Baseline Assessment of Physical and Biological Conditions Within Waterways on Big Springs Ranch, Siskiyou County, California

Jeffrey Mount, Peter Moyle, and Michael Deas, Principal Investigators

Report prepared by:

Carson Jeffres (Project lead), Randy Dahlgren, Joseph Kiernan, Aaron King, Robert Lusardi, Andrew Nichols, Sarah Null, Stacy Tanaka, Ann Willis

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UC DAVIS Center for Watershed Sciences

Watercourse Engineering Inc.

Center for Watershed Sciences
University of California, Davis • One Shields Avenue • Davis, CA 95616-8527

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1.0 Executive Summary

Introduction

The Shasta River in Siskiyou County, California may be one of the Lower Klamath River's more exceptional tributaries (CDFG 2004, Deas 2004, NRC 2004). The river receives more than half of its annual flow from spring complexes. These springs, fed by recharge from Mount Shasta, are nutrient-rich and fuel highly productive aquatic food webs. The natural resilience of the Shasta River, coupled with its high primary productivity, suggests a high potential for significant and immediate response to restoration and conservation actions supporting salmonids.

From 2006 to 2008, the University of California, Davis (UC Davis) Center for Watershed Sciences, in cooperation with Watercourse Engineering Inc., conducted a baseline assessment of aquatic ecosystems within the Shasta River basin. With support from The Nature Conservancy, California (TNC) and the U.S. Bureau of Reclamation, these assessments provided the first-of-their kind comprehensive evaluation of factors limiting salmonid spawning and rearing in the Shasta River and related changes in aquatic ecology over the course of a year (Jeffres et al. 2008). A principal finding of these studies was that degradation of water quality and physical habitat in Big Springs Creek, a large spring-fed tributary, coupled with loss of access to the upper reaches of the Shasta River, were significant limiting factors affecting salmonid spawning and rearing. In particular, the principle limiting factor for coho salmon was high summer temperatures in reaches downstream of Big Springs Creek.

These studies, along with earlier work by Deas et al. (2004), identified Big Springs Creek and the spring complex that feeds it as the highest priority restoration location in the Shasta River. In March 2008 TNC acquired an option to purchase the Busk Ranch (TNC exercised this option on 5 March 2009 and has since renamed the property Big Springs Ranch). The property is approximately 4,100 acres with an additional 400 acres retained by the property owner (with a conservation easement). Along with numerous cold water springs, the property encompasses 4.0 km (2.5 mi) of the Upper Shasta River, 3.5 km (2.2 mi) of Big Springs Creek, 1.6 km (1 mi) of Little Springs Creek, and portions of Parks Creek and Hole in the Ground Creek (a spring creek). The steady influx of cold ($12^{o}C/54^{o}F$) water makes the Big Springs Ranch's spring complex a natural haven for native fishes. Cold, clear water and almost 9.7 km (6 mi) of potential prime salmonid habitat make the Big Springs Ranch one of the most ecologically important parcels in the entire Klamath River watershed.

Big Springs Creek Project

The UC Davis Center for Watershed Sciences and Watercourse Engineering Inc. conducted a baseline assessment of physical and biological conditions within waterways on the Big Springs Ranch. The focus of this effort was to document baseline conditions in the previously undescribed Big Springs Ranch and provide guidance to resource managers in restoration efforts. Specifically, the goal of the baseline assessment program was to support conservation and restoration planning throughout the Shasta River directed toward management of coho and Chinook salmon, and steelhead trout. To that end, the objectives of our study were four-fold:

1) document baseline aquatic habitat conditions in Big Springs Creek and other springs and spring creeks from Summer 2008 through Winter 2009. The sample sites adequately represent lateral and longitudinal gradients as well as ranges of conditions that affect salmonids in these unique systems.
2) establish a monitoring infrastructure and protocols that capture, to the extent possible, seasonal changes in habitat conditions and food web structure.
3) identify and, where possible, quantify factors that limit salmonid production downstream in the Shasta River
4) identify a range of options that may be viable for improved water resource and habitat management that will directly improve salmonid spawning and rearing conditions

This baseline assessment greatly increases understanding of hydrological and ecological processes not only within Big Springs Ranch, but also provides important insights into the Shasta River. Using an interdisciplinary approach, we were able to describe physical and ecological limiting factors affecting salmonids in Big Springs Creek. Study elements included a wide range of field investigations, laboratory investigations, and computer model simulations. We collected physical data documenting hydrology, water temperature, water quality, geomorphology and physical habitat; obtained ecological data through surveys of primary producers, aquatic macroinvertebrates, and, using light stable isotopes, food web structure; and conducted extensive fish surveys to determine seasonal habitat utilization. This data, along with knowledge gained from previous and ongoing studies on the Shasta River, were used to develop preliminary restoration strategies for Big Springs Creek. A two-dimensional hydrodynamic and water temperature model was developed to identify potential flow and thermal benefits associated with selected passive (no direct actions in the stream channel) and active (direct in-channel activities) restoration actions after 1, 5, and 20 years.

Key observations and conclusions identified in this seminal investigation of Big Springs Creek include:

Streamflow

Findings

- During the non-irrigation season portion of the assessment period (1 October 2008, to 8 January 2009) streamflows in Big Springs Creek (mean = 82 ft^3/s) were minimally variable and nearly five times mean streamflows recorded in the Upper Shasta River (above the Big Spring Creek confluence and including Parks Creek inflows; mean = 15 ft^3/s).

- During irrigation season (1 April 2008 to 1 October 2008) Big Springs Creek streamflow declined by 35% (mean = 54 ft^3/s). This seasonal reduction was derived almost entirely from water diversions from Big Springs Lake and apparent reduced spring flow contributions associated with seasonal groundwater pumping local to and upgradient (east and south) of the Big Springs complex. Streamflow magnitudes in the Upper Shasta River above the Parks Creek confluence fluctuated little during the study period. Streamflows in Parks Creek were variable during the irrigation season and doubled (increasing from approximately 6 ft^3/s to 13 ft^3/s) following cessation of irrigation season on 1 October 2008.

Streamflow: Conclusion

The Big Springs complex forms a considerable and important component of baseflow for the Shasta River downstream of Big Springs Creek. Seasonal depletion is evident during summer periods when diversions and possibly groundwater withdrawals deplete flows not only in Big Springs Creek, but also upstream reaches of the Shasta River and Parks Creek.

Water Temperature

Findings

- Big Springs Lake, an artificial impoundment intended to provide water to irrigated agriculture, forms the headwater temperature boundary condition for Big Springs Creek. The lake is fed by a spring complex at the eastern shoreline. Water that discharges from the lake into the creek ranges from approximately 10°C in winter to over 15°C in summer. Considerable spring inflows averaging approximately 11°C contribute to the baseflow of the creek.

- Big Springs Creek is prone to high thermal loading. This stems, in part, from water management practices prior to ownership by TNC, where depletion and tailwater return contributed to heat gain. However, much of the heating is associated with the degraded channel form. More than a century of intense grazing has removed all riparian vegetation. Additionally, grazing has led to erosion of channel banks and the formation of a broad, shallow channel that maximizes heating due to insolation. During daytime in spring and summer, water discharged from Big Springs Lake and the adjacent spring complex warms at a rate greater than 3^{o}C per km (5^{o}C per mi) before reaching the confluence with the Shasta River. However, due to a short transit time, waters within the creek are completely replaced during the night by spring flow. Coupled with local nighttime meteorological conditions, the result is daily lows throughout the summer averaging 11^{o}C to 12^{o}C. These low nighttime values are a potentially valuable attribute for anadromous fish.

- Tailwater associated with flood irrigation practices adjacent to Big Springs Creek can exacerbate thermal loading associated with degraded channel form and lack of riparian shading. Temperatures over 30^{o}C were measured in tailwater return flows in May, 2008, indicating that water management practices are a likely source of warming. However, more information is needed to quantify the water quality impacts of tailwater.

- During summer 2008, the previous landowner reduced grazing pressures within Big Springs Creek. This allowed observation of the impact of seasonal growth of aquatic vegetation on channel form, hydrology and water quality. Left undisturbed, aquatic macrophyte growth caused a narrowing and deepening of the channel through increased river stage, which improved physical habitat for salmonids. Shading associated with extensive vegetation growth, a reduced air-water interface available for heat exchange, and a shorter transit time due to channel narrowing all led to moderated water temperatures in the creek. These observations indicate that even basic cattle exclusion practices can markedly modify channel form and provide direct thermal benefits.

- Modifications to Big Springs Creek include crossings for water pipelines and roads, and modest bank stabilization efforts. One such feature, an inoperative water wheel structure, creates a large backwater in the upper creek, leading to marked increased channel widths (90 m) and shallow depths (0.5 m) resulting in increased thermal loading. Water temperatures increase as much as 3.9^{o}C in the 420 m reach upstream of the water wheel. Model results indicate that removing the water wheel could decrease the maximum heating rate by 1^{o}C; however, temperatures at the mouth of Big Springs Creek would be largely unaffected by the removal of this structure.

Conclusion: Water Temperature

The spring complex that feeds Big Springs Creek is a distributed collection of springs encompassing a fairly large spatial area. Considerable accretions of spring flow at a stable temperature of approximately 11^{o}C create a wide range of thermal conditions. During summer, the springs provide cool water inputs into an otherwise warm system, and in winter the spring flow provides relatively warm waters to a system that would typically be notably colder. Given that historic land and water use practices on the ranch have created a thermally degraded condition, there is considerable potential for restoration of cooler thermal conditions through cattle exclusion and improved irrigation water management.

Water Quality

Findings

- Unlike most rivers, where elevated nitrogen and phosphorous levels are caused by anthropogenic sources, elevated inorganic nitrate (0.39 mg/l) and inorganic orthophosphate (0.16 mg/l) levels in Big Springs Creek are naturally derived from geologic sources along the groundwater flowpath (i.e. from source or recharge area to the Big Springs complex).

- A longitudinal attenuation of nitrate was observed during the spring and summer months as distance increased from the spring source. This decrease is likely inversely proportional to the abundance of aquatic macrophytes in the channel as determined from qualitative macrophyte biomass observations throughout the year. A similar rate of attenuation was not observed in orthophosphate, suggesting that the system experiences nitrogen limitation in Shasta River reaches downstream from Big Springs Creek.

Conclusion: Water Quality

Unique water chemistry in Big Springs Creek includes large, dispersed springs of constant temperature with notable inorganic nitrogen and phosphorus concentrations. These high nutrient levels result is unusually high primary production, which forms a critical base of the food web. This food web is an important element of ecology of Big Springs Creek and is capable of supporting juvenile salmonids.

Geomorphology

Findings

- Cross-sectional channel forms in Big Springs Creek are characterized by predominantly rectangular geometries with large width-to-depth ratios. Mean width to depth ratios observed in Big Springs Creek are more than double those observed in spring-fed creeks throughout eastern Oregon and western Idaho.

- Big Springs Creek exhibits three discrete longitudinal differences in channel slope. Gradient differences are controlled by erosion-resistant bedrock outcroppings in the channel bed and channel margins.

- Qualitative observations suggest fine sediment (sand and silt) transport and depositional dynamics are strongly influenced by aquatic macrophyte growth. Water velocities within and adjacent to macrophyte beds are reduced, resulting in increased sedimentation, particularly within dense macrophyte beds along the channel margins. However, increased water velocities between macrophyte stands (i.e., main channel) promote suspension of fine sediment and the winnowing of fine sediment from available spawning gravels – a beneficial outcome.

- In-channel cattle grazing influenced channel morphology through bank erosion and fine sediment mobilization. Bank trampling appeared to be the dominant source of fine sediment in Big Springs Creek. Furthermore, the removal of aquatic macrophytes through cattle grazing appeared to mobilize fine sediment trapped in macrophyte beds.

- The rock structure that supported the historic water wheel creates a large backwater resulting in extremely wide and shallow channel geometries for approximately 420 meters upstream. Slow water velocities throughout this reach appear to promote fine sediment accumulation.

<u>Conclusion: Geomorphology</u>

The relatively stable spring-dominated hydrology of Big Springs Creek (i.e., the predominance of groundwater-derived baseflows and a lack of large, precipitation-driven flood events) results in stable channel morphologies exhibiting moderate gradients and high cross-sectional width-to-depth ratios. Natural channel change in Big Springs Creek appears largely limited to alterations in bedform configuration due to the growth (and destruction by grazing) of submerged aquatic macrophytes. Channel restoration activities in spring-fed creeks like these require different approaches from snowmelt- and rainfall-dominated systems. Stable, predictable flows enable the use and management of a wide variety of passive actions (e.g., riparian fencing to promote macrophyte and riparian growth) during restoration.

Food Webs

<u>Findings</u>

- Standing crops of both epilithon and aquatic plants increased throughout the study period with the submergent aquatic macrophytes *Myriophyllumsibericum* (northern watermilfoil) and *Polygonumamphibium* (water smartweed) accounting for the bulk of the macrophyte biomass.

- The aquatic macroinvertebrate communities in Big Springs Creek and the Shasta River were dominated by members of the collector-gatherer feeding guild while shredders and invertebrate predators were relatively rare.

- Amphipods (*Hyalella* sp.) were especially abundant in Big Springs Creek during the summer and fall sample periods with densities exceeding 80,000 individuals per square meter of streambed during the fall.

- Natural abundance stable isotope analysis indicated that most primary consumers in Big Springs Creek were deriving their carbon from sources of fine particulate organic matter, epilithic biofilms and attached algae.

- The diets of juvenile salmonids during the spring sample period could not be accurately assessed using stable isotope analysis due to the presence of residual maternal yolk in their body tissues. However, juvenile salmonids had clearly reached isotopic equilibrium with their riverine diets by the summer and fish appeared to be feeding opportunistically on the invertebrate assemblage.

<u>Conclusion: Food Webs</u>
Abundant growth of submergent and emergent macrophytes was a salient feature of Big Springs Creek throughout much of the year. While these plants serve as important habitat for macroinvertebrates and fish, they make limited contributions to carbon flow in the food web prior to senescence, decomposition and entry into the detrital pool. Fine particulate organic matter was the major source of carbon fueling secondary production in Big Springs Creek and members of the collector-gatherer functional feeding group dominated the invertebrate assemblage. While overall taxonomic richness was low, aquatic macroinvertebrate densities are remarkably high throughout much of the year. Collectively, our results suggest that Big Springs Creek has a unique intrinsic potential to provide high-quality rearing-habitat for juvenile salmonids.

Fish and Fish Habitat
<u>Findings</u>

- When water temperatures increased in late May, 2008, approximately 225 juvenile coho from Big Springs Creek and the Shasta River migrated to the pool at the outlet of Big Springs Lake, where they remained throughout the summer and fall. This was the only location where juvenile coho were observed in Big Springs Creek during the summer months.

- Food was never limiting for oversummering coho salmon. Primary production, as fueled by naturally elevated levels of inorganic nutrients (nitrogen and phosphorus) from the springs complex, provides abundant food sources that, coupled with cool summer water temperatures, lead to optimal conditions for growth of coho salmon, albeit in a very small area.

- Relatively warm waters during winter result in the early emergence and rapid growth of juvenile salmonids in Big Springs Creek. Further, warm winter water temperatures allow for growth of aquatic vegetation and benthic invertebrates that provide cover and food for juvenile salmonids rearing in Big Springs Creek.

- During October, adult Chinook salmon returned to spawn in the lower section of Big Springs Creek. Several active redds were found on the lower creek, the only location where suitable gravels currently exist. With adult Chinook present, mature male Chinook parr were observed in the redds and participating in spawning activities. Maturation as parr is a relatively unique life history strategy and is likely the result of the productive spring-fed system.

- A school of adult and juvenile steelhead was observed immediately above the water wheel throughout the study period. The steelhead utilized the relatively deep backwater upstream of the water wheel. It is unknown if these are resident rainbow trout or steelhead oversummering in Big Springs Creek.

Conclusion: Fish and Fish Habitat

Conditions throughout much of Big Springs Creek are too warm for oversummering of juvenile coho salmon. Currently, localized cool water sources with adequate depth are where coho find habitat throughout the summer months. Despite current degraded conditions, attributes that could potentially provide unique and valuable habitat for anadromous fishes, and in particular coho salmon, include nutrient rich spring inflows and unique habitat conditions along upper Big Springs Creek. Springs moderate temperatures in the creek, with relative cool water in summer and warm water in winter. Naturally elevated levels of inorganic nitrogen and phosphorus result in substantial primary production, which in turn fuels the food web that provides abundant, high-quality food for juvenile salmonids rearing in Big Springs Creek.

Restoration Strategies

The data collected and detailed observations made in this study allow for development and evaluation of an array of restoration strategies. These fall into two categories: passive restoration strategies that include actions where no direct in-channel work is carried out, and active restoration actions that include direct in-channel activities.

- Passive restoration strategies
    - Riparian fencing- Excluding and/or management of livestock in the riparian zone can reduce channel bank degradation, allow woody and herbaceous riparian vegetation growth, and in-channel vegetation growth to narrow and deepen the channel. A narrower, deeper channel will reduce heating through a smaller air-water interface and reduced travel time. Coupled with more effective shading from riparian vegetation on a narrower stream, the new channel morphology will lead to reduced temperature throughout the system.

- Tailwater management- Irrigation management actions, such as capture of agricultural tailwater for reuse to eliminate warm inputs to Big Springs Creek will be beneficial to instream water temperatures. Tailwater could also be managed to discharge waters that are not elevated in temperature.

- Management and irrigation efficiency- Improved conveyance, water application rates, field rotation (e.g., hay vs. grazing), retirement of unsuitable lands (e.g., avoid flood irrigating steep lands adjacent to creek), etc., can reduce diversions or modify diversion timing, leaving more cool water in the creek to support anadromous fishes.

- Active restoration
  - Planting emergent and riparian vegetation-Planting of emergent and riparian vegetation to stabilize stream banks and help trap fine sediment. Vegetation should be established above the water wheel to reduce sediment flux to Big Springs Creek prior to removal of the structure.

  - Placement of large woody debris- Currently, instream structure in Big Springs Creek is largely absent, yet has been shown to be a vital component in high quality coho salmon habitat. Instream structures such as large woody debris (LWD) placed in a spring-fed creek will have a much longer lifespan than instream structures placed in a non-spring-fed river due to the absence of high-flow events. Trees placed in the stream will create velocity refugia and overhead cover for rearing juvenile salmonids. Geomorphic impacts of LWD placement will include localized scour of fine sediments, which will increase depths near the LWD.

  - Sediment Management - If active restoration is to take place in the channel, a fine sediment management plan should be in place to monitor sediment flux as a result of restoration activities. This will allow for real-time management to strike a balance between long term restoration of habitat with short term sediment management.

- Modeling Potential Restoration Actions:
  - One element of this study was the development of a two-dimensional water flow and temperature model to assess potential impacts of various actions including increasing flows, narrowing the stream, and providing riparian shading. This proof of concept application has provided key insight into rates of heating along the creek and the implications of different prescriptions on thermal conditions along the creek during summer periods (e.g., the impact of additional riparian shading versus narrowing of the channel). The model is limited to Big Springs Creek and does not include downstream effects in the Shasta River.

- In addition to identifying potential implications of increased flow, reduced channel width, and shade from riparian vegetation, specific modeling assumptions associated with an approximate time frame of when restoration prescriptions would be effective or provide benefit was completed. Time frames of 1, 5, and 20 years were assumed and different assumptions on the extent and efficacy of restoration measures was applied.

- Simulation results suggest an immediate response to cattle exclusion (1 year) with eventual reductions of up to 4^{o}C in mean daily maximum temperatures for long-term restoration conditions (20 years). These conditions will not only provide benefit to Big Springs Creek, but to downstream reaches of the Shasta River.

- The flow and temperature model can interface with existing TMDL models or be extended to include additional water quality parameters. As such, this tool would be available to assess and identify TMDL implementation plan activities, determine potential efficacy of specific actions, and prioritize actions for completion.

- Assumptions employed in the Shasta River TMDL relating to Big Springs Creek were reviewed. Flow assumptions in the TMDL were confirmed with field observations from the 2008 field study – summer flows in Big Springs Creek contribute on the order of 60 ft^{3}/s to the Shasta River. However, TMDL assumptions regarding heating in Big Springs Creek between the lake and the Shasta River were low. Field studies indicate that water released from the lake can exceed 15^{o}C (versus the assumed 12^{o}C). Furthermore, assumptions made about inflow temperatures to the Shasta River under existing conditions and, in particular, during future scenarios were several degrees lower than those observed and modeled under a restored condition. These findings can be incorporated into the TMDL implementation plan activities as appropriate.

- Monitoring is a critical element of any restoration program. To assess the efficacy of restoration prescriptions, baseline monitoring programs must be in place prior to, during, and after restoration. A comprehensive monitoring plan will allow for real-time information gathering that will measure the success of restoration activities and provide guidance if restoration/ranch management actions need to be altered. The report provides specific recommendations for flow, temperature, water quality, geomorphology, food webs, and fish monitoring.

Conclusion: Restoration Strategies

Big Springs Creek and associated springs complex provide multiple attributes that support coho salmon and other fish species of interest. However, land and water use has degraded streamflow and water temperatures, limited seasonal sequestering of nutrients in plant biomass, modified the geomorphology, disrupted food webs, and limited coho and other salmonid production in the creek and in downstream Shasta River reaches.

Beyond formulating baseline conditions for habitat and habitat usage in Big Springs Creek, this project introduces a limited set of potential passive and active restoration actions. These actions are not intended to be exhaustive, but provide fodder for future exploration of opportunities to restore this unique aquatic system as additional research sheds light on critical elements of the creek and associated land use actions (both on site and in the general local area).

Summary and Recommendations

Ecologic, hydrologic and geomorphic assessment activities at Big Springs Ranch indicate that salmonid habitat conditions in Big Springs Creek are severely degraded due to past ranch management. However, during the course of this study, Big Springs Creek demonstrated high resiliency, with significant improvements in conditions with only minor changes in management. Aquatic macrophyte growth was prolific in Big Springs Creek when cattle were excluded from the stream. The aquatic macrophytes added habitat complexity, increased depth, and trapped fine sediment in the margins, revealing suitable spawning gravels in the channel. Despite degraded conditions in much of Big Springs Creek, isolated locations currently exist where juvenile coho are able to grow at rates nearly double that of an adjacent watershed. Using physical and ecological data, a hydrodynamic and temperature model was built to assess restoration alternatives. The model will help ranch managers prioritize restoration options for a rapid recovery of Big Springs Creek.

Despite the large amount of information collected during this study, many questions remain about the unique ecologic conditions in Big Springs Creek, how those conditions will change in response to a range of restoration activities, and how those changes will impact downstream reaches of the Shasta River. For this reason, we recommend continued investment in improving the ecologic and hydrologic models for Big Springs Creek. The baseline dataset developed during this study will be the foundation of a monitoring program that should accompany any restoration effort. This monitoring program will be used to determine degrees of success in restoring Big Springs Creek and to help guide ranch management and restoration activities. The quality of the baseline data and models allows for a novel approach to real-time monitoring and assessment that can be used elsewhere in the Shasta River and the Klamath River basin.

2.0 Introduction

The 1997 National Marine Fisheries Service (NMFS) listing of the SONCC (Southern Oregon/Northern California Coast) evolutionary significant unit of coho salmon (*Oncorhynchus kisutch*) as threatened under the Endangered Species Act (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 River studies 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 for anadromous fish, little information is available about the unique hydrologic and ecologic conditions that exist in the Shasta River.

The Shasta River in Siskiyou County may be one of the more resilient tributaries due to its unique hydrologic/geomorphic conditions and high productivity (Deas et al. 2004, NRC 2004, CDFG 2004), suggesting a high potential for significant and immediate response to restoration and conservation actions. For the past two years, the University of California, Davis Center for Watershed Sciences, in cooperation with Watercourse Engineering (see Jeffres et al. 2008), has been conducting a baseline assessment of aquatic ecosystems on the Shasta River, principally on the Nelson Ranch (approximately 48 kilometers upstream from the confluence with the Klamath River). With support from The Nature Conservancy, California (owners of the Nelson Ranch) and the U.S. Bureau of Reclamation, these assessments provided the first-of-its kind comprehensive evaluation of factors limiting salmonid spawning and rearing habitat and usage of those habitats over the course of a year (Jeffres et al. 2008).

A principal finding of the Nelson Ranch studies was that water volumes and temperatures inherited from upstream sources were the dominant factors limiting the availability of salmonid spawning and rearing habitat in the Shasta River on the Nelson Ranch, particularly for coho salmon. These observations, combined with previous work by Deas et al. (2004), suggested that the magnitude and quality of water sourced from the upstream tributary Big Springs Creek and associated natural springs (herein referred to as the Big Springs complex) played dominant roles in limiting salmonid habitat availability downstream in the Shasta River. The comprehensive field studies at the Nelson Ranch confirmed the Deas et al. (2004) identification of the Big Springs complex region as the highest priority restoration property in the Shasta River basin., Findings of the Nelson Ranch studies helped prompt The Nature Conservancy, California (TNC) to acquire an option to purchase approximately 4,100-acres of ranch land surrounding Big Springs Creek and portions of the upper Shasta River previously identified as the Busk Ranch and herein referred to as Big Springs Ranch (TNC exercised this option on 5 March 2009). Approximately 400 acres of the Busk Ranch were retained by the previous property owner (with a conservation easement purchased by TNC). Along with numerous cold water springs, the Big Springs Ranch property encompasses 4 km of the Upper Shasta River, 3.5 km of Big Springs Creek, 1.6 km of Little Springs Creek, and portions of Parks Creek and Hole in the Ground Creek (spring creek). The steady influx of cold ($12^{o}C/54^{o}F$) water from the Big Springs Complex makes the Big Springs Ranch a natural haven for native fishes. Cold, clear water and almost six miles of potential prime salmonid streams make the Big Springs Ranch one of the most ecologically important parcels in the entire Klamath River watershed.

This project provides the first comprehensive assessment of physical and ecological conditions in the Big Springs Creek region throughout various life stages of salmonids, and complements the previous Nelson Ranch baseline assessment (Jeffres et al. 2008) and an ongoing system-wide wide baseline assessment (funded by U.S. Bureau of Reclamation). A principal objective of this effort was to provide the baseline information necessary to guide and evaluate restoration efforts designed to improve salmonid populations. This research takes advantage of two important events in the watershed. First, this work was coincident with the 2008 coho cohort, the largest of the three brood years in the Shasta River. This relatively large cohort provided the unique opportunity to collect meaningful observations regarding seasonal usage of key habitat types by juvenile coho salmon. Second, by securing an option to purchase a large portion of the former Busk Ranch, The Nature Conservancy (TNC) obtained access to the Big Springs complex and creek.

As identified in the previous work at the Nelson Ranch, the principal limiting factor in coho salmon production is availability of cold water rearing habitat during the over summering lifestage. Observations from the Shasta River indicate that local differences in summer water temperatures force juvenile coho to exhibit two different life history strategies. One life history strategy is for juvenile coho to rear in the Shasta River until increasing springtime water temperatures (4 to 5 months following emergence) prompt downstream emigration in search of cooler water in the Klamath River, tributaries to the Klamath River or the Klamath River estuary. The second life history strategy is for coho salmon to remain in the Shasta River for more than one year, with emigration occurring the second spring following emergence. Due to a dependence on the availability of cool water temperatures throughout the summer, this second life history strategy appears only utilized by fish born near cold water sources in the upper reaches of the Shasta River. As identified herein, considerable cold water resources are available in the Big Springs Creek region, but land and water use practices have severely degraded local conditions. This report focuses on the identification and quantitative characterization of these cold water features and associated aquatic habitat, documentation of cold water habitat use by salmonids, and the development of tools to assess potential restoration strategies in support of TNC's long-term goal to restore Big Springs Creek and the Shasta River to improve and maintain populations of coho salmon and other native fishes. .

2.1 *Report Organization*

The Big Springs Creek region baseline habitat and habitat usage 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.

2.2 *Acknowledgments*

The authors would like to acknowledge the following people who provided support, either directly or indirectly, to the Big Springs Creek baseline assessment and study. 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, and fish collection in the Shasta River; U.S. Bureau of Reclamation, Klamath Basin Area Office for providing funding to compliment this project; Sue Maurer for study design and data collection; Nick Corline and Ben Lamb for sample collection and tireless laboratory work; Robyn Suddeth, Nickilou Krigbaum, Bruce Hammock for field data collection.

We would like to acknowledge TNC for their support throughout this project. Their proactive and bold action in acquiring one of the most important restoration parcels in the Shasta River watershed will potentially have profound impacts on anadromous fisheries in the basin. Thank you Henry Little, George Stroud, Amy Hoss, Chris Babcock, Ada Fowler, and all those behind the scenes at TNC.

We would also like to extend our sincere appreciation to Irene Busk who accommodated a wide range of researchers on her ranch during the project.

Finally we would like to thank the State Water Resources Control Board for funding this project and supporting this work in the Shasta River basin.

3.0 Project Description

The goal of the UC Davis/Watercourse Engineering baseline assessment program is to support conservation and restoration planning throughout the Shasta River directed toward management of coho and Chinook salmon, and steelhead. The objectives of this study necessary to meet these goals are four-fold:

1) document baseline aquatic habitat conditions on Big Springs Creek, and other springs and spring creeks from Summer 2008 through Winter 2009. The sample sites will allow adequate representation of lateral and longitudinal gradients and ranges of conditions that impact salmonids in these unique systems.
2) establish a monitoring infrastructure and protocols that capture, to the extent possible, seasonal changes in habitat conditions and food webs
3) identify and, where possible, quantify factors that limit salmonid production in the Shasta River
4) identify a range of options that may be viable for improved water resource and habitat management that will directly improve salmonid spawning and rearing conditions

Summarized below are the project scope of work and project area. In addition, we discuss some of the challenges associated with working at the Big Springs Ranch during the transition period where TNC held an option to the property, but did not control land and water use activities and had limited access.

3.1 *Scope of Work*

The scope of work outlined herein includes the physical habitat data/observations, water quality characterizations, food web data, and fish abundance and habitat.

Physical habitat data/observations included streamflow monitoring, water temperature observations, geomorphic reconnaissance and habitat mapping throughout the Big Springs Ranch. Streamflow monitoring was completed at locations in the Shasta River, Parks Creek, Big Springs Creek and selected spring-fed tributaries including Hole in the Ground Creek and Little Springs Creek to define the hydrology and quantify spring flow accretions. A wide range of water temperature observations were collected throughout Big Springs Creek to quantify thermal gradients. Geomorphic reconnaissance and habitat mapping were completed within Big Springs Creek to support all aspects of the project.

Water quality characterization included systematically sampling water quality at multiple sites to capture seasonal variations, particularly in nutrients (nitrogen and phosphorus), as well as to source spring waters emanating from the Big Springs Complex.

Food web monitoring included collection and analysis of primary producers (epilithon and aquatic macrophytes), benthic macroinvertebrates, and fishes to examine the temporal and spatial dynamics of the aquatic food web.

Fish abundance and habitat were quantified through extensive snorkel surveys of the Big Springs Ranch. Surveys were tied to physical and chemical habitat characterizations described previously to determine seasonal distribution of salmonids of different age, life history and environmental tolerance.

3.2 *Project Area*

Big Springs Ranch encompasses approximately 4500 acres and part or all of five rivers or creeks: Big Springs Creek, Shasta River, Parks Creek, Little Springs Creek, and Hole in the Ground Creek. Big Springs Creek, the primary focus of the study, is 3.7 km (2.3 mi) long and enters the Shasta River at river kilometer 54.2 (rm 33.7; Figure 1). The Shasta River flows approximately 97 km (60 mi) 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 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 a 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 basalitic lava flows and volcaniclastic 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 25.4 cm (10 in) and 45.7 cm (18 in) (Vignola and Deas 2005), much of which falls as snow in higher elevations during the winter months.

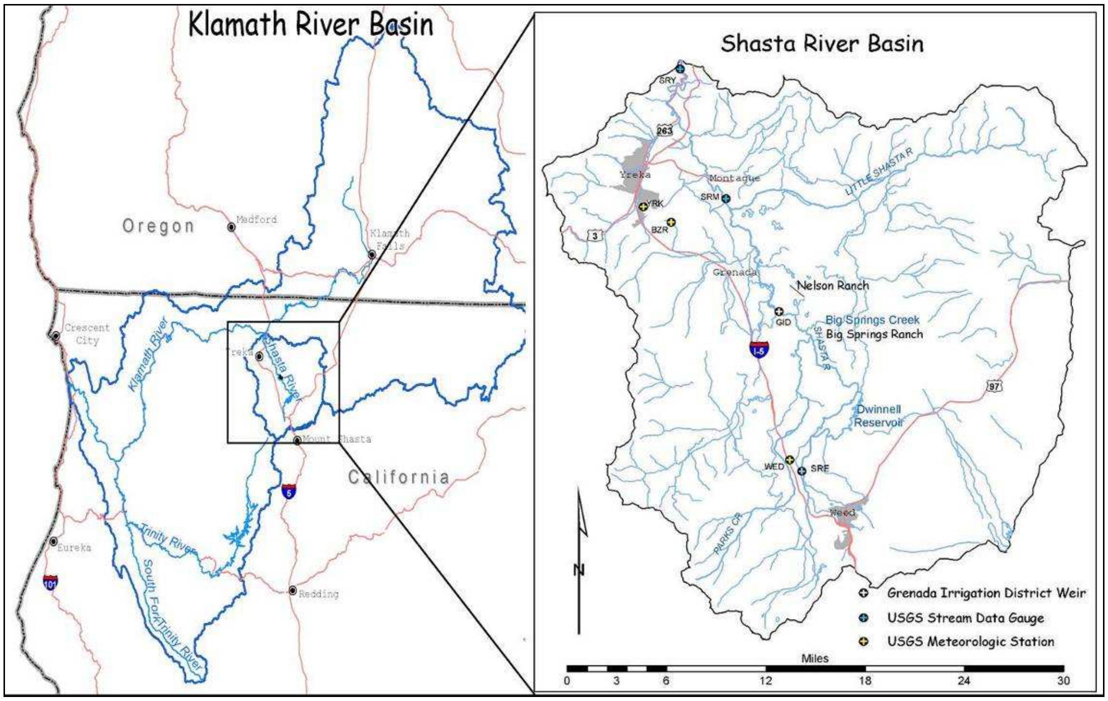


Figure 1. Location of the Shasta River within the Klamath Basin and Big Springs Ranch within the Shasta River Basin.

Big Springs Creek joins the Shasta River at Rkm 54.2 as a major tributary. The creek itself emanates from Big Springs Lake and several discrete springs and flows westward for approximately 3.5 km. Big Springs Lake was impounded in approximately 1875 to support irrigation activities on adjacent lands, and inundated the easternmost portion of the springs complex (i.e., the source water for the lake). Through time an extensive network of irrigation canals and associated features evolved to the current conditions.

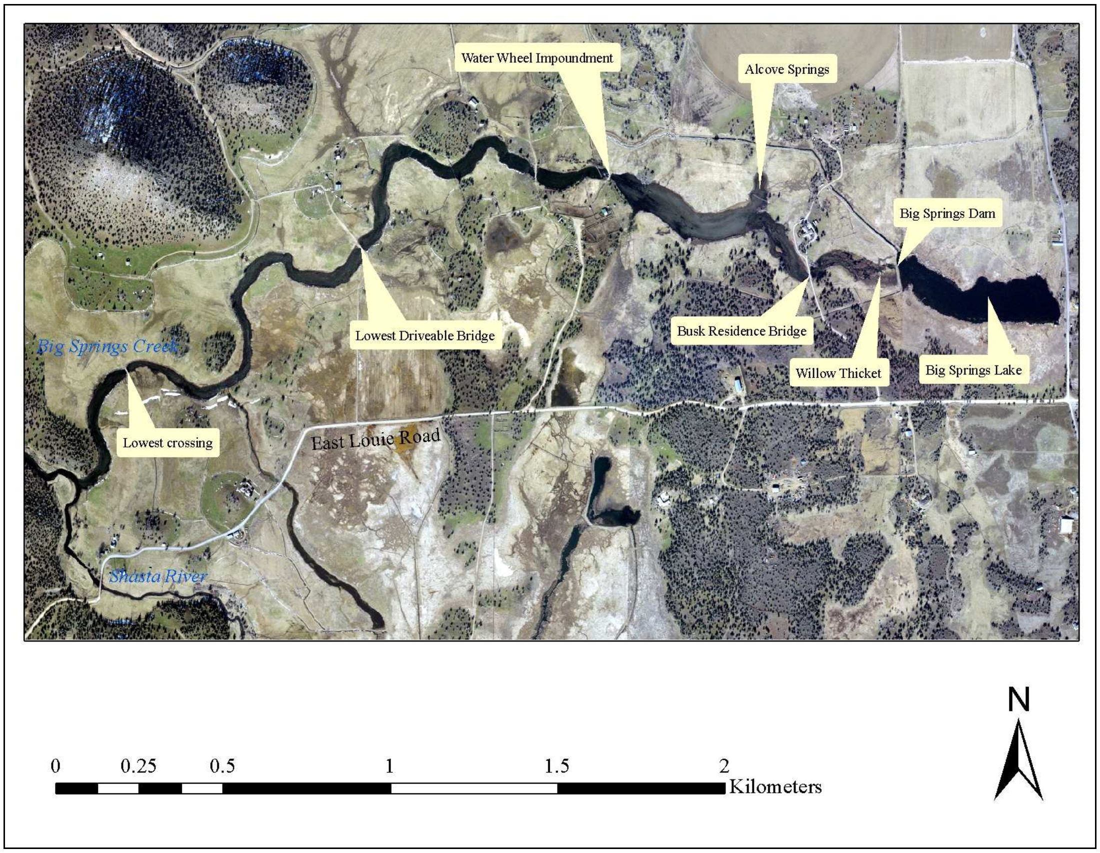


Figure 2. Important locations on Big Springs Creek.

3.3 *Basic Conditions During the Project Period*

During the project, TNC held an option on the Big Springs Ranch, but agreements with the landowner allowed ranching operations to continue. Similarly site visits were limited in number of people and frequency. Research and acquisition related visits were carefully scheduled and clear communications with the landowner were paramount to this project and related efforts. In retrospect, the opportunity to observe conditions under ongoing ranch operations as well as under scaled-back operations provided a unique perspective on the restorative potential of Big Springs Creek.

Figure 3. Example of aquatic macrophyte growth from (a) March, after being grazed throughout the winter and (b) July, after having relatively little cattle grazing pressure.

For example, conditions changed dramatically on Big Springs Creek throughout the project monitoring period, primarily due to land and water use activities associated with cattle management in the ranch. For example, when sampling commenced, Big Springs Creek was devoid of aquatic vegetation and habitat complexity was very low (Figure 3a) in response to long-term cattle management practices. Due to a reduction in the number of livestock maintained on the ranch during summer of 2007, cattle were excluded from most of Big Springs Creek throughout the summer. The result was that extensive growth of aquatic and emergent plants became established and notably altered channel morphology, increasing complexity within the channel (Figure 3b). When cattle were reintroduced to portions of the creek in September, direct grazing on aquatic vegetation resulted in considerable reduction in standing crop, and by late January much of Big Springs Creek looked similar to when sampling began except above the water wheel, where cattle were excluded until early February.

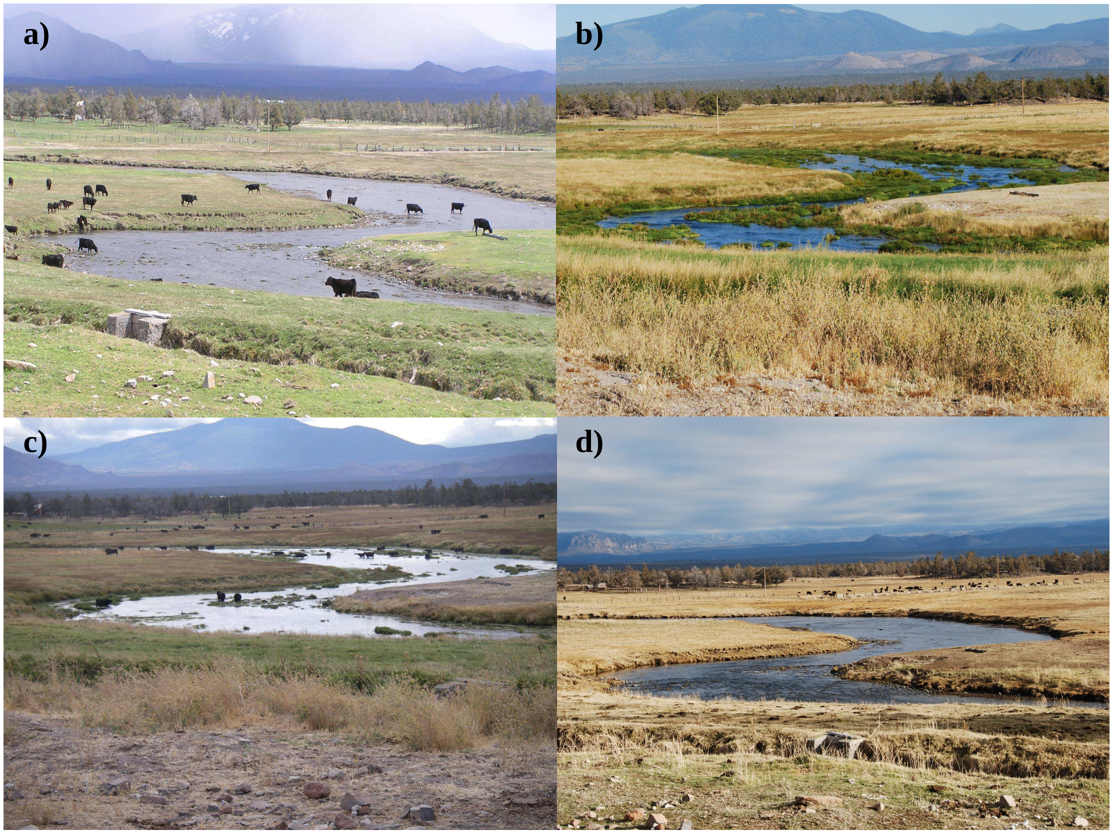


Figure 4. Photo mosaic documenting temporal changes in Big Springs Creek aquatic macrophyte growth. Big Springs Creek during (a) initial conditions after instream cattle grazing in March 2008, (b) after five months of no grazing in September 2008, (c) three weeks after cattle were reintroduced, and (d) after four months of instream cattle grazing.

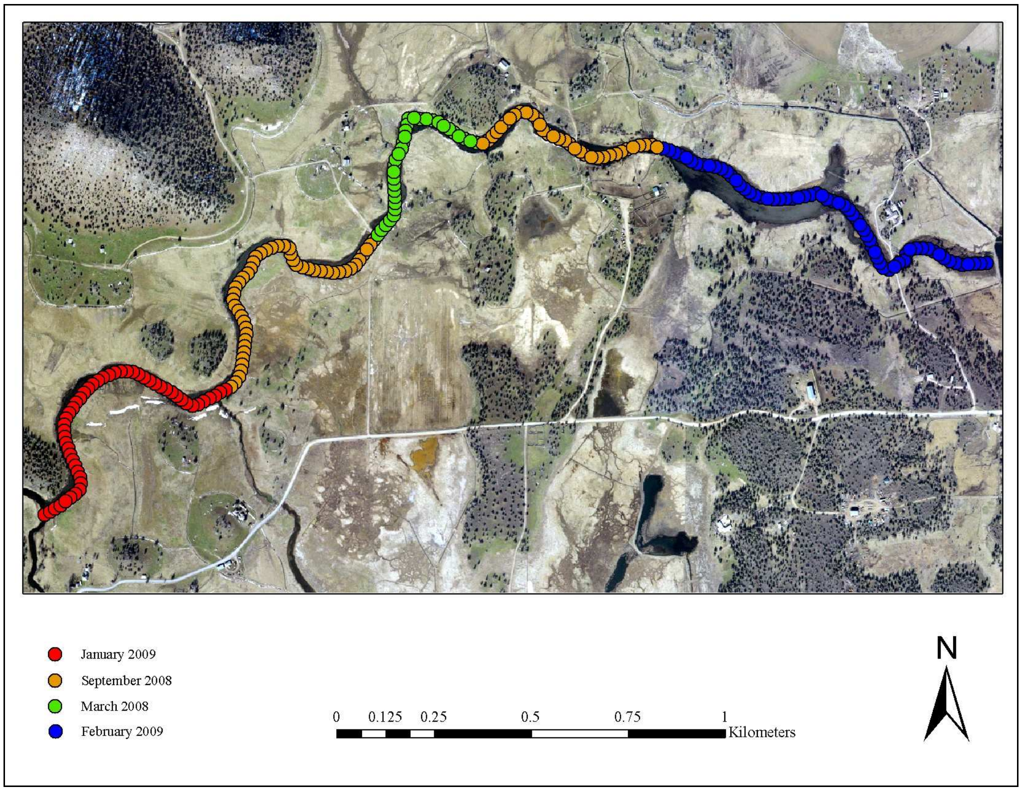


Figure 5. Approximate times when cattle were allowed access to Big Springs Creek.

Another unique condition for this project was that access had to be scheduled in advance. Access to Big Springs Ranch for research purposes was generally conducted every other week throughout the study. However, from 7 October 2008 to 13 November 2008, five weeks of sampling were missed due to restricted access. After 13 November biweekly access was resumed throughout the remainder of the study period.

4.0 Physical habitat data/observations

Physical habitat data/observations were collected based on protocols established at Nelson Ranch (Jeffres et al. 2008) and included flow monitoring, temperature observations, geomorphic reconnaissance and habitat mapping (habitat mapping is addressed under fish abundance and habitat survey).

4.1 *Flow Monitoring*

Nearly all of the water in the Shasta River flows through the Big Springs Ranch. Along the southern ranch boundary, snowmelt and rainfall runoff, as well as spring-fed streamflows in Parks Creek combine with the predominantly spring-fed streamflows in the Upper Shasta River[1] and Hole in the Ground Creek (Figure 6). Approximately 2 km (1.2 mi) downstream, substantial contributions from the spring-fed Big Springs Creek nearly quadruple mean annual discharge in the Shasta River. Quantifying streamflows throughout Big Springs Ranch was a critical step in understanding the spatial and temporal variability of water supplied to the Shasta River, and associated water temperature and quality conditions.

4.1.1 Methods

To assess streamflow conditions in the Shasta River, Big Springs Creek, and associated tributaries, nine streamflow gauging stations were installed on Big Springs Ranch (Figure 6), augmenting existing streamflow monitoring efforts on the Nelson Ranch approximately 2.7 kilometers downstream on the Shasta River. Four stream gauges were installed in Big Springs Creek, one gauge in the northerly Big Springs Ranch irrigation diversion from Big Springs Lake, and one in each of the tributaries to Big Springs Creek and the Shasta River: Upper Shasta River, Parks Creek, Hole in the Ground Spring and Little Springs Creek. Timing of gauge installment and duration of use are provided in Table 1.

Table 1: Dates of operation for stream gauges on Big Springs Ranch. With the exception of gauges in the Big Springs Lake North Diversion and Lake Outlet, all stream gauges continue to be operated.

Stream Gauge	Dates of Operation
Big Springs Lake - North Diversion	6/9/2008 to 10/2/2008
Big Springs Lake - Outlet	6/9/2008 to 8/1/2008
Big Springs Creek - Busk Residence Bridge	7/22/2008 to 1/9/2009
Big Springs Creek - Water Wheel	3/26/2008 to 1/9/2009
Big Springs Creek - Lowest Bridge	3/26/2008 to 1/9/2009
Little Springs Creek	10/1/2008 to 1/9/2009
Upper Shasta River	3/26/2008 to 1/9/2009
Parks Creek	3/26/2008 to 1/9/2009
Hole in the Ground Creek	6/9/2008 to 1/9/2009

[1] The Shasta River above Parks Creek and Parks Creek are impaired due to diversion to and impoundment at Dwinnell Dam. Under predevelopment conditions precipitation and snowmelt events probably provided appreciable flow to the Shasta River above Big Springs Creek during the winter and spring. However in the late summer and fall – even under pre-development conditions – flows in Parks Creek and the Shasta River above Big Springs Creek probably fell to seasonal lows. Thus Big Springs Creek provided the majority of baseflow to downstream Shasta River reaches during critical summer and fall periods. Additional details can be found in Deas et al. (2004)

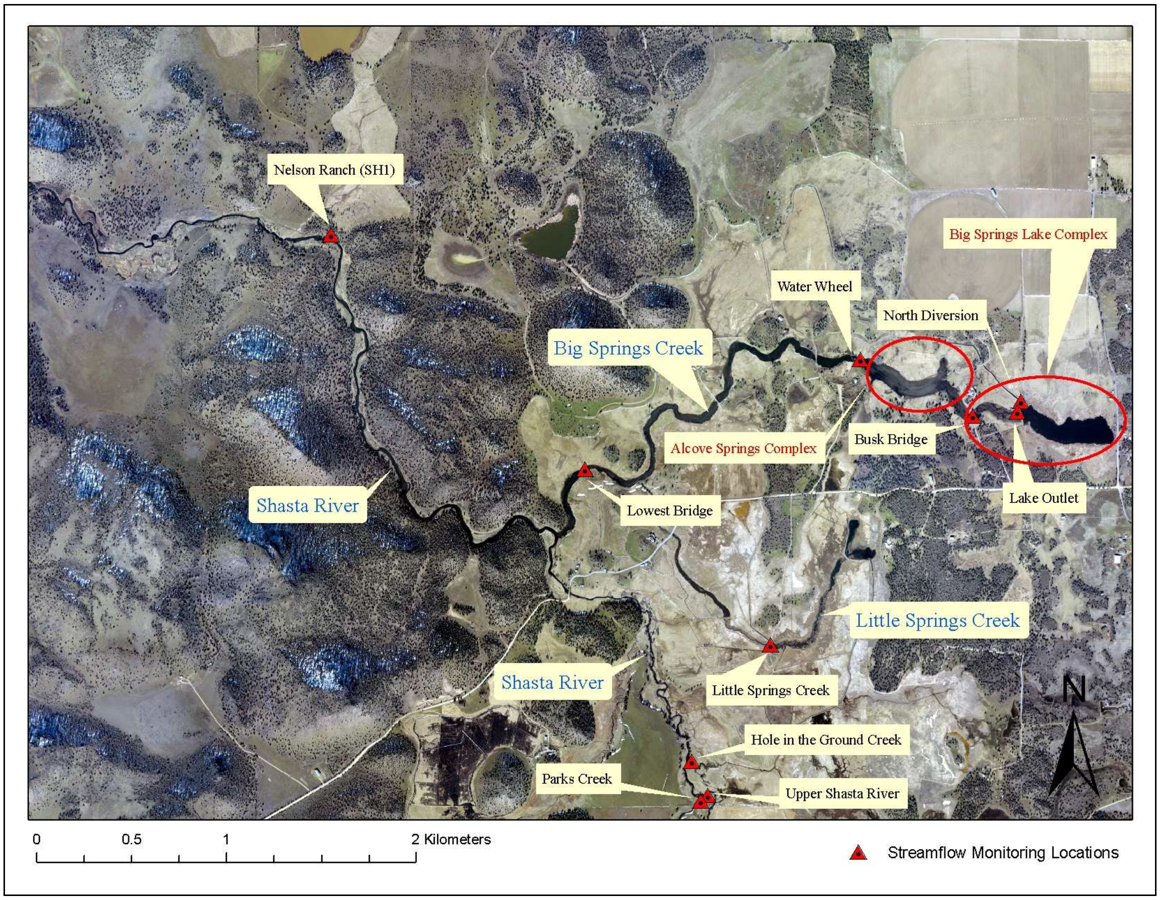


Figure 6. Big Springs Ranch stream gauge locations.

Stream gauge locations in Big Springs Creek were chosen to quantify flow rates and changes in flow through time with respect to 1) streamflow releases from Big Springs Lake; 2) spring/groundwater accretion between Big Springs Lake and the Busk Residence Bridge; 3) spring/groundwater accretion from a large spring complex adjacent to the Busk residence; and 4) streamflow accretions from Little Springs Creek. Stream gauges in the Upper Shasta River, Parks Creek, and Hole in the Ground Creek allowed timing and quantification of tributary inputs to the Shasta River above the Big Springs Creek confluence.

Streamflow was measured using standard methodologies (Rantz 1982). Point velocities were measured at 0.6 of the stream depth using a Marsh-McBirney Flo-Mate electromagnetic velocity meter attached to a top-set wading rod. Vertical cells were such that no more than 10 percent of the flow at a cross section was within a single cell. USGS mid-section velocity-area methods (Rantz 1982) were used to calculate discharge by integrating water velocity and depth across each vertical. Measured discharges and river stage data collected at 10-minute intervals with Global Water WL-16 submersible pressure transducers were used to quantify stage-discharge relationships (i.e. rating curves) for each stream gauge. Rating curves were subsequently used to estimate streamflow at each gauge location.

Construction of a beaver dam approximately 100 meters below the Big Springs Lake outlet forced an abandonment of the Big Springs Lake Outlet stream gauge on 1 August 2008. As such, the Busk Residence Bridge gauge was used to quantify and characterize streamflow from the dam outlet, and thus the Big Springs Lake spring complex. Also, extensive aquatic macrophyte growth around the lowermost gauge on Big Springs Creek prevented the development of an accurate stage-discharge rating curve at this location. Interestingly, this same confounding factor was identified during flow quantification for the adjudication in the 1920's (DPW 1925). A consequence of the observed macrophyte growth was a doubling of river stage (i.e. depth) for nearly identical streamflow magnitudes (Figure 7) Furthermore, a strong correlation between discharge and stage on Little Springs Creek was not established due to lack of flow and limited variability in flow measured to date. While discharge measurements and corresponding river stage measurements for both gauges are provided (appendix), associated preliminary rating curves and 10-minute streamflow data are not included in this report.

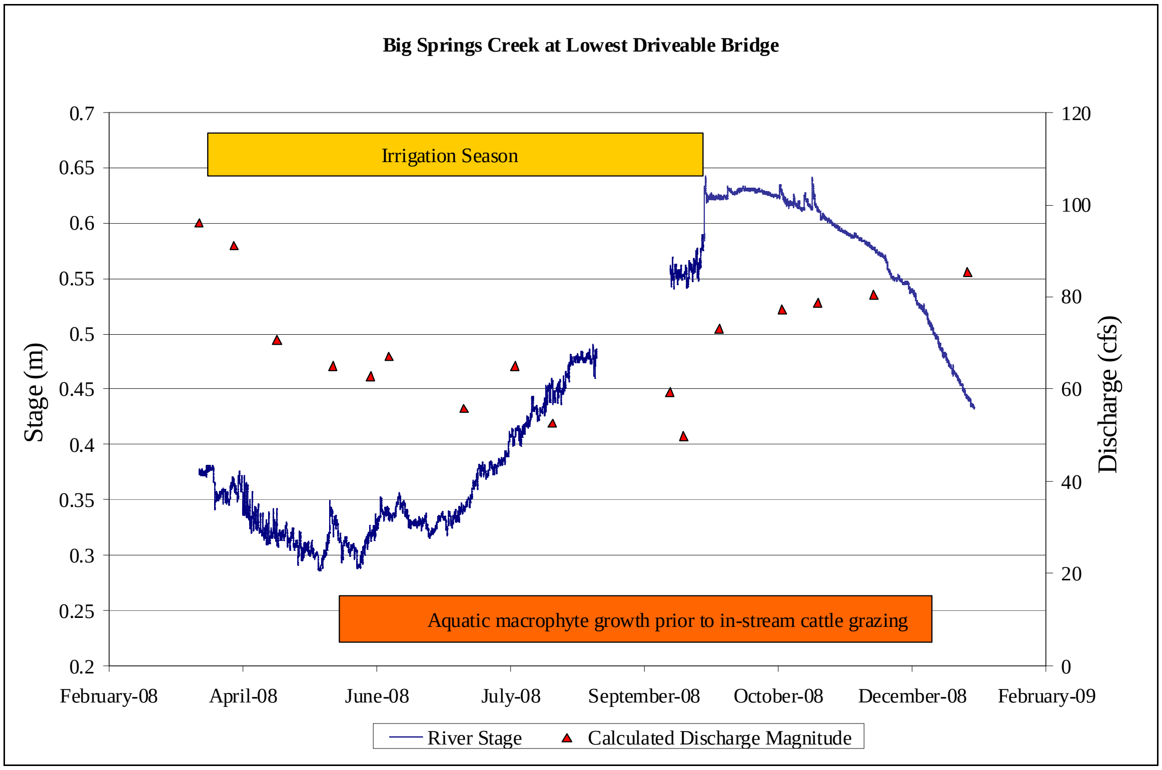


Figure 7. Extensive growth of aquatic macrophytes around the lowest stream gauge in Big Springs Creek approximately doubled observed river stage (i.e. depth) for nearly identical streamflows magnitudes. Removal of aquatic macrophytes through in-stream cattle grazing reduced stream depth in January 2009.

4.1.2 Data Analysis

Although the project area focuses largely on Big Springs Creek, both the creek and the Shasta River and tributaries on Big Springs Ranch are discussed because of their overall relation and importance to the downstream Shasta River reaches.

Big Springs Creek

Big Springs Creek exhibits a complex hydrologic regime dominated by stable spring-fed baseflows upon which are imposed temporally and spatially variable surface water diversions and groundwater pumping associated with irrigation season (1 April to 30 September) (Figure 8). Streamflow data identified two large natural spring complexes in the upper 1.5 km (0.9 mi) of Big Springs Creek. The first spring complex (herein referred to as the Big Springs Lake complex) enters the creek system as distributed or diffuse inputs from the top of Big Springs Lake to the bridge crossing at the Busk residence (Figure 6). Mean non-irrigation streamflows at the Busk Residence Bridge, which likely represent the magnitude of current unimpaired discharge from the Big Springs Lake complex, were 35.5 ft^3/s ($\sigma = 2.64$). The second spring-complex (herein referred to as the Alcove Springs complex) also exhibits a distributed or diffuse inflow, extending from the Busk Residence Bridge to the water wheel impoundment in Big Springs Creek (Figure 6). The difference in streamflow magnitude between the Busk Residence Bridge gauge and the water wheel gauge was used to quantify discharge from the Alcove Springs complex. Mean non-irrigation streamflow at the water wheel impoundment was 82.4 ft^3/s ($\sigma = 3.86$), indicating unimpaired streamflow accretion from the Alcove Springs complex is approximately 47 ft^3/s (Table 2). Streamflow accretion below the waterwheel impoundment and across the lower 2.7 kilometers of Big Springs Creek is minimal, principally reflecting nearly constant 5.5 ft^3/s non-irrigation season streamflows from Little Springs Creek. Flows for Little Springs Creek are likely underestimated due to unregulated head gates that diverted water (estimated at 1-2 ft^3/s) from Little Springs Creek to fields to the north and south of the Creek even after the end of irrigation season.

The exercising of riparian water rights during the irrigation season substantially reduced mean streamflows in Big Springs Creek below Big Springs Dam. Additionally, it appears that seasonal groundwater extraction in the Big Springs area and possibly throughout upgradient regions to the east of Big Springs, may have a direct effect on the production of these large spring complexes as identified by Watercourse (2006). Big Springs Ranch diverted an average of 8.1 ft^3/s from Big Springs Lake for transmission along the property's northern diversion ditch between June and September 2008. Additional diversions by other water users from Big Springs Lake are unquantified; however documented water rights to Big Springs Lake total approximately 47.5 ft^3/s. Mean irrigation-season streamflows immediately below Big Springs Dam and at the Busk Residence Bridge were 7.6 ft^3/s ($\sigma = 2.59$) and 10.3 ft^3/s ($\sigma = 3.39$), respectively. While mean irrigation season streamflows derived from the Big Springs Lake spring complex were notably reduced (-70%) and significantly more variable than non-irrigation streamflow conditions, spring accretions from the Alcove Springs complex exhibited minimal variability throughout the entire period of record, with average irrigation-season spring accretions (44 ft^3/s) exhibiting only a 6% reduction from non-irrigation season magnitudes (47 ft^3/s). This suggests observed irrigation-season streamflow variability in Big Springs Creek was largely derived from operations of Big Springs Dam and associated water diversions. While groundwater pumping in the vicinity of Big Springs Lake likely impacted irrigation streamflows measured in Big Springs Creek, this impact could not be quantified with available data. It should also be noted that temporally and spatially diffuse irrigation tailwater returns represented an unquantified volume of streamflow input along the entire length of Big Springs Creek.

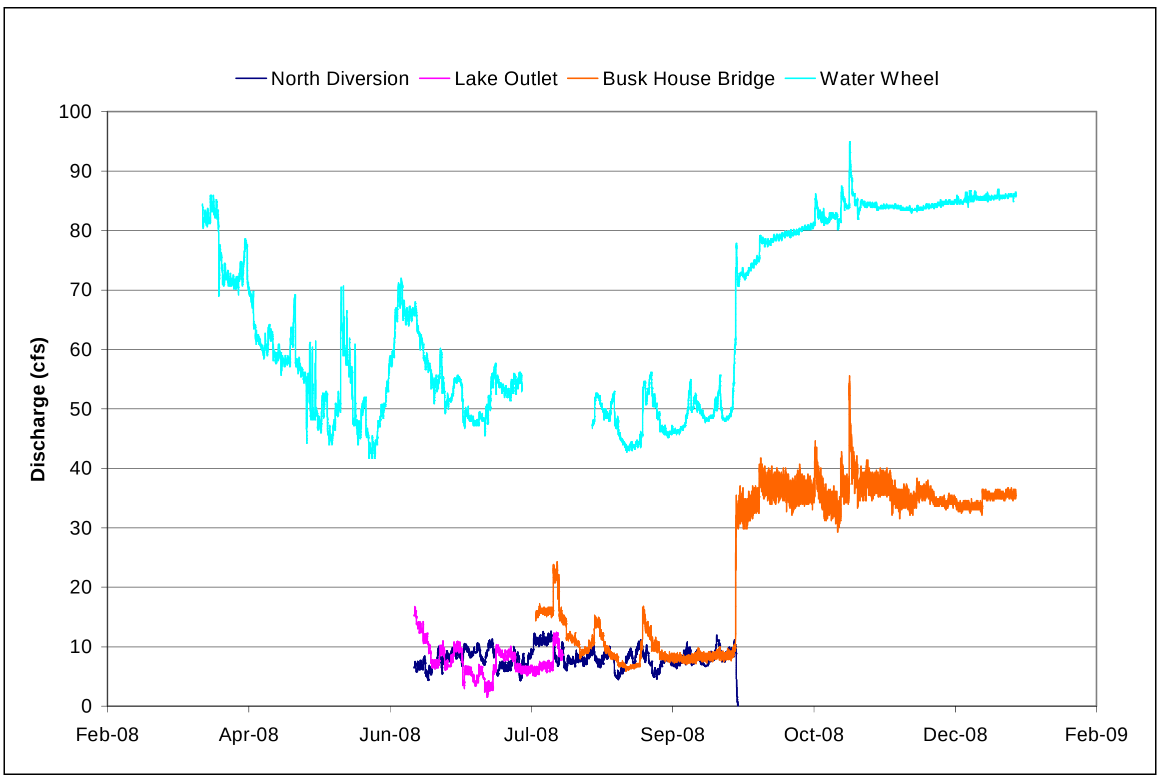


Figure 8. Calculated discharge at the North Irrigation Diversion from Big Springs Lake, Big Springs Creek below Big Springs Lake, Busk Residence Bridge, and Water Wheel.

Table 2. Streamflow statistics for gauges on Big Springs Creek. All measurement units are cubic feet per second (ft^3/s).

	Big Spring Lake - North Diversion	Big Springs Lake - Outlet	Busk Bridge	Water Wheel
All Data (March 26, 2008 to January 9, 2009)				
Mean	8.06	7.60	25.07	65.53
Median	8.02	6.99	33.63	61.93
Max	12.62	16.71	55.56	94.91
Min	0.00	1.47	5.90	41.77
Standard Deviation	1.59	2.59	12.78	15.32
Irrigation Season (April 1, 2008 to September 30, 2008)				
Mean	8.08	7.60	10.32	54.30
Median	8.02	6.99	8.89	52.18
Max	12.62	16.71	24.29	83.42
Min	4.25	1.47	5.90	41.77
Standard Deviation	1.52	2.59	3.39	8.07
Non-Irrigation Season (March 26 to 31, 2008; October 1, 2008 to January 8, 2009)				
Mean	--	--	35.54	82.43
Median	--	--	35.48	83.93
Max	--	--	55.56	94.91
Min	--	--	9.28	59.92
Standard Deviation	--	--	2.64	3.86

Shasta River and Tributaries

Stream gauges located in the Upper Shasta River, Parks Creek, and Hole in the Ground Creek (Figure 6) measured what is estimated to be 90-95% of the streamflow in the Shasta River above Big Springs Creek. Unquantified streamflow accretions come from numerous small springs and a secondary channel of Parks Creek (Figure 9).

Upper Shasta River

The Upper Shasta River stream gauge was used to quantify streamflows derived from the combination of water releases out of Dwinnell Dam and unmeasured spring accretion between the dam and the southern boundary of Big Springs Ranch. Releases from Dwinnell Dam are typically 10 ft^3/s throughout the year and solely used to meet volumetric needs of riparian water rights holders downstream. However, the magnitudes of streamflow releases between March 2008 and January 2009 are unknown and may even be significantly less than 10 ft^3/s due to a lack of water in Lake Shastina. Streamflows measured in the Upper Shasta River over the period of records were small (mean = 5.04 ft^3/s), minimally variable (σ = 1.39) (Table 3) and may largely reflect natural spring accretion between Dwinnell Dam and Big Springs Ranch.

Parks Creek
Streamflows measured at the mouth of Parks Creek reflect a hydrologic regime characterized by spring-fed baseflows augmented by snowmelt and rainfall runoff. However, upstream water resources development, including the regulation of numerous springs tributary to Parks Creek, appeared to strongly regulate observed downstream flows. Highly variable streamflows dominated by rapidly increasing and decreasing discharge magnitudes during the spring snowmelt and observed flows approaching zero flow occurred during the summer months (Table 3). Moderate non-irrigation period streamflows (mean 10.16 ft^3/s; $\sigma = 4.90$) reflected minimal precipitation and snowmelt over the gauged period of record, particularly between October 2008 and January 2009 (Table 3). Consequently, streamflows measured over the period of record largely reflect spring-fed baseflows. An anomalous and temporary streamflow peak (46 ft^3/s) on 4 October, 2008 may have resulted from upstream operations because no precipitation occurred during this period.

Hole in the Ground Creek
Streamflows in Hole in the Ground Creek strongly reflect its existence as a regulated spring-fed creek. Measured streamflows were minimally variable throughout the period of record (Table 3). However, upstream diversions reduced mean streamflow from 6.22 ft^3/s to 4.83 ft^3/s during the irrigation season.

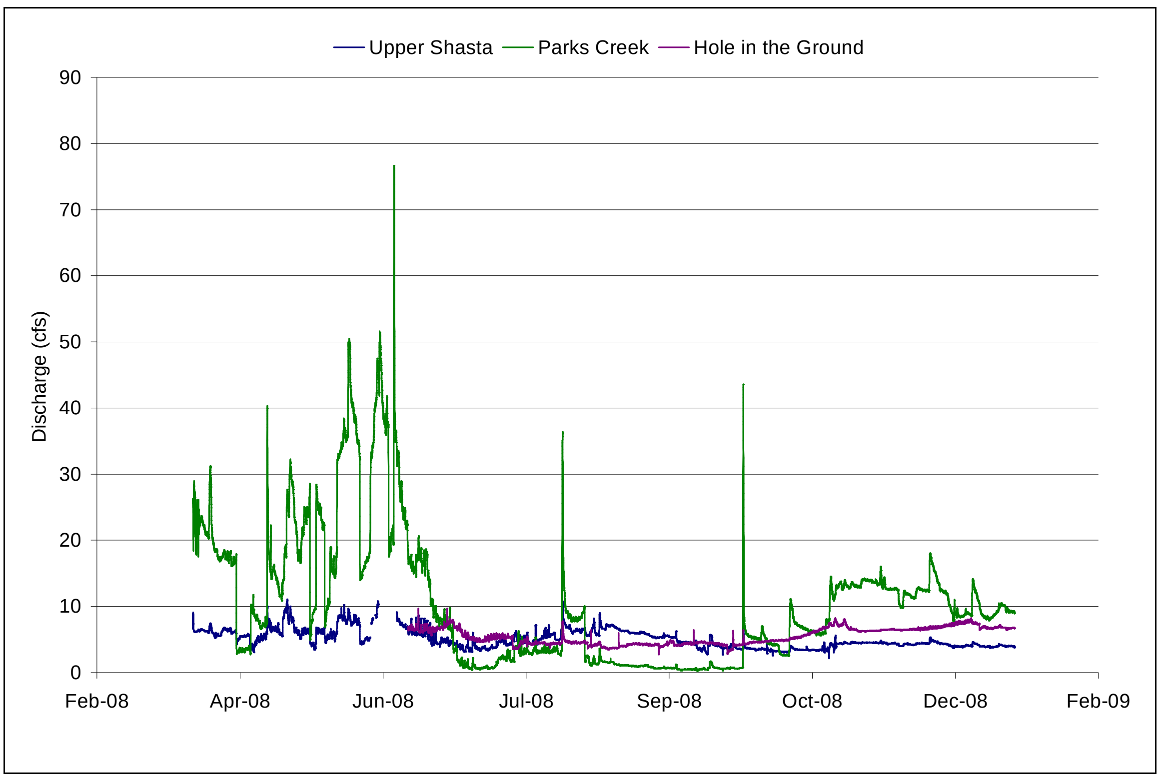


Figure 9. Calculated discharge at Shasta River above Parks Creek, Parks Creek, and Hole in the Ground Creek.

Table 3. Streamflow statistics from gauges located on the Upper Shasta River, Parks Creek and Hole in the Ground Creek. All measurement units are cubic feet per second (ft^3/s).

	Upper Shasta River	Parks Creek	Hole in the Ground Creek
All Data (March 26, 2008 to January 9, 2009)			
Mean	5.04	10.30	5.48
Median	4.55	8.35	5.18
Max	11.08	76.65	9.68
Min	2.15	0.24	2.70
Standard Deviation	1.39	9.94	1.22
Irrigation Season (April 1, 2008 to September 30, 2008)			
Mean	5.60	10.39	4.83
Median	5.58	3.76	4.43
Max	11.08	76.65	9.68
Min	2.68	0.24	2.70
Standard Deviation	1.39	11.89	1.02
Non-Irrigation Season (March 26 to 31, 2008; October 1, 2008 to January 8, 2009)			
Mean	4.11	10.16	6.22
Median	4.06	9.97	6.48
Max	9.36	43.58	8.27
Min	2.15	0.54	3.41
Standard Deviation	0.72	4.90	0.99

4.1.3 Conclusions

Streamflows measured on the Big Springs Ranch effectively capture contributions to the Shasta River from Parks Creek, Hole in the Ground Creek and Big Springs Creek. Observed differences in hydrologic regime characteristics (i.e. streamflow magnitude and variability) throughout the period of record for each tributary appear largely derived from: 1) differences in sub-watershed scale streamflow generation processes (i.e. snowmelt/rainfall runoff versus spring flow); and 2) irrigation season water management along each tributary. Such differences in hydrologic regime and water management, which are inherited by the Shasta River below Big Springs Creek, are summarized below:

- Streamflows in the Shasta River above Parks Creek were small, minimally variable and represented the combination of small releases from Dwinnell Dam, unquantified natural spring accretions below the impoundment, and unquantified irrigation return flows.

- Streamflows in Parks Creek reflected a hydrologic regime characterized by spring-fed baseflows augmented by snowmelt and rainfall runoff. However, unquantified upstream water resources development between 1 April 2008 and 1 October 2008, including both in-stream diversions and regulation of numerous springs tributary to Parks Creek, appeared to strongly regulate observed flows. Streamflow magnitudes measured during the period of record peaked during the spring-snowmelt (77 ft^3/s), and decreased rapidly in the early summer to less than 1 ft^3/s.

- Streamflows measured in Big Springs Creek reflected a hydrologic regime dominated by large magnitude spring-fed baseflows (80 to 90 ft^3/s), which are reduced by approximately 35% between 1 April 2008 and 1 October 2008 by temporally variable (5 to 12 ft^3/s) irrigation water diversions from Big Springs Lake and unquantified groundwater pumping locally and upgradient (east and south) from the Big Springs complex. Measured mean non-irrigation season, spring-fed baseflows (82 ft^3/s) were nearly five times the magnitude of mean non-irrigation season streamflows derived from both Parks Creek and the Upper Shasta River (~15 ft^3/s) during the period of record.

- Streamflows in Big Springs Creek, Parks Creek and Hole in the Ground Creek increased following cessation of irrigation season practices on 1 October, 2008. Given the lack of snowmelt or rainfall-derived runoff during early October, the observed rapid increase in streamflow suggests large contributions from stable springs and groundwater sources in these tributaries to the Shasta River.

- Combined streamflows measured in the Upper Shasta River, Parks Creek, Hole in the Ground Creek and Big Springs Creek comprise approximately 90% of streamflows measured in the Shasta River 2.7 kilometers downstream on the Nelson Ranch. The difference between streamflows quantified on the Busk Ranch and those measured downstream in the Shasta River at the Nelson Ranch is attributable to several small (and unquantified) springs entering the Shasta River throughout the Busk Ranch property.

4.2 *Water Temperature*

The Big Springs complex has been previously identified as producing notable flows at near constant temperatures (NCRWQCB 2004). As noted before, these circumstances produce temperatures of approximately 11°C, which are near ideal for anadromous fish, particularly coho salmon, due to relatively warm temperatures in winter and cool temperatures in summer. Several temperature investigations occurred during the study period in the Big Springs Creek project area. These included monitoring temperatures in Big Springs Lake, monitoring selected longitudinal locations in the creek and Shasta River, identification of spring sources and assessments of thermal diversity through direct observation and thermal imagery, and simulation modeling. Some of these programs were completed in cooperation with other agencies and funding sources, including The Nature Conservancy and U.S. Bureau of Reclamation, Klamath Basin Area Office. These investigation and principal findings are outlined below.

As part of the project a two-dimensional flow and temperature model was constructed to assess thermal conditions associated with different flow regimes, potential future riparian vegetation shading configurations, variable channel geomorphic forms (e.g., narrower and deeper) to support potential restoration strategies. The temperature monitoring work addressed herein also supported modeling. The modeling element is addressed more fully under Restoration Strategies and in the appendix.

4.2.1 Methods

Water temperature field monitoring occurred primarily through the direct deployment of remote logging thermistors. HOBO® Pro v2 Water Temperature Data Loggers from Onset Computer Corporation were used to collect information at 30 minute increments throughout the project area. These loggers have a resolution of approximately 0.03^{o}C (0.02^{o}C at 25^{o}C) and an accuracy of ±0.2^{o}C over the range from 0^{o}C to 40^{o}C, and a 90% response time of 5 minutes in water (Onset 2009). Instruments were deployed consistent with protocols developed on the Nelson Ranch (Jeffres et al., 2008).

Other instruments and approaches used to measure water temperature include handheld temperature devices and high resolution thermal infrared (TIR) imagery. Handled devices were used to spot check return flows, identify cold water sources, and generally explore thermal conditions and diversity throughout the project area. For handheld investigations the water depth was measured with a Global Water pressure transducer (model WL 16) accurate to +/-0.2% in the 0-21°C range, and a Tech Instrumentation model TM99A temperature unit with a model 2007 probe was used for temperature. The TM99A temperature unit is accurate to ± 0.1°C in the 0-40°C range. The pressure transducer and TM99A temperature unit were mounted to Plexiglas on a 1.8 m rod (6 ft). Probe tips were attached to the end of the rod, and the rod was marked in 0.3 m (1 ft) increments. Temperature and depth measurements could then be taken simultaneously in water up to 1.5 m. The handheld device allowed quick assessment of vertical distribution of water and streambed temperature, with the ability to explore under overhanging vegetation, cutbanks, and into other types of cover elements.

TIR was flown for morning and afternoon conditions as part of the U.S. Bureau of Reclamation funded project (for a larger spatial area, but including Big Springs). These data were useful in furthering our understanding of thermal conditions in Big Springs Creek. Details of TIR work can be found in Watershed Sciences (2009).

4.2.2 Data/Analysis

The temperature data collection efforts provided a basis for several assessments of thermal conditions in Big Springs Lake, longitudinal characteristics of Big Springs Creek, and general thermal diversity of the creek (e.g., springs sources and lateral variability).

Big Springs Lake

Although not explicitly included in the scope of work for this project, an opportunity to monitor temperatures in Big Springs Lake provided invaluable information on the source water temperatures for Big Springs Creek.

Big Springs Lake is an impounded reach of Big Springs Creek. The reservoir is approximately 520 m long, a maximum width of 150 m, and is generally east-west in orientation (Figure 10). Field reconnaissance suggests that there is a large springs complex (the aforementioned Big Springs Lake Complex) at the east end of the lake, although there may be additional sources along the reservoir. Three vertical strings of three thermistors each were deployed in the lake at the east end, middle, and west end with loggers placed near surface, mid-water column, and near bottom. Not all data were available at the time of publication. Nonetheless there are clear findings from this supplemental study.

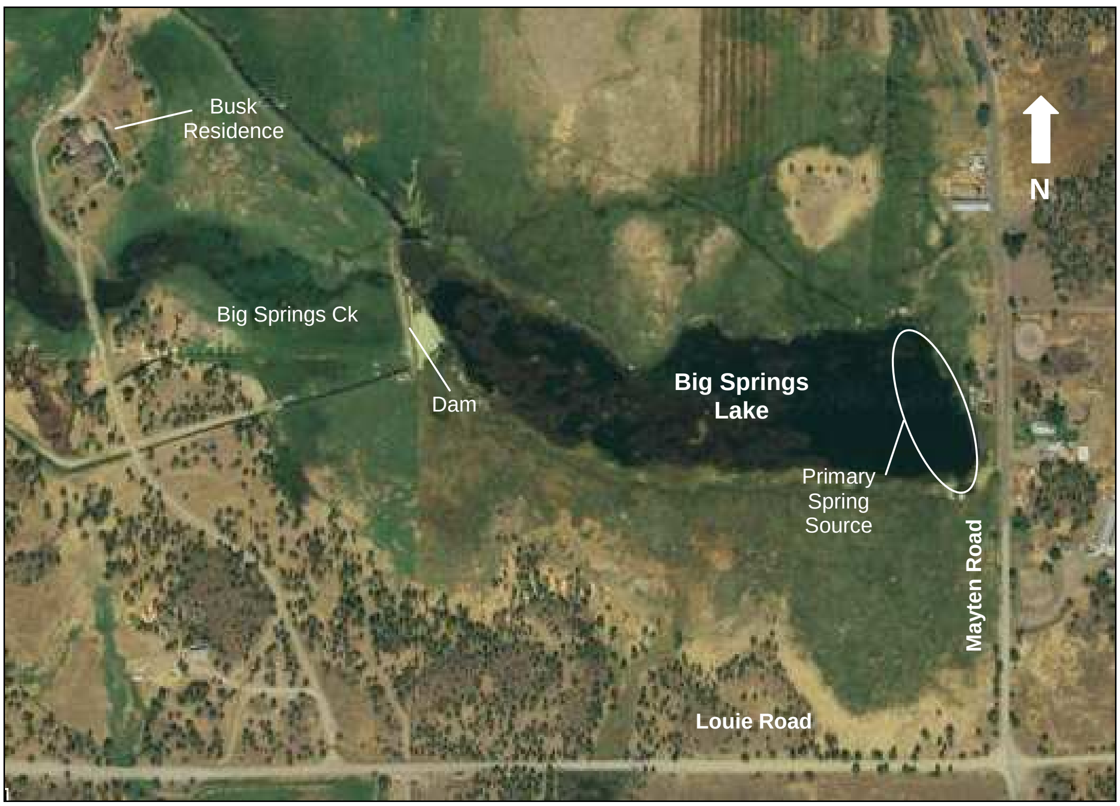


Figure 10. Big Springs Lake

Initial findings suggest:

- Loggers located at the east end of lake clearly indicate a consistent, cool temperature from the springs complex at the head of the lake.

- During mid-summer, loggers in the center of the lake were consistently warmer than those at the east end. During fall, the conditions were reversed, with slightly cooler temperatures in the center of the lake, suggesting that waters cooled as distance from the spring source increased.

- Surface loggers were consistently warmer than bottom loggers in the summer period, showing clear stratification. In winter water temperatures were similar top to bottom.

- Surface loggers typically show a larger diurnal range than bottom loggers. There is considerable aquatic vegetation growth in the lake, which may impede vertical mixing and probably limits light penetration. These conditions, coupled with replenishment of cool water from the spring complex at the east end of the lake, lead to a persistent stratification through the summer period – an anomalous condition for such a shallow lake.

- Cool waters in the summer appear to traverse the bottom of the reservoir as density driven flows. The result of passing through the lake is that there is a phase shift in the diurnal signal. Peak water temperatures in Big Springs Creek below the dam during summer periods typically occur between 1:00 to 3:00 a.m., versus the typical summer period peak between 4:00 and 6:00 p.m. Once in the creek, local meteorological conditions imposed on the creek largely remove this inverted signal by the time the water reaches the Shasta River or commingles with other spring inflows downstream.

- A second point associated with transit across Big Springs Lake is that release temperatures from the dam are several degrees warmer than the springs at the east end of the lake during summer. Measurements of springflow at the east end of the lake are on the order of 11^{o}C, while maximum release temperatures exceeded 15^{o}C during certain periods of the summer.

A more complete synopsis of the role of Big Springs Lake on downstream thermal conditions will be the subject of future analyses.

<u>Longitudinal Variability in Big Springs Creek</u>

The longitudinal temperature monitoring program was implemented to assess changes in temperature from upstream to downstream and included the deployment of remote logging thermistors (Onset HOBO® Pro v2 Water Temperature Data Logger) at multiple locations between Big Springs Dam and the Shasta River. Some of the data presented herein are augmented with data from ongoing monitoring efforts in Big Springs Creek.

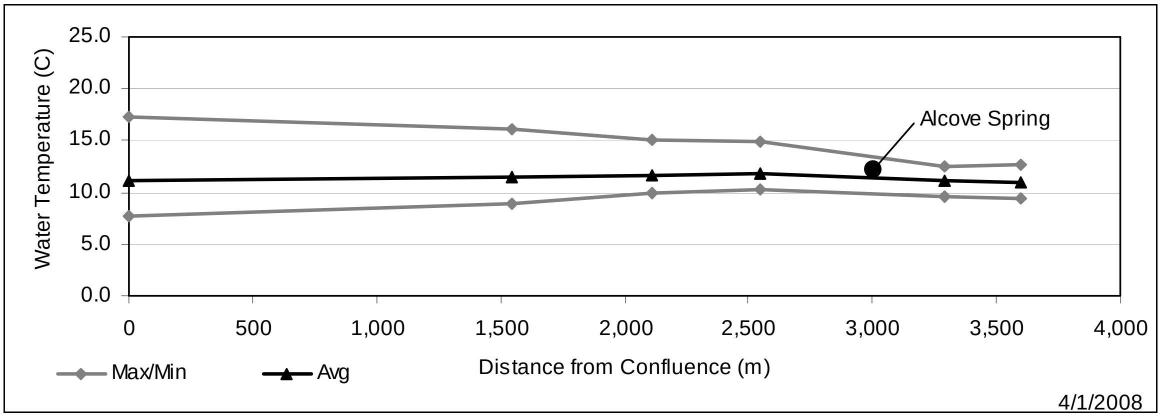
Review of spring time temperatures in Big Springs Creek between the dam and the confluence with the Shasta River suggest that under the conditions of 2008 there was considerable heating from source to mouth.

Spring: March-June

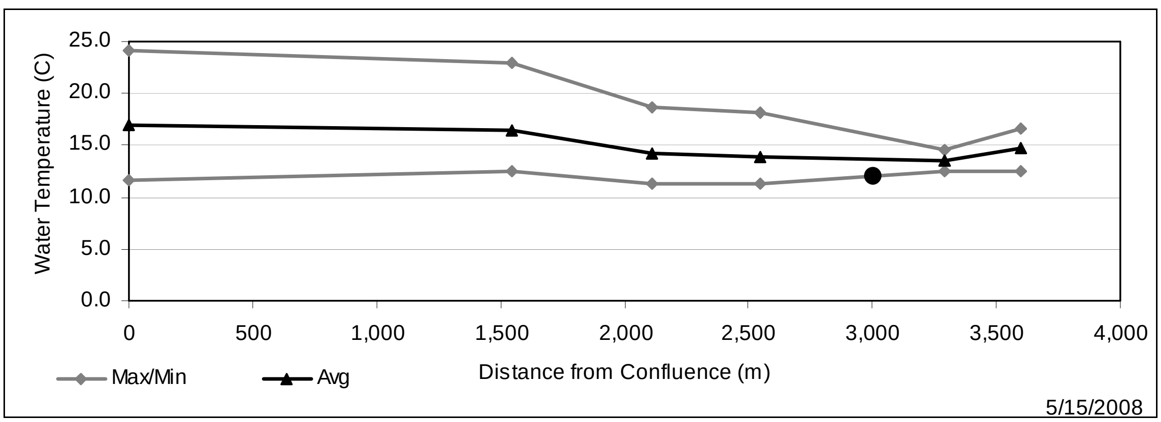
Seasonal temperatures vary considerably for the spring period. Longitudinal profiles of daily maximum, minimum, and average for 1 April indicate that average daily temperatures at the dam were on the order of 11^{o}C, and showed little heating en route to the mouth (Figure 11(a)). However review of daily maxima and minima indicate there is a fair amount of heating and cooling as diurnal range increases from approximately 3^{o}C at the dam to nearly 10^{o}C at the mouth. The suppressed diurnal range at the dam is representative of Big Springs Lake, a relatively deep body of water with a large thermal inertia compared to the shallow and wide creek channel which responds rapidly to atmospheric conditions – illustrating considerable heating above and cooling below the average. Note that the alcove spring (which for purpose of this discussion includes other nearby accretions associated with the aforementioned Alcove Spring complex – see Figure 6) is a source of heat where it enters the main stem Big Springs Creek on 1 April.

As the spring season progresses, releases from Big Springs Lake increase in temperature to approximately 15^{o}C and heating between the dam and mouth, as represented by daily average temperature, increases to about 3^{o}C. The alcove spring becomes a source of cool waters during this period (Figure 11(b) and (c)). The diurnal range is well over 10^{o}C at the mouth with maximum daily temperatures approaching 25^{o}C. Daily minima are stable at temperatures nearly equal to source waters at approximately 11^{o}C. This regime is possible based on several factors. First, transit time is sufficiently short in Big Springs Creek resulting in essentially all the water in the creek being replaced by cool spring flows over the night time period. Second, meteorological conditions in this portion of the Shasta Valley indicate that nighttime equilibrium temperature[2] is on the order of 11 to 12^{o}C. Finally, source waters are close to this equilibrium temperature. Due to the high specific heat and density of water, changes in temperature in response to meteorological conditions often lags loading by several hours. Thus, in streams that lack a strong groundwater signal (e.g., dominant spring inflow) nighttime temperatures may never attain equilibrium with nighttime meteorological conditions – daytime temperatures are simply too high and the nights too short to attain this condition (particularly around summer solstice). However, because the transit time is short, spring flow dominates, and spring flow temperatures are near nighttime equilibrium temperature in Big Springs Creek, the entire stream for the season (as well as through summer) drops down to 11 to 12^{o}C nearly every night. The implications for this from a salmonid fishery perspective can be considerable: although warm temperatures may occur during daytime periods, nighttime temperatures drop to optimal ranges values for salmonids.

[2] Equilibrium temperature is the water temperature that would result from exposure to a specific set of meteorological conditions, i.e., the water temperature is in equilibrium with meteorological conditions. In reality, equilibrium temperature is a moving target over the period of a day in response to varying meteorological conditions. Nonetheless, the theoretical construct of an equilibrium condition is a useful tool to interpret water temperature information.



(a)



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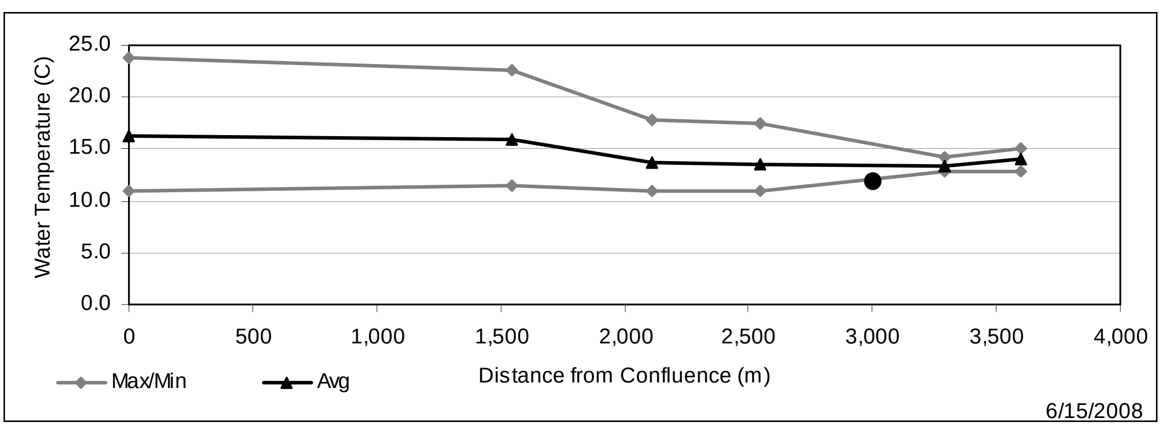


(c)

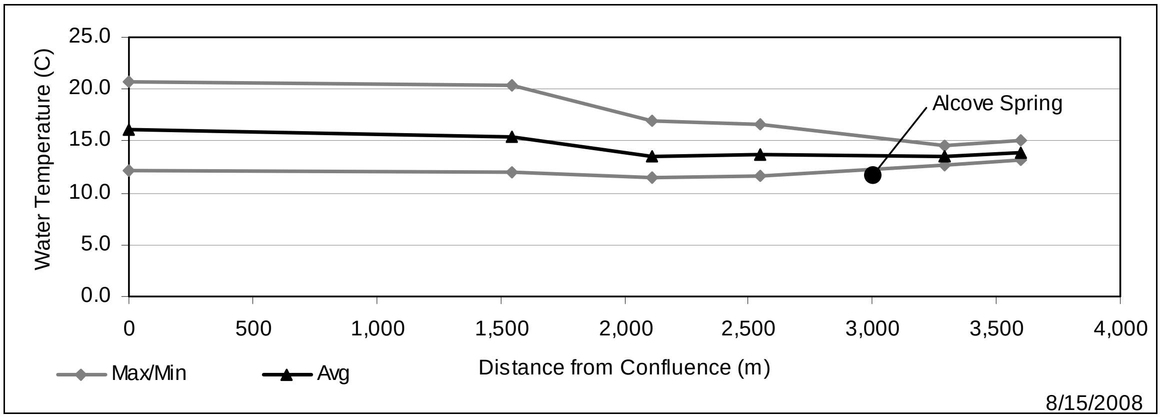
Figure 11. Longitudinal temperature patterns for Big Springs Creek in maximum, minimum, and average daily temperatures for (a) 4/1/08, (b) 5/15/08, and (c) 6/15/08. The water wheel is located 2550 m from the confluence.

Summer: July-September

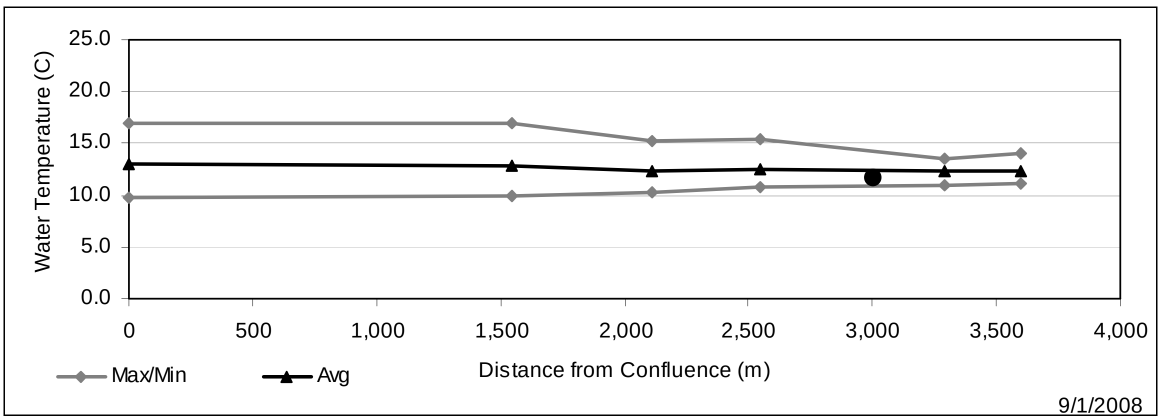
Summer conditions in Big Springs Creek are similar to late spring. Namely, release temperatures at the dam are between 13 and 15^{o}C, the alcove spring provides considerable flow at cooler temperatures than the creek, and heating in the wide, shallow channel increases in the downstream direction (Figure 12). There is reason to believe that thermal loading was moderated in 2008 due to reduced grazing in the creek channel, allowing aquatic vegetation to colonize near shore areas and effectively narrow and deepen the channel. This process of narrowing and deepening the channel leads to increased velocity and reduced travel time, a smaller air-water interface for thermal loading, and a larger thermal mass which moderated thermal changes – all factors that lead to reduced heating. Finally, the longitudinal profiles indicate that persistent thermal conditions in Big Springs Creek are amenable to coho salmon and other anadromous fishes.

Late Fall- Early Winter: December and January

Fall creates an inverse condition to spring. As days shorten the rate of heating longitudinally reduces to no net gain on a daily average basis, while daily maxima and minima may show a larger range at the mouth than the headwaters. However, but late fall and into winter, the Alcove Springs complex acts as a source of heat in an otherwise cool or cooling system. As illustrated in Figure 13, the December and January period temperatures exhibit little diurnal range in response to low solar altitude and short day length. Releases from Big Springs Dam are typically 10^{o}C or less on a daily average basis (suggesting cooling from the springs complex at the east end of Big Springs Lake to the dam as noted above). However, considerable accretion (springs) to the creek below the dam (i.e. the Alcove Springs complex) indicate an increase in creek temperature from 1 to 2^{o}C above dam release temperatures. Subsequently temperatures begin to cool downstream of these accretion sources en route to the Shasta River; however, the overall winter "warming" associated with the Big Springs complex extends well beyond the confluence with the Shasta and has direct implications of food web and fish production..



(a)



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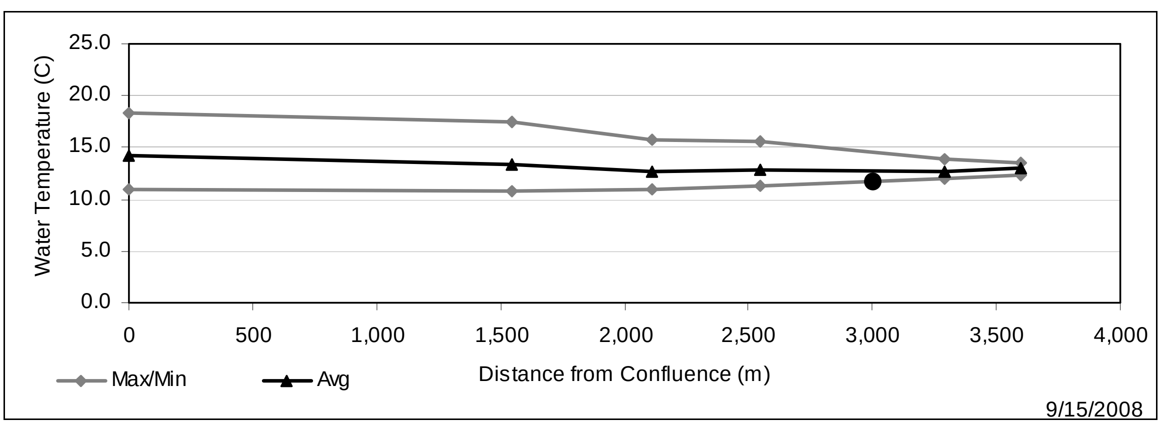
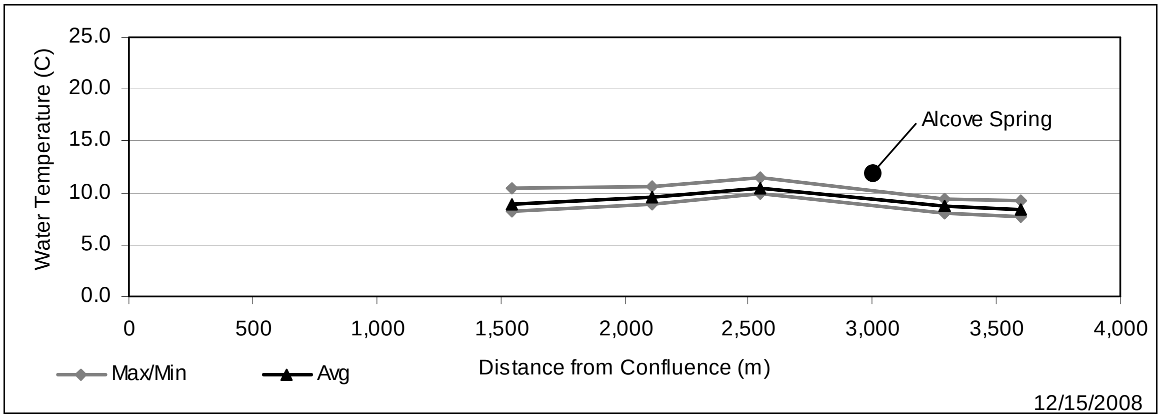
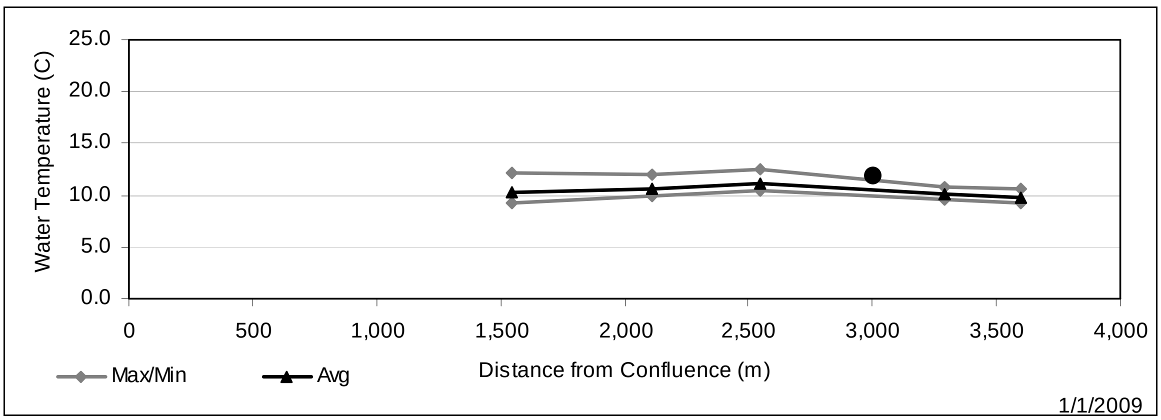


(c)

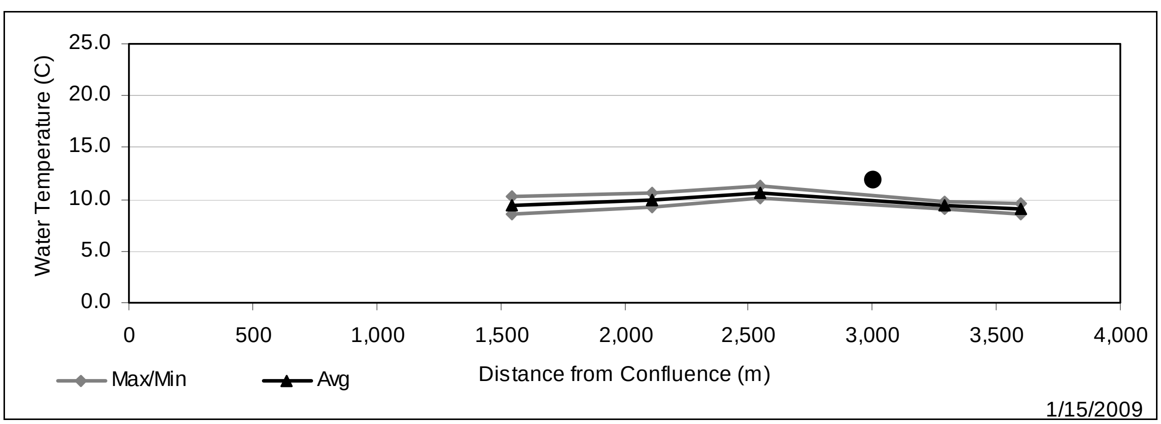
Figure 12. Longitudinal temperature patterns for Big Springs Creek in maximum, minimum, and average daily temperatures for (a) 8/15/08, (b) 9/1/08, and (c) 9/15/08. The water wheel is located 2550 m from the confluence.



(a)



(b)



(c)

Figure 13. Longitudinal temperature patterns for Big Springs Creek in maximum, minimum, and average daily temperatures for (a) 12/15/08, (b) 1/1/09, and (c) 1/15/09. The water wheel is located 2550 m from the confluence.

Thermal Diversity of Big Springs Creek

Because Big Springs Creek baseflow is derived from a large spring complex (principally the combined inputs of the Big Springs Lake and Alcove Springs complexes) that is spatially distributed over approximately 0.8 km or more of stream length, there is a wide range of temperature conditions present in addition to the basic longitudinal variability described above. The head of the springs complex is actually at the east end of Big Springs Lake; however, below the impoundment, there is nearly a continuous accretion of spring flow for approximately 0.5 km, with springs ranging from seeps to locations with considerable spring flow, and even artesian features.

To identify and quantify cool water sources, as well as rates of heating in the system, lateral variability and thermal diversity were explored through direct temperature observations. Additionally, observations in Big Springs Lake illustrate some unique features of this spring complex and thermal conditions entering the creek at the dam. System variability was assessed by deploying thermistors at multiple cross sections along the creek. Additionally, handheld measurements were completed with the instrumentation presented in the methods section, above.

Thermal diversity is a feature present in most streams, particularly during periods of increased thermal loading (e.g., spring and summer). Features that produce thermal gradients include shading, depth, velocity, aquatic vegetation and residence time. Shading affects differential heating rates. Different water depths lead to different heating rates due to the specific heat and density of water. Different velocity fields can transport heat energy from one location to another. Aquatic vegetation affect local residence time of water. Other factors, including exchange with ground water, also affect thermal loading, though they are currently beyond the scope of this project. Such lateral variability was clearly evident in Big Springs Creek. What makes this system unique is the imposition of multiple spring inflows (e.g., cold sources of water in summer) on the system.

Transects

Transects and an extensive exploration of the creek were completed with handheld instrumentation. This work provided the opportunity to explore large areas over relatively short periods of time and assess attributes that may or may not play important roles in the system. An example of a cross section assessment of temperature and depth with associated field observations is shown in Figure 14. These types of field forays proved invaluable in setting final deployments for long-term transect assessment using temperature loggers.

In August 2008 ten transects were placed in Big Springs Creek from the dam to downstream approximately two miles (Figure 15). Depending on the stream width, two to six loggers were deployed in a cross section. Loggers were set to record temperature every 30 minutes. These data provided clear insight into the complexity of the system and assisted in corroboration and interpretation of the TIR data. Several transects will be presented herein to illustrate the lateral variability apparent in the field data.

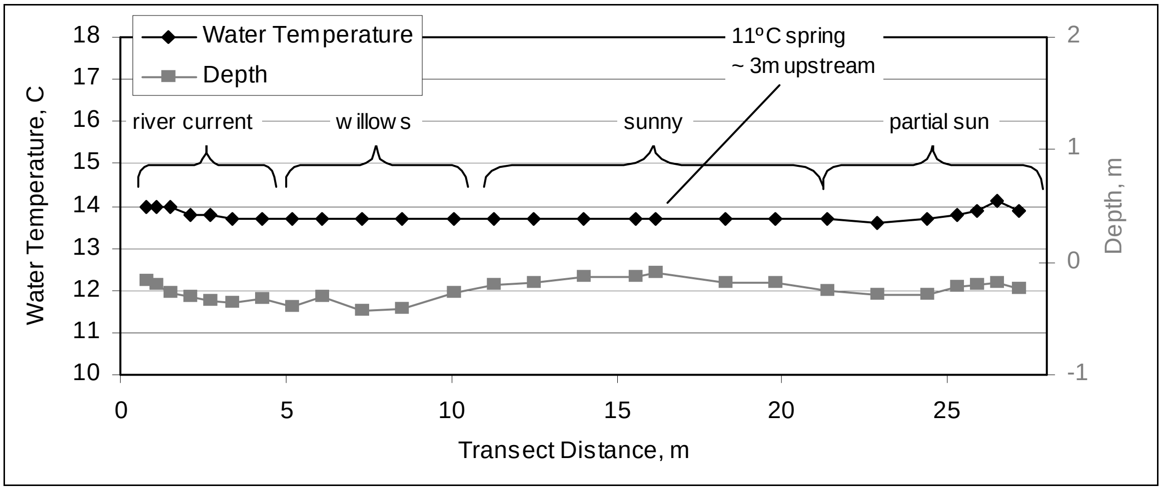


Figure 14. Busk Ranch transect 1 water temperature and depth (6/25/08 15:00). Stream depth assumes water surface stage datum is 0.0, thus depths are negative.

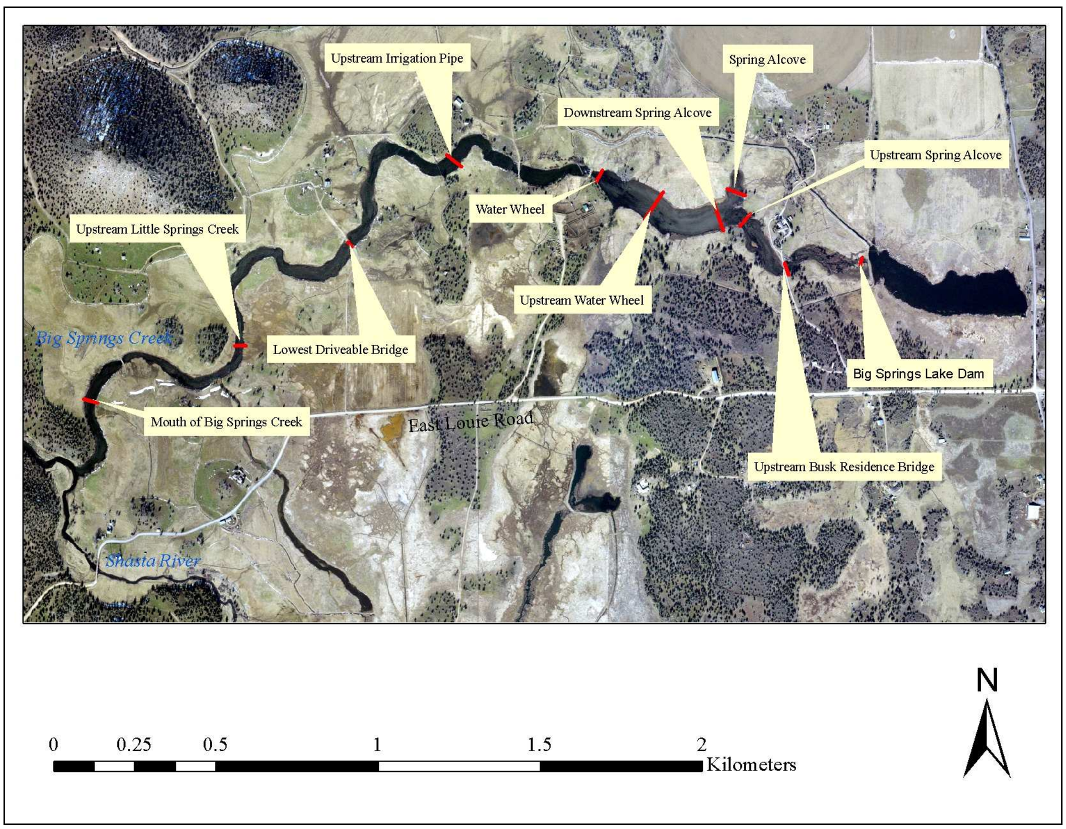


Figure 15. Location of temperature logger transects.

Starting at the upstream most cross section, Big Springs Lake Dam, lateral diversity was minimal. This site is less than 10 meters in width and largely dominated by releases from the dam. Also, there is appreciable riparian shading and spring flow accretions in the area that are largely diffuse and difficult to quantify. As a result, there is little lateral variability (Figure 16).

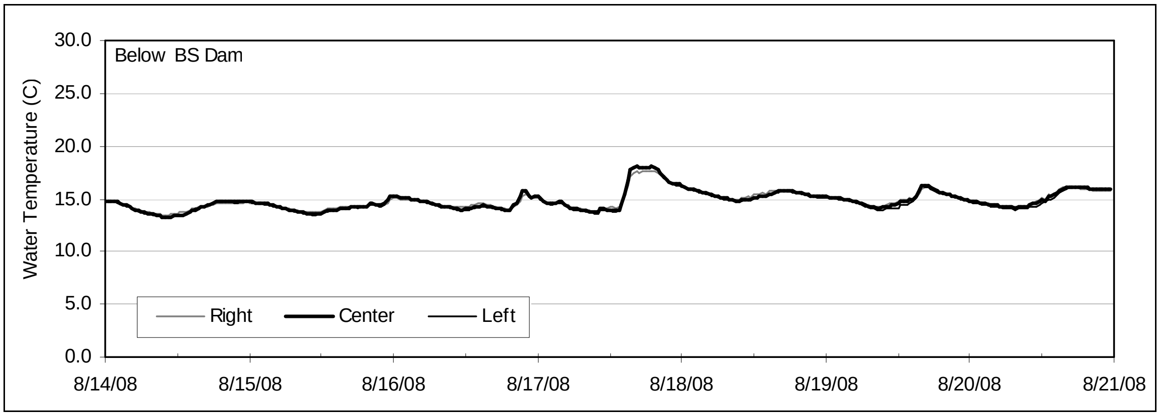


Figure 16. Lateral temperature time series for Big Springs Creek below Big Springs Lake dam: 30 minute data 8/14 to 8/20/08.

Below the Busk residence bridge, discrete spring sources are readily apparent. The cross section above the Alcove Springs complex illustrated conditions unique to spring fed streams: namely that at this location the left and right bank areas are notably cooler and generally exhibit a smaller diurnal range than the mid-channel (center left and center right) locations (Figure 17). This condition is the result of streamside springs on both the left and right banks entering Big Springs Creek. Other factors that play a role in these conditions are local flow paths through extensive aquatic vegetation.

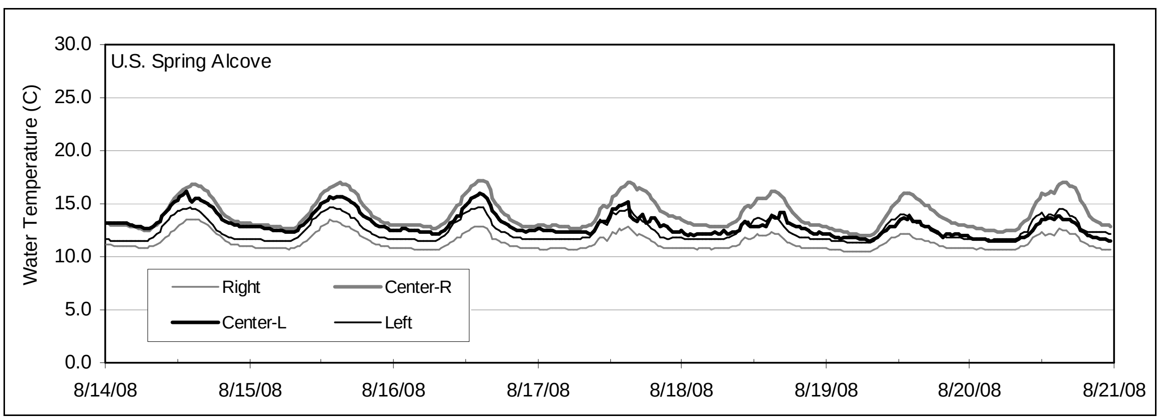


Figure 17. Lateral temperature time series for Big Springs Creek above the alcove springs: 30 minute data 8/14 to 8/20/08.

Below the alcove springs in the wide section above the water wheel, the influence of multiple spring sources, to commingling of main stem creek water from upstream, and variable transit times manifest themselves in a complex thermal picture (Figure 18). At this location, left to right differences vary well over 15^{o}C. Additionally, the time of maximum daily temperature and the different rates of heating and cooling among the traces indicate variable transit times for parcels of water moving through the system. It appears that the center and center-right locations have notable longer flow paths and or shallower waters, and possibly may be segregated to some degree from the influence of source waters from springs. Examining the center-right time series suggest that these waters do not get replaced at night by upstream cool spring waters and do not attain the minimum daily temperatures of center and center-left locations.

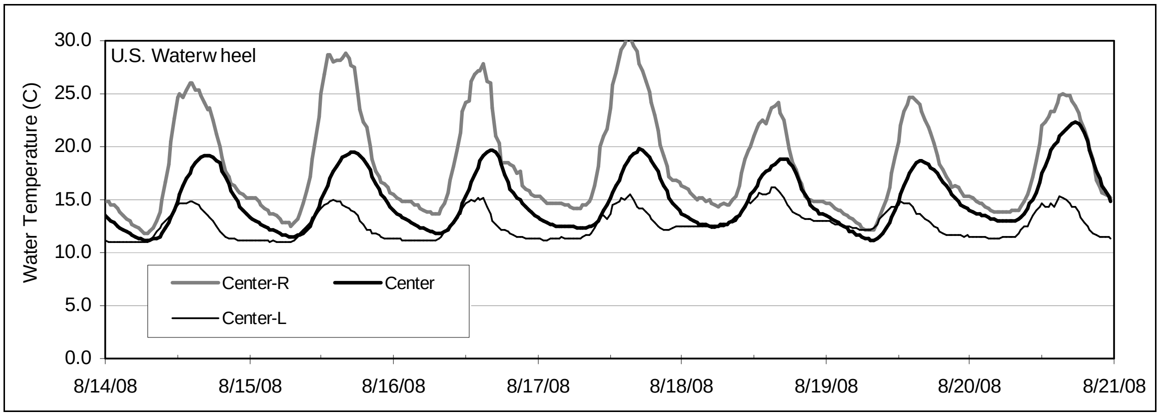


Figure 18. Lateral temperature time series for Big Springs Creek below the alcove springs and above the waterwheel: 30 minute data 8/14 to 8/20/08.

Downstream at the waterwheel, there is a considerable constriction and the entire creek passes through two openings at a bridge (waterwheel site). The total width here is approximately 10 meters. Thus all waters from upstream must commingle as they pass this point. Although there are appreciable velocities at this constriction and there is potential for mixing, the left and right channels clearly segregate by temperature (Figure 19).

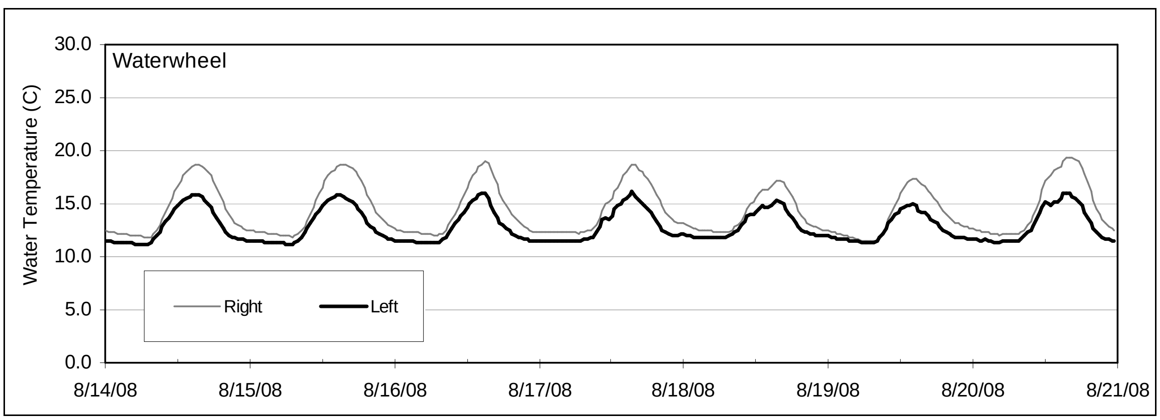


Figure 19. Lateral temperature time series for Big Springs Creek at the waterwheel: 30 minute data 8/14 to 8/20/08.

Thermal Infrared (TIR) Imagery

Airborne thermal imagery (TIR) remote sensing is an effective method for mapping spatial temperature patterns in rivers and streams. TIR imagery illustrates the location and thermal influence of point sources, tributaries, and surface springs (Watershed Sciences 2009). In 2008 Watershed Sciences, Inc. was contracted to provide TIR imagery for approximately thirty river miles in the Upper Shasta River basin. The TIR acquisition included an early dawn flight and late afternoon flight for the Upper Shasta River, Big Springs Creek, Little Springs Creek, Parks Creek (East and West) and Spring Creek. These data were collected under a U.S. Bureau of Reclamation grant and are only briefly introduced herein. A more comprehensive interpretation of these data will be forthcoming in a separate report.

Images were collected with TIR (7.5-13.0) attached to a gyro-stabilized mount on the underside of a helicopter. The aircraft was flown longitudinally along the stream corridor in order to have the river in the center of the display. The objective was for the stream to occupy 30-60% of the image. The TIR sensor is set to acquire images at a rate of 1 image every second resulting in 40-70% vertical overlap between images. A flight altitude of 2,000 ft (609 m) above ground level resulted in a pixel ground sample distance of 1.6 ft (0.5 m). The flight altitude was selected in order to optimize resolution while providing an image ground foot print wide enough to capture the active channel and immediate floodplain (Watershed Sciences 2009).

Unique to this application of TIR was collecting data at approximately maximum and minimum daily temperatures. The afternoon (maximum) flight was intended to contrast cool water sources in an otherwise warm system. The dawn flight was intended to capture transient thermal conditions well down the Shasta River. This latter point is beyond the scope of this study; however, the dawn flight did substantiate that spring flow temperatures dominate the daily minimum thermal conditions. TIR images for the reach between the alcove springs and the waterwheel is shown in Figure 20 for both the afternoon (top) and dawn (bottom). These data indicate that there is considerable thermal diversity in the reaches where springs are present adjacent to and within the channel. Further, preferential flow paths are apparent through this reach and waters are well segregated left to right in the channel depending on their source locations and temperatures. Further, day time minimum temperatures are largely uniform throughout the creek reach. The conditions are consistent with thermal refugia temperature response observed in the Klamath River (Deas and Tanaka, 2005) wherein refugial areas varied spatially throughout the day. These findings suggest that portions of Big Springs Creek under a restored condition could perform with thermal refugia-like characteristics. Specifically, Big Springs Creek could provide extensive thermally-favorable conditions in the early morning hours and allowing fish to forage widely, and then congregate in refugial areas during the warmer portions of the day when conditions are undesirable. This feature may extend the salmonid carrying capacity of the stream, but has yet to be quantified.

TIR imagery of the confluence of Big Springs Creek and the Shasta River is shown in Figure 21. Here the cooler waters of Big Springs Creek dominate temperatures along the right side of the Shasta River with little lateral mixing for some distance downstream. This is another example of thermal diversity both laterally and longitudinally – strong gradients exist laterally in the channel below the confluence and these gradients diminish with distance downstream as the two streams slowly commingle, with the Shasta River ultimately experiencing overall cooling due to the relatively large spring flow contributions to the smaller Shasta River flows.

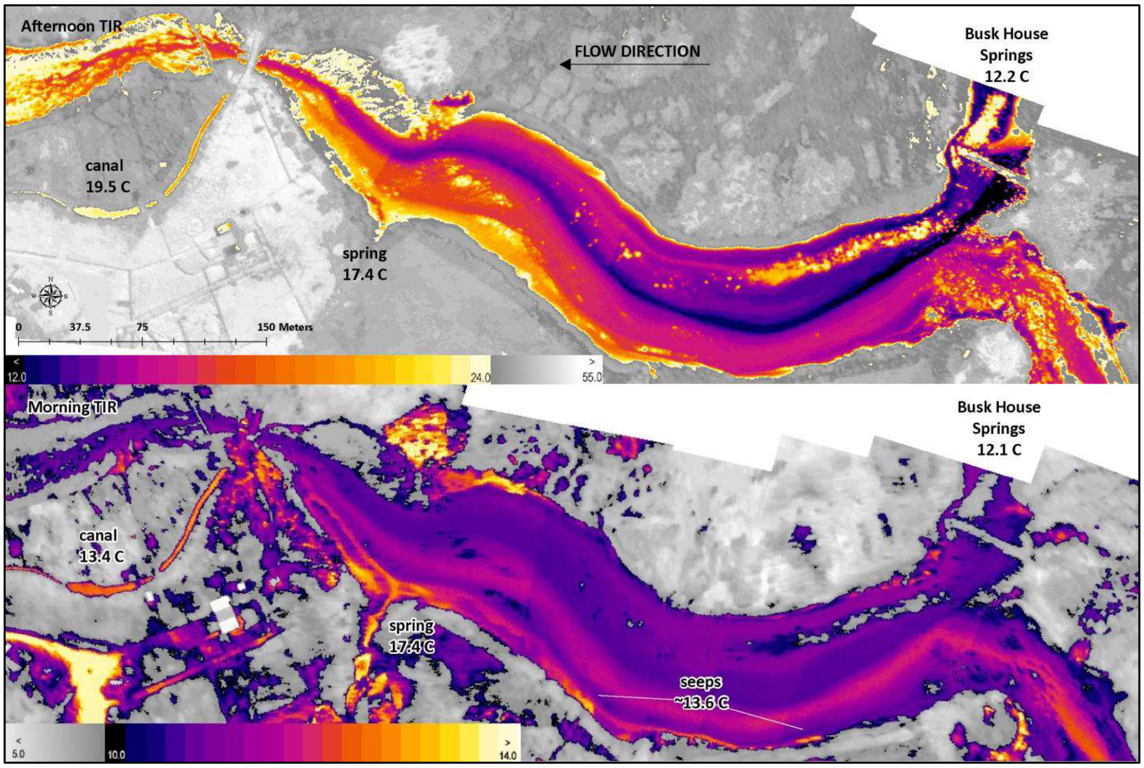


Figure 20. TIR imagery for July 16 afternoon (top), and July 17 at Dawn (bottom).

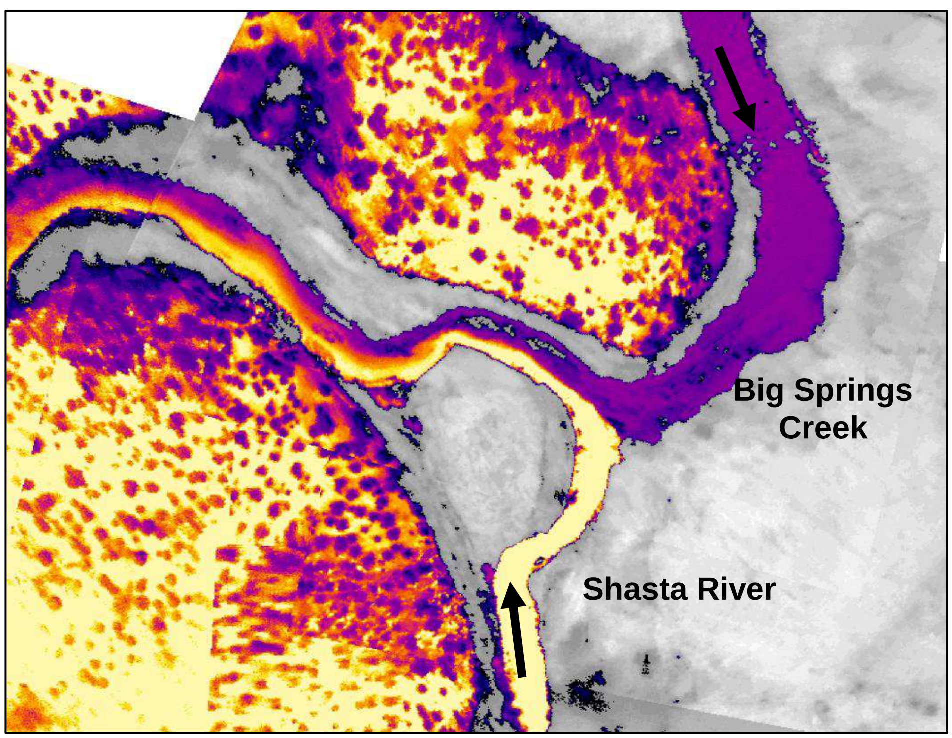


Figure 21. TIR imagery for July 17 at dawn of the confluence of Big Springs Creek and the Shasta River

The 2008 FLIR and hydrology data indicate that no significant accretions occur between the confluence of Big Springs Creek with the Shasta River and the southern area of Nelson Ranch (approximately 2.7 kilometers downstream). Rather, the cold water signal observed during both the 2003 and 2008 FLIR surveys is caused by advection of Big Springs Creek water downstream. The hydrology data indicates that flow accretions in this reach are minimal (see section 4.1.3). This data is supported with field observations of only small springs (<<1 ft^3/s) in this reach. The thermal signal in the Shasta River near the boundary between Big Springs Ranch and Nelson Ranch was dominated by water that leaves Big Springs Creek. The 2003 FLIR data was taken during an afternoon flight on 26 July 2003. A more recent FLIR survey was flown during the afternoon of 16 July 2008. The longitudinal thermal profiles of the Shasta River illustrate almost the identical temperature signal in the approximate 12.9 km reach of the Shasta River below Big Springs Creek. Each of these flights shows a temperature decrease downstream of the confluence with Big Springs Creek (Figure 22 and Figure 23). Though both the afternoon surveys flown in 2003 and 2008 show a temperature decrease in the Shasta River below Big Springs Creek, the morning FLIR survey flown on 17 July 2008 shows that this signal is due to cold water inputs from Big Springs Creek. When the signals recorded during the morning and afternoon flights are compared, the longitudinal profile differs substantially (Figure 23).

During the afternoon flight, the depression in water temperatures near the Grenada Irrigation District/Huseman Ditch diversion structure is the result of advection of cold water produced by Big Springs Creek that morning. Note how during the morning survey, water temperatures immediately below Big Springs Creek are approximately 13^{o}C. Several hours later, during the afternoon survey, water temperatures downstream of the confluence with Big Springs Creek are slightly warmer (approximately 17^{o}C) – this is due to thermal loading of Big Springs Creek’s cool morning water as it travels downstream. During the day, water also heats in Big Springs Creek as it travels from its source to its mouth. This warmer water is what creates the warmer thermal signal downstream in the Shasta River during the morning flight.

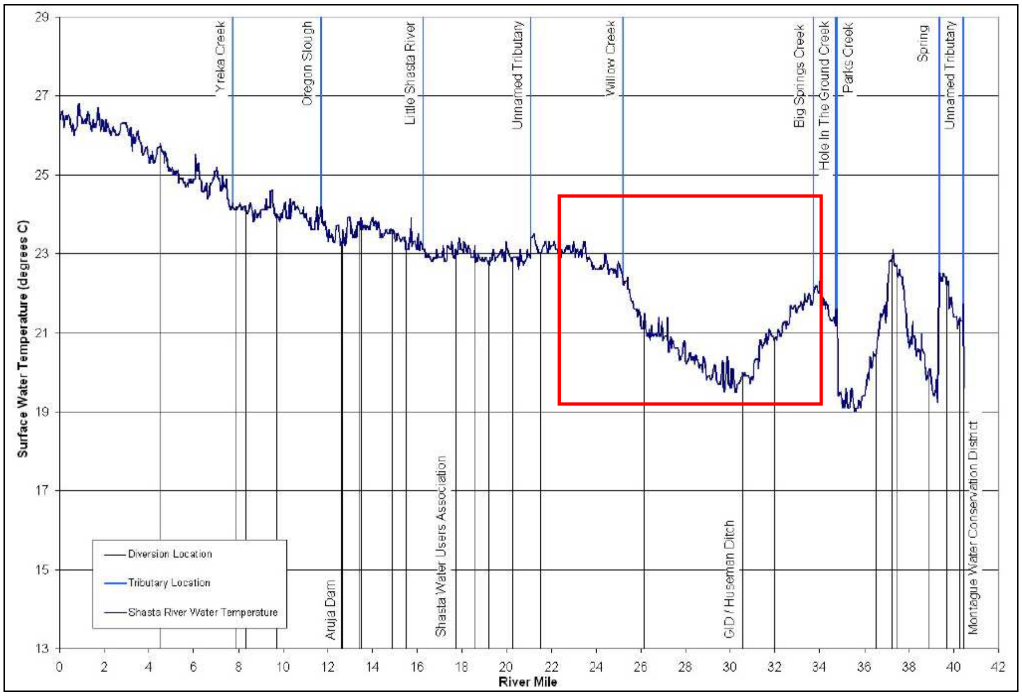


Figure 22. Shasta River longitudinal surface water temperature profile, and locations of tributaries and diversions, July 26, 2003. The red box highlights the reach of interest. (Source: NCRWQCB (2006)).

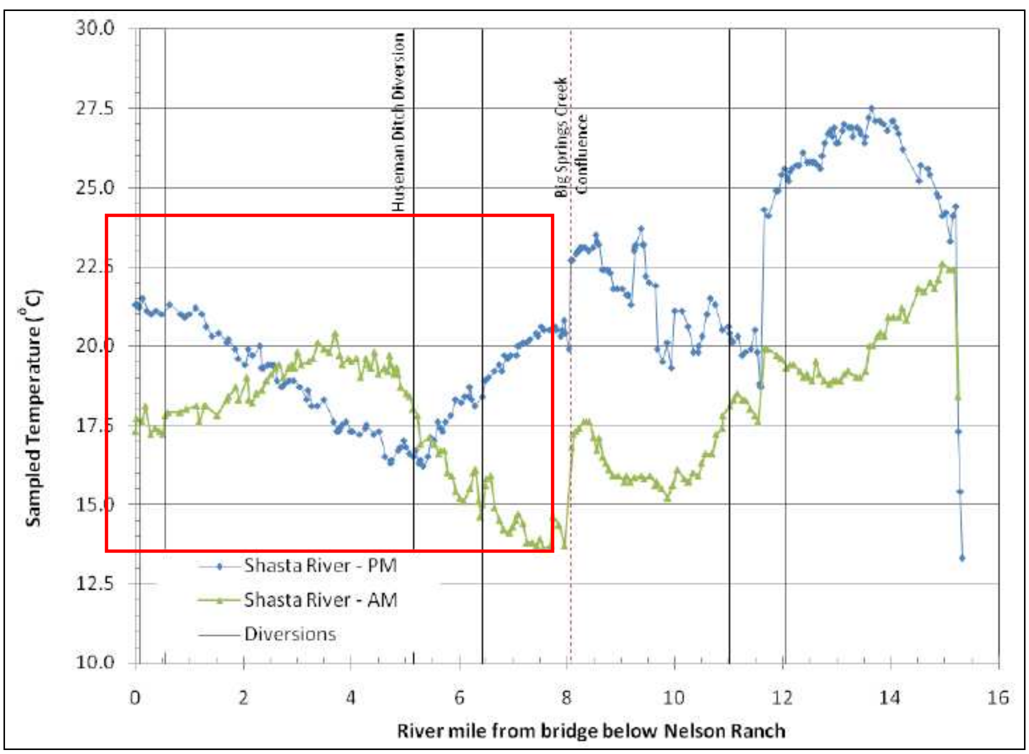


Figure 23. Comparison of the Upper Shasta River morning (AM) and afternoon flights (PM) on July 16-17, 2008. The red box highlights the reach of interest. (Source: (Watershed Sciences 2009)).

4.2.3 Conclusions

Overall, the cold water resources of the Big Springs complex are impressive. The fact that the complex is spatially diverse adds a level of complexity that may prove useful in restoration actions. This springs complex consists of a longitudinal feature over 0.8 km in length and that spring flow contributions enter the creek on both river left and right, within the channel itself, and through unique features (e.g., the spring alcove). In many cases resource managers are limited to small discrete cool patches or other thermal refugia type settings for salmonid over summer habitat, and these features often lack appreciable/sufficient size and persistent cold water temperatures necessary to provide robust, sustainable conditions. Big Springs Creek has an extensive spatial extent and robust cold water supplies, creating high potential for restoration of oversummering cool water habitat for anadromous and resident fishes. Individual conclusions of the various studies are summarized below.

Big Springs Lake

- Big Springs Lake is an artificial impoundment intended to provide water to irrigated agriculture.

- There is a large springs complex (Big Springs Lake Complex) located along the eastern edge of the lake and water temperatures are approximately 11^{o}C. During winter this water may cool en route to the dam and during summer it heats en route to the dam.

- Persistent source of cool water retains thermal stratification in this lake – a condition that would not persist without such cold water replenishment.

Longitudinal Characteristics

- During the spring season, the creek can experience considerable thermal loading, particularly under historic land and water use practices where the creek was wide and shallow.

- Summer periods are similar to late spring, with considerable heating between the Big Springs Lake Dam and the Shasta River. However, due to a short transit time, spring waters completely replace the waters within the creek during nighttime periods. Coupled with local nighttime meteorological conditions the result is daily lows throughout the summer on the order of 11 to 12^{o}C. These low nighttime values are a potentially valuable attribute for anadromous and resident fish.

- Advection of cool morning waters from Big Springs Creek result in a noticeable temperature decrease in the Shasta River during the afternoon. Similarly, as the water temperature in Big Springs Creek increase during the day due to thermal loading, downstream water temperatures in the Shasta River also increase as the warmer water flows from the creek to the river.

- During winter the springs complex serves as a heat source for the creek and downstream Shasta River. The influence of these relatively warm waters on the food web and fish production are an important element of Big Springs Creek.

Thermal Diversity

The work undertaken here was a preliminary assessment of the thermal diversity of Big Springs Creek.

- Both the handheld work and the TIR data were used to identify spring sources in the creek and complex system.

- Generally, waters appeared well mixed vertically in the water column, with few temperature differences identified between the surface and the bottom in this shallow system.

- Transect data indicates water temperature may vary considerably left to right depending on depth, current, and local spring influences. Data suggests that aquatic vegetation can act to buffer water temperatures or slow/reduce heating from solar radiation.

4.3 *Water Quality*

The unique water quality of the Big Springs complex, and presumably other spring complexes associated with the Shasta River south of the Big Springs Creek-Shasta River confluence, was likely one of the largest contributing factors to high historical abundances and productivity of salmonids in the Shasta River. The combination of ancient marine sediments overlain by volcanic rock in the Shasta Valley allows for inputs of natural sources of nitrogen (N) and phosphorus (P) to be incorporated into the groundwater that eventually daylights as springs in Big Springs Creek. Nitrogen and phosphorous are key components of primary productivity and one or the other are often limiting in natural aquatic ecosystems (when both limit primary productivity, the condition is termed colimitation). When N and P are available in sufficient quantities, primary production in aquatic systems can be appreciable. The result is enhanced growth rates at higher trophic levels in the food web from primary producers up through salmonids. In addition to nutrient availability, the large groundwater inputs strongly buffer water temperatures. Specifically, temperatures are warmer in the winter and cooler in the summer than they would otherwise be, attenuating stress from extreme temperatures. This moderation of stream temperatures maintain conditions in a more biologically advantageous range, further enhancing productivity throughout the food web. Herein we highlight nitrate (NO_3) and soluble-reactive PO_4 (SRP), the most biologically important inorganic forms of nitrogen and phosphorous, respectively, present in Big Springs Creek. A full suite of water quality constituents from the sampling program are included in the appendix.

4.3.1 *Methods*

Water samples were collected at 19 locations throughout the Shasta Valley on a biweekly or monthly basis[3]. Samples were collected in acid-washed 125 ml high-density polyethylene bottles. Bottles were thoroughly rinsed with the environmental water three times prior to collection of each sample. Samples were placed on ice and transported back to the University of California, Davis where samples were refrigerated throughout completion of processing. Samples were analyzed for pH, electrical conductivity (EC), total N, NO_3+NO_2-N, NH_4-N, total P, soluble-reactive phosphate (SRP), dissolved organic carbon (DOC), turbidity, and major cations (Ca^{2+}, Mg^{2+}, K^+, Na^+) and anions (Cl^-, SO_4^{2-}).

4.3.2 *Nitrate (NO_3) and Orthophosphate (SRP)*

Although a wide suite of analyses were collected at multiple sites, one aspect of the water chemistry that is clearly unique and has a direct influence on the aquatic system is that of nutrients. In many aquatic systems nitrogen and phosphorous play a dominant role in primary production; this in turn plays a key role in aquatic food webs. Nitrogen is an essential nutrient for plant growth, yet is often described as a pollutant (e.g., anthropogenic sources such as fertilizers, animal wastes, municipal and industrial discharges, etc.), along with orthophosphate, in many freshwater systems and subject to total daily maximum loads (TMDLs) due to its role in eutrophication. In rivers with elevated nutrient levels (N & P), abundant primary productivity can result in eutrophication. Eutrophication is defined as high levels of nutrients and high primary productivity. In certain cases these conditions can lead to other undesirable water quality conditions including elevated pH (in weakly buffered systems), associated unionized ammonia toxicity (if ammonia is present in sufficient concentrations), and subsaturation dissolved oxygen conditions (if, for example natural reaeration rates are insufficient to counter demand). Results of this study indicate that elevated levels of nitrogen and phosphorus are most likely not related to anthropogenic sources, but rather elevated nitrate levels in Big Springs Creek are naturally derived from geologic sources along the groundwater flow path.

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[3] A portion of this work was funded under a separate contract through the U.S. Bureau of Reclamation, Klamath Basin Area Office (Reclamation). The intent of the Reclamation study was to assess conditions over a broader spatial and temporal scale, and to specifically support and compliment other studies in the Shasta Valley. As such, some of the data presented herein are folded into the Big Springs Creek analysis to illustrate both characterize conditions within Big Springs, as well as the influence that Big Springs has on the greater Shasta River system.

Carbon, nitrogen, and phosphorus in algal tissues typically occur in a 106:16:1 molar ratio, known as the Redfield ratio (Redfield et al. 1963). Generally, a ratio less than 16:1 is associated with a nitrogen limitation (Allan 1999). The ratio of N:P at the source springs on Big Springs Creek (i.e. Big Springs Lake and Alcove Spring complexes) is 2.5:1, suggesting that nitrogen will potentially be the most limiting nutrient to aquatic primary productivity (Figure 24). This hypothesis was tested by tracking the availability of both NO_3-N and SRP downstream from spring sources in Big Springs Creek. Ammonium concentrations were very low at all sites throughout the study period and therefore NO_3-N is the primary source of biologically-available nitrogen in the water column. Flows in the Shasta River above the confluence with Big Springs Creek were typically less than 25 percent of Big Springs Creek inflows. Thus, the primary source of nutrients in Big Springs Creek and the Shasta River below the confluence were derived from the Big Springs complex.

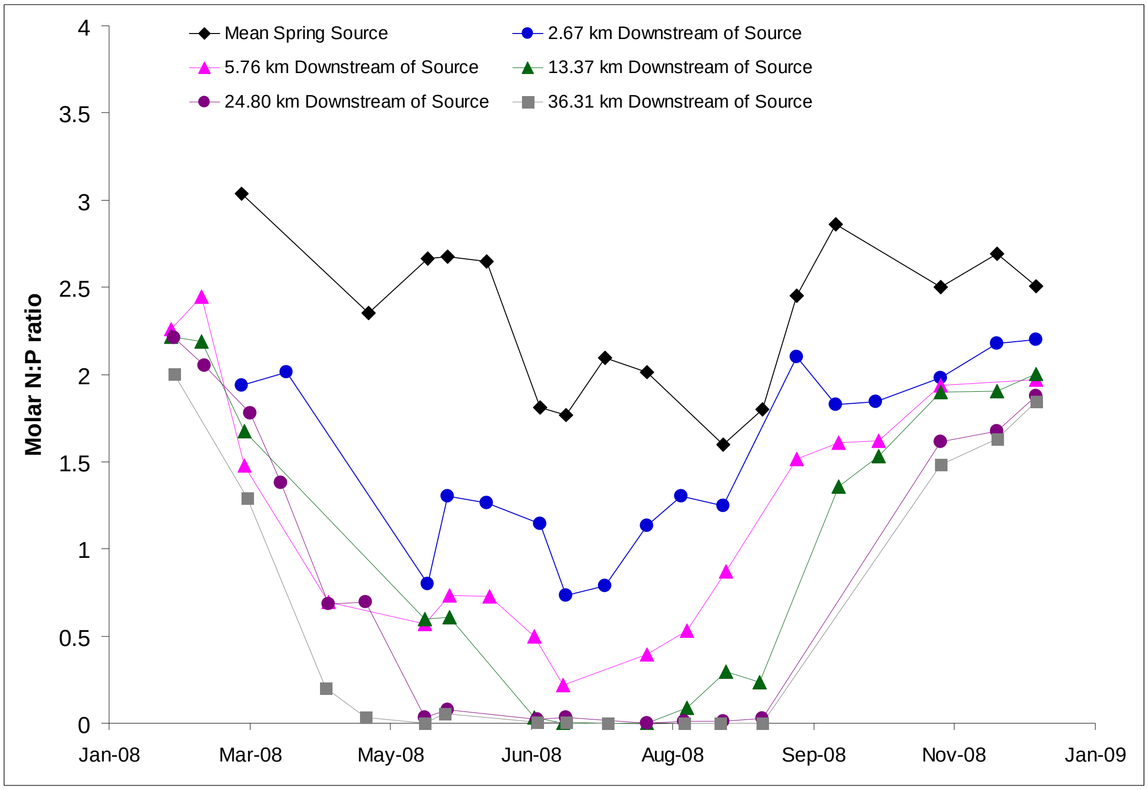


Figure 24. Nitrogen to Phosphorous molar ratio.

Water samples were collected longitudinally over a 42.8 km distance from the headwaters of Big Springs Creek downstream into the Shasta River (under the U.S. Bureau of Reclamation project). When sampling began in February, little aquatic vegetation was present and nitrate levels were relatively similar throughout Big Springs Creek and the Shasta River reaches at concentrations on the order of 0.25 to 0.30 mg/l. As day length increased into the spring time period, aquatic macrophytes proliferated in an environment where nitrate and orthophosphate were available in surplus quantities. However, as biomass of this aquatic vegetation increased throughout Big Springs Creek and in downstream Shasta River reaches during the spring and summer months, a longitudinal attenuation in nitrate concentrations was observed as distance increased from the spring source (Figure 25). These findings suggest that the seasonal decrease (June, July, and August) in nitrate is inversely proportional to the abundance of aquatic macrophytes in the channel as determined from qualitative macrophyte biomass observations throughout the year. Aquatic plants closest to the spring source were able to largely meet their nitrate demands. Nearly continuous, extensive macrophyte growth both in Big Springs Creek and the Shasta River downstream of the confluence with the creek appear to have systematically removed nitrate through the primary growth season (Figure 25). The result is that nitrate was depleted to a level where nitrogen limitation was probably prevalent in the Shasta River only a few miles below the Big Springs Creek confluence.

While phosphorus in freshwater systems is often the limiting nutrient to plant growth, this was not the case in Big Springs Creek and the Shasta River below the creek confluence. Throughout the summer sampling period SRP fluctuated only modestly – from approximately 0.12 mg/l to 0.18 mg/l (Figure 26). Further, seasonal longitudinal attenuation of SRP was not observed as it was for nitrate. Chemical equilibrium modeling of Big Spring waters indicated that the groundwater SRP concentrations are controlled by rock-water reactions with the mineral apatite. Thus, SRP concentrations generally fall within a narrow range near 0.15 mg-P/l. It is interesting to note that a level of 0.01 mg-P/l is suggested by the EPA as a maximum level to limit eutrophication in freshwater aquatic ecosystems (Kelly 2001). Given the concentrations of SRP in the system and the fact that phosphorus is required by macrophytes in much lower concentrations (16N:1P) than nitrogen, the lack of seasonal longitudinal attenuation in SRP concentrations due to macrophyte uptake is not unexpected. Further findings supporting this condition are the N:P ratios throughout the year (Figure 24), suggesting N limitation is seasonally prevalent during the growth season at downstream locations.

As day length shortens in the late summer and into fall, plants begin to senesce and require considerably less nitrate (and SRP) for daily maintenance and growth. The result is more nutrients are allowed to pass to downstream reaches. This considerable reduction in demand is clearly illustrated in Figure 25, as nitrate levels between mid-August and the end of September increase rapidly. By late September nitrate concentrations return to pre-irrigation levels. Because SRP is not limiting, concentrations show modest changes through this same period. The small increase in SRP concentrations that were observed beginning in August and into September may reflect decreased uptake by macrophytes and/or release of phosphorus from the senescence of aquatic vegetation. Some of the increase in nitrate may also result from late summer and fall senescence of macrophytes.

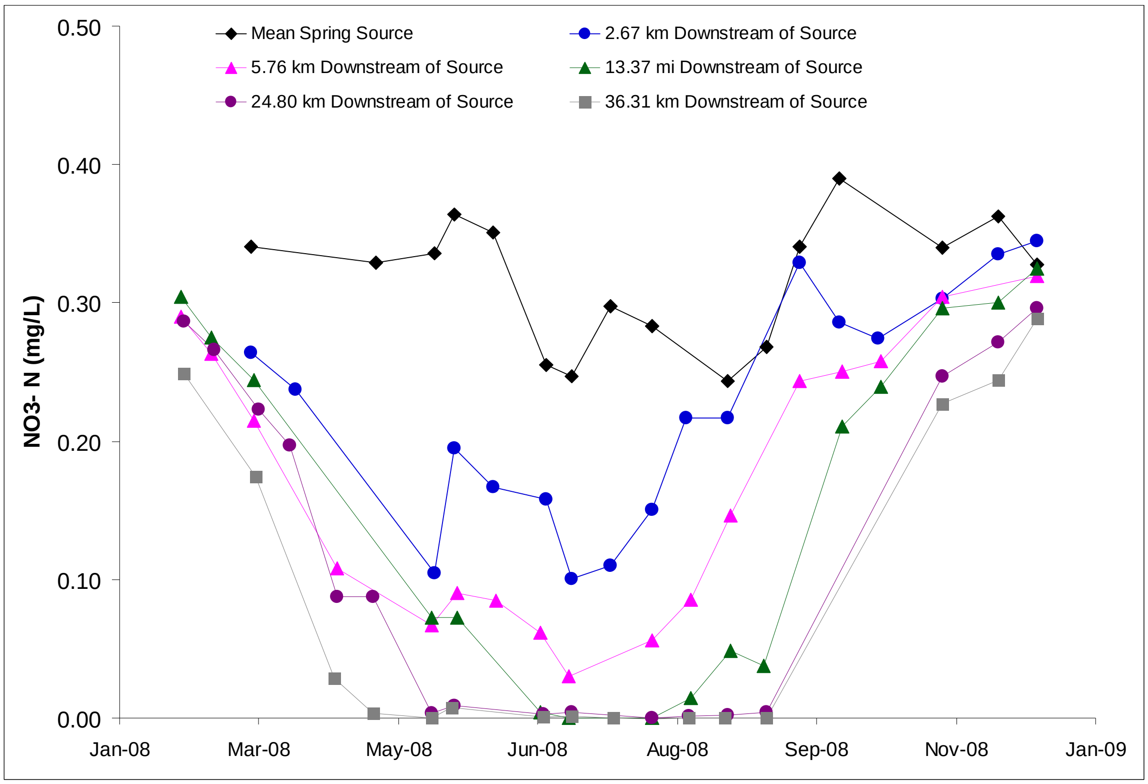


Figure 25. Nitrate concentration longitudinally from Big Springs Creek source downstream 26.6 miles into the Shasta River. Note attenuation of nitrate during summer months due to uptake by aquatic macrophytes.

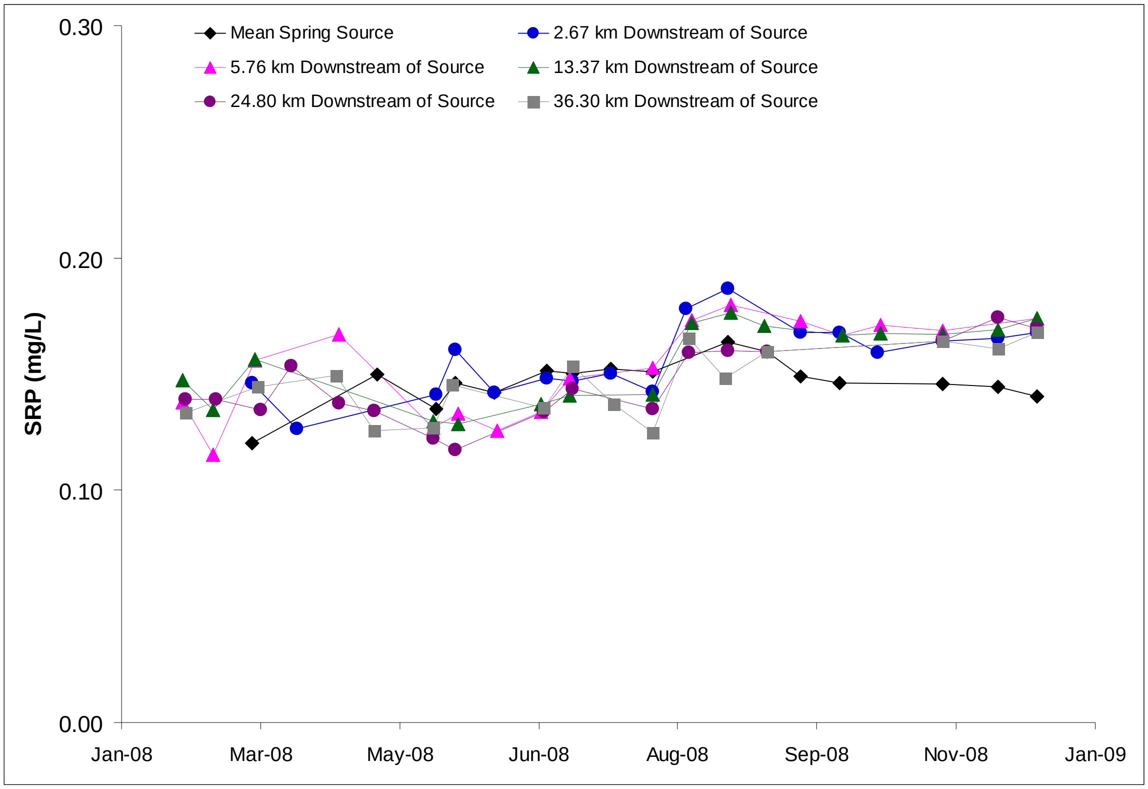


Figure 26. Soluble-reactive phosphorus (SRP) concentration longitudinally from Big Springs Creek source downstream 26.6 miles into the Shasta River. Note very little seasonal variation in concentrations, showing that SRP is likely not the limiting nutrient for primary productivity in Big Springs Creek and in the Shasta River downstream of Big Springs Creek.

4.4 *Geomorphology*

Geomorphic studies provide key insights into understanding the physical conditions and processes that help establish the template upon which aquatic ecological communities develop and function. Furthermore, geomorphic data also document existing channel conditions, providing a critical foundation for riverine restoration projects. Such data can also be used to provide boundary conditions for physically-based models used to assess potential outcomes of restoration strategies.

On Big Springs Creek, geomorphic surveys were conducted between July and October 2008 to achieve the following research objectives:

- Understand longitudinal variations in existing geomorphic conditions along Big Springs Creek,
- Provide topographic data to populate a two-dimensional hydrodynamic model (see Section 7.4), and
- Establish baseline geomorphic conditions from which to compare outcomes of potential future restoration activities.

Longitudinal variability in existing baseline conditions are discussed below. Topographic data used in populating the 2-D models are provided in the appendix.

4.4.1 Methods

Channel morphology was characterized through local field topographic surveys of Big Springs Creek using a TOPCON HiperLite Plus Real-Time Kinematic (RTK) survey unit. Longitudinal profiles of the channel bed and water surface elevation were conducted along the channel thalweg while wading. Additionally, sixty-four (64) cross-sections were surveyed across straight reaches and at meander bend apexes from the confluence of the Shasta River to Big Springs Dam (Figure 27). Cross-sections were surveyed across the channel bottom using at least 13 points, with survey point densities greater at locations with topographic variability (i.e. channel margins). Elevations of the bankfull surface were estimated at each cross-section based on topographic breaks in the channel bank. However, extensive bank trampling by cattle and a lack of evidence of bankfull surface inundation hindered clear identification of this surface. Channel width-to-depth ratios were calculated by dividing the bankfull channel width by mean bankfull depth.

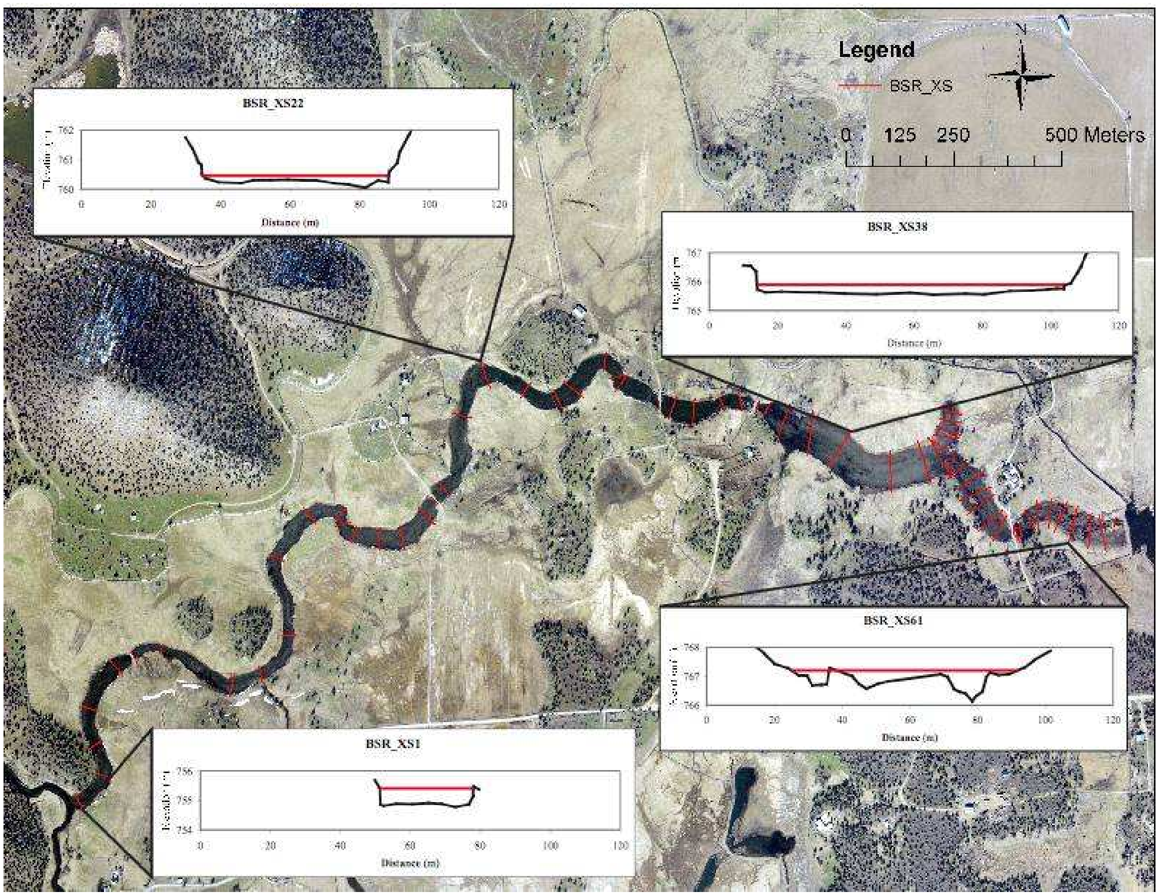


Figure 27. Locations and examples of channel cross sections surveyed on Big Springs Creek showing cross sectional profile and water surface elevation when surveyed.

4.4.2 Data/Analysis

Longitudinal Profile

Longitudinal channel bed and water surface elevation topographic surveys identify reach-scale trends in channel gradient along Big Springs Creek. Discrimination by slope reveals four (4) geomorphologically distinct channel reaches extending from Big Springs Dam to the confluence with the Shasta River: 1) Big Springs Dam to the northern spring alcove; 2) northern spring alcove to the water wheel impoundment; 3) waterwheel impoundment to river kilometer 1.90 and; 4) River kilometer 1.90 to the Shasta River (Figure 28).

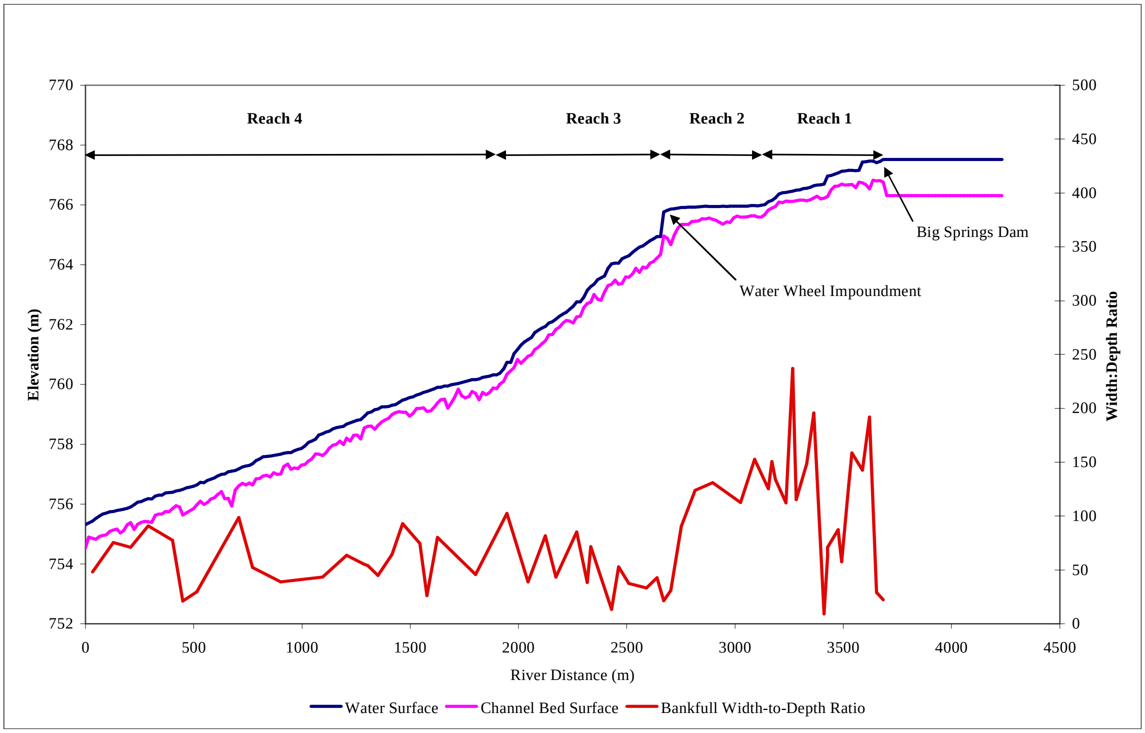


Figure 28. Water surface and channel bed longitudinal profiles plotted with estimated cross-section bankfull width-to-depth ration along Big Springs Creek.

The channel reach from Big Springs Dam to the northerly spring alcove exhibits water surface and channel bed slopes approaching 0.003. Moderate local fluctuations in gradient and water depth are observed, largely in response to the presence of a large beaver dam and localized hydraulic interactions with in-channel woody vegetation (i.e. arroyo willows) extending for approximately 200 meters immediately below the dam. Channel gradient abruptly decreases by a factor of ten downstream from the northerly spring alcove and maintains a relatively constant slope (0.0003) for approximately 300 meters. This gradient reduction appears largely a geomorphic response to a flow-through impoundment at a location known as the “water wheel”. While water surface gradient throughout this channel reach remains largely constant due to streamflow impoundment, streambed elevations decrease towards the downstream end of the reach, creating one of only two pools throughout Big Springs Creek deeper than 1 meter.

Following an approximately 1-meter hydraulic drop through the water wheel structure, channel bed and water surface gradient increases to 0.006 and remain relatively stable for approximately 750 meters downstream. Basaltic bedrock outcrops in the channel bed and on channel margins throughout this reach. Furthermore, fractured basalt provides angular cobble and boulder-sized substrate in this geomorphologically homogenous river reach.

The lowest two kilometers of Big Springs Creek shallows in gradient, with water surface and channel bed slopes approaching 0.003. The bed profile across this reach is considerably more variable, resulting in large longitudinal variability in local water depths.

Cross-Section Surveys

Cross-sectional channel morphologies throughout Big Springs Creek are remarkably wide and shallow. Width-to-depth ratios range from less than nine (9) at laterally-confining road crossings to 237 across a low-gradient riffle immediately upstream from the northern spring alcove (Figure 28). The mean value across all reaches is 84 (including road crossings), with a standard deviation (σ) of 50. Minimum width-to-depth ratios are found at road crossings, which laterally constrict the channel and increase mean stream depth (Figure 28). Reach-averaged width-to-depth ratios remain relatively stable between the mouth of Big Springs Creek and the water wheel (mean = 61; σ = 21), only to nearly double in reaches above the water wheel (mean = 117; σ = 54). Reach-averaged ratios measured in Big Springs Creek are significantly greater than those measured in selected spring-fed streams in Oregon and Idaho, where average width-to-depth ratios are 34 (σ = 24) (Whiting and Moog 2001).

Longitudinal trends in cross-sectional channel form are apparent in Big Springs Creek. Throughout the 2.5 river kilometers from the mouth of Big Springs Creek to the water wheel impoundment, channel geometries are largely rectangular and exhibit minimal lateral asymmetry. Excluding channel road crossings, width-to-depth ratios across this reach are high (mean = 61) and moderately variable (σ = 21) (Figure 28). Water depths also remain consistently shallow through this reach, with a mean depth during the summer of 0.581 meters (σ = 0.15 meters). Large deviations from the mean water depth principally occur across shallow, bedrock-dominated riffles and at deeper bridge crossings (Figure 28). A localized, mid-channel pool not associated with any in-channel structures is present below the confluence between Big Springs and Little Springs Creeks.

The impoundment structure at the water wheel forces a unique set of localized geomorphic conditions over approximately 400 meters from the water wheel to the northerly spring alcove (Figure 28). While channel width remains largely stable across this reach, the gradual reduction in mean water depth away from the impoundment results in a large upstream increase in width-to-depth ratios (Figure 28). Upstream from the northerly spring alcove, width-to-depth ratios increase slightly, but also exhibit much greater variation (mean = 120; σ = 60). Extensive growth of arroyo willows throughout the 200 meters below Big Springs Dam appears largely responsible for this variation.

Willow growth has created numerous islands throughout this channel reach, resulting in a multi-threaded channel with strong lateral variations in channel depth and cross-sectional area (Figure 28).

4.4.3 Conclusions

Big Springs Creek can largely be divided into discrete reaches based on channel slope and cross-section geometry. With the exception of the reach between Big Springs Dam and the northerly spring alcove, localized channel morphologies within each reach are remarkably homogenous – exhibiting relatively stable gradient, water depth, and channel width. Discrete longitudinal differences in channel slope appear dependent upon external geologic conditions, namely erosion resistant bedrock outcroppings in the channel bed and channel margins. Stable morphological characteristics observed across each reach likely reflect stable spring-fed streamflow conditions and resultant sediment transport processes. Cross-sectional channel morphologies throughout Big Springs Creek are remarkably wide and shallow with the exception of five laterally-confining road crossings.

5.0 Food Web and Aquatic Macrophytes

5.1 Introduction

Food web studies provide a framework for understanding many of the key interactions that structure ecological communities. In the spring, summer, and fall of 2008, we conducted biotic sampling intended to document the distribution and abundance of various food web components within Big Springs Creek. Specifically, we examined the temporal dynamics of primary producers (epilithon and aquatic macrophytes), benthic macroinvertebrates, and fishes (see section 6) to elucidate the structure of the aquatic food web and understand important conduits for the flow of energy and material. The common objective of our biotic sampling was to provide a baseline understanding of the important trophic pathways that support juvenile salmonids in Big Springs Creek.

5.1.1 Autochthonous Production

Macrophytes (vascular aquatic plants) and epilithon (matrix of algae, bacteria, fungi, protozoans and non-living organic matter specific to rock surfaces) are the two major aquatic primary producers that serve as the base of the Big Springs Creek food web. Abundant growths of submergent and emergent macrophytes are an especially salient feature of Big Springs Creek throughout much of the year and these organisms play a central role in the ecology of the creek. Macrophytes provide important habitat for macroinvertebrates and fish, regulate the cycling of biologically important nutrients, and influence channel roughness and river stage (Carpenter and Lodge 1986, Sand-Jensen and Mebus 1996, Diehl and Kornijow 1997). Moreover, macrophytes, both as fresh plant material and detritus following plant senescence, serve as a potential food resource to the aquatic food web and as a substrate for epiphyton and invertebrate growth (Suren and Lake 1989, Newman 1991). Similarly, epilithon is often an important source of carbon and energy in riverine ecosystems and has been reported to support the production of many aquatic consumers, including juvenile salmonids (e.g., Bilby and Bisson 1992). Herein we present data on seasonal macrophyte community composition and temporal changes in epilithon and macrophyte standing crops in Big Springs Creek. We define standing crop as the weight of biota per unit area at a given point in time. Such estimates provide a snapshot of system productivity and yield important insight into the trophic basis of production.

5.1.2 Macroinvertebrates

Macroinvertebrates 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 epilithon and detritus) and fishes. Furthermore, certain macroinvertebrate taxa are known to be extremely sensitive to environmental conditions (e.g., temperature, dissolved oxygen, turbidity, etc.) and provide valuable insights into the general health of freshwater ecosystems (Barbour et al. 1999, Davis et al. 2001). We collected macroinvertebrates and quantified rates of exchange (i.e., insect emergence and terrestrial to aquatic input) between Big Springs Creek and the adjacent terrestrial environment. Our objectives were to (*1*) generate macroinvertebrate taxonomic lists for various reaches of Big Springs Creek and the Shasta River, (*2*) document seasonal changes in the macroinvertebrate community; and (*3*) elucidate the key taxa that potentially serve as prey for stream resident salmonids.

5.1.3 Stable Isotope Analysis

Based on results from the seasonal biotic studies introduced above, key members of the food web were subsequently analyzed for natural abundance stable isotope ratios to establish trophic relationships and the flow of carbon and nitrogen within the Big Springs Creek food web. Stable isotope analysis has been widely applied in ecological studies to identify 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 ($\delta^{13}C$) isotopes is based upon the observation that the ratio of the heavy (^{13}C) to light (^{12}C) isotope changes little with each trophic transfer (DeNiro and Epstein 1978, Fry and Sherr 1984). Hence, $\delta^{13}C$ values are effectively conserved up the food chain and may be used to differentiate between alternative carbon resources when the $\delta^{13}C$ values of the potential resources are sufficiently distinct. In contrast to carbon, stable nitrogen isotope ratios (^{15}N:^{14}N or $\delta^{15}N$) 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 $\delta^{15}N$ signature provides an indirect measure of its relative trophic position and ecological role in the community. Unlike traditional gut content analysis, stable isotope ratios provide 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 Big Springs Ranch (BSR) were to identify the important sources of organic matter to stream consumers and determine temporal variability in the structure of the aquatic food web in Big Springs Creek.

A fundamental goal of many stream and watershed restoration programs is to enhance the abundance and growth of juvenile salmonids. Thus, a critical underpinning for the development of effective restoration strategies is a robust understanding of the sources of energy and organic matter on which salmonid production is based. In the following sections we provide novel data concerning the dynamics of epilithon, aquatic macrophytes, and macroinvertebrates in Big Springs Creek, and their contributions to the cycling of energy and material within the aquatic ecosystem.

5.2 *Autochthonous Production*

5.2.1 Methods

Epilithon

Epilithon was sampled from the surfaces of 10 randomly selected cobbles during the spring sample period and from 10 unglazed ceramic tiles (38.4 cm^2) on subsequent sample dates (summer and fall). In the laboratory, all substrates were examined under 10× magnification and any invertebrates encountered were removed and discarded. A rubber template was then used to delineate a known area (8.0 cm^2) and the inside of the template was scrubbed with a stiff-bristled brush. Dislodged material was suspended in a small volume of water and collected on pre-combusted (500˚C for 4 h) Whatman GF/F filters (47 mm diameter; 0.7 µm effective pore size). For each sample period, 5 filters were used to quantify epilithon standing stock and 5 were reserved for natural abundance stable isotope analysis (see section 5.4). For epilithon ash free dry mass (AFDM) determination, filters were transferred to aluminum weigh dishes, oven-dried to a constant weight (48-72 h at 60°C), ashed for 4 h at 500°C, and reweighed. Epilithon standing stock is reported as grams ash-free dry mass per square meter (g AFDM·m^{-2}).

Aquatic Macrophytes

We characterized the aquatic plant assemblage during the spring, summer, and fall of 2008. On each date, six sample sites were randomly selected within the study reach. A square PVC-frame quadrat was used to delineate an area of 0.37 m^2 and all above-ground biomass within the quadrat was removed. Harvested plant material was vigorously agitated in the stream to reduce the presence of clinging macroinvertebrates (epibiota) and other detrital material prior to being placed in individually labeled bags and returned to the laboratory. In the laboratory, samples were separated by species and the individual fractions were dried to a constant mass at 65ºC for ≥ 72h and weighed. Samples were then ashed in a muffle furnace for 4 h at 475ºC, cooled to a constant mass and reweighed to derive AFDM. Mean standing stock for each macrophyte species is reported as grams ash-free mass dry per square meter (g AFDM·m^{-2}).

Our aquatic macrophyte study site was the one location on Big Springs Creek where cattle had complete access to the river throughout the study period. This was unknown at the time of site selection and cattle grazing likely altered (reduced) the standing crop of aquatic macrophytes across all seasons. We observed that other locations in Big Springs Creek where cattle were excluded exhibited higher abundances of macrophytes relative to those documented at our study location (Figure 3). It is recommended that future studies be conducted to accurately estimate plant standing crop and productivity in reaches not impacted by cattle grazing.

5.2.2 Data/Analysis

Epilithon Standing Crop

Epilithon standing crop increased throughout the year yielding statistically significant seasonal differences (ANOVA, p = 0.005; Figure 29). Mean (±1 SE) standing crop was similar during the spring (5.7 ± 1.3 g AFDM·m^{-2}; range = 2.9 to 10.6) and summer (7.0 ± 1.3 g AFDM·m^{-2}; range = 4.7 to 11.9) sample periods. However, epilithon standing crop more than doubled by fall averaging 15.7 ± 2.7 g AFDM·m^{-2} (range = 7.8 to 23.3; n = 5 on all dates).

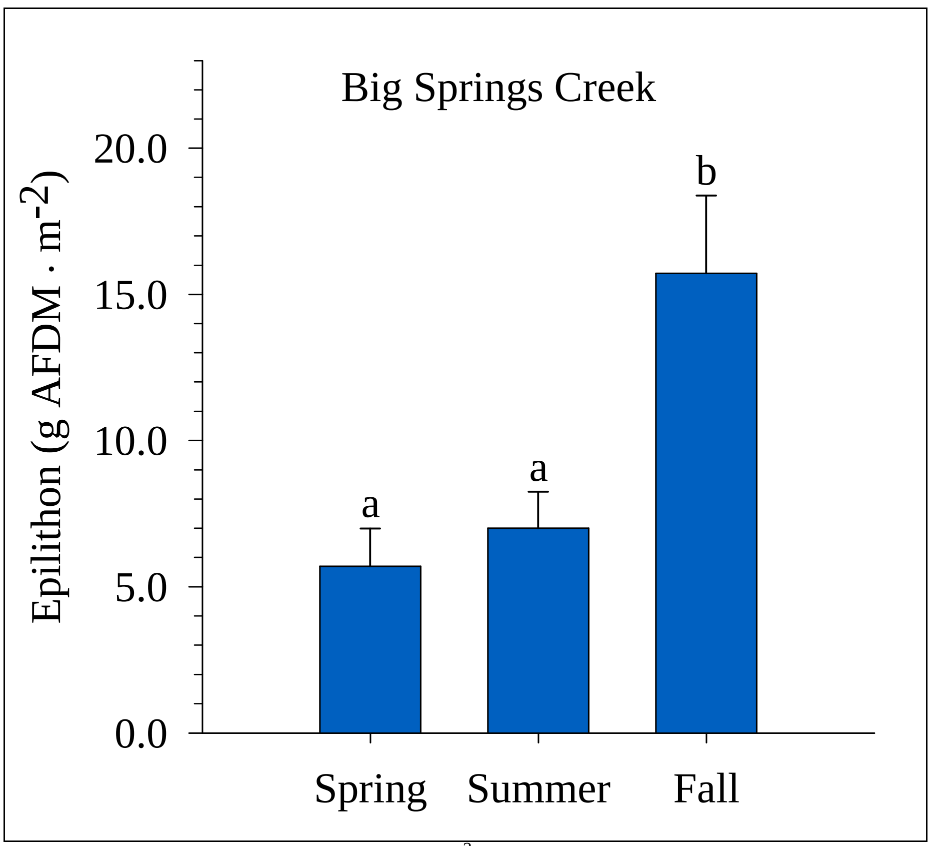


Figure 29. Epilithon standing crop (g AFDM·m^{-2}) during each sample period. Bars represent the mean (± 1SE) of 5 replicate samples. Bars that do not share a common letter are significantly different at α = 0.05.

Macrophyte Standing Crop

The standing crop of aquatic plants (i.e., macrophytes + filamentous algae) increased throughout the year (Figure 30). The lowest total standing crop was observed during the spring, averaging 35.7 ± 10.7 g AFDM·m^{-2} (n = 6; A, Figure 3). Mean total standing crop increased by 282% (136.2 ± 33.0 g AFDM·m^{-2}; n = 6) between the spring and summer sample dates, and by an additional 34% (182.1 ± 60.6 g AFDM·m^{-2}; n = 6) between the summer and fall dates (A). While this temporal increase in plant biomass is ecologically relevant, differences were not statistically different (ANOVA, p = 0.06) due to high variability among the replicate samples.

The Big Springs Creek macrophyte assemblage at our study location was dominated by two taxa: northern watermilfoil (*Myriophyllum sibericum* Kom.) and water smartweed (*Polygonum amphibium* L.). Mean *Myriophyllum* standing crop accounted for 26% (9.2 ± 2.1 g AFDM·m^{-2}), 81% (110.0 ± 30.9 g AFDM·m^{-2}) and 66% (120.2 ± 37.5 g AFDM·m^{-2}) of the entire aquatic plant assemblage in spring, summer, and fall, respectively (ANOVA, p = 0.03; B). *Polygonum* was the dominant macrophyte during the spring accounting for 55% (19.6 ± 8.0 g AFDM·m^{-2}; Figure 31C) of the entire aquatic plant biomass during this sample period. The mean standing crop of *Polygonum* decreased between spring and summer (Δ = -7.0 g AFDM·m^{-2}) but significantly increased between the summer and fall sample periods (Δ = +49.4 g AFDM·m^{-2}, +269 %; ANOVA, p = 0.04, Tukey's HSD p = < 0.05). Filamentous algae were present during the spring and summer and accounted for the majority of the standing stock classified as "other aquatic plants" (Figure 28D).

The prolific growth of aquatic macrophytes in Big Springs Creek makes it a unique ecological environment. Aquatic macrophytes act as a substrate for benthic macroinvertebrates and provided complex habitat for fish, something lacking in Big Springs Creek when aquatic macrophytes are absent. Along with providing direct benefits to invertebrates and fish, growth of aquatic macrophytes increase roughness in the creek resulting in increased depth and a reduced transit time. When depth is increased and transit time reduced, water does not heat as rapidly. By removing the disturbance associated with cattle grazing from Big Springs Creek and allowing the natural growth of aquatic macrophytes, habitat conditions will improve within the creek and downstream into the Shasta River.

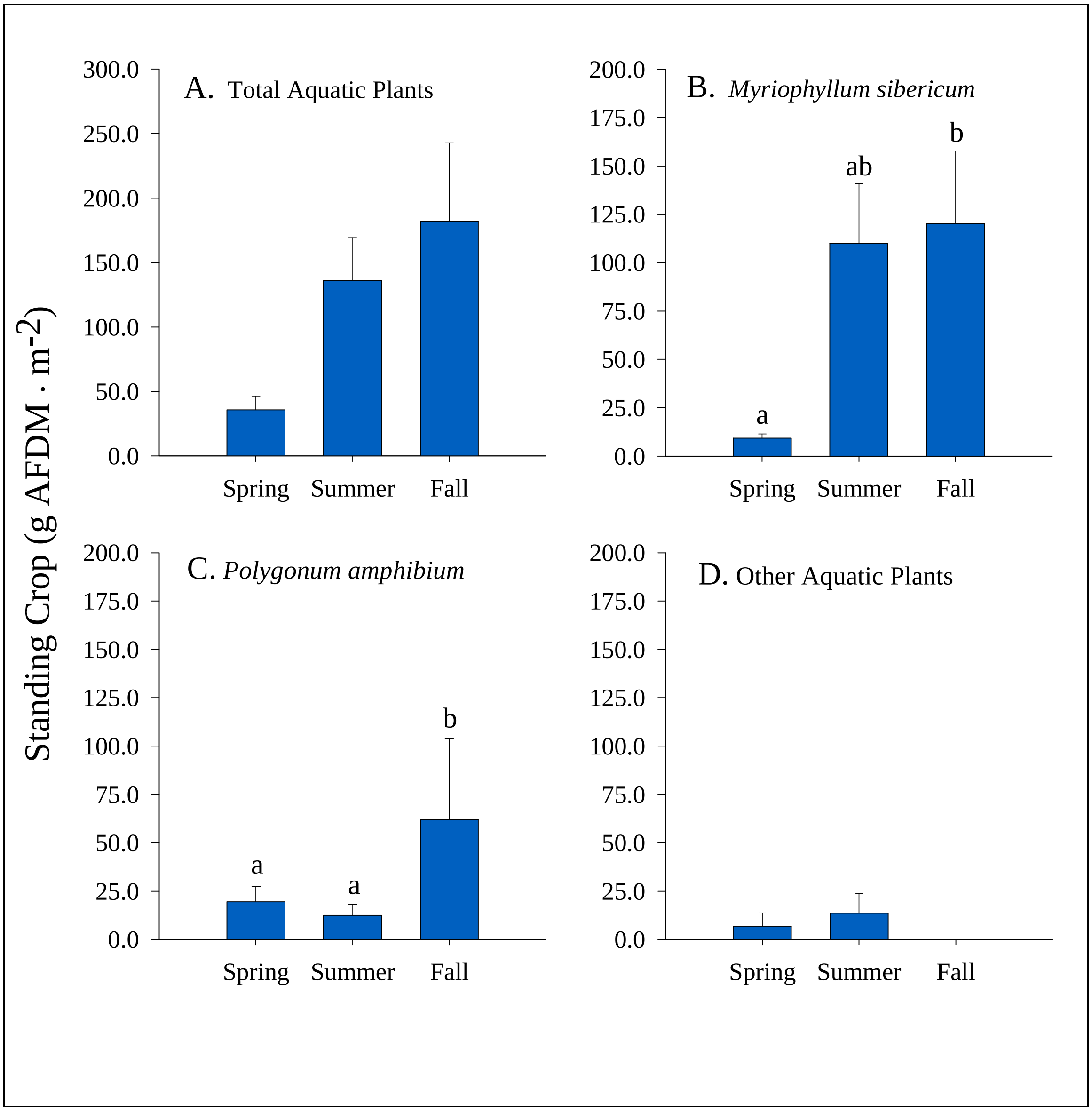


Figure 30. *Mean standing crop (g AFDM·m-2) for all aquatic plants (A), Myriophyllum sibericum (B), Polygonum amphibium (C), and the remainder of the plant community (D) during each sample period. Bars represent the mean (±1SE) of 6 replicate quadrat samples. Bars that do not share a common letter are significantly different at α = 0.05. Note the different scale for panel A.*

5.3 *Aquatic Macroinvertebrates*

5.3.1 Methods

Aquatic macroinvertebrates were collected from Big Springs Creek and Shasta River during March, June, and September of 2008 (spring, summer, and fall) to determine community compositions and temporal changes in the assemblages. Multiple sample sites were selected in an effort to understand the spatial arrangement of macroinvertebrates in Big Springs Creek and the effects of a large spring creek tributary on the invertebrate assemblages of the Shasta River. Sample sites in Big Springs Creek included the spring alcove (BS-Up), a middle river reach (BS-Mid), and the most downstream reach of the creek occurring directly above its confluence with the Shasta River (BS-Low). Additionally, samples were collected from the Shasta River upstream (SR-US) and downstream (SR-DS) of its confluence with Big Springs Creek (Figure 31). At each sample site, a kick net was used to acquire macroinvertebrates, except for the middle reach of Big Springs Creek where a Hess sampler was employed so that qualitative density estimates could be generated.

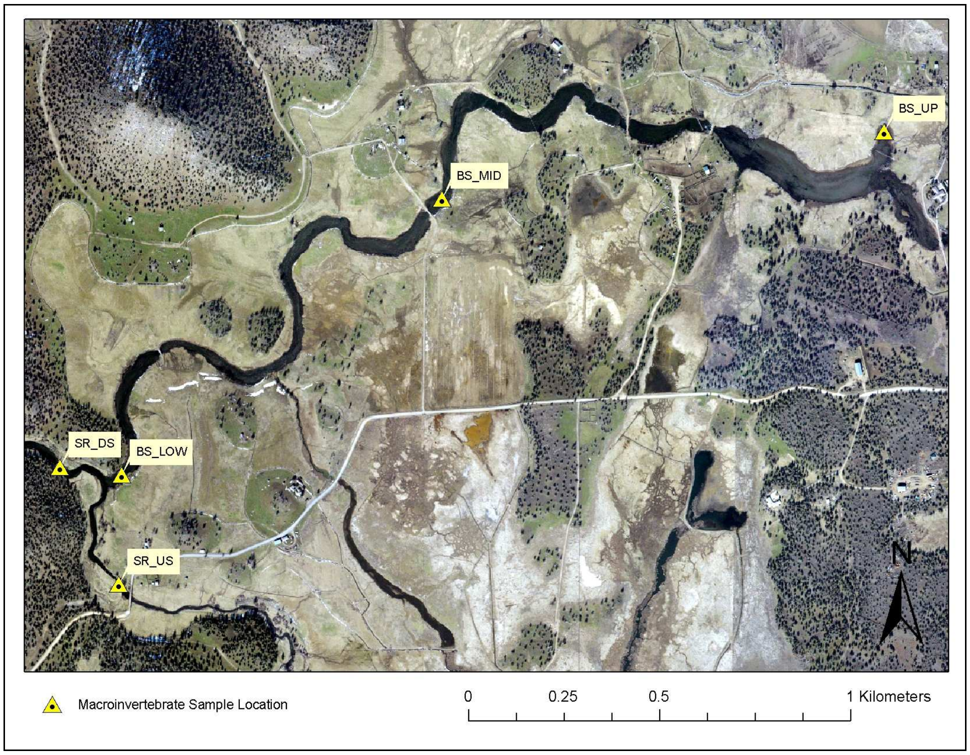


Figure 31. Location map showing benthic macroinvertebrate sample sites in Big Springs Creek (BS) and the Shasta River (SR), Siskiyou County, California.

Qualitative Sampling

Aquatic macroinvertebrate samples were collected from the upper headwater reach of Big Springs Creek, the most downstream reach of Big Springs Creek (BS-Low), and the Shasta River above and below the Big Springs Creek confluence. At each location we established three transects and collected nine individual kick samples corresponding to right, center and left channel for each transect. A standard D-framed kick net (500 µm mesh) was placed immediately downstream of the target sample area and approximately 0.09 m^2 of the streambed was vigorously disturbed for a period of one minute. The nine individual kick samples were then combined in a bucket and the entire sample was elutriated to remove sand, silt, and gravel. The composite sample was preserved in 95 percent ethyl alcohol and returned to the laboratory for processing and identification. Collection locations and methods remained constant across dates, allowing us to examine temporal variation in the relative abundance and diversity of macroinvertebrates.

Quantitative Sampling

Macroinvertebrate samples from the middle reach of Big Spring Creek (BS-Mid) were collected using a modified 21.6 cm diameter Hess sampler (335 µm mesh). We used a tape measure and number table to randomly select the location for a single transect line during each sample period. Five subsamples were then collected at evenly spaced intervals across the length of the transect. For each sample, substrate within the area delineated by the Hess sampler was vigorously disturbed to a depth of 5 cm for one minute. The five resultant subsamples were combined in a bucket and elutriated to remove sand, silt, and gravel. The composite sample was passed through a 250 µm sieve and all retained material was preserved in 95 percent ethyl alcohol and returned to the laboratory for processing and identification.

Taxonomic Determination

In the laboratory, macroinvertebrate samples were evenly distributed over a standardized sorting grid and randomly subsampled to reach a minimum count of 500 organisms. The remainder 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 using Merritt et al. (2008), Thorp and Covich (2001), Smith (2001), Wiggins (1996), as well as various taxonomic-specific references. Ostracoda, Oligochaeta, and Arachnida were identified to class, while Chironomidae were identified to family. Specimens in poor condition or in very young instars were left at the next highest taxonomic level. We selected 12 common macroinvertebrate 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. 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: (*i*) *collectors* which gather or filter fine particulate organic matter; (*ii*) *shredders* which consume coarse particulate organic matter; (*iii*) *scrapers* (grazers) which consume epilithon; (*iv*) *predators*, which capture and feed on other consumers (*v*) *omnivores*, which consume both plant and animal matter; and (*vi*) *parasites* which live in or derive nourishment from other aquatic animals. A description of the specific metrics examined in this study is provided in Table 4.

Table 4. Benthic macroinvertebrate metrics and their expected responses to ecological perturbation.

Macroinvertebrate Metric	Metric Description	Expected Response to Disturbance
Percent EPT	Percent macrobenthos in the orders Ephemeroptera, Plecoptera, and Trichoptera	Decrease
Percent Sensitive Taxa	Percent macrobenthos with tolerance values of 0, 1, or 2 (scale of 10; least to most tolerant)	Decrease
Percent Tolerant Taxa	Percent macrobenthos with tolerance values of 8, 9, or 10 (scale of 10; least to most tolerant)	Increase
Hilsenhoff's Biotic Index	Measure of community tolerance to organic pollution (based on tolerance values and relative abundance).	Increase
Percent CG and CF	Percent of the macrobenthos that collect and gather (CG) or filter (CF) fine particulate organic matter (FPOM)	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 Shredders	Percent of the macrobenthos that shred coarse particulate organic matter (CPOM)	Decrease
Percent Non-Insect Taxa	Percent of macrobenthos that are not insects	Increase
Total Density	Total number of macrobenthos per square meter	Decrease
Taxonomic Richness	Total number or richness of taxa found in sample	Decrease
Simpson's Evenness	Measure of the relative abundance of different species contributing to the taxa richness of a sample. Values range between 0 (least diverse) and 1 (most diverse).	Decrease

5.3.2 Aquatic Macroinvertebrate Results

Our macroinvertebrate sampling was designed to generate three seasonally-specific taxonomic lists for reaches of Big Springs Creek and the Shasta River. Further, sampling was conducted in both Big Springs Creek and the Shasta River in order to understand how a large spring creek system contributes to macroinvertebrate community dynamics in the Shasta River. Due to cost constraints and the nature of the study goals, all sampling events were unreplicated. However, several interesting observations were made based on the data and are discussed below.

The EPT index measures the percentage contribution of Ephemeroptera, Plecoptera, and Trichoptera to the total benthic invertebrate assemblage. These aquatic insects orders are known to be sensitive to environmental stressors and perturbation and therefore, are often used to assess the relative condition and health of lotic communities (Merritt et al. 2008). For each season and among all sample sites, percent EPT was greatest during the spring and summer in the Shasta River above the Big Springs Confluence (47% and 54%, respectively; Figure 32).

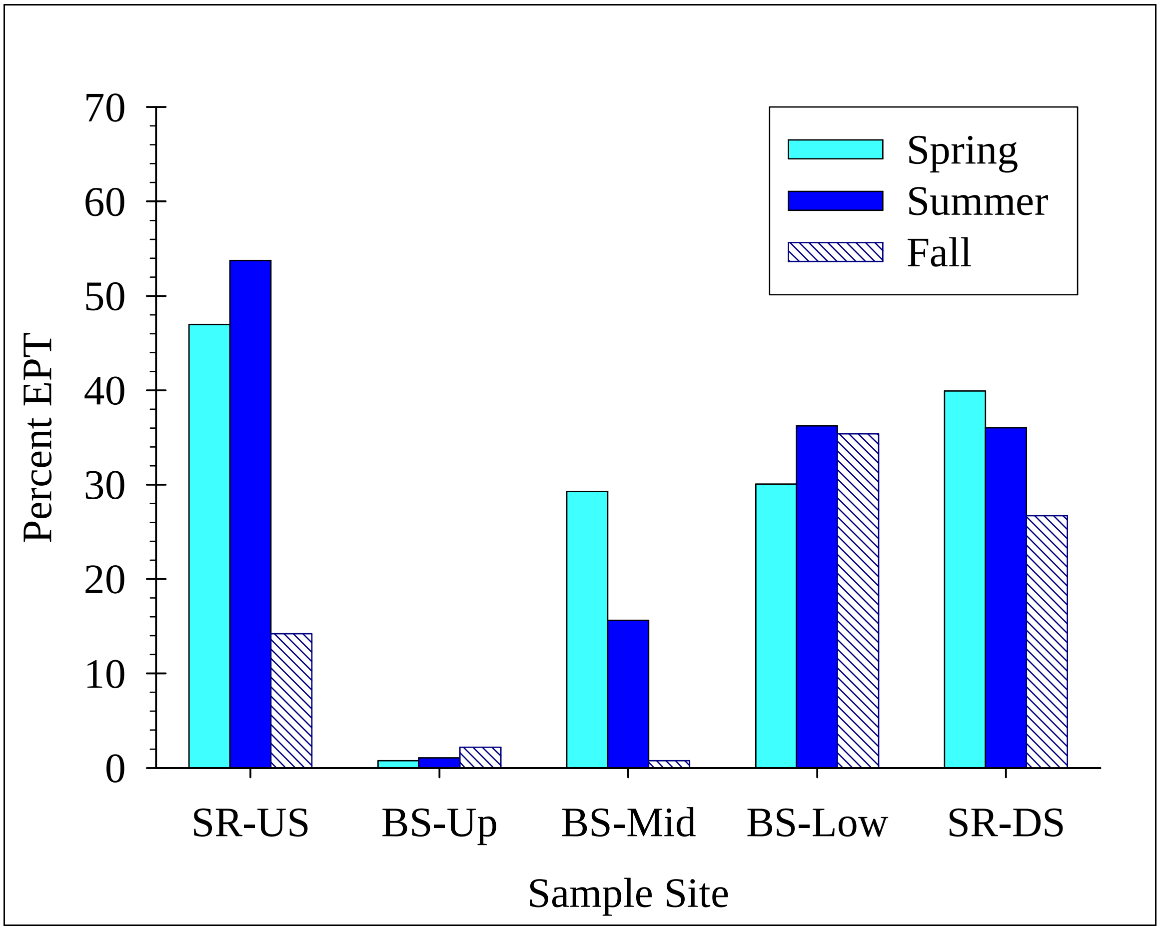


Figure 32. *Percentage of the total macroinvertebrate assemblage represented by the aquatic insect orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) at five sample locations: the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.*

In contrast, EPT contributions to benthic assemblages in the upper reach of Big Springs Creek were consistently the lowest across all seasons. We interpret these low percentages of EPT to be primarily a function of low overall variability of the abiotic factors in this spring-fed reach (e.g., thermal regime, water chemistry, discharge, and habitat heterogeneity)(Vannote et al. 1980). Most strikingly, we observed an abrupt temporal decline in EPT index at the middle Big Springs Creek reach from 29% in the spring to 15% in the summer to less than 1% in the fall. Strong declines in EPT taxa are often associated with increases in fine sediment inputs and water temperature from poor grazing and agricultural management strategies (Miller et al. 2007, Larsen et al. 2009). Seasonal declines in EPT taxa may be associated with current cattle and irrigation management practices on Big Springs Creek.

In contrast to the EPT index, the abundance of tolerant organisms (those with published tolerance values > 8) and non-insect taxa have been associated with several anthropogenic impacts to lotic systems, including organic pollution (Hachmöller et al. 1991, Klemm et al. 2003, Miller et al. 2007). Herbst et al. (2008) suggested that the unique water chemistry of some spring creeks (primarily elevated specific conductivity) may also be associated with natural increases in some non-insect taxa such as gastropods. Tolerant organisms and non-insect taxa accounted for a larger percentage of the overall benthic community in Big Springs Creek across all reaches when compared with the Shasta River (Figures 29, 30). With the exception of lower Big Springs Creek and the Shasta River below Big Springs Creek, percentages of tolerant organisms and non-insect taxa increased throughout the year at each sample reach. Tolerant organisms accounted for approximately 87% and 94% of the entire assemblage at the upper and middle Big Springs sites, respectively, during the fall sampling period. Additionally, non-insect taxa in those same reaches accounted for 95% and 97%, respectively, of the entire macroinvertebrate assemblage during the same sampling period. Conversely, the Shasta River exhibited lower overall percentages of tolerant organism and non-insect taxa, with peaks in these metrics occurring at approximately 30% and 53%, respectively, during the summer sampling period.

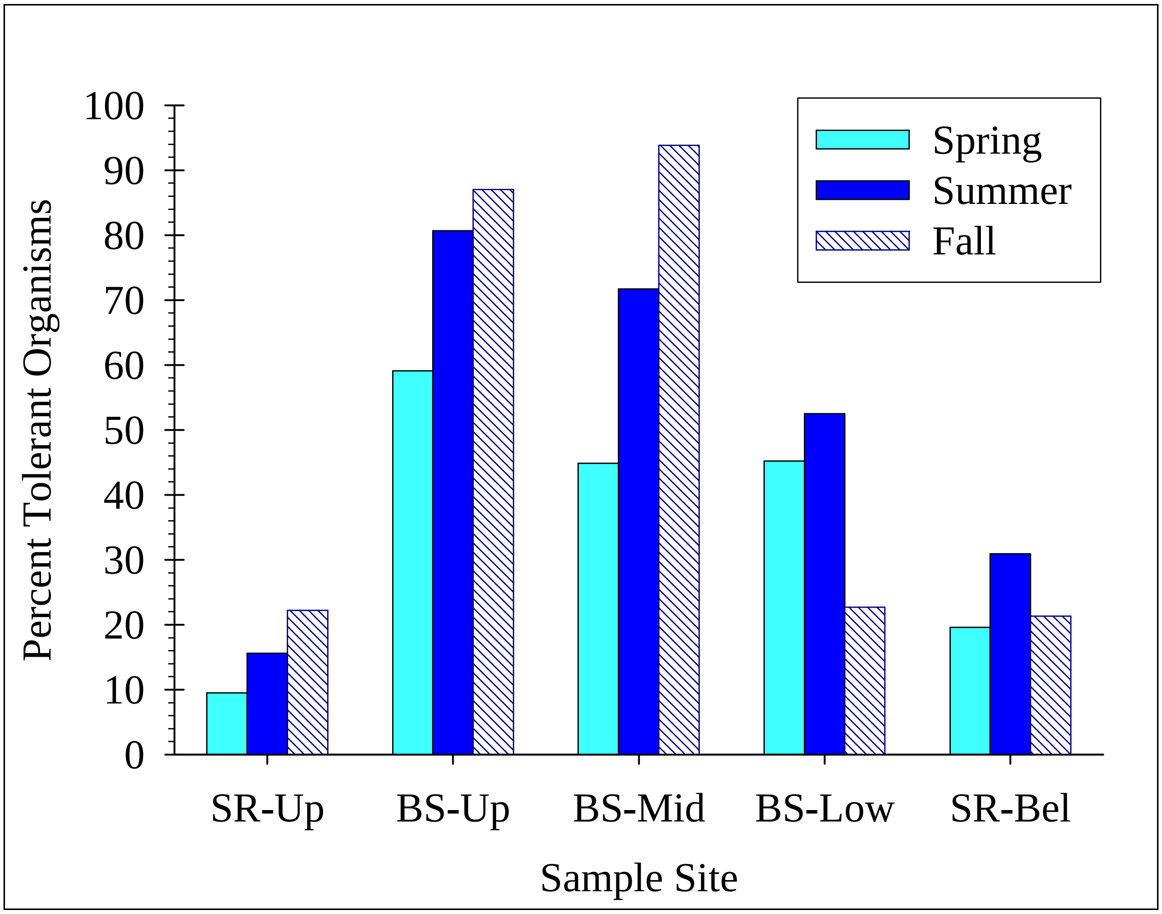


Figure 33. Percentage of the total macroinvertebrate assemblage represented by tolerant organisms (i.e., those taxa with published tolerance values ≥ 8; see Table 5). Sample sites are the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

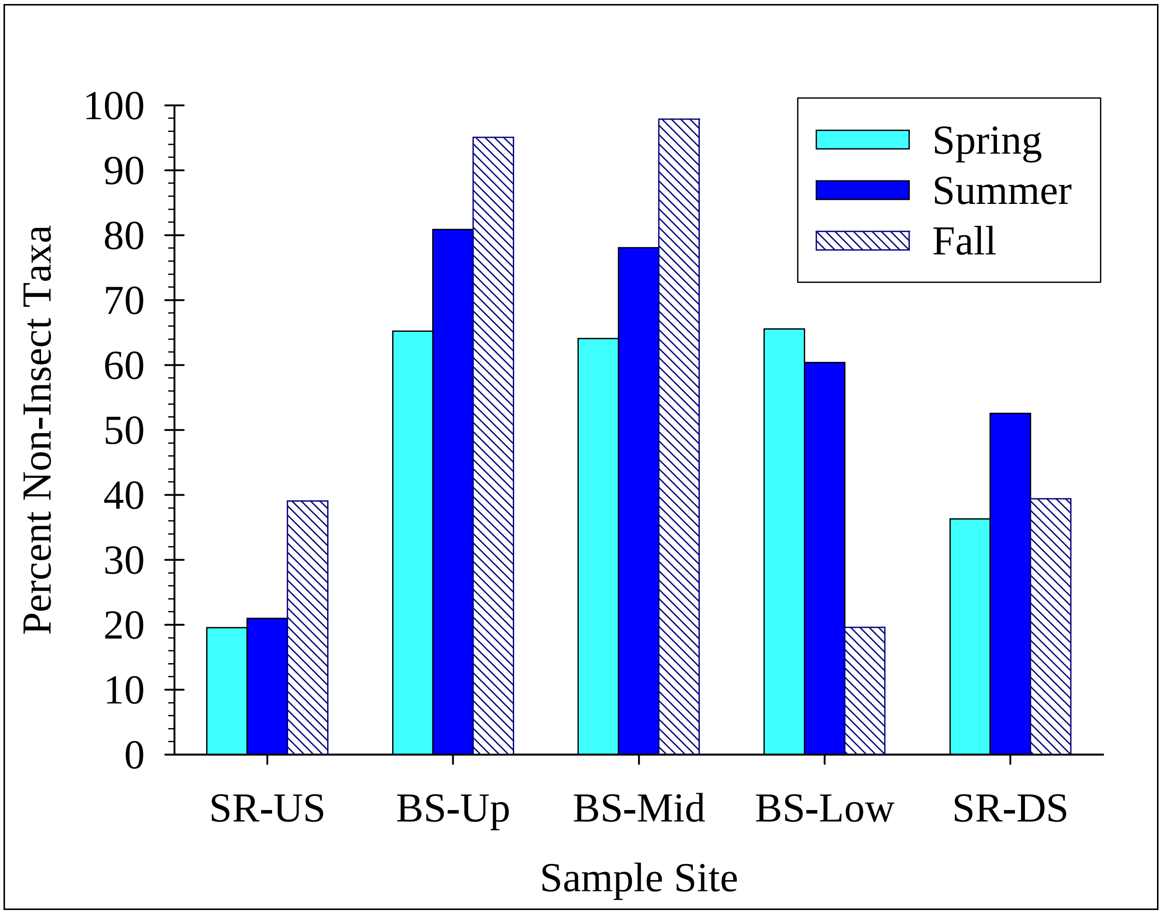


Figure 34. Percentage of the total macroinvertebrate assemblage comprised of non-insect taxa. Sample sites are the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

Macroinvertebrates have evolved several different functional feeding strategies in order to exploit various carbon sources, both allochthonous and autochthonous in origin. The abundance or absence of particular functional feeding groups provides direct insight into the types of organic matter available for uptake by particular macroinvertebrates. Collector-gatherer insects dominated the macroinvertebrate assemblage at all sites on both Big Springs Creek and the Shasta River over all sample periods (Table 5), at times accounting for nearly 98% of the entire assemblage (BS-Mid during the summer sampling period). However, Shasta River sample sites showed a greater overall abundance of scrapers during all seasons relative to Big Springs Creek Figure 35), suggesting that epilithon may be an important carbon source for macroinvertebrates in these reaches. Shredding macroinvertebrates were rare in Big Springs Creek and the Shasta River during all seasons, never accounting for more than 0.4% of the entire macroinvertebrate assemblage for each reach (Table 5). The ubiquitous nature of collector-filterers, coupled with an absence of shredders, implies that CPOM-FPOM breakdown processes and transport may not follow traditional pathways associated with the river continuum. Rather, shredder-mediated breakdown of CPOM may be replaced by sources of FPOM associated with bank destabilization and fecal matter from grazing in these reaches (Scrimgeour and Kendall 2003).

Table 5. Seasonal of macroinvertebrate metrics calculated for the Shasta River and Big Spring Creek reaches. Individual macroinvertebrate metrics are defined in the Table 4.

	Spring Sample Period				
Macroinvertebrate Metric	**SR-US**	**BS-Up**	**BS-Mid**	**BS-Low**	**SR-DS**
Percent EPT	46.9	0.6	8.6	9.0	15.9
Percent Sensitive Taxa	2.0	0.0	0.0	0.0	0.6
Percent Tolerant Taxa	9.5	34.9	20.1	20.4	3.8
Hilsenhoff's Biotic Index	5.1	7.1	6.4	6.6	5.5
Percent CG and CF	72.8	93.1	89.4	92.7	79.8
Percent Predators	2.0	3.6	1.0	0.4	3.5
Percent Scrapers	24.9	2.5	4.9	5.2	15.4
Percent Shredders	0.0	0.0	0.0	0.0	0.0
Percent Non-Insect Taxa	19.5	65.2	64.1	65.5	36.3
Density (Organisms•m^{-2})	1427	1925	8219	1469	1780
Taxonomic Richness	18.0	11.0	14.0	15.0	23.0
Simpson's Evenness	0.8	0.6	0.7	0.6	0.8

	Summer Sample Period				
	SR-US	**BS-Up**	**BS-Mid**	**BS-Low**	**SR-DS**
Percent EPT	53.7	1.1	15.6	36.2	36.0
Percent Sensitive Taxa	22.9	0.0	0.7	10.3	10.5
Percent Tolerant Taxa	15.6	80.7	71.7	52.5	30.9
Hilsenhoff's Biotic Index	4.4	7.6	7.1	6.1	5.4
Percent CG and CF	70.0	86.6	97.6	84.9	79.5
Percent Predators	4.9	13.2	0.1	3.6	4.3
Percent Scrapers	21.8	0.2	1.3	2.7	15.1
Percent Shredders	0.0	0.0	0.0	0.2	0.2
Percent Non-Insect Taxa	21.0	80.8	78.1	60.3	52.5
Density (Organisms•m^{-2})	4299	3025	56750	1086	1271
Taxonomic Richness	26.0	7.0	15.0	22.0	26.0
Simpson's Evenness	0.9	0.5	0.5	0.7	0.9

	Fall Sample Period				
	SR-US	**BS-Up**	**BS-Mid**	**BS-Low**	**SR-DS**
Percent EPT	14.2	2.2	0.8	35.4	26.7
Percent Sensitive Taxa	4.0	2.4	0.2	4.8	11.6
Percent Tolerant Taxa	22.2	87.0	93.8	22.7	21.3
Hilsenhoff's Biotic Index	5.9	7.4	7.8	5.8	4.9
Percent CG and CF	80.7	85.6	96.2	88.3	53.4
Percent Predators	2.1	10.6	0.5	2.7	1.1
Percent Scrapers	14.1	3.7	0.3	4.4	41.4
Percent Shredders	0.1	0.0	0.0	0.4	0.2
Percent Non-Insect Taxa	39.0	95.1	97.9	19.6	39.4
Density (Organisms•m^{-2})	9006	6542	82750	1555	1907
Taxonomic Richness	24.0	11.0	11.0	23.0	24.0
Simpson's Evenness	0.8	0.4	0.1	0.8	0.9

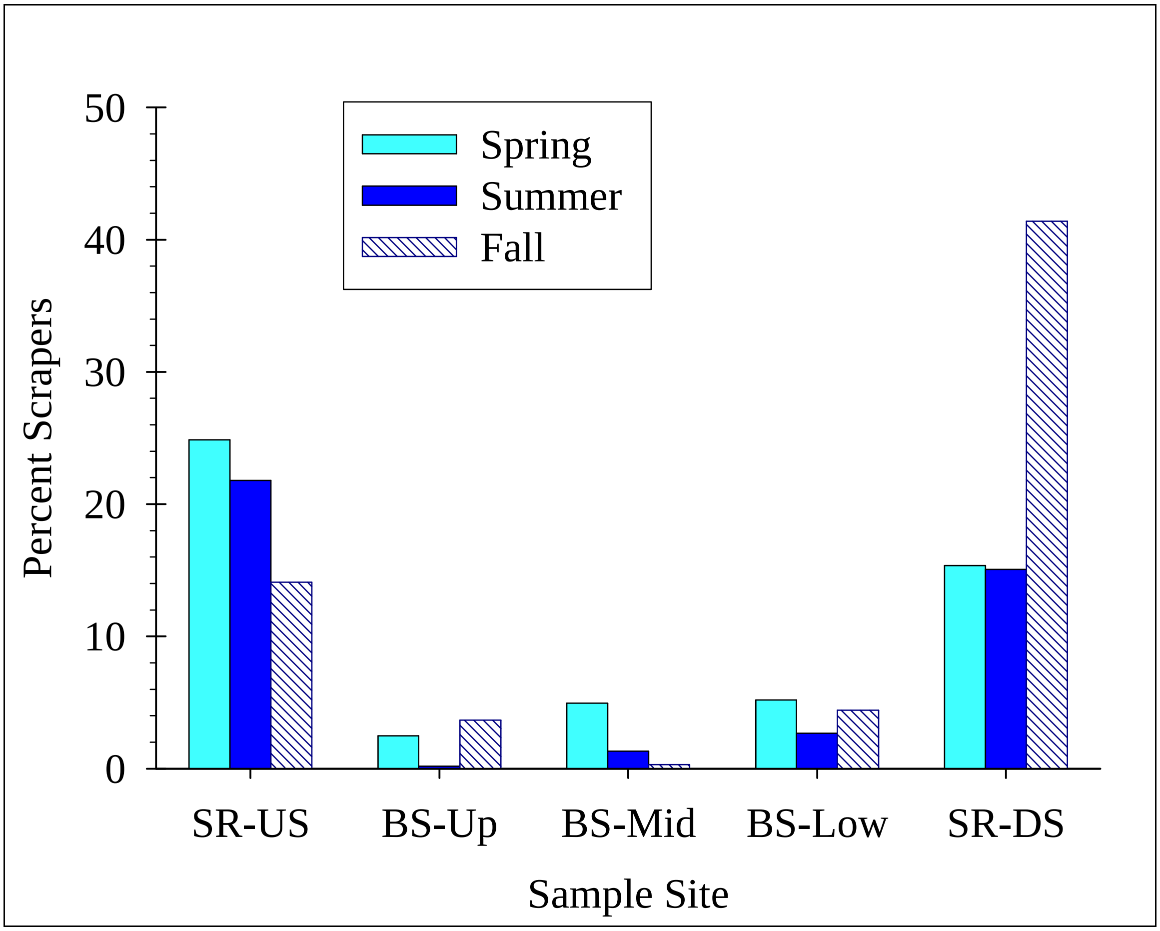


Figure 35. Percentage of the total macroinvertebrate assemblage represented by members of the scraper functional feeding group. Sample sites are the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

Another interesting aspect of the macroinvertebrate data was the apparent paucity of Plecoptera (stoneflies) from both Big Springs Creek and the Shasta River. This finding is especially notable because previous macroinvertebrate surveys conducted within the basin (DWR unpublished data, Great Northern Corporation 1999) reported the presence of multiple plecopteran families. Similarly, data collection on the Shasta River above Dwinnell Dam showed that three families of Plecoptera (Nemouridae, Chloroperlidae, and Perlodidae) accounted for greater than 5% of the macroinvertebrate assemblage during the summer sampling period in 2008 (R. Lusardi, unpublished data). Conversely, plecopterans never accounted for more than 1.7% and 1.0% of the macroinvertebrate assemblages in the Shasta River below Dwinnell Dam and in Big Springs Creek, respectively (Figure 36). Plecoptera are regarded as a highly sensitive order of aquatic insects that require cold, well-oxygenated water with low turbidity, and stable substrates (Merritt et al. 2008). Reductions in plecopteran abundance to overall macroinvertebrate assemblage structure have been correlated with increased fine sediment production, water temperature, and organic pollution from grazing and agricultural management (Scott et al. 1994, Miller et al. 2007, Larsen et al. 2009). A range of abiotic factors may contribute to declines in plecopteran abundance below Dwinnell Dam. However, it should be noted that the riparian corridor and general channel morphology of the study reach above Dwinnell Dam remain in good condition.

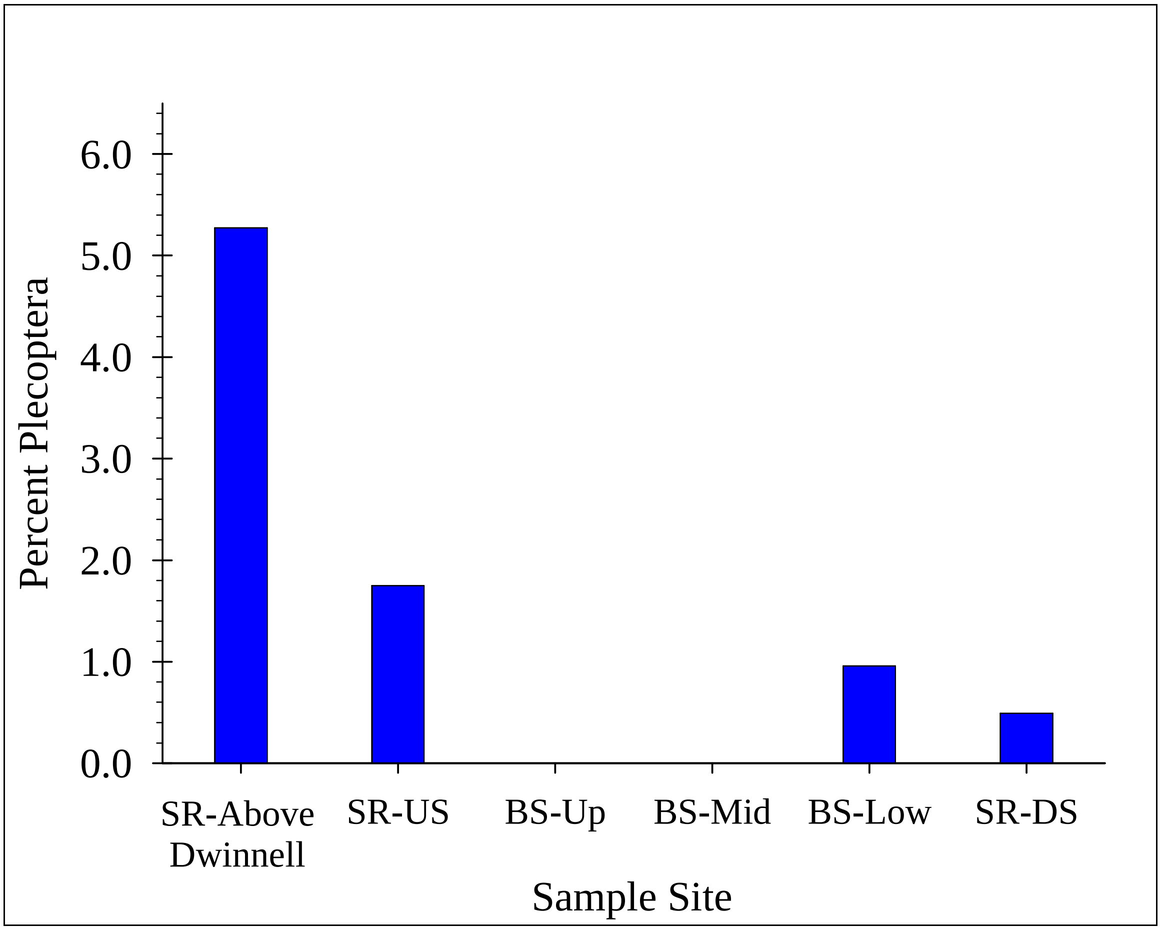


Figure 36. Percentage of the total macroinvertebrate assemblage represented by organisms belonging to the insect order Plecoptera. Data are presented for the summer sample period only. The six sample sites are: the Shasta River (SR) upstream of Dwinnell Dam, upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

Biological diversity consists of two primary metrics: species richness and species evenness. Each metric may be influenced by a number of structuring forces such as biotic competition, colonization rates, anthropogenic impacts, frequency of the natural disturbance regime, overall ecosystem stability, and habitat heterogeneity (Townsend 1997). Consequently, it is often difficult to discern the driving forces associated with changes in biological diversity. The Shasta River generally exhibited higher macroinvertebrate richness and evenness relative to Big Springs Creek during all sample periods (Figure 37, Figure 38). Temporal declines in evenness and richness were most apparent in the middle Big Springs Creek reach between the spring and fall. Evenness declined from 0.7 to 0.1 (1.0 being the highest evenness attainable) and richness declined from 14 to 11 individuals. However, it is important to note that five of the 11 taxa in fall were represented by a single organism, suggesting that a taxonomic richness of 11 may be misleading.

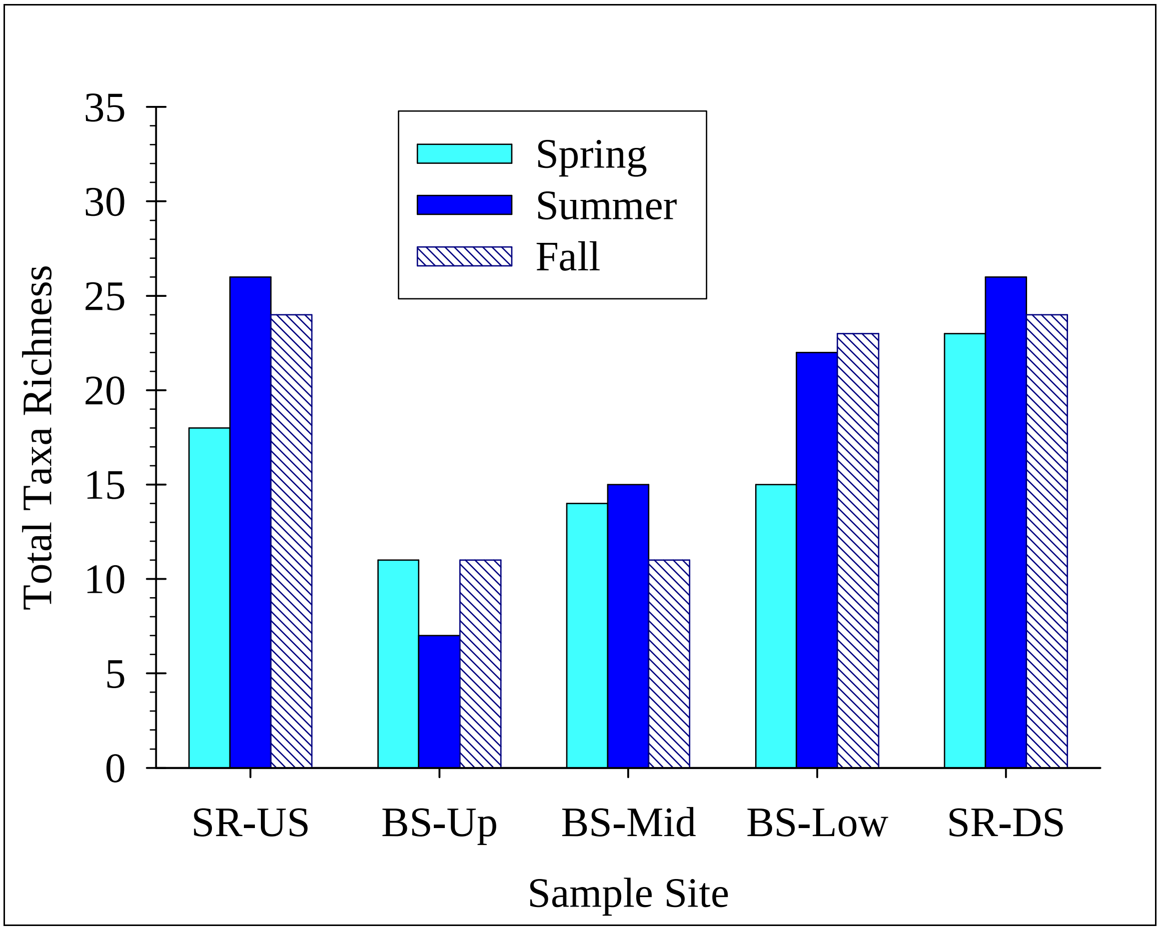


Figure 37. Benthic macroinvertebrate total taxa richness by season. Sample sites are the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

Relative to Big Springs Creek, the Shasta River experiences a higher frequency of natural disturbance events such as flooding, and therefore, may be expected to support higher biological diversity consistent with the intermediate disturbance hypothesis (Connell 1978, Townsend et al. 1997). However, others have shown that while high species evenness may be directly correlated with the frequency of the disturbance regime, species richness may increase with ecosystem stability (Death and Winterbourn 1995). This is contrary to our results for Big Springs Creek and may suggest that grazing and agricultural impacts have reduced overall habitat heterogeneity and altered macroinvertebrate community structure. Further, 93% of the fall sample from the middle reach of Big Springs Creek (BS-Mid) was dominated by the tolerant amphipod *Hyalella*, while EPT taxa accounted for less than 1% of the entire assemblage. Taxonomic richness and evenness values may be naturally lower on Big Spring Creek relative to the Shasta River (consistent with the intermediate disturbance hypothesis), but strong seasonal declines in evenness, richness, and EPT taxa may indicate direct perturbations to the system.

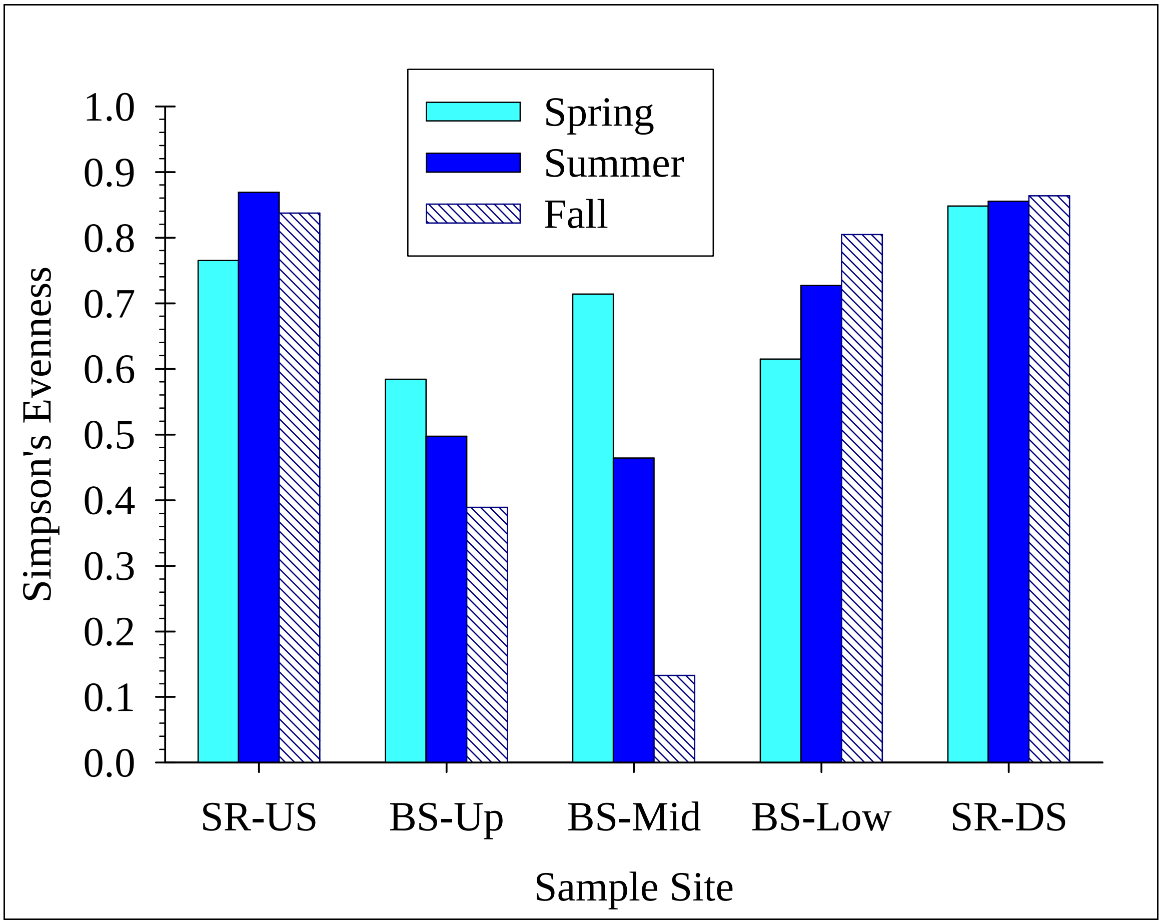


Figure 38. Simpson's evenness values for the benthic macroinvertebrate community. Sample sites are the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

The biotic index, a composite measure of benthic community tolerance to organic pollution, can be used to describe the general condition of lotic habitats (Hilsenhoff 1987). We used a modified biotic index scoring system (Table 6) which included tolerance values associated with genus level identification.[4] Overall, both the Shasta River and Big Springs Creek exhibited low biotic index values indicating that water quality was of "poor" to "fair" condition. The lowest biotic index value of all sites over all seasons (7.8) occurred during the fall on the middle reach of Big Springs Creek, this was concurrent with heavy cattle grazing in the creek. In fact, this reach of Big Springs Creek showed consistent declines in water quality between the spring and fall sample periods. Overall, the Shasta River exhibited higher biotic index values than Big Springs Creek (average over all samples and seasons) at 5.5 (fair water quality) and 6.9 (fairly poor water quality), respectively (Figure 39).

[4] The biotic index was calculated by summing the product of each organism's known tolerance value and abundance and dividing by the total number of organisms in the sample.

Table 6. Criteria for the evaluation of water quality using Hilsenhoff's Biotic Index (HBI; Hilsenhoff 1987). HBI values are derived from macroinvertebrate tolerance values weighted by the number of individuals of each taxa in the total sample.

HBI Value	Water Quality	Degree of Organic Pollution
0.00-3.50	Excellent	No Apparent Organic Pollution
3.51-4.50	Very Good	Possible Slight Organic Pollution
4.51-5.50	Good	Some Organic Pollution
5.51-6.50	Fair	Fairly Signficant Pollution
6.51-7.50	Fairly Poor	Signficant Organic Pollution
7.51-8.50	Poor	Very Signficant Organic Pollution
8.51-10.0	Very Poor	Severe Organic Pollution

Extremely high densities of macroinvertebrates, mainly amphipods, were observed during the summer and fall on the middle reach of Big Springs Creek (Table 5). Densities were approximately 23 and 17 times greater, respectively, in this reach when compared with all other reaches of Big Springs Creek and the Shasta River. The densities (>80,000·m^{-2} in the fall) can only be described as extraordinary and may be a strong indicator of the intrinsic potential of this spring fed system to support juvenile salmonids. Implementation of best management practices, including riparian fencing may improve overall conditions on Big Springs Creek and the Shasta River by improving habitat heterogeneity, channel form, riparian recruitment, and overall water quality conditions (see section 7 for more details). Such improvements are subsequently expected to improve overall macroinvertebrate community structure and ecological function.

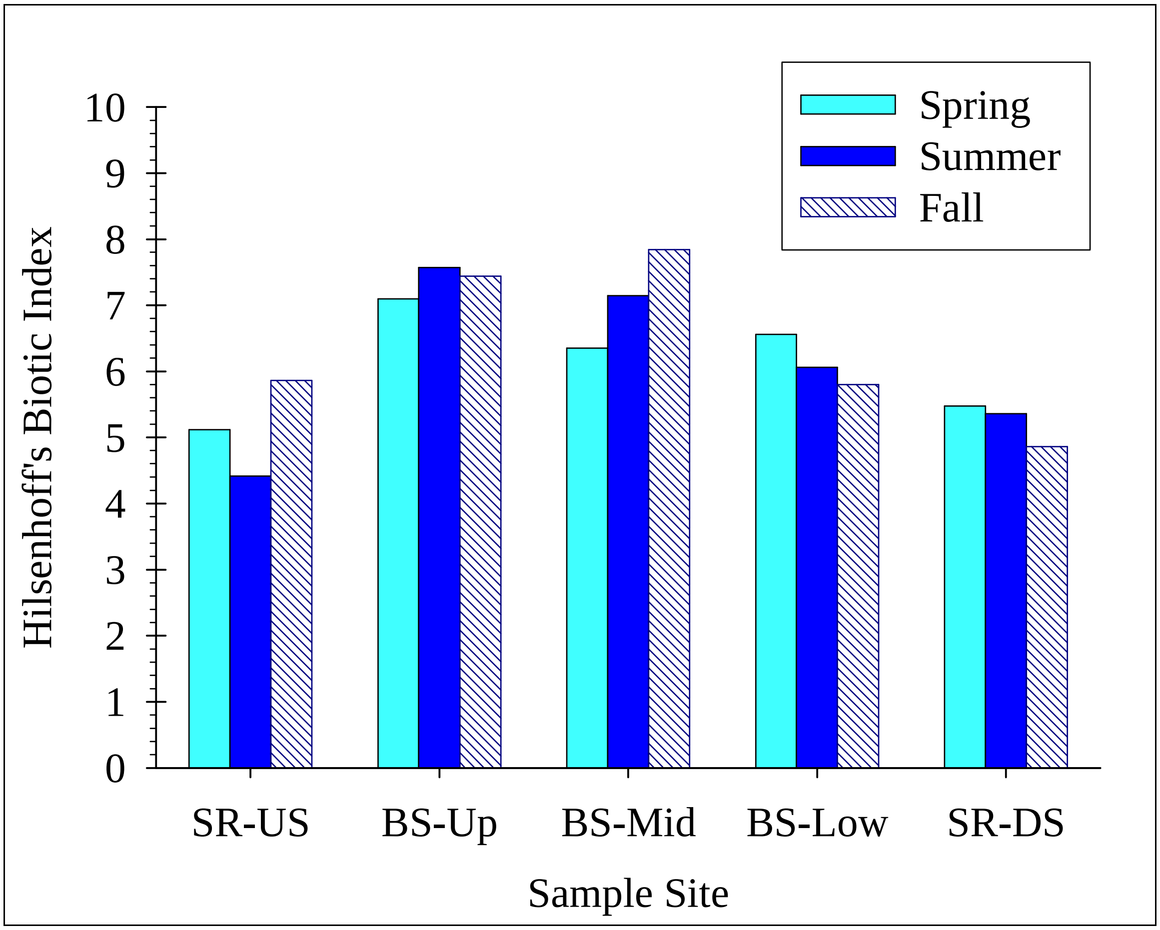


Figure 39. Hilsenoff's Biotic Index values generated from the benthic macroinvertebrate assemblage during each season. Sample sites are the Shasta River upstream (SR-US) and downstream (SR-DS) of the confluence with Big Springs Creek, and upper (BS-Up) middle (BS-Mid) and lower (BS-Low) Big Springs Creek.

5.4 *Food Web Analysis*

5.4.1 Methods

Allochthonous Inputs

Clear plastic floating pan traps (55 cm long × 40 cm wide × 14 cm deep) were used to quantify inputs of allochthonous material, both plant material and terrestrial insects, to Big Springs Creek. We partitioned the width of the stream into three subsections (left, center, and right) and randomly positioned a single pan trap in each subsection. This process was repeated at three locations longitudinally to yield nine replicate pan trap samples per sample period. Once positioned, each pan trap was filled with approximately 3.0 cm of water and a small amount of surfactant was added to retain captured invertebrates. Collection dates were 3-8 July (summer) and 2-7 October (fall). At the conclusion of each sample period, trap contents were sieved through 250 µm mesh and retained material was immediately frozen on dry ice. In the laboratory, samples were thawed, rinsed with distilled water and dried to a constant mass (55 ˚C for ≥ 48 h). Dried samples were examined under 10-20× magnification and separated into three broad categories: plant, animal, and other materials. The individual fractions were then weighed (± 0.1 mg) using a Mettler AE-160 digital balance (Mettler-Toledo International, Greifensee, Switzerland), ashed for 4 h at 500°C, and reweighed to determine ash-free dry mass (AFDM). Allochthonous input categories are expressed as mean AFDM per square meter per day (g $AFDM \cdot m^{-2}\ day^{-1}$).

Insect Emergence

To assess the flux of emerging aquatic invertebrates, we deployed six square-pyramid floating emergence traps (250 µm mesh) that sampled 0.093 m^2 of stream surface. Six emergence traps were haphazardly positioned longitudinally throughout the study reach and anchored in place using rebar. Emergence was quantified for five consecutive days beginning on 3 June (summer) and 2 October (fall). Following each collection period, captured invertebrates were frozen on dry ice and returned to the laboratory for biomass determination. In the laboratory, samples were dried to a constant mass (55 ˚C for ≥ 48 h), weighed (± 1.0 µg) using a PerkinElmer AD-4 auto-balance (PerkinElmer, Waltham, MA, USA), ashed, and reweighed to determine AFDM. No attempt was made to establish the taxonomic composition of the invertebrate emergence samples. Emergence data are expressed as AFDM per square meter of stream surface per day (mg $AFDM \cdot m^{-2}\ day^{-1}$).

Food Web Sampling

To understand the seasonal dynamics of carbon sources that serve as the energetic base of the Big Springs Creek food web, we collected five types of organic matter on each sampling date: epilithon (i.e., matrix of algae, bacteria, fungi, protozoans and non-living organic matter), seston, detritus, aquatic macrophytes, and filamentous algae.

Seston (suspended fine particular organic matter (FPOM); particles > 0.45 µm to < 1.0 mm) was sampled by filtering stream water through pre-combusted GF/F filters until the filters were lightly colored. In all cases, ≤1.5 liters of filtered water produced sufficient material for isotopic analyses. Seston filters were immediately placed in individually labeled opaque bags and frozen on dry ice. Ten replicate seston samples were collected on each sampling date. Five samples were analyzed for natural abundance stable isotope ratios and five were used to quantify seston concentration (mg $AFDM \cdot L^{-1}$). Epilithon sample collection and processing methods are detailed in section 5.2.1.

Detrital samples (coarse particulate organic matter (CPOM); particles >1.0 mm in diameter) were handpicked from the streambed and consisted mainly of decomposing aquatic macrophyte fragments and conditioned terrestrial leaf litter. Aquatic macrophytes were collected by hand from various locations in the study reach. Harvested samples were vigorously agitated in a bucket of 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 examined microscopically (10-20× 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 D-framed kick net and by handpicking organisms from the substrate. Macroinvertebrate samples were passed through a 500 µm sieve and all retained material was frozen (-80˚C) until taxonomic identification and stable isotope preparation. Analysis of the macroinvertebrate community was restricted to representative taxa from each major functional feeding group (Cummins 1973, Cummins and Klug 1979). Functional feeding group (FFG) designations are based on how an organism acquires food and included: (*i*) *collectors*, which gather or filter fine particulate organic matter; (*ii*) *scrapers* (grazers) which consume epilithic biofilms; (*iii*) *predators*, which capture and feed on other consumers; (i*v*) *omnivores*, which consume both plant and animal matter; and (v) *parasites*, which live in or derive nourishment from other aquatic animals. A list of the specific taxa analyzed during each sample period is provided in Table 7**.**

Table 7. Food web constituents analyzed for natural abundance C and N stable isotope ratios during each sample period. Functional feeding group (FFG) abbreviations are collector-filterers (CF); collector-gatherers (CG); scrapers (SC); omnivores (OM); predators (P); and parasites (PA). Taxon codes are used to identify organisms in the stable isotope bi-plots (Figure 43, Figure 47, and Figure 50).

		Taxon	Life		Sample Period		
Food Web Component		Code	Stage	FFG	Spring	Summer	Fall
Organic Matter							
Epilithon		1		n/a	x	x	x
Detritus (CPOM)		2		n/a	x	x	x
Seston (FPOM)		3		n/a	x	x	x
Macrophytes	*Myriophyllum sibiricum*	4		n/a	x	x	x
	Polygonum amphibium	5		n/a	x	x	x
Filamentous Algae		6		n/a	x	x	
Macroinvertebrates							
Diptera	Simuliidae (*Simulium* sp.)	7	Larvae	CF	x	x	
Coleoptera	Elmidae	8	Larvae	CG	x		x
Diptera	Chironomidae	9	Larvae	CG	x	x	x
Ephemeroptera	Baetidae	10	Larvae	CG	x	x	x
Amphipoda	Hyalellidae (*Hyalella*)	11	Adults	CG	x	x	x
Oligochaeta		12	Adults	CG		x	x
Ephemeroptera	Heptageniidae	13	Larvae	SC		x	
Gastropoda	Pleuroceridae (*Juga* sp.)	14	Adults	SC	x	x	x
Trichoptera	Brachycentridae	15	Larvae	OM	x	x	x
Plecoptera	Perlodidae	16	Larvae	P	x	x	
Hirudinea	Glossiphoniidae (*Helodbella*)	17	Adults	PA	x	x	x
Fishes							
Chinook Salmon		18	Juvenile	P	x		
Coho Salmon		19	Juvenile	P	x	x	
Steelhead Trout		20	Juvenile	P	x		

Fish samples utilized in our study were obtained from the California Department of Fish and Game, Yreka office. Our reliance on donated samples resulted in different members of the fish community being available for analysis during each sample period. Consequently, while we had adequate replication during the spring sample period, we only received a pair of coho salmon (*Oncorhynchus kisutch*) during the summer sampling period, and no fish were provided for analysis during the fall. All fish samples were frozen and later 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. Excised muscle samples were placed in individually labeled vials and prepared for natural abundance stable isotope analysis as detailed below. Fish species analyzed during the course of this investigation included: Chinook salmon (*O. tshawytscha*), coho salmon, and steelhead trout (*O. mykiss*).

5.4.2 Laboratory Methods

Stable Isotope Analyses

Samples for natural abundance stable isotope analysis were dried at 55°C for ≥ 48 h and ground to a fine powder using a Wig-L-Bug® dental amalgamator (Crescent Dental Corp., Chicago, IL. USA). For epilithon samples, dried material was removed from filters when possible. Otherwise, entire filters were ground, package in 9×5 mm tin capsules, and combusted. Snail body tissues were excised from their shells to avoid potential carbonate interference. Sample weights were approximately 1.0, 3.0, and 40.0 mg for animals, plants and filters, respectively. Isotopic analyses were carried out at the stable isotope facility at the University of California, Davis using a PDZ Europa 20/20 continuous flow isotope ratio mass spectrometer (PDZ Europa Ltd., Sandbach, United Kingdom). 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 parts per thousand or per mil (‰) deviation from standard reference material (PeeDee belemnite for $\delta^{13}C$ and atmospheric nitrogen for $\delta^{15}N$) according to the following equation:

(Eqn 1)
$$\delta X = \left(\frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right) \times 1000$$

where X=^{13}C or ^{15}N and R =^{13}C:^{12}C or ^{15}N:^{14}N. Under this convention a more positive δ value (or less negative for carbon) is deemed isotopically enriched and indicates that the sample contains more of the heavier isotope (e.g., ^{13}C or ^{15}N). Analysis of replicate (within-run) standards yielded standard deviations of 0.03‰ and 0.15‰ for $\delta^{13}C$ and $\delta^{15}N$, respectively (n = 41 each). Replicate blank pre-ashed GF/F filters were also analyzed for quality control. Blank filters contained no measurable N but contributed small amounts of background carbon. Therefore, epilithon $\delta^{13}C$ values were corrected using a carbon-specific variant of the equation presented in Torn et al. (2003):

(Eqn. 2)
$$\delta^{13}C_{\text{Corrected}} = \frac{\left[\left(\delta^{13}C_{\text{Sample}}\right)\left(\mu g\ C_{\text{Sample}}\right) - \left(\delta^{13}C_{\text{Filter}}\right)\left(\mu g\ C_{\text{Filter}}\right) \right]}{\left[\left(\mu g\ C_{\text{Sample}}\right) - \left(\mu g\ C_{\text{Filter}}\right) \right]}$$

where, $\delta^{13}C_{Sample}$ = uncorrected epilithon $\delta^{13}C$, μg C_{Sample} = the carbon content of the uncorrected sample, $\delta^{13}C_{Filter}$ = $\delta^{13}C$ of the blank filter, and μg C_{Filter} = the carbon content of the blank filter. Mean $\delta^{13}C_{Filter}$ and μg C_{Filter} values were determined to be -26.3‰ and 21.2 μg, respectively ($n = 4$). All isotope samples containing < 100 μg C or < 10 μg N were considered unreliable and excluded from analysis (David Harris, UC Davis Stable Isotope Facility, personal communication).

5.4.3 Data/Analysis

Data were $\log_{10}(x+1)$ or arcsine square-root transformed as appropriate to correct for heteroscedasticity and non-normality. One-way analysis of variance (ANOVA) was used to test for seasonal differences among biotic variables. We set our experiment-wide Type I error rate (α) at 0.05 and significant ANOVAs were followed by Tukey's Honestly Significant Difference (HSD) test. Statistical tests were conducted using NCSS software version 2004 (NCSS, Kaysville, UT, USA) or SAS version 9.1.3 (SAS Institute Inc., Cary, NC, USA).

Trophic relationships within the Big Springs Creek food web were inferred using graphical interpretation (bi-plots) of carbon and nitrogen stable isotopes in conjunction with stoichiometric data (molar C:N) on potential carbon resources. Plant C:N ratios provide insight into 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).

Results are presented for several elements of the study including seston concentration (representing fine particulate organic matter), invertebrate emergence, allochthonous inputs, and stable isotope analysis

Seston (FPOM) Concentration

There was an overall significant seasonal difference in the amount of suspended fine particulate organic matter (seston) in Big Springs Creek (ANOVA, $p = 0.016$; Figure 40). Mean (± 1SE) organic seston concentrations were 1.8 ± 0.5, 3.3 ± 0.3, and 3.1 ± 0.3 mg AFDM·L^{-1} during the spring, summer, and fall sample periods, respectively. However, mean concentrations for the summer and fall did not differ statistically (Tukey's HSD, $p > 0.05$).

In most low-order salmon-bearing streams, organic seston originates primarily as allochthonous material (e.g., leaf litter) which is processed into successively smaller particles through biological and physical breakdown (Cummins 1975, Hynes 1975, Vannote et al. 1980). However, in Big Springs Creek, as is true throughout much of the Shasta River, seston is largely derived from the detrital processing of submerged and emergent aquatic macrophytes (i.e., autochthonous material). In addition to serving as the primary source material of FPOM to the ecosystem, macrophytes reduce water velocity and increase the retention of particulate organic matter (POM). The low hydrologic variability of Big Springs Creek contributes to POM being locally retained until it has been biologically processed into very small, low density particles (Wallace et al. 1982, Webster 1983).

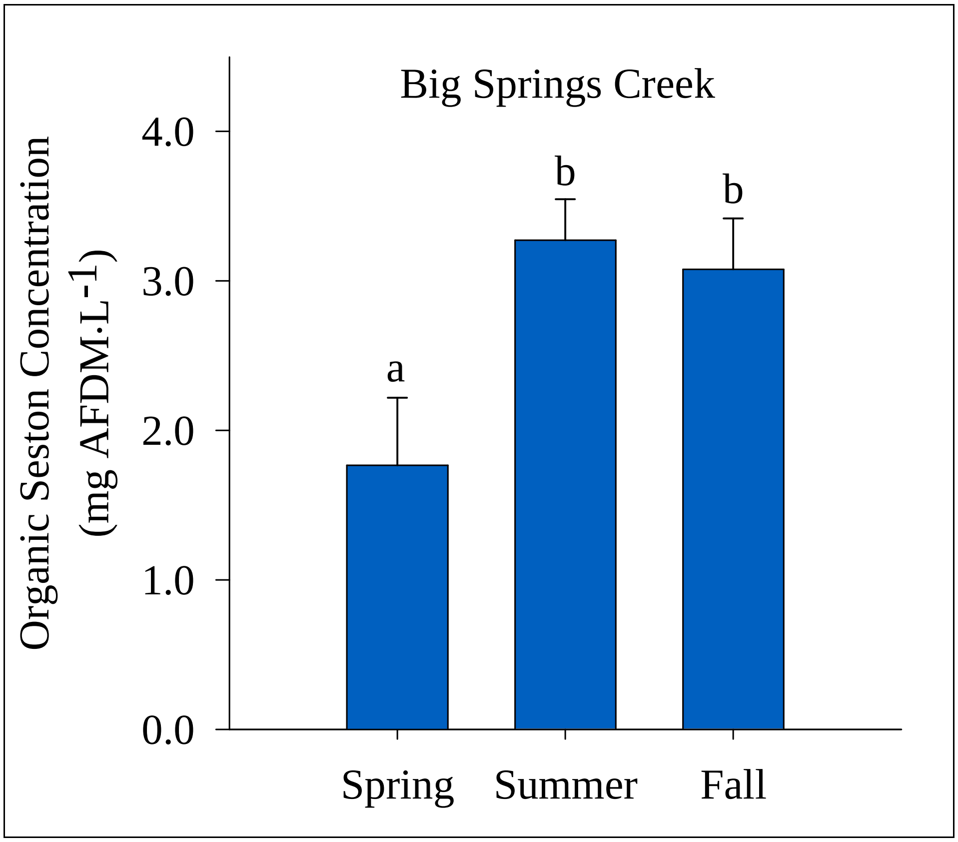


Figure 40. Mean total organic seston (FPOM; particles > 0.45 μm to < 1.0 mm) concentration during each sample period. Bars represent the mean (± 1SE) of 8 replicate water samples. Bars that do not share a common letter are significantly different at α = 0.05.

Our efforts to understand the natural seston dynamics of Big Springs Creek was greatly hindered by cattle grazing, both along the stream banks and in the active channel, which began in early June prior to our summer sampling. The removal of macrophytes and physical disruption of the streambed by cattle resulted in the entrainment of both sediments and organic matter and contributed to the significant increase in seston observed during the summer and fall sample periods. The physical disturbance created by cattle grazing fundamentally alters the organic matter budget of Big Springs Creek and potentially influences the productivity of the entire aquatic food web.

<u>Invertebrate Emergence</u>

Aquatic insect emergence traps were deployed during the summer and fall sample periods. Mean emergent biomass was 204% greater during the fall (mean = 0.07 ± 0.05 g AFDM$\cdot$m$^{-2}\cdot$d^{-1}; n = 6) than the summer (mean = 0.02 ± 0.01 g AFDM$\cdot$m$^{-2}\cdot$d^{-1}; n =5) sample period. Despite this disparity, the high variability among individual traps during the fall (range = 0.003 to 0.24 g AFDM$\cdot$m$^{-2}\cdot$d^{-1}) resulted in non-significant differences between the two sample periods (ANOVA, p = 0.260; Figure 41).

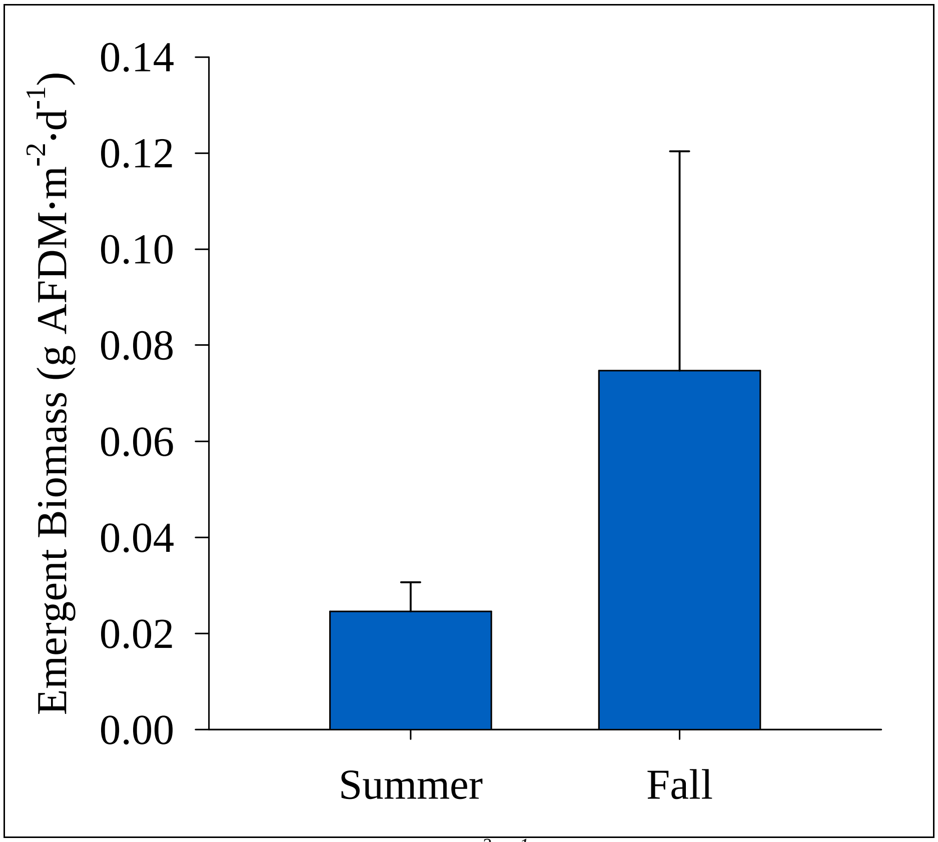


Figure 41. Mean daily biomass (grams AFDM·m^{-2}·d^{-1}) of aquatic insects emerging from Big Spring Creek during the summer and fall. Mean (± 1 SE) biomass estimates were derived from 6 floating emergence traps that were deployed for 5 consecutive days during each season.

<u>Allochthonous Inputs</u>

Invertebrates were the dominant component of the aerial input to Big Springs Creek during both seasons examined (Figure 42). Invertebrate input was significantly higher in the fall averaging 1.2 ± 0.4 g AFDM·m^{-2}·d^{-1}, compared to 0.2 ± 0.1 g AFDM·m^{-2}·d^{-1} during the summer (ANOVA, $p = 0.03$). As a percentage of the total mass being delivered to the creek, invertebrates represented 63% and 92% of the allochthonous input during the summer and fall sample periods, respectively (Figure 42B). These temporal differences were chiefly driven by the capture of many large-bodied adult caddisflies (primarily *Dicosmoecus* sp.) during the fall. Although plant material represented a greater proportion of the total input to Big Springs during the summer period (Figure 42B), the mean AFDM of this material did not differ among seasons (ANOVA, $p = 0.54$; Figure 42A). Plant material contributed 0.06 ± 0.01 g AFDM·m^{-2}·d^{-1} in the summer and 0.04 ± 0.01 g AFDM·m^{-2}·d^{-1} in the fall.

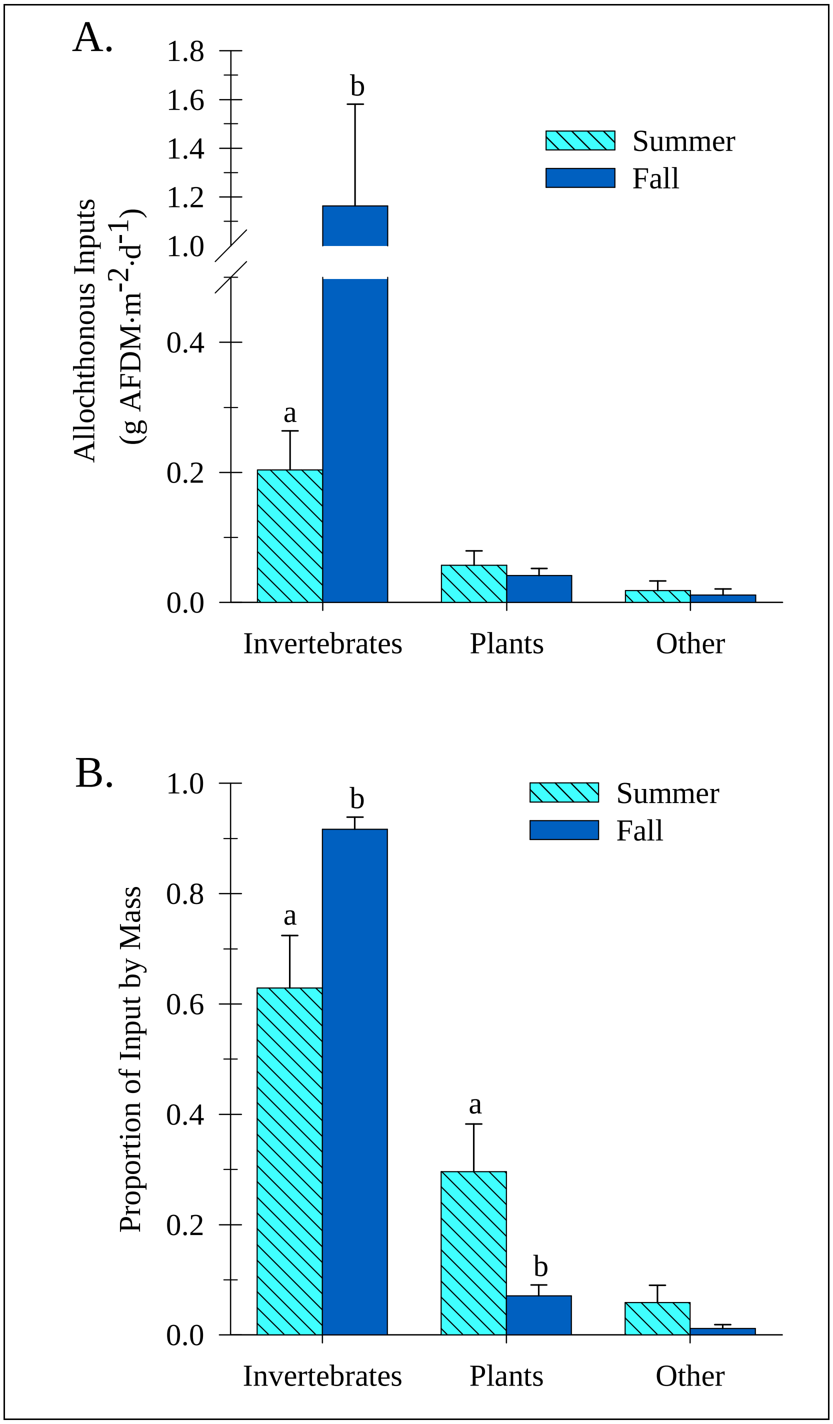


Figure 42. Aerial input of allochthonous materials to Big Springs Creek during the summer and fall sample periods. Data are presented as the mean ± 1 SE input rate (A), and proportion of total input (B) for invertebrates, terrestrial plant matter, and other material. Bars that do not share a common letter are significantly different.

Stable Isotope Analysis

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 basal sources of organic matter to macroinvertebrates to fish). Plots of $\delta^{15}N$ versus $\delta^{13}C$ are presented to 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., (Figure 43, Figure 45, and Figure 48). A key to the numbering convention is provided in Table 7.

Spring

Basal sources of organic matter exhibited a wide range of $\delta^{13}C$ values during the spring period (Table 8). Filamentous algae were the most ^{13}C-depleted source with a mean (±1 SE) $\delta^{13}C$ value of -33.2 ± 0.02‰ (range = -33.3 to -33.1; $n = 5$). Epilithon and the submerged aquatic macrophyte *Polygonum* were the most ^{13}C-enriched of the organic matter sources with mean $\delta^{13}C$ values of -25.0 ± 0.8‰ (range = -27.3 to -22.1; $n = 6$) and -24.5 ± 0.02‰ (range = -24.5 to -24.4; $n = 5$), respectively. Mean $\delta^{13}C$ signatures for seston (-29.9 ± 0.2‰) and detritus (-28.3 ± 0.2‰) were positioned intermediate to the range of values observed for primary carbon sources during spring. The overall mean $\delta^{13}C$ signature across all samples of basal organic matter was 28.1‰ (± 0.6‰; $n = 29$).

Mean $\delta^{15}N$ values of organic matter ranged from 2.2 ± 0.02‰ for filamentous algae to 7.1 ± 0.04‰ for the macrophyte *Myriophyllum* (Table 8). Seston $\delta^{15}N$ values were fairly ^{15}N-depleted (mean = 2.7 ± 0.3‰, range = 2.0 to 3.7; $n = 5$) and showed some degree of overlap with the $\delta^{15}N$ values obtained for both filamentous algae and detritus (mean = 3.3 ± 0.6‰, range = 2.7 to 3.8; $n = 3$). The overall mean $\delta^{15}N$ signature across all basal sources of organic matter was 4.2‰ (± 0.3‰; $n = 28$).

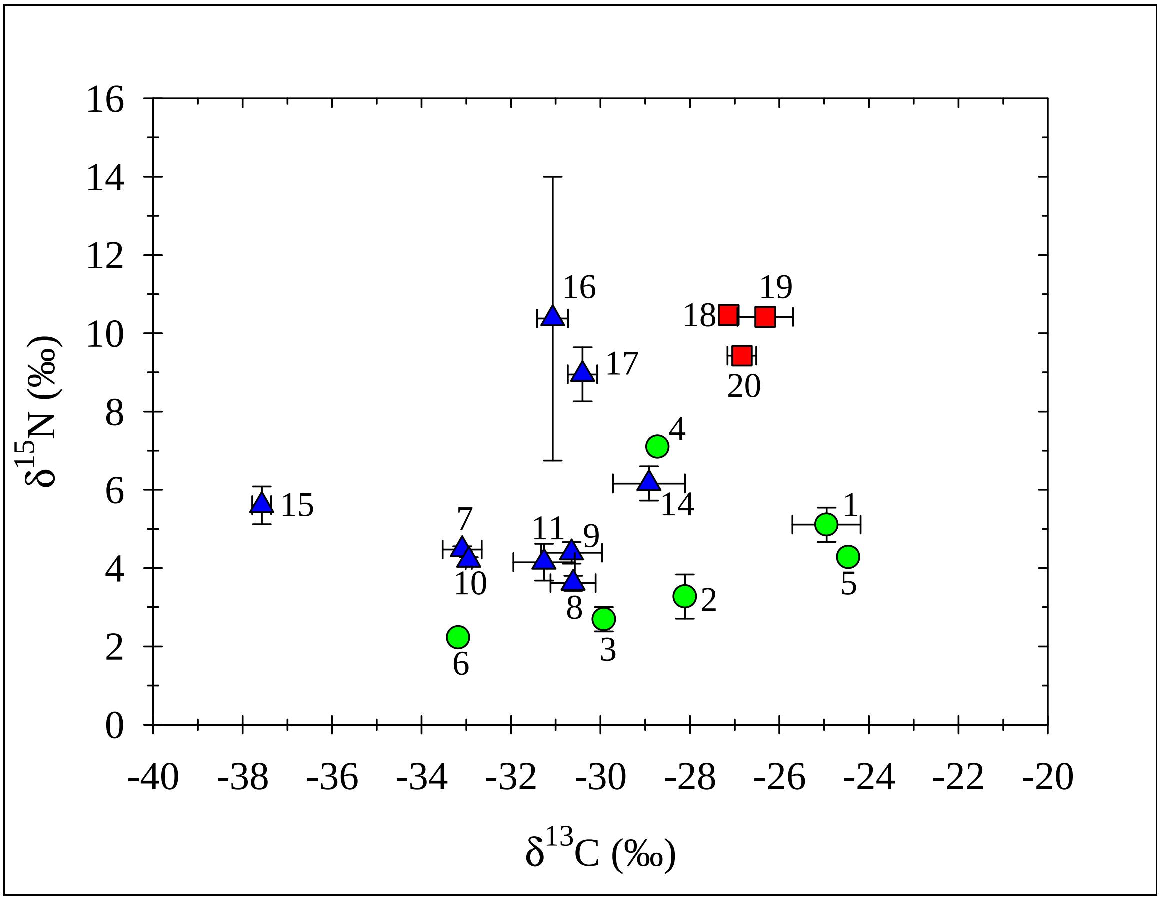


Figure 43. Stable carbon (δ13C) and nitrogen (δ15N) isotope ratios for key members of the Big Springs Creek aquatic food web during the spring of 2008. Circles designate basal carbon resources, triangles represent macroinvertebrate taxa, and squares signify salmonid species. Data are presented as mean values ± 1 standard error. A key to numerical codes is provided in Table 7.

Carbon to nitrogen (C:N) molar ratios of organic matter serve as an indicator of food quality, with high C:N ratios signifying nutritionally poor (refractory) food resources. High C:N ratios can result from extremely recalcitrant starting materials such as 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) reported that macroinvertebrate consumers preferentially selected food resources with C:N ratios below 10, and that the maximum C:N ratio for maintaining the growth of primary consumers was approximately 17. Among the primary organic matter sources analyzed during the spring sample period, epilithon and detritus (CPOM) had the lowest C:N (Figure 44). Mean epilithon C:N was 7.2 ± 0.7, but individual samples were highly variable ranging from 5.1 to 9.2 (n = 6). Detrital C:N was slightly higher averaging 7.6 ± 0.3 (range = 6.9 to 7.9; n = 3). The remainder of the basal sources examined had mean C:N molar ratios between 11.1 (organic seston or FPOM) and 14.5 (*Polygonum amphibium*). All of the organic matter sources analyzed during the spring sample period had mean C:N ratios below 17 and potentially contributed to carbon flow within the Big Springs Creek aquatic food web.

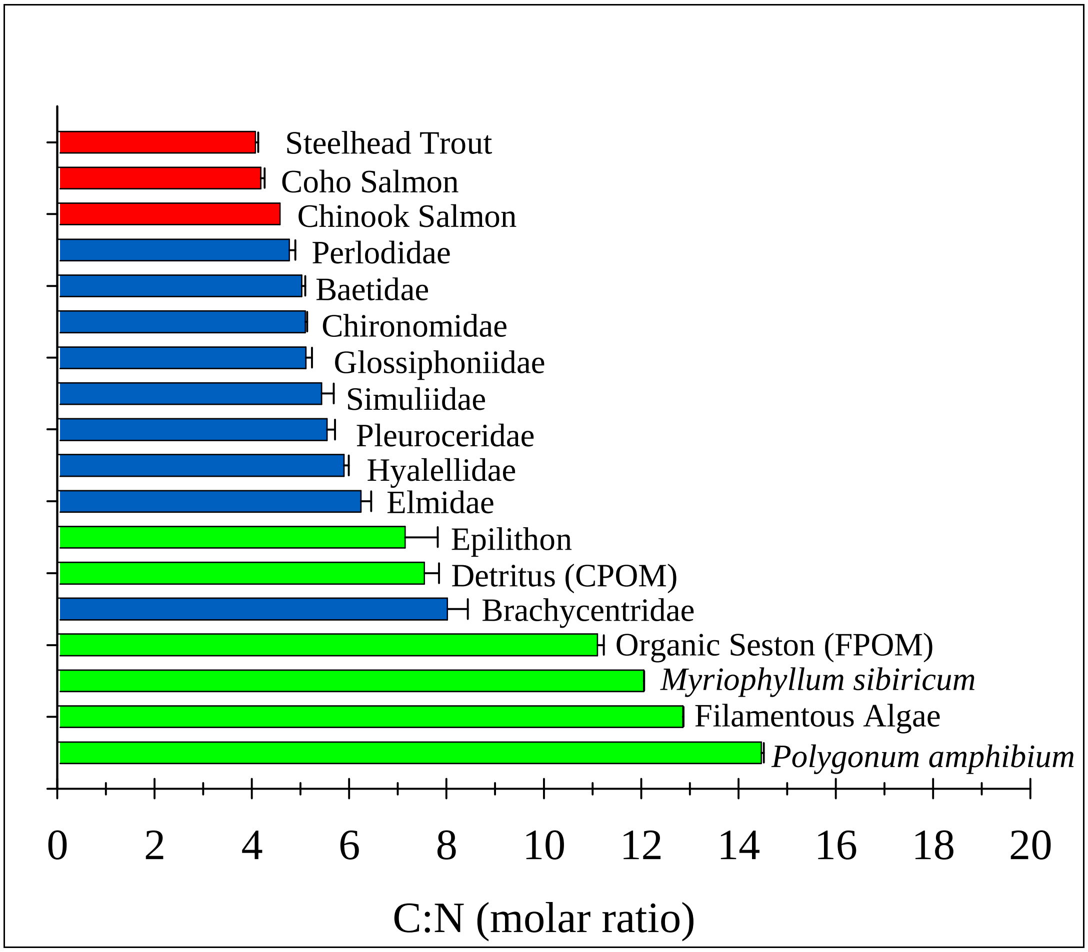


Figure 44. Carbon to nitrogen molar ratios for select food web components during the spring sample period. Green bars indicate basal sources of organic matter, blue bars represent benthic macroinvertebrates, and red bars signify fishes. Bars represent the mean C:N ± 1 standard error.

Mean macroinvertebrate $\delta^{13}C$ values ranged from -37.6 ± 0.2‰ for omnivorous brachycentrid caddisflies (primarily *Brachycentrus* sp. and *Amiocentrus aspilus*) to -28.9 ± 0.8‰ for scraping pleurocerid snails (*Juga* sp.). The enriched $\delta^{13}C$ value of pleurocerids directly reflects their functional feeding role as scrapers and the incorporation of epilithic carbon. Mean $\delta^{13}C$ values for the other macroinvertebrate functional feeding groups were -33.1 ± 0.4‰ (range = -34.8 to -32.5; n = 5) for collector-filterers, -31.5 ± 0.3‰ (range = -33.1 to -28.6; n = 17) for collector-gatherers, -31.1 ± 0.3‰ (range = -31.4 to -30.7; n = 2) for predators, and -30.4 ± 0.3‰ (range = -31.2 to -29.5; n = 5) for parasites.

***Table** 8. 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., $^{13}C/^{12}C$ and $^{15}N/^{14}N$) and are expressed as the per mil (‰) deviation from the standards PeeDee Belemnite and atmospheric N_2 for C and N, respectively. Values for each food web component are presented as the mean ± 1 standard error of the mean. Dashed lines indicate that no data were collected.*

		Taxon	Spring		Summer		Fall	
Food Web Component		Code	δ^{13}C (‰)	δ^{15}N (‰)	δ^{13}C (‰)	δ^{15}N (‰)	δ^{13}C (‰)	δ^{15}N (‰)
Organic Matter								
Epilithon		1	-24.95 ± 0.76	5.11 ± 0.44	-24.80 ± 0.70	5.17 ± 0.25	-23.72 ± 0.61	5.55 ± 0.21
Detritus (CPOM)		2	-28.33 ± 0.22	3.27 ± 0.56	-28.15 ± 0.17	5.59 ± 0.03	-28.97 ± 0.90	5.17 ± 1.12
Seston (FPOM)		3	-29.92 ± 0.15	2.69 ± 0.31	-29.04 ± 0.39	6.72 ± 2.42	-28.96 ± 0.39	2.91 ± 0.94
Macrophytes	*Myriophyllum sibiricum*	4	-28.72 ± 0.01	7.10 ± 0.04	-26.13 ± 0.08	3.50 ± 0.04	-25.45 ± 0.01	4.03 ± 0.02
	Polygonum amphibium	5	-24.46 ± 0.02	4.28 ± 0.03	-20.49 ± 0.01	1.83 ± 0.06	-27.53 ± 0.01	3.46 ± 0.05
Filamentous Algae		6	-33.18 ± 0.02	2.23 ± 0.02	-38.22 ± 0.09	2.36 ± 0.07	---	---
Macroinvertebrates								
Collector-Filterers								
Diptera	Simuliidae (*Simulium*)	7	-33.09 ± 0.44	4.47 ± 0.08	-30.68 ± 0.10	5.31 ± 0.06	---	---
Collector-gatherers								
Coleoptera	Elmidae	8	-30.61 ± 0.51	3.61 ± 0.19	---	---	-28.57 ± 0.27	3.02 ± 0.19
Diptera	Chironomidae	9	-30.65 ± 0.68	4.39 ± 0.27	-31.53	4.96	-31.71 ± 1.42	2.44 ± 2.56
Ephemeroptera	Baetidae	10	-32.94 ± 0.07	4.20 ± 0.08	-30.89 ± 0.30	5.24 ± 0.29	-30.20 ± 0.28	---
Amphipoda	Hyalellidae (*Hyalella*)	11	-31.26 ± 0.69	4.15 ± 0.47	-31.01 ± 0.24	4.29 ± 0.09	-26.57 ± 0.10	4.05 ± 0.16
Oligochaeta		12	---	---	-31.41 ± 0.24	6.09 ± 0.52	-28.30	4.08
Scrapers								
Ephemeroptera	Heptageniidae	13	---	---	-31.52 ± 0.67	5.66 ± 1.12	---	---
Gastropoda	Pleuroceridae (*Juga* sp.)	14	-28.92 ± 0.81	6.16 ± 0.44	-29.09 ± 0.24	6.54 ± 0.03	-25.04 ± 1.43	7.00 ± 0.24
Omnivores								
Trichoptera	Brachycentridae	15	-37.57 ± 0.21	5.60 ± 0.48	-34.76 ± 0.12	4.48 ± 0.11	-32.77 ± 0.11	5.41
Predators								
Plecoptera	Perlodidae	16	-31.06 ± 0.35	10.37 ± 3.63	-30.71 ± 0.26	7.40 ± 0.49	---	---
Parasites								
Hirudinea	Glossiphoniidae (*Helodbella*)	17	-30.40 ± 0.33	8.95 ± 0.69	-30.21 ± 0.05	7.70 ± 0.03	-28.65 ± 0.37	8.18 ± 0.07
Fishes								
Chinook Salmon		18	-27.13	10.47	---	---	---	---
Coho Salmon		19	-26.32 ± 0.62	10.42 ± 0.25	-30.31 ± 0.62	9.76 ± 0.38	---	---
Steelhead Trout		20	-26.84 ± 0.32	9.43 ± 0.18	---	---	---	---

Macroinvertebrate $\delta^{15}N$ values were highly variable among individual taxa. Larval riffle beetles (Coleoptera: Elmidae) were the most ^{15}N-depleted invertebrate (mean $\delta^{15}N = 3.6 \pm 0.2‰$, range = 3.2 to 4.1; $n = 4$) during the spring. Conversely, leeches (Hirudinea: Glossiphoniidae) and predatory stoneflies (Plecoptera: Perlodidae) were the most ^{15}N-enriched invertebrates with mean $\delta^{15}N$ signatures of $9.0 \pm 0.7‰$ and $10.4 \pm 3.6‰$, respectively (Table 8). Although leeches are principally classified as temporary ectoparasites of fish, amphibians, and waterfowl (Davies 1991, ABL 2003), members of the family Glossiphoniidae have also been reported to frequently prey on other aquatic macroinvertebrates. The position of leeches in food web space (i.e., the $\delta^{13}C$ versus $\delta^{15}N$ bi-plot, Figure 43) indicates they were largely consuming benthic macroinvertebrates during the spring and effectively functioning as the dominant invertebrate predator in the Big Springs Creek food web.

The community-wide mean $\delta^{15}N$ value of primary consumers was $4.8 \pm 0.2‰$ (range = 3.2 to 8.0; $n = 35$) an enrichment of only +0.6‰ over the mean $\delta^{15}N$ for organic matter sampled during this same period. Given the expected isotopic fractionation with each trophic transfer, the dominant (numerically) macroinvertebrate consumers in Big Springs Creek appeared to be deriving their carbon from fine particulate organic matter (seston) and filamentous algae. While macrophytes were present during the spring (>25.0 g AFDM·m^{-2}; see section 9.2.2) their enriched stable isotope ratios (specifically, $\delta^{15}N$ for *Mriophyllum* and $\delta^{13}C$ for *Polygonum*; Figure 43) suggest 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 stream food webs by serving as substrata for epiphytic biofilm or as refugia from predators (France 1995). *Myriophyllum*, in particular, may be of limited use as a carbon source to the Big Springs Creek food web. Members of the genus *Myriophyllum* (Haloragaceae) have been reported to contain and release allelochemicals that target epiphytes, cyanobacteria, and invertebrate herbivores (Gross 2003). While the allelopathic effects of *Myriophyllum sibericum* are not well documented (Linden and Lehtiniemi 2005), they have the potential to alter community composition as well as flows of energy and material within the ecosystem.

An especially notable finding during our spring stable isotope sampling was a larval perlodid stonefly with an extremely elevated $\delta^{15}N$ signature (+14.0‰). We are not aware of any native source of nitrogen in the Big Springs Creek drainage that is sufficiently ^{15}N-enriched to produce such an elevated signal in biota. We hypothesize that this enriched $\delta^{15}N$ value stems from the incorporation of marine-derived inputs vectored to Big Springs Creek by spawning anadromous salmon. A biogenic consequence of feeding in the marine environment is that anadromous salmonids are uniquely enriched with the heavier isotopic forms of many elements (e.g., C, N and S) relative to terrestrial or freshwater sources of these same elements. These marine-derived nutrients are ultimately liberated to freshwater ecosystems through the excretion of metabolic waste products, deposition of gametes, and decomposition of post-spawning carcass (Cederholm et al. 1999, Naiman et al. 2002). Kiernan (2009) reported that adult coho salmon in a coastal

California river had mean $\delta^{13}C$ and $\delta^{15}N$ values of -16.9 ± 0.2‰ (range = -17.3 to -15.7‰) and 15.1 ± 0.1‰ (range = 14.5 to 15.7‰), respectively. Many aquatic macroinvertebrate taxa (Piorkorwski 1995, Kline et al. 1997, Minakawa 1997, Wipfli et al. 1998, Minakawa and Gara 1999) have been reported to readily scavenge and ingest salmon carcasses, gametes, or dead fry when these items are present and empirical studies have shown that incorporation of such materials by consumers leads to significant isotopic enrichment of their body tissues (Bilby et al. 1996, Chaloner et al. 2002, Hicks et al. 2005, Kiernan 2009). While adult anadromous salmon were not directly observed in Big Springs Creek prior to our initial sampling (due to site access), the occurrence of this highly enriched stable isotope value suggests that spawning activity occurred in the creek during the months preceding our sampling.

We determined stable isotope ratios for 13 juvenile coho salmon (mean FL = 47 mm), 8 juvenile steelhead trout (mean FL = 47 mm), and a single juvenile Chinook salmon (FL = 53 mm) during the spring sample period. All salmonid species had mean $\delta^{13}C$ values that were within 1.0‰ of each other (Table 8). Coho salmon were the most ^{13}C-enriched of the salmonids (mean $\delta^{13}C$ = -26.3‰) while the lone juvenile Chinook was the most ^{13}C-depleted ($\delta^{13}C$ = -27.1‰). With respect to nitrogen, juvenile Chinook and coho salmon had nearly indistinguishable $\delta^{15}N$ values of 10.5‰ and 10.4 ± 0.3‰, respectively. Juvenile steelhead were ^{15}N-depleted by approximately 1.0‰ (mean $\delta^{15}N$ = 9.4 ± 0.2‰) relative to the other two salmonid species (Figure 43; Table 8).

It is difficult to interpret our salmonid stable isotope results within the context of the rest of the Big Springs Creek food web. We found significant relationships between fish size (fork length) and stable isotope ratios, with smaller (and presumably younger) fish exhibiting enriched $\delta^{13}C$ and $\delta^{15}N$ values. These results indicate the presence of marine-derived nutrients in juvenile fish, specifically in the form of residual maternal yolk. Following emergence from the gravels, the C and N isotope ratios of juvenile salmon systematically decline as they deplete their maternal yolk and begin to feed exogenously (Doucett et al. 1996). However, the time required for young salmon to reach isotopic equilibrium with their riverine diet is highly variable and remains very poorly understood. Power and Finlay (2001) reported that juvenile steelhead in the South Fork Eel River watershed maintained a maternal (marine) signal until they reached standard lengths >50 mm. Our data for Big Springs Creek show a significant negative relationship between salmonid fork length and muscle stable isotope ratios for fish with fork lengths up to 55 mm (Figure 45). Unfortunately, we lack equivalent data for fish between 55 and 75 mm, and are unable to predict the size at which juvenile $\delta^{13}C$ and $\delta^{15}N$ values begin to accurately reflect their freshwater diets. While such isotopic enrichment in juvenile fish is ultimately transient, it greatly obscures the interpretation of both diet and trophic position. Consequently, efforts to understand trophic linkages in salmonid food webs must recognize that the presence and assimilation of marine-derived nutrients and biomass, be it in the form of dissolved nutrients, gametes, carcass material, or residual maternal yolk, can alter the stable isotope ratios of biota at all trophic levels. Presently, the extent to which marine-derived nutrient subsidies influence food web structure and salmonid productivity in the greater Shasta River basin is unknown and warrants additional investigation.

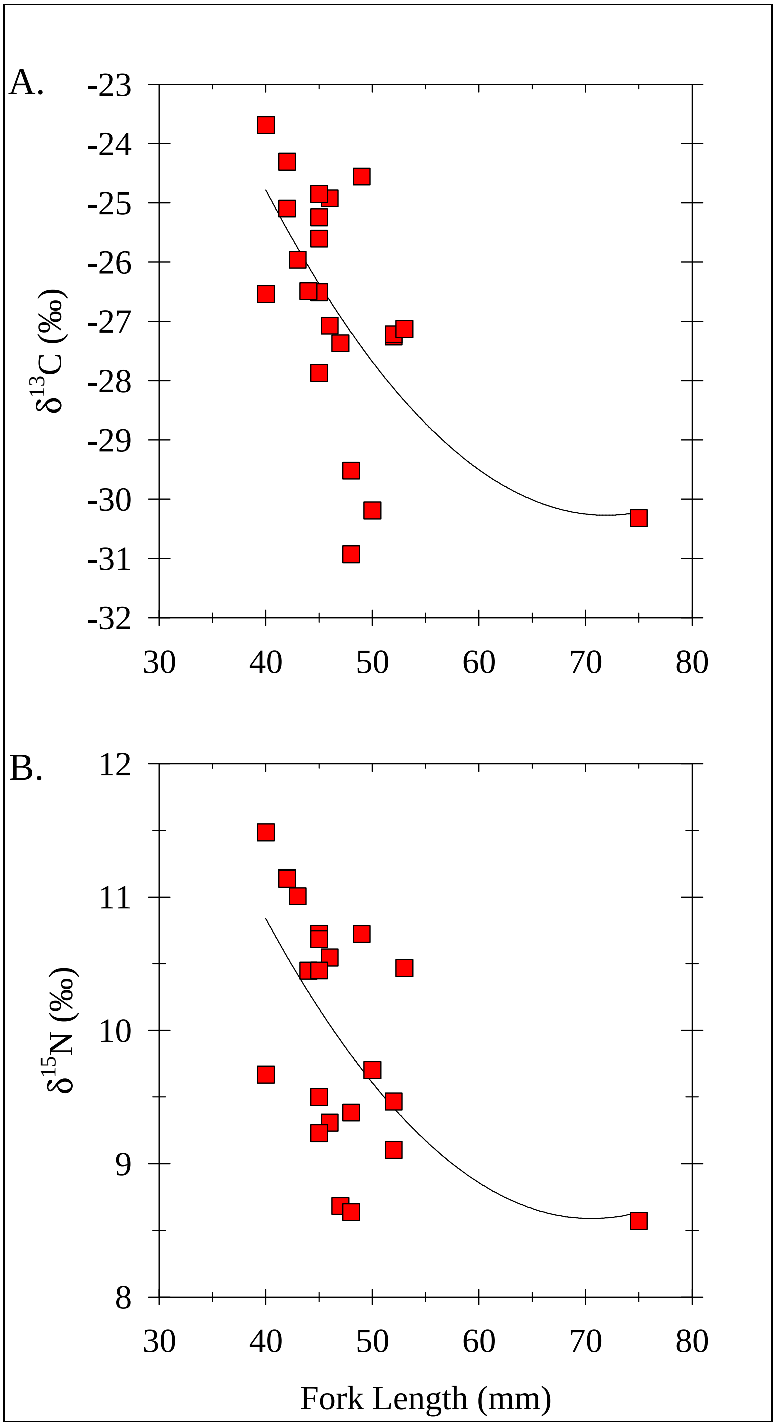


Figure 45. Relationship between juvenile salmonid fork length (mm) and muscle tissue stable carbon (A) and nitrogen (B) isotope ratios. Plots illustrate that smaller, and presumably younger, salmonids exhibit an isotopically enriched maternal (marine) signal that dissipates as they grow and begin to equilibrate with their freshwater diets.

Summer

Basal sources of organic matter demonstrated considerable isotopic separation during the summer sample period. Nearly all sources analyzed were more ^{13}C-enriched relative to mean $\delta^{13}C$ values obtained during the spring sample period (Figure 46; Table 8). Filamentous algae were again the most ^{13}C-depleted of the basal carbon resources with a mean $\delta^{13}C$ value of -38.2 ± 0.1‰ (range = -38.5 to -38.0; *n* =5). This mean value represented an isotopic shift of more than -5.0‰ from the previous sample period (Figure 46). Summer samples of the macrophyte *Polygonum* produced the most ^{13}C-enriched carbon measurements obtained in our entire study. *Polygonum* $\delta^{13}C$ averaged -20.5 ± 0.01‰ during the summer and individual plants demonstrated surprisingly little variability (range = -20.50 to -20.46; *n* =5). Excluding *Polygonum*, the overall mean $\delta^{13}C$ of all basal carbon sources during the summer period was -28.7 ± 0.9‰ (range = -38.5 to -22.1; *n* = 29).

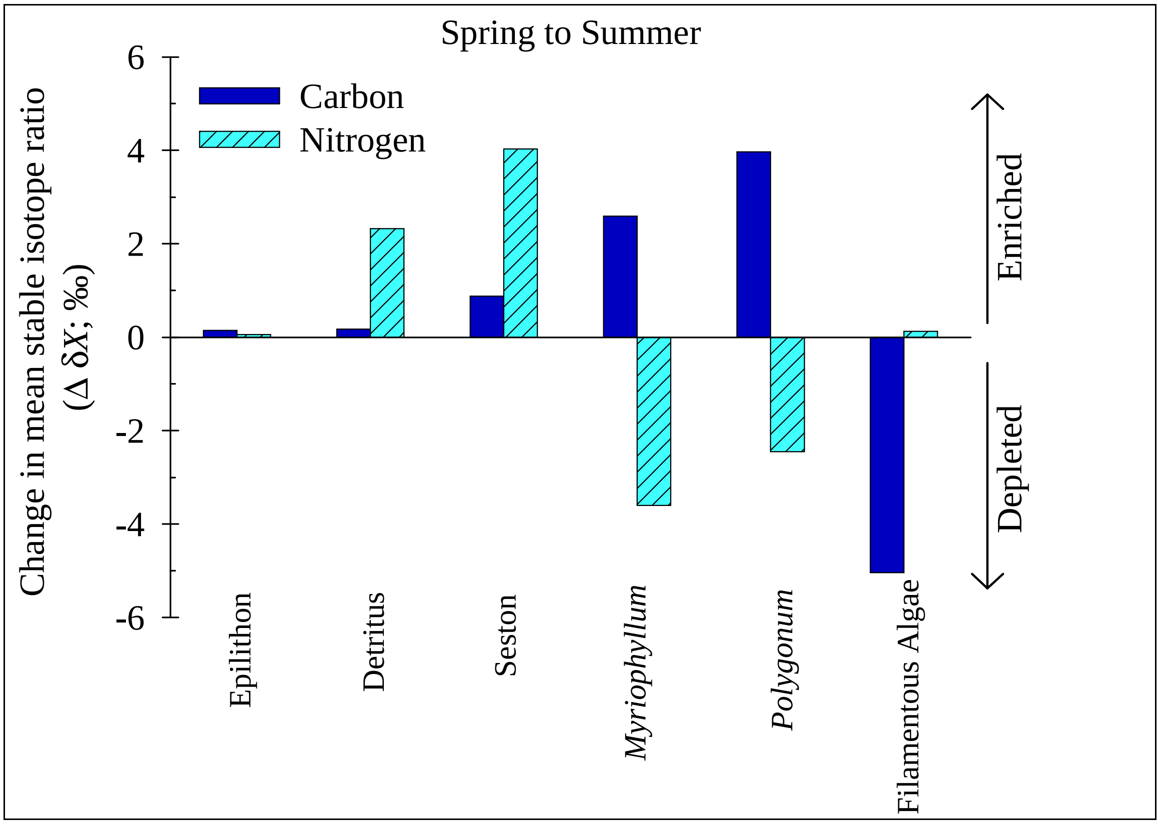


Figure 46. Per mil (‰) change in the mean stable carbon (δ13C) and nitrogen (δ15N) isotope ratios for basal sources of organic matter between the spring and summer sample periods.

As a group, freshwater autotrophs can demonstrate extremely variable $\delta^{13}C$ values ranging from -50 to -10‰ (Boutton 1991). Although the fundamental reasons for this variability are not well understood, they likely stem from the source (HCO_3^- or CO_2), concentration and isotopic composition of the dissolved inorganic carbon (DIC) in the water column (Keeley and Sandquist 1992). Additionally, hydrologic parameters such as water velocity have been shown to strongly influence epilithon and macrophyte $\delta^{13}C$

values (Finlay et al. 1999, Trudeau and Rasmussen 2003). Under low velocity (i.e., low turbulence) conditions, boundary layers are thicker and rates of CO_2 and HCO_3^- diffusion are reduced (MacLeod and Barton 1998). This results in reduced discrimination against the heavier ^{13}C and more enriched $\delta^{13}C$ values (Osmond et al. 1981, Trudeau and Rasmussen 2003). Other environmental variables such as temperature and light intensity have also been shown to affect carbon isotopic fractionation through changes in metabolic activity (MacLeod and Barton 1998). Finally, $\delta^{13}C$ may be influenced by community composition, plant growth rates (Laws et al. 1995) and biomass (Singer et al. 2005, Hill and Middleton 2006). Given the broad range of mean $\delta^{13}C$ values observed for primary producers in our study (from -38.2‰ for filamentous algae to -20.5‰ for *Polygonum amphibium*), an improved understanding of carbon cycling will require detailed sampling of dissolved inorganic carbon isotope values ($\delta^{13}C$ of Σ DIC), and how these values influence the $\delta^{13}C$ of autotrophic organisms and other carbon sources at the base of the food web.

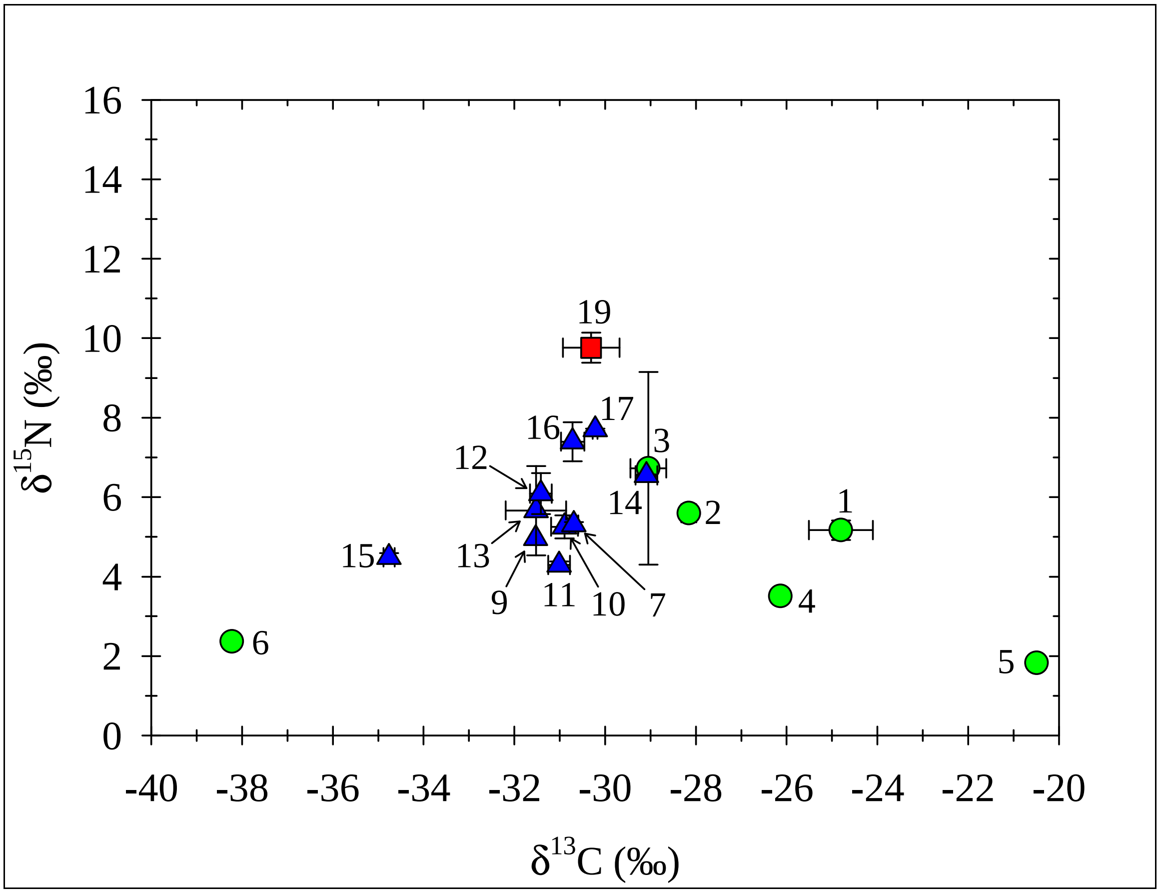


Figure 47. Stable carbon (δ13C) and nitrogen (δ15N) isotope ratios for key members of the Big Springs Creek aquatic food web during summer of 2008. Circles designate basal carbon resources, triangles represent macroinvertebrate taxa, and the square indicates juvenile coho salmon. Data are presented as mean values ± 1 standard error. A key to numerical codes is provided in Table 7.

Basal organic matter sources clustered into two distinct groups with respect to mean $\delta^{15}N$ values. *Polygonum* (1.8 ± 0.1‰), filamentous algae (2.4 ± 0.1‰), and *Mriophyllum* (3.5 ± 0.04‰) were all significantly ^{15}N-depleted relative to mean values obtained for epilithon (5.2 ± 0.3‰), detritus (5.6 ± 0.03‰) and seston (6.7 ± 2.4‰) (Figure 47; Table 8). Mean seston $\delta^{15}N$ was surprisingly enriched and increased by more than +4.0‰ compared to samples collected during the spring (Figure 46; Table 8). It should be noted, however, that the mean $\delta^{15}N$ value derived for seston was influenced by one extraordinarily enriched sample with a $\delta^{15}N$ value of 16.2‰. While this observation represented a statistical outlier, we opted to retain it in our data set given the dearth of information that currently exists regarding seston and FPOM dynamics in Big Springs Creek. However, with this point excluded, mean seston $\delta^{15}N$ decreased from 6.7‰ to 4.4 ± 0.6‰ (n = 4), an enrichment of +1.7‰ relative to seston in the spring. Contrary to the general trend of organic matter ^{15}N-enrichment between the spring and summer sample periods, both submerged macrophyte species were found to be ^{15}N-depleted (Figure 46). *Myriophyllum*, the dominant plant in terms of standing crop during the summer period (see Figure 29), was ^{15}N-depleted by 3.6‰ and *Polygonum* was depleted by 2.5‰ relative to values obtained for the same taxa during the spring (Table 8).

Carbon to nitrogen ratios were again generally low for the various basal organic matter sources, ranging from 8.1 ± 0.5 (range = 4.9 to 9.6; n =9) for epilithon to 20.8 ± 1.6 (range = 16.9 to 23.5; n =5) for detritus (Figure 48). The elevated C:N of detritus during the summer sample period was due to the increased occurrence of terrestrially-derived materials (predominantly grasses and some leaf litter) in the conditioned detrital pool.

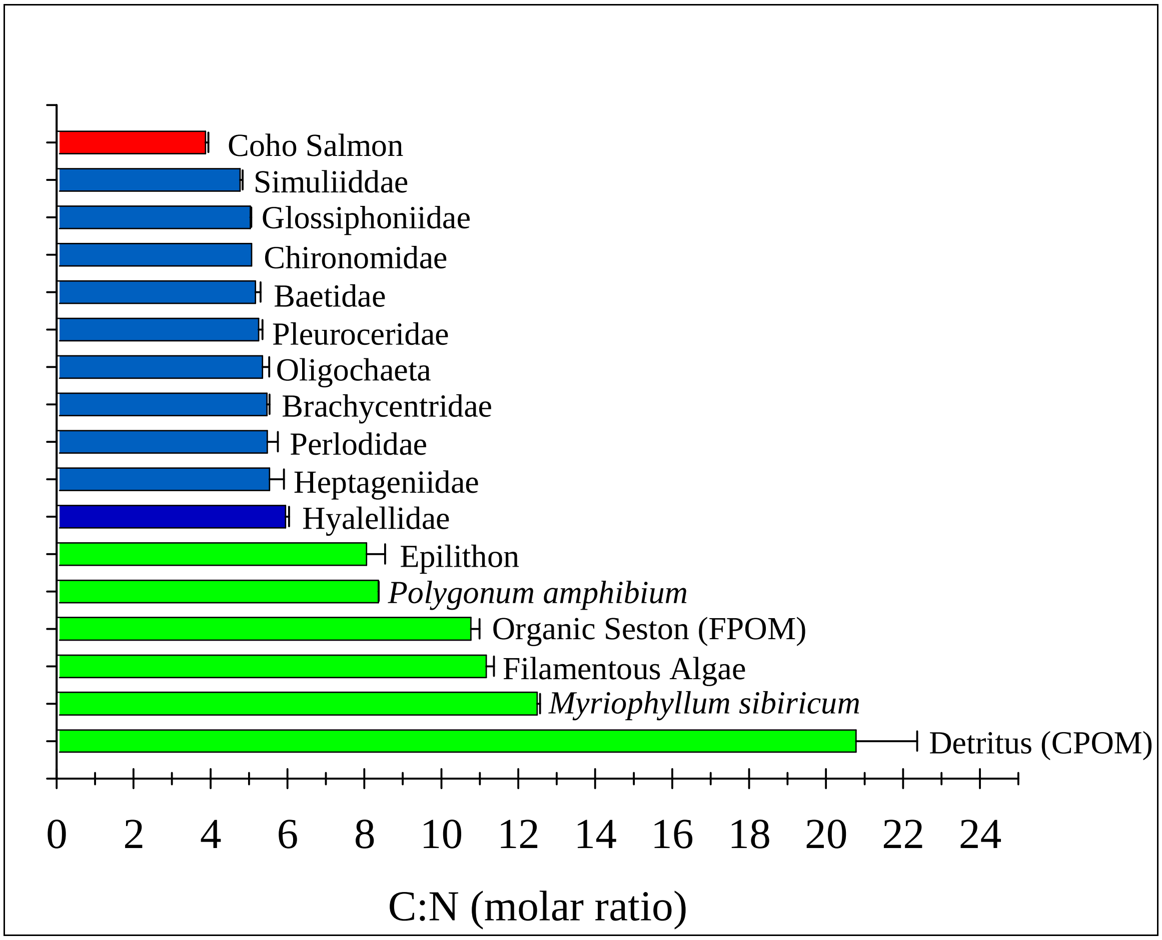


Figure 48. Carbon to nitrogen molar ratios for select food web components during the summer sample period. Green bars indicate basal sources of organic matter, blue bars represent benthic macroinvertebrates, and red bars signify fishes. Bars represent the mean C:N ±1 standard error.

There was surprisingly little variability in $\delta^{13}C$ among macroinvertebrate taxa during the summer sample period. Only omnivorous brachycentrids exhibited a relatively distinct (depleted) mean $\delta^{13}C$ value of -34.8 ± 0.1‰ (range = -35.1 to -35.4; *n* = 5). All other macroinvertebrate taxa had mean $\delta^{13}C$ signatures that fell between approximately -29.0 and -31.5‰ (Figure 47, Table 8). Mean $\delta^{13}C$ values for the major functional feeding groups were -31.1 ± 0.1‰ (range = -32.1 to -29.9; *n* = 16) for collector-gatherers, -30.7 ±0.3‰ (range = -31.5 to -30.0; *n* = 5) for invertebrate predators, -30.7 ± 0.1‰ (range = -31.0 to -30.4; *n* = 5) for collector-filterers, -30.6 ± 0.7 (range = -32.6 to -28.9; *n* = 5) for scrapers, and -30.2 ± 0.1‰ (range = -30.4 to -30.1; *n* = 5) for parasites.

In contrast to carbon, macroinvertebrate $\delta^{15}N$ values were highly variable during the summer sample period. Amphipods were the numerically dominant consumer taxon and exhibited the most ^{15}N-depleted isotope signature (mean $\delta^{15}N$ = 4.3 ± 0.1‰, range = 4.0 to 4.5; *n* = 5). Predictably, invertebrate predators (i.e., perlodid stoneflies) and parasites (i.e., leeches) were again the two most ^{15}N-enriched feeding guilds with mean $\delta^{15}N$ values of 7.4 ± 0.5‰ (range = 5.5 to 8.1; *n* = 5) and 7.7 ±0.03‰ (range = 7.6 to 7.8; *n* =

5), respectively (Table 8). Mean $\delta^{15}N$ values for the other functional feeding groups were 4.5 ± 0.1‰ (range = 4.3 to 4.9; *n* = 5) for omnivores, 5.2 ± 0.3‰ (range = 4.0 to 7.5; *n* = 16) for collector-gatherers, 5.3 ± 0.1‰ (range = 5.2 to 5.5; *n* = 5) for collector-filterers, and 6.0 ± 0.7‰ (range = 4.3 to 7.9; *n* = 5) for scrapers.

As mentioned previously, only two juvenile coho salmon were available for analysis during the summer sample period. Mean coho $\delta^{13}C$ and $\delta^{15}N$ signatures were -30.3‰ (individual values = -30.9 and-29.7) and 9.8‰ (values = 9.4 and 10.1), respectively. In contrast to juvenile coho salmon analyzed during the spring period, theses mean values were depleted by -4.0‰ for ^{13}C and -0.7‰ for ^{15}N. This directional shift, coupled with their position in food web space (see taxon 19 in Figure 47) indicates that the tissues of juvenile fish were in isotopic equilibrium with their diets during this sample period.

Fall

There was no clear pattern of isotopic enrichment or depletion among organic matter sources between the summer and fall sample periods (Figure 49). Epilithon exhibited the most depleted mean $\delta^{13}C$ (-23.7 ± 0.6‰) and most enriched mean $\delta^{15}N$ value (5.6 ± 0.2%) among the basal resources (Table 8). Mean $\delta^{13}C$ signatures of detritus and seston were nearly identical at -28.97‰ (± 0.9‰; *n* = 4) and -28.96‰ (± 0.4‰; *n* = 5), respectively. However, mean $\delta^{15}N$ values for these two resources differed by more than 2.2‰ with detritus (mean $\delta^{15}N$ = 5.1 ± 1.1‰) being significantly more enriched than seston (mean $\delta^{15}N$ = 2.9 ± 0.9‰). Curiously, the macrophyte *Polygonum* was ^{13}C-depleted by more than 7.0‰ relative to the mean value derived for this same taxon during the summer (Figure 49; Table 8). No other organic matter source exhibited a temporal shift in $\delta^{13}C$ of greater than ± 1.1‰ between the summer and fall periods (Figure 49).

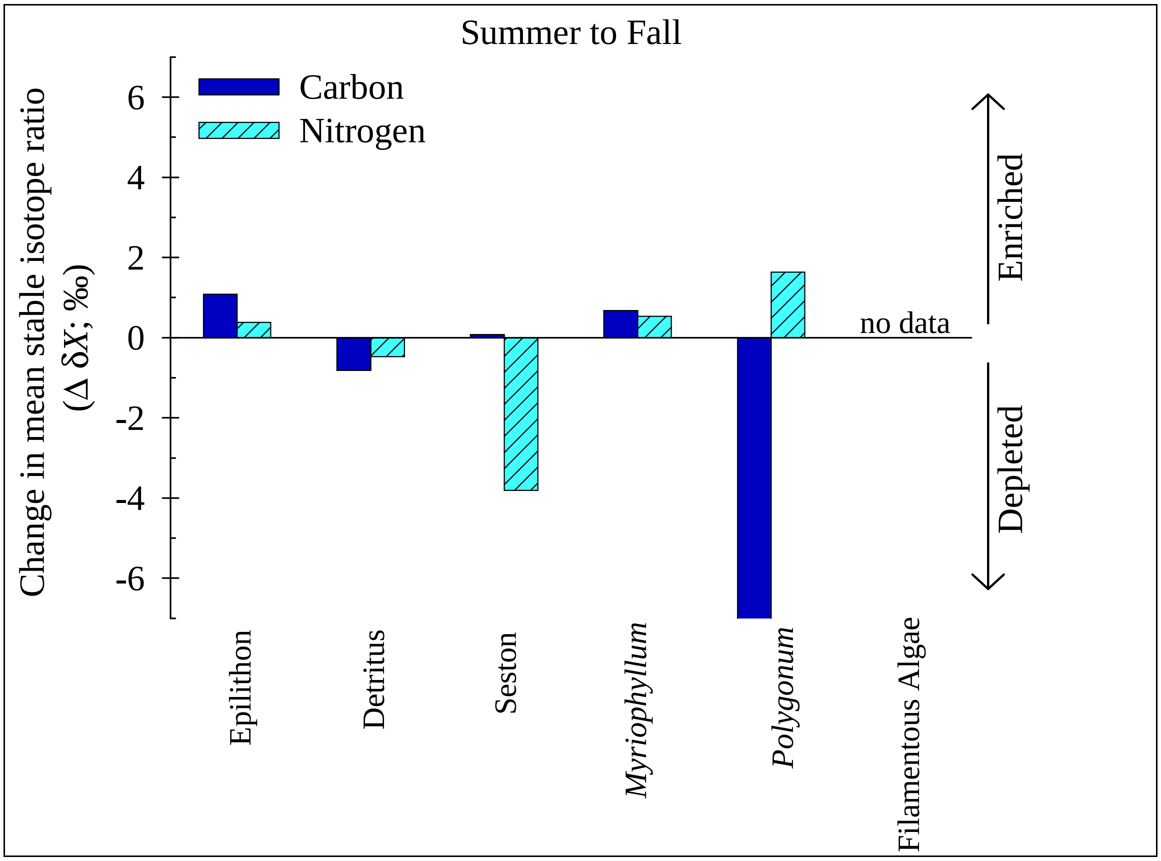


Figure 49. Per mil (‰) change in the mean stable carbon (δ13C) and nitrogen (δ15N) isotope ratios for basal sources of organic matter between the summer and fall sample periods.

As discussed in the macroinvertebrate results presented earlier (see section 5.3), the Big Springs Creek invertebrate assemblage was notably depauperate during the fall sample period. The community was dominated, both numerically and in terms of biomass, by amphipods (*Hyalella sp.*) and other collector-gatherers, while collector-filterers and predatory invertebrates were rare or absent (Table 8). Pleurocerid snails (*Juga* sp.) were the most ^{13}C-enriched invertebrate taxa during the fall with a mean δ^{13}C value of -25.0 ± 1.4‰ (range = -27.9 to -20.5; *n* = 5) reflecting their incorporation of epilithic carbon. *Hyalella* were the next most ^{13}C-enriched taxon with a mean δ^{13}C signature of -26.6 ± 0.1‰ (range = -26.8 to -26.2; *n* = 5). This carbon signature was notably ^{13}C-enriched (> 4.0‰) when compared to mean values obtained for conspecifics during the previous sample periods (i.e., -31.3‰ in spring and -31.0‰ in summer; Table 8). Since few of the organic matter sources analyzed were appreciably ^{13}C-enrichment between summer and fall (see Figure 50), it is unclear what contributed to the marked change in *Hyalella* δ^{13}C.

As was the case during the spring and summer sample periods, brachycentrids were the most ^{13}C-depleted of the invertebrate taxa analyzed. However, the mean $\delta^{13}C$ signature of brachycentrids (-32.8 ± 0.11%; *n* = 5) was far less isolated in food web space during the fall and shifted (via ^{13}C-enrichment) toward the balance of the macroinvertebrate community. The $\delta^{13}C$ values of larval chironomids were especially variable with individual observations ranging between -32.1‰ and -29.3‰ (mean = -30.7‰). As a group, the mean $\delta^{13}C$ value of collector-gatherers was -28.5 ± 0.4‰ (range = -32.1 to -26.2; *n* = 16) and the mean signature for all primary consumers during the fall sample period was -28.7 ± 0.6‰ (range = -33.1 to -20.5; *n* = 26).

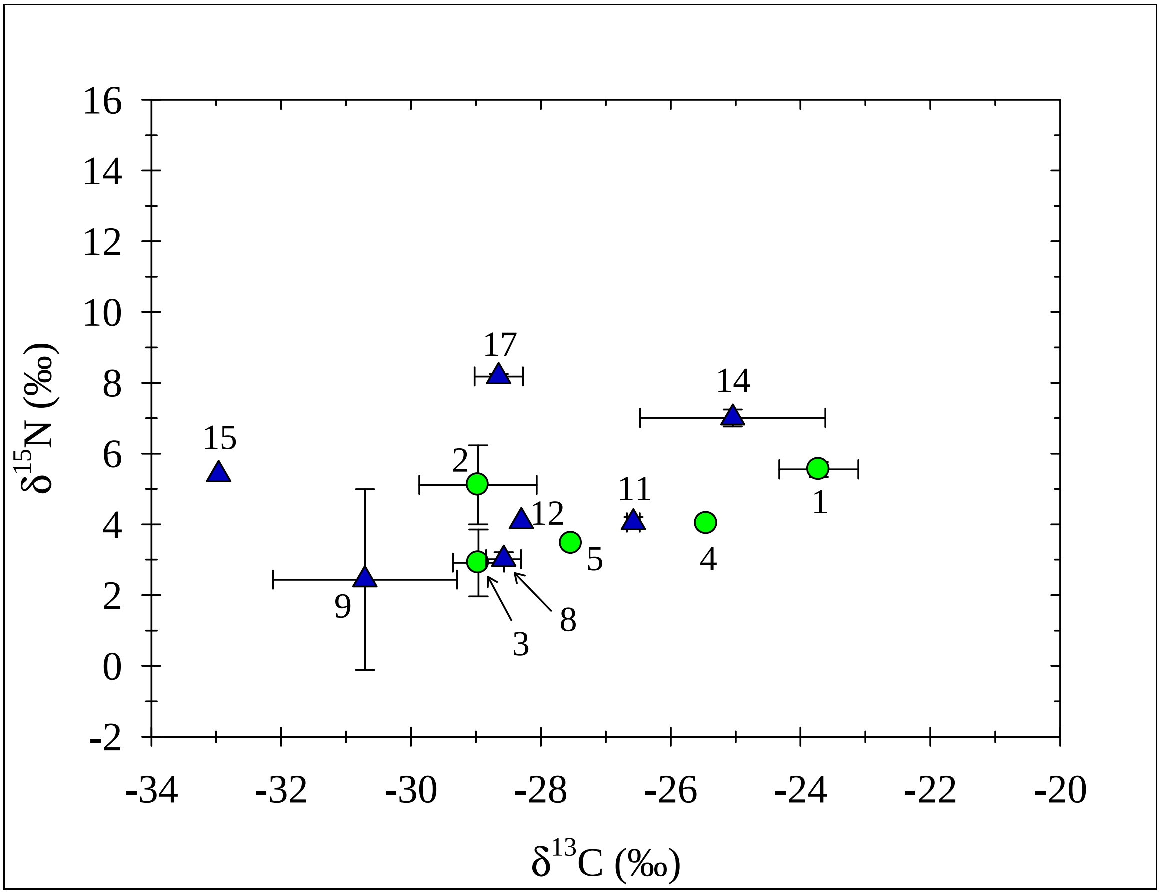


Figure 50. Stable carbon (δ13C) and nitrogen (δ15N) isotope ratios for key members of the Big Springs Creek aquatic food web during the fall of 2008. Circles designate basal carbon resources and triangles represent macroinvertebrate taxa. Data are presented as mean values ± 1 standard error. A key to numerical codes is provided in Table FW4.

Leeches had the highest $\delta^{15}N$ value among the fall macroinvertebrates averaging 8.2 ± 0.1‰ (range = 8.0 to 8.4; *n* = 5). Pleurocerid snails had a mean $\delta^{15}N$ signature of 7.0 ± 0.2‰ (range = 6.3 to 7.6; *n* = 5) indicating a trophic fractionation factor of approximately 1.5‰ between the snails and their primary food source (i.e., epilithon; $\delta^{15}N$ = 5.6‰). There was considerable variability in $\delta^{15}N$ signatures among members of the collector-gatherer feeding guild during the fall. Larval chironomids and riffle beetles were the most ^{15}N-depleted collector-gatherers with mean $\delta^{15}N$ values of 2.4 ± 2.6‰ (range = -0.1 to 5.0; *n* =2) and 3.0 ± 0.2‰ (range = 2.5 to 3.5; *n* =5), respectively. The numerically dominant taxon, *Hyalella sp.*, exhibited a slightly more enriched mean $\delta^{15}N$ value of 4.1 ± 0.2‰ (range = 3.6 to 4.5; *n* =5).

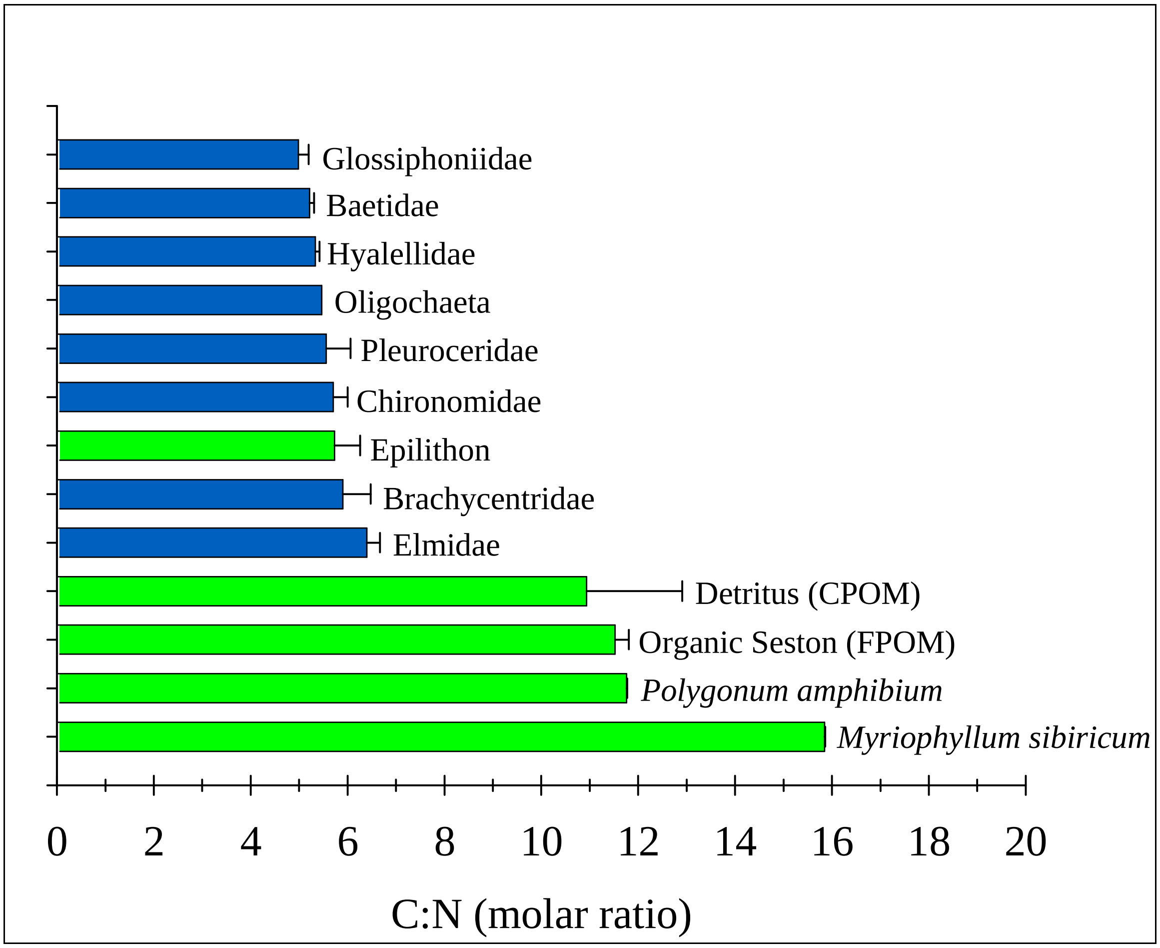


Figure 51. Mean (± 1 SE) carbon to nitrogen molar ratio for select food web components during the fall sample period. Green bars represent basal sources of organic matter and blue bars indicate benthic macroinvertebrate taxa.

Epilithon obtained its lowest C:N during the fall sample period with a mean molar ratio of 5.7 ± 0.5 (range = 3.8 to 8.4; *n* = 9). This C:N closely matched the stoichiometry of all primary consumers during this period (Figure 51). Mean C:N for conditioned detritus, seston, and the macrophyte *Polygonum* were similar ranging between 11 and 12 (Figure 48).

5.5 *Conclusions*

Standing crops of both epilithon and aquatic plants in Big Springs Creek increased throughout the study period. While total aquatic plant standing crop exhibited a step-wise increase over successive seasons (i.e., from spring to summer to fall), significant increases in epilithon biomass did not occur until the fall sample period. The submergent aquatic macrophytes *Myriophyllum sibericum* (northern watermilfoil) and *Polygonum amphibium* (water smartweed) were the dominant plant species during all sample periods. Abundant growths of macrophytes are central to the ecological integrity of Big Springs

Creek as these organisms provide complex habitat for fish and aquatic invertebrates and serve as key food resources to the aquatic food web following senescence.

The aquatic macroinvertebrate communities in Big Springs Creek and the Shasta River were dominated, both numerically in terms of biomass, by members of the collector-gatherer feeding guild. At times, collector-gatherers accounted for more than 97% of the entire macroinvertebrate assemblage. Conversely, shredding macroinvertebrates (organisms that process coarse particulate organic matter) were surprisingly rare in both Big Springs Creek and the Shasta River, never accounting for more than 0.4% of the total assemblage on any date. Tolerant organisms (those with published tolerance values $\geq$8 out of 10) and non-insect taxa were more abundant at our Big Springs sample sites than in the Shasta River. The Shasta River generally exhibited higher taxonomic richness and community evenness relative to Big Springs Creek during all sample periods. We documented extremely high densities of the amphipod *Hyalella* sp. during both the summer and fall in the middle reach of Big Springs Creek (BS-Mid). Amphipod densities exceeded $80{,}000 \cdot m^{-2}$ in the fall and abundances were considerably greater in BS-Mid than at all other Big Springs and Shasta River sample sites. The biomass of aquatic insects emerging from Big Springs Creek was 204% greater during the fall than the summer. Similarly, aerial inputs to the creek were significantly greater in the fall and consisted predominantly of large-bodied invertebrates.

Sources of organic matter at the base of the food chain exhibited variable stable carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) isotope ratios during the all sample periods. With the exception of detritus during the summer, all potential sources of organic matter during each season had mean carbon to nitrogen (C:N) molar ratios below 17:1, a reported critical maximum ratio for maintaining the growth of primary consumers. In general, the numerically dominant aquatic macroinvertebrate taxa in Big Springs Creek appeared to be deriving their carbon from sources of fine particulate organic matter, epilithic biofilms and attached algae. While primary consumers exhibited only modest variability in $\delta^{13}C$ values during the spring and summer, values during the fall sample period were highly variable and did not track any single food source. During the spring sample period the muscle tissue of juvenile salmonids (<55 mm fork length) contained enriched carbon and nitrogen stable isotope ratios due to the presence of residual maternal yolk. These elevated values greatly hindered our interpretation of both diet and trophic position during the spring. By summer, however, juvenile salmon were in isotopic equilibrium with their riverine diets and appeared to be feeding opportunistically on the invertebrate assemblage.

Our findings provide important and heretofore unknown information regarding the structure and function of the aquatic community in Big Springs Creek. 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. Future studies should seek to (1) quantify ecological rates such as primary and secondary production, invertebrate drift, emergence, and aerial input; (*2*) characterize the contributions of epiphytic biofilm and the different size fractions of benthic organic matter to primary consumers; (3) document temporal changes in dissolved carbon and nitrogen isotope ratios and how these values influence the isotopic signatures of key carbon sources at the base of the food web; (4) determine stable isotope values for the complete fish assemblage across all seasons; and (5) incorporate traditional gut content analysis to confirm trophic relationships inferred from the stable isotope studies.

6.0 Fish Abundance and Habitat Surveys

Snorkel surveys were conducted in Big Springs Creek from April 2008 to January 2009 to determine fish relative abundance and habitat usage. Seven distinct reference reaches in Big Springs Creek were established and a total of 97 surveys were conducted. Six species of fish were documented including; coho salmon, Chinook salmon, steelhead, speckled dace, Klamath small-scale sucker, and marbled sculpin. Due to cattle grazing during the previous winter, followed by removal of cattle from the creek for the summer, habitat conditions changed dramatically throughout the study period. Below we discuss how temporal changes in habitat affected fish abundance and habitat usage in Big Springs Creek.

6.1 *Methods*

We conducted non-invasive snorkel surveys to determine the relative abundance of fish in Big Springs Creek and their utilization of different habitats. Seven distinct reaches throughout Big Springs Creek were surveyed bi-weekly from April through January as conditions and access allowed (Figure 52). Reaches ranged from 24 to 114 m in length. During each survey, a single snorkler moved upstream through the length of the reach and enumerated fish by species and age class on a wrist slate. After a reach survey was completed, instream cover, substrate type and exposed substrate were qualitatively estimated and recorded. Further, temperature, dissolved oxygen, turbidity, pH, and conductivity were measured using a YSI 6820 data sonde.

Due to the uniform nature of Big Springs Creek habitats, survey reaches were not chosen by habitat type as was done for previous surveys of the Shasta River (Jeffres 2008). Rather, survey reaches were selected to include as much variability as possible longitudinally throughout Big Springs Creek. Reaches included uniform reaches with little variability, bridge crossings, above and below the water wheel and in the willows below Big Springs Lake. The only survey reach with complex habitat was near the top of the Big Springs Creek at the outlet of the lake where willows are present in the channel providing velocity refuge and overhead cover. During early spring when snorkel surveys began, habitat was homogeneous throughout Big Springs Creek with only minor changes in depth at constriction points. As spring/summer progressed, water volume was reduced due to irrigation withdrawal and aquatic and emergent macrophytes became more abundant (Figures 3, 29). With reduced flow, one survey reach was abandoned due to insufficient water to snorkel. As aquatic macrophytes grew, the channel narrowed and in some places no single channel was visible. This created varying habitat complexity in the survey reaches throughout the sampling effort.

Cattle were excluded from most of Big Springs Creek and all of the survey sites from April through September. This allowed aquatic and emergent vegetation to establish, trapping fine sediment on the margins ultimately creating complex habitat for rearing fish. In mid-September cattle were allowed into Big Springs Creek below the water wheel and above Little Springs Creek and they began feeding on the aquatic and emergent vegetation. This allowed fine sediment and organic matter to be mobilized and entrained in the water column and made it extremely difficult to observe fish at the three downstream-most snorkel sites. In mid-December the cattle were moved to downstream of Little Springs Creek where they began feeding on aquatic and emergent vegetation. Due to low visibility, fewer surveys were performed downstream of the cattle in the creek in the fall and winter months.

Figure 52. Snorkel survey locations on Big Springs Creek.

6.2 *Analysis*

We observed six fish species during our snorkel surveys of Big Springs Creek (coho salmon, Chinook salmon, steelhead trout, speckled dace, Klamath small-scale sucker, and marbled sculpin). In the following section, we report our findings on seasonal habitat use by coho, Chinook, steelhead, and speckled dace. Our results suggest that water temperature, habitat complexity, and physical barriers are the dominant drivers of fish distribution in Big Springs Creek.

6.2.1 Coho

Very little habitat was available for juvenile coho salmon in early spring due to a lack of cover and depth in Big Springs Creek,. Coho that emerged in Big Springs Creek likely moved from spawning grounds either downstream to the Shasta River or upstream toward the water wheel where adequate cover (from blackberry brush) and depth (from the constriction) existed. During spring, when the coho were small and water velocities were high, the water wheel was a migration barrier for juvenile coho. As temperatures increased in Big Springs Creek and Shasta River in May (Figure 11), juvenile coho moved into the upper portions of Big Springs Creek near the lake outlet (Figure 53). Many of the juvenile coho at the lake outlet migrated from downstream sections of Big Springs Creek and Shasta River (B. Chesney pers. comm.). During times of warm water in Big Springs Creek and Shasta River, primarily due to irrigation practices and stream degradation, the lake outlet provided an important over summering thermal refuge for juvenile coho salmon. The pool formed from the outlet of the lake provided adequate depth and cover from nearby willow trees. Additionally, the outlet contained a very high abundance of amphipods that get transported out of the lake and provide the juvenile coho with an abundant food source. Juvenile coho that reared in the outlet of Big Springs Lake grew at a very high rate compared to the neighboring Scott River. In fact, when juvenile coho were sampled from the lake outlet in November they were found to be roughly twice the length of coho sampled from the Scott River the previous week (B. Chesney pers. comm.).

From late May through December, juvenile coho were only found in the lake outlet and adjacent downstream willows in Big Springs Creek, likely due to high water temperatures in the rest of the creek. In late July, a beaver dam was constructed in the willows and increased the water stage throughout the reach. During surveys conducted after the beaver dam was established, juvenile coho were observed using deeper water habitat throughout the willow reach. During December, when air temperatures were very cold, Big Springs Creek temperatures also cooled with distance from the spring source. The outlet from Big Springs Lake was much cooler than downstream at the water wheel, which lies immediately below a large influx of spring water. This cold-water input altered coho distribution as they moved from the willow reach near the outlet of the lake to downstream of the water wheel where temperatures were warmer. These observations provide valuable insight into how coho utilize seasonally limited habitat in Big Springs Creek.

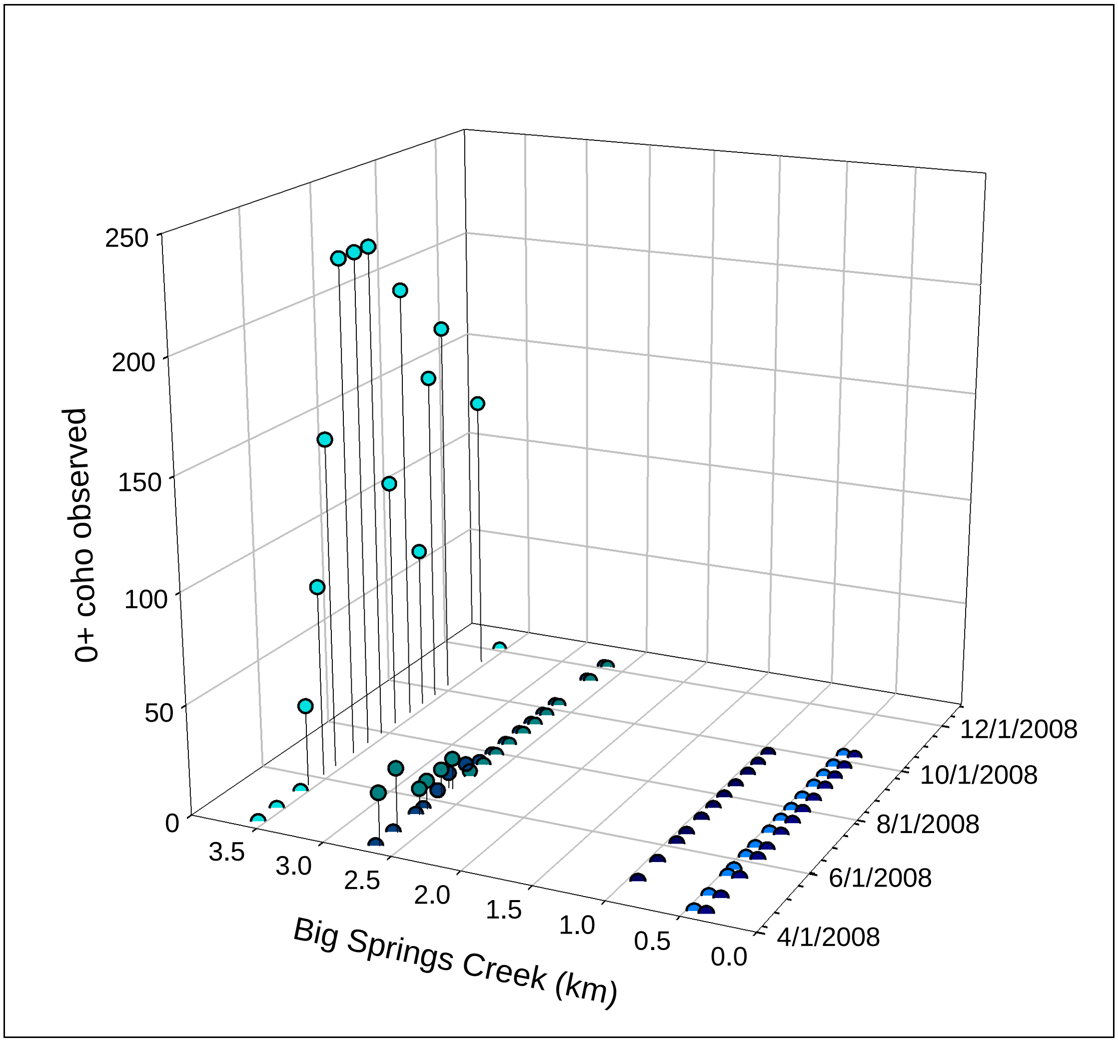


Figure 53. Abundance of juvenile coho salmon in Big Springs Creek at the various sample locations throughout the study period. Juvenile coho were most abundant at the outlet of Big Springs Lake (km 3.55) after water temperatures increased downstream in Big Springs Creek and the Shasta River.

6.2.2 Chinook Salmon

When surveys began in early April, no juvenile Chinook were observed in Big Springs Creek. This is likely due to warm temperatures in the creek during winter months allowing for early hatching and rapid growth of Chinook eggs and fry. Upon emerging from the gravels, Chinook fry left Big Springs Creek in search of suitable rearing habitat. On one occasion during the summer (3 July 2008), a large juvenile Chinook was observed in lower Big Springs Creek. At the time of the observation, aquatic macrophytes had grown in and stream depth was greater than during spring. After this observation, however, no Chinook were observed in Big Springs Creek until the fall when adults returned to spawn.

Adult Chinook returned to Big Springs Creek during October and began spawning in the lower portion of the creek (river km 0 to 1.6). During this time, cattle were excluded from Big Springs Creek below Little Springs Creek to the confluence of the Shasta River. However, cattle were not excluded from the channel above Little Springs Creek and were observed trampling Chinook redds on multiple occasions. Similar, observations were also made below Little Springs Creek in late December. The presence of cattle in the creek can adversely effect Shasta River salmonid populations. Trampling of eggs and fry while they are in the gravels can be a significant source of mortality. Additionally, removal of aquatic and emergent vegetation increases the amount of fine sediment mobilized in the creek. Increased fine sediment reduces the quality of spawning gravels and the removal of aquatic macrophytes reduces the amount of rearing habitat for those fish that do emerge from the gravels.

Snorkel surveys conducted below Little Springs Creek yielded many important observations concerning Chinook spawning behavior. Most significantly, we observed large numbers of sexually mature male Chinook parr and documented several of them participating in spawning activities. Using an underwater camera, we successfully recorded this rare behavior. To our knowledge, our recording represents the first time that mature male parr have been video taped participating in spawning activities in the wild. Mature male parr were 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 did not die after spawning (iteroparity) and produced viable progeny when crossed with an adult female. Mature male parr are very rare in the wild and are most often found in hatchery populations where growth rates are high due to an abundance of food resources (Larsen et al. 2004). While the extent to which mature parr contribute to the population in the Shasta River or Klamath Basin is unknown, this life history strategy may help the population 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. Mature parr highlight the growth potential of juvenile fish in the Big Springs Creek if thermal refuge is found during critical over summer rearing.

6.2.3 Steelhead

Steelhead are regarded as the most thermally tolerant of the salmonid species and thus widely distributed in the Shasta River and its tributaries. Steelhead were observed throughout the length of Big Springs Creek during our 10 month study. Young-of-the-year steelhead (age 0+) were observed using margin and aquatic macrophyte habitat downstream of the water wheel until they were approximately 80 mm when they moved to deeper water. In general, numbers of age 0+ steelhead in Big Springs Creek decreased throughout the summer (Figure 54).

A school of 1+ steelhead was observed in the deep run immediately above the water wheel throughout the study. These fish ranged from approximately 20 to 61 cm (8 to 24 inches) in length and were observed holding in relatively deep water (1.2 m deep) containing aquatic macrophytes for cover. Moreover, these fish were frequently observed feeding at an extremely high rate (approximately every 10-15 seconds) and were markedly robust for fish of their length likely due to optimal temperature and abundant food source.

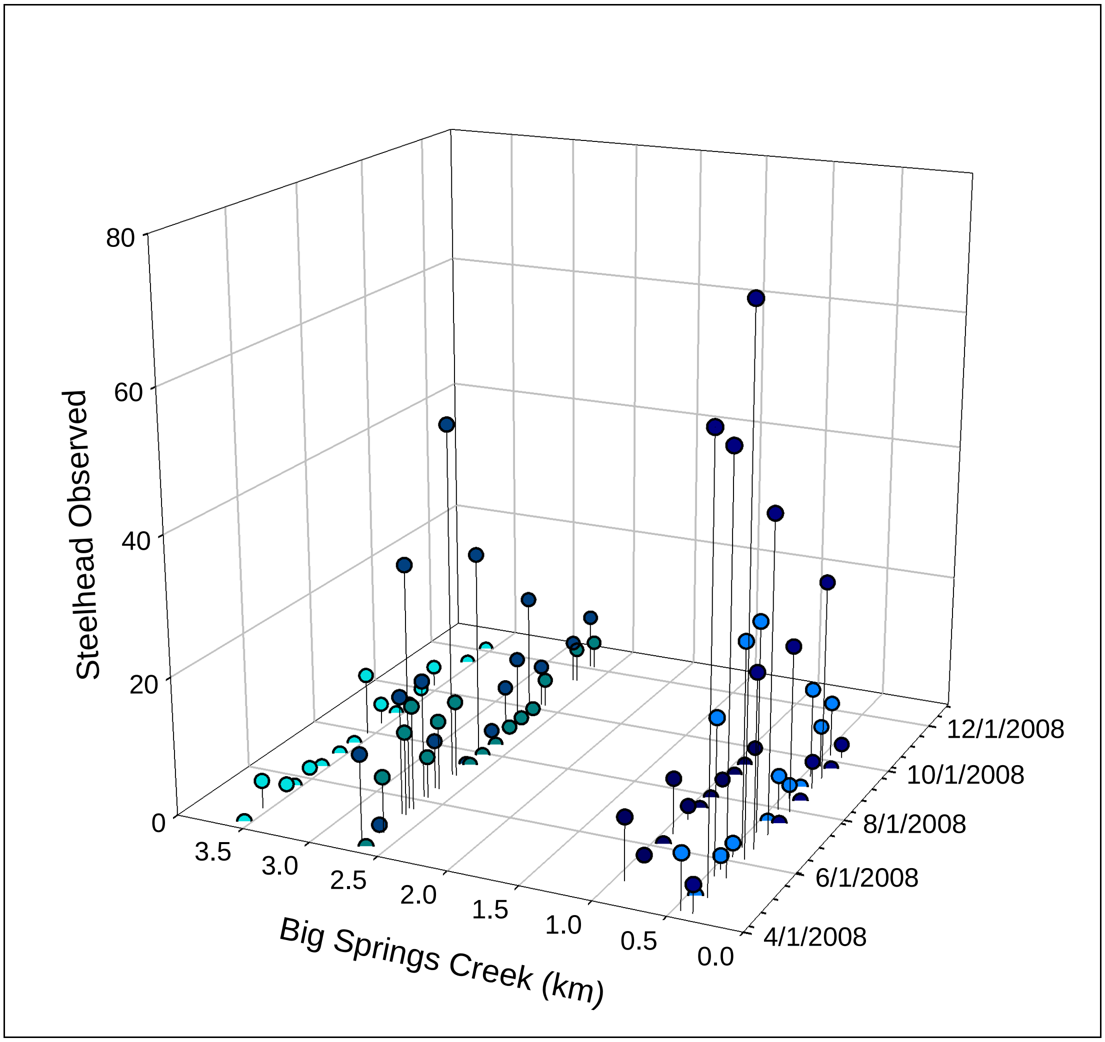


Figure 54. Abundance of steelhead observed in Big Springs Creek at various survey locations. Many of the steelhead observed at the water wheel location (2.67 km) were adult (1+), while most steelhead observed at the downstream locations (.37-.93 km) were 0+.

6.2.4 Non-Salmonids

Speckled dace, Klamath small-scale sucker, and marbled sculpin were the three non-salmonid species observed in Big Springs Creek during our study. Observations of speckled dace, the most abundant of the non-salmonids, declined throughout the summer months (Figure 55). It must be noted, however, that speckled dace were typically observed among aquatic macrophytes near the creek margin. Use of this habitat made accurate censusing difficult and may have led to an underestimation of true population abundance. Speckled dace were never observed above the water wheel, which likely functions as a migration barrier. Klamath small-scale suckers and marbled sculpin were rarely observed in Big Springs Creek, presumably due to a lack of suitable habitat and warm water temperatures.

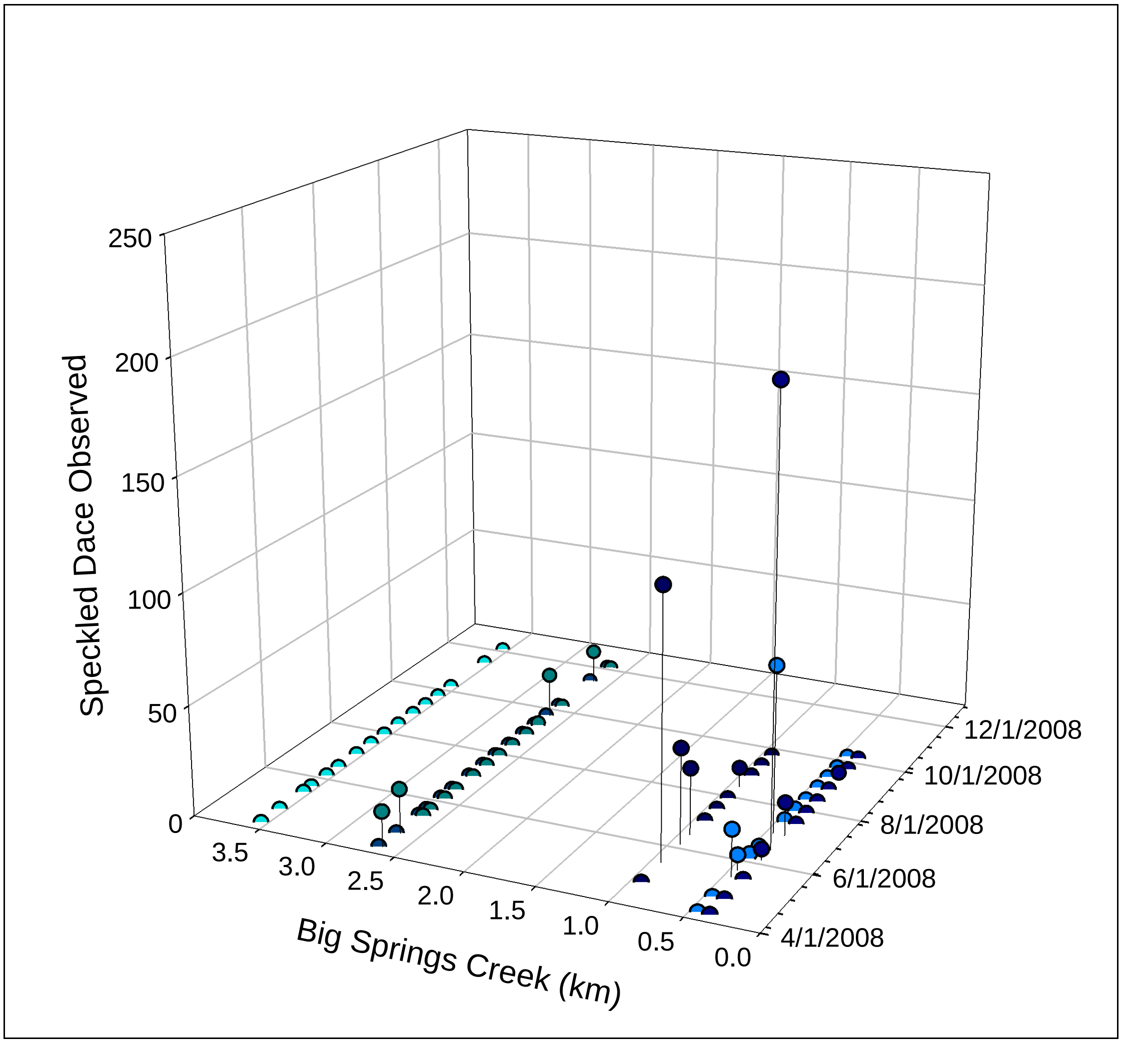


Figure 55. Abundance of speckled dace observed in Big Springs Creek at various survey locations. No speckled dace were observed above the water wheel (2.67 km) which likely is a migration barrier for the relatively small fish.

6.3 *Conclusions*

Physical conditions (water temperature, habitat complexity, and migratory barriers) are the principal factors affecting fish distribution in Big Springs Creek. There was little suitable rearing habitat available to salmonids during the spring as a result of intensive cattle during the preceding winter months. When water temperatures warmed in May, juvenile coho moved to the outlet of Big Springs Lake where suitable habitat and cool temperatures existed. Coho rearing in the lake outlet benefited from an abundant food source (see section 5 for more details) and grew at rates nearly double those of coho in the adjacent Scott River watershed (B. Chesney pers. Comm.). If high quality oversummering conditions like those found at the lake outlet can be replicated through restoration activities at other locations in Big Springs Creek, over summer habitat, a key limiting factor in the coho life history in the Shasta River watershed will be ameliorated.

7.0 Restoration Strategies

Ecologic, hydrologic and geomorphic assessment activities at Big Springs Ranch indicate that salmonid habitat conditions in Big Springs Creek are severely degraded as a result of past and present ranch management. Based on current knowledge, we identified three principle factors that limit the maintenance of self-sustaining salmonid populations in Big Springs Creek and much of the Shasta River downstream:

- Seasonal water temperature impairment;
- diminished habitat complexity and availability in Big Springs Creek; and
- downstream propagation of seasonally elevated water temperature into the habitat-rich Shasta River below Big Springs Creek

These primary limiting factors are inter-related and understood to various degrees. The role that elevated seasonal temperatures play is the best understood, but implications under future restoration conditions are still largely undefined. The various factors that affect habitat and the associated interrelationship are complex, and certain elements are not completely defined. For example, there are most likely inter-relationships (i.e., feedback mechanisms) between irrigation practices and cattle grazing within the wetted channels and along the channel margins of Big Springs Creek and its tributaries with channel margin features (e.g., riparian vegetation communities), geomorphic conditions, and habitat complexity. These feedbacks result in key indicators of stream and salmonid habitat degradation in Big Springs Creek, including:

- Seasonally elevated water temperatures
- Reduced streamflows during irrigation season (1 April to 30 September)
- Channel bank erosion and fine sediment introduction
- Absence of aquatic macrophytes and emergent vegetation

Monitoring of physical habitat conditions and ecological functioning is imperative to improve current knowledge about the stream system and to inform and support decisions about restoration strategies. A principal objective of this process was to provide this information explicitly to fulfill these needs and bring these analyses to bear on potential restoration actions.

A principal element of any long-term restoration program where information is limited is a monitoring plan. Subsequently, a range of restoration actions should be considered. Herein, both passive and active restoration activities are recommended for Big Springs Ranch to mitigate habitat and aquatic system degradation. To help prioritize these restoration activities, the hydrodynamic model (described in Section 7.4) was used to simulate projected restoration configurations after 1, 5 and 20 years of restoration.

7.1 *Monitoring Recommendations*

UC Davis Center for Watershed Sciences and Watercourse Engineering have monitored hydrologic, geomorphic, water quality and ecological (i.e. fish, aquatic macrophytes and benthic macroinvertebrates) conditions in Big Springs Creek since March 2008. This effort has provided an invaluable baseline data set documenting habitat conditions as described in previous sections. A comprehensive monitoring plan will allow for real-time information gathering that will measure the success of restoration activities and improve the performance of the hydrodynamic model, both of which will provide guidance if restoration/ranch management actions need to be altered. Recommendations for future monitoring efforts at Big Springs Ranch are outlined below.

7.1.1 Flow

Monitoring streamflow is vital to quantifying spring output and water use on Big Springs Ranch. We propose that the current array of stage gauges, with the exception of the lowest bridge, remain in place throughout restoration activities. Due to aquatic vegetation growth and a lack of a weir-type structure, the lowest crossing cannot provide an accurate stage-discharge relationship. Discharge from the water wheel location and Little Springs Creek combined provides a proxy for discharge at the lowest crossing and should be used for future studies of flow in Big Springs Creek. Maintaining stage gauges at Upper Shasta River, Parks Creek, Hole in the Ground Creek, and the top of the Nelson Ranch will provide an adequate water budget for Big Springs Ranch.

Also, an effort should be made to isolate and quantify flow contributed by individual springs and assess the spring output during spring, summer, fall, and winter to determine seasonal changes as well as their potential response to seasonal groundwater withdrawals. Spring discharge can be quantified by isolating flows and completing a discharge measurement. These data would improve the discharge distribution in the hydrodynamic and temperature modeling effort, allowing a better representation of accretions and associated temperature.

7.1.2 Water Temperature

Currently, water temperature is the largest threat to juvenile coho rearing in Big Springs Creek and the Shasta River. As one of the primary goals is to reduce water temperatures to improve salmonid habitat, monitoring water temperature is essential to determining the success of various restoration activities on Big Springs Creek. Thermistors should be maintained in a longitudinal array in Big Springs Creek and the Shasta River throughout the Big Springs and Nelson Properties. Upstream and downstream property boundaries are key monitoring sites; existing intermediate sites should be maintained.

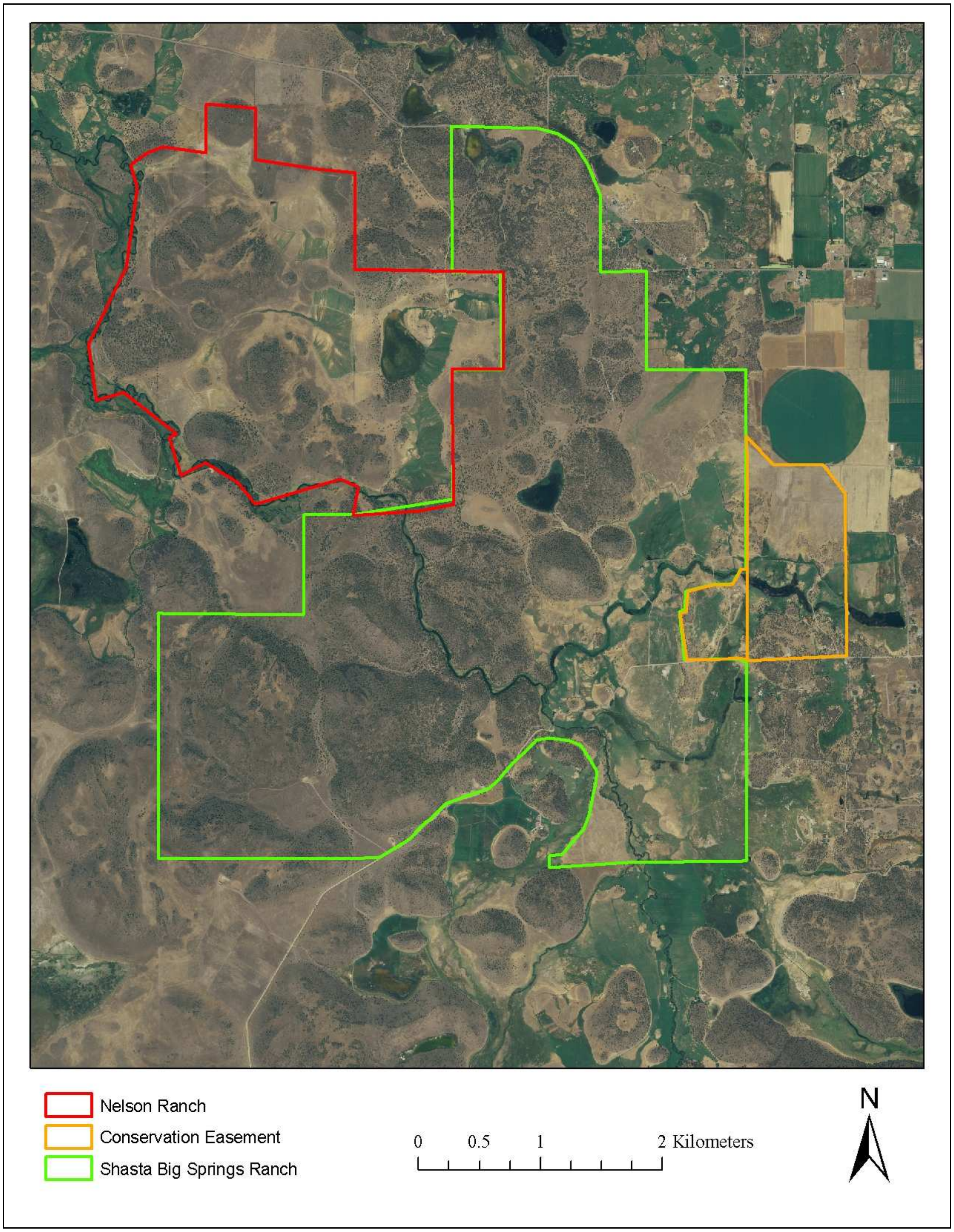


Figure 56. Map showing Nelson Ranch, Shasta Big Springs Ranch, and Conservation easement on Big Springs Creek.

Thermistors should also be placed at selected return flow locations and where other tributaries enter Big Springs Creek or the Shasta River (i.e. Little Springs Creek, Parks Creek, Hole in the Ground, etc.). Monitoring return flow and tributary temperatures will improve our understanding of the temperature profiles at the stream's edge. Finally, thermistors should be placed in spring sources throughout Big Springs Creek to determine the individual thermal contributions. As well as improving our current understanding of the thermal regime, these data would identify valuable habitat areas, potential thermal refugia and support temperature modeling.

7.1.3 Geomorphology

Several geomorphology components are important to monitor when assessing restoration actions. Measurable physical parameters (e.g. channel width and depth) principally reflect geomorphic responses to ranch management as opposed to underlying hydrologic or geomorphic processes inherited from upstream catchments. Furthermore, geomorphic responses (e.g. channel widening and shallowing) to land management induce complex reactions in the form of degraded water quality and increased fine sediment accumulation. Because the stream's geomorphology primarily responds to land management changes, initial stream restoration activities should be assessed based on first the stream's geomorphic response and then on the more complex metrics of ecological function that represent the biotic response to restoration. Also, because the rate of heating in the hydrodynamic model directly depends on channel geomorphology, maintaining an accurate model configuration is vital to providing assessments of restoration activities.

Channel morphology characteristics such as width, depth and channel pattern are easily measured physical parameters that are likely to respond to both passive and active restoration activities (Graf 2001). Currently 64 cross sections have been established in Big Springs Creek from which baseline geomorphic characteristics have been established. Furthermore, documented longitudinal bed and water surface elevation profiles determine current hydraulic energy gradients. As the quality of spawning gravels in Big Springs Creek has not been evaluated, such data should be acquired to help identify future responses to restorative activities.

Seasonal reoccupation of selected channel cross sections and thalweg longitudinal profiles should be performed to monitor and evaluate restoration activities. Mapping channel geometries at different periods (e.g. summer and winter) will demonstrate whether channel geometries will narrow permanently or seasonally with vegetation senescence. Furthermore, the quality of spawning gravels in Big Springs Creek should be monitored over time using surface and/or bulk sampling techniques to quantify changes in particle size distributions. Replicable geomorphic survey activities will enable documentation of physical habitat changes (e.g. channel widths and depths) in response to restoration. Survey activities will also facilitate quantification of the direction, magnitude and rates of change for important geomorphic characteristics such as channel gradient, sediment storage, and coarse gravel recruitment and transport.

7.1.4 Water Quality

Further research should continue to refine water quality monitoring to ascertain seasonal variations in water quality conditions consistent with restoration objectives. Restoration of the creek through narrowing will most likely result in reduced stream width, increased velocity and depth, and reduced transit times. This condition will limit growth of aquatic macrophytes to the margins and a shift in nutrient availability in the downstream direction will theoretically occur. Monthly water quality sampling will be necessary to quantify this potential trophic shift in downstream river reaches and be critical to ascertain the implications of TMDL implementation actions and overall water quality conditions in downstream Shasta River reaches.

7.1.5 Aquatic Macrophytes

Quantifying longitudinal changes in aquatic macrophyte species composition and biomass will be an important component in determining the success of cattle exclusion. Seasonal assessments of aquatic macrophyte abundance will show how plants recover with the removal of cattle grazing. In most spring-fed systems fine sediment trapping by aquatic macrophytes is often followed by a release of fine sediment when plants senesce in the fall (Cotton et al. 2006). Quantifying seasonal senescence without grazing pressure will be important in determining how much fine sediment the plants can capture annually. Fine sediment capture by aquatic macrophytes will improve spawning gravels and facilitate habitat for establishment by emergent vegetation. It will also lead to channel narrowing, deepening, meandering, and lower residence time, which will result in lower rates of water heating.

The amount of shade provided by aquatic macrophytes should also be monitored to improve the performance of the hydrodynamic model. Quantifying shade contributions by aquatic macrophytes will be accomplished by using handheld solar pyronometer and/or PAR sensors for the subsurface work. Shade attributes of the various shade species expected in the project area under restored conditions (e.g. woody and riparian species, terrestrial species of interest) will be identified. Information will be collected at different periods of the year to capture variations in vegetative cover and the effects of ambient temperature changes.

7.1.6 Benthic Macroinvertebrates

Aquatic macroinvertebrates should be monitored 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 restoration success. Because of the unique physical conditions, seasonal longitudinal sampling should be performed in Big Springs Creek as well as in the Shasta River above and below Big Springs Creek. This will quantify how aquatic macroinvertebrates respond to restoration actions in Big Springs Creek.

7.1.7 *Fish*

Continuing to monitor fish populations in Big Springs Creek is necessary to determine success of the primary goal of the purchase of the Big Springs Ranch, the recovery of the federally listed SONCC coho salmon. Currently, snorkel surveys and PIT tags have been used to provide information about salmonids' movements, preferred habitats and survival rates. Snorkel surveys should be continued to determine habitat use by salmonids and other resident fish as conditions change due to restoration activities. Also, monitoring of juvenile salmonids using PIT tags over time should be continued to determine the salmonids' movement in Big Springs Creek and the Shasta River, gather data describing juvenile to adult survival rates, and indicate the habitat locations that contribute to the high survival rates. Data regarding changes in survival rates and increased habitat locations are both strong indicators of successful restoration strategies.

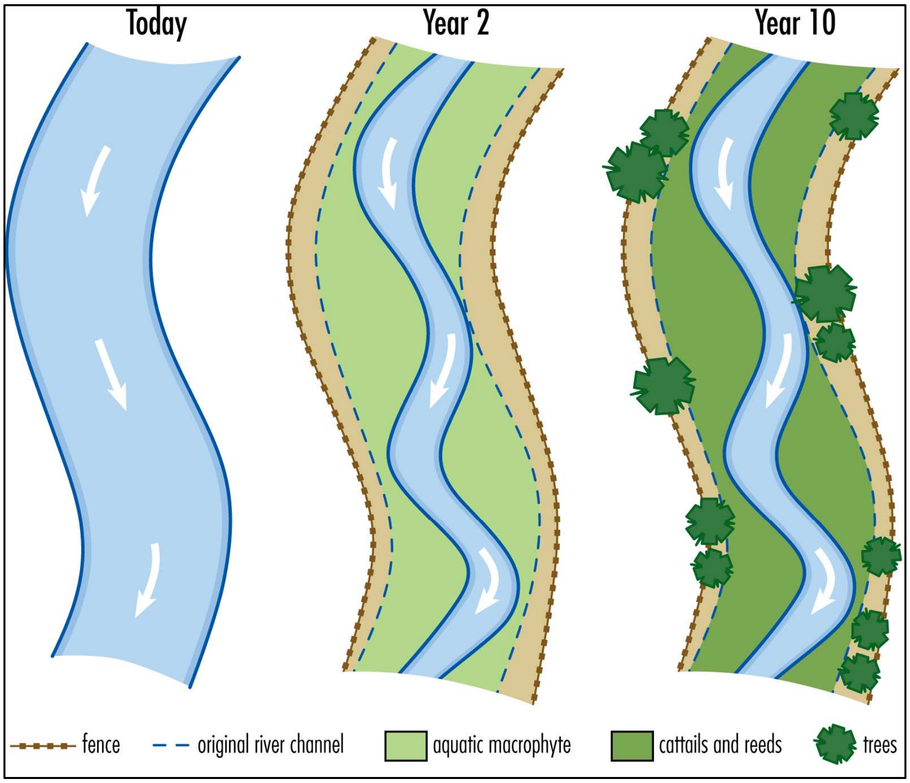
7.2 *Passive Restoration*

Ongoing monitoring efforts suggest passive restoration activities (i.e. activities that take place outside of the stream channel) are likely to yield benefits to habitat conditions in Big Springs Creek. However, current management policies regarding cattle grazing return flow, and tailwater returns to Big Springs Creek prevent passive restoration from occurring. Addressing these land management issues are critical to allow passive restoration to occur in Big Springs Creek. Cattle exclusion and tailwater management have been identified as critical priority actions that will enable passive restoration activities in Big Springs Creek.

7.2.1 *Cattle Exclusion*

Excluding cattle from Big Springs Creek may be the most cost effective and beneficial restoration activity available to ranch managers. When cattle are allowed to graze on aquatic and emergent vegetation, salmonid habitat conditions are adversely affected. Aquatic macrophytes are removed, stream banks destabilize, fine sediment is introduced to the stream and the channel widens and shallows. Because cattle were kept in pastures away from the creek during the previous summer, we were able to observe rapid short-term recovery of aquatic and emergent vegetation within the channel and margin habitat. As aquatic vegetation grew, the channel narrowed, stage increased and fine sediment mobilized from the creek bottom and collected in the margins revealing gravels in the mid-channel. Though aquatic macrophytes do senesce somewhat in the winter months in Big Springs Creek, the relatively warm spring water allows aquatic plants to continue to grow throughout the winter and retain their fine sediments. If aquatic macrophytes continue to grow and accumulate sediment and organic matter, they will create conditions conducive to the establishment of more permanent emergent plants such as tules (*Scirpus sp.*) and cat tails (*Typha sp.*)(Figure 57). With a more established, narrower channel, habitat conditions will be much more complex and suitable for all life stages of salmonids in the Big Springs Creek.

Establishment of aquatic vegetation provides more than physical habitat in Big Springs Creek; it can reduce water temperatures, too. By narrowing the channel and increasing depth, residence time and surface area are reduced. When the residence time and surface area are reduced, the rate of heating decreases, which reduces water temperatures in Big Springs Creek and downstream in the Shasta River. As reduced water temperatures are a key component of improved salmonid habitats, enabling the creek to naturally create conditions to maintain cool water temperatures through cattle exclusion is critical.



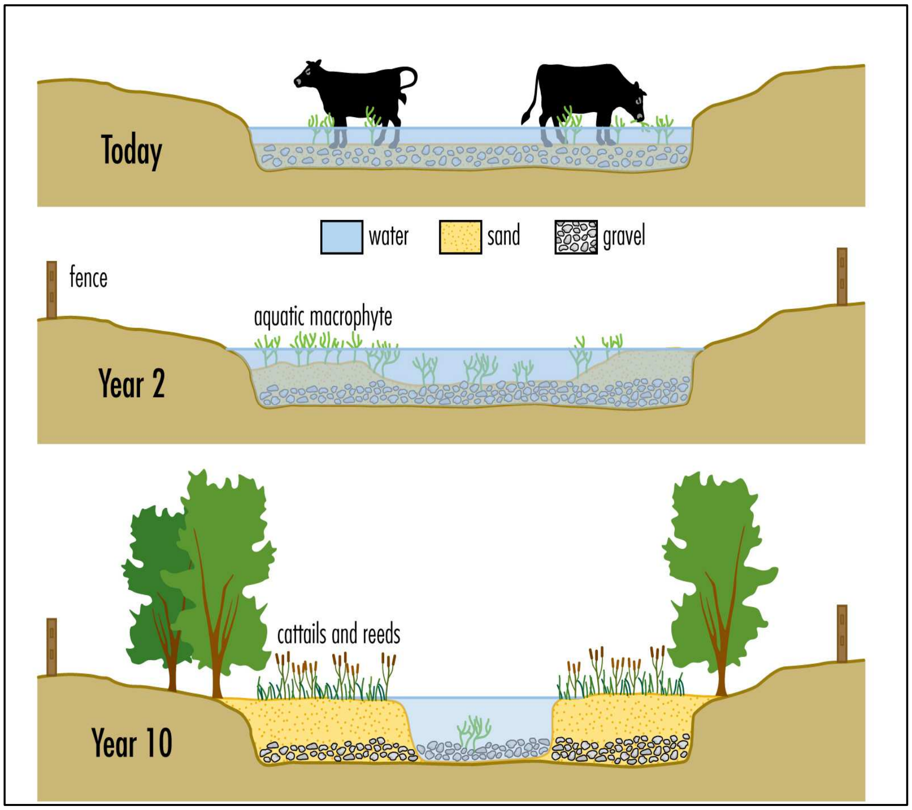


Figure 57. Conceptual model of future condition in a restored Big Springs Creek.

7.2.2 Water Management

Current irrigation practices on Big Springs Ranch create conditions that are detrimental to salmonids in Big Springs Creek and the Shasta River. Tailwater that enters Big Springs Creek from flood irrigation has been measured at temperatures in excess of 32^{o}C (~90^{o}F). As well as the heated water returning to the creek, the volume of water used to flood irrigate the fields contributes to the degradation of instream water temperatures.

Development of a ranch operations plan that accommodates conservation, conveyance, fields recapture systems, and ultimately reduces discharge to the creek will be paramount to successful water management. To prevent the heated return flow from draining directly into the creek, ditches can be constructed to capture return flows. These rock-filled ditches would run parallel to the creek to collect return flows. Then the collected irrigation runoff would collect in tailwater return ponds and either percolate into the soil and return to Big Springs Creek via groundwater exchange or be reused for irrigation. Coordinated management of irrigation practices will be paramount to reducing the amount of hot tailwater that enters the river. This entails increasing the efficiency of delivery ditches and head-gate structures so that proper, previously determined amounts of water can be placed on fields with less waste than current practices.

7.3 *Active Restoration*

Active restoration of Big Springs Creek will accelerate processes that would otherwise occur too late to benefit the dwindling salmon population in the Shasta River. By enhancing passive restoration by planting, removing constrictions or placing instream structures such as large woody debris, habitat conditions will improve more rapidly within Big Springs Creek than if only passive restoration occurs.

7.3.1 Planting of Emergent and Riparian Vegetation

When cattle were excluded from the creek during the previous summer, we observed abundant growth of aquatic macrophytes and a subsequent narrowing of the channel, increase in stream depth, trapping of fine sediment in the margins, and an increase in habitat complexity. While we saw the beginnings of seral stage growth, the distribution of these plants was limited to a fraction of the creek. By actively planting both emergent and riparian vegetation, passive restoration processes will be accelerated and greater ecological benefits (including extensive bank stabilization and habitat complexity) will occur sooner.

Preliminary restoration with various types of restoration plantings in different reaches of Big Springs Creek will provide information necessary for a complete and successful restoration of the creek. It will identify physical conditions where various vegetation species will be successful for future large scale plantings. From previous work we have defined four separate reaches in Big Springs Creek by channel gradient, substrate type, and volume of water. Restoration should entail detailed topographic surveys to establish baseline conditions and hydrodynamic modeling of different restoration configurations in different reaches. Once a planting approach is implemented, the biotic response to reestablishment of emergent and riparian vegetation will be surveyed. From information gathered by preliminary plantings, we will determine successful restoration strategies and implement those strategies where deemed appropriate throughout the creek.

7.3.2 Removal of Water Wheel

Since the late 1800s a partial impoundment located approximately one kilometer below Big Springs Lake has altered streamflow on Big Springs Creek. Historically utilized as a water wheel to generate power and provide a lift for irrigation water, the impoundment is now used to structurally support a road crossing and an irrigation water delivery pipe. The hydraulic head maintained by the flow-through impoundment provides an upstream migration barrier for federally and state-listed juvenile salmonids. Furthermore, the structure retards in-stream water velocities, resulting in the trapping of fine sediment and the widening and shallowing of the wetted river channel for approximately 400 meters upstream. Reduced water velocities (and associated transit times) and increased surface area of the wetted channel facilitate rapid thermal loading of the stream, with resultant detrimental impacts on cold-water salmonids. Removal of the water wheel impoundment will facilitate juvenile salmonid access to cold water spring-sources that provide critical over-summer rearing habitat. Additionally, impoundment removal will also reduce streamflow transit times, reduce the rate of thermal loading, and propagate cold water through Big Springs Creek and into the Shasta River.

7.3.3 Large Woody Debris/Instream Structure Placement

Currently instream structure in Big Springs Creek is very limited, yet has been shown to be a vital component in high quality coho salmon habitat (Cederholm et al. 1997). Instream structures such as large woody debris (LWD) placed in a spring-fed creek will have a much longer lifespan than instream structures placed in a non-spring-fed river due to the absence of high-flow events (Whiting and Moog 2001). Currently there is an abundance of dead juniper trees near the creek that could be placed in the stream experimentally. Trees placed in the stream will create velocity refugia and overhead cover for rearing juvenile salmonids. Geomorphic impacts of LWD placement will include localized scour of fine sediments, which will increase local depths. Snorkel surveys will determine the levels of scour and presence of fish near the LWD. If LWD placement is successful in experimental locations, a plan will be developed for large scale LWD placement throughout Big Springs Creek.

7.4 *Two-Dimensional Hydrodynamic and Temperature Modeling of Restoration Elements*

Given the scope of habitat degradation and the limited funds available for immediate action, a hydrodynamic model was developed to help assess restoration alternatives and identify priority actions (details of the model development are provided in Section 9.1). Such a model allows planners to simulate different water management and irrigation strategies and examine their effects on water temperatures. Instream grazing effects can be simulated by altering roughness and shading factors in different areas. Similarly, the impacts of reduced solar radiation due to riparian shading on water temperature can be simulated. Alternative channel geometries can also be tested to examine their impacts on water temperatures. Results can be tabulated or presented graphically to identify spatial and temporal conditions throughout the creek. To illustrate how local stream velocity and water temperature results are graphically illustrated, present temperature conditions were simulated for the reach from Big Springs Dam to the waterwheel (Figure 58 and Figure 59). To illustrate how broad temperature trends are graphically illustrated over the entire creek reach, present temperature conditions were also simulated (Figure 60). Simulations of present temperature conditions represent the geometries, flow and vegetative growth observed during the 2008 field season.

Based on our current knowledge of the creek, several passive and active restoration actions were identified and described in the previous section. Several restoration scenarios based on those actions were developed to estimate the effects of each action on water temperatures in Big Springs Creek after 1, 5, and 20 years. Alternative restoration configurations were built using the same techniques employed for the base case hydrodynamic model. A summary of the restoration configurations that were simulated for each target year is provided in the sections below. A discussion of the results from these simulations follows.

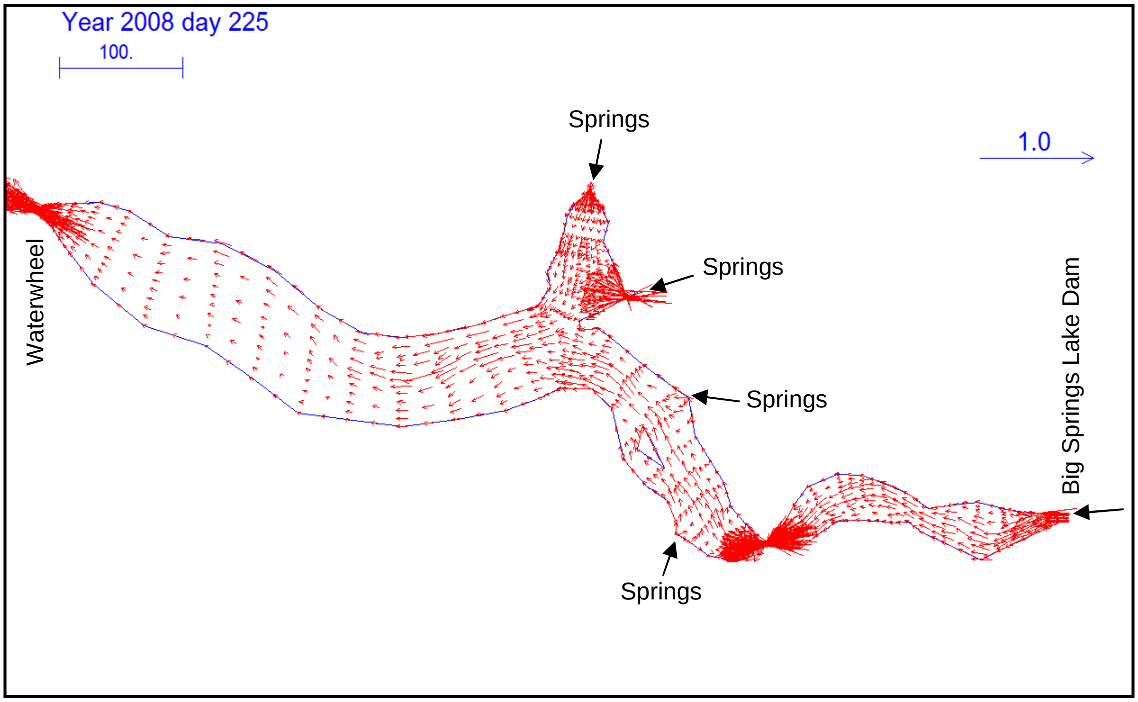


Figure 58. RMA-2 Simulated velocity vectors: present conditions (velocities in m/s)

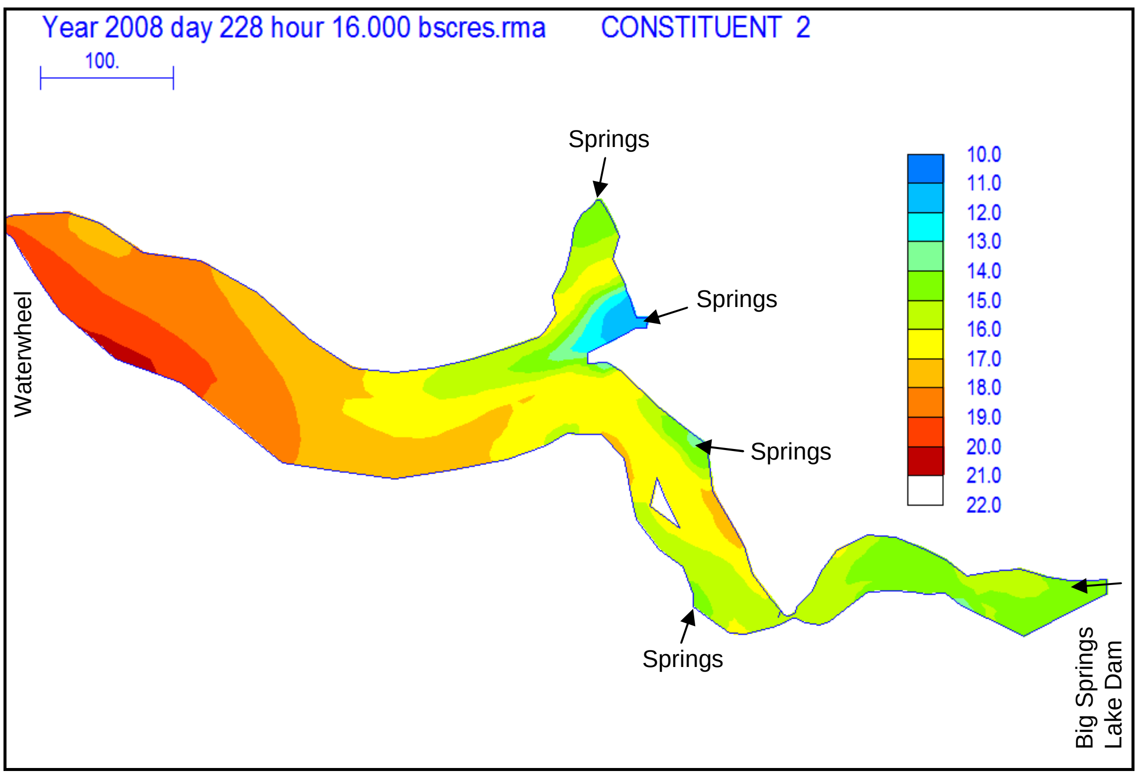


Figure 59. RMA-11 Simulated water temperatures: present conditions (temperatures in ^oC)

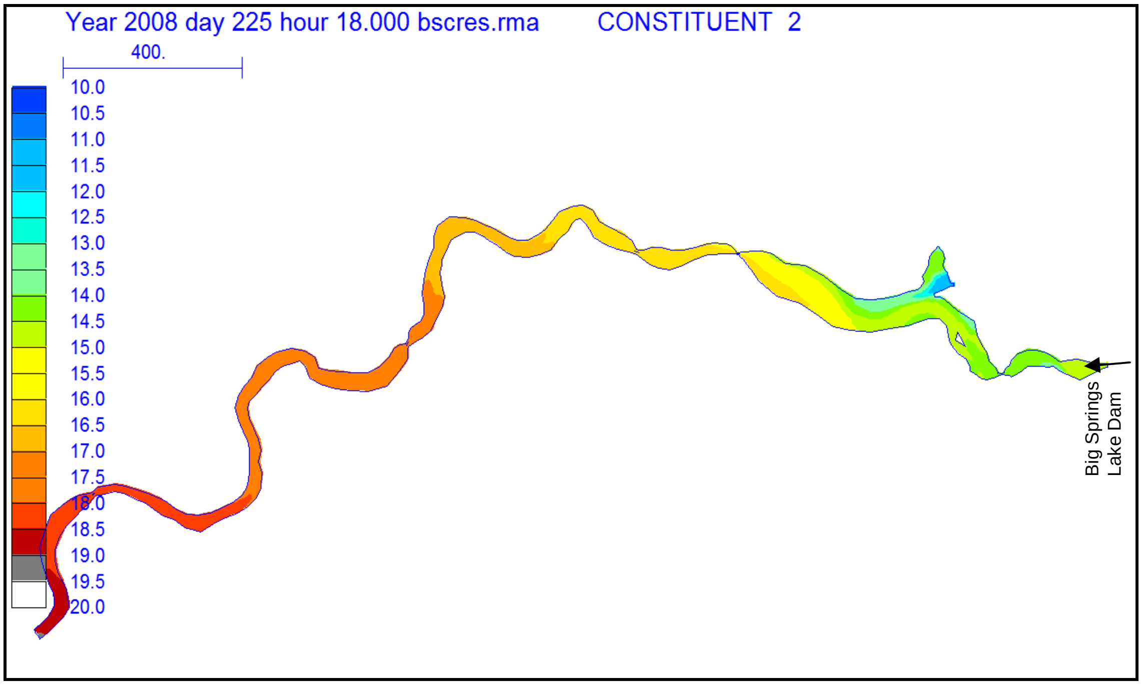


Figure 60. RMA-11 Simulated water temperatures: present conditions (temperatures in ^{o}C)

In addition to employing models to assess different boundary flows and temperature, model grids can be modified to accommodate alternative stream configurations under a proposed restoration action. RMA-2 and RMA-11 were used to model a proposed future condition with modified channel widths and islands, as shown in Figure 61 and Figure 62. In the narrowed channel, higher velocities are apparent and isolated channels are cooler than in the historic conditions. These examples are but one of a wide range of conditions that can be assessed with the model. Details of scenario assumptions are provided in section 7.4.1.

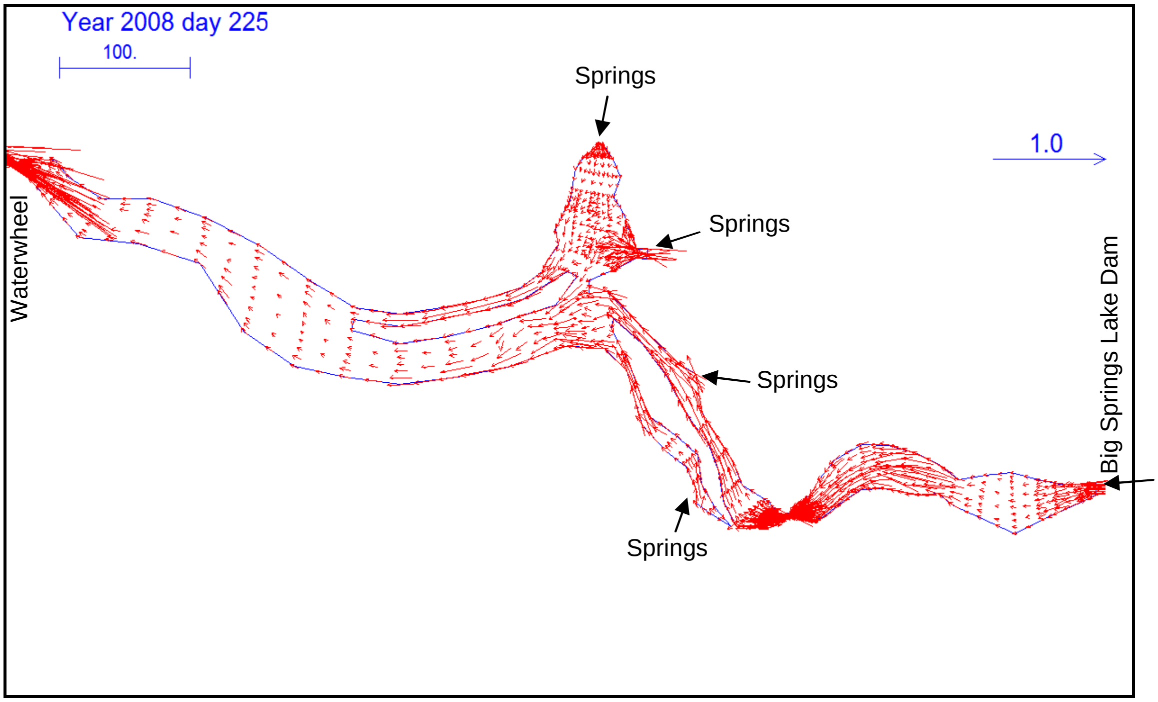


Figure 61. RMA-2 Simulated velocity vectors: modified channel under 2030 conditions (velocities in m/s)

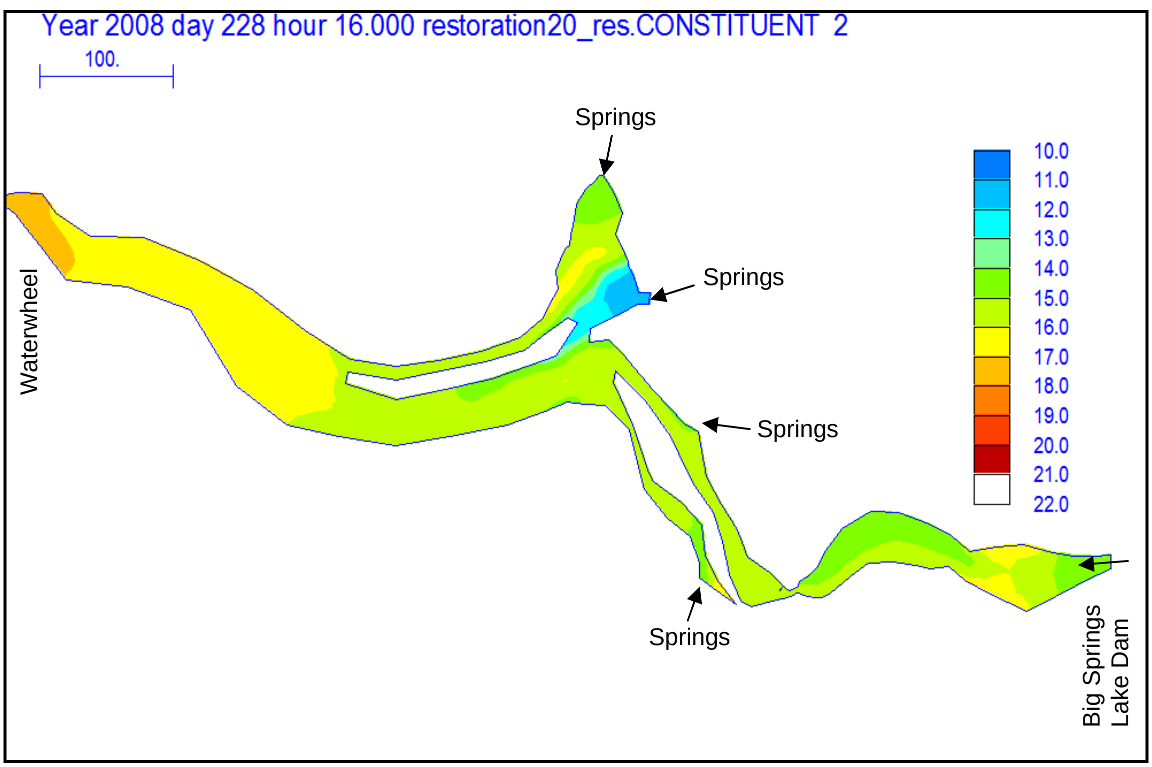


Figure 62. RMA-11 Simulated temperatures: modified channel under 2030 conditions (temperatures in ^{o}C)

7.4.1 Restoration Alternatives: Approach and Assumptions

Several model configurations and options were considered for preliminary assessment of restoration actions at Big Springs Creek. For the purposes of this report, we focus on three characteristics: streamflow, channel geometry, and vegetation growth. Changes in each of these categories were tested to assess the response of water temperatures in the creek. Streamflows were changed to test the thermal response in Big Springs Creek to release changes from Big Springs Dam, e.g., under a reduced irrigation delivery scenario. Channel geometry was changed to test the creek's thermal response to channel narrowing. A second configuration was assessed wherein the waterwheel was removed. Vegetation growth simulated the effects of increased roughness and shading in the stream channel due to aquatic and riparian vegetative succession. Currently, with the exception of configurations that include the removal of the water wheel, these configurations reflect a largely passive set of restoration activities. One could argue that aggressive planting of larger woody riparian species is active restoration. Summaries of the restoration configurations for each recovery year are provided in the sections below.

The approach used herein was to define conditions and activities to represent the response of passive and active restoration through time. Three time horizons are identified, approximately representing initial response of the system to cattle exclusion (1 year); an intermediate condition where the stream is presumed to be evolving to a dynamic equilibrium with exclusion fencing in place, but the long-term measures have not had time to provide appreciable benefit (5 years); and a long-term conditions where the channel is established and riparian vegetation is maturing and providing shade for temperature management (20 years). Simulations with and without the waterwheel in place were completed. In general, simulation results for with and without the waterwheel indicated minimal improvements in water temperature. However, model results identify that this reach experiences some of the largest rates of heating. After considerable review of model results and field data, we determined that additional, site specific information was required to more fully explore and represent this restoration option effectively in the model. Runs herein include the waterwheel.

Specific details for the 1 year, 5 year, and 20 year configurations are discussed below. Specific assumptions for flow, channel width, and vegetation are summarized in Table 9, Table 10, and Table 11, respectively. Restoration conditions are presented on a sub-reach basis. Each sub-reach is defined as follows:

- Big Springs Dam to Busk residence bridge
- Busk residence bridge to alcove springs
- Alcove springs area
- Alcove springs to waterwheel
- Waterwheel to corral crossing immediately downstream
- Lower creek (below corral crossing)

<u>Year 1 restoration configuration</u>

The first restoration configuration focused on the immediate benefits to water temperatures given one year of passive restoration activities. This first-year configuration assumes that livestock are excluded from the entire creek reach and that no return flows enter the creek as surface runoff. Streamflow, channel geometry and vegetative growth changes were predicted based on field data collected over the past year.

Streamflow alternatives focused on release changes made from Big Springs Dam. Data collected from the stage gauge just below Big Springs Dam indicates that during the irrigation season (when maximum water temperatures are observed) the minimum amount of water released into Big Springs Creek is approximately 5-7 ft^3/s. The water right held by Big Springs Ranch allows it to divert 10 ft^3/s from Big Springs Lake. Streamflow alternatives for year 1 recovery conditions test the effect of releasing the minimum and maximum amount of water into the creek. That is, the minimum discharge released from the dam is assumed to be 5 ft^3/s. The maximum possible discharge is assumed to be 15 ft^3/s. This discharge volume was determined by adding 10 ft^3/s to the minimum flow; this assumes that irrigation diversions are released into the creek instead of diverted to irrigation canal system. These two flow regimes are intended to bracket the available water at Big Springs Dam.

Channel geometry changes represent a narrowed stream channel that results from one year aquatic and riparian vegetation growth (for a conceptual illustration of this idea, see Year 2 in Figure 57). Because the livestock were excluded from the stream channel last spring, we were able to observe the rapid growth of aquatic macrophytes. These observations provide invaluable observations of the location and extent of first-year growth in the creek. The location and dimensions of the new stream channel are determined based on photographs and field observations made during spring and summer 2008 when vegetative growth was at a maximum. Channel depth changes were determined based on early data from the stream gauge at the lowest bridge and estimates based on field observations.

Observations of rapid aquatic vegetation also allowed us to determine locations where roughness and shading associated with instream aquatic vegetation would potentially occur. Thick vegetative growth increases the roughness of those areas and prevents water from flowing easily. Though the water moves slowly through the vegetation, the vegetation provides shade and limits the amount of solar radiation that can heat the water. Roughness and shading factors are estimated based on field observations.

Year 5 restoration configuration

The second restoration configuration simulated changes in the stream channel after five years of restoration activities. Streamflow alternatives are the same that were described for the year 1 restoration configuration, i.e., minimum flows released from Big Springs Dam are simulated as 5 ft^3/s, and maximum flows are 15 ft^3/s.

Channel geometries reflect increased narrowing in specific creek reaches. After five years of passive restoration activities, we assume that sediment will be transported and trapped by the increased emergent and intermediate stage aquatic macrophyte growth, further increasing channel narrowing. The new stream channel was estimated based on available stream bed data and field observations.

Aquatic vegetation was assumed to progress from emergent to seral species in some locations, especially along the south bank. Seral stage growth, including bulrush and cattail species, would provide modest but increased shading, roughness and bank stability. Emergent aquatic macrophyte species were also included in some of the creek reaches. Roughness and shading factors for the intermediate growth species are estimated based on field observations.

A restoration configuration was also developed to simulate the removal of the water wheel. The same streamflows and vegetation growth that are described above were used to isolate the effect of removing the water wheel. Stream geometry also remained the same with the exception of the area around the water wheel. At the water wheel's location, the stream width was increased from 8 m to 18 m to reflect estimated stream geometries prior to the water wheel's construction. Upstream of the water wheel, a narrower channel from the spring alcove to the water wheel was simulated to eliminate the backwater effects.

Year 20 restoration configuration

The third restoration configurations simulated water temperatures in Big Springs Creek after 20 years of restoration activities. As in the year 5 simulations, passive restoration activities included modified streamflow, channel geometry, and aquatic vegetation changes. Streamflow alternatives are the same as the flows simulated in year 1 and year 5 configurations.

Channel geometries reflect further narrowing from the year 5 configurations. In year 20, we assume that midstream bars have matured to marsh areas, splitting the creek channel to further narrow the flow areas. Also, we assume that downstream areas that previously trapped sediment have now established new and stable stream banks with woody riparian growth.

Extensive woody riparian vegetation is assumed to exist after twenty years of restoration activities. This new growth contributes extensive shade to the creek. However, since we assume it exists on established bars and stream banks, the woody riparian growth does not contribute any roughness to the flow channel. Some emergent and seral aquatic macrophyte growth is included in some of the creek's reaches.

A restoration configuration was also developed to simulate further recovery after the removal of the water wheel in year 5. The same streamflows and vegetation growth projected for the year 20 configuration that included the water wheel were applied to the alternative that did not. Stream geometries were also consistent except for the reach from the spring alcove to the water wheel, where a narrower and meandering channel was simulated.

Table 9. Assumptions for restored flow conditions in year 1 for individual sub-reaches of Big Springs Creek

Reach	Description	Year 1	Year 5	Year 20
A	Big Springs Lake Dam to Busk residence bridge	10 ft^3/s minimum, 15-17 ft^3/s max, net spring accretion of 10 ft^3/s	Same	Same
B	Busk residence bridge to alcove springs	Spring accretion of 16 ft^3/s	Same	Same
C	Alcove springs area	Spring accretion of 16 ft^3/s	Same	Same
D	Alcove springs to waterwheel	Spring accretion of 2.5 ft^3/s	Same	Same
E	Waterwheel to corral crossing	No local inflow	Same	Same
F	Lower creek (below corral crossing)	No local inflow	Same	Same

Table 10. Assumptions for restored channel width conditions in year 1 for individual sub-reaches of Big Springs Creek

Reach	Description	Year 1	Year 5	Year 20
A	Big Springs Lake Dam to Busk residence bridge	Down through willows, no change. 25% narrowing from willows to road	Same	Same
B	Busk residence bridge to alcove springs	25% narrowing	50% narrowing	More narrowing due to development of marsh island that splits the channel
C	Alcove springs area	No change	No change	No change
D	Alcove springs to waterwheel	25% narrowing	Reach tested to simulate presence and absence of waterwheel. Removal of waterwheel will yield reductions in width of 50 to 75%.	Reach tested to simulate presence and absence of waterwheel. Additional narrowing due to marsh island below alcove is tested.
E	Waterwheel to corral crossing	30% narrowing tested to simulate space occupied by extensive macrophyte growth.	Same	Same
F	Lower creek (below corral crossing)	25% reduction in width (or more), downstream depth of 0.42 m	35% reduction in width (or more), downstream depth of 0.56 m	50% reduction in width (or more), downstream depth of 0.66 m

Table 11. Assumptions for restored vegetation/shade configuration conditions in year 1 for individual sub-reaches of Big Springs Creek

Reach	Description	Year 1	Year 5	Year 20
A	Big Springs Lake Dam to Busk residence bridge	Existing willow thicket in top half of reach. In bottom half of reach, macrophyte A* growth along banks.	Add bulrush/cattail shading to elements adjacent to the south bank.	Complete willow thicket to road.
B	Busk residence bridge to alcove springs	Macrophyte A growth along banks. Mid-stream patches of macrophyte B.	Add bulrush/cattail shading south, near-shore elements, islands and bars.	Woody riparian shade on "new" banks. Notable shade in portions of this reach and on south and north shores.
C	Alcove springs area	Above old rock berm, 100% macrophyte A growth. Open water channel around left side of berm. Mix of macrophyte A and B downstream of berm.	Same	Same
D	Alcove springs to waterwheel	Extensive distribution of macrophyte A as per aerial photos and estimated flow paths. Less growth as velocities increase near waterwheel.	On shores and formed islands/bars, add with bulrush/cattail shading south, near-shore elements.	Woody riparian shade along south bank and along mid-channel bar.
E	Waterwheel to corral crossing	Extensive macrophyte A growth.	Same	Convert shading on south, near-shore elements to bulrush/cattail.
F	Lower creek (below corral crossing)	Extensive macrophyte A growth.	Add bulrush/cattail shading to the elements adjacent to the south bank.	Riparian shade on 50 percent of reach length.

*Macrophyte A is simulated using shading and roughness, while macrophyte B is in the water and is simulated using roughness only.

7.4.2 Results

The RMA-2 and RAM-11 model geometry and input files were modified to represent restoration configurations for the 1, 5, and 20 year scenarios. The models were then used to simulate flow velocities, water depths, and water temperatures. Initial water temperatures and surrounding meteorological conditions were simulated using the same period of data applied to the base case configuration: 12 August to 20 August, 2008 (meteorological conditions from this period are applied to all simulations). This period was chosen because the most complete data set was available against which to calibrate the model. Though maximum temperatures commonly occur during the last week of July through the first week of August in this area, periods adjacent to that time frame can also experience high temperatures and are suitable for calibration. Herein, results compare mean daily maximum water temperatures during 12 August to 18 August, 2008.

Once the simulations were completed, results for specific downstream locations were extracted to illustrate water temperatures along the longitudinal profile for each configuration. These downstream locations (measured by their distance from the confluence of Big Springs Creek with the Shasta River) are:

1. Big Springs Dam (River kilometer 3.6)
2. Upstream Busk Residence Bridge (R. km. 3.3)
3. Upstream Springs Alcove (R. km 3.1)
4. Downstream Springs Alcove (R. km 3.0)
5. Upstream Water Wheel (R. km 2.8)
6. Water Wheel (R. km. 2.6)
7. Upstream Irrigation Pipe (R. km. 2.1)
8. Lowest Drivable Bridge (R. km. 1.5)
9. Upstream Little Springs Creek (R. km 0.9)
10. Mouth of Big Springs Creek (R. km 0.1)

Multiple locations were selected to identify reaches which experience the highest heating rates and to capture areas where appreciable spring inflows occur. This approach allows restoration activities to be targeted where actions will provide the greatest benefit.

1 Year: Immediate Response

Simulation results under assumed conditions associated with letting the creek respond to one year of passive restoration activities suggest a decrease in peak water temperatures throughout the system (Figure 63). Differences under a low flow condition (5 ft^3/s) indicated a modest decrease of typically less than 1^oC. Under increased flows (15 ft^3/s) at Big Springs Dam, an additional decrease in temperature was realized throughout much of the creek. Rates of heating were highest in between the alcove springs and the waterwheel as a result of cold water rapidly seeking equilibrium temperature and channel geometry in this sub-reach (wide and shallow flows). Note that heating was minimal in all cases through the heavily shaded willow thicket below Big Springs Dam. The alcove springs (approximately 3.0 km (1.9 mi) upstream from the mouth) provided additional cool water inflows during this summer period, which lowered local water temperatures.

Once below the waterwheel the stream steadily heated in the downstream direction, but rates were lower with additional flows.

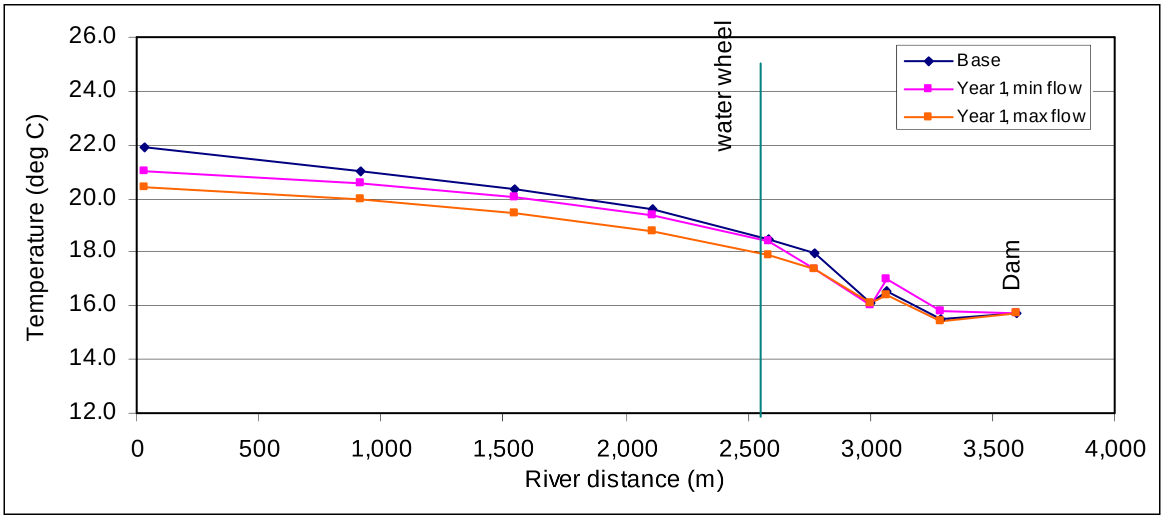


Figure 63. Year 1 restoration simulation results – longitudinal profile of mean daily maximum water temperature during August 12-18

5 Years: Intermediate Response

Simulation results under assumed conditions associated with letting the creek respond to five years of restoration activities suggest a decrease in peak water temperatures throughout the system (Figure 64). Under these conditions mean daily maximum temperatures for low flow reduced temperatures notably – particularly below the water wheel. For the high flow conditions, mean daily maximum temperatures did not exceed 20^{o}C. Overall differences under a low flow condition (5 ft^3/s) and increased flows (15 ft^3/s) were modest. Again, rates of heating were highest in between the alcove springs and the waterwheel as a result of cold water rapidly seeking equilibrium temperature and channel geometry in this sub-reach (wide and shallow flows).

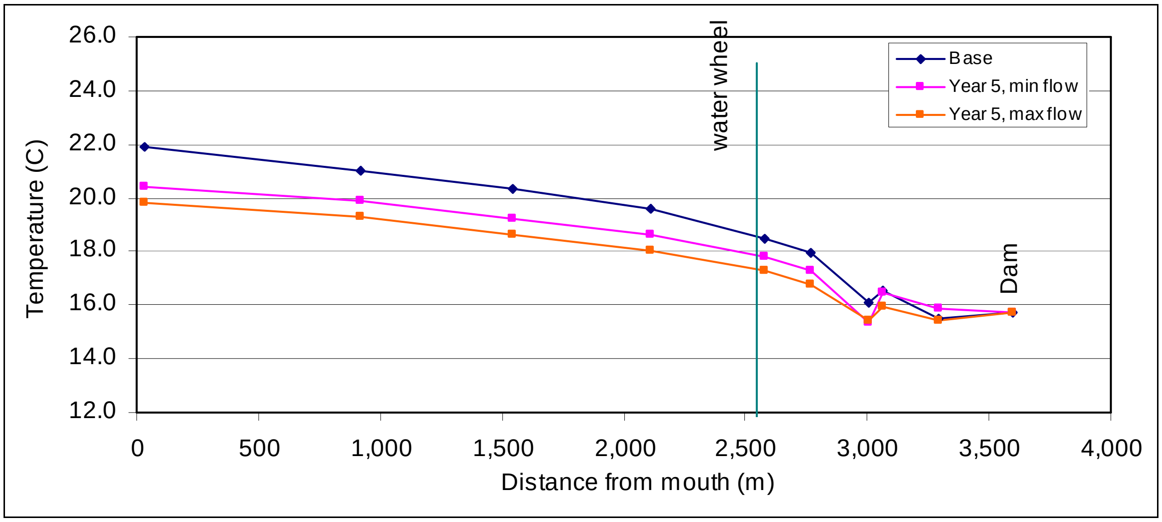


Figure 64. Year 5 restoration simulation results – longitudinal profile of mean daily maximum water temperature during August 12-18

A second 5-year simulation was run to examine the effect of removing the water wheel given minimum flows (5 ft^3/s). Results suggest that while water temperatures decrease locally, the effects of removing the water wheel do not propagate to the mouth of Big Springs Creek (Figure 65). Local water temperatures near the water wheel decrease approximately 0.5^oC when the structure is removed. By the time the water reaches the mouth, cooling effects from the water wheel's removal are negligible. Simulating increased flows from Big Springs Dam (15 ft^3/s) yielding similar results.

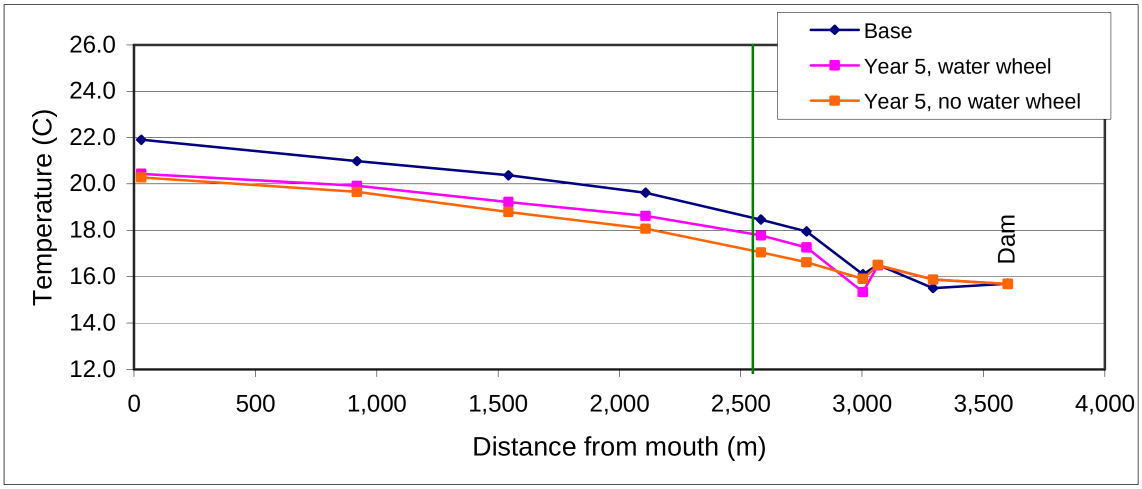


Figure 65. Year 5 restoration simulation results illustrating the removal of the water wheel – longitudinal profile of mean daily maximum water temperature during August 12-18

20 Years: Long-term Response

Simulation results under assumed conditions associated with letting the creek respond to twenty years of restoration activities suggest a decrease in peak water temperatures throughout the system (Figure 66). Under these conditions mean daily maximum temperatures for low flow reduced temperatures notably throughout the creek. Overall differences between a low flow condition (5 ft^3/s) and increased flows (15 ft^3/s) were modest, but notable. An important aspect of this simulation is that the slope of the longitudinal temperature trace for the 20 year conditions (low or high flow) is less than that for the base case, indicating that restoration prescriptions are having a positive benefit throughout the system. Additional narrowing and shading in the upper-most reach yields marked benefit in the uppermost 1,000 meters of the creek – mean daily maximum temperatures are on the order of 15°C.

Rates of heating were highest in between the alcove springs and the waterwheel as a result of cold water rapidly seeking equilibrium temperature and channel geometry in this sub-reach (wide and shallow flows). However, removing the water wheel had negligible effects both locally and at the mouth of Big Springs Creek (Figure 67).

Additional sensitivity testing was completed with the model for further width reductions. Preliminary results suggested that mean daily maximum temperatures on the order of 18°C were achievable – approximately a 4°C reduction over existing condition. Mean daily maximum temperatures would be below 15°C and mean daily average temperatures would be approximately 12.5°C (data not shown). These conditions would extend into the Shasta River for a considerable distance downstream of the confluence with Big Springs Creek, providing additional benefits to and expanding the available cool water habitat for coho salmon and other anadromous fish species.

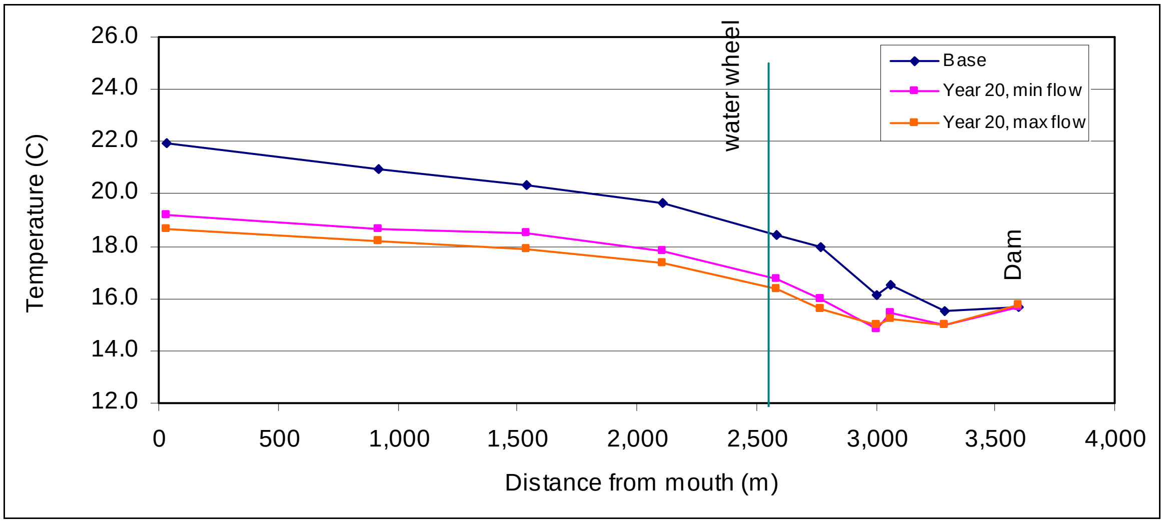


Figure 66. Year 20 restoration simulation results – longitudinal profile of mean daily maximum water temperature during August 12-18

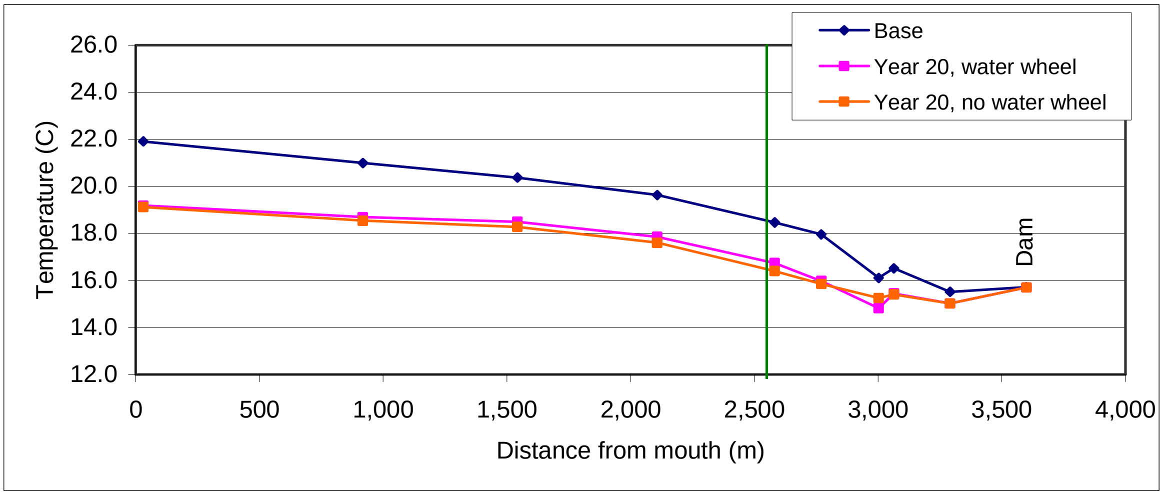


Figure 67. Year 20 restoration simulation results illustrating the effect of removing the water wheel – longitudinal profile of mean daily maximum water temperature during August 12-18

7.4.3 Recommendations for future model applications

These results reflect a modest set of potential restoration actions that may affect water temperature management at Big Springs Creek and in the Shasta River downstream of Big Springs Creek. However, these actions are primarily passive and do not explore the effects of more active approaches. These initial findings are intended to prompt TNC and other stakeholders and regulatory agencies to envision a broader range of restoration prescriptions for testing in the model.

To better guide the implementation of active restoration options, specific improvements should be made to the model and restoration configurations. These improvements are:

- Assessing long-term potential for stream narrowing given the geomorphology of the creek and ability of aquatic vegetation to trap fine sediment; and sediment supply conditions in the creek.

- Identifying the potential for woody riparian vegetation to be used as a temperature control approach in Big Springs Creek. This work would include identifying limiting factors to riparian vegetation colonizing the various sub-reaches of the creek (e.g., bedrock), preferred species (leaf out timing considerations), desired location of woody vegetation, and the time to establish and grow trees.

- Identifying the role of herbaceous riparian vegetation in Big Springs Creek. The approach identified herein explicitly considered succession of one sort (in channel aquatic vegetation, cattail/bulrush, and ultimately large woody vegetation). There may be areas where wetlands are desirable from bank stabilization perspective, as a desired habitat component, and/or providing shading benefits (modest).

- Explicitly identify land and water resource management options to support or refute the minimum and maximum flow conditions employed herein. This also relates to return flow management.

- Considering real time operations of waters on Big Springs Ranch to ameliorate adverse thermal conditions. The modeling effort has focused on one week in early August, but more extreme thermal conditions have certainly occurred. Water resources management in the form of reducing return flows and increasing creek flows based on short-term forecasts could benefit coho salmon and other temperature sensitive fishes and accelerate the pace of recovery and support long-term maintenance of these species.

- Further exploring the removal of the waterwheel. Identification of potential restoration options above this structure will assist in representing this activity in the model, which in turn will provide insight on appropriate timing for removal.

- Based on the sediment characteristics in the system, consider modeling sediment to manage sand and other fine material during restoration activities.

- Given the unique water quality conditions of the spring sources – elevated forms of inorganic nitrogen and phosphorus – consider expanding the model to include other water quality parameters that would lend insight into food web dynamics and assist in TMDL implementation activities in the Shasta River below the Big Springs Creek confluence.

By including these restoration options and simulation capabilities in the hydrodynamic and temperature models, we can improve our understanding of the effects of each restoration action and use funds and resources more effectively and efficiently.

7.5 *Conclusions*

Reducing water temperatures is a key component to improving rearing habitat for juvenile salmonids. Both passive and active restoration activities result in lower water temperatures throughout Big Springs Creek. Within the first year of restoration, peak water temperatures in Big Springs Creek may decrease by nearly 2^{o}C in the lower portions of the creek provided livestock exclusion and return flow management is implemented. By year 20, peak water temperatures are estimated to decrease by 4^{o}C at the mouth and show marked improvements throughout the creek. Examining the low and high flow conditions suggests that during times when forecasted conditions are likely to lead to increased thermal loading, increased discharge volume from Big Springs Dam may reduce peak temperatures by just under 1^{o}C. Removing the water wheel will have little effect on temperatures at the mouth of the creek; over time the water quality benefits of removing the water where are negligible. Overall, simulations of long-term restoration suggest that much of Big Springs Creek will experience thermal benefit from actions taken on Big Springs Ranch and other TNC lands. These benefits will extend into the Shasta River, expanding potential habitat for coho salmon and other salmonid species.

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9.0 Appendix: Hydrodynamic and Water Temperature Modeling

9.1 *Flow and Temperature Modeling*

One element of the Big Springs Creek study was the development of a flow and water temperature model to characterize existing conditions and assess and prioritize potential future restoration actions. Current conditions in the study area have been heavily impacted through land and water use activities.

9.1.1 Purpose

Though the relative effects of existing water management and grazing practices are broadly understood, little data exists that quantifies these effects on Big Springs Creek. Historically, neither discharge nor groundwater pumping data was recorded, nor were there even general records of the timing, location or volume of irrigation diversions or return flows. This critical lack of data prevented any quantified assessment of the effects of the different management practices on water temperatures and challenged cold water management plans to evaluate the benefits of alternative solutions. To overcome this paucity of data a monitoring program and associated modeling project were identified as a means to assess potential cold water management alternatives.

Specifically, a two-dimensional numerical model was developed to describe temperature conditions in Big Springs Creek and quantify the benefits of different restoration alternatives (described later in this document). Such a model allows planners to simulate different water management and irrigation strategies and examine their effects on water temperatures. Instream grazing effects can be simulated by altering roughness and shading factors in different areas. Similarly, the impacts of reduced solar radiation due to riparian shading on water temperature can be simulated. Alternative channel geometries can also be tested to examine their impacts on water temperatures. By using a model to gain a better understanding of current conditions, specific prescriptions can be tested to develop a cold water management plan. The model assessments help identify high priority actions and define an effective restoration strategy, resulting in an efficient use of resources and funds.

9.1.2 Methods

The steps to develop a model that accurately represented Big Springs Creek were as follows:

- Conceptualization
- Data assembly/organization
- Implementation
- Calibration
- Production

Conceptualization helped identify what kind of model would best represent the system. Data assembly and organization helped refine the conceptualization and define baseline conditions over a period of several months. Implementation involved creating a running model of the system. After a running model was created, it was calibrated by using field data to test its accuracy. Finally, once the model was adjusted to accurately simulate observed baseline conditions, assessment of restoration alternatives was completed.

9.1.3 Conceptualization

Before the model was constructed, a conceptual understanding of Big Springs Creek was developed to identify the type of model that would best represent the system. The conceptual understanding was achieved by gathering preliminary data of Big Springs Creek's geometry, hydrology, water temperature and meteorological conditions. Understanding the general characteristics of the creek helped identify appropriate model characteristics. For example, early temperature data suggested that the model should simulate temperature changes in two dimensions: transect and longitudinal changes (i.e., x and y directions). While lateral and longitudinal thermal variability existed, the wide and shallow geometry of the creek yielded well-mixed, uniform vertical (i.e., z-direction) temperatures in the water column. Therefore, a two-dimensional model would simulate appropriate thermal complexity.

A suite of modeling software, RMA-2 for hydrodynamics (v8.1(a)) and RMA-11 (v8.1(b)) for water temperature, was selected to represent Big Springs Creek as a two-dimensional, depth-averaged, finite element model. RMAGEN (v7.3(g)) was used to create a geometry file of Big Springs Creek that was used by both the hydrodynamic and water temperature models. RMA-2 is a two-dimensional, finite element, depth-averaged numerical model that calculates velocity, water surface elevation and depth at defined nodes on the boundary of each grid element in the geometry file. RMA-11 is a finite element water quality model that uses the depth and velocity results from RMA-2 to solve advection diffusion constituent transport equations. Details of each of these applications are provided below.

RMAGEN

RMAGEN is a preprocessor program, used to construct the numerical mesh used in RMA-2 and RMA-11. RMAGEN assigns spatial information to each node within the mesh (x-y location and elevation), interpolating values from the topographic description. The mesh consists of triangular and polygon elements of variable size and configuration. A triangular element consists of six nodes – three at the vertices and three mid-side nodes. Similarly, polygon elements consist of eight nodes – one node at each corner and one node at the midpoint of each side.

RMA-2 Model

RMA-2 is a two-dimensional, depth-averaged, finite element hydrodynamic numerical model. It computes water surface elevations and horizontal velocity components for subcritical, free-surface, two-dimensional flow fields. The model computes a finite element solution of the Reynolds form of the Navier-Stokes equations for turbulent flows. Friction is calculated with the Manning's or Chezy equation, and eddy viscosity coefficients are used to define turbulence characteristics. Both steady and unsteady (dynamic) problems can be analyzed. RMA-2 is a general-purpose model designed for far-field problems in which vertical accelerations are negligible and velocity vectors generally point in the same direction over the entire depth of the water column at any instant of time.

RMA-2 has been applied to calculate water levels and flow distribution around islands; flow at bridges having one or more relief openings, in contracting and expanding reaches, into and out of off-channel hydropower plants, at river junctions, and into and out of pumping plant channels; circulation and transport in water bodies with wetlands; and general water levels and flow patterns in rivers, reservoirs, and estuaries. For complete details about RMA-2 see King (2008).

RMA-11 Model

RMA-11 is a finite element water quality model capable of simulating one and two-dimensional approximations to systems either separately or in combined form. It is designed to accept input of velocities and depths, either from an ASCII data file or from binary results files produced by the two-dimensional hydrodynamic model, RMA-2. Results in the form of velocities and depth from the hydrodynamic models are used in the solution of the advection diffusion constituent transport equations. Additional terms for each constituent represent source or sinks and growth or decay.

The model solves the advection diffusion equations include sources, sinks, and reactions. The governing transport equations may be integrated over the vertical dimension with the assumption that C is independent of elevation (z).

In RMA-11 the dependent variable modeled when simulating heat transport is temperature, T (^{o}C). (The truly consistent parameter should be concentration of stored heat or heat content of water, $C_{H,}$ which has units of kJ/m^3.) The approach used in RMA-11 consistent with QUAL2E and other literature and is to assume that heat is transferred from various energy sources. So that:

$H_N = H_{SN} + H_{AN} - (H_B + H_E + H_C)$

where

H_{SN} = Net short-wave influx, (kJ/m^2/hr)

H_{AN} = Net long-wave influx, (kJ/m^2/hr)

H_B = Long-wave back radiation, (kJ/m^2/hr)

H_E = Conductive flux, (kJ/m^2/hr)

H_C = Evaporative flux, (kJ/m^2/hr)

For comprehensive details about RMA-11 see King (2008).

9.1.4 Data overview

Field data describing Big Springs Creek's geometry, hydrology, water temperature and meteorology were required to

- build a conceptual model and the two-dimensional model;
- describe baseline conditions in Big Springs Creek; and
- provide a measure of accuracy with which to test the model.

Data were available from a previous project (started in March 2008) and augmented with data from this project. Data include information describing the longitudinal and cross-sectional profile of the river, instream temperature observations, and flow velocities. At that time, stage gages and data loggers were also deployed to monitor flow and water temperature conditions. Currently, flow and water temperature data are still being collected as part of ongoing monitoring. As well as instream data loggers, infrared aerial imagery was used to describe water temperature conditions in Big Springs Creek in 2003 (NCRWQCB, 2004) and 2008 (Watershed Sciences, 2009). Meteorological data was gathered using the California Department of Forestry's gage at Weed Airport. Details about data gathering methods are provided below.

9.1.5 Geometry

Longitudinal and cross-sectional bathymetry data were gathered by surveying the length of the creek to define the shoreline and specific cross sections, noting relative network coordinates and elevation of each point in x-, y-, z-coordinates. Data describing 2,448 points were recorded along the 3.5 km (2.2 mi) creek reach, including measurements of 63 cross-sections (see Geomorphology section for more details). Survey data were gathered using a TOPCON HiperLite Plus Real-Time Kinematic (RTK) survey unit. The RTK survey unit is accurate up to 0.01-0.02 m (0.3-0.7 ft).

9.1.6 Hydrology

In July 2008, velocity measurements were made across 11 transects. Point velocities were measured at 0.6 of the stream depth using a Marsh-McBirney Flo-Mate electromagnetic velocity meter attached to a top-set wading rod. Measurements were not made at regular intervals, but rather at points where the creek cross-section changed (e.g., edges of vegetation, noticeable flow paths, etc.). A total of 201 point velocity and stage measurements were made. The number of measurements at each site ranged from 10 to 32, depending on the width of the channel. Additional stage data was gathered at five locations and used to generate stage-discharge rating curves (see Hydrology section for more details).

9.1.7 Temperature

Forty nine data loggers were deployed to gather water temperature data along the longitudinal profile of Big Springs Creek as well as points upstream and downstream of Big Springs Creek's confluence with the Shasta River. Data was downloaded from loggers three times throughout the study. Cross-sectional water depth and temperature data were also gathered at several sites during August, 2008. Some loggers were not recovered due to removal or vegetative growth (i.e., lost). Currently, loggers are still deployed to collect longitudinal temperature data.

In June 2008, three transects in the willow thicket below the lake outlet were sampled to monitor thermal diversity. Locations of transects were chosen for representative places to capture margin warming, local temperature differences, and riparian shading. Cross-sectional temperature data was recorded with simultaneous measurements of water depth. Water depth was measured with a Global Water pressure transducer (model WL 16) accurate to +/-0.2%. A Tech Instrumentation model TM99A temperature unit with a model 2007 probe was used for most handheld temperature sampling. The TM99A temperature unit is accurate to +/- 0.1°C in the 0-40°C range. The pressure transducer and TM99A temperature unit were mounted to Plexiglas on a 1.8 m (6 ft) rod. Probe tips were attached to the end of the rod, and the rod was marked at 0.3 m (1 ft) increments. Temperature and depth measurements could then be taken simultaneously in water up to 1.5 m (5 ft). The handheld device allowed quick assessment of vertical distribution of water and bed temperature, with the ability to explore areas under overhanging vegetation, cutbanks, and other types of cover elements.

In July, thermistors were deployed in 11 transects along the river corridor to monitor temperature over a three-week period. Water temperature was recorded using HOBO Water Temp Pro and Hobo Water Temp Pro V2 thermistors, manufactured by Onset Computer Corporation (Onset, 2007). These devices are accurate to +/- 0.2°C in the –20-50°C range (temperatures typically experienced on Big Springs Creek, Little Springs Creek and the Shasta River fall within this range). All devices were launched prior to deployment using a computer with HOBOware software.

Finally, aerial thermal infrared radiometer (TIR) imagery provided a longitudinal profile of water temperatures in Big Springs Creek as well as temperature estimates for individual springs. This imaging was collected using a FLIR System SC660 TIR sensor, which is accurate within +/-0.02^{o}C, and a high resolution camera. Three flights occurred: one was flown in 2003, which provided a longitudinal profile of water temperatures. The second and third were flown on 07/16/2008 (evening) and 07/17/2008 (morning) to compare diurnal water temperature trends and estimate the water temperatures of individual spring inflows.

9.1.8 Meteorology

To model water temperatures in Big Springs Creek, RMA-11 required information about local meteorological conditions for the following categories:

- Atmospheric dust attenuation
- Cloudiness
- Dry bulb temperature (^{o}C)
- Dewpoint temperature (^{o}C)
- Atmospheric pressure (mb)
- Wind speed (m/s)
- Wind direction (radians from x-axis)
- Solar radiation (W/m^2)

These data were required at sub-daily (e.g. hourly) frequency. Because this study focused on water temperatures that benefit coho salmon during summer rearing, data extending from June through August was necessary. Though one goal for extending this study was to include other seasons, currently that goal is outside the scope of work. Therefore, while data was compiled for the entire 2008 water year (1 October, 2007 – 30 September, 2008), ensuring data quality for the summer months was a priority. By employing meteorological data for that period, the temperature model provided water temperature results that could be compared to water temperatures gathered by field work for that same period, i.e., calibration.

Meteorological data for water year 2008 were used because this period brackets available flow and water temperature data in Big Springs Creek. Data from the Weed Airport were used in the modeling study. The meteorological station is located approximately nine miles south of Big Springs Creek. Data were downloaded from the California Data Exchange Center (CDEC) website.

The Weed Airport weather station included data for dry bulb temperature, relative humidity, wind speed, wind direction and solar radiation. The data records were complete except for several missing hourly measurements (and, in two cases, two hours). The data gaps were identical for all meteorological categories. Missing data were determined using linear interpolation between measurements for the hour before and after.

Cloudiness and dewpoint temperature data were calculated based on solar radiation and relative humidity data downloaded from CDEC. Cloudiness was estimated using the daily maximum solar radiation at Weed Airport and the sine curve function (Figure 68). A sine wave function was developed to simulate the maximum possible solar radiation for a given time. The difference between the observed maximum radiation and the possible maximum radiation provided an estimate of the cloud cover for that day. Because there is some subjectivity in fitting a sine wave to the meteorological data, a sensitivity analysis was performed with regards to cloud cover. The results were nearly identical to simulations made using the sine wave function illustrated in Figure 68. Fires in the region during the summer of 2008 would have a direct impact on solar radiation, i.e., reduction. This is captured in the process of fitting a theoretical sine function to the data and calculating percent of maximum theoretical solar radiation.

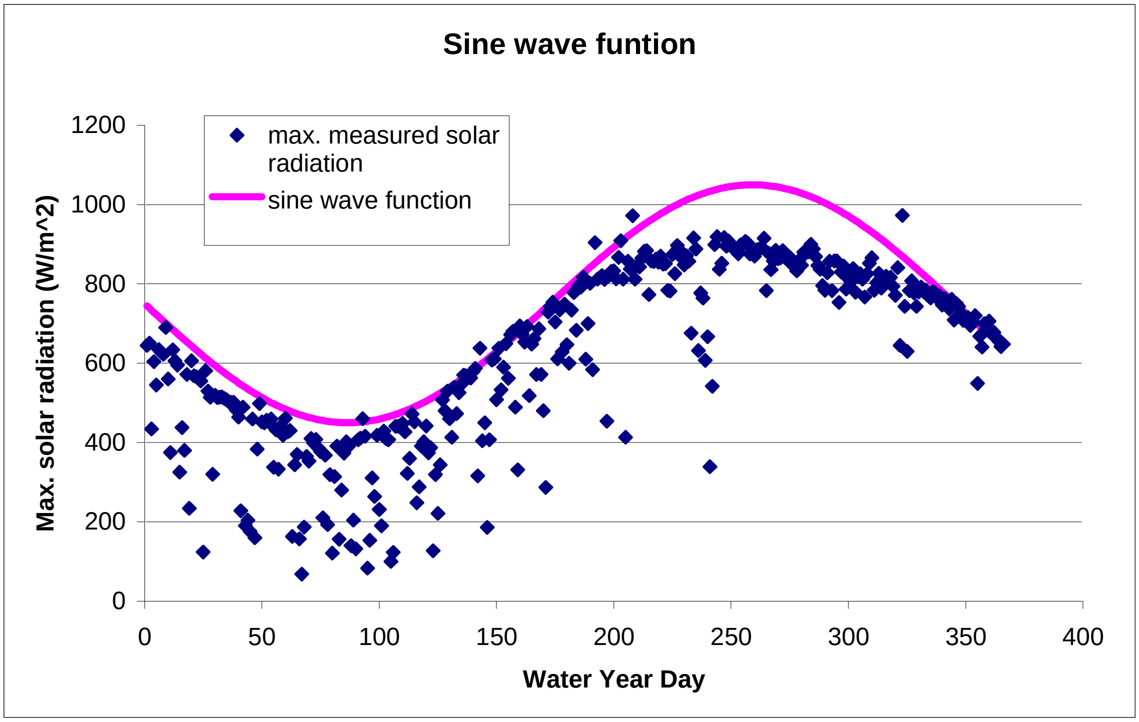

Figure 68. Sine wave function used to estimate cloud cover. Water year begins on 1 October and ends on 30 September.

Dewpoint temperatures were calculated based on relative humidity measurements made at Weed Airport. The relative humidity data contained the same data gaps as the other data downloaded from CDEC for WY 2008. These gaps were also resolved using linear interpolation. Another data gap existed for the entire last day of WY 2008. This gap was filled using the average relative humidity for the past three days and following two days. As this study focuses on the period from May through August, the missing data at the end of September was not critical.

Wind speed was converted to m/s and used in the temperature model. Wind direction is predominately used for wind set-up in water bodies, which can affect flow patterns. Such conditions were not modeled at Big Springs Creek.

Atmospheric pressure was calculated based on elevation. Though the Weed Airport gage also records atmospheric pressure, data were only available until 31 July, 2007, which is outside the identified modeling period. However, since the data during the two-year measured period does not vary more than 0.2 in., we assume that pressure is constant – a reasonable assumption during summer periods. A sensitivity analysis was performed using the atmospheric pressure data at Brazie Ranch (approximately 200 m higher than Big Springs Creek). The results were within 4 percent of results that used Weed Airport atmospheric pressures.

9.1.9 Model Implementation

Once field data was gathered and processed, the two-dimensional model was implemented. Model implementation followed several steps. First, a geometry file describing the bathymetry of the creek was developed. Next, flow data was added to the model and tested. After a flow simulation was successfully completed, temperature data were added to the model, and tested again. Once the model was successfully ran a flow and temperature simulation, it was calibrated (described in more detail below).

The first step in building the temperature model was to construct a grid that replicated the bathymetry of the creek. The profile, water edge, and cross section data were used to construct a contour map of the stream bed using Surfer Version 8 software (Figure 69). The contour map was generated using 1 m^2 (11 ft^2) resolution and defined contours for every 0.5 m (1.6 ft) change in elevation. This contour map was imported to RMAGEN (v7.3(g), which was used to construct a two-dimensional description of the creek bed that was subsequently used to simulate flow (RMA-2) and water temperature (RMA-11) conditions in Big Springs Creek (Figure 70).

The model grid was created using polygons and triangle elements, to describe the shape of the river. The size of each element was determined based on the amount of detail that needs to be described in each area to accurately represent the creek's characteristics, while balancing the computational requirements of the model. Big Springs Creek was represented through much of the stream using five elements to describe the cross-section of the river. In areas where less resolution is required (for example, reaches that are wide, shallow and flat, with uniform bed substrate), element dimensions were larger than in areas where more detail was needed to describe the creek. This level of detail represents the creek with sufficient accuracy for uniform sections, while those areas with more complexity are represented with considerably greater spatial detail. Finer resolution was used to describe features such as spring inputs, confluences, channel constrictions and islands.

Streambed characteristics were assigned to each element. Manning roughness, shading, and other attributes could thus vary on an element-by-element basis. The model includes six different types of elements. Each element type is identified based on a unique roughness factor. The elements types are listed below:

1 – clear channel, sandy bottom
2 – clear channel, gravel bottom
3 – macrophyte vegetation: this includes aquatic plants that extend through part or the entire water column
4 – willow: water can flow through part of the tree structure (i.e. exposed roots, overhanging limbs)
5 – bedrock
6 – rock berm in spring alcove

Solid, impermeable structures, such as bridge pilings and islands, are represented by creating blank spaces in the grid structure. The model automatically routes water around these blank areas.

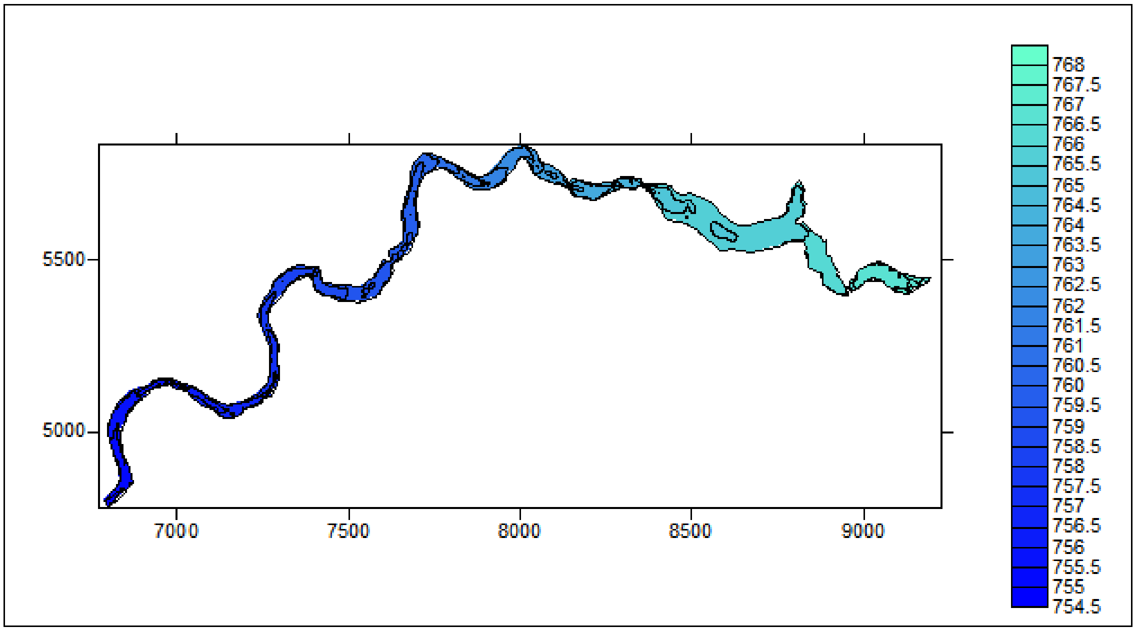

Figure 69. Big Springs Creek contour map developed using Surfer. The axes indicated truncated UTM coordinate locations.

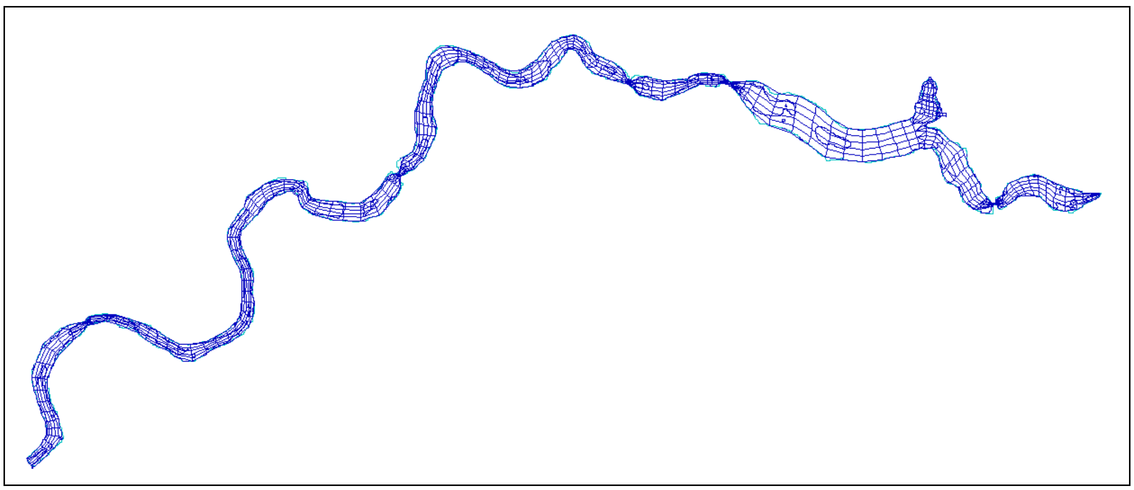

Figure 70. The grid created in RMAGEN is laid over the contour map created in Surfer. This grid is the basis of the two-dimensional flow and temperature simulations made by RMA-2 and RMA-11, respectively.

Once the grid describing the channel geometry was created in RMAGEN, flow data was simulated using RMA-2. In this model, flow sources simulate discharge from the Big Springs Lake outlet and flow contributions from spring inputs. Discharge volumes are estimated based on stage-discharge relationship data. Locations of spring inputs are based on observations made in the field. Discharge volumes that the springs contribute are made using mass balance estimates based on stage data. Flow simulations are successful if the model produces reasonable values for water depths and velocities. Refinements to the grid were made in certain cases based on discussion with field crews, field observations, and aerial photos. Return flows were not quantified in 2008 and are not explicitly included in the model.

Upon completion of the flow model, the temperature model was employed. Temperature boundary conditions were assigned to all inflows, and appropriate meteorological conditions defined. In the case of surface water inputs, such as inflows from the Big Springs Lake outlet, a time series of temperatures were added to define an inflow temperature boundary condition. The source of this temperature time series is observed data from temperature loggers placed longitudinally in Big Springs Creek. For ground water inputs, such as the spring sources, temperatures are assumed to be constant. Estimates of individual spring inputs are based on the FLIR report (Watershed Sciences, 2008).

9.1.10 Model Calibration

Once the model was successfully implemented, it was calibrated against field observations. Calibration occurred between 12 August, 2008 and 20 August, 2008 when sufficient flow, stage, and water temperature data were available. Flow calibration was assessed by comparing model inflows to outflows. Simulated flows were reproduced to within 1 percent of measured flows. Depths throughout the system were shallow in observed data (often on the order of 0.2 to 0.5 meters). Simulated depths were within in this range; however, direct comparisons were difficult because the channel was modified by aquatic vegetation through the study period (access issues resulted in field surveys being separated by more than a month). Two criteria were determined to evaluate the simulated water temperature. First, calculated water temperature should be within approximately 1^{o}C of observed conditions in the main channel areas of the creek. Second, the model should reproduce the diurnal phase within one hour of the observed signal.

Calibration was tested by comparing observed temperature data from the data loggers to calculated water temperature results at nodes in similar locations on the model grid. This method established 37 points of calibration and assessed accuracy along the longitudinal and lateral profiles. The 37 points of calibration comprise 10 longitudinal locations, with two to five points at each cross-section (the number of points at each cross-section depends on the number of data loggers recovered from those locations).

Currently, the temperature model performs well throughout much of the reach. Nodes near the Big Springs Lake outlet and the spring alcove simulate temperatures within 1.0^{o}C. The model also performs well simulating the longitudinal temperature profile in Big Springs Creek. The model simulates temperatures within 1^{o}C of the observed data for the longitudinal cross sections; however, at certain cross section points deviations are greater. The longer residence time in off channel areas of certain sections and the variable vegetation growth in the channel confounded field observations. Overall, the tool has been calibrated consistent with the identified purpose of making preliminary assessments of restoration actions. Temperature calibration figures for multiple locations (included at the end of this appendix) indicate that simulated temperatures reproduce diurnal variations effectively, reproducing amplitude at most locations. Phase is shifted at locations from the vicinity of the alcove springs and downstream locations. Model testing indicates that this shift is most likely due to a combination of factors, the most important of which is allocating inflow quantities to the various discrete spring locations and diffuse sources. Currently, only the bulk accretions have been quantified between the dam and the Busk Residence bridge and from this bridge down to the water wheel. Nonetheless, the model effectively represents longitudinal heating, diurnal range/amplitude, and approximate timing. A comparison of simulated and observed longitudinal profile of mean weekly average, minimum, and maximum daily temperatures shows that the model reproduces these conditions closely throughout the reach (Figure 71).

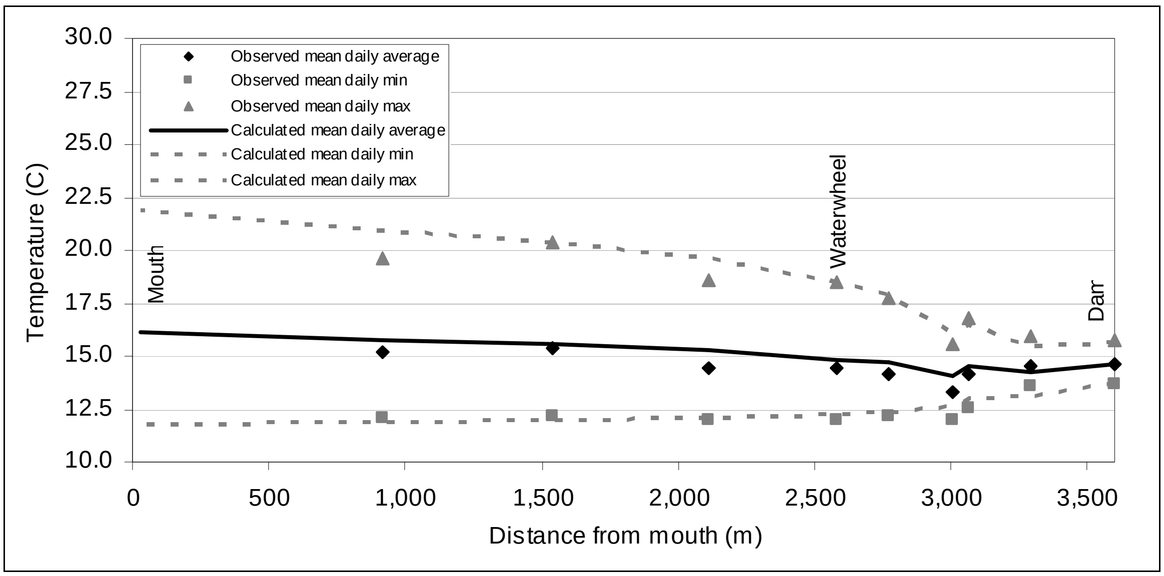

Figure 71. Longitudinal profile of simulated and observed mean weekly average, maximum, and minimum daily water temperatures for Big Springs Creek: August 12-18, 2008.

Also included in the calibration plots are several cross section locations for the wide reach above the waterwheel. Examination of these plots shows that the model deviates in these off channel areas. Additional information (e.g., geometry and vegetation characteristics) on non-thalweg areas of the channel will improve these results.

Additional detailed observations of geomorphology; flow, velocity, and depth; water temperature; vegetation growth; and associated data would improve temperature model performance. For example, the location of many diffuse springs between Big Springs Dam and the alcove spring are incompletely characterized in location and quantity at this time. The result is that simulated diurnal phase is inconsistent with field observations.

Also, more comprehensive representation of aquatic vegetation growth (and subsequent capturing of fine sediment) would assist in representing lateral variability in simulated creek temperatures. Stream edges are not readily comparable because vegetation dominated these areas and was not comprehensively mapped for inclusion into the model. Finally, effective monitoring of return flow location, timing, quantity, and temperature are required to fully develop the predictive value of the numerical model. Overall, the model has a wide range of capabilities to accommodate these features at small spatial and temporal scales. Future data collection is recommended to support the model and improve system representation. Specific recommendations on how to improve the predictive capabilities of the model include:

- More comprehensive mapping of the dynamic nature of the channel morphology in response to seasonal vegetation growth. This will change from current conditions when grazing was allowed directly within the creek to a condition where cattle exclusion is practiced. Multiple years of surveying will be required to determine if the creek attains a dynamic equilibrium within a single year or over multiple years.

- Identification of diffuse spring flows upstream of the waterwheel. There are a notable amount of accretions that are manifest as non-point inputs of groundwater (e.g., areas of boiling sands). Identifying these areas with detailed handheld measurements and associated water temperatures will aid in proper representation within the model.

- Quantification of individual spring sources. Although temperature at discrete spring sources of the complex are well defined, the individual flows are largely unquantified. The project team has identified approaches to quantifying these springs and completion of this flow and temperature work will assist in understanding of diversity (or similarity) among springs as well as support modeling applications

- Long term monitoring to assess thermal diversity throughout the creek will be useful to assessing habitat conditions and support future modeling. Approximately 50 temperature devices were deployed in 2008 prior to maximum vegetation growth. Many of these were overtaken with vegetation during the deployment. Continued work, with an understanding of system dynamics, will yield valuable information on how thermal habitats can change through time and support modeling applications.

- Examination of the TIR data indicates considerable thermal diversity downstream of the confluence with the Shasta River. Extension of the two-dimensional model downstream to the vicinity of the Nelson Ranch will provide a means to assess the influences of Big Springs Creek on the Shasta River and include several minor springs that enter the Shasta River downstream of the confluence.

- RMA-11 is a full water quality model and can simulate sediment transport, nutrient dynamics, benthic algae dynamics, dissolved oxygen, organic matter, and other constituents. Given the complex water quality conditions present in the creek and Shasta River, and the dynamic nature of the system, it is recommended that modeling of other constituents be explored to assess nutrient fate and transport and determine potential downstream impacts of spring flows.

9.2 *Conclusion*

Maintaining cool water temperatures from the Big Springs Creek springs downstream into the Shasta River is a key component to restoring juvenile salmonid rearing habitat. Restoration alternatives will be evaluated based on their ability to accomplish this goal. The Big Springs Creek hydrodynamic model is a valuable tool that can be used to assess restoration alternatives. These assessments will help direct funds and resources efficiently throughout the restoration process. While the Big Springs Creek two-dimensional temperature model is ready to make preliminary assessments of restoration options, additional data would improve this tool's performance. Recommendations to improve the hydrodynamic model are included in Section 11.2.

10.0 Appendix: Review of the Shasta River TMDL Analysis – Big Springs Flow and Temperature Boundary Condition Assumptions

10.1 *Introduction*

The Shasta River TMDL report was developed in 2006 to evaluate the water quality components of the Shasta River including the effects of flow and temperature contributions from Big Springs Creek. When the TMDL was developed, data measurements of flow and water temperature at the mouth of Big Springs Creek were unavailable. Therefore, to simulate the effect of the creek on the mainstem, assumptions were made about its flow and temperature contributions and used to run a one-dimensional hydrodynamic model of the Shasta River. Current work that assesses the baseline physical and ecological conditions on Big Springs Creek provides the opportunity to reevaluate the assumptions made concerning the Big Springs Creek flow and temperature boundary conditions and recommend additional monitoring for future TMDLs. For detailed discussion of the TMDL boundary conditions assumption see (NCRWQCB 2006).

10.2 *Overview of the Shasta River TMDL Assumptions*

To assess the effect of Big Springs Creek on the flow and water temperature characteristics of the Shasta River, data describing these water quality characteristics was required. However, access to the property where the mouth of Big Springs Creek is located was unavailable during the TMDL analysis. Therefore, estimates or assumptions about the flow and thermal contributions that the creek made to the river were necessary.

10.2.1 Flow

Flow assumptions estimated the volume of water discharged from Big Springs Creek into the Shasta River (details of the flow assumptions in the TMDL are provided in (NCRWQCB 2006), Appendix G). Due to access restrictions, flow measurements at the mouth of Big Springs Creek were not available for the 2006 Shasta River TMDL. Instead, flows were estimated based on information from documented water rights to Big Springs Lake and Little Springs Creek, historical water master reports of irrigation diversions, and data gathered by the California Department of Public Works in 1922 and 1923. While the Department of Public Works had access to the mouth of Big Springs Creek during its study, flow measurements were limited due to extensive vegetation growth. Flow contributions from the creek were estimated based on flow measurements taken on the Shasta River directly upstream and downstream of the creek mouth. The TMDL assumed that flow from Big Springs Creek was approximately 60 ft^3/s during the irrigation season and 100-125 ft^3/s during the non-irrigation season.

10.2.2 Water temperature

Water temperature assumptions estimated both the current and future thermal conditions in Big Springs Creek (details of the flow assumptions in the TMDL are provided in NCRWQCB 2006, Chapter 6 and Appendix E). Again, due to access limitations, water temperature data was not available for the 2006 Shasta River TMDL. Instead, temperatures were estimated using data collected from the GID intake facility on the Shasta River, approximately 4.5 km (2.8 mi) downstream from the mouth of Big Springs Creek. Average daily water temperatures were assumed to be 17^{o}C. Calibration of the model illustrated that using water temperatures measured at the GID intake as water temperatures from Big Springs Creek reproduced a reasonable simulation of observed conditions.

A second water temperature assumption was made to describe the thermal conditions in a restored Big Springs Creek. Under restored conditions, average water temperatures were assumed to decrease by 4^{o}C, resulting in average temperatures of 13^{o}C. This estimate was based on water temperature data from the Big Springs source (10^{o}C-11^{o}C) and the estimated rate of heating between the source and mouth of the creek given unspecified restoration activities.

10.3 *Assessment of Shasta TMDL Assumptions*

Data gathered during the Big Springs Creek baseline monitoring study provides an opportunity to reevaluate the flow and water temperature assumptions made concerning the Big Springs Creek boundary conditions during the Shasta River TMDL study. Data describing the flow and water temperature characteristics of the creek from top to bottom, including variations during the irrigation season, were recorded during the baseline assessment. While the TMDL assumptions describing flow contributions to the Shasta River were generally accurate, those made to describe water temperature characteristics in Big Springs Creek deviated from conditions identified in 2008. Descriptions of the flow and water temperature findings are presented in sections 4.1 and 4.2, respectively.

10.3.1 Flow

Flow conditions were consistent with TMDL assumptions, primarily because previously available flow data generally were representative of flow in the Shasta River above and below the Big Springs Creek confluence. Flows during the 2008 irrigation season (1 April 2008 through 1 September 2008) averaged 54 ft^3/s; after the irrigation season ended, flows averaged 82 ft^3/s with some flows increasing to 100 ft^3/s. These volumes are comparable to the estimates used for the 2006 TMDL described above.

10.3.2 Temperature

Several assumptions were made regarding water temperature in the TMDL. First, the Big Springs Creek boundary condition in the hydrodynamic model was defined using data measured at the GID intake structure – approximately 4.5 km (2.8 mi) downstream of Big Springs Creek. Second, average water temperatures under a restored condition were assumed to decrease by 4°C. based on 2008 observations, these assumptions appear optimistic.

After collecting data during the 2008 field season at the mouth of Big Springs Creek, differences between that location and the Shasta River at the GID intake were apparent. The baseline assessment study determined that during late summer/early fall (the period simulated in the TMDL), average temperatures at the mouth of Big Springs Creek were approximately 21°C – not 17°C as was previously assumed. Given the thermal characteristics of Big Springs Creek's headwaters, the relatively short distance from the source to the mouth, and the calibration results that closely simulated the observed signal in the Shasta River downstream of Big Springs Creek, such a heating rate between Big Springs Creek's source and mouth was not anticipated during the TMDL study.

These findings suggest that Big Springs Creek imposes a thermal signal on the Shasta River that are often distinct from the trends observed at the GID intake. Further, this signal is possibly modified by land and water use on the Big Springs Ranch as illustrated in notable differences between downstream temperature in 2007 and 2008. In 2008 grazing was considerably reduced compared to 2007 (from an estimated 800 head to 200 head of cattle) on the ranch. Certain fields were used as hay production as opposed to grazing pasture, resulting in different water uses and different return flow characteristics. These land and water use modifications were not quantified due to access limitations. Quantifying these elements will help improve knowledge about the relationships between ranch management and instream temperatures.

As part of this study, a two-dimensional hydrodynamic model was developed to simulate the effects of specified restoration actions on water temperatures. Comparing the simulation results to the TMDL assumptions of water temperatures under restored creek conditions indicates that projected temperature decreases are less than previously assumed. Though maximum temperatures under restored conditions decreased approximately 4°C (the temperature decrease projected for average temperatures in the TMDL), average temperatures decreased between 1°C-2°C. Two primary reasons for this reduced average temperature decrease are the assumed water temperature below Big Springs Lake at the headwater of Big Springs Creek and the amount of heating that would occur under an implemented TMDL en route to the Shasta River. Due to a lack of data describing water temperature at the Big Springs Lake outlet, the upstream boundary conditions identified in the TMDL were notably cooler than those observed during the baseline assessment study. In the TMDL, temperatures at the outlet were assumed to be approximately 12^{o}C; the baseline assessment study indicates that temperatures are closer to 15^{o}C. Therefore the upstream Big Springs Creek temperature is higher than previously assumed and temperatures will not decrease as much under restored conditions. Also, the TMDL did not identify specific restoration actions that would reduce average temperatures by 4^{o}C. Due to access limitations, site-specific data was not available that would have described heating patterns or areas of high restoration potential. Data gathered during the baseline assessment study helped the project team identify specific restoration actions (described in Section 7.4.1). Current assumptions implemented in the baseline assessment study indicate that improved irrigation efficiency and limited tailwater discharge to the creek would improve overall management of cold water on the ranch. However, under these specific restoration prescriptions, mean daily temperatures decreased by a considerably smaller margin – on the order of 1^{o}C.

10.4 *Recommendations and Conclusions*

Given the recent change in ranch management and the commencement of restoration activities, water temperatures are expected to shift from the patterns observed in the baseline assessment study. The accuracy of future TMDLs will rely on the continued monitoring of Big Springs Creek to track temperature and flow patterns as they change due to continued restoration actions. This continued monitoring should included concurrent data records of water temperatures at the mouth of Big Springs Creek and the GID intake to ensure that accurate simulations of warming trends are produced. Also, a comprehensive assessment of the irrigation conveyance facilities and associated infrastructure, land and water use practices, should be completed to improve understanding of the feedback relationship between ranch management and flow and water temperatures.

The Shasta River TMDL was developed using the best assumptions given the available data. Flow assumptions appear to be reasonable and consistent with 2008 observations. Temperature data gathered during the baseline assessment suggests that TMDL analysis assumptions should be revisited or TMDL implementation plan activities should incorporate the more recent data and findings. Though flow assumptions were consistent with observed data, changes in ranch management may affect the volume of flows during the irrigation season.