The Upper Sacramento River Anadromous Fish Habitat Restoration Project: Monitoring of Habitat Restoration Sites in the Upper Sacramento River in 2020-2021

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Note: *Some sections of this report are derived from the Monitoring Plan (Tussing and Banet, 2017) and previous annual reports (Banet et al., 2018, 2020, 2021). Information that has remained consistent between years (such as background information or methodology) may be excerpted from these earlier reports without alteration.*

SUMMARY

The Upper Sacramento River Anadromous Fish Habitat Restoration Project restores spawning and juvenile rearing habitat in the Upper Sacramento River. The project approach assumes that restoring or creating side channels that are connected at a range of flows will recreate the historical biological and geologic characteristics that support anadromous salmonid populations, leading to increased survival and condition. This report presents data from monitoring efforts from December 2015 through April 2021 and addresses project objectives 2-5: increasing the areal extent of rearing habitat meeting juvenile salmonid rearing habitat suitability criteria; increasing salmonid juvenile abundance/density at restoration sites after implementation, as compared to before implementation; improving size and average condition of salmonids using the side channels, as compared to those that have not been documented using the side channels; and increasing available macroinvertebrate prey abundance.

At the time of reporting, seven sites had been restored between 2014 and February 2021 (Painter's Riffle, North Cypress, Kapusta, Anderson River Park, Reading Island, Lake California, and Rio Vista). Control sites near the restorations were chosen from historical side channels, which are thought to be the highest quality habitat nearest the restoration sites. When side channel controls were not available, mainstem controls were chosen from nearby areas that exhibited characteristics that could support juvenile salmon. The monitoring team aimed to collect data from project and control sites before and after restoration. However, due to logistical constraints (e.g. timing of restoration relative to availability of monitoring funding and resources), before data is limited to a subset of data types and sites.

To analyze fish abundance, we first used a BACI (before-after-control-impact) approach to analyze total observed fish number from the restoration sites and their nearby controls that had adequate before data (restoration sites: Anderson River Park, Lake California, and Rio Vista; control sites: Bourbon Island, Mainstem North, and Mainstem South). A zero-inflated linear mixed model showed that restoration sites had a significantly larger increase in observed fish number after restoration, as compared to the controls, indicating a positive effect of the restoration. When broken down by run, this pattern was significant for fall run Chinook Salmon (*Oncorhynchus tshawytscha*) and steelhead/Rainbow Trout (*Oncorhynchus mykiss*). Winter run and late-fall run Chinook Salmon showed similar, nonsignificant trends. Spring run Chinook Salmon could not be analyzed due to poor model fit. We used a similar model to analyze fish density (fish-per-acre) in these same sites. Steelhead/Rainbow Trout showed a significant increase in density in response to restoration. Fall run Chinook, late-fall run Chinook, and all salmonids pooled together showed similar, nonsignificant trends. Winter and spring run Chinook Salmon could not be analyzed due to poor model fit. We then analyzed the full dataset (including those sites without before data). The lack of data taken before restoration makes it more challenging to make decisive conclusions. Fish counts, in particular, are difficult to analyze and interpret without adequate before data for comparison, so our dependent variable in these

analyses is estimated fish density (fish-per-acre). The trends for estimated density show that control sites and restored sites are similar, and consistently have more estimated fish than baseline (unrestored) sites. However, these results were only statistically significant in steelhead/Rainbow Trout. These results suggest that the benefit of the restorations comes largely from the addition of new habitat and that restored sites support similar densities of habitat as control sites.

Linear mixed models applied to habitat mapping data (depth, velocity, and cover) show that restored and control sites had similar proportions of suitable and optimal habitat (as defined by Goodman *et al.*, 2015). Baseline sites were not included in this analysis because the habitat criteria we used during mapping gave misleading results. The Goodman *et al.* criteria were created on the Trinity River, and did not include backwater habitats or disconnected side channels. Disconnected pre-restoration sites can sometimes show large proportions of suitable habitat due to artifacts of classification (e.g. an entire backwater pool could be classified as suitable using depth, velocity, and cover criteria, but still not be appropriate habitat based on oxygen levels, potential of stranding fish, etc.). When looking at absolute area of each habitat (rather than proportions), restoration did indeed increase the areal extent of suitable and optimal habitat. The estimated acreage of habitat gained varies depending on the underlying assumptions of the calculations; these are detailed in the discussion.

The habitat mapping described above used depth, velocity, and cover criteria that were created using juvenile salmonid (>50 mm) data from the Trinity River. To examine how well these criteria fit data from the Sacramento River, we created study habitat suitability curves (HSCs) using salmonid observations from this study. When looking at pooled data for all salmonid juveniles (>50 mm) and fry (≤ 50 mm), distance to cover appears to be very similarly represented, but Sacramento River populations exhibit a notably narrower range of depth and velocities than the mapping criteria. This indicates that the estimates of habitat described above may be slightly overestimating habitat with suitable depth and velocity. When looking at data for salmonid fry, suitable ranges of all criteria for Sacramento River populations are substantially smaller than the mapping criteria that we used. This indicates that the habitat mapping described above may be overestimating fry habitat. Juveniles in the Sacramento River use a similar range of velocities and distance to cover, but a narrower range of depth, as compared to fish from the Trinity River. This indicates that our mapping is closely approximating the amount of suitable velocity and distance to cover, but may be overestimating the amount of suitable depth. We provide updated suitability criteria to help inform future restoration efforts.

Microhabitat surveys also provided information on preference for different cover types (boulders, fine woody debris, branches/small woody debris, logs/large woody debris, overhead cover, undercut banks, and rip rap). Separate analyses were run for fry and juveniles of each salmonid species. All groups showed similar preference trends, but significance varied slightly between groups. Fine woody debris was the preferred cover; fry and juveniles from both Chinook Salmon and steelhead/Rainbow Trout significantly preferred it over all other cover types besides undercut banks. Undercut banks were the second most favored cover type; preference scores were strong enough that it was not statistically distinguishable from fine woody debris, but the degree of significance between undercut banks and other cover types varied between species and size groups. No strong preference was shown for other cover types.

The lowest preference scores were found for branches/small woody debris, though these were only statistically distinguishable from other cover types for certain comparisons. Details of these analyses are provided in the main body of the report.

Fish size and condition collected via seining did not yield consistent results between runs. Fish from restored side channels had significantly larger fork sizes for some runs (e.g. winter run Chinook Salmon and steelhead/Rainbow Trout had significantly longer fork lengths in restored side channels as compared to the mainstem), but other runs showed the opposite relationship. While this may indicate run-specific benefits of restoration on growth, there are factors (detailed in the discussion) that make conclusive interpretation difficult.

Macroinvertebrate monitoring provides information on taxonomic diversity and sampling rate between sites, but unfortunately due to sample deterioration, we were unable to compare macroinvertebrate biomass at each site type. Descriptive statistics were calculated for other metrics. Baseline sites had the lowest values for three EPT (Ephemeroptera, Plecoptera, and Trichoptera) metrics, mainstem sites had the highest values, and control and restoration side channels performed similarly. Baseline sites also showed the lowest overall macroinvertebrate diversity, taxonomic richness, and had a lower rate of individuals captured over time.

The datasets used in the analyses reported above vary in quality and size. Results obtained from the highest quality datasets all suggest that the Upper Sacramento River Anadromous Fish Habitat Restoration Project has effectively produced additional high quality juvenile salmonid habitat (objective 2) that supports higher numbers of fish (objective 3) in the upper Sacramento River. The effects of restoration on fish size and condition (objective 4) varied between runs when looking at seining data. The seining data was likely confounded by several other factors (detailed in the text of the report), and data collection of enclosure study growth rates were unfortunately not completed due to COVID-19 shut downs. The higher number of macroinvertebrates (determined by sampling rate) observed in restored side channels as compared to baseline channels suggests that there may be a positive effect of restoration on food availability (objective 5), but without biomass and diet information, firm conclusions can’t be drawn. Addressing the logistical challenges of collecting data for objectives 4 and 5 can help paint a clearer picture of how side channel restoration affects salmonid growth.

Continued monitoring of completed and future restorations will provide additional insight into the effectiveness of side channel restoration, as well as information about how side channel characteristics evolve over time. We recommend that future habitat mapping efforts measure dissolved oxygen levels, temperature, and access to habitat in addition to depth, velocity, and cover in order to get a more accurate picture of the habitat added by restoration. Future restorations may also benefit from including the preferred cover types of fine woody debris and undercut banks, as well as considering the habitat suitability criteria created from data collected in our study sites. Recommendations for increasing the cost-effectiveness of future restoration monitoring are presented at the end of the report.

INTRODUCTION

Project Overview

Central Valley anadromous salmonid populations have seen dramatic declines in the past century, largely due to anthropogenic habitat alterations (Katz *et al.*, 2013). In the upper Sacramento River, the largest impacts have been attributed to loss of floodplains, riparian habitat, and instream cover; increased competition and predation; and alterations to morphologic function (NMFS, 2014). Historic off-channel habitat has largely been lost due to flood control and associated geologic processes; the Central Valley Project Improvement Act Science Integration model (CVPIA SIT) estimates in-stream habitat to be 26 acres at median flows (8311 cfs), far below the number needed to aid in recovery of threatened and endangered populations of Central Valley salmonids (Gill, n.d.).

The Upper Sacramento River Anadromous Fish Habitat Restoration Project (hereafter, the Project) restores spawning and juvenile rearing habitat in the Upper Sacramento River. The project approach assumes that restoring or creating side channels that are connected at a range of flows will recreate the historical biological and geologic characteristics that support salmon populations, leading to increased survival and condition. The conceptual model underlying this hypothesis, which forms the basis for the monitoring plan approach, is provided below (Figure 1). An in-depth discussion of this conceptual model is available in the Upper Sacramento River Anadromous Fish Habitat Restoration Project Monitoring Plan and Protocols (Tussing and Banet, 2017), hereafter referred to as the Monitoring Plan.

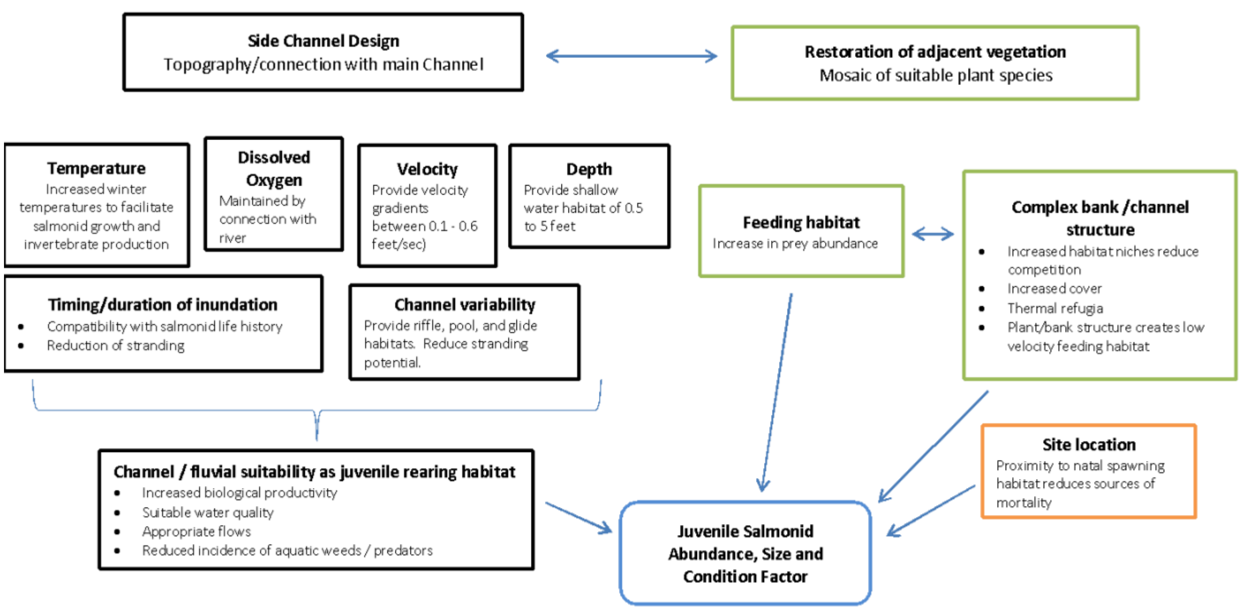

Figure 1. Conceptual model of design-related elements and their influence on biotic and abiotic juvenile salmonid habitat elements, from Banet and Tussing (2017).

Restoration Goals and Objectives

The primary goals of the Project, as stated in the Monitoring Plan (Tussing and Banet, 2017), are to:

1. Increase the availability, quality and quantity of spawning and rearing habitat for Sacramento River Basin Chinook Salmon and steelhead/Rainbow Trout
2. Restore, maintain or enhance natural system processes whenever possible
3. Determine project effectiveness, including cost, project longevity and maintenance requirements, with an efficient and scientifically-robust monitoring program
4. Demonstrate a positive, detectable salmonid population response to habitat enhancement activities
5. Contribute to the long-term health of the river ecosystem (water quality, invertebrate and fish assemblages, riparian and floodplain habitat function, etc.)
6. Incorporate information learned to improve future projects (adaptive management)
7. Contribute to scientific understanding of aquatic ecology
8. Work collaboratively with partners to identify and implement projects that are cost effective and benefit aquatic resources, emphasizing anadromous salmonids, in the short and long term

The primary objectives of the Project, as stated in the Monitoring Plan (Tussing and Banet, 2017) are to provide:

1. An increase in the areal extent of spawning habitat meeting suitability criteria and the use of spawning habitat
2. An increase in the areal extent of rearing habitat meeting juvenile salmonid rearing habitat suitability criteria
3. An increase in salmonid juvenile abundance/density at restoration sites after implementation, as compared to before implementation
4. Improved size and average condition of salmonids using the side channels, as compared to those that have not been documented using the side channels
5. An increase in available prey abundance, including both drift and benthic macroinvertebrates
6. Increased extent and quality of riparian habitat at Sand Slough

Purpose of Annual Reporting

The purpose of annual reporting, as described in the Monitoring Plan (Tussing and Banet, 2017), is to determine if monitoring data collection methods are effective at achieving data objectives; to modify field protocols as needed to effectively meet those objectives; to perform preliminary tests of hypotheses as data allows; and, to inform restoration efforts where a biological response to restoration can be established. More extensive analyses and reporting are to be performed when there is sufficient data to analyze the full suite of hypotheses as described in the primary study design. This annual report addresses objectives 2, 3, 4 and 5 using data collected between December 2015 and April 2021. Objective 1 is not addressed because resource limitations prevented the collection of sufficient data.

Monitoring Site Selection

Project sites (Figure 2, Table 1) were identified and prioritized for construction through the CVPIA habitat restoration process. Restoration sites are side channels that were either previously connected to the river and have since been cut off to fish due to increased channelization, or side channels that are only available to juvenile fish during certain times of year (i.e. during high flow releases from Keswick dam). The Project prioritized sites for construction based on a multitude of factors which may include but are not limited to: stranding potential at lower Keswick releases, feasibility of construction, land-owner cooperation, site longevity and maintenance requirements, and overall perceived benefit to juvenile salmonids, with emphasis on benefits to listed species. Baseline snorkel data was taken from restoration sites when possible, but this data is limited, either due to logistical constraints, or because many restored sites were not consistently connected to the mainstem prior to restoration. More detailed descriptions of project sites are available in Appendix A.

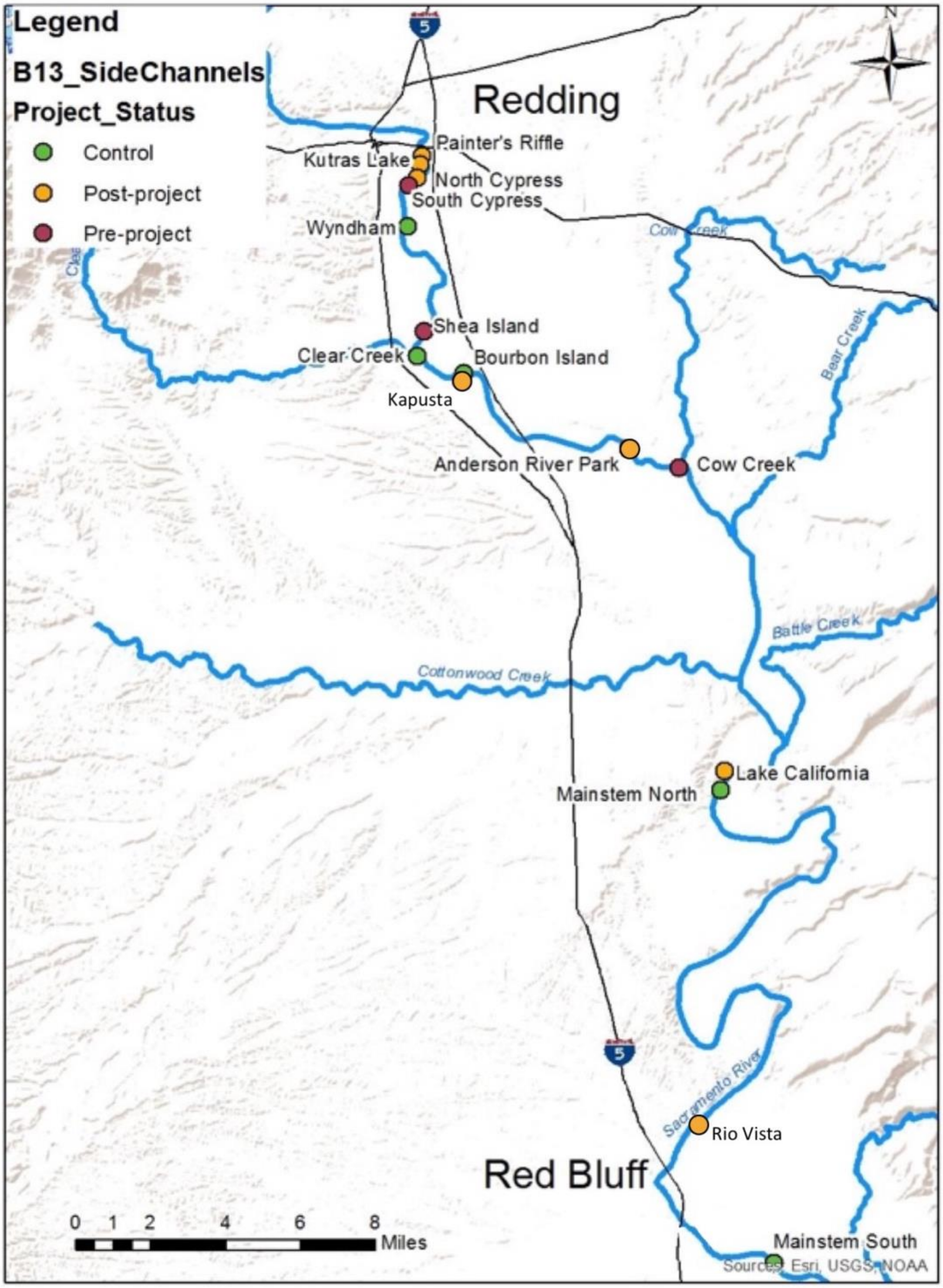

Figure 2. Map of control, pre-project (pre-restoration) and post-project (restored) side channels surveyed as part of the Project.

Table 1. Name, status (as of April 2021), and approximate river mile (RM) of Project Sites. *Note that Kutras Lake is not a side channel, and is thus not addressed in this report. Pre-project status refers to project sites that are slated for restoration, but were not restored at the end of this reporting period. Post-project status refers to sites that have been restored. Control status refers to existing habitat that is not scheduled for restoration.*

Site Name	Status	Restoration Date(s)	RM
Painter's Riffle	Post-project	2014	296
Kutras Lake	Post-project	May 2017	296
North Cypress	Post-project	December 2016	295.5
South Cypress *(Nur Pon Open Space)*	Pre-project	N/A	294
Wyndham	Control	N/A	293.5
Shea Island	Pre-project	N/A	290
Clear Creek	Control	N/A	289
Bourbon Island	Control	N/A	287.5
Kapusta	Post-project	May 2018 (Kapusta 1A only)	287.5
Anderson River Park	Post-project	December 2019 (Phase I) February 2021 (Phase II/III)	282
Cow Creek	Pre-project	N/A	280
Reading Island	Post-project	August 2019 (Phase I) December 2019 (Phase II)	274
Lake California	Post-project	January 2018	269.5
Mainstem North	Control	N/A	268.5
Rio Vista	Post-project	October 2019	247
Mainstem South	Control	N/A	242

In order to examine the performance of the restored side channels, the monitoring team identified five control sites. To select control sites, we consulted with experts from the project team to identify habitat geographically located near restoration (or future restoration) sites that was thought to be the highest quality nearby habitat (based on estimated depth, velocity, cover, and prior fish observations). When possible, currently functioning side channels with flow year-round were selected as controls. In areas of the river where functioning side channels were not available to use as controls, mainstem control sites were selected. This process resulted in three side channel controls, and two mainstem controls (Figure 2, Table 1).

FISH ABUNDANCE INDEX

Field Methods to Estimate Fish Abundance

An index of fish abundance was collected via snorkel surveys when conditions permitted. Surveys were conducted at each site between 9AM and 3PM, generally every two weeks. Data was classified as control, baseline (pre-restoration), or impact (restored). The order in which control, impact, and baseline sites were surveyed were randomized whenever possible, in order to reduce the likelihood that fish abundance was confounded with time of day. We recorded several physical variables each time a site was surveyed (Table 2). Visibility, weather, and water temperature were recorded on site. Flow was calculated in the office using data from nearby gauging stations.

Table 2. Physical variables collected in conjunction with snorkel counts.

Variable	Description
Visibility	Visibility is measured using a secchi disk. A member of the crew submerges his or her face into the water and extends the pole upstream along the plane of their eye level until the disc can no longer be seen. The distance from the disc to the swimmer's eye is recorded in feet.
Weather	Weather is measured on a numeric scale as follows: 1- Clear, 2 - Partly Cloudy, 3 - Cloudy, 4 - Rain, 5 - Snow, 6 - Fog. For this report, monthly weather scores are reported both as mean and mode numeric values.
Water Temperature	Water temperature is measured in Fahrenheit during each survey.
Calculated Flow	Flow is determined using data from nearby gauging stations. Lake California, Mainstem North, Mainstem South, and Rio Vista use data from the Bend Bridge (BND) gauging station in Red Bluff, CA. All other sites use data from the Keswick (KWK) gauging station in Keswick, CA.

Each swimmer calibrated his or her vision prior to commencing a snorkel survey in order to account for the visual distortion that occurs in water. To do this, the swimmer submerged their face and mask in the water, and another crew member held a calibration tool equipped with a model fish of known length in front of the swimmer for a short period of time. This process was repeated until the swimmer was comfortable with the calibration.

Flows and conditions at some sites were not amenable to snorkeling upstream. Because of this, all surveys were conducted downstream to maintain consistency. Swimmers formed a line perpendicular to flow prior to the start of the survey and recorded the start time of the survey. At most sites, two snorkelers surveyed edge habitat along each bank of a side channel. For mainstem sites, one snorkeler surveyed the edge of the main river bank. Swimmers maintained their line in order to reduce the likelihood of double counting fish. Juvenile salmonids were identified to species, classified by size, and counted as they passed by each snorkeler. In order to gather information on species richness and the presence of predators, other fish species were noted and counted as well. After the survey was completed, an end time was recorded.

For analysis, steelhead and Rainbow Trout juveniles were classified together, and Chinook Salmon were categorized into runs using the Central Valley length-to-date chart (See Appendix

B). Some analyses broke fish down into size classes of juveniles (>50 mm) and fry (≤ 50 mm). Graphs of Raw Data can be found in Appendix C.

Fish Abundance in Habitats with Before-After-Control-Impact (BACI) Data

Data analysis

In an ideal study, we would have paired data of all restoration sites with nearby controls, before and after restoration. This approach allows analysis of the data using a before-after-control-impact (BACI) design, which can be used to disentangle effects of restoration from that of natural temporal variation (Smith, 2014). This design provides more statistical power to detect differences between treatment types than studies that compare only post-restoration data from control and impact sites. We were able to collect and analyze sufficient before-after-control-impact data on a subset of our sites (restoration, or impact sites: Anderson River Park, Lake California, Rio Vista; Nearby control sites: Bourbon Island, Mainstem North, Mainstem South).

All surveys from each restored site had a complementary survey taken at its nearby control within a short time frame (typically within two days from one another, and never more than a week). Comparison of paired control and restored sites within the same time frame allows us to account for variation in escapement across years. Anderson River Park and Bourbon Island each had 11 before surveys (collected between 5/6/18 and 9/18/19) and 20 after surveys (1/9/2020 – 4/19/2021). Lake California and Mainstem North each had 11 before surveys (7/24/17 – 12/29/17) and 46 after surveys (1/11/18 – 4/19/21). Rio Vista and Mainstem South each had 21 before surveys (7/24/17 – 9/19/19) and 14 after surveys (11/18/19 – 4/19/21). These data were analyzed using the glmmTBD package in R (R Core Team, 2016), which can fit mixed effect generalized linear models to datasets with large numbers of zeros. We analyzed two metrics: fish counts and fish density (fish-per-acre). These two metrics give insight into different aspects of the restoration. Higher fish counts are suggestive of a positive effect of restoration. Density can be more challenging to interpret. Higher densities suggest a positive effect of restoration, but we could see lower densities even if the overall effect of restoration is positive (e.g., if restoration greatly increases suitable/optimal habitat availability, we may see lower densities of fish even if overall the system has more juvenile salmonids).

To analyze fish counts, we used a zero-inflated linear mixed model (Brooks *et al.*, 2017) to examine the effects of site classification (restoration/control), restoration timeline (before/after), visibility, quarter (January-March, April-June, July-September, and October-December), and the interaction between site classification and restoration timeline on fish count. Because geographic location may impact fish count, we paired restoration sites with their nearest control (Anderson River Park with Bourbon Island, Lake California with Mainstem North, and Rio Vista with Mainstem South) and used these pairs as a random effect in the model. We used a type I negative binomial distribution in the model, because it had the lowest Akaike Information Criterion (AIC) score compared to the other distributions considered: Normal and type II negative binomial. The negative binomial distributions were considered because they are designed to handle count data, unlike the normal distribution. The interaction term in this model (restoration timeline x site classification) is the key output for understanding the effect of restoration. A greater increase in fish count in the restored side channels after restoration, relative to the control sites, would indicate that the restoration was successful in increasing the number of fish. Separate models were run for all salmonids (pooled into a single dataset), each Chinook Salmon run, and steelhead/Rainbow Trout.

Fish density (fish-per-acre) was estimated by the following equation:

$$Fish\text{-}per\text{-}acre = N \div \frac{L*V*S}{43{,}560}$$

where *N* was total fish count during the survey, *L* was the length of the survey in feet, *V* was the visibility surveyed in feet (a proxy for survey width), and *S* was the number of snorkelers. For example, if two snorkelers observing the stream margins floated a survey of 100 feet with 10 feet visibility, then the area observed would be estimated as 2000 sq. ft., or approximately 0.046 acres. This approach assumes that both snorkelers were oriented toward the margins of the stream at a distance equal to the visibility on a given day. If ten fish were observed during the survey, density would be calculated as 10/0.046, or approximately 217.4 fish-per-acre.

To analyze fish density, we used a similar statistical approach as fish counts, but employed two models to examine the robustness of our results. In the first model, density (fish-per-acre) was rounded to the nearest whole number, and a zero-inflated linear mixed model was used to examine the effects of site classification (restoration/control), restoration timeline (before/after), quarter (January-March, April-June, July-September, and October-December), and the interaction between site classification and restoration timeline on fish count. Visibility was not included in this model because it was included in the density calculation. Because geographic location may impact fish density, we paired restoration sites with their nearest control (Anderson River Park with Bourbon Island, Lake California with Mainstem North, and Rio Vista with Mainstem South) and used these pairs as a random effect in the model. A type I negative binomial distribution was used in the model because it had the lowest AIC score compared to the other distributions considered: Normal and type II negative binomial. The negative binomial distributions were considered because they are designed to handle count data, unlike the normal distribution. The second model was similar with two exceptions: fish-per-acre was not rounded to the nearest whole number, and a normal distribution was used (negative binomials can only be used with data represented as whole numbers). As with the fish count analysis described above, the interaction term in these models (site classification x restoration timeline) is the key output for understanding the effect of restoration. A greater increase in fish density in the restored side channels after restoration, relative to the control sites, would indicate that the restoration was successful in increasing the density of fish. Separate models were run for all salmonids (pooled into a single dataset), each Chinook Salmon run, and steelhead/Rainbow Trout.

Below we report on results for populations that showed the same trend for both statistical approaches (All salmonids, fall run Chinook, late-fall run Chinook, and steelhead/Rainbow Trout). If the two density modelling approaches did not show the same trend (winter run), the data was not considered robust and the model results are not reported below. For simplicity, graphs and tables report just the results of the model with the non-rounded numbers when both modeling approaches showed the same trends. For all runs, this type of model more conservatively estimated the positive effects of restoration.

Results

Results of the zero-inflated linear mixed models used for the BACI analyses of total observed fish are shown in Table 3 and Figure 3. The interaction term (site classification x restoration timeline) indicates whether the restoration was successful in increasing the number of fish. Restoration significantly increased fish count in restored sites relative to control sites when examining all salmonids, fall run Chinook Salmon, and steelhead/Rainbow Trout. Late-fall and winter run Chinook Salmon showed similar, non-significant trends. Our limited data on spring run Chinook Salmon did not allow a reliable model fit and is thus not included in this analysis. Note that the main effects of site classification and restoration timeline in each model cannot be interpreted directly due to the presence of an interaction in the model. Because the above models include comparison of paired control and restored sites within the same time frame, variation in escapement across years is accounted for.

Table 3. BACI (before-after-control-impact) analyses of fish counts for three restoration sites (Anderson River Park, Lake California, and Rio Vista) and nearby controls (Bourbon Island, Mainstem North, and Mainstem South). *Details of the zero-inflated linear mixed models used in these analyses are provided in the methods. A significant interaction term (Restoration Timeline x Site Classification) indicates a significant impact of restoration. Note that the main effects of site classification and restoration timeline in each model cannot be interpreted directly due to the interaction in the model.*

Run	**Quarter**	**Visibility**	**Restoration Timeline**	**Site Classification**	**Interaction: Restoration Timeline x Site Classification**
All salmonids	**$\chi2 = 23.943$ df = 3 $p < 0.001$**	**$\chi2 = 11.520$ df = 1 $p < 0.001$**	$\chi2 = 3.581$ df = 1 $p = 0.058$	**$\chi2 = 17.305$ df = 1 $p < 0.001$**	**$\chi2 = 8.907$ df = 1 $p = 0.003$**
Fall run Chinook	**$\chi2 = 51.134$ df = 3 $p < 0.001$**	$\chi2 = 1.310$ df = 1 $p = 0.252$	$\chi2 = 1.304$ df = 1 $p = 0.254$	$\chi2 = 0.875$ df = 1 $p = 0.350$	**$\chi2 = 6.361$ df = 1 $p = 0.012$**
Late-fall run Chinook	**$\chi2 = 31.051$ df = 3 $p < 0.001$**	**$\chi2 = 5.622$ df = 1 $p = 0.018$**	**$\chi2 = 9.569$ df = 1 $p = .002$**	$\chi2 = 3.501$ df = 1 $p = 0.061$	$\chi2 = 1.091$ df = 1 $p = 0.296$
Winter run Chinook	**$\chi2 = 31.900$ df = 3 $p < 0.001$**	**$\chi2 = 10.666$ df = 1 $p < 0.001$**	**$\chi2 = 16.568$ df = 1 $p < 0.001$**	**$\chi2 = 16.906$ df = 1 $p < 0.001$**	$\chi2 = 3.198$ df = 1 $p = 0.073$
Trout	**$\chi2 = 38.867$ df = 3 $p < 0.001$**	$\chi2 = 1.135$ df = 1 $p = 0.2868$	$\chi2 = 0.100$ df = 1 $p = 0.7521$	**$\chi2 = 23.228$ df = 1 $p < 0.001$**	**$\chi2 = 22.864$ df = 1 $p < 0.001$**

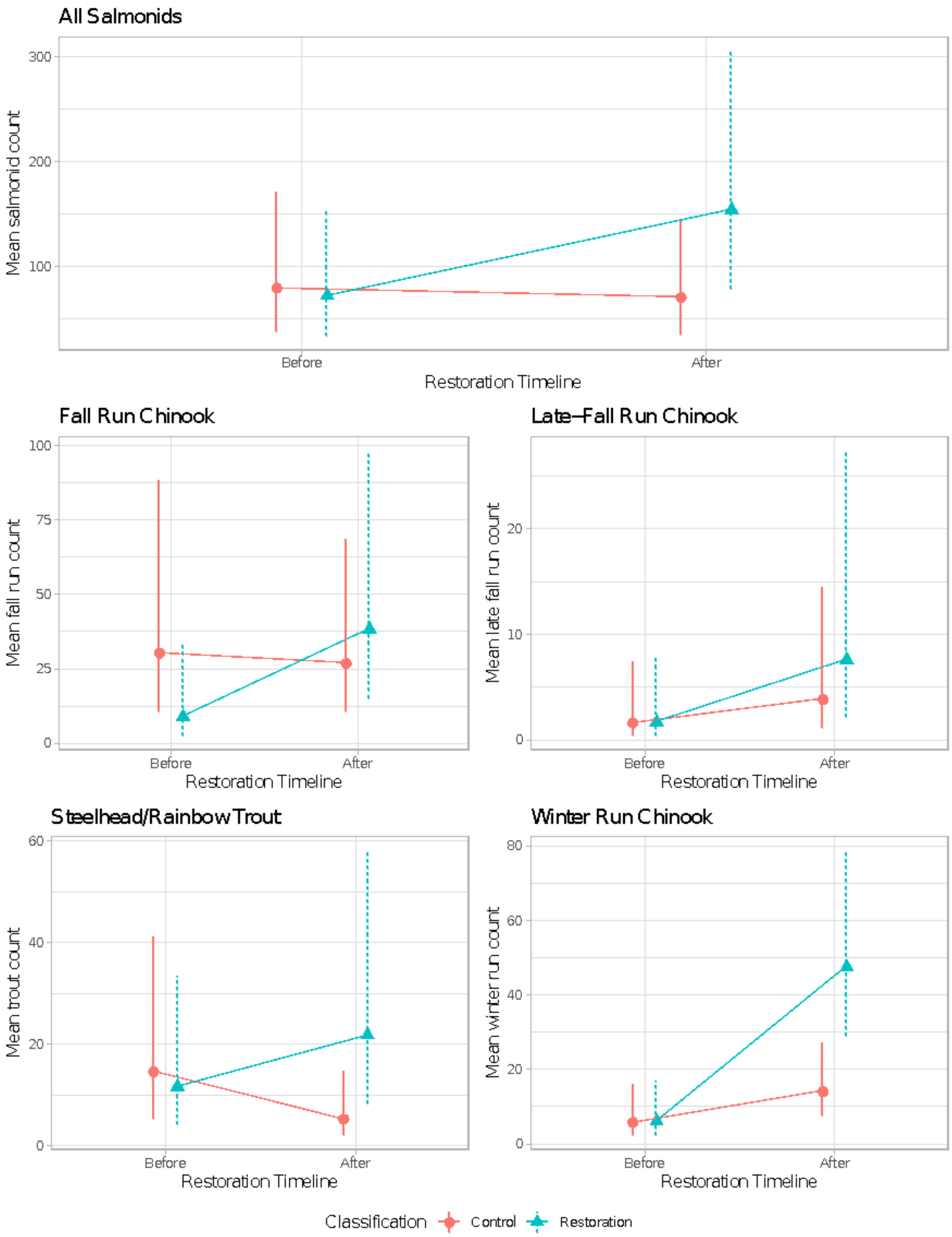

Figure 3. Estimated marginal means of fish count before and after restoration for three restoration sites (Anderson River Park, Lake California, and Rio Vista) and nearby controls (Bourbon Island, Mainstem North, and Mainstem South). *Run was classified using the Central Valley length-to-date chart. A larger slope in restoration sites, as compared to control sites, indicates a positive effect of restoration on fish count. Error bars are 95% confidence intervals. Details of the zero-inflated linear mixed models used to generate this data are provided in the methods.*

Results of the zero-inflated linear mixed models used for the BACI analysis of fish-per-acre are shown in Table 4 and Figure 4. The interaction term (site classification * restoration timeline) indicates whether the restoration affected fish density. Restoration showed no effect on density when examining all salmonids, fall run Chinook, late-fall run Chinook and steelhead/Rainbow Trout. Winter run and spring run Chinook Salmon data did not allow a reliable model fit and is thus not reported below. Note that the main effects of site classification and restoration timeline in each model cannot be interpreted directly when the interaction is significant.

Table 4. BACI (before-after-control-impact) analyses of fish density for three restoration sites (Anderson River Park, Lake California, and Rio Vista) and nearby controls (Bourbon Island, Mainstem North, and Mainstem South). *Run was classified using the Central Valley length-to-date chart. Details of the zero-inflated linear mixed models used in these analyses are provided in the methods. A significant interaction term indicates a significant impact of restoration. Note that the main effects of site classification and restoration timeline in each model cannot be interpreted directly due to the interaction in the model.*

Run	**Quarter**	**Restoration Timeline**	**Site Classification**	**Interaction: Restoration Timeline * Site Classification**
All salmonids	$\chi2 = 10.081$ df = 3 p = 0.018	**$\chi2 = 4.236$ df = 1 p = 0.040**	**$\chi2 = 13.001$ df = 1 p < 0.001**	$\chi2 = 2.868$ df = 1 p = 0.090
Fall run Chinook	$\chi2 = 5.419$ df = 3 p = 0.144	**$\chi2 = 4.450$ df = 1 p = 0.035**	**$\chi2 = 6.626$ df = 1 p = 0.010**	$\chi2 = 0.674$ df = 1 p = 0.412
Late-fall run Chinook	$\chi2 = 0.615$ df = 3 p = 0.893	$\chi2 = 0.495$ df = 1 p = 0.482	$\chi2 = 0.152$ df = 1 p = 0.696	$\chi2 = 0.276$ df = 1 p = 0.600
Trout	$\chi2 = 7.398$ df = 3 p = 0.060	$\chi2 = 0.788$ df = 1 p = 0.375	**$\chi2 = 10.897$ df = 1 p < 0.001**	$\chi2 = 0.6794$ df = 1 p = 0.410

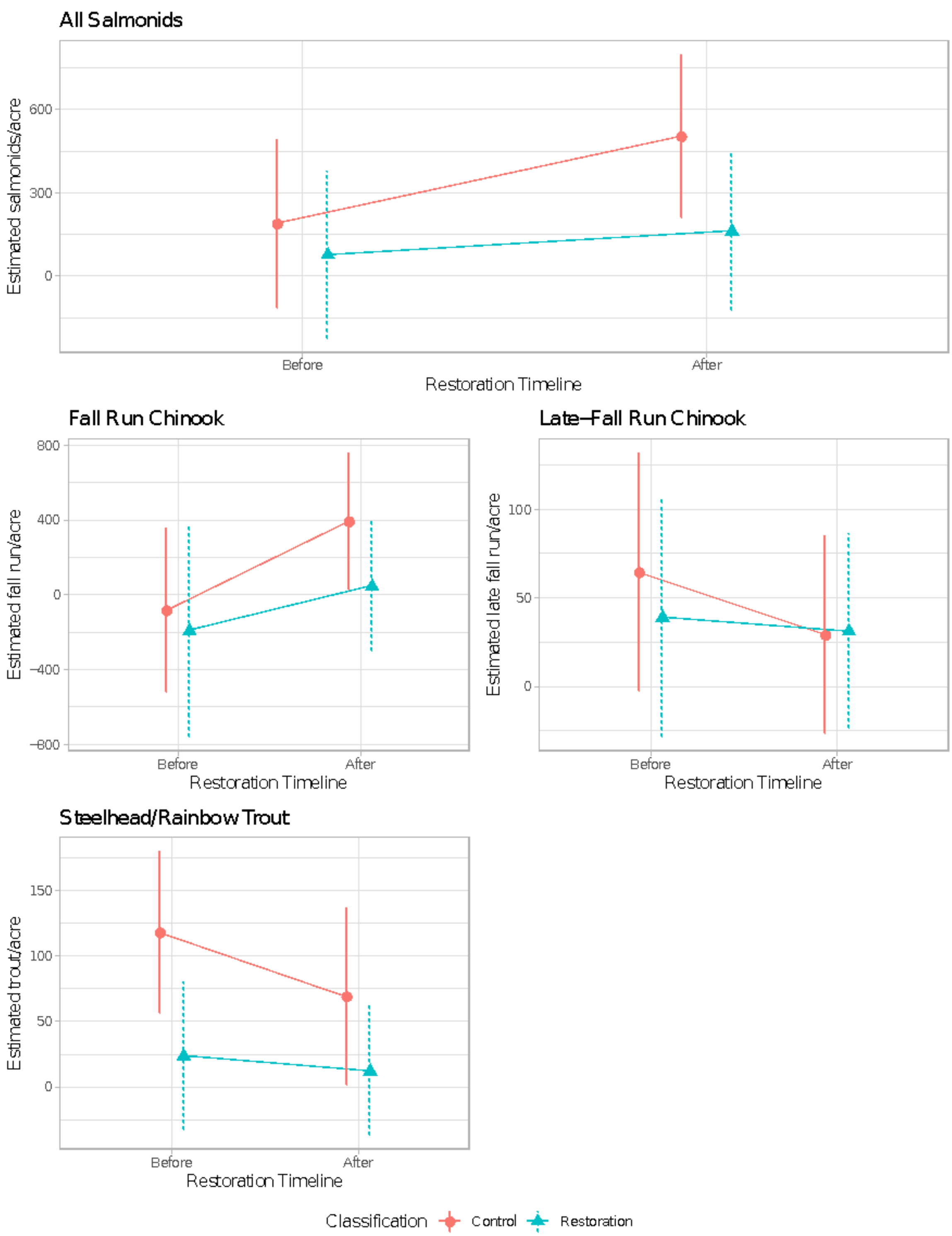

Figure 4. Estimated marginal means of fish-per-acre before and after restoration for three restoration sites (Anderson River Park, Lake California, and Rio Vista) and nearby controls (Bourbon Island, Mainstem North, and Mainstem South). *Run was classified using the Central Valley length-to-date chart. A larger slope in restoration sites, as compared to control sites, indicates a positive effect of restoration on density. Error bars are 95% confidence intervals. Details of the zero-inflated linear mixed models used to generate this data are provided in the methods.*

Fish Abundance in the Full Dataset

Data analysis

The full dataset is more challenging to analyze due to lack of paired before and after data at many of the study sites. Thus, we urge extreme caution with the interpretation of the analyses described below. Fish counts, in particular, cannot be analyzed and interpreted without adequate before data for comparison, so our dependent variable in these analyses is estimated fish density (fish-per-acre). This was calculated as described in the previous section.

To analyze fish density of the full dataset, we compared baseline (pre-restoration), impact (post-restoration), and control sites. This approach allows us to include more data in our analyses because we can include sites that did not have baseline data, as well as sites that have not yet been restored; however, a caveat to interpretation of the full dataset is that since the group of sites that have baseline data does not have fully overlapping membership with the group of sites that have impact data, it is difficult to disentangle the effects of restoration from natural variation between the sites. Additionally, data collection during earlier years of the project were biased toward control or baseline data since many restorations had not yet taken place, meaning that treatment is partially confounded with escapement. Together, this makes it more challenging to detect any effects of restoration that may be present when analyzing the full dataset.

To analyze fish density of the full dataset, a zero-inflated linear model was used to examine the effects of treatment (baseline/impact/control) on fish density. Year and quarter (October-December, January-March, April-June, and July-September) were included as fixed effects to account for temporal correlation in fish densities. Likewise, because geographic location may impact fish density, we included river mile as a fixed effect in the model. We used a normal distribution in the model.

Results from these models are presented below; again, we urge extreme caution with interpretation due to the challenges described above. These results alone should not be used to make future management decisions.

Results

Results of the zero-inflated models used for the analysis of fish-per-acre are shown in Tables 5, and 6, as well as Figures 5, 6, and 7. Site Classification (baseline/impact/control) did not have a detectable effect on fish density, with one exception: Steelhead/Rainbow Trout were found at higher densities in impact sites than in control sites.

Table 5. Analysis of Deviance table produced by a zero-inflated model of fish density for the full dataset. *Run was classified using the Central Valley length-to-date chart. Details of the zero-inflated linear mixed models used in these analyses are provided in the methods.*

Run	**Site Classification**	**Year**	**Quarter**	**River Mile**
All salmonids	$\chi^2 = 5.354$ df = 2 p = 0.069	**$\chi^2 = 36.431$** **df = 6** **p < 0.001**	**$\chi^2 = 22.282$** **df = 3** **p < 0.001**	**$\chi^2 = 99.586$** **df = 1** **p < 0.001**
Fall run Chinook	$\chi^2 = 4.055$ df = 2 p = 0.132	**$\chi^2 = 23.939$** **df = 6** **p < 0.001**	**$\chi^2 = 33.167$** **df = 3** **p < 0.001**	**$\chi^2 = 50.714$** **df = 1** **p < 0.001**
Late-fall run Chinook	$\chi^2 = 1.426$ df = 2 p = 0.490	$\chi^2 = 7.580$ df = 6 p = 0.270	$\chi^2 = 2.837$ df = 3 p = 0.417	**$\chi^2 = 3.945$** **df = 1** **p = 0.047**
Winter run Chinook	$\chi^2 = 2.262$ df = 2 p = 0.322	**$\chi^2 = 14.079$** **df = 6** **p = 0.029**	**$\chi^2 = 28.532$** **df = 3** **p < 0.001**	**$\chi^2 = 15.372$** **df = 1** **p < 0.001**
Steelhead / Rainbow Trout	**$\chi^2 = 8.674$** **df = 2** **p = 0.013**	$\chi^2 = 7.482$ df = 6 p = 0.279	**$\chi^2 = 36.721$** **df = 3** **p < 0.001**	**$\chi^2 = 44.991$** **df = 1** **p < 0.001**

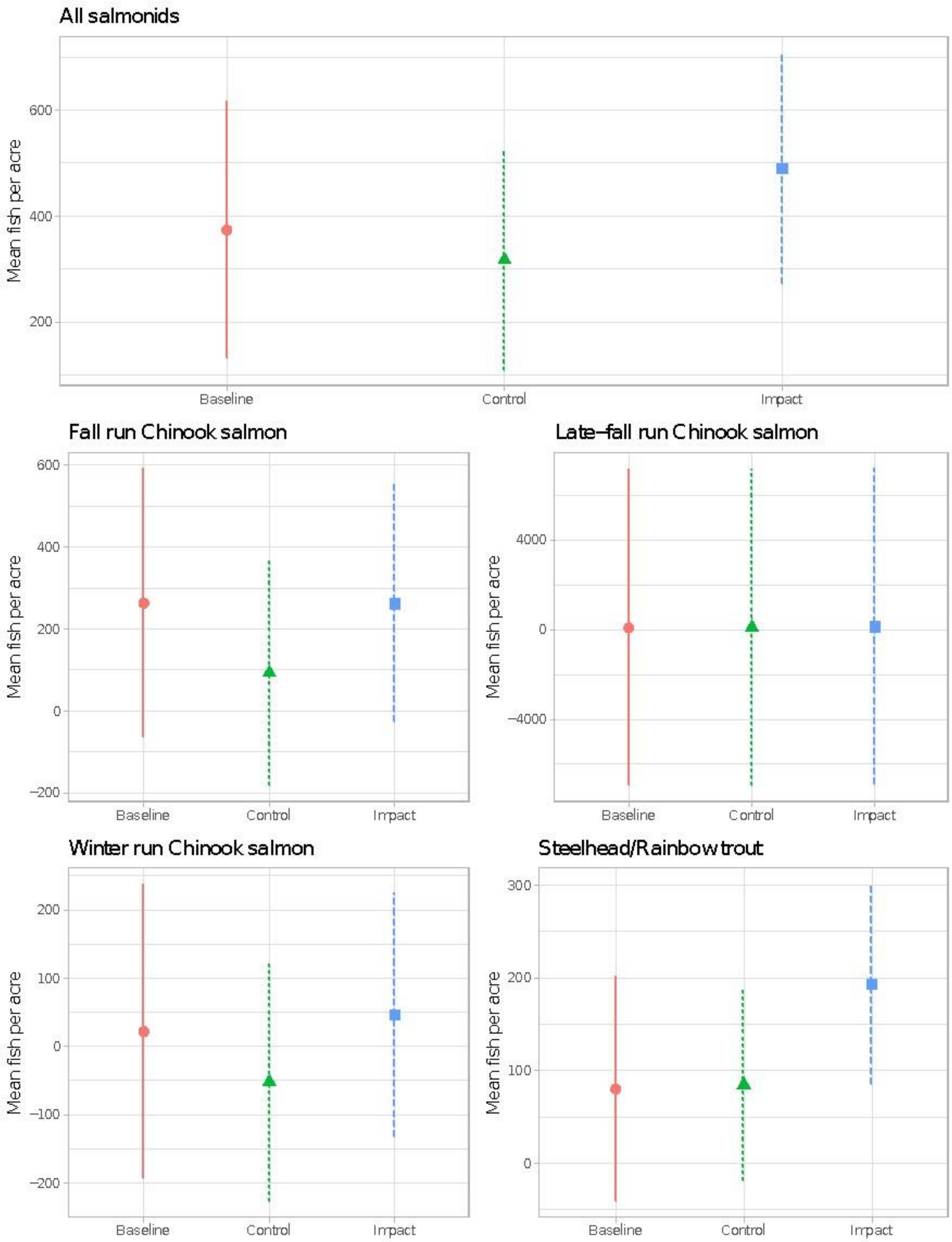

Figure 5. Estimated marginal means of fish density for the full dataset in baseline, control, and impact sites. *Run was classified using the Central Valley length-to-date chart. Details of the zero-inflated models used in these analyses, as well as caveats for the interpretation are detailed in the data analysis text.*

Table 6. Post-hoc comparisons of different site classifications for the zero-inflated model of fish density for the full dataset. *Run was classified using the Central Valley length-to-date chart. Details of the zero-inflated models used in these analyses are provided in the methods. A positive difference value indicates that the first channel status listed has a higher fish density, while a negative difference value indicates the opposite. P-values indicate whether differences in fish density are statistically significant.*

Run	**Difference**	**SE**	**df**	**t-ratio**	**p-value**
All salmonids					
Baseline - Control	-56.4	93.4	983	0.603	0.818
Baseline - Impact	-116.4	89.1	983	-1.036	0.392
Control - Impact	-172.8	76.5	983	-2.259	0.062
Fall run Chinook					
Baseline - Control	170.3	115.9	983	1.469	0.306
Baseline - Impact	1.9	111.1	983	0.018	0.999
Control - Impact	-168.3	89.5	983	-1.882	0.144
Late-fall run Chinook					
Baseline - Control	-19.1	46.9	983	-0.407	0.913
Baseline - Impact	-49.0	45.3	983	-1.080	0.526
Control - Impact	-29.9	35.2	983	0.849	0.673
Winter run Chinook					
Baseline - Control	73.6	90.4	983	0.814	0.694
Baseline - Impact	-24.2	83.8	983	-0.289	0.955
Control - Impact	-97.8	65.3	983	-1.497	0.293
Steelhead / Rainbow Trout					
Baseline - Control	-4.5	51.6	983	-0.089	0.996
Baseline - Impact	-112.9	48.3	983	-2.338	0.051
Control - Impact	**-108.3**	**43.0**	**983**	**-2.520**	**0.032**

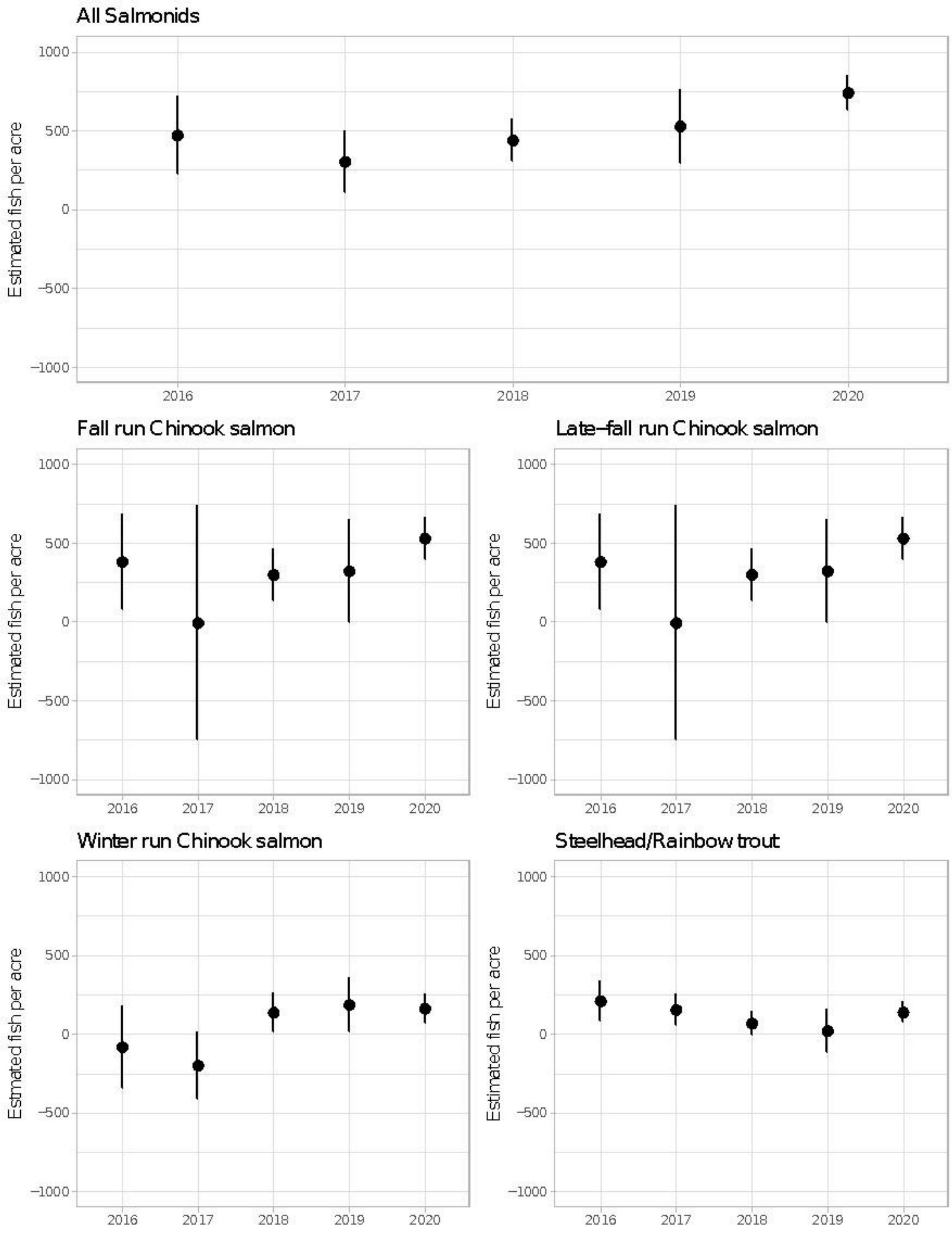

Figure 6. Estimated marginal means of fish density for the full dataset in across years. *Only years with a full calendar year of data are presented. Run was classified using the Central Valley length-to-date chart. Caveats for the interpretation of the modeling approach that these graphs are based on are detailed in the text.*

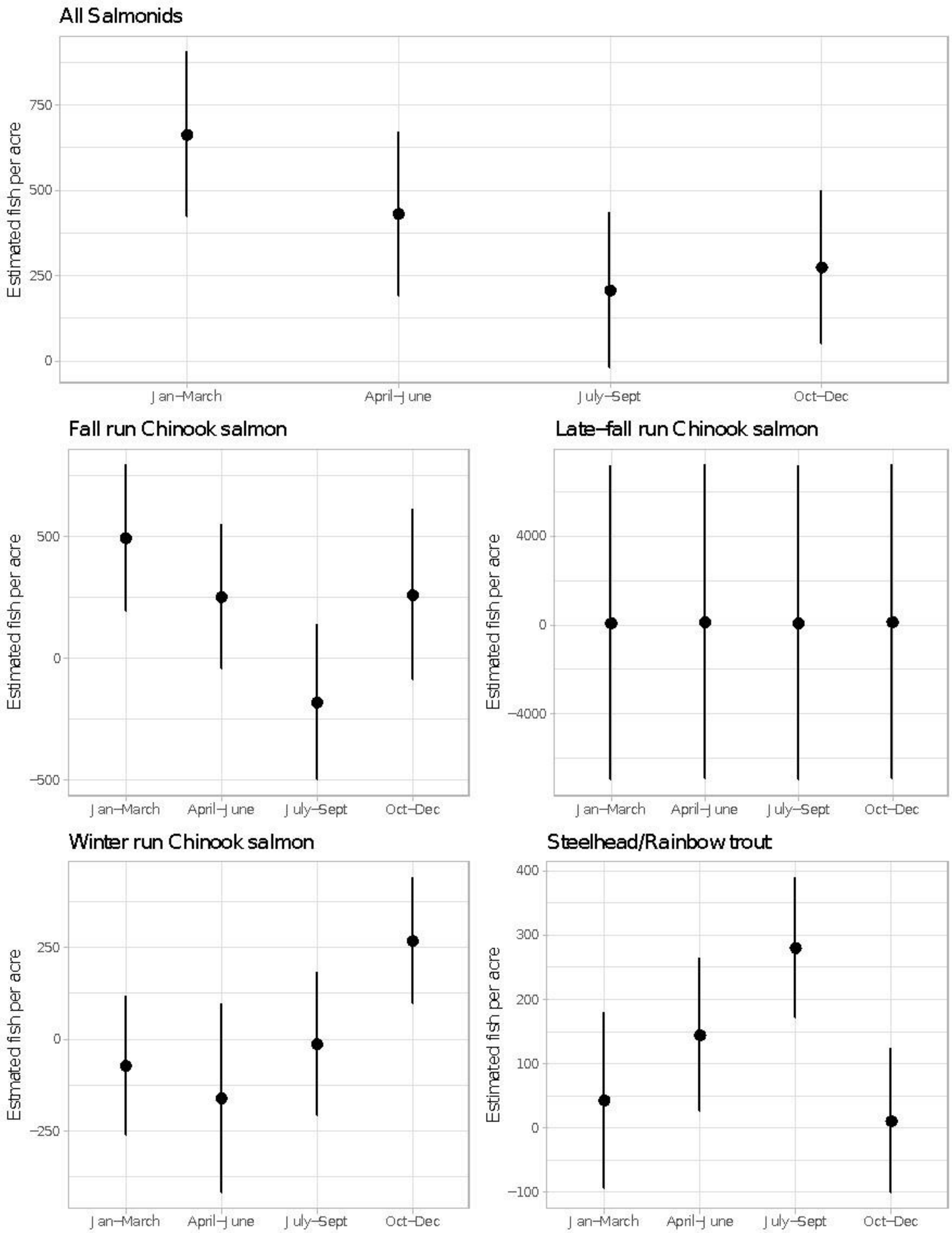

Figure 7. Estimated marginal means of fish density for the full dataset across quarters. *Data is averaged across years. Run was classified using the Central Valley length-to-date chart. Caveats for the interpretation of the modeling approach that these graphs are based on are detailed in the text.*

HABITAT MAPPING: DEPTH, VELOCITY, AND COVER

Field methods to measure depth, velocity, and cover

Habitat mapping was implemented on a schedule that allowed us to map a range of flows. Limited flow regimes during the study period and crew safety concerns prevented us from collecting data at the full range of target flows laid out in the Monitoring Plan (Tussing and Banet, 2017). Instead, data was collected at three flow ranges: low (3,250-3,700 cfs), medium (5,000-7,800 cfs), and high (8,000-11,000 cfs).

Juvenile habitat mapping efforts followed the juvenile habitat suitability criteria of Goodman *et al.* (2015) and apply to age-0 presmolt (>50mm) Chinook Salmon. These criteria include depth, velocity, and distance to cover (Table 7). Cover types were mapped followed the primary cover types previously identified during the study of Flow-Habitat Relationships for Chinook Salmon Rearing in the Sacramento River between Keswick Dam and Battle Creek (USFWS, 2005; Holmes *et al.*, 2014) (Table 8).

Table 7. Juvenile Chinook Salmon Habitat Suitability Criteria **(Goodman *et al.*, 2015)**

Parameter	Upper Range (m)	Upper Range (ft)
Depth	1	3.3
Velocity (m/s)	0.24	0.8
Distance to Cover	0.6	2.0
Definitions		
Unsuitable habitat	Does not meet depth, velocity, or cover criteria	
Suitable habitat	Meets depth & velocity criteria ***or*** cover criteria, but not both	
Optimal habitat	Meets depth, velocity, and cover criteria	

Table 8. Juvenile Salmonid Habitat Cover Types **(USFWS, 2005; Holmes *et al.*, 2014)**

Cover Type	Definition
No cover	No cover
Cobble	3”-12” particle size, < 50% embedded
Boulder	>12” particle size
Fine wood vegetation	<1” Diameter
Branches, small woody debris (SWD)	< 12” Diameter
Log, large woody debris (LWD)	> 12” Diameter
Overhead cover	> 2’ above substrate, < 1.5’ off water surface
Undercut banks	Undercut banks
Aquatic vegetation	In-water vegetative cover
Rip rap	Rip rap

To map depth and velocity, the field crew used a Trimble Geo7x Handheld GPS. Data were collected when the accuracy of the Trimble unit allowed mapping to occur at a scale of one meter or less. Using juvenile depth and velocity suitability criteria identified in Table 7, the crew outlined areas of suitable habitat by measuring depth and velocity using hand-held flow meters on top-setting rods. This allowed identification of discrete polygons throughout the side channel that simultaneously met both depth and velocity criteria (i.e., depth and velocity were not mapped independently). We excluded small habitat areas less than $2m^2$ from perimeter mapping in order to reduce geo-spatial error.

The Trimble GPS was also used to map cover. Using juvenile cover suitability criteria and cover types listed in Tables 7 and 8, the crew outlined the perimeter of in-water escape cover, and georeferenced locations of this outline using the Trimble GPS. The in-water escape cover was mapped separately for each of the cover types without overlapping polygons. In some cases where cover types overlapped, and separate mapping of types was not feasible (e.g., minimum size criteria), the polygon was classified by the dominant cover type. The mapping of unembedded cobble as a cover type is the one exception to the general rule, and was mapped independently and often overlapped with other cover types. Similar to the depth and velocity mapping, we excluded small areas of cover less than $2m^2$ to reduce geo-spatial error from perimeter mapping. The data were processed using Trimble GPS Pathfinder Office software, and imported into ArcGIS in order to determine the proportion of each side channel that met the Goodman *et al.* criteria for depth & velocity, cover, suitable habitat, and optimal habitat for age-0 presmolt (>50mm) Chinook Salmon.

By the end of July 2020, twelve side channels had been mapped at a range of flows: three control sites (Bourbon, Clear Creek, and Wyndham), three restoration sites with both baseline and impact data (Anderson River Park, Reading Island, and Rio Vista), four sites with only impact data (Kapusta, Lake California, Painter’s Riffle, and North Cypress), and two sites with only baseline data (Shea Island and South Cypress). Mapping within each channel occurred over a range of flows, but did not always meet the full range of target flows due to logistical constraints. The statistical analyses reported below exclude cobble and aquatic vegetation as cover types. For cobble, this is because we believe our early estimates of cobble may have been biased due to difficulty detecting cobble in deeper water. Aquatic vegetation was excluded because it created a relationship between flow and cover that was an artifact of seasonal changes in vegetation, making the results of the model misleading. Appendix D presents maps without cobble and aquatic vegetation for all side channel mapping completed by the end of the reporting period. Appendix E presents a complementary set of maps that exclude cobble, but include vegetation.

Data analyses

Proportion of Habitat

Statistical analyses were conducted using R (R Core Team, 2016). The proportion of each habitat classified as suitable or optimal was calculated for each side channel mapped. We initially used linear mixed effects models to determine the effect of channel status (control, baseline, and impact) and flow from Keswick Dam on the proportions of optimal habitat, suitable habitat, and the sum of the two. Because each side channel was measured multiple times at different flows, these models included side channel ID as a random effect in order to account for correlations between measurements within sites. However, the results for baseline data were misleading when using this approach because the suitability criteria from Goodman *et al. (2015)* were not created using backwater habitats or disconnected side channels. Compared to habitat with continuous flow, these types of habitats more often have temperature or oxygen levels that do not fall with acceptable levels for juvenile salmonids. This was not captured in our data, which only focused on depth, velocity, and cover. Because of this, the proportion of suitable and optimal habitat was overestimated in baseline sites. We chose to report the results of models that included only control and restored sites, so that this artifact of data collection does not lead to misinterpretation of the quality of baseline sites.

We used similar linear mixed models to determine the effect of restoration and flow on the proportion of suitable depth & velocity, and suitable cover, which are the component habitat characteristics used to define suitable and optimal habitat. We used the emmeans package in R to conduct post-hoc pairwise comparisons (Lenth, 2018).

Acres of Habitat

Three restoration sites had both baseline and impact data at flows that ranged from 3100 – 5000 cfs. We compiled the total acres of suitable, optimal, and suitable + optimal habitat measured pre-restoration and post-restoration for these sites. These data are presented below with the caveat that baseline sites tended to be backwater or disconnected channels, so the amount of high-quality habitat estimated using the criteria from Goodman *et al.* (2015) is most certainly overestimated in these sites. Because of this, the total habitat gained by restoration within the sites may be considerably underestimated.

For restoration sites that did not have baseline data at a similar flows, we present the total number of acres in the channels after restoration at three different flow regimes: low (3,250-3,700 cfs), medium (5,000-7,800 cfs), and high (8,000-11,000 cfs). We typically did not collect baseline data at these sites because they were dry, disconnected, or otherwise unable to be mapped. However, these data come with a caveat that the total number of acres in restored sites may be overestimating the impact of the restoration if some small amounts of habitat were available prior to restoration. Some sites had data available from multiple days with similar flows; habitat criteria proportions collected at similar flows within each site were averaged in these cases.

Results

Proportion of Habitat

Linear mixed model analyses show that channel status (restored vs control) had no effect on the proportion of any habitat criteria. Flow from Keswick Dam significantly affected all habitat criteria except suitable cover (Table 9, Figure 8).

Table 9. Linear mixed model analyses of the effects of channel status (restored vs control) and flow from Keswick on the amount of habitat available. *Comparisons with significant differences are bolded. Habitat criteria are from Goodman et al. (2015). Analyses include three control sites (Bourbon, Clear Creek, and Wyndham) and impact data from seven restoration sites (Anderson River Park, Reading Island, and Rio Vista, Kapusta, Lake California, Painter's Riffle, and North Cypress). Details are in text. P-values were estimated using Kenward-Rogers degrees of freedom.*

Habitat Classification	**Channel Status**	**Flow**
All Suitable	$F_{1,8.5504} = 0.018$ $p = 0.897$	**$F_{1,31.5041} = 57.623$** **$p < 0.001$**
All Optimal	$F_{1,9.8364} = 0.0206$ $p = 0.889$	**$F_{1,31.5125} = 9.083$** **$p = 0.005$**
Suitable + Optimal	$F_{1,9.4866} = 0.0226$ $p = 0.8836$	**$F_{1,31.5110} = 58.753$** **$p < 0.001$**
Suitable Depth & Velocity	$F_{1,9.303} = 0.0079$ $p = 0.9311$	**$F_{1,31.663} = 78.539$** **$p < 0.001$**
Suitable Cover	$F_{1,9.5278} = 0.2402$ $p = 0.6352$	$F_{1,30.1149} = 0.0010$ $p = 0.9744$

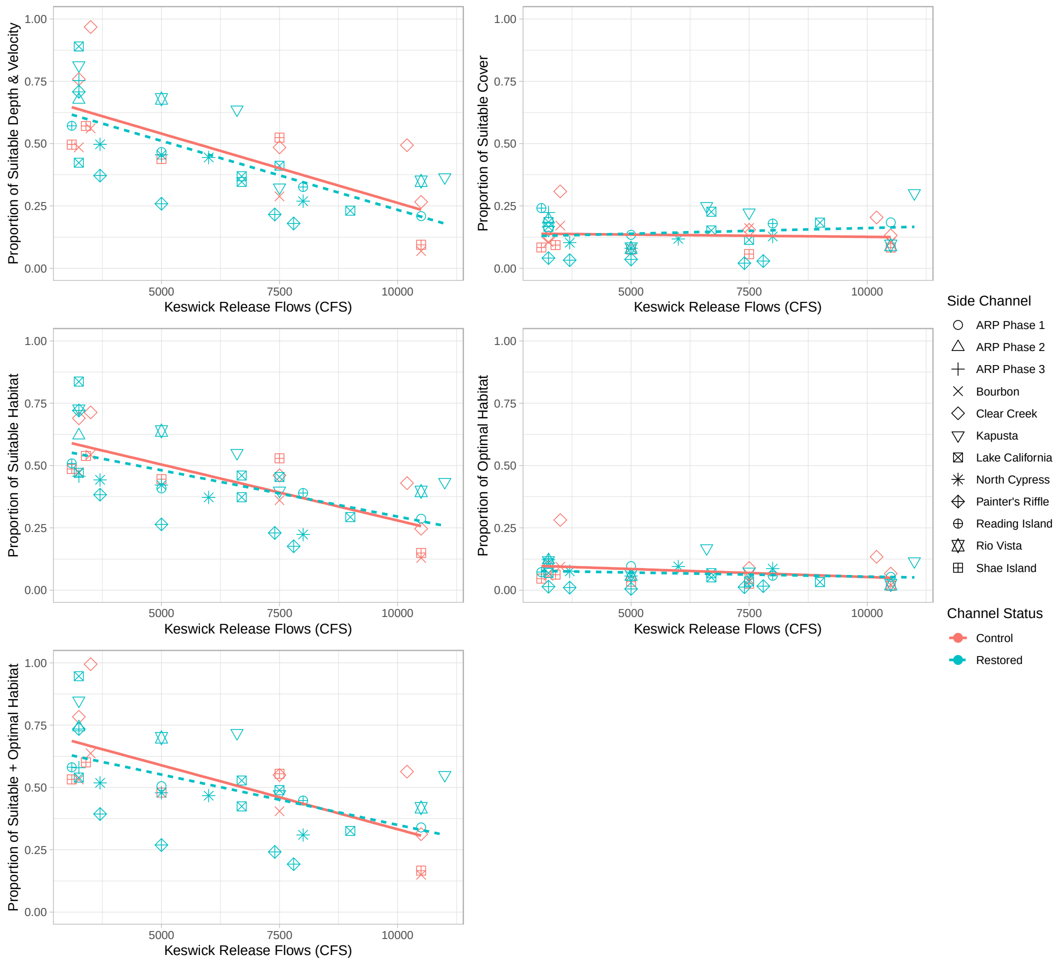

Figure 8. Proportion of habitat that has suitable depth & velocity, suitable cover, suitable habitat, optimal habitat, and suitable + optimal habitat found across a range of flows*. Habitat criteria are from Goodman et al. (2015). See text for more detail. Points represent individual sampling days and sites. Shading represents the 95% confidence bands.*

Acres of Habitat

In sites that had sufficient paired baseline/impact data (Rio Vista, Reading Island, and Anderson River Park phase 1), the total amount of suitable, optimal, and suitable + optimal habitat increased after restoration (Figure 9). However, as discussed above, these data likely underestimate the positive effect of restoration because baseline sites were backwater or disconnected areas, which are more likely to have oxygen and temperature ranges outside of an acceptable range. Because our data only focuses on depth, velocity, and cover, the amount of suitable and optimal habitat are most certainly overestimated in baseline sites.

Baseline habitat mapping data was not collected at Lake California, North Cypress, Painter's Riffle, and Kapusta. Before restoration, the sites were dry, disconnected, or assumed to be uninhabitable, with no suitable or optimal habitat. After restoration, the documented amount of suitable and optimal habitat in these sites ranged from 6.9 to 4.0 acres, with lower flows having more available habitat (Figure 10). Note that if our initial assumption (that these sites were uninhabitable) was incorrect, then we may be slightly overestimating the impact of the restoration in these sites.

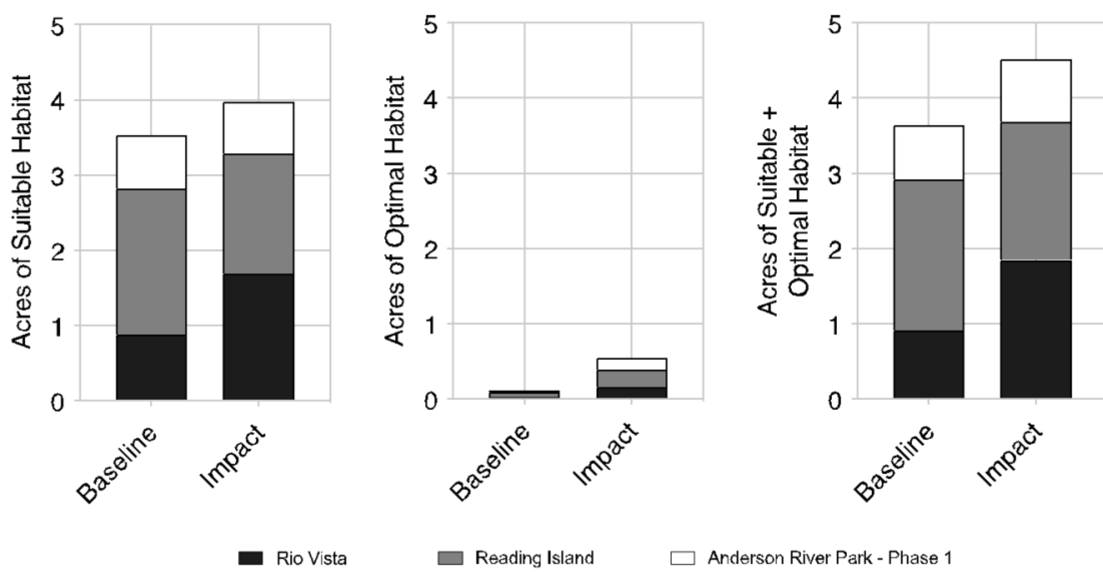

Figure 9. Acres of habitat from three restoration sites that have both baseline and impact data at flows ranging from 3100-5000 cfs. *Habitat criteria are from Goodman et al. (2015) and do not consider factors such as temperature and oxygen, which may differ between baseline and restored data. For this reason, the actual acreage of high-quality habitat in baseline sites may be overestimated. <u>This could lead to an underestimate of the positive effect of restoration.</u>*

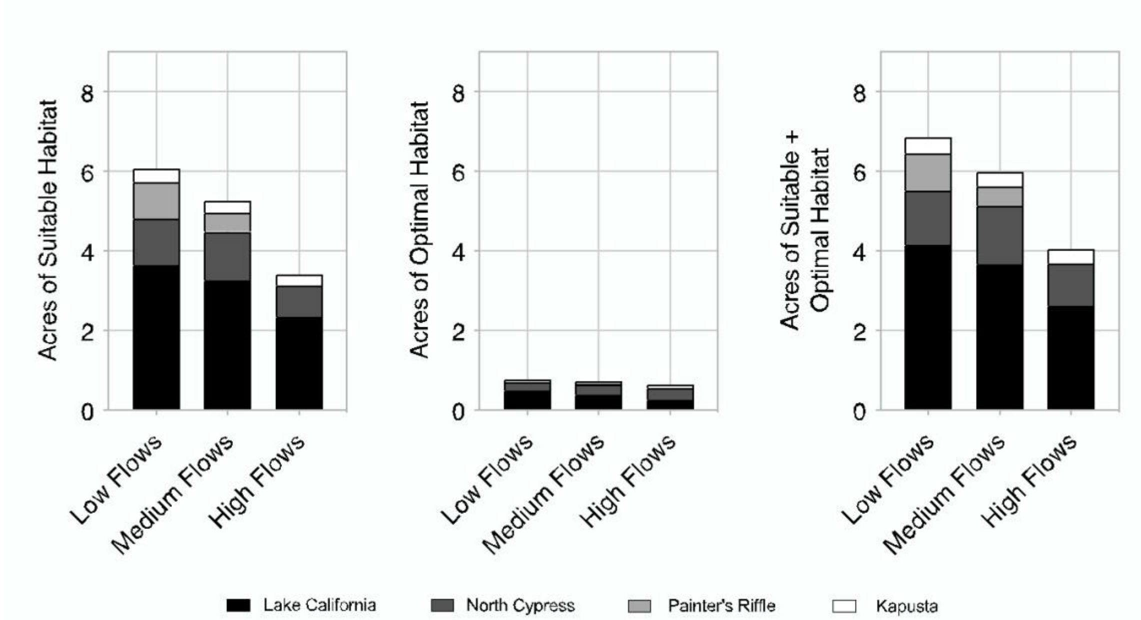

Figure 10. Acres of habitat available across flows from four restored sites that did not have baseline data collection. *Low flows ranged from 3,250-3,700 cfs, medium flows ranged from 5,000-7,800 cfs, and high flows range from 8,000-11,000 cfs. Painter's Riffle was not mapped at high flows. Baseline data were not collected at these sites because they were dry, disconnected, or assumed to be uninhabitable. Thus, these numbers may slightly overestimate the habitat created by restoration if some small amounts of habitat were available prior to restoration.*

MICROHABITAT USE: HABITAT SUITABILITY CURVES AND COVER PREFERENCE

Field Methods to Measure Microhabitat Use

We use the microhabitat data to address two main questions. First, we ask whether the habitat suitability criteria described and applied above (Goodman *et al.*, 2015), which were originally developed for juvenile salmonids > 50mm in the Trinity River, perform similarly when applied to our river system. Second, we ask whether fry and juvenile Chinook Salmon and steelhead/Rainbow Trout display preference for specific cover types. In order to evaluate these questions, microhabitat-use snorkel surveys were conducted across a range of flows. We used stratified random sampling to select habitats for inclusion in data collection for microhabitat use, in order to ensure the full range of available habitat types were captured and that a commensurate amount of surface area was sampled for each habitat type. Surveys focused on both suitable and unsuitable habitat (as defined in Table 7) in order to establish the difference between fish use of preferred vs. available habitat.

For selected habitat units, snorkelers worked in an upstream direction and at a slow pace to observe the point locations of undisturbed fish. The location of fish observed was marked with a weighted tag on the stream bottom. The species, run, size, and number of fishes were recorded on tags for any observed salmonids less than 201mm in fork length. Estimates of fish size and selection of the appropriate size class bin was aided by the use of a dive cuff with photographs of salmonids at class bin lengths. Size class bins included fork lengths of <41mm, 41-50mm, 5160mm, and then by 20mm bin widths up to a maximum of 200mm. These were subsequently categorized as fry (≤ 50 mm) or juveniles (>50 mm) for analyses.

After the habitat unit was surveyed, flagged locations were revisited, and data was collected on fish attributes, GPS point location, habitat type, depth (total water column), distance to bank, distance to cover, cover type, mean water column velocity, and substrate. Due to safety concerns, snorkeling surveys were restricted to flows below 13,000 CFS. This resulted in a shortage of late-fall run observations, as they are typically most abundant at high flows.

Habitat Suitability Curves

Data Analysis

To determine whether the habitat suitability criteria (HSC) developed on the Trinity River by Goodman *et al.* (2015) are appropriate to map juvenile salmonid habitat on the side channels of the Sacramento River, HSC specific to the side channels of the study area were developed (hereafter referred to as "study habitat suitability criteria" or "study HSC") for each respective run (fall-run, late fall-run, winter-run, spring-run, or steelhead trout) and size-class (fry ≤ 50 mm fork length, or juveniles >50 mm fork length) of salmonid. Microhabitat-use surveys were carried out on 12 side channels in the study area and used to produce habitat utilization curves (Bovee, 1986). From these utilization curves we are able to produce nonparametric tolerance limits, or ranges of a habitat parameter that would capture 75% of the observed population at 95% confidence to comprise our study HSC. The nonparametric tolerance limits were computed using programming language Julia (Bezanson *et al.*, 2017) and were cross-checked against the tables in Somerville (1958). In order to mimic the proportion of observations captured by the criteria in Goodman's 2014 study, 75% was chosen as the proportion of observations to fall within the parameter ranges for the study HSC. Examination of the study HSC can be a good indicator of whether the area mapped might be overestimating or underestimating the total area of suitable habitat for a given run or size class. For this analysis, observations from reference, restored, and baseline side channels were combined because we did not expect habitat preferences of a population of fish to depend on the category of side channel in which it was observed. Each site was snorkeled at three respective flows each year, in order to give equal snorkeling effort to each available side channel.

Results

Study habitat suitability criteria, as determined by 75% nonparametric tolerance limits, differed between run and life stage (Table 10). Histograms depicting the distribution of observations across velocities, depths, and distances to cover are displayed with associated study HSC (Figures 11, 12, and 13).

Table 10. Nonparametric tolerance limits are reported as a range of values that could be used to characterize 75% of the observations at the 95% confidence level*. For each demographic group, tolerance limits are reported for water velocity, depth, and distance to the nearest suitable cover of the location of each observation. The number of individual fish observations for each group is listed in the far-right column. Additionally, ranges for microhabitat mapping criteria from Goodman et al. (2015) are reported in bold at the top for comparison.*

	Velocity (m/s)	**Depth (m)**	**Distance to Cover (m)**	**Number of Fish (*n*)**
Goodman et al. Suitability Criteria	**0-0.24**	**0-1.0**	**0-0.6**	
All Salmonids	0.000-0.174	0.183-0.658	0.000-0.610	25,914
All Salmonid Fry	0.000-0.095	0.152-0.427	0.000-0.427	12,595
All Salmonid Juveniles	0.000-0.222	0.244-0.762	0.000-0.701	13,319
Chinook salmon				
Fall Run Fry	0.000-0.094	0.152-0396	0.000-0.366	7,523
Late Fall Run Fry	0.012-0.070	0.183-0.366	0.000-0.061	547
Winter Run Fry	0.000-0.101	0.198-0.506	0.000-0.634	946
Spring Run Fry	0.003-0.122	0.152-0.427	0.000-0.786	622
Fall Run Juveniles	0.000-0.216	0.229-0.792	0.000-0.457	5,485
Late Fall Run Juveniles	0.000-0.311	0.305-0.731	0.000-0.518	338
Winter Run Juveniles	0.000-0.146	0.244-0.640	0.000-1.320	3,159
Spring Run Juveniles	0.006-0.290	0.335-1.067	0.000-0.640	854
Steelhead/Rainbow Trout				
Rainbow Trout Fry	0.000-0.104	0.183-0.427	0.000-0.305	2,996
Rainbow Trout Juveniles	0.003-0.302	0.244-0.853	0.000-0.610	2,951

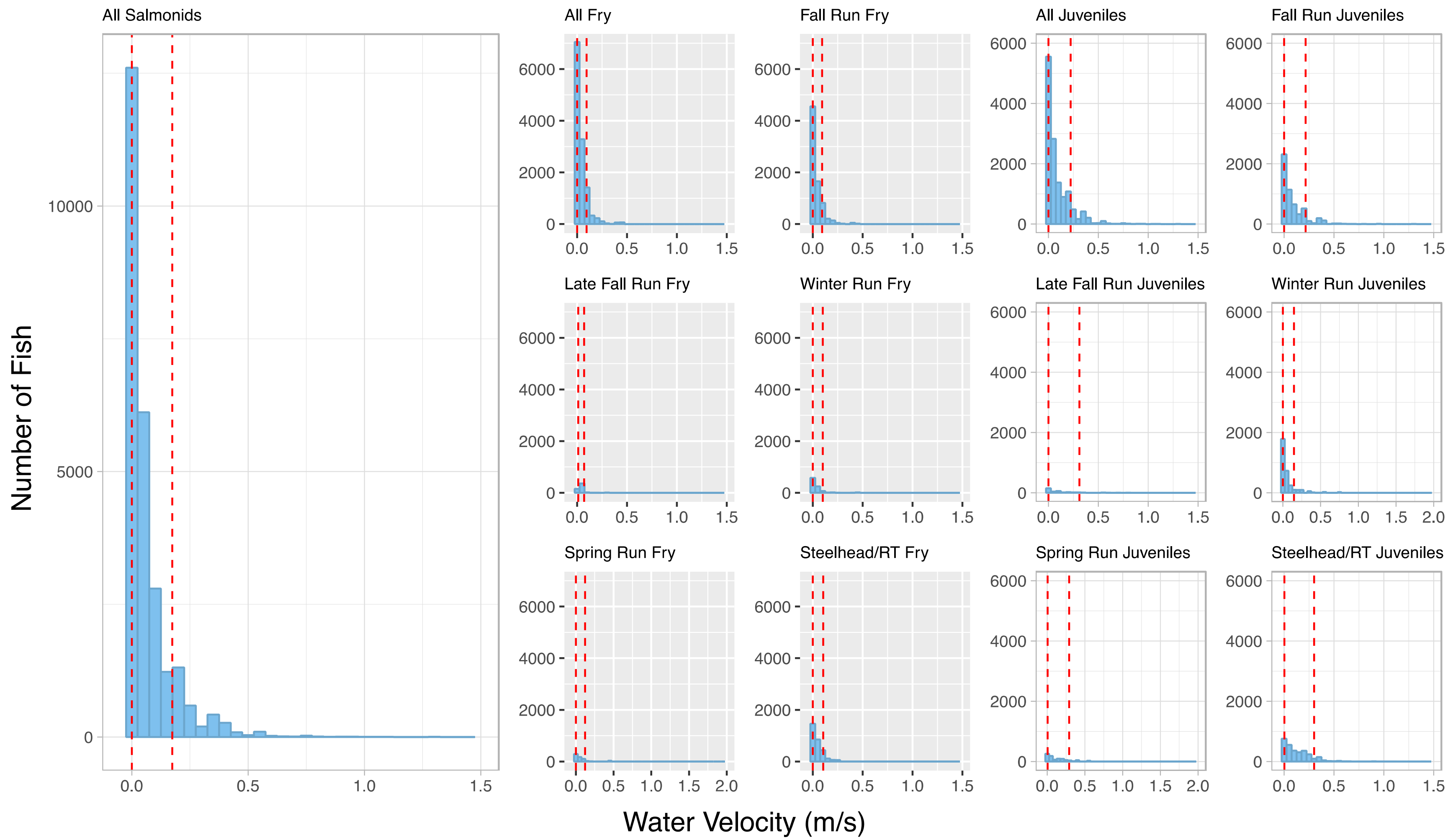

Figure 11. Water velocity tolerance limits for different runs and size classes. *Red lines represent the tolerance limits that capture 75% of the observed population at a 95% confidence level.*

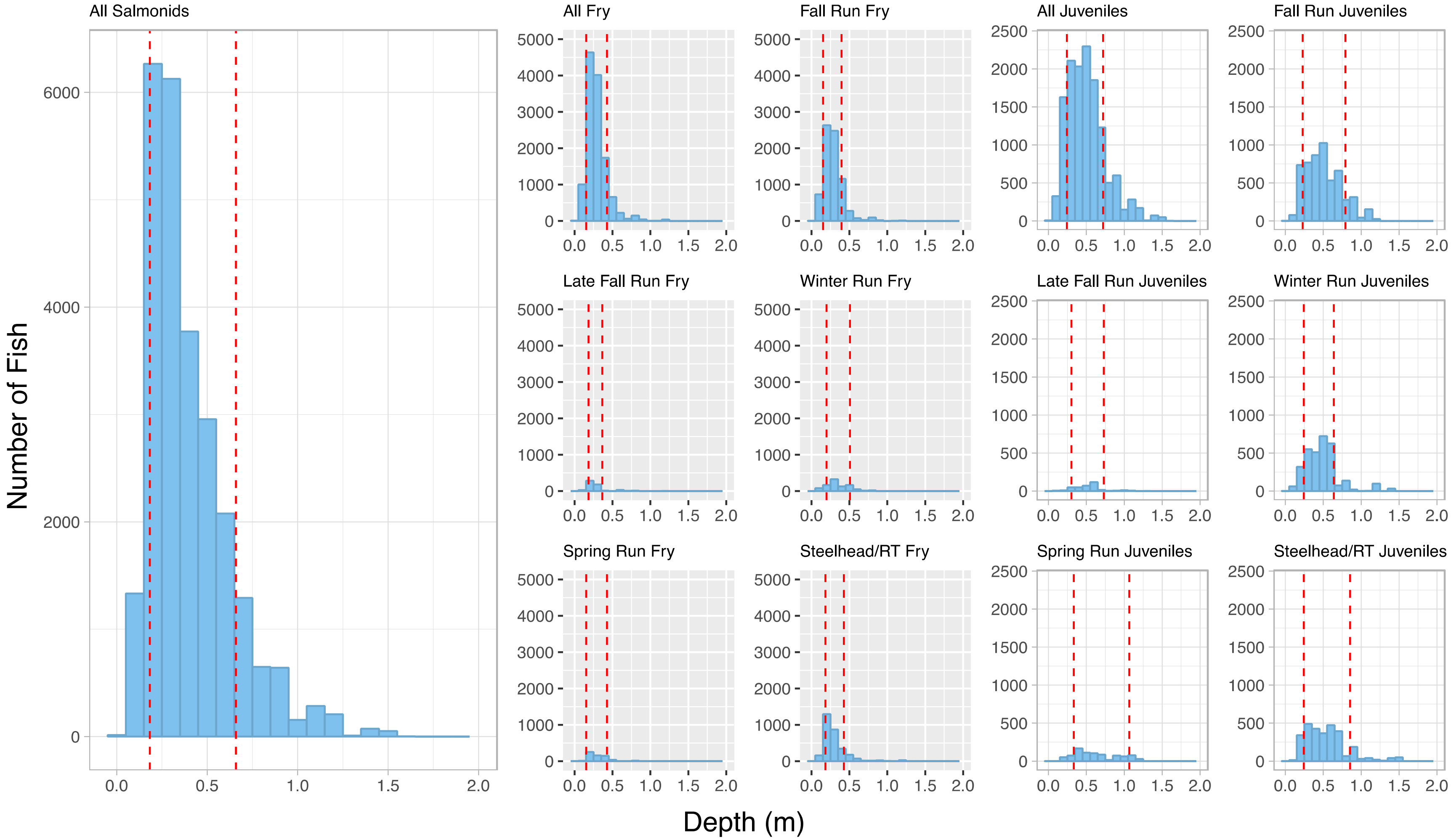

Figure 12. Water depth tolerance limits for different runs and size classes. *Red lines represent the tolerance limits that capture 75% of the observed population at a 95% confidence level.*

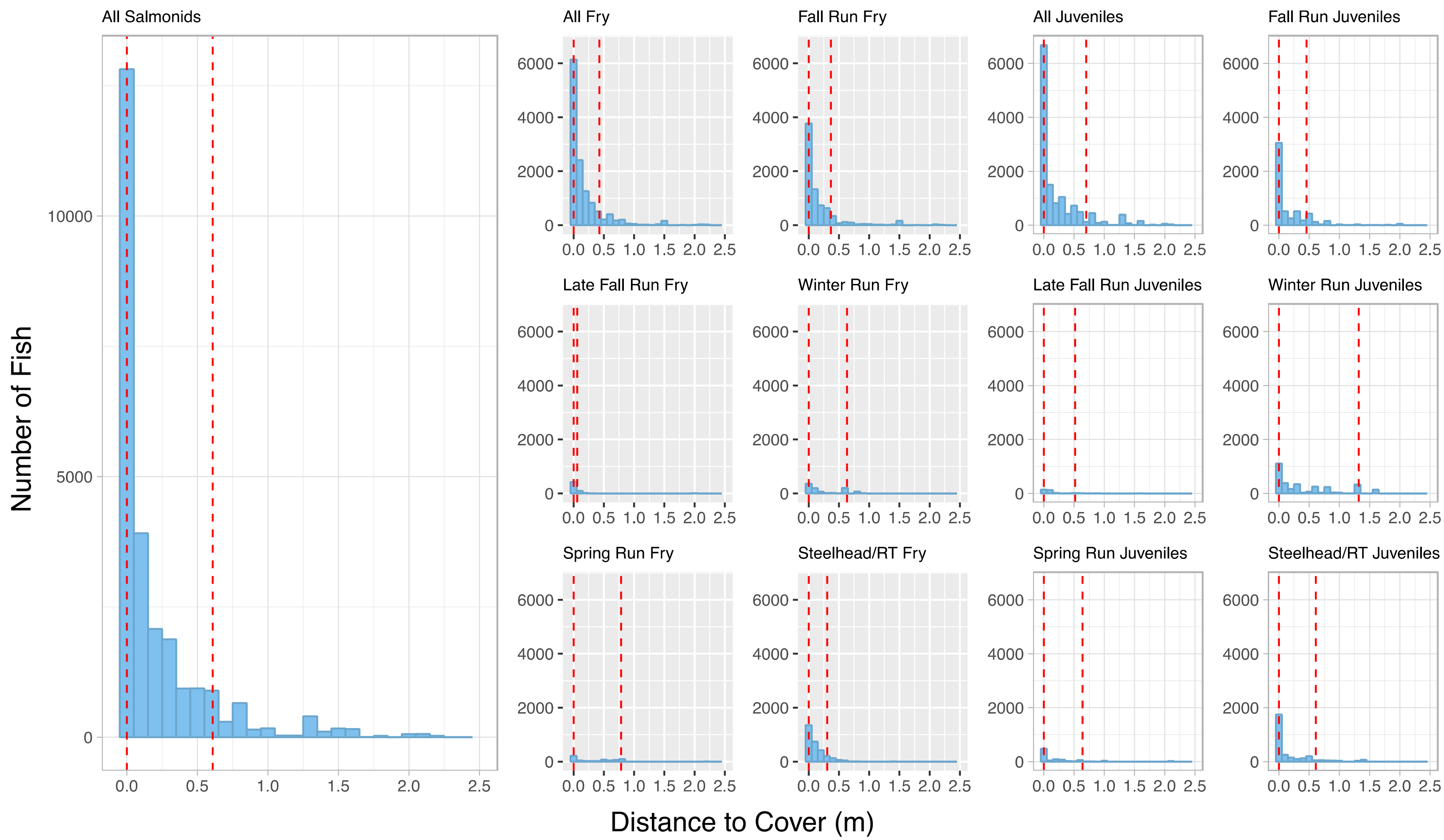

Figure 13. Distance to cover tolerance limits for different runs and size classes. *Red lines represent the tolerance limits that capture 75% of the observed population at a 95% confidence level.*

Cover Preference

Data Analysis

Preference for different cover types was explored by comparing the proportion of fish found in each cover type with the proportion of area each cover type occupies at a specific site. We assume that a higher proportion of fish found in cover types that make up relatively less square footage of a site indicates preference for that cover type. Thus, preference is defined as:

$$\text{Preference} = \frac{F_{cover}}{F_{total}} - \frac{A_{cover}}{A_{total}}$$

where F_{cover} represents the number of fish observed in a given cover type, F_{total} represents the number of fish observed in all cover types, A_{cover} represents the area of a given cover type, and A_{total} represents the total area surveyed.

Analysis of cover preference data was constrained due to the inherent issues of analyzing groups that make up a proportion of a whole. We ran an ANOVA that examined whether fish preference was a function of cover type or the interaction between channel status and cover type. Separate tests were run for Chinook fry, Chinook juveniles, steelhead/Rainbow Trout fry, and steelhead/Rainbow Trout juveniles. When an ANOVA identified a significant difference, we performed additional post-hoc pairwise comparisons to determine which mean(s) are different. Combinations that are of interest are reported below. All p-values were adjusted to control for multiple comparisons and maintain a family-wise confidence level of 95% using Tukey's Honest Significant Difference.

Results

All groups (Chinook Salmon fry, Chinook Salmon juveniles, steelhead/Rainbow Trout fry, and steelhead/Rainbow Trout juveniles) showed a significant difference in preference scores between cover types, and these differences were consistent between restored and control sites (Table 11, Figure 14). Post-hoc tests showed that all groups significantly preferred fine woody debris to all other cover types except undercut banks, for which they showed a similar preference (Tables 13 and 14). Chinook Salmon fry and juveniles significantly preferred undercut banks to overhead cover (Table 12). Fry of both species significantly preferred undercut banks to branches and small woody debris (Tables 13 and 14).

Table 11. ANOVA examining the effect of channel status*cover type on cover preference for Chinook fry, Chinook juveniles, steelhead/rainbow trout fry, and steelhead/rainbow trout juveniles.

	Chinook fry (all runs)	**Chinook juveniles (all runs)**	**Steelhead/ Rainbow trout fry**	**Steelhead/ Rainbow trout juveniles**
Cover Type	$\mathbf{F_{7,70}= 8.7098}$ **p < 0.001**	$\mathbf{F_{7,70}= 6.2953}$ **p < 0.001**	$\mathbf{F_{7,70}= 6.9356}$ **p < 0.001**	$\mathbf{F_{7,70}= 3.6789}$ **p < 0.001**
Channel Status* Cover Type	$F_{7,70}= 2.0651$ $p = 0.0588$	$F_{7,70}= 0.8685$ $p = 0.5357$	$F_{7,70}= 0.4337$ $p = 0.8774$	$F_{7,70}= 0.9240$ $p = 0.4935$

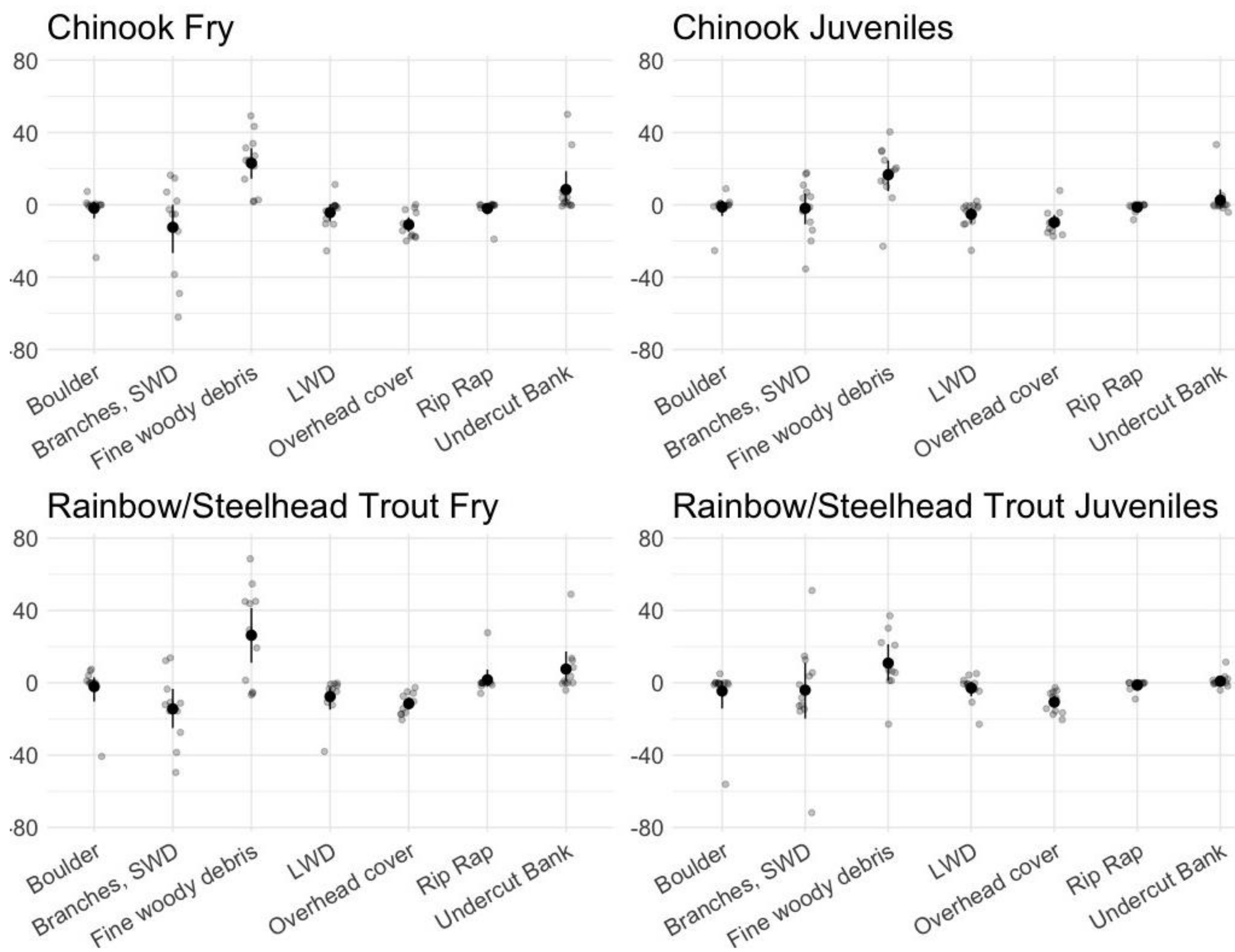

Figure 14. Cover preference index for Chinook salmon (all runs) and steelhead/rainbow trout fry and juveniles in control and impact habitat. Values above zero indicate fish were found at those cover types more than expected based on a random distribution, indicating a positive preference. Negative values suggest the inverse relationship. SWD is small woody debris, and LWD is large woody debris.

Table 12. Post-hoc comparisons of Chinook Salmon fry and juvenile preferences for different cover types. *A positive difference value means the first cover type listed in the contrast is preferred. A negative value indicates the second cover type listed is preferred. The magnitude of the difference indicates the strength of the preference, and P-values indicate whether these preferences are statistically significant.*

Run & Contrast	**Difference**	**SE**	**df**	**t-ratio**	**p-value**
Chinook Salmon Fry					
Boulder – Branches, SWD	12.748	5.52	70	2.310	0.254
Boulder – Fine Woody Debris	**-25.194**	**5.52**	**70**	**-4.565**	**<0.001**
Boulder - LWD	2.514	5.52	70	0.455	0.999
Boulder – Overhead Cover	9.895	5.52	70	1.793	0.557
Boulder – Rip Rap	0.115	5.52	70	0.021	1.000
Boulder – Undercut Bank	-11.300	5.52	70	-2.047	0.395
Branches, SWD – Fine Woody Debris	**-37.943**	**5.52**	**70**	**-6.875**	**<0.001**
Branches, SWD – LWD	-10.235	5.52	70	-1.854	0.5169
Branches, SWD – Overhead Cover	-2.854	5.52	70	-0.517	0.999
Branches, SWD – Rip Rap	-12.634	5.52	70	-2.289	0.2639
Branches, SWD – Undercut Bank	**-24.049**	**5.52**	**70**	**-4.375**	**<0.001**
Fine Woody Debris - LWD	**27.708**	**5.52**	**70**	**5.020**	**0.001**
Fine Woody Debris – Overhead Cover	**35.089**	**5.52**	**70**	**6.358**	**<0.001**
Fine Woody Debris – Rip Rap	**25.309**	**5.52**	**70**	**4.586**	**<0.001**
Fine Woody Debris – Undercut Bank	13.894	5.52	70	2.517	0.1690
LWD – Overhead Cover	7.381	5.52	70	1.337	0.832
LWD – Rip Rap	-2.399	5.52	70	-0.435	1.000
LWD – Undercut Bank	-13.814	5.52	70	-2.503	0.174
Overhead Cover – Rip Rap	-9.780	5.52	70	-2.503	0.174
Overhead Cover – Undercut Bank	**-21.195**	**5.52**	**70**	**-3.840**	**0.005**
Rip Rap – Undercut Bank	-11.415	5.52	70	-2.068	0.3827
Chinook Salmon Juveniles					
Boulder – Branches, SWD	0.993	4.39	70	0.227	1.000
Boulder – Fine Woody Debris	**-16.878**	4.39	**70**	**-3.848**	**0.005**
Boulder - LWD	3.8131	4.39	70	0.869	0.976
Boulder – Overhead Cover	9.117	4.39	70	2.079	0.377
Boulder – Rip Rap	0.087	4.39	70	0.020	1.000
Boulder – Undercut Bank	-4.323	4.39	70	-0.986	0.956
Branches, SWD – Fine Woody Debris	**-17.872**	4.39	**70**	**-4.075**	**0.002**
Branches, SWD - LWD	2.819	4.39	70	0.643	0.995
Branches, SWD – Overhead Cover	8.123	4.39	70	1.852	0.518
Branches, SWD – Rip Rap	-0.907	4.39	70	-0.207	1.000
Branches, SWD – Undercut Bank	-5.317	4.39	70	-1.212	0.887
Fine Woody Debris - LWD	**20.691**	4.39	**70**	**4.718**	**<0.001**
Fine Woody Debris – Overhead Cover	**25.995**	4.39	**70**	**5.927**	**<0.001**
Fine Woody Debris – Rip Rap	**16.965**	4.39	**70**	**3.868**	**0.004**
Fine Woody Debris – Undercut Bank	12.555	4.39	70	2.863	0.077
LWD – Overhead Cover	5.304	4.39	70	1.209	0.888
LWD – Rip Rap	-3.737	4.39	70	-0.850	0.979
LWD – Undercut Bank	-8.136	4.39	70	-1.855	0.517
Overhead Cover – Rip Rap	-9.030	4.39	70	-2.059	0.388
Overhead Cover – Undercut Bank	**-13.440**	4.39	**70**	**-3.064**	**0.046**
Rip Rap – Undercut Bank	-4.4093	4.39	70	-1.005	0.095

Table 13. Post-hoc comparisons of steelhead/Rainbow trout fry and juvenile preferences for different cover types. *A positive difference value means the first cover type listed in the contrast is preferred. A negative value indicates the second cover type listed is preferred. The magnitude of the difference indicates the strength of the preference, and P-values indicate whether these preferences are statistically significant.*

Run & Contrast	Difference	SE	df	t-ratio	p-value
Steelhead / Rainbow Trout Fry					
Boulder – Branches, SWD	12.83	6.92	63	1.853	0.519
Boulder – Fine Woody Debris	**-28.61**	**6.92**	**63**	**-4.132**	**0.002**
Boulder - LWD	5.42	6.92	63	0.782	0.986
Boulder – Overhead Cover	9.81	6.92	63	1.417	0.7907
Boulder – Rip Rap	-3.42	6.92	63	-0.493	0.999
Boulder – Undercut Bank	-9.13	6.92	63	-1.318	0.841
Branches, SWD – Fine Woody Debris	**-41.44**	**6.92**	**63**	**-5.984**	**<0.001**
Branches, SWD – LWD	-7.41	6.92	63	-1.071	0.934
Branches, SWD – Overhead Cover	-3.02	6.92	63	-0.436	0.999
Branches, SWD – Rip Rap	-16.24	6.92	63	-2.346	0.239
Branches, SWD – Undercut Bank	**-21.95**	**6.92**	**63**	**-3.171**	**0.036**
Fine Woody Debris - LWD	**34.02**	**6.92**	**63**	**4.914**	**0.001**
Fine Woody Debris – Overhead Cover	**38.42**	**6.92**	**63**	**5.549**	**<0.001**
Fine Woody Debris – Rip Rap	**25.19**	**6.92**	**63**	**3.638**	**0.001**
Fine Woody Debris – Undercut Bank	19.48	6.92	63	2.814	0.087
LWD – Overhead Cover	4.40	6.92	63	0.635	0.995
LWD – Rip Rap	-8.83	6.92	63	-1.275	0.861
LWD – Undercut Bank	-14.54	6.92	63	-2.100	0.365
Overhead Cover – Rip Rap	-13.23	6.92	63	-1.910	0.482
Overhead Cover – Undercut Bank	-18.94	6.92	63	-2.735	0.106
Rip Rap – Undercut Bank	-5.71	6.92	63	-0.825	0.982
Steelhead / Rainbow Trout Juveniles					
Boulder – Branches, SWD	-0.891	7.29	70	-0.122	1.000
Boulder – Fine Woody Debris	**-24.641**	7.29	**70**	**-3.378**	**0.020**
Boulder - LWD	-1.145	7.29	70	-0.157	1.000
Boulder – Overhead Cover	7.205	7.29	70	0.988	0.955
Boulder – Rip Rap	-2.805	7.29	70	-0.385	1.000
Boulder – Undercut Bank	-5.126	7.29	70	-0.715	0.991
Branches, SWD – Fine Woody Debris	**-23.750**	7.29	**70**	**-3.256**	**0.028**
Branches, SWD - LWD	-0.254	7.29	70	-0.035	1.000
Branches, SWD – Overhead Cover	8.096	7.29	70	1.110	0.923
Branches, SWD – Rip Rap	-1.914	7.29	70	-0.262	1.000
Branches, SWD – Undercut Bank	-4.325	7.29	70	-0.593	0.997
Fine Woody Debris - LWD	**23.496**	7.29	**70**	**3.221**	**0.030**
Fine Woody Debris – Overhead Cover	**31.845**	7.29	**70**	**4.366**	**<0.001**
Fine Woody Debris – Rip Rap	**21.836**	7.29	**70**	**2.994**	**0.056**
Fine Woody Debris – Undercut Bank	19.425	7.29	70	2.663	0.123
LWD – Overhead Cover	8.349	7.29	70	1.145	0.912
LWD – Rip Rap	-1.660	7.29	70	-0.228	1.000
LWD – Undercut Bank	-4.071	7.29	70	-0.558	0.998
Overhead Cover – Rip Rap	-10.009	7.29	70	-1.372	0.814
Overhead Cover – Undercut Bank	-12.421	7.29	70	-1.703	0.616
Rip Rap – Undercut Bank	-2.411	7.29	70	-0.331	1.000

FISH SIZE AND CONDITION

Field Methods to Estimate Fish Size and Condition

Fish size and condition data were collected through the use of seining at a variety of sites both within side channels and in the mainstem Sacramento River in the vicinity of side channels. Within each side channel, three permanent seining sites were established. We chose sites that were free of in-water obstructions; would be seinable at the range of targeted flows (3,250 to 13,000 cfs Keswick releases); and represented a riffle, flatwater, and a pool habitat types. Three permanent seining sites were also selected in the mainstem river in the vicinity of side channels that met the same criteria. Mainstem sites captured the diversity of velocity and depth characteristics present rather than specific habitat types, which occur on much larger spatial scales.

Seining occurred routinely from 12/2018 – 6/2019 and 12/2019 – 3/2020. Each pair of side channel/mainstem sites were sampled on the same day, and it took approximately 10 days to sample all side channel/mainstem paired sites for each sampling event. Two seine pulls were applied at each permanent sampling site and all salmonids captured were identified to run, enumerated, and measured for fork lengths (mm) and weights (to the nearest 0.01 g). Seines used were of a wandering pole type with a purse and 30’ in total length. Surface area seined and average depths were measured and recorded. When seining at fixed sites did not yield sufficient numbers of fish to establish size and condition, roving seining consisting of single seine sets were applied anywhere that was conducive to sampling in side channels and the mainstem.

Data analyses

We used fork length and Fulton’s condition factor (K) as our metrics for fish size and condition (Ricker, 1975). Fulton’s Condition Factor is represented by the equation:

$$K = 10^5 \left(\frac{w}{L^3}\right)$$

where *L* equals the fork length of the fish in centimeters and *w* is the mass of the fish in grams. A larger *K* indicates a “chunkier” fish with presumably more fat reserves and a better condition for outmigration.

No additional data on size and condition was collected this reporting year, but in the 2019-20 report, we were unable to fit models to data from the 2019-20 reporting year. This year, we present results from analyses that pool all size and condition data, ranging from December 2018 to March 2020. We used a linear model to analyze the effect of site type (control side channel, mainstem, and post-restoration side channel) on fork length. Month and year were included as fixed effects to adjust for temporal variation within the dataset for all runs except Late-fall Chinook Salmon. For these fish, only month was included because all data was from a single calendar year.

Results

An important caveat to the results reported below is that fish were classified into run based on fork lengths, not genetic analyses. This means that we are analyzing size in groups that were classified by size. If these classifications have errors, the reported analyses may produce misleading results.

For all runs, there was a significant effect of site type on fork length (Table 14, Figure 15). Post hoc analysis (Table 15) shows that the trends between runs were not consistent. Fall run Chinook Salmon in pre-restoration side channels had larger fork lengths than all other site classifications, and control side channels had longer fish than both mainstem or post-restoration sites. Post hoc analyses for Late-fall run Chinook Salmon did not detect significant differences in fork length between any of the site classifications. Winter run Chinook Salmon had significantly larger fork lengths in restored side channels, as compared to mainstem sites or control side channels. Spring run Chinook Salmon had larger fork lengths in control side channels than in mainstem sites or post restoration side channels. Steelhead/Rainbow Trout had significantly higher fork lengths in both control and restored side channels, as compared to mainstem sites.

For all Chinook Salmon runs, there was a significant effect of site type on Fulton's Condition Factor (K) (Table 16, Figure 16). No effect was seen for Steelhead/Rainbow Trout (Table 16, Figure 16). As with fork length, post hoc analysis (Table 17) shows that the trends between runs were not consistent. Fall run Chinook Salmon in control side channels had significantly higher K values than both mainstem and post-restoration sites. Late-fall run Chinook Salmon in control side channels had significantly higher K values than both mainstem and post-restoration sites, while those in post-restoration sites had significantly higher K values than mainstem sites. Post hoc analyses for Winter run Chinook Salmon did not detect significant differences in K between any of the site classifications. Spring run Chinook Salmon had significantly higher K values in control side channels than in post restoration side channels.

***Table 14. Analysis of variance table showing results of fork length models**. Run was classified using the Central Valley length-to-date chart. Details of the models used in these analyses are provided in the methods.*

Run	**Site Classification**	**Month**	**Year**	**River Mile**
Fall run Chinook (n=3029)	$\mathbf{F_{3,3023} = 75.493}$ **p < 0.001**	$F_{1,3023} = 14.107$ p = 0.274	$\mathbf{F_{1,3023} = 1.197}$ **p < 0.001**	$\mathbf{F_{1,3023} = 28.449}$ **P < 0.001**
Late-fall run Chinook (n=295)	$\mathbf{F_{3,290} = 2.865}$ **p = 0.037**	$\mathbf{F_{1,290} = 1059.35}$ **p < 0.001**	N/A	$F_{1,290} = 0.309$ p = 0.579
Winter run Chinook (n=240)	$\mathbf{F_{2,235} = 20.422}$ **p < 0.001**	$\mathbf{F_{1,235} = 88.867}$ **p < 0.001**	$F_{1,235} = 3.113$ p = 0.079	$\mathbf{F_{1,235} = 12.730}$ **p < 0.001**
Spring run Chinook (n=174)	$\mathbf{F_{2,169} = 11.569}$ **p < 0.001**	$\mathbf{F_{1,169} = 143.122}$ **p < 0.001**	$F_{1,169} = 0.307$ p = 0.581	$F_{1,169} = 0.474$ p = 0.492
Steelhead / Rainbow Trout (n=66)	$\mathbf{F_{3,60} = 10.233}$ **p < 0.001**	$F_{1,60} = 1.599$ p = 0.211	$F_{1,60} = 2.789$ p = 0.100	$F_{1,60} = 0.210$ p = 0.649

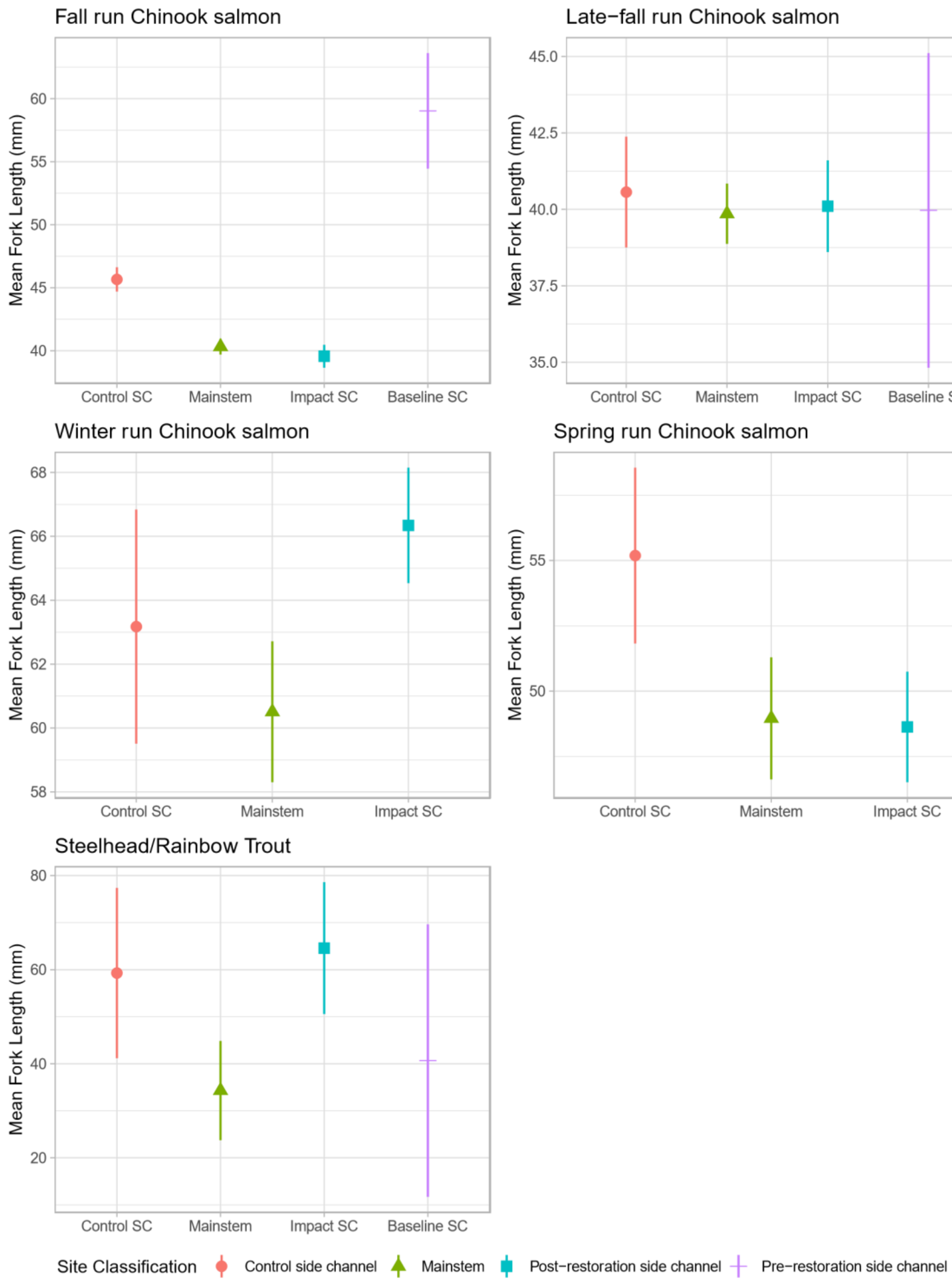

Figure 15. Estimated marginal means of fork length from seined fish captured between December 2019 and March 2020. *Run was classified using the Central Valley length-to-date chart.*

Table 15. Post-hoc comparisons of different site classifications for fork-length models. *Positive ratios indicate that the first site classification listed had larger fork lengths. Negative ratios indicate the second site classification listed had larger fork lengths. Significance of these trends are indicated by the p-value. Run was classified using the Central Valley length-to-date chart. Details of the models used in these analyses are provided in the methods.*

Run & Contrast	**Difference**	**SE**	**df**	**t-ratio**	**p-value**
Fall run Chinook (n=3029)					
Control SC - Mainstem	**5.338**	**0.467**	**3023**	**11.423**	**<0.001**
Control SC - Impact SC	**6.097**	**0.550**	**3023**	**11.087**	**<0.001**
Control SC - Baseline SC	**-13.377**	**1.888**	**3023**	**-7.084**	**<0.001**
Mainstem - Impact SC	0.759	0.438	3023	1.731	0.3075
Mainstem - Baseline SC	**-18.715**	**1.858**	**3023**	**-10.075**	**<0.001**
Impact SC - Baseline SC	**-19.474**	**1.873**	**3023**	**-10.399**	**<0.001**
Late-fall run Chinook (n=295)					
Control SC - Mainstem	0.709	0.829	290	0.855	0.8279
Control SC - Impact SC	0.461	0.930	290	0.496	0.9599
Control SC - Baseline SC	0.598	2.176	290	0.275	0.9927
Mainstem - Impact SC	-0.247	0.727	290	-0.340	0.9864
Mainstem - Baseline SC	-0.110	2.091	290	-0.053	0.9999
Impact SC - Baseline SC	0.137	2.138	290	0.064	0.9999
Winter run Chinook (n=240)					
Control SC - Mainstem	2.67	1.75	235	1.520	0.2835
Control SC - Impact SC	-3.17	1.91	235	-1.657	0.2239
Control SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Mainstem - Impact SC	**-5.83**	**1.21**	**235**	**-4.841**	**<0.001**
Mainstem - Baseline SC	N/A	N/A	N/A	N/A	N/A
Impact SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Spring run Chinook (n=174)					
Control SC - Mainstem	**6.234**	**1.75**	**169**	**3.555**	**0.0014**
Control SC - Impact SC	**6.559**	**1.72**	**169**	**3.818**	**<0.001**
Control SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Mainstem - Impact SC	0.325	1.33	169	0.244	0.9676
Mainstem - Baseline SC	N/A	N/A	N/A	N/A	N/A
Impact SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Steelhead / Rainbow Trout (n=66)					
Control SC - Mainstem	**24.98**	**8.09**	**60**	**3.085**	**0.0158**
Control SC - Impact SC	-5.30	9.46	60	-0.561	0.9433
Control SC - Baseline SC	18.60	13.39	60	1.388	0.5114
Mainstem - Impact SC	**-30.28**	**6.99**	**60**	**-4.332**	**<0.001**
Mainstem - Baseline SC	-6.38	12.03	60	-0.530	0.9514
Impact SC - Baseline SC	23.90	12.45	60	1.920	0.2306

Table 16. Analysis of variance table showing results of Fulton's Condition Factor (K) models. *Run was classified using the Central Valley length-to-date chart. Details of the models used in these analyses are provided in the methods.*

Run	**Site Classification**	**Month**	**Year**	**River Mile**
Fall run Chinook (n=1133)	**$F_{3,1127} = 23.131$ $p < 0.001$**	**$F_{1,1127} = 27.181$ $p < 0.001$**	**$F_{1,1127} = 27.744$ $p < 0.001$**	$F_{1,1127} = 0.013$ $P < 0.911$
Late-fall run Chinook (n=130)	**$F_{3,125} = 19.820$ $p < 0.001$**	$F_{1,125} = 0.0138$ $p = 0.907$	N/A	$F_{1,125} = 0.301$ $p = 0.584$
Winter run Chinook (n=120)	**$F_{2,115} = 5.860$ $p < 0.001$**	**$F_{1,115} = 16.060$ $p < 0.001$**	$F_{1,115} = 0.412$ $p = 0.522$	$F_{1,115} = 0.252$ $p = 0.617$
Spring run Chinook (n=62)	**$F_{2,57} = 4.004$ $p = 0.024$**	**$F_{1,57} = 8.756$ $p = 0.004$**	$F_{1,57} = 0.296$ $p = 0.588$	**$F_{1,57} = 7.899$ $p = 0.007$**
Steelhead / Rainbow Trout (n=15)	$F_{3,10} = 0.7332$ $p = 0.5556$	$F_{1,10} = 0.4782$ $p = 0.5050$	N/A	$F_{1,10} = 0.2269$ $p = 0.6441$

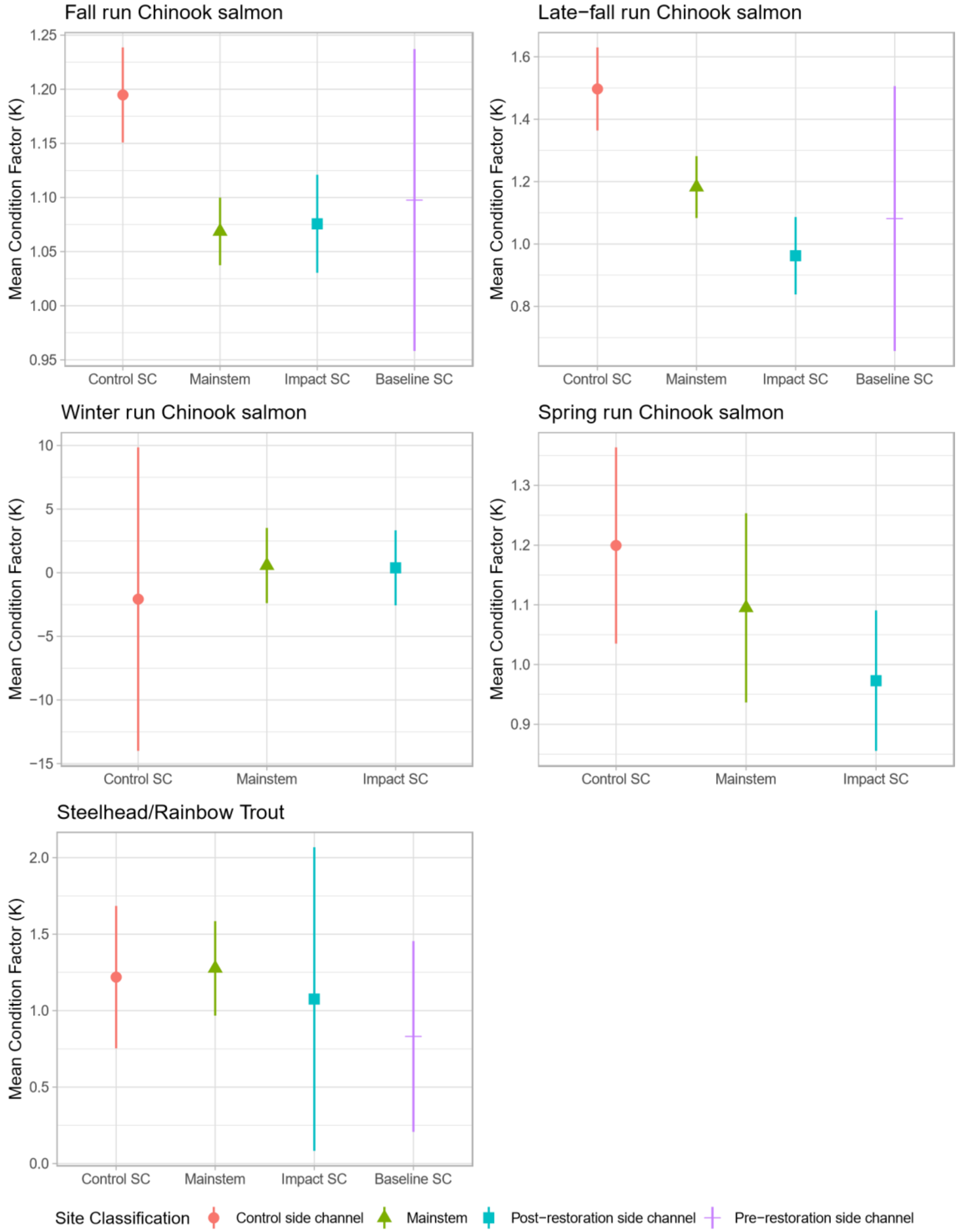

Figure 16. Estimated marginal means of Fulton's Condition Factor (K) from seined fish captured between December 2019 and March 2020. *Run was classified using the Central Valley length-to-date chart. Note that while K cannot be negative, but the estimated marginal mean of K may appear negative when model fit is poor.*

Table 17. Post-hoc comparisons of different site classifications for Fulton's Condition Factor (K) models. *Positive differences indicate that the first site classification listed had larger fork lengths. Negative differences indicate the second site classification listed had larger fork lengths. Significance of these trends are indicated by the p-value. Run was classified using the Central Valley length-to-date chart. Details of models used in these analyses are provided in the methods.*

Run & Contrast	Difference	SE	df	t-ratio	p-value
Fall run Chinook (n=3029)					
Control SC - Mainstem	**0.126**	**0.022**	**1127**	**5.692**	**<0.001**
Control SC - Impact SC	**0.119**	**0.027**	**1127**	**4.476**	**<0.001**
Control SC - Baseline SC	0.097	0.058	1127	1.666	0.3423
Mainstem - Impact SC	-0.007	0.027	1127	-0.324	0.9883
Mainstem - Baseline SC	-0.029	0.057	1127	-0.507	0.9575
Impact SC - Baseline SC	-0.022	0.059	1127	-0.370	0.9827
Late-fall run Chinook (n=295)					
Control SC - Mainstem	**0.315**	**0.067**	**125**	**4.734**	**<0.001**
Control SC - Impact SC	**0.535**	**0.072**	**125**	**7.476**	**<0.001**
Control SC - Baseline SC	0.416	0.176	125	2.364	0.0895
Mainstem - Impact SC	**0.220**	**0.065**	**125**	**3.386**	**0.0052**
Mainstem - Baseline SC	0.101	0.172	125	0.586	0.9361
Impact SC - Baseline SC	-0.119	0.176	125	-0.678	0.9053
Winter run Chinook (n=240)					
Control SC - Mainstem	-2.644	3.712	115	-0.712	0.757
Control SC - Impact SC	-2.456	3.711	115	-0.662	0.786
Control SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Mainstem - Impact SC	0.185	0.082	115	2.263	0.065
Mainstem - Baseline SC	N/A	N/A	N/A	N/A	N/A
Impact SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Spring run Chinook (n=174)					
Control SC - Mainstem	0.105	0.099	57	1.059	0.5435
Control SC - Impact SC	**0.227**	**0.081**	**57**	**1.792**	**0.0192**
Control SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Mainstem - Impact SC	0.122	0.079	57	1.552	0.2749
Mainstem - Baseline SC	N/A	N/A	N/A	N/A	N/A
Impact SC - Baseline SC	N/A	N/A	N/A	N/A	N/A
Steelhead / Rainbow Trout (n=66)					
Control SC - Mainstem	-0.0573	0.281	10	-0.203	0.9968
Control SC - Impact SC	0.3881	0.408	10	0.950	0.7795
Control SC - Baseline SC	0.1433	0.576	10	0.249	0.9943
Mainstem - Impact SC	0.4454	0.288	10	1.548	0.4471
Mainstem - Baseline SC	0.2006	0.433	10	0.463	0.9654
Impact SC - Baseline SC	-0.2448	0.441	10	-0.555	0.9431

MACROINVERTEBRATE SAMPLING

Field Methods to Collect Macroinvertebrates

Macroinvertebrates were collected via drift samples at control side channels, baseline side channels, impact side channels, and mainstem sites near each side channel that we sampled. This resulted in samples from 3 control side channels (Bourbon Island, Clear Creek, and Wyndham), one baseline side channel (Shea Island), six restored side channels (Anderson River Park, Kapusta, Lake California, North Cypress, Reading Island, and Rio Vista), and 10 mainstem sites (one near each side channel sampled). Sampling methods were adapted from the Columbia Habitat Monitoring Program (CHaMP, 2015).

Two drift nets (20x40 cm with 500μm mesh) were deployed perpendicular to flow. Net placement was chosen to meet the following criteria when possible 1) center of the side channel, 2) near the downstream end of a riffle habitat, 3) depth between 15-30 cm, and 4) velocities between 0.3 and 0.6 m/s. Nets were anchored with rebar and suspended 2cm off the stream bed using spacers. The top of the net extended above the water's surface in order to capture surface drift. Nets were deployed for 3 hours during midday. At the end of each sampling period, the drift sample was collected and transferred to jars with 95% ethanol in a 1:1 ratio to sample size volume. Samples were delivered to the CDFW Aquatic Bioassessment Lab (ABL) for taxonomic identification and enumeration. Unfortunately, the amount of ethanol was not sufficient in many jars to prevent deterioration of the samples, making mass measurements unreliable. Because of this, mass is not reported below.

Data analyses

The CDFW Aquatic Bioassessment Lab identified each individual to the lowest taxonomic level possible given the sample condition and taxonomist knowledge.

We first filtered the dataset to include only aquatic insects, and calculated metrics related to the proportions of sensitive EPT (Emphemeroptera, Plecoptera, and Trichoptera) taxa for each site type (control side channels, baseline side channels, restored side channels, and mainstem sites). The EPT Individual Index is the percent of individuals found from EPT taxa, relative to the total number of aquatic insects sampled. The EPT family index is the percent of unique families within EPT taxa, relative to the total number of aquatic insect families found. Higher EPT metrics are generally correlated with better water quality, though note that the metrics will be most comparable when comparing habitats of the same type (e.g. side channel to side channel).

We then used the whole dataset to look at a number of diversity metrics. We chose to group orders together for analyses, with the exception of a few taxa that were not identified down to order. This was primarily the case for non-insect invertebrates such as spiders, snails, and hydrozoans. In these cases, we used the lowest taxonomic level available for analysis.

For each site type, we looked at the data for all sampling events, and for sampling events that fell within each quarter of the year (January-March, April-June, and July-September). We calculated the percentage of individuals present from each taxa identified, and the mean number of individuals (of any taxonomic group) captured per sampling day. The vegan package (Oksanen *et al.*, 2020) in R was used to calculate two diversity metrics – the Shannon and Simpson Indices (Hill, 1973).

$$\text{Shannon Index (H)} = -\sum_{i=1}^{s} p_i \ln p_i$$

$$\text{Simpson Index (D)} = 1 - \sum_{i=1}^{s} p_i^2$$

In these equations, *p* is the proportion of individuals of taxonomic group *i* and *s* is the number of taxonomic groups. We also used the vegan package (version 2.5-7) in R to count the total number of taxa observed at each site type, and to calculate the corresponding rarified number of taxa. Because number of taxa is a function of sampling effort, rarefaction can provide an estimate of taxa number when sample sizes are small or uneven (Hurlbert, 1971; Heck *et al.*, 1975). Finally, we calculated an average number of macroinvertebrates per sampling day by dividing the total number of individuals collected by the number of sampling days. We were unable to compare macroinvertebrate biomass between sites because deterioration of samples prevented us from taking accurate weights.

Results

EPT metrics are shown in table 19. Family richness is reported for reference, but note that this number is not adjusted for sampling effort. Mainstem sites had the highest numbers for all metrics examined. Within side channel sites, control and restored side channels had similar EPT indices. Baseline side channels, which had substantially fewer samples, had the lowest EPT indices. Taxonomic proportions are shown in figure 17. Diptera made up the highest proportion of sampled invertebrates for all site classifications. Trichoptera were the second most abundant taxon in mainstem sites (13%), but were rarely found in any of the side channel types (<1%). Ephemeroptera were found in similar proportions at all sites. Table 19 shows diversity and richness indices, as well as the average number of individuals sampled per day. Baseline sites also showed the lowest overall macroinvertebrate diversity, taxonomic richness, and had a lower rate of individuals captured over time.

Table 18. Three EPT (Ephemeroptera, Plecoptera, and Trichoptera) indices. *EPT family richness is the number of unique EPT families found in each habitat type. Note that this metric is not adjusted for sampling effort. EPT Individual Index is the percent of individuals from EPT taxa relative to the total number of aquatic insect individuals in the sample. EPT Family Index is the percent of unique EPT families, relative to the total number of unique aquatic insect families found in the sample. Sample sizes refer to the number of drift samples taken in each site classification.*

Site Classification	**EPT Family Richness**	**EPT Individual Index**	**EPT Family Index**
Baseline Side Channel (n=2)	4	18.12%	33.33%
Mainstem (n=19)	14	44.07%	47.62%
Control Side Channel (n=6)	10	34.41%	42.42%
Restored Side Channel (n=22)	13	35.10%	40.63%

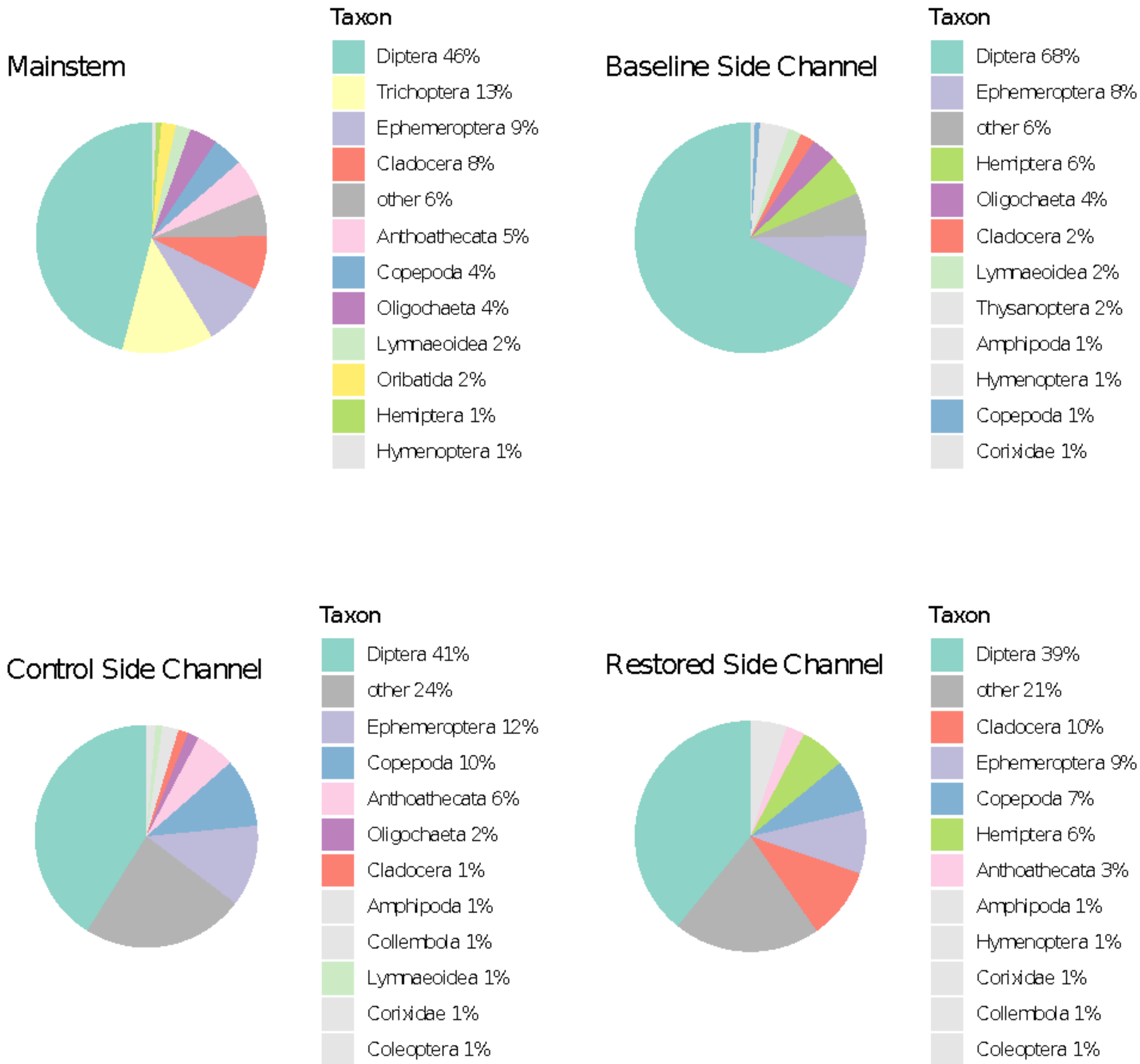

Figure 17. Distribution of the top 12 taxa identified at each site type. *Colors are assigned based on the full dataset with all sites pooled. If a taxa represented more than 1% in the full dataset, then it was assigned a unique color. Taxa that represented 1% or less of the full dataset are represented in light gray. Note that due to variation between site types, some taxa may represent less than 1% at a given site type, but still be assigned a unique color since they represent more than 1% in the pooled dataset.*

Table 19. Diversity, Richness, and Abundance metrics for macroinvertebrates at each site classification. *Calculations were made for the entire data set, and for each quarter of the year that samples were collected. Sample sizes refer to the number of samples taken in each site classification. See text for a description of each metric.*

Site Classification	**Simpson Index**	**Shannon Index**	**Number of Taxa**	**Rarefied Number of Taxa**	**# Individuals Sampled/ Sampling Days**
Baseline Side Channel (n=2)	0.5289	1.4170	23	23.00	356.5
Jan-March (n=0)	*N/A*	*N/A*	*N/A*	*N/A*	*N/A*
April-June (n=1)	*0.500*	*1.201*	*17*	*17.00*	*140*
July-Sept (n=1)	*0.531*	*1.208*	*12*	*12.00*	*573*
Mainstem (n=19)	0.7511	1.9539	29	24.12	595.5
Jan-March (n=9)	*0.692*	*0.692*	*13*	*12.33*	*470.5*
April-June (n=9)	*0.735*	*1.732*	*23*	*16.17*	*582.7*
July-Sept (n=1)	*0.5877*	*1.179*	*9*	*8.88*	*599.0*
Control Side Channel (n=6)	0.7883	2.0757	33	23.51	548.0
Jan-March (n=2)	*0.800*	*1.746*	*14*	*14.00*	*534.1*
April-June (n=3)	*0.723*	*1.713*	*18*	*14.59*	*655.3*
July-Sept (n=1)	*0.547*	*1.133*	*11*	*10.87*	*609.0*
Restored Side Channel (n=22)	0.8088	2.2206	26	26.96	430.0
Jan-March (n=9)	*0.720*	*1.614*	*17*	*15.09*	*484.8*
April-June (n=9)	*0.757*	*1.846*	*21*	*16.48*	*375.0*
July-Sept (n=4)	*0.792*	*1.796*	*18*	*13.53*	*430.3*

DISCUSSION

The goal of this monitoring report was to examine the effects of restoration on juvenile salmonids in the upper Sacramento River by focusing on objectives 2-5 of the Upper Sacramento River Anadromous Fish Habitat Restoration Project: increasing the areal extent of rearing habitat meeting juvenile salmonid rearing habitat suitability criteria; increasing salmonid juvenile abundance/density at restoration sites after implementation, as compared to before implementation; improving size and average condition of salmonids using the side channels, as compared to those that have not been documented using the side channels; and increasing available macroinvertebrate prey abundance. To that end, we examined the effect of restoration on juvenile salmonid abundance; habitat suitability and area, size and condition; and macroinvertebrate food availability. We also refine what habitat characteristics are preferred by juvenile salmonids within the upper Sacramento River system.

We were able to conduct BACI (before-after-control-impact) analyses using fish abundance data from three restored side channels (Anderson River Park, Lake California, and Rio Vista) and three control sites (Bourbon Island, Mainstem North, Mainstem South). This design is ideal because it controls for differences between sites at the start of the project, as well as temporal variation in salmonid numbers (e.g. seasonal or annual variation). The impact of restoration differed when looking at two metrics – fish counts and fish densities (Figures 3 and 4, Tables 3 and 4). Juvenile salmonid count models showed that restoration significantly increased total juvenile salmonid number. When broken down by run, these results were statistically significant for fall-run Chinook Salmon and steelhead/Rainbow Trout. Late-fall and winter run Chinook Salmon showed similar, non-significant trends. There was not adequate data to examine the effect on spring run Chinook Salmon. The greater numbers of salmon observed in restored sites vs control sites following the restoration could be the result of an increase in salmonid carrying capacity (Lepori *et al.*, 2005). Before the restoration, many of the impacted sites were either disconnected during low flows, or partially connected backwater habitats, resulting in uninhabitable areas. The increase in accessible habitat that restorations provided could enable the side channel to hold greater numbers of fish. Juvenile salmonid density models did not detect a significant impact of restoration on fish-per-acre. This was true for all salmonids, fall-run Chinook, late-fall run Chinook, and steelhead/Rainbow Trout. There was not adequate data to examine spring run Chinook Salmon, and data from winter run Chinook Salmon did not meet our robustness requirements (e.g. similar models produced conflicting results).

The differing results between the two abundance metrics could potentially be explained by the increase in available habitat that was created by restoration. The number of juvenile salmonids recorded in each channel pre-restoration was lower, and the area of habitat sampled was smaller; following the restoration, there were greater numbers of fish in a larger habitat, leading to similar densities of fish. A review of several North American river restoration techniques by Roni *et al.(2008)* suggests that following a restoration, fish density can be increased by improving habitat quality and by creating habitats that increase survivability with abundant cover and food for the target species. Our results show that the average density of fish was not statistically different in restored and control sites following a restoration, suggesting that the quality of habitat within the restored habitat is similar to control sites.

We also analyzed fish abundance in the full suite of sites, including project sites that do not have data from before and after restoration. The lack of data taken before restoration makes it more challenging to make decisive conclusions, in part because of reduced power to detect differences between treatments. Fish counts, in particular, are difficult to analyze and interpret without adequate data taken before restoration for comparison, so our dependent variable in these analyses is estimated fish density (fish-per-acre). Steelhead/Rainbow Trout had significantly more fish-per-acre in impact (post-restoration) sites than control sites and showed a near significant trend ($p = 0.051$) of impact sites having higher densities than baseline (pre-restoration) sites (Figure 5, Tables 5 and 6). These results are not surprising, given that the BACI models detected the major impact of restoration to be on fish count, rather than fish density. The creation of additional habitat in a side channel may increase the number of fish, even if there is no measurable change in fish density.

The results of the fish abundance surveys indicate that restoration is having an overall positive effect on fish abundance, though the extent of the benefit varies depending on which metric and runs are examined. The primary benefit appears to be through the production of additional habitat that supports similar densities of fish. In the Upper Sacramento River, juvenile salmon are severely habitat-limited, with only 26 of the estimated 331 acres of habitat needed to be selfsustaining (Gill, n.d.). Habitat suitability criteria can give insight into how habitat availability in restored sites compares to control and baseline sites, and can provide a rough estimate of the amount of habitat gained through restoration.

We examined habitat suitability first by comparing the proportion of each site that met the suitability criteria, and then by estimating the amount of habitat in acres that was gained through restoration. Consistent with results from previous annual reports, restored and control side channels had similar proportions of high-quality habitat for every criterion we examined, suggesting that the restoration has successfully recreated the depth, velocity, and cover characteristics of historical side channels (Figure 8 and Table 9). Flow from Keswick Dam was also included in this model, and had a significant negative effect on all criteria whose estimations included velocity as a component: suitable habitat, optimal habitat, suitable + optimal habitat, and suitable depth & velocity. Suitable cover did not show a significant relationship with flow from Keswick at our study sites. Though we had data from four baseline/unrestored sites, we chose not to present them in this analysis. This is because these sites showed an artificially high proportion of suitable habitat, which was likely an artifact of how the habitat suitability criteria used from Goodman *et al.* were created. Their criteria focused on depth, velocity, and cover and did not include backwater habitats or disconnected side channels. Disconnected pre-restoration sites can sometimes show large proportions of suitable habitat due to artifacts of classification. For example, a small backwater habitat with near-zero velocity will appear to have a high proportion of suitable habitat. Additionally, temperature and oxygen availability (which were not included in the criteria designed by Goodman and colleagues) are more likely to be outside of tolerable ranges in backwater and disconnected sites. Because of this, the approach used in Goodman et al. may not be appropriate for examining habitat availability in baseline sites; baseline and impact data within a site are best compared using absolute values of habitat availability at similar flows, particularly when the pre-restoration side channel was not connected to the mainstem river on both ends at all flows.

Temperature, oxygen levels, and access to habitat should also be incorporated into suitability criteria when comparing baseline sites with impact sites. Including distance to spawning grounds may also provide information on how accessible the sites are to juvenile salmon. Additionally, the field crew noted that few fish observations were made in zero velocity water (microhabitat use data shows ~ 5% of fish). Removing zero velocity water from the suitability criteria may allow more accurate comparisons between baseline and restored conditions.

We looked at the aquatic acreage gained by restoration in two ways. Rio Vista, Reading Island, and Anderson River Park (Phase 1) were each mapped before and after restoration in flows that ranged from 3,100 to 5,000 cfs. Comparing the acreage mapped before and after restoration can provide a very rough estimate of the acreage gained due to restoration (Figure 9). As previously mentioned, the criteria from Goodman *et al.* (2015) likely overestimate the amount of suitable and optimal habitat in baseline sites, meaning that the estimates of acreage gained from these sites is almost certainly an underestimate of the actual amount of habitat gained. Using this conservative approach, the habitat gained from restoration in these three sites at the aforementioned flows created 0.88 acres of habitat that was suitable, optimal, or both. However, initial site visits from crew members prior to restoration found prior to restoration, these sites were warmer backwater habitat or disconnected pools which contained no viable habitat. Using this information, we can use a less conservative approach to calculate the habitat gained. If we assume backwater and disconnected sites contain no viable habitat, then the habitat gained from restoration is equal to the amount of suitable and/or optimal habitat available after restoration: 4.52 acres. These numbers may overestimate the habitat created by restoration if some small amounts of habitat were available prior to restoration.

Baseline data were not collected at the remaining mapped sites (Lake California, North Cypress, Painter's Rifle, and Kapusta) because they were dry, disconnected, or otherwise assumed to be uninhabitable. Each of these sites were mapped at multiple flows after restoration, and the data was split into three groups for analysis using the less conservative approach described above: Low flows ranged from 3,250-3,700 cfs, medium flows ranged from 5,000-7,800 cfs, and high flows range from 8,000-11,000 cfs. Sites were mapped at each flow regime with the exception of Painter's Riffle, which was not mapped at high flows (Figure 10). The amount of suitable, optimal, or suitable + optimal habitat measured at these sites was 6.86 acres at low flows (all four sites), 6.00 acres at medium flows (all four sites), and 4.05 acres at high flows (three sites, excluding Painter's Riffle). As described above, these numbers may slightly overestimate the habitat created by restoration if some small amounts of habitat were available prior to restoration.

Habitat suitability criteria (HSC) used for evaluation should have direct links to microhabitat-use of the populations of interest (Goodman *et al.*, 2015). The amount of suitable and optimal habitat described above was determined using habitat suitability criteria originally created from microhabitat-use observations on the Trinity River, and supported by microhabitat-use observations of the entire upper Sacramento River (USFWS, 2005; Goodman *et al.*, 2015). We were particularly interested in habitat-use within the side channels of the river to reflect the focus of restoration efforts. Because of this, we created study HSC snorkel surveys in the side channels of our study area. Much like the mapping HSC derived from Goodman *et al.* (2015), study HSC represent 75% of observations at the 95% confidence level. Study HSC are not directly relatable to the mapping HSC due to a lack of habitat availability data from our sites, but study HSC can be a good indicator as to whether mapping HSC overestimates or underestimates habitat for the populations of interest. Generalizing study HSC for all salmonid observations in our study, distance to cover appears to be very similarly represented, but exhibit a notably narrower range of depth and velocities than our mapping criteria (Table 10, Figures 11-13). This indicates that our estimates of habitat may be slightly overestimating habitat in respect to suitable depth and velocity. In comparing mapping HSC to our study HSC of all fry in our study, we can see that cap of suitable ranges for Sacramento River populations are 39.5-71% of mapping HSC (Table 10). This indicates that mapping HSC are likely overestimating fry habitat. This is not unexpected, because the mapping criteria we used were developed for juveniles >50mm in size. In examining our study HSC for all juveniles in our study, we can see a similar range of velocities and distance to cover, and a narrower range of depth compared to mapping HSC (Table 10). This indicates that we may be overestimating the amount of habitat with suitable depth, but closely representing suitable velocity and distance to cover. While mapping HSC might be slightly over or underestimating available habitat, it is important to point out that mapping criteria is still adequate for making comparisons between restored and unrestored habitat or among flows because the trends in mapping HSC should track those of the available habitat. In order to inform future restoration designs, discrete ranges of suitable habitat for each respective run, species, and life stage of juvenile salmonid in the area has been provided (Table 10). Within-study comparisons can be made between Chinook and steelhead/Rainbow Trout so long as variability between sample sizes is accounted for. Another caveat that must be taken into consideration is that this study does not take into account habitat availability. This means some variation between species, run, and life stages could be biased by habitat available at time of observation.

Microhabitat surveys also provided information on preference for different cover types (Figure 14). Unlike the habitat suitability curves described above, this dataset took cover availability in order to generate a preference score. Fry and juveniles of both species showed similar preference trends, but significance varied slightly between groups. Fry and juveniles from both Chinook Salmon and steelhead/Rainbow Trout showed a distinct preference for fine woody debris over all other cover types besides undercut banks. Chinook Salmon fry and juveniles significantly preferred undercut banks to overhead cover (Table 12). Fry of both species significantly preferred undercut banks to branches and small woody debris (Tables 13 and 14). This information can inform future restorations, and suggests that similar cover criteria can be applied to salmonid juveniles and fry of both species.

Fish size and condition collected via seining did not yield consistent results between runs (Figures 15 and 16, Tables 15-18). Fish from restored side channels had significantly larger fork sizes for some runs (e.g. winter run Chinook Salmon and steelhead/Rainbow Trout had significant longer fork lengths in restored side channels as compared to the mainstem), but other runs showed the opposite relationship. While this may indicate run-specific benefits of restoration on growth, there are factors that make conclusive interpretation difficult. First, fish are mobile, so the location the fish seined may not be representative of where they spend the majority of their time. Second, runs were classified using the Central Valley Length-to-Date chart (Appendix B), not genetic analyses, meaning that we analyzed the size within groups that were classified by size. Mistakes in classification could therefore make it difficult to accurately detect trends. Finally, it is possible that fish are choosing habitat based on their size, rather than a particular habitat producing fish of a certain size. In this case, we may see significant correlations between size and site type that are not indicative of the growth potential within a site type. We previously attempted to disentangle these issues by rearing hatchery fish in netted enclosures that were anchored in each habitat type, so that we could ensure that growth was a product of the environment. Data in the pilot enclosure study (presented in a previous annual report) was confounded due to high mortality in mainstem sites, leading to reduced densities and higher growth potential in those enclosures. We subsequently modified the study design to address these issues and redeployed the enclosures. Unfortunately, enclosure deployment occurred in early March 2020, and we were forced to terminate the experiment shortly after deployment due to COVID-19. Because of this, we are unable to report on this work.

Macroinvertebrate monitoring provides some information on taxonomic diversity within macroinvertebrates, but unfortunately due to sample deterioration, we were unable to determine macroinvertebrate biomass at each site type. Biomass would provide a stronger indicator of food availability for juvenile salmonids. We looked at three EPT (Ephemeroptera, Plecoptera, and Trichoptera) metrics. These three orders can provide an indicator of stream water quality (Barbour *et al.*, 1992; Lenat and Penrose, 1996; Wallace *et al.*, 1996). Baseline sites had the lowest values for all three metrics, mainstem sites had the highest values, and control and restoration side channels performed similarly (Table 18). Baseline sites also showed the lowest overall macroinvertebrate diversity, taxonomic richness, and had a lower rate of individuals captured over time (Table 19). The most dominant taxon at all sites was Diptera; it was found in the highest proportions in baseline sites (68%), followed by mainstem sites (46%), control side channels (41%), and (39%). Determining what these values mean for salmonid growth is challenging. The lower rate of individuals captured in baseline sites relative to restored sites could indicate that restoration was successful in creating more macroinvertebrate prey, but this can't be confirmed without information on biomass and salmonid diet. Jeffres *et al.* (2008) examined salmonid growth and diet in floodplain and river locations, and found that fish reared in river sites had lower growth rates and higher proportions of Diptera in their gut contents than those from floodplain sites. While this may indicate that the high proportion of Diptera found in baseline sites indicates it provides a lower quality food source, there could also be other factors in their study unrelated to taxonomic classification that differed between sites (e.g. food abundance) that influenced growth.

The datasets used in the analyses reported above vary in quality and size. Results obtained from the highest quality datasets all suggest that the Upper Sacramento River Anadromous Fish Habitat Restoration Project has effectively produced additional high quality juvenile salmonid habitat (objective 2) that supports higher numbers of fish (objective 3) in the upper Sacramento River. Through the monitoring efforts, we were also able to provide refined habitat suitability criteria and cover preferences for juvenile salmonids found in side channels in the upper Sacramento River, which can be used to inform future restoration. However, some metrics need additional data collection in order to draw definitive conclusions. For future restorations, we emphasize the need for data collection before restoration occurs, in order to increase our ability to detect the effects of restoration. The effects of restoration on fish size and condition (objective 4) varied between runs when looking at seining data. The seining data was likely confounded by several other factors, and data collection of enclosure study growth rates were unfortunately not completed due to COVID-19 shut downs. The higher number of macroinvertebrates (determined by sampling rate) observed in restored side channels as compared to baseline channels suggests that there may be a positive effect of restoration on food availability (objective 5), but without biomass and diet information, firm conclusions can't be drawn. Addressing the logistical challenges of collecting data for objectives 4 and 5 can help paint a clearer picture of how side channel restoration affects salmonid growth. Continued monitoring of completed and future restorations will provide additional insight into the effectiveness of side channel restoration, as well as information about how side channel characteristics evolve over time.

RECOMMENDATIONS FOR FUTURE WORK

- Incorporate preferred cover types in channel design: fine woody debris (<1" diameter) and undercut banks. Branches and small woody debris (1" up to 12" diameter) were the least preferred, so this cover type should be deprioritized.

- Use the non-parametric tolerance limits presented in this report to help inform channel design in terms of depth, velocity, and cover.

- The data presented in this monitoring report requires significant time, effort, and resources. In order to increase cost effectiveness, future monitoring could be refocused to collect data that provides the most information relative to effort. Below is a brief summary of our recommendations. Further details on the justification, pros, and, cons of these suggestions can be found in the *Recommended Monitoring Plan Revisions for 2022-2026* (Tussing and Banet, 2022)

    - <u>Timing of Biological Sampling:</u> Current biological monitoring is implemented in all months across all baseflows. We recommend focusing on salmon stocks of most interest when they are at higher abundances. This includes the 8-month period from September through April, as well as a sampling event in July at the peak of steelhead/Rainbow Trout use. This would capture the majority of the data, and reduce the number of zero observation snorkel surveys.

    - <u>Timing of Physical Habitat Sampling:</u> Physical habitat monitoring currently targets a range of three baseflows to capture habitat at high summer baseflows, mid-baseflows in the spring/fall, and low winter baseflows. We suggest focusing physical habitat monitoring components on the 6-month period from November through April, which coincides with low winter baseflow releases from Keswick. Historically, we have been unable to capture mid-baseflows at all sites due to their fleeting nature, and high baseflows coincide primarily with steelhead/Rainbow Trout habitat.

    - <u>Duration of Monitoring:</u> As many analyses rely on a BACI design, we recommend data collection one year before and one year after restoration.

However, the one year of post-project monitoring of salmonid juveniles and their habitats would begin the fall after construction and extend for 8 months (Sept-Apr). The full window of before/after monitoring efforts would span 2.5 years. BACI analyses with approximately one year before/after index data have been successful at demonstrating effectiveness for pooled analyses of all salmonids, as well as fall run Chinook. Similar trends have been found for less abundant runs. This approach also provides an opportunity to discontinue monitoring of some sites as new restoration projects are implemented.

- Snorkel Index Methods: We recommend foregoing bi-weekly snorkel index surveys and relying on Microhabitat Use surveys to generate an index of abundance. Current side channel monitoring efforts employ two snorkel-based surveys: 1) A rapid Snorkel Index which enumerates salmonids by run and is focused solely on the channel margin proceeding in a downstream direction, and 2) An intensive Microhabitat Use Survey in the upstream direction, which enumerates salmonids by size class, covers the entire channel area, and captures precise fish locations and related habitat attributes. Though microhabitat surveys are more intensive and thus conducted less frequently, they can provide a representation of fish observed per unit area of the entire channel rather than just the channel margin. They also remove the need to adjust fish/area metrics based upon visibility, provide a better chance at detecting fish at low densities (meaning less zeros in the data set), are more consistent with generally accepted snorkel survey methods, and provide cost savings.

REFERENCES

Banet AI, Tussing S, Doolittle G (2020) Salmonid Monitoring of Habitat Restoration Sites in the Upper Sacramento River in 2018-2019 (Annual Report).

Banet AI, Tussing S, Memeo M (2018) Salmonid Monitoring of Habitat Restoration Sites in the Upper Sacramento River in 2017-2018 (Annual Report).

Banet AI, Tussing S, Roualdes EA, Doolittle G, Nielsen DS (2021) Salmonid Monitoring of Habitat Restoration Sites in the Upper Sacramento River in 2019-2020 (Annual Report).

Barbour M, Graves C, Plafkin J, Wisseman R, Bradley B (1992) Evaluation of EPA's Rapid Bioassessment Benthic Metrics: Metric Redundancy and Variability Among Reference Stream Sites. *Environmental Toxicology and Chemistry* 11: 437–449.

Bezanson J, Edelman A, Karpinski S, Shah VB (2017) Julia: A Fresh Approach to Numerical Computing. *SIAM Rev* 59: 65–98.

Bovee KD (1986) Development and Evaluation of Habitat Suitability Criteria for Use in the Instream Flow Incremental Methodology. National Ecology Center, Division of Wildlife and Contaminant Research, Fish and Wildlife Service, U.S. Department of the Interior.

Brooks ME, Kristensen K, Van Benthem KJ, Magnusson A, Berg CW, Nielsen A, Skaug HJ, Machler M, Bolker BM (2017) glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *The R journal*, *9*(2), 378-400.

CHaMP (2015) Scientific Protocol for Salmonid Habitat Surveys within the Columbia Habitat Monitoring Program.

Gill S (n.d.) Central Valley Project Improvement Act Science Integration Team Chinook Carrying Capacity Calculator.

Goodman DH, Som NA, Alvarez J, Martin A (2015) A mapping technique to evaluate age-0 salmon habitat response from restoration. *Restoration Ecology* 23: 179–185.

Heck KL, van Belle G, Simberloff D (1975) Explicit Calculation of the Rarefaction Diversity Measurement and the Determination of Sufficient Sample Size. *Ecology* 56: 1459–1461.

Hill MO (1973) Diversity and Evenness: A Unifying Notation and Its Consequences. *Ecology* 54: 427–432.

Holmes RA, Allen MA, Bros-Seeman S (2014) Seasonal microhabitat selectivity of juvenile steelhead in a central California coastal river 100.

Hurlbert SH (1971) The Nonconcept of Species Diversity: A Critique and Alternative Parameters. *Ecology* 52: 577–586.

Jeffres CA, Opperman JJ, Moyle PB (2008) Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook Salmon in a California river. *Environ Biol Fish* 83: 449–458.

Katz J, Moyle PB, Quiñones RM, Israel J, Purdy S (2013) Impending extinction of salmon, steelhead, and trout (Salmonidae) in California. *Environ Biol Fish* 96: 1169–1186.

Lenat D, Penrose (1996) History of the EPT Taxa Richness Metric. *Bulletin of the North American Benthological Society* 13: 305–306.

Lenth RV (2018) Emmeans: Estimated Marginal Means, Aka Least-Squares Means.

Lepori F, Palm D, Brännäs E, Malmqvist B (2005) Does Restoration of Structural Heterogeneity in Streams Enhance Fish and Macroinvertebrate Diversity? *Ecological Applications* 15: 2060–2071.

NMFS (2014) Recovery plan for the evolutionarily significant units of Sacramento River Winterrun Chinook Salmon and Central Valley Spring-run Chinook Salmon and the distinct population segment of California Central Valley steelhead.

Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, *et al.* (2020) Vegan: Community Ecology Package.

R Core Team (2016) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Ricker WE (1975) Computation and interpretation of biological statistics of fish populations. *Bull Fish Res Bd Can* 191: 1–382.

Roni P, Hanson K, Beechie T (2008) Global Review of the Physical and Biological Effectiveness of Stream Habitat Rehabilitation Techniques. *North American Journal of Fisheries Management* 28: 856–890.

Smith EP (2014) BACI Design. In: Wiley StatsRef: Statistics Reference Online. American Cancer Society.

Somerville PN (1958) Tables for Obtaining Non-parametric Tolerance Limits. *The Annals of Mathematical Statistics* 29: 599–601.

Tussing S, Banet A (2017) Upper Sacramento River Anadromous Fish Habitat Restoration Project Monitoring Plan and Protocols.

USFWS (2005) Flow-Habitat Relationships for Chinook Salmon Rearing in the Sacramento River between Keswick Dam and Battle Cree.

Wallace JB, Grubaugh JW, Whiles MR (1996) Biotic Indices and Stream Ecosystem Processes: Results from an Experimental Study. *Ecological Applications* 6: 140–151.