

Big Chico Creek

2020 Fish Population Survey



Prepared By:
Michael Hellmair
Matt Peterson



FISHBIO
180 E 4th Street, Ste. #160
Chico, CA 95928
530.892.9686
www.fishbio.com

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Introduction

Steelhead, the anadromous life history form of rainbow trout (*Oncorhynchus mykiss*), were once abundant throughout California's Central Valley (CV). A combination of anthropogenic factors has resulted in severely reduced abundance of these ocean-going trout, including dam construction, mining, and logging. In addition, modification of creeks and rivers for navigation and flood protection has reduced and degraded available spawning- and rearing habitat (McEwan 2001). As a consequence, steelhead have become relatively rare in the Central Valley, prompting their listing as "Threatened" under the Endangered Species Act (ESA) in 1998, a status that was reaffirmed in 2006, 2011, and 2016.

Though this listing has afforded the species special protection and made resources available to facilitate its recovery, management of steelhead is complicated by the complex life history of the species (Satterthwaite et al., 2010; Kendall et al. 2015). Most populations of *O. mykiss* in anadromous watersheds are partially migratory: some individuals emigrate to the ocean where they grow to adulthood before returning to freshwater habitats to reproduce, while others remain in their natal riverine habitat and reach maturity without undergoing long-distance migrations. For anadromous individuals (steelhead), the food-rich marine environment offers the potential for faster growth, larger size, and higher fecundity. However, migration to the ocean and the typically later age at maturity of these individuals reduce their probability of surviving until reproduction (Fleming and Reynolds 2004). Resident rainbow trout typically mature at a younger age and smaller size, and they have a higher chance of surviving until reproduction, as well as a higher rate of iteroparity (repeat spawning; Fleming and Reynolds 2004, Schill et al. 2010).

Further challenges to fisheries conservation and management stem from considerable plasticity within life-history types. For example, individuals that migrate to sea can do so at various ages, and some may migrate out to the Sacramento-San Joaquin estuary or San Francisco Bay and return to spawn without spending any time in the open ocean (Teo et al. 2011, Null et al. 2012). Additionally, in rivers where steelhead and rainbow trout are sympatric, migratory and resident forms interbreed and may produce offspring with a life history different from their own (Zimmerman and Reeves 2000, Heath et al. 2008, Zimmerman et al. 2009, Christie et al. 2011, Courter et al. 2013). These various aspects of the species' complex and variable life history illustrate the difficulty in assessing steelhead population viability in the Central Valley.

Management of steelhead depends on the relative prevalence of the migratory and resident polymorphisms in a population. Within the CV Distinct Population Segment (DPS), it may not be possible to manage one life-history morph without reference to the other (Williams et al. 2007). Without information regarding the abundance of *O. mykiss* or the prevalence of various life-history morphs, it is difficult to examine how changes in the environment may affect the population abundance as a whole. Consistent and robust population monitoring is necessary to document trends and natural variation in *O. mykiss* abundance and to understand whether certain actions may negatively or positively affect population size (Eilers et al. 2010). While the life history plasticity

of *O. mykiss* raises substantial challenges for management and recovery of the anadromous population segment, it unequivocally underlines the importance of including resident rainbow trout in status assessment and recovery planning of anadromous steelhead.

While comprehensive monitoring plans are in place to track and assess most larger remaining populations of other anadromous salmonids (typically Chinook salmon, *Oncorhynchus tshawytscha*), nearly all of the 81 historical populations of steelhead in the CV are considered data deficient (Lindley et al. 2006, Lindley et al. 2007, National Marine Fisheries Service 2009). Despite, or perhaps as a result of, management focus on anadromous salmonids, other native species are often only given ancillary consideration in assessment of fish populations in California. However, it has been apparent for decades that the decline of native fish fauna in lotic waters of inland California has been paralleling that of Central Valley steelhead (Moyle and Nichols 1974, Moyle and Williams 1990). To alleviate data deficiencies associated with abundance of *O. mykiss* and to track the distribution and demographic characteristics of other native species, the study of smaller watersheds that remain relatively undeveloped is of great importance. The study of these systems can help provide insights regarding large-scale population trends and patterns that may be masked by the effects of localized changes in riverine conditions resulting from infrastructure developments (i.e., dam construction and operation).

Big Chico Creek is well suited for such an assessment as it supports populations of both steelhead and resident rainbow trout in the 24 miles of the stream accessible to anadromous fish (USFWS AFRP; Figure 1). No large water storage infrastructure exists on Big Chico Creek, and a natural barrier (Higgin's Hole) blocks the upstream migration of anadromous fish species. Past efforts to enhance the salmonid populations in Big Chico Creek have included a rotenone treatment (a piscicide) by the California Department of Fish and Wildlife (CDFW; then California Department of Fish and Game), applied in 1986 between Higgin's Hole and Iron Canyon, in response to perceived competition from non-game fish. This treatment was followed by the release of large numbers of hatchery-reared Chinook salmon and steelhead fry for several years. Following the toxic treatment, Dr. Paul Maslin (California State University, Chico) and his students conducted annual surveys in the affected reach from 1986 to 1998. The overall effectiveness of the rotenone treatment for restoring anadromous fish populations remains unclear. While populations of rainbow trout generally increased slightly over the study period, this may be attributable to the large stocking efforts and stochastic weather events (i.e., winter flooding) rather than the result of reduced competition/predation by native species (Maslin 1997a). Native non-game fishes were extremely slow to re-colonize the affected area of the creek, and only California roach (*Hesperoleucus symmetricus*) have been observed at high abundances since the treatment (Maslin 1997a). Riffle sculpin (*Cottus gulosus*) later rebounded close to pre-treatment levels, with considerable recruitment being observed after the 1997 flood events (Maslin 1997a).

To address the increasingly recognized need to incorporate the resident rainbow trout populations in status assessment and management of the steelhead CV ESU, and to assess the demography and distribution of native fish species in the upper anadromous reaches of Big Chico Creek, we conducted quantitative assessments of Big Chico Creek fish populations in the summers of 2013,

2014, 2018, and 2019. The survey in 2014 covered the full extent of over-summering habitat in the watershed accessible to anadromous species, whereas the surveys in 2013, 2018 and 2019 covered the extent of Big Chico Creek within the boundaries of the Big Chico Creek Ecological Reserve (BCCER). Abundance estimates reported herein are based on direct observation dive counts (i.e., snorkel surveys), a cost-effective, non-invasive method of estimating abundance. It does not require fish handling and can provide counts similar to depletion electrofishing under conditions such as those found on Big Chico Creek during the summer months (Mullner et al. 1998, Allen and Gast 2007).

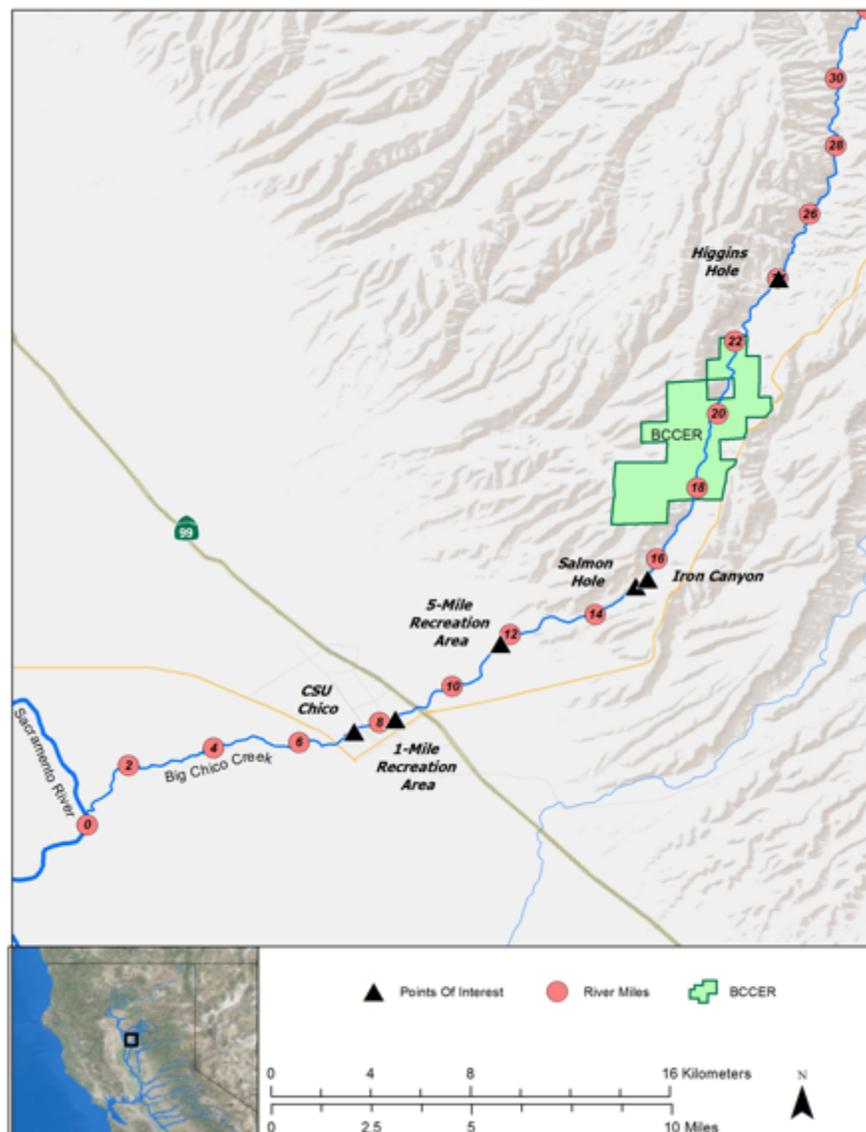


Figure 1. Map of the Big Chico Creek watershed.

Material and Methods

Study Site

Big Chico Creek originates on the western slope of Colby Mountain, at an elevation of 5,400 feet, and flows 45 miles to its confluence with the Sacramento River. It is one of several small eastside tributaries to the Sacramento River (along with Butte -, Deer -, Mill - and Antelope creeks) with comparable topography and annual discharge patterns. All of these creeks flow into the Sacramento River within approximately 40 miles of one another, are mostly undammed, and are all considered high-priority watersheds for conservation and restoration of anadromous fish populations.

Big Chico Creek can be roughly divided into three different zones, based on both, geological barriers and the composition of the fish community: valley zone, foothill zone, and mountain zone.

The valley zone is the lowermost zone in the watershed and extends from the confluence with the Sacramento River upstream to Iron Canyon, located in Upper Bidwell Park (Figure 1). In this narrow canyon, as the creek flows over a geologic formation known as the Lovejoy basalt, years of erosion have resulted in an assemblage of large basalt boulders in the middle of the creek. The arrangement of these boulders has formed impassable barriers to anadromous fish during typical flows, but during high flows, upstream migration past Iron Canyon is possible (DWR 2002). The fish community in the valley zone is dominated by introduced centrarchids (black bass and sunfishes, *Micropterus spp.* and *Lepomis spp.*, respectively), native Sacramento pikeminnow (*Ptychocheilus grandis*), hardhead (*Mylopharodon conocephalus*), and Sacramento sucker (*Catostomus occidentalis*). The valley zone does not provide much suitable spawning habitat for salmonids, has larger populations of predatory fish, and experiences seasonally warm water temperatures in excess of the physiological tolerance of salmonids (BCCWA 1997).

The foothill zone extends upstream from Iron Canyon to Higgin's Hole, where a large waterfall forms the upstream barrier to anadromous fish migration on Big Chico Creek (though it may be possible for spring-run salmon and steelhead to navigate past this waterfall during unusually wet years). The timing of high flows and fish migrations has a significant effect on the accessibility of the foothill zone to various fish species. Although a fish ladder was built in Iron Canyon to permit more frequent access to this the foothill zone in the 1950s, years of deterioration in absence of maintenance have rendered it ineffective. Steelhead, migrating predominantly between November and February, can typically overcome this partial migration barrier. Other species, such as spring- and fall-run Chinook salmon (with different migration times), often have difficulty accessing this section of the creek (DWR 2002). Historically, anadromous fishes dominated the foothill zone, and Chinook salmon, steelhead, and Pacific lamprey (*Lampetra tridentata*) were prominent in this reach. Populations of native cyprinids, including hardhead, Sacramento pikeminnow, and California roach (*Hesperoleucus symmetricus*), as well as brown trout (*Salmon trutta*), Sacramento sucker, and riffle sculpin were also found in the foothill zone (Maslin 1997a, BCCWA 1997). It is

unclear whether the resident species mentioned above can migrate upstream through Iron Canyon; however, the apparent lack of recolonization of the foothill zone following the rotenone treatment suggests that these resident species have difficulty accessing this area from the valley zone. No hardhead and only two Sacramento pikeminnow were observed in the study area after the treatment, though limited numbers of Sacramento suckers were documented after the treatment (all less than 300 mm in length; Maslin 1997a). Between 1987 and 1991, over 1.5 million Chinook salmon fry and several hundred thousand steelhead fry (Feather River stock) were planted in the foothill zone (just below Higgin's Hole) to bolster populations of these species following the piscicide treatment (BCCWA 1997).

The mountain zone extends from Higgin's Hole upstream to the headwaters of Big Chico Creek and only supports resident rainbow and brown trout. In the small headwaters, where tributaries are spring-fed and most precipitation falls as snow, winter flooding is not severe. This is thought to favor the reproductive success of exotic brown trout (over native rainbow trout), which have come to dominate the uppermost reaches of the Big Chico Creek watershed and are typically the only fish species found in this area (Maslin 1997b).

The Big Chico Creek Ecological Reserve, a 3,950-acre parcel of land which encompasses an estimated four and a half miles of Big Chico Creek, is located in the foothill zone about two miles downstream of Higgin's Hole (Figure 1). The BCCER was chosen as the location for this survey as it provides accessible habitat along a relatively large section of the creek located directly below the barrier to anadromy. This portion of Big Chico Creek likely provides the highest quality habitat for anadromous fish in the watershed.

Habitat Mapping and Unit Selection

In order to obtain an accurate estimate of fish abundance, the entire reach of Big Chico Creek between the downstream boundary of the BCCER and Higgin's Hole was surveyed on foot and categorized into habitat units based on a four-category classification in 2013 (i.e., riffle, run, pool, and cascade). Global Positioning Satellite (GPS) waypoints were taken at the boundaries of each habitat unit using a handheld Garmin® GPS unit (Garmin International Inc., Olathe, KS) in order to accurately locate each habitat unit during subsequent surveys. In addition, the length and width of each unit was measured with a Bushnell® rangefinder (Bushnell Outdoor Products, Overland Park, KS), and the maximum water depth of each unit was determined with a stadia rod. Other measurements recorded during habitat mapping included dominant substrate, dominant cover type, and presence of large woody debris. Stream sections classified as "cascades" are often hazardous or do not permit sufficient visual coverage due to turbulence and were excluded from this survey. Classification and size of distinct habitat units was verified during surveys in subsequent years. Of note, classification of individual units, their length, and prominent characteristics and/or landmarks used to identify the units in the field have remained consistent, despite several high-flow events following heavy and prolonged precipitation in the 2016/2017 and 2018/2019 wet seasons.

Within each habitat category (i.e. “stratum”) conducive to visual surveys (run, riffle, pool), units were sampled systematically by generating a random number between 1 and 5, and subsequently surveying every k^{th} unit in an upstream direction. A sub-sample of the surveyed units was randomly selected for calibration of dive counts using the Method of Bounded Counts (MBC), as described in more detail below.

According to our classification, the reach of Big Chico Creek between the upstream and downstream boundaries of BCCER consists of 208 distinct habitat units (51 pools, 55 riffles, 73 runs, and 29 cascades; Table 1). Snorkel surveys were conducted in 9 pools, 15 runs, and 10 riffles in 2020. Additionally, 16 of the 34 surveyed units were selected for bounded counts.

Table 1. Habitat composition and percentage surveyed during snorkel surveys conducted on Big Chico Creek in August/September 2020.

Habitat Type	Count of Type	Sum of Length (m)	Percent by Length	Units Surveyed	Length of Units Surveyed (m)	Percent of Type Surveyed
Pool	51	3010	41.8	9	443	14.72
Riffle	55	1483	20.6	10	279	18.81
Run	73	2230	30.9	15	476	21.35
Cascade	29	484	6.7	0	0	0
Total	208	7207	100	34	1198	16.62

Dive Counts

There are numerous methods for estimating the total abundance of fish in freshwater systems, the majority of which depend on handling the fish during enumeration (e.g., electrofishing, seining etc.). Direct observation dive counts (i.e., snorkel surveys) are a cost-effective, non-invasive means of estimating abundance based on visual counts, which do not require fish handling (Allen and Gast 2007). Therefore, this method is preferred for species of special conservation concern. In some situations, when water visibility is excellent and conditions are good, snorkeling can provide counts similar to depletion electrofishing (Mullner et al. 1998). However, visual estimates are typically negatively biased (i.e., underestimation) due to imperfect detection (e.g., visibility, temperature, time of day, species-specific behaviors and fish size) (Northcote and Wilkie 1963 as cited by Hagen and Baxter 2005, Mullner et al. 1998, Bradford and Higgins 2001, Hagen and Baxter 2005, O’Neal 2007, Hagen et al. 2010). Therefore, without estimates of observer bias (which generally require depletion estimates of abundance for a subsample of the reaches under study), single-pass snorkel surveys cannot provide an estimate of absolute abundance. Rather, they provide an unbiased index of abundance with associated confidence intervals. A viable alternative to obtaining accurate population size estimates by traditional methods (such as depletion electrofishing or mark-resighting experiments) is the Method of Bounded Counts. This approach relies on repeated counts of fish from the same unit (generally four passes) and produces nearly unbiased estimates of abundance if fish abundance in respective survey units is relatively low

(Mohr and Hankin 2005). As such, this method provides a non-invasive (no fish handling required) alternative to traditional methods that is highly applicable to stream surveys involving species of special concern.

Snorkel surveys were conducted on August 31 – September 2, 2020. A standardized protocol was followed to ensure comparability of survey results with previous and future results and to minimize variation due to sampling error. The number of divers needed for a snorkel survey was dependent on the width of the stream, but was chosen to ensure complete visual coverage of the stream during upstream snorkeling. If the stream section to be surveyed required more than two divers for complete visual coverage of the stream width, parallel dive lanes were established prior to snorkeling. Dive lanes were assigned randomly to divers at each survey unit to minimize the effects of diver familiarity with the physical habitat and fish population on dive counts. Care was taken to minimize disturbance of fish prior to sampling each unit.

Divers entered the stream at the downstream border of the survey reach and counted fish within their respective dive lanes as they proceeded upstream in unison with the other divers. Divers recorded fish counts on a wrist-mounted dive slate and assigned a size category to each observation (less than 150 mm, 150-300 mm, and greater than 300 mm). Divers were equipped with two reference dowels (150 mm and 300 mm in length) to facilitate the correct estimation of fish size and account for underwater size distortion. When approaching the upstream boundary of the survey unit, divers carefully monitored fish holding close to the unit boundary and included fish that crossed the unit boundary in an upstream direction. Any fish that was observed moving between lanes was noted immediately after the dive to avoid multiple counts of the same fish. To minimize potential observer bias during all snorkel passes, the units selected for additional passes were not revealed to the divers until the first dive pass was completed. In sampling units that were selected for calibration of single-pass dive counts, a minimum of five minutes was allowed to elapse between each of the three subsequent dives.

Obtaining accurate counts of *O. mykiss* and *S. trutta* was the priority of this survey. Other observed species (and their lengths) were recorded, so long as this did not compromise counts of the focal species.

Fish Abundance

To estimate total abundance of focal fish species, a two-phase estimator was used in each stratum surveyed (runs, riffles, and pools) to “calibrate” single-pass counts. Error in abundance estimation can occur in the first and second phase of estimation, termed sampling error and measurement error, respectively. Error that occurs in the first phase is called sampling variance, which results from selecting any sample from a sampling universe. Sampling variance can be minimized by selecting an adequately large number of samples from all units that are available in a given stratum. In the second phase (in units selected for bounded counts), there is error associated with the

measurement of any particular unit abundance (measurement error or precision) due to variation of dive counts within units surveyed multiple times.

For each unit selected for a bounded count (multiple passes), individual pass counts were ordered from highest to lowest, and unit abundance was estimated as

$$\tilde{y}_{Bk} = d_m + (d_m - d_{m-1})$$

where \tilde{y}_{Bk} = the bounded count estimate of “true” abundance in unit k , d_m is the largest of the four counts for the unit, and d_{m-1} is the second largest of the four counts.

For example, if a unit was snorkeled four times with pass counts of 6, 7, 9, and 6 fish, the ordered counts would be 9, 7, 6, and 6. The difference between the highest count (9) and the next highest count (7) is 2, which is added to the highest pass count of 9, for an abundance estimate of 11 fish in the unit.

The estimate of error, or mean square error (MSE), around the unit abundance estimate was calculated as

$$\widehat{MSE}_{\tilde{y}_{Bk}} = (d_m - d_{m-1})^2$$

In the preceding example, the MSE would equal the squared difference between the highest count (9) and the next highest count (7), which would equal 4. The 95% confidence intervals would be twice the square root of MSE, again, which would equal 4, for a final unit abundance estimate of 11 ± 4 (7 – 15).

For each stratum in which surveys were conducted, the total stratum abundance (\hat{Y}_D) is estimated as

$$\hat{Y}_D = N \bar{\tilde{y}}_{BD} \frac{\bar{x}_1}{\bar{x}_2}$$

where N is the total number of habitat units within stratum D , and $\bar{\tilde{y}}_{BD}$ is the mean estimated total abundance for all units in stratum D for which bounded counts were performed. The last term in the equation is the mean of the first pass counts in habitat units that were dove only once (\bar{x}_1) divided by the mean of the first pass counts in habitat units that were dove four times (\bar{x}_2). This is an adjustment factor that accounts for the observation probability during the snorkel surveys (i.e., the difference between a unit abundance derived from a single-pass survey versus a four-pass survey).

Estimates of error around the total stratum abundance were calculated as

$$\hat{V}(\hat{Y}_D) = N^2(1 - f_1) \frac{s_{\tilde{y}}^2}{n_1} + N^2(1 - f_2) \left(\frac{\bar{x}_1}{\bar{x}_2}\right)^2 \frac{s_{\tilde{y}|x}^2}{n_2}$$

where f_1 and f_2 are the sampling fractions for the first and second phases, respectively; n_1 and n_2 are the numbers of units that are sampled in the first and second phases, respectively. The variation in the unit counts in the first phase, $s_{\tilde{y}}^2$, was calculated as

$$s_{\tilde{y}}^2 = \frac{1}{n_2 - 1} \sum_{k=1}^{n_2} (\tilde{y}_{Bk} - \bar{\tilde{y}}_{BD})^2$$

where \tilde{y}_{Bk} is the estimated abundance in the k^{th} second phase sample and $\bar{\tilde{y}}_{BD}$ is the mean abundance over all second phase samples in stratum D . The conditional variation (i.e., variation that arises from selecting particular second phase samples), $s_{\tilde{y}|x}^2$, was calculated as

$$s_{\tilde{y}|x}^2 = \frac{1}{n_2 - 1} \sum_{k=1}^{n_2} \left[MSE_{\tilde{y}_{Bk}} + (\tilde{y}_{Bk} - \bar{\tilde{y}}_{BD} \frac{x_{Bk}}{\bar{x}_2})^2 \right]$$

where x_{Bk} is the first pass dive count in unit k .

Sampling under a stratified design such as the one employed in this study is considered independent across the different habitat strata (run, riffle, pool; $D = 1, 2, 3$), so that estimates of total abundance for each of the habitat types, \hat{Y}_D , and their corresponding sampling variances, $\hat{V}(\hat{Y}_D)$, can be combined across strata (Thompson 2002):

$$\hat{Y} = \sum_{D=1}^3 \hat{Y}_D$$

and

$$\hat{V}(\hat{Y}) = \sum_{D=1}^3 \hat{V}(\hat{Y}_D)$$

Notably, though bias of this method is considered negligible at low abundances (less than approximately 30 individuals per unit), special scenarios can lead to a failure of this estimator. More specifically, at very low abundances of the target species, failure to observe the species (or size category) during the first pass of (all) bounded counts in a given stratum results in a zero in the denominator of the count ratio between single- and bounded count units (i.e., \bar{x}_2 , see formula for estimation of total stratum abundance). The chance of estimator failure (or unrealistic estimates) increases with the number of size classes and habitat categories for which abundance is to be estimated. In other words, the chance of observing *at least one* individual of a particular species during one or more first-pass MBC counts for a given habitat category is relatively high (given that a sufficiently large number of units are selected for bounded counts). In contrast, the chance of observing individuals belonging to *each* of several size classes is lower, which increases the chance of estimator failure for a given size class and habitat category.

Results

Fish Abundance

Overall, five species of fish were observed during the Big Chico Creek snorkel survey in 2020, namely rainbow trout, brown trout, riffle sculpin, Sacramento sucker and California roach. With the exception of California roach, which were too numerous to count in nearly every surveyed habitat, rainbow trout were the most abundant species observed, followed by Sacramento sucker, brown trout and riffle sculpin. We observed 157 rainbow trout, 35 Sacramento sucker, three brown trout, and three riffle sculpin during the first pass of snorkel surveys (Figure 2). No spring run Chinook salmon were observed; Henning Hole, where spring run Chinook have been observed in the past, was not snorkeled, but observed from an elevated position on the south bank. We estimated that there were approximately 1,451 rainbow trout in the reach of Big Chico Creek within the study area, or approximately 347 fish per mile. Note that this estimate of total abundance was calculated by summing estimates of individual size classes (see Materials and Methods - Fish Abundance). Estimates could not be calculated for the other species due to the low number of observations. Estimates of abundance do not account for cascade habitat units that were not sampled due to safety concerns and poor visibility. This habitat type accounted for 6.7 percent of the total length of the stream within the study area (Table 1).

Overall, the majority of rainbow trout were observed in riffles (45%), followed by pools (28%) and runs (27%). We estimated that there were approximately 649 trout inhabiting riffles, 396 rainbow trout inhabiting runs, and 406 rainbow trout inhabiting pools in the study area (Figure 3).

When the distinct size categories are taken into consideration for abundance estimation, we estimated that there were 634 juvenile rainbow trout (< 150 mm), 617 rainbow trout between 150 and 300 mm in length, and 200 rainbow trout larger than 300 mm in the study area (Figure 4). The three brown trout we observed were juveniles.

In addition to the fish species mentioned in preceding paragraphs, we also observed four Western pond turtles (*Actinemys marmorata*) during our survey. They were observed in all surveyed habitat types but were most frequently seen in pools.

Table 2. Percentage of habitat units in which each species (all size classes combined) was observed during snorkel surveys conducted on Big Chico Creek during August 31 – September 2, 2020.

Common Name	Scientific Name	Pool	Riffle	Run
Rainbow trout	<i>Oncorhynchus mykiss</i>	80.0	80.0	88.9
Brown trout	<i>Salmo trutta</i>	0	0	13.3
Riffle sculpin	<i>Cottus gulosus</i>	0	0	13.3
California roach	<i>Hesperoleucus symmetricus</i>	100.0	100.0	100.0

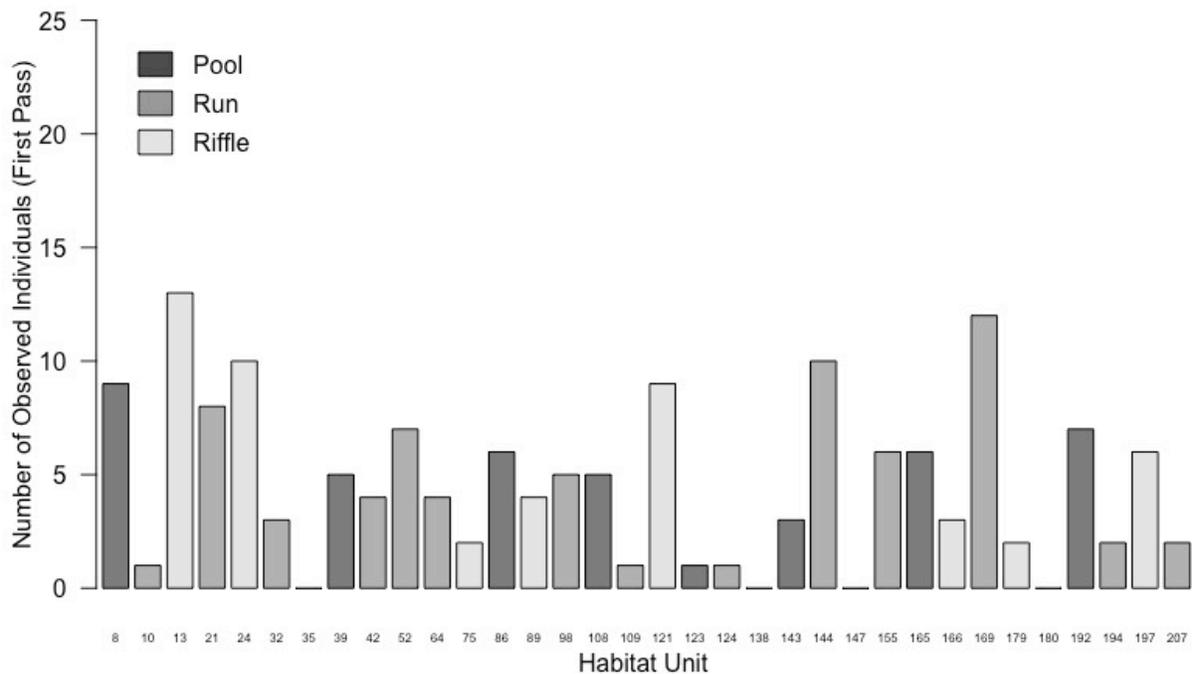


Figure 2. Number of *Oncorhynchus mykiss* (all size classes combined) observed during the first pass of snorkel surveys conducted on August 31 – September 2, 2020 on Big Chico Creek.

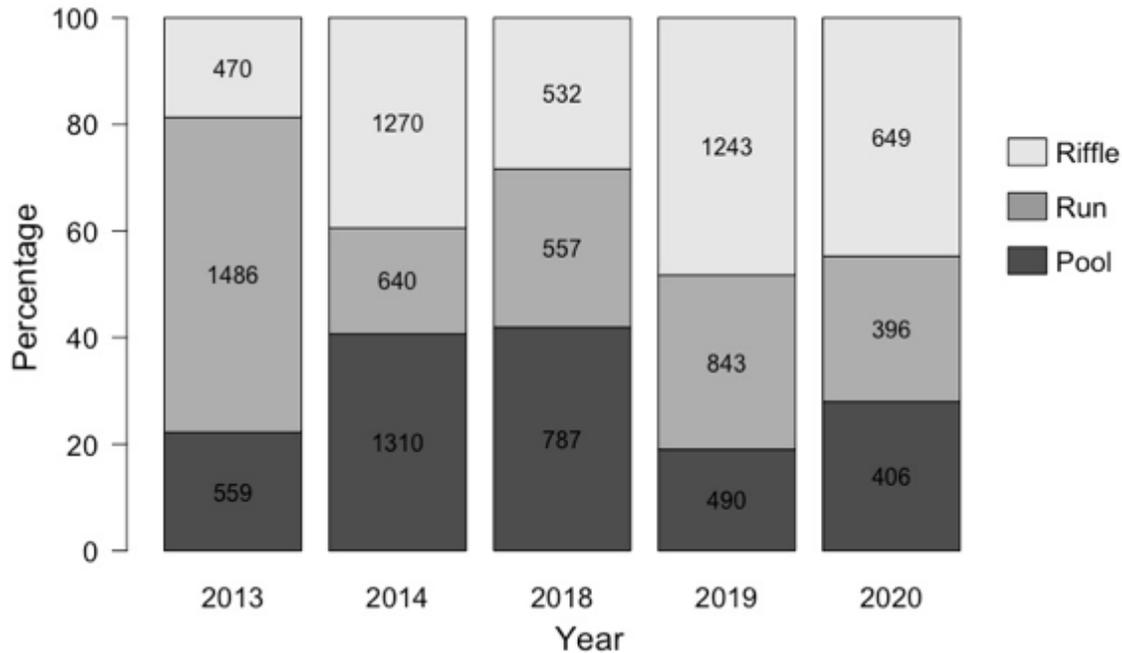


Figure 3. Occupancy of *O. mykiss*, by habitat type, estimated from snorkel surveys conducted on Big Chico Creek during in 2013, 2014, 2018, 2019 and 2020. Note that in 2014, the surveyed reach extended upstream to Higgins Hole, therefore total abundance estimates are not directly comparable.

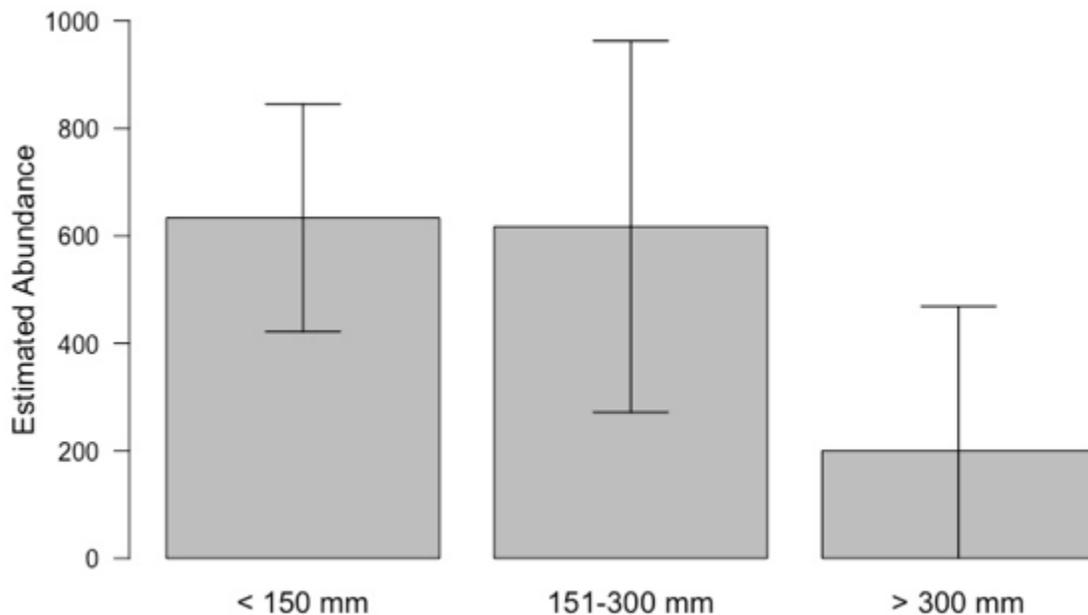


Figure 4. Estimated number of *Oncorhynchus mykiss*, by size category, on Big Chico Creek in August/September 2020. Error bars represent one standard error.

Stream Characteristics

Streamflow data for Big Chico Creek (BIC), located just upstream of Five-Mile Recreation Area in Upper Bidwell Park, was obtained from CDEC (<https://cdec.water.ca.gov>), indicating a discharge of ranging from 33 to 36 cfs between August 31 and September 2. Temperature data were recorded opportunistically throughout the survey, and instantaneous temperatures ranged from 18.2 – 19.0°C, depending on location and time of day, well within the physiological tolerances of *O. mykiss*.

Cobble (diameter 6.4–25 cm) and boulder (diameter >25 cm) were the two most common substrates within wetted areas of the mapped stream reach, and the dominant substrate category in 34% and 35% of habitat units, respectively. Bedrock, gravel, and sand characterized some of the units, though these rarely constituted the predominant substrate type. Most units (~44%) had no predominant cover type, and boulders, live vegetation, and bubble curtains (white water) were considered the predominant cover in 23%, 8%, and 8% of the units, respectively. Large woody debris (>10 cm diameter and >1 m length) was present in 26% of the surveyed units, although in low densities.



Figure 5. Big Chico Creek on August 31, 2020, illustrating the level of discharge during the survey period.

Discussion

Despite a growing body of scientific literature documenting the ability of resident rainbow trout to produce migratory offspring (e.g., Zimmerman et al. 2009, Christie et al. 2011, Courter et al. 2013), the amount of information regarding the abundance of rainbow trout in anadromous waters of Northern California continues to be very limited. To our knowledge, the surveys summarized herein are the only recent studies that attempt to quantify summertime abundance of rainbow trout in eastside tributaries of the Sacramento River.

With the exception of 2014, when the additional 2.5 miles between the BCCER and Higgin’s Hole, the barrier to anadromy, were included, survey occurred within the boundaries of the BCCER (FISHBIO 2014). The reach upstream of the reserve appeared to be more productive than the reach within the BCCER, indicated by the large proportion (approximately 57%) of all rainbow trout observed in this section (based on first pass dive counts only) despite its comparatively short length. As a consequence the larger overall sampling area between the 2014 survey and other years, direct comparisons of abundance are not possible. However, a crude approximation in abundance can be achieved by scaling the overall abundance estimate for 2013 (BCCER) by the relative fraction of first pass counts for the two years (347 individuals in 2013, compared to only 88 in 2014). This results in an abundance approximation of 638 individuals for the BCCER, a decrease of about 75% compared between 2013 and 2014. Similar drastic reductions in over-summer population size of *O. mykiss* were observed in other Central Valley streams during the recent multi-year drought (FISHBIO, unpublished).

The wet winters in 2016/2017 and 2018/2019 and resulting high flows appeared to benefit successful reproduction by *O. mykiss*, leading to a rebound in abundance approaching pre-drought levels in Big Chico Creek, and elsewhere. Lower flows over the 2019/2020 likely did not provide favorable conditions for reproduction, as suggested by a reduced abundance of juvenile (<150 mm) abundance estimated in 2020, despite a large number of presumed reproductively mature individuals in 2019 (Table 4) that could have reproduced in the winter and spring 2019/2020. The decrease in overall abundance, however, is also driven by a 75% decrease in absolute abundance of larger individuals (>300 mm) compared to the previous year. Individuals in the medium size category (150-300mm), however, increased compared to the previous year, likely driven by the strong juvenile year class estimated in 2019.

Table 3. Summary of estimated *O. mykiss* abundance since 2013, by habitat type. Note that 2014 included all habitat to Higgins Hole. See text for description.

	2013	2014	2018	2019	2020
Pool	559	1,310	787	490	406
Run	1,486	640	557	843	396
Riffle	470	1,270	532	1243	649
Total	2,515	3,220	1,876	2,576	1,451

Table 4. Summary of estimated *O. mykiss* abundance since 2013, by size class (percentage in parentheses). Note that 2014 included all habitat to Higgins Hole. See text for description.

	2013	2014	2018	2019	2020
<150 mm	1,638 (65%)	1,067 (35%)	823 (44%)	1,346 (52%)	634 (44%)
150 – 300 mm	711 (28 %)	1,511 (50%)	907 (48%)	429 (17%)	617 (42%)
>300 mm	176 (7%)	458 (15%)	146 (8%)	801 (31%)	200 (14%)
Total	2,515	3,036*	1,876	2,576	1,451

*The total used to evaluate abundance by size class *does not* include the estimate for *O. mykiss* larger than 300 mm in pools, and smaller than 150 mm in runs, in 2014.

Compared to historical accounts of the fish community of Big Chico Creek, several species continue to be conspicuously absent from the surveyed reach. No Pacific lamprey, Sacramento pikeminnow or hardhead, were observed in 2020. Sacramento sucker, not observed in 2019, continue to be rare in 2020. Historically, these species comprised a large percentage of the fish community in the foothill zone of Big Chico Creek. Our findings and historical observations made by Dr. Maslin, are suggestive of long-term detrimental effects of the rotenone treatment on native, non-salmonid fish species.

In recent years, spring-run Chinook salmon escapement to Big Chico Creek has been intermittent, ranging from zero to 350 since 2001, and is probably most affected by the timing of high flows through Iron Canyon. In 2014, two adult Chinook salmon were reported to be present on the BCCER, and ten spring-run Chinook were observed holding in Salmon Hole, a large pool just downstream of Iron Canyon, in late spring. Neither the salmon observed on the BCCER nor those seen in Salmon Hole persisted through the summer, therefore no escapement has been recorded for Big Chico Creek since 2011. In 2019, however, favorable flow conditions encouraged several hundred spring-run Chinook to ascend Big Chico Creek, and could be observed migrating through the valley reach of Big Chico Creek and the City of Chico. The official count listed in CDFW’s Grand Tab is 350. The majority of these fish appeared to be holding in Salmon Hole, just downstream of Iron Canyon, and only 4 individuals were observed in Henning Hole, the most suitable holding habitat within the BCCER. In 2020, no spring-run Chinook salmon were observed in Henning Hole.

It should be noted that a lack of observations during snorkel surveys does not necessarily constitute absence of a particular species. As only about 17% of the total creek length within the study area was surveyed in 2020, it is possible that additional species are present in Big Chico Creek (in habitat units not included in our sample). Furthermore, as snorkel surveys are designed to detect a particular suite of species, observations of non-target species can often be affected by species-specific behaviors (e.g., burrowing by lamprey ammocoetes, hiding in interstitial spaces by sculpin). Other factors that affect fish detectability (visibility, temperature, time of day, and fish size) are likely negligible considering the relatively small size and low turbidity of Big Chico Creek during the summer months.

Despite the above-described limitations of visual surveys, it appears that the relative species composition of Big Chico Creek was again slightly different during the 2020 survey compared to earlier years. Of note, very few exotic brown trout were observed on the reserve in 2020 (n=3; compared to 2 in 2019, 5 in 2018 and 16 in 2013), all of them juveniles. Thirty-five Sacramento sucker were observed on the reserve (compare to n=0 in 2019, n=7 in 2018, n= 36 in 2013 and n=38 in 2014). These differences, however, could be attributable to differences in distribution or sample unit selection, rather than reflect actual differences in abundance. Similar to previous surveys, California roach were ubiquitous throughout the reserve.

The abundance estimates reported in this study demonstrate that substantial populations of *O. mykiss* exist even in relatively small tributaries to the Sacramento River. This highlights the importance of such streams as reservoirs of genetic diversity and potential source populations of the steelhead life-history type in California's Central Valley. Furthermore, the continued presence of resident populations should warrant the consideration of these population segments and such small streams in future status assessments of *O. mykiss*. In the meantime, population estimates presented herein can serve to help evaluate trends and inter-annual comparisons of fish community composition and abundance. An expansion of the study area to other comparable, nearby tributaries of the Sacramento River (e.g., Deer Creek, Butte Creek, and others) would serve to create a more complete assessment of the trout population in the northern Central Valley. Such an expansion would provide for comparative abundance estimates and better quantify the status and recovery potential inherent to populations of *O. mykiss*.

While we do not know how many, if any, of the rainbow trout inhabiting Big Chico Creek within the study area may emigrate and assume the migratory life history that typifies steelhead, distribution and behavior of *O. mykiss* in Big Chico Creek suggest that some individuals emigrate from the creek during the winter and spring months. Each year, *O. mykiss* – often displaying the characteristic silvery coloration of smolts – can be observed in the lower, valley-floor reaches of Big Chico Creek, sometimes in great numbers. These fish are likely a mixture of resident trout that have moved to lower stream sections during fall and winter, and juvenile steelhead (the migratory form of the same species) during their migration to the marine environment. Regardless, rising water temperatures in spring limit the duration of their residence in the lower reaches of Central Valley streams, eventually forcing a behavioral response (migration) or, alternatively, resulting in mortality. It has been suggested that the likelihood of juvenile *O. mykiss* to assume a migratory/anadromous life history is greater when water temperatures are elevated (Sloat and Reeves 2014), yet the more immediate behavioral response to rising temperatures has not been studied in detail. As such, the reach of Big Chico creek downstream of the flood-control overflow (at 5-mile Recreation Area) is ideally suited to investigate *O. mykiss* behavior, for a number of reasons, including sufficient abundance (in spring), accessibility, and applicability of insights to a range of similar watersheds in the Central Valley.

In summary, the abundance estimates presented in this study provide a much-needed and current quantification of “potential steelhead” in the Northern Central Valley. As this survey encompassed a large part of suitable over-summering habitat for *O. mykiss* in the anadromous portion of Big

Chico Creek, we consider the abundance estimates reported herein to be an important reference for comparison to other local tributaries of the Sacramento River. The relatively high density, albeit with fluctuations from year to year, of *O. mykiss* estimated in this study is a testament to the importance of Big Chico Creek, and specifically the Big Chico Creek Ecological Reserve, to the conservation and recovery of Central Valley steelhead. The number of trout observed in this survey confirms the importance of this habitat to the threatened species, and the need for continual monitoring and conservation of resident trout populations.

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