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Lower Columbia River Fisheries and Escapement Evaluation in Southwest Washington, 2010

Edited by Dan Rawding, Bryce Glaser, and Thomas Buehrens

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Dan Rawding, Bryce Glaser and Thomas Buehrens, editors

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Executive Summary

In 2010, The Washington Department of Fish and Wildlife (WDFW) began implementation of an expanded monitoring program for Chinook and coho salmon populations in the Lower Columbia River (LCR) region of Southwest Washington (WDFW's Region 5) and fishery monitoring in the lower mainstem of the Columbia River. The focus of this expanded monitoring was to 1) gather data on Viable Salmonid Population (VSP) parameters – spawner abundance, including proportion of hatchery origin spawners (pHOS), spatial distribution, diversity, and productivity and 2) to increase the Coded Wire Tag (CWT) recovery rate from spawning grounds to meet regional standards, and 3) to evaluate the use of PIT tags to develop harvest rates for salmon and steelhead populations by having fishery samplers recovery PIT tags from fish being sampled for CWT in existing fisheries monitoring programs. Monitoring protocols and analysis methods utilized were intended to produce unbiased estimates with measurements of precision in an effort to meet NOAA monitoring guidelines (Crawford and Rumsey 2009).

Funding for this program came from multiple sources: 1) the Bonneville Power Administration (BPA) through the Lower Columbia Coded Wire Tag (CWT) Recovery Project (BPA Project #: 2010-036-00) ; 2) the National Oceanic and Atmospheric Administration (NOAA) via Mitchell Act Monitoring, Evaluation and Reform (MA MER) funds; 3) NOAA via Pacific Coastal Salmon Recovery Funds (PCSRF) (administered thru the Washington State Recreation and Conservation Office (RCO); 4) Washington State; 5) PacifiCorp (Lewis River Basin) and 6) Tacoma Power (Cowlitz River Basin).

This report is structured into four components:

1) Fall Chinook Salmon Escapement Estimates and Coded-Wire-Tag Recoveries in Washington's Lower Columbia River Tributaries in 2010

- Key Results
  - Adult fall Chinook abundance was estimated using weir counts, open and closed mark-recapture models, Area-Under-the-Curve (AUC), redd counts, and peak count expansion depending on resources and survey conditions.
  - We estimated 38,460 adult Tule, 65 adult Rogue River Bright hatchery, 9,612 adult Lewis River Bright natural origin , and 1,657 adult Bonneville (BON) Pool Bright fall Chinook salmon in the Washington portion of the LCR ESU.
  - For Tules and BON Brights the proportion of marked adults was 79% and 58%, respectively. Age structure varied by population but most Tule Chinook salmon were age 3 or 4.
  - For Tules most populations were comprised primarily of hatchery fish except the Coweeman (70% unmarked), Lewis (73% unmarked), Lower Gorge (77% unmarked), and the White Salmon (73% unmarked).
  - A total of 184 snouts were collected from the field and examined for CWT. CWT recoveries were uploaded to the regional coded-wire-tag database (RMIS). Unexpanded CWT recoveries indicate most Tule hatchery fish returned to the basin of release or an adjacent basin.
  - BON Brights are not native to this ESU and are successfully spawning in the Upper Gorge and White Salmon populations. Rogue River Brights, also not native to this ESU, are successfully spawning in the Grays River.

2) Coho Salmon Escapement Estimates and Coded-Wire-Tag Recoveries in Washington's Lower Columbia River Tributaries in 2010

- Key Results
  - The adult coho salmon population monitoring program used trap and haul census counts, mark-recapture, smolt expansion, and redd-based methods to monitor adult coho salmon.
  - We estimated a mean escapement of 57,666 (95% CI 48,240 - 70,751) adults and 4,428 jacks (95%CI 3,522 - 5,816) for the Washington portion of this ESU below Bonneville Dam excluding the mainstem Lower Cowlitz, mainstem Lower North Fork Lewis, mainstem Toutle/ lower North Fork Toutle (below the Sediment Retention Structure), and Salmon Creek populations.
  - The total mean estimate of unmarked coho salmon adults was 25,942 (95%CI 19,430 – 35,510).

- As expected, populations with an operating coho salmon hatchery, including the Grays, Elochoman, Upper Cowlitz/Cispus, Green, Kalama and Washougal rivers, had high proportions of hatchery spawners—(mean = 81%, 73%, 87%, 60%, 99%, and 44%, respectively). The converse was true for populations without hatcheries, such as the Mill-Abernathy-Germany, Lower Cowlitz, Coweeman, and South Fork Toutle populations, where we observed low percentages of marked adults—(mean = 12%, 15%, 10%, and 21%, respectively).
- From carcass recoveries on stream surveys, a total of 84 CWTs were recovered from coho salmon in 2010.

3) Detection Probabilities for Passive Integrated Transponder (PIT) Tags in Adult Salmon and Steelhead with Hand Held Scanners

- Key Results
  - We completed a study to evaluate detection rates for PIT tags in adult salmonids in a fisheries sampling setting using a variety of tag scanner types under conditions similar to those expected in sampling of fisheries catch.
  - From 45 trials, the mean detection rate among all species using either a Destron Fearing FS2001F-ISO Reader Base Unit (DF) with racquet antenna or All Flex Model RS601-3 (AF) was 99.4% (95% CI = 98.5% - 100.0%), and was >99% for each species-reader type combination.
  - Short-term PIT tag retention estimates were approximately 98% for all three species (Chinook, coho and steelhead).

4) Estimates of Columbia River Salmon and Steelhead Harvest Rates for the 2010 Fall Commercial and Treaty Fisheries based on Passive Integrated Transponder (PIT) Tags

- Key Results
  - The fall commercial fishery below Bonneville Dam (BON) and the fall treaty fishery above BON were sampled in 2010. Estimates of harvest rates below BON in the commercial fishery were less precise than above BON due to lower tagging rates of stocks caught in these fisheries and smaller numbers of tag recoveries in this area, in addition to a late start in sampling for PIT tags.
  - For the treaty fishery, coho salmon harvest rate estimates were similar for four hatchery groups and provided a precise estimate for the aggregate of ‘early’ coho salmon (17% with a 95%CI from 14% to 20%). Using a mixture model we estimated the harvest rate for the ‘late’ coho salmon group (4% with a 95%CI from 2% to 4%).
  - For steelhead, a total of 26 individual population and multiple DPS harvest estimates were calculated. Our harvest estimate for Snake Group B steelhead (8%) was lower than the Columbia River Technical Advisory Committee (TAC)

estimate (15%), while our wild Snake Group A estimate (5%) and TAC's Group A estimate (4%) were similar, but we noticed variation in Distinct Population Segment (DPS) estimates. For example, our Middle Columbia River DPS estimate was 3%, while our Upper Columbia River DPS estimate was 10%.

- Our estimated Chinook salmon harvest rates were higher for Tule fall Chinook (64%) and lower for Snake River fall Chinook (9%), compared to the TAC estimates (51% and 18%, respectively).
- We detected size selectivity in the fishery catch relative to the escapement based on PIT tag analysis. Since we did not estimate harvest rates by age or size, our harvest rate estimates were likely biased low for adults and large fish and biased high for jacks and smaller fish due to higher catch rates of larger fish in fisheries.

Relationship to the 2008 Federal Columbia River Power System Biological Opinion

Work conducted by the BPA Lower Columbia River CWT Recovery Project (#2010-036-00) supports the following Reasonable and Prudent Actions (RPA) as identified in the 2008 Federal Columbia River Power System Biological Opinion (FCRPS BiOp). http://www.nwr.noaa.gov/hydropower/fcrps_opinion/federal_columbia_river_power_system.html

RPA:

50.4: Fund pilot studies in Wenatchee/Methow/Entiat

Fund status and trend monitoring as a component of the pilot studies in the Wenatchee, Methow, and Entiat river basins in the Upper Columbia River, the Lemhi and South Fork Salmon river basins, and the John Day River Basin to further advance the methods and information needed for assessing the status of fish populations. (Initiate in FY 2007-2009 Project Funding)

Relationship: This project provided PIT tag-based estimates of run timing and fishery-specific harvest rates for these salmon and steelhead populations at hierarchical spatial scales.

50.5: Provide additional status monitoring of SR B-Run Steelhead populations

Provide additional status monitoring to ensure a majority of Snake River B-Run Steelhead populations are being monitored for population productivity and abundance. (Initiate by FY 2009)

Relationship: This project provided PIT tag-based estimates of run timing and fishery-specific harvest rates for wild and hatchery B-run steelhead populations at hierarchical spatial scales.

50.6: Review/modify existing fish pop status monitoring projects

Review and modify existing Action Agencies fish population status monitoring projects to improve their compliance with regional standards and protocols, and ensure they are prioritized and effectively focused on critical performance measures and populations. (Initiate in FY 2008)

Relationship: Through increased monitoring conducted under this project, WDFW was able to develop the first comprehensive LCR ESU wide VSP monitoring estimates of NOR abundance, pHOS, spatial distribution, and the first data points for several abundance and productivity metrics (adult to adult recruitment, smolt to adult returns, and natural origin spawners) for Washington Chinook and coho populations; including estimates of precision for the LCR ESU and for individual populations in an effort to meet NOAA monitoring guidelines (Crawford and Rumsey 2009).

51.1: Report available information on population viability metrics in annual and comprehensive evaluation reports. (Initiate in FY 2008).

Relationship: Reported population viability metrics and indicators for Washington's portion of the LCR Chinook and coho salmon. These indicators are formatted to be entered into the Coordinated Assessments data exchange standard.

51.1: Synthesize fish population metrics thru Regional Data Repositories. Support the coordination, data management, and annual synthesis of fish population metrics through Regional Data Repositories and reports such as the CBFWA State of the Resource. (Annually).

Relationship: In 2010, WDFW began implementing standardized data collection and storage protocols for information collected at fish traps and weirs, and during spawning ground surveys. Data was stored in corporate databases including WDFW's Spawning Ground Survey (SGS) and Age & Scales (A&S) databases. Also, WDFW began development of a regional relational database, entitled Traps, Weirs, & Surveys (TWS), to store all monitoring data in a single location and to further facilitate standardization of data collection, data entry, and quality assurance, in order to increase quality, efficiency, and improve analysis/reporting timeliness. This database will feed statewide corporate databases and regional reporting platforms as they are developed.

62.1: Evaluate the feasibility of obtaining PIT-tag recoveries between Bonneville and McNary dams (Zone 6) to determine whether recoveries can help refine estimates of in-river harvest rates and stray rates used to assess adult survival rates. For FY 2009, focus on a pilot to test the feasibility of PIT-tag recoveries of harvested fish in this reach (spring, summer, and fall Chinook salmon and summer steelhead). (Initiate in FY 2007-2009 Projects).

Relationship: This project developed PIT tag harvest rates by modifying the current mainstem lower Columbia River fisheries sampling program to include the collection of PIT tag information along with CWT recovery. Our PIT tag harvest rates provided harvest rates for fall Chinook and steelhead at a finer resolution than previously available. Using PIT tag methods we provided harvest rates for natural origin populations that are currently not available using CWT.

62.4: Support coded-wire tagging and coded-wire tag recovery operations that inform survival, straying, and harvest rates of hatchery fish by stock, rearing facility, release treatment, and location. (Initiate in FY 2007-2009 Projects)

Relationship: This project increased the frequency and intensity of spawning ground surveys in LCR tributaries for CWT recoveries. This lead to additional CWT recoveries and more precise CWT expansion estimators to estimate survival, straying and harvest rates by release group.

References

Crawford, B.A. and S. Rumsey. 2009. Guidance for monitoring recovery of salmon and steelhead listed under the federal Endangered Species Act (Idaho, Oregon, and Washington). Draft June 12, 2009.

Fall Chinook Salmon Escapement Estimates and Coded-Wire-Tag Recoveries in Washington's Lower Columbia River Tributaries in 2010

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Abstract

The Lower Columbia River (LCR) Chinook salmon Evolutionary Significant Unit (ESU) is composed of spring and fall Chinook populations split between the states of Washington and Oregon. Washington has been estimating abundance and age structure for all its fall Chinook salmon populations for decades but most were not meeting the accuracy and precision guidance for salmon recovery and there was no standardized reporting of important management and salmon recovery indicators. In 2010, the Washington Department of Fish and Wildlife (WDFW) initiated a monitoring program to estimate Chinook salmon spawner abundance, the proportion of hatchery origin spawners (pHOS), proportion by age, percent females, and spawning time, and to recover Coded Wire Tags (CWT). Due to difficulties in obtaining sufficient numbers of jacks the estimates we reported only include adult Chinook salmon (>59 cm) estimates. Adults were estimated using weir counts, open and closed mark-recapture models, Area-Under-the-Curve (AUC), redd counts, and peak count expansion depending on resources and survey conditions. We estimated 38,460 adult Tule, 65 adult Rogue River Bright hatchery, 9,612 adult Lewis River Bright natural origin , and 1,657 adult Bonneville (BON) Pool Bright fall Chinook salmon in the Washington portion of the LCR ESU. For Tules and BON Brights the proportion of marked adults was 79% and 58%, respectively. Operation of weirs successful reduced the proportion of marked spawners for some populations. The marked proportion is a slight underestimate of pHOS because approximately 98% of the hatchery juveniles were mass marked. For Tules most populations were comprised of primarily hatchery fish except the Coweeman (70% unmarked), Lewis (73% unmarked), Lower Gorge (77% unmarked), and the White Salmon (73% unmarked). Age structure varied by population but most Tule Chinook salmon were age 3 or 4. The adult sex ratio was skewed toward males for most populations. A total of 184 snouts were collected from the field and examined for CWT. CWT recoveries were uploaded to the regional coded-wire-tag database (RMIS) and unexpanded CWT recoveries indicate most Tule hatchery fish returned to the basin of release or an adjacent basin. BON Brights not native to this ESU are successfully spawning in the Upper Gorge and White Salmon populations. Rogue River Brights are successfully spawning in the Grays River. Assumption testing indicated our abundance and proportion estimates were relatively unbiased. Most adult abundance estimates had a coefficient of variation (CV) of less than 15%, and our proportion estimates had a 95% confidence interval of less than 5% except for natural origin ages in small populations or populations with few unmarked fish. Thus, the majority of adult Chinook monitoring meets the NOAA guidelines for accuracy and precision. This Chinook salmon monitoring program is the first in Washington to estimate multiple high level indicators and the associated uncertainty in these indicators at the population and ESU scales. The data was analyzed to report on important high level indicators for salmon recovery and monitoring.

Introduction

Chinook salmon (*Oncorhynchus tshawytcha*) in the Lower Columbia River (LCR) Evolutionary Significant Unit (ESU) were listed for protection under the Endangered Species Act (ESA) in 1998. In a recent five-year review, the National Oceanic and Atmospheric Administration (NOAA) Fisheries concluded that these fish should remain listed as threatened under the ESA (NOAA 2011). The LCR Chinook Salmon ESU is composed of spring and fall populations split between the states of Washington and Oregon (Myers et al. 2006). The Washington Department of Fish and Wildlife has monitored these populations for decades (WDFW 2011) and focused primarily on providing an abundance estimate. However, the need for monitoring of additional indicators and more accurate and precise estimates of these indicators, especially for the fall Chinook populations, has been identified as a high priority for salmon management and recovery (Rawding and Rodgers 2013, LCFRB 2004).

The coastwide Coded-Wire-Tag (CWT) program was developed in the 1970's to evaluate the contribution of different salmonid populations and hatchery programs to various fisheries and to estimate salmon fishery harvest rates, along with evaluation of hatchery rearing practices. The initial protocols for the CWT program included the insertion of a CWT into the snout of a juvenile hatchery salmon, which was accompanied by an adipose fin clip. A proportion of hatchery fish released from selected facilities had a CWT inserted. When salmon were recovered from fisheries and spawning areas, the snout of fish with missing adipose fins were taken to fisheries agency labs for decoding. Later the purpose of the CWT program was expanded to include forecasting run sizes to meet conservation and harvest objectives. For conservation purposes, the vast majority of Chinook salmon released from hatcheries are now adipose fin clipped (sometimes referred to as mass marked) and WDFW has implemented selective fisheries, which require the release of all adipose-intact (assumed to be natural origin) fish. CWTs are now detected electronically by scanning fish with handheld or stationary detectors, rather than using the adipose fin clip as an indicator of CWT presence.

In 2010, the Washington Department of Fish and Wildlife (WDFW) initiated a program to sample LCR spawning grounds for Chinook salmon. This program had dual objectives: 1) to estimate Viable Salmonid Population (VSP) indicators (McElhaney et al. 2000) and measure specific indicators to assess Chinook salmon viability (Rawding and Rodgers 2013) including Chinook salmon spawner abundance, the proportion of hatchery origin spawners, the proportion of spawning reaches occupied, spatial distribution, and sex ratio including the proportion of jacks; and 2) to recover CWT from spawning fish to provide complete accounting of CWTs, so that harvest rates could accurately be determined. The first objective addressed a salmon recovery monitoring gap while the second objective addressed a gap identified from the CWT expert panel (Hankin et al. 2005). This report summarizes population monitoring of VSP indicators for LCR Chinook salmon returns and CWT recoveries in 2010.

Methods

Study Area

The LCR Chinook salmon ESU extends from the mouth of the Columbia up to and including the Big White Salmon River in Washington and Hood River in Oregon, and includes the Willamette River to Willamette Falls, Oregon. Within this ESU, there are a total of 13 Washington populations, 8 Oregon populations, and 2 populations (Lower and Upper Gorge) that are split between the states (Figure 1). In this document we report on 11 populations in Washington. The Salmon Creek population is believed to have been extirpated, and it is unclear if the Lower Gorge historically supported a Chinook population, but if it did it is likely extirpated. The Lower Cowlitz and North Fork Lewis populations are surveyed in conjunction with hydropower companies and their results have been reported in other reports (Roler et al. 2011). In addition, we report on Rogue River Brights which has been established in the Grays River (Roegner et al. 2010), and Bonneville Pool Brights populations, which have become established in the Lower Gorge and White Salmon Rivers.

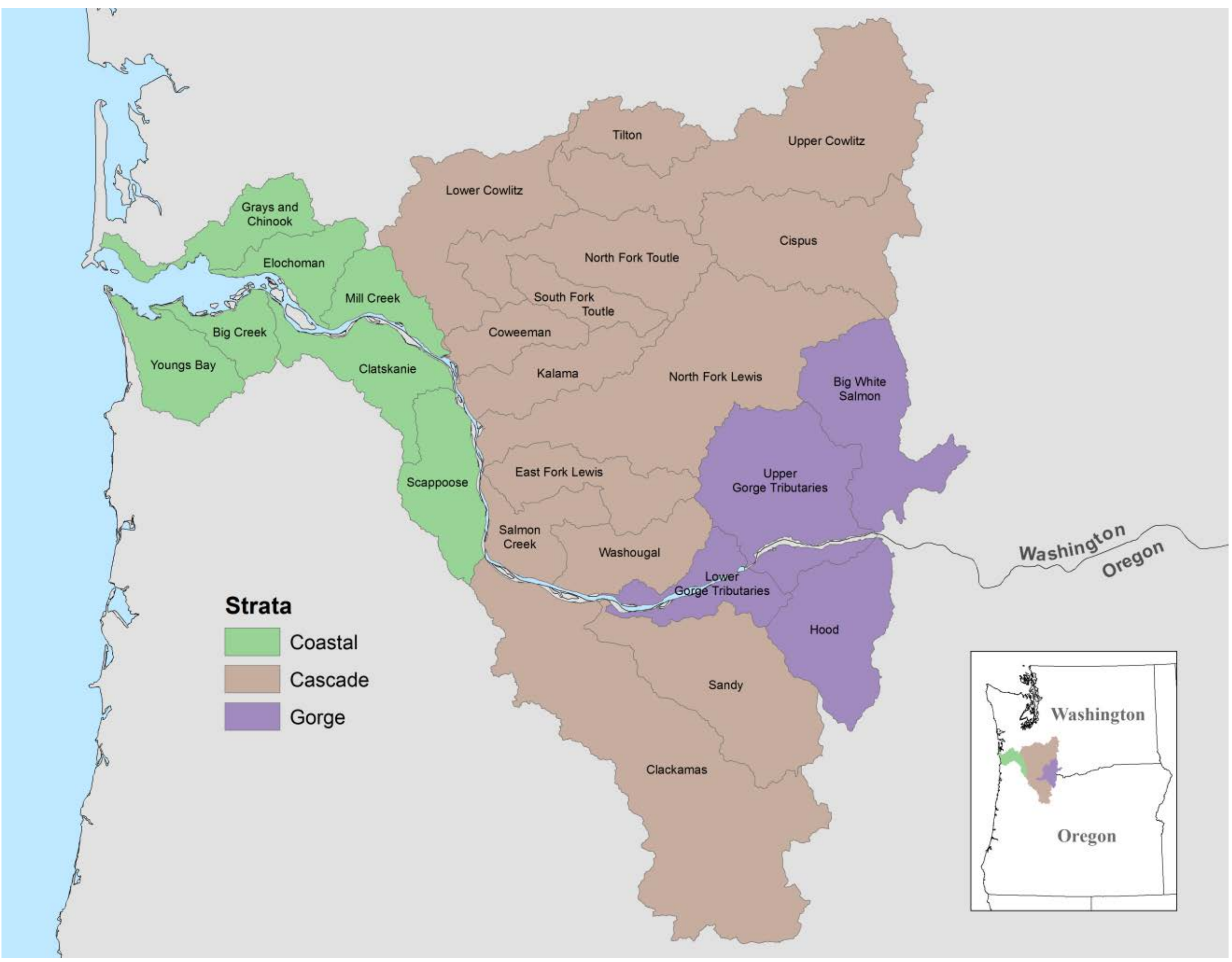

Figure 1. Lower Columbia River Chinook salmon populations and the regional groupings (i.e., strata) in which they occur within the LCR subunit recovery domain.

Monitoring Design

The Chinook salmon monitoring design for the study area used a variety of methods including weir counts, mark-recapture estimates based on live and carcass tagging, redd counts, and periodic counts of live spawners to estimate abundance (Schwarz and Taylor 1998; Sykes and Botsford 1986; Gallagher and Gallagher 2005; Parken et al. 2003) (Figure 2). When facilities existed we used weir counts because these are counts and not estimates, and provide the most accurate measure of escapement (Cousens et al. 1982). Weirs are located on the Grays River (RM 9), Elochoman River (RM 2), Cowlitz River (RM 51), Green River (RM 1), Kalama River (RM 2), and the Washougal River (RM 10). However, only census estimates were potentially available from the Elochoman, Green, Tilton, and Upper Cowlitz/Cispus because a portion of Chinook salmon by-passed the weirs at these other sites. We anticipated that the weirs would not provide a census, so all weir operations simultaneously implemented a mark-recapture design (Schwarz and Taylor 1998), where fish were tagged at the weir and recovered on spawning ground surveys (Grays, Elochoman, and Green rivers). We implemented carcass tagging mark-recapture studies (Sykes and Botsford 1986) in Grays, Skamokawa, Mill, Germany, Abernathy, Lower Green, South Fork (SF) Toutle, Coweeman, and Washougal basins. In some of our mark-recapture studies, we tagged and/or recovered too few fish to obtain a reliable estimate for mark-recapture abundance (Grays, Skamokawa, Abernathy, SF Toutle, Coweeman) and in these cases we counted live and dead fish, and redds during typical spawning ground survey to estimate escapement based on Area-Under-the Curve (AUC) and/or redd surveys using survey life and females per redd data from previous LCR studies or from 2010 where we obtained a reliable estimate of survey life. In, addition we obtained redd and AUC estimates for the Kalama and EF Lewis populations in 2010. An exception to this is we conducted a trans-generational genetic mark-recapture study (tGMR) for the Coweeman river (Rawding et al. 2013). For the Wind, Little White Salmon, and Big White Salmon rivers we used historic peak count expansion factors from JS carcass tagging projects in the 1960's and 1980's. As mentioned above estimates for the Lower Cowlitz and NF Lewis were conducted injunction with hydropower companies and are not reported here available and can be found in Roler et al. (2011).

In addition to spawner abundance, we report on the proportion of adult males and females, the proportion of marked and unmarked adults, and the proportion of age three to five year old Chinook salmon. To estimate these parameters, we obtained biological and tag data from handled fish or carcasses. Tag data included physical tags from mark-recapture studies, and coded-wire-tags (CWT). Biological data included scales for age, length, sex, mass mark status, and spawning success for females. Since almost all hatchery fish are adipose fin clipped, the proportion of marked fish is a surrogate for the proportion of hatchery fish.

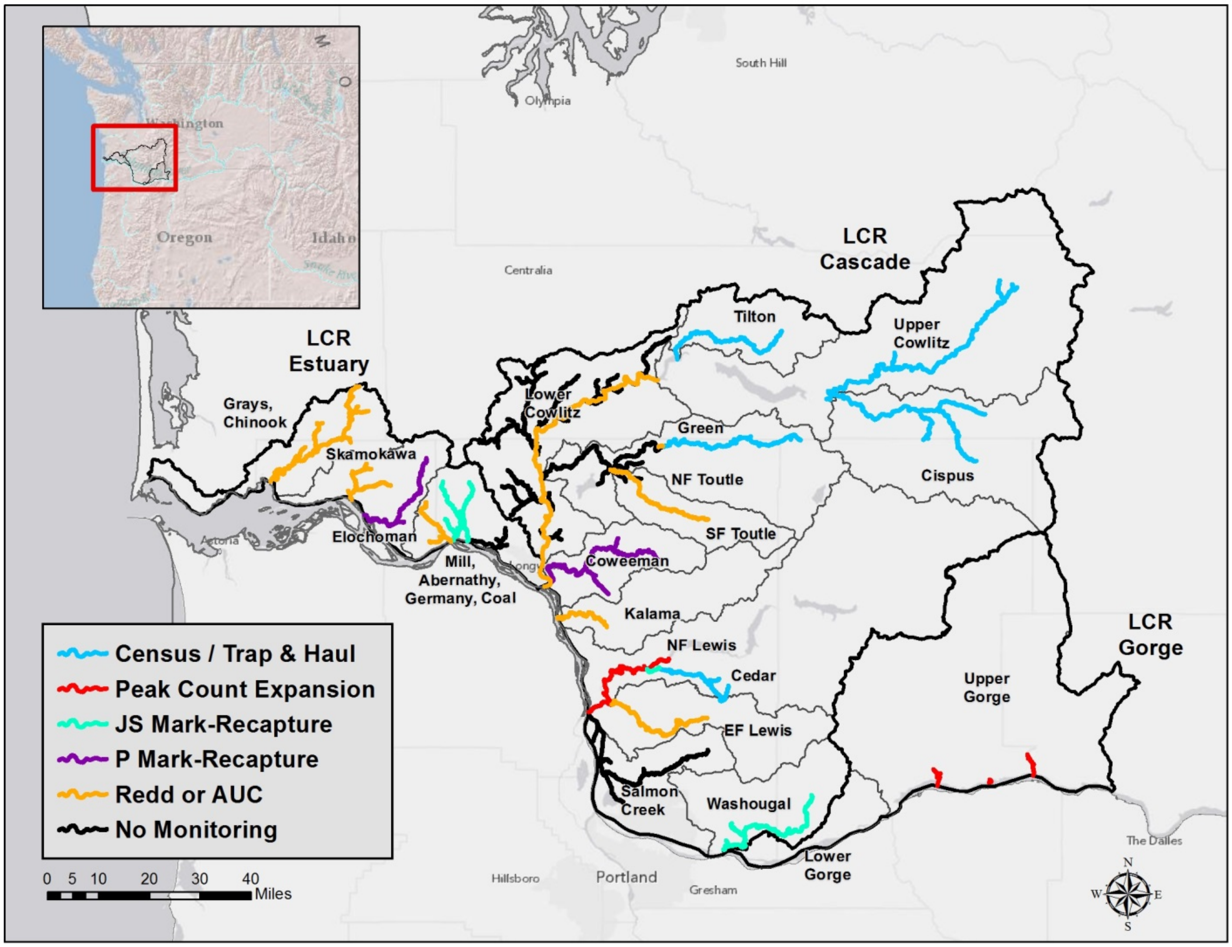

Figure 2. Watersheds containing the Washington populations of the Lower Columbia River Chinook salmon ESU and the methods WDFW used to estimate their abundance.

Weir

Temporary weirs were successfully operated to estimate escapement and obtain biological data in the Elochoman River (RM 2) and Green River (RM 1). The Barrier Dam on the Cowlitz River (RM 51), and the Toutle Fish Collection Facility (TFCF) on the NF Toutle River (RM 12) were also operated. No Chinook salmon were transported into the NF Toutle River because there are no mainstem release sites above the Sediment Retention Structure (SRS) for Chinook salmon trapped at the TFCF. In Cedar Creek, a tributary to the NF Lewis River, a trap was operated in the fishway (RM 2). Since the recapture event above the trap was not successful, we assumed the ladder count was a census.

Depending on management objectives Chinook salmon collected at these facilities were used for hatchery broodstock, donated to food banks, surplused, used for nutrient enhancement, or transported and released above the facility. We made the following key assumptions for the weir program: 1) the count of all transported fish was without error, 2) all unmarked fish released survived to spawn except on the Green and Elochoman where we had estimates of pre-spawning mortality, 3) transported fish spawned in the watershed they were released (there was no fall back), 4) when fisheries in the Elochoman, Upper Cowlitz, Cispus, and Tilton Rivers occurred only marked (adipose clipped fish) were harvested in accordance with regulations, 5) there was no illegal harvest of salmon, 6) survival of all unmarked caught and released fish was 100%, and 6) the WDFW methodology to expand catch record card (CRC) reported catch to total harvest and variance are correct.

Closed Population Models

To validate the success of weir operations in the Elochoman and Green rivers, we successfully implemented mark-recapture studies. Chinook salmon were captured in adult traps located adjacent to the temporary weirs. After release, fish were recaptured at upstream traps or as carcasses during spawning ground surveys. The study design at all three locations was based on the Darroch estimator, which was developed for time stratified Petersen mark-recapture abundance estimates (Darroch 1961, Seber 1982). Schwarz and Taylor (1998) indicate that the following assumptions must be met to provide a consistent estimate of abundance for the Petersen estimator: 1) no tag loss, 2) no handling mortality, 3) all tagged and untagged fish are correctly reported, 4) the population is closed, and 5) equal capture probability during the tagging or recapture events, or marked fish mix uniformly with unmarked fish.

We also used tGMR to estimate abundance on the Coweeman River (Rawding et al. 2013). The marks were the genotyped carcasses collected from the spawning area during the first sampling event. The second sampling event consisted of a collection of juveniles from a downstream migrant trap located below the spawning area. The parents that assigned to the juveniles through parentage analysis were considered the recaptures, which was a subset of the genotypes captured in the second sample. The same mark-recapture assumptions reported above are needed for tGMR except the last assumption is modified because of heterogeneity in the number of offspring per spawner (Anderson et al. 2012). Therefore, assumption 5, the equal capture assumption, is equal probability of being tagged in the first sample or the probability of being captured in both samples are independent or not correlated (Lukcas and Burnham 2005, Schwarz and Taylor 1998, Rawding 2013).

Jolly Seber Model for Open Populations

The Jolly-Seber (JS) model estimates population abundance in mark-recapture studies where the population is open (Jolly 1965; Seber 1965) and has been widely used in estimating Pacific salmon spawning escapement from live fish (Schwarz et al. 1993; Jones and McPherson 1997; Rawding and Hillson 2003) but also using salmon carcasses (Parker 1968; Stauffer 1970; Sykes and Botsford 1986) The carcass tagging model has been used extensively in the Lower Columbia to estimate Chinook salmon abundance (McIssac 1977; Rawding et al. 2006). The JS model utilized carcass tagging for mark-recapture and **w**as the primary method of estimating escapement in this study. In 2010, we developed estimates for Mill, Germany, Lower Cedar Creek, and the Washougal basins. Seber (1982) and Pollock et al. (1990) provide details of study design, assumptions, and analysis of mark-recapture experiments using the JS model. Five assumptions of the Jolly Seber model must be met in order to obtain unbiased population estimates from the model (Seber 1982) are: 1) equal catchability 2) equal survival between periods, 3) no handling mortality, 4) no tag loss, and 5) instantaneous sampling.

Peak Count Expansion

We used historic JS estimates to develop peak count expansion (PCE) factors for the Wind, Little White Salmon, and Big White Salmon populations. There are a number of ways to estimate the peak count expansion factor including the mean of the ratios (Parken et al. 2003), the ratio of the means, calibrated regression, and inverse prediction (Parsons and Skalski 2009). We divided the posterior distribution of the abundance estimate by the peak count of live fish and carcasses to obtain a PCE. Rawding and Rogers (2013) list the following critical assumptions for the PCE method: 1) the peak day of abundance is known and the survey takes place on the peak, 2) if the entire spawning distribution is not surveyed, the proportion of fish used in the index or indices is similar to that of the years used to develop the peak count expansion factor, 3) observer efficiency is similar in all years, and 4) the proportion of fish observed on the peak day is similar over all years.

Spawning Ground Surveys

The purpose of spawning ground surveys was to collect the data required to estimate abundance, to tag fish and recover tag data, and to collect biological information from sampled fish. Surveys were scheduled weekly from the beginning of fish entry into the system (August to September) until completion of spawning (October to December), depending on the population, and over the entire spawning distribution as developed by Rawding et al. (2010). Exceptions were cases where PCE was used to estimate abundance; for those areas, surveys were scheduled for 3 weeks near the historical peak spawning week in the index area to identify the actual peak week. Since we had no successful weir or mark-recapture estimates on the Grays, Skamokawa, Abernathy, Lower Green, SF Toutle, Coweeman, Kalama, and EF Lewis rivers we needed alternate methods to estimate abundance. We used previous JS estimates in conjunction with redd surveys to develop estimates of apparent redds per female on the Coweeman and EF Lewis rivers from 2003-2009, where the apparent redds per female is the redd count divided by the mark-recapture estimate of females. In addition, we used the 2010 mark-recapture data to develop estimates of apparent survey life from the Elochoman, Mill, Abernathy, Coweeman, and Washougal populations, where apparent residence time is the estimate of Area-Under-the-Curve in fish days divided by the mark-recapture adult escapement estimate. These redds per female and apparent survey life estimates were applied to redd and live fish counts to estimate abundance in the Grays, Skamokawa, SF Toutle, Lower Green, Kalama, and EF Lewis rivers.

Rawding and Rodgers (2013) listed the critical assumptions for redd surveys used to estimate abundance: 1) representative spatial and temporal sampling throughout the spawning period, 2) estimates of apparent females-per-redd, which are from adjacent populations or from the same population in previous years, are consistent between the study population (one used to derive the redds per female estimate) and the treatment population,. This assumes the methods used to identify and enumerate females-per-redd follow standard redd survey protocols. The third critical assumption is that the apparent females-per-redd and sex ratio from other streams or years accurately represent the females-per-redd and sex ratio for the population in the current year.
For the AUC method, Rawding and Rodgers (2013) identified the first assumption is that representative spatial and temporal sampling occurs throughout the spawning period. If concurrent observer efficiency and survey life estimates are made, the second critical assumption for AUC is that these estimates are spatially and temporally representative of the survey area and occur throughout the spawning period. Concurrent observer efficiency and survey life studies are

costly, so AUC spawner abundance estimates often rely on observation efficiency and survey life from adjacent populations or from the same populations in other years. The use of these kinds of surrogate estimates in calculations of spawner abundance should be carefully considered. Finally, survey frequency should occur every 7 to 10 days and surveys should not be missed during peak spawning time (Hill 1997).

The methods used to estimate escapement for each monitoring unit are found in Table 1.

Table 1. Methods used to estimate fall Chinook salmon escapement in 2010.

Subpopulation	Escapement Method
Grays	Redds with mean FpR and population specific % females
Skamokawa	AUC with mean survey life from 2010
Elochoman	2010 Petersen Estimate minus prespawn mortality & CRC harvest
Mill	Jolly Seber model based on carcass tagging
Abernathy	AUC with mean survey life from 2010
Germany	Jolly Seber model based on carcass tagging
Lower Cowlitz (Tule)	Redd based estimate from Roler 2013
Tilton	2010 census count minus CRC harvest
Upper Cowlitz/Cispus	2010 census count minus CRC harvest
Green	Above weir using a census minus prespawn mortality and CRC harvest & below weir AUC with mean survey life from 2010
SF Toutle	Redds with mean FpR and population specific % females
Coweeman	tGMR using Petersen (Binomial) estimate
Kalama	AUC with mean survey life from 2010
NF Lewis (Tule)	Carcass expansion by age based on carcass recovery rates from 2003 JS carcass tagging estimate. Assume recovery rates by age are constant over the season.
Cedar	Census count at ladder with Jolly Seber model based on carcass tagging below ladder adjust for ladder fallbacks
EF Lewis	AUC with mean survey life from 2010
Washougal	plus above census count above weir
Wind	PCE based on 1964 Tule Jolly-Seber carcass tagging model
Little White Salmon	PCE of carcasses based on 1966 Jolly-Seber Tule carcass tagging model
Big White Salmon	PCE based on 1989 Jolly-Seber Bright carcass tagging model
Grays (Rogue)	Redds with mean FpR and population specific % females and the % or Rogue Bright based on right ventral clip
NF Lewis (Brights)	Carcass tagging expansion from Roler 2013
Hamilton (Brights)	PCE based on professional judgment (Roler et al. 2013)
Col. R Ives (Brights)	AUC based on historic residence time
Wind (Brights)	PCE based on 1964 Tule Jolly-Seber carcass tagging model
L. White Salmon(Brights)	PCE of carcasses based on 1966 Jolly-Seber Tule carcass tagging model
B. White Salmon (Brights)	PCE based on 1989 Jolly-Seber Bright carcass tagging model

Data Collection

Traps and Weirs

Data collection at weirs was similar to the standardized methods for collecting salmon data at weirs described in Zimmerman and Zubkar 2007. Chinook salmon populations originating above dams in the Cowlitz watershed were trapped and hauled into the Tilton and Upper Cowlitz /Cispus rivers allowing for their enumeration and the collection of biological data. Cowlitz River Chinook salmon captured at the Barrier Dam were anesthetized using electro-anesthesia and sampled for sex and origin. In addition, male Chinook salmon were classified as jacks or adults based on size. Adult salmon captured at the Barrier Dam were released to their natal watersheds based upon differential marking they received as smolts when they were transported downstream of the Cowlitz dams; since outmigrants caught at the Mayfield trap were tagged with blank CWT and not adipose fin clipped, these fish were released in the Tilton River which empties into Mayfield Lake, whereas unmarked adipose intact fish were transported to the Upper Cowlitz and Cispus Rivers where they presumably originated. In addition, adipose clipped hatchery Chinook salmon were released in the Tilton, Upper Cowlitz and Cispus rivers to provide recreational fishing opportunity and spawners to seed the available habitat. This action is needed because the current juvenile collection rate for fall Chinook at the Cowlitz falls dam is too low to support self-sustaining runs (Serl and Morrill 2009). Other Chinook salmon collected at the Barrier dam were used for broodstock, surplused, and/or used for nutrient enhancement.

Temporary weirs were installed on the Elochoman and Green rivers on August 8, 2010 and August 26, 2010, respectively. The Elochoman weir was removed during a high water event on October 10, and not reinstalled because Chinook entry into the Elochoman was believed to be complete. The Green River weir operated into mid-November for a coho salmon study which is well past the last observed Chinook salmon. Because of the possibility of weir failure due to high water events, we tagged fish to implement a mark-recapture study at both locations. Fish in good condition were bio-sampled, double Floy tagged (FD 68BC T-bar Anchor tags; Floy Tag & Mfg., Inc. Seattle, WA) and released upstream in Elochoman and Green rivers. Tags were uniquely numbered and placed adjacent to the posterior edge of the dorsal fin, with one on each side of the fish. Opercle punches were applied as a secondary mark, and punch shapes were rotated weekly, allowing assessment of Floy tag (ft) loss and assignment of a recovered fish back to the weekly release group if both tags were lost. Bio-sampling consisted of sex determination, measuring the fork length, and recording fin clips to determine mass mark status (hatchery or natural origin). All fish captured at the Elochoman River were tagged and every other fish captured was tagged on the Green River. A trap was operated in the fishway at the Grist Mill falls on Cedar Creek the entire migration period.

Except for the Cowlitz trap, scales were taken from live fish from the preferred area of the fish, as described in Crawford et al. (2007b) and repeated below in the Spawning Ground Survey section. For a bio-sampled fish, the fork length is taken by running the tape measure from the tip of the snout to the fork in the tail. Sex is determined based on morphometric differences between males and females, mass mark status was determined by the presence or absence of the adipose fin.

Spawning Ground Surveys.
These surveys consisted of three components: 1) biological sampling, 2) fish tagging and tag recovery, and 3) periodic counts of live fish, carcasses and redds, which are used to estimate abundance. Data collection during scheduled weekly spawning ground surveys was similar to the standardized methods for collecting salmon data from carcass counts, redd surveys, and foot-based visual counts (Crawford et al. 2007a, Gallagher et al. 2007, and Crawford et al. 2007b).

All carcasses that were not totally decomposed were sampled for external tags (Floy T-bar or opercle tags) and biologically sampled for fork length, sex, adipose fin presence, and condition (extent of decomposition). Sex is determined based on morphometric differences between males and females. If necessary, the abdominal cavity is cut open to confirm sex and determine spawning success. The spawning success was approximated based on visual inspection, ranging from 100% to 0% success. A fish with 0% spawning success or 100% egg retention was considered a pre-spawning mortality. Carcass condition and gill color were recorded to qualitatively rate carcass (Sykes and Botsford 1986). Scale samples were collected by selecting scales from the preferred area as described in Crawford et al. (2007b). Preferred scales are samples in an area ~ 1-6 scale rows high, and ~15 scale rows wide, above the lateral line in a diagonal between the posterior insertion of the dorsal fin and anterior insertion of the anal fin. Scale samples are removed with forceps with special care to select scale samples that are of good quality (round shape, non-regenerated) and not adjacent to one another (to minimize the effects of regeneration) as described in a WDFW technical report (Cooper et al. 2011). Scales are placed on the gummed portion of WDFW scale cards with their exterior surfaces facing up. The scale card number, position number, date, and location create a unique code in the age and scales (A&S) database. Due to a high number of carcasses on Washougal and Kalama these fish were systematically samples for scales.

For Chinook salmon carcasses, fish were enumerated by the following categories: unmarked, marked, and unknown. Unmarked fish are Chinook with intact adipose fins and snout, marked fish have their snout but are missing their adipose fin, and unknown fish are salmon with either a damaged caudal peduncle (e.g. adipose fin area unexaminable) or missing snout. All unmarked and marked fish are sampled for CWT following standard protocols (NWMFT 2001). The surface of the CWT wand with radiating arrows is placed in contact with the snout and moved from the right to the left eye, and then up and over the snout area. The wand is also inserted into the mouth with the radiating arrows rubbed against the roof of the mouth in vertical strokes. If a CWT is detected, the red LED will light up and a beep is emitted from the wand. When a CWT is detected, the snout is severed by cutting across the head straight down behind the eyes (Crawford et al. 2007b). The snout is placed in a plastic bag with a tag number linking the snout to biological data (length, sex, fin clips, spawning success for females, and scale sample number) recorded on the scale card, stream survey card, or other datasheet. Snouts are stored in a freezer and periodically delivered to the WDFW CWT lab in Olympia.

All carcasses were inspected for tags. Untagged carcasses were tagged on both opercles with uniquely numbered plastic tags (McIssac 1977). Tags were placed on the inside of the opercle to limit predation and potential bias in recovery rates due to observation of brightly colored tags. Tagged carcasses were then placed into moving water to facilitate mixing with untagged carcasses (Sykes and Botsford 1986). When tagged carcasses were recovered, surveyors recorded the tag numbers , the tags were removed and fish were marked by removing the tail (denoted as loss on capture in the Jolly-Seber model).

In addition, all live adult and jack salmonids were identified to species based on physical characteristics unique to each species and recorded by species (Crawford et al. 2007a). We used a 60cm cut off between adult and jack salmon although this cut off is difficult to accurately determine during visual surveys. However, since few fish are near 60cm the misclassification errors are believed to be low. Salmon were identified as either spawning or holding. A fish was identified as holding if it was observed in an area not considered spawning habitat, such as pools or large cobble and boulder riffles (Parken et al. 2003). Salmon were classified as spawners if they were on redds or not classified as holders. Counts of live Chinook, coho, and chum salmon were recorded separately for each survey reach.

Redd surveys in the Grays, Coweeman, EF Lewis, and the SF Toutle, followed the protocols of Gallagher et al. (2007). The start and end of each survey reach was geo-referenced and its coordinates were recorded on a on a Garmin Oregon 550 unit set in NAD 83. Surveyors typically located the upper most point in the reach and walked downstream to the coordinates at the end of the reach. Surveys were scheduled weekly and followed methods in Rawding et al. (2006, 2006b). All identifiable redds were flagged, and their location (latitudinal and longitudinal coordinates) was recorded. GPS units were allowed to acquire satellite locations until an accuracy of ± 100 feet or less was obtained, most often accuracies averaged 5 to 50 feet. In subsequent surveys, previously flagged redds were inspected to determine if they should be classified as “still visible” or “not visible”. A redd was classified as “still visible” if it would have been observed and identified without the flagging present, and was recorded as “not visible” if it did not meet this criteria. These data were collected to allow us to estimate the time period redds were visible to surveyors.

Experienced field personnel were employed for this project when possible; all personnel were trained in adult salmon identification, redd identification, and sampling/tagging protocols (Crawford et al. 2007a, Gallagher et al. 2007, and Crawford et al. 2007b). Training took place in orientation meetings and with field supervisors. When possible field supervisors also walked behind surveyors to check on redd identification and enumeration, carcasses tagging, and live counts.

Trans-generational Genetic Mark-Recapture

Carcass in the Coweeman River were sampled as described above except a small portion of tissue from all carcasses was excised from the operculum and placed in a labeled vial with 100% non-denatured ethanol for subsequent genetic analysis. The following year a downstream outmigrant trap was operated at Rkm 10 from the February through October following American Fishery Society protocols (Lamperth et al 2013,Volkhardt et al. 2007). The weekly outmigration estimate was adjusted it based on instantaneous mortality and juvenile outmigrants were proportionally sub-sampled for genotyping by excising a small portion of fin tissue.

Sample Processing

Scale Analysis

Scale preparation and analysis followed WDFW protocols (Cooper et al 2011). Acetate impressions are made of the scale samples by a scale card press, where samples are covered with clear acetate (0.5mm thickness) and pressed under 1200-1300 PSI @ 100 degrees C for 30 seconds to one minute. Acetate impressions are then slightly cooled and removed from the scale card. Acetate impressions of scale samples are aged using a modified Gilbert/Rich ageing notation (Groot and Margolis 1991), where annuli are counted along with the scale edge to produce a total age in years. Annuli are defined as an area of narrowly spaced circuli that represent winter/early spring growth. Age is recorded as the total age in years followed by the year at outmigration. For example a typical Chinook salmon adult is age 4_1. This notation indicates a total age of 4 and the juvenile salmon left its natal freshwater habitat within its first year of life. After being aged in Olympia by an aging specialist, scale samples were returned for entry into the Age & Scales database.

CWT Lab Analysis

The recovery of CWT tags at the WDFW lab follows the procedures outlined in the tag recovery chapter (Blankenship and Hiezer 1978) of the Pacific Coast Coded Wire Tag Manual and is briefly repeated here. Each snout is passed through a magnetic detector to determine tagged and untagged snouts. Untagged snouts are set aside and rechecked after magnetizing the tag. To ensure the tag is magnetized the length of the tag must pass through the horseshoe magnet in a plane parallel with a straight line collecting the poles. If the tag angle is off more than 40 degrees the tag may not be magnetized. Therefore, the head is passed through the magnet in three positions corresponding to the X, Y, and Z axes and then through the detector. Large heads are often dissected to maximize tag detections. Snouts with no tags are saved and an x-ray machine is periodically used to determine tag presence in these “no tagged” snouts. After determining a tag is present, the snout is dissected, and the tag is located by process of elimination. After recovering the tag, the binary code is determined by careful observation under a microscope. CWT data is then entered into WDFW CWT access database, and provided to managers as needed and uploaded into the Regional Mark Information System (RMIS).

Genetic and Parentage Analysis

Polymerase chain reaction (PCR) amplification was performed on carcass and juvenile tissues using the 13 fluorescently end-labeled microsatellite marker loci standardized as part of the Genetic Analysis of Pacific Salmonids (GAPS) project (Seeb et al. 2007) and an additional locus *Ssa*197 (O’Reilly et al. 1996). Carcass and juvenile samples were analyzed independently by two people to reduce the genotyping errors. Categorical allocation methods were used to identify the single most-likely parent from a group of non-excluded parents because these methods have been shown to be accurate and perform better than exclusion (Wang 2004, Kalinowski et al. 2007, Harrison et al. 2013). FRANz was used to assign parents to offspring using the multilocus genotypes and maximum likelihood methods (Riester at al. 2009). Although Rawding et al. (2013) used COLONY 2.0 for assignments, FRANz was used for this report because the algorithms in COLONY may take months to converge and final estimates were not available at the time of this report. In addition, there has been no statistical difference in the overall assignment between the two different software (Small et al. 2013, Seamons et al. 2013). The summarized output from FRANz provided the summary statistics needed to estimate spawner abundance. In tGMR these include the marks which are the number of genotyped marked carcasses, recapture which are the number of juvenile genotypes assigned based to these marked carcasses, and the captures which are the total number of juvenile genotypes.

Data Analysis

Overview

Chinook salmon abundance estimation was relatively straightforward for mark-recapture, trap and haul, and peak count expansion areas, but required combining multiple sources of information for AUC and redd survey areas. Briefly, a spawning habitat model was developed for the ESU to predict the extent of spawning habitat (i.e. the spawning habitat sampling frame) (Rawding et al. 2010). Either the entire sampling frame was surveyed weekly or an index reach was surveyed weekly with the entire sampling frame surveyed near peak abundance. The estimate for the remainder of the frame was based on the ratio of the total count within the index compared to the count within the index on the day the entire sampling frame was surveyed. For the Grays, Skamokawa, SF Toutle, Coweeman, EF Lewis, and Washougal rivers some surveys were missed or ineffective due to high turbid water, and redd and AUC counts were adjusted based on interpolation between survey dates. The total redd count was converted to a total female count by applying a LCR estimate of females per redd (FpR) based on the ratio of female abundance to census redd counts in mark-recapture basins. Female abundance was converted to adult abundance based on sex ratios. The estimated abundance of marked (hatchery fish) and unmarked fish (offspring of natural origin spawners) was based on marked to unmarked ratios from carcass recoveries during spawning ground surveys.

Modeling Approach

Data analysis was conducted using a Bayesian framework. Bayes rule states the posterior distribution, $p(\theta|y)$, is the product of the prior distribution, $p(\theta)$, and the probability of the data given the model or likelihood, $p(y|\theta)$, which is expressed by

$$p(\theta \mid y) = \frac{p(\theta)\, p(y \mid \theta)}{p(y)} \tag{1}$$

Where y are the data, θ are the parameters, and $p(y) = \Sigma_\theta\, p(\theta)p(y|\theta)$ for all discrete values or $p(y) = \int p(\theta)\, p(y \mid \theta) d\theta$ for continuous data (Gelman et al. 2004). The formula of the posterior distribution may be complex and difficult to directly calculate. Samples from the posterior distribution can often be obtained using Markov chain Monte Carlo (MCMC) simulations (Gilks et al. 1995). WinBUGS is software package that implements MCMC simulations using a Metropolis within Gibbs sampling algorithm (Spiegelhalter et al. 2003) and has been used to estimate fish abundance (Rivot and Prevost 2002, Su et al. 2001, Link and Barker 2010). For the Bayesian methods we tested the sensitivity of the prior and convergence based on the Brook-Gelman-Rubin statistic (Su et al. 2002, appendix 1).

We specified vague priors for parameters because this was the first comprehensive study to estimate fall Chinook salmon in the Washington's portion of LCR, there was little prior information and we wanted and objective analysis to "let the data speak for themselves". There are not consensus reference priors for objective Bayesian analysis, although there has been much work in this area (Tuyl et al. 2013). For the binomial or multinomial distributions we chose to evaluate the beta and Dirichlet priors parameterized with α = β = 1 & 0.5, which are the Bayes-LaPlace uniform prior and the Jefferies prior, respectively. We adopted the Bayes/LaPlace prior for our analysis but conducted a sensitivity analysis by the comparing the results of the two priors in select cases. For abundance estimates in mark-recapture we chose a uniform prior, so that the minimum and maximum bounds did not truncate the posterior distribution.

Bayes rule was used for null hypothesis testing based on the Bayes factor, which is similar to the likelihood ratio. However the likelihood ratio is based on the parameters that maximize the likelihood while the Bayes Factor (BF) integrates over the values of the parameter specified by the prior distribution. The BF can be expressed as

$$\Pr(H_i \mid D) = \frac{\Pr(H_i)\,\text{x}\,\Pr(D \mid H_i)}{\sum_j \Pr(H_j)\,\text{x}\,\Pr(D \mid H_j)} \qquad (2)$$

where D is the data, H is the hypothesis, Pr is the probability, and subscipts i and j indicate individual and multiple hypotheses, respectively. Zhou (2002) and Murdoch et al. (2010) noted that Chinook salmon carcass recoveries are biased toward longer fish and females. For the equal capture assumption, we used a logistic regression to test the null hypothesis that there is no difference in recapture probabilities by sex or length (Link and Barker 2006).

$$\text{logit}(\pi_i) = B_0 + B_1 x_{1i} + \ldots B_k x_{ki}\,. \qquad (3)$$

where model 1:constant, model 2: sex, model 3: length, and model 4:sex + length. The covariates were centered and standardized to allow for better mixing and interpretation of the covariates and we used a reverse jump markov chain monte carlo (RJMCMC) to simultaneously evaluate all models and calculate the posterior model probabilities. Due to the sensitivity of Bayes Factors to vague priors, we flowed the approach in Link and Barker (2006) to assign mean zero normal priors with variance $V/(k_i + 1)$ to the regression coefficients. This way, regardless of the number of parameters in the model, the total prior uncertainty in the linear predictor is fixed. We used an inverse Gamma prior (3.2890/7.8014) for the variance because this distribution for the logit(π_i) so that π_i are approximately uniform (0,1) . The Bayes Factors were calculated from the model posteriors (Link and Barker 2006, 2010).

To test independence between the mark and recovery events for the tGMR (Rawding et al. 2013), we estimated covariate inclusion and posterior model probabilities from a negative binomial regression where offspring per spawner was the dependent variable and location, age, sex, and origin were the independent variables using the Kuo-Mallick sampler (Kuo and Mallick 1998). Covariates were centered and standardized and with the regression coefficient considered part of a hierarchical model (O' Hara and Sillanpaa 2009). For these regressions, we evaluated support for the null and alternative hypotheses and interpreted Bayes factors based on the scale provided by Kass and Rafferty (1995) found in Table 2.

Table 2. Bayes factor interpretation from Kass and Rafferty (1995).

B_{10}	Evidence against H_0	B_{10}	Evidence for H_0
1 – 3	Negligible Support	1-0.33	Negligible Support
3 – 20	Positive Support	0.33 -0.05	Positive Support
20-150	Strong Support	0.05-0.0067	Strong Support
> 150	Very Strong Support	<0.0067	Very Strong Support

The Bayes factors for proportions were computed analytically from the marginal likelihood using the beta binomial distribution (Ntzoufras 2009, page 399). Bayes factors are known to be sensitive to the prior distribution to estimate θ. However, since our analysis is limited to comparison of binomial models, we adopted the standard reference prior which is a uniform distribution between 0 and 1. We tested the null hypothesis using this method to test the complete mixing and equal proportion hypothesis tests (Schwarz and Taylor 1998) to determine if the pooled Petersen was appropriate (see closed population abundance estimates below). These null hypothesis tests are: 1) there is no difference in the proportion of marked fish by recovery period, and 2) there is no difference the proportion of recovered fish by release periods. In addition, we also tested the null hypothesis that the there was no difference in the proportion of marked fish by recovery location.

Due to computational challenges it is difficult to estimate Bayes Factors when using MCMC approaches (Ntzoufras et al. 2009, Lunn et al. 2012) and this occurred in mark-recapture model selection. As a practical solution we limited the number of JS models to four (Table 3), and used the Deviance Information Criteria (DIC) developed by Spiegelhalter et al. (2002) for model selection:

$$DIC = Dev(\theta_m) + pv \tag{4}$$

where $D(\theta_m)$ is the posterior mean deviance for the model and $pv = Var(D(\theta|Y))/2$ and is a measure of the number of effective terms in the model. We choose pv over the more commonly used pD for an estimate of effective parameters, because pv performs well when there is weak prior information and is invariant to parameterization (Gelman et al. 2004). DIC is a Bayesian analog of Alkiake Information Criteria (AIC) but based on MCMC outputs. Similar to the model support scale developed by Burnham and Anderson (2002), Spiegelhalter et al. (2002) suggested that models ΔDIC of less than 2 have considerable support, models with ΔDIC having 3-7 have less support, and models with ΔDIC > 10 have negligible support.

Table 3. Model notation used for JS carcass tagging (from Lebreton et al. 1992). Models names indicate whether capture, survival, or entrance probabilities were allowed to vary over time ("t") or were held constant ("s = same").

Model	Probability of capture (p)	Probability of survival (φ)	Probability of entry (b^*)
t t t	varies over periods	varies over periods	varies over periods
s t t	equal over periods	varies over periods	varies over periods
t s t	varies over periods	equal over periods	varies over periods
s s t	equal over periods	equal over periods	varies over periods

Goodness of Fit (GOF) Tests

The purpose of a GOF test is to identify potential inadequacies in the fit of the model to the observed data. One Bayesian approach used for GOF testing is posterior predictive checking, which is a comparison of the posterior predictive distribution of replicated data from the model with the data analyzed by the model (Gelman et al. 2004). In other words, the predictive data ($y.rep_i$) is the expected observation after replicating study having observed the data (y_i) and assuming the model is true. When using MCMC simulations a measure of discrepancy (D) is computed for the actual and replicated datasets for each iteration. An assessment of the posterior distributions of $D(y^{rep},\theta)$ and $D(y,\theta|y)$ provides individual and overall GOF measures. The posterior or Bayesian P-value = $\Pr(D(y^{rep},\theta) > D(y,\theta|y)$. The interpretation of the Bayesian P-value is the proportion of the times the discrepancy measure of the replicated data is more extreme than the observed data. If there is a good fit of the model to the data, we would expect the observed data to be similar to the replicated data, resulting in a Bayesian P-value of 0.50 while values near 0 or 1 indicate that the model does not fit the data.

There are many possible types of discrepancy measures including the Freeman-Tukey, standardized Pearson residual, chi-square, and deviance statistics (Brooks et al. 2000, Lunn et al. 2012). Since mark recapture counts consist of many zero and this test statistic does not require the pooling of bins with small or zero values we used the Freeman-Tukey statistic (Brooks et al. 2000), which is expressed as

$$d_i(\theta) = \sqrt{y_i} - \sqrt{E(y_i \mid \theta)} \tag{5}$$

where d_i is an individual discrepancy, y_i is an individual data point, and $E(y_i|\theta)$ is the fitted value of y_i based on the function to determine the parameter θ. When estimating independent values such as the proportion of hatchery fish or the age of hatchery fish in a single population, Bayesian P-values are typically near 0.5. Although Bayesian P -values are commonly used for model checking, there have been criticisms of this approach. First, it uses the data twice to build and check the model, which may not be as robust as other methods for testing model adequacy (Carlin et al. 2009, Kery 2010). Second, it is unclear what cut off values to use for the interval (5% to 95%) to indicate lack of model fit. Third, the posterior distribution is influenced by the prior distribution, thus a Bayesian P-value is influenced by the prior distribution (Brooks et al. 2000). These concerns are addressed but beyond the scope of this paper (Gelman et al. 2004, Carlin et al. 2009, and Brooks et al. 2000). Due to these concerns we used posterior predictive model checking as a qualitative measure of model adequacy and if a Bayesian P-value indicated the model did not fit the data, we considered this to indicate significant lack of model fit (Link and Barker 2009). We primarily used GOF to test the recapture portion of the JS model, which is similar to the RELEASE GOF test or parametric bootstrapping in the program MARK (Citation), and to test the recapture portion of the Darroch model.

The mode, median, and mean are commonly reported measures of central tendency for posterior distributions, which are reported in the form of point estimates. The mode is the most frequent value in the dataset. The middle value of the data is the median and the mean is the sum of the numbers in the dataset divided by the numbers in the dataset. For symmetrical distributions these measures of central tendency are the same. However, for asymmetric distributions it is not always clear on which measure of central tendency to report. The median is often used because it is intermediate to the mode, which can be a poor choice when it is distant to the middle of the distribution, and the mean, which can be give substantial weight to extreme values (Carlin and Louis 2009). Many of our estimates include the combination of two distributions (e.g. the number of fish by age which include the multinomial distribution for age and various distributions for abundance). Because these two distributions are often asymmetrical for fish monitoring data when we sum the medians of abundance by age they may not equal the median abundance estimate. Therefore, to limit confusion we have decided that the reported estimate will be the mean, which has a property that the individual estimates sum to the total estimate. The summary table will also include the median, the standard deviation based on the posterior distribution. We reported the equal-tailed or symmetrical 95% confidence intervals which exclude 2.5% from each tail of the posterior distribution rather than the highest probability interval, which is the shortest 95% interval of the posterior mass and is sometimes preferred (Lee 2004).

Tag Loss

There was no need to assess tag loss in our closed population experiments because we used a permanent opercle punch that was rotated weekly as secondary mark. We conducted double tagging/marking experiments to address tag loss in open population studies, except in Cedar Creek, where salmon were only single tagged. In this double tagging experiment we assumed tag loss was the same for each of the tags because the tag type and location were the same, and tag loss was independent. The summary statistics include the tag recoveries with one and two tags, and the number of tag releases (Table 4). The fundamental parameter is probability of losing a tag and the derived parameters include the probability of recovering a fish with one or two tags, the probability of losing two tags, the probability of retaining at least one tag, and the number of tags adjusted for tag loss (Table 5). The likelihoods for single and double tag recoveries are found in Table 6.

Table 4. Summary statistics used in a double tagging experiment to estimate tag loss.

Statistic	Definition/Equation
t_1	Number of fish recovered with one tag
t_2	Number of fish recovered with two tags
t_all	Number of fish with one or two tags, *t_all* = *t_1* + *t_2*
Tags	Number of tags release

Table 5. Fundamental and derived parameters in a double tagging experiment to estimate tag loss assuming the probability of losing a tag was the same for each tag, and loss of each tag was independent.

Parameter	Definition/Equation
p_tl	Probability of losing a tag
p_1	Probability of recovering a fish with 1 tag, *p_1* = ((2**p_tl*)*(1-*p_tl*))/(1-*p_tl***p_tl*)
p_2	Probability of recovering a fish with 2 tags, *p_2* = ((1-*p_tl*)*(1-*p_tl*))/(1-*p_tl***p_tl*)
p_0	Probability of losing two tags, *p_0* = *p_tl* * *p_tl*
q_0	Probability of retaining at least 1 tag, *q_0* = 1 – *p_tl***p_tl*
T_adj	Number of tags released adjusted for tag loss, *T_adj* = *Tags* * *q_0*

Table 6. The likelihoods for the independent tag loss model when using the same tag type.

Description	Likelihood
Pr(capture of fish with 1 tag) Pr(capture of fish with 2 tags)	*t_1* ~ Binomial(*p_1*,*t_all*) *t_2* ~ Binomial(*p_2*,*t_all*)

Weir Abundance Estimates

A census count of Chinook salmon occurred in the Cowlitz River at the Barrier Dam. Based on a Cowlitz management plan, Chinook salmon are trucked for release into the upper Cowlitz and Tilton rivers. At the Green and Elochoman weirs, a portion of the trapped fish was released above the weir, depending on WDFW management objectives for that basin (Table 7). In some cases, not all fish released above a weir successfully spawn. To estimate the proportion of successful spawners (*pSuc*) female carcasses are inspected for spawning success. In some cases a fishery may occur above the weir, which is estimated through a statistical expansion of catch record card (CRC) returns (Kraig 2012) (Table 8). The number of spawners above the weir (*WeirSpawners*) is the weir count (*count*) times the proportion of successful spawners minus the estimated harvest.

Table 7. Summary statistics used to estimate spawners above weirs.

Statistic	Definition/Equation
count	Number of fish released and passed above the weir
Fcarc	Number of females examined above the weir for spawning success
Fsuc	Number of females examined that had spawned (i.e., egg retention < 100%)

Table 8. Likelihoods and derived parameters to estimate spawner abundance above weirs.

Description	Likelihood/Derived Estimates
Mu	*mu* is the mean catch from CRC harvest estimates
Prec	*prec* = 1/variance from the CRC harvest estimates
Pr(catch)	*catch*~ Normal(*mu,prec*) estimated from CRC returns
Pr(spawn success)	*Fsuc*~ Binomial(*pSuc*,*Fcarc*)
WeirSpawners	*WeirSpawners* is the number of fish above the weir that attempted to spawn, *WeirSpawners = count*pSuc-catch*

Closed Population Abundance Estimates

Our study design was developed based on stratified Petersen or Darroch closed population mark-recapture models because they are relatively robust to heterogeneity in capture and movement probabilities (Seber 1982). Using this study design we recorded the number of fish marked and released above the trap, and the number of captured and recaptured fish (carcasses) per week during spawning ground surveys (Table 9).

Table 9. Summary statistics used to estimate abundance using the Darroch (1961) model.

Statistic	Definition/Equation
d_m_i	Number of fish marked and released at sample time *i*.
d_r_{ij}	Number of marked fish recaptured at sample time *ij*, *i* = 1,…, *s*, *j* = 1,…, *s*.
$d\_m_i - d\_r_i$	Number of marked fish not recaptured
d_u_j	Number of fish captured are sample time *j* that were not previously marked.

The fundamental parameters including the probability of capture, probability of movement between strata, probability a fish is caught in a stratum, and the population estimate at the time of tagging (Table 10), and are estimates based on the likelihoods in Table 11.

Table 10. Fundamental and derived parameters for the Darroch (1961) model.

Parameter	Definition/Equation
d_s	Number of sample times
d_p_j	Probability of capture at sample time j, $j = 1,\dots, s$.
d_θ_{ij}	Probability that a fish from m_i moves to stratum j, $i = 1,\dots, s$, $j = 1,\dots, s$. Since the population is closed & no mortality the $\sum \theta_{ij} = 1$, $i = 1,\dots, s$.
d_ψ_{ij}	Probability that a fish from m_i is caught at time j, $i = 1,\dots, s$, $j = 1,\dots, s$. $\psi_{ij} = \theta_{ij}\, p_{j.}$ Probability of not being captured, $\psi_{ij} = (1-\sum\psi_{ij})$, $i = 1,\dots, s$, $j = s + 1$.
d_U_j	Number of fish at sample time j. $N = \sum U_j$, which is the population estimate

Table 11. The likelihoods for the Darroch (1961) model.

Description	Likelihood/Derived Estimates
Pr(capture)	$d\_u_j \sim$ Binomial(d_p_j,d_U_j), $j = 1,\dots, s$.
Pr(capture at time$_j$)	$d\_r_{ij} \sim$ Binomial(d_ψ_{ij}, d_m_i), $i = 1,\dots, s$, $j = 1,\dots, s$.

When there is an equal probability of capture during the tagging event, an equal probability of capture during the second tagging event, or there is complete mixing of tagged and untagged fish between events, all releases, recoveries and captures may be combined into a "pooled" Petersen estimator (Schwarz and Taylor 1998). The summary statistics include the number of marks, recaptures, and captures (Table 12). The fundamental parameter is the population size estimated from the summary statistics and hypergeometric distribution (Table 13). The hypergeometric distribution is appropriate to use when there is sampling without replacement as salmon carcasses captured in the second event were mutilated and not available for recapture in future sampling events. When there is sampling with replacement the binomial model is appropriate (Table 14 and 15) and this model was used for the trans-generational genetic mark-recapture estimates (Rawding et al. 2013).

Table 12. Summary statistics used in the hypergeometric Petersen model where Chinook salmon were live tagged and recovered as carcasses.

Statistic	Definition/Equation
m_h	Number of fish marked in the first sample (*n1*) for the hypergeometric model
r_h	Number of marked fish recaptured in the 2nd sample (*m2*) for the hypergeometric model
c_h	Number of fish captured in the second sample (*n2*) for the hypergeometric model

Table 13. The fundamental parameters and likelihoods for the hypergeometric Petersen model.

Description	Definition/Likelihood
Nh	The population size *Nh*
Pr(Recapture)	r_h ~Hypergeometric(m_h,c_h,N_h)

Table 14. Summary statistics used in the trans-generational genetic mark-recapture binomial Petersen model where genotyped Chinook salmon carcasses were the marks, genotyped juveniles were the captures, and juvenile genotypes assigned back to the marks were the recaptures.

Statistic	Definition/Equation
m_b	Number of fish marked in the first sample (*n1*) for the Binomial model
r_b	Number of marked fish recaptured in the second sample (*m2*) for the Binomial model
c_b	Number of fish captured in the second sample (*n2*) for the Binomial model

Table 15. The fundamental parameters and likelihoods for the tGMR binomial Petersen model.

Description	Likelihood
Pr(Proportion Marked)	$r_b \sim \text{Binomial}(p,c_b)$
Pr(Recapture)	$m_b \sim \text{Binomial}(p,N_b)$

Open Population Abundance Estimates

We parameterized the Schwarz et al. (1993) "super population" JS model into a Bayesian framework. Rather than using individual capture histories we used summary statistics to increase the computational speed (Table 16). It is important to note that in more popular Schwarz and Arnason (1996) model the super population and other fundamental parameters are based on births while in the Schwarz et al. (1993) model the super population is the total of gross births or salmon escapement (Table 17). This model allows salmon escapements to be hierarchically modeled (Rivot and Prevost 2002) and probability of entry to be modeled based on various distributions (Hilborn et al. 1999). Derived parameter estimates in Table 18 are based on the Schwarz et al. (1993) and Manske and Schwarz (2000). We included the later author's derived estimates for cases when a the mark-recapture study ends early, as they proposed a method to estimate escapement based on the residence time estimated from the mark-recapture data and AUC method, which is a plot of the population size at each sampling period. The JS likelihood is the product of 3 likelihoods: 1) the probability of first capture based on a super population (N) that enter the population (b^*_i) following a multinomial distribution, 2) the probability of release on capture (v_i) from a binomial distribution using total fish sampled (n_i) and number of n_i that are released (R_i), and 3) the probability of recapture which is the product of two binomial distribution to estimate the probability of capture (p_i) and survival (φ_i) (Burnham 1991) (Table 19).

Table 16. Summary statistics used in the Jolly-Seber model.

Statistic	Definition/Equation
m_i	Number of fish captured at sample time *i* that were previously marked.
u_i	Number of fish captured at sample time *i* that were unmarked.
n_i	Number of fish captured at sample time *i*. $n_i = m_i + u_i$.
l_i	Number of fish lost on capture at time *i*.
R_i	Number of fish that were released after the *i*th sample. R_i need not equal n_i if there were losses on capture or injections of new fish at sample time *i*.
r_i	Number of R_i fish released at sample time *i* that were recaptured at one or more future sample times.
z_i	Number of fish captured before time *i*, not captured at time *i*, and captured after time *i*.
T_i	Number of fish captured at before time *i* and captured at or after time *i*. $T_i = m_i + z_i$.

Table 17. Fundamental parameters for the Jolly-Seber model under the salmon escapement super population model (Schwarz et al. 1993).

Parameter	Definition/Equation
s,tm	Number of sample times and length of interval between samples
p_i	Probability of capture at sample time *i*, $i = 1,\ldots, s$.
φ_i	Probability of a fish surviving and remaining in the population between sample time *i* and sample time $i + 1$, given it was alive and in the population at sample time *i*, $i = 1,\ldots, s$-1.
b^*_i	Probability that a fish enters the population between sample times *i* and *i* +1, $i = 0,\ldots, s$-1 under the constrain that $\sum b^*_i = 1$. These are referred to as entry probabilities.
v_i	Probability that a fish captured at time *i* will be released, $i = 1,\ldots, s$-1.
N	Total number of fish that enter the system before the last sample time or the escapement. This is referred to as the super population.

Table 18. Derived parameters for the Jolly-Seber model under the salmon escapement super population model (Schwarz et al. 1993) and the stream residence time model (Manske and Schwarz 2000).

Parameter	Definition/Equation
λ_i	Probability that a fish is seen again after sample time *i*, $i = 1,\ldots, s$. $\lambda_i = \varphi_i\, p_{i+1} + \varphi_i\,(1 - p_{i+1})\,\lambda_{i+1}$, $i = 1,\ldots, s$-1; $\lambda_s = 0$.
τ_i	Conditional probability that a fish is seen at sample time *i* given that it was seen at or after sample time *i*, $i = 1,\ldots, s$. $\tau_i = p_i/(\,p_i + (1\text{-}p_{i+1})\,\lambda_i)$.
ψ_i	Probability that a fish enters the population between sample time *i*-1 and *i* and survives to the next sampling occasion. $\psi_i = b^*_0$, $\psi_{i+1} = \psi_i\,(1 - p_i)\varphi_i + b^*_i(\varphi_i - 1)/\log(\varphi_i)$
B_i	Number of fish that enter after sample time *i* and survive to sample time *i* +1, $i = 0,\ldots, s\text{-}1$. These are referred to as net births. $B_0 = B^*_0$, $B_i = B^*_i(\varphi_i - 1)/\log(\varphi_i)$.
B^*_i	Number of fish that enter between sampling occasion *i*-1 and *i*, $i = 0,\ldots, s$-1. These are referred to as gross births. $B^*_i = N\,(b^*_i)$
N_i	Population size at time *i*, $i = 1,\ldots, s$. $N_1 = B_0$, $N_{i+1} = (N_i - n_i + R_i)\,\varphi_i + B_i$
N^-_i	Number of fish alive immediately before sample time i, $i = 1,\ldots, s$. $N^-_1 = B_0$; $N^-_{i+1} = N^+_i\,\varphi_i + B_i$
N^+_i	Number of fish alive immediately after sampling time i, $i = 1,\ldots, s$. $N^+_i = (N^-_i - n_i + R_i)$. N^+_i may differ from N^-_i if there were losses on capture or injections of new fish.
RT	Average residence time; for $i = 1,\ldots, s$-1. $\mathrm{RT} = 0.5\sum tm_i N^+_i(\varphi_i + 1) + 0.5\, tm_s N^+_s + 0.5 tm_0 B_0 + \sum B_i tm_i(\varphi_{i/}\,\varphi_{i-1} - 1/\log(\varphi_i))$
AUC	Aggregate residence time over all spawner. This is referred to as the total fish days or Area-Under-the-Curve. $\mathrm{AUC} = 0.5\, tm_0 N^-_1 + \sum 0.5\, tm_i(N^+_i + N^-_i) + 0.5\, tm_s N^+_s$.
ESC	Escapement. ESC = AUC/RT. This is slightly greater than N, which is also a measure of escapement due to accounting for fish before and after sampling.

Table 19. The likelihoods for the Schwarz et al. (1993) model

Description	Likelihood
Pr(first capture part a)	$u. \sim$ Binomial($\sum\psi_i p_i$,N), i = 0,…, s-1. $u. = \sum u_i$
Pr(first capture part b)	$u_i \sim$ Multinomial ($\psi_i p_i / \sum\psi_i p_i$,$u.$)), i = 0,…, s-1.
Pr(release on capture)	$R_i \sim$ Binomial(v_i,n_i), i = 1,…, s-1.
Pr(recapture part a)	$m_i \sim$ Binomial(τ_i,T_i), i = 2,…, s-1.
Pr(recapture part b)	$r_i \sim$ Binomial(λ_i,R_i), i = 1,…, s-1.

Spawning Ground Survey Abundance Estimates

Three types of abundance estimates may be obtained from weir or mark-recapture estimates when counts of redds, fish, or at the peak are collected concurrently (Table 20). Using the summary statistics we estimate redds per female, survey life, and peak count expansion factor (Table 21 and 22). Spawner abundance based on redd counts, periodic fish counts (AUC), and peak counts are estimates (Gallagher et al. 2007, Parken et al. 2003)

Table 20. Summary statistics used from spawning ground surveys.

Statistic	Definition
Redd_tot	Total number of new redds observed during the spawning period
$Spawners_i$	Number of fish classified as spawners on day i
PC	The greatest number of live fish and/or carcasses observed on a single day during the spawning period

Table 21. Derived parameters for spawning ground abundance methods.

Parameter	Definition/Equation
F	Number of females in the population, $F = pF * N$ or Females from tGMR
RpF	Number of redds per female, $RpF = Redd\_tot / F$
AUCsp	The total number of fish days for spawners or Area-Under-the-Curve. $AUCsp = 0.5\ t_0 Spawner_1 + \sum 0.5\ t_i(Spawner_i + Spawner_{i+1}) + 0.5\ t_s Spawner_{s+1}$. For days i = 1,…, s+1.
SL	The apparent survey life, which is the average number of days a fish remains in the survey area; survey life equals residence time when the survey area is the entire spawning distribution, $SL = AUCsp / N$
PCEF	Peak count expansion factor , PCEF = N/PC

Table 22. Derived parameters for spawning ground abundance methods.

Parameter	Definition/Equation
Nredds	Redd-based spawner abundance, $Nredds = Redd\_tot / RpF / pF$
Nauc	AUC-based spawner abundance estimate, $Nauc = AUCsp / SL$
Npc	Peak count-based spawner abundance estimate, $Npc = PC * PCEF$

Proportions

Important indicators for salmon populations included the number of females and marked (hatchery origin) fish (Rawding and Rodgers 2013). In addition, ages are a measure of diversity and are needed to reconstruct salmon runs for forecasting and spawner-recruit analysis (Rawding and Rodgers 2013, Hilborn and Walters 1992). When the data allow for only two possibilities, such as the sex being male or female, the binomial distribution is an appropriate model for analysis but when there are more than two possibilities such as adult ages, the multinomial model is appropriate. The summary statistics and likelihoods for the proportions are found in Tables 23 and 24. The total number of marked and unmarked adults, adult males and females, and subtotals of marked and unmarked fish by age were estimated by multiplying these proportions by the total escapement estimates.

Table 23. Summary statistics from spawning ground surveys to estimate proportions.

Statistic	Definition/Equation
Females	Number of adults that were females
Males	Number of adults that were females
Adults	Number of adults examined for sex and origin
Marked	Number of adults that were mass marked (adipose fin clipped)
Unmarked	Number of adults that were not mass marked (adipose fin intact)
M_Age_i	Number of marked adults that are age *i*, *i*=3,4,5
U_Age_i	Number of unmarked adults that are age *i*, *i*=3,4,5
pF	Proportion of adults that are females
PM	Proportion of adults that are males
pMS	Proportion of adults that are mass marked
pUS	Proportion of adults that are not mass marked
pM_Age_i	Proportion of adults that are marked adults that are age *i*, *i*=3,4,5
pU_Age_i	Proportion of adults that are un marked adults that are age *i*, *i*=3,4,5

Table 24. The likelihoods for sex, origin, and age proportions.

Description	Likelihood
Pr(Females)	*Females* ~Binomial(*pF*,*Adults*)
Pr(Males)	*Males* ~Binomial(*pM*,*Adults*)
Pr(Marked)	*Marked* ~Binomial(*pMS*,*Adults*)
Pr(Unmarked)	*Unmarked* ~Binomial(*pUS*,*Adults*)
$Pr(M\_age_i)$	M_age_i ~Multinomial(pM_Age_i,*Adults*)
$Pr(U\_age_i)$	U_age_i ~Multinomial(pU_Age_i,*Adults*)

ESU abundance

The ESU estimates by reporting group are the sum of the population estimates.

Cross Validation of AUC and Redd Estimates

To test the robustness of our redd- and AUC-based estimates we used a leave-one-out cross validation approach. In places where we had concurrent mark-recapture and AUC or Redd based estimates, we compared the results of the AUC or redd estimate to the mark-recapture estimate. In this case we used the mean SL and FpR to adjust develop abundance estimates from AUC and redds, respectively. These mean SL and FpR estimates did not include population specific estimate of SL or FpR (Parken et al. 2003). For comparison where we tested the hypothesis that the redd or AUC estimate was greater than the mark-recapture estimate by monitoring this in node in WinBUGS.

Timing

We used period (weekly) counts of spawners and divide these counts by the total count of spawners to estimate the cumulative timing of spawning for each Tule Chinook salmon population.

Results

Model Convergence and Diagnostics

We ran two chains using the Gibbs sampler in WinBUGS saving a total of 10,000 iterations for the posterior distribution of each parameter. Visual inspection of the trace and history plots suggested the chains mixed and converged. The Brooks-Gelman-Rubin (BGR) diagnostic test for convergence yielded values of less than 1.03 for each parameter, which is less than the recommended value of 1.2. While it is impossible to conclusively demonstrate a simulation has converged, the above diagnostic tests did not detect that the simulations did not converge. The MCMC error rate was less than 5% of the standard deviation of the parameter estimates, which suggests our posterior distributions were accurate. Our population abundance estimates were similar for the different vague priors and the proportion results were not sensitive to the priors except when we had few observations. The results reported here used the based LaPlace/Bayes prior.

Grays Population

The Grays Chinook salmon consists of three stock components; a marked Tule, a marked Rogue Bright population, and an unmarked (wild) population likely comprised of Tules, naturalized Rogue River Brights, and their hybrids. The escapement estimates are based on redd surveys and include the historic distribution below the Grays River canyon, and spawning in areas altered to allow passage above the canyon. A total of 153 adults (95%CI 114-233) were estimated. Abundance, origin, sex, and age estimates for the Tule and unmarked component are found in Table 25.

Table 25. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Grays Tule Chinook salmon population.

Parameter	mean	sd	2.50%	median	97.50%
Escapement	89	23.41	51	86	142
Marked Esc.	19	10.90	4	17	46
Unmarked Esc.	70	20.90	36	68	118
Female Esc.	54	16.96	27	52	94
Male Esc.	35	12.86	15	33	65
Marked Esc. Age3	11	7.81	2	10	31
Marked Esc. Age4	4	4.19	0	2	15
Marked Esc. Age5	4	4.21	0	2	16
Unmarked Esc. Age3	49	17.22	22	46	88
Unmarked Esc. Age4	16	9.37	3	14	39
Unmarked Esc. Age5	5	5.44	0	4	20
Prop. of Marked Esc.	0.212	0.106	0.049	0.199	0.454
Prop. of Unmarked Esc.	0.788	0.106	0.546	0.802	0.951
Prop. of Female Esc.	0.609	0.099	0.411	0.612	0.793
Prop. of Male Esc.	0.391	0.099	0.207	0.388	0.589
Prop. of Marked Esc. Age3	0.601	0.200	0.191	0.617	0.930
Prop. of Marked Esc. Age4	0.198	0.163	0.006	0.156	0.604

Table 25. (continued)

Parameter	mean	sd	2.50%	median	97.50%
Prop. of Marked Esc. Age5	0.201	0.163	0.007	0.161	0.596
Prop. of Unmarked Esc. Age3	0.693	0.122	0.433	0.703	0.898
Prop. of Unmarked Esc. Age4	0.230	0.112	0.056	0.217	0.484
Prop. of Unmarked Esc. Age5	0.077	0.072	0.002	0.056	0.265

A total of 67 adults (95%CI 36-118) in the Grays River were stray hatchery origin Rogue River Brights (Table 26). Of all carcass recoveries in the basin these Brights represent 44%. Age 3 comprised 70% of the Brights recovered. Since few carcasses were recovered after mid-October, it is likely that the bright component was underestimated because they comprise a greater proportion of late-timed Chinook in the Grays.

Table 26. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Grays River "Rogue Bright" Chinook salmon population.

Parameter	mean	sd	2.50%	median	97.50%
Escapement	67	20	35	65	115
Prop. of Marked Esc.	0.424	0.095	0.243	0.423	0.611
Prop. of Marked Esc. Age3	0.692	0.123	0.432	0.701	0.901
Prop. of Marked Esc. Age4	0.231	0.113	0.056	0.215	0.488
Prop. of Marked Esc. Age5	0.077	0.070	0.002	0.056	0.258

Elochoman/Skamokawa Population

The Elochoman/Skamokawa or Elochoman population is composed of the Skamokawa and Elochoman subpopulations. Historically, WDFW monitored these subpopulations separately so we continue to report on these separately but we will also report a combined estimate.

For the Skamokawa subpopulation the logistic regression favored the null model (constant carcass recoveries by length and sex) but there was similar support for the length model (BF=2.04) (Table 27). Since there was similar support for both models we chose the null model and did not stratify estimates by length. The abundance estimate obtained by dividing AUC by mean SL was 530 adults (95%CI 489-567). Based on adipose fin clips, the proportion of marked fish was 93.44% and 46.7% of the adults were females. Most adults were age three with very few age five fish. Subpopulation abundance, origin, sex, and age estimates are found in Table 28. We considered reporting the estimate from the JS model (311 adults, 95%CI 224-513), which had 95%CI that overlapped the point estimate from the AUC model. However, we chose to report the AUC estimate due to the few recaptures in most periods and poor precision of the JS estimate (CV=23%).

Table 27. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 adult (>59cm) Skamokawa Chinook salmon.

Model	Constant	Sex	Length	Sex + Length
Posterior Model Probabilities	0.53	0.12	0.26	0.08
Bayes Factor	1.00	4.29	2.04	6.55

Table 28. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Skamokawa Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	529	20.16	489	530	567
Marked Esc.	495	23.30	447	495	539
Unmarked Esc.	35	13.85	13	33	67
Female Esc.	247	29.23	191	247	305
Male Esc.	282	29.49	225	282	341
Marked Esc. Age3	400	29.60	340	400	456
Marked Esc. Age4	74	20.86	38	73	119
Marked Esc. Age5	20	11.54	5	18	49
Unmarked Esc. Age3	21	11.28	5	19	48
Unmarked Esc. Age4	7	6.70	0	5	25
Unmarked Esc. Age5	7	6.64	0	5	24
Prop. of Marked Esc.	0.934	0.026	0.874	0.937	0.976
Prop. of Unmarked Esc.	0.066	0.026	0.024	0.063	0.126
Prop. of Female Esc.	0.467	0.052	0.365	0.467	0.570
Prop. of Male Esc.	0.533	0.052	0.430	0.533	0.636
Prop. of Marked Esc. Age3	0.808	0.046	0.709	0.811	0.890
Prop. of Marked Esc. Age4	0.150	0.042	0.078	0.147	0.240
Prop. of Marked Esc. Age5	0.041	0.023	0.009	0.037	0.098
Prop. of Unmarked Esc. Age3	0.599	0.201	0.193	0.612	0.933
Prop. of Unmarked Esc. Age4	0.201	0.164	0.006	0.163	0.609
Prop. of Unmarked Esc. Age5	0.200	0.164	0.006	0.157	0.601

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

For the Elochoman subpopulation, the logistic regression model to test for bias in carcass recoveries favored the null model over the length model, which was the second best model (BF=12.7), which indicated that the carcass recovery rates were not sex- or length-dependent (Table 29). Therefore all adults were analyzed as a single group (Table 30). The weir operated from the start of the run through October 9, when it was removed to prevent damage during a high water event. Since the run was just about complete the weir was not was re-installed after this date. A total of 1,018 adults were trapped and release above the weir of which 1,013were tagged. The BF, computed from the marginal likelihood, for the null hypothesis that a constant proportion of recovered fish were tagged at the weir and during carcass surveys rates was 3.4, which provided support for pooling weir and spawning survey carcass recoveries for abundance estimation. The BF for the null hypothesis of constant proportion of recovery marked by time period was 3,469, which is very strong support for pooling recoveries by time period. These tests indicate the pooled Petersen estimator would provide a consistent estimate of adult abundance above the weir, and this estimate was 1,047 (95% CI 1030 - 1075) with a CV of 1%. This indicated that approximately 2% of the run bypassed the weir. For a comparison the Darroch model provide an estimate of 1,064 (95%CI 841 – 1425) with a CV of 20%. The GOF test for the recapture portion of the Darroch model yielded a Bayesian *P*-value of 0.42 indicating the Darroch model fit the data. To estimate the number of spawners we multiplied the run size estimate by the proportion of successful spawners and subtracted the estimated sport harvest, which leaves an estimate of 797 adults (95%CI 593 to 932). The proportion of marked fish was 85% and most marked and unmarked fish were age 3. Subpopulation abundance, origin, sex, and age estimates are found in Table 30.

Table 29. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Elochoman Chinook salmon >59cm.

Model	Constant	Sex	Length	Sex + Length
Posterior Model Probabilities	0.86	0.06	0.07	0.01
Bayes Factor	1.00	13.44	12.71	130.53

Table 30. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Elochoman Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	788	88.36	593	797	932
Marked Esc.	669	75.61	501	677	793
Unmarked Esc.	119	16.00	87	120	149
Female Esc.	352	41.35	263	356	423
Male Esc.	436	50.44	326	440	522
Marked Esc. Age3	524	66.30	387	529	644
Marked Esc. Age4	129	32.40	72	126	199
Marked Esc. Age5	16	11.31	2	13	45
Unmarked Esc. Age3	77	17.06	46	77	111
Unmarked Esc. Age4	28	12.45	8	27	57
Unmarked Esc. Age5	14	9.15	2	12	37
Prop. of Marked Esc.	0.849	0.011	0.826	0.849	0.870
Prop. of Unmarked Esc.	0.151	0.011	0.130	0.151	0.174
Prop. of Female Esc.	0.447	0.016	0.417	0.447	0.478
Prop. of Male Esc.	0.553	0.016	0.522	0.553	0.583
Prop. of Marked Esc. Age3	0.784	0.045	0.687	0.786	0.864
Prop. of Marked Esc. Age4	0.192	0.043	0.116	0.190	0.283
Prop. of Marked Esc. Age5	0.024	0.017	0.003	0.020	0.065
Prop. of Unmarked Esc. Age3	0.648	0.112	0.418	0.654	0.850
Prop. of Unmarked Esc. Age4	0.235	0.099	0.074	0.225	0.454
Prop. of Unmarked Esc. Age5	0.117	0.075	0.016	0.103	0.301

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Both the Skamokawa and Elochoman estimates were summed to obtain an estimate for the Elochoman (Elochoman/Skamokawa) population. The total population estimate was 1,329 (95%CI 967-1483). As with the two subpopulations, the marked fraction was 88% and most fish were age 3. Population abundance, origin, sex, and age estimates are found in Table 31.

Table 31. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Elochoman/Skamokawa Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	1317	102.20	1092	1329	1483
Marked Esc.	1163	89.53	967	1172	1312
Unmarked Esc.	154	21.79	113	153	199
Female Esc.	599	55.42	483	603	699
Male Esc.	718	63.59	582	722	830
Marked Esc. Age3	924	80.09	755	929	1068
Marked Esc. Age4	203	38.96	132	201	285
Marked Esc. Age5	37	16.30	12	34	75
Unmarked Esc. Age3	98	20.65	60	97	141
Unmarked Esc. Age4	35	14.15	12	33	68
Unmarked Esc. Age5	21	11.46	4	19	49
Prop. of Marked Esc.	0.883	0.013	0.855	0.884	0.906
Prop. of Unmarked Esc.	0.117	0.013	0.094	0.116	0.145
Prop. of Female Esc.	0.455	0.023	0.411	0.455	0.501
Prop. of Male Esc.	0.545	0.023	0.499	0.545	0.590
Prop. of Marked Esc. Age3	0.794	0.032	0.727	0.796	0.854
Prop. of Marked Esc. Age4	0.174	0.030	0.119	0.173	0.238
Prop. of Marked Esc. Age5	0.031	0.014	0.011	0.029	0.063
Prop. of Unmarked Esc. Age3	0.637	0.099	0.437	0.641	0.819
Prop. of Unmarked Esc. Age4	0.228	0.086	0.085	0.220	0.416
Prop. of Unmarked Esc. Age5	0.135	0.070	0.030	0.125	0.301

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Mill/Abernathy/Germany Population

The population is composed of three subpopulations that enter the Columbia within two miles of each other. We continue to report on these subpopulations separately due to the historic reporting structure and because they are part of Washington State's Intensively Monitored Watershed (IMW) program, which requires reporting at this scale. For the Mill Creek population the test for length and sex selectivity in carcass recoveries for all adults provided positive support for the length-dependent recovery model over the null model (BF=3.5) (Table 32). Therefore, we stratified the carcass recovery at 80cm and re-tested the same hypothesis. For the large adults (≥80 cm), there was positive support for the null compared to the sex model (BF=8.48). For small adults (<80 cm) there was similar support for null, length (BF=1.03), and sex + length (BF=1.84) models. Based on these results we estimated abundance for the large and small adults separately.

Table 32. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Mill Chinook salmon >59cm.

Population	Model	Constant	Sex	Length	Sex + Length
Mill	Posterior Model Probabilities	0.19	0.03	0.66	0.12
Mill	Bayes Factor	3.47	19.37	1.00	5.48
Mill ≥80cm	Posterior Model Probabilities	0.81	0.10	0.08	0.01
Mill ≥80cm	Bayes Factor	1.00	8.48	9.83	63.23
Mill <80cm	Posterior Model Probabilities	0.35	0.11	0.34	0.19
Mill <80cm	Bayes Factor	1.00	3.26	1.03	1.84

We used DIC for to select the best model for large and small adults (Tables 33 and 34). In these tables we report on the model deviance, pv-the effective number of parameters in the model, the DIC value (lower DIC indicates the preferred model), change in DIC and DIC weights for model comparison, abundance estimates from the model, and CJS-pvalue-which is a Bayesian *P*-value for our GOF test. DIC favored the stt model for both groups but there was similar support for the ttt model. Bayesian *P*-values of 0.37 and 0.61 do not indicate lack of fit. Abundance estimates were relatively similar across all models thus not very sensitive to model choice. The abundance estimate for the Mill Creek subpopulation was 1,042 adults (95%CI 957-1153). The proportion of marked adults was 95%. The proportions of age 3 marked and unmarked fish were 88% and 53%, respectively. Subpopulation abundance, origin, sex, and age estimates are found in Table 35.

Table 33. JS model selection for large Chinook salmon adults (≥80cm) in Mill Creek, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	142.3	24.9	167.2	1.1	0.366	819	0.50
stt	144.0	22.1	166.1	0.0	0.634	795	0.37
tst	170.4	20.4	190.8	24.7	0.000	801	0.33
sst	183.0	16.9	199.9	33.8	0.000	770	0.00

Table 34. JS model selection for small Chinook salmon adults (<80cm) in Mill Creek, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC_weights	Abundance	CJS_pvalue
ttt	84.4	17.3	101.7	2.8	0.178	249	0.60
stt	83.2	15.7	98.9	0.0	0.724	245	0.61
tst	89.2	15.3	104.5	5.6	0.044	254	0.42
sst	91.6	12.5	104.1	5.2	0.054	246	0.02

Table 35. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Mill Creek Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	1046	49.77	957	1042	1153
Marked Esc.	993	48.18	908	990	1097
Unmarked Esc.	52	8.82	37	52	71
Female Esc.	420	28.00	368	419	478
Male Esc.	626	35.25	561	625	699
Marked Esc. Age3	873	44.40	793	870	968
Marked Esc. Age4	81	11.83	59	80	106
Marked Esc. Age5	40	8.23	25	39	57
Unmarked Esc. Age3	27	6.29	17	27	41
Unmarked Esc. Age4	13	4.28	6	13	23
Unmarked Esc. Age5	12	4.04	5	11	21
Prop. of Marked Esc.	0.950	0.008	0.933	0.950	0.964
Prop. of Unmarked Esc.	0.050	0.008	0.036	0.050	0.068
Prop. of Female Esc.	0.402	0.018	0.366	0.402	0.438
Prop. of Male Esc.	0.598	0.018	0.562	0.598	0.634
Prop. of Marked Esc. Age3	0.879	0.013	0.851	0.880	0.904
Prop. of Marked Esc. Age4	0.081	0.011	0.060	0.081	0.105
Prop. of Marked Esc. Age5	0.040	0.008	0.026	0.039	0.057
Prop. of Unmarked Esc. Age3	0.522	0.079	0.365	0.523	0.676
Prop. of Unmarked Esc. Age4	0.251	0.069	0.130	0.247	0.397
Prop. of Unmarked Esc. Age5	0.227	0.066	0.113	0.222	0.371

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The Germany Creek subpopulation was estimated by applying the JS model to the carcass tagging data. For this subpopulation, the test for sex and length selectivity provided positive support for the length-dependent model compared to the constant model (BF=12.3). After stratifying the Germany subpopulation into large and small adults BF provided positive support for the constant model over the sex model (BF=5.0) for the large adults and the BF provided similar support for the null and length model (BF=2.4) for the small adults (Table 36). Since there was similar support for both models we chose to the null model and did not stratify age estimates by length for small adults.

Table 36. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Germany Creek Chinook salmon >59cm.

Population	Model	Constant	Sex	Length	Sex + Length
Germany	Posterior Model Probabilities	0.06	0.03	0.69	0.22
Germany	Bayes Factor	12.27	27.18	1.00	3.08
Germ ≥80cm	Posterior Model Probabilities	0.74	0.15	0.09	0.02
Germ ≥80cm	Bayes Factor	1.00	4.97	8.24	30.62
Germ <80cm	Posterior Model Probabilities	0.57	0.11	0.24	0.08
Germ <80cm	Bayes Factor	1.00	5.04	2.37	7.49

Model selection was based on DIC. For the small adults DIC favored the ttt model with some support for the stt model (Table 37). This was reversed for the large adults the stt model found more support than the ttt model with little support for the remaining models (Table 38). GOF tests based on Bayesian *P*-values were 0.64 and 0.36 for small and large fish, respectively, indicating the models fit the data. The abundance estimate for the Germany Creek population was 1182 (95%CI 1053-1398). As with the Mill Creek population over 90% of the sampled fish were marked. Over 80% of the marked and unmarked adults were age 3. Subpopulation abundance, origin, sex, and age estimates are found in Table 39.

Table 37. JS model selection for small Chinook salmon adults (<80cm) in Germany Creek, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	92.5	17.5	110.0	0.0	0.870	346	0.64
stt	96.8	17.0	113.8	3.8	0.130	327	0.51
tst	141.6	14.4	156.0	46.0	0.000	348	0.00
sst	153.3	12.4	165.7	55.7	0.000	304	0.00

Table 38. JS model selection for large Chinook salmon adults (≥80cm) in Germany Creek, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	148.0	26.9	174.9	2.0	0.269	769	0.42
stt	151.5	21.4	172.9	0.0	0.731	722	0.26
tst	213.6	19.8	233.4	60.5	0.000	719	0.00
sst	237.1	16.1	253.2	80.3	0.000	680	0.00

Table 39. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Germany Creek Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	1193	88.63	1053	1182	1398
Marked Esc.	1091	82.70	958	1081	1283
Unmarked Esc.	102	20.44	68	100	147
Female Esc.	661	61.52	558	654	802
Male Esc.	532	48.18	448	528	638
Marked Esc. Age3	924	77.28	800	915	1103
Marked Esc. Age4	157	20.17	121	155	200
Marked Esc. Age5	10	5.02	4	8	23
Unmarked Esc. Age3	82	18.55	51	80	124
Unmarked Esc. Age4	13	6.55	4	11	29
Unmarked Esc. Age5	7	4.57	2	6	19
Prop. of Marked Esc.	0.914	0.016	0.880	0.915	0.942
Prop. of Unmarked Esc.	0.086	0.016	0.058	0.085	0.120
Prop. of Female Esc.	0.554	0.027	0.500	0.553	0.606
Prop. of Male Esc.	0.447	0.027	0.394	0.447	0.500
Prop. of Marked Esc. Age3	0.847	0.018	0.809	0.848	0.880
Prop. of Marked Esc. Age4	0.144	0.018	0.113	0.143	0.181
Prop. of Marked Esc. Age5	0.009	0.005	0.003	0.008	0.021
Prop. of Unmarked Esc. Age3	0.803	0.070	0.648	0.811	0.911
Prop. of Unmarked Esc. Age4	0.126	0.058	0.044	0.116	0.266
Prop. of Unmarked Esc. Age5	0.071	0.042	0.019	0.061	0.179

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

For the Abernathy population the BF provided similar support for the null model compared to the length-dependent carcass recovery model (BF=2.7). Since there was similar support for both models, we chose the null model and did not stratify estimates by length (Table 40). The abundance estimate based on AUC divided by mean SL was 541 adults (95%CI 498-579). Based on adipose fin clips, the proportion of marked fish was 93.44% and 46.7% of the adults were females. Most adults were age 3 with very few age 5 fish. Subpopulation abundance, origin, sex, and age estimates are found in Table 41. We considered the estimate from the JS model (361adults with 95%CI 298-546), which had 95%CI that overlapped the AUC model estimate. However, we chose to report the AUC estimate due to the small number of recaptures in most periods and poor precision of the JS estimate (CV=17%).

Table 40. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Abernathy Chinook salmon >59cm.

Model	Constant	Sex	Length	Sex + Length
Posterior Model Probabilities	0.55	0.17	0.21	0.07
Bayes Factor	1.00	3.21	2.68	8.11

Table 41. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Abernathy Creek Tule Chinook salmon adult subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	540	20.57	498	541	579
Marked Esc.	500	21.40	457	501	542
Unmarked Esc.	40	9.61	23	39	60
Female Esc.	214	12.72	189	214	239
Male Esc.	326	15.78	294	327	357
Marked Esc. Age3	332	22.02	289	332	376
Marked Esc. Age4	150	17.68	117	150	186
Marked Esc. Age5	18	6.81	7	17	34
Unmarked Esc. Age3	26	7.99	13	26	44
Unmarked Esc. Age4	11	5.13	3	10	23
Unmarked Esc. Age5	3	2.67	0	2	10
Prop. of Marked Esc.	0.926	0.018	0.888	0.928	0.957
Prop. of Unmarked Esc.	0.074	0.018	0.043	0.072	0.112
Prop. of Female Esc.	0.396	0.018	0.361	0.396	0.431
Prop. of Male Esc.	0.604	0.018	0.569	0.604	0.639
Prop. of Marked Esc. Age3	0.664	0.034	0.595	0.664	0.728
Prop. of Marked Esc. Age4	0.300	0.033	0.238	0.299	0.366
Prop. of Marked Esc. Age5	0.036	0.014	0.015	0.035	0.067
Prop. of Unmarked Esc. Age3	0.665	0.116	0.421	0.672	0.868
Prop. of Unmarked Esc. Age4	0.268	0.110	0.085	0.258	0.510
Prop. of Unmarked Esc. Age5	0.067	0.063	0.002	0.048	0.233

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The Mill, Abernathy, and Germany Creek estimates were summed to obtain an estimate for the MAG (Mill/Abernathy/Germany) population. The total population estimate was 2770 (95%CI 2598-3024). The marked fraction was 93% and greater than 70% of the adults were age 3. Population abundance, origin, sex, and age estimates are found in Table 42.

Table 42. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Mill/Abernathy/Germany Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	2779	111.50	2589	2770	3024
Marked Esc.	2584	105.00	2404	2575	2817
Unmarked Esc.	195	24.46	151	193	247
Female Esc.	1294	71.14	1171	1288	1450
Male Esc.	1484	65.34	1366	1480	1619
Marked Esc. Age3	2130	95.69	1963	2122	2341
Marked Esc. Age4	387	30.20	331	386	451
Marked Esc. Age5	67	11.85	46	67	92
Unmarked Esc. Age3	136	21.25	99	134	182
Unmarked Esc. Age4	37	9.47	21	36	58
Unmarked Esc. Age5	22	6.66	11	21	37
Prop. of Marked Esc.	0.930	0.008	0.913	0.930	0.945
Prop. of Unmarked Esc.	0.070	0.008	0.055	0.070	0.087
Prop. of Female Esc.	0.466	0.014	0.439	0.466	0.494
Prop. of Male Esc.	0.534	0.014	0.506	0.535	0.561
Prop. of Marked Esc. Age3	0.824	0.012	0.801	0.824	0.846
Prop. of Marked Esc. Age4	0.150	0.011	0.130	0.150	0.172
Prop. of Marked Esc. Age5	0.026	0.005	0.018	0.026	0.036
Prop. of Unmarked Esc. Age3	0.699	0.052	0.589	0.702	0.792
Prop. of Unmarked Esc. Age4	0.189	0.044	0.114	0.186	0.286
Prop. of Unmarked Esc. Age5	0.112	0.033	0.058	0.109	0.187

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Toutle Population

The Toutle basin contains two subpopulations; the Green River and the SF Toutle. A third subpopulation may exist in the main stem/NF Toutle but this area was not surveyed due to high sediment loads, which cause poor survey conditions, and the low probability that spawners are present.

The Green population was comprised of spawners above and below the weir that were summed together. Above the weir carcass recoveries were not length- or sex-dependent (Table 43). Below the weir, the sex-dependent recovery model was better than the null model (BF=1.4). However, since both models had similar support so we chose not stratify age estimates by sex below the weir.

Table 43. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Green Chinook salmon >59cm.

Population	Model	Constant	Sex	Length	Sex + Length
Green>Weir	Posterior Model Probabilities	0.81	0.08	0.09	0.02
Green>Weir	Bayes Factor	1.00	10.50	8.66	39.86
Green<Weir	Posterior Model Probabilities	0.31	0.45	0.07	0.17
Green<Weir	Bayes Factor	1.42	1.00	6.02	2.66

The weir operated over the entire spawning run. A total of 1,954 adults were trapped and released above the weir of which 965 were tagged. The tests for variable marked carcass recovery proportions among survey areas and time periods provided positive support for pooling of recoveries among locations and time periods (BF of null model was 6.5 and 107 greater than the next best spatial and temporally stratified recovery model, respectively). These tests indicated that the pooled Petersen estimator would provide an consistent estimate of adult abundance, and this estimate was 1,963 (95% CI 1788 - 2189) with a CV of 5%. Since this estimate differed by two fish from the weir count, we have no reason to believe the weir was a not total barrier, and we therefore reported the weir count and considered it a census. For a comparison the mode of the Darroch model provide an estimate of 1,987 (95%CI 1606 – 4071) with a CV of 26%. The GOF test for the recapture portion of the Darroch model yielded a Bayesian *P*-value of 0.74 indicated the Darroch model fit the data. To estimate the number of spawners we multiplied the run size estimate by the proportion of successful spawners an subtracted the estimated sport harvest, which leaves an estimate of 1,685 adults (95%CI 1559 to 1782). The AUC estimate below the weir was 31 (95%CI 29-32). We conducted a JS estimate for the same area but had to subtract the carcasses that passed downstream over the weir. Although the JS and AUC estimates were similar, the CV for the JS estimate was near 100%. Therefore we used the AUC estimate. The proportion of marked fish for the Green subpopulation was 89% and most marked and unmarked fish were age 3 and 4. Subpopulation abundance, origin, sex, and age estimates are found in Table 44.

Table 44. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Green River Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	1713	57.43	1590	1717	1813
Marked Esc.	1528	58.77	1408	1531	1638
Unmarked Esc.	184	29.47	131	183	245
Female Esc.	646	34.10	580	646	713
Male Esc.	1066	44.46	975	1068	1150
Marked Esc. Age3	936	50.85	836	937	1034
Marked Esc. Age4	425	36.48	355	424	499
Marked Esc. Age5	168	23.80	124	167	218
Unmarked Esc. Age3	84	18.58	52	83	125
Unmarked Esc. Age4	76	17.03	46	74	113
Unmarked Esc. Age5	24	8.56	11	23	44
Prop. of Marked Esc.	0.892	0.017	0.858	0.893	0.923

Table 44. (continued)

Parameter*	mean	sd	2.50%	median	97.50%
Prop. of Unmarked Esc.	0.108	0.017	0.077	0.107	0.142
Prop. of Female Esc.	0.377	0.015	0.348	0.377	0.408
Prop. of Male Esc.	0.623	0.015	0.592	0.623	0.652
Prop. of Marked Esc. Age3	0.612	0.023	0.566	0.613	0.658
Prop. of Marked Esc. Age4	0.278	0.021	0.237	0.278	0.321
Prop. of Marked Esc. Age5	0.110	0.015	0.082	0.109	0.141
Prop. of Unmarked Esc. Age3	0.458	0.066	0.331	0.457	0.587
Prop. of Unmarked Esc. Age4	0.410	0.065	0.289	0.409	0.541
Prop. of Unmarked Esc. Age5	0.132	0.042	0.063	0.127	0.226

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

A carcass tagging study was designed for the SF Toutle but too few tags and recoveries occurred to produce an estimate. The test for sex and length selectivity provided positive support for the null model compared to the length- or sex-dependent models (Table 45).

Table 45. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 SF Toutle Chinook salmon >59cm.

Model	Constant	Sex	Length	Sex + Length
Posterior Model Probabilities	0.61	0.15	0.18	0.06
Bayes Factor	1.00	3.98	3.45	10.72

Redd surveys and AUC estimates were conducted concurrently with the carcass tagging. The redd based estimate was 415 (95%CI 302-624), which is slightly greater than the AUC based estimate of 205 (95%CI 189-219). We used the redd-based estimate because this estimate required making fewer assumptions that may have been violated during difficult environmental conditions (i.e. missed fish on surveys). Redd-based abundance estimates may be slightly biased high due to the small number of carcass recoveries and a higher than average proportion of males. As with the Green subpopulation we encountered a high proportion of marked fish (79%) and most adults were age 3 and 4. Subpopulation abundance, origin, sex, and age estimates are found in Table 46.

Table 46. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) SF Toutle Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	427	82.30	302	415	624
Marked Esc.	336	71.99	224	326	507
Unmarked Esc.	91	35.42	36	86	175
Female Esc.	185	48.33	108	179	298
Male Esc.	242	56.46	152	236	374
Marked Esc. Age3	133	40.48	69	128	228
Marked Esc. Age4	142	42.37	75	137	241
Marked Esc. Age5	61	25.98	22	57	123
Unmarked Esc. Age3	41	21.36	11	38	94
Unmarked Esc. Age4	25	15.77	4	22	64
Unmarked Esc. Age5	25	15.66	5	21	64
Prop. of Marked Esc.	0.787	0.071	0.631	0.793	0.907
Prop. of Unmarked Esc.	0.213	0.071	0.093	0.207	0.369
Prop. of Female Esc.	0.433	0.074	0.293	0.433	0.581
Prop. of Male Esc.	0.567	0.074	0.419	0.567	0.707
Prop. of Marked Esc. Age3	0.395	0.083	0.240	0.393	0.565
Prop. of Marked Esc. Age4	0.423	0.085	0.260	0.422	0.593
Prop. of Marked Esc. Age5	0.182	0.066	0.073	0.175	0.329
Prop. of Unmarked Esc. Age3	0.455	0.144	0.185	0.452	0.739
Prop. of Unmarked Esc. Age4	0.272	0.129	0.066	0.259	0.559
Prop. of Unmarked Esc. Age5	0.273	0.129	0.067	0.259	0.560

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Both the Green and SF Toutle estimates were summed to obtain an estimate for the Toutle population. The total population estimate was 2132 (95%CI 1962-2361). The marked fraction was 87% and most fish were age 3 or 4. Population abundance, origin, sex, and age estimates are found in Table 47.

Table 47. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Toutle River Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	2140	100.40	1962	2132	2361
Marked Esc.	1865	93.25	1694	1859	2065
Unmarked Esc.	275	45.86	195	272	375
Female Esc.	831	59.52	727	827	959
Male Esc.	1308	72.00	1177	1304	1463
Marked Esc. Age3	1069	65.25	946	1067	1204
Marked Esc. Age4	567	56.36	467	564	688
Marked Esc. Age5	229	35.32	168	226	307
Unmarked Esc. Age3	126	28.25	79	123	189
Unmarked Esc. Age4	100	23.25	62	98	152
Unmarked Esc. Age5	49	17.76	23	46	92
Prop. of Marked Esc.	0.872	0.020	0.830	0.873	0.907
Prop. of Unmarked Esc.	0.129	0.020	0.093	0.127	0.170
Prop. of Female Esc.	0.389	0.020	0.351	0.388	0.428
Prop. of Male Esc.	0.612	0.020	0.572	0.612	0.649
Prop. of Marked Esc. Age3	0.574	0.026	0.522	0.574	0.623
Prop. of Marked Esc. Age4	0.304	0.024	0.259	0.303	0.353
Prop. of Marked Esc. Age5	0.123	0.017	0.092	0.122	0.159
Prop. of Unmarked Esc. Age3	0.457	0.066	0.330	0.457	0.588
Prop. of Unmarked Esc. Age4	0.366	0.063	0.248	0.363	0.495
Prop. of Unmarked Esc. Age5	0.177	0.053	0.090	0.172	0.296

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Upper Cowlitz/Tilton Population

Fall Chinook salmon are captured at the Barrier Dam, trucked, and released into the Tilton, Upper Cowlitz, and Cispus rivers. Prior to being transported Chinook salmon are classified as males, females, and jacks and their mark status is recorded. However, scales are not taken to determine ages. We subtracted the angler harvest from the number of salmon released and assumed no fall back or no mortality due to transportation. A total of 4,246 adults (95%CI 4164-4319) we estimated to spawn in the Tilton River with 69% of these being unmarked. Subpopulation abundance, origin, and sex estimates are found in Table 48.

Table 48. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Tilton River Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	4245	39.72	4164	4246	4319
Marked Esc.	1319	30.55	1259	1319	1379
Unmarked Esc.	2925	50.27	2827	2925	3022
Female Esc.	1771	35.12	1702	1771	1840
Male Esc.	2474	38.37	2398	2474	2548
Prop. of Unmarked Esc.	0.689	0.008	0.674	0.689	0.705
Prop. of Marked Esc.	0.311	0.008	0.295	0.311	0.326
Prop. of Female Esc.	0.417	0.007	0.403	0.417	0.432
Prop. of Male Esc.	0.583	0.007	0.569	0.583	0.597

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

A total of 5,449 adults (95%CI 5449-5619) were estimated to spawn in the upper Cowlitz and Cispus Rivers. The proportion of marked fish was 86%. Subpopulation abundance, origin, and sex estimates are found in Table 49.

Table 49. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Upper Cowlitz/Cispus River Tule Chinook salmon adult subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	5525	43.53	5449	5525	5619
Marked Esc.	4750	50.78	4655	4750	4854
Unmarked Esc.	775	25.80	725	775	827
Female Esc.	2391	37.71	2317	2391	2466
Male Esc.	3135	41.38	3056	3134	3218
Prop. of Marked Esc.	0.860	0.005	0.850	0.860	0.869
Prop. of Unmarked Esc.	0.140	0.005	0.131	0.140	0.150
Prop. of Female Esc.	0.433	0.006	0.421	0.433	0.444
Prop. of Male Esc.	0.567	0.006	0.556	0.567	0.579

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The Tilton and Upper Cowlitz/Cispus were summed to obtain the population abundance, origin, and sex estimates, and are found in Table 50.

Table 50. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Upper Cowlitz combined Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	9770	62.81	9666	9764	9914
Marked Esc.	7676	74.73	7541	7672	7835
Unmarked Esc.	2094	39.84	2015	2094	2172
Female Esc.	4162	52.51	4061	4161	4269
Male Esc.	5608	58.17	5498	5607	5725
Prop. of Marked Esc.	0.786	0.004	0.777	0.786	0.794
Prop. of Unmarked Esc.	0.214	0.004	0.206	0.214	0.223
Prop. of Female Esc.	0.426	0.005	0.417	0.426	0.435
Prop. of Male Esc.	0.574	0.005	0.565	0.574	0.583

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Coweeman Population

This population was estimated using tGMR based on binomial sampling. To test tGMR the assumption of independent probability of capture of marked and unmarked individuals between the marking and recovery events, we used the Kao-Mallick approach for Bayesian variable selection based on a negative binomial regression with offspring per spawner as the dependent variable and age, location, sex, and origin of carcasses the independent variables. The variable inclusion probabilities for age, location, sex, and origin were 0.93, 0.67, 0.58, and 0.39, respectively. This suggests that models with age had better explanatory power than other variables. The resulting model probabilities are found in Table 51. There was similar support for the first four models that included age. The full model had positive support compared to the null model (BF=7.76). Examination of the data indicated that the four fish that produced the most offspring (19, 17, 15, 15) were very influential in regression and some of these would be considered outliers. Removing these possible outliers suggest the age and the null models were the most probable models supported by this reduced data set.

Table 51. Posterior model probabilities, Bayes factors, and support for the negative binomial model, Coweeman River Chinook salmon 2010..

Model	Prob.	BF	Support
loc,sex,or,age	0.244	1.00	Similar
loc,sex,age	0.190	1.29	Similar
loc,age	0.182	1.34	Similar
age	0.103	2.36	Similar
sex,age	0.071	3.41	Positive
loc,or,age	0.055	4.44	Positive
sex,or,age	0.053	4.61	Positive
null	0.031	7.76	Positive
or,age	0.028	8.78	Positive
sex,	0.018	13.47	Positive

Table 51. (continued)

Model	Prob.	BF	Support
loc	0.008	30.22	Strong
loc,sex	0.005	48.68	Strong
sex,or	0.005	50.60	Strong
or	0.005	51.67	Strong
loc,sex,or	0.001	236.80	Very Strong
loc,or	0.001	280.34	Very Strong

The population estimate was 635 (95%CI 494-810). The proportion of unmarked fish was 70%. In contrast to other populations, there tended to be older fish in the Coweeman population. Population abundance, origin, sex, and age estimates are found in Table 52.

Table 52. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Coweeman River Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	639	80.52	494	635	810
Marked Esc.	193	40.11	123	190	280
Unmarked Esc.	446	64.76	330	442	585
Female Esc.	350	55.83	250	346	469
Male Esc.	289	50.69	201	286	398
Marked Esc. Age3	28	13.89	8	26	61
Marked Esc. Age4	76	24.16	37	73	131
Marked Esc. Age5	90	26.24	46	87	149
Unmarked Esc. Age3	118	31.11	66	115	187
Unmarked Esc. Age4	265	48.54	179	262	369
Unmarked Esc. Age5	63	22.25	28	60	115
Prop. of Marked Esc.	0.303	0.050	0.210	0.301	0.404
Prop. of Unmarked Esc.	0.697	0.050	0.596	0.699	0.790
Prop. of Female Esc.	0.547	0.054	0.440	0.548	0.650
Prop. of Male Esc.	0.453	0.054	0.350	0.452	0.560
Prop. of Marked Esc. Age3	0.144	0.065	0.042	0.135	0.292
Prop. of Marked Esc. Age4	0.392	0.092	0.222	0.388	0.579
Prop. of Marked Esc. Age5	0.464	0.093	0.288	0.464	0.650
Prop. of Unmarked Esc. Age3	0.264	0.058	0.158	0.261	0.384
Prop. of Unmarked Esc. Age4	0.595	0.064	0.468	0.595	0.719
Prop. of Unmarked Esc. Age5	0.141	0.045	0.066	0.137	0.241

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Kalama Population

Since there was no tagging in 2010 for this population, we could not test sex for length selectivity of carcass recoveries. Since the BF favored the null model in 12 of 15 populations we assumed that carcass recoveries were not influenced by length and sex, and did not attempt to stratify abundance estimates based on these covariates. A total of 7,065 (95%CI 6513-7565) adults are estimated to have spawned in the Kalama based on the AUC model. The proportion of marked fish was 88%, and most spawners were age 4. Population abundance, origin, sex, and age estimates are found in Table 53.

Table 53. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Kalama River Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	7057	268.80	6513	7065	7565
Marked Esc.	6225	244.40	5731	6231	6691
Unmarked Esc.	832	67.57	706	830	969
Female Esc.	3603	228.80	3160	3604	4054
Male Esc.	3454	226.30	3014	3453	3916
Marked Esc. Age3	1723	166.80	1402	1720	2059
Marked Esc. Age4	2763	201.50	2374	2759	3162
Marked Esc. Age5	1739	166.70	1431	1734	2080
Unmarked Esc. Age3	327	56.11	225	325	445
Unmarked Esc. Age4	354	58.40	248	351	477
Unmarked Esc. Age5	151	40.38	81	148	239
Prop. of Marked Esc.	0.882	0.008	0.865	0.882	0.898
Prop. of Unmarked Esc.	0.118	0.008	0.102	0.118	0.135
Prop. of Female Esc.	0.511	0.026	0.459	0.510	0.562
Prop. of Male Esc.	0.490	0.026	0.438	0.490	0.541
Prop. of Marked Esc. Age3	0.277	0.025	0.230	0.277	0.326
Prop. of Marked Esc. Age4	0.444	0.027	0.392	0.444	0.497
Prop. of Marked Esc. Age5	0.279	0.025	0.234	0.279	0.329
Prop. of Unmarked Esc. Age3	0.393	0.060	0.281	0.392	0.513
Prop. of Unmarked Esc. Age4	0.425	0.060	0.310	0.423	0.546
Prop. of Unmarked Esc. Age5	0.181	0.046	0.101	0.179	0.279

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Lewis Population

This Tule population is composed on three subpopulations including Cedar, EF Lewis, and NF Lewis and we report on the first two components in this population. Cedar Creek estimates are the sum of the JS model for fish spawning below the trap and the trap count, which is assumed to be a census of upstream spawners. Bayes factors slightly supported the length-dependent carcass recovery model compared to the null model (BF=1.06), which equates to similar support for both models (Table 54). As with other subpopulations we used the null model and did not stratify the estimates based on length. DIC favored the ttt model but was not very sensitive to model choice (Table 55). The GOF test yielded a Bayesian *P*-value of 0.45 indicating the model fit the data. The abundance estimate based on the fish ladder count and the JS model was 551 adults (95%CI

483-674). The proportion of unmarked escapement was 61% with a high proportion of age 3 adults. Subpopulation abundance, origin, sex, and age estimates are found in Table 56.

Table 54. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Cedar Chinook salmon >59cm.

Model	Constant	Sex	Length	Sex + Length
Posterior Model Probabilities	0.40	0.08	0.42	0.10
Bayes Factor	1.06	5.11	1.00	4.10

Table 55. Model selection for the JS model for large adults (>80cm) adults in Cedar Creek, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	69.7	13.4	83.1	0.0	0.667	378	0.45
stt	71.4	13.3	84.7	1.6	0.300	328	0.26
tst	77.0	12.6	89.6	6.5	0.026	330	0.43
sst	81.9	10.1	92.0	8.9	0.008	314	0.00

Table 56. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Cedar Creek Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	558	48.80	483	551	674
Marked Esc.	219	24.48	178	216	274
Unmarked Esc.	339	33.02	287	335	416
Female Esc.	211	23.04	172	208	262
Male Esc.	347	34.63	292	343	429
Marked Esc. Age3	100	12.58	78	99	126
Marked Esc. Age4	87	15.45	61	86	123
Marked Esc. Age5	32	8.57	18	31	50
Unmarked Esc. Age3	187	20.30	152	185	232
Unmarked Esc. Age4	119	16.30	91	117	155
Unmarked Esc. Age5	34	9.24	19	33	55
Prop. of Marked Esc.	0.392	0.027	0.341	0.392	0.444
Prop. of Unmarked Esc.	0.608	0.027	0.556	0.608	0.659
Prop. of Female Esc.	0.378	0.027	0.327	0.378	0.431
Prop. of Male Esc.	0.622	0.027	0.569	0.623	0.673
Prop. of Marked Esc. Age3	0.457	0.045	0.368	0.457	0.545
Prop. of Marked Esc. Age4	0.399	0.045	0.314	0.398	0.488
Prop. of Marked Esc. Age5	0.145	0.034	0.086	0.143	0.215
Prop. of Unmarked Esc. Age3	0.551	0.035	0.482	0.551	0.618
Prop. of Unmarked Esc. Age4	0.350	0.033	0.287	0.349	0.417
Prop. of Unmarked Esc. Age5	0.100	0.023	0.060	0.098	0.150

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The EF Lewis escapement estimates were based on AUC methods. Since there was no tagging in 2010 for this population, we could not test sex or length selectivity of carcass recoveries. Since the BF favored the null model in 12 of 15 populations we assumed that carcass recoveries were not influenced by length and sex, and did not attempt to stratify abundance estimates based on these covariates. The abundance estimate was 427 adults (95%CI 394-457). The unmarked fraction was high (89%) and age proportions were greatest for age 4 fish. Subpopulation abundance, origin, sex, and age estimates are found in Table 57.

Table 57. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) EF Lewis River Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	426	16.24	394	427	457
Marked Esc.	48	12.19	27	47	75
Unmarked Esc.	378	18.68	341	379	413
Female Esc.	219	21.12	178	218	260
Male Esc.	208	20.80	168	208	249
Marked Esc. Age3	18	7.42	7	17	35
Marked Esc. Age4	24	8.53	10	23	43
Marked Esc. Age5	6	4.28	1	5	17
Unmarked Esc. Age3	129	17.98	95	128	166
Unmarked Esc. Age4	179	19.88	141	178	218
Unmarked Esc. Age5	71	14.27	46	70	101
Prop. of Marked Esc.	0.113	0.028	0.064	0.110	0.174
Prop. of Unmarked Esc.	0.887	0.028	0.826	0.890	0.936
Prop. of Female Esc.	0.513	0.045	0.423	0.513	0.601
Prop. of Male Esc.	0.488	0.045	0.400	0.487	0.577
Prop. of Marked Esc. Age3	0.376	0.118	0.166	0.370	0.618
Prop. of Marked Esc. Age4	0.499	0.121	0.267	0.499	0.733
Prop. of Marked Esc. Age5	0.125	0.081	0.016	0.109	0.319
Prop. of Unmarked Esc. Age3	0.340	0.045	0.257	0.339	0.430
Prop. of Unmarked Esc. Age4	0.472	0.047	0.380	0.472	0.563
Prop. of Unmarked Esc. Age5	0.188	0.037	0.122	0.186	0.264

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The combined EF Lewis and Cedar Creek population estimate was 979 (95%CI 901-1103). As with the two subpopulations, there was a high proportion of unmarked fish (73%) and most adults were age 3. Population abundance, origin, sex, and age estimates are found in Table 58.

Table 58. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Lewis River Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	984	51.38	901	979	1103
Marked Esc.	267	27.37	220	265	328
Unmarked Esc.	718	37.94	651	715	802
Female Esc.	429	31.15	371	428	493
Male Esc.	555	40.16	485	552	645
Marked Esc. Age3	118	14.65	92	117	149
Marked Esc. Age4	111	17.70	81	110	151
Marked Esc. Age5	38	9.61	21	37	59
Unmarked Esc. Age3	315	27.22	266	314	372
Unmarked Esc. Age4	297	25.66	249	296	350
Unmarked Esc. Age5	105	17.03	75	104	141
Prop. of Marked Esc.	0.271	0.021	0.232	0.270	0.312
Prop. of Unmarked Esc.	0.729	0.021	0.688	0.730	0.768
Prop. of Female Esc.	0.436	0.025	0.387	0.436	0.486
Prop. of Male Esc.	0.564	0.025	0.515	0.564	0.613
Prop. of Marked Esc. Age3	0.442	0.043	0.358	0.442	0.527
Prop. of Marked Esc. Age4	0.417	0.043	0.333	0.416	0.504
Prop. of Marked Esc. Age5	0.141	0.031	0.085	0.139	0.209
Prop. of Unmarked Esc. Age3	0.439	0.029	0.383	0.439	0.496
Prop. of Unmarked Esc. Age4	0.414	0.030	0.357	0.414	0.474
Prop. of Unmarked Esc. Age5	0.146	0.022	0.106	0.145	0.193

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Washougal Population

The selectivity of carcass recoveries was tested using a logistic regression model with sex and length as covariates. For all carcasses the best model based on BF was the length-dependent model, although there was also support for the model that included both length and sex (BF=3.0). We therefore stratified carcasses into large and small adults (Table 59). For large adults the BF provided similar support for the null and length-dependent model (BF=1.06) and for small adults the BF favored the null model but provided similar support for the length-dependent model (BF=1.8). Therefore, the Jolly-Seber model was run for the small and large adults separately.

Table 59. Results from model selection using the logistic model used to assess factors affecting carcass recapture probabilities for 2010 Washougal Chinook salmon >59cm.

Population	Model	Constant	Sex	Length	Sex + Length
Washougal	Posterior Model Probabilities	0.06	0.03	0.69	0.22
Washougal	Bayes Factor	12.27	27.18	1.00	3.08
Wash ≥80cm	Posterior Model Probabilities	0.40	0.06	0.46	0.08
Wash ≥80cm	Bayes Factor	1.14	7.38	1.00	5.86
Wash <80cm	Posterior Model Probabilities	0.49	0.14	0.27	0.10
Wash <80cm	Bayes Factor	1.00	3.58	1.79	4.85

We used DIC for to select the best model for large and small adults (Tables 60 and 61). DIC favored the ttt model for both groups. Abundance estimates were relatively similar across all models, except the sst model, and thus estimates were not very sensitive to model choice. The Bayesian *P*-value for the smaller adults indicted the model fit the data (*P*= 0.40) but there was lack of fit for the larger adults (*P* = 0.02). The estimated adult escapement in the Washougal from the size stratified JS models combined with the census of fish passed above the hatchery weir was 6,075 adults (95%CI 5627-6642). The proportion of marked fish was 87%. Marked fish tended to be primarily age 3 and 4 while unmarked fish tended to be age 4 and 5. Population abundance, origin, sex, and age estimates are found in Table 62.

Table 60. JS model selection for large Chinook Salmon adults (≥80cm) in Washougal River, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	137.5	21.5	159.0	0.0	1.000	2735	0.02
stt	171.8	19.1	190.9	31.9	0.000	2432	0.00
tst	170.5	17.4	187.9	28.9	0.000	2612.5	0.00
sst	280.0	15.0	295.0	136.0	0.000	2041	0.00

Table 61. JS model selection for small Chinook salmon adults (≤80cm) in Washougal River, 2010.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	120.7	19.2	139.9	0.0	1.000	3358	0.40
stt	163.8	17.8	181.6	41.7	0.000	2829	0.00
tst	148.1	14.5	162.6	22.7	0.000	3249	0.00
sst	265.6	13.4	279.0	139.1	0.000	2515	0.00

Table 62. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Washougal River Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	6087	262.00	5627	6075	6642
Marked Esc.	5254	255.40	4784	5243	5788
Unmarked Esc.	833	118.20	626	824	1089
Female Esc.	3134	147.40	2866	3126	3441
Male Esc.	2953	143.50	2694	2946	3256
Marked Esc. Age3	3098	217.30	2699	3089	3559
Marked Esc. Age4	2042	195.70	1676	2034	2447
Marked Esc. Age5	115	57.04	31	105	254
Unmarked Esc. Age3	172	52.97	80	168	285
Unmarked Esc. Age4	301	77.38	167	295	472
Unmarked Esc. Age5	360	94.48	199	353	566
Prop. of Marked Esc.	0.863	0.019	0.824	0.865	0.897
Prop. of Unmarked Esc.	0.137	0.019	0.103	0.136	0.176
Prop. of Female Esc.	0.515	0.010	0.494	0.515	0.535
Prop. of Male Esc.	0.485	0.010	0.465	0.485	0.506
Prop. of Marked Esc. Age3	0.590	0.031	0.528	0.590	0.651
Prop. of Marked Esc. Age4	0.389	0.031	0.328	0.388	0.450
Prop. of Marked Esc. Age5	0.022	0.011	0.006	0.020	0.048
Prop. of Unmarked Esc. Age3	0.209	0.065	0.096	0.204	0.346
Prop. of Unmarked Esc. Age4	0.362	0.080	0.212	0.359	0.527
Prop. of Unmarked Esc. Age5	0.429	0.079	0.277	0.429	0.586

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Upper Gorge Population

The Upper Gorge Tule population consists of the Wind and Little White Salmon populations. Since spawning ground surveys occurred near the peak of spawning, the Wind population was estimated using a peak count expansion factor estimates from the 1,965 carcass tagging study based on the JS model. Based on DIC the best model was stt and the model GOF test based in Bayesian *P*-values was 0.47, which indicates the data fit the model (Table 63).

Table 63. JS model selection for Chinook salmon adults (≥59cm) in the Wind River, 1965.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC_weights	Abundance	CJS_pvalue
ttt	113.4	22.4	135.8	4.9	0.071	237	0.57
stt	111.6	19.3	130.9	0.0	0.818	236	0.47
tst	119.8	19.0	138.8	7.9	0.016	237	0.29
sst	119.5	15.7	135.2	4.3	0.095	236	0.45

The expansion factor for this population was 1.19 (95%CI 1.13-1.28). There are approximately two miles of spawning habitat accessible to Chinook salmon and the Tule population estimate was 83 adults (95%CI 79-90). The proportion of marked fish was high (76%) and greater than 60% of the adults were age 3. Population abundance, origin, sex, and age estimates are found in Table 64.

Table 64. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Wind River Tule Chinook salmon subpopulation.

Parameter*	mean	Sd	2.50%	median	97.50%
Escapement	84	2.82	79	83	90
Marked Esc.	63	6.26	50	63	74
Unmarked Esc.	21	5.94	11	21	34
Female Esc.	41	7.17	27	41	55
Male Esc.	43	7.14	29	43	57
Marked Esc. Age3	38	10.00	18	38	57
Marked Esc. Age4	19	8.92	4	18	39
Marked Esc. Age5	6	5.72	0	5	21
Unmarked Esc. Age3	17	5.13	9	17	29
Unmarked Esc. Age4	2	1.35	0	2	5
Unmarked Esc. Age5	1	1.09	0	1	4
Prop. of Marked Esc.	0.750	0.070	0.600	0.755	0.874
Prop. of Unmarked Esc.	0.250	0.070	0.126	0.245	0.400
Prop. of Female Esc.	0.486	0.084	0.324	0.485	0.649
Prop. of Male Esc.	0.514	0.084	0.351	0.515	0.677
Prop. of Marked Esc. Age3	0.605	0.147	0.301	0.611	0.867
Prop. of Marked Esc. Age4	0.296	0.138	0.074	0.280	0.597
Prop. of Marked Esc. Age5	0.099	0.090	0.003	0.074	0.334
Prop. of Unmarked Esc. Age3	0.828	0.069	0.676	0.835	0.940
Prop. of Unmarked Esc. Age4	0.103	0.055	0.022	0.094	0.233
Prop. of Unmarked Esc. Age5	0.069	0.046	0.009	0.060	0.182

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

For the Little White Salmon population, we developed a peak count expansion factor based on the 1,966 carcass tagging study. Model selection using DIC favored the ttt model and there was good fit (Bayesian *P*-value = 0.60). The 1966 expansion factor for this population based only on carcasses was 3.28 (95%CI 2.71-4.31).

Table 65. JS model selection for Chinook salmon adults (≥59cm) in the Little White Salmon River, 1966.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	70.7	15.6	86.3	0.0	0.840	290	0.60
stt	82.3	14.5	96.8	10.5	0.004	258	0.03
tst	76.1	13.8	89.9	3.6	0.139	254	0.44
sst	82.5	11.7	94.2	7.9	0.016	230	0.02

Based on this expansion factor, we estimated 341 adult Tule Chinook salmon spawned in the Little White Salmon River (95%CI 281-448). Over 90% of the spawners were unmarked and most were age 3. Subpopulation abundance, origin, sex, and age estimates are found in Table 66.

Table 66. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Little White Salmon River Tule Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	348	43.28	281	341	448
Marked Esc.	35	9.94	19	34	57
Unmarked Esc.	313	39.95	251	307	406
Female Esc.	107	19.08	75	105	151
Male Esc.	241	32.65	188	237	315
Marked Esc. Age3	26	8.29	13	25	45
Marked Esc. Age4	4	3.18	1	4	13
Marked Esc. Age5	4	3.27	1	4	13
Unmarked Esc. Age3	275	36.22	219	271	359
Unmarked Esc. Age4	27	8.63	13	26	46
Unmarked Esc. Age5	10	5.13	3	9	22
Prop. of Marked Esc.	0.102	0.025	0.057	0.100	0.157
Prop. of Unmarked Esc.	0.899	0.025	0.843	0.900	0.943
Prop. of Female Esc.	0.308	0.038	0.235	0.307	0.385
Prop. of Male Esc.	0.692	0.038	0.615	0.693	0.765
Prop. of Marked Esc. Age3	0.751	0.105	0.521	0.761	0.920
Prop. of Marked Esc. Age4	0.125	0.080	0.017	0.109	0.316
Prop. of Marked Esc. Age5	0.125	0.081	0.017	0.108	0.319
Prop. of Unmarked Esc. Age3	0.881	0.029	0.819	0.883	0.931
Prop. of Unmarked Esc. Age4	0.087	0.025	0.045	0.085	0.141
Prop. of Unmarked Esc. Age5	0.032	0.016	0.009	0.029	0.069

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The Upper Gorge tule Chinook population (Wind and Little White Salmon Rivers combined) abundance, origin, sex, and age estimates can be found in Table 67. The estimates abundance for this population is 425 adults (95%CI 365-532). The unmarked fraction was 78% with most fish being age 3.

Table 67. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Upper Gorge tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	432	43.34	365	425	532
Marked Esc.	98	11.60	77	97	122
Unmarked Esc.	333	40.43	271	329	427
Female Esc.	148	20.44	113	146	193
Male Esc.	284	33.41	229	280	360
Marked Esc. Age3	64	12.91	40	64	90
Marked Esc. Age4	23	9.42	8	22	44
Marked Esc. Age5	11	6.54	2	9	27
Unmarked Esc. Age3	293	36.60	235	288	377
Unmarked Esc. Age4	29	8.73	15	29	49
Unmarked Esc. Age5	11	5.25	4	10	24
Prop. of Marked Esc.	0.229	0.028	0.177	0.228	0.284
Prop. of Unmarked Esc.	0.771	0.028	0.716	0.772	0.823
Prop. of Female Esc.	0.343	0.035	0.275	0.342	0.412
Prop. of Male Esc.	0.657	0.035	0.588	0.658	0.725
Prop. of Marked Esc. Age3	0.656	0.103	0.444	0.661	0.842
Prop. of Marked Esc. Age4	0.235	0.095	0.081	0.225	0.444
Prop. of Marked Esc. Age5	0.109	0.065	0.021	0.096	0.270
Prop. of Unmarked Esc. Age3	0.877	0.027	0.819	0.879	0.926
Prop. of Unmarked Esc. Age4	0.088	0.024	0.048	0.087	0.139
Prop. of Unmarked Esc. Age5	0.034	0.015	0.012	0.032	0.070

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

A fall Chinook salmon “Bright” population has been established in the Wind and Little White Salmon Rivers. We estimated 246 adults (95%CI 232-264) in the Wind Bright subpopulation, which is more than the Tule abundance estimate of 83 adults. The proportion of unmarked fish was 70% with most adults being age 4 and 5. Subpopulation abundance, origin, sex, and age estimates are found in Table 68.

Table 68. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Wind River bright Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	246	8.31	232	246	264
Marked Esc.	74	11.46	53	73	97
Unmarked Esc.	172	12.59	148	173	197
Female Esc.	165	12.46	141	165	190
Male Esc.	81	11.43	60	81	105
Marked Esc. Age3	18	6.16	8	17	32
Marked Esc. Age4	29	7.80	16	29	46
Marked Esc. Age5	27	7.44	14	26	43
Unmarked Esc. Age3	33	8.17	18	32	50
Unmarked Esc. Age4	72	11.16	52	72	96
Unmarked Esc. Age5	68	10.99	48	67	91
Prop. of Marked Esc.	0.301	0.045	0.217	0.299	0.392
Prop. of Unmarked Esc.	0.700	0.045	0.608	0.701	0.783
Prop. of Female Esc.	0.670	0.045	0.578	0.671	0.755
Prop. of Male Esc.	0.330	0.045	0.245	0.329	0.422
Prop. of Marked Esc. Age3	0.241	0.073	0.113	0.236	0.398
Prop. of Marked Esc. Age4	0.395	0.084	0.237	0.392	0.564
Prop. of Marked Esc. Age5	0.364	0.083	0.212	0.361	0.531
Prop. of Unmarked Esc. Age3	0.189	0.045	0.108	0.187	0.288
Prop. of Unmarked Esc. Age4	0.419	0.057	0.311	0.418	0.533
Prop. of Unmarked Esc. Age5	0.392	0.056	0.284	0.391	0.505

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

For the Little White Salmon bright subpopulation, we estimated 259 adults (95%CI 214-340). This estimate is lower than the Tule estimate of 341 adults. The marked fraction was 57% with most adults being age 4. Subpopulation abundance, origin, sex, and age estimates are found in Table 69.

Table 69. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Little White Salmon River bright Chinook salmon subpopulation.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	264	32.88	214	259	340
Marked Esc.	149	23.02	111	147	201
Unmarked Esc.	115	19.69	83	113	160
Female Esc.	157	23.46	119	155	209
Male Esc.	107	18.49	76	105	149
Marked Esc. Age3	51	12.39	31	50	79
Marked Esc. Age4	63	13.94	39	61	93
Marked Esc. Age5	35	10.10	18	34	58
Unmarked Esc. Age3	48	12.00	28	47	75
Unmarked Esc. Age4	53	12.56	32	52	81
Unmarked Esc. Age5	13	6.08	4	13	28
Prop. of Marked Esc.	0.564	0.051	0.463	0.565	0.661
Prop. of Unmarked Esc.	0.436	0.051	0.339	0.435	0.537
Prop. of Female Esc.	0.595	0.048	0.500	0.595	0.687
Prop. of Male Esc.	0.405	0.048	0.313	0.405	0.500
Prop. of Marked Esc. Age3	0.344	0.063	0.225	0.343	0.471
Prop. of Marked Esc. Age4	0.419	0.066	0.292	0.419	0.551
Prop. of Marked Esc. Age5	0.236	0.057	0.134	0.233	0.358
Prop. of Unmarked Esc. Age3	0.420	0.074	0.279	0.419	0.567
Prop. of Unmarked Esc. Age4	0.464	0.075	0.319	0.463	0.609
Prop. of Unmarked Esc. Age5	0.117	0.048	0.040	0.111	0.226

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

The combined Lower Gorge Bright estimate was 506 adults (95%CI 458-589). This is larger than the tule adult estimate of 425. Over 40% of the adults from this population were age 4 and 56% were unmarked fish. Population abundance, origin, sex, and age estimates are found in Table 70.

Table 70. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) Upper Gorge bright Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	511	33.82	458	506	589
Marked Esc.	223	25.66	178	221	280
Unmarked Esc.	288	23.33	246	286	338
Female Esc.	322	26.38	275	320	380
Male Esc.	189	21.64	150	187	235
Marked Esc. Age3	69	13.81	45	68	99
Marked Esc. Age4	92	16.03	64	91	126
Marked Esc. Age5	62	12.61	40	61	89
Unmarked Esc. Age3	81	14.54	55	80	113
Unmarked Esc. Age4	126	16.84	95	124	162
Unmarked Esc. Age5	81	12.49	58	80	107
Prop. of Marked Esc.	0.437	0.035	0.368	0.436	0.506
Prop. of Unmarked Esc.	0.564	0.035	0.494	0.564	0.632
Prop. of Female Esc.	0.631	0.033	0.566	0.631	0.695
Prop. of Male Esc.	0.369	0.033	0.306	0.369	0.434
Prop. of Marked Esc. Age3	0.310	0.049	0.217	0.309	0.410
Prop. of Marked Esc. Age4	0.411	0.052	0.311	0.411	0.515
Prop. of Marked Esc. Age5	0.279	0.048	0.191	0.277	0.377
Prop. of Unmarked Esc. Age3	0.281	0.042	0.202	0.279	0.366
Prop. of Unmarked Esc. Age4	0.437	0.046	0.349	0.436	0.529
Prop. of Unmarked Esc. Age5	0.282	0.041	0.206	0.281	0.366

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

White Salmon Population

Historically, tule Chinook salmon spawned in the White Salmon River but there are tule and bright populations currently spawning in this river. As with the Lower Gorge population spawning ground surveys occurred near the peak of the Tule and Bright spawning times, thus we used a peak count expansion factor to estimate abundance. Based on the Bright expansion factor for this river (described below), the Tule estimate was 1,503 adults (95%CI 1427-1592). The proportion of unmarked fish is 73% with age 3 fish accounting for over 90% of the adults. Population abundance, origin, sex, and age estimates are found in Table 71.

Table 71. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) White Salmon River Tule Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	1505	41.78	1427	1503	1592
Marked Esc.	408	60.01	297	406	531
Unmarked Esc.	1097	66.60	967	1098	1226
Female Esc.	552	65.15	428	550	684
Male Esc.	954	68.20	819	954	1086
Marked Esc. Age3	364	57.32	257	362	483
Marked Esc. Age4	33	18.85	7	30	79
Marked Esc. Age5	11	11.02	0	8	40
Unmarked Esc. Age3	1029	67.95	895	1030	1162
Unmarked Esc. Age4	57	24.96	19	54	114
Unmarked Esc. Age5	11	11.23	0	8	41
Prop. of Marked Esc.	0.271	0.039	0.198	0.270	0.351
Prop. of Unmarked Esc.	0.729	0.039	0.649	0.730	0.803
Prop. of Female Esc.	0.367	0.042	0.286	0.366	0.451
Prop. of Male Esc.	0.634	0.042	0.549	0.634	0.714
Prop. of Marked Esc. Age3	0.892	0.050	0.775	0.899	0.968
Prop. of Marked Esc. Age4	0.081	0.044	0.018	0.074	0.187
Prop. of Marked Esc. Age5	0.027	0.026	0.001	0.019	0.098
Prop. of Unmarked Esc. Age3	0.938	0.025	0.882	0.940	0.977
Prop. of Unmarked Esc. Age4	0.052	0.023	0.017	0.049	0.102
Prop. of Unmarked Esc. Age5	0.010	0.010	0.000	0.007	0.037

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

We re-analyzed the 1989 "bright" Chinook carcass tagging data to develop a peak count expansion factor. DIC and Bayesian *P*-values favored the ttt model. Graphical examination of the data indicated these estimates of gross births were high (indicating they may have been Tules) during the first period and the remaining estimates of gross births approximated a normal distribution. Therefore, we did not include the first period in the "Bright" escapement estimate. Based on this our expansion factor estimate is 2.40 (95%CI 2.28-2.54), which we used for both the Tule and Bright populations in this river.

Table 72. JS model selection for Chinook salmon adults (≥59cm) in the White Salmon River, 1989.

Model	Deviance	JS_pv	JS_DIC	Δ DIC	DIC Weights	Abundance	CJS_pvalue
ttt	204.9	32.9	237.8	0.0	0.991	1150	0.09
stt	231.4	27.1	258.5	20.7	0.000	1127	0.00
tst	221.0	26.2	247.2	9.4	0.009	1145	0.04
sst	310.0	20.5	330.5	92.7	0.000	1147	0.00

The White Salmon Bright population estimate was 1,148 adults (95%CI 1090-1216). The marked fraction was 57% and most fish were age 3. The Bright population represents ~ 40% of the total population in the White Salmon, while the Tule population comprises ~ 60%. Population abundance, origin, sex, and age estimates are found in Table 73.

Table 73. Abundance, origin, sex, and age estimates for the 2010 adult (>59 cm) White Salmon River Bright Chinook salmon population.

Parameter*	mean	sd	2.50%	median	97.50%
Escapement	1150	31.92	1090	1148	1216
Marked Esc.	650	44.97	562	650	738
Unmarked Esc.	500	43.37	417	499	587
Female Esc.	694	42.49	611	694	778
Male Esc.	456	40.41	379	456	537
Marked Esc. Age3	239	33.89	177	238	309
Marked Esc. Age4	291	36.31	222	289	365
Marked Esc. Age5	120	25.08	76	118	173
Unmarked Esc. Age3	128	24.79	83	126	180
Unmarked Esc. Age4	225	31.84	166	224	290
Unmarked Esc. Age5	148	26.43	100	146	204
Prop. of Marked Esc.	0.565	0.036	0.494	0.566	0.634
Prop. of Unmarked Esc.	0.435	0.036	0.367	0.434	0.506
Prop. of Female Esc.	0.604	0.033	0.537	0.604	0.668
Prop. of Male Esc.	0.396	0.033	0.332	0.396	0.463
Prop. of Marked Esc. Age3	0.368	0.045	0.282	0.367	0.458
Prop. of Marked Esc. Age4	0.447	0.046	0.359	0.447	0.540
Prop. of Marked Esc. Age5	0.184	0.036	0.119	0.182	0.260
Prop. of Unmarked Esc. Age3	0.255	0.044	0.174	0.253	0.348
Prop. of Unmarked Esc. Age4	0.450	0.050	0.353	0.449	0.549
Prop. of Unmarked Esc. Age5	0.295	0.046	0.208	0.294	0.389

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.

Population Summary

The individual population estimates were summed to provide a Lower Columbia River (LCR) ESU-scale estimate for Washington populations of Tule, Lewis Brights, Rogue Brights, and Bonneville (BON) Pool Brights. We estimated 38,460 adult Tule Chinook salmon in the WA LCR ESU. The proportion of unmarked Tules was 21%, which yields an estimate of 8,238 unmarked Tules. Since the high hatchery mark rate this is very close to the hatchery origin spawner (HOS) estimate. A total of 67 Rogue River Bright strayed into the Grays River. Roler et al. (2013) reported 9,612 NF Lewis Brights. BON Pool Brights in the WA LCR ESU totaled 1,657 of which 53% were marked.

Table 74. Abundance and origin, and run type estimates for the 2010 adult (>59 cm) WA LCR Chinook salmon populations.

Parameter*	mean	sd	2.50%	median	97.50%
WA Adult Tules #	38465	601	37290	38460	39650
WA Marked Tules #	30222	521	29220	30220	31250
WA Unmarked Tules #	8243	203	7857	8239	8655
Prop. Of Marked Tules	0.786	0.004	0.777	0.786	0.794
Prop. Of Unmarked Tules	0.214	0.004	0.206	0.214	0.223
Rogue Brights in Grays R.	67	20	35	65	115
Lewis River Brights	9612	NA	NA	NA	NA
BON Pool Brights	1660	47	1578	1657	1761
Marked BON Brights	873	52	775	872	976
Unmarked BON Brights	788	49	695	787	886
Prop. Of Marked BON Brights	0.526	0.027	0.473	0.526	0.578
Prop. Of Unmarked BON Brights	0.474	0.027	0.422	0.474	0.527

*The sum of abundance by marked status, sex, and age may not equal the abundance estimate due to rounding errors.
includes Cowlitz and NF Lewis Tule estimate from Roler et al. 2013.

Tag Loss

Tag loss for carcasses tags was evaluated using a double tagging experiment for seven tagging groups. The median probability of losing a single tag was 0.9% to 5.4% and the probability of retaining at least one tag ranged from 99.7% to 99.9% (Table 75). Based on this analysis no adjustment for tag loss was required for Jolly-Seber population estimates. The median probability for Floy tag loss was higher ranging from 8% to 32% (Table 76). However, the median probability of retaining at least one tag ranged from 84% to 99%. We reported tag loss for Floy tags but tag loss did not affect our estimate because we used a permanent opercle mark that was rotated weekly for our mark-recapture estimates.

Table 75. Tag loss for Chinook salmon tagged with carcasses tags, where t_2 is the number of fish retaining two tags, t_1 is the number of fish retaining one of two tags, p_i is the estimated probability of losing a single tag and q_0_i is the probability of a tagged fish retaining at least one tag.

Population	*t_2*	*t_1*	Parameter	mean	sd	2.50%	median	97.50%
Mill	360	25	p[1]	0.034	0.005	0.026	0.034	0.044
Abernathy	85	8	p[2]	0.048	0.012	0.028	0.047	0.073
Germany	377	22	p[3]	0.029	0.004	0.021	0.029	0.038
Coweeman	12	1	p[4]	0.060	0.035	0.012	0.054	0.144
Washougal	457	24	p[5]	0.026	0.004	0.020	0.026	0.034
Green	51	2	p[6]	0.024	0.011	0.008	0.023	0.049
Skamokawa	19	0	p[7]	0.014	0.014	0.000	0.009	0.049
Mill	-	-	q_0[1]	0.999	0.000	0.998	0.999	0.999
Abernathy	-	-	q_0[2]	0.998	0.001	0.995	0.998	0.999

Table 75. (continued)

Population	*t_2*	*t_1*	Parameter	mean	sd	2.50%	median	97.50%
Germany	-	-	q_0[3]	0.999	0.000	0.999	0.999	1.000
Coweeman	-	-	q_0[4]	0.995	0.006	0.979	0.997	1.000
Washougal	-	-	q_0[5]	0.999	0.000	0.999	0.999	1.000
Green	-	-	q_0[6]	0.999	0.001	0.998	1.000	1.000
Skamokawa	-	-	q_0[7]	1.000	0.001	0.998	1.000	1.000

Table 76. Tag loss for Chinook salmon tagged live with Floy tags and recovered as carcasses, where t_2 is the number of fish retaining two tags, t_1 is the number of fish retaining one of two tags, and t_0 is the number of fish retaining no tags, p_i is the estimate of tag loss and q_0_i is the probability of a tagged fish retaining at least one tag.

Population	*t_2*	*t_1*	*t_0*	Parameter	mean	sd	2.50%	median	97.50%
Eloch_Males	63	46	11	p[1]	0.194	0.026	0.146	0.193	0.246
Eloch_Females	79	16	10	p[2]	0.080	0.019	0.048	0.079	0.120
Green_Males	70	60	38	p[3]	0.237	0.023	0.193	0.236	0.283
Green_Females	67	19	6	p[4]	0.328	0.034	0.261	0.327	0.396
Eloch_Males	-	-		q_0[1]	0.962	0.010	0.940	0.963	0.979
Eloch_Females	-	-		q_0[2]	0.993	0.003	0.986	0.994	0.998
Green_Males	-	-		q_0[3]	0.943	0.011	0.920	0.944	0.963
Green_Females	-	-		q_0[4]	0.892	0.023	0.843	0.893	0.932

Cross Validation of AUC and Redd Estimates

Since the female per redd (FpR) estimates were based on the female abundance estimate from the JS model in previous years, we used DIC for model selection and Bayesian *P*-values to test GOF for the historic redd data sets. These include the 2003 and 2004 Coweeman data along with the 2005 and 2006 EF Lewis data. DIC favored models tst, ttt, tst, and ttt, respectively. The Bayesian *P*-values were 0.05, 0.86, 0.76, and 0.65, respectively. This indicated some lack of fit between the JS model and data for the 2003 Coweeman estimate. For the redd surveys the estimates of RpF are found in Table 77. The mean was ~1.1 (excluding the Coweeman 2010 estimate). The redd-based abundance estimate for the Coweeman was 657 compared to the mark-recapture estimate of 635. The probability that the redd-based estimate was greater than the mark-recapture estimate was 0.59, which indicated no difference between the two estimates. Although limited to one study, this suggests applying the 1.1 FpR to other populations should yield reasonable abundance estimates.

Table 77. Female per Redd (FpR) estimates for Lower Columbia Tule Fall Chinook mark-recapture studies.

Parameter	mean	sd	2.50%	median	97.50%
Coweeman 03	1.019	0.049	0.926	1.017	1.118
Coweeman 04	1.281	0.156	1.034	1.265	1.640
EF Lewis 05	1.052	0.209	0.752	1.014	1.552
EF Lewis 06	0.958	0.187	0.679	0.929	1.403
Coweeman 09	1.175	0.114	0.967	1.169	1.418
Coweeman 10	1.185	0.207	0.822	1.171	1.625
Mean 03-09	1.097	0.069	0.978	1.091	1.248
Mean 03-10	1.112	0.067	0.993	1.107	1.254

We repeated the same procedure using SL. The Green River population had a mean survey life of 3.8 days, which was lower than the SL calculated for all of the other populations. This was likely due to weir operations, which routed all fish into the hatchery released them once per week. Given that fish were delayed on average 3.5 days, we believe this altered fish behavior (necessitated accelerated spawning) yielding a SL estimate that was lower than we calculated for other populations. Therefore, the Green River data was not used in the SL comparison analysis.

The mean SL for the five populations was ~ 5.0 days (95%CI 4.66-5.40) (Table 78). The longest SL was observed in Mill Creek (5.8 days) and the shortest SL (excluding the Green) was 4.3 days for the Washougal population. Cross validation estimates (bottom of Table 78) ranged from 4.8 to 5.2 days. There was lack of fit for the Mill Creek population (Bayesian *P*-value = 0.99) but the other *P*-values did not indicate lack of fit. The *P*-values for the Elochoman, Germany, Washougal, Coweeman were 0.37, 0.37, 0.09, and 0.60, respectively. This analysis, with the exception of the Mill Creek population, suggests that reasonable estimates of abundance should be achieved when applying the mean SL estimates to other populations.

Table 78. Estimates of apparent survey life (SL) for Lower Columbia River Tule populations from weir census and mark-recapture studies in 2010.

Parameter	mean	sd	2.50%	median	97.50%
Elochoman	4.930	0.617	4.108	4.806	6.461
Mill	5.772	0.271	5.224	5.780	6.294
Germany	4.880	0.346	4.145	4.899	5.505
Washougal	4.363	0.553	3.389	4.319	5.555
Coweeman	5.060	0.222	4.619	5.062	5.477
Green	3.791	0.132	3.575	3.778	4.086
mean 2010-Green	5.001	0.193	4.658	4.988	5.412
mean 2010-Elo	5.019	0.185	4.670	5.013	5.395
mean 2010-Mill	4.808	0.232	4.393	4.791	5.306
mean 2010-Germ	5.031	0.224	4.645	5.015	5.514
mean 2010-Wash	5.161	0.198	4.815	5.146	5.599
mean 2010-Cowe	4.986	0.234	4.578	4.967	5.495

Timing

The cumulative timing for Tule Chinook salmon subpopulations are shown in Figure 3. These were not adjusted for missed counts, which occurred for some of the later spawning populations such as the Grays, Coweeman, EF Lewis, and Washougal. The first timing group consisted of populations from the LCR Coastal strata including Skamokawa, Elochoman, Mill, Abernathy, and Germany subpopulations. These populations have historically had high hatchery fractions and CWT recoveries indicated most of the hatchery fish were from the Elochoman and Big Creek hatcheries. The mean spawning date for this group was mid-September. The exception to the trend of early spawning in the Coastal strata was the Grays population where a weir removed Tule hatchery strays and timing was more influenced by Rogue River Brights, which tend to enter tributaries and spawn later than coastal Tule subpopulations. The next timing group included the Green, SF Toutle, Kalama, Coweeman, and EF Lewis populations from the LCR Cascade strata. There was more diversity in mean spawning dates within this group, which ranged from early to mid-October. The latest spawning population was the Washougal. Since peak counts were conducted in the LCR Gorge strata, we have no spawning time estimates for these Chinook populations.

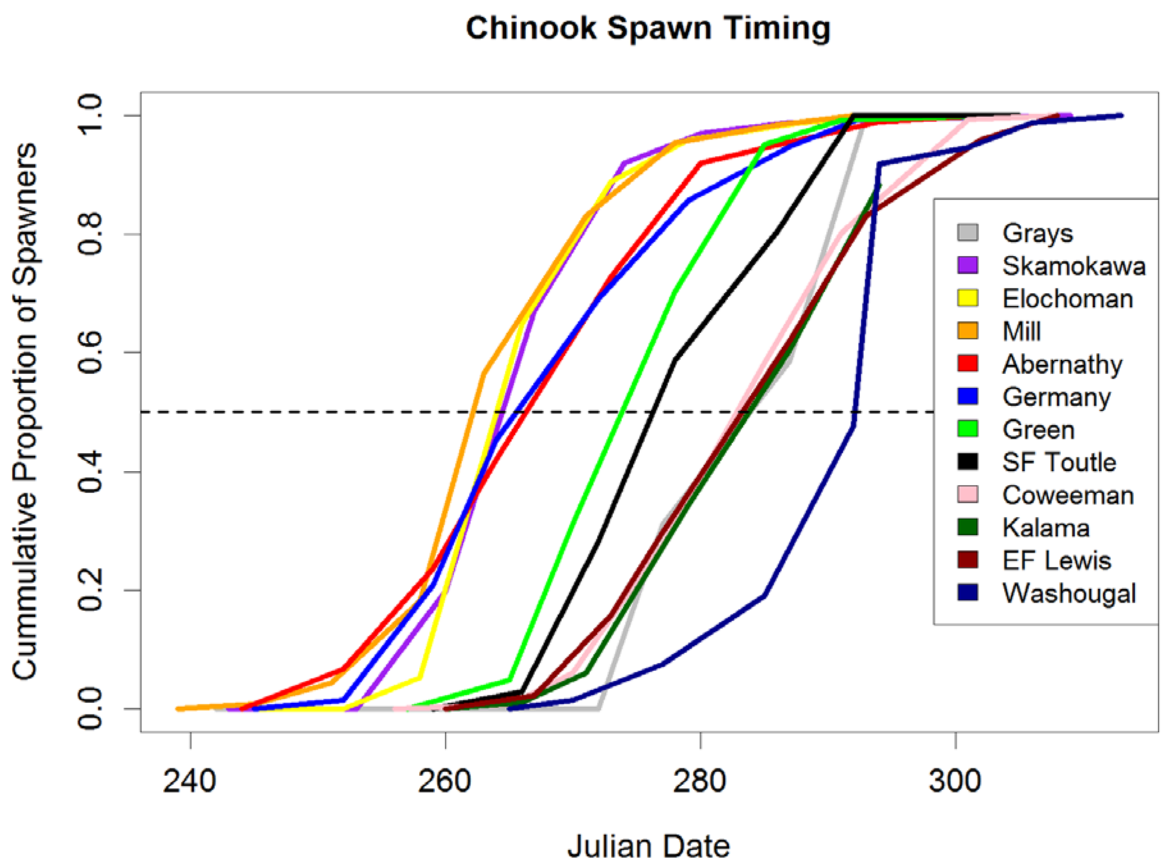

Figure 3. Tule fall Chinook salmon timing for Washington's Lower Columbia River populations based on period (weekly) counts of Chinook salmon classified as spawners.

CWT Program

The CWT recoveries in the fall of 2010 were uploaded to the RMIS system during 2011-12. The uploaded data include: 1) freshwater sport fishery recoveries on November 8, 2011, hatchery facility Chinook recoveries on January 12, 2012, and Chinook spawning ground recoveries on April 12, 2012. RMIS is a coastwide database that stores CWT tag and release data along with recovery and sampling data.

Based on carcass recoveries from stream surveys, there were no CWT recoveries in the Lower Cowlitz and Lower Gorge populations and only one in the Coweeman population (Table 26). These recoveries are consistent with the low proportion of marked fish sampled in these populations (Table 76). For some Washington Hatcheries most tags were recovered in the basin they were released, including North Toutle, Fallert, Kalama Falls, and Washougal, but for other hatcheries most recoveries occurred not in their release basins Elochoman, Cowlitz, and Little White Salmon. The CWT release with the largest number of hatchery recoveries occurred from the Washougal River (76). Fish released from Oregon hatcheries were generally recovered at a low rate in Washington, with the largest number of recoveries coming from the Big Creek Hatchery. There were a few fish tagged at distant location such as the Naselle and Umatilla Hatcheries that were recovered during our surveys. CWT data for fisheries and carcass recoveries are presented in annual reports for missing production groups (e.g. Roler 2012).

Table 79. Unexpanded CWT recoveries by population and hatchery for Chinook salmon in 2010. Gray boxes indicate CWT was recovered in the same basin as released. Spring Cr. Hatchery recoveries in the White Salmon were considered in basin recoveries.

		Release Basin															
Recovery Basin	Population	Naselle H.	CEDC Youngs H	Big Cr. H.	Eloch. H.	Cowlitz H.	North Tou. H.	Fallert H.	Kalama Falls H.	Wash. H.	BON H.	LWS H.	Spring Cr. H.	Klick. H.	Uma. H.	Blank	Total
	Grays	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Elochoman	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	2
	Mill	0	0	1	11	0	0	0	0	0	0	0	0	0	0	0	12
	Abernathy	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	3
	Germany	0	0	5	2	0	0	0	0	0	0	0	0	0	0	1	8
	Lower Cowlitz	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4
	Coweeman	0	0	0	1	0	0	0	1	0	0	0	0	0	0	9	11
	Toutle/Green	0	0	0	0	0	11	0	1	0	0	0	0	0	0	0	12
	SF Toutle	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Kalama	0	0	0	0	0	0	8	11	0	0	0	0	0	0	1	20
	NF Lewis	0	0	0	0	4	0	4	2	1	0	0	0	0	0	70	81
	EF Lewis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Washougal	0	1	0	0	1	0	0	0	26	0	0	0	0	0	0	28
	Wind	0	0	0	0	0	0	0	0	0	0	3	0	1	1	0	5
	L White Salmon	0	0	0	0	0	0	0	0	0	1	8	0	0	0	0	9
	B White Salmon	0	0	0	0	0	0	0	0	0	1	13	3	2	0	0	19
	TOTAL	1	1	6	19	9	11	12	15	27	2	24	3	3	1	81	
	%Out on Basin	100%	100%	100%	89%	56%	0%	33%	27%	4%	100%	67%	0%	100%	100%	NA	

Discussion

This is the first comprehensive estimate of tule fall Chinook salmon in Washington's portion of the Lower Columbia ESU. We used weir census and mark-recapture estimates when possible, AUC and redd-based estimates for other populations, and used peak count expansion when other methods were not possible. We used Bayesian hypothesis testing to develop mark-recapture models consistent with required assumptions for unbiased estimates. We developed statistical methods to estimate the uncertainty in the escapement and the proportion of natural and hatchery origin spawners by age. We adopted a Bayesian approach, but with vague priors, which lets the results be driven by the data.

We conducted sensitivity analysis to compare the results using our two vague prior (Jefferies and the uniform prior) for models that used proportion such as the JS model, Darroch model, estimates of proportions by sex, origin, and age, and to adjust spawner counts if AUC surveys were missed. The result of the sensitivity analysis (not shown) indicated in general the Jefferies prior yielded estimates with slightly more fish in most cases, although they were not significantly different then the estimates with the uniform prior. In the logistic regression, the vague uniform prior and the standard vague gamma (0.001,0.001) for the standard deviation of regression coefficients produced similar results, which was the same conclusion reached by Link and Barker (2006).

We used two Bayesian approaches for multi-model inference including Bayes Factors and Deviance Information Criteria (DIC) (Kass and Rafferty 1995, Spiegelhalter et al. 2002). When using DIC for JS model selection the uniform and Jefferies priors yield the same results. BF are sensitive to the choice of priors, which is often referred to as the Lindley-Bartlett paradox (Link and Barker 2006, Ntzoufras 2009). This paradox is if the priors are too vague the BF will select more parsimonious models. To avoid this in our logistic model, we implemented the priors from Link and Barker (2006) that constrained the total prior variability to be constant between models. We also used a uniform prior in estimating BF when comparing proportions (Ntzoufras 2009, Link and Barker 2010, Lodewyckx et al. 2011). There are other approaches for selecting reference priors for model selection that include posterior, fractional, and intrinsic BF that we did not use (Aitkin 1991, O'Hagan 1995, and Beregr and Pericchi 1996).

We tested if stratification of data into homogenous groups for open and closed population models was needed through testing if the recapture probability was sex or length dependent (Link and Barker 2006) by computing posterior model probabilities and BF. We also tested the various assumptions need for the pooled Petersen estimator (Schwarz and Taylor 1998) using BF calculated from the marginal likelihood of the betabinomial model (Ntzoufras 2009). In the fifteen test regarding recovery probabilities with uniform prior model weights, BF indicated there was support for models in which recovery probabilities were constant for 12 populations and were length dependent in 3 population. When recapture probability models with length as a covariate models had significant support we split the data into two groups and conducted separate abundance estimates by length. We explored the sensitivity of our prior on model selection by considering models that favored parsimonious models by developing prior model probabilities equal to exp(-k) where k is the number of covariates in the model, and complex models where the prior model probability equal to exp(k). When we used a prior that favored a more parsimonious model, the BF always favored constant recovery model. When using a prior that favored complex models, the BF (<3) found evidence of positive support of the constant recovery model for 8 populations, and favored other models in 7 cases. So, our prior model probability was somewhat sensitive to the model selection when considering prior model probabilities that favor complexity.

Due to the difficulty in estimating BF for open population models, we used DIC for model selection. Since we used vague priors we used pv instead of pD when estimating DIC because this is accurate when priors are vague and invariant to reparameterization (Gelman et al 2004). DIC is a Bayesian analog of AIC and used the Kullback-Leibler prior, which tends to favor more complex models (Link and Barker 2006, Ward 2008). Of the seven current year models and the seven historic populations used in this analysis the more complex model (t,t,t) was favored by DIC in nine models, model (s,t,t) and the model (tst) were favored in five cases. However, in most cases abundance estimates were similar indicating that JS model selection using DIC did not influence the abundance estimate in most cases. Model selection, choosing between models, is a continuing area of research for statisticians and there is no consensus among statisticians regarding model selection (Ward 2008). We used vague priors for this analysis, thus the sensitivity of our priors on had little effect on the results except when data was spare in the proportion of ages for unmarked fish. There is some hesitancy in using Bayesian methods and most of this is due to how priors are developed. We used an objective analysis where prior were vague, with the intent that they would have little influence of the posterior distribution. There is often little difference in parameter estimates between maximum likelihood and Bayesian methods as long as the posterior is data driven, which occurred in this analysis (McCarthy 2006, Kery 2010).

Weir Estimates

Although the use of weirs in this study was primarily for broodstock collection and management purposes, such as limiting the number of hatchery fish on spawning grounds, in some cases, the weirs were able to provide a census or very precise estimates of the Chinook salmon runs. We subtracted harvest estimates and estimates of pre-spawn mortality to provide an estimate of the number of adult Chinook salmon spawners. We did not have an estimate of the pre-spawn mortality and the number of transported fish that fell back and did not spawn on the Tilton and Upper Cowlitz/Cispus rivers. Therefore, in these cases, it is likely we overestimated the number of spawners.

We were not successful in operating weirs on the Washougal and Grays rivers to obtain abundance estimates of Chinook salmon. The run timing of the Chinook population in the Washougal is later than other LCR populations (Figure 3) and operating the weir later in the season makes it more susceptible to freshets and, thus, more challenging. On the Grays River, we were able to successfully operate the weir during the Tule time period. However, Rogue River “Bright” Chinook salmon from the Oregon’s Selective Area Fisheries Enhancement (SAFE) program have a broader run timing and stray into the Grays. As with the Washougal, our ability to successfully operate a weir on the Grays River after mid-October is challenging. . Appendix A provides a summary and evaluation of WDFW’s fall Chinook management weirs in 2010.

Zhou (2002) and Murdoch et al. (2010) indicated that Chinook salmon carcass recoveries were sex and length dependent. However, their study basins are generally larger and have a different flow pattern than many streams in the Lower Columbia region. We used a logistic regression analysis to test if carcass recoveries were sex or length dependent in two basins with weirs (Elochoman and Green). Bayes Factors favored the null model, which indicated sex and length were not significant factors in predicting the recovery of adult salmon carcasses (> 60 cm). This is an important finding because in these basins representative age structure and CWT recoveries can be obtained from well designed carcass recoveries program. Since only a few basins are larger than these basins (i.e. Cowlitz and NF Lewis) unbiased carcass recoveries may occur in most other Lower Columbia River populations, but this needs further exploration.

Closed Population Estimates

We developed two closed population estimates for adult Chinook salmon in the Green and Elochoman rivers, when comparing these estimates using the Darroch model, they were not statistically different from the adjusted weir count. Using the “pooled” Petersen model, the estimate for the Green was not statistically different than the census count, but the Petersen model estimate on the Elochoman was significantly different than the weir count because the weir was removed to prevent damage during the October 10th freshet and not reinstalled. Assuming the pooled Petersen estimate on the Elochoman is correct, the bias due to removing the weir was 31 fish or -2.4% relative bias. Schwarz and Taylor (1998) indicate that the following assumptions must be met to provide a consistent estimate of abundance: 1) Tag Loss - there is no mark loss, 2) Handling Mortality - there are no marking effects, 3) Tag Reporting - all marked and unmarked fish are correctly identified and enumerated, 4) Closure - the population is closed, and 5) Equal Capture - all fish in the population have the same probability of being tagged; or all fish have the same probability of being captured in the second sample; or marked fish mix uniformly with unmarked fish.

We addressed tag loss by adding a permanent secondary mark, which was a shaped opercle punch that was rotated weekly. All Chinook were handled carefully to minimize mortality, but even if it did occur it did not affect our results since the population was closed and we estimated abundance at the time of tagging. All surveyors were trained and carefully inspected all carcasses for marks, so we believe there was high probability that all marked and unmarked fish were correctly identified and enumerated. The weirs were fish tight so we meet the closure assumption except at the end of the Elochoman study. We conducted Bayesian hypothesis testing for the equal capture assumption. The results from our logistic model indicated that sex and length did not influence recapture probabilities, and Bayes Factors favored models with a constant marked proportion of carcasses by location and by recovery period, suggesting the pooled Petersen estimator was appropriate. Finally, we compared the weir census and closed population estimates. These results support that our estimate was accurate and unbiased with the exception of the loss of the weir at the end of the Elochoman study.

Parsons and Skalski (2010) did not recommend the pooled Petersen estimate because they believed that it was unreasonable that the assumptions could be met. However, under these conditions based on assumption testing and empirical results, the Petersen estimator can provide accurate and unbiased estimates of adult salmon. In both cases, the Petersen estimate had less bias than the Darroch estimator, which was recommended by Parsons and Skalski (2010). We also note that matrix algebra or least squares methods are commonly used to provide estimates of the Darroch model (Seber 1982, Arnason et al. 1996), but for our data this method did not provide admissible estimates because there were negative abundance estimates in the recovery stratum and the probability of capture for some stratum was outside of the plausible range between 0 to 1. Extensive pooling was needed to obtain an admissible estimate, which defeats the purpose of the Darroch estimator. This was due to the linear dependency in the row (Arnason et al. 1996, Schwarz and Taylor 1998). Our Bayesian methods, naturally constrained the stratum population estimates to be positive and the capture probabilities from 0 to 1, and with our data the Bayesian approach was more robust than least squares or matrix algebra methods.

tGMR estimates

The first two assumption of no tag loss and handling mortality are met in tGMR studies because genetic marks are permanent (Lukcas and Burnham 2005) and genetic tissue is obtained from carcasses which are dead and non-lethal from juveniles as they emigrate from the study area. The closure assumption requires that within the study period there are no additions to the population through births or immigration and no deletions through death or emigration. Regarding juveniles, it is unlikely that immigration occurred since our trap was located at Rkm10, which would be a substantial upstream migration for juvenile Chinook salmon for other populations. In this study, the closure assumption is not testable and likely violated by adults failing to produce offspring; however, the Petersen estimator is unbiased with respect to the population at time of tagging if mortality was random (Seber 1982; Williams et al. 2002). Laboratory procedures and simulation results, reported in Rawding et al. (2013), suggest the tGMR estimator would not be biased with regard to correct identification and reporting of tagged/marked fish, which is also supported by parentage algorithms power analysis (Wang 2004; Kalinowski et al. 2007).

We implemented a representative carcass sampling design to obtain a sample of parents to meet the fifth assumption of equal capture probabilities, which is considered the Petersen estimator's "Achilles heel" (Arnason et al. 1996). Although we cannot formally test for equal probability of capture in the first event we believe violation of this assumption are minor. The few published carcass selectivity studies for Chinook salmon indicate carcass recoveries appear biased toward larger and possibly more productive fish, which would cause an under estimate in abundance using tGMR with proportional sampling in the second event (Zhou 2002; Murdoch et al. 2010).

However, our logistic regression analysis of recapture probabilities from weirs on the Elochoman and Green rivers suggested that sex and length did not influence carcass recovery. In addition, the bias in the abundance estimate due to size selectivity in carcass recoveries is likely to be small because of the weak relationship between reproductive success and body size for salmon (Dickerson et al. 2005, Williamson et al.2010; Richard et al. 2013, Rawding et al. 2013). It is likely that carcass recoveries were selective based on timing due to incomplete surveys at after peak spawning due to environmental conditions. BF for the regression model suggests support for models with age as a covariate. Given twelve of fifteen logistic regression models including the two weir models supported that carcass recovery was not sex or length selective, it is likely that our carcass recoveries (marks) in the Coweeman were not selective by age or sex. Further evidence that the tGMR model was relatively unbiased is that the redd and AUC estimates are not statistically different than the tGMR estimates. Our preliminary conclusion is the tGMR estimate is a consistent estimate of abundance in the Coweeman. However, the tGMR estimate includes Chinook salmon jacks, which have a reproductive success of ~ ¼ that of adult males (Schroeder et al. 2012) and we have not developed methods to partition the adult and jack estimates. We can more fully explore the tGMR analysis when COLONY runs are complete.

Open Population Estimates

In the JS model, all parameters are not identifiable including the probability of capture (p) in the first and last periods. Therefore to obtain a salmon population estimate the p's were modeled as $p_2 = p_1$, and $p_s = p_{s-1}$ unless survivals were modeled as a constant (Schwarz et al. 1993). Also the probability of entry (b^*_i) must be constrained to sum to one. The recruitment parameters (B^*) at the beginning and end of the sampling periods cannot be estimated without further assumptions. At the start of the study, the JS model is not able to directly estimate births (B_0) but Schwarz et al. (1993) assume that a well-designed mark-recapture study should commence before a significant number of fish enter the stream or spawning area; thus $B_0 = N_1$ is a reasonable assumption in our study since we started survey before spawning started. Also if studies extend to the end of recruitment, Schwarz et al. (1993) suggest that net births (B_{s-1}) should approach zero, with little effect on the population estimate.

Assumptions to recruitment between sampling occasions are needed to estimate annual salmon escapement from the JS model. One assumption is that recruitment takes place at the mid-point (Sykes and Botsford 1986); the adjustment factor for this assumption is (1/ sqrt(φi)), where φi = the probability that an animal alive at sampling occasion i will be alive at sampling occasion (i+1). An alternative assumption is uniform recruitment (Crosbie and Manly 1985; Schwarz et al. 1993) with an adjustment factor of (log φi / (φi -1)). Schwarz et al. (1993) conducted a sensitivity analysis to these and other distributions of adult recruitment. Adjustment factors are similar when survival is high because most fish survive to the next sampling occasion. When survival is low, the adjustment factors varied considerably. Schwarz et al. (1993) noted the actual distribution of recruitment is unknown and care should be taken in choosing a recruitment adjustment factor. In their analysis, the performance of the mid-point and uniform adjustment factors was similar and the uniform recruitment distribution was used in this analysis.

The JS model based on carcass tagging is not often used to estimate salmon escapement. Among the different carcass tagging mark-recapture models, the JS model is accurate, precise, and robust method for estimating salmon spawning escapement (Boydstun 1994) but may be slightly biased due to heterogeneity, no abundance estimate available for the last period, confounding parameters during the first and last period, and assumptions about the pattern/distribution of the arrivals within a period (Schwarz et al. 1993, Law 1994). However, Schwarz (1993) and Law (1994) found the Jolly-Seber model was robust to these violations through simulations. Sykes and Botsford (1986) found no difference in the adult Chinook salmon abundance estimate based on carcass tagging when compared to a census count at the weir on Bogus Creek, California. Five assumptions of the Jolly Seber model must be met in order to obtain unbiased population estimates from the model (Seber 1982) are: 1) Equal Catchability - every animal in the population whether tagged or untagged, has the same probability of being caught (p_i) in the i^{th} sample given that it is alive and in the population when the sample is taken, 2) Survival- every tagged animal has the same probability of surviving (φ_i) from the i^{th} to the $(i+1)^{th}$ sample and of being in the population at the time of the $(i+1)^{th}$ sample, given that it is alive and in the population immediately after the ith release, 3) Handling Mortality - every animal caught in the i^{th} sample has the same probability of being tagged and returned to the population, 4) Tag Loss & Reporting - tagged animals do not lose their marks and all marks are recognized on recovery, and 5) Instantaneous Sampling- all samples are instantaneous, i.e., sampling time is negligible and each release is made immediately after the sample.

With respect to our study the JS model requires that all fish have identically independently distributed survival and capture probabilities, which are the equal catchability and survival assumptions. We addressed this through the use of a logistic regression model to develop homogeneous groups with respect to sex and length because these two covariates are known to influence recapture (survival and capture) probabilities. To help meet the equal capture assumption of tagged and untagged fish, we placed carcasses in flowing water to ensure mixing of fish, and placed opercle tags on the inside of the operclum so surveyors would not be attracted to the tagged fish at a higher rate than untagged fish. The survival assumption may be violated when tagging live fish because survey life or residence time is positively correlated with date of entry (Schwarz et al. 1993), but this is not the case for carcasses. However, Chinook salmon carcasses decompose over time and may be available for capture for three weeks or more (Law 1994).To address the potential dissimilar survival of old and new carcasses assumption, we did not tag decomposed carcasses as they may have a lower probability of “surviving” to the next period. These decomposed fish had their tails cut off and were treated as loss on capture. In addition, all tagged fish that were recaptured also had their tails cut and were also treated as loss on capture. Finally, we used a Bayesian GOF test, similar to the GOF test in the program RELEASE, to test the combined equal capture and survival assumptions.

In the Washougal, the GOF test indicated a lack of fit and this is related to different survival and captures probabilities for some releases, which we will illustrate with the Washougal 2010 m-array (Table 80). Our sample design calls for all recaptured fish to be treated as loss on capture. Therefore, if capture probabilities are consistent and high for every sampling occasion and all recaptures are lost on capture, the resulting m-array has a strong diagonal component (light gray cells), with few fish recaptured after the first recapture event. However, during a flood event (recovery period 4) the higher stream flows reduce capture probabilities and increase the proportion of marked fish surviving to subsequent sampling occasions. This problem is compounded in carcass tagging because all recaptured fish are not released (lost on capture) to meet the survival assumption, and don’t have the chance to survive to an additional period. In fact, for release period four more fish were recovered later than the first recovery period than for all other releases, which, in part, leads to the lack of fit. The affect on the abundance estimate from the lack of fit is usually that the estimate is unbiased but the confidence interval coverage is biased low (Nichols 2005), but may be explored through simulations (Schwarz et al. 1993, Law 1994).

Table 80. The m-array for 2010 Washougal Chinook salmon CJS model. Cells are the number of released marked individuals recaptured by period and the last column is the number of marked individuals that were never recovered.

Release Period	Recovery Period 1	2	3	4	5	6	
1	1	0	0	0	0	0	14
2	0	9	0	0	1	0	46
3	0	0	220	3	5	2	115
4	0	0	0	32	31	23	320
5	0	0	0	0	104	11	141
6	0	0	0	0	0	39	151

The third assumption requires that there is no handling mortality, which is true for carcasses. The fourth assumption is that there is not tag loss and all tags are recognized and reported on recovery. We assessed tag loss through double tagging experiments, and surveyors followed protocol to inspect the inside of both opercula from every carcass to ensure all tagged and untagged fish were correctly identified and reported. However, if tag loss is a major concern (Arnason and Mills 1981, MacDonald et al. 2003) we recommend the use of Jolly-Seber models that account for tag loss (Cowan and Schwarz 2006). Sampling was not instantaneous but usually occurred during a single day within a weekly period. Schwarz et al. (1993) indicated that in these cases, this violation of instantaneous sampling is not believed to be a serious violation.

PCE Estimates

When weirs and mark-recapture were not available we used alternate methods to estimate abundance. WDFW has been using PCE factors for over 40 years because they are the most cost-effective method for estimating abundance. The PCE factors for the Wind, Little White Salmon, and White Salmon rivers were based on a single study from 20 to 40 years ago depending on the basin. Except for some concerns regarding the GOF test in the JS abundance estimates from carcass tagging, the PCE factors provide a statistical based population estimate. The following assumptions are used in the PCE method: 1) the peak day of abundance is known and the survey takes place on the peak, 2) if the entire spawning distribution is not surveyed, the proportion of fish used in the index or indices is similar to that of the years used to develop the peak count expansion factor, 3) observer efficiency is similar in all years, and 4) the proportion of fish observed on the peak day is similar over all years. Since the expansion factors are old there are concerns about the similar proportions of spawners using the index reaches, changes in proportion of carcasses and live fish present on the peak day due to changes in run timing, and the lack of replication to better estimate the variability in the peak count expansion factor. Representative biological data and CWT samples do not usually occur using the peak count method because it relies on a single or a few surveys near the peak, when the population may be comprised of mixture of different populations (i.e., hatchery & natural origin) with different timing. Since other methods (AUC or redd counts) require surveys from the beginning to the end of spawning, more representative biological and CWT data will be collected with these methods.

Redd and AUC Estimates

We successfully implemented concurrent census or mark-recapture estimates and periodic counts of spawners in Elochoman, Germany, Mill, Green, Coweeman, and Washougal for salmon populations draining areas from 25 to 250 sq miles to estimate the mean of apparent survey life. We did not use the Green population because the survey life was less than the others due to holding of fish in the hatchery for up to a week before release. Following the recommendation of Gallagher et al. 2010, we used annual estimates of apparent survey life to account for potential differences in environmental conditions and observer efficiency that may affect survey life. Since in 2010 we had only one population where we had concurrent mark-recapture and redds counts (Coweeman), we used five previous redd and mark-recapture estimates on the Coweeman and EF Lewis to estimate mean redds per female. Our CV-1 analysis supported that SL and FpR are relatively consistent across populations thus application to other populations should yield consistent results

Proportions

We used the binomial distribution to estimate the proportion of unmarked Chinook salmon adults from carcass recoveries. This pMark estimate is close to the actual estimate of pHOS because almost 98% of the hatchery Chinook salmon are mass marked based on sampling of hatchery juvenile prior to release. We used a pooled estimate except in the Washougal, Mill, and Germany basins because recapture probabilities were different by length. We used the multinomial distribution to estimate the proportion of adult Chinook salmon by age. There were different length at age relationships by sex and river (data not shown). These should be investigated using approaches such as comparing scale and CWT ages from the same fish, and examining the use of mixture models (Marin and Robert 2007) with all lengths and known ages to more accurately estimate proportions by age.

Recommendations

Over the last decade or so there has been a significant shift in the monitoring of Chinook populations in the LCR to estimate VSP parameters (McElhany et al. 2000), and other important management indicators (Rawding and Rogers 2013). While great progress has been made in the LCR region, opportunities remain for improvement of estimates of Chinook salmon VSP and indicator parameters. Therefore, we recommend the following: 1) a radio tag study to assess pre-spawn mortality and fall back in Tilton and Upper Cowliz/Cispus River, 2) upgrade of the Modrow weir and trap facility on the Kalama River for broodstock management and sampling, 3) change of weir facilities to more flexible designs such as resistance board weirs that could allow for more successful operation during freshets, 4) change of weir locations (Grays River) if possible to be more effective at trapping during fall freshets, 5) as funding allows transition away from PCE estimates on the Wind, Little White Salmon, and the Big White Salmon rivers and more representatively sampling of carcasses for biological and CWT data, 6) continue to improve current modeling that estimates abundance and proportions by exploring covariates, hierarchical and space-state models, 7) scan Chinook salmon for CWT at weir locations to improve CWT recovery rates rather than relying only on spawning ground recoveries.

Acknowledgements

We compiled a single fall Chinook salmon monitoring and CWT recovery report because all the information from study designs to results are in a single reference and it was cost-effective to do this. The major funders for Chinook salmon monitoring and reporting are the Bonneville Power Administration (BPA), the National Marine Fisheries Service (NMFS) through the Mitchell Act and Pacific Coast Salmon Recovery Fund (PCSRF). The later was administered through the Washington RCO. However, we received significant funding through the Pacific Salmon Treaty (PST), specifically from the Coded Wire Tag Implementation Team (CWTIT), Chinook Technical Committee (CTC) and the Southern Fund. This PST funding was used for study design development, improvements in WDFW's internal CWT database, and purchase of CWT wands used to recover CWT on spawning ground surveys, and the development and implementation of the trans-generational mark-recapture (tGMR) program on the Coweeman River along with Coweeman spawning ground surveys. Tacoma Public Utilities provided the number of Chinook salmon released into the Tilton and Upper Cowlitz/Cispus Rivers. We adapted WinBUGS code for the logistic regression model from Link and Barker (2010) and the CJS model from King et al. (2010). Martin Liermann (NWFSC) provided assistance with the code for the Jolly-Seber model as parameterized by Schwarz et al. (1993). We thank Sewall Young of the WDFW Genetics Unit for genotyping and parental assignments analysis; Erik Kraig for CRC salmon harvest estimates; Bob Woodard, Steve VanderPloeg, Michelle Groesbeck, Danny Warren, Ari Storm, and Gill Lensgrav from the Biological Data Systems Unit for assistance with WDFW's CWT, Spawning Ground Survey, and the Juvenile Outmigrant database development and management; staff from the WDFW Coded Wire Tag laboratory for decoding CWTs; John Sneva and Lance Campbell from the WDFW Age laboratory for assigning ages to readable scales; (names) Region 5 hatchery staff for operation and assistance in sampling at weirs, and Vancouver biologists and technicians for implementation of the study design, data collection, and data entry. We would to thank Rick Golden (BPA), and Rob Jones (NOAA) and Scott Rumsey (NOAA), and various committees of the PST for their support of this project. Mention of trade names or commercial products does not constitute endorsement for their use.

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Appendix A. LCR Chinook Management Weirs

Appendix A. LCR Chinook Management Weirs

Lower Columbia River Chinook Management Weirs –

2010 Summary and Evaluation

Jeremy Wilson and Bryce Glaser
WDFW Region 5 Fish Management

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Introduction

Chinook salmon (*Oncorhynchus tshawytcha*) in the Lower Columbia River (LCR) Evolutionary Significant Unit (ESU) were listed for protection under the Endangered Species Act (ESA) in 1999. In a recent five-year review, the National Oceanic and Atmospheric Administration (NOAA) Fisheries concluded that these fish should remain listed as threatened under the ESA (NOAA 2011). The LCR Chinook Salmon ESU is composed of spring and fall populations split between the states of Washington and Oregon (Myers et al. 2006).

The Lower Columbia Fish Recovery Board's (LCFRB) Recovery Plan (2010) describes a recovery scenario for Lower Columbia River Chinook salmon. The plan identifies each population's role in recovery as a primary, contributing, or stabilizing population generally based on its baseline viability level and the desired recovery viability level. In 2007, the Hatchery Scientific Review Group's (HSRG) memo to the Columbia River Hatchery reform Steering Committee stated that one of the key factors limiting recovery of naturally spawning populations is interaction with hatchery-origin fish on the spawning grounds. The HSRG recommended management targets of less than 5% hatchery-origin spawners for primary populations and less than 10% hatchery-origin spawners for contributing populations without integrated hatchery programs. For populations with integrated hatchery programs, the goal is less than 30% hatchery-origin spawners (HSRG 2009).

In an effort to reduce the proportion of hatchery-origin spawners (pHOS) to meet HSRG standards and improve abundance estimates to meet NOAA's accuracy and precision guidelines, WDFW began installing and operating river-spanning weirs for fall Chinook management in LCR basins in 2008. This coincided with the phased implementation of LCR fall Chinook mass marking (adipose clipping of hatchery production) which began in 2005 and was fully realized in 2012 with all age 2-6 year old returns being marked. The Grays River Weir was the first LCR weir focused on fall Chinook management, which was installed in the fall of 2008. In the fall of 2009, the Elochoman River Weir was added, followed by the Green in the fall of 2010.

This appendix reports on the weirs operated in the fall/winter of 2010 on the lower Grays, Elochoman, and Green rivers. For all three weir locations, operations are primarily focused on fall Chinook abundance monitoring, management, and broodstock collection (Green River only); however, information gathered from other returning salmonids (chum, coho, and steelhead) is also used to improve monitoring and management when possible.

At all three locations removal of known hatchery fish (identified by a fin mark) is utilized as a tool to promote recovery of wild stocks and meet management guidelines and objectives. The proportion of hatchery fish removed at each weir varies to meet management goals and objectives in the basin and, in some cases, is used to evaluate hatchery reform actions. WDFW annually conducts fall Chinook spawning ground surveys on the Grays, Elochoman, and Green rivers. Staff funded by these weir projects assist in these surveys to collect data necessary to estimate total abundance of fall Chinook populations, estimate proportions of hatchery and natural-origin Chinook, and evaluate weir effectiveness.

These projects have three objectives: 1) to complement existing adult salmonid monitoring efforts by developing accurate and precise estimates of total abundance, especially for fall Chinook salmon, 2) to promote recovery of fall Chinook salmon populations by meeting management guidelines/objectives for control of hatchery-origin Chinook allowed to spawn naturally and 3) for collection of hatchery broodstock in the Green River for WDFW's North Toutle Hatchery.

Methods

Study Site

The LCR Chinook salmon ESU extends from the mouth of the Columbia up to and including the Big White Salmon River in Washington and Hood River in Oregon, and includes the Willamette River to Willamette Falls, Oregon. Within this ESU, there are a total of 13 Washington populations, 8 Oregon populations, and 2 populations (Lower and Upper Gorge) that are split between the states. As of 2010, WDFW has installed temporary weirs in three of these populations in Washington for the purpose of fall Chinook management- the Grays/Chinook population, the Elochoman/Skamokawa population, and the Toutle Population (Green River) (Figure 1). The Grays River/Chinook population is comprised of two subpopulations: Grays River and Chinook River and is identified as a contributing population with pHOS target of less than 10%. The weir is located on the lower Grays River and is therefore only controlling pHOS within the Grays basin. The Elochoman/Skamokawa Chinook population is comprised of two subpopulations: Elochoman River and Skamokawa Creek and is identified as a primary population with a pHOS target of less than 5%. The weir is located on the lower Elochoman River and is only controlling pHOS for the Elochoman River subpopulation. The Toutle River Chinook population is made up of three subpopulations within the basin: Green River, SF Toutle River, and NF Toutle River. The Toutle population is identified as a primary population with a pHOS target of less than 5%. The weir is located on the lower Green River and is therefore only controlling pHOS for the Green River subpopulation.

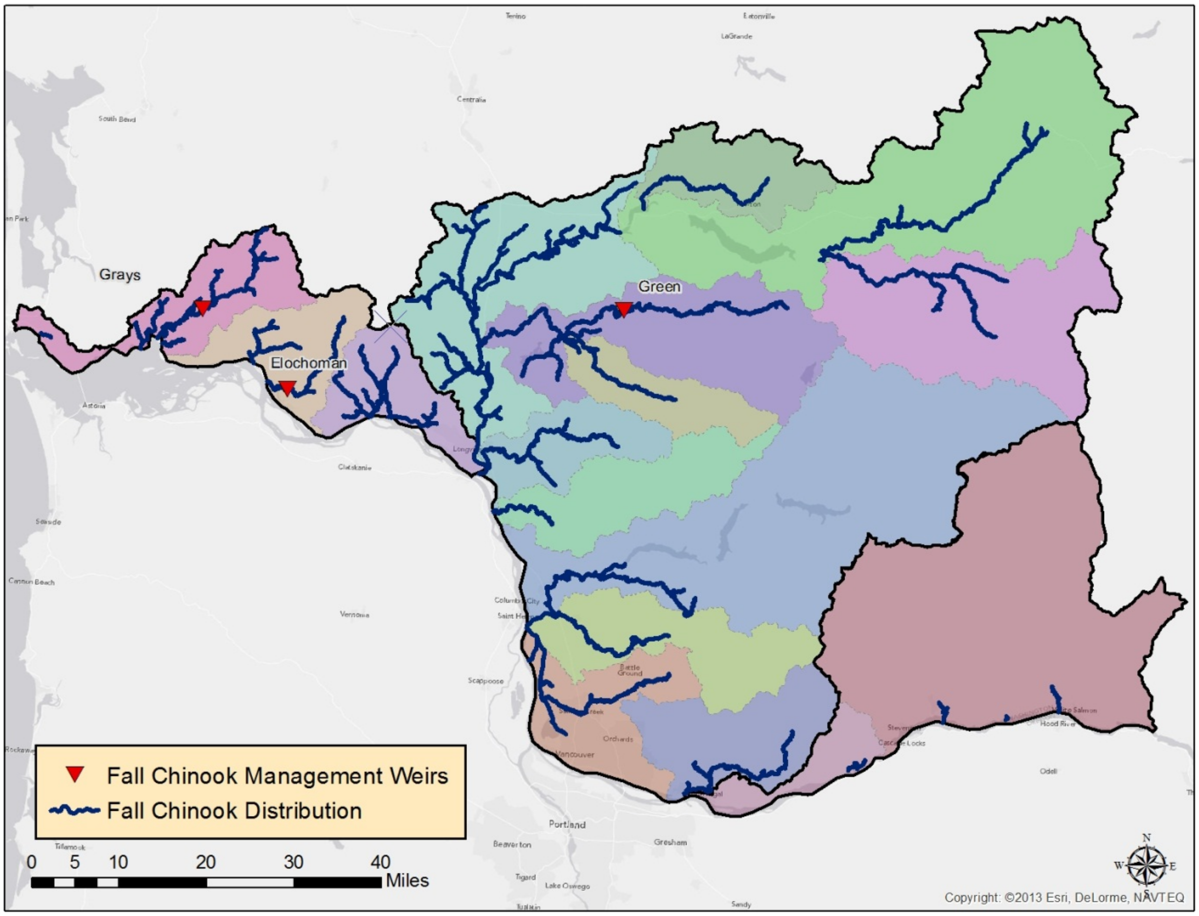

Figure 1. Location of weirs used for fall Chinook management in the Lower Columbia River.

Fish Capture

River-spanning weirs were installed in the three LCR tributaries in the fall of 2010. Weir designs varied based on the available infrastructure and goals of the specific weir. In general, three weir designs were used: a fixed panel design, a resistance board design, and a hybrid fixed/resistance design. Fixed panel weirs have been used for decades in LCR tributaries to meet hatchery broodstock needs. Fixed panel weirs can be highly effective at low, constant flows especially when paired with a concrete sill. This design was used in the Elochoman River with an existing concrete sill and trap box at river kilometer (rkm) 4.35. A hybrid resistance board/fixed panel design utilizes fixed wooden panels on the perimeter and a floating resistance board section constructed primarily of PVC pipe in the center. This design was used in the Green River with an existing concrete sill and fish ladder which diverted fish into the N. Toutle Hatchery adult holding pond at rkm 0.60. A resistance board design utilizes a floating resistance board section made of PVC pipe river-wide. It is typically anchored using duckbill anchors and cables (Figure 2). This design was used in the Grays at rkm 16.42. All weirs had 3.8 cm spacing to limit any size bias.

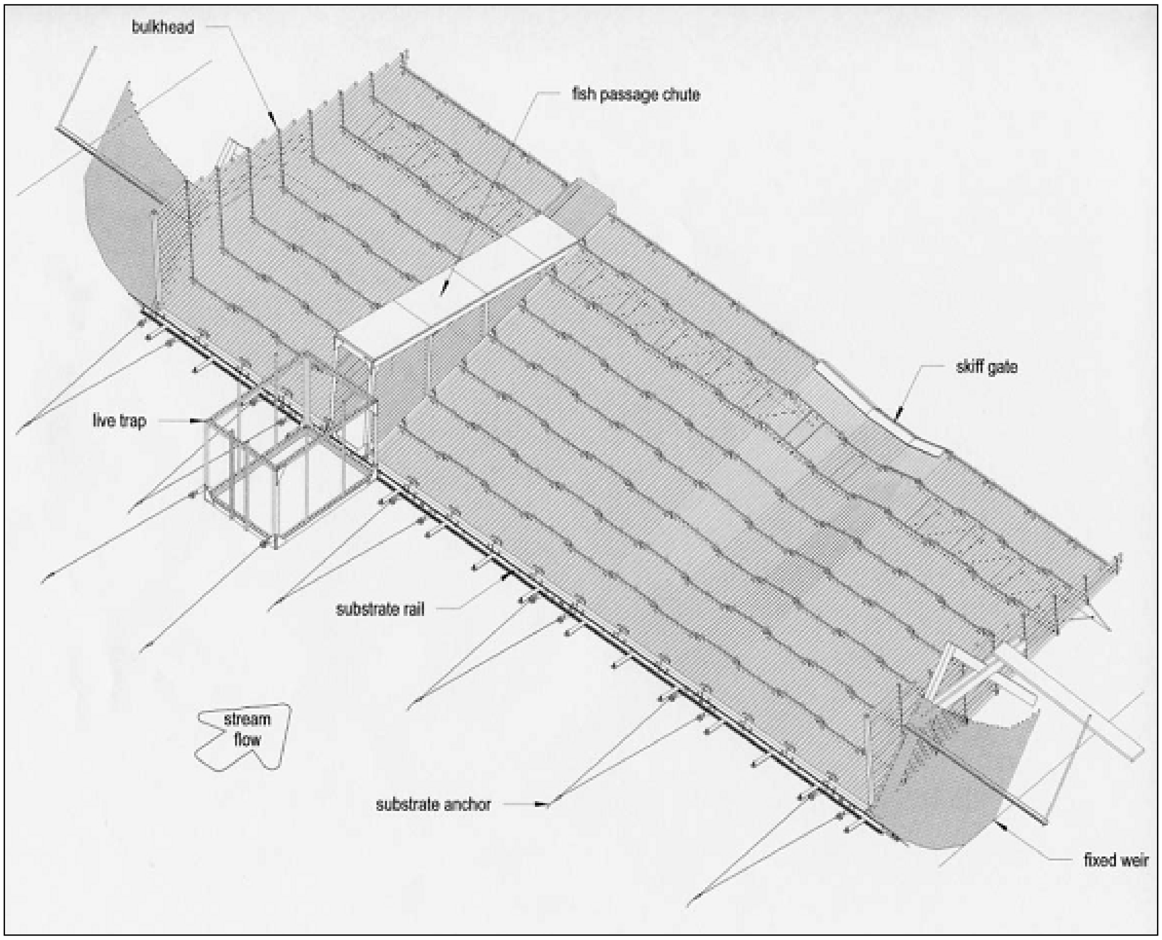

Figure 2. Schematic of a Resistance Board Weir (Stewart 2003).

Weir Operation and Sampling Protocols

Weirs and traps were staffed continuously while installed and the trap box was checked daily (multiple times per day when necessary). Close attention was paid to the recruitment of fish into trap boxes and the accumulation of fish below the trap. When the abundance of salmonids exceeded the ability of staff to efficiently work through fish, modifications were made to trapping protocols to facilitate passage without handling. This was accomplished by opening the upstream gate on the trap box and allowing fish to pass through without handling or submerging a panel section of the resistance weir to allow fish passage around the trap box.

Stream flow and weather forecasts were monitored closely to ensure the well-being of captured fish in the live box. The Washington Department of Ecology (WDOE) operates telemetry stream flow gauges that provide near real-time information on stream flows. Stream flow and weather forecast information, and ultimately direct observation, determined when flows began to limit accessibility to the trap box. When these conditions were encountered, the trap box was opened on both the upstream and downstream end to allow direct passage through the trap.

Marking/tagging of fish combined with stream surveys provided a means for estimating abundance and weir efficiency when fish were allowed through the trap unsampled and/or high flows compromised the ability to trap fish at the weir.

Adult fall Chinook captured at each weir were sampled and marked/tagged prior to release above the weir to evaluate weir efficiency and generate population estimates. Marking/tagging was coordinated with spawning ground surveys to re-sight/recover these marks. Independent estimates of spawner abundance were made for fall Chinook via mark/recapture, redd count expansion and/or Area-Under-the Curve (AUC) methods for comparison to weir estimates. All adult salmonids that were bio-sampled, except those able to be retained in sport fisheries upstream of weir sites, were anaesthetized (MS-222) prior to handle/tagging at the weir. All anaesthetized fish were allowed to fully recover before releasing upstream of the weir. Table 1 outlines the planned disposition by species and origin at the Grays, Elochoman, and Green weirs in 2010.

Table 1. Planned disposition of adult salmonids by species and origin for the Grays, Elochoman, and Green weirs in 2010.

Species	Origin	Grays	Elochoman	Green
Fall Chinook	Unmarked	U	U	1 in 2 U*
	Marked	R	R	1 in 3 U
Coho	Unmarked	U	U	U*
	Marked	U	R	R*
Chum	Unmarked	U	U	U
	Marked	U	U	U
Steelhead	Unmarked	U	U	U
	Marked	U	U	R

Unmarked fish are assumed to be of natural origin (NOR) and marked fish are assumed to be of hatchery origin (HOR)
U=Upstream, R=Removed
* denotes in excess of weekly broodstock needs
N. Toutle (Green) has integrated fall Chinook and coho programs – any unmarked Chinook and coho not placed upstream were taken for brood.
All LV-clipped fall Chinook were removed at all weirs.

Data Analysis

Weir data analysis was conducted by adding additional code to the analysis done in the main body of this report. A Bayesian framework was utilized using WinBUGS (Spiegelhalter et al. 1999) from R using the R2WinBUGS package (Sturtz et al. 2005). We use pMark as a surrogate to pHOS in the main body of the report and in this appendix. pMark is the proportion of marked spawners based on presence of a fin clip and/or coded wire tag while pHOS is the proportion of hatchery-origin spawners. In order to accurately calculate pHOS, you must account for unmarked hatchery releases which is typically ~2% of mass mark releases from facilities that have mass marked releases. However, not all returning LCR fall Chinook hatchery releases were fully mass marked in 2010 (5 year olds from N. Toutle Hatchery and Washougal Hatchery were not mass marked). Therefore, we did not make this adjustment in 2010. We calculated adult weir capture efficiency, estimated pMark without hatchery Chinook removals, change in pMark, and the proportion of Chinook spawning occurring downstream of the weir site. Tables 2 and 3 outline the summary statistics, parameters, and equations used to calculate these metrics.

Table 2. Summary statistics used from weir reporting.

Statistic	Definition
N_{aw}	Total Chinook abundance above the weir site
N_{bw}	Total Chinook abundance below the weir site
W_{up}	Adults passed upstream at weir
W_{hrem}	Hatchery adults (>59 cm) removed at weir
W_{wrem}	Unmarked adults taken for brood or trap mort
H_{swim}	Chinook swim-ins to hatchery facility above weir
pMark	Proportion of marked spawners based on the presence of an adipose and/or ventral fin clip and/or coded wire tag (CWT)
MS_{aw}	Marked spawners above weir

Table 3. Derived parameters for weir reporting.

Parameter	Definition/Equation
W_{eff}	Weir Capture Efficiency $((W_{up}+W_{hrem}+W_{wrem})/(N_{aw}+W_{hrem}+W_{wrem}+H_{swim}))$
nwpMark	Estimated pMark without hatchery removals. $((pMark_{aw}+W_{hrem})/(N_{aw}+W_{hrem}+W_{wrem}))$
cpMark	Estimated change in pHOS from removal of hatchery fish at the weir site. *nwpMark-pMark*
% spbw	Proportion of the spawning population that spawned downstream of the weir site. $N_{bw} / (N_{bw} + N_{aw})$

We provide estimates of age structure by mark type for Chinook removed at each of the weirs based scale readings.

Results and Discussion

The three weirs were installed prior to the start of fall Chinook upstream migration and were operated throughout the migration period. A total of 59, 4,597, and 3,443 Chinook were captured at the Grays, Elochoman, and Green weirs, respectively. Table 4 lists the catch at each weir site by species, origin, and disposition. Weir totals represent total number of fish that were captured at each weir site. Total spawning escapement may be more or less than weir totals depending on weir capture efficiency, sport harvest above weir sites, and pre-spawning mortality. Escapement reported in the main body of the report is less removal from fisheries and/or pre-spawning mortality.

Table 4. 2010 Weir capture totals by location, species, origin, and disposition.

		Number Trapped (Males/Females/Jacks)			
Species	Mark	Grays	Elochoman	Green	Disposition
Chinook	LV or ADLV*	45 (19/16/10)	13 (7/3/3)	0	Removed
	AD only	11 (2/6/3)	3567 (1938/1621/8)	505 (354/127/24)	Removed
	AD only	0	864 (467/395/2)	1593 (1048/499/46)	Released upstream
	AD only	0	0	730 (379/325/26)	Held for Brood
	AD only	0	0	1 (1/0/0)	Trap Mortality
	None	3 (0/3/0)	153 (93/57/3)	409 (183/224/2)	Released upstream
	None	0	0	204 (77/125/2)	Held for Brood
	None	0	0	1 (0/1/0)	Trap Mortality
Coho	AD	0	0	8431 (4709/3688/34)	Removed
	AD	341 (194/139/8)	53 (27/26/0)	0	Released upstream
	AD	0	0	1203 (550/651/2)	Held for Brood
	None	86 (54/30/2)	14 (8/6/0)	621 (343/277/1)	Released upstream
	None	0	0	89 (44/45/0)	Held for Brood
Steelhead	AD	0	0	26 (17/9/0)	Removed
	AD	1 (1/0/0)	20 (9/11/0)	4 (2/2/0)	Released upstream
	AD	0	0	22 (8/14/0)	Trap Mortality
	None	2 (0/2/0)	6 (1/5/0)	0	Released upstream
Chum	None	2 (0/2/0)	0	0	Released upstream

AD only = Fish with an adipose fin clip – indicates hatchery-origin
LV or ADLV = Fish with a left ventral or a left ventral and an adipose fin clip – indicates hatchery-origin Select Area Brights (SABs) from Oregon Select Area Fisheries Enhancement (SAFE) releases.
None = All fins intact – indicates natural-origin or a fish that was not mass marked (not all LCR 5 year old fall Chinook hatchery releases were mass marked)

It should be noted that fish called jacks in the weir totals table (Table 4) are based off a fork length cutoff. Chinook 60 cm and larger are considered adults while Chinook <60 cm are considered jacks. All abundance, pMark, and weir efficiency estimates in this report are based off of this fork length cutoff.

Grays River Weir

The Grays River weir was initially established and operated in the fall of 2008 using Pacific Coast Salmon Restoration Fund (PCSRF) dollars; in 2009, funding to install and operate the weir shifted to Mitchell Act MER dollars. The Grays River Weir is a hybrid resistance board/fixed panel design with 3.8 cm spacing between panel bars. Since its initial installation in the fall of 2008, the weir has been located at rkm 17.2, just downstream from the Grays River Covered Bridge. The weir configuration has changed slightly each year to try and improve fish recruitment and to adapt to changing site conditions. For the fall 2010 season, the weir was weir installed and operational on August 24, 2010. The first and last Chinook salmon were captured on August 28, 2010 and September 26, 2010, respectively. In 2010, no substantial time was lost due to high flow events until a large rain event on October 10th and 11th. This event caused trap boxes to dislodge and, as a result of this damage, we never resumed trapping operations. The weir was removed as flows from this high water event subsided. For the first time in its three years of operation, both Chinook and coho were observed jumping over the weir on September 20th when flows were ~720 CFS. Figure 3 shows the 2010 Grays River weir configuration.

Figure 3. 2010 Grays River Weir configuration. Photo credit: Jeremy Wilson (WDFW).

A total of 59 Chinook salmon were trapped at the Grays River weir in 2010. Over 76% of the catch were Select Area Brights (SABs). SABs are hatchery fall Chinook that are released into Youngs Bay as part of the Select Area Fisheries Enhancement (SAFE) program. This is a non-local stock that originated from Rogue River stock. Table 4 shows the weir catch by species, sex, mark type, and disposition. Adult weir capture efficiency was 15.4% (95%CI 10.1 – 21.3%) and removal of marked (adipose and/or LV-clipped) Chinook salmon at the weir site reduced the proportion of marked spawners, or pMark, by 12.3%. No spawning occurred downstream of the weir site (Table 5). Age 3 and 4 fish dominated the age structure of fish removed at the weir (Tables 6). We were unable to examine run timing by mark type past the weir site due to the lower than expected weir capture efficiency.

Table 5. Adult weir capture efficiency, pMark, and percent spawning below weir for Grays River Chinook salmon population.

Parameter	Mean	SD	2.50%	Median	97.50%
W_{eff}	15.5%	2.9%	10.1%	15.4%	21.3%
pMark	54.1%	10.0%	34.5%	54.1%	73.2%
nwpMark	66.5%	9.7%	48.4%	66.3%	86.1%
cpMark	12.4%	13.9%	0.0%	12.3%	40.2%
% spbw	0.0%	0.0%	0.0%	0.0%	0.000

Table 6. Age structure by mark type of Chinook salmon removed at the Grays River Weir in 2010 based on scale readings. Not all Chinook removed were sampled for scales.

Age Read	AD-clipped		LV-clipped	
	Scale samples	Proportion	Scale samples	Proportion
Age 2	3	27.3%	22	51.2%
Age 3	6	54.5%	18	41.9%
Age 4	2	18.2%	3	6.9%
Age 5	0	0.0%	0	0.0%
Total n	11		43	

Elochoman River Weir

The Elochoman River weir currently utilizes a full fixed panel weir design that is installed annually on a permanent concrete sill with adjoining live box (Figure 4). The site is located just above Risk Road near the head of tide at rkm 4.39. For several decades, this site and configuration were used to trap broodstock for the WDFW Elochoman Salmon Hatchery fall Chinook program. In 2009, after closure of the Elochoman Hatchery (2008) and discontinuation of the fall Chinook program, responsibility for operation of the weir transferred to WDFW Region 5 fish management under Mitchell Act MER funding. During this transition, weir panels were re-built with 3.8 cm spacing (instead of the previous 7.6 cm spacing) between panel bars. For the fall 2010 season, weir installation began on August 8, 2010 and was operational later that same day. The first and last Chinook were captured on August 11, 2010 and October 9, 2010, respectively. On October 10th high flows topped the weir and the upstream gates to the trap box were opened. Fish tight weir operations never resumed in 2010 and the weir was later removed as flows subsided.

Figure 4. 2010 Elochoman River Weir configuration. Photo credit: Claire Landry (WDFW).

A total of 4,597 Chinook salmon were trapped at the Elochoman River weir in 2010. Table 4 shows the weir catch by species, sex, mark type, and disposition. Adult weir capture efficiency was 99.5% (95%CI 98.9 – 99.8%) and removal of marked (adipose and/or left ventral-clipped) Chinook at the weir site reduced pMark at the subpopulation level by 12.4%. No spawning occurred downstream of the weir site (Table 7). Age 3 dominated the age structure of Chinook removed at the weir (Tables 8). There was very little difference in the run timing between adipose clipped and unmarked Chinook at the weir; however, the 50% passage date for Select Area Brights was one week later than Tules (Figure 5).

Table 7. Adult weir capture efficiency, pMark, and percent spawning below weir for the Elochoman River Chinook salmon subpopulation.

Parameter	Mean	SD	2.50%	Median	97.50%
W_{eff}	99.4%	0.2%	98.9%	99.5%	99.8%
pMark	88.3%	1.3%	85.5%	88.4%	90.6%
nwpMark	97.3%	0.3%	96.7%	97.3%	98.0%
cpMark	12.4%	1.0%	10.6%	12.4%	14.3%
% spbw	0.0%	0.0%	0.0%	0.0%	0.0%

Table 8. Age structure by mark type of Chinook salmon removed at the Elochoman River Weir in 2010 based on scale readings. Not all Chinook removed were sampled for scales.

Age Read	AD-clipped		LV-clipped	
	Scale samples	Proportion	Scale samples	Proportion
Age 2	4	1.1%	3	23.1%
Age 3	267	72.0%	9	69.2%
Age 4	97	26.1%	1	7.7%
Age 5	3	0.8%	0	0.0%
Total n	371		13	

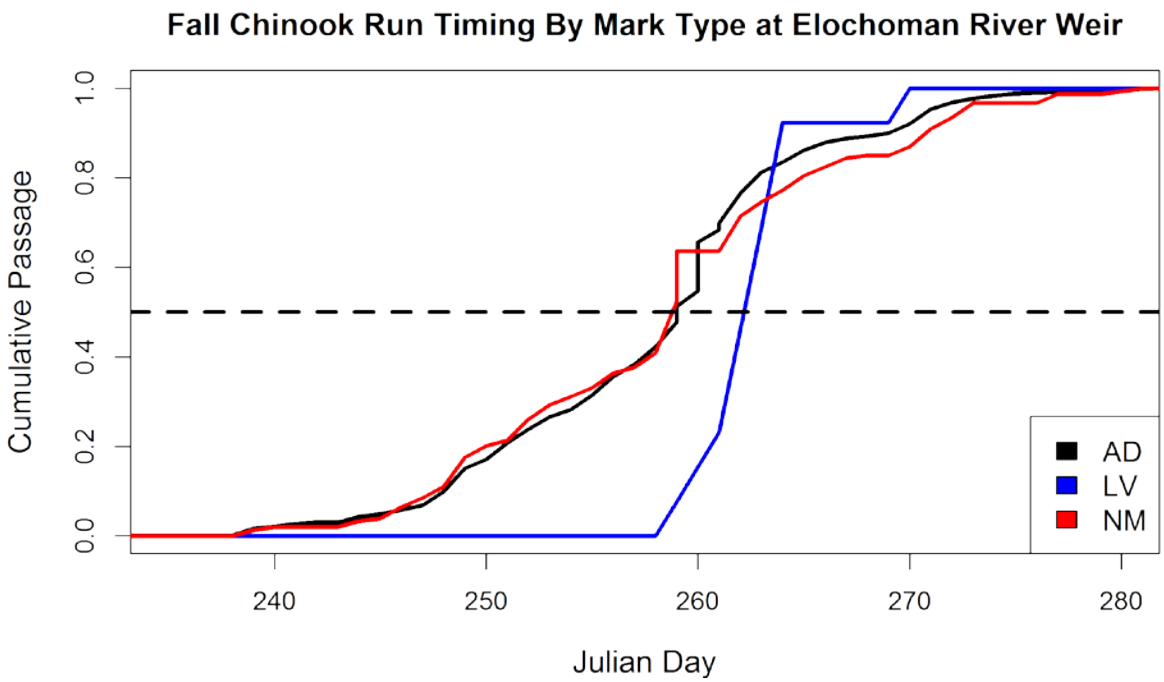

Figure 5. Run timing of Chinook salmon by mark type, Elochoman River Weir 2010. AD = adipose fin clipped (indicates hatchery-origin), LV = left ventral fin clipped (indicates hatchery-origin SAB stock), NM = no mark or all fins intact (assumed to be natural-origin).

Green River Weir

The Green River weir has been used to collect broodstock for the hatchery production program at North Toutle Hatchery for many years. In 2010, in addition to collecting broodstock, the fish management objective of controlling pHOS was implemented. This weir currently utilizes a hybrid fixed/resistance board design with 3.8 cm spacing between panel bars. The site is located at rkm 0.64 at the entrance to the North Toutle Hatchery adult pond. For the fall 2010 season, weir installation began on August 23, 2010 and was fully operational on August 25, 2010. The first and last Chinook were captured on September 1, 2010 and November 3, 2010, respectively. The weir was topped during a high water event on October 25, 2010 and submerged for multiple days. The weir was removed on November 29, 2010. Figure 6 shows the 2010 Green River weir configuration.

Figure 6. 2010 Green River Weir configuration. Photo credit: Amanda Danielson (WDFW).

A total of 3,443 Chinook salmon were trapped at the Green River weir in 2010. Table 4 shows the weir catch by species, sex, mark type, and disposition. Adult weir capture efficiency was 98.3% (95%CI 91.0 – 100.0%). While some marked Chinook were removed at the weir in an attempt to reduce pHOS, weir operations actually increased pHOS (pMark) slightly (~2%) due to unmarked Chinook being integrated into the hatchery program. Only 1.8% (95%CI 1.6 – 2.0%) of the Chinook spawning in the Green River subpopulation occurred below the weir site (Table 9). The 50% passage date past the weir site was three days later for unmarked Chinook compared to marked Chinook (Figure 7). Age data from Chinook removed at the weir was combined with Chinook spawned at N. Toutle Hatchery. Therefore, we did not report on age structure of weir removals in 2010.

Table 9. Adult weir capture efficiency, pMark, and percent spawning below weir for the Green River Chinook salmon subpopulation.

Parameter	Mean	SD	2.50%	Median	97.50%
W_{eff}	98.7%	3.9%	91.0%	98.3%	100.0%
pMark	89.2%	1.7%	85.8%	89.3%	92.3%
nwpMark	87.5%	0.9%	85.6%	87.6%	89.2%
cpMark	-1.7%	0.7%	-3.1%	-1.8%	-0.2%
% spbw	1.8%	0.1%	1.6%	1.8%	2.0%

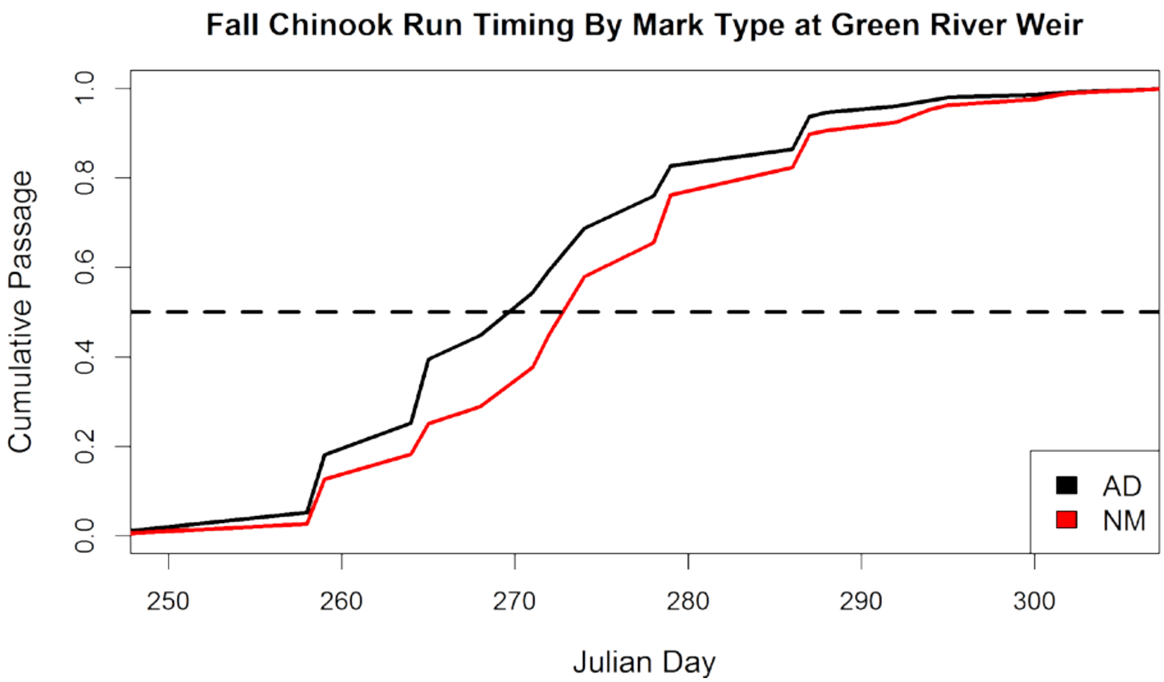

Figure 7. Run timing by mark type of Chinook salmon, Green River Weir 2010.

Weir efficiencies were greater than 99% and 98% at the Elochoman River and Green River weirs, respectively. Based on these weir efficiency estimates, we believe we could have reduced pHOS to a much greater degree with different management objectives (e.g. reducing pHOS vs. seeding habitat) at both of these sites. Both sites have solid infrastructure (concrete sills and large trap boxes and/or holding ponds) from old or existing hatchery programs. Weir operations at the Elochoman and Grays had the greatest impact of the three weir sites at reducing the proportion of marked Chinook on the spawning grounds in 2010 (Figure 8).

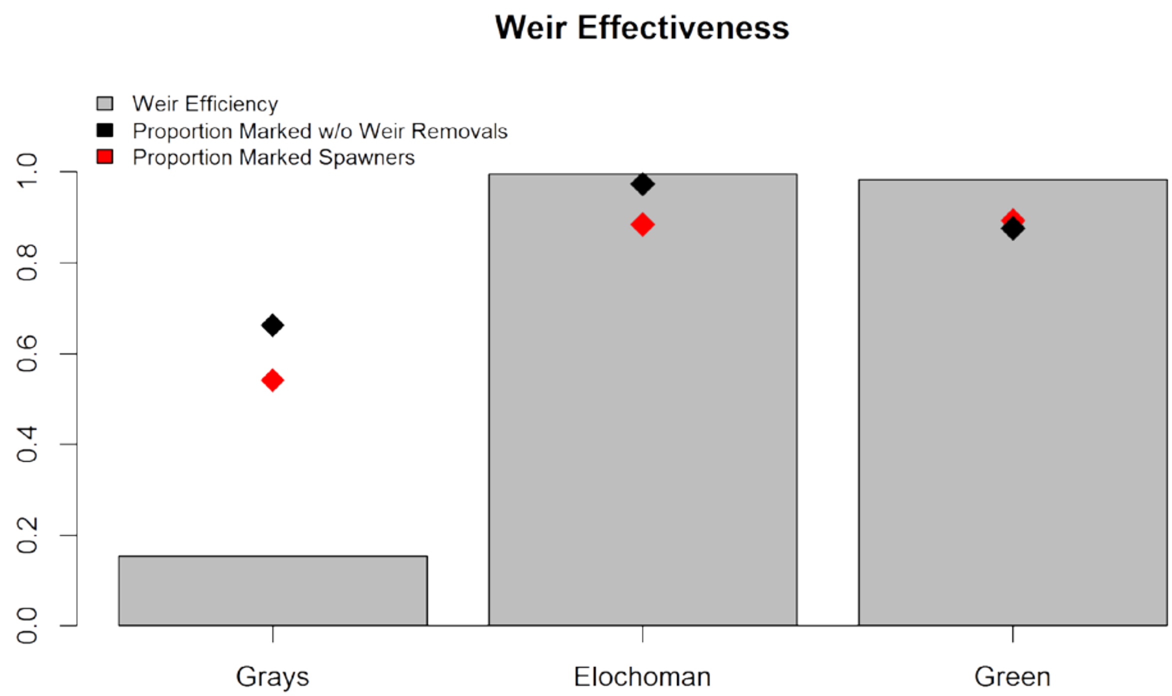

Figure 8. Adult weir capture efficiency, estimated pMark, and estimated pMark without hatchery origin weir removals for Chinook Salmon in 2010 by subpopulation.

SAFE program releases made up 100% of the CWT recoveries at the Grays River weir. At Elochoman weir, CWT recoveries were primarily made up of remaining returns from the recently terminated Elochoman Hatchery program (84% of unexpanded CWTs), with Big Creek releases (15% of unexpanded CWTs) and Deep River net pen releases (1% of unexpanded CWTs) making up the rest. The Green weir/N. Toutle Hatchery CWT recoveries were almost entirely from hatchery program releases within the basin (95%) with one unexpanded CWT from a Kalama River release. Table 10 shows the CWT recoveries by weir site in 2010.

Table 10. Unexpanded Coded Wire Tag (CWT) recoveries from surplus Chinook salmon at Grays, Elochoman, and Green weirs in 2010.

		Recovery Location		
		Grays Weir	Elochoman Weir	Green Weir / N. Toutle Hatchery
Release Basin	SF Klaskanine (OR)	2		
	Deep River		1	
	Big Creek (OR)		14	
	Elochoman		80	
	N. Toutle			18
	Kalama			1
	Total CWT Recoveries	2	95	19

Displaced spawning due to the weir structure was not an issue at any of the weir sites. No spawning activity was observed below the Grays and Elochoman weirs. Only 1.8% of the Chinook spawning in the Green subpopulation occurred below the weir site.

The permitted take levels associated with each weir operation were sufficient to allow unimpeded trapping operations without worry of exceeding the permitted take in 2010. Table 11 lists the permitted and actual take levels by species and origin for each weir location.

Table 11. Permitted and actual take levels of ESA-Listed salmonids by weir location, species, and origin in 2010.

		Grays	Elochoman	Green
Marked Chinook	Permitted Non-Lethal Take	NA	NA*	NA*
	Actual Non-Lethal Take	NA	864	2,323
	Permitted Intentional Mortality	500	NA*	NA*
	Actual Intentional Mortality	56	3,580	506
Unmarked Chinook	Permitted Non-Lethal Take	750	NA*	NA*
	Actual Non-Lethal Take	3	153	614
	Permitted Unintentional Mortality	23	NA*	NA*
	Actual Unintentional Mortality	0	0	1
Marked Coho	Permitted Non-Lethal Take	4,500	NA*	NA*
	Actual Non-Lethal Take	341	53	9,634
	Permitted Intentional Mortality	NA	NA*	NA*
	Actual Intentional Mortality	0	0	0
	Permitted Unintentional Mortality	135	NA*	NA*
	Actual Unintentional Mortality	0	0	0
Unmarked Coho	Permitted Non-Lethal Take	800	NA*	NA*
	Actual Non-Lethal Take	86	14	710
	Permitted Unintentional Mortality	24	NA*	NA*
	Actual Unintentional Mortality	0	0	0
Unmarked Chum	Permitted Non-Lethal Take	8,500	NA*	NA*
	Actual Non-Lethal Take	2	0	0
	Permitted Unintentional Mortality	255	NA*	NA*
	Actual Unintentional Mortality	0	0	0
Winter Steelhead	Permitted Non-Lethal Take	Not Listed	Not Listed	NA*
	Actual Non-Lethal Take	0	0	0
	Permitted Unintentional Mortality	Not Listed	Not Listed	NA*
	Actual Unintentional Mortality	0	0	0

*Draft Hatchery and Genetic Management Plans (HGMPs) were submitted to NOAA in 8/2004 which permitted take tables included. However, this was prior to mass marking of tule fall Chinook releases and no information was available yet on natural origin vs. hatchery origin abundance and proportions. HGMPs for these programs are scheduled to be resubmitted in the next year and will have specific take limits in them.

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Coho Salmon Escapement Estimates and Coded-Wire-Tag Recoveries in Washington's Lower Columbia River Tributaries in 2010

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Abstract

The Lower Columbia River (LCR) Coho Salmon Evolutionary Significant Unit (ESU) is composed of 24 populations split between the states of Washington and Oregon. The Oregon Department of Fish and Wildlife (ODFW) began comprehensive monitoring of coho salmon populations in this ESU in 2002. Minimum adult coho salmon estimates in Washington's portion of this ESU have been limited primarily to counts at hatchery facilities and a few fish ladder traps. In 2010, the Washington Department of Fish and Wildlife (WDFW) initiated a monitoring program to estimate coho salmon spawner abundance, the proportion of hatchery origin spawners (pHOS), the proportion of spawning reaches occupied, spatial distribution, sex ratio including the proportion of jacks, and to recover Coded Wire Tags (CWT). We recovered 84 CWT including at least one from each watershed containing a coho population with the exception of the Lower Cowlitz and the Lower Columbia Gorge. The adult coho salmon population monitoring program used trap and haul census counts, mark-recapture, smolt expansion, and redd-based methods to monitor adult coho salmon. We estimated a mean escapement of 57,666 (95% CI 48,240 - 70,751) adults and 4,428 jacks (95%CI 3,522 - 5,816) for the Washington portion of this ESU below Bonneville Dam excluding the mainstem Lower Cowlitz, mainstem Lower North Fork Lewis, mainstem Toutle/ lower North Fork Toutle (below the Sediment Retention Structure), and Salmon Creek populations. Individual population estimates for natural spawners ranged from a high of 21,745 adults for the combined upper Cowlitz/Cispus population to a low 488 adults for the Kalama population. As expected, populations with an operating coho salmon hatchery, including the Grays, Elochoman, Upper Cowlitz/Cispus, Green, Kalama and Washougal rivers, had high proportions of hatchery spawners—(mean = 81%, 73%, 87%, 60%, 99%, and 44%, respectively). The converse was true for populations without hatcheries, such as the Mill-Abernathy-Germany, Lower Cowlitz, Coweeman, and South Fork Toutle populations, where we observed low percentages of marked adults—(mean = 12%, 15%, 10%, and 21%, respectively). The total mean estimate of unmarked coho salmon adults was 25,942 (95%CI 19,430 – 35,510). Estimates of precision for the aggregate estimate for all adults and unmarked adults as measured by the coefficient of variation (CV) were 10% and 16%, respectively. The precision of individual estimates of unmarked adults based on redd surveys was low—the most precise estimate occurring in the Green River with a CV of 24%. In two study streams redd surveys were conducted in conjunction with mark-recapture to estimate the number of redds per female in order to expand redd counts throughout the ESU. The median estimate of redds per female was 0.471 (95%CI 0.351 – 0.607), which indicates that we likely only observed about 47% of the redds, assuming each female constructed one redd. This estimate was consistent with poor water clarity and periods of high discharge which limited observer efficiency and erased physical evidence of redds. Trap counts and mark-recapture estimates of coho salmon abundance were more precise than the redd-based estimates. To improve the precision of adult coho salmon redd-based estimates, we recommend obtaining more precise estimates of redds per female, increasing the number of reaches surveyed per population, and exploring possible density-based stratification of the Generalized Random Tessellation Stratified (GRTS) sampling design used to select redd survey reaches.

Introduction

Coho salmon (*Oncorhynchus kitsuch*) in the Lower Columbia River (LCR) Evolutionary Significant Unit (ESU) were listed for protection under the Endangered Species Act (ESA) in 2005. In a recent five-year review, the National Oceanic and Atmospheric Administration (NOAA) Fisheries concluded that these fish should remain listed as threatened under the ESA (NOAA 2011). The LCR coho salmon ESU is composed of 24 populations split between the states of Washington and Oregon (Myers et al. 2006). The Oregon Department of Fish and Wildlife (ODFW) began comprehensive monitoring of LCR coho salmon populations in 2002 (Suring et al. 2006). However, estimates of adult coho salmon escapement in Washington's portion of the LCR were limited primarily to counts at hatchery facilities and a few fish ladder traps until this project was funded in 2010.

The coastwide Coded-Wire-Tag (CWT) program was developed in the 1970's to evaluate the contribution of different salmonid populations and hatchery programs to various fisheries and to estimate salmon fishery harvest rates, along with evaluation of hatchery rearing practices. The initial protocols for the CWT program included the insertion of a CWT into the snout of a juvenile hatchery salmon, which was accompanied by an adipose fin clip. A proportion of hatchery fish released from selected facilities had a CWT inserted. When salmon were recovered from fisheries and spawning areas, the snout of fish with missing adipose fins were taken to fisheries agency labs for decoding. Later the purpose of the CWT program was expanded to include forecasting run sizes to meet conservation and harvest objectives. For conservation purposes, the vast majority of coho salmon released from hatcheries are now adipose fin clipped (sometimes referred to as mass marked) and WDFW has implemented selective fisheries, which require the release of all adipose-intact (assumed to be natural origin) fish. CWTs are now detected electronically by scanning fish with handheld or stationary detectors, rather than using the adipose fin clip as an indicator of CWT presence. Upon implementation of mass marking, standard CWT protocols were modified to include inserting a CWT into a proportion of a hatchery release that was not adipose fin clipped—referred to as a Double Index Tag (DIT) group. The DIT groups were released from a few select hatcheries. These DIT groups are unmarked hatchery fish with a CWT that allow the evaluation of the harvest rates specific to selective fisheries and these DIT groups serve as surrogates for wild coho harvest rates for stocks subject to selective fisheries.

In 2010, the Washington Department of Fish and Wildlife (WDFW) initiated a program to sample LCR spawning grounds for coho salmon. This program had dual objectives: 1) to estimate Viable Salmonid Population indicators (McElhany et al. 2000) and measure specific indicators to assess coho salmon viability (Rawding and Rodgers 2013) including coho salmon spawner abundance, the proportion of hatchery origin spawners, the proportion of spawning reaches occupied, spatial distribution, and sex ratio including the proportion of jacks; and 2) to recover CWT from spawning fish to provide complete accounting of CWT, so that harvest rates could accurately be determined. The first objective addressed a salmon recovery monitoring gap while the second objective addressed a gap identified from the CWT expert panel (Hankin et al. 2005). This report summarizes population monitoring of VSP indicators for LCR coho salmon returns and CWT recoveries in 2010.

Methods

Study area

The LCR coho salmon ESU extends from the mouth of the Columbia up to and including the Big White Salmon River in Washington and Hood River in Oregon, and includes the Willamette River to Willamette Falls, Oregon. Within this ESU, there are a total of 15 Washington populations, 7 Oregon populations, and 2 populations (Lower and Upper Gorge) that are split between the states (Figure 1). In this document we report on 14 populations in Washington. The upper Cowlitz and Cispus populations are combined into a single population because there is currently no way to determine spawning locations from the Cowlitz River trap and haul program for fish that are placed above Cowlitz Falls Dam. In 2010, the Salmon Creek and Upper Gorge populations (including the Big White Salmon River), were not surveyed; nor were fish spawning in mainstem areas of the lower Cowlitz, lower NF Lewis, lower NF Toutle, and Toutle, though of these areas, only the mainstem NF Lewis is known to support significant numbers of spawners. It should be noted that coho salmon in the Lower Gorge population spawn in both states, but this report only contains information on the Washington proportion of the Lower Gorge population.

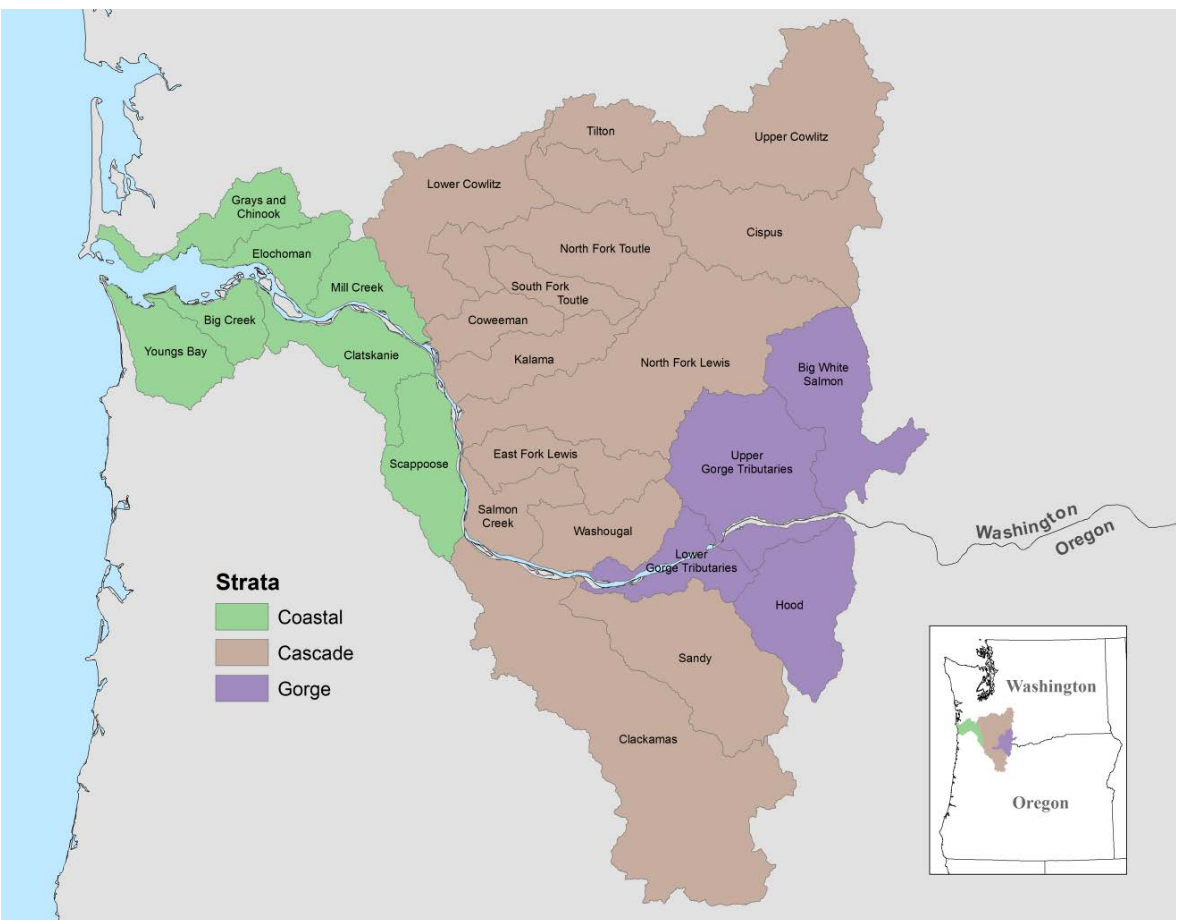

Figure 1. Lower Columbia River coho salmon populations and the regional groupings (i.e., strata) in which they occur within the LCR subunit recovery domain. The White Salmon population is considered part of the Upper Gorge Population.

Monitoring Design

We used dam counts and trapping, mark-recapture, and spawning ground surveys to estimate population parameters of LCR coho salmon (Figures 2 & 3). Field personnel were experienced and/or trained on adult salmon identification. Field data collection protocols varied but were based on the methods from the American Fisheries Society for salmon monitoring (Johnson et al. 2007). Coho salmon redd, live fish, and carcass counts along with environmental and header information collected during coho salmon surveys were stored in the WDFW Spawning Ground Survey (SGS) database. Biological data collected on spawning ground surveys was stored in the WDFW Region 5 Age and Scales (A&S) database. Individual trap counts, tagging, and recovery data were stored in individual watershed databases or spreadsheets.

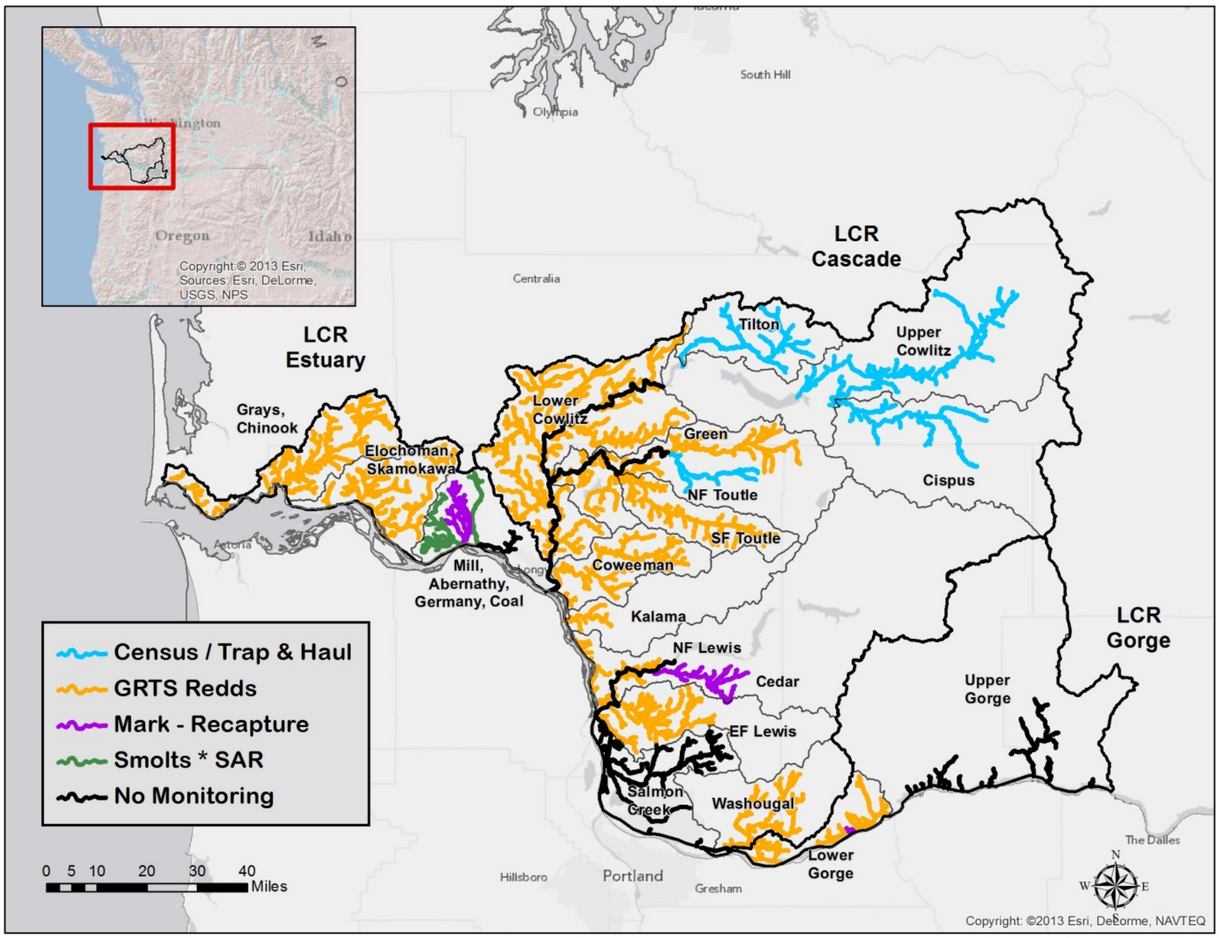

Figure 2. Watersheds containing the Washington populations of the Lower Columbia River coho salmon ESU and the methods WDFW used to estimate their abundance.

Trap & Haul

Dam counts were used at the Barrier dam on the Cowlitz River (RM 50), and the Toutle Fish Collection Facility (TFCF) on the NF Toutle River (RM 12). Depending on management objectives, coho salmon collected at these facilities were used for hatchery broodstock, surplused (donated to food banks, sold to the state fish buyer, or used for nutrient enhancement) or transported and released above the facility. We made the following key assumptions for the trap and haul programs: 1) the count of all transported fish was without error, 2) all unmarked fish released survived to spawn, 3) transported fish spawned in the watershed where they were released (there was no fall back), 4) when fisheries in the Upper Cowlitz, Cispus, and Tilton Rivers occurred only marked (adipose clipped fish) were harvested in accordance with regulations and there was no illegal harvest, 5) survival of all unmarked released fish was 100% (catch and release mortality was negligible), and 6) the WDFW methodology to expand catch record card (CRC) reported catch to total harvest and variance are correct.

Mark-Recapture

Petersen mark-recapture estimates for adult coho salmon were attempted in the Green River and Abernathy, Cedar, and Duncan creeks. However, high water undermined the weir at the Green River so escapement in this basin was determined using spawning ground surveys. Coho salmon were captured in adult traps located adjacent to the resistance board weir in Abernathy Creek at RM 0 (Kinsel et al. 2009), the fishway in Cedar Creek (RM 2), and Duncan Creek fishway (RM 0). Traps were installed prior to immigration of adult coho and fished through the end of migration in January or February. Recapture of fish occurred at upstream traps or during spawning ground surveys. A study design at all three locations was based on the Darroch estimator, which was developed for time stratified Petersen mark-recapture abundance estimates (Darroch 1961, Seber 1982). Schwarz and Taylor (1998) indicate that the following assumptions must be met to provide a consistent estimate of abundance: 1) there is no mark loss, 2) there are no marking effects, 3) all marked and unmarked fish are correctly identified and enumerated, 4) the population is closed, and 5) all fish in the population have the same probability of being tagged or all fish have the same probability of being captured in the second sample; or marked fish mix uniformly with unmarked fish.

Smolt Expansion

The number of coho salmon smolts emigrating from Mill, Abernathy, and Germany creeks are estimated annually based on a mark-recapture design (Kinsel et al. 2009). In addition, adult coho salmon returning to Abernathy Creek are estimated based on a mark-recapture study design. All three creeks enter the Columbia within a few miles of each other. We assumed that the smolt to adult return (SAR) and stray rates were the same between the three sites, and were therefore able to estimate the number of returning unmarked adults in Mill and Germany creeks by dividing their smolt estimates by the estimated Abernathy Creek (SAR).

Spawning Ground Surveys

The monitoring design components for spawning ground surveys consist of basic elements (Stevens et al. 2007). These included: 1) the development of the sampling frame covering the entire spawning area, 2) a probabilistic sampling design to representatively survey the spawning area, 3) a temporal component to ensure the entire spawning period was sampled, and 4) a decision on the metric (e.g., live fish, carcass, or redd counts) used to estimate escapement, the observer efficiency, and the relationship between the metric and the escapement.

Spawning Distribution

The upper extent of the coho salmon spawning distribution was estimated based on the methods of Fransen et al. (2006). The upper extent of adult and juvenile coho salmon presence was estimated from focused and randomly selected surveys over two years. For sampled streams, fish presence protocols from the Washington Forest Practices Board (WFPB) for juveniles were followed. Following AFS electroshocking protocols (Temple et al. 2007), juveniles were continuously sampled moving in an upstream direction until fish were not observed for at least ¼ mile or a waterfall was encountered. This protocol was adapted for adult salmon except fish presence was based on visual sampling of live or dead adult coho salmon or their redds. The uppermost presence of fish was recorded using a global position satellite (GPS). This location was plotted on the WDFW Geographic Information Systems (GIS) stream and attribute layer. GIS attributes were recorded for the last reach where coho salmon were found, as well as the seven reaches downstream and eight reaches upstream of that point. Using logistic regression, a model was developed to predict the upper extent of coho distribution as a function of the GIS covariates including drainage area, mean annual flow, annual precipitation, confinement, elevation, and gradient. Akaike's Information Criterion (AIC) was used to compare models and select the best model following Burnham and Anderson (2002). The coefficients for GIS covariates included in the best predictive model of upstream coho extent were drainage area, gradient, and elevation. The model was used to predict upstream extent throughout the ESU; the upstream extent was then further truncated by applying the WDFW fish passage barriers layer. The lower extent of coho salmon spawning was defined by the lowest location surveyed for steelhead or Chinook redds in previous years; typically the downstream most extent of gravel in each watershed. More complete details of the upstream extent model development for Chinook salmon are provided in Rawding et al. (2010), which was adapted as described above for coho salmon. The spawning distribution drainage network as described above was the sampling frame used to develop the spatial sampling design for redd surveys.

GRTS Survey Sampling Design

A spatial sampling design was developed for 12 of the 17 coho salmon populations in Washington. The Upper Gorge population was excluded due to limited resources, the Mill-Abernathy-Germany (MAG) Creeks, Upper Cowlitz, Cispus, and Tilton River populations were excluded because we used alternate methods. For each population a Generalized Random Tessellation Stratified (GRTS) sampling design was used to establish a set of random, spatially balanced sample points for coho salmon surveys (Stevens 2002). Reach selection was based on the LCR GRTS web based sampling tool developed by Oregon State University (OSU) through the Pacific Northwest Aquatic Monitoring Partnership (PNAMP) with assistance from Don Stevens (OSU). Reaches, one mile in length, were established based on these points. In a few cases the reach length was less than one mile. This occurred when the GRTS point was located in a small tributary less than one mile in overall length or there was an anadromous barrier falls less than a mile from the mouth. In the case of a tributary being less than 1 mile in length, the reach length was extended to the top fork of the 24k Washington Lakes and Rivers Information System (WLRS) stream layer regardless of sample frame.

A three-year rotating panel design was established for each coho salmon population (Firman and Jacobs 2004). In this design about 1/3 of the surveys for the 9-year period are repeated annually, 1/3 are repeated every third year, and new points are chosen each year for the remaining 1/3 of all surveys. For Oregon coastal coho salmon, the Oregon Department of Fish and Wildlife (ODFW) surveys 30 sites for each population, or enough sites to cover 30% of the coho spawning habitat for each coho salmon population (Lewis et al. 2009). The 30 sites or 30% of the habitat, whichever is lower, is expected to yield an average coefficient of variation (CV) near 15% (Jeff Rodger, ODFW, pers. comm.). However, due to limited resources, WDFW only sampled from 2 to 25 reaches per population (Appendix 1).

Weekly spawning ground surveys were scheduled for each reach from the start of spawning in mid-to late October until there was no observed spawning activity, which usually occurred in December or January depending on the population. However, due to high turbid flows and personnel challenges the designed temporal pattern did not always occur and some scheduled weekly surveys were missed.

Data Collection

Trap & Haul

Coho salmon populations originating above dams in several Lower Columbia watersheds were trapped and fish were hauled above those dams allowing for their enumeration and the collection of biological data. These watersheds included areas above the Sediment Retention Structure (SRS) on the North Fork Toutle River, and the upper Cowlitz watershed above the Barrier Dam, including separate populations in the Tilton River and the Upper Cowlitz/Cispus Rivers. NF Toutle River coho salmon were trapped, anesthetized in CO_2, and sampled for biological data including length, sex, origin, and age. Adipose intact fish were transported and released in Alder and Bear Creeks, while hatchery fish, those with adipose fin clips, were recycled below the TFCF. Cowlitz River coho salmon captured at the Barrier Dam were anesthetized using electro-anesthesia and sampled for sex and origin. In addition, male coho salmon were classified as jacks or adults based on size. Adult salmon captured at the Barrier Dam were released to their natal watersheds based upon differential marking they received as smolts when they were transported downstream of the Cowlitz dams; since smolts and parr caught at the Mayfield trap were tagged with blank CWT and not adipose fin clipped, these fish were released in the Tilton River which empties into Mayfield Lake, whereas unmarked adipose intact fish were transported to the Upper Cowlitz and Cispus Rivers where they presumably originated. In addition adipose clipped hatchery coho salmon were released in the Tilton, Upper Cowlitz and Cispus rivers to provide recreational fishing opportunity and spawners to seed the available habitat. This action is needed because the current juvenile collection at the Cowlitz falls dam is approximately 30% and too low to support self-sustaining runs (Serl and Morrill 2009). Other adipose clipped coho salmon collected at the Barrier dam were used for broodstock, or surplused.

Mark-recapture

Fish in good condition were anesthetized, bio-sampled, double Floy (FD 68BC T-bar Anchor tags, Floy Tag & Mfg., Inc. Seattle, WA) tagged and released upstream in Duncan Creek at RM 0, Abernathy Creek at RM 0, and Cedar Creek at RM 2. Opercle punches were applied as a secondary mark, and were rotated weekly, allowing assessment of Floy tag (ft) loss and assignment of a recovered fish back to the weekly release group in Duncan and Abernathy creeks. In Cedar Creek, the opercle punch was not used but a third plastic tag was stapled to the opercle (op) to assess tag loss. The recapture events occurred at the Abernathy Fish Technology Center (AFTC) located at RM 4, the Cedar Creek resistance board weir (RM 6). In addition to the recapture events described above, carcass recovery events occurred in Duncan and Abernathy creeks along with a resight event of live tagged and untagged spawning fish during scheduled weekly spawning ground surveys. Recovery events were concurrent with spawning ground surveys in all creeks. Due to their small size, the sample frame for Duncan and Abernathy

Creeks was the entire spawning distribution, which resulted in a redd census rather than a probabilistic sample (e.g., from a GRTS design).

Spawning Ground Surveys.

Redd surveys followed the protocols of Gallagher et al. (2007). The start and end of each survey reach based on the GRTS design was geo-referenced and its coordinates were recorded on a Garmin Oregon 550 unit set in NAD 83. Surveyors typically located the upper most point in the reach and walked downstream to the coordinates at the end of the reach. Surveys were scheduled weekly and followed methods in Rawding et al. (2006a) and Rawding (2006b). All identifiable redds were flagged, and their location (latitudinal and longitudinal coordinates) was recorded. GPS units were allowed to acquire satellite locations until an accuracy of $\pm$ 100 feet or less was obtained, most often accuracies averaged 5 to 50 feet. In subsequent surveys, previously flagged redds were inspected to determine if they should be classified as “still visible” or “not visible”. A redd was classified as “still visible” if it would have been observed and identified without the flagging present, and was recorded as “not visible” if it did not meet this criteria. These data were collected to allow us to estimate the time period redds were visible to surveyors.

In addition, all live adult and jack salmonids were recorded by species (Crawford et al. 2007a). Salmon were identified as either spawning or holding. A fish was identified as holding if it was observed in an area not considered spawning habitat, such as pools or large cobble and boulder riffles (Parken et al. 2003). Salmon were classified as spawners if they were on redds or not classified as holders. Counts of live Chinook, coho, and chum salmon were recorded separately for each survey reach. During these surveys, counts of tagged and untagged coho salmon occurred in the Abernathy, Green, Cedar, and Duncan watersheds to provide one potential method to estimate abundance based on a mark-resight estimator (Jacobs et al. 2002).

All carcasses that were not totally decomposed were sampled for external tags (Floy T-bar or opercle tags) and biologically sampled for fork length, sex, adipose fin presence, and condition (extent of decomposition). Sex was determined based on morphometric differences between males and females and/or by cutting open the abdominal cavity to confirm sex and determine spawning success. The spawning success was approximated based on visual inspection, ranging from 100% to 0% success. A fish with 0% success was considered a pre-spawning mortality. Scale samples were collected by selecting scales from the preferred area as described in Crawford et al. (2007b). Preferred scales are samples in an area ~ 1-6 scale rows high, and ~15 scale rows wide, above the lateral line in a diagonal between the posterior insertion of the dorsal fin and anterior insertion of the anal fin. Scale samples were removed with forceps with special care to select scale samples that were of good quality (round shape, non-regenerated) and not adjacent to one another (to minimize the effects of regeneration) as described in a WDFW technical report (Cooper et al. 2011). Scales were placed on the gummed portion of WDFW scale cards with their exterior surfaces facing up. The scale card number, position number, date, and location create a unique code in the A&S database.

All adipose fin marked fish were sampled for CWT following standard protocols (NWMFT 2001). The surface of the CWT wand with radiating arrows is placed in contact with the snout and moved from the right to the left eye, and then up and over the snout area. The wand is also inserted into the mouth with the radiating arrows rubbed against the roof of the mouth in vertical strokes. If a CWT is detected, the red LED will light up and a beep is emitted from the wand. When a CWT is detected, the snout is severed by cutting across the head straight down behind the eyes (Crawford et al. 2007b). The snout is placed in a plastic bag with a tag number linking the snout to biological data (length, sex, fin clips, spawning success for females, and scale sample number) recorded on the scale card, stream survey card, or other datasheet. Snouts are stored in a freezer and periodically delivered to the WDFW CWT lab in Olympia for processing.

Sample Processing

Scale Analysis

Scale preparation and analysis followed WDFW protocols (Cooper et al 2011). Acetate impressions are made of the scale samples by a scale card press, where samples are covered with clear acetate (0.5mm thickness) and pressed under 1200-1300 PSI @ 100 degrees C for 30 seconds to 1 minute. Acetate impressions are then slightly cooled and removed from the scale card. Acetate impressions of scale samples are aged using a modified Gilbert/Rich ageing notation (Groot and Margolis 1991), where annuli are counted along with the scale edge to produce a total age in years. Annuli are defined as an area of narrowly spaced circuli that represent winter/early spring growth. Age is recorded as the total age in years followed by the year at outmigration. For example a typical coho salmon adult is age 3_2. This notation indicates a total age of 3 and the juvenile salmon left its natal freshwater habitat within its second year of life. After being aged in Olympia by an aging specialist, scale samples were returned to Vancouver for entry into the Age & Scales database.

CWT Lab Analysis

The recovery of CWT tags at the WDFW lab follows the procedures outlined in the tag recovery chapter (Blankenship and Hiezer 1978) of the Pacific Coast Coded Wire Tag Manual and is briefly repeated here. Each snout is passed through a magnetic detector to determine tagged and untagged snouts. Untagged snouts are set aside and rechecked after magnetizing the tag. To ensure the tag is magnetized the length of the tag must pass through the horseshoe magnet in a plane parallel with a straight line collecting the poles. If the tag angle is off more than 40 degrees the tag may not be magnetized. Therefore, the head is passed through the magnet in three positions corresponding to the X, Y, and Z axes and then through the detector. Large heads are often dissected to maximize tag detections. Snouts with no tags are saved and an x-ray machine is periodically used to determine tag presence in these “no tagged” snouts. After determining a tag is present, the snout is dissected, and the tag is located by process of elimination. After recovering the tag, the binary code is determined by careful observation under a microscope. CWT data is then entered into WDFW CWT access database, and provided to managers as needed and uploaded into the Regional Mark Information System (RMIS).

Data Analysis

Overview

Coho salmon abundance estimation was relatively straightforward for mark-recapture areas and trap and haul areas, but required combining multiple sources of information for GRTS survey areas (Figure 3). Briefly, a spawning habitat model developed for the ESU was parameterized with data from each GRTS survey sub-basin to predict the spawning habitat sampling frame. A subsample of reaches in this area was surveyed for redds, live fish, and carcasses, and the mean redd density was multiplied by the spawning habitat frame to estimate total redds for the GRTS sub-basin. Total redds were converted to total females by applying an ESU-wide estimate of redds per female based on the ratio of female abundance to census redd counts in mark-recapture basins. Female abundance was converted to adult abundance which could also be assigned marked and unmarked proportions based on hierarchically modeled sex ratios and marked to unmarked ratios from carcass recoveries in GRTS surveys. Jack abundance was estimated based on total male abundance from an ESU-wide estimate of the proportion of males that were jacks from adult fishway traps.

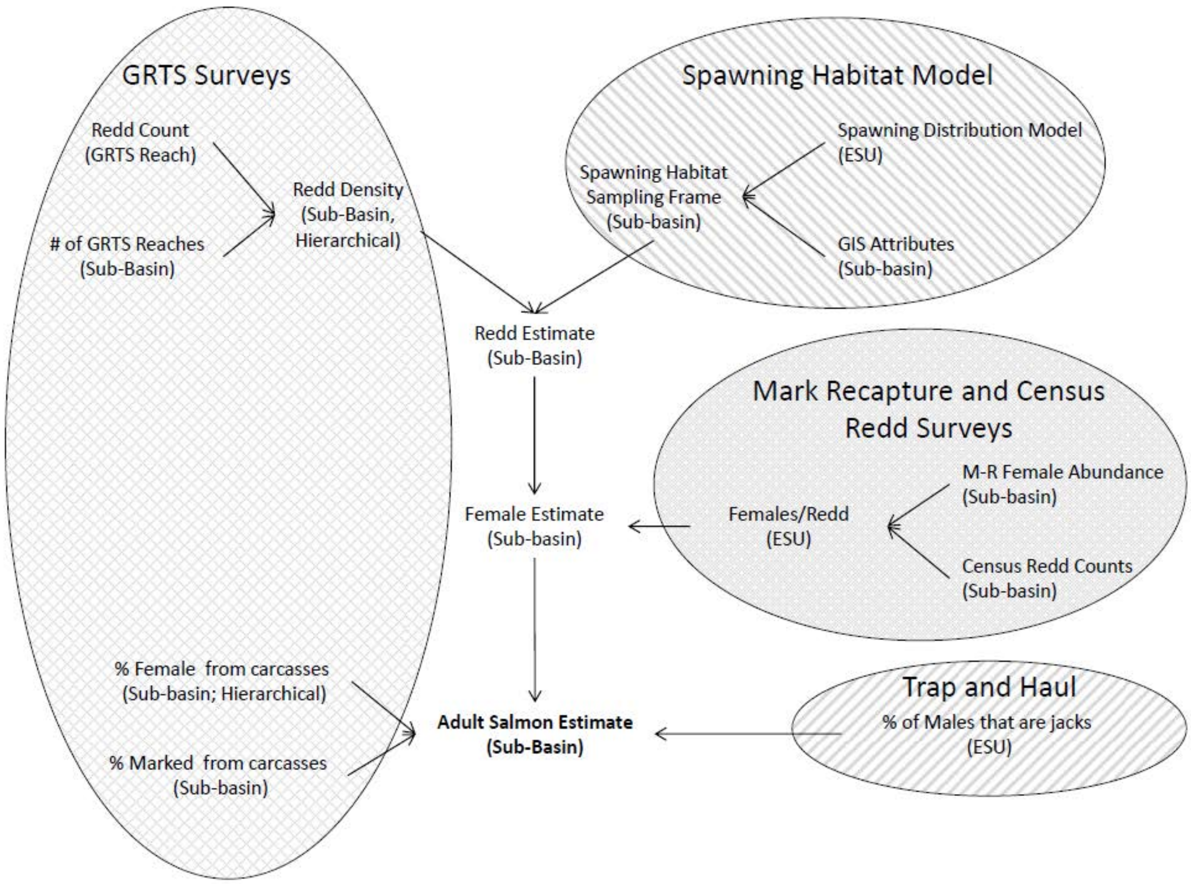

Figure 3. Overview of study design and data inputs from used to generate coho salmon abundance estimates for GRTS survey sub-basins. Shaded circles show general information sources while wording describes specific parameters and arrows show how specific parameters were combined to generate estimates. Spatial scales are listed below parameters and hierarchically modeled parameters are noted.

Modeling Approach

Data analysis was conducted using a Bayesian framework. Bayes rule states the posterior distribution, $p(\theta|y)$, is the product of the prior distribution, $p(\theta)$, and the probability of the data given the model or likelihood, $p(y|\theta)$, which is expressed by

$$p(\theta \mid y) = \frac{p(\theta)\, p(y \mid \theta)}{p(y)} \tag{1}$$

Where y are the data, θ are the parameters, and $p(y) = \Sigma_\theta\, p(\theta)p(y|\theta)$ for all discrete values or $p(y) = \int p(\theta)\, p(y \mid \theta) d\theta$ for continuous data (Gelman et al. 2004). The formula of the posterior distribution may be complex and difficult to directly calculate. Samples from the posterior distribution can often be obtained using Markov chain Monte Carlo (MCMC) simulations (Gilks et al. 1995). WinBUGS is a software package that implements MCMC simulations using a Metropolis within Gibbs sampling algorithm (Spiegelhalter et al. 2003) and has been used to estimate abundance and densities in fish and wildlife studies (Rivot and Prevost 2002, Wyatt 2000, Link and Barker 2010). All of the modeling results described in this paper have undergone tests to assess chain convergence and the uncertainty in the parameter estimates due Markov Chain variability (Plummer et al. 2005, Su et al. 2001). We used multiple chains starting at divergent initial values and monitored the chains until they reached equilibrium, which was assessed visually and using the Brooks-Gelman-Rubin statistic (Lunn et al. 2013). Values less than 1.1 are considered to have converged (Gelman et al. 2004). After discarding the burn-in, iterations before convergence, we monitored the Monte Carlo standard error (MCSE) until it was less than 5% of the standard deviation to obtain accurate parameter estimates (Lunn et al. 2013). We also monitored the estimate of effective parameters, which Rafferty and Lewis (1992) suggested that if the effective parameters equaled 4,000 than the estimate of the 2.5% quantile is within ± 0.005 with a 95% probability. It is therefore assumed that our reported estimates are accurate and represent the underlying stationary distributions of the estimates parameters.

The mode, median, and mean are commonly reported measures of central tendency for posterior distributions, which are reported in the form of point estimates. The mode is the most frequent value in the dataset. The middle value of the data is the median and the mean is the sum of the numbers in the dataset divided by the numbers in the dataset. For symmetrical distributions these measures of central tendency are the same. However, for asymmetric distributions it is not always clear on which measure of central tendency to report. The median is often used because it is intermediate to the mode, which can be a poor choice when it is distant to the middle of the distribution, and the mean, which can be give substantial weight to extreme values (Carlin and Louis 2009). Many of our estimates include the combination of two distributions (e.g. the number of fish by age which include the multinomial distribution for age and various distributions for abundance). Because these two distributions are often asymmetrical for fish monitoring data when we sum the medians of abundance by age they may not equal the median abundance estimate. Therefore, to limit confusion we have decided that the reported estimate will be the mean, which has a property that the individual estimates sum to the total estimate. The summary table will also include the median, the standard deviation based on the posterior distribution. We reported the equal-tailed or symmetrical 95% confidence intervals which exclude 2.5% from each tail of the posterior distribution rather than the highest probability interval, which is the shortest 95% interval of the posterior mass and is sometimes preferred (Lee 2004)

We specified vague priors for parameters. First, because this was the first study to estimate coho salmon in the Washington's portion of LCR, there was little prior information. Second, vague priors are developed not to influence the posterior distribution and therefore "let the data speak for themselves". We chose Beta and Dirichlet priors parameterized with $\alpha = \beta = 1$, 0.5, or 0.01 for binomial or multinomial distributions, which are referred to as the Bayes-LaPlace, Jefferys', and Haldane, respectively. We used the Jefferys' prior in the model and tested sensitivity using the other priors. For abundance estimates in mark-recapture, we chose a Uniform prior, so that the minimum and maximum bounds did not truncate the posterior distribution. When hierarchical modeling of binomial proportions, we chose the logit-normal model with mean having a vague Normal(0,100) and a Uniform (0,100) for the standard deviation (Gelman 2006). We also considered a Gamma (0.001, 0.001) constrained to less than 100 for each of the alpha and beta hyper-priors in the hierarchical models to test the sensitivity of the logit-normal priors.

Null hypothesis tests were used to test some of the closed population mark-recapture assumptions including: 1) capture probabilities are constant between periods, 2) the proportion of marked fish in the second sample is constant between periods, and 3) the proportion of marked fish captured between methods is constant. Bayes rule was used to test these hypotheses based on the Bayes factor, which is similar to the likelihood ratio. However the likelihood ratio is based on the parameters that maximize the likelihood while the Bayes factor integrates over the values of the parameter specified by the prior distribution. The Bayes factor can be expressed as

$$\Pr(H_i \mid D) = \frac{\Pr(H_i) \text{x} \Pr(D \mid H_i)}{\sum_j \Pr(H_j) \text{x} \Pr(D \mid H_j)} \tag{2}$$

where D is the data, H is the hypothesis, Pr is the probability, and subscripts i and j indicate individual and multiple hypotheses, respectively. The Bayes factors were computed analytically using the beta binomial distribution (Ntzoufras 2009, page 399). Bayes factors are known to be sensitive to the prior distribution to estimate θ. However, since our analysis is limited to comparison of binomial models, we adopted the standard reference prior which is a uniform distribution between 0 and 1. In addition, we evaluated support for the null and alternative hypotheses and interpreted Bayes factors based on the scale provided by Kass and Rafferty (1995) found in Table 1.

Table 1. Bayes factor interpretation from Kass and Rafferty (1995).

B_{10}	Evidence for H_0	B_{10}	Evidence against H_0
1-3	Negligible Support	1-0.33	Negligible Support
3-20	Positive Support	0.33-0.05	Positive Support
20-150	Strong Support	0.05-0.0067	Strong Support
>150	Very Strong Support	<0.0067	Very Strong Support

In some cases we used hierarchical models (e.g., Gelman et al. 2004). For example, we believe that the sex ratio of adult coho salmon in each LCR population should be near 50% females (Sandercock 1991, Dittman et al. 1998), but may vary slightly between populations, and may be subject to measurement error due to small carcass sample sizes. In this case, hierarchical models should adequately describe the percentage of females in each spawning population while allowing the hierarchical posterior distribution of sex ratio estimates to reduce the influence of small sample sizes in contributing to measurement error. Following this same logic, hierarchical models were used to estimate the percentage of jacks within the male population from trap data and to estimate watershed redd density based on the negative binomial distribution, which is appropriate for over-dispersed count data. The hierarchical redd density model was also necessary for some populations because the small number of reaches surveyed resulted in challenges obtaining stable numerical redd density estimates unless the method of moments was used to estimate parameters for the negative binomial distribution.

A key assumption in hierarchical models is that of exchangeability (Kery and Schaub 2012). In our sex ratio model, this means that all the individual population sex ratios are assumed to come from a common distribution of sex ratios for all LCR coho salmon populations, and their ordering does not affect the results. An important characteristic of hierarchical models is that the individual estimates borrow strength from the group estimate. This results in shrinkage of the individual estimates toward the population mean (Gelman et al. 2004). The amount of shrinkage depends on the variance between the populations and their sample sizes. We chose the hierarchical approach as a compromise between treating each population's sex ratio independently and pooling all data to estimate a single sex ratio. An advantage of this approach is less over-fitting of the data than would occur in generating independent estimates for each population, while still accounting for individual variability to influence estimates for a particular population (Kery and Schaub 2012).

Goodness of Fit (GOF) Tests

The purpose of GOF tests are to identify potential inadequacies in the fit of the model to the observed data. One Bayesian approach used for GOF testing is posterior predictive checking, which is a comparison of the posterior predictive distribution of replicated data from the model with the data analyzed by the model (Gelman et al. 2004). In other words, the predictive data ($y.rep_i$) is the expected observation after replicating the study, having observed the data (y_i) and assuming the model is true. When using MCMC simulations, a measure of discrepancy (D) is computed for the actual and replicated datasets for each iteration. An assessment of the posterior distributions of $D(y^{rep},\theta)$ and $D(y,\theta|y)$ provides individual and overall GOF measures. The posterior or Bayesian p-value = $\Pr(D(y^{rep},\theta) > D(y,\theta|y)$. The interpretation of the Bayesian p-value is the proportion of the times the discrepancy measure of the replicated data is more extreme than the observed data. If there is a good fit of the model to the data, we would expect the observed data to be similar to the replicated data, resulting in a Bayesian p-value of 0.50 while values near 0 or 1 indicate that the model does not fit the data.

There are many possible types of discrepancy measures including the Freeman-Tukey, standardized Pearson residual, chi-square, and deviance statistics (Brooks et al. 2000, Lunn et al. 2012). Residuals measure the difference between observed and fitted data. The standardized Pearson residual is one measure of this difference and is expressed by

$$r_i(\theta) = \frac{y_i - E(y_i \mid \theta)}{\sqrt{Var(y_i \mid \theta)}} \qquad (3)$$

where r_i is an individual residual, y_i is an individual data point, and $E(y_i|\theta)$ is the fitted value of for y_i based on the function to determine the parameters θ. We used standardized Pearson residuals to assess GOF in hierarchical binomial models following Kery (2010). To assess the GOF for redd densities, we used the Freeman-Tukey statistic (Brooks et al. 2000). Our binned redd count data consisted of many zero counts and this test statistic does not require the pooling of bins with small or zero values. The Freeman-Tukey statistic is expressed as

$$d_i(\theta) = \sqrt{y_i} - \sqrt{E(y_i \mid \theta)} \qquad (4)$$

where d_i is an individual discrepancy, y_i is an individual data point, and $E(y_i|\theta)$ is the fitted value of y_i based on the function to determine the parameter θ. When estimating independent values, such as the proportion of hatchery fish in a single population, Bayesian *p*-values are typically near 0.5. Therefore, we conducted GOF tests for hierarchical estimates and not independent estimates. Although Bayesian *p*-values are commonly used for model checking, there have been criticisms of this approach. First, it uses the data twice to build and check the model, which may not be as robust as other methods for testing model adequacy (Carlin and Louis 2009, Kery 2010). Second, it is unclear what cut off values to use for the interval (5% to 95%) to indicate lack of model fit. Third, the posterior distribution is influenced by the prior distribution, thus a Bayesian *p*-value is influenced by the prior distribution (Brooks et al. 2000). These concerns have addressed, but are beyond the scope of this paper (Gelman et al. 2004, Carlin and Louis 2009, and Brooks et al. 2000). Due to these concerns, we used posterior predictive model checking as a qualitative measure of model adequacy—if a Bayesian *p*-value indicated the model did not fit the data, we considered this to indicate significant lack of model fit (Link and Barker 2010).

In some cases, we tested the probability that one estimate was greater than another. These test included if tag loss was greater in males than females, if females density was greater than the density needed to seed habitat, if the escapement estimate was greater than the forecast, and greater than NOAA proposed occupancy rate (Brewin et al. 1995, Zimmerman 2010, and Crawford and Rumsey 2010). In these cases we monitored the difference between these two variables and assigned a value of 1 when the first estimate was higher than the second for each iteration. The proportion of times the first estimate is higher than the second estimate is the sum of the ones divided by the total iterations. We refer to this probability as a *p*-value, which is different than the Bayesian *p*-value described above.

In our coho salmon study, data were sparse and thus formal model selection techniques were unlikely to be very informative. Therefore, model development relied more on our knowledge of Lower Columbia River (LCR) coho salmon biology and population dynamics than formal model selection (Mäntyniemi and Romakkaniemi 2002).

Trap & Haul Escapement Estimates

The coho salmon abundance estimate for unmarked adults was simply the number of unmarked coho salmon trapped, hauled, and released into the upper NF Toutle, Tilton, and Upper Cowlitz/Cispus rivers. There were no marked adults released into the upper Toutle River. In the Tilton and Upper Cowlitz/Cispus rivers a recreational fishery occurred for marked coho salmon. All anglers retaining a marked coho salmon are required to record the fish on a CRC. At the end of the season, CRCs are returned to WDFW. However, successful anglers are more likely to return CRCs than unsuccessful anglers (Bob Leland, WDFW, pers. comm.). To account for this bias, WDFW contacts a random sample of anglers not returning their CRC by mail and they are reminded to turn in their CRC. Phone calls are then made to a random set of anglers receiving the reminder that still did not return their CRC in order to obtain their harvest information (Eric Kraig, WDFW pers. comm.). For each month the mean catch and variance are estimated (Kraig 2013). To obtain the total marked catch, the means and variances are summed. Therefore, we estimate the marked catch of adults and jacks by

$$Ad\_Catch_j \sim Normal(a\mu_j, asd_j) \tag{5}$$

$$J\_Catch_j \sim Normal(j\mu_j, jsd_j) \tag{6}$$

where *Ad_Catch* and *J_Catch* is the estimated catch assuming a normal distribution, $a\mu$ and $j\mu$ are the means for the adult and jack marked catch, *asd* and *jsd* are the standard deviation for the adult and jack marked catch, and *j* is an index for the Tilton and Upper Cowlitz/Cispus populations. The escapement of marked adults and jacks is the number trapped and hauled minus the catch

$$ATHm_j = AHm_j - Ad\_Catch_j \tag{7}$$

$$JTHm_j = JHm_j - J\_Catch_j \tag{8}$$

where *ATHm* and *JTHm* is the estimated adult and jack escapement and *AHm and JHm* are the number of transported adults and jacks, respectively.

Mark-Recapture Adult Escapement Estimates

Adult salmon escapement estimates were made using Peterson mark-recapture methods in Duncan, Abernathy, and Cedar creeks. The tagging event occurred near the mouth and the recovery events consisted of recoveries of live fish at adult traps upstream of the tagging site in Abernathy and Cedar creeks, carcass recoveries in all three creeks, and resighting of live fish during scheduled spawning ground surveys. Ignoring the resighting data, a weekly Darroch table was developed based on the number of tags and recoveries. Due to low recovery rates of jacks, they were addressed using other methods. Darroch's estimator (Darroch 1961, Seber 1982) can be expressed by

$$f(d.\theta_{ij}, d.p_j \mid d.r_{ij} \mid d.m_i) = \prod_{i=1}^{s} \left\{ \frac{d.m_i!}{(d.m_i.)!\prod_{j=1}^{s} d.r_{ij}!} \left(1 - \sum_j d.\theta_{ij} d.p_j\right)^{d.m_i - d.r_i.} \prod_{j=1}^{s} \left(d.\theta_{ij} d.p_j\right)^{r_{ij}} \right\} \tag{9}$$

$$d.u_j \sim Binomial(d.p_j, d.U_j) \tag{10}$$

where *s* is the number of strata, $d.m_i$ is the number of tags released in stratum *i*, $d.r_{ij}$ is the number of $d.m_i$ tags recovered in stratum *j*, $d.u_j$ is the number of fish in stratum *j*, $d.\theta ij$ is the probability that a tag from $d.m_i$ moves to stratum *j*, $d.r_i. = \sum d.r_{ij}$. Since the population is closed the $d.\theta_{ij}$ for each stratum *j* must sum to one. The total population is the sum of the $d.U_{j.}$

The minimum value was the number of tagged fish released and maximum value was sufficiently large that it did not constrain the upper bound of the population estimate ($d.U_j$) and expressed by

$$d.p_j \sim Beta(a, b) \tag{11}$$

$$d.U_j \sim Uniform(\min, \max) \tag{12}$$

$d.\theta_{ij}$ ~*Dirichlet*(*a*,*a*,…*a*) for each startum *j*. (13)

A series of null hypothesis tests were conducted for each trap site. The first null hypothesis tested was that there was no difference in tag recoveries by sex. Failing to reject this hypothesis allows for the pooling of males and females into an adult category. The next two null hypotheses are that there was no difference in the proportion of tag recoveries by time period and no difference in the proportion of tag recoveries by tag group. The pooled Petersen estimator is an unbiased estimator if not more than one of these null hypotheses are rejected (Schwarz and Taylor 1998). Finally, we tested the null hypothesis that there was no difference in the proportion of marked fish by recovery type (live fish at adult trap, carcasses, and resighting). If this null hypothesis was not rejected all fish data were used in a pooled Petersen estimator. Since an individual fish may be recaptured more than once, a Binomial distribution version of the Petersen model (Seber 1982, Korman et al. 2002) was used to estimate abundance by

$$rb_h \sim Binomial(q_h, tb_h) \tag{14}$$

$$cb_h \sim Binomial(q_h, Nb_h) \tag{15}$$

where *tb*, *rb*, and *cb* are the number of tagged, recaptured or resighted fish, and fish captured or observed in the second sample, respectively. The recapture efficiency and the population estimate are denoted by *q* and *Nb* and estimated by:

$$q_h \sim Beta(a, b) \tag{16}$$

$$Nb_h \sim Uniform(\min, \max). \tag{17}$$

Tag loss will positively bias mark-recapture estimates because tag loss causes and under reporting of recaptures (Rajawani and Schwarz 1997). We assessed tag loss at Duncan and Abernathy creeks with double tagging experiments. This experimental design assumes that loss of each tag is independent. From Seber (1982), we expanded the two-tag loss model to account for loss of three tags by

$$M = \sum_{i=1}^{n} m_i \tag{18}$$

$$m_i \sim Multinomial(ptr_i, M) \tag{19}$$

$$ptr_1 = ((1-otl)*(1-ftl)^2)/(1-ftl*ftl*otl) \tag{20}$$

$$ptr_2 = ((1-otl)*2ftl*(1-ftl))/(1-ftl*ftl*otl) \tag{21}$$

$$ptr_3 = ((1-otl)*ftl^2)/(1-ftl*ftl*otl) \tag{22}$$

$$ptr_4 = (otl*(1-ftl)^2)/(1-ftl*ftl*otl) \tag{23}$$

$$ptr_5 = (otl*2ftl*(1-ftl)/(1-ftl*ftl*otl) \tag{24}$$

$$p1plus = 1-(otl*ftl*ftl) \tag{25}$$

$$otl \sim Beta(a, b) \tag{26}$$

$$ftl \sim Beta(a, b) \tag{27}$$

where *otl* is the probability of losing an opercle tag (ot), *ftl* is the probability of losing a Floy tag (ft), m_1 are the recoveries with 1 ot and 2 ft, m_2 are the recoveries with 1ot and 1 ft, m_3 are the recoveries with 1ot, m_4 are the recoveries with 2 ft, m_5 are the recoveries with 1ft, the ptr_i are the probability of tag retention given the number and type of tags recovered, and $p1plus$ is the probability of retaining at least one tag. The prior distribution for the probability of losing a tag (*otl* and *ftl*) was a Beta distribution. For Duncan and Abernathy creeks, *ftl* was estimated by:

$$m \sim Beta(ftl, M) \qquad (28)$$

where *m* is the total number of Floy tags recovered, *M* is the total number of fish sampled with opercle punches multiplied by two. The same Beta prior was used for *ftl*.

Expanded Smolts Abundance Estimates

Weekly redd surveys were not conducted over the entire spawning area in Mill and Germany creeks so adult abundance in these watersheds was estimated by applying the smolt-to-adult return rate (SAR) from neighboring Abernathy Creek to smolt estimates from these basins. Smolt estimates are available for Mill, Germany, and Abernathy Creeks, and a mark-recapture estimate is available for adults in Abernathy Creek. The Abernathy Creek SAR was estimated for Abernathy Creek by

$$Ab\_SAR = AMRum_2 / Ab\_smolts \qquad (29)$$

where *Ab_SAR* is the Abernathy Creek smolt to adult return rate, *Ab_smolts* is the estimated smolt outmigration in 2009, and $AMRum_2$ is the mark-recapture estimate of adult abundance for unmarked fish. The unmarked adult abundance for Mill and Germany creeks is estimated by

$$ASum_f = smolts_f / Ab\_SAR \qquad (30)$$

where *ASum* is the adult abundance estimate using the smolt expansion method and *smolts* is the estimated smolt abundance from Mill and Germany creeks in 2009 based on a stratified estimator (Volhardt et al. 2007).

Redd based Abundance Estimates

To estimate the adult coho spawning escapement, the following estimates are required: 1) the number of redds per female, 2) the proportion of adult spawners that are females, and 3) the total number of redds in the population. In Duncan and Abernathy creeks we estimated the total abundance based on a mark-recapture study above trapping sites located at the mouth of these creeks (equations 9, 10, 14 and 15). Morphometric characteristics of tagged live fish and carcass recoveries of untagged fish were used to estimate the proportion of females and the number of female spawners by

$$FMR_h \sim Binomial(pFMR_h, AMR_h) \qquad (31)$$

$$NbF_h = pFMR_h * Nb_h \qquad (32)$$

$$pMC_k \sim Beta(a, b) \qquad (33)$$

where *FMR* and *AMR* are the number of unique females and adults sampled in the mark-recapture study, respectively, while *pFMR* is the proportion of female adults in the mark-recapture study, and *NbF* is the estimated number of female spawners in the population. The redd counts and female spawners from Duncan and Abernathy creeks were summed and the redds per female was estimated using the Binomial distribution with a Beta distribution for the proportion of females by

$$MRF = \sum_{h=1}^{2} NbF_h \tag{34}$$

$$DA\_redds \sim Bin(RpF, MRF) \tag{35}$$

$$RpF \sim Beta(a, b) \tag{36}$$

where *MRF* is the sum of the mark-recapture estimate of females in Duncan and Abernathy creeks, *DA*_redds is the sum of the Duncan and Abernathy creek redd counts, and *RpF* is the estimated number of redds per female.

For each one mile redd survey reach, the sum of the new redds counted was the redd density for that reach. To estimate the redd density for the sampled reaches, parametric statistics were not considered due to concerns about the lack of fit using standard sampling theory (Courbois et al. 2008). The starting point for analysis of count data is often the Poisson distribution. However, in the Poisson distribution the mean is equal to the variance, which is often an unrealistic assumption for count data. The negative binomial distribution is a more flexible distribution for the analysis of count data and allows for over dispersion in count data (Link and Barker 2010). The Poisson distribution is a special case of the negative binomial distribution as the over dispersion parameter approaches ∞ the Poisson distribution is recovered (Hilborn and Mangel 1997). Redd counts were modeled using a hierarchical negative binomial distribution, with an adjustment to accommodate WinBUGS parameterization by

$$y_{ik} \sim NegativeBinomial(p_k, r_k) \tag{37}$$

$$\mu_k = r_k * (1 - p_k / p_k) \tag{38}$$

where *y* is the number of redds in reach *i* for population *k*, with hyperparameters *p* and *r*. Both hyperparameters were assigned vague hyperpriors including

$$logit(p_k) \sim Normal(p\_mu, p\_sd) \tag{39}$$

$$r_k \sim Gamma(a_1, a_2) \tag{40}$$

$$p\_mu \sim Normal(0{,}100) \tag{41}$$

$$p\_sd \sim Uniform(0{,}100) \tag{42}$$

where the *a, b, p_mu*, and *p_sd* are the hyperpriors.

The redd density for each population is estimated by

$$\lambda_k \sim NegativeBinomial(p_k, r_k) \tag{43}$$

where lambda is the redd density. Ntzoufras (2009) noted that the dispersion index is equal to Var(Y)/E(Y). This is estimated by

$$DI_k = 1/p_k \tag{44}$$

where DI is the dispersion index and p is the hyperparameter of the negative binomial distribution. Since by definition the variance equals the mean for the Poisson distribution, a dispersion index greater than one indicates support for the Negative Binomial over the Poisson distribution for each population, which was assessed with a Bayesian GOF test. The female and adult redd density, and the proportion of females are estimated by

$$FD_k = \lambda_k / RpF \tag{45}$$

$$AD_k = FD_k / pF_k \tag{46}$$

$$FC_k \sim Binomial(pF_k, AC_k) \tag{47}$$

where FD is the female density, AD is the adult density, pF is the proportion of females, FC is the number of female carcasses, and AC is the number of adult carcasses. We estimate p-values to estimate the probability that our observed female redd densities for each population was greater than the mode of the female density required to seed freshwater habitat from Bradford et al. (2000). The proportion of females was hierarchically modeled by

$$logit(pF_k) \sim Normal(pF\_mu, pF\_sd) \tag{48}$$

$$pF\_mu \sim Normal(0{,}100) \tag{49}$$

$$pF\_sd \sim Uniform(0{,}100) \tag{50}$$

where pF_mu and pF_sd are the hyperpriors.

The total escapement based on redd surveys was estimated by

$$AT_k = \left(y_k + yc_k + \sum_{i=1}^{mis\_miles} \lambda_k \right) / RpF \tag{51}$$

where y is the observed number of redds in GRTS reaches, yc is the number of redds in non GRTS reaches (typically index Chinook or chum salmon reaches), λ is the redd density, and mis_miles is the number of unsurveyed miles in the GRTS sampling frame from which to expand redd counts in GRTS reaches.

Estimates of Hatchery and Wild Adults

We used the carcasses collected during the stream surveys to estimate the proportion of marked and unmarked adults by

$$MC_k \sim Binomial(pMC_k, SC_k) \tag{52}$$

$$pUMC_k = 1 - pMC_k \tag{53}$$

$$pMC_k \sim Beta(a,b) \quad (54)$$

where MC_k is the number of marked adult carcasses sampled, SC_k is the number of sampled adult carcasses, pMC_k is the proportion of marked adults based on the carcasses sampled, and $pUMC_k$ is the proportion of unmarked adults. The estimated number of marked and unmarked adult coho salmon based on the stream surveys was estimated by

$$ARm_k = pMC_k * AT_k \quad (55)$$

$$ARum_k = pUMC_k * AT_k \quad (56)$$

where ARm_k is the estimate of marked adult coho and $ARum_k$ is the estimate of adult unmarked coho salmon. The same equations (52-53) were used to estimate the proportion of marked and unmarked adults in the mark-recapture studies except the subscript was used *h* instead of *k* to denote the difference between the mark-recapture and redd based proportions. Equations 55 and 56 were used to estimate the marked and unmarked adult abundance in the mark-recapture estimates.

Combining Data Sources to Generate Population Estimates

Finally, the adult marked and unmarked abundance estimates from redd, mark-recapture, and trap and haul methods were summed as needed to estimate population abundance. These population abundances were summed to estimate the total adult coho salmon abundance below Bonneville Dam except for areas not sampled including Salmon Creek and the mainstem NF Lewis, Toutle, lower NF Toutle, and Cowlitz Rivers.

Estimating the Proportion of Males that are Jacks

Due to differential capture probabilities between jack and adult coho salmon carcasses during spawning ground surveys, we applied an aggregate estimate of the proportion of males that were jacks obtained from weirs and trap and haul operations to areas where redd counts were used. We used a hierarchical model to estimate the proportion of male coho salmon that were jacks based on trap data at Cedar Cr, Barrier Dam, and TFCF, and we used the subscript *g* to denote these 4 groups. The proportion of jacks was estimated by

$$TJ_g \sim Binomial(pJ_g, TM_g) \quad (57)$$

$$logit(pF_g) \sim Normal(pJ\_mu, pJ\_sd) \quad (58)$$

$$pJ\_mu \sim Normal(0,100) \quad (59)$$

$$pJ\_sd \sim Uniform(0,100) \quad (60)$$

where TJ_g is the count of jacks at a trap, TM_g is the number of trapped males, pJ_g is the proportion of jacks. We used the same hierarchical equations and priors based on the logit-normal distribution as in equations 48-50. The jack abundance for each population was estimated by

$$Jtot_k = AT_k * (1 - pF_k) * mean(hier\_pJ) \quad (61)$$

where $Jtot_k$ is the estimate of jacks within the population where AT_k is the adult abundance from the redd or mark-recapture estimate, pF_k is the proportion of females, and *mean(hier_pJ)* is the mean of the hierarchical estimate of the proportion of males that are jacks. .

Estimating the Proportion of One-Mile Reaches Occupied by Coho Salmon Spawners

The spawning reach occupancy rate of coho salmon was based on the redd surveys and estimated by

$$Oc_k \sim Bin(pOc_k, m_k) \qquad (62)$$

where OC_k is the number of reaches in which at least one redd was observed, m_k is the number of reaches, and pOC_k is the percent of reaches occupied. In addition, we estimated the probability that 80% of the reaches in a population were occupied by recording the number of iterations the occupancy rate exceeded 80% by

$$p80Oc_k = pOc_k - 0.80 \qquad (63)$$

where $p80OC_k$ is the probability that 80% of the surveyed reaches were occupied.

Results

Model convergence and diagnostics
We ran two chains with a thinning rate of 10 using the Gibbs sampler in WinBUGS. After discarding the 5,000 burn-in iterations, a total of 20,000 iterations for the posterior distribution of each parameter were saved. Visual inspection of the history plots suggested the chains mixed and converged. The BGR diagnostic test for convergence yielded values of less than 1.02 for each parameter, which is less than the recommended value of 1.1. While it is impossible to conclusively demonstrate a simulation has converged, the above diagnostic tests did not detect that the simulations did not converge. The MCSE was 1% of the standard deviation of the parameter estimates, which suggests our posterior distributions were accurate. In addition, the estimates effective number of parameters ranged from 2,400 to 20,000 for all monitored parameters but for the reporting parameters the minimum was 6,500, suggesting sufficient iteration for accurate estimates of 95% CI. It should be noted that some numbers of the report may not sum due to rounding errors.

We tested the sensitivity of our analysis based on the various priors. We used three vague priors for the beta distribution ($\alpha = \beta = 0.5$, 1, and 0.1), which correspond to Jeffreys, LaPlace-Bayes, and Haladine priors. We used vague hyper-priors for the binomial and negative binomial hierarchical models based on the gamma distribution (0.001, 0.001) and normal distribution (0, 0.001) for the mean, and a uniform distribution (0, 100) for the standard deviation for logit-normal model. Our results were not sensitive to the priors or hyperpriors except when we had few observations, such as tag loss in Duncan Creek. In addition for hierarchical models, the logit-normal provided slightly better mixing than the Gamma distribution. Our population abundance estimates were similar for all priors and the results reported here are based on Jeffreys prior for the beta distribution, and the logit-normal priors for the hierarchical binomial and negative binomial models mentioned above.

Trap and Haul Abundance Estimates
A total of 202 unmarked adult and 14 jack coho salmon were collected and released above the TFCF on the NF Toutle River (Table 2). These numbers are the total escapement above the TFCF. A total of 3,616 and 207 marked adult and jack coho salmon, respectively, were collected at Barrier Dam on the Cowlitz River and released into the Tilton River. Subtracting the fishery impacts for mass marked coho salmon left a mean Tilton River escapement of 2,523 and 194 marked adults and jacks, respectively. In addition, 978 unmarked adults and 99 unmarked jacks were released in the Tilton and were not available for harvest, so were assumed to have spawned. For the Cowlitz/Cispus population a total of 21,008 and 2,130 marked adults and jacks, respectively, were captured at Barrier Dam and released above Cowlitz Falls Dam (CFD). Subtracting the expanded CRC catch of marked coho salmon leaves a mean escapement of 18,839 and 2,102 marked adults and jacks, respectively. Since we assumed no fishery impacts for unmarked fish, the Upper Cowlitz/Cispus escapements were the same as the release totals of 2,906 and 203 unmarked adults and jacks, respectively.

The proportion of male coho salmon that were classified as jacks was relatively consistent and ranged from 8.5% in Cedar and Abernathy creeks to 12.1% at the Barrier Dam on the Cowlitz River (Table 3). The Bayesian *p*-values ranged from 0.503 to 0.607 for these four populations based on an analysis of Pearson's residuals, which do not indicate any problems with the GOF test for these data using the hierarchical model. The mean proportion of males that were jacks based on the hierarchical model for the trap data was 10.5%.

Table 2. Trap and haul counts at North Toutle Fish Collection Facility (TFCF), Cowlitz Barrier Dam counts transported to the Tilton and Upper Cowlitz/Cispus Rivers, estimate of recreational harvest of marked fish, and marked and unmarked escapement.

Parameter	mean	sd	2.50%	median	97.50%
Toutle FCF Unmarked Adult Release	202				
Toutle FCF Unmarked Jack Release	14				
Upper Cowlitz Unmarked Adult Release	2,906				
Upper Cowlitz Unmarked Jack Release	203				
Tilton Unmarked Adult Release	978				
Tilton Unmarked Jack Release	99				
Upper Cowlitz Marked Adult Release	21,008				
Upper Cowlitz Marked Jack Release	2,130				
Tilton Marked Adult Release	3,616				
Tilton FCF Marked Jack Release	207				
Upper Cowlitz Marked Adult Catch	2,169	125	1,922	2,170	2,413
Upper Cowlitz Marked Jack Catch	31	13	6	30	57
Tilton Marked Adult Catch	1,093	86	923	1,092	1,260
Tilton Marked Jack Catch	13	7	1	13	29
Tilton Marked Adult Escapement	2,523	86	2,356	2,524	2,693
Tilton Marked Jack Escapement	194	7	178	194	206
Upper Cowlitz Marked Adult Escapement	18,839	125	18,590	18,840	19,090
Upper Cowlitz Marked Jack Escapement	2,102	13	2,076	2,103	2,127

Table 3. Estimates of the proportion on male coho salmon that are jacks from trap data at the North Toutle Fish Collection Facility (TFCF), Cowlitz at Barrier dam, Abernathy trap and Cedar trap.

Subpopulation	mean	sd	2.50%	median	97.50%
Proportion of jacks (Toutle FCF)	12.1%	0.2%	11.8%	12.1%	12.4%
Proportion of jacks (Cowlitz-Barrier Dam)	9.0%	1.8%	5.5%	8.9%	12.6%
Proportion of jacks (Abernathy)	8.8%	1.2%	6.5%	8.7%	11.3%
Proportion of jacks (Cedar)	11.0%	2.3%	7.0%	10.8%	16.0%
Mean proportion of jacks	10.5%	4.5%	5.4%	10.2%	17.2%

Adult Mark-Recapture: Assumption Tests

We conducted a series of hypothesis tests to test the equal capture assumption of the Petersen estimator for the Duncan, Abernathy, and Cedar populations using Bayes Factors. There was positive support for the null hypotheses that there was no difference in recapture probabilities or proportion of marked fish recovered by sex in the second sample for all Duncan and Abernathy populations (Table 4). There was negligible support for different recovery probabilities of marks by period at Abernathy Creek. These results of these diagnostic null hypothesis tests indicate a pooled Petersen estimator is an appropriate model to estimate adult coho salmon abundance in Abernathy and Duncan creeks.

For Cedar Creek there was also positive support for the null hypotheses that there were no differences in recapture probabilities or proportion of marked fish recovered by sex in the second sample, but there was very strong positive support that recovery probabilities of marks by period at Cedar Creek were different (Table 5). Recovery probabilities at Cedar Creek were stratified into early and late periods and there was positive and very strong support for the null hypothesis of no difference in recapture probabilities of marks within each recovery period. Therefore, we estimated the adult coho salmon abundance in Cedar Creek using the Darroch estimator stratified by early and late periods.

Table 4. Tests for violations of the equal capture probability assumption for the Petersen estimator using Bayes Factors (BF) in Duncan and Abernathy creeks.

Population	Null Hypothesis	BF	Comments
Duncan	Ho: no difference in the recovery probability of marks by sex	0.55	Negligible support against Ho
Duncan	Ho: no difference in the proportion of recaptured carcasses at the weir & on spawning grounds, & live fish on spawning grounds	1.32	Negligible support for Ho
Abernathy	Ho: no difference in the recovery probability of marks at AFTC by early and late periods	0.62	Negligible support against Ho
Abernathy	Ho: no difference in the recovery probability of marks by sex at AFTC	3.04	Positive support for Ho
Abernathy	Ho: no difference in the recovery probability of marks by sex as weir wash-ups	1.91	Negligible support for Ho
Abernathy	Ho: no difference in the recovery probability of marks by sex as carcasses	3.37	Positive support for Ho
Abernathy	Ho: no difference in the proportion of recaptured carcasses at the weir & on spawning grounds, & live fish at AFTC & on spawning grounds	14.50	Positive support for Ho

Results for the tag loss and retention study are found in Table 6. The mean probability of losing an opercle tags was less 3.4%. The mean probability of losing a Floy tag was less 2.4 to 7.3% for females, but ranged from 7.0 to 15.3% for males. The probability that a fish retained at least one of it tags ranged from 97.1 to 100%. The probability that females retained their opercle tag at a higher rate than males was 35.0% (Cedar Creek only) but for Floy tags this probability was greater; 90.0% in Abernathy Creek and 96.7% in Cedar Creek. Although males appeared to lose more tags than females, tag retention on the whole was very high, and consequently, mark-recapture abundance estimates were not adjusted for tag loss.

Table 5. Tests for violations of the equal capture probability assumption for the Petersen estimator using Bayes Factors (BF) in Cedar creek.

Population	Null Hypothesis	BF	Comments
Cedar	Ho: no difference in the recovery probability of marks at the weir during the ten periods	-3.7E+13	Very strong support against Ho
Cedar	Ho: no difference in the recovery probability of marks at the weir in the early period	2.11	Positive support for Ho
Cedar	Ho: no difference in the recovery probability of marks at the weir in the late period	6.2E+05	Very strong support for Ho
Cedar	Ho: no difference in the recovery probability of marks at the weir by sex	14.70	Positive support for Ho
Cedar	Ho: no difference in the proportion of recaptured fish at the weir by sex	2.12	Positive support for Ho

Table 6. Tag loss and retention for Floy and opercle tags at Cedar, Abernathy, and Duncan creeks.

Subwatershed	Tag Type	Sex	mean	sd	2.50%	median	97.50%
Probability of losing a tag							
Cedar Cr.	Opercle	Male	2.5%	2.0%	0.2%	1.9%	7.6%
Cedar Cr.	Opercle	Female	3.4%	2.1%	0.6%	3.0%	8.4%
Cedar Cr.	Floy	Male	7.0%	2.3%	3.2%	6.8%	12.0%
Cedar Cr.	Floy	Female	2.4%	1.3%	0.6%	2.2%	5.4%
Duncan Cr.	Floy	Male	NA	NA	NA	NA	NA
Duncan Cr.	Floy	Female	7.3%	9.3%	0.0%	3.7%	33.7%
Abernathy Cr.	Floy	Male	15.3%	7.4%	3.9%	14.2%	32.5%
Abernathy Cr.	Floy	Female	5.6%	4.3%	0.4%	4.5%	16.6%
Probability of retaining at least one tag							
Cedar Cr. (3 tags)	Both	Male	100.0%	0.0%	100.0%	100.0%	100.0%
Cedar Cr. (3 tags)	Both	Female	100.0%	0.0%	100.0%	100.0%	100.0%
Duncan Cr. (2 tags)	Floy	Female	98.6%	3.4%	88.7%	99.9%	100.0%
Abernathy Cr.	Floy	Male	97.1%	2.8%	89.5%	98.0%	99.8%
Abernathy Cr.	Floy	Female	99.5%	0.8%	97.3%	99.8%	100.0%

Adult Mark Recapture Results

The tagged adult recovery efficiency, which included resighted fish, was over 36% in Duncan and Abernathy Creeks. The overall recovery efficiency in Cedar Creek, which included only weir recoveries, was much lower (12.2%), but high in the early portion of the run (74.6%) and very low in the later portion of the run (7.4%). The adult abundance estimates ranged from a low of 59 in Duncan Creek to 2,180 in Cedar Creek. The proportion of unmarked adults was high in both Abernathy (87.5%) and Cedar Creek (94.3%). The estimates of marked and unmarked adult abundance with 95% CI for these creeks are found in Table 7. The number of female spawners in Duncan and Abernathy creeks was 27 and 269, respectively. The proportion of females was similar at both locations (Table 7). The estimate of redds per female for our study was 0.471 (95%CI 0.351 - 0.607). This estimate (0.471) equates to a detection efficiency (number of redds observed out of those actually constructed) of 47% if we assume one redd per female coho salmon.

Table 7. Results for mark-recapture populations in 2010 including estimates of mark-recapture tag recovery efficiency, total, unmarked, and marked adult escapement, proportions of marked, unmarked spawners, and female escapement, proportion of females and redds per female.

Parameter	mean	sd	2.50%	median	97.50%
Mark Recovery Efficiency					
Abernathy Creek	56.4%	5.9%	44.5%	56.4%	67.6%
Cedar Creek (Early)	74.6%	17.2%	42.9%	75.3%	99.6%
Cedar Creek (Late)	7.4%	0.9%	5.8%	7.4%	9.2%
Duncan Creek	36.5%	8.6%	21.7%	35.9%	54.8%
Adult Escapement					
Abernathy Cr. (Total)	553	64	449	546	700
Abernathy Cr.(Unmarked)	486	58	393	480	618
Abernathy Cr. (Marked)	67	15	42	66	99
Cedar Cr. (Total)	2180	284	1685	2157	2791
Cedar Cr. (Unmarked)	2056	266	1592	2035	2631
Cedar Cr. (Marked)	124	35	67	120	201
Duncan Cr. (Total)	59	16	34	56	94
Proportion of Marked and Unmarked Adults					
Abernathy Cr.(Unmarked)	87.9%	2.2%	83.2%	88.0%	91.9%
Abernathy Cr. (Marked)	12.1%	2.2%	8.1%	12.0%	16.8%
Cedar Cr. (Total)	94.3%	1.3%	91.5%	94.4%	96.6%
Cedar Cr. (Unmarked)	5.7%	1.3%	3.4%	5.6%	8.5%
Female Escapement, Proportions, and Redds per Female					
Duncan Cr. Female Escapement	27	9	13	26	49
Abernathy Cr. Female Escapement	269	37	207	265	350
Proportion of Females (Duncan)	45.9%	9.8%	27.3%	45.8%	65.5%
Proportion of Females (Abernathy)	48.6%	3.4%	41.9%	48.6%	55.4%
Redds per Female (Duncan & Abernathy)	0.471	0.065	0.351	0.469	0.607

SAR Expansion Adult Estimates

The 2009 coho salmon smolt emigration estimates ranged from 2,634 in Germany Creek to 7,023 in Mill Creek (Table 8; Kinsel et al. 2009). The estimate in Cedar Creek was an order of magnitude higher (69,393) than the other estimates (Zimmerman 2010). The 2009 smolt to adult return (SAR) ranged from a low of 3.0% in Cedar Creek to a high of 12.1% in Abernathy Creek. The SAR for Cowlitz adult coho salmon was 3.3% and the mean SAR was 6.1%. Coho salmon escapement based on smolt abundance and Abernathy Cr. SAR in Mill and Germany creeks was 968 and 363 adults, respectively (Table 8). Unmarked and marked adult abundance in these basins based on the proportion of marked adults in Abernathy Cr. is also provided in Table 8.

Table 8. Estimates of 2009 smolt abundance and Smolt to Adult Return rate (SAR), and 2010 total, unmarked, and marked coho spawner abundance based on SAR expansions.

Parameter	mean	sd	2.50%	median	97.50%
2009 Smolt Abundance Estimate					
Abernathy Cr.	4020	240	3549	4020	4491
Germany Cr.	2634	201	2240	2634	3028
Mill Cr.	7023	291	6452	7023	7594
Cedar Cr.	69393	3480	62572	69393	76214
2009 Smolt to Adult Return					
Cowlitz R. (Mayfield)	3.3%	0.1%	3.1%	3.3%	3.5%
Abernathy Cr.	12.1%	1.6%	9.5%	11.9%	15.8%
Cedar Cr.	3.0%	0.4%	2.3%	2.9%	3.9%
Mean Smolt to Adult Return	6.1%	0.6%	5.2%	6.1%	7.4%
2009 Adult Escapement Estimates based on SAR Expansions					
Germany Cr. (Total)	363	55	271	357	487
Germany Cr. (Unmarked)	319	49	237	314	431
Germany Cr. (Marked)	44	11	27	43	67
Mill Cr. (Total)	968	134	747	954	1272
Mill Cr. (Unmarked)	851	119	653	838	1122
Mill Creek (Marked)	118	27	72	115	178

Redd Based Estimates

A total of 183 reaches across 13 populations were surveyed as part of the GRTS design (Appendix 1). The number of sites ranged from 2 to 24 per population and averaged 13 (Appendix 1). The mean dispersion index for the redd data ranged from 8.68 to 15.2 and the lower 95% CI exceeded 2.65, which all exceeded the expected dispersion index of 1, which is consistent with the Poisson distribution (Table 9). The GOF test indicated there was a less than a 0.09% probability that the replicated dispersion index based on the Poisson model was more extreme than the dispersion index based on the observed data. This provides strong evidence that the data are not consistent with the Poisson distribution.

Table 9. The estimated dispersion index for the negative binomial distribution from GRTS surveys in 2010. The last column is a Bayesian *p*-value for a GOF test to measure if the dispersion index is less than 1, which would favor the Poisson distribution.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%	GOF
Grays	9.60	4.08	3.39	9.34	18.51	0.000
Elochoman	12.46	4.58	6.67	11.62	23.22	0.000
L Cowlitz	10.66	3.54	5.32	10.25	18.86	0.000
Coweeman	13.10	4.21	7.50	12.32	23.42	0.000
Toutle Tribs	11.11	7.17	2.65	10.35	25.56	0.009
Green	14.62	4.70	8.69	13.57	26.44	0.000
SF Toutle	18.15	12.05	8.69	14.92	45.80	0.000
Kalama	13.55	9.72	5.42	11.69	32.59	0.000
NF Lewis	15.21	11.32	6.53	12.71	38.94	0.004
Cedar	8.68	3.14	3.89	8.35	15.47	0.000
EF Lewis	11.89	5.29	5.40	11.02	24.68	0.000
Washougal	13.75	8.05	6.22	12.07	32.13	0.000
L Gorge	12.80	8.21	4.60	11.29	31.34	0.000

The hierarchical modeled redd densities followed a highly skewed (right-tailed) distribution, resulting in a mean being greater than the median. The observed mean redd density (range 1.73 to 13.07 (Table 9)). The NF Lewis and the Coweeman populations had Bayesian *p*-values of 0.947 and 0.932 indicating some lack of fit. We sampled only two sites from the NF Lewis population (where hatchery fry releases occurred) and found 8 and 24 redds at each site, respectively, which may explain the poorer fit for this population. For the Coweeman population there were fewer observed counts of low numbers (0, 1, 2, etc.) than expected under the Negative Binomial distribution. However, the Bayesian *p*-values for the hierarchical Negative Binomial model for count data ranged from 0.235 to 0.947, indicating no significant lack of fit for the GOF test based on Freeman-Tukey test statistics.

Based on the mark recapture estimates and redd census, we expanded the redd counts by 0.471 observed redds per female (Table 7) to convert the estimated redds to females (Table 11). The females per mile ranged from a low of 3.75 to 28.39. Based on a meta-analysis, Bradford et al. (2000) found the mode of female coho salmon per mile needed to seed freshwater habitat was 15 females. The probability of our population estimates exceeded 15 females per miles ranged from 0.049 to 0.590.

Table 10. Observed coho salmon redds per mile based on the negative binomial distribution from GRTS surveys in 2010. The last column is a Bayesian *P* -value for a GOF test.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%	GOF
Grays	2.18	4.86	0	0	16	0.470
Elochoman	5.84	8.95	0	2	31	0.444
L Cowlitz	3.80	6.81	0	1	23	0.539
Coweeman	13.07	13.35	0	9	48	0.932
Toutle Tribs	1.21	4.33	0	0	12	0.235
Green	12.95	14.44	0	8	52	0.743
SF Toutle	5.04	10.70	0	1	34	0.588
Kalama	2.30	6.29	0	0	19	0.646
NF Lewis	12.47	18.04	0	7	56	0.947
Cedar	4.48	6.62	0	2	23	0.547
EF Lewis	5.05	8.61	0	2	28	0.521
Washougal	5.11	9.46	0	2	31	0.678
L Gorge	1.73	5.61	0	0	15	0.634

Table 11. The estimated number of coho salmon females/mile based on GRTS surveys in 2010. The p-value is the probability the observed female density is greater than the mode of the full habitat seeding density based on Bradford et al. 2000.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%	*p*-value
Grays	4.71	10.63	0.00	0.00	34.38	0.092
Elochoman	12.63	19.74	0.00	5.25	68.65	0.269
L Cowlitz	8.23	14.86	0.00	2.34	49.11	0.173
Coweeman	28.29	29.45	0.00	19.70	106.60	0.590
Toutle Tribs	2.63	9.45	0.00	0.00	25.30	0.049
Green	28.05	31.86	0.00	18.20	115.10	0.558
SF Toutle	10.93	23.44	0.00	2.18	72.09	0.212
Kalama	4.98	13.81	0.00	0.00	40.31	0.098
NF Lewis	27.05	39.86	0.00	14.64	123.20	0.494
Cedar	9.71	14.63	0.00	4.47	49.43	0.214
EF Lewis	10.91	18.81	0.00	4.07	61.50	0.229
Washougal	11.07	20.74	0.00	3.41	66.96	0.228
L Gorge	3.75	12.35	0.00	0.00	32.50	0.071

Using the hierarchical model we estimated the mean proportion of females among all adult coho was 44.6% based on carcass recoveries. Population-specific estimates ranged from 41.0% to 49.3% and all 95% credible intervals overlapped with 50%, which may be expected since the sex ratio should be near 1:1 (Table 12). The Toutle and Kalama populations had the most extreme GOF test values of 0.919 and 0.893. However, this is due to the few females observed in the few examined carcass 0/1 and 1/7 for the Toutle and Kalama populations, respectively. The GOF test based on Bayesian *p*-values ranged from 0.245 to 0.919 for the 13 populations, which indicated no concern with model fit.

Table 12. Estimates of the proportion of adult females in the 2010 population based on carcass recoveries during redd surveys. The last column is a Bayesian *p*-value for a GOF test.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%	GOF
Grays	45.0%	4.6%	35.5%	45.1%	54.3%	0.698
Elochoman	41.5%	5.6%	29.0%	42.3%	50.7%	0.444
L Cowlitz	49.3%	6.2%	39.3%	48.2%	64.1%	0.304
Coweeman	44.4%	4.2%	35.8%	44.6%	52.5%	0.649
Toutle Tribs	43.8%	7.4%	27.0%	44.4%	58.3%	0.919
Green	48.5%	3.1%	42.9%	48.3%	54.8%	0.455
SF Toutle	43.0%	3.9%	34.8%	43.3%	50.1%	0.552
Kalama	44.7%	6.4%	30.8%	44.9%	58.1%	0.893
NF Lewis	41.5%	7.3%	23.8%	42.7%	53.4%	0.245
Cedar	42.3%	4.6%	32.1%	42.7%	50.4%	0.525
EF Lewis	46.4%	6.0%	34.9%	46.1%	59.7%	0.642
Washougal	41.0%	6.4%	26.1%	42.2%	51.2%	0.343
L Gorge	49.1%	5.7%	39.9%	48.2%	62.4%	0.356
Mean Females	44.6%	3.2%	37.8%	44.7%	50.5%	

The female density estimates were expanded by the population-specific estimates of the proportion of females to estimate the adult densities (Table 13). The mean adults per mile ranged from a high of 68.24 in the Coweeman to a low of 6.16 in the Toutle tributaries. A total of 11 of 13 population estimates had mean adult densities greater than 10 per mile.

Table 13. Expanded coho salmon adults per mile based on GRTS surveys in 2010.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%
Grays	10.60	24.14	0.00	0.00	77.08
Elochoman	31.17	49.70	0.00	12.87	169.61
L Cowlitz	16.91	30.73	0.00	4.90	101.20
Coweeman	64.33	67.38	0.00	44.37	243.41
Toutle Tribs	6.16	22.11	0.00	0.00	59.41
Green	58.18	66.45	0.00	37.46	239.01
SF Toutle	25.63	55.12	0.00	5.11	170.50
Kalama	11.35	31.68	0.00	0.00	93.53
NF Lewis	68.24	107.78	0.00	35.78	318.10
Cedar	23.26	35.53	0.00	10.67	119.40
EF Lewis	23.86	41.37	0.00	8.70	135.41
Washougal	27.79	53.63	0.00	8.21	167.92
L Gorge	7.73	25.27	0.00	0.00	66.88

The adult densities were then expanded by the proportion of the sample frame surveyed to estimate the adult abundance (Table 14). Adult coho salmon abundance estimates followed variably skewed right-tailed distributions; the mean adult coho salmon abundance estimated from redd surveys ranged from a low of 255 for the tributaries of the mainstem Toutle population to a high of 7,783 adults for the Lower Cowlitz population.

Table 14. The estimated adult coho salmon escapement based on redd surveys in 2010.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%
Grays	2005	806	903	1859	3970
Elochoman	3093	1091	1603	2895	5750
L Cowlitz	7383	2594	3593	6966	13600
Coweeman	3915	1014	2364	3768	6311
Toutle Tribs	255	247	15	186	885
Green	3186	769	1991	3080	5000
SF Toutle	1920	991	761	1717	4301
Kalama	488	358	111	400	1376
NF Lewis	1924	1456	473	1586	5356
Cedar	695	198	409	661	1176
EF Lewis	2014	883	812	1840	4170
Washougal	1426	834	475	1251	3382
L Gorge	486	210	248	438	1014

Based on carcass surveys we estimated the percentage of marked (clipped adipose fin or CWT) and unmarked adult coho salmon. Populations with hatcheries such as the Grays, Elochoman, Green, Kalama, and Washougal Rivers had mean estimates of 81%, 73%, 67%, 99%, and 44% marked carcasses, respectively (Table 15). Other populations had lower marking rates including the Lower Cowlitz, Coweeman, Toutle, SF Toutle, EF Lewis, and the Lower Gorge, which had median estimates of 85%, 90%, 75%, 79%, 68%, and 71% unmarked fish, respectively (Table 16).

There was a low percentage of marked fish in the NF Lewis and Cedar Creek, but these basins are heavily supplemented with hatchery fry releases from intensive use of remote site incubators (RSI) (Table 17). Since these hatchery releases are not visually marked our estimates of unmarked fish are biased high in these basins. In addition, it should be noted that we reported on marked fish, which include hatchery fish that are adipose fin-clipped or CWT only, such as the NF Lewis River hatchery double index group. There are also a small percentage of hatchery fish that are released unmarked due to machine or human error during marking. Therefore estimates of the true proportion of hatchery fish would increase if adjusted for the percentage of unmarked hatchery fish.

Table 15. The estimated percentage of adult coho salmon that are marked (adipose fin clipped or CWT) based on carcass recoveries during GRTS surveys.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%
Grays	80.6%	5.6%	68.5%	81.1%	90.3%
Elochoman	73.0%	7.0%	58.1%	73.5%	85.5%
L Cowlitz	15.0%	6.4%	4.8%	14.3%	29.7%
Coweeman	9.9%	3.4%	4.2%	9.6%	17.6%
Toutle Tribs	24.7%	24.7%	0.0%	16.2%	85.0%
Green	66.7%	3.0%	60.6%	66.7%	72.5%
SF Toutle	20.9%	4.0%	13.7%	20.7%	29.2%
Kalama	99.1%	1.3%	95.3%	99.6%	100.0%
NF Lewis	6.2%	8.0%	0.0%	3.0%	29.2%
Cedar	0.8%	1.1%	0.0%	0.4%	3.9%
EF Lewis	32.3%	11.0%	13.1%	31.5%	55.3%
Washougal	44.2%	9.7%	26.0%	44.1%	63.3%
L Gorge	29.4%	7.2%	16.5%	29.1%	44.5%

Table 16. The estimated percentage of adult coho salmon that are marked (adipose fin clipped or CWT) based on carcass recoveries during GRTS surveys.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%
Grays	19.4%	5.6%	9.7%	19.0%	31.5%
Elochoman	27.0%	7.0%	14.5%	26.5%	41.9%
L Cowlitz	85.0%	6.4%	70.3%	85.7%	95.2%
Coweeman	90.1%	3.4%	82.4%	90.4%	95.8%
Toutle Tribs	75.3%	24.7%	15.0%	83.8%	100.0%
Green	33.3%	3.0%	27.5%	33.3%	39.4%
SF Toutle	79.1%	4.0%	70.8%	79.3%	86.3%
Kalama	0.9%	1.3%	0.0%	0.4%	4.7%
NF Lewis	93.8%	8.0%	70.8%	97.0%	100.0%
Cedar	99.2%	1.1%	96.1%	99.6%	100.0%
EF Lewis	67.7%	11.0%	44.7%	68.5%	86.9%
Washougal	55.8%	9.7%	36.7%	55.9%	74.0%
L Gorge	70.6%	7.2%	55.5%	70.9%	83.5%

Table 17. Off station hatchery releases (primarily from remote site incubators) of coho salmon expected to return in 2010.

Brood Yr	Month	River	Stage	UnMark	MassMark	Totals
2007	Mar	Cowlitz	Fry	234,100	0	234,100
2007	July	Cowlitz	Fingerling	0	1000	1000
2007	Mar	Lewis	Fry	810,000	0	810,000
2007	Mar	Salmon Cr.	Fry	93,400	0	93,400
2007	Apr	Salmon Cr.	Fry	860,000	0	860,000
						1,998,500

Estimates of marked and unmarked adult coho abundance for GRTS areas followed variably-skewed distributions and were generally right-tailed. The mean estimate of marked adult abundance ranged from 5 in Cedar Creek to 2,259 in the Elochoman (Table 18). Basin releasing hatchery fish had some of the highest number of marked fish especially the Grays, Elochoman, and Green. Mean estimates of unmarked adult abundance ranged from a low of 5 for the Kalama population to a high of 6,274 for the Lower Cowlitz population (Table 19). In the Cowlitz, Lewis, and Salmon Creek populations a total of 1,998,500 fry were released primarily from RSIs (Table 17). Since these hatchery fish are not externally marked, they are likely included in some of the unmarked samples in these populations and possibly other populations. At this time there is no straightforward method to determine the percentage of unmarked RSI releases in these populations.

Table 18. Estimated adult marked coho salmon abundance from 2010 GRTS surveys.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%
Grays	1617	662	714	1497	3267
Elochoman	2259	830	1120	2108	4305
L Cowlitz	1109	633	284	979	2704
Coweeman	388	173	139	357	810
Toutle Tribs	63	107	0	24	354
Green	2125	524	1314	2052	3361
SF Toutle	402	227	143	354	963
Kalama	483	355	110	396	1366
NF Lewis	121	219	0	45	681
Cedar	5	8	0	2	28
EF Lewis	651	373	177	568	1591
Washougal	631	400	184	543	1605
L Gorge	143	73	57	127	323

*The sum of abundance by marked status may not equal the abundance estimate due to rounding errors.

Table 19. Estimated adult unmarked coho salmon abundance from 2010 GRTS surveys.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%
Grays	388	197	133	347	881
Elochoman	834	371	333	764	1733
L Cowlitz	6274	2268	2950	5913	11690
Coweeman	3528	923	2113	3397	5687
Toutle Tribs	192	205	8	131	712
Green	1060	273	637	1021	1704
SF Toutle	1518	785	594	1354	3407
Kalama	5	9	0	2	27
NF Lewis	1804	1370	433	1480	5029
Cedar	689	197	407	656	1167
EF Lewis	1363	647	497	1238	2980
Washougal	795	495	242	688	1929
L Gorge	344	153	167	308	727

*The sum of abundance by marked status may not equal the abundance estimate due to rounding errors.

The percentage of GRTS reaches having at least one redd ranged from 16% for the Lower Gorge population to 95% for the Green population (Table 20). The high occupancy rate in the Green is probably a result of the hatchery program since many of the recovered carcasses in this basin were marked fish (Table 15). Other populations with high occupancy rates include Elochoman, NF Lewis, and Cedar, which also have intensive hatchery programs. Some populations with low hatchery influence, as measured by the percentage of marked fish, also had high occupancy rates including the Cowlitz and Coweeman. We calculated the probability that 80% of the reaches were occupied based on observed redd counts, which is the NOAA occupancy rate standard (Table 20). The Green, Coweeman and NF Lewis were the only populations for which there was a greater than 80% probability that 80% of reaches were occupied (Table 20). For most populations the probability that the occupancy rate was greater than 80% was near 0%, indicating that most populations were below the NOAA guideline. It should be noted that we are reporting on the observed occupancy rate based on redds. This is less than the true occupancy rate because our redd detection rate was about 47%, assuming 1 redd per female, and males and jacks were not included in the occupancy rate.

Table 20. Occupancy rate or the percentage of GRTS reaches that were occupied (had at least one redd) and the probability that the occupancy rate was above 80% (last column).

GRTS Area	mean	sd	2.50%	median	97.50%	*P*- value
Grays	42.3%	10.7%	22.2%	42.0%	63.6%	0.000
Elochoman	72.7%	9.6%	52.1%	73.5%	89.1%	0.238
L. Cowlitz	62.0%	9.5%	42.5%	62.3%	79.6%	0.023
Coweeman	88.1%	7.0%	71.6%	89.4%	97.9%	0.871
Toutle Tribs	16.6%	11.7%	1.3%	14.0%	44.9%	0.000
Green	93.4%	5.1%	80.6%	94.7%	99.5%	0.980
SF Toutle	46.0%	13.7%	20.5%	45.6%	73.2%	0.005
Kalama	27.7%	14.3%	5.4%	26.0%	59.0%	0.000
NF Lewis	83.2%	18.6%	33.8%	90.2%	100.0%	0.683
Cedar	75.2%	8.3%	56.9%	75.8%	89.5%	0.293
EF Lewis	62.5%	13.4%	34.7%	63.2%	86.0%	0.091
Washougal	56.3%	16.4%	24.0%	56.9%	86.2%	0.079
L. Gorge	18.7%	12.9%	1.5%	16.0%	48.9%	0.000

We also estimate the unmarked females density (Table 21). The mean density ranged from 0.05 to 25.49 in the Kalama and Coweeman, respectively. Using a significance level of 0.05, there is a low probability that the Grays, Elcohoman, Toutle Tribs, Kalama, and Lower Gorge have unmarked seeding levels that exceed the mode from Bradford's analysis. The mean unmarked females densities in the NF Lewis, Cedar, and Lower Cowlitz are influenced by an unknown number of unmarked hatchery fish.

Table 21. The estimated number of unmarked coho salmon females/mile based on GRTS surveys in 2010. The p-value is the probability the observed wild female density is greater than the mode of the full habitat seeding density based on Bradford et al. 2000.

GRTS Survey Basin	mean	sd	2.50%	50%	97.50%	*p*-value
Grays	0.91	2.15	0.00	0.00	6.73	0.003
Elochoman	3.40	5.56	0.00	1.38	18.99	0.042
L Cowlitz	7.00	12.74	0.00	2.00	41.94	0.146
Coweeman	25.49	26.54	0.00	17.72	96.09	0.556
Toutle Tribs	1.98	7.38	0.00	0.00	19.41	0.035
Green	9.33	10.65	0.00	6.02	37.94	0.200
SF Toutle	8.64	18.48	0.00	1.73	57.14	0.174
Kalama	0.05	0.21	0.00	0.00	0.46	0.000
NF Lewis	25.38	37.79	0.00	13.78	116.90	0.476
Cedar	9.63	14.52	0.00	4.44	48.99	0.212
EF Lewis	7.36	12.85	0.00	2.69	42.42	0.151
Washougal	6.17	11.91	0.00	1.74	37.40	0.121
L Gorge	2.65	8.86	0.00	0.00	23.05	0.047

Coho Salmon Escapement Estimates Using Multiple Methods by TRT Population

Population estimates were calculated by summing redd-based, mark-recapture, and trap and haul based estimates as appropriate. The mean estimates ranged from 488 to 21,745 for the Kalama and Upper Cowlitz/Cispus populations, respectively (Table 21). For the MAG, Upper Cowlitz/Cispus, and Tilton populations the CVs were less than the NOAA guideline of 15% (Crawford and Rumsey 2011). Precise estimates were obtained for these populations because of the trap and haul program and for the MAG Creek stock group based on the combination of mark-recapture and smolt trap expansion method. In contrast, the CV for all redd based estimates did not meet the standard and had CV ranging from 26% to 73% on the Coweeman and Kalama, respectively. The NF Lewis and Green estimates are redd based estimates but were supplemented with a mark-recapture estimate on Cedar Creek and trap and haul census on the NF Toutle River. These combined methods has CV’s that were generally less than the redd based estimate and were 23% and 36%, respectively.

Table 22. Adult coho salmon population estimates by TRT population. Salmon Creek and the Upper Gorge populations were not monitored in 2010 and there were no GRTS surveys for the mainstem NF Lewis, mainstem Toutle/mainstem NF Toutle, or mainstem Lower Cowlitz populations.

Population	mean	sd	2.50%	50%	97.50%	CV
Grays	2005	806	903	1859	3970	40%
Elochoman	3093	1091	1603	2895	5750	35%
MAG	1884	239	1493	1859	2429	13%
Cowlitz	7383	2594	3593	6966	13600	35%
Coweeman	3915	1014	2364	3768	6311	26%
Green	3642	829	2352	3526	5580	23%
SF Toutle	1920	991	761	1717	4301	52%
U Cispus/Cowlitz	21745	125	21500	21740	21990	1%
Tilton	3501	86	3334	3502	3671	2%
Kalama	488	358	111	400	1376	73%
NF Lewis	4104	1484	2497	3787	7563	36%
EF Lewis	2014	883	812	1840	4170	44%
Washougal	1426	834	475	1251	3382	58%
L Gorge	545	211	300	496	1073	39%

*The sum of abundance by marked status may not equal the abundance estimate due to rounding errors.

As expected the marked population estimates were highest in basins with hatcheries (Table 22; Appendix 2). The population with the greatest number of marked fish was in the Upper Cowlitz/Cispus population where hatchery fish are released as part of a program to re-establish natural production above Cowlitz Falls dam. This population estimate of 18,839 adults accounted for over 50% of the total marked population (Table 22). The largest producers of unmarked adults include the Lower Cowlitz (6,274), NF Lewis (3,860), Coweeman (3,528), and the MAG (1665) populations (Table 23). The CVs for redd based estimates exceeded the NOAA guideline but trap & haul and mark-recapture estimates meet the NOAA guideline.

Table 23. Marked adult coho salmon population estimates by NOAA Technical Recovery Team population. Note Salmon Creek and the Upper Gorge populations were not surveyed in 2010 and there were no surveys for the mainstem NF Lewis, mainstem Toutle/mainstem NF Toutle, or mainstem Lower Cowlitz populations.

Population	mean	sd	2.50%	50%	97.50%	CV
Grays	1617	662	714	1497	3267	41%
Elochoman	2259	830	1120	2108	4305	37%
MAG	229	51	143	224	341	22%
Cowlitz	1109	633	284	979	2704	57%
Coweeman	388	173	139	357	810	45%
Green	2188	540	1349	2115	3461	25%
SF Toutle	402	227	143	354	963	56%
U Cispus/Cowlitz	18839	125	18590	18840	19090	1%
Tilton	2523	86	2356	2524	2693	3%
Kalama	483	355	110	396	1366	73%
NF Lewis	244	222	84	179	802	91%
EF Lewis	651	373	177	568	1591	57%
Washougal	631	400	184	543	1605	63%
L Gorge	160	75	68	145	345	47%

*The sum of abundance by marked status may not equal the abundance estimate due to rounding errors.

Table 24. Unmarked adult coho salmon population estimates by TRT population. Note Salmon Creek and the Upper Gorge populations were not surveyed in 2010 and there were no surveys for the mainstem NF Lewis, mainstem Toutle/mainstem NF Toutle, or mainstem Lower Cowlitz populations. An unknown number of NF Lewis and L. Cowlitz adults are from RSI releases.

node	mean	sd	2.50%	50%	97.50%	CV
Grays	388	197	133	347	881	51%
Elochoman	834	371	333	764	1733	45%
MAG	1655	214	1305	1633	2142	13%
Cowlitz	6274	2268	2950	5913	11690	36%
Coweeman	3528	923	2113	3397	5687	26%
Green	1454	353	933	1399	2295	24%
SF Toutle	1518	785	594	1354	3407	52%
U Cispus/Cowlitz	2906					0%
Tilton	203					0%
Kalama	5	9	0	2	27	189%
NF Lewis	3860	1396	2339	3556	7159	36%
EF Lewis	1363	647	497	1238	2980	47%
Washougal	795	495	242	688	1929	62%
L Gorge	385	155	202	350	773	40%

*The sum of abundance by marked status may not equal the abundance estimate due to rounding errors.

Applying the estimate of the percentage of males that were jacks from Table 3 leads to an estimate of jacks by population (Table 25). Except for the Upper Cowlitz/ Cispus population, the mean estimate of jacks ranged from 26 to 398 fish. Except for trap and haul programs the jack estimates were imprecise due the low sampling rate of jacks and the few sampling locations (a total of 4 locations) for jacks.

Table 25. Jack coho salmon population estimates by TRT population. Note Salmon Creek and the Upper Gorge populations were not surveyed in 2010 and there were no surveys for the mainstem NF Lewis, mainstem Toutle/mainstem NF Toutle, or mainstem Lower Cowlitz populations.

node	mean	sd	2.50%	50%	97.50%	CV
Grays	117	74	38	102	273	63%
Elochoman	192	119	68	169	452	62%
MAG	102	46	50	97	176	45%
Cowlitz	398	243	123	355	904	61%
Coweeman	230	120	92	211	465	52%
Green	203	97	94	188	381	48%
SF Toutle	115	79	34	98	295	69%
U Cispus/Cowlitz	2305	13	2279	2306	2330	1%
Tilton	293	7	277	293	305	3%
Kalama	29	27	5	22	94	95%
NF Lewis	213	104	106	190	449	49%
EF Lewis	115	79	32	99	284	69%
Washougal	90	72	22	73	254	80%
L Gorge	26	17	9	23	64	66%

The LCR ESU estimates for WA populations are found in Table 26. We estimated a median total of 57,666 adults, 4,428 jacks, 31,724 marked adults, and 25,942 unmarked adults. The CV for marked adults met the NOAA standard and the CV for the total adults was 7% compared to the 15% NOAA standard. The CV for unmarked adults was 16%, which is only 1% more than the NOAA standard. It should be noted that no adult population estimates were made for the mainstem NF Lewis, mainstem Toutle/mainstem NF Toutle below the SRS, mainstem Lower Cowlitz, Salmon Creek, or Upper Gorge populations. Without these, the total ESU population estimate should be considered a minimum. We believe these missed areas do not represent substantial production except for the mainstem NF Lewis due to the observation of coho salmon spawning during Chinook surveys there.

The proportion of unmarked and marked adult coho salmon are found in Table 27. As described above basins with hatcheries tended to have higher proportions of marked fish while basins without hatcheries tend to have higher proportions of marked fish. The precision estimates generally exceeded the 95%CI half width of 5% except for populations with traps.

Table 26. Washington's LCR coho salmon population estimates for 2010. Note Salmon Creek and the Upper Gorge populations were not surveyed in 2010 and there were no surveys for the mainstem NF Lewis, mainstem Toutle/mainstem NF Toutle, or mainstem Lower Cowlitz populations.

node	mean	sd	2.50%	50%	97.50%	CV
Marked Adults	31724	2155	28250	31450	36630	7%
Unmarked Adults	25942	4141	19430	25400	35510	16%
Total Adults	57666	5840	48240	56930	70751	10%
Total Jacks	4428	780	3522	4323	5816	18%

*The sum of abundance by marked status may not equal the abundance estimate due to rounding errors.

Table 27. Estimates of the unmarked and marked adult coho salmon proportion by TRT population. An unknown number of NF Lewis and L. Cowlitz unmarked fish are included in the proportions are from RSI releases.

Proportion Unmarked					Proportion Marked		
Population	mean	sd	95%CI -1/2 width		mean	sd	95%CI -1/2 width
Grays	19.4%	5.6%	10.9%		80.6%	5.6%	10.9%
Elochoman	27.0%	7.0%	13.7%		73.0%	7.0%	13.7%
MAG	87.9%	2.2%	4.4%		12.1%	2.2%	4.4%
Cowlitz	85.0%	6.4%	12.6%		15.0%	6.4%	12.6%
Coweeman	90.1%	3.4%	6.7%		9.9%	3.4%	6.7%
Green	40.0%	4.2%	8.2%		60.0%	4.2%	8.2%
SF Toutle	79.1%	4.0%	7.8%		20.9%	4.0%	7.8%
U Cispus/Cowlitz	13.4%	0.1%	0.2%		86.6%	0.1%	0.2%
Tilton	27.9%	0.7%	1.3%		72.1%	0.7%	1.3%
Kalama	0.9%	1.3%	2.6%		99.1%	1.3%	2.6%
NF Lewis	94.1%	3.7%	7.3%		5.9%	3.7%	7.3%
EF Lewis	67.7%	11.0%	21.6%		32.3%	11.0%	21.6%
Washougal	55.8%	9.7%	18.9%		44.2%	9.7%	18.9%
L Gorge	70.6%	7.2%	14.1%		29.4%	7.2%	14.1%

CWT Program

The CWT recoveries in the fall of 2010 were uploaded to the RMIS system during 2011-12. The uploaded data include: 1) freshwater sport fishery recoveries on November 8, 2011, hatchery facility coho recoveries on January 12, 2012, and coho spawning ground recoveries on April 12, 2012. RMIS is a coastwide database that stores CWT tag and release data along with recovery and sampling data.

CWT recoveries from carcass recoveries found during stream surveys are presented in Table 28. These do not include hatchery recoveries, thus the recoveries and percent of out of basin recoveries only apply to coho salmon that spawned in stream. There were no CWT recoveries in the Lower Cowlitz and Lower Gorge populations and only one in the Coweeman population. These recoveries are consistent with the low proportion of marked fish sampled in these populations (Table 15). Most hatchery fish were recovered in the basin they were released in with the exception of Elochoman Hatchery where 11(68%) of the tags were recovered in the adjacent Grays population and Lewis Hatchery where 4(80%) of the tags were recovered in the adjacent EF Lewis population. The CWT release with the largest number of hatchery recoveries occurred from the Kalama River (18). An experimental rearing program occurs in Mill Creek, an EF Lewis River tributary, where natural origin coho fry are collected in the late spring/summer as they become stranded during a period of declining surface flow. The fish are transferred to a rearing pond and fed until release in the following spring. These fish accounted for the highest number of CWT recoveries. CWT data for fisheries and carcass recoveries are presented in annual reports for missing production groups (e.g. Roler and Olk 2012).

Table 28. Unexpanded CWT recoveries by population and hatchery for adult coho salmon in 2010. Note there were no jacks with CWT recovered. Gray boxes indicate CWT was recovered in the same basin as released. The (W) indicates wild smolts and (E) indicates experimental rearing of wild fry collected from dewatered reaches on Mill Creek, a tributary to the EF Lewis.

		Release Basin									
	Population	Eloch. H.	MAG (W)	Toutle H.	Kalama H.	Lewis H.	Lewis Mill(E)	Wash. H.	Clackamas Eagle H.	Columbia Tanner H.	**Total**
Recovery Basin	Grays	11	0	0	0	0	0	0	0	1	12
	Elochoman	5	0	1	0	0	0	0	0	0	6
	MAG	0	3	0	0	0	0	0	0	0	3
	Lower Cowlitz	0	0	0	0	0	0	0	0	0	0
	Coweeman	0	0	0	1	0	0	0	0	0	1
	Toutle/Green	0	0	11	0	0	0	0	0	0	11
	SF Toutle	0	0	2	1	0	0	0	0	0	3
	Kalama	0	0	0	18	0	0	0	0	0	18
	NF Lewis	0	0	0	0	1	0	0	0	0	1
	EF Lewis	0	0	1	0	4	20	0	0	0	25
	Washougal	0	0	0	0	0	0	2	1	0	3
	Lower Gorge	0	0	0	0	0	0	0	0	0	0
	Columbia	0	0	0	0	0	0	0	0	1	1
	Total	16	3	15	20	5	20	2	1	2	84
	% Out of Basin	69%	0%	27%	10%	80%	0%	0%	NA	50%	

Discussion

One of the most controversial aspects of the Bayesian approach is the specification of priors. We used vague priors for this analysis with the intent that the posterior distribution be dominated by the observed data. When this occurs the results obtained from Bayesian and maximum likelihood methods yield similar results (Kery 2010). We used vague priors because this is the first study on LCR coho and because it was unclear if other coho salmon information was applicable to the LCR (see below) given differences in climate, habitat, and experience of surveyors in conducting coho salmon surveys. The Bayesian framework provides an approach to account for this type of information in future years.

We used census counts and mark-recapture estimates to estimate coho population abundance where feasible. These methods are preferred because all the data needed to make an abundance estimate are collected annually. However, this left a large area for which alternate methods had to be used to estimate abundance. Other methods such as Area-Under-the-Curve (AUC) and redd surveys require assumptions about observer efficiency, survey life, and redd identification that may or may not be applicable to data collected from other basins or from other years (Irvine et al. 1992). We considered the use of AUC estimates, but survey life (e.g., the duration of time that live coho remained in the survey area) and observer efficiency (of live coho adults) for LCR coho salmon are unknown. Suring et al. (2006) assumed the estimates for the Oregon coast are applicable for the Lower Columbia. However, Jacobs et al. (2002) noted that for some years the mark-recapture estimates for the Smith River, an Oregon coastal stream, were higher than the AUC estimates for the same area. If the Smith River mark-recapture estimates are correct, one possible explanation for this discrepancy is that the standard Oregon coastal estimates of survey life and/or observer efficiency are biased high for those years on the Smith River. Therefore, application of the standard Oregon coastal observer efficiency and survey life estimates may lead to underestimates of LCR coho salmon abundance.

In addition, a review of the literature by Perrin and Irvine (1990) demonstrated high variability in survey life for coho salmon. Gallagher et al. (2010) estimates of survey life for coho were approximately two times or greater than those used by ODFW. However, Gallagher et al. 2010 estimates include residence time from entry, while ODFW estimates (~11 days) focus more on residence time in spawning tributaries (Willis 1954). Lestelle and Weller (2002) used an average residence time of 15 days. Estimates of observer efficiency for coho salmon averaged 75.5% for Oregon coastal streams (Solazzi 1984), 22% (range 20-24%) for a Northern California stream (Szerlong and Rundio 2007), 65% (range 22-100%) for an Alaskan stream (Hetrick and Nemeth 2003), and 86.5% for a coho stream on Vancouver Island (Holt 2002). Gallagher et al. 2010 indicated that AUC estimates were very sensitive to survey life and observer efficiency estimates; consequently concluding they were less reliable than redd counts. Lestelle and Weller (2002) believed that AUC estimates under estimated escapement at low density because it is difficult to observe fish when their abundance is low. However at higher escapement they believed redd counts are likely to underestimate abundance due to superimposition and the difficulty in identifying individual redds.

After our AUC review we were uncomfortable in applying this method without LCR-specific observer efficiency and residence time estimates over varying spawning escapements, so we opted to use redd-based estimates because we could obtain a specific annual LCR estimate of redds per female. While redd surveys are widely used (WDFW 2011) and can provide unbiased estimates, they have their own set of challenges (Muhlfeld et al. 2003, Dunham et al. 2001). The key assumptions for redd surveys are: 1) the spatial spawning distribution is known and either sampled completely or expanded for in an unbiased manner as part of the sampling design, 2) surveys cover the entire temporal spawning period, 3) all redds are consistently identified with the same protocols, and 4) the variability in redds per adult or female is measured annually for that population or if derived from other population or years is similar to the population where redd surveys are conducted.

The first two assumptions indicate that redd surveys must be spatially or temporally complete otherwise redd abundance will be under estimated. We believe that we had a good spatial survey design based on using GRTS. However our temporal coverage was more problematic and there were missed scheduled surveys, particularly due to high and turbid water conditions. Training was provided to all staff to help with consistent redd identification (Crisp and Carling 1989) and to differentiate coho, chum, Chinook, and steelhead redds, which were all visible during the coho spawning time. We used physical differences in substrate size and location within the basin to help classify redds from different species (e.g., Gallagher and Gallagher 2005). In addition, we used two locations to estimate redds per female and these locations were geographically distant from each other, and had different habitat, and survey conditions. A key assumption was that the number of redds constructed by females in these basins, and the observer efficiency in identifying these redds together were representative of redds per female in all other redd survey reaches in the ESU. Provided this assumption was met, our design did address the key assumptions needed for an unbiased redd survey.

Curbois et al. (2008) noted that the 95% CI based on normally distributed data and large sample theory was not adequate to estimate redd abundance. This resulted from the clumpiness of the redd data and many reach counts of zero, particularly when population sizes were low. To address this problem, we used the Negative Binomial distribution, which is commonly used for over dispersed count data (Hilborn and Mangel 1997). Based on Bayesian *p*-values, the negative binomial distribution adequately fit the data. However, the precision of our estimates was worse than we anticipated. This occurred because the data were over dispersed resulting in large variances, which were consistent with our observations. Another factor that affected precision was the reach sample sizes, which were fewer than expected due to a later than anticipated start of the contract, delays in developing the GRTS design, and limited resources. Finally, our escapement estimates include most sources of uncertainty. Our redd based estimates included spatial uncertainty as with the Oregon coast surveys, but also include uncertainty in redds per female, adult sex ratio, and jack to adult male ratio. Mark-recapture estimates include uncertainty in tag loss and smolt to adult survival. The trap and haul estimates included uncertainty associated with harvest of marked fish.

We explored a number of approaches to see if our estimates of adult coho abundance seemed reasonable. One approach we used was to compare our estimate of redds per female with other studies. For example, Gallagher et al. (2010) found that adult coho salmon redd-based abundance estimates were positively correlated with, and similar to, mark-recapture estimates in northern California streams. However, they noted that the coho salmon spawner to redd ratio varied annually and with the exception of 2006 the average adults per redd was 2.2; assuming a 1:1 sex ratio this would equate to 1.16 females per redd, which equates to an average of 0.86 redds per female. However, their annual point estimates for redds per female ranged from 0.55 to 1.67. In 2006 Gallagher et al. (2010) observed ~0.20 redds per female for each of three surveyed populations because of challenging observation conditions, which likely decreased redd detectability and life. Lestelle and Weller (2002) estimated coho salmon escapement in two Washington coastal streams between 1996 and 2000. They judged four mark-recapture experiments to be successful and in these years the redd based estimates were positively biased by ~15% in three of the four years. In one year the redd based estimate was negatively biased by ~7% compared to the mark-recapture estimate. Assuming equal sex ratio the redds per female from Lestelle and Weller (2002) was approximately 0.87, which is similar to the average estimate from Gallagher et al. (2010).

Our estimate was 0.47 redds per female (95%CI 0.35 to 0.61) and is lower than the averages from the California and Washington studies, but within the lower range observed by Gallagher et al. (2010) in average conditions and higher than the range observed under the poor observation conditions. Given the missed surveys and periodic poor observation conditions during the season our lower estimate of redds per female relative to Gallagher et al. (2010) is not unexpected. However, if our estimate of redds per females is biased low and the true estimate is closer to 0.86 redds per female from the other studies or 1.0 for Chinook salmon from Murdoch et al. (2009), the true population estimates would be approximately 41 to 53% less than those reported.

Chinook carcass recoveries may be biased by sex, age, and origin (Zhou 2000, Parken et al. 2003, and Murdoch et al. 2010). To minimize possible size bias in carcass recoveries for coho salmon, we estimated adult and jack abundance separately and used only trap data to estimate the proportion of males that were jacks. For coho salmon, carcass recoveries may be biased because females tend to guard the nest after spawning (Sandercock 1991). If we assume sex ratio for coho salmon should be approximately 50% females (Dittman et al. 1998), our hierarchical estimate for adults from carcasses surveys (46% females) may indicate a bias that males are recovered at a higher rate. If this is the case, our redd-based population estimates may be slightly biased high; however some populations of coho salmon are known to maintain female-biased sex ratios (Holtby and Healey 1990), in which case our estimates may remain unbiased. Most coho spawning carcasses recoveries occurred in small streams and were based on weekly surveys using a spatially balanced design so both the size of streams and the representative sampling design should minimize carcass recovery bias by origin.

We provided direct estimates of marked and unmarked coho salmon adults as surrogates for hatchery and natural origin spawners. If all hatchery origin juveniles were adipose fin clipped and/or CWT, then we could make the assumption that marked fish were hatchery origin fish and all unmarked fish were wild origin fish. However, when examining the actual hatchery marking data ~ 98% of the hatchery fish were mass marked and/or CWT. Therefore our estimates of unmarked and marked fish as surrogates for hatchery and natural origin spawners are slightly biased. In addition, we found no reliable method for correction for the RSI and unfed fry releases, which may decrease the number of natural origin spawners reported in the NF Lewis and Lower Cowlitz rivers.

The last NOAA status review suggested that all coho salmon populations in Washington's portion of the Lower Columbia ESU were at high risk for extinction because limited surveys suggested that the ESU was comprised of greater than 90% hatchery spawners. However, there was great uncertainty in the status due to the lack of comprehensive coho salmon surveys (NMFS 2011). In this report we estimate that ~26,000 unmarked adult coho salmon spawned in the WA portion of the LCR ESU in 2010. The actual estimate is likely higher since we did not include the mainstem Cowlitz, mainstem NF Lewis, and mainstem Toutle/lower NF Toutle rivers, along with the Salmon Creek, and the Upper Gorge populations in our estimates. It is likely a small percentage of the unmarked population is comprised of unmarked hatchery fish. The total proportion of hatchery origin spawners (pHOS) estimate, not corrected for missing mass marks, was 55%. If we subtract the 18,800 hatchery adults released to spawn in the upper Cowlitz to maintain that population until better juvenile passage exists, the total pHOS estimate decreases to 33% for the remainder of the Washington portion of the ESU. In contrast to the status review, we found only the Kalama population had greater than 90% hatchery spawners. Excluding the NF Lewis due to the release of unmarked hatchery fish, a total of five WA populations had proportions of natural origin spawners (pNOS) greater than 69%, with the Cowlitz and Coweeman populations having greater than near 85% pNOS. In addition, Zimmerman (2010) estimated the natural origin smolt outmigration was almost 400,000 smolts. This report suggests coho salmon are more abundant than NOAA status reviews indicate, and that some populations that are dominated by natural origin fish.

Zimmerman (2010) estimated that approximately 395,500 coho salmon smolts emigrated from Washington's LCR below BON in 2009. The smolt to adult return rates (SAR) were 2.9%, 3.1%, and 12.1% for the Cedar, Tilton, and Abernathy populations, respectively. The mean SAR for the 2009 smolt outmigration was estimated to be 6.2%. This would produce a post-harvest run of approximately 24,155 adults (95%CI 20,460-29,160), which is greater but not significantly different (p-value = 0.333) than our escapement estimate of 25,942 (95%CI 19,430 – 35,510). This suggests that our total adult population estimates at the ESU level are in line with our expectations based on estimated smolt production and an index of smolt to adult returns. For the 2010 return year, the ocean exploitation rate was approximately 6% (PFMFC 2011) with a harvest rate of 14.6% for LCR coho (WDFW and ODFW 2011).

Recommendations

This was the first year WDFW conducted coho salmon spawning ground surveys. This was an enormous undertaking and it was met with considerable success, however many improvements can be made to reduce possible bias, improve precision, and improve repeatability of our study and results. Our recommendations for improvement include:

1) Develop a standardized manual of protocols for conducting coho salmon spawning ground surveys. This should at a minimum cover species and redd identification, a detailed description of the methods used to conduct surveys and how to record information, methods for data storage and frequency of downloading information, and the methods used to establish and modify survey reaches based on GRTS points.
2) One key assumption is that the redd identification methods in Duncan and Abernathy creeks are the same as those used in all other basins. While difficult to test, supervisors and crew leaders should schedule periodic surveys following surveyors to ensure standard techniques described in the manual are being implemented during surveys. Standardized methods and proper training can minimize differences between surveyors (Willis 1964).
3) It is likely that early-timed hatchery coho salmon that spawned in the Grays, Elochoman, Kalama, and possibly the Lewis River are under-represented by redd counts since these fish may be spawning in the same areas and at the same times as Chinook salmon. The coho sampling design should be refined to address this issue.
4) Redd locations are recorded electronically. However, the remainder of the data is transcribed on field datasheets then entered into electronic databases after the surveys are completed. WDFW should pursue the use of technology to electronically record data in the field to save time and reduce error generation during data entry. However, data storage devices must be rugged and waterproof to minimize loss of data in these difficult survey conditions.
5) Currently data for this analysis is obtained from different ARC-GIS databases, trapping spreadsheets, the WDFW corporate spawning ground survey database, and a regional age and scales database. Efforts should be made to consolidate databases when possible and move toward unified corporate databases (WDFW 2011b).
6) The current coho survey design used GRTS location draws developed for other purposes. One result is that there were a limited number of data points available to develop the coho salmon spawning ground survey design. A denser GRTS draw for the LCR area would eliminate this problem and should be pursued.
7) We recommend that the Upper Gorge and Salmon Creek populations be monitored for redds, and other methods be explored to develop estimates for the mainstem NF Lewis and Lower Cowlitz rivers.
8) The precision of the redd-based estimates is low due to sampling a low fraction of the spawning area, over dispersed data, and the sampling design. To address these concerns we recommend increasing the number of samples per population and considering stratification of sampling effort corresponding to higher and lower density coho spawning areas. Stratification may lead to more precise estimates if a denser GRTS draw is available and homogeneous strata can be developed.

9) Since the precision for the mark recapture estimates was low, the resulting redds per female estimate had low precision. Besides the clumpiness in our redd densities, the redds per female estimate is the largest source of error in our abundance estimates. We recommend efforts to improve trap operations at these sites to mark more fish to improve estimates or consider alternate approaches for estimating escapement such as Labelle et al. (1994) or modification of Korman et al. (2007).
10) Over 1.7 million unmarked fry releases occurred in the Lewis River and over 200 thousand unmarked fry releases occurred in the Cowlitz. Since these groups are too small to be mass marked, we cannot use the mass mark to identify hatchery origin fish. Analysis should be undertaken to determine the extent to which these plants are contributing to adult returns and whether the receiving waters they are planted in are being fully seeded by natural spawning. If these programs are to be continued, we recommend funding of otolith marking and recovery to identify hatchery origin fish. Rawding et al. (2007) used this method in Cedar Creek to estimate the proportion of hatchery and wild fish in the coho smolt outmigration. These methods could be extended to adults. Alternatively parental based tagging using genetic markers could be used (Anderson and Garza 2006).
11) Rawding and Rodgers (2013) suggested that efficiencies may be obtained by the WDFW and ODFW working together on salmon and steelhead escapement estimates in the LCR ESU. Since both agencies are estimating coho salmon abundance, we suggest annual workshops/coordination meetings to review and learn about different study designs, protocols, database management, and statistical analysis to explore these efficiencies would be beneficial.
12) The original coho sampling frame for redd surveys was developed based on a few years of adult and juvenile survey data. There are 3 additional years of adult and juvenile data, since the frame was developed. The sample frame should be updated based on these additional data.

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We compiled a single coho salmon monitoring and CWT recovery report because all the information from study design to results are in a single reference and it was cost-effective to do this. The major funders for coho salmon monitoring and reporting are the Bonneville Power Administration (BPA), the National Marine Fisheries Service (NMFS) through the Mitchell Act and the Pacific Coast Salmon Recovery Fund (PCSRF). The later was administered through the Washington Recreation and Conservation Office (RCO). Significant WDFW resources contributed to the success of this project. Pacific Salmon Treaty funding was used to assist with study design development, improvements in WDFW's internal CWT database, and purchase of CWT wands used to recover CWT on spawning ground surveys. Tacoma Public Utilities provided the number of adult coho salmon released into the Tilton and Upper Cowlitz/Cispus Rivers. We acknowledge the support of Region 5 and Olympia database managers including Robert Woodard, Michelle Groesbeck, Danny Warren, Ben Warren, Are Storm, Gil Lensegrav, Catie Mains, and Leslie Sikora. Lance Campbell and John Sneva provided scale ages and the Olympia CWT lab staff extracted and read CWT. We thank Region 5 hatchery staff for operation and assistance in sampling at weirs, and the WDFW biologists and technicians for the implementation of the study design, data collection, and data entry. We thank Rick Golden (BPA), Rob Jones (NOAA), Scott Rumsey (NOAA), Brian Abbott (RCO), Tara Galuska (RCO), Kat Moore (RCO), and various committees of the PST for their support of this project. We thank Don Stevens from Oregon State University for creating the Lower Columbia web-based GRTS sampling tool proto-type and developing the spatial sample draws for the coho spawning ground surveys, and Martin Liermann (NOAA) for statistical advise. In addition, we thank Jen Bayer from the United States Geological Survey (USGS) and Pacific Northwest Aquatic Monitoring Partnership (PNAMP) and Russell Scranton (BPA) for their support of in the development of the Lower Columbia River GRTS sampling tool. Mention of trade names or commercial products does not constitute endorsement for use.

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Appendices

Appendix 1

Appendix 1 a. GRTS reaches surveyed for Grays River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
BLA	Grays	Blaney Creek	0.41	0.00	0.41
BLAB	Grays	Blaney Creek	0.59	0.41	1.00
BLB	Grays	Blaney Creek	0.41	1.00	1.41
CRJ	Grays	Crazy Johnson Creek	1.00	0.05	1.05
EGB	Grays	East Fork Grays River	1.00	1.86	2.86
EGT	Grays	East Fork Grays River Tributary	1.00	0.68	1.68
FOA	Grays	Fossil Creek	0.82	1.25	2.07
FOAB	Grays	Fossil Creek	0.18	2.07	2.25
FOB	Grays	Fossil Creek	0.82	2.25	3.07
GRC	Grays	Grays River	1.00	19.91	20.91
HUB	Grays	Hull Creek	1.00	2.97	3.97
IMA	Grays	Impie Creek	1.00	0.17	1.17
KLB	Grays	Klints Creek	1.00	1.69	2.69
MTA	Grays	Mitchell Creek	0.44	0.18	0.62
MTAB	Grays	Mitchell Creek	0.56	0.62	1.18
MTB	Grays	Mitchell Creek	0.44	1.18	1.62
NIA	Grays	Nikka Creek	1.00	0.00	1.00
STA	Grays	South Fork Grays River Tributary	0.06	0.00	0.06
STAB	Grays	South Fork Grays River Tributary	0.94	0.06	1.00
STB	Grays	South Fork Grays River Tributary	0.06	1.00	1.06
WGA	Grays	West Fork Grays River	0.86	2.40	3.26
WGAB	Grays	West Fork Grays River	0.14	3.26	3.40
WGB	Grays	West Fork Grays River	0.86	3.40	4.26
WGC	Grays	West Fork Grays River	1.00	4.39	5.39

Appendix 1 b. GRTS reaches surveyed for Elochoman River coho salmon in 2010

Section	Population	Stream Name	Miles	RM Start	RM Stop
BVA	Eloch/Skam	Beaver Creek	0.93	1.63	2.56
BVAB	Eloch/Skam	Beaver Creek	0.07	2.56	2.63
BVB	Eloch/Skam	Beaver Creek	0.93	2.63	3.56
CLA	Eloch/Skam	Clear Creek	1.00	0.06	1.06
ELB	Eloch/Skam	Elochoman	0.30	9.27	9.57
ELBC	Eloch/Skam	Elochoman	0.70	9.57	10.27
ELC	Eloch/Skam	Elochoman	0.30	10.27	10.57
ELF	Eloch/Skam	Elochoman	1.00	13.67	14.67
ELG	Eloch/Skam	Elochoman	1.00	15.96	16.96
ETA	Eloch/Skam	Elochoman Trib	1.00	0.00	1.00
NFA	Eloch/Skam	NF Elochoman	0.59	0.00	0.59
NFAB	Eloch/Skam	NF Elochoman	0.41	0.59	1.00
NFB	Eloch/Skam	NF Elochoman	0.59	1.00	1.59
WFA	Eloch/Skam	WF Elochoman	1.00	1.42	2.42
WFC	Eloch/Skam	WF Elochoman	1.00	2.80	3.80
FAA	Eloch/Skam	Falk Creek	1.00	1.60	2.60
KEL	Eloch/Skam	Kelly Creek Trib	0.43	0.00	0.43
NTA	Eloch/Skam	Nelson Creek Trib	0.65	0.15	0.80
RTB	Eloch/Skam	Risk Creek Trib B	0.66	0.00	0.66
SKT	Eloch/Skam	Skamakowa Trib	1.00	0.00	1.00
WID	Eloch/Skam	Wilson Creek	0.24	5.53	5.77
WIDE	Eloch/Skam	Wilson Creek	0.76	5.77	6.53
WIE	Eloch/Skam	Wilson Creek	0.24	6.53	6.77

Appendix 1 c. GRTS reaches surveyed for Lower Cowlitz River coho salmon in 2010

Section	Population	Stream Name	Miles	RM Start	RM Stop
5SA	Lower Cowlitz	Trib 5 Salmon Creek	1.00	1.00	2.00
ARB	Lower Cowlitz	Arkansas Creek	1.00	5.46	6.46
ARC	Lower Cowlitz	Arkansas Creek	1.00	7.57	8.57
BCA	Lower Cowlitz	Becker Creek	1.00	0.23	1.23
BLC	Lower Cowlitz	Blue Creek	0.93	4.44	5.37
BND	Lower Cowlitz	NF Brim Creek	0.32	1.38	1.70
BNE	Lower Cowlitz	NF Brim Creek	0.68	1.70	2.38
BNO	Lower Cowlitz	NF Brim Creek	0.32	2.38	2.70
BRM	Lower Cowlitz	Brim Creek Trib	1.00	0.35	1.35
CMA	Lower Cowlitz	Campbell Creek	1.00	1.40	2.40
DLB	Lower Cowlitz	Delameter Creek	1.00	2.45	3.45
GDA	Lower Cowlitz	Gardner Creek	0.28	0.79	1.07
GDO	Lower Cowlitz	Gardner Creek	0.58	1.07	1.65
GDB	Lower Cowlitz	Gardner Creek	0.42	1.65	2.07
KIT	Lower Cowlitz	King Creek Trib	1.00	2.15	3.15
OLC	Lower Cowlitz	Olequa Creek	1.00	6.20	7.20
OLD	Lower Cowlitz	Olequa Creek	0.97	7.51	8.48
OLO	Lower Cowlitz	Olequa Creek	0.03	8.48	8.51
OLE	Lower Cowlitz	Olequa Creek	0.97	8.51	9.48
OLF	Lower Cowlitz	Olequa Creek	1.00	10.59	11.59
OSB	Lower Cowlitz	Ostrander Creek	1.00	5.44	6.44
RAP	Lower Cowlitz	Rapid Creek	1.00	0.00	1.00
RCK	Lower Cowlitz	Rock Creek	1.00	0.00	1.00
SMO	Lower Cowlitz	Salmon Creek	0.78	16.41	17.19
SMR	Lower Cowlitz	Salmon Creek	1.00	20.59	21.59
SOB	Lower Cowlitz	SF Ostrander Creek	1.00	0.91	1.91
STA	Lower Cowlitz	SF Ostrander Creek Trib	0.87	0.00	0.87

Appendix 1 d. GRTS reaches surveyed for Coweeman River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
BDC	Coweeman	Baird Creek	1.00	0.75	1.75
CAF	Coweeman	Coweeman	0.41	26.30	26.71
CAFG	Coweeman	Coweeman	0.59	26.71	27.30
CAG	Coweeman	Coweeman	0.35	27.30	27.65
CAGJ	Coweeman	Coweeman	0.06	27.65	27.71
CAJ	Coweeman	Coweeman	0.94	27.71	28.65
CAL	Coweeman	Coweeman	0.43	28.69	29.12
CALM	Coweeman	Coweeman	0.57	29.12	29.69
CAM	Coweeman	Coweeman	0.43	29.69	30.12
CWC	Coweeman	Coweeman	0.79	7.54	8.33
CWCE	Coweeman	Coweeman	0.21	8.33	8.54
CWE	Coweeman	Coweeman	0.79	8.54	9.33
CWT	Coweeman	Coweeman	0.91	19.62	20.53
CWTU	Coweeman	Coweeman	0.09	20.53	20.62
CWU	Coweeman	Coweeman	0.91	20.62	21.53
CWY	Coweeman	Coweeman	1.00	22.46	23.46
GBA	Coweeman	Goble Creek	0.55	0.24	0.79
GBAC	Coweeman	Goble Creek	0.45	0.79	1.24
GBC	Coweeman	Goble Creek	0.55	1.24	1.79
GBF	Coweeman	Goble Creek	1.00	2.41	3.41
GBJ	Coweeman	Goble Creek	0.90	5.11	6.01
GBJL	Coweeman	Goble Creek	0.10	6.01	6.11
GBL	Coweeman	Goble Creek	0.90	6.11	7.01
GLB	Coweeman	Goble Creek	1.00	0.04	1.04
MUHJ	Coweeman	Mulholland Creek	1.00	3.04	4.04
NGD	Coweeman	NF Goble Creek	0.46	0.67	1.13
NGDF	Coweeman	NF Goble Creek	0.54	1.13	1.67
NGF	Coweeman	NF Goble Creek	0.46	1.67	2.13

Appendix 1 e. GRTS reaches surveyed for Green River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
CAA	Green	Cascade Creek	0.14	0.00	0.14
CAO	Green	Cascade Creek	0.67	0.14	0.81
CAB	Green	Cascade Creek	0.19	0.81	1.00
CAV	Green	Cascade Creek	0.14	1.00	1.14
CAC	Green	Cascade Creek	0.67	1.14	1.81
DT1	Green	Devils Trib 1	1.00	0.00	1.00
DVA	Green	WF Devils Creek	1.00	0.31	1.31
DTA	Green	Devils Creek	1.00	0.44	1.44
DVB	Green	Devils Creek	1.00	2.17	3.17
ELA	Green	Elk Creek	0.03	0.11	0.14
ELO	Green	Elk Creek	0.97	0.14	1.11
ELB	Green	Elk Creek	0.03	1.11	1.14
ELC	Green	Elk Creek	1.00	3.76	4.76
G2A	Green	Green 2550 Trib	0.37	0.00	0.37
G2O	Green	Green 2550 Trib	0.63	0.37	1.00
G2B	Green	Green 2550 Trib	0.37	1.00	1.37
GNA	Green	Green River	1.00	3.84	4.84
GNB	Green	Green River	0.53	4.84	5.37
GNO	Green	Green River	0.47	5.37	5.84
GNC	Green	Green River	0.31	5.84	6.15
GNV	Green	Green River	0.22	6.15	6.37
GND	Green	Green River	0.66	6.37	7.03
GNL	Green	Green River	0.12	7.03	7.15
GNE	Green	Green River	0.88	7.15	8.03
GNG	Green	Green River	1.00	13.24	14.24
GNK	Green	Green River	0.29	19.32	19.61
GNP	Green	Green River	0.46	19.61	20.07
GNT	Green	Green River	0.25	20.07	20.32
GNH	Green	Green River	0.29	20.32	20.61
GNI	Green	Green River	0.09	20.61	20.70
GNR	Green	Green River	0.37	20.70	21.07
GNJ	Green	Green River	0.63	21.07	21.70

Appendix 1 f. GRTS reaches surveyed for mainstem and NF Toutle River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
HEE	NF Toutle	Hemlock Creek	1.00	5.13	6.13
N1C	NF Toutle	NF Toutle Trib 1	0.19	1.00	1.19
N1D	NF Toutle	NF Toutle Trib 1	0.80	1.19	1.99
S2A	NF Toutle	Silver Trib 2	0.77	0.66	1.43
T3A	NF Toutle	Toutle Trib 3	1.00	0.31	1.31
WTB	NF Toutle	Wyant Trib 1	1.00	0.78	1.78
WYA	NF Toutle	Wyant Creek	1.00	0.29	1.29
WYI	NF Toutle	Wyant Creek	0.68	5.26	5.94
WYK	NF Toutle	Wyant Creek	0.68	6.26	6.94

Appendix 1 g. GRTS reaches surveyed for SF Toutle River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
FLY	SF Toutle	Flye Creek	1.00	0.00	1.00
HAR	SF Toutle	Herrington Creek	1.00	0.17	1.17
JOH	SF Toutle	Johnson Creek	1.00	0.75	1.75
LOC	SF Toutle	Loch Creek	1.00	0.00	1.00
SFA	SF Toutle	SF Toutle River	0.49	0.00	0.49
SFO	SF Toutle	SF Toutle River	0.51	0.49	1.00
SFB	SF Toutle	SF Toutle River	0.49	1.00	1.49
STU	SF Toutle	SF Toutle River	1.00	3.38	4.38
SFC	SF Toutle	SF Toutle River	1.00	6.71	7.71
SFD	SF Toutle	SF Toutle River	0.90	8.77	9.67
SFV	SF Toutle	SF Toutle River	0.10	9.67	9.77
SFE	SF Toutle	SF Toutle River	0.90	9.77	10.67
SFI	SF Toutle	Studebaker Creek	1.00	19.03	20.03

Appendix 1 h. GRTS reaches surveyed for SF Toutle River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
FLY	SF Toutle	Flye Creek	1.00	0.00	1.00
HAR	SF Toutle	Herrington Creek	1.00	0.17	1.17
JOH	SF Toutle	Johnson Creek	1.00	0.75	1.75
LOC	SF Toutle	Loch Creek	1.00	0.00	1.00
SFA	SF Toutle	SF Toutle River	0.49	0.00	0.49
SFO	SF Toutle	SF Toutle River	0.51	0.49	1.00
SFB	SF Toutle	SF Toutle River	0.49	1.00	1.49
STU	SF Toutle	SF Toutle River	1.00	3.38	4.38
SFC	SF Toutle	SF Toutle River	1.00	6.71	7.71
SFD	SF Toutle	SF Toutle River	0.90	8.77	9.67
SFV	SF Toutle	SF Toutle River	0.10	9.67	9.77
SFE	SF Toutle	SF Toutle River	0.90	9.77	10.67
SFI	SF Toutle	Studebaker Creek	1.00	19.03	20.03

Appendix 1 i. GRTS reaches surveyed for Kalama River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
BRK	Kalama	Burke Creek	0.21	0.40	0.61
FSA	Kalama	Fish Pond Creek	0.88	0.12	1.00
KAD	Kalama	Kalama River	0.47	1.20	1.67
KADA	Kalama	Kalama River	0.53	1.67	2.20
KAA	Kalama	Kalama River	0.47	2.20	2.67
KAE	Kalama	Kalama River	1.00	8.74	9.74
NFA	Kalama	NF Schoolhouse	1.00	0.00	1.00
SCA	Kalama	Schoolhouse	1.00	0.00	1.00
SPA	Kalama	Spencer Creek	1.00	0.00	1.00

Appendix 1 j. GRTS reaches surveyed for NF Lewis River coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
HOD	Lower Lewis	Houghton Creek	1.00	1.51	2.51
RSC	Lower Lewis	Ross Creek	1.00	0.60	1.60

Appendix 1 k. GRTS reaches surveyed for Cedar Creek coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
BEA	Cedar Creek	Beaver Creek	1.00	0.22	1.22
BIA	Cedar Creek	Bitter Creek	0.69	0.20	0.89
BIAB	Cedar Creek	Bitter Creek	0.31	0.89	1.20
BIB	Cedar Creek	Bitter Creek	0.69	1.20	1.89
CEF	Cedar Creek	Cedar Creek	1.00	6.02	7.02
CEG	Cedar Creek	Cedar Creek	0.32	7.02	7.34
CEH	Cedar Creek	Cedar Creek	1.00	7.34	8.34
CEI	Cedar Creek	Cedar Creek	0.39	8.59	8.98
CEIJ	Cedar Creek	Cedar Creek	0.61	8.98	9.59
CEJ	Cedar Creek	Cedar Creek	0.39	9.59	9.98
CEK	Cedar Creek	Cedar Creek	1.00	10.07	11.07
CEL	Cedar Creek	Cedar Creek	0.40	11.30	11.70
CELM	Cedar Creek	Cedar Creek	0.60	11.70	12.30
CEM	Cedar Creek	Cedar Creek	0.39	12.30	12.69
CEMN	Cedar Creek	Cedar Creek	0.01	12.69	12.70
CEN	Cedar Creek	Cedar Creek	0.99	12.70	13.69
CEO	Cedar Creek	Cedar Creek	0.97	13.94	14.91
CEOP	Cedar Creek	Cedar Creek	0.03	14.91	14.94
CEP	Cedar Creek	Cedar Creek	0.52	14.94	15.46
CEPQ	Cedar Creek	Cedar Creek	0.45	15.46	15.91
CEQ	Cedar Creek	Cedar Creek	0.55	15.91	16.46
CER	Cedar Creek	Cedar Creek	0.32	18.11	18.43
CERS	Cedar Creek	Cedar Creek	0.68	18.43	19.11
CHA	Cedar Creek	Chelatchie Creek	1.00	0.00	1.00
CHE	Cedar Creek	Chelatchie Creek	0.12	4.42	4.54
CHEF	Cedar Creek	Chelatchie Creek	0.88	4.54	5.42
CHF	Cedar Creek	Chelatchie Creek	0.12	5.42	5.54
JOAB	Cedar Creek	John Creek	0.85	0.00	0.85
NCA	Cedar Creek	NF Chelatchie Creek	0.47	0.00	0.47
NCB	Cedar Creek	NF Chelatchie Creek	1.00	0.47	1.47
PTR	Cedar Creek	Pup Creek Trib	1.00	0.00	1.00
PUA	Cedar Creek	Pup Creek	1.00	0.48	1.48
PZAB	Cedar Creek	Protzman Creek	1.00	0.00	1.00

Appendix 1 l. GRTS reaches surveyed for EF Lewis coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
BKL	EF Lewis	Brickle Creek	0.32	0.00	0.32
DEB	EF Lewis	Dean Creek	1.00	0.23	1.23
DEE	EF Lewis	Dean Creek	1.00	2.07	3.07
JEN	EF Lewis	Jenny Creek	0.23	0.00	0.23
LTR	EF Lewis	Lockwood Trib	0.65	0.00	0.65
LWA	EF Lewis	Lockwood Creek	1.00	0.00	1.00
LWE	EF Lewis	Lockwood Creek	1.00	3.55	4.55
MIC	EF Lewis	Mill Creek	1.00	1.20	2.20
MNE	EF Lewis	Mason Creek	1.00	2.68	3.68
MNK	EF Lewis	Mason Creek	1.00	6.78	7.78
RIB	EF Lewis	Riley Creek	1.00	0.00	1.00

Appendix 1 m. GRTS reaches surveyed for Washougal coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
BOA	Washougal	Boulder Creek	1	1	2
DTA	Washougal	Dougan Trib	1	0	1
JNB	Washougal	Jones Creek	1	0.41	1.41
LWA	Washougal	Little Washougal	1	0.93	1.93
WAF	Washougal	Washougal River	1	9.43	10.43
WAV	Washougal	Washougal River	1	20.21	21.21
WFI	Washougal	WF Washougal	1	4.41	5.41

Appendix 1 n. GRTS reaches surveyed for Lower Gorge coho salmon in 2010.

Section	Population	Stream Name	Miles	RM Start	RM Stop
CMP	L. Gorge	Campen Creek	1.00	0.35	1.35
EWB	L. Gorge	EF Woodward Creek	1.00	0.02	1.02
GFA	L. Gorge	Greenleaf Creek	1.00	0.00	1.00
HMC	L. Gorge	Hamilton Creek	0.28	1.72	2.00
HMCE	L. Gorge	Hamilton Creek	0.72	2.00	2.72
HME	L. Gorge	Hamilton Creek	0.10	2.72	2.82
HMEF	L. Gorge	Hamilton Creek	0.18	2.82	3.00
HMF	L. Gorge	Hamilton Creek	0.52	3.00	3.52
HMFH	L. Gorge	Hamilton Creek	0.30	3.52	3.82
LNA	L. Gorge	Lawton Creek	1.00	0.00	1.00

Appendix 2

Appendix 2 a. Hatchery coho salmon smolt releases occurred in 2009 from BY 2007. The two large releases of unmarked fish included a release of early and late stock coho salmon from the Lewis River hatchery. These unmarked fish were all CWT and are part of the double index tag (DIT) group used to evaluate selective fisheries.

Brood Yr	Month	River	Tributray	Stage	Release MassMark	Release UnMark	Release Totals
2007	Spring	COWLITZ R 26.0002	All	Smolt	2810144	12015	2822159
2007	Spring	DEEP R 25.0071	All	Smolt	702018	4132	706150
2007	Winter	ELOCHOMAN R 25.0236	All	Smolt	81655	0	81655
2007	Spring	ELOCHOMAN R 25.0236	All	Smolt	84039	961	85000
2007	Spring	FALLERT CR 27.0017	All	Smolt	325532	2452	327984
2007	Spring	GRAYS R -WF 25.0131	All	Smolt	157336	664	158000
2007	Spring	GREEN R 26.0323	All	Smolt	483670	5971	489641
2007	Winter	KALAMA R 27.0002	All	Smolt	116683	943	117626
2007	Spring	KALAMA R 27.0002	All	Smolt	235808	1904	237712
2007	Spring	KLICKITAT R 30.0002	All	Smolt	60296	360	60656
2007	Spring	LEWIS R -NF 27.0168	All	Smolt	1567737	155931	1723668
2007	Spring	WASHOUGAL R 28.0159	All	Smolt	234256	2744	237000
2008	Spring	COWLITZ R 26.0002	All	Smolt	2899368	9036	2908404
2008	Spring	DEEP R 25.0071	All	Smolt	747000	0	747000
2008	Spring	FALLERT CR 27.0017	All	Smolt	115670	479	116149
2008	Spring	GRAYS R -WF 25.0131	All	Smolt	153000	0	153000
2008	Spring	GREEN R 26.0323	All	Smolt	144433	2799	147232
2008	Spring	KALAMA R 27.0002	All	Smolt	550064	545	550609
2008	Spring	KLICKITAT R 30.0002	All	Smolt	1833110	11065	1844175
2008	Spring	LEWIS R -NF 27.0168	All	Smolt	1542130	159912	1702042
2008	Spring	WASHOUGAL R 28.0159	All	Smolt	158003	1274	159277
2008	Spring	TOTAL	All	Smolt	15001952	373187	15375139

Detection Probabilities for Passive Integrated Transponder (PIT) Tags in Adult Salmon and Steelhead with Hand Held Scanners

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Abstract

Passive Integrated Transponder (PIT) tags are used throughout the Columbia River basin for measuring survival, migration patterns, and predation rates of salmon and steelhead. The use of PIT tags for estimating salmon and steelhead harvest rates in fisheries is a potential new application for PIT tags. However, the efficiency with which these tags may be detected in landed catch must be known for these estimates to be unbiased. We implemented a study to evaluate PIT detection rates for tagged adult salmonids using a variety of tag scanner types under conditions similar to those expected in sampling of fisheries catch. A total of 198 fall Chinook salmon, 197 coho salmon, and 137 summer steelhead hatchery adults were tagged with PIT tags and released into hatchery raceways in the fall of 2010. The salmon were held for 7 days after tagging and the steelhead were held for 32 days after which they were sacrificed for sampling. A total of 518 individual salmon and steelhead with PIT tags were sampled multiple times in 54 trials with Destron Fearing (DF) FS2001F-ISO Reader Base Unit (DF) with racquet antenna, All Flex (AF) Model RS601-3, Destron Fearing (DF) FS2001F-ISO Reader Base Unit with Flat Plat (FP) antenna, and Destron Fearing (DF) FS2001F-ISO Reader Base Unit with 24 inch square (SQ) antenna. After removing nine questionable trials due to equipment malfunction or samplers not implementing protocols, a total of 7,398 PIT tags (99.6%) were detected out of 7,423 PIT tags samples. The mean detection rate based on the 45 trails was 99.4% (95% confidence interval (CI) = 98.5% - 100.0%). The short-term PIT tag retention estimates were approximately 98% for all three species and the short-term survival of PIT tagged fish ranged from 95% for Chinook to 98% for steelhead. We found no statistical differences in detection efficiency between any of the units ($P = 0.468$), tag retention ($P = 0.880$) between species, and survival ($P = 0.186$) between species. The data were analyzed using a general linear mixed model (GLMM) and the best model based on Bayesian Information Criteria (BIC) included a random effect for the intercept and a fixed effect for species resulting in a detection probabilities of 99.20% (95% CI 98.56% - 99.56%), 99.78% (95% CI 99.32% - 99.93%), and 99.83% (95% CI 99.37% - 99.95%) for Chinook salmon, coho salmon, and summer steelhead, respectively. When Akaike Information Criteria (AIC) was used for model selection, a fixed effect was also added for detector type, indicating the DF had a higher detection rate than the AF. Regardless of the model selection criteria used, our results suggested that PIT tag detection rates with hand held PIT tag scanners were consistent with detection means above 98.8% when standardized protocols were implemented. Application of our detection rates to adult PIT tag fishery sampling programs allows for a bias correction due to PIT tags that are not detected, although this bias is negligible.

Introduction

Passive Integrated Transponder (PIT) tagging of salmonids in the Columbia River Basin has served a variety of purposes including estimation of juvenile and adult survival and the investigation of mechanisms affecting survival (Connor et al. 1998, Zabel and Accord 2004, Buchanan et al. 2006) such as predation by bird and northern pikeminnow (Collis et al. 2001, Petersen and Barfoot 2003), and habitat characteristics (Paulsen and Fisher 2005). The number of applied PIT tags has increased from less than 20,000 tags applied in 1988 to over 2,000,000 in 2009. In addition, the number of returning PIT tagged adults, as measured by detections at Bonneville Dam, has exceeded 30,000 individuals. Detection systems have advanced from hand held devices (Busby and Deegan 1999) to juvenile bypass systems to adult traps and ladders (Harmon 2003) to instream arrays capable of detecting adult and juvenile passage (Connolly et al. 2008). Almost $4,000,000 annually is dedicated to the purchase of PIT tags, and millions more are spent capturing and tagging fish, recovering tags, and storing data in PTAGIS, a coastwide PIT tag database. Harvested PIT tagged fish represent one of the largest sources of unaccounted mortality for PIT tagged fish in the Columbia basin and could provide valuable information to researchers, managers, and policy makers. This is especially true of harvest rates for groups of fish that are generally not tagged with coded-wire tags (CWT) but are often PIT tagged. These groups often include natural origin salmon and steelhead. Thus, PIT tag recoveries in fisheries may also allow managers to better estimate harvest impacts on at risk natural origin populations and to shape fisheries to reduce impacts on ESA listed fish based on their spatial and temporal occurrence.

However, before harvest estimates can be calculated based on PIT tags, the probability of detecting a PIT tag during catch sampling of harvested adult salmon or steelhead must be estimated. While it may be convenient to assume that all PIT tagged fish sampled will be detected, violation of this assumption would result in systematic underestimates of harvest rates on PIT tagged fish. Therefore, the Washington Department of Fish and Wildlife (WDFW) initiated a study to estimate the probability of detecting PIT tags in harvested fish using a variety of readily available portable PIT tag detectors and antennae. In addition to evaluating the effect of scanner type, we explored the effects of fish length, girth, sex, species, and sampling personnel on detection rates. The purpose of this study was to estimate PIT tag detection rates, which may be applied to expand the number of PIT tags from sampled fisheries. Without expansion, harvest estimates derived from PIT tag sampling may be biased and not defensible.

Methods

Tagging
Adult salmon and steelhead were tagged at WDFW Hatcheries to test short-term PIT tag retention, mortality, and detection. Standard Columbia Basin PIT tagging procedures from CBFWA (1999) developed for juveniles were adapted for adults. Briefly, fish were crowded in the hatchery raceway with seines or screens, netted, and released into a holding tank. The tank was filled with water and had a concentration of ~ 40mg of MS-222 per liter. After fish had become docile, they were tagged with a Destron Fearing TX1411SST PIT tag (12mm 134.2 kHz) primarily in the peritoneal cavity, scanned with a Destron Fearing (DF) FS2001F-ISO Reader Base Unit with a racket antenna to record the tag number, and released back into the raceway. At the time of tagging we recorded species and sex, which was identified using morphological characteristics (Groot and Margolis 1991). A measuring board was used to measure fork length and a plastic tape measure was used to determine girth by measuring body circumference anterior to the dorsal fin.

Detection Trials
At the end of the study, fish were sacrificed to mimic sport and commercial sampling of dead fish. The salmon were held for 7 days after tagging and the steelhead were held for 32 days. Fish were sacrificed prior to full maturation of the gonads because high rates of tag loss may occur immediately prior to spawning (Prentice et al. 1994). Carcasses were laid out in single file as one might sample a group of fish at a boat ramp or in a commercial fishery. The salmon were placed parallel to each other in the same orientation, and the distance between the noses of adjacent fish was approximately 0.75 meters to avoid potential interference between adjacent PIT tags.

An initial carcass scan was conducted with a Destron Fearing (DF) FS2001F-ISO Reader Base Unit with a racket antenna to determine tag presence. This process was repeated with a hand held All Flex (AF) Model RS601-3. The process was repeated with different DF and AF readers. At the end of the day, all fish were dragged over a 12 x 26 inch flat plate (FP) and passed through a 24 inch square (SQ) antennae. Both were attached to a FS2001F-ISO Reader Base Unit with a 20-inch cable. Data was not collected on individual samplers as they passed fish through the SQ and FP detectors (Figure 1). Salmon were scanned for PIT tags using an oval shaped pattern initiated near the chin, moving back toward the tail near the ventral surface, and then moving forward near the dorsal surface toward the head (Figure 2).

A single trial involved scanning every individual of a particular species with a particular detector unit. For DF readers, all Chinook salmon, coho salmon, and steelhead were scanned with each of the four readers, but within each trial, scanning was split among samplers so that each sampler scanned approximately 25% of the fish. For AF readers, sampling of Chinook salmon followed the same protocol as was used in DF trials, whereas for coho salmon and steelhead, 100% of the fish were scanned with each of the four AF readers by each sampler, resulting in four times the number of AF trials conducted for coho salmon and steelhead. Chinook salmon were also each scanned once with a Psion (PS) data logger with RFID (Agrident Air 200) and a BioMark hand held Pocket Reader (PR) detector. At the end of the trials, all fish were scanned once with the FP and SQ readers.

Figure 1. Picture of PIT tag detection antennae used in this study include a) Destron Fearing (DF) FS2001F-ISO Reader Base Unit (DF) with racquet, b) All Flex (AF) Model RS601-3, c) Destron Fearing (DF) FS2001F-ISO Reader Base Unit (DF) with Flat Plat (FP) antennae, d) Destron Fearing (DF) FS2001F-ISO Reader Base

Unit (DF) with 24 inch square (SQ) antennae, e) Psion Teklogic data logger with RFID (PS), and a f) BioMark hand held Pocket Reader (PR).

During efficiency trials most Chinook and coho salmon retrieved from the holding raceway contained PIT tags. This resulted in few untagged coho and Chinook salmon in the scan sample, potentially conditioning samplers to expect tagged fish. This could have resulted in a tendency for samplers to use scanners in a manner ensuring high detection efficiency since most fish were known to contain tags, whereas similar scan patterns would be less likely in the field where tag rates would be substantially lower. To counteract this tendency, samplers used the protocols described in the appendix and were limited to one reader pass to sample each fish, preventing them from repeatedly scanning fish that they believed had tags that were undetected during a first pass. As an additional control for the potential of such biased sampling, trials with summer steelhead contained approximately 60% untagged fish; therefore, samplers could not target tagged fish.

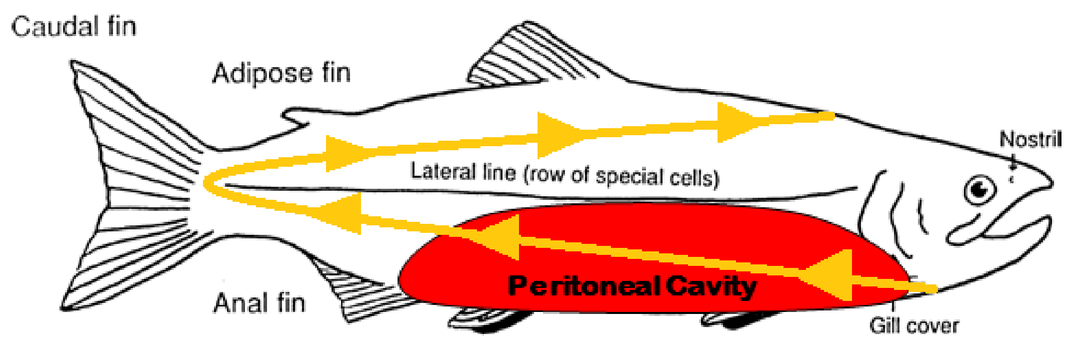

Figure 2. Pass over detection method for scanning adult salmon for PIT Tags.

Statistical Analysis

The study was designed to collect PIT tag presence/absence (binary) response data along with explanatory variables relating to the fish, samplers, and equipment. We analyzed the data using a generalized liner model (GLM) to accommodate the non-Gaussian distribution (O' Hara and Kotze 2010). The appropriate analysis for this data type is a binomial regression (Faraway 2006), which is a type of GLM that is composed of three components: 1) the binomial distribution to describe the random variation in the response, 2) the logit function is applied to the expectation of the response, and 3) a linear predictor is composed of a combination of covariates that describe the system. Both fixed and random effects were considered in the analysis. Fixed effects are traditionally used when there is interest in the studied factors or most conceivable levels of the factors are studied (Kery 2010). A fixed effect is a parameter to be estimated from the data. A random effect is often used when we do not have an interest in the factor levels in the study or when these levels form a random sample of possible levels that could have been included within the study. Random effects are used to make generalizations about this larger population and there is a greater interest in the variation among all factor levels. However, in some cases there is a desire to estimate the random effect and use it in regression assessment.

When both fixed and random effects are included in a model it is referred to as a mixed or mixed-effects model. The data were analyzed as grouped in Table 4 based on the following equation:

$$\text{Logit}\ (Detection/Samples_{ijk}) = \mu + Trial_i + Species_j + Type_j + Serial_{ijk} \quad (1)$$

Where µ is the intercept, *Trial* is the complete sampling of a species with each scanner type and serial number, *Type* is the scanner unit, *Species* is Chinook, coho, or steelhead, and *Serial* is the serial number from 1 through 7 for each of the scanner types. The fixed effects included species and scanner type. PIT tag scanners were classified into two types the DF used for portable commercial fisheries sampling, and AF used for sport fisheries sampling. A block is a property of the experimental unit, which is the trial in our study design. Since we were not interested in the block effects but had to account for their effect, the blocks were treated as random effects for the intercept. Since we used multiple readers with different serial numbers for each reader type in the trials, we considered the serial as a nested within type and also treated this as a random effect. GLM analysis was conducted in R (R Development Core Team 2010) using lme4 and glmer. Following the guidance from Bolker et al. (2009) we tested the random effects using the likelihood ratio test (LRT), and then we proceeded to test fixed effects using multi-model inference based on information criteria (Burnham and Anderson 2002).

We analyzed detection efficiencies of the other types of tag readers (FP, SQ, PR, and PS) separately due to small sample sizes and lack of individual sampler information. In addition we used the binomial distribution to report on the estimated tag loss, and mortality to ensure that we did not miss any PIT tagged fish because this would bias our estimate of the probability of detection. We used the methods described in Connolly et al. (2008) to estimate the probability that we missed a tag assuming independent sampling events (in our case multiple scans of individual fish with different readers).

$$x_i \sim \text{Binomial}(n_i, p.undet_i)$$

$$CumDet = 1 - \prod_{i=1}^{trials} (p.undet_i) \qquad (2)$$

where CumDet= the probability of detecting a PIT tag if it was present after multiple independent trials, *p.undet* = probability the fish was not detected in a trial, *x* = fish with a missed PIT tags in a trial, *n* = PIT tagged fish sampled in a trial, and *i* = trial number. WinBUGS (Lunn et al. 2000) was used to estimate probabilities of detection, tag loss and retention, and mortality and survival of PIT tagged fish using vague independent priors. Due to problems with the hypothesis testing for detection by trial, mortality, and tag loss by species were compared using contingency tables and χ^2 tests.

Results

PR and PS Readers

The PR and PS were only used on Chinook salmon for one trial each. The PS had a high detection probability of greater than 98% while the PR had a very low detection probability (6%) (Table 1). The PR was borrowed for the trials and was an older model. The low PR detection probabilities are likely the result of older or damaged equipment. The PS and PR were only used in a single trial due to the low detection efficiency of the PR and because we had to return the PS to the manufacturer. Due to the limited use these reader types were not used in further data analysis.

Table 1. Detections, samples, and detection probability estimates for 2 PIT tag trials with the Pocket Reader (PR) and Psion (PS) data logger with RDIF.

Type	Species	Detections	Samples	Mean	SD	2.50%	Median	97.50%
PS	Chinook	193	196	0.9822	0.0097	0.9596	0.9837	0.9957
PR	Chinook	12	196	0.0634	0.0175	0.0339	0.0620	0.1025

Uncensored Results and Data Censoring

The total number of fish tagged, detected, mortalities and tag loss is summarized by species and location (Table 2). A total of 518 individual salmon and steelhead with PIT tags were sampled multiple times in 54 trials for a total of 9,044 detection samples. A total of 8,408 tags were detected, which equates to a 93% detection rate.

Table 2. Summary data from tagging and detections at the hatchery PIT tag study.

Species	Fall Chinook	Coho	Summer Steelhead
Location	Skamania Hatchery	Kalama Falls Hatchery	Skamania Hatchery
Tag Date	9/24/10	10/19/10	10/15/10
Tagged	198	197	132
Not Tagged	2	3	169
Mortality	~9	~4	2 w/tag, 1 wo/tag
Sample Date	10/1/10	10/26/10	11/16/10
Detected	196	195	127
Sampled	200	200	299 w/out morts
Tag Loss	3	2	2

However, during the study we had some problems with equipment malfunction and implementation of protocols, which necessitated censoring data collected during these trials (Table 3). Trials 10-15 were censored due to poor data resulting from reader battery loss and the use of an older tag reader, AF-2, which consistently detected fewer PIT tags and stored detections at a slower rate than other AF units. On many of the Chinook and coho trials when ~100% of the fish were tagged it could not store data fast enough resulting in non-detections. The unit performed much better for steelhead when only 40% of the fish were tagged. In trial 48 the weather was cold and the sampler with gloved hands had difficulty pressing the button to start scanning on DF-4. After this was brought to their attention, the detection efficiencies improved during the remainder of the trial but this data required censoring.

The weather was rainy and wet during trial 50. The FP detection test occurred at the end of the day and samplers were in a hurry to get done. They did not follow protocol and did not keep the whole fish in contact with the FP, resulting in a lower detection efficiency than other trials (49 & 51). The first time the SQ was used in trial 52 we tossed fish through the square into a tote to mimic commercial sampling. However, the detection rate was very low. Therefore, fish were then passed through the SQ, which improved the detection rate to 87%. In addition, we had problems tuning the unit during our first sampling occasion. The efficiency improved to almost 100% after we constructed a slide in subsequent trials (53 & 54). Consequently we censored trial 52.

Results after Data Censoring

After removing nine questionable trials (10-15, 48, 50, & 52), a total of 7,398 PIT tags (99.6%) were detected out of 7,423 PIT tags samples. The sample mean for these trials was 99.4%. These detection rates were not significantly different between detector types ($\chi2 = 43.66$, df = 44, $P = 0.486$). Given the few missed detections, these data may not follow $\chi2$ distribution. However, graphical examination of the data demonstrates overlapping 95% CI, which implies similar detection rates between tag reader types (Figure 3).

Table 3. Detections, samples, and detection probability estimates for 54 PIT tag trials.

Trial	Type/#	Species	Detections	Samples	mean	sd	2.50%	median	97.50%
1	AF/1	Chinook	195	196	0.9924	0.62%	97.62%	0.994	0.9995
2	AF/1	Coho	195	195	0.9974	0.0036	0.9868	0.9988	1.0000
3	AF/1	Coho	194	195	0.9924	0.0062	0.9762	0.9940	0.9995
4	AF/1	Coho	194	195	0.9924	0.0062	0.9764	0.9940	0.9995
5	AF/1	Coho	194	195	0.9924	0.0062	0.9763	0.9941	0.9994
6	AF/1	Steelhead	127	127	0.9961	0.0055	0.9801	0.9982	1.0000
7	AF/1	Steelhead	126	127	0.9882	0.0095	0.9634	0.9906	0.9992
8	AF/1	Steelhead	127	127	0.9961	0.0055	0.9803	0.9982	1.0000
9	AF/1	Steelhead	127	127	0.9961	0.0055	0.9807	0.9982	1.0000
10*	AF/2	Chinook	120	196	0.6118	0.0347	0.5423	0.6123	0.6777
11*	AF/2	Coho	91	195	0.4671	0.0355	0.3980	0.4669	0.5363
12*	AF/2	Coho	97	195	0.4977	0.0356	0.4288	0.4976	0.5677
13*	AF/2	Coho	70	195	0.3593	0.0341	0.2936	0.3591	0.4267
14*	AF/2	Coho	49	195	0.2532	0.0311	0.1951	0.2525	0.3156
15*	AF/2	Steelhead	120	127	0.9414	0.0205	0.8948	0.9435	0.9748
16*	AF/2	Steelhead	126	127	0.9883	0.0094	0.9633	0.9907	0.9992
17*	AF/2	Steelhead	127	127	0.9962	0.0053	0.9805	0.9982	1.0000
18*	AF/2	Steelhead	126	127	0.9883	0.0095	0.9641	0.9908	0.9992
19	AF/3	Chinook	192	196	0.9771	0.0108	0.9518	0.9788	0.9931
20	AF/3	Coho	194	195	0.9924	0.0061	0.9769	0.9940	0.9994
21	AF/3	Coho	193	195	0.9872	0.0080	0.9671	0.9887	0.9979
22	AF/3	Coho	195	195	0.9975	0.0035	0.9876	0.9988	1.0000
23	AF/3	Coho	195	195	0.9974	0.0037	0.9867	0.9988	1.0000
24	AF/3	Steelhead	127	127	0.9960	0.0056	0.9803	0.9982	1.0000
25	AF/3	Steelhead	126	127	0.9883	0.0095	0.9634	0.9908	0.9992
26	AF/3	Steelhead	127	127	0.9961	0.0055	0.9802	0.9982	1.0000
27	AF/3	Steelhead	127	127	0.9962	0.0053	0.9811	0.9983	1.0000
28	AF/4	Chinook	195	196	0.9923	0.0063	0.9760	0.9939	0.9995
29	AF/4	Coho	195	195	0.9974	0.0037	0.9869	0.9988	1.0000
30	AF/4	Coho	195	195	0.9975	0.0036	0.9875	0.9988	1.0000
31	AF/4	Coho	195	195	0.9975	0.0035	0.9875	0.9989	1.0000
32	AF/4	Coho	194	195	0.9923	0.0062	0.9765	0.9939	0.9994
33	AF/4	Steelhead	127	127	0.9961	0.0055	0.9809	0.9982	1.0000
34	AF/4	Steelhead	127	127	0.9961	0.0054	0.9804	0.9982	1.0000
35	AF/4	Steelhead	127	127	0.9961	0.0055	0.9806	0.9982	1.0000
36	AF/4	Steelhead	127	127	0.9961	0.0055	0.9803	0.9982	1.0000
37	DF/1	Chinook	194	196	0.9872	0.0081	0.9672	0.9889	0.9979
38	DF/1	Coho	195	195	0.9974	0.0036	0.9873	0.9988	1.0000
39	DF/1	Steelhead	127	127	0.9961	0.0055	0.9810	0.9982	1.0000
40	DF/2	Chinook	195	196	0.9923	0.0062	0.9765	0.994	0.9995
41	DF/2	Coho	195	195	0.9974	0.0036	0.9873	0.9988	1.0000
42	DF/2	Steelhead	127	127	0.9961	0.0056	0.9801	0.9982	1.0000
43	DF/3	Chinook	195	196	0.9924	0.0061	0.9768	0.994	0.9995
44	DF/3	Coho	195	195	0.9974	0.0036	0.9876	0.9988	1.0000
45	DF/3	Steelhead	127	127	0.9961	0.0056	0.9801	0.9982	1.0000
46	DF/4	Chinook	195	196	0.9873	0.0081	0.9672	0.9889	0.9979
47	DF/4	Coho	195	195	0.9975	0.0035	0.9876	0.9989	1.0000
48*	DF/4	Steelhead	110	127	0.8625	0.0303	0.7987	0.8645	0.9157
49	FP	Chinook	194	196	0.9873	0.0078	0.9681	0.9888	0.9978
50*	FP	Coho	182	195	0.9311	0.01807	0.892	0.9327	0.9623
51	FP	Steelhead	127	127	0.9961	0.0054	0.9808	0.9982	1.0000
52*	SQ	Chinook	171	196	0.8708	0.0238	0.8207	0.8723	0.9136
53	SQ	Coho	194	195	0.9923	0.006249	0.9762	0.9939	0.9994
54	SQ	Steelhead	127	127	0.9961	0.0055	0.9808	0.9982	1.0000

* Indicates potential problem in the trial, red lines were censored

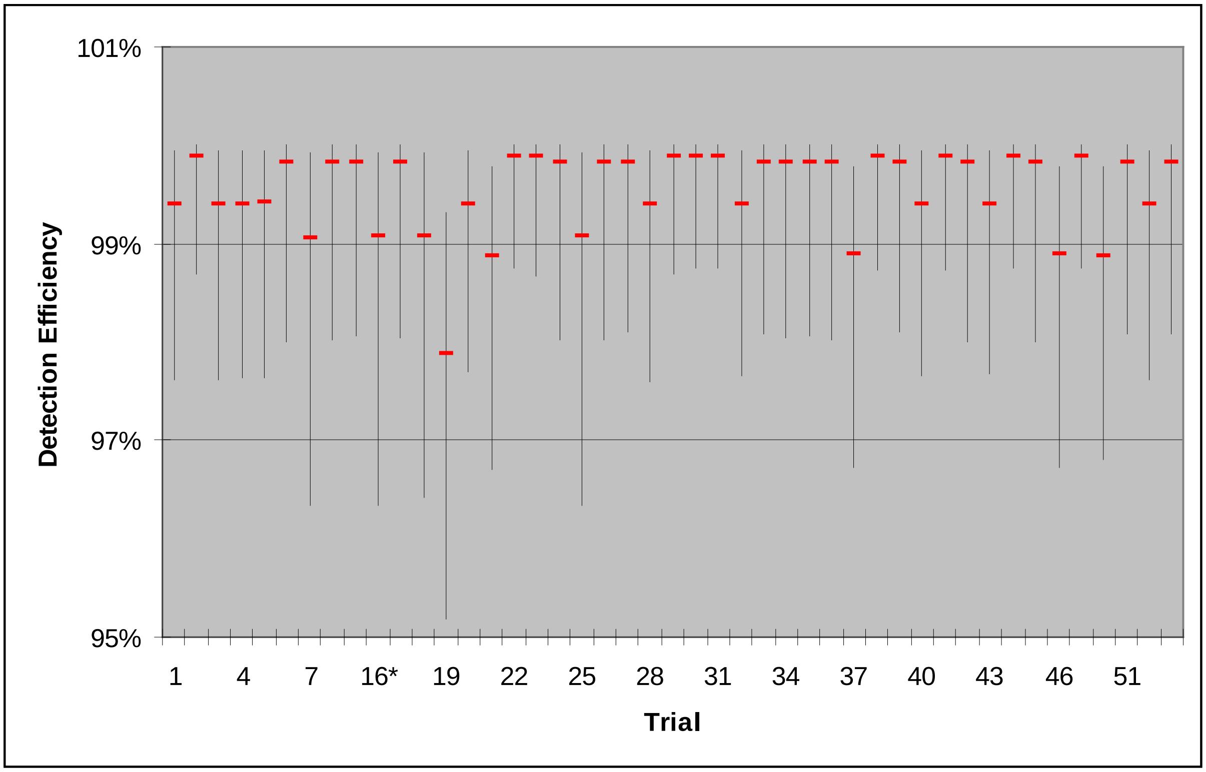

Figure 3. Detection efficiency of known PIT tagged fish in 45 trials using DF, AF, FP, and SQ readers. The red bar is the median for the 95% CI.

Tag Retention, Mortality, and Probability of Missing Tagged Fish

The estimated probability of detecting a tag in multiple trials was equal to 100% assuming 10, 22, and 22 independent passes for Chinook, coho, and steelhead, respectively. Tag retention estimates were approximately 98% (Table 4). Tag retention was not significantly different between species (χ^2 = 0.256, df = 2, P = 0.880). The estimated PIT tag survival rates ranged from 95% for Chinook salmon to 98% for steelhead (Table 5) and were not significantly different (χ^2 = 3.360, df = 2, P = 0.186). As in single pass trials (above), these data may not follow a χ^2 distribution but 95% CI overlap, implying no significant difference in tag loss or survival rates between species (Figure not shown).

Table 4. PIT tag retention rates for adult salmon and steelhead held for 7 to 32 days in the hatchery.

Species	Days	Tagged	Sampled	mean	sd	2.50%	median	97.50%
Chinook	7	195	198	0.9823	0.009338	0.9601	0.9839	0.9957
Coho	7	195	197	0.9874	0.007878	0.9681	0.989	0.9978
Steelhead	32	125	127	0.9805	0.01219	0.9499	0.9831	0.9968

Table 5. PIT tag survival rates for adult salmon and steelhead held for 7 to 32 days in the hatchery.

Species	Days	Alive	Sampled	mean	sd	2.50%	median	97.50%
Chinook	7	189	198	0.9521	0.01518	0.918	0.9539	0.9767
Coho	7	193	197	0.9773	0.01049	0.9529	0.9789	0.9931
Steelhead	32	129	131	0.9811	0.0118	0.9515	0.9835	0.9968

Detection Probabilities

Detection rates were analyzed using a general linear mixed model (GLMM) based on equation 1, which is the full model. The likelihood ratio test for the random effects found that adding tag scanner (Serial) number did not improve model fit ($P = 1.0$) so this variable was dropped from model 1. The best model (2) based on Akaike Information Criteria (AIC) includes the intercept, trial, species, and type. The best model (3) using Bayesian Information Criteria (BIC) included intercept, trial and species (Table 6). Model 4, which only considered the intercept, had a higher information criteria score than models 2 and 3. Since the penalty term for BIC includes both the number of parameters and the sample size, BIC selects more parsimonious models compared to AIC (Ntzoufras 2009). Using the intercept and species model, preferred by BIC, the predicted detection probability was estimated at 99.20%, 99.78%, and 99.83% for Chinook salmon, coho salmon, and summer steelhead, respectively. Since all three models have ΔBIC of less than 2, the results from each model are reported (Table 6). These results support similar detection probabilities using all models (Table 7) with the DF detecting PIT tags at a slightly higher detection rate than the AF, and coho salmon and steelhead having higher PIT tag detection rates than Chinook salmon.

Table 6. Results from mixed model selection using treatment contrasts or corner constraints to estimate regression coefficients.

Model 2	Estimate	Std. Error	z value	Pr(>|z|)	AIC	BIC
(Intercept)	4.4001	0.3571	12.323	< 0.001	37.18	45.75
Sp2	1.5358	0.5007	3.067	0.00216		
Sp3	1.8417	0.6037	3.051	0.00228		
Type2	0.9065	0.5347	1.695	0.09001		
Model 3	Estimate	Std. Error	z value	Pr(>|z|)	AIC	BIC
(Intercept)	4.8181	0.3027	15.916	< 0.001	38.39	45.25
Sp2	1.2794	0.4846	2.64	0.00829		
Sp3	1.5284	0.5849	2.613	0.00897		
Model 4	Estimate	Std. Error	z value	Pr(>|z|)	AIC	BIC
(Intercept)	5.7271	0.2135	26.82	< 0.001	44.21	47.64

Table 7. Estimates of detection probability by species and scanner type from the three different PIT tag detection models.

Model	Species/Type	2.50%	mean	97.50%
2	CK w/AF	97.59%	98.79%	99.39%
2	CK w/DF	98.28%	99.51%	99.66%
2	CO w/AF	99.13%	99.74%	99.81%
2	CO w/DF	99.48%	99.89%	99.96%
2	ST w/AF	99.24%	99.81%	99.87%
2	ST w/DF	99.56%	99.92%	99.97%
3	CK	98.56%	99.20%	99.56%
3	CO	99.32%	99.78%	99.93%
3	ST	99.37%	99.83%	99.95%
4	Adults	99.51%	99.68%	99.79%

Discussion

Our results suggested that individual PIT tag detection rates with hand held PIT tag detectors were greater than 98% and were consistent between scanner types. Using the same PIT tag with AF and DF readers, Hauser (2003) examined PIT tag detection rates in Pacific Halibut that were PIT tagged in the cheek and found each PIT tag reader had detection rates greater than 96% and no significant difference between readers ($P > 0.05$) in 19 trials. Although tag location in that study was different from ours (cheek for halibut versus peritoneal cavity for salmon), both yielded similar results, although our study had more trials and a slightly better detection rate. Since our detection rate approached 100%, using the assumption of perfect detection efficiency for salmon and steelhead would lead to only a slight negative bias in sampled PIT tags. Our results support that PIT tag detection rates in commercial and recreational fisheries can be high if thoughtful, well-designed protocols are developed and implemented. We only observed lower detection rates when there were equipment problems or protocols were not implemented correctly. This underscores the importance of properly maintaining testing equipment, proper training of staff, and quality assurance/quality control (QA/QC) programs.

Our tag retention and mortality results were similar to previous estimates. Knudsen et al. (2009) conducted a double tagging study on spring Chinook salmon. They found PIT tag loss for juveniles in the short term was 2%, which is similar to our 2% short-term estimate for adults but less than their long-term 18% estimate for returning adults. We demonstrated that short-term mortality was ~3% for adults held for 7 to 32 days, which is greater than the long-term average mortality rate of 10% estimated for fish tagged as juveniles and returning as adults. However, it should be noted that pre-spawning mortality for adult salmonids is common in the hatchery and our mortality rates were similar to those observed in untagged fish.

Our scanning protocols (Figure 2) call for the use of the pass over method. However, many samplers using the DF with the racquet use the pass through method, where fish are passed through the open racquet (Charlie Cochran, WDFW per. comm.). It is likely that the pass through method has high detection rates. We implemented the pass over methods because we wanted a consistent detection method in order to compare among reader types, and because we wanted our results to be applicable to real catch sampling, which sometimes does not allow for pass through detection. For example, large Chinook salmon cannot fit through the opening in the racquet, eliminating the possibility of consistent use of the pass through method for all adult salmon with the DF. In addition, the pass through method cannot be implemented for the smaller hand held detectors like the AF. We chose the pass over method because it can be consistently implemented under all hand held adult fishery sampling situations. WDFW adult PIT tag sampling protocols for commercial and sport fisheries are listed in Appendix 1 and 2, respectively.

Study Limitations

Bolker et al. (2009) in a review of GLMM noted that there are accurate techniques to estimate parameters in simple cases, but complex GLMM are challenging to fit, and model selection and hypothesis testing remains difficult. Our original study design was set up as a repeated measures design with each fish as a random effect along with individual fish covariates such as length, girth, sex, and species, and was analyzed using logistic regression based on the binary data (Hilbe 2009). However, we had challenges in estimating mixed effects with the logistic regression packages in R. Venables and Ripley (2002) noted the Hauk-Donner phenomenon can cause convergence problems when the fitted probabilities are close to 1, which occurred in our data set. We were unable to get reasonable results using the glmer in the lme4 using Laplace approximation of the likelihood or in the glmmPQL in the MASS package for the binary data. To address this problem we analyzed the data by blocking each trial as a random effect, but this led to a loss of individual data such as length, girth, sex, and tag location. We developed a Bayesian approach for the blocking effect in WinBUGS, which yielded the almost identical to our reported results to the results. Bolker et al. (2009) indicated that Bayesian methods might offer a solution in challenging cases in this study for binary data and we are currently exploring this approach.

One of the major limitations in this study was our use of fish PIT tagged as adults rather than as juveniles, which is when the majority are tagged in the Columbia River. However, seeding PIT tags to estimate efficiency is an accepted practice for hand held detectors (Hauser 2003) and for flat plate detectors (Evans et al. 2012). We chose seeding as a practical solution to ensure sufficient sample sizes. A key assumption in seeding tags is that the tag location and orientation in newly tagged adults is similar to returning adults or that if tag placement and orientation are different the read range in the detectors allows for the same detection rate for sown adults and juvenile-tagged fish. The read range is the manufactures estimate of the distance from which a tag can be detected. Many factors contribute to the read range including the tag type, operation frequency, antenna power, tag orientation and interference from other devices. The read range table from the manufacturers is presented for the 125 kHz and 134 kHz tags (Table 8). Read range for the 134 kHz tag used in this study with the DF, SQ, and FP is greater than 9 inches which is much further than the 2-6.7 inches for the lighter and more portable handheld units (AF, PR, and PS). Therefore, the DF, SQ, and FP should provide higher detection efficiencies. For this reason and based on the results of this study, WDFW uses the DF for all commercial sampling. The read range of the AF units was listed at 3.2 inches, which suggests it may have a lower probability detecting tags in larger fish such as Chinook salmon. However, field observations suggest the AF read range may be an underestimate (Steve VanderPloeg WDFW pers. comm.). WDFW sport samplers use the AF because samplers cannot carry the heavy and larger DF units and these DF units are substantially (6 times) more expensive than the AF units. Based on AIC model selection, separate detection rates are appropriate for the AF and DF, where AF had a lower detection rate, and separate detection rates for the three species examined with Chinook salmon having the lowest detection rate. However, based on BIC model selection separate detection rates are appropriate for each species but not scanner type.

Table 8. Read ranges in inches for PIT tag detectors used in this study with two different tags.

Model	Read Range (12.5mm, 125 kHz)	Read Range (12.5mm, 134.2 kHz)
FS2001F-ISO Reader with a racket antenna	5.0 - 8.5	9.0 - 14.5
FS2001F-ISO Reader with a flat plate	9.0 -12.5	12.0-16.5
FS2001F-ISO Reader with a 24" square	12.0	18.5
All Flex Model RS601-3	2.0	3.2
Pocket Reader	1.5 to 2.5	2.0 - 4.0
Psion Teklogix Workabout Pro 7527C-G2 w/Agrident AIRE 200 RFID	NA	6.7

A second concern could be our approach to estimating PIT tag loss, which was based on multiple independent detections of the same fish. We estimated tag loss at approximately 2% for each species in this study (Table 4). However, PIT tag location, orientation, and other factors could lead to some PIT tags not being detected. If this occurred than our assumption about independence may have been violated. Assuming there was no tag loss in our study and all the tags were present but missed, our reported PIT tag detection probabilities should be reduce from approximately 99% (Table 7) to 97%. Other PIT tag studies have documented adult PIT tag loss (Prentice et al. 1994, Knudsen et al. 2009) and our estimates of tag loss results are consistent with their findings.

Conclusions and Recommendations

This analysis indicates that applying our detection rates to PIT tag fishery sampling programs, where our protocols are followed, is appropriate. However, it should be noted that some fish are dressed or cleaned after they are caught, but before they can be sampled. Since these fish have likely expelled tags inserted into the peritoneal cavity, this leads to fewer fishery samples than coded-wire-tags (CWT) because CWT placement is in the snout and tag loss is not an issue for dressed fish. Samplers need to exclude dressed fish from samples they scan for PIT tags.

We make the following recommendations: 1) samplers should continue to implement protocols that were developed during this study that led to consistent and high PIT tag detection rates, 2) a quality control and quality assurance program is needed to ensure implementation of sampling protocols, 3) continued periodic tagging of hatchery fish and replication of this study (or one similar) to update detection rates should be conducted as needed, 4) sampling should be expanded to adult hatchery salmon tagged as juveniles to address the concern mentioned above, and 5) since we had a significant fixed effect for species using AIC for model selection it makes sense to expand sampling to include sockeye salmon and other races of Chinook salmon.

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Appendix 1: Procedures/Protocol for PIT sampling in the Columbia River Commercial/Treaty Fisheries, Version 4.0

Procedures/Protocol for PIT sampling in the Columbia River Commercial/ Treaty Fisheries

It is essential to follow the protocols and procedures below. Failure to follow these will lead to undetected PIT tagged fish, which will provide biased estimates of harvest and stock composition. Therefore, all staff will follow these procedures and protocols daily. If you have any questions please contact: Ben Warren @ Cell: (360) 635-2318, Office: (360) 906-6700 Ex: 6844 and/or E-mail Benjmin.Warren@dfw.wa.gov or your supervisor.

I will assign units to crews, or have units available for crews to take with them on the table in the PIT station / CWT/DNA data summary cubicle. The unit should be ready for you to just turn on and start sampling.

1. **Enter the PIT detector unit number into the header comments of each form** – This number is found on the front and base of the Cheeseblock units. Use the prefix 'CB' for Cheeseblock units. Ie Sampler has **Cheeseblock # 3**, so sampler would enter **CB3** in the comments.
2. **Scan the dummy tag at the beginning of each session!!!** (Dummy tags kept in front zipper pouch of Cheeseblock bag) A session changes when you start a new day or switch sampling locations (i.e. Sport Locations: Section 1-10) Don't worry about changes between species.
3. Follow Scanning protocols below and Unit Operation on Page 2 to scan fish
4. **Scan the dummy tag at end of each day!!!**

Scanning Protocol

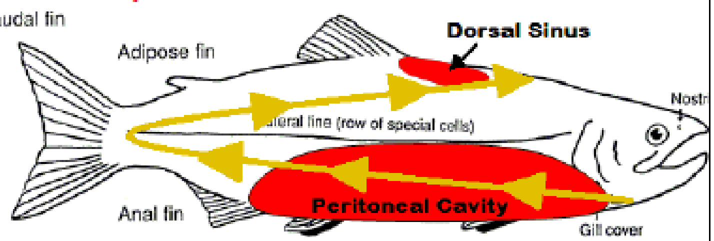

Figure 1

The preferred location for PIT tagging is the peritoneal cavity; gutted salmon should not be PIT tag scanned because they are likely to lose their tag. **Gutted or 'snout-missing' fish are not to be sampled,** just counted as fish caught.

We will be using the **2-swipe method** for scanning salmonids as illustrated below (Figure 1). The cheeseblock is set to: Continuous mode: Off, so you have to hold the button down during the entire scan. Speed can affect the detection rate, so try to scan both swipes with care.

When a PIT tag is detected, follow the steps below:

a) During sampling, PIT tagged fish are treated the same as CWT fish. They will be set aside to work up for bio data just like we treat CWT fish.

b) When the data-logger entry person is ready, the fish is scanned with the hand-held PSION and the tag # populates the field.

1

As seen in figure 1 below, a complete kit for Commercial PIT sampling will include:

a) Cheeseblock, b) Cable, c) Snuggly*, d) Racket and e) bag (which should have a dummy tag and protocol in the front pouch)

*We have front-pack baby carriers (Snugglies) for carrying/holding the cheeseblock. It's optional, but the preferred method. This keeps the unit and cables out of the way and close to the PIT sampler to better hear the beep and see the readout. If you don't use the snuggly you should cover unit in snout bag (Ripping a small hole for Antenna cable, and using a rubber band to keep bag closed works well) Make sure you can read the PIT display, because that is the ultimate detection: seeing the PIT number on display

You have to connect the racket to the cheeseblock in the correct configuration. The cables are labeled to minimize confusion, but as seen in figure 2, the end with the cable-shielding box (circled) goes to the transceiver (cheeseblock).

- Please be careful when attaching and detaching these cables. They are kind of sticky. You have to kind of wiggle them while screwing on and off. Screw on caps when not using.

3 Bars in upper left corner of screen (left to right) are: Battery level, Data memory, and interference.

Menu button – scrolls through menu settings.

Enter button – when you want to change a setting, hit Enter and follow the prompts. Hit Enter again when you've finished manipulating the setting. Hit Escape or Menu when you've finished.

When you're in the standby/scan screen (The screen it's in when you turn it on), hitting Enter scrolls though parameters and values (ie. Battery level, SIG level (interference), antenna power, time, etc..)

When you first turn the unit on, there should be a number in the upper right corner of the screen. That's the file number, when you can see this as a number (0 is a number), we should be ready to store PIT detections in a file. If you see dashes, there's no file created and you'll have to create one. See page 4 for creating a file. If it's just blank, then the store mode is set to off, and it won't store, you'll have to change store mode to 'All' (Pg 3).

PIT samplers: Do best to stay out of way, cable can really get in the way, be aware of this. Always be aware of surroundings and looking for ways to better position yourself. If needed, help the crew by setting unit down in a safe/out-of-the-way location and lending a hand. **Always be paying attention!!**

End of day: you can hose the racket off, but **don't spray the wire connections or the Cheeseblock**. Please wipe them off though.

Figure 1

Figure 2

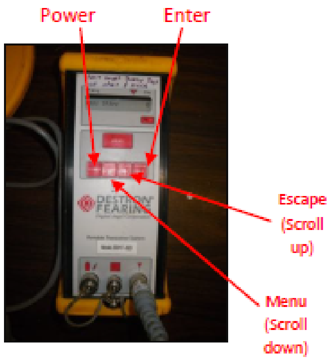

Figure 3

Appendix 2: Procedures/Protocol for PIT sampling in the Columbia River Sport Fishery, Version 3.0.

Procedures/Protocol for PIT sampling in the Columbia River <u>Sport</u> Fishery

It is essential to follow the protocols and procedures below. Failure to follow these will lead to undetected PIT tagged fish, which will provide biased estimates of harvest and stock composition. Therefore, all staff will follow these procedures and protocols daily. If you have any questions please contact: Ben Warren @ Cell: (360) 635-2318, Office: (360) 906-6700, Ex: 6844 and/or E-mail Benjamin.Warren@dfw.wa.gov or your supervisor.

Sampling Session Protocol

1) Enter the PIT detector unit number into the Header comments of each form – This number is found on the handle of the AllFlex units (Figure 1). <u>Use the prefix **AF** when entering unit number</u>.

ie. Sampler has AllFlex unit # 8, so he/she would enter <u>**AF8**</u> into the header comments.

2) Scan a Green dummy tag (Figure 2) at the beginning of each DAY.

Figure 2

The Unit number is here

Figure 1

3) Follow Scanning protocols and Unit Operation on Page 2 to scan fish.

4) Scan a Red dummy tag (Figure 3) at end of each DAY.

Figure 3

It is mandatory to scan tags at the beginning and end of the sampling day to ensure that the unit is working properly.

It is **important** to keep these dummy tags away from the unit when you're scanning a fish. If you detect one of these dummy tags while scanning a fish, you can mistake it for a PIT tag in the fish. Therefore, we recommend that dummy tags be stored in zip-closed pocket on back of vest or kept in truck. DO NOT LOSE DUMMY TAGS

If you change from <u>Mainstem sampling to Tributary sampling in one day</u> (or vice-versa), they need to be separate sessions. So when you change from one to the other, end session by scanning the Red tag, and start new session by scanning the Green tag again between the 2 different sampling types.

5) At end of EACH work week, leave units to be downloaded on large table in PIT cubicle labeled, "**PIT Detectors needing to be downloaded**" (Regardless of detections or not). When I've finished downloading, I'll put your unit on small white table labeled, "**AllFlex units downloaded and ready to go**". Samplers are assigned the same unit for the season, so you are responsible for that unit all season.

6) At end of EACH work week, I will download the units and replace your batteries with fully charged batteries. Use re-chargeable batteries (Figure 4) for everyday use, and keep the alkalines (Figure 7) for backup. You can use the carrying cases for your alkaline storage (Figure 8) located in PIT station.

Figure 4

Figure 7

Figure 8

Unit Operation

Figure 9

- **Turn unit on** : Press the Read Button. The display should say 'Reader Ready' (Figure 9).
- **Scan**: Press the Read Button and the display should say 'Reading' (Figure 10) The unit will scan for 5 seconds.
- If you detect a tag, the tag # (ie. 3D9$_{H}$ 1C2D48010B$_{H}$). should appear in the display (Figure 11); in this example the last 7 = D48010B.
- If no tag was found, the display will indicate that (Figure 12).
- See scanning protocol below for directions on entering last 7 digits of PIT tag # into form.

Figure 10

Figure 11

Figure 12

- The unit will automatically shut down after 2:00 minutes

- If you detect a tag, but didn't get the PIT #, scan it again, **or** if fish is no longer in hand, press menu button until you see 'STORED ID-CODES', then press scroll down **<u>exactly 4X</u>**. This will show you last tag detected. Be very careful not to delete any Stored ID_Codes.

Scanning Protocol

The preferred location for PIT tagging is the peritoneal cavity; gutted salmon should not be PIT tag scanned because they are likely to lose their tag. **<u>Gutted or 'snout-missing' fish are not to be sampled,</u>** just counted as fish caught.

We will be using the **2-swipe method** for scanning salmonids as illustrated below (Figure 13). The AllFlex units are set to scan for 5 seconds. Utilize the entire 5 seconds to scan the fish, so try and time it for 2.5 seconds on 1st swipe, and 2.5 seconds on 2nd swipe.

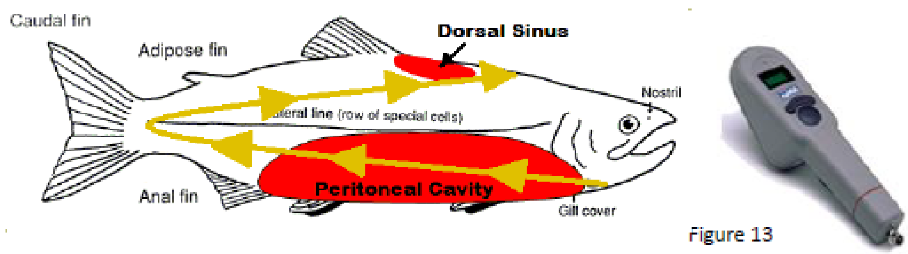

Figure 13

When a PIT tag is detected, follow the steps below:

a. In the **BioData** sub-form: In the **remarks** field, **following the DNA #**, enter the characters '**P1**'

b. In the **Effort** sub-form: In the **comments** field, enter the characters '**P1**' then a **space**, followed by the **last 7-digits** of the PIT # found in the AllFlex screen. (ie. P1 010B)

P1 denotes that was the 1st fish with a PIT tag for **that fishing party**. If by chance, a fishing party has more than 1 PIT tagged fish, you would denote the 2nd fish by entering 'P2' in the remarks field following the DNA # in the BioData sub-form. You would then enter 'P2' then a space, followed by the last 7-digits of the PIT # in the comments field of the Effort sub-form. Each fishing party should have their own Effort sub-form.

- 2 -

Estimates of Columbia River Salmon and Steelhead Harvest Rates for the 2010 Fall Commercial and Treaty Fisheries based on Passive Integrated Transponder (PIT) Tags

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November 7, 2014

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Extended Abstract

In 2010, the Washington Department of Fish and Wildlife (WDFW) and the Pacific States Marine Fisheries Commission (PSMFC) added Passive Integrated Transponder (PIT) tag sampling to the existing mainstem Columbia River fishery sampling program that collects biological data and recovers Coded Wire Tags (CWT) from sport and commercially harvested salmon and steelhead for use in salmon recovery and fisheries management decisions. We conducted a study to determine the feasibility of using PIT tags recovered from fisheries to estimate harvest rates for Columbia River salmon and steelhead. In our study, we focused on estimating PIT tag detection rates and expanding the PIT tags recovered in the fishery based on sample rates detected at Bonneville Dam (BON) to estimate harvest rates for PIT tagged upriver (above BON) populations. We conducted a PIT tag detection study in a hatchery and estimated detection rates in adult fish ladders at BON and found PIT tag detection rates for Chinook salmon, coho salmon, and steelhead exceeded 98% for handheld scanners and in the fish ladders. This harvest study was opportunistic in that it took advantage of the existing juvenile PIT tagging program throughout the Columbia Basin, to estimate harvest rates. As expected, harvest rates were variable and likely imprecise when adult PIT tag returns to BON were low or when tag recoveries were few in the fishery. To address this concern, we calculated harvest rates at a range of population scales including individual release groups and major tributaries, Evolutionary Significant Units (ESU)/Distinct Population Segments (DPS), and for larger aggregates. Pooled tag groups used to estimate harvest rates were supported both by life history attributes and 2010 run timing graphs for each species at BON. We sampled the fall commercial fishery below BON and the fall treaty fishery above BON in 2010. Estimates of harvest rates below BON in the non-Indian commercial fishery were less precise than above BON treaty fishery due to lower tagging rates and smaller numbers of tag recoveries in this area, in addition to a late start in sampling for PIT tags. For the treaty fishery, coho salmon harvest rate estimates were similar for the four hatchery groups and provided a precise estimate for the aggregate of ‘early’ coho salmon (17% with a 95%CI from 14% to 20%). Using a mixture model we estimated the harvest rate for the ‘late’ coho salmon group (4% with a 95%CI from 2% to 4%). For steelhead, a total of 26 individual population and multiple DPS harvest estimates were calculated. Our harvest estimate for Snake Group B steelhead (8%) was lower than the Columbia River Technical Advisory Committee (TAC) estimate (15%), while our wild Group A estimate (5%) and TAC’s (4%) were similar, but we noticed variation in Distinct Population Segment (DPS) estimates. For example, our Middle Columbia River DPS estimate was 3% while our Upper Columbia River DPS estimate was 10%. Our estimated Chinook salmon harvest rates were higher for Tule fall Chinook (64%) and lower for Snake River fall Chinook salmon (9%) compared to the TAC estimates (51% and 18%, respectively). We detected size selectivity in the fishery catch relative to the escapement based on PIT tag analysis. Since we did not estimate harvest rates by age or size, our harvest rate estimates were likely biased low for adults and larger fish and biased high for jacks and smaller fish due to higher catch rates of larger fish in fisheries. We recommend the continued recovery of PIT tags as part of the fishery sampling program as well as the development of age or size structured harvest estimates using PIT tag data. Since this was a feasibility study, final salmon and steelhead Columbia River harvest estimates by reporting group are available in WDFW and ODFW (2011).

Introduction

Passive Integrated Transponder (PIT) tagging of salmonids in the Columbia River Basin has served a variety of purposes including estimation of juvenile and adult survival and the investigation of mechanisms affecting survival (Connor et al. 1998, Zabel and Accord 2004, Buchanan et al. 2006) such as predation by birds and northern pikeminnow (Collis et al. 2001, Petersen and Barfoot 2003), and habitat characteristics (Paulsen and Fisher 2005). Based on the PIT tag database (PTAGIS), the number of applied PIT tags has increased from less than 20,000 tags applied in 1988 to over 2,000,000 in 2009. In addition, the number of returning PIT tagged adults, as measured by detections at Bonneville Dam (BON), has recently exceeded 30,000 individuals. Detection systems have advanced from hand held devices (Busby and Deegan 1999) to detectors located in juvenile bypass systems and adult traps and ladders (Harmon 2003), and instream arrays capable of passively detecting tagged adult and juvenile fish as they migrate (Connolly et al. 2008). Almost $4,000,000 annually is dedicated to the purchase of PIT tags, and millions more are spent capturing and tagging fish, recovering tags, and storing data in the Pacific States Marine Fisheries Commission (PSMFC) PIT Tag Information System (PTAGIS), a coastwide PIT tag database. Harvested PIT tagged fish represent one of the largest sources of unaccounted mortality for PIT tagged fish in the Columbia Basin and could provide valuable information to researchers, managers, and policy makers. This is especially true of harvest rates for groups of fish that are generally not tagged with code-wire tags (CWT) but are often PIT tagged. These groups often include natural origin salmon and steelhead. Thus, PIT tag recoveries in fisheries may also allow managers to better estimate harvest impacts on at-risk natural origin populations and to shape fisheries to reduce impacts on ESA listed fish based on their spatial and temporal occurrence. The purpose of this study was to estimate the harvest rate of PIT tags from commercial and treaty sampled fisheries in the fall of 2010 using the detection rate from hand held scanners. Scanner detection rates by species were available from an adult PIT tag detection study by Rawding et al. (2013). In this we report we estimated harvest rates for salmon and steelhead in mainstem Columbia River fisheries in the fall of 2010.

Methods

Data Collection

Fall commercial (non-Indian) fisheries for salmon occur annually below BON from August through October in Zones 1-5. The 2010 fall fishery was a drift gillnet fishery with the following mesh restrictions: 9 to 9.75 inches in August, 8-9.75 inches from September 22 to Oct 11, 6 inch maximum from October 12 to 14, 8-9.75 inches from October 14 to 19, no mesh restriction on October 20, and 8 inch minimum on October 21 (WDFW and ODFW 2011, Table 15). Treaty fisheries occur during the same months in Zone 6, which is the area from BON to McNary Dam (MCN). Treaty fisheries are mainly a gillnet fishery using set nets, however drift gillnets, hoopnets and hook and line are also gear types used to commercially fish. There were no mesh restrictions for treaty fisheries except an 8-inch minimum mesh size from September 27 to October 22 (WDFW and ODFW 2011, Table 19). These fishery zones are depicted in Figure 1. Fisheries are sampled through a coordinated effort between the Washington Department of Fish and Wildlife (WDFW), Oregon Department of Fish and Wildlife (ODFW), and the Pacific States Marine Fisheries Commission (PSMFC). Commercial catches in Zone 1-5 are sold to buyers with ODFW primarily sampling the Oregon catch, and WDFW and PSMFC sampling the Washington catch. In Zone 6, PSMFC and WDFW sample the majority of the treaty catch. The original purpose of the fishery sampling program was to estimate the individual stock contributions to the overall catch using CWT groups primarily from hatchery production. However, this same CWT sampling program has been used to collect data for mixed stock genetic fishery analysis (Kassler et al. 2002) and in this project WDFW and PSMFC sampling was expanded to include scanning of harvested adult salmon and steelhead for PIT tags.

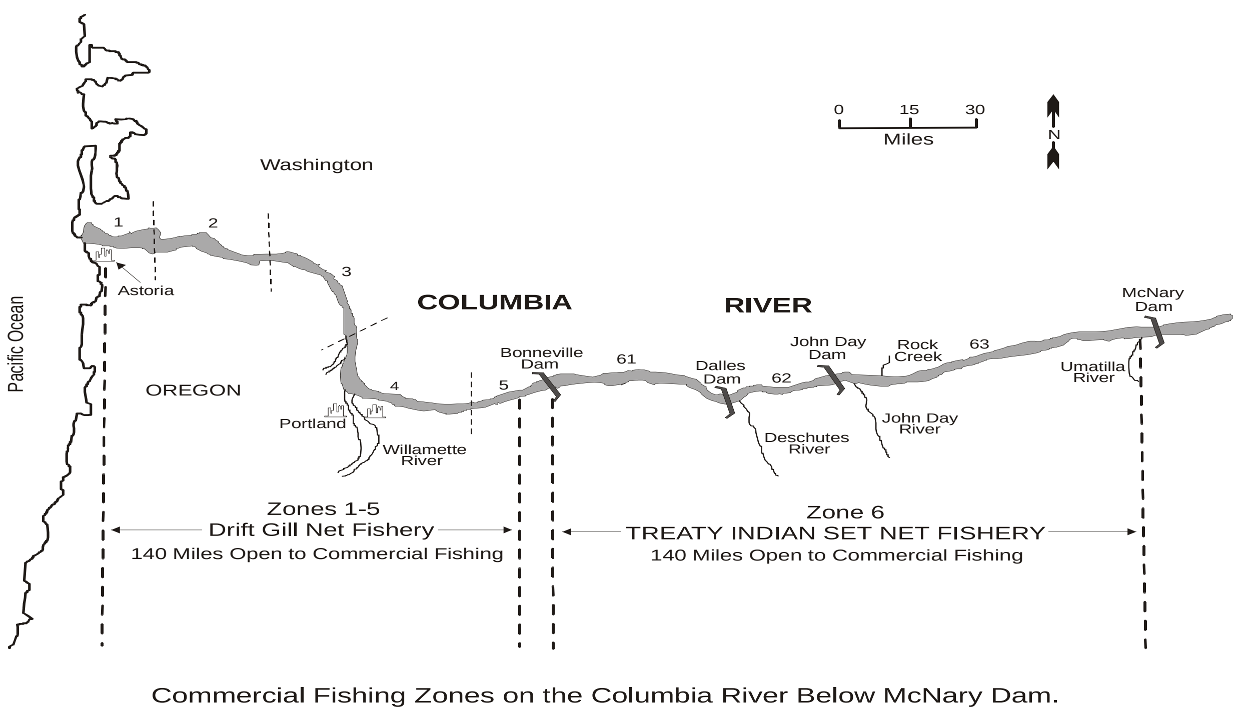

Figure 1. Map of commercial fishing zones sampled in fall non-Indian and treaty commercial fisheries.

Juvenile salmon and steelhead throughout the Columbia River are tagged using standard Columbia Basin PIT tagging procedures from CBFWA (1999). In 2010, adult salmon and steelhead in the fall fisheries that were not dressed (gutted) were sampled by WDFW and PSMFC samplers for CWT and PIT tags. Sampling for PIT tags was conducted with a Destron Fearing (DF) FS2001F-ISO Reader Base Unit with racquet antenna. More details on the fishery sampling can be found in Nandor et al (2011). Most smaller adults were scanned for PIT tags by passing the fish through the opening of the racquet (pass through method). However, larger fish especially Chinook salmon were scanned using the pass over method (Rawding et al. 2013). The PIT tag number and date/time of each detection was recorded on the scanner and later downloaded at the office.

If a sampler heard the scanner beep indicating a PIT tagged was detected the last four digits of the PIT tag number along with biological information including species, date, length, sex, and fin marks were recorded on a data form. This was done to link the PIT tag to the biological information. However, because samplers work in a noisy environment and sample a high volume of fish, many times the beep was not heard and only the PIT tag number and date stamp were recorded on the scanner. In addition to PIT tag detections, samplers maintained a count of the number of each species of fish sampled for PIT tags at each location for a specific time period called a session. At the end of the season, all fish detected with PIT tags were uploaded to PTAGIS.

The commercially sold Columbia River catch is recorded in pounds (lbs.) on fish tickets and this information is reported to WDFW and ODFW by commercial buyers. This total represents the reported catch in Zones 1-5. In Zone 6, not all of the catch is sold to commercial buyers and some of the catch may be sold “over the bank” to the public or retained for personal use. This catch, not sold to commercial buyers, is reported by treaty tribes as non-ticketed catch. In addition to the CWT and PIT sampling reported above, a systematic sample of weights of undressed harvested salmon and steelhead was taken to convert the landed pounds of fish on the fish tickets to an estimate of the total number of fish harvested by species and period.

The design of the CWT program used to estimate harvest has been well documented (Bernard and Clark 1996, Bernard et al. 1998). In the design phase, managers may conduct a power analysis to ensure that tagging and sampling programs release and recover enough CWTs to meet management goals. However, evaluating harvest is not primary purpose of juvenile PIT tagging, thus using PIT tags to estimate harvest rates is opportunistic, much like studies of bird predation on juvenile salmonids (Collis et al. 2001). Some small juvenile release groups have a low probability of surviving to the adult stage and an even a lower chance of being sampled in fisheries. Reporting harvest rates for these small individual tag groups may lead to imprecise harvest estimates. Although there are no exact guidelines for the minimum number of tags needed to estimate harvest rates per group, Hankin et al (2005) suggest ~10 recoveries per stratum, which is similar to the range of 5-10 tag recoveries for mark-recapture studies (Seber 1982, Schwarz and Taylor 1998). Therefore, we reported harvest for groups with a minimum of one recovered tag but we also pooled tag groups into larger aggregates by river, management group, and ESU or DPS for reporting purposes to address the potential bias in harvest rates caused by the low number of recoveries.

The catch or landings data were obtained from a variety of sources including: 1) WDFW and ODFW (2011 - Table 16), 2) landings for 2010 Columbia River Mainstem August Fisheries, and 3) landings for 2010 Columbia River Mainstem August Fisheries and Late Fall Fisheries available at from ODFW (http://www.dfw.state.or.us/fish/OSCRP/CRM/comm_fishery_updates_10.asp). Individual fish weights were based on sampling these fisheries and provided by ODFW. The PIT tag sampling rate was provided by PSMFC (Bryant Spellman, PSMFC, unpublished data) and all PIT tag information including fishery catch, mainstem dam, and other detections were queried from PTAGIS.

Harvest Reporting Groups

Arbitrary pooling of returning individuals from various release sites or populations can lead to aggregate groups of fish that may not experience homogenous harvest rates. However, the population structure of Columbia River salmon and steelhead has been summarized by Weitkamp et al. (1996), Busby et al. (1996), Myers et al. (1998), and McClure et al. (2003). These authors suggest that salmon and steelhead populations are hierarchically organized into major population groups and then Evolutionary Significant Units (ESU) or Distinct Population Segments (DPS).

We hypothesized that ESU or DPS membership would be a suitable surrogate for susceptibility of individuals and populations to harvest. Individuals and populations comprising an ESU or DPS share similar life history attributes, such as size and age at maturity, and migratory timing, both of which are known to affect susceptibility to harvest (Kendall et al. 2009). Therefore, in order to estimate harvest rates we needed to group individuals by run timing and other population membership information. We decided upon three kinds of harvest reporting groups organized around ESUs and DPSs at hierarchical spatial scales: 1) a series of "ESU/DPS" groups which included all returning PIT tagged fish that could be assigned to an "ESU/DPS" group based on their release site, species, and run type, or in the cases of unlisted populations (e.g., coho salmon above the White Salmon and Hood rivers) to spatial groups organized to mimic existing ESUs for other species; 2) a series of smaller spatial scale "Rivers" groups which included all releases in individual major rivers or other subpopulation aggregates comprising the various "ESU/DPS" groups; and 3) a series of "Large Aggregates" groups which pooled harvest rates at varying spatial scales greater than the "ESU/DPS" groups. At each of these levels, we also estimated harvest rates for hatchery and wild origin fish separately, as well as pooled. We restricted our dataset to only PIT tagged returns with release sites, species, rear types, and run types, that could unambiguously be assigned to one of our categories. For example, all fish tagged as at BON as jacks or adults were excluded; all Chinook with an unknown run type were excluded, steelhead tagged as smolts at mainstem Columbia River dams below the Snake were excluded since they could not be assigned to "ESU/DPS" groups and we did not calculate an overall "Above BON" harvest rate for steelhead.

In order to determine whether pooling of our harvest groups was appropriate, we developed a graphical analysis to determine whether candidate members of a potential group shared similar run timing at BON. We plotted the cumulative run timing of candidate groupings of returning PIT tagged salmon and steelhead. We compared the timing of fish from each "ESU/DPS" as well as each major tributary or subpopulation—"Rivers" group contributing to each "ESU/DPS" group with at least 10 returning adults detected at BON. Steelhead were further divided in the Snake River into Snake River A and B runs based on Busby et al. (1996). Chinook salmon were subdivided beyond the ESU level in a few cases based on traditional harvest accounting in the Columbia River (e.g., Lower Columbia Chinook were split into "Bright" and "Tule" groups, and Hanford Reach Chinook were separated from Upper Columbia and Middle Columbia River stocks). Based on the graphical analysis we developed groups with similar timings for harvest analysis.

Statistical Analysis

A Bayesian framework was used to estimate parameters in our analysis (Gelman et al. 1995). The goal of the Bayesian approach is to calculate the probability of a specific parameter (θ) given the data (x), written as $p(\theta|x)$. Bayes theorem is a conditional probability statement that proves the $p(\theta|x)$ is proportional to the sampling distribution for the data $p(x|\theta)$ multiplied by an independent probability distribution for the parameter, $p(\theta)$ (Gelman et al. 1995). The formula of the posterior distribution may be complex and difficult to derive. Samples from the posterior distribution can often be obtained using Markov chain Monte Carlo (MCMC) simulations (Gilks et al. 1995). WinBUGS is a software package that implements MCMC simulations using a Metropolis within Gibbs sampling algorithm (Spiegelhalter et al. 2003) and has been used in salmonid studies (Rivot and Prevost 2002, Link and Barker 2010).

One of the most controversial aspects of the Bayesian approach is the specification of priors. We used vague priors for this analysis with the intent that the posterior distribution be dominated by the observed data. When this occurs the results obtained from Bayesian and maximum likelihood methods yield similar results (Kery 2010). The priors are specified below and are standard vague priors which allow the data to form the posterior distribution. Convergence was tested using the Brook-Gelman-Rubin (BGR) diagnostic (Ntzoufras 2009) and precision was assessed by monitoring MC % error. The BGR diagnosis compares the between and within sample variability. Although convergence cannot be assured, a BGR value of less than 1.1 is generally acceptable, and indicates that the MCMC simulations have stabilized (Kery 2010). The MC % error measures the variation of a parameter due to simulation, and to obtain precise parameter estimates it is recommended that the MC% error divided by the standard deviation be less than 5% (Lunn et al. 2012).

We used a directed acyclic graph (DAG), which is a graphical representation of a statistical model that facilities understanding of the model structure. A DAG has three elements: nodes, plates, and edges. The node can be 1) a constant, such as data or a prior distribution in a Bayesian analysis, 2) a stochastic sampling distribution, such as normal, binomial, or beta distributions, or 3) a logical expression (such as a regression equation $y = a + bx$). In our DAGs rectangles represent constants and ellipses are observed and unobserved probabilistic nodes. The conditional dependencies between nodes are expressed as arrows called edges. Solid lines depict deterministic or probabilistic dependencies and hollow or dotted lines represent logical relationships or dependencies. The repeated part of the graph is represented by a frame or plate.

All quantities are depicted as nodes in the DAG, in which arrows run into nodes from their direct influences (parents). The model represents the assumption that, given its parent nodes pa[*v*], each node *v* is independent of all other nodes in the graph except descendants of *v*, where a descendant is defined. The conditional independence assumptions represented by the graph mean that the full joint distribution of all quantities *V* has a simple factorization in terms of the conditional distribution p(*v* | parents[*v*]) of each node given its parents. A crucial concept is that the parent-child distributions are all that is needed to fully specify the model. The BUGS software (Spiegelhalter et al. 2003) compartmentalizes the necessary sampling methods directly from the expressed graphical structure. More importantly, constructing the DAG in BUGS, which is sometimes referred to as a Doodle, provides the BUGS code for parameter estimation including a point estimate and Bayesian confidence interval. DAGs are used often to depict model structure in ecological analyses (Link and Barker 2010, King 2010).

The DAG (Figure2) with the underlying statistical frameworks are presented for the PIT tag detection and harvest models. The PIT tag detection analysis from Rawding et al. (2013) was modified from a frequentist to a Bayesian approach to be consistent with incorporating the uncertainty from the detection study into the uncertainty associated with the harvest estimates. Their detection study was designed to collect PIT tag presence/absence response data and relate it to explanatory variables including the species (e.g., coho, Chinook, steelhead) and scanner type—Destron Fearing (DF) FS2001F-ISO Reader Base Unit (DF) with racquet antenna or All Flex (AF) Model RS601-3). The data were analyzed using a generalized liner model (GLM) to accommodate the non-Gaussian distribution (O' Hara and Kotze 2010). When both fixed and random effects are included in a GLM it is referred to as a mixed or mixed-effects model (GLMM). The data were analyzed based on the following equation using corner constraints (Ntzoufras 2009):

$$\text{Logit}\,(p\_det_{ij}) = \mu + Trial_i + Species_j + Type_j + Serial_{ij} \qquad (1)$$

where μ is the intercept, *Trial* is the complete sampling of a species with each scanner type and serial number, *Type* is the scanner unit, and *Species* is Chinook salmon, coho salmon, or steelhead. The fixed effects included species and scanner type. PIT tag scanners were classified into two types the DF used for portable commercial fisheries sampling, and All Flex (AF) used for sport fisheries sampling. A block is a property of the experimental unit, which is the trial in our study design. Since we were not interested in the block effects but had to account for their effect, the blocks were treated as random effects.

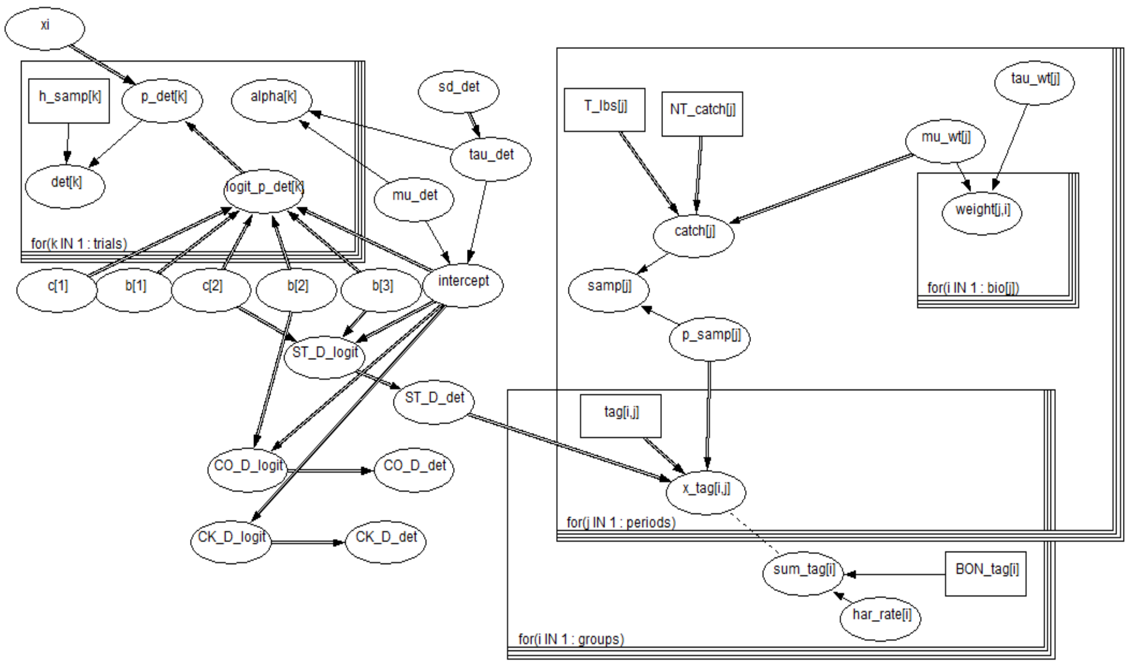

Figure 2. Directed acyclic graph for the PIT tag detection study (left side) and estimate of steelhead harvest from Zone 6 fisheries (right side). There were slight modifications of this DAG for other species: 1) the detection probabilities varied by species and PIT detector type used to sample a particular fishery; and 2) the run size serving as the denominator in harvest rate estimates differed between below-BON and above-BON fisheries. The run size above-BON was simply the number of PIT tags passing BON, whereas fishery removals below-BON were added to this number to generate a river mouth run size to serve as the denominator for lower river harvest rates.

Harvest Model Components

The estimate of catch in the treaty fishery is obtained through a creel census program but a variance for the catch is currently not available (Marianna McClure, CRITFC, pers. comm.). The catch from this fishery can be split into ticketed catch (i.e. sold to large buyers/processors) reported in pounds of fish by species, and non-ticketed catch reported counts of fish by species. Non-tribal commercially caught salmon and in Washington and Oregon are reported to the states in pounds on fish tickets by species. Non-treaty commercial sale of steelhead is below BON is prohibited. Biological data, including weight, is collected from a subsample of commercially sold fish. The mean weight can be estimated by

$$weight_{ji} \sim Normal(wt\_mu_j, wt\_tau_j) \tag{2}$$

where $_i$ is an individual $weight_i$ in period $_j$, wt_mu_j is the estimated mean weight, and wt_tau_j is 1/variance for each period *j*. The ticketed catch (count of fish) is estimated by

$$T\_catch_j = (lbs_j / wt\_mu_j) \tag{3}$$

where lbs_j are the pounds of fish. The non-ticketed catch (only treaty fishery), which is used for ceremonial and subsistence purposes or sold directly to the public, is estimated by subtracting the estimate of the ticketed catch from the treaty catch by

$$catch_j = T\_catch_j + NT\_catch_j \quad (4)$$

where NT_catch_j is the non-ticketed catch. Although not perfect, this method was used to account for uncertainty in the weekly ticketed catch in the Zone 6 treaty fishery, whereas $catch_j$ was exactly equal to T_catch_j in the commercial fishery. A subsample of ticketed catch is inspected for PIT tags as described in the sampling section:

$$samp_j \sim Binomial(p\_samp_j, catch_j) \quad (5)$$

where samp$_j$ is the weekly number of fish landed that were sampled, and p_samp$_j$ is the proportion of the weekly catch that was sampled. The expanded number of tags by group k is estimated by

$$x\_tag_{jk} = (tag_{jk} / p\_samp_j) / h\_\mathrm{det}) \quad (6)$$

where $_k$ denotes the harvest reporting group, x_tag_{jk} is the expanded number of weekly tags, tag_{jk} is the number of sampled tags in the fishery, and h_det is the PIT tag detection rated using handheld detectors. The total tags by group in the fishery is estimated by

$$sum\_tag_k = \sum_{k=1}^{K} (x\_tag_{jk}) \quad (7)$$

which is the sum of the weekly expanded tags by group. Since not all PIT tags are detected at BON we estimated the PIT tag detection rate at BON by

$$B\_tags \sim Binomial(pBON\_\mathrm{det}, M\_tags) \quad (8)$$

where $pBON\_det$ is the probability of being detected at BON, based on the number of unique PIT tags detected at MCN plus those missed at BON (M_tags), and B_tags the number of PIT tags detected at BON. The expanded number of tags by group passing BON is estimated by

$$xBON\_tag_k = BON\_tag_k / pBON\_\mathrm{det} \quad (9)$$

where $xBON\_tag_k$ is the expanded number of tags per group. The harvest rate in the Zone 6 treaty fishery was estimated by

$$sum\_tag_k \sim Binomial(HarvRate_k, x\_BON\_tag_k) \quad (10)$$

based on the expanded tags in the fishery and those passing BON, whereas equation 10 was modified to

$$sum\_tag_k \sim Binomial(HarvRate_k, x\_BON\_tag_k + sum\_tag_k) \quad (11)$$

in order for the denominator in below BON harvest rates to include fishery removals between BON and the river mouth as previously described, by adding the expanded PIT tag catch to the expanded BON estimate of tags. To complete the Bayesian analysis, we specified vague priors so the posterior distribution would be dominated by the data. We used the following vague priors for the weights and proportions

$$mu_j \sim Normal(0, 0.001) \quad (12)$$

$$tau_j \sim Gamma(0.001, 0.001) \quad (13)$$

$$p\_samp_j \sim Beta(0.5, 0.5) \quad (14)$$

$$pBON\_\mathrm{det} \sim Beta(0.5, 0.5) \quad (15)$$

$$HarvRate_k \sim Beta(0.5, 0.5) \tag{16}$$

for equations 2, 2,5,8, 10, and 11.

Coho Salmon Mixture Model

Graphical examination of the daily BON coho salmon count data indicated two peaks and suggested there may be two run timing components to the BON coho salmon population. Weinkamp et al. (1996) indicated that there are early and late timed salmon populations in the Columbia River. Further inspection of the data indicated that only the early portion of the run was PIT tagged. Using our approach detailed above can only estimate the harvest rate for the early run. However, the harvest rate of the late coho salmon population is also of interest to fishery and hatchery managers to assess the contribution of these fish to fisheries. Our basic approach was to use mixture models to estimate the proportion and total number of early and late coho salmon passing BON. We estimated the harvest rate for the early group based on PIT tags as described above and estimated the catch of early coho salmon by multiplying the early coho harvest rate by the number of early coho salmon passing the dam. The number of late fish harvested was estimated by subtracting the early coho catch from the total harvest, and the late coho salmon harvest rate is a function of the number of late coho salmon caught in the fishery and passing BON.

Mixture models are a class of models used to estimate subpopulations within the overall population using a different probability distribution to represent each subpopulation (Marin and Robert 2008). Finite mixture models have a specified number of subpopulations that sum to 100%. We used the PIT tag abundance and timing of the early coho salmon along with the BON count data to help define mixture components. This is referred to as the incomplete data model because not all early coho salmon are PIT tagged. Typical fishery application for mixture models include estimating ages based on length frequency data or timing of fish runs (Macdonald and Pitcher 1979; Flynn et al. 2006, Holt 2006, Anderson and Beers 2009). For coho salmon, we observed a bimodal migration pattern at BON and noticed the tags were only detected from the first mode. Therefore, we pursued the use of mixture models in conjunction with the harvest estimates to estimate harvest for the second mode. We assumed the mixture was comprised of two normally distributed run components for:

$$Nj_t = Pj\left[\frac{1}{\sqrt{2\pi\sigma_j^2}}\exp\left(-\frac{day_t - \mu_j}{2\sigma_j^2} -\right)\right] \tag{17}$$

j=1,2 for the first two components, Nj_t is the predicted count of salmon for each of the components at time t, day_t is Julian day , μ_j is the mean date of passage, and σ_j is the standard deviation of the day of arrival. The predicted number of fish present on each day is;

$$NT_t = (N1_t + N2_t)Total \tag{18}$$

where *Total* is the count of coho salmon passing the BON fish ladder. The statistical model allows normal process error in the counts is:

$$CT_t \sim Normal(NT_t, prec) \tag{19}$$

where CT_t is the daily count at BON and *prec* (e.g., 1/variance) allows a normal error structure in the counts and the number of coho salmon in each component is:

$$coho_j = P_j Total\,. \tag{20}$$

To allow better mixing the second mean date of passage is estimated as;

$$\mu_2 = \mu_1 + K \tag{21}$$

where *K* is the difference between the mean dates of arrival. The early catch is estimated by:

$$E\_catch = e\_HarvRate * coho_1$$

where $coho_1$ is the abundance estimate for early coho and *e_HarvRate* is the harvest rate based on PIT tags for the early component calculated from equation 11. The catch for the late period is estimated by:

$$L\_catch = \sum_{i=1}^{periods} catch - E\_catch\,. \tag{22}$$

where catch is estimated as from equation 4. The harvest rate for the late component is estimated by:

$$L\_HarvRate = L\_catch / coho_2 \tag{23}$$

where the $coho_2$ is the abundance for the second component. To finish specifying the model, we placed a prior on the proportions of the first components and estimate the second component by subtraction:

$$P_1 \sim Beta(1,1) \text{ and } P_2 \sim 1 - P_1 \tag{24,25}$$

And used a vague prior for both *K* and *prec*:

$$K \sim dgamma(0.001, 0.001)\,,\ \ prec \sim dgamma(0.001, 0.001) \tag{26,27}$$

and for the standard deviation of the second component, a vague uniform prior was used:

$$\sigma_2 \sim dunif(1,15)\,. \tag{28}$$

We used PIT tagged jack and adult coho salmon to estimate the prior for the first mixture component:

$$day_j \sim Normal(\mu_1, tau_1) \tag{17}$$

where day is the date for each PIT tag coho salmon passing BON, μ_1 is the estimated mean date of passage, and tau_1 is the 1/variance for the first period.

Size Selectivity

Assigning catch to specific age classes is an important aspect of estimating fishery exploitation rates and is necessary for accurate run reconstruction. It also allows for measurement of potential age-based fishery selection (Kendall et al. 2009). It is not possible to estimate size selectivity in below-BON fisheries because we did not have an estimate of PIT tagged ocean ages upon entry to the Columbia River, but we were able to evaluate age selectivity in treaty fisheries for coho and Chinook salmon and steelhead because PIT tag ages were available at BON. To do this we developed methods to compare the age distribution in the catch relative to the run as it passed BON prior to being subjected to the fishery. Our assignment of ages was based on PIT tags which were implanted in fish as juveniles, and later interrogated at BON, and in fisheries catch. PIT tags are implanted in juvenile salmonids at a variety of sizes and ages and specific freshwater ages are not always available. However, adult size in salmonids is predominantly explained by the amount of time spent at sea (e.g., Quinn 2005), which is a variable we can measure with PIT tags based on release dates and return dates. Since age selectivity principally operates through selectivity on different ocean age classes of differing sizes, we compared the ocean age of catch relative to the run as a whole.

We developed a set of criteria to assign steelhead, and coho and Chinook salmon to ocean ages based on the time of year at which they were tagged as juveniles, the duration between tagging and return to BON, and in the case of steelhead, their juvenile size at tagging. In order to confidently assign ocean ages to steelhead, we excluded all fish tagged as adults (during a previous spawning run), as well as all fish tagged during summer/fall as parr. After excluding these fish a length frequency histogram (not shown) revealed two clear and distinct modes, representing parr and smolts with a distinguishing length of ~100 mm. We further censored our dataset to remove steelhead <100 mm. We then used Fisher's exact tests to compare the relative proportion of age classes among the catch and overall run for each species.

Model Selection and Validation

We used Deviance Information Criterion (DIC) (Spiegelhalter et al. 2002) to compare GLMM models and select models which best fit the data. Information Criteria are often used for model selection but their application is complicated for mixed models, where determining the number of parameters and the number of observations is debatable (Burnham and Anderson 2002, Carlin et al. 2009). For Bayesian model selection Spiegelhalter et al. (2002) introduced Deviance Information Criteria (DIC) as an extension of Akaike Information Criteria (AIC) when MCMC methods are used as a method to estimate the number of effective parameters for fixed and random effects. Since our GLMM model included fixed and random effects we used DIC for model selection. Spiegelhalter et al. (2002) suggested that models ΔDIC of less than 2 have considerable support, models with ΔDIC having 3-7 have less support, and model > 10 have negligible support.

In our harvest study, tag recovery data was sparse and formal model selection techniques may not be very informative. Therefore, model development relied more on our knowledge of salmon biology and harvest rather than formal model selection (Mäntyniemi and Romakkaniemi 2002). Since formal model selection and validation with Bayesian mixture models is difficult, in addition to graphical inspection we developed an *ad hoc* model validation approach for the coho run timing mixture model, and did not use formal model selection. Our *ad hoc* approach involved comparing the proportions of early and late timed coho jacks and adults at BON based on dam counts, with the estimated number of early- and late-timed hatchery coho salmon smolts released above-BON in corresponding years. These proportions should be similar if the smolt to adult return (SAR) rate to BON is similar for the two groups. Since the proportion of natural origin coho above BON is assumed to be relatively small and hatchery coho releases are consistent among years, comparing the proportions of early- and late-timed hatchery smolts and early and late-timed adults should be equivalent.

To make this comparison, we estimated the total abundance of 2009 hatchery coho smolt releases above BON (http://www.cbr.washington.edu/dart/hatch.html) belonging to various early- and late-timed stocks. However, in order to compare the early and late proportions with adult timing proportions, it was necessary to adjust releases from each basin for their estimated downstream survival to BON, which was likely to have varied considerably with migration distance. We multiplied release numbers by estimated in-river reach survivals of spring Chinook salmon, which we believed to be the most suitable surrogate for coho survival, in the absence of empirical coho data. We used survival estimates based on hatchery releases from Dworshak Hatchery on the Clearwater River in Idaho to estimate survival of early coho salmon smolts above BON (http://www.fpc.org/survival/css_inriverannualsmoltsurvival_query_2011v2.html). For Snake River and Upper Columbia populations, which originated above 8 and 9 dams, we assumed the same reach survivals as the Dworshak smolts. For Umatilla and Yakima releases, we used the McNary Dam to BON survival of Dworshak smolts within that reach, and for the late timed Klickitat population we assumed a 90% survival to BON. New release totals were tabulated and the proportion of early- and late-timed releases was compared to our mixture model-estimated proportions of early- and late-timed adults.

Results

Our harvest models were estimated in WinBUGS with two MCMC chains. After the burn-in 200000 iteration, we sampled another 400,000 iterations, used a thin rate of 80 to reduce autocorrelation. We saved 10,000 independent samples of the posterior distribution for estimates. The MCMC output was monitored for convergence and yielded BGR value of less than 1.1. The MC % error was less than the recommended 5% needed to obtain precise estimate for most parameters (Lunn et al. 2012).

Detection study

This section provides a summary of the adult PIT tag study; for full results see Rawding et al. (2013). A total of 198 fall Chinook salmon, 197 coho salmon, and 137 summer steelhead hatchery adults were tagged with PIT tags and released into hatchery raceways in the fall of 2010. The salmon were held for 7 days after tagging and the steelhead were held for 32 days after which they were sacrificed for sampling. A total of 518 individual salmon and steelhead with PIT tags were sampled multiple times in 54 trials with DF and AF scanners. After removing nine questionable trials due to equipment malfunction or samplers not implementing protocols a total of 7,398 PIT tags (99.6%) were detected out of 7,423 PIT tags samples.

Detection rates were analyzed using a general linear mixed model (GLMM) based on equation 1, which is the full model. The best model based on DIC was the intercept, trial, species, and type (model 1) with a value of 71.5, followed by the intercept, trial and species (model 2) with a value of 72.7, and the intercept only (model 3) with a value of 78.6. The first two models have ΔDIC of less than 2, which indicates considerable support for each of the models, while the third model had a ΔDIC of 7, which indicates less support. These results support similar detection probabilities using all models (Table 1) with the DF detecting PIT tags at a slightly higher detection rate than the AF, and coho salmon and steelhead having higher PIT tag detection rates than Chinook salmon. For the harvest estimation in the commercial and treaty fisheries, we used model 1 and estimated the detection probability by species with the DF scanner, which was the only scanner type used in sampling these fisheries.

Table 1. Estimates of detection probability by species and scanner type from the two different PIT tag detection models (AF=Allflex, DF=Destron Fearing).

Species	Scanner	detection rate	detection rate (sd)	2.50%	median	97.50%
Chinook	AF	98.79%	0.42%	97.85%	98.84%	99.47%
Chinook	DF	99.52%	0.22%	98.99%	99.55%	99.85%
coho	AF	99.74%	0.10%	99.50%	99.75%	99.90%
coho	DF	99.89%	0.07%	99.71%	99.90%	99.98%
steelhead	AF	99.80%	0.10%	99.56%	99.82%	99.95%
steelhead	DF	99.92%	0.06%	99.75%	99.93%	99.99%

Timing Analysis

Graphical analysis of the run timing of (hatchery and wild) PIT tagged fall Chinook salmon, coho salmon, and summer steelhead at BON supported grouping of individuals for harvest analyses at hierarchical spatial scales developed around ESUs and DPSs (Figures 3-5). Chinook salmon from six “ESU/DPS” reporting groups were sampled in fall-season fisheries in 2010. Timing of all “ESU/DPS” groups and their contributing “Rivers” groups was very similar, with the vast majority of fish passing BON during a narrow period between the end of August and the middle of September. Lower Columbia ‘Tule’ fall Chinook (LC Tules) appeared to be slightly earlier than other fall Chinook salmon stocks, respectively (Figure 3).

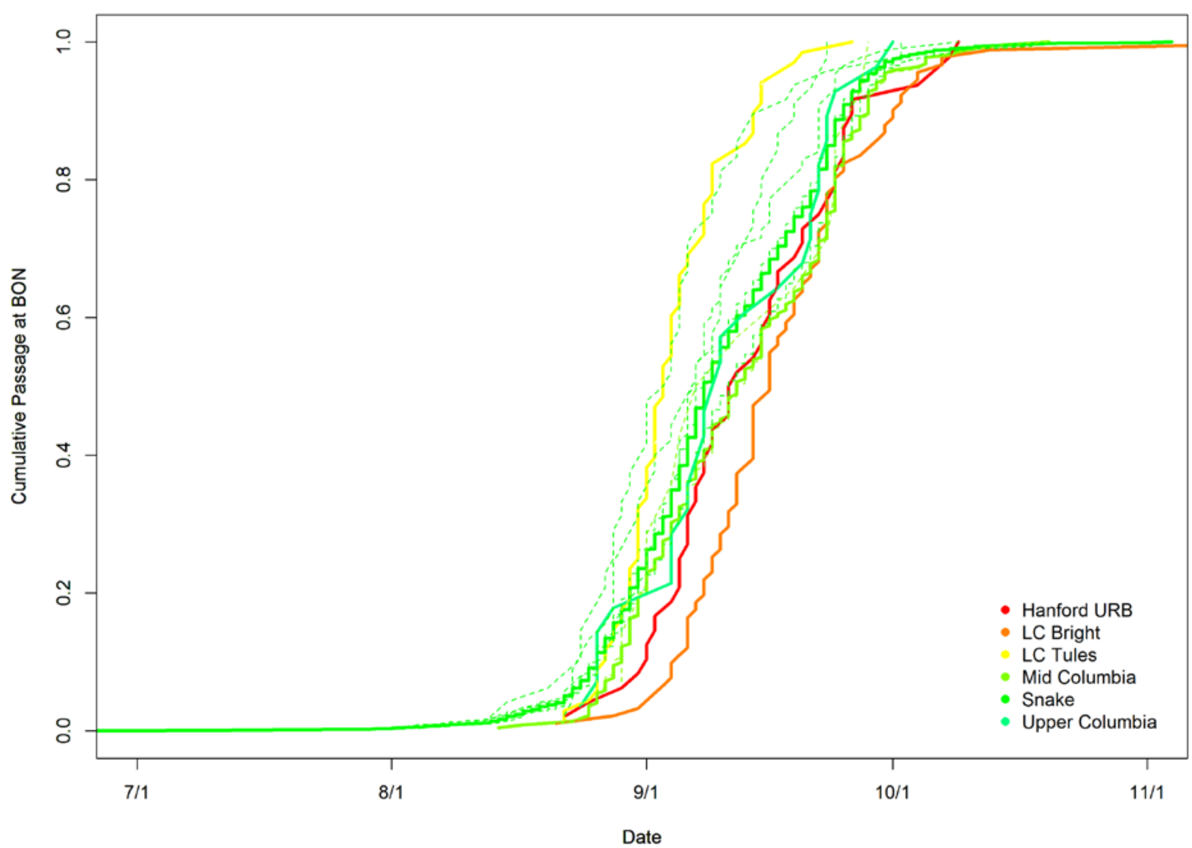

Figure 3 Cumulative passage of PIT tagged fall Chinook salmon at BON that belonged to release groups detected in sampled fall 2010 fisheries and other release groups belonging to the same ESUs. Continuous lines are timing groups by ESU whereas dotted lines are timing of groups by rivers contributing to ESUs of the same corresponding color.

Coho salmon were sampled from three “ESU/DPS” reporting groups in 2010 and all of these ESUs and their contributing rivers exhibited similar run timing at BON, passing BON primarily between the last week in August and the middle of September, similar to Chinook (Figure 4).

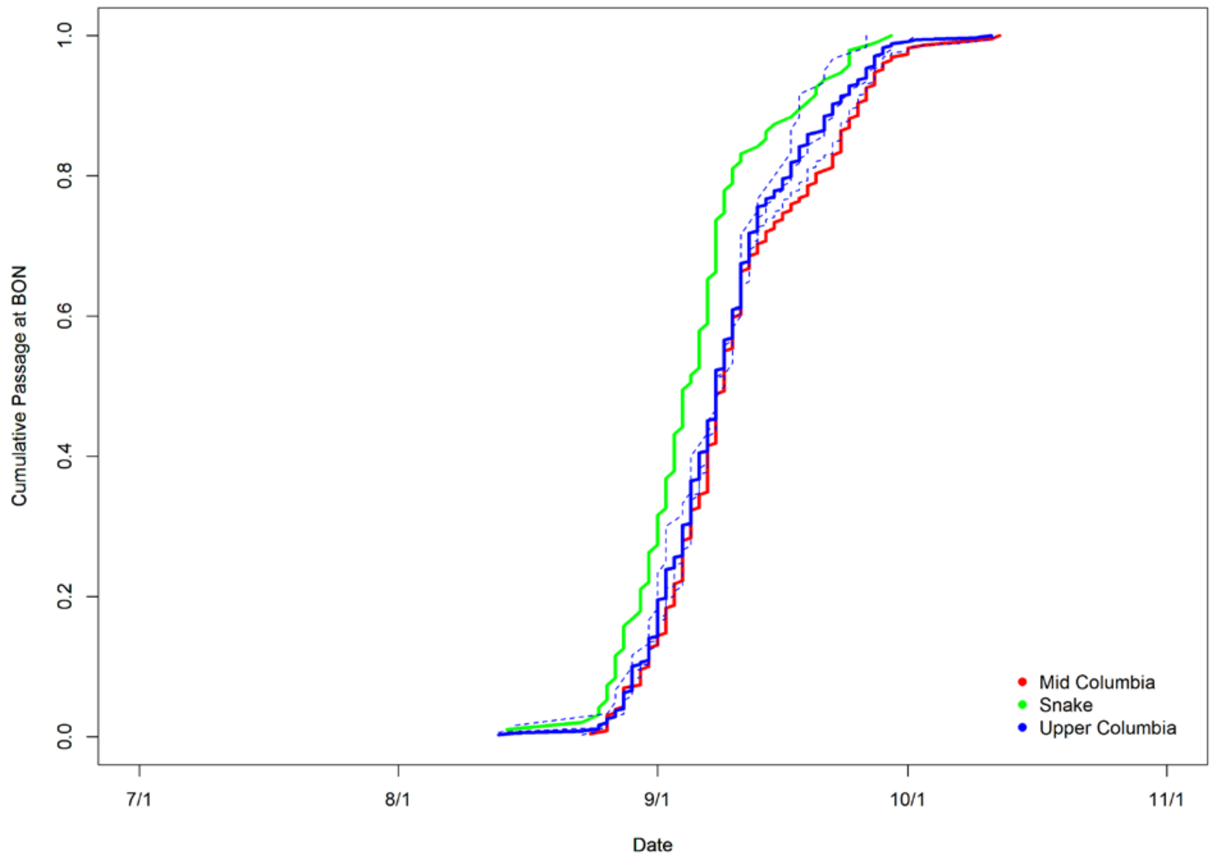

Figure 4 Cumulative passage of PIT tagged coho salmon at BON that belonged to release groups detected in sampled fall 2010 fisheries and other release groups belonging to the same 'ESU' grouping. Continuous lines are timing groups by 'ESU' whereas dotted lines are timing of groups by rivers contributing to 'ESUs' of the same corresponding color.

Summer steelhead were sampled from five "ESU/DPS" reporting groups in 2010. Unlike Chinook and coho salmon, run timing varied considerably among these groups (Figure 5a): Lower Columbia summer steelhead displayed a more protracted run which began much earlier than other groups (Figure 5b); Middle Columbia (MCR), Upper Columbia (UCR), and Snake River A-run (SNA) all had similar run timing with most passage occurring in July and August (Figure 5c,d&f) ; Snake River B-run steelhead were considerably later to arrive at BON, with most fish passing after August (Figure 5e). The individual "Rivers" groups comprising these "ESU/DPS" groups exhibited similar run timing at BON, suggesting that using these "ESU/DPS" groups for pooling harvest rates was appropriate.

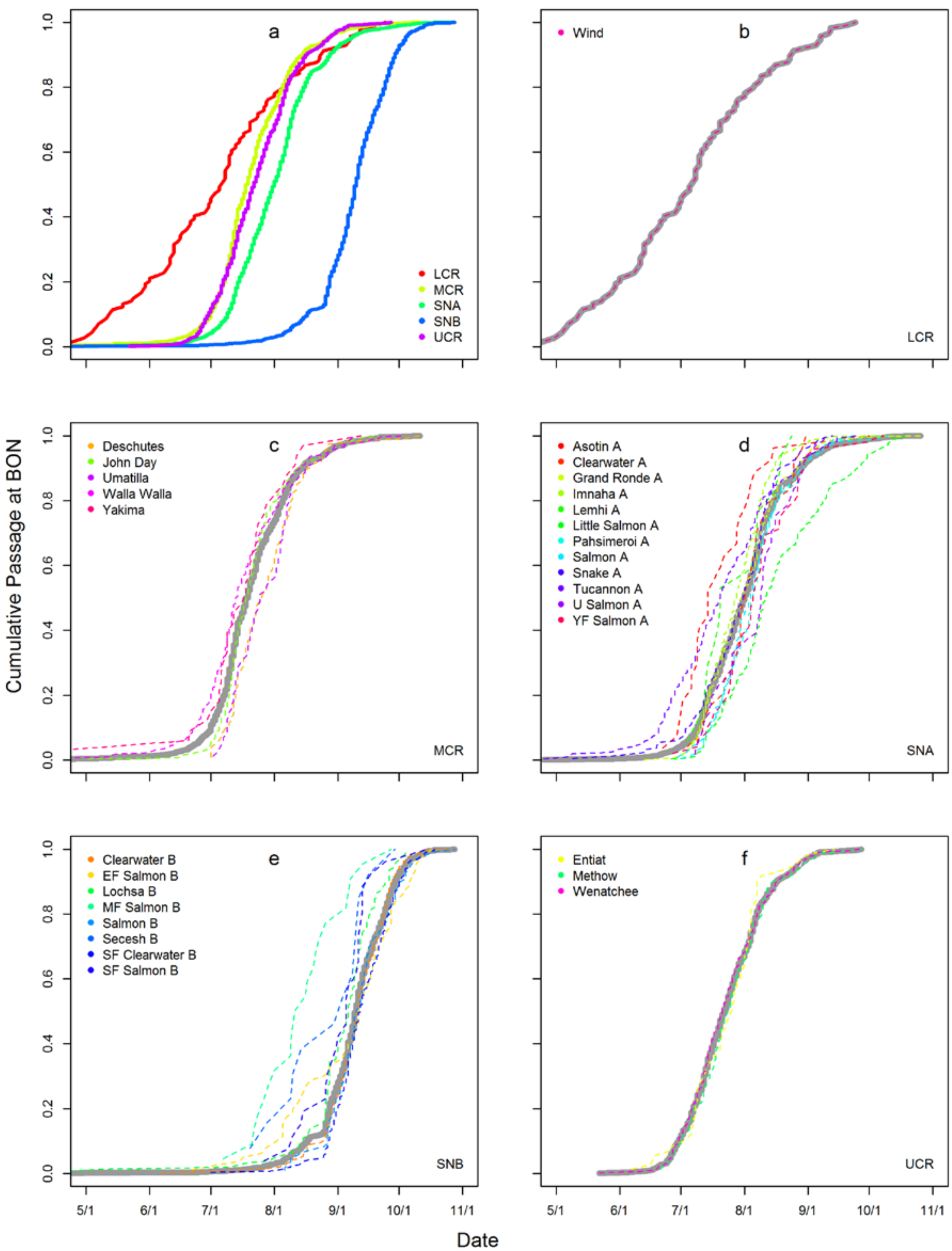

Figure 5 Cumulative passage timing of PIT tagged summer steelhead at BON that belonged to release groups detected in sampled fall 2010 fisheries and other release groups belonging to the same "ESU/DPS" groups. Timing is shown by for all "ESU/DPS" reporting groups (a), as well as comparing each "ESU/DPS" group with its contributing "Rivers" groups; Lower Columbia (b), Middle Columbia (c), Snake A-run (d), Snake B-run (e), and Upper Columbia (f). In panels b-f continuous gray lines depict the timing of "ESU/DPS" groups, whereas dotted lines show the timing of "Rivers" groups contributing to those "ESU/DPS" groups. Only data series with >10 tagged adults returning to BON were plotted.

PIT Tag Recoveries and Harvest Reporting Groups

We were able to assign a total of 5,851 jack and adult fall Chinook salmon detected at BON dam in 2010 from various release sites within the Columbia Basin to harvest reporting groups. PIT tagged fall Chinook salmon returning to BON in 2010 were assigned to 6 "ESU/DPS" harvest reporting groups which generally followed formal ESU boundaries with some modifications as previously described. These "ESU/DPS" harvest groups consisted of 34 unique combinations of release sites and rear types (Figure 6; Appendix Table 1).

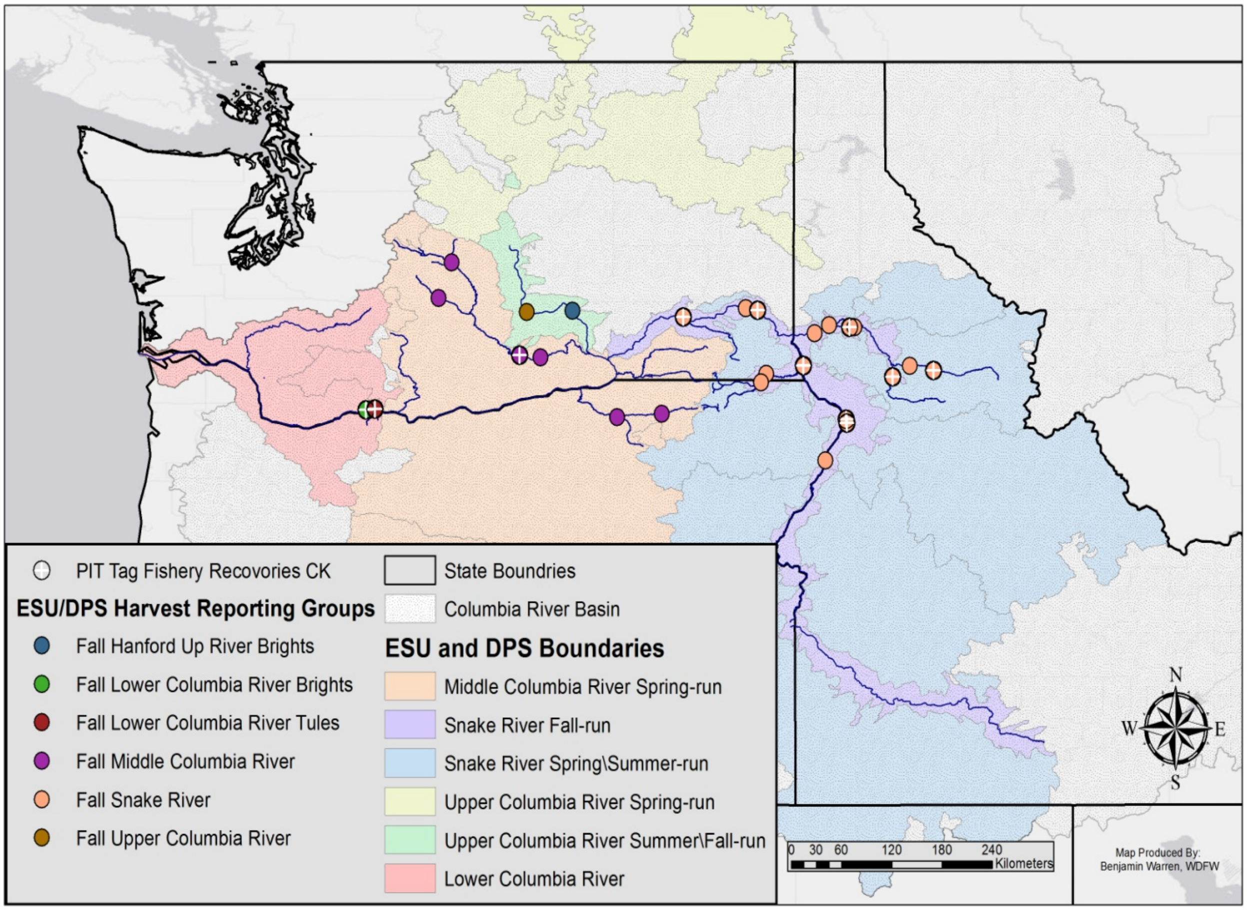

Figure 6. Map of the Columbia River Basin showing NOAA ESU boundaries for Chinook salmon, as well as the locations where PIT tagged jack and adult Chinook salmon that were used in generating harvest estimates in 2010 were released as juveniles. Colored circles show juvenile release locations for all returning adult fish (harvest + escapement) that were used to estimate harvest rates in 2010, colored according to the "ESU/DPS" harvest reporting groups they were assigned to, while white crosshatches represent the subset of those release sites for which harvested PIT tagged adults were sampled in the fishery.

We were able to assign a total of 675 jack and adult coho salmon detected at BON dam in 2010 from various release sites within the Columbia Basin to harvest reporting groups. PIT tagged coho salmon returning to BON in 2010 were assigned to 3 "ESU/DPS" harvest reporting groups which generally followed NOAA ESU boundaries of steelhead and Chinook since coho salmon populations above the White Salmon River (WA) and Hood River (OR) do not have formally defined ESUs. These "ESU/DPS" harvest groups consisted of 21 unique combinations of release sites and rear types (Figure 7 Appendix Table 2).

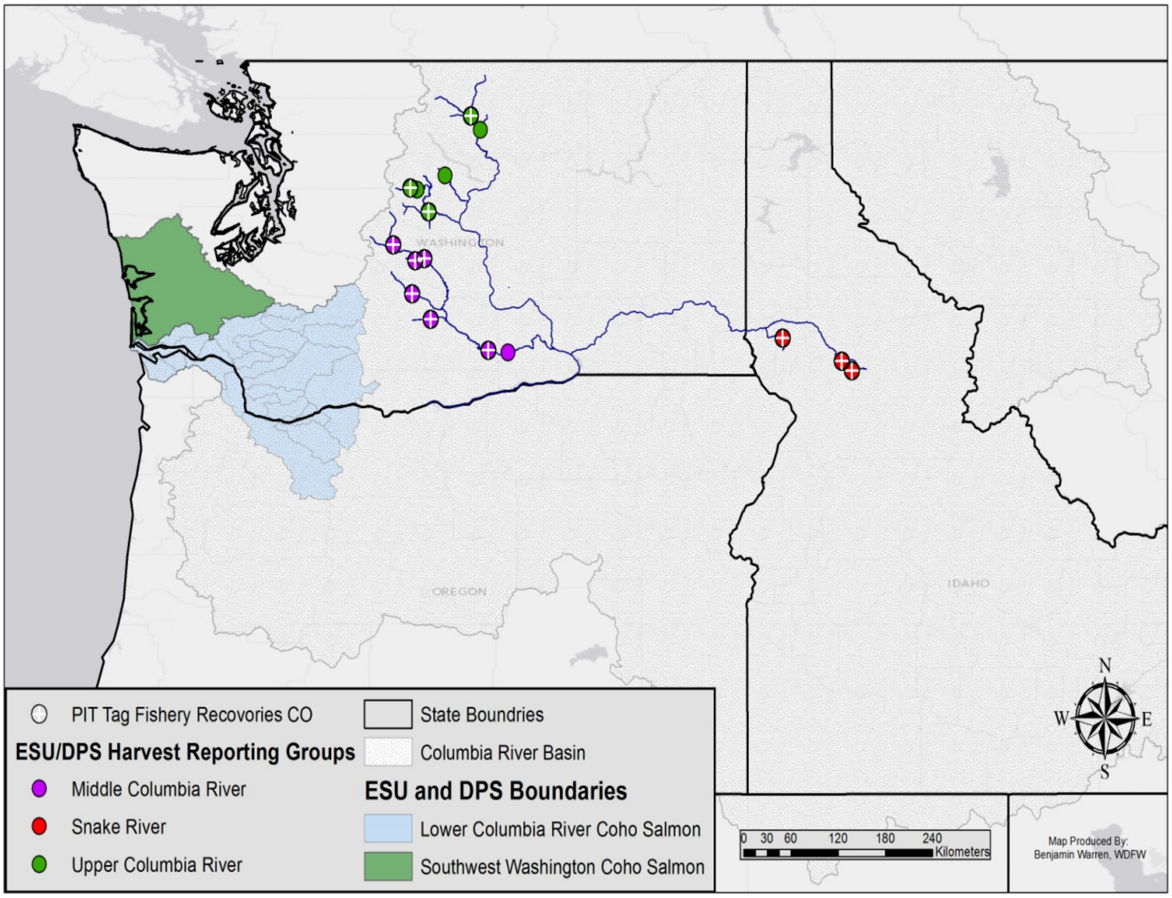

Figure 7. Map of the Columbia River Basin showing NOAA ESU boundaries for coho salmon, as well as the locations where PIT tagged jack and adult coho salmon that were used in generating harvest estimates in 2010 were released as juveniles. Colored circles show juvenile release locations for all returning adult fish (harvest + escapement) that were used to estimate harvest rates in 2010, colored according to the ESU harvest reporting group they were assigned to, while white crosshatches represent the subset of those release sites for which harvested PIT tagged adults were sampled in the fishery.

We were able to assign a total of 8,320 steelhead detected at BON dam in 2010 between April 1st and November 1st from various release sites within the Columbia Basin to harvest reporting groups. PIT tagged steelhead returning to BON in 2010 were assigned to 5 "ESU/DPS" harvest reporting groups which generally followed formal NOAA DPS boundaries for steelhead except for in the Snake River where A and B run steelhead were separated for harvest rate estimation. These "ESU/DPS" harvest groups consisted of 166 unique combinations of release sites and rear types (Figure 8 Appendix Table 3).

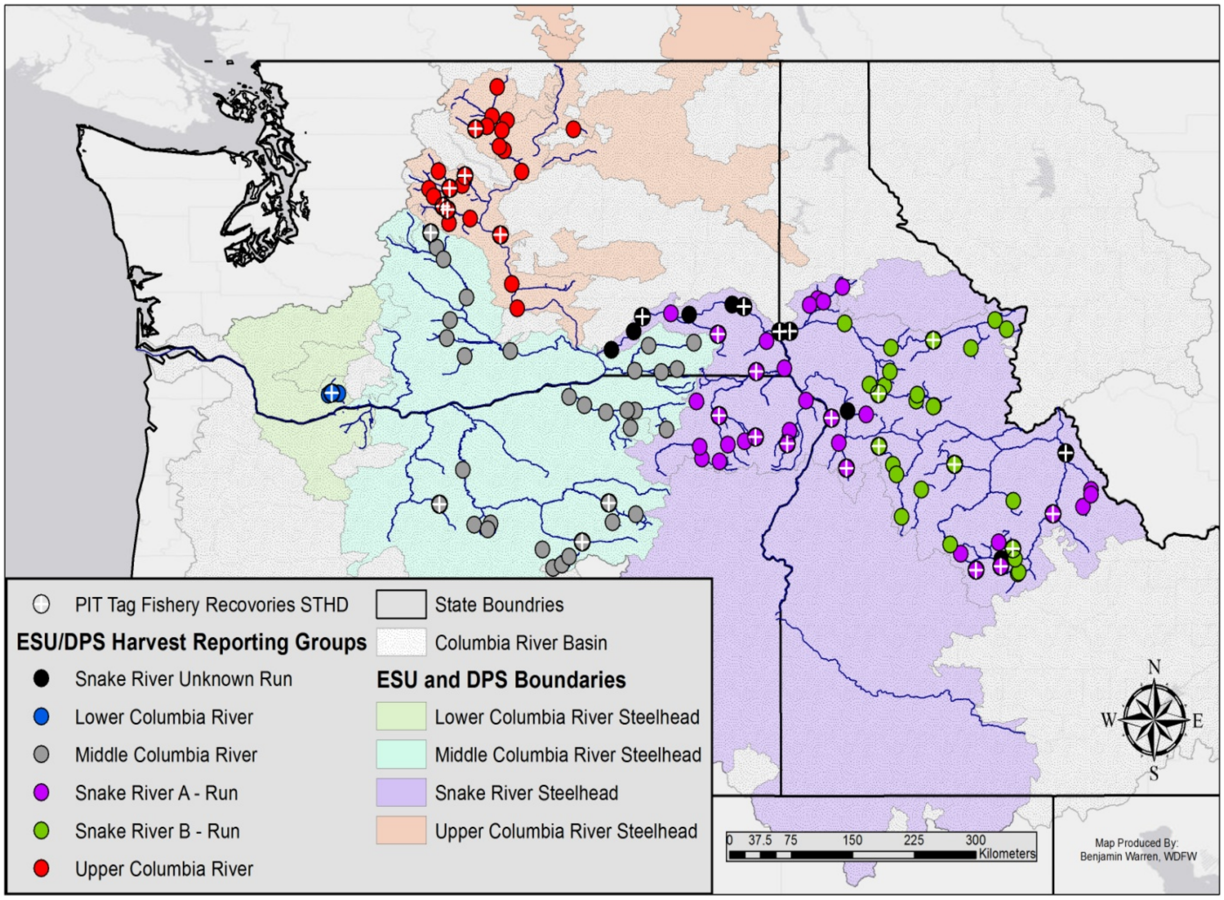

Figure 8. Map of the Columbia River Basin showing NOAA DPS boundaries for steelhead, as well as the locations where PIT tagged steelhead that were used in generating harvest estimates in 2010 were released as juveniles. Colored circles show juvenile release locations for all returning adult fish (harvest + escapement) that were used to estimate harvest rates in 2010, colored according to the "ESU/DPS" harvest reporting group they were assigned to, while white crosshatches represent the subset of those release sites for which harvested PIT tagged adults were sampled in the fishery. Snake River unknown run fish belonged to both Snake A and B run DPS reporting groups.

Fisheries Sampling and Harvest Rates

Zone 1-5

We sampled five of the six weeks comprising the 2010 fall fishery for Chinook salmon for CWT, PIT tags, and tissue samples for genetic mixed stock fishery analysis. Over 22,000 fall Chinook were caught during these five weeks sampled. Mean weights, estimated catch, and the proportion of the catch sampled for PIT tags are found in Table 2. The sample rate generally exceeded the 20% guideline (Nandor et al. 2011) for fishery sampling (Table 2). Few salmon are PIT tagged below BON, and we were unable to estimate the uncaught proportion of these release groups. Consequently lower river harvest rates were only estimated for PIT tagged groups originating from releases above BON. Harvest rates based on PIT tags were calculated for BON pool Tules and a few Snake River Fall Chinook groups because these were the only PIT tagged fall Chinook stocks for which tags were recovered. The estimated harvest rate was 14% for the Tule fall Chinook hatchery stock released from Spring Creek Hatchery. The estimated harvest rates for

Snake groups were between 2% and 3% for these populations and the “ESU/DPS” group (Table 3).

Table 2. Weights, catch, and PIT tag sample rate of fall Chinook salmon during the August and late fall commercial salmon fishery in Zones 1-5, 2010. Periods are fishery weeks, weights are mean weights in pounds, catch is the total catch of fish in a period, and p_samp is the proportion of the catch sampled for PIT tags.

period	weight	weight(sd)	catch	catch(se)	p_samp	p_samp(sd)
wk 35	20.54	5.69	12732	172.84	18.49%	0.43%
wk 39	16.01	5.82	5453	104.28	25.58%	0.78%
wk 40	16.42	5.94	2033	40.03	23.13%	1.04%
wk 41	12.83	6.21	1667	36.01	27.45%	1.25%
wk 42	17.02	7.97	485	19.49	20.14%	1.98%
total			22370			

Table 3. Sampled and expanded PIT tags, and harvest rates of fall Chinook salmon during the August and late fall commercial salmon fishery in Zones 1-5, 2010. Tags and ex_tags are the unexpanded and expanded number of PIT tags sampled in fisheries, respectively. Harvest rates for “Rivers” groups are lower case, whereas harvest rates for “ESU/DPS” groups are capitalized. H denotes to hatchery-origin groups.

ESU	Group	tags	ex_tags	harv. rate	harv rate (sd)	2.50%	median	97.50%
Separate Hatchery and Wild Rivers Groups								
LCR	Spring Cr._H	2	10.93	3.46%	0.55%	3.08%	3.42%	4.61%
SNK	Clearwater_H	7	38.05	14.23%	3.87%	11.53%	13.95%	22.68%
SNK	Snake_H	10	51.35	2.14%	0.30%	1.93%	2.13%	2.76%
Separate Hatchery and Wild ESU/DPS Groups								
LCR	LCR TULE_H	2	10.93	0.14	0.04	0.12	0.14	0.23
SNK	SNAKE_ESU_H	21	108.14	0.02	0.00	0.02	0.02	0.02

Coho salmon were also sampled during the Zones 1-5 fishery. Mean weights, total catch and the PIT tag sample rate are reported in Table 4. Over 18,500 fish were caught in this five week fishery. A total of two PIT tags from Yakima River hatchery, and a single Yakima wild coho salmon adult were recovered in this fishery. The harvest rates for Yakima River coho salmon was ~ 2% (Table 5).

Table 4. Coho data: weights, catch, and PIT tag sample rate during the August and late fall commercial salmon fishery in Zones 1-5, 2010. Periods are statistical weeks, weights are mean weights in pounds, catch is the total catch of fish in a period, and p_samp is the proportion the catch sampled for PIT tags.

period	weight	weight(se)	catch	catch(se)	p_samp	p_samp(sd)
wk 35	8.566	2.343	327	8.09	24.40%	2.44%
wk 39	10.78	2.539	679	10.67	30.86%	1.79%
wk 40	12.32	2.624	4468	42.79	26.05%	0.70%
wk 41	11.17	2.733	11410	101.7	40.65%	0.59%
wk 42	10.96	2.576	1670	27.1	20.51%	1.04%
total			18550			

Table 5. Coho data: sampled and expanded PIT tags, and harvest rates during the August and late fall commercial salmon fishery in Zones 1-5, 2010. Tags and ex_tags are the unexpanded and expanded number of PIT tags sampled in fisheries, respectively. H, W, and HW refer to hatchery, wild and combined hatchery and wild groups. Only "River" and "Large Aggregates" harvest estimates were computed for Zones 1-5 coho salmon.

ESU	Group	tags	ex_tags	harv. rate	harv rate (sd)	2.50%	median	97.50%
Separate Hatchery and Wild Rivers Groups								
NA	Yakima_W	1	4.13	1.26%	0.60%	0.37%	1.17%	2.69%
NA	Yakima_H	1	3.10	29.54%	12.63%	8.44%	28.48%	57.33%
Combined Hatchery and Wild Rivers Groups								
NA	Yakima_HW	2	7.22	2.06%	0.75%	0.87%	1.97%	3.76%

Zone 6

The 2010 treaty fall Zone 6 fishery was sampled simultaneously for CWT, PIT tags, and genetic samples. This fishery catches ~ 100,000 more Chinook salmon than the lower river commercial fishery (Table 2 and 6). Sample rates exceeded the 20% guideline in 3 of 7 weeks and were lower in the remaining four weeks. The probability of detecting an adult Chinook with a PIT tag passing BON was 99.45% with 95%CI (99.25% - 99.61%). The median harvest rate for hatchery Chinook released from Spring Creek Hatchery was 64%, 19% for hatchery Chinook released from the SF Clearwater River, and less than 10% for all other releases mostly from the Snake River (Table 7). Pooling PIT tags into homogenous groups at the "ESU/DPS" level resulted in a LCR harvest rate of 64% and a Snake River harvest rate of 9%.

Table 6. Weights, catch, and PIT tag sample rate during the fall treaty fishery in Zone 6, 2010. Periods are fishery weeks, weights are mean weights in pounds, catch is the total catch of Chinook in a period, and p_samp is the proportion the catch sampled for PIT tags.

period	weight	weight(se)	catch	catch(se)	p_samp	p_samp(sd)
wk 31-35	19.86	6.38	7047	162.41	15.86%	0.57%
wk 36	17.48	5.65	37713	460.12	11.02%	0.21%
wk 37	18.51	5.56	46220	664.73	13.45%	0.25%
wk 38	17.74	5.19	34620	466.36	24.74%	0.41%
wk 40	15.45	4.88	11156	219.09	18.25%	0.51%
wk 42	14.57	6.18	2281	76.35	24.07%	1.21%
wk 43+	14.20	6.10	1318	44.36	43.91%	2.00%
total			140356			

+ Fish in this period include platform hook and line subsistence fish caught late in the season and after week 43.

Table 7. Sampled and expanded PIT tags, and harvest rates during the fall treaty fishery in Zone 6, 2010. Tags and ex_tags are the unexpanded and expanded number of Chinook salmon PIT tags sampled in fisheries, respectively. Harvest rates for "Rivers" groups are lower case, whereas harvest rates for "ESU/DPS" groups are capitalized. H refers to hatchery groups, HW to combined hatchery and wild groups, and U to unknown origin groups.

ESU	Group	tags	ex_tags	harv. rate	harv rate (sd)	2.50%	median	97.50%
Separate Hatchery and Wild Rivers Groups								
LCR	Spring Cr._H	7	43.76	64.18%	5.84%	52.50%	64.26%	75.24%
NA	L White Sal._H	1	2.02	2.75%	1.68%	0.47%	2.43%	6.81%
MCR	Yakima_H	1	4.00	2.17%	1.00%	0.67%	2.03%	4.51%
SNK	Snake_H	33	215.90	9.18%	0.61%	8.03%	9.17%	10.41%
SNK	Clearwater_H	8	56.28	5.32%	0.68%	4.05%	5.30%	6.74%
SNK	SF Clearwater_H	1	9.02	19.33%	5.59%	9.78%	18.89%	31.62%
SNK	Selway_H	1	4.00	9.85%	4.38%	3.08%	9.26%	19.83%
Separate Hatchery and Wild ESU/DPS Groups								
NA	LCR_URBRIGHTS_H	1	2.02	2.77%	1.71%	0.46%	2.44%	6.92%
LCR	LCR_TULE_H	7	43.76	64.22%	5.80%	52.54%	64.36%	75.17%
MCR	MCR_H	1	4.00	2.03%	0.94%	0.62%	1.90%	4.21%
SNK	SNAKE_H	69	457.90	8.57%	0.40%	7.82%	8.56%	9.36%
SNK	SNAKE_U	1	4.00	10.37%	4.55%	3.24%	9.83%	20.63%
SNK	SNAKE_HW	70	463.20	8.60%	0.39%	7.85%	8.59%	9.38%

The adult steelhead detection probability at BON was 99.08% with a 95% CI (98.84% - 99.28%). Over 25,000 steelhead were caught in this fishery (Table 8). Weights increased through the fishery period as more large B-run fish became available. Sample rates were lower for steelhead than other species, and the sample rate was only 5% in week 40. These lower sample rates resulted in lower precision of our steelhead estimates relative to other species. Harvest rates were variable due to lower individual recovery rates and ranged from 2% to 54% for steelhead passing BON after July 1. Pooling release groups into DPSs decreased the variability of the harvest rate estimates (Table 10). There was only one wild Snake River A-run steelhead PIT tag recovered as an adult in the fishery, resulting in an estimated harvest rate of 1.7%, but with considerable uncertainty. For other ESUs the harvest rate ranged from 5% for the MCR to 25% for the UCR. Hatchery harvest rates ranged from 5% to 7% for the UCR and the SNK group A and B steelhead ESU groups (Table 10). Aggregation of both hatchery and wild steelhead by DPS yielded estimates of 7%, 3%, 11%, 5% and 8% for LCR, MCR, UCR, SNKA and SNKB management groups, respectively.

Over 11,500 coho salmon were harvested in the fall Zone 6 fishery. The mean weight ranged from 8 to 10 lbs. among weeks, and sample rates were generally above the 20% guideline (Table 11). The adult coho salmon detection probability at BON was 98.73% with 95%CI (97.38% - 99.55%). Individual hatchery harvest rates ranged from 14 to 24% and the aggregate harvest rate for this group was 17%.

Table 8. Weights, catch, and PIT tag sample rate during the fall treaty steelhead fishery in Zone 6, 2010. Periods are fishery weeks, weights are mean weights in pounds, catch is the total catch of fish in a period, and p_samp is the proportion the catch sampled for PIT tags.

period	weight	weight(se)	catch	catch(se)	p_samp	p_samp(sd)
wk 31-35	9.15	2.78	6594	58.41	16.00%	0.46%
wk 36	9.23	3.57	5023	101.91	13.66%	0.56%
wk 37	10.11	3.98	4940	208.73	10.42%	0.62%
wk 38	11.50	3.89	4961	98.31	22.61%	0.73%
wk 40	10.36	4.17	2712	59.87	5.35%	0.45%
wk 42	11.06	4.16	1530	36.56	19.57%	1.10%
wk 43+	11.38	3.32	816	7.85	12.38%	1.17%
total			26576			

[+] Fish in this period include subsistence fish late in the season and after week 43.

Table 9. Sampled and expanded PIT tags, and harvest rates by "Rivers" groups, during the fall treaty fishery in Zone 6 in 2010. Tags and ex_tags are the unexpanded and expanded number of PIT tags sampled in fisheries, respectively. Estimates are separated for hatchery (H) and wild (W) steelhead in each group. For Snake River populations, A, B and AB refer to A run, B run and combined A and B run estimates, respectively.

DPS	Group	tags	ex_tags	harv. rate	harv rate (sd)	2.50%	median	97.50%
Separate Hatchery and Wild Rivers Groups								
LCR	Wind_W	1	4.28	7.35%	3.29%	2.31%	6.88%	14.87%
MCR	Deschutes_W	1	5.09	4.88%	2.04%	1.70%	4.61%	9.63%
MCR	John Day_W	2	11.36	4.39%	1.25%	2.28%	4.29%	7.18%
MCR	Yakima_W	1	6.09	22.03%	7.49%	9.27%	21.41%	38.42%
UCR	Wenatche_W	4	28.42	40.61%	5.93%	29.29%	40.57%	52.41%
UCR	Wenatchee_H	4	26.12	5.58%	1.06%	3.68%	5.50%	7.80%
UCR	Entiat_W	2	19.28	29.09%	5.75%	18.39%	28.86%	40.96%
UCR	Methow_H	1	6.09	12.23%	4.46%	4.94%	11.69%	22.14%
SNK	Tucannon_AW	1	5.09	14.29%	5.58%	5.17%	13.66%	26.88%
SNK	Tucannon_AH	4	29.49	15.03%	2.56%	10.34%	14.94%	20.34%
SNK	Grand Ronde_AH	4	29.42	4.86%	0.90%	3.25%	4.82%	6.76%
SNK	Imnaha_AH	2	9.56	2.30%	0.72%	1.08%	2.23%	3.93%
SNK	Snake_AH	2	11.80	5.95%	1.64%	3.12%	5.82%	9.50%
SNK	Salmon_AH	1	9.63	6.74%	2.10%	3.27%	6.49%	11.40%
SNK	U Salmon_AH	1	6.09	10.65%	3.97%	4.24%	10.18%	19.57%
SNK	Little Salmon_AH	2	9.56	3.21%	1.00%	1.57%	3.12%	5.43%
SNK	Pahsimeroi_AH	2	14.68	5.83%	1.50%	3.26%	5.71%	9.12%
SNK	Salmon_ABH	2	11.80	6.05%	1.67%	3.20%	5.92%	9.62%
SNK	Salmon_BH	1	18.82	17.55%	3.90%	10.53%	17.31%	25.94%
SNK	SF Salmon_BW	1	7.29	28.78%	8.79%	13.35%	28.12%	47.12%
SNK	Secesh_BW	1	6.09	47.18%	13.05%	22.53%	47.01%	72.58%
SNK	MF Salmon_BW	1	6.09	28.62%	9.36%	12.20%	28.02%	48.71%
SNK	Lochsa_BW	1	8.16	5.56%	1.90%	2.42%	5.38%	9.80%
SNK	SF Clearwater_BH	2	11.80	4.11%	1.15%	2.17%	4.01%	6.63%
SNK	Clearwater_BH	6	28.41	7.68%	1.39%	5.19%	7.59%	10.66%
SNK	Clearwater_ABW	1	9.63	53.42%	11.70%	30.50%	53.59%	75.64%

Table 10. Sampled and expanded PIT tags, and harvest rates for "ESU/DPS" groups and "Large Aggregate" groups during the fall treaty steelhead fishery in Zones 6, 2010. Tags and ex_tags are the unexpanded and expanded number of PIT tags sampled in fisheries, respectively. H, W, and HW refer to hatchery, wild and combined hatchery and wild groups. For Snake River populations, A, B and AB refer to A run, B run and combined A and B run estimates, respectively.

DPS	Group	tags	ex_tags	harv. rate	harv rate (sd)	2.50%	median	97.50%
Separate Hatchery and Wild ESU/DPS Groups								
LCR	LCR_W	1	4.28	7.34%	3.30%	2.27%	6.88%	14.95%
MCR	MCR_W	4	22.78	4.54%	0.91%	2.90%	4.48%	6.48%
UCR	UCR_W	6	47.70	25.00%	3.21%	19.05%	24.90%	31.62%
SNK	SNAKE_AW	1	5.09	1.70%	0.72%	0.59%	1.61%	3.38%
SNK	SNAKE_BW	4	28.01	12.88%	2.30%	8.67%	12.76%	17.66%
UCR	UCR_H	5	32.37	5.55%	0.95%	3.86%	5.49%	7.58%
SNK	SNAKE_AH	19	120.38	5.28%	0.48%	4.37%	5.27%	6.25%
SNK	SNAKE_BH	9	59.00	6.88%	0.88%	5.26%	6.85%	8.71%
Combined Hatchery and Wild ESU/DPS Groups								
LCR	LCR_HW	1	4.28	7.37%	3.30%	2.31%	6.92%	14.98%
MCR	MCR_HW	4	22.78	3.05%	0.63%	1.93%	3.01%	4.39%
UCR	UCR_HW	12	87.41	10.38%	1.07%	8.36%	10.35%	12.56%
SNK	SNAKE_AHW	20	125.51	4.82%	0.43%	4.01%	4.81%	5.69%
SNK	SNAKE_BHW	12	87.02	8.08%	0.85%	6.48%	8.06%	9.83%
Large Aggregate Groups								
SNK	SNAKE_ABH	39	255.06	5.76%	0.37%	5.06%	5.75%	6.52%
SNK	SNAKE_ABW	21	153.56	8.84%	0.70%	7.52%	8.82%	10.26%
SNK	SNAKE_ABHW	60	408.61	6.61%	0.35%	5.94%	6.60%	7.30%

Table 11. Weights, catch, and PIT tag sample rate during the fall treaty coho salmon fishery in Zone 6, 2010. Periods are fishery weeks, weights are mean weights in pounds, catch is the total catch of fish in a period, and p_samp is the proportion the catch sampled for PIT tags.

period	weight	weight(se)	catch	catch(se)	p_samp	p_samp(sd)
wk 31-35	8.45	3.10	131	6.12	19.40%	3.56%
wk 36	9.09	2.06	443	7.46	25.57%	2.14%
wk 37	9.59	2.17	2122	28.95	14.99%	0.81%
wk 38	9.90	2.15	3653	43.10	13.40%	0.59%
wk 40	10.18	2.46	763	12.85	33.21%	1.79%
wk 42	9.87	2.47	2975	62.27	27.89%	1.02%
wk 43+	10.45	2.53	1472	17.52	30.11%	1.23%
total			11561			

+ Fish in this period include subsistence fish late in the season and after week 43.

Table 12. Sampled and expanded PIT tags, and harvest rates for "Rivers" and "Large Aggregates" groups during the fall treaty coho salmon fishery in Zones 6, 2010. Tags and ex_tags are the unexpanded and expanded number of PIT tags sampled in fisheries, respectively. H refers to hatchery groups.

ESU	Group	tags	ex_tags	harv. rate	harv rate (sd)	2.50%	median	97.50%
Separate Hatchery and Wild Rivers Groups								
NA	Methow_H	2.00	10.46	18.04%	5.04%	9.28%	17.73%	28.97%
NA	Wenatchee_H	5.00	21.48	13.90%	2.74%	8.99%	13.80%	19.53%
NA	Yakima_H	12.00	58.30	16.29%	2.01%	12.54%	16.23%	20.41%
NA	Clearwater_H	5.00	22.89	24.00%	4.51%	15.73%	23.87%	33.27%
Large Aggregates Groups								
NA	All Above BON	26.00	112.66	16.53%	1.47%	13.73%	16.51%	19.51%

Coho Salmon Mixture Model

The mixture model suggested that the mean date of passage at BON for the early (type S) and late (type N) coho salmon components (adults and jacks) was separated by about 31 days—days of the year (DOY) 251 and 282, respectively. The width of the normal curve (SD) is greater for the second component. The abundance estimates for the early and late components were 49,000 and 79,000 fish, respectively. The estimated catches were 8,100 and 3,500 for the early and late components, respectively. These correspond to harvest rates of 17% and 4%. Graphical examination of the observed pattern of daily counts is consistent with a bimodal distribution (Figure 9).

Two validation measures suggested the mixture model adequately fit the data. Graphical inspection suggested the mixture model was consistent with the data (Figure 9). A second validation measure based on the relative contribution of early- and late-timed smolts migrating past BON found 38% were early-timed and the remainder were late-timed. The mixture compositions from our analysis suggested the early and late components comprised 38.2% (95%CI 34.0% to 42.8%) and 61.8% (57.2% to 66.0%) of the run, respectively. Comparison of 95% CIs suggested the adult and smolt proportions of early- and late-timed components were not statistically different, indicating the smolt and mixture model proportions were consistent. This would be expected if early and late smolt groups had similar smolt to adult return rates (SAR).

Table 13. Parameter estimates for the early and late components of coho salmon passing BON using a normal mixture model.

Node	Parameter	Mean	SD	2.50%	median	97.50%
mu[1]	Early Run Mean Passage (Day of Year)	250.83	0.53	37.83	38.81	39.93
mu[2]	Late Run Mean Passage (Day of Year)	281.64	0.46	68.74	69.64	70.53
sd [1]	Early Run Passage sd (days)	8.47	0.67	7.26	8.44	9.89
sd [2]	Late Run Passage sd (days)	9.36	0.51	8.38	9.37	10.39
K	Diff. Between Early and Late (days)	30.82	0.57	29.67	30.82	31.92
coho[1]	Early Run Size (fish)	48,830	2,882	43,480	48,750	54,770
coho[2]	Late Run Size (fish)	79,040	2,882	73,100	79,130	84,400
E_catch	Early Run Catch (fish)	8,079	871	6,474	8,054	9,851
L_catch	Early Run Catch (fish)	3,475	869	1,718	3,505	5,075
E_HarvRate	Early Run Harvest Rate (%)	16.54%	1.48%	13.73%	16.52%	19.54%
L_HarvRate	Early Run Harvest Rate (%)	4.38%	1.03%	2.24%	4.43%	6.23%
P[1]	Early Run Proportion of Total Run (%)	38.19%	2.25%	34.00%	38.12%	42.83%
P[2]	Late Run Proportion of Total Run (%)	61.81%	2.25%	57.17%	61.88%	66.00%
sd_CT	Total Run Passage sd (days)	429.80	26.96	380.40	428.40	486.70

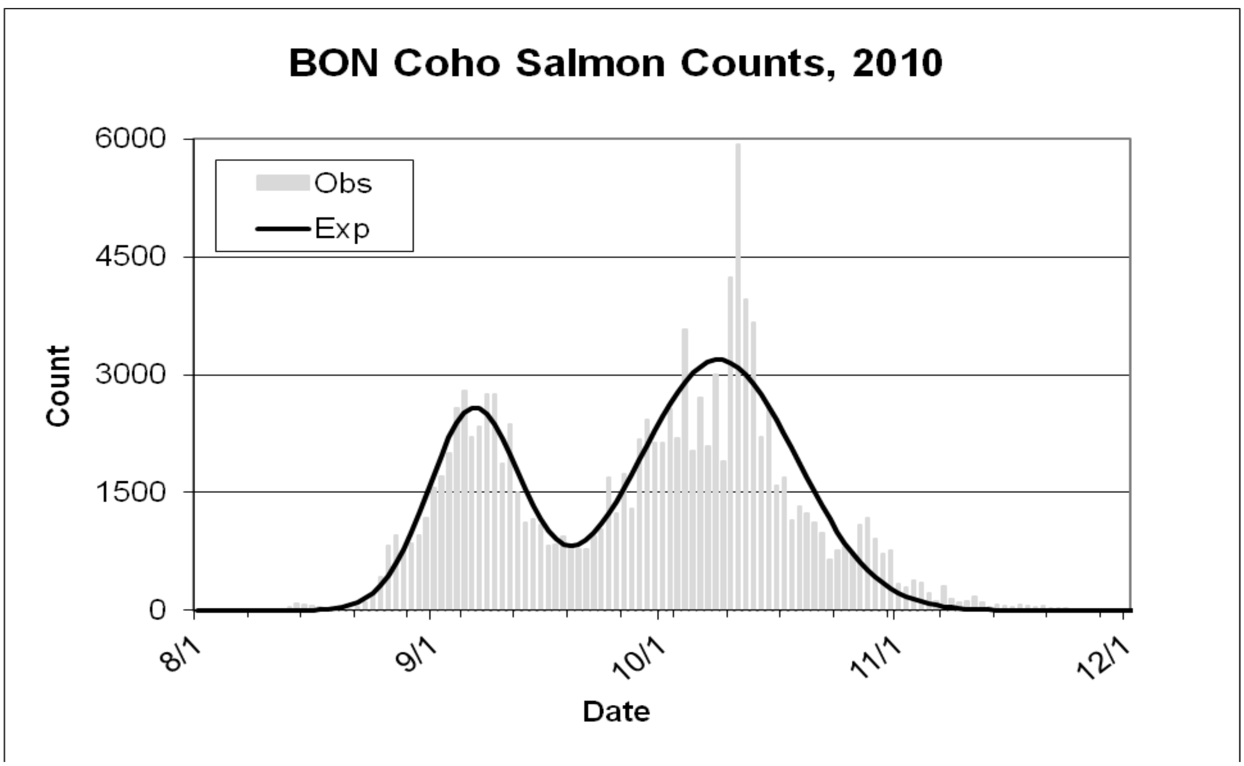

Figure 9. Fit of the mixture model to BON coho salmon counts. These results suggest the early proportion of the jack and adult return was approximately 38% of the return in 2010.

Comparison of Selected Harvest Estimates

We only compared harvest rates based on PIT tags with those developed using other methods for populations above BON since the entire fall fishery was not sampled below BON in 2010. For tule fall Chinook salmon, a total of 68 PIT tagged fish that were released from Spring Creek National Fish Hatchery (SCNFH) were detected at BON. We detected 7 fish in the fishery, which based on our weekly sampling rates expanded to 44 tags for the sum of all fishery periods (95% CI 43-45). Based on these 44 tags, we estimated a median harvest rate of 64% (95% CI 52-75%) for Lower Columbia River Tule Chinook originating from SCNFH. TAC estimated a harvest rate of 51% for adult Bonneville Pool Hatchery (BPH) based on CWT—the naming convention used by TAC referring to Lower Columbia River Tule Chinook originating from SCNFH and other Tule release in the Bonneville Pool. However, the TAC estimate does not account for the uncertainty in their estimate due to a variety of sources. The lower 95% CI for the PIT tag harvest estimate was 1% above the TAC point estimate, which suggests that our estimate was marginally greater than the TAC estimate. However, the number of tags passing BON was small and the recovery rate of tags was toward the lower limit recommended by Seber (1982). If we had recovered one less PIT tag our estimates would be similar.

The WDFW and ODFW (2011; Table 18) harvest rate estimate for the adult Snake River Wild Upriver Bright (URB) stock in Zone 6 was 18% based on a combined CWT analysis from multiple populations and the return to the Columbia River mouth. We only sampled one Snake River wild PIT tagged adult and only 7 passed BON. This yielded a harvest rate estimate of 56.7% (95% CI 23.6% to 86.4%). Since there was only one recovery, this estimate should be interpreted with caution. As a more robust alternative, we also made a harvest rate estimate using all adult (hatchery, wild, and unknown origin) Snake River fall Chinook PIT tags captured in the fishery and detected at BON. We sampled a total of 65 tags in the fishery with a total of 4,512 adult detections at BON. These data yield a harvest estimate of 8.3% (95% CI 7.5% – 9.1%). Our harvest rate point estimate based on all Snake River PIT tags was lower than TAC's but we can't formally compare the estimates because the TAC estimate does not calculate an estimate of uncertainty in their harvest rates. However, our harvest estimate was likely biased low due to size selectivity (see next section)

TAC manages fisheries in the Columbia River for impacts to upriver summer steelhead for the Skamania Group (<7/1 at BON), Group A (>7/1 at BON and <78 mm) and Group B (>7/1 and >78 mm. For 2010 fall treaty fisheries, TAC reported a 4% impact for wild Group A steelhead and a 15.7% impact for Group B steelhead based on harvest data and passage at BON (WDFW and ODFW 2011; Table 18). Our harvest rate estimates were 3%, 10%, 5%, and 8% for Middle Columbia River (MCR), Upper Columbia River (UCR), Snake River A-run (SNKA), and Snake River B-run (SNKB) steelhead groups, respectively, which provide a finer resolution for fishery management than the TAC management Group A index estimates. In addition, we estimated a 7% harvest rate for LCR steelhead, which would be considered part of the Skamania management group by TAC, and are assumed not to be impacted by fall fisheries above BON, given the timing of the Skamania stock and time-frame of the fall season.

Fishery Size Selectivity

For coho salmon, comparison of juvenile tagging dates with a frequency histogram (Figure 10) of the number of days between tagging and arrival as adults at Bonneville Dam revealed modes interpreted to represent fish of two ocean ages. Jacks (<1 seawinter) were defined as coho salmon tagged as smolts in spring that returned the same fall (100-225 travel days) or tagged in the fall that returned in the following fall (<450 travel days). Adults (1 seawinter) were defined as coho salmon tagged as smolts in spring that returned the following fall (450-600) travel days), those tagged in fall that returned 2 years in the fall (650-725 travel days), and finally those tagged in the spring that returned 2 years later in the fall (>750 travel days), presumably having spent an additional year in freshwater before smolting.

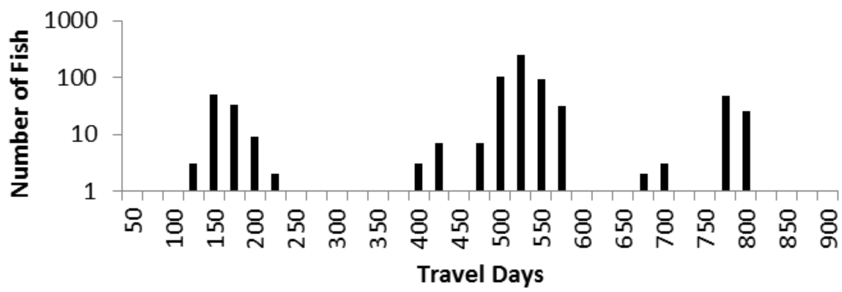

Figure 10. The number of days between juvenile tagging and adult detection at Bonneville Dam in 2010 for coho salmon.

For Chinook salmon comparison of juvenile tagging dates with a frequency histogram (Figure 11) of the number of days between tagging and arrival as adults at Bonneville Dam revealed modes interpreted to represent many ocean ages, which we simplified to three categories (MiniJacks, Jacks, and Adults). MiniJacks (<1 seawinter) were defined as chinook salmon tagged as smolts in spring that returned the same fall (100-250 travel days) or tagged in the fall and returned in the following fall (<400 travel days). Jacks (1 seawinter) were defined as Chinook salmon tagged as smolts in spring that returned the following fall (400-600) travel days. All other Chinook were (>650 travel days) were considered adults, likely representing fish that spent 2, 3 and 4 winters at sea.

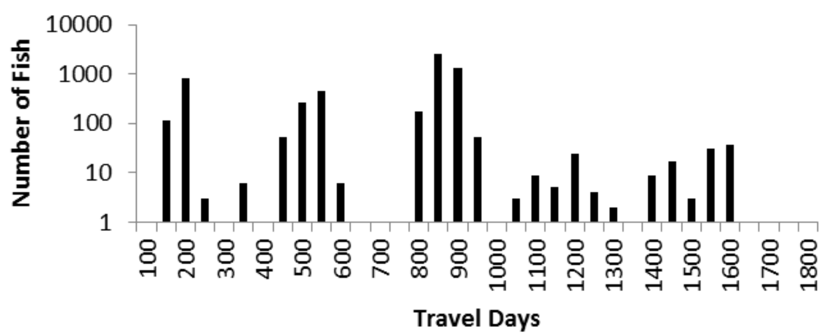

Figure 11. The number of days between juvenile tagging and adult detection at Bonneville Dam in 2010 for Chinook salmon.

For steelhead a frequency histogram of the number of days between juvenile tagging and arrival as adults at Bonneville Dam in conjunction with information about juvenile tagging dates was not sufficient to assign ocean ages without further data censoring (Figure 12).

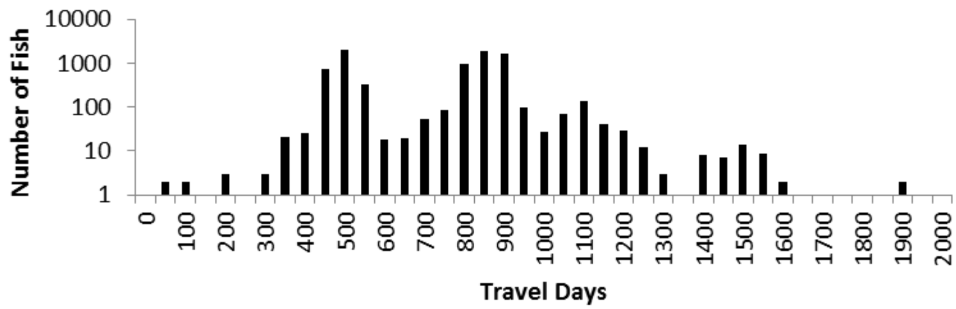

Figure 12. The number of days between juvenile tagging and adult detection at Bonneville Dam in 2010 for steelhead.

We excluded all fish tagged as adults, fish tagged during the summer/fall as parr, and juveniles for which length at tagging was not available. Using this restricted dataset, a length frequency histogram of juveniles at the time of tagging (not shown) revealed two clear and distinct modes, representing parr and smolts with a distinguishing length of ~100 mm. We further censored our dataset to remove steelhead <100 mm, for which ocean entry year was unknown, leaving a dataset with only spring smolts.

A travel days frequency histogram with this restricted dataset then revealed clear modes representing 1, 2, 3, and 4 salt fish, potentially including repeat spawners among the older age classes (Figure 13). We categorized the remaining steelhead as 1-salts (<600 travel days) or 2+ salts (>600 travel days).

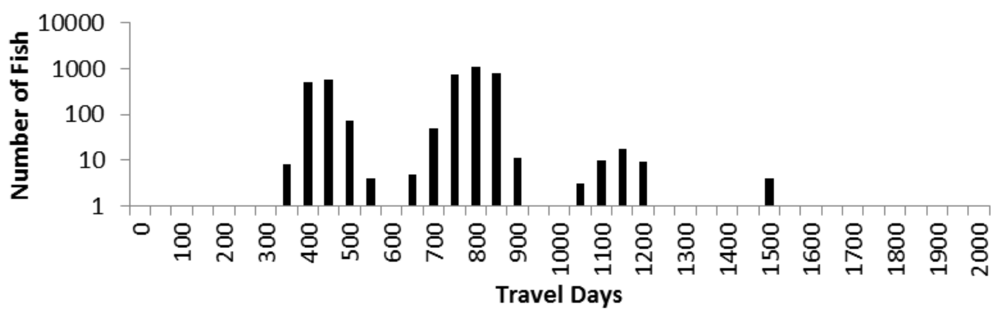

Figure 13. The number of days between juvenile tagging and adult detection at Bonneville Dam in 2010 for steelhead using a censored dataset as described above.

We found significant differences in the age distribution of the catch versus the run at large for all species (P = 0.037, 0.000004, and 0.01, for coho salmon, Chinook salmon, and steelhead, respectively), based on the data in Table 14.

Table 14. The number of steelhead and coho and Chinook salmon assigned to different ocean ages in the overall run above Bonneville as well as in the Zone 6 catch in 2010.

Life	Run Over Bonneville			Catch in Zone 6		
History	Coho	Chinook	Steelhead	Coho	Chinook	Steelhead
<1 Salt	113	923	NA	0	1	NA
1 Salt	562	771	1149	24	4	15
2+ Salt	NA	4157	2753	NA1	72	61

Using age assignments for coho salmon, we examined the effects of size biased harvest on age-structured harvest rates. In this fishery a total of 107 of the 675 coho salmon detected at BON were jacks but 0 of the 24 recoveries in Zone 6 were jacks. Accounting for age structure, the coho salmon adult (non-jack) harvest rate would increase from 17% to 20%, meaning the unadjusted PIT tag adult harvest rate was biased low by 3% in terms of total impacts and the adjusted harvest rate would be 118% of the unadjusted estimate. We did not generate age-structured harvest rates for Chinook or steelhead since our methods for assigning ages required restricting our dataset, which would have prohibitively limited sample sizes for purposes of generating group-specific age structured harvest rates. Based on the results of our selectivity test, our reported harvest estimates are biased high for jacks and younger adults and are biased low for older fish. This suggests that age structured models are a more defensible approach for estimating harvest.

Discussion

Hatchery Detection Study

The estimates of detection probabilities for handheld PIT tag detectors used to sample fisheries were similar using frequentist (Rawding et al. 2013) and Bayesian (in this report) approaches. Kery (2010) noted that Bayesian and maximum likelihood methods often yield similar results when there is sufficient data because vague priors have little influence on the posterior distribution. Model selection based on AIC in Rawding et al. (2013) and the Bayesian approach with DIC for the GLMM also provided a similar ranking of models. This is not surprising since DIC is considered a Bayesian analog of AIC (Link and Barker 2010). As pointed out by Rawding et al. (2013), regardless of the model selection criteria used, the average detection probabilities by species with the two different scanner types were above 99%. Therefore, model averaging was not pursued because it would make little practical difference in the detection rate. See Rawding et al. (2013) for the major limitations in the hatchery PIT tag detection study. It should be noted that some fish are dressed or cleaned after they are caught but before they can be sampled. Since these fish have likely expelled PIT tags inserted into the peritoneal cavity, this leads to fewer fishery PIT tag samples than CWTs because CWT placement is in the snout and tag loss is not an issue for dressed fish. In addition, samplers need to exclude dressed fish from samples they scan for PIT tags.

Assumptions

Since this was a feasibility study, the official estimates of harvest and harvest rates in 2010 fall Columbia River mainstem fisheries are available from TAC. Our harvest rate estimates rely on a number of untested assumptions regarding reported or estimated catch, and random or representative sampling: 1) All commercial buyers in both states are required to report commercially sold catch by species and weight. This study assumed the reporting by weight was accurate for all species; 2) Non-ticketed catch in the Zone 6 treaty fisheries is provided by tribal fisheries managers. We assume their methods are unbiased but we have not incorporated the uncertainty in their non-ticketed estimates into our harvest rate estimates because this is unknown; 3) Only PIT tagged fish that were sold on the Washington shore below BON were sampled for PIT tags in 2010. If PIT tagged fish from all populations are equally susceptible to being caught regardless of net location, our harvest estimates should remain unbiased. However, if PIT tagged fish from certain populations tend to be sold at a higher rate to Washington or Oregon buyers, our sampling of Washington in the lower Columbia River only catch could lead to biased harvest estimates.

We aggregated fishery catch by statistical week, and combined catch across spatial strata: Zones 1-5 and Zones 61, 62, and 63 (Zone 6) were combined. We did this because it was difficult to obtain daily data, some catch sampled at buyers came from a mixture of zones, and PIT tagged fish recoveries were few, necessitating pooling. Furthermore, we assumed that fish sampled for PIT tags and weights were a representative sample of the catch. Representative fisheries sampling from buyers is difficult to ensure because we cannot predict how many commercial buyers are present for any fishery. To best address this situation, we concentrated our sampling effort on known larger buyers with lesser effort on smaller buyers. However, when we sampled at a buyer we sampled all available undressed fish for PIT tags, and used a systematic sample to collect weights and other biological information. Although random sampling is a better option, systematic sampling is used because it easier to consistently implement in the field.

We were hopeful of developing small portable fixed stations for commercial sampling. However, we ran into some challenges with electrical interference causing variable detection rates and found it difficult to set up the small portable stations in most confined fish buying areas. Alternatively, additional PIT tag recoveries could be obtained by working with fish buyers to set up detectors as fish move through the facility or as fish are cleaned, because most PIT tags will come out with the body cavity contents. However, it would be difficult to expand these numbers at fish buyers because the number of fish sampled for tags is unknown and a study to estimate detection rates would be needed at each facility. If regional fish managers decide it is important to obtain more adult detections additional funds should be set aside to work with larger stationary buyers.

Size Selectivity

Size selectivity in many fisheries should be the expected and not the exception (Bernard and Clark 1996, Kendall et al. 2009). The results of our selectivity test and comparison of age-structured and non-age structured estimates for coho salmon suggest that our reported harvest estimates are biased high for jacks and younger adults, which are caught at low rates, and are biased low for older fish, which are caught at relatively higher rates. This suggests that age structured models are a more defensible approach for estimating harvest rates in fall mainstem Columbia River fisheries. Since some of the juveniles that are PIT tagged are not actively migrating smolts and may rear in freshwater for an extended period before emigrating to the ocean, estimates of ocean ages are not straightforward with PIT tags. The denominators in our harvest estimates are the numbers of PIT tagged adult fish detected at BON, so we must be able to assign ages to these fish (for which we have no data other than their tagging information) as well as the catch (for which we have lengths and weights). One of the only data types available to assign ocean age to both these groups using comparable methods is number of days since tagging and the date a fish was tagged.

Data sharing

All PIT tags sampled in fisheries were uploaded to PTAGIS after the season. Therefore, this fishery mortality information is available to fishery managers and researchers. In this report, we purposely provided estimates of the fishery sample rate, PIT tag scanner detection probabilities, the PIT tag detection rate at BON, and the species-specific release site-rear/run types used (Appendices 1-3) to make harvest estimates in this paper. Interested parties may query PTAGIS to obtain the number of adults and jacks detected at BON along with the PIT tags sampled in the fishery belonging to our harvest groups. With this information, they can estimate harvest rates for their own releases our part of a different aggregate. However, when estimating harvest rates they should be aware of the bias caused by the few PIT tag recoveries in fisheries. Despite these caveats we have demonstrated that PIT tag sampling can provide reasonable harvest estimates, finer scale population harvest estimates, and provide estimates of uncertainty, which are not available using current harvest estimation methods in the Columbia River.

Mixture Models

The use of mixture models to estimate harvest rates for coho salmon was necessary because of the bimodal return timing of early and late coho populations (Weinkamp et al. 1996). Inference using mixture models is difficult because there is often no data regarding the number of subpopulations, their timing, or whether the variance in Gaussian mixture models should be equal for both subpopulations (Carlin et al. 2009). However, Figure 9 suggested our PIT recoveries contained information on the mean date of passage and the variance for the early population, since these recoveries substantially overlapped the early mode of the adult coho salmon run at BON, which we used as prior in the mixture model. Since the different subpopulations were well defined due to timing differences, a vague prior provided similar results for the estimate of the late harvest rate. More work is needed in mixture model selection and goodness of fit (GOF) tests.

Conclusions

We reported the first estimates of salmon and steelhead harvest rates based on PIT tags in the Columbia River. It is a natural extension to recover these PIT tags given the small incremental cost to fisheries sampling because fisheries are already sampled (e.g., for CWT and genetic samples) and PIT tags provide fishery managers finer scale harvest estimates along with harvest estimates for wild populations, which are currently not directly available for most populations. We missed the first few periods of the lower Columbia River commercial fishery due to the starting date of our contract. Future sampling will not miss these periods and will be expanded to include all mainstem Columbia River fisheries below MCN.

Large aggregate harvest estimates compared favorably with those developed by Columbia River fishery managers in some cases, but differed in others. In addition, we provided fine scale harvest estimates at the "River", "ESU/DPS", and management group scales. These fine scale estimates are not available with traditional sampling programs, but ESU level estimates are available through genetic sampling (Kassler et al. 2002). These fine scale estimates allow fishery managers greater flexibility in managing for weaker stocks or populations at high risk. However, our estimates were only made possible where there was sufficient juvenile PIT tagging of populations or ESUs of interest. It appears that many populations of steelhead in each Columbia River DPS are PIT tagged, but fewer populations of fall Chinook and coho salmon are PIT tagged. Therefore, making an inference about harvest for these population using PIT tags requires assumptions about the susceptibility to harvest of tagged and untagged populations. However, harvest rates for hatchery populations of Chinook and coho salmon are still available based on CWT.

We calculated harvest rates specific to each fishery, as opposed to exploitation rates. For example, in the Zone 1-5 fishery the harvest rate is based on the abundance of fish entering the Columbia River (Zone 1); and in the Zone 6 fishery the harvest rate based on the abundance at BON. This differs from the TAC, which calculates harvest rates for fisheries based on fish entering the Columbia River. We did not standardize harvest estimates to the mouth of the Columbia River in 2010 because the number of PIT tags detected below BON was small and the fishery sampling was incomplete; this should be considered in future years.

Recommendations

We recommend the following improvements to PIT tag-based harvest estimates:
1) Standardize PIT tag fishery sampling to use the pass over method to be consistent with the methods used in the detection study,
2) Develop statistical methods to estimate “over the bank” or non-ticketed catch in Zone 6 treaty fisheries,
3) Obtain numbers of dressed and whole fish when sampling and explicitly using this information in adjustment of ticketed catch,
4) Stratify Zone 6 sampling by pool especially for summer steelhead due to their holding behavior,
5) Explore the feasibility of reporting fish numbers instead of pounds on fish tickets, which would eliminate the uncertainty in the derived catch estimate,
6) Sample PIT tagged fish in the Oregon landings,
7) Pursue further development of statistical methods that combine harvest information from multiple sources such as PIT tags, CWT tags, and genetic markers using maximum likelihood or Bayesian approaches to provide a single harvest estimate,
8) Consider the use of hierarchical modeling as an alternative to pooled/aggregate estimates,
9) Consider a power analysis for important fishery management groups to ensure sufficient PIT tagging and sampling to meet managers precision goals, and
10) Develop methods for age structured harvest rates based on PIT tags.
11) Explore alternate model parameterization to reduce autocorrelation and allow better mixing of the chains. In addition, we used the “zeros trick” in WinBUGS to estimate harvest and the sensitivity of harvest estimates should be explored.

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Many WDFW and PSMFC staff contributed to the success of this study, through their hard work in developing protocols, coordinating logistics, and sampling fisheries. We thank the various PIT tag coordinators who allowed WDFW to use their PIT tag data for this study. Bob Woodard and the Region 5 data management team provided assistance in PIT tag and fishery database development and management. We thank Rick Golden (BPA) for his support of this project.

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Appendix

Appendix Table 1. The number of PIT tagged Chinook salmon detected passing upstream through Bonneville Dam used in harvest estimates by hierarchical reporting groups at three spatial scales (coarse, medium, and fine). Release Site refers to the release site code from PTAGIS. Only Chinook with a run type of 3 (fall) in PTAGIS were used in analyses.

ESU/DPS	River	Release Site	Rear Type	Number of Tags 1/1—12/31
Hanford URB	Hanford URB	COLR6	W	48
LC Bright	L.W. Salmon Bright	LWSH	H	91
LC Tules	SCNFH	SPRC	H	68
Mid Columbia	Natches	NATCHR	H	1
Mid Columbia	Umatilla	THOP	H	8
Mid Columbia	Umatilla	UMAR	H	6
Mid Columbia	Yakima	YAKIM1	H	197
Mid Columbia	Yakima	YAKIM2	H	8
Mid Columbia	Yakima	CHANDL	W	1
Snake	Clearwater	BCCAP	H	1037
Snake	Clearwater	CLWR	H	1
Snake	Clearwater	NLVP	H	23
Snake	Clearwater	NPTH	H	12
Snake	Clearwater	CLWR	W	6
Snake	Grande Ronde	COUGRC	H	2
Snake	Grande Ronde	GRAND1	H	86
Snake	MF Clearwater	CLWRMF	H	4
Snake	Selway	CEFLAF	H	45
Snake	SF Clearwater	LUGUAF	H	48
Snake	Snake R	CJRAP	H	942
Snake	Snake R	HCD	H	2
Snake	Snake R	LYFE	H	800
Snake	Snake R	PLAP	H	611
Snake		LGRRBR	H	1
Snake		LGRRRR	H	2
Snake		LGRRTR	H	5
Snake		SNAKE3	H	863
Snake		SNAKE4	H	862
Snake		LGRRRR	U	3
Snake		LGRRTR	U	28
Snake		SNAKE2	U	10
Snake		SNAKE3	U	1
Snake		LGRRBR	W	1
Upper Columbia	Columbia R	PRDH	H	28

Appendix Table 2. The number of PIT tagged coho salmon detected passing upstream through Bonneville Dam used in harvest estimates by hierarchical reporting groups at three spatial scales (coarse, medium, and fine). Release Site refers to the release site code from PTAGIS.

Large Aggregate	ESU/DPS	River	Release Site	Rear Type	Number of Tags 1/1—12/31
Abv BON	Mid Columbia	Yakima	AHTANC	H	6
Abv BON	Mid Columbia	Yakima	CHANDL	H	1
Abv BON	Mid Columbia	Yakima	CLEFBY	H	1
Abv BON	Mid Columbia	Yakima	YAKIM1	H	105
Abv BON	Mid Columbia	Yakima	YAKIM2	H	108
Abv BON	Mid Columbia	Yakima	CHANDL	W	2
Abv BON	Mid Columbia	Yakima	YAKIM1	W	1
Abv BON	Mid Columbia	Yakima	YAKIM2	W	5
Abv BON	Snake	Clearwater	CLEARC	H	12
Abv BON	Snake	Clearwater	KOOS	H	31
Abv BON	Snake	Clearwater	LAPC	H	52
Abv BON	Upper Columbia	Entiat	ENTIAR	W	1
Abv BON	Upper Columbia	Methow	WINT	H	5
Abv BON	Upper Columbia	Methow	WINTBC	H	54
Abv BON	Upper Columbia	Methow	METHR	W	1
Abv BON	Upper Columbia	Wenatchee	BUTCHP	H	14
Abv BON	Upper Columbia	Wenatchee	LEAV	H	112
Abv BON	Upper Columbia	Wenatchee	NASONC	H	1
Abv BON	Upper Columbia	Wenatchee	ROLFIP	H	28
Abv BON	Upper Columbia	Wenatchee	NASONC	W	1
Abv BON	Upper Columbia	Yakima	NATCHR	H	134

Appendix Table 3. The number of PIT tagged steelhead detected passing upstream through Bonneville Dam used in harvest estimates by hierarchical reporting groups at three spatial scales (coarse, medium, and fine). Release Site refers to the release site code from PTAGIS. Run management designations A, B, and combined AB are listed for all Snake River stocks.

Large Aggregate	ESU/DPS	River	Release Site	Rear Type	Number of Tags 7/1—10/31	Number of Tags 4/1—10/31
	LCR	Wind	PANT2C	W	5	11
	LCR	Wind	TROUTC	W	11	26
	LCR	Wind	WIND2R	W	48	77
	MCR		MXWLCN	U	1	1
	MCR	Deschutes	TROU2C	W	114	114
	MCR	John Day	BEAR2C	W	2	3
	MCR	John Day	BRIDGC	W	32	33
	MCR	John Day	CAMP2C	W	4	4
	MCR	John Day	GABLEC	W	1	1
	MCR	John Day	GRBLDC	W	1	1
	MCR	John Day	JDAR1	W	5	5
	MCR	John Day	JDAR2	W	48	51
	MCR	John Day	JDARMF	W	22	22
	MCR	John Day	JDARSF	W	124	129
	MCR	John Day	JSFBC	W	7	9
	MCR	John Day	JSFDC	W	2	2
	MCR	John Day	JSFMC	W	20	20
	MCR	John Day	MCKAYC	W	1	1
	MCR	Umatilla	FEEDCN	H	1	1
	MCR	Umatilla	MEACHC	H	3	3
	MCR	Umatilla	MEACHC	W	4	4
	MCR	Umatilla	MINP	H	22	22
	MCR	Umatilla	MXWLCN	H	4	4
	MCR	Umatilla	PENP	H	15	15
	MCR	Umatilla	UMAR	H	31	31
	MCR	Umatilla	UMAR	W	8	8
	MCR	Walla Walla	DAYP	H	53	66
	MCR	Walla Walla	DAYP	W	22	22
	MCR	Walla Walla	MILLC	W	7	8
	MCR	Walla Walla	TOUCHR	H	61	82
	MCR	Walla Walla	TOUCHR	W	5	5
	MCR	Walla Walla	WALLAR	H	61	78
	MCR	Walla Walla	WALLAR	W	49	55
	MCR	Walla Walla	YELHKC	W	5	5
	MCR	Yakima	AHTANC	W	1	1
	MCR	Yakima	NFTEAN	W	6	6

Appendix Table 3. Continued

Large Aggregate	ESU/DPS	River	Release Site	Rear Type	Number of Tags 7/1—10/31	Number of Tags 4/1—10/31
	MCR	Yakima	ROZTAL	W	7	7
	MCR	Yakima	SATUSC	W	1	1
	MCR	Yakima	TEANAR	W	1	1
	MCR	Yakima	TOPPEC	W	10	12
	MCR	Yakima	YAKIM1	W	2	3
	MCR	Yakima	YAKIM2	W	1	2
	UCR		PRDLD1	U	9	9
	UCR		PRDTAL	U	11	12
	UCR		RI2BYP	H	51	53
	UCR		RI2BYP	U	18	18
	UCR		RI2BYP	W	33	39
	UCR		RISTAL	U	15	15
	UCR		TWISPW	U	2	2
	UCR		WANTAL	U	7	10
	UCR		WELFBY	U	1	2
	UCR	Entiat	ENTIAR	W	61	65
	UCR	Entiat	MADRVR	W	6	8
	UCR	Methow	BEAV2C	W	6	7
	UCR	Methow	CHEWUR	W	2	2
	UCR	Methow	GOLD2C	W	1	1
	UCR	Methow	METHR	H	7	7
	UCR	Methow	METHR	W	1	2
	UCR	Methow	METTRP	H	1	1
	UCR	Methow	PESHAR	W	1	1
	UCR	Methow	TWISPR	H	10	11
	UCR	Methow	TWISPR	W	11	12
	UCR	Methow	WINT	H	35	38
	UCR	Okanogan	OMAKC	H	9	9
	UCR	Wenatchee	CHIWAC	W	2	2
	UCR	Wenatchee	CHIWAR	H	33	38
	UCR	Wenatchee	CHIWAR	W	1	1
	UCR	Wenatchee	CHIWAT	W	11	12
	UCR	Wenatchee	NASONC	H	161	172
	UCR	Wenatchee	NASONC	W	14	17
	UCR	Wenatchee	TUMFBY	H	10	10
	UCR	Wenatchee	TUMFBY	W	8	9
	UCR	Wenatchee	WENATR	H	273	313
	UCR	Wenatchee	WENATR	W	26	32
	UCR	Wenatchee	WENATT	W	8	10
SNK AB			IHRBYP	H	11	12

Appendix Table 3. Continued

Large Aggregate	ESU/DPS	River	Release Site	Rear Type	Number of Tags 7/1—10/31	Number of Tags 4/1—10/31
SNK AB			IHRBYP	U	1	1
SNK AB			IHRBYP	W	1	1
SNK AB			IHRTAL	H	3	3
SNK AB			LGRLDR	W	23	23
SNK AB			LGRRBR	H	680	707
SNK AB			LGRRBR	W	849	910
SNK AB			LGRRRR	H	238	245
SNK AB			LGRRRR	U	1	1
SNK AB			LGRRRR	W	247	255
SNK AB			LGRTAL	W	2	3
SNK AB			LGSTAL	H	5	5
SNK AB			LMNBYP	H	92	101
SNK AB			LMNBYP	U	12	12
SNK AB			LMNBYP	W	32	34
SNK AB			LMNTAL	U	1	1
SNK AB			SNAKE1	H	4	4
SNK AB			SNAKE2	H	3	3
SNK AB			SNAKE2	U	2	2
SNK AB			SNKTRP	H	46	47
SNK AB			SNKTRP	W	21	22
SNK AB		Clearwater AB	CLWTRP	W	18	18
SNK AB		Salmon AB	SALR3	H	139	140
SNK AB		Salmon AB	SALR4	H	41	41
SNK AB		Salmon AB	SALTRP	H	22	22
SNK AB		Salmon AB	SALTRP	W	2	2
SNK AB	SNA	Asotin A	ASOTIC	W	24	27
SNK AB	SNA	Clearwater A	BIGBEC	W	17	17
SNK AB	SNA	Clearwater A	LBEARC	W	1	1
SNK AB	SNA	Clearwater A	PINE2C	W	2	2
SNK AB	SNA	Clearwater A	POTREF	W	4	4
SNK AB	SNA	Grand Ronde A	BCANF	H	190	190
SNK AB	SNA	Grand Ronde A	CATHEC	W	8	8
SNK AB	SNA	Grand Ronde A	COTP	H	113	113
SNK AB	SNA	Grand Ronde A	GRAND2	W	11	11
SNK AB	SNA	Grand Ronde A	GRNTRP	H	68	69
SNK AB	SNA	Grand Ronde A	GRNTRP	W	27	27
SNK AB	SNA	Grand Ronde A	LCATHC	W	1	1
SNK AB	SNA	Grand Ronde A	LOOKGC	W	12	12
SNK AB	SNA	Grand Ronde A	LOSTIR	W	12	13
SNK AB	SNA	Grand Ronde A	MINAMR	W	9	11

Appendix Table 3. Continued

Large Aggregate	ESU/DPS	River	Release Site	Rear Type	Number of Tags 7/1—10/31	Number of Tags 4/1—10/31
SNK AB	SNA	Grand Ronde A	WALH	H	242	246
SNK AB	SNA	Imnaha A	BSHEEC	H	146	148
SNK AB	SNA	Imnaha A	IMNTRP	W	121	124
SNK AB	SNA	Imnaha A	LSHEEF	H	291	293
SNK AB	SNA	L Salmon A	SLATEC	H	5	5
SNK AB	SNA	Lemhi A	LEMHIR	W	9	9
SNK AB	SNA	Lemhi A	LEMHIW	W	8	8
SNK AB	SNA	Little Salmon A	LSALR	H	312	313
SNK AB	SNA	Little Salmon A	RPDTRP	W	6	6
SNK AB	SNA	Pahsimeroi A	PAHTRP	H	260	260
SNK AB	SNA	Pahsimeroi A	PAHTRP	W	7	7
SNK AB	SNA	Salmon A	HAYDNC	W	8	8
SNK AB	SNA	Salmon A	SAWT	H	44	44
SNK AB	SNA	Salmon A	SAWTRP	H	105	107
SNK AB	SNA	Salmon A	SAWTRP	W	1	1
SNK AB	SNA	Snake A	LYFE	H	39	52
SNK AB	SNA	Snake A	SNAKE4	H	167	168
SNK AB	SNA	Tucannon A	TUCR	H	199	266
SNK AB	SNA	Tucannon A	TUCR	W	38	45
SNK AB	SNA	U Salmon A	SLAT2C	H	28	28
SNK AB	SNA	U Salmon A	VALEYC	H	33	33
SNK AB	SNA	YF Salmon A	YANKFK	H	48	49
SNK AB	SNB	Clearwater B	CLEARC	H	40	41
SNK AB	SNB	Clearwater B	CLEARC	W	1	1
SNK AB	SNB	Clearwater B	DWORMS	H	313	314
SNK AB	SNB	Clearwater B	LOLOC	H	23	23
SNK AB	SNB	EF Salmon B	SALEFT	H	18	19
SNK AB	SNB	EF Salmon B	SALREF	H	61	61
SNK AB	SNB	Lochsa B	CFCTRP	W	31	31
SNK AB	SNB	Lochsa B	COLTKC	W	12	12
SNK AB	SNB	Lochsa B	FISTRP	W	111	113
SNK AB	SNB	MF Salmon B	BIG2C	W	19	19
SNK AB	SNB	MF Salmon B	CAMASC	W	2	2
SNK AB	SNB	MF Salmon B	MARTRP	W	1	1
SNK AB	SNB	Salmon B	SQAW2C	H	105	105
SNK AB	SNB	Salmon B	SQUAWP	H	4	4
SNK AB	SNB	Secesh B	LAKEC	W	3	3
SNK AB	SNB	Secesh B	SECESR	W	5	5
SNK AB	SNB	Secesh B	SECTRP	W	5	5
SNK AB	SNB	Selway B	MOOS2N	W	1	1

Appendix Table 3. Continued

Large Aggregate	ESU/DPS	River	Release Site	Rear Type	Number of Tags 7/1—10/31	Number of Tags 4/1—10/31
SNK AB	SNB	SF Clearwater B	CLWRSF	H	171	171
SNK AB	SNB	SF Clearwater B	CROOKR	H	49	49
SNK AB	SNB	SF Clearwater B	CROTRP	W	3	3
SNK AB	SNB	SF Clearwater B	MEAD2C	H	14	14
SNK AB	SNB	SF Clearwater B	MILL2C	H	1	1
SNK AB	SNB	SF Clearwater B	REDP	H	7	7
SNK AB	SNB	SF Clearwater B	REDR	H	56	56
SNK AB	SNB	SF Salmon B	JOHTRP	W	8	8
SNK AB	SNB	SF Salmon B	KNOXB	W	13	13
SNK AB	SNB	SF Salmon B	SFSTRP	W	5	5

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