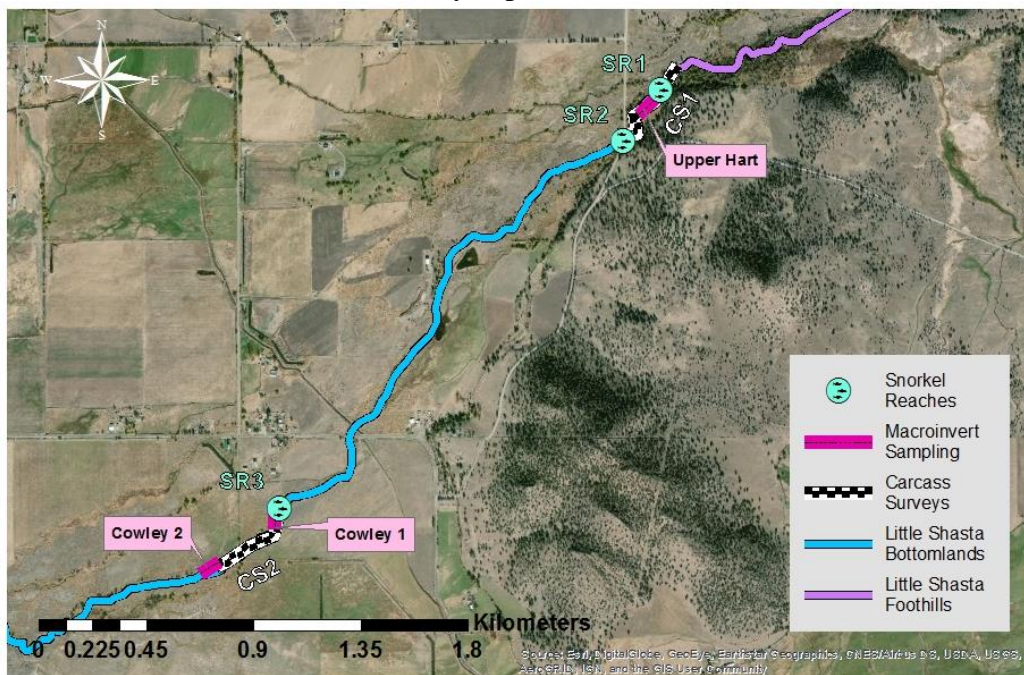
study is provide in

Figure 3. Little Shasta River snorkel reach, carcass survey, and macroinvertebrate sampling locations.
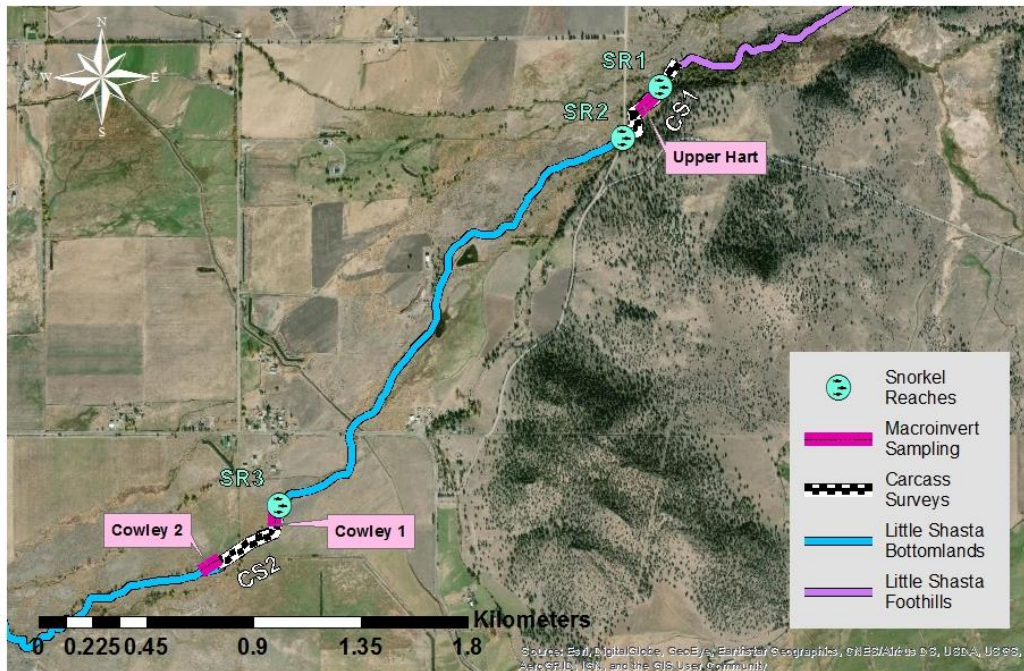
Table 3.

Figure 3. Little Shasta River snorkel reach, carcass survey, and macroinvertebrate sampling locations.

Table 3. Benthic macroinvertebrate metrics and their expected response to ecological perturbation.

Macroinvertebrate Metric	Metric Description	Expected Response to Disturbance
Total Density	Total number of macroinvertebrates per meter squared	Variable
Taxonomic Richness	Total number or richness of taxa found in sample	Decrease
EPT Index	Relative abundance of macrobenthos in the orders Ephemeroptera, Plecoptera, and Trichoptera	Decrease
Shannon-Wiener Index	Measure of macroinvertebrate diversity *and* evenness within a sample	Decrease
Tolerant Species	Relative abundance of species with tolerance values of 8, 9, or 10 (scale of 10; least to most tolerant)	Increase
Sensitive Species	Relative abundance of species with tolerance values of 0, 1, or 2 (scale of 10; least to most tolerant)	Decrease
Hilsenhoff's Biotic Index	Measure of community tolerance to organic pollution or degraded habitat (based on tolerance values and relative abundance)	Increase

Fish Presence/Absence

Adult and juvenile salmon presence/absence monitoring was conducted during the period using non-intrusive methods at delineated reaches above and below the project site (Figure 3). Adult spawning and carcass surveys were conducted monthly (or bi-monthly, depending on weather and hydrologic conditions) between October 2017 and January 2018 to coincide with adult salmon returns. Walking each study reach, the presence and location of adult anadromous salmonids (live or carcasses) or redds was recorded using a GPS and mapped following the methods of Gallagher (2001).

Snorkel surveys were conducted monthly or bimonthly (depending on hydrologic conditions) during spring of 2018 and 2019 between the months of April and September. Initially, we delineated three snorkel reaches: one reach located on the Cowley Ranch property and two reaches on the upper Hart property (Figure 3; Appendix C, Figure C- 1). However, due to a lack of flow at the Cowley Ranch during spring, we were precluded from conducting snorkel surveys at this location. Each snorkel site was delineated based on similarities in geomorphic class and included at least two riffle pool sequences per monitoring reach with each reach measuring between 50 m and 60 m. Snorkel surveys were conducted following the procedures of Apperson et al. (2015). During each survey, a single snorkeler moved upstream through entire reach and enumerated fish by species and age class on a wrist slate, then transferred the information to a data sheet.

Results

Hydrologic Year Type

The average snowpack recorded at this site (last 50-year span) was 16.6 inches. The 2017 and 2018 water years were classified as "normal" based on snow-water content exceedance evaluations, while the 2019 and 2016 water years were classified as "wet" (Figure 4). The 2017 water year had an April 1 average snowpack depth of 15.5 inches, placing it at 93% of the average for this site. The following 2018 water year had an April 1 average snowpack depth of 13.0 inches (78% of the average). In contrast, the 2016 water year had 23.5 inches of snowpack (142% of the average), and the 2019 water year had an April 1 average snowpack of 22.0 inches (133% of the average).

These "normal" and "wet" water years were preceded by a "dry" 2015. This water year was the last in a 5-year drought observed throughout the state, and marked the first recorded zero-snow condition on April 1 (0% of the average).
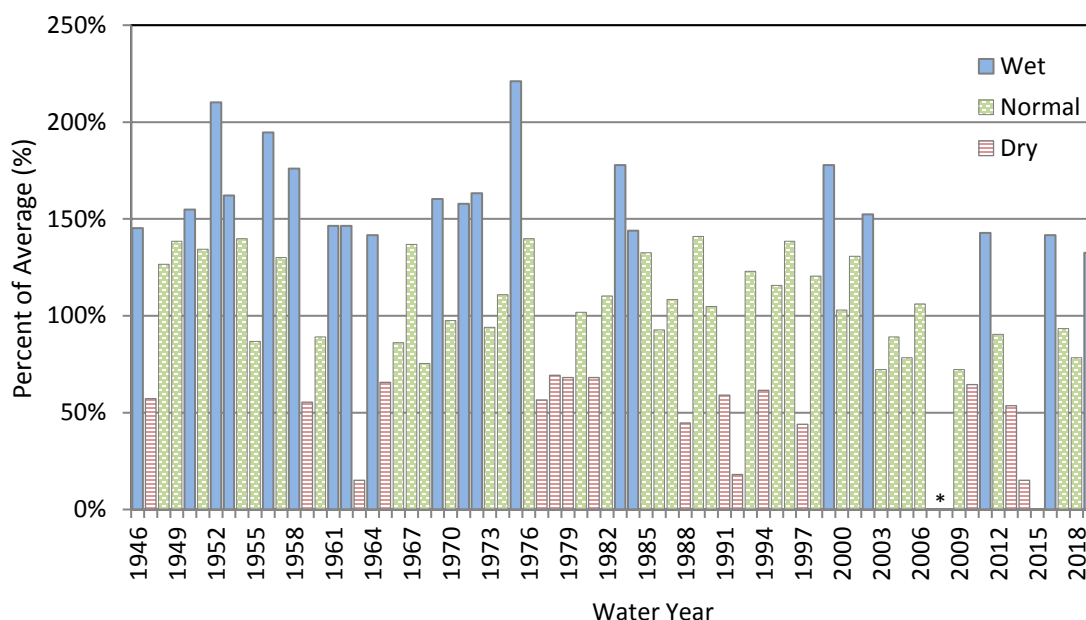
Figure 4. Snow-water exceedance (SWE) evaluations for the Little Shasta River from the 1946 water year to most recent (2019), given in percent of the average (16.6 in). Data obtained from the CDEC station "LSH". (*Note: no data was recorded for the 2007 and 2008 water years.)

Streamflow

Flow monitoring occurred at three sites to track hydrologic conditions due to natural variability and local diversions: LSR, LSM, and LSH (Figure 5). Previous studies showed that surface water diversions led to the disconnection of sites downstream of LSH from the upper reach (Nichols et al. 2016). Attempts were made to monitor seasonal flows at CWL; however, high-flow winter events destroyed two stilling wells; game cameras that had been mounted to visually monitor

flow conditions were knocked down. Thus, discharge monitoring was limited to the foothills reach.

The rating curves developed for each site extended from summer baseflows to winter high-flow events. At LSR, measured streamflow ranged from 3.9 to 71.6 ft^3/s; at LSM, from 3.2 to 56.1 ft^3/s; and at LSH, from 0.8 to 51.1 ft^3/s (Table 4). Rating curves were developed at each site and exhibited R^2 values greater than 0.9, indicating a good fit between stage and discharge data (Figure 6, Figure 7, and Figure 8). Multiple measurements were taken near the upper and lower bounds of each rating curve. At LSH, discharge measurements above 51.1 ft^3/s showed poor fit with the curve (Figure 8); however, a closer examination of both stage and discharge data showed no indication of erroneous sampling. Thus, while the data was preserved to show the full dataset, the rating curve was only applied up to 51.1 ft^3/s.
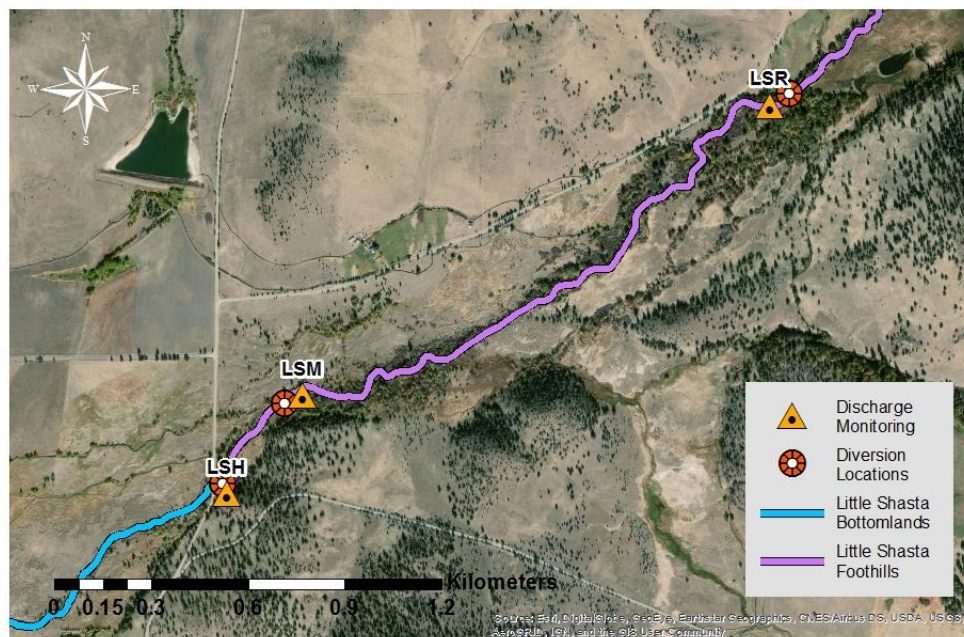
Figure 5. The three reaches of the foothills section of the Little Shasta River where discharge measurements are collected (LSR, LSM, LSH) and the locations of the irrigation diversions.

Table 4. A summary of stage vs. discharge relationships for the three sites in discharge monitoring section of the foothills reach of the Little Shasta.

Site	River km	Rating Equation	R^2 value	Min Flow (ft^3/s)	Max Flow (ft^3/s)
LSR	20.9	$y = 472.25x^{3.6528}$	0.9752	3.9	71.6
LSM	18.9	$y = 1016.4x^{3.3888}$	0.9575	3.2	56.1
LSH	18.5	$y = 0.559x^{4.9211}$	0.9159	0.8	51.1
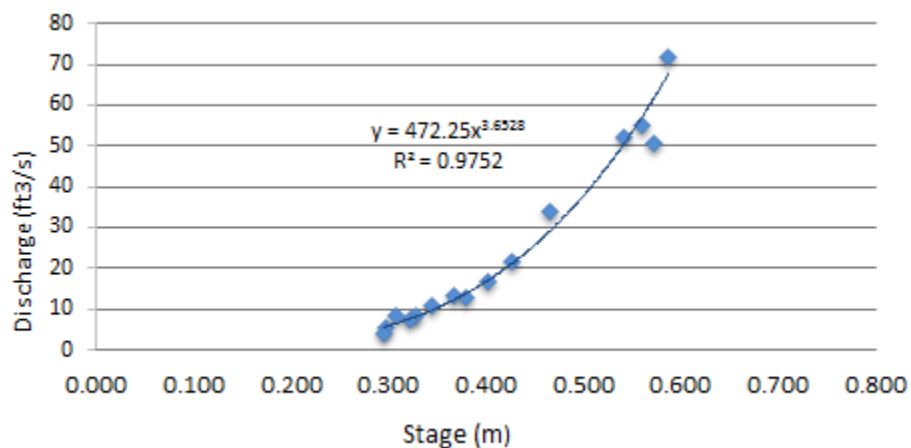
Figure 6. The rating curve developed to calculate discharge at LSR. Stage and discharge data are available online via the California Data Exchange Center (CDEC).

Figure 7. The rating curved developed to calculate discharge at LSM.

Figure 8. The rating curve developed to calculate discharge at LSH. Given the poor fit of discharge points greater than 51.1 ft³/s, the rating curve was only used to calculate discharge up to this threshold.

Streamflow at LSR showed relatively stable baseflows from June through September, followed by event-driven peaks in the winter and early spring (Figure 9). Flashy rainfall runoff events extended from late November and continued while snowmelt began in March and April. Av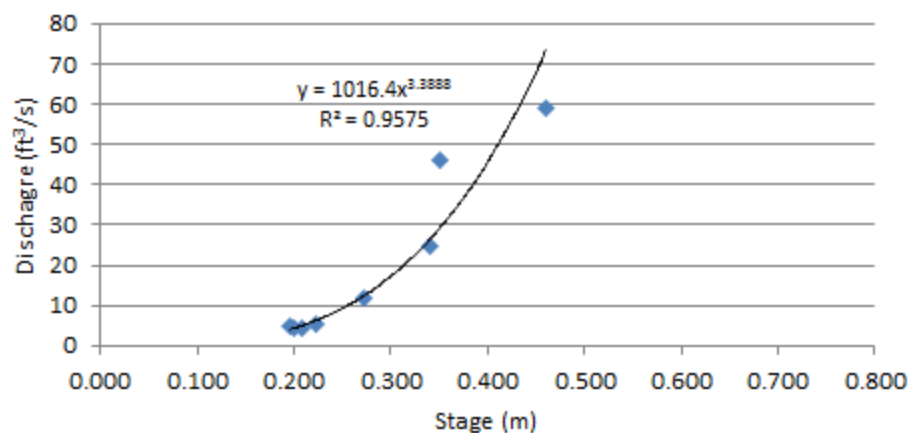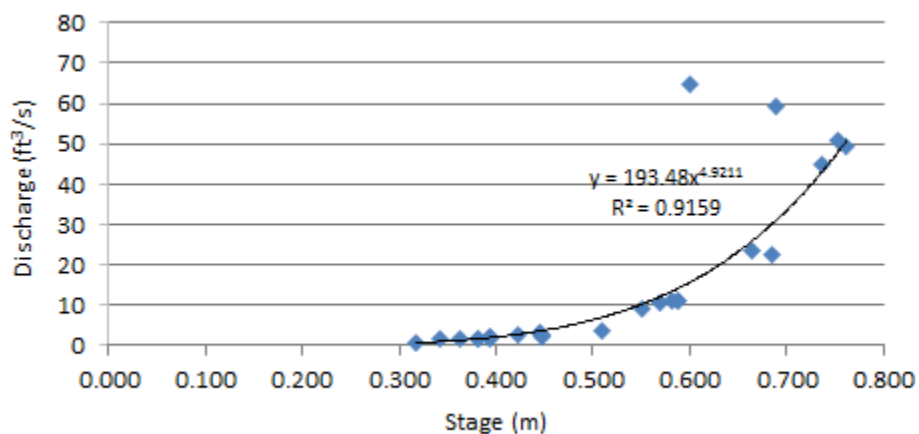erage streamflow ranged between 6.5-12.0 ft^3/s during the irrigation season (i.e., March 1-Octber 31) and 13.7-14.7 ft^3/s during the non-irrigation season (November 1-February 28), while peak event flows ranged between 60-70 ft^3/s and, in 2018, exceeded the maximum-rated flow of 71.6 ft^3/s (Table 5).

Figure 9. Discharge (measured and calculated) and temperature data for LSR throughout the 2017-2019 water year monitoring period. The dashed line shows the highest rated flow, 71.6 ft^3/s.

Table 5. The mean daily averages for the LSR site, presented for no irrigation (November 1-February 28) and irrigation seasons (March 1-October 31).

Water year	Mean (ft^3/s)		Annual max (ft^3/s)	Annual min (ft^3/s)
	Nov 1 - Feb 28	Mar 1 - Oct 31[†]		
2017*	NA	6.5	28.2	6.0
2018	13.6	11.6	71.6	5.9
2019*	14.5	29.9	67.4	9.1

Incomplete water year records for 2017 and 2019 irrigation seasons: July 24, 2017 through Sept. 30, 2017 and March 1, 2018 through June 30, 2019.
[†]*Irrigation season begins in March and extends through October of the following water year.*

Streamflow at LSM showed similar patterns to those observed at LSR, though at lower magnitudes. While no diversions are located between LSR and LSM, the channel is obstructed by debris just downstream of snorkel reach 2 (see Figure 3), which forces water out of the main channel. Therefore, data presented in this report describes the water that remains in the main channel, but does not quantify the multi-channel flow that occurs at this site.

Baseflows remained relatively stable from June through September, with event-driven peaks in the winter and early spring (Figure 10). Rainfall runoff events began in November and continued

into the beginning of snowmelt runoff in the March and April. Average streamflow ranged between 7.1-13.5 ft^3/s during the irrigation season (i.e., March 1-October 31) and 4.8-14.9 ft^3/s during the non-irrigation season (November 1-February 28), while peak event flows ranged between 48.3-96.7 ft^3/s, exceeding the maximum rated flow of 56.1 ft^3/s in 2018 (Table 6).

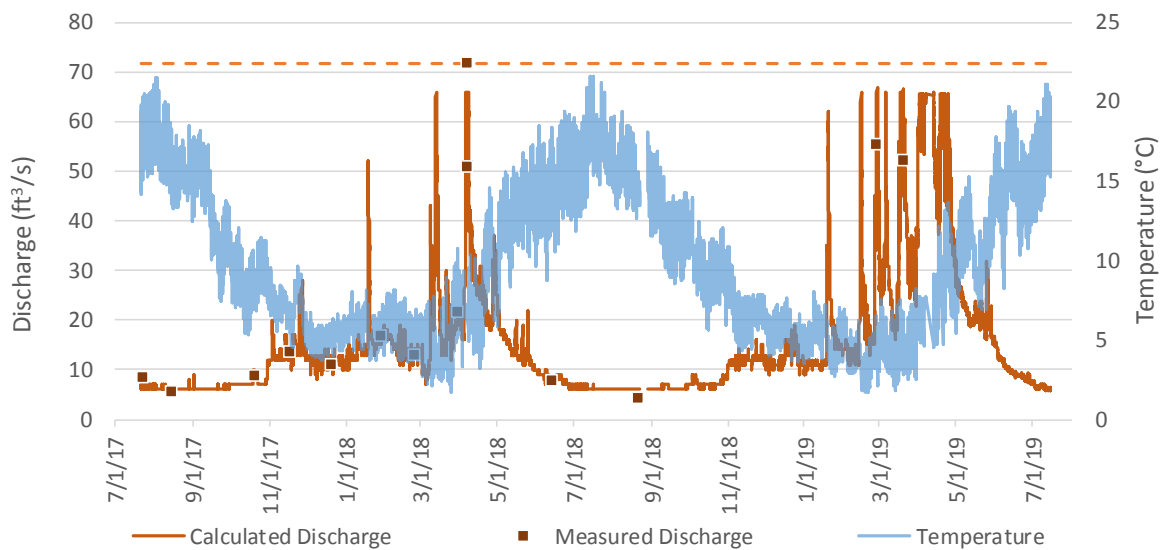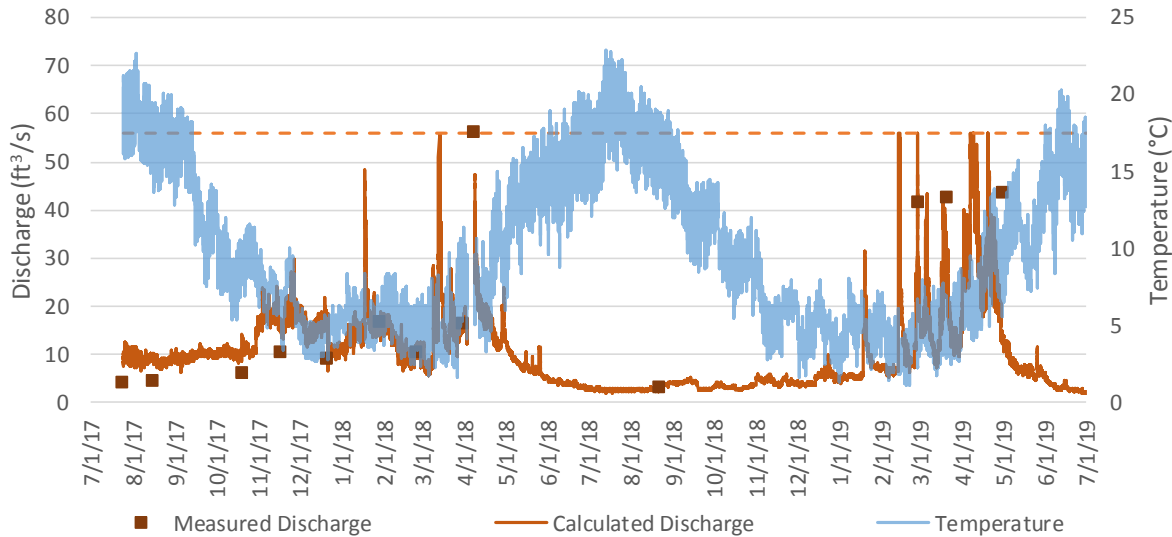*Figure 10. Discharge (measured and calculated) and temperature data for the LSM site for the 2017-2019 monitoring period. The dashed line shows the highest rated flow, 56.1 ft^3/s.*

Table 6. The mean daily averages for the LSM site, presented for no irrigation (November 1-February 28) and irrigation seasons (March1-October 31).

Water year	Mean (ft^3/s)		Annual max (ft^3/s)	Annual min (ft^3/s)
	Nov 1-Feb 28	March 1-Oct 31[†]		
*2017	NA	9.8	48.3	6.3
2018	14.9	7.1	96.7	2.0
*2019	4.8	13.5	56.1	1.2

Incomplete water year records for 2017 and 2019 irrigation seasons: July 24, 2017 through Sept. 30, 2017 and March 1, 2018 through June 30, 2019.
[†]*The irrigation season begins in March and extends through October of the following water year.*

An analysis of streamflow and water temperature at LSH was complicated by diversion operations at this site. The rating curve was developed during periods when the diversion was inactive; however, a backwater pool created during periods when the diversion was active prevented an accurate calculation of discharge from the time series stage data collected at this site. Thus, only measured streamflow is included in this report.

The manually measured flow data is shown in Figure 11. Average measured streamflow ranged between 3.8-26.8 ft^3/s during the irrigation season (i.e., March 1-October 31) and 13.1-51.8 ft^3/s during the non-irrigation season (November 1-February 28), while peak measurements ranged between 7.2-51.8 ft^3/s (

Figure 11. Measured discharge and temperature data for the LSH site for the 2017-2019 monitoring period. Water temperature data was available through Oct. 25, 2018.
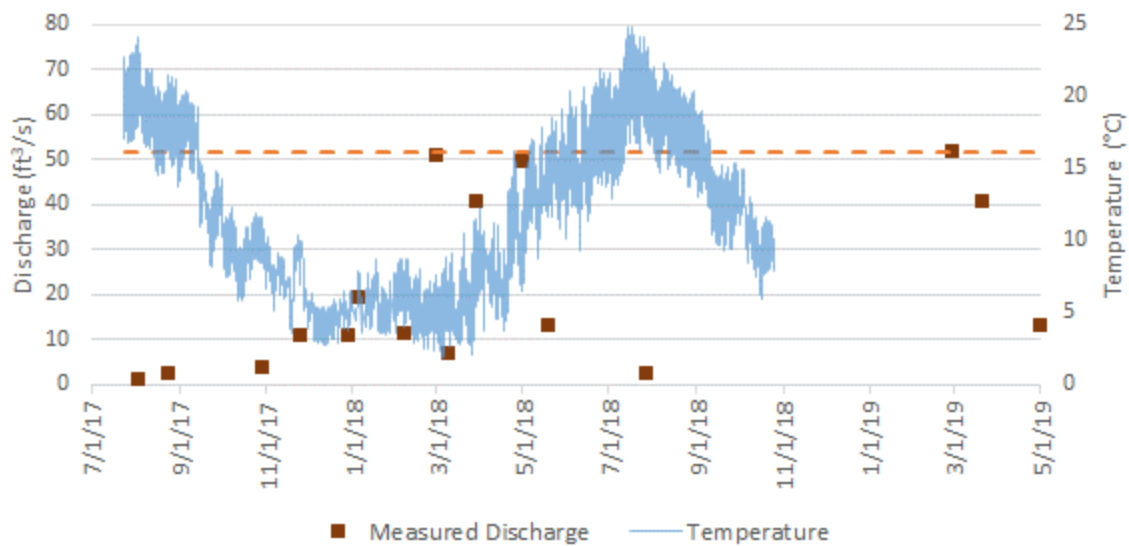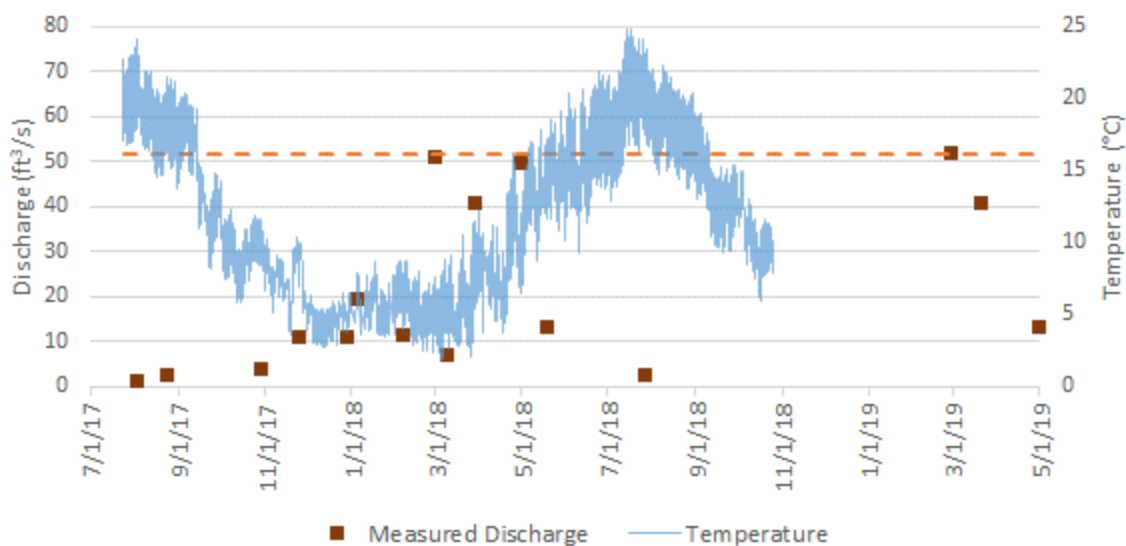
Table 7).

Figure 11. Measured discharge and temperature data for the LSH site for the 2017-2019 monitoring period. Water temperature data was available through Oct. 25, 2018.

Table 7. The mean daily averages for the LSH site, presented for no irrigation (November 1-February 28) and irrigation seasons (March 1-October 31).

Water year	Mean (ft^3/s)		Annual max (ft^3/s)	Annual min (ft^3/s)
	Nov 1-Feb 28	Mar 1-Oct 31[†]		
*2017	NA	3.8	7.2	2.6
2018	13.1	22.3	51.1	2.7
*2019	51.8	26.8	51.8	13.1

Incomplete water year records for 2017 and 2019 irrigation seasons: July 24, 2017 through Sept 30, 2017 and Oct 1, 2018 through Oct 25, 2018. Stage and water temperature data were available through Oct 25, 2018.
[†]*Diversion season begins in November and extends through October of the following water year.*

Water Temperature

Time series plots of water temperature trends at the three foothills sites followed similar seasonal patterns (Figure 9, Figure 10, Figure 11Figure 11). Monthly maximum, minimum, and average water temperatures were summarized with streamflow conditions (Table 8). At all three sites, water temperatures peaked during July and were coldest in March. Cool March temperatures are likely due to the onset of snowmelt, which is indicated by the increasing streamflow as compared to February. Minimum water temperatures were the same at LSR, LSM, and LSH (1.7°C, 1.7°C, and 1.8°C, respectively). Maximum temperatures showed that warming occurred in the 2.4 km from LSR to LSH, and peaked at 21.6°C (LSR), 22.9°C (LSM), and 24.9°C (LSH). As well as heating that would have occurred naturally as water flowed downstream during summer, lower streamflows at each site may have contributed to warming. Maximum streamflows at LSR were 7.2 ft3/s in July, but decreased to 3.2 ft^3/s at LSM, then increased to 3.9 ft^3/s at LSH. As no diversions occur between LSR and LSM, lower streamflows at LSM could be due to losses into the stream bed or portions of flow diverted out of the main channel due to the upstream debris jam. The slightly higher flows measured at LSH indicate that a portion of those surface flows may return to the channel as the stream flows more closely to the hill bounding the south bank, forcing water to return to the main channel.

Table 8. A summary of the monthly maximum, minimum, and average values recorded for streamflow and temperature in the three discharge monitoring locations. Since the 2017 and 2019 water year records are incomplete, only data from the 2018 water year is included.

	Streamflow (ft³/s)			Water Temperature (°C)		
LSR	**Max**	**Min**	**Avg**	**Max**	**Min**	**Avg**
Feb	19.1	10.0	14.1	8.2	3.6	5.3
Mar	66.0	7.3	18.6	10.8	1.7	4.9
Apr	66.2	16.0	26.4	14.6	3.2	7.9
May	25.0	10.1	14.6	17.3	5.8	12.6
Jun	10.3	6.3	8.1	19.3	8.7	14.5
Jul	7.2	6.1	6.2	21.6	11.8	16.6
Aug	6.0	6.0	6.0	19.4	12.4	15.3
Sep	7.4	5.9	6.2	16.9	8.5	12.2
LSM	**Max**	**Min**	**Avg**	**Max**	**Min**	**Avg**
Feb	18.1	6.7	11.5	8.4	2.4	5.1
Mar	55.8	5.6	14.5	11.4	1.7	5.1
Apr	47.5	10.8	17.4	15.0	3.3	8.1
May	15.3	5.0	7.9	17.8	6.0	12.9
Jun	5.4	2.6	3.9	20.2	8.8	15.1
Jul	3.2	2.0	2.7	22.9	12.1	18.0
Aug	4.4	2.2	3.0	20.5	12.6	16.7
Sep	5.1	2.6	3.7	17.7	8.5	12.7
LSH	**Max**	**Min**	**Avg**	**Max**	**Min**	**Avg**
Feb	24.2	10.4	16.5	8.7	2.3	5.3
Mar	>51.1	5.5	27.3	12.3	1.8	5.6
Apr	>51.1	6.2	28.7	16.2	3.7	8.6
May	34.4	6.6	11.4	19.1	6.4	13.6
Jun	8.5	2.6	4.7	21.9	9.2	16.0
Jul	3.9	2.0	2.7	24.9	12.9	19.3
Aug	4.1	2.3	2.9	22.3	13.6	17.8
Sep	5.7	2.7	3.8	19.0	9.2	13.6

The number of days when water temperatures exceeded 20°C was quantified to examine whether water temperatures met U.S. EPA (2003) guidance criteria for oversummering coho salmon. Both the total number of days over 20°C and the longest consecutive number of days over 20°C were identified. Data was included from this study, as well as water temperature data from LSM that was previously gathered and analyzed in Nichols et al. (2016).

Water temperature exceeded 20°C at all sites for all years that data was available (Table 9). Data from 2016, 2017, and 2018 showed that the number of over 20°C days increased from upstream to downstream. LSH has the most days exceeding 20°C (67 days in 2018), as well as the longest period of consecutive days (45 days in 2018). Interestingly, LSM showed more days over 20°C during 2016, which was a wet year, than in 2015, which was the driest year on record: 33 days in 2015 and 56 days in 2016. At the same site, the fewest number of over 20°C days were observed in 2017 (17 days) and 2018 (29 days), which were both normal years. At LSR, the most upstream site, the number of days over 20°C decreased from 2016 to 2018, with only 7 days observed in 2018.

Table 9. An overview of the total days per year that daily maximum water temperatures at the discharge monitoring sites on the Little Shasta surpass 20°C as well as the longest run of consecutive days in which temperatures surpass 20°C. No data was recorded for LSR during the 2015 water year, or at LSH during the 2015 and 2016 water years.

	# days > 20°C (most consecutive days)		
Water Year	**LSR**	**LSM**	**LSH**
2015	--	33 (13)	--
2016	21 (9)	56 (27)	--
2017	11 (5)	17 (8)	40 (26)
2018	7 (3)	29 (17)	67 (45)

Water Quality

Water quality varies in response to an area's geology, hydrology, land use, and aquatic system processes. This analysis focuses on nutrients (nitrogen and phosphorus) because of their biological importance in aquatic systems and the potential role of these constituents in restoration actions. Tabulated data for all analyzed parameters are included in appendix B.

Sample Timing

Samples were collected concurrently with discharge measurements, which were designed to capture a range of flows. As such, the water quality data provided insight to both background constituent concentrations during baseflows, as well as constituent loading during high-flow events. A plot of the water quality sampling events and discharge at LSR shows that samples taken during 2017 and 2018 generally occurred during seasonally stable flows, whereas samples collected during 2019 were coincident with high-flow events and the spring recession (Figure 12). Sampling at CWL and PMK were added in 2018; however, due to seasonal disconnection from the foothills reach, spring and summer samples at these sites were unrelated to flow conditions at the upstream sites.
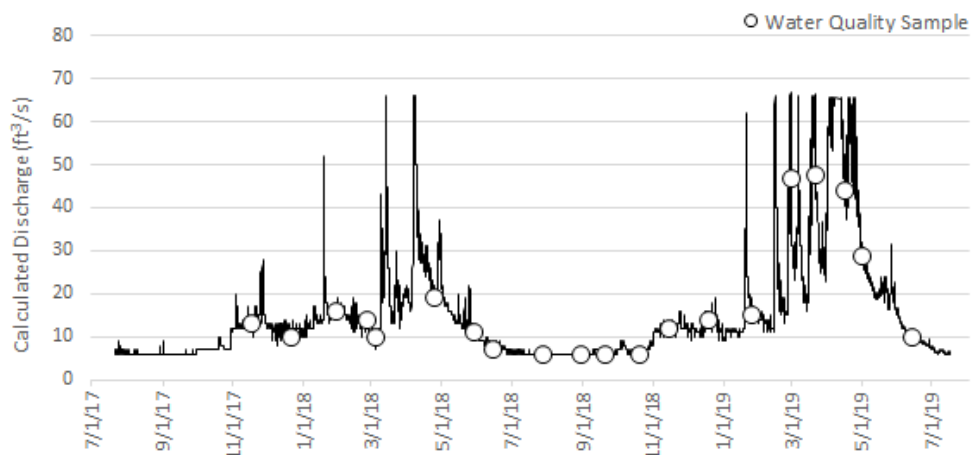
Figure 12. An overview of the timing of each water quality sample and the corresponding streamflow levels.

Nitrogen

Nitrogen and phosphorous are key components of primary productivity; one or the other are often limiting in natural aquatic ecosystems. When nitrogen and phosphorous are available in sufficient quantities, primary production in aquatic systems can be appreciable. Nitrogen is an essential nutrient for plant growth, yet is often described as a pollutant (e.g., from fertilizers and animal wastes) in many freshwater systems. In rivers with elevated nutrient levels, abundant primary productivity often results in a high biological oxygen demand, which can lead to undesirable dissolved oxygen concentrations. Inorganic nitrogen is available for uptake by aquatic plans and consists of ammonium, nitrite, and nitrate. When oxygen is present, nitrite is largely absent; thus, total inorganic nitrogen is calculated as ammonium plus nitrate. Organic nitrogen is produced from the breakdown of organic materials (i.e., plants and animals). An analysis of both total nitrogen (organic and inorganic) and total inorganic nitrogen are presented.

Total nitrogen (TN) and total inorganic nitrogen (TIN) varied between sites and seasonally. LSR, LSM, and LSH all showed similar levels of TIN, which decreased seasonally from fall to spring (Figure 13). In addition, TIN at these sites was generally higher during the 2018 water year than 2019. Lower TIN concentrations during spring are generally associated with primary productivity (i.e., demand and uptake by aquatic plants and/or algae); increases suggest that demand and uptake decreased. However, given the relatively short distance between LSR, LSM, and LSH (approximately 2.4 km), similar TIN values at each site in the foothills reach during each season are consistent with the limited productivity within the study reach itself (see the macroinvertebrate section for more details on reach productivity). Concentrations generally decreased at downstream locations CWL and PMK, suggesting that the nitrogen is the limiting factor on productivity in the reach. A relatively high TIN at CWL during spring 2019 is due to the May 1, 2019 sample, which had a concentration of 0.24 mg/L. While this concentration is relatively high compared to the other sites, it is consistent with values observed in other unregulated watersheds with rangeland (Ahearn et al. 2005).

Total nitrogen (TN) showed a different response than TIN. No clear trend was observed from upstream to downstream across the three sites in the foothills reach. In fall 2018 and winter 2019, TN decreased from LSR to LSH. In fall and spring 2019, TN increased. In spring and summer 2018, no consistent trend was observed. Across seasons, a more general trend emerged: TN

18

generally decreased from fall to spring, then increased until the following winter. The relative proportion of TIN to TN also changed from the 2018 to 2019 water year. During the 2018 and 2019 water years, TIN accounted for 19-69% and 6-29% of total nitrogen at each foothills reach site, respectively. Thus, total organic nitrogen loads were higher in 2019 than 2018. This increase may be due to the sampling schedule, which included some of the first-flush, high-flow events during the 2019 water year. Samples taking during this time would have captured elevated levels of TIN as nitrogen stored in banks and floodplains would have been entrained to the main channel (Ahearn et al. 2004). In addition, higher concentrations of TN at PMK mainly consist of organic nitrogen, which is consistent with runoff containing organic matter (i.e., plant and animal material) flowing into the channel. As zero-flow conditions were observed downstream of LSH, all water in the channel would have been due to either groundwater accretion or return flow.

Figure 13. A plot of total inorganic nitrogen (TIN) and total nitrogen (TN) at each monitoring site. Data are arranged within each season from upstream to downstream. LSR, LSM, and LSH are located in the foothills reach; CWL and PMK are located in the bottomlands reach. Seasons are defined as: fall: October, November, and December; winter: January, February, and March; spring: April, May, and June; and summer: July, August, and September. Years indicate water year, which extend from October 1 through September 30.

Phosphate

Like nitrogen, phosphorus is an essential nutrient for plant growth, and is often described as a pollutant (e.g. from fertilizers, pesticides, detergents). In combination with nitrogen, phosphorus can lead to abundant primary productivity; inorganic phosphorus (i.e., orthophosphate) is available for uptake by aquatic plants. Data for both total phosphorus (organic and inorganic) and inorganic phosphorus are presented.

Similarly to TIN, TIP remained relatively consistent at each site in the foothills reach (LSR, LSM, and LSH) during each season, and concentrations ranged between 0.03-0.08 mg/L (Figure 14). TN showed similar consistency from upstream to downstream sites, and ranged between 0.05-0.13 mg/L. The seasonal and spatial stability suggests that phosphorus is not limiting primary productivity in the Little Shasta River. The proportion of TIP to TP supports trends expected with seasonal growth and senescence. During the winter, TIP accounts for less TP; this is consistent

with senescence, when organic material from plants and bacteria are decaying. During the summer, TIP comprises the majority of TP, when plants typically experience peak growth and less decay.
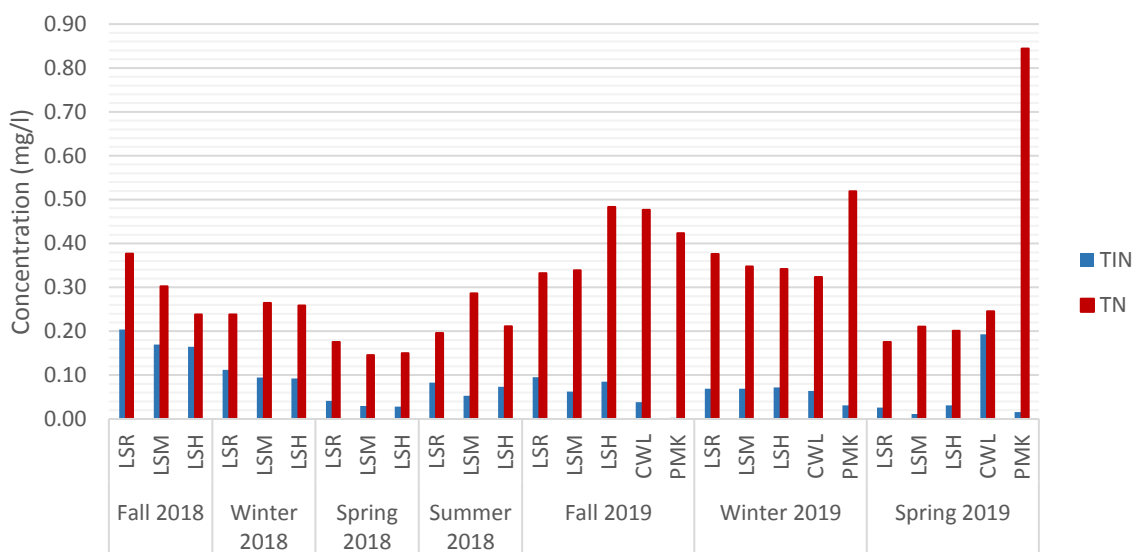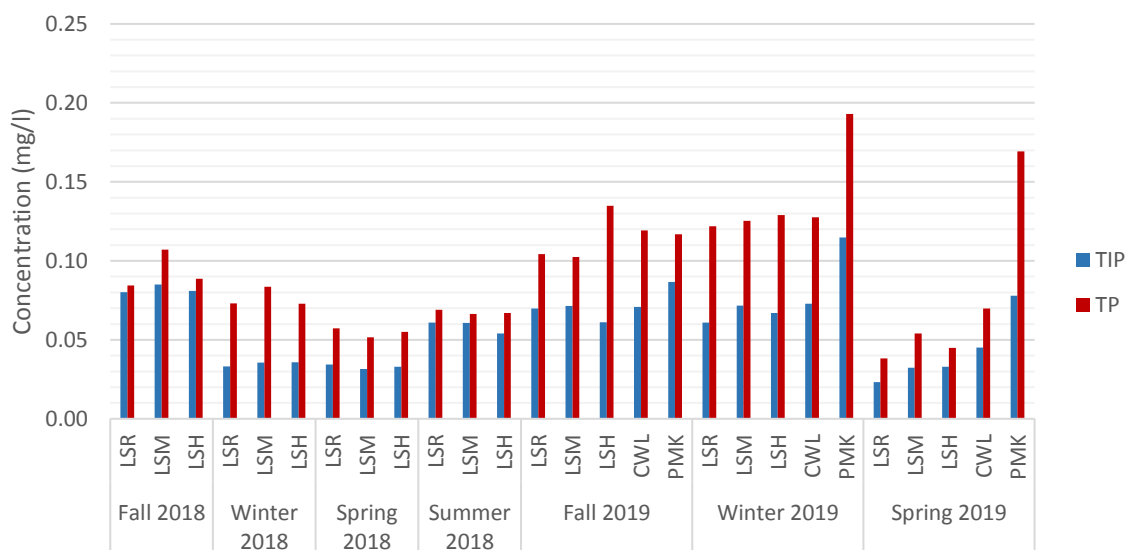

Figure 14. A plot of total inorganic phosphorus (TIP) and total phosphorus (TP) at each monitoring site. Data are arranged within each season from upstream to downstream. LSR, LSM, and LSH are located in the foothills reach; CWL and PMK are located in the bottomlands reach. Seasons are defined as: fall: October, November, and December; winter: January, February, and March; spring: April, May, and June; and summer: July, August, and September. Years indicate water year, which extend from October 1 through September 30.

N:P Ratio

The relationship between Nitrogen (N) and Phosphorus (P), also considered the Redfield ratio (Redfield 1934), is an important indication of water quality in terms of both ecosystem health and function. The primary productivity of a body of water is in large part determined by N:P mass ratios. Higher rates of primary productivity directly affect each successive trophic level and lead to greater abundance of food for salmonids. However, excessive rates of primary production can lead to the eutrophication of a water system through the rapid increase of phytoplankton biomass (algal blooms), the respiration of which depletes dissolved oxygen from the system. Generally, a ratio less than 7:1 by mass is associated with a nitrogen limitation; a ratio greater than 7:1 translates to a phosphorus limitation (Kalff 2002), although local conditions can lead to deviations in these ratios.

Using the inorganic forms (i.e., those available for plant uptake) the nitrogen to phosphorus ratio (by mass) was calculated for each location by season. Throughout the project area, the TIN:TIP ratio was well under 7:1, indicating that the system is generally nitrogen-limited (Figure 15). The upstream boundary of the study reach, LSR, showed the highest ratio, 3.4:1 during winter 2018. Subsequent seasonal ratios showed far lower values, with a minimum of 0.4:1 at LSM during spring 2019.
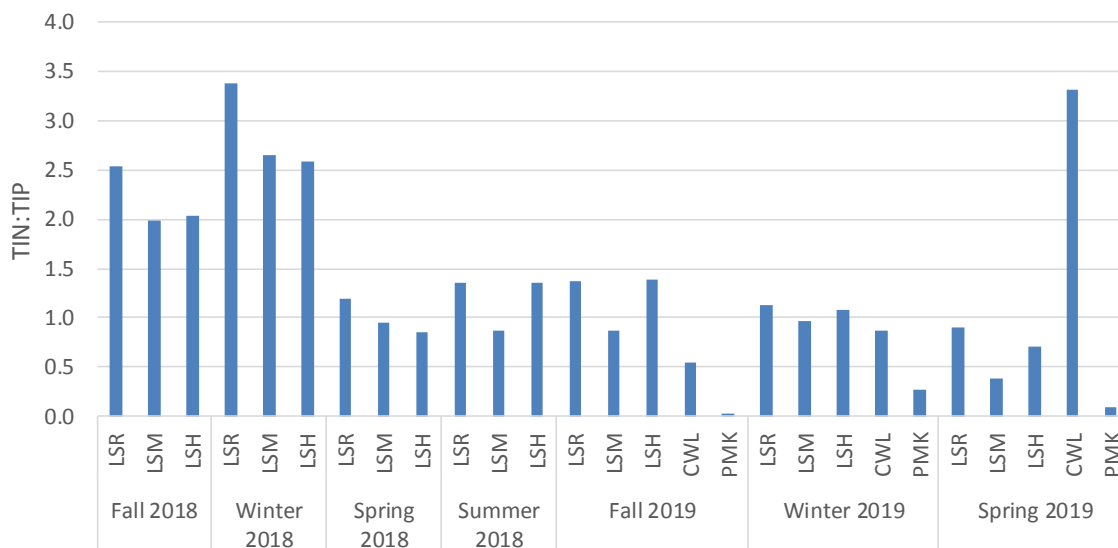
Figure 15. Total inorganic nitrogen to total inorganic phosphorus (TIN:TIP) by location and season in the Little Shasta River. Data are arranged within each season from upstream to downstream. LSR, LSM, and LSH are located in the foothills reach; CWL and PMK are located in the bottomlands reach. Seasons are defined as: fall: October, November, and December; winter: January, February, and March; spring: April, May, and June; and summer: July, August, and September. Years indicate water year, which extend from October 1 through September 30.

Carbon

Carbon is an essential nutrient for plant growth, and an important limiting factor in macroinvertebrate production; it also lends insight to the fate and transport of organic matter in a river system. Dissolved organic carbon (DOC) in the Little Shasta River varied little from LSR to LSH, consistent with the trends observed for nitrogen and phosphorus (Figure 16). However, the magnitude of DOC concentrations varied considerably from baseflow to event-driven samples. DOC in samples taken during stable flow conditions (i.e., fall 2018 through fall 2019) ranged from 1.3 to 3.0 mg/l, with the exception of PMK (4.7 mg/l in fall 2019). However, concentrations at all sites increased during high flow events, and ranged from a minimum of 4.26 mg/l at LSR to 14.35 mg/l at PMK during spring 2019. For context, previous analysis of DOC in the Shasta River upstream of its confluence with the Little Shasta River has been on the order of 1.5 to 2.5 mg/l throughout the year (Jeffres et al. 2010).
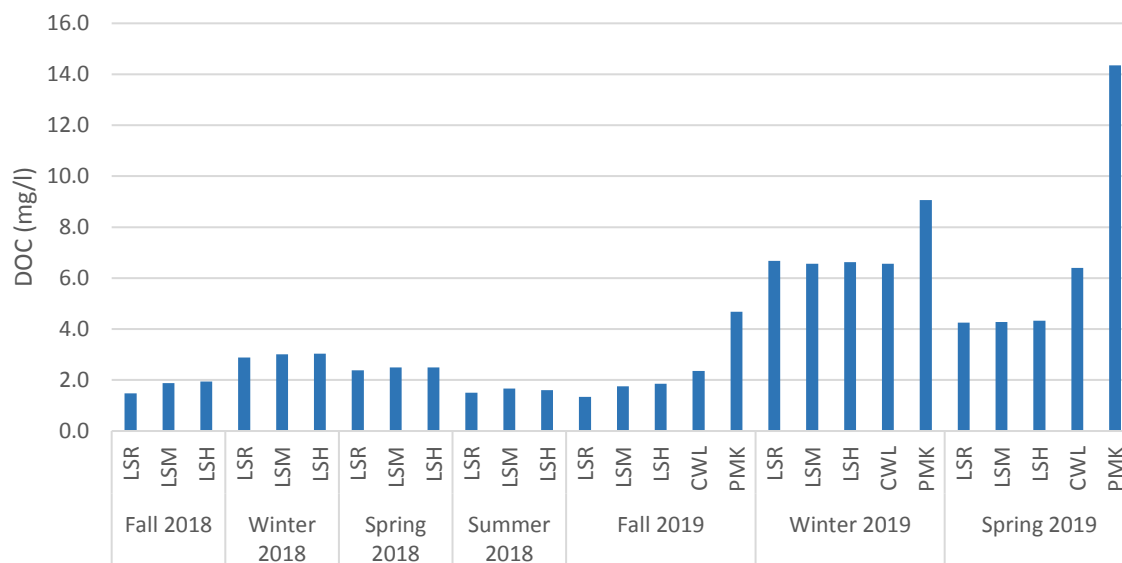
Figure 16. Dissolved organic carbon (DOC) by location and season in the Little Shasta River. Data are arranged within each season from upstream to downstream. LSR, LSM, and LSH are located in the foothills reach; CWL and PMK are located in the bottomlands reach. Seasons are defined as: fall: October, November, and December; winter: January, February, and March; spring: April, May, and June; and summer: July, August, and September. Years indicate water year, which extend from October 1 through September 30.

Macroinvertebrates

Stream macroinvertebrates were collected during June 2019 from three locations in the Little Shasta River (lower Cowley Ranch, upper Cowley Ranch, and upper Hart – see Figure 3) to determine community composition and assess the potential for differences in assemblages between locations pre- and post-project. Stream macroinvertebrate density was greatest at the most upstream Cowley Ranch site (C1; 5,285 invertebrates·m^{-2}), followed by the downstream most Cowley Ranch site (C2; 4,540 invertebrates·m^{-2}) and the upper Hart Property (H1; 2,360 invertebrates·m^{-2}), respectively (Figure 17). Despite higher densities associated with the Cowley Ranch property, several richness indices indicated that the Hart Ranch sampling site (H1) exhibited a more diverse macroinvertebrate community (Figure 17). This is contrary to what is typically observed in a natural systems where the relationship between density and richness is often positive (Gotelli and Colwell 2001, Lusardi et al. 2016).
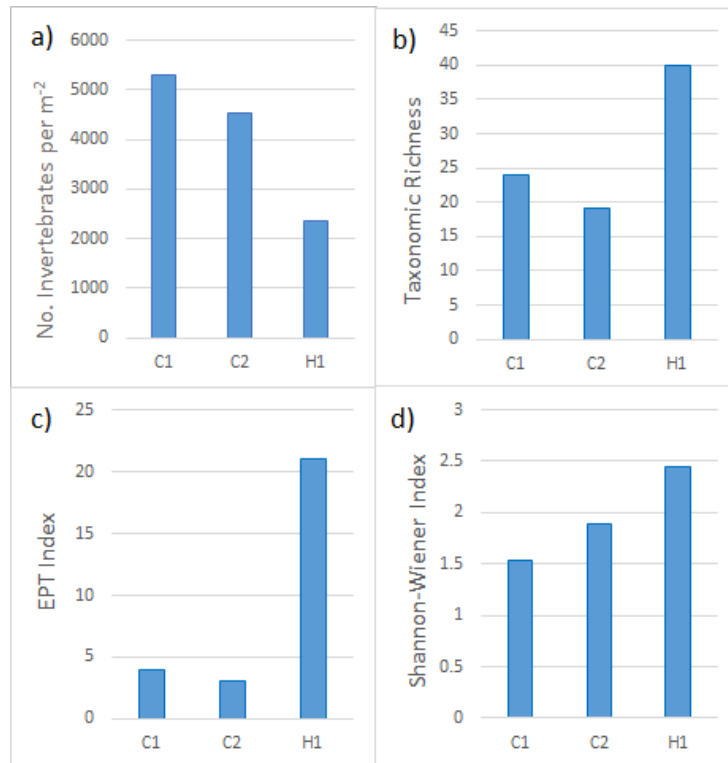
Figure 17. Aquatic macroinvertebrate data at three locations on the Little Shasta River: (a) aquatic invertebrate density (no.·m⁻²), (b) taxonomic Richness, (c) EPT index, and (d) Shannon-Wiener Index. H1 = upper Hart Ranch; C1 = Cowley Ranch at Cattle Crossing; C2 = lower Cowley Ranch.

Despite lower densities associated with H1, total taxonomic richness values were up to two-fold greater (H1: 40 vs. C2: 19) when compared to the assemblage patterns on the Cowley Ranch (C1 and C2). Similarly, the EPT index, a measure of the relative contribution of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) was significantly greater at H1 than C1 and C2 (Figure 17). H1 exhibited an EPT index of 21, while C1 and C2 exhibited an EPT index of 4 and 3, respectively. EPT taxa are sensitive to anthropogenic disturbance and require cold, well-oxygenated water, and substrate clean of fine sediment. Such low EPT Index scores at C1 and C2 suggest significant aquatic habitat impairment. The H1 sampling site also scored significantly higher than C1 and C2 using the Shannon-Wiener diversity index, which combines species richness and evenness (Figure 17). The Shannon-Wiener index at H1 was 2.45 versus 1.53 and 1.89 at C1 and C2, respectively. When decoupling between density and diversity occurs, as observed at C1 and C2, it may be reflective of poor habitat quality and the dominance of a few species or genera that are capable of both colonizing and surviving under stressful conditions. All three of the major indices used here (taxonomic richness, EPT index, and the Shannon-Wiener Index) suggest such patterns are evident at C1 and C2, but not at H1.

The abundance of tolerant organisms (those with published tolerance values > 7) are generally associated with anthropogenic disturbance or poor aquatic habitat conditions (Klemm et al. 2003; Merrit, Cummins, and Berg 2008). Tolerant species dominated sites C1 and C2, accounting for approximately 25% and 60% of the entire macroinvertebrate assemblage at those locations, respectively (Figure 16). Conversely, the H1 sampling site exhibited relatively few tolerant species, but numerous sensitive species (those with published tolerance values < 2). Sensitive species are indicative of good water quality and habitat. H1 exhibited a 52-fold increase in

sensitive species compared to sites C1 and C2. Such sensitive species accounted for approximately 18% of the entire assemblage at H1 (Figure 18). Sensitive species were prevalent at H1 and included numerous stonefly genera including *Calineuria sp.*, *Hesperloperla sp.*, *Pteronarcys californica*, and multiple genera from the families Nemouridae and Chloroperlidae. No stoneflies were observed at sites C1 and C2. Finally, the Hillsenhoff biotic index (HBI), a composite measure of benthic community tolerance to organic pollution, can also be used to describe the general condition of lotic habitats (Hilsenhoff 1987). We used a modified biotic index scoring system (Table 10) which included tolerance values associated with genus level identification. Overall, C1 and C2 exhibited low biotic index values (6.35 and 7.0) indicating that water quality was "fairly poor" at both sites. Alternatively, H1 exhibited a biotic index score of 4.69 indicating "good" water quality (Table 10; Figure 18).

Table 10. Criteria for the evaluation of water quality using Hilsenhoff's Bioti Index (HBI; Hilsenhoff 1987). HBI values are derived from macroinvertebrate tolerance values weighted by the number of individuals of each taxa in the total sample.

HBI Value	Water Quality	Degree of Organic Pollution
0.00-3.5	Excellent	No apparent organic pollution
3.51-4.5	Very good	Possible slight organic pollution
4.51-5.5	Good	Some organic pollution
5.51-6.5	Fair	Fairly signficant organic pollution
6.51-7.5	Fairly Poor	Significant organic pollution
7.51-8.5	Poor	Very significant organic pollution
8.51-10.0	Very Poor	Severe organic pollution
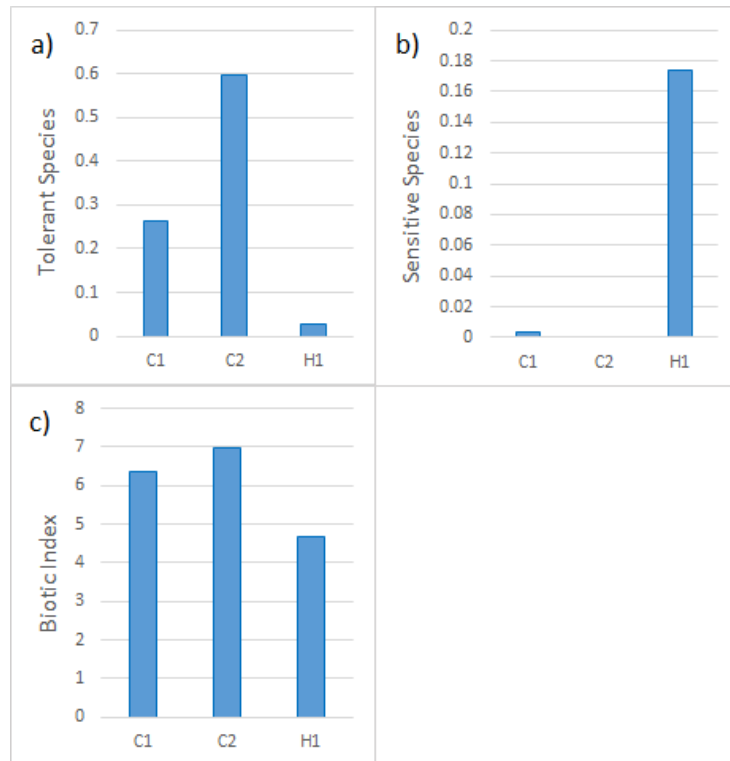
Figure 18. Aquatic macroinvertebrate data at three locations on the Little Shasta River: Relative abundance of tolerant macroinvertebrate species, relative abundance of sensitive macroinvertebrate species, and Hilsenhoff biotic index (HBI). H1 = upper Hart Ranch; C1 = Cowley Ranch at Cattle Crossing; C2 = lower Cowley Ranch.

Fish Presence/Absence

Spawning and Carcass Surveys

Adult spawning and carcass surveys were conducted during December and January of 2017 and again during October, November, December and January 2018. Spawning and carcass surveys were conducted at two primary reaches on the Hart property (Figure 3). No salmon redds, live salmon, or carcasses were observed during the period of study.

Snorkel Surveys

Numerous fishes were observed during snorkel surveys on the upper Hart property including rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), Klamath smallscale sucker (*Catostomus rimiculus*), marbled sculpin (*Cottus klamathensis)*, speckled dace (*Rhinichthys osculus)*, and adult lamprey (*Lampetra sp*.; likely Klamath Brook lamprey or Klamath River lamprey) (Figure 19). Over all reaches and surveys, *O. mykiss* were observed most frequently followed by speckled dace, Klamath small-scale sucker, brown trout, marbled sculpin, and lamprey. No coho salmon (*Oncorhynchus kisutch*) or Chinook salmon (*Oncorhynchus tshawytscha)* were observed during the survey period. Due to poor water quality conditions, we were unable to dive at snorkel site 3 (SR3) (Figure 3).
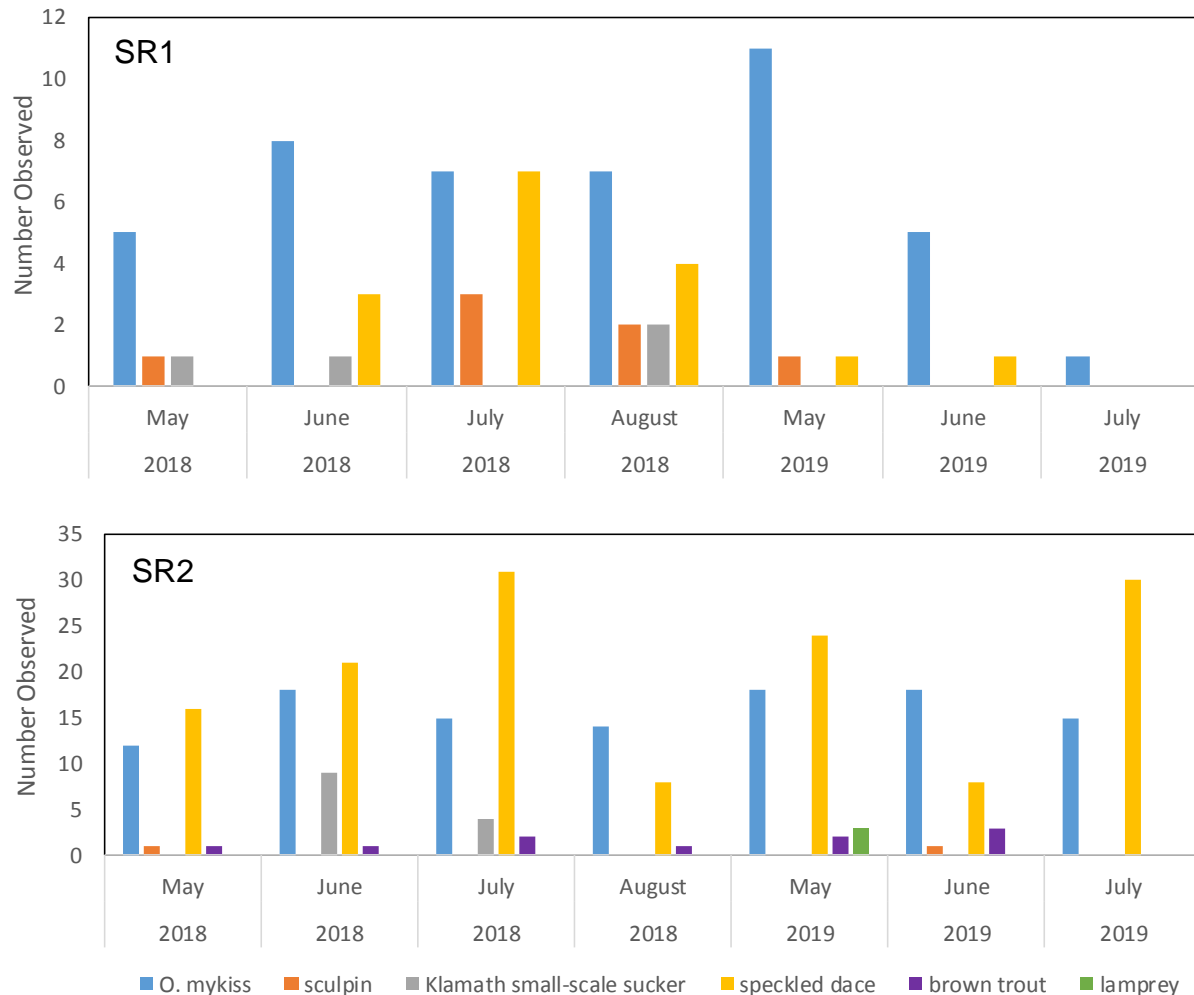
Figure 19. Relative number of each species of fish observed in snorkel reach 1 (SR1) and 2 (SR2) at the Little Shasta River throughout the study period.

We observed differences in assemblage dynamics between the two snorkel reaches on the upper Hart property (i.e., above and below the Musgrave diversion). In general, both the abundance and diversity of fishes above the Musgrave diversion were greater than below the diversion (Figure 20). Over the entire study period, reach 2 (SR2) supported an average diversity of 3.6 species and 39 individuals observed per dive compared with an average of 2.7 species and 10 individuals observed per dive in Reach 1 (SR1, Figure 20). Reach 2 also supported multiple age classes of *O. mykiss*, the most abundant species observed, while Reach 1 generally supported older O. mykiss (> 1+) individuals, although young of year were occasionally observed. During spring of 2018, Klamath small-scale sucker were observed in both reaches, but not during 2019. Brown trout, an introduced species, were observed in reach 2, but not reach 1 (Figure 19). Brown trout were represented by multiple age classes during most dives suggesting a robust population.
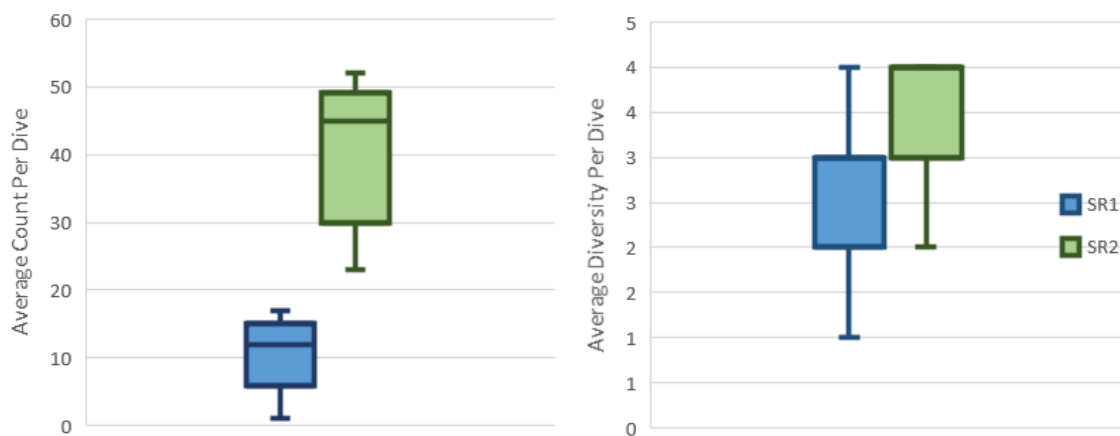
Figure 20. Average count of fish observed per dive in two snorkel reaches (left), and the average diversity of fishes observed per dive in each snorkel reach (right). Center line: median; box limits: upper and lower quartiles; whiskers: maximum and minimum interquartile range.

Conclusions

This report summarizes the physical, chemical, and biological conditions in the Little Shasta River, with particular focus on stream reaches above and below the Hart-Haight diversion (monitoring site LSH). The objective of this report is to document stream conditions prior to the replacement of the diversion and channel restoration, establishing a baseline from which project effectiveness can be assessed. In addition, because this data also provides the first comprehensive assessment of the Little Shasta River, it has also been used to understand how this stream fits into the broader ecosystem function and conservation strategy of the Shasta Basin. A discussion of the Little Shasta River's role in the Shasta watershed, as well as recommendations for future activities in the Little Shasta River, are included following the summary of analyses and findings.

Summary

As the Little Shasta River is predominately driven by winter runoff and spring snowmelt, the headwaters provided the greatest source of streamflow to the river. Steep cascades form a natural barrier that prevent fish migration into the headwaters (SVRCD, McBain & Trush 2013). Below these cascades, the Little Shasta River transitions from its headwaters to the foothills reach. In the foothills reach, physical, chemical, and biological conditions indicated a healthy cold-water ecosystem that could support coho salmon. Elevated water temperatures that would coincide with juvenile oversummering and limited productivity from low nutrient content appeared to be the main limiting factors to this reach, though both macroinvertebrate and fish data suggest that current conditions are sufficient to support cold-water ecosystem function. The original Hart-Haight diversion marks a transition in the Little Shasta River from the foothills reach to the bottomlands. Below this diversion, seasonally occurring low-flow conditions disconnected the bottomlands from the foothills. In addition to the zero-flow conditions observed in this reach during the summers, macroinvertebrate and fish data indicated degraded ecological function. Near the mouth of the Little Shasta River, water quality showed elevated levels of organic material in the form of nitrogen, phosphorus, and carbon.

These data and subsequent analyses suggest that each reach in the Little Shasta River – headwaters, foothills, and bottomlands – plays a distinct and important role in the stream's overall potential function and management strategy to support the broader recovery of coho salmon. Monitoring in the headwaters provides a valuable assessment of each water year's hydrologic type. The foothills provide potential year-round habitat for coho salmon; however, data collected during this and previous studies suggests that the foothills reach is likely the downstream limit of year-round habitat. The bottomlands potentially function a migratory corridor into and out of the foothills reach. We did not assess the bottomlands potential for spawning habitat.

Despite the presence of functioning year-round habitat in the foothills, our data suggest that the Little Shasta River does not currently support anadromous salmonid populations. Throughout the study period, we did not observe coho or Chinook salmon. We speculate that the primary limiting factor on salmon production in the Little Shasta River is the lack of connectivity and habitat throughout the bottomlands to the foothills. Access to additional reaches could improve understanding of issues like water quality dynamics. Existing habitat remains disconnected from the bottomlands throughout much of spring and into the fall, which would coincide with outmigrating juvenile coho and early returning adults. Encouraging coho to utilize the Little Shasta River will require sufficient flow during the adult migration period (late fall through winter) and again during the spring juvenile outmigration period. Since the timing of flows is an important restoration feature, a functional flows approach (Yarnell et al. 2015) may provide a valuable framework to approach restoration in the Little Shasta River. Additional work may be needed to address water quality in the bottomlands.

Secondary limitations may exist in the foothills as well. Low productivity and limited food resources suggest that juvenile salmonids may be less able to tolerate elevated water temperatures in the reach, even for short periods. Thus, unlike the food-habitat-temperature nexus observed in the upper Shasta River and Big Springs Creek (Lusardi et al. 2019), coho in the Little Shasta River may not be able to compensate for slight increases in temperature with food resources. Ultimately, this suggests that cold water (i.e., Welsh et al. 2001) during the oversummer period will be paramount to coho recovery in the Little Shasta River.

Role of the Little Shasta River in the Shasta watershed

Within the broader context of coho conservation in the Shasta watershed, the Little Shasta River could play an important role in the recovery of listed salmonids in the basin. The differences, from an ecological and physical system perspective, between the Shasta River and Little Shasta River are immense. The river's hydrologic and thermal regimes are largely dictated by rainfall and snowmelt runoff events (rather than springs), which is reflected in the physical and biological data presented in this report. This is contrary to the mainstem Shasta River, which functions as a large, volcanic, spring-fed river, exhibiting relatively stable flow and thermal regimes, and abundant, geologically derived nutrients that enhance instream productivity. These fundamental differences in hydrology, geomorphology, and ecology, however, will likely be important factors influencing the recovery of listed salmonids in the basin. Moyle et al. (2017) suggested that restoration of diverse and productive habitats provides the best hope for the recovery of salmonids. The Little Shasta River is nested within the broader Shasta River watershed; because of the basin's unique geology, these rivers and others provide a portfolio of habitat types and conditions, flow, thermal, and productivity regimes known to influence life history timing and salmonid production. Restoring these habitats and connections between these

habitats is fundamental to improving population resiliency and, ultimately, the status of salmon in the basin.

Recommendations

On-going monitoring will continue to assess the stream's response to the removal of the Hart-Haight diversion and channel restoration. Given the findings of this report, additional work is recommended to improve understanding of the physical, chemical, and biological dynamics in the Little Shasta River. Specifically, we recommend:

- Maintain hydrologic year-typing based on snow survey data in the headwaters to better quantify available water resources each year.

- Establish permanent flow gages at LSH and CWL, and a temporary site at PMK, to understand connectivity through the reach and track future flow dedications planned under this project.

- Develop a water temperature model to identify the main drivers of stream warming during summer in the foothills reach.

- Monitor streamflow, water temperature, food webs, and water quality (nutrients and dissolved oxygen) in the bottomlands reach to understand how the Little Shasta influences the mainstem Shasta River, and potentially identify limiting factors to its ability to attract anadromous fish.

- Explore potential connections to groundwater that might influence summer baseflows.

- Collect water quality samples in the bottomlands reach at both baseflows and high flows to examine potential differences in the amount of exported DOC to the Shasta River.

References

Ahearn, D. S., Sheibley, R. W., Dahlgren, R. A., Anderson, M., Johnson, J., & Tate, K. W. (2005). Land use and land cover influence on water quality in the last free-flowing river draining the western Sierra Nevada, California. Journal of Hydrology, 313(3), 234-247. doi:https://doi.org/10.1016/j.jhydrol.2005.02.038

Ahearn, D. S., Sheibley, R. W., Dahlgren, R. A., & Keller, K. E. (2004). Temporal dynamics of stream water chemistry in the last free-flowing river draining the western Sierra Nevada, California. Journal of Hydrology, 295(1), 47-63. doi:https://doi.org/10.1016/j.jhydrol.2004.02.016

Apperson KA, Copeland T, Flinders J, Kennedy P, Roberts RV. 2015. Field protocols for stream snorkel surveys and efficiency evaluations for anadromous parr monitoring. Idaho Department of Fish and Game.

Bash, J., Bolton, S., Berman, C. 2001. Effects of turbidity and suspended solids on salmonids. Center for Streamside Studies, University of Washington.

Carter, K. 2005. The effects of dissolved oxygen on Steelhead trout, Coho salmon, and Chinook salmon biology and function by life stage. California Regional Water Quality Control Board, North Coast Region.

Carter K. 2008. Effects of temperature, dissolved oxygen/total dissolved gas, ammonium, and pH on Salmonids. North Coast Regional Water Quality Control Board. Appendix 4, 53 pg.

Gallagher SP. 2001. Results of the 2000-2001 Coho salmon (*Onchorhynchus kisutch*) and Steelhead (*Onchorhynchus mykiss*) spawning surveys on the Noyo River, California. California State Department of Fish and Game. Fort Bragg, California. 44 pp.

Gotelli, N. J., and R. K. Colwell. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecology Letters 4:379–391.

Hilsenhoff WL. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist, 20: 31-39

Ikuta K, Suzuki Y, Kitamura S. 2003. Effects of low pH on the reproductive behavior of salmonid fishes. Fish Physiology and Biochemistry, 28: 407-410.

Jeffres C, Dahlgren R, Kiernan J, King A, Lusardi R, Nichols A, Null S, Tanaka S, Willis A. 2009. Baseline assessment of physical and biological conditions within waterways on Big Springs Ranch, Siskyou County, California. Report prepared for: California State Water Resources Control Board.

Kalff J. 2002. Limnology. Inland Water System. Prentice Hall, NJ. 592 pp.

Kennedy CJ, Stecko P, Truelson B, Petkovich D. 2012. Dissolved organic carbon modulates the effects of copper on olfactory-mediated behaviors of Chinook salmon. Environmental Toxicology and Chemistry, 31(10): 2281-2288.

Klemm DJ, Blocksom KA, Fulk FA, Herlihy AT, Hughes RM, Kaufmann PR, Peck DV, Stoddard JL, Thoeny WT, and Griffith MB. 2003. Development and evaluation of a macroinvertebrate biotic integrity index (MBII) for regionally assessing Mid-Atlantic Highlands streams. Environmental Management 31:656-669.

Lusardi, R. A., Bogan, M. T., Moyle, P. B., & Dahlgren, R. A. 2016. Environment shapes invertebrate assemblage structure differences between volcanic spring- fed and runoff rivers in northern California. Freshwater Science, *35*(3), 1010–1022. https://doi.org/10.1086/687114

Lusardi, R. A., Hammock, B. G., Jeffres, C. A., Dahlgren, R. A., & Kiernan, J. D. 2019. Oversummer growth and survival of juvenile coho salmon (Oncorhynchus kisutch) across a natural gradient of stream water temperature and prey availability: an in situ enclosure experiment. Canadian Journal of Fisheries and Aquatic Sciences. In press.

McMahon T.E. 1983. Habitat suitability index models: Coho salmon. Western Energy and Land Use Team, U.S. Fish and Wildlife Service. 40 pp.

Merrit RW, Cummins KW, Berg MB. 2008. An introduction to the aquatic insects of North America. 4th Ed. Iowa, USA. Kendall/Hunt Publishing Company.

Moyle, P. B., Lusardi, R. A., Samuel, P., & Katz, J. (2017). *State of the Salmonids: Status of California's Emblematic Fishes 2017*. Report prepared for California Trout.

(NCRWQCB) Total Maximum Daily Load Development Unit Staff. 2005. Salmon River, Siskyou County, California: Total maximum daily load for temperature and implementation plan. North Coast Regional Water Quality Control Board. 51 pp.

Nichols A, Willis A, Lambert D, Limanto E, Deas M. 2016. Little Shasta River baseline hydrologic and water temperature assessment: April to December 2015. 24 pp.

Nichols A, Lusardi R, Willis A. 2017. Little Shasta River aquatic habitat assessment. 28 pp.

NOAA. 2018. California Nevada River Forecast Center: Monthly Precipitation Summary Water Year 2018.

Rantz SE. 1982. Measurements and computation of streamflow: Volume 1, measurement of stage and discharge. U.S. Geological Survey, Water Supply Paper 2175.

Redfield AC. 1934. On the proportions of organic derivatives in sea water and their relation to the composition of plankton. James Johnstone Memorial Volume, University Press of Liverpool, 176-192.

SVRCD, McBain & Trush. 201). Study Plan to Assess Shasta River Salmon and Steelhead Recovery Needs. Report prepared for U.S. Fish and Wildlife Service: Arcata Fish and Wildlife Office.

Smith RA, Alexander RB, Schwarz GE. 2003. Natural background concentrations of nutrients in streams and rivers of the conterminous United States. Environmental Science & Technology, 37:14, pp. 3039-3047

Sturm M, Taras B, Liston GE, Derksen C, Jonas T, Lea J. 2010. Estimating snow water equivalent using snow depth data and climate classes. Journal of Hydrometerorology, Vol. 11: pp 1380-1394.

(SWAMP) Ode PR, Herbst D. 2007. Standard operating procedures for collecting benthic macroinvertebrate samples and associates physical and chemical data for ambient bioassessments in California. California State Water Resources Control Board.

Thorp JH, Covich AP. 2001. Ecology and classification of North American freshwater invertebrates. Second Edition. Academic Press.

USEPA. (2003). EPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards. (EPA 910-B-03-002). Seattle, WA Retrieved from https://www3.epa.gov/region10/pdf/water/final_temperature_guidance_2003.pdf

Willis A, Deas M. 2012. Response to restoration: water temperature conditions in Big Springs Creek and surrounding waterways, 2009-2011. Watercourse Engineering, Inc.

Willis A, Nichols A. 2015. Little Shasta River experimental flow: July 2015.

Yarnell, S. M., Petts, G. E., Schmidt, J. C., Whipple, A. A., Beller, E. E., Dahm, C. N., Goodwin, P., Viers, J. H. (2015). Functional Flows in Modified Riverscapes: Hydrographs, Habitats and Opportunities. BioScience, 65(10), 963-972. doi:10.1093/biosci/biv102

A Rating Curve Data

Table A- 1. Discharge and stage measurements recorded for the three sites in the upper reach of the Little Shasta River (LSR, LSM, and LSH) during monitoring from 2017 through 2019.

Discharge measurements		LSR			LSM			LSH	
Date	Time	Stage (m)	Discharge (ft³/s)	Time	Stage (m)	Discharge (ft³/s)	Time	Stage (m)	Discharge (ft³/s)
7/24/2017	14:00	0.3078	8.35	14:44	0.152	4.24	15:33	0.448	2.66
8/15/2017	14:20	0.2987	5.40	17:05	0.151	4.27	18:00	0.448	2.31
10/20/2017	12:00	0.3292	8.48	12:30	0.158	6.14	12:58	0.509	3.63
11/17/2017	8:02	0.3688	13.41	9:04	0.167	10.25	9:30	0.570	10.76
12/21/2017	10:03	0.3444	10.84	11:00	0.166	8.96	11:50	0.582	10.93
1/29/2018	10:08	0.4023	16.52	10:55	0.188	16.82	10:08	-	19.49
2/25/2018	11:00	0.3810	12.67	12:00	0.175	10.44	12:30	0.588	11.33
3/31/2018	12:36	0.4267	21.44	11:25	0.183	16.49	11:58	0.701	7.10
4/8/2018	11:00	0.5873	71.62	12:22	0.220	56.06	13:45	0.762	49.41
6/14/2018	7:45	0.5723	7.54	-:--	-	-	-	-	-
8/22/2018	11:31	0.3231	3.89	11:31	0.134	3.21	14:00	-	1.31
2/28/2019	17:45	0.2957	55.09	12:30	0.000	41.46	-	-	51.77
3/21/2019	13:18	0.5608	51.84	12:26	0.000	42.70	11:31	0.448	40.68

B Water Quality Data

Electrical Conductivity (EC)

Table B- 1. A summary of the electrical conductivity (EC) measurements from each monitoring site from 2017-2018, given in Siemens per meter (S/m).

	Electrical Conductivity (EC)																		
	2018 Water Year											**2019 Water Year**							
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	253	133	120	125	166	203	119	139	140	126	133	131	103	102	101	105	103	77.2	76.4	88.6
LSM	328	208	183	157	121	125	140	273	310	157	151	170	167	149	156	149	92	94.9	82.7	127.0
LSH	362	233	201	190	97	97	212	147	159	347	304	430	227	176	166	95	94	108.1	118.3	200.1
CWL													261	192	192	163	95	115.8	144.5	331.1
PMK													466	373	445	412	229	341.6	480.9	775.7

Acidity/Alkalinity (pH)

Table B- 2. The pH measurements from each monitoring site from 2017-2018

	Acidity/Alkalinity (pH)																		
	2018 Water Year											**2019 Water Year**							
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	7.82	7.99	7.83	8.04	7.67	7.57	7.56	7.71	7.74	7.75	8.23	8.02	7.79	7.54	7.33	7.63	7.99	7.84	7.80	7.72
LSM	7.66	8.00	7.85	7.87	7.61	7.67	7.61	7.59	7.73	7.73	8.20	7.98	7.74	7.49	7.34	7.94	7.65	7.66	7.88	7.65
LSH	7.54	7.83	7.67	7.88	7.56	7.63	7.58	7.62	8.06	7.71	7.96	7.50	7.52	7.36	7.29	7.90	7.52	7.83	7.67	7.55
CWL													7.91	7.42	7.45	7.88	7.50	7,78	7.88	8.12
PMK													8.08	7.87	7.97	8.32	8.17	8.18	8.33	8.67

Turbidity

Table B- 3. Turbidity measurements from each monitoring site from 2017-2018.

	Turbidity (NTU)																		
	2018 Water Year											**2019 Water Year**							
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	1.45	1.17	4.11	1.60	10.6	6.46	1.79	1.52	1.73	1.15	1.5	1.8	1.0	2.3	8.3	23.4	12.1	8.04	6.42	NA
LSM	1.77	2.72	4.26	1.30	9.97	5.81	2.03	1.33	28.4	2.23	1.8	2.6	2.5	3.2	6.8	22.4	13.3	7.95	5.07	NA
LSH	1.03	1.33	3.9	1.10	10.2	4.37	1.46	1.81	2.26	1.57	1.2	2.0	5.1	8.4	7.3	23.9	13.1	8.48	3.59	NA
CWL													1.8	5.1	6.7	22.2	12.6	7.76	3.61	NA
PMK													2.9	3.0	5.8	24.6	14.5	8.16	10.50	NA

Dissolved Organic Carbon (DOC)

Table B- 4. The dissolved organic carbon (DOC) measurements (in mg/L) for each monitoring site from 2017-2018.

Dissolved Organic Carbon (DOC-mg/l)																			
	2018 Water Year											2019 Water Year							
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	1.8	1.1	2.7	1.3	4.7	3.4	2.0	1.7	2.0	1.4	1.2	1.5	0.9	1.6	4.0	8.6	7.4	6.8	4.2	1.8
LSM	2.2	1.6	2.9	1.3	4.8	3.3	2.2	2.0	2.0	1.5	1.5	1.8	1.7	1.8	3.8	8.3	7.6	6.9	4.0	1.9
LSH	2.2	1.7	2.9	1.4	4.9	3.1	2.1	2.2	2.0	1.5	1.3	1.9	1.9	1.8	3.8	8.3	7.8	6.8	4.1	2.1
CWL													2.3	2.4	3.9	8.2	7.7	7.3	5.0	6.9
PMK													5.3	4.0	6.8	11.3	9.0	12.8	15.0	15.3

Ammonium

Table B- 5. The ammonium (NH$_4$) measured at each monitoring site from 2017-2018, measured in mg/L.

Ammonium (NH$_4$)																			
	2018 Water Year											2019 Water Year							
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	0.00	0.05	0.02	0.03	0.01	0.04	0.04	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.01	0.01	0.02	0.01	0.01	0.02
LSM	0.02	0.05	0.04	0.03	0.01	0.02	0.04	0.00	0.03	0.00	0.01	0.00	0.06	0.00	0.03	0.01	0.02	0.01	0.00	0.00
LSH	0.03	0.04	0.03	0.03	0.01	0.02	0.03	0.00	0.01	0.03	0.04	0.05	0.02	0.01	0.04	0.01	0.02	0.01	0.02	0.03
CWL													0.01	0.05	0.05	0.01	0.01	0.01	0.23	0.31
PMK													0.00	0.00	0.03	0.01	0.01	0.00	0.01	0.03

Nitrogen

Table B- 6. The nitrate concentrations measured monthly in each monitoring site in the Little Shasta River during the 2018 and 2019 water years.

Nitrates (NO$_3$-ppm)																			
	2018 Water Year											2019 Water Year							
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	0.16	0.20	0.10	0.14	0.04	0.01	0.03	0.00	0.03	0.08	0.08	0.04	0.14	0.11	0.11	0.06	0.00	0.00	0.02	0.02
LSM	0.10	0.16	0.09	0.09	0.02	0.01	0.02	0.00	0.03	0.02	0.06	0.00	0.06	0.06	0.10	0.06	0.00	0.00	0.01	0.01
LSH	0.10	0.16	0.10	0.09	0.03	0.01	0.02	0.00	0.02	0.11	0.05	0.04	0.06	0.08	0.09	0.05	0.00	0.00	0.02	0.01
CWL													0.00	0.02	0.08	0.05	0.00	0.01	0.01	0.01
PMK													0.00	0.00	0.01	0.04	0.00	0.00	0.00	0.00

Table B- 7. *The total concentration of nitrogen (TN) measured in each study site during the 2017-2019 monitoring period. TN is calculated through the combined concentrations of all nitrates within a water column.*

	Total Nitrogen (TN-ppm)																			
	2018 Water Year											**2019 Water Year**								
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	0.36	0.40	0.23	0.27	0.21	0.19	0.20	0.14	0.18	0.13	0.28	0.12	0.20	0.68	0.34	0.43	0.35	0.19	0.17	0.17
LSM	0.31	0.30	0.29	0.29	0.21	0.19	0.16	0.09	0.48	0.13	0.25	0.11	0.18	0.72	0.28	0.44	0.32	0.16	0.24	0.24
LSH	0.23	0.24	0.27	0.29	0.22	0.20	0.16	0.09	0.21	0.13	0.29	0.20	0.13	1.12	0.28	0.45	0.29	0.24	0.18	0.18
CWL													0.03	0.92	0.27	0.42	0.28	0.28	0.23	0.23
PMK													0.20	0.64	0.38	0.76	0.42	0.54	1.00	1.00

Phosphorous

Table B- 8. *The phosphate concentrations measured monthly in each monitoring site in the Little Shasta during the 2018 and 2019 water years.*

	Phosphates (PO$_4$-ppm)																			
	2018 Water Year											**2019 Water Year**								
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	0.075	0.086	0.004	0.063	0.032	0.024	0.037	0.042	0.071	0.065	0.047	0.058	0.088	0.064	0.053	0.103	0.038	0.028	0.015	0.027
LSM	0.081	0.089	0.008	0.065	0.034	0.026	0.040	0.029	0.062	0.062	0.058	0.056	0.088	0.071	0.063	0.103	0.050	0.035	0.027	0.035
LSH	0.078	0.084	0.014	0.063	0.030	0.023	0.037	0.039	0.075	0.046	0.041	0.036	0.079	0.067	0.063	0.101	0.037	0.036	0.029	0.033
CWL													0.071	0.071	0.063	0.106	0.050	0.040	0.039	0.057
PMK													0.090	0.084	0.087	0.187	0.071	0.071	0.089	0.073

Table B- 9. *The total concentration of phosphorus (TP) measured throughout the 2017-2019 monitoring period. This value is calculated through the combined concentrations of all phosphates within a water column.*

	Total Phosphorous (TP-ppm)																			
	2018 Water Year											**2019 Water Year**								
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	0.079	0.090	0.082	0.077	0.060	0.057	0.060	0.054	0.081	0.065	0.061	0.104	0.116	0.097	0.097	0.214	0.072	0.055	0.027	0.033
LSM	0.090	0.124	0.108	0.073	0.070	0.050	0.057	0.047	0.069	0.064	0.066	0.088	0.125	0.097	0.097	0.196	0.083	0.062	0.050	0.050
LSH	0.084	0.093	0.095	0.067	0.057	0.054	0.050	0.061	0.088	0.061	0.053	0.075	0.212	0.100	0.100	0.214	0.072	0.065	0.033	0.036
CWL													0.162	0.107	0.107	0.189	0.086	0.068	0.057	0.084
PMK													0.138	0.134	0.134	0.314	0.131	0.115	0.196	0.196

N:P Ratio

Table B- 10. The ratio between nitrate and phosphorus concentrations measured monthly in the Little Shasta River during the 2018 and 2019 water years.

	N:P ratio (TIN/TIP-ppm)								
	\multicolumn{11}{c}{2018 Water Year}	\multicolumn{8}{c}{2019 Water Year}
	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
LSR	2.15	2.89	27.36	2.66	1.47	2.10	1.89	0.06	0.53	1.88	1.88	0.62	1.60	1.70	2.30	0.73	0.45	0.42	1.83	5.19
LSM	1.54	2.41	16.91	1.91	0.93	0.96	1.53	0.11	0.96	0.35	1.31	0.01	1.40	0.91	2.00	0.64	0.31	0.37	0.42	4.74
LSH	1.68	2.36	8,71	1.85	1.28	1.42	1.31	0.09	0.32	2.39	2.10	2.32	1.05	1.29	2.19	0.61	0.45	0.37	1.12	5.06
CWL													0.15	0.94	1.95	0.56	0.18	0.52	6.19	2.70
PMK													0.03	0.03	0.41	0.27	0.09	0.04	0.12	5.07

C Macroinvertebrate and snorkel survey reach images

Figure C- 1. Snorkel reach 1 (SR1) (left) and 2 (SR2) (right), Little Shasta River.

Figure C- 2. Snorkel photographs taken during the period of study. Top: brown trout (nearest) and rainbow trout holding near large woody debris at SR2. Bottom: juvenile O. mykiss finding velocity refuge near large wood and aquatic macrophytes, SR2.

Figure C- 3. Adult lamprey redd building at SR2 in the Little Shasta River.

Figure C- 4. Macroinvertebrate sampling reach at upper Hart (H1).

Figure C- 5. Macroinvertebrate sampling reach at Cowley Ranch Cattle Crossing (C2).

Figure C- 6. Macroinvertebrate sampling reach at Cowley Ranch (C2).