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Multi-Scale Action Effectiveness Research in the Lower Columbia River and Estuary, 2012

FINAL ANNUAL REPORT

Pacific Northwest National Laboratory
Oregon Department of Fish and Wildlife
National Marine Fisheries Service
University of Washington
U.S. Fish and Wildlife Service

November 2013

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FINAL ANNUAL REPORT

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Preface

Pacific Northwest National Laboratory (PNNL), the National Marine Fisheries Service (NMFS), the Oregon Department of Fish and Wildlife (ODFW), the University of Washington (UW), and the U.S. Fish and Wildlife Service (USFWS) conducted this study for the U.S. Army Corps of Engineers, Portland District (USACE). The study was coordinated regionally under the USACE's Anadromous Fish Evaluation Program, study code EST-P-11-01. The goal of the study was to evaluate the ecological benefits of restoration actions for juvenile salmon in the lower Columbia River and estuary (LCRE; rkm 0–234). The PNNL project manager was Gary Johnson. The USACE technical lead was Cynthia Studebaker. For more information about the study, please contact Cynthia Studebaker (503 808 4788).

This study originated with research funded by the Bonneville Power Administration (BPA) on juvenile salmon ecology in tidal freshwater during 2007–2010. In 2010, the project was transferred from BPA to the USACE under provisions of the Washington Memorandum of Agreement for estuary habitat restoration. The first annual report (2011) for the USACE-funded work was delivered in 2012. The present annual report is the second in the USACE series.

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Acknowledgments

This study was made possible by contributions made by many individuals to whom we extend our gratitude. Jan Salter (PNNL) managed the contracts. The U.S. Forest Service allowed property access to sites at the Sandy River delta. From PNNL, Sarah Aspens and Cynthia Wright provided data collection and management support. Susan Ennor and Kathy Neiderhiser edited and formatted the report, respectively; Shon Zimmerman, Matt Hennen, and James Hughes were instrumental in the acoustic telemetry study. We especially acknowledge the dedication and hard work put forth in the field by Elizabeth Torrey from ODFW. Funding for the study was provided by the USACE under the Congressionally-appropriated Columbia River Fish Mitigation project.

Executive Summary

The study reported herein was conducted for the U.S. Army Corps of Engineers, Portland District (USACE) by researchers at the Pacific Northwest National Laboratory (PNNL), National Marine Fisheries Service (NMFS), Oregon Department of Fish and Wildlife (ODFW), University of Washington (UW), and U.S. Fish and Wildlife Service (USFWS). The goal of the study was to evaluate the ecological benefits of restoration actions for juvenile salmon in the lower Columbia River and estuary (LCRE; rkm 0–234).

This multi-year study (2011–2018) addresses the ecological benefits of restoration actions at multiple spatial scales over time. The spatial scales include the 1) site scale as a result of an individual project, 2) landscape scale as a result of multiple restoration actions located within a ~50-km segment of the LCRE, and 3) estuary scale as a result of the cumulative effects of multiple restoration actions estuary-wide.

2012 Objectives

Objective 1, Site Scale – a) Continue pre-restoration action effectiveness research to evaluate effects of the upcoming dam removal/rechannelization at the Sandy River delta; b) continue post-restoration action effectiveness research to evaluate effects of the tide gate replacements at the Julia Butler Hansen National Wildlife Refuge (JBHNWR) mainland and Tenasillahe Island.

Objective 2, Landscape Scale – a) Estimate juvenile salmon density in shallow water habitats between St. Helens and Longview (rkm 110–141); b) estimate residence time for tagged juvenile Chinook salmon during winter 2012 in Carroll's Channel behind Cottonwood Island.

Objective 3, Estuary Scale – Prepare a compendium of tag release-recapture technologies to inform planning for future action effectiveness studies.

Summary of Findings

The following summary of findings for the 2012 Multi-Scale Action Effectiveness study is organized by the objectives. The study period is October 2011 through December 2012, unless noted otherwise.

Objective 1, Site Scale

Sandy River delta – Continue pre-restoration action effectiveness research to evaluate effects of the upcoming dam removal/rechannelization at the Sandy River delta.

- Environmental conditions during the study period entailed low flow conditions (75–125 kcfs) that persisted from late summer through fall. Peak discharge (350–450 kcfs) occurred from spring to summer months. Within a given season, variability in both water-surface elevation and water temperature was observed among the four Sandy River delta sites, especially when river discharge was low.
- Fish community composition during the study period consisted of 27 species, of which 15 were non-native fishes. In terms of total numbers of fish, catches predominantly comprised native taxa;

non-native constituents accounted for approximately 16% of the total catch. Threespine stickleback composed 90% of the total number of fish sampled.

- Three species of unmarked and marked salmon were captured: chum, coho, and Chinook. Unmarked Chinook salmon were the only species captured during every season and the most abundant salmon species captured. Seasonally, the lowest mean densities for unmarked Chinook salmon (<0.002 fish/m^2) occurred during fall; the highest densities (~0.015 fish/m^2) were observed during winter.
- The lengths of juvenile salmon captured ranged from 33 to 127 mm fork length. Seasonally, the smallest mean size of all salmonids occurred during winter (65 mm) and the largest occurred during fall 2011 (96 mm). Unmarked Chinook salmon primarily comprised fry size fish (<60 mm) during winter months, with size classes <100 mm present during spring. Overall, the sizes of unmarked Chinook salmon sampled at the Sandy River delta increased during summer and fall months.
- Stock composition estimates from the genetics analysis of 192 *unmarked* Chinook salmon sampled in the Sandy River delta showed that most fish were from the Spring Creek Group Tule Fall (39%) and the Upper Columbia Summer/Fall (33%) stock groups. Smaller proportions were estimated for the Willamette River Spring (8%), West Cascade Tributary Fall (7%), Snake River Fall (6%), West Cascade Tributary Spring (5%), Deschutes River Fall (1%), and Mid and Upper Columbia River Spring (1%) stock groups. Most *marked* Chinook salmon were from the Upper Columbia Summer/Fall (56%) and the Spring Creek Group Tule Fall (36%) stock groups.
- The diets of juvenile Chinook salmon were dominated by dipterans (primarily chironomids and ceratopogonids) and amphipods. Of these prey taxa, dipterans were most frequently consumed in large proportions, accounting for more than 20% of the diet during 77% of sampling episodes in which non-empty gut content samples were collected. Across sites, amphipods were encountered regularly in the diet, accounting for greater than 20% of consumed biomass during 27% of sampling episodes.
- Prey electivity index scores for benthic, drifting, and winged or terrestrial taxa available to juvenile salmon for consumption showed that, when present in the diet and/or environment, dipterans commonly were selected against despite constituting large proportions of the gut content biomass. Large-bodied amphipods were also selected against; however, it is possible these results may at least partially reflect a relative increase in amphipod production in the environment.
- Bioenergetics modeling to evaluate energy acquisition by juvenile salmon in shallow tidal freshwater showed that during all applicable sampling episodes, growth and gross conversion efficiency values were positive (i.e., fish gained biomass). Thus, despite certain sampling episodes when environmental conditions may constrain fish production, the current forage base and physical habitat at our sites generally appear to be suitable to support juvenile Chinook salmon.

JBHNWR (mainland and islands) − Continue post-restoration action effectiveness research to evaluate effects of the tide gate replacements at the JBHNWR and sloughs on Tenasillahe Island.

- Comparison of the presence and distribution of fish inhabiting mainland and Tenasillahe Island sloughs at JBHNWR to those observed at reference sloughs showed that 1) juvenile salmon had increased access to sloughs after installation of self-regulating tide gates at JBHNWR, and 2) juvenile salmon were captured in more treatment sloughs after self-regulating tide gates were installed than before.

- Water temperatures of sloughs at JBHNWR were similar to reference sloughs with 7-DADM exceeding 18°C in the same months and at similar cumulative days.

Objective 2, Landscape Scale

Juvenile Salmon Density – Estimate juvenile salmon density in shallow water habitats between St. Helens and Longview (rkm 110–141).

- Estimates of juvenile Chinook salmon density (mean and variance) at the landscape scale revealed that the highest densities (~0.08 fish/m^2) for unmarked Chinook salmon occurred during winter and spring 2012 and the lowest densities (<0.01 fish/m^2) occurred during fall 2011 and 2012. Except during winter 2012, the densities for unmarked Chinook salmon were lowest in the wetland habitat compared to main channel and off-channel habitats.
- The genetic stock identities for a subset of unmarked Chinook salmon sampled for landscape density indicated that most fish were estimated to be from the West Cascade Tributary Fall stock group (68%). Upper Columbia Summer/Fall fish composed an estimated 20% of the samples.

Residence Time During Winter – Estimate residence time for tagged juvenile Chinook salmon during winter 2012 in Carroll's Channel behind Cottonwood Island.

- The median residence time was approximately 17 d for the 14 tagged Chinook salmon in this study in Carroll's Channel. The mean residence time was 22 d, with a range from 0.03 to 62 d. Residence times were reasonably consistent between the Sandy River delta (2010 and 2011) and Carroll's Channel (2012) study areas.

Objective 3, Estuary Scale

Tag Release-Recapture Compendium – Prepare a compendium of tag release-recapture technologies to inform planning for future action effectiveness studies.

- The compendium provides an overview of statistical designs using mark-recapture techniques to assess juvenile salmon performance in the LCRE. It is intended to serve as a basis for instituting field research studies.

Acronyms and Abbreviations

°C	degree(s) Celsius
µm	micron(s)
7-DADM	seven-day average daily maximum
AA	Action Agencies
AER	action effectiveness research
BiOp	Biological Opinion
BPA	Bonneville Power Administration
CEERP	Columbia Estuary Ecosystem Restoration Program
d	day(s)
FCRPS	Federal Columbia River Power System
FL	fork length
ft	foot(feet)
g	gram(s)
GCE	gross conversion efficiency
h	hour(s)
IRI	Index of Relative Importance
JBHNWR	Julia Butler Hansen National Wildlife Refuge
JSATS	Juvenile Salmon Acoustic Telemetry System
kcfs	thousand cubic feet per second
km	kilometer(s)
L	liter(s)
LCRE	lower Columbia River and estuary
LRR	Lower River Reach
m	meter(s)
m^2	square meter(s)
m^3	cubic meter(s)
mg	milligram(s)
mm	millimeter(s)
MS-222	tricaine methanesulfonate
NMFS	National Marine Fisheries Service
ODFW	Oregon Department of Fish Wildlife
PNNL	Pacific Northwest National Laboratory
rkm	river kilometer(s)
RME	research, monitoring, and evaluation
RPA	Reasonable and Prudent Alternative
s	second(s)

s.d.	standard deviation
SRD	Sandy River delta
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service
UW	University of Washington

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1.0 Introduction

The study reported herein was conducted for the U.S. Army Corps of Engineers, Portland District (USACE) by researchers at the Pacific Northwest National Laboratory (PNNL), Oregon Department of Fish and Wildlife (ODFW), National Marine Fisheries Service (NMFS), University of Washington (UW), and U.S. Fish and Wildlife Service (USFWS). The goal of the study was to evaluate the ecological benefits[1] of restoration actions for juvenile salmon in the lower Columbia River and estuary (LCRE; rkm 0–234).

1.1 Study Objectives

This multi-year study (2011–2018) addresses the ecological benefits of restoration actions at multiple spatial scales over time. The spatial scales include the 1) site scale as a result of an individual project, 2) landscape scale as a result of multiple restoration actions located within a ~50-km segment of the LCRE, and 3) estuary scale as a result of the cumulative effects of multiple restoration actions estuary-wide.

Specific objectives for the 2012 study year were as follows (report sections and appendixes in which the topics are discussed are in parentheses):

Objective 1, Site Scale

a. Continue pre-restoration action effectiveness research to evaluate the effects of the upcoming dam removal/rechannelization at the Sandy River delta (SRD) (Section 2.0, Appendix A).

b. Continue post-restoration action effectiveness research to evaluate effects of the tide gate replacements at the Julia Butler Hansen National Wildlife Refuge (JBHNWR) mainland and Tenasillahe Island (Section 3.0, Appendix B).

Objective 2, Landscape Scale

a. Estimate juvenile salmon density in shallow water habitats between St. Helens and Longview (rkm 110–141) (Section 4.0).

b. Estimate residence time for tagged juvenile Chinook salmon during winter 2012 in off-channel habitats near Cottonwood Island (Carroll's Channel) (Section 5.0).

Objective 3, Estuary Scale

a. Prepare a compendium of tag release-recapture technologies to inform planning for future action effectiveness studies (Appendix C).

[1] For the purposes of this study, "ecological benefit" is defined as a net ecosystem improvement, across space and time (=trajectory of change) relative to key response variables: controlling factors (e.g., hydrology, water quality), structural attributes (e.g., habitat type, vegetation, substrate), and biological community presence and response (e.g., genetic stock identification, native and non-native species interactions, growth and diet, residence, migration, bioenergetics, mean fish density).

1.2 Background

The USACE's Anadromous Fish Evaluation Program includes estuary research that is used to adaptively manage decision-making for the federal Columbia Estuary Ecosystem Restoration Program (CEERP; Figure 1.1). The CEERP is conducted by the Action Agencies (Bonneville Power Administration and USACE) in response to mandates in the Biological Opinions (BiOps) on operation of the Federal Columbia River Power System (FCRPS) (e.g., NMFS 2008a). The study addresses Reasonable and Prudent Alternative (RPA) Actions 60.2 and 60.3 as they relate to evaluating the effects of restoration actions at project site, landscape, and estuary scales (NMFS 2008a). The study is also relevant to RPAs 37, 59, and 61.

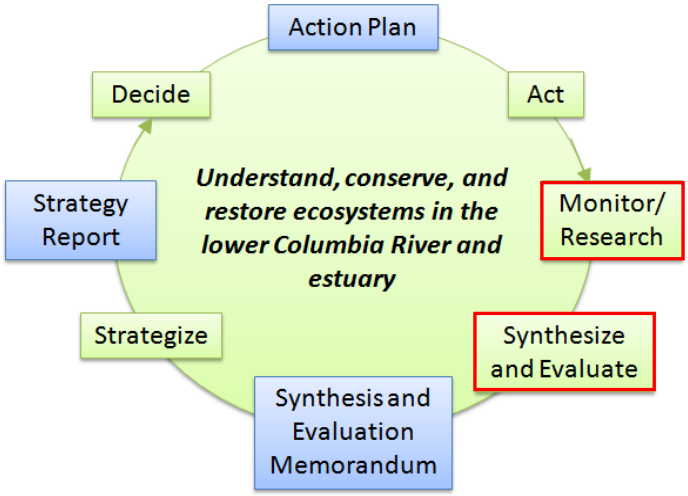

Figure 1.1. CEERP Adaptive Management Process. Green and blue boxes signify adaptive management phases and deliverables, respectively. Red outlines denote adaptive management phases to which the Multi-Scale Action Effectiveness study pertains. Modified from Thom et al. (2012a).

Annually, the CEERP is a large-scale effort to restore LCRE ecosystems for the benefit of juvenile salmon stocks listed as endangered or threatened under the Endangered Species Act (ESA). Evaluation of the ecological benefits of the restoration actions is essential to inform decision-makers about findings related to questions, such as: Did a particular action have the desired effect and, if not, why not? Which restorations actions are most effective at improving habitat access, capacity, and realized functions supporting juvenile salmon? Where are restoration actions most effective? Is the trend in juvenile salmon density increasing over time? Are multiple restoration actions having a positive effect on juvenile salmon ecosystems estuary-wide? Definitive answers to these and other basic questions about CEERP's effectiveness are not well-understood. Multi-scale action effectiveness research (AER) is focused on remedying this situation with science for the Monitor/Research and Synthesize and Evaluate phases of CEERP adaptive management (Figure 1.1). Thom et al. (2013) and the Independent Scientific Advisory Board (ISAB 2012) called for more work to determine the effectiveness of restoration actions in the LCRE.

In the LCRE (Figure 1.2), the substantial loss of shallow water habitats (e.g., Thomas 1983) through diking, filling, dredging, and development has been associated with the decline of salmon in the

Columbia River basin (Bottom et al. 2005). Shallow water habitats in the tidal freshwater and estuarine portions of the LCRE are important to the many life history strategies adopted by juvenile salmon (Fresh et al. 2005; Roegner et al. 2008). Restoration of shallow water habitat could enhance performance (e.g., foraging success and growth) and, thus, increase the survival of juvenile salmon (NMFS 2008a). The federal listing status of several salmonid stocks within the Columbia River basin and the resulting BiOps elucidated the need for a comprehensive understanding of salmon ecology within the LCRE. Improved understanding has resulted from key studies of juvenile salmon ecology in the LCRE, including studies by Johnson G. et al. (2009a, 2009b, 2011b), Campbell (2010), Haskell and Tiffan (2011), Maier and Simenstad (2009), Roegner et al. (2008), Bottom et al. (2005), and Dawley et al. (1986). Unlike basic juvenile salmon ecology in the LCRE, questions surrounding the effectiveness of restoration actions remain under investigation. Literature describing AER in the LCRE include studies by Diefenderfer et al. (2008), Diefenderfer and Montgomery (2009), Diefenderfer et al. (2010a), Haskell and Tiffan (2011), Johnson G. et al. (2009a, 2009b), Johnson J. et al. (2011), and Thom et al. (2013). Restoration is costly and outcomes are often uncertain. Without AER, resource managers will not be able to evaluate past restoration actions within the context of salmon recovery efforts. Furthermore, the planning and implementation of future actions may be hindered by the inability to link restoration actions and subsequent ecosystem responses.

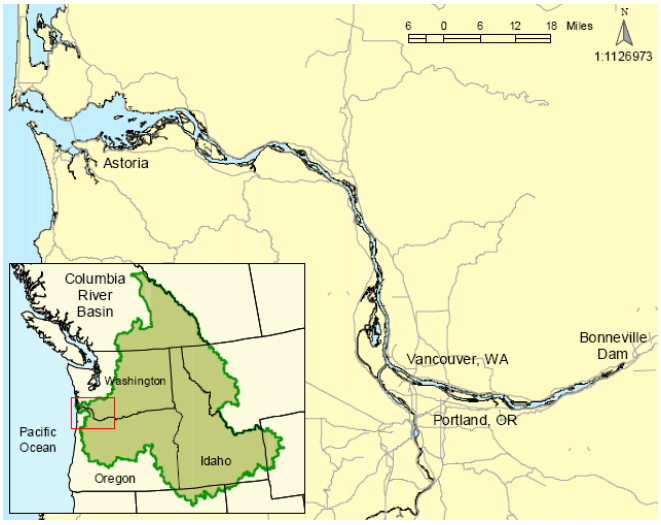

Figure 1.2. Map of the lower Columbia River and estuary (Bonneville Dam rkm 234 to the mouth rkm 0). The tidal freshwater region is about rkm 56–234.

1.3 General Approach

An integrated study design was informed by the ecological relationships defined in the Columbia River Estuary Conceptual Model: environmental stressors → controlling factors → habitat structure → habitat processes → ecosystem functions (Thom et al. 2005) and the relevant methods developed in the Salmon Benefits Study (Diefenderfer et al. 2010b). Data collection methods and sampling across multiple restoration sites (e.g., SRD, JBHNWR, and Tenasillahe sites) were coordinated and integrated for consistency to accomplish the study's goals and objectives. The study included matched restoration and reference/control sites, also as appropriate and available. Target habitats included main channel, tributary confluence, off channel, wetland channel, and others in which juvenile salmon rearing has been

documented (e.g., Johnson G. et al. 2011b). The null hypotheses, in terms of ecological benefits, were: pre-restoration conditions were equal to post-restoration conditions (site scale), and juvenile salmon density and seasonal distribution were not changing over time for a given estuary segment (landscape scale).

The SRD, JBHNWR, and Tenasillahe sites were chosen because they are part of ongoing AER in the LCRE. Similarly, due to the likelihood of restoration projects in the river segment from Longview to St. Helens (rkm 110–141), evaluating trends in juvenile salmon density across multiple habitat strata at a landscape scale has been ongoing since 2009.

This study provided a systematic assessment of physical and biological response ("ecological benefit") resulting from restoration actions in the LCRE. Ecological benefits, based on ecological relationships and responses at site, landscape, and estuary-wide scales, informed the Action Agencies' adaptive management process for LCRE restoration, including project selection and prioritization, project and alternatives development, and project evaluation.

1.4 Report Contents and Organization

This report contains six main sections and three appendixes. The site-scale AER data for SRD and JBHNWR (including Tenasillahe) are presented in Sections 2.0, and 3.0, respectively. Landscape density data and residence time data are presented in Sections 4.0 and 5.0, respectively. Section 6.0 contains a summary of 2012 findings and discussion. References for the literature cited in each chapter are listed in Section 7.0. Appendix A provides a synopsis of pre-restoration action effectiveness data from SRD. Appendix B is a synopsis of post-restoration action effectiveness data from the JBHNWR. Appendix C presents a compendium of mark-recapture techniques applicable to the action effectiveness work in the LCRE.

2.0 Site-Scale: Sandy River Delta

Prepared by Nichole Sather, Adam Storch, and David Teel

During 2012, we continued pre-restoration AER at the SRD. The restoration action (dam removal in the old Sandy River channel) is scheduled for summer 2013. The restoration action at the SRD is expected to increase habitat opportunity and capacity for juvenile salmon. In general, the intent of an AER investigation at the site scale is to quantify ecological benefits resulting from restoration actions. The null hypothesis is that, in terms of ecological benefits, pre-restoration conditions are equal to post-restoration conditions. The study objectives were as follows:

1. Characterize environmental conditions during the study period.
2. Characterize fish community composition of observed native and non-native fishes.
3. Estimate juvenile salmon density.
4. Describe the length frequency distribution of sampled salmon.
5. Estimate genetic stock identities of observed juvenile Chinook salmon.
6. Characterize juvenile Chinook salmon diet (number and biomass).
7. Estimate prey electivity for benthic, drifting, and winged or terrestrial taxa available to juvenile salmon for consumption.
8. Model bioenergetics to evaluate energy acquisition by juvenile salmon in shallow tidal freshwater by summarizing predicted growth, consumption, and gross conversion efficiency (GCE).

Pre-restoration AER at the SRD has been ongoing since June 2007 when our research in the SRD began. At that time, and since then, planning and design have been undertaken to reconnect the old Sandy River channel to the Columbia River. The low degree of connectivity between the Sandy River and the historic confluence likely constrains the functional integrity of this floodplain-deltaic ecosystem. Removal of the dam is intended to reestablish the connectivity of the Sandy River channel to its historic confluence. In pre-restoration sampling of fish and habitat characteristics within a formal Before-After-Control-Impact design, we noted the low degree of surface-water connectivity was correlated with low dissolved oxygen within the remnant channel, yet the absence of elevated water temperatures indicated the remnant channel maintains some degree of hyporheic connection with the Sandy River (Johnson J. et al. 2011, Appendix A, Figure A.7). Vegetation surveys near the remnant channel indicate a large proportion of obligate wetland species (Johnson J. et al. 2011, Appendix B, Figure B.11). Compared with other sites closer to the Columbia River, the remnant channel was also noted to have the greatest amount of submerged aquatic vegetation. We sampled juvenile Chinook and coho salmon in the remnant channel during previous research (Johnson J. et al. 2011). Given construction planned for summer 2013, sampling during 2012 concludes the pre-restoration phase of AER at the SRD.

2.1 Methods

The study area and sampling design and the methods for capturing fish and determining the genetic stock composition, fish diet, and prey availability are described in the following sections.

2.1.1 Study Area

The tidally influenced freshwater portion of the Columbia River extends from approximately Tenasillahe Island to Bonneville Dam (rkm 56–234). We sampled within two distinct areas of the tidal freshwater segment of the LCRE: the SRD and the Lower River Reach (LRR) area (Figure 2.1). Sites at the SRD are representative of off-channel habitats, with the exception of a wetland channel at Site N, which is located in a remnant channel of the historic SRD (Figure 2.2). These sites were selected for the purposes of the Before-After-Reference-Impact design (Johnson G. et al. 2011b). Full descriptions of habitat characteristics for each of the SRD sites are provided by Sobocinski et al. (2008) and Sather et al. (2009). Data from LRR sites are reported in Section 4.0.

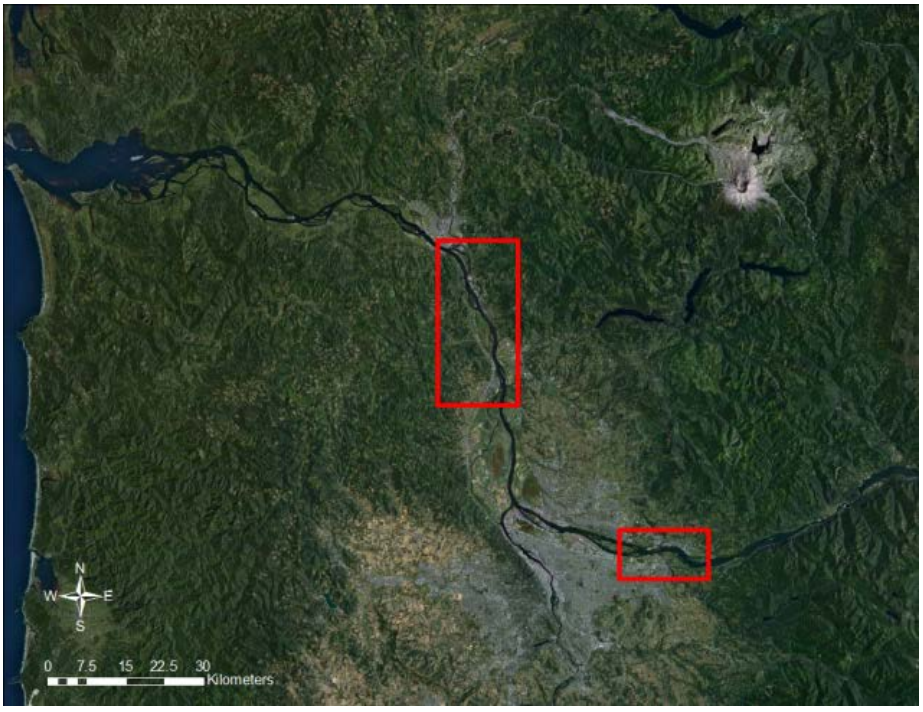

Figure 2.1. Location of the SRD (bottom rectangle; rkm 188–202) and LRR (top rectangle; rkm 110–141) study areas in the LCRE tidal freshwater. This section of the report concerns the SRD. The LRR is reported in Section 4.0.

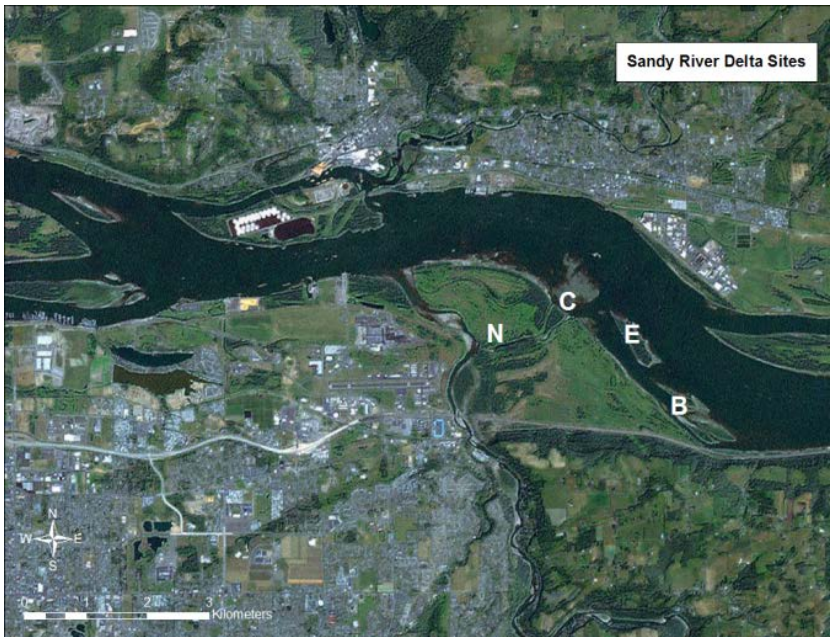

Figure 2.2. Sampling sites in the SRD study area (rkm 188–202).

2.1.2 Sampling Design

The SRD study area (rkm 188–202) includes four sites (Figure 2.2) that were selected as part of a Before-After-Reference-Impact experiment design (Sobocinski et al. 2008). These sites were typically sampled monthly, except when high flow conditions prevented data collection efforts (e.g., May and June 2012).

2.1.3 Fish Capture

To capture fish, we deployed either a 46-m beach seine (1.5–3 m depth; 13-mm knotless mesh wings; 3-mm knotless mesh purse; 15-m haul lines; sites B, C, and E) or a 30.5-m beach seine (3 m depth; 5-mm knotless mesh; site N). At sites B, C, and E, the beach seine was set by boat except when water depths were prohibitively low or site accessibility was poor at the time of sampling; during these instances the net was deployed by foot. Due to the inaccessibility by boat throughout the year and space constraints at site N, the smaller beach seine (30.5 m) was always set by foot. Two non-overlapping hauls were performed at each site with a minimum interval of 30 min between sets. After each haul, all salmon and steelhead were removed immediately from the net and placed in holding buckets filled with sufficiently oxygenated river water at ambient temperature. The remaining individuals (i.e., non-salmon taxa) were placed in separate holding buckets until processing. When catches were large, non-salmon fishes were subsampled according to the protocol described by Sather et al. (2011). To minimize handling stress, salmon and steelhead were anesthetized using a 40-mg/L solution of MS-222 (tricaine methanesulfonate) prior to processing.

We used boat and backpack electrofishing during May and June 2012. High flow during these months prevented effective sampling with a beach seine. While electrofishing doesn't allow us to calculate density estimates as is done with the beach seine data, we were still able to obtain samples for

other analyses such as diet composition and genetic stock identification. Boat and backpack electrofishing was conducted using a 20-ft electrofishing boat and an LR-24 Electrofisher (Smith-Root, Inc, Vancouver, WA), respectively. For both gear types, current was set to 4 amperes (pulsed direct current electrical output). After capture, fish were placed immediately in containers filled with aerated water at ambient temperature.

2.1.4 Genetic Stock Identification for Chinook Salmon

Samples collected for genetic mixture analysis were obtained from fin clips on a subset of juvenile Chinook salmon sampled from the SRD and LRR sites. All fin clips were preserved in ethanol until analysis. We used standard methods of genetic stock identification and individual assignment (reviewed by Manel et al. 2005). Chinook salmon were genotyped using the methods described by Teel et al. (2009). Data were collected for 13 microsatellite loci that have recently been standardized among several West Coast genetics laboratories (Seeb et al. 2007). Genetic mixture analysis and the relative probability of stock origin of each sample were estimated using the genetic stock identification computer program ONCOR (Kalinowski et al. 2007). Confidence intervals of the mixture proportions were estimated using ONCOR by re-sampling mixture and baseline data 100 times. Population baseline data were from the multilaboratory standardized Chinook salmon genetic database described by Seeb et al. (2007). Mixture proportions and assignment probabilities for individual baseline populations were summed to 10 Columbia River basin stock groups (Sather et al. 2011).

2.1.5 Fish Diet and Prey Availability

2.1.5.1 Field Processing – Fish Diet

Data were collected at four sites (B, C, E and N [Figure 2.2]; see Sather et al. 2011 for site descriptions) adjacent to the SRD near Troutdale, Oregon, from October 2011 through December 2012 to 1) characterize the diets of juvenile Chinook salmon, and (2) describe the compositions of specific prey pools in tidal freshwater habitats of the LCRE. Throughout the study period, no juvenile Chinook salmon were captured at site N; thus, specific methodologies applied at this site are not described.

Morphometric attributes and diets of juvenile Chinook salmon were sampled monthly at each site according to procedures detailed by Storch and Sather (2011). Anesthetized fish were measured to the nearest millimeter (fork length [FL]) and weighed (nearest 0.01 g). Gastric lavage was then performed on up to 20 fish, greater than 50 mm in length, to remove stomach contents. After lavage, samples were preserved, and salmon were allowed to recover before being released.

2.1.5.2 Field Sampling − Available Prey

To characterize community compositions of specific prey pools, we applied a combination of benthic, drift, and terrestrial/winged prey sampling methodologies as outlined by Storch and Sather (2011). At all sites, duplicate samples from each prey pool were collected and preserved. Benthos was sampled at two points parallel to the shore using a standard Ekman dredge (232 cm^2). Drifting invertebrates were sampled with plankton nets (363-µm mesh) placed 3–6 m from the waterline, midway in the water column, and facing upstream. Nets were set for approximately 24 h and, when possible, instantaneous flow readings were recorded near the mouth of each net at both the beginning and the end of sampling

periods. Terrestrial or winged organisms were sampled using floating fallout traps (0.2 m^2) filled with a solution of filtered river water and liquid detergent/surfactant. Traps were set for 48 h and positioned downstream of the drift nets.

2.1.5.3 Laboratory Procedures

Fish diet and prey community samples were processed in the laboratory following procedures described by Storch and Sather (2011). In the laboratory, prey items in samples from each site-sampling period (hereafter sampling episode) were identified to the lowest classification practicable using standard taxonomic keys (e.g., Merritt and Cummins 1996). Partially degraded organisms were identified based on paired or individual characteristic structures. Prey items of the same taxon and life history stage were counted and placed in labeled centrifuge vials containing 70% ethanol solution. Subsequently, whole animals stored in the centrifuge vials were weighed (blotted dry), individually or as a group depending on size, to the nearest 0.001 g.

As in diet samples, organisms in prey community samples were identified to the lowest feasible taxonomic resolution. Whenever possible we enumerated entire samples; however, when prey densities were appreciably large, we subsampled according to accepted protocols. Benthos was subsampled following methods adapted from Boward and Friedman (2000). For each benthic sample, randomly selected subsamples (i.e., fractions of the entire sample partitioned using a gridded tray) were enumerated successively until 120 organisms were counted or the entire sample had been processed. Organisms encountered in both drift and fallout samples were subsampled, when necessary, using standard procedures (Mills et al. 1992; Storch et al. 2007). Total sample counts were extrapolated from subsamples following relationships described in Storch and Sather (2011).

We estimated densities of taxa in the environment using methodologies applied previously (Storch and Sather 2011). For benthos, prey densities (individuals$\cdot m^{-2}$) were calculated by dividing sample counts by the area of the dredge opening. To estimate densities of drifting prey, the total volume of water flowing through each plankton net was first approximated using hydrographs for the lower Columbia River, recorded over the respective sampling periods and adjusted to beginning and end instantaneous flow measurements. Sample counts were then divided by the total volume of water flowing through each net to arrive at final total densities (individuals$\cdot m^{-3}$). Densities of terrestrial and/or winged prey were calculated by dividing fallout sample counts by both the area of the fallout trap and the sampling time (individuals$\cdot m^{-2} \cdot h^{-1}$).

2.1.5.4 Data Analyses

Relative Importance

The Index of Relative Importance (IRI; Pinkas et al. 1971) is a compound model combining information about a consumer's diet in terms of number, biomass, and frequency. To assess the importance of specific prey items in the general Chinook diet, we calculated IRI values by averaging the numbers and biomasses of individual prey found in gut contents during each site-sampling period combination (hereafter sampling episode) and then calculating a single composite score (Storch et al. 2007). These IRI scores were then standardized (%IRI) to fall within a discrete scale (i.e., 0–100%;

Cortés 1997) allowing for direct comparisons among different food types. Further details of IRI calculations are presented by Storch and Sather (2011).

Prey Selection

To characterize the foraging behavior of juvenile Chinook salmon at the study sites, we used the same stepwise approach described by Storch and Sather (2011), where 1), selectivity coefficients (Wi; Vanderploeg and Scavia 1979a) were calculated to summarize the relative proportion of prey items within a particular site in relation to the proportion of those prey items within the diets, and 2) Wi values were then standardized using the Relativized Electivity Index (Ei*;Vanderploeg and Scavia 1979b), representing the degree to which salmon were selecting or avoiding a particular prey item. Similar to the calculation of %IRI, single electivity coefficients were calculated by averaging numbers of individual prey found in gut contents during each sampling episode to represent generalized foraging behavior.

Electivity index values were calculated for each of the three potential prey sources sampled: benthos, drift, and fallout (i.e., terrestrial or winged prey). To achieve this, based on the life stage of prey items and/or knowledge of their general behaviors, diet data were coded according to where in the environment particular prey items were most likely to be encountered by a juvenile salmon. For example, although it is possible that a predator could encounter Daphnia spp. in the benthos, because the crustacean is planktonic, the likelihood is greater that the invertebrate was consumed in the drift.

Many prey items encountered in gut content samples could not be easily assigned to a specific habitat. To account for the uncertainty associated with prey taxa that could be encountered by a fish either in the benthos or the drift (hereafter termed “ambiguous” taxa), the electivity index was applied to gut content data matrices where 1) 50% of ambiguous prey were attributed to foraging in the drift and 50% of ambiguous prey were attributed to foraging in the benthos, 2) 100% of ambiguous prey were attributed to foraging in the drift, and 3) 100% of ambiguous prey were attributed to foraging in the benthos.

2.2 Results

The results include data on environmental conditions, fish community composition, salmon densities and lengths, genetic stock identification for Chinook salmon, salmon diet, prey electivity, and juvenile salmon bioenergetics.

2.2.1 Environmental Conditions

The shallow water habitats sampled as part of this study were influenced by the seasonal fluctuation in river discharge. While inter-annual variability affects the timing and magnitude of discharge, general seasonal patterns in the LCRE were such that lower flow conditions (75–125 kcfs) persisted from late summer through fall and peak discharge (350–450 kcfs) occurred from spring to summer months (Figure 2.3).

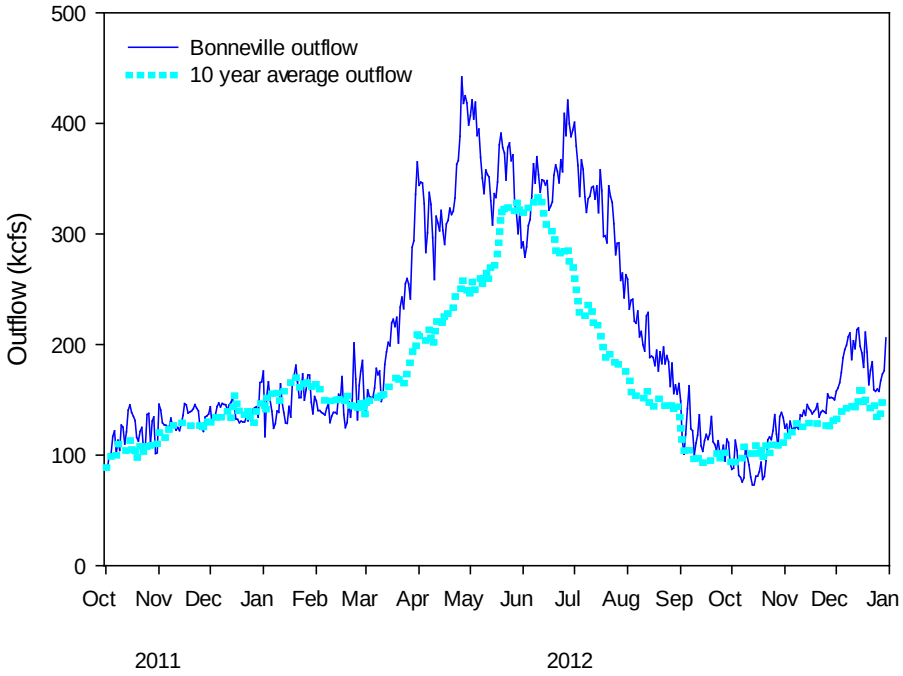

Figure 2.3. Daily average total discharge (kcfs) measured at Bonneville Dam, October 2012–December 2011. The 10-year average outflow (2004–2013) is displayed as the dotted light blue line (data from Columbia River Data Access in Real Time [2013]).

Seasonally, the four SRD sites were generally similar to each other in water elevation and temperature (Figure 2.4). The patterns observed across site-scales emulated seasonal patterns in river discharge. However, within a given season, variability in both water-surface elevation and water temperature was observed among the four SRD sites, especially when river discharge was low. The degree to which sites respond to river conditions is linked to a site's relative position to the main channel, as indicated by the seasonal pattern of water-surface elevation and temperature at site N, the furthest removed of the four sites. Site N conditions were different from those observed at the other three sites because of the relative lack of hydraulic connectivity.

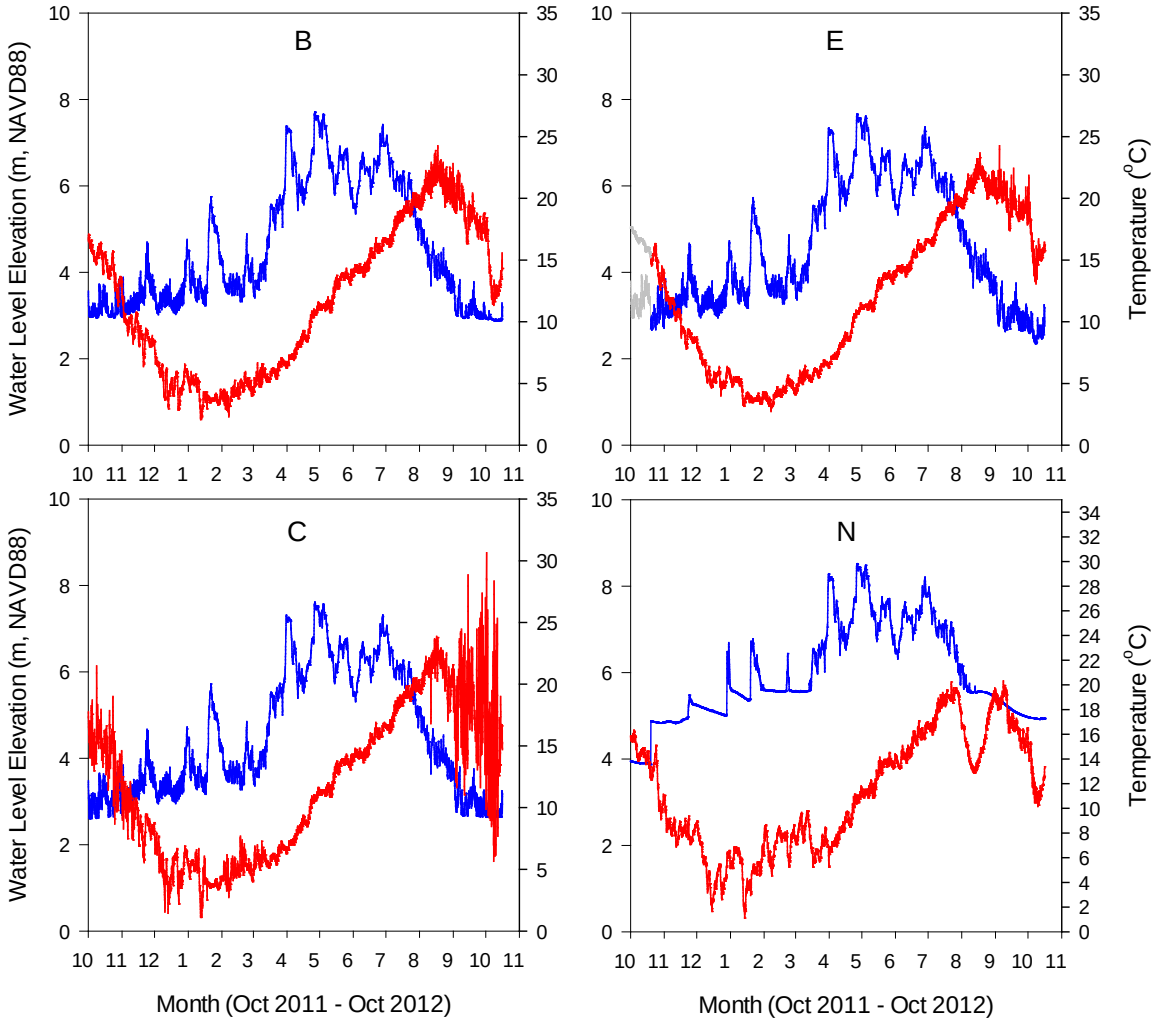

Figure 2.4. Water-surface elevation (blue) and water temperature (red) from Hobo data loggers at sites B, C, E, and N at the SRD. The Hobo sensor at site E was discovered to have moved from its installation location and been buried in sediment during a portion of October 2011; displayed in gray. Therefore, data corresponding to this time period should be considered uncorrected.

2.2.2 Fish Community Composition

Total catch at the SRD from October 2011 to December 2012 consisted of 27 species, of which 15 were non-native fishes (Table 2.1). While non-native fishes accounted for more than half of the taxa captured, they composed less than 2% of the total number of individuals caught. Threespine stickleback (*Gasterosteus aculeatus*), a native fish, accounted for 90% of the total number caught. Northern pikeminnow (*Ptychocheilus orgonensis*) and peamouth (*Mylocheilus caurinus*) each accounted for 3% of the total catch while sucker and sculpin spp. each accounted for 1%.

Table 2.1. Percentage of total catch for fish captured at the SRD sites. Catches were based on beach seine sampling efforts spanning October 2011–December 2012.

Scientific Name	Family Name	Common Name	Status	% of Total Catch	% of Total Excluding Threespine Stickleback
Gasterostedus aculeatus	Gasterosteidae	threespine stickleback	Native	90	-
Ptychocheilus oregonensis	Cyprinidae	Northern pikeminnow	Native	3	28
Mylocheilus caurinus	Cyprinidae	peamouth chub	Native	3	27
Catostomus spp.	Catostomidae	Sucker spp.	Native	1	13
Cottus spp.	Cottidae	sculpin spp.	Native	1	10
Fundulus diaphanus	Catostomidae	banded killifish	Non-native	0.7	7
Lepomis macrochirus	Centrarchidae	bluegill	Non-native	0.4	4
Oncorhynchus tshawytscha	Salmonidae	Chinook salmon	Native	0.2	2
Cyprinus carpio	Cyprinidae	common carp	Non-native	0.1	1
Lepomis spp.	Centrarchidae	sunfish spp.	Non-native	0.1	1
Micropterus dolomieu	Centrarchidae	smallmouth bass	Non-native	0.1	1
Micropterus salmoides	Centrarchidae	largemouth bass	Non-native	0.1	1
Oncorhynchus keta	Salmonidae	chum salmon	Native	0.09	0.9
Lepomis gibbosus	Centrarchidae	pumpkinseed	Non-native	0.08	0.8
Oncorhynchus kisutch	Salmonidae	coho salmon	Native	0.05	0.4
Oncorhynchus tshawytscha	Salmonidae	Hatchery Chinook salmon	Native	0.04	0.4
Richardsonius balteatus	Cyprinidae	redside shiner	Native	0.04	0.4
Cottus asper	Cottidae	prickly sculpin	Native	0.02	0.1
Cyprinidae	Cyprinidae	minnow spp.	Native	0.01	0.1
Catostomus macrocheilus	Catostomidae	largescale sucker	Native	0.007	0.07
Platichthys stallatus	Pleuronectidae	starry flounder	Native	0.006	0.06
Rhinichthys spp.	Cyprinidae	dace spp.	Native	0.006	0.06
Alosa sapidissima	Clupeidae	American shad	Non-native	0.006	0.06
Percopsis transmontana	Percopsidae	sandroller	Native	0.005	0.05
Oncorhynchus kisutch	Salmonidae	Hatchery coho salmon	Native	0.005	0.05
Rhinogobius brunneus	Gobiidae	Amur goby	Non-native	0.004	0.04
Gambusia affinis	Poecilidae	mosquito fish	Non-native	0.003	0.03
Perca flavescens	Percidae	yellow perch	Non-native	0.003	0.03
Ameriurus natalis	Ictaluridae	yellow bullhead	Non-native	0.001	0.01
Carassius auratus	Cyprinidae	goldfish	Non-native	0.001	0.01
Pomoxis nigromaculatus	Centrarchidae	black crappie	Non-native	0.0006	0.006
Ameiurus nebulosus	Ictaluridae	brown bullhead	Non-native	0.0006	0.006
Notemigonus crysoleucas	Cyprinidae	golden shiner	Non-native	0.0006	0.006
Prosopium williamsoni	Salmonidae	mountain whitefish	Native	0.0006	0.006

Fall 2012 yielded the highest mean densities (0.75 fish/m^2) for native taxa (excluding salmon) and salmon were most predominant during winter and spring (Figure 2.5). Peak densities for non-native taxa (0.01 fish/m^2) were observed during fall 2012. Spring 2012 yielded the lowest densities for native and non-native fish sampled at the SRD.

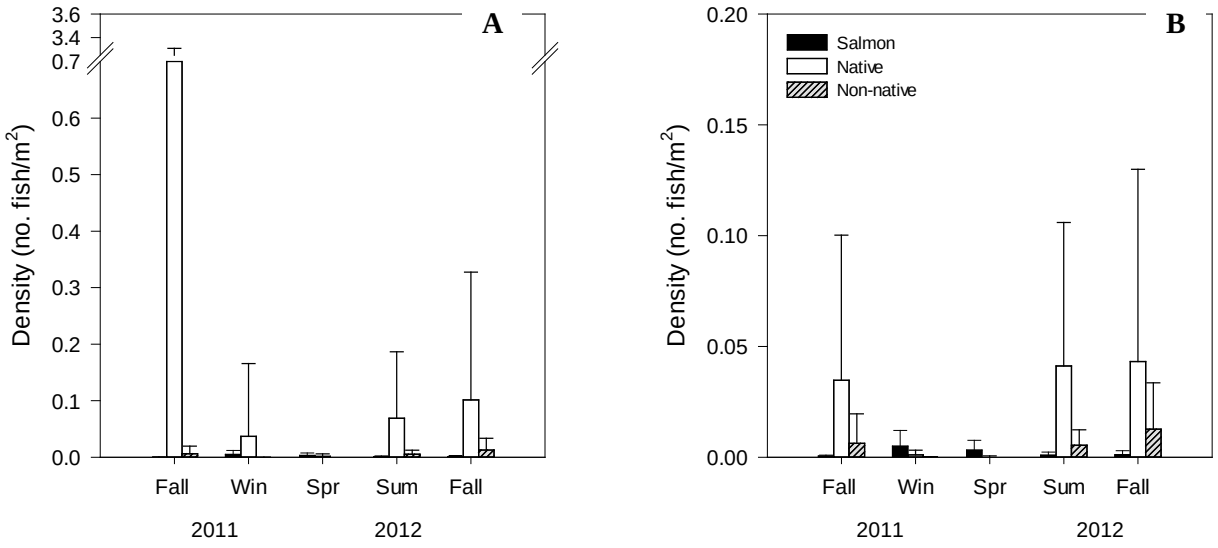

Figure 2.5. Mean density for salmon, native (excluding salmon), and non-native taxa at all SRD sites during the sample period from fall 2011 through fall 2012. Error bars represent one standard deviation. The panel on the left (A) depicts mean density for all taxa sampled during the study period and the panel on the right (B) represents mean density for all taxa, excluding threespine stickleback from the native catch.

2.2.3 Salmon Density

Three species of unmarked and marked salmon were captured at the SRD sites during the October 2011–December 2012 time period: chum, coho, and Chinook salmon (Figure 2.6). Unmarked Chinook salmon (Figure 2.7) were the only species captured during every season and were the most abundant salmon species captured at the SRD sites. Hatchery coho were the most infrequently captured taxa. The lowest salmon densities at the SRD occurred during fall.

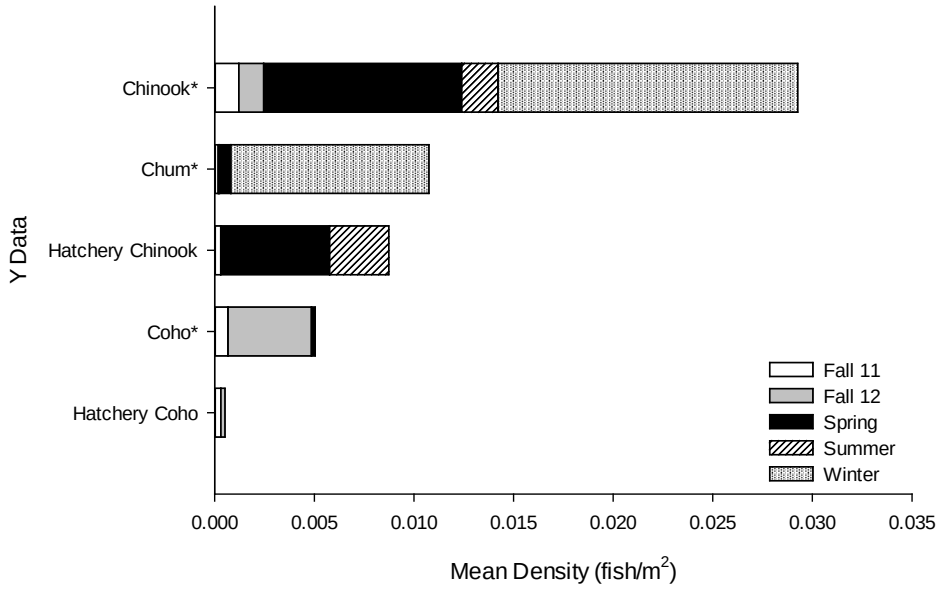

Figure 2.6. Mean density for salmonids at all SRD sites during the sampling period from fall 2011 through fall 2012. An asterisk (*) denotes fish were unmarked. Sampling spanned from fall 2011 through fall 2012; therefore, the two fall time periods are distinguished in the figure.

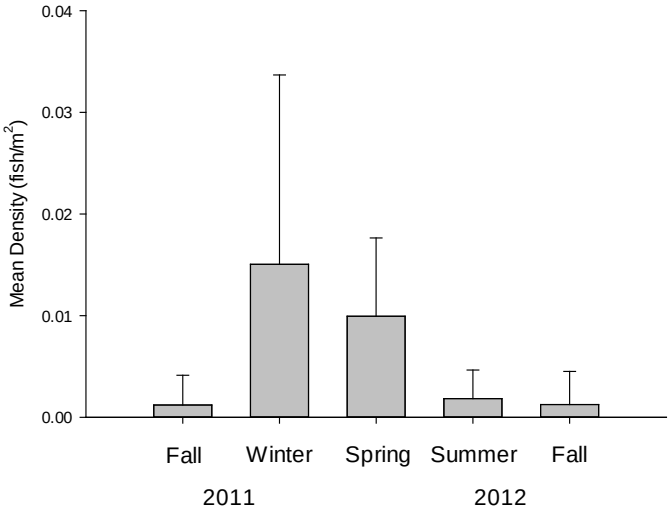

Figure 2.7. Mean seasonal density of unmarked Chinook salmon sampled at the SRD during the sampling period from fall 2011 through fall 2012. Error bars represent one standard deviation.

2.2.4 Salmon Lengths

Sizes for all salmonids captured during our study period ranged from 33 to 127 mm FL, but species-specific sizes tended to span narrower ranges, dependent on season (Table 2.2). The smallest mean size of all salmonids across seasons occurred during winter (mean 65 mm) and the largest occurred during fall 2011 (mean 96 mm). Chum salmon were the smallest salmonids, mean 41 and 44 mm during winter and spring, respectively.

Table 2.2. Size summary for salmonid species captured using a beach seine in the SRD study area from 2011–2012. Sizes are expressed as fork lengths (mm). Marked salmon were those without adipose fins and/or with coded wire tags.

Fall 2011												
Taxon	Common Name	N	Mean	Std Dev	Std. Error	C.I. of Mean	Range	Max	Min	Median	25%	75%
O. tshawytscha	Chinook salmon	21	106	13	2.8	5.9	49	127	78	108	99	114
O. tshawytscha	Chinook salmon (hatchery)	5	95	6.0	2.7	7.5	16	104	88	95	90	100
O. kisutch	Coho salmon	12	87	7.3	2.1	4.6	26	97	71	89	82	92
O. kisutch	Coho salmon (hatchery)	5	96	6.3	2.8	7.9	16	104	88	93	90.5	102
Winter												
Taxon	Common Name	N	Mean	Std Dev	Std. Error	C.I. of Mean	Range	Max	Min	Median	25%	75%
O. tshawytscha	Chinook salmon	153	42	9.7	0.8	1.6	82	115	33	40	38	44
O. keta	chum salmon	49	41	2.6	0.4	0.7	11	47	36	41	39	42
O. tshawytscha	Chinook salmon (hatchery)	1	113	--	--	--	0	113	113	113	113	113
Spring												
Taxon	Common Name	N	Mean	Std Dev	Std. Error	C.I. of Mean	Range	Max	Min	Median	25%	75%
O. tshawytscha	Chinook salmon	41	45	7.8	1.2	2.5	40	78	38	43	41	47
O. keta	chum salmon	3	40	1.7	1.0	4.3	3	42	39	39	39	42
O. tshawytscha	Chinook salmon (hatchery)	29	69	4.8	0.9	1.8	26	81	55	69	68	72
O. kisutch	Coho salmon	1	52	--	--	--	0	52	52	52	52	52
Summer												
Taxon	Common Name	N	Mean	Std Dev	Std. Error	C.I. of Mean	Range	Max	Min	Median	25%	75%
O. tshawytscha	Chinook salmon	28	79	6.3	1.2	2.4	28	92	64	79	74	82
O. tshawytscha	Chinook salmon (hatchery)	43	78	6.9	1.1	2.1	38	103	65	77	73	82
Fall 2012												
Taxon	Common Name	N	Mean	Std Dev	Std. Error	C.I. of Mean	Range	Max	Min	Median	25%	75%
O. tshawytscha	Chinook salmon	30	105	11.7	2.1	4.4	49	127	78	104.5	99	113
O. keta	chum salmon	1	41	--	--	--	0	41	41	41	41	41
O. tshawytscha	Chinook salmon (hatchery)	5	95	6.0	2.7	7.5	16	104	88	95	90	100
O. kisutch	Coho salmon	79	85	9.9	1.1	2.2	49	110	61	86	78	93
O. kisutch	Coho salmon (hatchery)	9	98	7.8	2.6	6.0	24	112	88	97	91	103

The patterns associated with length frequency distributions of unmarked Chinook salmon captured in shallow water habitats indicated distinct temporal trends (Figure 2.8). During winter months, unmarked Chinook salmon primarily comprised fry size fish (<60 mm), while few fish occupied larger sizes (>100 mm), which are indicative of different life stages. Spring months corresponded to times in which small size classes (<100 mm) were present. After the occurrence of small sizes of unmarked Chinook during the spring months, the overall sizes of unmarked Chinook salmon sampled at the SRD increased during summer and fall months.

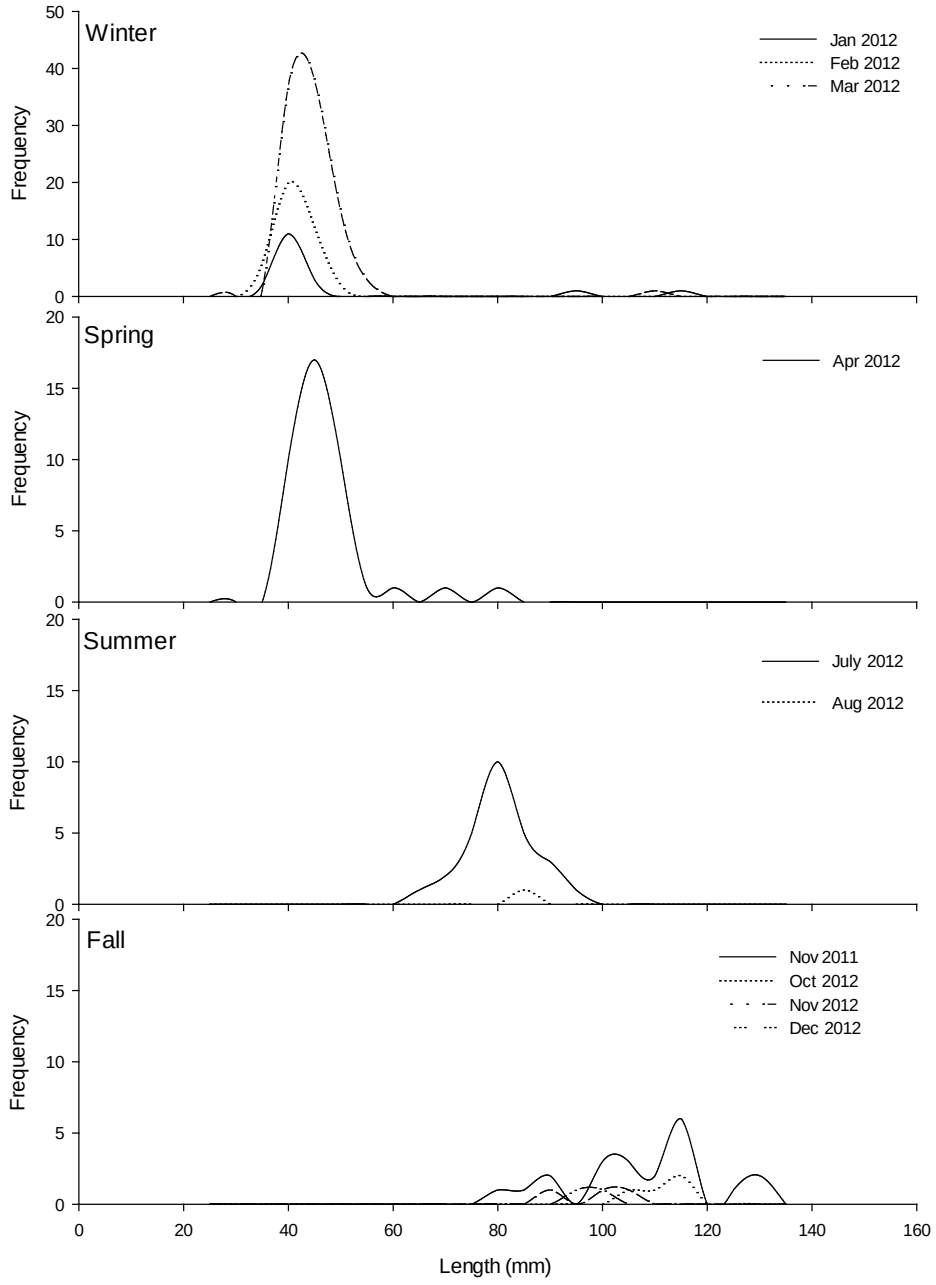

Figure 2.8. Seasonal length frequency distribution for unmarked Chinook salmon sampled at the SRD study area between October 2011 and December 2012.

2.2.5 Genetic Stock Identification for Chinook Salmon

A total of 269 Chinook salmon were genotyped at 7 or more of the 13 microsatellite loci and used in genetic stock identification analysis. Stock identification estimates from the analysis of 192 unmarked Chinook salmon sampled in the SRD are presented in Table 2.3. Most of the fish were from the Spring Creek Group Tule Fall (39%) and the Upper Columbia Summer/Fall (33%) stock groups. Smaller proportions were estimated for the Willamette River Spring (8%), West Cascade Tributary Fall (7%), Snake River Fall (6%), West Cascade Tributary Spring (5%), Deschutes River Fall (1%), and Mid and Upper Columbia River Spring (1%) groups. A total of 77 marked (known hatchery origin) Chinook salmon captured in the SRD were analyzed genetically (Table 2.4). Most of the hatchery fish were also from the Upper Columbia Summer/Fall (56%) and the Spring Creek Group Tule Fall (36%) stock groups. Four other stock groups contributed small proportions to the marked fish mixture (1%–5%). Individual fish genetic stock assignments of samples of unmarked SRD fish were grouped by survey month and are presented in Figure 2.9. Genetic sample sizes of the surveys ranged from 1 to 61 individuals. The data revealed strong seasonal shifts in stock compositions with samples early in the year (January–March) dominated by Spring Creek Group Fall juveniles, followed by increased proportions of Upper Columbia Summer/Fall fish (April–July). Although very few samples were analyzed from sampling later in the year, juveniles in November and December were mostly spring Chinook salmon.

Table 2.3. Estimated percentage genetic stock composition and 95% confidence intervals of 192 unmarked juvenile Chinook salmon sampled at SRD sites from November 2011 through December 2012.

Genetic Stock Group	Estimated Contribution (%)	95% Confidence Interval	
Upper Columbia River Summer/Fall	32.9	22.4	40.4
West Cascade Tributary Fall	6.9	4.0	15.3
Spring Creek Group Tule Fall	39.1	28.4	42.5
Snake River Fall	6.3	1.9	12.1
Willamette River Spring	8.2	4.1	12.6
Deschutes River Fall	1.2	0.0	7.1
West Cascade Tributary Spring	4.5	1.9	9.2
Mid and Upper Columbia River Spring	0.5	0.0	1.9
Snake River Spring	0.0	0.0	0.0
Rogue River	0.0	0.0	0.9

Table 2.4. Estimated percentage genetic stock composition and 95% confidence intervals of 77 marked juvenile Chinook salmon sampled at SRD sites from November 2011 through December 2012.

Genetic Stock Group	Estimated Contribution (%)	95% Confidence Interval	
Upper Columbia River Summer/Fall	55.7	42.5	63.9
West Cascade Tributary Fall	4.5	0.0	11.1
Spring Creek Group Tule Fall	36.1	26.5	47.2
Snake River Fall	0.6	0.0	8.2
Willamette River Spring	0.0	0.0	0.0
Deschutes River Fall	1.8	0.0	7.9
West Cascade Tributary Spring	1.3	0.0	4.3
Mid and Upper Columbia River Spring	0.0	0.0	1.9
Snake River Spring	0.0	0.0	2.6
Rogue River	0.0	0.0	0.0

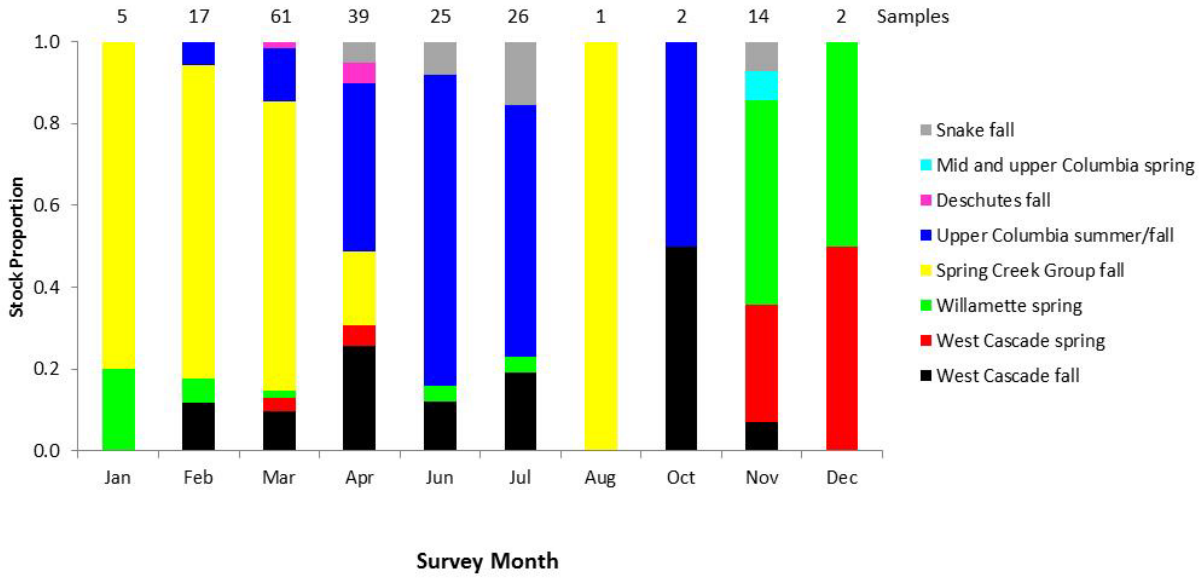

Figure 2.9. Estimated stock group proportions, sample sizes, and month of sampling of unmarked juvenile Chinook salmon at SRD sites during 2012. November proportions include samples collected in 2011. Snake River spring and Rogue River fall stock groups were not estimated to contribute to the samples.

2.2.6 Salmon Diet

Despite variability in space and time, the diets of juvenile Chinook salmon sampled at our sites from November 2011 through December 2012 generally were dominated by dipterans (primarily chironomids and ceratopogonids) and amphipods. Of these prey taxa, dipterans were most frequently consumed in large proportions; accounting for more than 20% of the diet during 77% of sampling episodes in which

non-empty gut content samples were collected. Across sites, amphipods were encountered regularly in the diet, accounting for more than 20% of consumed biomass during 27% of sampling episodes, with the maximum proportion during any one sampling episode occurring at site C (0.63, August 2012; Figure 2.10).

Considerable proportions of hemipterans were encountered in gut contents at each site, accounting for more than 20% of gut content biomass at site B during 25% of sampling episodes, at site C during 20% of sampling episodes, and at site E during 11% of sampling episodes. Both Coleopteran and hymenopteran biomass was represented in the gut contents of juvenile Chinook salmon at all sites during at least one sampling episode but generally accounted for relatively small proportions (Coleoptera, range = 0.00–0.05; Hymenoptera, range = 0.00–0.22). Appreciable diet proportions of the group consisting of other aquatic insects (Collembola, Ephemeroptera, Megaloptera, Plecoptera and Trichoptera) were encountered at sites B and E (maximum = 0.42 and 0.41, respectively), but were restricted to few applicable sampling months (>20% of the diet during 14% of all sampling episodes). Known terrestrial insects (Psocoptera and Orthoptera) were relatively uncommon in the diet, accounting for no more than 5% of consumed biomass during any one sampling episode. Insects that could not be identified beyond class (i.e., "Unidentified Insecta") due to degradation resulting from digestive processes infrequently contributed more than 20% to the diet of juvenile Chinook salmon (5% of sampling episodes). Although mysids were encountered in the diet at all sites, the macrocrustacean constituted more than 20% of the diet only at sites B and E (25% and 11% of sampling episodes, respectively). While cladocerans rarely constituted more than 20% of the diet (9% of sampling episodes), the crustaceans were at least marginally present in gut contents during several sampling episodes across sites. Copepods generally were underrepresented, accounting for less than 6% when present in the diet. Combined biomass proportions of prey items included in the "Other" category (Arachnida, unidentified crustaceans, fish, mollusks, and Nemata/Nematomorpha) were encountered periodically at all sites; the maximum proportion occurred at site B (0.15, November 2012; Figure 2.10).

Trends in %IRI for taxa associated with weighted mean values ≥10% (see Storch and Sather 2011 for justification) largely mirrored those described by biomass proportions (c.f., Figure 2.10 and Figure 2.11). Dipterans and amphipods were commonly the most important prey taxa; they had combined %IRI values greater than 50% during 68% sampling episodes. Of the two taxa, Diptera was most commonly associated with the greatest %IRI scores (81% of sampling episodes), which exceeded 50% during 55% of all applicable sampling events (mean = 52.71% ± 29.64 s.d.; range = 0.0%–100.0%; Figure 2.2). Among sites, the largest %IRI values for dipterans were calculated for site C (mean = 70.57% ± 19.52), followed by sites E (mean = 56.02% ± 34.14) and B (mean = 37.81% ± 24.31). When encountered in gut contents, on average, amphipods appeared to be of greater importance in the diets of juvenile Chinook salmon at sites B (mean = 13.06% ± 28.01) and C (mean = 15.71% ± 24.70) than at site E (mean = 5.22% ± 9.00), although values varied considerably among sampling months.

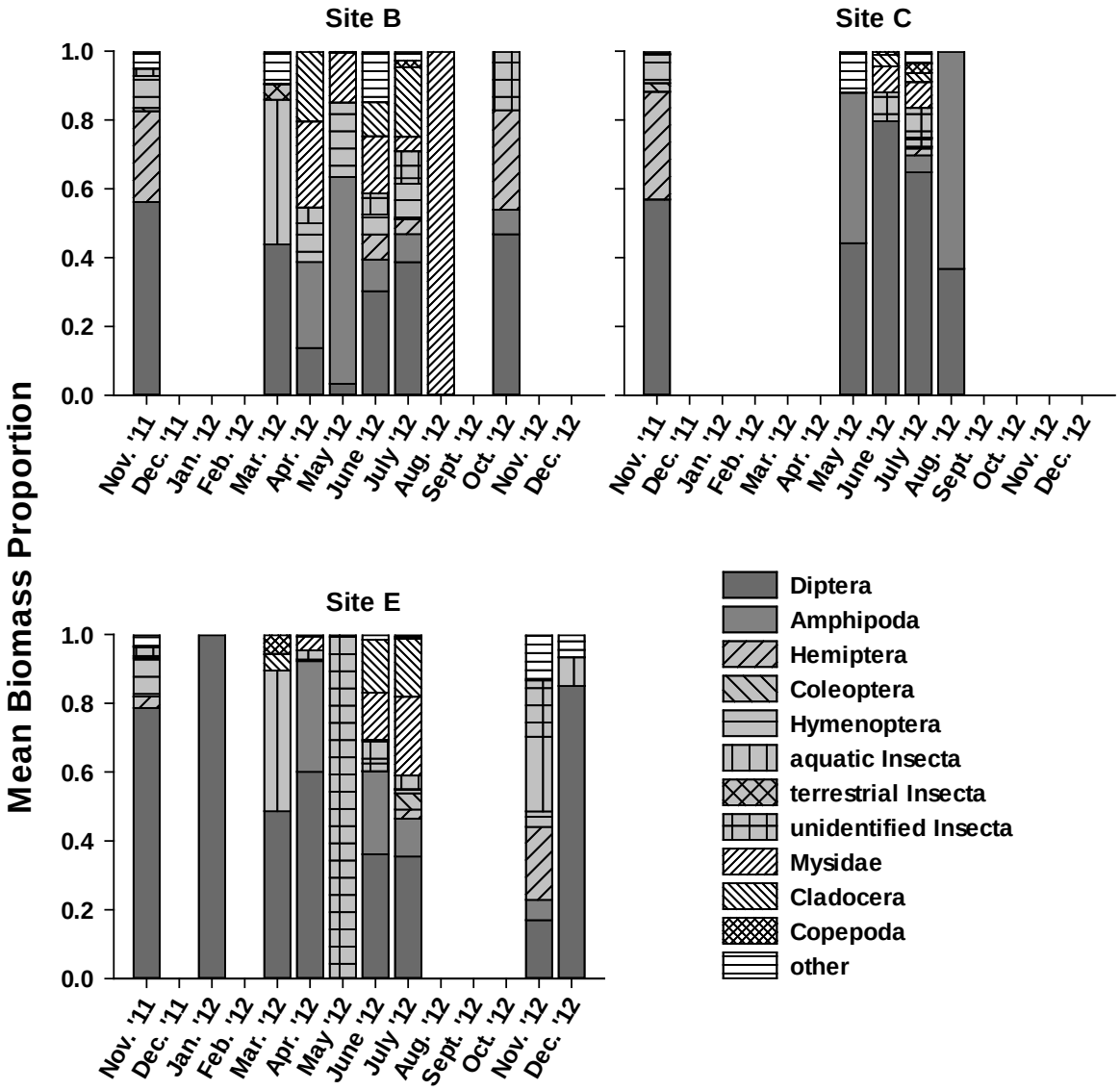

Figure 2.10. Distribution of mean biomass proportions for major prey categories found in the gut contents of Chinook salmon. Missing data indicate episodes in which sampling was not conducted or no Chinook salmon of a size appropriate for gastric lavage were encountered.

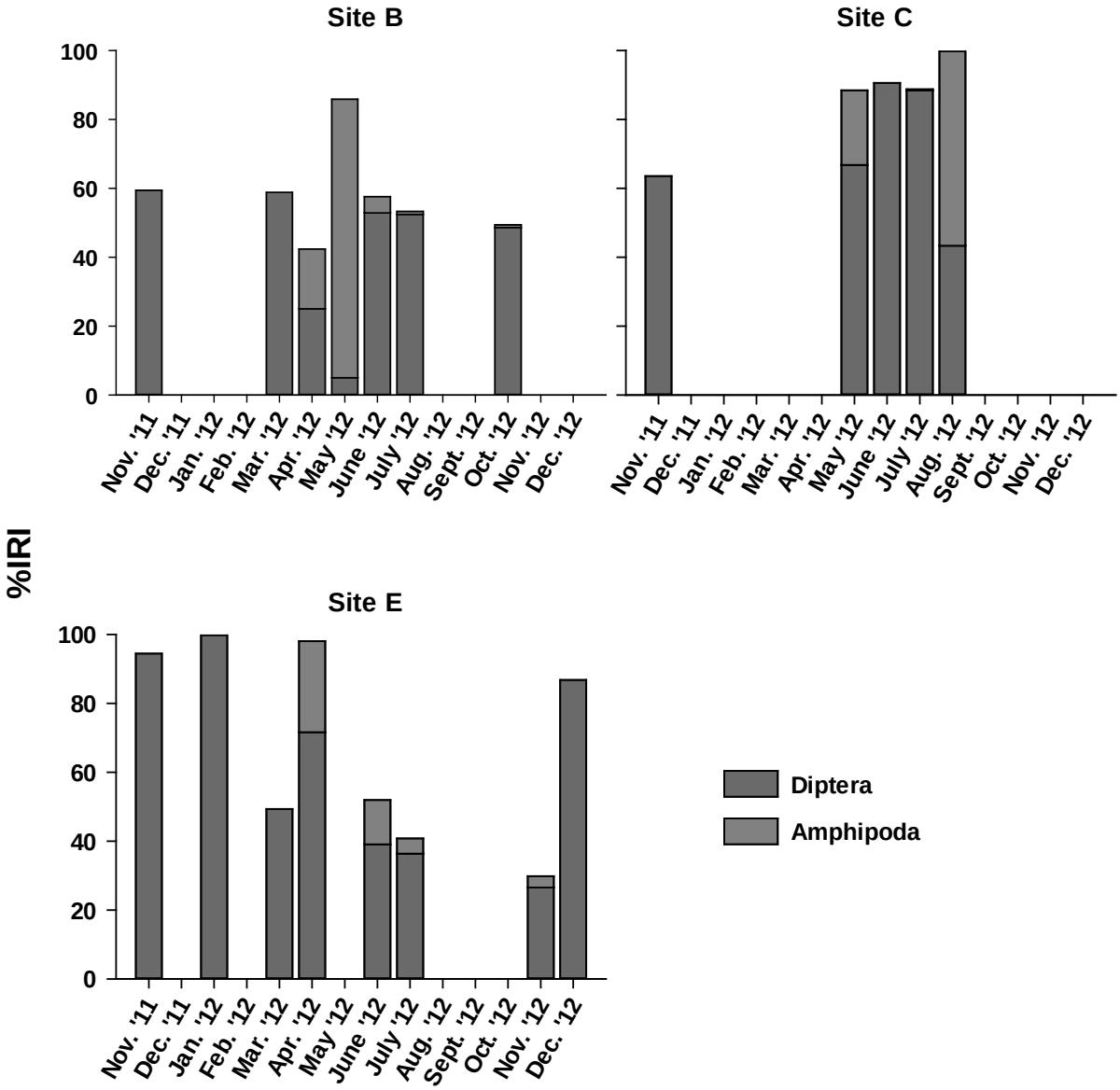

Figure 2.11. Distribution of %IRI values for major prey categories found in the gut contents of juvenile Chinook salmon. Missing data indicate episodes in which sampling was not conducted or no Chinook salmon of a size appropriate for gastric lavage were encountered.

2.2.7 Prey Electivity

Apportioning ambiguous diet items had little effect on electivity scores and, in turn, no impact on conclusions that may be drawn from the index output. Thus, no contrasts between 100% and 50% scenarios (see Methods) are described below. Because %IRI values identified two groups generally to be most important across sampling episodes (Diptera and Amphipoda; Figure 2.2), electivity values for only these prey items are presented.

2.2.7.1 Benthic Prey

When dipterans were encountered in gut contents and/or benthic samples at sites B and E, the taxon was selected against. While dipterans were selected against at site C during November 2011, during June and July 2012, the prey item was associated with positive electivity index values. At site B, amphipods were a preferred prey item during June 2012 and avoided during July 2012. Amphipods were always associated with negative electivity scores at site C, whereas at site E, the taxon was selected against during June and July of 2012 and preferred November of 2012 (Figure 2.12 and Figure 2.13).

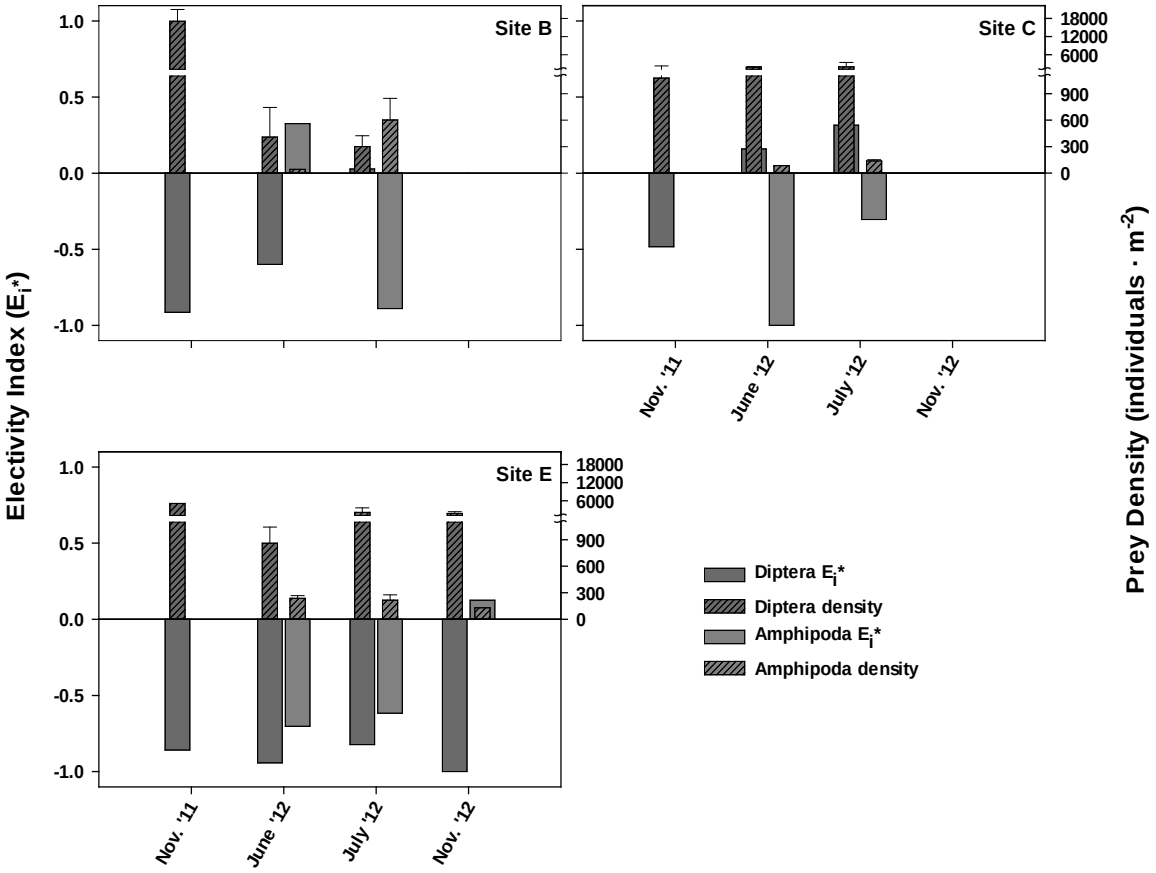

Figure 2.12. Relativized electivity index values and prey densities calculated for major benthic prey items. Values were calculated with 100% of the "ambiguous" prey items allocated to benthic production. Across sampling episodes, these taxa were never consumed in proportion to their abundances in the environment (E* = 0.0). No Chinook salmon of a size appropriate for gastric lavage were encountered at site N during months indicated in the figures. Error bars represent one standard deviation.

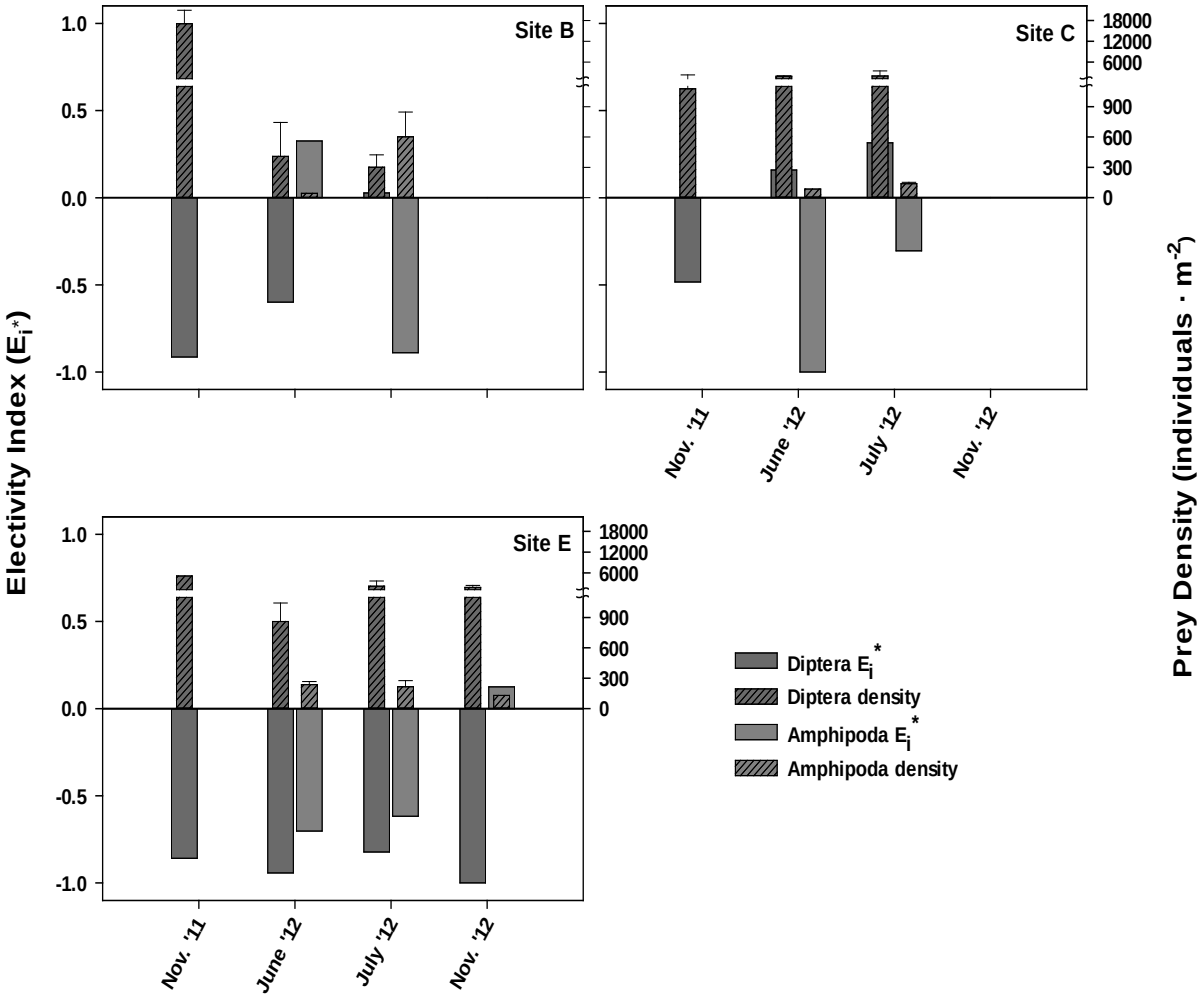

Figure 2.13. Relativized electivity index values and prey densities calculated for major benthic prey items. Values were calculated with 50% of the "ambiguous" prey items allocated to benthic production. Across sampling episodes, these taxa were never consumed in proportion to their abundances in the environment (E* = 0.0). No Chinook salmon of a size appropriate for gastric lavage were encountered at site N during months indicated in the figures. Error bars represent one standard deviation.

2.2.7.2 Drifting Prey

Regardless of site or sampling month, electivity index values calculated for dipteran prey in the drift were negative, indicating the taxon was avoided in the water column. When amphipods were encountered in gut contents and/or the environment at sites C and E, the macro-crustaceans were avoided invariably, while at site B, amphipods were selected against during November 2011 and June 2012 and selected for during July 2012 (Figure 2.14 and Figure 2.15).

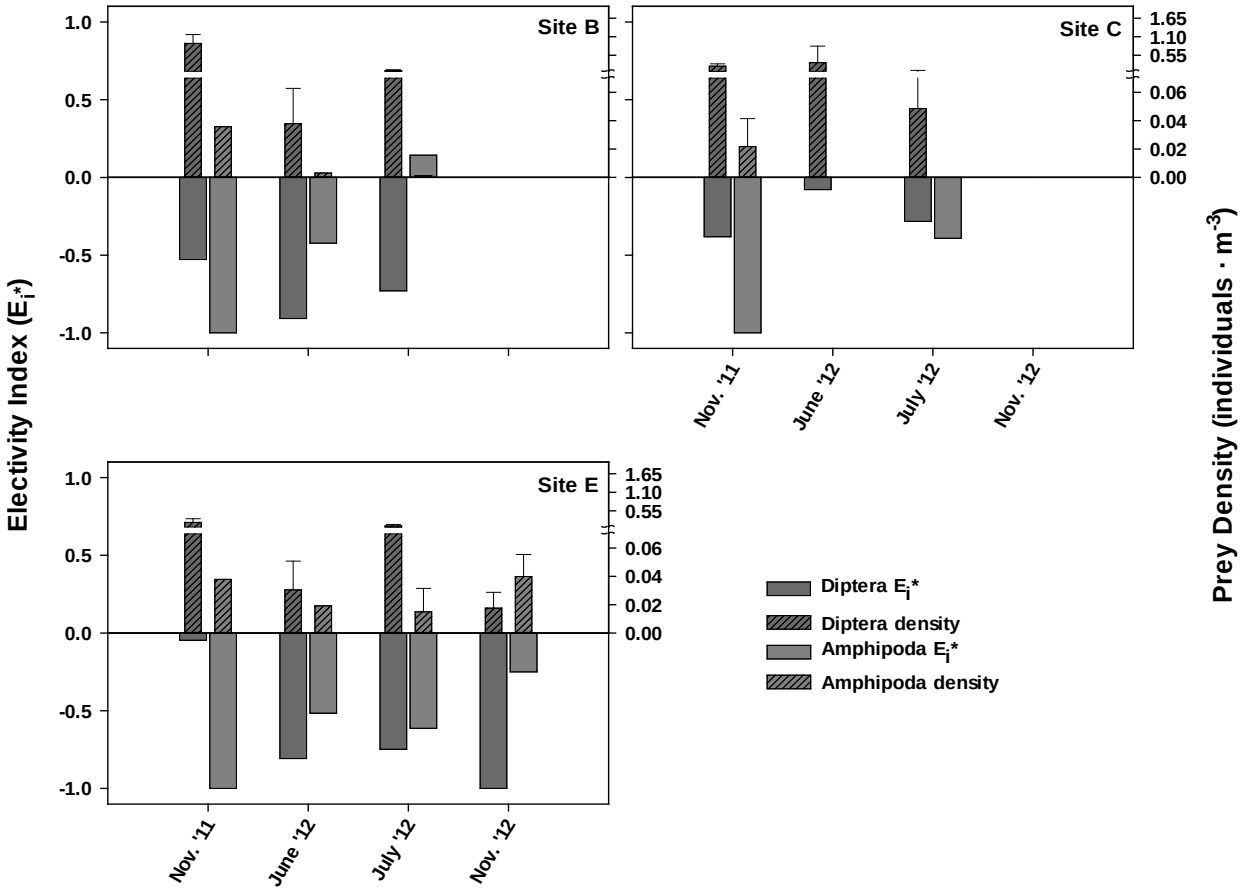

Figure 2.14. Relativized electivity index values and prey densities calculated for major drifting prey items. Values were calculated with 100% of the "ambiguous" prey items allocated to benthic production. Across sampling episodes, these taxa were never consumed in proportion to their abundances in the environment (E* = 0.0). No Chinook salmon of a size appropriate for gastric lavage were encountered at site N during months indicated in the figures. Error bars represent one standard deviation.

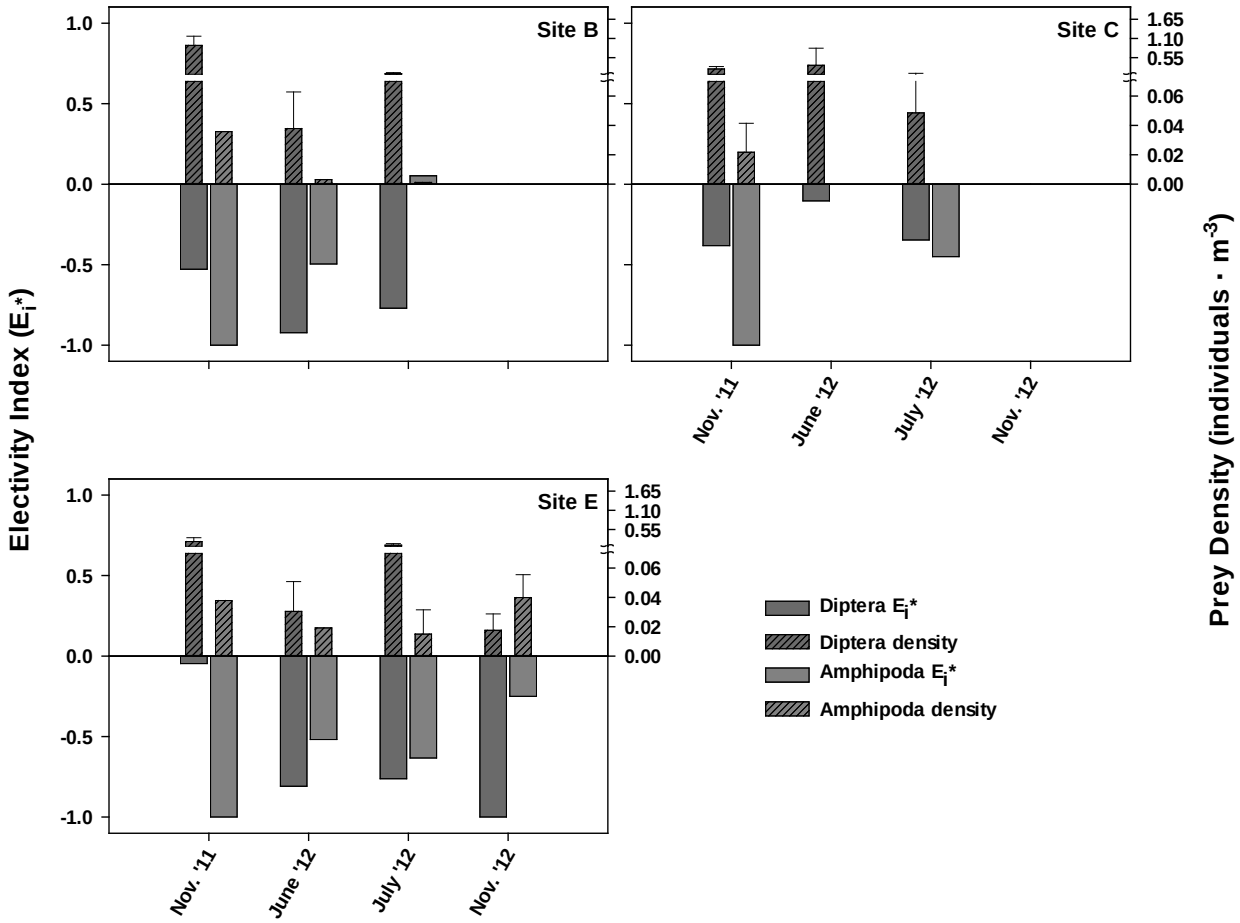

Figure 2.15. Relativized electivity index values and prey densities calculated for major drifting prey items. Values were calculated with 50% of the "ambiguous" prey items allocated to benthic production. Across sampling episodes, these taxa were never consumed in proportion to their abundances in the environment (E* = 0.0). No Chinook salmon of a size appropriate for gastric lavage were encountered at site N during months indicated in the figures. Error bars represent one standard deviation.

2.2.7.3 Terrestrial and Winged Prey

The behavior of juvenile Chinook salmon foraging on winged or terrestrial dipterans was relatively consistent, varying little across sampling episodes except during June 2012 at site C, where the invertebrates were selected for and dipterans were avoided (Figure 2.16).

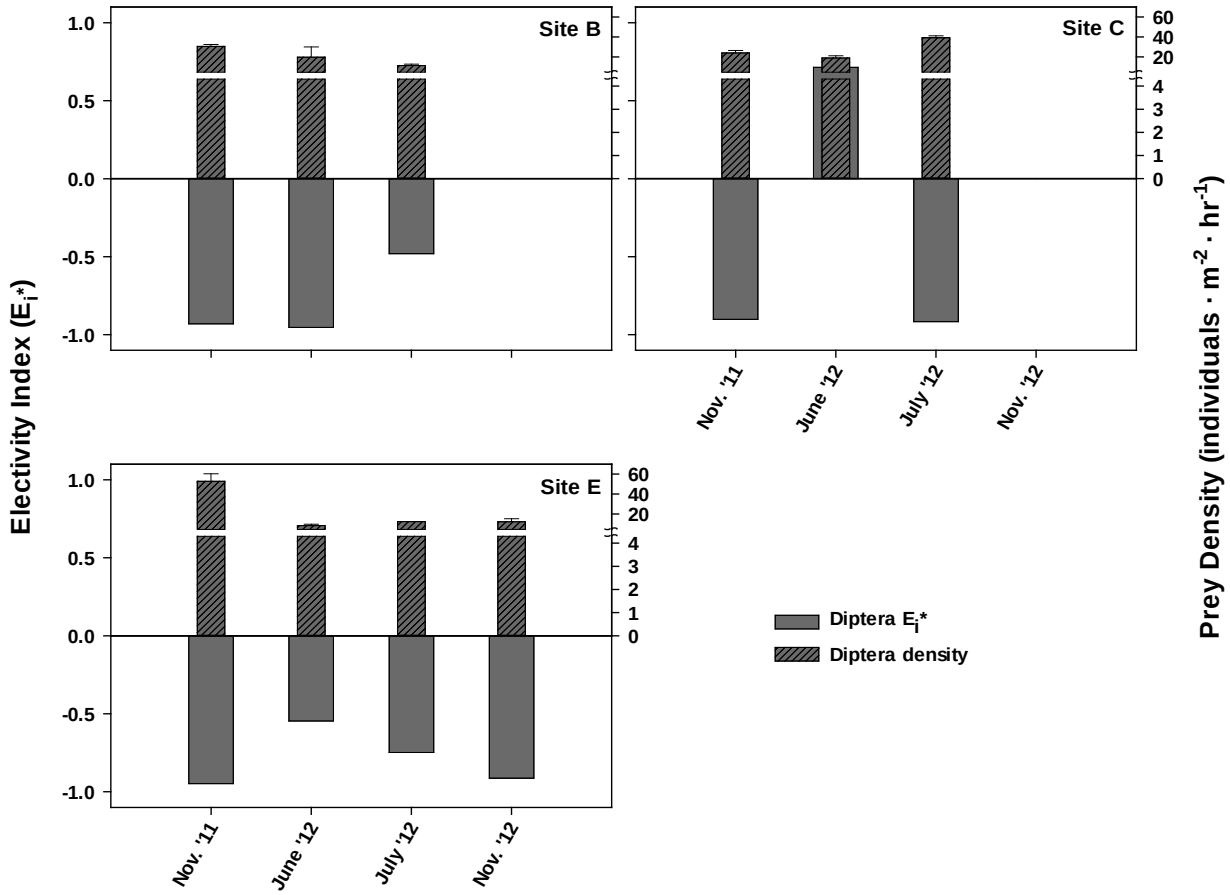

Figure 2.16. Relativized electivity index values and prey densities calculated for major terrestrial or winged prey items. Across sampling episodes, these taxa were never consumed in proportion to their abundances in the environment (E* = 0.0). No Chinook salmon of a size appropriate for gastric lavage were encountered at site N during months indicated in the figures. Error bars represent one standard deviation.

2.2.8 Juvenile Chinook Salmon Bioenergetics

2.2.8.1 Initial and Final Body Mass

The length-biomass regression models applied to estimate initial (W_{ti}) and final (W_{tf}; Table 2.5) were all significant at $\alpha = 0.05$. All models, including the combined model, fit the data well, with coefficients of determination (R^2) ranging from 0.729 to 0.982 (Table 2.5). The combined model was used to estimate W_{ti} or W_{tf} for months where inadequate sample sizes precluded development of period-specific equations (c.f., Table 2.1 and Table 2.2).

Table 2.5. Parameters and fit statistics for length-biomass regression models used to estimate initial and final masses for bioenergetics simulations. All data were log-transformed prior to analysis. Models were considered significant at $\alpha = 0.05$.

Model	RMSE	R2	Model prob > F	Length (FL)	Intercept
Nov 2011	0.079	0.962	<0.0001	2.987	-11.440
Feb 2012	0.142	0.729	<0.0001	3.374	-13.077
Mar 2012	0.155	0.895	<0.0001	3.315	-12.771
Apr 2012	0.128	0.979	<0.0001	3.344	-12.938
May 2012	0.051	0.995	<0.0001	3.091	-11.850
June 2012	0.138	0.874	<0.0001	3.149	-12.116
July 2012	0.077	0.897	<0.0001	2.801	-10.477
Combined	0.161	0.982	<0.0001	3.309	-12.781

2.2.8.2 Feeding Rate

Simulated site-specific feeding rates (P values) generally were moderate and varied substantially among cohorts (mean = 0.57±0.18 s.d.; range = 0.42–0.94). Although the largest mean feeding rate occurred at site C (mean = 0.60±0.23), followed by sites E (mean = 0.57±0.18) and B (mean = 0.55±0.17), fitted P values were not significantly different among sites (Kruskal-Wallis rank sum test, $\chi 2 = 0.1229$, $p = 0.9404$). Simulated feeding rates exceeded the theoretical maximum (1.00) during three residence periods: 1 Aug 2012–15 Sept 2012 (site B), 1 Aug 2012–16 Sept 2012 (site C), and 1 Jan 2012–21 Jan 2012 (site E). At both sites B and C, these simulation periods were associated with thermal peaks (mean = 20.56°C±1.19 and 20.09°C±1.77, respectively). Alternatively, at site E, the one simulation period in which a simulated P value exceeded 1.0 was associated with the lowest mean temperature observed (mean = 4.48°C±0.75).

2.2.8.3 Growth

Predicted $\overline{SGR}$ values were positive for all simulation cohorts irrespective of site. For cohorts where fitted P values (i.e., those not set to 1.0, see methods) were applied, modeled $\overline{SGR}$ was positive even after perturbing rates of feeding by –10%. Across all site-cohort combinations, mean specific growth rates modeled using fitted P values ranged from 0.010 $g \cdot g^{-1} \cdot d^{-1}$ and 0.022 $g \cdot g^{-1} \cdot d^{-1}$ (mean = 0.015 $g \cdot g^{-1} \cdot d^{-1}$ ± 0.004 s.d.), with both minimum and maximum values occurring at site E (minimum: simulation cohort = 1, residence period = 8 Nov 2011–28 Nov 2011; maximum: simulation cohort = 5, residence period = 1 Mar 2012–24 Apr 2012; Figure 2.17).

For residence periods where fitted feeding rates exceeded the theoretical maximum, and consequently P values were set at 1.00 in bioenergetics simulations, the period-specific thermal experience and the magnitude of growth response appeared to vary among sites. At site B, a conspicuous relative decrease (% change from previous simulation cohort = –92%) in $\overline{SGR}$ occurred during the simulation conducted from 1 Aug 2012–15 Sept 2012 while during a comparable period (1 Aug 2012–15 Sept 2012) at site C, reduced growth was much less pronounced (% change from previous simulation cohort = –23%). As noted previously, these simulated decreases in $\overline{SGR}$ were concomitant with the greatest mean

temperatures observed at each site (i.e., 20.56°C±1.19 and 20.09°C±1.77, respectively). Alternatively, the lowest predicted $\overline{SGR}$ encountered at site E (1 Jan 2012–21 Jan 2012) was associated with the thermal minimum. Even during these periods of temperature extremes, juvenile Chinook salmon gained biomass (Figure 2.17).

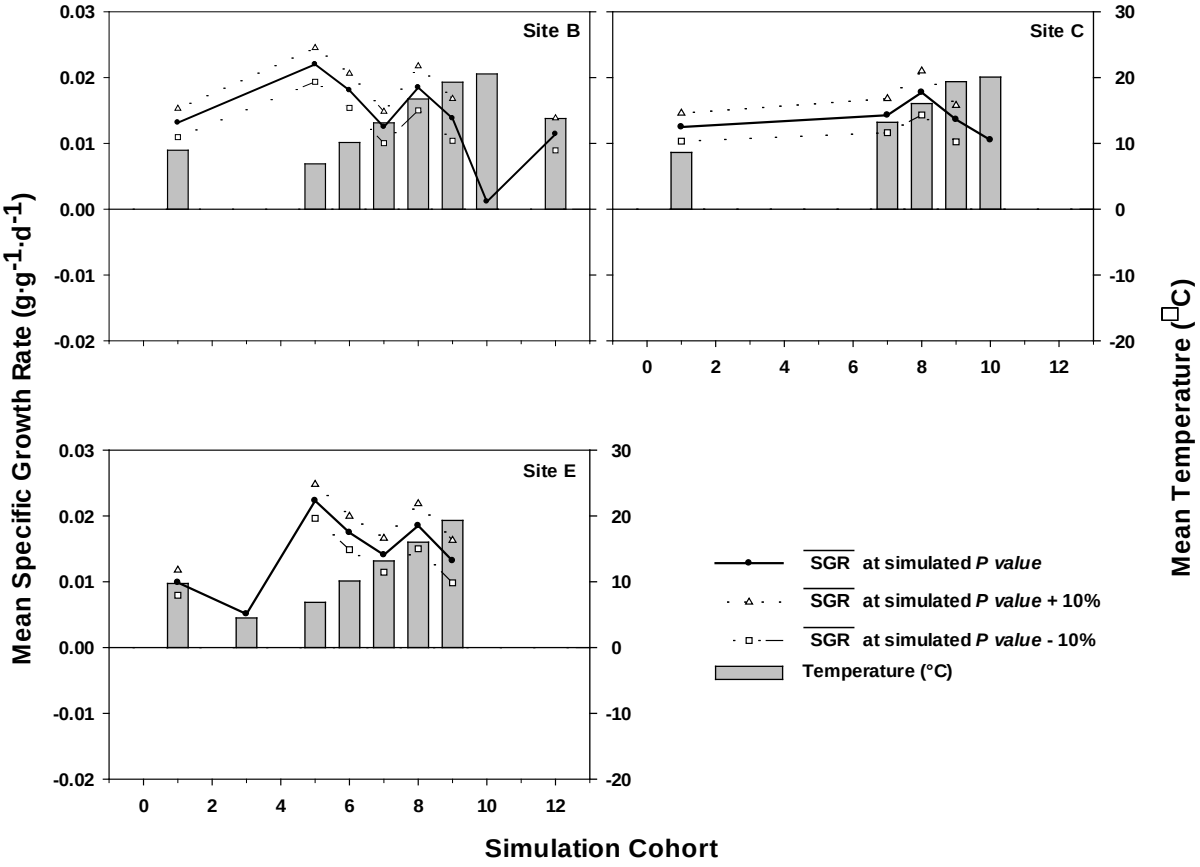

Figure 2.17. Mean predicted specific growth rates ($g \cdot g^{-1} \cdot d^{-1}$) for juvenile Chinook salmon cohorts. Simulations were conducted for each site over the periods specified in Table 2.2, corresponding with the simulation cohorts identified on the x-axis. Simulations were run using proportions of maximum consumption (P value) predicted from initial and final mass estimates (baseline, filled circles), baseline estimates plus 10% (open squares), and baseline estimates minus 10% (open triangles). Baseline values not associated with mean specific growth rates predicted after initial *P values* were perturbed ±10% are for those cohorts where simulated P values exceeded 1.0.

2.2.8.4 Gross Conversion Efficiency

Across sites, the GCE varied considerably among simulation cohorts (mean = 17%±7 s.d.; range = 1%–28%) with the highest and lowest values occurring at site B (1 Mar 2012–24 Apr 2012 and 1 Aug 2012–15 Sept 2012, respectively). Like predicted $\overline{SGR}$, GCE values were always positive (i.e., fish were predicted to gain body mass given the integrated effects of food quality and quantity and temperature dependence). Not surprisingly given substantial within-site variability, GCE values were not statistically different among sites (Kruskal-Wallis rank sum test, $\chi 2 = 0.8513$, $p = 0.6533$; Table 2.6).

Table 2.6. Simulation cohorts, habitat parameters, fish size, bioenergetics model output, and gross conversion efficiency for juvenile Chinook salmon. Asterisks indicate cohorts where a proportion of maximum consumption of 1.00 was assumed for bioenergetics simulations because fitted values exceeded the maximum theoretical P value (i.e., 1.00). Fork lengths without associated error values represent a sample size of one.

Site	Simulation cohort	Residence period	Simulation period	Mean temperature (°C; s.d.)	Mean fork length (mm; s.d.)	Wt_i	Wt_f	*P value*	C	GCE (%)
B	1	8 Nov. 11 - 28 Nov. 11	312 - 332	8.98 (0.91)	96.6 (8.3)	10.4	13.7	0.44	12.8	24
	5	1 Mar. 12 - 24 Apr. 12	426 - 480	6.91 (1.48)	51.7 (2.9)	1.4	4.5	0.61	11.4	28
	6	1 Apr. 12 - 21 May 12	457 - 507	10.15 (2.04)	66.3 (7.2)	3.0	7.4	0.45	17.9	24
	7	1 May 12 - 15 June 12	487 - 532	13.13 (1.22)	92.5 (29.0)	8.5	15.1	0.44	35.1	18
	8	1 June 12 - 30 July 12	518 - 577	16.76 (1.95)	59.0 (6.1)	2.1	6.2	0.58	28.8	14
	9	1 July 12 - 16 Aug. 12	548 - 594	19.32 (1.62)	75.2 (6.8)	5.1	9.6	0.91	46.7	9
	10*	1 Aug. 12 - 15 Sept. 12	579 - 624	20.56 (1.19)	103.0	12.9	22.4	1.00	42.6	1
	12	1 Oct. 12 - 21 Oct. 12	640 - 660	13.78 (1.55)	111.0 (1.4)	16.5	20.9	0.45	22.5	19
C	1	9 Nov. 11 - 29 Nov. 11	313 - 333	8.63 (1.00)	91.2 (10.8)	7.7	10.0	0.42	9.3	23
	7	1 May 12 - 16 June 12	487 - 533	13.22 (1.25)	79.5 (6.4)	5.3	10.4	0.50	29.7	17
	8	1 June 12 - 17 July 12	518 - 564	16.07 (1.52)	63.8 (7.5)	2.6	6.0	0.56	22.3	15
	9	1 July 12 - 16 Aug. 12	548 - 594	19.38 (1.65)	76.5 (5.1)	5.3	10.0	0.94	48.6	9
	10*	1 Aug. 12 - 16 Sept. 12	579 - 625	20.09 (1.77)	83.0	6.3	12.5	1.00	32.5	11
E	1	8 Nov. 11 - 28 Nov. 11	312 - 332	9.73 (1.16)	110.6 (9.4)	13.7	17.0	0.42	16.3	19
	3*	1 Jan. 12 - 21 Jan. 12	366 - 386	4.48 (0.75)	115.0	18.5	23.4	1.00	17.3	11
	5	1 Mar. 12 - 24 Apr. 12	426 - 480	6.87 (1.45)	51.0	1.3	4.4	0.64	11.3	27
	6	1 Apr. 12 - 21 May 12	457 - 507	10.11 (2.04)	69.0 (5.9)	3.4	8.2	0.48	20.7	23
	7	1 May 12 - 16 June 12	487 - 533	13.15 (1.25)	81.0	5.7	10.9	0.46	28.8	18
	8	1 June 12 - 17 July 12	518 - 564	16.00 (1.51)	60.9 (10.8)	2.3	5.4	0.52	19.1	16
	9	1 July 12 - 16 Aug. 12	548 - 594	19.31 (1.64)	79.3 (6.0)	5.9	10.8	0.90	51.0	9

2.3 Discussion

The trends in fish community composition noted at the SRD are similar to those observed during previous years of sampling (Sather et al. 2011, 2012). During the 2012 sampling effort, beach seine catches were dominated by native taxa, but non-native species composed 15 of the 27 species captured. The highest densities of native taxa (excluding salmon) occurred during fall. Threespine stickleback accounted for 90% of the total catch during the sampling period, which accounts for a higher proportion noted during previous years of sampling (Sather et al. 2012).

Three species of salmon, unmarked and marked, were captured at the SRD sites during the October 2011–December 2012 time period. Juvenile steelhead, while rare in beach seine catches at the SRD (Sather et al. 2012, 2011), were not captured via beach seine during 2012. However, these species were captured via boat electrofishing during May and June. Unmarked Chinook salmon were the only species captured during every season, and were the most abundant salmon species captured at the SRD sites. Unlike previous years, the densities of salmon were not highest during spring months at the SRD. This may be an artifact of sampling effort because high flows prevented beach seine sampling efforts during two of the three spring months. Size and timing of salmon captured at the SRD were variable across seasons, indicative of subyearling and yearling life history groups.

Genetic stock groups were characterized by patterns associated with life history attributes as well as geographic patterns (Waples et al. 2004; Narum et al. 2010). However, transfers of hatchery stocks in the Columbia River basin have confounded our ability to definitely link some genetic stock groups with natal sources. Examples include the Spring Creek Group Fall, Upper Columbia Summer/Fall and the Willamette River Spring stock groups (see Sather et al. 2009 and 2011 for additional discussion).

Despite the confounding factors limiting our ability to discern geographic origins of some fish in our samples, the genetic stock compositions presented in this report are similar to results obtained earlier (Sather et al. 2011, 2012). The percentage of Upper Columbia River Summer/Fall juveniles in our unmarked SRD samples collected in 2012 (33%) is consistent with previous years' results (33%–35%). The proportion of Spring Creek Group Tule Fall (39%) was also similar to earlier estimates (31%–35%). The five stock groups contributing minor percentages to the 2012 samples (1%–8%) were also the same in the two previous periods, as were the three stock groups estimated to be absent or nearly absent. In addition, the seasonal shifts in SRD habitat use reported by Sather et al. (2011, 2012) was also apparent in our current results with Spring Creek Group Tule Fall run fish most abundant from February to April and Upper Columbia River Summer/Fall stock group in April to July. While very few Chinook salmon juveniles were analyzed from November and December sampling (n = 16), these included spring run juveniles from the Willamette River stock group which were also identified in previous years' fall collections. These fish are most likely from the nearby Sandy River, which has a spring run stock group with a genetic profile consistent with the sustained introductions of Willamette River stock group. And, as in earlier years, our fall SRD samples also included juveniles assigned to the West Cascade Tributary Spring stock group. These, too, may originate in the Sandy River because spring Chinook salmon in that river show genetic affinities with West Cascade Tributary spring run populations in addition to those in the Upper Willamette River Evolutionarily Significant Unit (Myers et al. 2006). Other potential sources of West Cascade Tributary spring run juveniles that are upstream of the SRD include the Big White and Hood rivers in the Columbia River gorge, as well as the Klickitat River, which is genetically intermediate to the basin's lower river and interior spring Chinook salmon lineages (Narum et al. 2010).

Genetic estimates for LRR unmarked samples collected in 2012 were also similar to those obtained in sampling conducted from 2009 through 2011 (Sather et al. 2011, 2012). The recent samples were largely (68%) West Cascade Tributary Fall stock group, although in lower proportion than in earlier years (75%). While our unmarked samples may include both naturally produced and unmarked hatchery fish (Sather et al. 2009), we also analyzed marked fish sampled at sites in the SRD (n = 77) and LRR (n = 150). As in previous years, the Spring Creek Group Fall and Upper Columbia River Summer/Fall stock groups were the major contributors to the 2012 SRD samples of known hatchery juveniles. However, in earlier years substantial proportions of our SRD marked samples were collected in spring and early summer, whereas in 2012 a much greater proportion (52%) was collected in July. This seasonal shift in samples likely explains lower proportions of Spring Creek Group Fall in 2012 (36% vs 49%–69%) and greater proportions of Upper Columbia River Summer/Fall stock group (56% vs 20%–35%). Seasonal differences in stock compositions were also evident in analyses of marked juveniles collected at LRR sites. Greater proportions of West Cascade Tributary Fall and Upper Columbia River Summer/Fall hatchery fish were identified when samples included summer collections (this study, Sather et al. 2012) than when sampling occurred only in February and May (Sather et al. 2011).

The consistency of the new data with those from previous years suggests that major stock distribution patterns in Columbia River tidal freshwater habitats may remain relatively stable across years. These consistencies include both temporal (seasonal) and spatial (SRD vs LRR) patterns for several different stocks. And while the Chinook salmon juveniles in these habitats are primarily from three fall run stock groups (Upper Columbia River Summer/Fall, Spring Creek Group Tule Fall, and West Cascade Tributary Fall), our samples also consistently include much smaller numbers of spring run fish from both lower river and interior basin sources.

Trends in diet composition and %IRI described by data collected during the current study period were similar to those identified previously (Storch and Sather 2011; Sather et al. 2012), where the gut contents of juvenile Chinook salmon were dominated by dipterans in addition to other large-bodied invertebrates (e.g., amphipods). Comparison of these data across study periods appears to characterize a relatively stable forage base at sites near the SRD, providing opportunity for juvenile Chinook salmon to regularly encounter and consume high-quality prey (see Storch and Sather 2011). However, whether forage opportunity is at times constrained by prey quantity (e.g., via competitive interactions) remains equivocal (but see, Sather et al. 2012).

When present in the diet and/or environment, dipterans commonly were selected against despite constituting large proportions of the gut content biomass. Similar results were uncovered by Storch and Sather (2011) and Sather et al. (2012), and as concluded in these past studies, this may highlight the productivity of the dipteran prey resource at our sites. Unlike this previous work, however, in the current study we found large-bodied amphipods commonly were selected against, indicating a shift in preference from previous years. Yet like dipterans, amphipods periodically accounted for large proportions in the gut content biomass. While it is possible this represents a behavioral change, as was suggested for dipteran prey, these results may at least partially reflect a relative increase in amphipod production in the environment (i.e., selection against amphipods may be an artifact of their abundance in the environment).

General conclusions that can be drawn from bioenergetics output are largely consistent with previous years. During certain sampling episodes—associated temperature extremes—P values were comparatively high, indicating fish were unable to meet putative energetic demands without feeding at rates beyond maximal. During the same periods, in certain cases, predicted growth and GCE appeared to

suffer. This may suggest the amount of energy consumed (i.e., the energetic composition of the diet) was insufficient to mitigate completely for thermal constraints. Nonetheless, during all applicable sampling episodes, growth and GCE values were positive (i.e., fish gained biomass). Thus, despite certain sampling episodes when environmental conditions may constrain fish production, the current forage base and physical habitat at our sites generally appear to be suitable to support juvenile Chinook salmon. Future restoration efforts should seek to maintain current prey production and moderate temperature fluctuations.

3.0 Site-Scale: Julia Butler Hansen National Wildlife Refuge − Post-Construction Assessment of Fish, Habitat, and Tide Gates

Prepared by Jeffrey Johnson, Taylor Brewer, David Teel, and Timothy Whitesel

The primary goal of this study was to assess the effect of habitat restoration on fish, fish communities, and aquatic habitat at JBHNWR. Habitat restoration has focused on replacement of traditional style, side-hinged tide gates with new side-hinged, self-restrained tide gates at sloughs not connected to the Columbia River. This section covers 2012 post-restoration AER at JBHNWR mainland (tide gate retrofits in 2008 and 2009) and Tenasillahe Island (tide gate retrofit in 2007) sites. The study objectives were as follows:

1. Describe the presence and distribution of fish inhabiting mainland and Tenasillahe Island sloughs at JBHNWR and compare them to those observed at reference sloughs.
2. Characterize the habitats of sloughs at JBHNWR and compare them to those observed at reference sloughs.

The JBHNWR consists of island and mainland areas of the lower Columbia River. These areas are managed primarily for the protection of the endangered Columbian white-tailed deer. Islands adjacent to mainland refuge areas are relatively pristine. Sloughs are not diked or controlled by tide gates and have unimpeded connection to surrounding waters and tidal action. Aquatic habitats on the mainland portion of JBHNWR historically included the lower reaches of three tributaries (i.e., Risk Creek, Nelson Creek, and an unnamed creek), wetlands, and eight tidally influenced sloughs to which adult and juvenile salmonids likely had access (NMFS 2008b). Now, the accessibility of slough habitats is largely impeded by dikes and tide gates. Conditions prior to restoration actions reduced tidal influence on sloughs and caused poor habitat conditions for native salmonid species.

To improve fish passage, ingress and egress, in 2003 the USACE replaced a failing culvert and traditional top-hinge wooden tide gate at slough W201+30 with a new culvert and a side-hinge aluminum gate. The gate is equipped with a float and cam system that is designed to hold the gate partially open during incoming tides until the buoyancy of the float rotates the cams and closes the gate. Operation of this culvert and tide gate was compromised by damage to the culvert caused by 2006 winter flooding.

In 2007, the USACE initiated a hydrologic and hydraulic feasibility study to analyze options for modifying existing tide gates to improve flood control, increase fish passage into sloughs and improve slough habitat quality on the refuge (NMFS 2008b). The feasibility study focused on eight sloughs, four with existing tide gates, and four sloughs that are isolated from the Columbia River by dikes without tide gates (Figure 3.1). As a result of this study, the USACE proposed installing tide gates at three sloughs currently blocked by dikes (Hampson, Indian Jack, and Winter) and replacing tide gates in two other sloughs (Brooks and Duck Lake). The replacement gates would be side-hinge aluminum and equipped with a hydraulic arm assembly that controls gate closing. This assembly blocks the gate at a fully open position until the water level within the slough rises to a predetermined elevation at which point the hydraulic arm allows the gate to close. Tide gates were installed in the summer of 2008 and 2009. The gates were designed and installed to allow attenuated tidal cycle but still protect Columbian white-tailed deer habitat.

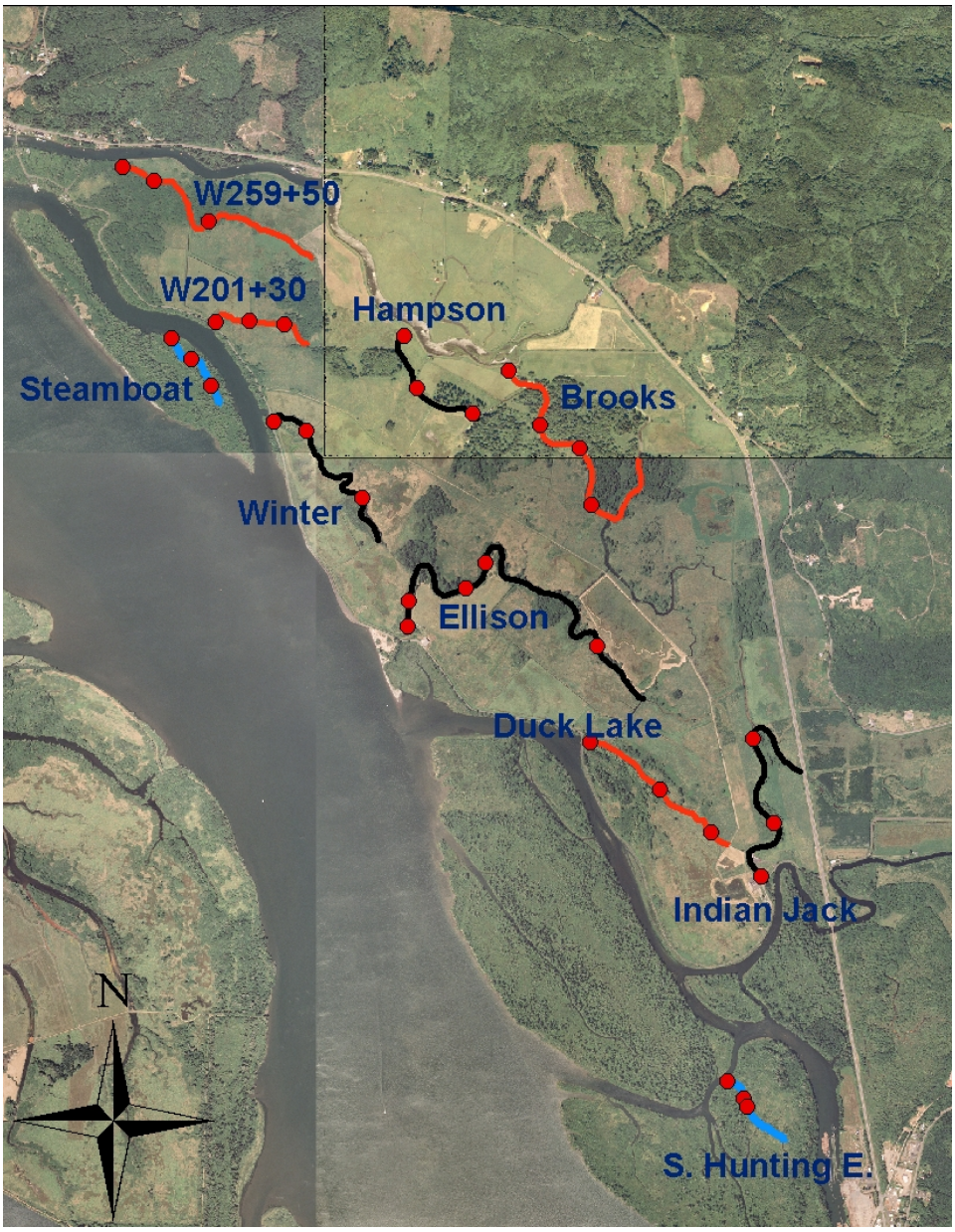

Figure 3.1. Area map of JBHNWR showing the location of sloughs and sample reaches (red circles) surveyed in 2007, 2008, 2010, 2011, and 2012. Black, red, and blue lines indicate closed, gated, and reference sloughs, respectively.

3.1 Methods

The study area, process to identify study sloughs and sample reaches, study design and analysis methods, and field methods and protocols are described in the following sections.

3.1.1 Study Area

3.1.1.1 JBH Mainland

JBHNWR was established in 1972 for the protection and management of endangered Columbian white-tailed deer. The refuge complex contains over 5,600 acres of pastures, forested tidal swamps, brushy woodlots, marshes, and sloughs along the Columbia River in both Washington and Oregon. The mainland portion of JBHNWR ("mainland" JBHNWR) is located near the town of Cathlamet, Washington, on the Columbia at rkm 54.7–57.9. Mainland JBHNWR is bordered by the Columbia River to the west, the Elochoman River to the south, Brooks slough and the town of Skamokawa to the north, and Washington Highway 4 to the east. The refuge has been altered by homesteading, wetland drainage, agricultural production, flood control construction, and grazing by cattle. There are eight sloughs on mainland JBHNWR, which was historically influenced by tides and is currently interconnected by a series of drainage ditches and channels. Until 2009, four of these sloughs were connected to the Columbia River by culverts with tide gates and four were not connected because of flood control levees. The four gated sloughs—Brooks, Duck Lake, W201+30, and W259+50—had tide gates that controlled the discharge of water from the mainland interior. Brooks slough had three 1.5- × 1.5-m, top-hinge aluminum tide gates. Duck Lake had a single 1.8-m diameter, top-hinge steel tide gate. W201+30 has a 1.2-m diameter side-hinge aluminum tide gate equipped with a cam and float system that holds the gate partially open during incoming tide until the float system disengages the cams and allow the gate to close completely. W259+50 has a 1.5- × 1.5-m, top-hinge wooden tide gate. The four closed sloughs—Ellison, Hampson, Indian Jack, and Winter—were not connected to the Columbia River and its side channels because of flood control levees, but they were interconnected with other sloughs on the JBHNWR by drainage ditches. Construction in 2009 installed culverts and the new tide gate design at Hampson and Winter sloughs, replaced one of the three gates at Brooks slough with the new tide gate design, and fixed a heaved culvert at W201+30 that was thought to affect tide gate operation (Figure 3.1).

JBHNWR includes islands that do not have dikes and that are adjacent to the mainland JBHNWR. The Hunting Islands are a group of three islands on the Washington side of the Columbia River immediately downstream of the town of Cathlamet at rkm 54.7. The tidal marsh habitat on South Hunting Island is relatively pristine with no evidence of human habitation or landscape alterations. The slough on the eastern edge of South Hunting Island was selected as a control site (Figure 3.1). Price Island is also part of the JBHNWR. The island is located on the Washington State side of the Columbia River at rkm 56.3. Steamboat slough separates the island from the mainland JBHNWR on the Washington shore. The tidal marsh and tidal spruce swamp habitat remain intact with no apparent evidence of human settlement. There are no water control structures on the island. The large slough on the north (interior) side of the island was selected as a control site on Price Island (Steamboat slough) (Figure 3.1).

3.1.1.2 Tenasillahe and Welch Islands

Tenasillahe Island is an 809-hectare island located in the lower Columbia River at rkm 56 (Figure 3.2). Much of the tidal marsh habitat historically occurring at Tenasillahe Island was altered by the construction of dikes around the island during the course of the last century. Aquatic habitat on the island currently consists primarily of two interior sloughs connected to the Columbia River via tide gates. Until the summer of 2007, the aquatic habitat on the island consisted primarily of a network of interior

sloughs connected to the Columbia River via steel top-hinged tide gates. These gates were designed to close when the river water elevation reaches the slough water elevation. When gates were closed, water flow into sloughs was limited to that which leaks through the gates. Tide gates limited fish passage into or out of the sloughs to times when water was flowing out of the slough. Connection of the smaller of the two sloughs to the Columbia River was controlled by a single top-hinge steel tide gate, but now connection of the larger of the sloughs to the Columbia River is controlled by three side-hinge aluminum tide gates equipped with a manually controlled fish orifice. These gates, installed in 2007, replaced three top-hinge steel tide gates.

Welch Island is part of the Lewis and Clark National Wildlife Refuge (also managed by USFWS), which was established in 1972. Welch Island is a 429-ha island located in the lower Columbia River at rkm 55, adjacent to and just downstream of Tenasillahe Island (Figure 3.2). The natural tidal marsh habitat on Welch Island is relatively pristine. We have not found any evidence that Welch Island was settled by humans. Sloughs are not diked or controlled by tide gates and have unimpeded connection to surrounding waters and tidal action.

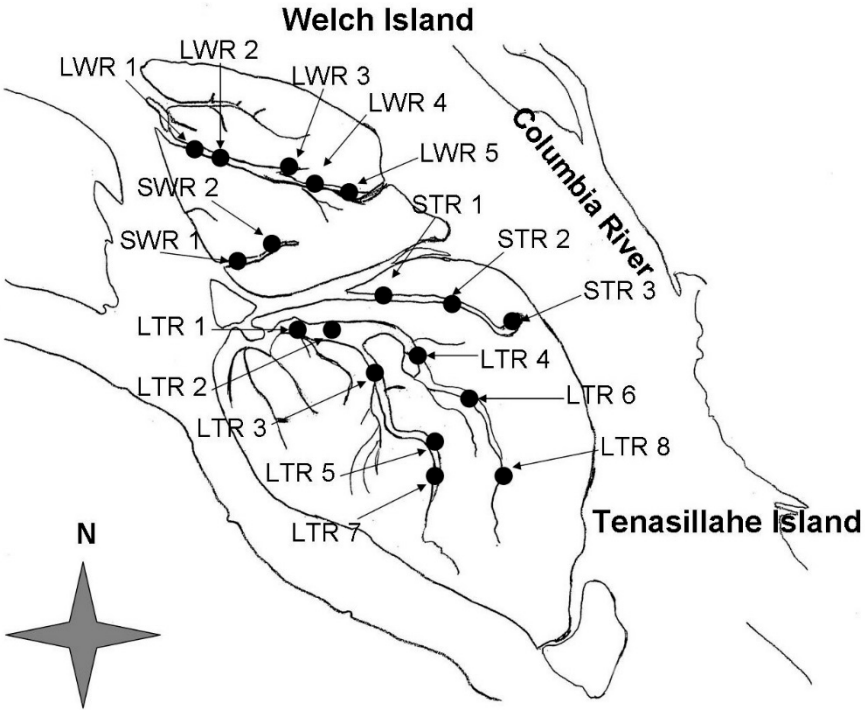

Figure 3.2. Area map of Tenasillahe Island and Welch Island showing locations of reference sloughs (large Welch slough, small Welsh slough), treatment sloughs (Tenasillahe slough, small Tenasillahe slough), and sample reaches within sloughs.

3.1.2 Identification of Study Sloughs and Sample Reaches

All sloughs proposed for restoration actions were included in this study. Treatment sloughs included three sloughs with traditional style tide gates and three closed sloughs on mainland JBHNWR. Two control sloughs, W259+50 and Ellison, were to receive no modifications during the study. Reference sloughs selected for this study showed no evidence of human impact, no water control, and were within

2 km of the treatment sloughs. One natural (unmodified) slough from Price Island (Steamboat slough) and one from South Hunting Island (S. Hunting E.) were designated as reference sloughs (Figure 3.1). All treatment, control, and reference sloughs are located within a 2-km reach of the Columbia River on the Washington side of the shipping channel and therefore, likely experience the same pool of migrating fish. Although the inclusion of unaffected, mainland control sloughs would have been preferred for this study, none were available within the vicinity (within 2 km) of the treatment sloughs. Therefore, we selected control sloughs that experience full tidal influence and would likely represent conditions that treatment sloughs would approach without tide gate influences. In addition, Ellison slough (closed) and W259+50 slough (gated) will not be modified during this project, so they function as additional controls.

Sample reach selection was designed to ensure random and spatially balanced data collection representing at least 10% of the total slough length. Each treatment and reference slough was divided into 50-m sample reaches. If 10% of these reaches was less than two reaches, then the slough was split into 25-m reaches. The sample reach closest to the mouth, tide gate, or historic connection to the Columbia River was sampled in each slough. Additional sample reaches (within each slough) were selected using a random, spatially balanced approach to ensure that various habitats and conditions were represented (see Stevens and Olsen 2004). Three 25-m sample reaches were established in W201+30 and Hampson sloughs; three 50-m sample reaches were established in Indian Jack, Duck Lake, W259+50, and Winter sloughs; four 50-m reaches were established in Brooks slough; and five 50-m reaches were established in Ellison slough (Figure 3.1). In reference sloughs, three 25-m sample reaches were established in S. Hunting E. and Steamboat sloughs (Figure 3.1). The result was that a minimum of 10% of slough length was represented and at least three reaches were sampled in each slough. Sampling effort in 2007 and 2008 (pre-construction) and 2010 (post-construction) focused on the same sets of reaches.

For Tenasillahe and Welch Island sloughs, sample reaches within each slough were randomly selected using a random, spatially balanced approach to ensure that various habitats and conditions were represented (Poirier et al. 2005). Eight 50-m sample reaches were established in LTS (LTS) and five 25-m reaches were established in large Welch slough (LWS, Figure 3.2). These reaches were sampled from 2005 through 2009 during the original restoration assessment. Efforts in 2012 focused on reaches closest to the confluence with the Columbia River.

3.1.3 Study Design and Analysis

Our study design is based upon comparing fish community and habitat conditions in treatment sloughs to reference sloughs and control sloughs before and after treatment. We selected two reference sloughs and two control sloughs for the mainland and one reference slough for the Tenasillahe Island treatment slough. One mainland control, W259+50, is a gated slough and the other, Ellison, is a closed slough. We refer to W259+50 as a “positive control” and Ellison as a “negative control.” The expectation is that conditions at closed sloughs that receive a new tide gate will move toward those of the positive control and further away from the negative control. W259+50 was subsequently removed from the sampling design due to its modification through dredging and the installation of a tide gate during the study. Selection of sloughs and slough reaches are explained above. The reference sloughs are considered the ideal condition and are expected to be independent of treatments on the mainland or Tenasillahe Island. Conditions in the reference sloughs are assumed to reflect natural or system-wide variation in estuary quality. We would expect conditions at treatment sloughs to trend toward that at reference sloughs but fall short of ideal conditions. The difference between conditions at reference

sloughs and treatment sloughs post-construction might reflect the extent that the new tide gates allow sloughs to reach ideal conditions (e.g., what level of restoration has occurred).

3.1.4 Field Methods and Protocols

3.1.4.1 Sampling Schedule

To minimize any spatial or temporal bias, the order in which reaches were sampled was randomized. Sampling effort was distributed evenly throughout the field season. This sampling regime was used to ensure the various habitats and conditions present within each slough were represented, as well as to capture the seasonal variation and changes in fish community composition and distribution.

3.1.4.2 Fish Community and Distribution

Beach seines (15-m × 1.8-m with 0.6-cm mesh) were the primary fish sampling method used during the 2012 field season. Each seine was held on shore and either walked by foot or towed into the channel by boat making a sweep along the shore. The size of the encircled area was estimated. In general, five non-overlapping seine hauls were performed in each sample reach in 2012. When excess fine substrate coupled with warm air temperatures put fish health at risk, fewer than five seine hauls were performed.

All captured fish were placed in an aerated live well, identified, enumerated, and released. In addition, fork length and weight were recorded for most fish other than threespine stickleback. Individual fish were anaesthetized in a 0.3 g/L solution of MS-222, measured, weighed, and examined for external marks. Prior to release, fish were allowed to recover in an aerated live well for 15 to 30 min.

3.1.4.3 Habitat Characterization

Water temperature was recorded hourly in the lowest reach of each slough using Onset StowAway Tidbits. Recorders were deployed in April 2012 in all mainland JBHNWR treatment, reference, and control sloughs. The 7-DADMs for water temperature (a rolling average of seven consecutive daily maximum water temperatures recorded within a stream) were calculated from the temperature logger data. The 7-DADM levels were compared to threshold criteria above which juvenile salmonids exhibit sub-lethal effects (Richter and Kolmes 2005; EPA 2003).

3.2 Results

The results include data on fish community composition, juvenile salmon, and habitat characterization (water temperature).

3.2.1 Fish Community Composition

3.2.1.1 JBH Mainland

One thousand eighty-four seine hauls were performed in 20 sample units between May 14, 2012 and February 12, 2013. A total of 26,004 fish representing 21 taxa were captured in three mainland sloughs

(Duck Lake, Indian Jack, Winter), one control slough (Ellison), and two reference sloughs (South Hunting and Steamboat; see Table 3.1). Nine of the 11 (82%) species captured in reference sloughs Steamboat and South Hunting were native. Nine of 18 (50%) species captured in Duck Lake, Indian Jack, and Winter sloughs (treatment sloughs) were native. Seven of 15 (47%) species captured in Ellison slough (control) were native. Threespine stickleback represented the most prevalent species in all sloughs.

Table 3.1. Fish species and numbers captured from JBHNWR mainland sloughs. Captures from all months are combined.

	Duck	Ellison	Indian Jack	South Hunting	Steamboat	Winter	Total
Black crappie						1	1
Bluegill	1	66				1	68
Chinook salmon	166	10	204	75	37	195	684
Chum salmon				3			3
Coho salmon	31		37	38	8	12	126
Common carp	1	3	20			3	27
E. banded killifish	129	146	81	2	4	220	582
Largemouth bass	3	164				11	178
Northern pikeminnow	122	17	46	36	6	96	323
Peamouth	104	45	36	101	23	74	383
Pumpkinseed	6	11	7			6	30
Redside shiner	33	72	20			26	151
Sculpin	41	2	87	97	38	54	319
Starry flounder	1				1		2
Sucker	1	7	15	8		7	38
Threespine stickleback	5,881	1,498	3,192	4,035	2,538	5,679	22,823
Western brook lamprey			1				1
White crappie	1	102	3				106
Yellow bullhead	2	16					18
Yellow perch	45	64	4	2	2	21	138
Grand Total							**26,004**

3.2.1.2 Tenasillahe and Welch Islands

One-hundred twenty-six seine hauls were performed in four sample units between July 18, 2012 and February 12, 2013. A total of 10,930 fish representing 18 taxa were captured in two treatment sloughs (LTS and small Tenasillahe slough [STS]), and two reference sloughs (LWS and small Welch slough [SWS]; see Table 3.2). Eight of the 10 (80%) species captured in reference were native. Five of 13 (38%) species captured in treatment sloughs were native. Threespine stickleback was the most prevalent species in all sloughs except STS where pumpkinseed was the most prevalent.

Table 3.2. Fish species and numbers captured from Tenasillahe and Welch Island sloughs. Captures from all months are combined.

	Large Ten	Large Welch	Small Ten	Small Welch	Grand Total
Bluegill			12		12
Chinook salmon		116		5	121
Coho salmon		1			1
Common carp	33				33
Killifish	39	44	62	6	151
Largemouth bass	63		22		85
Northern pikeminnow	1		1		2
Peamouth	7	12		85	104
Pumpkinseed	1		173		174
Redside shiner			25		25
Sculpin	46	23	2	9	80
Starry flounder		138		2	140
Sucker		15		3	18
Top minnow			2		2
Threespine stickleback	198	6,980	28	2,741	9,947
Unknown sunfish			14		14
Western brook lamprey		1			1
Yellow bullhead			12		12
Yellow perch	1	4	3		8
Grand Total					**10,930**

3.2.2 Juvenile Salmon

3.2.2.1 JBH Mainland

Juvenile salmon were captured throughout the mainland JBHNWR. They were captured in every treatment, control, and reference slough. Salmonid species captured included juvenile Chinook, coho, and chum. Juvenile salmonids were captured in all sample reaches of all treatment, reference, and control sloughs. Salmonids were captured in all months sampled except September in treatment sloughs and August, September, and December in reference sloughs. Salmonids were captured in Ellison slough (control) only in May, June, and July (Figure 3.3).

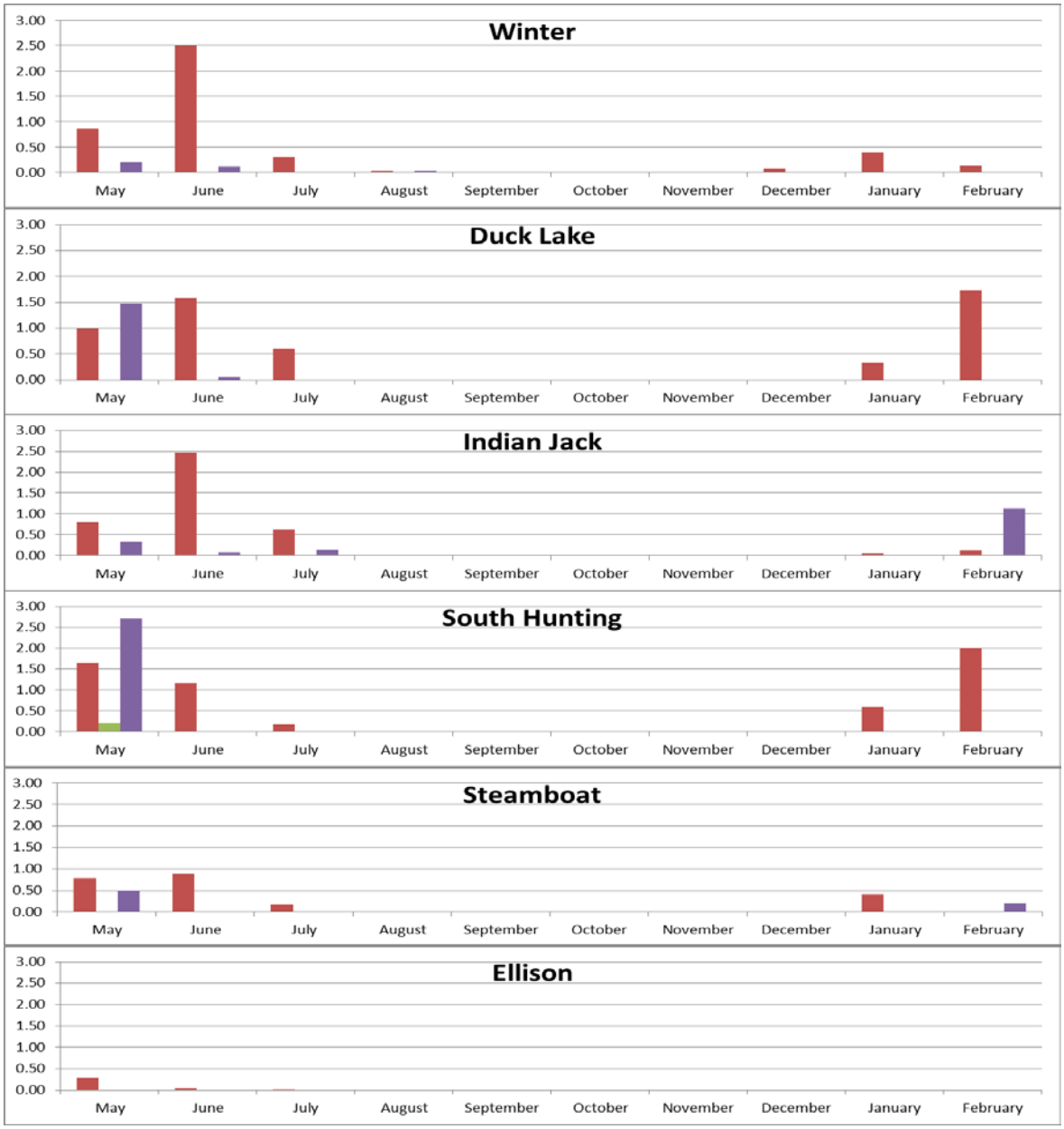

Figure 3.3. Number of juvenile Chinook (red), chum (green), or coho (purple) salmon per seine pull by month from May 2012 to February 2013 from mainland JBHNWR treatment, reference, and control sloughs. No sampling took place October and November 2012.

Genetic Stock Identification for Chinook Salmon: JBH Mainland

A total of 190 Chinook salmon sampled at JBH mainland sites were genotyped at 7 or more of the 13 microsatellite loci and used in genetic stock identification analysis. Samples were collected in May in 2010 and 2011 and in July, August, and December 2012. Stock composition estimates are presented in Table 3.3. Most of the fish were from the West Cascade Tributary Fall (65%) and Spring Creek Group Tule Fall (26%) stock groups. The West Cascade Tributary Spring (4%) and Upper Columbia Summer/Fall (2%) stock groups were also estimated to contribute to the JBH samples. Small proportions

(<2%) were also estimated for Deschutes River Fall, Willamette River Spring, and Rogue River stock groups, although the lower 95% confidence intervals for these three stocks were zero. Individual fish genetic stock identifications of JBH samples were grouped by survey month and site type (reference and treatment) and are presented in Figure 3.4. Individuals from four stock groups (West Cascade Tributary Fall, Spring Creek Group Tule Fall, West Cascade Tributary Spring, and Upper Columbia Summer/Fall) were identified in both the reference and treatment sites. Additional stock groups were estimated to contribute to samples from treatment sites but not from reference sites.

This result may be due to the number of samples we analyzed from the two site types (n = 164 vs n = 26). Moreover, the Willamette River Spring and Rogue River contributions consisted of single fish and the Deschutes Fall assignments (n = 4) had relatively low assignment probabilities (<0.82). Our analysis included six unmarked fish collected at JBH in December and all of these individuals were assigned with high probabilities (P >0.95). The smallest individuals (FL < 45 mm) were estimated to be from West Cascade Tributary Spring (N = 2), West Cascade Tributary Fall (n = 1), and Spring Creek Group Tule Fall (n = 1) stock groups. Two larger fish in the December samples were from the West Cascade Tributary Fall (FL = 125 mm) and Spring Creek Group Tule Fall (FL = 116 mm) stock groups.

Table 3.3. Estimated percentage genetic stock composition and 95% confidence intervals of 190 juvenile Chinook salmon at JBH sites from May 2010 through December 2012.

Genetic Stock Group	Estimated Contribution (%)	95% Confidence Interval	
Upper Columbia River Summer/Fall	2.3	1.5	7.3
West Cascade Tributary Fall	64.8	52.4	72.2
Spring Creek Group Tule Fall	26.1	13.6	30.6
Snake River Fall	0.0	0.0	1.9
Willamette River Spring	0.5	0.0	3.3
Deschutes River Fall	1.6	0.0	3.5
West Cascade Tributary Spring	4.0	2.9	16.9
Mid and Upper Columbia River Spring	0.0	0.0	0.0
Snake River Spring	0.0	0.0	0.0
Rogue River	0.8	0.0	2.2

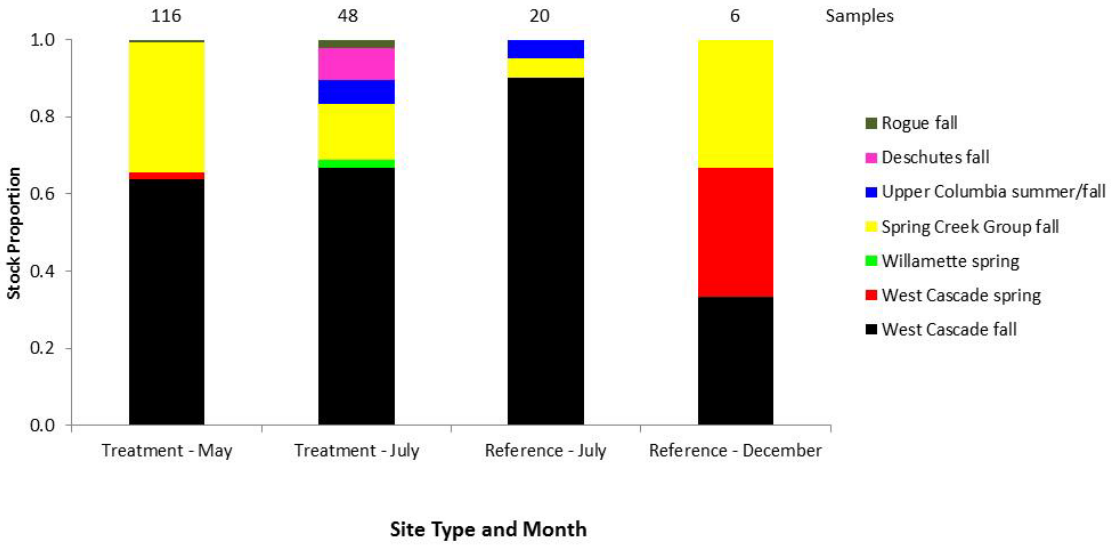

Figure 3.4. Estimated stock proportions, sample sizes, month of sampling for juvenile Chinook salmon at JBH reference and treatment sites from 2010 through 2012. July proportions include samples collected at reference (N = 9) and treatment (N = 1) sites in August. Snake River Spring, Snake River Fall, and Mid and Upper Columbia Spring stock groups were not estimated to contribute to the samples.

3.2.2.2 Tenasillahe and Welch Islands

No salmonids were captured in Tenasillahe Island treatment sloughs, but they were captured in both reference sloughs. They were captured in large Welch slough in all months samples except September and in small Welch slough in July and January. Juvenile salmonid species captured include juvenile Chinook and coho salmon (Figure 3.5).

Figure 3.5. Number of juvenile Chinook (red) or coho (green) salmon per seine pull by month from July 2012 to February 2013 from large Welch and small Welch reference sloughs. No sampling took place October and November 2012.

3.2.3 Habitat Characterization – Water Temperature

Water temperature consistently exceeded 18°C 7-DADM during July and August in all sloughs (Table 3.4). The earliest month of this temperature was May in Duck Lake (treatment) and Steamboat (reference) sloughs. The latest month was October in Winter treatment and both reference sloughs. The highest cumulative days above 18°C 7-DADM was 107 d in the treatment slough Duck and the lowest (65 d) was in treatment slough Indian. Both South Hunting and Steamboat reference sloughs showed similar cumulative days exceeding threshold temperature (90 and 94 d). Interestingly, the control slough Ellison had the second fewest days exceeding threshold temperature (67 d).

Table 3.4. Days per month that 7-DADM water temperatures exceeded 18°C for JBHNWR mainland treatment (Winter, Duck, Indian), reference (South Hunting, Steamboat), and control (Ellison) sloughs in 2012.

	Winter	Duck	Indian	South Hunting	Steamboat	Ellison
January	0	0	0	0	0	0
February	0	0	0	0	0	0
March	0	0	0	0	0	0
April	0	0	0	0	0	0
May	0	6	0	0	1	0
June	2	13	0	0	4	0
July	28	31	21	26	27	24
August	31	31	30	30	31	31
September	30	26	14	30	29	12
October	6	0	0	4	2	0
November	0	0	0	0	0	0
December	0	0	0	0	0	0
Total	**97**	**107**	**65**	**90**	**94**	**67**

3.3 Discussion

Our ability to witness changes in fish community and salmon densities are limited by the high variance among fish collections. Salmon numbers are relatively low in seine and trap collections. It is not uncommon to capture zero Chinook salmon in multiple seine pulls, then subsequently capture several. In addition, chum and coho salmon captures were rarer than Chinook salmon captures. It is logical that presence data are presented with high confidence and density data may have such high variance as to be unusable to witness the level of changes that may occur.

Inter-annual and month-to-month variation in weather makes meaningful habitat comparisons difficult on the limited temporal scale of this study. As with salmon density, high inter-annual variance makes witnessing meaningful changes difficult on the temporal scale of this study. Two years of pre-restoration data and two years of post-restoration data are not sufficient to draw conclusions about temperature changes.

Although we have found an increase in the presence and distribution of juvenile salmonids in JBH refuge sloughs since tide gate retrofitting, we do not know the survival rate or physical condition of these fish or the duration of rearing within the refuge sloughs. From other work at Tenasillahe Island, we found that some juvenile Chinook salmon survive months (during summer) and with high growth rates within refuge sloughs even with high water temperatures (7-DADM >18°C). Better information about summer use (duration, growth, prey availability) will allow us to understand juvenile salmon life history and habitat limitation here and throughout the LCRE.

The large proportion of fish from the West Cascade Tributary Fall stock group (65%) in the JBH genetic samples of juvenile Chinook salmon is similar to our results from LRR sites. We also observed a substantial proportion of Spring Creek Group Tule Fall fish at JBH (26%). Although both of

these stocks are produced in rivers and hatcheries well upstream of JBH, nearby sources exist as well. The most proximate potential source to JBH is the Elochoman River, which is a component of the Spring Creek Group Tule Fall genetic stock (Teel et al. 2009). Although genetic samples of JBH juveniles were limited, these two stock groups as well as small proportions of the Upper Columbia Summer/Fall and West Cascade Tributary Spring stock group were identified in both reference and treatment sites. Overall, these data provide the initial documentation of multiple genetic stocks occupying JBH habitats.

4.0 Landscape-Scale: Baseline Characterization of Juvenile Salmon Density

Prepared by Nichole Sather, Gary Johnson, and David Teel

Juvenile salmon are typically sampled locally (at the site-scale; e.g., Roegner et al. 2010) or at multiple sites over a broad area (at the landscape scale; e.g., Roegner et al. 2012). Data are reduced and summarized accordingly; however, juvenile salmon density has not estimated across a landscape in the LCRE. This study is the first to estimate juvenile salmon density at the landscape scale using a statistically robust sampling design. Landscape density estimates have potential to be applied in baseline characterizations (reference conditions) for action effectiveness studies.

An evaluation of migratory patterns and juvenile salmon density across the landscape of shallow water habitats of the LCRE provides a means for measuring the response of juvenile salmon to restoration actions. Our 2012 research to estimate juvenile salmon density in shallow water habitats between St. Helens and Longview (rkm 110–141) provides a baseline for evaluating response to restoration actions within the vicinity, which are expected to increase habitat availability. The anticipated response to an increase in habitats is measured by examining change in salmon density across specific habitats at seasonal scales within a given year and at landscape scales across multiple years. However, detailed statistical methods to apply landscape scale data on juvenile salmon density as a reference or baseline for purpose of site-scale AER remain to be developed.

The study objectives were as follows:

1. Estimate juvenile Chinook salmon density (mean and variance) at the landscape scale for the sampling region as a whole and by habitat type.
2. Estimate the genetic stock identities for a subset of Chinook salmon sampled for landscape density.

4.1 Methods

In the LRR, a random stratified sampling design was used to estimate fish density. Fifteen sites were randomly sampled seasonally across three habitat strata (main channel, off channel, and wetland channel; Figure 4.1) within a rotational panel design (Table 4.1). Details pertaining to site selection criteria are described by Sather et al. (2011). The mean and variance of unmarked Chinook salmon density across habitat strata and the LRR landscape were estimated using statistical approaches outlined by in Sather et al. (2012).

Table 4.1. Depiction of the sampling design for the landscape scale baseline characterization of juvenile Chinook salmon density. Different sampling sites are designated by letters. An "x" means the site was sampled. The table shows the shift in the rotational panel design occurring from spring 2012 to summer 2012.

		Main Channel Sites							Off Channel Sites							Wetland Channel Sites						
Year	Season	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U
2011	Summer	x	x	x	x	x			x	x	x	x	x			x	x	x	x	x		
2011	Fall	x	x	x	x	x			x	x	x	x	x			x	x	x	x	x		
2012	Winter	x	x	x	x	x			x	x	x	x	x			x	x	x	x	x		
2012	Spring	x	x	x	x	x			x	x	x	x	x			x	x	x	x	x		
2012	Summer			x	x	x	x	x			x	x	x	x	x			x	x	x	x	x
2012	Fall			x	x	x	x	x			x	x	x	x	x			x	x	x	x	x

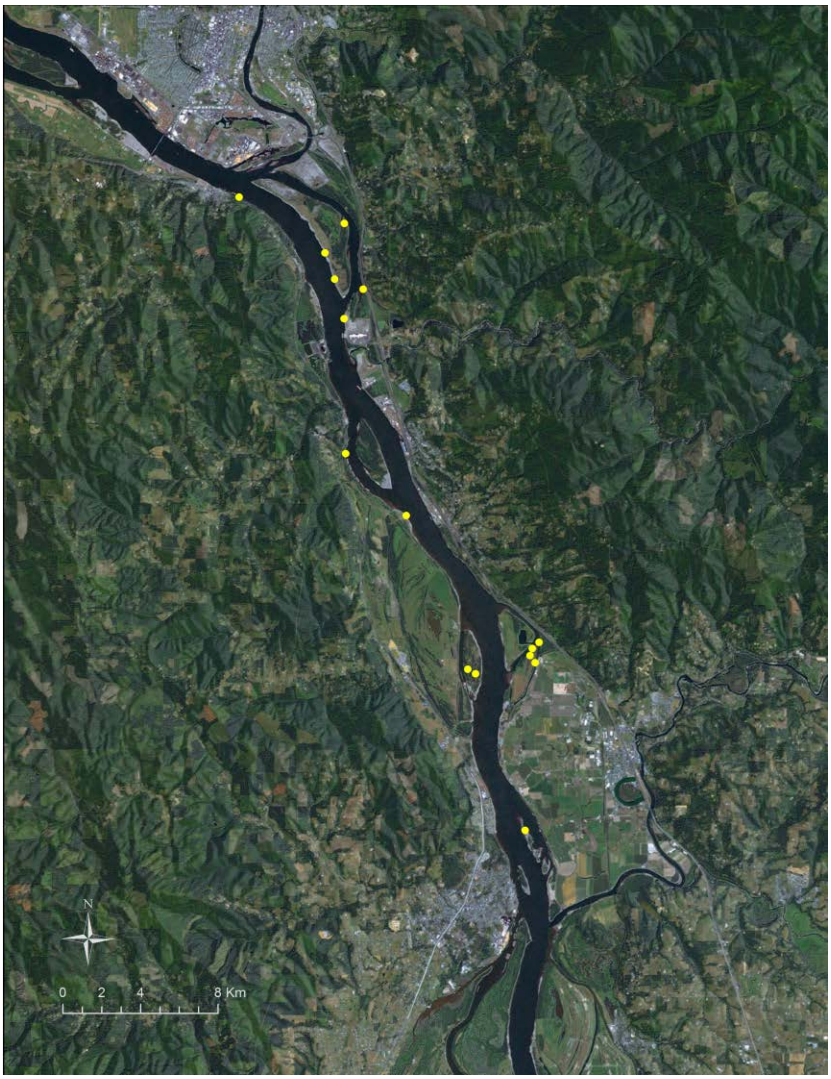

Figure 4.1. Sites sampled in the LRR (rkm 110–141) during 2012. See Figure 2.1 for the location of the LLR in the LCRE.

4.2 Results

Results are presented for unmarked salmon density and genetic stock identification across the LLR landscape.

4.2.1 LRR Unmarked Chinook Salmon Density

Densities for unmarked Chinook salmon across the LRR landscape were greatest during the winter and spring of 2012. Lowest densities occurred during fall 2011 and 2012 (Figure 4.2). Among habitat strata, densities varied through time (Figure 4.3). Except for during winter 2012, densities for unmarked Chinook salmon were lowest in the wetland habitat compared to main channel and off-channel habitats. Across seasons and habitats, main channel densities of unmarked Chinook salmon exceeded those measured in other habitats during summer and fall.

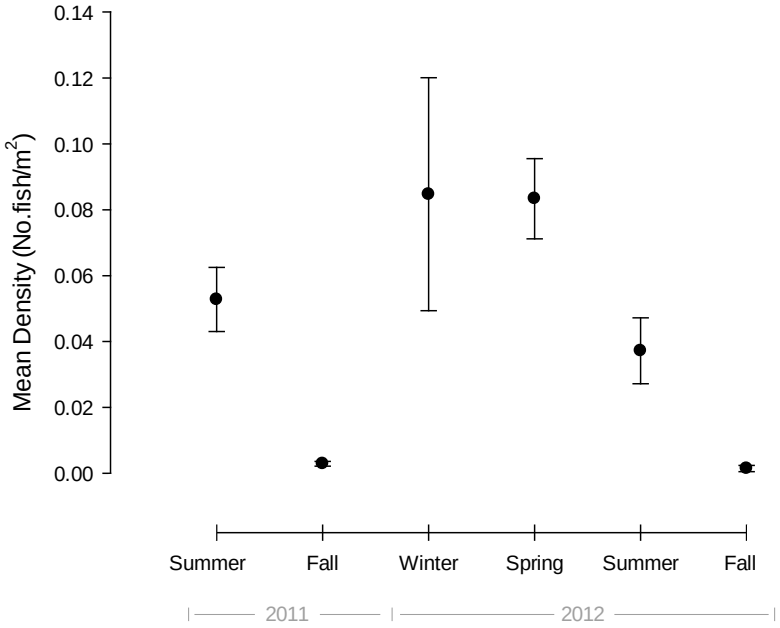

Figure 4.2. Mean landscape-scale density of unmarked Chinook salmon sampled at the LRR from summer 2011 to all 2012. Error bars depict the standard error of the mean.

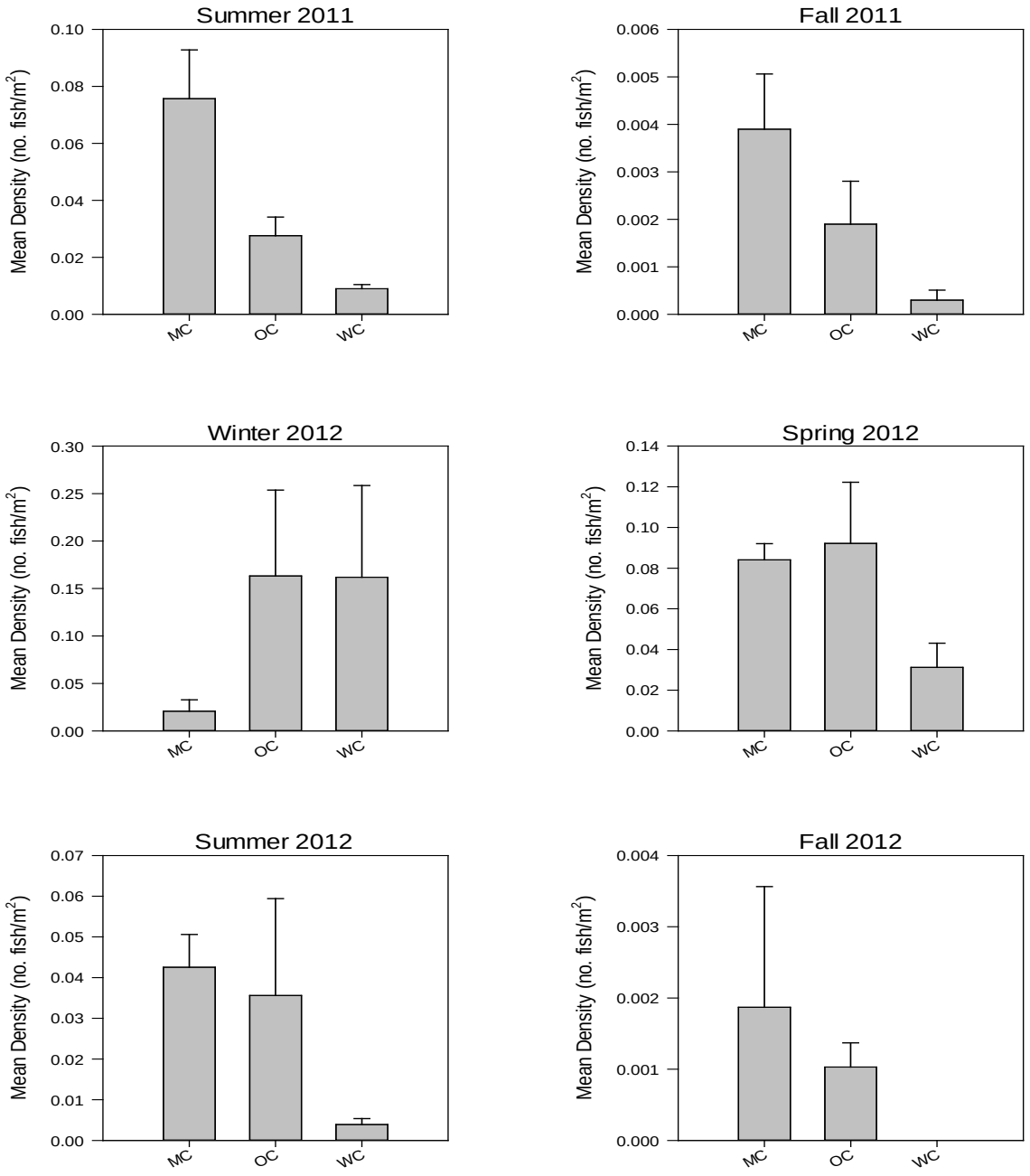

Figure 4.3. Mean density of unmarked Chinook salmon sampled in main-channel (MC), off-channel (OC), and wetland-channel (WC) habitats at the LRR from summer 2011 to fall 2012. Error bars depict the standard error of the mean.

4.2.2 LRR Genetic Stock Identification for Chinook Salmon

Estimated stock proportions of unmarked Chinook salmon sampled in the LRR (n = 569) are reported in Table 4.2. Most fish were estimated to be from the West Cascade Tributary Fall stock group (68%). Upper Columbia Summer/Fall stock group composed an estimated 20% of the samples. Much smaller proportions were estimated for the West Cascade Tributary Spring, Spring Creek Group Tule Fall, and

Deschutes River Fall stock groups (2%−4%). The largest proportions of fish in the sample of marked Chinook salmon from the LRR (n = 150) were from the West Cascade Tributary Fall (73%), Spring Creek Group Tule Fall (12%) and Upper Columbia Summer/Fall (9%) stock groups (Table 4.3). Smaller contributions were estimated for the West Cascade Tributary Spring (5%) and Snake River Fall (1%) stock groups. Individual fish genetic stock assignments of samples of unmarked LRR fish were grouped by survey month and are presented in Figure 4.4. Genetics sample sizes of the surveys ranged from 18 to 241 individuals. West Cascade Tributary Fall Chinook salmon were the major contributors in all five survey months. The largest proportions of Upper Columbia Summer/Fall juveniles were estimated in June and July, the months with our greatest number of LRR genetic samples.

Table 4.2. Estimated percentage genetic stock composition and 95% confidence intervals of 569 unmarked juvenile Chinook salmon sampled at LRR sites from November 2011 through November 2012.

Genetic Stock Group	Estimated Contribution (%)	95% Confidence Interval	
Upper Columbia River Summer/Fall	19.5	15.0	24.1
West Cascade Tributary Fall	68.2	57.8	68.8
Spring Creek Group Tule Fall	2.1	0.9	4.6
Snake River Fall	3.1	1.3	6.5
Willamette River Spring	0.5	0.0	1.3
Deschutes River Fall	2.3	0.4	4.9
West Cascade Tributary Spring	4.4	3.5	9.1
Mid and Upper Columbia River Spring	0.0	0.0	0.0
Snake River Spring	0.0	0.0	0.0
Rogue River	0.0	0.0	0.6

Table 4.3. Estimated percentage genetic stock composition and 95% confidence intervals of 150 marked juvenile Chinook salmon sampled at LRR sites from November 2011 through November 2012.

Genetic Stock Group	Estimated Contribution (%)	95% Confidence Interval	
Upper Columbia River Summer/Fall	8.9	4.0	15.4
West Cascade Tributary Fall	72.8	56.6	78.6
Spring Creek Group Tule Fall	12.0	4.2	16.3
Snake River Fall	1.3	0.0	6.5
Willamette River Spring	0.0	0.0	0.5
Deschutes River Fall	0.0	0.0	2.1
West Cascade Tributary Spring	5.2	0.8	15.6
Mid and Upper Columbia River Spring	0.0	0.0	0.0
Snake River Spring	0.0	0.0	0.0
Rogue River	0.0	0.0	0.9

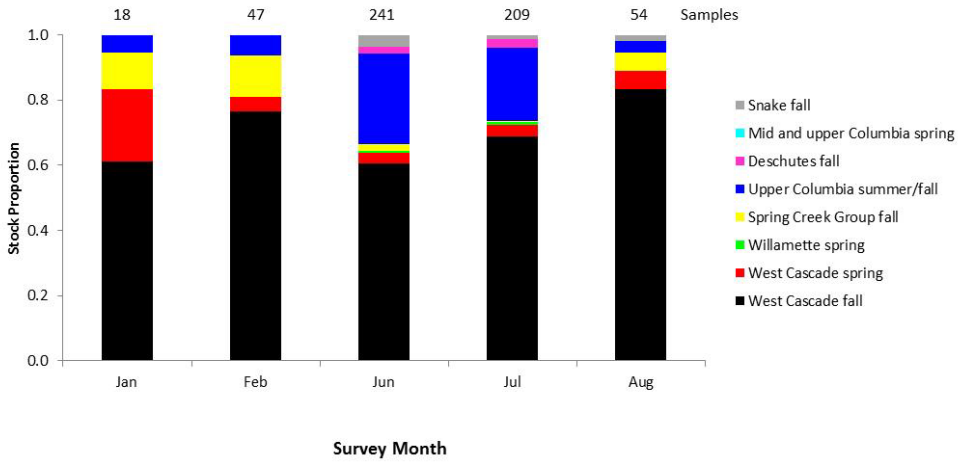

Figure 4.4. Estimated stock proportions, sample sizes, and month of sampling of unmarked juvenile Chinook salmon at LRR sites during 2012. November proportions include samples collected in 2011. Snake River Spring and Rogue River Fall stocks were not estimated to contribute to the samples.

4.3 Discussion

Our genetic results from landscape-scale sampling revealed distinct juvenile Chinook salmon stock compositions between regions (LRR compared to SRD). Much of the overall difference may be explained by the proximity of each region to the points of entrance into the LCRE for the different genetic stock groups. For example, unmarked fish sampled at our LRR sites were largely juveniles from West Cascade fall and spring run stock groups, whose major natural spawning populations are in adjacent tributaries such as the Lewis, Kalama, and Cowlitz rivers. In contrast, samples from the SRD, which is upstream of much of the West Cascade fall production, had much smaller proportions of the stock, likely juveniles originating in the nearby Sandy and Washougal rivers. SRD fish were predominately Spring Creek fall stock group in winter and spring with increasing proportions of Upper Columbia Summer/Fall fish in summer. Natural production of both of these stocks occurs relatively near our SRD sites in areas downstream of Bonneville Dam (e.g., in main stem habitats near Ives and Pierce islands). In addition, fish from the two stocks are also produced in nearby tributaries to the Columbia River Gorge and therefore enter the estuary via Bonneville Dam as do more distant Upper Columbia Summer/Fall fish from major production areas further upstream. While the contrasting stock proportions in the LRR and SRD regions highlight the use of nearshore estuary habitats by juveniles from nearby natal sources, shifts in seasonal composition patterns, and the overall stock diversity in both regions indicate that fish from sources further upstream also occupy these habitats during their downstream migrations.

One of the difficulties in action effectiveness research attempting to statistically compare ecosystem responses at the restoration (impact) and reference sites is finding a suitable reference site in the vicinity of the restoration site (Thom et al. 2013). A before-after-reference-impact design for action effectiveness research requires being able to compare to some reference condition. In lieu of a formal reference site, juvenile salmon densities at the restoration site could be compared to the landscape density estimates as a reference. The landscape density estimates we produced have potential to be applied in baseline characterizations (reference conditions) for action effectiveness studies in the landscape we sampled. Detailed statistical procedures would need to be developed, but the concept has promise.

5.0 Landscape-Scale: Residence Time

Prepared by Gary Johnson, Amanda Bryson, and Nichole Sather

From February through April 2012, we used acoustic telemetry to estimate the residence time of 95- to 190-mm Chinook salmon that were present during winter months in the tidal freshwater portion of the LCRE.[1] Our particular study area was in Carroll's Channel behind Cottonwood Island (~rkm 112) (Figure 5.1).

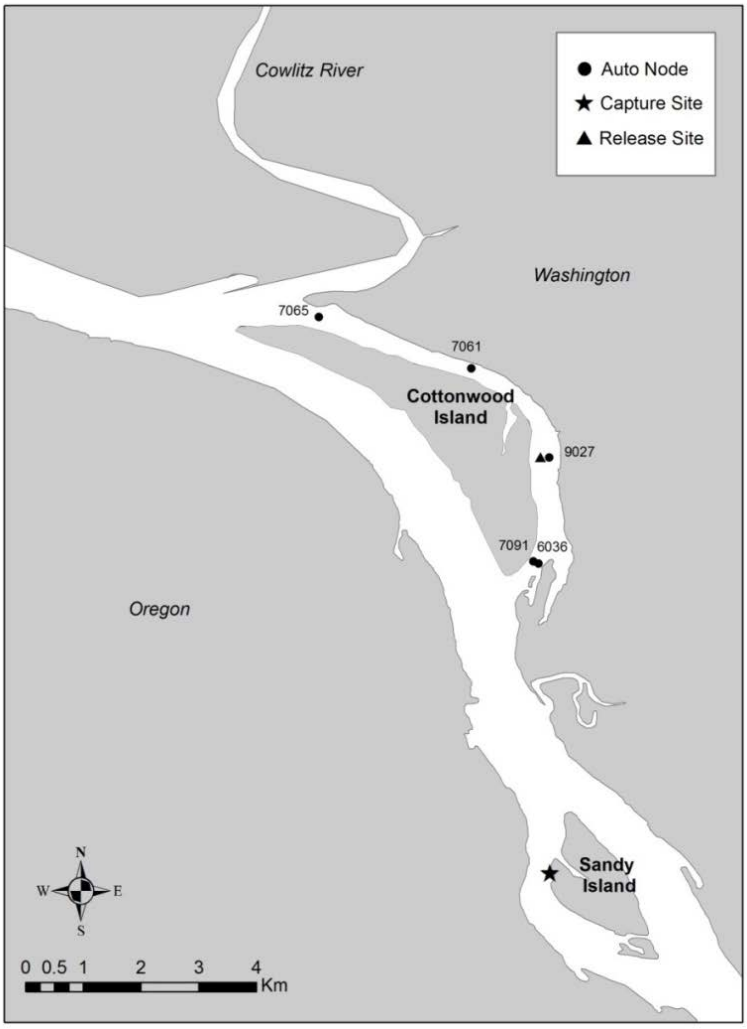

Figure 5.1. Map of 2012 residence time study area in the LCRE.

[1] This work also included an objective to estimate the extent to which tagged juvenile salmon from the main-stem LCRE might migrate up the Cowlitz River during their emigration to the sea. This objective could not be met because excessive shoaling at the mouth of the Cowlitz River prevented boat access from our LCRE study area into the Cowlitz.

Juvenile Chinook salmon exhibit a bi-modal size distribution in tidal freshwater during winter (Sather et al. 2012)—modes at ~40 mm and ~105 mm. Fish in the larger mode are big enough to be tagged with acoustic transmitters. Because residence times of Chinook salmon in shallow off-channel areas of the LCRE are not well-understood, we first estimated residence times of 95- to 125-mm Chinook salmon in an off-channel area near the SRD (~rkm 198) during February through April 2010 and 2011 (Johnson et al. 2011a). During 2011, coho salmon were also sampled. In 2012, we changed the sampling location to the Cottonwood Island area to determine whether juvenile salmon residence times were comparable in late winter and early spring for the two locales, which are approximately 86 rkm apart.

5.1 Methods

Juvenile Salmon Acoustic Telemetry System (JSATS) acoustic transmitters manufactured by Advance Telemetry Systems were surgically implanted in juvenile Chinook salmon obtained by beach seining (for seining methods, see Sather et al. 2011) (Table 5.1). Fish were weighed, measured, and a sample of fin tissue was collected from each fish for genetic stock identification.

Table 5.1. Summary of acoustic telemetry methods for residence time studies during 2010, 2011, and 2012 in the LCRE. Species code: CH = juvenile Chinook salmon.

Factor	2010	2011	2012
Study Period (receivers deployed)	Jan 27 to April 23	February 3 to May 17	February 2 to August 23
Tag Manufacturer	ATS	Sonic Concept	ATS
Tag Weight in Air (g)	0.43	0.63	0.43
Tag Dimensions (mm; W × H × L)	5.21 × 3.8 × 12	5.5 × 4 × 16.9	5.21 × 3.8 × 12
Species Tagged	CH	CH and Coho	CH
Source of Tagged Fish	In situ beach seine	In situ beach seine	In situ beach seine
Marked/Unmarked	Unmarked	Marked/Unmarked	Marked/Unmarked
Mean Fish Fork Length (mm)	CH = 103	CH = 115, Coho = 116	CH = 121
Mean Weight (g)		CH = 16, Coho = 16	CH = 19
Genetic Stock Estimate	Yes	Yes	Yes
Number of Tagged Fish Potentially Available for Detection	51	50	16
Number of Release Sites	(1) Sandy River delta (SRD) vicinity	(1) Sandy River delta (SRD) vicinity	(1) Cottonwood Island, Carroll's Channel

Surgeries were performed in the field to insert acoustic transmitters in 18 fish. During surgery, each fish was anesthetized (40 mg/L) and placed ventral side up on a foam cradle. A tube was placed in the mouth of each fish to allow for a gravity-fed dilution of the "maintenance" anesthesia. A micro-sharp was used to make a 5- to 7-mm incision on the linea alba between the pelvic girdle and pectoral fin. An active (transmitting) acoustic tag was placed into the coelomic cavity of the fish. The incision was closed with two interrupted sutures using a 5-0 Monocryl suture. Two fish died during the 48-h post-surgery holding period. The 16 remaining fish had an average weight of 19 g and fork length of 121 mm, with ranges of 9 to 77 g and 98 to 190 mm, respectively (Table 5.2).

Table 5.2. Characteristics of the tagged Chinook salmon released in Carroll's Channel in February 2012. The codes are WC_F for West Cascades Fall Chinook salmon, WC_S for West Cascades Spring Chinook salmon, and WR_S for Willamette River Spring Chinook salmon. Probability refers to the chances that a given genetic stock estimate is correct. Analysis of genetic stock identification was provided by D. Teel, National Marine Fisheries Service.

Fish No.	Length (mm)	Weight (g)	Ad Clipped	Genetic Stock	Probability
1	112	13.5	No	WC_F	0.9997
2	120	16.1	No	WC_F	1
3	113	12.5	Yes	WC_F	0.5005
4	121	16.1	No	WC_F	0.9921
5	108	10.5	No	WC_F	1
6	100	8.6	No	WC_F	1
7	115	13.5	No	WC_F	1
8	107	10.6	No	WC_F	1
9	100	8.9	No	WC_F	0.9971
10	129	21.3	Yes	WC_F	0.9942
11	113	13.6	Yes	WC_F	0.6514
12	98	10.7	No	WC_F	0.9985
13	190	77.3	No	WC_Sp	0.7
14	138	23.3	No	WC_Sp	0.7934
15	151	30.7	No	WR_Sp	1
16	126	18	No	WR_Sp	0.697

Based on genetic stock identification (Teel et al. 2009), the 16 Chinook salmon we tagged and released were mostly West Cascades Fall Chinook salmon (12 fish) (Table 5.2). Tagged fish also included West Cascades Spring Chinook salmon (2 fish) and Willamette River Spring Chinook salmon (2 fish). All fish tagged during the 2012 study originated from watersheds downstream of Bonneville Dam. In all, 16 acoustic-tagged fish were released on February 2–3, 2012 in Carroll's Channel of the LCRE (Figure 5.1).

Five autonomous acoustic receivers were placed to cover almost the entire 8-km off-channel area behind Cottonwood Island (Figure 5.1) to detect signals from the transmitters in the tagged fish. Two of the receivers (#6036 and #7091) were deployed near the upstream entrance to the channel to ensure detection of tagged fish in this area. The receiving nodes were in place from February 2 through August 23, 2012. Data were downloaded monthly. A concurrent tag-life study in a tank at PNNL offices in North Bonneville showed tag life was more than 89 d with the pulse repetition interval of 10 s used in the study. This finding indicates no effect of transmitter battery life on final detection events and residence times.

A total of 14 tagged fish composed the residence time data set; for reasons not known, two tagged fish were never detected after release. Methodologies associated with analysis of residence time data are available from Johnson G. et al. (2011a). The residence time investigation has the following assumptions and caveats: 1) the residence time estimates are conservative because we do not know how long a given fish was in the study area before it was captured and tagged; 2) tagged fish behavior is not affected by the

tag; i.e., tagged fish are representative of untagged fish; 3) the date/time of last detection on a receiving node indicates when fish left the study area; and 4) the tagged fish have not been eaten.

5.2 Results

The analysis of residence time data included the identification of genetic stocks, time juvenile salmon spent in residence, their exit timing and distribution, and fish length and weight relationships to residence time.

5.2.1 Residence Time

Median residence time was approximately 17 d for the 14 tagged Chinook salmon in this study (Table 5.3). The mean residence time was 22 d, with a range from 0.03 to 62 d.

Table 5.3. Residence time (d) statistics for tagged Chinook salmon behind Cottonwood Island from February through April 2012. Monitoring occurred until August 23, but the last of the tagged fish exited the study area on April 27.

Statistic	Time (d)
Minimum	0.03
Maximum	62.3
Mean	22.4
Median	17.5
n	14

5.2.2 Exit Timing and Distribution

Acoustic-tagged Chinook salmon exited the study over a 3-month period—February through April 2012 (Figure 5.2). Nine of the 14 tagged fish exited in February, 3 fish exited in March, and 2 in April. All tagged fish had vacated the study area by the end of April.

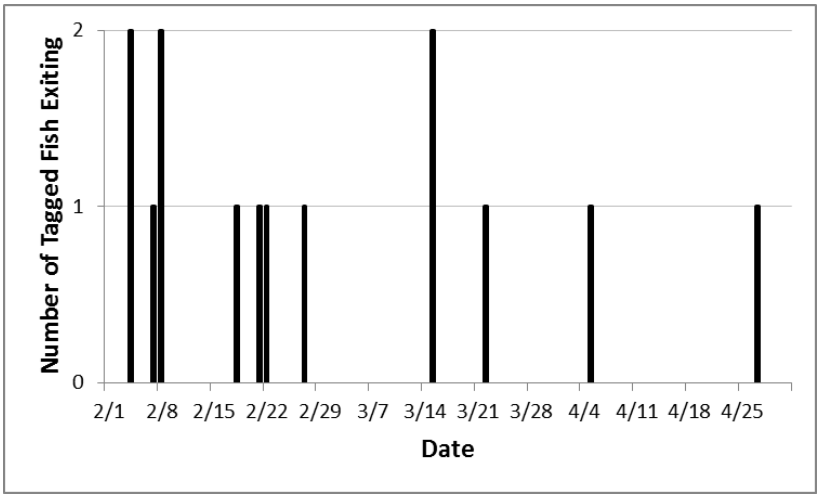

Figure 5.2. Exit timing (fish per day).

The exit distribution of tagged Chinook salmon, as indicated by the node of last detection, showed no apparent pattern (Figure 5.3). Five fish were last detected on the two downstream nodes and six fish were last seen on the upstream nodes in the study area. Last detections for the other three fish were at the middle node.

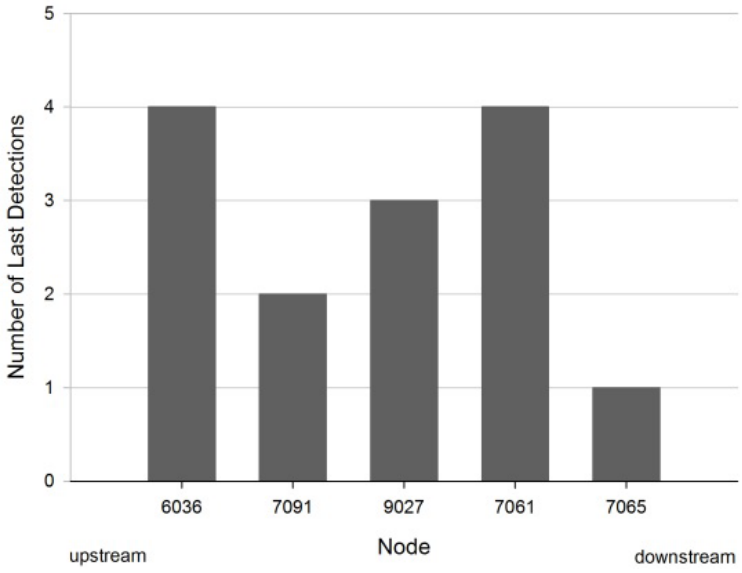

Figure 5.3. Node where tagged fish were last detected.

5.2.3 Fish Length and Weight Versus Residence Time

There was a non-significant negative relationship between fish length and residence time for Chinook salmon (P = 0.4925, Figure 5.4). Conversely, there was a significant negative correlation between Chinook salmon weight and residence time (P = 0.0166), although this relationship is largely the result of one large fish with a short residence time.

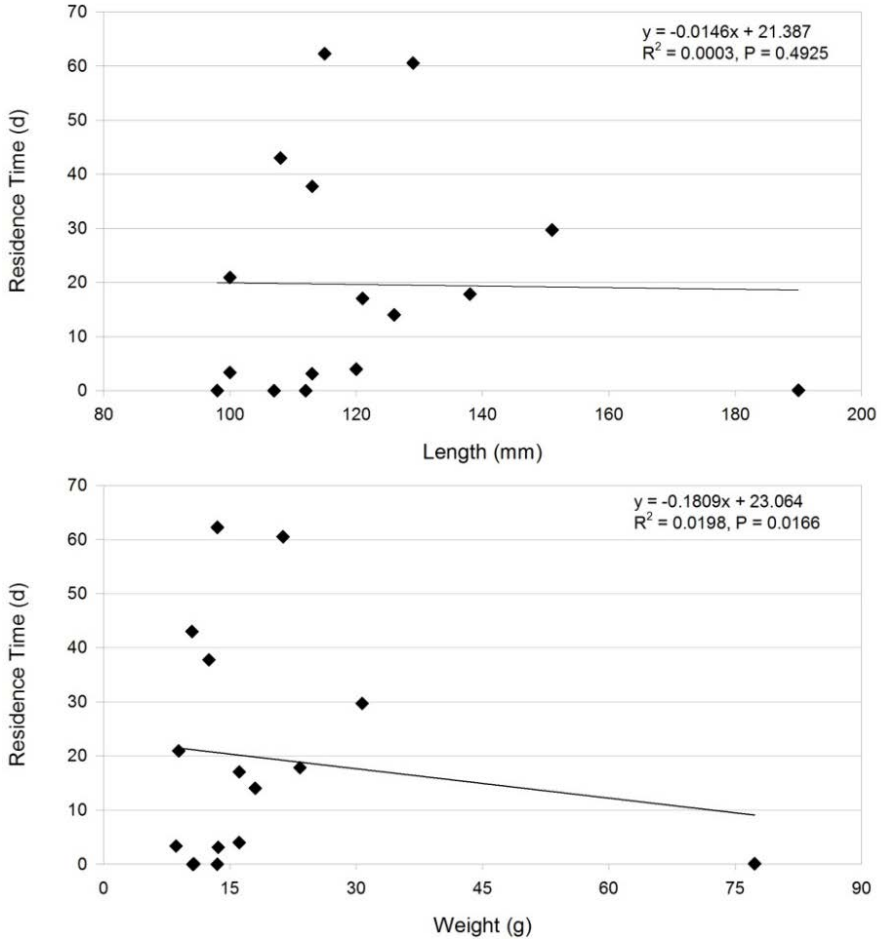

Figure 5.4. Relationship between residence time and fish length (top panel) and weight (bottom panel) at the time of tagging for Chinook salmon.

5.3 Discussion

For this type of residence time study, it is ideal to capture fish for tagging and release tagged fish in the same general area. During 2012, we moved to the large, off-channel area behind Cottonwood Island (Carroll's Channel) to investigate whether residence times for juvenile salmon there were similar to those at the SRD 80 km upstream in the LCRE, but, despite a dedicated effort we did not capture taggable fish (>95 mm FL) in Carroll's Channel. Therefore, by necessity, we shifted sampling to the nearest suitable seining location—Sandy Island 8 km upstream from Cottonwood Island—where we were able to capture fish, then tag, transport, and release them back in Carroll's Channel. We assumed that the transport operation did not affect fish behavior and that the residence times we estimated were representative of fish present volitionally in Carroll's Channel. These assumptions seem reasonable because standard, well-established transport procedures were used and the residence times were comparable to those from previous studies conducted elsewhere (Table 5.4).

For juvenile Chinook salmon, mean residence times in Carroll's Channel during 2012 were similar to those in the SRD during 2011 (22.42 d and 24.68 d, respectively; Table 5.4). The mean residence time during 2010, however, was about 10 d longer than in 2011 or 2012. The median residence time of 17.5 d

during 2012 at Carroll's Channel was between medians of 26.3 d and 11.61 d during 2010 and 2011 at SRD, respectively. Residence times were reasonably consistent between the SRD and Carroll's Channel study areas.

Future studies should address a similar, but slightly smaller, size mode in fall (October–December; see Figure 2.8) than we did in this study, which focused on the larger size mode (90–120 mm) in the length frequency distribution consistently observed in winter (January−March) at the SRD (Figure 2.8). Such a study could address residence times from fall into winter and early spring. This could be an improvement over the current study design where the residence time of fish in the area *before* we captured and tagged them is unknown. Moreover, smaller size classes of juvenile salmon should be studied when acoustic transmitters are downsized, which is expected for field application during 2014. As acoustic telemetry technology advances, detailed study designs based on tagging and release-recapture methods (Appendix C) should be considered to provide functional, action effectiveness data on juvenile salmon responses to habitat restoration.

Table 5.4. Estimates of residence time for juvenile salmon during late winter and early spring 2010, 2011, and 2012 at two study areas in the Lower Columbia River.

Year	2010	2011		2012
Study Area	SRD	SRD		Carroll's Channel
Species	Chinook	Chinook	Coho	Chinook
Minimum	1.11	0.09	0.02	0.03
Maximum	78.39	73.68	89.78	62.26
Mean	34.25	24.68	28.56	22.42
Median	26.31	11.61	11.22	17.46
n	48	12	36	14

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Note: Reference lists for citations in the appendices are included in the respective appendix.

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Appendix A

Synopsis of Pre-Restoration Action Effectiveness Data from Sandy River Delta

Appendix A

Synopsis of Pre-Restoration Action Effectiveness Data from Sandy River Delta

In this appendix, we provide a synopsis of the action effectiveness research (AER) for pre-restoration conditions at the Sandy River delta (SRD) site of the proposed restoration action to remove a dam and rechannelize the historic Sandy River channel.

A.1 Introduction

Background: The Multi-Scale Action Effectiveness study, from its inception in 2007 as a Bonneville Power Administration (BPA) study to its current role as a U.S. Army Corps of Engineers (USACE) study, was structured to determine the effectiveness of the proposed restoration action to remove the earthen dam and rechannelize the historic Sandy River. Because the restoration action was a distinct possibility when the study started, we developed a statistical design (Skalski 2007, 2008) and commenced collecting pre-restoration data. In total, we have collected 59 months of pre-restoration data at the SRD study area. Because of complicating issues unrelated to the research, restoration has yet to occur, but is now scheduled for summer 2013.

Restoration Action: According to the USACE (Expert Regional Technical Group Project Template dated January 25, 2013), "The proposed project removes a dam blocking the east distributary on the lower Sandy River and will restore natural hydrologic connectivity to approximately 51 acres of disconnected tidal floodplain back to the lower Columbia River. Project elements include removal of the concrete and rock structure built in the 1930s, approximately 10 acres of tidal channel excavation to connect approximately 1,000 feet of the old channel to the Columbia River. Additionally some planting, invasive plant control, dredge material removal will occur and large woody debris placements will occur."

A.2 Study Design

Working Hypotheses: The working hypotheses for the dam removal/rechannelization restoration are that it will increase access for juvenile salmon, including upriver stocks, to the shallow water habitats in the SRD; improve prey availability for juvenile salmon; and, increase juvenile salmon growth rates.

Statistical Design and Analysis: The original statistical design for the action effectiveness evaluation was a before-after-control-impact (BACI) design.[1] This BACI design (Skalski and McKenzie 1982) entails two pairs of impact/control sampling sites (N/E and C/B; Figure A.1). As Skalski[1] noted, "…In the BACI analysis of restoration effects, it will be important to know on what mathematical scale (e.g., arithmetic, log, etc.) are location and seasonal effects additive. Besides graphical analysis, a

[1] Skalski JR. 2007. *Statistical Considerations for the Tidal Water Monitoring Pro*ject. Prepared by JR Skalski (UW) for GE Johnson and KL Sobocinski (PNNL) and Skalski JR. 2008. *Statistical Considerations for the Tidal Water Monitoring Study, 2008–2009*. Prepared by JR Skalski (UW) for GE Johnson and KL Sobocinski (PNNL).

two-way ANOVA test for location-by-season interactions will be used to test for additivity. It is recommended that only seasonal data [monthly data will be reduced to seasonal values] be used to help minimize the potential effects of autocorrelation on the repeated measures through time. Using the log of the impact-control ratios should also minimize any perceived autocorrelations…The F-test for the main effect of the monitoring phase tests the overall effect of the mitigation action…"

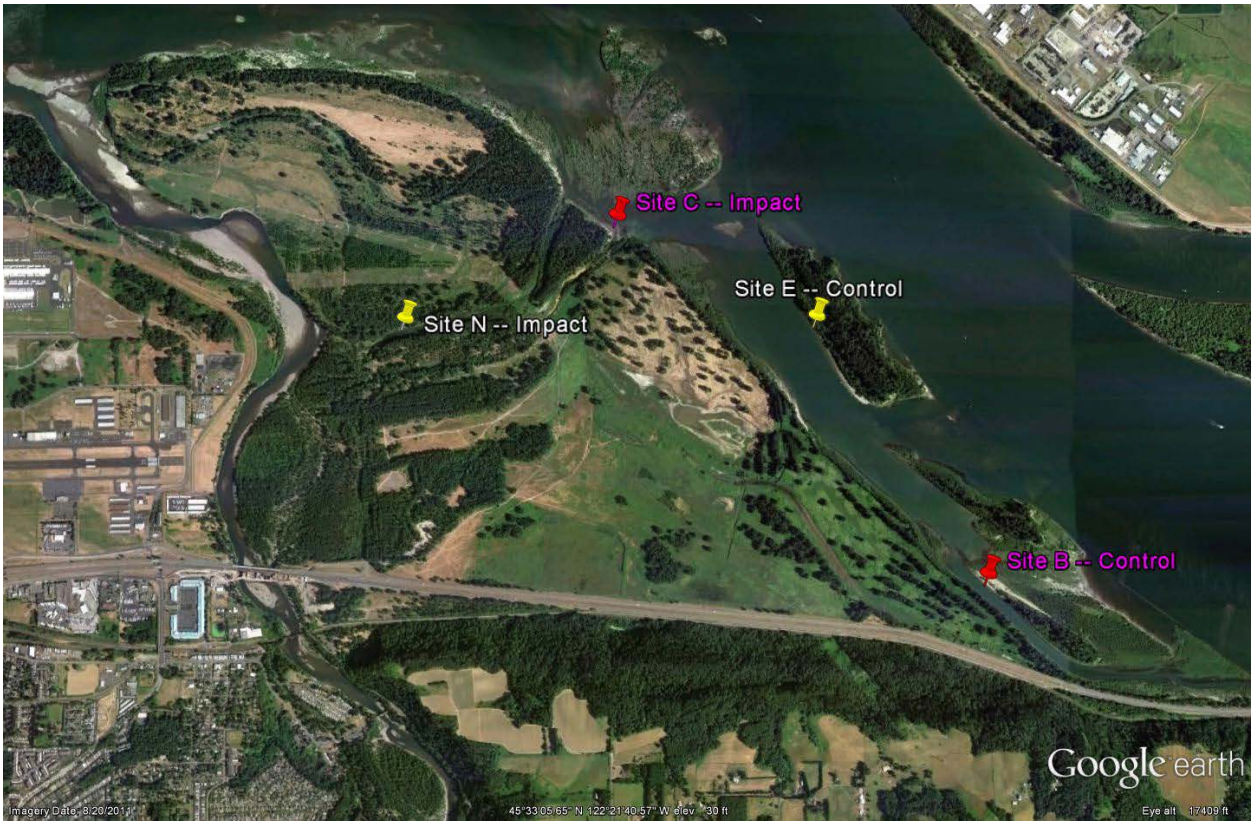

Figure A.1. SRD sampling sites: Impact site N is paired with control site E (N/E) and impact site C is paired with control site B (C/B).

- Site N is a wetland habitat located within the remnant SRD. Site N is within the upper extent of the remnant channel that drains to site C, the former mouth of the Sandy River. Site N will be directly affected by the dam removal/rechannelization.
- Site E, on the west side of Gary Island, is a wetland site characterized by a gradual sloping beach face, fine sediments, and fringing emergent vegetation. It is upstream of site C and on the opposite side of the channel between the Oregon shore and Gary Island. Site E will serve as a control for restoration at site N.
- Site C, at the historic mouth of the Sandy River, is a river confluence site that maintains connection to a small channel from the remnant delta. The topography of this site is higher in elevation compared to the other sampling locations and this is the only site that completely dewaters during periods of low flow. Like site N, this area will be directly affected by the dam removal/rechannelization.
- Site B, on the southwest side of Chatham Island, is an off-channel site that maintains a steeply sloping beach face adjacent to a fairly deep channel. The thalweg of the channel adjacent to site B is

fairly deep; the inlet and outlet to this channel maintain a higher elevation, causing the site to be disconnected during low-flow conditions. Site B will serve as a control for restoration at site C.

Sampling and Response Variables: Sites B, C, E, and N have been sampled nearly every month since September 2007, river conditions permitting (Figure A.1). Response variables (=monitored indicators) in the original design focused on juvenile salmon density and non-native fish density (Skalski 2007, 2008). Response variables reflecting ecosystem processes and realized functions are sampled to increase the intensity of the AER in terms of ecosystem responses to the restoration action beyond fish densities (Table A.1).

Table A.1. Variables *planned for use* in the SRD Action Effectiveness Evaluation.

Response Variable	Metric	Ecosystem Component
Juvenile salmon density	#/m^2	structure
Channel cross-sectional area	m^2 (NAVD 88)	structure
Prey availability	#/m^2	process
Salmon bioenergetics	mean specific growth rate	realized function
Covariate		
Water-surface elevation	m (NAVD 88)	controlling factor
Water temperature	°C	controlling factor
Dissolved oxygen	mg/L	controlling factor
Genetic stock identification	genetic stock diversity	structure

NAVD = North American Vertical Datum of 1988.

A.3 Pre-Restoration Conditions

In addition to the 2012 annual report for the Multi-Scale Action Effectiveness study (Section 2), the material that follows is based on *Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats of the Lower Columbia River, 2007–2010* by Johnson et al. (2011) and *Multi-Scale Action Effectiveness Research in the Lower Columbia River and Estuary, 2011* by Sather et al. (2012). Results were reasonably consistent across study years.

Juvenile Salmon Density: During 2007–2010, juvenile salmon density on a seasonal basis was highest in spring (mean ~0.01 fish/m^2) (Figure A.2). The season with the second highest density was winter (mean ~0.005 fish/m^2). Chinook salmon was only salmonid species encountered during every season. Unmarked juvenile Chinook salmon were the most abundant salmonid captured (74% of the total salmonid catch), followed by chum (10%) and coho (8%) salmon, and steelhead (<1%). Marked Chinook salmon composed 8% of the total salmonid catch. Densities were relatively low (mean <0.005 fish/m^2) at the sampling sites during summer and fall. These patterns were observed in 2011 and 2012.

Genetic Stock Identification: A majority of the unmarked juvenile Chinook salmon were from the Spring Creek Tule Fall and the Upper Columbia Summer/Fall stock groups. Smaller proportions were estimated for the West Cascade Tributary Fall and Willamette River Spring groups. Snake River Fall,

Deschutes River Fall, and West Cascade Tributary Spring fish were also present. Most of the marked, hatchery fish were also from the Spring Creek Group Tule Fall and Upper Columbia Summer/Fall stock groups.

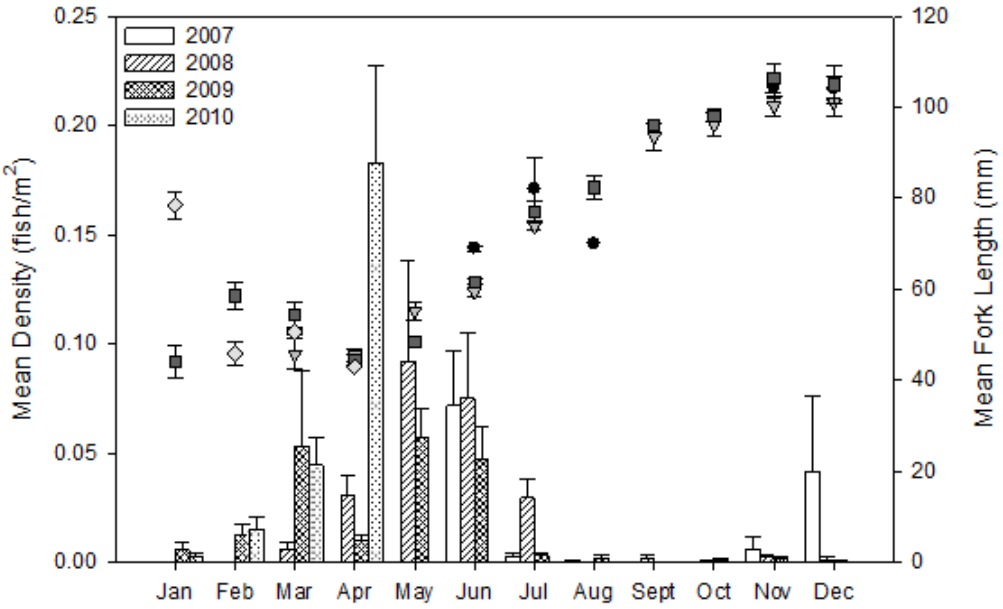

Figure A.2. Monthly juvenile salmon densities 2007–2010. (Figure 2.14 from Johnson et al. 2011; Mean Monthly Density of Unmarked Chinook Salmon Sampled at the SRD Study Area During the 2007–2010 Study Period and Average Fork Length for Unmarked Chinook Salmon During 2007 (circles), 2008 (triangles), 2009 (squares), and 2010 (diamonds). Error bars represent the standard error of the mean.)

Channel Cross-Sectional Area: Pre-restoration data have yet to be collected for channel cross-sections.

Prey Availability: Prey collected from the sampled environment consisted of benthic, drifting, and winged or terrestrial organisms. Benthic samples were composed primarily of several insect groups, mollusks, and large crustaceans including scuds and opossum shrimp. Drift samples were dominated by small crustaceans (e.g., water fleas, copepods, and seed shrimps), various insect groups, and arachnids. Although present, large crustaceans such as those found in the benthos were encountered infrequently in the drift. Samples collected using traps designed to help characterize winged or terrestrial prey items, consisted almost exclusively of insects. The diets of juvenile Chinook salmon were generally dominated by dipterans (primarily chironomids and ceratopogonids), hemipterans, and malacostracans (Amphipoda and Mysidae).

Salmon Bioenergetics: At each SRD sampling site, mean predicted specific growth rates for simulation cohorts generally were positive, indicating juvenile Chinook salmon typically gained biomass throughout residence periods. Feeding rates and estimates of gross conversion efficiency generally were moderate to high at the sampled sites. Over time, predicted growth was positive for most cohorts, and there were few instances during which a cohort lost biomass over a simulation period.

Water-Surface Elevation: Site-specific water-surface elevations generally followed annual, seasonal, weekly, and hourly patterns similar to those observed at Bonneville Dam; e.g., power peaking at Bonneville Dam caused corresponding rises in water level 40 km downstream at the SRD study area. Site-scale hydrodynamics were also influenced by topography and lateral connectivity with the main channel. Example water-surface elevation data for sites C and N, the impact sites, are shown in Figure A.3.

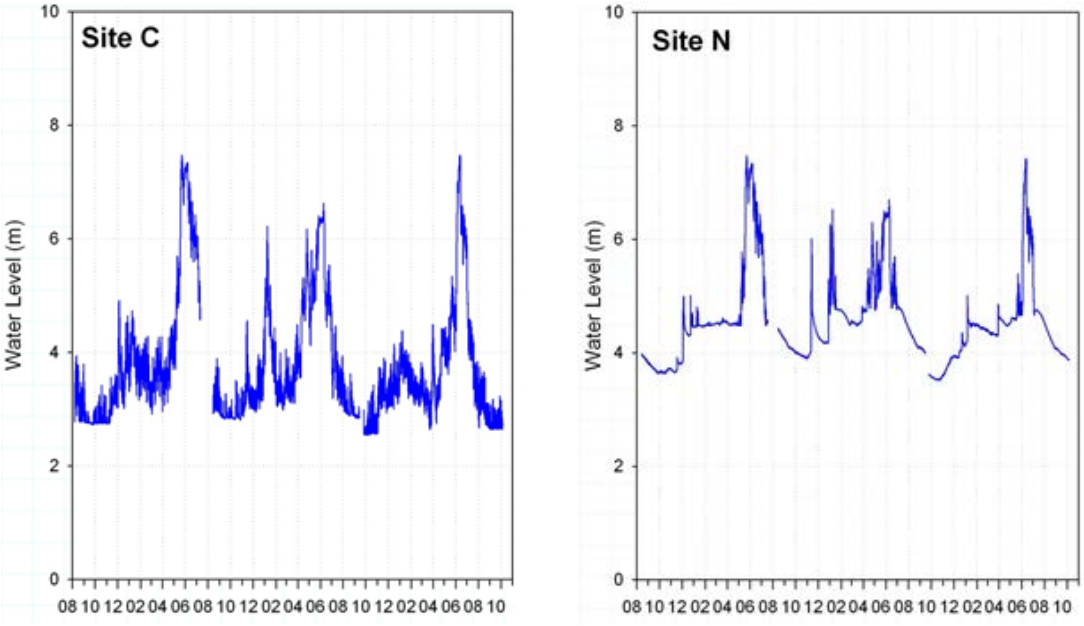

Figure A.3. Water-surface elevation for sites C and N, August 2007 through October 2010. (From Johnson et al. 2011, Figures A.2 and A.4, respectively.)

Dissolved Oxygen: These data have not been summarized yet.

Water Temperature: Water temperature peaked during August through October (~25°C) and gradually declined through the fall and winter months. While the overall seasonal patterns were similar, thermal conditions differed among sites. Example water temperature data for sites C and N, the impact sites, are shown in Figure A.4.

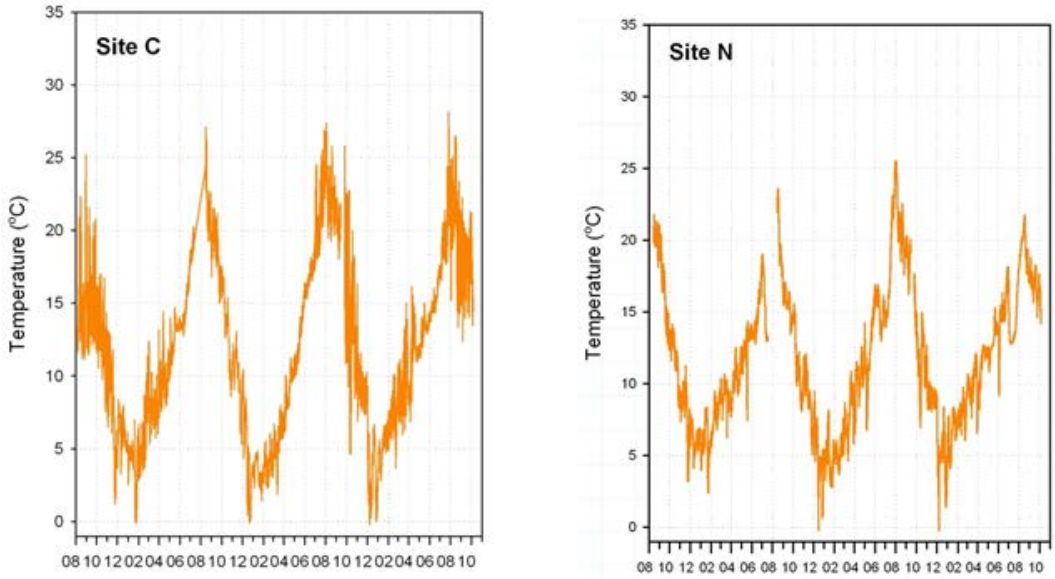

Figure A.4. Water temperature for sites C and N, August 2007 through October 2010. (From Johnson et al. 2011, Figures A.5 and A.7, respectively.)

A.4 Conclusion

Site-specific understanding of the SRD gained from pre-restoration monitoring supports discussion of the efficacy of the proposed reconnection of the old Sandy River to the Columbia River. Removal of the dam and rechannelizing the historic Sandy River could increase fish accessibility to this channel, as well as to other rearing habitats. Changes in the flow regime, coupled with riparian plantings as part of other restoration efforts in the delta, could increase water quality and flux of salmon prey items from the SRD to the main-stem Columbia River. Confluences offer sources of heterogeneity in main-stem rivers by influencing morphological features and aquatic habitats. Reconnecting the old Sandy River channel to the Columbia River will likely increase the opportunity and capacity of habitats for aquatic biota, including juvenile salmon. To address this expectation, post-restoration AER should be conducted at a minimum during 2014 and 2015.

A.5 References

Johnson GE, NK Sather, AJ Storch, DJ Teel, JR Skalski, EM Dawley, AJ Bryson, GR Ploskey, C Mallette, TA Jones, AB Borde, SA Zimmerman, ES Van Dyke, DR Kuligowski, and KL Sobocinski. 2011. *Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats of the Lower Columbia River, 2007–2010.* PNNL-20083, Pacific Northwest National Laboratory, Richland, Washington.

Sather NK, AJ Storch, GE Johnson, DJ Teel, JR Skalski, AJ Bryson, RM Kaufmann, J Blaine, DL Woodruff, DR Kuligowski, RK Kropp, EM Dawley. 2012. *Multi-Scale Action Effectiveness Research in the Lower Columbia River and Estuary, 2011.* PNNL-SA-86024, DRAFT annual report submitted to the U.S. Army Corps of Engineers, Portland District, Portland, Oregon, by Pacific Northwest National Laboratory, Richland, Washington.

Skalski JR and DH McKenzie. 1982. "A design for aquatic monitoring programs." *Journal of Environmental Management* 14:237–251.

Appendix B

Synopsis of Pre- and Post-Restoration Action Effectiveness Research from Julia Butler Hansen National Wildlife Refuge Mainland and Islands

Appendix B

Synopsis of Pre- and Post-Restoration Action Effectiveness Research from Julia Butler Hansen National Wildlife Refuge Mainland and Islands

In this appendix, we provide a synopsis of the action effectiveness research for pre- and post-restoration conditions at tide gate restoration sites in the Julia Butler Hansen National Wildlife Refuge (JBHNWR), which includes the JBH mainland and Tenasillahe Island.

B.1 Introduction

Background: Habitat restoration at JBHNWR has focused on replacement of traditional style tide gates with side-hinged, tide gates or side-hinged, self-restrained tide gates and installation of these new style tide gates at diked sloughs without connection to the Columbia River. Prior to restoration actions, there was reduced tidal influence in JBHNWR sloughs and poor habitat conditions for salmonid species.

Restoration Actions: At the JBHNWR mainland during summer 2008 and 2009, the U.S. Army Corps of Engineers (USACE) installed tide gates at three sloughs previously blocked by dikes (Hampson, Indian Jack, and Winter) and replaced tide gates in two other sloughs (Brooks and Duck Lake) (Figure B.1). These gates were designed and installed to allow attenuated tidal influence but still protect Columbian white-tailed deer habitat. At JBHNWR's Tenasillahe Island during 2007, the USACE installed three side-hinged aluminum tide gates equipped with a manually controlled fish orifice (Figure B.2) to improve fish passage and water quality conditions.

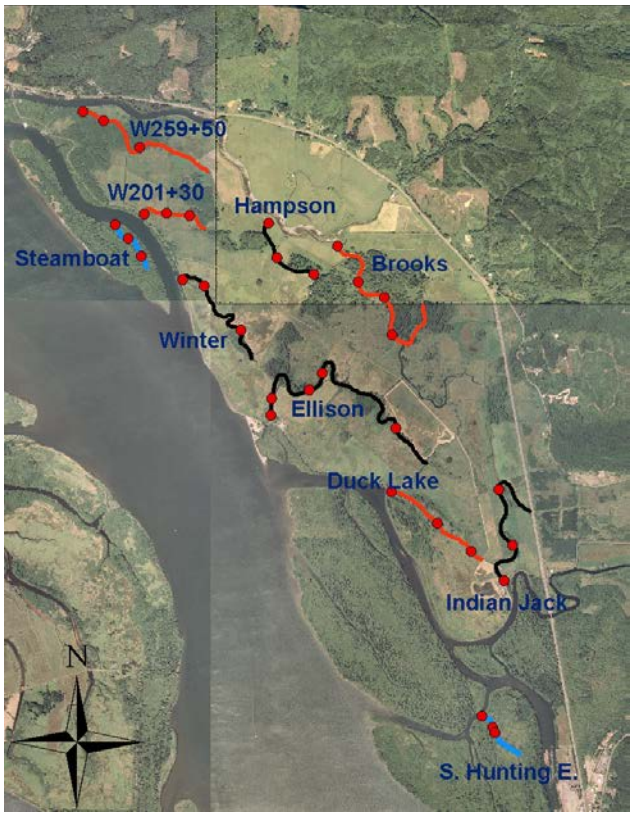

Figure B.1. Area map of JBHNWR showing the location of sloughs and sample reaches (red circles) surveyed in 2007, 2008, 2010, 2011, and 2012. Black, red, and blue lines indicate closed, gated, and reference sloughs, respectively.

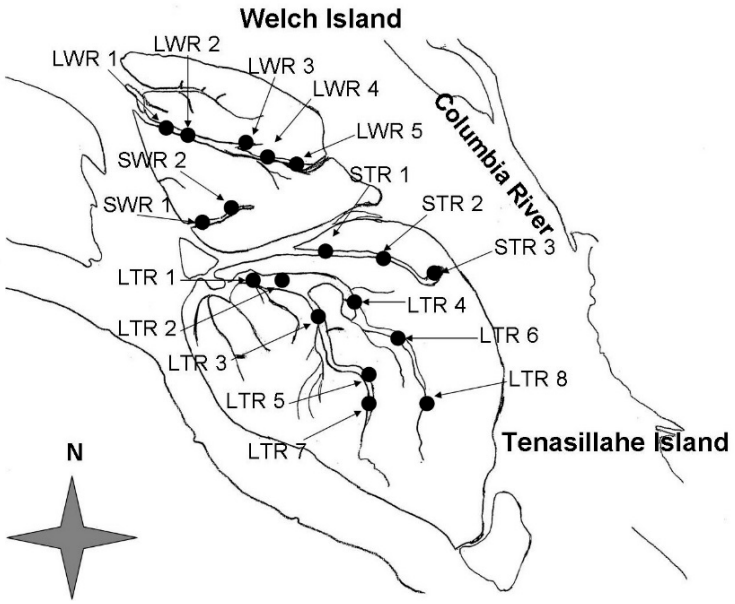

Figure B.2. Area map of Tenasillahe Island (treatment sites) and Welch Island (referenced sites) showing locations of reference sloughs (large Welch slough, small Welch slough), treatment sloughs (large Tenasillahe slough, small Tenasillahe slough) and sample reaches within sloughs.

B.2 Study Design

Working Hypothesis: The working hypothesis was that the tide gate installations would increase access for juvenile salmon to the rearing habitats in JBHNWR sloughs and improve water quality conditions (temperature).

Statistical Design: The study design involved comparing fish community and water quality conditions in treatment sloughs to reference sloughs and control sloughs before and after treatment. The reference sloughs are considered the ideal condition and are expected to be independent of treatments. At the JBHNWR mainland, all treatment, control, and reference sloughs were located within a 2-km reach of the Columbia River on the Washington side of the shipping channel and therefore, likely experienced the same pool of migrating fish. Sample reach selection was designed to ensure random and spatially balanced data collection representing at least 10% of the total slough length. At Tenasillahe Island, sample reaches within each slough were randomly selected using a random, spatially balanced approach to ensure that various habitats and conditions were represented. Adjacent to Tenasillahe Island, Welch Island provided appropriate reference sites. The difference between conditions at reference sloughs and treatment sloughs post-construction could reflect the extent that the new tide gates allow sloughs to reach ideal conditions.

Sampling and Response Variables: To minimize any spatial or temporal bias, the order in which reaches were sampled was randomized. Sampling effort was distributed evenly throughout the field season. This sampling regime was used to ensure the various habitats and conditions present within each slough were represented, as well as to capture the seasonal variation and changes in fish community composition and distribution. The key response variables measured during field sampling were fish community composition, juvenile salmon, and habitat characterization (water temperature).

B.3 Pre-Restoration Conditions

The material that follows is based on annual reports submitted by U.S. Fish and Wildlife Service to the USACE from 2007 through 2009 (Johnson et al. 2007, 2009a, 2009b) and Ennis (2009).

B.3.1 Julia Butler Hansen NWR Mainland

Water temperatures in gated sloughs may be more limiting to juvenile salmon than temperatures in reference sloughs (Figure B.3–Figure B.6). This was evidenced by the seven-day average daily maximum (7-DADM)—a rolling average of seven consecutive daily maximum water temperatures recorded within a stream. Water temperature levels in W259+50, W201+30, Brooks, Hampson, Ellison, and Steamboat sloughs remained below 16°C until late May or early June 2007. Temperature in Indian Jack and Duck Lake exceeded 16°C in early May. Water temperature in S. Hunting E. was still within the acceptable range when the temperature logger was removed on 13 June.

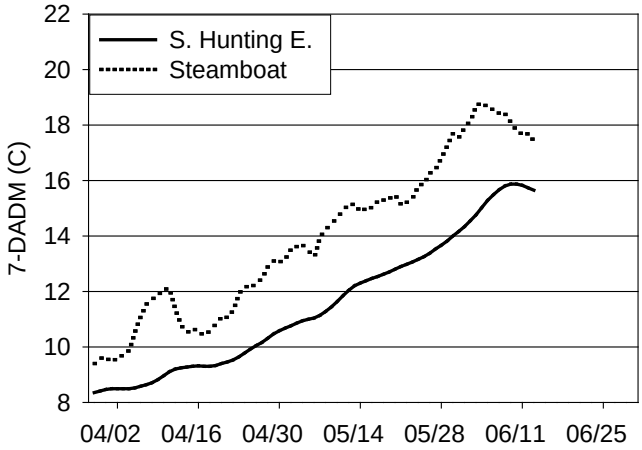

Figure B.3. Seven-day average daily maximum (7-DADM) water temperature for lowermost sampling reach within reference sloughs S. Hunting E. and Steamboat at JBHNWR, 2007 pre-restoration conditions.

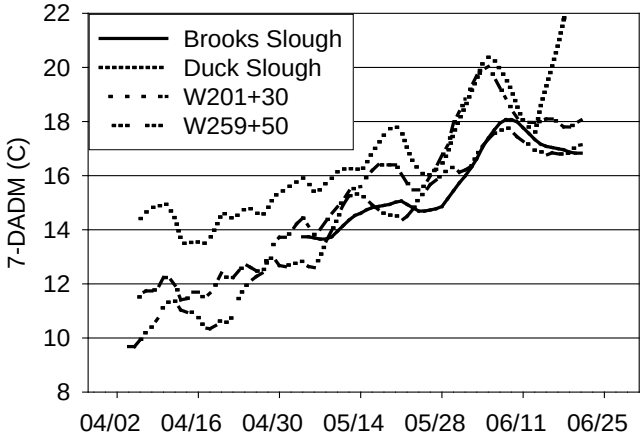

Figure B.4. Seven-day average daily maximum (7-DADM) water temperature for lowermost sampling reach within gated sloughs Brooks, Duck, W201+30, and W259+50 at JBHNWR, 2007 pre-restoration conditions.

Juvenile salmon were able to enter through pre-restoration tide gates (Table B.1 and Table B.2). The numbers of juvenile salmonids captured entering through these tide gates were fewer than those captured entering into reference sloughs. However, sampling efficiency may be different among sloughs and slough types. Thus, while tide gates did not prevent juvenile salmon from entering sloughs, it was unclear whether the tide gates influenced the number of juvenile salmon entering sloughs.

Juvenile Chinook salmon entered both reference and gated sloughs (Table B.1 and Table B.2). Of Chinook captured entering gated sloughs, the proportion longer than 110-mm fork length was more than that entering reference sloughs. This suggests that smaller fish may have more difficulty entering gated sloughs.

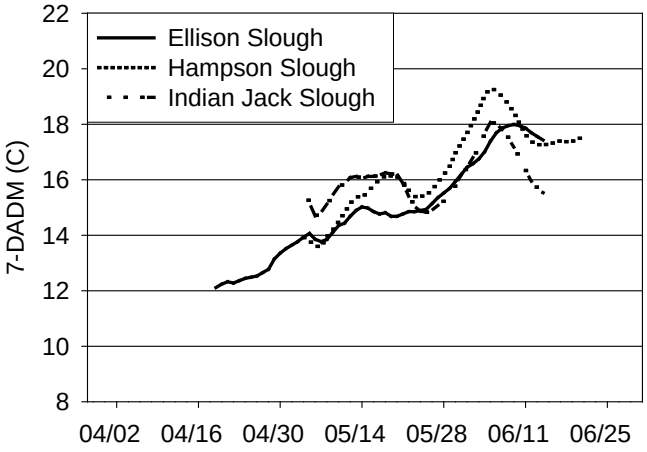

Figure B.5. Seven-day average daily maximum (7-DADM) water temperature for lowermost sampling reach within closed Ellison, Hampson, and Indian Jack sloughs at JBHNWR, 2007 pre-restoration conditions.

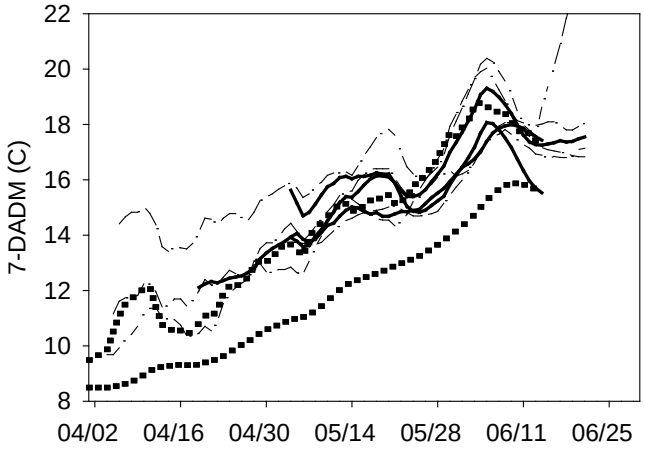

Figure B.6. Seven-day average daily maximum (7-DADM) water temperature for lowermost sampling reach within closed sloughs (solid lines), gated sloughs (dashed lines) and reference sloughs (dotted lines) JBHNWR, 2007 pre-restoration conditions.

Reference sloughs appeared to contain more salmon species than either pre-restoration closed or gated sloughs (Table B.1 and Table B.2). Three species of salmon (Chinook, coho, and chum salmon) were captured in reference sloughs, whereas chum salmon were not captured in closed or gated sloughs. Both hatchery origin (adipose fin clipped) and unmarked Chinook salmon were captured in gated and reference sloughs. Gated and closed sloughs appeared to contain more total fish species, specifically non-natives species, than reference sloughs. All 10 species captured in reference sloughs were native species. In gated sloughs, 6 of 14 species were non-native and in closed sloughs 10 of 16 species were non-native. Non-native species captured include those known to prey on juvenile salmonids (e.g., smallmouth bass). Juvenile salmon were captured in Ellison slough (closed slough) indicating that salmon can move among sloughs using interconnecting ditches.

Table B.1. Species type and percentage (number) of total fish captured (all sampling methods combined) in four closed sloughs (Indian Jack, Ellison, Winter, and Hampson), four gated (Duck Lake, W201+30, W259+50, and Brooks), and two reference sloughs, 2007 pre-restoration conditions.

Species		Closed	Gated	Reference
Threespine stickleback		73.9 (99)	57.3 (220)	80.4 (295)
Bluegill		2.2 (3)	1.3 (5)	0.0 (0)
Chinook salmon	Total	0.7 (1)	22.7 (87)	18.0 (66)
	Unmarked	0.7 (1)	20.1 (77)	16.6 (61)
	Adipose clipped	0.0 (0)	2.5 (10)	1.3 (5)
Chum salmon		0.0 (0)	0.0 (0)	0.3 (1)
Coho salmon		0.7 (1)	3.1 (12)	0.8 (3)
Dace		0.0 (0)	0.0 (0)	0.3 (1)
E. banded killifish		3.0 (4)	7.6 (29)	0.0 (0)
Largemouth bass		1.5 (2)	1.3 (5)	0.0 (0)
Largescale sucker		0.7 (1)	0.0 (0)	0.0 (0)
N. pikeminnow		2.2 (3)	0.0 (0)	0.0 (0)
Peamouth		6.7 (9)	0.2 (1)	0.0 (0)
Pumpkinseed		0.0 (0)	1.6 (6)	0.0 (0)
Sculpin		0.0 (0)	0.5 (2)	0.3 (1)
Smallmouth bass		0.7 (1)	0.0 (0)	0.0 (0)
Unknown sunfish		5.2 (7)	4.4 (17)	0.0 (0)
Yellow bullhead		0.7 (1)	0.0 (0)	0.0 (0)
Yellow perch		1.5 (2)	0.0 (0)	0.0 (0)

Table B.2. Species type and percentage (number) of total fish captured (all sampling methods combined) in two closed sloughs (Indian Jack and Winter), two gated sloughs (Duck Lake and W259+50), and two reference sloughs, 2008 pre-restoration conditions.

Species		Closed	Gated	Reference
Threespine stickleback		79.6 (296)	84.2 (976)	95.6 (2,158)
Bluegill		0.8 (3)	0.2 (2)	0.0 (0)
Brown bullhead		0.5 (2)	0.0 (0)	0.0 (0)
Chinook salmon	Total	0.0 (0)	5.0 (58)	1.5 (33)
	Unmarked	0.0 (0)	4.9 (57)	1.3 (30)
	Adipose clipped	0.0 (0)	0.1 (1)	0.1 (3)
Coho salmon		0.0 (0)	0.1 (1)	0.0 (0)
Common carp		0.0 (0)	0.4 (5)	0.0 (0)
E. banded killifish		2.7 (10)	1.6 (18)	0.0 (0)
Largemouth bass		0.3 (1)	0.0 (0)	0.0 (0)
Largescale sucker		0.0 (0)	0.3 (3)	0.2 (5)
Northern pikeminnow		0.0 (0)	0.5 (6)	1.3 (30)
Peamouth		0.3 (1)	1.2 (14)	0.9 (20)
Pumpkinseed		2.4 (9)	1.2 (14)	0.0 (0)
Sculpin		0.0 (0)	3.1 (36)	0.3 (7)
Shiner		3.0 (11)	0.1 (1)	0.0 (0)
Starry flounder		0.0 (0)	0.0 (0)	0.2 (5)
Unknown sunfish		10.5 (39)	2.2 (25)	0.0 (0)

B.3.2 Tenasillahe Island

Temperatures reached sub-lethal threshold levels earlier in May 2007 in Tenasillahe Island sloughs than in Welch Island sloughs during pre-restoration conditions (Figure B.7). This was most pronounced in large Tenasillahe slough (LTS) where the 7-DADM reached the threshold criterion of 16°C 18 d earlier than large Welch slough (LWS).

Tide gates had a significant influence on fish community structure during pre-restoration conditions (Table B.3 and Table B.4). There are fundamental differences in species composition and relative abundance between Tenasillahe Island and Welch Island sloughs. The greatest overall species richness occurred in un-gated LWS (12 species) followed by LTS (10 species). Small Welch slough contained three times the number of species that were found in small Tenasillahe slough. A higher percentage of non-native species were captured in both Tenasillahe Island sloughs than in either of the Welch Island sloughs. The relative abundance of individuals was higher in Welch Island sloughs than in Tenasillahe Island sloughs. These differences are likely related to Tenasillahe Island sloughs' lack of tidal influence and the water quality parameter values resulting from limited water exchange in addition to access issues caused by tide gate operation.

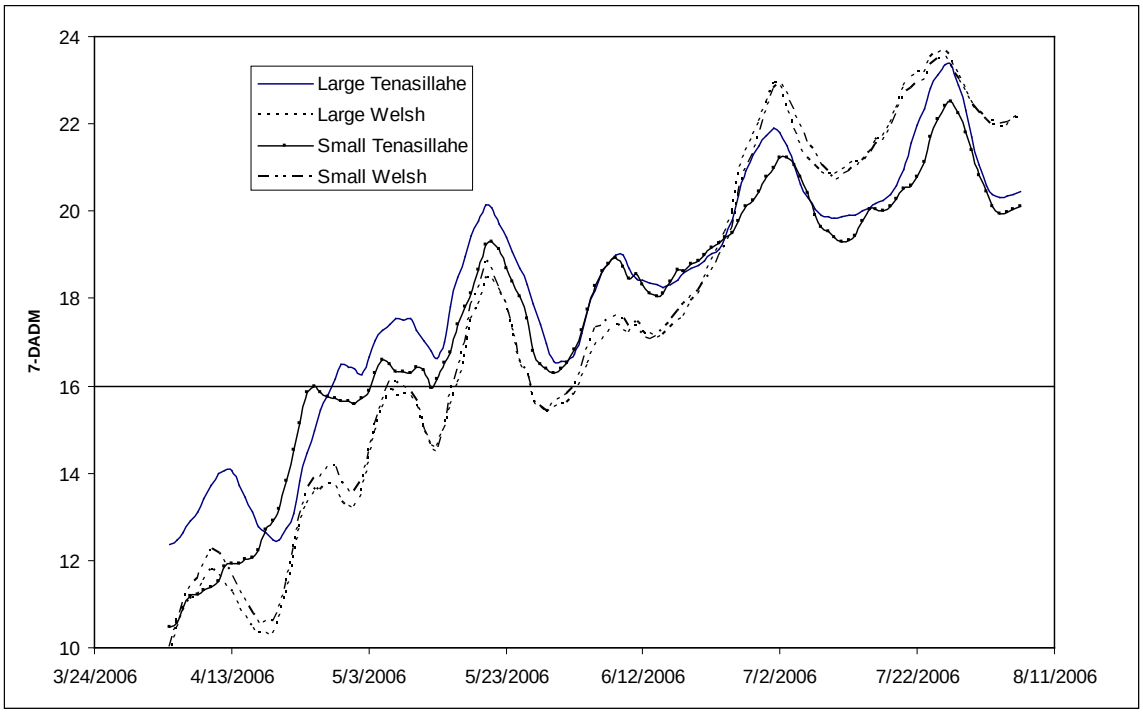

Figure B.7. Seven-day average daily maximum (7-DADM) water temperature for lowermost sampling reach within large Tenasillahe slough, small Tenasillahe slough, large Welch slough, and small Welch slough, 4 April through 6 August 2006 pre-restoration conditions. Horizontal line represents 16°C.

Table B.3. Species, number, and size range of salmonids that were captured in Tenasillahe Island and Welch Island sloughs, 2006 pre-restoration conditions.

Island	Species	Total	Size Range (mm)
Tenasillahe	Chinook	1*	46
	Chum	1*	46
Welch	Chinook	270	36–195
	Chum	6	44–50
	Coho	1	47

* Caught in submersible traveling screen during non-scheduled sampling when tide gate was blocked open.

Table B.4. Salmonid capture per seine pull in Tenasillahe and Welch Island sloughs, 2006 pre-restoration conditions. Small Tenasillahe Island slough was not sampled with seines.

		Chinook	Coho	Chum
March 27–April 14	Large Tenasillahe	0	0	0
	Large Welch	14.3	0	0.5
	Small Welch	12.5	0	1
May 8–May 26	Large Tenasillahe	0	0	0
	Large Welch	55.0	0	0
	Small Welch	9.5	0	0

B.4 Post-Restoration Conditions

The material that follows is based on the 2012 annual report for the Multi-Scale Action Effectiveness study (Section 3), and progress reports submitted to USACE from 2009 to 2011 (Johnson et al. 2011).

B.4.1 JBHNWR Mainland

Temperature consistently exceeded the 18°C 7-DADM during July and August in all sloughs (Table B.5). (Note: 16°C 7-DADM was used in earlier work.) The earliest month of this temperature was May in Duck Lake (treatment) and Steamboat (reference) sloughs. The latest month was October in Winter (treatment) and both reference sloughs. The highest cumulative days above the 18°C 7-DADM was 107 in the treatment slough Duck and the lowest was in the treatment slough Indian. Both S. Hunting E. and Steamboat reference sloughs showed similar cumulative days exceeding threshold temperature (90 and 94 days). Interestingly, the control slough Ellison had the second fewest days exceeding threshold temperature (67 days).

Table B.5. Days per month that the 7-DADM exceeded 18°C for JBHNWR mainland treatment (Winter, Duck, Indian), reference (South Hunting, Steamboat), and control (Ellison) sloughs in 2012. (Repeated from Table 3.4 in the main body of this report.)

	Winter	Duck	Indian	South Hunting	Steamboat	Ellison
January	0	0	0	0	0	0
February	0	0	0	0	0	0
March	0	0	0	0	0	0
April	0	0	0	0	0	0
May	0	6	0	0	1	0
June	2	13	0	0	4	0
July	28	31	21	26	27	24
August	31	31	30	30	31	31
September	30	26	14	30	29	12
October	6	0	0	4	2	0
November	0	0	0	0	0	0
December	0	0	0	0	0	0
Total	**97**	**107**	**65**	**90**	**94**	**67**

In previously closed sloughs that were retrofitted with self-restrained tide gates, juvenile salmon were captured in all reaches post-restoration (Table B.6). Chinook salmon were captured in all reaches of Ellison slough (closed slough). In addition, chum and coho salmon juveniles were captured in Ellison slough.

Table B.6. Fish species and number of individuals captured by seining in JBHNWR mainland sloughs and two reference sloughs, 2010 post-restoration conditions.

	Control		Treatment			Reference	
Species	W259+50	Ellison	Brooks	Hampson	Winter	S. Hunting E.	Steamboat
3-spine stickleback	1,395	1,251	1,229	927	3,804	3,857	603
Bluegill	1	157	55	11			
Brown bullhead		1					
Chinook salmon	6	16	10	24	280	36	26
Chum salmon		2			2		
Coho salmon	7	2	5	23	13	18	
Common carp		3					
Crappie		151	80	8			
E. banded killifish	1	49	3	22	44		1
Largemouth bass		82	42	16	22		
Largescale sucker		15	1	1		10	
N. pikeminnow		29	5	7	5	15	
Peamouth	3	39		43	27	18	1
Pumpkinseed		9	2	7		1	
Redside shiner		107	4	11	9		
Salmonid			1				
Sculpin	10				4	3	
Starry flounder						1	
Unknown sunfish		33	10	3			
Yellow bullhead				1			
Yellow perch		61	4	8			
Total	**1,423**	**2,007**	**1,451**	**1,112**	**4,210**	**3,959**	**631**

B.4.2 Tenasillahe Island

Randomized Intervention Analysis and Monte Carlo tests revealed no significant difference in slough water temperatures after tide gate replacement, although minimum temperatures dropped up and downstream of the tide gates (Ennis et al. 2009).

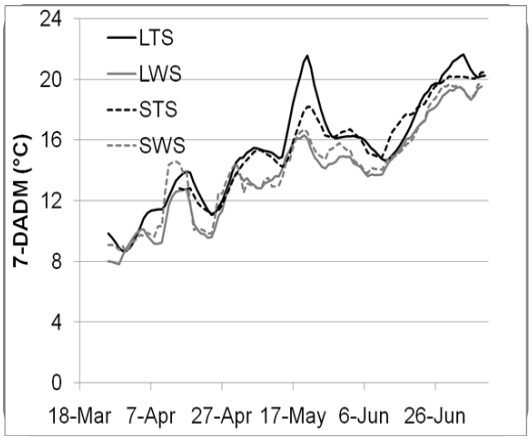

Figure B.8. Seven-day daily average maximum (7-DADM) water temperature of large Tenasillahe (LTS), large Welch (LWS), small Tenasillahe (STS), and small Welch (SWS) sloughs, 2008 post-restoration.

Fish, including Chinook salmon, do enter and exit LTS through tide gates during post-restoration conditions (Table B.7). However, juvenile salmonid access to Tenasillahe Island sloughs appears to be limited. Their access into Tenasillahe Island sloughs is dependent upon tide gate opening. Although all three tide gates on LTS open in response to tidal fluctuation, the duration of opening is limited to times when slough water elevation is above that of river water elevation. As such, fish have access to LTS on average less than 20% of any given day. In addition, fish must swim against the water flow to enter the slough through the gates.

More salmon species were captured in reference sloughs than in gated sloughs during post-restoration conditions (Table B.7). Three species of salmonid (Chinook, chum salmon, and steelhead trout) were captured in reference sloughs, whereas only Chinook salmon were captured in LTS. Both hatchery origin (adipose fin clipped) and unmarked Chinook salmon were captured in gated and reference sloughs. More non-natives species were captured in gated sloughs than in reference sloughs. Non-native species captured include those known to prey on juvenile salmonids (e.g., smallmouth bass).

At the mainland JBHNWR, juvenile salmon were captured throughout the study area in every treatment, control, and reference slough. Salmonid species captured included juvenile Chinook, coho, and chum salmon. At Tenasillahe Island, juvenile salmonid species were not captured at any treatment site after tide gate retrofit. In addition, no juvenile salmonids were captured entering or leaving LTS during passage trials after tide gate retrofit.

Table B.7. Species type and percentage (number) of total fish captured (all sampling methods combined) in large Tenasillahe, large Welch, small Tenasillahe, and small Welch sloughs, during 2008 post-restoration.

Fish Species	# of Individuals	% of Total
Large Tenasillahe Slough		
3-spine stickleback	185	38.95%
Sculpin	91	19.16%
Eastern banded killifish	61	12.84%
Largescale sucker	53	11.16%
Peamouth	25	5.26%
Bluegill	22	4.63%
Unknown sunfish	13	2.74%
Yellow bullhead	8	1.68%
Common carp	6	1.26%
Pumpkinseed	6	1.26%
Largemouth bass	3	0.63%
Chinook salmon	2	0.42%
Total	**475**	
Large Tenasillahe Slough		
3-spine stickleback	296	66.22%
Shiner	45	10.07%
Unknown sunfish	44	9.84%
Bluegill	22	4.92%
Pumpkinseed	22	4.92%
Sculpin	12	2.68%
Eastern banded killifish	4	0.89%
Smallmouth bass	2	0.45%
Total	**447**	
Large Welch Slough		
3-spine stickleback	35,002	98.80%
Chinook salmon	175	0.49%
Peamouth	126	0.36%
Eastern banded killifish	53	0.15%
Sculpin	31	0.09%
Chum salmon	23	0.06%
Starry flounder	10	0.03%
Largescale sucker	3	0.01%
Northern pikeminnow	3	0.01%
Unknown sunfish	2	0.01%
Total	**35,428**	
Small Welch Slough		
3-spine stickleback	12,375	98.77%
Peamouth	63	0.50%
Chinook salmon	54	0.43%
Sculpin	26	0.21%
Eastern banded killifish	4	0.03%
Largescale sucker	4	0.03%
Pacific lamprey	1	0.01%
Steelhead trout	1	0.01%
Western brook lamprey	1	0.01%
Total	**12,529**	

B.5 Conclusion

Our ability to witness changes in fish community and salmon densities was limited by the high variance among fish collections. We have high confidence in fish presence data, but our density data have high variance that may preclude their use for discerning between treatment and reference conditions. It was clear, however, that improving tidal influence by installing side-hinged, self-restrained tide gates resulted in improved fish passage, fish distribution, and water quality conditions at the JBHNWR mainland. Installation of side-hinged tide gates with a manually operated fish orifice had no positive effect on fish passage or water quality at Tenasillahe Island.

B.6 References

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Appendix C

Compendium of Tag Release-Recapture Designs for Studying Juvenile Salmonid Performance in the Lower Columbia River and Estuary

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Compendium of Tag Release-Recapture Designs for Studying Juvenile Salmonid Performance in the Lower Columbia River and Estuary

Prepared by John R. Skalski

This appendix contains a reprint of a submittal from John Skalski to Gary Johnson dated March 22, 2013, to fulfill a deliverable for a subcontract to University of Washington from Pacific Northwest National Laboratory for statistical support for the 2012 Multi-Scale Action Effectiveness study.

Compendium of Tag Release-Recapture Designs for Studying Juvenile Salmonid Performance in the Lower Columbia River

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Table of Contents

1 Introduction

The purpose of this report is to provide a compendium of available mark-recapture designs applicable for studies of juvenile salmon performance in the lower Columbia River and estuary (LCRE). Juvenile salmon performance includes survival rates, migration pathways and rates, residence times, spatial and temporal distributions, etc. This general material is intended to serve as a reference to aid implementation of research, monitoring, and evaluation (RME) in support of the Action Agencies' (BPA and Corps) Columbia Estuary Ecosystem Restoration Program. The RME effort and relevant tagging studies provide critical data and information on juvenile salmon performance to program managers, who are especially interested in evaluation of the effectiveness of restoration actions. Detailed sampling designs depend on study objectives and study area characteristics, and are not included here.

The design of efficient and effective tagging studies is the result of careful integration of fish biology, tag technology, statistical methods, and study objectives. Investigators must consider fish behavior and how it may affect study design. Tag technologies must be matched with fish size, study objectives, and statistical model assumptions. Finally, statistical release-recapture models must be tailored to the fish behavior, study objectives, and limitations of tag technology. Failure to appreciate the integrative nature of tagging studies risks the possibility of assumption violations and estimability of desired performance measures.

A crucial concern in the tagging of juvenile salmonids in the Lower Columbia River is fish size. The lower Columbia River estuary has juvenile salmonids from the size of fry to 2+-year-old steelhead. Generally, the smaller the fish, the smaller the requisite tag size and the information content from the tag. Improvements in technology are shrinking the sizes of the tags, but limitations still exist and must be taken into consideration when designing mark-recapture studies.

In this report, three categories of tag types will be reviewed for their ability to provide assessments of salmon performance in the LCRE.

- Active tags, including radio and acoustic tags, which can be used to tag juvenile salmonids ≥95 mm in fork length.
- PIT tags that can be used on juvenile salmonids ≥75 mm in fork length
- Inert tags used to mark fish down to 35 mm in fork length.

Although active tags and PIT tags carry unique identifiers, inert tags used on the smallest of fish often only have, at best, group-level identifiers (i.e., the marking technique cannot be used to identify individuals). This is a very important distinction that has major impacts on study design and estimability of demographic and movement parameters. The other distinction is that fish usually need to be physically recaptured to read an inert tag while noninvasive detectors can acquire information from PIT and active tags. However, the detection field for PIT tags is measured in feet, while for active tags the detection zone is in hundreds of meters.

Table 1.1 provides a cross-listing of tag type versus study objectives, which will provide the basis for the structure of this report. After the Introduction (Chapter 1), I present a Review of Marking Techniques (Chapter 2), then proceed to material for common study objectives of juvenile salmon performance: Survival Estimation (Chapter 3), Movement and Migration (Chapter 4), Entrance Efficiency (Chapter 5), Residualization and Overwintering (Chapter 6), and Residence Time/Travel Time Measurements (Chapter 7).

Table 1.1. Cross-reference of report sections based on study objectives and types of tags used in marking fish.

Objective	Tag Types		
	Inert	PIT	Active
Survival	3.1 Models M_2, M_4, M_5 M_7, M_8, M_9	3.1 Models M_1, M_2, M_3, M_5 M_6	
Movement	4.1	4.2	4.3
Entrance efficiency	5.1	5.2	5.3
Residualization/overwintering	N/A	N/A	6
Residence/travel time	7.1	7.2	7.2

2 Review of Marking Techniques[a]

This section provides a brief review of methods for marking salmonid fry and smolts. Fry here is defined as the stage of development between alevin and parr, and generally of fork lengths < 61 mm (Roegner et al. 2012). Studies that use marking techniques are widely varied, and the type of mark is dependent on study objectives, the period of time over which the mark is required to be detectable, and sample sizes required (Nielson 1992). Marking technologies are classified into three categories of detection (Pacific Salmon Commission 2006).

A. Inert Tags
   1. Immediate visual: Marks that can be immediately seen by the unaided eye.
   2. Immediate Specialized Detection: Marks that can be immediately detected with the proper sampling equipment. Every fish must be analyzed, because these marks do not have a visual identifier.
   3. Delayed Detection: Marks that require sacrificing the fish or sampling harvested fish to obtain the tag or tissue for specialized laboratory analysis.

These three categories of inert tagging methods (i.e., non-electronic) are generally used for the smallest of fish. As fish size increases, electronic tags are more likely to be used because of the detection

[a] This section reproduced in part from Skalski, J. R., and J. Griswold. 2006. A summary of methods for conducting salmonid fry mark-recapture studies for estimating survival in tributaries. Volume XXI in the Design and Analysis of Salmonid Tagging Studies in the Columbia Basin. Bonneville Power Administration, Portland, OR

advantages they provide. In addition to this category of inert tags, we will also review two types of electronic tags.

B. PIT Tags
C. Active Tags (i.e., radio tags and acoustic tags)

PIT tags provide passage tag detection where tagged fish must be interrogated in close physical proximity to obtain identifiers, while acoustic tags send out an identification signal that can be detected some distance (e.g., hundreds of meters) from the tagged fish.

Each category of tag is further subdivided at the level to which the mark can be distinguished: individuals or groups, and whether the method is suitable for mass marking or selective studies. This overview of tag methodologies considers externally visible marks, internal marks, external marks, internal tags, natural marks, biotelemetric tags, genetic identifiers, and chemical marking (Table 2.1).

2.1 External Marks

Externally visible tags suitable for marking salmonid fry include visible implant elastomer (VIE) and visible implant filament (VIF) tags.

Table 2.1. Summary of marking techniques for juvenile salmon and the availability of unique codes for ease of identifiability, permanency (stability) of the mark, and minimum fish size or life stage requirements.

Mark technique	Unique codes possible	Suitable for mass mark	Category of detection*	Mark longevity	Minimum fish size or life stage
External Marks					
Fluorescent elastomer (VIE)	240	No	IV or ISD	Variable	26 mm
Fluorescent filament (VIF)	3-character alpha-num	No	IV or ISD	Variable	50 mm
Pigments	Limited to few	Yes	IV, NS	Temporary	Fry
Immersion dyes	4 or 5	Yes	IV, NS	Low	25 mm
Fluorescent	Limited	Yes	ISD	Low	25 mm
Adipose clip	None	Yes	IV	Permanent	50 mm
Ventral clip	None	Yes	IV	Permanent	50 mm
Adipose clip & CWT	Unlimited	Yes	DD	Variable	<2.1 g HLCWT* >2.1 g FLCWT*
Ventral clip & CWT	Unlimited	Yes	DD	Variable	<2.1 g HLCWT >2.1 g FLCWT
Tattoos	Limited	No	IV, NS	Low	100 mm
Freeze branding	Limited	No	IV, NS	Low	100 mm
Internal Marks					
Half-length CWT (HLCWT)	Unlimited	Yes	DD, S	High	<2.1 g

Mark technique	Unique codes possible	Suitable for mass mark	Category of detection*	Mark longevity	Minimum fish size or life stage
Full-length CWT (FLCWT)	Unlimited	Yes	DD	High	>2.1 g
Natural Marks					
Strontium isotope ratios	None	Yes	DD, NS	Permanent	None
Chemical Marks					
Oxytetracycline	Limited	Yes	DD, S	High	None
Strontium chloride	Limited	Yes	DD, S	12-16 mos.	None
Calcein immersion	Limited	Yes	ISD, S, or NS	12-16 mos.	None
Tetracycline	Limited	Yes	DD, S	High	None
Otolith Marks					
Otolith thermal	Nearly unlimited	Yes	DD, S	Permanent	Emergent fry – advanced yearling
Dry mark otolith (eggs)	Unlimited	Yes	DD, S	Permanent	Only for eggs
Genetic	Unlimited	Yes	DD, NS	100%	N/A
Biotelemetric					
PIT	Unlimited	Yes	ISD, NS	85 – 100%	50 mm
Acoustic	≈32,000	Yes	ES	Variable tag life	95 mm
Radio	≈32,000	Yes	ES	Variable tag life	95 mm

***Detection categories**

IV – Immediate visual: marks that can be easily and immediately seen by the unaided eye.
ISD – Immediate specialized detection: marks that can be immediately detected with the proper equipment. Every fish must be analyzed because these fish do not have a visual identifier.
DD – Delayed detection: marks that require sacrificing the fish or sampling harvested fish to obtain the tag or tissue for specialized laboratory analysis.
S – Sacrificing the fish is required.
NS – No Sacrifice of the fish is required.
ES – Electronic signal.
FLCWT – Full-length, coded-wire tag.
HLCWT – Half-length, coded-wire tag.

2.1.1 VIE Tags

The VIE tag consists of a biocompatible, two-part, fluorescent, silicone, elastomer material that is mixed and injected into tissue as a liquid with a hypodermic syringe. After 24 h at room temperature, it cures into a pliable solid, providing an externally visible internal mark that fluoresces under ultraviolet light. The fluorescent elastomer is available in four colors, and recognition of individuals is possible through the use of different body locations and colors (Bonneau et al. 1995, Choe and Yamazaki 1998). VIE -tagged wild Age 0 brown trout (*Salmon trutta)* (26 – 70 mm) experienced negligible mortality, and all marks were recognizable upon recapture 39–83 day after marking (Olsen and Vollestad 2001). Green

and yellow VIE post-ocular tagged rainbow trout became undetectable when a blue-filtered flashlight and amber glasses were used to aid in mark detection and rates of detection were found to be related to marking skill (Close 2000). Advantages of this type of tag are low tag mortality, the ability to mark very small fish in the field with little training needed to recognize marks, and not requiring sacrificing the fish. Disadvantages are the inability to distinguish more than about 240 individuals, the possibility of tissue growth occluding visibility of marks, and the reliance on highly trained techs in order to avoid excessive tag losses. VIE tags may be appropriate for short-term survival and movement studies.

2.1.2 VIF Tags

VIF tags are made of plastic and coded with a three-digit alphanumeric code. Tag placement by syringe in transparent periocular eye tissue exhibited excessive stress whereas tag placement in the tissue between fin rays improved the ability to successfully tag fish smaller than 150-mm fork length (Shepard et al. 1996, Wenburg and George 1995). Shepard et al. (1996) found a retention rate of 58% for VIF tags in wild westslope rainbow trout (*Oncorhynchus* sp.) as small as 100 mm; tag loss rate was inversely related to FL. Bailey et al. (1998) reported retention rates of 73% after 2 years in coho marked at a mean length of 108 mm. Recognition of tags may not be a significant problem as inexperienced technicians successfully detected the body locations of VI tags 91% and 98% of the time after only 1 h of training, though tag retention is thought to be closely related to technician skill (Hale and Gray 1998). Advantages of the system are little or no effects of the tag on survival or growth, the ability to mark large numbers of fish in the field with unique codes, immediate detection of marks with minimal training, and the ability to release recaptured fish after recording the tag. Disadvantages are special training and experience needed to successfully mark fish and the possibility of tissue growth occluding marks. Types of studies suited for VIF tags are short-term mortality and growth as well as movement studies and abundance estimation.

2.1.3 Dye Marking

Dye marking may be suitable for mass marking for short-duration studies where it is necessary to distinguish only a few experimental lots. Dussault and Rodriguez (1997) found that Alcian Blue dye mark retention was low for individuals recaptured 10-14 months after injection and that dye applied to pelvic or pectoral fin locations induced high mortality in smaller fish ~ 55 mm. Bismark brown dye has been used successfully applied in short term (<3 months) abundance estimates of migrating sockeye salmon (*Oncorhynchus nerka)* smolt populations (Carlson et al. 1998). Gaines and Martin (2004) dual-marked Chinook salmon (*Oncorhynchus tshawytscha)* fry (mean fork length = 57.7 mm) with spray-dye fluorescent pigments and Bismarck brown stain, and applied single marks of each type. Daily mortality was less than 0.15% for all marked fish for 3 d after marking. The authors concluded that the dual-marking technique provides a feasible method to differentially mass-mark fish with minimal mortality for short-term studies. It was found that dual-marking improved mark recognition. This technique is efficient, inexpensive, produces an immediately recognizable mark, and can be applied to large numbers of fish in the field with little training. Disadvantages include lack of unique codes and short lifespan, i.e., months, of the mark.

2.1.4 Fin Excision

One of the oldest and simplest of methods of marking fish is fin-clipping. Johnson (2004) used a pelvic fin clip on Atlantic salmon fry to provide a means of distinguishing first summer survival and growth in salmon planted as eggs versus those planted as fry. The adipose fin clip is the external mark of choice used to help recognize CWT-marked salmon (*Oncorhynchus* spp.) in commercial fishery sampling and was sequestered for that purpose until 1996. Delayed mortality of clipped fish is a function of size. Coble (1967) suggested that salmonids smaller than 90 mm FL are especially vulnerable. Mortality is lower for adipose and pelvic fin clips (McNeil and Crossman 1979). In order to avoid biased estimates, studies involving adipose fin clips should be accompanied by an assessment of the rate of naturally missing fins (Blankenship 1990). Advantages are low cost, efficiency of application, and immediate visibility of mark. Disadvantages include lack of unique codes, fin regeneration, and delayed mortality due to the fin clip. Fin clipping may be appropriate for flagging interior marks and movement, abundance estimation, growth studies in situations where groups and individuals need not be identifiable, and where there are no other uses of the same mark to which it could be confused.

2.1.5 Freeze Branding

Freeze branding may provide a useful mark for short-term (less than a year), fry-marking studies not requiring individual capture histories. Advantages include ease of application, low cost, and ability to mass mark as many as 1000 fish per hour. A disadvantage of the technique is that marks fade and become unrecognizable with time (Bryant et al 1990). The authors used a brand 1 mm x 5 mm for young coho salmon (*Oncorhynchus kisutch)* less than 50 mm total length. Straight-line letters—T, V, X, U or I—were used and found to provide the best level of correct recognition upon recapture. By altering the orientation of these letters and changing the side of the fish marked, 30 distinct marks can be made. The freeze brand may work well for short-term studies requiring identification of only a few groups.

2.2 Internal Marks

2.2.1 Coded Wire Tags (CWT)

Peltz and Miller (1990) concluded that half-length coded wire tags (HLCWT) can be used to estimate return proportions from pink salmon (*Oncorhynchus gorbuscha)* hatchery releases numbering in the hundreds of millions. The authors emphasized the importance of the maintenance of a constant proportion of marked fish among all release groups. Possible sources of error using CWT tagging are differential mortality between tagged and untagged fish, tag loss, regeneration of clipped adipose fins, straying due to olfactory damage caused by the tagging procedure, nonrandom distribution of marks in the population (Seber 1982), occurrence in the population of naturally missing adipose fins, the presence of wild fish in the returning broodstock, and error in determination of the proportion of marked fish among the original hatchery releases (Peltz and Miller 1990, Habicht et al. 1998). Evidence that CWT placement in pink salmon fry is related to straying was found to be inconclusive, giving mixed results for the two-year study carried out by Habicht et al. (1998). Blankenship (1990) found that by holding CWT-tagged pink salmon fry for 29 d after tagging a final level of tag loss can be ascertained.

The same study recommended that in order to avoid excessive and prolonged tag loss, fish smaller than 2.1 g be tagged with HLCWT, while fish larger than 2.1 g receive full-length coded wire tags (FLCWT). Blankenship (1990) reported production size releases averaged less than 5% tag loss 30 d after tagging and no significant tag loss 200-300 d after tagging. Kaill et al. (1990) evaluated the use of HLCWT on newly emergent pink salmon fry (mean weight, 0.26 g) and found that estimates of short-term retention rates ranged from 93 to 100% using experienced taggers. Estimated long-term retention rates were 75, 50, 65, and 84% for the years 1983-1986. However, the estimates did not take into consideration human error in recognizing the adipose fin clip nor was there an adjustment for the rate of naturally occurring missing adipose fins. Advantages of CWTs are low cost (8-9 cents/tag), availability of unique codes, and the apparent minimal effect on growth and survival. Disadvantages are the possibility of lost tags and expensive delayed laboratory detection requiring sacrificing the fish.

2.2.2 Natural Marks

Natural geochemical signatures have been found to be useful as a population marking technique (Campana and Thorrold 2001, Barnett-Johnson et al. 2005, Bacon et al. 2004). In a study of Atlantic salmon populations in tributaries of the Connecticut River, Kennedy et al. (2002) found stream-specific Sr isotopic ratios ($^{87}Sr/^{86}SR$) in calcified tissues of salmon parr within 3 months of stocking and were able to differentiate fish from different geographical areas. The authors point out that the site-specific uptake and incorporation of isotopic signatures makes this technique useful for distinguishing fish populations in both wild and managed settings. Kennedy et al. (2002) used micromilling techniques to extract strontium (Sr) isotopic signatures from the otoliths of four returning Atlantic salmon and detected distinct signatures from four life-cycle stages, including prefeeding hatchery development, rearing stream growth, smolt outmigration, and ocean residence.

2.2.3 Chemical Marks

Oxytetracycline (OTC), calcein, and strontium are routinely used in fisheries programs to mark otoliths and other calcified tissue in fish as a way to evaluate fish management strategies. Calcein (2,4-bis[N,N-di(carbomethyl)-amino-methyl]fluorescein; molecular weight, (622) marking can be accomplished by immersing very young fish in a bath containing either (1) 125-250 mg/L calcein for 1 – 6 h; or (2) 2.5-5.0 g/L for 1-7 min. A pre-treatment immersion of fish in a 1-5% solution of non-iodized salt for ~ 3.5 min facilitates the osmotic transfer of calcein into calcified tissues (Johnson 2003). The study found that marks faded on exposure to direct sunlight. Frenkel et al. (2002) and Bart et al. (2001) noted that when immersion was preceded by a 30-s ultrasound exposure mark endurance in caudal fin rays was increased in small rainbow trout (*Oncorhynchus mykiss*) (~ 0.2–0.3 g). A general positive relationship was found between mark endurance and fish size. Differences were not found in growth rates between control fish and the different treatments within any of the size groups (0.2, 0.3, and 1.0 g). Brook trout (*Salvelinus fontinalis)* (~ 1 g) and Atlantic salmon (0.8 g) fed calcein for 5 d showed calcein scale marks 7-10 d postmarking (Honeyfield et al. 2006). Brook trout were marked twice with distinct bands when fed calcein 5 months apart. Increased concentration of calcein in food produced increased mean pixel luminosity in brook trout scales. Longer-term retention of calcein marks has been reported in fish injected or immersed in calcein. Rainbow trout retained their external marks for at least

12 months in young fish (Negus and Tureson 2004, Frenkel et al. 2002). Calcein-marked Atlantic salmon have been recovered from the wild after 16 months (Mohler 2004). Strontium and calcein otolith marking has an advantage over thermal marking in that wild fish can be marked by holding fish in large immersion vats or raceways (Alaska Department of Fish and Game 2005).

Tetracycline exposure appears to be an inferior otolith-marking technique compared to temperature manipulation. Marks can be faint and difficult to distinguish, and the number of patterns is more limited; incident-light fluorescence microscopy is also required (Brothers 1990).

2.2.4 Otolith Marks

Otolith banding as an identification mark can be produced by exposing fish to cycles of high and low temperatures or alternating five-day periods of feeding and starvation (Buckley and Blankenship 1990). The method produces a permanent mark. The most practical use of this system is to identify large groups of fish from artificial production, which is especially useful in the management of terminal-area salmonid fisheries that harvest mixed stocks and where identification of groups can be effective in controlling exploitation rates (Volk et al. 1990). Advantages to otolith marking when it is necessary to assess early life stages where it is required to discriminate between experimental lots include: (1) It is applicable to the very youngest and smallest stages of all species, including embryos. (2) It produces a permanent mark. (3) It is accomplished in batches with minimal or no manipulation or handling of the fish. (4) Groups or lots can be uniquely marked (Brothers 1990). Disadvantages or limitations of otolith marking include: (1) Fish must be sacrificed to remove and examine otoliths or even to detect the presence of the mark unless there is an external marker such as an adipose clip. (2) Otolith marking does not allow recognition or coding of individuals. (3) The production of marks and the preparation of otoliths for viewing those marks require the development of special techniques and skills which go well beyond that required by most marking systems. (4) Otolith marking is not easily applied to the marking of wild fish in the field. The dry method of otolith marking is based on periodic changes of the water regime during incubation of the eggs. The eggs are dried in incubators, usually at 24-h intervals. One dark and one light ring are formed for each marking cycle during which the eggs are kept dry for 24 h and washed with water during the next 24 h. A disadvantage of the dry marking method is that it cannot be used for marking salmon larvae and fry. However, the method is simple, convenient, and requires no special equipment (2005, Alaska Department of Fish and Game 2005). The technique was developed in Russia where it is used extensively (Akinicheva and Rogatnykh 2000).

2.2.5 Genetic Marking

Genetic marking uses selective breeding to alter frequencies of alleles in the marked population so it can be distinguished from unmarked populations. Gharret and Seeb (1990) list the following factors necessary for consideration of marker alleles: (1) Information on the range and time of spawning and the sizes of the target population and the populations from which it is to be discriminated are needed to determine the utility of a mark. (2) Life history information is needed to determine the extent of follow-up marking necessary. (3) Select a relatively large brood stock so that genetic variability

will be sustained. (4) Adequate resources to mark the population and subsequently to detect the mark in mixtures. (5) Selection for single allele markers can produce optimum genetic marks.

2.3 Biometric Tags

2.3.1 PIT Tags

Passive integrated transponder (PIT) tags can be injected into juvenile salmon as small as 50 mm without jeopardizing growth or survival (Prentice et al. 1990). PIT-tag releases were successfully used to estimate survival and to estimate sampling variability of survival estimates for comparison with model-based variance estimates and to assess mixing of detected and nondetected Chinook salmon smolts (Skalski et al. 1998). Portable PIT readers have been developed (Destron Fearing Corporation) and field tested for use with 2.1 mm X 11.5 mm PIT tags on brown trout in shallow streams (Cucherousset et al. 2005). The detection range was 36 cm and 73.3 ± 5.8% to 93.3 ± 11.5% of age-0 trout were detected depending on the stream section. Advantages of PIT tags are the ability to tag large numbers of fish in the field, identify individual fish, expect high tag retention, experience tag longevity of around ten years, and have minimal impact on growth. A disadvantage of the system is the requirement that a tagged fish be within a distance of less than one meter of a tag interrogation system for successful detection of the signal.

2.3.2 Acoustic Tags

Numerous manufacturers produce acoustic tags for juvenile-size fish (Table 2.1, McMichael et al. 2010). The weight-in-water of acoustic tags ranges from 0.29 to 1.0 g in a survey conducted by McMichael et al. (2010). Since that review, tag size has further decreased, allowing tags to be implanted in juvenile fish using a hypodermic needle, rather than surgically. In the case of the JSATS tags (McMichael et al. 2010), each tag can transmit a uniquely coded 31-bit binary phase-shift keyed (BPSK) signal. Tags typically have a programmable pulse rate interval that affects both the detection probability and battery life. The shorter the pulse rate interval, the shorter the tag life. Current approach is to work with manufacturers and to tailor the tag fabrication for the size and tag-life requirements of the study. Autonomous acoustic receivers record the acoustic-tag signs for a distance up to 500 m. The receivers can be deployed individually or in transect arrangements spanning the width of the river or estuary. The buoyant receivers are mounted underwater in a variety of methods depending on water velocity, substrate type, and boat traffic (Titzler et al. 2010). Frequency of the acoustic sign varies with manufacturer but also varies deliberately between freshwater and saltwater applications. Welch and Jackson (2007) and Welch et al. (2011) describe applications of acoustic tags in saltwater studies of fish movement and survival.

2.3.3 Radio Tags

Adams et al. (2012) provide a review of radio telemetry methods in fisheries with special focus on salmonids. The radio- and acoustic-tag technologies in many ways are similar with regard to study design capabilities. A major tagging difference is that while the body of the radio tag is surgically implanted in the fish, there is an external antenna to transit the signal. Radio tags have unique identifier

codes similar to acoustic tags but the number of distinct codes is virtually limitless. Radio tags tend to work better in shallow and turbulent waters, where acoustic signals are difficult to transmit. Radio-tag signals can be difficult to receive in noisy industrial environments and not suitable for saltwater studies. Adams et al. (2012) provide a good tabular comparison of acoustic- and radio-tag capabilities (Table 2.3). Manufacturers produce a wide range of radio tags with varying size and battery life to accommodate fish size and study objectives. In the case of survival studies where absolute survival is to be estimated, tag-life studies should be performed to allow tag-life-adjusted survival estimates to be calculated. Program ATLAS (Active Tag-Life-Adjusted Survival; http://www.cbr.washington.edu/paramest/atlas/) can provide these adjustments for both acoustic- and radio-tag studies.

Table 2.2. Comparison of microacoustic transmitters currently available (January 2009) on the market [reprinted from McMichael et al. (2010)]. PRI = pulse repetition rate.

System	Model	Frequency (kHz)	PRI(s)	Weight in air (g)	Weight in water (g)	Dimensions (mm)	Tag life (d)	Detection range (m)	Power (dB)
JSATS	SS130[a]	416.7	3 5 10	0.43	0.29	5.2 × 12.0 × 3.7	22 30 55	300[b]	156
HTI	795s	307	3 16	0.65	0.34	6.7 × 16.4 × 6.7	15 28	up to 1000[c]	142
HI	795m	307	3 16	0.75	0.4	6.8 × 16.5 × 6.8	15 28	up to 1000[c]	142
HTI	795e	307	1.3 5.1 15.4	1.5	0.8	6.8 × 21.0 × 6.8	25 35	up to 1000[c]	148–150
Lotek	MAP6_1	200	1.3 5.1 15.4	0.9	0.6	6.2 × 13.0 × 6.2	7 24 51	NA	NA
Lotek	MAP6_2	200	1.3 5.1 15.4	1.1	0.7	6.2 × 15.0 × 6.2	4 14 34	NA	NA
Vemco	V7-1L	69	30	1.4	0.7	7 × 18 × 7	24	292[d]	136
Vemco	V7-2L	69	30	1.6	0.75	7 × 20 × 7	52	292[d]	136
Vemco	V7-4L	69	30	1.8	1.0	7 × 22.5 × 7	77	292[d]	136
Vemco	V9-6L	69	60	2.9	1.6	9 × 21 × 9	79	418[d]	142
Vemco	V9-6L[e]	69	60	3.1	2.0	9 × 20 × 9	120	418[d]	142
Sonotronics	PT-1	69–83	1.0	1.25	0.6	7.1 × 16 × 7.1	7	300	129–133
Sonotronics	PT-2	69–83	1.0	1.7	1.0	7.1 × 19 × 7.1	12	500	132–136
Sonotronics	PT-3	69–83	1.0	2.0	1.0	7.8 × 19 × 7.8	21	750	132–136

a Vendor: Advanced Telemetry Systems
b Distance at which 20% of expected detections were received in freshwater. Transmitters have been detected at distances of 800 m.
c As reported on HTI website.
d As provided by the range calculator on the Vemco website.
e From Rechisky et al. (2009).
f From vendor website.
NA – Not available from manufacturer.

Table 2.3. Comparison of radio- and acoustic-tag performance under different study conditions (reprinted from Adams et al. [2012]).

Characteristic	Acoustic	Radio
Site conditions		
Saline/high conductivity water (>600 EC)	Excellent	(Not usable)
Low conductivity water	Excellent	Excellent
Deep water (>20 m)	Excellent	Poor
Turbulent water	Poor	Excellent
Dense aquatic vegetation	Poor	Very good
In water obstructions	Poor	Poor to marginal
Turbid water	Good	Very good
Algae	Poor	Excellent
Thermocline/temperature gradient	Poor to marginal	Good
Ice	Poor	Good
Study species		
Fast/mobile species	Poor (mobile tracking) Good (stationary arrays)	Excellent
Long migrations	Poor (mobile tracking) Good (stationary arrays)	Excellent
Number of animals	Good	Good
Equipment		
Tracking options	Hydrophone in water Boat, stationary array	Antenna usually in air On foot, vehicle, boat, aircraft
Automated stations	Good	Excellent
3D positioning	Established methods	Difficult
Archival tags	Established methods	Possible
Directionality	Poor to fair	Good to excellent (dependent on frequency and antenna type)
Power usage	Good	Good

3 Survival Estimation

Sixteen different marking and release-recapture designs were examined to determine their utility in estimating fry and smolt survival. The objective of all the study designs was to estimate survival in the initial river reach or sampling period (i.e., S_1) of interest. These designs were examined in conjunction with either unique identifier marking methods or batch-marking techniques. Consideration included whether fish were either rereleased or not rereleased after capture. In other words, whether examination for marks required destructive (i.e., without rerelease) or nondestructive (i.e., with rerelease) sampling techniques to identify marked fish.

The most powerful and flexible design is the single release-recapture model with uniquely marked fish. In this scenario, each fish produces a complete capture history which can be used to estimate survival probabilities and detection probabilities in all reaches but the last. These survival estimate procedures are associated with PIT-tag and acoustic-tag studies.

Staggered-entry designs allow new fish to enter the study at downstream detection sites. The infusion of new fish into the design can improve estimation processes and/or allow survival to be estimable in case where it otherwise may not. These approaches can be used with both batch-marked and unique-identifier tags.

Similar in appearance to the staggered-entry designs are the paired release-recapture designs. In these approaches, fish are released above and below the river reach of interest with subsequent recaptures downstream. Emphasis of this design is estimation of survival in the first reach. However, estimation of survival downriver is also possible, depending on the marking and recapture approach used in the study. This paired-release design can be used with either unique identifier or batch-mark technique.

The final release-recapture designs considered are the release-remark-rerelease designs. In this study approach, batch-marked fish are released at the top of the river reach of interest. First-time recaptured fish are given a second mark for subsequent identification. Should this fish be recaptured a second time, it is removed from the study. Two alternative protocols using the partial capture history data are reviewed.

For each of the 16 study protocols reviewed , the ability to estimate survival in the first one or few reaches was examined based on the properties of minimum sufficiency and separability of parameters. In other words, the protocols were examined to determine whether there was sufficient information permitting survival estimation or not. For those models that provided a valid means of estimation, details are presented.

Of the 16 different combinations of marking and release-recapture designs evaluated for fry survival studies, 11 approaches provided estimates of survival for one or more reaches (Table 3.1). Five of the feasible approaches required uniquely marked individuals. Four of the other feasible approaches used multiple batch marks. The last two feasible approaches require applying an additional batch mark

to fry or smolts recaptured and rereleased. None of the methods which relied on a single common batch mark to identify study fish provided a valid means of estimating fry or smolt survival. Subsequent discussion describes in greater detail the statistical and logical approaches of the methods capable of estimating survival.

3.1 Model M_1: Single release – individual marks – nondestructive sampling (Scenario 2)

This study design with uniquely marked fish and nondestructive sampling provides maximum estimation capability. (Nondestructive sampling means the fish do not have to be sacrificed to read the mark.) The single release with subsequent downriver recapture and rereleases permits survival and capture probabilities to be uniquely estimated in all reaches but the last (Figure 3.1). Only the joint probability of surviving and being detected (i.e., $\lambda = Sp$) can be estimated for the last reach. The model is a special case of the full capture history model of Burnham et al. (1987:112-116) when only one of release in a paired-release is considered. Skalski et al. (2001) has applied the model to estimate salmonid smolt survival using PIT tags in the Columbia River. The summary detections are the number of fry in each of the 2^k possible capture histories in a k-reach investigation.

The statistical model (Burnham et al. 1987; Skalski et al. 1998) provides closed-form estimators for the survival and capture probabilities. Burnham et al. (1987) provides two goodness-of-fit statistics called T_2 and T_3 that can be used to assess whether upstream detection history has an effect on subsequent downstream survival. The release-recapture design has also received considerable attention where survivals are subsequently regressed against environmental covariates to study the survival relationships (Lebreton et al. 1992; Skalski et al. 1993). Two statistical software packages, SURPH (http://www.cbr.washington.edu/paramEst/SURPH/) and SURGE (http://www.phidot.org/software/surge/surge.html) can be used to provide survival estimates, standard errors, and subsequent survival analyses. Hoffmann and Skalski (1995) extended the model to examine the relationship between individual covariates and survival and detection processes. Program SURPH allows regression analyses using both group covariates and individual-based covariates.

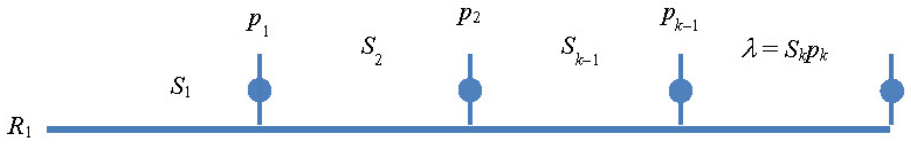


Figure 3.1. Schematic of Model M_1 using a single release of uniquely marked individuals and nondestructive sampling. Using this method, survival (*S*) can be estimated in all but the last reach (Burnham et al. 1987, Skalski 1998). • denotes rerelease/nondestructive sampling. *R* refers to the release location, S_i refers to survival rate, and pi refers to probability of detection, lamda is the joint probability of surviving and being detected in the last sampling reach.

Table 3.1. Alternative approaches to conducting fry survival studies and their ability to provide valid estimates of reach survival. Marking and release-recapture scenarios identified in parentheses.

Scenario	Survival estimable	Model
I. Single release-recapture		
A. Unique individual marks		
1. Destructive sampling (1)	No	--
2. Nondestructive sampling (2)	Yes	M_1
B. Common batch mark		
1. Destructive sampling (3)	No	--
2. Nondestructive sampling (4)	No	--
II. Staggered entry		
A. Unique individual marks		
1. Destructive sampling (5)	Yes	M_2
2. Nondestructive sampling (6)	Yes	M_3
B. Common batch mark		
1. Destructive sampling (7)	No	--
2. Nondestructive sampling (8)	No	--
C. Unique batch marks		
1. Destructive sampling (9)	Yes	M_2
2. Nondestructive sampling (10)	Yes	M_4
III. Paired release		
A. Unique individual marks		
1. Destructive sampling (11)	Yes	M_5
2. Nondestructive sampling (12)	Yes	M_6
B. Unique batch marks		
1. Destructive sampling (13)	Yes	M_5
2. Nondestructive sampling (14)	Yes	M_7
IV. Single release – remark – rerelease		
A. Two batch marks (15)	Yes	M_8
B. Multiple batch marks (16)	Yes	M_9

3.2 Model M_2: Staggered entry – individual or unique batch marks – destructive sampling (Scenarios 5, 9)

Destructive sampling to examine individual fry for marks results in no individual being recaptured more than once during the course of the study. (Destructive sampling means the fish has to be sacrificed to obtain the tag identification.) For this reason, there is no effective advantage of unique marks over that of batch-specific marks. It is adequate to simply be able to identify a fry to a specific release group in this staggered-entry design. Hence, whether individual or batch marks are used, the statistical model is the same.

To estimate survival in the first reach, marked fry must be released upstream and sampled at a minimum of two downstream locations. Fry captured at the first downstream sampling location are examined for marks and the number enumerated. At this site, a new and distinctive batch of fry are released. Both the initial (R_1) and secondary (R_2) releases are then susceptible to destructive sampling at a second downstream site (Figure 3.2). To estimate survival in additional reaches, new and distinctive batches of marked fry must also be released at subsequent detection sites. At least one detection site must exist below the last river reach of interest. Survival cannot be estimated in that last reach.

The likelihood model for a three-reach design with staggered entry only at the first downstream recapture location can be expressed as a product of two multinomial distributions, where

$$\begin{aligned} L\left(S_1, p_1, \theta_1, \theta_2 \,|\, \underset{\sim}{x}, \underset{\sim}{y}\right) = & \\ & \binom{R_1}{\underset{\sim}{x}} \left(S_1 p_1\right)^{x_1} \left(S_1(1-p_1)\theta_1\right)^{x_2} \left(S_1(1-p_1)\theta_2\right)^{x_3} \\ & \cdot \left(1 - S_1 p_1 - S_1(1-p_1)(\theta_1+\theta_2)\right)^{R-x_\cdot} \\ & \cdot \binom{R_2}{\underset{\sim}{y}} \theta_1^{y_1} \theta_2^{y_2} \left(1-\theta_1-\theta_2\right)^{R_2-y_\cdot} \end{aligned} \tag{1}$$

Where

S_i = probability of fry recovery in the i th reach $(i = 1, \ldots, 3)$;

p_i = probability a fry is recovered at the i th recovery site $(i = 1, \ldots, 3)$;

θ_1 = $S_2 p_2$;

θ_2 = $S_2(1-p_2)S_3 p_3$;

x_i = number of fry recovered at the i th recapture site $(i = 1, \ldots, 3)$ for the first release of size R_1;

y_i = number of fry recovered at the i th recapture site $(i = 1, \ldots, 3)$ for the second release of size R_2.

The likelihood model has four parameters and four minimum sufficient statistics, permitting closed-form estimators. Because there are only two staggered entries, only survival in the first reach between the two release locations can be estimated, where

$$\hat{S}_1 = \frac{R_2(x_2 + x_3) + x_1(y_2 + y_3)}{R_1(y_2 + y_3)} \tag{2}$$

$$\hat{p}_1 = \frac{x_1(y_2 + y_3)}{R_2(x_2 + x_3) + x_1(y_2 + y_3)} \tag{3}$$

$$\hat{\theta}_1 = \frac{x_2 + y_2}{R_1\hat{S}_1(1 - \hat{p}_1) + R_2} \tag{4}$$

$$\hat{\theta}_2 = \frac{x_3 + y_3}{R_1\hat{S}_1(1 - \hat{p}_1) + R_2} \tag{5}$$

The variance of $\hat{S}_1$ can be estimated using the delta method, where

$$\mathrm{Var}(\hat{S}_1) \doteq \frac{S_1}{R_1(\theta_1 + \theta_2)} \cdot \left[1 - S_1(\theta_1 + \theta_2) - p_1(1 - \theta_1 - \theta_2) + R_2(\theta_1 + \theta_2)^2(1 - \theta_1 - \theta_2)S_1(1 - p_1)^2\right] \tag{6}$$

and the variance estimated by substituting in the parameter estimates.

Assumptions of Model M_2 include the following:

1. All fry have equal and independent fates.
2. Marked fry are correctly identified and designated to the correct release group.
3. Release groups have equal downstream survival probabilities.
4. Release groups have equal downstream detection probabilities.

Goodness-of-fit to model M_1 can be tested using an R x C contingency table test (Zar 1999) of the form:

		Release group	
		R_1	R_2
Recovery Site	2nd	x_2	y_2
	3rd	x_3	y_3

(7)

with one degree of freedom. Program USER (http://www.cbr.washington.edu/paramEst/USER/) can be programmed to numerically analyze likelihood model (1) and other special cases of the staggered-entry design.

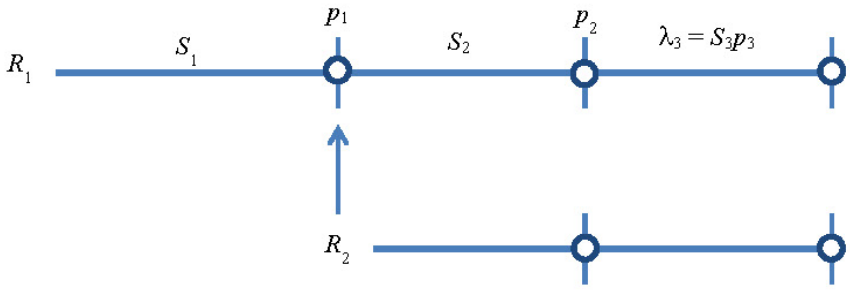


Figure 3.2. Schematic of Model M_2 using staggered entry with uniquely marked individuals or batch marks and destructive sampling. Using this method, survival (S) can be estimated only between staggered entry locations R_1 and R_2. ○ denotes removal/destructive sampling.

3.3 Model M_3: Staggered entry – individual marks – nondestructive sampling (Scenario 6)

This staggered-entry design using uniquely marked individuals and rerelease of captured individuals is the release-recapture model of Cormack (1964). This model is also a special case of the Jolly (1965) - Seber (1965) model where only numbers of marked animals recaptured and released are recorded, and mark-to-unmark ratios ignored.

Unique survival and capture probabilities can be estimated for all but the last reach. In the last reach, only the joint probability of surviving and being capture (i.e., $\lambda = S_k p_k$) at the last location can be estimated (Figure 3.3). Although closed-form estimation for the survival and capture probabilities exist, statistical software such as SURPH, SURGE, or SURVIVE can be used to numerically estimate the parameters and standard errors. Program SURPH will provide profile likelihood confidence intervals.

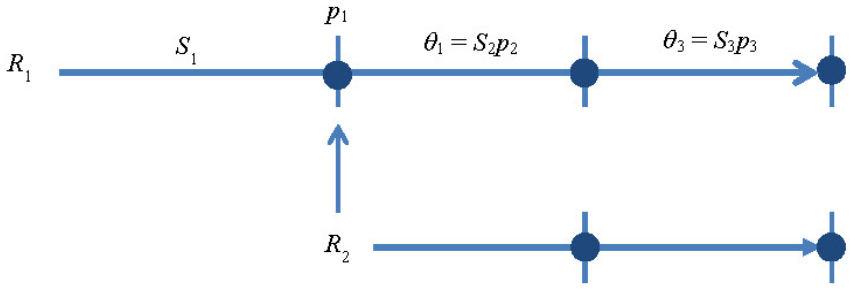


Figure 3.3. Schematic of Model M_3 using a staggered entry with uniquely marked individuals and nondestructive/rerelease sampling. Using this method, survival (S) can be estimated for all reaches but the last. • denotes rerelease/nondestructive sampling.

3.4 Model M_4: Staggered entry – unique batch marks – nondestructive sampling (Scenario 10)

In this variation of the staggered-entry design, survival can be estimated between release sites for all but the last reach. In Figure 3.4, only the uppermost reach is available for survival estimation. The nondistributive sampling, combined with batch-level marking, results in capture data that is no longer mutually exhaustive and exclusive. For example, fry first detected at recapture location 2 cannot be distinguished from fry first recaptured at location 3.

The likelihood model describing the staggered-entry release-recapture design of Figure 3.4 can be parsimoniously written as follows:

$$
\begin{aligned}
L\left(S_1, p_1, \gamma_1, \gamma_2 \,|\, \underset{\sim}{x}, \underset{\sim}{y}\right) = & \\
& \binom{R_1}{x_1} \left(S_1 p_1\right)^{x_1} \left(1 - S_1 p_1\right)^{R_1 - x_1} \binom{R_1}{x_2} \left(S_1 \gamma_1\right)^{x_2} \left(1 - S_1 \gamma_1\right)^{R_1 - x_2} \\
& \cdot \binom{R_1}{x_3} \left(S_1 \gamma_2\right)^{x_3} \left(1 - S_1 \gamma_2\right)^{R_1 - x_3} \binom{R_2}{y_2} \gamma_1^{y_2} \left(1 - \gamma_1\right)^{R_2 - y_2} \\
& \cdot \binom{R_2}{y_3} \gamma_2^{y_3} \left(1 - \gamma_2\right)^{R_2 - y_3},
\end{aligned}
\tag{8}
$$

where

$\gamma_1 = S_2 p_2$,

$\gamma_2 = S_2 S_3 p_3$.

The likelihood has four parameters $\left(S_1, p_1, \gamma_1, \gamma_2\right)$ and five minimum sufficient statistics, requiring numerical estimation. Program USER can be readily programmed to estimate the model parameters, standard errors, and profile likelihood confidence intervals.

The model assumptions include the following:

1. All fry have equal and independent fates.
2. Marked fry are correctly identified and designated to the correct release group.
3. Release groups have equal downstream survival probabilities.
4. Release groups have equal downstream detection probabilities.

Goodness-of-fit to Model M_4 can be tested using the 2 x 2 contingency table test (7).

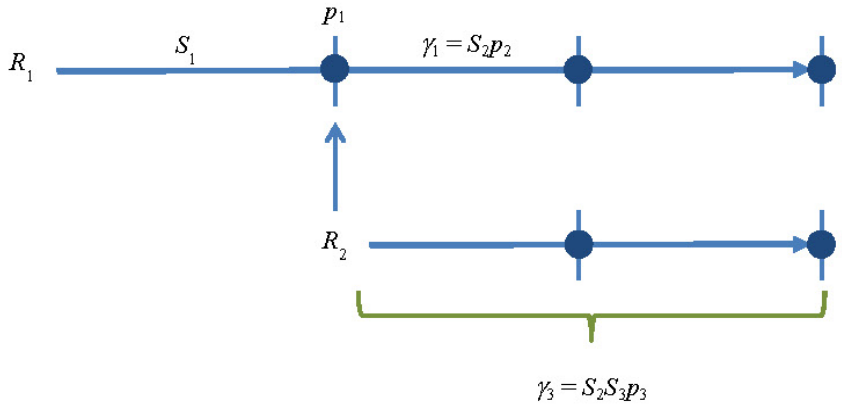


Figure 3.4. Schematic of Model M_4 using staggered entry with unique batch marks and nondestructive sampling. Using this method, survival (*S*) can be estimated only for the reaches between batch releases. • denotes rerelease/nondestructive sampling.

3.5 Model M_5: Paired-release – individual marks or unique batch marks – destructive sampling (Scenarios 11, 13)

The destructive sampling to identify fry and designate the fry to specific batches eliminates the possibility of capturing a fish more than once. Hence, whether a fry is individually marked or simply batch marked does not change the nature of the recorded data (Figure 3.5). This model was first recommended by Ricker (1958) and is sometimes referred to as the relative recovery method. Burnham et al. (1987:78-84) designated the approach as the "first capture history" method.

The general likelihood model for this paired design, regardless of the number of downstream recovery sites, can be written as:

$$L(S_1, \varphi | x_\bullet, y_\bullet) = \binom{R_1}{x_\bullet} (S_1\varphi)^{x_\bullet} (1 - S_1\varphi)^{R_1 - x_\bullet} \binom{R_2}{y_\bullet} \varphi^{y_\bullet} (1-\varphi)^{R_2 - y_\bullet}, \quad (9)$$

where

φ = probability of a fry surviving from release location R_2 and being recaptured downstream,

$x_\bullet = \sum_{i=1}^{k} x_i$ = total number of R_1 fry recovered downstream,

$y_\bullet = \sum_{i=1}^{k} y_i$ = total number of R_2 fry recovered downstream.

The model has two parameters (S_1, φ) and two minimum sufficient statistics, permitting closed-from estimators.

Survival in the first reach can be estimated by the quotient

$$\hat{S}_1 = \frac{R_2 x_\cdot}{y_\cdot R_1}, \tag{10}$$

$$\varphi = \frac{y_\cdot}{R_2} \tag{11}$$

The survival estimator has the sampling variance of

$$\mathrm{Var}\left(\hat{S}_1\right) = \frac{S_1}{\varphi}\left[\frac{(1-S_1\varphi)}{R_1} + \frac{S_1(1-\varphi)}{R_2}\right], \tag{12}$$

which can be estimated by

$$\widehat{\mathrm{Var}}\left(\hat{S}_1\right) = \hat{S}_1^2\left[\frac{1}{x_\cdot} - \frac{1}{R_1} + \frac{1}{y_\cdot} - \frac{1}{R_2}\right]. \tag{13}$$

The assumptions of Model M_5 are essentially the same as those previously stated for Models M_2 and M_4. However, the dimensionality of the model does not permit an independent test of model assumptions based on the summaries $x_\cdot$ and $y_\cdot$. Instead, the assumption of shared probability φ can be tested on the basis of the arrival patterns of the release groups to the downstream detection sites.

Either a chi-squared test of homogeneity (Zar 1999:488-491) or a Kolmogorov-Smirnov test of homogeneous distribution (Conover 1980:368-377) can be used to assess whether arrival timing was the same for both release groups. The inference from the tests is that if the release groups arrived downstream at the same time(s), they experienced the same recapture environment and capture probabilities. These tests of homogeneity cannot, however, discern differential survival probabilities among release groups.

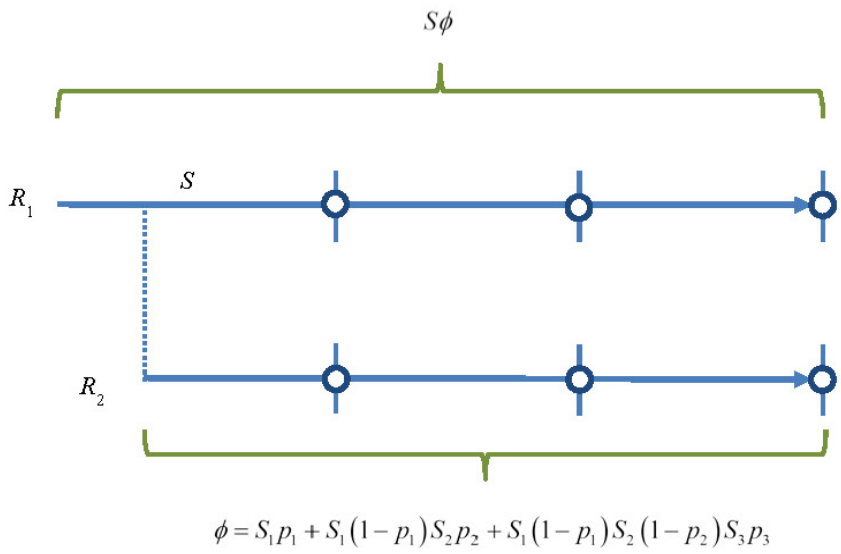


Figure 3.5. Schematic of Model M_5 using a paired release with unique individual marks or unique batch marks and destructive sampling. Using this method, survival (S) can only be estimated between release locations of R_1 and R_2. ○ denotes removal/destructive sampling.

3.6 Model M_6: Paired release – individual marks – nondestructive sampling (Scenario 12)

This model is an extension of Scenario 2 described by Burnham et al. (1987:112-129) as the "complete capture history" model. In essence, each release group functions as an independent, single release-recapture model with uniquely marked individuals that are nondestructively sampled (Figure 3.6). Release R_1 estimates survival S_{11} and release R_2 estimates survival S_{21} (Figure 3.6). Then the survival in the reach between release locations is estimated by the quotient

$$\hat{S} = \frac{\hat{S}_{11}}{\hat{S}_{21}} \tag{14}$$

with associated variance estimator

$$\widehat{\mathrm{Var}}\left(\hat{S}\right) = \hat{S}^2\left[\frac{\mathrm{Var}\left(\hat{S}_{11}\right)}{\hat{S}_{11}^2} + \frac{\mathrm{Var}\left(\hat{S}_{21}\right)}{\hat{S}_{21}^2} - \frac{2\,\mathrm{Cov}\left(\hat{S}_{11},\hat{S}_{21}\right)}{\hat{S}_{11}\hat{S}_{21}}\right]. \tag{15}$$

With multiple downstream detection sites, sequential model testing and Akaike information criterion (AIC) (Burnham and Anderson 1998) can be used to identify the most parsimonious statistical model to describe the joint releases. The preferred model would share common downstream detection and

survival rates where the values are equal, thereby improving the precision of the remaining model parameters.

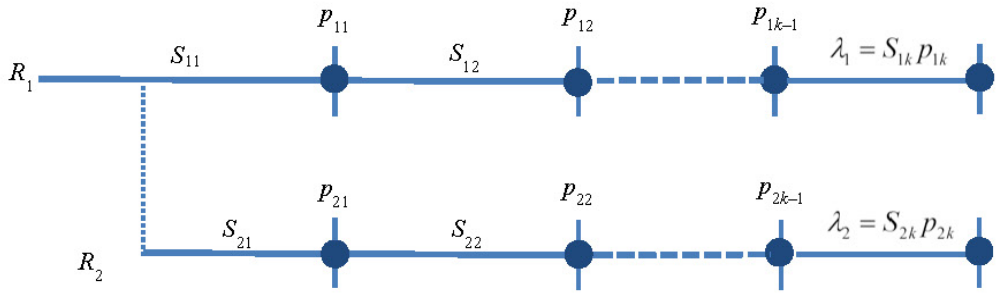


Figure 3.6. Schematic of Model M_6 using a paired release with unique individual marks and nondestructive sampling. Survival (*S*) can be estimated for each reach and detection location except the last. Survival between release locations is estimated as the quotient, $\hat{S}_{11}/\hat{S}_{21}$. • denotes rerelease/nondestructive sampling.

3.7 Model M_7: Paired release – batch marks – nondestructive sampling (Scenario 14)

In this approach, each release group receives a different batch mark that does not distinguish between individuals. Fry are recaptured downstream at one or more downstream locations are rereleased without further marking (Fig. 4.7). Hence, a fish may be caught multiple times without the investigator's knowledge. Burnham et al. (1987:100-105) designated this approach as the "unknown capture history" method. The method is complicated by the fact that individual fish cannot be categorized into mutually exclusive and exhaustive capture histories.

The joint likelihood model for the paired releases can be written as

$$L = \prod_{i=1}^{k} \binom{R_1}{x_i} (S_1\theta_i)^{x_i} (1 - S_1\theta_i)^{R_1 - x_i} \cdot \prod_{i=2}^{k} \binom{R_2}{y_i} \theta_i^{y_i} (1 - \theta_i)^{R_2 - y_i} \qquad (16)$$

where

x_i = number of R_1 fish recaptured and re-released at the *i*th recapture location $(i = 1, \ldots, k)$;

y_2 = number of R_2 fish recaptured and re-released at the *i*th recapture location $(i = 1, \ldots, k)$;

θ_i = joint probability of surviving to and being captured at the *i*th recapture location $(i = 1, \ldots, k)$ for R_2 fish.

Model (16) is appropriate as long as all recaptured fish are re-released alive (i.e., no handling mortality) or handling mortality is independent of release group (Burnham et al. 1987:106). Burnham et al. (1987) suggest using an R × C contingency table to determine whether loss rates are homogeneous between release groups. The method of moments estimator for *S* is Eq. (10) with variance estimator (12). However, Burnham et al. (1987:105) suggest the slightly "better" variance formula

$$\widehat{\mathrm{Var}}\left(\hat{S}\right)=\hat{S}^2\left\{\frac{1}{x_\bullet}-\frac{1}{R_1}\left[\sum_{j=2}^{k}\left(\frac{x_j}{x_\bullet}\right)^2\right]+\frac{1}{y_\bullet}-\frac{1}{R_2}\left[\sum_{j=2}^{k}\left(\frac{y_j}{y_\bullet}\right)^2\right]\right\}, \tag{17}$$

where

x_j = number of fish in release R_1 detected at recapture site $j\ (j=2,\ldots,k)$;

y_j = number of fish in release R_2 detected at recapture site $j\ (j=2,\ldots,k)$.

Burnham et al. (1987:104) generally do not recommend this study approach because of the model nonspecificity problems and recommend instead the use of the first capture history protocols (i.e., Model M_5) if feasible. However, if survival of the study fish is important as in the case of listed (endangered) species, then this method is performed.

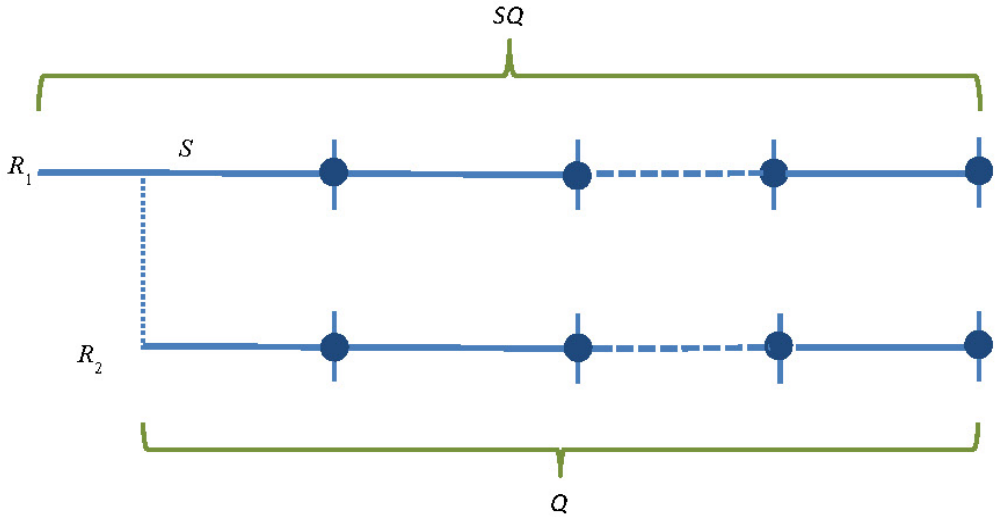


Figure 3.7. Schematic of Model M_7 using a paired release with batch-specific marks and nondestructive sampling. This method can only estimate survival (*S*) between release locations based on quotient of relative detections between release groups. • denotes rerelease/nondestructive sampling.

3.8 Model M_8: Single release-remark-rerelease – two batch marks

This scenario falls under the general category of "partial capture history" methods of Burnham et al. (1987:146-172). There are numerous ways of implementing this general procedure. Each variation has its own likelihood model and associated survival estimators. The general process beings with single release of a common batch-marked group of fish. Upon first recapture, the fish acquire an additional mark and are subsequently re-released. Upon second recapture, the fish are removed from the population. Burnham et al. (1987) describe two alternative schemes A and B. In Scheme A, a fish is given a second mark that is site-specific with mark-releases occurring at all site locations but the last. In Scheme B, a fish is given a second mark if recaptured at the first downstream recapture site. At all other locations, the fish is simply examined for the marking code(s) and removed (Figure 3.8).

Scheme B is the simplest to implement, requiring just two distinguishing markers and, consequently, will be discussed first. Define the following terms:

R_1 = number of fish initially released,

S_1 = probability of survival in the reach between release R_1 and the first downstream recovery site,

p_1 = probability of capture at the first recovery site,

λ = probability a fish survives below the first recovery site and is recaptured somewhere downstream,

x_1 = number of fish recaptured at the first recovery site,

x_{23} = number of fish recovered for the first time at the second or subsequent recovery sites,

R_2 = number of fish among x_1 that are given a second mark and rereleased,

y_{23} = number of double-marked fish from R_2 that are subsequently recovered.

The likelihood model for the release-remark-rerelease method can then be written as follows:

$$L = \binom{R_1}{x_1, x_{23}} (S_1 p_1)^{x_1} \left(S_1 (1-p_1)\lambda\right)^{x_{23}} \left(1 - S_1 p_1 - S_1 (1-p_1)\lambda\right)^{R_1 - x_1 - x_{23}} \cdot \binom{R_2}{y_{23}} \lambda^{y_{23}} (1-\lambda)^{R_2 - y_{23}}. \quad (18)$$

It should be noted that Model (18) is a compressed version of Model (1), yielding essentially the same survival estimator. The maximum likelihood estimates are

$$\hat{S}_1 = \frac{R_2 x_{23} + x_1 y_{23}}{R_1 y_{23}}$$
$$\hat{p}_1 = \frac{x_1 y_{23}}{R_2 x_{23} + x_1 y_{23}} \qquad (19)$$
$$\hat{\lambda} = \frac{y_{23}}{R_2}.$$

The variance of $\hat{S}_1$ is approximated by the delta method to be

$$\mathrm{Var}\left(\hat{S}_1\right) \doteq \frac{S_1}{R_1 \lambda}\left[1 - S_1\lambda - p_1(1-\lambda) + R_2\lambda^2(1-\lambda)S_1(1-p_1)^2\right], \qquad (20)$$

with variance estimated by substituting the MLEs into Eq. (20).

The key assumptions of this release-remark-rerelease method are the following:

1. All fish have equal and independent probabilities of survival and capture.
2. Marking and remarking have no effect on survival and recapture.

For these assumptions to be true, the recapture and remarking techniques at the first downstream recovery site must be benign. For this protocol, only survival in the first reach can be estimated. A goodness-of-fit test can be constructed, using an R × C contingency-table test of homogeneity of the recovery counts at the removal sites for single- and double-marked fish of the form:

Recovery	Single mark	Double mark
Site 2	x_2	y_2
Site 3	x_3	y_3
⋮	⋮	⋮

analogous to (7).

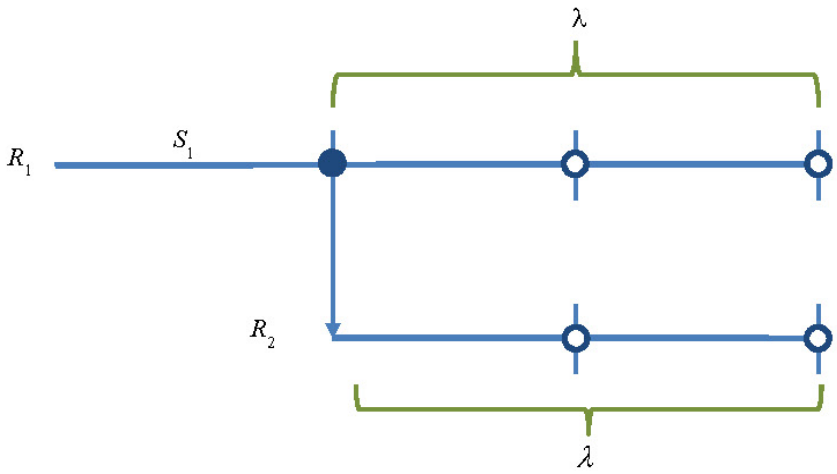


Figure 3.8. Schematic of Model M_8 using a release-remark-rerelease method. An initial release of batch-marked fish (R_1) with remark-release capabilities at the first recovery site and removal sampling only. Fish caught at the initial site are given a second mark and rereleased (R_2). • denotes rerelease/nondestructive sampling; ○ denotes removal/destructive sampling.

3.9 Model M_9: Single release-remark-rerelease – multiple batch marks

In the previous method (i.e., Model M_8), recaptured fish were remarked at only the first downstream recovery site. This allows estimation of survival only between the initial release location and the first detection site. However, if first-time recaptured fish are given a site-specific second mark, then survival can be estimated in all reaches but the last. This method is designated as Scheme A in Burnham et al. (1987:149-349). In this approach, any fish recaptured a second time (i.e., with two marks) is removed from the population (Fig. 4.9). In the case of *k* reaches, there needs to be *k* uniquely identified batch marks that can be applied two at a time. Consequently, the logistics of multiple batch marks and multiple remarking locations add complexity beyond the simple Scheme B described earlier.

In the case of three reaches, the joint likelihood model for the release-remark-rerelease scheme can be written as follows:

$$
\begin{aligned}
L = & \binom{R_1}{\underset{\sim}{x}} (S_1 p_1)^{x_1} \left(S_1(1-p_1)S_2 p_2\right)^{x_2} \left(S_1(1-p_1)S_2(1-p_2)\lambda\right)^{x_3} \\
& \cdot \left(1 - S_1 + S_1(1-p_1)\left[(1-S_2) + S_2(1-p_2)(1-\lambda)\right]\right)^{R_1 - x_\cdot} \\
& \cdot \binom{R_2}{\underset{\sim}{y}} (S_2 p_2)^{y_2} \left(S_2(1-p_2)\lambda\right)^{y_2} \left(1 - S_2 + S_2(1-p_2)(1-\lambda)\right)^{R_2 - y_\cdot} \\
& \cdot \binom{R_3}{\underset{\sim}{z}} \lambda^{z_3} (1-\lambda)^{R_3 - z_3},
\end{aligned}
\tag{21}
$$

where

R_i = number of fish released at the *i*th release location;

x_i = number of R_1 fish caught for the first time at the ith recovery location $(i = 1, \ldots, 3)$;

$\sum_{i=1}^{3} x_i = x_\cdot$;

y_i = number of R_2 fish caught for the first time at the *i*th recovery location $(i = 2, 3)$;

$\sum_{i=1}^{3} y_i = y_\cdot$;

z_i = number of R_3 fish caught for the first time at the *i*th recovery location $(i = 3)$;

S_i = probability of survival in the *i*th reach $(i = 1, 2)$;

p_i = probability of recapture at the *i*th recovery site $(i = 1, 2)$;

$\lambda = S_3 p_3$ = joint probability of surviving the last reach and being detected.

The maximum likelihood estimators are as follows:

$$\hat{\lambda} = \frac{z_3}{R_3}$$

$$\hat{p}_2 = \frac{(x_2 + y_2) z_3}{(x_2 + y_2) z_3 + (x_3 + z_3) R_3}$$

$$\hat{p}_1 = \frac{x_1}{x_1 + \dfrac{(x_2 + x_3)}{\hat{S}_2\left(\hat{p}_2 + (1 - \hat{p}_2)\hat{\lambda}\right)}}$$

$$\hat{S}_2 = \frac{y_2 + y_3}{R_2\left(\hat{p}_2 + (1-\hat{p}_2)\hat{\lambda}\right)}$$

$$\hat{S}_1 = \frac{x_1}{R_1} = \frac{x_2 + x_3}{R_1\hat{S}_2\left(\hat{p}_2 + (1-\hat{p}_2)\hat{\lambda}\right)}.$$

The model assumptions are essentially the same of those of Model M_8. Again, R × C contingency-table tests of homogeneity of downstream recovery patterns can be used as a test of goodness of fit.

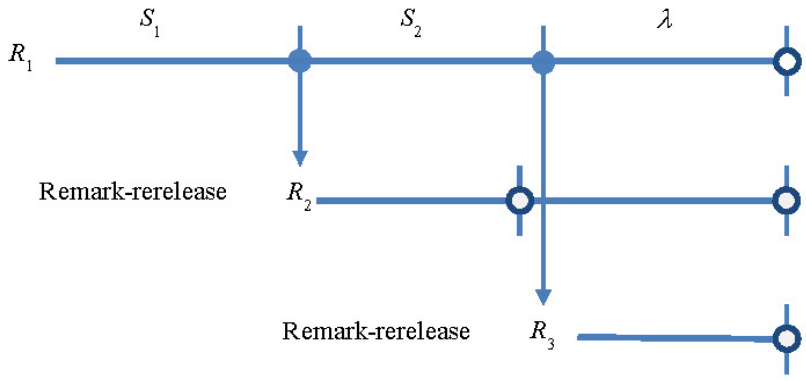


Figure 3.9. Schematic of Model M_9 using a release-remark-rerelease method with multiple-batch marks. First-time recaptured fish from release R_1 receive a second site-specific mark. All fish recaptured for the third time are removed from the population. • denotes rerelease/nondestructive sampling; ○ denotes removal/destructive sampling.

3.10 Summary of Survival Study Options

The choice of design for the fry survival study will depend on a number of considerations, including:

1. Marking capability and ability to read mark(s).
2. Recovery methods.
3. Desired precision and sample size requirements.
4. Model assumptions.

The art of implementing a successful fry survival study will be in the integration of these interrelated demands and constraints.

Typically, estimation precision will be improved the more detailed the release-recapture data. This means using unique fish identification methods will be preferable to batch marks, all else being equal. No release-recapture method is feasible with a single batch mark. The necessity to use multiple release groups or the ability to obtain partial capture history data from double marking fish is required at a minimum. However, double marking fish (i.e., M_8 and M_9) can result in undue stress on the rereleased individuals, biasing estimation techniques.

The more detailed release-recapture data permits tests of model assumptions often unavailable in simpler procedures and also allows more model parameters to be estimated, including capture rates and multiple reach survival estimates.

The choice between a single-release and a paired-release approach depends on more than logistical convenience. In a single-release, any post-release handling mortality will be incorporated in the survival estimates for the first one or few reaches. Paired-release models potentially eliminate this source of bias, assuming both upstream and downstream release groups experience similar handling effects. It should be noted that all of the single-release methods presented here can be arranged as a paired release to estimate survival in the intervening reach between initial release locations (Burnham et al. 1987). The presence and degree of post-release handling mortality should therefore be taken into account when selecting between single and paired releases.

All of the model options presented in this report can be readily programmed to provide survival estimates using Program USER (http://www.cbr.washington.edu/paramest/user/)./ The software provides a flexible model-building capability to determine the estimability of the approach and also provides estimates of survival and associated standard errors. Determining the estimability of the model should be a necessary first step in any well-designed, release-recapture investigation. Program SampleSize (http://www.cbr.washington.edu/analysis/apps/samplesize) provides a convenient means to perform sample size and precision calculations for a variety of alternative release-recapture designs used in estimating fish survival.

4 Movement and Migration

One objective of estuary RME is to assess movement rates of salmonids at landscape (1-10s km) and estuary-wide scales. Movement models where fish have alternative paths of travel fall under the category of multistate or branching models for mark-recapture studies. Ideally, these models are capable of estimating the probability of a fish selected a particular path of travel (e.g., entering an estuary) unconfounded by survival or detection probabilities. The ability to distinguish movement, survival, and detection processes requires unique-identifier tags in order to obtain full detection histories through a complex of detection arrays specifically designed for this purpose. Consequently, inert tags or batch-marked tags will have limited utility for purposes of estimating movement parameters.

4.1 Inert Tags

A single release of batch-marked fish prior to an estuary[2] opening (Figure 4.1) with detection/recapture in the estuary mouth can only estimate the joint probability of survival, movement to the entrance and detection, where

$$E\left(\frac{x}{R}\right) = S_0 \psi p_{IN} \tag{22}$$

where

S_0 = probability of survival from release to the estuary;

ψ = probability of a fish surviving to the estuary, entering the estuary;

p_{IN} = probability of detection at the mouth of the estuary upon entrance;

x = number of marked fish detected;

R = number of marked fish released.

In other words, the study can only estimate the joint probability of surviving, entering, and being detected at the estuary mouth. This assumes the detection process only occurs upon entrance, based on the configuration of the collection device.

[2] As used here, the term "estuary" refers to an off-channel area or wetland that is connected to the main stem Columbia River.

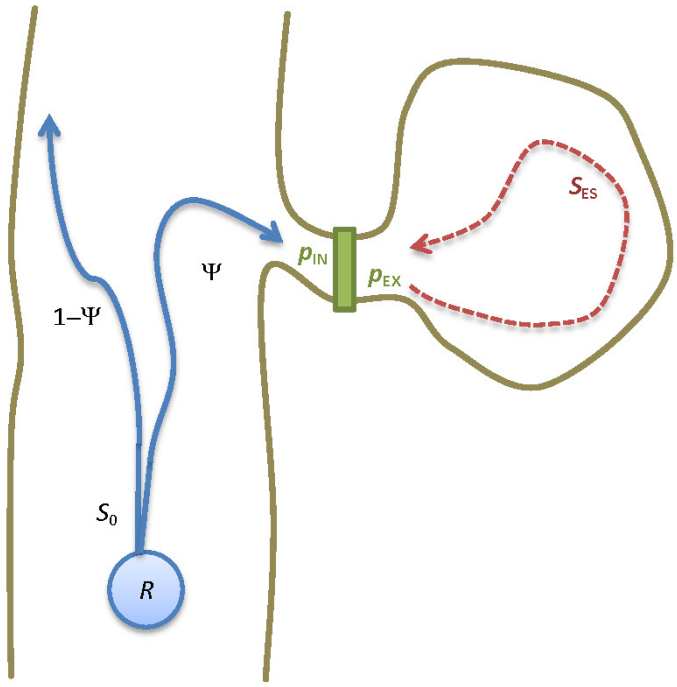


Figure 4.1. Schematic of a batch release of marked fish with associated movement (Ψ), survival (S_0, S_{ES}), and detection (p_{IN}, p_{EX}) probabilities for a study of estuary entrance efficiency.

Should the collection/detection facility at the mouth of the estuary detect fish upon entrance (p_{IN}) or exit (p_{EX}), then the joint probability of finding marked fish at the estuary becomes

$$E\left(\frac{x}{R}\right) = S_0\psi\left[p_{IN} + (1-p_{IN})S_{ES}p_{Ex}\right] \quad (23)$$

where

S_{ES} = probability of surviving while within the estuary, given the fish will eventually leave the estuary;

p_{EX} = probability of detection at the mouth of the estuary upon exit.

In this case, the probability of movement (ψ) into the estuary is even more confounded by other survival and detection processes. Consequently, a single batch release of marked fish has very limited value in understanding movement into estuaries in the Lower Columbia River.

4.2 PIT Tags

Should the entrance to the estuary be shallow, relatively narrow and well-defined, a pair of PIT-tag flat-plate detectors might be used to estimate the detection probability at the entrance to the estuary (Figure 4.2). Estimating entrance detection probability $(\hat{p}_{\text{IN}})$ would allow the estimation of the joint probability of survival to and entering the estuary based on Equation (22), when

$$E\left(\frac{x}{R\hat{p}_{\text{IN}}}\right) \doteq S_0\psi. \tag{24}$$

However, isolation of ψ would still not be possible. The joint likelihood model for this estimate can be written as

$$\begin{aligned} L = &\binom{R}{x}(\theta p_{\text{IN}})^x (1-\theta p_{\text{IN}})^{R-x} \\ &\cdot\binom{x}{n}\left(\frac{p_1 p_2}{1-(1-p_1)(1-p_2)}\right)^{n_{11}}\left(\frac{p_1(1-p_1)}{1-(1-p_1)(1-p_2)}\right)^{n_{10}}\left(\frac{(1-p_1)p_2}{1-(1-p_1)(1-p_2)}\right)^{n_{01}}, \end{aligned} \tag{25}$$

where

$\theta = S_0\psi$;

$p_{\text{IN}} = 1-(1-p_1)(1-p_2)$;

n_{11} = number of fish detected at both arrays;

n_{10} = number of fish detected at first array, not second;

n_{01} = number of fish detected at second array, not first.

The probability of survival from release to the mouth of the estuary (S_0) can be asymptotically equal to to 1 by having the release even closer to the entrance of the estuary. Investigators, however, must contend with the prospect that ψ may no longer represent the ψ of run-of-river fish.

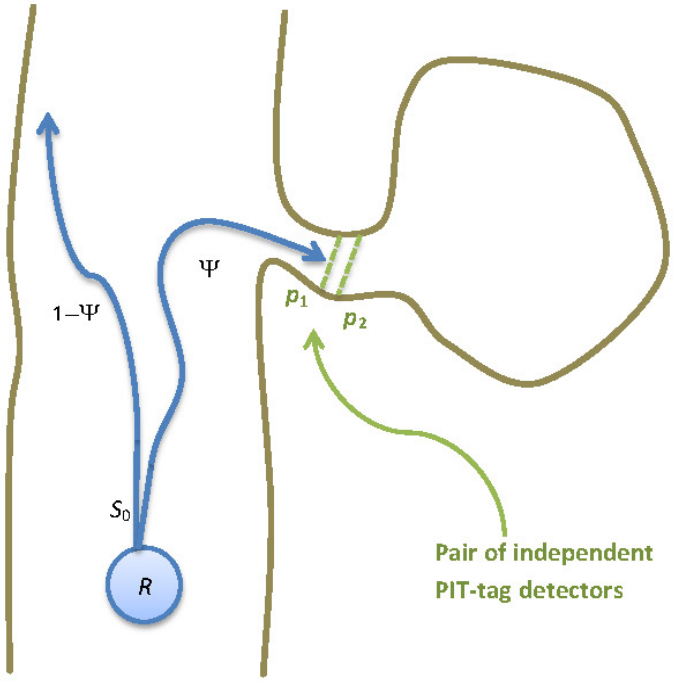


Figure 4.2. Schematic of PIT-tag releases of marked fish with associated (Ψ), survival (S), and detection (p_1, p_2) probabilities for a study of estuary entrance efficiency.

4.3 Active Tag[3]

Wetland restoration in the LCRE typically involves reconnecting wetlands with the estuary by removing barriers in channels that block water flow and fish passage into and out of wetlands. By providing fish passage, tagged fish may freely move into wetlands through a restored channel. These movements may be monitored by use of replicate hydrophone arrays within the restored channel leading to a wetland (Figure 4.3). The replicate hydrophone array consists of two closely spaced arrays that can be used to determine the direction of movement of fish passing through the restored channel connecting the wetland to the estuary. The key feature of the replicate hydrophone array is the ability to obtain information about (1) the time of entry into the wetland, (2) the time of exit out of the wetland, and (3) survival within the wetland. With just a single hydrophone array at the entrance to the wetland, it is impossible to distinguish whether fish are entering or exiting the wetland, and thus impossible to estimate residence time and survival within the wetland. By combining this information with hydroacoustic arrays already in the main stem Columbia River, a release-recapture model can be

[3] This section reproduced from Perry, R. W., and J. R. Skalski. 2008. Evaluating wetland restoration projects in the Columbia River estuary using hydroacoustic telemetry arrays to estimate movement, survival, and residence times of juvenile salmonids. Volume XX in the Design and Analysis of Salmonid Tagging Studies in the Columbia Basin. Bonneville Power Administration, Portland, OR.

developed to estimate important demographic parameters that measure the success of wetland restoration projects.

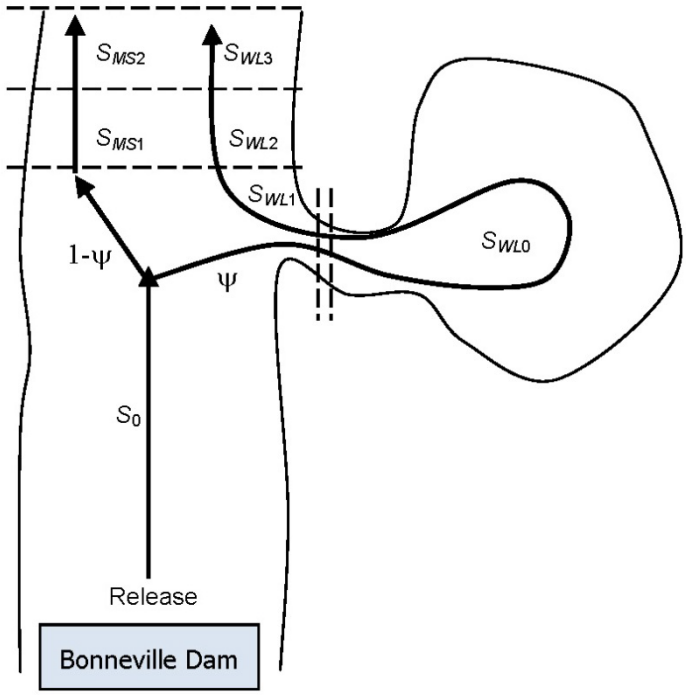


Figure 4.3. Schematic of the study area showing two possible migration pathways: (1) a juvenile salmon that visits the restored wetland with probability ψ and (2) a juvenile salmon that remains in the mainstem river with probability 1-ψ. Dashed lines show possible locations of telemetry stations. Survival parameters include probability of surviving the wetland (S_{WL0}), probabilities of surviving downstream of the wetland for fish that visited the wetland (S_{WL1}, S_{WL2}, S_{WL3}) and for fish that remained in the mainstem (S_{MS1}, S_{MS2}).

As smolts migrate downstream, some will remain in the mainstem Columbia River, while others will enter the wetlands where they will reside for some amount of time before resuming their journey to the ocean (Figure 4.3). The following parameters estimated through a release-recapture model will quantify movement and survival in the main stem river and the restored and monitored wetland:

ψ – Probability of entering the wetland conditional on fish surviving to this point in the river. In other words, this parameter estimates the proportion of the population that visited the wetland of those that passed the entrance to the wetland. Its complement, 1-ψ, estimates the fraction of fish remaining in the main stem Columbia River.

S_0 – Probability of surviving from release to the arrays at the wetland or to the channel cross-section in the main stem river just downstream of the entrance to the wetland.

S_{MSi} – Probability of surviving in each of k reaches (i = 1, ..., k) downstream of the wetland, conditional on fish having remained in the main stem (MS) Columbia River.

S_{WL0} – Probability of surviving from the time of entering the wetland to the time of exiting the wetland (WL).

S_{WLi} – Probability of surviving in each of k reaches (i = 1, ..., k) downstream of the wetland, conditional of fish having used the wetland.

These parameters directly measure the success of the restoration project in terms of the entire population of tagged fish migrating through the estuary. The parameter ψ directly measures the fraction of the population that visits the wetland, while S_{WL0} estimates the proportion of fish that survive their visit to the wetland. Further, if use of the wetland by juvenile salmon improves their survival by facilitating growth or improving their condition, then this benefit of the wetland may be reflected in the subsequent survival of smolts in the main stem river after they leave wetland (i.e., S_{WLi} compared to S_{MSi}). Thus, within a given reach, the survival probabilities downstream of the wetland can be compared between fish that remain in the main stem river (S_{MSi}) and those that visit the wetland and subsequently continue their migration in the main stem river (S_{WLi}). However, whether use of the wetland by smolts improves population-level survival depends on the mortality incurred in the wetland relative to the subsequent improvement in survival downstream of the wetland. This hypothesis can be expressed by comparing total passage survival through the estuary for fish remaining in the main stem river and visiting the wetland:

$$H_0: S_{MS} = S_{WL} \quad (1)$$

where $S_{MS} = S_{MS1}S_{MS2}$

and where $S_{WL} = S_{WL0}S_{WL1}S_{WL2}S_{WL3}$

These functions of model parameters quantify survival through the estuary to the last array where survival is estimated, but the term on the left-hand side of Eq. (1) is estuary survival for fish that remain in the main stem river, and the term on the right-hand side is estuary survival for fish that visit the wetland. The parameter ψ also indicates the relative contribution of each of these terms to the population, and thus, survival through the estuary for all tagged fish is:

$$S_{Estuary} = (1-\psi)\, S_{MS} + \psi\, S_{WL}$$

In addition to the demographic parameters described above, the replicate hydroacoustic arrays provide information about the amount of time fish spend in the wetland:

$$\bar{T}_{WL} = \frac{1}{n}\sum_{j=1}^{n} T_{WL,j} \quad (26)$$

where $\overline{T}_{WL}$ is the mean time spent in the wetland, $T_{WL,j}$ is the elapsed time between entry and exit from the wetland for individual j, and n is the number of fish with entry and exit times.

Using the parameters described above, another integrated performance measure combines information about (1) the fraction of the population using the wetland, (2) survival within the wetland, and (3) time spent within the wetland:

$$\overline{T}_{WL}\,\psi S_{WL0} + \overline{T}_{WL}\,(1-\psi)\,S_{MS} + \psi\overline{T}_{WL}\,(1-S_{WL0}) + \overline{T}_{WL}\,(1-\psi)(1-S_{MS})$$

Because time spent in the wetland is zero for fish remaining in the main stem and non-surviving fish do not contribute to the population, the above equation reduces to:

$$\overline{T}_{WL}\psi S_{WL0} \tag{27}$$

This population-level performance measure integrates the three key parameters of residence time, survival, and fractional use of the wetland and can be used to compare the success of multiple restoration projects in the estuary.

The release-recapture model consists of two independent likelihoods, each based on a multinomial probability distribution. The first likelihood uses information from only the replicate hydrophone array monitoring movement into the wetland to estimate detection probabilities and survival within the wetland (Figure 4.4). The second likelihood estimates the movement parameter ψ, detection probabilities in the main stem Columbia River (P_i), and survival probabilities in the main stem river (Figure 4.4). It is important to note that information contained in the replicate arrays at the mouth of the wetland is sufficient to estimate all detection probabilities, as well as survival within the wetland. Thus, Likelihood 1 can be fit to the data independent of any information provided by hydroacoustic arrays in the main stem Columbia River. The second likelihood, describing migration in the main stem, is overparameterized and does not contain enough information to estimate P_{WL}, the probability of being detected at least once by the replicate hydrophone array. Thus, a joint likelihood is used to estimate all parameters where information from Likelihood 1 is used to estimate P_{WL}. Given P_{WL}, all parameters in Likelihood 2 then become estimable.

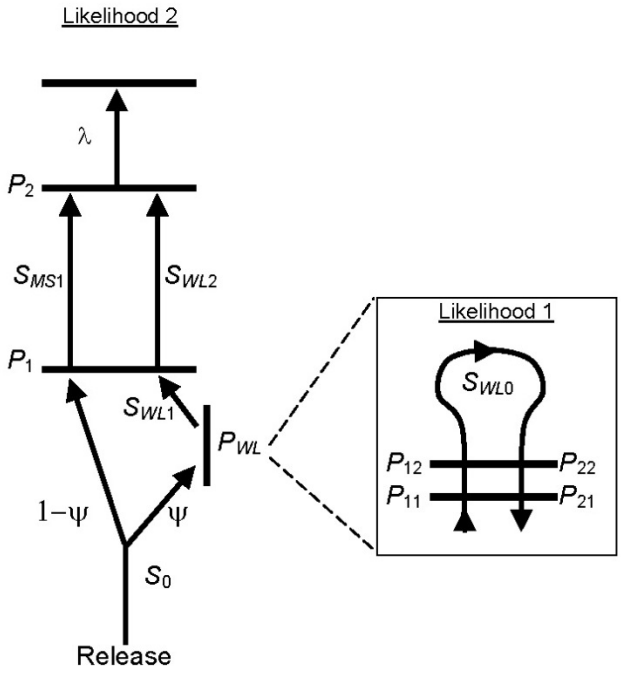


Figure 4.4. Schematic of the release-recapture model showing parameters estimated by Likelihoods 1 and 2. Solid horizontal lines show where detection probabilities (P_i) are estimated at detection stations, the forked arrows show where fish move from the main stem river to the wetland (ψ) or remain in the main stem river (1-ψ), and the remaining arrows show reaches where survival (S_i) is estimated. Likelihood 1 is shown as the inset schematic at the location in the Likelihood 2 where information is used from Likelihood 1 to estimate P_{WL}. In the last reach, λ is the joint probability of surviving and being detected at the last hydroacoustic array.

4.3.1 Likelihood 1: Survival within the Wetland

The replicate hydrophone array contains all the information necessary to estimate detection probabilities and survival in the wetland. Consider a fish that enters and subsequently exits the wetland (see Figure 4.3). Further, suppose this fish is detected on the first and second detection stations as it enters the wetland, and the second then first station as it exits (say, detection history "1221"). The probability of this event is $P_{11}\,P_{21}\,S_{WL0}\,P_{12}\,P_{22}$. That is, the fish was detected with probability P_{11} at the first station and with probability P_{21} at the second station as it entered the wetland, survived with probability S_{WL0}, and was then detected at both stations as it exited with probability P_{12} and P_{22}. The likelihood is formed by identifying all unique detection histories and their probability of occurrence in terms of the parameters (Table 4.1). However, some detection histories, such as 1000 and 0001 (where "0" indicates nondetection) are impossible to distinguish because with only a single detection, we cannot differentiate whether a fish was entering or exiting the wetland. By modeling the series of

events as a single likelihood, both possibilities are accounted for in the probability structure of this detection history (Table 4.1). Finally, the detection history "0000" is not observable, so the likelihood is constructed conditional on being detected at least once on any detection station during either entrance or exit events. Since *Pr*("0000") is the probability of not being detected, 1–*Pr*("0000") is the probability of being detected one or more times. Thus, the conditional likelihood is formed simply by dividing each multinomial cell probability by 1–*Pr*("0000"). There are 13 unique detection histories, with counts of each detection history and associated probabilities of occurrence forming the 13 cell probabilities of a multinomial likelihood model (Table 4.1):

$$L_1 = \binom{R_1}{\underset{\sim}{n}} \prod_{i=1}^{13} \pi_i^{n_i} \tag{28}$$

where L_1 is the likelihood, R_1 is the total number of fish detected at the replicate arrays, $\underset{\sim}{n}$ is the vector of observed frequencies for each detection history, π_i is the probability of occurrence of the ith detection history, and n_i is the number of fish with the ith detection history.

Table 4.1. Multinomial cell probabilities for the Likelihood 1, which estimates detection probabilities and survival within the wetlands (see Fig. 2.2). The probability of being detected at least once at the wetlands telemetry stations (P_{WL}) is (1–(1–P_{11}) (1–P_{21}) (1–S_{WL0}+S_{WL0} (1–P_{22}) (1–P_{12}))).

Detection history	Probability of occurrence (π_i)
1221	$P_{11}\ P_{21}\ S_{WL0}\ P_{22}\ P_{12}/P_{WL}$
0221	$(1-P_{11})\ P_{21}\ S_{WL0}\ P_{22}\ P_{12}/P_{WL}$
1021	$P_{11}\ (1-P_{21})\ S_{WL0}\ P_{22}\ P_{12}/P_{WL}$
0021	$(1-P_{11})\ (1-P_{21})\ S_{WL0}\ P_{22}\ P_{12}/P_{WL}$
1201	$P_{11}\ P_{21}\ S_{WL0}\ (1-P_{22})\ P_{12}/P_{WL}$
0201	$(1-P_{11})\ P_{21}\ S_{WL0}\ (1-P_{22})\ P_{12}/P_{WL}$
1001	$P_{11}\ (1-P_{21})\ S_{WL0}\ (1-P_{22})\ P_{12}/P_{WL}$
1	$(1-P_{11})\ (1-P_{21})\ S_{WL0}\ (1-P_{22})\ P_{12}/P_{WL}+P_{11}\ (1-P_{21})\ (1-S_{WL0}+S_{WL0}\ (1-P_{22})\ (1-P_{12}))/P_{WL}$
1220	$P_{11}\ P_{21}\ S_{WL0}\ P_{22}\ (1-P_{12})/P_{WL}$
0220	$(1-P_{11})\ P_{21}\ S_{WL0}\ P_{22}\ (1-P_{12})/P_{WL}$
1020	$P_{11}\ (1-P_{21})\ S_{WL0}\ P_{22}\ (1-P_{12})/P_{WL}$
2	$(1-P_{11})\ (1-P_{21})\ S_{WL0}\ P_{22}\ (1-P_{12})/P_{WL}+(1-P_{11})\ P_{21}\ (1-S_{WL0}+S_{WL0}\ (1-P_{22})\ (1-P_{12}))/P_{WL}$
1200	$P_{11}\ P_{21}\ (1-S_{WL0}+S_{WL0}\ (1-P_{22})\ (1-P_{12}))/P_{WL}$

Model Assumptions

1. Each fish has an independent fate.
2. The probability of detection at one array is independent of the probability of detection at the second array.

3. The direction of movement of fish (i.e., entering or exiting) can be determined based on the time series of detections at each array.
4. Fish move through both arrays and enter the wetland.

The second assumption can be fulfilled by ensuring that the detection zone of each array completely encompasses the channel cross-section. This assumption is likely to be fulfilled given the small size of these channels relative to the typical detection range of telemetry receivers. Assumption 3 is necessary because probabilities of occurrence for each detection history are based on the order of detection at the replicate arrays. This assumption can be fulfilled by separating each hydrophone array by a distance sufficient to yield spatial and temporal resolution among detection times at each array. However, the arrays should be in close enough proximity to ensure that little mortality occurs between the replicate arrays. The last assumption may be violated if fish do not completely pass through both arrays and enter the wetland. For example, if the replicate arrays are situated too close to the main stem Columbia River, then fish in the main stem river passing by the entrance of the wetland may be detected at the replicate arrays. As another example, if a fish enters the channel, passes the first array, but then turns around and exits into the main stem river, then the fourth assumption will be violated. The consequence of violating this assumption is positive bias in ψ and negative bias in S_{WL0}.

4.3.2 Likelihood 2: Movement and Survival within the Main Stem

The primary likelihood proceeds in similar fashion to a standard Cormack-Jolly-Seber model with the additional complexity of incorporating a movement probability (ψ) and estimating survival probabilities conditional on previous migration history (i.e., fish that remain in the main stem versus those that use the wetland). The primary likelihood ignores the replicate array structure and treats each telemetry station as if it were a single detection station, considering only the presence or absence of detections at each station. For illustration, we constructed a model with three telemetry stations (i.e., two reaches) downstream of the wetland and a single reach upstream of the wetland. However, a minimum configuration consists of two telemetry stations (i.e., one reach) downstream of the wetland. Under this minimum configuration, only the ratio S_{WL2}/S_{MS1} can be estimated with the assumption that detection probabilities at the last telemetry station are the same for these two groups of fish. All other parameters can be estimated with this minimum configuration. Beyond the minimum configuration, this model can accommodate any number of reaches upstream and downstream of the wetland. The likelihood is constructed by listing all possible detection histories and writing the probability of each detection history as a function of the model parameters (Figure 4.4, Table 4.2). To distinguish detections in the wetland from those in the main stem, detection histories for the main stem are coded with an "A" while those at the entrance to the wetland are denoted by "B". Downstream of the wetland, detections or absence thereof are denoted by a "1" or "0" respectively. Thus the detection history "BA11" indicates a fish was detected either entering or exiting the wetland (B), was then detected in the main stem river just downstream of the wetland (A), and was detected at the two downriver telemetry stations (Figure 4.3 and Figure 4.4). The probability of this detection history is

simply the joint probability of parameters that describe this pathway through the system (Fig. 2.2): S_0 ψ P_{WL} P_1 S_{WL1} P_2 λ.

Another important feature of the primary likelihood is the inability to distinguish among some of the possible detection histories. For example, the detection history "A11" cannot be distinguished from "0A11". In other words, from the detection data there is no way to distinguish whether a fish migrated only in the main stem, or entered the wetland, survived, and exited the wetland without being detected. The probability structure of this detection history must incorporate the possibility that either event could have occurred (see Table 4.2). For this likelihood, there are 16 unique detection histories, each forming the 16 cell probabilities of a multinomial distribution:

$$L_2 = \binom{R_2}{\underset{\sim}{n}} \prod_{j=1}^{16} \pi_j^{n_j} \tag{29}$$

and R_2 is the total number of fish released, $\underset{\sim}{n}$ is the vector of observed frequencies for each detection history, π_j is the probability of occurrence of the jth detection history, and n_j is the number of fish with the jth detection history.

Table 4.2. Multinomial cell probabilities for the Likelihood 2, which estimates detection, movement, and survival probabilities within the main stem Columbia River (see Fig. 2.2). The probability of being detected at least once at the wetland telemetry stations (P_{WL}) is (1– (1–P_{11}) (1–P_{21}) (1–S_{WL0}+S_{WL0} (1–P_{22}) (1–P_{12}))).

Detection history	Probability of occurrence (π_j)
BA11	S_0 ψ P_{WL} S_{WL1} P_1 S_{WL2} P_2 λ
A11	S_0 ψ (1–P_{WL}) S_{WL1} P_1 S_{WL2} P_2 λ+S_0 (1–ψ) P_1 S_{MS1} P_2 λ
B011	S_0 ψ P_{WL} S_{WL1} (1–P_1) S_{WL2} P_2 λ
011	S_0 ψ (1–P_{WL}) S_{WL1} (1–P_1) S_{WL2} P_2 λ+S_0 (1–ψ) (1–P_1) S_{MS1} P_2 λ
BA01	S_0 ψ P_{WL} S_{WL1} P_1 S_{WL2} (1–P_2) λ
A01	S_0 ψ (1–P_{WL}) S_{WL1} P_1 S_{WL2} (1-P_2) λ+S_0 (1–ψ) P_1 S_{MS1} (1–P_2) λ
B001	S_0 ψ P_{WL} S_{WL1} (1–P_1) S_{WL2} (1–P_2) λ
001	S_0 ψ (1–P_{WL}) S_{WL1} (1–P_1) S_{WL2} (1–P_2) λ+S_0 (1–ψ) (1–P_1) S_{MS1} (1–P_2) λ
BA10	S_0 ψ P_{WL} S_{WL1} P_1 S_{WL2} P_2 (1–λ)
A10	S_0 ψ (1–P_{WL}) S_{WL1} P_1 S_{WL2} P_2 (1–λ)+S_0 (1–ψ) P_1 S_{MS1} P_2 (1–λ)
B010	S_0 ψ P_{WL} S_{WL1} (1–P_1) S_{WL2} P_2 (1–λ)
010	S_0 ψ (1–P_{WL}) S_{WL1} (1–P_1) S_{WL2} P_2 (1–λ)+S_0 (1–ψ) (1–P_1) S_{MS1} P_2 (1–λ)
BA00	S_0 ψ P_{WL} S_{WL1} P_1 (1–S_{WL2}+S_{WL2} (1–P_2) (1–λ))
A00	S_0 ψ (1–P_{WL}) S_{WL1} P_1 (1–S_{WL2}+S_{WL2} (1–P_2) (1–λ))+S_0 (1–ψ) P_1 (1–S_{MS1}+S_{MS1} (1–P_2) (1–λ))
B000	S_0 ψ P_{WL} (1–S_{WL1}+ S_{WL1} (1–P_1) (1–S_{WL2}+S_{WL2} (1–P_2) (1–λ)))
000	1–S_0+S_0 ψ (1–P_{WL}) (1–S_{WL1}+ S_{WL1} (1–P_1) (1–S_{WL2}+S_{WL2} (1–P_2) (1–λ)))+S_0 (1–ψ) (1–P_1) (1–S_{MS1}+S_{MS1} (1–P_2) (1–λ))

Model Assumptions

1. Each fish has an independent fate.
2. Capture, survival, and movement are not affected by previous capture history.
3. Movements defining fish that remain in the main stem or move into the wetland occur over short distances such that mortality is zero.

The last assumption can be fulfilled by placing a hydroacoustic array in the main stem river as close as possible to the entrance to the wetland. This assumption is particularly important, as the consequence of failing this assumption is biased movement probabilities. For example, consider a hydroacoustic array that is placed considerable distance downstream of the wetland entrance. Now, a fish passes by the entrance to the wetland but remains in the main stem river with probability $(1-\psi)$, and from that point, it survives with probability <1 to the next array downriver. Since there is no array at the point of transition between the main stem and wetland, the movement and survival process cannot be separated, resulting in biased estimates of ψ. However, for fish that enter the wetland, we can estimate survival between the exit of the wetland and the first downriver array (S_{WL1} in Figure 4.3 and Figure 4.4). This survival probability can act as a check on assumption 3, since if assumption 3 is fulfilled, we would expect the estimate of S_{WL1} to be close to 1.

4.3.3 Joint Likelihood: Movement and Survival within the Main Stem and Wetland

As discussed above, the primary likelihood does not contain enough information to estimate P_{WL}, the probability of being detected at least once during a visit to the wetland. Thus, P_{WL} is estimated as a function of parameters in the Likelihood 1:

$$P_{WL} = 1-(1-P_{11})\,(1-P_{21})\,(1-S_{WL0}+S_{WL0}\,(1-P_{22})\,(1-P_{12}))$$

Given P_{WL} is estimated from Likelihood 1, all remaining parameters in Likelihood 2 become estimable and the joint likelihood for estimating all parameters is simply the product, L_1L_2.

5 Entrance Efficiency

There are numerous situations where some form of estimates of entrance efficiency is desired including culvert passage and entrance into and out of specific habitat types such as tidal channels. In all cases, the proportion of fish that successfully entered or passed through a location depends on knowing the number of fish available or at risk of passage.

In Section 4, unconditional probabilities of movement were calculated. These probabilities express the propensity for a fish to select a particular pathway, given a choice is to be made. In this section, entrance efficiency expresses the probability of entering a locale, given a fish has arrived at the locale.

These probabilities may be considered near-field probabilities of successful entrance, given arrival to the entrance.

5.1 Inert Tags

Tags that cannot be remotely identified to an individual fish will generally not be suitable for estimating entrance efficiency. Without this capability, it will be difficult to determine which fish were at risk of entering a culvert or estuary, and which fish successfully entered/passed. At best, using batch-marked fish, it might be possible to assess whether the route is susceptible to entrance or not, but not the rate of passage. Using different batch marks, it might be feasibly to compare the relative entrance propensity (REP) of different groups or classifications of fish, i.e., the ratio

$$\text{REP} = \frac{\left(\frac{x_2}{n_2}\right)}{\left(\frac{x_1}{n_1}\right)}$$

where

n_i = number of fish marked and released in the i th group $(i = 1, 2)$;

x_i = number of fish recovered from among the n_i released $(i = 1, 2)$.

5.2 PIT Tags

In the case of culvert entrance efficiency, PIT-tag technology can be an ideal approach to studying fish movement in the near field (Figure 5.1). PIT-tag, flat-plate technology has a relatively narrow detection field that can be located at both the lower and upper ends of a culvert. These narrow near-field detection arrays at either end can be used to identify those fish that approach or began entering the culvert, and which of these fish successfully passed through the culvert.

Passage efficiency is then estimated using a binomial model where

$$\hat{\theta} = \frac{x}{n} \tag{30}$$

where

$\hat{\theta}$ = estimate of passage efficiency,

n = number of fish detected approaching or entering culvert,

x = number of the n fish that successfully traversed the culvert,

with estimated sampling variance

$$\widehat{\mathrm{Var}}(\hat{\theta}) = \frac{\hat{\theta}(1-\hat{\theta})}{n}. \qquad (31)$$

This binomial model assumes the detection probability at the upper end of the entrance is 1.0.

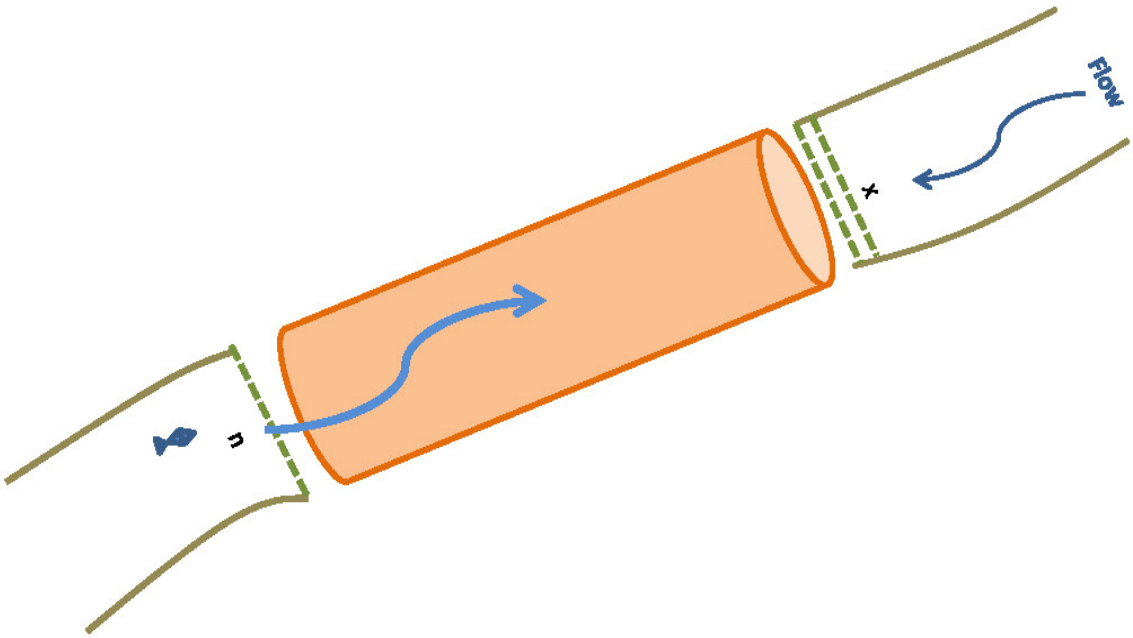


Figure 5.1. Schematic of a passage efficiency design at a culvert with numbers of fish approaching the entrance (*n*) and successfully exiting (*x*) identified with PIT-tag, flat-plate detection arrays (dashed lines).

In the case of replicated trials and alternative passage conditions, the values of $p_i\,(i=1,\ldots,m)$ can be modeled using generalized linear models (GLM) based on a binomial sampling error structure and logit or log link (McCullagh and Nelder 1989). This analysis assumes again that the detection probability at the upper end of the structure is 1.0. Otherwise, the analysis is modeling the percent probability of passage and detection.

With a double-detection array at the upper end of the passage structure, it is possible to independently estimate the probability of passage from the probability of detection. Assuming the

detection processes are independent of the two elements of the double array, the joint likelihood for estimating entrance efficiency can be written as

$$L = \binom{n}{x}(\theta p)^{x}(1-\theta p)^{n-x}\binom{x}{m}\left(\frac{p_1 p_2}{1-(1-p_1)(1-p_2)}\right)^{m_{11}} \cdot\left(\frac{(1-p_1)p_2}{1-(1-p_1)(1-p_2)}\right)^{m_{01}}\left(\frac{p_1(1-p_2)}{1-(1-p_1)(1-p_2)}\right)^{m_{10}} \tag{32}$$

where

$p = 1-(1-p_1)(1-p_2)$;

m_{11} = number of x with detections at both arrays;

m_{01} = number of x with no detections at first array, detected at second array;

m_{10} = number of x detected at first array but not at second array.

5.3 Active Tags

The unique individual identification capabilities of active tags allow the same form of analysis of entrance efficiency at PIT tags. However, their often large detection radii (i.e., 100–500 m) can make analysis of near-field movements very difficult. Their utility will depend on an investigator's ability to deploy narrow detection fields. With proper data collection, entrance efficiency can be calculated as illustrated in Section 5.2

Alternatively, cabled 2D or 3D acoustic arrays can be used to track fish movements in and about the vicinity of passage entrances. Should detection probabilities be 1, the entrance probability can be calculated according to Equation (30) with variance estimate Equation (31). In less certain situations, a double 2D or 3D array can be used to estimate the probability of detection $(\hat{p})$ based on likelihood model (25). The estimate of entrance efficiency is then calculated as

$$\hat{\theta} = \frac{x}{n\hat{p}}$$

With the variance estimated by

$$\widehat{\mathrm{Var}}(\hat{\theta}) = \frac{\theta(1-\theta)}{n} + \mathrm{Var}(\hat{p})\left[\frac{\theta(1-\theta)}{n} + \theta^2\right]. \tag{33}$$

6 Residualization and Overwintering[4]

6.1 Active Tags Only

Below Bonneville Dam to the mouth of the Columbia River, juvenile salmon may residualize and overwinter, outmigrating the following spring. Connor(2001) suggests this may be an important life-history strategy for fall Chinook salmon. The current difficulty in quantifying the significance of this strategy is that the PIT-tag bypass system at John Day and Bonneville dams are dewatered in winter and there are no other detection arrays to monitor outmigration near the mouth of the river. These limitations could be mitigated by using radio telemetry or acoustic arrays across the river and near its mouth.

Lowther and Skalski (1998) developed a multinomial model to estimate overwinter residualization and survival of outmigrants over a two-year period. The model uses two downstream detection arrays and is suited for the study of fall Chinook salmon smolts in the Lower Columbia River (Figure 6.1). A more generalized form of this model, permitting multiple downstream arrays and multiple age classes of outmigrants, has been developed and is available from the University of Washington in the form of the freeware program TribPit (http://www.cbr.washington.edu/analysis/apps/tribpit). Closed-from estimators are available for the Lowther and Skalski (1998) residualization and migration model. Iterative solutions are available for maximum likelihood estimates in Program TribPit.

[4] This section reproduced in part from Lowther, A. B., and J. R. Skalski. 1998. A multinomial likelihood model for estimating survival probabilities and residualization for fall chinook salmon (*Oncorhynchus tshawytscha*) smolt using release-recapture methods. Journal of Agricultural Biology and Environmental Statistics 3:223-236.

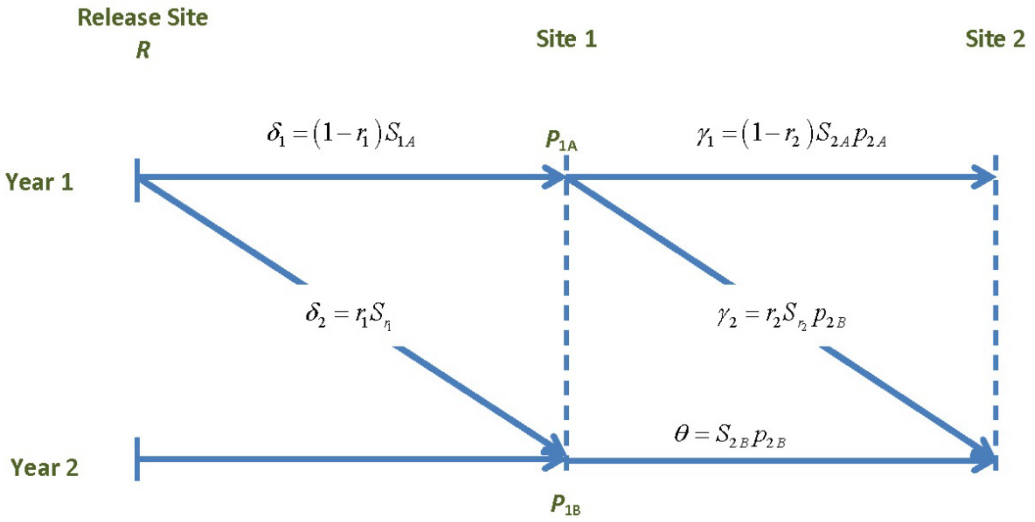


Figure 6.1. Schematic of the Lower and Skalski (1998) multistate, release-recapture model with seven estimable parameters (i.e., δ_1, δ_2, p_{1A},p_{1B}, γ_1, and γ_2) as a function of residualization (*r*), survival (*S*), and detection processes (*p*).

The Lowther and Skalski (1998) model with two river reaches can estimate seven parameters (Figure 6.2) as follows:

1. $\delta_1 = (1-r_1)S_{1A}$ = the joint probability of not residualizing in the first reach and survival to the first downstream detection array in the first year;
2. $\delta_2 = r_2 S_{r_1}$ = the joint probability of residualizing in the first reach and survival to the first downstream array in the second year;

where

p_{1A} = probability of detection at the first array in year 1;

p_{1B} = probability of detection at the first array in year 2;

$\gamma_1 = (1-r_2)S_{2A}p_{2A}$ = the joint probability of not residualizing in the second reach in the first year of migration, survival through the reach, and being detected at the second (i.e., last) array;

$\gamma_2 = r_2 S_{r_2} p_{2B}$ = joint probability of residualizing in the second year in the first year, surviving and migrating the second year, and being detected at the second (i.e., last) array;

$\theta = S_{2B}p_{2B}$ = joint probability of surviving migrating in the second reach in the second year, for a fish that residualized the first year, and being detected at the second (i.e., last) array.

The sum $\delta_1 + S_2 = (1-r_1)S_{1A} + r_1 S_{r_1}$ is the overall probability of a smolt surviving the first reach regardless of migration strategy. For a two-reach study, survival can only be directly estimated in the first reach. Like other release-recapture designs, only the joint probability of survival and detection can be estimated in the last reach.

The multinomial likelihood model for the fall Chinook salmon outmigration can be written as:

$$L\left(n_{100}, n_{101}, n_{111}, n_{110}, n_{112}, n_{102}, n_{120}, n_{122} \middle| R, \underset{\sim}{\delta}, \underset{\sim}{p}, \theta, \underset{\sim}{\gamma}\right)$$
$$= \binom{R}{n_{100}, n_{101}, n_{111}, n_{110}, n_{112}, n_{102}, n_{120}, n_{122}} (\pi_{101})^{n_{101}} (\pi_{110})^{n_{110}} (\pi_{111})^{n_{111}} \quad (34)$$
$$\cdot (\pi_{112})^{n_{112}} (\pi_{102})^{n_{102}} (\pi_{120})^{n_{120}} (\pi_{122})^{n_{122}},$$

where $n_{100} = R - (n_{101} + n_{111} + n_{110} + n_{112} + n_{102} + n_{120} + n_{122})$ and the π_{ijk} are as defined in Table 6.1. The capture histories denote the three locations and the detection fish present in year 1, year 2, or never (i.e., 0).

Table 6.1. Expected values for the cell (capture history) probabilities under the full parameterization (center column and under the reduced parameterization (right column).

Probability of capture history	Expected value given the original parameterization	Expected value with reduced parameterization
π_{111}	$(1-r_1)S_{1A}p_{1A}(1-r_2)S_{2A}p_{2A}$	$\delta_1 p_{1A}\gamma_1$
π_{101}	$(1-r_1)S_{1A}(1-p_{1A})(1-r_2)S_{2A}p_{2A}$	$\delta_1(1-p_{1A})\gamma_1$
π_{112}	$(1-r_1)S_{1A}p_{1A}r_2 S_{R_2}p_{2B}$	$\delta_1(1-p_{1A})\gamma_1$
π_{102}	$(1-r_1)S_{1A}(1-p_{1A})r_2 S_{r_2}p_{2B} + r_1 S_{r_1}(1-p_{1B})S_{2B}p_{2B}$	$\delta_1(1-p_{1A})\gamma_2 + \delta_2(1-p_{1B})\theta$
π_{120}	$r_1 S_{r_1}p_{1B}(1-S_{2B}p_{2B})$	$\delta_2 p_{1B}(1-\theta)$
π_{122}	$r_1 S_{r_1}p_{1B}S_{2B}p_{2B}$	$\delta_2 p_{1B}\theta$
π_{110}	$(1-r_1)S_{1A}p_{1A}\left[1-S_{2A}p_{2A}(1-r_2)-r_2 S_{r_2}p_{2B}\right]$	$\delta_1 p_{1A}(1-\gamma_1-\gamma_2)$
π_{100}	$1-\sum^{\pi_{ijk}}$	$1-\sum^{\pi_{ijk}}$

NOTE: Here π_{ijk} is used to represent the probability that an individual fish has capture history *ijk*.

Model assumptions include:

1. The acoustic tags are not lost and remain active.
2. The space of the resampling is small relative to the interval (i.e., a river reach) of the study.
3. All previously tagged fish alive in the population at the beginning of a given period (river reach) have the same probability of surviving until the end of that period (river reach). However, within a reach, fish that overwinter may have a different survival probability than those that do not overwinter.
4. The history of survival, capture, and overwintering of reach tagged fish is independent of all others.
5. All tagged fish alive at a particular sampling location have the same probability of being captured.
6. The probability of capture or survival of any individual is not affected by its previous history of captures.
7. The probability that a fish overwinters in a reach is the same for all tagged fish alive at the beginning of the reach.
8. The probability that a fish overwinters in a reach is not affected by its previous history of captures.
9. Fish that overwinter either migrate in the second year or die.
10. The test fish are representative of the population of interest.
11. Test conditions are representative of the conditions of interest.

Fish that overwinter in the last reach may experience quite different river conditions depending on the exact location of overwintering and the timing of the resumption of migration. However, this does not violate the assumptions 2, 3, and 5 of the model, as they assert that overwintering fish experience the same conditions in expectation, not that conditions experienced by each fish must be identical. The first nine assumptions are necessary for the construction of the multinomial likelihood model, while the final two assumptions allow statistical inference from the release group to the population of fall Chinook salmon.

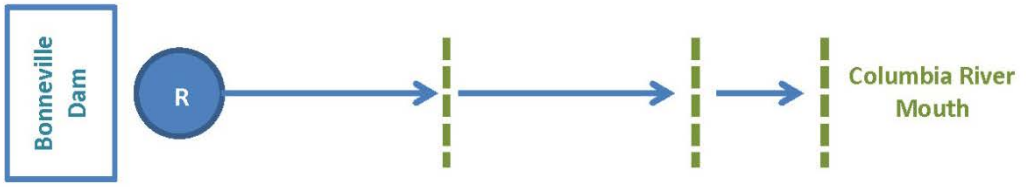
In practice, the acoustic tags would need to have an extended tag life of ≈180 days or more to allow the study to run from fall to the following spring. Overwintering fall Chinook salmon are generally large enough to accommodate a larger tag with more batteries. Nevertheless, the original Lowther and Skalski (1998) model assumes no tag failure due to manufacturing defects. An extension of the model to accommodate a tag-life study and tag-failure probabilities could be readily produced.

Figure 6.2 illustrates some alternative deployment designs, using either the two-reach Lowther and Skalski (1998) model or a three-reach, multistate extension of the model as found in Program TribPit. One aspect of the design is how close to the mouth of the Columbia River are inferences sought. Winter conditions at the mouth of the Columbia River could make maintaining arrays there difficult to achieve.

1. Two-reach Lowther and Skalski (1998) model with first reach being majority of estuary

2. Two-reach Lowther and Skalski (1998) model with first reach being entire estuary and second array surrounding mouth of the river

3. Three-reach model (Program TribPit) model with first two reaching being majority of estuary



4. Three-reach model (Program TribPit) with first two reaches being entire estuary and third array surrounds mouth of the river

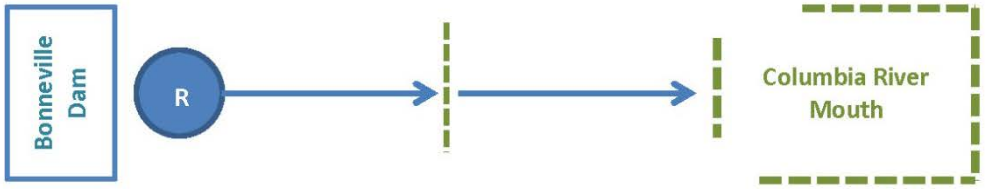


Figure 6.2. Schematics of possible multistate, release-recapture designs to estimate survival and residualization in the Lower Columbia River below Bonneville Dam.

7 Residence Time/Travel Time Measurements

7.1 Inert Tags

Passive tags with group marks do not permit individual travel times to be observed. As an alternative, the difference in mean or median arrival times is used as an estimate of residence/travel time where

$$\Delta t = \bar{t}_2 - \bar{t}_1$$

or

$$\Delta t = M_{t_2} = M_{t_1},$$

where $\bar{t}_j$ is the average and M_{t_j} is the median time at the j th location over time. These expressions will be biased estimators of the true mean or median time unless the fish arrive in the same rank order through the two events. Because mixing of individuals between events is most likely, these estimators need to be used with caution.

The variance of Δt can be calculated as

$$\mathrm{Var}(\Delta t) = \mathrm{Var}(\bar{t}_1) + \mathrm{Var}(\bar{t}_2)$$

under the assumption of independence.

7.2 Active and PIT Tags

The individual identifiers associated with active and PIT tags allow residence time or travel time between two locations to be recorded on an individual basis. The mean time can be estimated by the arithmetic mean

$$\bar{t} = \frac{\sum_{i=1}^{n} t_i}{n}$$

where t_i = residence/travel time of the i th individual $(i = 1, \ldots, n)$, with an associated variance estimate of

$$\widehat{\mathrm{Var}}(\bar{t}) = \frac{\sum_{i=1}^{n} (t_i - \bar{t})^2}{n(n-1)}.$$

A $(1-\alpha)$ 100% confidence interval of mean time, μ_t, can be calculated using a *t*-statistic with $n-1$ degrees of freedom.

In many cases, residence/travel times are right skewed—in which case, median travel time is a better measure of central tendency. The median is defined as

$$t_i: \quad P(t \geq t_i) = 0.50.$$

The median is a nonparametric measure of central tendency. The variance of the median can be estimated based on the distribution of the median (m) for sample size n from a population with density function $f(x)$ being asymptotically normal with mean m and variance (Rider 1960)

$$\left\{4n\left[f(m)\right]^2\right]^{-1}.$$

However, this estimator can be poorly behaved for small sample sizes and particular distributions (Rider 1960). An alternative method of variance calculation is to use bootstrap techniques which should be suitable for most underlying distributions (Efron and Tibshirani 1993).

8 Next Steps

This compendium provides an overview of statistical designs using mark-recapture techniques to assess juvenile salmon performance in the LCRE. It is intended to serve as a basis to institute field research studies. For a given study, the next steps would include formulation of specific objectives, identification of the appropriate mark-recapture technique (if applicable), and development of a detailed statistical plan specific to that study. In the case of branching or multistate investigations of migration, study preparation should include development of the statistical release-recapture model and to assure all intended parameters are estimable. This effort should also include Monte Carlo investigations to determine anticipated precision of the study and the calculability of the estimates. Spare data sets may allow all parameters to be calculable in certain circumstances. Simulations studies therefore provide an opportunity to determine the feasibility and precision of planned investigations.

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