

**Keeping it Cool: Sustainable Stream Conservation Using Process-Based
Thermal Regime Management**

By

ANN DENISE WILLIS
DISSERTATION

Submitted in partial satisfaction of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

Civil and Environmental Engineering

in the

OFFICE OF GRADUATE STUDIES

of the

UNIVERSITY OF CALIFORNIA

DAVIS

Approved:

Jay Lund, Chair

Alexander Forrest

Steven Sadro

Committee in Charge

2020

*To my father, John Fissekis,
who encourages me with infinite love, and
my mother, Jennifer Fissekis, who took me on that fateful trip
down the Middle Fork Salmon River*

*To my husband, Major,
who has been the bedrock supporting
my thrilling and turbulent pursuits*

*To my daughters, Grace and Lela:
the power of water doesn't begin to compare with
the love I feel for you*

ACKNOWLEDGEMENTS

Every quest is a journey of a million steps. My parents planted the seed of this achievement through their relentless insistence that I strive for excellence and the revelation of the natural world. But achievement is only possible when ambition and hard work meet opportunity. Jay Lund and Stu Townsley kicked open the door of opportunity when no one else saw the engineering potential of a raft-guiding English major.

Jeff Mount, Peter Moyle, and Jay Lund created a home for mavericks with the Center for Watershed Sciences, then filled it with a new generation of scientists who have become family. Thank you for gambling your professional capital on an idea that has inspired and nurtured so many students like me, and for welcoming me into this community.

Thank you to Rob Lusardi, my best friend. Your friendship, support, and dissertation coaching have been the greatest gifts throughout my time at Watershed. You have become part of the fabric of my history, and I hope our partnership is long.

Thank you to Ryan Peek and Eric Holmes: Chapters 2 and 4 stand on your shoulders. I'm so excited for our next collaboration.

Thank you to the many partners and collaborators on Big Springs Creek. Working on that project was the first iteration of a dream come true for me: using science to achieve effective, resilient stream restoration.

Thank you to Andrew Rypel for asking the question that emboldened me to think beyond what I already knew I could accomplish. Thank you to Alex Forrest and Steve Sadro for all your interest, encouragement, and advice. Thank you to Nick Santos, who epitomizes what a good teacher should be. Thank you to Ben Houlton, who provided a sympathetic ear and support when all seemed hopeless: you are a valuable role model, and I am honored to learn from you.

Thank you to Karen Swan, Sarah Null, Rachel Johnson, Amy Campbell, Alison Whipple, Lily Tomkovic, Miranda Tilcock, Mollie Ogaz, Anna Sturrock, Tessa Hill, Kat Kerlin, Sallie Poggi, and every woman who gave me faith and strength in the face of hardships just by being. Knowing each of you has reminded me to lean in with my full, authentic self every single day.

ABSTRACT

Stream temperature is a major aspect of stream ecology. For ecosystems that depend on cold water, major disturbances from water extraction, dam regulation, and climate change have blocked or eliminated critical cold-water habitat. In California, cold-water conservation in streams is particularly critical as it is the southern extent of many cold-water species currently at risk of extinction. Research showing the importance of managing stream temperature from a regime-based perspective is disconnected from the more common practice of managing streams for threshold-based temperature criteria. This research connects the conceptual framework of regime-based stream temperature management and practice. Classifying California streams' thermal regimes identifies unique temperature patterns in groundwater-dominated stream reaches, suggesting that they should be prioritized for long-term cold-water conservation. The resilience of groundwater-dominated stream reaches contrasts with thermal regimes below dam outlets and in regulated reaches, which show an artificial pattern unable to replicate desirable cold-water regimes. A case study of Big Springs Creek, one stream in the classification analysis, shows that its thermal regime is controlled by a previously undocumented mechanism: aquatic macrophytes. Extensive, emergent aquatic macrophyte growth provides seasonal cover to over 70% of the stream surface, creating a riverine canopy. This riverine canopy is the dominant seasonal control of Big Springs Creek's thermal regime. Spatial and temporal growth was quantified with Unmanned Aerial Vehicle (UAV) imagery and used to develop a numerical model, which showed that monthly surveys were sufficient to accurately simulate the effects of the seasonal canopy on the thermal regime. The results of this research show that modeling thermal regimes to identify thermally resilient reaches, and numerical modeling of stream reaches to assess alternative management actions, is a viable strategy to prioritize conservation investments to restore and sustain cold-water ecosystems.

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CHAPTER 1: INTRODUCTION

Stream temperature is a major aspect of stream ecology. Water temperature is a principle determinant of habitat suitability for juvenile salmonids and other organisms in stream (i.e., lotic) ecosystems (Bjornn and Reiser 1991, Caissie 2006). High water temperatures can increase stress and mortality of juvenile salmonids that rear in stream habitats throughout the summer months (Marine and Cech Jr 2004, De Brabandere et al. 2007). So, many management and recovery efforts for threatened or endangered salmonids focus on reducing high summer water temperatures (Poole et al. 2004, Richter and Kolmes 2005).

Regulators commonly use water temperature metrics (e.g. 7-day running average of daily maximum temperatures) to assess macrohabitat quality for each freshwater salmonid life stage (McCullough 1999, USEPA 2003, USBR 2008); other research provides more stringent thresholds to define suitable thermal conditions for species (Rahel and Olden 2008). These metric-based water temperature targets imply that desirable temperatures, independent of other physical and biological feedbacks, are sufficient to maintain or enhance anadromous fish. For example, Shasta Dam on the Sacramento River blocks access to naturally cool headwater streams historically used by anadromous fish (Moyle 2002). To mitigate for the loss of headwater habitat, dam releases are used to meet temperature criteria at downstream compliance points (USBR 2008, Yates et al. 2008). Despite these efforts to maintain a cold-water environment, some targeted species remain unviable (Lindley et al. 2007).

Contrary to the implications of metric-based temperature management, stream temperature is not an independent feature of a stream system, but rather a product of its own complex spatial and temporal feedbacks of the heat budget. Conceptually, the heat budget refers to the feedback between drivers, buffers, and insulators that control the amount of heat energy in a stream (Poole and Berman 2001). These include heat directly absorbed from solar radiation, heat gained (lost) through condensation (evaporation), heat exchanged with the atmosphere and stream bed (i.e. sensible heat, conduction, and friction), heat transported from other inflows (i.e., advection), and the influences of topography, vegetation, and climate (Dugdale et al. 2017).

Mathematically, the heat budget is represented as:

$$Q_n = \pm Q_r \pm Q_l \pm Q_s \pm Q_c \pm Q_f \pm Q_a$$

where Q_n = net heat exchange, Q_r = net radiative heat flux, Q_l = latent heat flux due to condensation or evaporation, Q_s = sensible heat flux due to transfer between air and water, Q_c = heat flux due to conduction with the stream bed and banks, Q_f = heat flux due to friction, and Q_a = heat flux due to advection (Webb 1996). Drivers refer to the dominant element of the heat budget; buffers or insulators can shift or moderate the extent to which drivers dominate the heat budget. Feedbacks between drivers, buffers, or insulators are used to define a stream's thermal regime, which describes the magnitude, timing, duration, and frequency of change in a stream's water temperature patterns (Steel et al. 2017). While the significance of stream temperature and changes to thermal regimes are widely appreciated, the underlying feedbacks, or processes, that regulate thermal regimes are less well-defined (Dugdale et al. 2017, Steel et al. 2017).

Process-based temperature management considers the spatial and temporal feedbacks of the heat budget and recommends actions that target dominant temperature drivers at a given place and time. This approach requires both a robust understanding of underlying processes controlling the heat budget, and sufficient data to quantify the spatial and temporal variability of those feedbacks. Recent decades have brought numerous empirical studies on the complex interactions

that fundamentally control thermal regimes (Webb et al. 2008, Steel et al. 2017). Though monitoring methods have rapidly advanced, water temperature data quality ranges widely, making it hard to determine the underlying processes controlling thermal regimes (Webb et al. 2008, Hannah and Garner 2015, Dugdale et al. 2017).

Regardless of the inadequate data quantifying thermal regime feedbacks, process-based temperature management has long been recommended as the best approach for environmental management (Sinokrot and Stefan 1993, Poole and Berman 2001, Poole et al. 2004, Dugdale et al. 2017). Poole and Berman (2001) presented a conceptual framework to assess the relative role stream characteristics play in thermal regimes depending on stream size. While useful, the framework lacks important elements that are necessary to apply it to cold-water conservation. Also, it did not apply the conceptual framework to quantify site-specific thermal regimes, nor develop management recommendations that are defined by the underlying thermal regime processes. For conservation objectives targeting cold-water ecosystems, a better framework that identifies management actions based on dominant controls of cold-water thermal regimes is required.

This dissertation presents research that explores process-based water temperature management in California, beginning with a characterization of the state's thermal regimes and ending with a case study of a small, spring-fed tributary that is the focus of considerable conservation investments. Chapter 2 focuses on the classification of California streams' thermal regimes and quantifies the diversity of cold-water habitat in the state. Chapter 3 focuses on a single stream within the state-wide classification, Big Springs Creek, and shows how its thermal regime is dominated by novel ecohydrologic feedbacks resulting from a riverine canopy of aquatic vegetation. Chapter 4 explores novel methods to quantify the temporal and spatial growth of aquatic vegetation in Big Springs Creek, and the resolution of data that would be needed to support a numerical model capable of guiding management decisions. Chapter 5 presents conclusions of this research and recommendations for further work. The findings of this research provide a roadmap on how to approach process-based water temperature management and achieve sustainable, successful conservation that balance human water use with environmental objectives.

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CHAPTER 2: CLASSIFYING CALIFORNIA'S STREAM THERMAL REGIMES FOR COLD-WATER CONSERVATION

Abstract: Stream temperature research is rapidly shifting from single-metric quantification of stream temperatures to multi-metric, thermal regime characterizations for larger scale streamscapes. Given the considerable investments in the recovery of cold-water fishes such as salmon, understanding where cold water is likely to persist, and how cold-water thermal regimes vary, is important for successful conservation. California's unique position at the southern end of cold-water ecosystems, highly variable geography and hydrology, and near-ubiquitous dam regulation requires a systematic approach to thermal regime classification. The results of this study of multi-metric thermal regime classification show that cool- and cold-water thermal regimes vary across California; groundwater-dominated streams are an important, but as yet poorly explored class of thermal regimes; dam regulation imposes an artificial thermal regime on downstream ecosystems; and California's dams do not contain sufficient cold-water storage to replicate desirable, reach-scale thermal regimes. While the barriers to cold-water conservation are considerable and the trajectory of cold-water species towards extinction is dire, reaches demonstrating resilience to climate warming are more worthy of investment.

Key words: Thermal regime, conservation, California, stream temperature, cold-water habitat, dams

Introduction

Water temperature influences biological, physical, and chemical processes in stream ecosystems (Poole and Berman 2001, Caissie 2006, Webb et al. 2008). Whereas some research explores the behavioral response of aquatic organisms across stream temperature thresholds, particularly in a regulatory or management context (e.g., Welsh et al. 2001, USEPA 2003), other work considers annual stream temperature patterns, or thermal regimes, to characterize the dynamics between stream temperature and aquatic ecosystems (Poole et al. 2004, Caissie 2006). Analogous to flow regimes, thermal regimes characterize the magnitude, frequency, duration, timing, and rate of change in water temperature (Poff et al. 1997). An annual time series of water temperature data defines a thermal regime for a specific location whereas thermal landscapes consider the pattern of thermal regimes over an entire region (Poole et al. 2004, Steel et al. 2017). Thermal regime research from the refugia- to reach-scale has explored the relationship between the timing, magnitude, and extent of exposure to both warm and cool water temperatures for the limits and overall productivity of aquatic ecosystems (Bjornn and Reiser 1991, Eaton and Scheller 1996, Sutton et al. 2007, Fraser et al. 2019, Lusardi et al. 2019).

Given overall trends of stream warming from climate change and land and water management, the loss of habitat that supports cold-water species, such as salmonids, is a particular concern (Eaton and Scheller 1996, McCullough 1999, Sharma et al. 2007, Isaak et al. 2015). Globally, warming of thermal landscapes are a direct result of climate change (Van Vliet et al. 2011, Arora et al. 2016, Michel et al. 2019). Across the United States, projections show nearly 50% of cold-water habitat could be lost to climate change (Eaton and Scheller 1996). These changes are compounded by regulation effects of dams. For example, changes in the timing and magnitude of peak temperatures from climate warming and dam regulation in the Columbia River and its tributaries have contributed to declining salmon populations (McCullough 1999). For fish like salmonids, regulated thermal regimes shift the timing of important cues for life history strategies that evolved in unregulated regimes (Bjornn and Reiser 1991, Angilletta et al. 2008).

Given limited resources and the desire to target conservation investments for maximum environmental benefits (Wu and Skelton-Groth 2002), identifying long-term, viable cold-water habitats is critical (Isaak et al. 2015). Thermal regime modelling and classification has been widely used to characterize spatial and temporal thermal variability within and across watersheds and regions (Ward 1963, Cluis 1972, Caissie et al. 2001, Maheu et al. 2016, Jones and Schmidt 2018, Daigle et al. 2019). However, natural resource management agencies have struggled to integrate the concepts of thermal regimes and landscapes into strategies that target species of conservation or economic importance (Steel et al. 2017).

In California, cold-water conservation is complicated by geography and engineering. California's Mediterranean climate includes extreme climatic and hydrologic variability (Lane et al. 2017). As the southern extent of many cold-water fish species, climate warming is likely to shrink the extent of unregulated cold-water habitat (Null et al. 2013). But unregulated reaches account for a small fraction of existing cold-water habitat: over 1,400 dams are on streams relevant to native fish conservation, making available habitat highly regulated (Grantham et al. 2014). Water management and land use changes have already changed thermal regimes throughout the state, with warmer temperatures reducing the distribution and survival of cold-water species (Moyle 2002). Previous studies of thermal regimes for cold-water ecosystems in California have generally neglected regulated reaches (Moyle and Ellison 1991, Eaton and Scheller 1996, Null et al. 2013, Maheu et al. 2016), and either explored California as part of a national analysis (Eaton and Scheller 1996, Maheu et al. 2016) or have focused on a specific region within the state (Yates et al. 2008,

Null et al. 2013, Fullerton et al. 2015, Nichols et al. 2020). In addition, methods used for some of these analyses rely on numerical modelling (Yates et al. 2008, Null et al. 2013) or costly data collection (Fullerton et al. 2015): resource-intensive approaches that are impractical for a statewide analysis. Other, more data- and computationally efficient approaches use short records (< 5 years) that bring considerable uncertainty in the results (Eaton and Scheller 1996, Chu et al. 2010, Rivers-Moore et al. 2013).

To direct conservation resources effectively to reaches with regulated cold-water regimes in California, strategies should account for extensive regulated influences and capture nuances of highly variable geography and hydrology. A classification framework to support cold-water conservation should meet several criteria: sufficient spatial resolution to capture the vast geographic and hydrologic diversity of California's watersheds and ecosystems; sufficient temporal resolution (i.e., daily or sub-daily) to meet the standard of thermal regime modeling and be ecologically relevant; include both regulated and unregulated reaches; and use a method that can be transferred easily to other watersheds, but provides sufficient characterization of local thermal regimes to support site-specific conservation decisions.

This study develops a classification framework for California's thermal regimes that allows for the rapid identification of stream reaches likely to sustain viable cool- and cold-water thermal regimes. In doing so, several fundamental questions related to cold-water conservation are addressed. First, what constitutes a cold-water thermal regime, and how does it vary across a region? Second, do dams reset the longitudinal evolution of thermal regimes along a streamscape? Finally, can dams be managed to replicate desirable cold-water regimes? While this study focuses on cold-water habitat in California, the results can be applied to any region and ecosystem to explore how their thermal regimes may be distinct from alternative locations. The study results can help evaluate which stream reaches should be targeted for cold-water conservation, and whether alternative, regulated reaches are suitable trade-offs to historic, unregulated habitat where historic habitat may be inaccessible as a result of dams.

Data and Methods

Data sources and site selection criteria

Stream temperature data were used to model the thermal regime for 77 sites throughout California. Data were downloaded from the United States Geological Survey (USGS) and California Data Exchange Center (CDEC), publicly accessible databases. Monitoring sites were initially filtered to exclude those in the Sacramento-San Joaquin Delta to focus the analysis on freshwater thermal regimes and minimize the influence of tidal dynamics. Recent studies have recommended at least 8 to 12 years of continuous, daily average data to reduce uncertainty from interannual variability (Jones and Schmidt 2018, Daigle et al. 2019). Monitoring stations were filtered to identify those with at least 8 years of daily average stream temperature data to balance the desire for reduced uncertainty with sufficient spatial representation to explore California's geographic diversity. Additional data were used from a long-term (10-year) monitoring network in the Shasta River watershed in Siskiyou County, northern California (Nichols et al. 2014, Willis et al. 2017, Nichols et al. 2020). These sites were in 7 of 10 hydrologic regions of the state (as defined by the California Department of Water Resources: North Coast, North Lahontan, Sacramento River, San Francisco Bay, San Joaquin River, South Lahontan, Tulare Lake); no sites in the southern range of the state (Central Coast, South Coast, and Colorado River) had sufficient periods of record for our analysis. All data were reviewed to remove flagged data and obvious outliers. Daily average stream temperature was calculated from sub-daily data sets.

Thermal regime modelling and classification

Thermal regime modelling used the mosaic package in R Studio. The reviewed datasets were used to calculate annual thermal regimes, defined by the daily mean temperature for each day of the water year (October 1 through September 30). Annual thermal regimes were modelled with a sine function (Cluis 1972, Caissie et al. 2001):

$$T_w = a + b \sin \frac{2\pi}{365} (n + n_o)$$

where T_w is water temperature, n is the day of water year, and a , b , and n_o are coefficients that were optimized using least square regression (Figure 2.1). Model fitness for each site was quantified using residual standard error; values closer to zero indicated better fit. Modelled thermal regimes were classified based on clustering and statistical analysis methods developed in Maheu et al. (2016), which are briefly summarized here: mean, amplitude (i.e., the difference between the annual mean and annual maximum water temperature), and phase (i.e., day of water year when annual maximum occurs) metrics were calculated from each thermal regime model (Figure 2.1), then classified using Ward's method, an agglomerative nesting (agnes) algorithm. Each class includes a cluster of individual sites, defined based on unique features of parameters to show similar annual variability and amplitudes distinct from other classes. The number of classes was determined using Calinski and Harabasz's (CH) index and the sum of squares ("Elbow") method (Milligan and Cooper 1985). We also used the silhouette method to validate the appropriate number of classes (Rousseeuw 1987). Classes were examined for stability using the Jaccard coefficient, with stable clusters indicated by coefficients greater than 0.75 (Hennig 2007). Clustering and statistical indices were computed using R packages cluster, factoextra, and fpc.

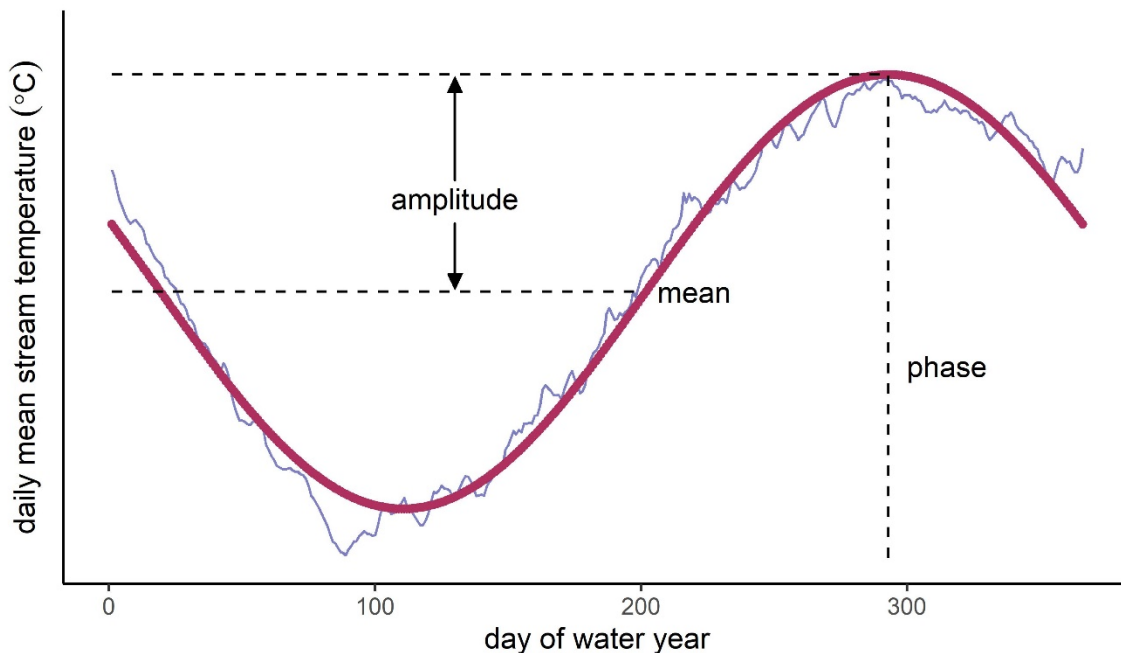


Figure 2.1. An example of a thermal regime model fit to observed data, using data from USGS site 11390500 (Sacramento River below Wilkins slough near Grimes, CA). Cluster analysis is based on annual mean, amplitude, and phase metrics for each thermal regime model. Amplitude and phase are analogous to annual maximum and day of annual maximum metrics. Figure adapted with permission from Maheu et al. (2016), figure 1.

Following classification and statistical testing, we visualized the distribution of the clusters by plotting the first two principle components that were calculated during the clustering analysis. We examined the relative contributions of each parameter (annual mean, amplitude, and phase) to each principal component to determine which was more important to the final clustering results. In addition, we calculated the distance of each cluster member relative to the centroid of its respective cluster to identify weak members. For each class, a histogram was made to examine the distribution of distance to centroid across all members. Weak members were defined as sites furthest from the centroid. Additional clustering analysis was done using the same methods to assess whether weak members had fundamentally different dynamics that were lost in higher-order clustering, or were simply farther downstream on the same gradient as the strong members.

Influence of dam regulation

Dam regulation effects were examined by quantitatively and qualitatively assessing thermal regime patterns downstream of dams. Previous research showed that large dams in California's Central Valley influence thermal regime 30-60 km downstream of release outlets (Angilletta et al. 2008, Yates et al. 2008); Shasta Dam, impounding California's largest reservoir, was shown to influence temperature patterns up to 250 km downstream (Lowney 2000). Given the range of dam sizes and the uncertainty of reaches influenced by multiple dams may show compounded and extend downstream effects, a histogram was generated for each thermal regime detected within the reach 100 km downstream of dams to explore the relationship to distance below dam and regime type within the range of magnitude generally influenced by dam releases. Finally, thermal regimes in these reaches were examined for member strength of each below-dam site to its respective regime class.

Results

Modeling results showed that the sine curve was a reasonable fit for the sites included in the study. Of the 77 sites, 53 had residual standard errors less than 1.0 °C, and all but two had residual standard errors less than 2.0 °C (Figure 2.2). Poorer model fit tended to occur at sites with larger annual and daily variation.

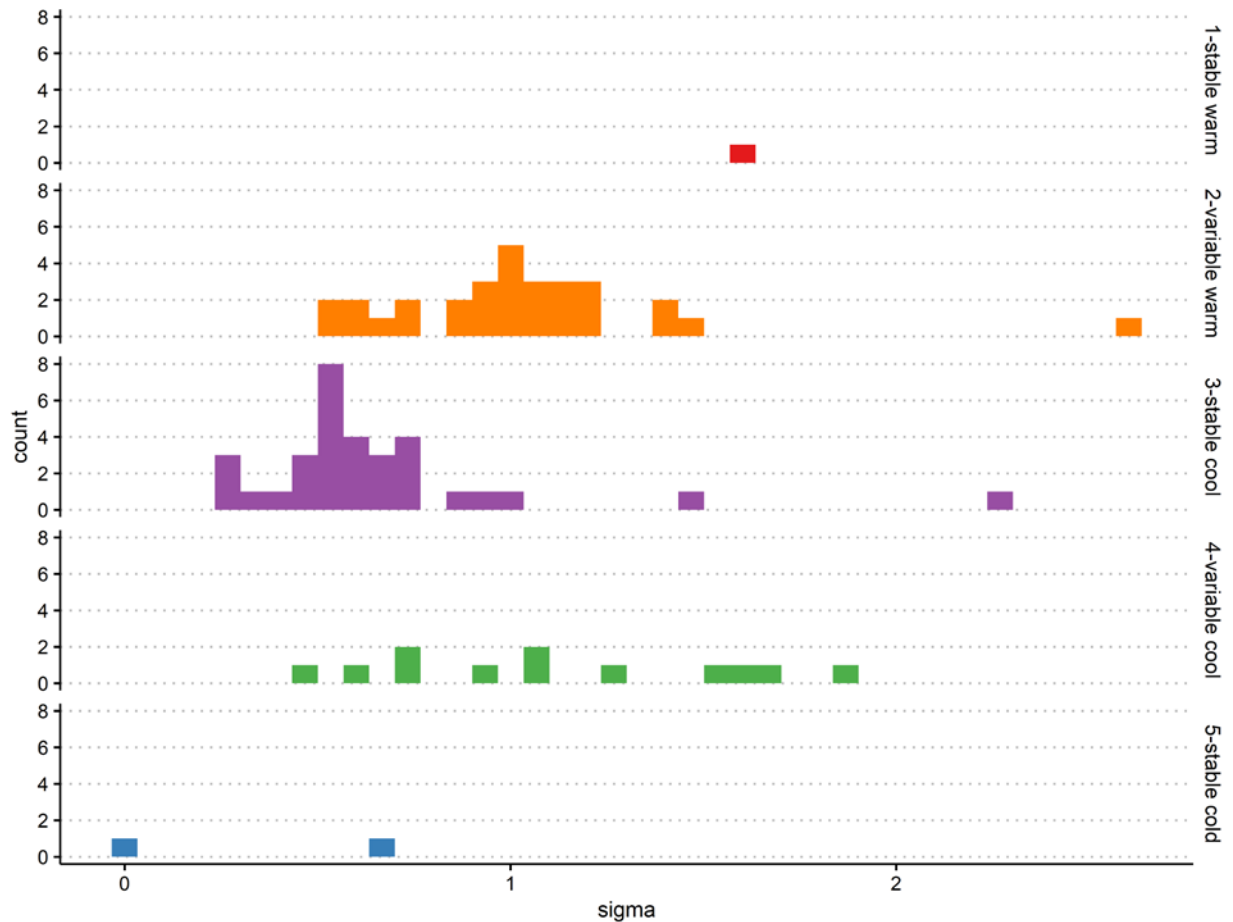


Figure 2.2. A histogram of average residual standard error for each site, grouped by thermal class.

The clustering analysis results showed that California’s thermal regimes were best divided into either three or five classes. An inspection of each result showed that $k = 3$ produced generally coarse groupings with little insight to the nuances of various thermal regimes. The five-class system was preferred due to its strong coefficients (Ward’s agglomerative coefficient = 0.96; CH index ranked $k = 5$ next favorable behind $k = 3$), and more refined characterization of the thermal landscape. To visualize the five clusters, the sites were plotted using the first two principal components of the clustering analysis, Dim1 and Dim2, then grouped by cluster (1-5, Figure 2.3A).

The two principal components represent different proportions of the three parameters (annual mean, annual amplitude, and phase) used in the clustering analysis, and together accounted for 88.6% of the distinguishing features of each thermal regime. Dim1 primarily represents the effect of annual mean temperature (46.6% of this principal component) and annual amplitude (46.8%) on a thermal regime; Dim2 represents the effect of annual maximum timing (phase; 93.4%). The five clusters generally occupy unique regions, with the exception of groups 2 and 4. Groups 1 and 3 were the most stable, with Jaccard coefficients (J_c) of 0.86 and 0.95, respectively. Groups 2 and 4 had considerable overlap, and were less stable ($J_c = 0.63$ and 0.59, respectively); Group 5 was similarly unstable ($J_c = 0.63$). Despite this instability, further examination of results showed a strong physical basis for each grouping. An examination of the elbow and silhouette results also supported selecting $k = 5$ as the appropriate number of clusters (Figure 2.3B and C).

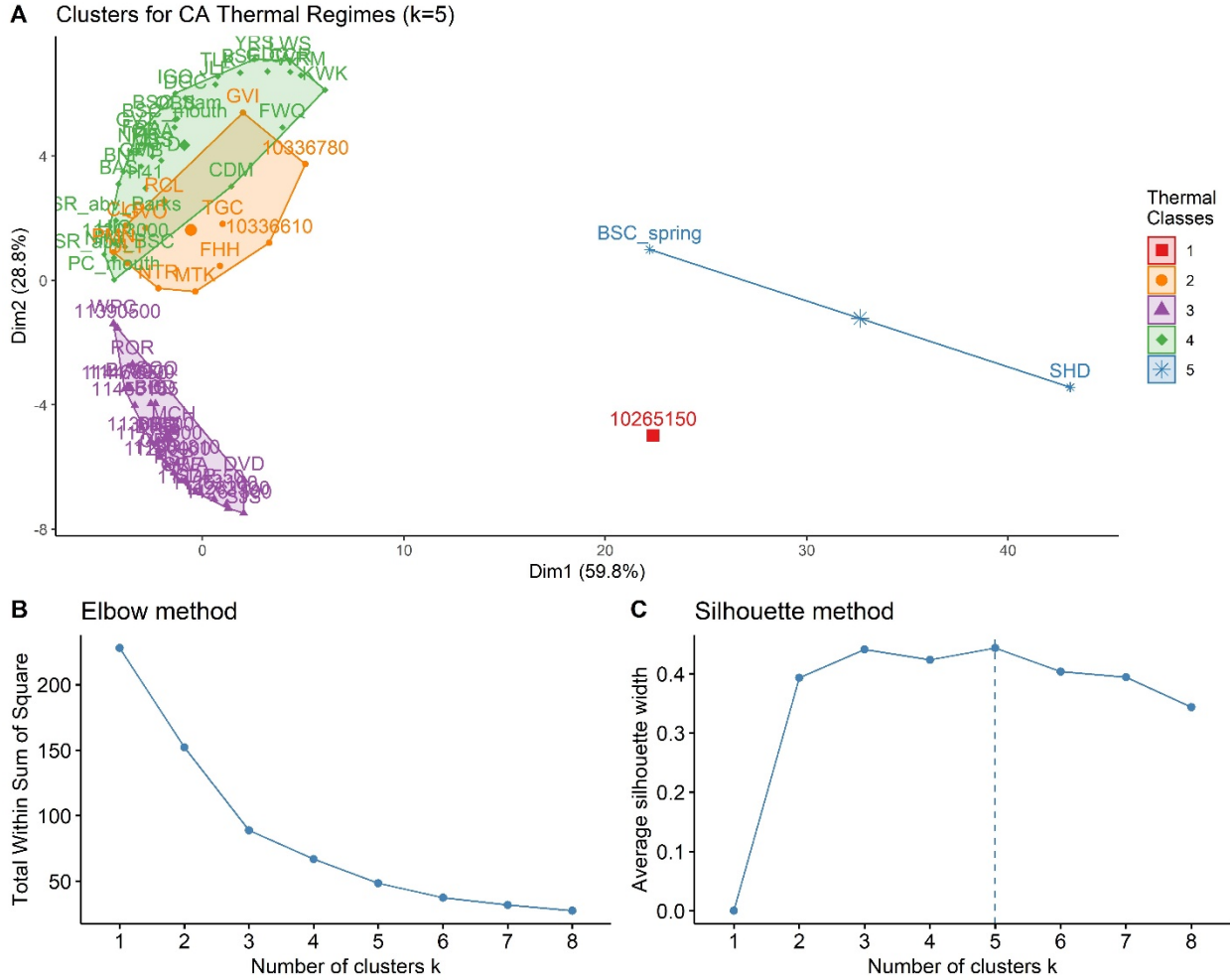


Figure 2.3. A) California’s thermal regimes were grouped into five clusters, with the centroid of each cluster marked by relatively larger symbols designated for each cluster. The percent influence of each principle component, Dim1 and Dim2, on the overall clustering is noted in parentheses. The inflection points at $k = 5$ in the B) elbow and C) silhouette analyses further support the selected groupings.

Because group 1 had a single member (USGS gage 10265150, Hot Creek in South Lahontan hydrologic region), it was not included in the weak-member analysis. Of group 5’s two members, SHD was almost twice as far from the group’s centroid (43.2 units away from the centroid) as the other member, BSC_spring (22 units, Figure 2.3). The remaining groups showed a tighter distribution of members around their centroids. The cluster plot of group 2 showed that most members were on the perimeter of the cluster, with distances from the centroid ranging from 0.5-6.3 units, suggesting a range of member strength to the cluster. Group 3 was tightly distributed around its centroid (4.0-7.8 units) despite having a larger population ($n = 30$). Group 4 was slightly more dispersed, with member distances from the centroid ranging from 3.3-8.6 units; unlike groups 2 and 3, members were scattered both within the cluster and around the perimeter. When the sites were reclassified to allow for additional clusters ($k = 6$), all groups retained the same membership except for group 4, which split into two clusters ($n=20$ and $n=12$).

Thermal regimes were plotted by the five original clusters and named based on mean annual maximum temperatures ($\bar{T}_{max} > 20\text{ }^{\circ}\text{C}$ was warm; $15\text{ }^{\circ}\text{C} > \bar{T}_{max} > 20\text{ }^{\circ}\text{C}$ cool; $\bar{T}_{max} < 15\text{ }^{\circ}\text{C}$ cold) and their relative annual variability (Figure 2.4A). Groundwater-fed springs each established their

own thermal regimes (stable cold and stable warm), differing in magnitude and timing of annual maximum temperature. The stable warm class was populated by a single site, with $\bar{T}_{max} = 28.7$ °C that occurred on DOWY = 353 (Sep. 19, the latest of all thermal regimes), and an annual mean (\bar{T}_{mean}) of 27.2 °C (Figure 2.4A, B, and C). As this thermal regime described only one site, no assessment could be made of potential variability in this regime. The stable cold regime similarly described a groundwater-fed site, as well as the outlet of Shasta Dam. The stable cold regime was characterized by $\bar{T}_{max} = 11.9$ °C (DOWY = 89, Dec. 22 – the earliest of all thermal regimes) and $\bar{T}_{mean} = 11.1$ °C. The two members of this regime showed little variability in annual maximum and mean temperatures, but high variability in the timing of the annual maximum: at the Shasta Dam outlet, the annual maximum occurred on DOWY 16 (Oct. 17); at the groundwater spring, it occurred on DOWY 148 (Feb. 26). Interestingly, the thermal regime at the Shasta Dam outlet showed the same annual pattern as the stable warm groundwater spring, while the stable cold groundwater spring showed a generally uniform temperature throughout the water year (Figure 2.4A).

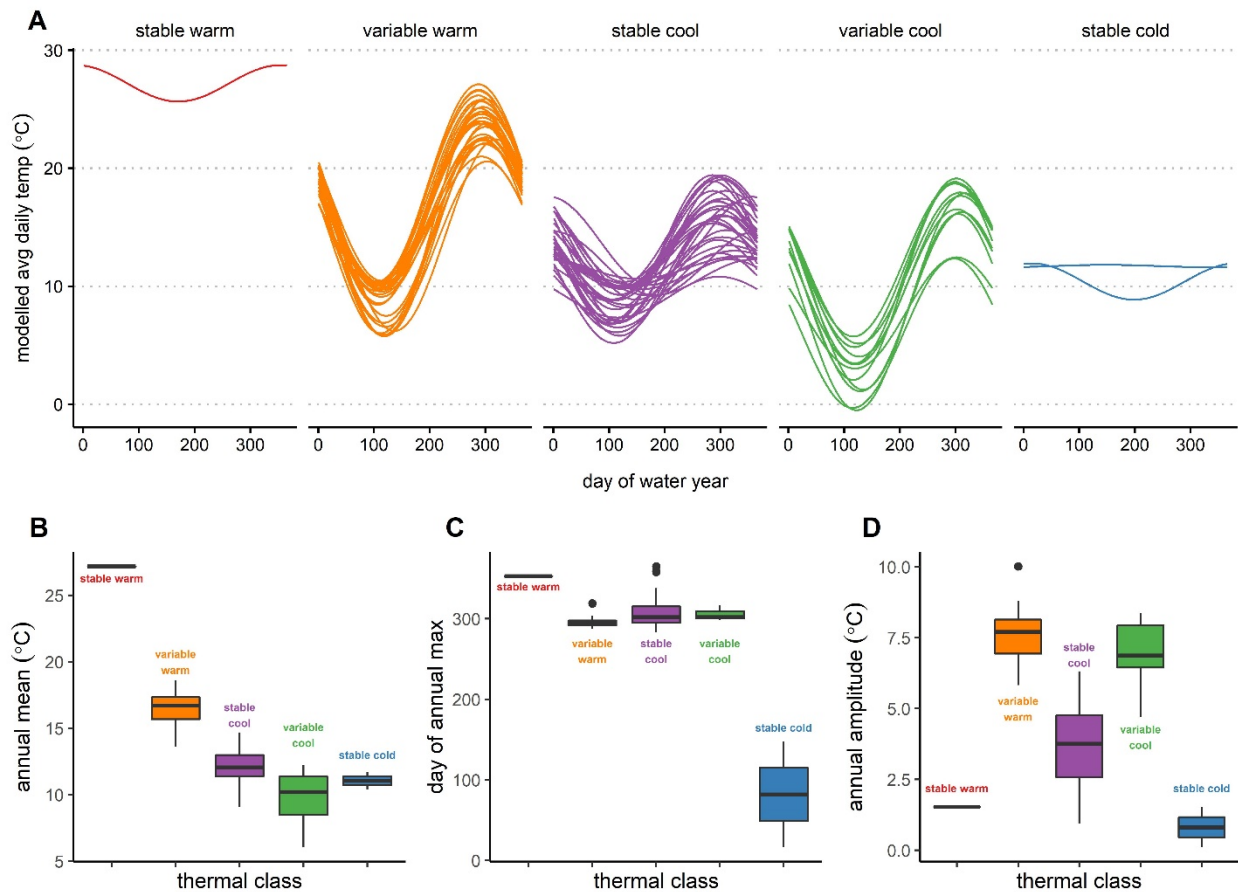


Figure 2.4. A) Classified models and box plots of B) annual mean, C) day of annual maximum, and D) annual amplitude; based on Maheu et al. (2016) Figure 3, with permission. Thermal regimes were characterized based on their mean annual maximum (warm, cool, or cold) and relative annual variability.

The variable warm regime included 30 sites, with a $\bar{T}_{max} = 24.0$ °C (DOWY = 295, Jul. 23) and $\bar{T}_{mean} = 16.4$ °C. This thermal regime showed the highest range of annual amplitude (Figure 2.4D) and second highest annual mean temperature (Figure 2.4B). Annual maximum and mean temperatures ranged from 20.6-27.1 °C and 13.6-18.5 °C, respectively. Of the classes with multiple members, the variable warm regime had the most consistent day of annual maximum, ranging from

DOWY 287-298 (Jul. 15-Jul. 26), with a single site showing its day of annual maximum on DOWY 319 (Aug. 16 at site SCQ, the Tule River at the outlet of Success Dam).

The stable cool regime included 32 sites, with a $\bar{T}_{max} = 15.7$ °C (DOWY 309, Aug. 6) and $\bar{T}_{mean} = 12.1$ °C. While the stability was observed in terms of the overall range of annual temperatures across this thermal regime (Figure 2.4A), each classifying metric showed variability across the regime's member sites: annual maximum temperatures showed a range of 10.8-19.4 °C; mean temperatures, 9.1-14.7 °C; and DOWY 283-365 (Jul. 11-Sep. 30). The variable cool regime included 12 sites, with a $\bar{T}_{max} = 16.9$ °C (DOWY 305, Aug. 2) and $\bar{T}_{mean} = 9.8$ °C. In contrast with the stable cool regime, the variable cool regime had a greater variable annual temperature pattern (i.e., the range of temperatures illustrated by the annual trend), but less variable range of annual maximum and mean temperatures, and day of annual maximum. Annual maximum and mean temperatures ranged from 12.3-19.1 °C and 6.0-12.2 °C, with the annual maximums from DOWY 298-317 (Jul. 26-Aug.14).

With the exception of the stable warm regime (which only described a single site), each regime occurred in several hydrologic regions and multiple thermal regimes occurred between the headwaters and mouth of each watershed (Figure 2.5, Table 2.1). Stable cold regimes were found in the North Coast and Sacramento River hydrologic regions; the variable warm and stable cool regimes occurred in the North Coast, Sacramento River, San Francisco Bay, San Joaquin River, and Tulare Lake hydrologic regions. The variable cool regime occurred in the North Coast, North Lahontan, Sacramento River, and San Joaquin River hydrologic regions. The frequency of variable warm sites increased towards inland, southern areas. While the North Coast and Sacramento River hydrologic regions had the same number of stable cool and stable cold sites ($n = 12$ and $n = 1$, respectively), the Sacramento River had more variable warm ($n = 9$ versus $n = 3$) and fewer variable cool ($n = 2$ versus $n = 4$) sites (Table 2.1). These differences increased in the San Joaquin River hydrologic region, with 15 variable warm sites and 8 stable cold; however, the San Joaquin had more variable cool sites than the Sacramento River ($n = 3$ versus $n = 2$).

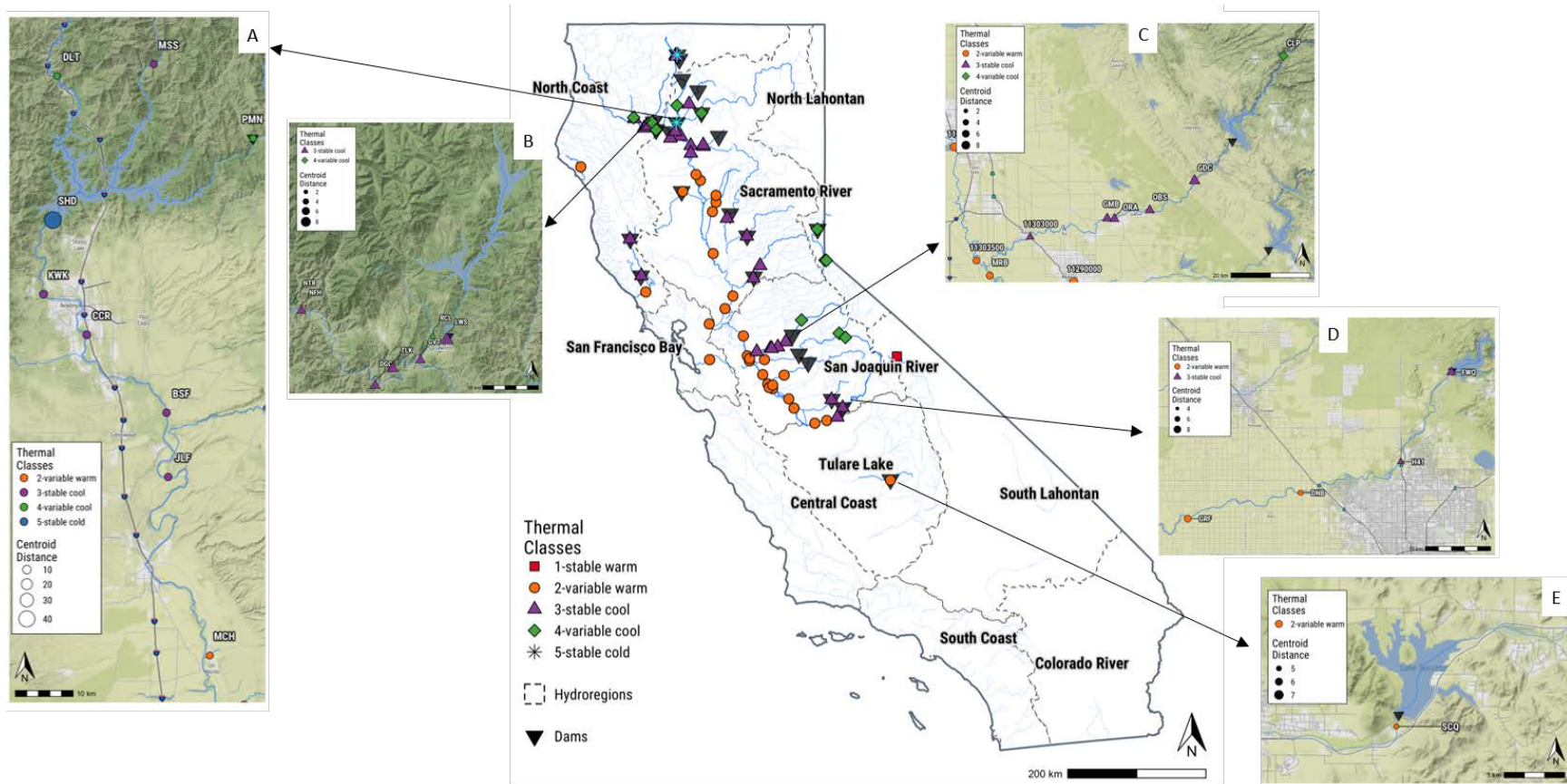


Figure 2.5. Map of classified thermal regimes and dams located upstream of study sites in California, with panels of thermal regimes below a) Shasta, b) Lewiston, c) New Melones, d) Friant, and e) Success dams. Dotted lines show the borders of California's hydrologic regions as defined by the state Department of Water Resources.

Table 2.1. A summary of the thermal regimes occurring in the seven hydrologic regions included in this study. Hydrologic regions are defined the California Department of Water Resources; the Central Coast, South Coast, and Colorado River are not included as no study sites were located in those regions.

	Stable warm	Variable warm	Stable cool	Variable cool	Stable cold
North Coast	0	3	12	4	1
North Lahontan	0	0	0	3	0
Sacramento River	0	9	12	2	1
San Francisco Bay	0	2	0	0	0
San Joaquin River	0	15	8	3	0
South Lahontan	1	0	0	0	0
Tulare Lake	0	1	0	0	0

The relative location of a site to a dam appeared to influence its thermal regime more than its hydrologic region. In general, sites upstream of reservoirs or in unregulated tributaries tended to have a variable cool thermal regime; stable cool regimes often occurred at dam outlets and extended downstream before transitioning to variable warm regimes (Figure 2.5). The outlet of Shasta Dam and Success Dam were two exceptions: Shasta Dam produced a stable cold regime at its outlet, while Success Dam (the southern-most site) created a variable warm regime (Figure 2.5A and E). Above California’s Central Valley rim dams, thermal regimes were exclusively variable cool. In the Central Valley, stable warm regimes generally occurred in the mainstem Sacramento and San Joaquin rivers, despite stable cool regimes their respective tributaries.

In addition, the stream length affected by an upstream dam varied. Stable cool regimes tended to occur closer to dams, while variable warm regimes were more frequent farther away (Figure 2.6). Below Shasta, Lewiston, and New Melones dams, stable cool regimes were observed tens of kilometers downstream (Figures 2.4A, B, and C). New Melones produced a stable cold thermal regime 83 km below its outlet (site 11303000), the farthest range of influence observed below any of the dams included in this study (Figure 2.5C, Figure 2.6). Success and Black Butte dams produced the shortest range to variable warm thermal regimes at 0.6 km and 1.9 km, respectively (Figure 2.5E; Black Butte panel not shown). The remaining dams could maintain stable cool regimes at least 40 km downstream from their outlets before transitioning to variable warm regimes (Figure 2.6).

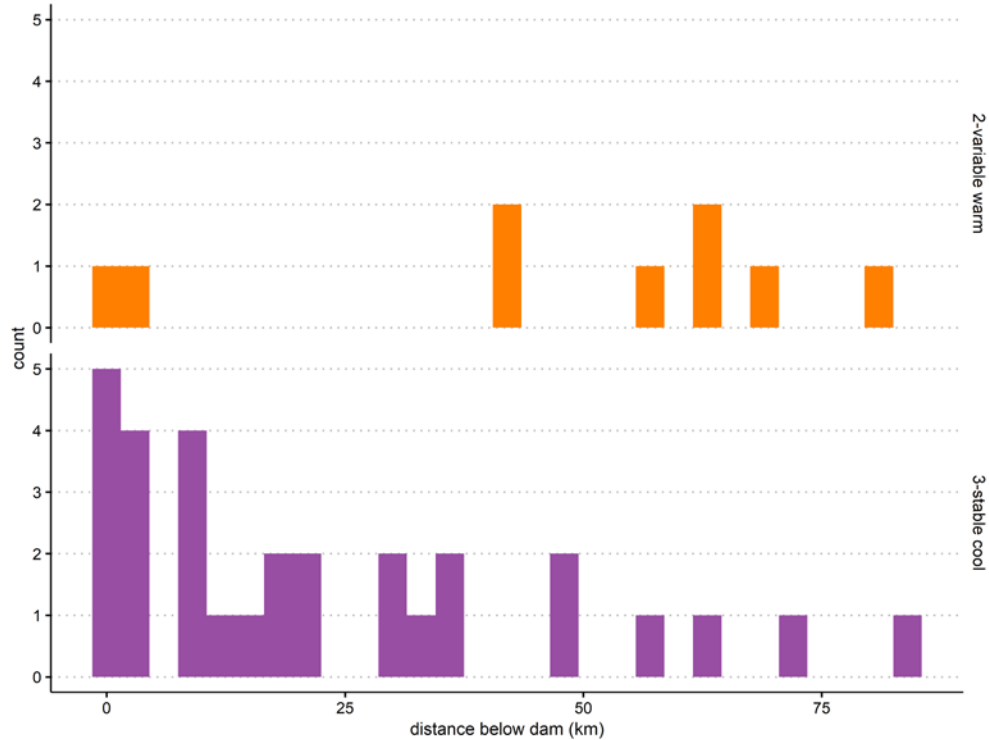


Figure 2.6. Histogram of thermal regime location relative to nearest upstream dam.

Discussion

For cool and cold-water ecosystem conservation, what constitutes a functional thermal regime for a desired ecosystem and whether it can be replicated in both space and time through managed dam releases is a critical question. This study developed a framework to identify thermal regimes throughout California, compare cool and cold-water regimes, and explore the effects of dams. We show how a few simple metrics can classify thermal regimes in a diverse hydrologic and geographic streamscape, provide insights to the importance of groundwater-dominated reaches, and distinguish cool thermal regimes in regulated and unregulated stream reaches. Specifically, we found that “cold” regimes vary throughout northern California, which complicates efforts to restore or replicate cold-water ecosystems through reservoir regulation. Where dams are managed to maintain desirable temperatures for cold-water species, they introduce an artificial regime that disrupts natural thermal patterns, generally reproduce summer patterns of unregulated regimes, but fail to replicate similar winter patterns. These results have important implications for questions about dam regulation, dam removal, and the likelihood of some species’ recovery.

Thermal regime classification method

The study shows how only three metrics – annual mean, annual amplitude, and phase (i.e., day of annual maximum) – describe most (>88%) differences in thermal regimes with a large enough number of clusters for meaningful differentiation. The variability of thermal regimes illustrated by the principal component analysis supports the idea that stream temperature management should consider several features of the annual thermal regime. Considering only annual mean or amplitude would have made it difficult to classify members into discrete groups; including day of annual maximum added a valuable dimension to the solution space occupied by

each group (Figure 2.3). Just as a stream's overall flow regime is defined by several variables (magnitude, timing, frequency, rate of change, and duration) important to ecological stream function (Poff et al. 1997, Yarnell et al. 2015), our results suggest the same is true for stream temperature management. In contrast, a metric-based threshold, like annual maximum or mean stream temperature, may appear to replicate a particular thermal regime, but can miss other ecologically significant aspects of stream temperature. Future work could explore additional metrics to assess which variables best characterize thermal regimes.

The period of record needed to classify thermal regimes with reasonable (>75 %) certainty is another consideration of this method. At the time of this analysis, only 36 % of publically available gages (77 out of 216) had sufficient data for this analysis, and none were in the southern portion of the state. The dearth of long-term, daily data for thermal regime analysis in southern California is reflected in other studies that use publicly available stream gages or crowd-sourced stream temperature database (Maheu et al. 2016, Isaak et al. 2020). Nevertheless, salmonids and other cold-water species have been documented as far south as Mexico (Williams et al. 2015), indicating that cool- and cold-water regimes can extend further south than shown in this study. Given the variability of cool- and cold-water thermal regimes illustrated in this study and their influence from hydrology, geography, and regulation, new regimes may be in the extreme southern boundary of cold-water species' ranges.

To expand this analysis for a broader geographic area, uncertainty could be quantified for sites with less data to indicate the likelihood of correct classification. Distance from centroid also seems a useful indicator of weak members and can track whether a site departs from the cluster centroid as more data is gathered. Nevertheless, the simplicity of the current method, combined with the common metrics used to define thermal regimes, illustrates how easily this technique could be applied beyond this study's area to develop insights about regional thermal regimes.

What constitutes a cold-water thermal regime?

In ecological terms, "cool" and "cold" is typically used to classify species based on temperatures that support optimal growth, and are often simplified to static thresholds. These thresholds vary by region: Magnuson et al. (1997) identified 10-15 °C for cold-water fishes, 21 °C for cool, and 30 °C for warm in the Great Lakes region; Rahel and Olden (2008) suggested <20 °C, 20-28 °C, and >28 °C for more general cold, cool, and warm-water optima. Other studies use additional criteria to classify thermal regimes (Maheu et al. 2016, Steel et al. 2017, Isaak et al. 2020), but still rely on threshold-based definitions for cool and cold. Our results show that, based on the Rahel and Olden (2008) criteria, all but two of California's thermal regimes would be considered cold; yet these supposedly co-equal cold regimes demonstrate a range of ecological performance related to targeted cold-water species (FitzGerald et al. 2019). Regulatory guidance has trended toward a more temporally refined thresholds based on target species and their life histories (EPA 2003). Refined classification, whether based on variability (Maheu et al. 2016), geography (Isaak et al. 2020), or some other feature, are important to distinguish thermal regimes that would be considered equally supportive of cold-water ecosystems using a threshold system.

In addition to developing a classification that captures the differences among California's cool- and cold-water regimes, our objective was to provide a classification to support conservation decisions. To understand the differences between cold-water regimes, agglomerative nesting showed comparable statistical strength of classifying California's thermal regimes into three or five classes. Fewer classes may provide a simpler framework that could be applied more easily, but are not necessarily useful for management decisions, which typically occur at state or local levels. Null et al. (2013) found that thermal regimes of California's western Sierra Nevada rivers did not show

the same shifts in desirable cold water habitat as in national-scale studies. Because California is the southern end for the range of many cold-water species (Moyle 2002), has diverse geographic and hydrologic streamscapes (Lane et al. 2017), and is strongly influenced by dam regulation (Grantham et al. 2014), having high resolution of its thermal regimes and effects of regulation is desirable for conservation planning and investment. Thus, while three classes had slightly stronger statistical support, five classes provided more insightful differences between cool- and cold-water thermal regimes, particularly relative to groundwater and dam releases.

The five-class system revealed nuanced differences between cool- and cold regimes, and highlighted the potential of groundwater-fed streams to support cold-water conservation. Warm and cold groundwater-fed springs accounted for two of California's five thermal regimes: stable warm and stable cold. Although each class contained a single groundwater-fed site, these regimes illustrated a unique thermal pattern dominated by groundwater-fed spring sources. The stable cold regime, which included both a groundwater-fed spring and the outlet of Shasta Dam, was relatively unstable as indicated by its Jaccard coefficient and large spread of members from the cluster centroid. Additional data describing stable cold sources would improve understanding of this regime by indicating whether it is a stable class with high variability (which would account for the large spread of the initial two members) or better broken down into separate classes, possibly defined by large groundwater-fed springs and dams with large cold-water storage volumes. Despite studies that have classified thermal regimes in California (Maheu et al. 2016; Isaak et al. 2020), none explicitly identified spring-fed thermal regime sites in the state. However, the presence of slightly thermal groundwater-fed streams in California (Nathenson et al. 2003, Nichols et al. 2014) suggests that such a class may be more prevalent than currently known. A separate, stable cold class dominated by releases from reservoirs, though, is unlikely given the dearth of reservoirs in California with the cold-water capacity comparable to Shasta Reservoir.

Existing research also points to important contributions of groundwater-dominated reaches to cold-water ecosystems, further supporting the decision to classify them separately from runoff-dominated and regulated classes. Previous studies showed that, even in regions sensitive to climate warming, watersheds with larger flow volumes and groundwater contributions, like California's Feather River, are less vulnerable (Null et al. 2013, Maheu et al. 2016). Also, some Californian spring-fed streams have novel hydroecological feedbacks that drive their thermal regimes (Willis et al. 2017), influence reaches tens of kilometers downstream from spring sources (Nichols et al. 2014, Nichols et al. 2020), and support robust ecological productivity and conservation potential (Lusardi et al. 2016, Lusardi et al. 2019). Given the ecological importance of spring-fed streams and their resilience to climate warming, designating a class of groundwater-dominated thermal regimes is critical to developing long-term conservation strategies for cold-water ecosystems. Data from additional spring-fed sites would better describe the variability across exurgent volumes and temperatures.

Do dams "reset" the longitudinal pattern of a stream's thermal regimes?

Our study shows that dams do not reset thermal regimes: rather, they create an artificial regime generally not found in unregulated reaches that may persist for 10s or even 100s of kilometers downstream from a dam's outlet (Lowney 2000, Olden and Naiman 2010). The differences between regulated and unregulated regimes (excluding groundwater-dominated regimes) are illustrated by their annual magnitudes and variability. Variable cool regimes occurred exclusively in unregulated reaches, had more variable annual patterns (i.e., warmer annual maximums and cooler minimums), and had more predictable annual means, maximums, and day of annual maximum than stable cool regimes in regulated reaches – as a result of this variability, the

sine model was a poorer fit for unregulated sites compared to regulated sites. Stable cool regimes were strongly influenced by upstream dams and showed less annual variability, but higher variability among the three classifying metrics. Thus, although the overall annual pattern was more stable in regulated reaches, the annual mean, annual maximum, and day of annual maximum varied more within this regime than in variable warm and cool regimes. This variability may relate to storage capacity, operational objectives, or geography (Maheu et al. 2016b, Isaak et al. 2020). Stable cool regimes transitioned to variable warm regimes as the downstream distance from a dam increased. Variable warm regimes generally occurred at least 40 km downstream of dam outlets and may reflect a transition from regulated influences to dynamic equilibrium, when stream temperatures are dominated by heat flux due to ambient meteorological conditions (Lowney 2000). Additional analysis of thermal regimes given modeled, unregulated stream temperatures (e.g., Null et al. 2013) could show the historic fate of thermal regimes over stream reaches currently dominated by dam releases and identify if the transition to equilibrium included regimes similar to those produced by dams, or a more gradual shift in annual mean from variable cool to variable warm regimes.

Most notably, while stable cool regimes successfully mitigated potentially elevated summer stream temperatures, they similarly constrained winter minimum temperatures and maintained artificially warm conditions. Research on the effects of dam regulation on stream temperatures tends to focus on the summer season (Olden and Naiman 2010), when elevated stream temperatures may lead to stress or increased mortality of cold-water species (McCullough 1999, Moyle 2002). Fewer studies have focused on the potentially negative effects of sustained periods of elevated winter temperatures on cold-water species (Richter and Kolmes 2006); we are unaware of studies that focus on the potential to replicate colder winter patterns with dam regulation.

Can dams be managed to replicate desirable cold-water regimes?

In stream reaches that lack resilience to climate warming, cool- and cold-water habitat may be unachievable through dam regulation. In particular, the stable cool regime may present the greatest challenge to cold water conservation as it generally lacks the cooler winter temperatures of unregulated variable cool regimes. One notable result was the classification of the Shasta Dam outlet (site SHD) – the only reservoir to produce a stable, cold thermal regime. At 4.6 million acre feet (MAF), Shasta Lake is California’s largest reservoir and maintains its cold pool both through cold-water inflows, cooling that occurs during the winter, thermal stratification, and operational decisions (Nickel et al. 2004). Despite the large capacity of New Melones (2.4 MAF, 4th largest reservoir in California), it, or any other dam included in this analysis, was unable to produce a stable cold regime at its outlet. Thus, the storage volume and operational objectives of Shasta Dam may define the threshold below which stable cold thermal regimes cannot be produced.

Dams that impound or block access to natural groundwater-fed reaches seem unlikely to support the stable cold regimes through dam releases, and may be incapable or providing thermal regimes to support migratory cold water species. In addition, this study only considered the effect of the nearest upstream dam to a study site. Many streams have several dams, perhaps with compounded thermal effects (Ward and Stanford 1983). For dams that lack both the capacity to produce a stable or variable cold regimes and lack passage above the dam, these barriers may be insurmountable for species’ recovery. While reservoir operation to support cold-water habitat has shown promise (Yates et al. 2008, Kiernan et al. 2012, Phillis et al. 2018), our results suggest that improving passage or dam removal may be needed to reunite species with thermal regimes in which their life history strategies evolved. Potential constraints are considerable, though, given the fundamental shift in underlying, unregulated thermal patterns as a result of climate warming,

particularly in mid-elevation streams (Null et al. 2013). Where passage above a dam is infeasible, dam removal might make the most sense in watersheds with streams that are resilient to climate warming and have the most viable, long-term, and unregulated cool- and cold regimes.

Nevertheless, groundwater-fed spring and dam-influenced reaches showed several similarities. Streams like the McCloud River, Pit River, Battle Creek, and Big Springs Creek are highly influenced by groundwater-fed springs, and were the only above-reservoir reaches to replicate the same stable cool thermal regimes found below reservoirs. The regulated reach below Shasta Dam also illustrates a distinct antinode-node pattern characteristic of large-volume, groundwater-fed streams (Lowney 2000, Nichols et al. 2014). Despite similar thermal regimes, other research has shown how other aspects of these groundwater-dominated streams differ from runoff and regulated reaches (Lusardi et al. 2016), historically out-produced non-groundwater-dominated streams (Lusardi et al. 2016, Lusardi et al. 2018), and, for the streams still accessible to fish, are preferentially selected (Lusardi et al. 2018, Phillis et al. 2018). Thus, other factors, such as water quality (nutrients), physical habitat, flow regime, and novel ecohydrological feedbacks may still make spring-fed reaches more desirable habitat than regulated reaches despite their similar thermal regimes.

Thermal regimes and conservation

Conservation planning for cold-water species can be a risky investment in California. The combination of California's location at the southern range of cold-water species, vulnerability to climate warming, and highly regulated streams all pose challenges to the long-term success of conservation strategies that prioritize stream temperature conditions. Extinction is likely for most (78%) of California's native salmonids; though altered or degraded thermal regimes are a major stressor, they are not the only limitation (Katz et al. 2013). Bold conservation actions are required to reverse the trend towards extinction. This study shows that such actions are worthwhile as high-quality, cold-water habitat is likely to persist in certain areas of the state. Future work should integrate ecological data for cold-water species and compare their historic thermal landscapes to currently available habitat, particularly where current habitat is restricted to regulated reaches. As more long-term data becomes available, the thermal regime classification developed in this study can be used to identify new areas where conservation investment will support the recovery and persistence of valued native species.

Acknowledgments

Funding for this work was provided by the S.D. Bechtel, Jr. Foundation and John Muir Institute for the Environment. The authors confirm there are no real or perceived financial conflicts of interests. Data and code are available via GitHub repository: https://github.com/ucd-cws/streamtemp_classification. We thank Jay Lund, Alexander Forrest, and Steven Sadro for their valuable comments; this manuscript is greatly improved by their reviews.

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CHAPTER 3: SEASONAL AQUATIC MACROPHYTES REDUCE WATER TEMPERATURES VIA A RIVERINE CANOPY IN A SPRING-FED STREAM

Abstract: Maximum water temperatures in streams throughout the western USA typically occur in late summer and early autumn, coinciding with low stream flow. However, in the spring-fed Big Springs Creek in northern California, where constant-temperature groundwater springs provide relatively stable stream flow throughout the year, peak water temperatures and maximum diurnal variability occur in spring. We attribute this anomaly to the riverine canopy provided by emergent aquatic macrophytes (e.g., *Polygonum hydropiperoides* and *Nasturtium officinale*), which mimics the shade function of a riparian canopy. Macrophyte biomass increased 264% between January and August 2011. This increase coincided with a 111% reduction in flow velocity and a 53% increase in stream depth. Solar radiation was reduced by an average of 88% in patches of macrophytes that covered ~50% of the water surface during the summer. Decreased solar radiation reduced rates of stream heating, maximum temperatures, and temperature variability. We tested the riverine canopy hypothesis analytically based on a 2-dimensional hydrodynamic and water temperature model. The model predicted that emergent aquatic macrophytes reduce maximum water temperatures by an average of 5.1°C ($p < 0.001$) during late summer, when water temperatures in northern California streams typically increase. Our study shows the influence of a riverine canopy on naturally occurring temporal patterns of water temperature in a spring-fed stream. Our results could inform basin-scale management or regulatory strategies to address water-temperature conditions.

Key words: riverine canopy, water temperature, aquatic macrophyte, spring-fed, model, conservation, management

Citation: Willis, A. D., A. L. Nichols, E. J. Holmes, C. A. Jeffres, A. C. Fowler, C. A. Babcock and M. L. Deas (2017). Seasonal aquatic macrophytes reduce water temperatures via a riverine canopy in a spring-fed stream. *Freshwater Science* 36(3): 508-522.

Introduction

Water temperature is a principal determinant of habitat suitability for juvenile salmonids and other organisms in lotic ecosystems (Bjornn and Reiser 1991, Caissie 2006). Elevated water temperatures have been linked to increased stress and mortality of juvenile salmonids that rear in freshwater lotic habitats throughout the summer months (Marine and Cech 2004, de Brabandere et al. 2007). Therefore, many management and recovery efforts for threatened or endangered salmonids have been focused on reducing elevated summer water temperatures (Poole et al. 2004, Richter and Kolmes 2005). Regulatory personnel commonly use water temperature metrics (e.g., 7-d running average of daily maximum temperatures) to assess macrohabitat quality during all freshwater salmonid life stages (USEPA 2003). However, threshold-based water temperature targets often do not include complex spatial and temporal water temperature dynamics that have important implications for regulatory compliance strategies or mitigating factors, such as food availability, that have direct effects on the thermal tolerance thresholds of coldwater fish (Poole et al. 2004).

Stream temperatures are expected to rise throughout western North America in response to warming under climate change (Null et al. 2013). Coldwater fishes are anticipated to respond to such warming trends by altering life-history strategies, such as contracting their ranges by moving to higher-elevation reaches with cooler water temperatures when migratory pathways are present (Isaak and Rieman 2013, Eby et al. 2014). However, spring-fed streams may provide a unique hedge to this anticipated range contraction. Under most climate-change scenarios, groundwater inflows to spring-fed streams are anticipated to remain generally stable and cool, albeit with reduced volumes (Tague et al. 2008). Thus, spring-fed streams that are accessible to salmonids may be ideal targets for conservation efforts focused on salmon recovery.

Spring-fed streams can exhibit water temperature patterns that are temporally and spatially unique. During the summer, spring-fed streams in the western Oregon Cascades were cooler, exhibited less temperature variability, and were less sensitive to variations in air temperature than streams with water sourced from runoff and shallow subsurface flow (Tague et al. 2007). Unique spatial patterns of thermal nodes and antinodes occur near the source of thermally stable groundwater inputs to a northern California spring-fed stream (Nichols et al. 2014). Such observations suggest that features of spring-fed streams, such as stable baseflow and constant-temperature water sources, are important to understanding how water temperatures in such streams may respond to climate change or river-restoration activities.

Observations from Big Springs Creek, a spring-fed stream in northern California, suggest that seasonal growth patterns of aquatic macrophytes may be an important, but often overlooked component of water temperature dynamics. The seasonal growth of emergent aquatic macrophytes in rivers influences habitat conditions in myriad ways. Interactions between macrophytes and streamflow influence channel hydraulics (Green 2005a, Bal et al. 2011, O'Hare et al. 2011), sediment transport and depositional processes (Madsen et al. 2001, Gurnell et al. 2010, O'Hare et al. 2011), nutrient dynamics (de Brabandere et al. 2007), and water quality (Madsen and Cedergreen 2002, Wilcock et al. 2004).

However, these interactions also could influence water temperatures. The relationships among stream temperature, channel hydraulics, and riparian canopy are well-established (Poole and Berman 2001, Caissie 2006, Webb et al. 2008). Channel hydraulics influence stream temperatures by changing heating capacity, either via changes to the volume of water (i.e., inflows and outflows), streambed heat exchanges, or narrowing and deepening by altered channel forms (Caissie 2006, Webb et al. 2008). Riparian vegetation is credited primarily with reducing thermal

loading to stream channel by blocking solar radiation from the water surface (Poole and Berman 2001, Cassie 2006, Webb et al. 2008). However, riparian canopies are less effective moderators of stream temperature in wider stream channels because the canopies block a smaller portion of the channel (Poole and Berman 2001).

Where riparian canopies are absent or ineffective, riverine canopies created by aquatic macrophytes may offer an important, analogous function. Aquatic macrophytes are quintessential “ecosystem engineers” (Jones et al. 1994, O’Hare et al. 2012), and are particularly influential in lowland, spring-fed streams with largely stable flow regimes sourced from large groundwater springs (Champion and Tanner 2000). However, despite the important roles of channel hydraulics and shading in regulating spring-fed stream temperatures (Whitledge et al. 2006), the effects of seasonal emergent macrophyte growth on water temperature patterns have not been widely studied. Microthermal gradients were observed in patches of aquatic macrophytes in Dorset rivers (UK), but these variations were limited to patches that covered a few centimeters to meters of stream channel (Clark et al. 1999) and the authors did not explore system-scale aquatic macrophyte communities and their potential effects on water temperature. System-scale studies across a range of stream sizes in Australia and Austria show that patchy shade plays a significant role in moderating maximum water temperatures (Rutherford et al. 2004, Kalny et al. 2017), but these studies were limited to riparian, not riverine, canopies.

We used a combination of empirical and analytic assessments to explore relationships among the seasonal growth of aquatic macrophytes in Big Springs Creek, hydraulic conditions, and water temperature patterns. Our objective was to identify the critical factors that influenced water temperature to help inform on-going conservation activities in Big Springs Creek and the downstream Shasta River. We used empirical data to quantify relationships among aquatic macrophytes, channel hydraulics, reduction in solar radiation by macrophyte shading, and water temperature. Our hypothesis was that seasonal aquatic macrophyte growth moderated maximum water temperatures during a period when they typically reach their annual peak by creating a riverine canopy. We developed a 2-dimensional (2-D) hydrodynamic and water temperature model to test the dynamics observed in field data over a more extensive spatial and temporal domain. Investigators have used theoretical and empirical models to examine the influence of aquatic macrophytes on flow conditions (Champion and Tanner 2000, Green 2005b, Gurnell 2014) and water quality (Cox 2003, Srivastava et al. 2008), but we are not aware of studies in which investigators quantified the system-scale influence of aquatic macrophytes on water temperature. Our goal was to quantify the relationship between emergent aquatic macrophytes and water temperature and to improve understanding of potentially effective approaches to managing elevated water temperatures in spring-fed streams. Our results can be applied broadly to streams that support extensive aquatic macrophyte growth and may be considered for conservation actions or water-resource management based on water temperature metrics.

Background

Big Springs Creek is a 3.7-km-long tributary to the Shasta River in Siskiyou County, California (USA; Figure 3.1). It is characterized by relatively large volumes ($\sim 2.3 \text{ m}^3/\text{s}$) of cool ($10\text{--}12^\circ\text{C}$) spring-fed discharge (Nichols et al. 2014). Rainfall and snowmelt contributions are negligible. Modest surface-water diversions ($<0.3 \text{ m}^3/\text{s}$) and regional groundwater extraction occur between 1 April and 1 October (Null et al. 2010, Nichols et al. 2014). The creek is entirely contained on 2 private properties, both of which support cattle-ranching activities, and has been the focus of restoration efforts because of its robust potential to support coldwater fishes in the creek and downstream in the Shasta River for tens of kilometers (Jeffres et al. 2009, Null et al. 2010, Nichols et

al. 2014). These efforts have been focused on reducing elevated water temperatures during spring and summer.

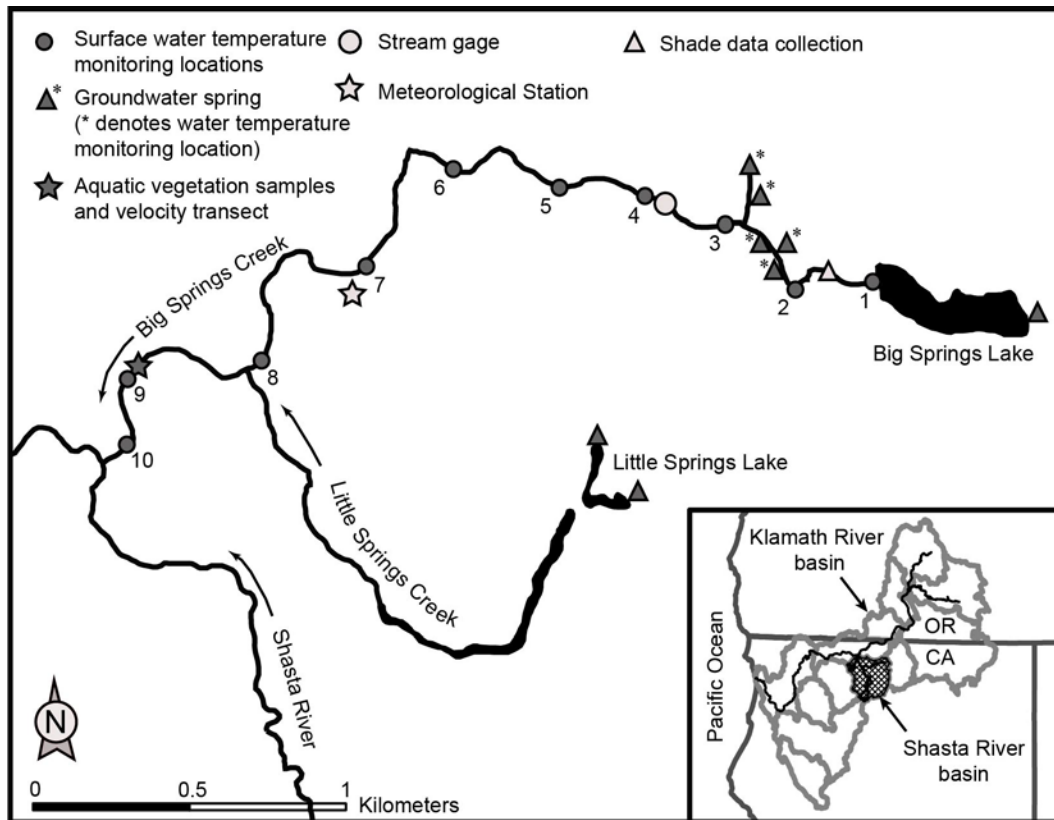


Figure 3.1. Map of the study area and monitoring sites.

Historical land use practices included unrestricted cattle grazing in the stream channel. Previous investigators have illustrated the relationship between livestock grazing and elevated water temperatures (Belsky et al. 1999, Agouridis et al. 2005), primarily via removal of riparian vegetation. Prior to restoration actions in Big Springs Creek, maximum daily water temperatures peaked $>25^{\circ}\text{C}$ (Jeffres et al. 2009, Nichols et al. 2014). Since restoration activities were initiated in 2009 to exclude cattle from the stream channel, annual maximum stream temperatures have declined as much as 4°C (Willis and Deas 2012), and generally remain within optimal growth ranges for juvenile rearing salmon and trout (USEPA 2003). The stream supports an array of anadromous salmonids, including Coho Salmon (*Oncorhynchus kisutch*, a federally threatened species), autumn-run Chinook Salmon (*Oncorhynchus tshawytscha*), and Steelhead Trout (*Oncorhynchus mykiss*) (Willis et al. 2012).

Before the start of restoration activities on Big Springs Creek, a monitoring program was designed and implemented to track the response of a broad array of physical, chemical, and biological variables (Jeffres et al. 2009). The objective of the monitoring program was to characterize baseline conditions of each variable, support long-term monitoring to track each variable's response to conservation actions, and identify areas where targeted, short-term experiments could improve understanding of key processes in the stream. The monitoring program also was designed to support the development of a 2-D, depth-averaged hydrodynamic and water temperature model to test a range of potential conservation activities across an extensive spatial and temporal domain. Solar radiation loads are a principal component of the heat budget because of

limited woody riparian vegetation and high width:depth ratios throughout Big Springs Creek (Nichols et al. 2014). The model was developed in 2 dimensions to capture spatial, volumetric, and thermal variability of groundwater spring inflow sources and to enable simulations of potential conservation activities that could alter channel forms substantially to reduce incoming radiation (Jeffres et al. 2009). Analysis of data collected in 2011 suggested that the seasonal growth of aquatic macrophytes moderated water temperature dynamics (Willis et al. 2012) and provided the basis for our study.

Methods

Aquatic macrophyte biomass

Aquatic macrophytes were harvested from 6 randomly selected sampling locations along a 100-m stream reach extending downstream from Big Springs Creek site 9 at river kilometer 0.19 (Figure 3.1) on each of 4 sampling dates in 2011: 20 January, 29 March, 24 May, and 22 August. Previous transect surveys of % cover and aquatic macrophyte species provided by The Nature Conservancy (unpublished data) and systematic photographic documentation of aquatic macrophyte growth conditions throughout Big Springs Creek since 2009 suggested that site 9 was representative of average macrophyte conditions in the creek. Visual evidence of previous harvests prevented the duplication of sampling locations. At each sampling location, all above-streambed biomass rooted within a 0.37-m² polyvinyl chloride (PVC)-frame quadrat was harvested. Samples were agitated in the stream to reduce the presence of clinging macroinvertebrates and other detrital material, then placed in individually labeled bags and returned to the laboratory. In the laboratory, plants were dried at 65°C for ≥ 72 h and weighed. Samples were then combusted in a muffle furnace for 4 h at 475°C, cooled, and reweighed to derive ash-free dry mass (AFDM). Mean standing macrophyte stock from each sampling date was reported as g AFDM/m². Cohen's *d* was used to explore the effect size between periods of minimum and maximum biomass. It was calculated by subtracting the means of macrophyte biomass samples collected in January and August 2011, and dividing that value by the pooled standard deviations of the samples (Cohen 1988).

Macrophyte shading and % cover

Shading effects associated with aquatic macrophytes were quantified based on measurements of solar radiation made with a solar pyranometer (Kipp and Zonen CMP 3, directional error <20 W/m²; Kipp and Zonen, Delft, The Netherlands). Hourly measurements were taken between 1000 and 1600 h at the water surface at 2 locations along Big Springs Creek on 20 July and 23 August 2011 (Figure 3.1). Each location was selected to assess a cover type that was typical throughout the stream: open water or emergent aquatic macrophytes. For each cover type, solar radiation values were reported for each sampling event as was % solar radiation reduction relative to solar radiation measured over open water.

Percent cover was quantified based on a combination of surveys and photographs to estimate the areal extent of emergent macrophytes during periods of maximum biomass. Where access permitted, surveys were conducted at 18 cross sections distributed throughout Big Springs Creek on 7 September 2011. At 2-m increments across each channel cross section, a 1-m² quadrat was surveyed visually for species present and macrophyte cover (aquatic macrophytes that emerged above the water surface) to the nearest 10%. Data for all sampling points were averaged to calculate % cover for each cross section. Calculated % cover was compared to visual estimates

based on photographs of each site to establish a consistent approach for locations where surveys were not permitted. Upstream and downstream photographs were taken from the middle of the stream channel to provide images of transects. These transect images were used to make visual estimates of macrophyte cover as per field surveys. Monthly photographs were taken at all locations to monitor the progression of emergent growth (i.e., growth above the water surface) from 1 April through 30 September 2011.

Channel hydraulics

Flow velocity measurements were made systematically across a single channel cross section 20 m upstream from the biomass sampling location (site 9; Figure 3.1). Measurements typically were completed during aquatic macrophyte sampling and occurred on 19 January, 23 March, 2 June, and 23 August 2011. Beginning at the stream margin, velocity sampling locations were identified at 1.0-m horizontal increments. At each sampling location, 6 vertical velocity measurements were collected: immediately above the stream bed and below the water surface, and at intermediate depth increments of 80, 60, 40, and 20 of total measured water depth. Velocities were measured using a Marsh–McBirney Flomate 2000 (Marsh–McBirney, Frederick, Maryland), which has an accuracy of $\pm 2\%$ of the total velocity measurement and ± 0.02 m/s at 0 flow. Two-dimensional velocity profiles were created in ArcMap (version 10; Environmental Systems Research Institute, Redlands, California) using inverse distance weighted (IDW) interpolations. Mean cross-section velocities for each sampling date were calculated by dividing mean daily discharge (obtained from an upstream gage) by cross-sectional area. The quotient of cross-sectional area and wetted width was used to represent mean channel depth. Manning's n was calculated for the cross section during each sampling event as:

$$n = \frac{\sqrt[3]{R \cdot S}}{v} \quad (\text{Eq. 1})$$

where R is hydraulic radius (m), S is bed slope (derived from channel long profile survey) (m/m), and v is mean cross-section velocity (m/s).

Manning's n was calculated for vegetated and unvegetated portions of channel because of the spatial variability of aquatic macrophytes in the sampled cross section. The hydraulic radius for each vegetated or unvegetated portion of the cross section was calculated as:

$$R = \frac{A}{2d+w} \quad (\text{Eq. 2})$$

where A is area of vegetated or unvegetated portion of the channel (m^2), d is mean water depth (m), and w is width of vegetated or unvegetated portion of the channel (m).

Mean flow velocities were calculated from point-velocity measurements collected in the vegetated or unvegetated portions of the cross section, whereas bed slope values were unchanged from those used in the total cross-section Manning's n calculation. Vegetated and unvegetated values were applied in the hydrodynamic model to represent lateral and longitudinal variable roughness in areas of aquatic macrophytes or open channel, respectively.

Hydrology

Stream flow in Big Springs Creek was quantified using standard discharge measurement and computational methods (Rantz 1982). A stream gage was established at site 4 (Figure 3.1), which was upstream of a grade-control structure and minimized the effects of aquatic macrophytes on stage. Access to discharge-monitoring cross sections was limited to monthly sampling events. Monthly measured discharges and continuous river stage data collected with a Global Water WL-16 submersible pressure transducer (Global Water, College Station, Texas) were used to quantify a stage–discharge relationship based on a power function. This relationship was used subsequently with continuous river stage data to develop a continuous (10-min interval) streamflow time series for Big Springs Creek. We used the hydrologic data to quantify hydraulic parameters and to define boundary conditions in the hydrodynamic model.

Channel geometry

Big Springs Creek channel morphology was characterized based on topographic survey data collected in 2008. We used these data to develop the 2-D hydrodynamic and water temperature model. Local field topographic surveys were completed using a TOPCON HiPer Lite+ Real-Time Kinematic Global Positioning System survey unit (Topcon Positioning Systems, Inc., Livermore, California), which has a horizontal and vertical accuracy of <2 cm. Longitudinal profiles of the channel bed and water surface were conducted along the channel thalweg while wading. In addition, 64 channel cross sections were surveyed across straight reaches and at meander bend apexes throughout Big Springs Creek. Each cross-section survey contained ≥ 13 points, with survey-point densities greater at locations with higher topographic variability. Cross-section surveys repeated at selected monitoring locations in 2011 identified only minor changes in bed topography, indicating that the more comprehensive 2008 cross-section survey data adequately represented topographic conditions throughout Big Springs Creek in 2011.

Meteorology

Meteorological conditions were monitored at 30-min intervals at a meteorological station near site 7 (Figure 3.1). Air temperature, relative humidity, precipitation, wind speed, wind direction, and solar radiation were monitored with a Campbell Scientific WXT520 weather station (Campbell Scientific, Inc., Logan, Utah). We used these data to implement the water temperature model.

Water temperature

Water temperature was monitored at 5 springs and 10 stream locations throughout Big Springs Creek (Figure 3.1) for the period 1 April–30 September 2011 with HOBO® Pro v2 water temperature data loggers (Onset Computer Corporation, Bourne, Massachusetts). The data loggers have an accuracy of $\pm 0.2^\circ\text{C}$ over the range from -40 to 50°C and recorded at 30-min sampling intervals. We used these data to implement and calibrate the water temperature model.

2-D hydrodynamic and water temperature model application

We generated a 2-D, depth-averaged, finite-element hydrodynamic and water temperature numerical model of Big Springs Creek with the aid of the RMA suite of finite-element modeling software (RMA-2, version 8.1b; RMA-11, version 8.7f; Resource Modeling Associates, Sydney, Australia). We simulated hourly flow and water temperature conditions for the period 1 April through 30 September 2011. We used cross-sectional topographic data to generate a bathymetric

map of the creek with Surfer (version 8; Golden Software, Golden, Colorado). A finite element mesh was generated using RMAGEN, from which we simulated flow (RMA-2) and water-temperature (RMA-11) conditions at hourly time steps. We applied hourly flow and water-temperature boundary conditions at all inflows (dam release and springs) and used hourly meteorological data. We represented aquatic macrophytes in model elements (Figure 3.2) based on a roughness coefficient (Manning's n) and shading (solar radiation reduction) factor. We used data from the macrophyte cover surveys and photographic images to define the distribution of aquatic macrophytes in the model and to determine which elements represented macrophytes that were submerged below or emerged above the water surface. Only elements representing macrophytes that emerged above the water surface were assigned an average solar radiation reduction factor based on the shade monitoring results (88% solar radiation reduction). To simplify the computational process, we used the average shade calculated from all measurements and assumed that elements representing submerged macrophytes have negligible shading effects. We assigned elements devoid of aquatic macrophytes a roughness coefficient associated with an open channel (i.e., nonvegetated) estimated from the channel hydraulics monitoring. For model elements representing mixed-substrate open channel, Manning's $n = 0.07$; for macrophytes, Manning's $n = 0.3$. Other substrates present included bedrock, tules, and willows, with Manning's $n = 0.02$, 0.3, and 0.02, respectively. We assigned a rock berm in one portion of the model domain a value of 0.5.

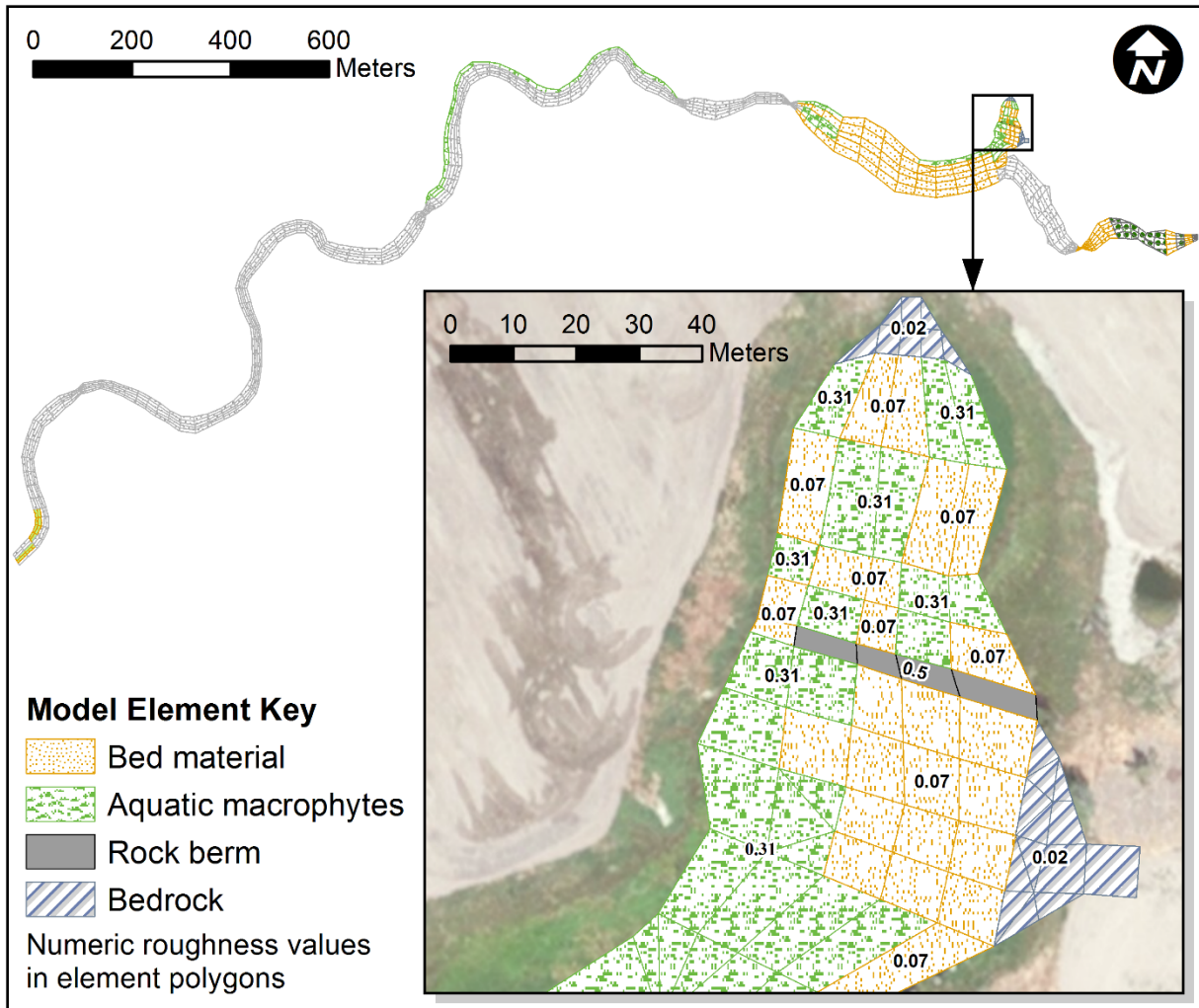


Figure 3.2. A graphical representation of the model mesh. Each element is assigned a roughness and shade factor to represent shaded or open channel. Cold numbers show roughness values for each element type.

We simulated 3 configurations to assess the relationship between aquatic macrophyte growth and water temperature (Table 3.1): no aquatic macrophytes, submerged aquatic macrophytes, and seasonal emergent aquatic macrophyte growth (i.e., representative of observed conditions). For each scenario representing aquatic macrophytes, we assigned roughness and shade elements based on observed distribution of aquatic macrophyte growth. We used the monthly photographs taken to monitor the progression of emergent aquatic macrophyte growth to identify the period when seasonal growth shifted from predominantly submerged aquatic macrophytes to emergent aquatic macrophytes. To simplify the computational process, we represented shade as a binary function: no shade was simulated from 1 April through 30 June (pre-emergent), and shade was simulated from 1 July through 30 September (post-emergent).

Table 3.1. A summary of the three configurations simulated using the 2D hydrodynamic and water temperature model.

Configuration	Roughness elements	Shade elements
1. No aquatic macrophytes	bed material (n=0.07)	open channel (no shade)
2. Submerged aquatic macrophytes	aquatic macrophytes (n=0.3), bed material (n=0.07)	open channel (no shade)
3. Seasonal emergent aquatic macrophyte growth	aquatic macrophytes (n=0.3), bed material (n=0.07)	aquatic macrophytes (88% solar radiation reduction), open channel (no shade)

We compared hourly simulation results for 1 April through 30 September ($n = 4392$) to observed data at 10 locations distributed longitudinally throughout Big Springs Creek, and analyzed the comparisons to assess model performance via statistical metrics and a graphical review of results. Statistical metrics used for calibration included mean bias, mean absolute error (MAE), and root mean square error (RMSE) (Maidment 1993, Null et al. 2010). We used mean bias to identify systematic over- or under-prediction by the model (mean bias $< 1^{\circ}\text{C}$ was desirable). We used MAE to segregate different management actions for coldwater fish. A resolution of $< 1^{\circ}\text{C}$ was sufficient for those decisions. Management applications were tested in a study subsequent to this preliminary model development and assessment. RMSE was used to identify large deviations from observed conditions (desired model performance was $\text{RMSE} < 1.5\text{MAE}$). In selecting these targets, we considered criteria established for other hydrodynamic and water-temperature modeling applications in the Shasta Basin (Null et al. 2010) and other hourly, deterministic water-temperature models (Caissie 2006), including those that met the criteria for regulatory applications (NCRWQCB 2006). In addition, the targets reflected the need for increased accuracy to assess the major physical processes controlling water temperatures on a refined spatial and temporal scale. We plotted simulated results and observed data to examine performance graphically throughout the time series.

Once the major physical processes were identified via calibration, we further analyzed results by comparing the magnitude, timing, and variability of maximum water temperatures for each configuration to the calibrated results. We compared changes in magnitude based on daily maximum water temperatures simulated for each configuration. We statistically analyzed for pairwise differences between means with a Tukey multiple comparison procedure. Significance was declared at $\alpha = 0.05$. We used the timing of the maximum weekly maximum water temperature (MWMT; the annual maximum of the 7-d average of daily maximum temperature [7DADM]) to identify the seasonal shift from increasing to decreasing maximum water temperatures. We analyzed the variability of daily maximum water temperatures based on the standard deviation (SD) of the 7-d moving average.

Results

Aquatic macrophyte growth influences on flow velocity, depths, and shading

During the monitoring period, emergent macrophyte species (e.g., *Polygonum hydropiperoides* and *Nasturtium officinale*) progressively emerged from the water column while submerged species (e.g., *Myriophyllum sibiricum*) occupied the understory (Figures 3.3A–3.3B). In

general, all identified species were native. Macrophyte biomass was lowest in the winter (mean = 35.5 g AFDM/m²) and highest in the late summer to early autumn (mean = 390.9 g AFDM/m²) (Figure 3.4A). This strong, positive seasonal growth pattern was confirmed with a simple exponential regression model ($R^2 = 0.9886$; Figure 3.4A) based on sample means and was further supported by a large (1.77) Cohen's *d* value (effect size).

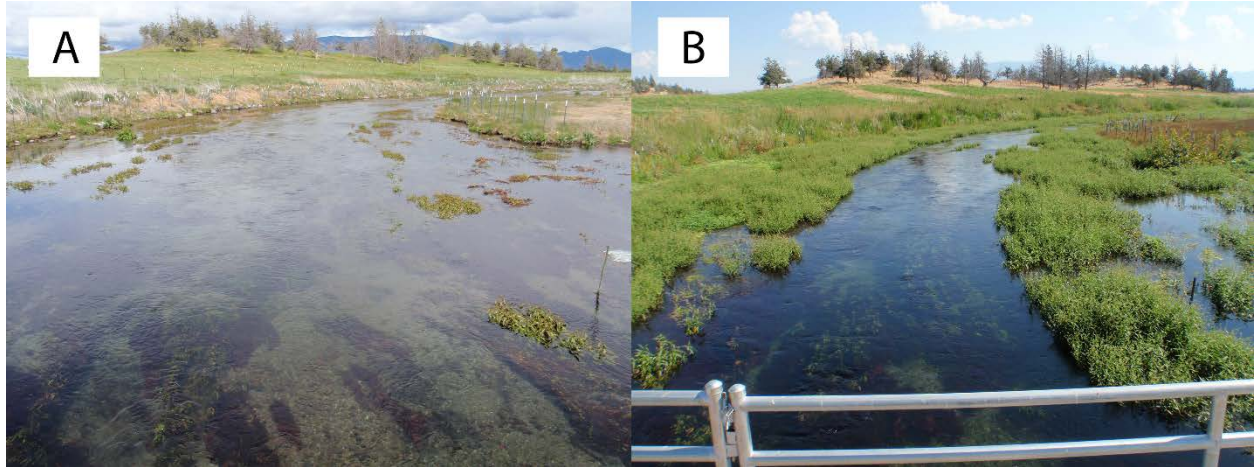


Figure 3.3. Seasonal macrophyte growth in Big Springs Creek in (A) May and (B) September 2011.

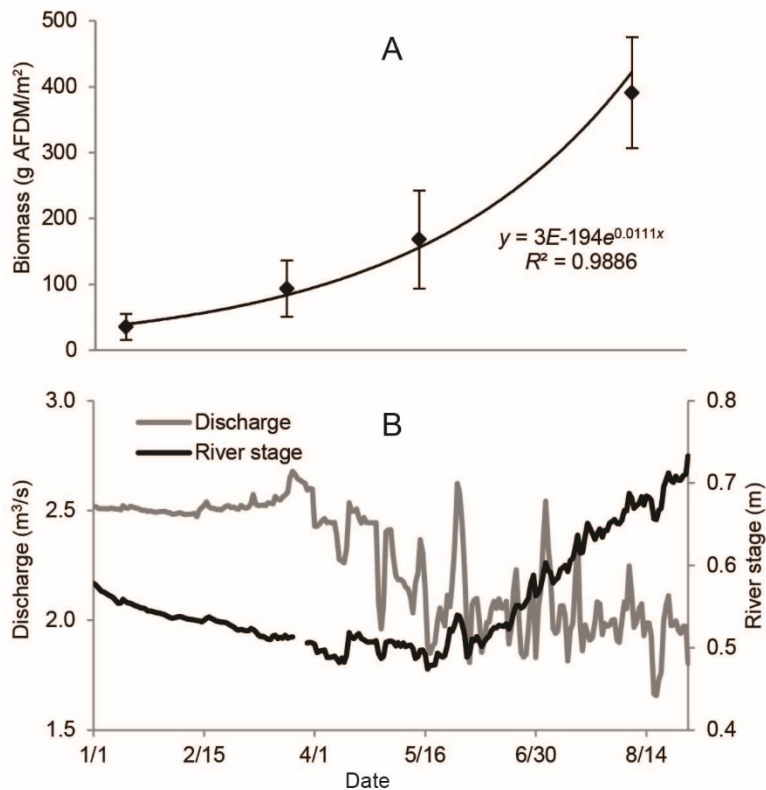


Figure 3.4. Mean (\pm SE, $n = 6$) standing crop of aquatic macrophytes (A) and continuous river stage and discharge (B) measured during the 2011 sampling period. AFDM = ash-free dry mass. Dates are formatted m/dd.

Seasonal aquatic macrophyte growth progressively slowed flow velocities throughout the growing season (Figure 3.5A, B). Flow velocities during early spring (March and April) averaged

0.38 m/s, and decreased to a mean of 0.18 m/s in the late summer (August and September). A logarithmic relationship ($-439.6 \ln x - 417.27$; $R^2 = 0.8378$) correlating cross section-averaged velocity and measured macrophyte biomass ($n = 4$) confirmed the negative relationship between velocity and biomass. River stage (depth) was slightly negatively correlated (Pearson's $r = -0.47$) with streamflow magnitudes in Big Springs Creek (Figure 3.4B). Average cross-sectional Manning's n values at the sampling site ranged from 0.08 in March 2011 to 0.21 in August 2011.

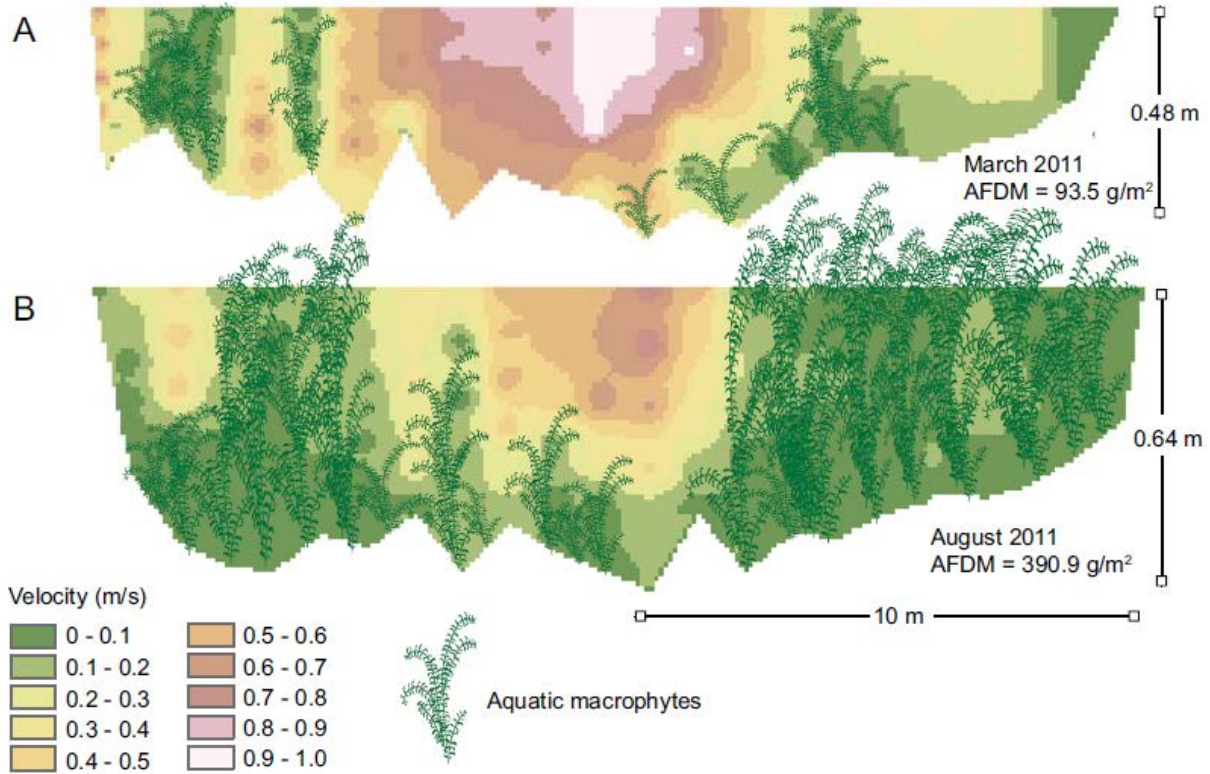


Figure 3.5. Flow velocity contour profiles for the Big Springs Creek sampling site, representing periods of seasonal minimum (A) and maximum (B) macrophyte biomass in 2011. Approximate locations of aquatic macrophytes within the water column are illustrated. AFDM = ash-free dry mass.

We next examined how emergent aquatic macrophytes influenced stream temperature by reducing incoming solar radiation. Emergent macrophytes reduced solar radiation loads in vegetated channel areas by an average of 88% (Table 3.2). Aquatic macrophytes covered 51% of the water surface in Big Springs Creek, but average % cover for each cross section ranged from 14 to 90%.

Table 3.2. A summary of solar radiation (SR) measured at the water surface of Big Springs Creek under open (i.e., unimpaired) and aquatic macrophyte (i.e., shaded) areas.

Date and time (PST)	Open channel	Aquatic macrophyte	
	Solar radiation (W/m ²)	Solar radiation (W/m ²)	Shade (%)
7/20/11 9:00	700	35	95
7/20/11 10:00	866	134	85
7/20/11 11:00	970	151	84
7/20/11 12:00	1020	165	84
7/20/11 13:00	991	179	82
7/20/11 14:00	934	180	81
7/20/11 15:00	821	176	79
8/23/11 9:00	605	46	92
8/23/11 10:00	769	45	94
8/23/11 11:00	867	86	90
8/23/11 12:00	919	82	91
8/23/11 13:00	915	22	98
8/23/11 14:00	845	114	87
8/23/11 15:00	684	52	92
Average shade			88

2-D hydrodynamic and water temperature model development and application

We applied the 2-D hydrodynamic and water temperature model to resolve the spatial and temporal discontinuities of the empirical data and to explore the relationship between water temperature dynamics and aquatic macrophyte influence on hydraulics and shade. Model simulations that neglected the roughness and shade effects of aquatic macrophytes failed to reproduce observed water temperature timing and magnitudes at multiple sites along the creek within the performance criteria (Table 3.3). For all simulations and calibration sites, RMSE performance criteria were satisfied (RMSE < 1.5MAE). However, for the ‘no aquatic macrophytes’ and ‘submerged aquatic macrophyte’ simulations, mean bias and MAE performance criteria were not met at all sites. Mean bias and MAE criteria were met at sites near boundary conditions (sites 1–3), but performance failed to meet the identified criteria by site 4, and showed progressively poorer performance at downstream locations. For both simulations, water temperatures were consistently overestimated, with MAE as high as 1.9°C. However, when both shade and roughness attributes of seasonal emergent aquatic macrophyte growth were represented in the model, performance criteria were met at all longitudinal locations.

Table 3.3. A summary of performance statistics for the Big Springs model for simulations that represent roughness and/or shade features of seasonal aquatic macrophyte growth, as well as a control simulation that includes neither roughness nor shade features.

Site	River km	1. no aquatic macrophytes			2. submerged aquatic macrophytes			3. seasonal emergent aquatic macrophytes		
		Mean bias (°C)	MAE (°C)	RMSE (°C)	Mean bias (°C)	MAE (°C)	RMSE (°C)	Mean bias (°C)	MAE (°C)	RMSE (°C)
1 ^a	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	3.3	-0.4	0.6	0.7	-0.3	0.6	0.7	-0.8	0.8	1.0
3	3.0	0.6	0.8	1.0	0.6	0.8	1.0	0.3	0.5	0.7
4	2.7	1.0	1.3	1.8	1.1	1.3	1.8	0.5	0.8	1.0
5	2.6	1.0	1.3	1.9	1.1	1.4	1.8	0.4	0.8	1.0
6	2.3	1.0	1.4	1.9	1.0	1.4	1.8	0.3	0.8	0.9
7	1.5	1.1	1.6	2.4	1.3	1.7	2.2	0.3	0.8	1.0
8	0.8	1.2	1.8	2.7	1.3	1.8	2.4	0.2	0.9	1.1
9	0.4	0.8	1.7	2.5	0.9	1.5	2.0	0.2	0.8	1.1
10	0.0	0.7	1.9	2.7	0.9	1.5	2.0	0.1	0.9	1.1

^aBoundary condition

Graphical examination of model results suggests that MAE in the ‘seasonal emergent aquatic macrophyte’ configuration have two main causes: shifts in phase timing (i.e., the timing of daily maximum and minimum water temperatures) and poorer performance during April and May. We examined observed data and simulated results at the mouth of the creek (site 10; Figure 3.1) to assess the cumulative effect of the major physical processes that influenced water temperature in Big Springs Creek and to minimize local effects associated with proximity to discrete groundwater-fed spring sources. We explored phase timing by comparing the timing of observed daily maximum water temperatures at site 10 (Figure 3.1) with simulated daily maxima. The ‘seasonal emergent aquatic macrophyte’ configuration generally shifted the timing of simulated daily maximum temperatures later by an average of 0.7 h over the simulation period (Figure 3.6)—a difference of <1 time step. Phase timing differed more often during the 1 April through 30 June simulation period, prior to the simulated onset of seasonal shading by emergent aquatic macrophytes. Relatively poorer performance during this period suggests that a more refined representation of seasonal aquatic macrophyte growth (e.g., monthly) may be desirable.

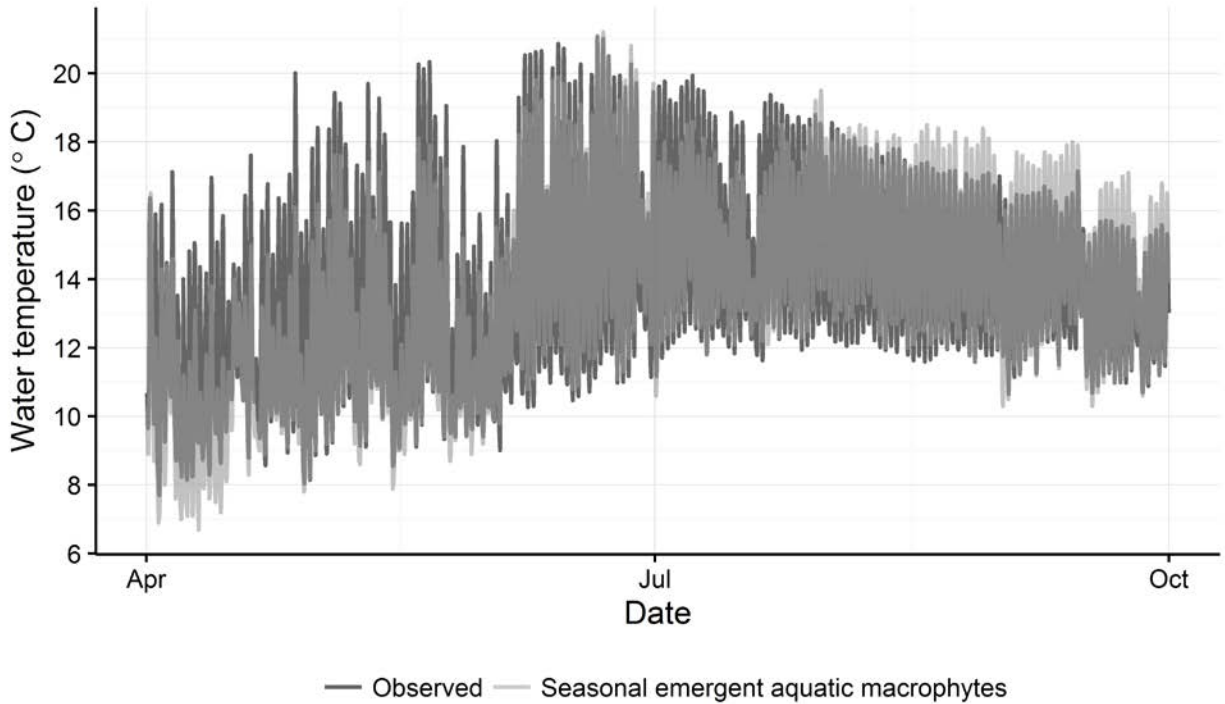


Figure 3.6. Observed and simulated water temperatures at site 10, near the mouth of Big Springs Creek.

Once an analysis of the model simulations confirmed that hydraulic and shade features of emergent aquatic macrophytes play an important role regulating seasonal water temperatures, we used additional analyses at site 10 to quantify their relative influence on daily maximum water temperatures. Daily maximum water temperature results for each configuration were compared for the period 1 July through 30 September 2011, the period during which shade was simulated in the 'seasonal emergent aquatic macrophyte' configuration. The Tukey multiple comparison procedure showed that hydraulics and shade each significantly affected daily maximum water temperatures ($p < 0.001$). The hydraulic effects of submerged aquatic macrophytes reduced average daily maximum water temperatures by 1.2°C (95% confidence interval [CI] = 1.1–1.3°C) (Figure 3.7). When shade was applied, average daily maximum water temperatures were an additional 3.9°C (95% CI = 3.8–3.9°C) cooler. In total, shade and hydraulic effects of seasonal emergent aquatic macrophytes reduced daily maximum water temperatures in Big Springs Creek by an average of 5.1°C.

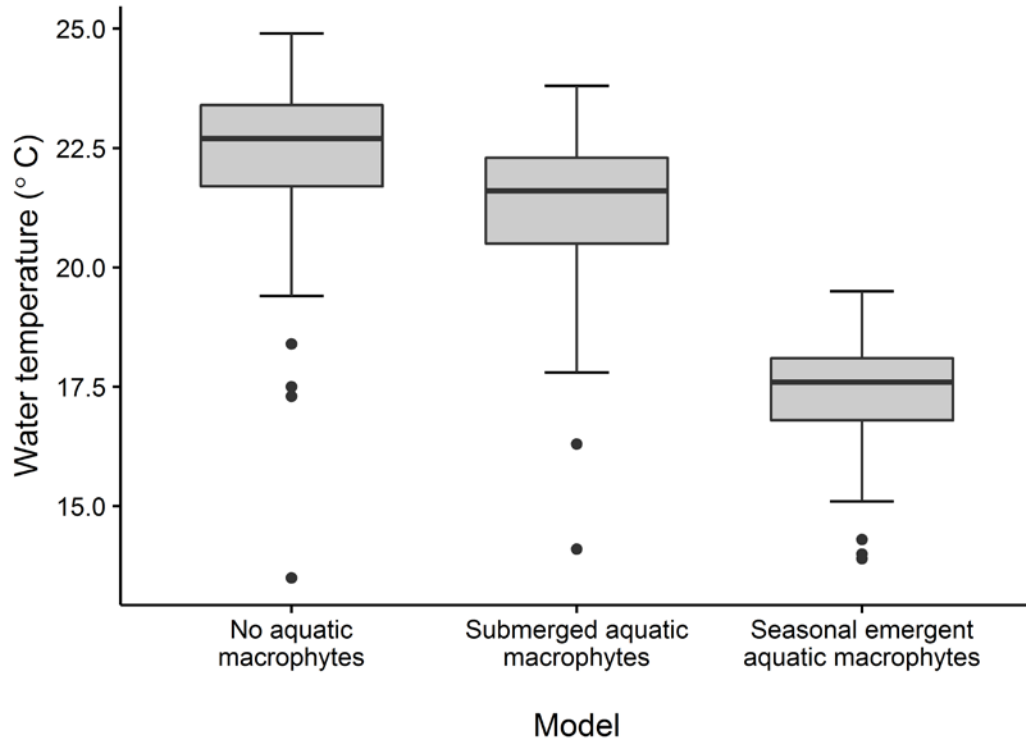


Figure 3.7. Box-and-whisker plots of the daily maximum water temperatures at site 10 for each aquatic macrophyte configuration comparing discrete effects of hydraulic and shade features of aquatic macrophytes. Lines in boxes are medians, box ends are quartiles, whiskers are 10th and 90th percentiles, and circles show outliers for each configuration's modeled results. Results are for the period 1 July to 30 September 2011.

A review of 7DADM time series for each model configuration suggest that seasonal emergent aquatic macrophytes also affect the timing of the seasonal shift from warming to cooling and the variability of the 7DADM. The timing of the seasonal shift from warming to cooling was identified based on the date of the MWMT. In the 'no aquatic macrophytes' and 'submerged aquatic macrophytes' simulations, the MWMT occurred on 29 July 2011 (Figure 3.8). However, the results of the 'seasonal emergent aquatic macrophytes' were substantially different for the timing and magnitude of MWMT. When shade features of emergent aquatic macrophytes were taken into account, MWMT occurred on 21 June 2011, 39 d earlier than the scenarios in which the effects of emergent aquatic macrophytes were neglected. Last, the variance in water temperature was analyzed using the SD of the 7DADM. In the 'seasonal emergent aquatic macrophyte' simulation, the SD prior to the onset of shade (i.e., only hydraulic effects of aquatic macrophytes were active) averaged 1.6°C. After shade was applied (i.e., both shade and hydraulic effects were active), the average SD decreased to 0.7°C.

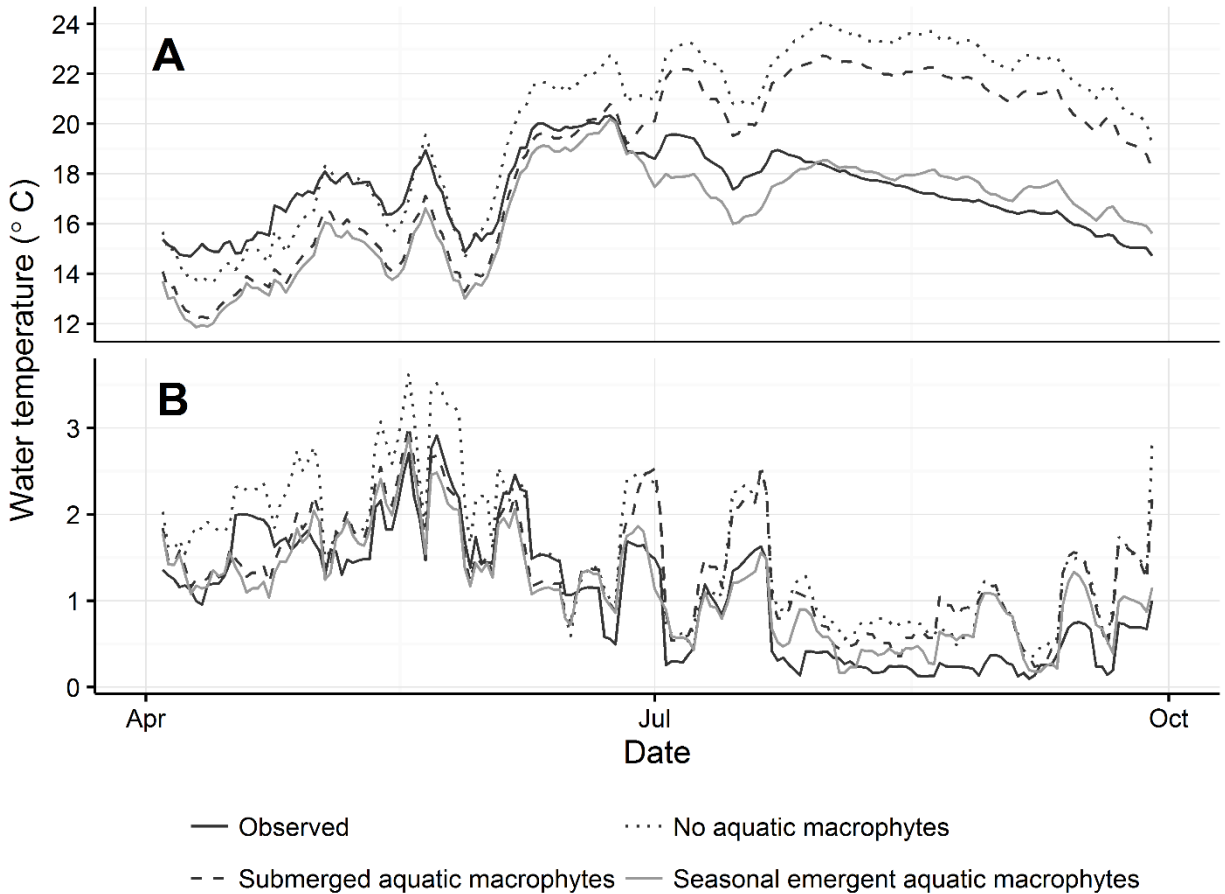


Figure 3.8. Seven-day average daily maximum temperatures (7DADM) (A) and standard deviation (SD) for 7DADM (B) for observed data and each configuration's modeled results.

Discussion

Riverine canopy

We investigated whether seasonal aquatic macrophyte growth can reduce seasonal water temperature trends in a spring-fed stream. Emergent aquatic macrophytes are not typically considered a significant component of a stream's thermal dynamics, but our results illustrate that aquatic macrophytes play an important role in regulating stream temperatures. Specifically, our results support the hypothesis that seasonal aquatic macrophyte growth moderates maximum water temperatures via the shading and hydraulic effects of its riverine canopy.

Field data illustrated how seasonal growth of aquatic macrophytes is a major factor controlling water temperature conditions in Big Springs Creek. Biomass results and large Cohen's *d* values provided evidence of large, positive changes in aquatic macrophyte biomass throughout the 2011 growth season, resulting in substantial hydraulic and shading effects. Velocity reductions were induced by increased channel roughness associated with macrophyte growth (de Doncker et al. 2009), an observation largely confirmed by a general positive correlation between channel depth (normalized by streamflow) and biomass. The slightly negative correlation between stage and stream flow magnitudes suggest that macrophyte-induced roughness was more dominant than

streamflow magnitude in controlling channel depth. This field-based evidence indicates that seasonal increases in biomass and associated macrophyte roughness decreased flow velocities and increased channel depths throughout the macrophyte growing season in Big Springs Creek, whereas flow rates declined during the period of local and regional agricultural water use. As a result, maximum water temperatures showed a notably cooler pattern, with less variability, and a smaller diurnal range after emergence of aquatic macrophytes. In addition, water temperatures began to cool more than a month earlier than would have occurred without the riverine canopy.

Riverine canopies function similarly to riparian canopies, but have important and distinct advantages. Riparian canopies are most effective for narrow channels, but their ability to mitigate elevated water temperatures decreases as channel width increases (Poole and Berman 2001). Because riverine canopies occupy the channel itself, rather than being restricted to the banks, they mitigate solar radiation loading over a larger range of stream sizes. In addition, riverine canopies expand the category of streams that may be managed for water temperature to include those with limited ability to support riparian growth, but high potential for emergent aquatic macrophyte growth. Such streams have been identified in the UK and New Zealand (Clark et al. 1999, Champion and Tanner 2000, Tague et al. 2008, Nichols et al. 2014) as well as in northern California and the Oregon Cascades.

Our study illustrates the relationship between a riverine canopy and water temperature, but additional research is needed to explore the limits of this aquatic macrophyte-based process. The patchy distribution of aquatic macrophytes should be explored to estimate the density and distribution necessary for an effective riverine canopy. The 51% average cover provided by the riparian canopy exceeds the minimum coverage recommended for effective shade (Kalny et al. 2017), but the range of cover (14–90%) suggests that some areas may have insufficient macrophyte density to provide effective shade. In studies of patchy riparian shade in Australia and Austria, sudden decreases in vegetation density showed the potential to result in substantial water temperature changes that are likely to be ecologically significant (Rutherford et al. 2004, Kalny et al. 2017). In addition, the seasonal dynamics of macrophyte growth suggest that patchy areas of elevated water temperatures may occur early in the growing season when macrophytes are predominately submerged (Clark et al. 1999). Last, effective riverine canopies may be limited to systems that lack scouring flows (Chambers et al. 1991).

Modeling considerations

The role of the riverine canopy in water temperature dynamics leads to important considerations for model development. Simulating seasonal emergent growth was a critical component in the 2-D numerical hydrodynamic and water temperature model because of the effect of the riverine canopy on water temperatures in Big Springs Creek. Meaningful results were reproduced only when aquatic macrophyte characteristics (i.e., roughness and shade) were explicitly incorporated. These results further support the hypothesis that the riverine canopy affects the timing and variability of maximum water temperatures in Big Springs Creek.

However, the results also illustrated areas where future studies should improve upon the current understanding of riverine canopies and how they are modeled. Streams that are dominated by this process may require 2-D models to replicate aquatic macrophyte density and distribution sufficiently. This requirement would add to the monitoring, data, and computation requirements of a study. Temporal refinement is another area that would benefit from additional insight. Two configurations (pre-emergence and post-emergence) were sufficient to capture the general process. However, a coarse representation may not be sufficient for management decisions that target specific periods, such as the period of rapid change during the early growing season, which is

coincident with periods of potentially elevated water temperatures. Defining the management objective is critical to determining the appropriate level of spatial and temporal detail included in a simulation of a riverine canopy.

Management implications

The management implications for this riverine canopy–water temperature dynamic in Big Springs Creek are extensive. Cattle grazing, which was the primary cause of degraded stream habitat and elevated water temperatures in Big Springs Creek before restoration activities, has been estimated to degrade ~80% of stream and riparian systems in arid environments (Agouridis et al. 2005). The actual geographic scope of degradation may be higher once the systematic removal of the riverine canopy via livestock grazing is taken into account. Groundwater spring flows, channel geometry, and meteorological conditions are not factors that are easily, or even possibly, manipulated to change water temperatures to address this degradation. Aquatic macrophytes are more easily managed as part of a restoration strategy to improve water temperatures. Other approaches to providing shade along spring-fed creeks, like extensive riparian plantings, may be effective in the long-term, but probably would require decades to meet shading objectives (Caissie 2006). In comparison, passive recovery via aquatic macrophyte growth provides considerable short-term benefits, and should be considered in other waterways exhibiting emergent macrophyte growth.

The natural water temperature patterns observed in Big Springs Creek also suggest that additional restoration or water-management actions may be advisable during the spring period before the shading influence of the riverine canopy. In Big Springs Creek, this period of maximum heating coincides with the early rearing stages of recently emerged juvenile salmonids that are relatively vulnerable to the effects of exposure to elevated water temperatures (Marine and Cech 2004, Ebersole et al. 2006). Additional work based on the 2-D hydrodynamic water temperature model is underway to assess how water management alternatives could further ameliorate elevated water temperatures during the critical spring period.

Our study illustrates another important contribution by spring-fed streams, such as Big Springs Creek, to the long-term viability of coldwater species. Annual maximum water temperatures in Big Springs Creek occur in late spring and were measured 3.6 km downstream from coldwater sources. After emergence of the riparian canopy, stream temperatures generally remained below the recommended thresholds for juvenile Coho Salmon (USEPA 2003). This suggests that Big Springs Creek provides extensive, reach-scale coldwater habitat during a critical late-summer period when such habitat is typically limited in this watershed. Maintaining the passively restored condition is critical to supporting robust and resilient coldwater habitat, a key component of the long-term recovery and sustainability of coldwater fish like Coho Salmon.

The seasonal dynamics of aquatic macrophyte growth on water temperatures has important implications for regulatory management of Big Springs Creek. Big Springs Creek can influence water temperatures for tens of kilometers downstream from its confluence with the Shasta River (Nichols et al. 2014). Water temperatures have been identified as the key impairment limiting the survival of anadromous fish in the Shasta Basin. Spatially explicit total maximum daily loads (TMDLs) (NCRWQCB 2006) and single-value thresholds (Stenhouse et al. 2012) have been recommended to address thermal habitat degradation in the Shasta Basin, specifically for the federally threatened Coho Salmon. However, these regulatory criteria do not account for the seasonal changes in the rate of stream heating or potential effects of reduced velocities on the downstream extent of water exported from Big Springs Creek. Thus, conventional management or

conservation actions may not be effective because Big Springs Creek has an anomalous water temperature dynamic.

Acknowledgments

We thank the National Fish and Wildlife Foundation (Keystone Initiative Grant 2009-0077-000), the National Oceanic and Atmospheric Administration (ARRA Grant NA09NMF4630315), and the California State Water Resources Control Board (Grant 08-593-550) for funding this research. We also thank Irene Busk for providing access to her property. We thank Sarah Null and 2 anonymous referees for insightful comments and suggestions that greatly improved this manuscript.

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CHAPTER 4: EYE IN THE SKY: USING UAV IMAGERY OF SEASONAL RIVERINE CANOPY GROWTH TO MODEL WATER TEMPERATURE

Abstract: Until recently, stream temperature processes controlled by aquatic macrophyte shading (i.e., the riverine canopy) was an unrecognized phenomenon. This study aims to address the question of the temporal and spatial scale of monitoring and modeling that is needed to accurately simulate canopy-controlled thermal processes. We do this by using unmanned aerial vehicle (UAV) imagery to quantify the temporal and spatial variability of the riverine canopy and subsequently develop a relationship between its growth and time. Then we apply an existing hydrodynamic and water temperature model to test various time steps of canopy growth interpolation and explore the balance between monitoring and computational efficiencies versus model performance and utility for management decisions. The results show that riverine canopies modeled at a monthly timescale are sufficient to represent water temperature processes at a resolution necessary for reach-scale water management decisions, but not local-scale. As growth patterns were more frequently updated, negligible changes were produced by the model. Spatial configurations of the riverine canopy vary interannually; new data may need to be gathered for each growth season. However, the risks of inclement field conditions during the early growth period are a challenge for monitoring via UAVs at sites with access constraints.

Keywords: water temperature; thermal regime; UAV; riverine canopy; management; model; aquatic vegetation

Citation: Willis, A., and E. Holmes. 2019. Eye in the Sky: Using UAV Imagery of Seasonal Riverine Canopy Growth to Model Water Temperature. *Hydrology* 6(1):1-6.

Introduction

Stream temperature is a widely-studied feature of freshwater aquatic ecosystems (Caissie 2006, Steel et al. 2017). Stream temperature regulates organisms' metabolism, growth, phenology, survival, food webs, and community structure (Caissie 2006, Webb et al. 2008, Steel et al. 2017). Water temperature changes profoundly affect stream ecology, including nutrient processing capacity and food webs (Woodward et al. 2010, Davis et al. 2013). In addition, macroinvertebrates and other ectothermic organisms will move in both space and time as their preferred thermal regimes shift to increasingly constrained habitats (Harper et al. 2006, Isaak et al. 2017a). Because stream temperature is more closely correlated with air temperature than with discharge, streams are generally expected to warm with climate change. However, buffering sources are expected to moderate stream temperature into the foreseeable future (Webb et al. 2008, Isaak et al. 2017b).

While the significance of stream temperature and changes to thermal regimes are widely appreciated, the processes underlying unregulated thermal regimes are less well-defined (Dugdale et al. 2017, Steel et al. 2017). Thermal regime is a phrase used to describe patterns of magnitude, timing, duration, and frequency of change in a stream's water temperature patterns (Steel et al. 2017). Thermal landscapes are the spatial distribution of thermal regimes and are a product of the unique interactions between geography, hydrology, meteorology, climate, and myriad characteristics of the stream itself and its surrounding features (Steel et al. 2017).

The past few decades have seen an increase in empirical studies about the complex interactions that fundamentally control thermal regimes (Webb et al. 2008, Steel et al. 2017). Though monitoring methods have rapidly advanced, water temperature data quality ranges widely, making it hard to determine the underlying processes controlling thermal regimes (Webb et al. 2008, Hannah et al. 2015, Dugdale et al. 2017).

Shade is considered a major "second-order" control—behind first-order climate and hydrologic processes—on large-scale thermal regimes (Dugdale et al. 2017). The distinction between shade and cover is important, as each refers to different elements in thermal regimes. In this paper, shade refers to the amount of solar radiation reduction that results from cover over an area. Canopy refers to the amount of physical cover over an area. For example, while a quadrat of a stream's water surface might be 20% covered by riparian or riverine canopies, the light reduction in those covered areas (i.e., shade) might be 80%.

Stream temperature dynamics associated with shade have been long-recognized, though predominately in the context of riparian shading and the effects of forestry practices (Webb et al. 1996), and, to a lesser extent, snow and ice (Moore et al. 2006). Canopy cover has been associated with net cooling (Johnson et al. 2004, Garner et al. 2017) and reduced sensible and latent heat exchange (Garner et al. 2017, Moore et al. 2005). In addition to cooling via riparian canopies (Isaak et al. 2017b), riverine canopies that result from emergent aquatic vegetation have shown comparable solar radiation reductions to those achieved by riparian canopies (Kalny et al. 2017, Willis et al. 2017). Canopies that result in $\geq 70\%$ shade is the objective for temperature control (Garner et al. 2017, Rutherford et al. 2018).

Predictive (also called deterministic) water temperature models can provide useful insights to thermal processes (Dugdale et al. 2017, Steel et al. 2017). Early modeling studies emphasized the seasonal relevance of riparian shade to water temperature dynamics. In particular, the leaf-out and leaf-drop transitions in the riparian zone were simulated using coarse assumptions as an early example of process-based water temperature modeling (Sinokrot and Stefan 1993). Predictive models also possess several advantages over more simplified statistical modeling. Statistical

models can be useful to explore questions with limited data (Benyahya et al. 2007, Schabenberger and Gotway 2017). However, statistical models that cover broad spatial scopes overcome data limitations by relying on underlying process assumptions. Often, these process assumptions assume a close correlation between air and stream temperature (Benyahya et al. 2007, Isaak et al. 2017b)—a sometimes erroneous assumption and unreliable simplification when studying streams affected by human activities (Webb et al. 2008, Steel et al. 2017). Statistical models have limited abilities to identify specific mechanisms in the temperature process and can be unreliable as surrogates for streams that differ in either space or time (Dugdale et al. 2017).

Predictive models also have significant disadvantages. Such models tend to be data-intensive, restricting their application to larger spatial scales (Dugdale et al. 2017). When simulating riparian (or riverine) vegetation, the data provided to the model must be of a resolution equal to or better than the representation provided in the model. Historically, this has resulted in either coarse representations of canopy cover (Willis et al. 2017), time-intensive manual mapping (Trimmel et al. 2018), or expensive data collection methods such as light detection and ranging (LiDAR) (Loicq et al. 2018). Finally, predictive stream modeling methods are not widely adopted by managers, suggesting that currently available methods need improvement to become more widely accessible (McGrath et al. 2017).

Because stream temperature is more closely correlated with air temperature than with discharge (Wondzell et al. 2018), streams are generally expected to warm with climate change (Van Vliet et al. 2011, Arora et al. 2016, Ptak et al. 2016, Null et al. 2013). However, buffering sources are expected to moderate stream temperature into the foreseeable future (Webb et al. 2008, Isaak et al. 2017b); shade is predicted to be a potentially significant buffer (Garner et al. 2017, Trimmel et al. 2018, Wondzell et al. 2018). Though temperatures in cold streams are projected to increase less than warm streams, streams with small temperature changes may see a large biological response if they are located near warm-edge or cold-edge boundaries of thermal niches (Isaak et al. 2017b). Cold-water ecosystems in California, such as the Shasta watershed, are considered the lower boundary for species such as salmon (Moyle et al. 2017). Given the more frequent extreme thermal conditions expected with climate change and their subsequent effects on temperate species (Vasseur et al. 2014), understanding the mechanisms with which to mitigate those events is critical to the viability of cold-water ecosystems.

Extensive temperature modelling and analysis has occurred in the lower Shasta watershed to explore past and current stream conditions as they relate to salmon (Null et al. 2010, Willis et al. 2015, 2017; Nichols et al. 2014). Big Springs Creek is a spring-fed tributary to the Shasta River that influences the quality and extent of cold water habitat for tens of kilometers downstream of its confluence (Nichols et al. 2014). Strategic investments in habitat restoration have partially offset warming due to previous land and water use decisions (Null et al. 2010, Willis et al. 2017) and may further mitigate expected warming due to climate change (Isaak et al. 2017b). The primary objective of these investments has been to improve overwintering habitat quality and extent for federally and state-listed threatened coho salmon (*Oncorhynchus kisutch*). The result of these investments has been the fundamental shift of factors that control the thermal regime from meteorological conditions (Nichols et al. 2014) to a reach-scale riverine canopy created by aquatic plants (Willis et al. 2017). However, as the stream is located in an area where rangeland is the predominant land use (Null et al. 2010), additional questions remain regarding how stream flows may be managed to enhance desirable instream water temperature conditions.

The objective of this study is to explore the temporal and spatial scale of monitoring and modeling that is needed to accurately simulate thermal processes controlled by the riverine canopy and support management decisions. We do this by using unmanned aerial vehicle (UAV) imagery to

quantify the temporal and spatial variability of the riverine canopy, and subsequently develop a relationship between its growth and time. Then we apply an existing hydrodynamic and water temperature model using the refined canopy data and explore the results for changes in accuracy. The results of this study will help identify the balance between monitoring and computational efficiencies versus model performance and utility that are needed to support management decisions in streams where shade plays a major role in water temperature processes.

Materials and Methods

Study Site and Period

The study site included the 3.7 km reach of Big Springs Creek from the outlet of Big Springs Dam to the confluence with the Shasta River (Figure 4.1). The study period occurred between 1 April 2017 and 30 September 2017, during which the growing season coincided with the irrigation season on the ranches surrounding the creek. In addition, data were used from a previous flight in August 2015 to assess the spatial variability of interannual riverine canopy growth. Site access to the reach upstream from Site 2 (river kilometer (rkm) 2.7) extending to Site 1 (rkm 3.7) was limited to 1 day per month, with access dates negotiated with the landowner at least 6 weeks prior to the proposed sampling dates. Once agreed upon, access dates could not be rescheduled.

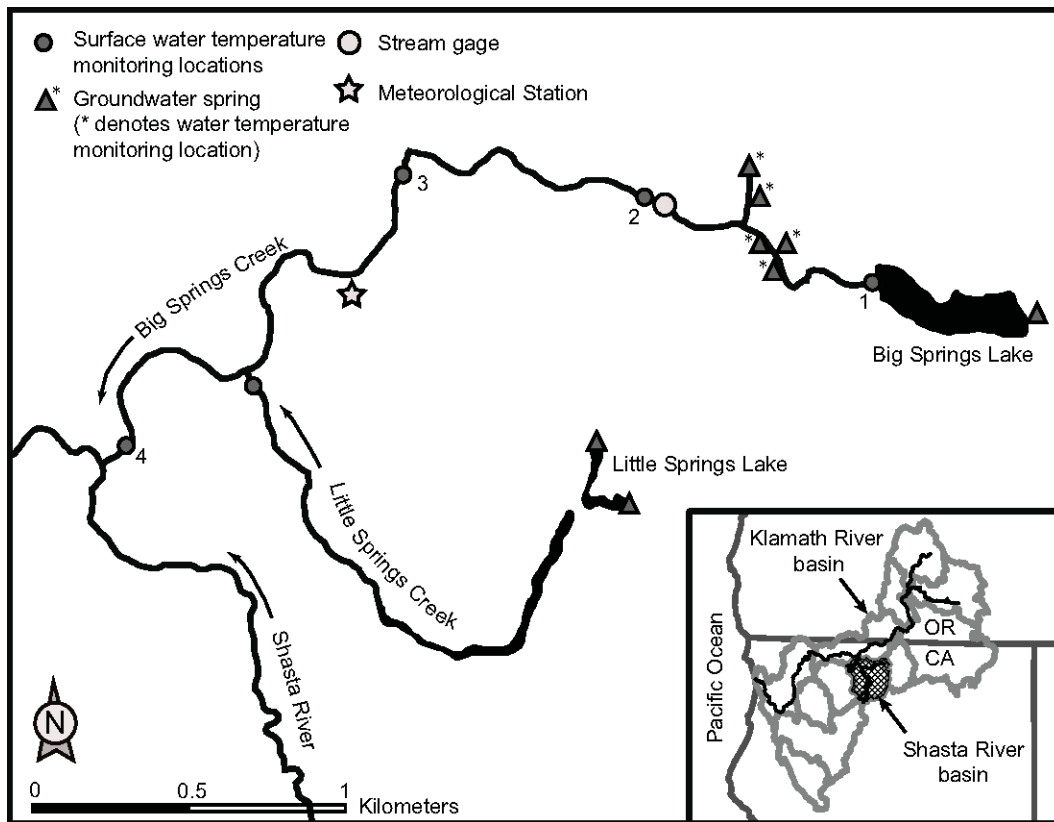


Figure 4.1. A map of the study area. Sites 1–4 were used to test water temperature modeling performance. Approximate centroid UTM coordinates are 10T 547,987.38 m E, 4,605,393.41 m N.

Riverine Canopy Surveys

Riverine canopy surveys were completed using a 3DR Solo Quadcopter, which was modified to attach a Canon Powershot S100 digital camera. A flight path of 84 north–south oriented transects covering 22.5 linear kilometers were flown from an altitude of 104 m, with a flight speed of 16 km per hour to achieve an approximate image side and end overlap of 70%. Control points were established at 36 locations by monumenting 13 cm² bolts upside-down in concrete, then surveying the top of the bolt using a Topcon Hiper V Real Time Kinematic GPS unit with 5 mm horizontal and 10 mm vertical accuracy. Control point targets were created using 0.6 m² wood boards that were painted white, then marked with 5 cm black lines across the diagonals, and finally had a 1.3 cm hole drilled in the center of the board. The targets were then mounted on the monumented bolts (targets were held in place using a washer and nut both above and below the board) to help visual identification of the control points in the UAV imagery. The camera was programmed to take images at a 5-s interval. Images were reviewed and adjusted for brightness, then stitched together using Agisoft Photoscan Professional (v 1.2.6, St. Petersburg, Russia). Completed Photoscan models were georeferenced by identifying the control point targets in the individual photos. Orthomosaic and digital elevation model (DEM) layers, each with a 0.05 m resolution, were then created and exported as georeferenced tiff files.

The spatial and temporal variability of riverine canopy growth was analyzed using supervised image classification and analysis of the tiff files in ArcMap 10.5 (Redlands, CA, U.S.A.). First, the orthomosaic image was clipped to include only the wetted channel. Also, polygons of a willow stand were made to mask the area from analysis of canopy extent due to aquatic macrophytes. Then, training samples of 40–50 merged polygons were created for each of two classes: open water and emergent aquatic vegetation. These training samples were then used to classify the clipped orthomosaic image of Big Springs Creek and to estimate the percent area covered by emergent aquatic vegetation. Misclassification was determined by extracting the classified image raster pixels in the training samples. Temporal changes were explored by comparing the percent area covered from one survey to the next during the 2017 monitoring period. Spatial changes were explored by comparing August surveys from 2015 and 2017, and analyzed to identify cover class areas that remained consistent.

Water Temperature Modeling

Once the temporal and spatial trends were analyzed, an existing hydrodynamic and water temperature model of Big Springs Creek (Willis et al. 2017) was used to simulate water temperature conditions given various frequencies of canopy growth interpolation. RMA-11 is a Fortran-based, proprietary model that has been applied in various water temperature studies (Lowney 2000, Dugdale et al. 2017, Willis et al. 2017). Riverine canopy surveys were used to develop element classes in the unstructured grid that represented various amounts of canopy cover using numerical representations of shade and roughness; no other changes to the grid were made so as to test model performance due solely to refined aquatic vegetation data.

The cover in each element was determined by extracting the classified image raster pixels of each class (open and cover) for the area covered by the element and calculating the percent cover for each element at each of the four survey dates. Weekly changes in percent cover for each model element were determined using a linear interpolation between survey points. To simplify the computational requirements of the numerical water temperature model, element cover types were binned into classes representing no cover (0–10% covered areas in the classified images), 20% cover (10–30%), 40% cover (30–50%), 60% cover (50–70%), 80% cover (70–90%), and full cover

(90–100% cover). A histogram of element classes was developed to review the trend of cover classes through the simulation period.

Shade for each element class was calculated by assigning the empirically observed solar radiation for covered areas (i.e., 12% of solar radiation was measured in covered areas; Willis et al. 2017) to the proportion of the element that represented its cover class, plus full solar radiation to the remaining area Equation (1).

$$\text{element shade} = (\text{percent cover} \times 0.12) + (\text{percent open} \times 1) \quad (1)$$

Roughness for each element class was similarly calculated, using empirically-based values for this site (Willis et al. 2017) Equation (2):

$$\text{element roughness} = (\text{percent cover} \times 0.31) + (\text{percent open} \times 0.07) \quad (2)$$

The water temperature model was run for a continuous period between 1 June and 15 August to simulate three different time-step adjustments to the riverine canopy: weekly, bi-weekly and monthly. Shade and roughness values were updated to reflect new values at the start of each step, with no smoothing applied. Results of each simulation were analyzed at four locations (Figure 4.1) using mean bias, mean absolute error (MAE) and root mean square error (RMSE), keeping with performance criteria developed for management decision-making applications (Null et al. 2010, Willis et al. 2017). RMSE was particularly useful as it remains unbiased by seasonal cycles (Dugdale et al. 2017); given the expected seasonal dynamics of canopy growth, controlling for seasonally-derived bias is a critical feature of this study.

Results

Riverine Canopy Surveys

Due to the long lead time necessary for scheduling access to the study site, field conditions were not always conducive to UAV flights. Inclement weather or wind speeds greater than 16 km per hour prevented riverine canopy surveys for the months of April and September. During the remaining visits, UAV flights of the entire reach were completed over 2 days (Table 4.1).

Table 4.1. A summary of the survey dates and percent cover of Big Springs Creek.

Survey	Flight Dates	Canopy Cover (%)	Canopy Cover (m²)
Survey 1	22–23 May 2017	38	61,668
Survey 2	19–20 June 2017	53	87,776

The supervised classification was able to distinguish between open channel and emergent plants (i.e., canopy) for all orthomosaic images produced via UAV monitoring (Figure 4.2). An analysis of misclassified pixels showed that the training samples were sufficient to classify cover type with 1.3% and 4.0% misclassification for open channel and emergent plant classes, respectively.

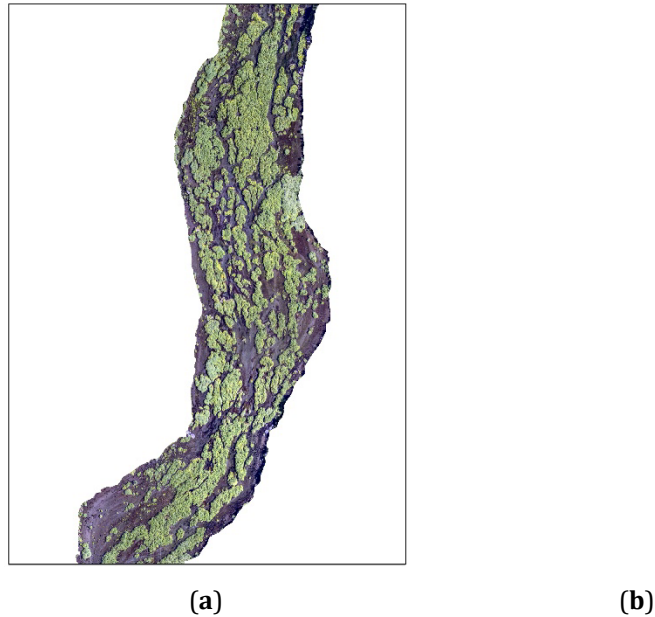


Figure 4.2. An example of the (a) orthoimagery; (b) results of the supervised classification, using data gathered during Survey 2 in June 2017.

Canopy cover changed both temporally and spatially in Big Springs Creek. Temporally, cover increased from 38% in May 2017 to 74% in August 2017 (36% increase), with the largest change occurring between June and July (18% increase) and the smallest change occurring between July and August (4% increase) (Table 4.1). For the spatial analysis, differences in the flight path used in 2015 resulted in poor image resolution or lack of coverage at the margins of the orthomosaic. Thus, of the total area surveyed in 2017, 16% (25,989 m²) could not be compared to data from the 2015 flight. Of the remaining area, the cover remained consistent over 66% (108,352 m²) of the stream from August 2015 to August 2017, while 16% (26,765 m²) shifted from the canopy to open channel and 2% (3,197 m²) shifted from the open channel to canopy (Table 4.2).

Table 4.2. A summary of the percent area that shifted cover classes from August 2015 to August 2017.

Class change	Area (m ²)	Area (%)
Canopy to open channel	25,989	16
Open channel to canopy	3197	2
No change	108,352	66
Area not analyzed	26,765	16

Water Temperature Modeling

The histogram of the element classes shows that during the beginning of the simulation period, element classes were dominated by areas with cover $\leq 40\%$ (Figure 4.3). As the simulation period progressed, classes were dominated by areas with cover $\geq 60\%$. While most classes showed steady trends either increasing or decreasing their frequency, the element class that represented 60% cover initially occurred more frequently, then declined. From its peak frequency, the 60% coverage class saw a net 30% transition to greater coverage classes. The highest and lowest cover

classes also showed indications of a plateau at the end of the modeled period, while areas in the 60% and 80% covered classes showed steady decreasing and increasing trends, respectfully.

Regardless of the frequency with which canopy growth was simulated, the water temperature model produced results that met the performance criteria (Table 4.3). Simulated water temperatures were generally warmer than observed water temperatures, as shown by the positive mean bias across all simulations. For all simulations, mean bias increased through location 3, then decreased towards the mouth, Site 4. This gradual increase in mean bias, then decline, suggests that better representation of local features such as groundwater inflow volumes may be necessary to apply the model for more refined management objectives. Mean absolute error (MAE) remained consistent at each site (with the expected exception of the boundary condition), showing no substantial changes in accuracy as the model progressed through the study area. Root mean squared error (RSME) also remained well within the $1.5 \times \text{MAE}$ threshold for all sites and simulations, indicating no anomalous, large errors.

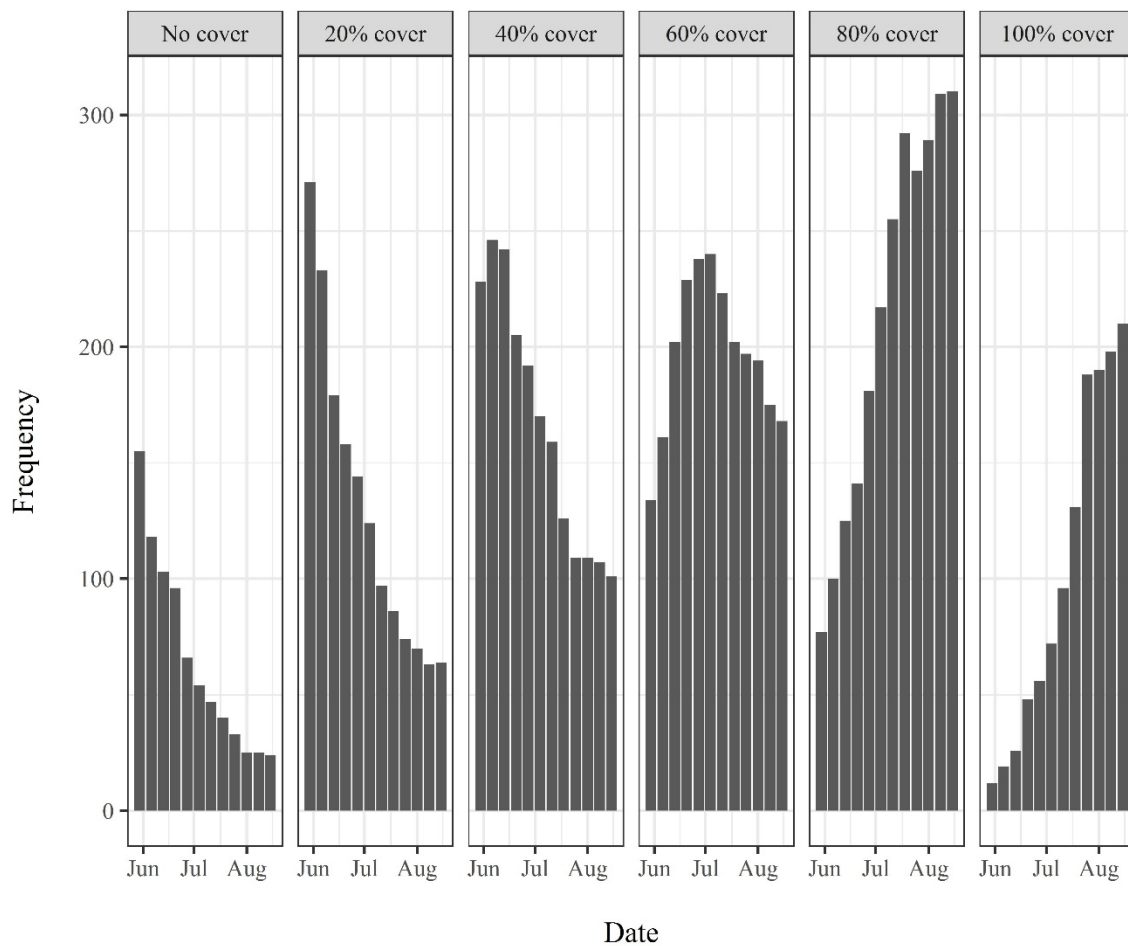


Figure 4.3. The distribution of element cover classes through the model simulation period.

Table 4.3. A summary of performance results for each simulation of water temperatures given various frequencies of interpolated canopy growth. All performance metrics are measured in °C.

Site	River Kilometer (rkm)	Weekly Growth			Biweekly Growth			Monthly Growth		
		Mean Bias	MAE ^b	RMSE ^c	Mean Bias	MAE	RMSE	Mean Bias	MAE	RMSE
1 ^a	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	2.6	0.3	0.9	1.2	0.3	0.9	1.1	0.4	0.9	1.2
3	1.7	0.6	0.8	1.0	0.7	0.8	1.0	0.7	0.9	1.1
4	0.0	0.3	0.7	0.9	0.4	0.7	0.8	0.5	0.7	0.9

^a Boundary condition; ^b Mean absolute error; ^c Root mean square error.

A closer review of the modeled water temperatures compared to observed water temperatures show where the likely sources of error occurred. Given the comparable performance of each simulation, only plots for the weekly growth simulation are presented and discussed. Though the performance metrics show a mean bias of up to 0.6 °C, the plots of modeled to observed water temperatures show that daily maximum water temperatures are generally overestimated while daily minimum water temperatures are well replicated (Figure 4.4). Site 1 represents a boundary condition of the model and is defined by the observed data at that site. At Sites 2 and 3, maximum water temperatures are generally over-estimated—a trend that remains consistent throughout the simulation period. At Site 2, observed water temperatures from 5 June to 13 June appear anomalous when compared to the other sites, suggesting that the observed data may not be an accurate record, contributing to the larger error at that site. By Site 4, the modeled diurnal water temperatures better match the observed record, with better agreement of both daily maximum and minimum water temperatures. However, agreement of modeled to observed daily maximum water temperatures declines towards the end of the simulation.

Sites 2–4 also show periodic underestimates of daily minimum water temperatures that occur coincidentally with the shift in coverage represented in the model. Similar results are shown in the bi-weekly and monthly results (results not presented). While the more frequent updates to canopy growth introduce more frequent errors for several hours after the new cover is introduced, results for the diurnal extremes (i.e., daily maximums and minimums) appear to be better simulated.

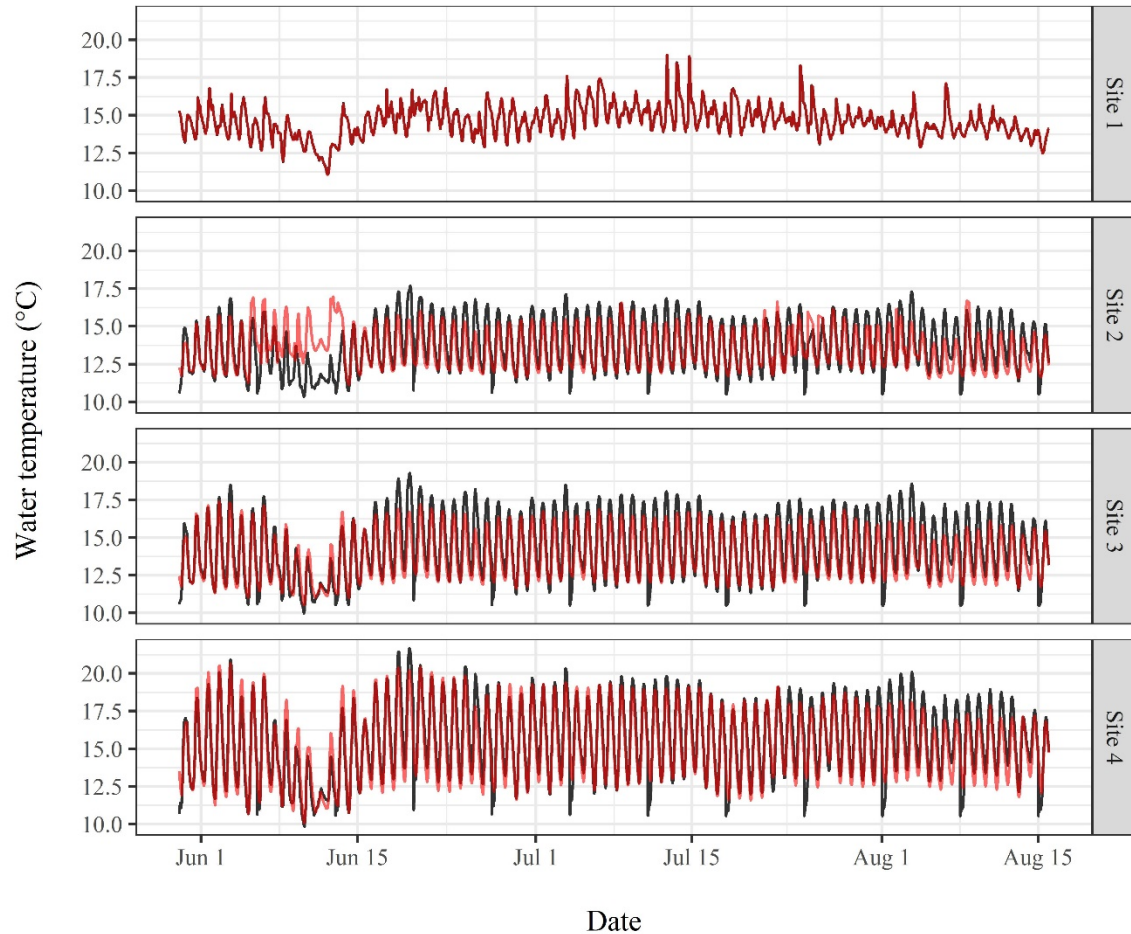


Figure 4.4. Plots of the modeled (black line) water temperature versus observed (red line) water temperatures for the weekly canopy growth simulation.

Discussion

As recently as the 1990s, local human activities were widely viewed as the dominant influence on thermal regimes in streams across the globe, rather than large-scale climate change (Webb 1996). By the early aughts, though, climate change was rapidly identified as a major driver of stream temperature changes (Webb et al. 2008). While many management strategies focus on water quantity, the relationship between riparian canopies and climate change have long been recognized as issues of greater consequence to thermal regimes (Webb 1996). Results from this and previous studies suggest that riverine canopies may play a similarly influential relationship in mitigating the predicted effects of climate change (Wondzell et al. 2018). The findings from this study have broader implications for three facets of water temperature management for cold-water ecosystems: extending canopy monitoring methodology, as shown by combining UAV and digital photography technology; the important role of riverine canopies in thermal regimes; and water temperature modeling for large-scale versus local management objectives.

UAV Survey Methods

The results of this study show that UAV survey methods provide an efficient approach (both in terms of equipment cost and person-time) to gather near-census data quantifying cover over a mid-sized stream. Prior to the use of UAVs for canopy cover monitoring, methods were limited by the spatial extent that could be covered due to most survey elevations, as well as the cost and time needed to perform the surveys. As recently as 2017, imagery taken from a height of 2–5 m above a stream's water surface was considered aerial imagery (Clark et al. 2008, Verschoren et al. 2017). Those survey altitude limits further constrained the size of stream that could be surveyed; at the upper bound of those aerial surveys, study sites were limited to streams up to 10 m wide (Verschoren et al. 2017). This study shows how utilizing the flight ability of UAVs greatly expands the area that can be surveyed: in 2 days, full surveys of Big Springs Creek were completed, which covered 3.7 km and included stream reaches as wide as 300 m (Nichols et al. 2014).

In addition, this study shows how digital photography methods used for low-elevation surveys can be extended by combining digital photography with UAV technology. Alternative methods that have been developed to assess cover, such as riparian surveys using canopy densitometers (Kelley et al. 2005), are impractical for riverine canopies, where emergent plants grow through and remain near the water surface. Digital photography provides a cost-effective approach that has been previously utilized for aquatic plant mapping at lower elevations (Verschoren et al. 2017); this study illustrates how similar methods are successful at higher survey elevations, extending this method to a wide range of stream sizes. Despite its classification as “low-spectral resolution,” the three-band (red-green-blue) survey was sufficient for classification accuracy, which makes this method a cost-effective alternative to others that use multi-spectral imaging.

Some limitations of this study can be overcome by standardizing the flight paths and survey extent of UAVs. A comparison of 2015 to 2017 data illustrated the need for wide spatial margins for survey areas, as well as the value in establishing repeatable flight paths to ensure reproducible and comparable survey areas. In addition, access limitations also highlighted how sites with more access flexibility are better suited to UAV monitoring, which requires dependable field conditions that typically are only well-forecast several days in advance. Finally, this method may benefit from additional ground-truthing by manually surveying randomly selected in-stream areas using an RTK or other comparable methods.

Riverine Canopy Growth

Understanding the thermal regime processes that will buffer against predicted climate change is critical to conserving and managing cold-water ecosystems. The results of this study show that riverine canopy processes can be well-characterized using monthly datasets and how its influence in the thermal regime becomes stronger as the canopy more fully develops throughout the growing season. Early in the growing season, the riverine canopy is dominated by relatively low-coverage areas: This period coincides with previous observations of annual maximum water temperatures in thermal regimes controlled by riverine canopies (Willis et al. 2017), when fish like juvenile coho may be more vulnerable to elevated water temperatures (Richter et al. 2005). Improving resolution around early season growth may be critical, as previous work has shown that water in and above submerged vegetation may be more sensitive to solar radiation than open-channel flow (Rutherford et al. 1993, Clark et al. 1999). As such, additional management actions may be necessary to mediate annual maximum water temperatures during those early growth periods.

As the growing season progresses, there's a transition across mid-coverage element areas as low-coverage areas become high coverage areas. Interestingly, lack of plateau in the 60% and 80% cover classes suggest that additional growth may be occurring and that the surveys completed in mid-August did not catch the transition from growth to senescence. Previous studies show that biomass continued to increase into September (Willis et al. 2017). Extending the surveys later in the year would help quantify the peak coverage provided by the riverine canopy, as well as the timing of when the canopy begins to senesce. As well as extending the data describing the seasonal trends of canopy growth, additional data to quantify the shade provided by the canopy would further improve understanding of those thermal regimes. As warm temperatures and dry conditions extend later into the year (Leung et al. 2004), understanding the full potential of the riverine canopy to act as a buffer against these conditions is critical to understanding the potential management challenges for thermally-sensitive ecosystems.

Once the riverine canopy transitions into predominately fuller coverage, its advantages over riparian shade as a solar radiation buffer are clear. Riparian shade needs both longer time frames and spatial scales to achieve similar effectiveness to riverine canopies (Kalny et al. 2017, Willis et al. 2017). Tree height and shape, channel width and shape (i.e., straight or meandering), and channel orientation are all factors that limit the effectiveness of riparian shade (Garner et al. 2017, Wondzell et al. 2018); riverine canopies have no limitations analogous to these riparian features. Despite these drawbacks of riparian canopies, research into the relationship between riparian cover and stream temperatures suggests useful considerations for future work. Microclimate changes due to extensive cover may shift energy fluxes in the heat budget and fluxes that are generally negligible in less densely covered reaches may become more influential in the overall thermal regime (Garner et al. 2017, Fabris et al. 2018). Examining the effects of canopy-controlled thermal regimes should include an analysis of daily extreme (i.e., maximum and minimum) water temperatures to ensure that sufficient minimum temperature conditions are maintained in streams targeted for salmonid or other cold-water species recovery. Also, the ability of aquatic plants to colonize 70% of the channel is consistent with the findings of other riparian studies for the cover extent needed to affect both temperature control (Garner et al. 2017, Rutherford et al. 2018) and macroinvertebrate recovery (Rutherford et al. 2018). Those findings are confirmed by the results of this and previous studies, which show that seasonal water temperatures in Big Springs Creek begin to cool in late June/early July (Willis et al. 2017), when the riverine canopy covers nearly 70% of the stream surface.

Given the larger stream-orders that may be affected by riverine canopies, vs. riparian, it would be useful to determine the geographic extent of these types of streams, as restoration of the riverine canopy process could influence water temperatures on the reach-scale (Nichols et al. 2014, Bartholow 2000) and mitigate for climate change (Wondzell et al. 2018). On a reach-scale, regression equations have been used to identify predictor variables for water temperature, such as riparian vegetation (Moore 2006), and could be useful tools to explore whether the riverine canopy-controlled thermal regime is representative of a class of rivers. Such findings could have important implications for mitigating the effects of climate change, as canopy-controlled thermal regimes may result in cooler stream temperatures than currently observed in spite of predicted climate warming (Garner et al. 2017, Wondzell et al. 2018). In addition, while this study focuses on the relationship between riverine canopies and thermal regimes, other studies have shown strong relationships between aquatic plants and channel hydraulics (Green 2005, Bal et al. 2011, O'Hare et al. 2011). However, because aquatic plants senesce each year, the role of riverine canopies and their seasonal effects on physical salmonid habitat, and, by extension, salmonid life history strategies, may show an interesting contrast to studies that focus on large woody debris and other semi-permanent features for cover and velocity utilization by juvenile coho (McMahon 1989, Lacey 2004).

Water Temperature Modeling

Finally, water temperature modeling is used to transform improved monitoring using UAV technology and the improved understanding of the role riverine canopies play in the thermal regime into a potential management tool. While model results show that monthly interpolated canopy growth is sufficient to model water temperatures, the performance metrics and comparative plots suggest that there is additional room for improvement. The negligible improvement that followed more refined temporal resolution of canopy growth suggests that further improvement is more likely to result from better representation of other processes in the thermal regime. Such processes include better representation of substantial groundwater inflows to the creek, both in the overall quantity and distribution of flows among discrete groundwater sources. Due to the dominant role that groundwater plays in spring-fed stream thermal regimes (Kurylyk et al. 2016, Caissie and Luce 2017), additional work is recommended to improve the understanding of the conductive and advective heat flux through the stream bed. Such work would also help clarify the issue of potential shifts in dominant heat flux processes given the microclimate effects of canopy cover. Also, the model showed some short-term (e.g., over a period of hours) sensitivity to the periodic update of canopy cover, and could benefit from additional refinement such as transitional smoothing between cover configurations. As such, the model is better suited for large-scale management objectives (e.g., managing water temperature conditions that are exported to the reach-scale habitat in the downstream Shasta River), but requires refinement before it could be confidently applied to managing the local habitat within Big Springs Creek.

Additional work that explores model performance in response to more refined grid structures would help illustrate the balance between computational efficiency and the data required to accurately simulate the heat exchange processes dictated by the riverine canopy. Models developed at fine spatial scales can be particularly useful for understanding the relationship between ecosystem dynamics and water temperature processes (Dugdale et al. 2017). Also, while the decision to use a proprietary model was influenced by considerable investments made in previous stages of this research, publicly available models would allow for more transparency. Future stages, particularly those with the objective of evaluating management decisions, should weigh the benefit of using currently available models against the desirability for more transparent, and potentially transferable, modeling methods.

Water resource and fisheries managers need to make decisions based on the thermal regime of a stream (Sinokrot and Stefan 1993), which may be controlled by factors other than stream flow or air temperature. In these cases, deterministic modeling may be necessary when longer-term datasets are unavailable, particularly where novel thermal processes have been identified. Future studies may want to explore statistical relationships between riverine cover and stream temperature to develop management tools that are less data-intensive than deterministic models.

Acknowledgments

We'd like to thank the Nature Conservancy and Irene Busk for providing access to their properties and permission to use the study site; Devon Lambert for his considerable field assistance; Ian King for his willingness to answer questions about implementing RMA; Sarah Yarnell for donating her UAV equipment for the surveys; and Carson Jeffres for his willingness to let Eric Holmes volunteer time and expertise for this study in addition to his valuable role in other projects. We'd also like to thank two anonymous reviewers for their thoughtful and detailed reviews, which considerably improved this manuscript. This research received no external funding.

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CHAPTER 5: CONCLUSION

Process-based stream temperature management differs from conventional approaches in important ways: it considers stream temperature as a holistic regime rather than a discrete temperature threshold. This research has shown that how we define “cold,” and what ecohydrologic feedbacks control the desired thermal regime, will lead us to focus on different stream systems and different management actions for long-term, sustainable cold-water conservation. These findings have important implications for cold-water management, particularly in California.

Cold-water streams’ thermal regimes

The cold-water thermal regimes throughout California are diverse. When threshold-based definitions are applied to streams ranging from the Oregon border to Bakersfield and coastal outlets to Sierra mountains, California appears to possess a multitude of suitable reaches to support cold-water ecosystems. A regime-based assessment illustrates the erroneous equivalence of that conclusion. Historic salmonid habitat in headwater reaches show unique thermal regimes compared to currently accessible, dam-regulated reaches. And while dam regulation has done a reasonably good job at mimicking desirable summer temperatures, the predominant focus on managing annually elevated stream temperatures has overlooked the implications poorly replicated winter temperatures. That California’s dams generally appear incapable of replicating stable cold-water regimes, with the exception of Shasta Dam, is of greater concern. California will have to confront the incompatibility of near-ubiquitous dam regulation of its streams and the desire to restore and sustain cold-water species like salmonids. Difficult decisions about dam removal versus species extinction and the collapse of cold-water ecosystems lay ahead.

Groundwater-fed streams produce stable, resilient cold-water thermal regimes and should be the focus of cold-water conservation. This and other research shows that groundwater-dominated reaches support robust cold-water ecosystems and are resilient to climate warming (Null et al. 2013, Lusardi et al. 2016, Willis et al. 2017). Prior to this research, though, the mechanisms controlling groundwater-dominated thermal regimes were poorly defined.

Process-based cold-water management

The case study of Big Springs Creek illustrates how the thermal regime of a tributary that provides high-quality cold-water habitat may be controlled by novel and previously undocumented drivers of the heat budget. These findings suggest that for streams that illustrate both desirable thermal regimes and support robust cold-water ecosystems, investments should be made to understand the specific dynamics of those systems, develop numerical models of those dynamics, and apply those models to identify effective stream temperature management actions. The Unmanned Aerial Vehicle (UAV) imagery used to quantify spatial and temporal changes to the riverine canopy was critical to developing a numerical model that could be applied to stream temperature management objectives. While numerical models can be costly to develop and update, their application can quantify the benefits of alternative management strategies, guide conservation investments, and provide assurances as to the realized value of restoration actions in a way that empirical studies simply cannot (Hall et al. 2020).

Next steps

Limited freshwater resources are considered the next frontier of resource conflict (Gunasekara et al. 2014). The human population and its water demands are growing (Cosgrove et al. 2000). Agriculture accounts for almost 70 percent of human water use (Cosgrove et al. 2000), and is a critical landscape for conservation efforts because of its dominant proportion of land use (Rosenzweig 2003). At the same time, conservation philosophy is changing. While previous conservation approaches have focused on reserving habitats from human use and restoring biocomplexity, current approaches look to create sustainable biodiversity within human-dominated landscapes (Western 2001). This idea is “reconciliation” ecology (Western 2001, Rosenzweig 2003).

Given the paradigm shift toward reconciliation ecology, future work should explore process-based stream temperature management in the context of working landscapes and more desirable ecosystems. Such research could explore the role of instream flow transfers, water diversion management, or cold-pool management in reservoirs. Most importantly, this research challenges the idea that restoring stream flow will necessarily result in desirable thermal regimes. Stream flow and flow regimes are critical aspects of functional lotic ecosystems (Poff et al. 1997, Yarnell et al. 2015). This research shows that thermal regimes are equally dynamic, though not necessarily directly linked to flow regimes. Thus, while stream flow management is an understandably intuitive strategy to address stream temperature conditions, it may not result in the desired outcomes. Future work should explore alternative restoration actions in the context of a process-based thermal regime strategy, specifically where stream flow does not control thermal regimes. Focusing on cold-water conservation within working landscapes provides the best path forward to sustainable human- and ecosystems across California and the arid western United States.

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