Russian River Biological Opinion Status and Data Report

Year 2016 - 2017

Sonoma Water

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Accessibility Statement

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Appendices

CHAPTER 1: Introduction

On September 24, 2008, the National Marine Fisheries Service (NMFS) issued a 15-year Biological Opinion for water supply, flood control operations, and channel maintenance conducted by the U.S. Army Corps of Engineers (USACE), Sonoma County Water Agency (Water Agency), and Mendocino County Russian River Flood Control and Water Conservation Improvement District in the Russian River watershed (NMFS 2008). The Biological Opinion authorizes incidental take of threatened and endangered Chinook salmon, coho salmon, and steelhead pending implementation of a Reasonable and Prudent Alternative (RPA) to status quo management of reservoir releases, river flow, habitat condition, and facilities in portions of the mainstem Russian River, Dry Creek, and Russian River Estuary. Mandated projects to ameliorate impacts to listed salmonids in the RPA are partitioned among USACE and the Water Agency. Each organization has its own reporting requirements to NMFS. Because coho salmon are also listed as endangered by the California Endangered Species Act (CESA), the Water Agency is party to a Consistency Determination issued by the California Department of Fish and Wildlife (CDFW) in November 2009. The Consistency Determination mandates that the Water Agency implement a subset of Biological Opinion projects that pertain to coho and the Water Agency is required to report progress on these efforts to CDFW.

Project implementation timelines in the Biological Opinion, and Consistency Determination, specify Water Agency reporting requirements to NMFS and CDFW and encourage frequent communication among the agencies. The Water Agency has engaged both NMFS and CDFW in frequent meetings and has presented project status updates on many occasions since early 2009. Although not an explicit requirement of the Biological Opinion or Consistency Determination, the Water Agency has elected to coalesce reporting requirements into one annual volume for presentation to the agencies. The following document represents the eighth report for year 2016-2017. Previous annual reports can be accessed at http://www.sonomawater.gov.

Water Agency projects mandated by the Biological Opinion and Consistency Determination fall into six major categories:

- Biological and Habitat Monitoring;
- Habitat Enhancement;
- California Environmental Quality Act (CEQA) Compliance and Permitting;
- Planning and Adaptive Management;
- Water and Fish Facilities Improvements; and
- Public Outreach.

This report contains status updates for planning efforts, environmental compliance, and outreach but the majority of the technical information we present pertains to monitoring and habitat enhancement. The Biological Opinion requires extensive fisheries data collection in the mainstem Russian River, Dry Creek, and Estuary to detect trends and inform habitat enhancement efforts. The report presents each data collection effort independently and the primary intent of this document is to clearly communicate recent results. However, because Chinook salmon, coho salmon, and steelhead have complex life history patterns that integrate all of these environments, we also present a synthesis section to discuss the interrelated nature of the data. Some monitoring programs are extensions of ongoing Water Agency efforts that were initiated a decade or more before receipt of the Biological Opinion.

References

National Marine Fisheries Service (NMFS). 2008. Biological Opinion for Water Supply, Flood Control Operations, and Channel Maintenance conducted by the U.S. Army Corps of Engineers, the Sonoma County Water Agency, and the Mendocino County Russian River Flood Control and Water Conservation District in the Russian River Watershed. September 24, 2008.

CHAPTER 2: Public Outreach

Biological Opinion Requirements

The Biological Opinion includes minimal *explicit* public outreach requirements. The breadth and depth of the RPAs, however, *implies* that implementation of the Biological Opinion will include a robust public outreach program.

RPA 1 (Pursue Changes to D1610 Flows) mandates two outreach activities. First, it requires the Water Agency, with the support of NMFS staff, to conduct outreach “to affected parties in the Russian River watershed” regarding permanently changing Decision 1610. Second, the RPA requires the Water Agency to update NMFS on the progress of temporary urgency changes to flows during Section 7 progress meetings and as public notices and documents are issued.

RPA 2 (Adaptive Management of the Outlet Channel) requires that within six months of the issuance of the Biological Opinion the Water Agency, in consultation with NMFS, “conduct public outreach and education on the need to reduce estuarine impacts by avoiding mechanical breaching to the greatest extent possible.”

Finally, RPA 3 (Dry Creek Habitat Enhancements, refers to public outreach in the following mandate, “Working with local landowners, DFG[1] and NMFS, Water Agency will prioritize options for implementation” of habitat enhancement.

The remaining RPAs do not mention public outreach.

Water Agency Public Outreach Activities – 2016

Meetings

Public Policy Facilitating Committee (PPFC) meeting - The PPFC met in March 2016 at the Westside Water Education Center. Notices for the meetings were sent out to approximately 800 individuals and agencies and a press release was issued. Approximately 80 people attended the meeting.

In 2016, the meeting included a field trip to the Mirabel Fish Passage Improvement Project. David Manning, Greg Guensch, Kelly Janes (USACE) and Bob Coey (NMFS) gave presentations about the Dry Creek Habitat Enhancement Project. Janes spoke specifically about the USACE’s CAP and General Investigation process, and Coey discussed the Dry Creek Safe Harbor Agreement. Jessica Martini Lamb provided an update on Estuary Management, the Final Jetty Study and the Fish Flow Project timeline. Following the meeting, members of the PPFC traveled to Dry Creek Vineyards for a Safe Harbor Agreement signing ceremony.

Community Meetings, Events & Tours – The eighth Russian River Estuary Lagoon Management Community Meeting was held in April 2016 at the Monte Rio Community Center. The meeting

[1] DFG (Department of Fish and Game) is now known as the California Department of Fish and Wildlife.

included discussions of 2015 Lagoon Management efforts and the 2015 plan (Martini Lamb); results from 2015 water quality monitoring and 2016 plans (Jeff Church); and the final jetty study (Matt Brennan, Environmental Science Associates-PWA). In addition, Natalie Manning, NOAA Fisheries, informed people about the NOAA Habitat Blueprint's Sea-Level Rise Workshops. About 60 people attended the meeting.

A community meeting on Dry Creek habitat enhancement was held in January 2016 at the Lake Sonoma Visitors Center. The meeting was co-hosted by the Dry Creek Valley Association, the Winegrape Growers of Dry Creek, the USACE and the Water Agency. Informational mailers were sent to more than 700 people and about 75 people attended. "Dry Creek Then and Now" was the topic of Neil Lassettre's presentation, while Gregg Horton discussed "Effectiveness Monitoring". In addition, there was an update on the status of Miles 2 & 3 (Cuneo and Guensch); Kelly Janes, USACE, discussed the Corps project; Manning informed people of Miles 4 through 6 conceptual plans; Dan Mason discussed Right-of-Way issues; and the meeting closed with a short update on the Safe Harbor Agreement from Coey.

Two community open houses were held in August 2016 in Cloverdale and Monte Rio on the Fish Habitat Flows and Water Rights Project Draft Environmental Impact Report. Approximately 25 people attended the Cloverdale open house and about 100 attended in Monte Rio. Public hearings on the DEIR were held in Santa Rosa in September and in Cloverdale and Guerneville in November. About 300 people attended the hearings, in total.

Tours held for public officials and others (coordinated with NMFS, DFG, Corps and Water Agency staff) included Eileen Sobeck - NMFS Assistant Administrator, Pat Montanio – Office of Habitat Conservation (OHC) Director, Donna Wieting – Office of Protected Resources Director, Carrie Selberg - OHC Deputy Director, Chris Doley – Restoration Center (RC) Chief, Shannon Dionne - RC Deputy Chief, Kara Meckley - Habitat Protection (HP) Chief, Chris Meaney - HP Deputy Chief, Jennifer Steger – North West RC Regional Supervisor, Peyton Robertson - Chesapeake Bay Office (CBO), Director Sean Corson – CBO- Deputy Director, Jennifer Lukens –Office of Policy Director, and Leslie Craig – Southeast RC Regional Supervisor. A special tour was held for a delegation from Colombia.

Several tours and events were held to celebrate the completion of the Mirabel Fish Passage Improvement Project, including for NMFS, CDFW and the construction contractors; Water Agency staff; VIPs, neighbors and elected officials; and the general public. In total, about 500 people visited the project in September and October.

Other Outreach

Free Media – Multiple articles about Biological Opinion projects (primarily the Fish Flow DEIR and the Mirabel Fish Passage Improvement Project) appeared in 2016 in The Press Democrat, the Russian River Times, the West County News and Review, and North Bay Bohemian, and the Russian River Gazette. In 2016, press releases were issued on Mirabel fishway construction, Dry Creek habitat construction, community meetings regarding the estuary and Dry Creek, Chinook returns, coho releases and the Public Policy Facilitating Committee meeting.

Electronic Media – The Water Agency continually updated its Biological Opinion webpage, including links on new documents and meetings. In addition, the Water Agency posted videos on YouTube regarding Dry Creek habitat construction, the Fish Flow DEIR and the Mirabel Fish Passage Improvement Project, which can be accessed via the agency's website. Email alerts regarding activities in the estuary were issued about a dozen times in 2016.

Materials – In 2016, flyers regarding the Dry Creek Demonstration Project and the Mirabel Fish Ladder projects were updated several times to reflect different stages of construction and completion. An FAQ and other explanatory materials were created for the Fish Flow DEIR. Other materials were updated and distributed at meetings, conferences, statewide forums, outreach events and through the Water Agency website.

Nearly 800 copies of the Dry Creek Habitat Enhancement Bulletin were mailed to residents throughout the Dry Creek Valley and distributed at meetings and during tours. The six-page newsletter covered topics including: Plans for construction of habitat features in the summer of 2016; updated timeline for completion of six miles of habitat enhancement projects by 2020; monitoring of habitat features; profiles of participating landowners and one of the project design consultants; and an article about the first Safe Harbor Agreement entered into with a property owner.

CHAPTER 3: Pursue Changes to Decision 1610 Flows

Two major reservoir projects provide water supply storage in the Russian River watershed: 1) Coyote Valley Dam/Lake Mendocino, located on the East Fork of the Russian River three miles east of Ukiah, and 2) Warm Springs Dam/Lake Sonoma, located on Dry Creek 14 miles northwest of Healdsburg. The Water Agency is the local sponsor for these two federal water supply and flood control projects, collectively referred to as the Russian River Project. Under agreements with the USACE, the Water Agency manages the water supply storage space in these reservoirs to provide a water supply and maintain summertime Russian River and Dry Creek streamflows.

The Water Agency holds water-right permits[1] issued by the State Water Resources Control Board (SWRCB) that authorize the Water Agency to divert[2] Russian River and Dry Creek flows and to re-divert[3] water stored and released from Lake Mendocino and Lake Sonoma. The Water Agency releases water from storage in these lakes for delivery to municipalities, where the water is used primarily for residential, governmental, commercial, and industrial purposes. The primary points of diversion include the Water Agency's facilities at Wohler and Mirabel Park (near Forestville). The Water Agency also releases water to satisfy the needs of other water users and to contribute to the maintenance of minimum instream flow requirements in the Russian River and Dry Creek established in 1986 by the SWRCB's Decision 1610. These minimum instream flow requirements vary depending on specific hydrologic conditions (normal, dry, and critical) that are based on cumulative inflows into Lake Pillsbury in the Eel River watershed.

NMFS concluded in the Russian River Biological Opinion that the artificially elevated summertime minimum flows in the Russian River and Dry Creek currently required by Decision 1610 result in high water velocities that reduce the quality and quantity of rearing habitat for coho salmon and steelhead. NMFS' Russian River Biological Opinion concludes that reducing Decision 1610 minimum instream flow requirements will enable alternative flow management scenarios that will increase available rearing habitat in Dry Creek and the upper Russian River, and provide a lower, closer-to-natural inflow to the estuary between late spring and early fall, thereby enhancing the potential for maintaining a seasonal freshwater lagoon that would likely support increased production of juvenile steelhead and salmon.

Changes to Decision 1610 are under the purview of the SWRCB, which retained under Decision 1610 the jurisdiction to modify minimum instream flow requirements if future fisheries studies identified a benefit. NMFS recognized that changing Decision 1610 would require a multi-year (6

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[1] SWRCB water-right permits 12947A, 12949, 12950 and 16596.
[2] Divert – refers to water diverted directly from streamflows into distribution systems for beneficial uses or into storage in reservoirs.
[3] Re-divert – refers to water that has been diverted to storage in a reservoir, then is released and diverted again at a point downstream.

to 8 years) process of petitioning the SWRCB for changes to minimum instream flow requirements, public notice of the petition, compliance with CEQA, and a SWRCB hearing process. To minimize the effects of existing minimum instream flows on listed salmonids during this process, the Russian River Biological Opinion stipulated that the Water Agency “will seek both long term and interim changes to minimum flow requirements stipulated by D1610.” The permanent and temporary changes to Decision 1610 minimum instream flow requirements specified by NMFS in the Russian River Biological Opinion are summarized in Figure 3.1.

Permanent Changes

The Russian River Biological Opinion requires the Water Agency to begin the process of changing minimum instream flows by submitting a petition to change Decision 1610 to the SWRCB within one year of the date of issuance of the final Biological Opinion. The Water Agency filed a petition with the SWRCB on September 23, 2009, to permanently change Decision 1610 minimum instream flow requirements. The requested changes are to reduce minimum instream flow requirements in the mainstem Russian River and Dry Creek between late spring and early fall during normal and dry water years and promote the goals of enhancing salmonid rearing habitat in the upper Russian River mainstem, lower river in the vicinity of the Estuary, and Dry Creek downstream of Warm Springs Dam. NMFS’ Russian River Biological Opinion concluded that, in addition to providing fishery benefits, the lower instream flow requirements “should promote water conservation and limit effects on in-stream river recreation.” NMFS’ recommended changes, based on observations during the 2001 interagency flow-habitat study and the 2007 low flow season, to achieve these goals are provided in the Russian River Biological Opinion (NMFS 2008) and are summarized in Figure 3.1.

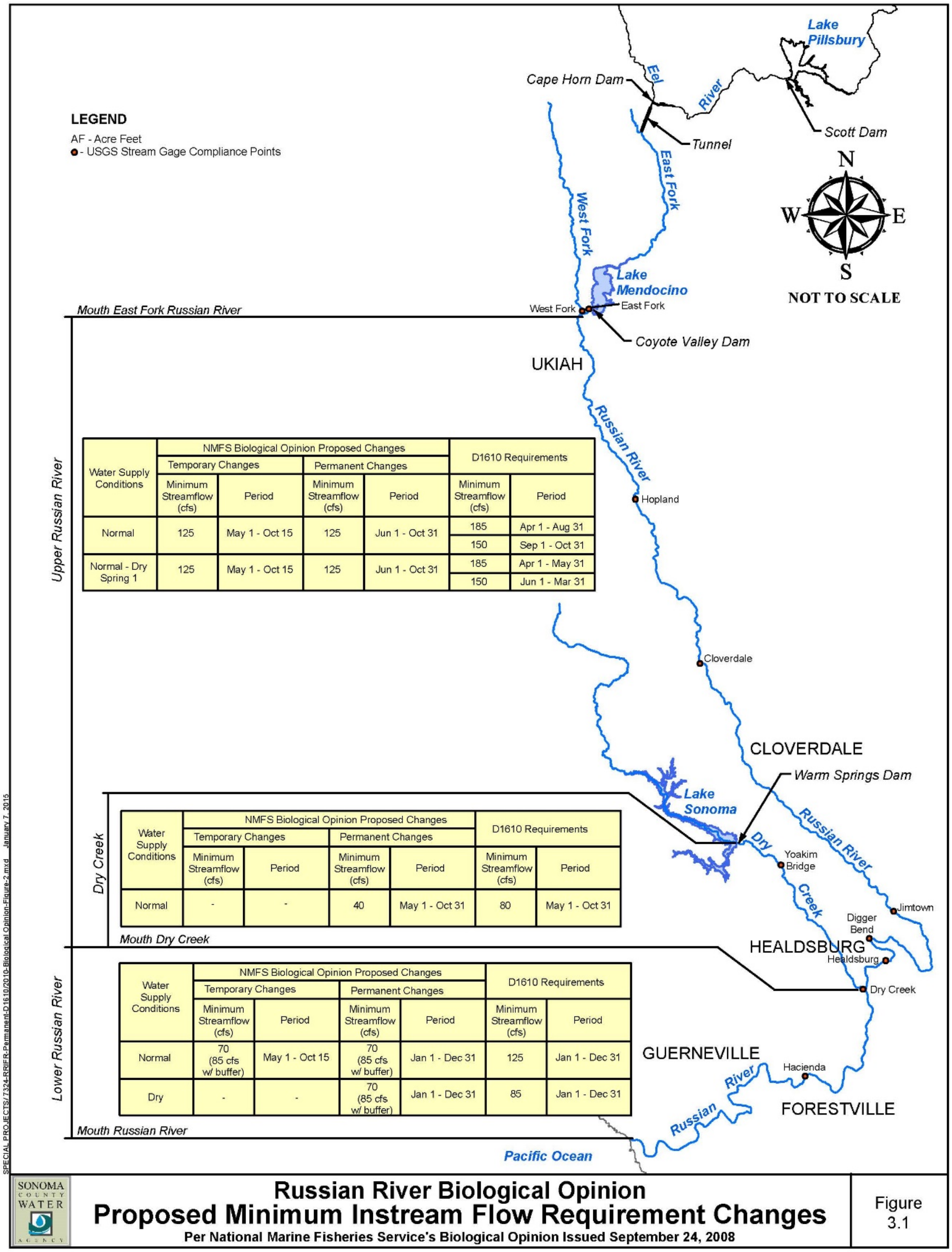
Summary Status

The SWRCB issued a second amended public notice of the Water Agency’s petition to modify Decision 1610 for public comment on March 29, 2010. Following filing of the petition to change Decision 1610, the Water Agency issued a Notice of Preparation (NOP) of an Environmental Impact Report (EIR) for the Fish Habitat Flows and Water Rights Project (Fish Flow Project).

A Draft Environmental Impact Report (EIR) was released for public review on August 19, 2016. The public comment period closed on March 10, 2017, after extending the comment period to allow additional time to review an errata released on January 26, 2017. Public hearings were held on September 13, 2016 (in Santa Rosa), November 16, 2016 (in Cloverdale), and November 17, 2016 (in Guerneville).

Temporary Changes

Until the SWRCB issues an order on the petition to permanently modify Decision 1610, the minimum instream flow requirements specified in Decision 1610 (with the resulting adverse impacts to listed salmonids) will remain in effect, unless temporary changes to these requirements are made by the SWRCB. The Russian River Biological Opinion requires that the Water Agency petition the SWRCB for temporary changes to the Decision 1610 minimum instream flow requirements beginning in 2010 and for each year until the SWRCB issues an



Upper Russian River

Water Supply Conditions	NMFS Biological Opinion Proposed Changes: Temporary Changes – Minimum Streamflow (cfs)	Temporary Changes – Period	Permanent Changes – Minimum Streamflow (cfs)	Permanent Changes – Period	D1610 Requirements: Minimum Streamflow (cfs)	D1610 Requirements: Period
Normal	125	May 1 - Oct 15	125	Jun 1 - Oct 31	185	Apr 1 - Aug 31
					150	Sep 1 - Oct 31
Normal - Dry Spring 1	125	May 1 - Oct 15	125	Jun 1 - Oct 31	185	Apr 1 - May 31
					150	Jun 1 - Mar 31

Dry Creek

Water Supply Conditions	NMFS Biological Opinion Proposed Changes: Temporary Changes – Minimum Streamflow (cfs)	Temporary Changes – Period	Permanent Changes – Minimum Streamflow (cfs)	Permanent Changes – Period	D1610 Requirements: Minimum Streamflow (cfs)	D1610 Requirements: Period
Normal	-	-	40	May 1 - Oct 31	80	May 1 - Oct 31

Lower Russian River

Water Supply Conditions	NMFS Biological Opinion Proposed Changes: Temporary Changes – Minimum Streamflow (cfs)	Temporary Changes – Period	Permanent Changes – Minimum Streamflow (cfs)	Permanent Changes – Period	D1610 Requirements: Minimum Streamflow (cfs)	D1610 Requirements: Period
Normal	70 (85 cfs w/ buffer)	May 1 - Oct 15	70 (85 cfs w/ buffer)	Jan 1 - Dec 31	125	Jan 1 - Dec 31
Dry	-	-	70 (85 cfs w/ buffer)	Jan 1 - Dec 31	85	Jan 1 - Dec 31

Figure 3.1. A summary of the permanent and temporary changes to Decision 1610 minimum instream flow requirements specified by NMFS in the Russian River Biological Opinion.

order on the Water Agency's petition for the permanent changes to these requirements. NMFS' Russian River Biological Opinion only requires that petitions for temporary changes "request that minimum bypass flows of 70 cfs be implemented at the USGS gage at the Hacienda Bridge between May 1 and October 15, with the understanding that for compliance purposes SCWA will typically maintain about 85 cfs at the Hacienda gage. For purposes of enhancing steelhead rearing habitats between the East Branch and Hopland, these petitions will request a minimum bypass flow of 125 cfs at the Healdsburg gage between May 1 and October 15."

Summary Status

The Water Agency submitted a Temporary Urgency Change Petition to the SWRCB on April 15, 2016, to comply with the requirements of the Russian River Biological Opinion (Appendix 3.1). The SWRCB issued an Order approving the Water Agency's TUCP on May 4, 2016 (Appendix 3.2).

The SWRCB's Order made the following changes to the Water Agency's permits until October 27, 2016: minimum instream flow in the upper Russian River (from its confluence with the East Fork of the Russian River to its confluence with Dry Creek) remained at or above 125 cfs; and minimum instream flow in the lower Russian River (from its confluence with Dry Creek to the Pacific Ocean) remained at or above 70 cfs. To allow the Water Agency to optimally manage flows in the Upper Russian River and Lower Russian River, the Order allowed for minimum instream flow requirements to be measured based on a 5-day running average of average daily stream flow measurements, provided that instantaneous flows in the upper Russian River would be no less than 110 cfs and in the lower Russian River no less than 60 cfs.

The Order included several terms and conditions, including requirements for fisheries habitat monitoring and regular consultation with National Marine Fisheries Service and California Department of Fish and Wildlife regarding fisheries conditions, preparation of a water quality monitoring plan and summary data report, reporting on hydrologic conditions of the Russian River system), and reporting of activities and programs implemented by the Water Agency and its contractors to assess and reduce water loss and promote increasing water use efficiency.

Reports to fulfill the terms of the Order were prepared and submitted to the SWRCB and the water quality and fisheries report are provided in Appendix 3.3. Water quality monitoring results were posted to the Water Agency website and are provided in Appendix 3.3. Water quality monitoring in the Russian River Estuary is further discussed in Chapter 4.

References

National Marine Fisheries Service (NMFS). 2008. Biological Opinion for Water Supply, Flood Control Operations, and Channel Maintenance conducted by the U.S. Army Corps of Engineers, the Sonoma County Water Agency, and the Mendocino County Russian River Flood Control and Water Conservation District in the Russian River Watershed. September 24, 2008.

CHAPTER 4: Estuary Management

The Russian River estuary (Estuary) is located approximately 97 kilometers (km; 60 miles) northwest of San Francisco in Jenner, Sonoma County, California. The Estuary extends from the mouth of the Russian River upstream approximately 10 to 11 km (6 to 7 miles) between Austin Creek and the community of Duncans Mills (Heckel 1994). When a barrier beach forms and closes the river mouth, a lagoon forms behind the beach and reaches up to Vacation Beach.

The Estuary may close throughout the year as a result of a barrier beach forming across the mouth of the Russian River. The mouth is located at Goat Rock State Beach (California Department of Parks and Recreation). Although closures may occur at any time of the year, the mouth usually closes during the spring, summer, and fall (Heckel 1994; Merritt Smith Consulting 1997, 1998, 1999, 2000; Sonoma County Water Agency and Merritt Smith Consulting 2001). Closures result in ponding of the Russian River behind the barrier beach and, as water surface levels rise in the Estuary, flooding may occur. The barrier beach has been artificially breached for decades; first by local citizens, then the County of Sonoma Public Works Department, and, since 1995, by the Water Agency. The Water Agency's artificial breaching activities are conducted in accordance with the Russian River Estuary Management Plan recommended in the Heckel (1994) study. The purpose of artificially breaching the barrier beach is to alleviate potential flooding of low-lying properties along the Estuary.

The National Marine Fisheries Service's (NMFS) Russian River Biological Opinion (NMFS 2008) found that artificially elevated inflows to the Russian River estuary during the low flow season (May through October) and historic artificial breaching practices have significant adverse effects on the Russian River's estuarine rearing habitat for steelhead, coho salmon, and Chinook salmon. The historical method of artificial sandbar breaching, which is done in response to rising water levels behind the barrier beach, adversely affects the Estuary's water quality and freshwater depths. The historical artificial breaching practices create a tidal marine environment with shallow depths and high salinity. Salinity stratification contributes to low dissolved oxygen at the bottom in some areas. The Biological Opinion (NMFS 2008) concludes that the combination of high inflows and breaching practices impact rearing habitat because they interfere with natural processes that cause a freshwater lagoon to form behind the barrier beach. Fresh or brackish water lagoons at the mouths of many streams in central and southern California often provide depths and water quality that are highly favorable to the survival of rearing salmon and steelhead.

The Biological Opinion's RPA 2, Alterations to Estuary Management, (NMFS 2008) requires the Water Agency to collaborate with NMFS and to modify Estuary water level management in order to reduce marine influence (high salinity and tidal inflow) and promote a higher water surface elevation in the Estuary (formation of a fresh or brackish lagoon) for purposes of enhancing the quality of rearing habitat for young-of-year and age 1+ juvenile (age 0+ and 1+) steelhead from May 15 to October 15 (referred to hereafter as the "lagoon management period"). A program of potential, incremental steps are prescribed to accomplish this, including adaptive management of a lagoon outlet channel on the barrier beach, study of the existing jetty and its potential influence on beach formation processes and salinity seepage through the barrier beach, and a feasibility study of alternative flood risk measures. RPA 2 also includes provisions for monitoring the response of water quality, invertebrate production, and salmonids in the Estuary to the management of water surface elevations during the lagoon management period.

The following section provides a summary of the Water Agency's estuary management actions required under the Russian River Biological Opinion RPA 2. These actions are also required by other regulatory permits issued for the Estuary Management Project, including the California Coastal Commission's Coastal Development Permit (CDP) and North Coast Regional Water Quality Control Board Clean Water Act Section 401 Water Quality Certification (Certification). References to the Biological Opinion's RPA are used to maintain consistency with previous annual reports.

Barrier Beach Management

Adaptive Management Plan

RPA 2 requires the Water Agency, in coordination with NMFS, California Department of Fish and Wildlife (CDFW), and the U.S. Army Corps of Engineers (USACE), to annually prepare barrier beach outlet channel design plans. The Water Agency contracted with Environmental Science Associates (ESA PWA) to prepare the Russian River Estuary Outlet Channel Adaptive Management Plan (Appendix 4.1). The approach of the plan was to meet the objective of RPA 2 to the greatest extent feasible while staying within the constraints of existing regulatory permits and minimizing the impact to aesthetic, biological, and recreational resources of the site. The annual meeting with regulatory agency staff to discuss the prior year's beach management activities and preparation of the updated 2016 annual Outlet Channel Adaptive Management Plan Estuary management for 2016 was discussed at a meeting on March 14, 2016, that included representatives from NMFS and CDFW, as well as the Water Agency, University of California, Davis's Bodega Marine Laboratory (Bodega Marine Lab), the USACE, the North Coast Regional Water Quality Control Board (NCRWQCB), and ESA PWA. Only minor updates to the prior year's plan were made in the 2016 plan, which includes a summary of physical processes from 2011 to 2015 as appendices to the plan. Prior to 2016, outlet channel implementation had occurred only in 2010 (summarized in Appendix F of the 2016 Outlet Channel Adaptive Management Plan; Appendix 4.1). An outlet channel was attempted twice in 2016, on June 7 and June 27. In both instances, water flowing through the outlet channel scoured the channel and, within a day, caused self-breaching of the barrier beach as described in the following sections.

Beach Topographic Surveys

A monthly topographic survey of the beach at the mouth of the Russian River is also required under RPA 2. Topographic data was collected monthly in 2016 and provided to NMFS and CDFW. The April 2016 topographic survey was cancelled due to the presence of neonate (less than 1 week old) harbor seals at the mouth of the Russian River. The beach topographic maps are provided in Appendix 4.2. The topographic maps provide documentation of changing beach widths and crest heights, which influence both flood risk and the need to respond to river mouth closures through beach management activities.

2016 Beach and River Mouth Conditions

Several inlet closure events occurred early in the management period: June 1 – 7, June 15 – 27, and July 1 – 12 (Figures 4.1 – 4.3). Two additional inlet closure events occurred later in the management period: September 11 - 30 and October 12 – 20 (this event ended after the conclusion of the lagoon management period).

A barrier beach was formed eleven times during 2016, during five of these closure events the Water Agency conducted water level management activities at the barrier beach (Table 4.1). The Russian River mouth was closed to the ocean for a total of 68 days (or 19%) in 2016, mostly during the fall months. As described in Appendix L of the 2017 Outlet Channel Adaptive Management Plan, during the 2016 management period, May 15 to October 15, Water Agency staff regularly monitored current and forecasted Estuary water levels, inlet state, river discharge, tides, and wave conditions to anticipate changes to the inlet's state (Appendix 4.3; ESA PWA 2016).

Lagoon Management Season Closures, Outlet Channel Implementation, and Self-Breaches

Time series of Estuary water levels, as well as the key forcing factors (waves, tides, and riverine discharge), are shown in Figure 4.1 for the entire 2016 management period. The lagoon water level time series (Figure 4.1a) summarizes the closure events at the beginning of the management period, as well as the subsequent tidal conditions and later closure events in fall . During the 2016 management period, Russian River flows were higher than the previous drought years, 2013-2015. Flows at Guerneville did not drop to 100 cfs until the end of July, which was more than a month later than in 2015, and two months later than in 2014 (Figure 4.1d). In August, flows increased to just above 100 cfs and remained above that for the rest of the management period.

As in prior years, wave heights declined through July and August (Figure 4.1b). However, in prior years closure events typically coincided with either moderately high waves (greater than 6 feet) having periods greater than 10 seconds, or with neap oceanic tide ranges of less than approximately 5 feet. Although all five closure events in the 2016 management period occurred during neap tidal conditions, wave heights were generally less than 5 feet. In all cases the waves were long-period swells, with periods of 12-17 seconds. Waves with longer periods are more effective at transporting sand on to shore and into the inlet.

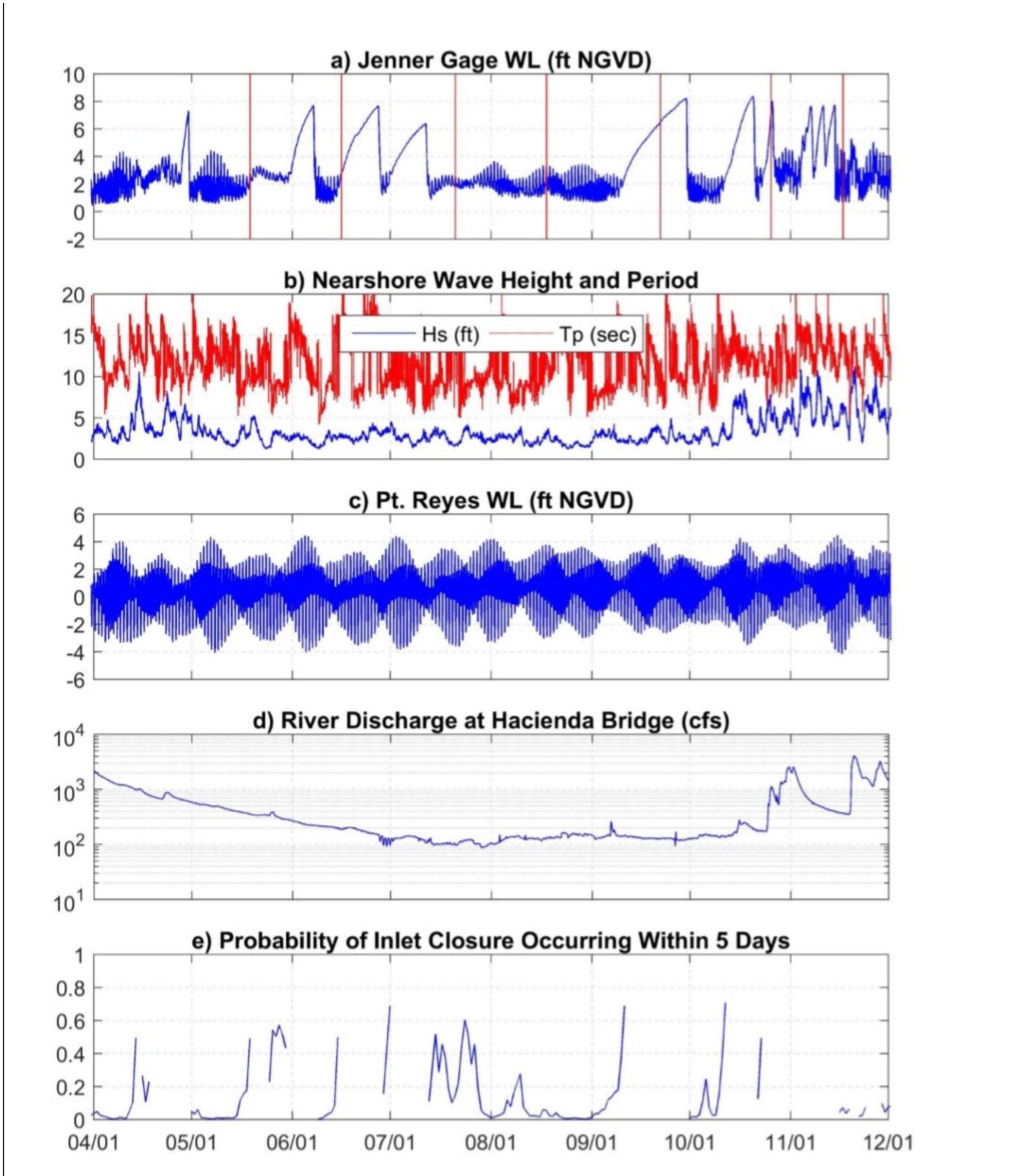


Figure 4.1. Estuary, Ocean, and River Conditions Compared with Closure Probability: April – November 2016.

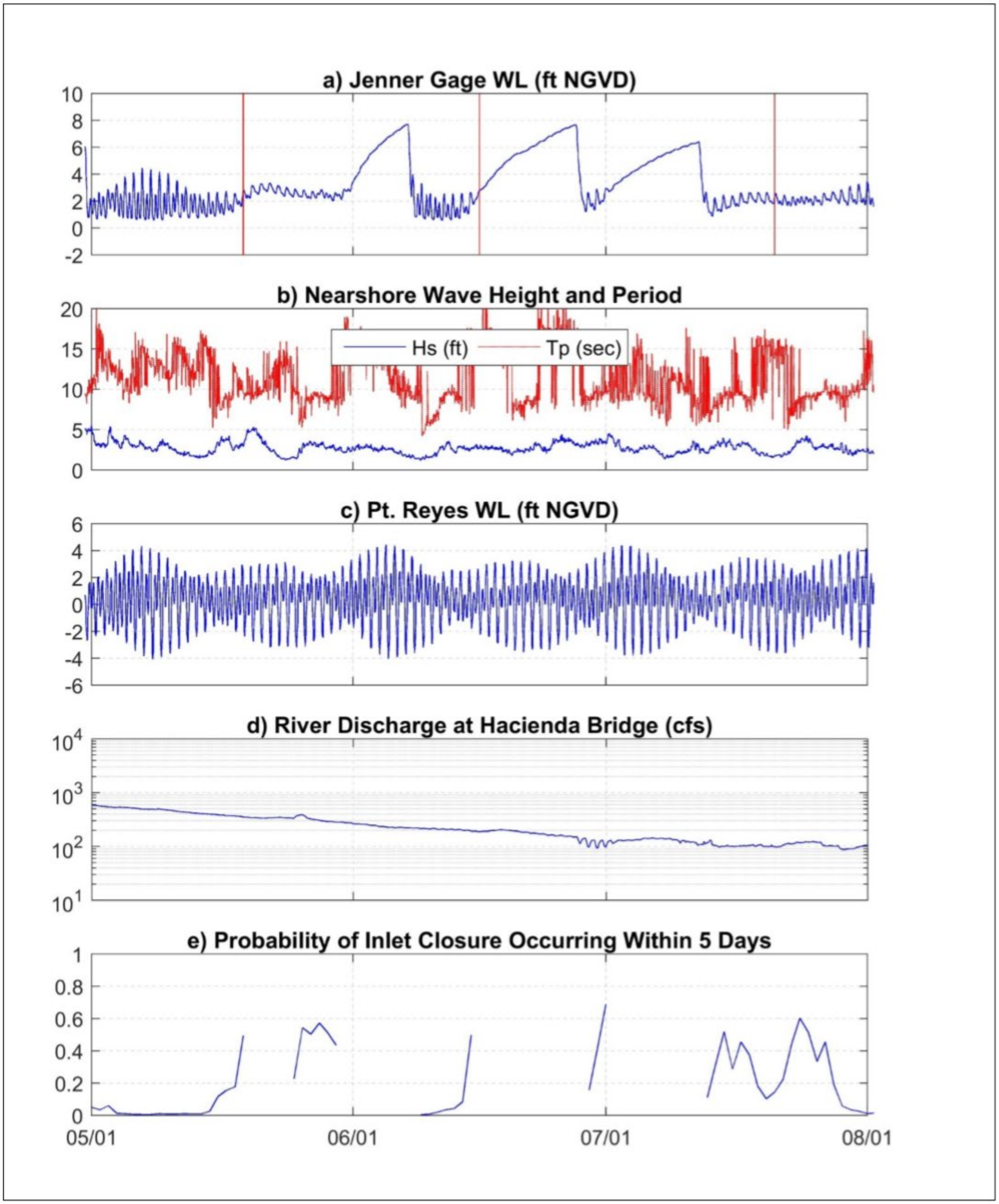


Figure 4.2. Estuary, Ocean, and River Conditions Compared with Closure Probability: May – July 2016.

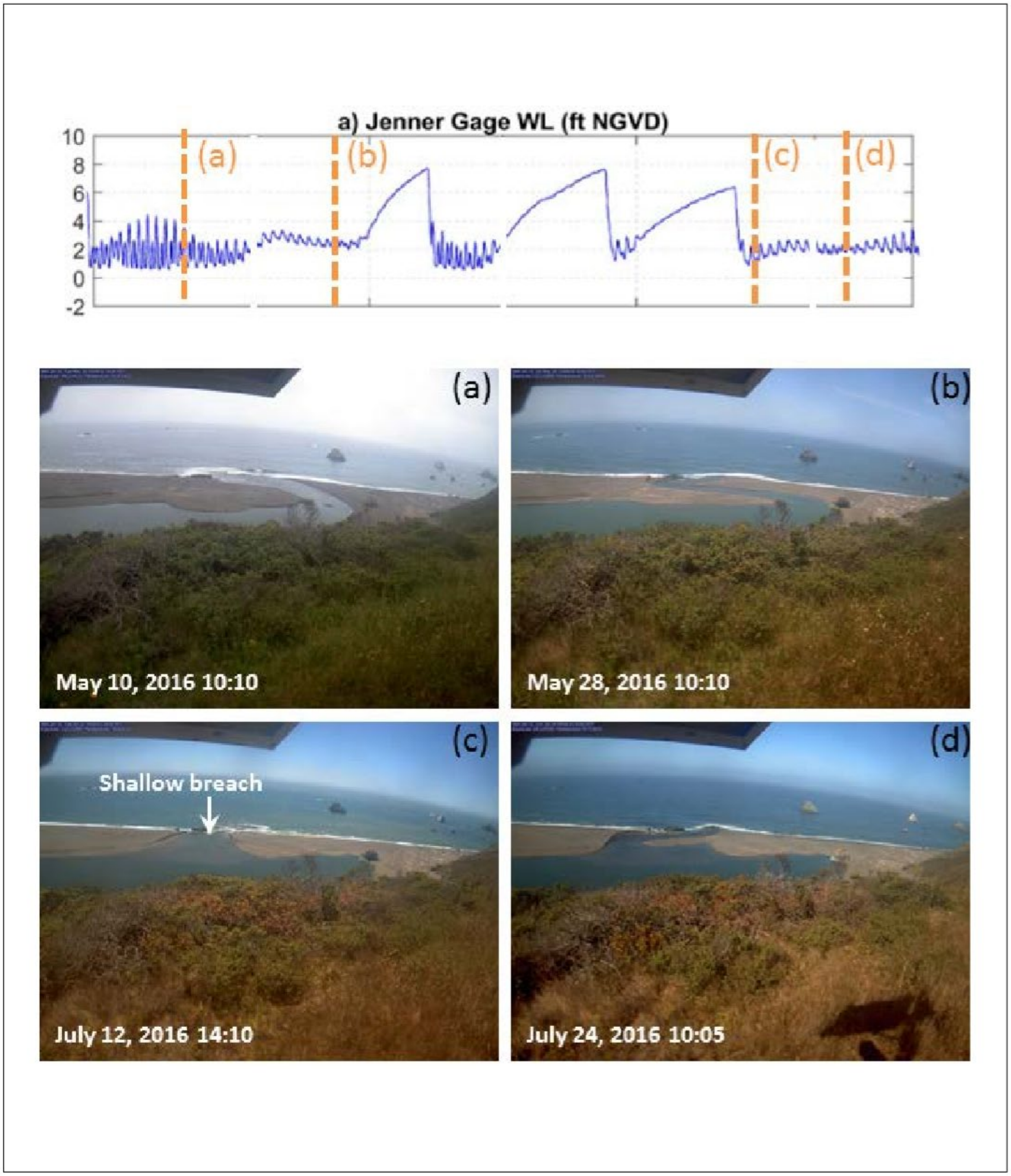


Figure 4.3. Russian River camera photographs showing some of the muted tidal conditions observed in 2016.

Table 4.1. Summary of beach management activities at Goat Rock State Beach for the Russian River Estuary Management Project, 2016. Location of activities are shown on Figure 4.1.

Closure Date	Beach Management Date	No. Days Closed	Activity Time[1]	Water Elevation (ft)[2]	Beach Management Activity[3]	Excavated Volume (CY)[4]
1-Jun	7-Jun	7	8:40am-10:55am	7.75	Lagoon Outlet Channel	660
15-Jun	27-Jun	12	7:55am-9:24am	7.75	Lagoon Outlet Channel	148
1-Jul	12-Jul	11	None	6.45	None. Self-breach	0
11-Sep	30-Sep	20	None	8.22	None. Self-breach	0
12-Oct	20-Oct	8	9:21am-10:50am	8.34	Pilot Channel	496
24-Oct	26-Oct	3	None	8.05	None. Self-breach	0
5-Nov	7-Nov	2	None	7.67	None. Self-breach	0
8-Nov	10-Nov	2	10:30am-2:28pm	7.71	Pilot Channel	1813
12-Nov	14-Nov	2	9:45am-12:05pm	7.75	Pilot Channel	725

[1] Estimated period that excavator/bulldozer equipment was on the beach.
[2] Water surface elevation recorded at the Jenner gage located at the Jenner Visitor's Center.
[3] Beach management activity consists of a pilot channel to initiate an artificial breach of the barrier beach or outlet channel to form a lagoon.
[4] Estimated volume of sand excavated with heavy equipment during artificial breach or lagoon management activity.

When the mouth closed on June 1, flows at the Guerneville gage were measured at 260 cfs, and these had tapered to 222 cfs by June 7, when the outlet channel was excavated. The outlet channel was excavated approximately 580 feet northwest of the jetty (Figure 4.4), angled to the northwest, with a bottom width of approximately 25 feet, a channel length of approximately 230 feet, and a channel bottom elevation of 7 feet National Geodetic Vertical Datum (NGVD). The estuary water surface elevation at the time of completion was 7.75 feet at the Jenner Visitor's Center, and the ocean tide level was approximately 1.9 feet and rising. The excavation was planned during rising tides in anticipation of rising tides conveying sand into the channel and

Figure 4.4. General location of outlet channel excavations for artificial breaching in 2016.

thereby reducing the potential for self-breaching. Less than a day after the outlet channel excavation, the channel scoured open (Figure 4.5) and estuary water surface elevations declined (Figure 4.2a).

River flows continued to decline into June, and although waves were generally moderate, neap tide conditions in mid-June preceded another closure event on June 15. Flows at the time of closure were measured at 193 cfs, and had declined to 151 cfs by June 27. Outlet channel excavation was implemented early on the morning of June 27. The outlet channel was excavated approximately 80 feet north of the jetty (Figure 4.4), angled to the northwest and parallel to the jetty, with a bottom width of approximately 20 feet, a channel length of approximately 150 feet, and a channel bottom elevation of 7 feet (as measured by Water Agency surveyor staff). The estuary water surface elevation at the time of completion was 7.7 feet, and the ocean tide level was approximately 2 feet and declining. By the afternoon, the outlet channel was scoured open and self-breached (Figure 4.5), such that estuary water surface elevations had declined quickly and the estuary became tidal (Figure 4.2a).

The mouth closure lasting from July 1 to July 12 happened at lower flows (110-140 cfs), but self-breached when water surface elevations were just over 6 feet, before an outlet channel could be implemented.

As with most years from 2010 to 2015, the mouth remained open for the remainder of July and August. The next closure event occurred on September 11, during a period of neap tides and relatively low-height (less than 4 ft), long-period (greater than 15 seconds) swell wave conditions. Flows were 95-140 cfs at the Guerneville gage during this closure event. A steep drop in topography adjacent to the jetty made the beach north of the jetty inaccessible to excavation equipment, and the mouth self-breached at a water surface elevation of 8.3 ft on September 30th. The last closure event during the 2016 management period occurred on October 12. The Guerneville discharge was 145 cfs at closure. Increasing flows and strong wave overwash contributed to rapid rise in estuary water surface elevation, and the mouth was artificially breached on October 20, after the end of the management period, with estuary water surface elevation at 8.3 ft.

Apart from having two outlet channel implementations, the 2016 management period was also notable for having several periods of muted tidal conditions (tide range less than 1 ft). These occurred for roughly ten days prior to the June 1 closure event, and for approximately two weeks after the July 13 self-breach. Wave conditions were generally moderate at the beginning of both conditions, with heights below 5 ft and periods below 14 seconds, although both began during neap tidal conditions. Figure 4.3 illustrates the channel shape during both periods.

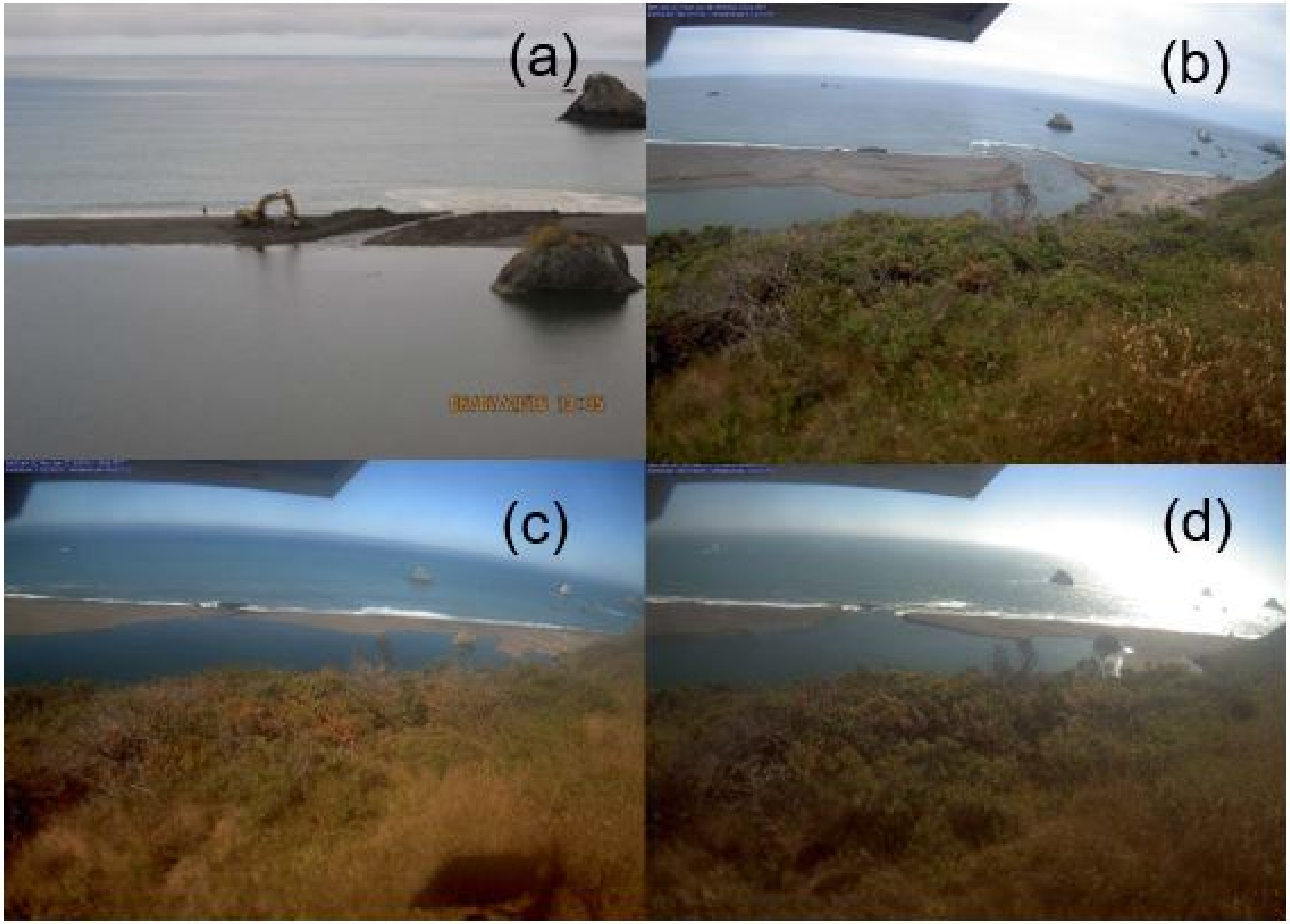


Figure 4.5. (a) Outlet channel after excavation on June 7 and (b) scoured inlet on June 8, 2016. (c) Outlet channel after excavation on June 27 and (d) scoured inlet on June 27.

Appendix L of the 2017 Russian River Estuary Outlet Channel Adaptive Management Plan offers lessons learned based on 2016 observations of the Estuary, associated physical processes, and the Water Agency's planning for outlet channel management. These are summarized here and may be found in Appendix 4.3 of this report for fuller context:

- During the 2016 management period, the beach 200 feet north of Haystack Rock remained stable between 17 and 19 ft NGVD. This is significantly higher than in 2015, when the inlet was observed to migrate farther north, and the beach crest ranged from 11 to 15 ft NGVD. This reinforces the idea that lack of migration can allow the beach to reach higher and more stable crest elevations.
- Peak annual river discharge has remained below 43,000 cfs for 10 consecutive winters (October 2007 to April 2016) preceding the management period, a streak unmatched in the 70-year flow record. This lack of larger fluvial discharge may contribute to the predominant inlet location near the groin.
- The beach width in 2016 at Transect 3 (near Haystack Rock) was similar to 2014 and wider than in 2015. This may suggest that beach width is closely tied to inlet migration – the lack of migration north of Haystack Rock for several years prior to 2015 had previously allowed the beach to grow at this end of the littoral cell.

Artificial Breaching

Outside of the management season, there were four mouth closures in 2016. The Water Agency artificially breached the barrier beach at the Russian River mouth outside the lagoon management period twice in 2016 (Table 4.1; Figure 4.6). Time series photographs of each event are shown in Figures 4.7 – 4.11. The breachings were necessary to minimize flood risk to low-lying structures, which occurs at or above an elevation of approximately 9 feet NGVD at the Jenner gage located at State Parks' Jenner visitor center. No artificial breaching activities occurred during the lagoon management period (May 15 – October 15).

Figure 4.6. Locations of beach management activities in 2016 at the Russian River mouth, Goat Rock State Beach. Lines crossing the barrier beach are pilot channels for artificial breaching (red) and outlet channels to form a lagoon (blue). Self-breach events are not shown.

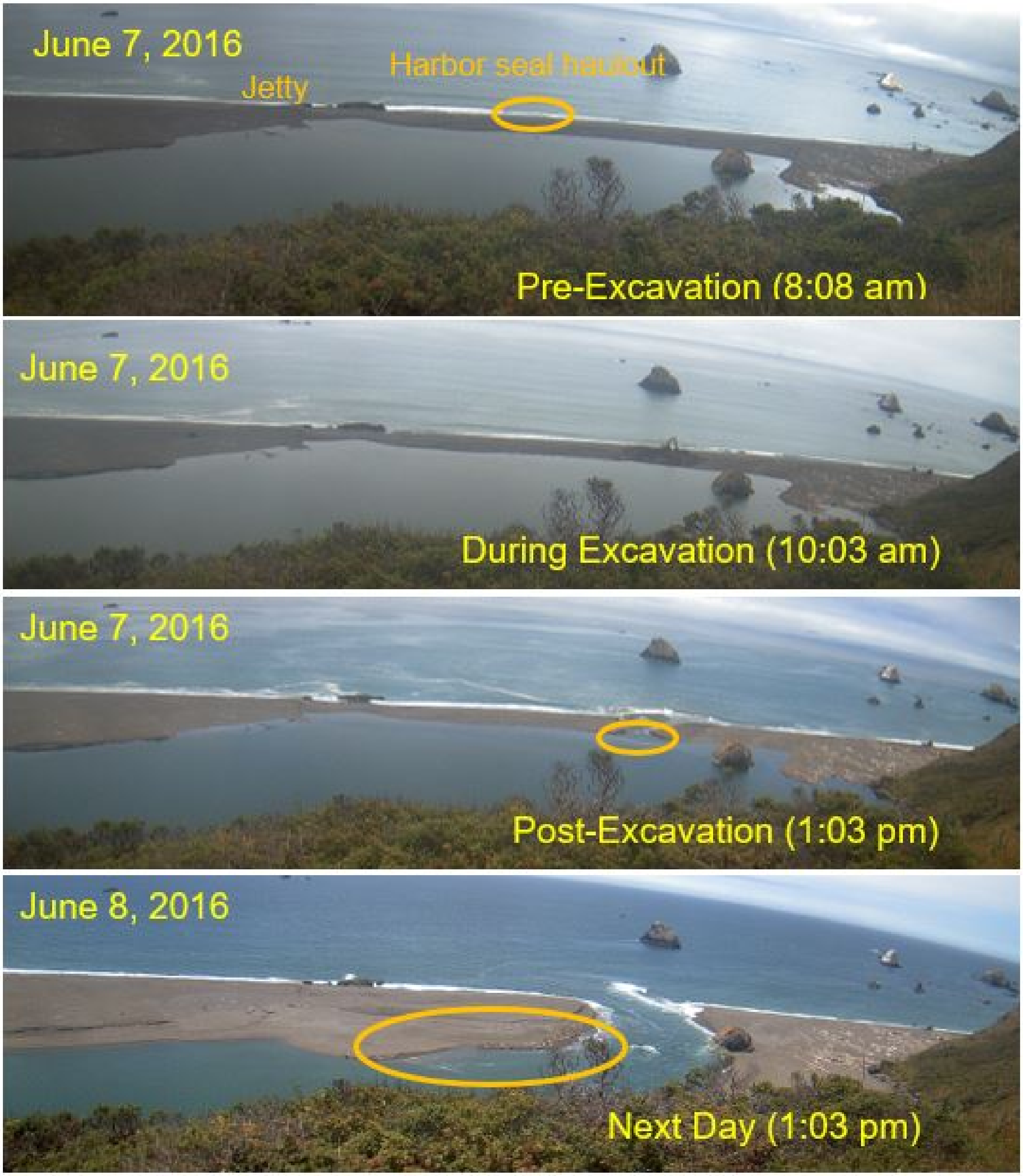


Figure 4.7. Lagoon outlet channel at the mouth of the Russian River Estuary, June 7, 2016. The outlet channel was excavated at the north end of the barrier beach. Photographs show pre-management activity through next day conditions. The outlet channel eroded causing a self-breach within a few hours.

Figure 4.8. Lagoon outlet channel at the mouth of the Russian River Estuary, June 27, 2016. Photographs show pre- management activity through next day conditions. The channel was excavated near the jetty. Morning fog obscured pre-excavation photos. The outlet channel eroded causing a self-breach within a few hours.

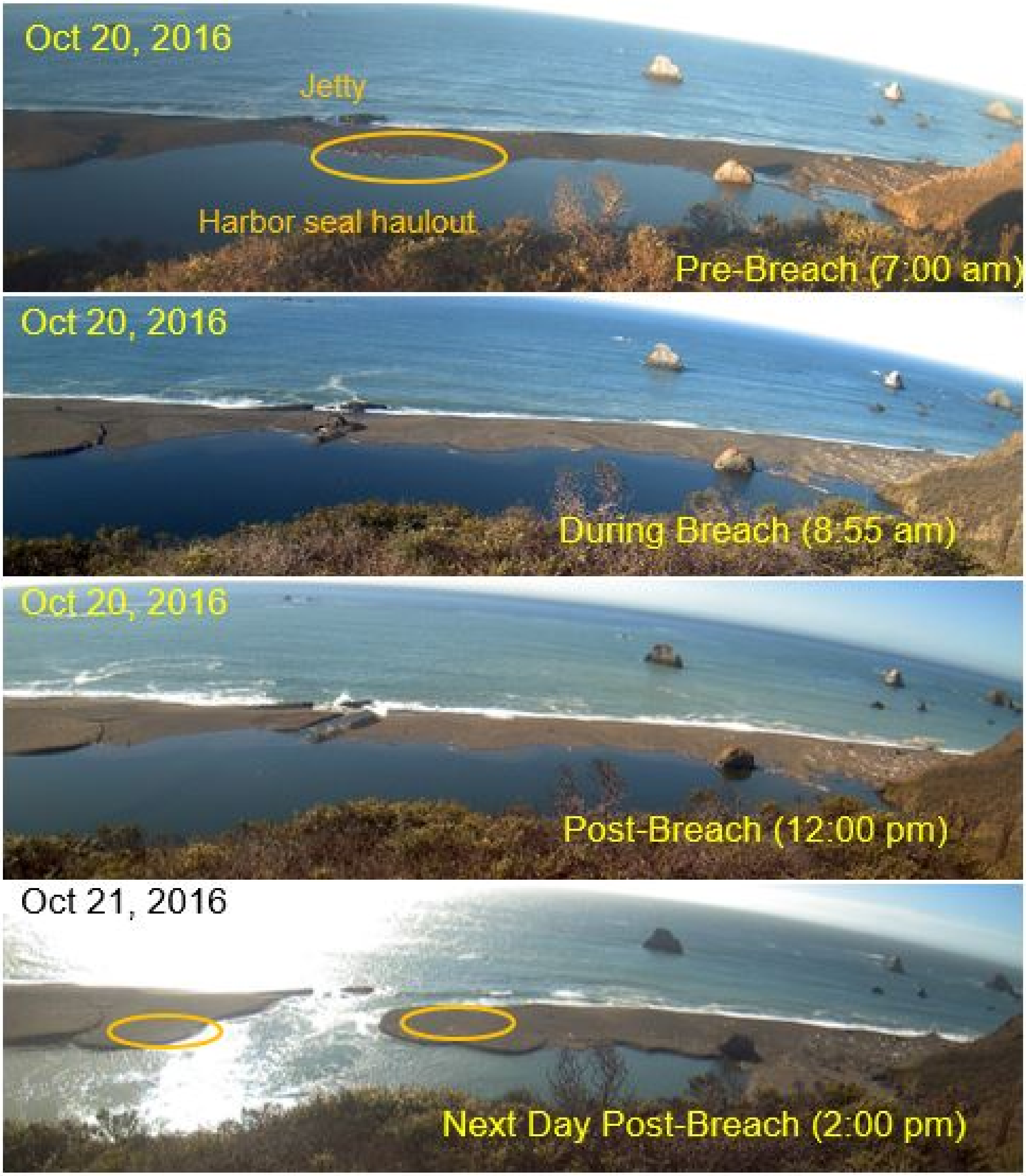


Figure 4.9. Artificial breaching at the mouth of the Russian River Estuary, October 20, 2016. Photographs show pre-management through next day conditions.

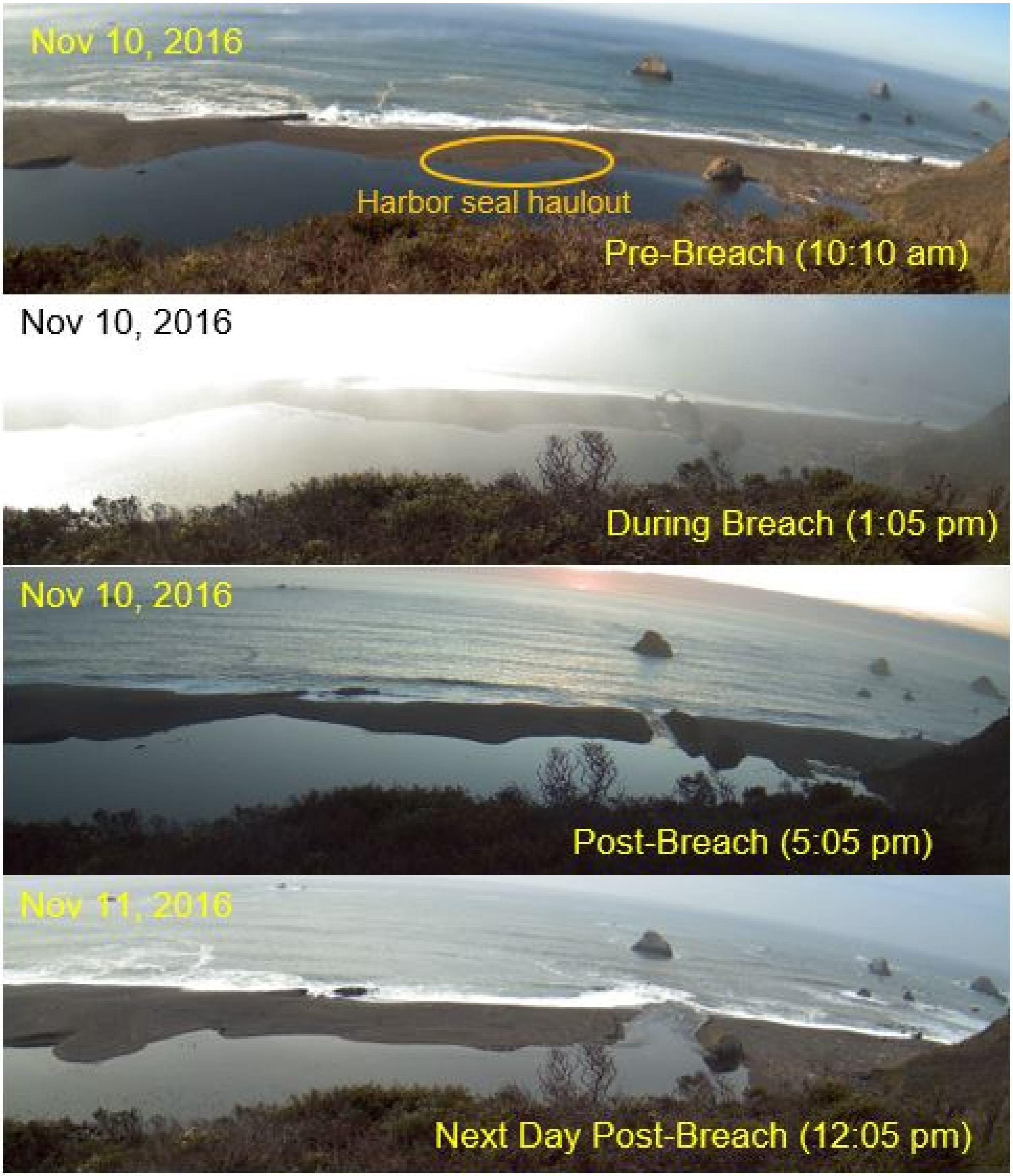


Figure 4.10. Artificial breaching at the mouth of the Russian River Estuary, November 10, 2016. The pilot channel was excavated at the north end of the beach to conserve sand deposits further to the south, per National Marine Fisheries Service (NFMS) request. Photographs show pre- through post-management activity conditions.

Figure 4.11. Artificial breaching at the mouth of the Russian River Estuary, November 10, 2016. The pilot channel was excavated at the north end of the beach to conserve sand deposits further to the south, per NFMS request. Photographs show pre- through post-management activity conditions. Morning fog obscured photos during excavation.

A pre-construction field meeting to discuss pinniped haulouts, permit conditions, and safety issues was held at the Highway 1 overlook in the morning with Water Agency staff prior to staff entering the beach (Figure 4.6) for each breaching event. Project activities were monitored by the project manager, breaching crew lead staff, and biological monitor at the Highway 1 overlook and were in radio contact with the breaching crew on the beach.

The Water Agency breaching crew was comprised of the equipment operator, two staff on foot monitoring safety conditions, and an additional staff member near the jetty and work area boundary to talk with any beach visitors. The excavator was escorted from the Goat Rock State Beach parking lot across the unvegetated sandbar to the river mouth. Excavation of a pilot channel across the sandbar took about 1 to 4 hours to complete, depending on the size of the barrier beach and water surface elevations. The excavator and field crew departed the beach once the barrier beach was breached.

Staff and equipment cautiously and slowly approached the breaching site and harbor seal haulout. The locations of harbor seal haulouts and numbers of seals are shown on Figures 4.7 through 4.11. Following a breaching event harbor seals returned to a haulout (usually at the location of the constructed pilot channel) within a day after a breach. Harbor seal numbers the day after breaching were similar, or higher, than observed prior to breaching. No seal pups were observed on the beach during any breaching event.

Jetty Study

The Russian River Biological Opinion, RPA 2, includes a step if adaptive management of the outlet channel as described, “is not able to reliably achieve the targeted annual and seasonal Estuary management water surface elevations by the end of 2010, Water Agency will draft a study plan for analyzing the effects and role of the Russian River jetty at Jenner on beach permeability, seasonal sand storage and transport, seasonal flood risk, and seasonal water surface elevations in the Estuary. That study will also evaluate alternatives for achieving targeted estuarine management water surface elevations via jetty removal, partial removal of the jetty, jetty notching, and potential use of the jetty as a tool in maintaining the estuary water surface elevations described above.”

ESA PWA, at the request of the Water Agency, developed a plan to study the effects of the Goat Rock State Beach jetty on the Estuary in 2011 (ESA PWA 2011). In addition, it described the recommended approach for developing and assessing the feasibility of alternatives to the existing jetty that may help achieve target estuarine water surface elevations. As such, this study plan fulfills a portion of the Water Agency’s obligations under the Biological Opinion. The Biological Opinion directs the Water Agency to change its management of the Estuary’s water surface elevations with the intent of improving juvenile salmonid habitat while minimizing flood risk. Geophysical field studies were completed in 2014. The draft report was reviewed by resources agencies in 2016. The final report was prepared in 2017 and was included as an appendix to the Russian River Estuary Management Project 2015 Annual Report submitted to the Coastal Commission.

Flood Risk Management Study

The Russian River Biological Opinion, RPA 2, includes a Flood Risk Reduction step if it proves difficult to reliably achieve raised water surface elevation targets based on implementation of a lagoon outlet channel or modification of the existing jetty. Should those actions be unsuccessful in meeting estuarine water surface elevation goals, RPA 2 states that the Water Agency "will evaluate, in coordination with NMFS and other appropriate public agencies, the feasibility of actions to avoid or mitigate damages to structures in the town of Jenner and low-lying properties along the Estuary that are currently threatened with flooding and prolonged inundation when the barrier beach closes and the Estuary's water surface elevation rises above 9 feet. Such actions may include, but are not limited to, elevating structures to avoid flooding or inundation."

The first effort to address flood risk management feasibility was compilation of a preliminary list of structures, properties, and infrastructure that would be subject to flooding/inundation as the result of sandbar formation and if the Estuary were allowed to naturally breach. As required by RPA 2, the Water Agency submitted a preliminary list of properties, structures, and infrastructure that may be subject to inundation if the barrier beach at the mouth of the Russian River was allowed to naturally breach. This preliminary list was updated for the California Coastal Commission Coastal Development Permit application process. Allowing Estuary water surface elevations to rise to between 10 and 12 feet NGVD (the estimated water surface elevation if the barrier beach was allowed to naturally breach per consultation with NMFS) may potentially inundate portions of properties.

The Water Agency was awarded federal funding from the National Oceanic and Atmospheric Administration (NOAA) under its Habitat Blueprint framework. The Habitat Blueprint is NOAA's strategy to integrate habitat conservation throughout NOAA, focus efforts in priority areas, and leverage internal and external collaborations to achieve measurable benefits within key habitats. The Russian River watershed was selected as the nation's first Habitat Focus Area under the Habitat Blueprint strategy. One of the federally-funded projects was an effort to expand the United States Geological Survey (USGS) sea level rise model (the Coast Storm Modeling System or CoSMoS) from Bodega Bay north along the Sonoma Coast to Point Area, including the Russian River Estuary up to Duncans Mills, to be used to inform adaptation planning and Estuary management efforts. In 2016, the USGS completed the Sonoma Coast and Russian River Estuary model scenarios that included an open Russian River mouth. These model scenarios were incorporated into the Our Coast, Our Future (OCOF) web platform by Point Blue Conservation Science (http://beta.ourcoastourfuture.org/index.php?page=russian-river-project-team). The draft scenarios and maps were reviewed by the partner organizations in 2016. Work continued on the model scenarios for a closed Russian River mouth and were scheduled for completion in 2017. This effort included staff of the County of Sonoma working on the Local Coastal Plan update. The County's Permit Resources and Management Department is updating its Local Coastal Plan, including consideration of sea level rise impacts to the lower Russian River. Sonoma Water hopes to use the CoSMoS and OCOF information to inform future flood risk feasibility studies of sea level rise and climate change effects on estuary flood risk and habitat management.

Pinniped Annual Monitoring

In addition to the Flood Management, Water Quality, and Habitat Conditions monitoring summarized in this report, Sonoma Water also monitors pinnipeds at the mouth of the Russian River.

An Incidental Harassment Authorization (IHA) was issued by the NMFS pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act (16 U.S.C 1361 et seq.) to take small numbers of marine mammals, by Level B harassment, incidental to the Water Agency's Estuary Management Project (issued April 21, 2016 , NMFS IHA). A summary of the results of 2016 pinniped monitoring as reported in the *Russian River Estuary Management Project, Marine Mammal Protection Act Incidental Harassment Authorization, Report of Activities and Monitoring Results – January 1 to December 31, 2016* (SCWA 2017; Appendix 4.4) are provided below.

Harbor seals (*Phoca vitulina richardsi*) regularly haul out at the mouth of the Russian River (Jenner haul-out). California sea lions (*Zalophus californianus*) and northern elephant seals (*Mirounga angustirostris*) are occasionally observed at the haul-out. There are also several known resting areas in the river at logs and rock piles.

Pinniped monitoring was performed in accordance with the requirements of the NMFS IHA issued April 21, 2016, and the Russian River Estuary Management Activities Pinniped Monitoring Plan (Sonoma County Water Agency and Stewards of the Coast and Redwoods 2016). Baseline monitoring was performed to gather additional information about the population of harbor seals utilizing the Jenner haul-out including population trends, patterns in seasonal abundance and the influence of barrier beach condition on harbor seal abundance. Pinniped monitoring was also conducted in relation to Water Agency water level management events (lagoon outlet channel implementation and artificial breaching). Estuary management monitoring occurred during the Water Agency's monthly topographic surveys of the barrier beach and biological and physical monitoring of the Estuary.

The purpose of the Russian River Estuary Management Project Pinniped Monitoring Plan (Sonoma County Water Agency and Stewards of the Coast and Redwoods 2016) is to detect the response of pinnipeds to estuary management activities at the Russian River estuary. Specifically, the following questions are of interest: 1) Under what conditions do pinnipeds haul out at the Russian River estuary mouth at Jenner?; 2) How do seals at the Jenner haul-out respond to activities associated with the construction and maintenance of the lagoon outlet channel and artificial breaching activities?; 3) Does the number of seals at the Jenner haul-out significantly differ from historic averages with formation of a summer (May 15th to October 15th) lagoon in the Russian River estuary?; and 4) Are seals at the Jenner haul-out displaced to nearby river and coastal haul-outs when the mouth remains closed in the summer?

The Estuary management and monitoring activities in 2016 resulted in incidental harassment (Level B harassment) of 1,915 harbor seals, well under the total allowed by NMFS IHA. The Russian River estuary management activities in 2015, 2014, 2013, 2012, 2011 and 2010 resulted in incidental harassment (Level B harassment) of 2,383, 2,121, 1,351, 208, 42 and 290 harbor seals, respectively

Harbor seals are found at the mouth of the Russian River (Jenner haul-out) throughout the year. They are observed on the beach throughout the tidal cycle and at any time of day. Our baseline pinniped monitoring concluded that tidal state and time of day influenced harbor seal abundance at the Jenner haul-out, with seals less abundant in the early morning and at high tide (SCWA 2012). Harbor seals were most abundant on the Jenner haul-out in July during their annual molt (SCWA 2012), with these same trends being observed in subsequent years (SCWA 2013, 2014, 2016). Seasonal variation in the abundance of harbor seals at their haul-out locations is commonly observed throughout their range (Allen et al. 1989, Stewart and Yochem 1994, Gemmer 2002). The variation in their abundance can mostly be explained by changes in their biological and physiological requirements throughout the year.

Harbor seals will use the beach when there is an open channel or when a barrier beach has formed, however, the number of seals at Jenner was influenced by river mouth condition. Daily average seal abundance was lower during closed conditions compared to open conditions. The closure of the barrier beach in September likely contributed to the low abundance of seals on the beach for the month. This effect is also closely related to time of year, since most closures occur during the fall and winter, when seal abundance is low.

The response of harbor seals at the Jenner haul-out to water level management activities in 2016 was similar to the responses observed in previous years of monitoring (Merritt Smith Consulting 1997, 1998, 1999, 2000; Sonoma County Water Agency and Merritt Smith Consulting 2001; SCWA 2011, 2012, 2013, 2014 and 2016). Harbor seals alerted to the sound of equipment on the beach and left the haul-out as the crew and equipment approached closer on the beach. When breaching activities were conducted south of the haul-out, or when seals were hauled out on the ocean side of the beach, seals often remained on the beach during all or some of the breaching activity. This indicates that seals are less disturbed by activities when equipment and crew do not pass directly past their haul-out.

Since the beginning of the modified estuary management as a result of the NMFS 2008 Biological Opinion a lagoon outlet channel has been implemented a few times (July 2010, twice in June 2016). In 2016 both attempts to create an outlet channel failed to an open river mouth condition within one day. Observations when a barrier beach has formed during the lagoon management period provide information as to how harbor seals respond when aquatic access between the estuary and the ocean is limited. A barrier beach has formed during the lagoon management period sixteen times, the longest incidence lasting 29 days, with an average duration of fourteen days. While seal abundance was lower during closed conditions, overall there continues to be a slight increasing trend in seal abundance. These results indicate that while seal abundance may exhibit a short term decline during closed conditions it has not inhibited seals from using the Jenner haul-out during any period of the year. We conclude that the effect of barrier beach condition on seal abundance represents only a short term response, and is not an indication that seals are less likely to choose Jenner as a haul-out overall. We do not yet know how seals would respond to a maintained lagoon outlet channel.

Conclusions and Recommendations

Five inlet closures occurred within the lagoon management period; two, 1-2 week long closures in June, a 2-week long closure in early July, and two, 1-3 week closures in September and October. In each closure, barrier beach formation was associated with a neap tide cycle and long period wave conditions. Outlet channels were excavated by the Water Agency during the two closures in June, but, in both cases, the outlet channel scoured and self-breached within a day. The first outlet channel excavation occurred during rising ocean tides, but this was not sufficient to avoid self-breaching. The July and September events ended with self-breaches, as access was an issue due to steep beach topography adjacent to the jetty groin. The last event led to an artificial breach, as the estuary water surface elevation approached flood stage and the closure had persisted past the end of the lagoon management period.

Outlet channels were attempted in two locations in 2016; one within 100 feet of the jetty groin and another roughly 600 feet north of the groin. Siting of the outlet channels was influenced by lack of inlet migration north of Haystack Rock, which led to high (17-19 ft NGVD) beach crest elevations, well above the target elevation for outlet channels.

Outside of the lagoon management season, there were four mouth closures in 2016. The Water Agency artificially breached the barrier beach at the Russian River mouth outside the lagoon management period twice in 2016, avoiding impacts to juvenile steelhead rearing habitat. The breachings were necessary to minimize flood risk to low-lying structures.

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4.1 Water Quality Monitoring

Water quality monitoring was conducted in the lower, middle, and upper reaches of the Russian River Estuary, including two tributaries and the Maximum Backwater Area (MBA), between the mouth of the river at Jenner and Vacation Beach near Guerneville. Water Agency staff continued to collect data to establish baseline information on water quality in the Estuary, gain a better understanding of the longitudinal and vertical water quality profile during the ebb and flow of the tide, and track changes to the water quality profile that may occur during periods of barrier beach closure, partial or full lagoon formation, lagoon outlet channel implementation, and sandbar breach.

Saline water is denser than freshwater and a salinity “wedge” (halocline) forms in the Estuary as freshwater outflow passes over the denser tidal inflow. During the Lagoon Management Period, the lower and middle reaches of the Estuary up to Sheephouse Creek are predominantly saline environments with a thin freshwater layer that flows over the denser saltwater. The upper reach of the Estuary transitions to a predominantly freshwater environment, which is periodically underlain by a denser, saltwater layer that migrates upstream to Duncans Mills during summer low flow conditions and barrier beach closure. Additionally, river flows, tides, topography, and wind action affect the amount of mixing of the water column at various longitudinal and vertical positions within the reaches of the Estuary. The Maximum Backwater Area encompasses the area of the river between Duncans Mills and Vacation Beach that is generally outside the influence of saline water, but within the upper extent of inundation and backwatering that can occur during tidal cycles and lagoon formation.

Methods

Continuous Multi-Parameter Monitoring

Water quality was monitored using YSI Series 6600 multi-parameter datasondes. Hourly salinity (parts per thousand), water temperature (degrees Celsius), dissolved oxygen (percent saturation), dissolved oxygen (milligrams per liter), and pH (hydrogen ion) data were collected. Datasondes were cleaned and recalibrated periodically following the YSI User Manual procedures, and data was downloaded during each calibration event.

Seven stations were established for continuous water quality monitoring, including three stations in the mainstem Estuary, two tributary stations, and two stations in the MBA near Monte Rio (Figure 4.1.1). One mainstem Estuary station was located in the middle reach at Patty’s Rock upstream of Penny Island (Patty’s Rock Station). One tributary station was located in the mouth of Willow Creek, which flows into the middle reach of the Estuary (Willow Creek Station). Two mainstem Estuary stations were located in the upper reach; downstream of Freezeout Creek in Duncans Mills (Freezeout Creek Station) and downstream of Austin Creek in Brown’s Pool (Brown’s Pool Station). The other tributary station was located downstream of the first steel bridge in lower Austin Creek, which flows into the mainstem Russian River above Brown’s Pool Station. Finally, two mainstem stations were located in the MBA: in a pool across from Patterson Point in Villa Grande (Patterson Point Station) and downstream of Monte Rio Beach (Monte Rio

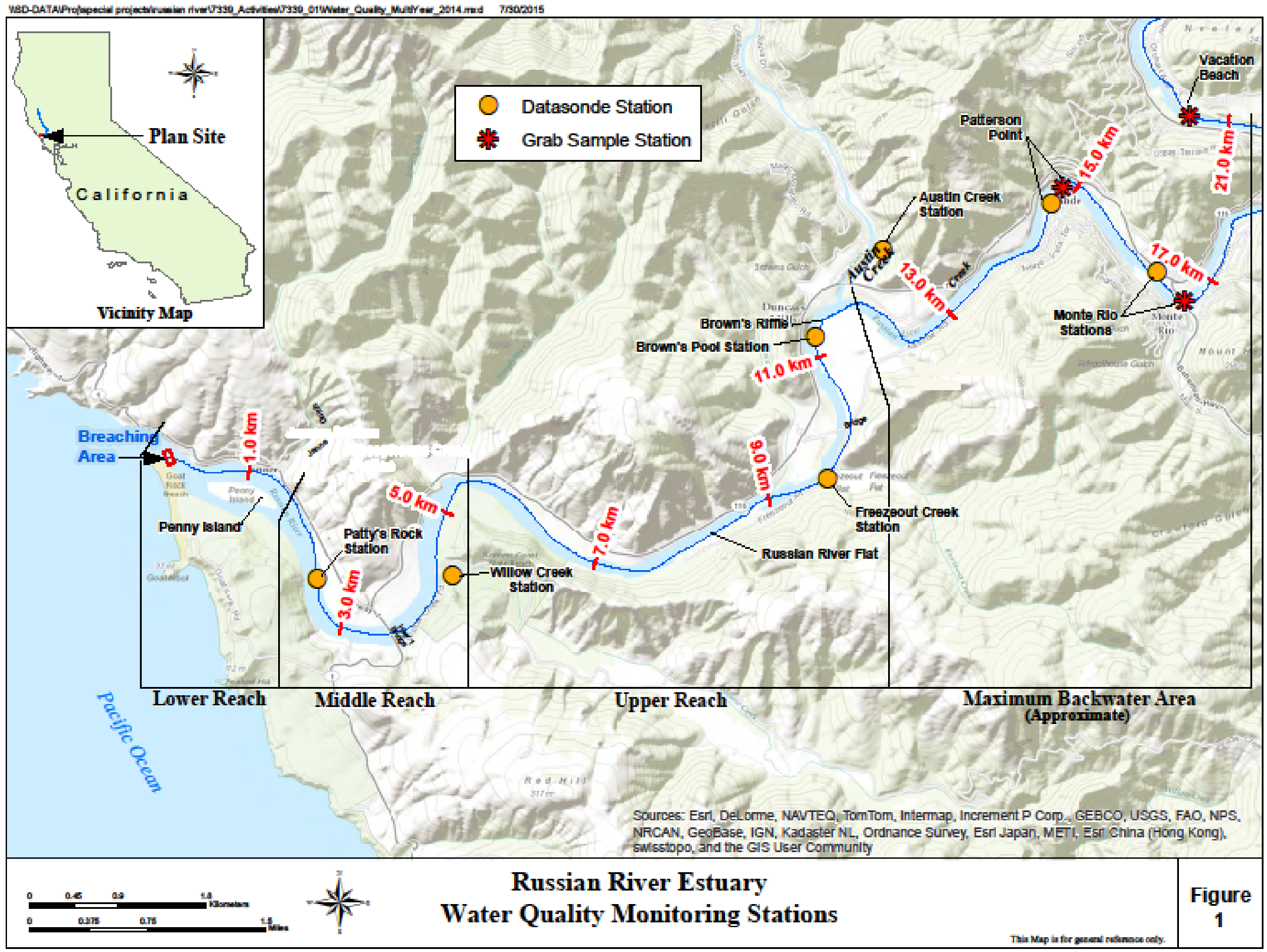

Figure 4.1.1. 2016 Russian River Estuary Water Quality Monitoring Stations

Station). An eighth station was established in the middle reach at Sheephouse Creek, however due to equipment malfunction no data was collected at this station in 2016. The rationale for choosing mainstem Estuary sites, including the Brown's Pool station, was to locate the deepest holes at various points throughout the Estuary to obtain the fullest vertical profiles possible and to monitor salinity circulation and stratification, hypoxic and/or anoxic events, and temperature stratification. Sondes were located near the mouths of Willow and Austin creeks to collect baseline water quality conditions and monitor potential changes to water quality (e.g. salinity intrusion) resulting from tidal cycling or inundation during partial or full lagoon formation. The Patterson Point and Monte Rio stations were established to monitor potential changes to water quality conditions (including potential salinity migration) in the MBA while inundated during lagoon formation (Figure 4.1.1).

Mainstem Estuary and MBA monitoring stations up to Patterson Point were comprised of a concrete anchor attached to a steel cable suspended from the surface by a large buoy (Figure 4.1.2).

The Patty's Rock, Freezeout Creek, Brown's Pool, and Patterson Point stations had a vertical array of two datasondes to collect water quality profiles. The Patty's Rock station, located in the middle reach of the Estuary, is predominantly saline and had sondes placed near the surface at approximately 1 meter depth (~1m), and at the mid-depth (~4-5m) portion of the water column. Stations in the upper reach of the Estuary, where the halocline is deeper and the water is predominantly fresh to brackish, had sondes placed at the bottom (~5-11m) and mid-depth (~3-6m) portions of the water column. The Patterson Point monitoring station, located in the MBA, also had datasondes placed at the bottom (~9-11m) and mid-depth (~6-7m) portions of the water column (Figure 4.1.2). Sondes were located in this manner to track vertical and longitudinal changes in water quality characteristics during periods of tidal circulation, barrier beach closure, lagoon formation, lagoon outlet channel implementation, and sandbar breach.

The monitoring stations in Austin Creek, Willow Creek, and at Monte Rio consisted of one datasonde suspended at approximately mid-depth (~1m during open conditions) in the thalweg at each respective site.

Most of the stations were deployed from April through late November. The Austin Creek and Willow Creek sondes were deployed from April to November.

Grab Sample Collection

In 2016, Water Agency staff continued to conduct nutrient and indicator bacteria grab sampling at three stations in the Russian River Estuary and MBA, including three stations established in 2010: the Jenner Boat Ramp (Jenner Station); Casini Ranch across from the mouth of Austin Creek (Casini Ranch Station); and just downstream of the Monte Rio Bridge (Monte Rio Station). The 2016 grab sampling effort represented the third year of collecting samples at

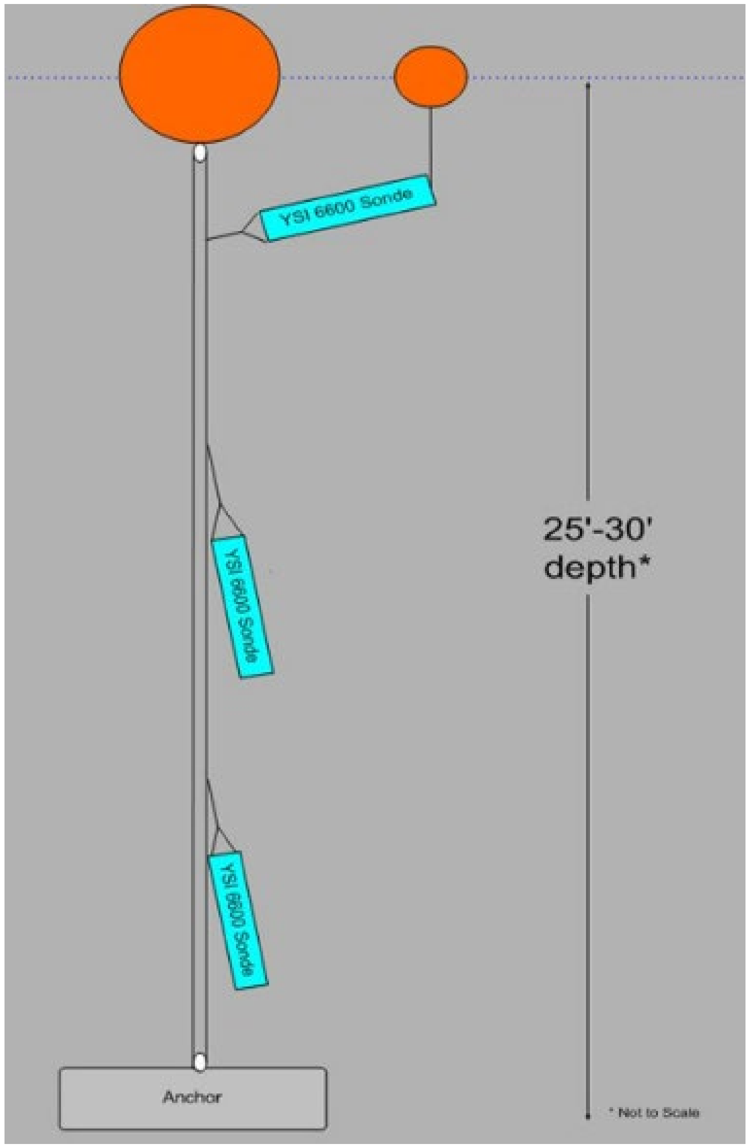

Figure 4.1.2. Typical Russian River Estuary monitoring station datasonde array.

Patterson Point in Villa Grande (Patterson Point Station); and just downstream of the Vacation Beach summer dam (Vacation Beach station). Refer to Figure 4.1.1 for grab sampling locations.

Water Agency staff collected grab samples weekly from May 10 to October 18. Additional focused sampling (collecting three samples over a ten day period) was conducted following or during specific river management and operational events including: barrier beach closure, lagoon outlet channel implementation, sandbar breach, or removal of summer recreational dams.

Nutrient sampling was conducted for total organic nitrogen, ammonia, unionized ammonia, nitrate, nitrite, total Kjeldahl nitrogen, total nitrogen, and total phosphorus, as well as for chlorophyll a, which is a measurable parameter of algal growth that can be tied to excessive nutrient concentrations and reflect a biostimulatory response. Grab samples were collected for the presence of indicator bacteria including total coliforms, E. coli and Enterococcus. These bacteria are considered indicators of water quality conditions that may be a concern for water contact recreation and public health. The results of sampling conducted for total orthophosphate, dissolved organic carbon, total organic carbon, total dissolved solids, and turbidity are included as Appendix 4.5; however, an analysis and discussion of these constituents is not included in this report. Temperature, dissolved oxygen, pH, salinity, specific conductance, and turbidity values were recorded using a YSI 6600 datasonde during grab sampling events and are included in Appendix 4.5.

Results

Water quality conditions in 2016 were similar to trends observed in sampling from 2004 to 2015. The lower and middle reaches are predominantly saline environments with a thin freshwater layer that flows over the denser saltwater layer. The upper reach transitions to a predominantly freshwater environment, which is periodically underlain by a denser, saltwater layer that migrates up and downstream and appears to be affected in part by freshwater inflow rates, tidal inundation, barrier beach closure, and subsequent tidal cycles following reopening of the barrier beach. The river upstream of Brown’s Pool is considered predominantly freshwater habitat. The lower and middle reaches of the Estuary are subject to tidally-influenced fluctuations in water depth during open conditions and inundation during barrier beach closure, as is the upper reach and the MBA to a lesser degree.

Table 4.1.1 presents a summary of minimum, mean, and maximum values for temperature, depth, dissolved oxygen (DO), pH, and salinity recorded at the various datasonde monitoring stations. Data associated with malfunctioning datasonde equipment has been removed from the data sets, resulting in the data gaps observed in the graphs presented as Figures 4.1.3 through 4.1.37. These data gaps may affect minimum, mean, and maximum values of the various constituents monitored in 2016, including temperature, dissolved oxygen, pH, and salinity at the Austin Creek sonde and the Brown’s Pool bottom sonde through the season, the Patty’s Rock surface sonde in August and early September, and the Monte Rio sonde after late August.

Table 4.1.1. Russian River Estuary 2016 Water Quality Monitoring Results. Minimum, mean, and maximum values for temperature (degrees Celsius), depth (meters), dissolved oxygen (percent) saturation, dissolved oxygen concentration (milligrams per Liter), hydrogen ion (pH units), and salinity (parts per thousand).

Monitoring Station	Temperature	Depth	Dissolved Oxygen	Dissolved Oxygen	Hydrogen Ion	Salinity
Sonde	(°C)	(m)	(mg/L)	(%) saturation	(pH)	(ppt)
Patty's Rock						
Surface						
May 19, 2016 - November 3, 2016						
Min	11.2	0.8	6.9	72.0	7.5	0.2
Mean	17.3	0.9	9.9	110.1	8.2	9.3
Max	**23.0**	**1.0**	**21.6**	**239.1**	**8.9**	**32.5**
Mid-Depth						
May 19, 2016 - November 3, 2016						
Min	11.5	4.3	0.8	9.5	7.1	0.2
Mean	14.2	4.6	7.8	90.7	7.7	29.4
Max	**19.2**	**4.7**	**17.7**	**212.3**	**8.3**	**33.0**
Willow Creek						
Mid-Depth						
April 15, 2016 - November 30, 2016						
Min	8.2	0.2	0.1	0.5	6.6	0.1
Mean	16.7	0.9	8.3	86.6	7.7	3.6
Max	**23.1**	**2.5**	**17.1**	**189.8**	**9.1**	**24.2**
Freezeout Creek						
Mid-Depth						
May 19, 2016 - November 3, 2016						
Min	13.9	3.2	2.1	24.1	6.9	0.1
Mean	20.6	3.5	8.4	93.7	7.9	1.1
Max	**24.3**	**3.7**	**11.5**	**130.4**	**8.5**	**18.8**
Bottom						
May 19, 2016 - November 3, 2016						
Min	14.0	5.1	0.0	0.0	6.9	0.1
Mean	20.5	6.3	4.2	46.2	7.6	3.3
Max	**24.3**	**7.6**	**11.4**	**131.9**	**8.5**	**19.2**
Brown's Pool						
Mid-Depth						
May 19, 2016 - November 3, 2016						
Min	14.0	4.9	3.9	45.1	7.3	0.1
Mean	20.5	5.2	8.2	91.2	7.7	0.3
Max	**24.8**	**5.5**	**10.9**	**126.7**	**8.3**	**10.7**
Bottom						
May 19, 2016 - November 3, 2016						
Min	14.0	9.0			5.4	0.1
Mean	18.7	9.9			6.8	2.2
Max	**24.7**	**10.5**			**8.2**	**11.1**

(continues on next page)

Table 4.1.1 (cont.). Russian River Estuary 2016 Water Quality Monitoring Results. Minimum, mean, and maximum values for temperature (degrees Celsius), depth (meters), dissolved oxygen (percent) saturation, dissolved oxygen concentration (milligrams per Liter), hydrogen ion (pH units), and salinity (parts per thousand).

Monitoring Station	Temperature	Depth	Dissolved Oxygen	Dissolved Oxygen	Hydrogen Ion	Salinity
Sonde	(°C)	(m)	(mg/L)	(%) saturation	(pH)	(ppt)
Austin Creek						
Surface						
April 26, 2016 - November 15, 2016						
Min	12.5	0.0	1.2	12.2	6.6	0.1
Mean	16.5	0.5	8.4	82.6	7.4	0.2
Max	**20.5**	**1.7**	**10.4**	**102.6**	**8.0**	**0.2**
Patterson Point						
Mid-Depth						
May 9, 2016 - November 7, 2016						
Min	14.2	6.3	3.2	36.5	7.2	0.1
Mean	20.3	6.6	7.7	84.5	7.7	0.1
Max	**25.5**	**7.1**	**10.9**	**122.3**	**8.2**	**0.2**
Bottom						
May 9, 2016 - November 7, 2016						
Min	12.1	8.8	0.1	0.9	5.4	0.1
Mean	17.1	10.2	3.6	36.8	6.9	0.2
Max	**21.5**	**11.3**	**10.9**	**114.2**	**8.1**	**0.3**
Monte Rio						
Mid-Depth						
May 9, 2016 - August 22, 2016						
Min	16.7	0.7	6.7	78.8	7.5	0.1
Mean	22.6	1.1	8.5	98.4	7.8	0.1
Max	**26.3**	**2.1**	**11.5**	**125.6**	**8.4**	**0.2**

Although gaps exist in the 2016 data that affect sample statistics, Agency staff has collected long time-series data on an hourly frequency for several years at most of these stations, and it is unlikely that the missing data appreciably affected the broader understanding of water quality conditions within the estuary. The following sections provide a brief discussion of the results observed for each parameter monitored.

Salinity

Full strength seawater has a salinity of approximately 35 parts per thousand (ppt), with salinity decreasing from the ocean to the upstream limit of the Estuary, which is considered freshwater at approximately 0.5 ppt (Horne, 1994). The Patty's Rock mid-depth sonde in the middle reach was located in a predominantly saline environment, whereas the surface sonde was located at the saltwater-freshwater interface (halocline or salt wedge) and recorded both freshwater and saltwater conditions. In the lower and middle reaches of the Estuary, salinities can range as high as 30 ppt in the saltwater layer, with brackish conditions prevailing at the upper end of the salt wedge, to less than 1 ppt in the freshwater layer on the surface. The Willow Creek sonde was located just upstream of the confluence with the Russian River, where predominantly freshwater conditions observed in the creek during higher springtime flows transitioned to a brackish environment during lower dry season flows.

In the upper reach, the Estuary typically transitions from predominantly saline conditions to brackish and freshwater conditions in the Heron Rookery area. Upstream, the Freezeout Creek station is located in a predominantly freshwater environment; however, brackish conditions can occur in the lower half of the water column during open estuary conditions with lower in-stream flows, as well as during barrier beach closure or perched conditions. The Brown's Pool station is located in predominantly freshwater habitat in the upper reach of the Estuary, just downstream of the confluence with Austin Creek and the beginning of the MBA; however, brackish water was observed to occur at the bottom of the pool periodically through the 2016 monitoring season and at mid-depth during a closure in late October.

The Austin Creek, Patterson Point and Monte Rio stations are located in the MBA in freshwater habitat that can become inundated during high tides, barrier beach closures, perched conditions, and lagoon formation. Elevated salinity levels were not observed at any of the stations in the MBA during either open river mouth or closed barrier beach conditions in 2016.

Lower and Middle Reach Salinity

The surface sonde at the Patty's Rock station was suspended at a depth of approximately 1 meter, and experienced frequent hourly fluctuations in salinity during open conditions. These fluctuations are influenced by freshwater inflows, tidal movement and expansion and contraction of the salt wedge. The freshwater layer was observed to deepen and become more persistent at the surface sonde during closed barrier beach conditions in the spring and fall (Figure 4.1.3). Concentrations ranged from 0.2 to 32.5 ppt at the Patty's Rock surface sonde with a mean salinity value of 9.3 ppt (Table 4.1.1).

The mid-depth sonde at the Patty's Rock station was suspended at a depth of approximately 4 to 5 meters, and also experienced frequent fluctuations in salinity during open and closed conditions, though to a lesser degree than the surface sonde. Concentrations ranged from 0.2 to 33.0 ppt at the Patty's Rock mid-depth sonde with a mean salinity value of 29.4 ppt (Table 4.1.1). Minimum concentrations were observed to occur at the Patty's Rock mid-depth sonde in November as flows increased during storm events (Figure 4.1.3).

The Estuary experienced five closures during the 2016 management period, including a closure that lasted 21 days from 10 September to 30 September before opening naturally (Figure 4.1.3). Declines in salinity during barrier beach closure and lagoon formation were due to a combination of freshwater inflows increasing the depth of the freshwater layer over the salt layer, a reduction in tidal inflow, the compression and leveling out of the salt layer, and seepage of saline water through the barrier beach. Salinity generally returned to pre-closure levels after the barrier beach reopened, although the time required to return to pre-closure conditions varied between closure events. This variability was related to the strength of subsequent tidal cycles, freshwater inflow rates, topography, relative location within the Estuary, and to a lesser degree, wind mixing.

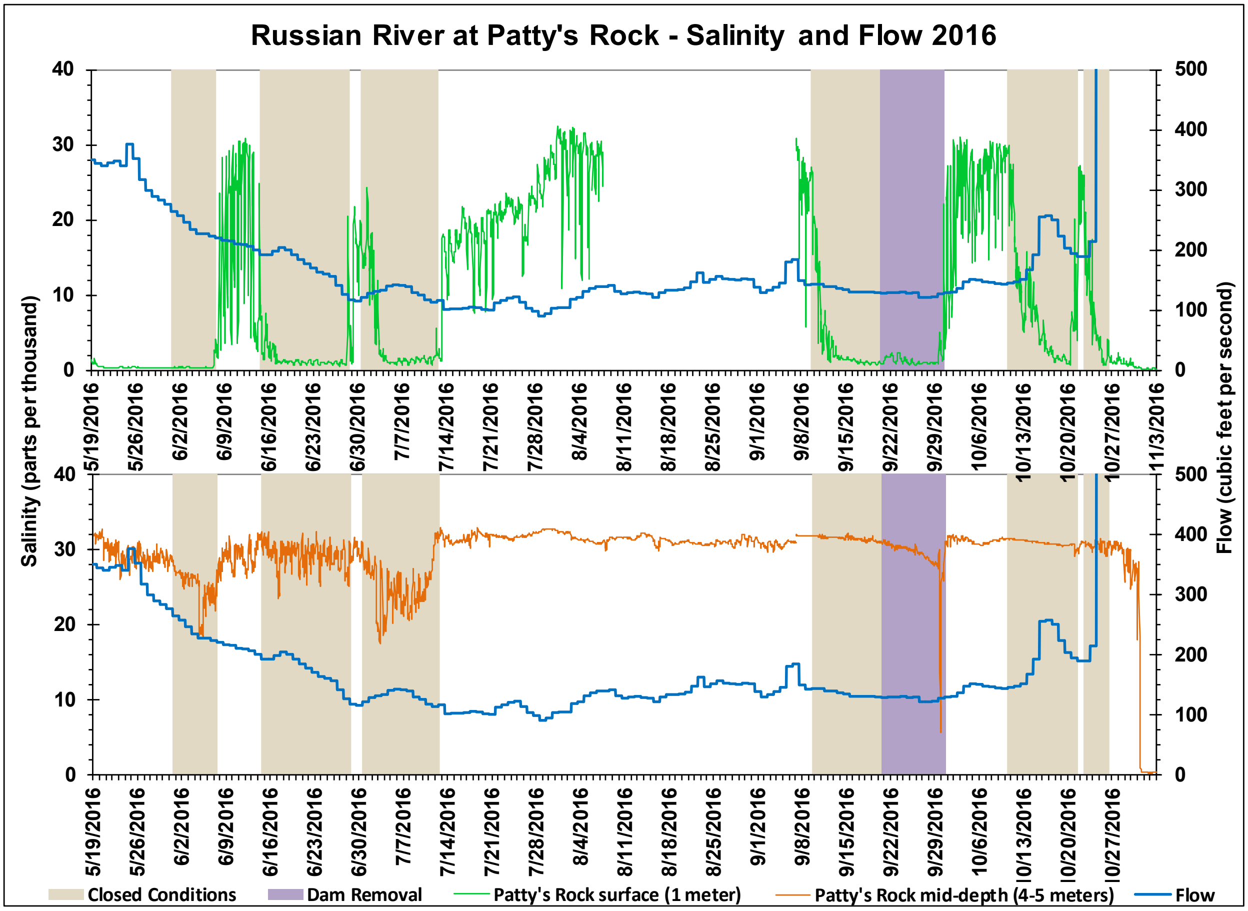

Figure 4.1.3. 2016 Russian River at Patty's Rock Salinity and Flow Graph

The Willow Creek station was located in predominantly freshwater habitat through May until spring flows receded below 250 cfs in the mainstem Russian River and increased tidal action allowed saline water to migrate to this station during open conditions. Salinity was observed to slightly decline during the two closures in late-June and early-July, but remained brackish through the rest of the monitoring season, including during late season closures (Figure 4.1.4). Salinity was also observed to decrease following the opening of the barrier beach in September and October, however, brackish conditions generally returned within a few days.

Salinity concentrations fluctuated significantly at times during open conditions with concentrations that ranged between 1 and 18 ppt from mid-July to early September. The mean salinity concentration observed at the Willow Creek station was 3.6 ppt, with a minimum concentration of 0.1 ppt, and a maximum concentration of 24.2 ppt (Table 4.1.1).

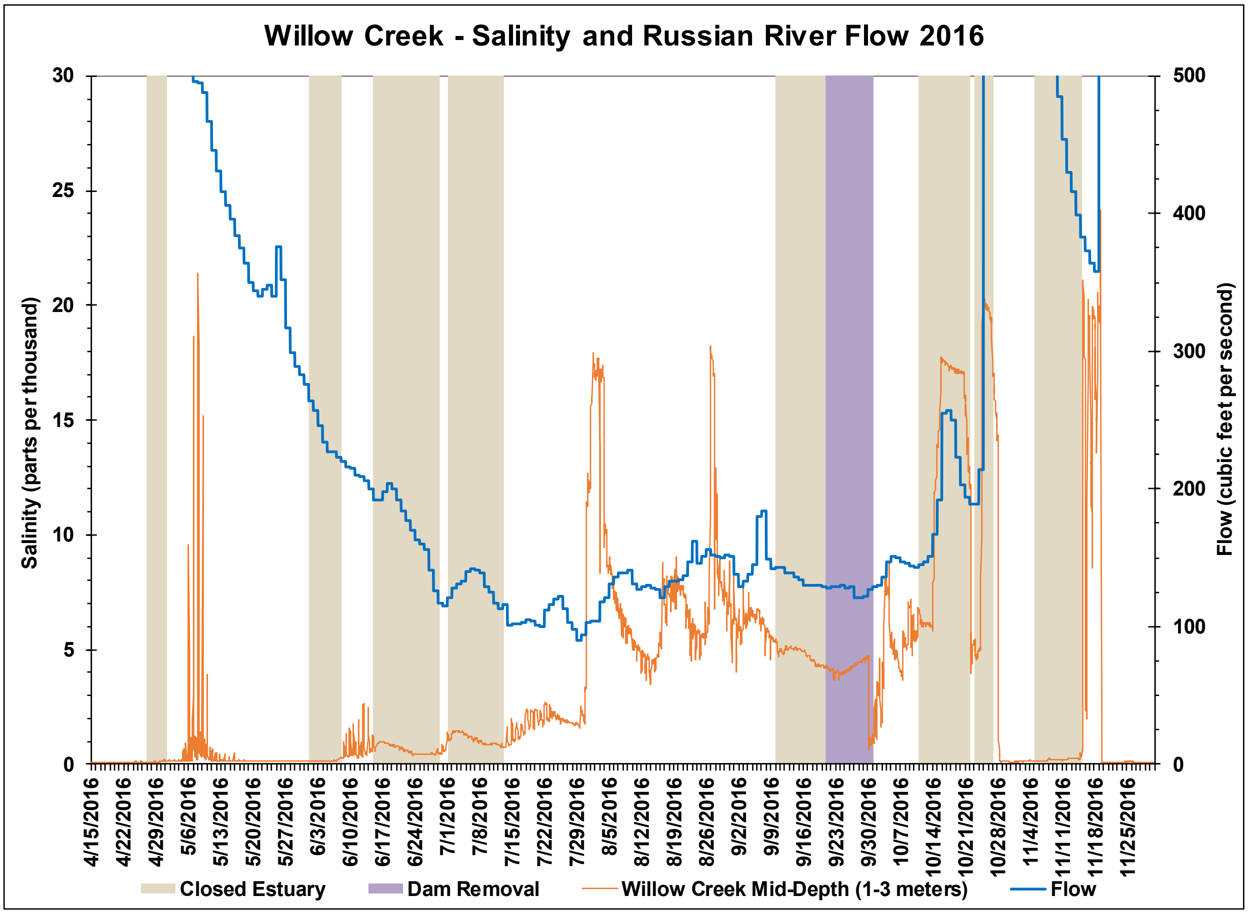

Figure 4.1.4. 2016 Willow Creek Salinity and Russian River Flow Graph

Upper Reach Salinity

Two stations were monitored in the upper reach in 2016; Freezeout Creek and Brown's Pool. Both stations included a bottom sonde and a mid-depth sonde. Sondes were located in this manner to track changes in the presence and concentration of salinity in the water column as well as the presence of thermal refugia for salmonids.

The Freezeout Creek station is located at River Kilometer 9.5 (RK 9.5), which is approximately 9.5 km upstream from the river mouth, in a pool approximately 300 meters downstream of the confluence of Freezeout Creek and the mainstem of the river. This station was located in a predominantly freshwater habitat that was subject to elevated salinity levels as the salt wedge migrated up the Estuary during both open and closed conditions (Figure 4.1.5). The elevated salinity levels were primarily observed at the bottom sonde, though elevated salinity was also seen at the mid-depth sonde during open and closed conditions. The bottom sonde at Freezeout Creek had a mean salinity concentration of 3.3 ppt, and salinity levels that ranged from 0.1 to 19.2 ppt (Table 4.1.1). The mid-depth sonde at Freezeout Creek had a mean salinity concentration of 1.1 ppt, and salinity levels that ranged from 0.1 to 18.8 ppt (Table 4.1.1).

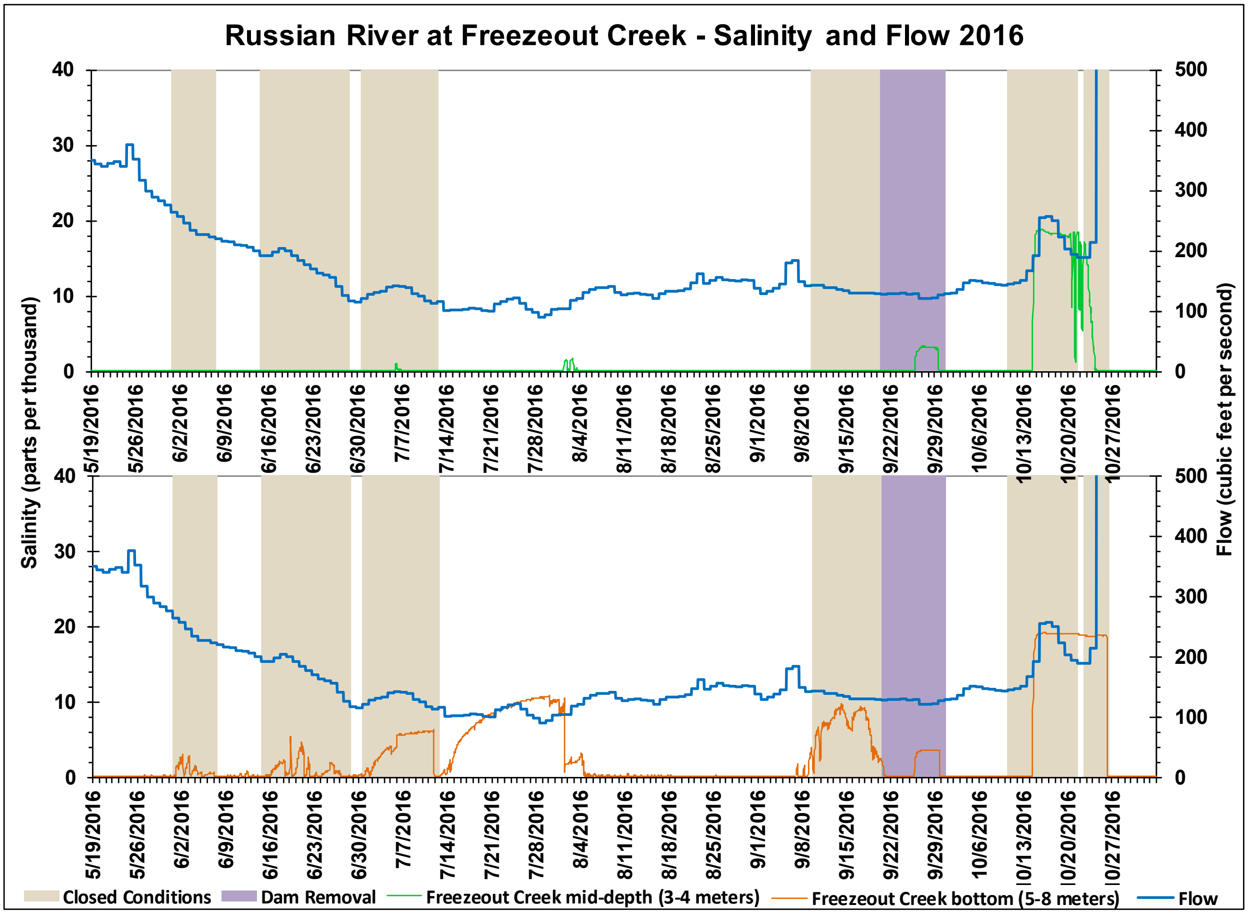

Figure 4.1.5. 2016 Russian River at Freezeout Creek Salinity and Flow Graph

The Brown's Pool station is located at RK 11.3 in a pool that is approximately 10m deep. Brown's Pool is located immediately downstream of Brown's Riffle (RK 11.4) and the confluence of Austin Creek and the mainstem Russian River, which is located at RK 11.65. Brown's Riffle is generally considered the demarcation between the Estuary and the MBA, where salinity levels have not been observed to occur past this point. This station was also located in predominantly freshwater habitat that was subject to elevated salinity levels as the salt wedge migrated up the Estuary during both open and closed conditions (Figure 4.1.6). Similar to Freezeout Creek, elevated salinity levels were predominantly observed at the Brown's Pool bottom sonde, though elevated salinity was also seen at the mid-depth sonde during closed conditions in October (Figure 4.1.6).

During the barrier beach closure in early October, salinity concentrations at Brown's Pool were observed to increase to approximately 11 ppt at the mid-depth and bottom sondes by 20 October. Salinity concentrations were observed to decrease to freshwater conditions at the mid-depth sonde before the barrier beach was opened on October 22. Salinity also briefly decreased at the bottom sonde before returning to brackish conditions, which persisted into the next closure until being replaced by freshwater on October 24 during elevated storm flows (Figure 4.1.6). The bottom sonde at Brown's Pool had a mean salinity concentration of 2.2 ppt, and

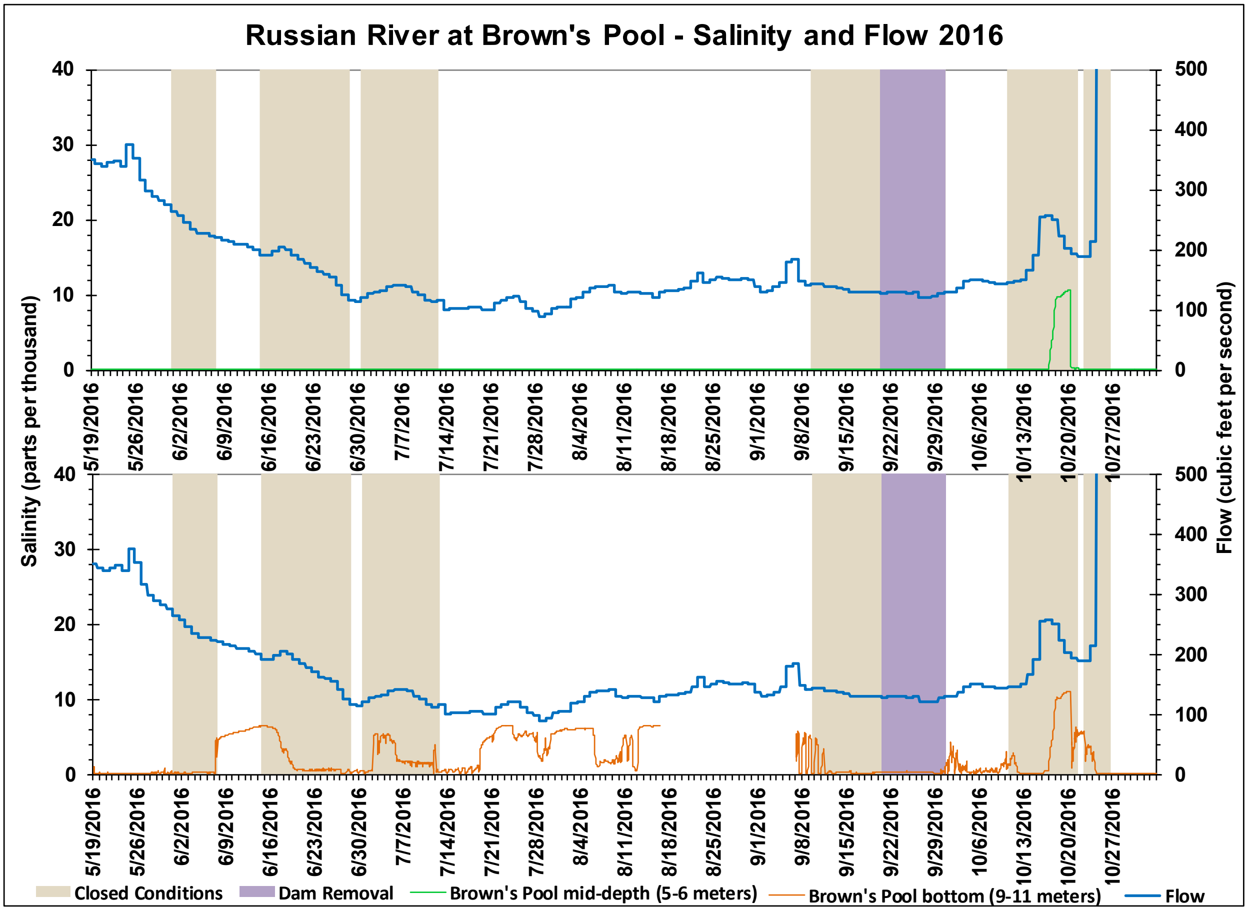

Figure 4.1.6. 2016 Russian River at Brown's Pool Salinity and Flow Graph

salinity levels that ranged from 0.1 to 11.1 ppt (Table 4.1.1). The mid-depth sonde at Brown's Pool had a mean salinity concentration of 0.3 ppt, and salinity levels that ranged from 0.1 to 10.7 ppt (Table 4.1.1).

Maximum Backwater Area Salinity

Three stations were located in the MBA, including one tributary station in lower Austin Creek and two mainstem Russian River stations located in Patterson Point (RK 14.9) and Monte Rio (RK 16.1) (Figure 4.1.1). None of these three stations were observed to have salinity levels above normal background conditions expected in freshwater habitats, during both open and closed barrier beach conditions (Figures 4.1.7 through 4.1.9). The Monte Rio sonde only has data through late August. The sonde was not recovered after that date and was lost in the subsequent high winter flows.

The Austin Creek station had a mean salinity concentration of 0.2 ppt, with a minimum of 0.1 ppt and a maximum of 0.2 ppt. The Patterson Point bottom sonde had a mean salinity concentration of 0.2 ppt, a minimum concentration of 0.1 ppt, and a maximum concentration of 0.3 ppt. The Patterson Point mid-depth sonde had a mean salinity concentration of 0.1 ppt, a minimum concentration of 0.1 ppt, and a maximum concentration of 0.2 ppt. The Monte Rio station had a mean salinity concentration of 0.1 ppt, a minimum concentration of 0.1 ppt, and a maximum concentration of 0.2 ppt.

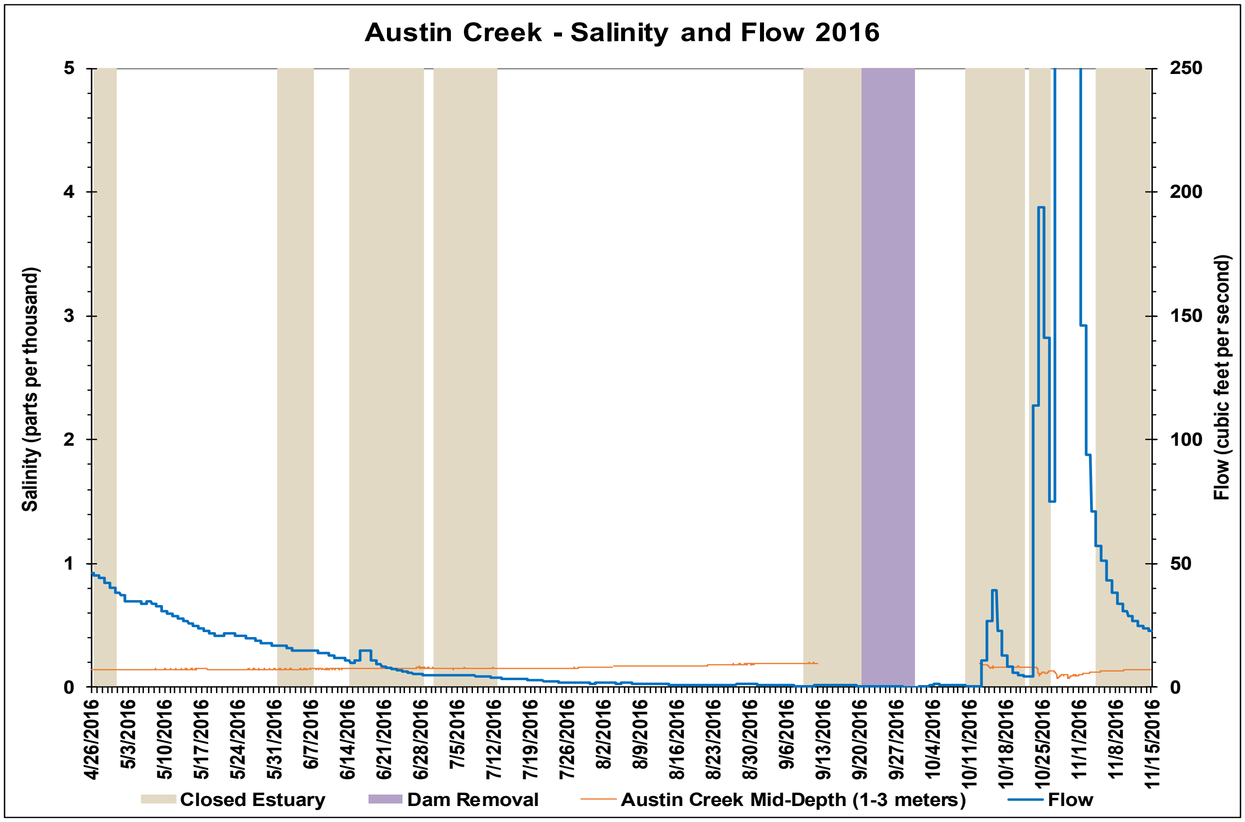

Figure 4.1.7. 2016 Austin Creek Salinity and Flow Graph

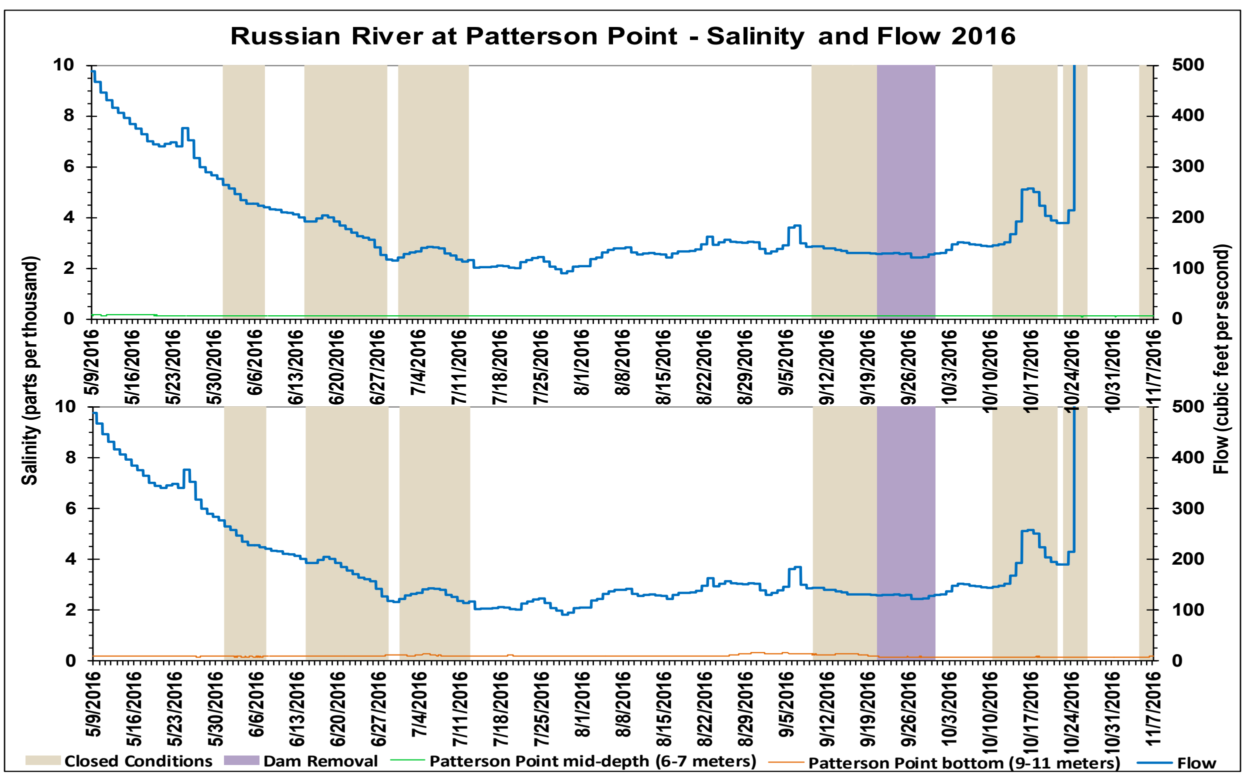

Figure 4.1.8. 2016 Russian River at Patterson Point Salinity and Flow Graph

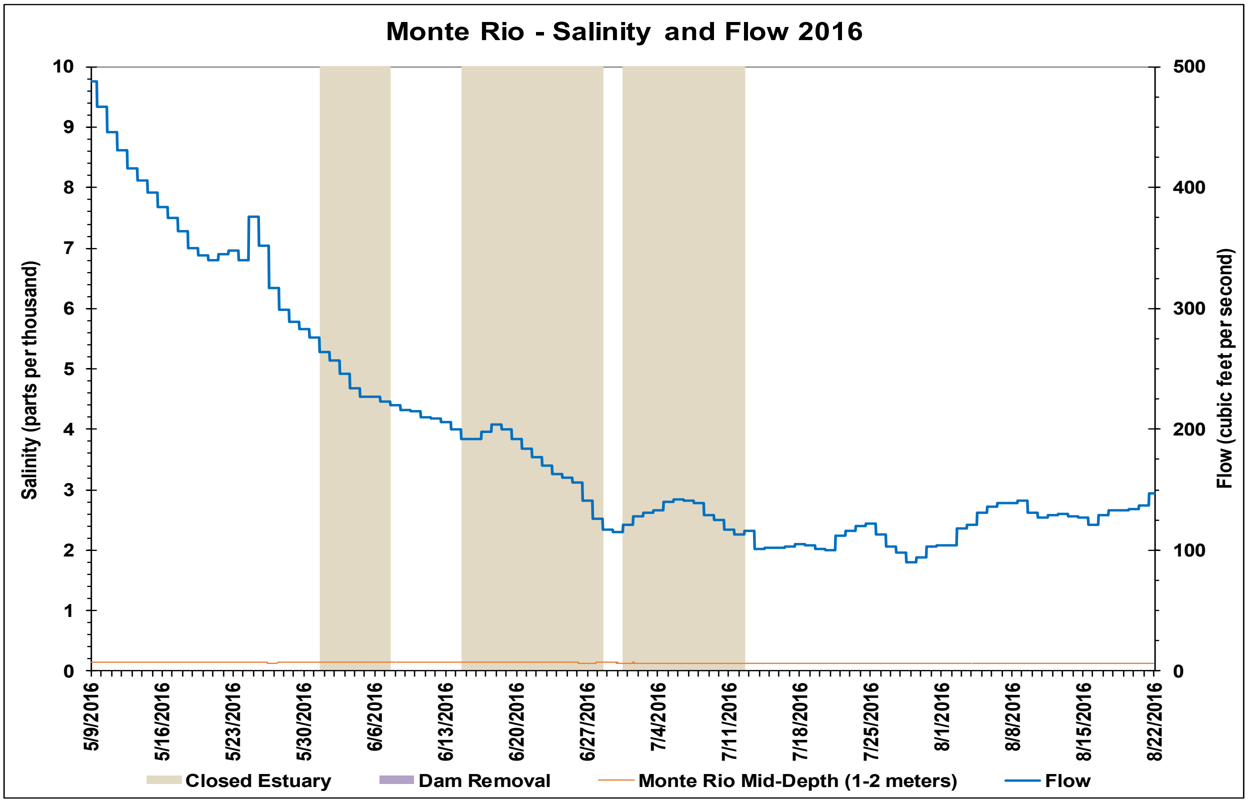

Figure 4.1.9. 2016 Russian River at Monte Rio Salinity and Flow Graph

Temperature

During open estuary conditions, mainstem water temperatures were reflective of the halocline, with lower mean and maximum temperatures typically being observed in the saline layer at the bottom and mid-depth sondes compared to temperatures recorded in the freshwater layer at the mid-depth and surface sondes (Figures 4.1.10 through 4.1.13). The differences in temperatures between the underlying saline layer and the overlying freshwater layer can be attributed in part to the source of saline and fresh water. During open estuary conditions, the Pacific Ocean, where temperatures are typically around 10 degrees Celsius (°C), is the source of saltwater in the Estuary. Whereas, the mainstem Russian River, with water temperatures reaching as high as 27 °C in the interior valleys, is the primary source of freshwater in the Estuary.

During closed Estuary conditions, increasing temperatures associated with fresh/saltwater stratification were observed to occur (Figure 4.1.10). Density and temperature gradients between freshwater and saltwater play a role in stratification and serve to prevent/minimize mixing of the freshwater and saline layers. When the estuary is closed, or the river mouth is perched and the supply of cool tidal inflow is reduced, solar radiation heats the underlying saline layer. Additionally, the overlying freshwater surface layer restricts the release of this heat, which can result in higher water temperatures in the underlying saline layer than in the overlying freshwater layer (Figure 4.1.10). Stratification based heating has also been observed to result in

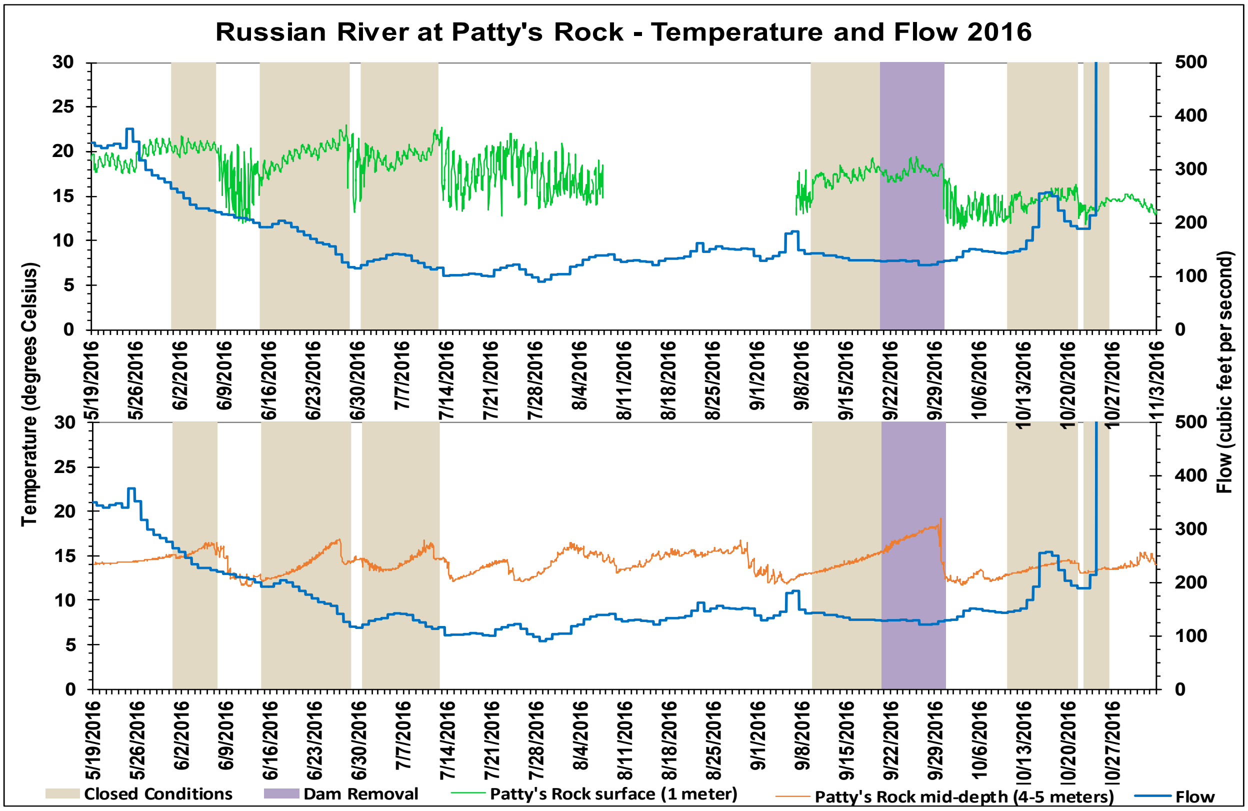

Figure 4.1.10. 2016 Russian River at Patty's Rock Temperature and Flow Graph

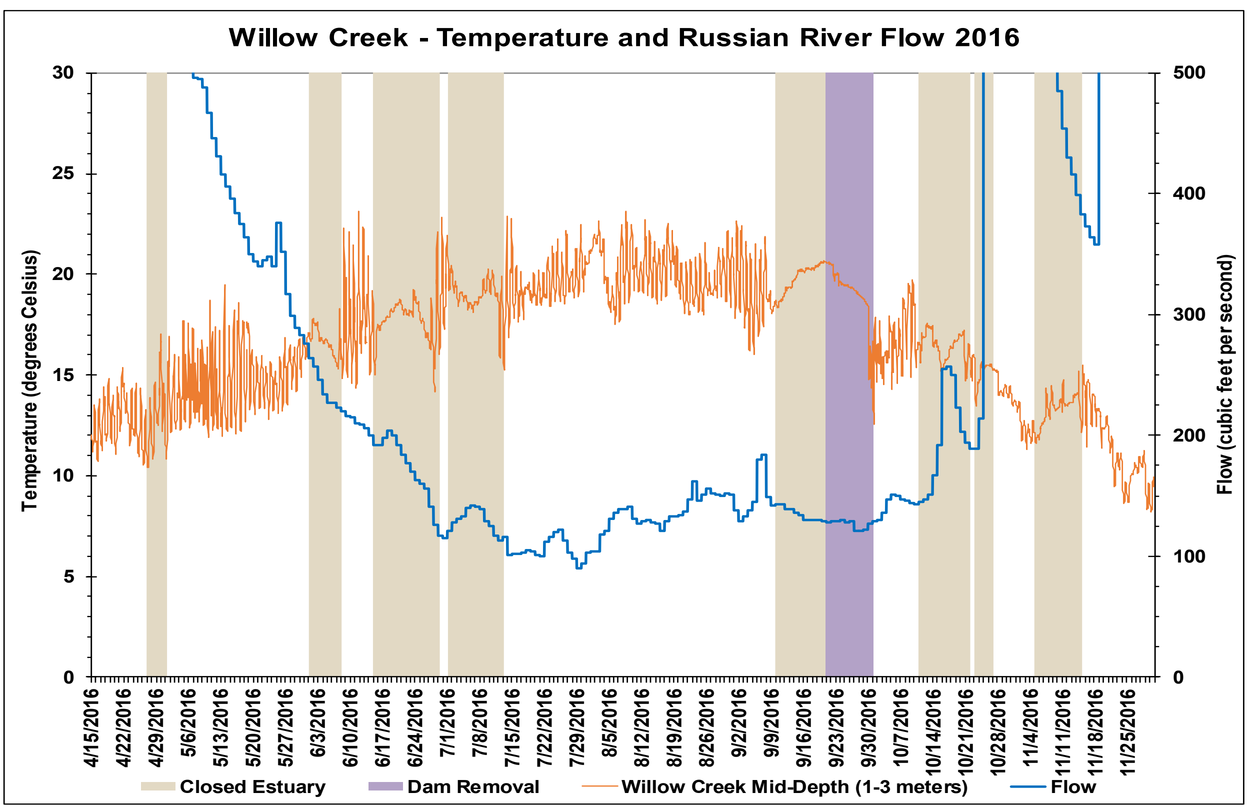

Figure 4.1.11. 2016 Willow Creek Temperature with Russian River Flow Graph

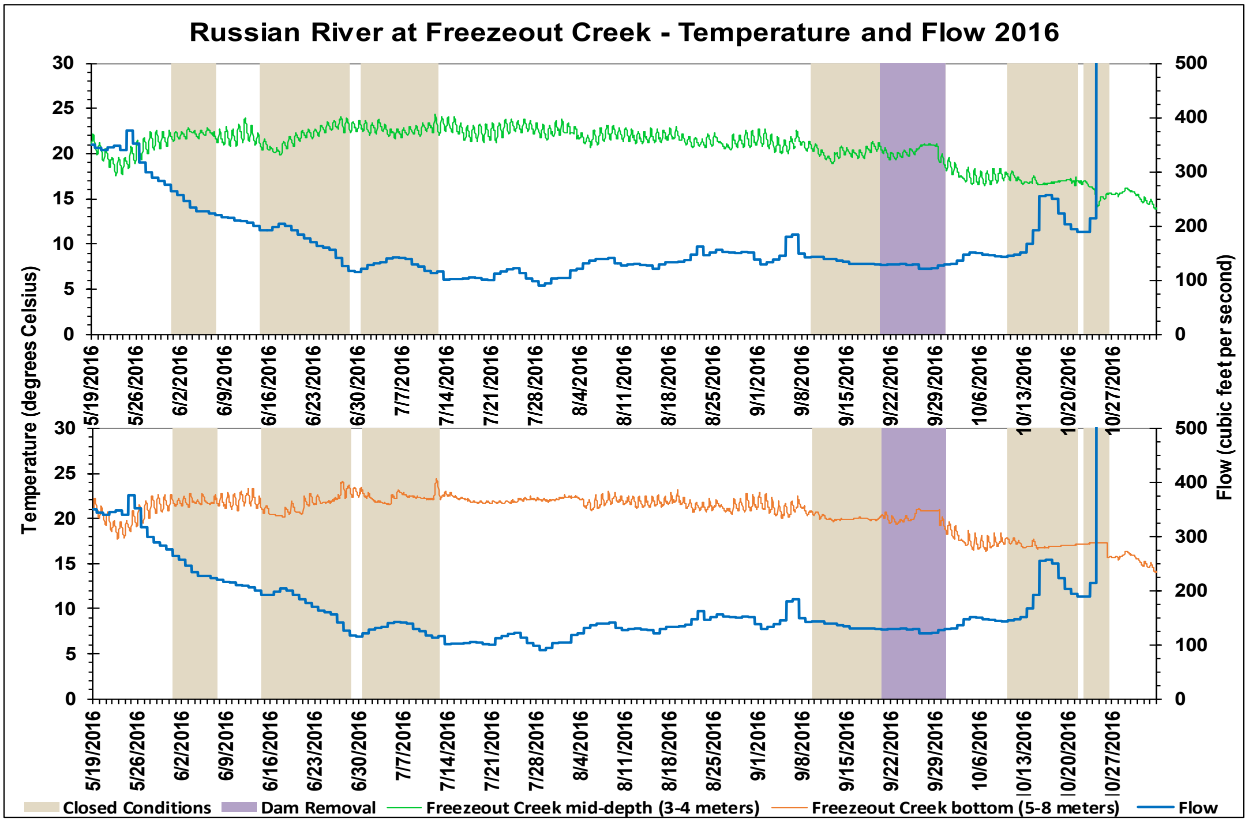

Figure 4.1.12. 2016 Russian River at Freezeout Creek Temperature and Flow Graph

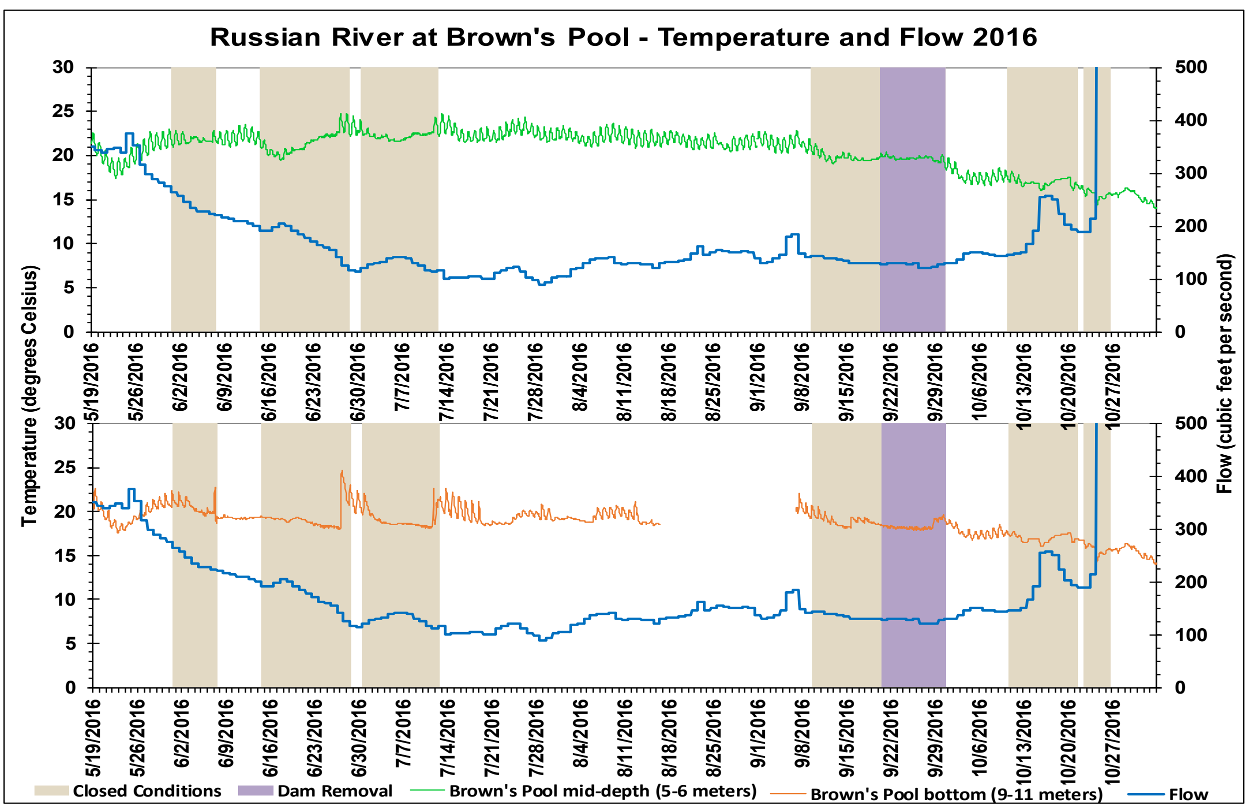

Figure 4.1.13. 2016 Russian River at Brown's Pool Temperature and Flow Graph

higher temperatures in the mid-depth saline layer compared to the bottom layer in deep pools, forming a three-layered system. This stratification-based heating can also contribute to higher seasonal mean temperatures in the saline layer than would be expected to occur under open conditions.

Lower and Middle Reach Temperature

The Patty's Rock surface sonde was located at the freshwater/saltwater interface and was observed to have a maximum temperature of 23.0 °C (Table 4.1.1). Whereas, the mid-depth sonde was located primarily in saltwater and had a maximum temperature of 19.2 °C. The Patty's Rock surface sonde had a mean temperature of 17.3 °C and a minimum temperature of 11.2 °C. The mid-depth sonde had a mean temperature of 14.2 °C and a minimum temperature of 11.5 °C.

The Willow Creek station had a maximum temperature of 23.1 °C, which occurred on 11 June in brackish water and open conditions (Figures 4.1.11 and 4.1.4). The mean temperature was 16.7 °C, and the minimum temperature was 8.2 °C. Elevated salinity was observed in early May with mainstem flows still above 400 cfs (Figure 4.1.4). However, the station returned to freshwater conditions within a week and remained that way until after the first closure of the monitoring season occurred in early June. After the barrier beach reopened, saline water migrated to the station, and it remained brackish during open and closed conditions through the rest of the monitoring season (Figure 4.1.4). Temperatures were observed to fluctuate with the movement of saline water into and out of the station, resulting in both heating and cooling during open and closed Estuary conditions (Figure 4.1.11). This was most apparent during several late season barrier beach closure events when warm brackish water was observed to signficantly decrease in temperature after freshwater or a fresh source of tidally migrating salt water migrated to the station during and between barrier beach closures (Figure 4.1.11).

Upper Reach Temperature

Overall estuarine temperatures in both the saline layer and freshwater layer were typically hottest at the upper reach stations, as observed at Freezeout Creek and Brown's Pool, and became progressively cooler as the water flowed downstream, closer to the cooling effects of the coast and ocean.

The bottom sonde at the Freezeout Creek station had a maximum temperature of 24.3 °C, a mean temperature of 20.5 °C, and a minimum temperature of 14.0 °C (Table 4.1.1). The mid-depth sonde had a maximum temperature of 24.3 °C, a mean temperature of 20.6 °C, and a minimum temperature of 13.9 °C. Minimum temperatures at the bottom and mid-depth sondes occured in freshwater during open conditions in November (Figure 4.1.12). The maximum temperatures were observed to occur at the bottom and mid-depth sondes in freshwater conditions during closed estuary conditions in July (Figures 4.1.5 and 4.1.12).

The bottom sonde at the Brown's Pool station had a maximum temperature of 24.7 °C, a mean temperature of 18.7 °C, and a minimum temperature of 14.0 °C (Table 4.1.1). The mid-depth sonde had a maximum temperature of 24.8 °C, a mean temperature of 20.5 °C, and a minimum temperature of 14.0 °C. Minimum temperatures at the Brown's Pool station were observed during the open conditions in early November after freshwater displaced the brackish water at the station during elevated storm flows (Figures 4.1.6 and 4.1.13). Early in the season, temperatures were observed to be lower at the bottom sonde compared to the mid-depth sonde when brackish water was present at the bottom sonde during open and closed conditions. Warmer freshwater from the MBA would periodically replace the cooler brackish water that was present at the bottom of the pool, resulting in higher temperatures, including the maximum temperature observed on 27 June (Figure 4.1.13). By contrast, temperatures were observed to increase during the closure in October as warm brackish water migrated to the station and displaced the cooler freshwater (Figures 4.1.6 and 4.1.13). Temperatures were then observed to decrease between the subsequent closures as the brackish water was displaced by cooler freshwater.

Maximum Backwater Area Temperature

Austin Creek had a maximum temperature of 20.5 °C, a mean temperature of 16.5 °C, and a minimum temperature of 12.5 °C (Table 4.1.1). A gradual increase in temperature through the summer months of the estuary management period coincided with increases in air temperatures (Figure 4.1.14). Closed estuary conditions did not appear to have a significant effect on the temperatures at the Austin Creek station,with slight increases and decreases in water temperature typically coinciding with increases and decreases in air temperatures (Figure 4.1.14).

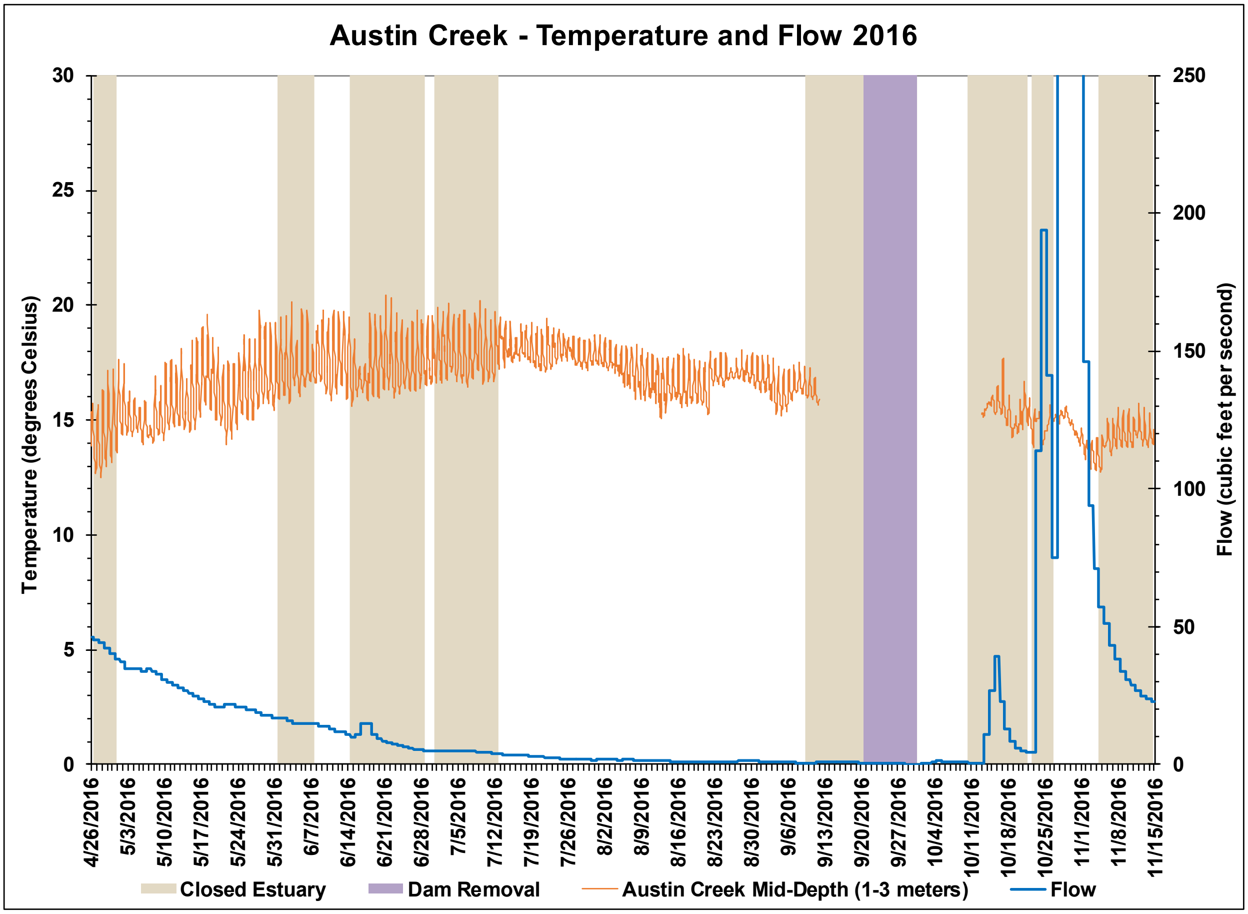

Figure 4.1.14. 2016 Austin Creek Temperature and Flow Graph

The Patterson Point bottom sonde had a maximum temperature of 21.5 °C, a mean temperature of 17.1 °C, and a minimum temperature of 12.1 °C (Table 4.1.1). The Patterson Point mid-depth sonde had a maximum temperature of 25.5 °C, a mean temperature of 20.3 °C, and a minimum temperature of 14.3 °C. Under open and closed conditions, daily temperatures were often lower at Patterson Point than at Brown's Pool and Monte Rio, which suggests that thermal stratification may be occurring at depth (Figure 4.1.15). It is also possible that a groundwater source could be contributing colder water at depth, or it could a combination of both effects occurring in tandem. Daily temperature fluctuations were significantly more stable when compared to Monte Rio (Figure 4.1.16) or Austin Creek (Figure 4.1.14), further suggesting some form of thermal stratification or regulation occurring. Temperatures continued to decline with atmospheric temperatures through the end of the season and did not appear to be affected by the extended closures (Figure 4.1.15).

The Monte Rio station had a maximum temperature of 26.3 °C, a mean temperature of 22.6 °C, and a minimum temperature of 16.7 °C during the abbreviated monitoring period (Table 4.1.1). Closed estuary conditions were not observed to have a significant effect on temperatures, which was consistent with data from previous monitoring efforts at Monte Rio and other stations within the MBA (Figure 4.1.16). Slight increases and decreases in water temperature during closure events typically coincided with increases and decreases in air temperatures.

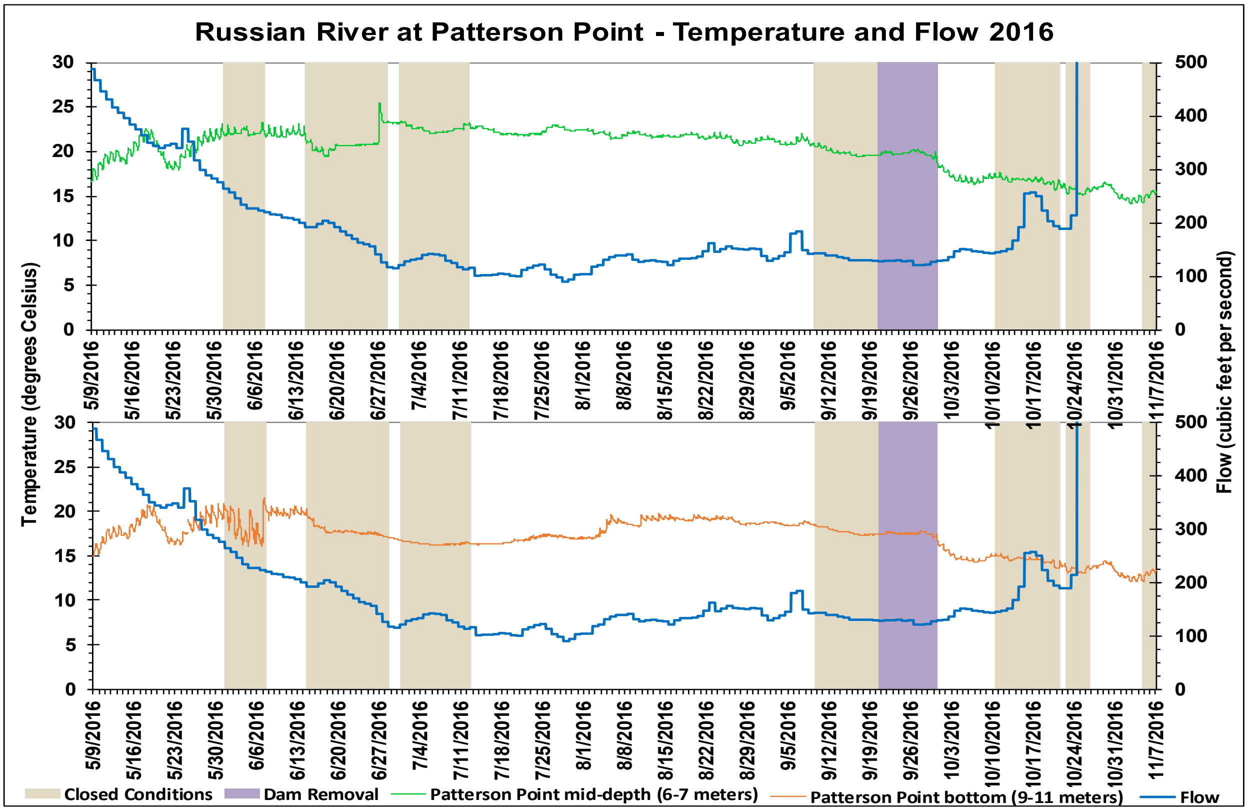

Figure 4.1.15. 2016 Russian River at Patterson Point Temperature and Flow Graph

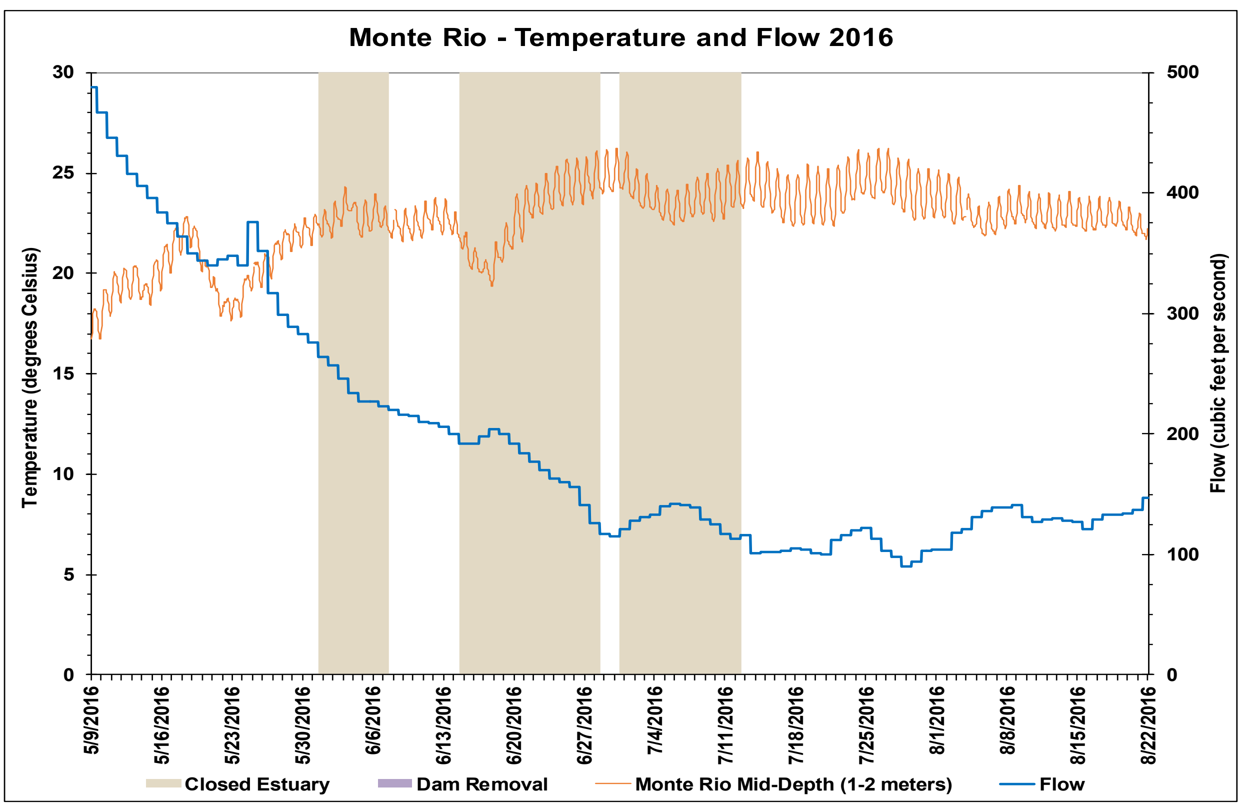

Figure 4.1.16. 2016 Russian River at Monte Rio Temperature and Flow Graph

Dissolved Oxygen

Dissolved oxygen (DO) levels in the Estuary, including the MBA, depend upon factors such as the extent of diffusion from surrounding air and water movement, including freshwater inflow. DO is affected by salinity and temperature stratification, tidal and wind mixing, abundance of aquatic plants, and presence of decomposing organic matter. DO affects fish growth rates, embryonic development, metabolic activity, and under severe conditions, stress and mortality. Cold water has a higher saturation point than warmer water; therefore cold water is capable of carrying higher levels of oxygen.

DO levels are also a function of nutrients, which can accumulate in water and promote plant and algal growth that both consume and produce DO during photosynthesis and respiration. Estuaries tend to be naturally eutrophic because land-derived nutrients are concentrated where runoff enters the marine environment in a confined channel[1]. Upwelling in coastal systems also promotes increased productivity by conveying deep, nutrient-rich waters to the surface, where the nutrients can be assimilated by algae. Excessive nutrient concentrations and plant, algal, and bacterial growth can overwhelm eutrophic systems and lead to a reduction in DO levels that can affect the overall ecological health of the Estuary.

Lower and Middle Reach Dissolved Oxygen

Mean dissolved oxygen concentrations at Patty's Rock were generally higher at the surface sonde compared to the mid-depth sonde. Whereas the Patty's Rock surface sonde had a mean DO concentration of 9.9 mg/L, the mid-depth sonde had a mean DO concentration of 7.8 mg/L (Table 4.1.1). Although the mid-depth and surface sondes were both observed to experience supersaturation conditions, the mid-depth sonde also experienced more frequent hypoxic and anoxic conditions that served to decrease the mean seasonal value. These supersaturation and hypoxic events were observed during open and closed conditions (Figure 4.1.17).

The effect of closed conditions at the surface sonde was variable as DO concentrations were observed to remain relatively unaffected, slightly decline, or increase in some instances (Figure 4.1.17). The Patty's Rock surface sonde had a minimum DO concentration of 6.9 mg/L (Table 4.1.1), which was observed in brackish water during open conditions in August (Figures 4.1.3 and 4.1.17).

DO concentrations were observed to become hypoxic and anoxic at the Patty's Rock mid-depth sonde during and immediately following river closures (Figure 4.1.17). The minimum DO concentration at the mid-depth sonde was 0.8 mg/L (Table 4.1.1).

[1] National Estuarine Eutrophication Assessment by NOAA National Centers for Coastal Ocean Science (NCCOS) and the Integration and Application Network (IAN), 1999.

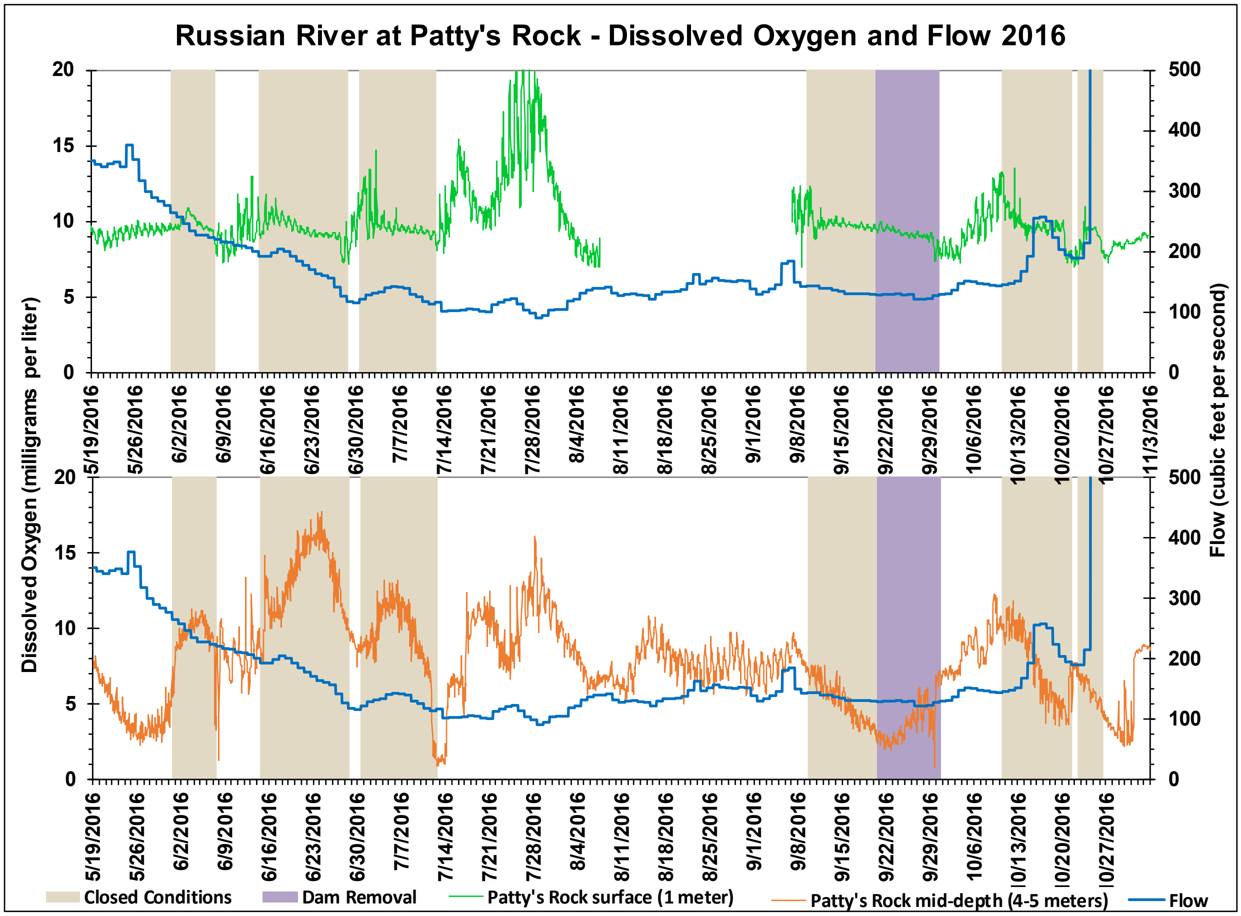

Figure 4.1.17. 2016 Russian River at Patty's Rock Dissolved Oxygen and Flow Graph

The Patty's Rock surface sonde, and mid-depth sonde to a lesser degree, experienced hourly fluctuating supersaturation events. Supersaturation events were observed at the surface and mid-depth sondes during open and closed estuary conditions (Figure 4.1.17). At times when oxygen production exceeds the diffusion of oxygen out of the system, supersaturation may occur (Horne, 1994). DO concentrations exceeding 100% saturation in the water column are considered supersaturated conditions. Because the ability of water to hold oxygen changes with temperature, there are a range of concentration values that correspond to 100% saturation. For instance, at sea level, 100% saturation is equivalent to approximately 11 mg/L at 10 °C, but only 8.2 mg/L at 24 °C. Consequently, these two temperature values roughly represent the range of temperatures typically observed in the Estuary.

The Patty's Rock surface sonde had a maximum DO concentration of 21.6 mg/L, which corresponded to 239% saturation (Table 4.1.1). The maximum DO concentration at the mid-depth sonde was 17.7 mg/L, which corresponded to 212% saturation (Table 4.1.1).

Dissolved oxygen concentrations in Willow Creek were observed to fluctuate in response to a variety of events including tidal water movement, saline intrusion, and open or closed Estuary conditions. Hypoxic events were observed to occur frequently in the presence of brackish water during open conditions from mid-July through early September and were frequently preceded or followed by supersaturation conditions as the day progressed through it's diurnal cycle (Figure

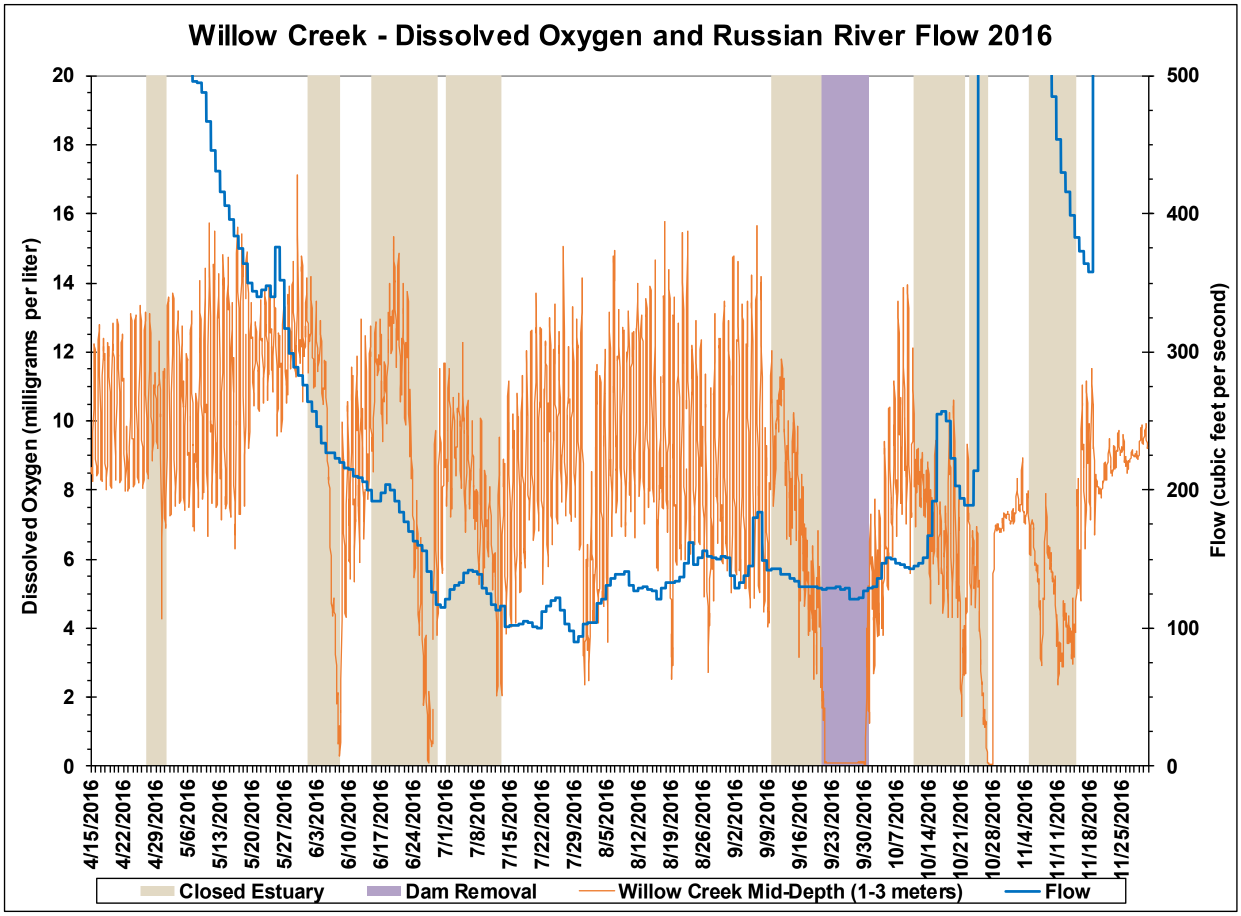

Figure 4.1.18. 2016 Willow Creek Dissolved Oxygen and Russian River Flow Graph

4.1.18). Whereas, dissolved oxygen concentrations were observed to steadily decline over a period of days during barrier beach closures in both brackish and freshwater conditions. However, dissolved oxygen concentrations were observed to recover between and after closures as oxygenated saline water or freshwater migrated back into the station (Figure 4.1.18).

The Willow Creek sonde had a minimum DO concentration of 0.1 mg/L, a mean DO concentration of 8.3 mg/L, and a maximum DO concentration of 17.1 mg/L (190%) (Table 4.1.1).

Upper Reach Dissolved Oxygen

Dissolved oxygen concentrations in the upper reach were influenced by the presence or absence of salinity, with lower minimum and mean DO concentrations observed in brackish water and higher minimum and mean concentrations observed in freshwater, especially during closed conditions. In 2016, the Freezeout Creek station was a predominantly freshwater habitat that was subject to elevated salinity levels as the salt wedge migrated up the Estuary during both open and closed conditions (Figure 4.1.5). The elevated salinity levels were predominantly observed at the bottom sonde, though elevated salinity was also seen at the mid-depth sonde during open and closed conditions. Similar to Freezeout Creek, the Brown's Pool station was predominantly freshwater habitat that was subject to elevated salinity levels as the salt wedge migrated up the Estuary during both open and closed conditions (Figure 4.1.6). The elevated salinity levels were predominantly observed at the bottom sonde, though elevated salinity was also seen at the mid-depth sonde during closed conditions in October. Hypoxic and anoxic conditions at both of these sites were observed to occur in brackish and freshwater conditions.

DO concentrations in the upper reach saline layer were also observed to be lower during open and closed conditions than DO concentrations observed in the saline layer in the lower and middle reaches. This effect was more pronounced at the bottom sondes with prolonged periods of hypoxia and anoxia observed to occur in the presence of salinity. This occurs as the saline layer becomes trapped at the bottom of deep holes where there is less circulation, especially further up in the estuary where the influence of the tidal cycle is reduced.

The Freezeout Creek bottom sonde had a minimum concentration of 0.0 mg/L, a mean DO concentrations of 4.2 mg/L, and a maximum concentration of 11.4 mg/L (132%) (Table 4.1.1). The mid-depth sonde at Freezeout Creek had a minimum concentration of 2.1 mg/L, a mean DO concentration of 8.4 mg/L, and a maximum concentration of 11.5 mg/L (130%) (Table 4.1.1).

DO concentrations at the Freezeout Creek bottom sonde fluctuated significantly and became hypoxic and anoxic during open and closed Estuary conditions when saline water was present (Figure 4.1.19).

The Freezeout Creek bottom sonde was in predominantly freshwater habitat during open and closed conditions, however there were several episodes of saline water migrating to the site in open and closed conditions (Figure 4.1.5). These fluctuations in salinity often occurred on a daily and even hourly basis. DO typically fluctuated with changing salinity concentrations, becoming depressed in saline water and recovering in freshwater (Figure 4.1.19). However, anoxic conditions were also observed to occur at the bottom of Freezeout Creek in freshwater habitat during open conditions in August (Figures 4.1.5 and 4.1.19). DO concentrations were observed to recover after the Estuary reopened in late-October as flows increased following a storm event (Figure 4.1.19).

The Freezeout Creek mid-depth sonde was observed to remain predominantly freshwater through the monitoring season until Estuary closures in September and October (Figure 4.1.5). Brackish conditions, when present, remained below 5 ppt until the October closure when salinity concentrations increased to approximately 19 ppt. DO concentrations were observed to remain stable at the mid-depth sonde in freshwater conditions, if not slightly depressed at times with concentrations as low as 5.5 mg/L. However, these depressed concentrations were typically part of a daily fluctuation in DO concentrations that also ranged as high as 11.5 mg/L (Figure 4.1.19). Conversely, DO concentrations were observed to become anoxic in brackish water during and between the two October Estuary closures. Similar to the bottom sonde, DO concentrations were observed to recover after the Estuary reopened in late-October as flows increased following a storm event (Figure 4.1.19).

The Brown’s Pool bottom sonde had a malfunctioning DO probe and therefore no DO data was available in 2016 (Figure 4.1.20).

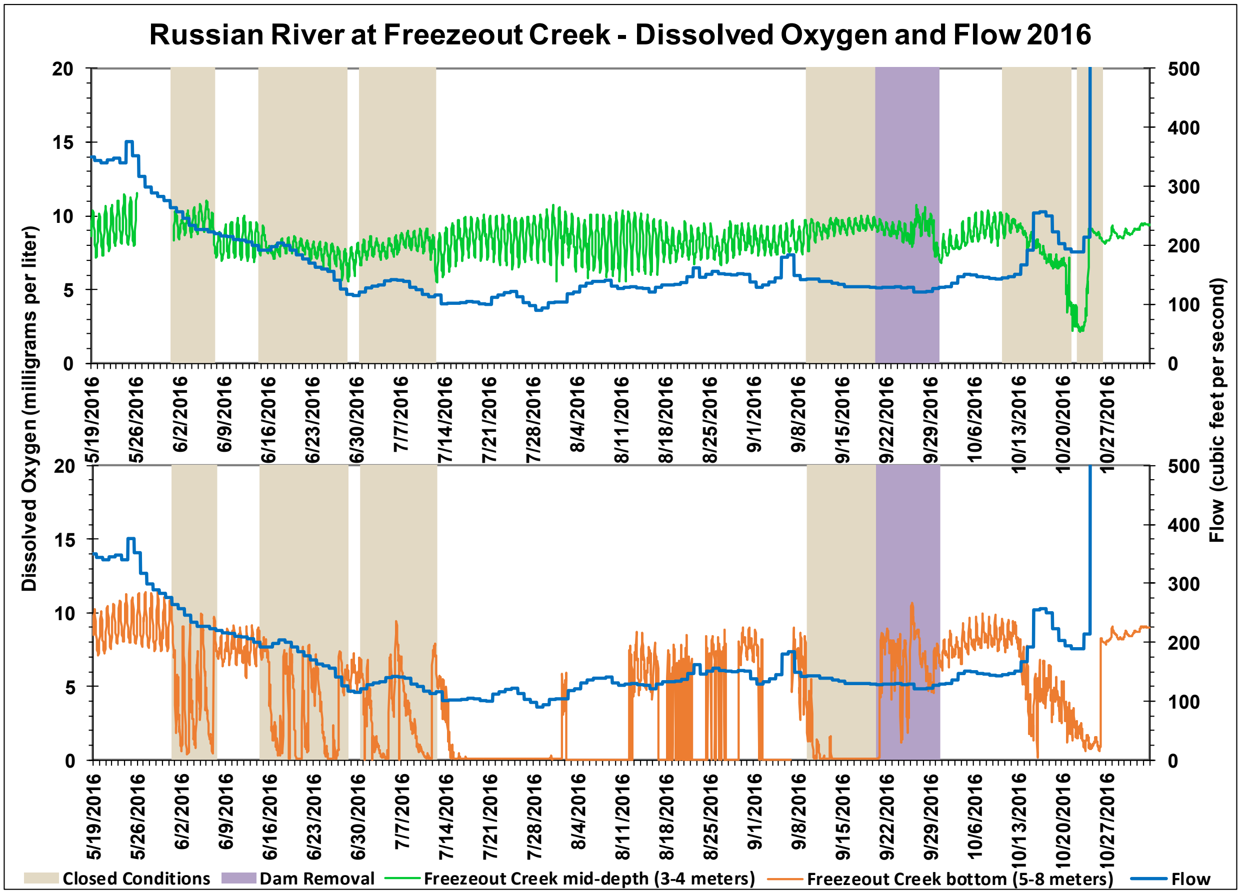

Figure 4.1.19. 2016 Russian River at Freezeout Creek Dissolved Oxygen and Flow Graph

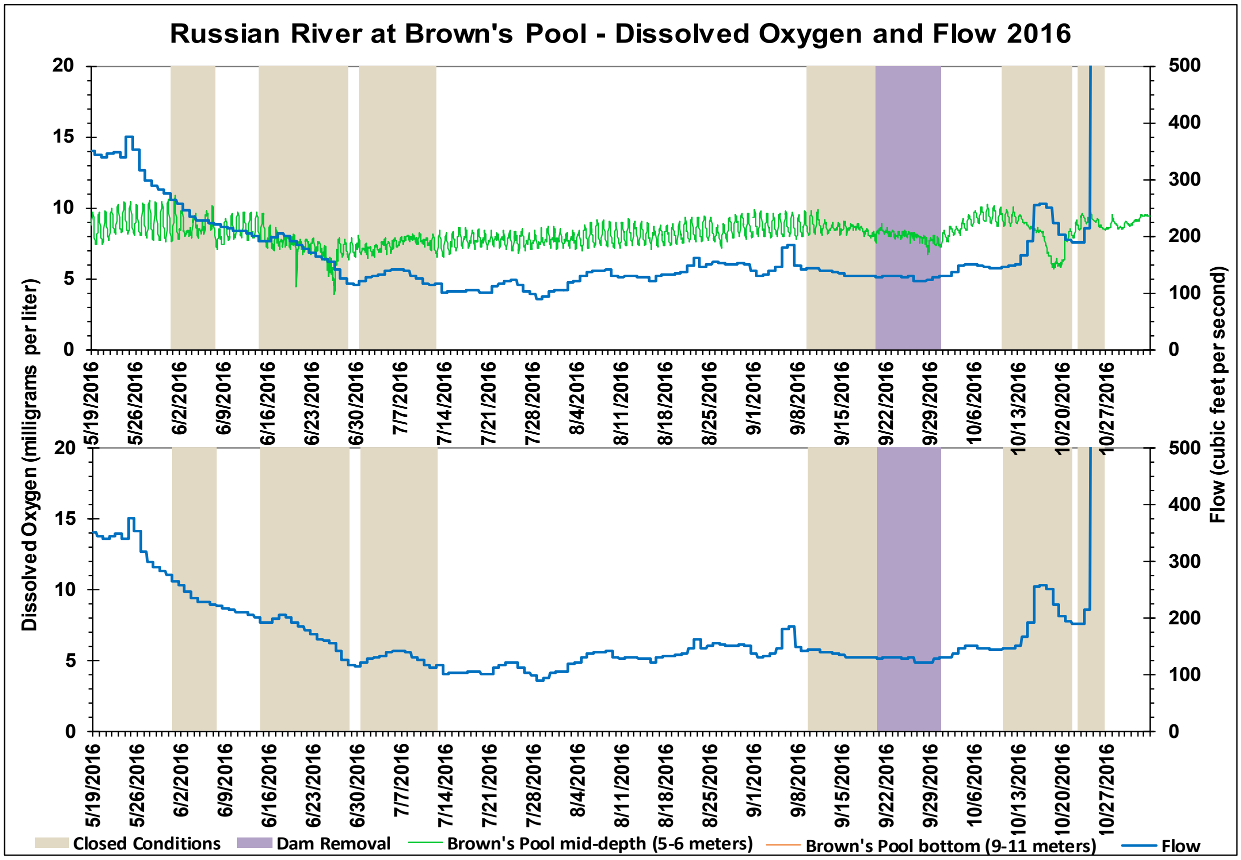

Figure 4.1.20. 2016 Russian River at Brown's Pool Dissolved Oxygen and Flow Graph

The Brown's Pool mid-depth sonde had a minimum concentration of 3.9 mg/L, a mean DO concentration of 8.2 mg/L, and a maximum concentration of 10.9 mg/L (127%) (Table 4.1.1). The mid-depth of Brown's Pool was predominantly freshwater during the entire monitoring season in open and closed conditions (Figure 4.1.6). DO concentrations were observed to remain relatively stable in freshwater conditions, with depressed concentrations as low as 3.9 mg/L being observed during estuary closure in June (Figure 4.1.20). These depressed concentrations were typically part of a daily fluctuation in DO concentrations that also ranged as high as 10.9 mg/L. As saline water migrated into the station during the first October closure, DO levels declined at the mid-depth sonde to approximately 5.8 mg/L (Figure 4.1.20). However, once freshwater conditions returned to the mid-depth, oxygen levels were observed to recover.

Maximum Backwater Area Dissolved Oxygen

The Austin Creek station had a malfunctioning DO probe that would calibrate, but stopped recording data. The sonde was replaced in October. As a result, there is only DO data available from the first day of deployment in May and in October and early November (Figure 4.1.21).

For the abbreviated monitoring period, the Austin Creek station had minimum, mean, and maximum DO concentrations of 1.2, 8.4, and 10.4 (103%) mg/L, respectively (Table 4.1.1). Flows were higher in 2016 compared to the drought year of 2015 and did not drop below 2 cfs at the upstream USGS gauging station until late July. The USGS gauging station was observed to have measurable flow all season, however flows were frequently below 1 cfs, resulting in

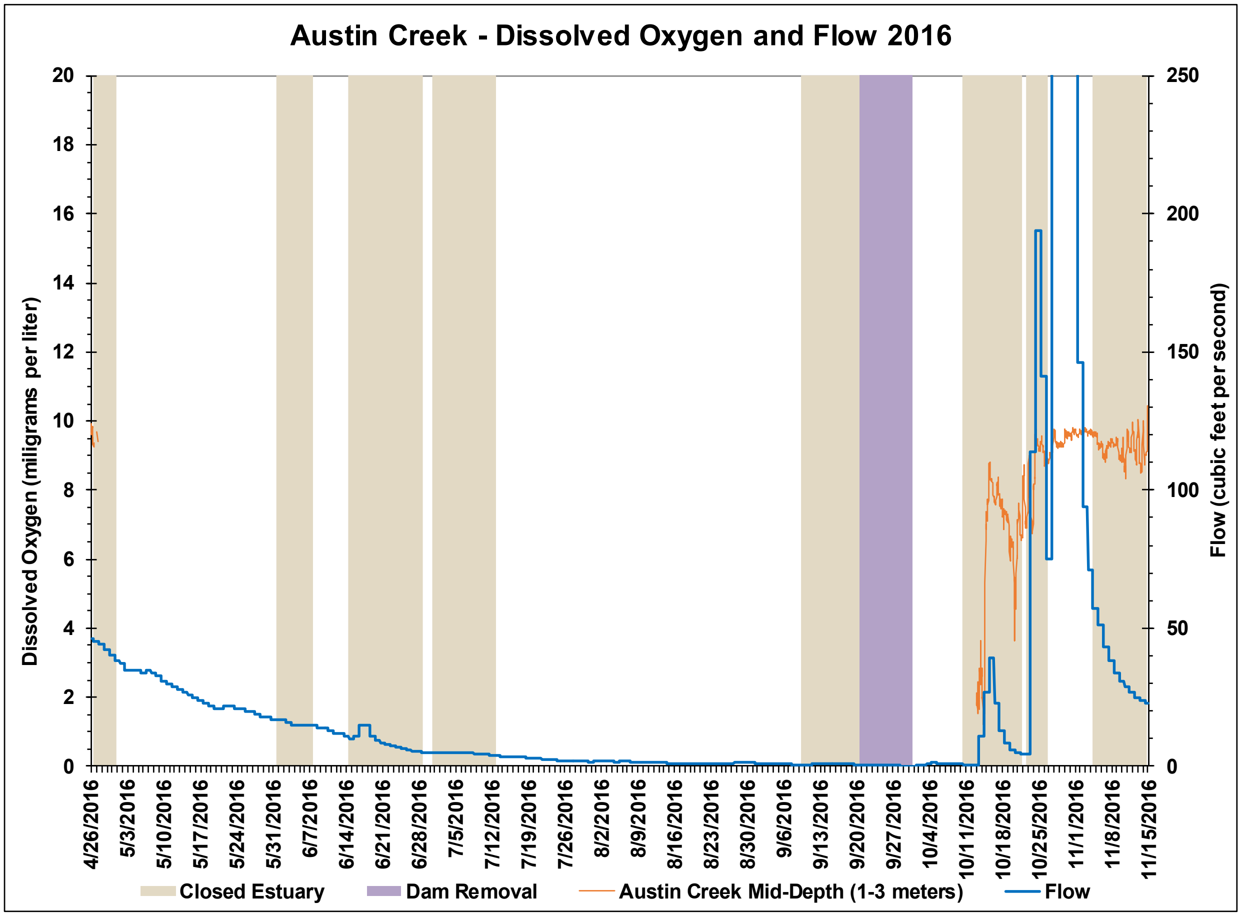

Figure 4.1.21. 2016 Austin Creek Dissolved Oxygen and Flow Graph

isolated pools. The replacement sonde was deployed on 13 October in an isolated pool where DO concentrations were hypoxic until a storm the following day raised flow rates and re-established surface flow between pools. The minimum value at Austin Creek was observed during closed conditions in October with a flow rate of 0.66 cfs measured at the upstream USGS gauging station (Figure 4.1.21). DO concentrations continued to recover to springtime levels as storm related flows continued through late October and early November.

DO response to estuary closures was variable at the Austin Creek station. Concentrations were observed to initially increase during the closure in October with rising storm related flows, but were also observed to decrease during the same closure as storm flows receded. Concentrations began to increase again during that same closure and then fully recovered during subsequent storm events.

The Patterson Point bottom sonde had a minimum DO concentration of 0.1 mg/L, a mean concentration of 3.6 mg/L, and a maximum concentration of 10.9 (114%). The bottom sonde remained predominantly hypoxic to anoxic through the monitoring season under both open and closed conditions until the beginning of October (Figure 4.1.22).

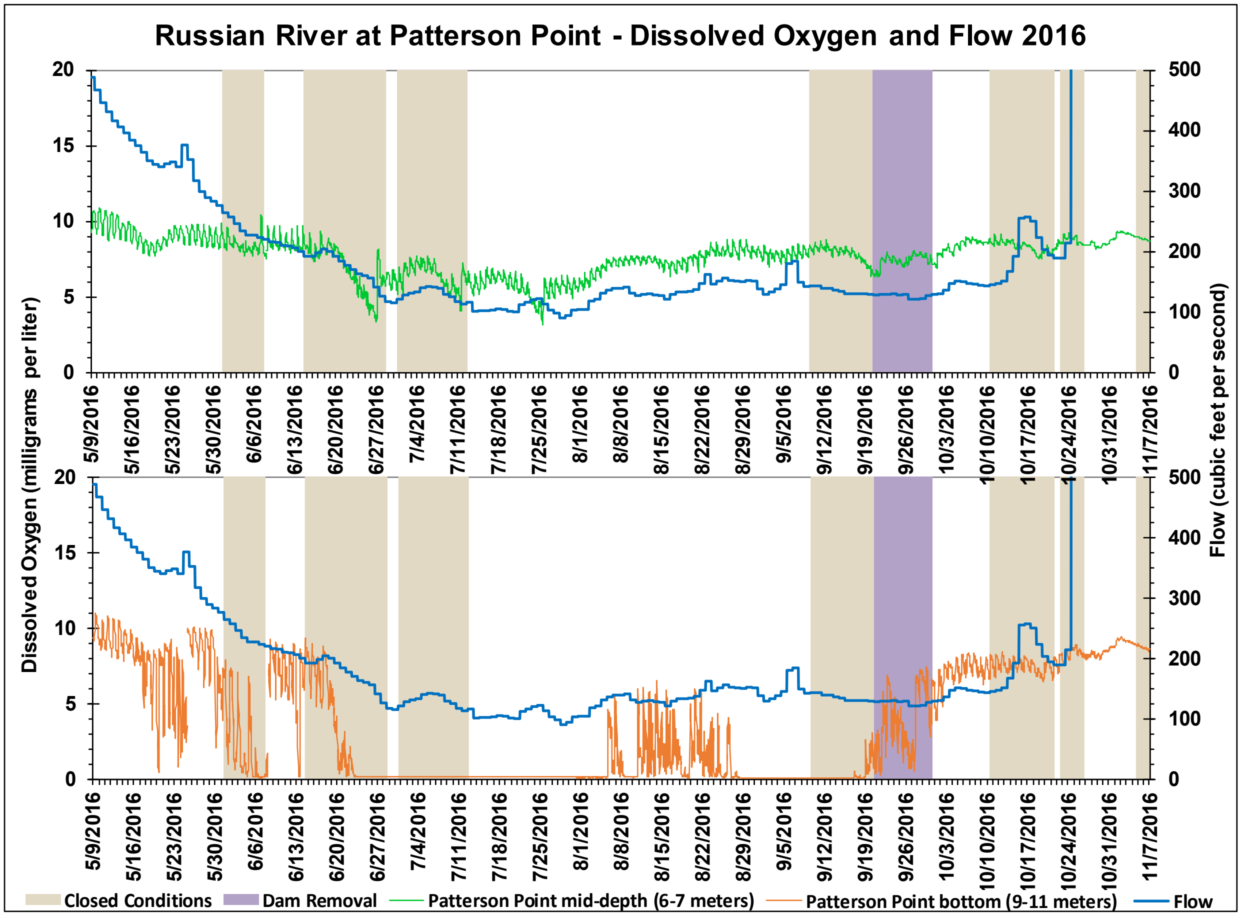

Figure 4.1.22. 2016 Russian River at Patterson Point Dissolved Oxygen and Flow Graph

Frequent fluctuations in DO concentrations were observed during higher spring flows, but the bottom sonde became anoxic during the June closure and remained predominantly anoxic during open and closed conditions before beginning to recover during Estuary closure and summer dam removal at the end of September (Figure 4.1.22). DO Concentrations recovered at the bottom sonde after the Estuary reopened and remained elevated during open and closed conditions through early November.

The Patterson Point mid-depth sonde had minimum, mean, and maximum DO concentrations of 3.2, 7.7, and 10.9 (122%) mg/L, respectively (Table 4.1.1). DO concentrations were observed to remain relatively stable in freshwater conditions, with depressed concentrations as low as 3.2 mg/L being observed during closed and open conditions in June and July (Figure 4.1.22). These depressed concentrations were typically part of a daily fluctuation in DO concentrations that also ranged as high as 10.9 mg/L. These depressed conditions also occurred when the lower half of the station was observed to experience thermal stratification (Figure 4.1.15).

The Monte Rio Station had a minimum concentration of 6.7 mg/L, a mean DO concentration of 8.5 mg/L, and a maximum concentration of 11.5 mg/L (126%) during the abbreviated monitoring period (Table 4.1.1). The minimum DO concentration occurred on 28 June during closed conditions (Figure 4.1.23). Although there were some temporally localized DO concentrations between 6 and 8 mg/L, DO concentrations did not appear to be significantly affected by summer

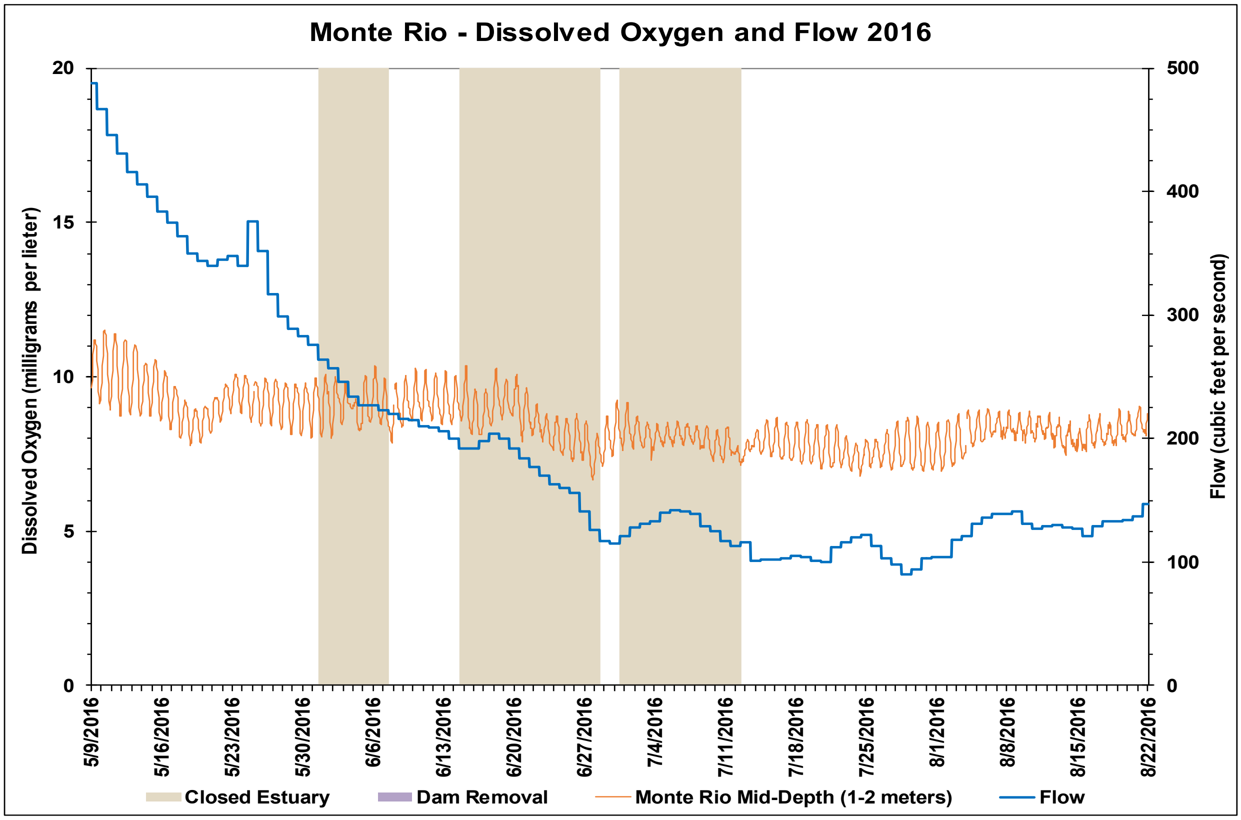

Figure 4.1.23. 2016 Russian River at Monte Rio Dissolved Oxygen and Flow Graph

flows or closed conditions and remained above 8 mg/L, on average, during both open and closed conditions (Figure 4.1.23).

Hydrogen Ion (pH)

The acidity or alkalinity of water is measured in units called pH, an exponential scale of 1 to 14 (Horne, 1994). Acidity is controlled by the hydrogen ion H+, and pH is defined as the negative log of the hydrogen ion concentration. A pH value of 7 is considered neutral, freshwater streams generally remain at a pH between 6 and 9, and ocean derived salt water is usually at a pH between 8 and 9. When the pH falls below 6 over the long term, there is a noticeable reduction in the abundance of many species, including snails, amphibians, crustacean zooplankton, and fish such as salmon and some trout species (Horne, 1994).

Lower and Middle Reach pH

The Patty's Rock surface sonde had a minimum pH value of 7.5, a mean pH value of 8.2, and a maximum pH value of 8.9 pH (Table 4.1.1). The Patty's Rock mid-depth sonde had a minimum pH value of 7.1, a mean pH value of 7.7, and a maximum pH value of 8.3 pH.

Patty's Rock pH values were observed to vary with increases and decreases of DO concentrations, with higher values generally observed during supersaturation conditions and lower values during hypoxic conditions (Figure 4.1.24). This was especially apparent when pH values briefly dropped to 7.1 at the Patty's Rock mid-depth sonde during a hypoxic event in July when the estuary reopened (Figures 4.1.17 and 4.1.24).

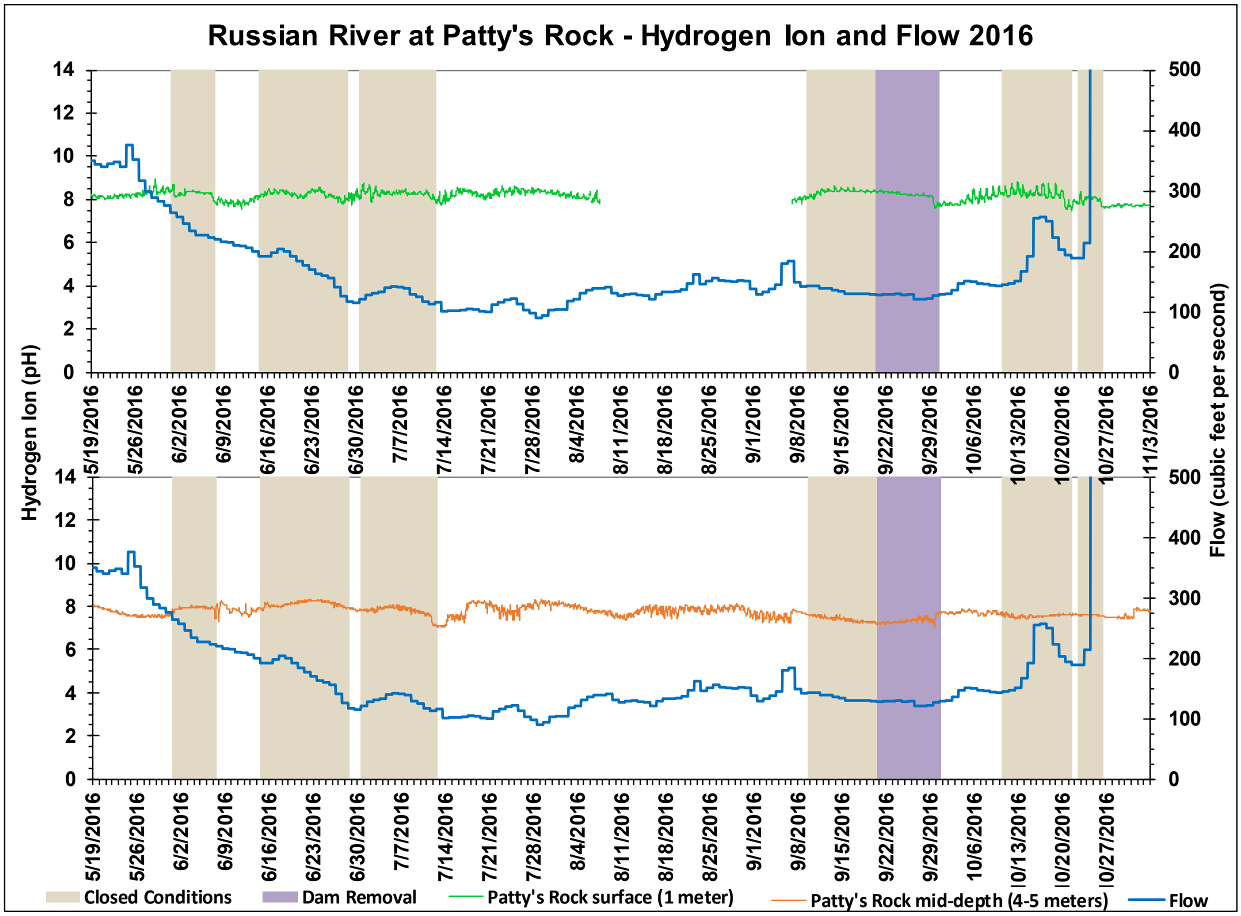

Figure 4.1.24. 2016 Russian River at Patty's Rock Hydrogen Ion and Flow Graph

The Willow Creek station had a minimum pH value of 6.6, a mean pH value of 7.7, and a maximum pH value of 9.1 (Table 4.1.1). The Willow Creek station also had pH values that were observed to vary with increases and decreases of DO concentrations, as well as with fluctuations in salinity associated with reduced freshwater flows, tidal influence, and Estuary closures (Figures 4.1.18 and 4.1.25).

Upper Reach pH

The Freezeout Creek bottom sonde recorded a minimum pH value of 6.9, a mean pH value of 7.6, and a maximum pH value of 8.5 (Table 4.1.1). The Freezeout Creek mid-depth sonde recorded a minimum pH value of 6.9, a mean pH value of 7.9, and a maximum pH value of 8.5 (Table 4.1.1). The Freezeout Creek station had pH values that were observed to vary with DO concentrations in the presence of both freshwater and brackish water (Figures 4.1.19 and 4.1.26).

The Brown's Pool bottom sonde had a minimum pH value of 5.4, a mean pH value of 6.8, and a maximum pH value of 8.2 (Table 4.1.1). The Brown's Pool mid-depth sonde had a minimum pH value of 7.3, a mean pH value of 7.7, and a maximum pH value of 8.3 (Table 4.1.1). Minimum pH values occurred at the mid-depth sonde during hypoxic conditions when the Estuary was closed (Figures 4.1.20 and 4.1.27).

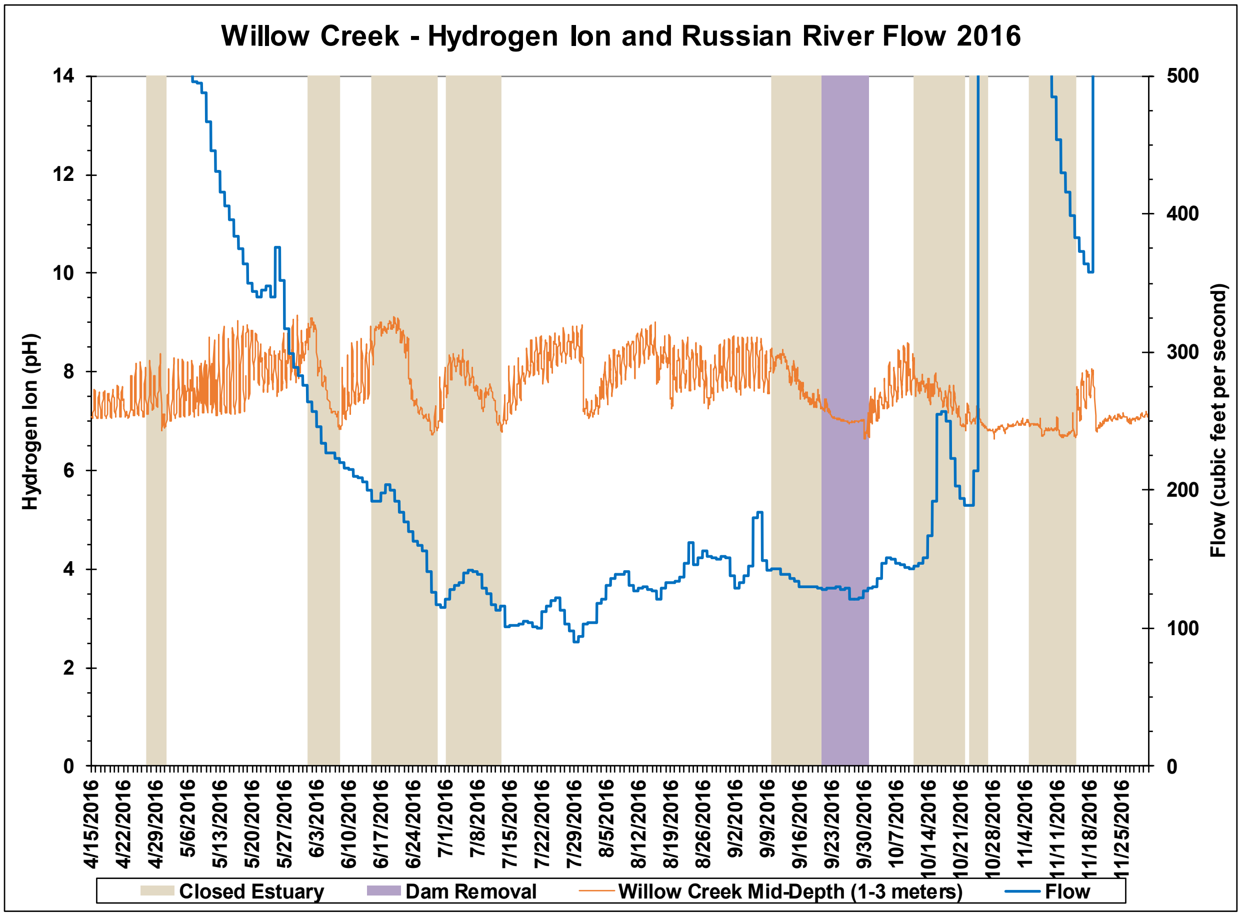

Figure 4.1.25. 2016 Willow Creek Hydrogen Ion and Russian River Flow Graph

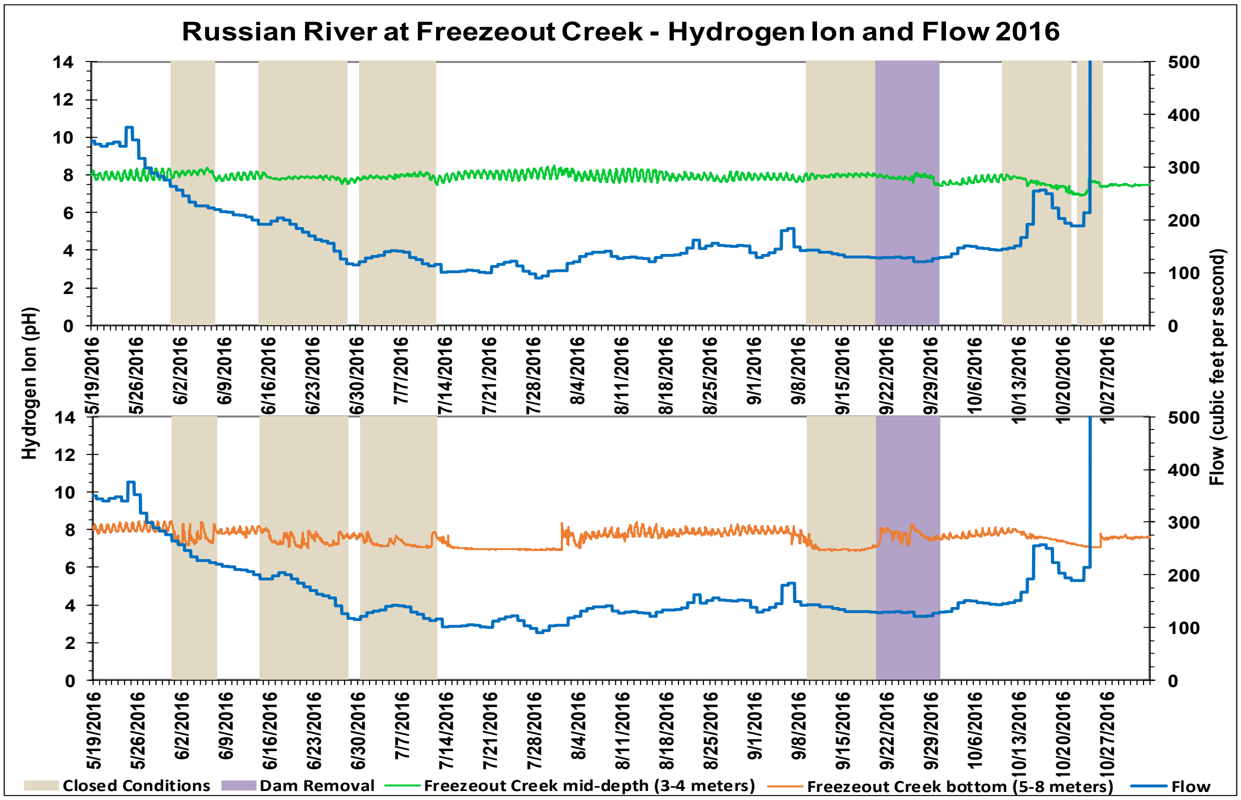

Figure 4.1.26. 2016 Russian River at Freezeout Creek Hydrogen Ion and Flow Graph

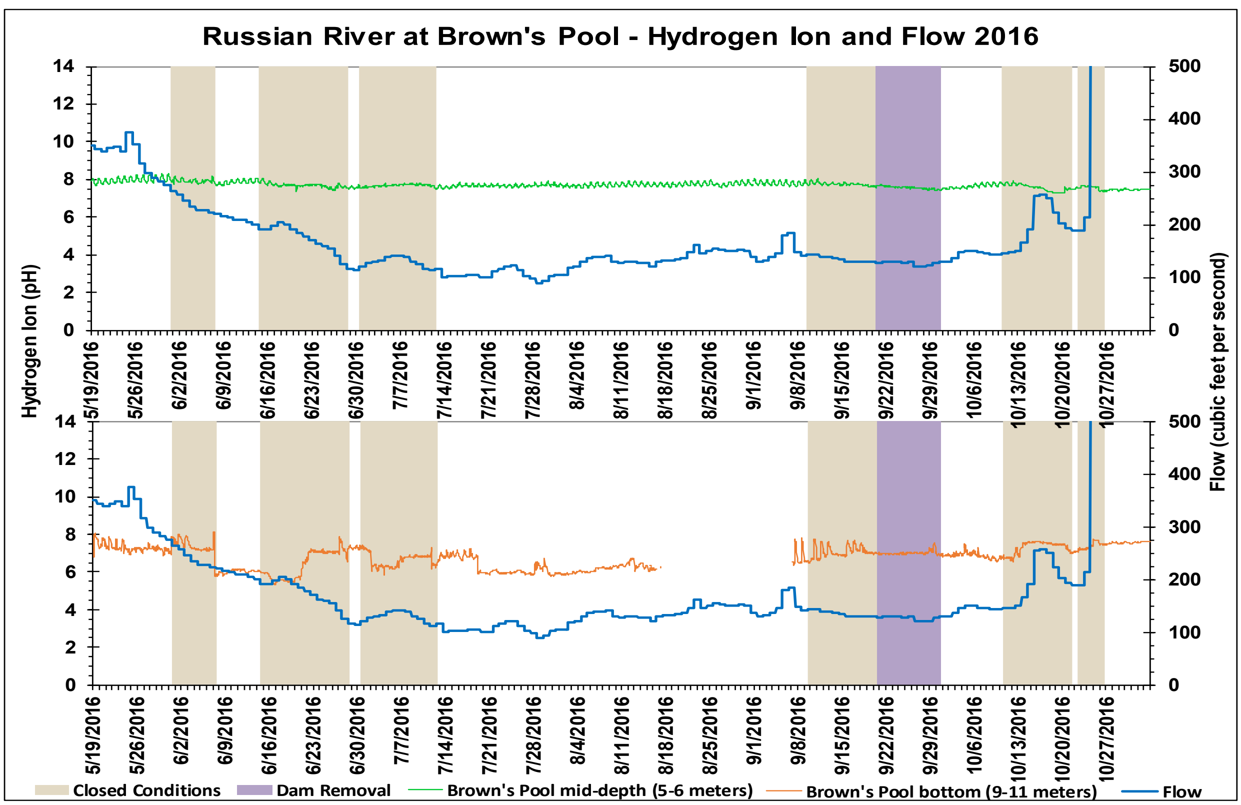

Figure 4.1.27. 2016 Russian River at Brown's Pool Hydrogen Ion and Flow Graph

Maximum Backwater Area pH

The Austin Creek sonde had a minimum pH value of 6.6, a mean pH value of 7.4, and a maximum pH value of 8.0 (Table 4.1.1). The Austin Creek sonde also had pH values that were generally observed to vary with increases and decreases of DO concentrations (Figures 4.1.21 and 4.1.28).

The Patterson Point bottom sonde had a minimum pH value of 5.4, a mean pH value of 6.9, and a maximum pH value of 8.1 (Table 4.1.1). The Patterson Point mid-depth sonde had a minimum pH value of 7.2, a mean pH value of 7.7, and a maximum pH value of 8.2 (Table 4.1.1). The Patterson Point sonde also had pH values that were generally observed to vary with increases and decreases of DO concentrations (Figures 4.1.22 and 4.1.29). Minimum concentrations were observed during hypoxic and anoxic conditions when the Estuary was closed.

The Monte Rio sonde recorded a minimum pH value of 7.5, a mean pH value of 7.8, and a maximum pH value of 8.4 (Table 4.1.1). Again, the sonde here recorded pH values that were generally observed to vary with increases and decreases of DO concentrations (Figures4. 1.23 and 4.1.30). Overall, pH concentrations did not appear to be significantly affected by summer flows or closed conditions and remained fairly stable through the monitoring period (Figure 4.1.30).

Grab Sampling

Water Agency staff conducted weekly grab sampling from May 10 to October 18 at three freshwater stations in the mainstem of the lower river including Patterson Point, Monte Rio, and Vacation Beach (Figure 4.1.1). Additional focused sampling was conducted during or after Estuary closures, as well as during summer dam removal in late September, where Agency staff would collect three samples in ten days (Tables 4.1.2 through 4.1.4). Samples collected and analyzed for nutrients, turbidity, *chlorophyll a*, and indicator bacteria are discussed below. Other sample results including organic carbon, and dissolved solids are not discussed, but are included as an appendix to the report.

Nutrients

The United States Environmental Protection Agency (USEPA) has established section 304(a) nutrient criteria across 14 major ecoregions of the United States. The Russian River was designated in Aggregate Ecoregion III (USEPA 2013a). USEPA's section 304(a) criteria are intended to provide for the protection of aquatic life and human health (USEPA 2013b). The following discussion of nutrients compares sampling results to these USEPA criteria. However, it is important to note that these criteria are established for freshwater systems, and as such, are only applicable to the freshwater portions of the Estuary. Currently, there are no numeric nutrient criteria established specifically for estuaries. However, Jenner will be included in the discussion for comparative purposes.

The USEPA desired goal for total nitrogen in Aggregate Ecoregion III is 0.38 mg/L for rivers and streams not discharging into lakes or reservoirs (USEPA, 2000). Calculating total nitrogen values requires the summation of the different components of total nitrogen; organic and

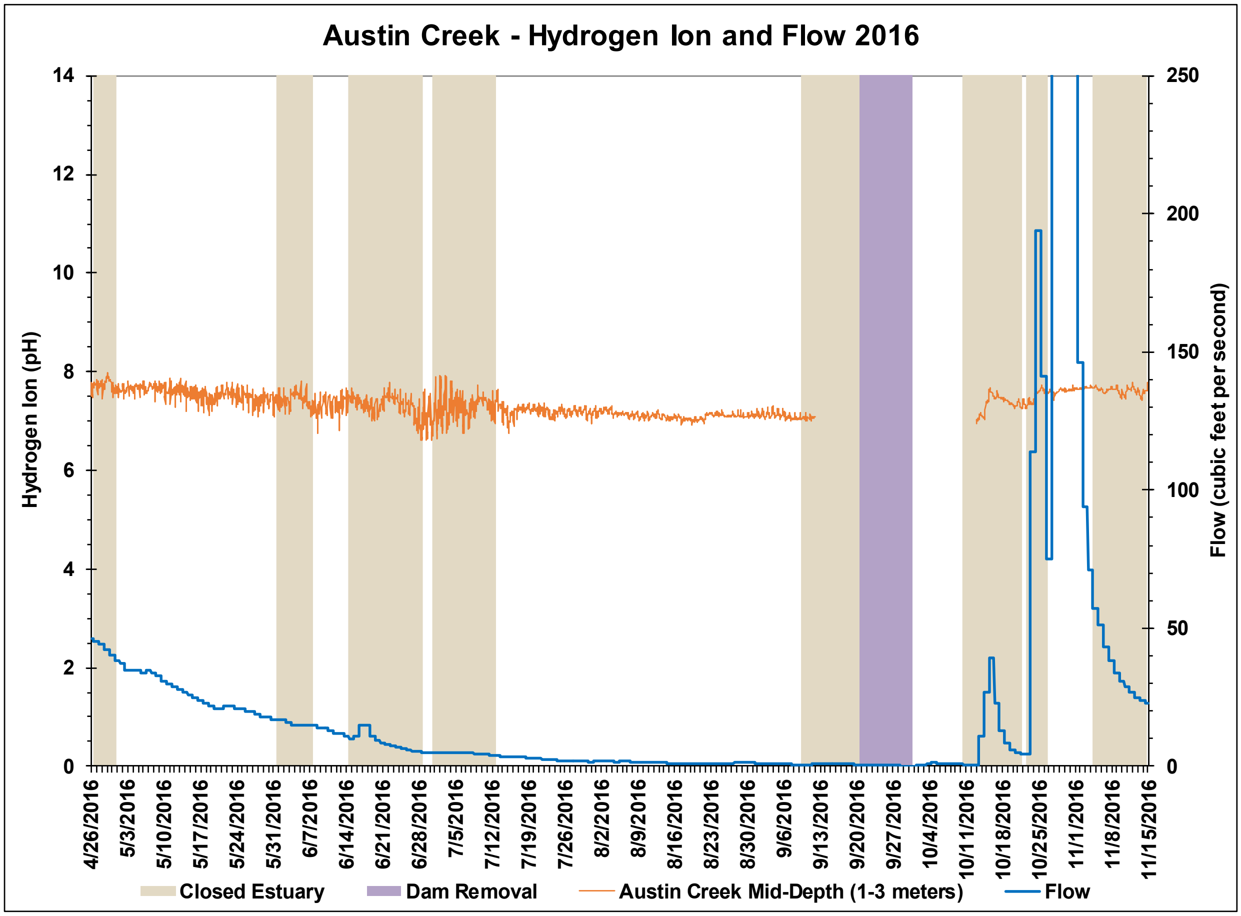

Figure 4.1.28. 2016 Austin Creek Hydrogen Ion and Flow Graph

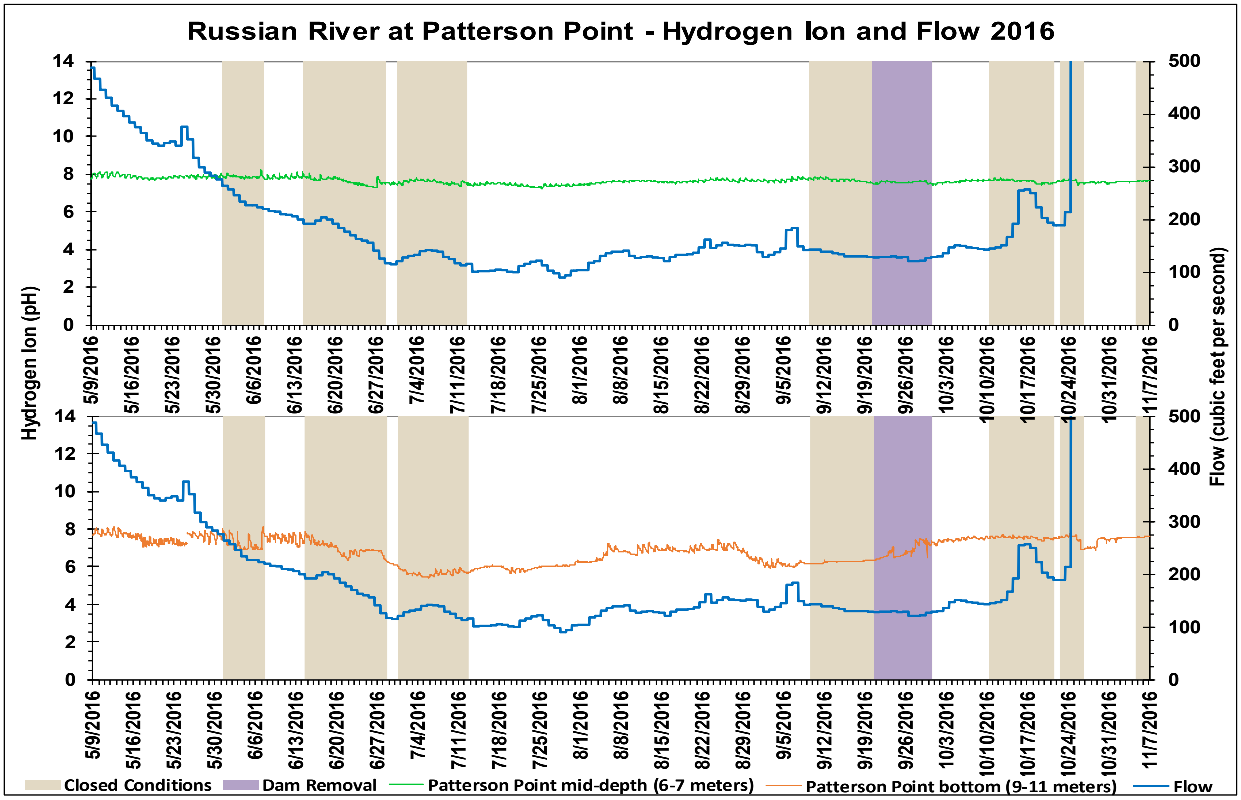

Figure 4.1.29. 2016 Russian River at Patterson Point Hydrogen Ion and Flow Graph

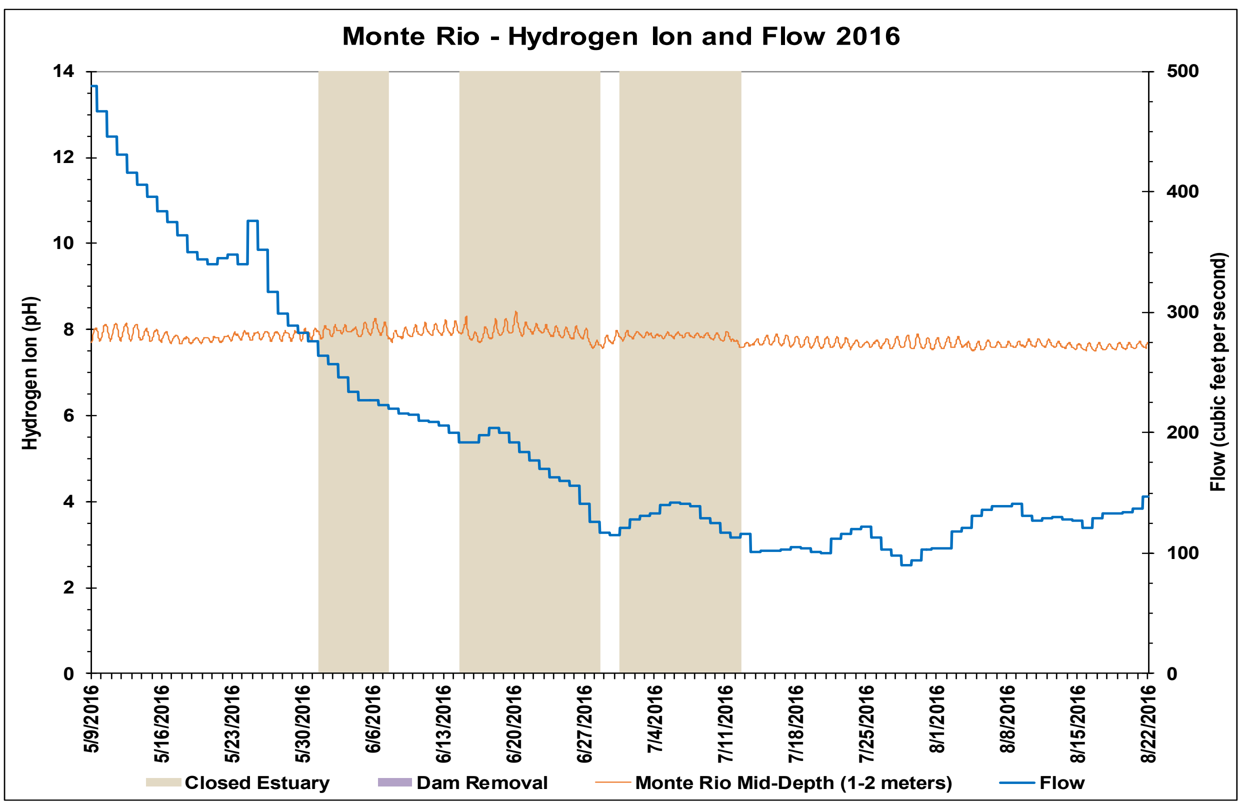

Figure 4.1.30. 2016 Russian River at Monte Rio Hydrogen Ion and Flow Graph

Table 4.1.2. 2016 Russian River at Patterson Point Station Grab Sample Results

Patterson Point*	Temperature	Total Nitrogen	Total Phosphorus	Turbidity	Chlorophyll-a	Total Coliforms (Colilert)	Total Coliforms Diluted 1:10 (Colilert)	E. coli (Colilert)	E. coli Diluted 1:10 (Colilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***		
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary	Jenner
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition	Gauge (ft)
5/10/2016	16.6	**0.50**	**0.040**	2.1	**0.0041**	686.7	908	12.1	<10	<10	468	Open	0.93
5/17/2016	20.1	**0.44**	**0.047**	1.8	0.0014	648.8	670	10	31	1.0	377	Open	1.77
5/24/2016	18.1	**0.49**	**0.031**	1.4	0.00073	547.5	455	8.4	<10	1.0	343	Open	2.57
5/31/2016	21.4	0.31	**0.036**	2.2	**0.0021**	1119.9	1178	18.9	<10	3.1	277	Open	2.91
6/2/2016	22.6	-----	-----	-----	-----	866.4	744	22.8	41	10	259	Closed	5.01
6/7/2016	21.7	0.24	**0.024**	2.1	**0.0058**	1553.1	2014	35.0	30	44.1	224	Closed	7.71
6/14/2016	21.3	0.31	**0.026**	1.4	**0.0024**	1732.9	1119	22.3	10	**63**	202	Open	1.56
6/21/2016	21.5	0.10	**0.036**	0.99	**0.0039**	>2419.6	2282	25.6	63	47.0	186	Closed	5.69
6/23/2016	22.6	0.21	**0.035**	1.7	**0.0027**	>2419.6	4611	43.2	74	28.2	170	Closed	6.45
6/28/2016	23.7	0.24	**0.043**	2.2	**0.0020**	>2419.6	3873	13.4	20	7.4	127	Open	1.43
7/5/2016	21.7	0.18	**0.036**	2.1	0.0015	>2419.6	2098	44.3	31	15.8	140	Closed	4.63
7/7/2016	22.6	0.14	**0.037**	1.6	**0.0035**	>2419.6	4352	43.2	41	21.3	141	Closed	5.31
7/12/2016	23.1	0.18	**0.038**	2.2	**0.0024**	>2419.6	3448	16.9	52	**73.3**	113	Open	3.58
7/19/2016	22.2	0.22	**0.034**	**3.0**	0.0011	1986.3	2613	1.0	<10	2.0	104	Open	2.15
7/26/2016	23.0	0.17	**0.035**	**2.4**	0.0013	2419.6	4106	6.3	10	14.5	113	Open	1.94
8/2/2016	22.7	0.24	**0.033**	**2.4**	0.0012	>2419.6	1956	29.9	41	21.6	104	Open	1.52
8/9/2016	22.1	0.10	**0.027**	2.2	0.0015	1732.9	2481	9.7	<10	10.8	141	Open	1.26
8/16/2016	21.9	0.070	**0.026**	1.2	0.0012	1413.6	1450	18.5	<10	4.1	121	Open	1.47
8/23/2016	21.7	0.10	0.021	1.8	0.0014	1299.7	1250	17.1	10	2.0	162	Open	0.97
8/30/2016	21.2	0.10	0.021	1.8	0.0016	1203.3	1236	12.0	20	3.1	152	Open	1.6
9/6/2016	20.8	0.21	ND	1.6	0.0012	1046.2	1145	16.1	20	5.2	181	Open	1.26
9/13/2016	19.8	0.10	0.021	1.0	0.00080	727.0	884	14.8	41	8.6	140	Closed	3.83
9/15/2016	20.0	0.10	**0.022**	1.5	0.00064	816.4	1374	15.8	31	17.3	136	Closed	4.59
9/20/2016	20.8	0.24	**0.024**	2.0	0.00060	1203.3	1723	34.5	52	16.0	129	Closed	6.07
9/22/2016	20.0	0.070	0.020	1.2	0.00090	1732.9	134	67.9	109	54.4	130	Closed	6.62
9/27/2016	20.3	0.14	0.025	1.4	0.0012	>2419.6	1789	66.3	41	39.9	121	Closed	7.71
9/29/2016	20.4	0.10	**0.026**	1.2	0.00050	1986.3	1396	38.9	52	18.3	122	Closed	8.18
10/4/2016	16.8	0.079	**0.030**	1.2	ND	1119.9	932	8.5	10	7.4	147	Open	0.97
10/11/2016	17.2	0.21	**0.027**	1.9	0.0012	547.1	399	25.0	20	6.3	142	Open	2.19
10/18/2016	16.6	0.15	**0.065**	0.97	0.00089	1299.7	1658	61.7	97	48.8	240	Closed	7.5

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III

Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L

Total Nitrogen: 0.38 mg/L

Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L

Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:

Beach posting is recommended when indicator organisms exceed any of the following levels:

Total coliforms: 10,000 per 100 ml

E. coli: 235 per 100 ml

Enterococcus: 61 per 100 ml

Table 4.1.3. 2016 Russian River at Monte Rio Station Grab Sample Results

Monte Rio*	Temperature	Total Nitrogen	Total Phosphorus	Turbidity	Chlorophyll-a	Total Coliforms (Collilert)	Total Coliforms Diluted 1:10 (Collilert)	E. coli (Collilert)	E. coli Diluted 1:10 (Collilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***		
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary	Jenner
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition	Gauge (ft)
5/10/2016	15.6	**0.56**	**0.040**	1.7	**0.0063**	908.4	1376	16.0	<10	<10	468	Open	0.93
5/17/2016	19.8	**0.44**	**0.037**	**2.4**	**0.0033**	866.4	857	4.1	20	1.0	377	Open	1.77
5/24/2016	17.9	0.34	**0.040**	1.8	0.0015	488.4	529	6.3	10	3.1	343	Open	2.57
5/31/2016	21.0	0.34	**0.036**	1.4	**0.0022**	770.1	1187	14.6	30	5.2	277	Open	2.91
6/2/2016	22.4	-----	-----	-----	-----	1203.3	822	48.0	52	**228**	259	Closed	5.01
6/7/2016	21.9	0.14	**0.026**	1.1	**0.0035**	>2419.6	1314	204.6	109	**387.3**	224	Closed	7.71
6/14/2016	21.4	0.15	**0.027**	1.5	0.0017	1119.9	1178	13.4	20	**63**	202	Open	1.56
6/21/2016	21.5	0.21	**0.034**	1.1	**0.0060**	>2419.6	2909	69.7	51	**62.4**	186	Closed	5.69
6/23/2016	22.9	0.22	**0.035**	1.9	**0.0035**	>2419.6	3784	**261.3**	**241**	**179.2**	170	Closed	6.45
6/28/2016	24.0	0.070	**0.049**	1.9	0.0017	>2419.6	4106	16.9	<10	5.2	127	Open	1.43
7/5/2016	21.9	0.10	**0.039**	2.2	**0.0040**	>2419.6	4106	22.4	10	12.8	140	Closed	4.63
7/7/2016	23.3	0.15	**0.032**	1.7	**0.0028**	>2419.6	3076	18.7	63	14.4	141	Closed	5.31
7/12/2016	23.4	0.14	**0.035**	1.4	**0.0022**	2419.6	4106	33.2	41	26.2	113	Open	3.58
7/19/2016	23.1	0.10	**0.032**	**2.6**	**0.0022**	>2419.6	3255	12.1	20	7.4	104	Open	2.15
7/26/2016	23.8	0.17	**0.039**	2.0	0.0016	2419.6	2909	2.0	<10	14.5	113	Open	1.94
8/2/2016	23.2	0.21	**0.032**	1.8	0.0016	571.7	1354	4.1	<10	7.2	104	Open	1.52
8/9/2016	22.4	0.070	**0.027**	2.0	0.0013	1553.1	1178	13.2	20	5.2	141	Open	1.26
8/16/2016	22.3	0.14	**0.029**	1.1	0.0012	1299.7	1198	7.5	20	<1.0	121	Open	1.47
8/23/2016	21.6	0.14	ND	1.3	0.0014	1732.9	1076	21.6	10	4.1	162	Open	0.97
8/30/2016	21.1	0.14	**0.029**	1.0	**0.0019**	1203.3	959	41	41	7.4	152	Open	1.6
9/6/2016	20.8	0.070	0.021	1.8	0.0010	1553.1	1187	16.7	20	6.2	181	Open	1.26
9/13/2016	19.8	0.10	**0.022**	1.2	0.00096	816.4	1126	8.6	10	3.1	140	Closed	3.83
9/15/2016	19.9	0.18	**0.025**	2.0	0.00096	980.4	657	20.1	10	3.0	136	Closed	4.59
9/20/2016	21.1	0.18	**0.024**	1.4	0.00030	1986.3	2187	104.3	121	52.0	129	Closed	6.07
9/22/2016	20.1	0.070	**0.024**	0.66	0.00060	1956.3	1860	72.7	110	53.7	130	Closed	6.62
9/27/2016	19.8	0.070	**0.022**	1.7	0.00045	1413.6	2187	99.0	41	43.1	121	Closed	7.71
9/29/2016	20.0	0.10	**0.030**	1.3	0.00017	>2419.6	4611	**980.4**	**884**	**290.9**	122	Closed	8.18
10/4/2016	16.8	0.19	**0.039**	1.3	0.00033	1203.3	933	8.5	10	13.5	147	Open	0.97
10/11/2016	17.1	0.18	**0.030**	**2.5**	0.0016	1119.9	1050	14.6	31	11.9	142	Open	2.19
10/18/2016	16.7	0.28	**0.072**	1.5	0.0014	1986.3	1670	77.1	97	**61.7**	240	Closed	7.5

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III

Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L

Total Nitrogen: 0.38 mg/L

Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L

Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:

Beach posting is recommended when indicator organisms exceed any of the following levels:

Total coliforms: 10,000 per 100 ml

E. coli: 235 per 100 ml

Enterococcus: 61 per 100 ml

Table 4.1.4. 2016 Russian River at Vacation Beach Station Grab Sample Results

Vacation Beach*	Temperature	Total Nitrogen	Total Phosphorus	Turbidity	Chlorophyll-a	Total Coliforms (Colilert)	Total Coliforms Diluted 1:10 (Colilert)	E. coli (Colilert)	E. coli Diluted 1:10 (Colilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***		
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary	Jenner
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition	Gauge (ft)
5/10/2016	17.3	**0.46**	**0.036**	2.2	**0.0051**	1299.7	1723	13.2	10	<10	468	Open	0.93
5/17/2016	20.5	**1.3**	**0.034**	**2.6**	**0.0029**	727.0	677	5.2	10	3.1	377	Open	1.77
5/24/2016	18.6	0.30	**0.033**	1.6	0.0010	387.3	529	8.6	<10	2.0	343	Open	2.57
5/31/2016	21.0	0.35	**0.036**	1.8	**0.0023**	686.7	816	16.6	<10	5.1	277	Open	2.91
6/2/2016	22.9	-----	-----	-----	-----	461.1	670	9.6	<10	30	259	Closed	5.01
6/7/2016	20.9	0.25	**0.031**	1.4	**0.0020**	980.4	1333	30.9	30	40.2	224	Closed	7.71
6/14/2016	20.8	0.22	**0.034**	1.3	**0.0024**	1553.1	4674	17.3	20	**141**	202	Open	1.56
6/21/2016	21.8	0.15	**0.031**	1.2	**0.0050**	>2419.6	2359	95.8	75	**248.9**	186	Closed	5.69
6/23/2016	22.9	0.18	**0.031**	**2.4**	**0.0034**	>2419.6	4106	57.1	63	**95.9**	170	Closed	6.45
6/28/2016	24.3	0.18	**0.028**	2.0	**0.0034**	>2419.6	2603	16.9	<10	41.4	127	Open	1.43
7/5/2016	21.9	0.14	**0.037**	**2.9**	**0.0024**	>2419.6	2755	24.6	10	47.4	140	Closed	4.63
7/7/2016	23.1	0.10	**0.029**	**2.5**	**0.0026**	1986.3	2909	13.5	10	7.4	141	Closed	5.31
7/12/2016	23.3	0.24	**0.030**	2.0	0.00087	>2419.6	4884	5.1	20	32.0	113	Open	3.58
7/19/2016	23.3	0.14	**0.030**	2.0	**0.0022**	>2419.6	3076	4.1	<10	6.3	104	Open	2.15
7/26/2016	23.5	0.14	**0.029**	1.8	0.0011	1732.9	3255	22.8	31	31.3	113	Open	1.94
8/2/2016	23.5	0.14	**0.031**	2.1	**0.0020**	412.0	2382	15.8	10	44.3	104	Open	1.52
8/9/2016	22.5	0.14	**0.023**	2.2	0.0012	1732.9	2613	25.9	20	8.6	141	Open	1.26
8/16/2016	22.5	0.14	**0.025**	1.7	0.0017	>2419.6	2064	18.3	20	7.3	121	Open	1.47
8/23/2016	21.8	0.10	0.021	2.0	0.0014	1299.7	1145	9.7	<10	9.7	162	Open	0.97
8/30/2016	21.5	0.10	ND	1.4	0.00069	920.8	932	<10	<10	10.9	152	Open	1.6
9/6/2016	21.2	0.18	ND	**2.7**	0.00052	866.1	1396	5.2	10	3.0	181	Open	1.26
9/13/2016	20.2	0.10	0.021	1.6	0.00064	1119.9	860	3.1	20	5.1	140	Closed	3.83
9/15/2016	20.0	0.092	0.020	**2.4**	0.00032	1046.2	933	20.1	41	2.0	136	Closed	4.59
9/20/2016	20.9	0.14	0.021	2.0	0.00030	1119.9	1063	26.2	41	9.7	129	Closed	6.07
9/22/2016	19.6	0.10	**0.024**	2.1	0.0011	1732.9	1291	17.5	31	12.8	130	Closed	6.62
9/27/2016	19.6	0.10	**0.022**	**3.5**	0.00045	1553.1	1019	27.5	41	41.6	121	Closed	7.71
9/29/2016	20.0	0.10	**0.026**	**2.7**	0.00067	980.4	1187	7.5	31	5.2	122	Closed	8.18
10/4/2016	16.9	0.14	**0.027**	**2.7**	0.001	1046.2	1112	20.3	41	14.4	147	Open	0.97
10/11/2016	17.2	0.10	**0.023**	**3.8**	**0.0020**	980.4	1050	32.3	31	40.4	142	Open	2.19
10/18/2016	16.3	0.21	**0.050**	**3.6**	**0.0018**	1732.9	934	65	85	22.8	240	Closed	7.5

* All results are preliminary and subject to final revision
** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.
*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III
Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L
Total Nitrogen: 0.38 mg/L
Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L
Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:
Beach posting is recommended when indicator organisms exceed any of the following levels:
Total coliforms: 10,000 per 100 ml
E. coli: 235 per 100 ml
Enterococcus: 61 per 100 ml

ammoniacal nitrogen (together referred to as Total Kjeldahl Nitrogen or TKN), and nitrate/nitrite nitrogen (Appendix 4.5).

Total nitrogen concentrations were observed to exceed the recommended USEPA levels three times at the Patterson Point station, and twice each at the Monte Rio and Vacation Beach monitoring stations (Tables 4.1.2 through 4.1.4). All of these exceedances occurred at the beginning of the monitoring season during open conditions with flows over 340 cfs (Figure 4.1.31). Whereas some of the lowest total nitrogen values observed at the freshwater stations occurred during closed conditions in September when flows were as low as 121 cfs (Tables 4.1.2 through 4.1.4). Overall, total nitrogen exceedances constituted 8.1% of all samples collected (Figure 4.1.31).

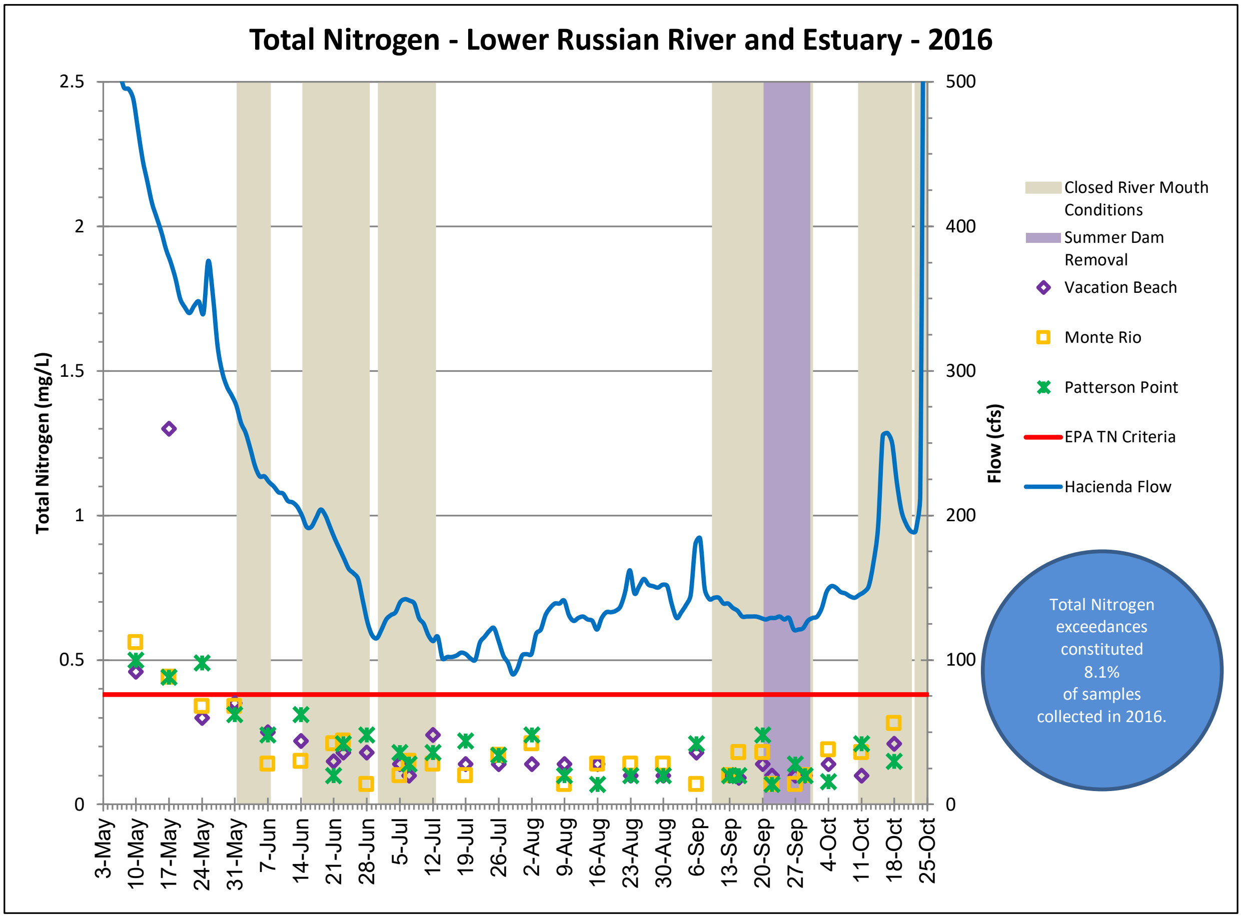

Figure 4.1.31. 2016 Russian River Grab Sampling Results for Total Nitrogen

The maximum total nitrogen concentration observed at Patterson Point was 0.50 mg/L on 10 May during open conditions with a flow of approximately 468 cfs (Table 4.1.2). The mean concentration at Patterson Point was 0.20 mg/L. The minimum concentration at Patterson Point was 0.070 mg/L, which occurred twice, on August 16 during open conditions with a flow of approximately 121 cfs, and on September 22 during closed conditions with a flow of approximately 130 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 0.22 and 0.24 mg/L, respectively (Table 4.1.2).

The maximum total nitrogen concentration observed at Monte Rio was 0.56 mg/L on May 10 during open conditions with a flow of approximately 468 cfs (Table 4.1.3). The mean concentration at Monte Rio was 0.18 mg/L. The minimum concentration at Monte Rio was 0.070 mg/L, which occurred five times during open and closed conditions and flows ranging from 121 to 181 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 0.10 and 0.21 mg/L, respectively (Table 4.1.3).

The maximum total nitrogen concentration observed at Vacation Beach was 1.3 mg/L on May 17 during open conditions with a flow of approximately 377 cfs (Table 4.1.4). The mean concentration at Vacation Beach was 0.21 mg/L. The minimum concentration at Vacation Beach was 0.092 mg/L, which occurred on September 15 during closed conditions and a flow of approximately 136 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 0.14 mg/L on both dates (Table 4.1.4).

The USEPA's desired goal for total phosphates as phosphorus in Aggregate Ecoregion III has been established as 21.88 micrograms per liter (µg/L), or approximately 0.022 mg/L, for rivers and streams not discharging into lakes or reservoirs (USEPA, 2000). Total phosphorus concentrations at the freshwater monitoring stations exceeded the U.S. EPA criteria approximately 83.9% of the time, continuing a trend of consistent exceedances observed in previous years.

Exceedances occurred during open and closed Estuary conditions, and in river flows ranging from 104 cfs to 468 cfs. Total phosphorus values were observed to generally be higher in the spring and early summer, trending downward through the rest of the season (Figure 4.1.32).

The maximum total phosphorus concentration observed at Patterson Point was 0.065 mg/L on October 18 during closed conditions with a flow of approximately 240 cfs (Table 4.1.2). The mean concentration at Patterson Point was 0.031 mg/L. The minimum concentration at Patterson Point was a non-detect (ND), which occurred on September 6 during open conditions with a flow of approximately 181 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 0.034 and 0.033 mg/L, respectively (Table 4.1.2).

The maximum total phosphorus concentration observed at Monte Rio was 0.072 mg/L on October 18 during closed conditions with a flow of approximately 240 cfs (Table 4.1.3). The mean concentration at Monte Rio was 0.032 mg/L. The minimum concentration at Monte Rio was ND, which occurred on 23 August during open conditions with a flow of approximately 162 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 0.032 mg/L recorded on both dates (Table 4.1.3).

The maximum total phosphorus concentration observed at Vacation Beach was 0.050 mg/L on October 18 during closed conditions with a flow of approximately 240 cfs (Table 4.1.4). The mean concentration at Vacation Beach was 0.028 mg/L. The minimum concentration at Vacation Beach was ND, which occurred on August 30 and September 6 during open conditions and flows of approximately 152 and 181 cfs, respectively. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 0.030 and 0.031 mg/L, respectively (Table 4.1.4).

Turbidity

There were three exceedances each of the Turbidity EPA criteria at the Patterson Point and Monte Rio stations, and 11 exeedances at the Vacation Beach station (Figure 4.1.33). These exceedances of the Turbidity criteria occurred under open and closed conditions in flows that ranged from 104 cfs to 377 cfs. In addition, Vacation Beach is subject to elevated turbidity from the effects of the summer dam over flow and fish ladder outflow occurring just upstream from the station.

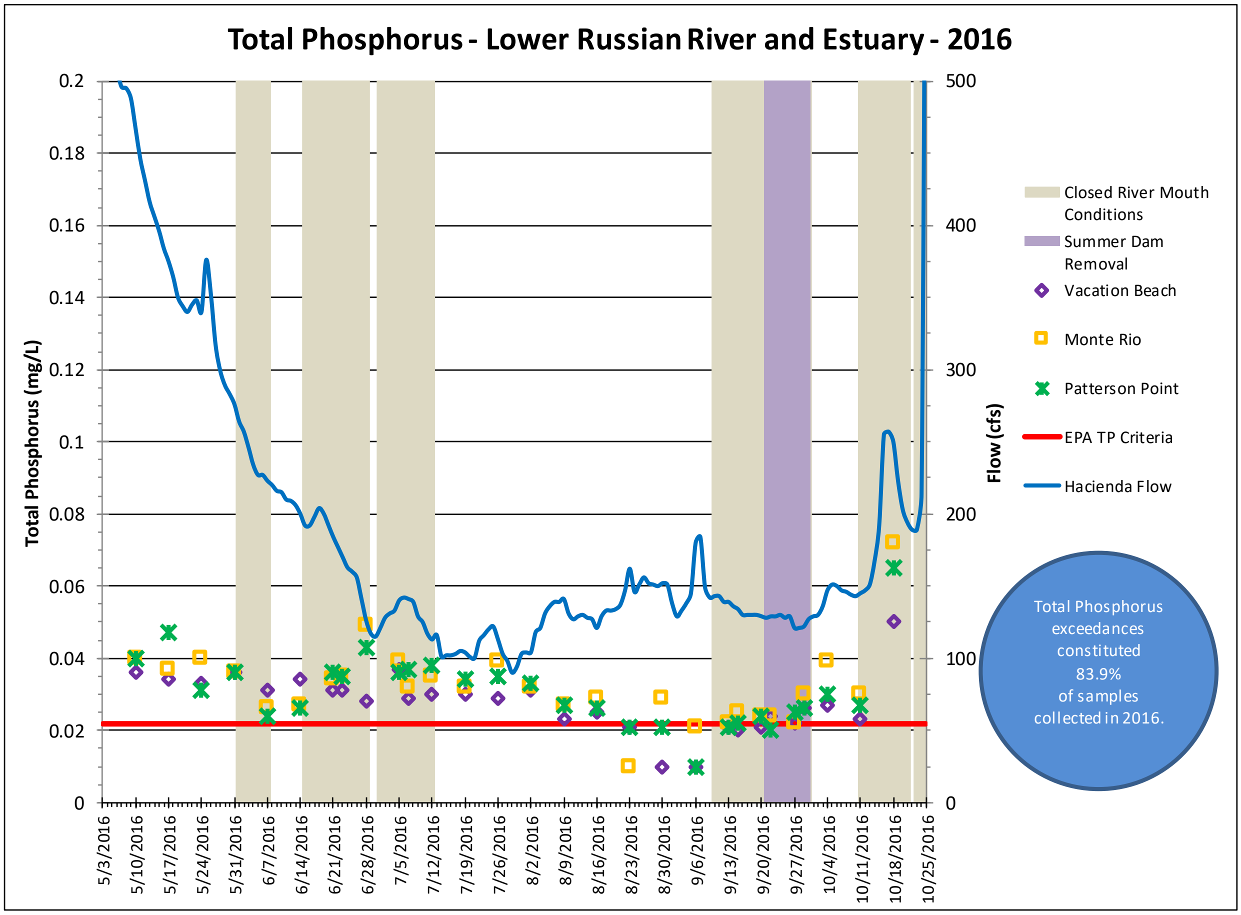

Figure 4.1.32. 2016 Russian River Grab Sampling Results for Total Phosphorus

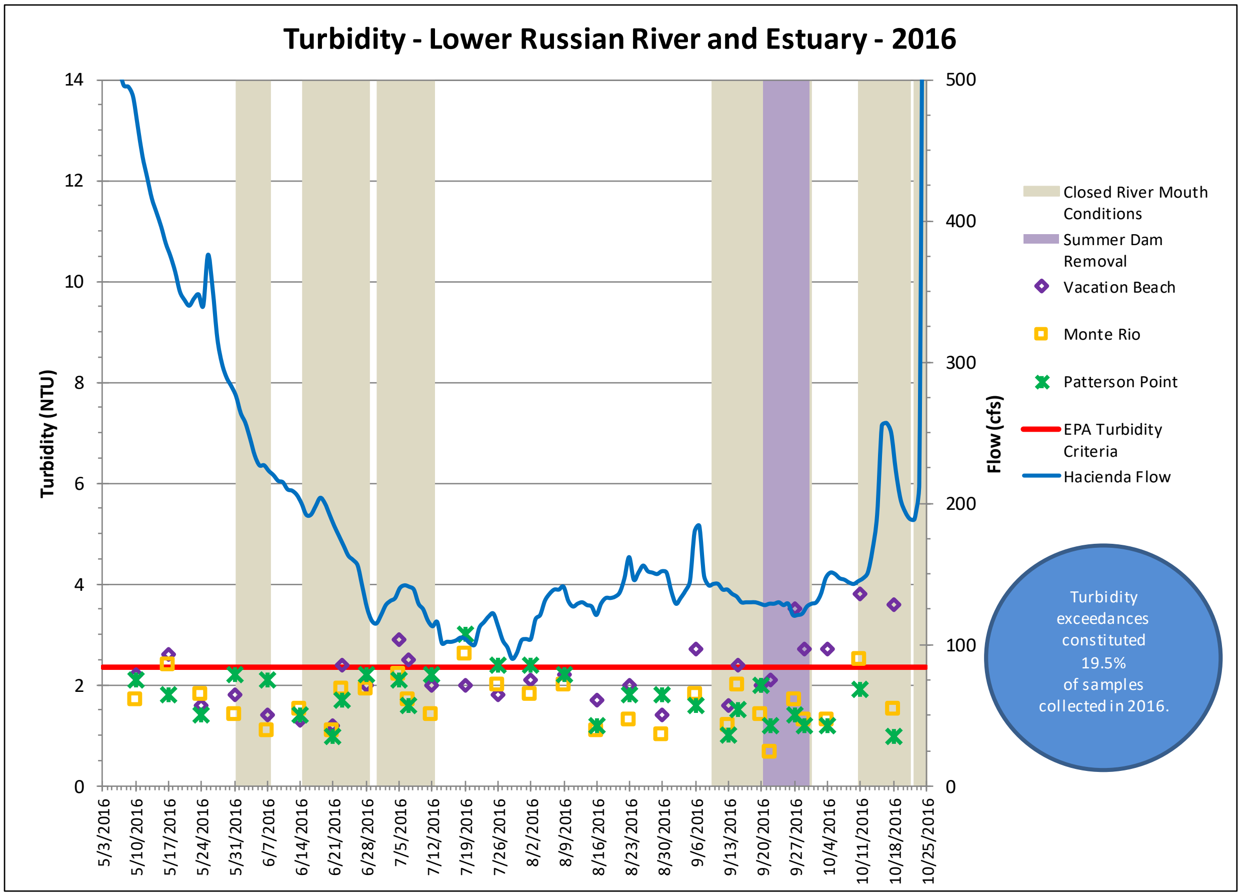

Figure 4.1.33. 2016 Russian River Grab Sampling Results for Turbidity

The maximum turbidity value observed at Patterson Point was 3.0 NTU on July 19 during open conditions with a flow of approximately 104 cfs (Table 4.1.2). The mean value at Patterson Point was 1.7 NTU. The minimum value at Patterson Point was 0.97 NTU, which occurred on October 18 during closed conditions with a flow of approximately 240 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with values of 3.0 and 2.4 NTU, respectively (Table 4.1.2).

The maximum turbidity value observed at Monte Rio was 2.6 NTU on July 19 during open conditions with a flow of approximately 104 cfs (Table 4.1.3). The mean value at Monte Rio was 1.6 NTU. The minimum value at Monte Rio was 0.66 NTU, which occurred on September 22 during closed conditions with a flow of approximately 130 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with values of 2.6 and 1.8 NTU, respectively (Table 4.1.3).

The maximum turbidity value observed at Vacation Beach was 3.8 NTU on October 11 during open conditions with a flow of approximately 142 cfs (Table 4.1.4). The mean value at Vacation Beach was 2.2 NTU. The minimum value at Vacation Beach was 1.2 NTU, which occurred on June 21 during closed conditions and a flow of approximately 186 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with concentrations of 2.0 and 2.1 NTU, respectively (Table 4.1.4).

Chlorophyll a

In the process of photosynthesis, *Chlorophyll a* - a green pigment in plants, absorbs sunlight and combines carbon dioxide and water to produce sugar and oxygen. *Chlorophyll a* can therefore serve as a measureable parameter of algal growth. Qualitative assessment of primary production on water quality can be based on *Chlorophyll a* concentrations. A U.C. Davis report on the Klamath River (1999) assessing potential water quality and quantity regulations for restoration and protection of anadromous fish in the Klamath River includes a discussion of *Chlorophyll a* and how it can affect water quality. The report characterizes the effects of *Chlorophyll a* in terms of different levels of discoloration (e.g., no discoloration to some, deep, or very deep discoloration). The report indicated that less than 10 µg/L (or 0.01 mg/L) of *Chlorophyll a* exhibits no discoloration (Deas and Orlob, 1999). Additionally, the USEPA criterion for *Chlorophyll a* in Aggregate Ecoregion III is 1.78 µg/L, or approximately 0.0018 mg/L for rivers and streams not discharging into lakes or reservoirs (USEPA, 2000). However, it is important to note that the EPA criterion is established for freshwater systems, and as such, is only applicable to the freshwater portions of the Estuary. Currently, there are no numeric *Chlorophyll a* criteria established specifically for estuaries.

Chlorophyll a concentrations were less than 0.01 mg/L at all stations during the monitoring period, the level recommended to prevent discoloration of surface waters (Tables 4.1.2 through 4.1.4). However, *Chlorophyll a* concentrations did exceed the EPA criteria approximately 39.1% of the time at the stations throughout the season under open and closed Estuary conditions, and during flows ranging from 104 cfs to 468 cfs (Figure 4.1.34). Similar to the trend for total phosphorus, *Chlorophyll a* values were observed to generally be higher in the spring and early summer, trending downward through the rest of the season (Figure 4.1.34).

The maximum *Chlorophyll a* concentration observed at Patterson Point was 0.0058 mg/L on 7 June during closed conditions with a flow of approximately 224 cfs (Table 4.1.2). The mean value at Patterson Point was 0.0017 mg/L. The minimum value at Patterson Point was ND, which occurred on October 4 during open conditions with a flow of approximately 147 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with values of 0.0011 and 0.0012 mg/L, respectively (Table 4.1.2).

The maximum *Chlorophyll a* concentration observed at Monte Rio was 0.0063 mg/L on May 10 during open conditions with a flow of approximately 468 cfs (Table4. 1.3). The mean value at Monte Rio was 0.0020 mg/L. The minimum value at Monte Rio was 0.00017 mg/L, which occurred on 29 September during closed conditions with a flow of approximately 122 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on July 19 and August 2, with values of 0.0022 and 0.0016 mg/L, respectively (Table 4.1.3).

The maximum *Chlorophyll a* concentration observed at Vacation Beach was 0.0051 mg/L on 10 May during open conditions with a flow of approximately 468 cfs (Table 4.1.4). The mean value at Vacation Beach was 0.0018 mg/L. The minimum value at Vacation Beach was 0.00030 mg/L, which occurred on September 20 during closed conditions and a flow of approximately 129 cfs. Finally, the lowest flow recorded during the sampling events was 104 cfs, which occurred on

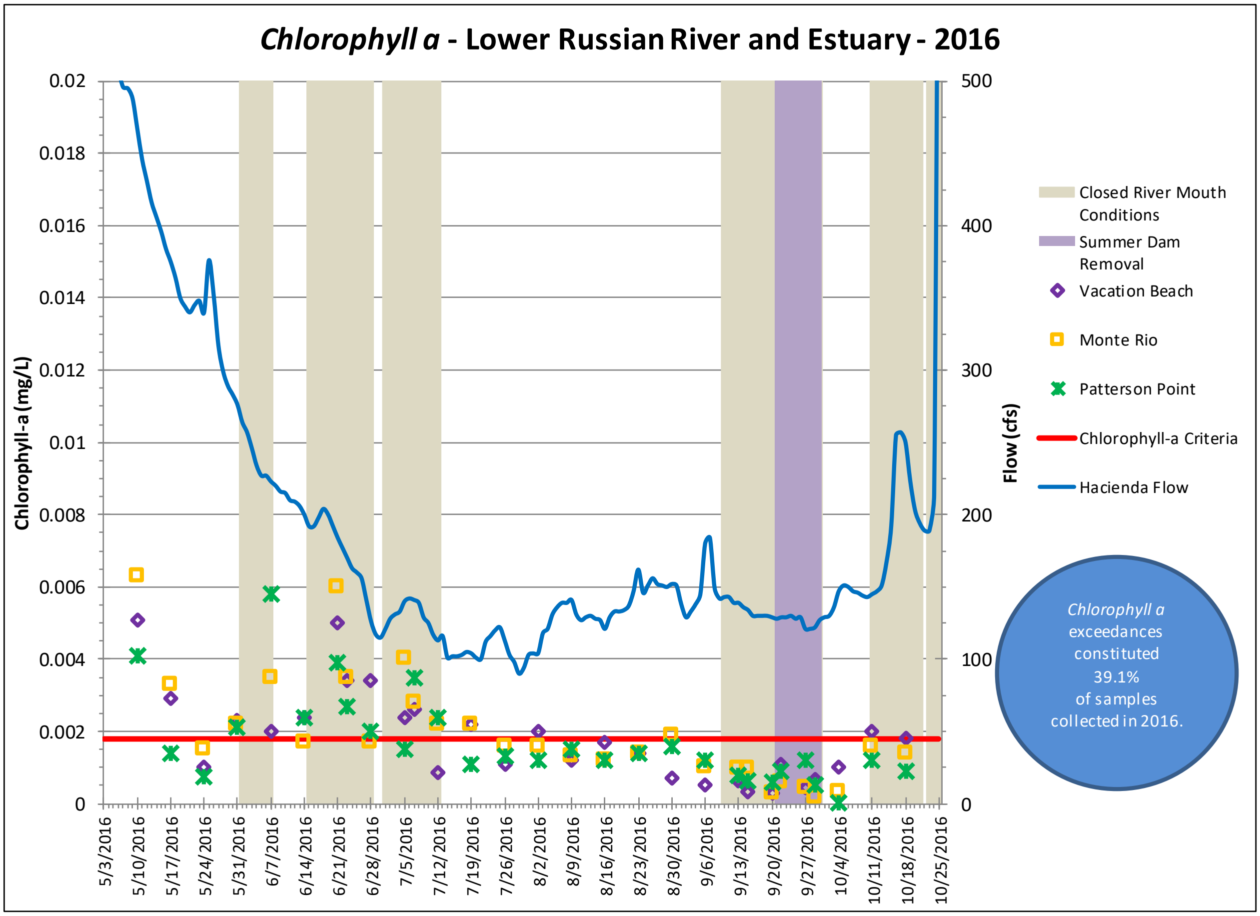

Figure 4.1.34. 2016 Russian River Grab Sampling Results for *Chlorophyll a*

July 19 and August 2, with concentrations of 0.0022 and 0.0020 mg/L, respectively (Table 4.1.4).

Indicator Bacteria

The California Department of Public Health (CDPH) developed the "Draft Guidance for Fresh Water Beaches," which describes bacteria levels that, if exceeded, may require posted warning signs in order to protect public health (CDPH 2011). The CDPH draft guideline for single sample maximum concentrations is: 10,000 most probable numbers (MPN) per 100 milliliters (ml) for total coliform, 235 MPN per 100 ml for E. coli, and 61 MPN per 100 ml for Enterococcus. In 2012, the United States Environmental Protection Agency (EPA) issued Clean Water Act (CWA) §304(a) Recreational Water Quality Criteria (RWQC) for States (EPA 2012). The RWQC recommends using two criteria for assessing water quality relating to fecal indicator bacteria: the geometric mean (GM) of the dataset, and changing the single sample maximum (SSM) to a Statistical Threshold Value (STV) representing the 75th percentile of an acceptable water-quality distribution. However, the EPA recommends using STV values as SSM values for potential recreational beach posting and those values are provided in this report for comparative purposes. It must be emphasized that these are draft guidelines and criteria, not adopted standards, and are therefore both subject to change (if it is determined that the guidelines and/or criteria are not accurate indicators) and are not currently enforceable.

Samples were collected during the monitoring season for diluted and undiluted analysis of E. coli and total coliform for comparative purposes and the results are included in Tables 4.1.2 through 4.1.4 and Figures 4.1.35 and 4.1.36. Samples collected for Enterococcus were undiluted only and results are included in Tables 4.1.2 through 4.1.4 and Figure 4.1.37. The Water Agency submitted samples to the Sonoma County DHS Public Health Division Lab in Santa Rosa for bacteria analysis. E. coli and total coliform were analyzed using the Colilert method and Enterococcus was analyzed using the Enterolert method. Samples for all other constituents were submitted to Alpha Labs in Ukiah for analysis. Total Coliform and E. coli data presented in Figures 4.1.35 and 4.1.36 utilize undiluted sample results unless the reporting limit has been exceeded, at which point the diluted results are utilized.

In 2014, staff at the NCRWQCB indicated that Enterococcus was not being utilized as a fecal indicator bacteria due to uncertainty in the validity of the lab analysis to produce accurate results, as well as evidence that Enterococcus colonies can be persistent in the water column and therefore its presence at a given site may not always be associated with a fecal source. Water Agency staff will continue to collect Enterococcus samples and record and report the data however, Enterococcus results will not be relied upon when coordinating with the NCRWQCB and Sonoma County DHS about potentially posting warning signs at freshwater beach sites or to discuss potential adaptive management actions including mechanical breaching of the sandbar to address potential threats to public health.

The Monte Rio station was observed to have two exceedances of the RWQC for E. coli, representing 2.2% of the total samples collected (Figure 4.1.35). The first exceedance was slightly higher than the RWQC with a value of 261.3 MPN. Estuary closures may have had an effect on E. coli, as values were observed to increase during closure, including the first Monte Rio exceedance which occurred on 23 June with a flow of approximately 170 cfs (Table 4.1.3). The second exceedance was significantly higher with a value of 980.4 MPN that occurred on 29 September with a flow of approximately 170 cfs. Summer dam removal may have also had an effect as this exceedance occurred during closed conditions and the removal of the Johnson's Beach summer dam (Figure 4.1.35).

There were no exceedances of the RWQC for total coliform at the three stations monitored for total coliforms in 2016 (Figure 4.1.36). Estuary closures during June may have had an effect on total coliform as values were observed to increase during closure, however not high enough to exceed the RWQC. Summer dam removal may have also had an effect on total coliform with an elevated concentration observed to occur on September 29 during the removal of the Johnson's Beach summer dam (Figure 4.1.36).

Based upon the recommended Enterococcus RWQC for fresh water beaches, several exceedances were observed representing 13.3% of the total samples collected (Figure 4.1.37). There were two exceedances at Patterson Point, seven exceedances at Monte Rio, and three exceedances at Vacation Beach. Estuary closures may have had an effect on E. coli, as values were observed to increase and exceed the RWQC during closures in June and in the latter half of the season, with flows varying from 113 cfs to 259 cfs. External factors likely had an effect on increasing Enterococcus concentrations including the removal of the summer dams in

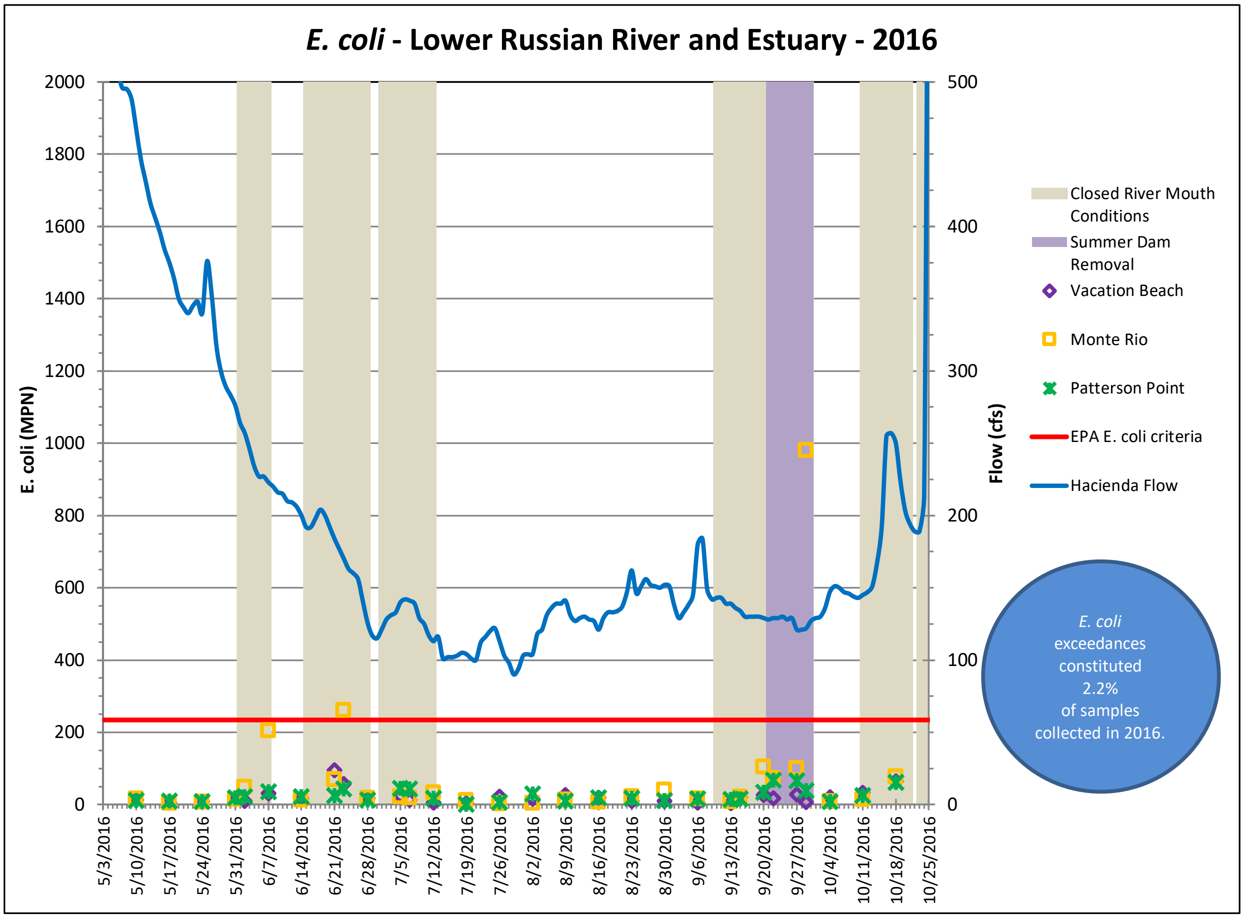

Figure 4.1.35. 2016 Russian River Grab Sampling Results for E. coli

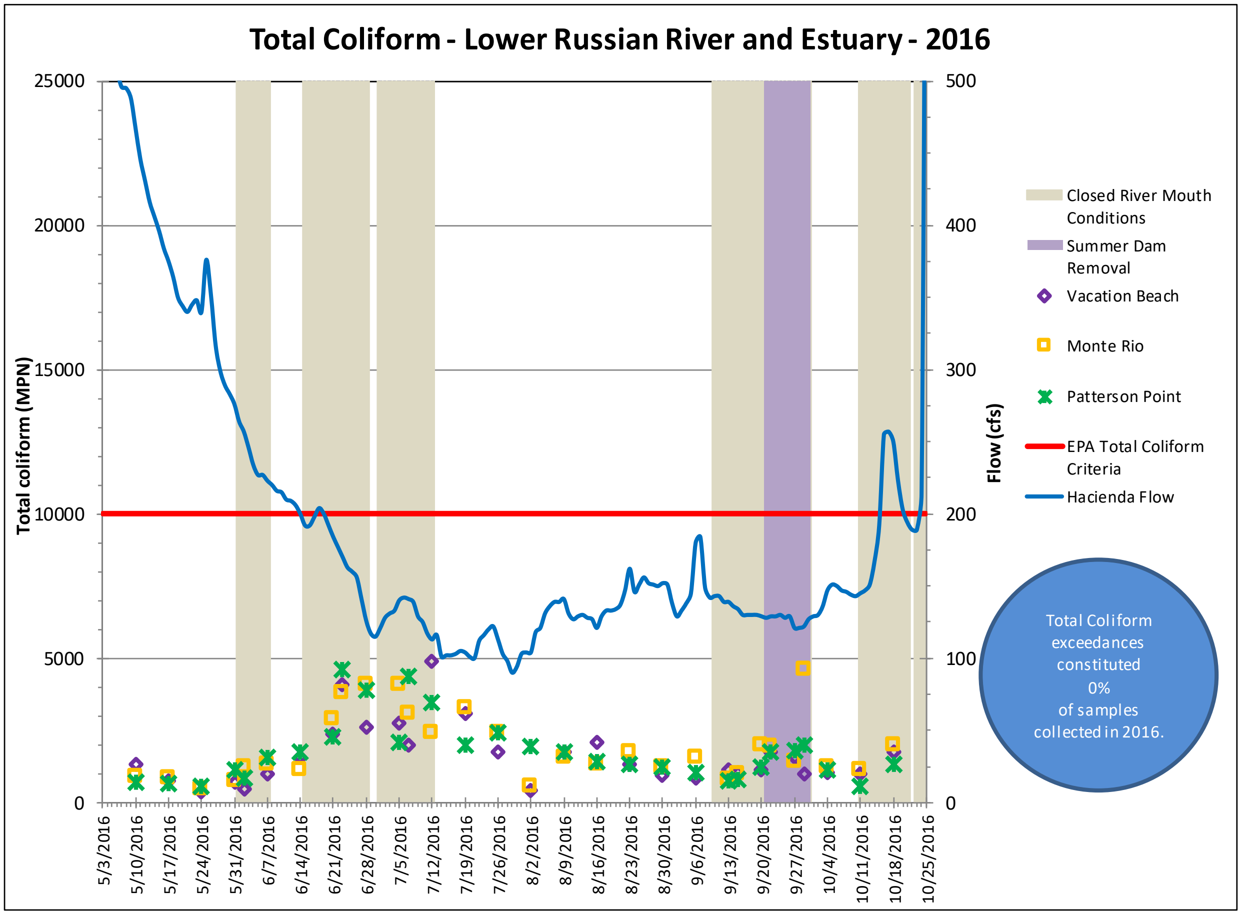

Figure 4.1.36. 2016 Russian River Grab Sampling Results for Total Coliform

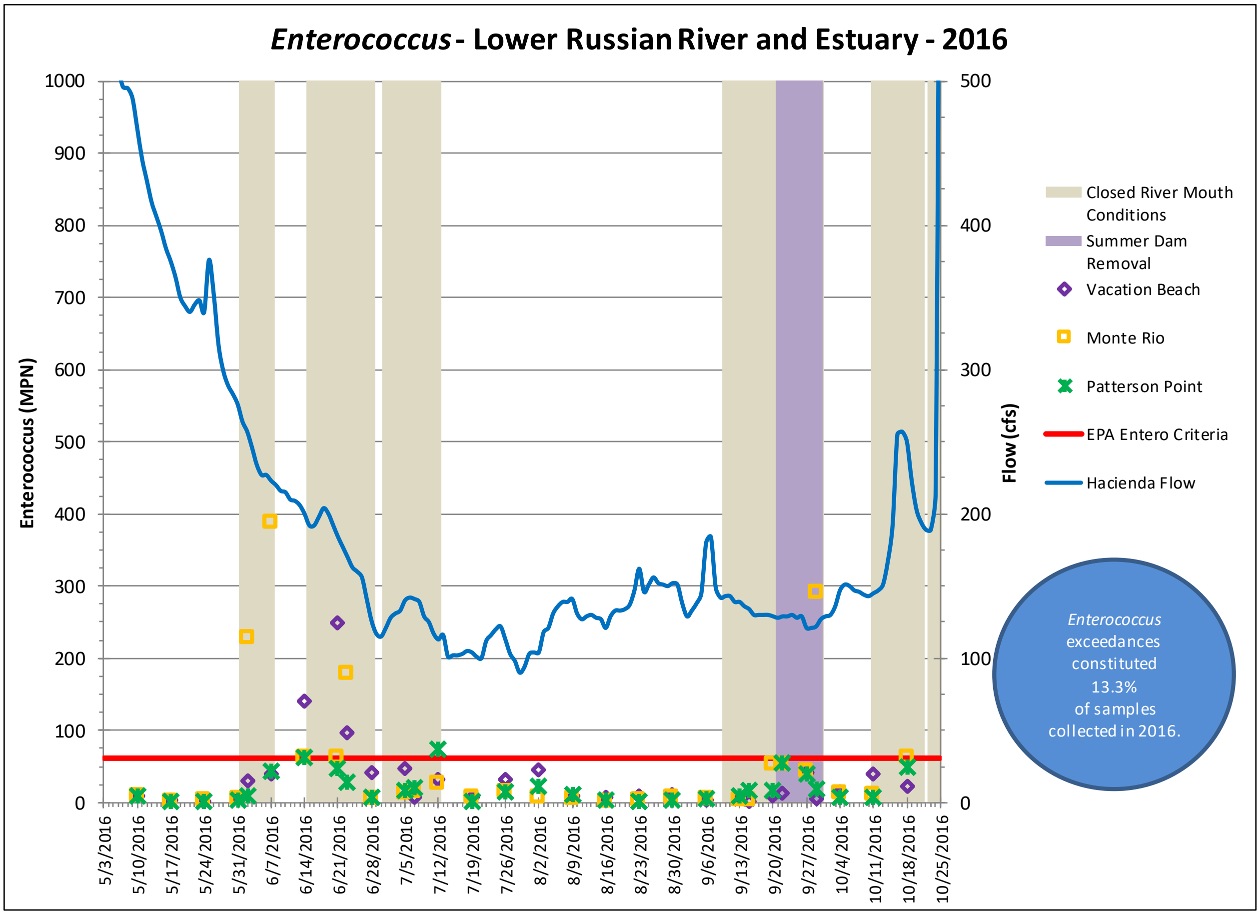

Figure 4.1.37. 2016 Russian River Grab Sampling Results for Enterococcus

Guerneville during an extended period of estuary closures. Similar to the E.coli and total coliform results, the Monte Rio station was observed to have an elevated concentration of 290.9 MPN that occurred during the removal of the Johnson's Beach summer dam on 29 September (Figure 4.1.37).

Conclusions and Recommendations

Continuous Water Quality Monitoring Conclusions

Water quality conditions observed during the 2016 monitoring season were similar to conditions observed during previous monitoring seasons, and similar to the dynamic conditions associated with an estuarine river system. The differing physical properties associated with freshwater versus those of saltwater play a pivotal role in the stratification that is common in the Russian River Estuary. Since the saltwater is denser than the freshwater inflow, the saltwater layer is observed below the freshwater layer, and the slope of the temperature and density gradients is typically steepest at the halocline. While this relationship is a key player in what shapes the water quality conditions in the estuary, there are other influences at work in the estuary as well, including wind mixing, river inflow, tidal influence, shape and size of the river mouth, air temperatures, and others.

There were five closures during the lagoon management season and Water Agency staff attempted to implement an outlet channel twice during the season. The first management period closure occurred for a period of 7 days between June 1 and 7. The second closure occurred for a period of 14 days between June 15 and 27. In both cases, outlet channel implementation was followed by self-scour of the channel within a day. The duration of these closures events and duration of the outlet channel were not of sufficient length to effectively compare the suitability of aquatic habitat for rearing salmonids between open and closed conditions. However, staff were able to collect data that provides a fuller understanding of salinity migration in the Upper Reach of the Estuary.

As freshwater flows in the Russian River decrease through spring, the salt layer typically migrates upstream. With the end of drought conditions in 2016, mainstem Russian River flows decreased later in the season than in 2013 through 2015, but were similar in timing to 2012. 2016 mainstem flows were observed to drop below 200 cfs by mid-June, whereas mainstem flows decresed below 200 cfs in mid-May during the drought years of 2013 through 2015.

Although salinity migration patterns in the upper reach of the Estuary were fairly similar to those in prior monitoring years, the Brown's Pool (RK 11.3) station had significantly more brackish water in 2016 than was observed in 2015. Whereas the bottom of Brown's Pool became predominantly brackish during open and closed conditions throughout the 2016 monitoring season, the bottom was only periodically brackish during open conditions in 2015. Concentrations in 2016 were as high as 6.5 ppt during open conditions, and occurred more frequently, compared to a maximum of approximately 4.3 ppt in 2015.

Brackish water had not been observed at Brown's Pool prior to the 2013 monitoring season, however Water Agency staff had only previously deployed a continuously monitoring sonde at this station in the 2011 season (Manning and Martini-Lamb, 2012). Even so, it is not unreasonable to expect salinity migration to periodically occur in this area, given the proximity of the Brown's Pool station to Moscow Road Bridge (RK 10.15), where brackish water has been observed to occur.

By contrast, monitoring conducted at the bottom of the Patterson Point station in Villa Grande did not detect any significant salinity migration into the site during open or closed conditions. Maximum salinity values observed at Patterson Point were approximately 0.3 ppt, and occurred during open conditions on at the end of August with flows of approximately 150 cfs. Water is considered fresh at approximately 0.5 ppt. These results correspond with the data collected in the Upper Reach of the Estuary and the MBA since 2010 and further supports the theory that Brown's Riffle and the confluence of Austin Creek provide a significant hydrologic barrier to salinity migration in the mainstem Russian River.

During prolonged barrier beach closures in 2016, overall water quality conditions were observed to be similar to those of previous years. Typically during a closure or perched event, the surface and mid-depth sondes in the lower and middle reaches of the Estuary would experience a decrease in salinity and an increase in temperature. Conversely, during prolonged closures or perched events, the mid-depth and bottom sondes in the upper reach of the Estuary typically experience increases in salinity as brackish water migrates into the area, with temperature responses that are variable. Conditions observed in the saline layer during the 2016 monitoring season were no exception.

DO response to Estuary closure events was variable in the Upper Reach and dependent on the presence and movement of salinity, the relative strength of stratification, circulation patterns, and flows in the Russian River. The presence of salinity would typically coincide with the presence of depressed DO levels, but not always (i.e. Freezeout Creek at the mid-depth sonde during the late September closure), suggesting that variability is dependent on relative DO concentrations in the migrating salt wedge, the length of time of Estuary closures, the timing of subsequent closure events, freshwater inflow rates, the DO concentration of inflowing freshwater, and subsequent tidal inundation and mixing.

Temperature, pH, and dissolved oxygen patterns during the 2016 monitoring season were also similar to those observed in previous monitoring years. While the Russian River Estuary is a dynamic estuarine system, the seasonal changes during the monitoring seasons have largely followed similar patterns each year since the implementation of the Biological Opinion in 2009.

To further illustrate the extent of salinity migration, a graphical representation of the maximum salinity levels recorded at various stations in the Russian River Estuary between 2009 and 2016 is being presented (Figure 4.1.38). The sondes chosen for this graph were situated in the lower portion of the water column at each station, where saline water would be expected to occur. This corresponds to approximately three to four meter depths for the Mouth, Patty's Rock, and Sheephouse Creek stations, six to nine meter depths at the Heron Rookery station, six to seven meter depths at the Freezeout Creek station, eight to ten meter depths at the Brown's Pool station, six to eight meter depths at Villa Grande, nine to eleven meters depth at Patterson Point, and one to two meters at the Monte Rio station. In the upper reaches of the Estuary and MBA, the sondes are located on the bottom of the river because the salt layer is typically thin when it occurs at these river locations. Excluding the depth variations, the graph depicts the decrease in salinity the further upstream in the Estuary and MBA the monitoring station is located.

The graph also illustrates the variable nature of salinity levels in the Upper Estuary. For instance, in 2014 and 2016, the maximum salinity concentrations observed at Brown's Pool were nearly identical, with concentrations of 11.3 and 11.1 ppt, respectively. However, at Freezeout Creek, the maximum salinity concentration was 14.1 ppt in 2014, whereas the maximum salinity concetration was 19.2 ppt in 2016. In addition, Brown's Pool has been observed to have maximum salinity concentrations that range from a low of 0.4 ppt in 2011 to a high of 11.3 ppt in 2014. Likewise, the maximum salinity concentrations observed at Freezeout Creek range from a low of 4.8 ppt in 2011 to a high of 25.9 ppt in 2013.

Note that there are no elevated salinity levels recorded in the Maximum Backwater Area for any monitoring seasons. As was mentioned above, it is possible that saline water does not migrate past the riffle between Brown's Pool and the confluence of Austin Creek due to hydrologic and/or geologic conditions that serve to define a transition from the Russian River Estuary and the beginning of the Maximum Backwater Area.

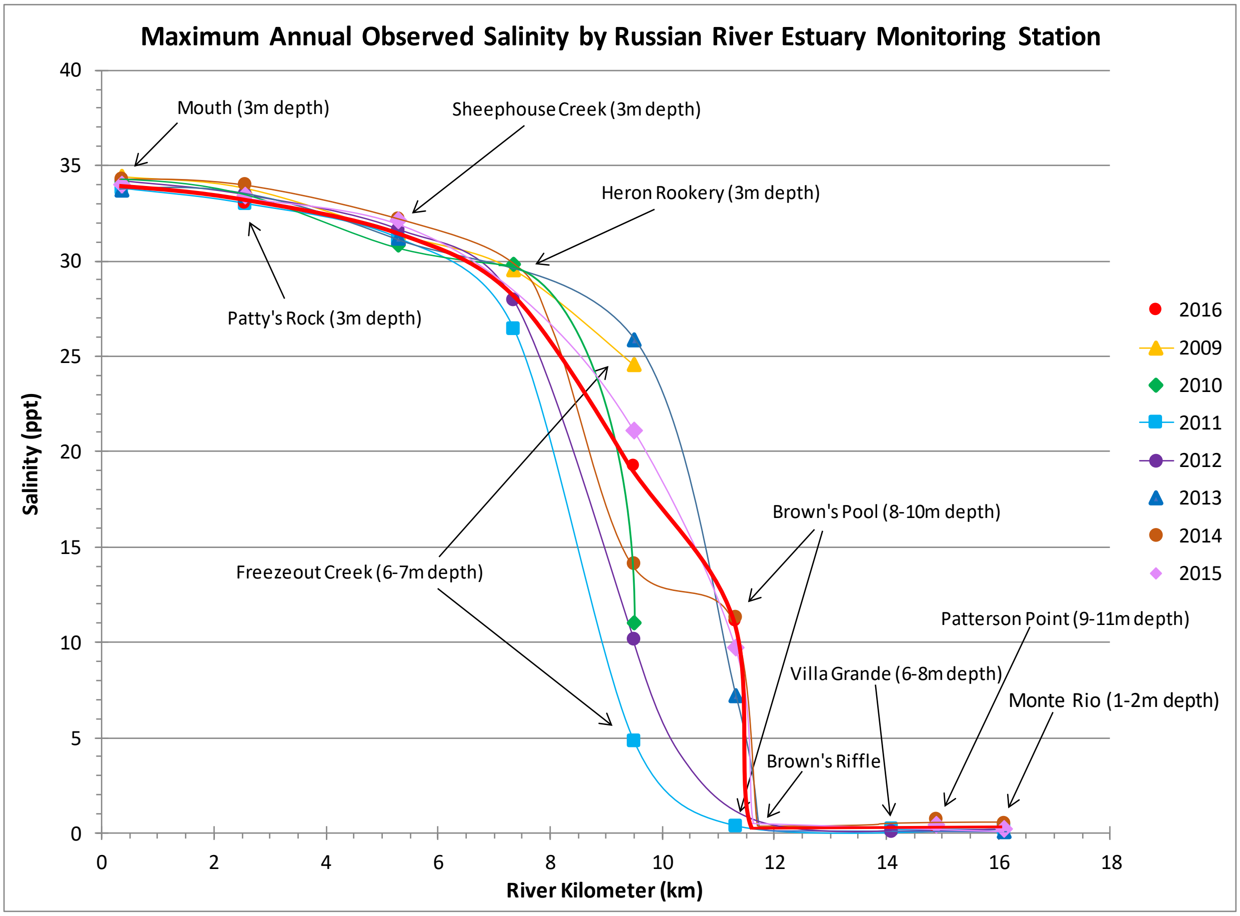

Figure 4.1.38. The maximum salinities at monitoring stations throughout the Russian River Estuary and Maximum Backwater Area between the years of 2009 and 2016.

The water quality conditions observed during the lagoon management season, particularly in the upper reach of the Estuary and in the MBA, indicates the expansion of freshwater and brackish water quality conditions during river mouth closures. These expanded aquatic habitat conditions may support additional rearing habitat for juvenile steelhead.

Water Quality Grab Sampling Conclusions

The 2016 grab sampling effort in the Russian River Estuary continued to collect a robust set of data similar in effort to the 2012 through 2015 monitoring seasons. Additional focused sampling was conducted during or after Estuary closures, as well as during summer dam removal in late September. Table 4.1.5 shows the total yearly number of sampling trips and the total number of samples collected within the freshwater portions of the Russian River Estuary and Maximum Backwater Area during each monitoring season since the implementation of the Biological Opinion in 2009.

The 2016 grab sampling effort observed Total Phosphorus exceedances in 83.9% of all samples collected (Table 4.1.6). This is not uncommon in the Russian River Estuary, and similar

Table 4.1.5. The total number of grab sampling trips per monitoring season and the total number of samples taken in the freshwater portion of the Russian River Estuary and Maximum Backwater Area per monitoring season. Note; duplicate and triplicate samples were counted as separate samples during the same sampling trip.

Estuary Monitoring Season	Total Number of Sampling Trips	Total Number of Samples
2009	7	7
2010	13	39
2011	13	52
2012	18	72-90
2013	33	98
2014	26-31	104-111
2015	26-27	104-107
2016	29-30	87-90

Table 4.1.6. The percentages of freshwater samples taken that were in exceedance of U.S. EPA water quality criteria for Total Phosphorus, Total Nitrogen, and Chlorophyll a. Note; Chlorophyll a was not quantified below 0.01 mg/L in 2009, and as such, cannot be verified against the U.S. EPA criteria of 0.00178 mg/L. Also, the Total Nitrogen values in 2009 were not quantified sufficiently against the criteria to make comparisons. The U.S. EPA criteria for Total Nitrogen is 0.38 mg/L, and the criteria for Total Phosphorus is 0.02188 mg/L.

Estuary Monitoring Season	Percentage of Total Phosphorus Samples in Exceedance	Percentage of Total Nitrogen Samples in Exceedance	Percentage of Total *Chlorophyll a* Samples in Exceedance
2009	100	N/A	N/A
2010	84.6	15.4	18.0
2011	92.3	30.8	23.7
2012	61.5	6.9	11.5
2013	99.0	15.3	44.9
2014	100	14.4	23.1
2015	86.5	1.9	26.0
2016	83.9	8.1	39.1

percentages of the samples analyzed for Total Phosphorus were in exceedance during previous monitoring seasons. Table 4.1.6 shows the percentage of samples that were in exceedance each season since 2009.

The Total Nitrogen and Chlorophyll a exceedances for samples taken during 2016 were also similar to percentages observed in previous monitoring years (Table 4.1.6). Year to year variability in the percentage of exceedances for these three constituents can be attributed in part to: the frequency and timing of storm events, fluctuating freshwater inflow rates, the frequency and timing of barrier beach closures, the strength of tidal cycles, summer dam removal, topography, relative location within the Estuary, and wind mixing.

The E. coli exceedances since the implementation of the Biological Opinion in 2009 until 2016 can be seen in Table 4.1.7. However, E. coli was not sampled for in 2010, with sampling being conducted for fecal coliforms instead. Samples collected in 2009 were analyzed using the multiple tube fermentation technique, whereas samples collected from 2011 through 2016 were analyzed using the Colilert Quanti-Tray method. Percentages for total coliform samples are not shown here since values were not quantified above 1600 MPN for 2010 and a portion of 2011, or above >2419.6 MPN for 2012, 2013 and a portion of the 2014 season. Both levels are below CDPH Guidelines, therefore it is impossible to establish percent criteria exceedances in this case.

Data collected through the grab sampling effort in 2016 appear consistent with data collected between 2009 and 2015. Further analysis could elucidate any trends that may exist temporally or longitudinally through the Russian River Estuary and guide water quality monitoring efforts in the future.

As described in the 2015 annual report, time series trend analyses of the grab sampling data collected could prove useful in the future. Trend analyses could determine if there have been changes over time for any of the constituents collected under this project. Certain trend tests are used for non-parametric data analysis such as water quality data, including the Sen Slope test, the Kendall-Theil test, the Seasonal Kendall test, or a variety of other suitable statistical tests. Analyses of this nature require both time and expert knowledge of environmental statistical analysis. As such, they are difficult to run and outside the scope of this project at this time. In the future, allocating resources to analyses of this nature, on these data, would likely give a better understanding of the existence, or absence, of trends in the data.

Table 4.1.7. The percentages of freshwater samples taken that were in exceedance of CDPH Guidelines for E. coli for the sampling years 2009 through 2016. Note that for 2009, the analyzing method was multiple tube fermentation, and for 2011-2016 the method was Colilert Quanti-Tray.

Estuary Monitoring Season	Percentage of Total E. coli Samples in Exceedance
2009	0
2010	N/A
2011	0
2012	0
2013	1.0
2014	6.3
2015	1.9
2016	2.2

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4.2 Algae Sampling

Monitoring of periphytic and planktonic algae was conducted to: document the algal response following estuary closure; and establish baseline ecological data for algal populations representative of habitats available in the Russian River. Monitoring for both was conducted as soon as river flows allowed a systematic investigation of abundance, cover, and successional processes. Surveys followed spring draw down, starting in June and continuing approximately every two weeks through October 2016.

Patterson Point was monitored to evaluate newly flooded shoreline areas in the Russian River Estuary following river mouth closures from May 15 to October 15. Patterson Point monitoring occurred along shallow habitat in the new littoral zone (depth light penetrates to allow for photosynthesis) that forms after water depths increase during river mouth closure. Follow up sampling was conducted at every 2 foot rise in water surface elevation following closure of the river mouth. For both sampling objectives, Water Agency staff implemented the field-based rapid periphyton sampling procedure described below.

Methods

Algal Estuary Response and Ambient Monitoring

Transects to monitor and assess periphytic micro- and macro-algal growth were established at four surface water stations selected to represent the range of algal habitats available in the Russian River. One station was retained in the maximum backwater area at Patterson Point (Figure 4.2.1) to continue data collection around the response of benthic algae following estuary closure. At Patterson Point, sampling was done along two, 50-foot transects for estuary response as well as along a single ambient transect established to collect additional baseline data from this location. Ambient algal monitoring for periphytic algae was conducted approximately every two weeks, as well as during river mouth closures at the Patterson Point station, between May 15 to October 15. Similar methods of estimating algal cover and abundance were utilized for both estuary response and ambient algal monitoring.

Estuary Response Monitoring

For closed estuary response monitoring, sampling methodology was developed based on modification of *Standard Operation Procedures for Collecting Stream Algae Samples and Associated Physical Habitat and Chemical Data for Ambient Assessments in California* (Fetscher, et al. 2009). This methodology is intended to address monitoring periphytic algae growth in newly flooded shoreline areas. Transect endpoint 0 was established at a 1 m depth in the main stem Russian River and extended 12.5 m landward or to a 9 foot elevation as diagramed in Figure 4.2.2. Transect locations avoided locations such as tributaries, outfalls, and man-made structures to minimize influence of algal growth from contributions in nutrients, temperature, or canopy cover from such sources.

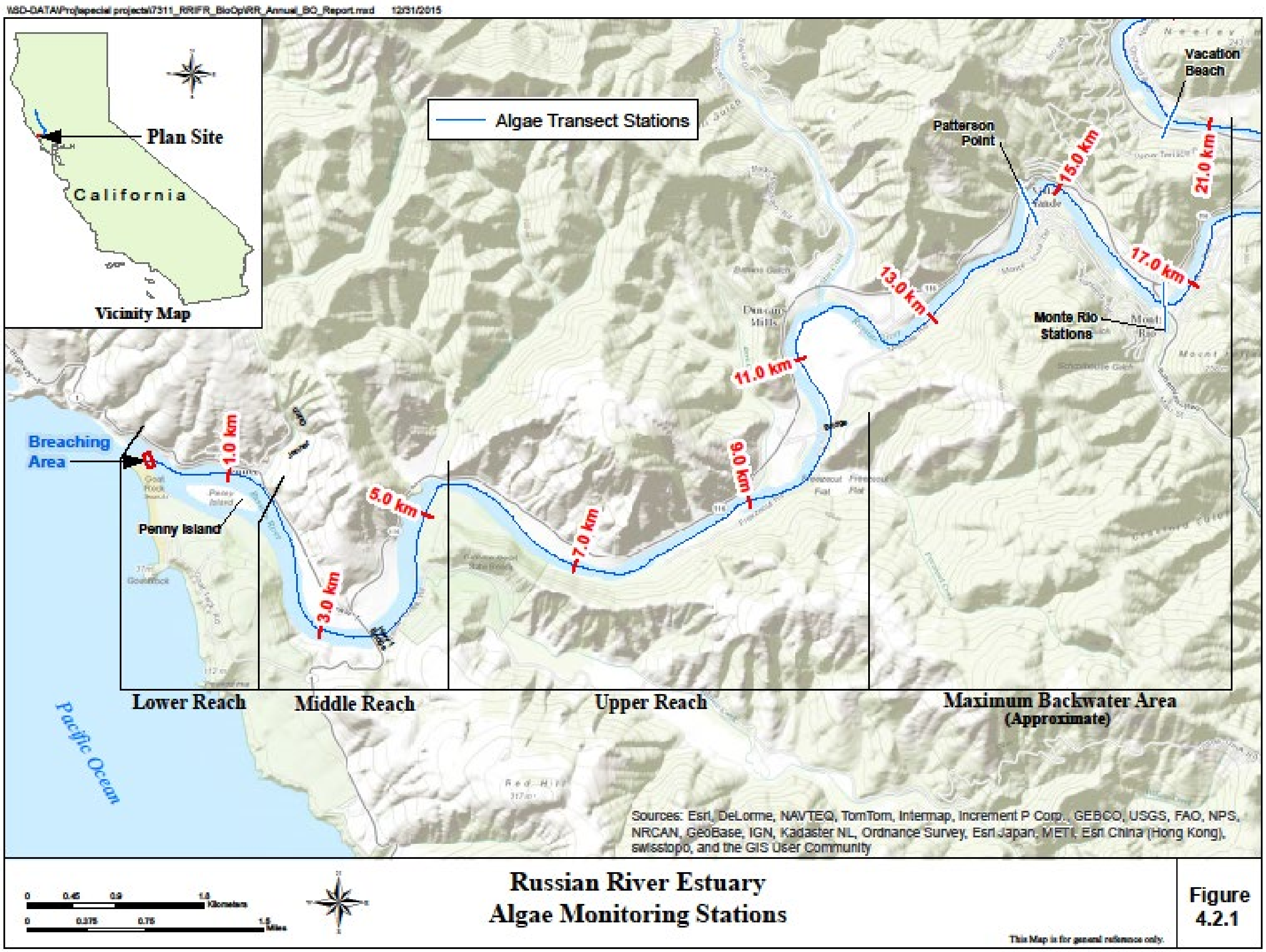

Figure 4.2.1. Russian River Estuary Algal Monitoring Stations.

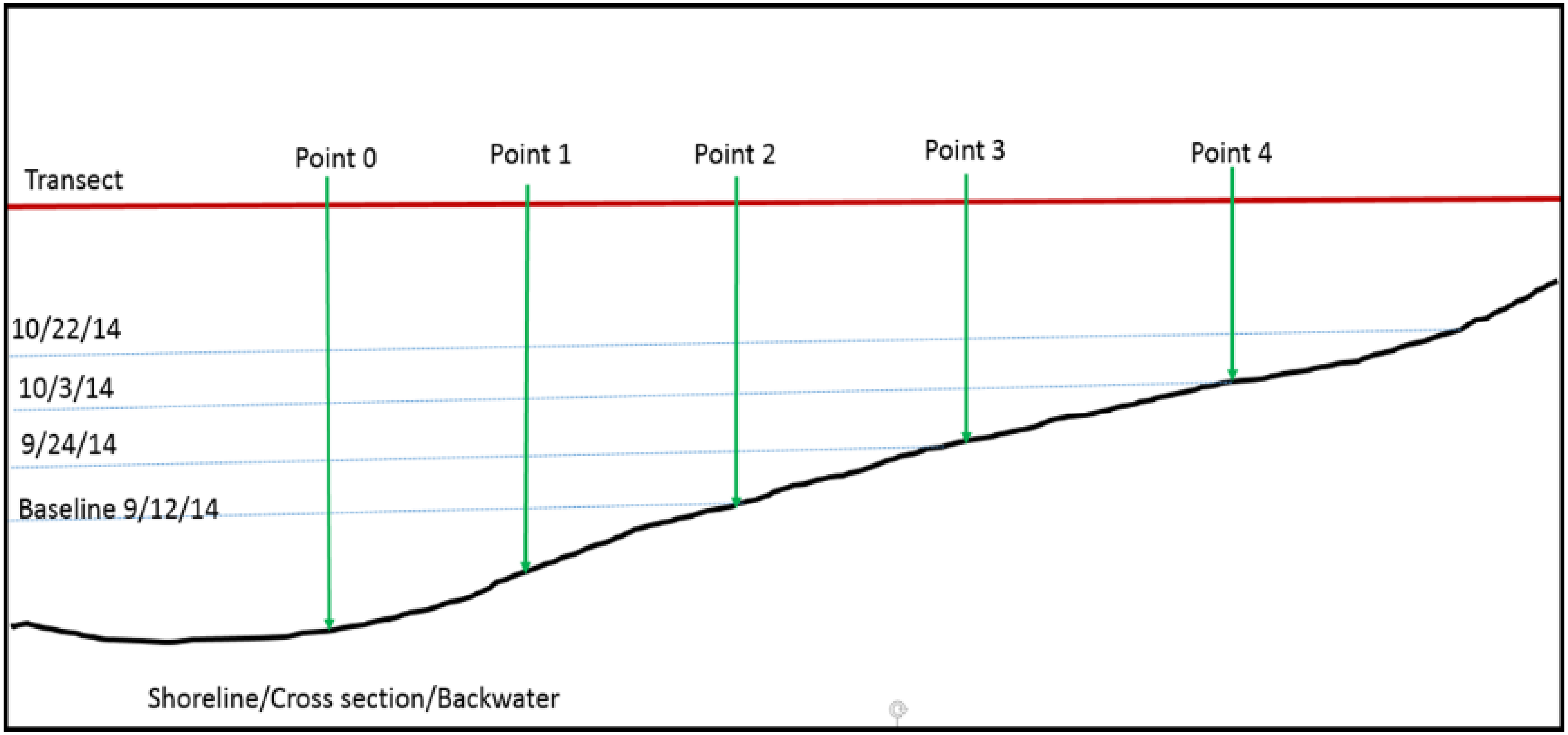

Figure 4.2.2. Transect schematic indicating transect sampling points and a representation of water levels following closure.

Percent algal cover was calculated as an algal indicator of productivity and was measured as algal abundance using a point-intercept collection methodology. Algal cover is the amount of microalgae coating and macroalgae taken at five (5) equidistant points along each transect. The number of points collected by category (in this case the two categories are macroalgae and microalgae) divided by the total number of points collected of every category provides an estimate of percent algal cover.

The presence of algae was recorded for each point along the transect and identified as microalgae or macroalgae. Microalgae is defined as a "film-like coating" of algae. Measurement of microalgae thickness followed the method identified in Fetscher, et al. 2009, and an estimate of film-like coating followed descriptions in Table 4.2.1. Thicker microalgae layers were measured using a ruler or rod with demarcations at 1, 5, and 20 millimeters (mm). The presence or absence of attached macroalgae or unattached, floating macroalgae was recorded at each point.

Table 4.2.1. Microalgal thickness codes and descriptions (from Fetscher, et al. 2009 and adapted from Stevenson and Rollins 2006).

Code	Thickness	Diagnostics
0	No microalgae present	The surface of the substrate feels rough, not slimy.
1	Present, but not visible	The surface of the substrate feels slimy, but the microalgal layers is too thin to be visible.
2	<1mm	Rubbing fingers on the substrate surface produces a brownish tint on them, and scraping the substrate leaves a visible trail, but the microalgal layers is too thin to measure.
3	1-5mm	
4	5-20mm	
5	>20mm	
UD	Cannot determine if a microalgal layer is present	

Ambient Monitoring

For ambient monitoring, transects were located to sample the range of algae habitat available at the sampling locations. Transects were subjectively placed to collect data from areas with different conditions in the littoral zone, including but not limited to depths, velocities, substrates, insolation, and emergent vegetation. One transect was established at each sample station. Existing transects were utilized as feasible. Percent algal cover was calculated as an algal indicator of productivity and was measured as algal abundance using a point-intercept collection methodology similar to the methodology used to evaluate the estuary response.

If bedload had moved significantly compared to prior monitoring, new transects were established. Sampling methodology was developed based on modification of *Standard Operation Procedures for Collecting Stream Algae Samples and Associated Physical Habitat and Chemical Data for Ambient Assessments in California* (Fetscher, et al. 2009) and the *California Watershed Assessment Manual* (Shilling, 2005).

Following data collection along the transect, multi-habitat algae samples were collected from the range of different habitat types (riffles, pools, shade, sun, sand, gravel, cobble) present along the transect. Each sample was collected from the substrate that is uppermost within the stream and has highest possibility of sun exposure (i.e. if a thick layer of macroalgae covers the substrate, collection included the layer). Samples were placed in a cooler to protect the algae from heat and desiccation and to preserve specimen integrity. Algal species present were identified to the lowest taxa, preferably species but at least genera. Successional changes in genera over the season should provide a metric to assess species (genera) richness as well as document the stages in development of the periphyton layer. Frequency of genera encountered will be evaluated as a proxy for abundance (i.e. more frequently encountered in samples equates to more abundant in habitat.

Photographs were taken of the sites to record site conditions at the time of sampling. Photographs were taken to document the morphologies of specific colonies and algal appearance underwater using a submersible digital camera. Oblique photographs at the shoreline were taken to document cyanobacteria (blue green algae) colonies occupying the accumulated drift and other edgeline periphyton. Photographs were also taken along the transects using an underwater photo bucket with a 50 dot matrix grid pattern to assess this methodology for effectiveness and use as an additional monitoring tool.

Samples were evaluated for presence of Chlorophyta (Green Algae), Chrysophyta (Golden Brown Algae (diatoms), and Cyanobacteria. Cyanobacterial target species were identified (including species of *Anabaena*, *Microcystis*, *Planktothrix*, *Oscillatoria, or Phormidium*), monitored for changes in cover successionally over the course of the season, and evaluated for the possibility of the presence of cyanotoxins. In addition, one sample was collected along each transect at a 1 foot depth in the flowing (in active flowing channel) water column using a plankton net (deployed for five minutes) to assess the presence and abundance of phytoplankton. Water chemistry measurements were recorded near the substrate at each monitoring station using a YSI 6600 datasonde and YSI 650MDS datalogger. Conditions measured include water temperature, dissolved oxygen, specific conductance, pH, and turbidity. Water depth was recorded using a stadia rod.

Results

Figures 4.2.3 and 4.2.4 illustrate the relationship and shift in relative cover by micro and macroalgae following estuary closure. Blue green algae cover was sampled as a total estimate along with other forms of microalgae including microscopic green algae and diatoms.

Estuary Closure Algal Response

Observations and cover data from 2014 to 2016 on the effect of estuary closure indicate that following estuary closure and the resulting increase in depth (with the corresponding change in what used to be photosynthetically active littoral zone) there is a shift in the location and composition of the benthic river algae. After spring drawdown and before any estuary closure, typical periphyton establishes in in the littoral zone. Typically these are assemblages composed of micro and macroalgae growing together. Often the dominant green alga is *Cladophora* sp. which is encrusted with single cell and/or tubular or colonial diatoms and cyanobacterial colonies as well as water fungi, bacteria, and detritus. As the water depths change some of the periphytic green algae detach and become planktonic, likely triggering a reproductive phase where numerous spores are produced to start the cycle anew or overwinter. This "drift" (the component of free floating filamentous macroalgae in a system) provides a habitat substrate for microalgae and deposits along shorelines. As the macroalgae starts to settle on shorelines or in aquatic vegetation and decompose it appears to provide an important method for dispersal of the taxa as well as providing significant shoreline habitat for colonizing microalgae, particularly cyanobacteria. As water depths change over colonies of microalgae a similar process unfolds. Microalgae that is motile (diatoms and motile greens) will simply move their cells to a more fortuitous position in the littoral zone with suitable light conditions. Non-motile colonies seem to follow a similar pattern as the filamentous greens. Whole colonies detach and become free floating. Often entraining together in the macro-drift. This shift is associated with all forms of algae and is triggered by environmental change. In this case the environmental change is the increasing water depth and the corresponding shift in the base elevation of the column of water that can be penetrated by sunlight.

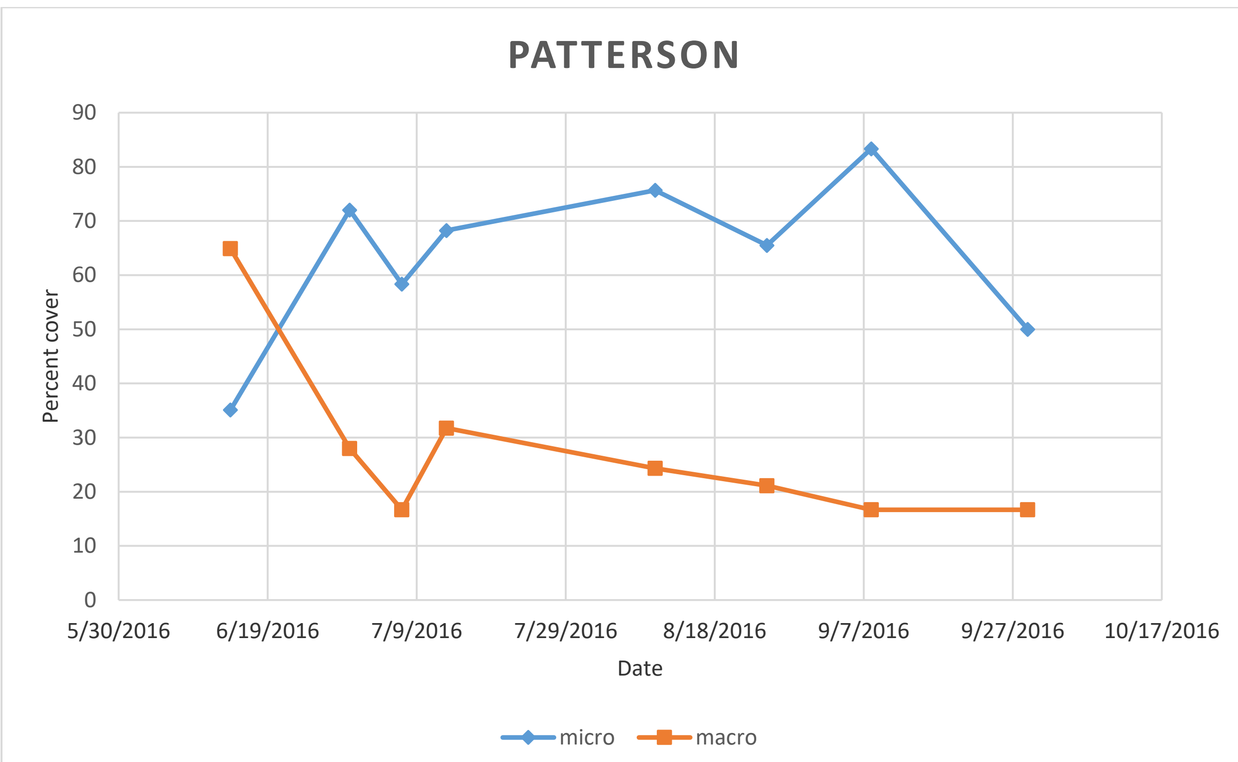

Figure 4.2.3. **Percent cover of micro- and macro-algae at Patterson Point.**

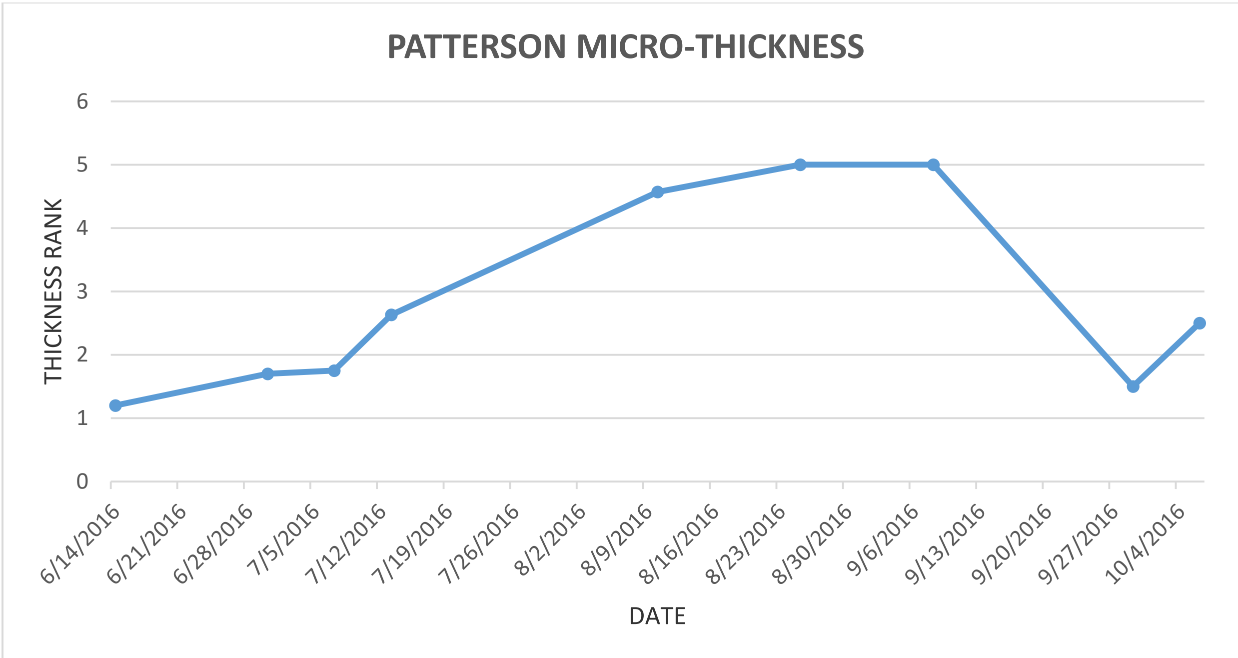

Figure 4.2.4. Micro-Algae thickness at Patterson Point.

Cover Shifts in the Estuary

Observations and data indicate that the shift in cover is triggered by water level increase when the Russian River Estuary mouth closes. Generally the data collected in 2016 is similar to finds in 2014-15. Generally data support the observation that water level rise causes the benthic mats of microalgae to detach from their locations in the littoral zone and through shoreline accumulation of floating colonies (and motile cells) begin to re-colonize the freshly wetted gravel bars, and other newly inundated low-lying areas. Figure 4.2.5 diagrammatically illustrates conditions before closure. Benthic algae is found in the photosynthetically active littoral zone but drops off in abundance quickly below the littoral zone. Figure 4.2.6 illustrates conditions following closure. In most cases, the area of habitat in the littoral zone increases as the water surface elevation increases. The benthic algae and periphyton break away from the substrate and drift onto the shoreline. Motile genera including diatoms start colonizing the new areas but where not observed re-developing into the thick crust present before estuary closure.

Discussion/Observations

Algae occurs in the lower Russian River and Estuary under a variety of conditions and species commonly found worldwide are present in the system. Conditions supporting algal abundance are largely driven by light, temperature, stream flow, and nutrient availability. Generally the most visible type of algae are filamentous Green Algae (Family Chlorophyta) initially growing on rocks and substrate (generally cobble, gravels, and occasionally finer grained sands and silts) (saxicolous) and then becoming planktonic during their reproductive phase, which is driven by largely by season, unless another environmental parameter changes and triggers the life cycle switch (light, temperature, nutrient availability, and changes in water depth). Figure 4.2.7 illustrates a representative cross section of a water body, showing the littoral, limnetic, and profundal zones. The profundal zone is below the area of active photosynthesis, and in the Russian River, generally in areas that exceed 3 feet in depth depending on water clarity. Depending on the annual conformation of the substrate following high flow events, the littoral zone may be larger or smaller depending on where the river moved the substrate during functional flows in the winter.

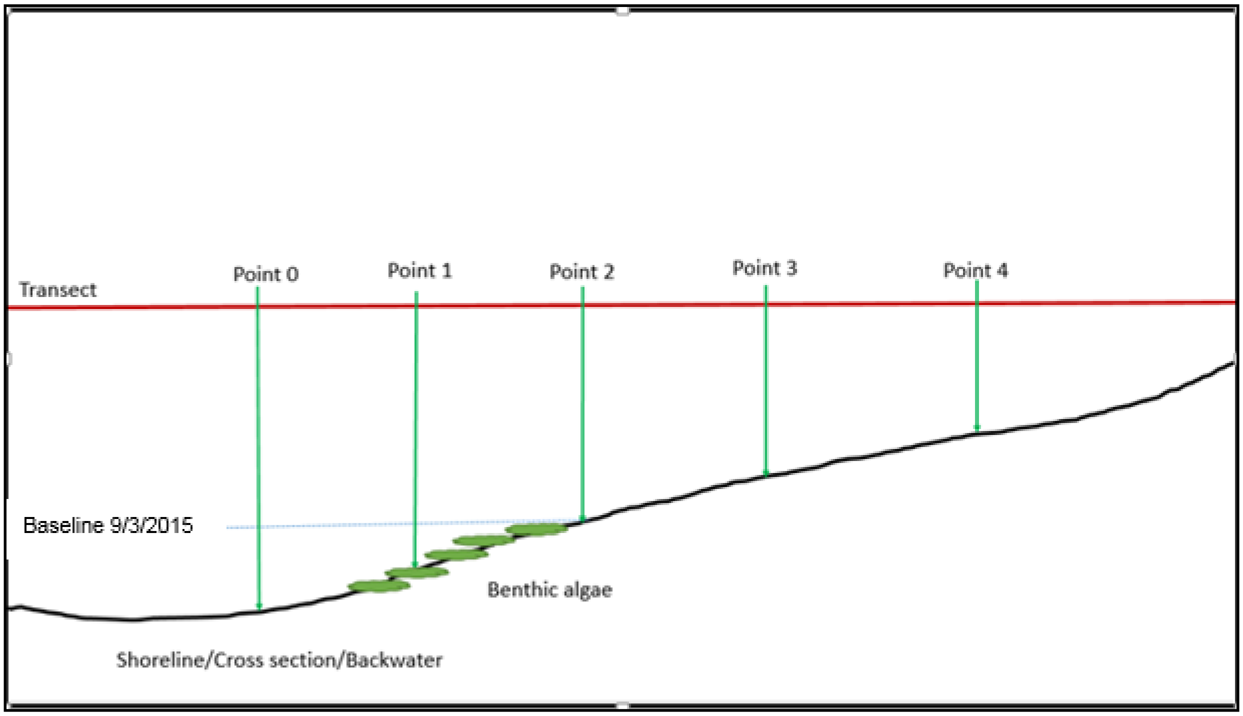

Figure 4.2.5. **Before the estuary closes algae is spread relatively evenly across the littoral zone.**

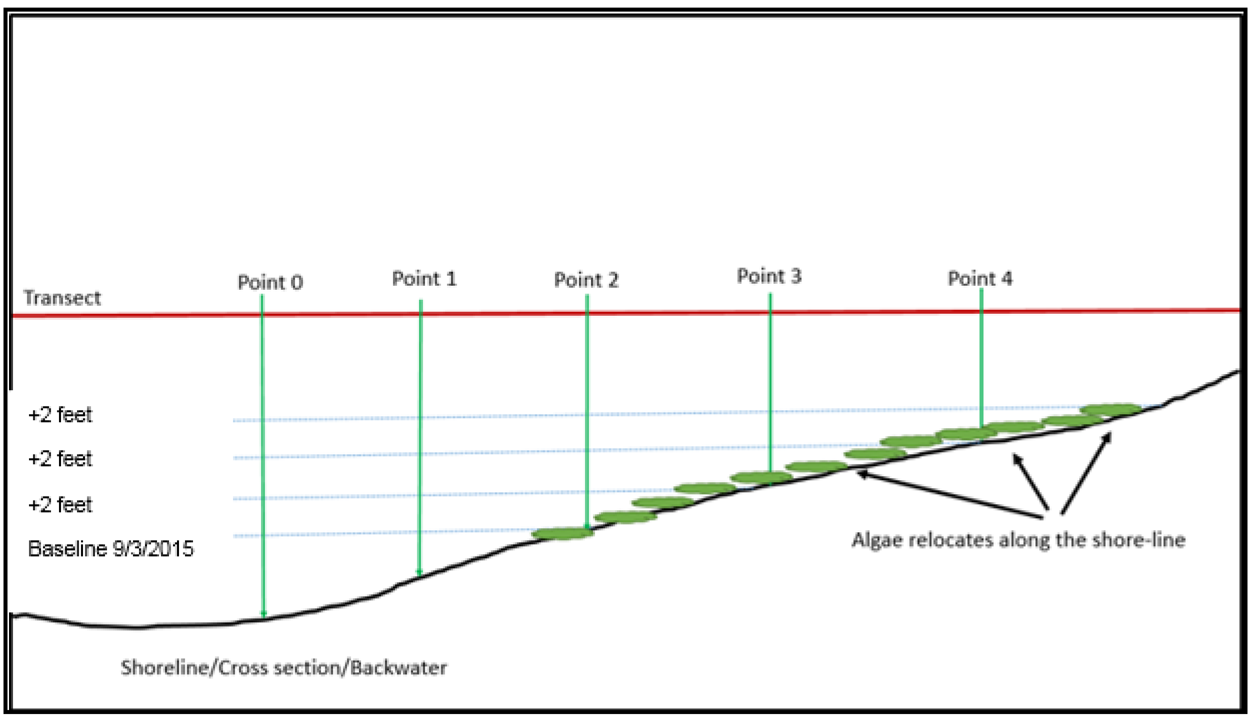

Figure 4.2.6. **After the estuary closes algae moves upslope either by drift or active motility and colonizes the newly wetted littoral zone.**

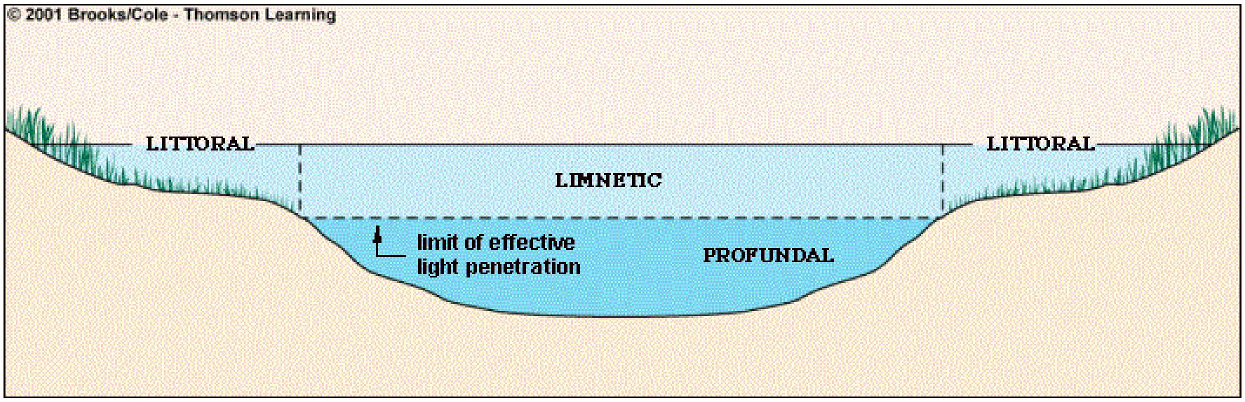

Figure 4.2.7. Diagram indicating littoral vs limnetic and profundal zones. Following river mouth closure, the profundal zone moves into the littoral zone and existing benthic algae either detach or if they have the means, move and re-colonize the newly wetted littoral zone.

Green Algae

Common green algae genera in the Russian River include *Chladophora* sp, *Spirogyra* sp, and *Zygnema* sp. Besides diatoms (described below), Green Algae is one of the most prevalent types of algae recognizably visible at the macro-scale. *Chladophora* is a common branching green alga (often slightly darker green) that grows on rocks and is observed in almost every habitat niche available (cobble, gravel, shallow, fast, deep, slow, shaded, direct sun, etc.) in the littoral zone. The greens and in particular (*Cladophora* sp.) appear to provide the substrate base for the periphyton (complex mixture of algae, detritus, and microbes). Early in the season the filaments are lightly colonized by diatoms and cyanobacterial colonies. Later in the season Cladophora filaments are densely encrusted with free living and tube dwelling diatoms and gelatinous cyanobacterial colonies. Flow also affects what can be retained in the periphyton. Fast water can preclude ultimate stature and size of the periphyton as velocity tends to shear off individual accumulations larger than four to six inches in length. Species diversity comparisons between samples collected in high flow areas indicate that high flows encourage filamentous and colonial forms over free living diatoms. Large substrate (submerged wood, cobble, large gravels, aquatic plants) allows filamentous greens and associated periphyton to reach their maximum sizes. In backwater areas, or locations with sluggish flow at the water edge, the *Chladophora* generally gets completely encrusted in diatoms and cyanobacteria colonies, which may trigger a reproductive cycle.

Green algae start their growth attached to the substrate but if physically disturbed (walking, swimming, rapid flow changes) or when forming reproductive propagules (generally in the Fall) the filaments detach and form large floating and visible rafts (these can negatively affect dissolved oxygen while they are decomposing). Often the green algae or emergent plants provides a substrate for other forms of algae, including diatoms, unicellular greens, and cyanobacteria. Floating mats dominated by green algae were observed to include in varying proportions a wide variety of other algal genera including diatoms, cyanobacteria, and other greens.

Golden Brown Algae

The most numerous and abundant type of algae found in most freshwater systems, and true for the Russian River as well, are diatoms, members of the Golden Brown Algae (Family Chrysophyta). These algae develop siliceous (glass) cell walls called “frustules” and display a wide range of shapes and sizes. Diatoms comprise the majority of the micro-algal crusts and fluffy brown growths found on submerged substrate in the photic zone (littoral). Diatoms have a variety of life styles and can be found as free-swimming (gliding) individuals, colonies of hundreds to thousands cells that form and live together in gelatinous tubes, and in long filaments. They make up a large part of the periphyton and were commonly observed mixed in the “planktonic drift” following river mouth closure. Diatoms are the first algal species along with cyanobacteria that colonize fresh substrate to form biofilms that support algal succession of the periphyton as flows reside and water levels drop in the spring (Bellinger 2015).

Cyanobacteria

Cyanobacteria or “blue green algae” are bacteria that, like plants, use solar energy and carbon dioxide to grow. As bacteria (procaryotes) they lack the complex cellular organization found in eucaryotic cells (nucleus, mitochondria, chloroplasts, endoplasmic reticulum, etc.). Cyanobacteria occur naturally in both freshwater and marine (salt) water bodies. Cyanobacteria are extremely common in the shallow water habitats along the Russian River. Dominant cyanobacterial genera sampled include Anabaena, Gleotrichia, Cylindrospermum, and Ocillatoria (Phormidium).

Toxic cyanobacteria are found worldwide in inland and coastal water environments. At least 46 species have been shown to cause toxic effects in vertebrates (WHO 2003). The most common toxic cyanobacteria in fresh water are Microcystis spp., Cylindrospermopsis raciborskii, Planktothrix (syn. Oscillatoria) rubescens, Synechococcus spp., Planktothrix (syn. Oscillatoria) agardhii, Gloeotrichia spp., Anabaena spp., Lyngbya spp., Aphanizomenon spp., Nostoc spp., some Oscillatoria spp., Schizothrix spp. and Synechocystis spp. Toxicity cannot be excluded for further species and genera (WHO 2003).

Cyanobacteria were observed at all sampling Russian River sites, including Patterson Point, during all sampling events. The years 2014-2016 were low rainfall drought years. During a drought year flows in the mainstem do not reach the sustained level of flow needed to scour periphyton biomass off the substrate. Subsequently, algal response during drought years is expected to be more rapid than a season with significant scouring flows. This response would be expected to be particularly significant with cyanobacteria because of their fast reproductive rates.

Blooms

Algae are photosynthetic microorganisms that are found in most habitats. Algae vary from small, single-celled forms to complex multi-cellular forms. An algal bloom is a rapid increase in the density of algae in an aquatic system. Algal blooms sometimes are natural phenomena, but their frequency, duration and intensity are increased by nutrient pollution. Algae can multiply quickly in waterways with an overabundance of nitrogen and phosphorus, particularly when the water is warm and the weather is calm. This proliferation causes blooms of algae that turn the water noticeably green, although other colors can occur. Some species of algae grow in clumps covered in a gelatinous coating and have the capability to float, allowing cells to stick together into large surface scums in calm weather. Other algae form thick mats that float on or just below the surface along the shoreline. In the lower Russian River, accumulations of algae floating at the surface have been observed to be composed of green algae, cyanobacteria, and diatoms. These “blooms” have been sampled and are composed of discrete aggregates of what used to be attached to the substrate as part of the periphyton (clumps of detritus mixed with whole colonies of different genera of cyanobacteria, green algal reproductive spores, partially decayed filamentous green algal genera, tube dwelling diatoms, and individual trichomes of Oscillatoria or Phormidium, etc.).

Most algae species go planktonic when entering their reproductive phase and can form large floating mats in backwater areas that locally affect dissolved oxygen as the thallus (algal body) disintegrates into propagules (resting spores, aplanospores, akinetes). Stimulus to convert algal metabolism from vegetative to reproductive is tied to light and substrate availability in conjunction with water quality, nutrient availability, and the average life cycle of the species in question. Spring through early fall are the times of year that water bodies typically exhibit the most visible response to water quality problems. Algal blooms can be dramatic and can be a result of excess nutrients from fertilizer, wastewater and storm water runoff, coinciding with lots of sunlight, warm temperatures and shallow, slow-flowing water. The challenge is separating a bloom caused through natural stimuli (reduced insolation from shorter days, increased shading due to inclination of the sun, leading to cooler water temperatures and slower metabolism) from the bloom caused from man-induced stimuli (un-natural fertilizer inputs, stirring up substrate, artificially modifying depth of littoral zone, etc.).

Rivers are not known for having cyanobacterial blooms that are composed of individual cells in the water column. Algal blooms in rivers are generally a result of the benthic genera (periphyton) going planktonic because of an environmental change or the end of the life cycle of a clone. These benthic mats can only grow in clear water where sunlight penetrates to the bottom, and reach their greatest development in locations with high light intensities. During sunny days, especially in the fall, photosynthesis drives oxygen production which forms bubbles in the colony mats that loosen parts of the mats and drives discrete clumps of them to the surface. Mats and broken bits of benthic cyanobacteria colonies wash up on the shore line and can be a hazard if ingested. These mats may be potentially lethal to animals when ingested, depending on the species and if toxins are released. The human impact of benthic cyanobacterial mats is less than from planktonic blooms in the water column, but is worth noting as these kinds of waters, or algae in this form is not generally recognized as producing cyanotoxins (WHO 2003).

Estuary Closure Algal Response

Observations and cover data from 2014 to 2016 on the effect of estuary closure indicate that following estuary closure and the resulting increase in depth (with the corresponding change in what used to be photosynthetically active littoral zone) there is a shift in the location and composition of the benthic river algae. After spring drawdown and before any estuary closure, typical periphyton establishes in in the littoral zone. Typically these are assemblages composed of micro and macroalgae growing together. Often the dominant green alga is *Cladophora* sp. which is encrusted with single cell and/or tubular or colonial diatoms and cyanobacterial colonies as well as water fungi, bacteria, and detritus. As the water depths change some of the periphytic green algae detach and become planktonic, likely triggering a reproductive phase where numerous spores are produced to start the cycle anew or overwinter. This “drift” (the component of free floating filamentous macroalgae in a system) provides a habitat substrate for microalgae and deposits along shorelines. As the macroalgae starts to settle on shorelines or in aquatic vegetation and decompose it appears to provide an important method for dispersal of the taxa as well as providing significant shoreline habitat for colonizing microalgae, particularly cyanobacteria. As water depths change over colonies of microalgae a similar process unfolds. Microalgae that is motile (diatoms and motile greens) will simply move their cells to a more fortuitous position in the littoral zone with suitable light conditions. Non-motile colonies seem to follow a similar pattern as the filamentous greens. Whole colonies detach and become free floating, often entraining together in the macro-drift. This shift is associated with all forms of algae and is triggered by environmental change. In this case the environmental change is the increasing water depth and the corresponding shift in the base elevation of the column of water that can be penetrated by sunlight.

Cover Shifts in the Estuary

Observations and data indicate that the shift in cover is triggered by water level increase when the Russian River Estuary mouth closes. Generally the data collected in 2016 is similar to finds in 2014-2015. Generally data support the observation that water level rise causes the benthic mats of microalgae to detach from their locations in the littoral zone and through shoreline accumulation of floating colonies (and motile cells) begin to re-colonize the freshly wetted gravel bars, and other newly inundated low-lying areas. Figure 4.2.8 is an illustration of conditions before closure. Benthic algae is found in the photosynthetically active littoral zone but drops off in abundance quickly below the littoral zone. Figure 4.2.9 illustrates conditions following closure. In most cases, the area of habitat in the littoral zone increases as the water surface elevation increases. The benthic algae and periphyton break away from the substrate and drift onto the shoreline. Motile genera including diatoms start colonizing the new areas but where not observed re-developing into the thick crust present before estuary closure.

Recommendations

There is a clear response exhibited by periphyton to estuary closure events that was observed and measured during algae sampling/monitoring. However, utilizing a point line intercept method to characterize macroalgae (present or not) has been observed to not accurately sample macroalgae conditions in the river. It is difficult using this method to discern between the smaller filaments of green algae and micro algae crusts and colonies.

Using photographic methods utilizing a 50 dot underwater viewing bucket may have additional merit as this sampling approach will measure lengths of macroalgae directly instead of simply noting presence. Further analysis during the winter and spring would be helpful to understand the shifts in algal cover by genera over the growth season. Studying initial recolonization following spring scour through to fall reproductive blooms would be helpful to better understand both the genera and successional processes involved. Line intercept data was largely unchanged over the season once periphyton established. This method is not likely useful to quantify algal abundance but rather algal substrate preference.

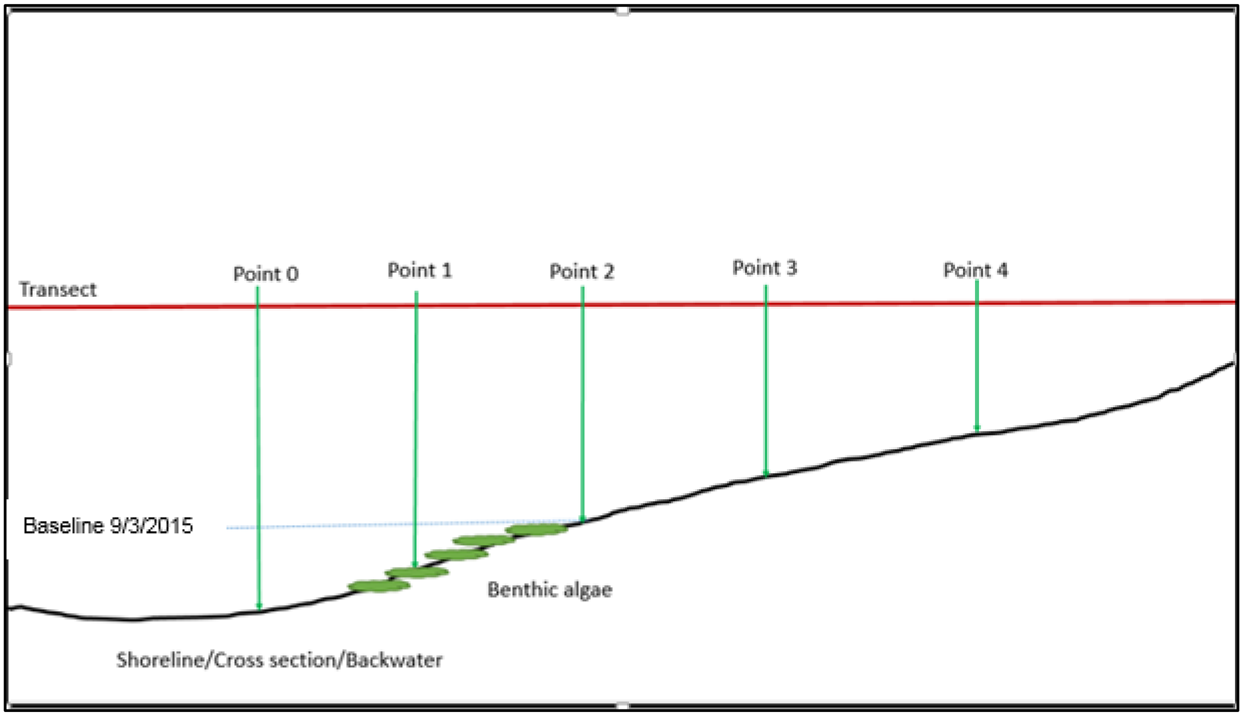

Figure 4.2.8. Before the river mouth closes algae is spread relatively evenly across the littoral zone.

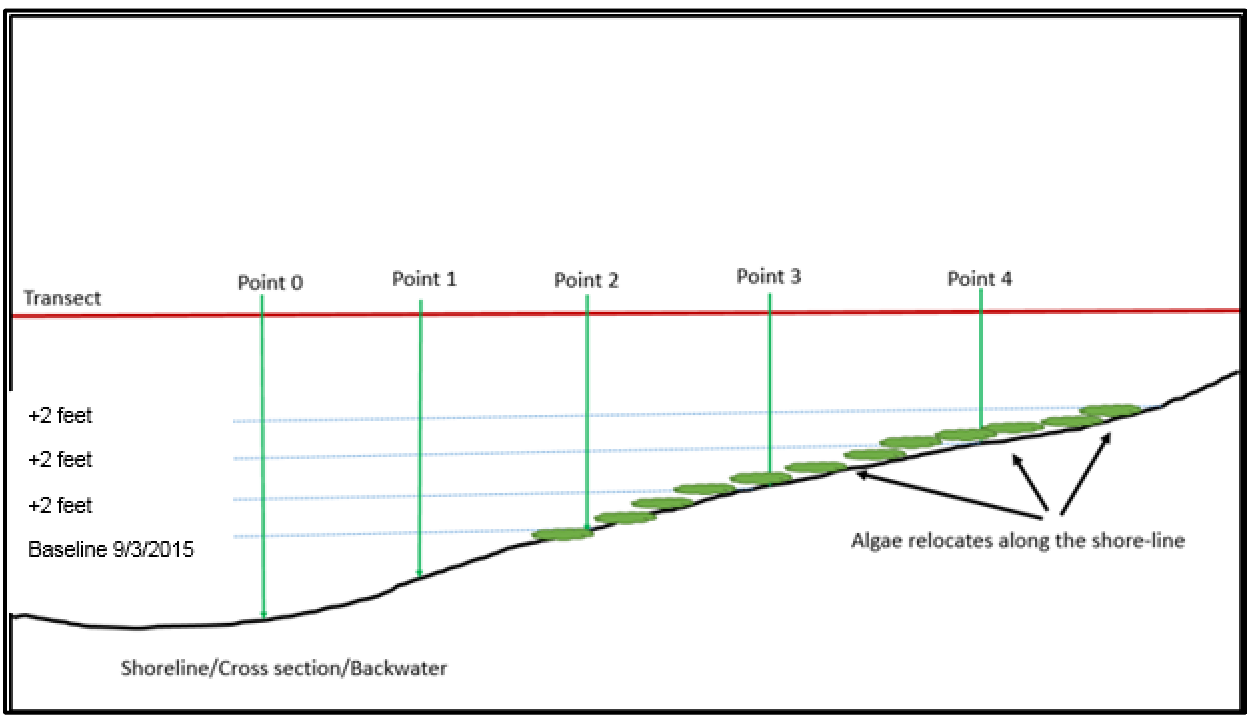

Figure 4.2.9. After the river mouth closes algae moves upslope either by drift or active motility and colonizes the newly wetted littoral zone.

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4.3 Invertebrate Prey Monitoring, Salmonid Diet Analysis and Juvenile Steelhead Behavior

The Russian River Biological Opinion requires the Water Agency to "monitor the effects of alternative water level management scenarios and resulting changes in depths and water quality (primarily salinity, dissolved oxygen concentration, temperature, and pH) on the productivity of invertebrates that would likely serve as the principal forage base of juvenile salmonids in the Russian River Estuary (NMFS 2008). Specifically, the Water Agency is determining the temporal and spatial distribution, composition (species richness and diversity), and relative abundance of potential prey items for juvenile salmonids in the Estuary, and evaluating invertebrate community response to changes in sandbar management strategies, inflow, estuarine water circulation patterns (stratification), and water quality. The monitoring of invertebrate productivity in the Estuary focuses primarily on epibenthic and benthic marine and aquatic arthropods within the classes Crustacea and Insecta, the primary invertebrate taxa that serve as prey for juvenile salmonids, especially steelhead (*Oncorhynhus mykiss*) that may be particularly characteristic of conditions unique to estuarine lagoons for which steelhead may be adapted in intermittent estuaries near the southern region of their distribution (Hayes and Kocik 2014). The monitoring effort will involve systematic sampling and analysis of zooplankton, epibenthic, and benthic invertebrate species" (NMFS 2008, page 254).

Commensurate with assessment of potential responses to Estuary conditions by the macroinvertebrate prey of juvenile salmonids, the Water Agency is also monitoring juvenile salmonid diet composition and behavior. Based on the hypothesis that both diet and behavior of juvenile salmonids will vary as a function of increased water level and rearing space when the mouth of the Estuary is closed, the potentially differential effects of density-dependent interactions on diet composition and consumption rate are being compared between open and closed Estuary conditions. To facilitate the synthesis of this information with more precise information on juvenile salmonid exposure to variability in Estuary salinity and thermal regime, the Water Agency is supporting hydroacoustic telemetry of their position, behavior and residence as a function of Estuary conditions. The purpose of this effort is to determine for juvenile steelhead in the Estuary between June-September the variation under different Estuary open-closure conditions in: (1) the Estuary's water quality environment and the specific water quality conditions experienced by the juvenile steelhead; (2) their behavior in terms of estuarine habitat, reach occupancy and intra-estuarine movement patterns; (3) diet composition; (4) potential (modeled) and empirical growth. These will be used to refine parameters used in the Seghesio (2011) bioenergetics model to generate more empirically-based potential growth estimates during juvenile steelhead response to changing conditions in this intermittent Estuary.

The Water Agency entered into an agreement with the University of Washington, School of Aquatic and Fishery Sciences' Wetland Ecosystem Team (UW-WET) to conduct studies of the ecological response of the Estuary to natural and alternative management actions associated with the opening and closure of the Estuary mouth. This component of the study is designed to evaluate how different natural and managed barrier beach conditions in the Estuary affect juvenile salmon foraging and their potential prey resources over different temporal and spatial scales. Systematic sampling is intended to capture the natural ecological responses (prey composition and consumption rate) of juvenile salmon and availability of their prey resources (insect, benthic and epibenthic macroinvertebrates, zooplankton) under naturally variable, seasonal changes in water level, salinity, temperature and dissolved oxygen conditions. A second approach, event sampling, was originally proposed in 2009 to contrast juvenile salmonid foraging and prey availability changes over Estuary closure and re-opening events. The hydroacoustic telemetry component was particularly adaptable and targeted for the event sampling.

Based on prior data on the foraging of juvenile salmonids in the region's estuaries, the dominant prey of juvenile steelhead can be generally classified as invertebrate organisms that are epibenthic and benthic infauna. All of these prey sources are vulnerable to the variable conditions imposed by river mouth conditions, but taxa composition, relative abundance and production may vary as a function of both longitudinal axis (reach) of the estuary and cross-channel distribution. Another potential invertebrate component, pelagic zooplankton, has not appeared in juvenile salmon diets in either open or closed estuary conditions. Epibenthic, benthic, and zooplankton invertebrate sampling has been conducted monthly from May to October since 2010. Most of these sampling events were completed during open river mouth, tidal conditions in the estuary providing a robust baseline dataset. The composition and abundance of invertebrates was consistent among monthly sampling and among years indicating that the current dataset is adequate to characterize the invertebrate fauna of the estuary. The main gap in data is sampling during prolonged lagoon conditions in the estuary, which is the continuing focus of the on-going research.

Methods

As a result of greater focus on changes in epibenthic and benthic prey availability during estuary closures, the Water Agency- UW-WET invertebrate monitoring protocols were revised in September 2016:

> *Monthly Estuary Surveys :*During years when no prolonged lagoon forms invertebrate surveys will be collected during May, June, and September. Under prolonged lagoon conditions surveys would be conducted monthly from May to October. This sampling schedule would be consistent with the Estuary fish seining schedule. There would be no change in the monthly number of epibenthic, benthic, and zooplankton invertebrate samples collected.
>
> *Mouth Closure Event Surveys:* Monitoring protocols will not change during estuary closure events. Samples would be collected approximately seven and 14 days after a river mouth closure and monthly during prolonged lagoon conditions.
>
> *Lab Processing:* The focus of invertebrate processing in the lab would include the primary steelhead prey taxa (based on years' results, approximately 12-15 taxa). These dominant prey would be sorted and enumerated in epibenthic and benthic samples. Zooplankton are not an important prey group and samples would not be processed. All

invertebrates from epibenthic, benthic, and zooplankton samples would be archived for further analysis if deemed important.

Sampling Sites

Sampling for fish diet and prey availability is designed to coincide with established Water Agency and other related sampling sites distributed in the lower, middle, and upper reaches of the Estuary during the Lagoon Management Period (May 15 to October 15). Since 2009, salmonid diet samples have been coincident with beach seining at 11 primary sites (Figure 4.3.1; modified from Largier and Behrens 2010) sampled for juvenile salmon by the Water Agency – (1) Lower Reach: River Mouth, Penny's Point and Jenner Gulch; (2) Middle Reach: Patty's Rock, Bridgehaven and Willow Creek; and, (3) Upper Reach: Sheephouse Creek, Heron Rookery, Freezeout Bar, Moscow Bridge and Casini Ranch. When possible, samples are specifically selected for diet analysis from the overall beach seine collections at Jenner Gulch to represent the lower Estuary reach, Bridgehaven to represent the middle reach and Casini Ranch, Freezeout Bar and Sheephouse Creek to represent the upper reach. Incidental steelhead diet samples also originated from Penny Point (lower), Willow Creek (middle), and Casini Ranch (upper) sites when there are not sufficient samples from the preferred reach sites. These locations also overlap with sites established by water quality measurements—dissolved oxygen, temperature and salinity.

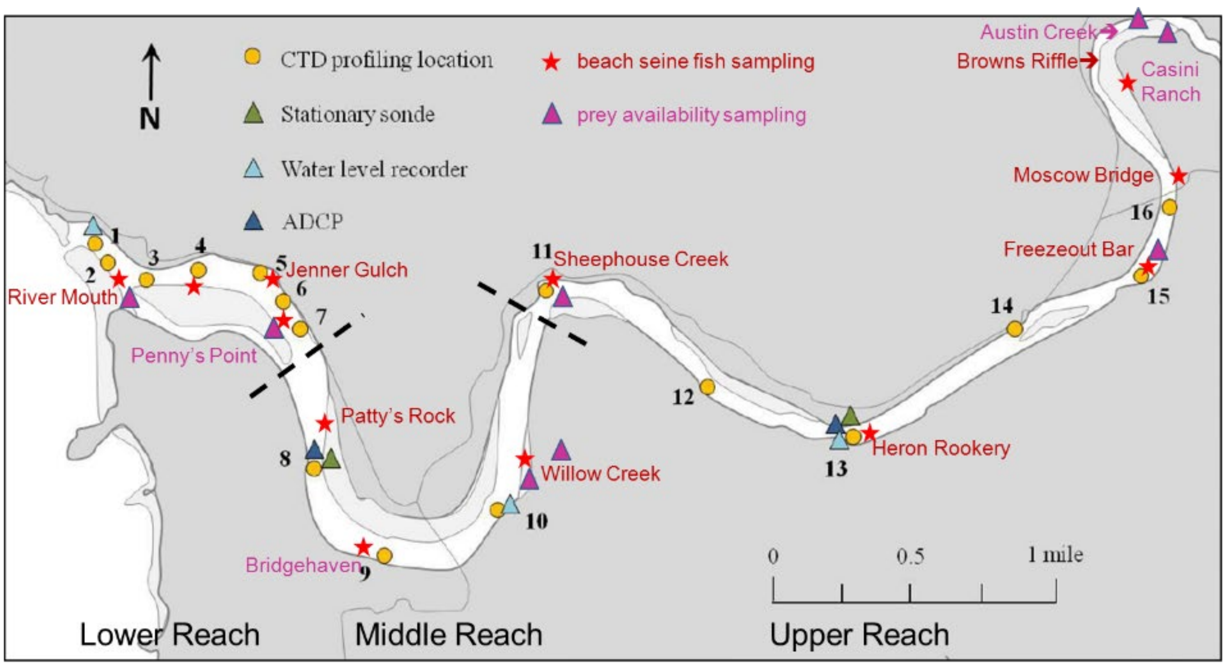

Figure 4.3.1. Locations of sampling stations for juvenile salmon diet (seining location) and prey resource availability (benthic infauna, epibenthos, zooplankton) in three reaches of the Russian River Estuary.

Prey resource availability sampling occurs at four sites distributed through the three estuarine reaches (Figure 4.3.1): Lower Reach—River Mouth and Penny Point; Middle Reach—Willow Creek; and Upper Reach—Freezeout Bar. Each of the sites includes three, lateral transects across the Estuary over which four sampling methods were deployed to sample availability of juvenile steelhead prey (Figures 4.3.2 – 4.3.7 for more specific locations by different sampling methods).

Juvenile Salmon Diet Composition

Systematic sampling of the diets of five or more ($n \geq 5$) juvenile steelhead ≥55 mm FL are derived, when available, from the beach seine sampling during the lagoon management period between May 15 and October 15. All fish designated for diet analysis are handled, gastric lavaged and released according to the University of Washington animal care protocols. If resources are available and sample sizes are less than five individual fish (n=<5) during systematic sampling, event sampling around scheduled beach management at the barrier beach are coordinated with Water Agency fisheries monitoring and physical measurements of estuarine response.

Stomach lavage follows Foster (1977) and Light et al (1983). Diet contents are preserved in 10% Formalin for later laboratory processing. As per Water Agency fisheries protocols, fork lengths and weights are taken from each fish. Each fish is scanned for a passive integrated transponder (PIT) tag and tagged if no previous PIT tag was detected.

Prey Resource Availability

Benthic infauna and epibenthos prey resource sampling were conducted once per month in the lagoon management period during open, tidal (baseline) conditions. If barrier beach conditions result in a closure, epibenthos and benthic infauna are sampled seven and 14 days after closure. Following an extended closure of 14 days or more, prey resource availability sampling of benthic infauna, epibenthos, and zooplankton will begin at day 14 and continue every three weeks after until the Estuary opens. Under Estuary conditions in 2016, a total of 369 individual samples were collected (Table 4.3.1).

Benthic Infauna

Replicate core samples (0.0024-m^2 PVC core inserted 10 cm in to the sediment) are taken at each transect of each site. The location of each core sample is consistent with each epibenthic sled and epibenthic net to shore sample, but no core samples are taken in between transects. This sample is repeated four times per transect (twelve times per site). Additional samples would be added along the transect with increasing water level (inundation of the shoreline) during closure or outlet channel implementation. The sediment cores are preserved in 10% buffered Formalin for laboratory analysis. During 2016, 144 benthic cores were acquired (Table 4.3.1).

Figure 4.3.2. Distribution of juvenile salmonid prey resource availability in three reaches of the Russian River Estuary.

Figure 4.3.3. Distribution of juvenile salmonid prey availability sampling transects and techniques at the River Mouth site in the Russian River Estuary.

Figure 4.3.4. Distribution of juvenile salmonid prey availability sampling transects and techniques at the Penny Point site in the Russian River Estuary.

Figure 4.3.5. Distribution of juvenile salmonid prey availability sampling transects and techniques at the Willow Creek site in the Russian River Estuary.

Figure 4.3.6. Distribution of juvenile salmonid prey availability sampling transects and techniques at the Freezeout Bar site in the Russian River Estuary.

Figure 4.3.7. Modification of sampling techniques during closed conditions for distribution of juvenile salmonid prey availability sampling transects and techniques at Willow Creek site in the Russian River estuary. The grey area is the inundation of area during closed conditions.

Table 4.3.1. Prey resource availability samples collected in 2016, Russian River Estuary.

Date	Mouth Condition	Jenner Gage Water Level (ft) (10am-2pm)	Benthic Core	Sled Channel	Epibenthic Net to Shore	Zooplankton Net
River Mouth						
5/18/2016	OPEN	1.8-2.0	12	9	5	3
6/23/2016	CLOSED (8th day of closure)	6.5	12	9	5	3
9/26/2016	CLOSED (16th day of closure)	7.5	12	12	5	3
Penny Point						
5/18/2016	OPEN	1.8-2.0	12	9	5	3
6/23/2016	CLOSED (8th day of closure)	6.5	12	12	5	3
9/26/2016	CLOSED (16th day of closure)	7.5	12	12	5	3
Willow Creek						
5/18/2016	OPEN	1.8-2.0	12	9	5	3
6/23/2016	CLOSED (8th day of closure)	6.5	12	12	5	3
9/26/2016	CLOSED (16th day of closure)	7.5	12	12	5	3
Freezeout Bar						
5/18/2016	OPEN	1.8-2.0	12	9	5	3
6/23/2016	CLOSED (8th day of closure)	6.5	12	12	5	3
9/26/2016	CLOSED (16th day of closure)	7.5	12	12	5	3
Subtotal by sample type			**144**	**129**	**60**	**36**
Total Number of Samples						**369**

Epibenthos

Epibenthic organisms at the sediment-water interface are sampled with two methods: 1) epibenthic net (net to shore); and, 2) epibenthic (channel) sled. The epibenthic net is a 0.5-m x 0.25-m rectangular net, equipped with 106-µm Nitex mesh that is designed to ride along the surface of the Estuary bottom substrate. It is deployed 10 m from shore and then pulled along the bottom perpendicular back to shore by an individual onshore. This is replicated five times per site (once at each transect and then once between Transects 1 and 2 and also between Transects 2 and 3). The epibenthic sled is equipped with a 0.125-m^2 opening, 1-m long 500-µm Nitex mesh net towed behind the boat against the current. The sled is dropped off of the bow of the boat and allowed to sink to the bottom. Once the boat has finished towing the sled (in reverse) 10 m against the current, it will be retrieved back onto the boat. This is replicated five times per site (once at each transect and then once between Transects 1 and 2 and also between Transects 2 and 3). The sled is used to obtain three samples per transect (nine per site under open conditions). Additional samples would be added along the shoreward margin of the transect with increasing water level (inundation of the shoreline) during closure or outlet channel implementation. Captured organisms are preserved in 10% buffered Formalin for laboratory analysis. During the 2016 study period, 60 epibenthic net shore and 129 epibenthic sled samples were acquired.

Zooplankton

Zooplankton are sampled at the same location as water quality (the deepest available depth per site) using a 0.33-m diameter ring net, 73-µm Nitex mesh and cod end cup. Replicated (n=3) vertical water column hauls are made by lowering the zooplankton net until the top ring of the net is just above the benthos and then pulled by hand vertically to the surface to obtain a sample of the entire water column. This sample set is repeated three times per site. Captured organisms are preserved in 10% buffered Formalin for laboratory analysis. During 2016, 36 zooplankton samples were acquired; as described in the revised monitoring methods above, these samples were archived instead of processed in the UW-WET laboratory.

Sample Processing and Analyses

Stomach contents from juvenile salmon are identified to the species level if possible under a dissecting microscope. Invertebrates found in the diets of steelhead and collected in the prey resource samples are identified to species level, except for insects which are identified to family level. Any invertebrate collected during prey sampling and not found to be part of the steelhead diet is identified to order or family level. Each of the identified prey taxa are counted (for numerical composition) and weighed (for gravimetric [biomass] composition) and the frequency of occurrence. The state of total stomach content biomass is normalized by individual fish weight to provide an additional index of relative consumption rate (“instantaneous” ration), which is the total biomass of prey found in individual fish stomach contents relative to the biomass of the fish expressed as $g\ g^{-1}$. It is recognized that this is only a short-term index of consumption, and will vary by fish size, time of day and other factors influencing foraging behavior. If fish are captured under the same general conditions, this index can provide an indication of differences in feeding performance. Under some conditions, the instantaneous ration can be used to develop an estimate of daily ration that can be used in bioenergetic modeling of potential growth.

In addition to individual metrics of diet composition, the Index of Relative Importance (IRI; Pinkas et al. 1971) is also calculated, wherein %Total IRI for each discrete prey taxa takes into account the proportion that prey taxa constitutes of the total number and biomass of prey and the frequency of occurrence of that taxa among in the total number of fish stomach samples:

$$IRI_i = FO_i*[NC_i + GC_i]$$

where NC is the percent numerical composition, GC is the percent gravimetric (biomass) contribution, FO is the percent frequency of occurrence for each of the prey taxa, and *i* is the prey taxa; results are expressed as a percentage of the total IRI for all prey items. We also interpret diet composition using just GC_i in order to better represent the bioenergetic contribution of prominent (from a FO_i standpoint) prey.

In accordance with a more recent revision of the IRI index, we calculated the Prey-Specific Index of Relative Importance (PSIRI) which substitutes NC and GC with their corresponding prey-specific abundances, %PNC and %PGC:

$$PSIRI_i = FO_i*[\%PNC_i + \%PGC_i]$$

PSIRI sums to 200% and therefore diving by 2 results in a version of the standardized %IRI (Amundsen et al. 1996; Cortės 1997), with an important distinction: the PSIRI is additive with respect to taxonomic levels, such that the sum of PSIRI for species will be equal to the PSIRI of the family containing those species.

Prey availability data are standardized to density per area or volume, i.e., m^2 for benthos and epibenthos and m^3 for zooplankton. Prior to analysis, density data are square root transformed to better equate group variances and compress positively skewed distributions to a more nearly normal distribution.

Multivariate analyses are also utilized to organize fish diet sample compositions and prey availability samples into statistically distinct groupings. Statistical analyses are performed using the PRIMER v6.0 multivariate statistics analysis package (Clarke and Gorley 2006) or the R 3.1.1 Vegan package (Oksanen et al., 2011). The primary analyses included non-metric multidimensional scaling (NMDS) and associated analyses of similarities (ANOSIM) and similarity percentages (SIMPER) of factors (in this case, organism taxa) that account for the similarity. Similarity is based on the Bray-Curtis similarity coefficient. The primary ANOSIM statistic for differences between groups is the Global R, which varies between 0 (no significant difference) to 1 (maximum difference). These analytical tools, and the PRIMER package in particular, are used extensively in applied ecology and other scientific inquiries where the degree of similarity in organization of multivariate data (e.g., species, ecosystem attributes) is of interest.

Results

Estuary Conditions

The Russian River Estuary experienced five mouth closures in 2016 during the lagoon management period (Figure 4.3.8). However, there were no opportunities for comparisons of sampling events between open and closed conditions due to the temporal spacing of the sampling events. The first sampling event took place during open estuary conditions on May 18, 2016, while the two subsequent sampling events took place during mouth closures. The second sampling event, June 23, occurred 8 days into the second closure with water levels at 6.5 feet and the third sampling event, September 26, occurred 16 days into the closure with water levels at 7.5 feet. These two estuary closure events both offer between a week and two weeks of water level inundation, so will provide some indication of macroinvertebrate response to the higher water levels. However, there are no samples during open estuary baseline conditions soon before or after these closure periods.

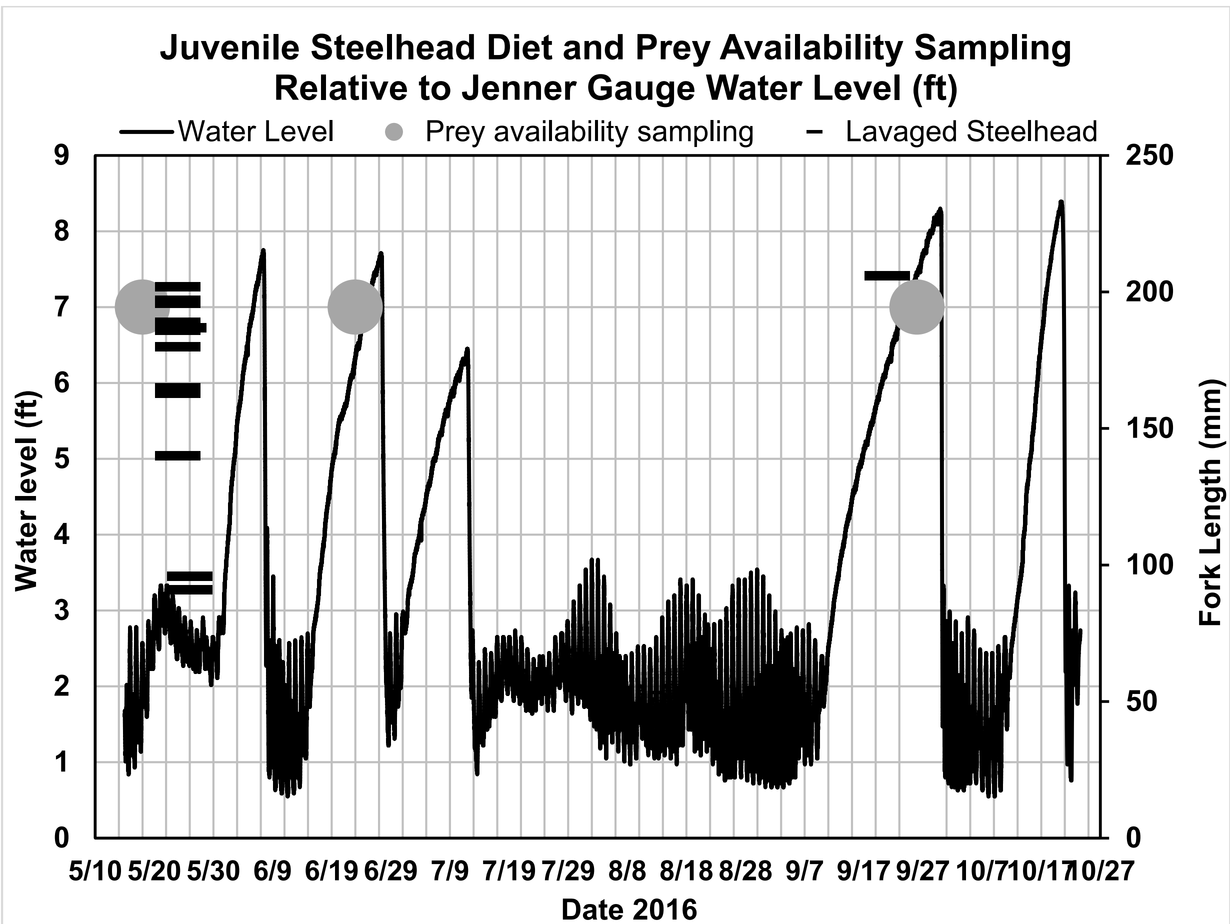

Figure 4.3.8. Occurrence of juvenile steelhead diets (lavage; with size, mm FL) and macroinvertebrate prey sampling relative to Jenner Gage water level (ft) at mouth of Russian River estuary, May 10-October 27, 2016.

Juvenile Steelhead Diet Composition

In 2016, 13 juvenile steelhead were sampled for diet composition. Twelve of the steelhead (91-202 mm FL) were sampled between May 24 and May 26 during open conditions and one steelhead (206 FL) during the fourth closure on September 21.

Prey Availability

Samples collected during the 2016 lagoon management period analyzed by UW-WET were prioritized for extended closed conditions. Benthic samples from 2016 are pending to be processed, June 23 (closed, 6.5 ft) and September 26 (closed, 7.5 ft); channel epibenthic sled and epibenthic net samples included the June and September dates while the May samples will be prioritized for overlap with the lavage samples; and, zooplankton samples are being archived.

Epibenthic Net to Shore

As described in methods above, the epibenthic net to shore sampling was completed within 10 m of the high water level and could be indicative of an expansion or shift in prey organism distribution as a function of estuary water level and volume. As water elevation rises above 2.1 ft (Jenner Gage) during a closure event, the epibenthic net to shore samples organisms expand (numerical response) or migrate (distributional response) into the recently inundated shallow water margin. However, because the epibenthic net to shore sampling does not repeatedly sample the same transect space, these data cannot detect whether the macroinvertebrates response is numerical or distributional.

In this littoral-edge habitat, the dominant macroinvertebrates during the June 23 sampling consisted of corixid (waterboatmen) beetles at Freezeout Bar with mean densities of individuals of up to 1600 m^{-2}, as well as gastropoda snails at Penny Point with mean densities of individuals of ~600 m^{-2} (Figure 4.3.9). Similar to the June 23 sampling, the September 26 Penny Point station identified gastropoda snails as a dominate taxa, with the Willow Creek samples having similar densities of ~800 individual m^{-2} (Figure 4.3.10). The mean densities of corixid beetles decreased in the September 26 samples at Freezeout Bar to less than ~100 individuals m^{-2}. Although several orders of magnitude less dense (e.g., 4-20 organisms m^{-2}), other typical prey, such as amphipods (*Americorophium spinicorne*, *Eogammarus confervicolus*), isopods (*Gnorimosphaeroma insulare*), mysids (*Neomysis mercedis*) and ephemeropteran (mayfly) nymphs were also prominent in the middle and upper reaches on June 23 (Figure 4.3.9). Only mysids occurred in any density at the River Mouth station. By the September 26 closure sampling, similarly modest densities (4-10 m-*2*) of the amphipods (*A. spinicorne, A. stimpsoni, E. confervicolus*) and isopods occurred predominantly at the Penny Point and Willow Creek stations (Figure 4.3.10). Chironomid larvae, which had only occurred at Freezeout Bar in June, were only prevalent at Willow Creek. The composition, richness and relative abundance of these selected prey taxa were comparable to epibenthic net to shore samples from late September 2014, when the estuary was closed at 4.2 ft water elevation.

Multivariate analysis of the taxa density composition among the four sites over the two sampling events during the estuary closure (Figure 4.3.11; 2D stress=0.181) indicated no significant difference between dates ($R = 0.10$) but a difference between sites ($R = 0.64$). Freezeout Bar and River Mouth are most dissimilar while Penny Point and Willow Creek had the most overlap (Figure 4.3.11).

Epibenthic Sled

Samples from the epibenthic sled distinguish potential macroinvertebrate prey availability in two respects: 1) the sled samples deeper habitats parallel to the thalweg; and 2) during prolonged closures, additional sled samples are added where newly inundated intertidal areas are available to foraging steelhead. The additional samples taken during the June 23 sampling excluded the River Mouth station. Additional samples were taken at all four stations during the September 26 sampling event.

Epibenthic sled samples from the June 23, 2016, sampling indicted similar general prey taxa distribution and densities as documented in the epibenthic net to shore with the exception of increased occurrence of the mysid *Neomysis mercedis* and corixid beetles, and greater overall abundances at Freezeout Bar, in the uppermost reach (Figure 4.3.12). In addition to *Neomysis mercedis* occuring at mean densities of between ~120 and 44 m^2 at River Mouth and Penny Point respectivally, they were found at high mean densities in the additional Freezeout Bar samples (190±397 m^2). Similar to the June epibenthic net to shore samples, the June 23 epibenthic sled samplies were dominated by gastropoda snails and corixid beetles. The gastropoda snails were primarily found in the additional samples from Willow Creek and Penny Point, in addition to the standard samples at River Mouth. The corixid beetles were found at relatively high densities in both the regular and additional samples of Freezeout Bar with mean densities of ~1200-1700 m^2.

Prey taxa that occurred in comparatively low densities and fewer stations in the epibenthic net to shore sampling were an order of magnitude more dense and more widely distributed in the epibenthic net samples. In particular, amphipods (*Americorophium* spp., *E. confervicolus*), isopods (*G. insulare*) and ephemeropteran nymphs were often prominent at all stations. Compared to the epibenthic net, sled densities of prey were more dense at the River Mouth, where *Americorophium* spp. occurred at 177 m^{-2}, *E. confervicolus* at 589 m^{-2}, *N. mercedis* at 126 m^{-2}, while polychaetes were found at relatively higher densities (~190 m^{-2}) in the Freezeout Bar additional samples.

Overall, where they co-occurred in both the regular epibenthic sled transects and the additional transect in the recently inundanted littoral zone, densities were equal or higher in the inundated shallows for specific taxa (14 occurrences; e.g., ephemeropteran numphs, corixid nymphs and adults, chironomids, isopods and fish) or less dense for others (6 occurrences; e.g., amphipods). This would suggest that the former types of macroinvertebrates may have uniformly redistributed their populations or experienced directed immigration into the newly inundated shallows. The latter prey taxa may have continued to sustain their populations in the deeper habitats.

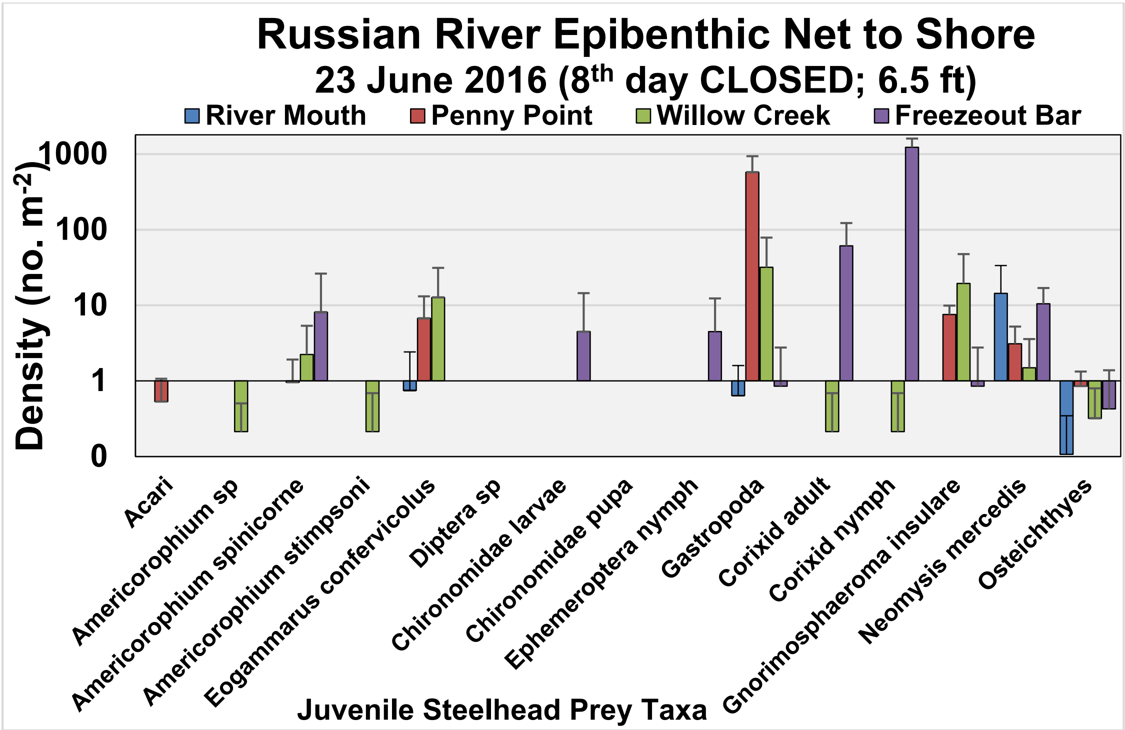

Figure 4.3.9. Density of epibenthic macroinvertebrates documented as juvenile steelhead prey from epibenthic net to shore sampling at four sites in the Russian River estuary, June 23, 2016. Note logarithmic scaling of density.

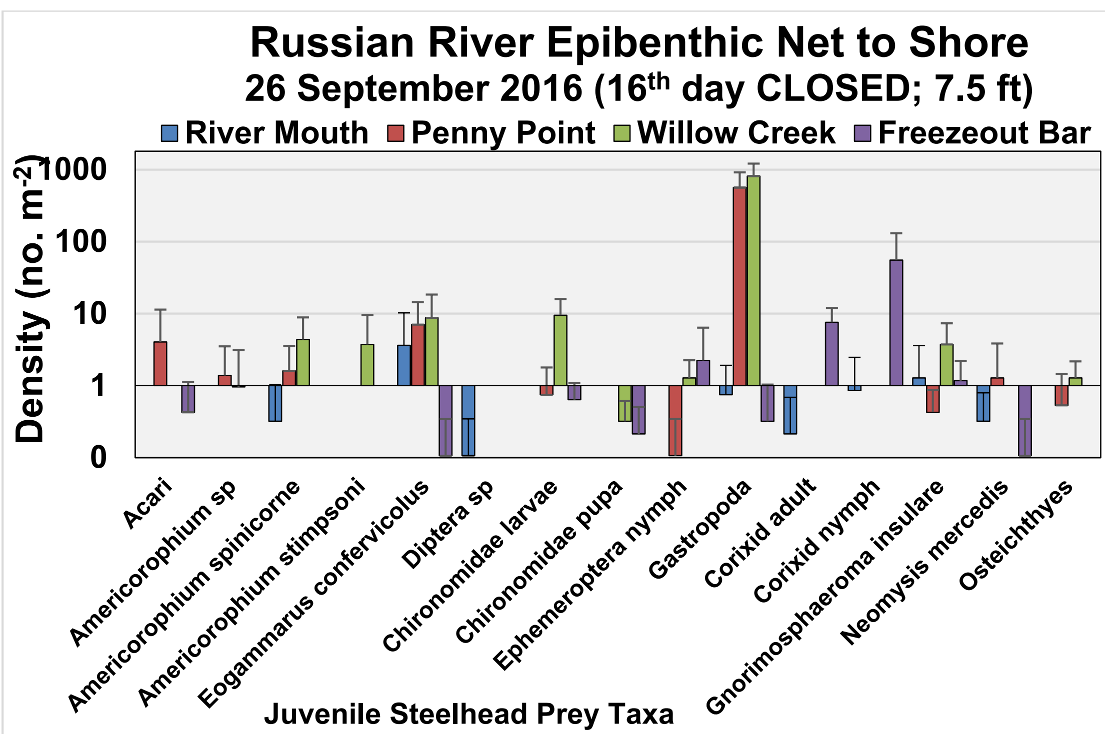

Figure 4.3.10. Density of epibenthic macroinvertebrates documented as juvenile steelhead prey from epibenthic net to shore sampling at four sites in the Russian River estuary, September 26, 2016. Note logarithmic scaling of density.

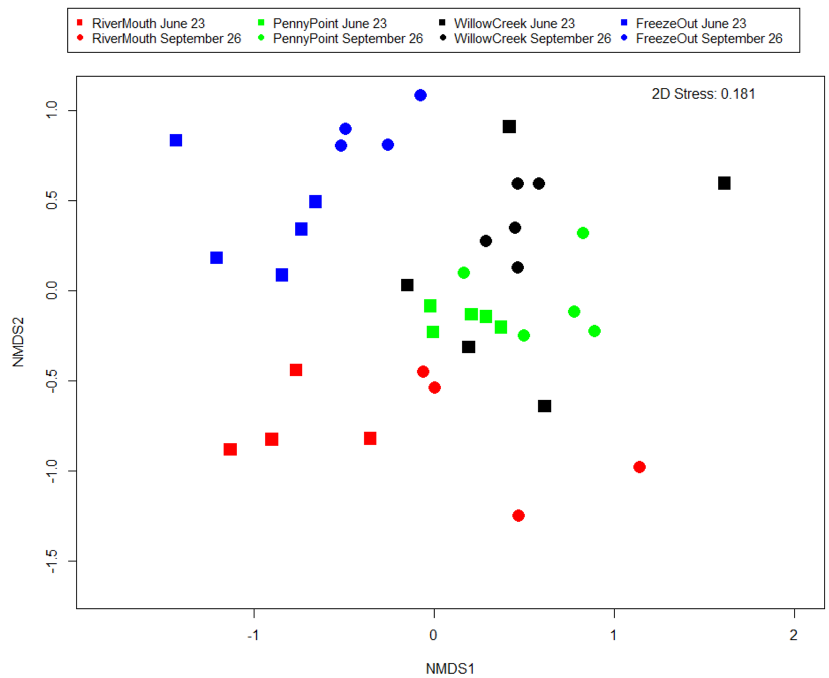

Figure 4.3.11. Multivariate analysis (NMDS) diagram of density composition of epibenthic net macroinvertebrate prey of juvenile steelhead in lower, middle and upper reaches of the Russian River estuary, 2016.

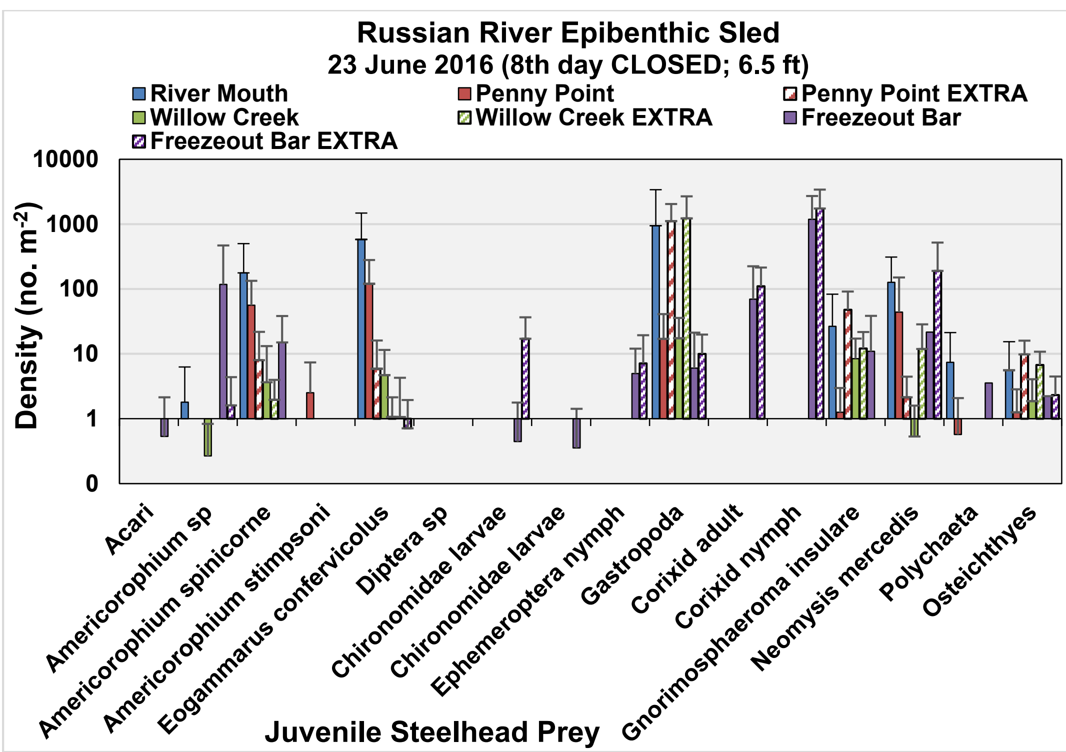

Figure 4.3.12. Density of epibenthic macroinvertebrates documented as juvenile steelhead prey from epibenthic sled sampling at four sites in the Russian River estuary, June 23, 2016. Note logarithmic scaling of density.

The September 26 epibenthic sled sampling found the highest overall densities of gastropod snails overall, and specifically at the additional Willow Creek station where greater than a mean of 5,000 m^{-2} occurred. Gastropod snails were also found in the River Mouth sample and additional samples (~200-1,000 m^{-2}). Most of the other prey taxa were broadly represented in all reaches but a slightly lower densities, many,10 organisms m^{-2}. Similar to the epibenthic net to shore samples, corixid nymphs were particularly dense at Freezeout Bar low densities at the other sites. All the amphipod taxa were somewhat uniformly dense at the lower to mid-estuary stations at River Mouth, Penny Point, and Willow Creek, but often <1 m^{-2} in the upper reach at Freezeout Bay.

Perhaps reflecting the higher water elevation (7.5 ft, as compared to 6.5 ft) and the twice longer period of rising water levels (16 d, as compared to 8 d) the densities of a majority of prey taxa sampled by the epibenthic sled were higher in the littoral zone (extra transect) on September 26 (Figure 4.3.13) than on June 23 (Figure 4.3.12). This was particularly the case for all the amphipod taxa and gastropods at Penny Point, which often illustrated densities an order of magnitude higher in the littoral shallows than along the deeper transects. Overall, densities were measurably higher along 18 of the inundated shallow littoral transects than in the deeper transects, compared to lower densities in 11 cases. Lesser occupation of the inundated littoral shallows was most notable among the River Mouth transects, even among the amphipod taxa.

Multivariate analysis (NMDS) of the taxa density composition among epibenthic sled samping stations in the four sites over the two sampling events (Figure 4.3.14; 2D stress=0.23) indicated no significant difference between dates ($R = 0.096$) and between the standard and additional samples ($R= 0.145$), and only a minimal difference among sites ($R = 0.268$). The NMDS (Figure 4.3.14) illustrates that the source of the main difference was the distinction of Freezeout Bar from the rest of the sites, likely from the presence of corixid beetles and the lack of many other microbenthic prey items.

Benthic Infauna

Among the prevalent prey of juvenile steelhead, the motile amphipod *Eogammarus confervicolus*, tubicolous amphipods *Americorophium* spp., and gastropod snails were most abundant on June 23, earlier in the season (Figure 4.3.15). Willow Creek was the location with consistently highest densities of *Americorophium* spp., with mean densities up to 11,000 individuals m^{-2}. Penny Point also had relatively high densities of A*mericorophium* spp. (~7,000 m^{-2}), but the highest mean densities of *Eogammarus confervicolus* (~6,600 m^{-2}), and gastropod snails (~10,000 m^{-2}). In addition, *Americorophium* spp. were found at mean densities as high as ~7,700 m^{-2} at Freezeout Bar. Common benthic macroinvertebrates at the River Mouth includes *Americorophium* spp. (~1,400 m^{-2} – 3,000 m^{-2}), *Eogammarus confervicolus* (~4,700 m^{-2}), gastropod snails (~1,000 m^{-2}) and polychaeta (~1,000 m^{-2}).

In comparison, benthic macroinvertebrates in late September 2016 had lower overall densities (Figure 4.3.16). Gastropod snails at Penny Point and *Americorophium* spp. at Willow Creek were similarly dominant, but decreased in density (~4,000 m^{-2} and ~7,000 m^{-2} respectively).

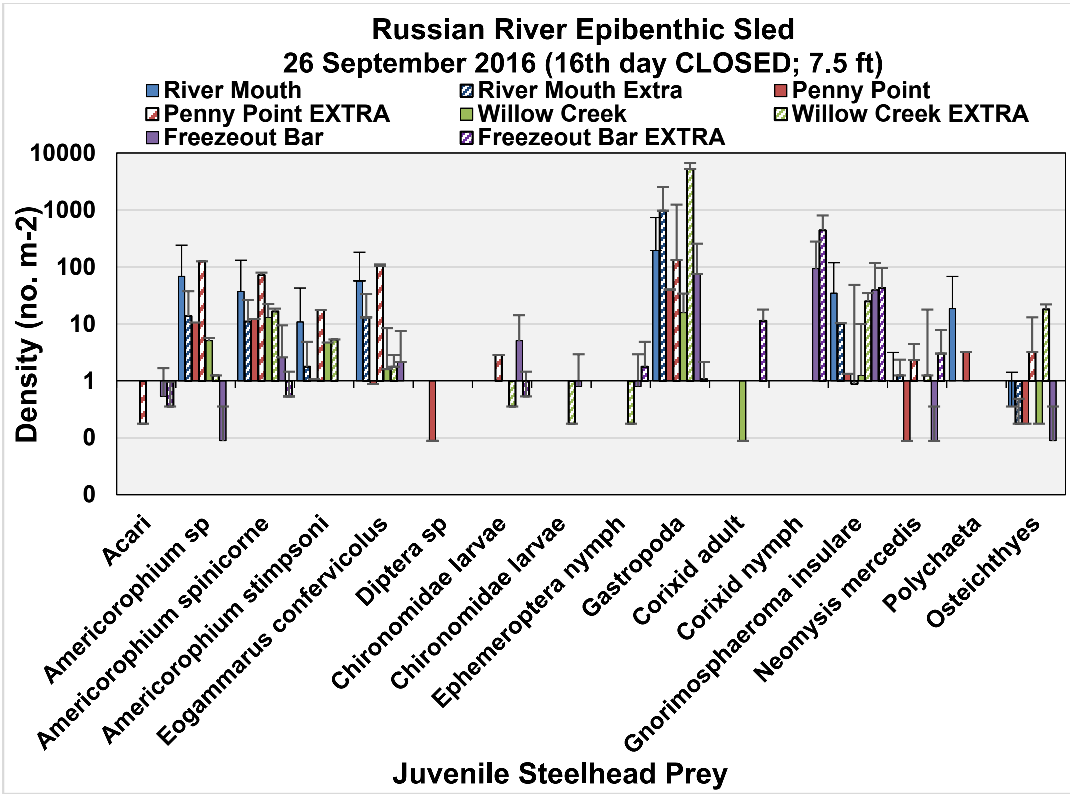

Figure 4.3.13. Density of epibenthic macroinvertebrates documented as juvenile steelhead prey from epibenthic sled sampling at four sites in the Russian River estuary, September 26, 2016. Note logarithmic scaling of density.

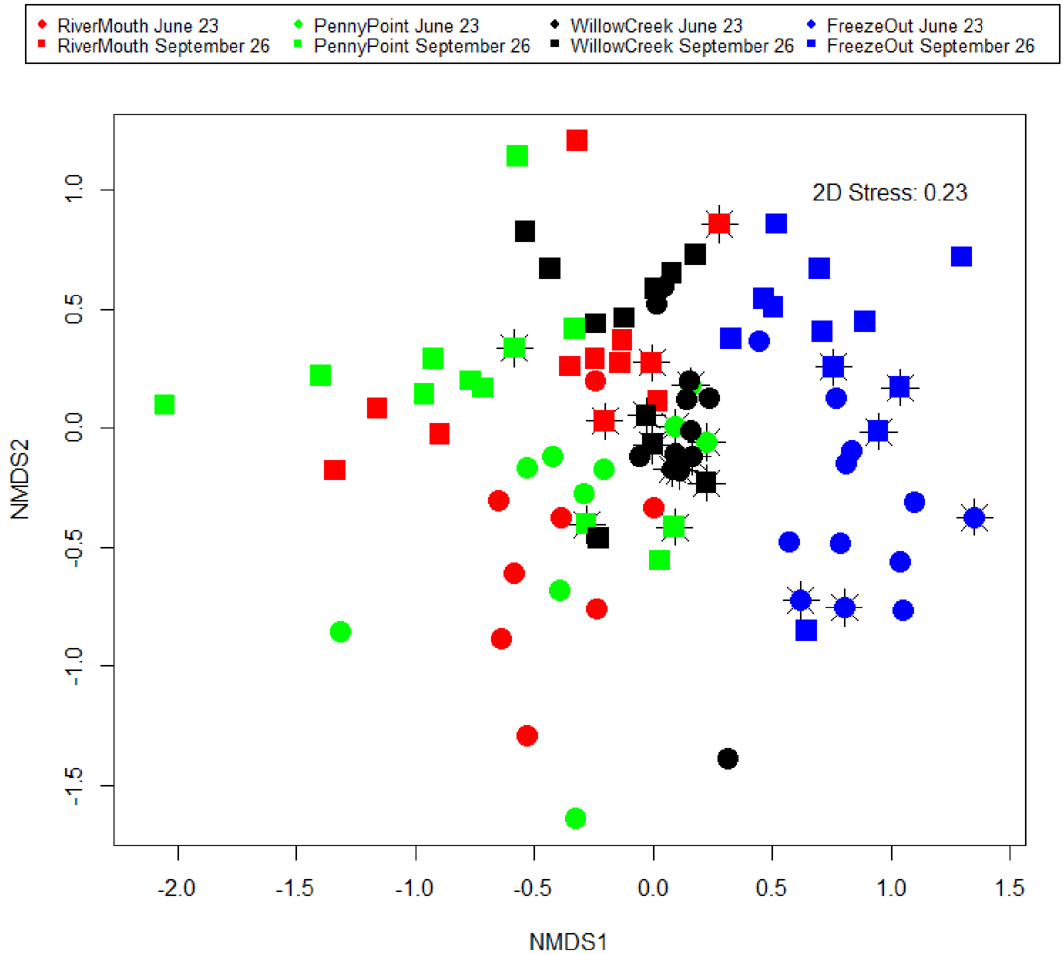

Figure 4.3.14. Multivariate analysis (NMDS) diagram of density composition of epibenthic sled macroinvertebrate prey of juvenile steelhead in lower, middle and upper reaches of the Russian River estuary, 2016; ☼ symbol outline designate additional samples from shallow water habitat inundated during estuary closure.

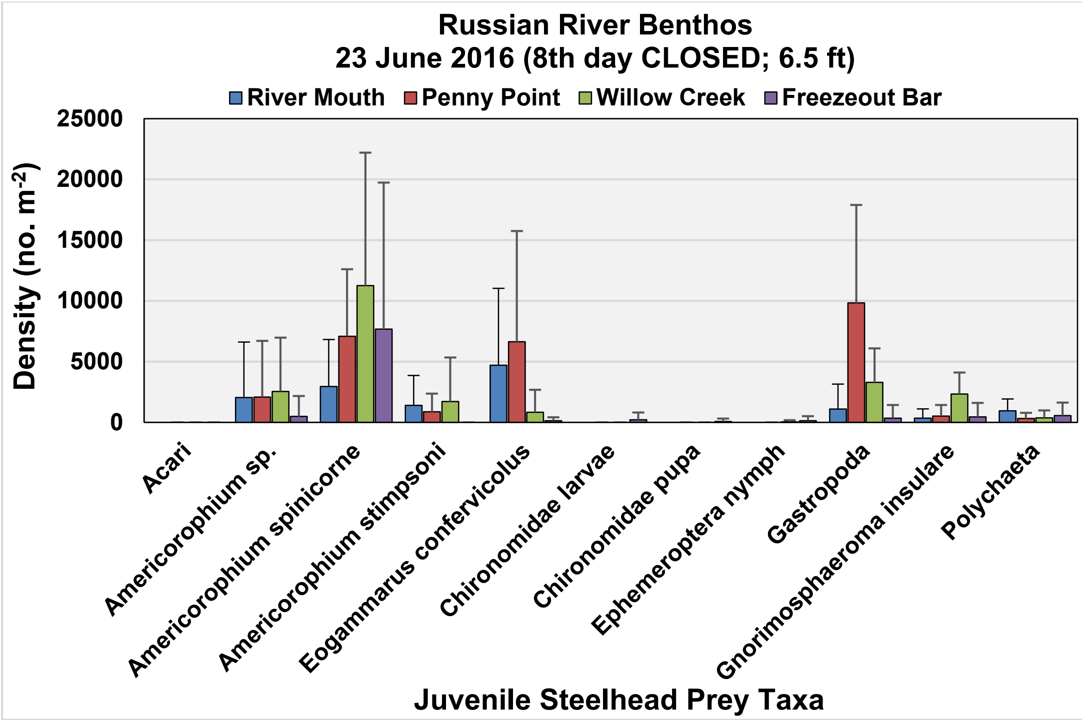

Figure 4.3.15. Density of benthic macroinvertebrates documented as juvenile steelhead prey, four sites in the Russian River Estuary, June 23, 2016.

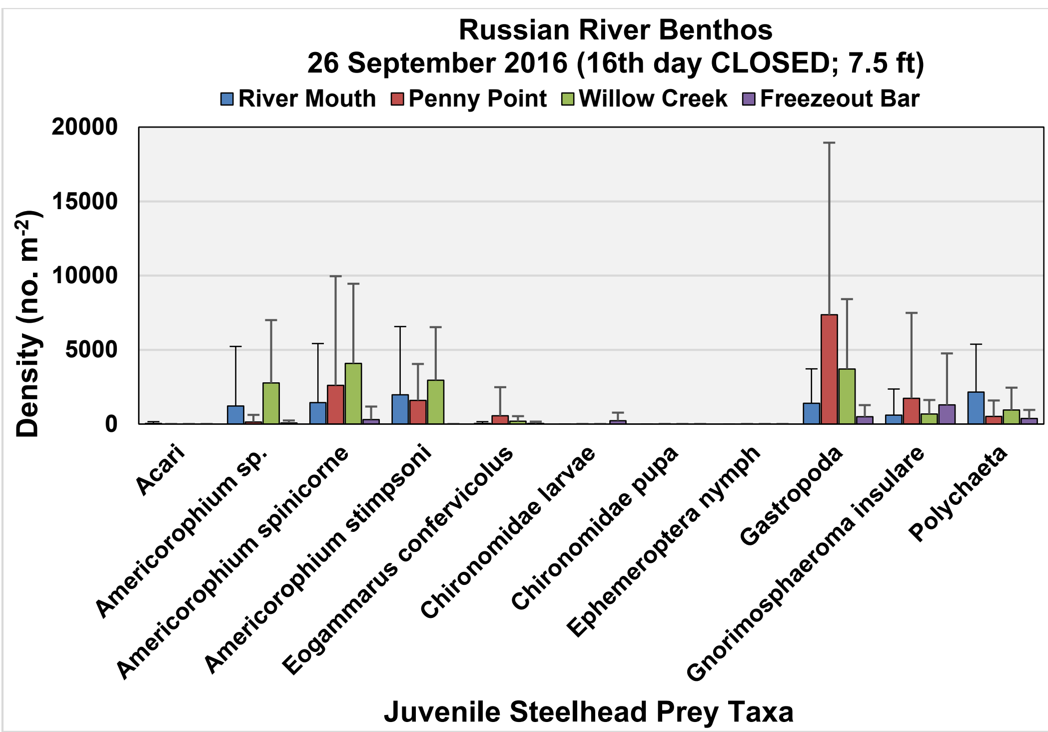

Figure 4.3.16. Density of benthic macroinvertebrates documented as juvenile steelhead prey, four sites in the Russian River Estuary, September 26, 2016.

At Penny Point and River Mouth, *E. confervicolus* were found at mean densities an order of magnitude smaller (<500 m^{-2}). At Freezeout Bar, the only macroinvertebrates found at moderate densities were *G. insulare*, which increased to ~1,300 m^{-2} from ~450 m^{-2} in June. Except for the density of polychetes doubling later in the season at the River Mouth (from ~1,000 m^{-2} to ~2,100 m^{-2}), polychetes were consistently found at mean densities less than 1,000 m^{-2}. Multivariate analysis of the taxa density composition among the four sites over the two dates bracketing the estuary closure (Figure 4.3.17; 2D stress =0.19) indicated no significant difference among sites (Global R = 0.14) or dates (Global R = 0.20).

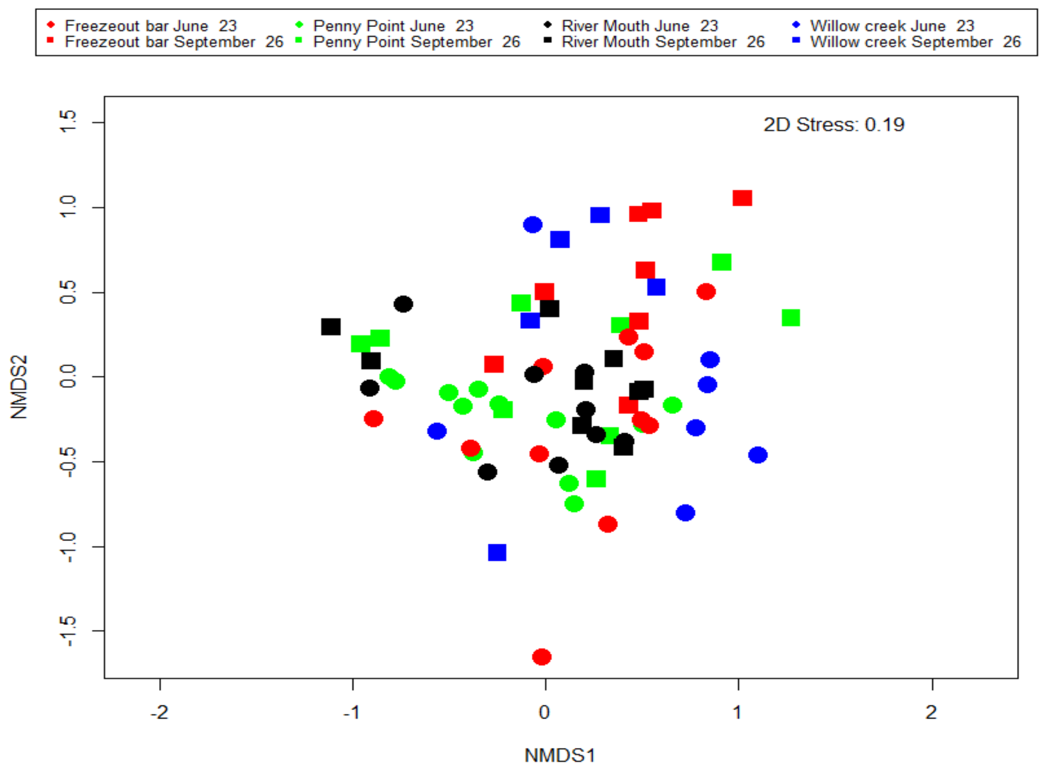

Figure 4.3.17. Non-metric multidimensional scaling diagram of benthic macroinvertebrate (juvenile steelhead prey) assemblages at four sites on two dates in the Russian River Estuary, 2016.

Conclusions and Recommendations

Findings

Prey densities were relatively comparable among the 2016 and prior years' results, implying a consistent estuarine prey community available for juvenile steelhead despite the variability in the occurrence and duration of estuary closures. Similarly, the composition and relative density distribution of macroinvertebrate prey in epibenthic net to shore and channel sled samples were similar, suggesting that there was equal or a relatively minor gradient of prey density distribution from their deeper channel to shallower marginal habitats, even under closed estuary conditions for eight to 16 days. Some prey species, especially gastropod snails, corixid beetles, mysids and some amphipods were found at higher densities in the additional littoral sleds transects than in the routine, deeper transects. Other than the June epibenthic sled River Mouth samples, gastropod snails were primarily found in the shallow epibenthic sled or net to shore samples. Except for the gastropod snails, the densities of epibenthic macroinvertebrates decreased later in the sampling season, similar to previous years. Although only speculative, there was also evidence of increased occupation of the recently flooded, shallow littoral habitat with increased duration of a closed estuary condition, between the eight day duration in June 23 and the 16-day duration of the September 16 sampling. However, the seasonal effects of samples three months apart confound any interpretation of these differences.

The most energy dense prey macroinvertebreate commonly consumed are the corixid beetles. These were found at the highest densities at Freezeout Bar in the upper estuary reach, especially in the shallow littoral habitat. The mysid *Neomysis mercedis* were only found at moderate densities (>25 m^2) at the standard River Mouth and Penny Point sled samples early in the season and the additional Freezeout Bar samples. The densities of mysids in the three additional Freezeout Bar samples consisted of 0.5 m^2, 2.1 m^2 and 568 m^2, suggesting that a single sample was able to capture a dense patch that is challenging to accurately characterize with our sampling methods.

Recommendations

Demography and Production of Prey Populations in Response to Estuary Closure

Despite revisions in the study design and sampling protocols that are more adaptive to assessing changes in prey availability with estuary closure, there is still considerable uncertainty about the effects of extended estuary closure on prey populations and the ability of juvenile steelhead to exploit them. As we have refined our understanding of the natural variability in patterns of juvenile steelhead foraging and prey availability over space and time in the open estuary, future monitoring and research should consider concentrated, real time investigations of immediate responses to estuary closures or, conversely, berm breaches reversing back to open estuary conditions. The purpose of this deeper delving into prey availability would be to address the present uncertainty about the source and consequence of epibenthic prey occupying shallow intertidal habitat with increasing water elevations after the estuary closes.

Enhanced Steelhead Diet and Foraging Rate Data Collection

Differences in potential juvenile steelhead prey consumption rate, indicated by patterns in prior years and in both 2015 and 2014, imply potential reach and estuary status differences in availability among the suite of preferred prey taxa. While the instantaneous ration is a viable index of consumption rate, consideration should be given to conducting periodic diet sampling of juvenile steelhead over a 24-hr or 30-hr period in order to obtain a more precise estimate of daily ration, which is a fundamental measurement for bioenergetic modeling of potential growth. We recognize that this involves periodic sampling during nocturnal hours, which may be unfeasible given Water Agency resources. Similarly, consideration should also be given to pulsed fish sampling during a prolonged estuary closure that enables fish samples from all four sites.

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4.4 Fish Sampling – Beach Seining

The Water Agency has been sampling fish in the Russian River Estuary since 2004 - prior to issuance of the Biological Opinion. An Estuary fish survey methods study was completed in 2003 (Cook 2004). To provide context to data collected in 2016, we present and discuss previous years of data in this report. Although survey techniques have been similar since 2004, some survey locations and the sampling extensity changed in 2010 as required in the Biological Opinion. The distribution and abundance of fish in the Estuary are summarized below. In addition to steelhead, coho salmon, and Chinook salmon, we describe the catch of several common species to help characterize fisheries habitat conditions in the Estuary.

Methods

Study Area

The Estuary fisheries monitoring area included the tidally-influenced section of the Russian River and extended from the sandbar at the Pacific Ocean to Duncans Mills, located 9.8 km (6.1 mi) upstream from the coast (Figure 4.4.1).

Fish Sampling

A beach-deployed seine was used to sample fish species, including salmonids, and determine their relative abundances and distributions within the Estuary. The rectangular seine consisted of 5 mm (¼ inch) mesh netting with pull ropes attached to the four corners. Floats on the top and weights on the bottom positioned the net vertically in the water. From 2004 to 2006, a 30 m (100 ft) long by 3 m (10 ft) deep purse seine was used. From 2007 to 2014 a conventional seine 46 m (150 ft) long by 4 m (14 ft) deep was used. Then in 2015 a 46 m by 3 m seine with a 3 m square pocket located in the center of the net was employed. The seine was deployed with a boat to pull an end offshore and then around in a half-circle while the other end was held onshore. The net was then hauled onshore by hand. Fish were placed in aerated buckets for sorting, identification, and counting prior to release.

Salmonids were anesthetized with Alka-Seltzer tablets or MS-222 and then measured, weighed, and examined for general condition, including life stage (i.e., parr, smolt). All salmonids were scanned for passive integrated transponder (PIT) tags or other marks. Steelhead and coho salmon were identified as wild or hatchery stock by a clipped adipose fin. Hatchery coho salmon were no longer clipped after spring 2013 and were either marked with a coded wire tag or PIT tag. Tissue and scale samples were collected from some steelhead. Unmarked juvenile steelhead caught in the Estuary greater than 60 mm fork length were surgically implanted with a PIT tag. Fish were allowed to recover in aerated buckets prior to release.

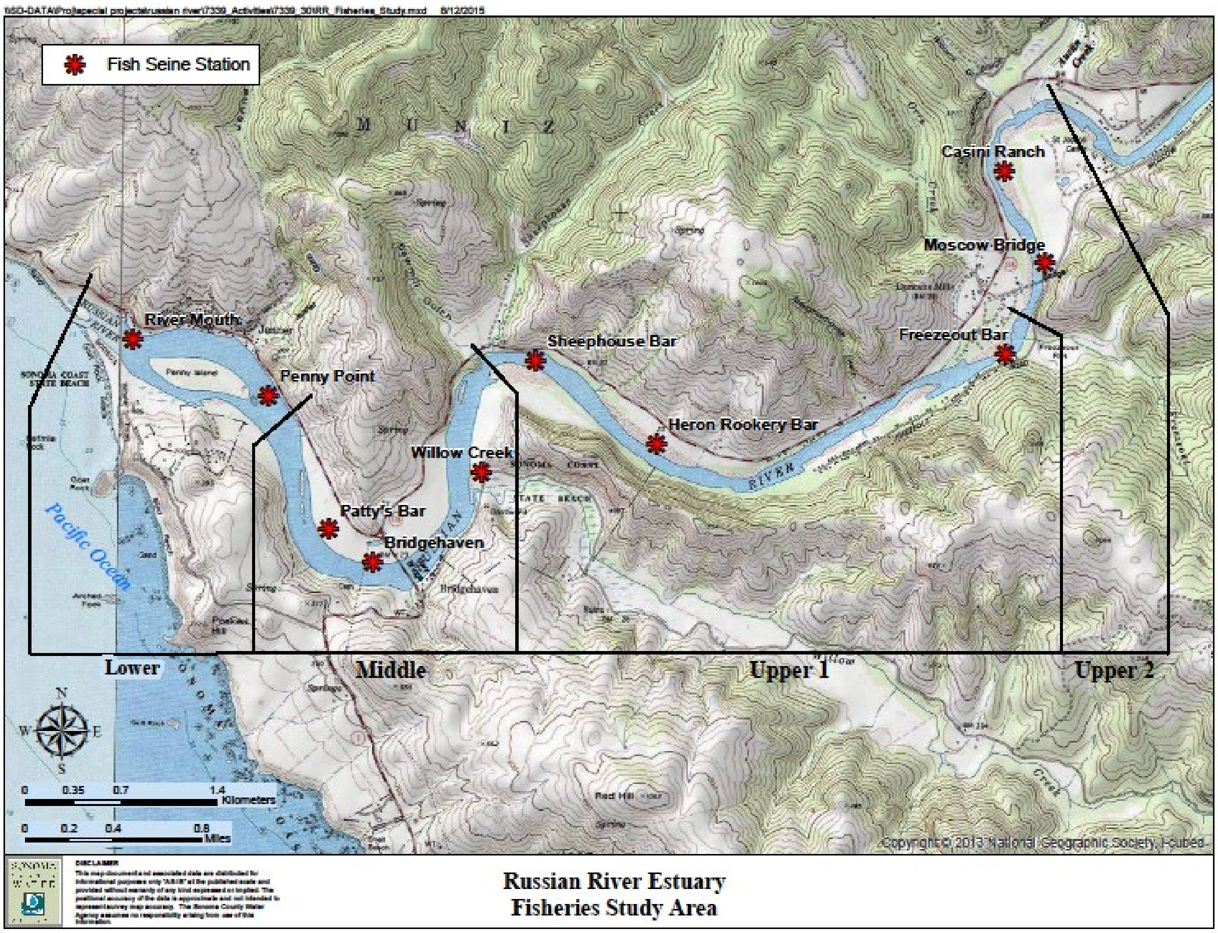

Figure 4.4.1. Russian River Estuary fisheries seining study reaches and sample sites, 2016.

From 2004 to 2009, eight seining stations were located throughout the Estuary in a variety of habitats based on substrate type (i.e., mud, sand, and gravel), depth, tidal, and creek tributary influences. Three seine sets adjacent to each other were deployed at each station totaling 24 seine sets per sampling event. Stations were surveyed approximately every 3 weeks from late May through September or October. Total annual seine pulls ranged from 96 to 168 sets.

Starting in 2010 fish seining sampling was doubled in effort with 300 sets completed for the season. Surveys were conducted monthly from May to October. Between 3 and 7 seine sets where deployed at 10 stations for a total of 50 sets for each sampling event. Twenty-five sets were in the lower and middle Estuary and 25 in the upper Estuary. In 2014 to 2016 the seining sampling effort was conducted in May, June, and September to characterize the Estuary under tidal conditions during the beginning and end of the lagoon management period. In 2014 seining was also conducted in October. Seining in July and August were not completed because a lagoon outlet channel could not be installed for an extended period of time to form a freshwater lagoon.

For data analysis the Estuary study area was divided into three reaches, including Lower, Middle, and Upper, which is consistent with study areas for water quality and invertebrate studies. For the fish seining study, the Upper Reach of the Estuary was divided into Upper1 and Upper2 sub-reaches to improve clarity on fish patterns. Fish seining stations were located in areas that could be sampled during open and closed river mouth conditions. Suitable seining sites are limited during closed mouth conditions due to flooded shorelines. Catch per unit effort (CPUE), defined as the number of fish captured per seine set (fish/set), was used to compare the relative abundance of fish among Estuary reaches and study years.

The habitat characteristics and locations of study reaches, fish seining stations, and number of monthly seining sets are below:

- Lower Estuary
    - River Mouth (7 seine sets): sandbar separating the Russian River from the Pacific Ocean, sandy substrate with a low to steep slope, high tidal influence.
    - Penny Point (3 seine sets): shallow water with a mud and gravel substrate, high tidal influence.
- Middle Estuary
    - Patty's Bar (3 seine sets): large gravel and sand bar with moderate slope, moderate tidal influence.
    - Bridgehaven (7 seine sets): large gravel and sand bar with moderate to steep slope, moderate tidal influence.
    - Willow Creek (5 seine sets): shallow waters near the confluence with Willow Creek, gravel and mud substrate, aquatic vegetation common, moderate tidal influence.
- Upper Estuary

    Upper1 Sub-Reach
    - Sheephouse Bar (5 seine sets): opposite shore from Sheephouse Creek, large bar with gravel substrate and moderate to steep slope, low to moderate tidal influence
    - Heron Rookery Bar (5 seine sets): gravel bank adjacent to deep water, low to moderate tidal influence.
    - Freezeout Bar (5 seine sets): opposite shore from Freezeout Creek, gravel substrate with a moderate slope, low tidal influence.

    Upper2 Sub-Reach
    - Moscow Bridge (5 seine sets): steep to moderate gravel/sand/mud bank adjacent to shallow to deep water, aquatic vegetation common, low tidal influence.
    - Casini Ranch (5 seine sets): moderate slope gravel/sand bank adjacent to shallow to deep water, upper end of Estuary at riffle, very low tidal influence.

Results

Fish Distribution and Abundance

Fish captures from seine surveys in the Russian River Estuary for 2016 are summarized in Table 4.4.1. During the 13 years of study 50 fish species were caught in the Estuary. In 2016, seine captures consisted of 10,964 fish comprised of 23 species. The freshwater smallmouth bass was captured in the Estuary for the first time 2016, although this non-native species is commonly found in the Russian River upstream of the Estuary.

The distribution of fish in the Estuary is, in part, based on a species preference for or tolerance to salinity (Figure 4.4.2). In general, the influence of cold seawater from the ocean under open mouth conditions results in high salinity levels and cool temperatures in the Lower Reach transitioning to warmer freshwater in the Upper Reach from river inflows (Figure 4.4.3). The water column is usually stratified with freshwater flowing over the denser seawater.

Fish commonly found in the Lower Reach were marine and estuarine species including topsmelt (*Atherinops affinis*), surf smelt (*Hypomesus pretiosus*), and staghorn sculpin (*Leptocottus armatus*). The Middle Reach had a broad range of salinities and a diversity of fish tolerant of these conditions. Common fish in the Middle Reach included those found in the Lower Reach and shiner surfperch (*Cymatogaster aggregata*) and bay pipefish (*Syngnathus leptorhynchus*). Freshwater dependent species, such as the Sacramento sucker (*Catostomus occidentalis*), Sacramento pikeminnow (*Ptychocheilus grandis*), and Russian River tule perch (*Hysterocarpus traskii pomo*) were predominantly distributed in the Upper Reach. Anadromous fish, such as steelhead (*Oncorhynchus mykiss*) and American shad (*Alosa sapidissima*), which can tolerate a broad range of salinities, occurred throughout the Estuary. Habitat generalists, such as threespine stickleback (*Gasterosteus aculeatus*) and prickly sculpin (*Cottus asper*), occurred in abundance in the Estuary, except within full strength seawater in the Lower Reach.

Estuary water conditions were fresher during the May and June seine sampling than in typical years from late spring rainfall and high river flows (Figure 4.4.3). This shifted the distribution of freshwater species into the Middle Estuary, which is usually dominated by estuarine species (Figure 4.4.2). The higher abundance of estuarine species in the Upper1 Reach, compared to the Middle and Lower reaches, was entirely from juvenile starry flounder, which tend to prefer lightly brackish to freshwater.

Table 4.4.1. Total fish caught by beach seine in the Russian River Estuary, 2016. Each station was sampled monthly during May, June, and September for a total of 150 seine sets for all sites. Monthly seine sets per station are shown in parentheses.

		Seining Station										
Life History	Species	River Mouth (7)	Penny Point (3)	Patty's Bar (3)	Bridge-haven (7)	Willow Creek (5)	Sheep-house Bar (5)	Heron Rookery Bar (6)	Freeze-out Bar (4)	Moscow Bridge (5)	Casini Ranch (5)	Total
Anadromous	American shad	11	12							9	33	65
	Chinook salmon	12	1		2	5		1	10		10	41
	coho salmon	3				1		1	2			7
	steelhead	2			2		2	1	20		6	33
Freshwater	bluegill									1		1
	common carp								1	3		4
	hitch					1			15	116	1	133
	largemouth bass					1		2	22	135	9	169
	Russian River tule perch					5		1	7	1167	9	1189
	Sacramento pikeminnow		1	1	12	174			13	40	4	245
	Sacramento sucker	10	17	632	1606	2308	478	393	310	505	51	6310
	smallmouth bass									9		9
	white catfish									2		2
Estuarine	bay pipefish	1										1
	shiner surfperch	9										9
	staghorn sculpin	51	2	4	11	3						71
	starry flounder	69	9	6	44	14	53	1165	296	12	33	1701
	topsmelt	39			1							40
Marine	cabezon	3										3
	Pacific herring	1										1
	Sebastes sp. (rockfish)	4										4
Generalist	prickly sculpin*	10	6	61	51	19	12	3		1		163
	sculpin sp.				8							8
	threespine stickleback	66	79	47	177	192	32	49	22	91		755
	Total	291	127	751	1914	2723	577	1616	718	2091	156	10964

*Prickly Sculpin counts may include small numbers of the freshwater-resident Coast Range sculpin (*Cottus aleuticus*) and riffle sculpin (*Cottus gulosus*), although neither of these species has been reported from the Estuary.

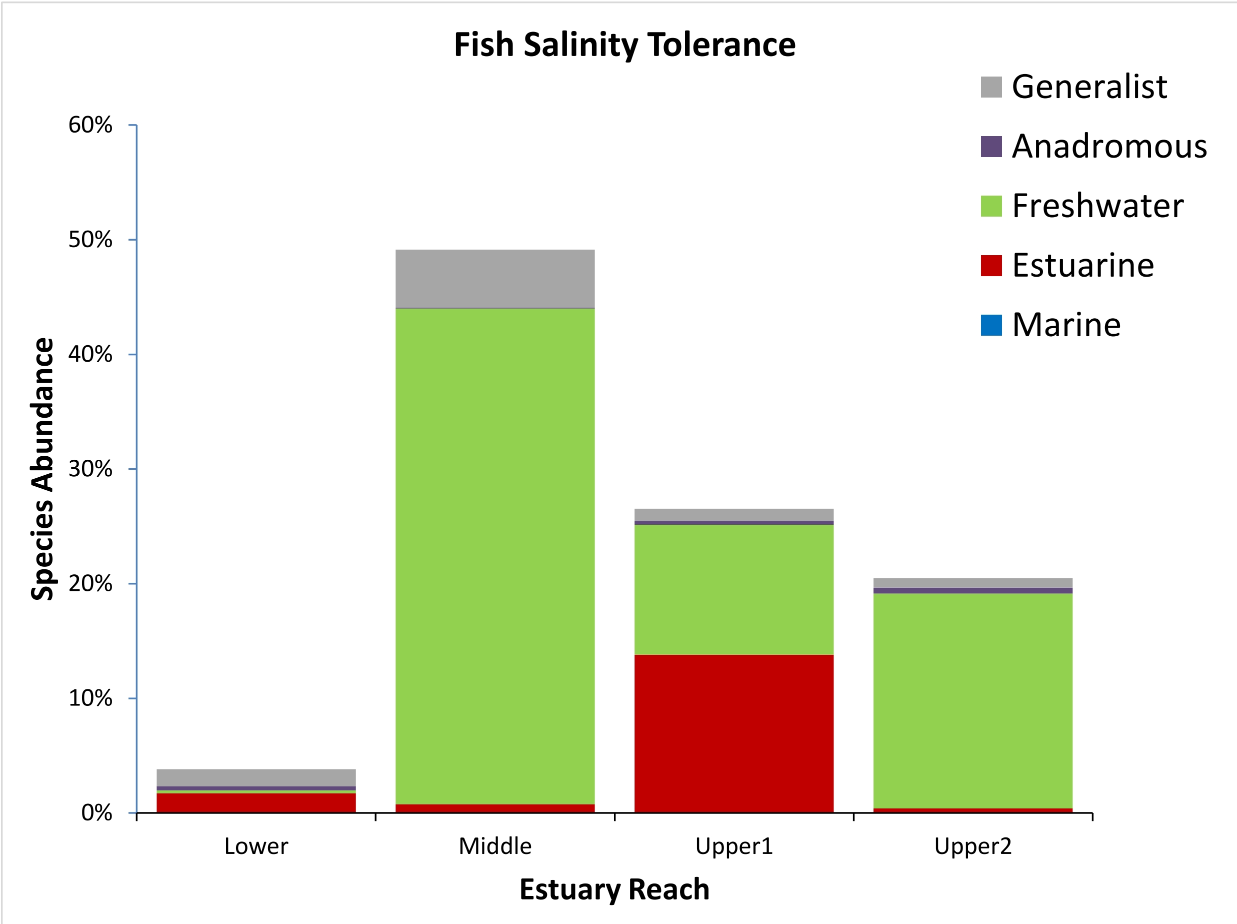

Figure 4.4.2. Distribution of fish in the Russian River Estuary based on salinity tolerance and life history, 2016. Data is from monthly seining during May, June, and September. Groups include: generalist species that occur in a broad range of habitats; species that are primarily anadromous; freshwater resident species; brackish-tolerant species that complete their lifecycle in estuaries; and species that are predominantly marine residents.

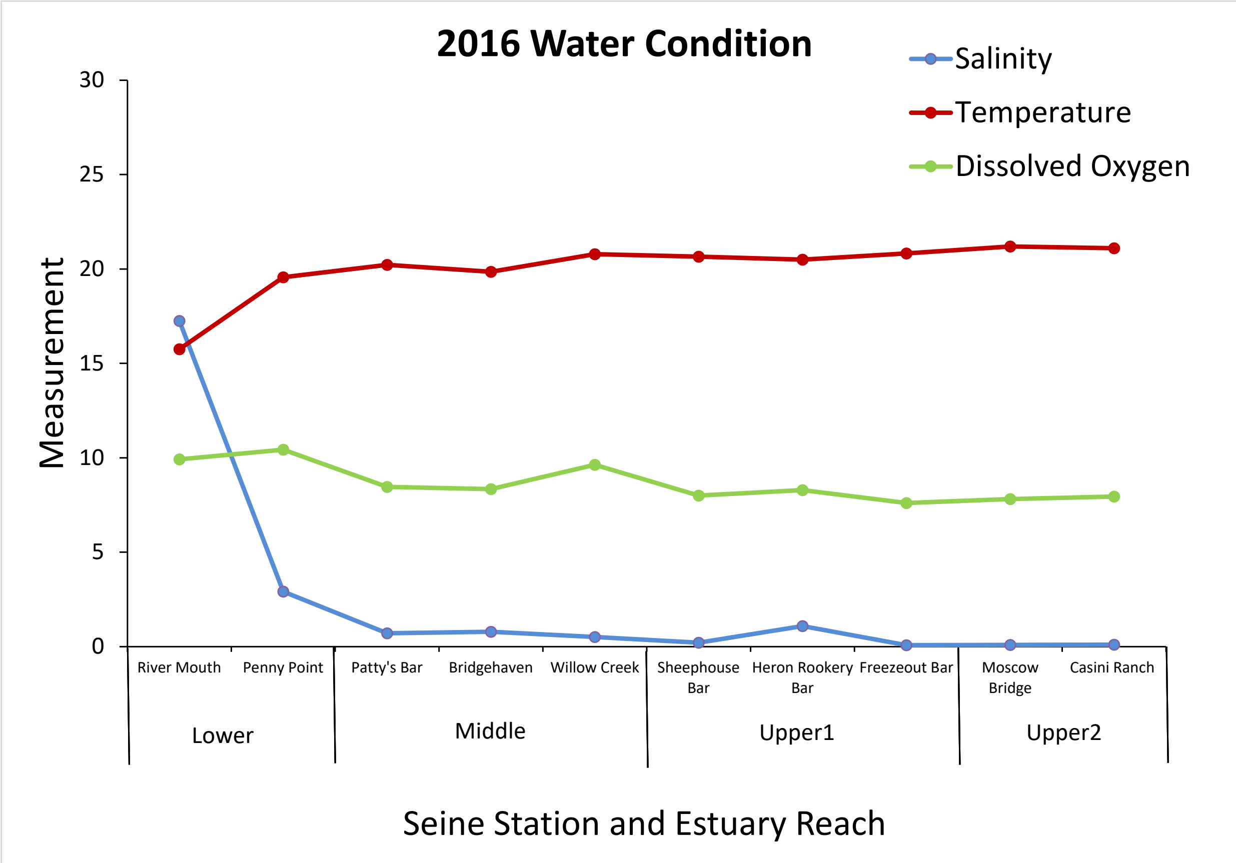

Figure 4.4.3. Generalized water conditions at fish seining stations in the Russian River Estuary, 2016. Values are averages collected at 0.5 m intervals in the water column during beach seining events from May, June, and September during primarily open mouth conditions. Water measurements are salinity in parts per thousand (ppt), dissolved oxygen in milligrams per liter (mg/L), and temperature in Celsius (C).

Steelhead

During 2016, a total of 33 steelhead were captured (Table 4.4.1) in 150 seine sets. The resulting CPUE was 0.22 fish/set (Figure 4.4.4). In comparison, during 2015, a total of 50 steelhead were captured in 150 seine sets for a CPUE of 0.33 fish/set. The highest CPUE for all study years was 1.32 fish/set in 2008. All steelhead captured in 2016 were wild (not from the Russian River watershed's hatcheries). The seasonal abundance of steelhead captures varied annually in the Estuary (Figure 4.4.5). Juvenile steelhead were captured during all three survey events in 2016. The highest steelhead abundances are typically in June and August. During 2016, steelhead captures were highest during June at 0.40 fish/set. The highest capture abundance among all study years was in August at 4.3 fish/set and June at 4.2 fish/set in 2008. Since seining surveys began in 2004, steelhead appear to have a patchy distribution and vary in abundance in the Estuary (Figure 4.4.6). Over all years surveyed, captures were typically highest in the Upper Reach with a high of 6.9 fish/set in the Upper1 Sub-Reach in 2008.

The temporal and spatial distribution of juvenile steelhead in the Estuary in 2016 was strongly influenced by relatively large captures in the Upper1 and Upper2 in May and June (Figure 4.4.7). Very few steelhead were caught in the Middle and Lower reaches.

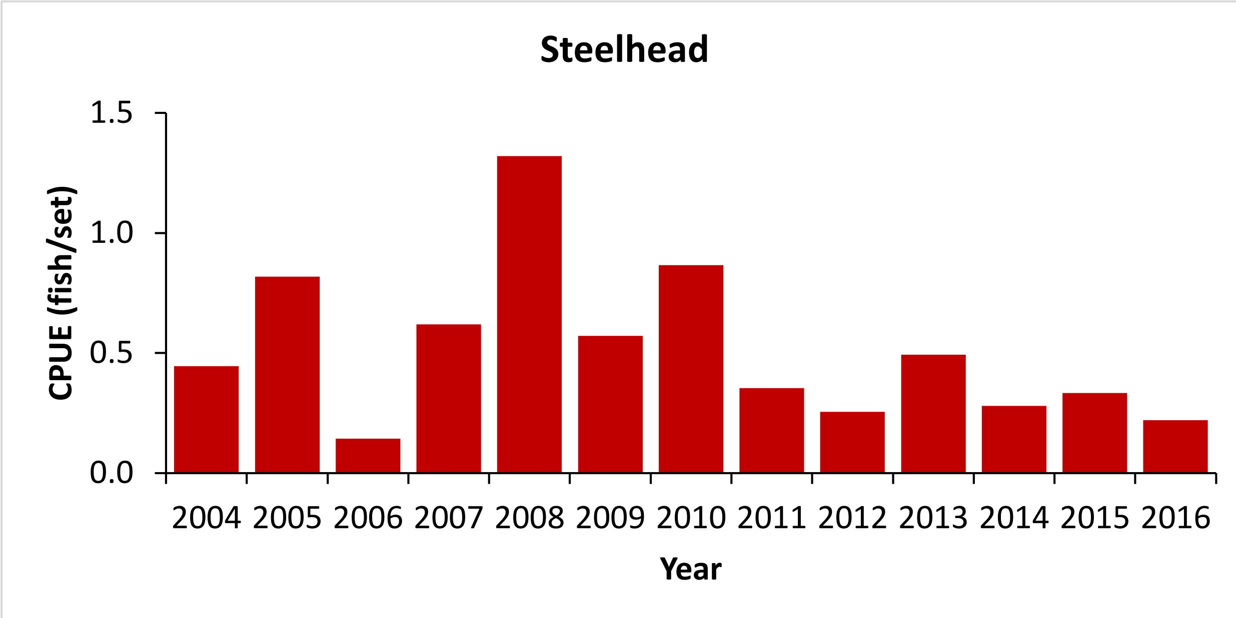

Figure 4.4.4. Annual abundance of juvenile steelhead captured by beach seine in the Russian River Estuary, 2004-2016. Samples are from 96 to 300 seine sets conducted yearly from May to October.

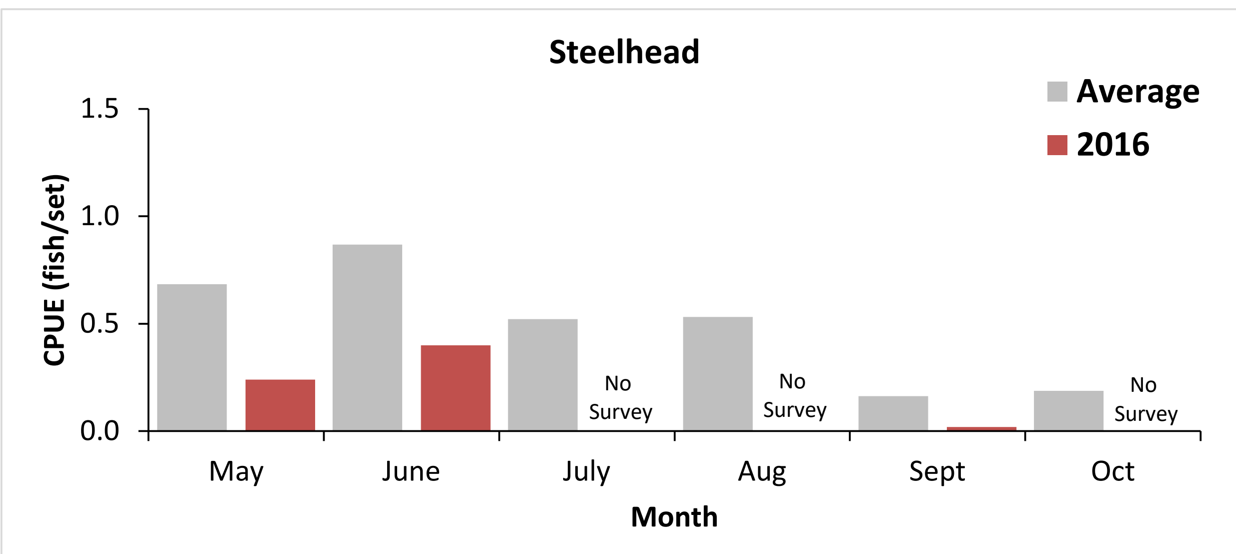

Figure 4.4.5. Seasonal abundance of juvenile steelhead captured by beach seine in the Russian River Estuary, 2004-2016. Seining events consisted of 21 to 50 seine sets approximately monthly. October surveys began in 2010.

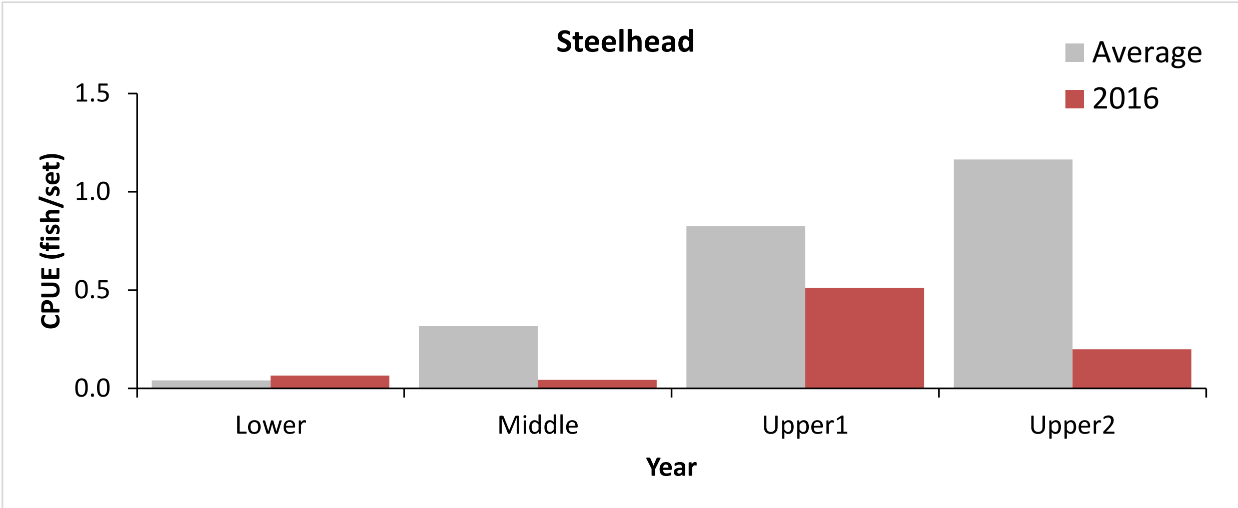

Figure 4.4.6. Distribution of juvenile steelhead in the Russian River Estuary, 2004-2016. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Sub-Reach (Casini Ranch and Moscow Bridge stations) from 2004 to 2009. Data from 2004 to 2015 were averaged.

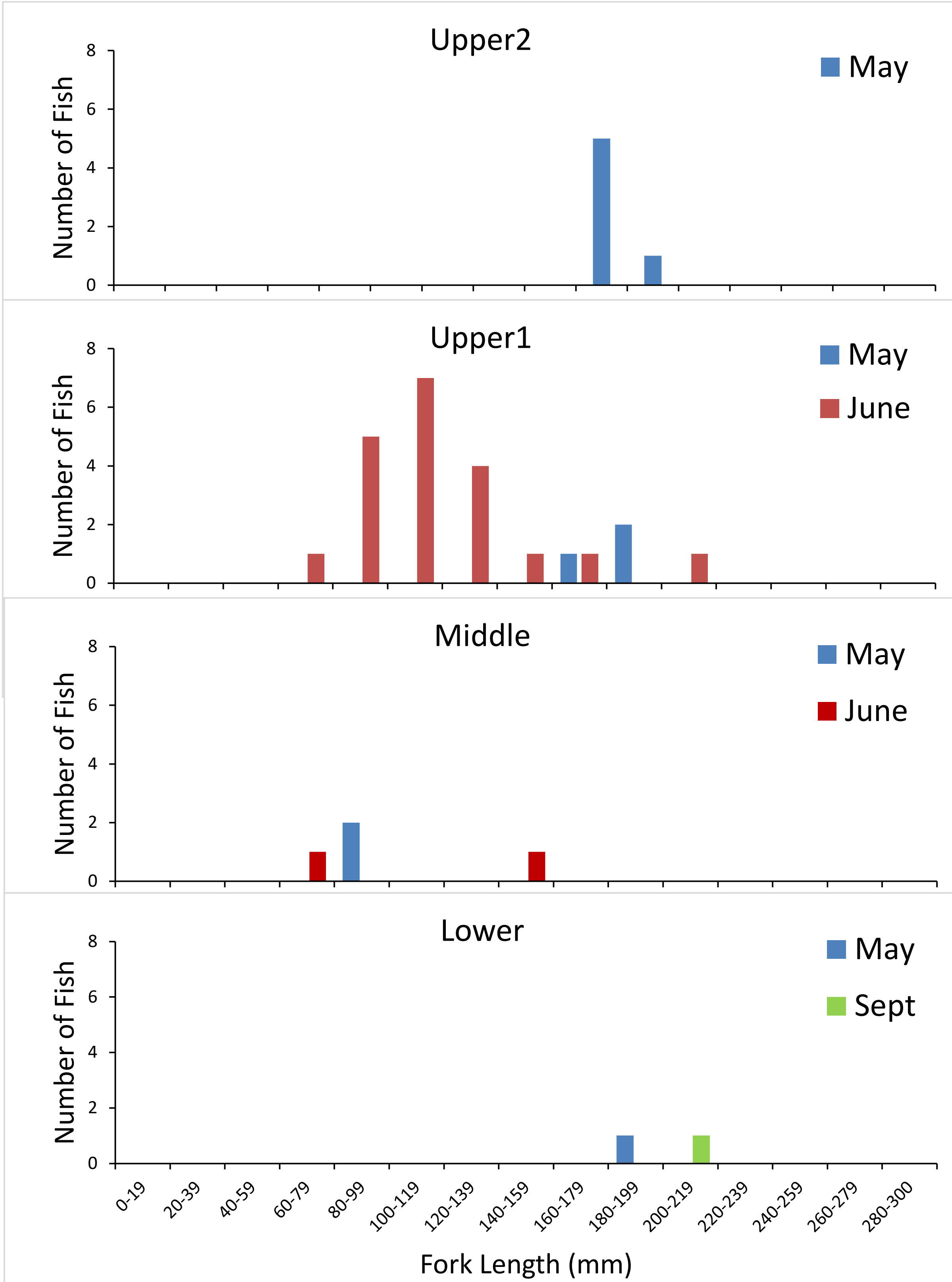

Figure 4.4.7. Length frequency of juvenile steelhead captured by beach seine in the Russian River Estuary, 2016. Fish captures are grouped by Estuary reach and month.

Most juvenile steelhead captured in the Estuary were age 0+ parr or age 1+ smolts and ranged in size from 67 mm to 206 mm fork length (Figure 4.4.8).

In 2016, 16 juvenile steelhead captured during Estuary seining surveys were implanted with PIT tags. Also, 1,797 juvenile steelhead where PIT-tagged during downstream migrant trapping studies in the Russian River and tributaries upstream of the Estuary. There were two PIT-tagged steelhead captured in the Estuary during 2016 seining. These steelhead parr were initially tagged in May at the Austin Creek downstream migrant trap and then recaptured on June 22 at the Freezeout Bar seining station. The size and growth patterns of steelhead are shown in Figure 4.4.9.

Chinook Salmon

A total of 41 Chinook salmon smolts were captured by beach seine in the Estuary during 2016 (Table 4.4.1). The abundance of smolts in the Estuary has varied since studies began in 2004 (Figure 4.4.10). The highest abundance of Chinook salmon smolts was in 2008 at 5.2 fish/set. The lowest abundance of Chinook smolts was in 2016 at 0.3 fish/set. Chinook salmon smolts are usually most abundant during May and June (Figure 4.4.11) and rarely encountered after July. Monthly smolt captures in 2016 were highest during May at 0.6 fish/set. Chinook salmon smolts were distributed throughout the Estuary with captures at most sample stations and reaches annually (Figure 4.4.12).

There were 2,994 Chinook smolts PIT-tagged at several downstream migrant trap sites in the Russian River and tributaries during spring 2016. However, none of these smolts were later recaptured in the Estuary.

Coho Salmon

There have been relatively few coho salmon smolts captured in the Estuary during our beach seining surveys (Figure 4.4.13). The first coho salmon smolt captured in the Estuary was a single fish in 2006. In 2011 and 2015 there were marked increases in abundances of coho smolts with a CPUE of 0.9 and 0.7 fish/set, respectively. During 2016 the total capture of coho was seven smolts, which is the least since coho were first detected in 2006. All of these fish contained a coded wire tag in their snout indicating they were hatchery raised. The relatively low coho salmon captures in the Estuary are related to their scarcity in the Russian River watershed, but also the timing of our seining surveys that begin in late-May or June when most smolts have already migrated to the ocean. Also, several years of severe drought likely reduced coho survival. Nearly all coho salmon smolts were captured by June (Figure 4.4.14). The spatial distribution of coho smolts has varied annually (Figure 4.4.15). In 2016 coho were captured in all reaches, except Upper2 Sub-Reach.

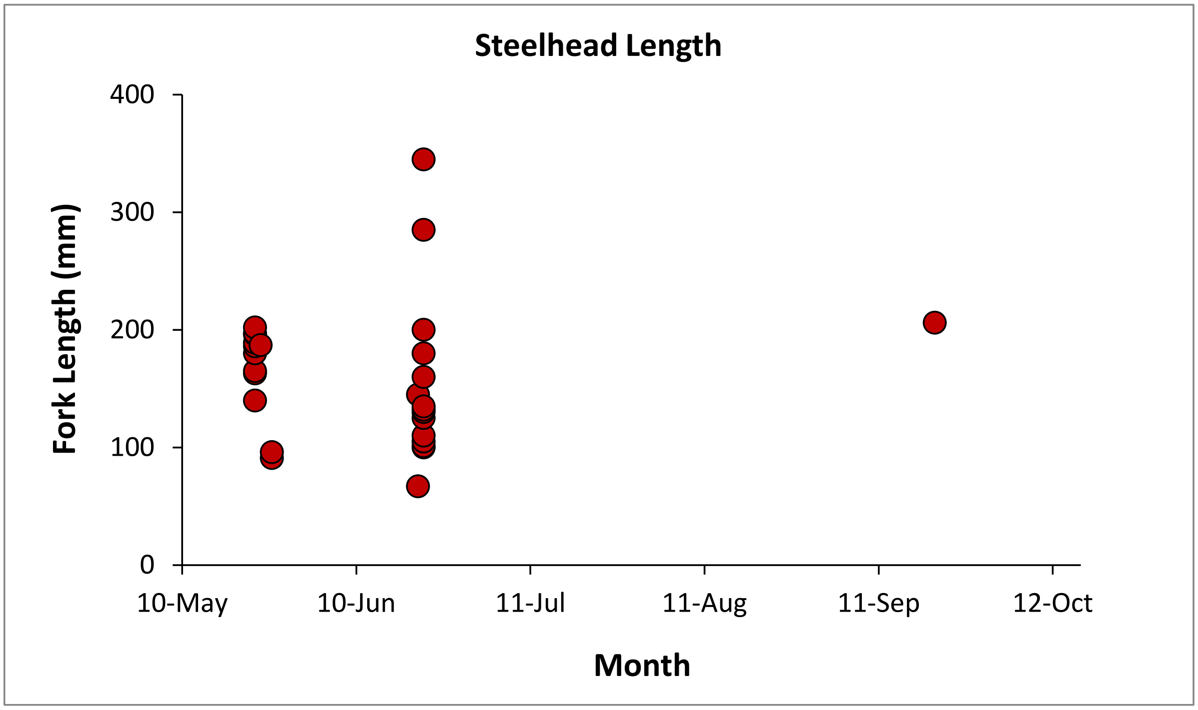

Figure 4.4.8. Juvenile steelhead sizes captured by beach seine in the Russian River Estuary, 2016.

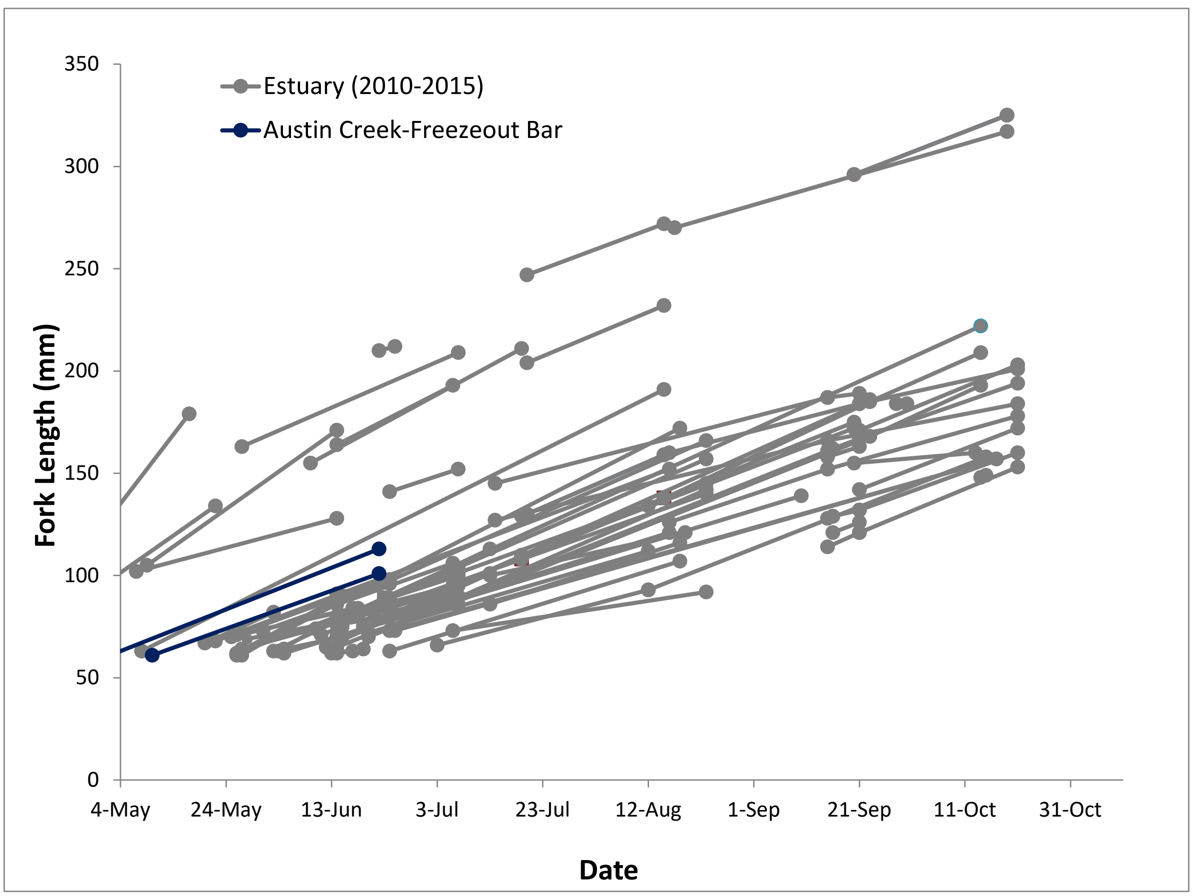

Figure 4.4.9. Growth rates of juvenile steelhead in the Estuary, 2010-2016. Fish were either PIT tagged in the Estuary or upstream and then recaptured in the Estuary. Fish from 2010-2015 are shown in gray. Other colors are steelhead from 2016.

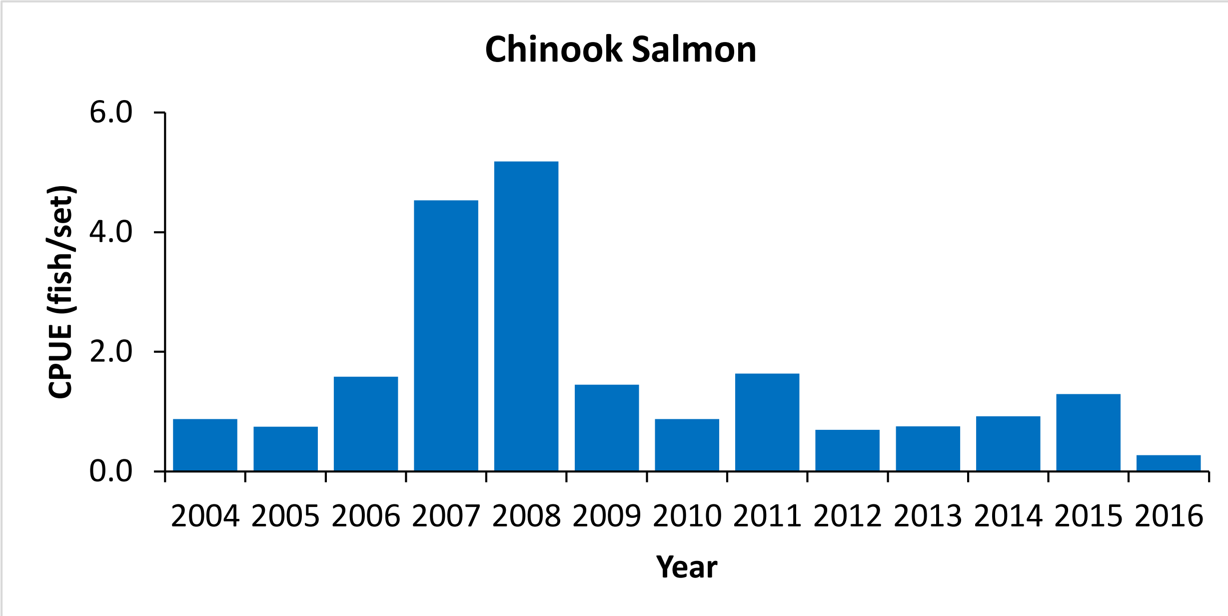

Figure 4.4.10. Annual abundance of Chinook salmon smolts captured by beach seine in the Russian River Estuary, 2004-2016. Samples are from 96 to 300 seine sets yearly from May to October.

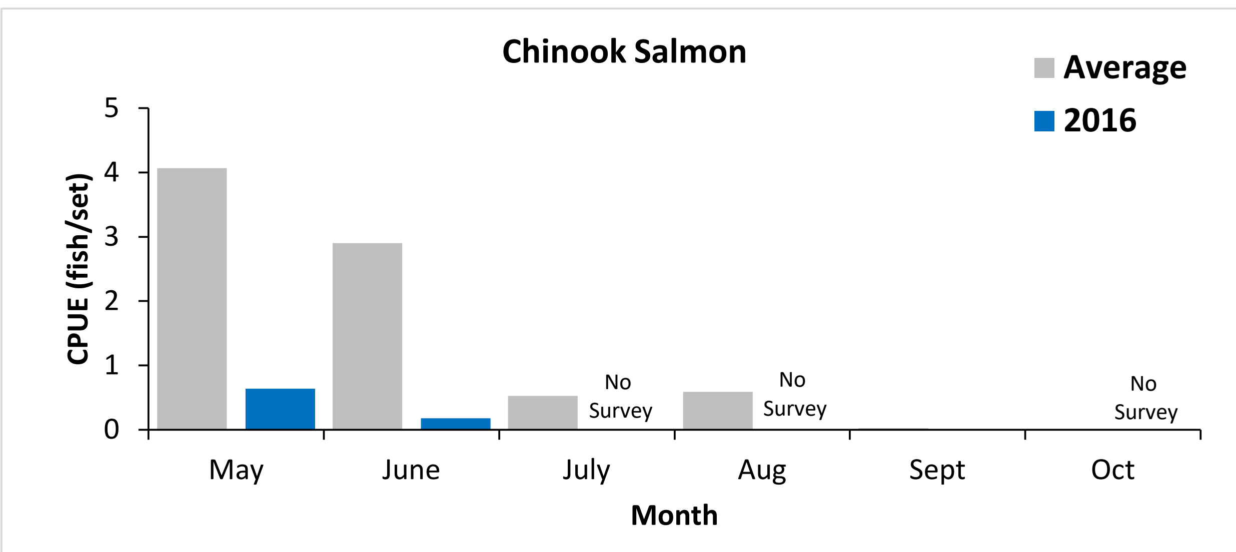

Figure 4.4.11. Seasonal abundance of Chinook salmon smolts captured by beach seine in the Russian River Estuary, 2004-2016. Seining events consisted of 21 to 50 seine sets approximately monthly. October surveys began in 2010. Data from 2004 to 2015 were averaged.

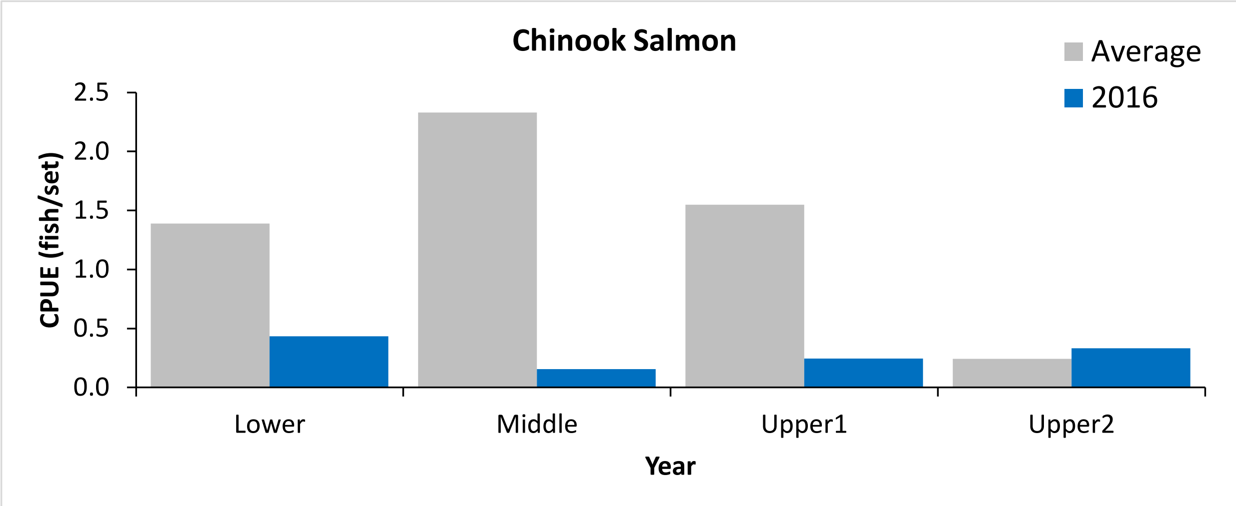

Figure 4.4.12. Spatial distribution of Chinook salmon smolts in the Russian River Estuary, 2004-2016. Fish were sampled by beach seine consisting of 96 to 300 sets annually. Data from 2004 to 2015 were averaged. No surveys were conducted in the Upper2 Sub-Reach (Casini Ranch and Moscow Bridge stations) from 2004 to 2009.

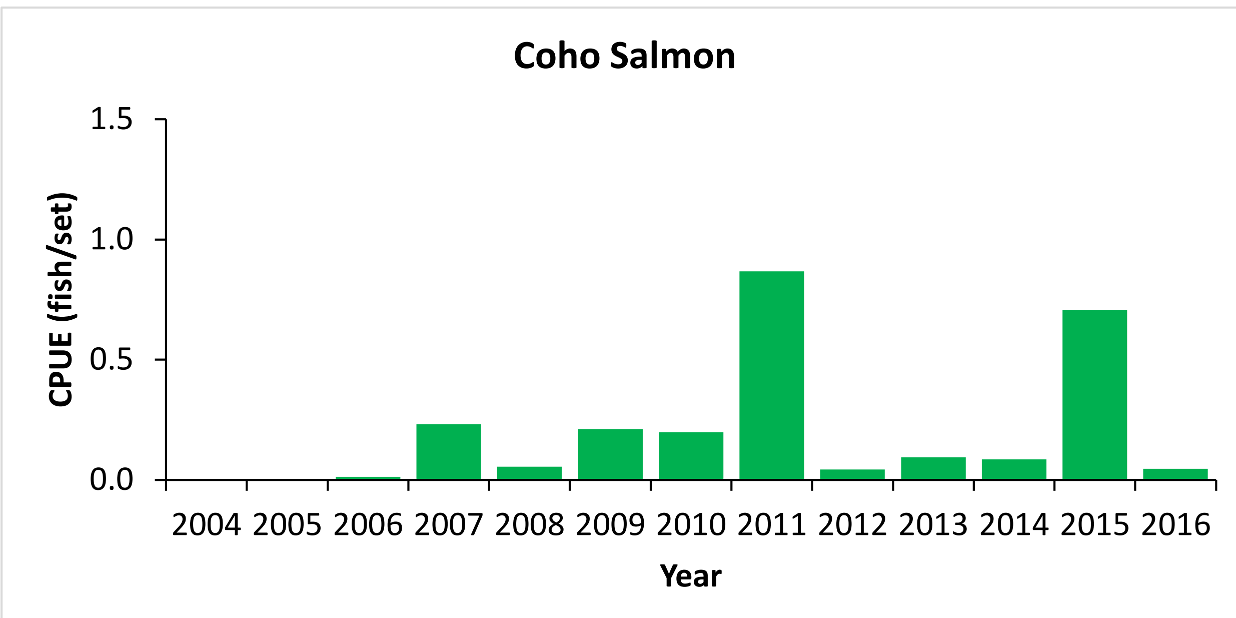

Figure 4.4.13. Annual abundance of coho salmon smolts captured by beach seine in the Russian River Estuary, 2004-2016. Samples are from 96 to 300 seine sets yearly from May to October.

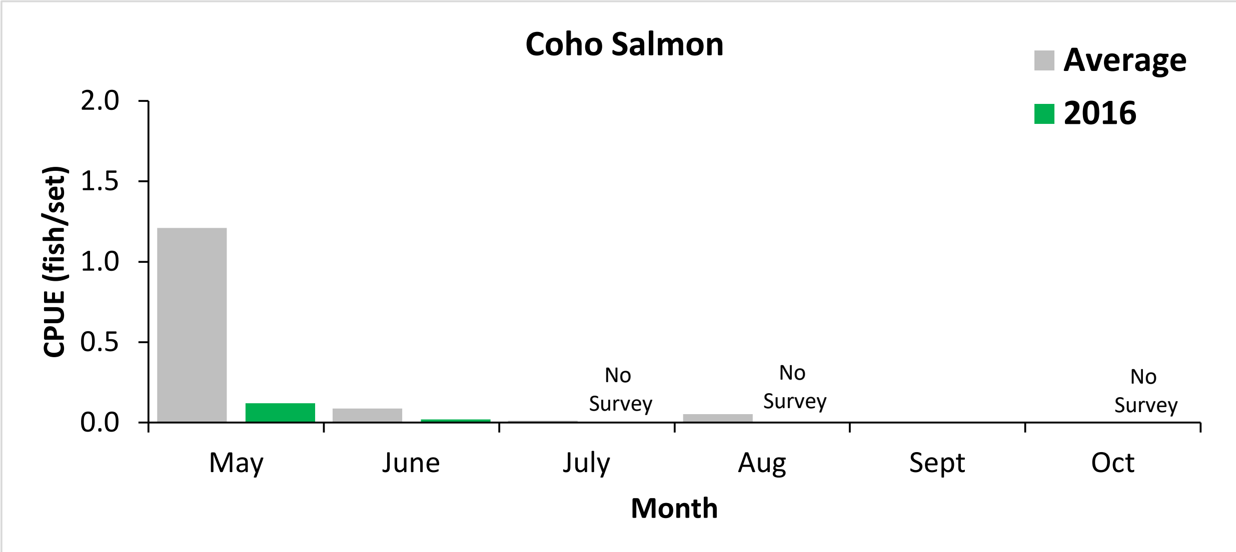

Figure 4.4.14. Seasonal abundance of coho salmon smolts captured by beach seine in the Russian River Estuary, 2004-2016. Seining events consisted of 21 to 50 seine sets approximately monthly. October surveys began in 2010. Data from 2004 to 2015 were averaged.

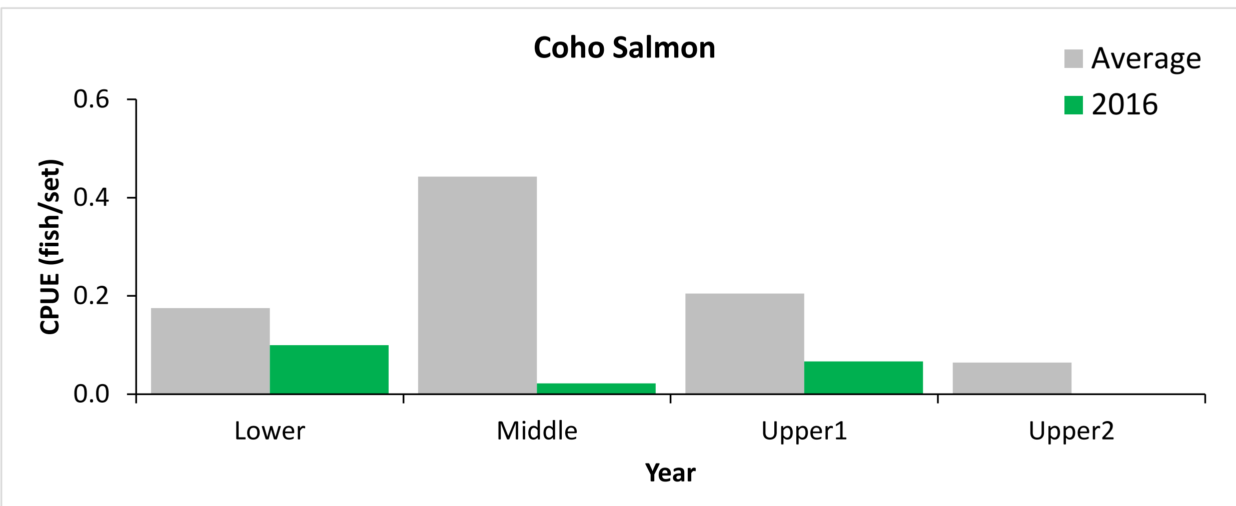

Figure 4.4.15. Spatial distribution of coho salmon smolts in the Russian River Estuary, 2004-2016. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Sub-Reach (Casini Ranch and Moscow Bridge stations) from 2004 to 2009. Data from 2004 to 2015 were averaged.

American Shad

American shad is an anadromous sportfish, native to the Atlantic coast. It was introduced to the Sacramento River in 1871 and within two decades was abundant locally and had established populations from Alaska to Mexico (Moyle 2002). Adults spend from 3 to 5 years in the ocean before migrating upstream to spawn in the main channels of rivers. Juveniles spend the first year or two rearing in rivers or estuaries.

The abundance of American shad in the Estuary during 2016 was low at 0.43 fish/set (Figure 4.4.16). This low abundance may have been influenced by the reduced seining effort in 2016

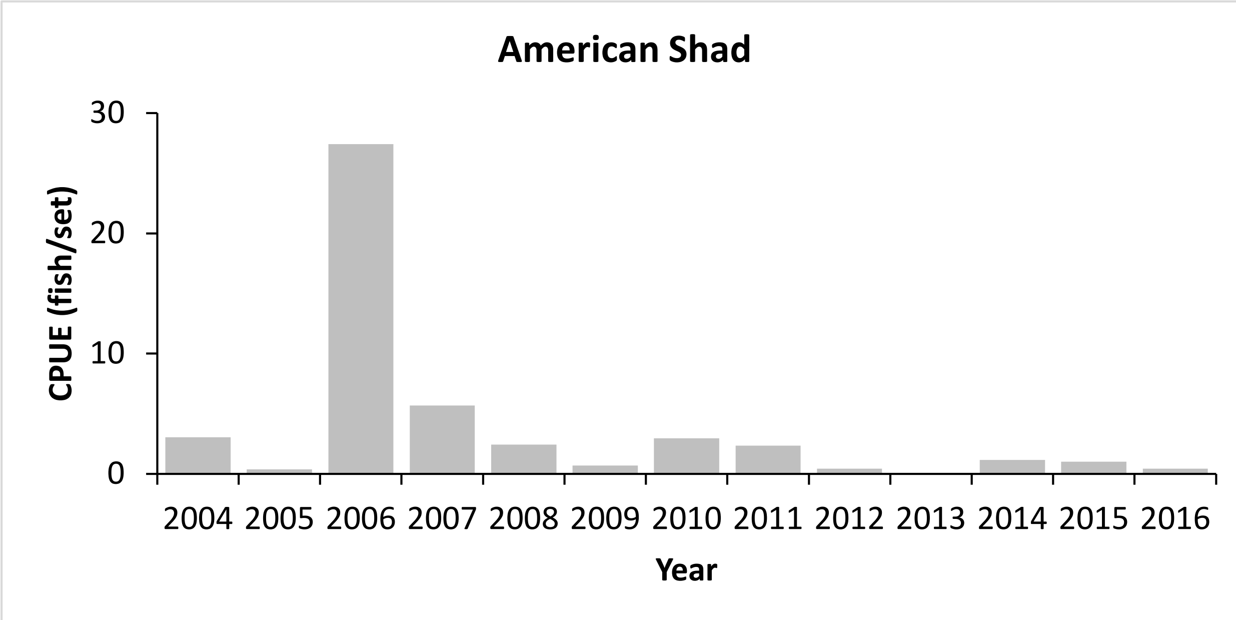

Figure 4.4.16. Annual abundance of juvenile American shad captured by beach seine in the Russian River Estuary, 2004-2016. Samples are from 96 to 300 seine sets yearly from May to October.

where no surveys were conducted during July and August. Typically, juvenile American shad first appear in relatively large numbers in July and the catch usually peaks in August. Shad are typically distributed throughout the Estuary, although in 2016 they were only found in the Lower and Upper1 reaches (Figure 4.4.17).

Topsmelt

Topsmelt are one of the most abundant fish in California estuaries (Baxter et al. 1999) and can tolerate a broad range of salinities and temperatures, but are seldom found in freshwater (Moyle 2002). They form schools and are often found near the water surface in shallow water. Sexual maturity is reached in 1 to 3 years and individuals can live as long as 7 to 8 years. Estuaries are used as nursery and spawning grounds and adults spawn in late spring to summer.

Topsmelt is a common fish in the Russian River Estuary. However, the abundance of topsmelt in the Estuary has varied substantially since 2004. There were peaks in abundance in 2006 and 2014 with a CPUE up to 17.9 and abundances as low as 0.3 fish/set in 2012 and 2016 (Figure 4.4.18). Also, the abundance of topsmelt in 2015 and 2016 may been an underestimate because no seining was conducted in July and August when the catch of topsmelt usually peaks. Topsmelt are mainly distributed in the Lower and Middle Reaches in the Estuary (Figure 4.4.19).

Starry Flounder

Starry flounder range from Japan and Alaska to Santa Barbara in coastal marine and estuarine environments. In California, they are common in bays and estuaries (Moyle 2002). This flatfish is usually found dwelling on muddy or sandy bottoms. Males mature during their second year and females mature at age 3 or 4 (Baxter et al. 1999). Spawning occurs during winter along the coast, often near the mouths of estuaries. Young flounders spend at least their first year rearing

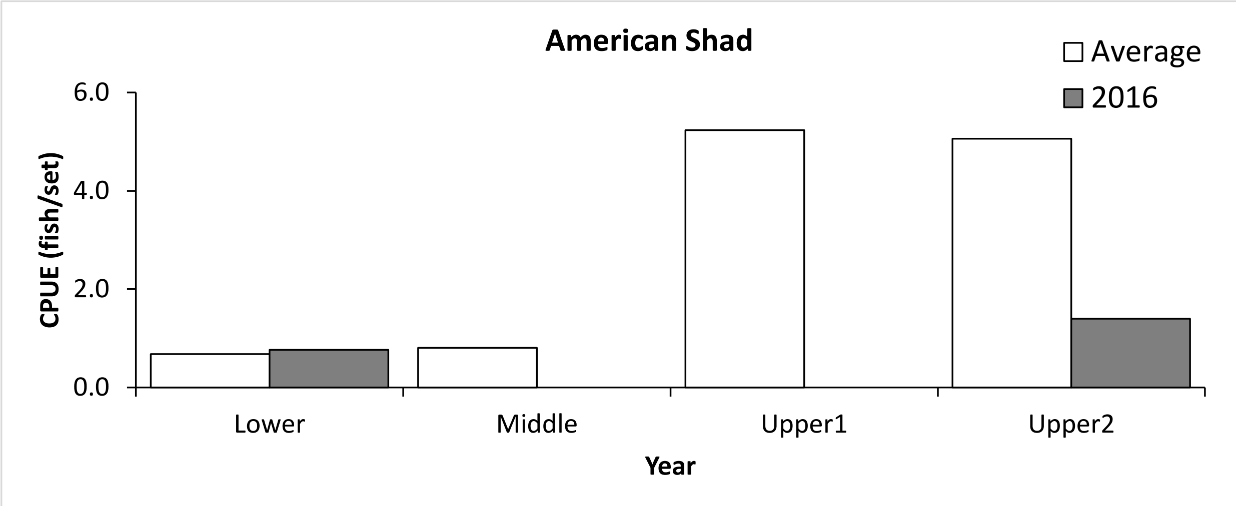

Figure 4.4.17. Spatial distribution of juvenile American shad in the Russian River Estuary, 2004-2016. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Reach during 2004 and 2009. Data from 2004 to 2015 were averaged. Whiskers indicate minimum and maximum values.

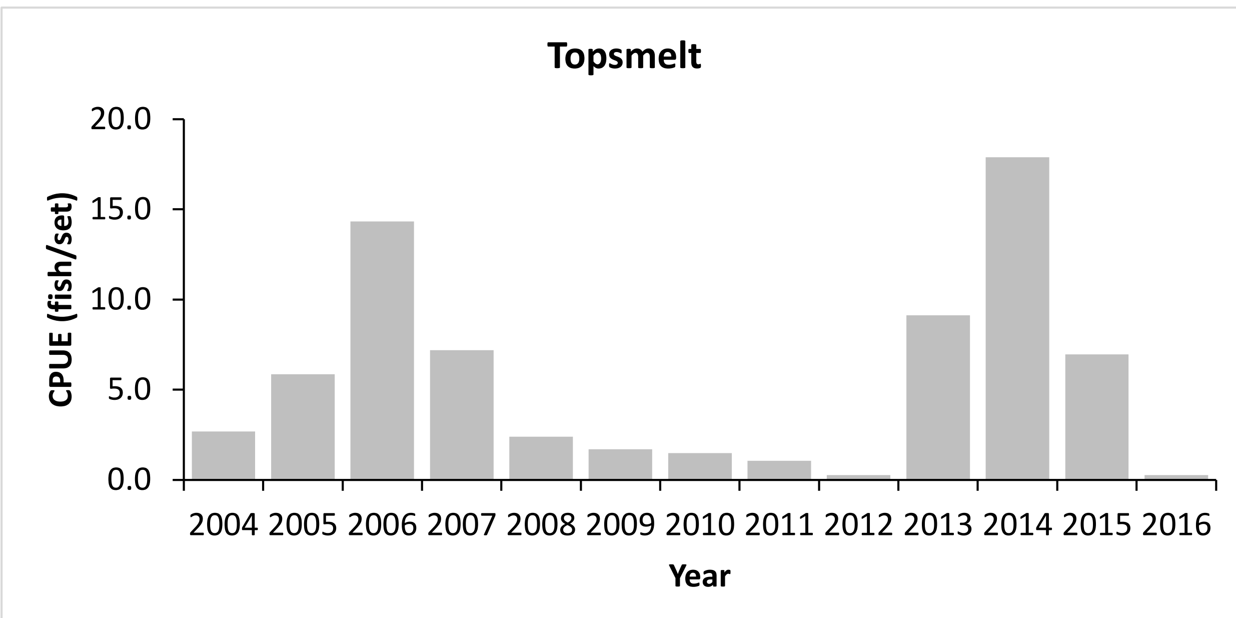

Figure 4.4.18. Annual abundance of topsmelt captured by beach seine in the Russian River Estuary, 2004- 2016. Samples are from 96 to 300 seine sets yearly from May to October.

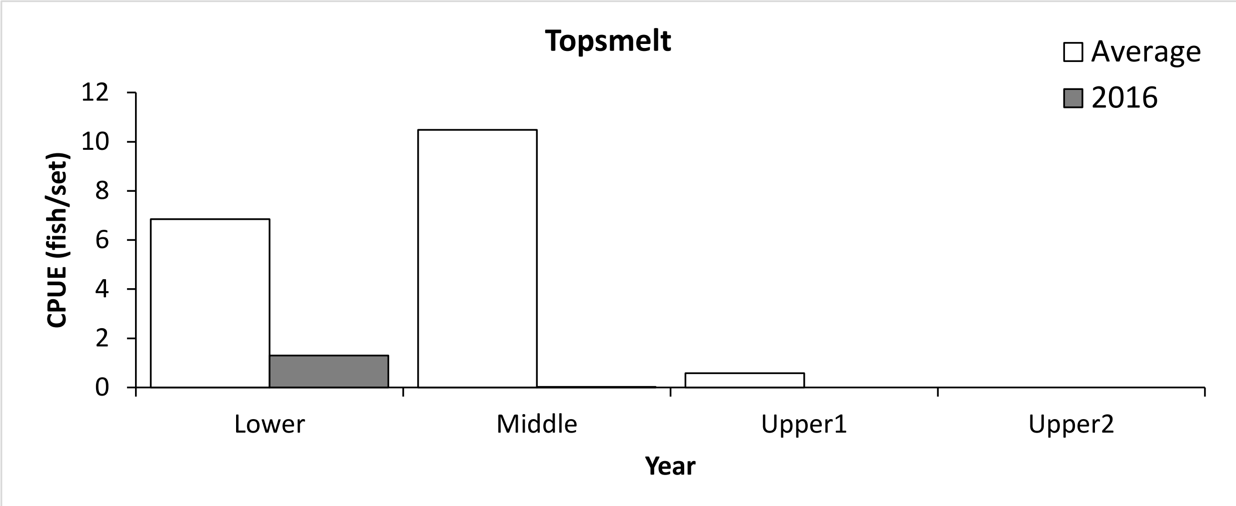

Figure 4.4.19. Spatial distribution of topsmelt in the Russian River Estuary, 2004-2016. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Reach during 2004 and 2009. Data from 2004 to 2015 were averaged.

in estuaries. They move into estuaries during the spring and generally prefer warm, low-salinity water or freshwater. As young grow, they shift to using brackish waters. The abundance of juvenile starry flounder in the Estuary has varied since studies began in 2004 (Figure 4.4.20). Juvenile flounder were relatively abundant in 2004, 2005 and 2016 with CPUEs greater than 10 fish/set. During the decade period from 2006 to 2015 abundances of flounder were below 2 fish/set. The Estuary appears to be utilized primarily by young-of-the-year fish where most flounder captures are less than 100 mm fork length. The seasonal occurrence of starry flounder was typically highest in May and June, and then gradually decreased through September and October when few were caught. Starry flounder were distributed throughout the Estuary ranging from the River Mouth in the Lower Reach, with cool seawater conditions, to the Upper Reach, with warm freshwater (Figure 4.4.21). Starry flounder have been detected as far as Austin Creek at the upstream end of the Estuary (Cook 2006).

Conclusions and Recommendations

The results of Estuary fish surveys from 2004 to 2015 found a total of 50 fish species from marine, estuarine, and riverine origins. The distribution of species was strongly influenced by the salinity gradient in the Estuary that is typically cool seawater near the mouth of the Russian River and transitions to warmer freshwater at the upstream end. Exceptions to this distribution pattern were anadromous and generalist fish that occurred throughout the Estuary regardless of salinity levels. The 2016 fish studies contribute to the 13-year dataset of existing conditions and our knowledge of a tidal brackish system. This baseline data will be used to compare with a closed mouth lagoon system.

The fluctuation in abundance of steelhead annually is likely attributed to the variability in adult spawner population size (i.e. cohort abundance), residence time of young steelhead before out-migration, and schooling behavior that affects susceptibility to capture by seining. A prolonged and severe drought likely contributed to the low abundance of steelhead and salmon in the Russian River Estuary in 2016. Chinook salmon smolts spent less than half the summer rearing in the Estuary and were usually absent after July. Based on the detection of these smolts at

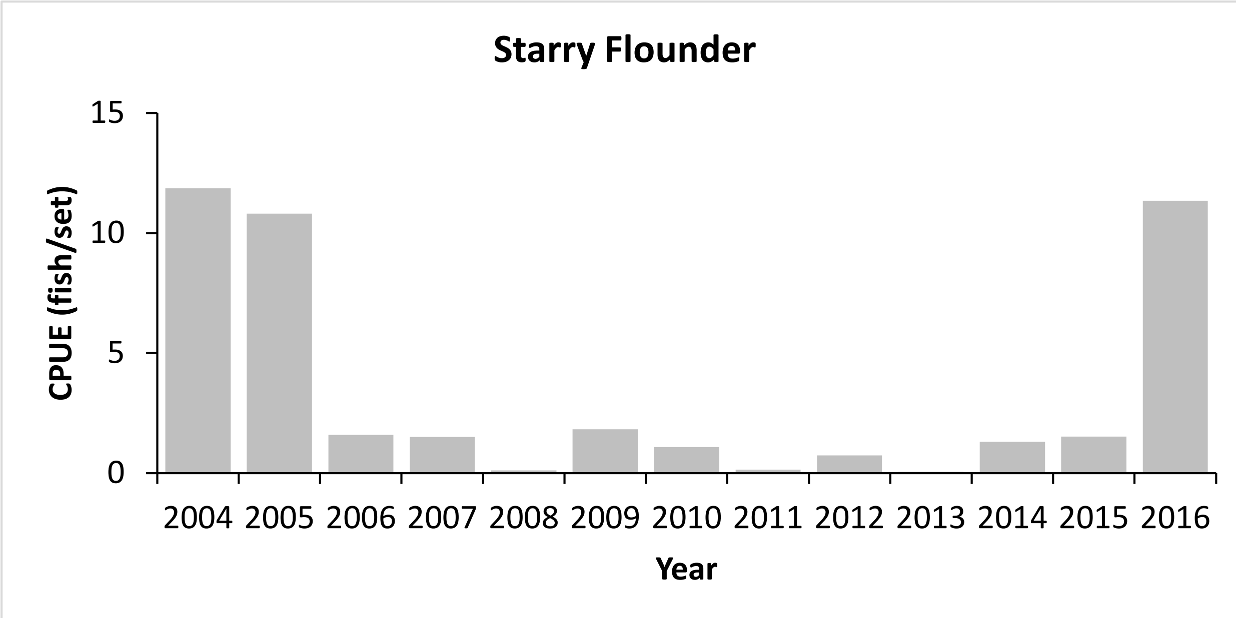

Figure 4.4.20. **Annual abundance of juvenile starry flounder captured by beach seine in the Russian River Estuary, 2004-2016. Samples are from 96 to 300 seine sets yearly from May to October.**

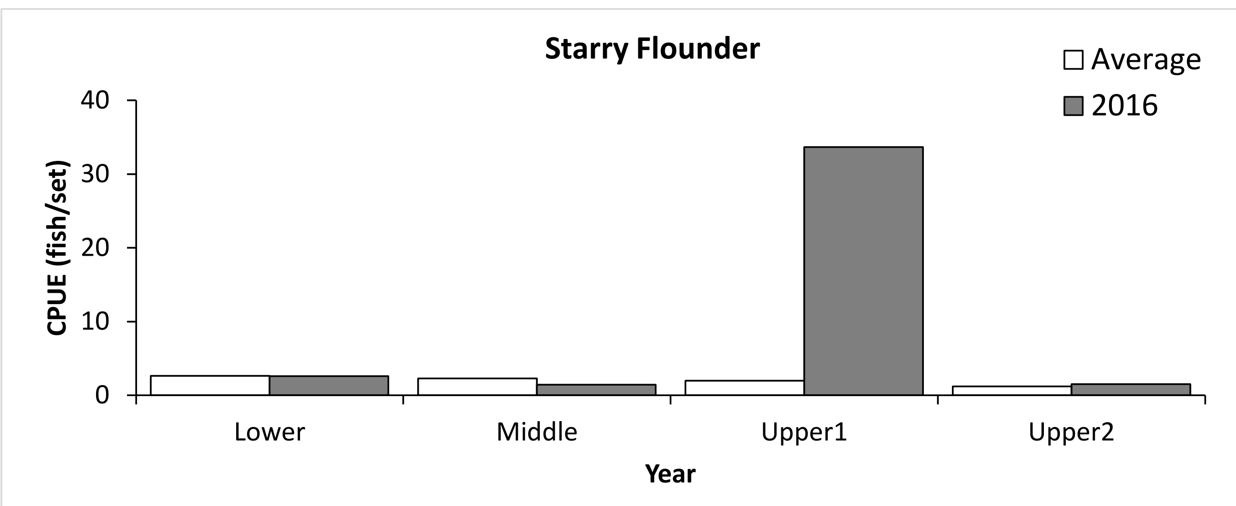

Figure 4.4.21. Spatial distribution of juvenile starry flounder in the Russian River Estuary, 2004-2016. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Sub-Reach during 2004 and 2009. Data from 2004 to 2015 were averaged.

most seining stations, they appear to use most estuarine habitats as they migrate to the ocean. In comparison, steelhead were found during the entire summer and were often found in the Upper Reach of the Estuary. However, there are sites in the Middle and Lower Estuary (e.g., Jenner Gulch confluence) where steelhead are consistently found.

Although beach seining is widely used in estuarine fish studies, beach seines are only effective near shore in relatively open water habitats free of large debris and obstructions that can foul or snag the net. Consequently, there is inherent bias in seine surveys (Steele et al. 2006). By design, our seining stations were located in areas with few underwater obstructions (i.e., large rocks, woody debris, etc.) and this likely influenced our assessment of fish abundance and habitat use. However, the spatial and temporal aspects of our sampling do allow quantitative comparisons among reaches and years.

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4.5 Downstream Migrant Trapping

The Reasonable and Prudent Alternative 2 in the Russian River Biological Opinion compels the Water Agency to provide information about the timing of downstream movements of juvenile steelhead into the Estuary, their relative abundance and the size/age structure of the population as related to the implementation of an adaptive management approach to beach management during the lagoon management season. The sampling design implemented by the Water Agency and described in this section specifically targets the detection and capture of anadromous salmonid young-of-the-year (YOY, age-0) and parr (≥age-1) (collectively referred to as juveniles) as well as smolts. In order to help accomplish the objectives listed above, the Water Agency undertook fish capture and PIT-tagging activities at selected trapping sites upstream of the estuary (Figure 4.5.1):

- Dry Creek (capture only)
- Mainstem Russian River at Mirabel (not operated in 2015)
- Mark West Creek
- Dutch Bill Creek
- Austin Creek

Stationary PIT antenna arrays were operated in the following locations:

- Mainstem Russian River at Northwood (19.16 rkm)
- Upstream end of the Russian River Estuary in Duncans Mills (10.46 rkm)
- Near the mouth of Austin Creek (0.5 rkm)

Implementation of the monitoring activities described here are the result of a continually-evolving process of evaluating and improving on past monitoring approaches. Descriptions and data from other monitoring activities conducted in the estuary (e.g., water quality monitoring, beach seining) are presented in other chapters of this report.

Methods

In 2016 we again relied on downstream migrant traps (DSMT) and stationary PIT antenna arrays at lower-Russian River basin trap sites to address the objectives in the RPA. Similar to 2010 through 2015, fish were physically captured at downstream migrant traps (rotary screw

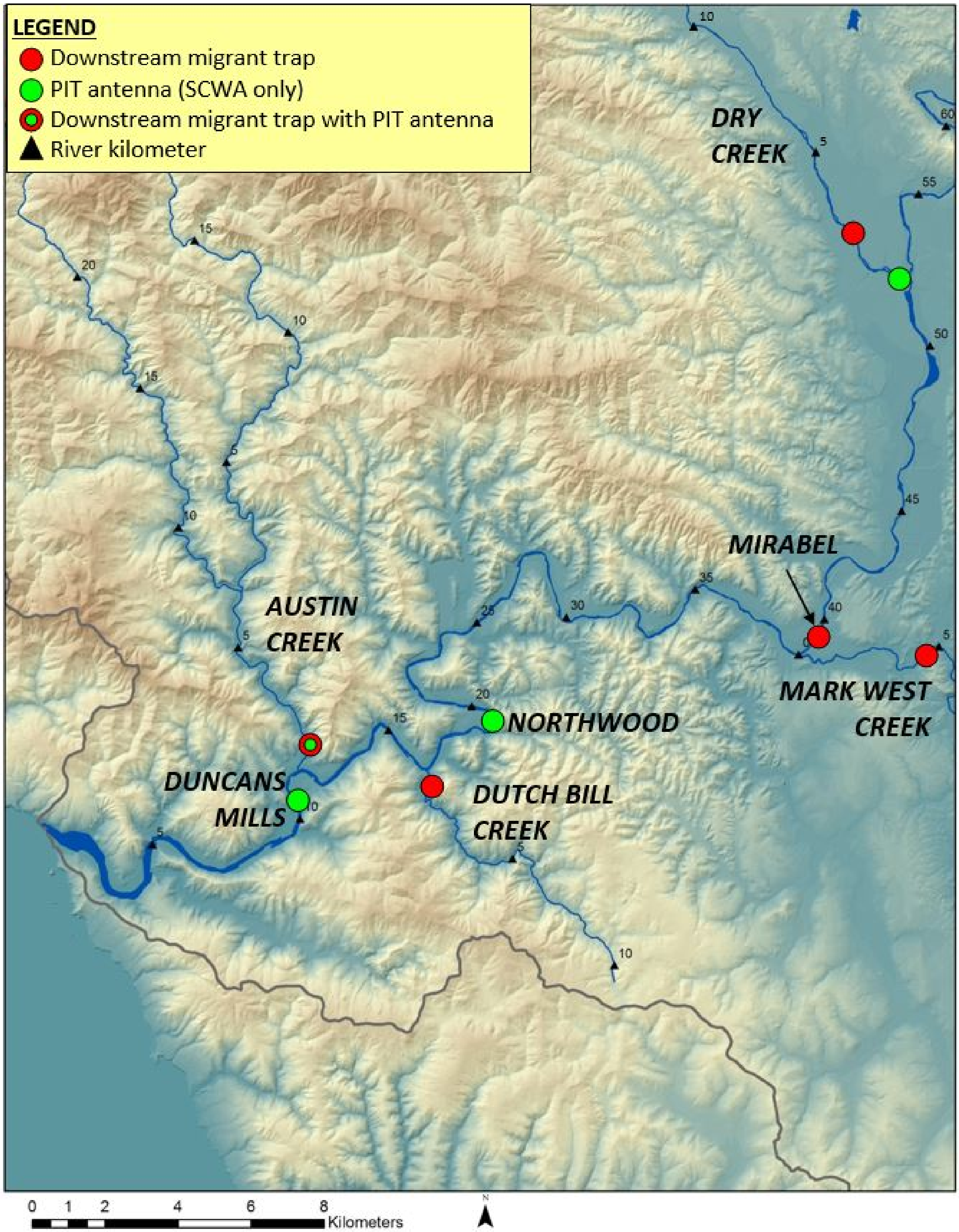


Figure 4.5.1. Downstream migrant detection sites in the lower Russian River, 2016. Numbered symbols along stream courses represent distance (rkm) from the mouth of each stream.

trap, funnel trap or pipe trap depending on the site), sampled for biological data and released. PIT tags were applied to a subset of age-0 steelhead captured at trap sites and fish were subject to detection at downstream PIT antenna arrays if they moved downstream into the estuary. In the sections that follow, we describe the sampling methods and analyses conducted for data collected at each site.

Estuary/Lagoon PIT antenna systems

Two antenna arrays with multiple flat plate antennas (antennas designed to lay flat on the stream bottom) were installed in the upper Russian River Estuary near the town of Duncans Mills (rkm 10.44 and 10.46) to detect PIT-tagged fish entering the estuary (Figure 4.5.2). Generally, 12 antennas were operated continuously from January 1 until May 28 (the period during which Austin Creek remained connected to the mainstem Russian River by surface flow). The orientation of the antennas consisted of 2 rows of six antennas with one row slightly upstream of the other. Each row contained 6 antennas placed side by starting at the west river bank and extending out into the channel.

Figure 4.5.2. Flat plate antenna arrays at Duncans Mills (rkm 10.44 and 10.46). Rectangles represent individual flat plate antennas.

As from 2013 to 2015, a dual flat plate PIT antenna array was operated in the mainstem Russian River in the vicinity of the golf course near the community of Northwood. The objective of this effort was to provide a means of detecting movements of juvenile steelhead that were PIT-tagged at upstream trap sites that may move into that portion of the mainstem of the Russian River that is non-tidal but can be inundated under perched lagoon or closed river mouth conditions. The antenna array consisted of two PIT antennas oriented so that they spanned approximately 75% of the wetted width of the river including the entire thalweg during open-mouth/non-perched conditions.

Lower River Fish Trapping and PIT tagging

Following consultation with NMFS and CDFW, the Water Agency identified three lower River tributaries (Mark West Creek, Dutch Bill Creek and Austin Creek, Figure 4.5.1) in which to operate fish traps as a way to supplement data collected from the Duncans Mills PIT antenna array and during sampling by beach seining throughout the estuary (Figure 4.5.2). In previous years downstream migrant traps were also operated at the Mirabel inflatable dam. However, a construction project to upgrade the fish ladder and water diversion intake screens precluded us from operating downstream migrant traps at this location. The Water Agency operated three types of downstream migrant traps in 2016: rotary screw trap, funnel trap and pipe trap depending on the stream, water depth, and velocity (Figure 4.5.3). Fish traps were checked daily by Water Agency staff during the trapping season (March through July). Captured fish were enumerated and identified to species and life stage at all traps. All PIT-tagged fish were measured for fork length (±1 mm) and weighed (±0.1 g). Additionally, a subset of all non-PIT-tagged individuals were measured and weighed each day. PIT tags were implanted in the majority of steelhead YOY and parr captured that were ≥60 mm in fork length.

Mainstem Russian River at Mirabel and Dry Creek at Westside Road

Typically two rotary screw traps (one 5 foot and one 8 foot) adjacent to one another have been operated on the mainstem Russian River immediately downstream of the Water Agency's inflatable dam site at Mirabel (approximately 38.7 rkm upstream of the river mouth in Jenner) (Figure 4.5.1). However, in 2016 active construction of a new fish ladder at Mirabel precluded operating a downstream migrant trap at this location. The Water Agency also operates a rotary screw trap at Dry Creek. The purpose of these trapping efforts is to fulfill a broader set of objectives in the Russian River Biological Opinion than what is described in the current section of this report.

Mark West Creek

A 5-foot rotary screw trap was installed on Mark West Creek approximately 4.8 km upstream of the mouth on April 6. On May 20 the rotary screw trap was removed and replaced with a pipe trap because of low water velocities. The pipe trap was removed and all trapping operations were suspended on June 23 when fish captures dropped off rapidly (Table 4.5.1).

Dutch Bill Creek

A pipe trap was installed on Dutch Bill Creek adjacent to the park in downtown Monte Rio (approximately 0.3 km upstream of the creek mouth) on March 29. The funnel net was removed and replaced with a pipe trap on May 23 because of low water velocity. The pipe trap was fished until the completion of trapping operations on June 23 when stream flow in lower Dutch Bill Creek became disconnected (Table 4.5.1).

Austin Creek

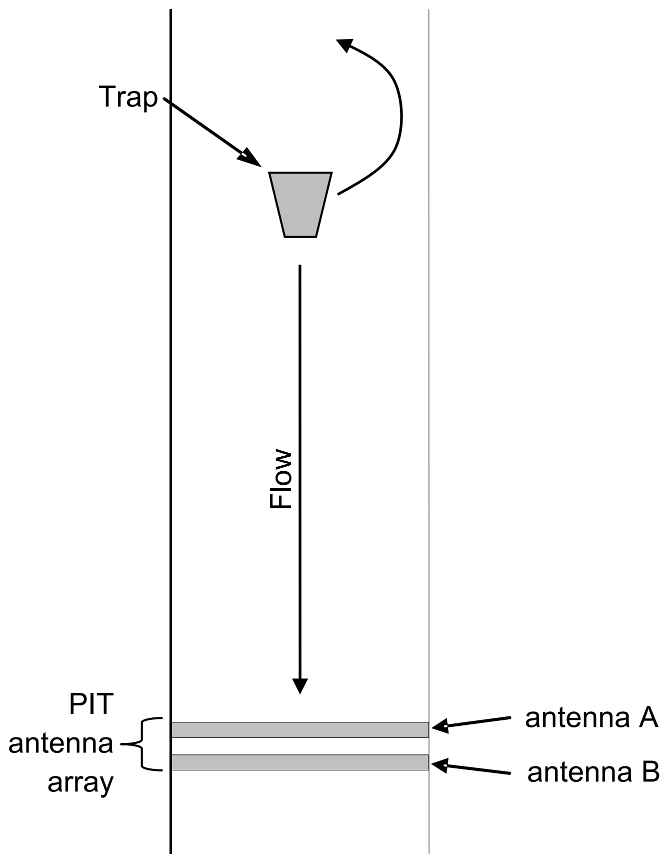
A rotary screw trap was installed in Austin Creek on April 8. Due to low water velocity this trap was changed to a funnel trap April 27. The funnel trap consisted of wood-frame/plastic-mesh weir panels, a funnel net and a wooden live box. Trapping continued until July 12 when surface flow in lower Austin Creek was no longer contiguous and daily catches of steelhead dropped to zero (Table 4.5.1).

Steelhead parr were marked with PIT tags and released upstream of the trap in order to measure trap efficiency and estimate population size of fish passing the trap site (Figure 4.5.4). We operated a dual PIT antenna array approximately 0.2 km downstream of the funnel trap and approximately 0.5 km upstream from the mouth of Austin Creek in order to detect PIT-tagged steelhead moving out of Austin Creek. The PIT antenna array was located at the upstream extent of the area that can be inundated by the Russian River during closure of the barrier

Figure 4.5.3. Photographs of downstream migrant traps operated by the Water Agency. Top: Mark West Creek rotary screw trap (operated April 6 – May 19) switched to pipe trap (operated May 20 - June 23). Middle: Dutch Bill Creek pipe trap (operated March 21-May 23). Bottom: Austin Creek funnel trap (operated April 8 - 27).

Table 4.5.1. Installation and removal dates, and total number of days fished for lower river monitoring sites operated by the Water Agency in 2016.

Monitoring site (gear type)	Installation date	Removal date	Number of days fished
Dry Creek (DSMT)	4/13	7/31	104
Mirabel (DSMT)	-	-	0
Mark West Creek (DSMT)	4/6	6/23	76
Northwood (PIT antenna array)	4/21	10/5	167
Dutch Bill Creek (DSMT)	3/29	6/23	87
Austin Creek (DSMT)	4/8	7/12	95
Duncans Mills (PIT antenna array)	continuous (not removed)	continuous (not removed)	entire downstream migration season



1. Methods:

Capture and PIT-tag juvenile steelhead, then release newly tagged fish upstream while releasing previously-tagged fish (recaptures) downstream.

2. Estimating trap efficiency:

Of the PIT-tagged fish released upstream of the trap, how many were recaptured in the trap before being detected on either antenna in the downstream antenna array?

3. Estimating antenna efficiency:

Of the PIT-tagged fish detected on the downstream antenna in the array (antenna B), how many were also detected on the upstream antenna (antenna A).

Figure 4.5.4. Diagram illustrating the relative location of the downstream migrant trap and PIT antenna array operated on Austin Creek and outline of how antenna efficiency was estimated.

beach; therefore, we assumed that once fish passed the antenna array they had effectively entered the estuary/lagoon. A second PIT tag antenna array located in the Russian River Estuary at Duncans Mills (approximately 1.5 km downstream) was used to calculate antenna efficiency for the PIT antenna array located in Austin Creek.

Results

Stream flow largely dictates when downstream migrant traps can be installed (Figure 4.5.5). Our sampling period most likely encompassed a high portion of the juvenile steelhead movement period but we probably missed a substantial portion of the steelhead smolt migration period.

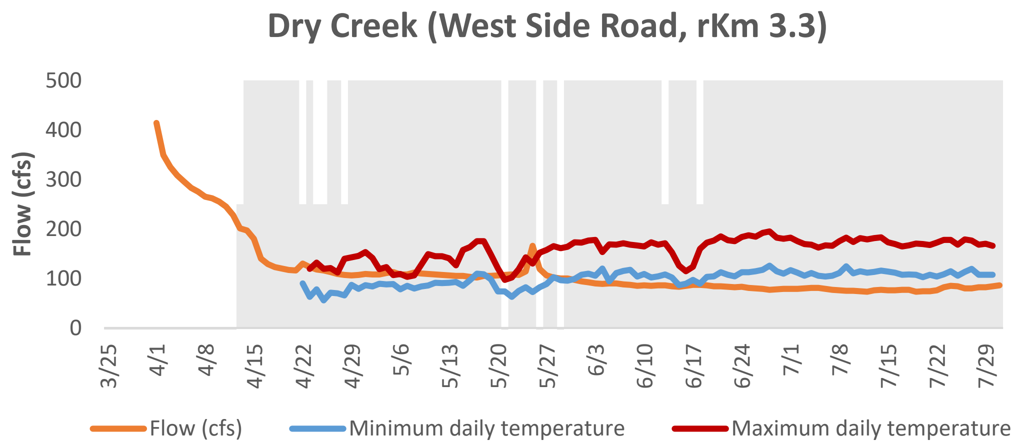
Estuary/Lagoon PIT antenna systems

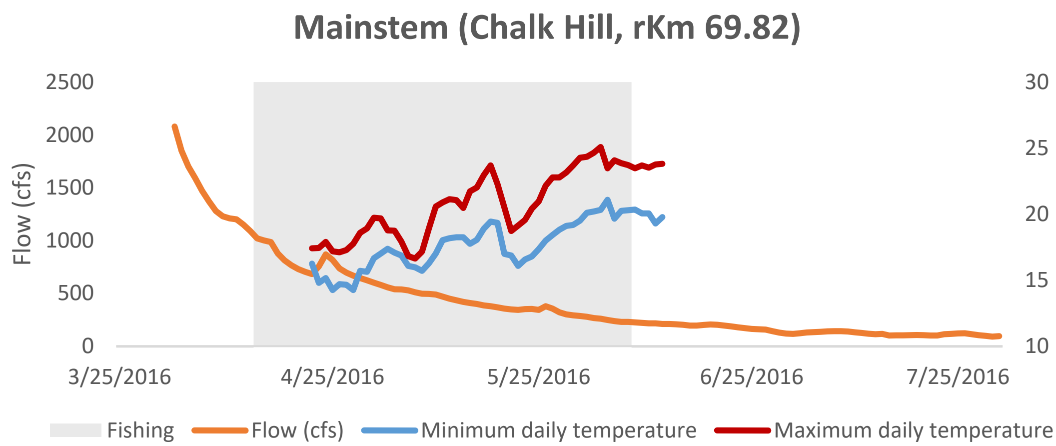
Steelhead

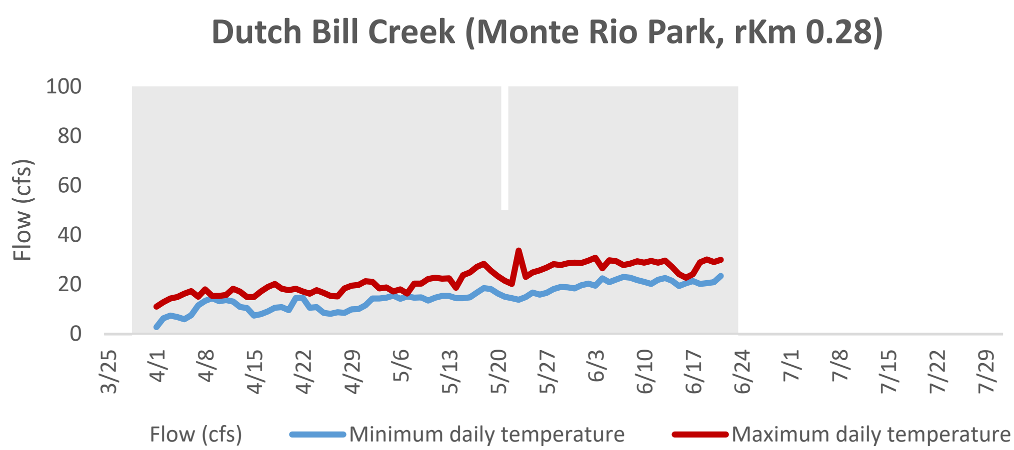
Steelhead were most frequently encountered at Dry Creek than any other trap. In total 4,221 YOY and parr, and 106 smolts were captured at the Dry Creek trap. In Austin Creek 3,798 juveniles and 201 smolts were captured while only 74 juvenile and 8 smolt steelhead were captured in Dutch Bill Creek. At Mark West Creek 509 YOY and parr, and 150 smolts were captured (Figure 4.5.6). Of the 1,797 juvenile steelhead that were PIT-tagged in downstream migrant traps in 2016, 137 (7.6%) were detected on the PIT antenna array at Duncans Mills, effectively entering the Estuary (Table 4.5.2). Reasons for non-detection include an unknown number of fish that simply did not move into the estuary as well as fish that moved into the tidal portion of the estuary but were not detected due to imperfect PIT antenna array detection efficiency at Duncans Mills.

Many steelhead juveniles were captured in Austin Creek in 2016. Over the course of the season, 3,999 steelhead were captured of which 3,520 were YOY (2,427 of the 3,520 YOY were ≥60 mm). Although we applied PIT tags to 1,205 total individuals (YOY+parr), we estimate that, based on their size, 993 of these PIT tagged fish were YOY. In total, 1,132 PIT-tagged steelhead were released upstream of the trap and 73 were released downstream of the trap (Table 4.5.4). Because 193 of the 1,132 PIT-tagged YOY were detected on the PIT antenna array just downstream of the trap in Austin Creek, we have high certainty that at least 17% (193/1,132) moved downstream into the estuary/lagoon. Because of imperfect antenna detection efficiency, we expanded those minimum counts that were based only on PIT-tagged YOY to the entire population of YOY in the vicinity of the Austin Creek trap (both tagged and untagged) as follows.

Of the 205 PIT tagged individuals (YOY+parr) detected on the downstream antenna in the array (Duncans Mills, Table 4.5.3), 76 were also detected on the upstream antenna array (Austin Creek) resulting in an estimated antenna efficiency of 37.1% (76/205). In order to estimate the number of YOY out of the original 1,132 that actually moved downstream of the Austin antenna array, we used this proportion to expand the 193 detections to 520 (193/37.1%).


Dry Creek (West Side Road, rKm 3.3)
Flow (cfs)
Temperature (°C)
500
400
300
200
100
0
30
25
20
15
10
3/25
4/1
4/8
4/15
4/22
4/29
5/6
5/13
5/20
5/27
6/3
6/10
6/17
6/24
7/1
7/8
7/15
7/22
7/29
Flow (cfs)
Minimum daily temperature
Maximum daily temperature


Mainstem (Chalk Hill, rKm 69.82)
Flow (cfs)
Temperature (°C)
2500
2000
1500
1000
500
0
30
25
20
15
10
3/25/2016
4/25/2016
5/25/2016
6/25/2016
7/25/2016
Fishing
Flow (cfs)
Minimum daily temperature
Maximum daily temperature


Dutch Bill Creek (Monte Rio Park, rKm 0.28)
Flow (cfs)
Temperature (°C)
100
80
60
40
20
0
30
25
20
15
10
3/25
4/1
4/8
4/15
4/22
4/29
5/6
5/13
5/20
5/27
6/3
6/10
6/17
6/24
7/1
7/8
7/15
7/22
7/29
Flow (cfs)
Minimum daily temperature
Maximum daily temperature

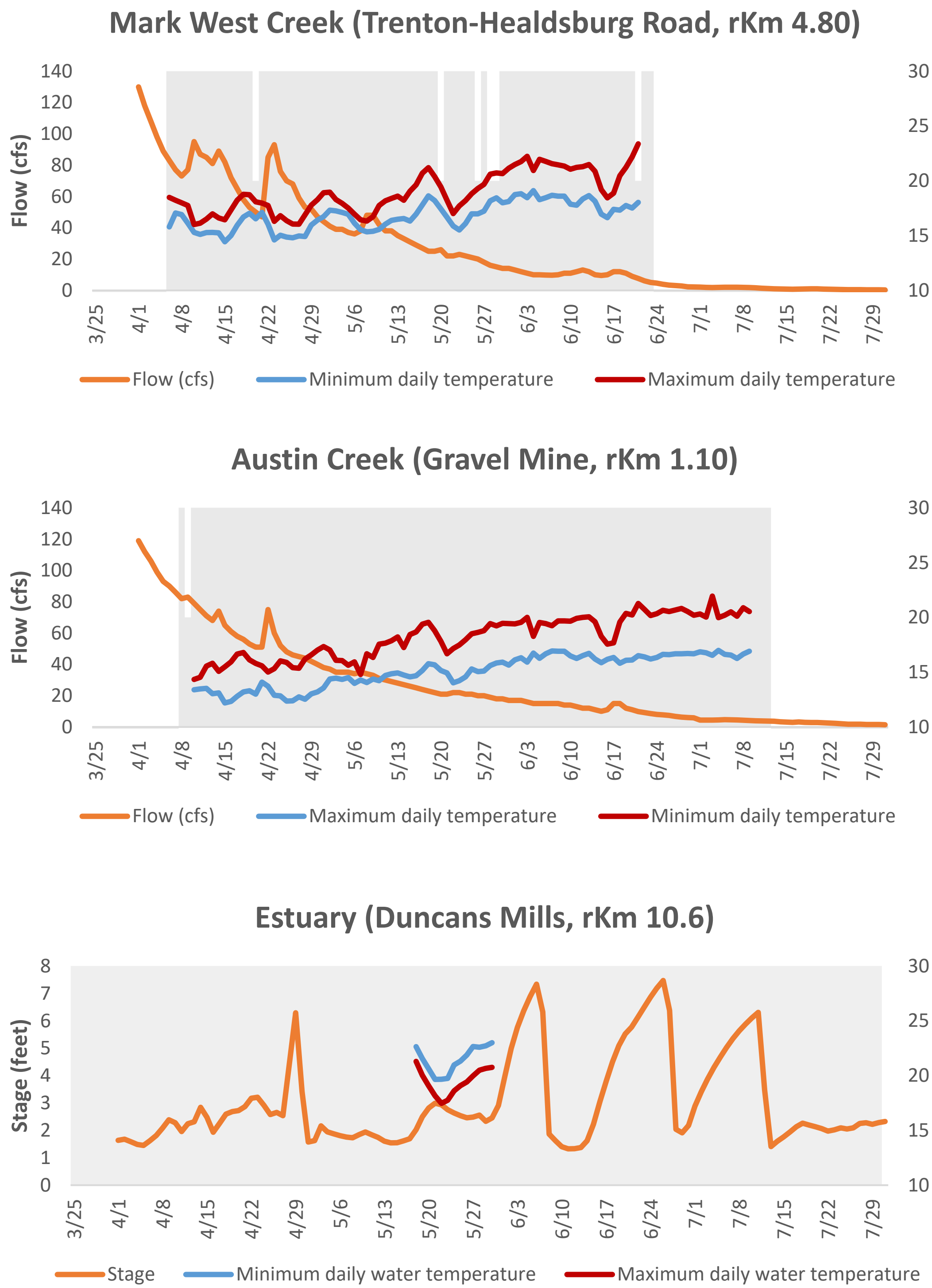
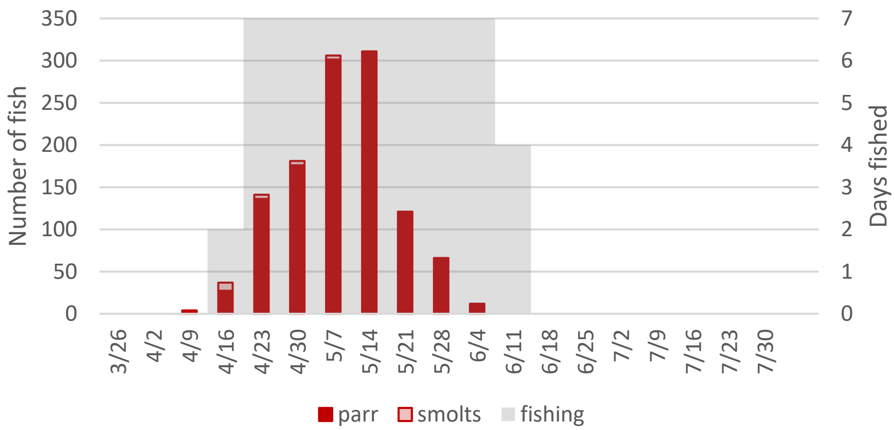
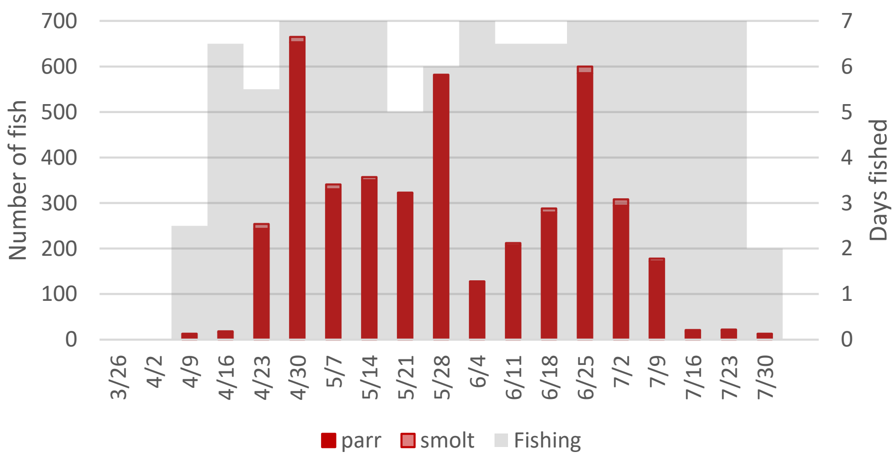


Figure 4.5.5. Environmental conditions at downstream migrant detection sites from March 25 to July 31. Gray shading indicates the proportion of each day that each facility was operated. Discharge data are from the USGS gage at Healdsburg (mainstem Russian, 11464000), the USGS gage at Trenton-Healdsburg Road (Mark West Creek, 11466800), a gage operated by CMAR on Dutch Bill Creek (data unavailable in 2016) and the USGS gage at Cazadero (Austin Creek, 11467200). Stage data for the estuary are from the Jenner gage. Temperature data are from the data loggers operated by the Water Agency at each monitoring site.

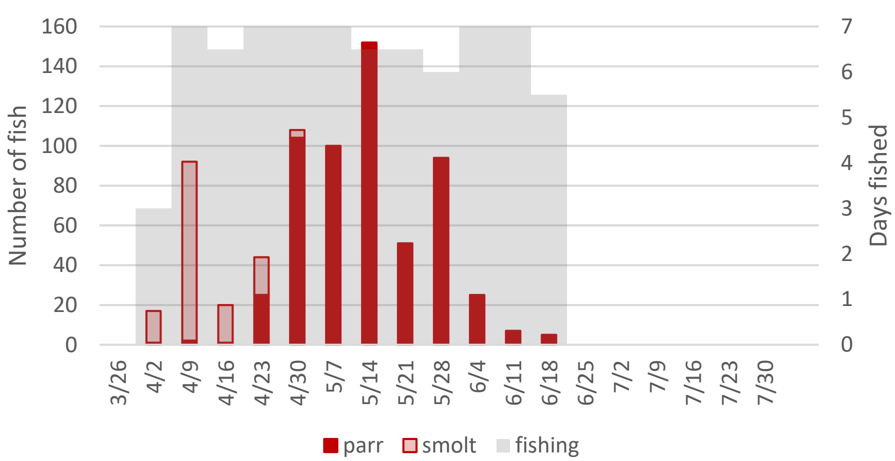
Mainstem Russian River (Chalkhill, RiverKm 70.11)
1,151 YOY+Parr, 28 Smolts

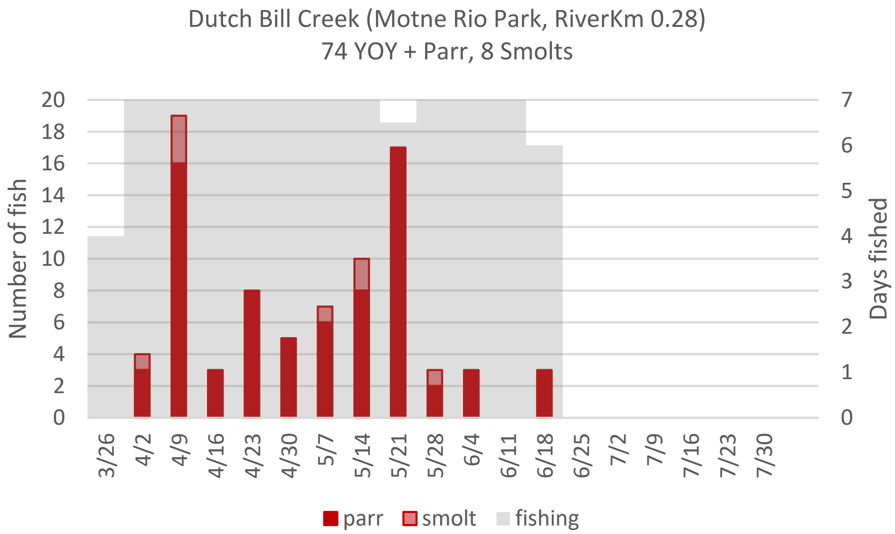


Dry Creek (West Side Road, RiverKm 3.3)
4,221 YOY+Parr, 106 Smolt



Mark West Creek (Trenton-Healdsburg Road, Riverkm 4.80)
590 YOY+Parr, 150 smolt





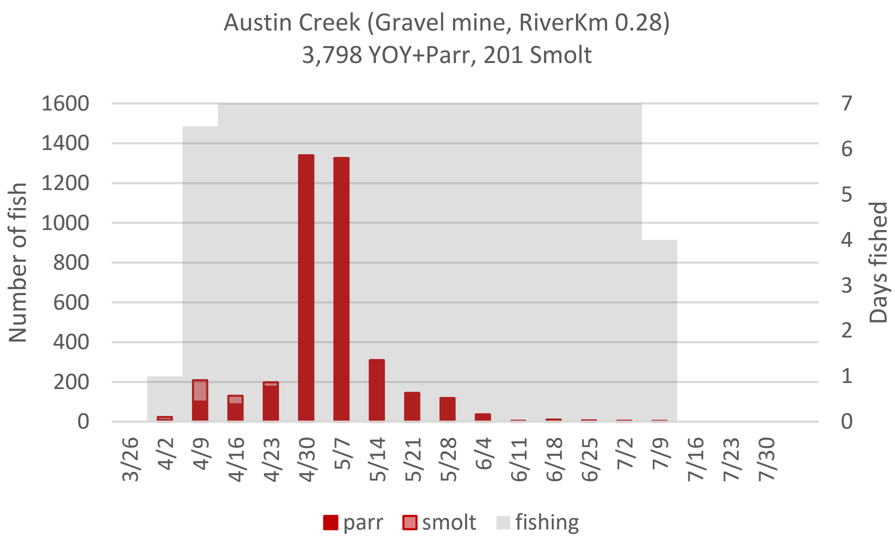


Figure 4.5.6. Weekly capture of steelhead by life stage at lower river downstream migrant trapping sites, 2016. Gray shading indicates portion of each week trap was fishing. Note the different vertical scale among plots for each site.

Table 4.5.2. Number of steelhead juveniles PIT-tagged at downstream migrant traps, 2009-2016.

Site	2009	2010	2011	2012	2013	2014	2015	2016
Dry Creek	no tagging	no tagging	no tagging	no tagging	2,703	1,348	no tagging	no tagging
Mainstem	5	96	99	315	100	101	not fished	not fished
Mark West Creek	not fished	not fished	not fished	43	135	18	19	546
Dutch Bill Creek	not fished	46	22	6	12	21	7	46
Austin Creek	not fished	996	500	1,636	1,749	590	107	1,205
Total	5	1,138	621	2,000	4,699	2,078	133	1,797

Table 4.5.3. Number of steelhead captured at downstream migrant traps, number PIT tagged and number detected on the Duncans Mills PIT tag detection system prior to October 15, 2016.

Site	Number Captured	Number PIT-Tagged	Number (proportion) Detected at Duncans Mills
Mainstem	-	-	-
Mark West Creek	590	546	1 (0.1%)
Dutch Bill Creek	74	46	1 (2.2%)
Austin Creek	3,798	1,205	135 (11.2%)
Total	**4,462**	**1,797**	**137 (7.6%)**

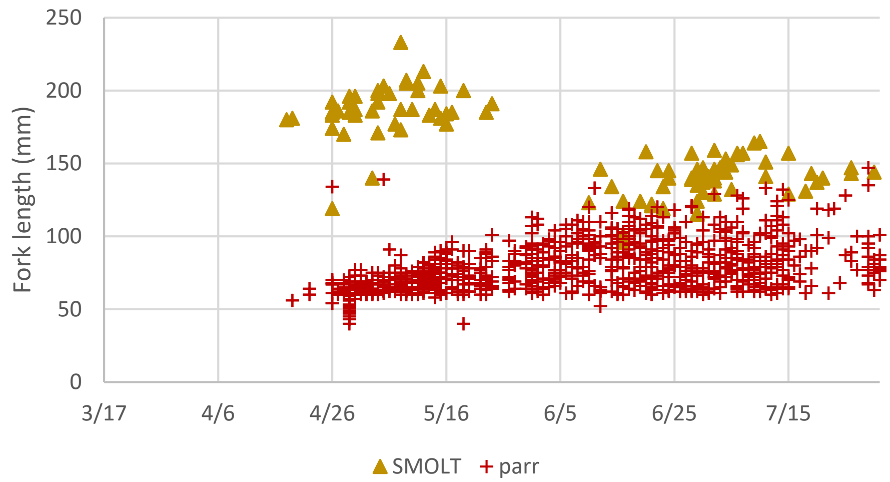
Of the YOY detected on either the downstream PIT antenna arrays that were also released upstream of the trap, 60 were recaptured in the trap resulting in a trap efficiency of 39.7%. Based on this trap efficiency we expand the 2,427 steelhead YOY captured at the trap to a population estimate of 6,113. Using the percentage of emigrants from the PIT tagged population we expect that 2,812 steelhead YOY (46% of the 6,113 steelhead YOY trap estimate) emigrated from Austin Creek to the estuary.

When compared to Austin and Dry Creeks fewer numbers of juvenile steelhead were captured at Mark West and Dutch Bill Creeks (Figure 4.5.6) meaning that fewer numbers of juvenile steelhead were PIT-tagged at these locations (Table 4.5.3). Fork lengths of fish caught at these traps show at least 3 year classes with steelhead YOY present at each of the trapping locations (Figure 4.5.7). As in other years, we assume that the few steelhead smolts captured at any of the trap sites was likely due to a large portion of the smolt outmigration occurring before trap installation and the generally low trap efficiencies for steelhead smolts that is well-documented in the Russian River and elsewhere. The season total catches of steelhead have been variable over the course of years monitored (Figure 4.5.8 through Figure 4.5.12).

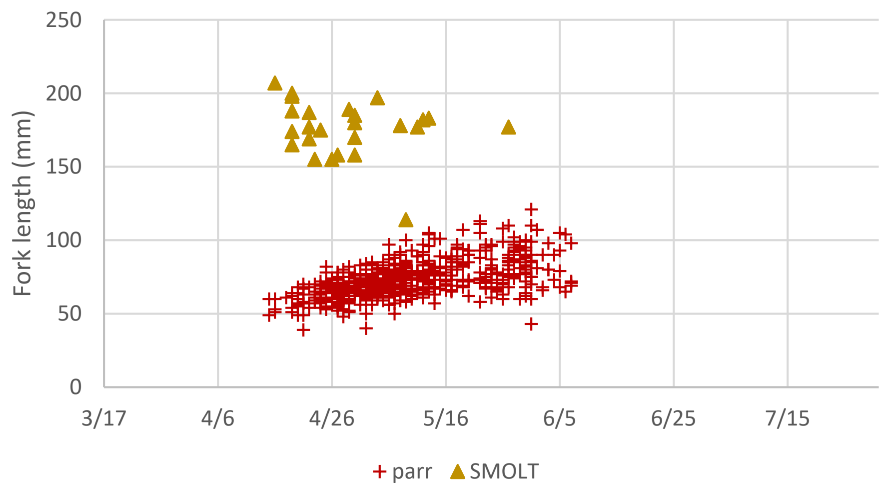
Coho Salmon

At Dry Creek 227 hatchery smolts, 13 wild smolts, 1 smolts of unknown origin, 16 parr of unknown origin and 22 wild parr were detected at the trap (Figure 4.5.8 and Figure 4.5.13). At Mark West Creek, 37 hatchery smolts, 16 smolts of unknown origin, and 5 wild YOY/parr were detected at the trap (Figure 4.5.10 and Figure 4.5.13). A total of 2,581 hatchery smolts, 15 smolt of unknown origin, 85 wild smolts, 1 hatchery YOY/parr and 15 YOY/parr of unknown origin, and 2 wild YOY/parr were captured at the Dutch Bill Creek trap (Figure 4.5.12 and Figure 4.5.13). At Austin Creek, 144 hatchery smolts, 3 smolts of unknown origin, 32 wild smolts, 809 hatchery YOY/parr, and 25 YOY/parr of unknown origin, and 105 wild YOY/parr were captured (Figure 4.5.12 and Figure 4.5.13). Based on length data collected at the lower river traps, there were at least two age groups (YOY: age-0 and parr/smolt: ≥age-1) of coho captured (Figure 4.5.14). For a more detailed analysis of downstream migrant trapping catches of coho from other Russian River streams see UCCE Coho Salmon Monitoring Program results for 2016.

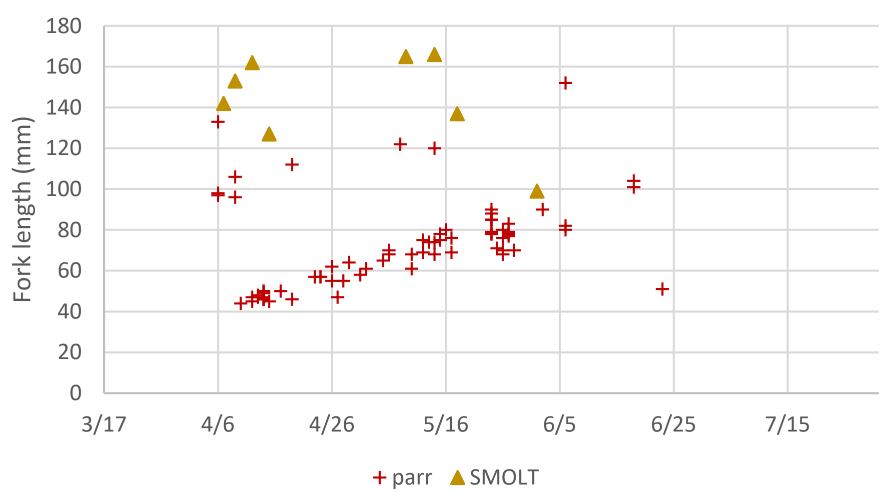
Dry Creek (West Side Road, RiverKm 3.30)

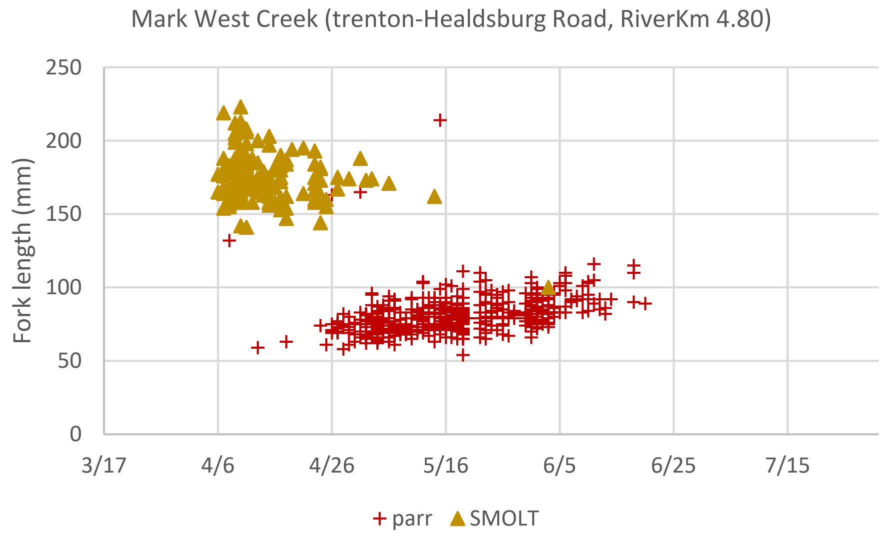


Mainstem Russian River (Chalk Hill, RiverKm 70.11)



Dutch Bill Creek (Monte Rio Park, RiverKm 0.28





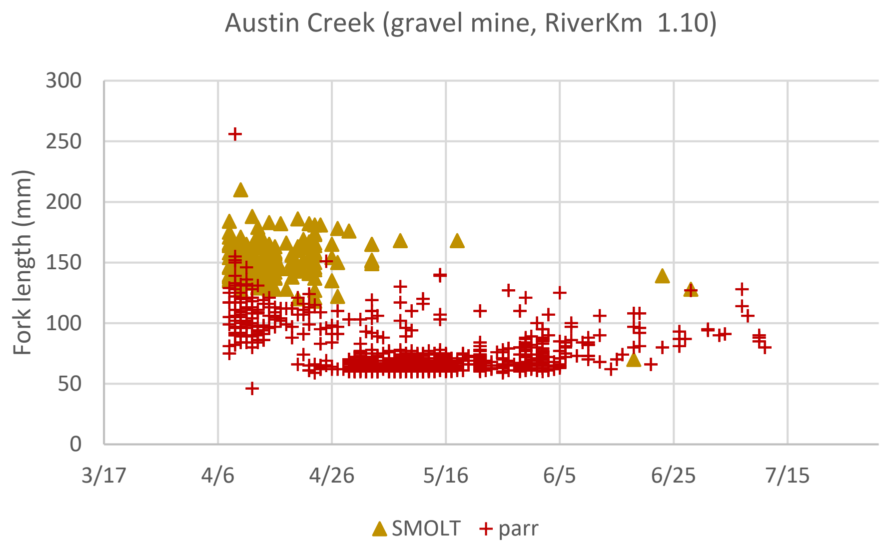
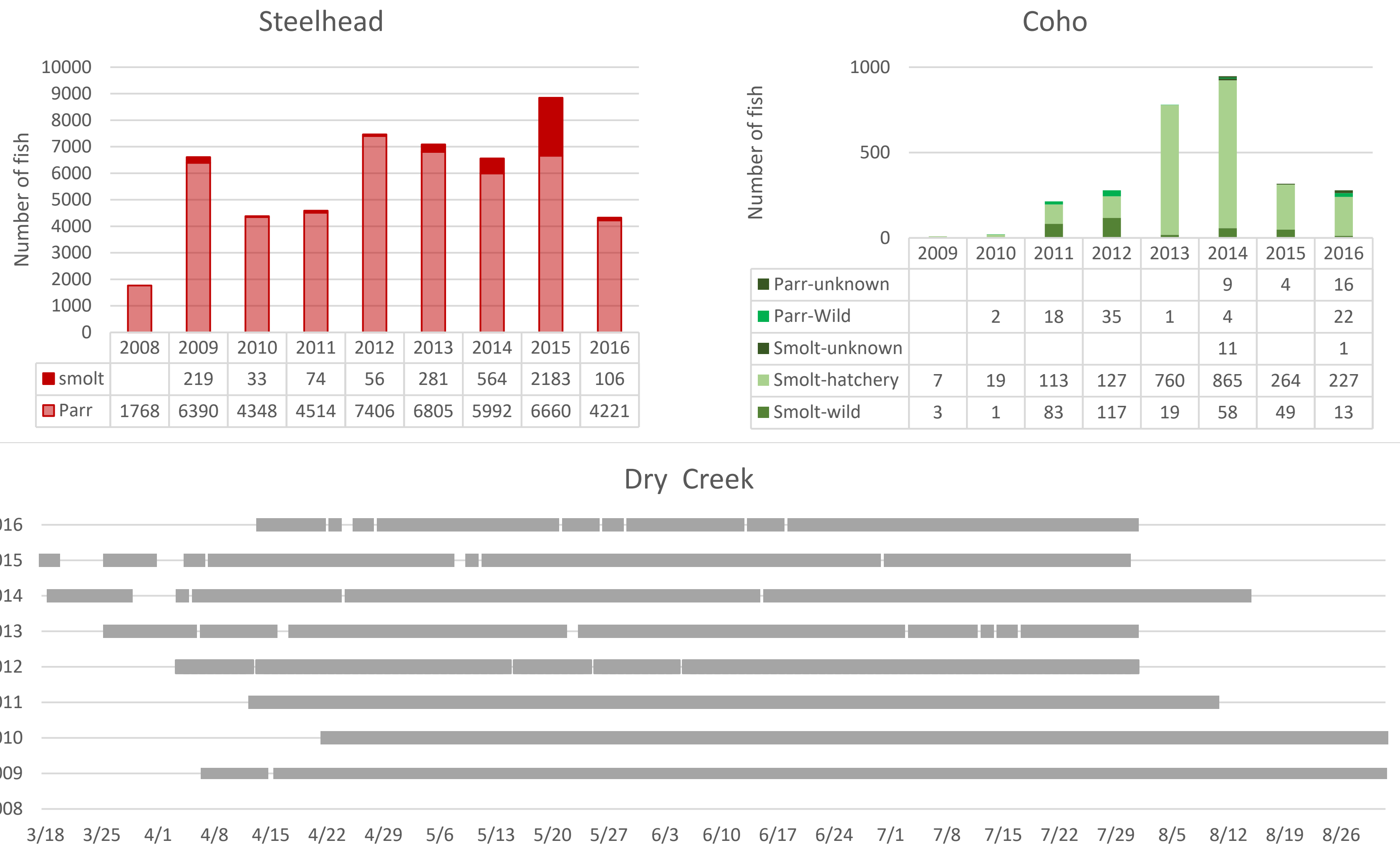


Figure 4.5.7. Weekly fork lengths of juvenile steelhead captured at lower river downstream migrant trap sites, 2016.

Table 4.5.4. PIT tag and trap capture metrics and values for YOY steelhead in Austin Creek. Note that 2010 numbers differ from Martini-Lamb and Manning (2011) because they have been adjusted to only include YOY.

Metric	2010	2011	2012	2013	2014	2015	2016
Number PIT-tagged YOY released upstream of trap	765	324	1,356	0	214	101	1,132
Number PIT-tagged YOY released downstream of trap	195	2	162	1,746	269	6	73
Number PIT-tagged YOY detected on antenna array that were tagged in Austin Creek	547	131	574	1,335	275	13	193
Number PIT-tagged YOY released upstream & detected on antenna array	389	131	486	0	57	13	151
Number released upstream & recaptured in trap & detected on antenna	47	8	196	0	2	0	60
ESTIMATED TRAP EFFICIENCY	12.1%	6.1%	40.3%	N/A	N/A	N/A	39.7%
Number YOY+parr detected on both antennas in array	241	93	85	399	129	34	76
Number YOY+parr detected on downstream antenna only	288	178	129	463	162	35	205
ESTIMATED ANTENNA EFFICIENCY	83.6%	52.2%	65.9%[1]	86.2%[1]	79.6%[1]	97.1%	37.1%[1]
Number YOY captured and PIT-tagged	960	324	1,518	1,746	483	42	993
Total number of YOY captured (≥60 mm only)	2,617	453	2,341	4,216	541	42	2,427
ESTIMATED NUMBER OF PIT-TAGGED YOY EMIGRANTS (≥60 mm only)	632	251	759	1,549	325	32	520
ESTIMATED PROPORTION OF PIT-TAGGED YOY THAT EMIGRATED (≥60 mm only)	65.8%	77.5%	50%	88.5%	67.3%	76.2%	46.0%
ESTIMATED POPULATION SIZE OF YOY AT TRAP	21,628	7,426	5,804	N/A	N/A	N/A	6,113
ESTIMATED NUMBER OF YOY IN POPULATION THAT EMIGRATED	14,231	5,755	2,901	N/A	N/A	N/A	2,812

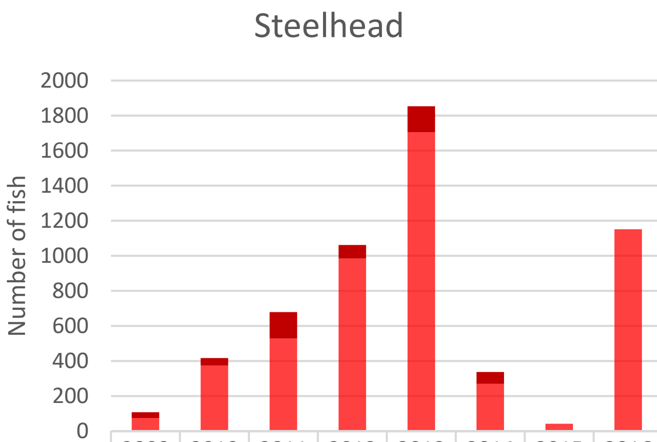
[1]Efficiency is based on detections of PIT-tagged fish at Duncans Mills



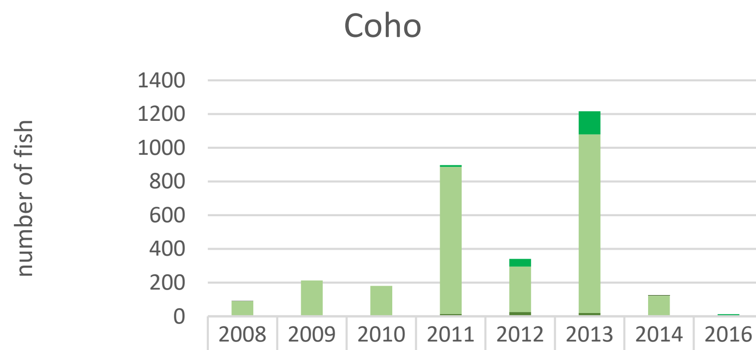
	2008	2009	2010	2011	2012	2013	2014	2015	2016
smolt		219	33	74	56	281	564	2183	106
Parr	1768	6390	4348	4514	7406	6805	5992	6660	4221

	2009	2010	2011	2012	2013	2014	2015	2016
Parr-unknown						9	4	16
Parr-Wild		2	18	35	1	4		22
Smolt-unknown						11		1
Smolt-hatchery	7	19	113	127	760	865	264	227
Smolt-wild	3	1	83	117	19	58	49	13

Figure 4.5.8. Number of steelhead and coho salmon captured by life stage and origin at the Dry Creek downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2016.



	2009	2010	2011	2012	2013	2014	2015	2016
smolt	33	42	151	78	148	67		
parr	75	375	528	984	1705	270	42	1151



	2008	2009	2010	2011	2012	2013	2014	2016
Parr-unknown							1	
Parr-wild				10	45	137	1	11
Smolt-unknown	3						1	
Smolt-hatchery	89	208	180	872	270	1059	117	
Smolt-wild	2	5	1	15	26	20	6	2

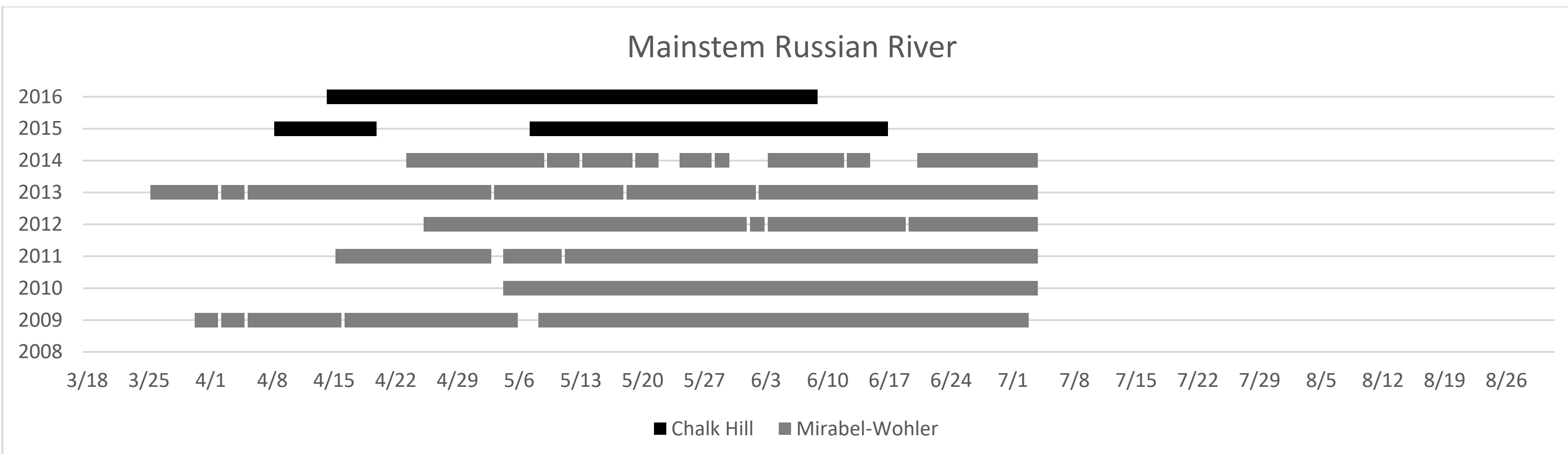
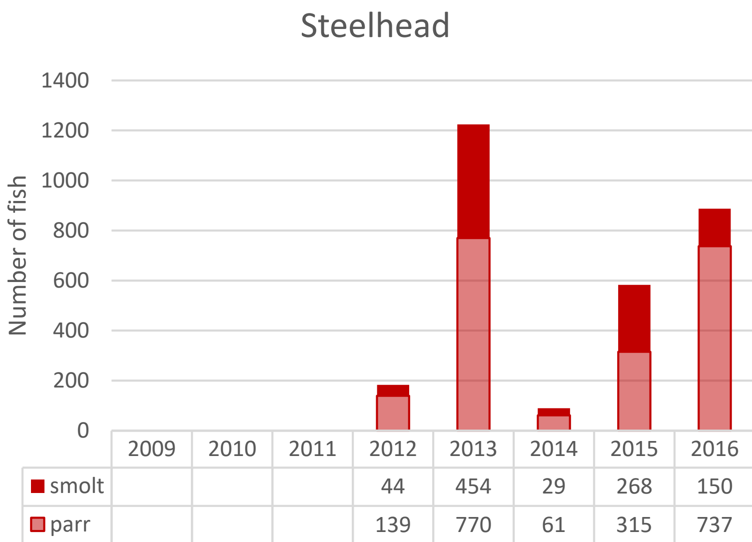
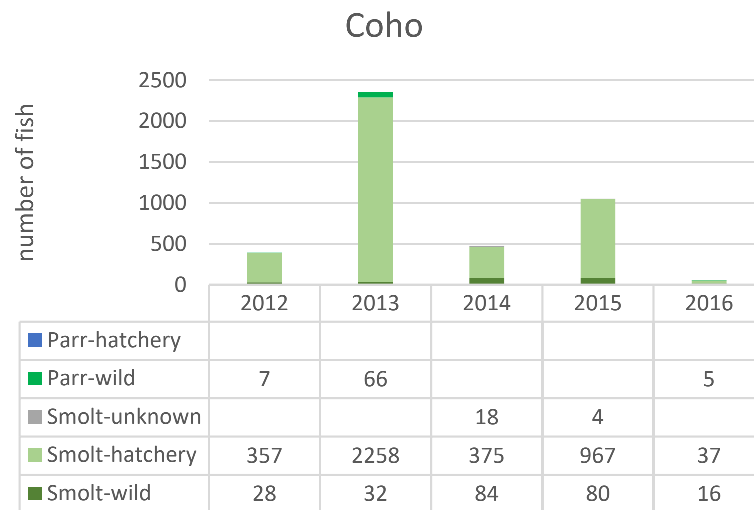


Figure 4.5.9. Number of steelhead and coho salmon captured by life stage and origin at the mainstem Russian River at Chalk Hill and Mirabel-Wohler downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2016.



	2009	2010	2011	2012	2013	2014	2015	2016
smolt				44	454	29	268	150
parr				139	770	61	315	737



	2012	2013	2014	2015	2016
Parr-hatchery					
Parr-wild	7	66			5
Smolt-unknown			18	4	
Smolt-hatchery	357	2258	375	967	37
Smolt-wild	28	32	84	80	16

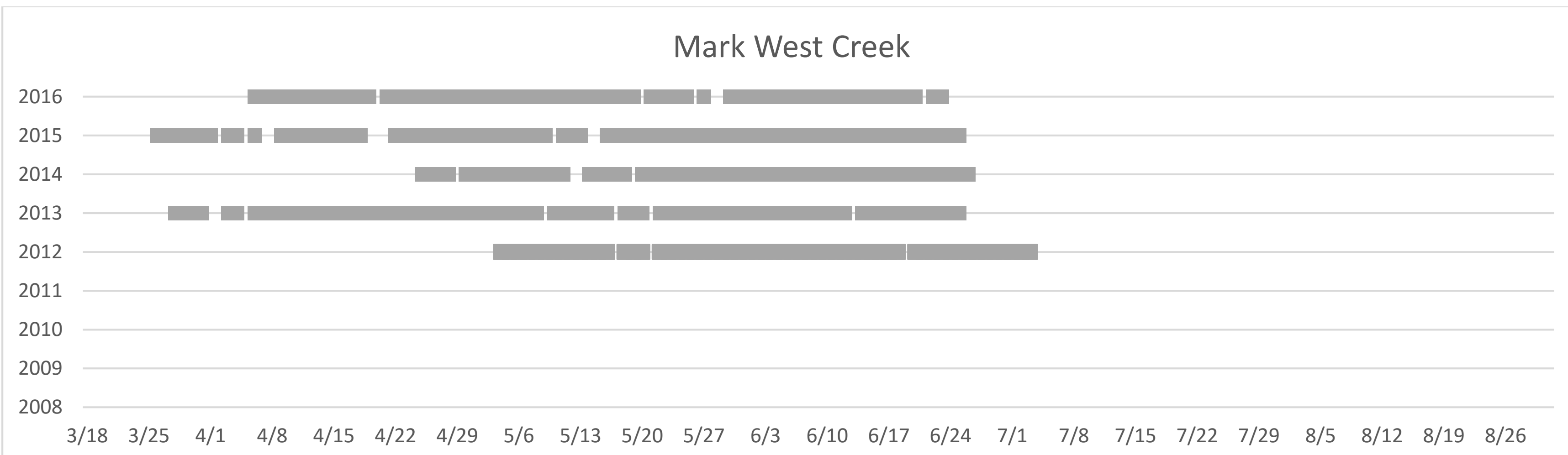


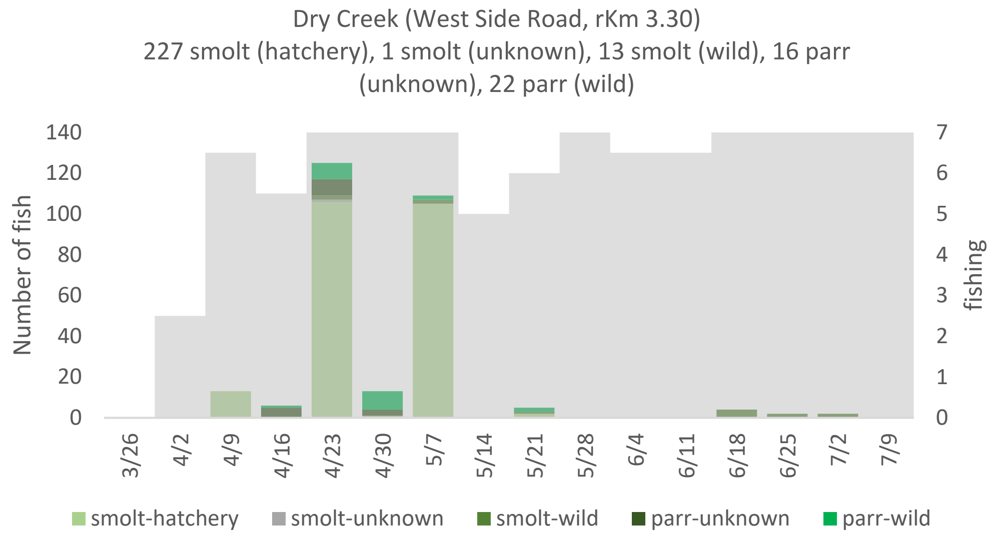
Figure 4.5.10. Number of steelhead and coho salmon captured by life stage and origin at the Mark West Creek downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2016.

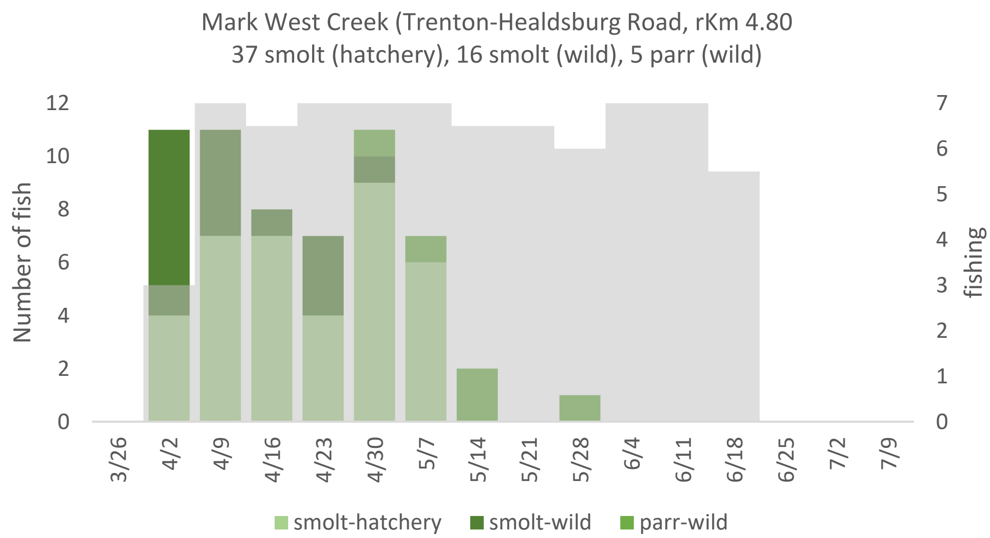
	2009	2010	2011	2012	2013	2014	2015	2016
smolt		5	47	11	18		3	
parr		58	31	33	79	1138	13	74

	2010	2011	2012	2013	2014	2015	2016
Parr-hatchery	39						1
Parr-unknown							15
Parr-Wild		5		2			2
Smolt-unknown		4			9	1	15
Smolt-hatchery	184	2904	1952	717	1668	192	2581
Smolt-wild	1		35	106	262	8	85

Figure 4.5.11. Number of steelhead and coho salmon captured by life stage and origin at the Dutch Bill Creek downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2016.

Figure 4.5.12. Number of steelhead and coho salmon captured by life stage and origin at the Austin Creek downstream migrant trap (upper panels), and duration and timing of trap operation (lower panel), 2009-2016.


Dry Creek (West Side Road, rKm 3.30)
227 smolt (hatchery), 1 smolt (unknown), 13 smolt (wild), 16 parr (unknown), 22 parr (wild)
Number of fish
fishing
140
120
100
80
60
40
20
0
7
6
5
4
3
2
1
0
3/26
4/2
4/9
4/16
4/23
4/30
5/7
5/14
5/21
5/28
6/4
6/11
6/18
6/25
7/2
7/9
smolt-hatchery
smolt-unknown
smolt-wild
parr-unknown
parr-wild


Mark West Creek (Trenton-Healdsburg Road, rKm 4.80
37 smolt (hatchery), 16 smolt (wild), 5 parr (wild)
Number of fish
fishing
12
10
8
6
4
2
0
7
6
5
4
3
2
1
0
3/26
4/2
4/9
4/16
4/23
4/30
5/7
5/14
5/21
5/28
6/4
6/11
6/18
6/25
7/2
7/9
smolt-hatchery
smolt-wild
parr-wild

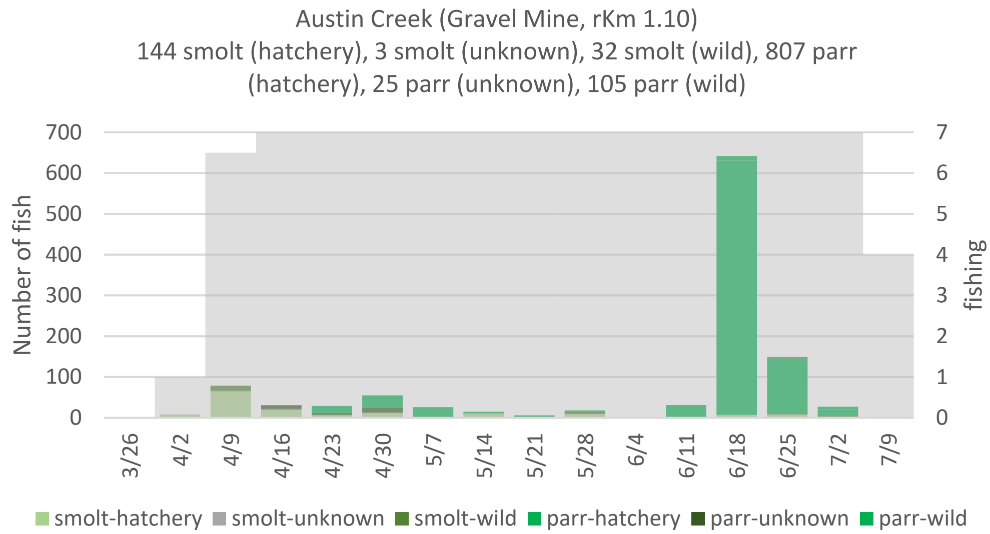
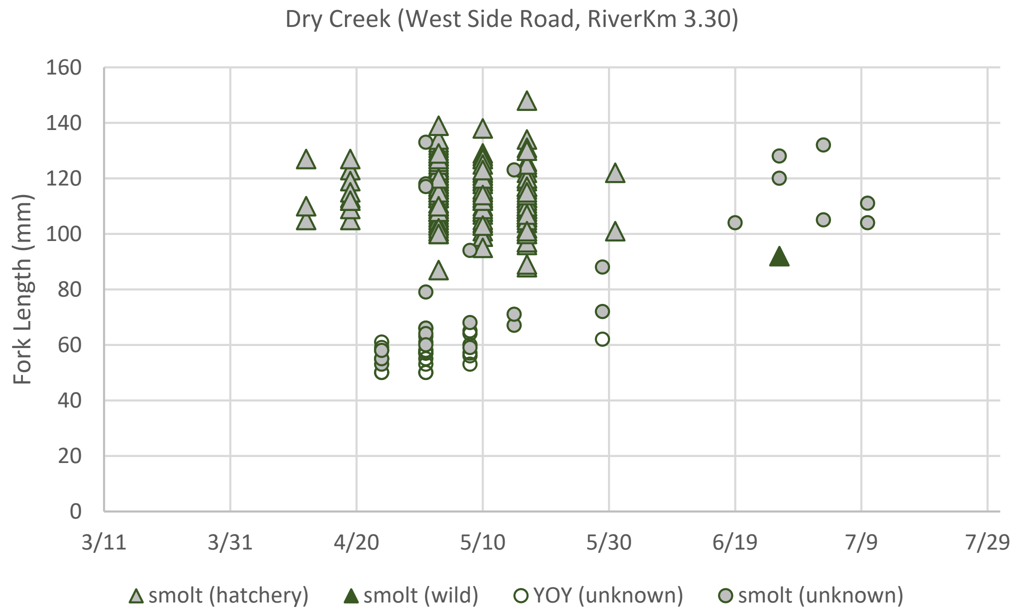
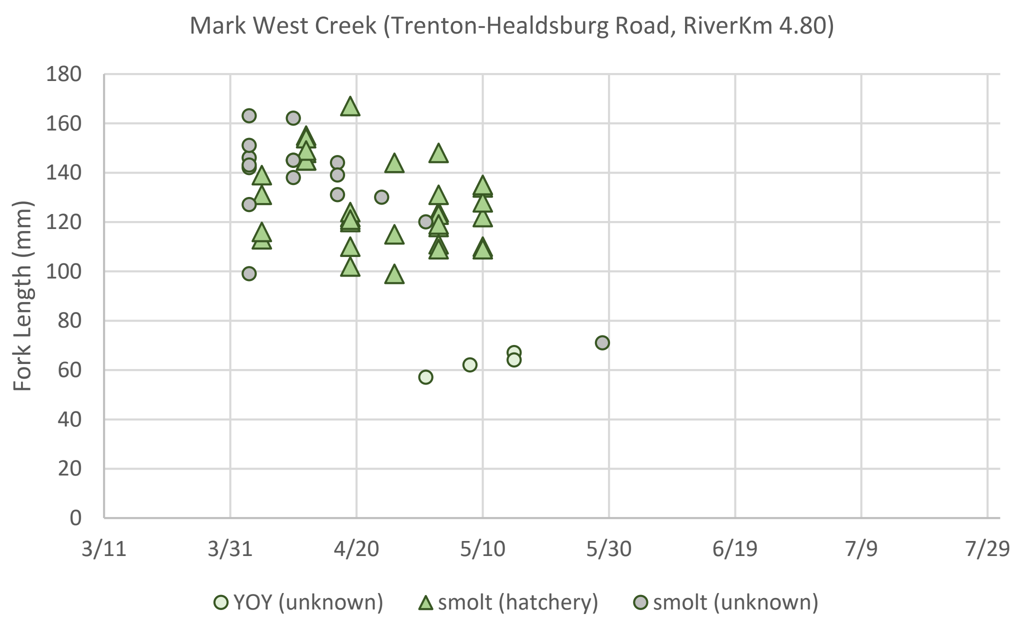
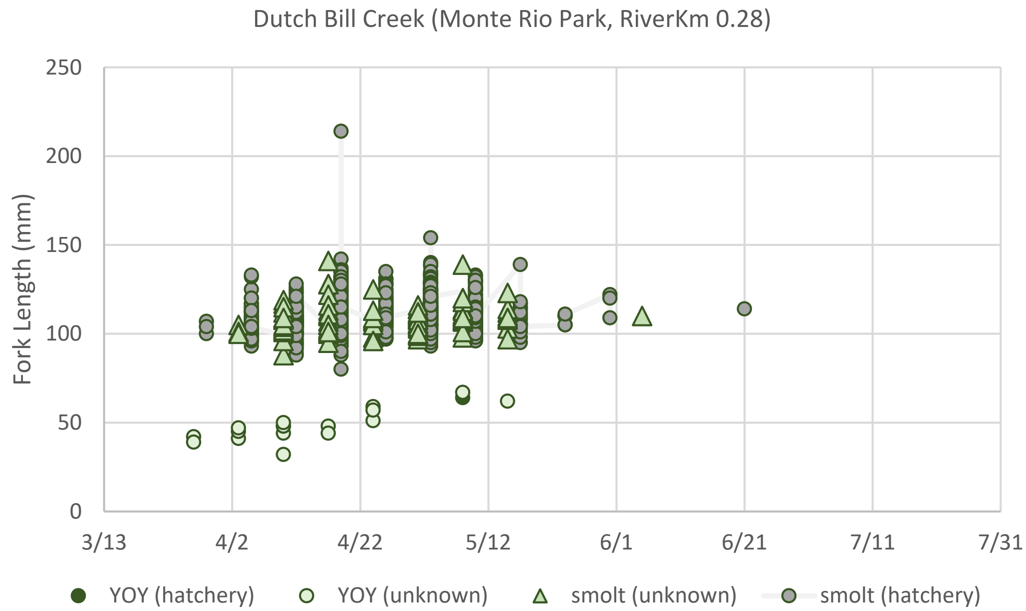


Figure 4.5.13. Weekly capture of coho salmon by life stage at lower river downstream migrant trapping sites, 2016. Gray shading indicates portion of each week trap was fishing. Note the different vertical scale among plots for each site.


Dry Creek (West Side Road, RiverKm 3.30)
Fork Length (mm)
0
20
40
60
80
100
120
140
160
3/11
3/31
4/20
5/10
5/30
6/19
7/9
7/29
smolt (hatchery)
smolt (wild)
YOY (unknown)
smolt (unknown)


Mark West Creek (Trenton-Healdsburg Road, RiverKm 4.80)
Fork Length (mm)
0
20
40
60
80
100
120
140
160
180
3/11
3/31
4/20
5/10
5/30
6/19
7/9
7/29
YOY (unknown)
smolt (hatchery)
smolt (unknown)



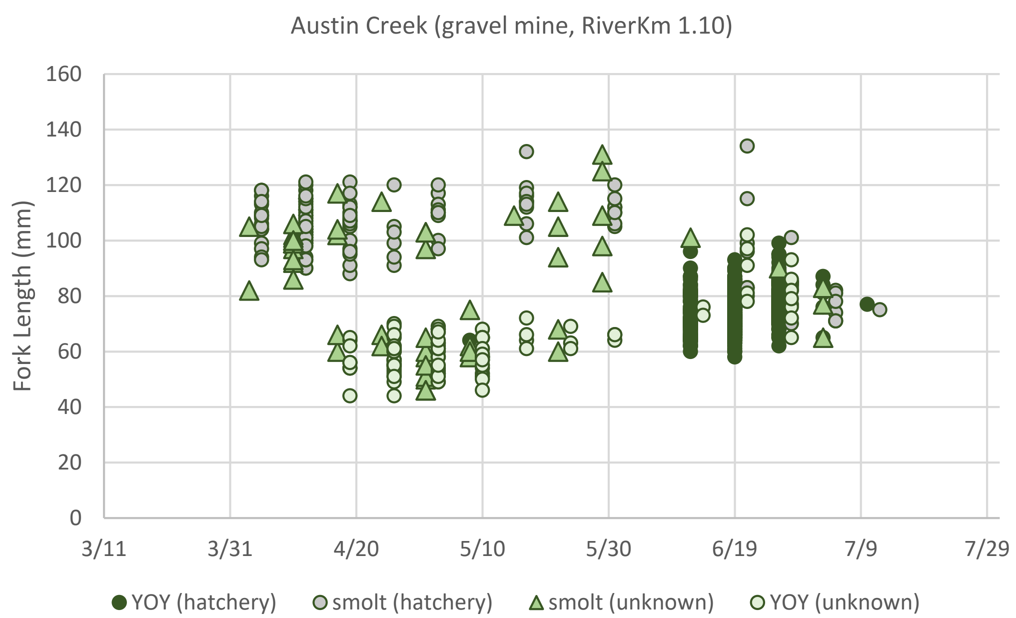


Figure 4.5.14. Weekly fork lengths of coho salmon captured at lower river downstream migrant trap sites, 2016.

Chinook Salmon

In 2016 relatively few Chinook smolts were captured in Austin Creek, Dutch Bill Creek, and Mark West Creek (14, 15 and 136, respectively).

Conclusions and Recommendations

The downstream migrant trapping monitoring objectives regarding the timing of estuary entry are partially met by using PIT tag detections from the paired antenna array in lower Austin Creek where antenna efficiency estimates are possible and where fish moving past that array have effectively entered the Estuary. In 2016, as in past years, many steelhead YOY were detected leaving Austin Creek and entering the Estuary. This same pattern was not seen at the other tributary monitoring sites.

While the PIT tag antenna at Duncans Mills spanned the Russian River for the 2016 outmigration season, detections of PIT tagged fish were not guaranteed because there are sections between antennas where fish could pass undetected. Fish orientation, and multiple PIT-tagged fish in the detection field of the same antenna at the same time, can also effect detection probability. Brackish water occasionally occurs at the antenna site, which causes decreases in antenna read range and water depths may exceed the detection field of some antennas. Collectively, these limitations all result in decreases in overall antenna efficiency; however, they are non-issues as long as detection efficiency can be estimated for use in expanding the number of fish detected. Unfortunately, efficiency estimates at Duncans Mills have not been possible because of the lack of a second antenna array in close proximity to the first (e.g., as is the case in Austin Creek, Figure 4.5.4). Regardless of these issues, PIT-tagging steelhead YOY at upstream locations and detecting those individuals if and when they move into the Estuary (along with beach seining in the Estuary itself) remain as the only viable method we know of for addressing the fish monitoring objectives in the Russian River Biological Opinion. Attempts continue to measure antenna efficiency so that expanded counts of PIT tagged individuals passing the antenna array can be constructed in future years.

References

Martini-Lamb, J. and D.J., Manning, editors. 2011. Russian River Biological Opinion status and data report year 2010-11. Sonoma County Water Agency, Santa Rosa, CA. p. 208

CHAPTER 5: Dry Creek Habitat Enhancement, Planning, and Monitoring

5.1 Dry Creek Habitat Enhancement Implementation

The Biological Opinion contains an explicit timeline that prescribes a series of projects to improve summer and winter rearing habitat for juvenile Coho Salmon and steelhead in Dry Creek (Figure 5.1.1). During the initial three years of implementation, 2008 to 2011, the Water Agency is charged with improving fish passage and habitat in selected tributaries to Dry Creek and the lower Russian River. The status of those efforts is described in Chapter 6 of this report. For the mainstem of Dry Creek, during this initial period, the Water Agency was directed to perform fisheries monitoring, develop a detailed adaptive management plan, and conduct feasibility studies for large-scale habitat enhancement and a potential water supply bypass pipeline. The pipeline feasibility study was completed in 2011 and is reported in Martini-Lamb and Manning 2011.

In 2012, the Water Agency began construction of the first phase of the Dry Creek Habitat Enhancement Demonstration Project. A second phase of the Dry Creek Habitat Enhancement Demonstration Project was constructed in 2013 with a third and final phase of the Demonstration Project constructed in 2014. The Dry Creek Habitat Enhancement Demonstration Project consists of a variety of habitat enhancement projects along a section of Dry Creek a little over one mile in length in the area centered around Lambert Bridge. Concurrently, the U.S. Army Corps of Engineers completed construction in 2013 of a habitat enhancement project on U.S. Army Corps of Engineers owned property just below Warm Springs Dam (Reach 15 area). In 2016, Sonoma Water began construction on the Dry Creek Habitat Enhancement Phase 2, Part 1 Project (centered approximately a mile upstream of the Demonstration Project) and the Dry Creek Habitat Enhancement Phase 3, Part 1 Project (centered in a lower reach area of Dry Creek just below the Westside Road Bridge crossing of Dry Creek). Construction activities for both the Phase 2, Part 1 and Phase 3, Part 1 projects were anticipated to be spread across two construction seasons with work starting in 2016 and then being completed in 2017.

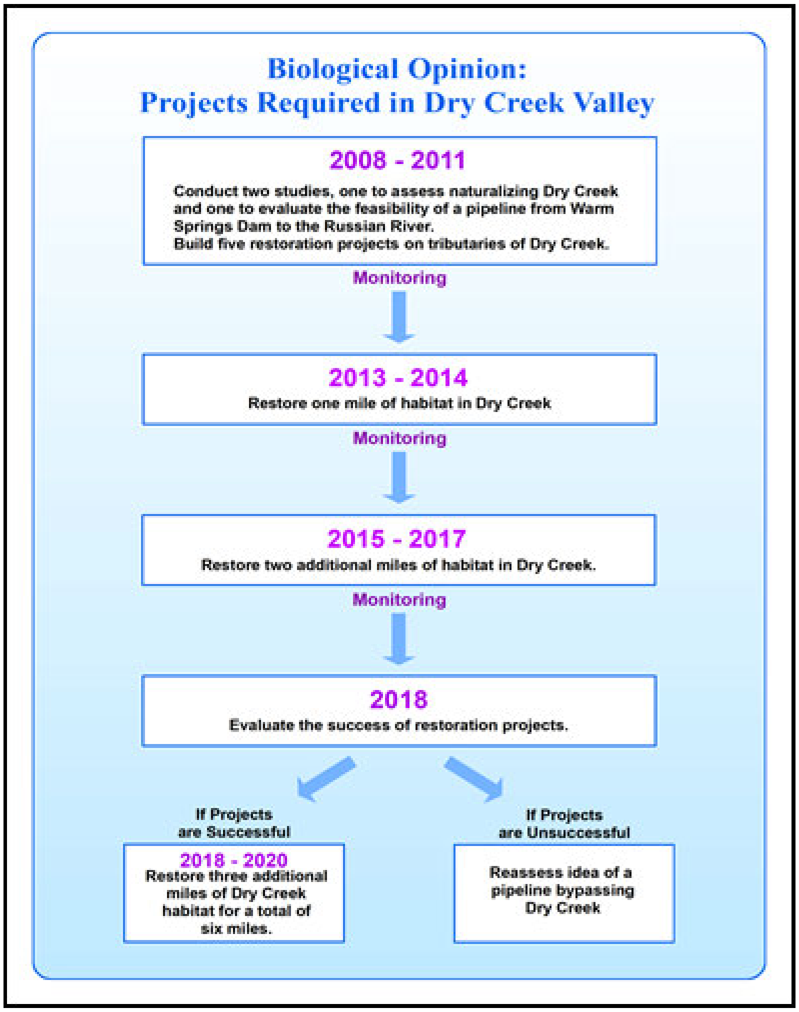


Figure 5.1. 1. Timeline for implementation of Biological Opinion projects on Dry Creek.

Habitat Enhancement Feasibility Study

The Water Agency regulates summer releases from Warms Springs Dam along a 14 mile reach of Dry Creek from Lake Sonoma to the Russian River. This abundant, cool, high quality water has tremendous potential to enhance the Russian River's Coho Salmon and steelhead population but it flows too swiftly to provide maximum habitat benefit. By modifying habitat conditions to create refugia from high water velocities along 6 miles of Dry Creek, NMFS and DFG assert that water supply releases can continue at current discharge levels of approximately 100 cubic feet per second (cfs) and potentially historic discharge levels up to 175 cfs.

To plan large scale enhancement of the Dry Creek channel, the Water Agency has retained Inter-Fluve, Inc. to conduct extensive field surveys and produce a series of reports detailing habitat enhancement opportunities along Dry Creek. Interfluve's work is being conducted in three phases: 1) inventory and assessment of current conditions; 2) feasibility assessment of habitat improvement approaches; and 3) conceptual design of habitat approaches deemed feasible. All three reports have been completed and can be viewed at http://www.scwa.ca.gov/drycreek/.

During 2011, Interfluve developed the Dry Creek Fish Habitat Enhancement Conceptual Design Report (Appendix 5.1). The final report was released to the public in July 2012 and identifies 26 sub reaches along Dry Creek as potential areas for construction of low velocity habitat with depth and cover characteristics conducive to rearing juvenile Coho Salmon and steelhead. The opportunities identified in the report are distributed throughout the 14 mile length of Dry Creek. However, different reaches of Dry Creek present unique geomorphic and hydrologic constraints and Interfluve divided the stream into upper, middle, and lower segments. In the upper segment (mile 11 to 13.7), the influence of Warm Springs Dam on streamflow, substrate, and channel dimensions is most pronounced. The stability of this reach provides opportunities for long lasting "constructed" habitat features such as side channels, backwaters, and log structures. In the lower segment between Westside Road Bridge and the confluence with the Russian River (mile 0 to 3), conditions are amenable to constructing projects designed to let natural river processes develop habitat over time. The middle segment between Pena Creek and Westside Road (mile 3 to 11), has opportunities for both constructed habitat and river process based approaches.

The Concept Design report includes a description of current habitat conditions, modeled inundations at high flow, maps and graphics depicted proposed summer and winter habitat features, and a preliminary cost estimate for each of the 26 enhancement sub reaches along Dry Creek (Figures 5.1.2 and 5.1.3 show conceptual design figures developed for two of these enhancement reaches). All of the sub reaches are ranked according to the potential quantity of summer and winter Coho rearing habitat they provide (Table 5.1.1). This ranking does not, however, include implementation considerations such as relative cost, landowner willingness and accessibility, and continuity or predicted longevity of constructed features. Figure 5.1.4 illustrates the two step process that will be employed to select enhancement reaches on Dry Creek.

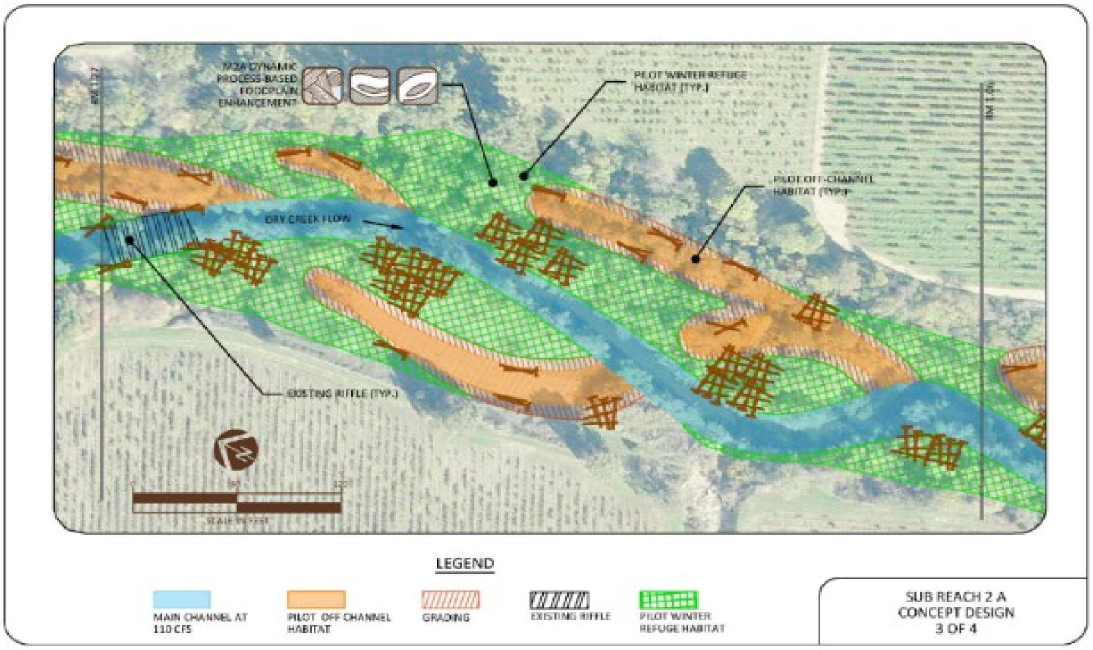


Figure 5.1. 2. Example of habitat enhancement conceptual design for Dry Creek reach 2A. Reach 2A is close the confluence of Dry Creek and the mainstem Russian River. In this highly dynamic environment, a "process" based approach that creates pilot habitat features the stream can adjust over time is proposed.

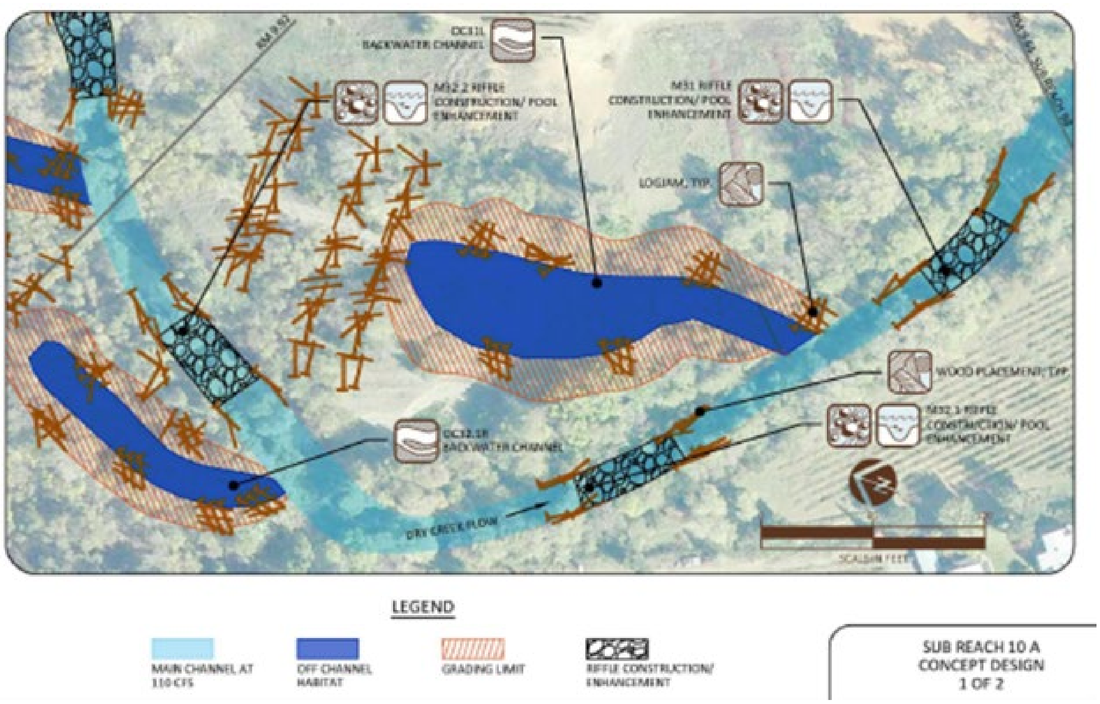


Figure 5.1. 3. Example of habitat enhancement conceptual designs for Dry Creek Reach 10A, illustrates proposed summer habitat enhancements using a static "constructed" habitat approach.

Table 5.1. 1. Ranking of enhancement subreaches in Dry Creek organized by Upper, Middle, and Lower segments.

Segment	Ranking Tier	(Sub) Reach	Coho Potential Coho Rearing Habitat Score	Winter Refuge & Rearing Habitat Score	Total Potential Habitat Score	Predicted Continuity Score
Upper	Tier I	14a	High	Medium	High	High
		13b	Medium	Medium	Medium	High
		15	Medium	Low	Low	High
		14b	Medium	Low	Low	High
	Tier II	12b	Low	High	Medium	High
		13a	Low	Low	Low	High
		12a	Low	Low	Low	High
Middle	Tier I	8b	High	Medium	High	Medium
		4a	High	Low	High	High
		5a	High	Low	High	Medium
		4b	High	Low	Medium	Medium
		8a	Medium	High	High	High
		5b	Medium	Medium	High	Medium
		10a	Medium	Low	Medium	High
		10b	Medium	Low	Medium	Medium
		4c	Medium	Low	Low	High
	Tier II	6	Low	High	High	Medium
		11	Low	Medium	High	Medium
		9b	Low	Medium	Low	Medium
		9a	Low	Low	Low	Medium
Lower	Tier I	2b	High	High	High	Low
		2a	High	High	High	Low
		1	High	High	High	Low
	Tier II	3b	Medium	Low	Medium	Medium
		3a	Medium	Low	Medium	Medium

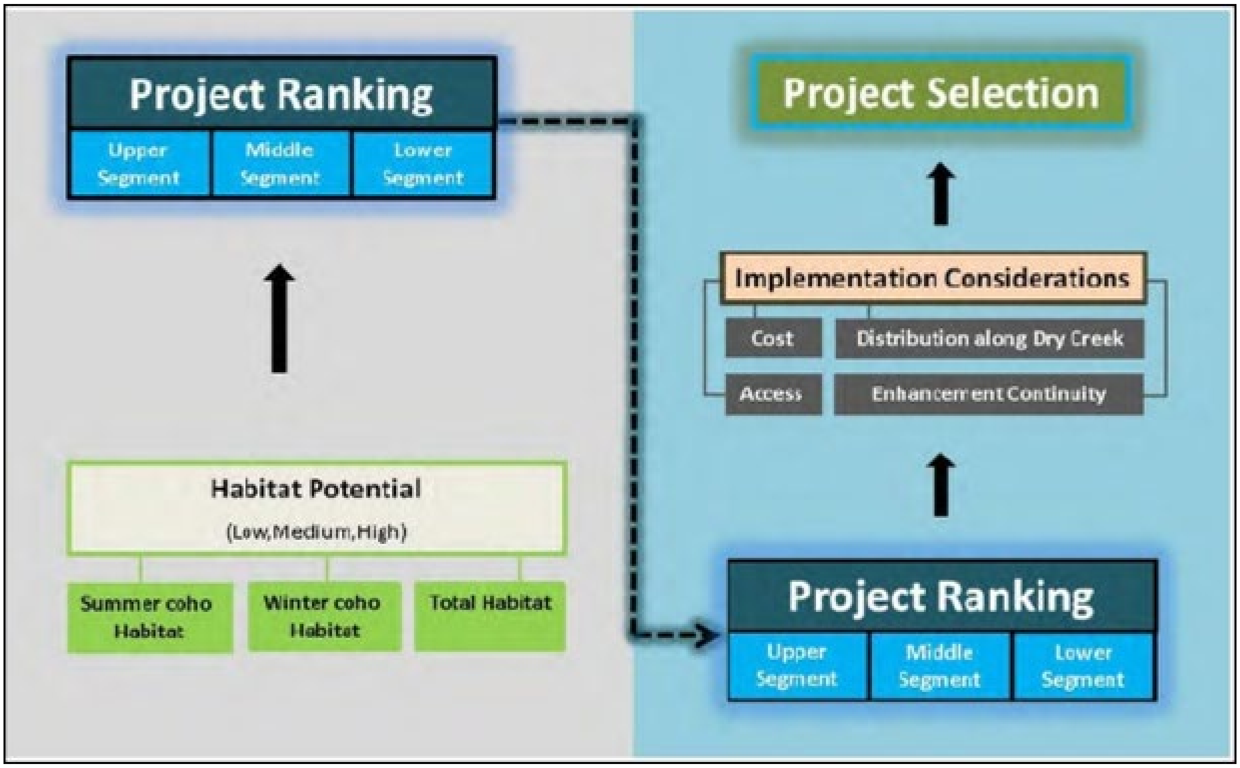


Figure 5.1. 4. Conceptual depiction of habitat project prioritization approach. The left side of the figure represents the first phase of the prioritization process which includes ranking of the enhancement subreaches based solely on their inherent potential for habitat enhancement. The second phase, project selection, includes implementation considerations such as access, distribution, and cost.

Demonstration Project

As described in the Public Outreach Chapter of this report, the Water Agency must engage a diverse group of stakeholders to implement the Biological Opinion. Dry Creek is held almost entirely in private ownership and Water Agency staff must work in concert with landowners of more than 170 parcels to study, plan, and construct habitat enhancements. The Biological Opinion’s 5 year timeline prior to construction of the first mile of habitat enhancement acknowledges this challenge and the depth of study, planning, and environmental compliance required for implementation. A forward looking group of property owners along a one mile stretch of the stream near Lambert Bridge, in the middle of Dry Creek Valley, approached the Water Agency with the opportunity to advance the schedule and demonstrate habitat enhancement techniques in their reach of the stream (Reach 7). The Water Agency welcomed this opportunity, and worked to implement the Dry Creek Habitat Enhancement Demonstration Project between 2012 and 2014. The U.S. Army Corps of Engineers also implemented a similar habitat enhancement (Reach 15 Project) on a 0.3 mile reach of Dry Creek immediately below Warms Springs Dam in 2013. A detailed summary of these two projects can be found in the 2015 Biological Opinion Annual Report.

Phase 2 and 3

Beyond the completion of the Demonstration Project (Reach 7) work and the Corps of Engineer's Reach 15 work, the Water Agency has continued to make progress towards the construction of the next two miles of habitat enhancement. Figure 5.1.5 shows the areas completed in Reach 15 and the Demonstration Project (Reach 7) and other areas either in design or under construction. These next two miles have been designated as Phase 2 and 3, with each of these phases to be constructed in parts. No construction activities occurred in 2015; however, construction of Phase 2, Part 1 (Reach 8) and Phase 3, Part 1 (Reach 2) began in June of 2016 (see photos below). The construction work for these two parts is anticipated to be completed in 2017. Design development and landowner negotiations continue for the future parts of both Phase 2 and Phase 3 design work. Phase 2, Part 2 (Reach 14) and Phase 3, Part 3 (Reach 5) are expected to be constructed in 2017 or 2018 by the Water Agency. Phase 3, Part 2 (Reach 4a) is expected to be constructed in 2017 or 2018 by the U.S. Army Corps of Engineers.

Figure 5.1. 5. This figure shows the habitat enhancement projects along Dry Creek that have been completed and projects that are being designed.

Photo 5.1.1. Dry Creek Habitat Enhancement Project Phase 2, Part 1. Photo shows new side channel inlet under construction at the Truett Hurst site (Reach 8). Mainstem of Dry Creek (looking downstream) can be seen at the right hand side of the photo. August 2016.

Photo 5.1.2. Dry Creek Habitat Enhancement Project Phase 2, Part 1. Photo shows new side channel backwater feature recently constructed at the Truett Hurst site (Reach 8). October 2016.

Photo 5.1.3. Dry Creek Habitat Enhancement Project Phase 3, Part 1. Photo shows new side channel feature recently constructed at the Geyser Peak site (Reach 2). October 2016.

5.2 Effectiveness monitoring

Effectiveness monitoring focuses on the physical response of Dry Creek to habitat enhancements and determines “whether habitat enhancement is having the intended effect on physical habitat quality” in Dry Creek (NMFS Russian River Biological Opinion 2008, pg. 266). NMFS (2008) concluded that sub-optimal water velocity, depth and instream cover limit juvenile coho salmon and steelhead and suggested optimal values for water velocity depth, and cover as part of the Reasonable and Prudent Alternative (NMFS 2008). The Joint Monitoring Team, consisting of representatives from NMFS, CDFW, USACE, and the Water Agency, refined these values within the Dry Creek Adaptive Management Plan (Porter et al. 2014) and developed primary performance metrics linked to the optimal values of water velocity, depth, and cover by which to evaluate the effectiveness of habitat features, sites, and reaches (Table 5.2.1). The Joint Monitoring Team also identified secondary performance metrics that help determine the effectiveness of habitat enhancements to influence non-target, ancillary conditions (e.g., water temperature, dissolved oxygen concentration). The Dry Creek Adaptive Management Plan also suggested target flows to represent seasonal variation critical to each life stage (Porter et al. 2014).

Table 5.2. 1. Primary and secondary performance measures from the Dry Creek Adaptive Management Plan.

Type of Performance Measure	Performance Measure	Life Stage	Spring Flow[1]	Summer Flow[2]	Winter Flow[3]
Primary	Velocity (ft/sec)	fry	0-0.5 ft/s	n/a	n/a
	Depth (ft)	fry	0.5-2.0 ft	n/a	n/a
	Velocity (ft/sec)	Summer/winter parr	0-0.5 ft/s	0-0.5 ft/s	0-0.5ft/s
	Depth (ft)	Summer/winter parr	2-4 ft	2-4 ft	2-4 ft
	Shelter value	Juvenile	≥80	≥80	≥80
	Pool: Riffle ratio	Juvenile	n/a	1:2 to 2:1	n/a
Secondary	Temperature (°C)	Juvenile	n/a	8-16° C	n/a
	Dissolved oxygen (mg/l)	Juvenile	n/a	6-10 mg/l	n/a
	Canopy (%)	Juvenile	80 %		
	Quiet water (< 0.5 ft/s) (%)	Juvenile	n/a	n/a	≥ 25%
	Off-channel access (off-ramps) (ft/sec)	Juvenile	Approx. 1.5 – 1.8 cm/s (Ucrit); Approx. 3.3 ft/s (burst speed)		
	Connectivity of habitats	Juvenile	Undefined		
	Substrate particle size (in.)	Adult	n/a	n/a	0.25-2.5 in.
	Depth (ft)	Adult	n/a	n/a	0.5-1.6 ft

[1] Target coho life stage during spring is newly-emerged feeding fry which use shallower depths than would be preferred later in the summer and winter when fish would be larger. Target spring flow (discharge within the enhancement reach) is 200 cfs (approximately double the summer “base” flow).

[2] Target summer flow is 105 cfs

[3] Target winter flow is 1000 cfs

Methods

The methods described below focus on data collection to assess the Dry Creek Habitat Enhancement Project against the primary performance measures of water depth (0.5-2 or 2-4 ft) and velocity (<0.5 ft/s), and amount of instream cover (shelter value) (Table 5.2.1). The remaining primary performance measure, pool to riffle ratio, is dependent on longer-term channel evolution in response to enhancement occurring after geomorphically effective flows and will be assessed in future monitoring reports. Monitoring project performance against secondary metrics is underway and will also be assessed in future reports. Depth, velocity, and shelter value provide a means to directly assess against primary metrics in the Dry Creek Adaptive Management Plan and against optimal habitat values suggested as part of the Reasonable and Prudent Alternative in the Russian River Biological Opinion (NMFS 2008, Porter et al. 2014).

Water depth and velocity

The Dry Creek Adaptive Management Plan (Porter et al. 2014) suggested collecting water depth and velocity at points along transects placed within constructed backwaters and main channel portions of Dry Creek, and “habitat feature mapping” near selected habitat enhancements (logjams, boulder fields). Habitat feature mapping would result in two-dimensional depictions of depth and velocity around habitat features and allow quantification of optimal habitat area adjacent to features. Upon consultation with NMFS, and through field experimentation with several mapping and survey tools (auto-level, differential global positioning system, total station), the Water Agency developed a robust habitat feature mapping method to characterize all portions of the Dry Creek channel, not just adjacent to enhancement features, obviating the need to collect cross-sectional data.

Field crews collected water depth and velocity at spatially referenced points across the streambed and banks using handheld flow meters and a total station. At each data point, we collected geographic location (latitude, longitude, elevation), and water depth and velocity by aiming the total station at a USGS topset rod fit with a survey prism and a flow meter (Figure 5.2.1). The technique allowed simultaneous collection of topographic and hydraulic data (water depth and velocity) that were highly spatially accurate and repeatable to enable comparison to future conditions, and allow collection of data across the streambed to create detailed relief maps. Field crews focused point collection on breaks in channel and bank slope and breaks in water velocity, and at a minimum collected points at the top of each bank, water’s edge (water surface elevation), toe of bank, thalweg, and at least two points in between the toe of bank and thalweg.

We processed the data within a Geographic Information System (GIS) to create detailed maps of stream topography (elevation) and hydraulic conditions (water depth and velocity) to spatially characterize habitat conditions and quantify optimal fry and juvenile habitat. The individual points were first used to create vector- (line) based representations of the stream channel, which were then smoothed to create raster (grid) based digital elevation models (DEMs). We classified hydraulic habitat conditions according to the primary metrics from Porter et al. (2014) (depth [0.5-2 ft or 2-4 ft], depending on life stage and velocity [<0.5 ft/s]) to identify the location of habitat falling within optimal depth, velocity, and depth and velocity ranges as polygons. Generating polygons within a GIS also allowed us to quantify the areas of optimal habitat.

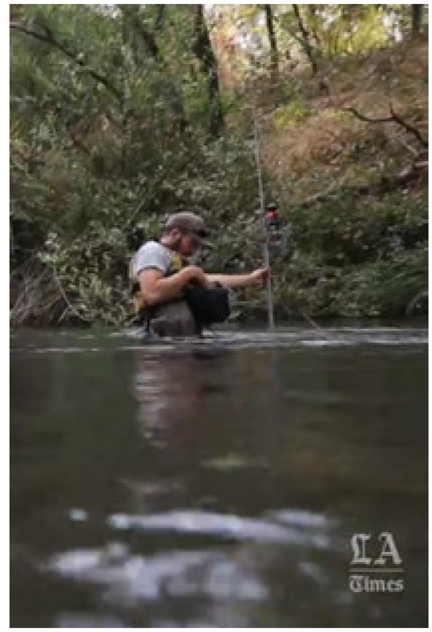


Figure 5.2. 1. Dry Creek effectiveness monitoring. At each data point, we collected geographic location (latitude, longitude, elevation), and water depth and velocity by aiming the total station at a USGS topset rod fit with a survey prism and a flow meter.

Shelter value

Field crews also determined the shelter value of individual habitat units within each enhancement site. The California Salmond Habitat Restoration Manual (Flosi et al. 2010) rates instream shelter by multiplying the complexity of available cover within a habitat unit (Table 5.2.1; 0 = no shelter, 3 = highly complex shelter) by the overhead area occupied by that cover (0 = 0% of overhead area covered, 100 = 100% of overhead area covered). The maximum shelter value is 300 (3 [complexity of available cover within a habitat unit] * 100 [area of habitat unit covered]), with a score of ≥80 considered optimal within the Dry Creek Adaptive Management Plan (Table 5.2.1) (Porter et al. 2014).

We inventoried instream habitat units using habitat types described in the California Salmonid Habitat Restoration Manual (Flosi et al. 2010). These habitat types are distinguished by differences in local channel gradient, water velocity, depth, and substrate size. Flosi et al. (2010) use four hierarchical levels of classification to describe physical fish habitat, with each successive level providing greater detail. The most elementary descriptions (Levels 1 and 2) break stream channels into pool, riffle, or flatwater habitat types. Successive levels differentiate habitat types by location within the stream channel (e.g., mid-channel pools, Level 3) or by cause or agent of formation (e.g., lateral-scour, log-formed pools, Level 4). In this survey, we inventoried to habitat types to Level 2 and delineated the upstream and downstream boundaries by placing flagged 10 inch nail spikes on the right and left bank. We surveyed the location of the nail spikes with a total station and processed the data within a GIS to create polygons of habitat unit types and cover complexity.

Results

Water depth and velocity

During summer and early fall 2016, we surveyed three enhancement reaches that make up parts of miles 2 and 3 of the Dry Creek Habitat Enhancement Project (Meyer and Truett Hurst [mile 2] and Geyser Peak [mile 3]). Pre- construction surveys of unenhanced mainstem areas (June/July 2016) and post-construction surveys (October/November 2016) of enhanced off-channel areas totaled 4,300 linear feet (188,000 ft^2). Pre-construction surveys mapped 113,000 ft^2 and recorded 15,000 ft^2 of optimal habitat. Post-construction surveys of enhanced areas took place over a smaller wetted area (75,000 ft^2), confined to off-channel side-channels and alcoves, but recorded over twice as much optimal habitat as mainstem areas (38,000 ft^2; Table 5.2.2, Figure 5.2.2-Figure 5.2.31). The Truett Hurst enhancement reach occupied the greatest enhanced wetted area and supported the greatest areas of optimal depth, velocity, and optimal habitat after enhancement (Table 5.2.2, Figure 5.2.22-Figure 5.2.31). The Geyser Peak enhanced wetted area occupied nearly half the enhanced wetted area of the Meyer enhancement reach (12,000 ft^2 vs 28,000 ft^2), but supported a proportionally greater amount of optimal habitat (7,000 ft^2 [57% of enhanced wetted area] versus 9,500 ft^2 [35% of enhanced wetted area] Table 5.2.2, Figure 5.2.2-Figure 5.2.21). Still, the enhanced area in the Meyer enhancement reach increased pre-enhancement optimal habitat five-fold. Enhanced area in Geyser Peak nearly doubled the area of optimal habitat post-enhancement and enhanced area in Truett Hurst nearly quadrupled the area of optimal habitat post-enhancement

Table 5.2. 2. Wetted area, area of optimal depth and velocity, and area of optimal habitat within Dry Creek enhancement reaches before enhancement (pre-enhancement), enhanced area constructed and total after construction in 2016.

Enhancement reach	Wetted area (ft^2)	Optimal depth (ft^2)			Optimal velocity (ft^2)	Optimal habitat (ft^2)		
		0.5 – 2.0 ft	2.0 – 4.0 ft	Total	<0.5 ft/s	0.5 – 2.0 ft < 0.5 ft/s	2.0 – 4.0 ft < 0.5 ft/s	Total
Geyser Peak (pre-enhancement)	37618	26238	5243	31481	11881	5946	1445	7391
Geyser Peak (enhanced area)	12252	9002	730	9733	9074	6429	628	7057
TOTAL (post-enhancement)	**49870**	**35240**	**5974**	**41214**	**20955**	**12375**	**2072**	**14448**
Meyer (pre-enhancement)	27073	21808	1135	22943	4092	1713	82	1795
Meyer (enhanced area)	27646	7706	8385	16091	15103	3705	5822	9527
TOTAL (post-enhancement)	**54720**	**29514**	**9520**	**39034**	**19195**	**5418**	**5904**	**11322**
Truett,Hurst (pre-enhancement)	48463	33040	10531	43571	8876	4520	1548	6067
Truett Hurst (enhanced area)	35574	12210	16843	29053	27128	8401	13403	21804
TOTAL (post-enhancement)	**84037**	**45250**	**27373**	**72624**	**36005**	**12921**	**14951**	**27872**
Pre-enhancement	113154	12179	3075	15254	3,217	754	1,873	15254
Enhanced area	75473	18535	19853	38388	24849	12179	3075	38388
TOTAL (post-enhancement)	**188626**	**30714**	**22928**	**53642**	**51306**	**18535**	**19853**	**53642**

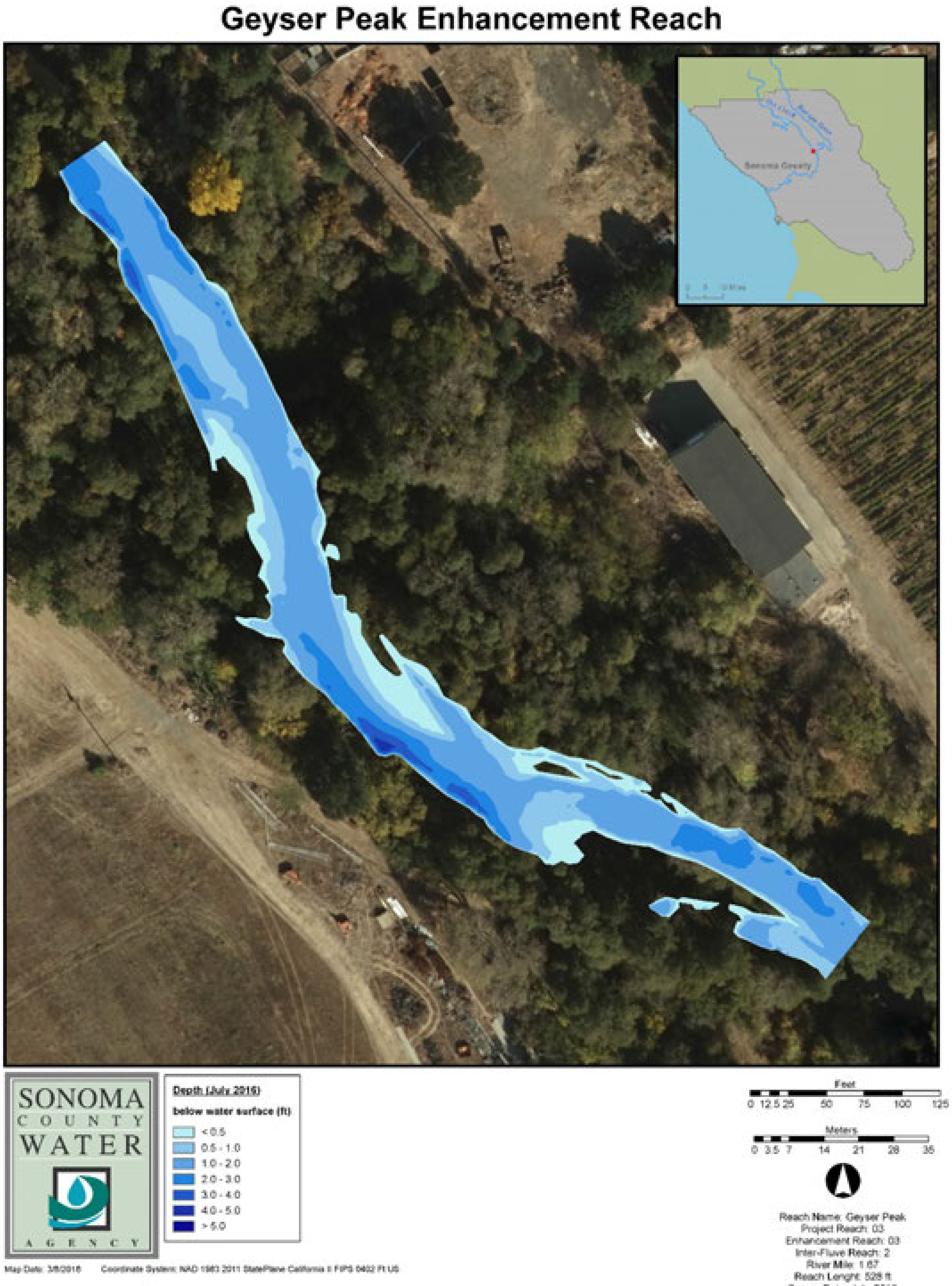


Figure 5.2.1. Measured water depth within the Geyser Peak habitat enhancement reach during July 2016.

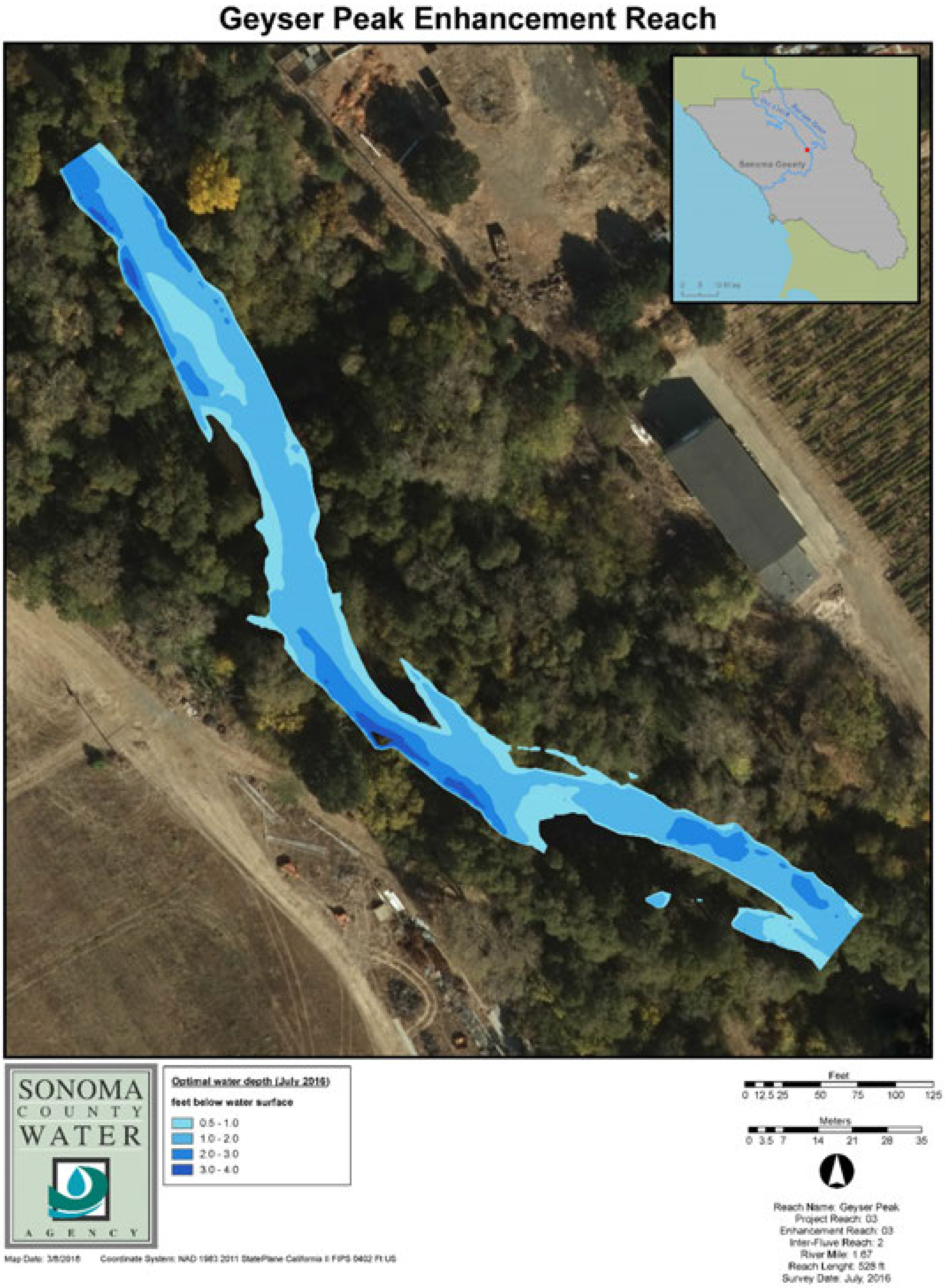


Figure 5.2.2. Area of optimal water depth within the Geyser Peak habitat enhancement reach during July 2016.

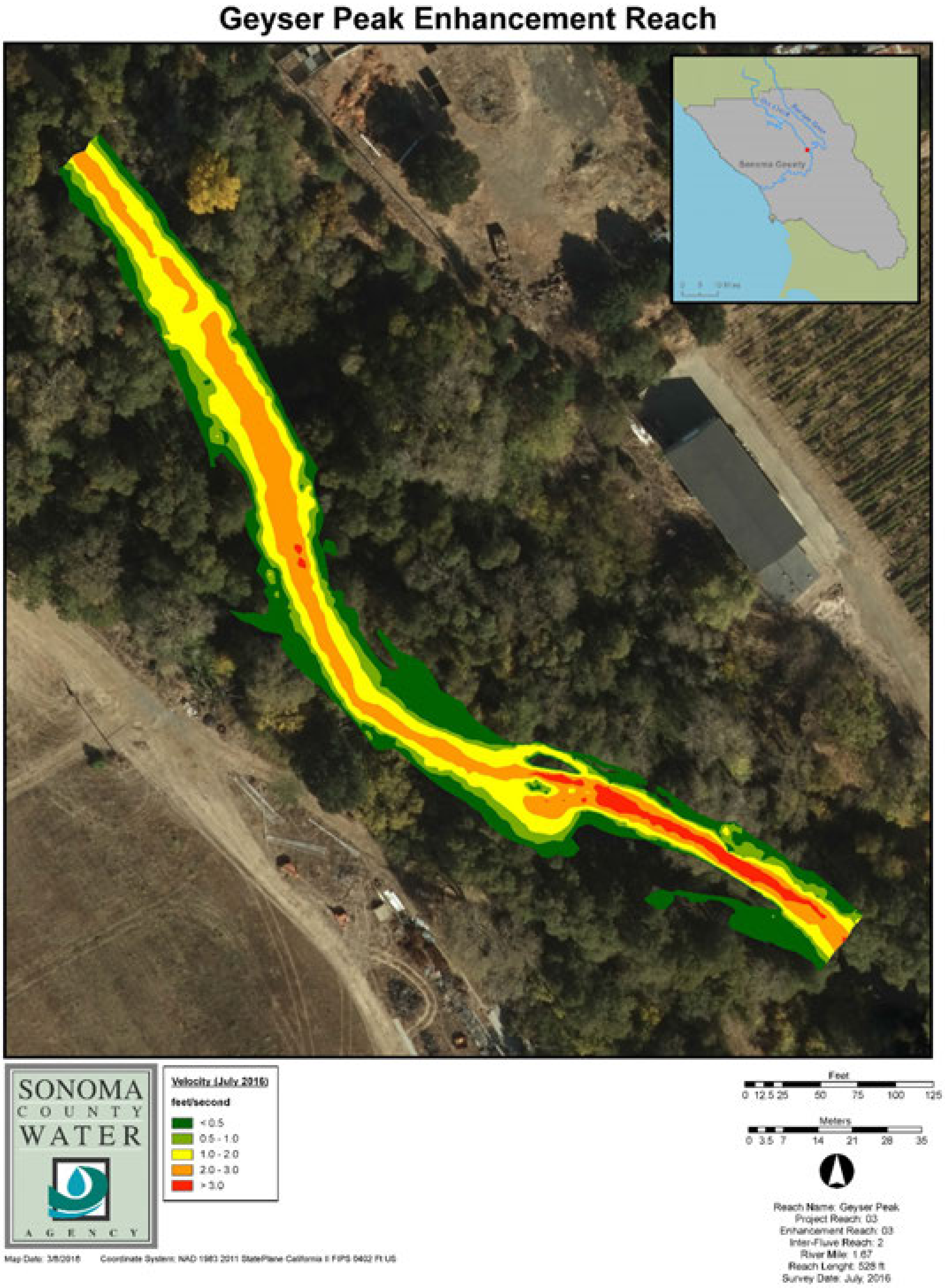


Figure 5.2.3. Measured water velocity within the Geyser Peak habitat enhancement reach during July 2016.

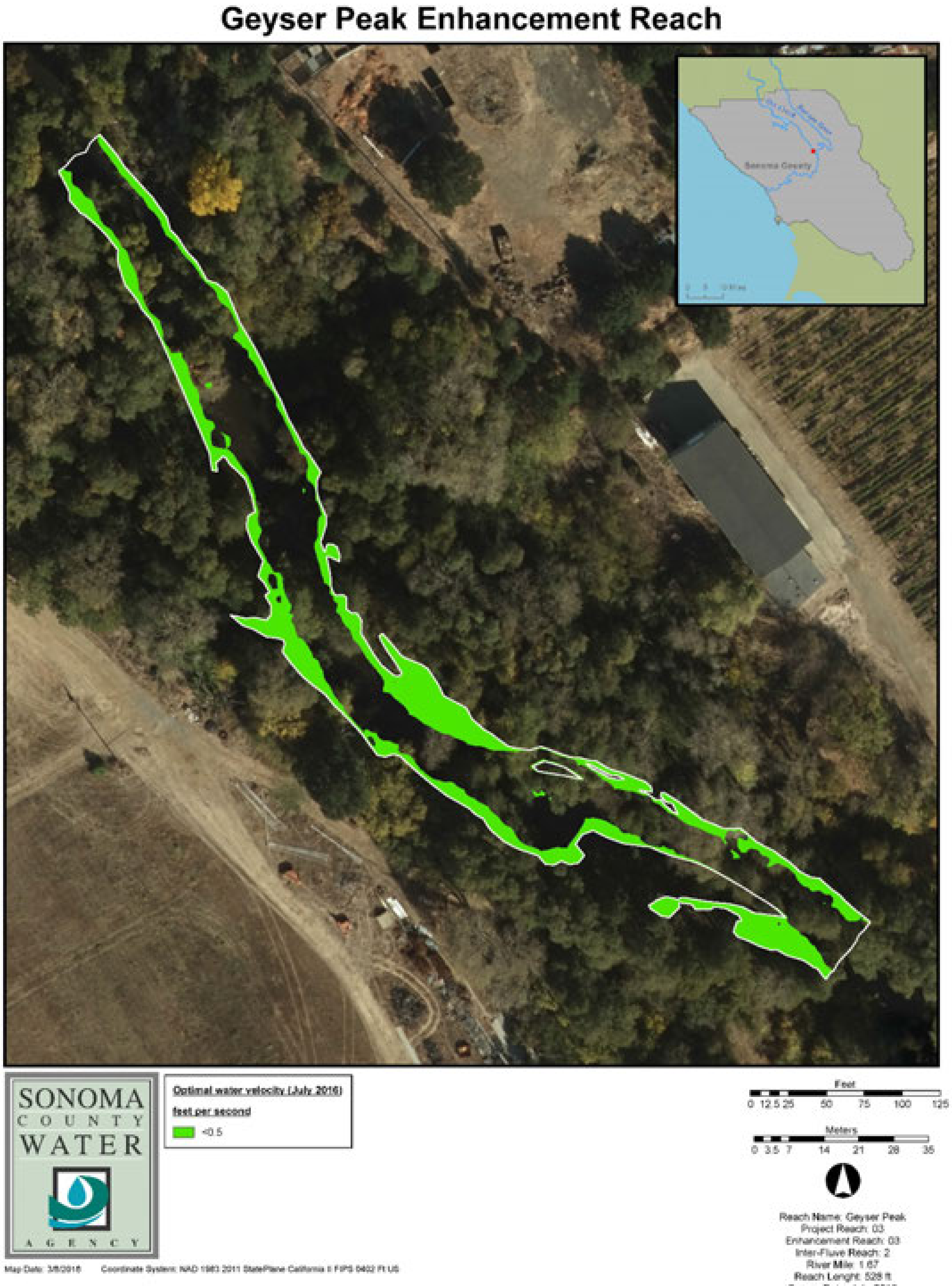


Figure 5.2.4. Area of optimal water velocity within the Geyser Peak habitat enhancement reach during July 2016.

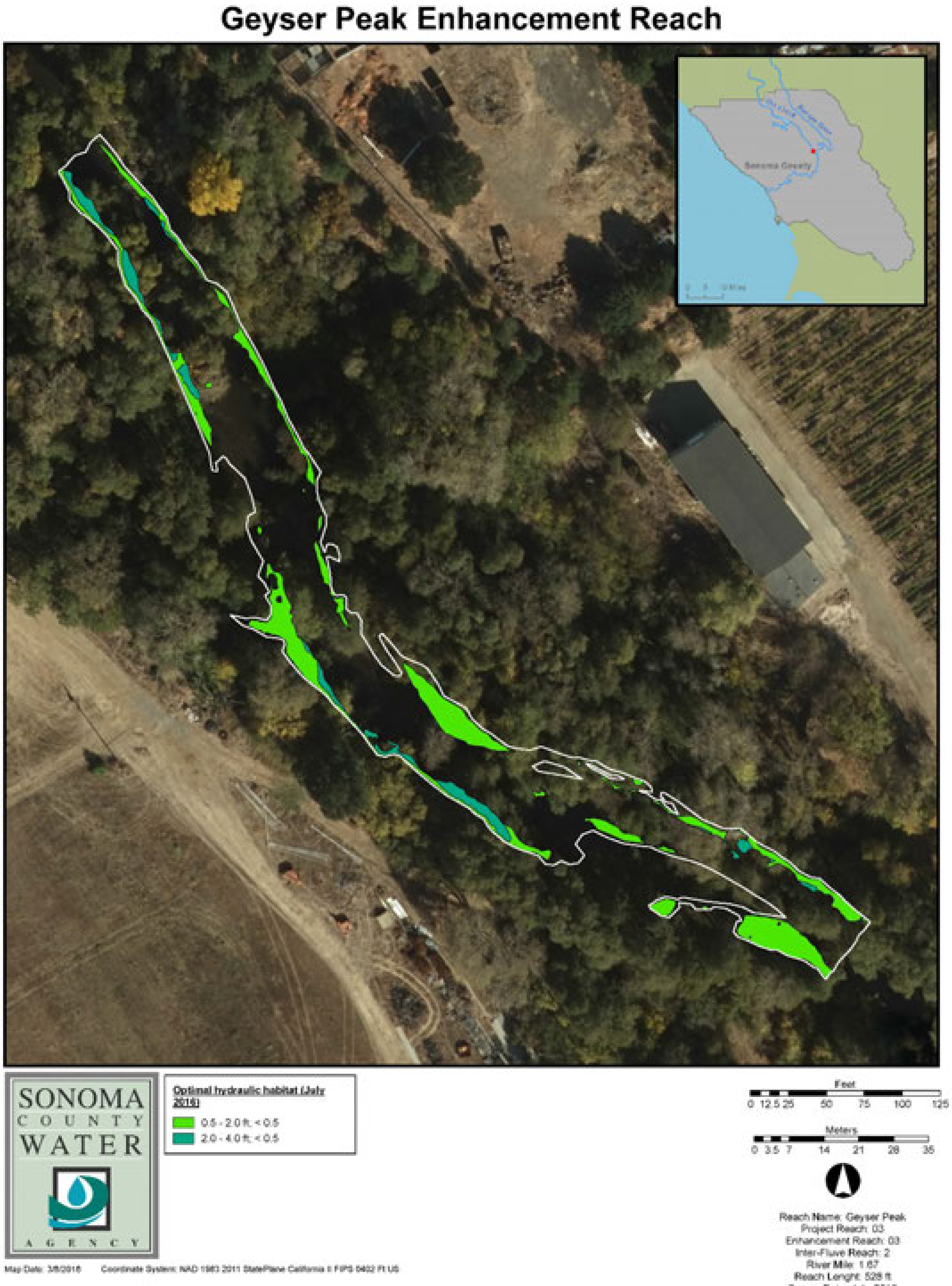


Figure 5.2.5. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Geyser Peak habitat enhancement reach during July 2016.

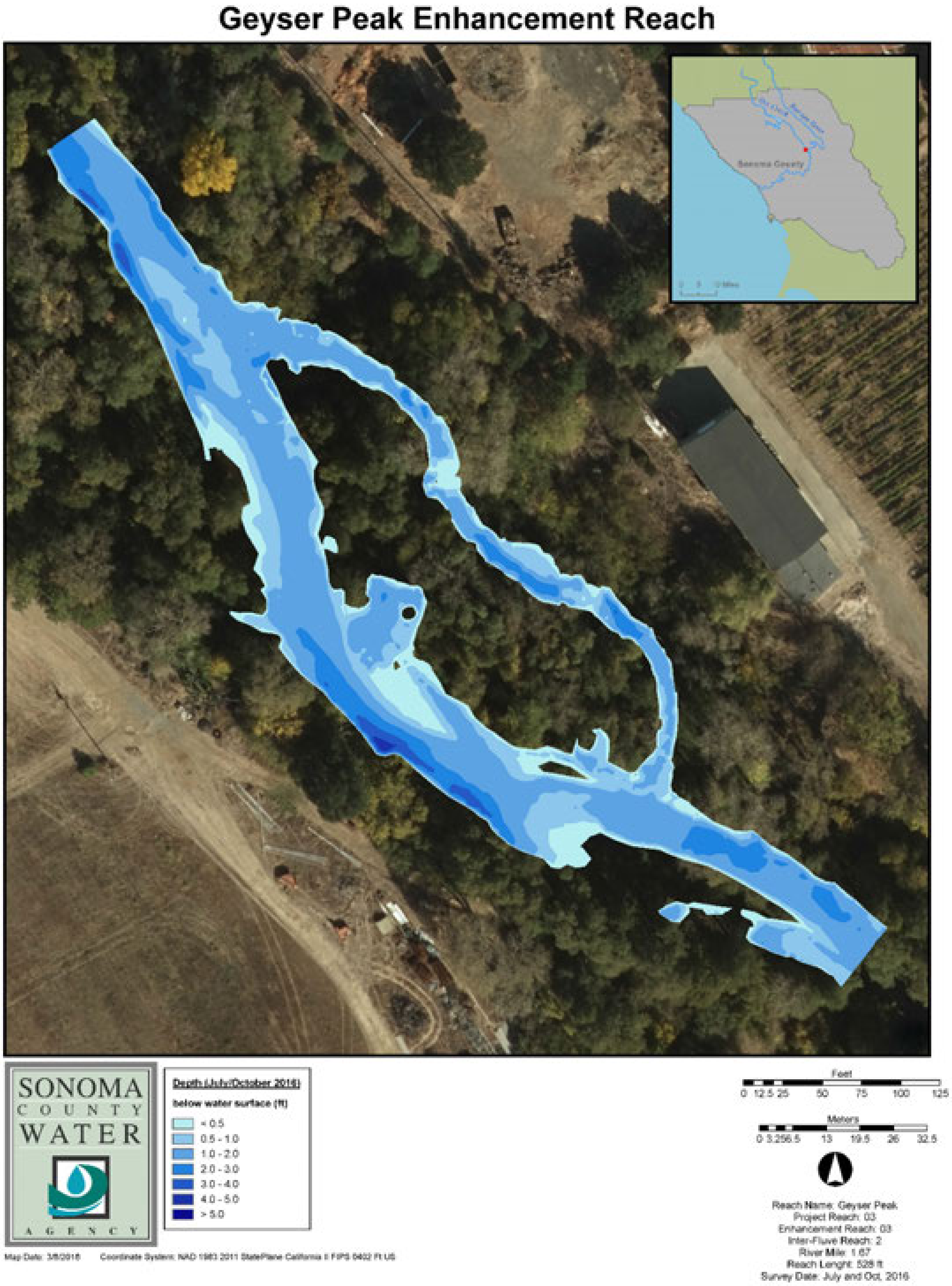


Figure 5.2.6. Measured water depth within the Geyser Peak habitat enhancement reach in November 2016.

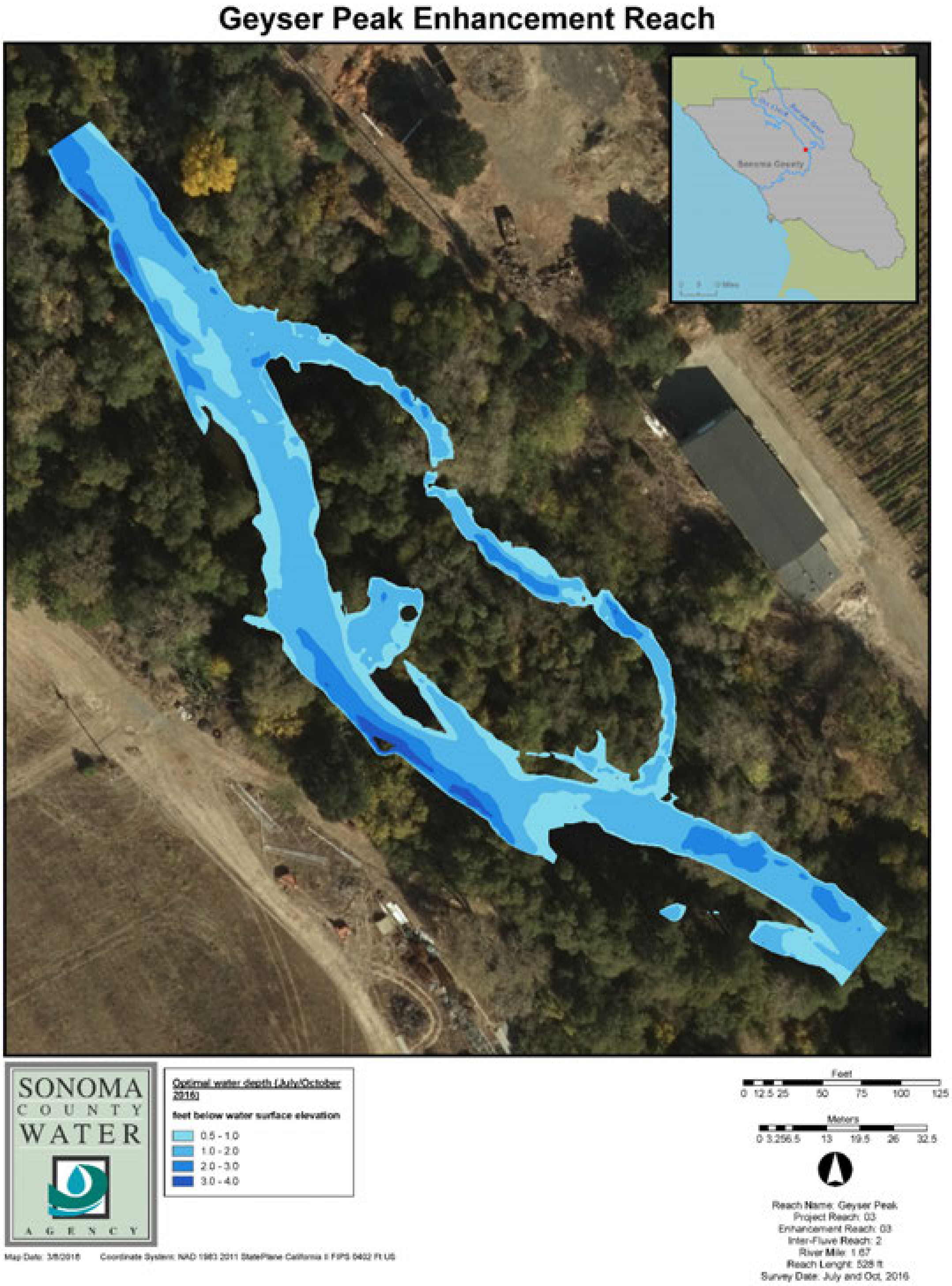


Figure 5.2.7. Area of optimal water depth within the Geyser Peak habitat enhancement reach in November 2016.

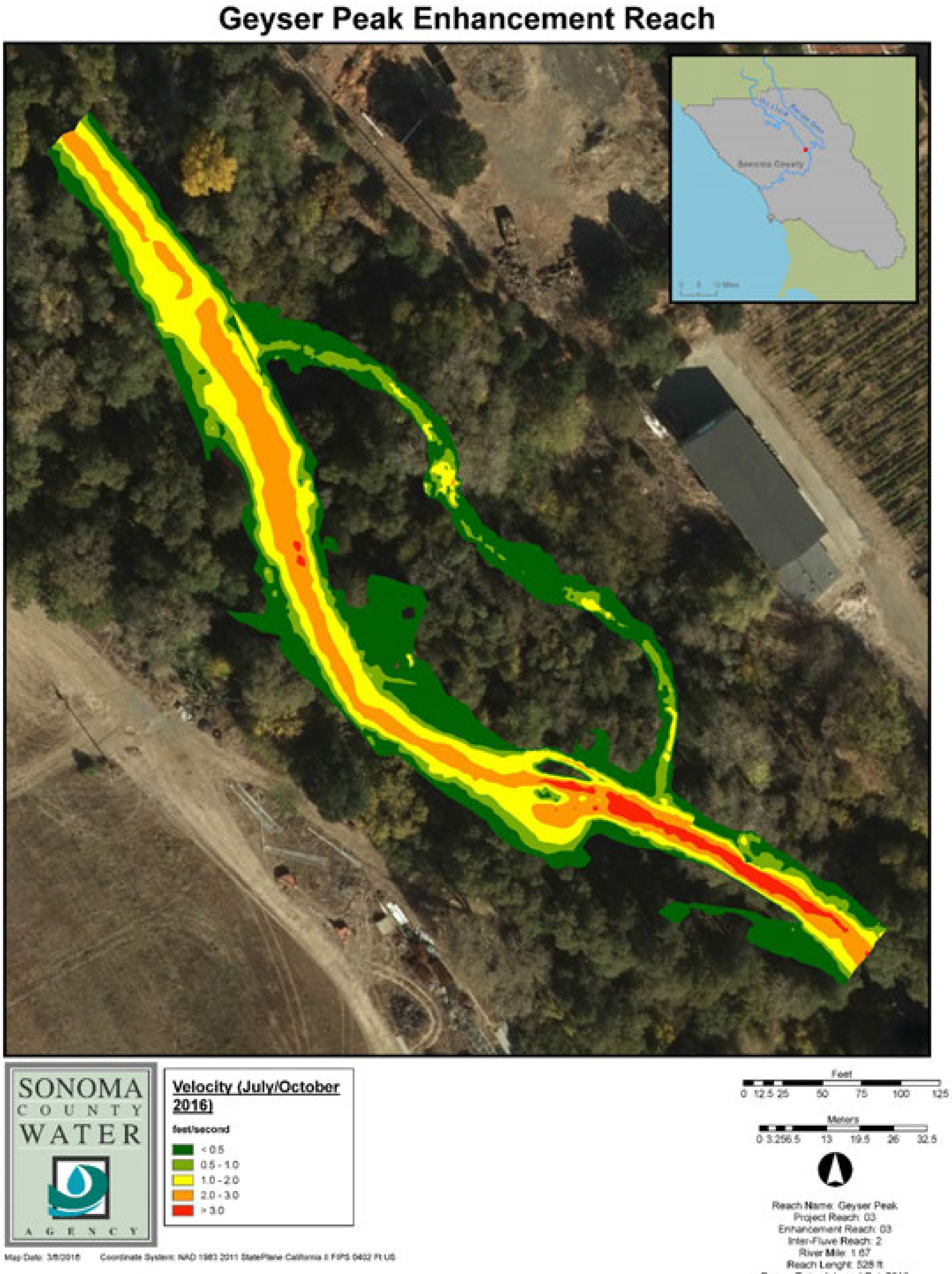


Figure 5.2.8. Measured water velocity within the Geyser Peak habitat enhancement reach in November 2016.

Figure 5.2.9. Area of optimal water velocity within the Geyser Peak habitat enhancement reach in November 2016.

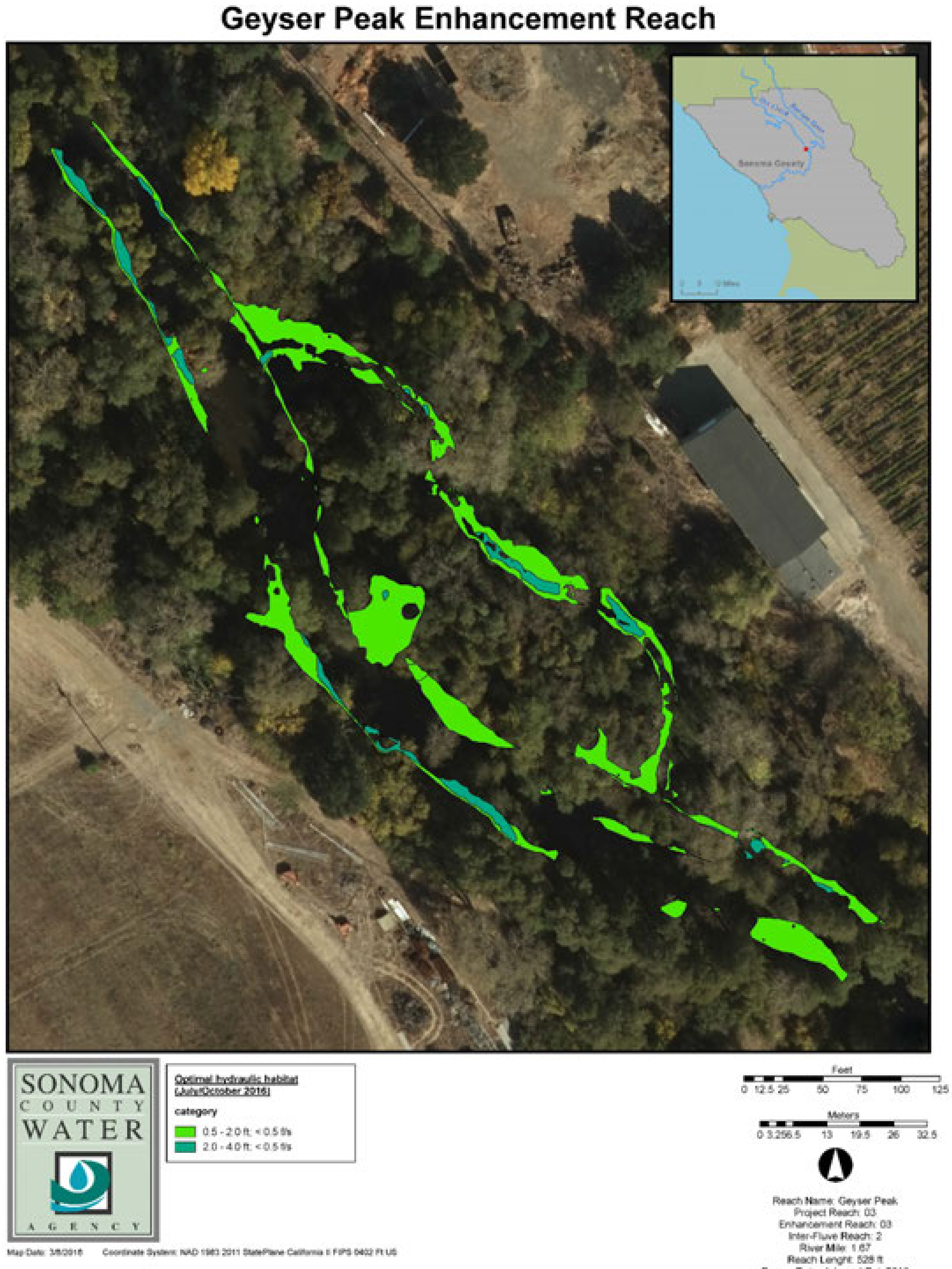


Figure 5.2.10. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within Geyser Peak habitat enhancement reach in November 2016.

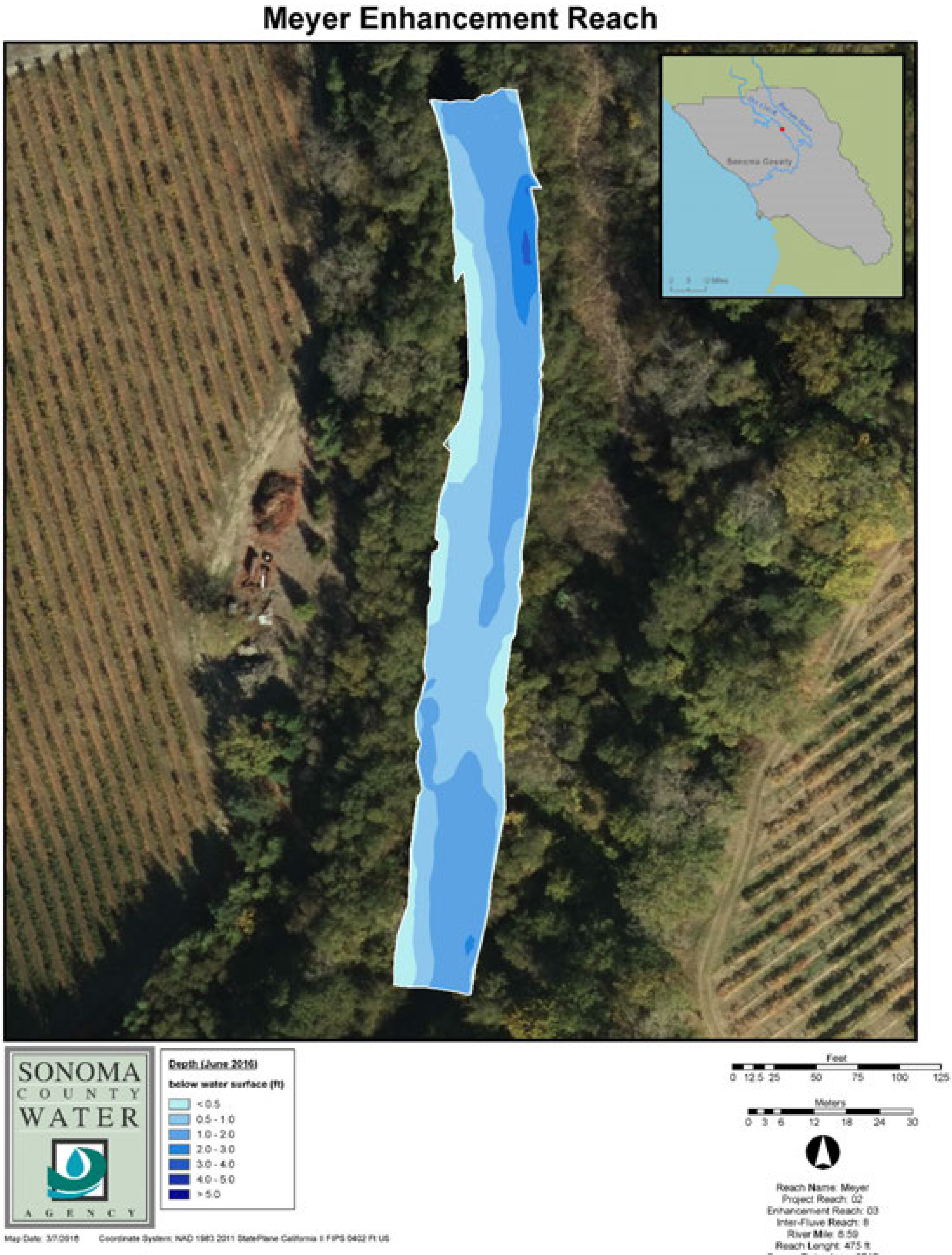


Figure 5.2.11. Measured water depth within the Meyer habitat enhancement reach during June 2016.

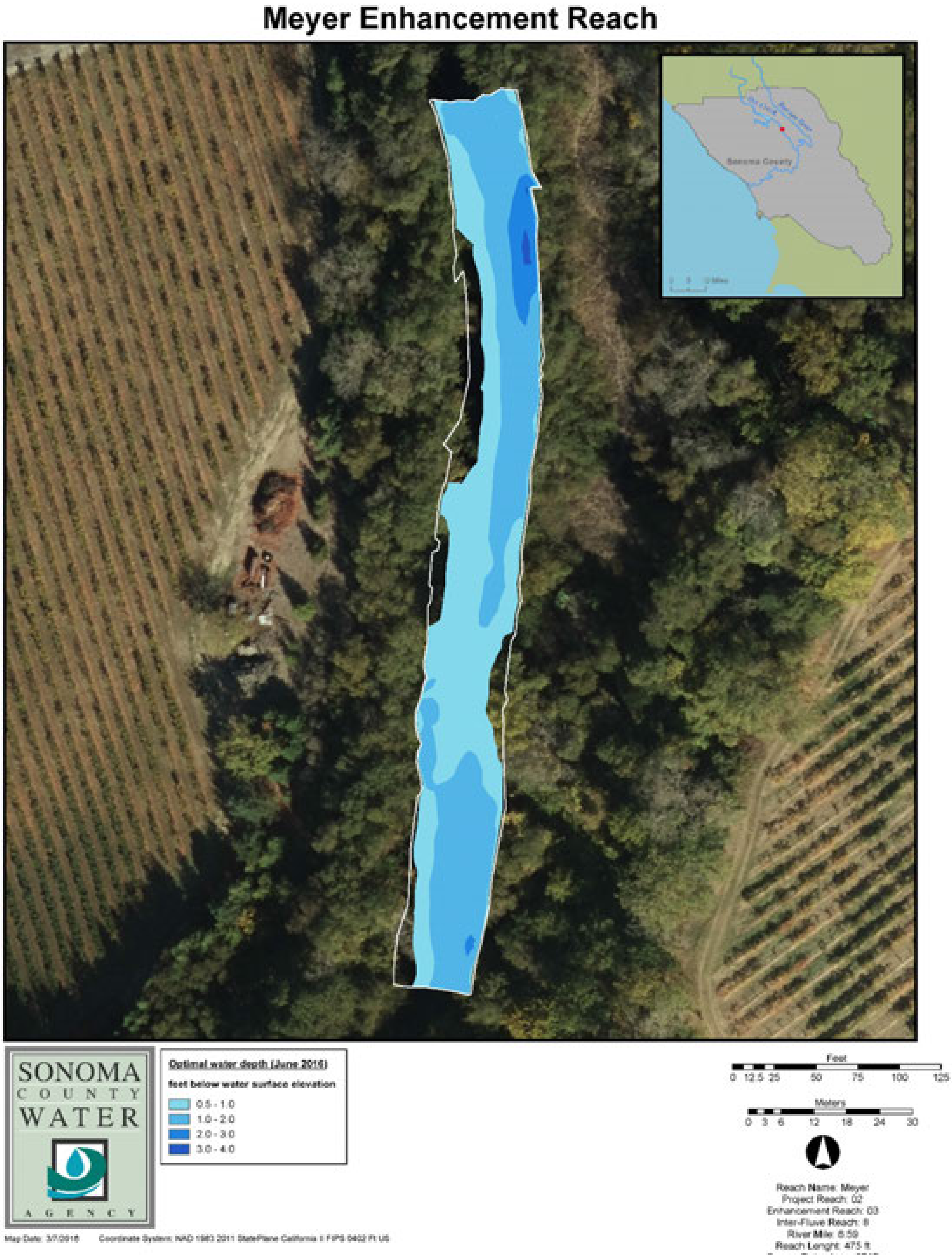


Figure 5.2.12. Area of optimal water depth within the Meyer habitat enhancement reach during June 2016.

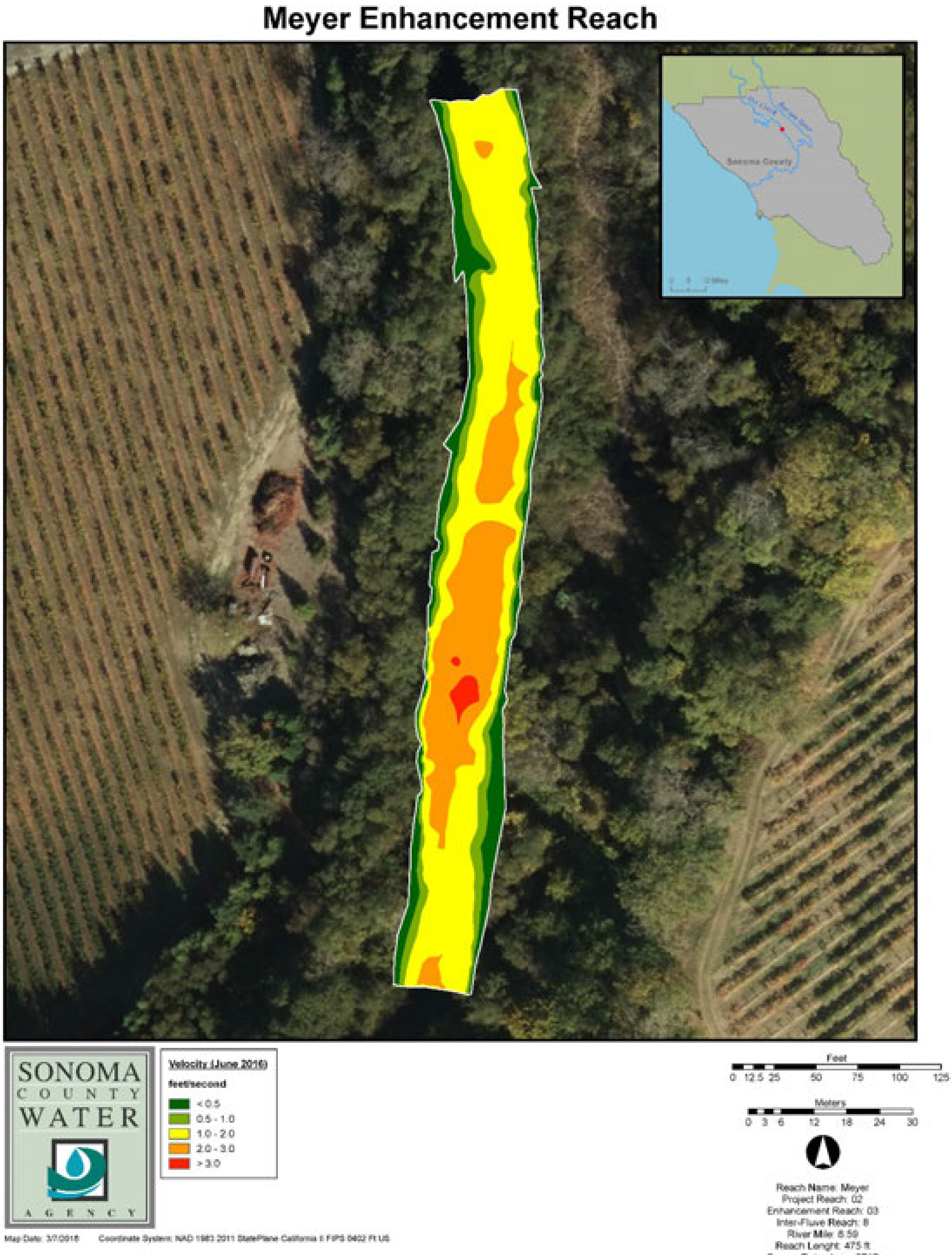


Figure 5.2.13. Measured water velocity within the Meyer habitat enhancement reach during June 2016.

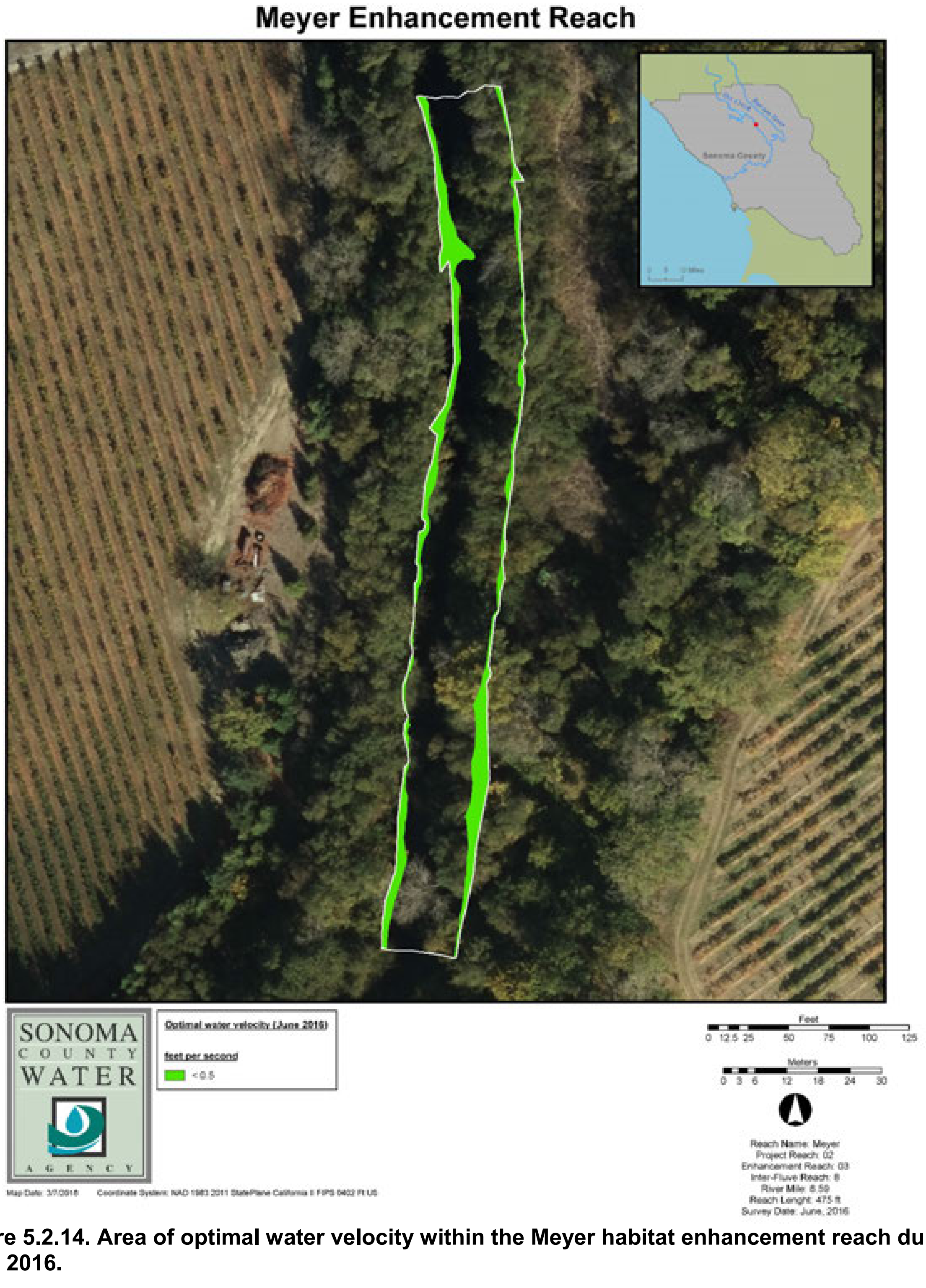


Figure 5.2.14. Area of optimal water velocity within the Meyer habitat enhancement reach during June 2016.

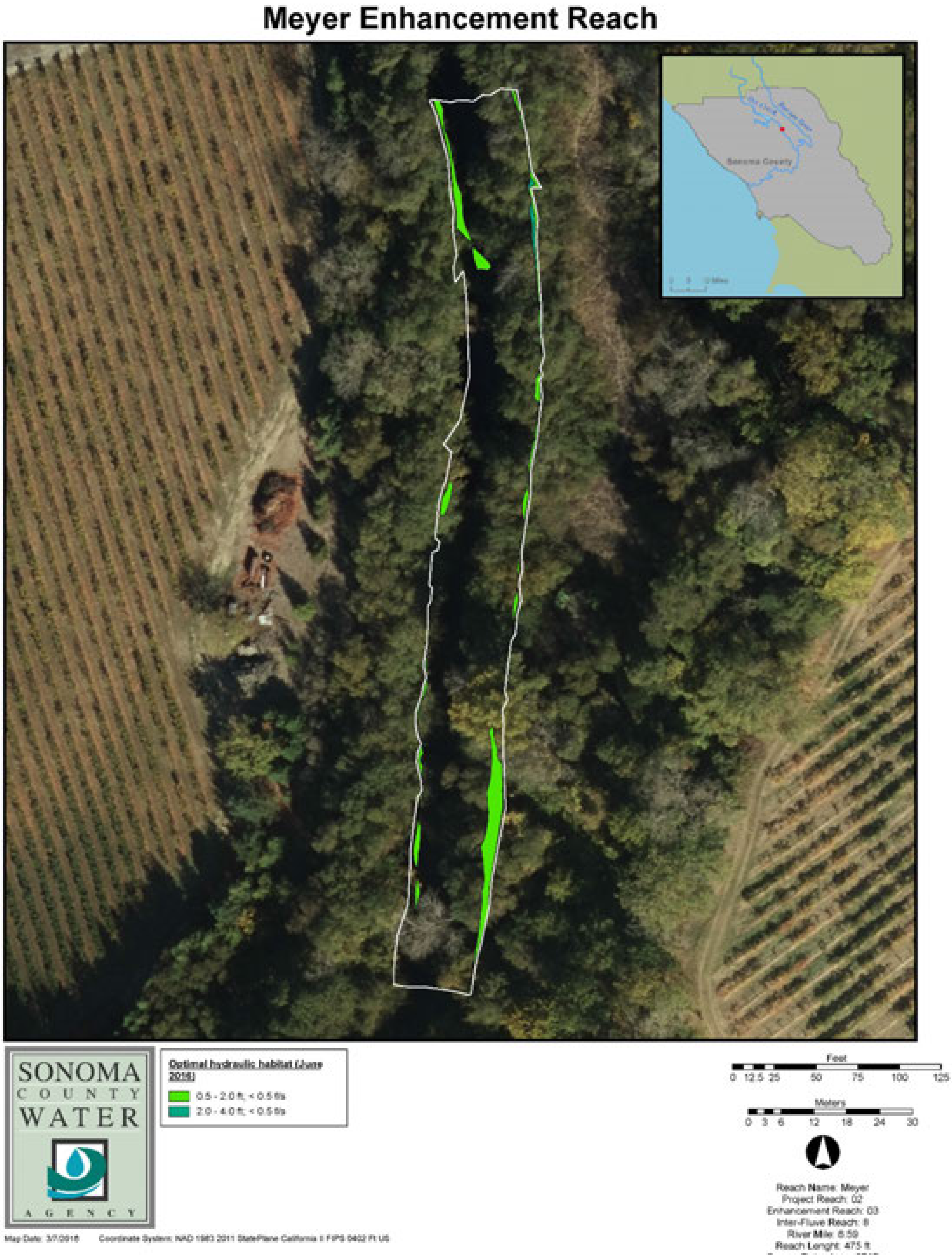


Figure 5.2.15. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Meyer habitat enhancement reach during June 2016.

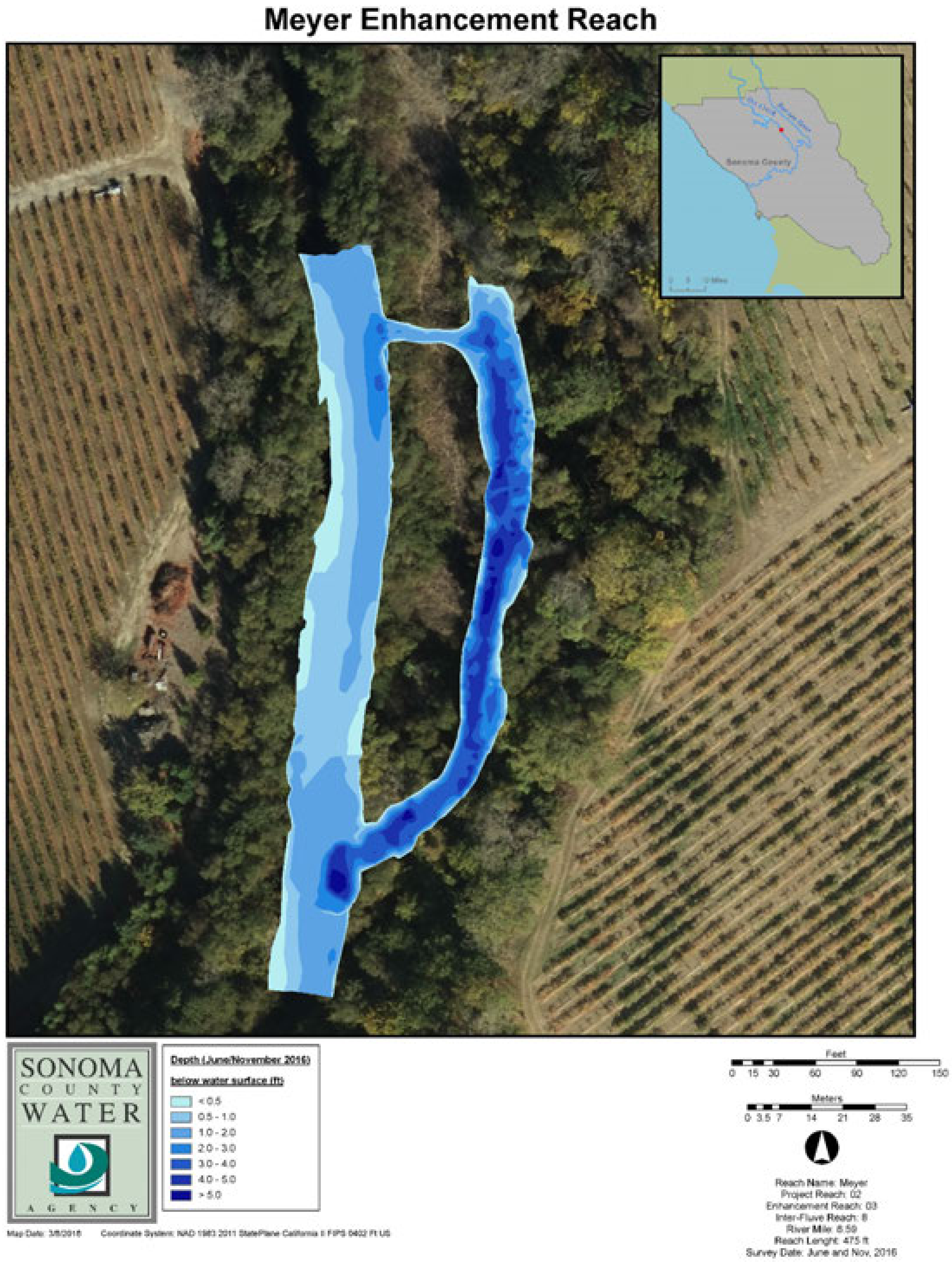


Figure 5.2.16. Measured water depth within the Meyer habitat enhancement reach during November 2016.

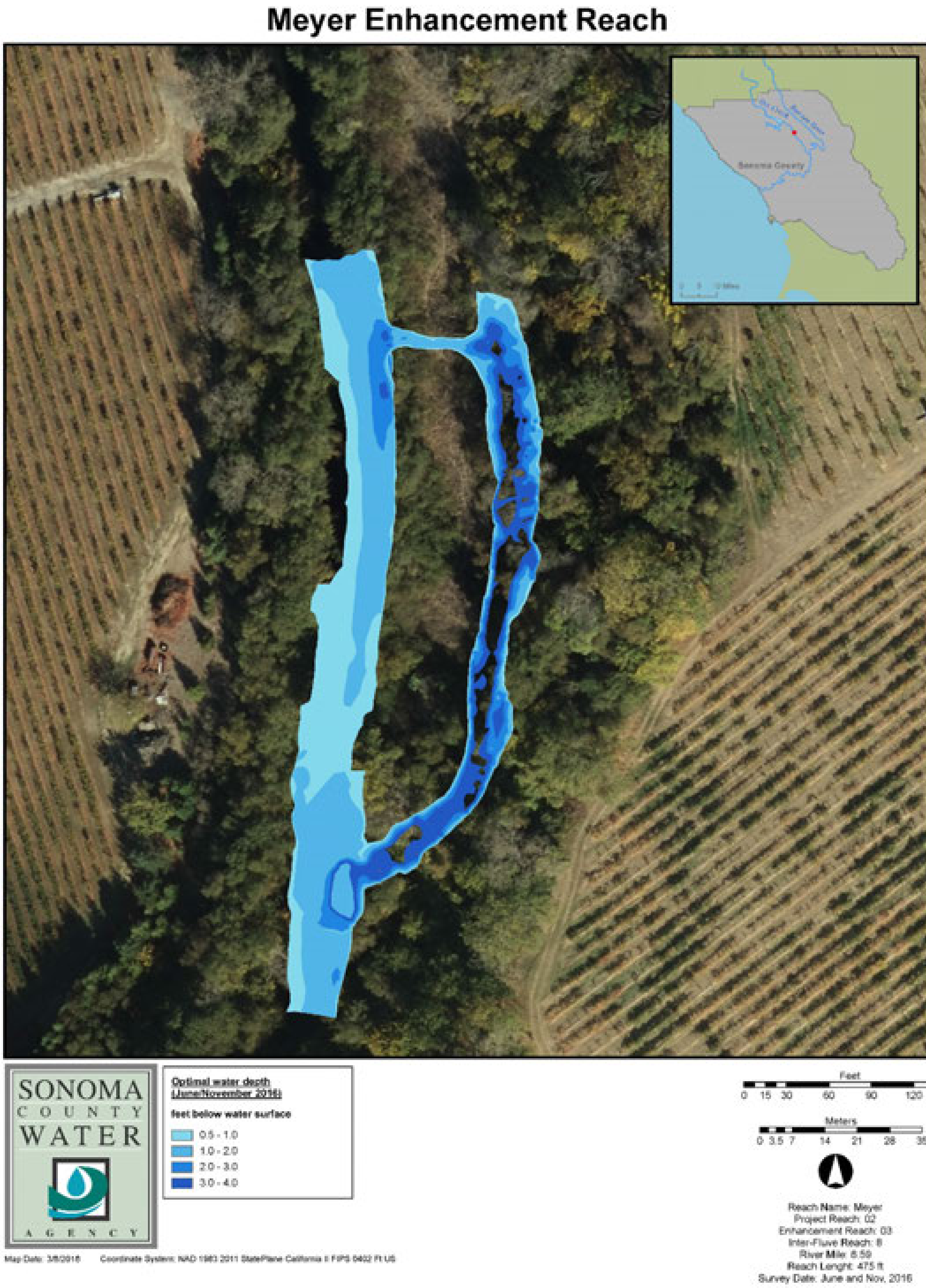


Figure 5.2.17. Area of optimal water depth within the Meyer habitat enhancement reach during November 2016.

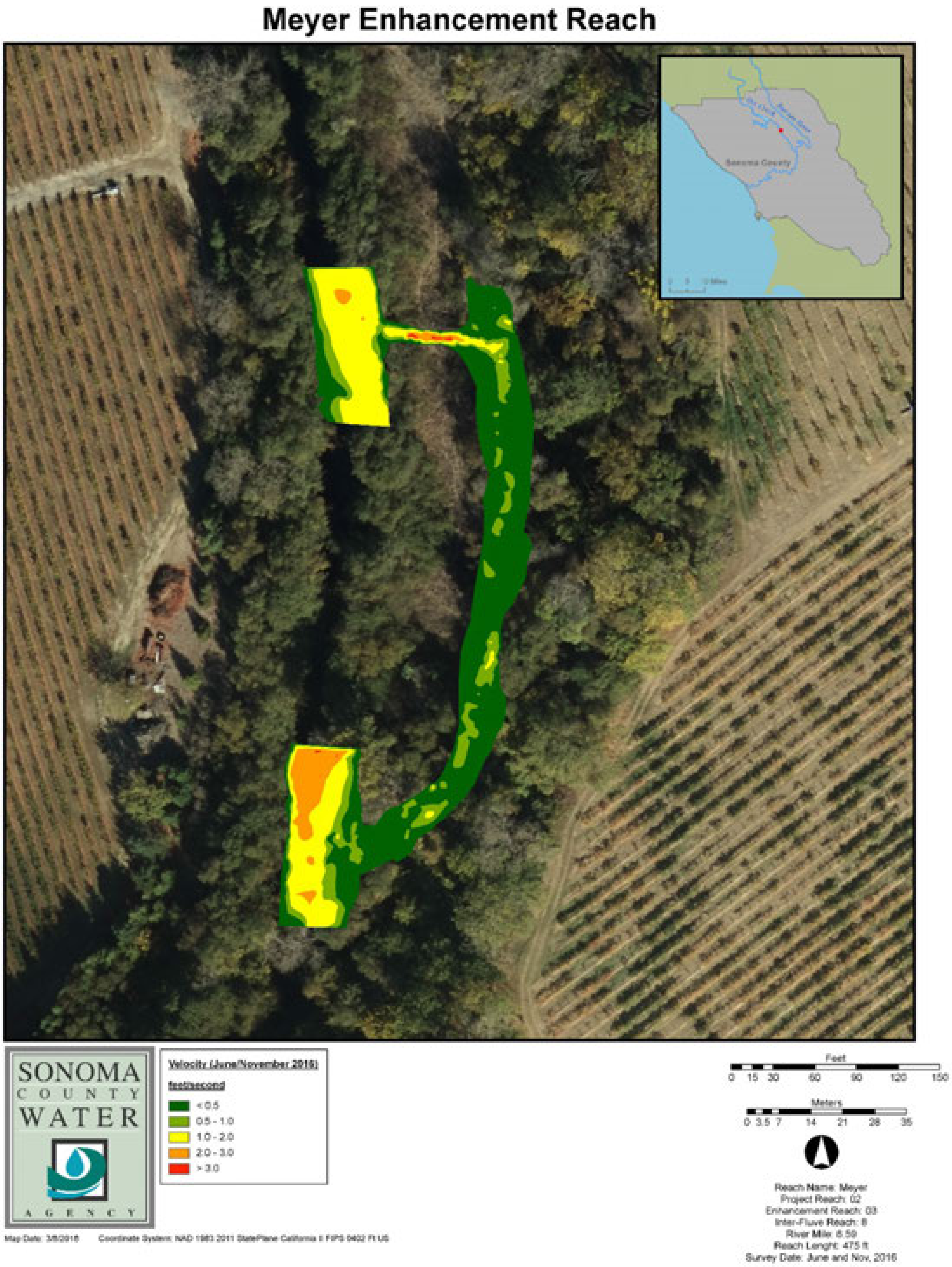


Figure 5.2.18. Measured water velocity within the Meyer habitat enhancement reach during November 2016.

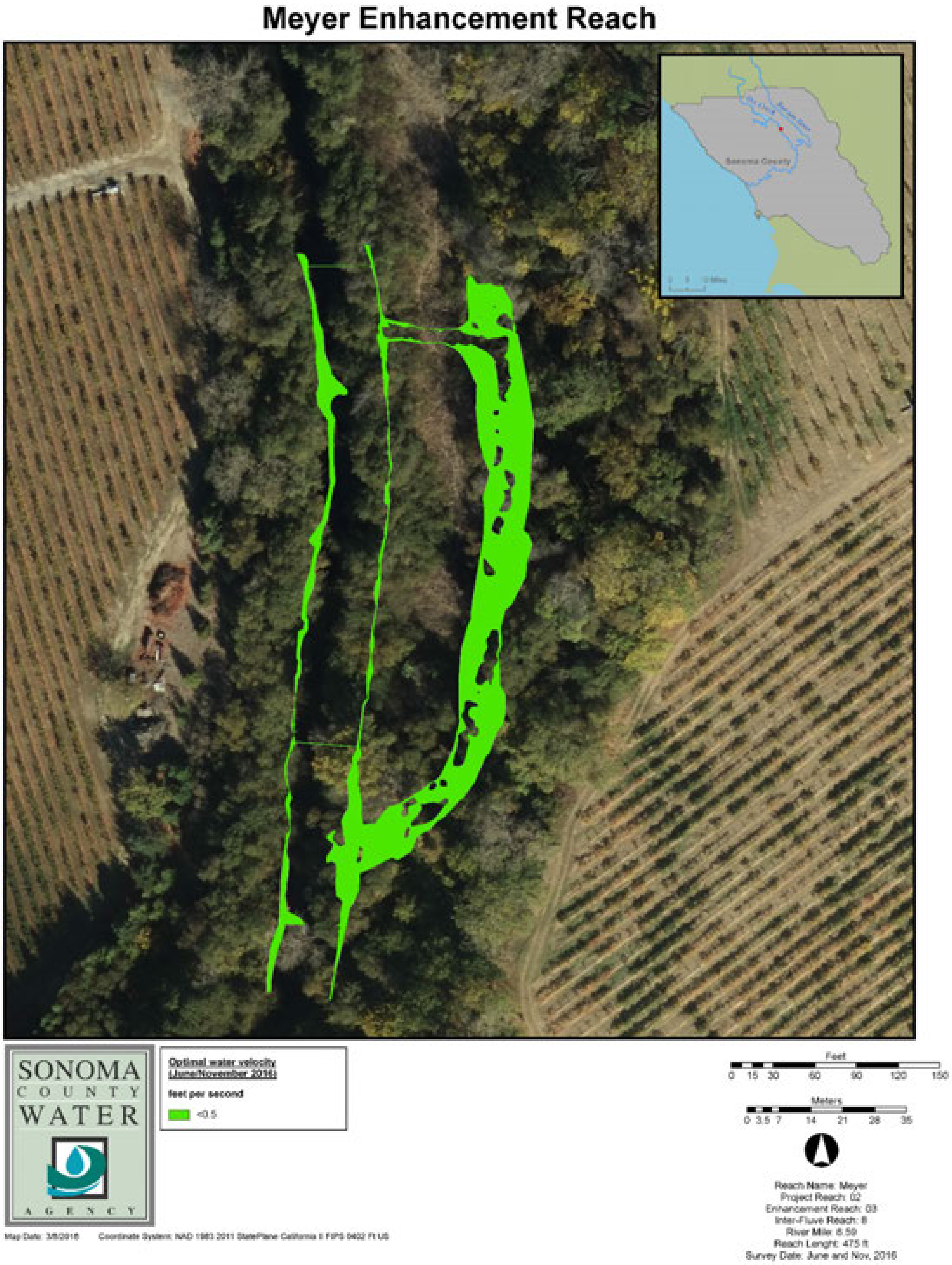


Figure 5.2.19. Area of optimal water velocity within the Meyer habitat enhancement reach during November 2016.

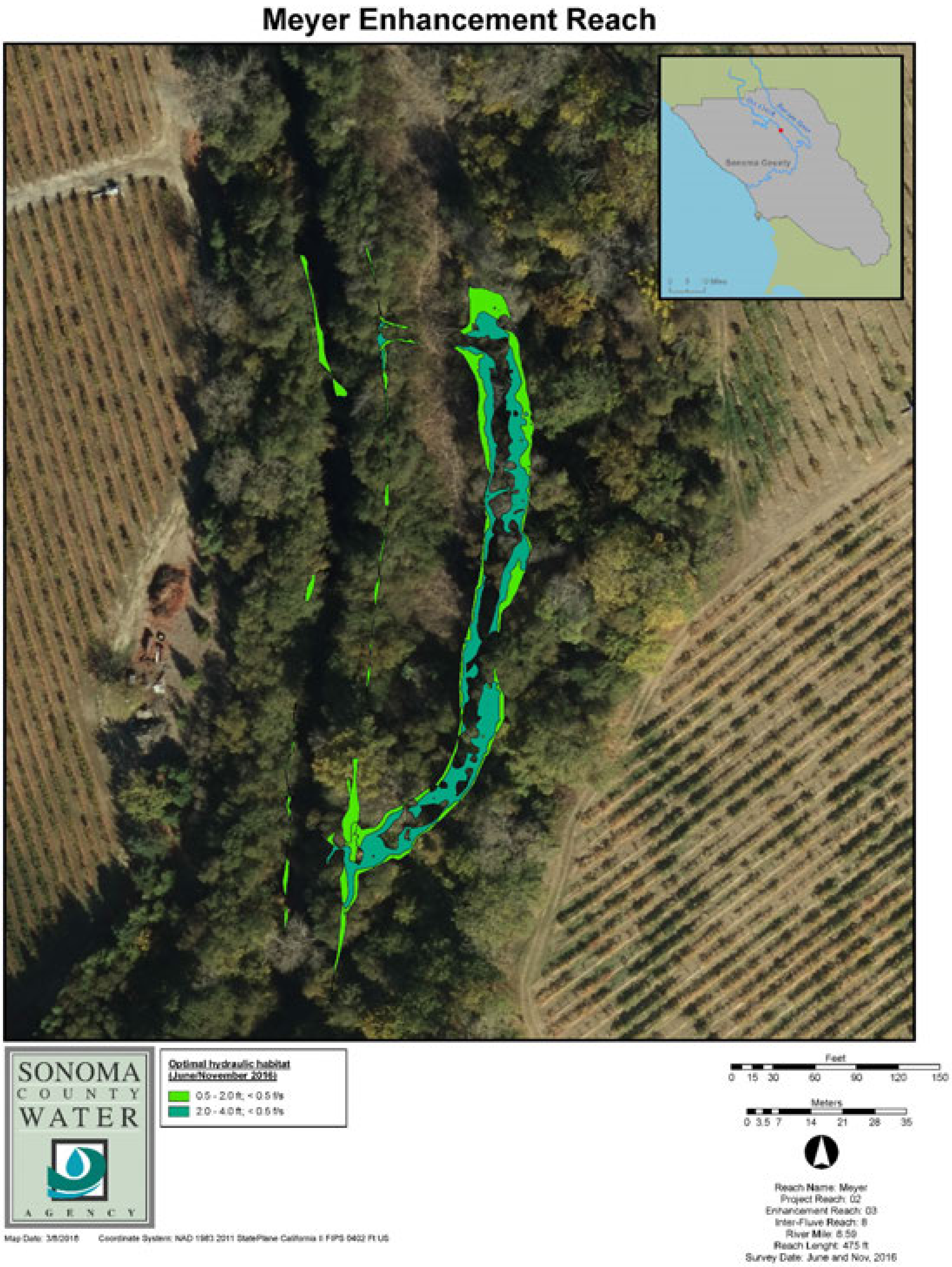


Figure 5.2.20. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within Meyer habitat enhancement reach during November 2016.

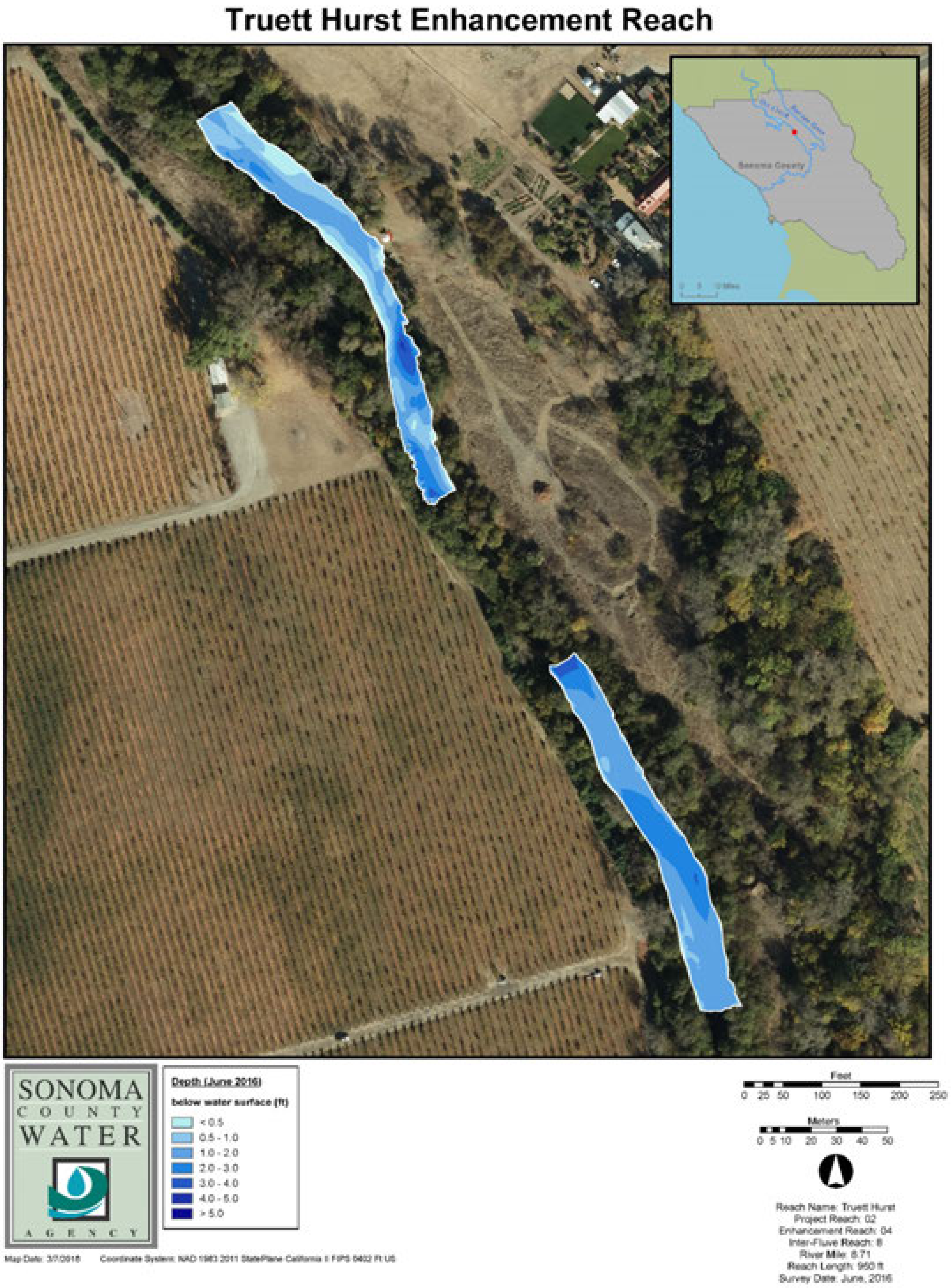


Figure 5.2.21. Measured water depth within the Truett Hurst habitat enhancement reach during June 2016.

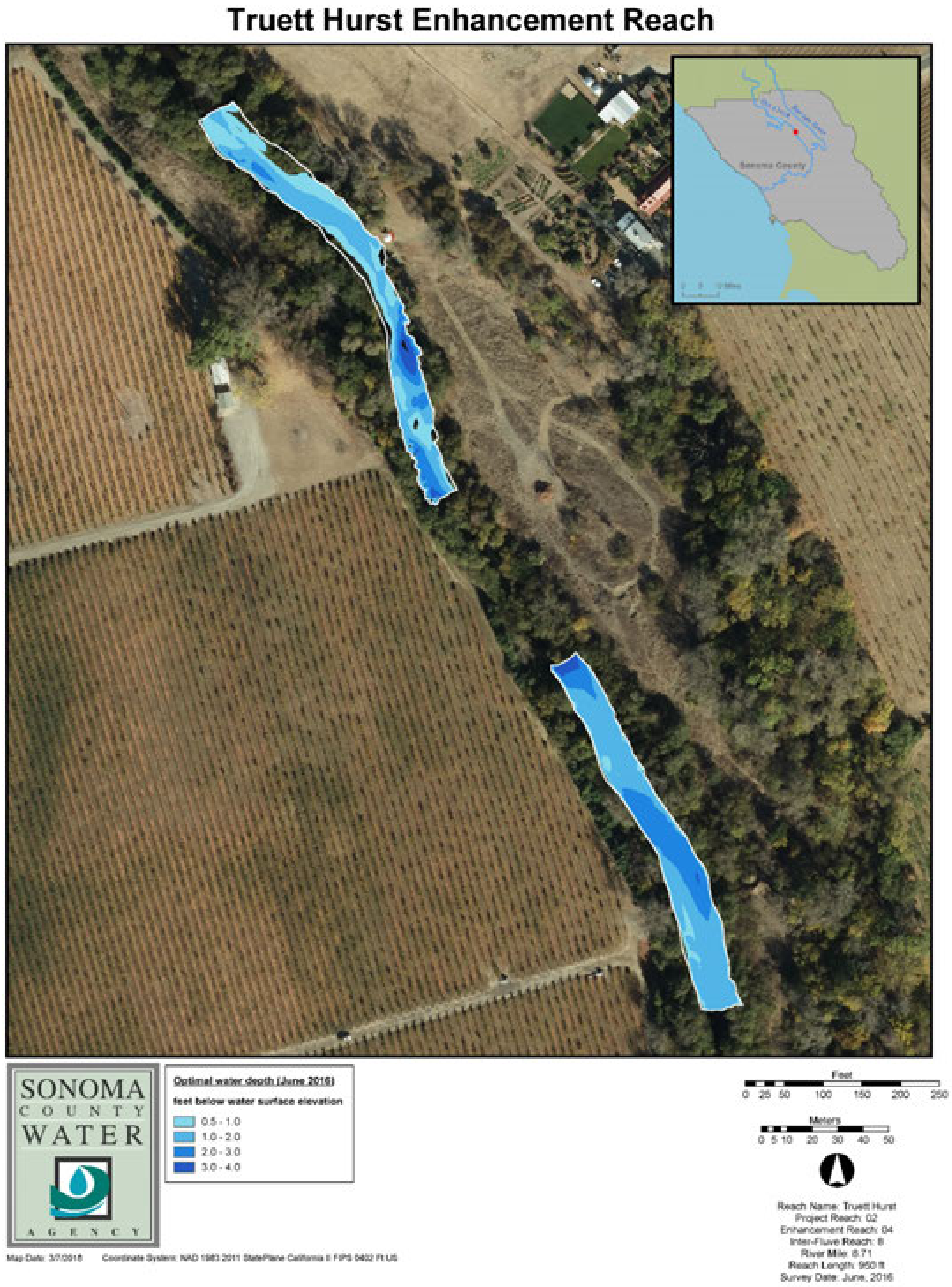


Figure 5.2.22. Area of optimal water depth within the Truett Hurst habitat enhancement reach during June 2016.

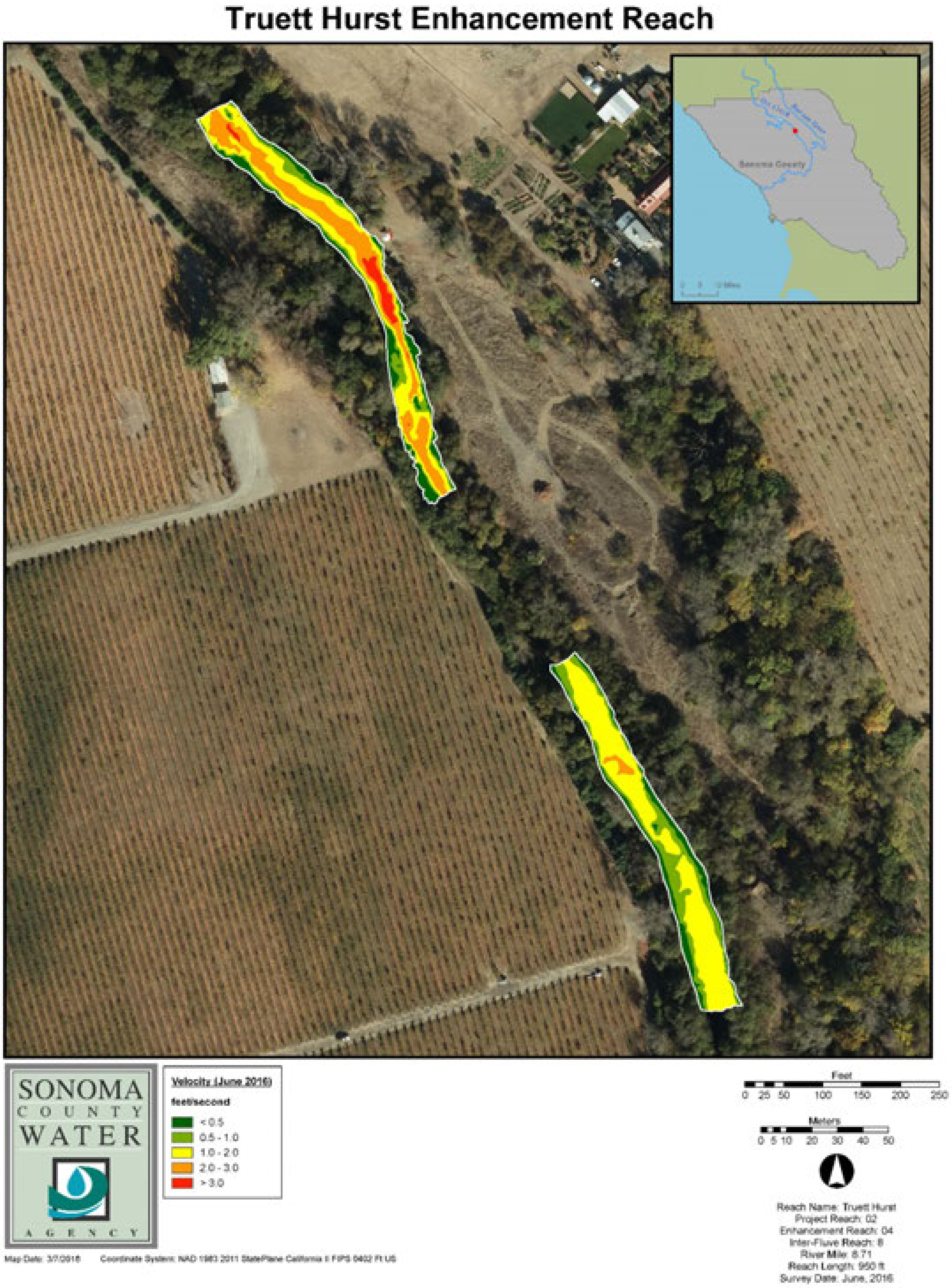


Figure 5.2.23. Measured water velocity within the Truett Hurst habitat enhancement reach during June 2016.

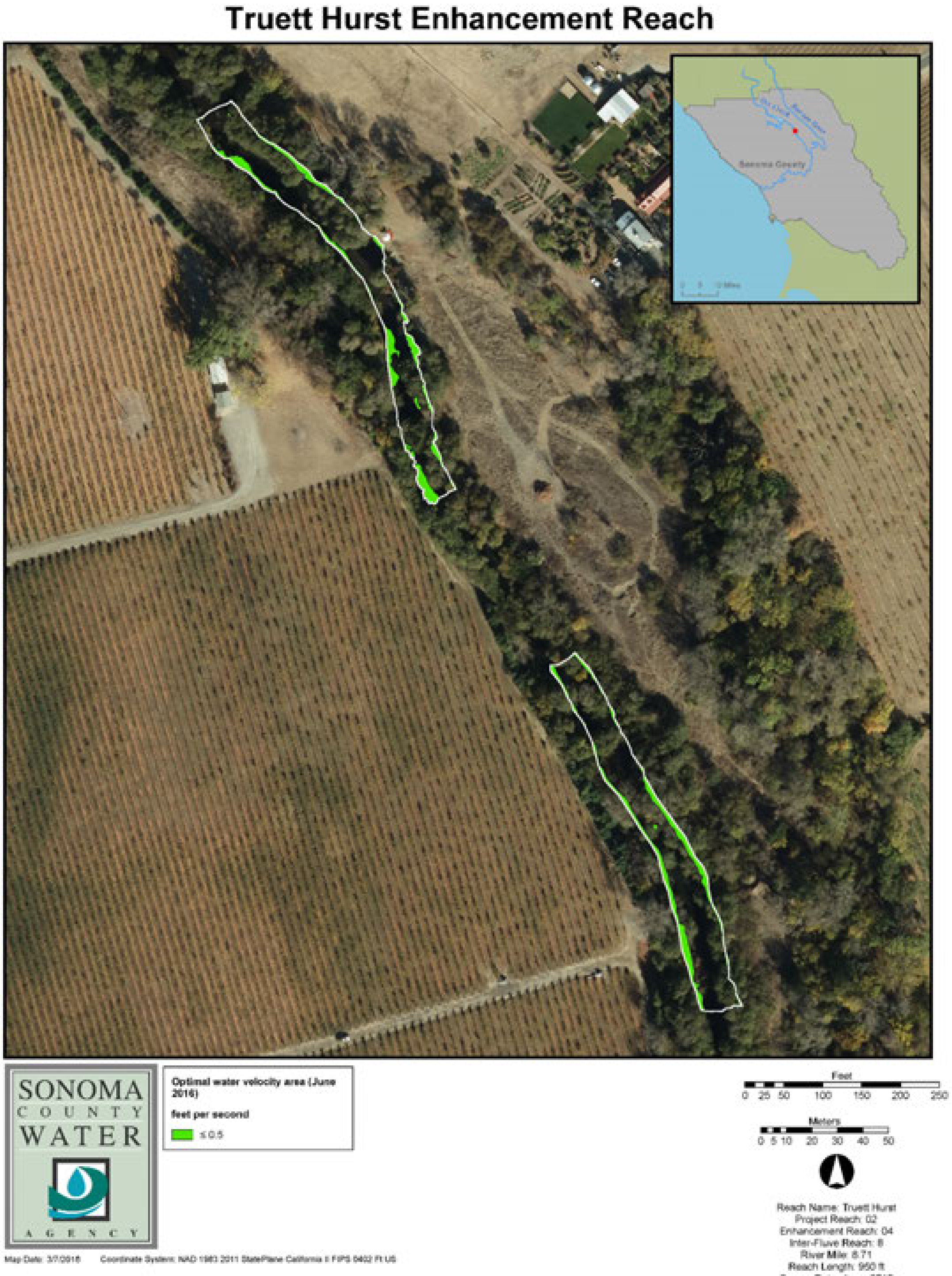


Figure 5.2.24. Area of optimal water velocity within the Truett Hurst habitat enhancement reach during June 2016.

Figure 5.2.25. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Truett Hurst habitat enhancement reach during June 2016.

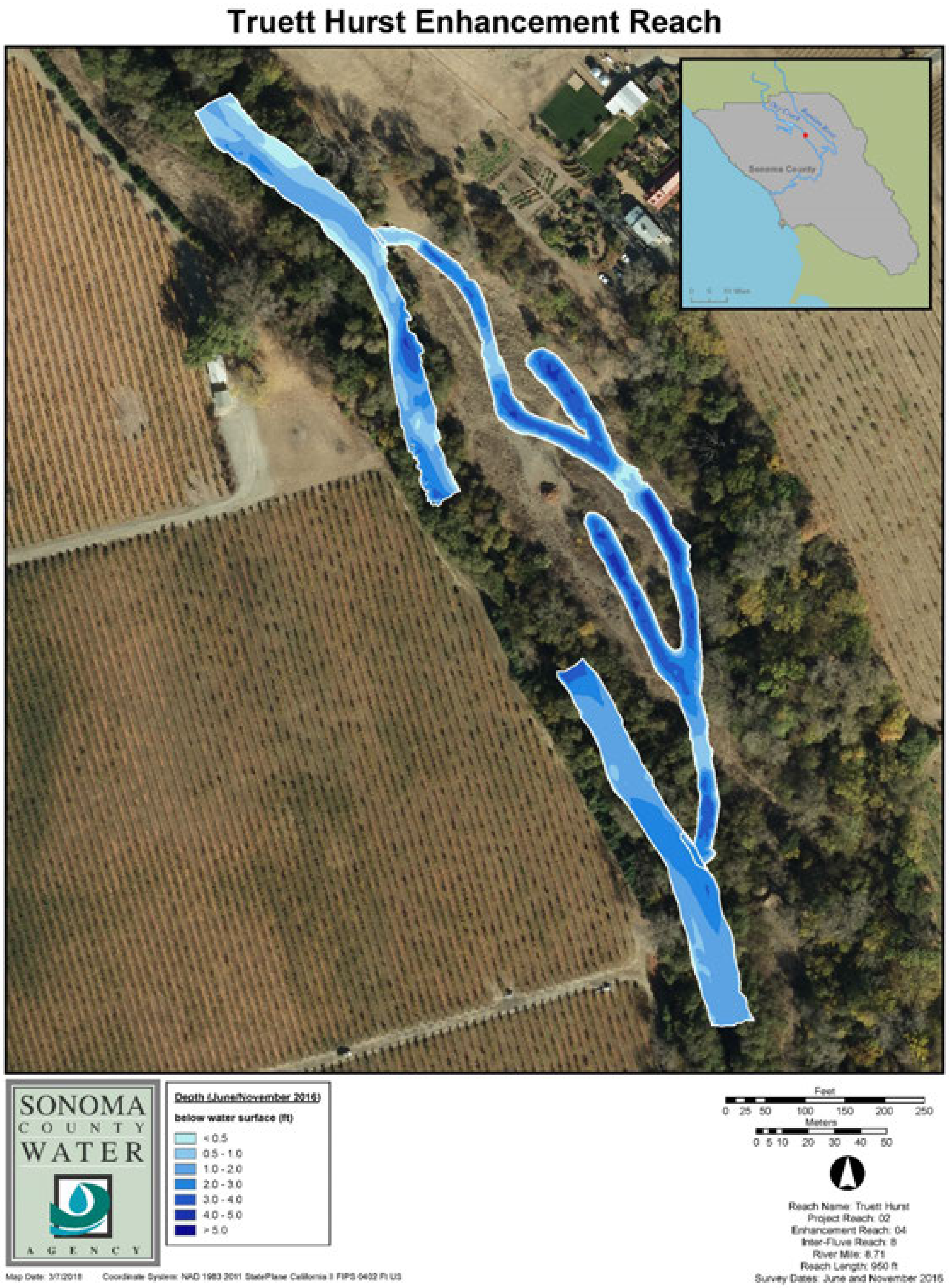


Figure 5.2.26. Measured water depth within the Truett Hurst habitat enhancement reach during November 2016.

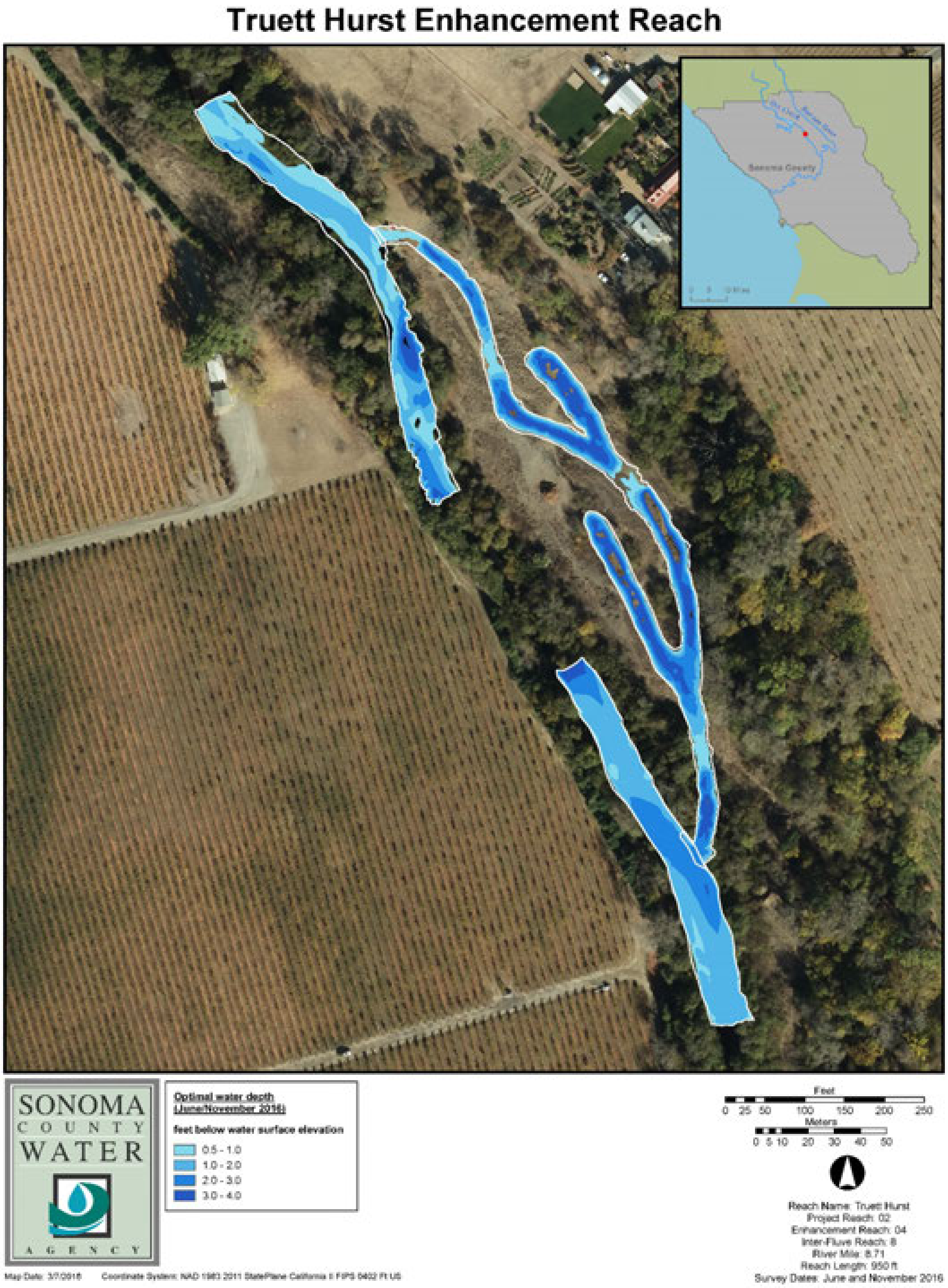


Figure 5.2.27. Area of optimal water depth within the Truett Hurst habitat enhancement reach during November 2016.

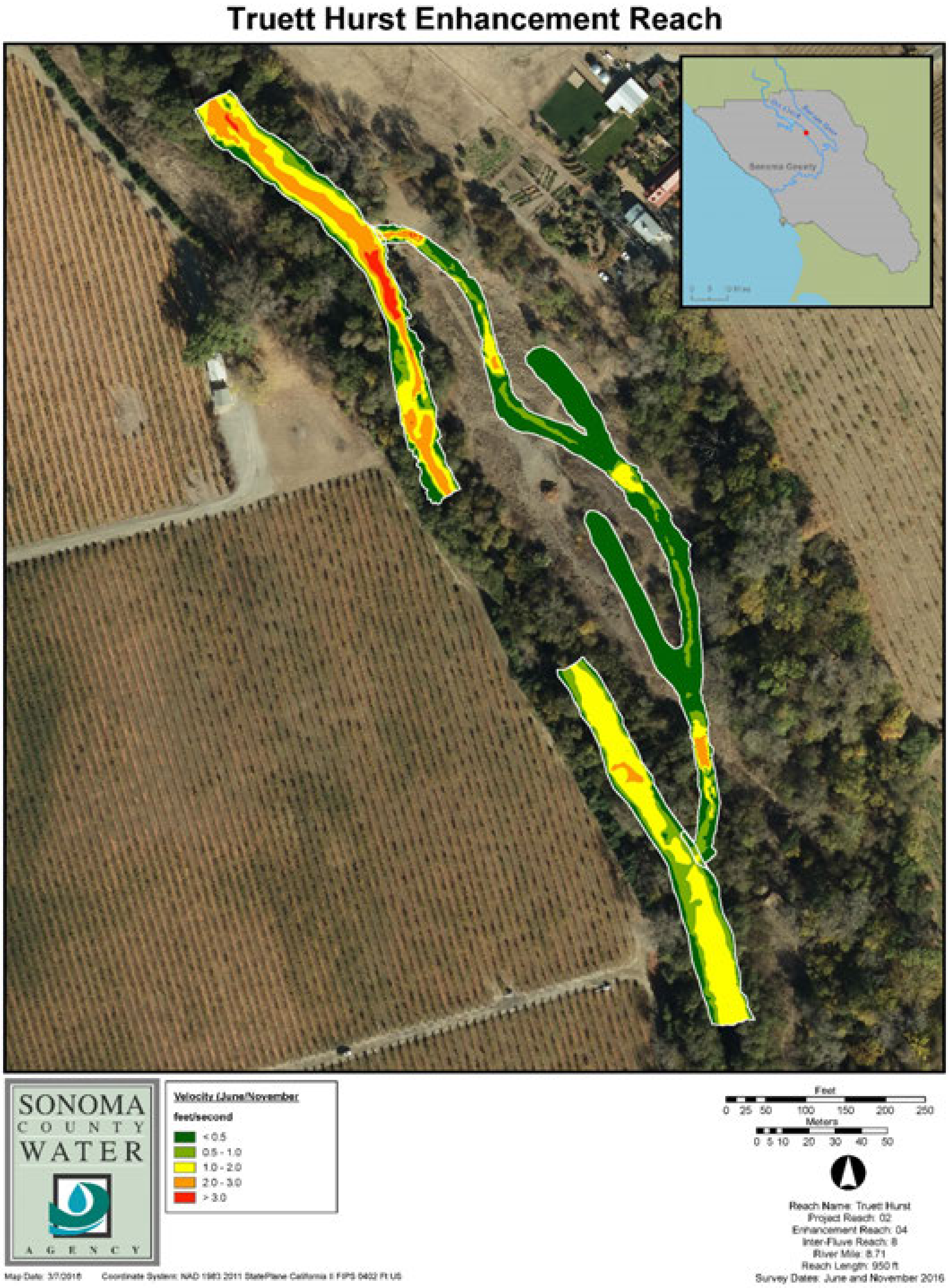


Figure 5.2.28. Measured water velocity within the Truett Hurst habitat enhancement reach during November 2016.

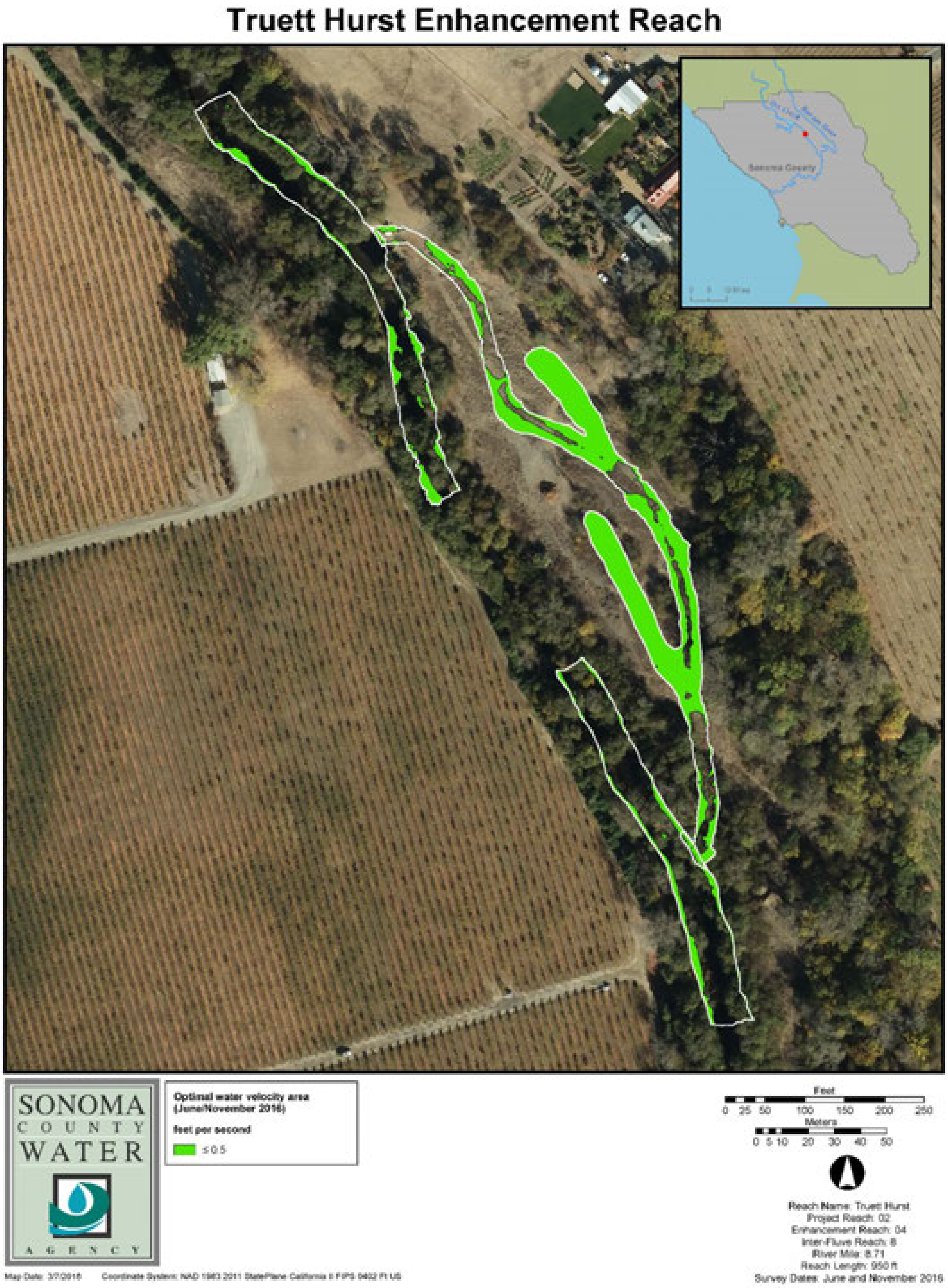


Figure 5.2.29. Area of optimal water velocity within the Truett Hurst habitat enhancement reach during November 2016.

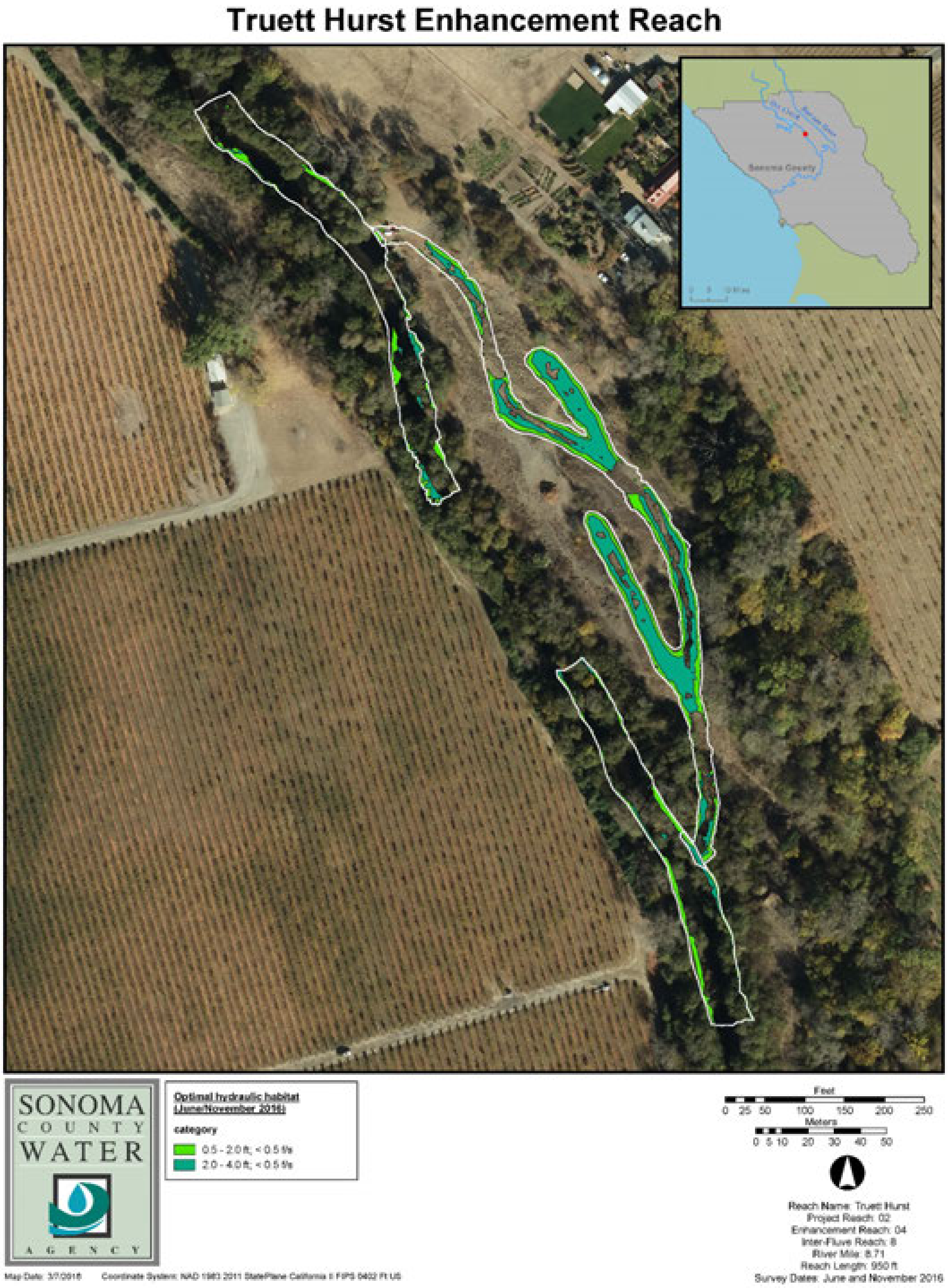


Figure 5.2.30. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within Truett Hurst habitat enhancement reach during November 2016.

Shelter Value

Field Crews inventoried instream habitat units in three enhancement reaches that make up parts of miles 2 and 3 of the Dry Creek Habitat Enhancement Project (Meyer and Truett Hurst [mile 2] and Geyser Peak [mile 3]) before habitat enhancement (June/July) and after habitat enhancement (October/November 2016) (Table 5.2.3-Table 5.2.5, Figure 5.2.32-Figure 5.2.43). The crews determined habitat type, and assigned shelter scores and estimated percent overhead cover to determine a shelter value for each habitat unit. Most habitat units observed within the enhanced areas nearly met or exceeded a shelter value of 80, while habitat units within the mainstem in unenhanced

Table 5.2. 3. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Geyser Peak enhancement reach in July (pre-enhancement) and October (with enhanced area) 2016.

Pre-enhancement (July 2016)				
Habitat Unit #	**Habitat Type**	**Shelter Score**	**Percent Cover**	**Shelter Value**
HU01	Flatwater	3	25	75
HU02	Riffle	2	15	30
HU03	Pool	3	10	30
HU04	Flatwater	2	10	20
HU05	Flatwater	2	20	40
HU06	Scour Pool	3	40	120
Enhanced area (October 2016)				
Habitat Unit #	**Habitat Type**	**Shelter Score**	**Percent Cover**	**Shelter Value**
HU01	Not recorded	Not recorded	Not recorded	Not recorded
HU02	Backwater	3	50	150
HU03	Backwater	3	40	120
HU04	Flatwater	3	30	90
HU05	Pool	3	25	75
HU06	Riffle	3	30	90
HU07	Pool	3	30	90
HU08	Riffle	3	25	75
HU09	Pool	3	25	75

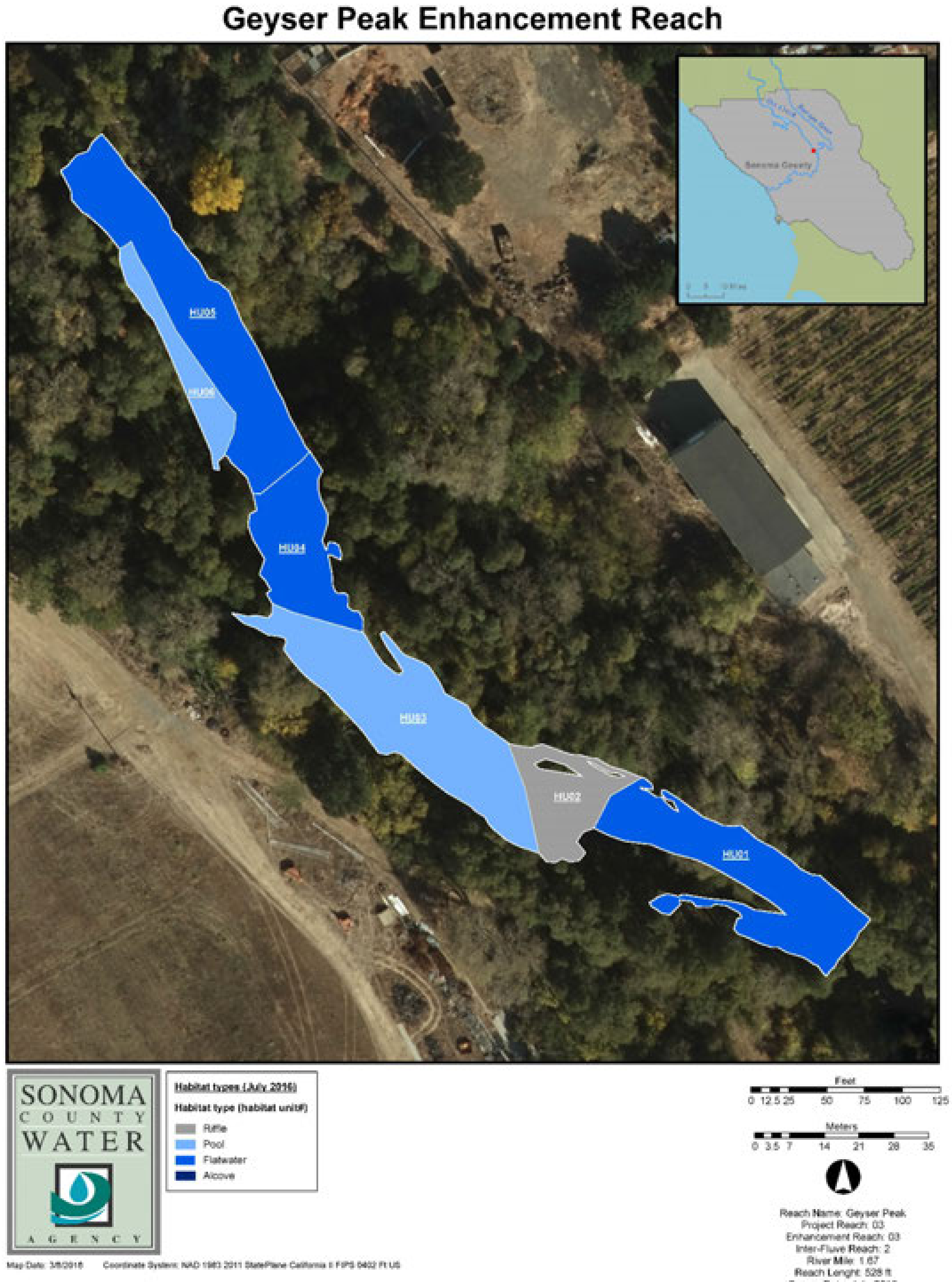


Figure 5.2.31. Habitat unit number and type within the Geyser Peak enhancement reach in June 2016.

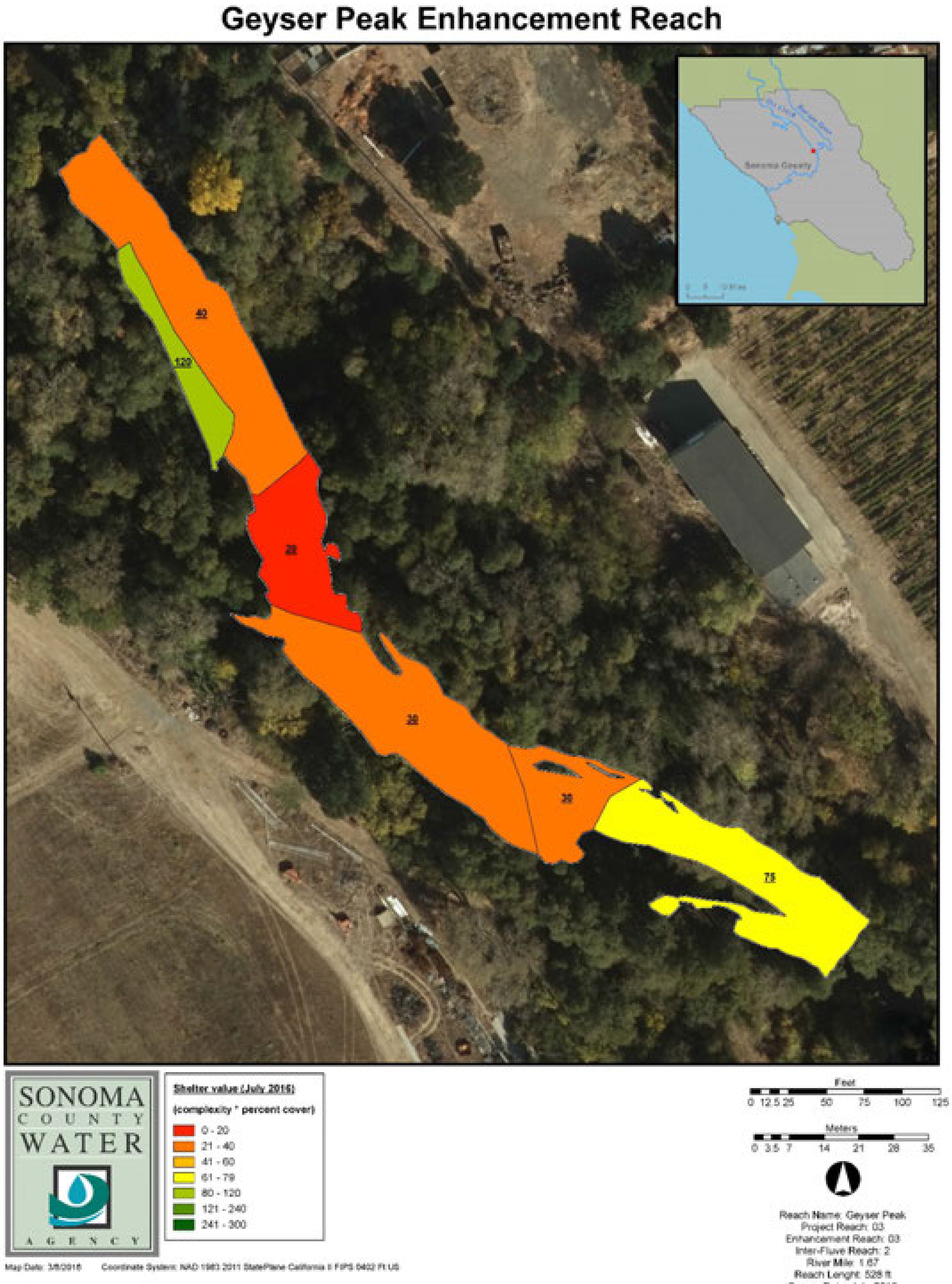


Figure 5.2.32. Habitat unit shelter values for the Geyser Peak enhancement reach in June 2016.

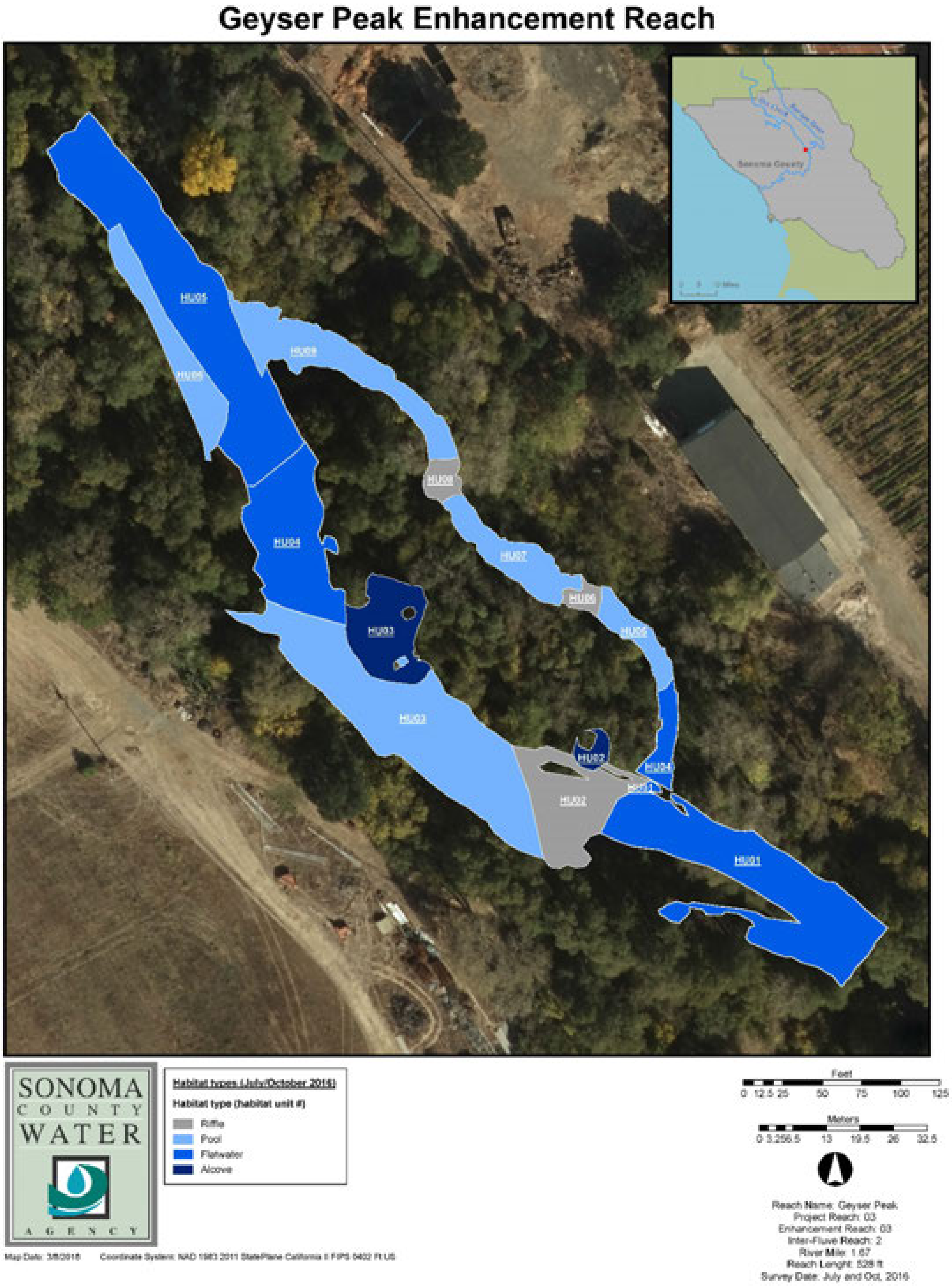


Figure 5.2.33. Habitat unit number and type within the Geyser Peak enhancement reach in June and October 2016.

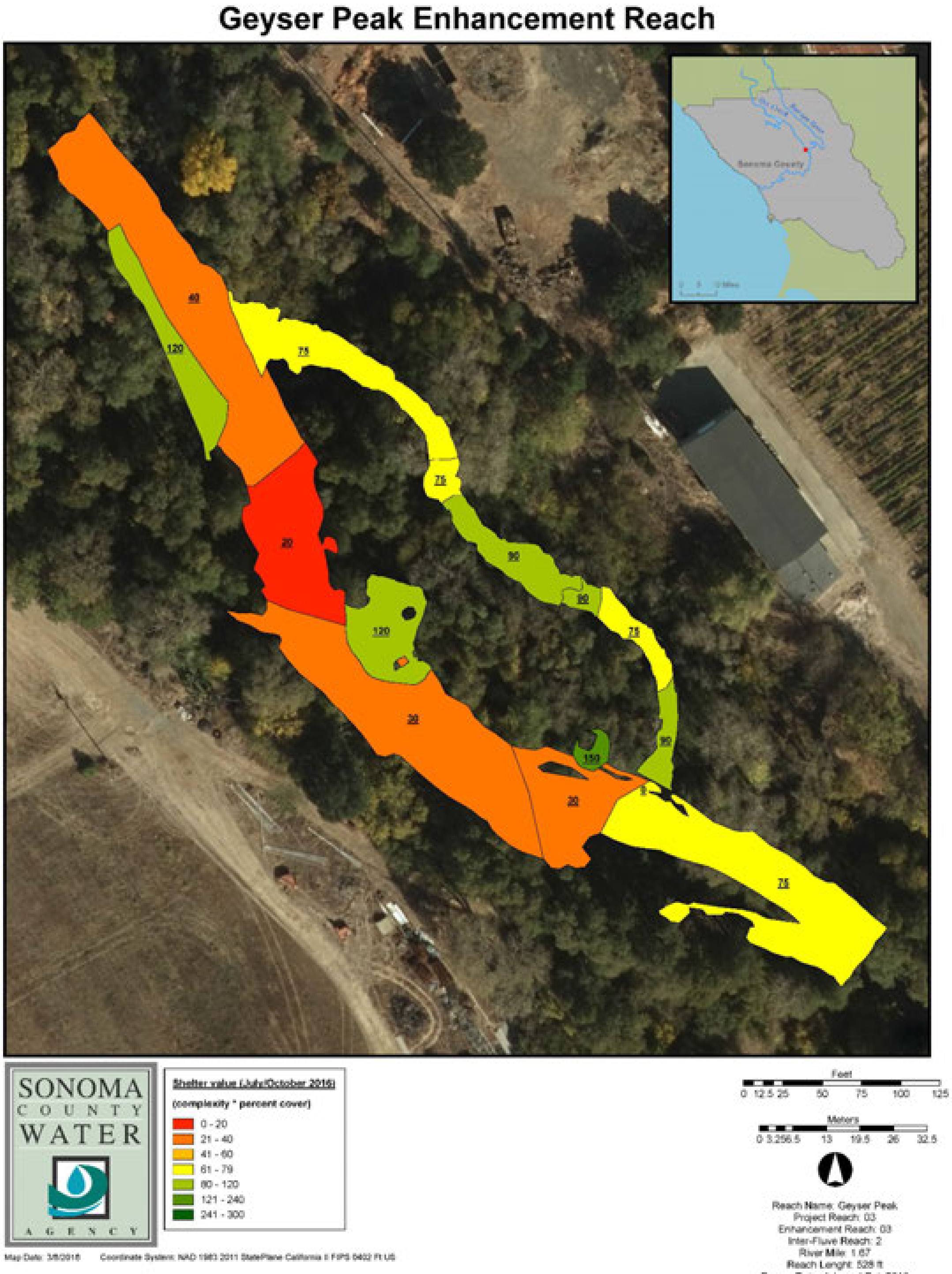


Figure 5.2.34. Habitat unit shelter values for the Geyser Peak enhancement reach in June and October 2016.

Table 5.2. 4. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Meyer enhancement reach in in June (pre-enhancement) and November (with enhanced area) 2016.

Pre-enhancement (June 2016)				
Habitat Unit #	**Habitat Type**	**Shelter Score**	**Percent Cover**	**Shelter Value**
HU01	Flatwater	2	5	10
HU02	Pool	2	5	10
HU03	Flatwater	2	25	50
Enhanced area (November 2016)				
Habitat Unit #	**Habitat Type**	**Shelter Score**	**Percent Cover**	**Shelter Value**
HU01	Pool	3	15	45
HU02	Flatwater	3	30	90
HU03	Backwater	3	45	135
HU04	Riffle	3	25	75
HU05	Flatwater	3	10	30

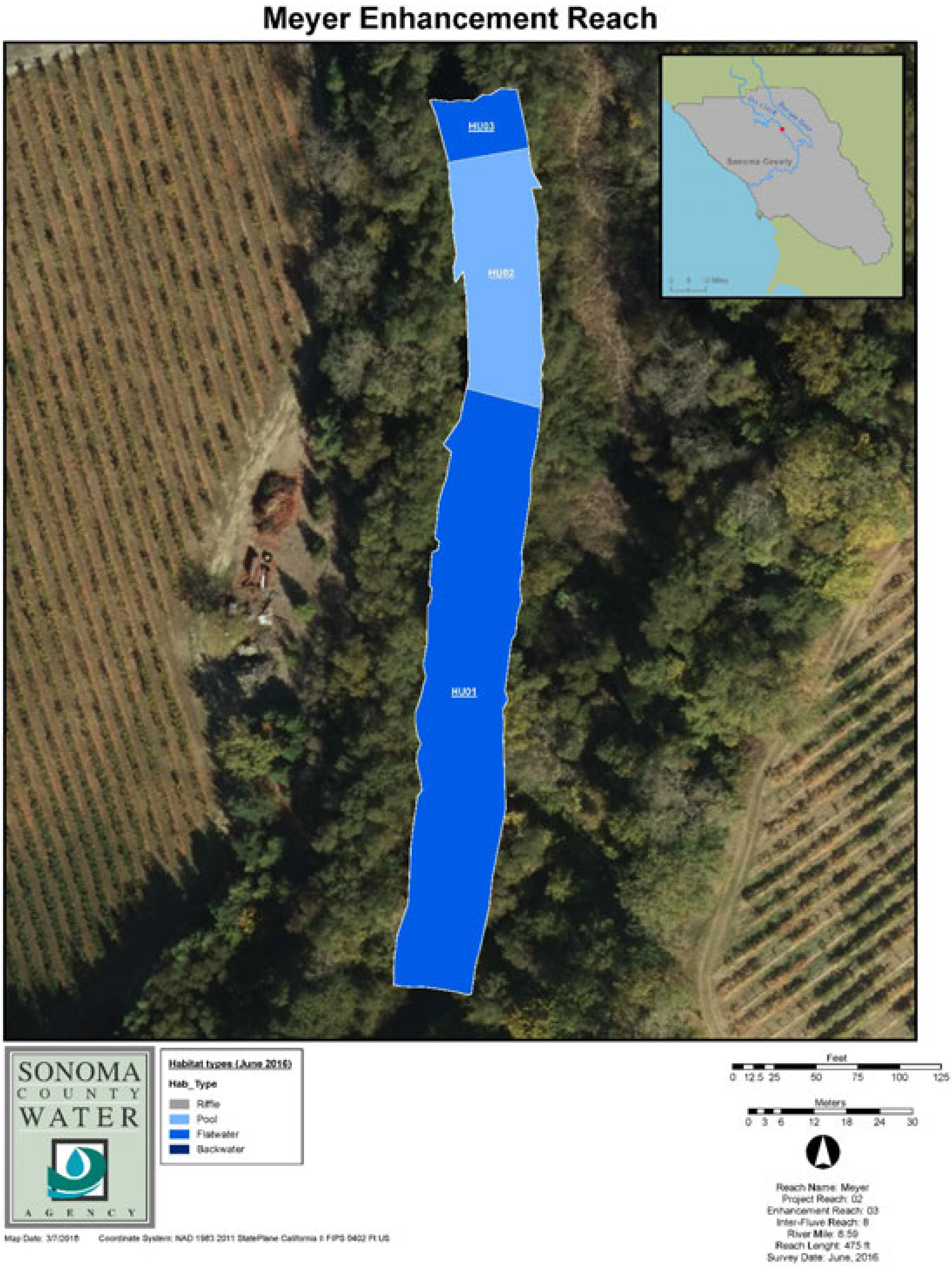


Figure 5.2.35. Habitat unit number and type within the Meyer enhancement reach in June 2016.

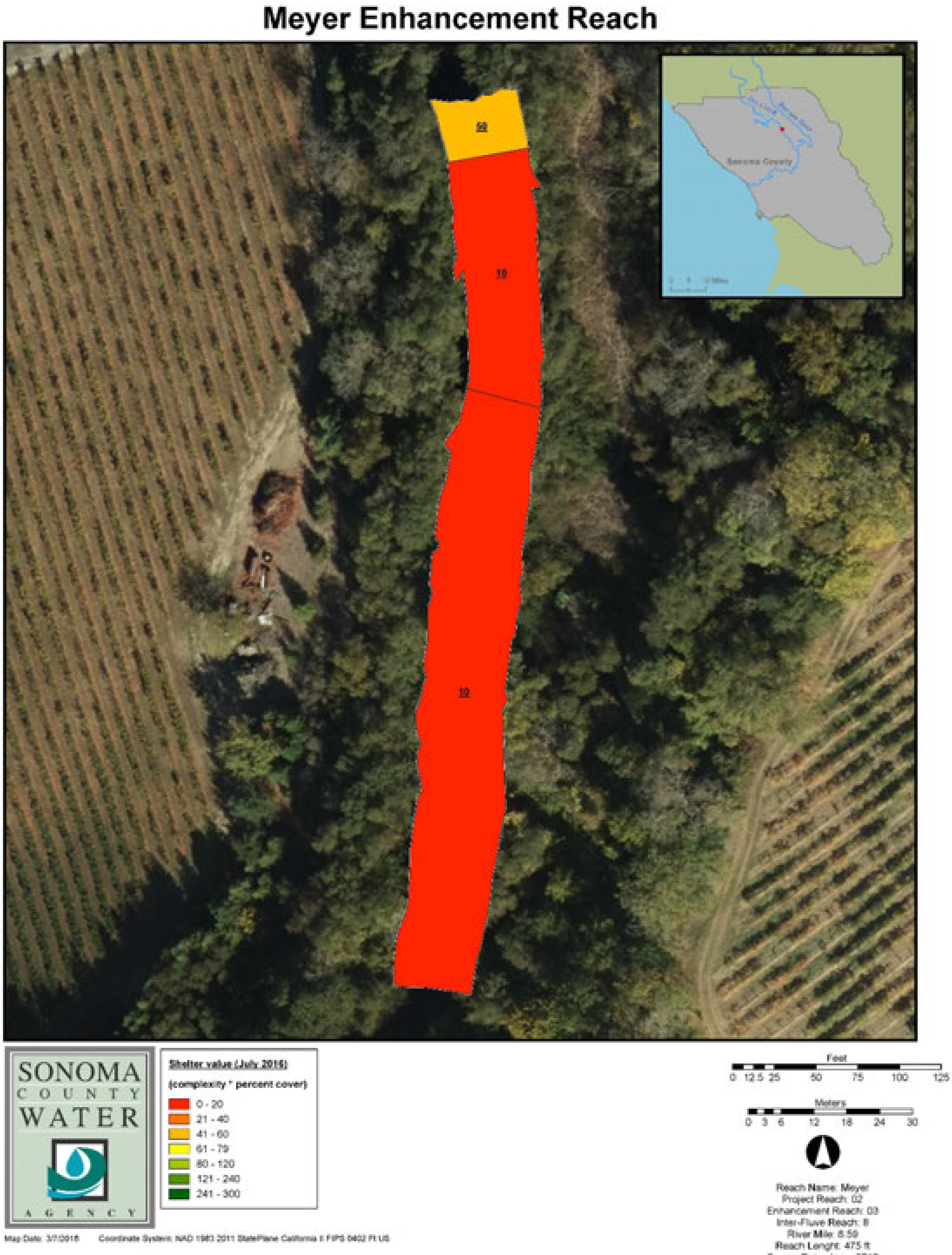


Figure 5.2.36. Habitat unit shelter values for the Meyer enhancement reach in June 2016.

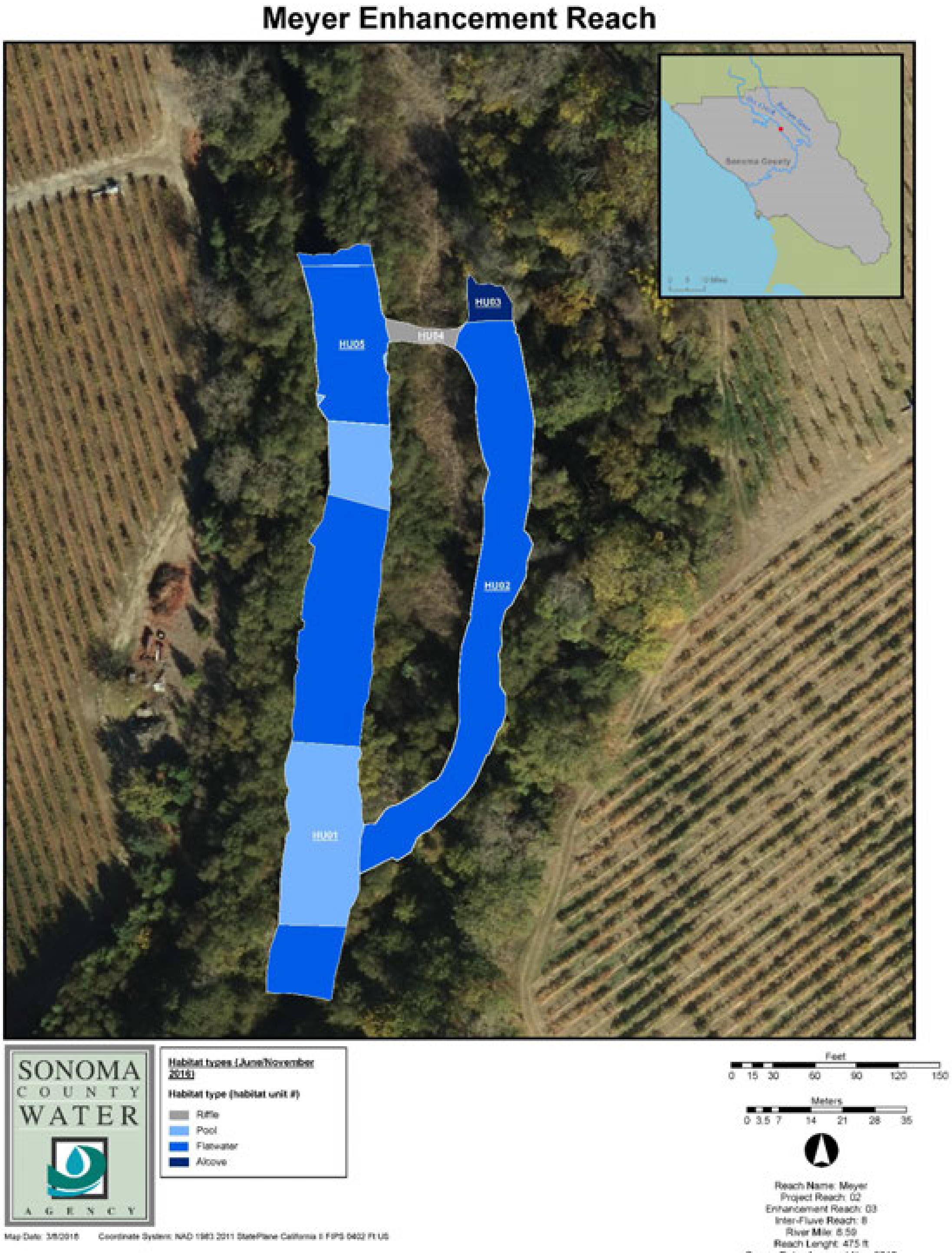


Figure 5.2.37. Habitat unit number and type within the Meyer enhancement reach in June and November 2016.

Meyer Enhancement Reach

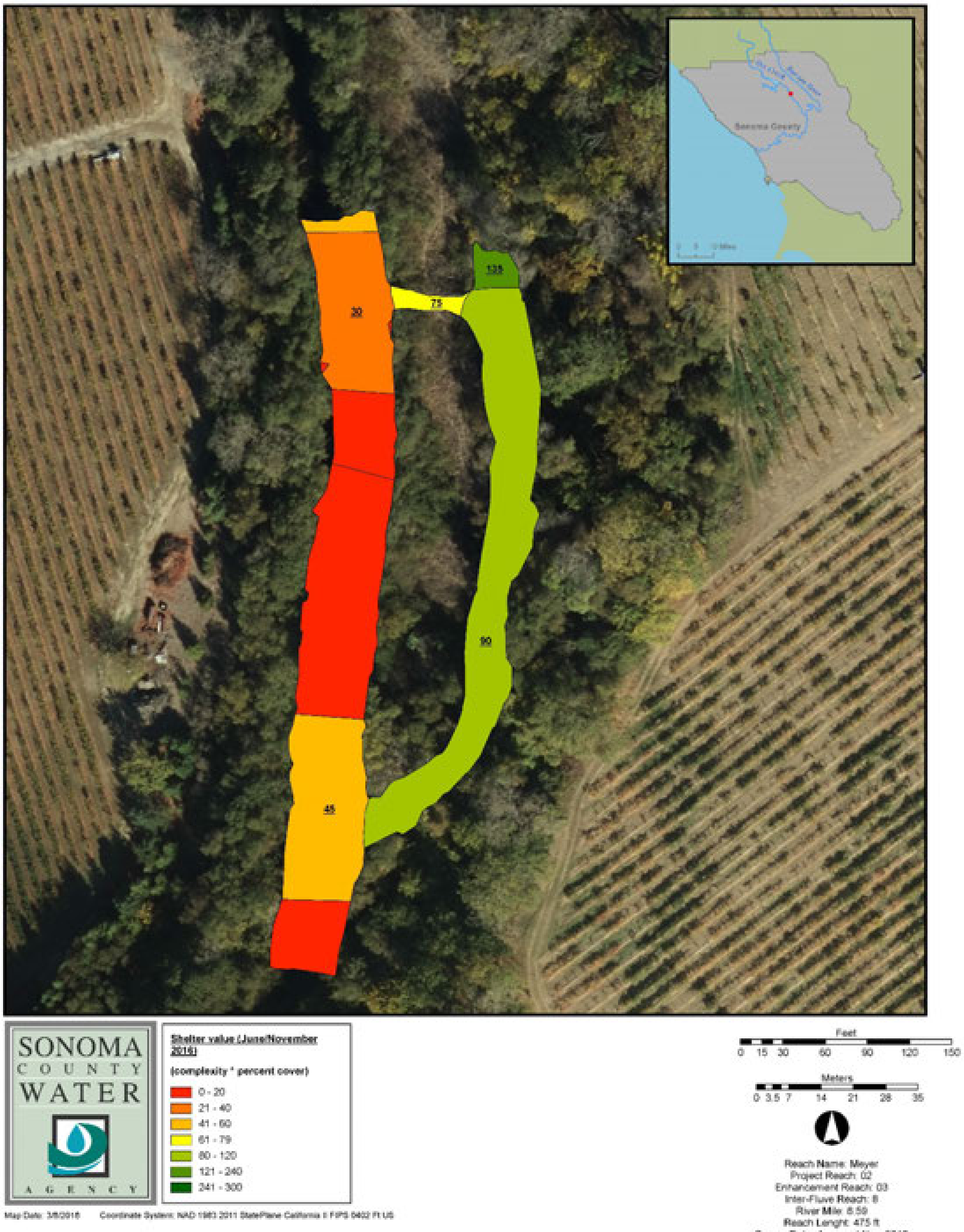


Figure 5.2.38. Habitat unit shelter values for the Meyer enhancement reach in June and November 2016.

Table 5.2. 5. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Truett Hurst enhancement reach in in June (pre-enhancement) and November (with enhanced area) 2016.

Pre-enhancement (June 2016)				
Habitat Unit #	**Habitat Type**	**Shelter Score**	**Percent Cover**	**Shelter Value**
HU03	Flatwater	2	25	50
HU04	Pool	3	35	105
HU05	Pool	2	20	40
HU06	Riffle	2	10	20
HU07	Flatwater	2	15	30
Enhanced area (November 2016)				
Habitat Unit #	**Habitat Type**	**Shelter Score**	**Percent Cover**	**Shelter Value**
HU01	Flat	3	20	60
HU02	Riffle	1	5	5
HU03	Pool	3	35	105
HU04	Alcove	3	50	150
HU05	Riffle	1	5	5
HU06	Pool	3	25	75
HU07	Alcove	3	45	135
HU08	Riffle	2	5	10
HU09	Pool	3	45	135
HU10	Riffle	3	20	60
HU01	Flat	3	20	60

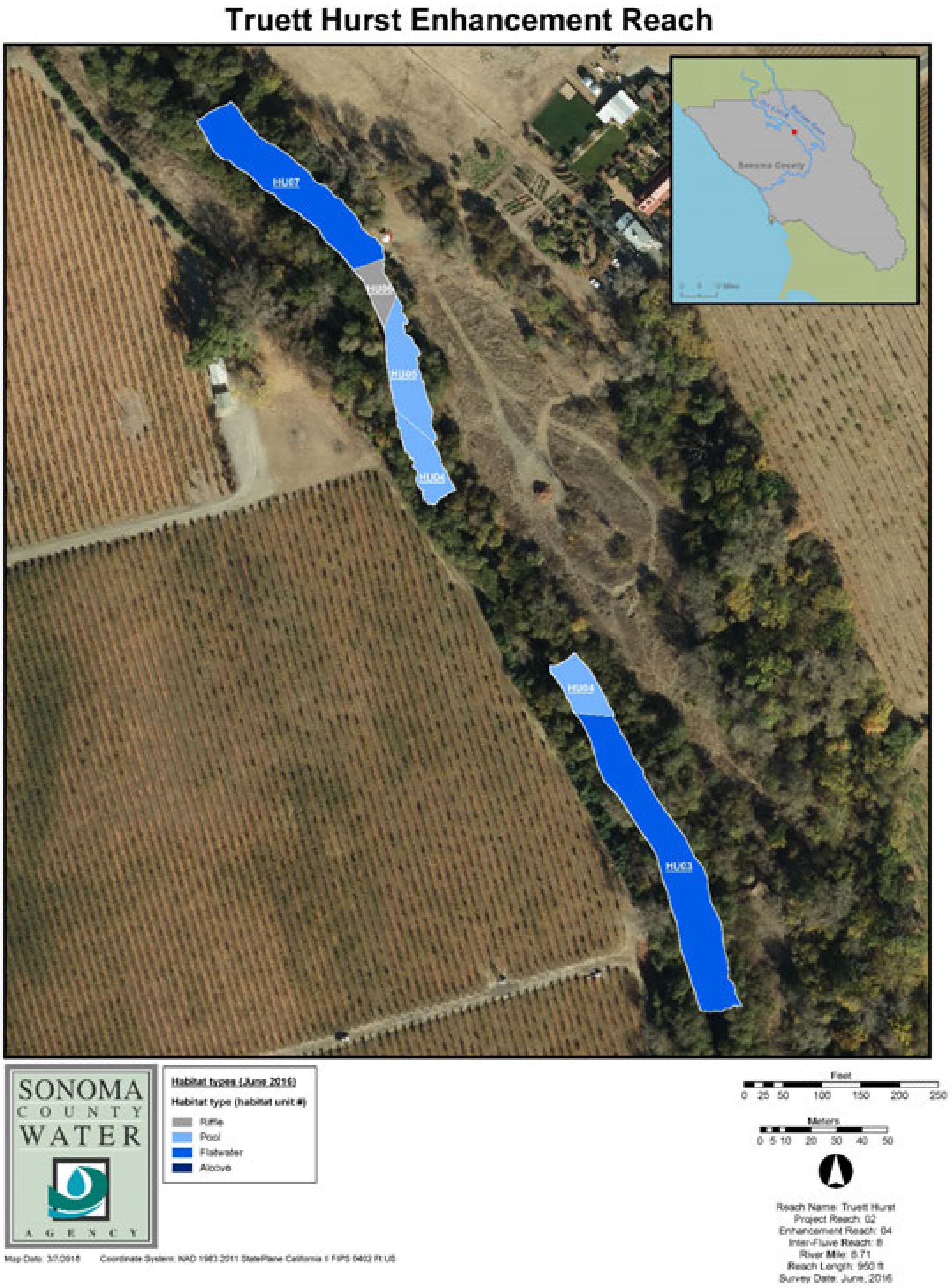


Figure 5.2.39. Habitat unit number and type within the Truett Hurst enhancement reach in June 2016.

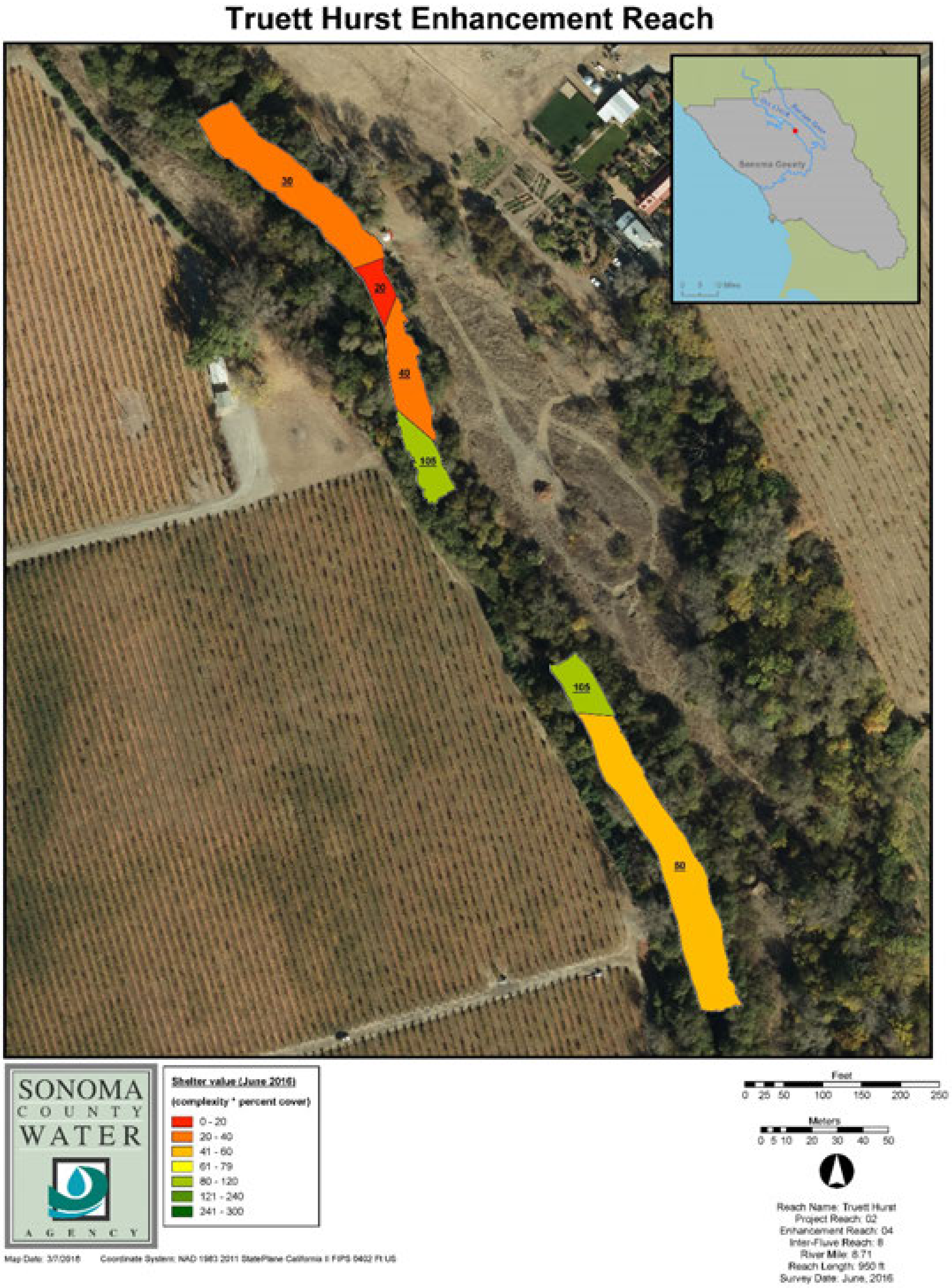


Figure 5.2.40. Habitat unit shelter values for the Truett Hurst enhancement reach in June 2016.

Figure 5.2.41. Habitat unit number and type within the Truett Hurst enhancement reach in June and November 2016.

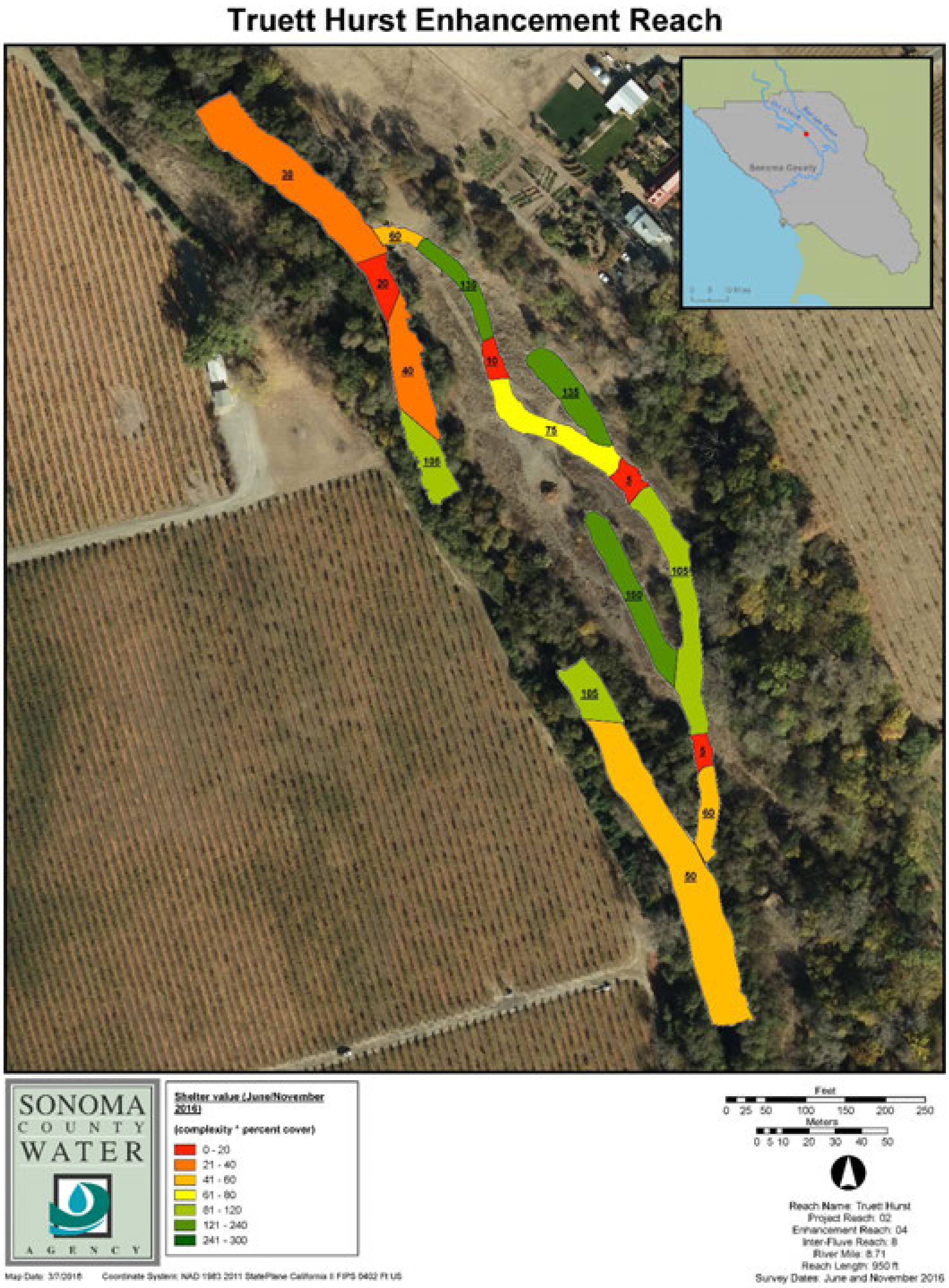


Figure 5.2.42. Habitat unit shelter values for the Truett Hurst enhancement reach in June and November 2016.

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National Marine Fisheries Service (NMFS). 2008. Endangered Species Act Section 7 Consultation: Water Supply, Flood Control Operations, and Channel Maintenance conducted by the U.S. Army Corps of Engineers, the Sonoma County Water Agency, and the Mendocino County Russian River Flood Control and Water Conservation Improvement District in the Russian River Watershed. Issued September 24, 2008.

Porter, M. D., D. M. Marmorek, D. Pickard, and K. Wieckowski. 2014. Dry Creek Adaptive Management Plan (AMP), Version 0.93. Final document prepared by ESSA Technologies Ltd., Vancouver, BC for Sonoma County Water Agency, Santa Rosa CA. 33 pp. + appendices.

5.3 Validation Monitoring

Part of the Adaptive Management Plan (AMP) for validating the effectiveness of habitat enhancement in mainstem Dry Creek calls for a multiscale monitoring approach in both space and time (Porter et al. 2013). The current section of this report focuses on the results of validation monitoring for juvenile and smolt salmonid populations in mainstem Dry Creek in 2016. These data are part of an ongoing pre-construction (baseline) monitoring effort begun in 2008 and outlined in the Reasonable and Prudent Alternative section of NMFS' Russian River Biological Opinion. Construction of the first mile of habitat enhancements in mainstem Dry Creek (the "demonstration project") was completed in 2014 allowing us to resume sampling efforts in the stream sections affected by construction in prior years. Construction on the second and third mile of habitat enhancements began in 2016 in the middle and lower sections of Dry Creek, limiting our ability to sample areas that had been monitored in previous years. Validation monitoring data collected in newly-constructed habitats are reported as well as continued efforts to monitor trends in juvenile and smolt abundance at the reach and watershed scale.

In the Russian River Biological Opinion status and data report year 2009-10 (Manning and Martini-Lamb 2011), the Water Agency outlined six possible metrics that could be considered for validation monitoring of juvenile salmonids with respect to eventual habitat enhancements in the mainstem Dry Creek: habitat use, abundance (density), size, survival, growth and fidelity (Table 5.3.1). In 2009-2010, a major focus of validation monitoring in Dry Creek was on evaluating the feasibility of sampling methods to accurately estimate each of those metrics while simultaneously attempting to understand how limitations in sampling approaches may affect our ability to validate project success. These same validation metrics and associated limitations and uncertainties have been discussed in the context of the results of those evaluations and are incorporated into the Dry Creek AMP (Porter et al. 2013). The methods currently employed for validation monitoring in Dry Creek are largely based on the outcome of that work (Manning and Martini-Lamb 2011; Martini-Lamb and Manning 2011).

Table 5.3.1. Proposed target life stages, validation metrics, spatiotemporal scale and monitoring tools for validation monitoring in mainstem Dry Creek.

Spatial scale	Target life stage	Target metric(s)	Temporal scale	Primary monitoring tools
Site/feature	Juvenile (non-smolt)	Habitat use, abundance (density), size, growth	Post-construction	Snorkeling, electrofishing, PIT tags and antennas
Reach	Juvenile (non-smolt)	Abundance (density), size, survival, growth, fidelity	Pre-construction (baseline) vs. post-construction	Electrofishing, PIT tags and antennas
Mainstem Dry Creek	Smolt	Abundance	Ongoing to capture long-term trend	Downstream migrant trap, PIT antennas

Methods

In order to address use of newly created habitat by juvenile salmonids at the site (feature) scale, sampling consisted of PIT-tagging in the summer, operation of stationary PIT antennas in the winter and snorkeling in summer and fall. We also conducted mark-recapture electrofishing in enhancement areas to estimate juvenile population density where possible. To better isolate how data collected at the site-scale indicate the effect of habitat enhancement, we also conducted backpack electrofishing in stream sections (reach-scale) that were not enhanced. Finally, we continued to operate a downstream migrant trap seasonally in lower Dry Creek to assess trends in smolt production over time. Broad-scale efforts that are part of the Coastal Monitoring Program (CMP) now being implemented in the Russian River provide a framework for placing our results in the context of watershed-scale patterns in those population metrics identified in Fish Bulletin 180 (the guiding document for California Coastal Salmonid Monitoring Program implementation, Adams et al. 2011).

Habitat utilization

In order for juvenile coho to take advantage of the habitat enhancements created in mainstem Dry Creek, fish will need to come from somewhere and although there is a substantial population of juvenile steelhead that rear in mainstem Dry Creek, coho are extremely scarce. Therefore, our strategy for juvenile coho validation monitoring must rely on hatchery releases coupled with visual observations of coho in the backwaters during snorkel surveys and observations on PIT antennas within habitat enhancement sites.

Summer / Fall

We conducted three snorkel surveys in Dry Creek habitat enhancement sites from August to November, 2016. Surveys were conducted with two snorkelers working in tandem. During site visits we measured water temperature and dissolved oxygen at 0.25 m depth increments throughout the water column allowing us to construct vertical temperature and dissolved oxygen profiles. Three enhancement sites (constructed riffles) were sampled in early September on a single pass electrofishing survey to further evaluate habitat utilization.

Winter

Similar to 2013, 2014 and 2015, we operated PIT antennas in newly constructed habitat enhancement sites during the winter at the downstream and upstream openings of the Geyser Peak (rkm 2.70, rkm 2.81), Meyer (rkm 13.81, rkm 13.94) and Truett Hurst side channels (rkm 14.05, rkm 14.30). Although antennas did not span the width of the channel openings, they did cover the majority of the wetted width (Figure 5.3.1). The source of PIT-tagged fish included: (1) PIT-tagged age-0+ coho from Warm Springs hatchery that were released directly into the three side channels in the fall; (2) PIT-tagged age-0+ coho from Warm Springs hatchery that were released in mainstem Dry Creek just upstream of the enhancement sites; (3) wild (natural-origin) juvenile steelhead that were PIT-tagged during mainstem Dry Creek electrofishing surveys.

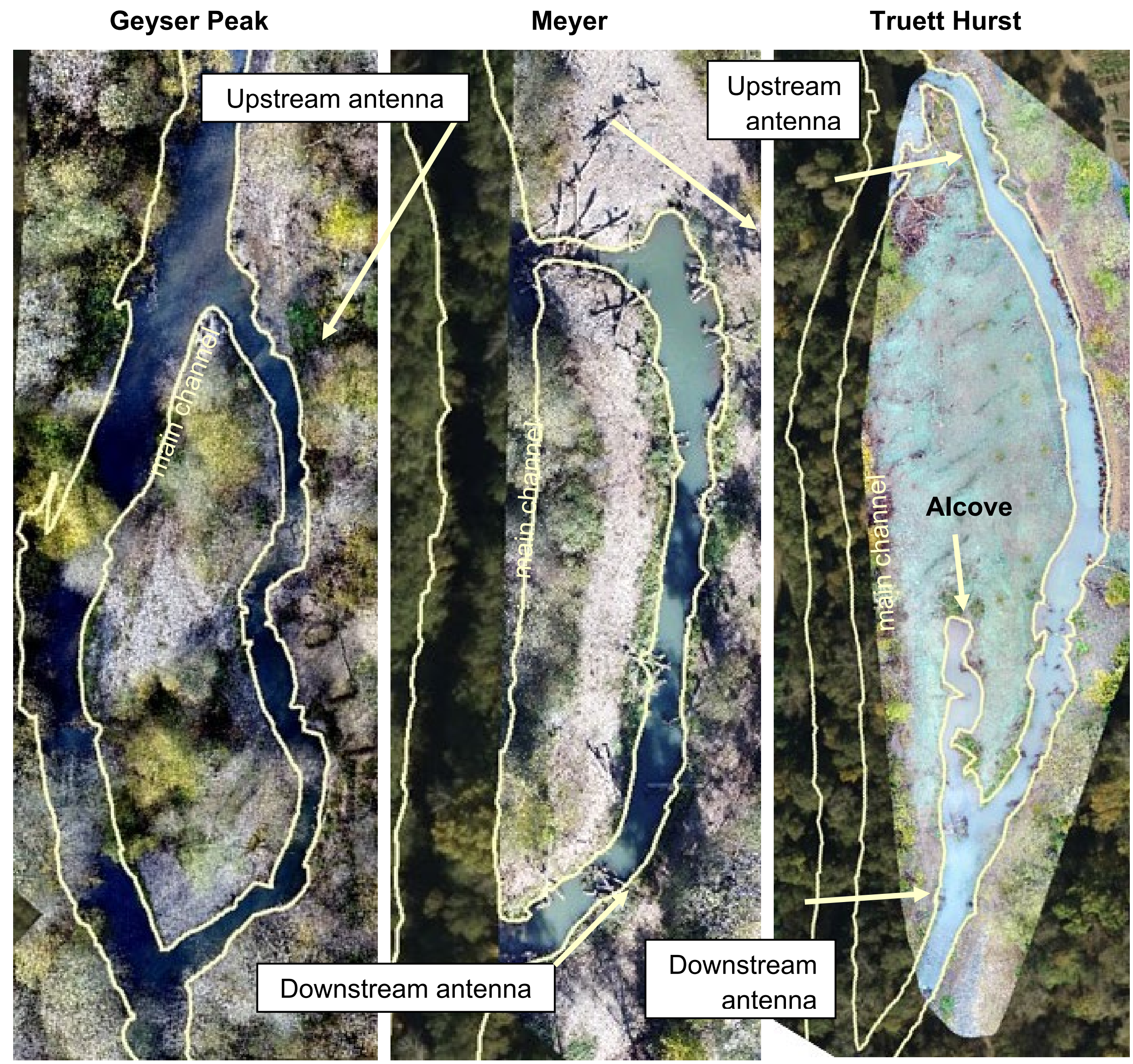


Figure 5.3.1. Location of PIT antennas in Dry Creek side channels completed in fall, 2016.

A total of 746 juvenile coho were released into the three side channels and another 502 were released within 2.5 to 3.0 km upstream of the enhancement sites in mainstem Dry Creek (Table 5.3.2). The residence time of PIT-tagged juvenile coho released into the backwater was calculated as the number of days between release date and their final detection date on the PIT antenna. We also detected some of these fish downstream of the backwaters on stationary PIT antennas at the mouth of Dry Creek as well as PIT antenna locations in other Russian River tributaries.

The experience with juvenile coho releases into tributaries of the Russian River has been for fish to flee the release area for a period lasting until fish settle in which usually occurs within 10 days (CA Sea Grant, unpublished data). To overcome this flight response, CA Sea Grant monitoring staff install block nets to act as physical barriers to this initial fish movement where possible. Unfortunately, it is extremely difficult or impossible to use block nets in this same way in the deep, wide backwater sites constructed on Dry Creek prior to 2016. However, of the three newly-completed side channels where we released fish in 2016, we were able to install a block net to contain 50% of the releases in an alcove inside the Truett Hurst side channel (Figure 5.3.1).

Table 5.3.2. Number of age-0+ coho released from Warm Springs Hatchery in 2016 (a) in the Truett Hurst, Meyer and Geyser Peak side channels completed in 2016 and (b) within ~3 km upstream of those side channels.

Mainstem or Off-channel	Release Site	Release rkm	Number of Fish
Off-channel	Truett Hurst side channel	14.16	253
	Meyer side channel	13.90	245
	Geyser Peak side channel	2.76	248
Mainstem	Yoakim Bridge	17.16	248
	Near downstream check dam	5.23	272
		Total	**1,266**

Late summer population density

Site-scale sampling

Due to the construction schedule, the Geyser Peak, Meyer and Truett Hurst side channels were not completed until after the 2016 electrofishing season and, as stated in previous reports, depths in the backwater ponds constructed in the demonstration reach preclude backpack electrofishing. Therefore, we were only able to conduct sampling to estimate population density in the U.S. Army Corps (USACE) constructed side channel (river km ~21.4). During late September through early October, we sampled with a backpack electrofisher by making a single pass through the entire side channel on day 1 (the marking event) followed by a second pass two days later (the recapture event). Individuals captured on day 1 were PIT-tagged, released near their capture location and subject to recapture on day 2. From these paired sampling events, we used the Petersen mark-recapture model in Program MARK (White and Burnham 1999) to estimate end-of-summer abundance ($\widehat{N}$). Provided recapture probability, mortality and the proportion of fish leaving the section between the marking and recapture events was the same for the marked group as it was for the unmarked group, the abundance estimates from the paired mark and recapture events in early autumn will be unbiased (White et al. 1982). Density estimates were calculated as the quotient of $\widehat{N}$ and wetted area of the site.

Reach-scale sampling

The Biological Opinion as well as the primary literature (e.g., Roni 2005) acknowledge the problem of biological monitoring that is too limited in time and space to accurately detect changes in population that may result from artificial habitat enhancements as opposed to larger scale factors. To overcome this we sought to place our results in a broader context. In 2015 and again in 2016, we added to our targeted site-scale sampling by employing a reach-based approach that relied on the spatially-balanced random sampling framework afforded by the generalized random tessellation stratified (GRTS) framework outlined for the CMP (Adams et al.

2011). Sampling reaches in this manner over time will allow us to place our results in a broader spatial context thereby facilitating more accurate validation of the effectiveness of habitat enhancement measures in Dry Creek (Figure 5.3.2). Towards that end, we sampled one randomly selected stream section in each of nine "GRTS" reaches defined in mainstem Dry Creek for CMP monitoring. We sampled using methods similar to those described for the paired sample, site-scale electrofishing so that we could estimate juvenile steelhead abundance using the Petersen mark-recapture model. Stream sections (sub-reaches) were typically longer (435 to 1350 feet) than sites sampled during site-scale sampling.

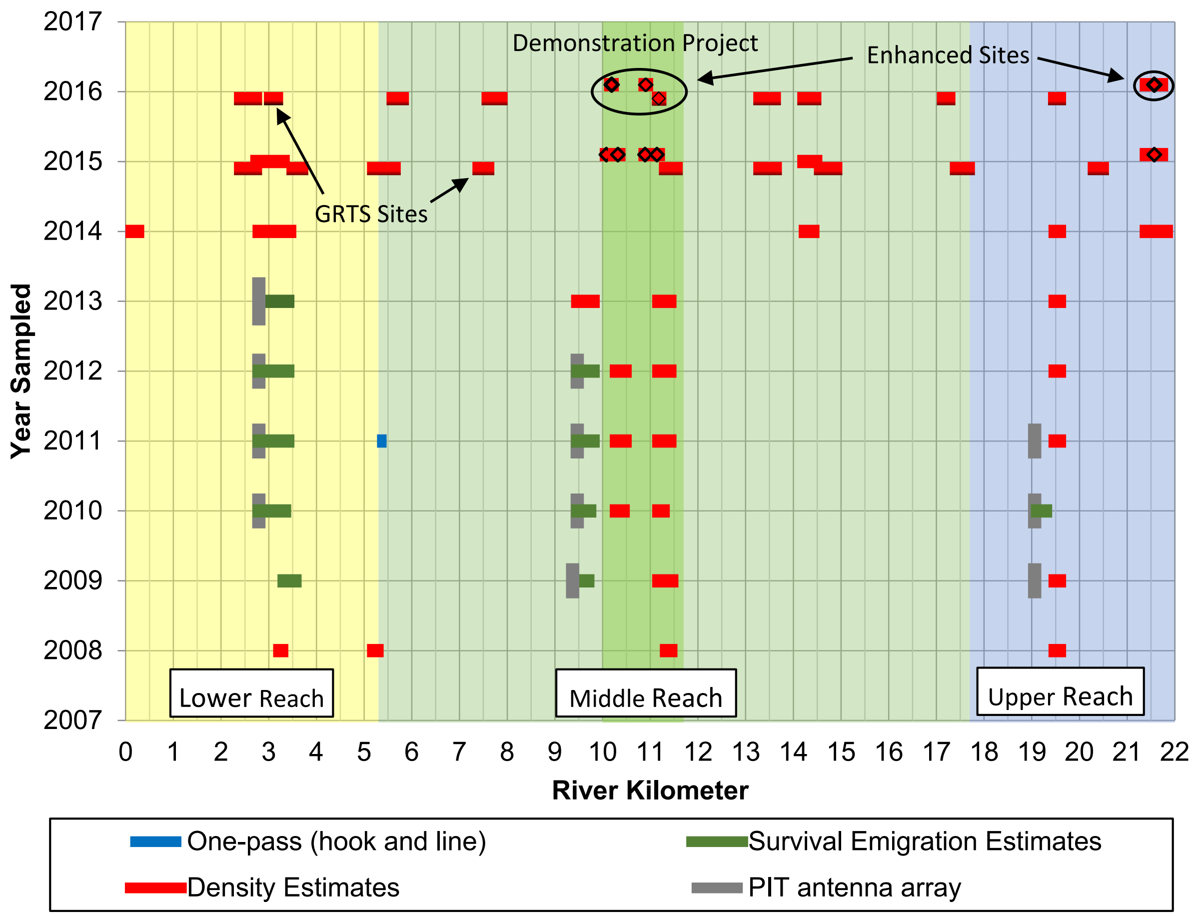


Figure 5.3.2. Years sampled and river kilometer (from the mouth) where juvenile steelhead populations were sampled in mainstem Dry Creek, 2008-2016. Line length for each site is scaled to the length of stream sampled. Data collected at the site scale were analyzed using mark-recapture (either a multiple-pass depletion or Petersen model) and reach-scale data collected in 2009 were analyzed with the core-sampling approach (see Manning and Martini-Lamb 2011 for details) while reach scale data collected in 2011-13 were analyzed with the multistate model using program MARK (White and Burnham 1999) to estimate survival and emigration. The darker green-shaded area indicates the stream section that has been targeted to receive the first mile of habitat enhancements (the "demonstration project"). We adopted the geomorphically-based reach designations identified by Inter-Fluve (2011) for defining reaches for use in summarizing density estimates.

Smolt abundance

A rotary screw trap with a 1.5 m diameter cone was anchored to the Westside Road bridge, located 3.3 km upstream from the confluence of Dry Creek and the Russian River. Wood-frame mesh panels were installed adjacent to the rotary screw trap in order to divert downstream migrating salmonids into the trap that may have otherwise avoided the trap.

Fish handling methods and protocols were similar to those used in previous years (see Manning and Martini-Lamb 2011). Fish captured in the trap were identified to species and enumerated. A sample of each species was anesthetized and measured for fork length each day, and a sample of salmonid species was weighed each week. With the exception of up to 50 Chinook salmon smolts each day, all fish were released downstream of the first riffle located downstream of the trap.

Each day, up to 50 Chinook smolts (≥60 mm) were marked and released upstream of the trap for the purpose of estimating trap efficiency and constructing a population estimate. Both fin clips and PIT tags were used to mark fish. Fin-clipped and PIT-tagged fish that were recaptured in the trap were noted and released downstream (the lengths and weights of recaptured fish were not recorded a second time). The population estimate of Chinook salmon smolts produced in the Dry Creek watershed upstream of the trap were based on recapture rates of PIT-tagged fish only. The abundance estimate of Chinook smolts reported in 2016 applies only to the period of trap operation (April 13-July 31). PIT-tagged fish provided the potential to evaluate migration mortality and migration time as fish were detected at downstream monitoring sites on mainstem Russian River.

Results

Habitat utilization

Summer / Fall

Counts from snorkel surveys of juvenile salmonids in the Dry Creek habitat enhancement areas were low with a total of only 49 juvenile steelhead observed in the six off-channel sites surveyed (Table 5.3.3). However, as in previous years (Manning and Martini-Lamb 2018), rooted aquatic vegetation, algae growth and high turbidity which resulted in poor visibility adversely affected our ability to observe juvenile salmonids in backwaters. Evidence of vegetation impacts on snorkeling visibility is clear in light of snorkel and electrofishing surveys conducted in 2014 in the USACE side channel. The number of fish observed on October 2014 during a snorkel survey (34) was far less than the 351 steelhead captured by electrofishing in the same site a few days earlier (Martini-Lamb and Manning 2016). In addition to juvenile salmonids, an adult Chinook was observed in the Meyer side channel during a dive survey on November 15, 2016 and an adult coho was observed constructing a redd on a riffle inside the Truett Hurst side channel on December 28, 2016 (Figure 5.3.3).

Table 5.3.3. Number of fish observed during snorkel surveys in Dry Creek habitat enhancement sites, 2016.

Date	Site	Visibility (ft)	Species	Life stage	Number of fish
8/25/2016	Farrow backwater	20	steelhead	Juvenile	0
	Wallace backwater	4	steelhead	Juvenile	20
	Van Alyea backwater	4	steelhead	Juvenile	0
10/6/2016	Farrow backwater	15	steelhead	Juvenile	0
	Wallace backwater	4	steelhead	Juvenile	14
	Van Alyea backwater	4	steelhead	Juvenile	0
11/15/2016	Geyser Peak side channel	4	steelhead	Juvenile	0
	Meyer side channel	4	Chinook	Adult	1
	Meyer Side channel	4	steelhead	Juvenile	0
	Truett Hurst side channel	4	steelhead	Juvenile	15

Figure 5.3.3. Adult Chinook holding in the Meyer side channel (left photo) and adult coho spawning in the Truett Hurst side channel (right photo).

In addition to the effect of low visibility on detectability of juvenile salmonids, we suspect that low dissolved oxygen levels also impacted use of backwaters by juvenile salmonids. Mean daily dissolved oxygen and vertical water quality profiles showed deteriorating conditions both seasonally and throughout the water column (Figure 5.3.4). However, dissolved oxygen was favorable in the other habitat enhancement sites (data not shown).

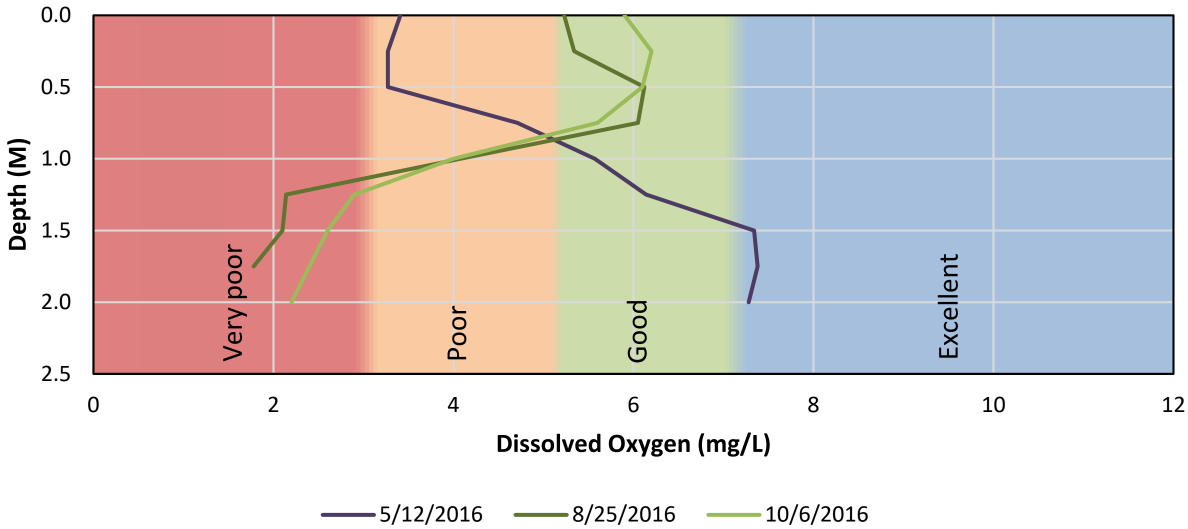


Figure 5.3.4. Dissolved oxygen from vertical water quality profiles collected with a handheld probe at 0.25 m depth increments in the Farrow backwater.

Summer / Fall

Juvenile steelhead were found at all three constructed riffles in the Demonstration Reach at rkm 10.17 (Farrow property), 10.89 and 11.14 (Van Alyea property) during single pass electrofishing surveys at each site. Only a few steelhead parr were observed at each riffle, ranging from 3 to 15 fish per site; however, our ability to sample these areas was hampered by the poor footing afforded by swift water velocities and slippery round-shaped boulders.

Winter

The only juvenile coho detected on PIT antennas in the three newly-constructed side channels during winter 2016-2017 were those that were released in or near those side channels. Of the 1,266 released (mainstem and side channels combined), 552 individuals (44%) were detected on PIT antennas located in the habitat enhancement sites (Table 5.3.4). Seventy-four (15%) of the 520 fish released in mainstem Dry Creek within 3 km upstream of the side channels were detected on one or more side channel antennas. Site-specific antenna efficiencies likely declined in early January following an extended period of high flows in Dry Creek caused by winter storms beginning approximately 30 days after fish were released. Based on side channel detections, residence time (days between release and final detection) was at least 14 days for 114 fish and at least 30 days for 65 fish. In the Meyer side channel where antennas likely weathered the high flows better than either Geyser Peak or Truett Hurst sites, 26 fish remained until at least March with the latest detection occurring on April 8. Because of imperfect antenna detection efficiency, these numbers should be considered minimums. Nevertheless, these data indicate that the three newly-constructed side channels were used by juvenile coho in December 2016 and beyond.

In addition to detections in the Dry Creek side channels, 71 individuals were detected on PIT antennas at locations downstream of Dry Creek. Those sites include 64 in Mill Creek (rkm 2.01), 2 in Porter Creek (rkm 0.20) and 5 in Willow Creek (rkm 0.41). Of the 71 individuals detected at these sites, almost half (31) were never detected on any of the Dry Creek side channel antennas.

Table 5.3.4. Number of PIT-tagged juvenile coho released and the subset of these fish that were detected on PIT antennas located in mainstem Dry Creek or in the three Dry Creek side channels completed in 2016.

Release Site	Release date	Release rkm	Number released	Number detected* Truett Hurst	Meyer	Geyser Peak
Mainstem Dry Creek at Yoakim Bridge	11/29/16	17.16	248	27	17	2
Truett Hurst side channel			128			
Side channel	11/29/16	14.05	125	50	19	0
Alcove^	12/8/16	14.05	245	59	23	0
Meyer side channel	11/29/16	13.81	272	105	160	3
Mainstem Dry Creek at lowest check dam	11/29/16	5.23	248	11	11	21
Geyser Peak side channel	11/29/16	2.70	248	3	4	173

*Individuals that were detected in more than one location are counted more than once.

^Fish were contained in the Truett Hurst alcove with a block net for 9 days before the net was removed on 12/8/2016.

Of the 1,473 steelhead parr that were PIT-tagged during backpack electrofishing surveys in mainstem Dry Creek, a total of 10% (151 individuals) were detected on PIT antennas in the three side channels that were completed in fall, 2016 (Table 5.3.5). Fish that were originally tagged in close proximity to these side channels (within 2.5 km) used them at a higher rate (16%) as compared to those captured and tagged further away (4.5%). Thirty-four PIT-tagged individuals moved upstream from as far away as 9 km from the site where they were originally tagged in mainstem Dry Creek into the Truett Hurst side channel. Even though a very small proportion of the juvenile steelhead population in Dry Creek were PIT-tagged, it is reasonable to conclude that a significant portion of all juveniles are making use of these enhanced off-channel habitats during the winter. In addition to juvenile salmonids, we detected 17 PIT-tagged adult coho and 14 PIT-tagged adult Chinook on side channel PIT antennas.

Table 5.3.5. Number of juvenile steelhead PIT-tagged during mainstem Dry Creek electrofishing surveys and subsequent number detected (and percent of total detected) on PIT antennas in habitat enhancement side channels, winter 2016. Reaches in bold italic text indicate reaches where constructed side channels with PIT antenna monitoring were located.

Reach						
Name	Upper (rkm)	Lower (rkm)	EF tagged (Fall 2016)	Truett Hurst (rkm=14.16)	Meyer (rkm=13.90)	Geyser Peak (rkm=2.76)
DRY 9	21.81	18.90	78	0 (0%)	0 (0%)	0 (0%)
DRY 8	18.90	17.07	232	3 (1.3%)	6 (2.6%)	0 (0.0%)
DRY 7	***17.07***	***13.93***	***206***	***25 (12.1%)***	***15 (7.3%)***	***0 (0.0%)***
DRY 6	***13.93***	***11.62***	***254***	***35 (13.8%)***	***42 (16.5%)***	***3 (1.2%)***
DRY 5	11.62	9.99	46	2 (4.3%)	3 (6.5%)	0 (0.0%)
DRY 4	9.99	6.87	164	9 (5.5%)	4 (2.4%)	2 (1.2%)
DRY 3	6.87	5.20	222	8 (3.6%)	8 (3.6%)	1 (0.5%)
DRY 2	5.20	2.82	172	0 (0.0%)	0 (0.0%)	20 (11.6%)
DRY 1	***2.82***	***0.00***	***99***	***0 (0.0%)***	***0 (0.0%)***	***15 (15.2%)***
		Totals	**1,473**	**82 (5.6%)**	**78 (5.3%)**	**41 (2.8%)**

Late summer population density

Site-scale sampling

The estimated density of juvenile steelhead in the USACE constructed side channel was 0.02 fish/m^2 (Figure 5.3.5). We captured a total of two wild coho YOY during electrofishing sampling of this enhancement site.

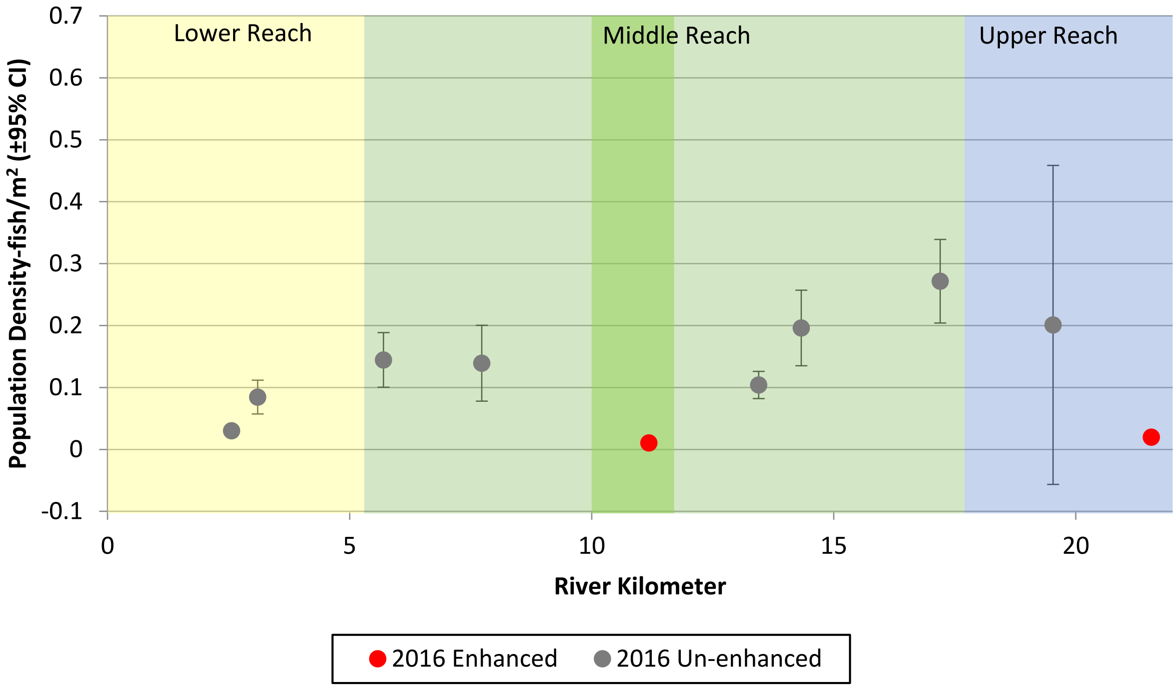


Figure 5.3.5. Estimated density of juvenile steelhead in mainstem Dry Creek, in habitat-enhanced habitat (site-scale monitoring) and un-enhanced habitat (reach-scale monitoring). Estimates are based on the Petersen mark-recapture model. Note there was an overlap with the GRTS reaches selected for reach-scale sampling and sections within the demonstration reach where habitat enhancements have been completed. Therefore one of the nine sites sampled for reach-scale sampling is identified as an enhanced site.

Reach-scale sampling

The average density of juvenile steelhead in GRTS sub-reaches was 0.13 fish/m^2 (range 0.01 fish/m^2 to 0.27 fish/m^2, Figure 5.3.5). When averaged for all sites within a year, densities in 2016 were 0.19 fish/m^2 higher than the eight year average from 2008-2015 (Figure 5.3.6). Unlike the previous year, the average population density for enhanced sites was lower than for un-enhanced sites (Figure 5.3.6).

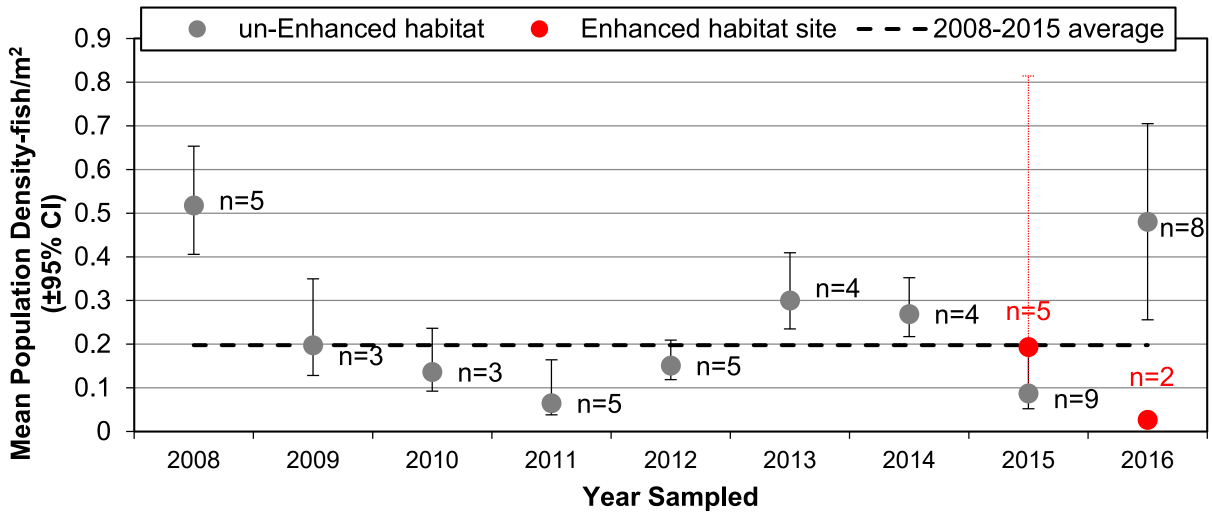


Figure 5.3.6. Mean juvenile steelhead density among all sites sampled within a year in mainstem Dry Creek, 2008-2016. "n" refers to the number of sites sampled per year.

Smolt abundance

We installed the rotary screw trap on April 14 (Figure 5.3.7). Except for brief periods when trapping was suspended because of high debris loading in the trap from high winds, the trap was checked daily during operation until it was removed on July 31. The peak capture of Chinook smolts (2,505) occurred during the week of 5/14 (Figure 5.3.8). Based on the estimated average weekly capture efficiency (range: 5% to 31%), the resulting population size of Chinook salmon smolts passing the Dry Creek trap between April 14 and July 31 was 64,384 (±95% CI: 8,578, Figure 5.3.9). This is the second smallest population estimate since we began trapping Dry Creek in 2009

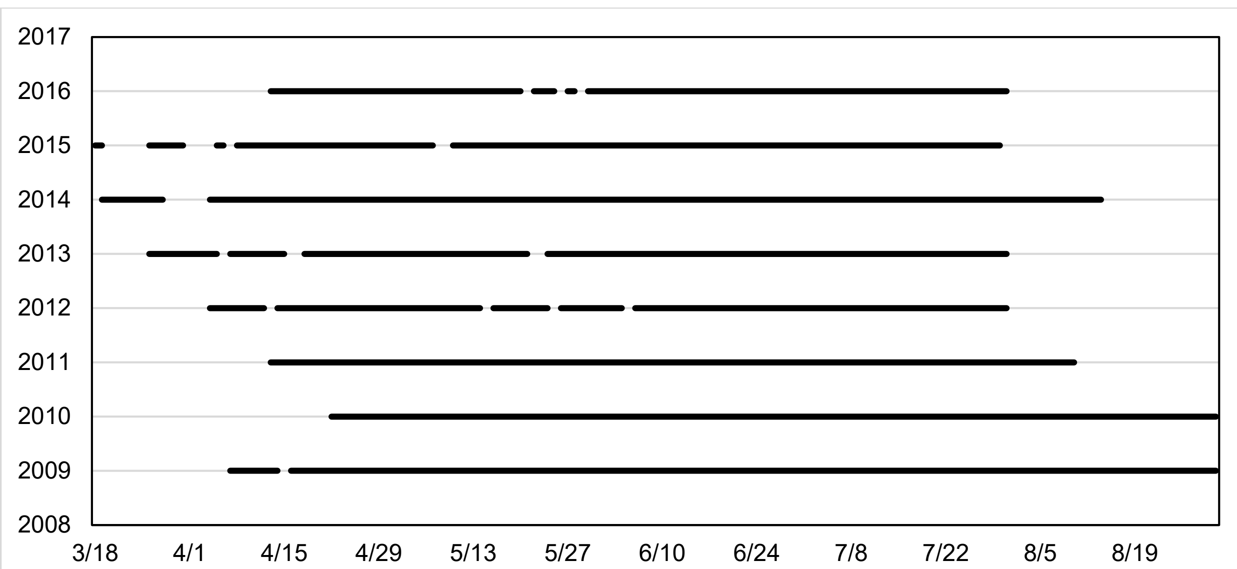


Figure 5.3.7. Begin and end dates and data gaps (spaces in lines) for operation of the Dry Creek downstream migrant trap, 2009-2016.

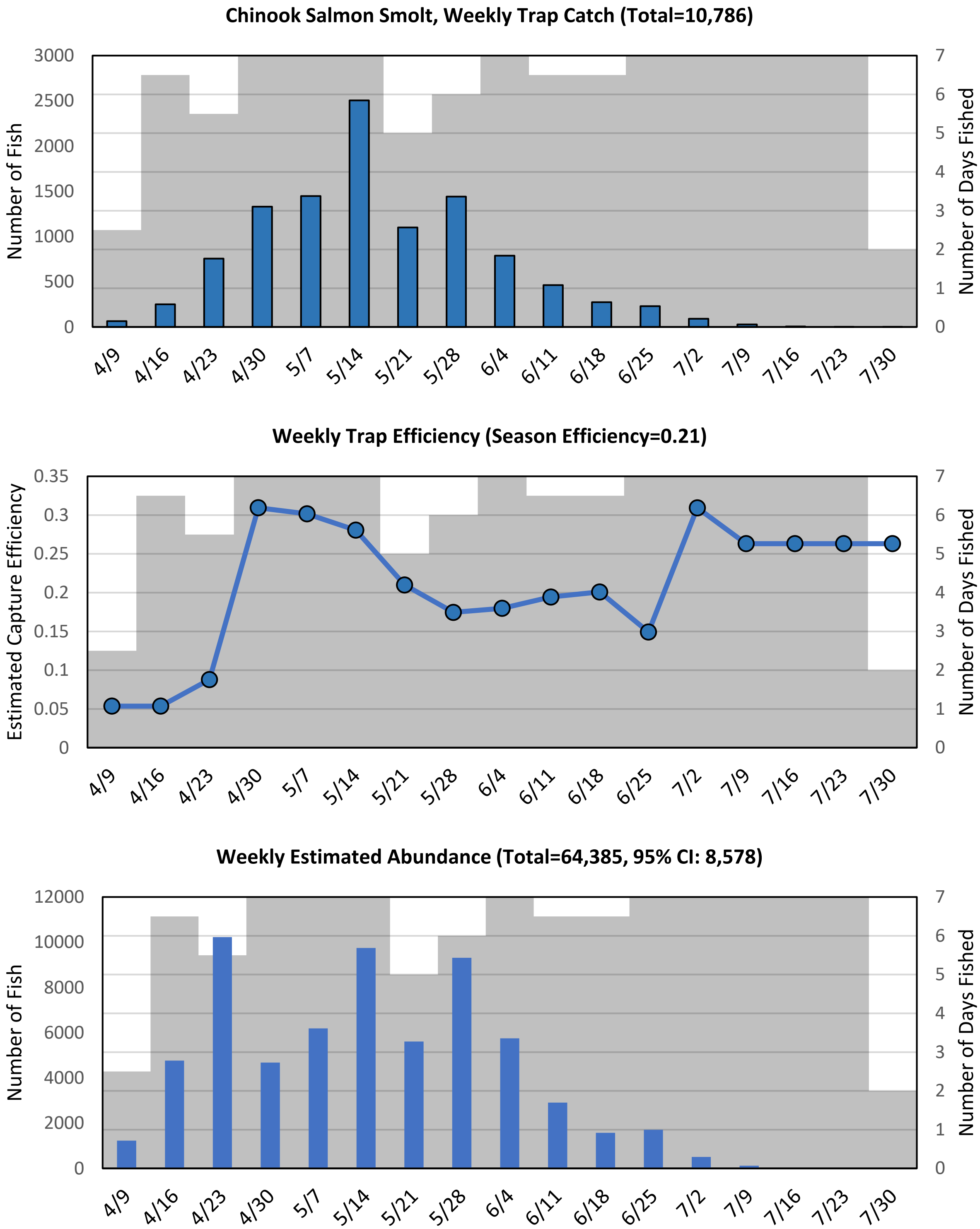
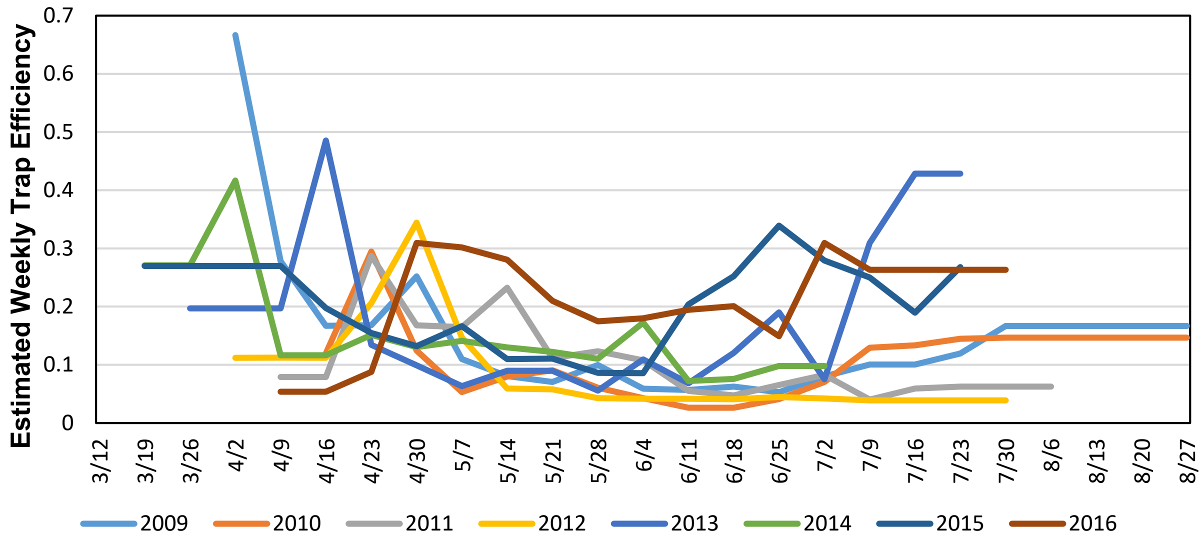


Figure 5.3.8. Weekly trap catch (upper panel), estimated average weekly capture efficiency (middle panel) and population estimate of Chinook salmon smolts in the Dry Creek rotary screw trap (lower panel), 2016. Estimates are from DARR (Bjorkstedt 2005). The number of days each week the trap was fished is represented by the shaded area.



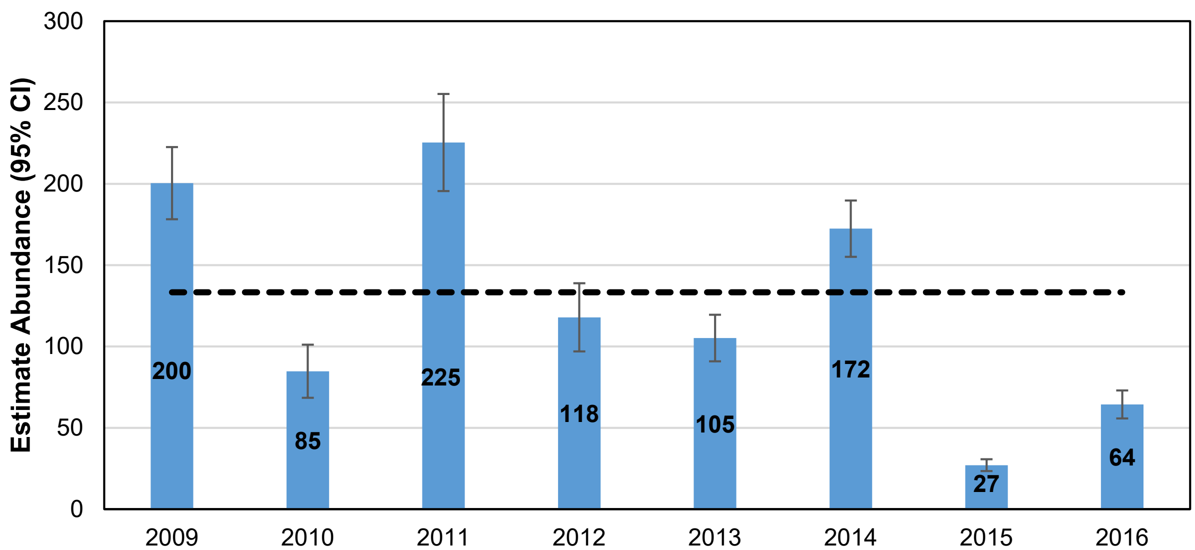
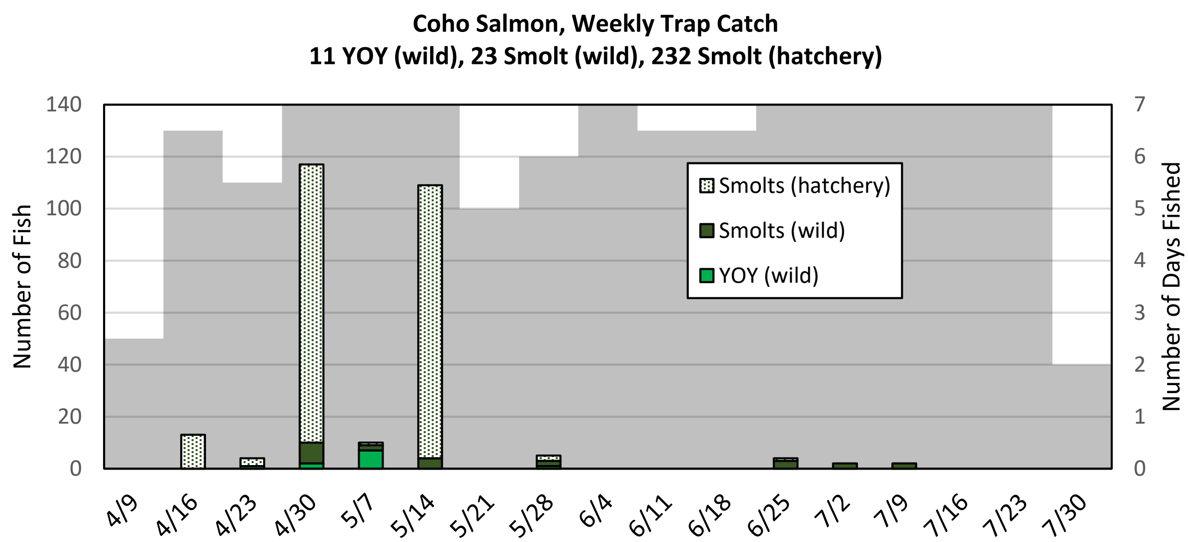


Figure 5.3.9. Estimated average weekly capture efficiency (upper panel) and population estimate of Chinook salmon smolts (x1000) produced from the Dry Creek watershed upstream of Westside Road smolt trap site (rkm=3.3) (lower panel), 2009-2016. Dashed line is the eight year average abundance for all years combined.

Coho were the least abundant of the three salmonid species captured. Hatchery smolts dominated the catch with a total of 232 individuals captured. Steelhead parr and smolt capture was highest in May (Figure 5.3.10).



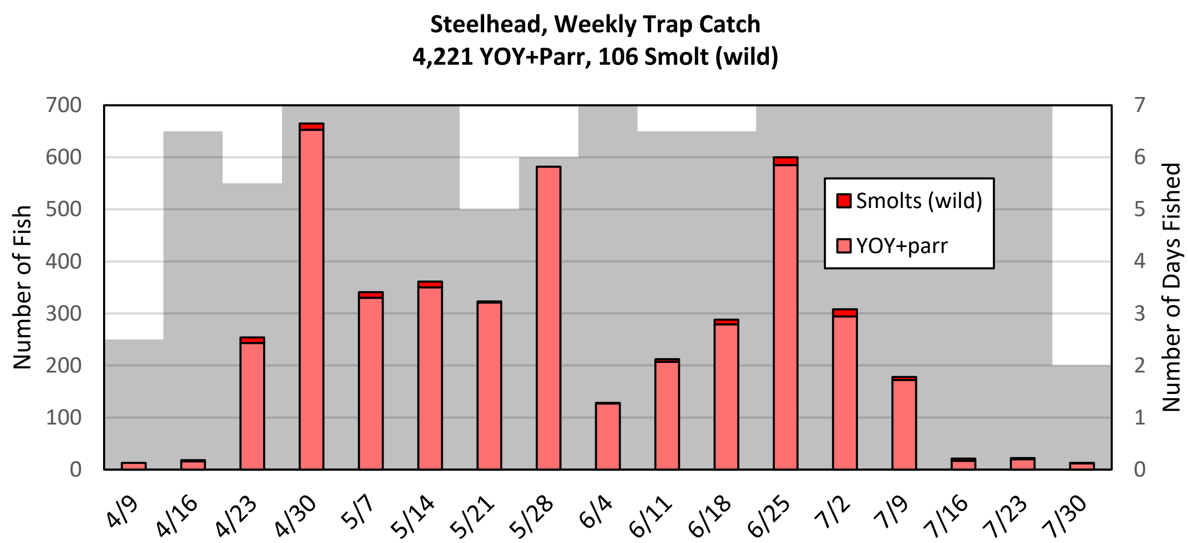


Figure 5.3.10. Weekly trap catch of juvenile coho salmon and steelhead in the Dry Creek rotary screw trap, 2016.

Coho smolt trap catch for the season was relatively low and similar to the catch in 2011, 2012 and 2015 (Figure 5.3.11). The capture of wild coho smolts was still quite low at 23 individuals and is similar to previous year's totals. Steelhead smolt and parr captures (106 and 4,221) were also similar to totals from previous years. Weekly sizes of all salmonids captured at the Dry Creek trap increased over the course of the trapping season in 2016 (Figure 5.3.12).

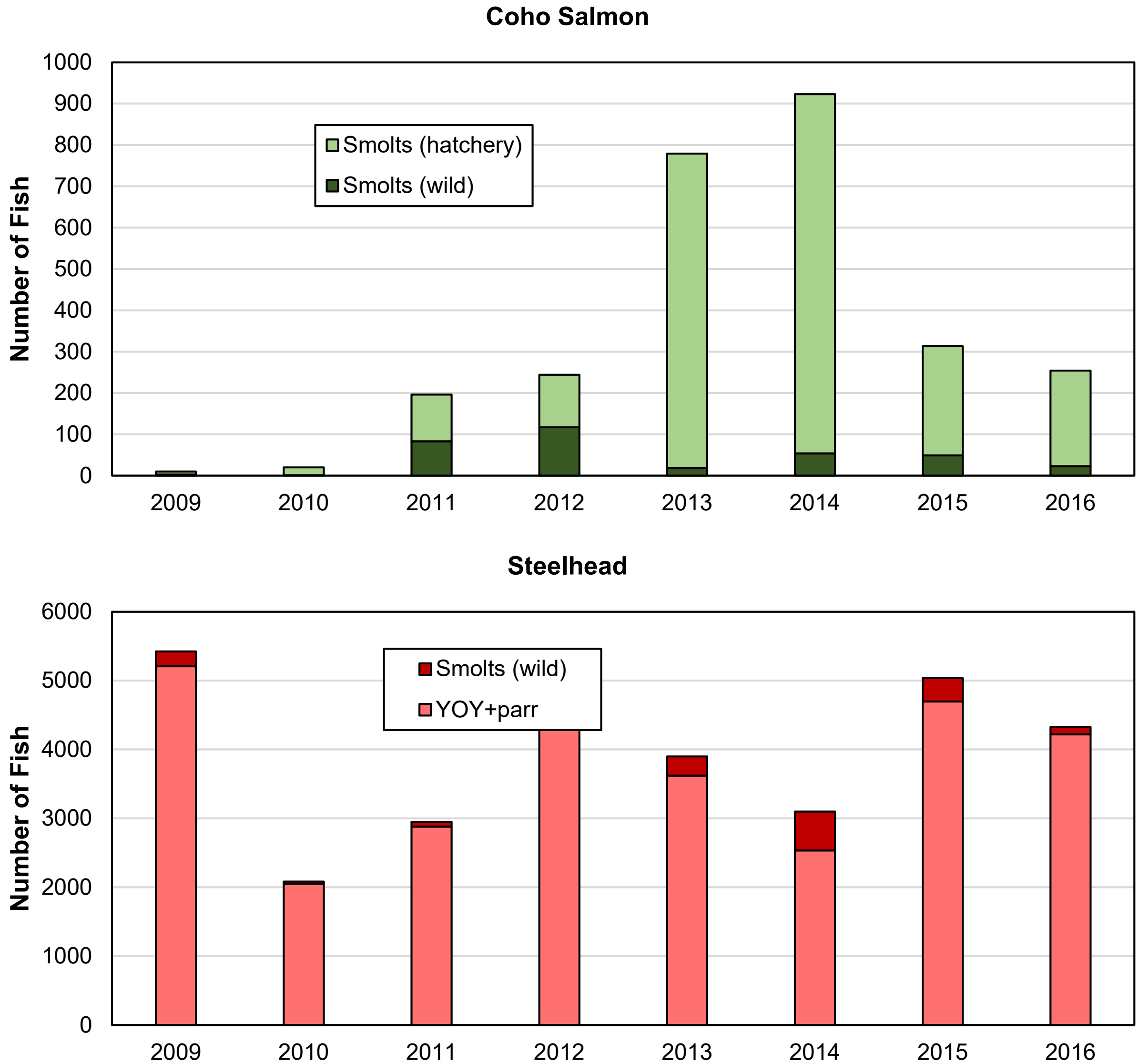


Figure 5.3.11. Trends in trap catch for coho smolts and steelhead smolts and parr, 2009-2016.

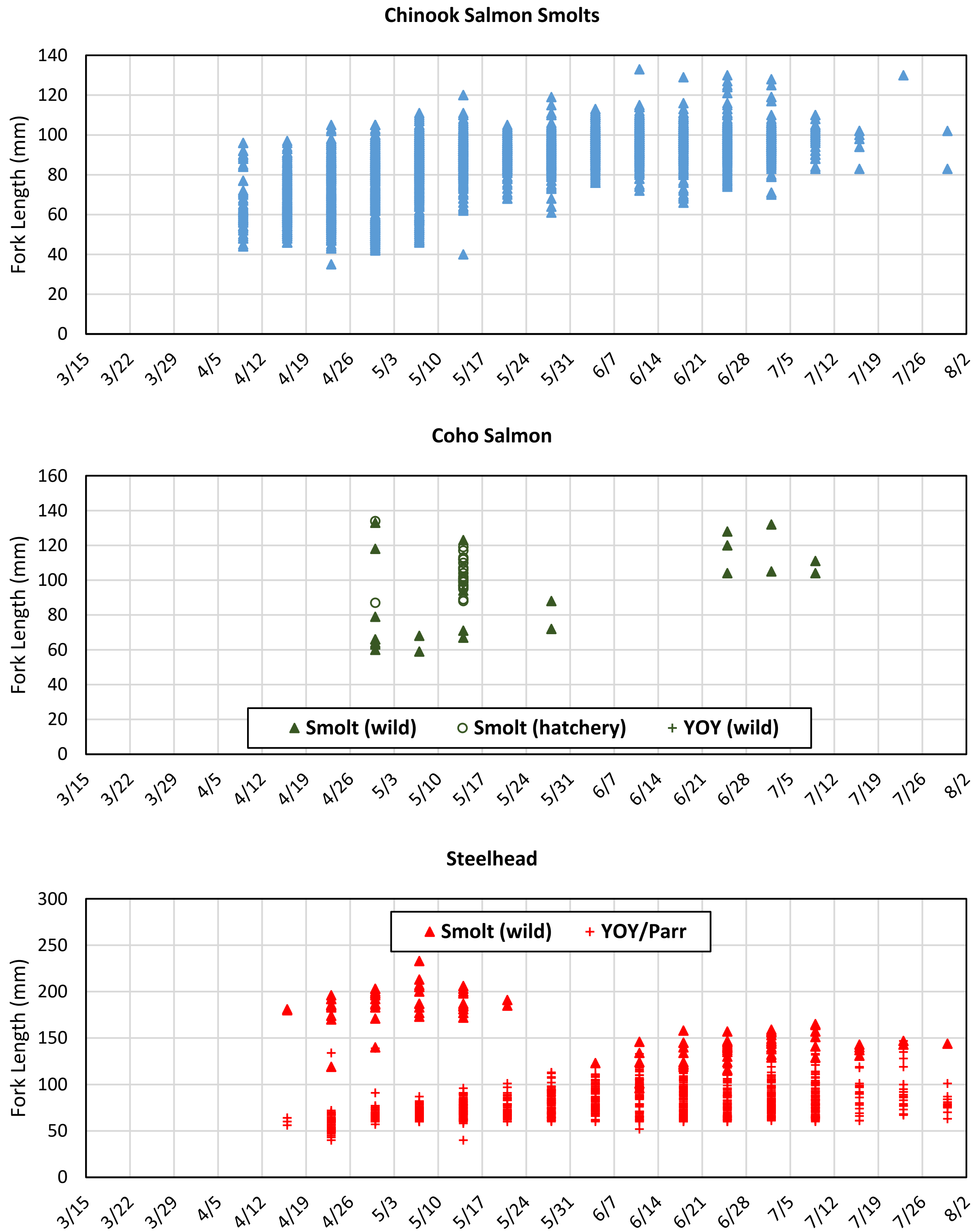


Figure 5.3.12. Fork lengths of juvenile salmonids captured in the Dry Creek rotary screw trap by week, 2016.

Conclusions and Recommendations

Our method of validating fish use in the late fall and winter through the use of PIT antennas within the backwaters continues to provided data that various life stages of all three species are using these habitats in the winter. The CMP sampling framework proved useful as a way of understanding our site-level data in a broader context. Unfortunately, marginal visibility due to high turbidity and vegetation growth in newly-created off-channel habitats continues to hamper our ability to effectively observe fish during summer/fall snorkel surveys and these features are largely too deep to sample with a backpack electrofisher. The difficulty in sampling specific enhancement features is highlighted by the variability observed in steelhead densities observed at enhanced versus un-enhanced areas. Overall steelhead density was higher as compared to 2015, while the density estimated at the enhanced sites were similar to last year. In the future, we will consider alternative methods for estimating summer use of these habitats by juvenile salmonids including PIT-tagging and stationary PIT antennas and, perhaps, radio telemetry.

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CHAPTER 6: Tributary Habitat Enhancements

Tributary Habitat Enhancement

One component of the reasonable and prudent alternative (RPA) identified in the Biological Opinion is the enhancement of salmonid rearing habitats in tributaries to Dry Creek and the Russian River. A total of ten potential tributary enhancement projects are listed in the Biological Opinion with the requirement that the Water Agency implement at least five of these projects by the end of year 3 of the 15 year period covered by the Russian River Biological Opinion. The five projects that the Water Agency intended to complete were 1) Grape Creek Habitat Improvement Project; 2) Willow Creek Fish Passage Enhancement Project; 3) Mill Creek Fish Passage Project; 4) Wallace Creek Fish Passage Project; and 5) Grape Creek Fish Passage Project. The Water Agency entered into agreements with the Sotoyome Resource Conservation District, now named Sonoma Resource Conservation District (RCD), to coordinate and implement two of these projects (the Grape Creek Habitat Improvement Project and Mill Creek Fish Passage Project), and with Trout Unlimited to provide funding towards the Willow Creek Fish Passage Enhancement Project. The Water Agency was also coordinating work with the Sonoma County Department of Transportation and Public Works to implement the Wallace Creek and Grape Creek Fish Passage Projects. After efforts to secure landowner access for the Mill Creek Fish Passage Project were unsuccessful, the Water Agency abandoned efforts on the Mill Creek Fish Passage Project and directed the Sotoyome Resource Conservation District to substitute the Crane Creek Fish Passage Project. The Water Agency also amended its agreement with the RCD to allow the RCD to oversee the implementation of the Grape Creek Fish Passage Project. The Wallace Creek Fish Passage Project, again after efforts to secure landowner access were unsuccessful, was abandoned. In the meantime, an agreement was reached between the National Marine Fisheries Service and landowners for the Mill Creek Fish Passage Project, although by this time the scope of the project had grown considerably beyond the project described in thew Russian River Biological Opinion. In April of 2015, the Water Agency received approval from the National Marine Fisheries Service to provide funding of $200,000 towards the construction of the Mill Creek Fish Passage Project now being implemented by the National Marine Fisheries Service and Trout Unlimited as a substitute for the Wallace Creek Fish Passage Project.

Grape Creek Habitat Improvement

Phase 1

The Grape Creek Phase 1 portion of the project consisted of installing 8 complex log and boulder structures along a 1,200 foot reach of Grape Creek upstream of the Wine Creek Road Crossing (Figures 6.1 and 6.2). Implementation of this work took place in July and August of 2009. All areas where vegetation was disturbed by heavy equipment were replanted with native plants prescribed by restoration staff from the RCD. Additional plantings were also installed per the request of the California Department of Fish and Wildlife, and permission of the landowner, in areas outside the active construction area in an effort to eventually expand the width of the riparian area. A total of 248 native trees and shrubs were planted along this reach of the project.

Figure 6. 1. Grape Creek – Phase 1. In-Stream Large Woody Debris Structure Example (2009 post construction).


Figure 6. 2. Grape Creek – Phase 1. In-Stream Large Woody Debris Structure Example. December 2014 winter flows.


Figure 6. 3. Grape Creek – Phase 1. February 2012.


Figure 6. 4. Grape Creek – Phase 1. December 2014.

Phase 2

The Grape Creek Phase 2 portion of the project consisted of installing 9 complex log and boulder structures and 2 bank layback areas along a 700 foot reach of Grape Creek upstream of the West Dry Creek Road Crossing (Figure 6.5). Implementation of this work took place over two construction seasons, in 2009 and 2010. Construction began in early October 2009 and was cut short due to rain. Revegetation took place in January 2010. In February 2010, portions of one structure (Site 5) were removed as an emergency measure to avoid bank erosion on the opposite bank as a result of the structure's movement during high flows. Construction resumed in late August 2010, with heavy equipment work completed in the first week of September, and final touches placed on erosion control in early October. The remaining vegetation was installed in early 2011 when the soil is sufficiently moist.

Figure 6. 5. Grape Creek – Phase 2. Large Woody Debris and Bank Layback Example.

Figure 6. 6. Grape Creek – Phase 2. February 2012.


Figure 6. 7. Grape Creek – Phase 2. December 2014.

Willow Creek Fish Passage Enhancement Project

Willow Creek is a tributary to the lower Russian River that once supported an abundant subpopulation of coho salmon. The creek continues to support significant potential spawning and rearing habitat; however, access to that habitat is blocked by impassable road culverts and a shallow braided channel that passes through forested wetland. To implement the Willow Creek Fish Passage Enhancement Project, the Water Agency contributed $100,000 in funding to Trout Unlimited towards the removal of a complete barrier in Willow Creek. On October 19, 2010, the Water Agency's Board of Directors approved the funding agreement with Trout Unlimited for the Willow Creek Fish Passage Enhancement Project. The $100,000 in funding was provided by the Water Agency to Trout Unlimited on January 26, 2011. During the summer of 2011, construction was completed for the Willow Creek Fish Passage Enhancement Project (Figures 6.8 and 6.9).

Figure 6. 8. Willow Creek Bridge Installation. September 2011.

Figure 6. 9. Willow Creek Bridge Installation. September 2011.

Crane Creek Fish Passage Project

The Water Agency originally intended to implement the Mill Creek Fish Passage Project. The Mill Creek Fish Passage Project required landowner permission from two property owners in order to design and construct the project. One of the property owners was willing to enter into an agreement to allow the project to move forward; however, the second landowner gave multiple indications that they would allow the project to move forward, but ultimately failed to ever sign any access agreements to allow project design to move forward. Multiple attempts at obtaining the necessary permissions from this landowner were made by the Sotoyome Resource Conservation District and the National Marine Fisheries Service. Still seeing no progress with this landowner, the Water Agency directed the Sotoyome Resource Conservation District in December 2010 to abandon its efforts on the Mill Creek Fish Passage Project and instead implement the Crane Creek Fish Passage Access Project (Figures 6.10 and 6.11). The Crane Creek Fish Passage Access Project consists of the removal of a barrier to fish passage caused by a bedrock outcropping at the lower end of Crane Creek near its confluence with Dry Creek. The proposed project design developed by Prunuske Chatham, Inc., consisted of creating a series of step pools through the bedrock outcropping to create sufficient depth and flow to allow fish passage (Figure 6.12). Design approval was obtained from National Marine Fisheries Service and the landowners in September of 2011. Construction began on October 1, 2011 and was completed on October 18, 2011.

Figure 6. 10. Crane Creek Fish Passage Access Project. Bedrock outcropping.

Figure 6. 11. Crane Creek Fish Passage Access Project. Chiseling pools in bedrock outcropping.

Figure 6. 12. Crane Creek Fish Passage Access Project. Expanded pools in bedrock outcropping (February 2012).

Grape Creek Fish Passage Project

The Grape Creek Fish Passage Project consists of the modification of a concrete box culvert where Grape Creek flows under West Dry Creek Road (Figure 6.13). As part of the permit review and design approval process, the National Marine Fisheries Service noted that the project design did not meet their maximum allowable 0.5-foot drop height for barrier passage. In October 2010, the Water Agency proposed re-designing the project to cut into the culvert bottom instead of placing curbs on top of the culvert bottom in order to meet the 0.5-foot maximum drop height requirement. Because the culvert-bottom is a structural portion of the bridge and culvert, cutting into the culvert bottom substantially increases the design complexity and costs of implementing the project. Between October 2010 and March 2011, the Water Agency coordinated with the Sonoma County Department of Public Works on the proposed re-design of the project. In April 2011, National Marine Fisheries Service indicated that the proposed re-design provided by the Sonoma County Department of Public Works was acceptable. Because of the increased complexity and cost, the revised project design was required to be put out to bid as a general construction contract, which required detailed project drawings and construction specifications. The Water Agency worked with a consultant through the Sotoyome Resource Conservation District to prepare the project construction drawings and specifications. Construction of the Grape Creek Fish Passage Project was completed in October of 2012 (Figures 6.14 and 6.15).

Figure 6. 13. Grape Creek Fish Passage Project – Flat culvert invert proposed for modification.

Figure 6. 14. Grape Creek Fish Passage Project – Newly Constructed October 2012.

Figure 6. 15. Grape Creek Fish Passage Project – First Flows November-December 2012.

Mill Creek Fish Passage Project

The Water Agency had been working towards the construction of the Wallace Creek Fish Passage Project, which would have consisted of the modification of a concrete box culvert where Wallace Creek flows under Mill Creek Road. Engineering designs were completed and the National Marine Fisheries Service had approved those engineering designs for the project. The County of Sonoma Permit and Resource Management Department had submitted permit applications and coordinated site visits with California Department of Fish and Wildlife, National Marine Fisheries Service, U.S. Army Corps of Engineers, and the North Coast Regional Water Quality Control Board. Unfortunately, the Water Agency was been unable to secure the necessary landowner permissions from two of the three landowners in the project area. Because of the inability to secure the necessary landowner permission for the project, the Water Agency abandoned efforts to construct the Wallace Creek Fish Passage Project and began working with the National Marine Fisheries Service on an alternative as a substitute for the Wallace Creek Fish Passage Project.

In April of 2015, the National Marine Fisheries Service acknowledged that a proposal by the Water Agency to provide $200,000 in funding towards the construction of the Mill Creek Fish Passage Enhancement Project would meet the intent of the Russian River Biological Opinion and would be considered as the completion of the fifth and final tributary enhancement project required under the Russian River Biological Opinion. The Mill Creek Fish Passage Enhancement Project is a high-value project that would restore coho salmon access into 11.2 miles of upper Mill Creek. The initial estimate for the Mill Creek Fish Passage Enhancement Project described in the Russian River biological Opinion estimated the cost of the project at $100,000 to $200,000; however, recent estimates place the costs closer to $1,500,000. The Water Agency will provide $200,000 towards the project costs, which is consistent with the original estimate. The remaining funding for the project will come from NOAA grant funding and California Department of Fish and Wildlife Fisheries Restoration Grant Program funding. The project, which was constructed in the summer of 2016, and is expected to allow for fish passage past an existing rock and mortar sill that is a barrier for fish passage under most flow conditions. See Appendix 6.1 for an October 30, 2017 Technical Memo from Prunuske Chatham, Inc. to Trout Unlimited providing detailed post-construction monitoring documentation for the Mill Creek Passage Project.

Figure 6. 16. Mill Creek Fish Passage Project. Existing passage barrier in Mill Creek. December 2009.

Figure 6. 17. Mill Creek Fish Passage Project. Showing completed new bypass channel and roughened ramp on downstream side of the passage barrier in Mill Creek. October 2016.

CHAPTER 7: Coho Salmon Broodstock Program Enhancement

NMFS' Russian River Biological Opinion compels the USACE to continue operation of a conservation hatchery to provide a source of genetically appropriate juvenile Coho Salmon to release into the Russian River watershed. The hatchery program is instrumental to Russian River Coho population recovery and Coho releases are widely recognized as the main reason the Russian River population was not extirpated. The Biological Opinion and Consistency Determination obligate Sonoma Water to provide hatchery support by increasing the production of Coho smolts. This support has primarily been in the form of funding for fish-rearing tanks, purchase of PIT tags, and technical staff to assist with hatchery operations including PIT-tagging of hatchery-reared juveniles. Sonoma Water has also contributed a significant amount of information through direct data collection, financial and staff support to partner entities, and consistent participation on the Russian River Coho Salmon Captive Broodstock Program (RRCSCBP) Technical Advisory Committee (TAC).

In addition to hatchery operations, USACE must also conduct annual monitoring of the distribution and survival of stocked juvenile salmon and the subsequent return of adult Coho to the Russian River. Much of the Coho monitoring in the Russian is implemented by CSG with base funding from USACE. However, Sonoma Water has and will continue to make significant contributions to the collection of monitoring data to allow evaluation of program success. These contributions include data collected at Sonoma Water operated fish monitoring sites (i.e., downstream migrant traps and stationary PIT antenna arrays) as well as assistance to CSG in conducting studies to identify population bottlenecks (e.g., low flow studies) and inform solutions to overcoming those bottlenecks (e.g., Russian River Coho Water Resources Partnership).

The technical aspects of Coho Salmon population recovery are complex, and it is often difficult to evaluate recovery strategies and program success in light of the host of factors operating at a variety of scales to shape Coho populations. The RRCSCBP TAC is a multi-partner effort involving USACE, CDFW, NMFS, CSG, and Sonoma Water. The TAC provides invaluable advice to ensure genetically sound broodstock management, and it develops annual plans for hatchery Coho releases with the primary objective of balancing survival of early life stages in the wild against the risk of artificial selection from releasing older life stages that are reared in the hatchery for a longer period of time. Many of the innovative monitoring methods spearheaded by CSG and Sonoma Water feedback to inform these plans while at the same time providing metrics of program success such as tributary-specific smolt production and numbers of adult returns (see CSG data reports 2004 through present) – both of which have been identified as key metrics in state and federal recovery plans.

A component that has been lacking until recently is a better understanding of the broader context in which salmonid demographic processes operate. In 2013, Sonoma Water and CSG began implementing CDFW's Coastal Monitoring Program (CMP, Adams et al. 2011). The broad-scale metrics from this coastwide effort have and will continue to inform Coho Salmon recovery in the Russian River watershed and elsewhere by helping to decouple those factors that are largely outside our control (e.g., marine survival) from in-watershed recovery efforts.

References

Adams, P. B., L. B. Boydstun, S. P. Gallagher, M. K. Lacy, T. McDonald, and K. E. Shaffer. 2011. California coastal salmonid population monitoring strategy design and methods. CA Department of Fish and Game, Sacramento, CA.

California Sea Grant (CSG). 2004-present. UC Coho Salmon and Steelhead Monitoring Reports. https://caseagrant.ucsd.edu/project/coho-salmon-monitoring/reports.

CHAPTER 8: Wohler-Mirabel Water Diversion Facility

Introduction

The Water Agency diverts water from the Russian River to meet residential and municipal demands. Water is stored in Lake Sonoma and Lake Mendocino, and releases are made to meet downstream demands and minimum instream flow requirements. The Water Agency's water diversion facilities are located near Mirabel and Wohler Road in Forestville. The Water Agency operates six Ranney collector wells (large groundwater pumps) adjacent to the Russian River that extract water from the aquifer beneath the streambed. The ability of the Russian River aquifer to produce water is generally limited by the rate of recharge to the aquifer through the streambed. To augment this rate of recharge, the Water Agency has constructed several infiltration ponds. The Mirabel Inflatable Dam (Inflatable Dam) raises the water level and allows pumping to a series of canals that feed infiltration ponds located at the Mirabel facility. The backwater created by the Inflatable Dam also raises the upstream water level and submerges a larger streambed area along the river. Three collector wells, including the Agency's newest and highest capacity well, are located upstream of Wohler Bridge. These wells benefit substantially from the backwater behind the Dam.

Mirabel Fish Screen and Ladder Replacement

To divert surface water from the forebay of Mirabel Dam, The Water Agency operates a pump station on the west bank of the river. The pump station is capable of withdrawing 100 cfs of surface flow through two rotating drum fish screens in the forebay. The fish screens have been functioning since the dam was constructed in the late 1970's. However, they fail to meet current velocity standards established by NMFS and CDFW to protect juvenile fish. The Biological Opinion requires the Water Agency to replace the antiquated fish screens with a structure that meets modern screening criteria. In 2009, the Water Agency employed the engineering firm of Prunuske Chatham, Inc. to prepare a fish screen design feasibility study. The report was completed in December 2009.

The feasibility study was conducted to develop a preferred conceptual design that meets many of the project objectives while ensuring that the fish screening facilities adhere to contemporary fish screening design criteria. A Technical Advisory Committee composed of the Water Agency engineering and fisheries biologist staff, NMFS, and CDFW provided guidance in refining the objectives and identifying alternatives. Six concept alternatives were evaluated for meeting the project objectives. Schematic designs and critical details were developed for these concept alternatives to assess physical feasibility and evaluate alternatives relative to the objectives. The preferred concept design alternative was determined through an interactive evaluation and was selected because it meets or exceeds the project objectives.

In 2010, the Water Agency solicited qualifications from engineering firms, and a list of qualified consultants was created from the responses. The Water Agency selected HDR Engineering (HDR) because of its demonstrated experience with this type of work and the strength of their proposed project manager, who has a proven track record with fish passage and screening projects. The Water Agency and HDR entered into an Agreement for Engineering Design Services for the Mirabel Fish Screen and Fish Ladder Replacement Project in June of 2011. In 2011 and 2012, HDR completed work on preliminary engineering, geotechnical analysis, hydraulic modeling, development of construction drawings and specifications. HDR's final construction drawings and specifications are anticipated in early 2013. HDR will also provide engineering support during bidding and construction. HDR's design process included consultation at different design steps with the Technical Advisory Committee described above.

Because the fish ladder enhancement identified in the feasibility study is not required by the Biological Opinion, the Water Agency applied for funds from CDFG's Fishery Restoration Grant Program (FRGP) in 2010 to help defray costs associated with fish ladder design. The Director of CDFG awarded the grant to the Water Agency in February 2011. The Water Agency also submitted a second application for FRGP funds in 2012 to help defray costs associated with fish ladder construction. In February of 2013, CDFW approved $1,184,049.00 in FRGP funds towards the construction of the new fishway at Mirabel to improve fish passage at the facility.

In January 2013, the Water Agency's Board of Directors approved and adopted an Initial Study and Mitigated Negative Declaration in accordance with the California Environmental Quality Act (CEQA).

The CEQA document for the project provided a discussion of potential environmental impacts related to the construction, operation, and maintenance of the proposed fish screen and fish ladder modifications. Project construction activities require isolating the work area from the active flow of the Russian River, demolishing the existing fish screen/intake and fish ladder structures on the western bank of the Russian River, and constructing the new fish screen and fish ladder structures. The new facilities will extend approximately 40 feet farther upstream and approximately 100 feet farther downstream than the existing facilities. This larger footprint is necessary to meet contemporary fish screen and fish passage design criteria. Figure 8.1 shows a plan view of the project design. Figure 8.2 shows a conceptual design drawing of the project components.

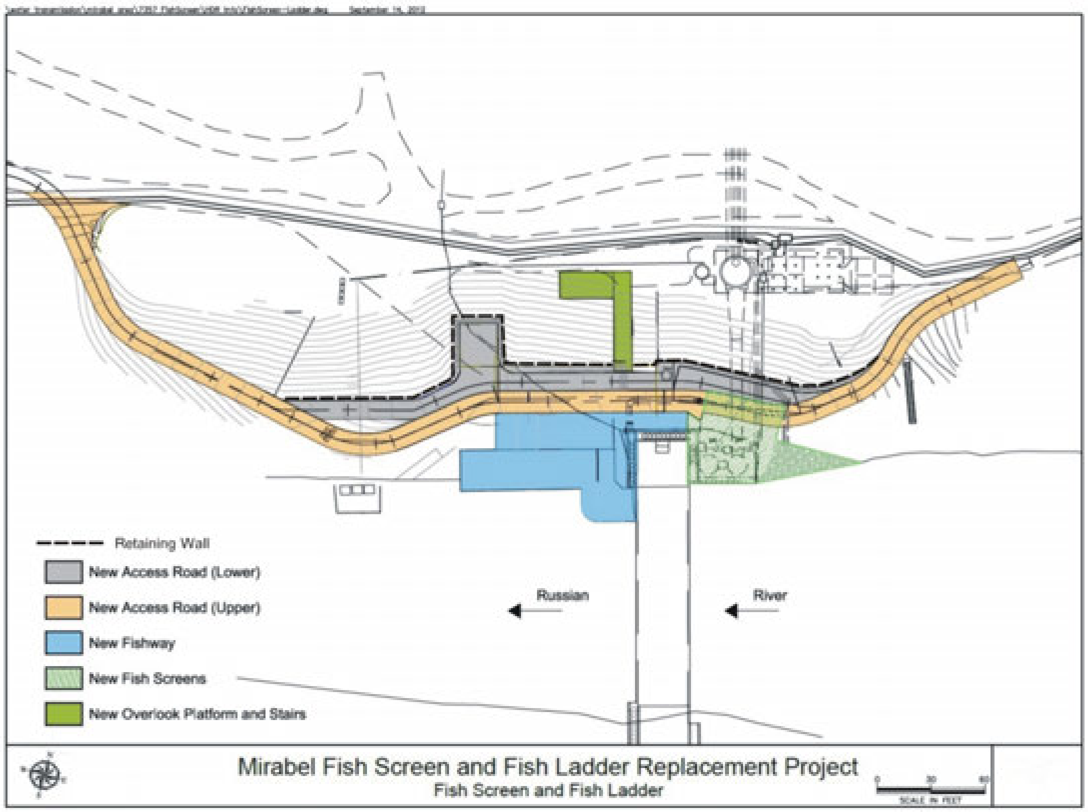

Figure 8. 1. Conceptual plan view drawing of new fish screen and fishway structure at Mirabel.

Figure 8. 2. Artist rendering of new fish screen and fishway structure at Mirabel.

Fish Screen

The proposed intake screen will consist of six 12-foot tall by 6-foot wide panels, with a total area of 432 square feet. The new fish screen will also incorporate a cleaning system to ensure that the screen material does not become clogged. Clogged screens result in higher flows through unclogged portions of the screen, which can lead to fish getting trapped against the screen. The cleaning mechanism is anticipated to be an electric motor-driven mechanical brush system that periodically moves back and forth to clean the intake screen structure.

Fish Ladder

A vertical slot type fish ladder was selected as the recommended design to provide passage for upstream migrating salmonids. Vertical slot fish ladders are commonly used for salmon and steelhead (among other fish species) throughout the world. A vertical slot fish ladder consists of a sloped, reinforced concrete rectangular channel separated by vertical baffles with 15-inch wide slots that extend down the entire depth of the baffle. The baffles are located at even increments to create a step-like arrangement of resting pools.

The design will be self-regulating and provide consistent velocities, flow depths, and water surface differentials at each slot throughout a range of operating conditions. It is anticipated that the ladder will be configured to accommodate a range of fish passage conditions while the Mirabel Dam is up and river flows ranging from 125 to 800 cubic feet per second. Fish passage while the Mirabel Dam is down will also be accommodated, but is not the primary focus of design. The fish ladder will extend approximately 100 feet further downstream than the existing fish ladder at the site.

Fisheries Monitoring Components

The Water Agency currently conducts a variety of fisheries monitoring activities at its Mirabel Dam facilities. The new fish ladder design will support these monitoring activities by providing a dedicated viewing window and video equipment room and a fish trapping and holding area built into the fish ladder. The monitoring information collected by Water Agency staff is critical in tracking population trends and movement of different species in the Russian River system.

Education Opportunities

The existing facility at Mirabel is visited every year by approximately 3,000 schoolchildren as part of the Water Agency’s water education efforts. The existing facility allows schoolchildren to see a critical component of the Water Agency’s water supply system, but the views of the top of the existing fish ladder do not offer much opportunity for observing and learning about the fisheries of the Russian River system. The project includes a viewing area, separate from the video monitoring viewing window, which will allow visitors to see into the side of the fish ladder. The educational experience for schoolchildren will be improved by having the opportunity to actually see fish travelling up or down the fish ladder.

Supporting Components

The project design includes a variety of other components that support the primary fish screen and fish ladder aspects of the project. These other components consist of items such as seismic stabilization of the soils around the Mirabel dam, replacement of the buoy warning line upstream of the Mirabel Dam, modification of the existing access road to the project site, and the installation of a viewing platform to allow visitors a safe location to view the overall facility. The existing access road down to the Mirabel Dam is a steep one-way road. Vehicles going down to the Mirabel Dam area must turn around or back up the road down to the project site. The proposed project includes a modification of the access road so that the road will not be as steep and will include both an entrance and exit ramp from the Mirabel Dam site. A stairway from the top of bank down to the Mirabel Dam will allow visitor access from the upper levee road area down to the Mirabel Dam.

Construction Status

In March 2014, Hayward Baker began construction on the first phase of site improvements at the Mirabel Dam. This work consisted of the seismic stabilization of the soil area around the area of the Mirabel intake screens and fish ladder on the west bank of the Russian River. Seismic stabilization consisted of the installation of approximately 300 compacted stone columns along the levee berm at the Mirabel facility. The Mirabel seismic improvement work was completed in July of 2014 by Hayward Baker, which then allowed the second phase of construction activities to begin. Once Hayward Baker had demobilized their equipment from the work area, a second contractor (F&H) mobilized to the site in July of 2014 to begin the construction of the fish screen, fish ladder, and viewing chamber project. By the end of 2014, demolition of the original intake structure and fish ladder was complete. At the beginning of 2015, high flows in the Russian River resulted in a temporary shut-down of construction activities; however, by mid-January 2015, construction activities were once again underway and continued uninterrupted for the remainder of 2015. At the end of December 2015 and early January, 2016, high flows in the Russian River again slowed construction progress.

Photo 8. 1. High flow events inundate job site. January 13, 2016.

Photo 8. 2. Progress is continuing to be made and the majority of the concrete pours are complete by February 17, 2016.

Photo 8. 3. The new flat-panel diversion screens are in place. May 4, 2016.

Photo 8. 4. The new viewing gallery windows are installed. May 18, 2016.

Photo 8. 5. The sheet pile in place since 2014 for isolating the job site from the Russian River is being removed. May 21, 2016.


Photo 8. 6. Sheet pile completely removed. June 8, 2016.


Photo 8. 7. The new Mirabel fish ladder, screen, and viewing galley structure. June 29, 2016.


Photo 8. 8. The Mirabel inflatable dam being raised again. July 8, 2016.


Photo 8. 9. First flow starting to go through the new vertical slot fishway. July 8, 2016.


Photo 8. 10. The new viewing gallery area. July 11, 2016.


Photo 8. 11. Some of the first visitors seen through the new viewing gallery windows. July 11, 2016.

Photo 8. 12. One of the first groups of Chinook salmon seen through the new viewing gallery windows. October 20, 2016.

CHAPTER 9: Adult Salmonid Returns

Adult Salmonid Escapement

Since 2000, the Water Agency has been operating video cameras in the east and west fish ladders to assess the adult Chinook salmon run passing the Mirabel inflatable dam located at river km 39 (rkm 39). In 2014 and 2015, however, construction of a new fish ladder and fish screens at Mirabel prevented inflation of the dam which impacted our ability to operate the video monitoring system in the Mirabel fish ladders for the full season. Because the logistics of operating a video camera in the new fish ladder were unknown, the Water Agency adjusted its sampling program by (1) installing and evaluating video operations in the new and old fish ladders at Mirabel, (2) installing and operating a video camera in the Russian River at the Healdsburg dam fish ladder just upstream of the Dry Creek confluence (rkm 55) and (3) a DIDSON camera in Dry Creek (USGS, rkm 0.36) just upstream of the confluence with the Russian River (Dry Creek/Russian River confluence rkm 52). Because little Chinook spawning habitat exists between Dry Creek, Healdsburg, and the Mirabel dam, conceptually, accurate counts of adult Chinook at Dry Creek and Healdsburg should represent the majority of the run.

Methods

A digital camera and lighting system was installed in the east and west Mirabel fish ladders and the Healdsburg fish ladder and video was recorded to a hard drive located in a nearby building. Individuals were counted as moving upstream once they exited the upstream end of the camera's view. For each adult salmonid observed, the reviewer recorded the species, date, and time of upstream passage. During periods of low visibility, it was not always possible to identify fish to species although identification as an adult salmonid was usually possible. Adult salmonids that could not be identified to species were lumped into a general category called "unknown salmonid." Unknown salmonids were then partitioned into species by taking the proportion of each species positively identified in the ladder on a given day and multiplying the number of unknown salmonids on that same day by these proportions. On days when no salmonids could be identified to species, an average proportion from adjacent days was used to assign species for the unidentified salmonids on that day.

Data collection in Dry Creek using a DIDSON was funded from a Fisheries Restoration Grant awarded to the Water Agency for implementation of the Coastal Monitoring Program in the Russian River. Because species identification is not possible from DIDSON, we relied on fish size, which can be reliably estimated with the DIDSON, to assign fish with body size of 2 feet or greater as a salmonid. Next, based on historical run-timing at Mirabel (years 2008-2013), we further apportioned salmonids counted prior to January 23, 2016 as Chinook, steelhead or coho. Finally, beginning January 23, 2016 all adult salmonids were assumed to be steelhead.

Results

In 2016, the Mirabel fish ladder cameras were in operation from September 9 to November 19 (west camera) and from November 10 to November 19 (east camera), although visibility was limited between October 25 and November 19 (Figure 9.1, Figure 9.2). The east side camera operated 24 hours/day after installation until it was removed.

Chinook Salmon

For the 2016 video monitoring season, 1,062 adult Chinook salmon were observed passing the Mirabel fish counting station (including “unknown salmonids” prorated as Chinook) (Table 9.1). However, poor visibility after October 25, and the early removal of the camera on November 19 because of high flows resulted in an underestimation of Chinook salmon passing through the fish ladder in 2016. A total of 51 fish were categorized as an “unknown salmonid” (i.e., they possessed the general body shape of an adult salmonid, but could not be identified to species). Of these 51 unknown salmonids 49 were partitioned to Chinook salmon.

During the monitoring period at Healdsburg, we arrived at 276 adult Chinook from a combination of positive identification of 253 individuals and proration of all 23 individuals initially identified as “unknown salmonids”. At Dry Creek we observed 2,550 fish with a length greater than or equal to 2 feet on the USGS DIDSON camera. Based on their size we assumed all of these fish were adult salmonids (however, this assumption may not be valid – see ‘Conclusions and Recommendations’ section). Using historical run timing information from Mirabel, 1,808 of these 2,550 unknown salmonids were prorated to Chinook; the remainder were likely steelhead and coho. For the reasons discussed below, the sum of Chinook counts (2,084) from Healdsburg and Dry Creek is not necessarily comparable to minimum counts for other years from the Mirabel fish ladder; however, it is within the range of counts from previous years (Table 9.1). By combining historical information from Mirabel with DIDSON and video data from 2016 we were able to make inferences about the Chinook run across a similar time frame that the Mirabel video camera is typically operated. The Chinook run in 2016 began to ramp up in mid-October, and, based on Dry Creek DIDSON data, likely peaked in late-October.

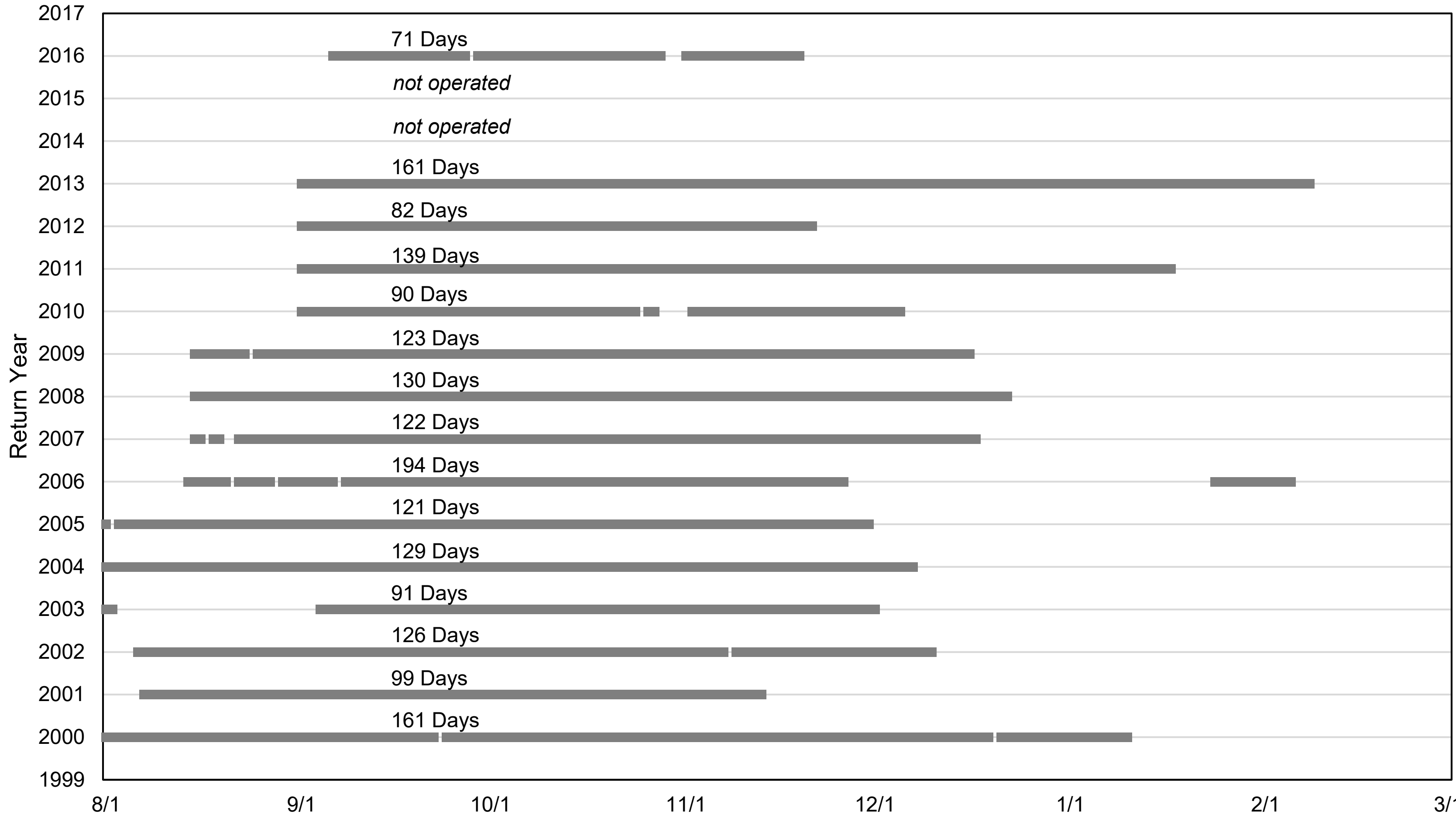

Figure 9.1. Period of operation by adult salmonid return year of video counting station at the Mirabel dam. 'Days' refer to the number of days of operation each year.

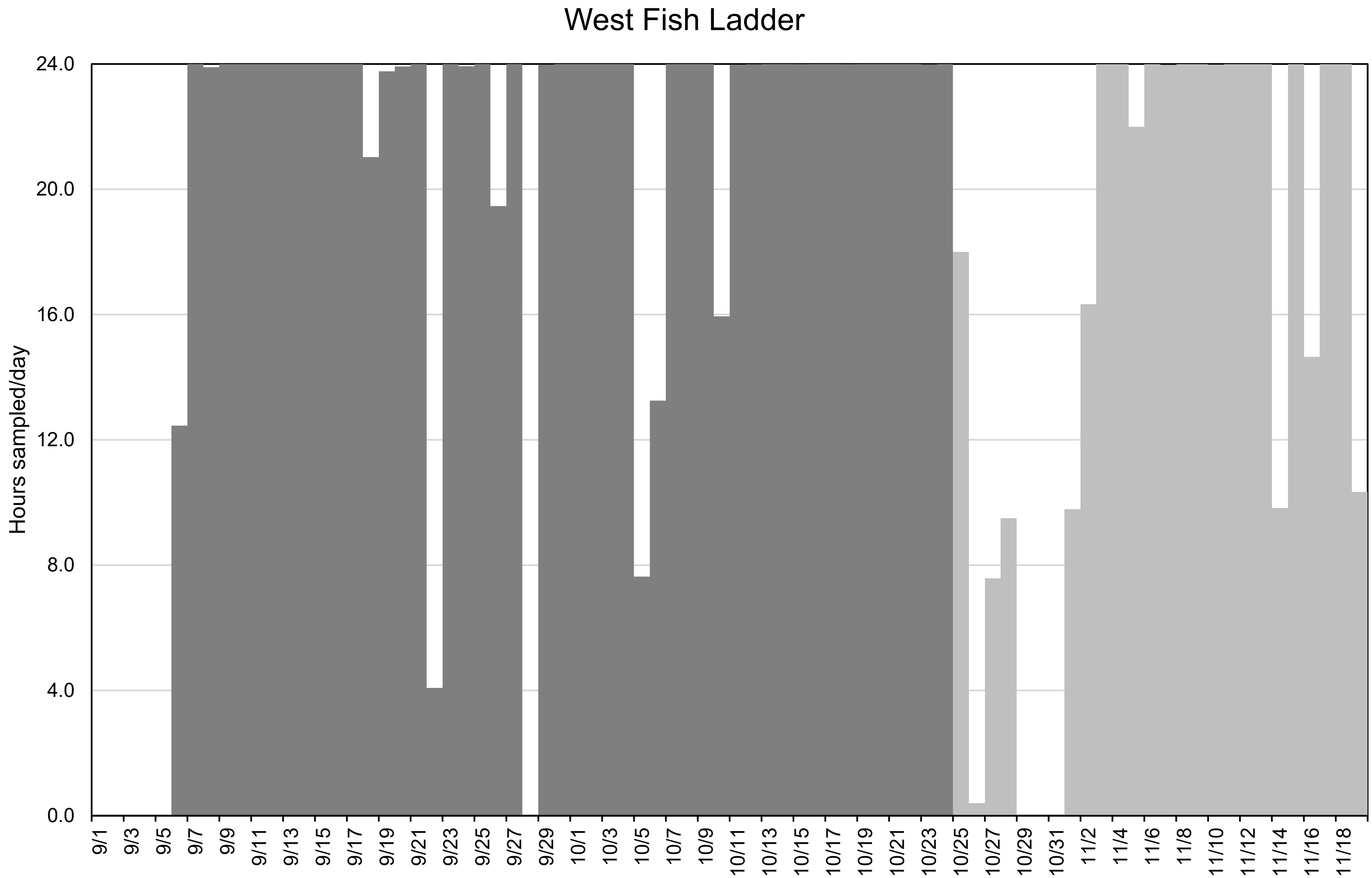

Figure 9.2. Number of hours/per day that the west fish ladder camera was in operation at the Mirabel dam in 2016. Dark grey represent hours when the cameras were operating and turbidity was low. Light gray represents periods of high turbidity which limited the accuracy of the counts.

Table 9.1. Weekly count of adult Chinook salmon at the Mirabel dam fish ladders, 2000-2016. Dashes indicate that no sampling occurred during that week.

Week	2000	2001	2002	2003	2004	2005	2006[1]	2007	2008	2009	2010	2011	2012	2013	2014[2]	2015[2]	2016[3]
15-Aug	0	0	1	--	0	0	0	0	0	0	--	--	--	--			--
22-Aug	1	0	8	--	0	1	1	0	0	0	--	--	--	--			--
29-Aug	0	3	7	2	1	4	0	0	1	0	0	0	0	1			--
5-Sep	9	1	18	7	1	4	0	0	0	0	0	0	1	1			0
12-Sep	36	7	19	20	3	14	3	0	2	0	0	0	2	2			0
19-Sep	25	12	65	23	8	14	4	1	17	0	3	1	0	1			0
26-Sep	50	17	1223	181	16	31	8	4	84	0	1	158	70	17			8
3-Oct	31	240	113	146	42	27	317	10	126	78	669	534	51	44			32
10-Oct	115	51	628	515	52	112	87	39	82	562	896	390	551	4			291
17-Oct	81	10	272	232	651	556	532	26	13	177	153	1070	1886	8			392
24-Oct	465	300	153	532	2287	309	114	106	22	285	280	273	996	27			131
31-Oct	64	661	505	2969	185	613	1531	250	511	135	94	223	1654	315			56
7-Nov	23	81	2337	1289	1189	699	298	429	174	335	169	90	619	731	Not Operated	Not Operated	50
14-Nov	182	--	20	47	221	127	459	154	15	38	43	120	851	1063			103
21-Nov	201	--	37	95	57	63	53	96	24	129	113	266	50	179			--
28-Nov	110	--	14	45	60	33	--	425	19	24	76	6	--	99			--
5-Dec	19	--	53	--	16	--	--	476	18	9	5	1	--	172			--
12-Dec	15	--	--	--	--	--	--	4	8	28	--	2	--	125			--
19-Dec	17	--	--	--	--	--	--	--	13	--	--	10	--	73			--
26-Dec	1	--	--	--	--	--	--	--	--	--	--	16	--	32			--
2-Jan	0	--	--	--	--	--	--	--	--	--	--	2	--	53			--
9-Jan	0	--	--	--	--	--	--	--	--	--	--	10	--	58			--
16-Jan	--	--	--	--	--	--	--	--	--	--	--	1	--	28			--
23-Jan	--	--	--	--	--	--	0	--	--	--	--	--	--	73			--
30-Jan	--	--	--	--	--	--	0	--	--	--	--	--	--	36			--
6-Feb	--	--	--	--	--	--	--	--	--	--	--	--	--	10			--
Total	**1,445**	**1,383**	**5,474**	**6,103**	**4,788**	**2,607**	**3,407**	**2,021**	**1,129**	**1,800**	**2,502**	**3,173**	**6,730**	**3,152**	**--**	**--**	**1,062**

[1] Video cameras were reinstalled and operated from 4/1-6/27/2007 but no Chinook were observed.
[2] Video cameras not operated in 2014 and 2015 because this site was under construction in order to construct the new fish screens and ladder.
[3] Typically 1 camera is operated in both fish ladders but in 2016 the video camera was only operated in the east ladder for the final 10 days of the season.

Coho Salmon

During the monitoring period for the 2016 return year, we observed 12 adult coho (10 from Mirabel and 2 from Healdsburg). These images were reviewed by fisheries biologist from the Water Agency, NMFS, and California Sea Grant (CSG). Because of the timing of camera operations, which are tied to dam operations, and the location of these monitoring sites upstream of significant amounts of coho habitat in the basin, these counts are not the best indicator of adult coho returns to the basin. Instead, we suggest the basinwide spawner survey estimate of 202 (95% CI: 123-281) as the most comprehensive and accurate indicator of all adult coho (hatchery- and natural-origin) returning to the Russian River basin in 2016-17. This estimate is based on spawner surveys in the coho stratum of the Russian River Coastal Monitoring Program sampe frame (see Adams et al. 2011 for details).

Steelhead

Based on hatchery returns, steelhead migrate and spawn in the Russian River primarily between December and March; however, we removed the Mirabel cameras in late November and there is significant uncertainty about the accuracy of the steelhead count in Dry Creek in 2016-17. In total, 15 steelhead were observed migrating through the Mirabel Fish ladder between September 29 and November 19. At Healdsburg, 1 adult steelhead was observed. Using historical run timing information from Mirabel, 761 of the 2,550 unknown salmonids observed on the Dry creek DIDSON were prorated to steelhead.

Conclusions and Recommendations

Data collected in 2016 at Mirabel are of limited value for assessing run size primarily because of the unusual hydrologic conditions that existed as opposed to issues related to the new fish ladder. Indeed, the new camera/lighting system offered a clear view of the entire fish ladder, facilitating an accurate count of fish migrating upstream past the cameras when water clarity was suitable. Although the viewing chamber is approximately 11 feet deep, the camera was configured to encompass the entire width and depth of the viewing chamber. Further refinements to the camera/lighting system will improve our ability to identify and count fish in the future.

As opposed to the past several years which were typified by drought-like conditions, streamflows in 2016 were unusually high resulting in reduced visibility caused by turbidity from early rain. Starting with the storm of October 25, which is statistically the middle of the run and close to the statistical peak of the run, turbidity increased to the point where our ability to see fish on the video cameras was compromised. Early high flows exceeding the level at which we can safely keep the dam inflated also caused us to drop the dam on November 19 which resulted in the second earliest date that video monitoring has been terminated since counts began in 2000. These factors likely resulted in an underestimation of the number of Chinook salmon returning to the Russian River 2016. Because of power issues, we were also unable to operate a camera in the east ladder except for 10 days near the end of the season.

Although sampling issues hampered our ability to accurately assess the Chinook run size in 2016, the appearance of Chinook salmon at the Mirabel and Healdsburg tracked fairly well, suggesting that at least some of the fish take no more than 1-2 days to traverse the approximately 14 kilometers between the two sites. From September 27 to October 24, 739 Chinook salmon were counted at the Mirabel dam and 241 were observed at the Healdsburg fish ladder. In addition, 395 large fish (assumed to be Chinook salmon based on the time of year), were observed in Dry Creek on the DIDSON during that same time period resulting in a combined total of 636 Chinook observed at the Healdsburg and the Dry Creek, combined. Evidence from dive surveys in past years suggests that many or all of these fish could have been holding in the mainstem Russian between the Mirabel and Healdsburg dams.

While the Dry Creek DIDSON camera performed well during the Chinook season, high flows in January greatly impacted the utility of the DIDSON during the steelhead season. A large storm during January misaligned the DIDSON camera, pointing it in a direction which made it difficult to accurately record fish passing the site. Furthermore, high turbidity related also likely reduced the range of the DIDSON which led to further undercounting. This issue is best illustrated by comparing the number of salmonids observed on the DIDSON that were prorated to steelhead to the number of adult steelhead trapped at the Warm Springs fish hatchery on Dry Creek. In total, 761 salmonids observed on the DIDSON were prorated to steelhead while 6,388 adult steelhead were trapped at the Warm Springs Hatchery, all of which would first had to have passed the DIDSON before arriving at the hatchery (Figure 9.3).

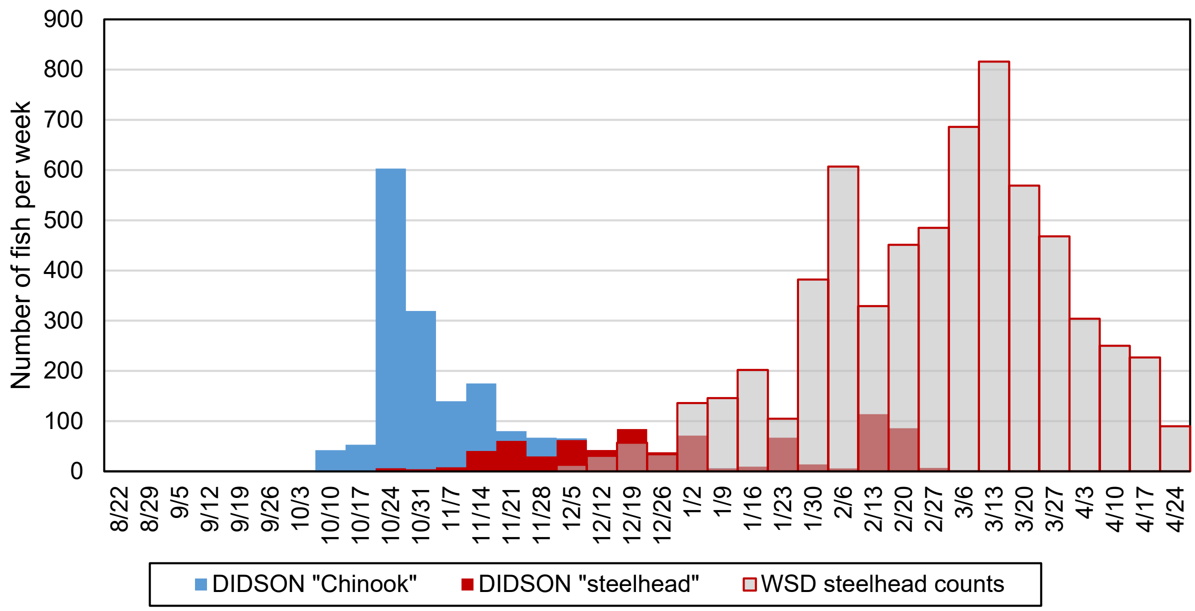

Figure 9.3. The weekly number Chinook and steelhead prorated from salmonids observed on the Dry Creek DIDSON in 2016-17. All upstream moving fish greater than 2 feet in length observed on the DIDSON were considered salmonids. Prior to January 23, 2016, salmonids observed on the DIDSON were prorated to Chinook and steelhead using historical (years 2008-2013) species ratios from Mirabel. From January 23, to April 15, 2016 (the removal date of the Dry Creek DIDSON) all salmonids were considered steelhead. Warm Springs Dam steelhead counts include both wild and hatchery adult steelhead trapped at the hatchery at Warm Springs Dam in 2016-17.

Chinook Salmon Spawning Ground Surveys

Although not an explicit requirement of the Biological Opinion, the Water Agency performs spawning ground surveys for Chinook salmon in the mainstem Russian River and Dry Creek. This effort compliments the required video monitoring of adult fish migration and has been stipulated in temporary D1610 flow change orders issued by the State Water Resources Control Board to satisfy the Biological Opinion (see Pursue Changes to D1610 flow chapter of this report). The Water Agency began conducting Chinook salmon spawning surveys in fall 2002 to address concerns that reduced water supply releases from Coyote Valley Dam (Lake Mendocino) may affect migrating and spawning Chinook salmon (Cook 2003). Spawner surveys in Dry Creek began in 2003.

Background information on the natural history of Chinook salmon in the Russian River is presented in the 2011 Russian River Biological Opinion annual report (SCWA 2011). The primary objectives of the spawning ground surveys are to (1) characterize the distribution and relative abundance of Chinook salmon spawning sites, and (2) compare annual results with findings from previous study years.

Spawner surveys were limited in the Russian River from Hopland to the Healdsburg area and Dry Creek during 2016. A late-season spawning run of Chinook salmon coupled with heavy rainfall and subsequent high river flows in early December 2016 prevented field studies to be conducted during the peak migration period of salmon in the Russian River mainstem. Spawner surveys were possible in Dry Creek due to regulated, clear water releases from Lake Sonoma during fall 2016.

Methods

Chinook salmon redd (spawning nest) surveys are conducted annually in the Russian River during fall. Typically, the upper Russian River basin and Dry Creek are surveyed (Figure 9.4). The study area includes approximately 114 km of the Russian River mainstem from Riverfront Park (40 rkm), located south of Healdsburg, upstream to the confluences of the East and West Forks of the Russian River (154 rkm) near Ukiah. River kilometer (rkm) is the meandering stream distance from the Pacific Ocean upstream along the Russian River mainstem and for Dry Creek the distance from the confluence with the Russian River upstream. In 2003, the study area was expanded to include 22 rkm of Dry Creek below Warm Springs Dam at Lake Sonoma to the Russian River confluence.

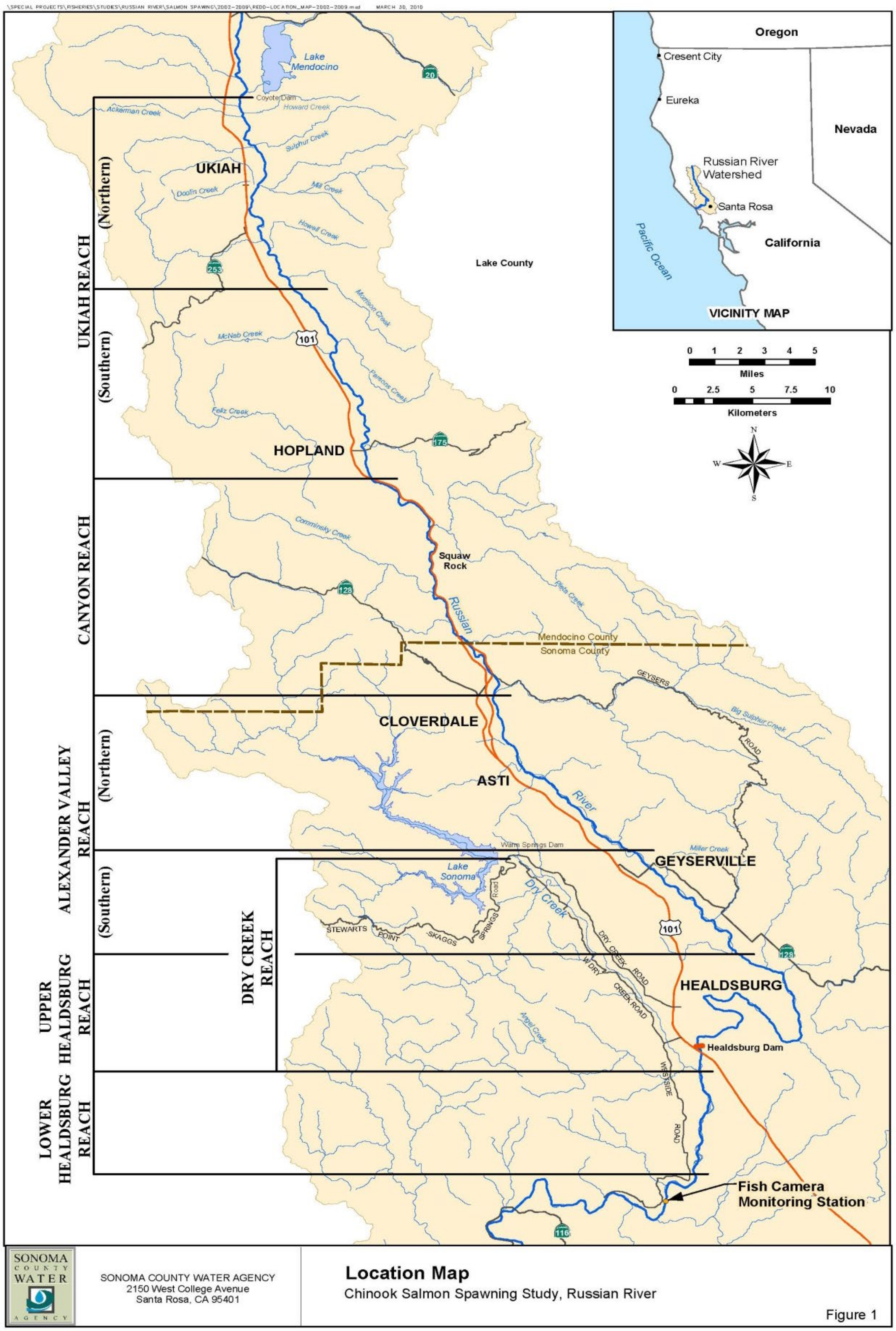

Figure 9.4. **Chinook salmon spawning survey reaches. Only Canyon, Alexander Valley, Upper Healdsburg, and Dry Creek reaches were surveyed in 2016.**

The Chinook salmon spawning ground study consists of a single-pass survey during the estimated peak of Chinook salmon fall spawning. A crew of two biologists in kayaks visually searched for redds along the streambed. Riffles with several redds are inspected on foot. The locations of redds are recorded using a global positioning system (GPS). Surveys are cancelled or postponed if increased turbidity from heavy rainfall obscures the detection of redds. Also, in recent years releases of highly turbid water from Lake Mendocino have prevented an accurate count of redds in Ukiah reach.

As mentioned above, Chinook salmon spawner surveys were curtailed during fall 2016 due to poor survey conditions. The Canyon, Alexander Valley, and Upper Healdsburg reaches of the Russian River were surveyed on December 7-8, 2016. To follow salmon spawning and determine peak activity in Dry Creek four bi-monthly surveys were conducted from November 2 to December 21, 2016. The survey conducted on November 16, 2016 along Dry Creek contained the largest count of redds and was selected as the single-pass visit to represent the abundance of redds in Dry Creek.

Results

Most of the Chinook salmon spawning typically occurs in the upper Russian River mainstem and Dry Creek (Table 9.2). During 2016, there were 58 redds observed in the Canyon, Alexander Valley, and Upper Healdsburg reaches of the Russian River, which is the lowest abundance recorded since surveys began in 2002. This survey was conducted prior to heavy winter rainfall that likely initiated additional spawning activity. Therefore, 58 redds is presumed to underestimate the actual number of redds produced in these reaches. During four surveys of Dry Creek a total of 186 individual Chinook salmon redds were detected. The highest count of redds on a single survey was 90 redds on November 16, 2016 (Table 9.2). This single pass number is the third lowest since 2003. In 2015 a similar number of redds, at 78 redds, were detected in Dry Creek.

Conclusions and Recommendations

Although Chinook salmon surveys were restricted to four reaches in 2016 the distribution and abundance of redds appear to be similar to or within the range of other redd numbers observed during previous study years. The abundance of Chinook salmon redds have shown a sharp decline in the past two fall runs. Although there are many factors that could be driving this trend, it is likely that three years of severe drought in the region is a major contributor.

Table 9.2. Chinook salmon redd abundances by reach, upper Russian River and Dry Creek, 2002-2016. Redd counts are from a single pass survey conducted during the peak of fall spawning activity. *Survey either not completed or incomplete.

Reach	Reach (rkm)	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Ukiah	33.1	511	464	284	*	248	118	20	38	*	*	90[2]	81	*	*	*
Canyon	20.8	277	190	169	*	68	88	36	38	*	*	*	43	*	*	16[2]
Alexander Valley	26.2	163	213	90	*	62	131	65	129	*	97	185	163	*	61[2]	41[2]
Upper Healdsburg	25.6	79	40	8	*	23	67	48	38	*	66	53	57	*	*	1[2]
Lower Healdsburg	8.2	6	0	7	*	1	2	9	30	*	7	4	18	*	*	*
Russian River	**113.9**	**1036**	**907**	**558**	*	**402**	**406**	**178**	**273**	*	**170**	**332**	**362**	*	*	*
Dry Creek	21.7	*	256	342	*	201	228	65[1]	223	269	229	362	325	*	78	90
Total	**135.6**	*	**1163**	**900**	*	**603**	**637**	**243**	**496**	*	*	*	*	*	*	*
Relative Contribution of Redds																
Russian River (%)	84.0	*	78.0	62.0	*	66.7	63.7	73.3	55.0	*	*	*	52.7	*	*	*
Dry Creek (%)	16.0	*	22.0	38.0	*	33.3	36.3	26.7	45.0	*	*	*	47.3	*	*	*

[1]Redd numbers are an estimate.

[2]Redd numbers are presumably an underestimate due to poor survey conditions.

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Chapter 10: Synthesis

Introduction

The Sonoma County Water Agency has collected a variety of fish and water quality monitoring data relevant to fulfilling the overall monitoring objectives in the Reasonable and Prudent Alternative (RPA) of the Russian River Biological Opinion. Those efforts have been detailed in portions of this report leading to this chapter. The objectives specific to this synthesis chapter are to relate these data sets to one another first by illustrating the spatial and temporal extent of monitoring activities in the basin and second by presenting and discussing emerging trends in salmonid abundance, movement and growth in streams encompassed by the RPA.

As in previous years of RPA implementation, we collected fish and related environmental data from a broad spatial and temporal extent in the Russian River Basin (Figure 10.1, Figure 10.2). We collected juvenile and smolt data from multiple locations in Dry Creek, Mark West Creek, Dutch Bill Creek, Austin Creek and the Russian River estuary. We counted adult salmonids with an underwater video system on mainstem Russian River at the Healdsburg dam, a DIDSON system on mainstem Dry Creek at the mouth and we conducted seven repeat Chinook salmon spawner surveys on the 22 km of stream length in mainstem Dry Creek downstream of Warm Springs Dam. Juvenile salmonids were sampled throughout the Russian River watershed using a variety of techniques. In mainstem Russian River, juvenile salmonids were sampled using beach seining at 8 fixed locations in the estuary and passive integrative transponder (PIT) antenna arrays operated near the upstream extent of the tidal portion of the estuary in Duncans Mills and adjacent to the golf course in Northwood and at points near the upstream extent of the river impounded by the Mirabel dam (Syar). Because of ongoing construction of a new fish ladder at the Water Agency's inflatable dam in Mirabel, neither downstream migrant trapping nor adult video monitoring could be conducted on the mainstem in 2016. In tributaries of the lower river, juvenile salmonids were sampled using downstream migrant trapping with rotary screw traps on Mark West Creek at Trenton-Healdsburg Road and Austin Creek at the gravel mine as well as a funnel net on Dutch Bill Creek in Monte Rio. PIT antennas were operated in conjunction with downstream migrant trap sites on Austin Creek and Dutch Bill Creek. In Dry Creek juvenile salmonids were sampled using downstream migrant trapping with a rotary screw trap and backpack electrofishing. PIT antennas were operated in conjunction with the downstream migrant trap and additional PIT antennas were operated in main-channel and off-channel sites in Dry Creek. Complementary data on water quality were collected by means of continuously-recording data sondes at multiple sites throughout the estuary/lagoon and from bi-weekly and weekly grab samples at additional sites. Details regarding the specifics of water quality and fisheries monitoring activities are covered in individual chapters of this report.

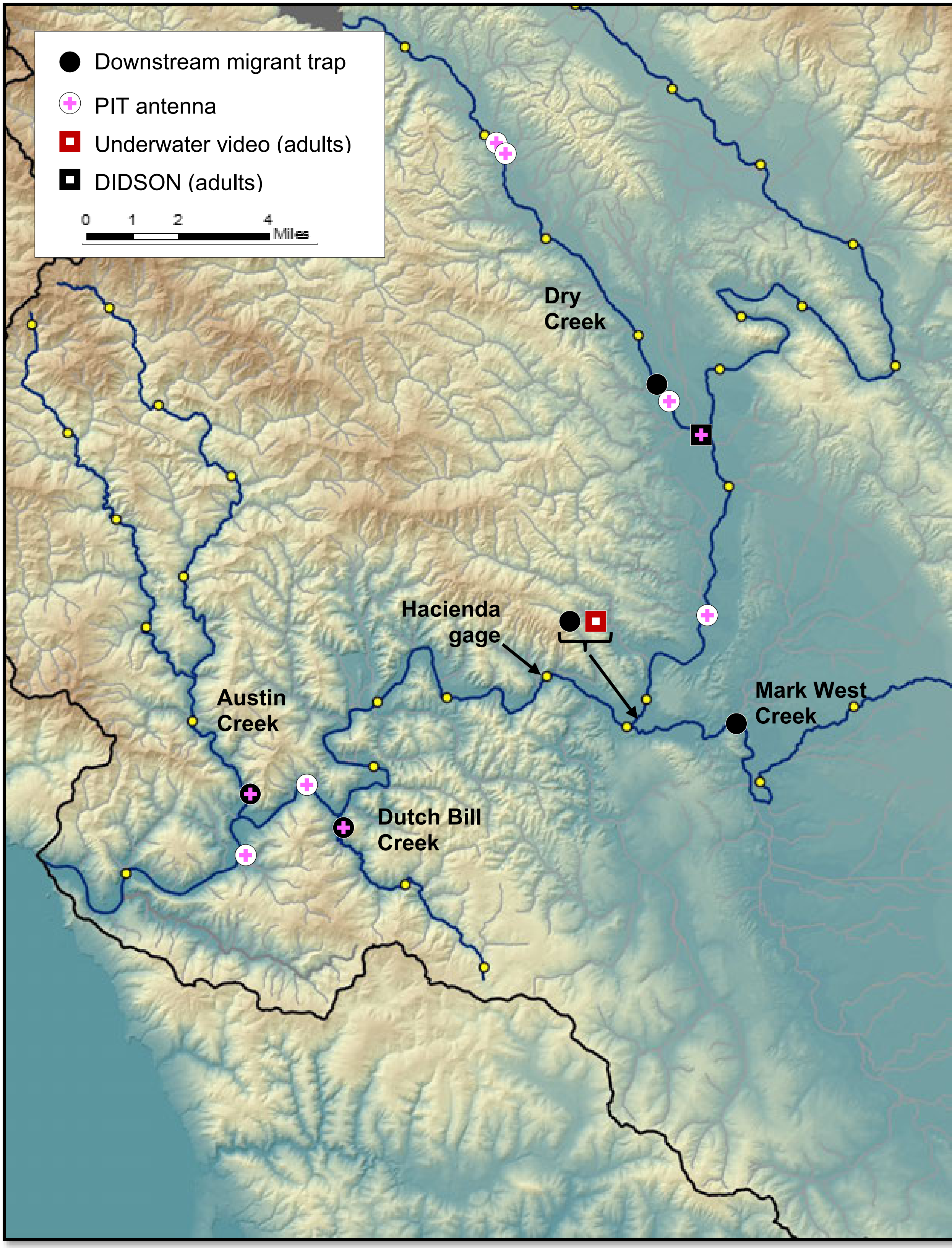

Figure 10.1. Location of fixed fisheries monitoring locations related to the Russian River Biological Opinion, 2016. Yellow dots represent 5 km increments along the stream course. PIT antenna and downstream migrant trapping sites operated by UC/California Sea Grant are not shown.

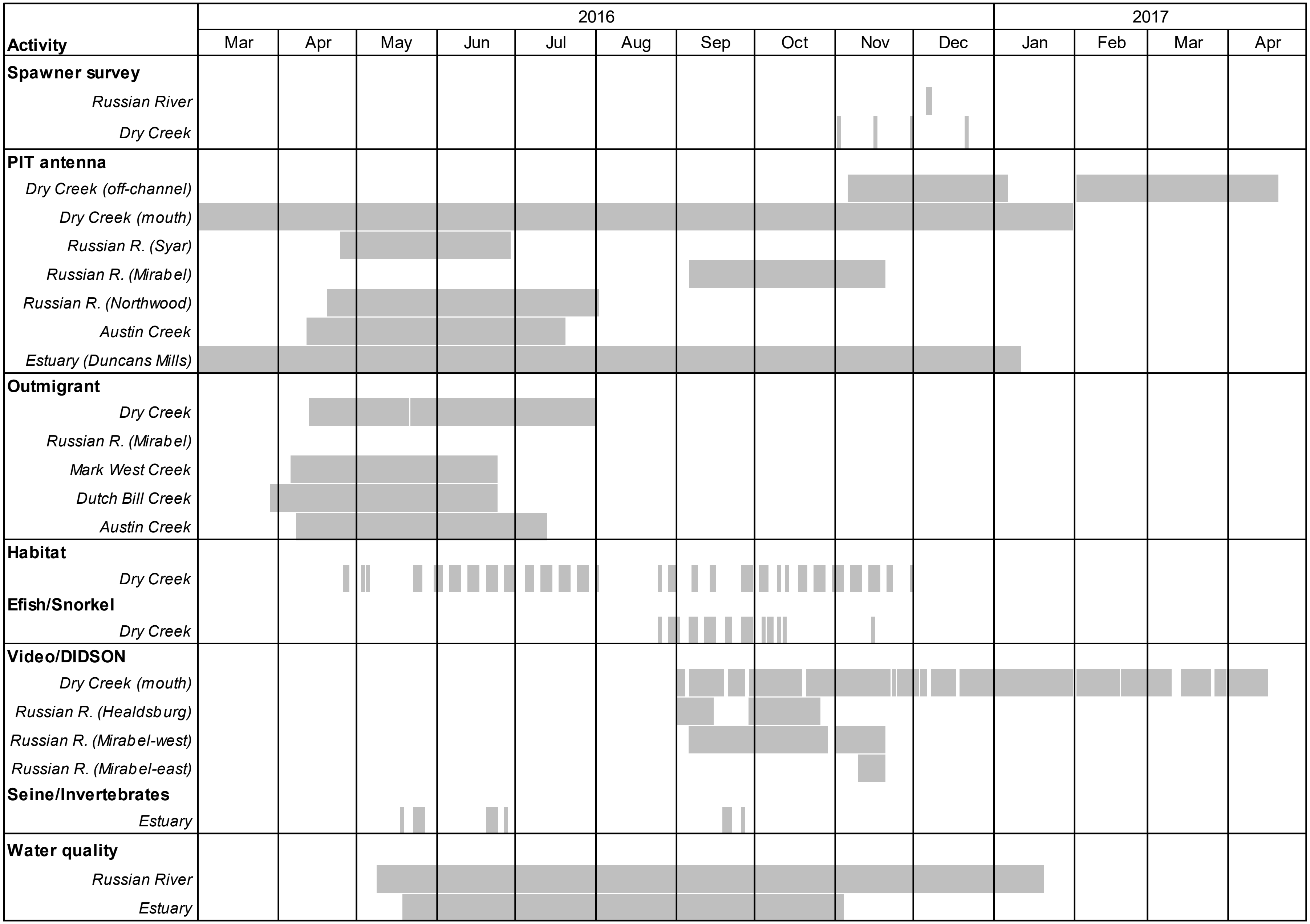

Figure 10.2. Temporal extent of sampling at fisheries and water quality monitoring sites related to the Russian River Biological Opinion, Spring 2016-Winter 2017.

In the sections that follow, we summarize population and movement dynamics of juvenile and smolt salmonids based on data from tributary and mainstem sites sampled in 2016. The Water Agency used PIT tags and fin-clipping as primary tools for characterizing population attributes such as growth and movement. As described in other sections of this report and reports from prior years, PIT-tagged fish were detected during sampling by beach seine in the estuary and at downstream migrant traps and stationary PIT antennas located throughout the system (Figure 10.1). In the first section below, we broadly summarize available indicators of abundance to describe some general temporal trends and variability in abundance. Following that, we focus specifically on movement of juvenile steelhead and Chinook salmon smolts from Dry Creek through the lower mainstem Russian River and estuary and the abiotic impacts of mainstem Russian River on juvenile and smolt survival.

Abundance

Combined juvenile steelhead downstream migrant trap (DSMT) catch at Dry Creek, Dutch Bill Creek and Austin Creek was higher in 2016 as compared to 2014 and 2015. The increase was most pronounced for Austin Creek (Figure 10.3) and this was, at least in part, due to higher springtime flow conditions in Austin which may have also led to higher juvenile steelhead production. Average juvenile steelhead density from backpack electrofishing on mainstem Dry Creek decreased relative to recent years, the Dry Creek Chinook smolt estimate showed only a slight increase over 2015 and captures of wild coho smolts were low everywhere (Figure 10.4). Due to construction of a new fish ladder and fish screens at Mirabel, the Mirabel smolt trap was not operated in 2016. Relative to 2014, adult returns increased for steelhead at both Russian River hatcheries. The minimum count of adult Chinook was approximately 600 fish lower than the long-term average while the number of hatchery coho increased significantly (Figure 10.5).

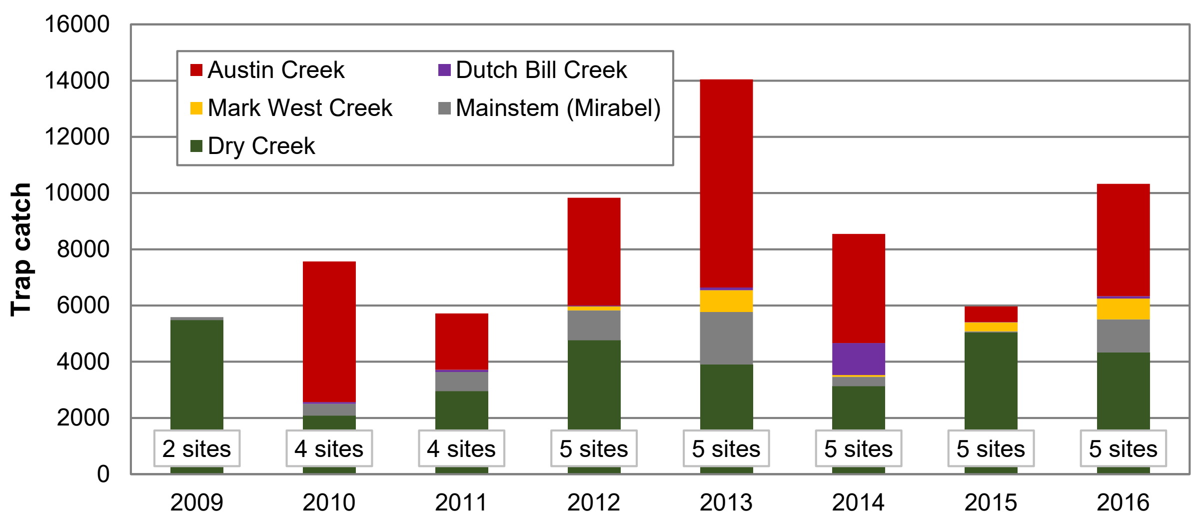

Figure 10.3. Number of juvenile (YOY + smolt combined) steelhead captured at downstream migrant trap sites operated by the Water Agency, 2009-2016 Note that downstream migrant trapping on the mainstem at Mirabel dam was suspended in 2016 due to construction of a new fish ladder.

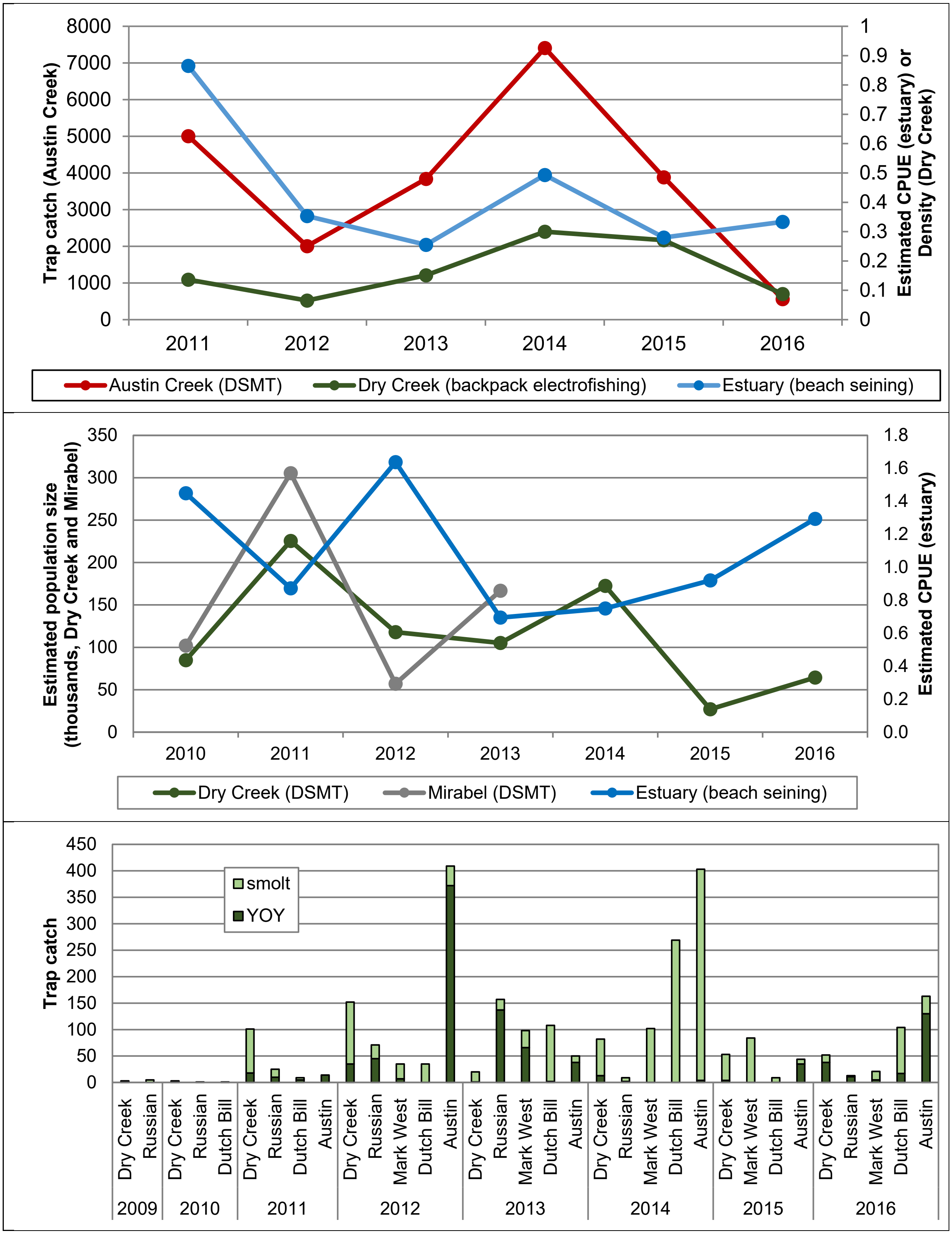

Figure 10.4. Indicators of juvenile steelhead (top panel), Chinook smolts (middle panel) and wild juvenile coho (lower panel) trends based on monitoring conducted by the Water Agency, 2009-2016.

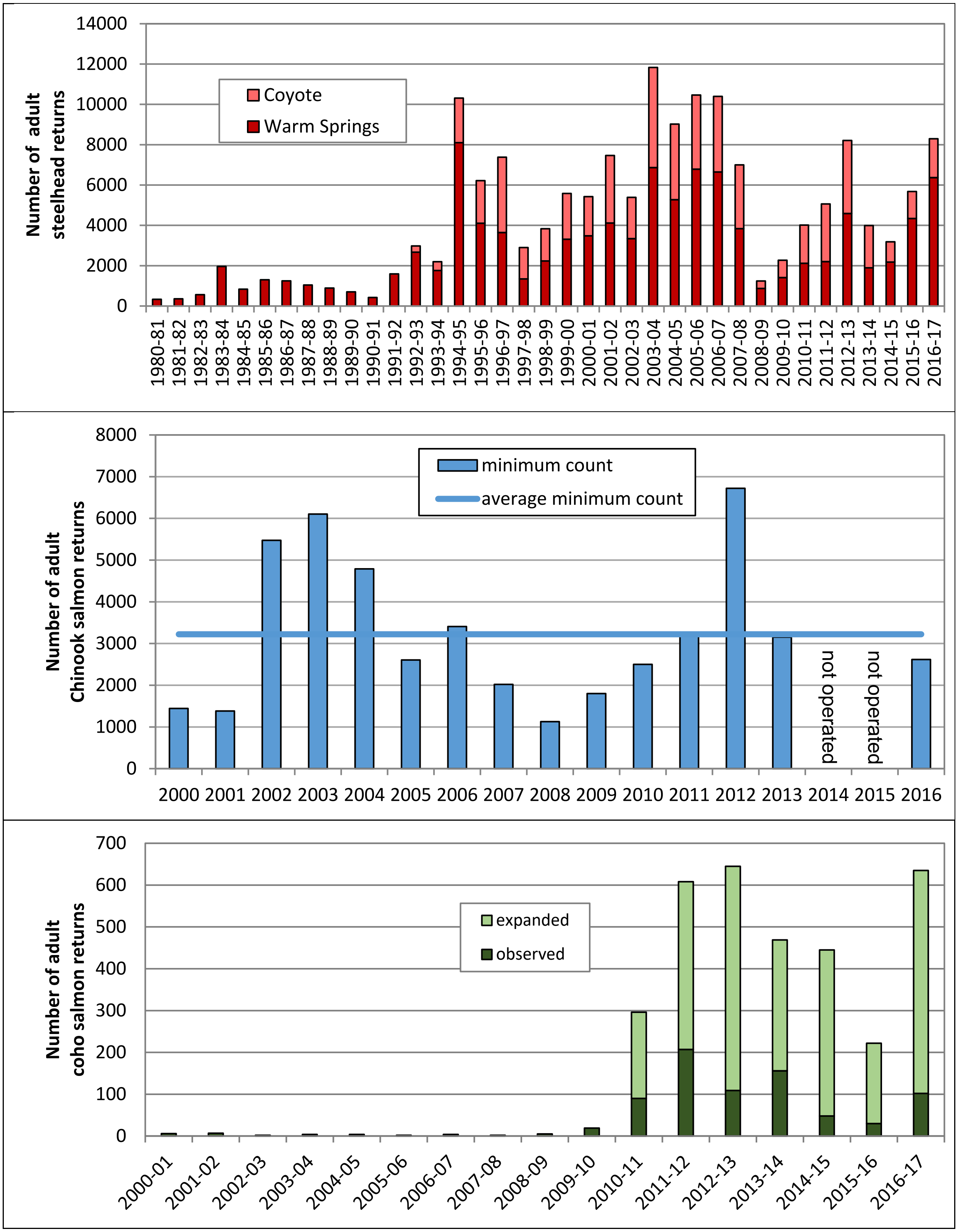

Figure 10.5. Indicators of adult steelhead (counted at Russian River hatcheries), adult Chinook (based on video-DIDSON counts at Wohler-Mirabel) and hatchery coho salmon returns (CA Sea Grant).

Juvenile Steelhead and Chinook Smolt Movement and Survival

Data from PIT-tagged fish and PIT antenna detections have been summarized in other chapters of this year's data report as well as data reports from other years. Collectively, these data have facilitated addressing some of the more salient points of the RPA including the following:

- Movement of juvenile steelhead out of Austin Creek and into the Russian River estuary is substantial with as many as 80% of fish PIT-tagged at the Austin Creek downstream migrant trap leaving Austin Creek each year (an estimated annual average of approximately 6,400 fish). Along with the very low numbers of fish detected leaving other lower river tributaries where downstream migrant trapping is conducted (i.e., Mark West, Green Valley, Dutch Bill and Willow Creeks), this finding strongly suggests that Austin Creek is a major contributor to the population of juvenile steelhead residing in the Russian River estuary.
- Although a substantial portion of the population of juvenile steelhead PIT-tagged at the Dry Creek downstream migrant move out of Dry Creek (80%), very few are detected at detection locations outside of Dry Creek (<1%).
- Juvenile steelhead PIT-tagged in mainstem Dry Creek use constructed off-channel habitat in the winter at a relatively high rate with approximately 20% of the fish PIT-tagged in close proximity to these features detected one or more times on PIT tag antennas at the entry/exit to a given constructed off-channel site.
- Individual growth rates of juvenile steelhead during the summer in Dry Creek and the Russian River estuary are consistently high averaging from approximately 0.4 mm/day in Dry Creek to well over 1 mm/day in the estuary.

If we are to understand the benefits of RPA implementation to salmonid populations in the Russian River, it is important to also understand the broader context that is outside the area of influence of the RPA that is also influencing these populations. The Water Agency is beginning to amass enough data to consider multiple lines of evidence in order to broadly evaluate which of those factors are significant. Between 2011 and 2016 we PIT-tagged 5,795 juvenile steelhead at the Austin Creek downstream migrant trap, 497 juvenile steelhead during beach seining in the estuary, 6,679 juvenile steelhead while backpack electrofishing in mainstem Dry Creek and 4,062 juvenile steelhead at the Dry Creek downstream migrant trap (Table 10.1). During that same time period, we also PIT-tagged 15,667 Chinook salmon smolts at the Dry Creek downstream migrant trap (Table 10.2).

Table 10.1. Number of juvenile steelhead that were PIT-tagged and observed with a PIT tag at all Water Agency fish capture sites, 2009-2016.

Tributary	Survey	Year	Applied	Observed
Dry Creek	Downstream migrant trap	2009	0	2
		2010	10	2
		2011	0	3
		2012	0	2
		2013	2,708	59
		2014	1,354	36
		2015	0	3
		2016	0	2
	Backpack electrofishing	2009	736	91
		2010	895	168
		2011	865	141
		2012	774	202
		2013	897	213
		2014	997	227
		2015	1,673	237
		2016	1,473	212
Mainstem	Downstream migrant trap	2009	17	0
		2010	99	51
		2011	99	1
		2012	327	3
		2013	505	37
		2014	102	7
		2015	not fished	
		2016	not fished	
Mark West Creek	Downstream migrant trap	2012	43	0
		2013	135	11
		2014	18	0
		2015	19	1
		2016	548	26
Dutch Bill Creek	Downstream migrant trap	2010	47	0
		2011	23	1
		2012	6	0
		2013	12	0
		2014	21	0
		2015	7	0
		2016	46	0

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Tributary	Survey	Year	Applied	Observed
Austin Creek	Downstream migrant trap	2010	1007	116
		2011	503	30
		2012	1,637	569
		2013	1,749	10
		2014	594	22
		2015	107	1
		2016	1,205	137
Estuary	Beach seining	2009	60	4
		2010	239	41
		2011	88	18
		2012	84	14
		2013	43	4
		2014	176	29
		2015	87	2
		2016	19	2
		Total	***22,252***	***2,746***

Table 10.2. Number of Chinook salmon smolts that were PIT-tagged and observed with a PIT tag at all Water Agency fish capture sites, 2011-2016.

Tributary	Survey	Year	Applied	Observed
Dry Creek	Downstream migrant trap	2011	1,849	242
		2012	1,326	110
		2013	3,677	439
		2014	4,775	641
		2015	1,369	278
		2016	2,671	525
Mainstem	Downstream migrant trap	2011	0	45
		2012	0	36
		2013	0	202
		2014	772	259
		2015	not fished	
		2016	not fished	
Estuary	Beach seining	2011	0	1
		2012	0	4
		2013	0	4
		2014	0	7
		2015	0	3
		2016	0	0
		Total	***16,439***	***2,776***

One important conclusion we can draw from our PIT-tagging efforts is that the lower 40 km of mainstem Russian River from Dry Creek to the estuary presents significant abiotic challenges to juvenile and smolt salmonid survival. Based on Chinook smolt migration patterns from Dry Creek (river km 52.0) to the Duncans Mills PIT antenna array (River km 10.6, Figure 10.1), data suggest that smolt mortality may be quite significant with levels for Chinook smolts exceeding 40% in some reaches (Manning and Martini-Lamb 2013). For the 13,818 Chinook PIT-tagged between 2012 and 2016 at the Dry Creek downstream migrant trap, we calculated the proportion that were detected at Duncans Mills and plotted that proportion as a function of water temperature at the lower mainstem USGS gage at Hacienda (Figure 10.6). The result showed a decided, monotonic and steep decrease in proportion detected when mainstem temperatures exceeded 19°C. Though mortality for coho smolts may differ, we expect that because of similarities in migration timing to Chinook smolts and the overall lower temperature tolerances of coho (Figure 10.7), the issue may be at least as important for Russian River coho populations.

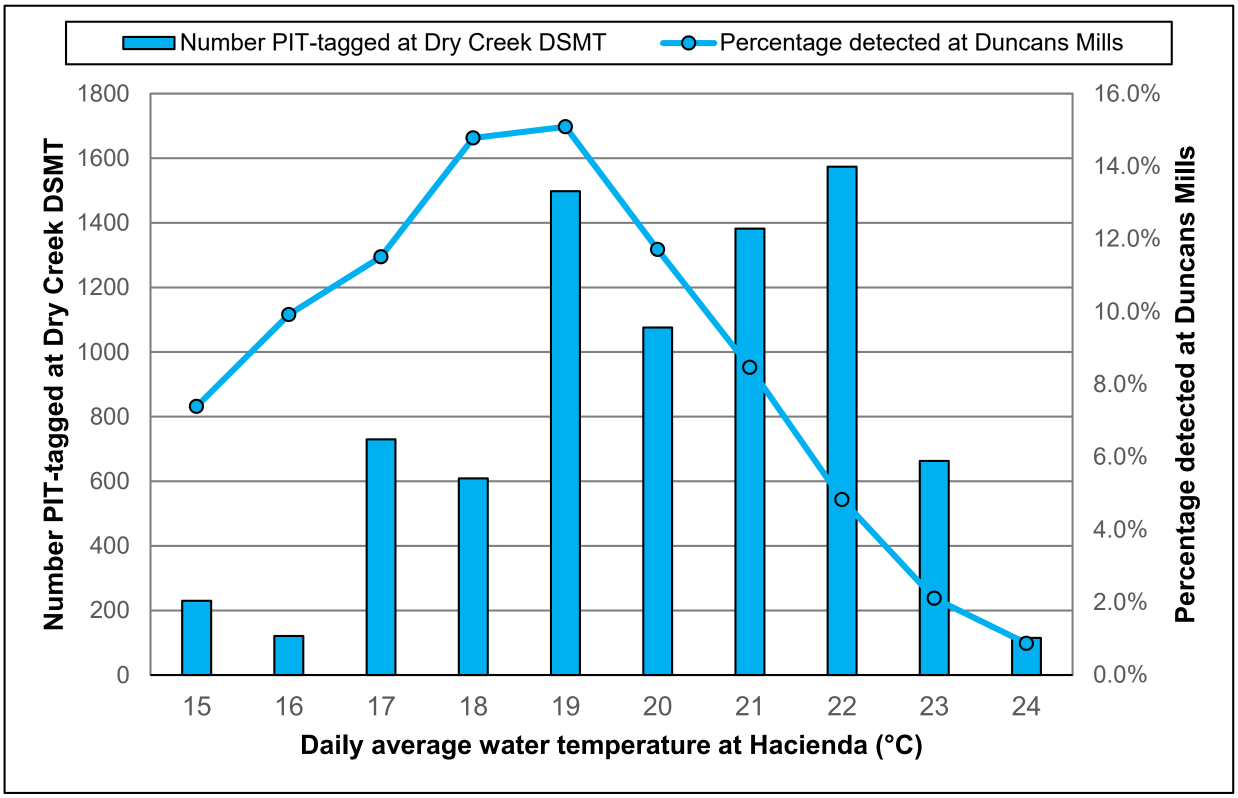

Figure 10.6. Number of Chinook salmon smolts PIT-tagged at the Dry Creek downstream migrant trap and percentage of those fish detected at the Duncans Mills PIT antenna array as a function of 1°C water temperature bins at Hacienda (USGS gage number 11467000, Figure 10.1). Data from 2012-2016 are combined.

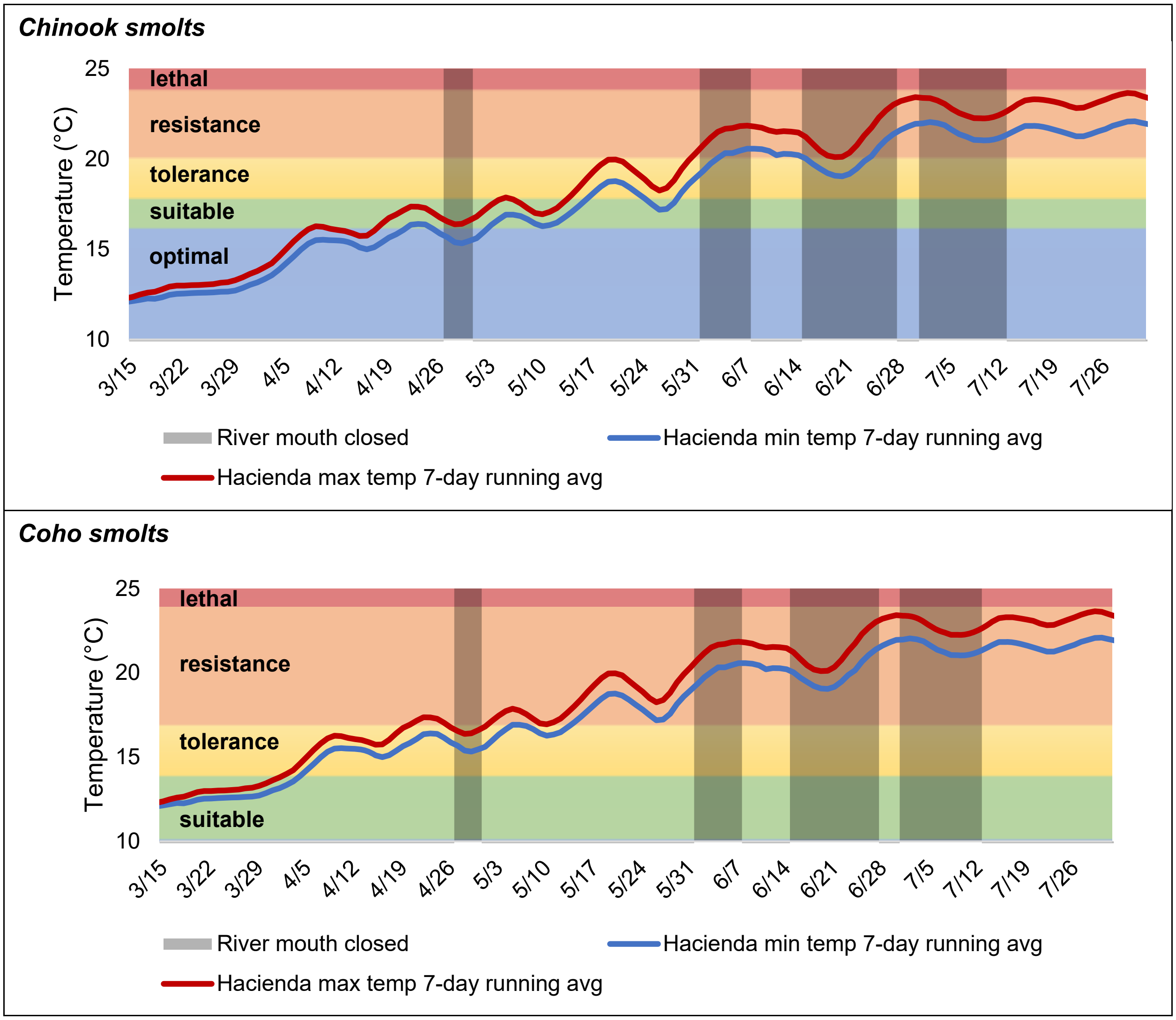

Figure 10.7. Water temperature at Hacienda (USGS gage number 11467000, Figure 10.1) and mouth closure periods, 2016. Temperature bins are from Sonoma County Water Agency (2016).

Further exacerbating the issues associated with water temperatures in mainstem Russian River for fish produced from Dry Creek are differences in water temperature between mainstem Dry Creek and mainstem Russian River. Average daily water temperatures in Dry Creek are at least 3-4°C cooler than the Russian during May through June when the bulk of salmon migration occurs. Consequently, mean size of Chinook smolts captured in the Dry Creek downstream migrant trap is approximately 4 mm smaller than fish captured in the mainstem Russian River downstream migrant trap at Mirabel. This is reflected in differences in size distribution at the two sites with 42% of the Dry Creek trap catch between 60 and 80 mm fork length as compared to only 30% at the Mirabel trap. Although median travel time of Chinook smolts through the 45 river km from the Dry Creek trap to Duncans Mills is 5 days, the median travel time is 9 days for fish between 60 and 80 mm (Figure 10.8). We suggest that this 12% difference in the size distributions between Dry Creek and the mainstem translates into more fish from Dry Creek being subjected to inhospitable mainstem Russian River environmental conditions for a longer period of time than fish produced in the mainstem Russian.

We are also beginning to gain insights into the effects of mainstem Russian River conditions on survival of steelhead young-of-the-year to the adult return stage. Based on 4,070 juveniles PIT-tagged at the Dry Creek downstream migrant trap, only 14 were were detected at the Duncans Mills PIT antenna as juveniles (or possible smolts) despite an estimated 80% (3,200) leaving Dry Creek within a few days of being PIT-tagged in the spring. Of those 4,070 PIT-tagged juveniles, only 1 (0.02%) was detected returning as an adult on the Duncans Mills PIT antenna. This is in contrast to the 30 adult steelhead returns detected at Duncans Mills out of the 5,795 juveniles PIT-tagged at the Austin Creek downstream migrant trap (0.5%). We suspect that this 95% difference in proportions may be at least partially attributable to high mortality of juveniles from marginal water quality conditions in mainstem Russian River.

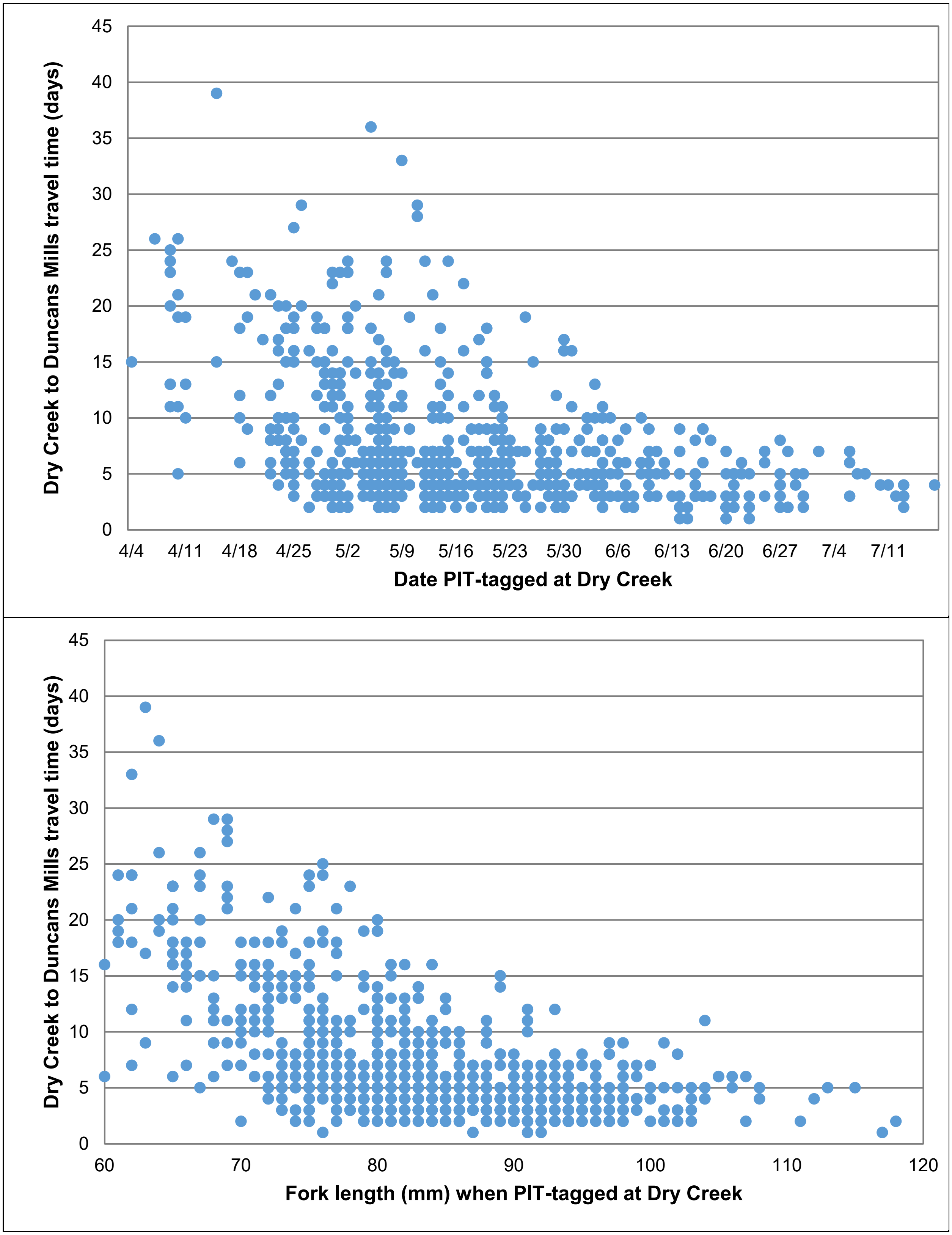

Figure 10.8. Travel time of individual Chinook salmon smolts from the Dry Creek downstream migrant trap to Duncans Mills as a function of date PIT-tagged (upper panel) and size when PIT-tagged at the Dry Creek downstream migrant trap. Data from 2012-2016 are combined.

Conclusions and Recommendations

Fish monitoring data collected in the lower mainstem Russian River and its tributaries from Dry Creek to Austin Creek illustrate a clear need to focus on the spatial and temporal context that is outside the influence of the RPA. While summer water flow in Dry Creek provides a steady supply of high quality water, the position of the Dry Creek confluence approximately 52 km from the ocean means that both juveniles and smolts that leave Dry Creek in late- or even mid-spring can be presented with serious challenges to survival. Many tributaries to the Russian River have the problem of being flow-impaired to the point where, even during the peak smolt migration in early-mid May, fish become trapped in pools that eventually dry completely. These issues were particularly apparent in recent years when drought conditions prevailed and fish either had to move to mainstem Russian or perish, but even in non-drought years natural and anthropogenic aggradation of stream channels can lead to similar albeit less severe issues.

If anadromous salmonid populations in the Russian River watershed are to persist, it is vitally important that connectivity between a diversity of habitat types be maintained and improved. Increasing evidence from PIT antenna arrays maintained by the Water Agency and California Sea Grant suggest that coho and steelhead are using habitats in surprising ways which leads us to conclude that life history diversity is being supported by the opportunities afforded by these connections. As we move forward, the combined efforts of entities conducting fisheries monitoring throughout the watershed and the data collected through RPA implementation, the Coastal Monitoring Program and Russian River Coho Salmon Broodstock Program monitoring will continue to be necessary for progress toward recovery.

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