Interagency Ecological Program for the San Francisco Estuary

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Gonzalo Castillo (USFWS), gonzalo_castillo@fws.gov, Matthew Dekar (USFWS), matthew_dekar@fws.gov, Lori Smith (USFWS), lori_smith@fws.gov

Following three years of dry conditions, this issue provides important information for understanding drought impacts on the San Francisco Estuary. This issue includes three contributed papers examining food web dynamics and fish abundance in the Yolo Bypass. Status and trends updates are provided for fish facilities and the Delta Smelt refuge population.

Jared Frantzich and **Ted Sommer** (DWR) characterized phytoplankton blooms observed in 2011 and 2012 in the lower estuary in connection with fall rice-field drainage flows in the Toe Drain of the Yolo Bypass. The authors further evaluated the variability of abiotic and biotic variables within the Toe Drain of the Yolo Bypass associated with rice-field drainage through 2013. Flow pulses from late summer and early fall in the Toe Drain matched the timing of the rice-field drainage, with the estimated discharge volume in 2011 and 2012 being two- to three-times greater than previously recorded. Water temperatures and conductivity were higher in the Yolo Bypass compared with the Sacramento River at Sherwood Harbor. Dissolved oxygen concentrations decreased in the summer in both regions, but concentrations appeared to be related to increased flow and chlorophyll *a* in the Yolo Bypass in 2011 and 2012. In all years, chlorophyll *a* concentrations and zooplankton densities were greater in the Yolo Bypass where flow was a significant predictor of adult calanoid copepod abundance. Results indicated that phytoplankton blooms in the fall of 2011 and 2012 in the Yolo Bypass contributed significantly to the estuarine food web.

Brian Mahardja (DWR) and colleagues examined the potential benefits of floodplain habitat to Delta Smelt rearing in the San Francisco Estuary. The Yolo Bypass Fish Monitoring Program data from 1998 to 2014 was summarized and contrasted with Delta Smelt data from other IEP fish monitoring efforts. Results demonstrated a long-term increase in Delta Smelt densities in the Yolo Bypass and suggested that Delta Smelt may have higher growth rates or earlier spawning times in the bypass relative to other regions in the Estuary. Additionally, the authors observed relatively high densities of Delta Smelt within the floodplain during recent drought years, indicating the benefits of the bypass extend beyond wet periods when the bypass is flooded.

In the third article on floodplain habitat, **Pascale Goertler** (DWR) and colleagues compared the relative abundance of juvenile Chinook Salmon, prey resources, and abiotic conditions in the Yolo Bypass to Sherwood Harbor on the Sacramento River from 2011 through 2014. Despite drought conditions, the density of prey resources were generally higher in the Yolo Bypass, with peak densities observed in 2013. The relatively high catches of juvenile Chinook Salmon in the Yolo Bypass in 2014 further suggested the potential benefit of the bypass during dry conditions.

Geir Aasen (DFW) provided the 2014 Water Year fish salvage report for the State Water Project and Central Valley Project. Coinciding with record low exports, the Skinner Delta Fish Protective Facility and the Tracy Fish Collection Facility reported record low total fish salvage (all fish species combined) with 236,846 and 160,681 individuals salvaged, respectively. Record and near-record low salvage was reported for species of management concern, including Delta Smelt, Longfin Smelt, Chinook Salmon, and Steelhead.

Finally, **Tewdros Ghebremariam** (UCD) and colleagues provided an update for the Delta Smelt refuge population held in 2014 at the Fish Conservation and Culture Lab in Byron, CA. The Delta Smelt captive breeding program is primarily focused on maintaining a viable population that is genetically and phenotypically identical to the wild population. Both genotypic and phenotypic (body weight, fork length, and egg number) differences between and within the cultured and wild Delta Smelt populations were evaluated. Genetic analysis based on neutral markers indicated successful maintenance of genetic diversity and minimization of inbreeding in the Delta Smelt refuge population, with low differentiation between the founding generation and the sixth generation. Initial results of the three phenotypic traits studied over five generations suggested no significant departure between cultured and wild fish.

Contributed Papers

Yolo Bypass as a fall food web subsidy for the Lower Estuary

Jared Frantzich (DWR), jared.frantzich@water.ca.gov,
Ted Sommer (DWR), ted.sommer@water.ca.gov

Introduction

Since 1998, the California Department of Water Resources (DWR) with support from the Interagency Ecological Program (IEP), has carried out a comprehensive monitoring effort in the Yolo Bypass to investigate the importance of floodplain habitat on lower trophic organisms and their implications for fishes, as part of the Yolo Bypass Fish Monitoring Program (YBFMP). Much of this past research has been focused on the importance of the Yolo Bypass during the winter and spring, when floodplain inundation is likely to occur (Sommer et al. 2001a, 2004; Schemel et al. 2004; Lehman 2007), but less is known about the role the Yolo Bypass may play during the drier months of summer and fall. In 2011, DWR began monitoring a suite of lower trophic metrics as part of a Yolo Bypass study, funded by the Ecosystem Restoration Project (ERP), to determine the contributions of future Yolo Bypass restoration efforts to the estuarine food web year-round.

This article describes the significance of phytoplankton blooms observed in 2011 and 2012 in the lower estuary as a likely result of increased fall rice-field drainage flows in the Toe Drain of the Yolo Bypass. In addition, we provide results from the multi-agency sampling effort in summer and fall of 2013, to investigate the spatial variability of: (1) water quality conditions, (2) flow conditions, (3) chlorophyll *a*, (4) phytoplankton species density and composition, (5) nutrient sources and concentrations, and (6) zooplankton density and composition within the Toe Drain of the Yolo Bypass pre- and post-rice field drainage.

Study Background

In 2011 and 2012, several Fall Low Salinity Habitat (FLaSH) studies observed a phytoplankton bloom in the lower Sacramento River shortly after a seasonal agricultural flow pulse passed through the Yolo Bypass. The lower Sacramento River had not experienced a fall phytoplankton bloom of this intensity for over 20 years ("Pulse of the Delta," 2012). Chlorophyll in relative fluorescence units (RFU) recorded at the Sacramento at Rio Vista Bridge (RVB) station (Figure 1) provided substantial evidence of a fall phytoplankton

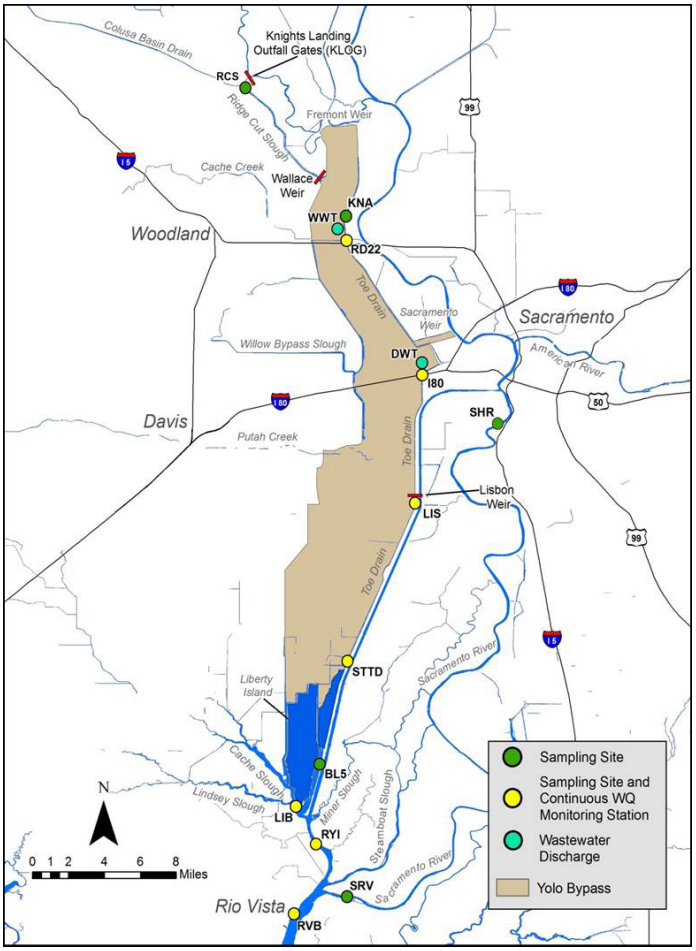

Figure 1 Map of the Yolo Bypass water sampling stations, wastewater treatment discharge sampling locations, and the sampling locations with continuous water quality monitoring. Sites include Sherwood Harbor (SHR), Ridge Cut Slough at Hwy 113 (RCS), Toe Drain at Knaggs Ranch (KNA), Toe Drain at Road 22 (RD22), Toe Drain at I80 (I80), Toe Drain below Lisbon Weir (LIS), Screw Trap at Toe Drain (STTD), Prospect Slough (BL5), Liberty Island (LIB), Cache Slough at Ryer Island (RYI), Sacramento River at Vieira's Marina (SRV), and Sacramento River at Rio Vista (RVB).

bloom occurring in the lower Sacramento River in both years (Figure 2a, 2b). The source of this increased phytoplankton biomass appeared to be from the Cache Slough Complex (CSC), as the Sacramento River at Hood (SRH) station did not have elevated levels of chlorophyll (< 3 RFU). Moreover, isotopic studies indicated that the bloom came largely as a result of contributions from the CSC, of which Yolo Bypass is a part (Kendall 2012 Interagency Ecological Program [IEP] Workshop oral presentation). The YBFMP year-round monitoring efforts in 2011 and 2012 observed increased chlorophyll *a* concentrations within the Toe Drain coinciding with increased flows from rice-field drainage in the fall of both years. In response to the observed bloom events, DWR, through a multi-agency collaboration with the United States Bureau of Reclamation (USBR), University of California at Davis (UCD), and the State Water Resources Control Board (SWRCB), initiated a comprehensive study in 2013 to further investigate the mechanisms and drivers in the development and transport of fall phytoplankton blooms from the Yolo Bypass to the lower San Francisco Estuary.

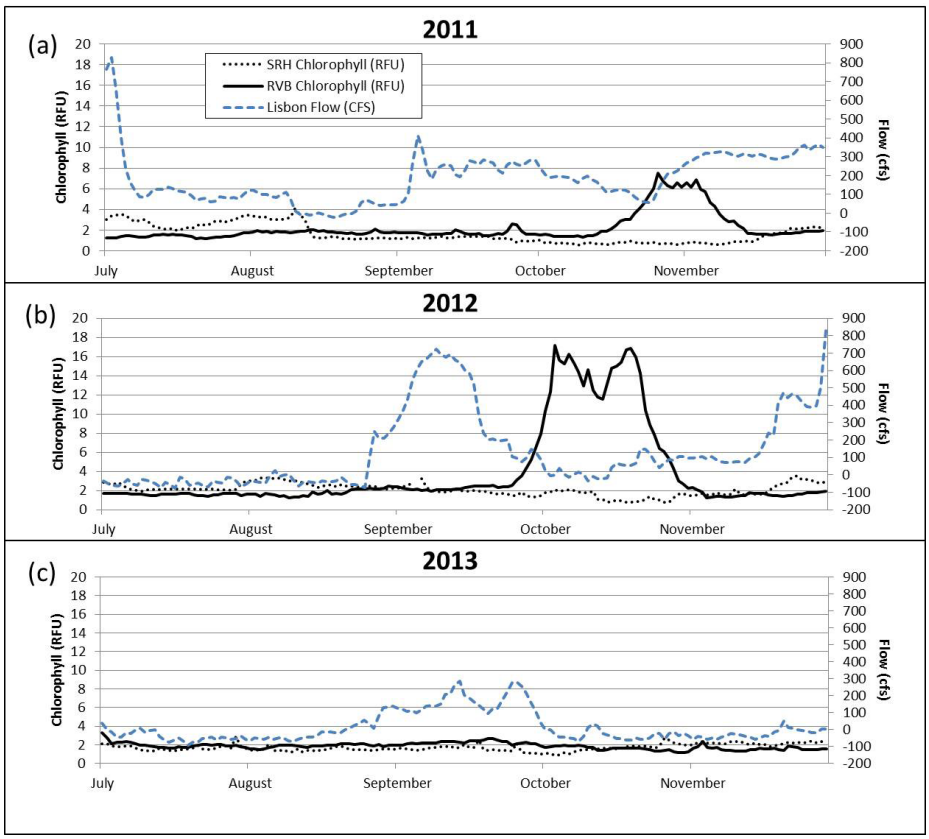

Figure 2 2011–2013 Daily average continuous chlorophyll (RFU) from Sacramento River at Hood (SRH), Sacramento River at Rio Vista Bridge (RVB) and Lisbon (LIS) flow (cfs).

Study Area

The Yolo Bypass is the primary floodplain (approximately 61 km long and 24,000 ha) of the San Francisco Estuary, and was engineered as a flood control system to convey up to 80% of Sacramento River basin flow during high water events (Sommer et al. 2001a, 2001b, 2004; Schemel et al. 2004). During the drier months of summer and early fall, connectivity of the Bypass to the San Francisco Estuary is maintained through the perennial riparian channel called the Toe Drain. The Toe Drain is a narrow (≤ 50 m wide) and shallow (≤ 5 m deep) linear channel, which flows along the east side of the leveed floodplain (Figure 1). During the summer and fall, the floodplain of the Yolo Bypass is heavily used for agriculture, of which a substantial portion is rice production. The downstream flow within the Toe Drain during this time period is maintained mostly by upstream inputs from agricultural discharges from the upper Colusa Basin Drain, and those flows are controlled by the DWR-operated Knights Landing Outfall Gates (KLOG) and further downstream through the local landowner-operated Wallace Weir in Ridge Cut Slough (Figure 1). There are additional sources of salinity and nutrients into the Toe Drain of the Yolo Bypass, including side tributary inputs (such as Putah Creek and Cache Creek) and wastewater treatment drainage (Woodland and Davis) (Figure 1). The Lisbon Weir, located in the tidal portion of the Toe Drain further downstream, is a permanent rock barrier that mutes tidal influence upstream (Figure 1). It assists with maintaining water levels for upstream water diversions used for both agriculture and the Yolo Wildlife Area. In most years, during the summer and fall, the Toe Drain below Lisbon Weir has extended periods of net negative outflow as a result of these local land use practices.

During August and early September every year, the northern Central Valley's estimated 500,000 acres of irrigated and flooded rice fields are drained for harvest. Much of this drainage water has two fates: (1) to enter the Sacramento River at Knight's Landing through the KLOG, or (2) to flow through Ridge Cut Slough and down into the Toe Drain of the Yolo Bypass and eventually into the greater CSC. Local rice fields within the leveed Yolo Bypass also begin draining fields in September and October, adding to the overall discharge volume. During these drainage events, flow throughout the Toe Drain of the Yolo Bypass can increase substantially, as recorded at the Lisbon Weir flow gage (LIS, Figure 1) each September since its installation in 2005.

Methods

In 2011, 2012, and 2013, DWR collected lower trophic data (e.g., chlorophyll *a* and zooplankton) year-round in the Yolo Bypass at the Screw Trap in the Toe Drain (STTD) and in the Sacramento River at Sherwood Harbor (SHR) (Figure 1). The collection of water samples for the spectrophotometric determination (Standard Method 10200 H-APHA et al. 2012) of chlorophyll *a* concentration was used as an indicator of phytoplankton biomass. The collection of biweekly zooplankton samples was achieved through the use of conical plankton nets (0.50 m mouth, 2 m length and 153 µm mesh) sampled for approximately 5 minutes during the mid to late morning on an ebb tide. Sample volume was estimated from flow measurements using General Oceanics Model 2030R flow meters. In 2013, additional water samples were collected for chlorophyll *a*, phytoplankton species composition and enumeration, and nutrient concentration. These additional samples were collected from late July through October at 11 sites in a north to south transect along the eastern edge of the Yolo Bypass to the Sacramento River at Rio Vista (Figure 1).

Flow, velocity, and stage measurements in the Yolo Bypass were obtained from gauges operated by DWR below Lisbon Weir (LIS; http://cdec.water.ca.gov/) and Sacramento River flow measurements from the USGS operated gauge at the Freeport Bridge (FPT; http://cdec.water.ca.gov/). The estimated volume of fall rice-field discharge in the Toe Drain of the Yolo Bypass was calculated for each year using the equation: discharge = area x velocity. The channel area was calculated using cross-sectional depth data from an acoustic doppler current profiler (ADCP) attached to a boat operated by DWR. The velocity data was obtained from the LIS gauging station. Continuous water temperature data for the sites in the Yolo Bypass (STTD) and Sacramento River (SHR) were collected using continuous onset temperature recorders. Additional discrete measurements were collected at all study sites for water temperature, electrical conductivity (µS/cm), pH, dissolved oxygen (mg/L), turbidity (Nephelometric Turbidity Units), and Secchi depth (m).

In March 2013, DWR installed and telemetered a Yellow Springs Instruments (YSI) 6600 multi-parameter water quality sonde at the LIS flow gauge to provide continuous data on water temperature, specific conductance, pH, dissolved oxygen, turbidity, and chlorophyll. In the summer of 2013, three YSI 6600 water quality sondes continuously measuring water temperature, specific conductance, and chlorophyll, were temporarily installed during July through November at three sites: (1) RD22, (2) I-80, and (3) STTD (Figure 1). The continuous chlorophyll RFU data along with lab-analyzed chlorophyll *a* was used to develop a regression model and ultimately to create a continuous estimated chlorophyll *a* (µg/L) value for all sites. In addition, continuous water quality data from both the DWR-operated SRH and RVB stations were used to determine changes in ambient water conditions downstream of CSC and in the upper reaches of the Sacramento River (Figure 2a, 2b, and 2c).

Statistical Analyses

For all years, a Mann-Whitney test was used to compare the concentrations of chlorophyll *a* and zooplankton between the Yolo Bypass (STTD) and Sacramento River (SHR). The effect of flow in the Yolo Bypass and Sacramento River on biological data (chlorophyll *a* concentration and zooplankton densities) was evaluated by calculating Pearson correlation coefficients for $Log_{10}(x)$ -transformed data.

For 2013, we used multiple linear regression to determine which physical and chemical parameters were related to the chlorophyll *a* concentrations during increased rice-field drainage flows in the Yolo Bypass. All data were $Log_{10}(x)$ -transformed and independent variables included: water temperature, specific conductance, dissolved oxygen, turbidity, and flow. A

two-sample t-test was used to characterize the significance between any differences in the mean concentration of measured constituents: chlorophyll, electrical conductivity, and water temperature, for above Lisbon Weir (I-80) and below Lisbon Weir (LIS). A Mann-Whitney test was also used to compare the concentrations of nutrients (i.e., ammonium, nitrate, and orthophosphate) at the I-80 (above Lisbon Weir) and LIS (below Lisbon Weir) sites.

Results

Flow

Flow pulses were observed in all years during the late summer and early fall in the Toe Drain of the Yolo Bypass (Figure 3a). These flow pulses were closely tied with the timing of rice-field drainage for fall harvest practices throughout the Central Valley. In August of all years, the Toe Drain below Lisbon Weir observed net negative (e.g., landward or northward) flows and a subsequent switch to net positive flows in late August and early September, with increased rice-field drainage flows (Figure 3a). The total estimated discharge volume in 2011 and 2012 was substantially greater (2–3 fold) than any of the previous years on record (Table 1). In 2012, DWR had to restrict discharge of rice-field water at the KLOG due to gate repairs, and this forced more water through Ridge Cut Slough into the Toe Drain.

Water Quality Monitoring

In all years, the summer and fall water temperatures for the Sacramento River and Yolo Bypass followed typical seasonal trends and closely tracked one another, although the Yolo Bypass consistently maintained a higher daily mean of 20.7 °C as compared to 18.4 °C in the Sacramento River at Sherwood Harbor (Figure3a, 3b). Electrical conductivity was considerably higher in the Yolo Bypass throughout the summer and fall as compared to the Sacramento River. In the Yolo Bypass, the electrical conductivity closely followed the changes in the flow intensity (Figure 3a and 3c). The dissolved oxygen concentrations decreased in both regions during the summer and increased in the late fall. The Sacramento River and Yolo Bypass dissolved oxygen concentrations coincided closely with the warming of waters in the summer (Figures 3b, 3d), but the Yolo Bypass in 2011 and 2012 also had dissolved oxygen concentration levels tied closely to the increases in flow and chlorophyll *a* (Figures 3a, 3d, and 5b); this suggests influence from respiration and primary production. The mean Secchi depth of the Yolo Bypass was considerably lower than the Sacramento River in all years (Figure 3e).

In 2013, continuous data from the I-80 and LIS sites were used in

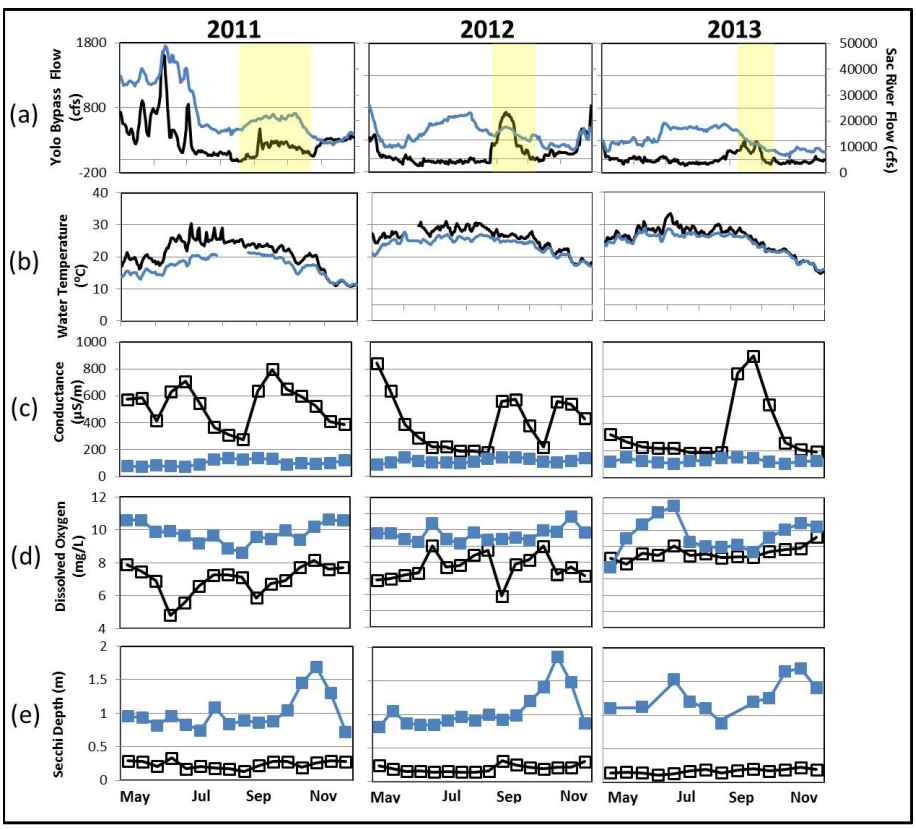

Figure 3 Physical conditions of Yolo Bypass (black line) and Sacramento River (blue line) during the months of May–Nov of 2011, 2012, and 2013. The variables from top to bottom: (a) mean daily flow (cfs); (b) water temperature (°C); (c) conductance (µS/cm); (d) dissolved oxygen (mg/L); (e) Secchi depth (m). Shaded area indicates increased flows in Yolo Bypass due to rice-field drainage.

a two-sample t-test to compare the differences in mean concentrations above and below Lisbon Weir before and after the observed increased rice-field drainage flows. The mean chlorophyll *a*, specific electrical conductivity, and water temperature above Lisbon Weir were all significantly higher than below Lisbon Weir until net positive flows on August 23 were observed at LIS (Table 2). The mean chlorophyll *a* below Lisbon Weir (LIS) continued to increase and maintain significantly higher levels than above Lisbon Weir (I-80) as the flow increased. Further results from a multiple regression model shows evidence that increased flow and specific conductance were good predictors of increased chlorophyll *a* at LIS and as far downstream as STTD (Table 3).

Table 1 Max flow (cfs) at Lisbon ADCP and estimated rice-field discharge volume through the Toe Drain of the Yolo Bypass in 2008–2011.

Year	*Max Flow at Lisbon (cfs)*	*Discharge Volume at Lisbon (ac-ft.)*
2013	284	10,800
2012	724	27,000
2011	400	21,400
2010	423	9,000
2009	183	3,000
2008	111	3,500

Table 2 Comparisons between mean values for three water constituents in relation to net flow from upstream of Lisbon Weir (I80) and downstream of Lisbon Weir (LIS) sampling sites in the Yolo Bypass (average n = 96) for 15 days during 2013.

Date	*Net Positive Flow at LIS*	*Mean Chl a (µg/L)*		*Mean EC (µS/cm)*		*Mean Temp. (°C)*	
		I80	*LIS*	*I80*	*LIS*	*I80*	*LIS*
15-Aug-13	-36	17.0*	13.8	758*	329	25.5*	23.6
19-Aug-13	-17	20.5*	11.5	789*	362	25.5*	24.5
23-Aug-13	+24	15.7*	11.6	914*	558	23.3	22.6
27-Aug-13	+7	13.9*	10.8	883*	592	23.8	22.9
31-Aug-13	+140	16.8	27.3*	913	929	24.2	24.0
4-Sep-13	+110	12.8	29.8*	856	967*	22.6	22.5
5-Sep-13	+100	12.7	30.8*	908	926	22.6	22.1
12-Sep-13	+212	17.4	31.5*	949*	927	22.3	22.4
13-Sep-13	+263	15.0	28.7*	964	957	22.5	22.3
14-Sep-13	+285	12.8	26.1*	920	994*	22.4	22.1
18-Sep-13	+143	11.2	21.8*	936	924	21.0	21.0
22-Sep-13	+126	15.4	17.2	1017	1030	19.2	19.8
26-Sep-13	+283	12.6	16.9*	676	746*	19.3	19.2
30-Sep-13	+106	12.5	14.6*	658	685*	19.6	18.9
4-Oct-13	-25	10.4	15.5*	683	690	16.5	16.5

* Indicates that the mean constituent concentration at I80 and LIS sites are significantly different as determined by a 2-sample t-test (P>0.05)

Chlorophyll a and Phytoplankton

In all years, chlorophyll *a* concentrations were higher in the Yolo Bypass than in the Sacramento River at Sherwood Harbor (Figure 5b). The levels in the Sacramento River were highest during periods of low flow, while the Yolo Bypass experienced its highest levels during the highest flows in the late summer and fall (Figure 5a and b). In August 2013, the continuous-estimated daily mean chlorophyll *a* values at the above Lisbon Weir sites of RD22 and I-80 were 9.8 µg/L and 16.8 µg/L, respectively. Chlorophyll *a* reached a mean daily max of 29.7 µg/L at I-80 on September 8, with a positive daily net flow at LIS of 143 cfs. On September 14, chlorophyll *a* reached a max of 46.4 µg/L, approximately 16 miles downstream at the STTD site, coinciding with a max net flow of 285 cfs at LIS (Figure 6).

The phytoplankton biovolume (µm³/L) during the 2013 bloom event at the LIS station was 90% diatoms, 6% flagellates, 3% green, and the remaining 1% were cyanobacteria and golden-brown. Approximately 50% of the diatom biovolume was made up of *Synedra ulna* (19%), *Thalassiosira* sp. (19%), and *Aulacoseira* sp. (10%).

Nutrients

In 2013, nutrient concentrations in the rice-field drainage water entering the Toe Drain of the Yolo Bypass were much lower than expected. Nutrient concentrations (mg/L) from sites RCS and KNA (Figure 1) ranged from 0.01–0.06 N for dissolved ammonia, 0.01–0.32 N for dissolved nitrate + nitrite, and 0.02–0.34 N for dissolved inorganic nitrogen (Figure 4). Dissolved orthophosphate concentrations (mg/L) ranged from 0.04–0.19 P (Figure 4). The dissolved inorganic nitrogen levels at I-80 were 0.45–1.8 N (mg/L), considerably higher than all the other sites sampled during the study period (Figure 4). Results of Mann-Whitney sign test (P > 0.05) comparing levels of nutrients above Lisbon Weir at I-80 and below Lisbon Weir at LIS showed that dissolved nitrate +

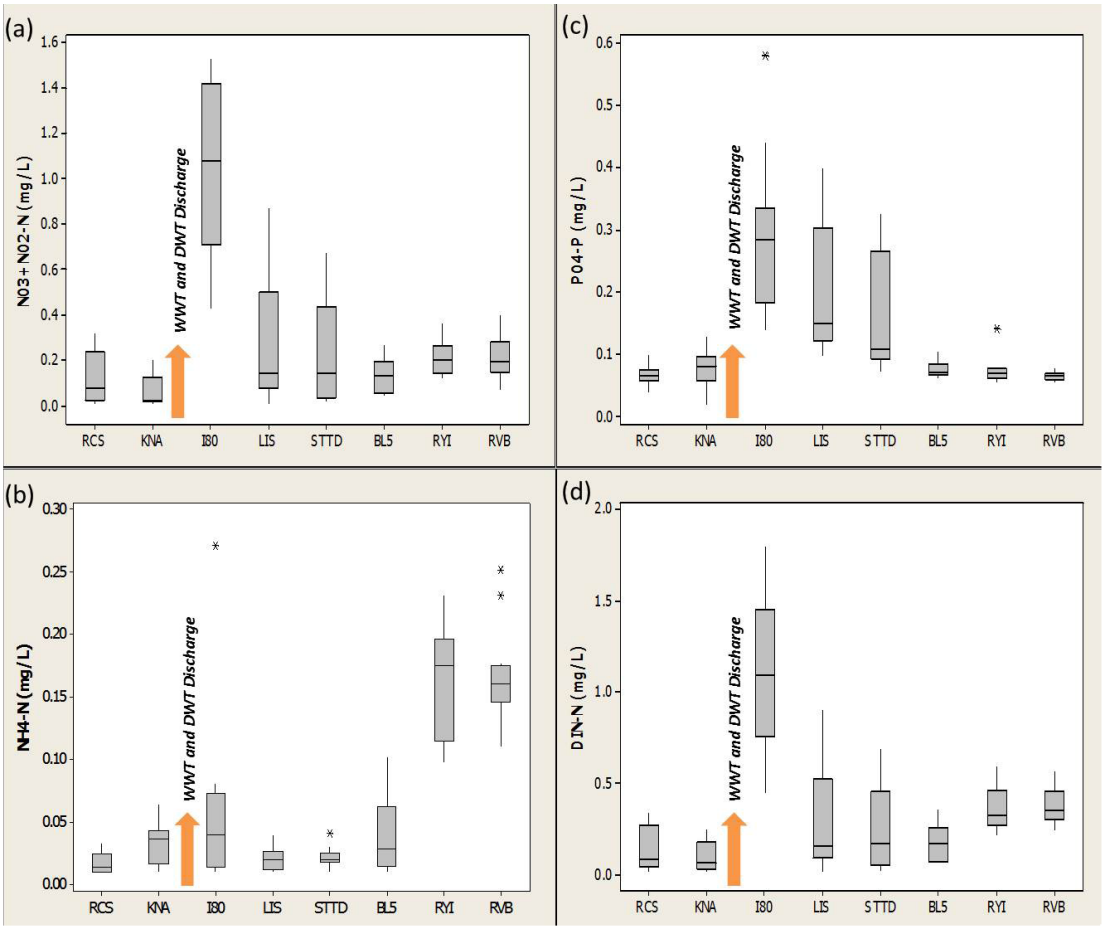

Figure 4 Boxplots displaying the differences in median concentrations of dissolved nitrate+nitrite (a), dissolved ammonia (b), dissolved orthophosphate (c) and the N : P ratios(d) in August–October 2013 (moving north to south in the Yolo Bypass). The orange arrow indicates the location of the wastewater treatment plant discharge at Woodland (WWT) and Davis (DWT).

nitrite and ammonia levels were both significantly higher ($P = 0.01$) above Lisbon Weir at I-80 than below, but the orthophosphate levels were not significantly different ($P = 0.09$).

Zooplankton

In all years, the zooplankton community in the Yolo Bypass was dominated by *Bosmina* (93% of the cladoceran in number/m^3), *Pseudodiamptomus forbesi* (55% of calanoid copepod adults), and *Sinocalanous doerri* (44% of calanoid copepod adults). In the Sacramento River, calanoid copepod adults were composed of *Osphranticum labranectum* (38%), *Sinolcalonus doerri* (25%), and *Diaptomidae* spp (22%). The Sacramento River cladoceran composition was more diverse than Yolo Bypass, with *Bosmina* (30%), *Ceriodaphnia* (25%), *Chydorus* (13%) and *Daphnia* (11%). The Mann-Whitney test showed significant differences in chlorophyll *a* concentration ($P < 0.001$), densities of calanoid copepod adults ($P < 0.001$), and cladocerans ($P = 0.0244$) between the Yolo Bypass and Sacramento River. The Sacramento River had much lower densities of zooplankton than the Yolo Bypass in all years (Figure 5). Flow was a significant predictor for adult calanoid copepod abundance in the Yolo Bypass, while flow did not seem to correlate well with Sacramento River adult calanoid copepod abundance during the summer and fall (Table 4). In all years, there were increases in the densities of both cladoceran and calanoid copepod adults in the Yolo Bypass at STTD during the month of October. The increased zooplankton densities were observed after increases in rice-field drainage flows and increased chlorophyll *a* concentrations (Figure 5).

Table 3 Results from a multiple linear regression analysis of mean $Log_{10}(x)$ -transformed estimated chlorophyll *a* (dependent variable) with a suite of physical independent water variables (EC, temperature, DO, turbidity, and flow).

Station	*Dates*	*n*	*r²*	*SE*	*Independent Variables*	*β*	*95% CI*	*P-level*
LIS	Aug 15 - Oct 31 2013	71	0.69	0.652	Intercept	-1.97	(-3.278, -0.673)	0.003
					EC (µS/cm)	0.68	(0.493, 0.874)	< 0.001
					Temperature (°C)	0.07	(-0.331, 0.472)	0.728
					DO (mg/L)	0.81	(0.256, 1.356)	0.005
					Turbidity (NTU)	0.30	(-0.027, 0.624)	0.072
					Flow (cfs)	0.05	(0.014, 0.081)	0.006
STTD	Aug 15 - Oct 31 2013	77	0.91	0.218	Intercept	-3.20	(-1.975, -1.409)	< 0.001
					EC (µS/cm)	0.05	(0.613, 0.754)	< 0.001
					Temperature (°C)	0.16	(0.497, 0.910)	< 0.001
					Flow (cfs)	0.02	(-0.001, 0.052)	0.057

Specifically in 2012, we saw exceptional densities of both cladoceran and copepod adults in early October with *Bosmina* (3.01E+05/m³) and *Pseudodiaptomus forbesi* (1.74E+04/m³) being the dominant taxa (Figure 5).

Discussion

The San Francisco Estuary has been shown to have poor pelagic biomass and primary productivity compared with other estuaries throughout the world (Cloern and Jassby 2008). Phytoplankton biomass and primary productivity has experienced a long-term decline since the mid-1970s (Jassby 2008). There are numerous contributing factors, such as hydrologic manipulation and channelization, loss of shallow water aquatic habitat, and changes in nutrient ratios (Lucas and Thompson 2012), but one of the more significant is the regime shift in the benthic community of the estuary. The invasion of *Corbicula fluminea*, the most dominant freshwater benthic organism within the upper estuary (Peterson and Vayssieres 2010) and later *Corbula amurensis* in the lower estuary, have both been linked to greatly reducing phytoplankton biomass (Alpine and Cloern 1992; Lucas et al. 2002; Jassby et al. 2002; Jassby 2008). In an estuary where the planktonic food web is primarily being driven by phytoplankton production (Sobczak et al. 2002, 2005; Mueller-Solger et al. 2002; Kimmerer 2002; and Kimmerer et al. 2005), this reduction in phytoplankton has resulted in changes in both the zooplankton and fish communities. These changes have resulted in significant zooplankton abundance and community shifts (Winder and Jassby 2011), and the collapse of several pelagic fishes including Delta Smelt, Longfin Smelt, Striped Bass, and Threadfin Shad (Feyrer et al. 2003, 2007; Sommer et al. 2007).

The occurrence of flood pulses has shown direct benefits on lower trophic levels in other river-floodplains (Junk et al. 1989), including those remaining in the San Francisco Estuary (Ahearn et al. 2006). The Yolo Bypass is the largest remaining floodplain in the estuary and has been identified as a significant source of phytoplankton biomass to the lower estuary in the winter and spring during floodplain inundation (Jassby and Cloern 2000; Schemel et al. 2004; Sommer et al. 2004; Lehman et al. 2007). In addition, the Yolo Bypass and CSC have been shown to be relatively higher in chlorophyll *a* and periodically higher in calanoid copepod densities compared with the rest of the estuary (Sommer and Mejia 2013). As a product of upstream water diversions by local land owners during the summer and fall, much of this higher productivity in the Yolo Bypass and CSC was assumed to be unavailable

Table 4 Correlations between flow and biological variables in the Yolo Bypass (STTD) and Sacramento River (SHR) for the months of May–Nov for all years combined (2011–2013). The Pearson correlation coefficients are shown, together with the number of observations and P-values in parentheses.

Biological Data	*Sacramento River (SHR)*	*Yolo Bypass (STTD)*
Chlorophyll *a*	0.028 (n=46, 0.853)	0.309 (n=46, 0.037)
Calanoid Copepod Adults	0.076 (n=46, 0.614)	-0.482 (n=46, <0.001)
Cladocera	0.965 (n=46, <0.001)	0.349 (n=46, 0.017)

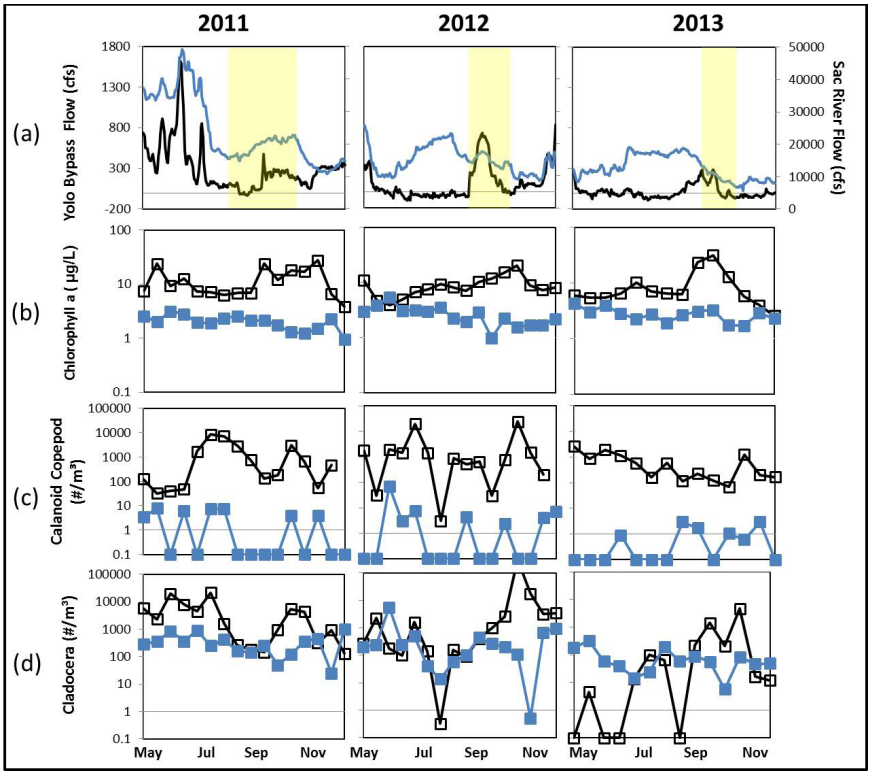

Figure 5 Trends in lower trophic levels in the Yolo Bypass (hollow black squares) and Sacramento River (solid blue squares) during May–November of 2011, 2012, and 2013. The variables from top to bottom are (a) Mean daily flow (cfs), (b) chlorophyll *a* (µg/L), (c) density of calanoid copepod adults (number/m³), (d) density of cladocerans (number/m³). Note that 0.1 = 0 density for copepods and cladocerans. The highlighted portions of the flow graphs represent the summer and fall increased rice field drainage flows in the Yolo Bypass.

to the rest of the estuary. Accordingly, the observed phytoplankton bloom events in fall of 2011 and 2012 in the Yolo Bypass and the eventual transport to the lower Sacramento River at Rio Vista were of great significance to the estuarine food web, as the upper estuary has not seen phytoplankton blooms at these intensities in over 20 years.

The observed intensity and volume of flow through the Yolo Bypass from rice-field drainage in 2011 and 2012 was larger than previously recorded over the past decade. In particular, the DWR KLOG repairs in 2012 caused significant increases in summer and fall flows being shunted down Ridge Cut Slough into the Toe Drain. The increased downstream flow from the Colusa Basin and local Yolo Bypass rice fields is vital in facilitating the exchange of the disconnected water sources from above Lisbon Weir to the lower Toe Drain, and eventually to the lower estuary, during the low flow periods of summer and fall. This upstream water source above Lisbon Weir maintains a high residence time throughout the summer months and consequently this results in the increase of water temperature, specific conductance, and chlorophyll *a*. In addition, upper sampling sites (RCS and KNA) near Colusa Drain rice drainage inputs were frequently below the limiting value of 0.07 N mg/L (Jassby 2005; Lehman et al. 2007) for dissolved inorganic nitrogen (DIN). The DIN concentrations were considerably higher at the downstream I-80 site location that was later discovered to be below both the Woodland and Davis wastewater treatment discharge points. Further investigations showed significant amounts of nitrate (> 40 N mg/L) in discharge water (Randy Dahlgren, UC Davis, unpublished data). The N:P ratios for much of the Toe Drain were ≤ 8 (except for I-80), indicating nitrogen as the limiting nutrient (Redfield 1958; Rhee 1978; Glibert 2010) during the study period. The sites from STTD southward had much higher concentrations of ammonia, suggesting that low Toe Drain outflows and stronger tidal movement may have transported Sacramento River wastewater treatment discharge into the CSC. In addition, subsequent increases in DIN concentrations were observed at lower Toe Drain sites (LIS, STTD, and BL5) after increased rice-drainage flows. The substantial nutrient loading by Woodland and Davis wastewater treatment plants appears to be highly influential in the timing and magnitude of local primary production and the availability of those nutrient concentrations for assimilation by phytoplankton, which is an essential factor controlling overall growth rate (Reynolds et al 1999; Glibert et al 2011). The shallow channel composition and lower flow rate observed in the upper Toe Drain fits the "Slower is Greener" and "Shallower is Greener" model (Lucas and Thompson 2012) that has been observed by others in the estuary (Jassby 2005; Ahearn et al. 2006) and specifically in the Yolo Bypass (Schemel et al. 2004; Sommer et al. 2004).

The phytoplankton species composition during the summer and fall blooms in the Toe Drain was dominated

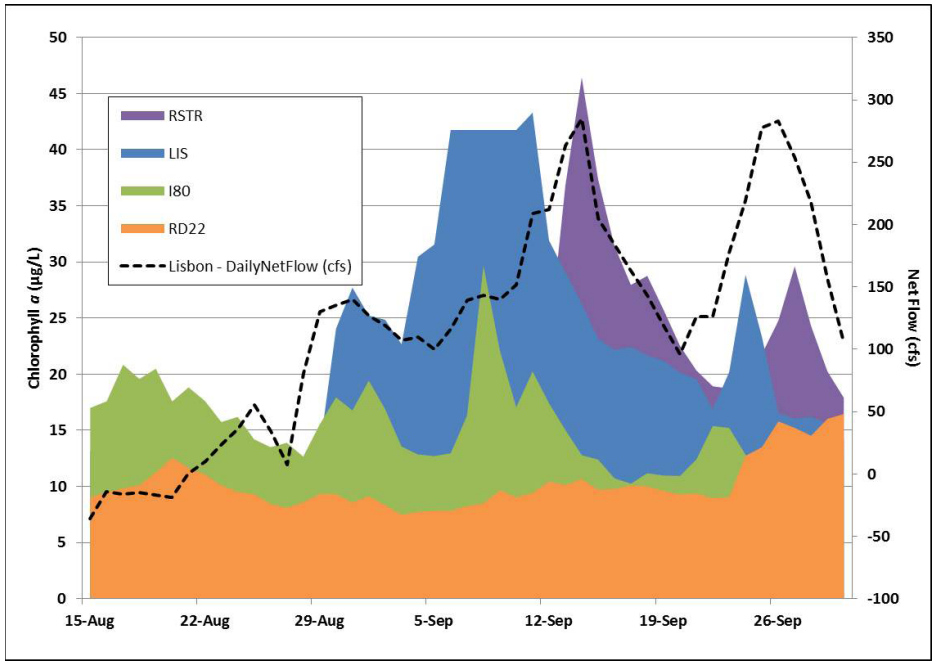

Figure 6 Daily average estimated continuous chlorophyll *a* (µg/L) and Lisbon gauge net flow (cfs) data at continuous water quality monitoring stations in the Yolo Bypass in 2013. LIS missing data 9/7–9/10.

by diatoms, which are typically of a higher nutritional quality (Lehman et al. 2007; Cloern and Dufford 2005; Brown 2009). The high concentration of diatoms in the overall phytoplankton community was representative of previously observed trends during winter and spring floodplain draining events in the Yolo Bypass (Lehman et al. 2007). Diatoms are considered a primary food source for calanoid copepods (Lehman 1992; Orsi 1995), which are an important diet item for many estuary larval and juvenile fishes (Cloern et al. 1983; Obreski et al. 1992; Orsi 1995) including Delta Smelt (Nobriga 2002; Moyle 2002; Sommer and Mejia 2013). Therefore, the observed increases in the densities of copepods, cladocerans, and chlorophyll *a* concentrations in the Toe Drain from higher fall flows, suggest potential direct benefits to higher trophic levels. This positive relationship between increased flow and improved plankton production has been reported within other estuaries, including the San Francisco Estuary (Cloern et al 1983; Kimmerer 2002). The fall phytoplankton bloom event observed at the CSC in 2011 was identified by the IEP Management, Analysis, and Synthesis Team (MAST) as an important component of the overall higher phytoplankton biomass in the low-salinity zone, and this has been hypothesized to have aided in the improved fall growth, survival, and fecundity of Delta Smelt as compared to previous years (FLaSH Synthesis Report 2014). In addition, MAST conceptual models identified food availability in the fall as a key driver in the growth and survival of Delta Smelt from sub-adults to adults (MAST Synthesis Report 2015).

Future Work

Both DWR and IEP have supported the continued effort of this study in 2014 and 2015. In the future, we plan to experimentally manage rice discharge pulse flows through the Yolo Bypass. The plan would be to replicate conditions observed in 2011 and 2012, thereby further enhancing fall phytoplankton biomass in the CSC and the lower estuary. The ultimate goal will be to model the fluxes of nutrients and phytoplankton through the Yolo Bypass and CSC corridor, and determine if it is possible with the existing water management infrastructure to stimulate an increase in the lower estuary food web during the fall period in certain years. Additional study components are being considered, such as contaminant monitoring and zooplankton growth rate and reproduction.

Acknowledgements

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Evidence for increased utilization of the Yolo Bypass by Delta Smelt

Brian Mahardja (DWR) brian.mahardja@water.ca.gov, Naoaki Ikemiyagi (DWR) naoaki.ikemiyagi@water.ca.gov, and Brian Schreier (DWR) brian.schreier@water.ca.gov

Introduction

As part of the Interagency Ecological Program (IEP), California Department of Water Resources (DWR) has operated the Yolo Bypass Fish Monitoring Program (YBFMP) since 1998. Over the years, data collected from this program has provided management agencies with crucial information on the ecology of native fishes and the Yolo Bypass floodplain. One of the most important findings from this continuous monitoring has been that off-channel floodplain habitat is highly beneficial for juvenile native fishes. However, past studies and reports demonstrating the advantages of floodplain rearing have mainly focused on Sacramento Splittail (*Pogonichthys macrolepidotus*) and Chinook Salmon (*Oncorhynchus tshawytscha*) (Sommer et al. 1997; Sommer et al. 2001; Sommer et al. 2002; Sommer et al. 2005; Feyrer et al. 2006), while little is known on how the imperiled Delta Smelt (*Hypomesus transpacificus*) utilize such habitat (though their presence has been previously reported by Sommer et al. 2004). The purpose of this report is to summarize the Delta Smelt catch data collected by the YBFMP from 1998 to 2014 and contrast our data with those observed in other IEP fish monitoring efforts. Specifically, we ask the following questions: (1) When, where, and at what abundance are Delta Smelt detected in the Yolo Bypass? (2) Does annual catch-per-unit-effort of Delta Smelt follow the annual abundance indices of Delta

Smelt from other IEP monitoring programs? (3) Is there a size difference between Delta Smelt found in the Yolo Bypass and those found elsewhere in the San Francisco Estuary?

Methods

Data Collection

For up to seven days a week from January to June, an 8-foot rotary screw trap located in the Yolo Bypass Toe Drain (Figure 1) has been used annually to sample small adult and juvenile fishes since 1998 (Sommer et al. 2004; 2005). To supplement this rotary screw trap data and better assess the near-shore fish community in the Yolo Bypass, beach seine surveys were also conducted at various locations along the Yolo Bypass Toe Drain (Figure 1). Beach seine surveys were conducted biweekly in the spring months from 1998 to 2009, and beginning in 2010, the sampling effort was increased to year-round sampling and additional locations were added. When Sacramento River water spills over the Fremont Weir (normally during months of high precipitation), four additional locations are sampled by beach seine. For all sampling, fish were identified to species, enumerated, and measured for fork length (FL) to the nearest millimeter. Fish under 25 mm FL were not measured to minimize species misidentification. In addition to fish length data, field crews also concurrently collect data on water quality parameters such as water temperature (°C) and Secchi depth (m). Electric conductivity (µS/cm), dissolved oxygen (mg/L), pH, and turbidity (Nephelometric Turbidity Units) are also measured as part of the YBFMP; however, collection of such data did not begin until the latter part of the YBFMP.

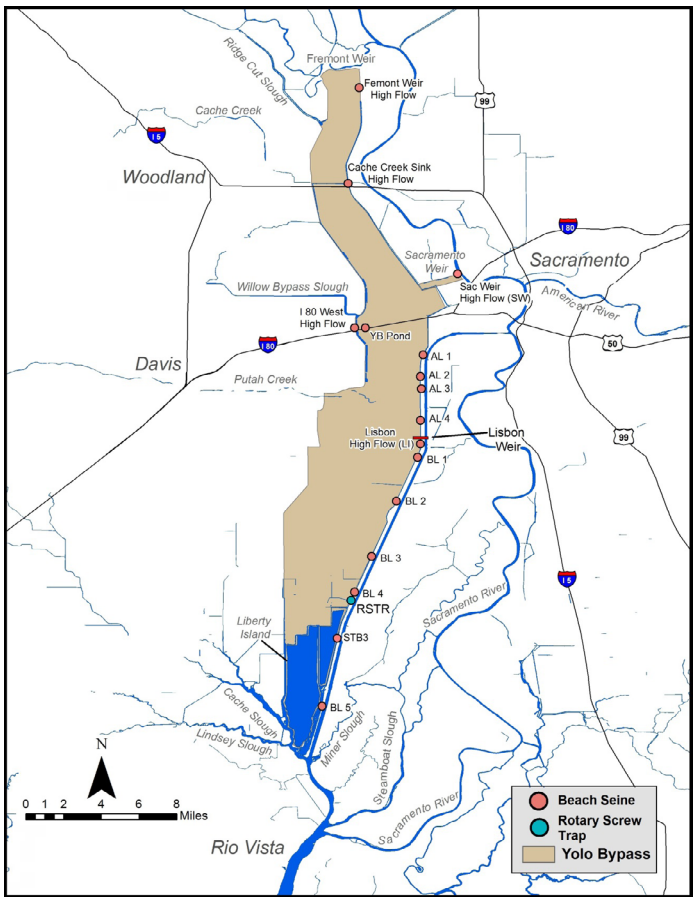

Figure 1 Overview map of Yolo Bypass including site type and locations.

Abundance Trends

Because consistent beach seine sampling at the Yolo Bypass Toe Drain did not occur until 2011 and a large portion of our Delta Smelt catch came from our rotary screw trap, we used the annual Delta Smelt catch per hour (CPH) for the rotary screw trap in order to best assess how the relative abundance of Delta Smelt in the Yolo Bypass varies from year to year. Our annual rotary screw trap CPH was then visually compared to the annual Delta Smelt indices from two other IEP fish monitoring programs: the 20 mm Larval Survey and the Fall Midwater Trawl Survey. The 20 mm Larval Survey monitors small, post-larval juvenile Delta Smelt from the Carquinez Strait to the Sacramento-San Joaquin Delta between the months of March and July (http://www.dfg.ca.gov/delta/projects.asp?ProjectID=20mm) each year. The Fall Midwater Trawl Survey samples pelagic species (including subadult Delta Smelt) at locations similar to that of the 20 mm Larval Survey, from September to December (http://www.dfg.ca.gov/delta/projects.asp?ProjectID=FMWT) annually. These indices were chosen because our trapping effort collects both adults (born in previous calendar year) and juveniles (young of year). Although the Spring Kodiak Trawl Survey (SKT) might provide a better abundance index for adult Delta Smelt, it was not included because the survey was initiated in 2002 (precluding comparison with our Yolo data for years prior to 2002). To explore if temporal variations in water quality parameters could explain

the pattern of Delta Smelt catch in the Yolo Bypass Toe Drain, we performed ordinary least squares regression on our annual rotary screw trap CPH data with the average recorded water temperature and Secchi depth at the rotary screw trap site for each six-month sampling period. Average water temperature ranged from 14.3 °C to 16.7 °C, and average Secchi depth ranged from 0.18 m to 0.21 m.

Size Comparison

To determine whether there was evidence of a size difference between Delta Smelt found in the Yolo Bypass and those found elsewhere in the San Francisco Estuary, we visually compared the FL-at-date of Delta Smelt caught in the Yolo Bypass rotary screw traps and beach seines with three other IEP fish-monitoring programs that sample Delta Smelt within a similar time frame, such as the 20 mm Larval Survey, Summer Townet Survey, and the SKT. The Summer Townet Survey samples young-of-year juvenile Delta Smelt from June to August (http://www.dfg.ca.gov/delta/projects.asp?ProjectID=TOWNET), while the SKT samples Delta Smelt adult spawners from January to May (http://www.dfg.ca.gov/delta/projects.asp?ProjectID=SKT). As a result of the low Delta Smelt catch in the Yolo Bypass prior to 2009 (total annual catch ranging from 4 to 26), we limited our comparison to the last seven years (2009 to 2014). Data from the Summer Townet Survey are limited from 2009 to 2013 because data for 2014 was not yet available.

We conducted an Analysis of Variance (ANOVA) to compare adult fish FL between the two groups (YBFMP and SKT) by way of ordinary least squares. We designated fish below 60 mm FL after the month of May as "juveniles" or "young-of-year fish," and those that did not fall under this category as "adults." For this analysis, juvenile fish were removed from the dataset and SKT fish were used as the reference group in the model (i.e., SKT data assigned to a dummy variable of 0 and YBFMP data assigned to a dummy variable of 1). For juvenile young-of-year size comparison, we opted not to conduct further analysis due to key differences in sampling methods between the monitoring programs that may severely bias the results (e.g., YBFMP does not measure fish smaller than 25 mm FL, larger disparity in sample size, etc.).

Results

From 1998 to 2014, the YBFMP caught a total of 707 Delta Smelt via rotary screw trap and beach seine, with the majority of the catch (78%) coming after 2008. A larger portion of Delta Smelt were captured using the rotary screw trap (89%, N=631) rather than beach seine (11%, N=76) over this period. Out of all Delta Smelt captured by beach seine as part of the YBFMP, a majority came from the sites closest to the Cache Slough Complex. Site BL5 (Figure 1) comprised the majority of the catch (68%), while BL4 (near the rotary screw trap) had the second-highest total (9 % of total catch). Few Delta Smelt were caught at other beach seine stations, including above Lisbon Weir (Figure 1), where all stations had a total of four or less Delta Smelt captured throughout the study period. Over the history of the program, only four

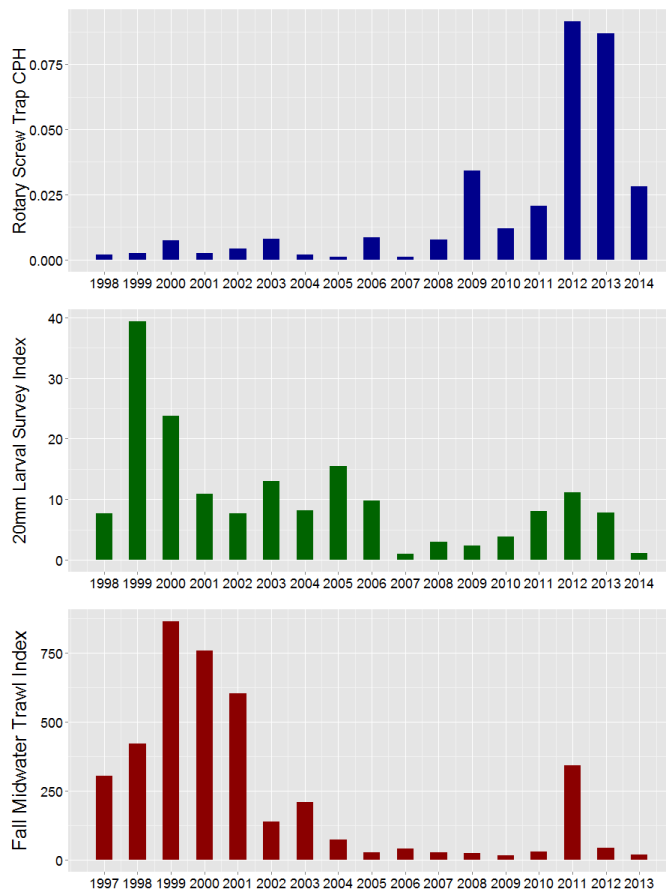

Figure 2 Annual Delta Smelt CPH of the Yolo Bypass rotary screw trap (top, blue) in comparison to the Delta Smelt annual 20 mm Larval Survey Index (middle, green) and the previous year's Fall Midwater Trawl index (bottom, red).

Delta Smelt have been observed in the fall (September to December); however, our rotary screw trap is not in operation during this period. All four fish were sampled by beach seine at site BL5, one individual in November of 2010, two in October of 2011, and one in November of 2011.

Abundance Trends and Water Quality

When the rotary screw trap data is standardized for effort, we observed a pattern of increased catch over time (Figure 2), with post-2008 annual CPH averaging at 0.046 ± 0.035 fish/hour in comparison to pre-2009 annual CPH average of 0.004 ± 0.003 fish/hour. Abundance trends in Yolo Bypass (lower CPH pre-2009 and higher CPH post-2008) contrasts with the 20 mm Larval Survey and Fall Midwater Trawl Survey, which had an overall declining trend. There appeared to be no linear relationship between annual rotary screw trap CPH and water temperature or Secchi depth at the sampling location. The single term temperature model resulted in multiple R^2 value of 0.003 with p-value of 0.825, while the single term Secchi depth model resulted in multiple R^2 value of 0.040 with p-value of 0.444. Similarly, the regression model with both terms included produced non-significant results with R^2 value of 0.041 and p-value of 0.748.

Size Comparison

During the most recent high abundance years in Yolo Bypass (2009–2014), the size data suggest that there were two cohorts annually (adults from previous year and young-of-year juveniles) (Figure 3). Adult Delta Smelt were generally collected between January and April, while those collected between May and July were smaller juveniles. In the period of May and June (Julian

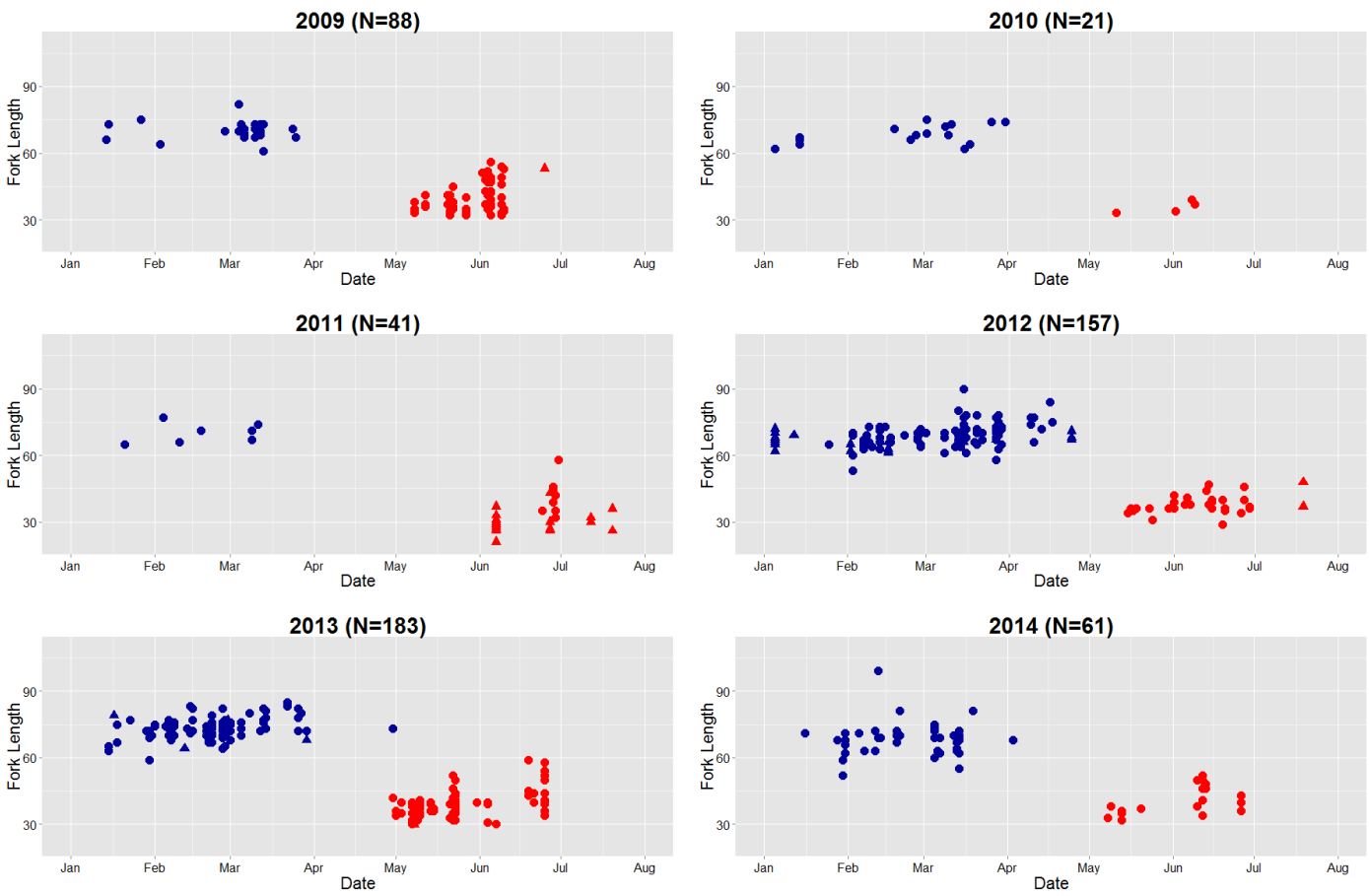

Figure 3 Delta Smelt FL distributions by date in Yolo Bypass for the six years with highest CPH (2009–2014). Blue color indicates adult fish and red indicates juvenile fish. Circles indicate that fish were sampled by rotary screw trap and triangles indicate that they were sampled by beach seine.

Day 121–181, Figure 4), juvenile Delta Smelt collected in the Yolo Bypass appear to be generally larger than those collected elsewhere in the San Francisco Estuary. While not as apparent, the length of adult Delta Smelt in the Yolo Bypass also seems to be longer on average in comparison to those captured everywhere else by the SKT (Figure 5). An ANOVA showed this difference to be significant ($P < 0.001$), with an estimated coefficient of 3.022.

Discussion

Delta Smelt utilization of the Yolo Bypass appears to have increased over time based on YBFMP total catch and rotary screw trap CPH. This increase in catch occurred despite the decline in Delta Smelt abundance elsewhere in the estuary during and after the Pelagic Organism Decline (POD) (Sommer et al. 2007). Moreover, relatively high catch numbers were observed in the Yolo Bypass even in dry years, when Delta Smelt are expected to be less abundant (Baxter et al. 2010). It is also important to note that our drought year sampling (2013–2014) may have underestimated actual use of the region. For example, the spread of water hyacinth rendered the BL5 site (our most productive beach seine site for Delta Smelt) inaccessible for beach seining throughout the entire 2013 and 2014 sampling periods.

There is evidence based on otolith microchemistry analysis suggesting that while Delta Smelt have exhibited migratory behavior in the past, they may be shifting towards freshwater residency over the last few years (James Hobbs, unpublished data). This shift in Delta Smelt life history could explain the increased presence of Delta Smelt in the Yolo Bypass. However, it remains unclear why Delta Smelt would target Yolo Bypass in recent years. For example, there were no major changes in water temperature and Secchi depth between early and later years at our sampling location. A more complex ecological shift might explain this pattern, but we hypothesize that in recent years the Yolo Bypass and the northern Delta region have become a more suitable habitat for Delta Smelt relative to other locations within the San Francisco Estuary.

Figure 4 Comparison of Delta Smelt FL in the Yolo Bypass (red) and elsewhere in the San Francisco Estuary (20 mm Larval Survey in light blue, Summer Townet Survey in dark blue, and SKT in yellow) between 2009 and 2014. Note that YBFMP does not record fish under 25 mm FL. Four observations of Delta Smelt in the Yolo Bypass BL5 site during the fall are not shown.

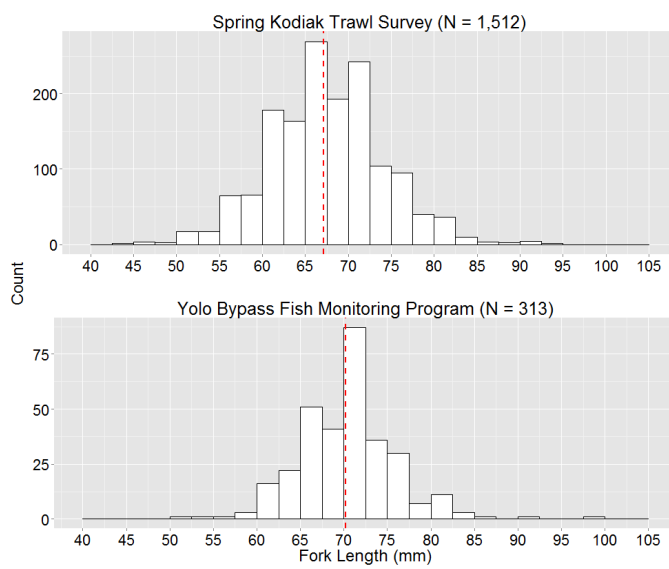

Figure 5 Histogram comparison of adult Delta Smelt FL for SKT (top) and YBFMP (bottom) from the time period between 2009 and 2014. Dotted red line denotes the mean for each group of fish.

Adult Delta Smelt found in the Yolo Bypass appear to be generally larger in recent years (2009–2014) than those found elsewhere by the Spring Kodiak Trawl Survey during the same period (Figure 5). It is unclear if these differences are due to habitat quality or other explanations, such as differences in sampling methods between the surveys or size-selective mortality in the Yolo Bypass. Juvenile Delta Smelt collected in the Yolo Bypass also appear to be larger earlier in the year than those collected by other IEP monitoring programs (Figure 4). Again, it is unclear if these differences are biological (e.g., earlier spawning or faster growth rates) or are related to differences in sampling methods in the Yolo Bypass versus the rest of the Delta. However, we have collected otoliths from ~200 Yolo Bypass Delta Smelt between 2012 and 2014, allowing for future work that may address this question more directly.

In summary, we have documented a long-term increase in the densities of Delta Smelt in the Yolo Bypass, particularly at the rotary screw trap location. While this trend is counter to those seen from the broader estuary, our findings compliment previous observations that Delta Smelt are increasingly utilizing the North Delta region (Sommer and Mejia 2013). Our observations are particularly important because, unlike other native floodplain-dependent fish species, such as the Sacramento Splittail, we found relatively high numbers of Delta Smelt during the recent drought years when the Yolo Bypass had minimal floodplain inundation. These results further confirmed that the Yolo Bypass provides various fisheries benefits to the San Francisco Estuary ecosystem (Sommer et al. 2001; Sommer et al. 2002). More importantly, our study indicated that these benefits are diverse, persistent, and not limited only to wet periods when the bypass is flooded.

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Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) occupy the Yolo Bypass in relatively high numbers during an extreme drought

Pascale Goertler (DWR) pascale.goertler@water.ca.gov, Jared Frantzich (DWR) jared.frantzich@water.ca.gov, Brian Schreier (DWR) brian.schreier@water.ca.gov, and Ted Sommer (DWR) ted.sommer@water.ca.gov

Introduction

Currently, California is experiencing a hydrological extreme to severe drought throughout most of the state (NOAA-NCDC). Additionally, 2014 was California's warmest calendar year on record (NOAA-NCDC). Furthermore, these extreme conditions preceded 2014, the SPI (Standardized Precipitation Index) has characterized the period from December 2012 to November 2014 as exceptionally dry (NOAA-NCDC). Although the current drought is extreme, droughts regularly occur throughout California's history. In the last 1200 years California has experienced a series of "mega droughts," all of which lasted longer than any drought recorded in the 20th century (Cook et al. 2004). This history of drought in California stresses the importance in assessing its vulnerability to precipitation deficits and understanding drought variability (Cook et al. 2004). These "mega droughts" also serve as lessons that: (1) longer duration droughts are possible, and (2) elevated aridity in the west may be a natural response to climate warming (Cook et al. 2004). With the inevitability of droughts in California and possibility of "mega droughts" it is essential to understand drought effects on natural resources.

Water deficits are a major concern for California's natural resource management; primarily agriculture (~$18 billion in 2012) (CDFA), municipal users, fisheries ($24,043,813 and 145,433 jobs in 2012) (NOAA-NMFS) and wildlife. Drought conditions may affect food supply and the prices of fruits, vegetables, tree nuts, and dairy throughout the US (USDA), as well as fish and wildlife throughout California. Chinook Salmon (*Oncorhynchus tshawytscha*) support both recreational ($100 million annually in Sacramento River) (Charbonneau and Caudill 2011) and commercial fisheries ($5,096,433 in California for 2011–2013) (NOAA-NMFS). However, the effects of drought on salmon and other fisheries are less understood than those on the agricultural industry. Drought will undoubtedly restrict access to habitat for aquatic organisms, but it may have additional effects on temperature, decrease connectivity and available refuge, deteriorate water quality, shift food resources, and modify the strength and structure of interspecific (Lake 2003) and intraspecific interactions (Mathews and Marsh-Mathews 2003). These alterations may increase mortality rates, decrease birth rates, and increase migration rates of fish (Magoulick and Kobza 2003), such as juvenile salmon. These alterations also have the potential for population declines (Mathews and Marsh-Mathews 2003).

Drought and water use are not the only restrictions on habitat for salmon. Many factors, including dams, levees, water diversions, invasive species, and contaminants have immensely changed the natural landscape for salmon. Salmon have been extirpated from much of their historically available habitat: approximately 2% of the historically dominant habitat in the Sacramento-San Joaquin River Delta is currently accessible to fish (SFEI-ASC 2014), and salmon are extirpated from approximately 75% of the upstream spawning and rearing habitats (Van Cleve 1945). In addition, many accessible habitats have been severely degraded by hydraulic mining and construction of dams and water-diversion projects, dredge mining, railroad construction, and logging (Yoshiyama et al. 1998). Therefore, drought is a compounding factor with the already extensive habitat loss. Not surprisingly, the current state of Chinook Salmon long-term resilience in California is bleak. Central Valley Chinook Salmon are listed as endangered and threatened under the Endangered Species Act (ESA) for two of their three federally recognized populations (Good 2005). Chinook Salmon in California are living at the most southern (and warmest) part of their natural range and may not be able to tolerate even slight increases in temperature; in addition to information on the effects of drought on salmon, drought studies may also produce insight to the effects of climate change on California's salmon populations.

In this study we examine juvenile Chinook Salmon, their prey resources, and the abiotic conditions during the most recent drought in the Yolo Bypass. In the Yolo Bypass, inundation events are the primary driver for

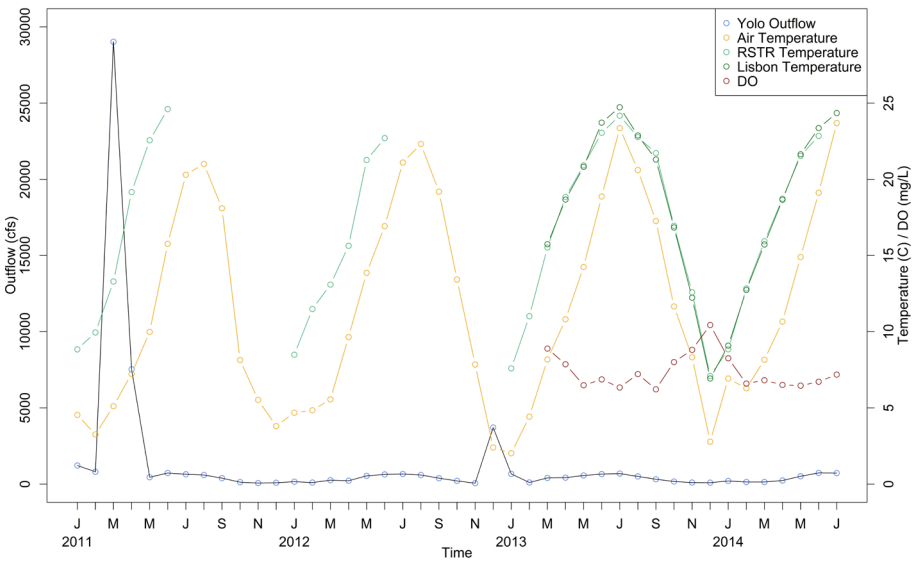

Figure 1 Discharge, water temperature, and dissolved oxygen within the Yolo Bypass during the sampling period. Air temperature is included to infer water temperature when it was not recorded (NOAA-NNDC-CDO).

expanding and contracting aquatic habitat. Without flooding, the majority of the Yolo Bypass is not accessible to fish and there is less floodplain habitat; a valuable foraging habitat for juvenile salmon (Sommer et al. 2001; Sommer et al. 2005). In addition, without flooding, juvenile salmon cannot enter the Yolo Bypass from the Sacramento River through the Fremont Weir, but only by moving up Prospect Slough. These dry conditions have been documented in the Department of Water Resources' (DWR) monitoring of the Yolo Bypass (Figure 1). DWR maintains a continuous monitoring program, which has documented a trend counter to what we may expect. In 2014, we found a relatively high number of juvenile Chinook Salmon occupying the Yolo Bypass despite the lack of flooding and presence of an extreme drought. Thus, it appears that the Yolo Bypass is used as rearing and migratory habitat for juvenile Chinook Salmon both during high flow as a floodplain (Sommer et al. 2001) and during low flow periods. Here we describe our recent observations and offer possible explanations.

Methods

We will be presenting results from DWR's fisheries and the Yolo Bypass Fish Monitoring Program (YBFMP), conducted as part of the Interagency Ecological Program (IEP). We have subset data from 2011–2014 during January–June monitoring efforts to capture juvenile Chinook Salmon habitat use during the current drought and most recent wet year (2011). The YBFMP has been collecting data since 1998.

Prey Resources

Invertebrates were sampled at the rotary screw trap (RSTR) and Sherwood Harbor (SHR) on the Sacramento River (Figure 2) on an ebb tide with a conical plankton net and rectangular drift net, monthly with the exception of inundation and draining periods when it may have been sampled weekly (Sommer et al. 2001, 2004). The plankton net was made of 153 µm mesh net, with a 0.50 meter diameter outer mouth, and is 2 meters in length. The drift net was a 500 µm mesh net, with a dimension of 0.46 meters by 0.3 meters at the mouth, and 0.91 meters long, harnessed to a floated stainless steel frame. After collection in the plankton net, a 1 milliliter subsample was extracted, identified, and counted under a compound microscope. All drift invertebrate samples were rinsed and passed through a 0.5 mm sieve. All the material remaining within the sieve was processed for identification. All the aquatic insects and non-insects were counted and identified to the family level. The terrestrial insects and non-insects were counted and identified to the order level. The number per cubic meter for each taxon of plankton (N) and number per cubic meter for each aquatic and terrestrial organism taken in the aquatic drift net (N) were then calculated. To simplify interpretation, drift invertebrates were reported by classification (Appendix Table 1) and zooplankton were reported in categories (Appendix Table 2).

Juvenile Chinook Salmon

Juvenile Chinook Salmon were sampled by beach seine and rotary screw trap (Sommer et al. 2011, 2004b, 2005). Beach seine sites were sampled monthly with a single haul from an 8.3 meter by 1.3 meter pole seine

(1/3 sq. cm mesh). The seine sites included one perennial pond, nine sites along the Toe Drain, and four high flow sites to capture floodplain inundation periods (Figure 2). A 2.6 meter diameter screw trap was installed near the lower end of the Toe Drain (Figure 2). The rotary screw trap was fished daily, but generally not operated on weekends unless the Bypass flooded substantially. When the Yolo Bypass flooded, four additional beach seine sites were sampled, and the screw trap could not always be operated. Captured fish were identified to species, and counted and measured for fork length to the nearest millimeter for up to 50 individuals of each species. We calculated a modified catch per unit effort (CPUE) based on the best data available for effort for both the beach seine and rotary screw trap sampling (Table 1). For the rotary screw trap, the number of hours fished is recorded for effort, not volume.

Additional Datasets

Abiotic monitoring was captured through a combination of DWR efforts and data from NOAA's National Climate Data Center (NNDC-CDO). Three temperature measures were used to illustrate the temperature range during the 2012–2014 drought: air temperature from the counties within the Sacramento watershed (NNDC-CDO), and the water temperature at the rotary screw trap and Lisbon Weir (Figure 2). Aquatic conditions within the Yolo Bypass were further characterized by dissolved oxygen at the Lisbon Weir and DWR's Dayflow calculations for the Yolo Bypass's flow addition to the Delta (QYolo: DWR-Dayflow 2014).

Table 1 A modified catch per unit effort (CPUE) based on number of hours fished (RSTR) and volume of beach seine (BSEIN) for each year and gear method.

Year	*Volume (m^3)*	*Operation Hour*	*Count BSEIN*	*Count RSTR*	*CPUE BSEIN*	*CPUE RSTR*
2011	9285	1992	102	265	0.011	0.133
2012	2792	1753	23	119	0.008	0.067
2013	4743	2109	42	88	0.009	0.042
2014	4709	2174	38	377	0.008	0.173

*Note: operation hours for the rotary screw trap was lower in 2011 and 2012 because of (1) damage from high debris loads resulting in several weeks of no operation in late March and April in 2011, and (2) high catches of ESA-listed species in 2012 resulting in shorter daytime-only sets taken intermittently from January through March.

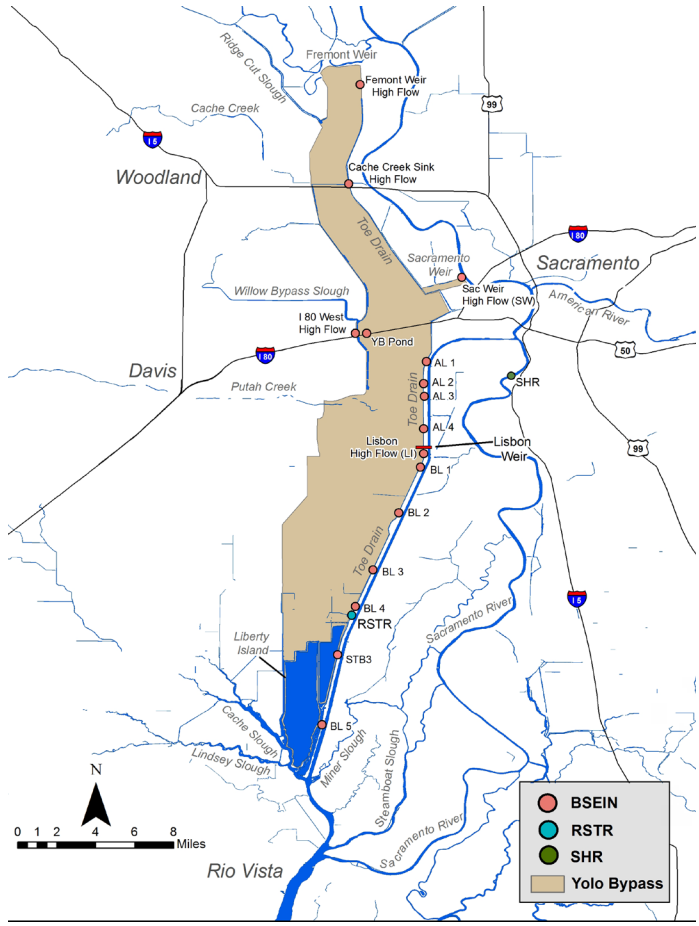

Figure 2 Map of the Yolo Bypass, showing the juvenile Chinook Salmon sampling locations and gauge locations.

In addition, data on adult Chinook Salmon escapement was obtained from the California Central Valley Chinook Population Report (Azat 2014).

Results

Prey Resources

The average density of drift invertebrates (N) varied by year and by location (Figure 3). Generally, the Sacramento River site had less drift invertebrates than the Yolo Bypass site. All categories were represented in all years and both sites. The relative proportion of each invertebrate category varied, and there was no obvious pattern between the number or relative proportion of each category with respect to drought years. Both 2013 and 2014 had substantially more drift invertebrates at both sites than 2011 and 2012. In the Yolo Bypass there were

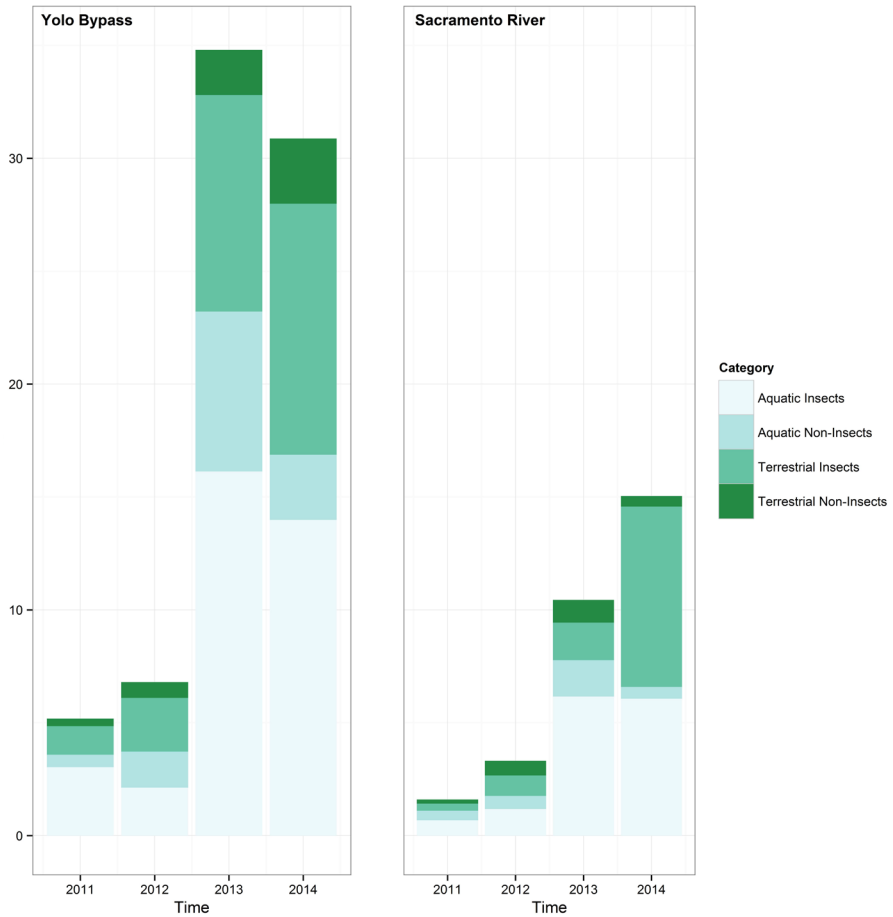

Figure 3 The average number of drift invertebrates per cubic meter (N) for each prey category by year and location.

considerably more aquatic insects and non-insects in 2013, and considerably more terrestrial insects and non-insects in 2014. In the Sacramento River site there was a sizeable increase in terrestrial insects in 2014, and many more aquatic non-insects in 2013 than any other year. In this analysis we focused primarily on yearly differences to capture drought effects, but there were also monthly differences among locations and years. For example, the large pulse of terrestrial insects in the Sacramento River in 2014 occurred during March, when 42 times as many terrestrial insects were captured as compared to any other month. March 2014 also had five times as many aquatic insects as any other month on the Sacramento River. Furthermore, in January of both 2013 and 2014, the Yolo Bypass had close to twice as many drift invertebrates on average than any other month.

The average density of zooplankton (N) was always greater in the Yolo Bypass when compared to the Sacramento River site (Figure 4). In the Yolo Bypass, Cladocera, Cyclopoids, and Microzooplankton/Nauplii dominated the catches, with the exception of 2012 when Calanoids had a noticeable presence (Figure 4). The highest catches of zooplankton in the Yolo Bypass were in 2013, and in the Sacramento River the highest catches were in 2014. The considerably higher catches in the Sacramento River in 2014 were driven by zooplankton catches in April; five times higher than any other month in our sampling period. The high catches in 2013 and 2014 in the Yolo Bypass were driven by January and February catches. These seasonal and annual differences between the Yolo Bypass and Sacramento River sites are similar for both zooplankton and drift invertebrates. Despite the drought, 2013 and 2014 produced higher densities of zooplankton and drift invertebrates, particularly in the winter and spring for the Yolo Bypass and Sacramento River, respectively.

Juvenile Chinook Salmon

Surprisingly, many more juvenile Chinook Salmon were captured in 2014 than in the previous two drought years (Figure 5). The number of sampling events per year and gear method did vary (Figure 5), but even when accounting for effort catches were high in 2014, particularly in the screw trap (Table 1). In addition to high catches, fish captured in 2014 displayed a different pattern of size and timing than those captured in 2012 and 2013 (Figure 5). The majority of the fish captured in 2014 occupied the Yolo Bypass in March at smaller size classes than 2012 and 2013 (average fork length in March, 2012: 71 mm; 2013: 75 mm; 2014: 53 mm). The high numbers of juvenile Chinook in the Yolo Bypass in 2014 was similar to catch totals in the most recent flooding year (2011) (Figure 5, Table 1). However, how the juveniles were distributed across months and sizes was very different in 2014 than 2011. In 2014, juveniles were first detected in the system later than 2011, and March catches

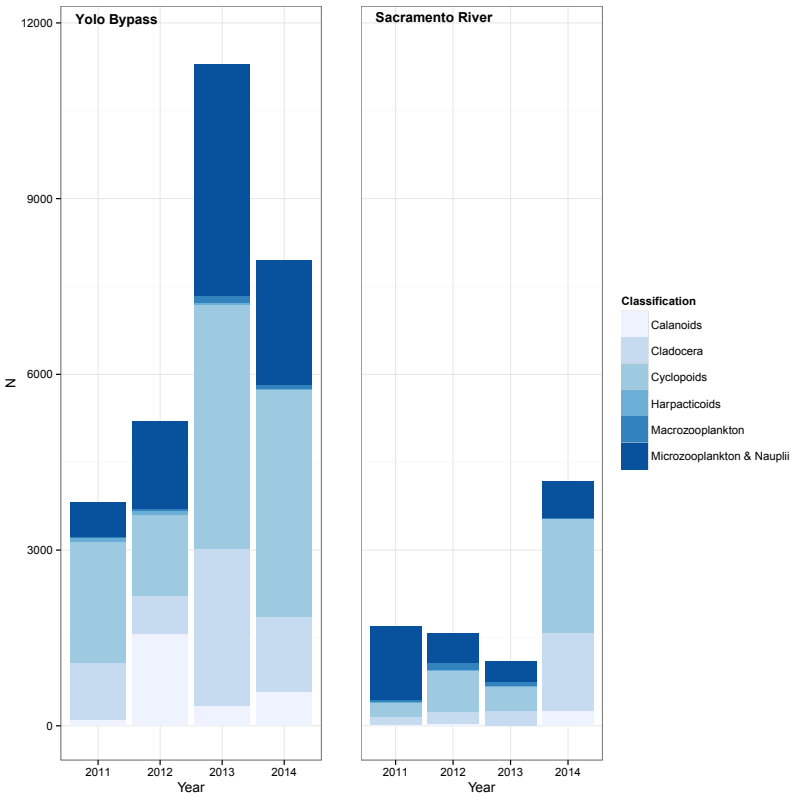

Figure 4 The average number of zooplankton per cubic meter (N) for each prey classification by year and location.

were much greater in 2014 than 2011. In contrast, in 2011 fish were present in the Yolo Bypass for a longer period than the other three lower water years; juvenile Chinook Salmon were caught until June in 2011, while juveniles seem to have left the system by May in drought years. In 2011, larger fish were present both earlier and later in the outmigration season for the Yolo Bypass. Additionally, the majority of juveniles captured in drought years were caught in March, while the majority of fish captured in 2011 were caught in April. These results show annual variability in juvenile Chinook Salmon catch in the Yolo Bypass, which is more complex than a simple association to local hydrology.

Discussion

During flood years, the Yolo Bypass has been shown to provide rearing habitat for juvenile salmon (Sommer et al. 2001; Sommer et al. 2005) and produce high levels of invertebrate prey (Sommer et al. 2004; Schemel et al. 2004). However, less is known about the relative value of this region during periods of drought. A key finding of our study was that juvenile Chinook Salmon occupied the Yolo Bypass in relatively high numbers in 2014 (Figure 5, Table 1), when the current drought was most severe. The timing and size of juvenile Chinook in the Yolo Bypass also varied among drought years and the most recent wet year (2011). To further investigate these results, we also examined: (1) catch efficiency, (2) the Central Valley Chinook Salmon population (e.g., parental escapement estimates), (3) abiotic conditions, and (4) prey resources in the Yolo Bypass and Sacramento River.

Drought conditions undoubtedly reduce habitat for aquatic organisms. However, a major limitation of our results is that it is unclear how the Yolo Bypass rotary screw trap efficiency varies with hydrology (e.g., Feyrer et al. 2006). In wetter years, analyses of rotary screw trap data do not support the hypothesis that trap efficiency is a primary driver of Chinook Salmon catch trends in Yolo Bypass (Sommer et al. 2005). Modeled flows and estimated area inundated for the sampling season showed an average of over 11 times more inundation in 2011 than 2012 (DWR, unpublished), meaning that the rotary screw trap simply samples a smaller percentage of the total available habitat during inundation. Moreover, it is possible that trap efficiency may be quite low during drought conditions because tidal velocities are correspondingly low, so the rotary screw trap only samples efficiently during a small portion of the day. Additionally, average inundation varies even within dry and critically dry years because of short, winter-flow pulses that can cause flooding despite overall low flows (i.e., December 2012). A related caveat is that it is unknown how variation in Chinook Salmon population size affects use of the Yolo Bypass. It is reasonable to assume that higher population levels could lead to increased catch. For example, fall-run adult escapement has steadily increased over our sampling period (Figure 6), and our high catches in 2014 may be partly attributed to more juveniles in the system in 2014. In addition to wild juvenile Chinook Salmon production, changes in hatchery release practices could have led to changes in catch, but total hatchery releases for all runs through 2014 are not currently available. Still,

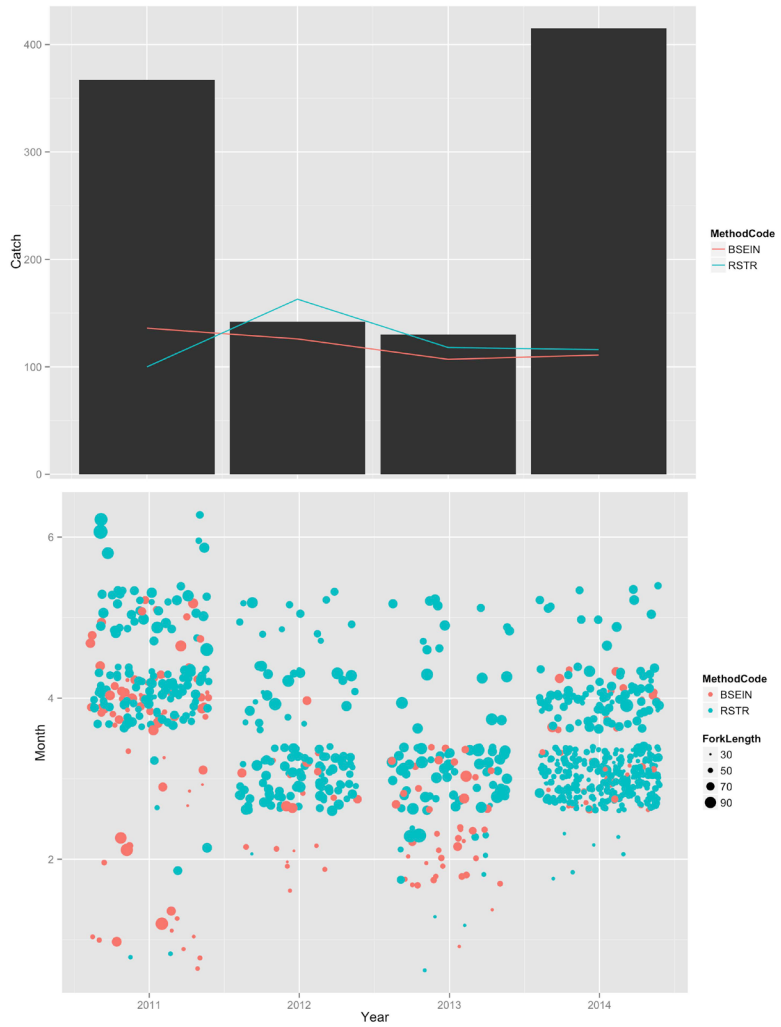

Figure 5 Juvenile Chinook Salmon catch from 2011 to 2014 in the Yolo Bypass using both rotary screw trap (RSTR) and beach seine (BSEIN) sampling. Top: total catch per year (black bars) and the number of sampling events for each sampling method per year (lines). Bottom: for those fish measured for fork length, the fork length (point size) and sampling method by month and year.

it is likely that much fewer hatchery-origin juveniles were released in the Sacramento River in 2013 and 2014 than in previous years as a result of concerns about high water temperatures. The availability of prey may have also been affected by population size through intraspecific competition (Levin et al. 2001; Daly et al. 2012). It has been shown that hatchery releases can negatively impact the survival of wild Chinook Salmon in years of poor ocean conditions (Levin et al. 2001), and smaller wild fish can be prey for larger hatchery fish (Seth and Sharpe 2012). Therefore, how the Central Valley Chinook Salmon population, management practices, and sampling techniques change for juvenile salmon during drought conditions may have unknown consequences on our results.

In addition to reducing habitat, droughts can affect water quality, such as temperature and dissolved oxygen (DO). Our results suggest that temperatures were suitable for salmon during much of the key winter–early spring salmon-rearing period (Figure 1). This is despite the fact that 2014 was the warmest calendar year on record for California. However, variations in temperature among years may have influenced outmigration patterns. For example, moderate-to-high water temperatures are thought to be a bioenergetic limitation for juvenile Chinook Salmon, and temperatures above 19 °C are associated with shallow wetland habitat exclusion (Bottom et al. 2011). Juvenile Chinook Salmon growth has been shown to begin declining at daily maximum water temperatures exceeding 23 °C (Geist et al. 2010). In the Yolo Bypass, periods of temperatures above 19 °C begin in May for 2011 and daily maximum temperatures exceeding 23 °C begin in June for 2011. For the drier years, however, both periods of temperatures above 19 °C and daily maximum temperatures exceeding 23 °C occurred in the same month: May for 2012, and April for 2013 and 2014. Therefore, the drought years had an earlier and more rapid transition into stressful water temperatures for juvenile salmon, which coincided with an earlier outmigration from the Yolo Bypass in drought years (Figure 5).

The uniquely high temperatures experienced by California in 2014 were not clearly reflected in Yolo Bypass water temperature, but DO began fluctuating into critically stressful levels one month earlier in 2014 than the previous drought year. For fish, DO can dictate the quality of habitat, and suboptimal oxygen concentrations can cause delayed development (Roussel et al. 2007), physical stress (Davis et al. 1963), and death during droughts (Mathews and Marsh-Mathews 2003). Monthly average DO levels range between 6 and 10 mg/L for the Yolo Bypass (Figure 1), and short duration periods fluctuate below 5 mg/L several days a month starting in May of 2013 and April of 2014. These stressful and

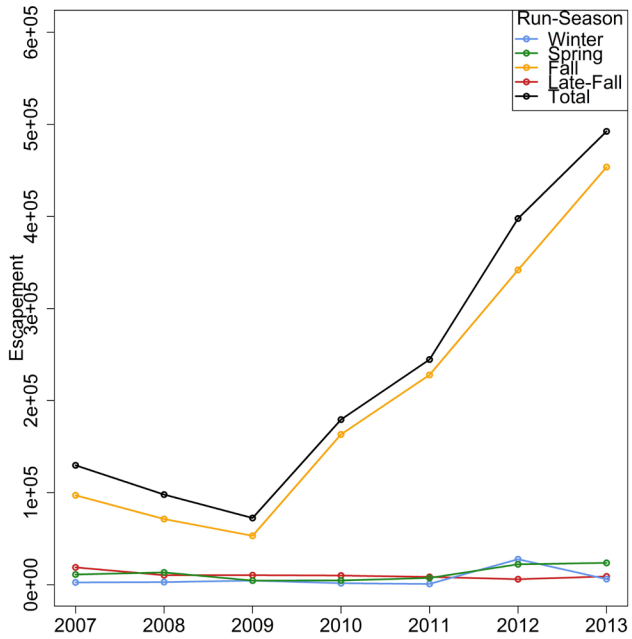

Figure 6 Adult escapement estimates for Central Valley Chinook Salmon, by return to river season (or run), for 2007-2013. These data have been synthesized from the *GrandTab 2014.04.22 California Central Valley Chinook Population Report* (Azat 2014).

sometimes lethal oxygen concentrations regularly occur in the spring and summer in the Yolo Bypass, and may affect habitat occupancy. DO and temperature could be major drivers in the emigration of juveniles in the late spring and summer.

Both the prey resources and Chinook Salmon catches were counter to what we might expect given the physical conditions during a drought. The exceptionally dry period experienced by California between December 2012 and November 2014 clearly reduced the available habitat in the Yolo Bypass as a result of a lack of inundation, and perhaps high temperatures and low DO. However, the remaining habitat in the Yolo Bypass may have been some of the most productive available to juvenile Chinook Salmon on their outmigration from the Sacramento River to the ocean. Our drift and zooplankton results show that the Yolo Bypass RSTR site had consistently higher densities of prey in all water years compared with Sherwood Harbor on the Sacramento River. Although temperature can negatively affect juvenile salmon and reduced habitat could crowd salmon into stressful densities, abundant prey can mitigate for these stresses. Simulations suggest that for smaller, younger life stages, growth is limited by feeding rate and prey quality more frequently than by temperature (Beauchamp 2009). Therefore, if more food is available, smaller fish may be able to withstand higher temperatures. Additionally, insects (aquatic and terrestrial) were the dominant prey category, which can have bioenergetic implications for juvenile salmon. Insects belonging to orders found in our study have been linked to higher modeled growth rates for juvenile salmon (Grey 2005). These high energy content insects, such as Diptera, Hemiptera, Homoptera, Hymenoptera, and Lepidoptera are estimated as two to five times as calorically valuable as some non-insects, such as Isopoda, Arachnida, Mysidae, and Amphipoda (Gray 2005). High temperatures were likely experienced throughout the watershed, but the exceptionally productive habitat in the Yolo Bypass may have provided juvenile salmon enough prey to endure the warmer temperatures. Even without inundation, the Yolo Bypass has more natural banks and riparian vegetation, and is better connected to tidal wetlands in Liberty Island and Cache Slough, compared to the more augmented and detached Sacramento River. Therefore, riparian vegetation and connectivity to tidal habitat may be important for floodplains even during droughts. This monitoring effort is a necessary tool to track changes over time, but special studies are needed to answer questions about how food-web processes might change during a drought and how droughts may impact the Central Valley Chinook Salmon population. Therefore, we are in the process of conducting special analyses to address some of our remaining questions.

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Appendixes

Appendix Table 1 Prey resource taxa in zooplankton classifications.

Calanoids	***Cladocera***	***Cyclopoids***	***Harpacticoids***	***Macrozooplankton***	***Microzooplankton & Nauplii***
Acartia spp.	Bosmina	Cyclopoid adult		Mysids	Rotifers
Acartia copepodid	Ceriodaphnia	Cyclopoid copepodid		Larval Fish	Barnacles
Diaptomidae spp.	Daphnia	*Acanthocylops robustus* copepodid		Fish eggs	*Copepod* nauplii
Diaptomidae copepodid	Chydorus	*Acanthocylops robustus* adult		*Corophium* spp.	*Cladocera* nauplii
Pseudodiaptomus forbesi adult	Camptocercus	*Oithona* Spp.		*Gammarus* spp.	*Unid* nauplii
Pseudodiaptomus forbesi copepodid	Scaphloberis	*Oithona similis*		Clam	Ostracods
Eurytemora affinis adult	Diaphanosoma	*Oithona davisae*		*Cumacean*	
Eurytemora affinis copepodid	Juvenile Daphnia	*Oithona* copepodid		Baby clam	
Sinocalanus doerrii adult	Alona	*Limnoithona* spp.		Snail	
Sinocalanus doerrii copepodid	Ilyocryptus	*Limnoithona tetraspina*		*Americorophium*	
Acartiella sinensis	Macrothrix	*Limnoithona sinensis*		Insect larvae	
Acartiella copepodid	Other Cladocera	*Limnoirhon* copepodid		Shrimp	
Tortanus spp.		*Acanthocyclops Vernalis*		Annelid	
Tortanus copepodid		Other Cyclopoids		Fish	
Osphranticum labronectum		*Acanthocyclops robustus* adult		Isopod	
Osphranticum copepodid		*Acanthocyclops robustus* copepodid			
Other Calanoid copepodids					
Other Calanoids					
Sinoclanus doerii					
Pseudodiaptomus spp.					
Eurytemora spp.					

Appendix Table 2 Prey resource taxa in the drift invertebrate categories.

Aquatic Insects			*Aquatic Non-Insects*			*Terrestrial Insects*			*Terrestrial Non-Insects*		
Class	***Order***	***Family***	***Class***	***Order***	***Family***	***Class***	***Order***	***Family***	***Class***	***Order***	***Family***
Insecta	Hydrachnida	Eylaidae	Hirudinea	Rhynchobdellida	Glossiphoniidae		Coleoptera		Arachnida	Acari	
Insecta	Hydrachnida	Lebertiidae	Oligochaeta	Haplotaxida	Enchytraeidae		Diptera		Arachnida	Araneida	
Insecta	Hydrachnida	Sperchontidae	Oligochaeta	Haplotaxida	Lumbricidae		Homoptera		Chilopoda	Geophilomorpha	
Insecta	Coleoptera	Dytiscidae	Oligochaeta	Haplotaxida	Naididae		Hymenoptera		Chilopoda	Lithobiomorpha	
Insecta	Coleoptera	Helophoridae	Oligochaeta	Haplotaxida	Tubificidae		Psocoptera		Crustacea	Isopoda	
Insecta	Diptera	Ceratopogonidae	Oligochaeta	Haplotaxida	Sparganophilidae		Collembola		Diplopoda	Julida	
Insecta	Diptera	Chaoboridae	Polychaeta	Aciculata	Nereidae		Dermaptera		Arachnida	Phalangida	
Insecta	Diptera	Chironomidae	Polychaeta	Canalipalpata	Sabellidae		Diplura		Arachnida		
Insecta	Diptera	Empididae	Malacostraca	Amphipoda	Corophiidae		Hemiptera		Gastropoda		
Insecta	Diptera	Ephydridae	Malacostraca	Amphipoda	Crangonyctidae		Isoptera		Arachnida	Opiliones	
Insecta	Diptera	Psychodidae	Malacostraca	Amphipoda	Gammaridae		Lepidoptera			Araneae	
Insecta	Diptera	Simuliidae	Malacostraca	Amphipoda	Taltridae		Neuroptera		Gastropoda		Stylommatophora
Insecta	Diptera	Tipulidae	Malacostraca	Mysidacea	Mysidae		Thysanura				
Insecta	Ephemeroptera	Caenidae	Crustacea	Ostracoda	Cypridae		Thysanoptera				
Insecta	Ephemeroptera	Baetidae	Malacostraca	Amphipoda	Hyalellidae	Chilopoda		Geophilomorpha			
Insecta	Ephemeroptera	Ephemerellidae	Malacostraca	Decapoda	Astacidae	Chilopoda					
Insecta	Ephemeroptera	Heptageniidae	Actinopterygii	Cyprinodontiformes	Poeciliidae		Ephemeroptera				
Insecta	Hemiptera	Corixidae	Hydrozoa	Anthoathecata	Hydridae		Orthoptera				
Insecta	Hemiptera	Notonectidae	Bivalvia	Veneroida	Corbiculidae	Insecta	Archaeognatha				
Insecta	Odonata	Coenagrionidae	Gastropoda		Physidae	Insecta	Archaeognatha				
Insecta	Odonata	Libellulidae	Gastropoda		Planorbidae	Insecta	Neuroptera	Sisyridae			
Insecta	Trichoptera	Hydropsychidae	Gastropoda		Ancylidae						
Insecta	Diptera	Culicidae	Adenophorea	Dorylaimida	Dorylaimidae						
Insecta	Diptera	Dixidae	Adenophorea	Mermithida	Mermithidae						
Insecta	Coleoptera	Gyrinidae	Enopla	Hoplonemertea	Tetrastemmatidae						
Insecta	Coleoptera	Hydraenidae	Turbellaria	Tricladida	Planariidae						
Insecta	Coleoptera	Melyridae	Arachnida	Hydrachnida	Hygrobatidae						
Insecta	Diptera	Sciomyzidae			Fish						
Insecta	Coleoptera	Staphylinidae		Diptera	Tabanidae						
Insecta	Trichoptera	Leptoceridae	Malacostraca	Isopoda	Asellidae						
Insecta	Ephemeroptera	Ephemeridae	Malacostraca	Decapoda	Crangonidae						
Insecta	Ephemeroptera	Tricorythidae	Clitellata		Branchiobdellidae						
Insecta	Coleoptera	Haliplidae	Oligochaeta	Haplotaxida	Glossoscolecidae						
Insecta	Coleoptera	Hydrophilidae	Hirudinea	Arhynchobdellida	Erpobdellidae						
Insecta	Trichoptera	Psychomyiidae	Arachnida	Hydrachnida	Unionicolidae						
Insecta	Trichoptera	Hydroptilidae	Crustacea	Ostracoda	Candonidae						
Insecta	Odonata	Aeshnidae	Arachnida	Hydracarina	Arrenuridae						
Insecta	Plecoptera	Capniidae	Malacostraca	Decapoda	Palaemonidae						
Insecta	Hemiptera	Veliidae	Gastropoda		Lymnaeidae						
Insecta	Diptera	Muscidae	Crustacea	Ostracoda							
Insecta	Diptera	Stratiomyidae	Arachnida	Trombidiformes	Limnesiidae						
Insecta	Coleoptera	Carabidae	Arachnida	Trombidiformes	Lebertiidae						
Insecta	Odonata	Gomphidae	Arachnida	Acari	Pionidae						
Insecta	Ephemeroptera	Leptohyphidae			Entoprocta						
Insecta	Diptera	Syrphidae	Arachnida	Oribatei							
Insecta	Ephemeroptera	Pseudironidae	Gastropoda	Neotaenioglossa	Hydrobiidae						
Insecta	Hemiptera	Belostomatidae	Arachnida	Trombidiformes	Mideopsidae						
Insecta	Trichoptera	Glossosomatidae									
Insecta	Lepidoptera	Pyralidae	Hirudinea								
Insecta	Coleoptera	Dryopidae	Crustacea	Amphipoda							
Insecta	Coleoptera	Elmidae	Bivalvia	Veneroida	Corbiculoidea						
Insecta	Coleoptera	Ptilodactylidae	Arachnida	Acari							
Insecta	Plecoptera	Perlodidae	Gastropoda								
Insecta	Plecoptera	Taeniopterygidae	Crustacea	Decapoda							
Insecta	Ephemeroptera	Ephemeroptera	Bivalvia	Veneroida	Sphaeriidae						
Insecta	Neuroptera	Sisyridae	Actinopterygii	Cypriniformes	Catostomidae						
Insecta	Trichoptera		Arachnida	Acari	Sperchontidae						
			Actinopterygii	Scorpaeniformes	Cottidae						
			Actinopterygii	Clupeiformes	Clupeidae						
			Actinopterygii	Perciformes	Gobiidae						
			Turbellaria								
			Branchiopoda	Cladocera	Daphniidae						
			Maxillopoda	Cyclopoida							
			Arachnida	Acari	Oribatei						
			Crustacea		Chydoridae						
			Crustacea	Decapoda	Corophiidae						

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Status and Trends

Fish Salvage at the State Water Project's and Central Valley Project's Fish Facilities during the 2014 Water Year

Geir Aasen (DFW) Geir.Aasen@wildlife.ca.gov

Introduction

Two facilities mitigate fish losses associated with water export by the federal Central Valley Project (CVP) and California's State Water Project (SWP). The CVP's Tracy Fish Collection Facility (TFCF) and the SWP's Skinner Delta Fish Protective Facility (SDFPF) divert (salvage) fish from water exported from the southern end of the Sacramento-San Joaquin Delta. Both facilities use louver-bypass systems to divert fish from the exported water. The diverted fish are periodically loaded into tanker trucks and transported to fixed release sites in the western Delta. Operations began in 1957 at the TFCF and in 1968 at the SDFPF.

Methods

This report summarizes the 2014 water year (10/1/2013–9/30/2014) salvage information from the TFCF and the SDFPF, and examines data from water years (WY) 1981 to 2014 for possible relevance to salvage trends in recent years. The following species were given individual consideration: Chinook Salmon (*Oncorhynchus tshawytscha*), Steelhead (*O. mykiss*), Striped Bass[1] (*Morone saxatilis*), Delta Smelt[1] (*Hypomesus transpacificus*), Longfin Smelt[1] (*Spirinchus thaleichthys*), Splittail (*Pogonichthys macrolepidotus*), and Threadfin Shad[1] (*Dorosoma petenense*).

Systematic sampling was used to estimate the numbers and species of fish salvaged at both facilities. Bypass flows into the fish-collection buildings were sub-sampled generally once every 1 or 2 hours for 1 to 60 minutes ($\bar{x}$ = 28.68, sd = 5.10) at the SDFPF, and once every 2 hours for 10 to 120 minutes ($\bar{x}$ = 29.39, sd = 4.27) at the TFCF. Fish 20 mm fork length (FL) or larger were identified, numerated, and measured. These fish counts were expanded to estimate the total number of fish salvaged in each 1- to 2-hour period of water export. For example, a subsample duration of 30 minutes over a 120-minute export period equals an expansion factor of 4, which was multiplied by the number of fish per species collected from the fish count. These incremental salvage estimates were then summed across time to develop monthly and annual species-salvage totals for each facility.

Chinook Salmon loss estimates were presented because the loss model has been widely accepted by regulatory agencies and has undergone extensive review. *Loss* is the estimated number of Chinook Salmon entrained by the facility minus the number of Chinook Salmon that survive salvage operations (California Dept.

[1] Pelagic Organism Decline (POD) species

of Fish and Game 2006). Salmon salvage and loss were summarized by origin (i.e., hatchery fish defined as adipose fin clipped or wild fish defined as non-adipose fin clipped) and race (fall, late-fall, winter, or spring). Race of wild and hatchery Chinook Salmon was determined solely by the Delta Model length-at-date table which is based on length at date of salvage (California Dept. of Fish and Wildlife 2014). It was created by the U.S. Fish and Wildlife Service, which further modified the California Department of Water Resources modified version of the Fisher Model by changing the upper and lower boundaries for winter-run Chinook Salmon. However, apparent growth rates and size ranges among races are variable, leading to potential misclassification with the Delta Model (Harvey and Stroble 2013).

Larval fish were also collected and examined to determine the presence of Delta Smelt and Longfin Smelt < 20 mm FL. Larval sampling at the SDFPF ran from February 24 through June 30 and from March 13 through June 7 at the TFCF. Larval samples were collected once for every 6 hours of water export. Duration of larval samples was the same as the duration for counts. To retain these smaller fish, the fish screen used in the routine counts was lined with a 0.5 mm Nitex net. Larval fish from the TFCF were identified to species by TFCF personnel, and larval fish from the SDFPF were identified to the lowest taxa possible by California Dept. of Fish and Wildlife personnel.

Water Exports

The SWP exported 1.12 billion m^3 of water which was a record low and a marked decrease from exports in WY 2013 (2.70 billion m^3), WY 2012 (3.25 billion m^3), and the record high in WY 2011 (4.90 billion m^3) (Figure 1). The CVP exported 1.17 billion m^3 of water which was a record low and well below exports in WY 2013 (2.27 billion m^3), WY 2012 (2.56 billion m^3), and substantially lower than the near record high in WY 2011 (3.13 billion m^3). The record low exports at both facilities coincided with 2014 being a critical water year and the third straight year of drought conditions in California. Exports in WY 2013–2014 at both facilities were below the WY 1981–2014 average (3.11 billion m^3 at SWP and 2.87 billion m^3 at CVP) while WY 2012 exports were near average at SWP and below average at CVP.

The exports of the two water projects generally followed a similar seasonal pattern. Exports at the SWP peaked in November–December 2013, March 2014, and again in August–September 2014 (Figure 2). During these periods, the SWP exported 684.45 million m^3, which represented 61.0% of annual export. Exports at the CVP peaked in October–November 2013, March–April 2014, and again in September 2014 (Figure 2). During this period, the CVP exported 790.38 million m^3, which represented 67.6% of annual export. CVP monthly exports ranged from 18.28 to 222.39 million m^3. SWP monthly exports ranged from 19.79 to million 152.52 m^3.

Total Salvage and Prevalent Species

Total fish salvage (all fish species combined) at the TFCF was a record low at 160,681 (Figure 3). This was

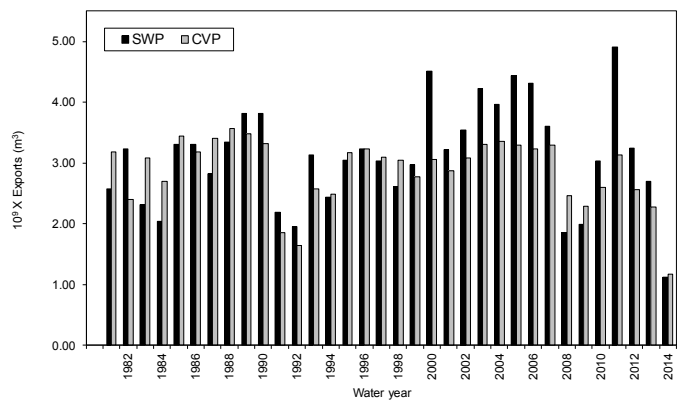

Figure 1 Annual water exports in billions of cubic meters for the SWP and the CVP, water years 1981 to 2014.

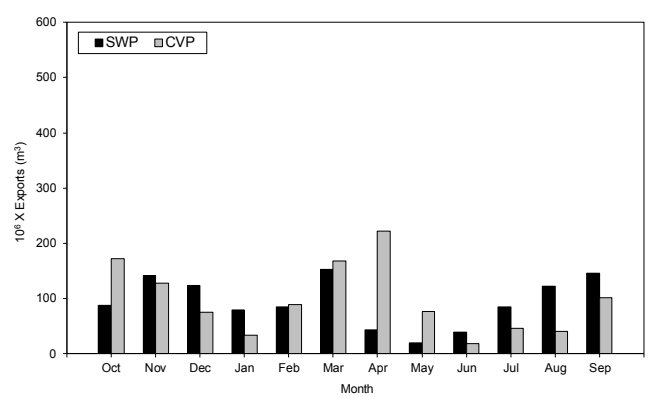

Figure 2 Monthly water exports in millions of cubic meters for the SWP and the CVP, WY 2014.

a marked decrease from WY 2013 (2,828,514) and the previous record low in WY 2012 (475,082) (Figure 3). Total fish salvage at the SDFPF was also a record low at 236,846. This was a marked decrease from WY 2013 (3,042,176) and WY 2012 (1,607,286). The record low total fish salvage at both facilities in WY 2014 was most likely affected by record low exports since salvage in recent years has been influenced by exports (i.e. lower salvage at low exports).

Threadfin Shad was the most salvaged species at SDFPF and Bluegill (*Lepomis macrochirus*) was the most salvaged species at TFCF (Figure 4 and Table 1). Bluegill and Inland Silverside (*Menidia beryllina*) were the 2nd and 3rd most salvaged fish at SDFPF, respectively. Threadfin Shad and Largemouth Bass (*Micropterus salmoides*) were the 2nd and 3rd most salvaged fish at TFCF, respectively. Native species comprised 3.2% of total fish salvage at SDFPF and 2.9% of total fish salvage at TFCF. Relatively few Chinook Salmon, Steelhead, Delta Smelt, and Longfin Smelt were salvaged at the SDFPF (< 0.2% combined of total fish salvage) and at the TFCF (< 1.0% combined of total fish salvage).

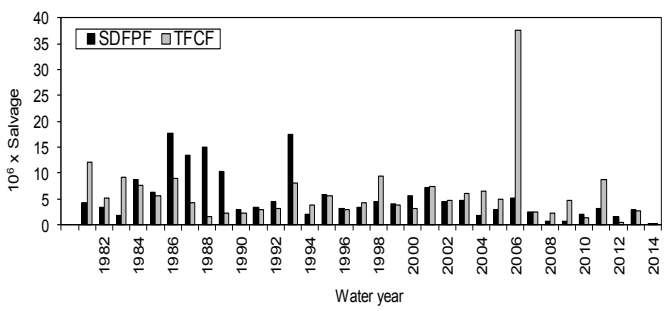

Figure 3 Annual salvage of all fish taxa combined at the SDFPF and the TFCF, water years 1981 to 2014.

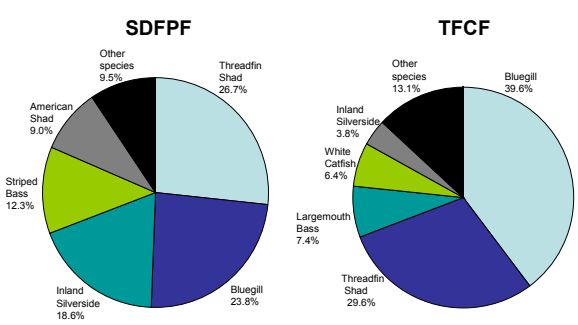

Figure 4 Percentages of annual salvage for the five most prevalent fish species and other fish species combined at the SDFPF and TFCF, WY 2014.

Table 1 Annual fish salvage and percentage of annual fish salvage (%) collected from the SDFPF and TFCF in WY 2014.

SDFPF			*TFCF*		
Species	***Salvage***	***%***	***Species***	***Salvage***	***%***
Threadfin Shad	63,237	26.7	Bluegill	63,667	39.6
Bluegill	56,458	23.8	Threadfin Shad	47,603	29.6
Inland Silverside	44,169	18.6	Largemouth Bass	11,961	7.4
Striped Bass	29,057	12.3	White Catfish	10,261	6.4
American Shad	21,423	9.0	Inland Silverside	6,163	3.8
Prickly Sculpin	5,425	2.3	Striped Bass	5,933	3.7
Largemouth Bass	5,278	2.2	Shimofuri Goby	4,382	2.7
Shimofuri Goby	5,080	2.1	Prickly Sculpin	2,494	1.6
Pacific Herring	1,604	0.7	Golden Shiner	1,367	0.9
Common Carp	1,325	0.6	Chinook Salmon	1,177	0.7
Rainwater Killifish	974	0.4	American Shad	1,080	0.7
Yellowfin Goby	873	0.4	Channel Catfish	972	0.6
White Catfish	533	0.2	Rainwater Killifish	835	0.5
Black Crappie	482	0.2	Black Crappie	667	0.4
Channel Catfish	176	<0.1	Western Mosquitofish	389	0.2
Western Mosquitofish	118	<0.1	Yellowfin Goby	352	0.2
Golden Shiner	101	<0.1	Rainbow / Steelhead Trout	330	0.2
Rainbow / Steelhead Trout	84	<0.1	Redear Sunfish	268	0.2
Lamprey Unknown	81	<0.1	Pacific Herring	204	0.1
Chinook Salmon	64	<0.1	Threespine Stickleback	154	0.1
Delta Smelt	62	<0.1	Pacific Lamprey	144	<0.1
Warmouth	60	<0.1	Black Bullhead	47	<0.1
Splittail	55	<0.1	Bigscale Logperch	35	<0.1
Threespine Stickleback	52	<0.1	Tule Perch	35	<0.1
Longfin Smelt	32	<0.1	Warmouth	32	<0.1
Bigscale Logperch	28	<0.1	Brown Bullhead	30	<0.1
Redear Sunfish	6	<0.1	Lamprey Unknown	24	<0.1
Tule Perch	4	<0.1	Delta Smelt	16	<0.1
Sacramento Blackfish	2	<0.1	Splittail	12	<0.1
California Roach	2	<0.1	Wakasagi	12	<0.1
Starry Flounder	1	<0.1	Longfin Smelt	8	<0.1
			Pacific Brook Lamprey	8	<0.1
			Green Sunfish	7	<0.1
			Sacramento Blackfish	4	<0.1
			White Crappie	4	<0.1
			Blue Catfish	4	<0.1

Chinook Salmon

Salvages of Chinook Salmon (all races and origins combined) at both facilities continued the low salvage trend since WY 2001 (Figure 5). SDFPF salvage (64) was a record low, which decreased substantially from WY 2013 (3,184) and WY 2012 (1,579). Mean WY 2001–2014 SDFPF salvage was about 9% of the mean salvages in the 1980s and the 1990s. Salvage of Chinook Salmon was also a record low at the TFCF (1,177) and decreased from WY 2013 (4,032) and WY 2012 (1,965). Mean WY 2001–2014 TFCF salvage was about 12% of the mean salvages in the 1980s and the 1990s.

Salvaged Chinook Salmon at the SDFPF were primarily wild winter-run–sized fish, which comprised 80.6% of wild fish. Salvaged Chinook Salmon at the TFCF were primarily wild fall-run–sized fish, which comprised 46.5% of wild fish (Table 2). The majority of wild winter-run fish at the SDFPF were salvaged in March and the majority of wild fall-run fish at the TFCF were salvaged in April (Figure 6).

Loss of Chinook Salmon (all origins and races) was higher at the TFCF (827) than at the SDFPF (278) (Table 2). Normally, greater entrainment loss occurs at the SDFPF than at the TFCF due to greater pre-screen loss (California Dept. of Fish and Game 2006). However, greater loss in WY 2014 at the TFCF was attributable to low numbers of fall-run fish and no spring-run fish salvaged at SDFPF. The low numbers of fall-run fish and no spring-run fish salvaged at SDFPF may have been influenced by low exports at SWP in April and May, whereas exports peaked in April at CVP when the majority of wild fall-run fish were salvaged (Figure 2).

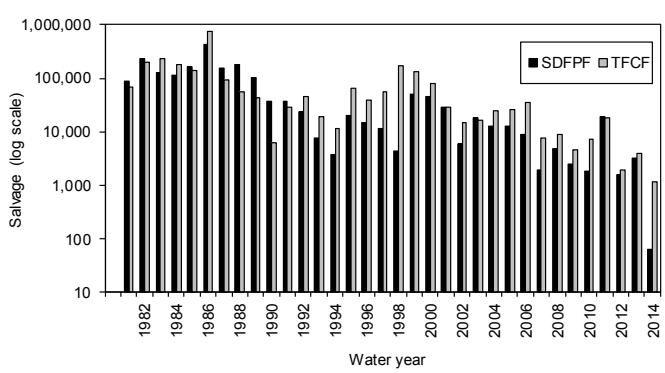

Figure 5 Annual salvage of Chinook Salmon (all races and wild and hatchery origins combined) at the SDFPF and the TFCF, water years 1981 to 2014. The logarithmic scale is log_{10}.

Table 2 Chinook Salmon annual salvage, percentage of annual salvage, race and origin (wild or hatchery), and loss at the SDFPF and the TFCF, WY 2014.

Facility	Origin	Race	Salvage	Percentage	Loss
SDFPF					
	Wild				
		Fall	4	6.5	16
		Late-fall	0	0.0	0
		Spring	8	12.9	33
		Winter	50	80.6	220
	Total Wild		***62***		***269***
	Hatchery				
		Fall	0	0.0	0
		Late-fall	0	0.0	0
		Spring	0	0.0	0
		Winter	2	100.0	9
	Total Hatchery		***2***		***9***
	Grand Total		***64***		***278***
TFCF					
	Wild	Fall	540	46.5	385
		Late-fall	0	0.0	0
		Spring	476	41.0	313
		Winter	141	12.2	118
		Unknown Race	4	0.3	*
	Total Wild		***1,161***		***816***
	Hatchery				
		Fall	0	0.0	0
		Late-fall	0	0.0	0
		Spring	12	75.0	8
		Winter	4	25.0	3
	Total Hatchery		***16***		***11***
	Grand Total		***1,177***		***827***

* No length was taken for Chinook Salmon and consequently race and loss could not be determined.

Steelhead

Salvage of Steelhead (wild and hatchery origins combined) continued the pattern of low salvage observed since WY 2005 (Figure 7). Salvage at the SDFPF (84) was a record low and substantially lower than in WY 2013 (861). Salvage at the TFCF (330) was also lower than in WY 2013 (646).

The SDFPF salvaged 47 hatchery Steelhead and 37 wild Steelhead. The TFCF salvaged 183 hatchery Steelhead and 147 wild Steelhead. Salvage of wild Steelhead at both facilities peaked around the middle of the water year (Figure 8). Wild Steelhead were salvaged most frequently in March at the SDFPF and in April at the TFCF.

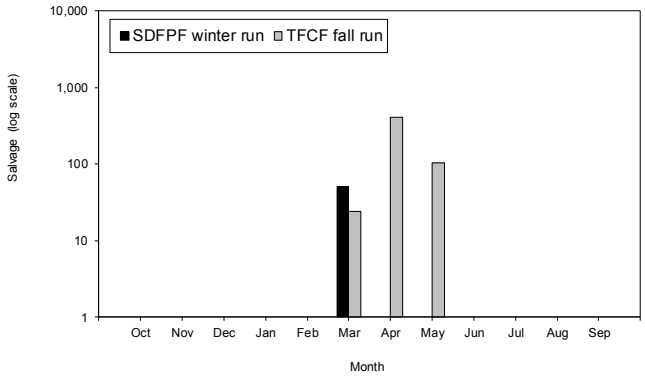

Figure 6 Monthly salvage of wild, winter-run Chinook Salmon at the SDFPF and wild, fall-run Chinook Salmon at the TFCF, WY 2014. The logarithmic scale is $\log_{10}$.

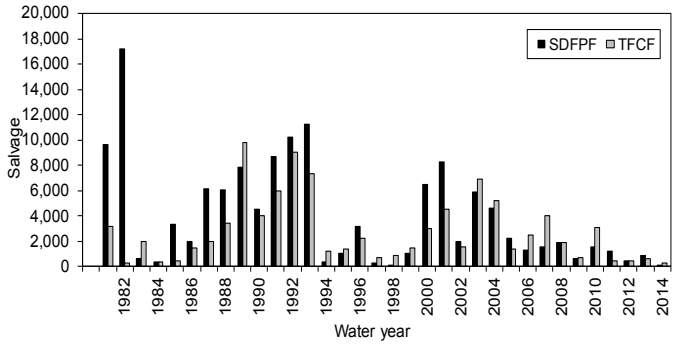

Figure 7 Annual salvage of Steelhead (wild and hatchery origins combined) at the SDFPF and the TFCF, water years 1981 to 2014.

Striped Bass

Salvage at the SDFPF (29,057) and the TFCF (5,933) were both record lows. Salvage at the SDFPF and the TFCF continued the generally-low trend observed since the mid-1990s (Figure 9). Prior to WY 1995, annual Striped Bass salvages were generally above 1,000,000 fish.

Most Striped Bass salvage at the SDFPF occurred in October–November and in June (Figure 10). Most Striped Bass salvage at the TFCF occurred from May–July. Salvage at the SDFPF in October (9,490), November (7,993), and June (4,166) accounted for 74.5% of annual salvage. At the TFCF, salvage in May (3,082), June (1,543), and July (550) accounted for 87.2% of annual salvage. Striped Bass was salvaged every month at both facilities, with the lowest monthly salvage occurring in September at the SDFPF (46) and in December at the TFCF (9).

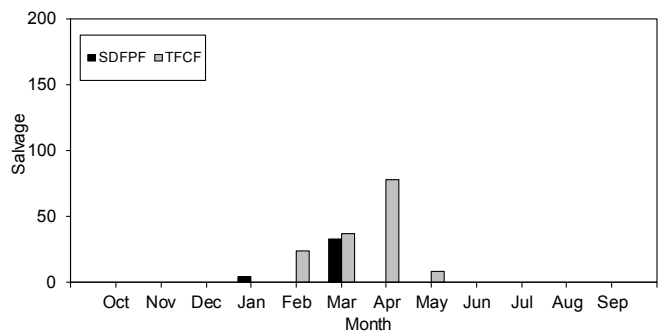

Figure 8 Monthly salvage of wild Steelhead at the SDFPF and the TFCF, WY 2014.

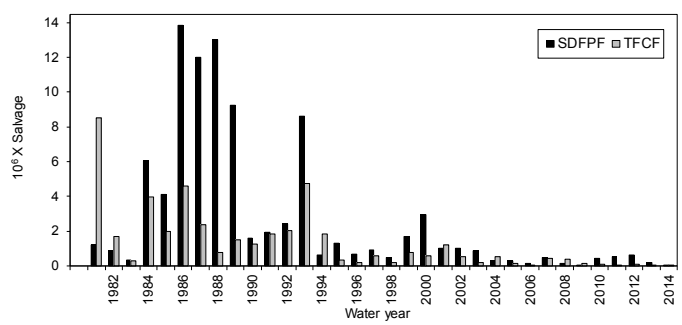

Figure 9 Annual salvage of Striped Bass at the SDFPF and the TFCF, water years 1981 to 2014.

Delta Smelt

Salvage of Delta Smelt continued the pattern of mostly low salvage observed since WY 2005 (Figure 11). Salvage at the TFCF (16) was a record low and decreased from WY 2013 (300) and the previous record low from WY 2011 (51). Salvage at the SDFPF (62) decreased markedly from WY 2013 (1,701) and increased slightly from the record low in WY 2011 (0).

No adult Delta Smelt were salvaged at either facility. Juvenile Delta Smelt at SDFPF were salvaged in April–May, where May salvage (42) accounted for 67.7% of the total WY salvage (Figure 12). Juvenile Delta Smelt at TFCF were also salvaged in April–May, where May salvage (12) accounted for 75.0% of the total WY salvage.

Delta Smelt less than 20 mm FL were first detected at the SDFPF on April 2 and were observed on 10 days of monitoring (Table 3). The longest periods of consecutive daily detections were April 18–19 and April 21–22. April was also the only month with < 20 mm FL detections.

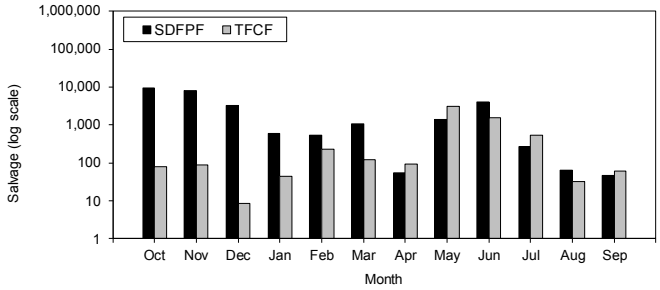

Figure 10 Monthly salvage of Striped Bass at the SDFPF and the TFCF, WY 2014. The logarithmic scale is $\log_{10}$.

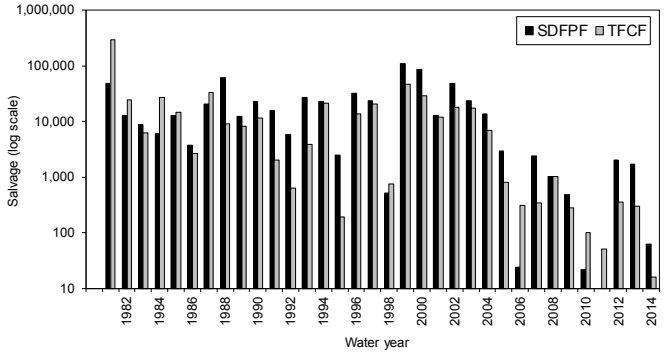

Figure 11 Annual salvage of Delta Smelt at the SDFPF and the TFCF, water years 1981 to 2014. The logarithmic scale is $\log_{10}$.

Delta Smelt less than 20 mm FL were first detected at the TFCF on April 19 and were observed on 5 days of monitoring (Table 3). The longest periods of consecutive daily detections were April 24–25 and May 2–3. April recorded the most daily detections (3 days).

Longfin Smelt

Salvage at the SDFPF (32) decreased from WY 2013 (659) but increased from the record low in WY 2011 (0) (Figure 13). Salvage at the TFCF (8) also decreased from WY 2013 (241) but increased slightly from WY 2011 (4).

Longfin Smelt was salvaged in February–March at the SDFPF (Figure 14). March salvage (28) accounted for 87.5% of the total WY salvage. Longfin Smelt was only salvaged in April at the TFCF which accounted for 100.0% of the total WY salvage.

Longfin Smelt less than 20 mm FL were first detected at the SDFPF on February 24 and were observed on 13 days of monitoring (Table 3). The longest period of consecutive daily detections was from February 24–March 1. March recorded the most daily detections (8 days).

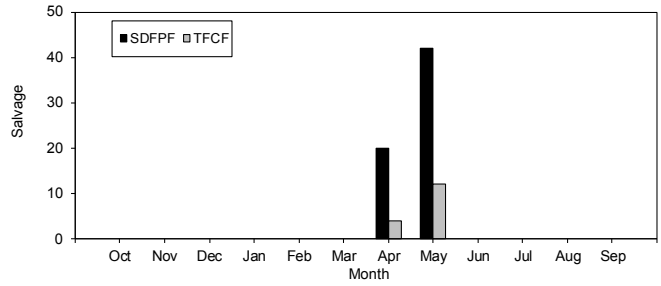

Figure 12 Monthly salvage of Delta Smelt at the SDFPF and the TFCF, WY 2014.

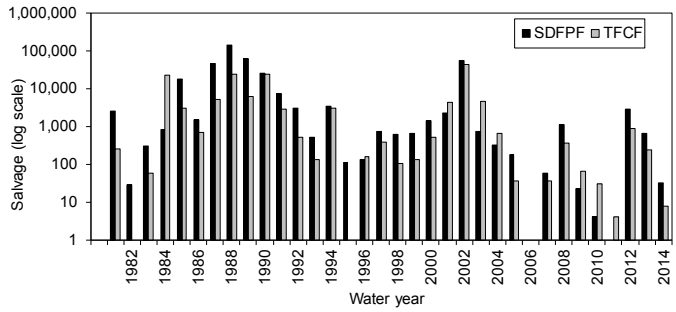

Figure 13 Annual salvage of Longfin Smelt at the SDFPF and the TFCF, water years 1981 to 2014. The logarithmic scale is $\log_{10}$.

Table 3 Smelt less than 20 mm fork length (FL) observed in larval samples collected from the SDFPF and the TFCF in WY 2014. Daily numbers of Delta Smelt and Longfin Smelt < 20 mm FL are recorded while an "N" indicates no detection. An "NS" indicates no sampling.

	SDFPF		*TFCF*	
DATE	*Delta Smelt larvae*	*Longfin Smelt larvae*	*Delta Smelt larvae*	*Longfin Smelt larvae*
2/24/2014	N	7	NS	NS
2/25/2014	N	2	NS	NS
2/26/2014	N	3	NS	NS
2/27/2014	N	1	NS	NS
2/28/2014	N	4	NS	NS
3/1/2014	N	3	NS	NS
3/3/2014	N	1	NS	NS
3/5/2014	N	1	NS	NS
3/7/2014	N	3	NS	NS
3/10/2014	N	1	NS	NS
3/19/2014	N	1	N	N
3/20/2014	N	1	N	N
3/25/2014	N	2	N	N
4/2/2014	1	N	N	N
4/12/2014	5	N	N	N
4/13/2014	N	N	N	1
4/14/2014	1	N	N	N
4/16/2014	2	N	N	N
4/18/2014	2	N	N	N
4/19/2014	1	N	1	N
4/21/2014	1	N	N	1
4/22/2014	2	N	N	N
4/24/2014	N	N	1	N
4/25/2014	N	N	1	N
4/28/2014	1	N	N	N
4/30/2014	4	N	N	N
5/2/2014	N	N	1	N
5/3/2014	N	N	2	N

Longfin Smelt less than 20 mm FL were first detected at the TFCF on April 13 and were observed on 2 days of monitoring (Table 3). April was also the only month with < 20 mm FL detections.

Splittail

Annual salvages of Splittail at both facilities were lower than in WY 2013 (Figure 15). Salvage at the TFCF was a record low (12), which was substantially lower than the previous record-low in WY 2013 (125) and the record high in WY 2011 (7,660,024). Salvage at the SDFPF was a record low (55), which was substantially lower than the previous record-low in WY 2013 (329) and the near record high in WY 2011 (1,326,065). Annual Splittail salvages have followed a boom-or-bust pattern, often varying year to year by several orders of magnitude.

Threadfin Shad

Annual salvage at the SDFPF (63,237) was slightly higher than at the TFCF (47,603) and both were record low salvages (Figure 16). Salvage at the SDFPF was much lower than in WY 2013 (2,535,117). Similarly, TFCF salvage was much lower than in WY 2013 (2,463,695). Similar to Splittail, annual salvages of Threadfin Shad have varied greatly through time.

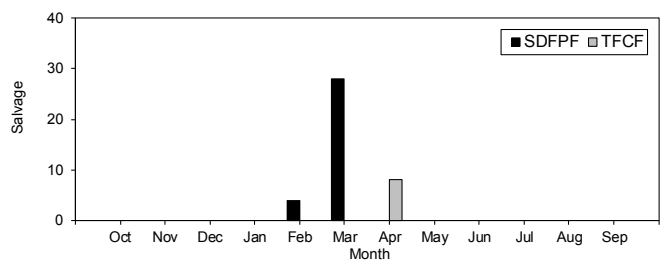

Figure 14 Monthly salvage of Longfin Smelt at the SDFPF and the TFCF, WY 2014.

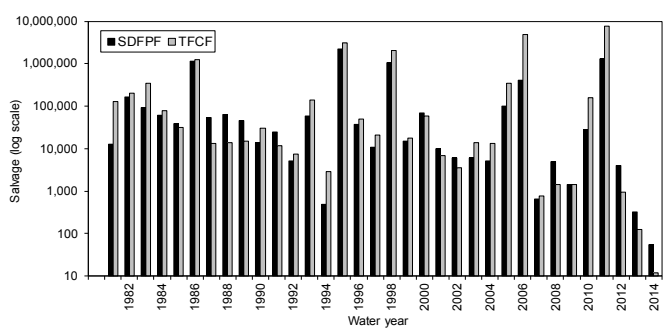

Figure 15 Annual salvage of Splittail at the SDFPF and the TFCF, water years 1981 to 2014. The logarithmic scale is log_{10}.

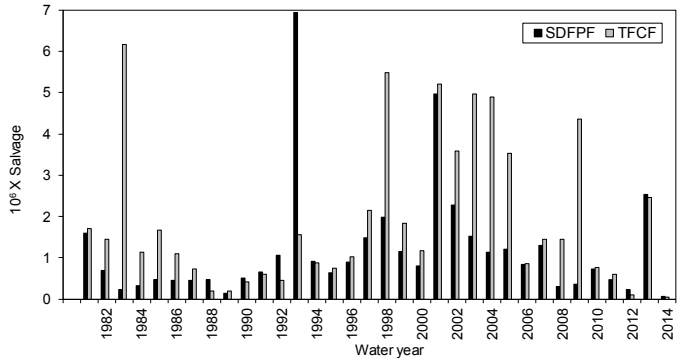

Figure 16 Annual salvage of Threadfin Shad at the SDFPF and the TFCF, water years 1981 to 2014.

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Delta Smelt refuge population: 2014 update and five-year trends in phenotypic traits

Tewdros Ghebremariam (UCD) tghebremariam@ucdavis.edu, Amanda J. Finger (UCD) ajfinger@ucdavis.edu, Luke Ellison (UCD) Luke ellison@ucdavis.edu, Galen Tigan (UCD) gttigan@ucdavis.edu, Joan Lindberg (UCD) jclindberg@ucdavis.edu, and Tien-Chieh Hung (UCD) thung@ucdavis.edu

Background

The University of California, Davis (UC Davis) Fish Conservation & Culture Laboratory (FCCL), in collaboration with the Genomic Variation Laboratory (GVL) of UC Davis, has maintained a refuge population of Delta Smelt (*Hypomesus transpacificus*) for seven generations (Fisch et al. 2009, 2010; Nagel et al. 2013; Lindberg et al. 2013; Ghebremariam et al. 2013). The refuge population was created in 2008 in response to the rapid decline of Delta Smelt abundance estimates. The main goals of the breeding program are to maintain a viable population of Delta Smelt, in captivity, that is phenotypically and genetically identical to the wild population, and to provide Delta Smelt for state and federal research programs and grants. These goals are attained, in part, with intensive genetic monitoring and management (Fisch et al. 2009). In addition, the FCCL collects phenotype information during spawning to detect and monitor any phenotypic changes in the refuge population over time. This article highlights the Delta Smelt breeding program in 2014 and discusses phenotypic (body weight, fork length, and egg number) and genotypic differences between and within the cultured and wild (wild caught and subsequently hatchery reared) Delta Smelt populations.

Spawning and larval survival 2010–2013

The FCCL has developed rearing methods and techniques for the cultured and wild Delta Smelt (Fisch et al. 2009, 2010; Nagel et al. 2013; Lindberg et al. 2013; and Ghebremariam et al. 2013). To maintain the refuge population, approximately 260 pair crosses are made each year between individual males and females. The progeny of one pair cross are full-siblings (FSG). Eggs from approximately eight FSGs are combined in equal numbers to form a "multi-family group" (MFG), which will be hatched and raised together until the following spawning season. In previous years, FSGs consisted of 750 eggs. However in 2014, due to improvements in larval survival over years (Figure 1), the number of eggs was reduced to 700/FSG. With good larval survival, the juvenile thinning introduced in the last two years has resulted in better fish growth and survival.

Tagging fin-clipping the adult fish

Starting two weeks before and continuing throughout the spawning season, tanks are sorted to find adults that are "ripe" (ready or nearly-ready to spawn). The refuge population becomes mature roughly in the order

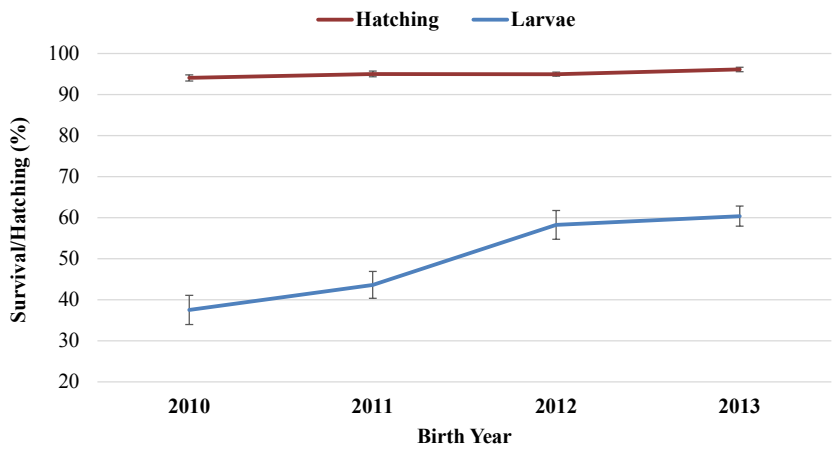

Figure 1 Hatching success and larval survival (40 days post hatch) expressed as a percentage of Delta Smelt multifamily groups (MFGs) during the 2010–2013 spawning seasons.

in which they were hatched. Normally, an abundance of ripe fish are available in the first several MFGs, with the later MFGs becoming ripe later in the season. Even so, all the fish in each MFG are checked to see if they are ripe. A sample of ripe fish (20 fish/MFG/week) is tagged for individual identification via alpha-numeric tags inserted dorsally under the skin. Fin-clips of the tagged fish are sent to the GVL for DNA analysis and pedigree reconstruction for the purposes of making pair crosses. Tagging occurs sequentially for each MFG. In 2014, a total of 2,391 adult fish (982 females : 1,409 males) in the F_6 generation were tagged and fin clipped from January 27 to May 7. The male to female tagging ratio varied weekly based on availability of ripe fish at the time of tagging (Figure 2 and Figure 3). Males were preferentially tagged to increase the options for mating (ripe males can be spawned at any time, while females cannot).

Spawning

Each year the FCCL makes it a priority to supplement the cultured population with wild fish in order to: (1) boost the genetic diversity of the refuge population, (2) reduce inbreeding in the refuge population, and (3) minimize differentiation from the wild population. A total of 100 wild fish were collected (December 5, 2013) from the lower Sacramento River near Sherman Island (N 38° 04' 09.7", W 121° 46' 56.6") with 78% survival. The FCCL preferentially makes crosses between wild and cultured fish (rather than crossing two wild fish), because offspring resulting from pair crosses between two wild fish have historically poor survival and recovery (Ghebremariam et al. 2013), as shown in the following section.

The creation of the F_7 generation from F_6 parents was conducted from January 7 to May 16, 2014, and 248 successful pair crosses were made out 295 total crosses. Of these successful pair crosses, 183 were cultured x cultured, and 65 were cultured x wild crosses. We were successful in spawning and incorporating 87% of all tagged wild fish caught in 2014 into the refuge population. Of note, seven wild females (30% of spawned wild females) had a second clutch at 63 (± 10) days after their first spawn. However, only the first clutch was used in the refuge population.

Recovery

Recovery is defined as having at least one member of a particular family identified and ripe. When all eight FSGs in an MFG are represented, the recovery of that MFG is 100%. Recovery of offspring from each FSG within an MFG is crucial for maintaining the genetic diversity of the refuge population (Fisch et al. 2009). In a typical year there is lower recovery in later-spawned FSGs

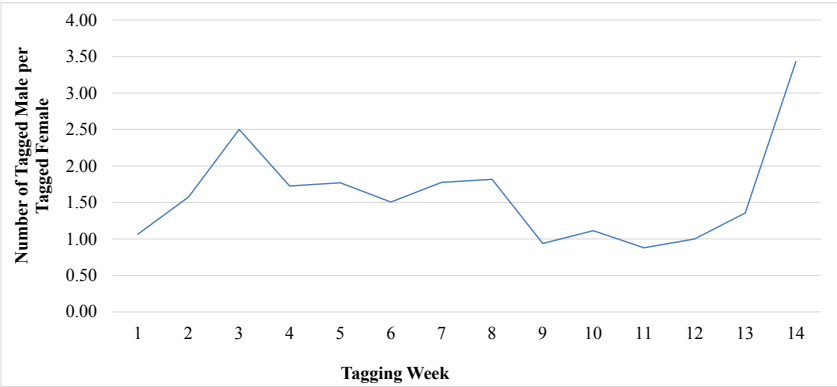

Figure 2 Distribution of tagged male : female ratio for the 2014 Delta Smelt spawning season. Tagging began on 01/27/2014 and ended on 05/08/2014 for a total of 14 tagging weeks.

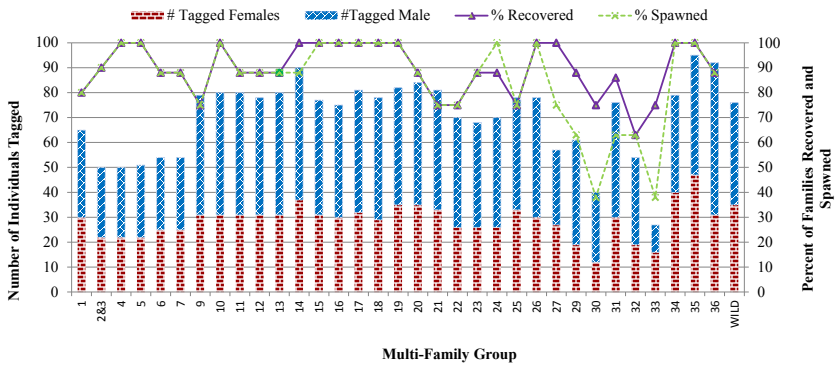

Figure 3 An overview of the 2014 Delta Smelt spawning season, including recovery, tagging, and spawning of MFGs of adults spawned in 2014 (F_6 MFGs). This figure also shows the male : female tagging ratio, and percent of F_6 FSGs recovered and spawned from each F_6 MFG during the 2014 spawning season.

compared to those spawned earlier (Lindberg et al. 2013). Low recovery in later MFGs may be due to a number of factors, including younger age, more fish handling, microenvironment differences (tank effects), and genetics (family effect). In an effort to improve the recovery of all MFGs, the late spawn MFGs are kept at their rearing temperature, 16 °C, longer in the fall and winter, while the first spawned fish are held at the spawning temperature, 12 °C.

In 2014, the 248 successful F_7 FSGs were created from 496 F_6 adult fish (both wild and cultured), and 234 of the 261 F_6 FSGs (90%) were recovered compared to 85% of the F_5 generation in 2013. Of these recovered F_6 FSGs, 222 (85%) were successfully spawned compared to 75 % in 2013 (Ghebremariam et al. 2013). The improvement in recovery may be the result of early termination of the spawning season in 2013, which avoids creating later FSGs that typically have poor recovery (Lindberg et al. 2013).

Genetic monitoring

Genetic Diversity

As with previous years, we used 12 microsatellite loci to conduct both parentage analysis and to analyze genetic diversity of the F_6 generation (the fish that created the F_7 generation in 2014) relative to previous generations at the FCCL. We calculated the mean number of alleles (A), allelic richness (A_r), observed and expected heterozygosity (H_o, H_e), mean inbreeding (*F*), and mean effective population size (mean N_e) for all wild fish, cultured fish, and wild + cultured fish combined from each generation of Delta Smelt at the FCCL (software: GenAlEx v. 6.5, Peakall and Smouse 2012, PMx, Ballou et al. 2010). Allelic richness was calculated with the software program HP-Rare (Kalinowski 2005) with 18 genes per locus (the lowest number of genes per locus in any group analyzed). For the founding generation (F_0), all fish were wild (Table 1).

Genetic diversity statistics indicate successful maintenance of genetic diversity and minimization of inbreeding in the FCCL refuge population. The number

Table 1 Genetic diversity statistics for groups from each generation of delta smelt: wild fish, cultured fish (cult), and wild and cultured combined (comb). Values include number of fish (N), mean number of alleles (A), allelic richness (A_r), observed heterozygosity (H_o), expected heterozygosity (H_e), mean inbreeding (Mean *F*), and mean effective population size (Mean N_e).

Gen.	*Grp.*	*N*	*Mean A*	A_r	H_o	H_e	*Mean F*	*Mean* N_e
F_0	all	290	23.58	9.44	0.84	0.86	0	-
F_1	cult	432	24.83	9.72	0.84	0.87		
	wild	51	15	9.41	0.82	0.84		
	all	494	24.17	9.72	0.84	0.86	0	539
F_2	cult	437	22.25	9.37	0.85	0.85		
	wild	32	16.08	9.43	0.83	0.83		
	comb	469	22.58	9.38	0.85	0.85	0	537
F_3	cult	407	21.33	9.33	0.83	0.85		
	wild	59	15.92	9.29	0.83	0.84		
	comb	466	21.58	9.33	0.83	0.84	0.0011	537
F_4	cult	515	21.33	9.31	0.83	0.85		
	wild	40	15.25	8.96	0.84	0.83		
	comb	555	21.75	9.3	0.84	0.85	0.0016	561
F_5	cult	439	22.17	9.32	0.83	0.85		
	wild	83	19.83	9.63	0.85	0.85		
	comb	522	23.25	9.38	0.83	0.85	0.0012	621
F_6	cult	385	22.17	9.38	0.84	0.85		
	wild	63	17.75	9.55	0.85	0.85		
	comb	448	22.83	9.42	0.84	0.85	0.0006	687
All	all	3,244	22.82	9.38	0.84	0.85	0.0008	2,482

of alleles (A) and A_r are nearly equal between the F_0 generation (A_r = 9.44) to the F_6 generation (A_r = 9.42) (Table 1). Mean *F* is low (when all fish are combined, mean *F* = 0.0008), and H_e and H_o are nearly constant from the founding F_0 generation to the F_6 generation. Further, H_e and H_o values of cultured fish are nearly equal to those of wild fish each year. In addition, mean N_e over all generations has increased due to the yearly incorporation of wild fish, and a concerted effort to equalize family size and founder representation in each generation. Typically, managers of captive populations attempt to maximize N_e, as this measure is an important predictor of the loss of genetic diversity over time (Lande and Barrowclough 1987). A steadily increasing N_e in the captive population is a positive indicator of successful maintenance of overall genetic diversity through intensive genetic management of the captive breeding program.

Genetic differentiation

We calculated pairwise F_{ST} values (F_{ST} is a measure of population genetic differentiation ranging from 0 = no differentiation, to 1 = complete differentiation) in the program FSTAT v 1.2 (Goudet 1995) between generations of wild Delta Smelt and wild + cultured fish combined (Table 2). Significance values were calculated using 1,000 bootstrap repetitions in FSTAT, using the Bonferroni correction for multiple comparisons (adjusted alpha value = 0.0024) (Rice 1989). We found that though there were some significant F_{ST} values between generations of wild fish and generations of cultured + wild fish at the FCCL, FST values are low (when F_{ST} < 0.05, groups show only little differentiation) (Wright 1978) (Table 2). Interestingly, there was significant low differentiation between generations of wild fish introduced into the FCCL. The F_1 generation of wild + cultured fish (hatched in 2008) showed low but significant (p < 0.0024) differentiation from each subsequent generation when wild + cultured fish were grouped. The F_4 generation of wild Delta Smelt (captured in 2011, a year known for a notable boost in the subadult Delta Smelt population abundance index) was also differentiated (p < 0.0024) from all other generations except F_2 and F_3. Differences in FST values between years of wild Delta Smelt are not an indicator of total population differentiation between years, as the captured smelt are merely a subsample of the population at large. However, it must be pointed out that there can be measurable significant, yet low, differentiation. Overall, there is very low differentiation between the founding generation and the current refuge population at the FCCL. While the goal is to minimize genetic differentiation from the wild population, some differentiation is unavoidable due to factors beyond the control of refuge managers, including genetic drift, differential survival, temporal population structure in collected wild fish, and domestication.

Table 2 F_{ST} values measuring genetic differentiation between each generation. Above the diagonal are values for wild and cultured fish combined. Below the diagonal are values for wild fish only. Bold indicates significant F_{ST} value (p < 0.0024) after Bonferroni correction. There were only wild fish in the founding generation (F_0).

	F_0	F_1	F_2	F_3	F_4	F_5	F_6
F_0	-	< 0.001	< 0.001	0.002	0.002	0.006	0.005
F_1	< 0.001	-	0.001	0.002	0.002	0.006	0.005
F_2	0.001	< 0.001	-	< 0.001	< 0.001	0.005	0.004
F_3	< 0.001	< 0.001	< 0.001	-	< 0.001	0.005	0.004
F_4	0.005	0.002	0.004	0.008	-	0.005	0.004
F_5	0.005	0.004	0.001	0.002	0.007	-	< 0.001
F_6	0.004	0.004	0.002	0.003	0.008	0.001	-

Phenotypic monitoring

The first generation of Delta Smelt refuge population was created in 2008 with 164 wild fish (the F_1 generation). We have progressed with the production of the 7th generation (F_7) in 2014. The majority of the broodstock used in the FCCL has been cultured fish (those born in captivity). Supplementation of wild adults accounts for roughly 12% of spawn population over the last five years. Though there has been little genetic differentiation between the refuge population and the wild population (genetic analysis using neutral genetic markers) (Fisch et al. 2009, 2010; Nagel et al. 2013; Lindberg et al. 2013; Ghebremariam et al. 2013), it is important to monitor phenotypic traits (Le Cam et al. 2015) that might help to identify differentiation (on the selected phenotypic trait) between the wild and refuge populations, allowing the hatchery to adaptively manage.

In order to get more insight about the phenotypic differentiation of the refuge population of Delta Smelt at FCCL, we measured fish for selected traits (body

length, body weight, and egg number (fecundity)) over generations. Data from wild born fish were included in the analysis. To compare phenotypic traits across generations, phenotypic data were collected from each parent fish in each spawning season from 2010–2014. After a fish is selected to spawn (but before spawning), each fish is anesthetized. The fork length (FL) is then measured using a measuring board (0.1 mm accuracy), and the body weight (BW) is determined using a digital balance (0.01 g accuracy). Three days after spawning, the fecundity of each fish is volumetrically estimated using the method developed by Lindberg et al. (2013). We compared the BW, FL, and fecundity of fish with various factors, including spawning season (year), sex, and fish origin (wild or cultured). Since the cultured Delta Smelt are normally kept at the FCCL for one year, one season represents one generation of Delta Smelt population. The mean differences were compared using ANOVA, and p values < 0.05 were considered significant.

We compared a total number of 2,892 fish (1,333 : 1,244 cultured males : females, and 115 : 200 wild males : females) spawned between 2010 and 2014. The overall distribution of the BW, FL, and fecundity of all sampled fish suggested unimodal distributions for each of these variables (Figure 4) despite some differences among years, as shown in the following sections.

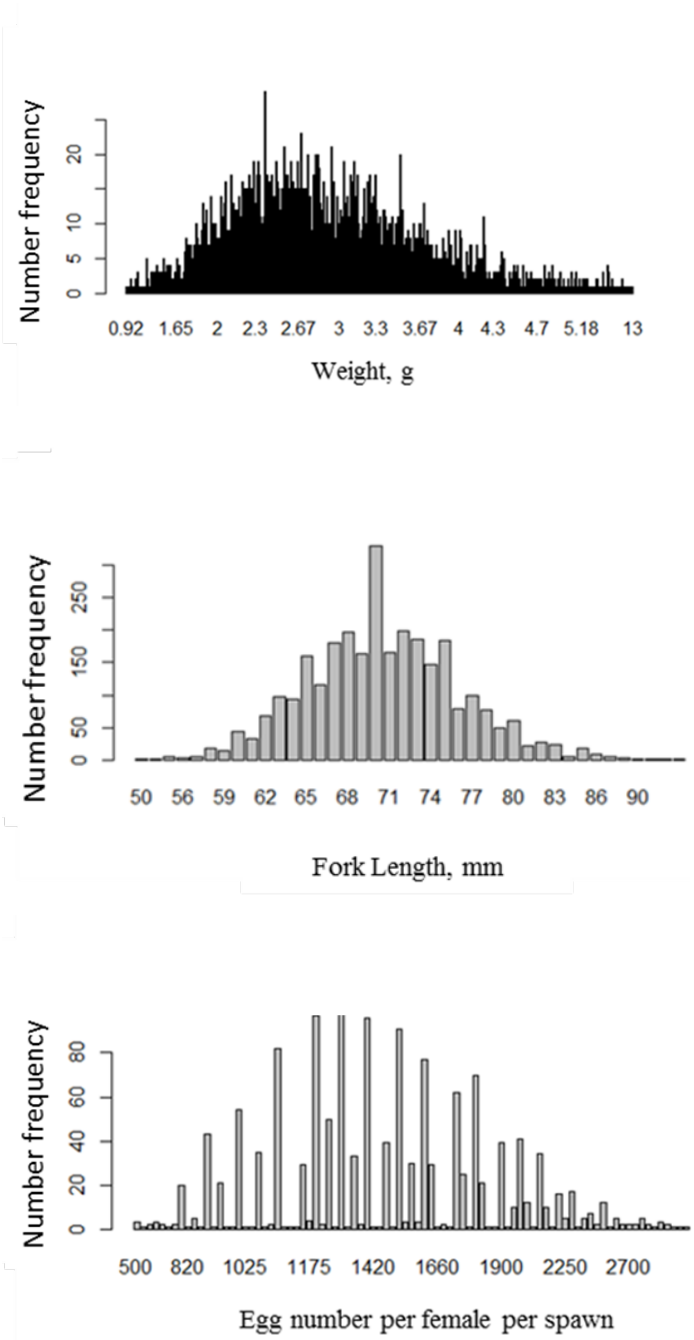

Figure 4 Distribution of body weight (BW), fork length (FL), and egg estimates of spawned Delta Smelt in the refuge population from 2010 to 2014.

Body weight

The preliminary results show that, except for the variation among years, we did not find a consistent difference in weight. The overall (2010–2014) mean weight of cultured Delta Smelt females was 3.32 g (± SD 0.93) and males was 2.63 g (± SD 0.68), while wild female and male were 2.84 g (± SD 0.49) and 2.47 g (± SD 0.61), respectively. Comparison of BW by sex and fish origin did not suggest strong temporal trends (Figure 5). Yet, the BWs of cultured females in the 2011 (3.15 g ± SD 1.07) and 2013 (3.04 g ± SD 0.78) spawning seasons were significantly lower than in the 2010 (3.49 g ± SD 0.93) and 2012 (3.56 g ± SD 1.01) seasons ($p < 0.001$). Mean weight of cultured males was significantly lower in 2013 (2.35 g ± SD 0.61) than in both the 2010 (2.79 g ± SD 0.72) and 2012 (2.84 g ± SD 0.87) spawning seasons ($p < 0.001$). There was no significant difference in BW between wild males and females. In addition, BW differences between cultured and wild fish were not significant, except for cultured females in 2012 and 2014 which were greater than wild females from the same season. However, the analysis of pooled male and female data showed that cultured Delta Smelt from the 2010, 2012, and 2014 spawning seasons had significantly ($p < 0.001$) heavier body weight than wild ones.

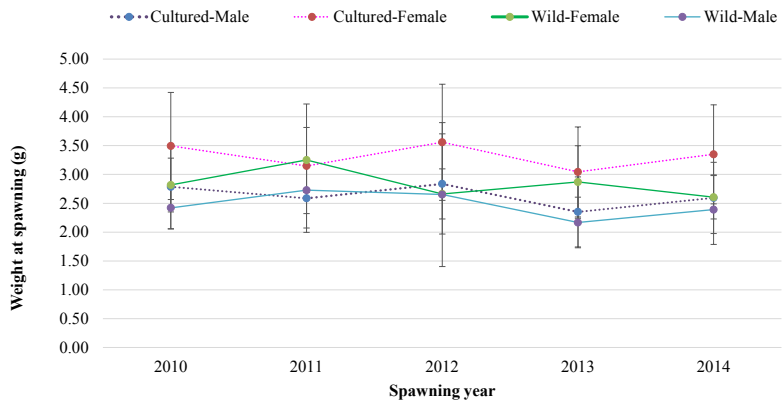

Figure 5 Mean body weight, (BW ± SD, g) of both cultured and wild male and female Delta Smelt spawned from 2010 to 2014.

Fork Length

The FL analyses show a variation among some spawning years (generations) (Figure 6); however, no consistent pattern was evident over the five generations of cultured Delta Smelt. The mean FL for the five generations of cultured female and male Delta Smelt were 71.60 mm (± SD 5.64) and 69.86 mm (± SD 5.36), respectively, while FL for wild fish were 69.16 mm (± SD 3.63) and 68.39 mm (± SD 4.12), respectively. The mean FL of cultured females spawned in 2011 (70.18 mm ± SD 6.20) and 2013 (70.21 mm ± SD 5.05) was significantly shorter than in 2012 (73.04 mm ± SD 6.04) and 2014 (72.55 mm ± SD 5.20) ($p < 0.001$). The mean FL of cultured males in 2013 (67.86 mm ± SD 5.30) was significantly shorter than in 2010 (70.56 mm ± SD 5.54), 2012 (71.22 mm ± SD 6.22), and 2014 (70.21 mm ± SD 5.06). In addition, spawned cultured females were significantly longer than males in the 2012, 2013, and 2014 spawning seasons ($p < 0.001$). No significant difference in FL was found between wild male and female fish. Moreover, spawned cultured fish in 2012 and 2014 were significantly longer than wild Delta Smelt.

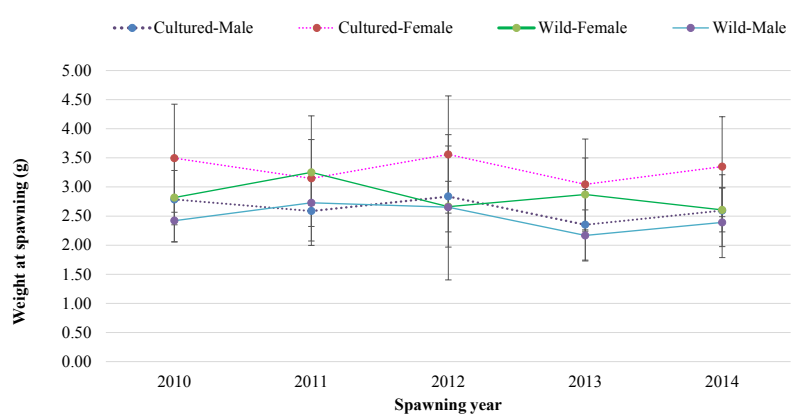

Figure 6 Mean fork length (FL ± SD, mm) of both the cultured and wild males and females Delta Smelt for years 2010–2014.

Fecundity

Our results show no consistent change in fecundity, i.e., the egg numbers produced per female per spawn for the five generations (F_2–F_6) analyzed (Figure 7). The average number of eggs produced in a single egg clutch by a cultured Delta Smelt over five generations is about 1,497 (± SD 415) while the average number of eggs produced by a wild Delta Smelt is about 1,350 (± SD 340). The mean number of eggs produced by cultured females (1,315 ± SD 342) was significantly lower in 2011 ($p < 0.001$) than all the other years. There was no significant difference in egg numbers produced from wild females in the five years. The difference in egg numbers between cultured and wild fish over the five years was not significant ($p < 0.001$), except for 2014, where cultured females produced an average of 1,600 (± SD 438) eggs, and the wild females produced 1,268 (± SD 256).

Conclusion

The overall objective of the refuge Delta Smelt breeding program is to reduce the likelihood of the extinction of the species. To achieve this goal, it is necessary to maintain a refuge population that is both genotypically and phenotypically similar to that of the wild population of Delta Smelt (Fisch et al. 2009; Lindberg et al. 2013). The FCCL and GVL of UC Davis do this by careful selection of parental pair crosses, which minimizes inbreeding, and by supplementing the refuge population with wild fish yearly. The rearing and breeding program for Delta Smelt progressed to a seventh generation, 2,391 fish were tagged, and 248 successful FSGs were

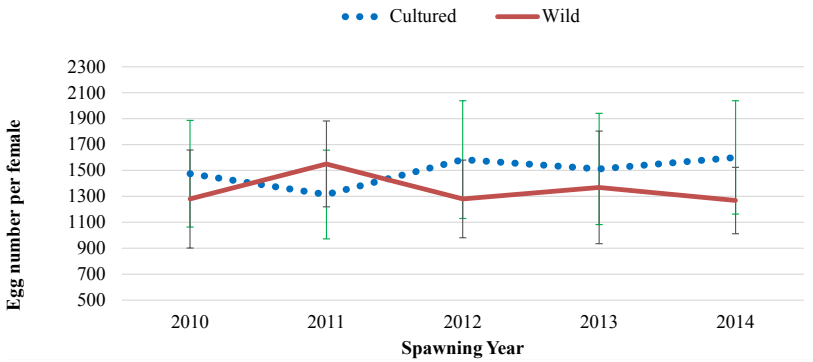

Figure 7 Mean estimated egg number (± SD) spawned per cultured and wild Delta Smelt for years 2010–2014.

made in 2014. The overall numbers of recovered and successfully spawned families were 90 and 85 percent, respectively. The genetic analysis using neutral markers indicates successful maintenance of genetic diversity and minimization of inbreeding in the Delta Smelt refuge population. The differentiation between the founding generation and F_6 population is very low. Our preliminary results of the three phenotypic traits studied indicate no significant departure between cultured fish over the five generations (F_2–F_6) and wild fish. We will continue effectively managing the Delta Smelt refuge population as a safeguard against extinction using the current genetic management methods and objectives.

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