

Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California Water Year 2007

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And
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1. Executive Summary

During water year 2007 (October 1, 2006 – September 30, 2007) the University of California, Davis Center for Watershed Sciences and Watercourse Engineering Inc. conducted an integrated assessment of the physical and biological factors affecting salmonids in the Shasta River, within the Nelson Ranch, Siskiyou County, California. The focus of this effort was to document a “year-in-the-life” of Shasta River aquatic ecology and to identify factors that limit salmonids during different life stages. This yearlong, detailed assessment increases understanding of seasonal habitat variability for the various salmonid species present in the river and the usage of those habitats. The study documents habitat conditions during salmonid immigration, spawning, incubation in gravels, emergence from redds, rearing and over-summering, and out-migration. Using an interdisciplinary approach to study the Nelson Ranch, we have been able to determine several physical and ecological constraints that probably impact salmonids of the greater Shasta River. We have also identified a suite of potential future studies to address key uncertainties in management of salmonids in the basin.

The year-in-the-life study included a wide range of field investigations, mapping, laboratory investigations, and associated work. We collected physical habitat data, including habitat mapping, geomorphology, hydrology, meteorology, and water temperature and quality. Ecological data collected includes surveys of aquatic macrophytes, macroinvertebrate communities and food webs, and seasonal habitat utilization by fish. All data collected during this study are included and incorporated into a project database (included CD).

The key conclusions and observations of this study are:

- Current hydrologic conditions on the Nelson Ranch are significantly affected by upstream water resource development and operations, including the impoundment of Lake Shastina and Dwinnell Dam, Parks Creek diversions to meet Montague Water Conservation District demands, and upstream irrigation practices in lands adjacent to the Shasta River and Parks Creek. Operations of the Grenada Irrigation District (GID) diversion, located adjacent to the Nelson Ranch, has direct impacts on reach hydrology during irrigation season.

- The Shasta River exhibits hybridized characteristics of both “spring-dominated” and “rainfall/snowmelt runoff-dominated” rivers. Historically, the geomorphology of the upper river (above Big Springs Creek) reflected runoff-dominated flow conditions, while the lower river (below Nelson Ranch) reflected spring-dominated flow conditions. The Nelson Ranch represents a geomorphic and hydrologic transition zone between the upper and lower Shasta River.
- Channel planform morphologies, particularly downstream from the GID diversion, remain largely unchanged across both the pre and post-Dwinnell Dam periods. This suggests channel geometries are scaled to largely invariable spring-fed baseflows sourced in Big Springs Creek, a hydrologic condition which has remained relatively unchanged since the early 1900’s.
- The proximity of the Nelson Ranch to Big Springs Creek results in water temperature conditions that exhibit seasonal variability imposed on a spring-stream dominated thermal regime. Coupled with this unique thermal regime, are impacts associated with upstream water resources development and management. Specifically, during spring and summer months, impacts of land and water use activities, coinciding with the maximum annual thermal loading, create warm water conditions on the Nelson Ranch.
- Mean weekly maximum water temperatures on the Nelson Ranch were greater than 18°C (64.4°F) for 151 days between 1 April and 30 September along the Nelson Ranch (82.5 percent of the period), which are above thresholds considered suitable for juvenile coho salmon.
- Aquatic macrophytes have a significant impact on the hydrology and aquatic habitats of the Nelson Ranch. Increased bed roughness from aquatic macrophytes increases river stage relative to discharge throughout the summer, increasing the availability or access to shallow water habitat. Aquatic macrophytes also create and alter mid-channel habitats available to fish throughout the seasonal growth and senescence cycle.
- Natural abundance stable isotope and food web sampling shows that the Shasta River along the Nelson Ranch is very productive, and the food web contains complex trophic interactions that vary seasonally. For instance, we found that instream (autochthonous) production supported food web productivity throughout the year.
- Aquatic macroinvertebrate sampling showed that during spring a large number of the highly tolerant Dipteran family Chironomidae (non-biting midges) were present in the samples. Large numbers of the Chironomidae are generally indicative of nutrient rich (e.g., eutrophic) water quality, and increased water temperatures.
- Juvenile coho were observed utilizing relatively fast deep-water habitats where instream woody debris was present on the Nelson Ranch. By early June, water temperatures warmed, and very few juvenile coho were observed only in a backwater habitat. After 3 July 2007, no coho were observed on the Nelson Ranch.

- Juvenile steelhead were the most abundant salmonid observed during snorkel surveys conducted on the Nelson Ranch. Adult steelhead were observed along the Nelson Ranch in June, and appeared to be fresh from the ocean. This is evidence that summer run steelhead reside in the Shasta River. Steelhead have higher temperature tolerances than coho, and are thus able to utilize habitat on the Nelson Ranch throughout the summer.
- During October 2006, while cooperating with CDFG, we observed mature 0+ male Chinook in redds with adult female Chinook. This is the first time that mature male parr have been observed in the Shasta River. How mature parr may contribute to the population is unknown, but this life history strategy may help the population hedge bets against poor migratory conditions downstream.

The Shasta River on the Nelson Ranch is a highly productive system with significant potential for restoration of salmonid habitat. The unique hydrology and abundant aquatic macrophytes provide various habitats for fishes during all life stages. Currently the primary limiting factor to salmonids on the Nelson Ranch is elevated water temperature. The quality of spawning habitat is also low. If water temperatures along the Nelson Ranch can be reduced (e.g., through management actions), then the abundant habitat and high natural productivity could support relatively large populations of salmonids, including the federally- and state-listed coho salmon.

The baseline assessment presented herein identifies the first multidisciplinary approach to studies the Shasta Valley, providing a framework for an integrated evaluation of physical and biological factors affecting the various salmonid life stages present throughout an annual period. Although these findings provide novel and important information, continued research is necessary to advance our understanding of salmonids in the Shasta River Basin, and to more effectively identify and assess alternative management practices and restoration activities. Specifically, future studies are recommended to:

- Extend the spatial extent of assessment to Shasta River reaches upstream and downstream of the Nelson Ranch. A short term study addressing a portion of the annual period at selected sites has been funded by the U.S. Bureau of Reclamation, and is currently in progress.
- In addition to extending investigations spatially, multiple year assessment is necessary to capture aquatic system response to hydrologic and meteorological variability, as well as to capture the range of year-class characteristics associated with coho salmon.
- Quantify upstream thermal characteristics associated with groundwater influence (springs), channel morphology, travel time, meteorological conditions, and other factors to quantify heat transport through Shasta River reaches in the vicinity of the Nelson Ranch.
- Quantify temporal and spatial changes in biomass (e.g., standing stocks of organic matter, macroinvertebrates) and important ecological rates (e.g., input of allochthonous material, invert emergence and drift, secondary production) to improve our understanding of ecosystem-level processes.

- Characterize the stable isotope signatures of additional sources of organic matter that have been identified as potentially important contributors to carbon flow in the Shasta River (e.g., particulate organic matter, epiphytic biofilms).
- Generate comprehensive information on the fish community during each season, including coupling fish distribution and abundance with stable isotope and gut content analysis to determine trophic relationships and important food web interactions. Future trophic investigation should target habitat types that are ecologically relevant to fish at different life history stages.

Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California Water Year 2007

2. Introduction

The 1997 NMFS listing of the SONCC (Southern Oregon/Northern California Coast) evolutionary significant unit of coho salmon (*Oncorhynchus kisutch*) as threatened under the ESA has resulted in an increased focus on the ecological and physical systems in the Klamath River Basin and particularly within the Shasta River. Several Klamath basin-wide reports and conservation plans have highlighted the importance of the Shasta River in preserving and restoring anadromous salmonid populations within the greater Klamath River Basin (CDFG 2004, NRC 2004, NMFS 2007, NRC 2007). Despite being a restoration priority, very little information is available about the unique hydrologic and ecologic conditions that exist in the Shasta River. The 2005 acquisition of the 1,700 acre Nelson Ranch by The Nature Conservancy allowed site-specific research and the establishment of a long-term data set that will help direct future resource management throughout the Shasta River watershed. As one of the federal agencies in the Klamath Basin, the Bureau of Reclamation contracted with the UC Davis Center for Watershed Sciences and Watercourse Engineering, Inc. to determine limiting factors of salmonids during a year-long period within the Nelson Ranch section of the Shasta River. This report is the summary of hydrology, water temperature, geomorphology, aquatic macrophyte, food web, and fish habitat usage data collected by UC Davis Center for Watershed Science (Watershed Center) and Watercourse Engineering, Inc. (Watercourse) personnel on the Nelson Ranch during the 2007 water year (1 October 2006 – 30 September 2007).

Report Organization

The year-in-the-life study included a wide range of field investigations, mapping, laboratory investigations, and associated work. Report elements include a general site description, followed by chapters addressing hydrology/meteorology, water temperature, geomorphology, habitat mapping, aquatic macrophyte, macroinvertebrate and food web sampling, and fish surveys. Each chapter ends with major findings and future recommendations. References are included, as are appendices addressing field data.

3. Acknowledgements

We would like to acknowledge the following people who provided support to the year-in-the-life study. Specifically, we would like to acknowledge the assistance of Bill Chesney, Mark Pisano, and Mark Hampton of the California Department of Fish and Game for sharing their expertise on coho sampling, habitat assessment, and general life history in the Shasta River; Dave Webb of the Siskiyou County RCD for his insights on general conditions in the Shasta River Basin; Chris Babcock and Ada Fowler of The Nature Conservancy for their patience and support during the extended periods of field work; Henry Little, Mark Reynolds, George Stroud, and Amy Haas of The Nature Conservancy for being fully engaged in our work and providing access to this critical reach of river; the Peters family and Dan Chase for providing access to the opposite bank of the Shasta River throughout much of the Nelson Ranch property; Curtis Knight and Drew Braugh of CalTrout for organizational and logistical support; Dave Fontius for property access; Sue Maurer for help in fish survey design and survey validation. Finally, we want to extend our appreciation to the U.S. Bureau of Reclamation, Klamath Area Office for funding this project.

4. Site Description

The 1,704 acre Nelson Ranch lies within the central Shasta River Valley in Siskiyou County, California (Figure 1). The Shasta River flows approximately 60 miles northwestward from its headwaters to its confluence with the Klamath River, and is the fourth largest tributary in the Lower Klamath River system (Figure 1). Bounded by the Scott Mountains to the west, Siskiyou Mountains to the north, and the Cascade Volcanic Range to the south and east, the Shasta River drainage basin exhibits considerable spatial variability in geologic and hydrologic characteristics. Tributaries from the Scott and Siskiyou Mountains flow northeast to the Shasta River, roughly perpendicular to the northerly strike of the Eastern Klamath Belt, a geologic province comprised of a complex assemblage of Paleozoic sedimentary and metamorphic rocks and Mesozoic intrusives (Hotz 1977). Northerly and westerly flowing tributaries to the Shasta River drain both the northern slopes of Mount Shasta, and the western slopes of the Cascade Range, regions largely underlain by porous volcanic rocks of the Western and High Cascades geologic provinces. The Shasta River flows for most of its length along the floor of Shasta Valley, an area underlain principally by a complex assemblage of High Cascade Plio-Pleistocene andesitic and basaltic lava flows and volcanoclastic materials derived from a Late Pleistocene debris avalanche from ancestral Mount Shasta (Wagner 1987, Crandell 1989). Low-gradient basalt flows (e.g., Plutos Cave Basalts) dominate the eastern portions of Shasta Valley, while western regions exhibit a mosaic of andesitic and dacitic hillocks and depressions formed by the aforementioned debris avalanche.

The local climate is semi-arid with mean annual precipitation varying between 10 inches (25.4 cm) and 18 inches (45.7 cm) (Clawson et al. 1986, Vignola and Deas 2005), much of which falls as snow in higher elevations during the winter months. The Shasta River has one major dam, Lake Shastina (Dwinnell Reservoir) at river mile 40.6. Drainage area for the Shasta River above Dwinnell Dam is 279 km² and 1,638 km² (632 mi²) below the dam. Current mean annual impaired runoff is approximately 168,000,000 m³ (136,000 acre-feet (af)).

Lake Shastina was impounded in 1928. The 1923 water right allowed 74,000,000 m³ (60,000 af) to be stored from October to June, although maximum operating capacity is 61,700,000 m³ (50,000 af) (Booher et al. 1960). The reservoir experiences substantial seepage losses through underlying volcanoclastic rocks (Vignola and Deas 2005). Direct reservoir outflow includes minimal controlled releases of up to 10 cfs (0.28 cms) and relatively infrequent uncontrolled winter spill events (e.g., 1964 and 1997; pers. comm., D. Webb 2007) (Vignola and Deas 2005). Consequently, measured streamflow downstream of Dwinnell Dam is primarily driven by inflow from tributaries (e.g., Parks Creek), discrete natural springs (e.g., Big Springs), and diffuse groundwater.

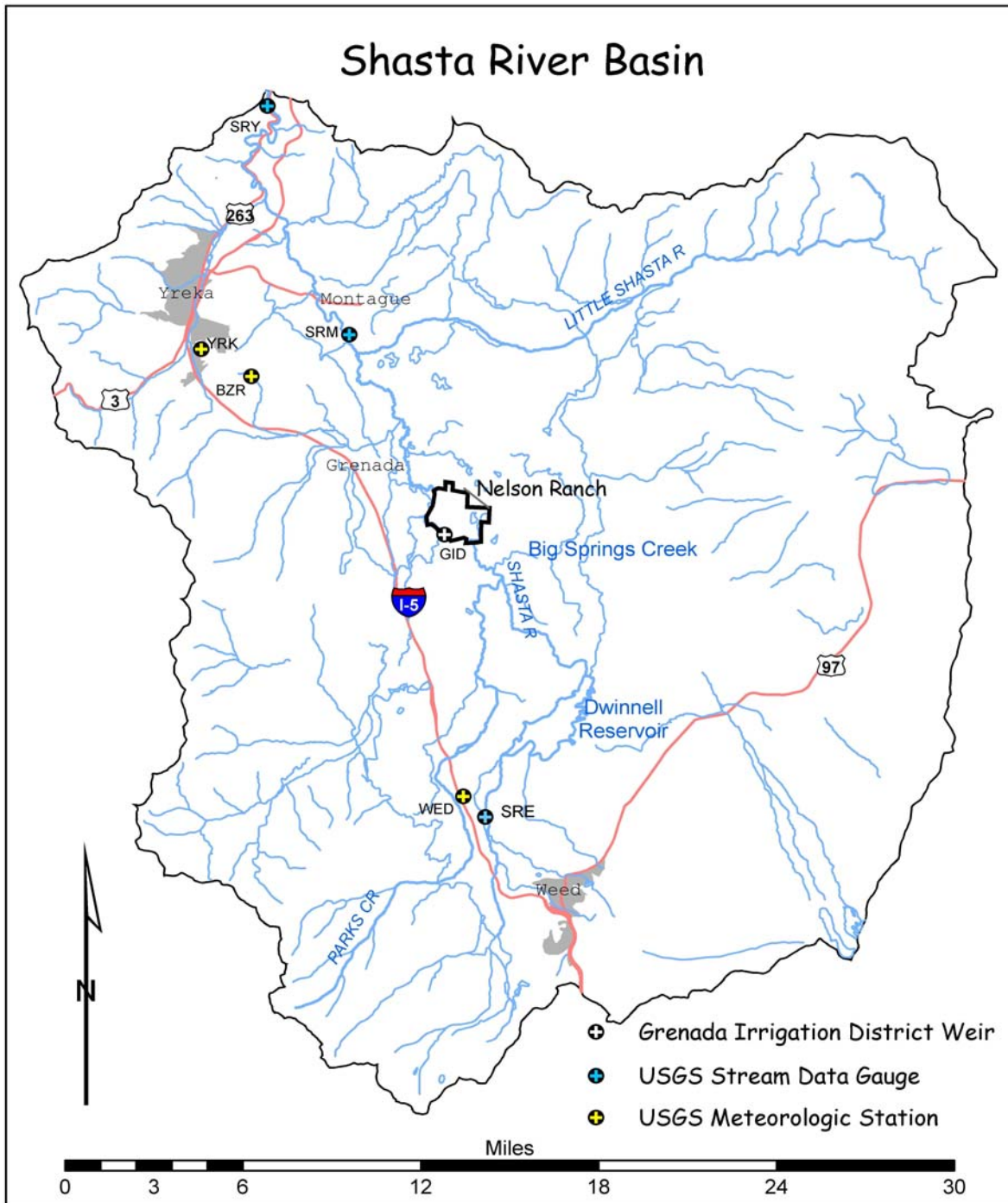


Figure 1. Shasta Valley hydrologic network and project site.

5. Hydrology

The Shasta River, like most California rivers, experiences seasonal minima flow in early summer and early fall in response to the Mediterranean climate that typifies the region. However, local spring inflows modify this typical seasonal hydrograph on the Nelson Ranch. The Nelson Ranch occupies a reach of the Shasta River between RM 32.0 and RM 27.5 (RKm 51.5 and RKm 44.3),

just below the confluence of Big Springs Creek. Big Springs (and other springs) impose a persistent baseflow on the typical seasonal hydrograph.

This reach has two diversions, Nelson Ranch irrigation of approximately 2 cfs (.05 cms), and a combined diversion of up to 52 cfs (1.47 cms) to supply up to 40 cfs (1.13 cms) to the Grenada Irrigation District (GID) and 12 cfs (0.34 cms) to the Huseman Ditch. The Nelson Ranch diversion is a screened pump, while the GID/Huseman Diversion includes a diversion dam and screened diversion facilities. Surface return flows from the Nelson Ranch averaged 0.14 cfs (0.004 cms) throughout the year, and peaked to 1.59 cfs (0.045 cms) during the 2007 irrigation season. Return flows from GID and Huseman Ditch have not been quantified.

To assess flow conditions in the Shasta River along the Nelson Ranch, two flow gauging stations were installed. The upstream boundary gauge (Shasta 1) was located at the upper end of the Nelson property in the vicinity of the California Department of Water Resource (DWR) gage (currently out of service). This location served to quantify flow entering the Nelson Ranch. Here, flow was primarily composed of discharge from the main-stem Shasta River below Dwinnell Dam, Parks Creek, and Big Springs Creek. A considerable volume of water was diverted during winter from Parks Creek into the Shasta River drainage above Lake Shastina for storage, which subsequently was diverted during spring and summer periods for use on Montague Water Conservation District (MWCD) lands with little return flow to the Shasta River. A second flow gauge at the downstream boundary of the Nelson Ranch (Shasta 4) captured Shasta River flow leaving the Nelson Ranch reach. Flow between the two gauging stations should differ by the quantity of water diverted to Nelson Ranch and GID, tailwater return flow, spring and seeps, and unquantified gains and losses (such as seepage, subsurface flow, evapotranspiration, and local precipitation and runoff).

Point velocity measurements were made with a Marsh McBirney Flomate electromagnetic flow meter mounted on a top set wading rod using the six-tenths-depth (0.6) method (Rantz 1982). Stream flow was calculated using the USGS mid-section velocity-area method (Rantz 1982). Stream stage measurements were collected using Druck 1830 series pressure transducers (resolution accuracy +/- 3 mm) connected on Campbell Scientific CR510 data loggers programmed to record water stage (m) at ten-minute intervals.

5.1. Stage-Discharge Rating Curves

Measured velocity and depth were used to calculate flow, which was then matched to corresponding river stage at the gauging station (Table 1). Throughout the study period such information was used to obtain stage-discharge relationships or rating curves at the two gaging locations. These rating curves were subsequently used to estimate flow during the 2007 water year (Figure 2).

Table 1. Computed discharge measurements and corresponding gage height at the two gauging stations located along the Nelson Ranch reach during water year 2007. Stage is height above the sensor at each location. Discharge measurements taken outside of water year 2007 are included in the appendix.

Date	Shasta 1: Upstream Nelson Ranch		Shasta 4: Downstream Nelson Ranch	
	Stage (m)	Discharge (cfs)	Stage (m)	Discharge (cfs)
10/2/2006	0.98	200.8	0.62	198.019
10/12/2006	0.97	209.4	0.66	212.1125
11/2/2006	0.97	205.2	0.64	208.101
11/9/2006	0.98	214.4	0.67	220.1355
12/7/2006	0.96	175.8	0.58	193.0135
12/29/2006	0.98	193.8	0.57	191.5225
1/4/2007	0.98	205.5	0.69	201.7465
1/11/2007	0.96	160.3	0.54	161.4185
1/18/2007	0.96	154.9	0.50	139.76705
1/25/2007	0.96	155.1	0.51	157.62
3/15/2007	0.97	160.1	n/a	n/a
3/30/2007	0.95	136.7	0.41	125.67
5/17/2007	0.91	136.3	n/a	n/a
5/24/2007	0.91	92.7	0.21	84.49
6/7/2007	0.94	112.5	n/a	n/a
9/13/2007	0.94	85.9	0.27	58.22
9/19/2007	0.93	85.6	0.26	60.35
9/26/2007	0.93	87.3	0.31	77.035

Notes:

- 1) The flow gage Shasta 1 is located at the upstream property boundary and Shasta 4 at the downstream boundary
- 2) Damage to data loggers yielded 28 missing days for Shasta 1 and 18 missing days for Shasta 4 during the 2007 water year.

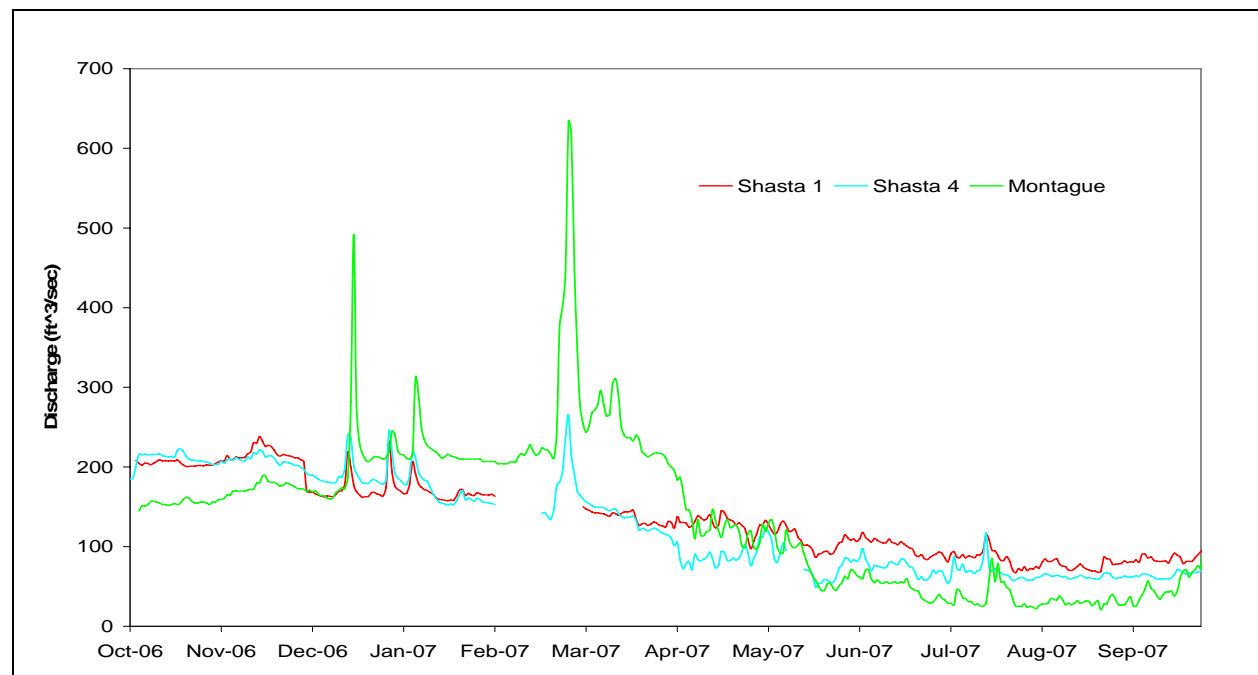


Figure 2. Calculated daily discharge during water year 2007 along the Nelson Ranch. USGS discharge data for Montague is included for comparison.

Variability in stage-discharge relationships occurred due to seasonal aquatic macrophyte growth. Submerged macrophyte growth can considerably reduce average flow velocities leading to increased stream depth and an increase in channel cross sectional area (Champion and Tanner 2000). Initial discharge measurements at the Nelson Ranch indicated that river discharge can vary by as much as 100 cfs at the same river stage during different times of the year due to seasonal aquatic macrophyte growth. To accommodate these variable conditions, stream flow records were segregated into discrete temporal periods defined by unique stage-discharge relationships: four periods for Shasta 1 and three periods Shasta 4 (Table 2). Temporal delineations for rating curves were determined by quantitative and qualitative estimates of aquatic macrophyte cover, assuming that stage-discharge relationships would be similar during times of similar vegetative cover. The various rating curves were then developed using discharge measurements collected during times of similar vegetative cover. Collected stage data was condensed into daily average stage values and stage-discharge equations were applied to create the daily discharge shown in Figure 2.

Table 2. Power functions for Shasta 1 and Shasta 4 used to estimate discharge where (x) is river stage relative to local datum.

Date	Shasta 1	Date	Shasta 4
10/1/06 - 11/30/06	$Q = 244.82(x)^{5.7545}$	10/1/06 - 2/28/07	$Q = 345.83(x)^{1.19}$
12/1/07 - 2/2/07	$Q = 201.89(x)^{5.2538}$	3/1/07 - 7/19/07	$Q = 301.12(x)^{1.0327}$
3/4/07 - 7/20/07	$Q = 167.43(x)^{5.7816}$	7/20/07 - 9/30/07	$Q = 104.81(x)^{0.3967}$
7/21/07 - 9/30/07	$Q = 133.23(x)^{6.6086}$		
The number of discrete periods for Shasta 1 and Shasta 2 differed in response to variable vegetative cover between the two sites			

5.2. Precipitation

Precipitation totals for the 2007 water year were below average. Average annual precipitation for the Shasta Valley varies from 10 inches (25.4 cm) to 18 inches (45.7 cm) (Clawson et al. 1986, Vignola and Deas 2005), and precipitation on the Nelson Ranch for the 2007 water year was measured at 11.14 inches (28.3 cm). Precipitation was measured at both stream gauging locations at the property boundaries of Nelson Ranch. Nelson Ranch precipitation totals equaled 11.14 in (28.3 cm) at the top of the property (Met 1) and 8.9 inches (22.6 cm) at the bottom of the property (Met 2) (Figure 3). Discrepancies in precipitation in January are due to interference by birds of the Met 2 precipitation gauge. Sixty-five percent of precipitation that fell along the study reach occurred between October and March, while thirty-four percent fell between April and August. This annual distribution of precipitation resembles large-scale weather events during the winter and spring months, and smaller more frequent convection based events taking place in summer months.

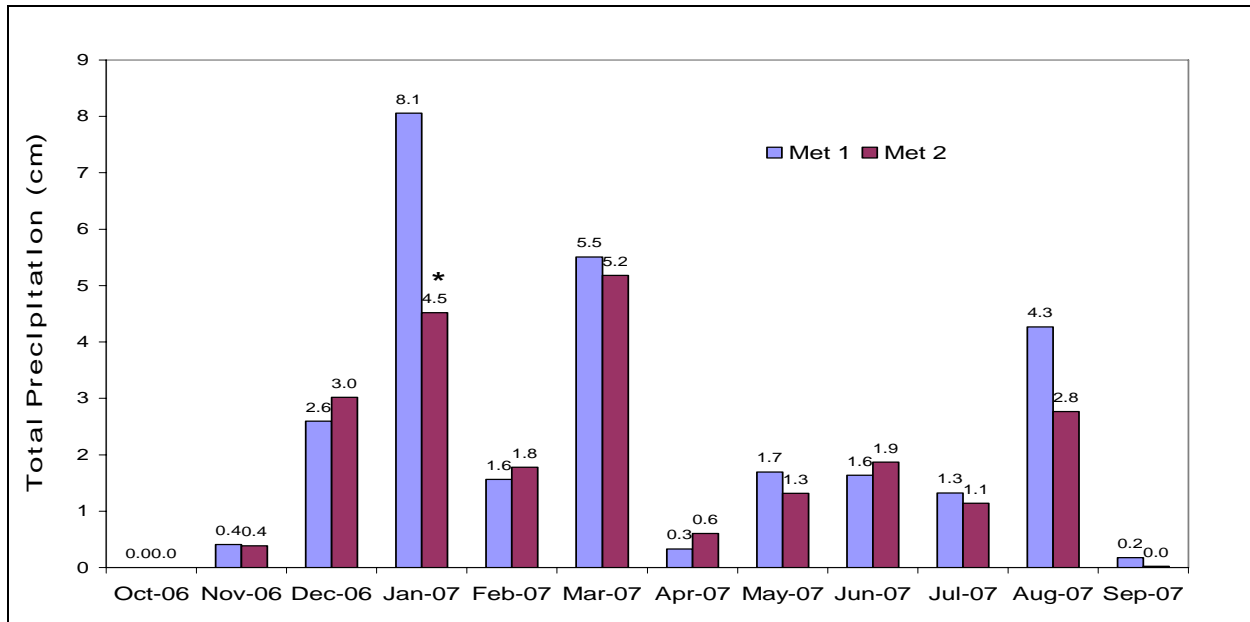


Figure 3. Monthly precipitation totals for the Shasta River along the Nelson Ranch. * indicates interference by birds with the precipitation gauge.

Most California rivers experience annual flow conditions in response to the Mediterranean climate that typifies the region. Winter storms bring precipitation as rain and snow which leads to the high flow conditions. During spring and into summer river stage and flow follow a typical descending hydrograph with minimum flows in late summer or fall. Year-in-the-life flow conditions on the Nelson Ranch are presented below.

Fall: October – November

Fall flow conditions for the Shasta River on the Nelson Ranch respond not only to season, but also to the termination of irrigation practices. Specifically, during the first week of October, stage increased markedly as irrigation season concluded, increasing depth and widening the river where it was shallow and inundating portions of benches and point bars that were exposed through the summer. In deeper sections, where the river channel is typically rectangular in form, there was little change in width. Some of the most stable flow regimes of the year occurred in October and November, at approximately 200 cfs, following irrigation season and prior to winter storms (Figure 2). This stability is a byproduct of inflow from springs upstream, and provides a valuable resiliency to baseflow in the Shasta River. This condition is absent in other tributaries of the Shasta River such as Parks Creek, which exhibits a hydrograph more typical of a precipitation and low elevation snowmelt stream.

Winter: December – March

Winter brings precipitation, typically as snow in the higher elevations and rain in the lower elevations. During winter 2007, baseflow remained near 200 cfs (5.66 cms) until mid-March, when flow dropped below 100 cfs (2.83 cms). Diversions from Parks Creek and upper Shasta River flows are stored in Lake Shastina during this period (and extending into spring), leaving modest flows and few peak storm events downstream at the Nelson Ranch. During winter, the Shasta River in the Nelson Ranch reach exhibited only modest increases in flow, largely derived

from local storm runoff. Appreciable overbank flow did not occur during the winter of 2007, with the exception infrequent inundation of low bars and benches

Lake Shastina did not spill during the winter of 2006-07, and on average spills approximately 1 out of every 10 years (B. Crabill, pers. comm. 4/07). In wetter years Parks Creek diversion to the reservoir may be significantly curtailed to avoid or minimize spill at Dwinnell Dam. The result during these years is that considerably larger flows pass through the Nelson Ranch. A longer dataset will help quantify flow differences at Nelson Ranch between dry and wet years and we recommend that gauges remain installed and continuously rated on the Nelson Ranch in future years.

Spring: April – June

Irrigation season generally begins on 1 April in the Shasta River, but certain lands have water rights that start on 1 March. When irrigation diversions from the Shasta River, tributaries, and springs commenced in 2007, flows along the Nelson Ranch were significantly reduced (Figure 2). Because the GID/Huseman Ditch diversion is located in the Nelson Ranch reach of the Shasta River, flows at any particular time often differed between the upstream and downstream gauges on Nelson Ranch. After irrigation season began, low flow conditions persisted below the GID diversion throughout the spring and summer period. Benches and point bars that had been inundated during winter were exposed by April.

Summer: July – September

Natural tributary runoff diminishes through spring and into summer. Coupled with continued irrigation demands through September, low flows persisted through summer. Small fluctuations in discharge, due to timing of various irrigation withdrawals and, on occasion, localized summer thunderstorm events were evident in the hydrograph. Overall, summer flows are restricted to the low flow channel.

5.3. Summary

The Shasta River experiences a complex hydrology. The precipitation patterns are typical of California's Mediterranean climate. However, the combined precipitation-snowmelt hydrology is modified by strong groundwater influences and water resources development. Impoundment and water management associated with operations of Lake Shastina alter the flow regime and reduce peak winter flows in most years. Groundwater influences, in the form of large spring complexes upstream of the Nelson Ranch, provide critical baseflow throughout the study reach, even during periods of stream flow diversion associated with irrigation practices. Diversions to meet demands of the GID and Huseman Ditch water rights impose a signal on the river that is apparent through much of the lower river.

6. Water Temperature

The impact of high summer water temperatures in the Shasta River reducing cold-water fish habitat and limiting fish survival has been well documented (DWR 1986, 2001; USFWS 1992; DFG 1996; NRC 2004; NCRWQCB 2006). The year-in-the-life assessment increased the understanding of spatial and temporal thermal variability throughout each season, and clearly defined small-scale thermal conditions and variability not evident in previous studies where monitoring locations were more widely distributed (Watercourse 2003b). The temperature

monitoring efforts on the Nelson Ranch illustrate clear seasonal changes, local cool and warm water regions, and overall spatial and temporal variability throughout the reach.

Outlined herein is a general description of Shasta River water temperature conditions, including discussion of thermal attributes of water entering Nelson Ranch, as well as effects of meteorological influences, spring inflow contributions, and detailed observations along the Nelson Ranch reach. Water temperature conditions over an annual period are presented under pre-water development conditions for all seasons, followed by a seasonal description of current thermal conditions and discussion of associated thermal habitat considerations. Comparisons are made between pre-development and current conditions, and changes explained or hypothesized.

6.1. General Water Temperature Conditions of Nelson Ranch

Temperature conditions in the Shasta River on the Nelson Ranch are largely driven by hydrology (and geohydrology) and meteorology. Unique attributes of the system are the temperature signals from substantial spring inflows, which may enter the river either notably warmer, nearly the same, or considerably cooler than ambient water temperatures depending on the time of year. These spring inputs create unique thermal conditions when compared to streams without springs.

In general, groundwater-dominated river systems, like the Shasta River, have a more stable flow and thermal regime than those not dominated by groundwater (Sear et al. 1999). Groundwater dominated systems can moderate the influence of meteorological conditions by direct dilution of stable inflow temperatures, as well as increasing the volume of the receiving water. The result is less seasonal variability (Caissie 2006). Big Springs Creek contributes the majority of spring-derived water, although smaller springs occur upstream of the Big Springs complex, some of which appear to be associated with the construction and operation of Lake Shastina (Crabill, pers. comm., 4/07). Thus, Shasta River water temperatures and flows are relatively stable in the reach immediately below Big Springs; however, meteorological conditions exert an increasing influence as distance from the Big Springs source increases.

Aside from spring inflows, meteorological conditions are a primary factor driving thermal conditions in the Shasta River. Water temperature response to solar radiation varies seasonally with maximum loading occurring during late spring and summer months when day length is long, solar altitude is at an annual maximum, and cloudy days are at a minimum. Air temperature reflects a similar response to seasonal solar radiation. During water year 2007, maximum air temperature was 39.7°C (103°F) on July 10, 2007, and minimum air temperature was -13.5°C (7.7°F) on January 13, 2007. Daily average air temperature typically exceed 25°C (77°F) in July and August (Figure 4).

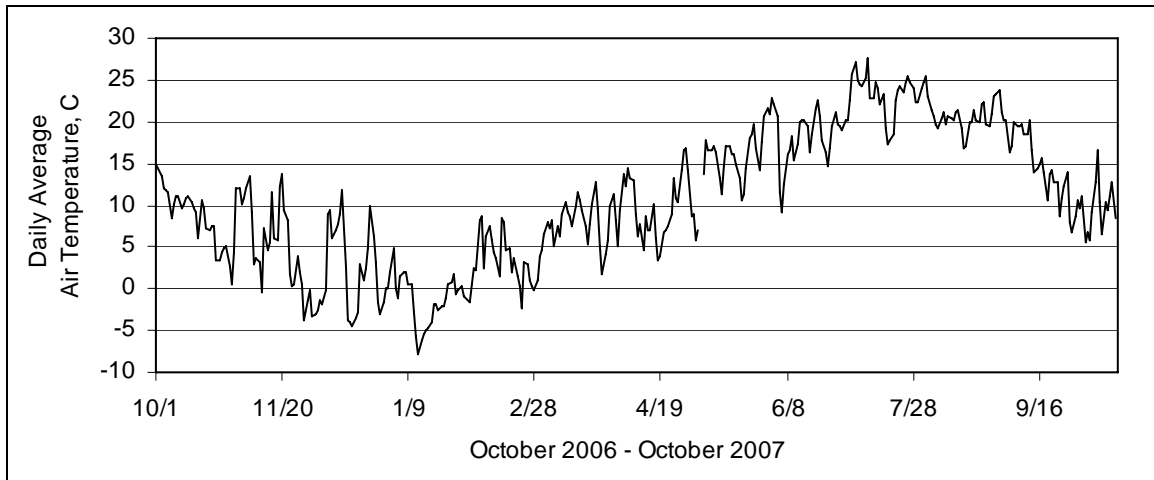


Figure 4. Nelson Ranch daily air temperature.

6.2. Detailed Observations on the Nelson Ranch

Exploratory temperature probing and lateral river transects were conducted during summer 2006 to improve understanding of small-scale thermal diversity in the Shasta River and identify possible cool-water habitat from small springs, subsurface flows, or seeps. Six sites were sampled for thermal diversity on upper Nelson Ranch on 22 August 2006 and 23 August 2006. Further, cross sectional water temperature was measured along three transects near the Nelson Ranch return flow ditch on 8 August 2006 (Figure 5) (Null 2007). Finally, winter monitoring of lateral diversity was conducted with multiple habitat cross sections and in side channels in the winter of in 2006-07.

These temperature observations throughout the year identified several key insights into smaller scale thermal conditions on the Nelson Ranch. Longitudinal thermal diversity exists on the Ranch, primarily in response to upstream conditions (both natural and anthropogenic) and meteorological influences on the river as it travels downstream. Longitudinal variability is presented in detail below, under the year-in-the-life discussion. Lateral variability was apparent near the river edge, and was most pronounced during summer. Riparian vegetation, both herbaceous and woody, may provide benefits for such margin habitat. During summer, small, localized cool water refugia associated with subsurface flow, seeps, and/or springs were identified on the Nelson Ranch with temperatures up to 1-2°C (1.8-3.6°F) cooler than mainstem river conditions. Irrigation return flows also occur on the Nelson Ranch. However, both cool refugia and return flows were generally small in size and/or magnitude and did not appear to have an appreciable influence on overall mainstem temperatures. Characterizing winter lateral variability identified that although mainstem temperatures were largely uniform, side channels differed dramatically. Side channels that were typically inundated under higher flows of winter and spring were generally frozen top to bottom during winter months. Further, soon after the ice melted, irrigation diversions reduced flows and stage, leaving the channels disconnected from the river and/or dry. Additional observations of potential refugia, springs and seeps, and return flows, would increase the information available and improve understanding of potential variability and impact these features have on anadromous fish production and associated management strategies on the Nelson Ranch as well as in other reaches.

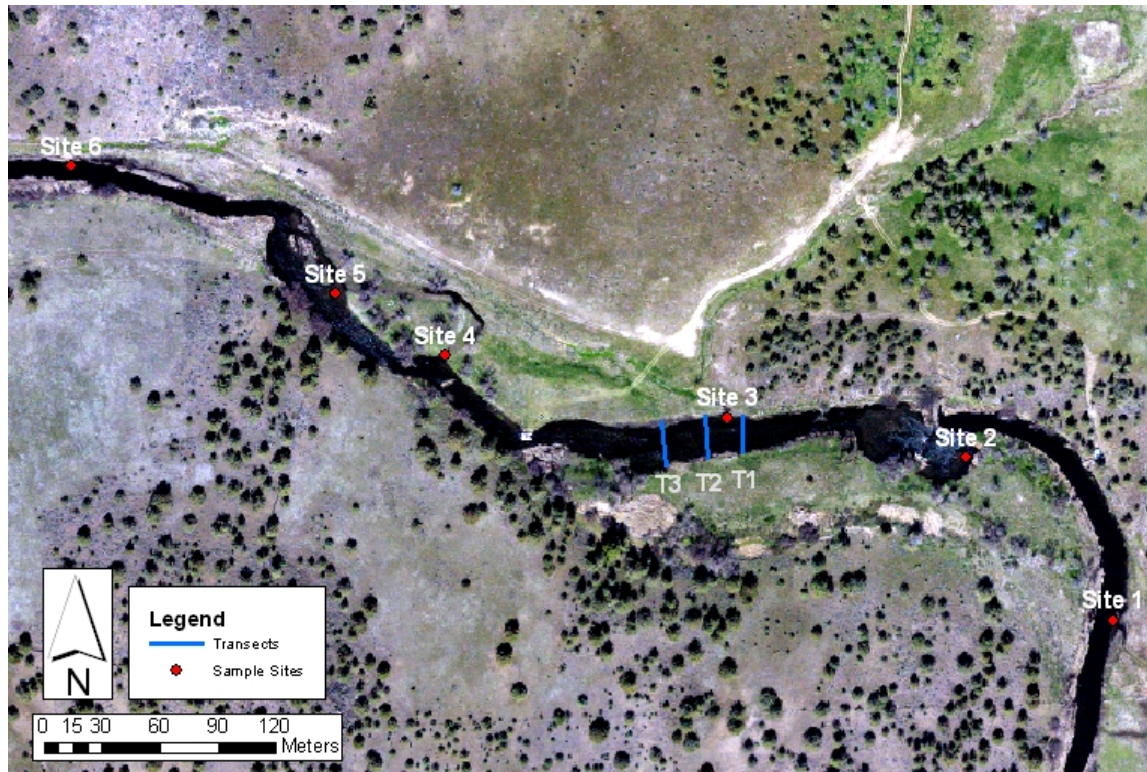


Figure 5. Exploratory temperature probing locations, August, 2006.

6.3. Pre-water Development Temperature Conditions

Pre-development conditions represent an estimate of the historic thermal regime of the Shasta River prior to groundwater pumping, construction of Dwinnell Dam, stream impoundments, diversions, and land use modifications. Historic Watermaster Service records and estimated quantification of the Big Springs complex were used to create an unimpaired hydrology of the Shasta River (DWR Watermaster service records 1930-1990; Deas 2006; Null 2007). Pre-development temperature conditions were estimated using historic flow data (NCRWQCB 2006), measured water temperatures throughout the Shasta River system, and equilibrium temperature theory (Martin and McCutcheon 1999) based on 2001 meteorological conditions. To provide a more comprehensive representation of water temperatures through space, the Tennessee Valley Authority's River Modeling System (TVA-RMS v.4) was used to simulate flow and water temperature under pre-water development conditions (Hauser and Schohl 2002; Null 2007). Estimated pre-development hydrologic and thermal conditions of the Shasta River are discussed in detail by season in the following paragraphs. Water temperature is necessarily linked to hydrologic conditions, thus, additional flow considerations are included herein with regards to assumptions regarding the unimpaired hydrology.

Fall: October – November

Fall flow regime under pre-development conditions was most likely similar to current conditions after irrigation season and before the onset of winter rains. The influence of spring inflow on local water temperatures were modest during this period of fall cooling and shorter day length because equilibrium temperature in the river approached that of the springs. Although there would be day-to-day variations in response to local meteorological conditions, water

temperatures would probably have ranged from 7°C to 15°C (44°F to 59°F), effectively bracketing local spring temperatures of 10°C to 12°C (50°F to 53°F) (NCRWQCB 2006). Warmer water temperatures were expected in early October and cooler water temperatures in late November. Modeling studies of pre-development conditions support the finding that inflow from springs would have a minor affect on water temperature during this period (Figure 6). Shasta River baseflow was approximately 200-300 cfs (5.66 - 8.5 cms), providing sufficient migratory conditions for returning Chinook and coho salmon. During late fall herbaceous riparian vegetation would have been modest and woody riparian species, if present, were most likely leafless, making riparian vegetation largely ineffective; however, water temperature was probably not limiting during this period.

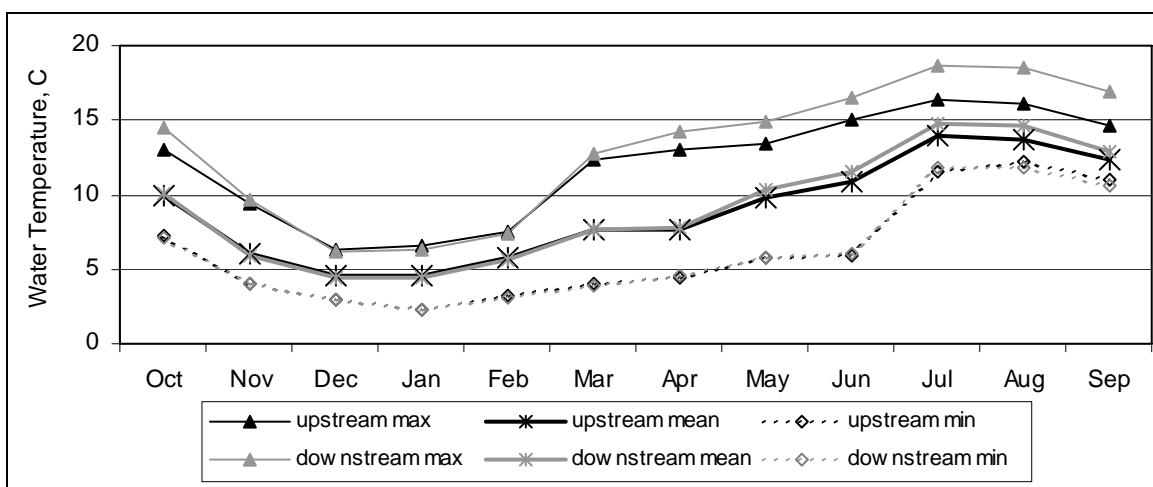


Figure 6. Simulated unimpaired max, mean, and min water temperature at Nelson Ranch upstream and downstream property boundaries

Winter: December – March

Historic data suggests winter baseflow was greater than 300 cfs (8.5 cms) (DWR Watermaster service records 1930-1990), and flows greater than 500 cfs (14.2 cms) probably occurred following storms. This larger and more consistent baseflow, coupled with more frequent and larger storm events, would lead to an increased the incidence of overbank flow onto the floodplain. Most likely the Shasta River on the Nelson Ranch was relatively warm during winter periods due to the influence of upstream springs (head of Big Springs is approximately 11°C (51.8°F) (NCRWQCB 2005)). However, during cold winter storms, precipitation events may have overwhelmed baseflow temperatures and the river may have exhibited notable cooling.

Modeling suggests winter temperatures were typically greater than 5°C, although this may be a low estimate (Null 2007) (Figure 6). The thermal regime of storm events under pre-development conditions was not assessed. Riparian vegetation was largely dormant during winter, providing only minimal stem shading.

Spring: April – June

Elevated baseflow and peak flows in response to winter storms and snowmelt events from the Parks Creek and upper Shasta River watersheds would have continued into spring for typical precipitation years under a pre-development condition. Flows would have remained elevated into late spring on the Nelson Ranch, inundating floodplain and side channel areas, creating a diverse range of thermal conditions and habitats.

Similar to the fall periods, spring inflow probably had a modest affect on water temperature because equilibrium temperature was close to the temperature of the springs through mid-May. However, later in May and into June, the importance of the spring (e.g., Big Springs) inflows and cool temperatures would have been vital to providing cues to young salmon that oversummering habitat would be available. Riparian vegetation leaf out would occur in mid-spring, and coupled with the thermal mass from a baseflow of approximately 200 cfs (5.66 cms) would have maintained cool temperatures (maximum daily temperatures less than approximately 16°C (60.8°F)) throughout the Shasta River on the Nelson Ranch.

Summer: July – September

Under unimpaired conditions, instream flow probably remained above approximately 150 cfs (4.25 cms) through summer on the Nelson Ranch. Because of the proximity of the Nelson Ranch site to Big Springs Creek, upstream conditions would have played a critical role in local temperature. During summer cool spring flows would be maintained well downstream in the Shasta River in response to the higher baseflow, reduced transit time, and potential riparian shading. Modeling results suggest that cool water conditions would have extended well beyond the downstream boundary of the Nelson Ranch, with daily maximum water temperature near or below 19°C (66.2 °F) during July and August, and nightly low temperatures around 12°C (53.6°F) (Figure 6) (Null 2007). Such conditions would provide a rich longitudinal and lateral thermal diversity on the Nelson Ranch.

6.4. Current Water Temperature Conditions

This section discusses seasonal changes to the thermal regime of the Shasta River under current conditions. Flow is discussed where it relates to temperature conditions and instream habitat. As noted, water year 2007 was a below normal year type based on rainfall and may not be representative of normal or wetter years. Temperature and flow monitoring was conducted at Nelson Ranch throughout the study period; however, due to instrument failure and loss, data from May 2006 to May 2007 was used to interpret thermal conditions and associated impacts on instream habitat (detailed included in Null and Deas 2007). Long term monitoring would lend further insight into the variability of the thermal regime in the Shasta River over variable year types.

Fall: October - November

During fall, the diurnal range of water temperature and response to meteorological conditions was generally similar throughout the Nelson Ranch (Figure 7). As day length shortened and solar altitude diminished, overall water temperatures declined and diurnal range diminished. Mean daily temperature in the fall differs little between upstream and downstream boundaries (Figure 8). The influences of spring inflow on local water temperatures were modest during this period because equilibrium temperature in the river approached that of the springs. Although the minimum and maximum daily temperatures are not notably different in October and November

(Figure 9), the timing of maximum daily water temperature was notably different at the upstream and downstream boundaries of the property. These differences are revealed by examining diurnal temperature variations over a 24-hour period using a series of monthly box and whisker plots. Maximum water temperature the upstream location occurred between 5:00 and 6:00 pm (Figure 10). At the lower property boundary, the daily maximum water temperature occurred between 10:00 pm and 1:00 am (Figure 11). The dominant thermal influence for the Shasta River is daytime solar radiation and advection of thermal energy from upstream sources. This implies that a volume of warm water, originating upstream of Nelson Ranch, was being transported downstream and was reaching Nelson Ranch's downstream boundary at night. Minimum daily water temperatures were similar at the upstream and downstream property boundaries, occurring between 7:00 – 9:00 am, and 8:00 – 11:00 am, respectively.

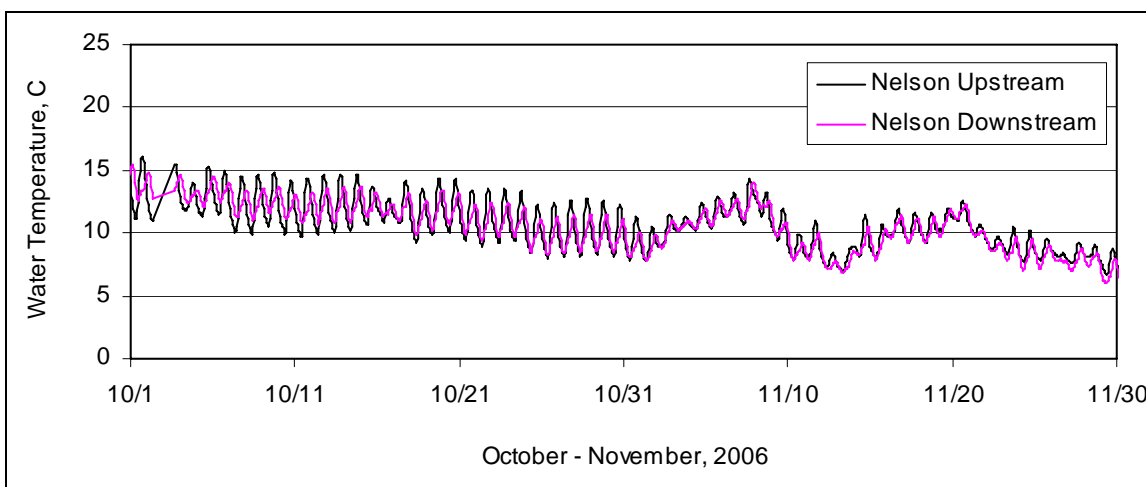


Figure 7. Fall hourly water temperature at Nelson Ranch property boundaries.

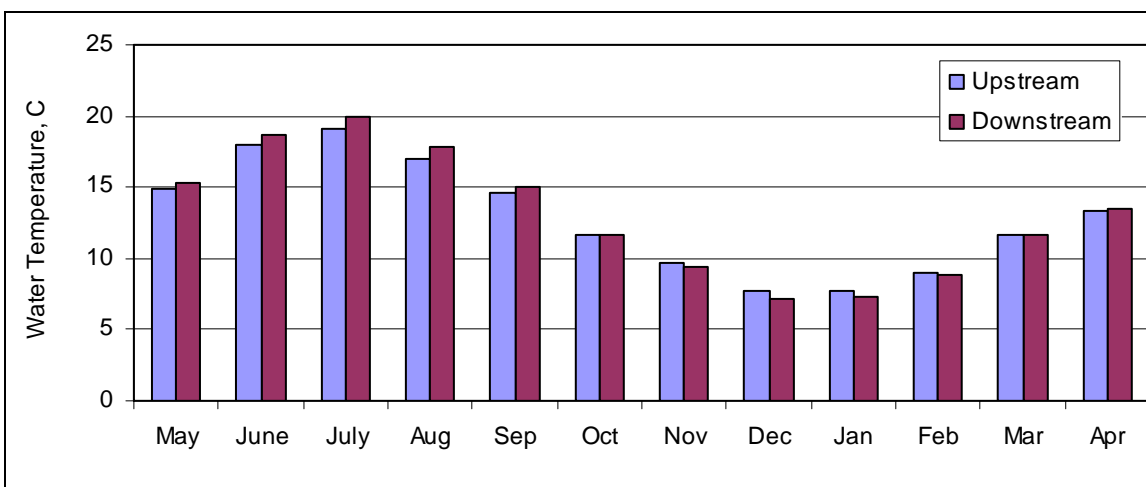


Figure 8. Mean monthly water temperature at Nelson Ranch property boundaries.

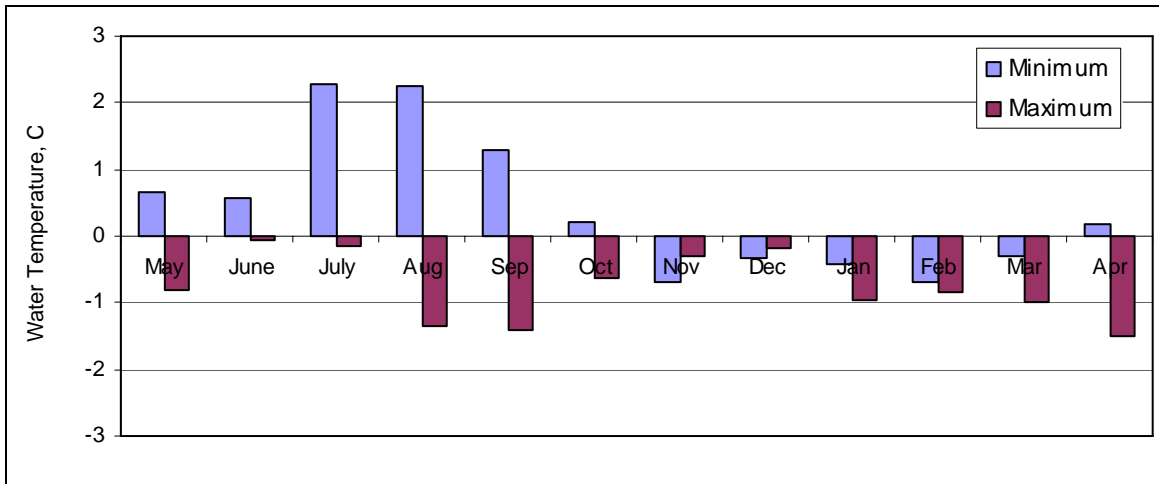


Figure 9. Difference in minimum daily and maximum daily temperature at Nelson Ranch property boundaries (downstream minus upstream, i.e., positive is warmer downstream and negative is warmer upstream).

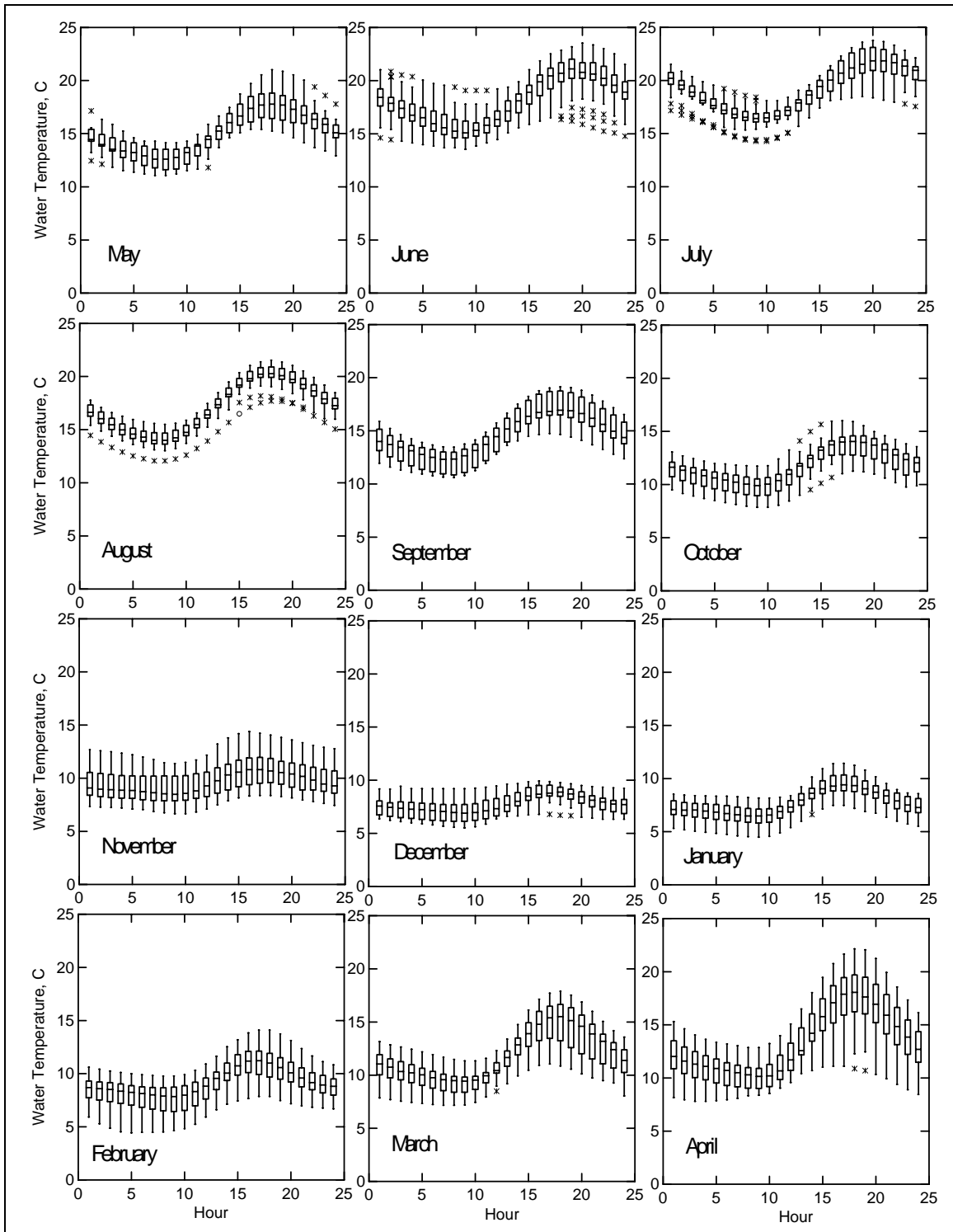


Figure 10. Hourly water temperature variability by month at Nelson Ranch upstream boundary.

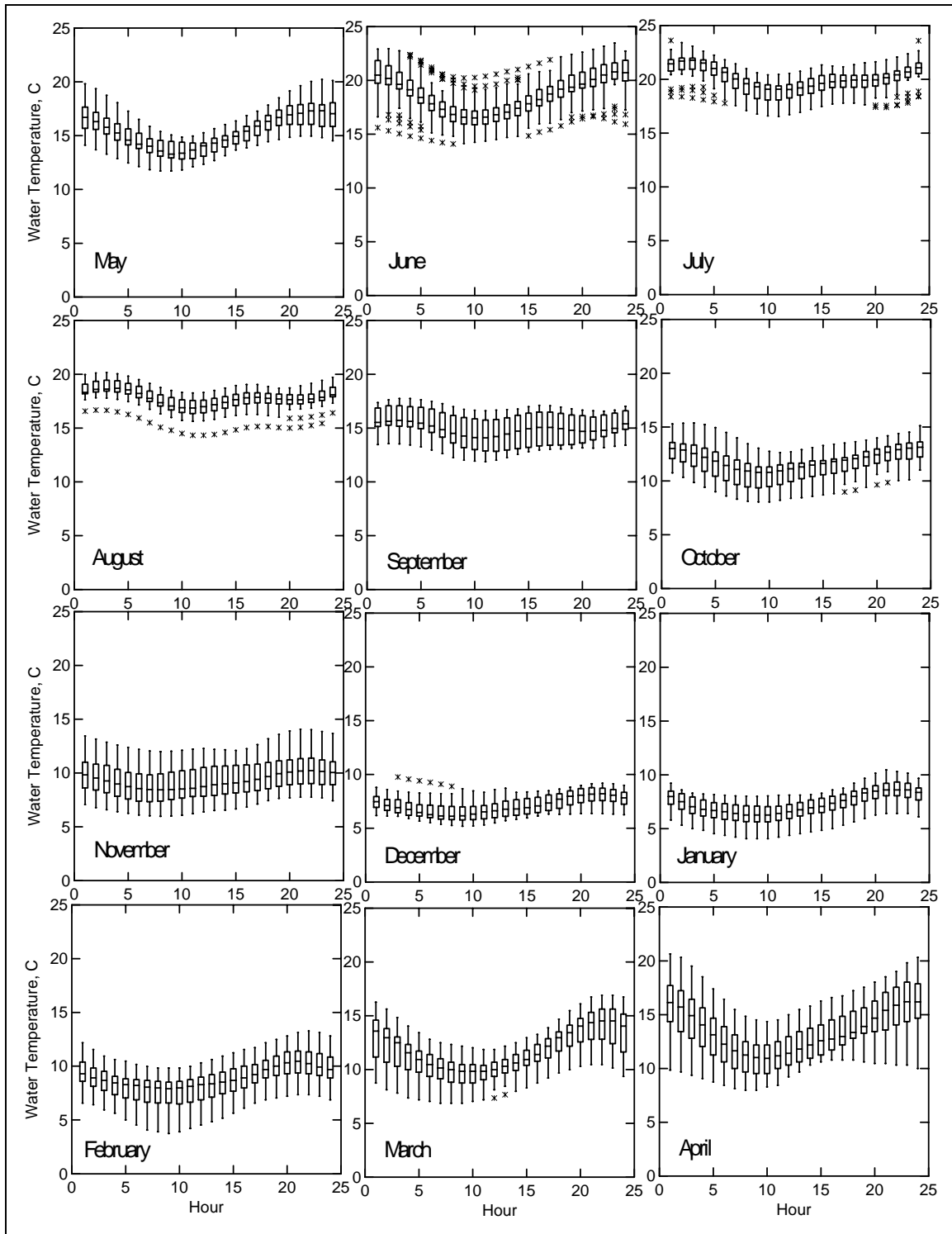


Figure 11. Hourly water temperature variability by month at Nelson Ranch downstream boundary.

Winter: December – March

During December and January, mean water temperature reached an annual minimum of approximately 5 to 7°C (41 to 44.6 °F) (Figure 12). Mean water temperature was approximately 0.5°C (0.9°F) warmer at the upstream property boundary than the downstream boundary (Figure 8). Warmer waters at the upstream boundary would be expected because of the proximity to Big Springs and subsequent cooling with distance from this warm water source, i.e., winter equilibrium temperatures are notable cooler than the springs. Diurnal range was similar at the two locations during all winter months, and increased throughout Nelson Ranch in late winter with increasing day length and solar altitude (Figure 12). Minimum and maximum temperatures were warmer at the upstream boundary than at the downstream boundary, by up to approximately 1.0°C (1.8°F) (Figure 9). Like fall, daily maximum water temperature occurred several hours later at the downstream boundary than the upstream boundary, near 10:00 pm, presumably in response to inherited upstream thermal conditions.

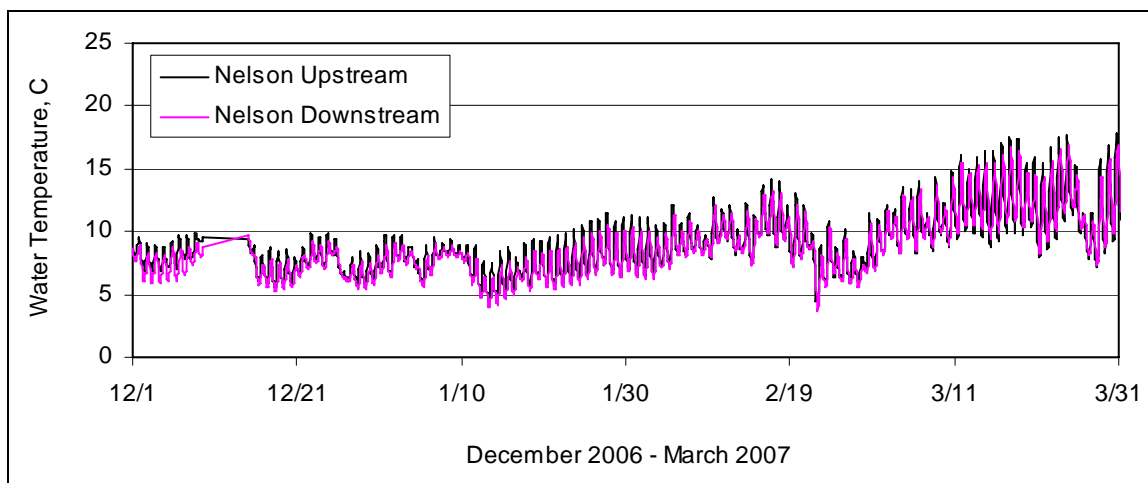


Figure 12. Winter hourly water temperature at Nelson Ranch property boundaries.

Spring: April – June

As with fall, meteorological conditions of late March and April result in water temperatures similar to the Big Springs and there is little net heat change through the Nelson Ranch reach. However, during May through June, atmospheric loading increases considerably and, in general, the system experiences net heating in the Nelson Ranch with mean water temperatures approximately 0.5°C (0.9°F) to 1.0°C (1.8°F) warmer at the bottom than the top (Figure 8). By mid- to late-spring a diurnal range of up to 5°C (9°F) was imposed on the system (Figure 13). Minimum and maximum temperature dynamics changed considerably, with minimum temperatures remaining slightly lower and maximum temperatures becoming notably higher at the upstream boundary (Figure 9). The increase in maximum daily temperatures is assumed to be related to upstream land and water use. Like fall and winter, maximum daily water temperature occurred at 10:00 – 11:00 pm at the lower property boundary, approximately four hours later than the upstream boundary (Figure 10, Figure 11).

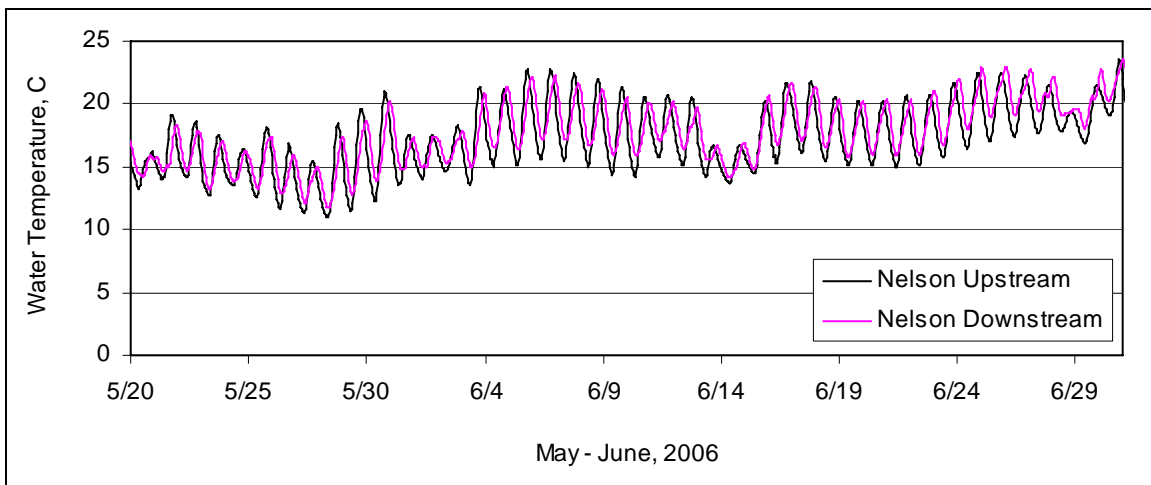


Figure 13. Spring hourly water temperature at Nelson Ranch property boundaries.

Summer: July – September

During summer, mean water temperature reached an annual maximum of 23.8°C (74.8°F) at the upstream property boundary and 23.6°C (74.5°F) at the lower property boundary (July) (Figure 14). Mean water temperature was approximately 0.5°C (0.9°F) to 1.5°C (1.3°F) warmer at the downstream property boundary than the upstream boundary (Figure 8), suggesting net heating. The springs remain approximately constant in temperature, so the summer condition presents the opposite effect of winter. Warmer waters at the downstream boundary would be expected because of the proximity to Big Springs and subsequent heating with distance from this cold water source, i.e., summer equilibrium temperatures are notably warmer than the springs. From mid-July through September, the diurnal range was greater at the upstream boundary than the downstream boundary (Figure 14). Minimum and maximum temperature dynamics were consistent with spring, but minimum temperatures were notably lower in August and September (over 1.0°C (1.8°F)) upstream and maximum temperatures were notably higher in the July through September period (over 2.0°C (3.6°F)) (Figure 14). Similar to other seasons, daily maximum water temperature occurred several hours later at the downstream boundary than the upstream boundary. Daily maxima and minima timing and magnitude differed remarkably at the property boundaries. Maximum temperatures at the downstream property boundary occurred up to eight hours after the upstream boundary maxima. Generally, water temperature peaked around 6:00 pm at the upper boundary, but not until 3:00 am at the lower boundary (Figure 10, Figure 11). Likewise, minimum daily water temperature occurred near 7:00 am and 11:00 am at the upstream and downstream boundaries, respectively (Figure 15). The increased travel time between property boundaries during summer was probably related to low flow conditions, increased channel roughness from macrophyte growth, as well as thermal signals advected into this reach from upstream conditions.

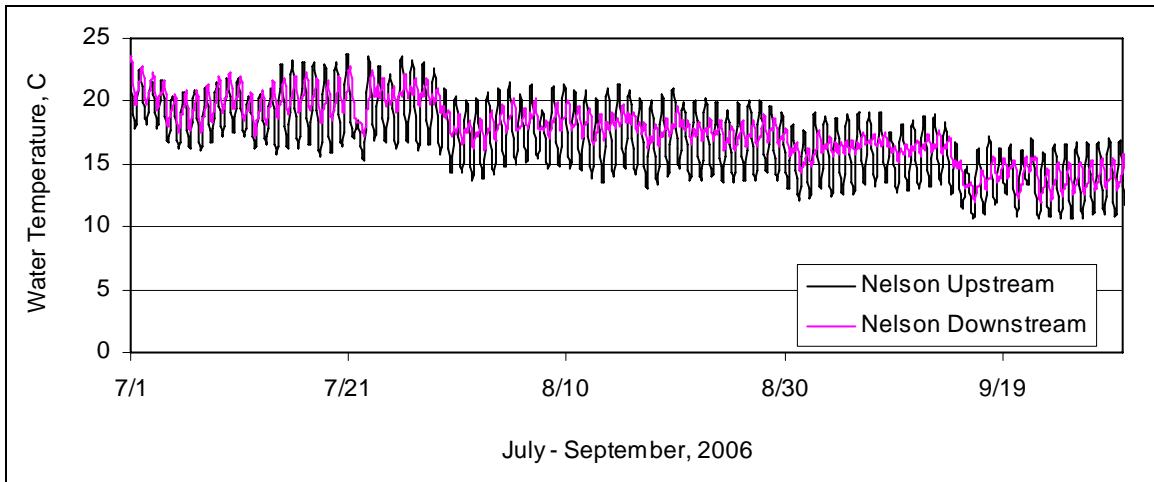


Figure 14. Summer hourly water temperature at Nelson Ranch property boundaries.

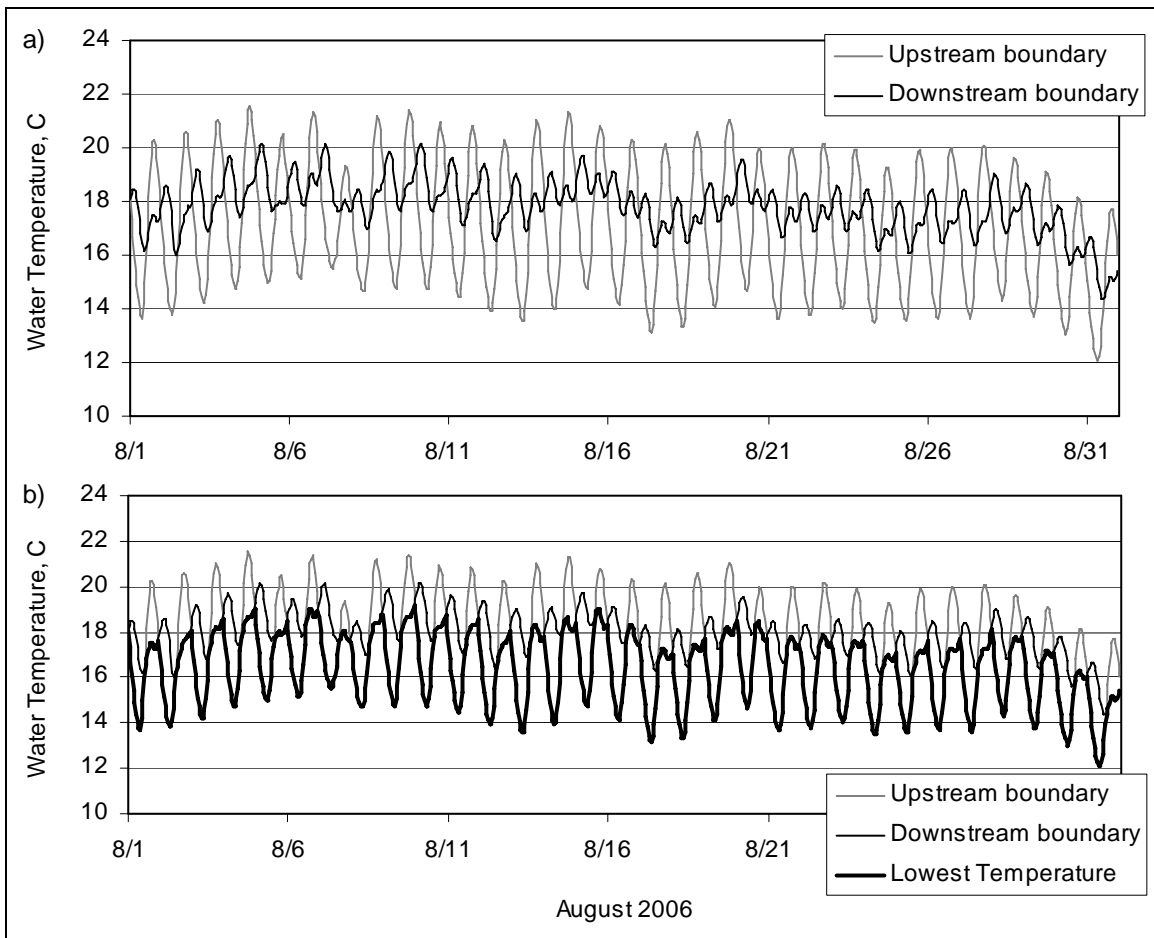


Figure 15. August water temperature at Nelson Ranch property boundaries (a); boundary temperatures with lowest temperature signal (b).

Land and water use developments along Big Springs Creek and possibly Parks Creek and the upper Shasta River contributed to this spatial and temporal thermal variability. Of particular interest are modifications to the Big Springs complex where an impoundment, diversions, return flow, and degradation of the Big Springs Creek channel have created conditions that led to appreciable thermal loading between the source water for the creek and the Shasta River.

During summer, small temperature differences occurred along the channel, such as localized areas of cool water refugia that were up to 1-2°C (1.8-3.6°F) cooler than surrounding river water, and were likely from springs, seeps, or possible subsurface flow. Additionally, slight margin warming occurred along river banks in reaches with shallow habitat and sparse riparian vegetation.

6.5. Summary

The Shasta River along Nelson Ranch is a unique river reach because of considerable influence from upstream spring contributions. However, during spring and summer seasons, the river is also strongly influenced by atmospheric heating as exhibited by increased daily mean temperatures and notable diurnal response. Water development and land use changes in the Shasta River basin, including low flow conditions, sparse riparian vegetation, diversion of springs near their sources, and warm water inflow from tailwater along the length of the river have fundamentally altered the thermal regime of the river (NRC 2004). Today, mean daily water temperature in the Shasta River along Nelson Ranch commonly exceeds 20°C (68°F) during summer months.

Of particular interest are the unique thermal signals observed on the Nelson Ranch. In the winter, upstream spring inputs form a warm water source and the river cools in the downstream direction. Conversely in the summer, the upstream spring inflows form a cool water source and the river warms in the downstream direction. Fall and winter are, in general, neutral. Of specific interest is the impact of upstream thermal conditions on minimum and maximum daily temperatures at the Nelson Ranch. There are periods of the year when maximum daily temperatures are considerably warmer (over 2°C (3.6°F)) at the upstream boundary of the ranch compared to the downstream boundary. The specific heat and density of water often result in thermal conditions from upstream reaches being transported to downstream reaches. Thus, developed land and water use, as well as spring inflow sources can impart specific thermal signatures on downstream reaches. Such conditions are hypothesized to occur at the Nelson Ranch, where relatively constant spring inflows at relatively constant temperatures, coupled with diversion practices and associated return flow at upstream locations, create unique thermal variability and daily maxima and minima patterns at downstream locations. As temperatures cooled in the fall and effects of land and water use practices abated, upstream and downstream temperatures took on attributes more typical of a stream with an upstream spring source. The most critical need at this time is to identify spring flow and temperature contributions to the Shasta River, as well as quantify the effects of land and water use in upstream reaches. The combination of these factors plays a critical role in the long term restoration of the Nelson Ranch reach for anadromous fish.

7. Geomorphology

Geomorphic processes and landforms play critical roles in determining physical habitat conditions in riverine settings. On the Shasta River, such processes and resultant channel morphological characteristics are largely controlled by spatially variable geologic (Wagner 1987) and hydrologic conditions. As such, a basin-scale understanding of longitudinally-variable geomorphic and hydrologic characteristics of the Shasta River is necessary to understand physical habitat conditions along the Nelson Ranch Reach.

7.1. Basin Geomorphologic and Hydrologic Characteristics

Headwater reaches (approximately RM 60 to 50) of the Shasta River drain the Eastern Klamath Belt rocks of the Scott and Siskiyou Mountains, with spring-fed tributary inflows from Boles and Beaughton Creeks, both sourced in the High Cascade rocks of Mount Shasta. River slopes approach 0.07, and bed materials are inferred to be dominated by cobbles and boulders. Across the southern end of Shasta Valley proper (approximately RM 50 to 33), Shasta River slopes steadily decrease from 0.005 to 0.004 as the river crosses the lithologic boundary between the rocks of the Eastern Klamath Belt and those of the High Cascades. In this river segment, commonly referred to as the Upper Shasta River, channel planforms exhibit braided to wandering morphologies and bed materials dominated by medium to fine sand, gravels, cobbles and boulders. Lower Shasta River reaches (approximately RM 33 to 15) maintain bed slopes of approximately 0.0009, exhibit tortuously meandering planform morphologies, and contain bed materials of silts, sands, and fine gravels. Channel reaches within Shasta Valley (approximately RM 50 to 15) flow across a floodplain of variable width (10-300 m) underlain by High Cascades materials and intermittently confined between 15 m to 200 m high conical hills and ridges - antecedent topography largely resulting from a late-Pleistocene debris avalanche sourced on an ancestral Mount Shasta (Crandell 1989).

Hydrologically, the Shasta River exhibits hybridized characteristics of both “spring-dominated” and “runoff-dominated” rivers (Whiting and Stamm 1995; Whiting and Moog 2001). Steady, seasonally-independent baseflow discharges suggest inflow sources from diffuse groundwater flow through a low-conductivity volcanic aquifer (Whiting and Stamm 1995; Whiting and Moog 2001), whereas both sharp and prolonged peaks in the hydrograph signify contributions from winter and spring rainfall and snowmelt. Hydrographs for the Shasta River at both Montague (RM 16) and Edgewood (RM 48), normalized by the mean annual flow for the period of record, show that the upper reaches of the Shasta River above Dwinnell Dam (RM 45 to 50) exhibit characteristics of a runoff-dominated stream, while lower reaches (RM 33 to 15) exhibit hydrologic characteristics of a spring-dominated stream periodically influenced by runoff-dominated flood events. Furthermore, the spring-dominated hydrologic characteristics of the Lower Shasta River appear largely derived from discrete spring-dominated tributary inputs (e.g. Big Springs Creek), which may be augmented by yet-unidentified diffuse groundwater sources.

Shasta River floodplain and gently sloping upland areas throughout Shasta Valley are intensely irrigated and utilized for agricultural practices including, pastureland and hay/alfalfa production (Vignola and Deas 2005). Cattle-grazing dominates riparian land-use with potential impacts on water quality and channel characteristics. The Shasta River is fully adjudicated, allowing riparian land owners and local irrigation districts to divert in-stream flow during the April 1 to September 1 irrigation season. Non-irrigation season water withdrawals are minimal.

7.2. Nelson Ranch Geomorphology

The Shasta River reach in the vicinity of the Nelson Ranch represents a unique geomorphic and hydrologic transition zone between the Upper (RM 60 to 34) and Lower Shasta River (RM 30 to 0). Characterized by changes in channel planform geometry, cross-sectional morphology, bed material size and hydrologic regime, this transition zone begins at approximately river mile 34 (2 river miles above the Nelson Ranch), ends at approximately river mile 30 and is strongly coincident with both a large decrease in valley slope (Figure 16) and voluminous effluent (groundwater) inflows from discrete (e.g. Big Springs Creek, Hole in the Ground Spring) sources which may be augmented by diffuse groundwater inflow. Channel gradient and hydrologic regime changes across the Upper and Lower Shasta River segments drive dramatic differences in observed channel morphology, and likely strongly influence habitat characteristics, including substrate composition, water quality, and riparian vegetation assemblages.

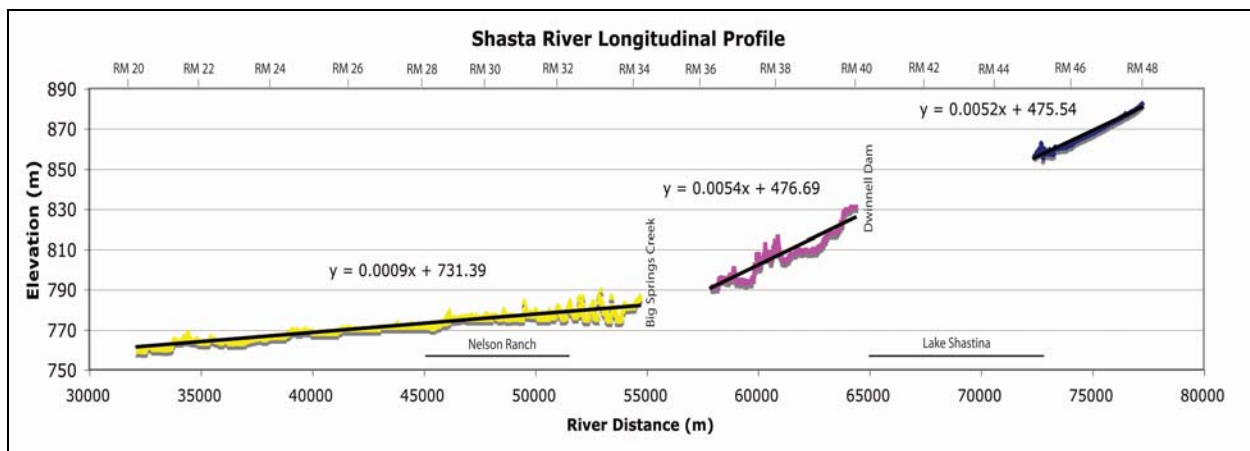


Figure 16. Longitudinal changes in DEM-derived valley slope between river miles 50 and 20.

Morphologically, the Upper Shasta River above Dwinnell Dam is characterized by wandering to meandering channel planform morphologies characterized by the presence of in-channel and lateral gravel bars, moderate to steep topographic gradients (0.005 to 0.06), riffle-pool bedform sequences, coarse bed materials and confined/narrow floodplains. Hydrologically, the upper river exhibits a hydrograph driven primarily by winter/spring rainfall and snowmelt, augmented by moderate summer baseflows sourced in several spring-fed tributaries (Beaughton Creek, Boles Creek) presumably sourced from High Cascade volcanic rocks on Mount Shasta (Nathenson et al. 2003). While access restrictions largely preclude extensive field observations of the Shasta River between Dwinnell Dam (RM 40) and the Nelson Property (RM 32) below Big Springs Creek (RM 34), initial site reconnaissance observations, historic photo and map analysis, and digital elevation model (DEM)-derived longitudinal profile data (Figure 16) suggest that this river segment was historically hydrologically and morphologically similar to upstream channel reaches. However, dam-driven flow regulation, particularly 90% reductions in mean annual flow and dramatic reduction in frequency and magnitude of flood flows, has strongly altered channel morphologies between Dwinnell Dam and Big Springs Creek. These changes have been dominated by reductions in meander bend wavelengths (Figure 17) and channel narrowing driven by vegetation encroachment (Pelzman 1973). The reduction in meander wavelength, measured as the straight-line distance between meander bend inflection

points, is expected following dam construction and flow regulation, due to established empirical relationships between meander wavelength and discharge (Carlston 1965; Dury 1965; Schumm 1967).

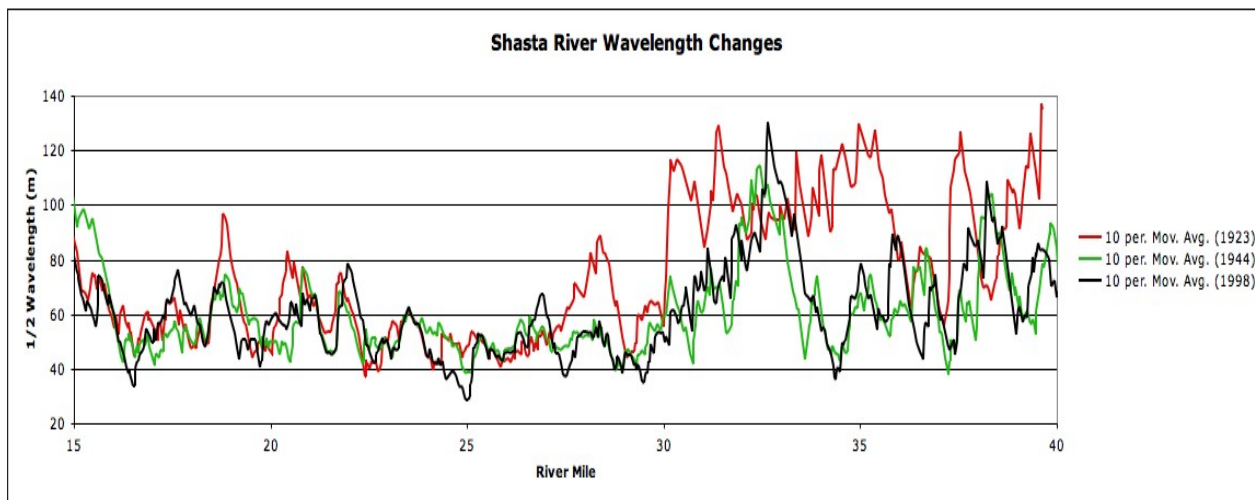


Figure 17. Historical longitudinal trends in Shasta River meander wavelength.

Contrasting with observations made on the Upper Shasta River, lower river reaches (including lower portions of the Nelson Ranch) exhibit tortuously meandering planform morphologies, shallow topographic gradients (< 0.001), plane bed bedform morphologies (i.e. relatively flat) and silt, sand and fine to medium-sized gravel bed materials. The approximately 16 river miles of lower river in the Shasta Valley proper (RM 30 to RM 14) above the Shasta River Canyon actively meanders across relatively unconfined floodplain reaches. While antecedent geologic structures (i.e. Pleistocene debris flow-derived andesite blocks) (Crandell 1989) locally constrain channel migration and planform geometry throughout this 20-mile long meander belt, unconfined reaches exhibit active bank erosion and lateral channel migration, resulting in numerous meander cutoffs and subsequent creation of oxbows and other ephemeral backwaters. Hydrologically, the lower river within Shasta Valley proper (in the absence of irrigation) exhibits minimally variable, spring-fed baseflows augmented by winter and spring flood events derived from both surface runoff and headwater snowmelt. Channel morphologies strongly reflect consistent spring-fed baseflow conditions and exhibit many similarities (e.g. high roughness values driven by vegetation growth) to previously studied spring-fed rivers in Idaho and Oregon (e.g. Whiting and Moog 2001).

As discussed above, Nelson Ranch channel reaches represent a geomorphic and hydrologic transition zone between the upper and lower rivers. Because the Nelson Ranch presents the first continuous stretch of laterally unconfined floodplain habitat downstream from Big Springs Creek, the reach not only helps dampen flood pulses from Parks Creek, but also provides fine-grained floodplain sediments into which the river can laterally erode. A consequence of this lateral erosion is both a large increase in channel sinuosity and coincident decrease in channel wavelength (Figure 17). Gradual decreases in meander wavelength throughout the Nelson Ranch (Figure 17) from upstream to downstream, further support the idea that the Nelson Ranch, and particularly channel reaches spanning river miles 32 to 30, is a transition zone between the hydrologically and geomorphically distinct upper and lower river segments.

7.3. Historic Year-in-the-life

Historic geomorphic processes along the Nelson Ranch Reach have remained largely unaltered since river adjudication in 1923 and construction of Dwinnell Dam in 1928. Upstream reservoir impoundment and diversion of Parks Creek into the Shasta River near Edgewood, above Lake Shastina, have diminished peak flood flows associated with winter rainfall events and spring snowmelt. Such hydrologic process alterations would be expected to produce substantial changes in downstream channel morphologies. However, relatively continuous baseflow conditions below Big Springs Creek have largely dampened the dam-induced hydrologic alteration and associated geomorphic process changes observed upstream (RM 40 to 34), thus minimizing post-dam geomorphic change along the Lower Shasta River. Based on historic geomorphic observations from the Nelson Ranch using decadal-series aerial photographs, several observations regarding historical channel planform behavior can be made:

- Geologically unconstrained portions of the Shasta River through the Nelson Ranch have exhibited progressive lateral channel migration at rates approaching 0.10 meters/year. When normalized by average channel width (~10m), lateral migration rates approach those observed on the Lower Sacramento River (Micheli et al. 2004), suggesting that the Lower Shasta River is actively meandering at a relatively high rate, a physical process essential for the creation of complex riparian and aquatic habitats. Lateral channel migration is directly responsible for the creation of cutbanks, resultant bank sloughing, and subsequent habitat availability within the Shasta River.
- Meander cutoff events and the creation of oxbow lakes/backwaters appear correlated with large winter flood flows.
- Meander bend wavelength magnitudes below the GID diversion (i.e., lower river) remain remarkably unchanged when compared using the pre-dam and post-dam record. This suggests geomorphic and hydrologic processes responsible for controlling channel planform geometry (i.e. meander wavelength) are largely unaltered by construction of Dwinnell Dam and the associated Parks Creek Diversion.

7.4. Current Year-in-the-life

Minimal, yet progressive, channel form change was observed throughout water year 2007 along the Nelson Ranch reach. Repeated channel cross-sectional surveys identified seasonally transient bedforms influenced predominantly by cutbank sloughing (i.e. bank failure) and in-channel sediment accumulation associated with seasonal rooted aquatic macrophyte growth. Additionally, lateral point bar aggradation and vertical sediment accretion along channel margins was observed. Channel margin sediment accretions are typically stabilized by emergent vegetation, facilitating the maintenance of rectangular channel cross-section morphologies during irrigation season flow regimes, while also continually creating low elevation floodplain benches typically inundated during higher, pre- and post-irrigation season flows (Figure 18). Seasonal variability in channel characteristics is discussed below.

Fall: October to November

The cessation of irrigation water withdrawals on 1 October allowed spring-fed baseflows to fill the river channel almost to bankfull capacity. Furthermore, while discharge magnitudes observed in October and November are lower than those observed in March (immediately prior

to the irrigation season), water stage is higher (see Figure 18), largely a result of channel roughness increases induced by in-channel growth of aquatic macrophytes. A consequence of this elevated river stage is channel margin geomorphic surfaces, stabilized by emergent vegetation during the irrigation season, become inundated until the rooted aquatic macrophytes senesce towards the end of the fall. Complex interactions between river stage and aquatic macrophyte senescence define the hydrogeomorphic characteristics along the Nelson Ranch Reach during this period of the year

Winter: December to March

In the absence of both irrigation withdrawals and aquatic macrophytes, water stage during the winter months exhibits minimal variation, largely reflecting consistent spring-fed baseflows. Water stage increases are typically only observed following rainfall events. During the winter, channel margin geomorphic surfaces (i.e. floodplain benches) provide critical, shallow and low-velocity rearing habitat for juvenile salmonids throughout the Nelson Ranch reach. Winter baseflows inundate these channel margin surfaces under 10 to 50 cm of water, thus extending available juvenile salmonid rearing habitat onto the channel margins (Figure 18).

Spring: April to June

The initiation of irrigation pumping on 1 April substantially dewateres channel margin geomorphic surfaces, thus creating hydrologic separation between these shallow, low-velocity channel margin areas from the largely homogenous (and higher velocity) main channel (Figure 18). Channel margin geomorphic surfaces remain largely dewatered until the initiation of in-channel aquatic macrophyte growth in late-spring/early summer. Geomorphic field mapping suggests that channel margin habitat is reduced by approximately 15,000 square meters along the 3 river miles downstream from the GID diversion following initiation of irrigation withdrawals beginning in April.

Summer: July to September

Late-spring/summer aquatic macrophyte growth forces considerable increases in river stage due to marked increases in channel roughness. However, observed macrophyte-induced increases in river stage still do not allow for channel margin geomorphic surfaces to be inundated, largely due to the magnitude of water withdrawals for irrigation. Such hydrogeomorphic conditions lead to reduced habitat complexity.

Summary

Stage and flow information presented in Table 1 and Figure 18 indicate complex interactions between river stage, irrigation, and aquatic macrophyte growth. For example, on 16 October the stage was approximately 0.15 meter higher than on 20 March, but the flow was 10 cfs lower. Such seasonal variations in the stage-discharge relationship directly impact habitat availability, particularly during the winter/early spring juvenile salmonid rearing period.

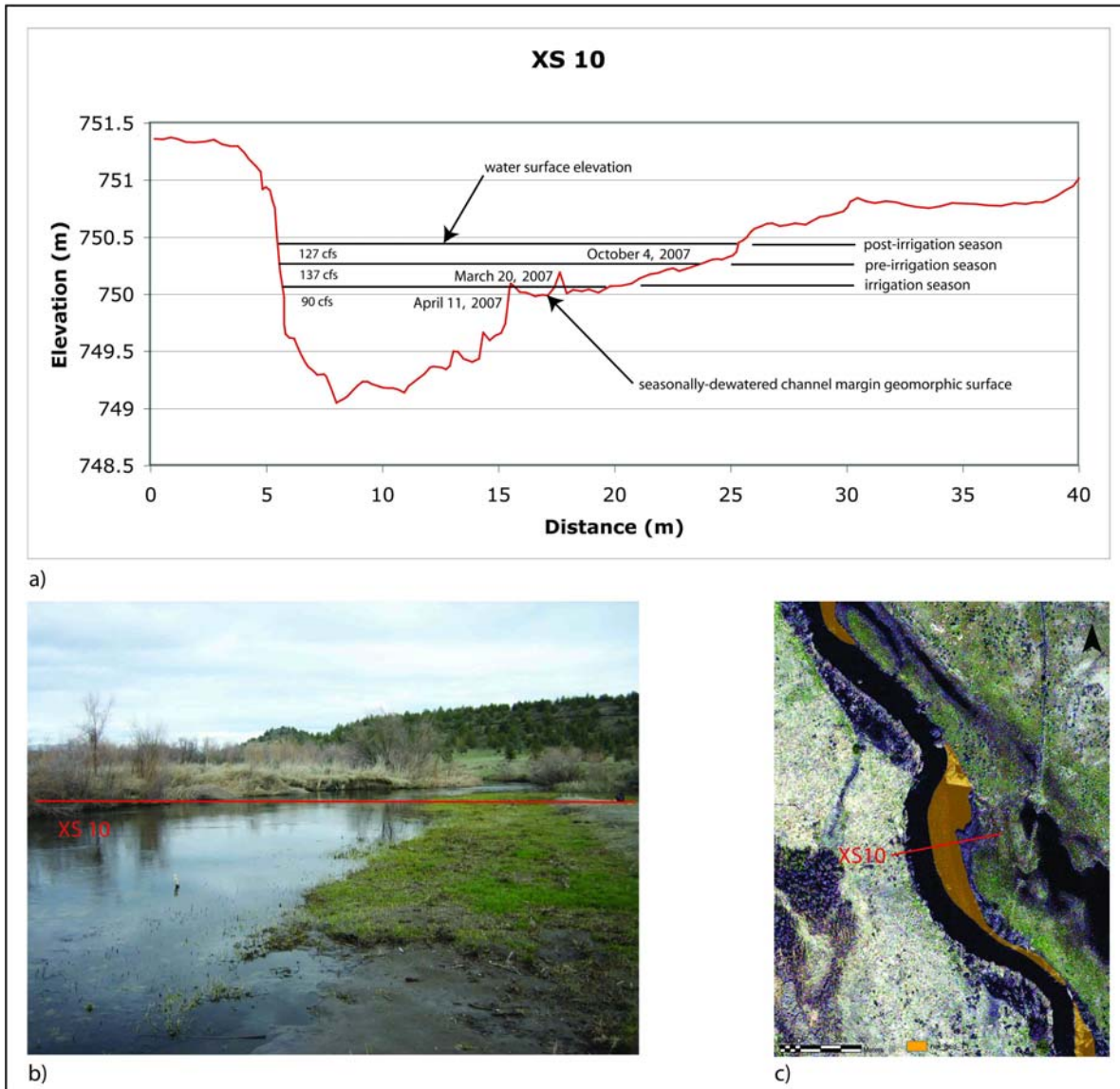


Figure 18. Habitat loss associated with irrigation-driven dewatering of channel margin geomorphic surfaces: a) irrigation season water withdrawals and seasonally variable aquatic vegetation growth substantially alter water surface elevations; b) channel margin geomorphic surface partially inundated prior to initiation of irrigation season (March 20, 2007); c) orange polygons in planform map represent geomorphic surfaces inundated during non-irrigation season and dewatered during the irrigation season.

Qualitative observations suggest sand-sized bed materials are continuously mobile throughout the water year, while gravel-sized bed materials are only mobile during flood events. Consequently, gravels patches are observed to accumulate sand-sized materials throughout the summer, often covering available gravels during the irrigation season. Fine-sediment accumulations are largely removed from gravel patches with increased flow following cessation of irrigation practices and decreased aquatic macrophyte growth during fall senescence.

8. Habitat Mapping

In July 2006, we performed a site reconnaissance of in-channel and floodplain localities throughout the Nelson Ranch. Reconnaissance goals included the identification, categorization, and mapping of in-channel and channel-margin aquatic habitat. Habitat mapping was performed to quantify the amount of different habitat types available to salmonids rearing along the Nelson reach. The described habitat types were then used as sample sites for the various studies (i.e. fish, invertebrate, food web, aquatic macrophyte, water temperature, and geomorphic) taking place along the Nelson reach.

8.1. Methods

Initial site reconnaissance indicated that habitat typing methods typically utilized in salmonid-bearing streams in California (Flosi et al. 1998) were inappropriate for the Nelson Ranch study reach. The relatively low-gradient and largely spring-fed Shasta River along the Nelson Ranch reach exhibits minimal variation in water surface gradient and substrate composition, largely precluding first-order habitat discrimination using the typical classification categories of riffles, runs and pools (Flosi et al. 1998). Following available CDFG typing methodologies (Flosi et al. 1998), initial reconnaissance observations typed the vast majority of available in-channel habitat as “glide” intermittently augmented by backwater and corner pools. Furthermore, initial fish snorkel survey efforts identified extensive use of local habitats not described using traditional typing methods, necessitating the identification and mapping of stream-specific habitat units for sampling purposes. Consequently, a site-specific habitat classification system was created to physically describe 100% of the wetted channel. The current classification system includes six types of channel margin and/or in-channel aquatic habitat, including emergent vegetation (EV), active cut banks (CB), point bars (PB), large woody debris (LWD), aquatic macrophytes (AM), and perennial and/or ephemeral backwaters (BWp/BWe). Similar to CDFG protocols, homogenous habitat areas with a length of one channel width or greater were considered distinct habitat units and thus mapped individually.

Emergent vegetation habitat type consists primarily of rushes (*Scirpus* sp.), cattails (*Typha latifolia*), and reeds (*Sparganium emersum*), typically found at rivers edge (Figure 19a). Cut bank habitat is located at the outside edge of meander bends in the river. In most cut bank habitat, large blocks of bank material have sloughed into the river providing a velocity refuge for fish (Figure 19b). Point bar habitat is found at the inside bend of meanders and consists of a gradual slope into the river (Figure 19c). Substrate on point bars consists of a mixture of gravel and sand. Large woody debris in the Shasta River is different from that normally classified as LWD in most rivers where juvenile salmonids rear (Figure 19d). Refuge provided by LWD in the Shasta River is defined as trees, four to six inches in diameter or larger that have fallen into the river, or submerged roots of trees growing at the rivers edge. These sources of LWD provide a local velocity refuge for fish and not necessarily geomorphic structure as found in other juvenile salmonid rearing rivers. Mid-channel aquatic macrophyte habitat is most often in predominately linear reaches of the river and typically has a gravel substrate (Figure 19e). On this gravel substrate, aquatic macrophytes provide a velocity refuge for fish and substrate for benthic macroinvertebrates. Backwater habitat areas, both perennial (BWp) and ephemeral (BWe), are typically characterized by zones of low flow velocity. Perennial backwater areas are generally found in large eddies on the downstream ends of point bars, in large pools formed

immediately downstream from irrigation diversions, or adjacent to excessively eroded cut banks at the outside of meander bends (Figure 19f). Ephemeral backwater areas, generally inundated only during winter baseflow and flood flows, are typically found in remnant channels/oxbows abandoned following meander cut-offs during apparent channel avulsion events. Ephemeral backwater areas are typically colonized by woody riparian vegetation.

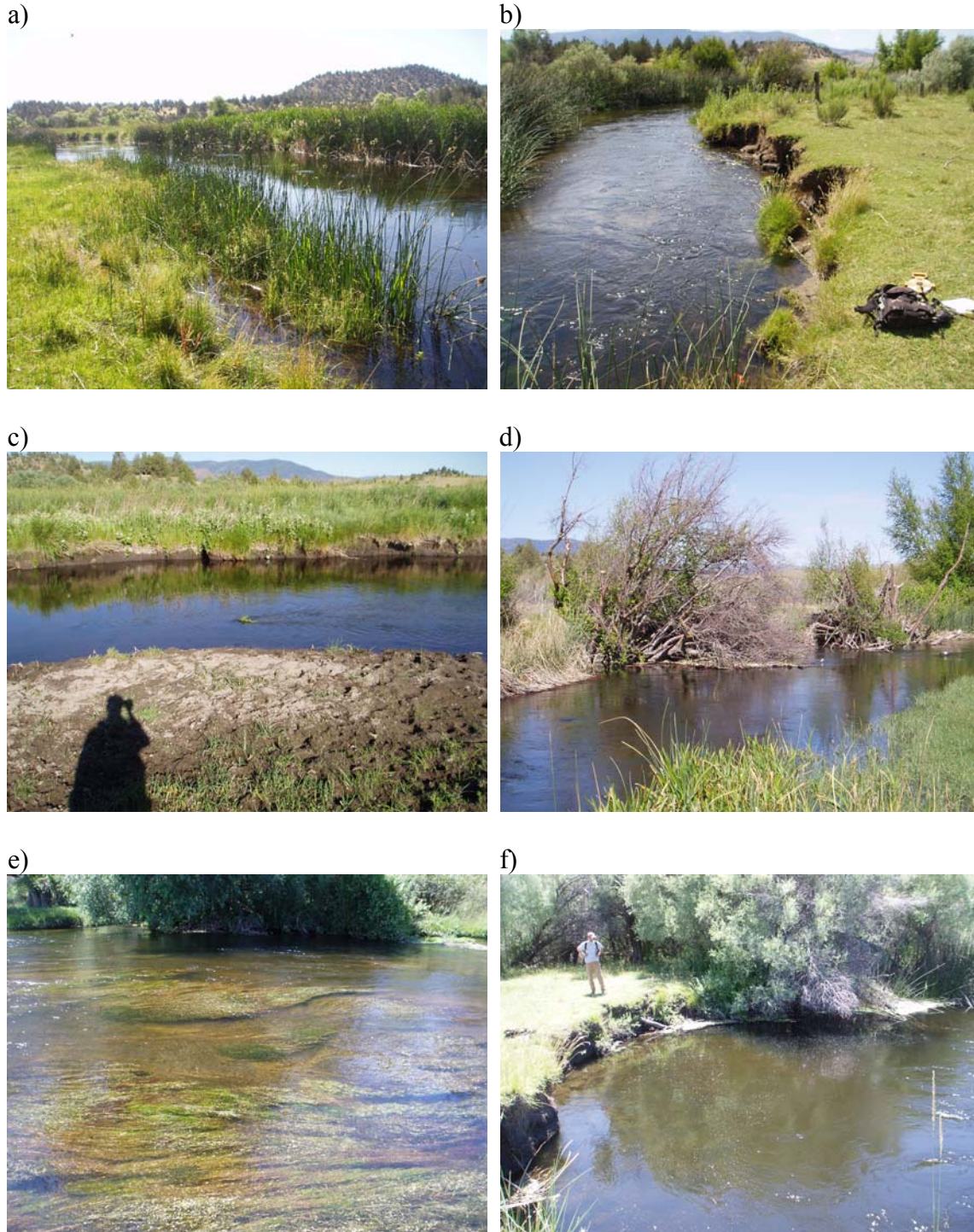


Figure 19. Habitat types described along the Nelson reach: a)Emergent vegetation, b)Active cut bank, c)Point bar, d)Large woody debris, e)Aquatic Macrophyte, and f)Backwater.

Field mapping of channel margin and/or in-channel aquatic habitat units was conducted on approximately 1:1,500 scale aerial photographs provided by The Nature Conservancy, California (TNC). Habitat areas reflect visual observations made in July 2006, and are not necessarily

representative of habitat availability during different water year types and at different flow magnitudes.

8.2. General Observations

The density and spatial distribution of habitat areas varies considerably. Upstream from the GID diversion, the lack of meander bends virtually eliminates perennial backwater, cut bank and point bar habitat areas in this upstream reach, with in-channel and channel margin habitat area dominated by emergent vegetation (EV) and aquatic macrophytes (AM). Factors potentially contributing to this observed lack of meander bends and associated habitat diversity include: 1) valley/channel slopes are greater and topographic constrictions more prevalent above the GID diversion, physically precluding formation of the tortuous meanders which facilitate backwater, point bar and cut-bank habitats observed downstream; and 2) backwater effects from the GID flashboard dam largely cover habitat areas dependent on channel bed/bank morphologies (e.g. point bar and cut-bank) resulting from stream meander processes. The dominant drivers of habitat simplification upstream from the GID diversion are presently unknown, but it is possibly a function of the backwater from the GID diversion. Currently the GID flashboard dam is slated for removal, an action which will hypothetically create more heterogeneous summer aquatic habitat upstream for at least a portion of the 1.6 km (1 mi) backwater effect from the dam. Downstream from the GID diversion, tortuously meandering channel morphologies appear to facilitate significantly greater variability in available habitat area (Figure 20, Figure 21).

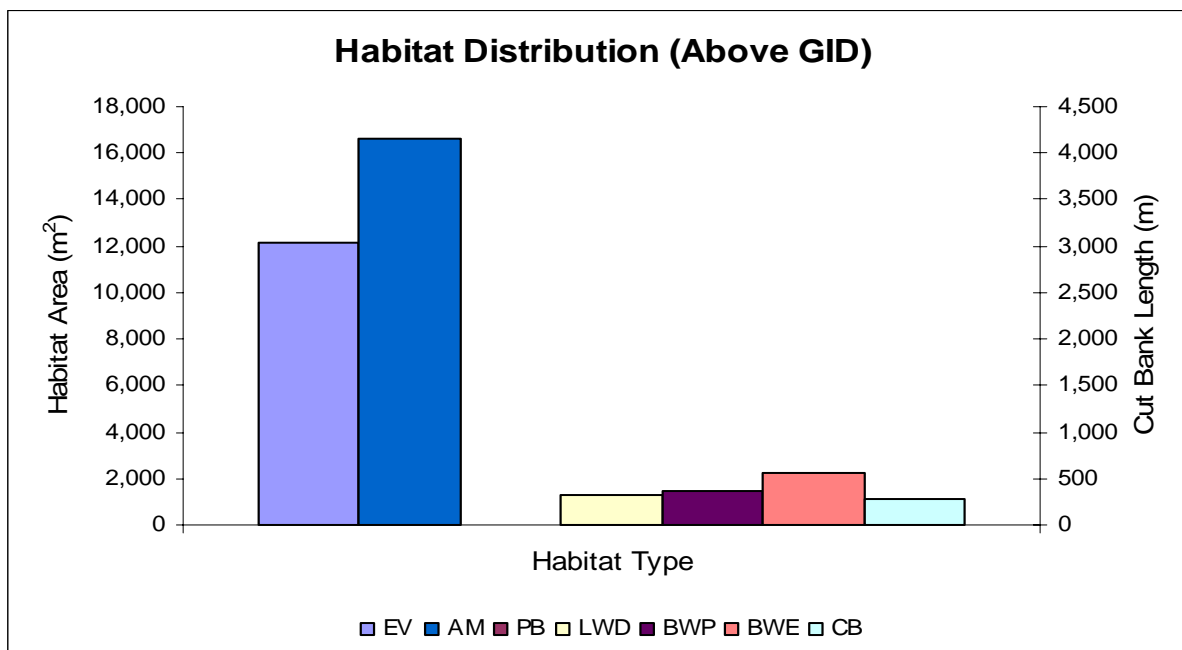


Figure 20. Habitat distribution above GID dam. Habitat types include emergent vegetation (EV), aquatic macrophytes (AM), point bar (PB), large woody debris (LWD), perennial back water (BWP), ephemeral backwater (BWE), and cut bank (CB).

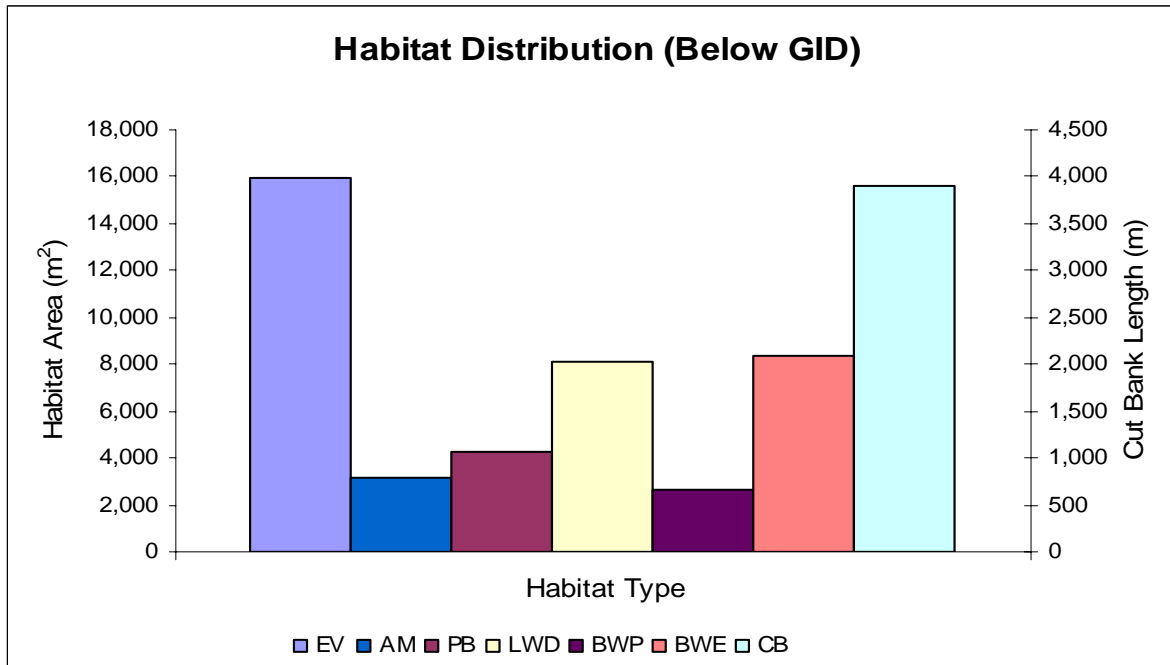


Figure 21. Habitat distribution below GID dam. Habitat types include emergent vegetation (EV), aquatic macrophytes (AM), point bar (PB), large woody debris (LWD), perennial back water (BWP), ephemeral backwater (BWE), and cut bank (CB).

9. AQUATIC MACROPHYTES

Aquatic plants are extremely abundant in the Shasta Valley reaches of the Shasta River. A survey of the stream conducted in August of 2003 (NCRWQCB 2004) found roughly 50% of the valley streambed covered with aquatic plants. Observations by Center for Watershed Science staff confirm that aquatic plants are widespread on the Nelson Ranch reach. During data collection efforts on the Nelson Ranch in 2006, aquatic macrophytes appeared to reach peak abundance by the end of September.

Aquatic macrophytes discussed in this section include plants rooted in the wetted stream bed, and which are either facultative or obligate submersed plants. This excludes plants such as *Typha latifolia* and *Juncus acuta* that never grow submerged, but includes plants like *Veronica* spp, that can grow submerged or emergent. Algae was not considered together with aquatic plants in this study.

The impetus to examine aquatic macrophytes was based on the hypothesis that these plants could drive other key processes in the reach, such as regulation of nutrients and provision of habitat for invertebrates and fish, as well as affecting channel roughness and river stage. The quantitative description of aquatic plant coverage on the ranch presented in this section was motivated by a need to gain insight into the role of aquatic macrophytes in the stream ecosystem.

9.1. Methods

To assess macrophyte distribution and growth, thirteen sites were established consisting of transects through the various habitat types to maximize coincidence with the fish-survey transects. Sites were selected to represent five of the six previously described habitat types (Table 3). The habitat types surveyed include:

- AM sites that are generally straight sections of the stream, with depth rarely exceeding one meter, and symmetrical in cross-section.
- PB sites with sharp bends and well-established point bars formed on the one bank and depths as great as 1.8 meters on opposite banks, and very asymmetrical cross-sections.
- The single EV site surveyed was a u-shaped channel, depth above 1.6 meters for most of the cross-section, and in a straight section of the stream.
- CB sites are bends in the stream with the right bank steeply eroding and occasionally sloughing chunks of bank.
- BW sites are sharp bends in the stream with several meters or more of flow reversal occurring on the right margin and depth to 1.8 meters.

No LWD sites were included because of safety considerations and the difficulty of observations. All of the AM sites were included. Each transect was surveyed once per month (with exceptions noted in Table 3 below) for eight months (March through October). The survey consisted of traversing the river along a tape-transect and identifying the beginning and end points for each patch encountered. The location, species, and depth were recorded throughout the transect. The longitudinal extent of each patch above and below the transect were also recorded starting in May. Left and right edges of the wetted channel were recorded for each transect.

Table 3. Sites surveyed for aquatic macrophytes.

Site	Habitat Type	River-mile	# of Surveys	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct
AM5	Aquatic Macrophyte	27.38	8	x	x	x	x	x	x	x	x
CB5	Cut Bank	27.45	6	x	x	x	x		x	x	
PB5	Point Bar	27.84	7	x	x	x	x	x	x	x	
AM4	Aquatic Macrophyte	28.17	7	x	x	x	x	x	x	x	
PB3	Point Bar	28.98	7	x	x	x	x	x	x	x	
BW3	Backwater	29.11	6		x	x	x	x	x	x	
AM3	Aquatic Macrophyte	29.31	7	x	x	x	x	x	x	x	
CB2	Cut Bank	30.07	7	x	x	x	x	x	x	x	
PB2	Point Bar	30.18	7	x	x	x	x	x	x	x	
AM2	Aquatic Macrophyte	30.35	8	x	x	x	x	x	x	x	x
EV1	Emergent Vegetation	31.15	4	x	x			x	x		
BW1	Backwater	31.35	6	x	x	x	x	x		x	
AM1	Aquatic Macrophyte	31.87	8	x	x	x	x	x	x	x	x

The program sampling frequency was approximately monthly at each site, but if the surveyor was unable to wade the stream safely, the effort was abandoned for that month. In September, four extra transects, two upstream and two downstream of the original transects (at six-foot intervals), were surveyed at sites AM1, AM3, and AM5.

Due to the low elevation of the late-fall sun, it was only feasible to consistently identify plants for a few hours of the day without significantly changing survey methods. In addition, the change in coverage between September and October appeared to be minor. For these reasons, the choice was made to collect multiple transects at three sites during the October sampling, rather than change methods to allow for data collection at single transects at all sites. The three sites sampled in October (AM1, AM2, and AM5) were surveyed along four extra transects in similar fashion to the extra transects surveyed in September. These extra transects surveyed in September and October were used to examine within-site variation.

To compare diversity between sites and months, Shannon's diversity index (Shannon, 1948) values were calculated using the formula:

$$H' = -\sum_{i=1}^n p_i \log p_i$$

where H' is the index, p_i is the proportion of total habitat area covered by the i^{th} category of coverage, and n is the number of categories. The total habitat area is taken to be the sum of the transect lengths, and the proportion is the intersected length of each transect, with each plant species counting as one category. This effectively treats each species as a separate habitat type, and all areas with no plant cover as a single habitat type, nonetheless, this measure of diversity can be used to compare the species distribution and diversity between different habitat types. This method was chosen rather than Simpson's Diversity Index because Simpson's method requires counting individual plants, which would have been prohibitive in this study.

Flow velocity measurements were taken above, at the upper margin, and within several patches of single species, during the April and October sampling events. These measurements are not intended for analysis but merely to provide some typical values; however, they are included for discussion below.

9.2. Aquatic Macrophyte Coverage

Averaged across all transects, total percent cover ranged from a low of 26 percent in March to a high of 73 percent in September then declined to 68 percent in October (Figure 22). All values fall within one standard deviation of each other. The mean value used to determine standard deviation was the mean of all transects surveyed in a particular month. Because the various habitat types sampled were diverse in their total coverage, the magnitude of this standard deviation is not unexpected.

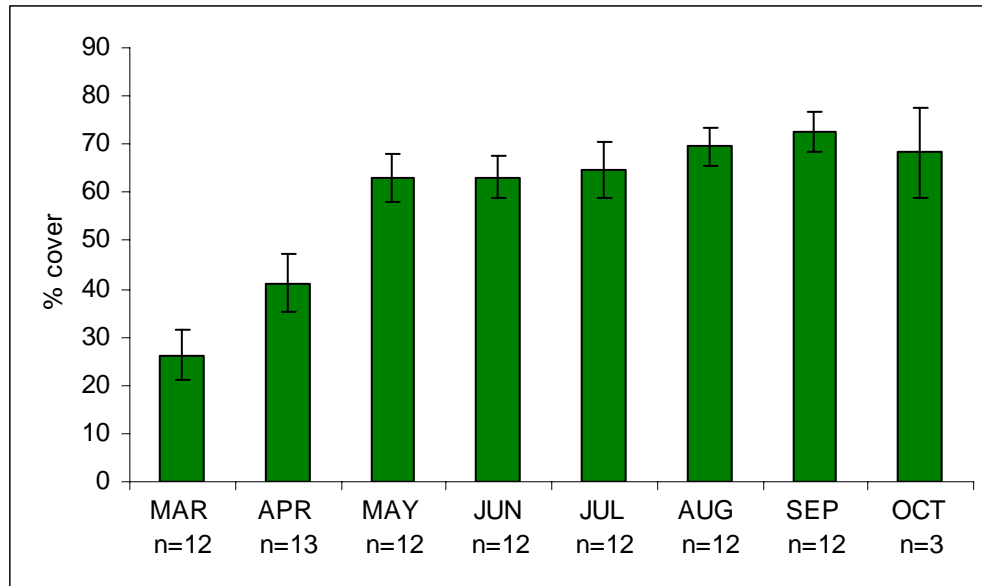


Figure 22. Percent cover averaged across all site types (with standard error).

When only the AM-type sites were taken into consideration, a similar pattern holds; however, the peak of the growth occurred during August, rather than September (Figure 23). Coverage increased rapidly in all transects during March, April, and May, then seemed to approach an equilibrium for the rest of the summer months. Typical late season aquatic plant coverage on small streams ranges between 15 percent and 70 percent (Wolfert 2001, Riis et al. 2001), putting the study region at the high end of the distribution.

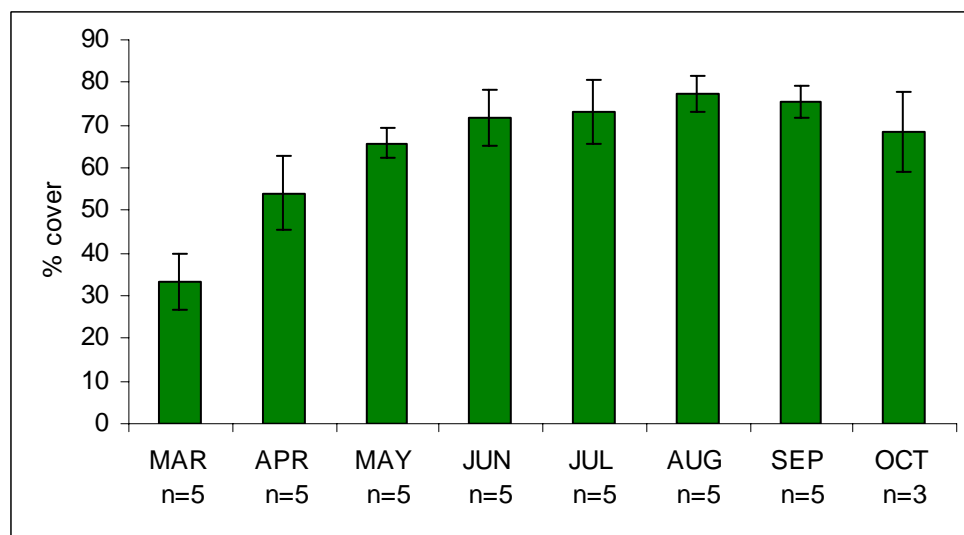


Figure 23. Percent cover averaged across all aquatic macrophyte type sites (with standard error).

Overall, the CB sites had the highest percent coverage, followed by AM sites, with PB and BW site types having similar, but usually lower coverage (Figure 24). Insufficient information on EV was available to make a meaningful comparison.

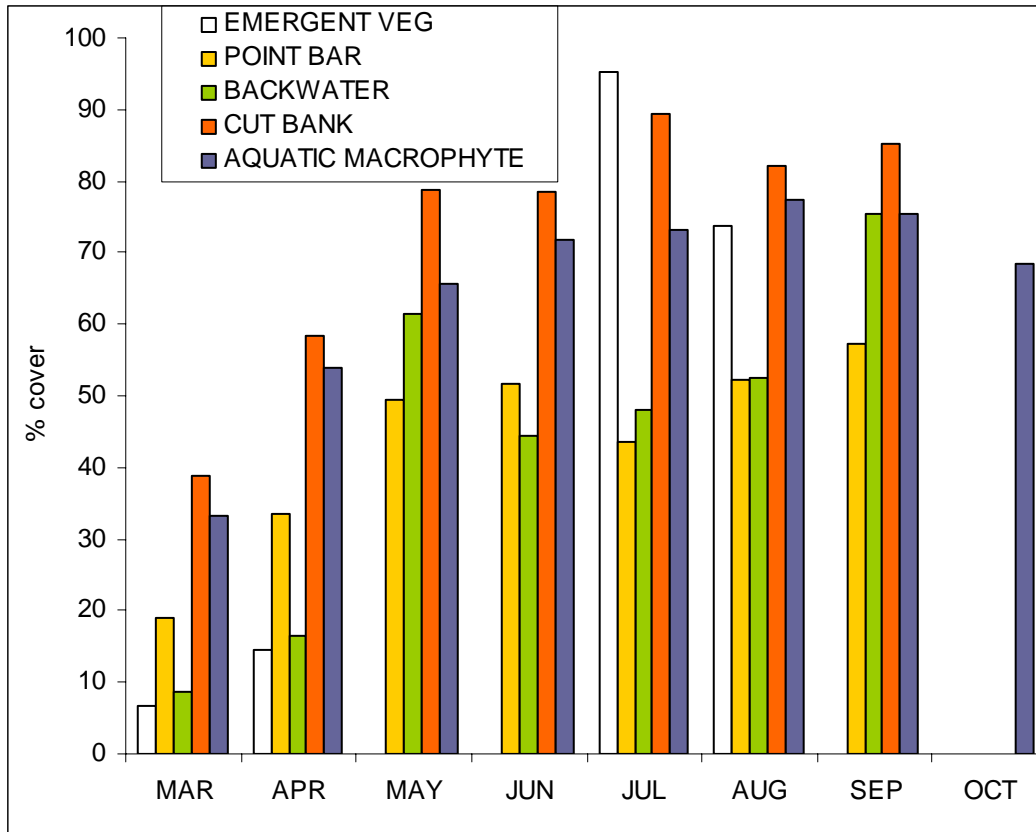


Figure 24. Percent cover of aquatic macrophytes by site type.

9.3. Species Present

Six species of rooted, fully aquatic macrophytes were differentiated in this study (Table 4). *Potamogeton pectinatus* and *Zannichellia palustris* were indistinct in the earlier part of the season and have very similar habits. *Zannichellia palustris* was not understood to be present until late in the season, and only occurred in a few transects, with low coverage. The two species have been treated as one for the purpose of this study. *Veronica anagalis-aquatica* and *V. catenata* are known to hybridize, essentially identical when not in flower, and were not identified to species in the field. The plant labeled as "Unknown 1" was never observed to flower and has not been identified. All of the identified rooted aquatic macrophytes observed were perennials, and all are native to California except for the *Veronica* spp.

Table 4. Species encountered in the aquatic macrophyte survey transects.

Name	Common	Habit
<i>Elodea canadensis</i>	Common Waterweed	Stoloniferous
<i>Potamogeton pectinatus</i>	Fennel-leaf Pondweed	rhizomatous, tuberous
<i>Zannichellia palustris</i>	Horned Pondweed	Rhizomatous
<i>Ranunculus aquatilis</i>	Water Buttercup	semi-stoloniferous
<i>Veronica anagalis-aquatica</i>	Water Speedwell	rhizomatous, crosses with <i>V. catenata</i>
<i>Veronica catenata</i>	Chain Speedwell	rhizomatous, crosses with <i>V. anagalis-aquatica</i>
<i>Myriophyllum sibiricum</i>	Milfoil	Rhizomatous
Unknown 1	?	stoloniferous?

It should be noted that this list is not exhaustive of all the aquatic plant species present in the Nelson Ranch section of the Shasta River. Emergent species such as *Typha latifolia* were not considered in this study; non-rooted species such as *Lemna minor* are not included in this analysis; and there are several additional submerged, rooted macrophytes known to be present in the stream that did not appear in the transects. Appendix A of NCRWCQB (2004) includes a more extensive list and description of aquatic and semi-aquatic plants in the Shasta River.

Averaged across the study area, from March to September, *Elodea canadensis* accounted for the largest transect coverage. In October, *Potamogeton pectinatus* was the dominant plant in the transects. These two species, along with *Ranunculus aquatilis*, made up about 87 percent of the observed plant distribution averaged across all sites and months. *Myriophyllum sibiricum* was generally the least common species observed, and occurred in small patches. The unknown species and the *Veronica* spp. followed roughly similar patterns over the study period, peaking in abundance much earlier, and much lower, than the more abundant species. Figure 25 shows the relative percent coverage of each species averaged across all habitat types for each month of the study.

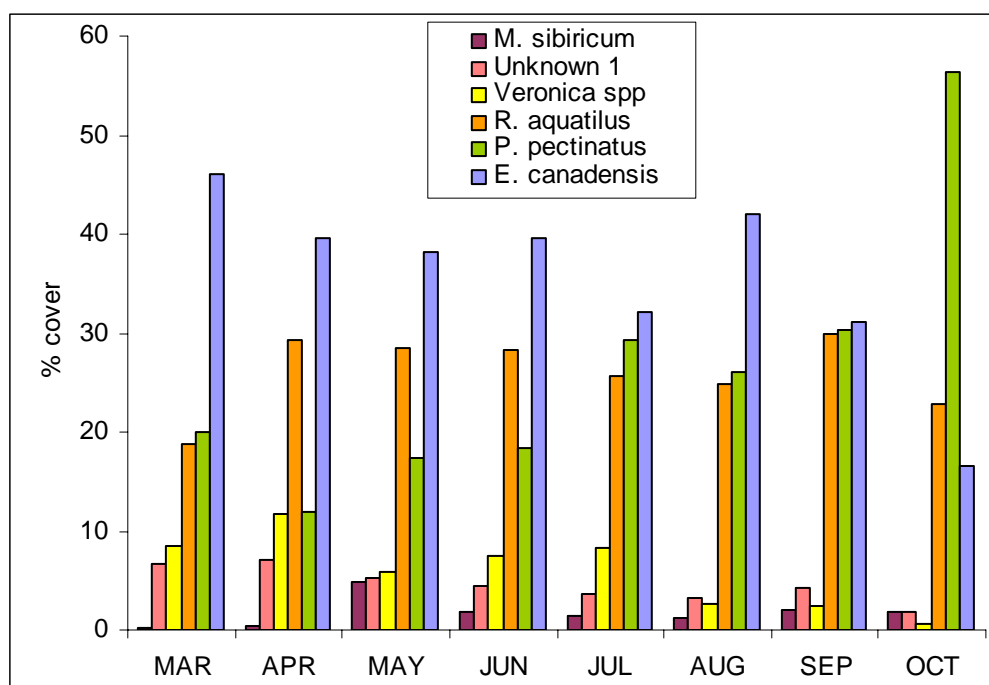


Figure 25. Percent cover of aquatic macrophytes by species and month across all sites.

Assemblages of plant species at the different site types are shown in Figure 26. In AM sites, *R. aquatilis* and *P. pectinatus* were the most abundant species, followed closely by *E. canadensis*. In BW and CB sites, and to a lesser extent, PB sites, *E. canadensis* was far more abundant than the next most abundant species. Because only one EV site was surveyed, it is difficult to say if it is representative of the type.

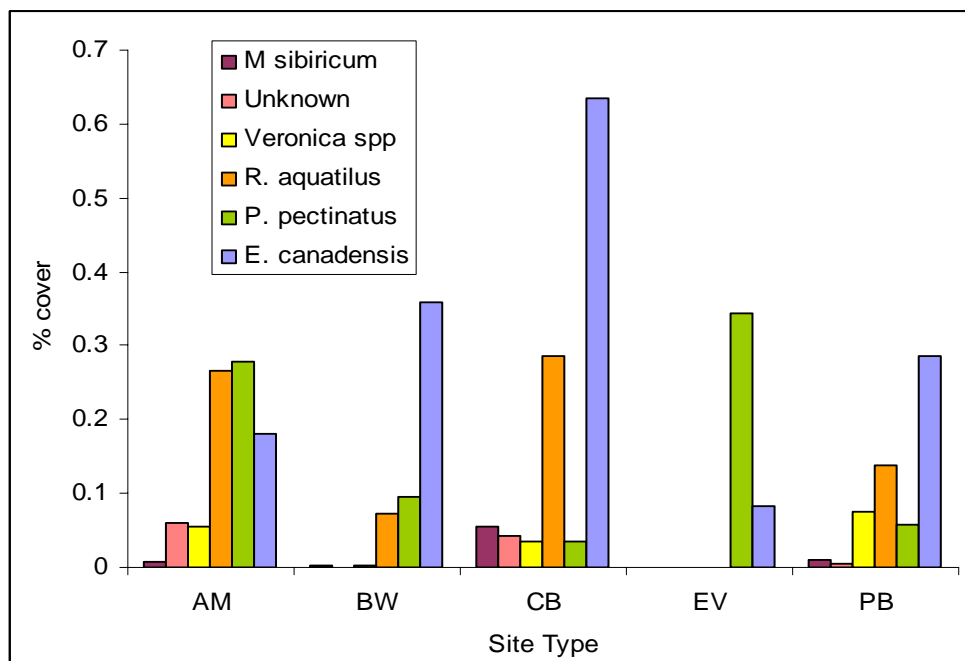


Figure 26. Percent cover of aquatic macrophytes by species and site across all months.

9.4. Aquatic Macrophyte Diversity

Shannon's Diversity Index values for each site type on a monthly basis are shown in Figure 27. For sites other than BW, diversity appears to peak in May and then decline slightly during the summer with an increase in late summer. The BW site data suggests that H' lags a month or two behind, with a peak occurring in June or July. The most diverse site types seem to be the CB and AM sites.

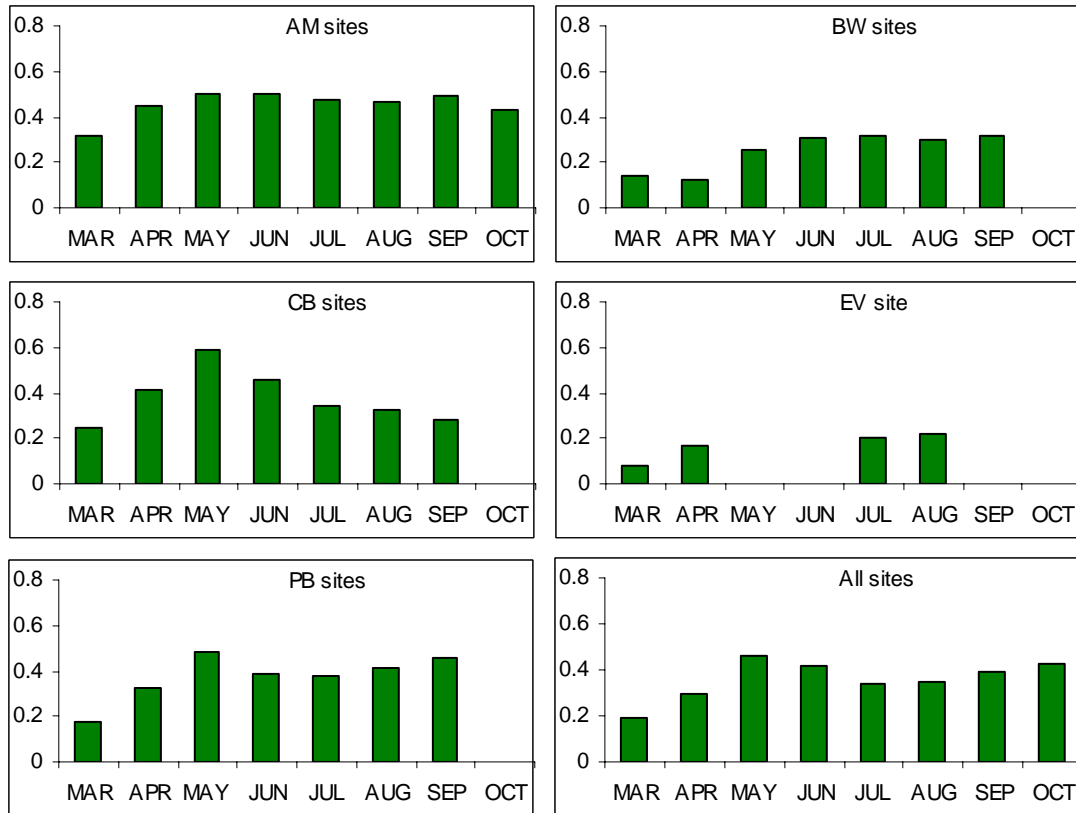


Figure 27. Shannon's diversity index for each site type.

The AM sites, shown individually in Figure 28, indicate a fairly wide distribution of values represented within this single site type, with AM2 having nearly twice the diversity of AM1 on average throughout the growing season.

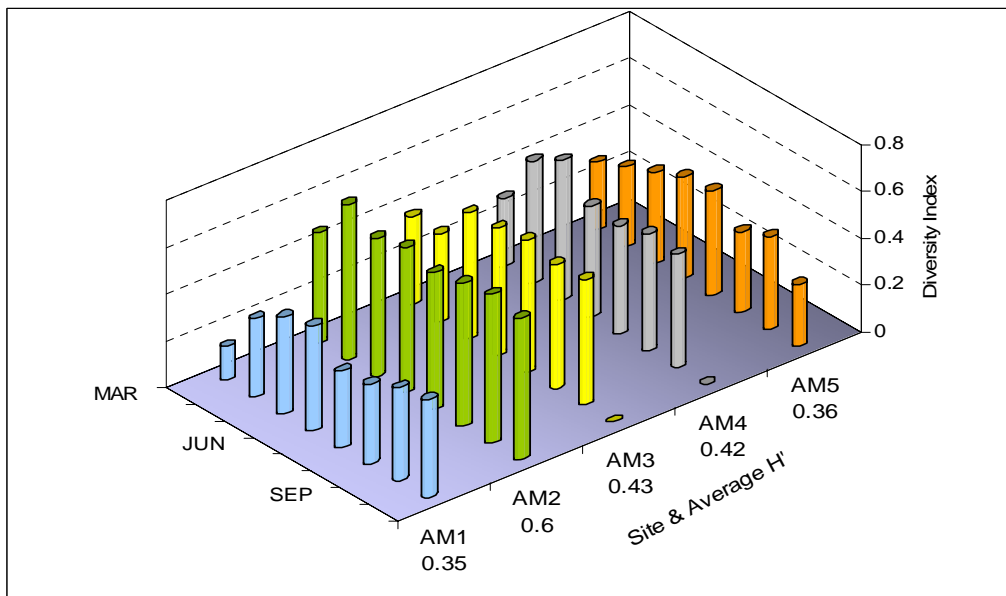


Figure 28. Shannon's diversity index for all five aquatic macrophyte sites (AM 3 and AM4 not surveyed in October).

Within-site variability was only examined in AM sites during September and October. For the sites that were sampled at multiple transects, the average standard deviation of per-species percent cover between transects at a site, was 0.056. In comparison, the average standard deviation between sites was 0.106. This means that nearly twice as much variability occurred between sites as occurred between transects at the same site, as an average of the percent cover of each species.

9.5. Patch Morphology and Plant Behavior

The various species can be grouped by their growth patterns in flowing water. *E. canadensis*, *R. aquatilis*, and the unknown sp. all have similar morphology at the patch scale, although they have different leaf and shoot morphology, and the unknown tended to occur in smaller patches (less than 5 square meters) than the other two (as large as 40 square meters for *E. canadensis*, and up to 12 square meters for *R. aquatilis*). Patches of these three species tend to be dense and close to the bed of the stream in places with appreciable flows; the stream bed is rarely visible below a patch. The upstream edges of patches tended to be sparsely rooted with larger stems. The downstream edges of patches tend to have a ragged appearance, and accumulate fine sediment. These species generally were only in motion at the downstream edge of the patch. Water velocities within patches of these species were always very near zero, and low velocity regions form at the downstream edge of each patch (Table 5). Patches of these species either overlapped other patches, or were surrounded by uncolonized substrate.

Potamogeton typically occurred in more open patches such that the bed surface was often visible beneath the patch. Patches were generally large, ranging up to 40 square meters. Usually, the entire patch was in motion. Upstream edges of patches were sparse, as with the three species discussed above, but downstream edges of *Potamogeton* were less ragged and more diffuse. Typically, water velocities within the patch were 10 percent or less of the velocity above the patch, but not as low as with the denser growth forms (Table 5). *Potamogeton* patches were sometimes isolated and sometimes overlapping with other species.

M. sibiricum tended to grow in very diffuse, small patches or as isolated individuals. Patches rarely overlapped other species. *Veronica* that grew as an emergent that tended towards larger, though still thinly distributed, patches. Emergent *Veronica* was often detached from the substrate and had floated downstream to catch in branches or stream edge vegetation. These detached plants were counted as cover when they included whole, relatively undamaged plants. The *M. sibiricum* patch for which velocities are given in Table 5 consisted of only a few plants in a very small patch, which likely explains the higher within-patch and downstream velocities observed.

Table 5. Water velocities above, at the top margin, within, and downstream of several species of aquatic plants. These data were not collected with controls for water velocity, depth, position in cross-section, or plant density. All values in meters per second.

Species	Above	Top Margin	Within	Downstream
<i>Elodea canadensis</i>	0.59	0.46	0.01	very low
<i>Myriophyllum sibiricum</i>	0.55	0.52	0.18	0.42
<i>Potamogeton pectinatus</i>	0.72	0.58	0.03	0.2
<i>Potamogeton pectinatus</i>	0.59	0.38	0.05	0.31
<i>Potamogeton pectinatus</i>	0.71	0.54	0.06	0.34
<i>Ranunculus aquatilis</i>	0.62	0.39	0	very low

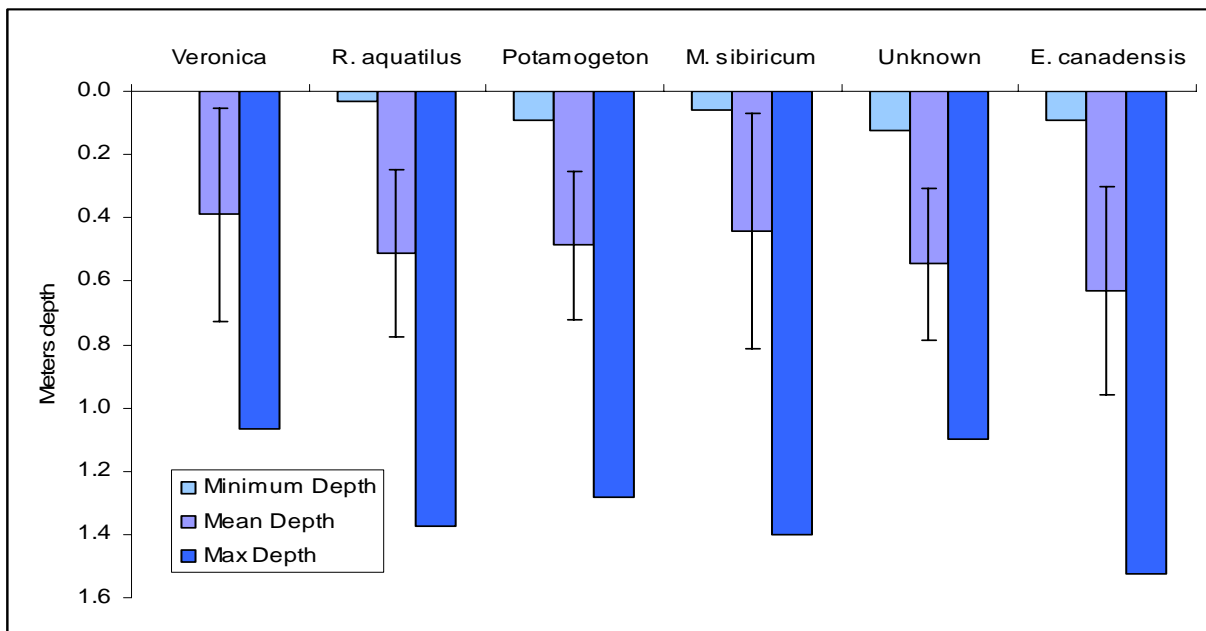


Figure 29. Depth of flow at center of aquatic macrophyte patch for all sites throughout the sampling period. Standard deviation is shown as error bars on the mean depth values.

9.6. Summary

Rapid growth of rooted plants that survive the winter high flows, and rapid colonization of bare ground, occurred during the months of March and April. During that time, *Elodea canadensis* dominated the coverage. Throughout the summer, *Potamogeton pectinatus* increased in coverage, appearing to take over space from some or all of the other species. *Ranunculus aquatilis* and *Elodea canadensis* both stayed relatively constant during the summer. *Myriophyllum sibiricum*, *Veronica*, and the unknown plant all diminished in percent cover during the summer. By October, *Potamogeton* began to dominate the coverage.

This apparent succession has not been examined closely in the Shasta River and specific causes are unknown at this time. One possible scenario is that *E. canadensis* has better establishment characteristics, or recovers from winter scouring sooner than other species. Barrat-Segretain and Bornette (2000) found *E. canadensis* had greater regeneration ability (development of propagules by fragments) than several other species, particularly in the spring.. These other species included

a *Potamogeton* species, *Sparganium emersum*, a *Ranunculus* species, and two other species that are not represented in the Shasta River.

Once a patch of any of the species is established, (i.e. *E. Canadensis* or *R. aquatilis*) sediment accumulates within the patch and in its downstream eddy (Sand-Jensen 1998). This sediment may provide substrate for establishment of plants that are better late-season performers. Downstream accumulation of sediment may also lead to downstream extension of the parent patch; however, more detailed longitudinal measurements may be needed to determine if this occurs in the Shasta River.

Transects were placed in such a way as to maximize coincidence with the fish-survey transects. In retrospect, this may not have been the best way to gather information about the distribution and abundance of aquatic macrophytes at the reach scale, because this resulted in some transects oriented perpendicular to the flow while others were not. Specifically, the AM, PB, and EV habitat type transects ran perpendicular to the flow of the stream, while the CB type transects ran parallel to flow, and the BW type transects ran across a chord of a circular flow pattern (i.e. an eddy). Because patches tend to be longer in the parallel-to-flow direction than in the normal-to-flow direction, and elongate over the growing season, the high abundance of the CB sites (Figure 24) may be an artifact of this sample design.

Flow velocities in macrophyte beds are often dramatically reduced (see Green 2005 for a review and discussion of this topic). Shih and Hughes (2002) conducted flume studies that produced values comparable to those observed during this study. This reduction of flow velocities amounts to a blockage of the stream channel, raising the water level at a given discharge. During the course of this study, it was observed that terraces above the normally wetted channel were inundated for a month or more during the late summer. These were regions vegetated entirely with terrestrial plants. Though this does coincide with the maximum coverage of aquatic plants, it is not possible to determine with the present dataset if the flow restriction caused by aquatic plants was critical to this wetting.

The ecological impact of increased stage to discharge ratio was not examined in this study. However, prolonged wetting of riparian terraces that might be dry without the contribution of aquatic plants may contribute to increased riparian vegetation growth, resulting in increased shade and reduced stream temperatures. Wetted edges may also provide low-velocity habitat for invertebrates and juvenile fish.

9.7. Methodological Modifications for Consideration

Although a considerable amount of information was derived under the current study design and field sampling, several lessons were learned from the effort. One change that would ease comparison of sites would be to include transects normal to flow at all site types. Also, throughout the sampling season, multiple same-day transects at a select number of sites would lend insight as to the reliability of the methodology.

A second consideration for future study would be to assess the impact of aquatic macrophyte coverage on water surface elevation as a function of flow. As noted previously, channel roughness increased considerably with increased macrophyte growth. In order to quantify the

contribution to stage increases of aquatic macrophyte-induced roughness, water surface slope and stage would need to be collected at each site at the time of each sample event.

Another observation of this study was that not all patches have the same thickness or density. There are several reasons to investigate this aspect of patch architecture, including the potential that the use of aquatic plants by stream fauna likely relates to the density of the patch. Additionally, surface observations may miss underlying layers of plant species that are not obvious from above, and patch density and form may affect roughness characteristics. To capture patch architecture, sufficient randomly located vertical transects should be made along each cross-stream transect to examine the vertical structure of the patches. This would provide a more accurate representation of abundance, could lend insight into succession, and allow further between-site and between-site-type comparisons. Due to the early-summer dominance of dense forms and the apparent late-summer dominance of the diffuse *Potamogeton*, a more thorough understanding of patch vertical density and extent will have direct bearing on important habitat variables for invertebrates and the fish that prey upon them.

Patterns of plant distribution should respond to flow velocity, substrate, and light availability (Riis et al. 2001, Chambers et al. 1991, Lacoul and Freedman 2006). Depth may be a useful proxy for light availability, but observations during this study indicate highly variable turbidity in the stream. Measurements of photosynthetically active radiation may be valuable for explaining the distribution of plants in the stream.

In order to understand nutrient cycling in the Shasta River, biomass and growth rates of aquatic plants must be assessed. This will require destructive sampling (harvest), and such sampling will have direct impacts on the site, and indirect impacts on downstream sites. Therefore, harvest should be conducted at sites other than the transect sites.

10. Macroinvertebrate and food web sampling

Food web studies provide a framework for understanding the key interactions that structure ecological communities. In streams and rivers, most organisms consume an array of food items and, in turn, ultimately serve as prey for other organisms. Food webs provide fundamental information concerning which species in a community interact and insights into how such interactions influence the flow of energy, dynamics of populations, and functioning of ecosystems.

We used seasonal macroinvertebrate surveys in conjunction with natural abundance stable isotope analysis to determine food web structure and the important energetic pathways that sustain juvenile salmonids in the Nelson Ranch reach of the Shasta River. Macroinvertebrates were targeted because they represent an ecologically important group of organisms that serve as the primary link between the energetic base of the food web (i.e., organic matter sources such as algae and detritus) and fishes. Moreover, certain macroinvertebrate taxa are known to be extremely sensitive to environmental conditions (e.g., temperature, dissolved oxygen, turbidity, etc.) and community assessments can provide valuable insights into stream health (Barbour et al. 1999, Davis et al. 2001).

Natural abundance stable isotope analysis has been widely applied in ecological systems to elucidate sources of organic matter and the trophic pathways through which this matter is transferred (Peterson and Fry 1987, Michener and Schell 1994, Pinnegar and Polunin 2000). The use of stable carbon isotopes is based upon the observation that the ratio of the heavy to light isotope ($^{13}\text{C}:^{12}\text{C}$; expressed as $\delta^{13}\text{C}$) changes little with each trophic transfer (DeNiro and Epstein 1978, Fry and Sherr 1984). Hence, $\delta^{13}\text{C}$ values are effectively conserved up the food chain and may be used to discriminate between alternative carbon resources when the $\delta^{13}\text{C}$ values of the potential sources are sufficiently distinct.

In contrast with carbon, nitrogen stable isotope ratios ($^{15}\text{N}:^{14}\text{N}$ or $\delta^{15}\text{N}$) increase by approximately 2-4‰ (mean = 3.4‰) with each step in the food chain (see Vander Zanden and Rasmussen 2001, Post 2002). Thus, an organism's $\delta^{15}\text{N}$ signature provides an indirect measure of its relative trophic position and ecological role in the community. Although stable isotope analysis provides less diet resolution than traditional gut content analysis, it addresses some of the limitations associated with diet studies. Stable isotope analysis provides information on those food items that are actually assimilated and converted to consumer biomass rather than those that are simply ingested. Moreover, stable isotope analysis provides time-integrated information on food preferences and is less subject to short-term bias (Creach et al. 1997).

Our specific research objectives at the Nelson Ranch site were to:

1. generate seasonally specific taxonomic lists for the macroinvertebrate community,
2. identify the important sources of organic matter to stream consumers,
3. determine temporal variability in the structure of the aquatic food web, and
4. develop a baseline understanding of the important trophic pathways that support juvenile salmonids.

To that end, macroinvertebrate and stable isotope samples were collected seasonally in 2006 and 2007. Because stable isotopes yield time-integrated dietary information (O'Reilly et al. 2002), we collected samples at the end of each season we sought to characterize. Specific collection dates were 9-10 November (fall), 9-10 March (winter), 7-8 June (spring), and 2-3 September (summer).

10.1. Methods

Macroinvertebrate Sampling

To determine the composition of the macroinvertebrate community and identify potential prey items for juvenile salmonids, we collected representative samples from the benthic environment (kick sampling), the water column (drift sampling), and a variety of additional habitat types (multi-habitat sampling).

Kick Sampling

Macroinvertebrate samples collected for taxonomic determination were comprised of nine individual kick samples. Samples were distributed in a 3 x 3 grid pattern that covered ~100 m² of fairly homogeneous substrate. A standard kick net (500 μm mesh) was placed immediately

downstream of the target sample area and approximately 0.09 m² of the streambed was vigorously disturbed for one minute. The nine individual kick samples were combined in a bucket and the entire sample was elutriated to remove sand, silt, and gravel. The composite sample was subsequently preserved in 95 percent ethyl alcohol and returned to the laboratory for additional processing. A duplicate macroinvertebrate sample was collected for analysis of natural abundance stable isotope ratios (described in detail below). Location of the sampling grid and collection methods remained constant across dates, allowing us to produce taxonomic lists and examine temporal variation in relative abundances.

Drift Sampling

Given the potential importance of invertebrate drift to salmonid production in the Shasta River, we initiated a pilot study during the final two collection periods (i.e., spring and summer) to quantify the relative contribution of terrestrial versus aquatic invertebrates to daytime drift. Three drift nets (363 μ m mesh; mouth opening = 0.12 m²) were positioned ~20 m upstream of the area used to collect benthic macroinvertebrate samples. Drift nets were evenly spaced across the channel, dividing the channel into quarters. To ensure that both animals floating on the water surface and suspended in the water column were sampled, each net was positioned so that the top was ~10 mm above the water surface. On 7 June, drift sampling was initiated at 15:44 h and terminated at 21:12 h (~30 min after sunset). Drift sampling was reinitiated for a second time on 8 June at 05:06 h (~30 min before sunrise) and terminated at 15:43 h. Material collected during the two sample periods was combined for laboratory identification and enumeration. Drift sampling on 2 September was similar, except sampling was completed in one day. Individual drift nets were treated as replicates and sub-sampled to a count of 150 organisms. Drifting animals were categorized as terrestrial or aquatic in origin, enumerated and processed for subsequent biomass determination and stable isotope analysis. All aquatic macroinvertebrates captured in the 2 September drift samples were identified to the lowest practical taxonomic unit.

Multi-habitat Sampling

Supplementary multi-habitat sampling was carried out during the Apr-Jun and Jul-Sep sampling dates. Our aim was to target habitat types not sampled as part of the normal taxonomy sampling. Specifically, the multi-habitat sample was a composite of samples collected from river margins, emergent vegetation, pools, and sandy substrate. Multi-habitat samples were thoroughly searched for additional taxa not found during the course of taxonomic sampling.

Stable Isotope Field Methods

To characterize important carbon sources to the Shasta River food web, we collected four types of organic matter on each sampling date: epilithic biofilms (i.e., matrix of algae, bacteria, fungi, protozoans and non-living organic matter), seston, detritus, and aquatic macrophytes. We randomly selected five stones from the study reach for biofilm sampling. The surface of each stone was scraped with a stiff brush and dislodged material was suspended in a small volume of distilled water. The resultant slurry was collected on pre-combusted (500°C for 4 h) Whatman GF/F filters (47 mm diameter; 0.7 μ m effective pore size) after being passed through a 250 μ m sieve to remove any small invertebrates present.

Seston (suspended fine particular organic matter (FPOM); particles > 0.45 μ m to < 1 mm) was sampled by filtering stream water through pre-combusted GF/F filters until the filters were

lightly colored. Generally, less than 2.5 liters of filtered stream water produced sufficient material for isotopic analyses. Five replicate seston samples were collected on each sampling date. Seston filters were immediately placed in individually labeled opaque bags and cryogenically frozen in the field. Detrital samples (coarse particulate organic matter (CPOM); particles >1 mm in diameter) were handpicked from the streambed and consisted mainly of decomposing aquatic macrophyte fragments, twigs, and conditioned terrestrial leaf litter.

Aquatic macrophytes were collected by hand from various locations in the study reach. Individual samples were selected randomly, but every attempt was made to target all species present during each sampling period. Harvested samples were vigorously agitated in stream water to dislodge clinging invertebrates (epibiota) before being placed in individually labeled polyethylene bags and frozen on dry ice. In the laboratory, samples were briefly thawed and thoroughly examined under 10X magnification to ensure the absence of epibiota that could potentially alter macrophyte stable isotope values. Only aboveground biomass was prepared and submitted for stable isotope analysis.

Aquatic macroinvertebrates for stable isotope analysis were qualitatively sampled using a kick net and by handpicking organisms from the substrate. Samples were then passed through a 500 μm sieve and all retained material was frozen (-80°C) until taxonomic identification and stable isotope preparation. We limited our analysis of macroinvertebrates to those taxa whose relative abundances represented greater than 1 percent of the entire invertebrate assemblage on each date.

Fish samples were obtained from California Department of Fish and Game and consisted of both incidental mortalities associated with rotary screw trap operation at the Nelson Ranch site and intentionally sacrificed fish. Our dependence on contributed samples resulted in different members of the fish community being available for analysis on each sample date. No samples were available for the April-June sampling period. All fish samples were frozen and transported to the laboratory for dissection and processing. Dissection protocols consisted of using a scalpel and forceps to remove muscle tissue from behind the dorsal fin and above the lateral line. All dissections were performed under 10X magnification to insure only dorsal muscle tissue was excised. Entire digestive tracts were concurrently removed and archived (-80°C) for future gut content analysis. Excised muscle samples were placed in pre-ashed (2 h at 400°C) glass scintillation vials and prepared for natural abundance stable isotope analysis as detailed below. Fish species analyzed during the course of this investigation included: Chinook salmon (*Oncorhynchus tshawytscha*), steelhead trout (*O. mykiss*), tui chub (*Gila bicolor*), Klamath smallscale sucker (*Catostomus rimiculus*), speckled dace (*Rhinichthys osculus*), marbled sculpin (*Cottus klamathensis*), and lamprey ammocoetes (*Lampetra* sp.).

Laboratory Methods

Macroinvertebrate Taxonomic Identification

Macroinvertebrate samples for taxonomic determination were repeatedly rinsed in a 500 μm mesh brass sieve to separate animals from silt and debris. All retained material was evenly distributed over a standardized sorting grid and randomly subsampled to reach a minimum count of 500 organisms (± 25 organisms). The balance of the sample was then searched for large and rare taxa (i.e., invertebrate taxa not found in the subsample but present nonetheless). Large and

rare taxa were excluded from subsequent quantitative analyses but included in the taxonomic list generated for each sample period (Appendix).

Aquatic macroinvertebrates were identified to the California Stream Bioassessment Procedure Level II (CSBP-II) standard, using Merritt and Cummins (1996), Smith (2001), and Thorp and Covich (2001) as well as various taxonomic-specific references. Larval Chironomidae were slide mounted using Euparal (BioQuip; Rancho Domingue, CA) and identified to genus when possible using Wiederholm (1983) and Merritt and Cummins (1996). Specimens in poor condition or in very young instars were left at the next highest taxonomic level. Ostracoda and Oligochaeta were identified to class. Taxonomic determinations and associated counts were entered into the California Environmental Data Analysis System (CaEDAS) database, a regional adaptation of software developed by the US EPA for processing macroinvertebrate data. CaEDAS stratifies information on benthic macroinvertebrates and generates a series of descriptive metrics that have known responses to the effects of pollutants or other environmental stressors (Barbour et al. 1999). We selected 10 common metrics that included various measures of taxonomic richness, functional feeding group membership, and organism tolerance values. Tolerance values are a measure of an organism's ability to survive and reproduce in the presence of known levels of stressors (Bressler et al. 2006). Tolerance values range from zero (highly intolerant) to 10 (highly tolerant). Functional feeding group designations are based on how an organism acquires food and include: (1) *collectors*, which gather or filter fine particulate organic matter; (2) *shredders* which consume coarse particulate organic matter; (3) *scrapers* (grazers) which consume epilithic biofilms; (4) *predators*, which capture and feed on other consumers; and (5) *omnivores*, which consume both plant and animal matter. A description of the specific metrics examined in this study is provided in Table 6.

Table 6. *Benthic macroinvertebrate metrics and their expected responses to ecological perturbation.*

Macroinvertebrate Metric	Metric Description	Expected Response to Disturbance
EPT Index	Number of taxa (genus or species) in the orders Ephemeroptera, Plecoptera, and Trichoptera	Decrease
Percent CG + CF Individuals	Percent of the macrobenthos that collect and gather (CG) or filter (CF) fine particulate organic matter	Increase
Percent Chironomidae	Percent of the macrobenthos that belongs to the family Chironomidae	Increase
Percent Predators	Percent of the macrobenthos that capture and consumes other animals	Variable
Percent Scrapers	Percent of the macrobenthos that grazes upon epilithic biofilms (periphyton)	Variable
Percent Tolerant	Percent of all tolerance-rated organisms in a sample that are classified as highly tolerant (tolerance values > 7 out of 10)	Increase
Tolerance Value	Value between 0 and 10 weighted by abundance of individuals with designated tolerances	Increase
Percent Shredders	Percent of the macrobenthos that shreds coarse particulate organic matter	Decrease
Taxonomic Richness	Total number of unique taxa (Genus level taxonomic resolution)	Decrease
Shannon Diversity Index	Measure of sample diversity that incorporates taxonomic richness and evenness	Decrease

Stable Isotope Analyses

Samples for stable C and N isotope analysis were dried at 60°C for ≥ 48 h in a laboratory oven. Dried samples were ground to a fine, homogenous powder using a mortar and pestle and packaged in 5 X 8 mm tin capsules prior to analysis. For samples collected on filters, dried material was transferred from filters when possible otherwise entire filters were ground in a Wig-L-Bug[®] amalgamator (Crescent Dental Corp., Chicago, IL, USA), encapsulated and analyzed. Snail body tissues were excised from their shells to avoid potential carbonate interference. Sample weights were approximately 1.0, 3.0, and 30.0 mg for animals, plants, and filters, respectively. All isotopic analyses were performed at the stable isotope facility at the University of California at Davis using a Europa Scientific Hydra 20/20 isotope ratio mass spectrometer. Analysis of replicate blank GF/F filters indicated that filters contributed negligible amounts of background C and no measurable N. Stable isotope results are presented using the delta (δ) value notation to reflect the ratio of the heavier to lighter isotope and expressed as the per mil (‰) deviation from accepted standards (PeeDee Belemnite limestone for δ¹³C and atmospheric nitrogen for δ¹⁵N) according to the following equation:

$$\delta^{13}\text{C or } \delta^{15}\text{N (‰)} = \left(\frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right) \times 1000$$

where $R = {}^{13}\text{C}:{}^{12}\text{C}$ or ${}^{15}\text{N}:{}^{14}\text{N}$. A more positive δ value (or less negative for carbon) is deemed as isotopically enriched and indicates that the sample contains more of the heavier isotope (e.g., ${}^{13}\text{C}$ or ${}^{15}\text{N}$).

Trophic relationships within the Shasta River food web were inferred using graphical interpretation of carbon and nitrogen stable isotopes in conjunction with stoichiometric data (C:N molar ratios) on potential carbon resources. Plant C:N ratios serve as indicators of food quality (Elser et al. 2000) and empirical studies have shown preferences by herbivorous invertebrate consumers for food items with lower C:N ratios (Burns and Ryder 2001, Menéndez et al. 2001)

Stable nitrogen isotope ratios ($\delta^{15}\text{N}$) were used to estimate the trophic position of fish and invertebrate taxa. Aquatic consumers rarely occupy discrete trophic positions (levels) as portrayed by simple food chain models (e.g., not all primary consumers = trophic position 2, secondary consumers = trophic position 3, etc.). Rather, organisms exist in reticulate food webs that often include complex ecological interactions such as omnivory, cannibalism, and reciprocal predation (Sprules and Bowerman 1998). $\delta^{15}\text{N}$ values are especially informative in that they provide a continuous measure of a consumer's realized (non-discrete) trophic position that integrates the flow of energy and material through multiple pathways (Vander Zanden et al. 1997). We estimated the trophic position of key consumer taxa during each sample period using the following equation:

$$\text{Trophic Position}_{\text{consumer}} = \left(\frac{\delta^{15}\text{N}_{\text{consumer}} - \delta^{15}\text{N}_{\text{baseline}}}{3.4} \right) + 2$$

where $\delta^{15}\text{N}_{\text{consumer}}$ is the $\delta^{15}\text{N}$ of the aquatic consumer for which trophic position is being estimated, $\delta^{15}\text{N}_{\text{baseline}}$ is the mean $\delta^{15}\text{N}$ signature of the baseline organism, the denominator value of 3.4 represents the average trophic fractionation (‰) per trophic level (Minagawa and Wada 1984, Post 2002) and the constant 2 is the expected trophic position of the baseline organism (i.e., primary consumer). We selected the riffle beetle *Optioservus* sp. (Coleoptera: Elmidae) as our baseline indicator in the Shasta River because they demonstrated low $\delta^{15}\text{N}$ values compared to other invertebrate scrapers and were common during all sampling events. *Optioservus* is classified as a scraper in California streams (California Department of Fish and Game 2000) and we assume the bulk of their diet is derived from autochthonous production (i.e., diatoms and green algae). Vander Zanden and Rasmussen (2001) and Anderson and Cabana (2007) provide detailed discussions on the use of baseline $\delta^{15}\text{N}$ values to estimate trophic position in aquatic food webs.

10.2. Macroinvertebrate Findings

Macroinvertebrate Survey

Our macroinvertebrate sampling was designed to generate four seasonally specific taxonomic lists. Due to cost constraints and the nature of the study goals, all sampling events are unreplicated. Therefore, drawing statistically significant conclusions beyond taxa presence is impossible. However, several interesting observations can be made based on the data and are discussed below.

The most striking seasonal change in community composition occurred in the tolerant Dipteran family Chironomidae (non-biting midges; family-wide tolerance value of 6). While virtually absent during three of the sampling periods, chironomid abundance jumped to 22.8 percent in Apr-Jun (chironomid abundance for Oct-Nov, Dec-Mar, and Jul-Sep was 1.4, 1.4, and 2.3 percent, respectively; Table 7). The marked increase in chironomids was driven by two genera in particular: the collector-gatherer *Cricotopus* sp. (tolerance value of 7) and the collector-filterer *Rheotanytarsus* sp. (tolerance value of 6; CDFG 2000).

Table 7. Seasonal comparison of macroinvertebrate metrics calculated for the Nelson Ranch reach of the Shasta River. Individual metrics are defined in Table 6.

Macroinvertebrate Metric	Sampling Period			
	Oct-Nov	Dec-Mar	Apr-Jun	Jul-Sep
EPT Index (%)	28%	68%	54%	25%
Percent CG + CF Individuals	68%	77%	84%	81%
Percent Chironomidae	1%	1%	23%	2%
Percent Predators	3%	3%	3%	2%
Percent Scrapers	28%	20%	10%	14%
Percent Tolerant	52%	5%	15%	48%
Tolerance Value	6.09	4.82	5.41	6.26
Percent Shredders	1%	0%	0%	0%
Taxonomic Richness	32	28	42	33
Shannon Diversity Index	2.58	1.70	2.82	2.45

It is important to note that high densities of *Cricotopus* and *Rheotanytarsus* are often associated with stressed ecosystems (CDFG 2000). Consequently, the large increase in their relative abundances may be indicative of a significant temporal change in stream health. One possible explanation for this increase is that high water temperatures caused an ecological perturbation, resulting in a rise in chironomid abundance. It is also possible that, by chance, the Apr-Jun sample included more patches of midges than the other three samples. However, we believe this explanation unlikely because the sample was a composite and the same location was sampled during all four sampling events. Nevertheless, without sufficient replication this possibility cannot be ruled out. A third explanation is that chironomid abundance naturally increases in the spring, independent of changes in environmental conditions. One way to parse out natural cycles of abundance versus changes in water quality would be to establish and sample one or more reference sites in conjunction with the Nelson Ranch site. In this way, natural variability in abundance could be factored out from shifts in abundance due to changing local conditions. If a suitable reference site can be established for the upstream Nelson Ranch site, the hypothesis that the aquatic community is being adversely impacted by warm water inputs could be empirically tested. Low water temperatures combined with low midge abundances at a reference site would provide strong evidence that the Nelson Ranch is highly influenced by warm water inputs from Big Springs.

Ecologists often consider high taxonomic richness to indicate high ecological integrity (and water quality). Therefore, ostensibly it appears that ecological integrity improved somewhat in Apr-Jun where taxonomic richness was notably higher than other seasons (Table 7). However,

one result of the dramatic increase in midge abundance during Apr-Jun was that the largest taxonomic richness value also occurred during this sampling period (Richness = 42 unique macroinvertebrate taxa). The elevated richness metric was largely driven by the fact that 20 different chironomid taxa were identified during this period. By comparison, the other three sampling periods only averaged 4.3 unique chironomid taxa each. Thus, the increase in taxonomic richness during the April-June period resulted from the addition of tolerant organisms to the community and may not signify higher ecological integrity.

Another interesting aspect of the macroinvertebrate data was the almost complete absence of Plecoptera (stoneflies) from the Nelson Ranch invertebrate community. This finding is especially notable because previous macroinvertebrate surveys conducted within the basin (e.g., DWR unpublished data, Great Northern Corporation 1999) reported the presence of multiple Plecopteran families. Stoneflies are regarded as a highly sensitive order of aquatic insects that require cold, well-oxygenated water with low turbidity, and stable substrates (Merritt and Cummins 1996). Because stoneflies were infrequently encountered during our sampling efforts, we believe that they are indeed rare at the Nelson Ranch site. However, as with chironomids, abundance data would be much more robust with the addition of reference sites.

Collector-gatherer insects dominated the macroinvertebrate assemblage at the Nelson Ranch site. In fact, during each sample period, the top three numerically dominant macroinvertebrates all belonged to the collector-gatherer feeding guild (Table 8). Both the Jul-Sep and Oct-Nov sampling dates were dominated by the scud *Hyaella* sp. (Amphipoda: Hyalellidae), while Dec-Mar and Apr-Jun were dominated by the mayfly *Baetis* spp. (Ephemeroptera: Baetidae). Although invertebrate predators were rare in the riffle habitats targeted by our taxonomic sampling, they were considerably more abundant in the multi-habitat samples. While multi-habitat samples precluded quantitative analysis, we observed that they were typically dominated by pool dwelling predators (i.e., Corixidae and Odonata) (see taxonomic list Appendix).

Table 8. The three most dominant macroinvertebrate taxa and their proportion of the total sample during each sample period. A complete list of taxa is presented in Appendix.

Oct- Nov		Dec- Mar		Apr- Jun		Jul- Sep	
<i>Hyaella</i> sp.	23%	<i>Baetis</i> spp.	59%	<i>Baetis</i> spp.	31%	<i>Hyaella</i> sp.	31%
Ostracoda	15%	Oligochaeta	10%	<i>Hyaella</i> sp.	11%	Oligochaeta	11%
<i>Baetis</i> spp.	14%	<i>Optioservus</i> sp.	10%	<i>Diphetor hageni</i>	8%	<i>Baetis</i> spp.	9%
Total	52%		80%		50%		51%

Invertebrate Drift

The Shasta River has very little woody and herbaceous riparian habitat upstream of our study site and hence limited potential for inputs of terrestrial invertebrates. Unsurprisingly, aquatic organisms dominated daytime invertebrate drift, both numerically and in terms of biomass (Table 9, Figure 30). Mean daytime invertebrate drift density was approximately 3.4 times greater during the Apr-Jun sampling event than in Jul-Sep (Table 9) suggesting a general decrease in food supply through the summer months. Taxonomic examination of the Jul-Sep drift samples revealed that drift was comprised mainly of Chironomidae (not identified past family), *Simulium* sp. (Diptera: Simuliidae), Baetidae, and *Hyaella* sp. (in order of relative dominance).

Table 9. Density and biomass of daytime invertebrate drift samples collected during the Spring (Apr-Jun) and Summer (Jul-Sep).

Drift Metric	Season	
	Apr-Jun	Jul-Sep
Total Invertebrates · 100 m ⁻³	63.77	17.22
Aquatic Inverts · 100 m ⁻³	58.67 (92%)	15.50 (90%)
Dry Mass Aquatic (mg · 100 m ⁻³)	7.10 (98%)	2.08 (97%)
Terrestrial Inverts · 100 m ⁻³	5.10 (8%)	1.72 (10%)
Dry Mass Terrestrial (mg · 100 m ⁻³)	0.17 (2%)	0.06 (3%)

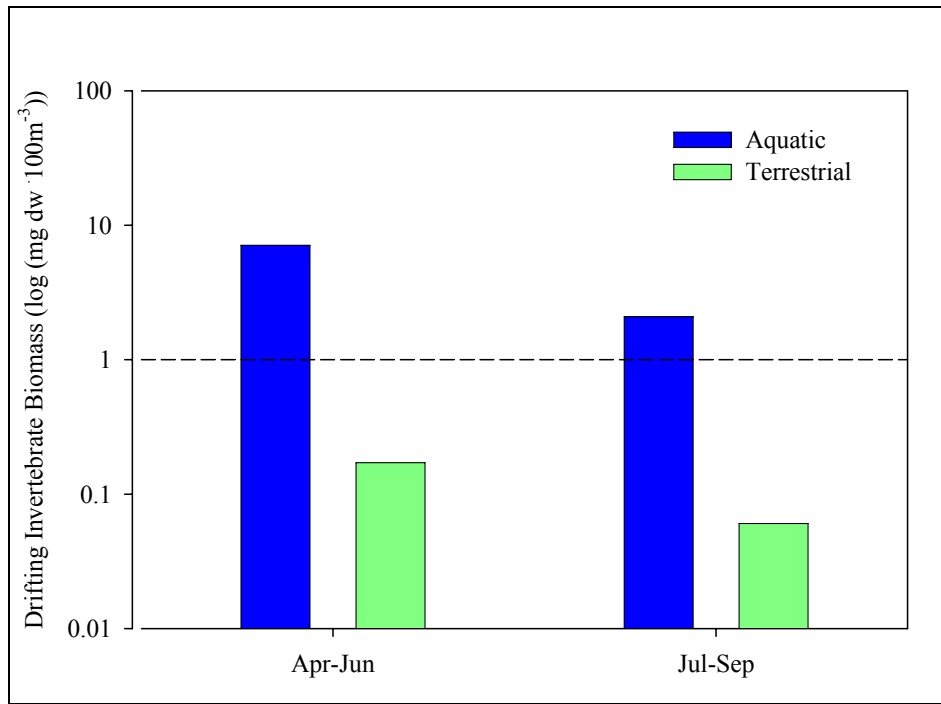


Figure 30. Drifting invertebrate biomass (mg dry weight · 100 m⁻³) for aquatic and terrestrial insects in the Shasta River. Drift biomass consisted mainly of larval aquatic insects. Note log scale.

10.3. Stable Isotope Findings

As outlined previously, carbon isotope values ($\delta^{13}\text{C}$) provide information on the primary energy sources that fuel aquatic production, while nitrogen isotopes ($\delta^{15}\text{N}$) allow discrimination among trophic levels. Furthermore, C:N molar ratios serve as measures of food quality with high C:N ratios indicating nutritionally poor (refractory) food resources. Collectively, seasonal variation in the C and N stable isotope ratios of organic matter (epilithic biofilms, seston, detritus, and macrophytes), macroinvertebrates, and fish provide critical insight into the structure of the food web and the energetic pathways that support anadromous salmonids.

In the following section we present and discuss the results of our seasonal stable isotope analyses. For each season, we begin our discussion by examining the energetic base of the food

web and move to progressively higher trophic levels (i.e., from sources of organic matter to macroinvertebrates to fish). Plots of $\delta^{15}\text{N}$ versus $\delta^{13}\text{C}$ are presented to help illustrate the flow of nutrients from sources to consumers. For visual clarity, unique numbers have been used in place of names to identify the isotopic position of the various food web components in all dual isotope plots (i.e., Figures 26, 28, 30 and 32). A key to the numbering convention is provided in Table 10.

Table 10. Key to the numbering convention used in C and N stable isotope diagrams (i.e., Figures 26, 28, 30, and 32). We limited our analysis of benthic macroinvertebrates to those taxa whose relative abundances represented greater than 1% of the entire invertebrate assemblage on each sample date.

No.	Food Web Component	No.	Food Web Component	No.	Food Web Component
Primary Carbon Sources		Macroinvertebrates Cont.		Macroinvertebrates Cont.	
1	Detritus [†]		Collector-Filterers		Predators
2	Epilithic Biofilms	14	<i>Simulium</i> sp.	24	<i>Hetaerina americana</i>
3	Macrophytes [‡]	15	Sphaeriidae	25	<i>Ophiogomphus</i> sp.
4	Seston		Scrapers		Drifting Invertebrates
	Macroinvertebrates	16	<i>Gyraulus</i> sp.	26	Aquatic Origin
	Collector-Gatherers	17	<i>Juga</i> sp.	27	Terrestrial Origin
5	<i>Amiocentrus aspilus</i>	18	<i>Optioservus</i> sp.		Fishes
6	<i>Baetis</i> spp.	19	<i>Physa</i> sp.	28	Chinook Salmon
7	<i>Dipheter hageni</i>	20	<i>Protoptila</i> sp.	29	Marbled Sculpin
8	<i>Gammarus</i> sp.	21	<i>Rhithrogena</i> sp.	30	Lamprey ammocoete
9	<i>Hexagenia limbata</i>	22	<i>Vorticifex</i> sp.	31	Speckled Dace
10	<i>Hyalella</i> sp.		Omnivores	32	Steelhead Trout
11	Oligochaeta	23	<i>Brachycentrus</i> sp.	33	Klamath Smallscale Sucker
12	<i>Paraleptophlebia</i> sp.			34	Tui Chub
13	<i>Tricorythodes</i> sp.				

[†]Detrital samples typically contained a mixture of conditioned organic matter derived from both aquatic (autochthonous) and terrestrial (allochthonous) sources.

[‡]Macrophyte samples consisted of multiple plant species.

Fall: October – November

The $\delta^{13}\text{C}$ of detritus ($-28.97 \pm 0.90\text{‰}$) differed significantly from that of epilithic biofilms ($-25.36 \pm 0.81\text{‰}$) during the Oct-Nov sample period ($p = 0.02$; Table 11). The detrital pool was comprised mainly of macrophyte fragments with small amounts of conditioned terrestrial leaf litter. The mean C:N ratio of detritus was both high (57.5) and extremely variable (1 SE = ± 16.28 ; range = 8.75 to 75.63; Figure 31). The high C:N ratio of detritus suggests that much of the coarse particulate organic matter available to macroinvertebrate consumers was of very low nutritive quality. High C:N ratios may result from extremely recalcitrant starting materials (e.g., macrophytes with high lignin or cellulose content) or plant matter that is in early stages of decomposition. A decrease in the C:N ratio is typically associated with colonization by heterotrophic organisms which add particulate nitrogen to the detrital pool (Thornton and McManus 1994, Pagioro and Thomaz 1999). Sheldon and Walker (1997) report that macroinvertebrate consumers preferentially select food resources with C:N ratios below 10, and that the maximum C:N ratio for maintaining the growth of primary consumers is approximately

17. Hence, a mean detrital C:N ratio of 57.53 suggests that much of the detritus during the late fall may be of insufficient quality to contribute to carbon flow through the Nelson Ranch aquatic food web.

Seston had a mean $\delta^{13}\text{C}$ value of $-28.23 \pm 0.17\text{‰}$ (Table 11). While mean seston $\delta^{13}\text{C}$ was similar to the mean $\delta^{13}\text{C}$ of detritus, the mean C:N ratio of seston was significantly lower (Figure 31). In general, most primary carbon sources had similar $\delta^{15}\text{N}$ values of approximately 5.0‰. However, seston samples were uniquely enriched with ^{15}N relative to other primary carbon sources ($\delta^{15}\text{N} = 7.17 \pm 3.16\text{‰}$) and the variability in seston $\delta^{15}\text{N}$ was exceptionally high (range = -0.62 to 18.18‰).

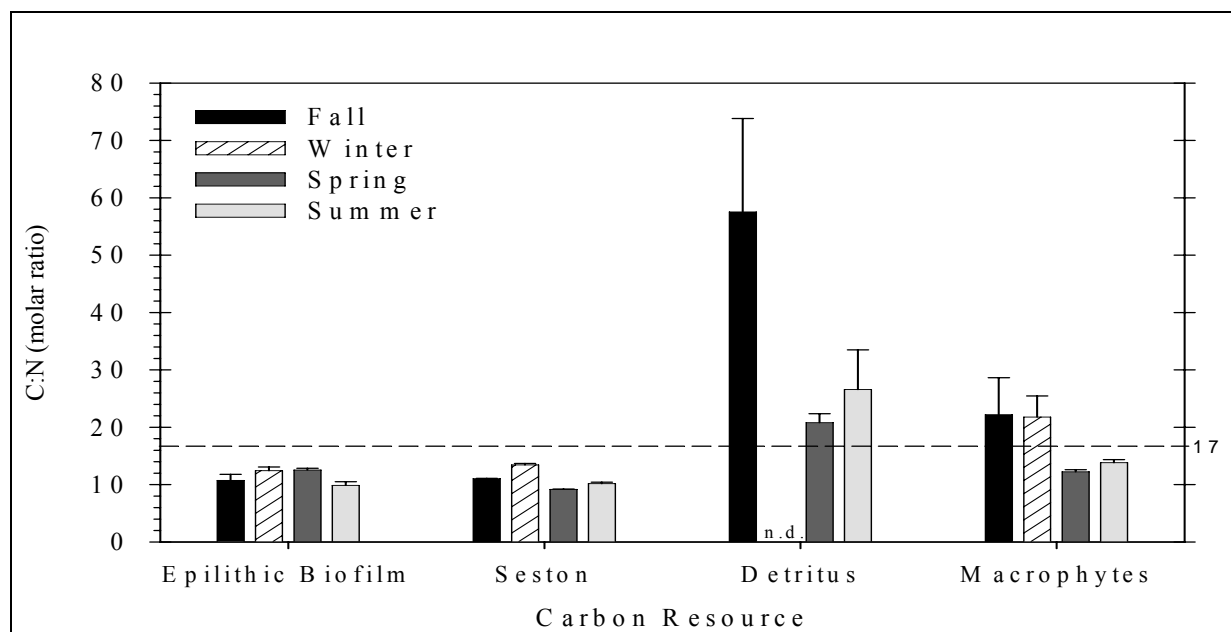


Figure 31. Seasonal changes in C:N ratios (molar) of select basal (primary) carbon resources in the Shasta River. Data are presented as means \pm 1 standard error. The dashed line represents a C:N of 17, a reported maximum ratio for maintaining the growth of macroinvertebrate consumers (see Sheldon and Walker 1997). n.d. = no data collected.

The majority of benthic macroinvertebrate $\delta^{13}\text{C}$ values fell between the mean values for detritus and periphyton (Table 11, Figure 32). This suggests that invertebrate production during the fall was fueled chiefly by autochthonous production. Mean macroinvertebrate $\delta^{13}\text{C}$ values ranged from -29.44‰ for the omnivorous caddisfly *Brachycentrus* sp. (Trichoptera: Brachycentridae) to -24.63‰ for the herbivorous snail *Gyraulus* sp. (Gastropoda: Planorbidae). $\delta^{15}\text{N}$ values ranged from 5.52‰ for *Hyaella* sp., the numerically dominant organism during the fall (Table 8) to 13.02‰ for oligochaete worms. Most invertebrates occupied the trophic role of primary consumers and had mean trophic positions between 1.9 and 2.3 (Figure 33). Larval dragonflies of the genus *Ophiogomphus* (Odonata: Gomphidae) had a mean trophic position of 2.9 reflecting their ecological role as invertebrate predators (secondary consumers) in the food web. The extremely elevated $\delta^{15}\text{N}$ signature of oligochaetes (Table 11) resulted in a high trophic position (TP) estimate (TP > 4.0) relative to the rest of the invertebrate assemblage (Figure 33).

Most fish taxa had very similar carbon and nitrogen isotope values (Figure 32). Fish $\delta^{13}\text{C}$ values generally fell within $\sim 1.0\%$ of the mean $\delta^{13}\text{C}$ value for the entire macroinvertebrate assemblage. This suggests that fish are feeding principally on the benthic macroinvertebrate community and that most of the carbon that fuels fish production is of autochthonous origin. However, Steelhead trout (mean FL = 108.5mm) were a notable exception as they had the most isotopically depleted mean $\delta^{13}\text{C}$ value of all the fishes analyzed (Table 11, Figure 32). This unique $\delta^{13}\text{C}$ signature suggests that a considerable portion of the carbon ultimately assimilated by juvenile steelhead was derived from either non-aquatic sources or some carbon source that we failed to characterize. It is possible that allochthonous inputs in the form of terrestrial insects represent an important component of the diet of juvenile steelhead at the Nelson Ranch site during the fall.

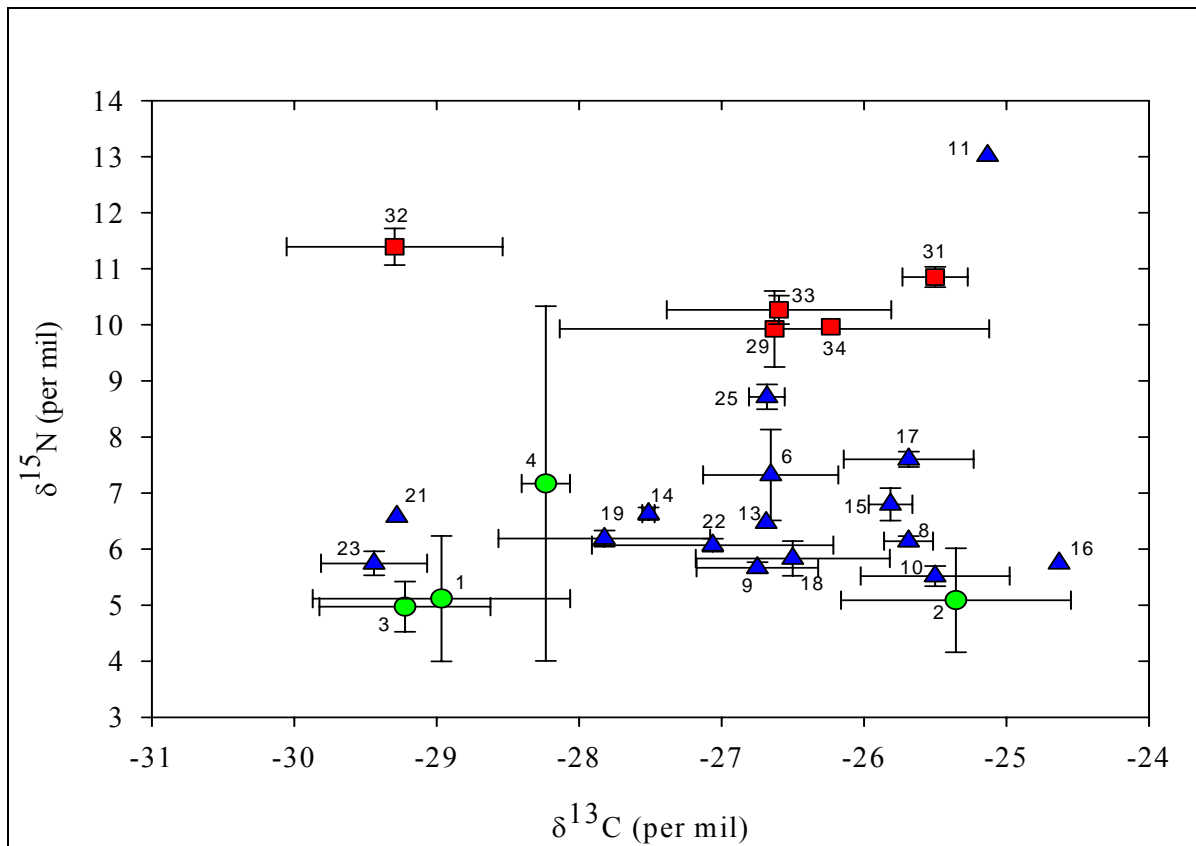


Figure 32. Stable carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) isotope ratios for key members of the Shasta River aquatic food web during Oct-Nov 2006. Circles designate basal carbon resources, triangles represent macroinvertebrates, and squares signify fish taxa. Data are presented as mean values ± 1 SE. A key to numerical codes is provided in Table 10.

Table 11. Food web components analyzed for C and N stable isotope analysis. Delta (δ) values reflect the ratio of the heavier to lighter isotopes (i.e., $^{13}\text{C}:^{12}\text{C}$ and $^{15}\text{N}:^{14}\text{N}$) and are expressed as the per mil (‰) deviation from the standards Pee Dee Belemnite and atmospheric N_2 for C and N, respectively. Values for each food web component are presented as the mean \pm 1 SE. n.d. = no data collected.

Food Web Component	Oct-Nov 2006		Dec 2006- Mar 2007		Apr -Jun 2007		Jul-Sept 2007	
	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)
Primary Carbon Sources								
Detritus [†]	-28.97 \pm 0.90	5.12 \pm 1.12	n.d.	n.d.	-28.15 \pm 0.17	5.59 \pm 0.03	-26.44 \pm 2.00	7.53 \pm 0.34
Epilithic Biofilm	-25.36 \pm 0.81	5.09 \pm 0.93	-27.98 \pm 0.82	5.06 \pm 0.39	-26.55 \pm 1.29	4.14 \pm 0.14	-26.72 \pm 0.61	4.55 \pm 0.27
Macrophytes [†]	-29.22 \pm 0.60	4.97 \pm 0.45	-26.63 \pm 0.59	7.18 \pm 0.20	-24.34 \pm 0.43	5.69 \pm 0.26	-25.06 \pm 0.62	5.39 \pm 0.27
Seston	-28.23 \pm 0.17	7.17 \pm 3.16	-28.94 \pm 0.08	5.15 \pm 1.34	-27.42 \pm 0.03	4.36 \pm 0.24	-26.40 \pm 0.14	5.46 \pm 0.37
Macroinvertebrates								
Collector-Gatherers								
<i>Amiocentrus aspilus</i>	n.d.	n.d.	-33.67 \pm 0.57	3.73 \pm 0.28	n.d.	n.d.	n.d.	n.d.
<i>Baetis</i> sp.	-26.66 \pm 0.47	7.32 \pm 0.81	-31.12 \pm 0.21	7.09 \pm 0.11	-26.42 \pm 0.05	6.20 \pm 0.03	-24.67 \pm 0.26	5.78 \pm 0.12
<i>Diphetera hageni</i>	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	-24.94 \pm 0.23	6.24 \pm 0.08
<i>Gammarus</i> sp.	-25.69 \pm 0.17	6.14 \pm 0.09	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
<i>Hexagenia limbata</i>	-26.75 \pm 0.47	5.67 \pm 0.11	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
<i>Hyalella</i> sp.	-25.50 \pm 0.52	5.52 \pm 0.18	-27.17 \pm 0.29	5.61 \pm 0.05	-24.29 \pm 0.05	5.38 \pm 0.06	-22.57 \pm 0.06	5.43 \pm 0.04
Oligochaeta	-25.13	13.02	-27.89	7.78	n.d.	n.d.	-25.22 \pm 0.19	6.27 \pm 0.12
<i>Paraleptophlebia</i> sp.	n.d.	n.d.	n.d.	n.d.	-26.87 \pm 0.11	5.89 \pm 0.04	n.d.	n.d.
<i>Tricorythodes</i> sp.	-26.69	6.47	n.d.	n.d.	-26.43 \pm 0.06	5.40 \pm 0.03	-25.17 \pm 0.13	4.82 \pm 0.48
Collector-Filterers								
<i>Simulium</i> sp.	-27.51 \pm 0.04	6.63 \pm 0.11	n.d.	n.d.	-26.71 \pm 0.11	5.81 \pm 0.03	-24.84 \pm 0.14	6.05 \pm 0.06
Sphaeriidae	-25.81 \pm 0.15	6.80 \pm 0.29	-26.19 \pm 0.14	5.71 \pm 0.17	n.d.	n.d.	-25.01	5.79
Scrapers								
<i>Gyraulus</i> sp.	-24.63	5.75	n.d.	n.d.	n.d.	n.d.	-24.75 \pm 0.16	5.52 \pm 0.09
<i>Juga</i> sp.	-25.69 \pm 0.46	7.60 \pm 0.14	-25.00 \pm 0.33	6.96 \pm 0.23	-22.86 \pm 1.48	7.87 \pm 0.25	-23.92 \pm 0.06	8.27 \pm 0.13
<i>Optioservus</i> sp.	-26.50 \pm 0.68	5.83 \pm 0.31	-26.82 \pm 0.35	5.12 \pm 0.13	-26.94 \pm 0.21	5.59 \pm 0.04	-26.31 \pm 0.17	5.25 \pm 0.10
<i>Physa</i> sp.	-27.82 \pm 0.74	6.19 \pm 0.14	n.d.	n.d.	n.d.	n.d.	-25.56 \pm 0.23	6.19 \pm 0.04
<i>Protophila</i> sp.	n.d.	n.d.	-31.37 \pm 0.09	6.92 \pm 0.37	-28.61 \pm 0.19	6.20 \pm 0.08	-30.39 \pm 0.10	5.85 \pm 0.00
<i>Rhithrogena</i> sp.	-29.28	6.58	-33.22 \pm 0.18	6.67 \pm 0.04	-31.36 \pm 0.11	5.85 \pm 0.07	n.d.	n.d.
<i>Vorticifex</i> sp.	-27.06 \pm 0.85	6.07 \pm 0.12	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
Omnivores								
<i>Brachycentrus</i> sp.	-29.44 \pm 0.37	5.75 \pm 0.22	n.d.	n.d.	-28.72 \pm 0.29	6.08 \pm 0.07	-25.63 \pm 0.16	6.38 \pm 0.06
Predators								
<i>Hetaerina americana</i>	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	-24.73 \pm 0.10	7.45 \pm 0.07
<i>Ophiogomphus</i> sp.	-26.68 \pm 0.13	8.72 \pm 0.22	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
Drift (Aquatic Insects)	n.d.	n.d.	n.d.	n.d.	-28.20 \pm 0.02	5.68 \pm 0.03	-25.57 \pm 0.15	6.41 \pm 0.41
Drift (Terrestrial Insects)	n.d.	n.d.	n.d.	n.d.	-27.11 \pm 0.55	5.81 \pm 1.12	-25.99 \pm 0.08	7.89 \pm 0.26
Fishes								
Chinook Salmon	n.d.	n.d.	-26.20 \pm 1.63	9.87 \pm 0.49	n.d.	n.d.	n.d.	n.d.
Marbled Sculpin	-26.63 \pm 1.51	9.93 \pm 0.68	n.d.	n.d.	n.d.	n.d.	-23.96 \pm 0.62	9.22 \pm 0.27
Lamprey ammocoete	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	-23.16	5.38
Speckled Dace	-25.50 \pm 0.14	10.85 \pm 0.20	-27.33 \pm 0.66	10.57 \pm 0.25	n.d.	n.d.	-24.64 \pm 0.48	10.24 \pm 0.36
Steelhead Trout	-30.06 \pm 0.90	11.23 \pm 0.59	-19.43	16.65	n.d.	n.d.	-26.03 \pm 1.40	10.61 \pm 0.69
Klamath smallscale sucker	-26.23	9.97	n.d.	n.d.	n.d.	n.d.	-24.46	9.44
Tui Chub	-26.60 \pm 0.96	10.27 \pm 0.31	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.

Trophic position estimates based on stable nitrogen isotope ratios indicate that steelhead trout occupy the highest trophic position among the fishes examined (mean trophic position = 3.6, Figure 33). In general, all fish had mean trophic position estimates of ≥ 3.2 , with marbled sculpin demonstrating the lowest mean trophic position (3.2) and the most variability among individuals (95 percent confidence interval = 2.3 to 4.1, $N=3$; Figure 33).

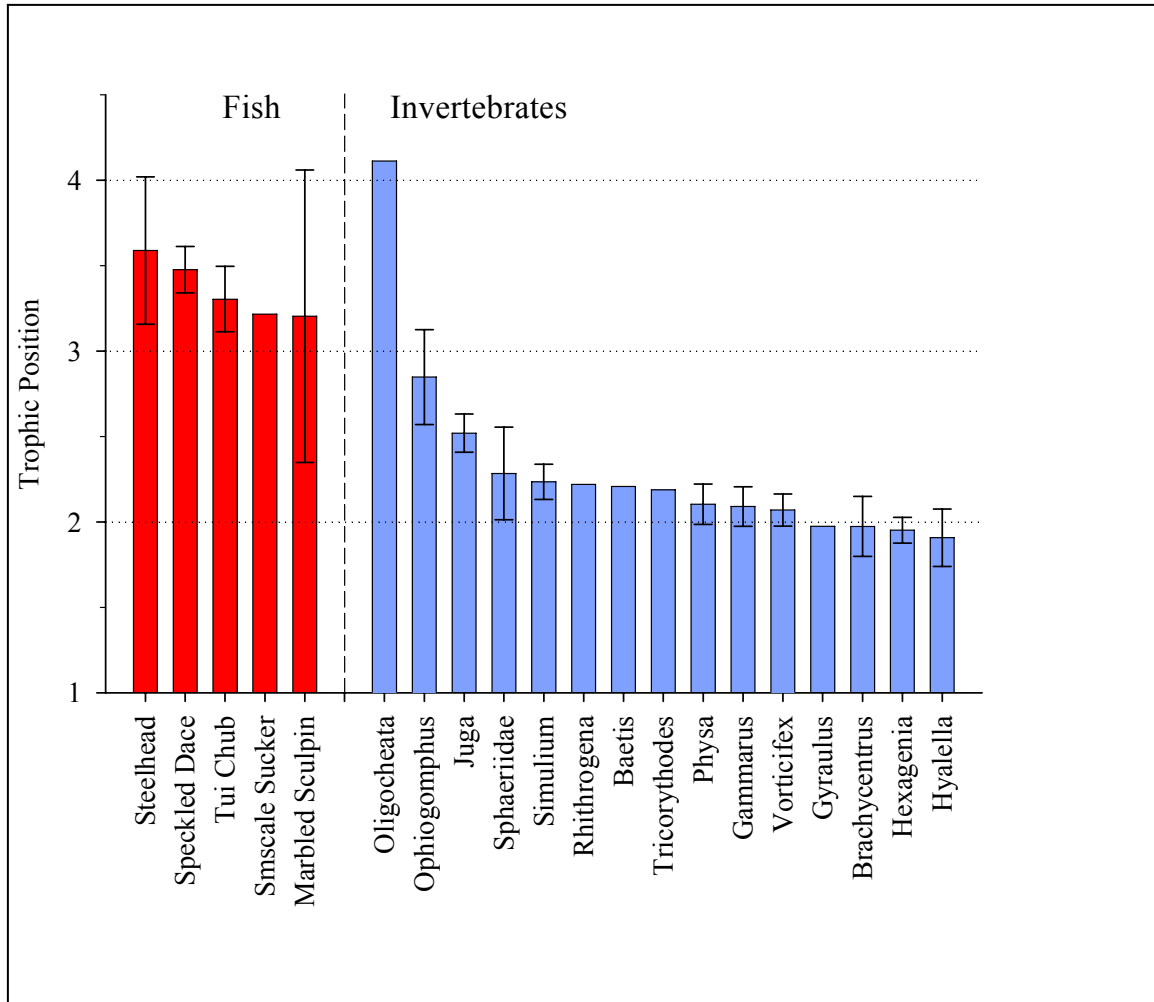


Figure 33. Mean trophic position estimates for aquatic consumers in the Shasta River during the fall 2006 sample period. Error bars indicate 95% confidence intervals.

Winter: December – March

The mean $\delta^{13}\text{C}$ signature of epilithic biofilm ($-27.98 \pm 0.82\text{‰}$) was intermediate between that of seston ($-28.94 \pm 0.08\text{‰}$) and aquatic macrophytes ($-26.63 \pm 0.59\text{‰}$; Table 11). However, individual biofilm samples were highly variable ranging from -30.40 to -25.64‰ . Macrophyte abundance was greatly reduced during this sampling period as much of the plant biomass had earlier senesced and been entrained during elevated flow events. Remaining macrophytes were significantly enriched in both ^{13}C ($+2.59\text{‰}$) and ^{15}N ($+2.21\text{‰}$) relative to the previous sampling period (Table 11).

Seston and epilithic biofilms had respective mean C:N ratios of 13.5 and 12.4, while macrophyte C:N was 21.8 (Figure 31). Mean $\delta^{15}\text{N}$ values of epilithic biofilm and seston were $5.06 \pm 0.39\text{‰}$ and $5.15 \pm 1.34\text{‰}$, respectively. The $\delta^{15}\text{N}$ signature of macrophytes ($7.18 \pm 0.20\text{‰}$) was more ^{15}N -enriched than values for any of the primary consumers in the food web except oligochaete worms (Table 11, Figure 34). The elevated mean $\delta^{15}\text{N}$ of macrophytes, coupled with their high C:N ratios (i.e., low nutritive value), suggests that live plants were not being directly utilized as a food source by primary consumers. Although some macroinvertebrate taxa have been reported to graze on live macrophytes (Berg 1949, Gower 1967, Suren and Lake 1989) direct consumption is thought to be fairly uncommon in lotic ecosystems (Mann 1988). Rather, live macrophytes principally contribute to carbon flow in food webs by serving as substrata for epiphytic biofilms or as refugia from predators (France 1995).

Aquatic macroinvertebrates demonstrated a wide range in $\delta^{13}\text{C}$ values (Table 11, Figure 34). Four invertebrate taxa had mean $\delta^{13}\text{C}$ values that were depleted (more negative) relative to the mean $\delta^{13}\text{C}$ of the potential basal carbon sources analyzed: *Amiocentrus aspilus* (Trichoptera: Brachycentridae), *Rhithrogena* sp. (Ephemeroptera: Heptageniidae), *Protophila* sp. (Trichoptera: Glossosomatidae), and *Baetis* spp. (Ephemeroptera: Baetidae; Figure 34). However, considering the large variability in epilithic biofilm $\delta^{13}\text{C}$, it is probable that these consumers were utilizing the epilithon as their primary carbon source. Moreover, because *Baetis* was the numerically dominant macroinvertebrate during this sample period (representing >59 percent of the assemblage; Table 8), epilithic biofilms likely supported a significant portion of the trophic productivity during the winter months.

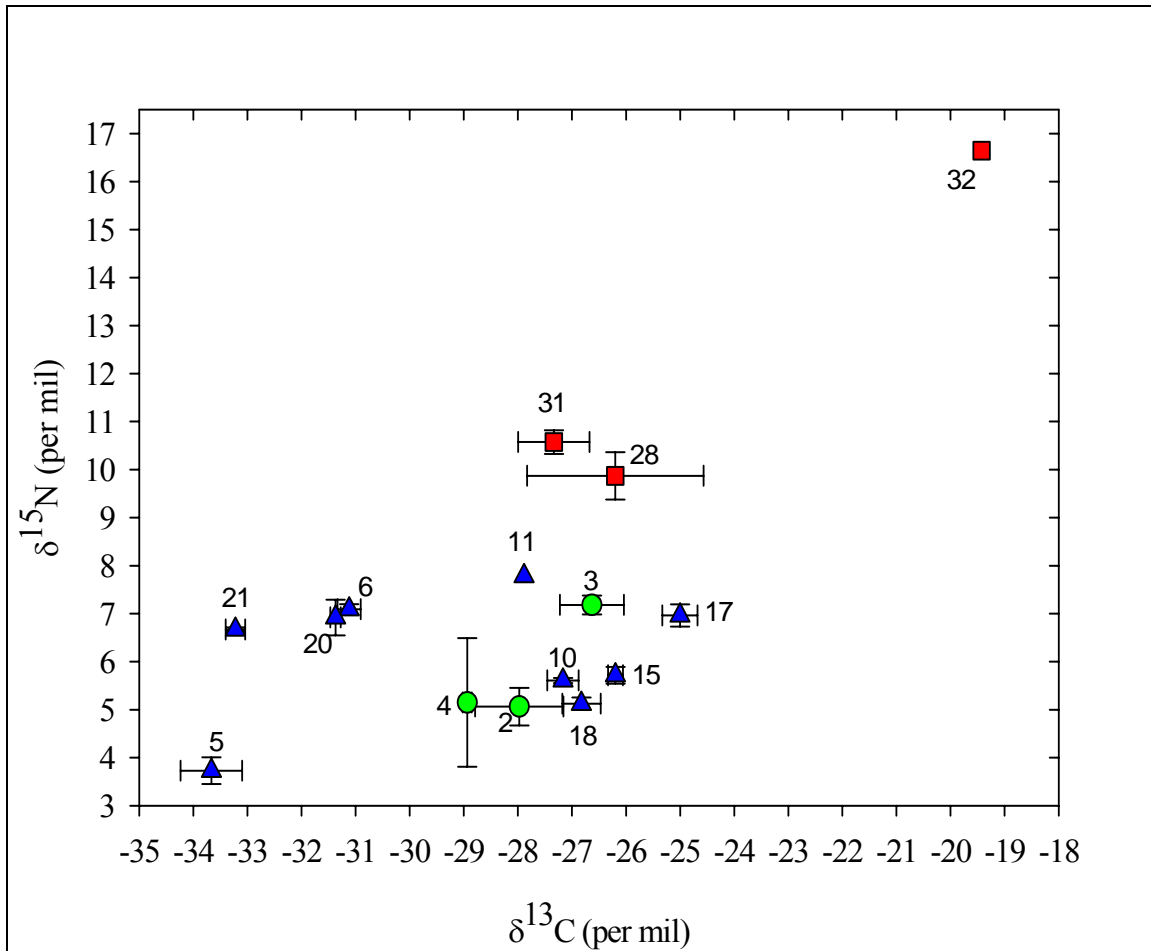


Figure 34. Stable carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) isotope ratios for key members of the Shasta River aquatic food web during the Dec 2006-Mar 2007 sampling period. Circles designate basal carbon resources, triangles represent macroinvertebrates, and squares signify fish taxa. All data are presented as the mean \pm 1 SE. Food web numerical codes are provided in Table 10.

The two most ^{13}C -enriched invertebrates were sphaeriid clams ($-26.19 \pm 0.14\text{‰}$) and the pleurocerid snail *Juga* sp. ($-25.00 \pm 0.33\text{‰}$). Sphaeriids are filter-feeding bivalves and have been reported to feed on both suspended and deposited organic matter (Hornbach et al. 1984). However, sphaeriid $\delta^{13}\text{C}$ signatures in the Shasta River did not match up well with the mean carbon signature for seston (Figure 34, Table 11). While this does not preclude the utilization and assimilation of seston by sphaeriids, it suggests that they are most likely deposit feeding and obtaining a significant portion of their carbon from fine particulate organic matter (FPOM) on the sediment surface. Detrital material typically becomes enriched with ^{13}C as it is processed and converted to increasingly smaller sized particles (i.e., from CPOM to FPOM). Although the mechanism behind this change in $\delta^{13}\text{C}$ during decomposition is unresolved (Finlay 2001) the alteration is both predictable and useful for inferring the exploitation of different particulate carbon sources (McNeely et al. 2006).

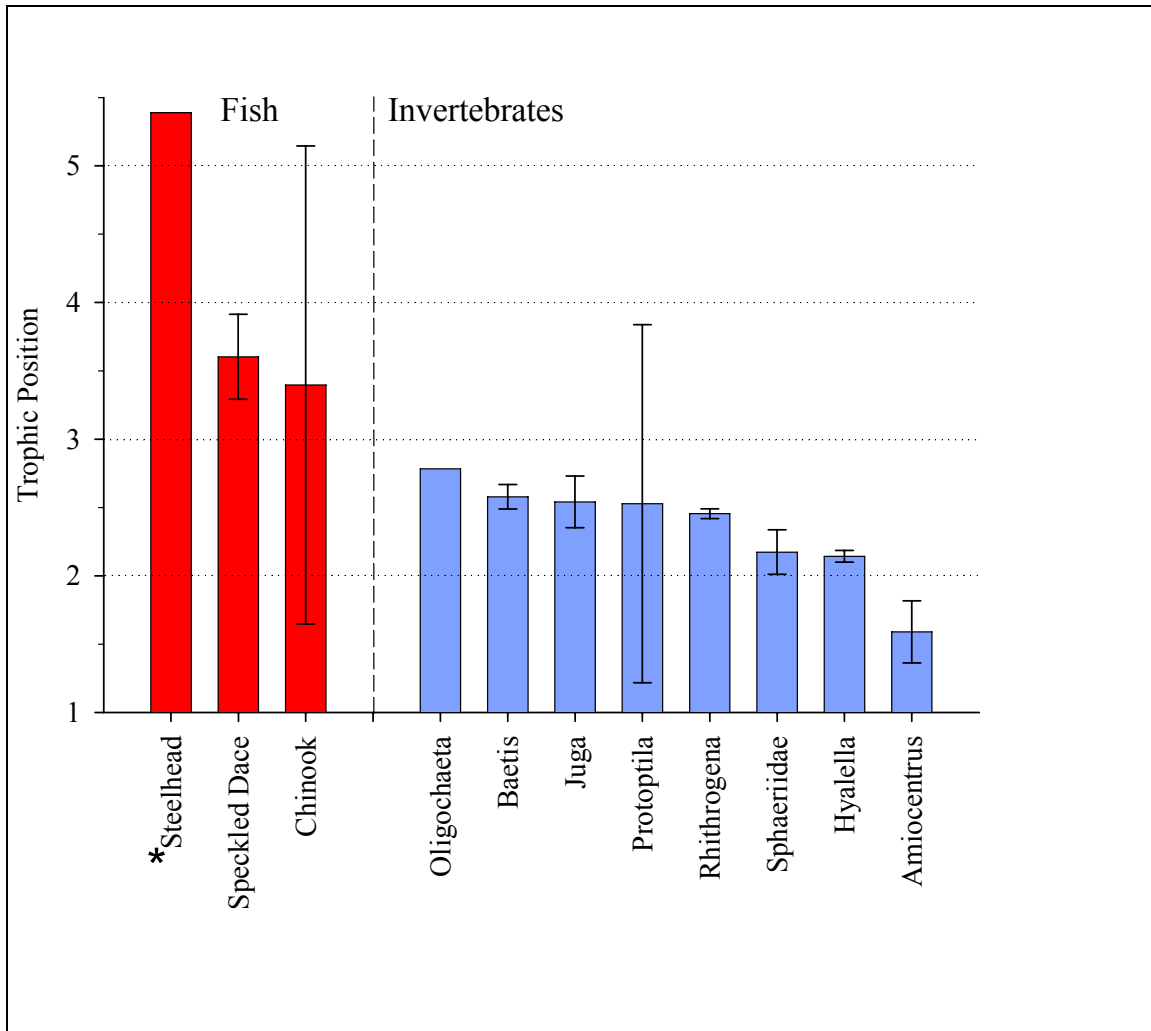


Figure 35. Mean trophic position estimates for aquatic consumers in the Shasta River during the Dec 2006 -Mar 2007 sample period. Error bars represent 95% confidence intervals. *The extreme trophic position estimate for steelhead trout is due to the presence of a residual maternal isotope signal in the tissues of the fish (N=1, FL = 27mm; see discussion in text).

Macroinvertebrate $\delta^{15}\text{N}$ signatures ranged from a high of 7.78‰ for oligochaete worms to a low of 3.73‰ for *Amiocentrus aspilus* (Figure 34). The mean $\delta^{15}\text{N}$ signature of *A. aspilus* was much lower than mean $\delta^{15}\text{N}$ values determined for any basal carbon source, but overlapped with the range of values determined for both seston and epilithic biofilms. Furthermore, trophic position (TP) estimates for *A. aspilus* were remarkably low with a mean TP of 1.6 (95 percent confidence interval = 1.4 to 1.8; N=5). Oligochaetes once again had the highest estimated trophic position among primary consumers (TP = 2.8) suggesting a general propensity for detritivory in the Shasta River (Figure 35). The remainder of the primary consumer guild had mean trophic position estimates ranging between 2.1 (*Hyalella* sp.) and 2.6 (*Baetis* spp.; Figure 35).

Stable isotope ratios were determined for three members of the fish community: speckled dace, Chinook salmon, and steelhead trout (Table 11). Speckled dace (mean FL = 74

mm; $N=3$) tissues had mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ signatures of $-27.33 \pm 0.66\text{‰}$ and $10.57 \pm 0.25\text{‰}$, respectively. Speckled dace were depleted by -1.83‰ relative to $\delta^{13}\text{C}$ values obtained for the previous (Oct-Nov) sample period, while both mean $\delta^{15}\text{N}$ values and trophic position estimates exhibited little change between dates (Table 11, Figure 35).

Two young-of-the-year (YOY) Chinook salmon (FL = 71 mm and 48 mm) were provided for isotopic analysis. Mean Chinook $\delta^{13}\text{C}$ was $-26.20 \pm 1.63\text{‰}$ and individual values were -27.82 and -24.56‰ , with the smaller fish being more ^{13}C -enriched (less negative). This was also the case with $\delta^{15}\text{N}$ as the smaller Chinook had a markedly higher $\delta^{15}\text{N}$ signature than the larger fish (10.37 vs. 9.38‰). We interpret the elevated $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values observed for the smaller (and presumably younger) fish to indicate the presence of marine-derived nutrients (MDN), specifically in the form of residual maternal yolk (see Gende et al. 2002, Naiman et al. 2002 for reviews of MDN in freshwater ecosystems). As YOY salmonids grow, their C and N isotope ratios systematically decline as they deplete their maternal yolk and begin to feed exogenously (Doucett et al. 1996). However, the time required for juvenile salmon to reach isotopic equilibrium with their riverine diet is highly variable and poorly understood. Power and Finlay (2001) found that juvenile steelhead in the South Fork Eel River drainage maintained a maternal (marine) signal until they reached standard lengths >50 mm. While such enriched isotope ratios are ultimately transient in YOY salmon, they can greatly obscure the interpretation of both diet and trophic position. For example, although the mean trophic position estimate for Chinook salmon produced a reasonable value (TP = 3.4), the 95 percent confidence interval around the mean was quite large (1.6 to 5.2; Figure 35). Further evidence of a marine-derived isotopic influence in the Shasta River food web comes from a single YOY steelhead trout (FL = 27 mm) with extremely elevated $\delta^{13}\text{C}$ (-19.43‰) and $\delta^{15}\text{N}$ (16.65‰) values (Figure 34). The enriched $\delta^{15}\text{N}$ value, in particular, produced a grossly inflated trophic position estimate of 5.2. Clearly, efforts to understand trophic linkages in salmonid food webs must recognize that assimilation of marine-derived nutrients and biomass, be it in the form of dissolved nutrients, gametes, carcass material, or predation on YOY fish, could lead to significant shifts in the stable isotope ratios of many organisms. However, the degree to which marine-derived nutrient subsidies influence food web structure and salmonid productivity in the Shasta River basin is presently unknown and warrants additional investigation.

Spring: April - June

There was considerable overlap among epilithic biofilms and seston in both $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values during the Apr-Jun sample period (Figure 36). Furthermore, biofilm $\delta^{13}\text{C}$ values were once again highly variable with individual samples ranging from -31.68 to -24.99‰ . Detritus during this period was comprised mainly of conditioned terrestrial leaf litter and contained very little macrophyte biomass. Mean detrital $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were $-28.15 \pm 0.17\text{‰}$ and $5.59 \pm 0.03\text{‰}$, respectively. Aquatic macrophytes displayed the most enriched $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of all the potential basal resources analyzed (Table 11, Figure 36). Interestingly, seston, detritus, and macrophytes all exhibited their lowest C:N ratios during this sampling period. Mean macrophyte C:N was especially notable as it decreased by approximately 9.5 from the prior sampling period (Figure 31).

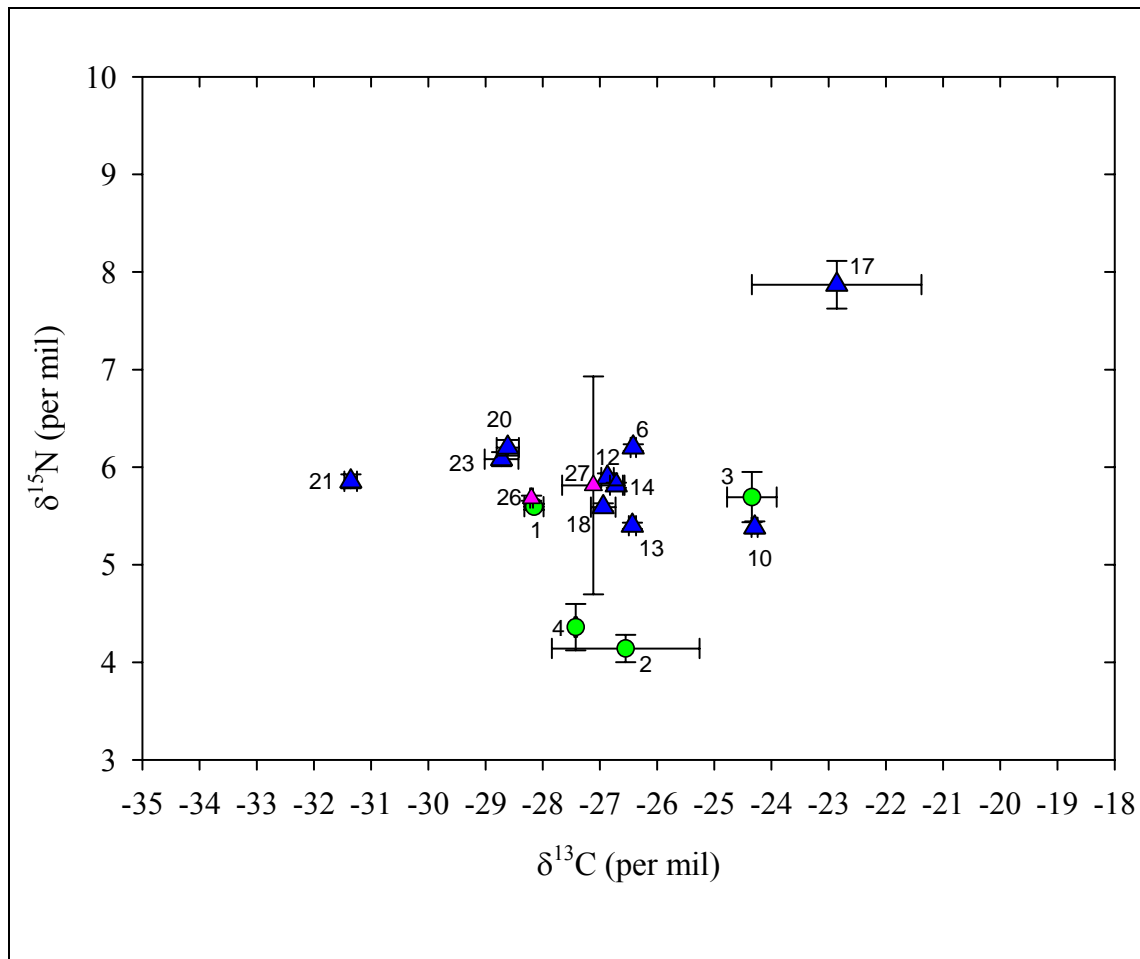


Figure 36. Stable carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) isotope ratios for selected members of the Shasta River aquatic food web during the Apr–Jun 2007 sampling period. Circles designate basal carbon resources and triangles represent macroinvertebrate taxa. Drifting aquatic and terrestrial invertebrates are identified by numerical codes 26 and 27, respectively. All data are presented as mean isotope ratios ± 1 SE. Numerical codes are provided in Table 11.

The majority of macroinvertebrate taxa had carbon signatures that closely matched the mean $\delta^{13}\text{C}$ of epilithic biofilms (Figure 36). In fact, mean $\delta^{13}\text{C}$ values for every macroinvertebrate analyzed fell within the range of carbon isotope values derived for epilithic biofilms. The heptageniid mayfly *Rhithrogena* sp. was the most isotopically depleted invertebrate with respect to carbon (mean $\delta^{13}\text{C} = -31.36 \pm 0.11\text{‰}$) while *Juga* sp. was the most enriched (mean $\delta^{13}\text{C} = -22.86 \pm 1.48\text{‰}$). *Juga* also had the highest mean $\delta^{15}\text{N}$ signature ($7.87 \pm 0.25\text{‰}$) and consequently the highest trophic position estimate (Figure 37) among the invertebrates sampled. Excluding *Juga*, the mean $\delta^{15}\text{N}$ value for primary consumers was $5.80 \pm 0.04\text{‰}$ ($N=48$), a trophic enrichment of 1.66‰ over mean epilithic biofilm $\delta^{15}\text{N}$. By comparison, *Juga* sp. was enriched in ^{15}N by 3.73‰ over mean biofilm $\delta^{15}\text{N}$. Trophic position estimates for primary consumers yielded surprisingly little variability among taxa. The highest mean trophic position (after *Juga*) was occupied by *Baetis* spp. (TP = 2.2), while *Tricorythodes* sp. (Ephemeroptera:

Leptohiphidae) and *Hyaletta* sp. shared the lowest estimated trophic position of 1.9 (Figure 37).

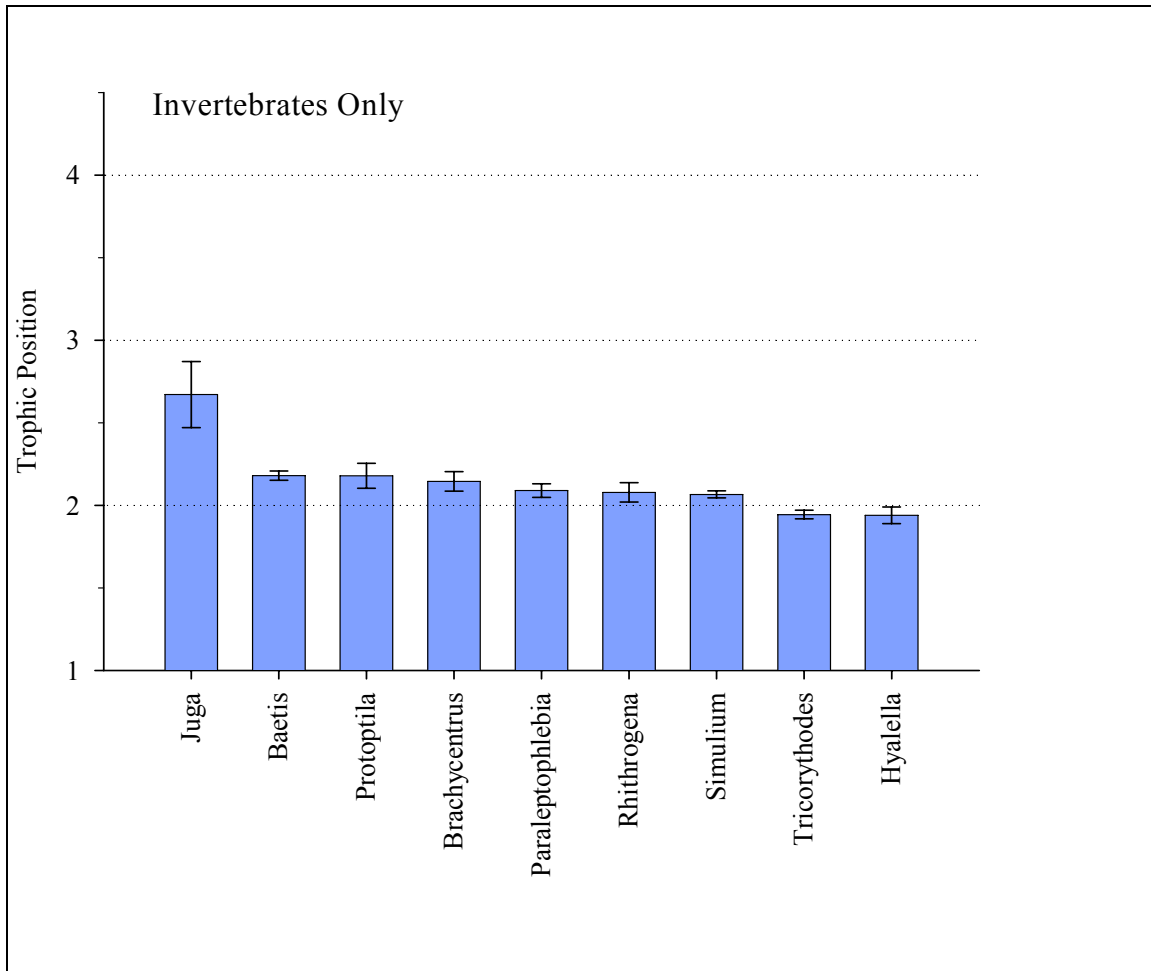


Figure 37. Mean trophic position estimates for aquatic consumers in the Shasta River during the Apr-Jun 2007 sample period. Only macroinvertebrates were analyzed during this sample period. Error bars indicate 95% confidence intervals.

Stream drift plays a central role in lotic food web dynamics as drifting insects are a commonly reported prey item of many stream fishes (Mundie 1974, Hunt 1975, Wipfli 1997, Nakano et al. 1999, Nakano and Murakami 2001). We sorted samples of drifting invertebrates into two factions according to origin (i.e., terrestrial vs. aquatic), and determined stable isotope ratios for each group independently. Drifting aquatic insects had a mean $\delta^{13}\text{C}$ value of $-28.20 \pm 0.02\text{‰}$ while those of terrestrial origin were slightly more ^{13}C -enriched with a mean $\delta^{13}\text{C}$ of $-27.11 \pm 0.55\text{‰}$ (Figure 36). The mean $\delta^{15}\text{N}$ signatures of drifting aquatic and terrestrial insects were very similar (5.68 ± 0.03 vs. $5.81 \pm 1.12\text{‰}$, respectively) but terrestrial $\delta^{15}\text{N}$ was highly variable (range = 4.70 to 6.93‰).

Summer: July - September

Primary carbon sources during this period were more variable with respect to $\delta^{13}\text{C}$ values than during previous sampling periods. Detritus had a mean $\delta^{13}\text{C}$ value of -26.44‰ (Table 11) but individual observations ranged from -22.45 to -28.49‰ . Moreover, the mean C:N ratio of detritus was approximately 27 (Figure 31) and detrital material was considerably ^{15}N -enriched relative to nearly all primary consumer taxa (Figure 38). Mean epilithic biofilm $\delta^{13}\text{C}$ was $-26.72 \pm 0.61\text{‰}$ and exhibited little change from the previous sampling date (Table 11). Biofilm C:N reached its lowest levels of the entire study with a mean C:N ratio of 9.9 (Figure 31).

Thirteen distinct invertebrate taxa were analyzed during this sample period (Table 11, Figure 38). Most primary consumers clustered into a single group centered on a $\delta^{13}\text{C}$ value of approximately -25‰ , suggesting exploitation of the same primary carbon source (Figure 38). Accounting for isotopic fractionation, epilithic biofilms are most likely fueling the bulk of invertebrate production during this period as well. However, interpretation is hindered by large variability in biofilm $\delta^{13}\text{C}$ and incomplete characterization of all potential organic matter sources (e.g., epiphytic biofilms or FPOM).

Among invertebrates, the caddisfly *Protophila* sp. was appreciably ^{13}C -depleted relative to the rest of the invertebrate assemblage (Figure 38). In fact, *Protophila* consistently yielded depleted mean $\delta^{13}\text{C}$ values across all sample periods (Table 11). *Juga* were again distinctively enriched with mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of -23.92 ± 0.06 and $8.27 \pm 0.13\text{‰}$, respectively. Curiously, both *Juga* and *Protophila* are classified as scrapers in California rivers (CDFG 2000). Despite belonging to the same functional feeding guild, these two taxa are clearly utilizing vastly different carbon sources in the Shasta River (Figure 38). The most ^{13}C -enriched taxon was *Hyalella* sp. with a mean $\delta^{13}\text{C}$ of $-22.57 \pm 0.06\text{‰}$. Although generally classified as a collector-gatherer in California (CDFG 2000), *Hyalella* is known to have extremely wide-ranging feeding habits. Published sources describe the consumption of detritus, epiphytic growth on rooted aquatic plants, and filamentous algae (Cooper 1965, Koslucher and Minshall 1973).

Drifting invertebrates of both aquatic and terrestrial origin had similar $\delta^{13}\text{C}$ signatures (Table 11). However, mean $\delta^{15}\text{N}$ values were distinct with terrestrial insects being enriched by $+1.48\text{‰}$ over drifting aquatic insects (Figure 38).

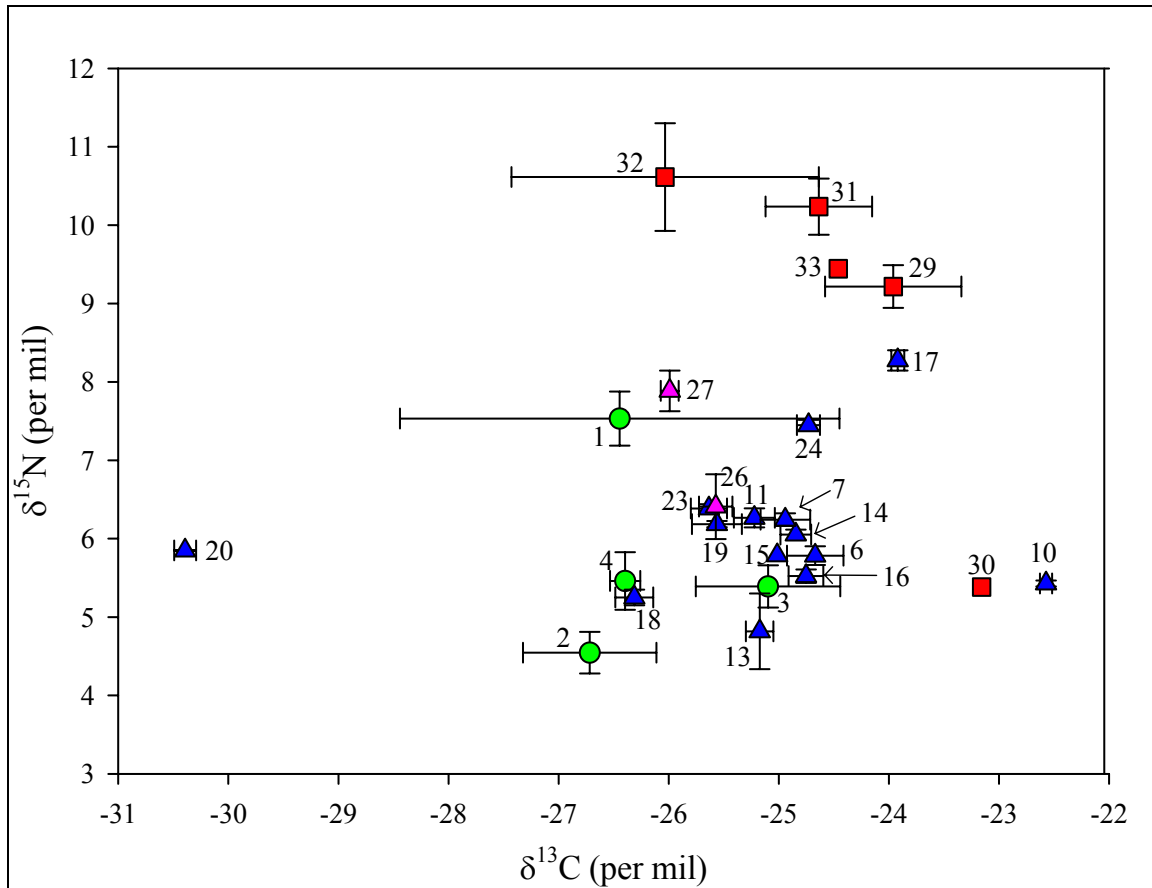


Figure 38. Stable carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) isotope ratios for key members of the Shasta River aquatic food web during the Jul–Sep 2007 sampling period. Circles designate basal carbon resources and triangles represent macroinvertebrate taxa. Drifting aquatic and terrestrial invertebrates are indicated by numerical codes 26 and 27, respectively. All data are presented as mean values ± 1 SE. A key to numerical codes is provided in Table 10.

The predatory damselfly *Hetaerina americana* (Odonata: Calopterygidae) had mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of -24.73 ± 0.10 and $7.45 \pm 0.07\text{‰}$, respectively. *H. americana*, as a secondary consumer in the food web (trophic position = 2.7; Figure 39) was predictably enriched in ^{15}N , but only by 1.55‰ over the mean $\delta^{15}\text{N}$ value for all primary consumers ($5.90 \pm 0.08\text{‰}$, $N=49$; *Juga* sp. omitted).

Steelhead trout had the most variable isotope values of the stream fishes analyzed. Steelhead $\delta^{13}\text{C}$ ranged from -23.23 to -29.90‰ (mean = $-26.03 \pm 1.40\text{‰}$) and $\delta^{15}\text{N}$ ranged from 9.22 to 11.89‰ (mean = $10.61 \pm 0.69\text{‰}$; Figure 38). All steelhead during this sample period were >65 mm FL and presumably beyond the influence of any confounding maternal isotope signal. As was the case during the fall (Oct–Nov), steelhead occupied the highest trophic position among the fishes analyzed (Figure 39). Speckled dace (mean FL = 65mm) $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ signatures demonstrated considerable overlap with steelhead, but dace were slightly ^{13}C -enriched (mean $\delta^{13}\text{C}$ = $-24.64 \pm 0.48\text{‰}$). Speckled dace mean $\delta^{13}\text{C}$ values varied by almost 2.5‰ across sample dates, while $\delta^{15}\text{N}$ values remained fairly constant over time (Table 11). Carbon isotope

signatures for marbled sculpin ($-23.96 \pm 0.62\text{‰}$) and a solitary Klamath smallscale sucker (-24.46‰) were similar to the mean $\delta^{13}\text{C}$ value for speckled dace (Figure 38). Sculpin and sucker mean $\delta^{15}\text{N}$ values were nearly identical at $9.22 (\pm 0.27\text{‰})$ and 9.44‰ , respectively. As was the case for the Oct-Nov 2006 sample period, $\delta^{13}\text{C}$ values for all fish taxa generally fell within $\sim 1.0\text{‰}$ of the mean $\delta^{13}\text{C}$ value for the entire macroinvertebrate assemblage. A single lamprey ammocoete was also analyzed during this sample period. Lamprey ammocoetes are generally found in soft sediments and thought to feed primarily on detrital material (Moyle 2002). The ammocoete was ^{13}C -enriched ($\delta^{13}\text{C} = -23.16\text{‰}$) relative to the bulk of the aquatic community (Figure 38). Furthermore, the ammocoete had a $\delta^{15}\text{N}$ value of 5.38‰ and a trophic position estimate of 2.0 (Figure 39) confirming its role as a primary consumer within the food web.

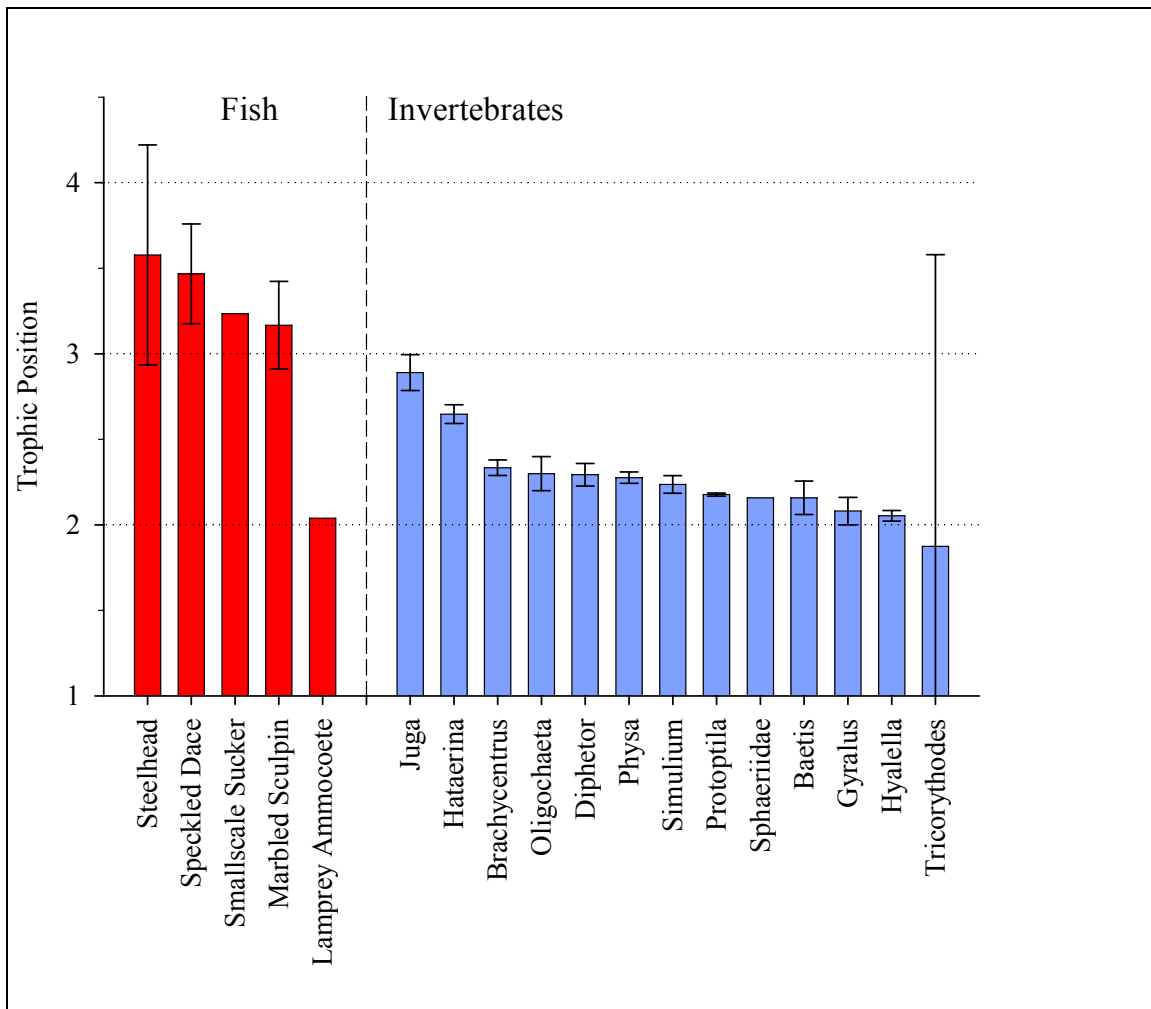


Figure 39. Mean trophic position estimates for aquatic consumers in the Shasta River during the Jul-Sep 2007 sample period. Error bars indicate 95% confidence intervals.

10.4. Summary

Our results demonstrate that natural abundance stable isotope analysis is a powerful tool for identifying important sources of organic matter and trophic linkages in the Shasta River watershed. We found that epilithic biofilms served as the primary energy source

fueling food web productivity throughout the year. Although aquatic macrophyte standing crops were high on most sample dates, the high lignin and cellulose content of live plants (i.e., high C:N ratios) rendered them a poor quality food resource and precluded their direct consumption by most aquatic herbivores. The detrital pool at the Nelson Ranch site consisted chiefly of macrophyte fragments, while allochthonous (terrestrial) materials generally represented a very minor fraction of the total pool. Similar to live macrophytes, detritus had elevated C:N ratios and was of poor nutritive value throughout the year.

We identified a total of 2,046 organisms representing 68 unique taxa as part of our seasonal macroinvertebrate surveys. Members of the collector-gather and collector-filterer functional feeding guilds consistently dominated the macroinvertebrate community. This suggests that fine particulate organic matter (FPOM; particles of organic matter < 1 mm) serves as a critical link between dissolved nutrients, epilithic biofilms, and secondary production. Shredders were conspicuously scarce on all dates, presumably due to the dearth of allochthonous material. While the relative abundances of most invertebrate taxonomic groups remained fairly consistent over time, non-biting midges (Chironomidae) exhibited a dramatic spike in abundance during the April-June sample period. Drifting invertebrates represent an important food resource for stream fishes and our results indicate that drift density (prey availability) decreases through the summer months at the Nelson Ranch site. We found that most drifting invertebrates were of aquatic origin as terrestrially-derived organisms accounted for $\leq 10\%$ of the total drift density and $\leq 3\%$ of the total drift biomass.

Our findings provide novel and important information regarding the structure and function of the aquatic community at the Nelson Ranch site. However, significant data gaps still exist and continued sampling is necessary to advance our understanding of the key ecological and trophic interactions that support juvenile salmonids in the Shasta River basin. Specifically, future studies should aim to:

1. Quantify temporal and spatial changes in biomass (e.g., standing stocks of organic matter, macroinvertebrates, etc.) and important ecological rates such as inputs of allochthonous material, invertebrate emergence and drift, and secondary production. Such knowledge would greatly improve our understanding of ecosystem-level processes and provide important baseline information from which the efficacy of alternative management practices and restoration activities may be assessed.
2. Characterize the stable isotope signatures of additional sources of organic matter that have emerged as potentially important contributors to carbon flow in the Shasta River (e.g., various size fractions of particulate organic matter and epiphytic biofilms).
3. Generate comprehensive information on the fish community during each season. Specifically, fish distribution and abundance data should be coupled with stable isotope and gut content analyses to confirm trophic relationships and quantify

important food web interactions. Future trophic investigations should target scales (habitat types) that are ecologically relevant to fish at different stages in their life cycle.

11. Fish Surveys

Snorkel surveys were conducted in the Shasta River to determine fish abundance and diversity within various habitats as hydrologic conditions changed throughout the 2007 water year. During water year 2007 we conducted 352 reach snorkel surveys along the Nelson Ranch on the Shasta River. During snorkel surveys we observed eight species of fish in the six different habitat types defined for the Nelson reach. Below we discuss observations of salmonids found in the Shasta River (coho salmon, Chinook salmon, and steelhead) during the seasons (fall, winter, spring, and summer) of the 2007 water year. Along with observations of fish, we introduce the ecological trap conceptual model for coho salmon within the Shasta River and highlight restoration goals for resource managers.

11.1. Methods

Snorkel surveys are a non-invasive method to determine relative abundance and habitat usage but are not a surrogate for population estimates. Because of the presence of coho (a federally threatened species), snorkel surveys were determined to be the method with the lowest level of impact when determining habitat usage by fishes. To conduct snorkel surveys, the Nelson Ranch was divided into five approximately 1.6 km sub-reaches and within each sub-reach, there was a replicate of each habitat type. As noted above, typical habitat typing methods did not provide sufficient variability to identify utilized habitats, and a site-specific habitat classification system was created to physically describe 100% of the wetted channel. Recall, the previously habitat defined classification system (Habitat Mapping) included six types of channel margin and/or in-channel aquatic habitat, including point bars (PB), active cut-banks (CB), perennial and/or ephemeral backwaters (BWp/BWe), large woody debris (LWD), emergent vegetation (EV) and aquatic macrophytes (AM) (Figure 40). The survey was completed moving upstream and fish were only counted within one meter of each side of the surveyor. We conducted snorkel surveys one to two times per month depending on hydrologic conditions (Table 12). River turbidity made sampling difficult during winter months and thus, few samples were conducted from November through March. As aquatic macrophytes became dense during summer, several sites were no longer readily surveyed. In this case, one downstream float survey was conducted in each of the 1.6 km reaches where the surveyor floated downstream, counting fish near the thalweg of the river. Reaches varied between 100 and 200 meters in length. During all surveys, the surveyor identified fish species and age class, and recorded the information on a wrist slate. A second snorkel surveyor occasionally conducted concurrent surveys to verify fish numbers and species counts. When the second surveyor was present, fish numbers and species counts were compared to the regular surveyor's data. On all occasions, data from both surveyors were similar. After a reach survey was completed, instream cover, substrate type and exposed substrate were qualitatively estimated and recorded. Water quality parameters were measured after

each survey using a YSI 6820 data sonde. Water quality parameters recorded were temperature, dissolved oxygen, turbidity, pH, and conductivity.

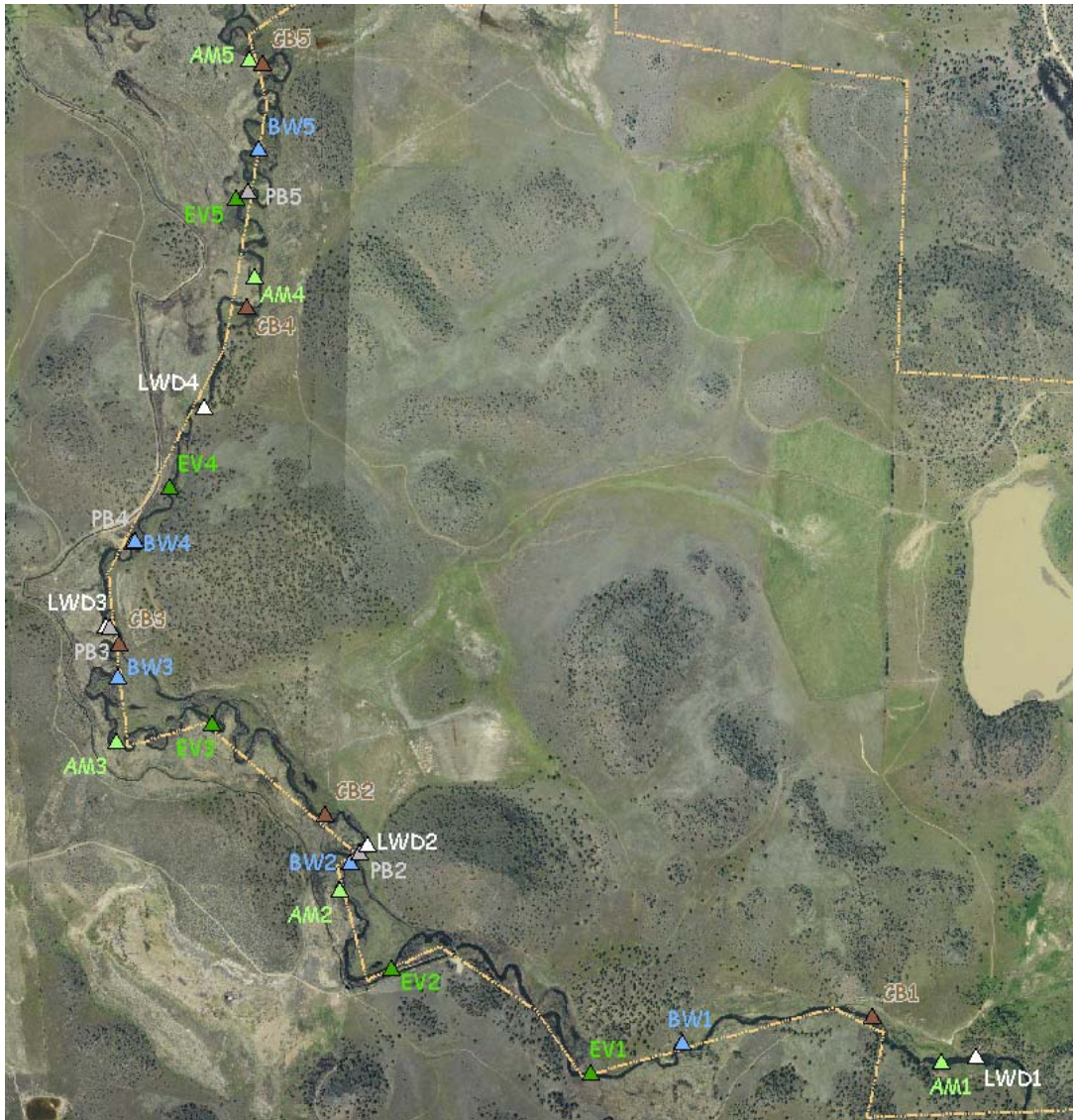


Figure 40. Map of snorkel survey sites along the Nelson Ranch. Different colors denote different habitat types.

Table 12. Number of snorkel surveys conducted in each habitat type during each sampling period. The number of surveys was dependent on hydrologic conditions, aquatic macrophyte density, and safety concerns.

	Aquatic Macrophyte	Backwater	Cut Bank	Emergent Vegetation	LWD	Point Bar
Oct. 2006-1	5	5	5	5	4	4
Nov. 2006-1	5	5	5	5	2	4
March 2007-1	2	0	1	5	2	4
April 2007-1	4	1	4	5	1	4
April 2007-2	5	5	5	5	2	4
May 2007-1	5	5	6	6	5	4
May 2007-2	5	5	6	6	5	4
June 2007-1	5	5	5	6	5	4
July 2007-1	5	5	5	6	5	4
July 2007-2	5	5	5	3	4	4
August 2007-1	5	5	4	3	4	3
August 2007-2	5	5	4	3	4	2
Sep. 2007-1	5	5	3	3	3	2
Sep. 2007-2	5	5	4	3	4	2

11.2. Coho Salmon (*Oncorhynchus kisutch*)

During fall 2006, 47 adult coho were counted entering the mouth of the Shasta River (CDFG unpublished data). Due to low numbers of returning adults, low juveniles counts were expected in the Shasta River. We observed juvenile coho on the Nelson Ranch during spring and early summer when visibility was good and water temperatures were relatively cool. We observed schools of 0+ and 1+ coho in a single backwater and LWD habitat in April and May 2007. Juvenile coho were often observed in schools of five to 15 individuals. Common features in the habitats where coho were observed were relatively deep water and the presence of woody debris. Often times, coho mixed with schools of juvenile steelhead and Chinook. When water temperatures warmed and stage dropped in mid-May, coho observations on the Nelson Ranch declined (Figure 41). A single coho was observed in a backwater habitat in June and early July. After 3 July, no coho were observed on the Nelson Ranch. The increase in temperature and decrease in water stage coincided with peak out migration at the mouth of the Shasta River (CDFG personal communication). Mean weekly maximum temperatures (MWMT) over 18°C (64.5°F) began in early April and remained over 18°C (64.5°F) for most of the summer through mid-September (Figure 42). Welsh et al. (2001) found that in the Mattole River watershed, streams with MWMT's greater than 18°C (64.5°F) contained no coho salmon during summer, while all streams sampled with MWMT's less than 16.3°C (61.3°F) contained coho during summer. Productivity within the Shasta River likely increases the maximum water temperature that coho may rear, but it is unknown what the maximum temperature is and how coho respond in the unique high productivity-elevated temperature conditions found in the Shasta River. Future studies on coho and habitat utilization should focus on large-scale seasonal movements and how water temperature affects movement between habitats longitudinally along the Shasta River.

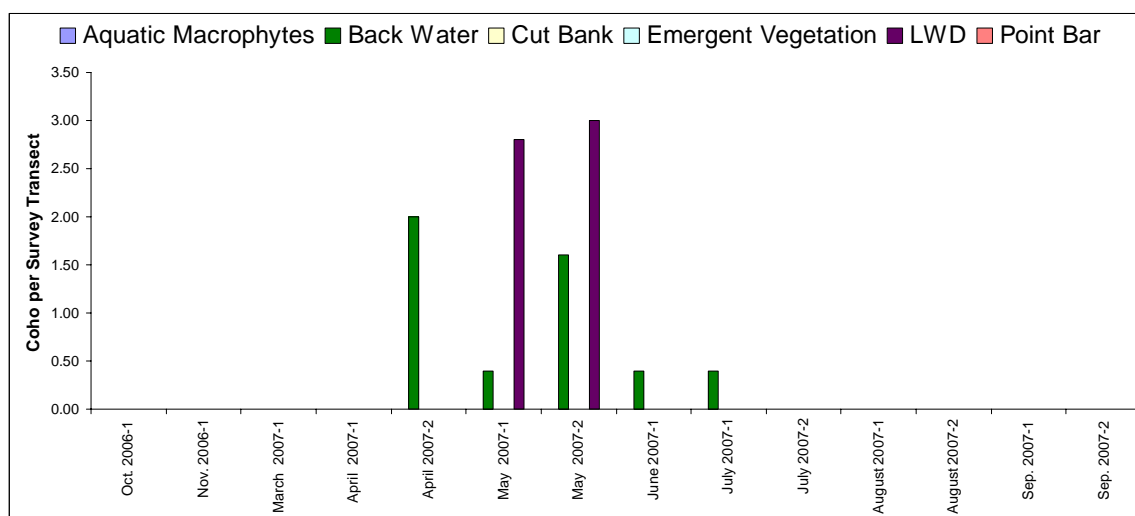


Figure 41. Juvenile coho salmon observed in various habitats during snorkel surveys on the Nelson Ranch. After water temperatures warmed in May, very few coho were observed along the Nelson Ranch. The June and July observations consisted of one fish each in a backwater habitat.

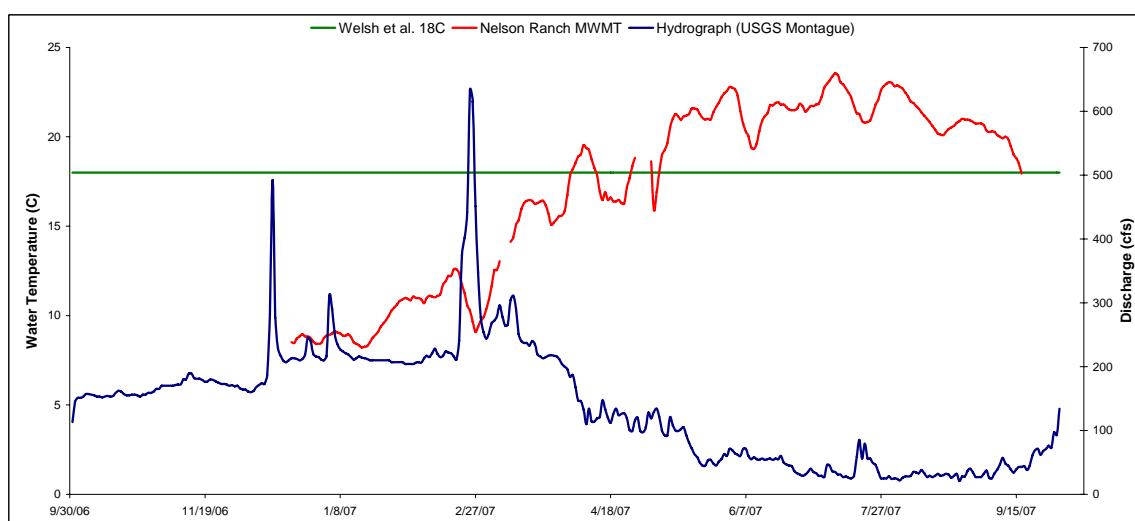


Figure 42. Mean weekly maximum temperature (MWMT) is from the upstream property boundary of the Nelson Ranch, with the 18°C (64.5°F) coho threshold line derived from Welsh et al. (2001). Hydrograph is from USGS Montague gauge (11517000).

11.3. Coho Salmon Ecological Trap Conceptual Model

Due to low number of returning adult coho, insufficient observations were made during the 2006-07 season to present year-in-the-life conditions on the Nelson Ranch. However, the coho salmon life history in the Shasta River is presented herein and the ecological trap conceptualization is discussed. Coho salmon typically return to the Shasta River between November and January. Due to the timing of returning adult coho, accurate counts are often difficult to make due to high winter flows. CDFG has been conducting adult counts as conditions allowed since 2001 and run sizes ranged from a low of 46 in 2006 to a high of 373 in 2004 (CDFG unpublished data). Spawning and egg incubation occurs between December and March and juveniles emerge between February and April. Coho juveniles can have several different life history strategies that result in differences

in rearing location and outmigration timing (Lastelle 2006). In the Shasta River juvenile coho exhibit two different life history strategies. One life history strategy is that after emergence coho rear in the Shasta until water temperatures increase, and then they leave the Shasta River and seek cooler temperatures downstream in the Klamath River, tributaries to the Klamath or the Klamath Estuary. The second life history strategy is remaining in the Shasta River throughout the summer and winter, and then emigrating the following spring. This second life history strategy is only available to fish born in the upper reaches where water temperatures remain relatively cool throughout the summer (see Ecological Trap discussion below).

Coho salmon in the greater Klamath River watershed have been reduced to such low population numbers that NOAA Fisheries has listed the population at “threatened” (NRC 2004). Coho populations in the Shasta River (a tributary to the Klamath) are very small and have been in decline for many years. A conceptual model that may help to explain the decline in coho populations is that the current flow regime and thermal conditions in the Shasta River acts as an “ecological trap” for the coho salmon life history. An ecological trap is a scenario that occurs when an animal preferentially chooses a habitat that ultimately reduces its survivability or reproductive success over a habitat of apparent similar quality where survivability and reproductive success are much higher (Robertson and Hutto 2006). The coho salmon life history strategy makes them more susceptible to anthropogenic ecological traps than other Pacific salmon species. Because juveniles rear for a year in rivers, they are more likely to perish prior to seaward migration due to anthropogenic alteration of the landscape than other salmon species that leave freshwater during their first year. To date there has been no described system in the literature where a fish population has been susceptible to an ecological trap. Identification of an ecological trap may help explain the cause of declining coho populations in the Shasta River and guide future management actions. Unless the underlying effects of the ecological trap conceptual model can be ameliorated, coho in the Shasta River have an increased risk of extinction over time.

Although data is sparse, it is hypothesized that the Shasta River, under current conditions, acts as an ecological trap for two life stages of coho salmon, returning adults and rearing juveniles. Adult coho returning to spawn in the Shasta River divide almost equally between two locations with suitable spawning habitat (B. Chesney, CDFG, personal communication 2006). One spawning area consists of the 7 km (4.3 mi) above the confluence with the Klamath River in the canyon section of the river. The other spawning area is 55 km (34 mi) above the Klamath confluence near the Big Springs complex (Ricker 1997). During late fall and winter when adults return, there is little difference in habitat quality between the two spawning areas. It is not until mid-spring and into summer that large differences in juvenile rearing habitat quality exist between the two locations. The primary degradation of habitat is seasonal elevated water temperatures, which are the result of the region’s Mediterranean climate, geographic location, and historical and current land and water use practices (NRC 2004, Watercourse 2003a)

One element of the ecological trap for adult coho salmon is the lack of environmental cues during winter about future summer conditions, i.e., where their progeny will rear, this applies to the population that spawns in the Shasta River canyon. During summer, temperatures in the canyon are lethal for juvenile coho salmon (Watercourse 2003b, G. Stutzer USFWS unpublished data 2007). The progeny of adult coho that spawn in the canyon cannot migrate upstream to relatively cool water reaches due to the considerable distance (48 km (30 mi)), steep river sections, and other barriers (e.g., flashboard dams installed for irrigation). This limits the options for juvenile coho hatched in the canyon to outmigration into the Klamath River. Unfortunately, mainstem conditions during summer in the Klamath River are not much better for rearing (e.g., elevated water temperatures, disease) than those found in the lower Shasta River. There are areas where thermal refugia are present in the Klamath, but those areas are small and competition is high between coho and other more competitive juvenile salmonid species (Sutton et al. 2007, NRC 2004). What causes the ecological trap for adult coho that choose to spawn in the Shasta River Canyon is reduced fitness, due to low survivability of their progeny. Because coho salmon are semelparous, this does not allow for experience-based learning behavior, which might act as a rescue from the ecological trap and promote future spawning in habitats that result in a higher fitness (Kokko and Sutherland 2001).

The Shasta River acts an ecological trap for juvenile coho that emerge from the gravels in the Upper Shasta River above the Nelson Ranch during late winter and early spring. After emergence, juvenile fish distribute downstream from upstream spawning locations. During this time of year, rearing conditions are good throughout the entire length of the Shasta River. Juvenile coho move downstream, leaving habitat that will provide good potential for survivability in summer, and entering habitat that will provide very low survivability during summer. Similar to adult coho choosing spawning locations in the Shasta River, there are no environmental cues for juvenile coho to discourage distribution into rearing areas that in a few of months will experience elevated water temperatures. As water temperature warms downstream, upstream migration opportunities are largely unavailable due to water and land use practices reducing baseflow and associated loss of habitat, increasing water temperatures, barriers (e.g., flashboard dams), and other factors. As with juvenile coho that emerge in the canyon reach, these fish must leave the Shasta River and enter the Klamath River. Thus, coho that migrate downstream early in the year will suffer from lower survivability and thus a reduced fitness compared to fish that remain in upstream rearing habitat where temperatures remain relatively cool year-round. Due to current low population numbers, density dependence is not an issue in upstream locations where potential over summer rearing habitat is available.

Currently, much of the Shasta River does not provide suitable over-summering water temperatures for juvenile coho with the exception of isolated reaches above the Nelson Ranch. If the ecological trap hypothesis holds unless upper reaches of the Shasta River are managed for cold water maintenance during summer periods and restored to suitable over-summering habitat, then coho face an increasing risk of extirpation from the Shasta River over time. If year-round cold water habitat is restored in the Shasta River, then the negative effects of the ecological trap could be minimized to the point where coho populations could potentially stabilize and even increase through time.

11.4. Chinook Salmon (*Oncorhynchus tshawytscha*)

Historically, the Shasta River was one of the most productive salmon streams in California, with runs of Chinook salmon over 80,000 returning adults in the 1930's (NRC 2004). Since the closure of Dwinnell dam in 1928, Chinook salmon numbers have decreased dramatically (Figure 43). Between 2001 and 2006, Chinook returns averaged 4,566 adults per year with a high of 11,093 and a low of 978 (CDFG unpublished data). A reduction in spawning habitat is likely one of the primary reasons for the decline of Chinook populations over time. Closure of Dwinnell Dam blocked 33 percent of river but likely, a much higher percentage of the high-quality spawning habitat (Wales 1951). Construction of Dwinnell Dam not only cut off access to spawning habitat upstream of the dam, but altered habitat conditions downstream. Through time, the combination of lower summer flows and less frequent and smaller magnitude peak winter flows, resulted in sedimentation of fine material within the gravels and encroachment of riparian vegetation. This reduction in stream size resulted in a considerable loss of spawning habitat in the reach from Dwinnell Dam to Big Springs Creek. It is possible that the gradual loss of spawning habitat below Dwinnell Dam allowed Chinook salmon populations to be maintained at relatively high numbers for several years after closure of Dwinnell, but ultimately the combined loss of both upstream and downstream habitat leads to numbers more consistent with current conditions. Recent spawning habitat surveys have shown that from Dwinnell Dam to the mouth, the quality of spawning gravels is poor (Ricker 1997). Currently little spawning takes place on the Nelson Ranch. During WY 2007, eight redds were observed along the Nelson reach. The limited spawning activity is due to the lack of high quality spawning gravels. Low gradient, lack of upstream gravel recruitment, and abundance of fine sediments within the Nelson reach limits suitable spawning gravels.

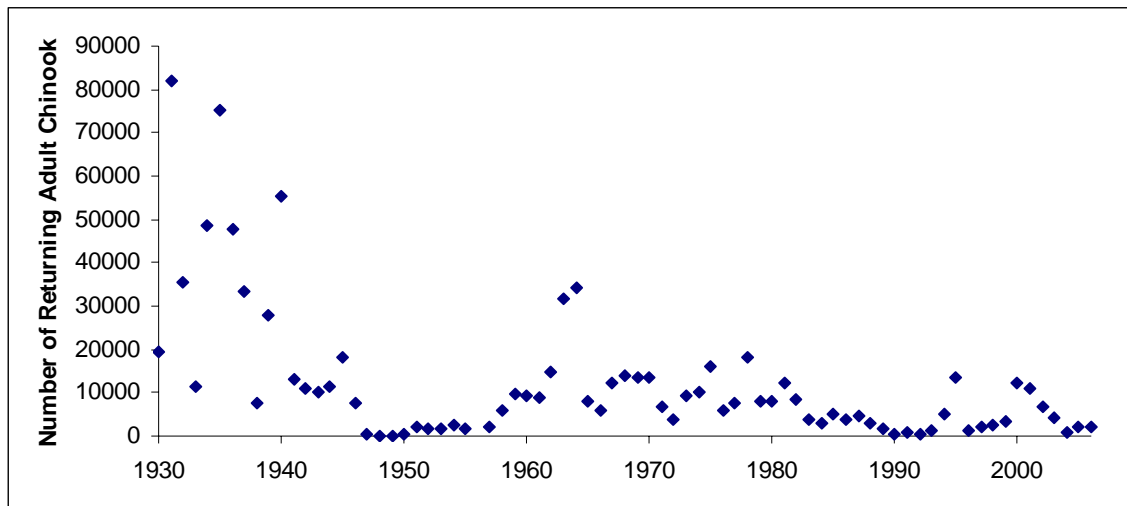


Figure 43. Number of adult Chinook entering the Shasta River between 1930 and 2006. Note the dramatic decline from the early 1930's to the mid 1940's. Data source KRISWEB and CDFG unpublished data.

Fall: October - November

Adult Chinook salmon returned to the Nelson Ranch starting in October as irrigation diversions were terminated and flashboard dams were removed following the end of

irrigation season on 1 October. During October and November, adult female Chinook find suitable locations to dig redds. During WY 2007, we observed eight Chinook redds on the Nelson Ranch, all of which were in the upper two miles of the reach. After Chinook dig redds and spawn, they die soon thereafter, providing nutrients to the aquatic and terrestrial ecosystems.

Atypical behavior during spawning was observed during a reconnaissance dive on 4 October 2006 near the top of the property. Specifically, juvenile Chinook salmon were observed in a Chinook redd with an adult female and two adult males present. Juvenile Chinook held immediately behind the female and were occasionally hit with her tail as she was digging, this was similar behavior to mature male parr observed in the Salmon River, Idaho (Gebhards 1960). CDFG personnel operating the screw trap immediately downstream of where the juvenile Chinook were observed noted that juvenile Chinook caught in the trap were sexually mature. Upon further otolith analysis, these mature juvenile Chinook salmon were found to be less than a year old (J. Reader, CDFG personal communication). This is the first time that mature male parr have been found in the Shasta River, but mature male parr have been documented in the Fall Creek hatchery, on the Klamath River above Iron Gate dam in the 1950's prior to the construction of Iron Gate dam in 1961 (Robertson 1957). Robertson (1957) also found that mature parr that spawned did not die after spawning and produced viable progeny when crossed with an adult female. Mature male parr are very rare in nature and are most often found in spring-run Chinook salmon that are born earlier than fall-run fish and thus are able to grow more rapidly and mature at an early age (Larsen et al. 2004). It is unknown how mature parr contribute to the population in the Shasta River or Klamath Basin in general, but the life history strategy may help the population hedge bets against poor migratory conditions downstream. More study is needed to determine what impact mature parr have in the overall Chinook population in the Shasta River.

Winter: December – March

As with late October and November, turbidity in the Shasta River was high throughout the winter season and did not allow for snorkel surveys. From December through February the Chinook eggs deposited in the gravels during October and November begin to mature and emerge from the gravels as alevins (sac fry). The alevins and fry seek slow moving water with cover until they are large enough to inhabit deeper faster habitats. During March, the river began to clear and snorkel surveys were again possible. In March, when snorkel surveys resumed, we observed juvenile Chinook in point bar, cut-bank and emergent vegetation habitat types (Figure 44). During this time, juvenile Chinook were the largest of the juvenile salmonids observed. The Chinook were often in habitats where relatively high velocities were observed.

Spring: April - June

As irrigation season began, water stage throughout the reach fell and juvenile Chinook left shallow emergent vegetation habitat and took residence primarily in point bar, backwater, and LWD habitats (Figure 44). These habitats provide deeper water, cover, and were close to higher velocities regions. A second reduction in water took place in May when the discharge was reduced from 131 cfs (3.71 cms) to 86 cfs (2.44 cms) at the

top of the Nelson reach. After this reduction in flow, we observed no schools of juvenile Chinook. It appears that as water temperature warms and river stage drops, most juvenile Chinook follow a typical fall-run ocean type life history and leave the Shasta River for the ocean. A small number of juvenile Chinook over-summer in the Shasta River, thus exhibiting a stream type life history.

Summer: July - September

Throughout the summer sampling season, juvenile Chinook were observed on several occasions. Juvenile Chinook were generally found in deep mid-channel aquatic macrophytes or on the edge of backwater habitat where velocities were relatively high. Juvenile Chinook observed during summer were often found mixed with schools of juvenile steelhead. Chinook observed during summer months were fish that could develop into mature male parr in fall or will leave for the ocean as 1+ smolts. Over wintering habitat for 1+ Chinook is not known at this time in the Shasta River, because of difficulties in winter surveying and relatively low numbers that leave as 1+ smolts.

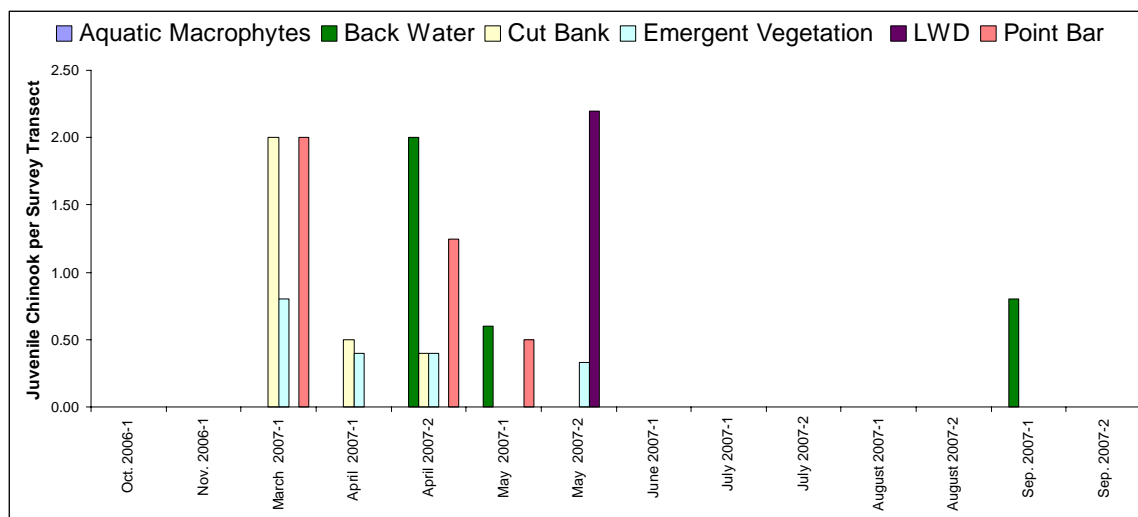


Figure 44. Juvenile Chinook salmon observed in various habitats during snorkel surveys on the Nelson Ranch. The majority of juvenile Chinook left by June, but several juveniles were observed to over-summer along the Nelson Ranch.

11.5. Steelhead (*Oncorhynchus mykiss*)

Steelhead trout are the most abundant salmonid on the Nelson Ranch. Steelhead reside in the river year-round and have high water temperature tolerances relative to coho salmon, the other salmonid the utilize the Shasta River throughout the year (Richter and Kolmes 2005). We observed several age classes of steelhead on the Nelson Ranch throughout the year. The most common age class was 0+ fry (fry that emerged the previous spring). 0+ steelhead were first observed in March, when the water first clears enough to begin snorkel surveys, and were observed through October (Figure 45). One year old and older steelhead were observed from June through September (Figure 46).

Fall: October - November

Steelhead have a close relationship with other species of anadromous fish in certain rivers and in the Shasta River the presence of fall-run Chinook salmon may play an important

role in the steelhead lifecycle. During the beginning of October, adult Chinook salmon returned to the Shasta River. Returning salmon provide a food source for juvenile steelhead in the form of dislodged invertebrates from the digging of the redd and from eggs that do not get buried and thus, become entrained in the drift. Salmon carcasses likely provide an additional food source in the form of direct feeding on the flesh and from invertebrates feeding on the salmon flesh (Bilby et al. 1998). During snorkel surveys in early October, schools of 0+ steelhead were observed feeding immediately behind the redds where female salmon were digging. Despite seeing 0+ steelhead behind salmon redds in October, we observed very few steelhead in the survey transects. This was consistent with a declining trend that was observed throughout the summer (Figure 45) and discussed in more detail below.

After the first week of October, turbidity in the river made snorkel surveys difficult. Several attempts were made at snorkeling during the winter season, but turbidity was too high to observe fish. Over-wintering habitat along the Nelson Ranch is an important element that has not been quantified at this time. To determine over-wintering habitat a more detailed study using radio telemetry, PIT tags, or other method will likely need to be implemented.

Winter: December – March

Adult Winter-run steelhead enter the river and spawn during December through March. As with late October and November, turbidity in the Shasta River was high throughout the winter season. During March, the river began to clear and snorkel surveys were again possible. During March, steelhead recently emerged from the gravels and were found in shallow water edge habitat along the Nelson Ranch. Juvenile steelhead were found during March in shallow water habitats associated with EV, CB, PB, and LWD. The common features of all of the shallow water habitats utilized by steelhead fry were that there was refuge from high velocities and protection from predators.

Spring: April - June

Irrigation season begins on 1 April, thus habitat in the Shasta River changes dramatically for steelhead fry, as well as other species. On 1 April, river stage dropped dramatically, reducing the amount of shallow-water habitat available (Figure 2). As river stage dropped, steelhead fry were displaced from many shallow-water habitats and were forced to find different, newly available shallow-water habitats in other locations.

As the spring season progressed, water temperatures warmed and aquatic macrophytes began to grow in the main channel. This provided an increase in habitat available to the growing steelhead fry. Point bar habitat is a transitional habitat with variable depth and often contained aquatic macrophytes, emergent vegetation, and woody debris in small patches. Point bar habitat was the most densely populated during May when steelhead fry were too large to utilize the remaining shallow-water habitat, yet not large enough to utilize mid-channel habitat (Figure 45). By June, aquatic macrophytes had become well established and juvenile steelhead had begun to utilize this productive and bioenergetically favorable habitat. Aquatic macrophytes provide a velocity refuge for

steelhead and a substrate for aquatic invertebrates, the primary food source for juvenile steelhead.

During June, several pods of adult summer-run steelhead were observed in survey reaches and while walking along the river. These fish appeared to be bright from the ocean and not resident rainbow trout. They were often observed feeding on an abundant hatch of *Hexagenia* mayflies, which hatched in large numbers from May through July. Summer/Spring-run steelhead are likely to be adversely affected by high water temperatures and barriers that limit migration throughout the Shasta River.

Summer: July - September

During summer along the Nelson Ranch, the dominant aquatic habitat feature is aquatic macrophytes. Aquatic macrophytes displaced certain other habitat types (emergent vegetation and backwaters), but provided mid-channel habitat that the now larger juvenile steelhead occupy. The majority of steelhead observed between July and September were in mid-channel aquatic macrophyte habitat (Figure 45). Another trend observed in juvenile steelhead on the Nelson Ranch was declining numbers from July through September. We currently do not know why numbers declined throughout the summer season. Possible explanations are migration to different reaches of the river, natural mortality, or some other unknown reason. Additional study is needed to determine large-scale movement of the steelhead in the Shasta River.

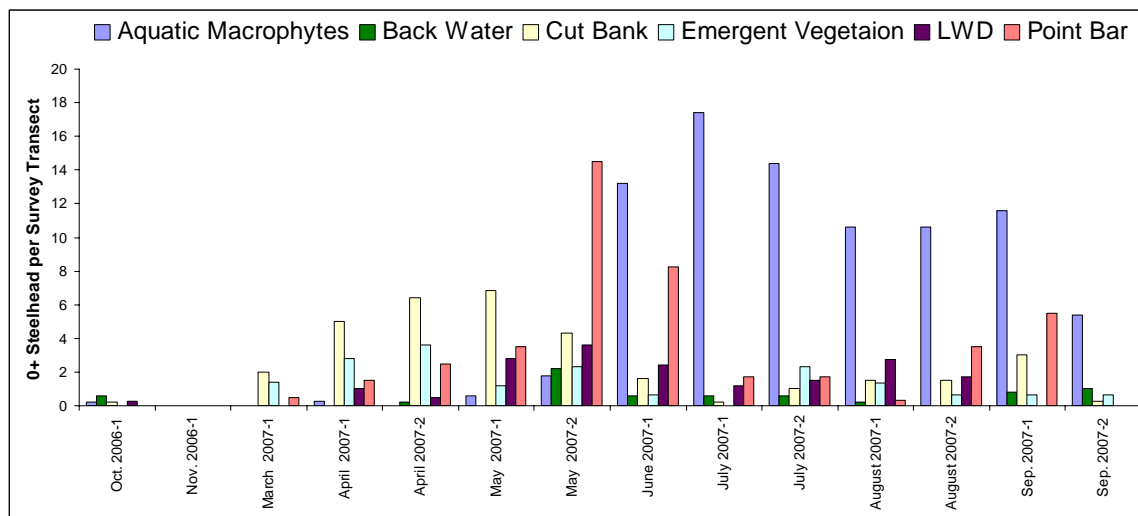


Figure 45. Young of the year steelhead observed in various habitats during snorkel surveys on the Nelson Ranch. Early in the season, newly emerged steelhead utilized protected cut bank and emergent vegetation habitats. As flows dropped and the steelhead grew, they utilized transitional point bar habitat then primarily utilized mid-channel aquatic macrophyte throughout the summer.

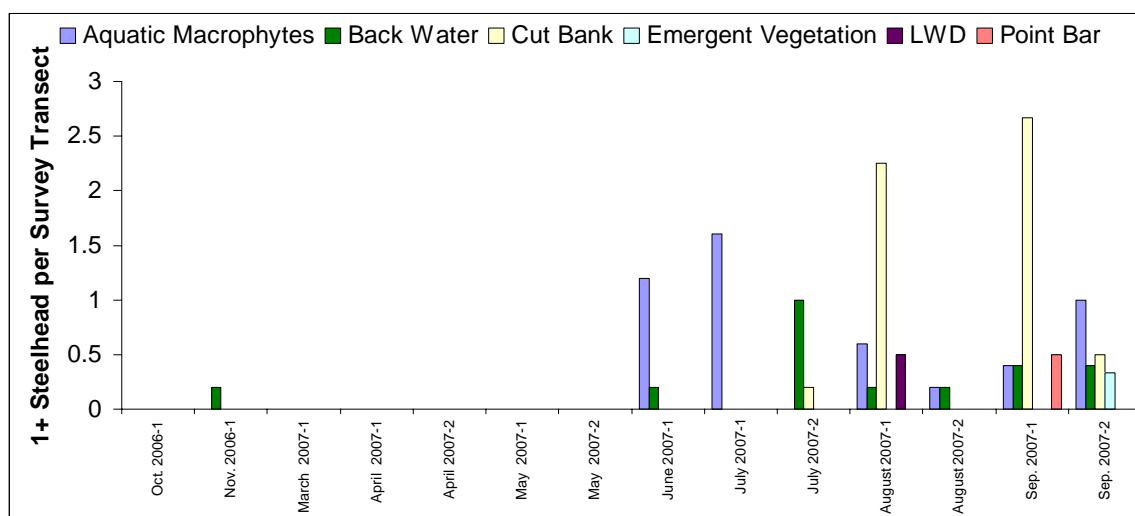


Figure 46. One-year-old or older steelhead observed in various habitats during snorkel surveys on the Nelson Ranch. 1+ steelhead are found primarily in deeper water habitats or locations where food is abundant (aquatic macrophytes).

11.6. Summary

The Nelson Ranch provides unique and potentially very high quality habitat for rearing juvenile salmonids. Of the many available habitats, seasonal growth of aquatic macrophytes creates bioenergetically favorable habitat that provides a substrate for aquatic macroinvertebrates (juvenile salmonid food source) and a velocity refuge for rearing salmonids during summer. For salmonids that have higher temperature tolerances (steelhead and Chinook), growth rates are high and in the case of male Chinook allows for sexual maturation during the first year after hatching. Juvenile coho were observed in deep water habitats where woody debris was present during late spring and early summer, but were not observed as temperatures increased. Despite the abundance of available habitat, water temperatures along the Nelson Ranch are a limiting factor for juvenile coho salmon rearing along the Nelson Ranch. These findings suggest that until water temperature conditions are reduced (e.g., through management actions), juvenile coho will be unable to over summer under current conditions on the Nelson Ranch or other locations downstream with the possible exception of currently unknown locations of thermal refugia.

12. Report Summary

The Shasta River on the Nelson Ranch is a highly productive system with significant potential for restoration of salmonid habitat. The unique hydrology and abundant aquatic macrophytes provide various habitats for fishes during all life stages. Currently the primary limiting factor to salmonids on the Nelson Ranch is elevated water temperature. The quality and quantity of spawning habitat is also low. If water temperatures along the Nelson Ranch can be reduced (e.g., through management actions), then the abundant habitat and high natural productivity could support relatively large populations of salmonids, including the federally- and state-listed coho salmon.

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