Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 1: Introduction and Methods

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Introduction

This report describes the current status of salmon and steelhead in Oregon Lower Columbia River tributaries, including the Willamette River. This region contains six groups of salmon and steelhead listed as threatened under the US Endangered Species Act (ESA): Lower Columbia River (LCR) chinook, Columbia River (CR) chum, LCR coho, LCR steelhead, Upper Willamette (UW) chinook, and UW steelhead. For salmon, the listed group is referred to as an Evolutionarily Significant Unit (ESU) and for steelhead, the listed group is a Distinct Population Segment (DPS) (Waples 1991, NMFS 2006). The LCR chinook ESU, CR chum ESU, LCR coho ESU and LCR steelhead DPS include populations that spawn in tributaries on both the Oregon and Washington sides of the Columbia River. This report, however, deals only with the populations spawning in Oregon tributaries. The status of Washington populations is discussed in the Washington Lower Columbia recovery plan (Lower Columbia Fish Recovery Board 2004) and elsewhere (McElhany et al. 2004). The UW chinook ESU and UW steelhead DPS are wholly contained in the Willamette River Basin of Oregon and all their component populations are addressed here.

The primary reason for conducting this assessment is to inform salmon recovery planning in Oregon. Information on individual population status is useful in scoping the level of effort needed to improve population status and reach recovery goals. It can also be useful in prioritizing populations and actions for recovery efforts. Another purpose of this report is to evaluate proposed viability criteria. Viability criteria describe what to measure to evaluate extinction risk ('metrics') and levels of the metrics associated with a low extinction risk ('thresholds'). These viability criteria are meant to inform delisting criteria for ESA listed species (NMFS 2000). In April 2006, the Willamette/Lower Columbia Technical Recovery Team (WLC-TRT) distributed revised draft viability criteria (McElhany et al. 2006a)[1]. By applying the viability report thresholds in this current status evaluation, we explored the utility of the 2006 draft criteria.

It is useful to consider the distinction between setting recovery goals and conducting a current status assessment. The viability criteria developed by the WLC-TRT are intended to inform recovery goals. Recovery goals are targets for the future and the goals tend to include either a very limited suite of metrics or are limited to describing guiding principles rather than quantitative thresholds. A current status evaluation, on the other hand, is concerned with providing an accurate view of where a population is at a given time and should utilize *all* available information. Existing data sets may contain information not identified as part of the viability goal metrics, but this information may still provide indicators of population status and should not be ignored. Accordingly, in this report we analyze both the viability criteria metrics and any other relevant data available.

Since the focus of the ESA is on extinction risk, in this assessment, we are equating the term "status" with "extinction risk." Although there may be alternative definitions of status (e.g., "harvestable"), this analysis is an evaluation only of population extinction

[1] The April 2006 WLC-TRT revised viability report [ref] built on a 2003 WLC-TRT viability report [ref]. Unless otherwise noted, references in this document to the "viability report" refer to the 2006 version [ref].

risk. There is clearly a link between extinction risk and other definitions of status, but we do not explicitly consider such links.

Because we need to evaluate a diverse array of information types, the ultimate estimation of risk involves some level of professional judgment. Although our analysis was systematic and evidence-based, it was not based on a single quantitative algorithm. While using only a fixed set of quantitative criteria might have the advantage of clear repeatability, and a perception of objectivity, it is likely to be less accurate because it fails to take into consideration population specific information and information that is not readily quantified.

Population status (i.e., extinction risk) is a continuous variable from almost 0% chance (no risk) to 100% chance (certain extinction). Following the methods in the viability report, we partition this continuum into the general risk categories shown in Table 1. A population with a persistence probability greater than 95% over a 100-year period is termed "viable". This level of risk is consistent with VSP guidelines (McEhany et al., 2000), the conservation literature (e.g., NRC, 1995), and with informal policy guidance indicating that, at least initially, the appropriate recovery target at the population level would be no more than a 5 percent risk of extinction within 100 years. Although the categories are defined in terms of quantitative extinction risk, we can rarely estimate extinction risk with precision and the categories are qualitative indicators. Estimating extinction risk is a challenging exercise – we are attempting to predict events far into the future. It is essential when presenting information on population status to include some assessment of the uncertainty associated with the prediction. We include both quantitative and qualitative assessments of the uncertainty in our extinction risk estimates.

Table 1: Population persistence categories (copied from McElhany et al. 2006a).

Population Persistence Category	Probability of population persistence in 100 years	Probability of population extinction in 100 years	Description
0	0–40%	60-100%	Either extinct or very high risk of extinction.
1	40–75%	25-60%	Relatively high risk of extinction in 100 years.
2	75–95%	5-25%	Moderate risk of extinction in 100 years.
3	95–99%	1-5%	Low ("negligible") risk of extinction in 100 years (viable salmonid population).
4	>99%	<1%	Very low risk of extinction in 100 years.

In parts of this report, we include a description of results from the Oregon Native Fish Report (ODFW 2005). Although comparison of our analysis to the Native Fish Report is interesting, it is important to note the scope and limitations of the Native Fish Report. These are best summarized in the words of the Native Fish Report itself:

> *"…This report summarizes risk assessment completed for native salmon and steelhead, most native trout, and other selected native fish species using the NFCP [Native Fish Management Policy] <u>interim criteria</u>. …. Risk, as used in this report, refers to the threat to the conservation of a unique group of populations in the near-term (5-10 years). …The NFCP <u>interim criteria</u> provide temporary guidance to ensure the conservation of native fish prior to completion of more detailed conservation plans for each species or group of populations. …The*

> *interim criteria* do not *describe long-term, extinction risks such as continuing downward trends, increasing threats, or extended intervals of unfavorable environmental conditions. Such long-term risks are better assessed with more in-depth analyses than was conducted for this report and will be considered in conservations plans."*

Our report is a more comprehensive analysis with a longer time horizon than the Native Fish Report.

This analysis has been conducted as a joint project of the NOAA Fisheries Northwest Fisheries Science Center (NWFSC), the Oregon Department of Wildlife (ODFW) and Cramer Fish Sciences (under contract to ODFW). Although the report has benefited from review and consultation with other biologists, both inside and outside our agencies, the final evaluations are those of the report authors, which may or may not reflect agency opinion.

Methods

Methods Overview

The majority of the methods used in the report are described in the WLC-TRT viability report (McElhany et al. 2006a), which builds on the basic framework in the NOAA Technical Memorandum on Viable Salmonid Populations (VSP (McElhany et al. 2000)) and a previous WLC-TRT interim viability report (McElhany et al. 2003). The methods described below are largely a summary of the viability report and readers are encouraged to examine the viability report for a more complete discussion. Since the viability criteria relate to evaluating risk status under the ESA, we are ultimately concerned with the status of the ESU/DPS (since the ESU/DPS is the listed unit, recovery criteria apply at the ESU/DPS scale). In the viability criteria, ESU/DPS status is assessed by examining the status of individual populations and groups of populations (called "strata") within a framework for ESU/DPS viability. Population boundaries for Pacific salmonids in the WLC have been identified in Myers et al. (Myers et al. 2006) and the population strata groupings are described in the viability report.

ESU/DPS Level Evaluation

Since this report is concerned only with the status of Oregon populations, it does not summarize status of the full Lower Columbia chinook and coho ESUs, steelhead DPS, or the Columbia River chum ESU, since those ESU/DPSs include some populations in Washington. The UW chinook ESU and steelhead DPS are both entirely in Oregon, so this report does analyze their status. The ESU/DPS criterion is that all historical strata need to be at a low risk of extinction. A low risk stratum is described as one with at least two viable populations (i.e. persistence category ≥3), where the average of the persistence categories for all historical populations is ≥2.25 based on the scale in Table 1, and there are sufficient viable populations to ensure that the stratum is buffered from the risks of catastrophic events, degraded metapopulation processes, and degraded evolutionary processes. Support for these recommendations is provided in the viability reports.

Individual population status is determined by examining three main attributes: 1) abundance and productivity (A&P); 2) spatial structure (SS); and 3) diversity (DV)[2]. These three primary attributes are sometimes referred to as the "biological" factors, or what we can learn from looking primarily at fish performance. A comprehensive evaluation of population status should also include an examination of the threats facing the population with an emphasis on future environmental conditions. Understanding future conditions is necessary to address the stationarity assumption inherent in the biological factor analysis. The stationarity assumption is that the recent past is a reasonable predictor of future fish performance. This assumption would be violated if future environmental conditions are different from the recent past (where "environment" is broadly defined to include anything that affects salmon). In this report, we do not conduct a complete assessment of likely future environmental conditions and their

[2] The VSP report (McElhany et al. 2000) separates abundance and productivity into two separate attributes for a total of four attributes. Because the effects of abundance and productivity on extinction risk are so interconnected, we analyze them together.

predicted impacts on population biological status, which would involve examination of both current and potential population threats. In conducting the analysis, we largely rely on the stationarity assumption, but make some adjustments to evaluations of the three population attributes if a violation of the assumption seems likely (e.g., with regard to global climate change). A more thorough evaluation of likely future environmental conditions would greatly enhance population status evaluation.

ESU/DPS status was evaluated for each population on the 0-4 persistence category scale shown in Table 1. We estimated the overall population score by first evaluating on the same 0-4 scale each of the three primary population attributes (abundance and productivity, spatial structure and diversity). The 0-4 score for the individual attributes was based on what risk would be suggested by examining that attribute in isolation. The individual attributes are likely to be correlated, so these are not independent factors; however, each does contribute some unique information.

We relied on professional judgment to reach overall conclusions on risk status associated with each population's attributes based on consideration of any and all quantitative metrics available. Using a single, quantitative method for combining all of the available information did not seem a practical approach. To capture the uncertainty in our assessment, we present our conclusions as a probability distribution in the form of "diamond graphs" (Figure 1). These graphs are presented with the population risk categories on the vertical axis. The thickness of the diamond at any particular point indicates the relative probability of that risk category. The most likely risk category is shown by the thickest part of the diamond and the maximum and minimum likely risks are indicated by the upper and lower tips of the diamond. Although the risk probability diamonds are not generated by any quantitative algorithm, the presentation of the multiple quantitative analyses and any qualitative considerations leading up to the risk conclusions are intended to make the evaluation as transparent as possible.

Overall population scores were estimated from individual attribute scores by using a modification of the weighted average algorithm developed by the WLC-TRT. In the weighted average method, the 0-4 scores are averaged, with abundance and productivity weighted twice as much as the sum of the other two attributes because it is considered the better predictor of extinction risk (Equation 1).

Equation 1: popScore = 4/6*abud&ProdScore + 1/6*spaceScore + 1/6*diverScore

The weighted average approach integrates all three of the population attributes, but may give a misleading result in cases where the abundance and productivity is low even though spatial structure and diversity are not excessively degraded. In these cases, the population is likely experiencing some risk factor driving down abundance and productivity that is not reflected in the spatial structure and diversity score. In these cases, it is appropriate to evaluate the status of the population based on the low abundance and productivity, rather than incorporating all the attributes in a weighted average. We therefore applied the following rule:

> *If the abundance and productivity risk estimate is lower than the spatial structure or diversity estimate, use the abundance and productivity rating as the overall population rating, otherwise, use the weighted average method to set the overall population rating.*

With this rule, spatial structure and diversity ratings might make a summary score lower than the abundance and productivity score, but spatial structure and diversity ratings will not make the summary score higher than the abundance and productivity score. This method is more precautionary than always applying the weighted average algorithm.

We present the overall population status in the form of diamond graphs like those used to present individual attribute status. If the weighted average method is applied, a Monte Carlo approach is used to generate the diamonds. Independent values are randomly drawn from the diamond graph distributions of the individual attributes then averaged using Equation 2. This is repeated 10,000 times and the resulting distribution of population scores are presented as a diamond graph.

Equation 2: popScore = W_a*abud&ProdScore + W_s*spaceScore + W_d*diverScore

In Equation 2, the parameters W_a, W_s and W_d replace the average weights of 4/6, 1/6 and 1/6 of Equation 1 because these weights themselves are estimated with uncertainty and are treated as random variables in the Monte Carlo process. The weights are constrained to sum to one and we used a random multinomial approach to describe the uncertainty in these parameters. This approach is described in Appendix A. We utilized a shape parameter of 50, which preserved the feature that abundance and productivity are generally weighted more than spatial structure and diversity. The TRT viability report did not include uncertainty in the attribute weights and this is a new feature of this analysis.

If the overall population summary is based on the abundance and productivity rating because it is lower than the spatial structure and diversity ratings, a different method for describing the overall population diamond is applied. The diamond graphs are a representation of a triangular distribution, which is define by three parameters: 1) mode, 2) lower bound, 3) upper bound. The mode is the point estimate or "most likely" value and is the fattest part of the diamond. If, after applying the rule above, the abundance and productivity mode will be used as the overall population mode, the lower and upper bounds on the overall population summary diamond are determined as the minimum lower or upper bound of all three attributes (Equations 3 and 4).

Equation 3: popLower=min(A&P_Lower, SS_Lower, DV_Lower)

Equation 4: popUpper=min(A&P_Upper, SS_Upper, DV_Upper)

This sets the most precautionary upper and lower bound for the overall population diamond considering all the population attributes.

The overall population status is presented in the form of the diamond graphs and we do not present the results in a "pass" or "fail" format. We prefer the diamond graph method because it retains more information (i.e., the uncertainty inherent in the analysis). If a pass or fail decision is required for a management decision, it is important that that decision be made with an understanding of the full range of possible risk status for the populations. By presenting the results as a distribution of possible extinction risks, the results of this analysis may be applied to different sorts of management problems, which may require different levels of precaution regarding risk.

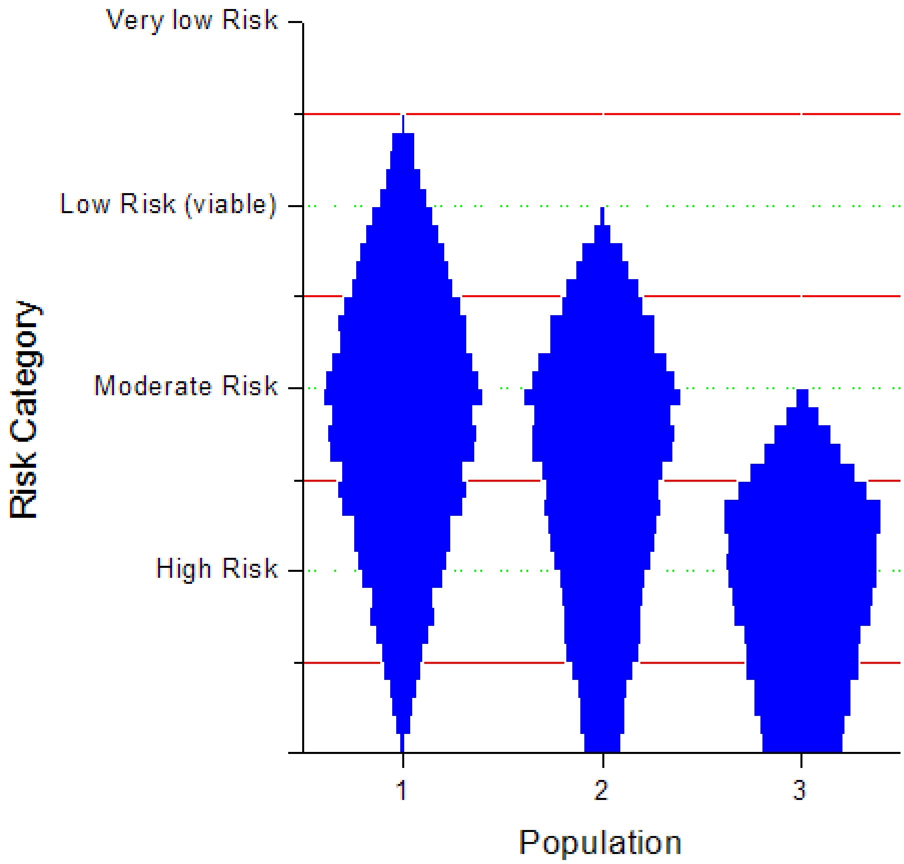

Figure 1: Example risk summary "diamond graph" for three populations with different risk profiles. The risk categories correspond to the probabilities in Table 1.

Abundance and Productivity

The abundance and productivity evaluation is predicated on two basic observations: 1) all else being equal, a larger population is less likely to go extinct than a small one and 2) all else being equal, a highly productive population is less likely to go extinct than a population with low productivity. Productivity is an indication of a population's "resilience" or tendency to return to high abundance if perturbed to low abundance. We typically measure productivity as the number of offspring per parent when there are very few parents (in fisheries parlance, "intrinsic productivity" or "recruits per spawner at extremely low spawner densities").

The quantity and quality of data available to evaluate the abundance and productivity varies dramatically among WLC populations. We can divide the populations into two basic groups: those with sufficient time series of abundance and related parameters for a quantitative evaluation and those without sufficient time series. For those with a time series, we explored a number of analytical approaches which are described in more detail below and in the viability report. For those without an adequate time series, we examined any available information (e.g., one-time surveys, qualitative reports) and often had to rely on extrapolation from assessments of similar populations where quantitative analysis was possible. Even for populations where a time series was available, we did not limit our analysis to the metrics described below, but examined any relevant piece of information. Time series used for the viability analysis are included in this report as Appendices B and C. For populations with adequate time series data, we present some general summary statistics, including comparison to a simple minimum abundance threshold, plus the results of three Population Viability Analysis (PVA) modeling approaches: 1) Viability Curve Criteria (VCC), 2) the Conservation Assessment and Planning Model (CAPM) and 3) a simple generic stochastic stock recruitment model (PopCycle). By exploring three different extinction risk models, we can develop better extinction risk estimate and understanding of the confidence around that estimate.

Summary Graphics and Statistics

Simply viewing a few summary graphs, like the abundance time series and a few simple statistics like the fraction of hatchery origin spawners can provide a lot of information for the abundance and productivity evaluation. For each population with adequate data, we present graphs of the time series of spawner abundance (distinguishing between total spawners and natural origin spawners), the time series of the fraction of hatchery origin spawners, the time series of harvest rate, and both escapement and pre-harvest recruitment curves. A table of summary statistics was also generated, showing the time period of the series, average abundance, average recruitment, growth rates, etc. Descriptions of the statistics estimated for every population with an available time series are shown in Table 2. These statistics were calculated for two different time periods: 1) the length of the entire available time series (which differs by population); and 2) the time series from 1990 to the most currently available year (typically 2004 or 2005). The 1990-current period is arbitrarily described as "recent". Where appropriate, statistics are also estimated based on both escapement and pre-harvest recruitment, since both sorts of calculations provide information for extinction risk analysis. In these analyses, the relative reproductive success of hatchery origin spawners is assumed to be the same as natural origin spawners (see viability report for a discussion of this issue). Many of the metrics presented in this summary table are likely to be highly correlated (e.g. Lambda and trend in ln(abundance)), and it would be reasonable to reduce the number of metrics to eliminate redundancy and shorten the table. However, all of these different metrics have been used in the past in different salmon assessments and we considered it useful to include all the metrics for comparative purposes.

Tables of the recruitment curve fits are also provided for both the escapement and preharvest analyses, where data were available. We estimated productivity, capacity and recruitment variance for the random walk, random walk with trend, constant recruitment, hockey-stick, Beaverton-Holt, Ricker, and MeanRS recruitment functions. The MeanRS recruitment function is described in the section below on viability curves. Equations for the other models are shown in Table 3. For all models except the MeanRS, parameters were fit using a Bayesian approach and we provide both point estimates and 95% posterior probability intervals. For the MeanRS method the 95% intervals were based on a bootstrap of 10,000 resamplings with replacement. We also present relative corrected Akaike information criterion (AICc) values to compare the 'fit' of the alternative models (Burnham and Anderson 1998). The model that is the "best" approximation has a relative AICc = 0. Models that are nearly indistinguishable from best have a relative AICc <2. Models that are possible, but less likely, contenders as best have 2 < relative AICc < 10. Models that are very unlikely to be the best approximating model have relative AICc > 10.

Table 2: Description of abundance and productivity statistics calculated for populations with abundance time series.

Statistic	Description
Time Series Period	Years used in the analysis
Length of Time Period	Number of Years used in the analysis
Geometric Mean Natural Origin Spawner Abundance	Geometric mean of natural origin spawners with 95% confidence intervals shown in parentheses. This parameter is compared to the minimum abundance threshold MAT and colored blue, green, orange, yellow or red for the very low risk, low risk, moderate risk, high risk or very high risk categories, respectively (see Figure 2)
Geometric Mean of Recruit Abundance	Geometric mean of natural origin recruits (either to escapement or pre-harvest) with 95% confidence intervals shown in parentheses. If recruits to escapement, will be similar, but not identical to geomean natural origin spawners. The geometric mean recruits is the "**Abundance**" parameter of the MeanRS method viability curve.
Lambda	Median annual population growth rate based on four-year running sum with 95% confidence interval. The variance estimate used to estimate the confidence interval uses the slope method approach of Holmes (2000). The statistic is the same used in recent NOAA status evaluations (Good et al.) Values above one indicate a growing population, values below one indicate a declining population. The statistic is corrected to hatchery fish to show the growth rate of the natural population if there had not been a hatchery subsidy.
Trend in Natural Origin Abundance	This is the exponentiated slope of the regression of ln(natural origin spawners) vs. year. The 95% confidence intervals are shown in

	parentheses. Values above one indicate an increasing number of natural origin spawners; values below one indicate a declining number of natural origin spawners. Hatchery origin spawners are ignored in the estimation of this statistic.
Geometric Mean Recruits per Spawner	Geometric mean of recruits per spawner using all brood years in the analysis period. The 95% confidence intervals are shown in parentheses.
Geometric Mean Recruits per Spawner for Broods below Median	Geometric mean of recruits per spawner using brood years where the spawner abundance is less than the median spawner abundance. The idea is to estimate recruits per spawner under conditions with reduced dependent effects. The 95% confidence intervals are shown in parentheses. This is the "**Productivity**" parameter of the MeanRS method viability curve.
Average Hatchery Fraction	The arithmetic average fraction of hatchery origin spawners on the spawning grounds over the time series period.
Average Harvest Rate	The arithmetic average harvest rate of natural origin fish over the time series period.
CAPM frequency distribution of estimated extinction probabilities	Median extinction probability for each population derived from 200 bootstrap samples of the raw data set. Included (in parentheses) are values for 5th and 95th percentiles associated with the median probability (50th percentile). This value is explained in more detail in the section on the CAPM model and in Appendix E.
PopCycle extinction risk estimate	This is the population extinction risk result from the PopCycle model as describe in the PopCycle section below and in Appendix F.

Table 3: Recruitment functions used for summary analysis of Oregon WLC salmon and steelhead populations.

Model Name	Equation [a]
Random walk	$R = S\exp(\sigma_0 Z)$
Random walk with drift; stochastic exponential growth or decline	$R = S\exp(a_1 + \sigma_1 Z)$
Constant recruitment	$R = b_2\exp(\sigma_2 Z)$
Stochastic hockey stick; stochastic exponential growth with a ceiling	$R = \min(S, b_3)\exp(a_3 + \sigma_3 Z)$
Ricker; stochastic logistic	$R = S\exp(a_4 + b_4 S + \sigma_4 Z)$
Beverton-Holt	$R = \dfrac{a_5 S}{1 + \dfrac{a_5}{b_5} S}\exp(\sigma_5 Z)$

[a] In the equations,

S_t = the number of spawners
R = the number of recruits
Z = a unit normal random variable
$\sigma_\#$ = the standard deviation of the process error
$a_\#$ and $b_\#$ = equation-specific parameters, with the $a_\#$ parameter relating in some way to "intrinsic productivity" and the $b_\#$ parameter relating in some way to "capacity"

Population Size Thresholds

The TRT viability report describes population minimum abundance thresholds (MATs) as one part of the abundance and productivity evaluation. Before placed in a particular risk category, a population should exceed the MAT criterion AND exceed the viability curve criteria (described below) AND exceed any of the TRT's qualitative criteria for that category. The MAT criteria are derived from a combination of general conservation biology literature recommendations and the results of the viability curve analysis. These thresholds apply to the estimated long-term geometric mean natural origin spawner abundance, and the viability report indicates that the threshold should meet with a reasonable level of confidence.

The viability report does not provide specifics on either "long term" or "reasonable," but suggests that at least 12 years of data are required and that simply observing a point estimate above a given threshold is not sufficient (i.e., the metric should be some statistical confidence limit.) The thresholds used in this analysis are presented in Figure 2 and Table 4. These thresholds differ from the thresholds presented in the viability report because newer estimates of population variability based on inclusion of additional data from Washington suggested a revision of the thresholds (see Appendix D). MAT evaluations are included in the population summary tables using a simple color coding as described in Table 2.

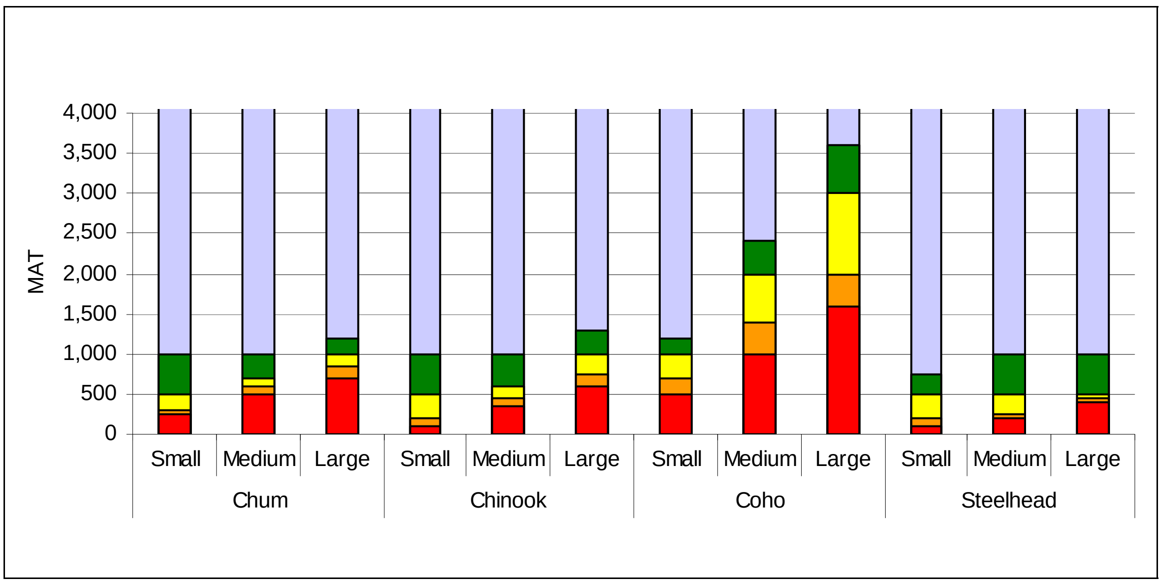

Figure 2: Abundance thresholds for population persistence categories by species and watershed size. The red, orange, yellow, green and blue bars show the ranges for persistence categories 0, 1, 2, 3, and 4, respectively. Figure data are shown in Table 4.

Table 4: Population abundance relative to persistence category. Data are graphed in Figure 2.

Species	Size Category	Persistence Category				
		0	1	2	3	4
Chum	Small	<250	250-300	300-500	500-1,000	>1,000
	Medium	<500	500-600	600-700	700-1,000	>1,000
	Large	<700	700-850	850-1,000	1,000-1,200	>1,200
Chinook	Small	<100	100-200	200-500	500-1,000	>1,000

	Medium	<350	350-450	450-600	600-1,000	>1,000
	Large	<600	600-750	750-1,000	1,000-1,300	>1,300
Coho	Small	<500	500-700	700-1,000	1,000-1,200	>1,200
	Medium	<1,000	1,000-1,400	1,400-2,000	2,000-2,400	<2,400
	Large	<1,600	1,600-2,000	2,000-3,000	3,000-3,600	>3,600
Steelhead	Small	<100	100-200	200-500	500-750	>750
	Medium	<200	200-250	250-500	500-1,000	>1,000
	Large	<400	400-450	450-500	500-1,000	>1,000

Viability Curves

This section contains a brief description of viability curve analysis, with a more detailed description available in the TRT viability report (McElhany et al. 2006a). Appendix D describes some modifications to TRT report viability curve methodology that apply to this status evaluation. The viability curve approach developed out of efforts to establish recovery criteria for threatened salmon and steelhead populations and was first described in McElhany et al. (2003). A viability curve describes a relationship between population abundance, productivity and extinction risk, with all the points on the curve showing abundance and productivity combinations that generate the same risk (Figure 3). Populations with productivity and abundance combinations above (to the right) of the viability curve have a lower extinction risk than that of the curve, while those below (to the left) have a higher risk.

Relating abundance, productivity and extinction risk is accomplished using a simulation model with a stochastic hockey-stick recruitment function having terms for productivity, carrying capacity, recruitment variability, age structure, future harvest rate, and a reproductive failure threshold (RFT). To estimate extinction risk for any particular set of input parameters, we run the model thousands of times and look at the fraction of simulations that drop below a critical risk threshold (CRT[3]). To draw the curve, we look for combinations of productivity and capacity (abundance) that are associated with a given level of risk. Drawing the curve for any particular group of fish requires appropriate estimates of recruitment variability, age structure, future harvest rate, and RFT. Note that we do not estimate productivity and capacity to draw the curve – in the curve we explore a range of hypothetical abundances and capacities (abundances). The viability curve can be thought of as a target for population abundance and productivity. The viability curve itself is not a complete evaluation of population status.

[3] The term ‘critical risk threshold’ (CRT) replaces the viability report term of ‘quasi-extinction threshold’ (QET) as described in Appendix D.

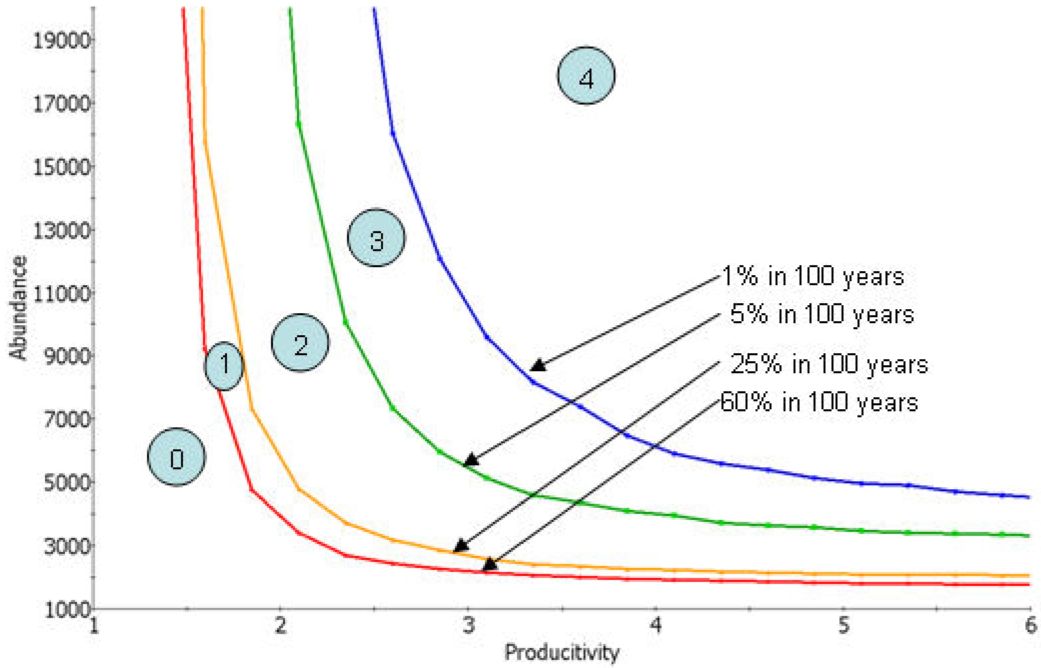

Figure 3: Viability curves showing relationship between risk levels and population persistence categories (example based on chinook curve). Each of the curves indicates a different risk level. The numbers in circles are the persistence categories associated with each region of the chart (i.e., the area between the curves). A population with a risk category 0 is described as a population that is nearly extinct and population with a risk category of 3 is described as "viable" (see Table 1).

In order to evaluate where any particular population is relative to the viability curve target, we must estimate the population's abundance and productivity. We used the MeanRS method described in the TRT viability report to estimate these parameters. Productivity is a measure of a population's resilience or tendency to return to higher abundance if the population declines to low abundance. Using the MeanRS method, this tendency is estimated as the geometric mean recruits per spawner for the brood years with the lowest half of spawner abundances. The abundance is estimated as the geometric mean recruitment over the time series. The characteristics of the MeanRS method compared other possible approaches are described in the viability report. The MeanRS methods are solidly based on the empirical data because they do not depend on extrapolation outside the observe ranges of recruitment and abundance.

Estimating a population's abundance and productivity requires input data on population spawner abundance, the fraction of hatchery origin spawners, harvest rates and the population age structure. All of these parameters are estimated with error – sometimes considerable error. We incorporate information about that error into our analysis by using a Monte Carlo approach of simulating many equally plausible data sets based on our understanding of the measurement errors and then calculating the MeanRS output for each simulated data set. This gives a distribution of possible abundance and productivity combinations for the current state of the population, which we present in the form of probability contours (a.k.a. "blobs") (Figure 4). We used the Salmon Population AnalyZer (SPAZ) computer program to generate viability curves and the current status distribution contours (McElhany et al. 2006b).

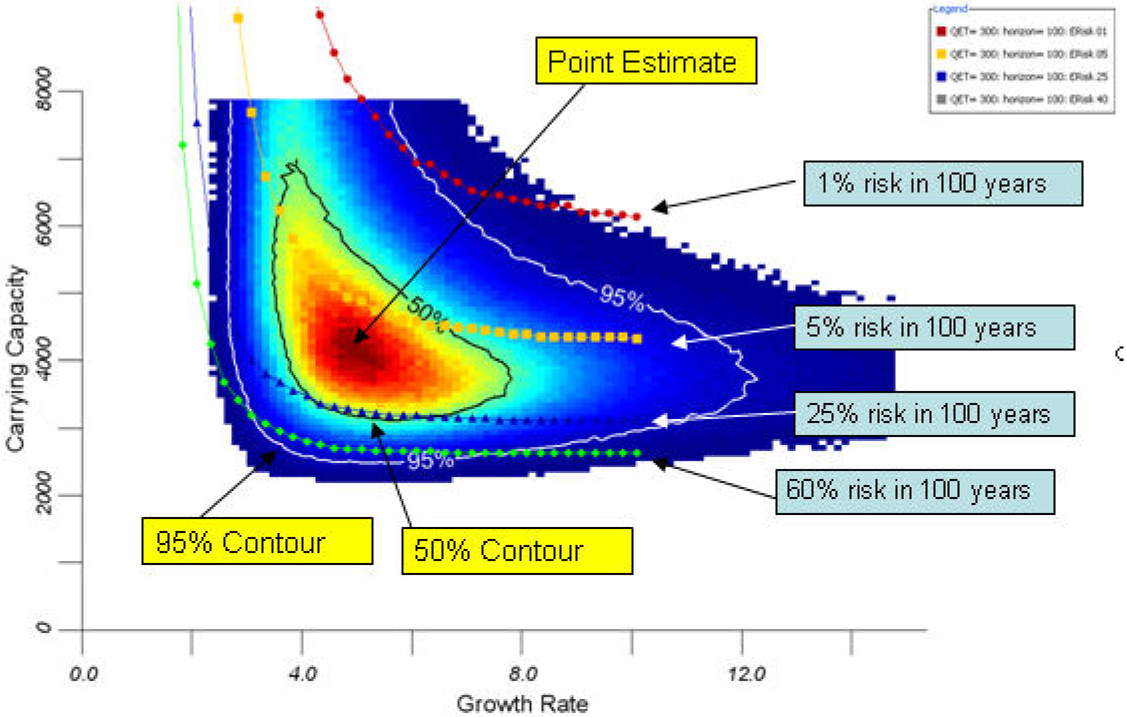

Figure 4: Example of current status contours combined with viability curves. In this example, the point estimate of the population indicates a persistence category of 2 (i.e., between 25% and 5% viability curves). To ensure at least a 50% chance that the population exceeds a given viability curve we would examine the 50% contour, which in this example suggests the population is in persistence category 1 (the bottom of the 50% contour is between the 40% and 25% viability curves).

CAPM Viability Model

Where appropriate time series were available, we also analyzed population viability using an extinction risk model that makes explicit use of information available over the recent past. This model, CAPM (Conservation Assessment and Planning Model) and its interpretation are described in Appendix E. A summary is provided here.

CAPM is a population viability model developed to assist salmonid conservation and recovery planning in Oregon. With the ability to define a wide range of possible future conditions the model lends itself to assessing both the likelihood of population extinction should conditions remain unchanged and also the likelihood of population extinction should these conditions change in response to implementation of successful recovery strategies. As is characteristic of all viability models, CAPM attempts to mimic the stochastic nature of population recruitment for a future period of time (e.g., the next 100 years). Simulations of this natural process are the basis for estimating probabilities of extinction, or in this case abundance less than CRT.

Although mechanically similar to other population viability models, several features of CAPM are unique. First, rather than using only one recruitment model to simulate population recruitment, CAPM uses three. It was assumed that in doing so, the adverse consequences of case-by-case inaccuracies of data fits to a particular recruitment function could be reduced. Secondly, in addition to the spawner abundance variable, all recruitment equations incorporate an independent index of environmental conditions. This second variable, called SNEG, was based on a 7-year moving average of high elevation maximum snow depth (see Appendix E). Inclusion of this variable not only improved recruitment model accuracy, but also had the effect of substantially reducing temporal autocorrelation of recruitment model residuals.

Another unique feature was that a probability of extinction was calculated for each set of recruitment function parameters estimated via the bootstrap process. This bootstrapping procedure was used to repeatedly sample each population data set (generally 200 times). A regression analysis was then performed on each data set sample using a nonlinear regression routine. This meant that for every bootstrap sample an estimate of recruitment equation parameters and associated standard deviations were generated for all three recruitment curves. Probabilities of the population becoming less than CRT levels were then estimated for each sample of parameters. The primary purpose of this extended bootstrap procedure was to better understand the range and magnitude of possible errors in estimating recruitment equation parameters. However, as a result of this process, the outputs from CAPM are not a single probability of CRT estimates, but rather distributions of CRT probabilities that can be visualized as frequency histograms. The median and percentile values from these distributions are used to characterize the population viability.

PopCycle Stochastic Stock-Recruitment Model

Oregon WLC populations were also evaluated using a generic risk analysis model (Popcycle) developed for application to Washington lower Columbia River salmon populations and fisheries. The model is described in more detail in Appendix F, but a brief summary is provided here. Popcycle is a simple stochastic stock-recruitment population model that projects annual run size, spawning escapement, and harvest numbers and frequency distributions based on user-defined population functions and parameter values. A simple interface page facilitates model use and review of results. The model includes optional inputs to apply fishing rates in each year to calculate harvest and fishery effects on population dynamics. Optional inputs are also included for analysis of demographic effects of natural spawning by hatchery fish based on inputs for hatchery releases, release to adult survival, and rates of natural spawning by hatchery fish. The model is built in Microsoft Excel using Visual Basic. In contrast to the viability curve and CAPM viability curve analyses, PopCycle estimates only expected averages and frequency distributions, and does use parameter uncertainty estimates to estimate confidence or plausibility regions about expected results. However, the simpler model formulation and ease of use of PopCycle facilitates exploration of population dynamics and model sensitivity to differences in population parameters and key assumptions.

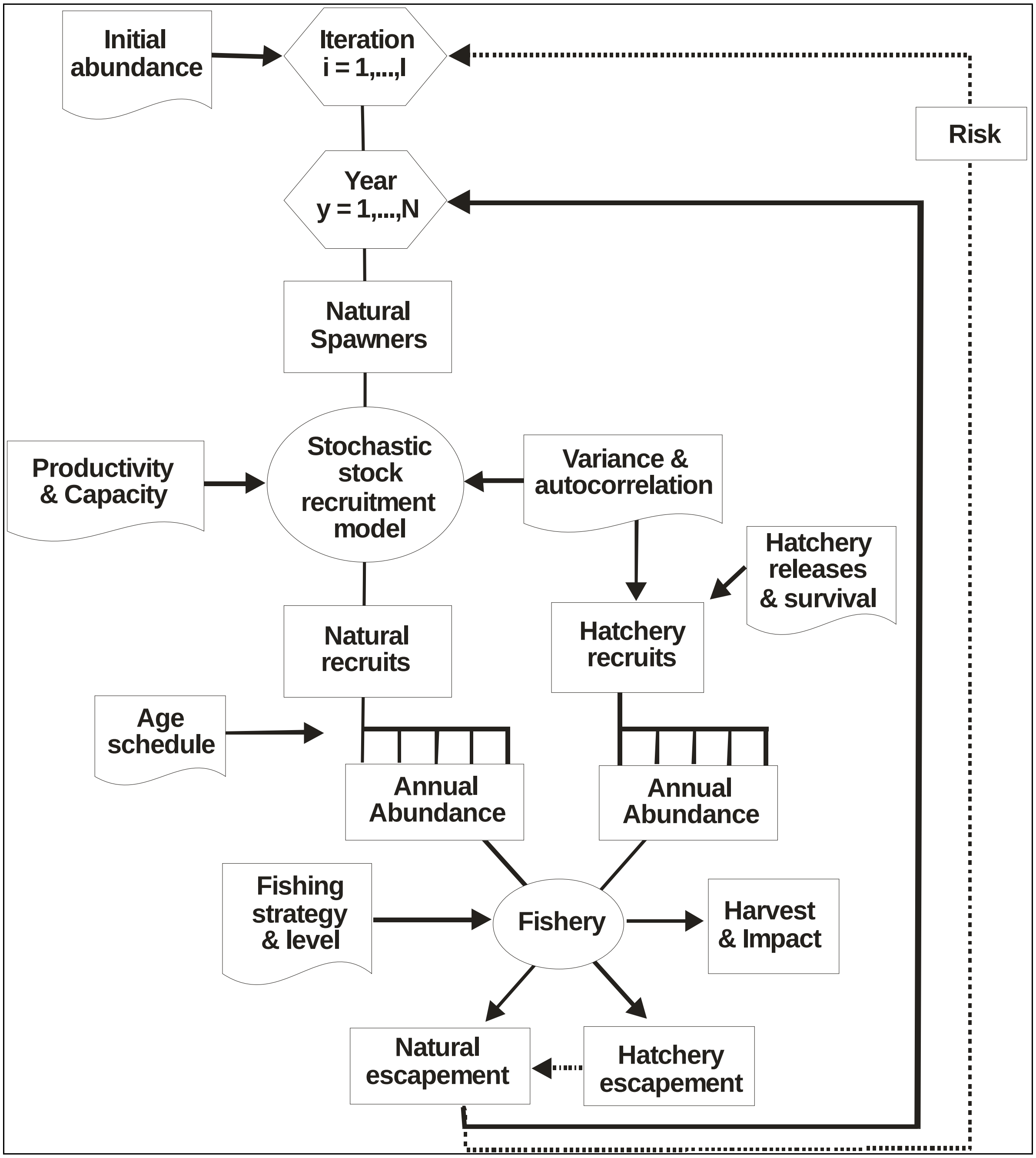

Figure 5: PopCycle model algorithm.

Viability Curve and CAPM Model Summary Table

A summary table from the viability curve, CAPM, and PopCycle analyses is provided for each population that had adequate data. This table provides estimates of the probability that the population is in each of the persistence categories. As is common with all extinction risk forecasts, the accuracy of these probability assessments depends upon the validity of the underlying model assumptions. For the viability curve analysis this statistic is estimated by integrating the fraction of the probability contour above a given viability curve. For example, if we are looking at a 5% extinction risk in 100 year viability curve and if the probability contour (“blob”) for a population is completely below the curve, the probability that the risk is less than 5% in 100 years is zero. Conversely, if the probability contour is completely above the curve, the probability that the risk is less than 5% in 100 years is 1. If the viability curve goes through the probability contour, there is some probability between zero and one that the risk is less than 5% in 100 years; the more of the contour above the curve, the closer to one. This gives a measure of how sure we are that the population is above a given risk threshold and is a quantification of the visual assessment of what fraction of the probability contour lies above a given viability curve. For the CAPM model, the probability that the population is above a given threshold is calculated as described in Appendix E.

Combining Abundance and Productivity Information

Combining information from the various summary statistics and extinction risk models was done using professional judgment rather than a quantitative algorithm. In general, all the information points to a similar conclusion about population status, so the overall result is fairly obvious. However, in some cases, the different analyses suggest different conclusions. In these cases, we discussed the alternative interpretations and generally indicate the increased ambiguity about the population’s status by increasing the amount of uncertainty displayed in the diamond figures used to show conclusions on population status.

Spatial Structure

Overview

Spatial structure of Oregon populations was assessed based on the application of basic principles and a coho example developed by the TRT (McElhany et al 2006a). Quantitative metrics address two of the key spatial structure issues: 1) total quantity of available habitat and 2) spatial distribution of accessible habitat. In addition, quantitative scores were adjusted based on qualitative considerations including habitat quality and life-stage specific spatial distribution. Adjustments are discussed in the text narrative for each population.

Spatial structure evaluations were primarily based on the evaluation of maps of accessible habitat developed in the Oregon WLC habitat atlas (Maher et al. 2005). These maps have some important limitations. They were developed using existing blockage databases and species-specific gradient thresholds. There is no consideration of habitat quality; the maps simply provide an estimate of where fish could go, not necessarily where the habitat can support fish or where fish currently are. Consequently, the maps likely overestimate current and historical use, perhaps substantially (see habitat atlas for discussion and comparison to potential use maps). The maps are also only as good as the blockage databases, which may contain some errors. In addition, the maps only address adult accessibility; they do not describe life stage specific habitat spatial distribution, such as the arrangement of habitat for juvenile rearing. Despite these caveats, the maps can provide useful information and as they where developed using a consistent protocol comparing current and historical potential distribution for the entire ESU/DPS, we have based the analyses on the maps. However, we do not rely solely on these maps and incorporate additional information in the final spatial structure evaluations. The refinement of maps describing current and historical habitat from a fish perspective should be a research priority.

Quantitative Metrics

A primary concern in evaluating spatial structure is whether the population has access to a sufficient quantity of habitat to survive catastrophic events. A viable population should not "put all its eggs in one basket." The TRT developed metric and threshold guidelines that are a function of both the amount of historically accessible habitat and the size of the watershed (Table 5). These thresholds are used in this current status evaluation. Historical accessibility is considered the appropriate reference value because the historic structure was assumed to be viable and the greater the deviation from the historical condition, the greater the risk. The guideline thresholds are a function of the watershed size because a smaller population is likely to be at a greater risk from a smaller relative loss than a larger population. These guidelines are not based on any quantitative model, but rather on the professional judgment of the TRT. The TRT included quantitative guidelines, not because they believed there is any quantitative precision in this assessment, but instead to provide a transparent presentation of how they view the relationship between the loss of habitat access and extinction risk.

Table 5: Guideline thresholds for relationship between persistence category and percent loss in accessible habitat.

Persistence Category	Watershed Size		
	Small	Medium	Large
0	50-100	60-100	75-100
1	25-50	40-60	50-75
2	15-25	20-40	25-50
3	5-15	10-20	15-25
4	0-5	0-10	0-15

Another key consideration is the spatial distribution of habitat loss. The TRT hypothesized that loss of access to an entire stream branch poses a greater risk to a population than a number of smaller losses that would produce the same total amount loss. The relative size of a stream branch loss can be evaluated as the percent of loss caused by each blockage. We apply the following guideline from the TRT viability report:

> If the largest single blockage results in a >10% loss for small watersheds or a >15% loss for medium and large watersheds, the persistence category is reduced by 0.5.

For example, a persistence category 3 would become a 2.5. This metric addresses some of the aspects of the arrangement of the loss in space, but is not a complete evaluation. The natural dendritic structure or "branchiness" of a stream and the exact location of the blockage can also be important. This aspect of spatial structure is difficult to quantify and set a priori thresholds. Therefore, we applied a qualitative evaluation based on consideration of the actual access maps.

Qualitative Spatial Considerations

In addition to the two spatial structure metrics described above, we applied adjustments to the scores based on qualitative considerations, which are discussed in the text narrative for each population. Qualitative factors considered are habitat quality and life-stage specific spatial distribution.

Diversity

The diversity evaluation follows the basic methods and approach of the viability report (McElhany et al. 2006a). However, the evaluation is organized slightly differently, with analyses divided into the following factors:

- Life history traits
- Effective population size
- Impact of Hatchery Fish
- Anthropogenic mortality
- Habitat diversity

Where data are available, we evaluate and assign a persistence score for each of these five diversity factors. These scores are then combined into a single diversity rating for each population. The overall diversity persistence score is estimated using expert judgment and considering all the individual diversity factor scores (i.e., there is no quantitative algorithm for combining the diversity factors). It should be noted that data are frequently insufficient to adequately evaluate one or more of the diversity factors.

Life History Traits

Measurable life history traits considered in our analyses include: 1) timing of return to fresh water, 2) age at maturation, 3) spawn timing, 4) outmigration timing, 5) smoltification timing, 6) developmental rate, 7) egg size, 8) fecundity, 9) freshwater distribution, 10) ocean distribution, 11) size at maturation and 12) timing of ascension to the natal stream. To assigned persistence scores for life history traits we generally relied on the risk guidelines developed by the Interior Columbia TRT (IC-TRT 2005)and modified by the WLC-TRT (McElhany et al. 2006a) (Table 6).

Table 6: Preliminary criteria describing risk levels associated with major life history strategies and change in phenotypic characteristics (from ICRTRT 2005).

Factor	Risk Level (Viability Score)			
	Very Low (4)	**Low (3)**	**Moderate (2)**	**High (1)**
Distribution of major life history strategies within a population.	No evidence of loss in variability or change in relative distribution	All historical pathways present, but variability in one reduce or relative distributions shifted slightly.	All historical pathways present, but significant reduction in variability or substantial change in relative distribution.	Permanent loss of major pathway.
Reduction in trait variability of traits, shift in mean value of trait, loss of traits	No evidence of loss, reduced variability, or change in any trait.	Evidence of change in mean or variability in 1 trait.	Loss of 1 trait or evidence of change in mean and variability of 2 or more traits.	Loss of 1 or more traits and evidence of change in mean and variability of 2 or more traits.

Effective Population Size

One of the indirect measures of diversity is effective population size. A population at chronic low abundance or experiencing even a single episode of low abundance can be at higher extinction risk because of loss of genetic variability, inbreeding and the expression of inbreeding depression, or the effects of mutation accumulation. The viability report identifies increased risk as significant when the effective population size drops below about 500. The relationship between effective population size, census population size, and estimated persistence category are shown in Table 7.

Table 7: Relationship between effective population size, census population size (in parentheses) and estimated persistence category. From (McElhany et al. 2006a).

Effective Population Size	Persistence Category				
	0	1	2	3	4
Ne < 12.5 (*N<25*)	x				
12.5<Ne < 25 (*25<N<50*)		x			
25<Ne <125 (*50<N<250*)			x		
125<Ne<500 (*250<N<1000)*				x	
500 < Ne (*1000<N*)					x

Impact of Hatchery Fish

Interbreeding of wild populations and hatchery origin fish can be a significant risk factor to the diversity of wild populations because of the potential genetic dissimilarities between these two groups of fish. We evaluate this risk based on two characteristics of the problem, the proportion of hatchery fish within the natural spawning population and the genetic similarity of these hatchery fish to the wild population. Our assumption is that the genetic risk to the wild population is greatest when the proportion of hatchery fish in the spawning population is high and their genetic similarity to the wild population is low. Conversely, the lowest risk occurs when the proportion of hatchery fish is low and they are genetically similar to the wild population.

We use three different methods to evaluate the potential impact of hatchery fish: 1) Proportion of Natural Influence (PNI) modeling for domestication in integrated hatchery programs; 2) Thresholds for introgression with out-of-stratum hatchery broodstocks; and 3) Synthetic approach based on fraction of hatchery origin spawners.

Domestication PNI Modeling

For interactions with locally derived hatchery brood stocks, we considered the hatchery and natural spawners as part of a potential "integrated" population. The approach to assessing risk is based on evaluating the Proportion of Natural Influence (PNI) index, a measure of potential domestication. The index is the ratio of the proportion of natural origin fish in the hatchery brood stock and the proportion of hatchery origin fish on the natural spawning grounds (Figure 6). The lower the PNI, the greater the population risk from domestication, because the majority of the breading takes place in the hatchery. Following the viability report, we related the PNI to potential fitness loss (Figure 7) and associated the fitness loss with a population persistence category (Table 8). As a precautionary measure the fitness loss measure is based on the *lower* confidence bound. In many cases hatcheries are run as "isolated" programs with no known inclusion of naturally-produced spawners into the hatchery broodstock, although there is generally some straying of hatchery origin fish onto the natural spawning grounds. Isolating the hatchery broodstock produces a PNI of 0, regardless of the proportion of hatchery fish on the natural spawning grounds. In these situations, the PNI approach is not applicable, and we rely on the other two methods for evaluating hatchery impacts on diversity.

The PNI model was developed to estimate the potential decline in fitness due to selection for hatchery conditions rather than natural conditions (aka domestication) and does not directly address the other possible consequences of hatchery/wild interaction. Domestication effects were modeled using empirical estimates from studies and estimates based on the professional opinion of a number of fisheries scientists. As such, the PNI model represents a work in progress and it is likely that further refinements will be made as more information on hatchery effects becomes available. While the focus was on "domestication" it is likely that non-domestication effects were incorporated into estimates of decline in fitness. Other effects include competition, predation, non-genetic domestication (behavioral and developmental), disease, etc. The impacts of these effects will generally be reflected in the assessment of population productivity, which integrates all factors affecting mortality. However, the PNI metric does provide some information on these factors, since the hatchery effects are largely a function of the fraction of hatchery origin fish on the spawning grounds, which is one factor in the PNI metric. We present information on how the domestication thresholds relate to the fraction of hatchery origin fish in Table 8. Often, populations with hatchery fish will show poor productivity estimates at hatchery fractions lower than those that cause significant domestication effects because of how hatchery fish enter the productivity equations (i.e., hatchery fish on the spawning grounds count as spawners, but not natural origin recruits.).

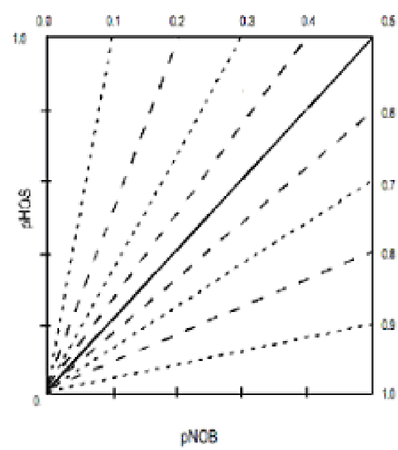

Figure 6: Proportion of Natural Influence (PNI) relationship between percent Hatchery Origin Spawners (pHOS) and percent Natural Origin Broodstock (pNOB). The numbers are the outside of the graphic represent the PNI score. Populations located toward the lower right corner are at relatively lower risk of domestication and populations located toward the upper left corner are at relatively higher risk.

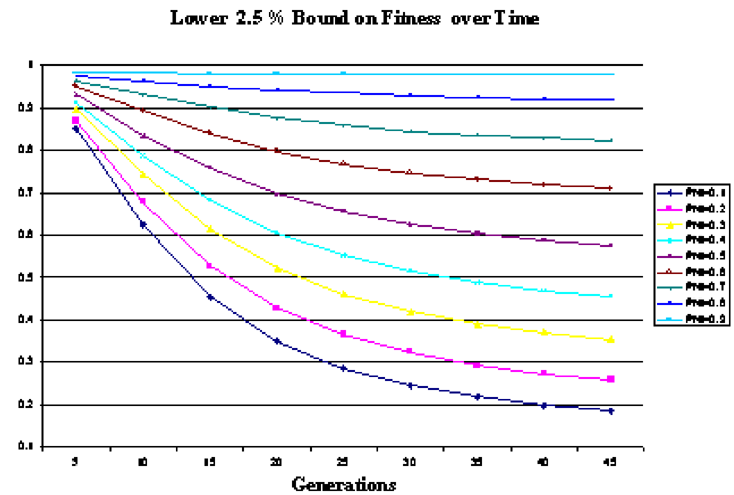

Figure 7: Influence of PNI on overall population fitness over time (generations). Fitness estimates are based on the lower 2.5% bound of the confidence intervals. (Graphic from C. Busack, WDFW)

Table 8: Loss of fitness over time (from Figure 7) and diversity score for populations affected by artificial propagation programs.

Percent Fitness Loss	***Diversity Score***	***PNI at 25 generations***	***pHOS at 50% pNOB***
0.0 -2.5	*4*	*0.9*	*10%*
2.5 – 5.0	*3.5*	*0.85*	*15%*
5.0 – 10.0	*3.0*	*0.8*	*20%*
10.0 – 15.0	*2.5*	*0.7*	*30%*
15.0 – 25.0	*2.0*	*0.6*	*40%*
25.0 – 45.0	*1.5*	*0.5*	*50%*
45.0 – 65.0	*1.0*	*0.4*	*60%*
65.0 – 85.0	*0.5*	*0.3*	*70%*
> 85.0	*0*	*0.1*	*90%*

Introgression Thresholds for Out of Stratum Stocks

If there is interbreeding between a natural population and hatchery or wild stocks from outside the stratum, the effects are not as easily estimated by the PNI/Hatchery Domestication approach. The genomes of the populations are likely to have differences not caused solely by domestication to the hatchery environment, but will also exhibit differences from local adaptation to other basins. We are concerned in this risk factor not just about hatchery fish from outside the stratum but also artificially high interbreeding with natural origin fish from outside the stratum. Although some interbreeding of fish from different strata occurs naturally, some human activities (like altering passage at Willamette Falls) can create elevated levels of interbreeding. The potential for reduced viability is greater for out of ESU/DPS interbreeding than for out of stratum, but within ESU/DPS, interbreeding. The relationship between stray rates and risk categories is shown in Table 9 (from McElhany et al. 2006a). The hatchery introgression tables are also used in situations where a local hatchery is operated as an isolated program—without the inclusion of naturally-produced fish into the broodstock. In these situations the PNI metric always produces a PNI value of zero, regardless of the hatchery stray rate onto the natural spawning grounds.

Table 9: Influence of non-local origin fish strays on the diversity status of the local population. For the diversity metric, strays are only considered if there is evidence of interbreeding, the effective stray rate. Where both within ESU and out-of-ESU strays are present, a weighted mean (using the proportional occurrence of both types of strays) should be calculated.

Diversity Score	**0**	**1**	**2**	**3**	**4**
Within ESU/Out of Strata Effective Stray Rate (m)[1]					
75% < m	x				
30% < m < 75%		x			
10% < m < 30%			x		
5% < m < 10%				x	
m < 5%					x
Out of ESU Effective Stray Rate (m)[1]					
50% < m	x				
20% < m < 50%		x			
5% < m < 20%			x		
2% < m < 5%				x	
m < 2%					x

For example, if 10% of the natural spawners in a basin were from a different strata within the ESU, and 5% were from outside of the ESU, the stray metric would be calculated as:
(.67) * (2 [w/i ESU@20%]) + (.33) * (3 [out of ESU@10%]) = 2.3.
Remember that the stray rate is based on the proportion of effective (spawning) non-local fish.

Synthetic Approach

The synthetic approach considers both domestication from integrated programs and introgression from out of strata fish within a single framework based on the proportion of hatchery origin spawners (Ph). This method was developed for this report to provide a streamlined metric based on empirical estimates of hatchery fish induced productivity declines (Chilocte 2003), rather than modeling genetic processes (i.e. PNI). To formulize the relationship between proportion of hatchery spawners and a persistence score we have adopted a modified version of the rating system in Table 9. This rating system differs from Table 9 in two important ways. First, rather than specifying an effective migration rate (m), the approach here is based on the proportion of hatchery origin spawners within the basin shared by wild fish. No distinction is made for spatial or temporal segregation of hatchery and wild spawners, only presence is counted. This is an adjustment based on the reality that in most cases it is exceedingly difficult to measure effective migration rate (m). In contrast, Ph can be determined easily if a means to discriminate between hatchery and wild fish is available and the data are collected.

Secondly, the rating assumes the baseline hatchery stock has a low genetic similarity to the local wild population. However, if evidence suggests a moderate to high similarity between the hatchery and wild fish, then the persistence score is incremented by one. In contrast, if the hatchery stocks involved likely have a very low genetic similarity to the wild population, a decrement of one persistence score category is applied. A matrix display of this rating system is presented in Table 10.

The classification of the hatchery stocks into one of three similarity categories was made largely on the basis of broodstock origin and incorporation of wild fish into the hatchery spawning cycle. Where possible, genetic analysis of hatchery and wild populations was examined to estimate the degree of similarity. The 'very low' genetic similarity classification was reserved for those hatchery stocks whose origin was from outside of the stratum or the ESU. The ‘low’ classification was assigned to the hatchery stock if its origin was within the same stratum. The ‘moderate’ classification was used for those hatchery stocks that were derived from the local wild population and for which more than 50% of the spawners used to each generation for hatchery broodstock were wild fish.

Table 10: Persistence scores for different proportions of hatchery fish within naturally spawning populations of mixed hatchery and wild fish.

Presumed Genetic Similarity to Wild Population	Proportion of Hatchery Fish (Ph) in Natural Spawning Population	Persistence Score				
		0	1	2	3	4
Moderate (Broodstock from same wild population and > 50% of the hatchery broodstock are wild fish)	Ph > 0.75		x			
	0.75 > Ph > 0.30			x		
	0.10 > Ph < 0.30				x	
	0.05 > Ph > 0.10					x
	Ph < 0.05					x
Low (Broodstock source is from same stratum or from same wild population but < 50% wild fish used as hatchery broodstock)	Ph > 0.75	x				
	0.75 > Ph > 0.30		x			
	0.10 > Ph < 0.30			x		
	0.05 > Ph > 0.10				x	
	Ph < 0.05					x
Very Low (Broodstock source is from different stratum or ESU)	Ph > 0.75	x				
	0.75 > Ph > 0.30	x				
	0.10 > Ph < 0.30		x			
	0.05 > Ph > 0.10			x		
	Ph < 0.05				x	

Anthropogenic Mortality

Anthropogenic mortality (e.g., from harvest or habitat alterations) is unlikely to be selectively neutral. The susceptibility to mortality will differ depending on size, age, run timing, disease resistance or other traits. The TRT developed general guidelines for relating anthropogenic mortality to extinction risk category (Table 11). Different types of mortality will certainly have different selective effects and therefore different impacts on extinction risk and these guidelines are only a starting point for the consideration of this risk.

Table 11: Relationship between anthropogenic mortality and persistence category.

Anthropogenic Mortality Rate (%)[1]	Persistence Category				
	0	1	2	3	4
> 95%	x				
80%-95%		x			
45%-80%			x		
20%-45%				x	
< 20%					x

[1] Includes anthropogenic factors that could potentially result in non-random mortality (harvest, hydro operations, etc.). Adjust +/- depending on the presumed strength of selection (e.g., seasonal temporal selection, gill net size selection).

Habitat Diversity

Habitat characteristics have clear selective effects on populations and changes in habitat characteristics are expected to eventually lead to genetic changes through selection for locally adapted traits (although habitat changes can occur at a much faster rate than genetic changes, as a result the fitness of a population is rarely optimized as it adjusts to a constantly moving target). Therefore, change in habitat diversity is a reasonable surrogate for evaluating potential changes in population diversity. In assessing risk associated with altered habitat diversity, we take the historical diversity as a reference point here and throughout this evaluation. The topic is discussed elsewhere in this report. In the viability report, we developed two simple habitat diversity metrics. One metric is based on the distribution of accessible habitats at different elevations and the other is based on the distribution of accessible habitats of different stream size. The viability report describes how these metrics are related to the persistence categories and provides a table of habitat diversity scores in the viability report Appendix I.

Integrating the Diversity Factors

Few of the population diversity assessments contained sufficient information on each of the factors to utilize a single mathematical algorithm to integrate the scores. For each population, those factors that were scored were averaged. Consideration was given to the quality of data used to determine each factor. Information on data quality is given in the diversity summary for each population.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 2: Lower Columbia Chinook

September 2007

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Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

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I. ESU Overview and Historical Range

Based on TRT analysis, the Oregon portion of the LRC ESU historically contained 12 populations: 9 fall run chinook (tules); 1 late fall run chinook (brights); and 2 spring run chinook (Figure 1 and Figure 2). The stratum composition is shown in Table 1. The Lower Gorge and Upper Gorge populations occur in both Washington and Oregon. In this report, we describe only the status of the Oregon portion of these two populations.

In general, naturally-produced chinook in the lower Columbia basin are thought to be substantially reduced compared to historic levels (Myers, et al. 1998). Coinciding with this decline in total abundance has been a reduction in the number of functioning wild populations, particularly in the case of Tule fall chinook. In addition the significant presence of stray hatchery fish is thought to be common throughout most of the range. Currently, only 2 of the historical 12 populations in the ESU show substantial natural production.

The presentation of our assessment begins with three sections, each of which evaluates one of the viability criteria (i.e., abundance/productivity, spatial structure, and diversity). We have pooled the results from these sections in a synthesis section for each population, where we derive a status rating for each population. We end our presentation with an interpretation of the population results in terms of the overall status of Oregon's LCR chinook populations.

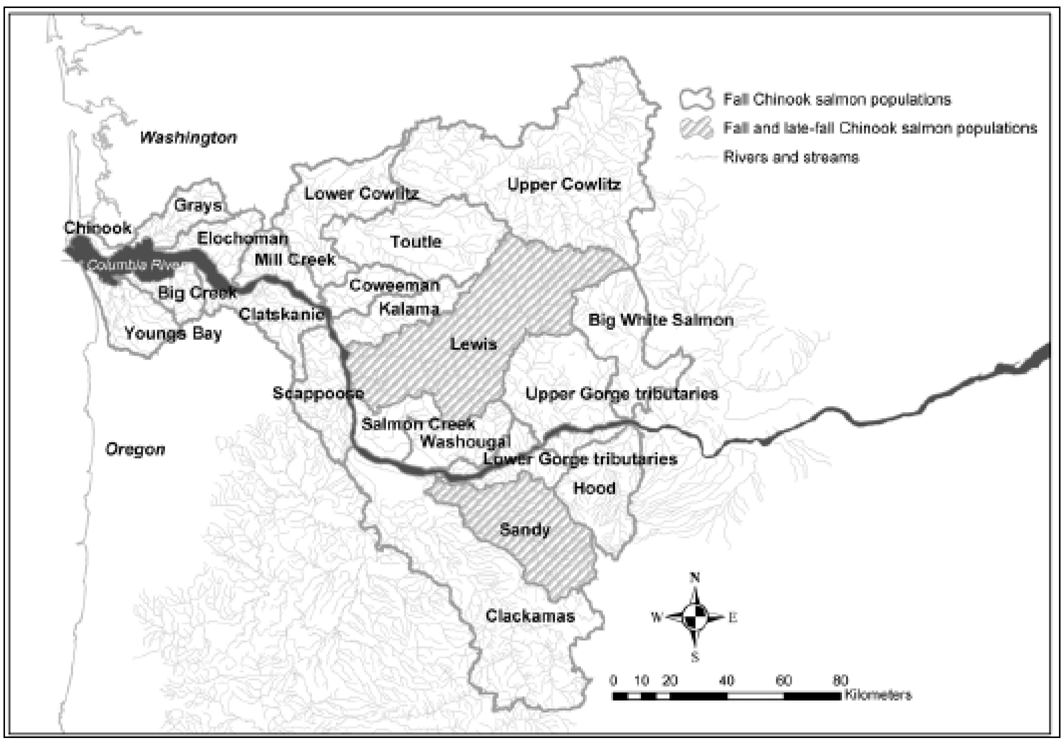

Figure 1: Map of LCR fall chinook salmon populations.

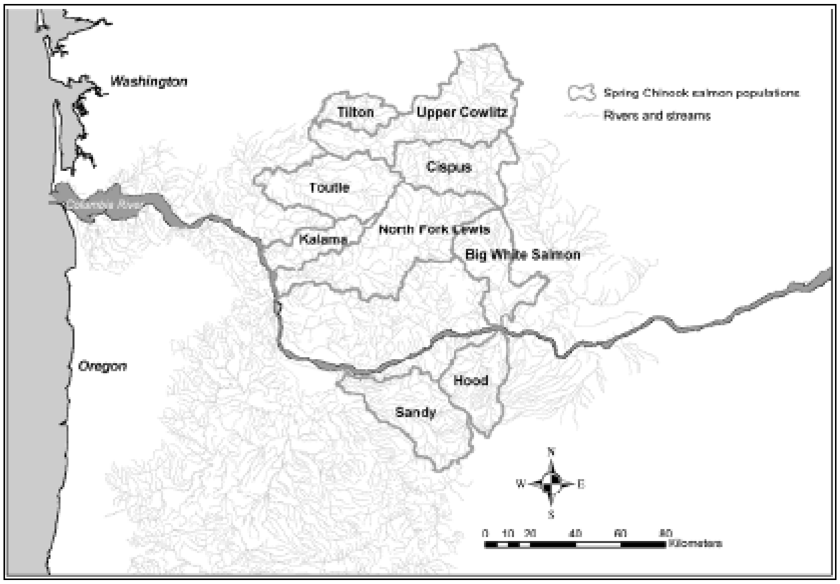

Figure 2: Map of LCR spring chinook salmon populations.

Table 1: Stratum composition of Oregon LCR chinook populations. Each ecozone and life history combination is a separate stratum, which results in six strata in this ESU.

EcoZone	Life History	Populations
Coastal	Fall (tule)	Youngs Bay Big Creek Clatskanie Scappoose
Cascade	Fall (tule)	Clackamas Sandy
	Late Fall (bright)	Sandy
	Spring	Sandy
Gorge	Fall	Lower Gorge Upper Gorge Hood
	Spring	Hood

II. Abundance and Productivity

A&P – Youngs Bay Fall Run (Tule)

A time series of abundance adequate for quantitative viability analysis is not available for the Youngs Bay fall chinook population. A time series of fish per mile for this population was included in the 2005 BRT status update (Good et al. 2005) (Figure 3), but the time series does not distinguish between hatchery and natural origin fish, so it is not very informative about the status of the natural population. However, the time series does indicate that no fish (of either hatchery or natural origin) were observed during some recent years, suggesting that the number of fish can get relatively low (assuming the survey was reasonably efficient at finding fish). A time series of abundance was analyzed for the nearby Clatskanie fall chinook population and that analysis indicated that the Clatskanie is at a high risk of extinction. Conditions in Youngs bay are not expected to be any more favorable to fall chinook than in the Clatskanie. In fact, conditions may be less favorable because of the presence of a large number of out of strata origin hatchery fish (discussed in the diversity section). Data in the 2005 Oregon Native Fish Status Report show a geometric mean return abundance for this populations in years 2000-2004 of 37 fish per mile (ODFW 2005). The report states that the "existing run is likely to be primarily hatchery fish." There is no abundance and productivity evidence supporting the existence of a viable natural origin population in Youngs Bay, and comparisons with populations in similar habitats suggest the population is at significant risk. The 2005 Oregon Native Fish Status Report listed this population as "failed" for abundance and productivity.

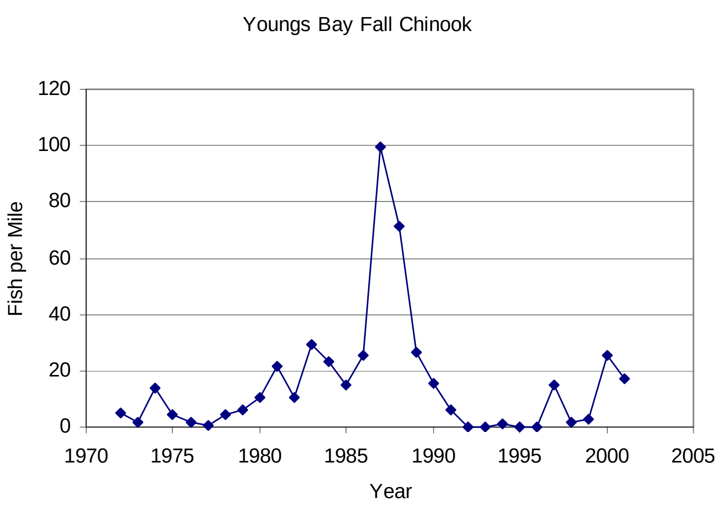

Figure 3: Youngs Bay chinook salmon per mile, 1972-2001.

A&P – Big Creek Fall Run (Tule)

A time series of abundance adequate for quantitative viability analysis is not available for the Big Creek fall chinook population. A time series of fish per mile for this population was included in the 2005 BRT status update (Figure 4), but the time series does not distinguish between hatchery and natural origin fish, so it is not very informative about the status of the natural population. However, the time series does indicate that very few fish (of either hatchery or natural origin) were observed during some recent years, suggesting that the number of fish can get relatively low (assuming the survey was reasonably efficient at finding fish). A time series of abundance was analyzed for the nearby Clatskanie fall chinook population and that analysis indicated that the Clatskanie is at a high risk of extinction. Conditions in Big Creek are not expected to be any more favorable to fall chinook than in the Clatskanie. In fact, conditions may be less favorable because of the presence of a large number of origin hatchery fish (discussed in the diversity section). Data in the 2005 Oregon Native Fish Status Report show a geometric mean return abundance for this population in years 2000-2004 of 413 fish per mile, but the report states that the "existing run is likely to be primarily hatchery fish" (ODFW 2005). There is no abundance and productivity evidence supporting the existence of a viable natural origin population in Big Creek and comparisons with populations in similar habitats suggest the population is at significant risk. The 2005 Oregon Native Fish Status Report listed this population as "failed" for abundance and productivity.

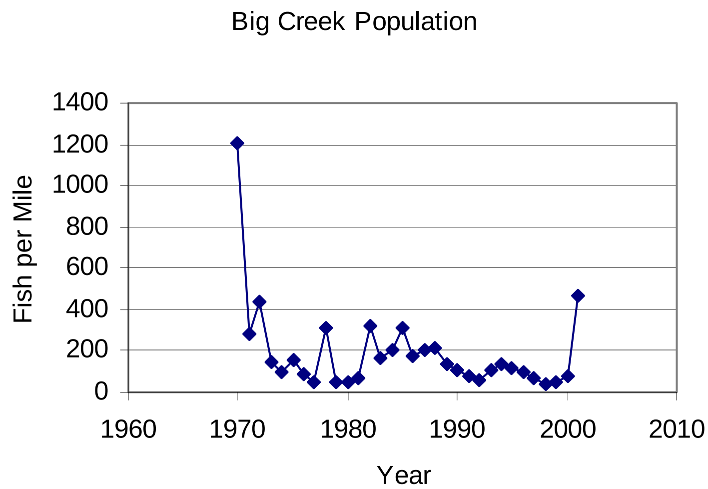

Figure 4: Big Creek chinook salmon per mile, 1970-2001.

A&P – Clatskanie Fall Run

Although there is likely to be substantial measurement error in the data, a time series of abundance was available for the Clatskanie fall chinook population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 5 to Figure 11 and in Table 2 and Table 5. These analyses suggest that this population is at substantial risk of extinction. As shown in the viability curve graphs, the population has been at a very low abundance of natural origin spawners. In more than half the years, the population was below 100 spawners, and in 9 of the years the abundance was less than 10 fish.

The viability curves suggest a relatively high productivity for this population. However, we believe that is likely a product of measurement error and does not reflect the true productivity of the population. With very low abundances, even small measurement errors in abundance estimates and hatchery fraction estimates or violations of the no migration assumptions will lead to erroneous (and upwardly biased) estimates of productivity. These analyses put the population in the very high risk category. The PopCycle model estimates a 56% risk level, which also puts it in the high risk category.

The CAPM model also indicates that the population is in the high risk category, with a median CRT risk probability of 53%. The escapement viability curve indicates that the population has very low chance of persistence if the pattern of harvest that occurred over the available time series were to continue (average harvest rate 66%). The 2005 Oregon Native Fish Status Report (ODFW 2005) states that the "existing run is likely to be primarily hatchery." However, in 2006 new information became available that the frequency of Fall chinook recovered during spawning surveys known to be hatchery fish as indicated from CWT recoveries was extremely low.

Expansion of these observations based on the CWT tagging rate of hatchery fish released from nearby hatcheries, indicated the likely fraction of all hatchery fish (with and without CWTs) within the Clatstkanie in recent years was in the range of 15%. The geometric mean natural origin spawners is 50 fish (Table 2), which is in the "extirpated or nearly so" minimum abundance threshold category. The 2005 Oregon Native Fish Status Report listed this population as "failed" for abundance and productivity.

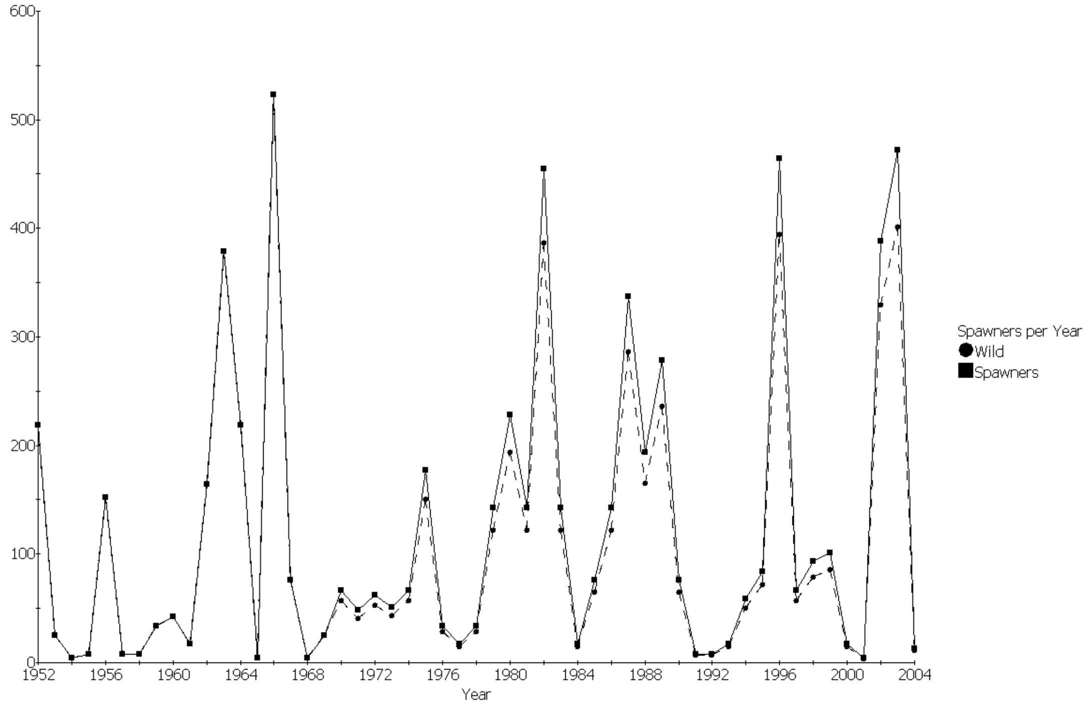

Figure 5: Clatskanie fall chinook abundance.

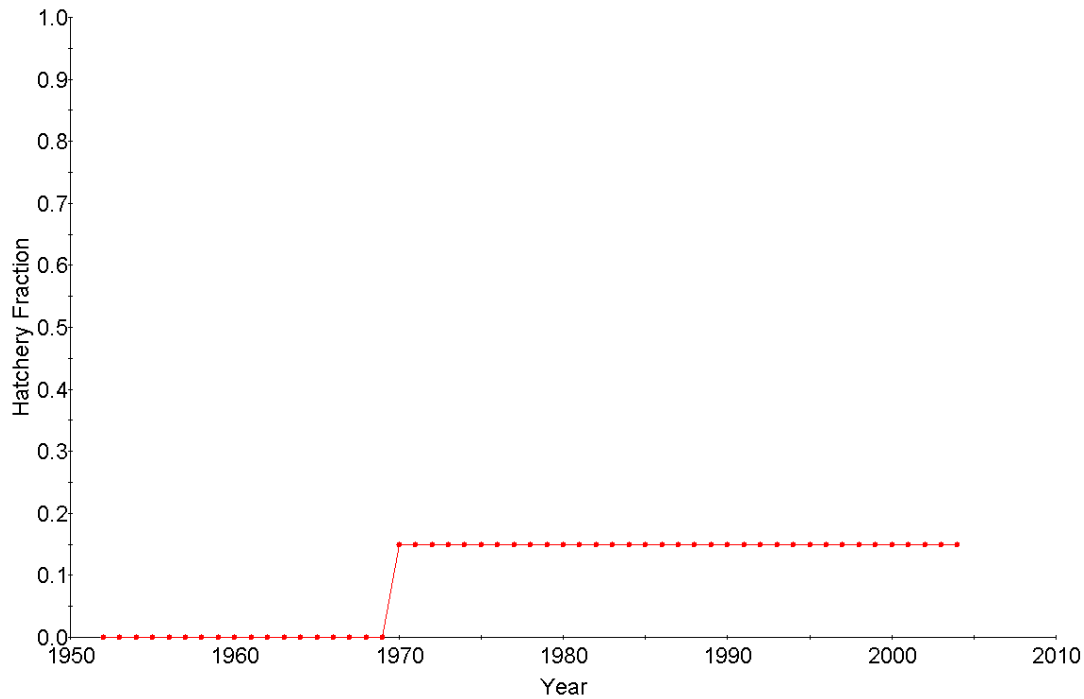

Figure 6: Clatskanie fall chinook hatchery fraction.

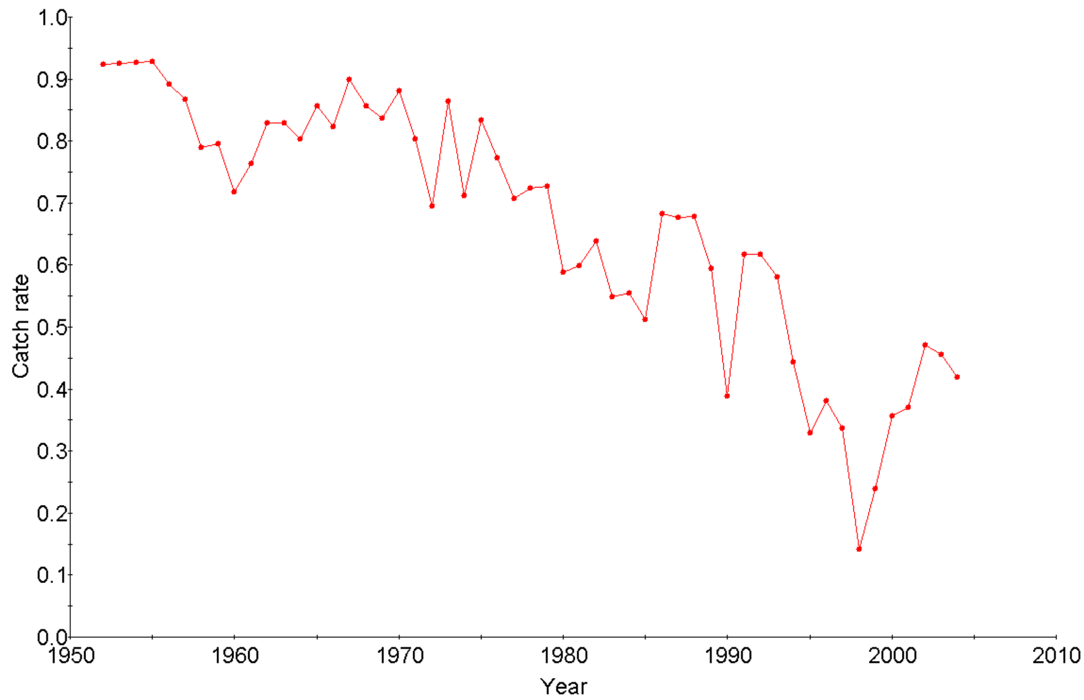

Figure 7: Clatskanie fall chinook harvest rate.

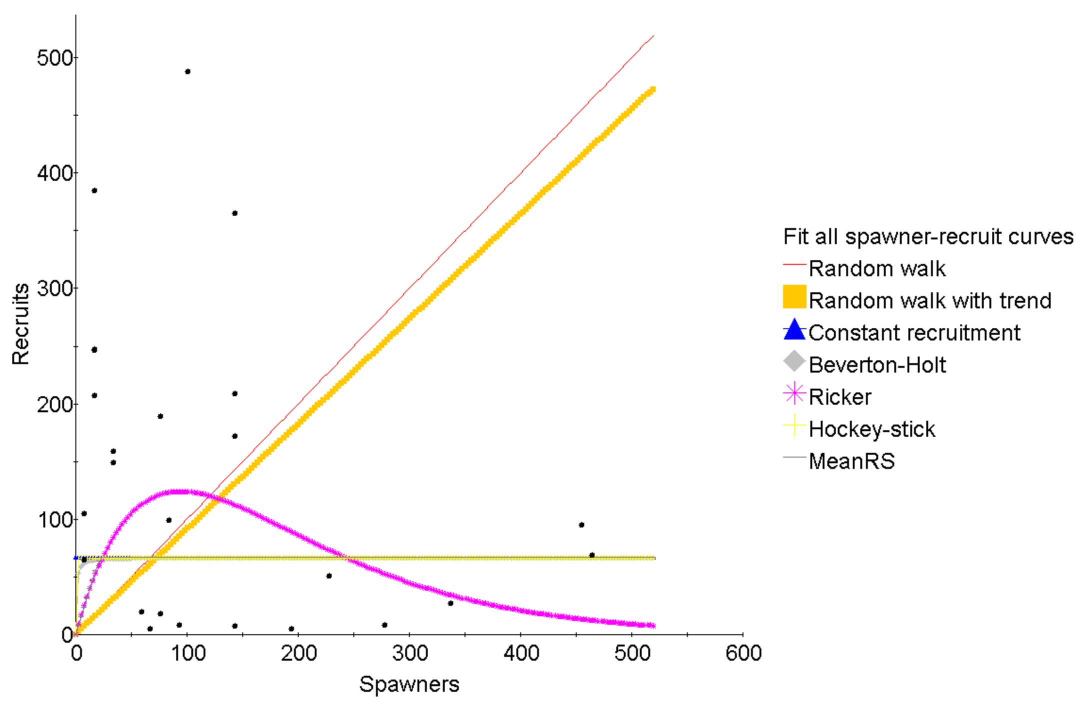

Figure 8: Clatskanie fall chinook escapement recruitment functions.

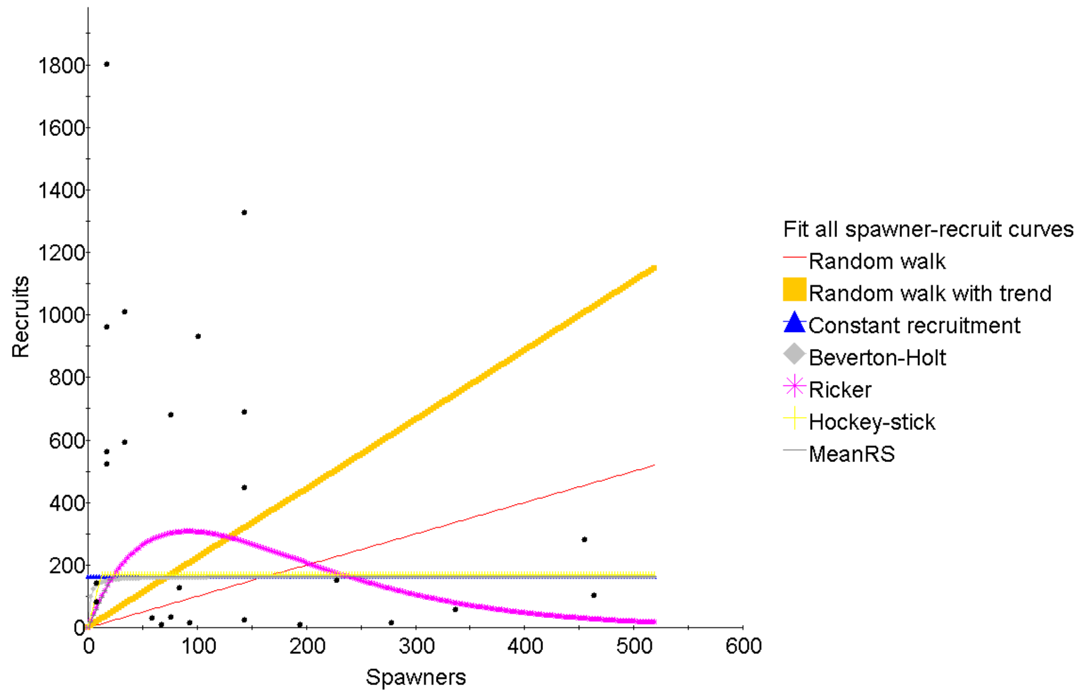

Figure 9: Clatskanie fall chinook pre-harvest recruitment functions.

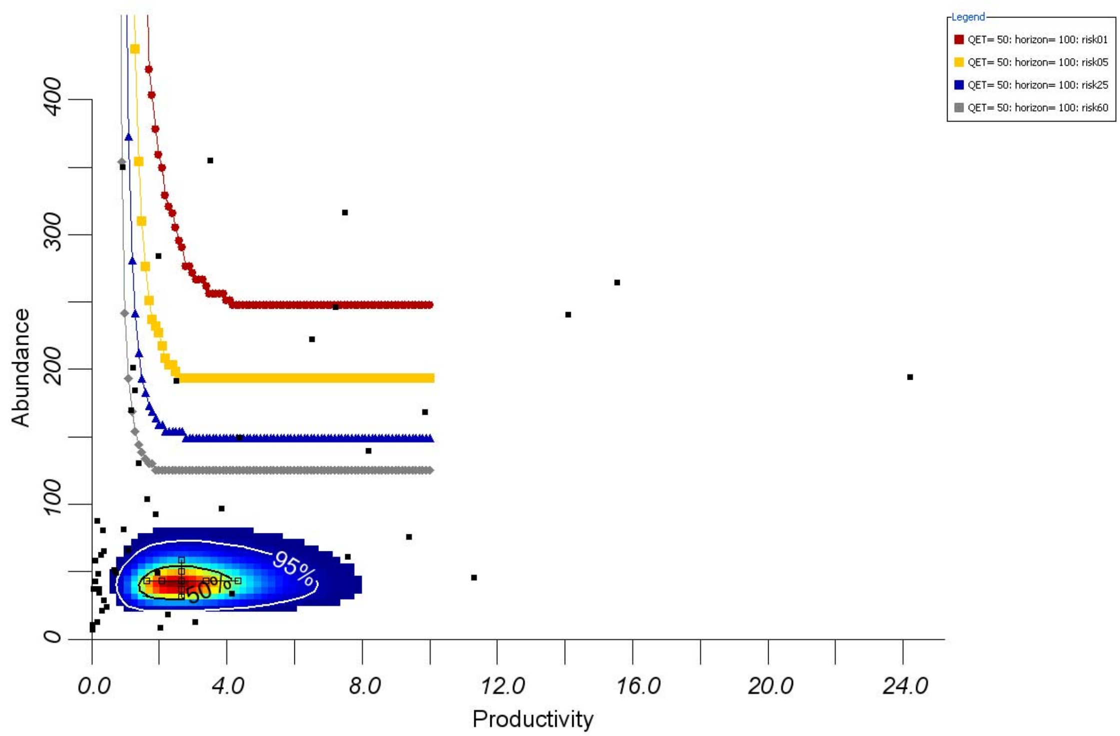

Figure 10: Clatskanie fall chinook escapement viability curve. Measurement error assumptions were: abundance ± 40%; hatchery fraction ± 70%; age structure shape parameter 20; catch abundance ± 40%. CRT = 50.

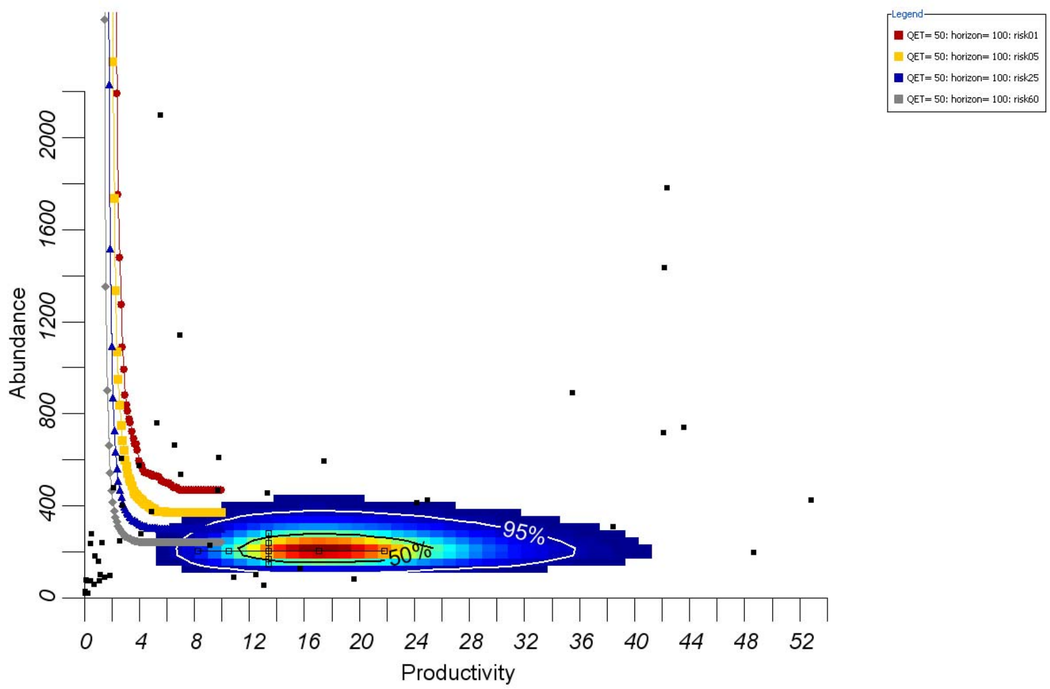

Figure 11: Clatskanie fall chinook pre-harvest viability curve. Measurement error assumptions were: abundance ± 40%; hatchery fraction ± 70%; age structure shape parameter 20; catch abundance ± 40%. CRT = 50.

Table 2: Clatskanie fall chinook summary statistics. The geometric mean natural origin spawner abundance (highlighted in red) is in the "extirpated or nearly so" viability criteria category. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	**Recent Years**	**Total Series**	**Recent Years**
Time Series Period	1952-2004	1990-2004	1952-2004	1990-2004
Length of Time Series	53	15	53	15
Geometric Mean Natural Origin Spawner Abundance	50 (34-74)	41 (18-96)	NA	NA
Geometric Mean Recruit Abundance	71 (52-96)	83 (40-173)	242 (173-337)	132 (62-280)
Lambda	0.99 (0.824-1.189)	1.152 (0.514-2.582)	1.397 (1.129-1.729)	1.33 (0.564-3.134)
Trend in Log Abundance	1.012 (0.987-1.039)	1.077 (0.882-1.314)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	1.232 (0.763-1.99)	1.628 (0.449-5.908)	4.214 (2.52-7.047)	2.592 (0.697-9.646)
Geometric Mean Recruits per Spawner (broods < median spawner abudance)	4.61 (2.998-7.088)	7.502 (0.861-65.372)	17.503 (11.436-26.789)	11.585 (1.173-114.452)
Average Hatchery Fraction	0.099	0.150	NA	NA
Average Harvest Rate	0.664	0.410	NA	NA
CAPM median extinction risk probability (5th-95th percentiles)	NA	NA	0.53	NA
PopCycle extinction risk	NA	NA	0.56	NA

Table 3: Escapement recruitment parameter estimates and relative AIC values for Clatskanie fall chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	0 (0-0)	0 (0-0)	2.14 (1.77-2.77)	18.5
Random walk with trend	0.91 (0.54-2.42)	0 (0-0)	2.14 (1.8-2.82)	20.5
Constant recruitment	0 (0-0)	67 (45-123)	1.42 (1.2-1.96)	0
Beverton-Holt	>100 (14.28->100)	68 (45-131)	1.42 (1.22-1.99)	2.2
Ricker	3.61 (1.91-13.44)	126 (94-384)	1.65 (1.44-2.41)	9.6
Hockey-stick	43.03 (9.05->100)	67 (44-123)	1.42 (1.2-1.97)	2
MeanRS	3.32 (1.43-7.11)	67 (41-106)	2.33 (1.48-2.94)	12.1

Table 4: Prehavest recruitment parameter estimates and relative AIC values for Clatskanie fall chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	0 (0-0)	0 (0-0)	2.43 (2.02-3.25)	17.6
Random walk with trend	2.21 (1.26-6.69)	0 (0-0)	2.3 (1.94-3.22)	16.8
Constant recruitment	0 (0-0)	162 (102-337)	1.64 (1.39-2.28)	0
Beverton-Holt	>100 (17.23->100)	166 (105-392)	1.65 (1.41-2.32)	2.4
Ricker	9.12 (4.41-35.76)	306 (234-1107)	1.81 (1.58-2.7)	7.1
Hockey-stick	14.13 (12.12->100)	168 (102-339)	1.63 (1.39-2.28)	1.9
MeanRS	7.5 (2.94-18.07)	162 (94-276)	2.95 (1.95-3.7)	14.4

Table 5: Clatskanie fall chinook CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.000	0.408	1.000
Probability the population is above "Moderate risk of extinction" category	0.000	0.114	0.158
Probability the population is above "Viable" category	0.000	0.023	0.005
Probability the population is above "Very low risk of extinction" category	0.000	0.000	0.000

A&P – Scappoose Fall Run

No abundance data were available on the Scappoose fall chinook population. While chinook salmon have been observed, the 2005 Oregon Native Fish Status Report states that the “existing run is likely to be primarily hatchery fish” and the population is categorized as “Fail” for abundance and productivity. A time series of abundance was analyzed for the nearby Clatskanie fall chinook population and that analysis indicated that the Clatskanie is at a high risk of extinction. Conditions in Scappoose Creek are not expected to be any more favorable to fall chinook than in the Clatskanie. There is currently no hatchery in this watershed, but there are large fall chinook hatchery releases in neighboring watersheds (discussed in the diversity section). There is no abundance and productivity evidence supporting the existence of a viable natural origin population in Scappoose Creek and comparisons with populations in similar habitats suggest the population is at significant risk.

A&P – Clackamas Fall Run (Tule)

No reliable abundance data were available on the Clackamas River fall chinook population. The 2005 BRT status update report (Good et al. 2005) contained a figure of spawner abundance for this population (Figure 12), but subsequent analysis has suggested that the data are unreliable. The Oregon Native Fish Status Report continued this time series through 2003 and the geometric mean abundance for 2000-2003 is 12 fish, with two of those years having an abundance estimate of 3 fish. Although the specific abundance estimates may not be accurate and there is no estimate of the fraction of spawners that are of hatchery origin, the figure does provide a suggestion of the order of magnitude for population size—present total spawners are likely to be in the single digits, tens or maybe hundreds. These numbers put the population in the "extirpated or nearly so" persistence category based on the minimum abundance threshold. The 2005 Oregon Native Fish Status Report listed the population as "failing" for abundance because of "chronically low returns". There is currently no hatchery in this watershed, but there are large fall chinook hatchery releases in neighboring watersheds (discussed in the diversity section). There is no abundance and productivity evidence supporting the existence of a viable natural origin population in the Clackamas, and comparisons with populations in similar habitats suggest the population is at significant risk.

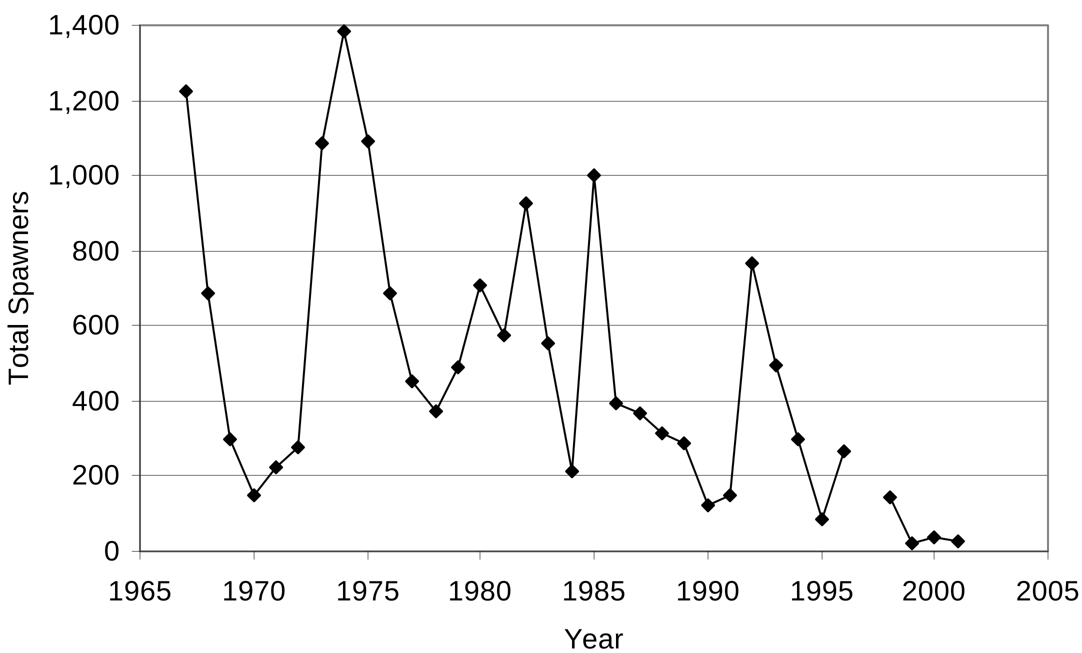

Figure 12: Spawner abundance estimates or Clackamas fall chinook copied from the 2005 BRT status update report. These data are considered unreliable, but are provided as an order of magnitude approximation. There is no estimate of the fraction of the fish that are of hatchery origin.

A&P – Sandy River Fall Run (Tule)

No abundance data were available on the Sandy River tule fall chinook population. The 2005 Oregon Native Fish Status Report does not list this population since there is uncertainty on the historical existence of a tule population in the Sandy River. The TRT list of historical populations adopted a more inclusive approach with populations of uncertain heritage (Myers et al. 2006). There is currently no hatchery in this watershed, but there are large numbers of hatchery-origin fall chinook released into neighboring watersheds (discussed in the diversity section). The neighboring Clackamas tule population is describe as being “chronically low abundance” in the 2005 Native Fish Status Report. There is no abundance and productivity evidence supporting the existence of a viable natural origin population in the Sandy River, and comparisons with populations in similar habitats suggest the population is at significant risk.

A&P – Lower Gorge Fall Run (Tule)

No abundance data were available for the Lower Gorge fall chinook population. The 2005 Oregon Native Fish Status Report did not assess mainstem populations (i.e. Ives Island), which is where much of the spawning for this population currently occurs. Part of the population exists on the Washington side of the Columbia. There are large hatchery releases in this population and it is expected that the majority of spawning fish that return are of hatchery origin. Historically, the nearby Clackamas population would have been much larger than the Lower Gorge population and given that the Clackamas population is currently at low abundance, it is likely that the Lower Gorge is at even lower abundance. There is no abundance and productivity evidence substantiating the existence of a viable natural origin population in the Oregon portion of the Lower Gorge population and the population is considered to be at significant risk.

A&P – Upper Gorge Fall Run (Tule)

No abundance data were available on the Upper Gorge fall chinook population. The 2005 Oregon Native Fish Status Report did not assess mainstem populations, which is where much of the spawning for this population is likely to have occurred. Historical spawning was also likely in the lower reaches of tributaries which have been inundated by Bonneville Dam. Part of the population also occurs on the Washington side of the Columbia. There are large hatchery releases into this population and it is expected that the majority of spawning fish that return are of hatchery origin. Historically, the nearby Hood River population may have been larger than the Upper Gorge population and so, given the Hood River population is currently at low abundance, it is likely that the Upper Gorge is at even lower abundance. There is no abundance and productivity evidence supporting the existence of a viable natural origin population in the Oregon portion of the Upper Gorge population and the population is considered to be at significant risk.

A&P – Hood Fall Run (Tule)

The 2005 Oregon Native Fish Status Report lists an average spawner abundance for the Hood River fall chinook population from 1992-2004 as 26 fish and the geometric mean from 2000-2004 as 36 fish (Figure 13). These numbers put the population in the "extirpated or nearly so" persistence category based on the minimum abundance threshold. The 2005 Oregon Native Fish Status Report puts the population in the "fail" category for abundance and productivity.

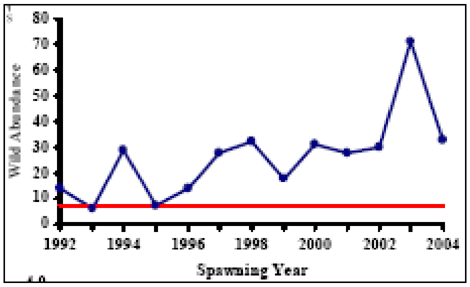

Figure 13: Estimate of Hood River fall chinook wild abundance based on Powerdale Dam count (from Oregon Native Fish Status Report 2005).

A&P – Sandy River late fall Run (Brights)

A time series of abundance sufficient for quantitative analysis is available for the Sandy River late fall run population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 14 to Figure 21 and in Table 6 and Table 9. The population is relatively large (recent geometric mean > 2,500 spawners). The population is also assumed to be relatively free of hatchery fish. The pre-harvest viability curve analysis, the PopCycle modeling and the CAPM Modeling suggest that the population is currently viable. The pre-harvest viability curves were run considering two different future harvest assumptions, 25% and 50%, in order to bracket the range of observed harvest rates in the population. The viability curve analysis assumes that a 25% future harvest indicates that the population is most likely viable, but there is considerable uncertainty in the assessment. If it is assumed that future harvest will be 50%, the population is most likely not viable. The escapement viability curve suggests that the population would not be sustainable in the long term if the harvest rates over the available time series, which averaged 43%, were extended into the future,. The geometric mean natural origin abundance is approximately 3,000 (Table 6), which is in the "very low risk" minimum abundance threshold category.

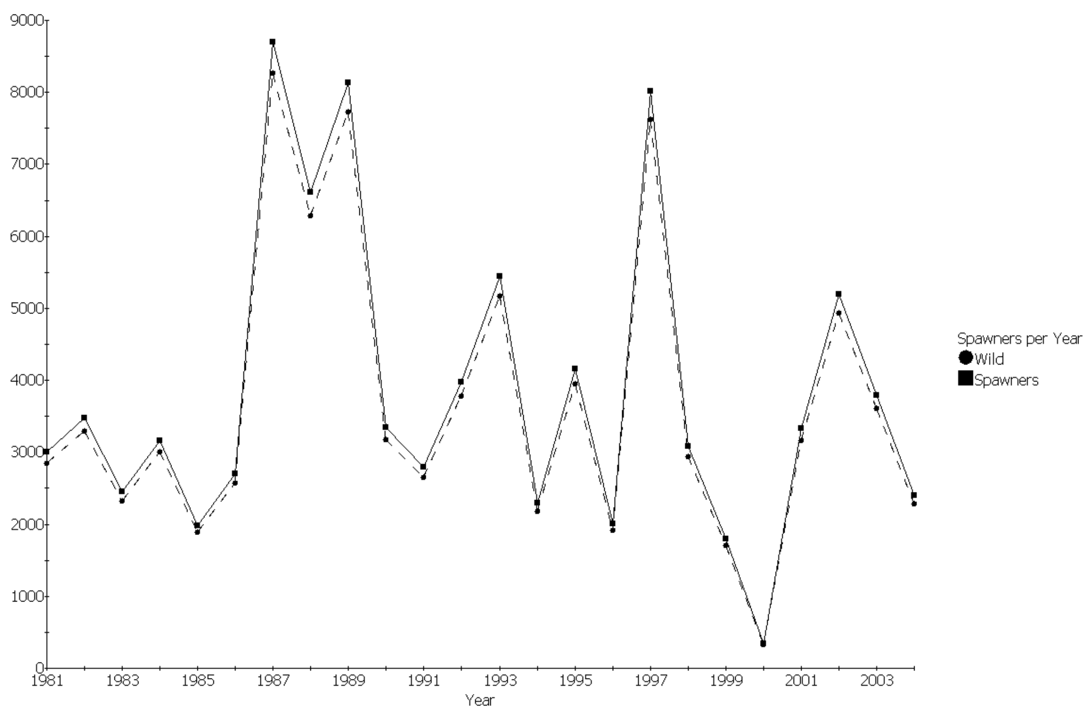

Figure 14: Sandy River late-fall chinook salmon abundance.

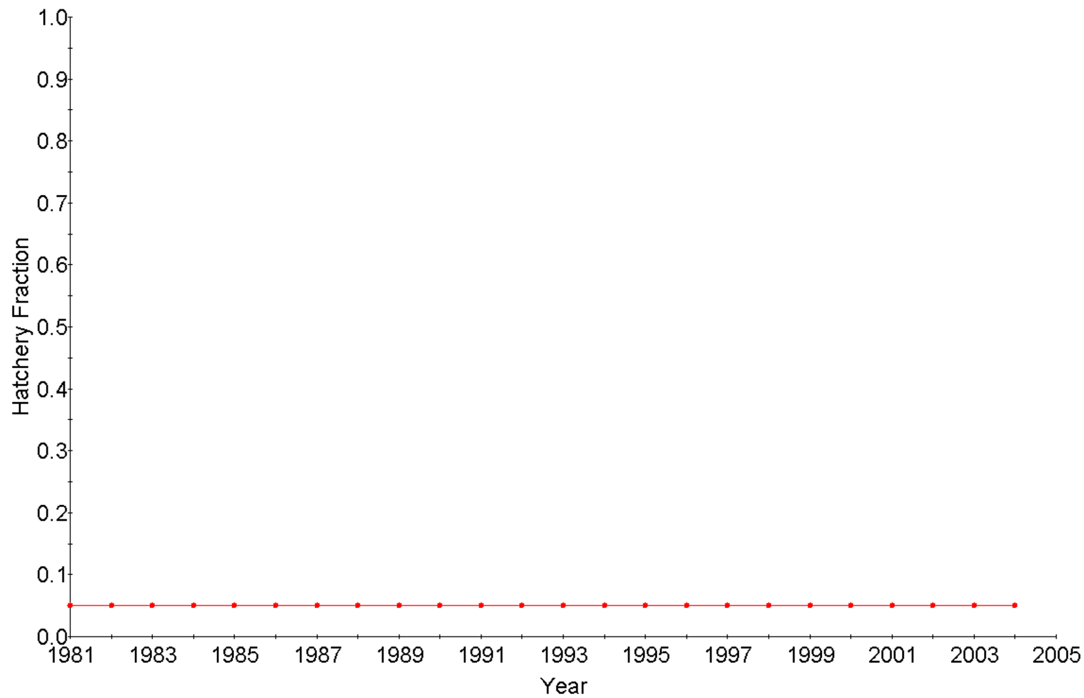

Figure 15: Sandy River late-fall chinook salmon hatchery fraction.

Figure 16: Sandy River late-fall chinook salmon harvest rate.

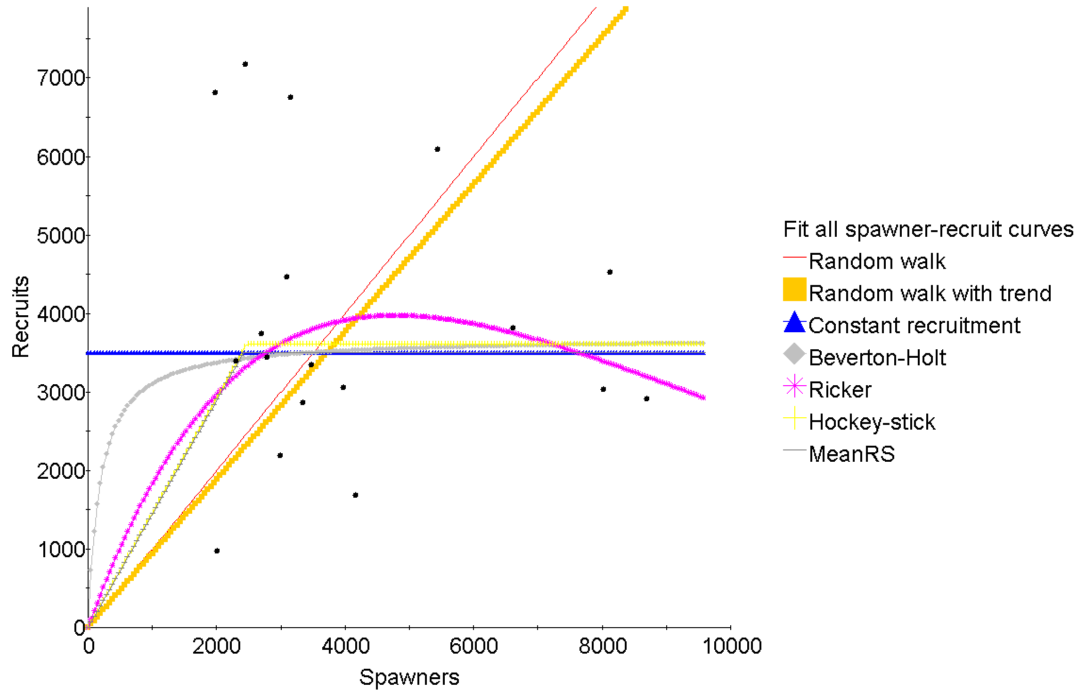

Figure 17: Sandy River late-fall chinook salmon escapement recruitment functions.

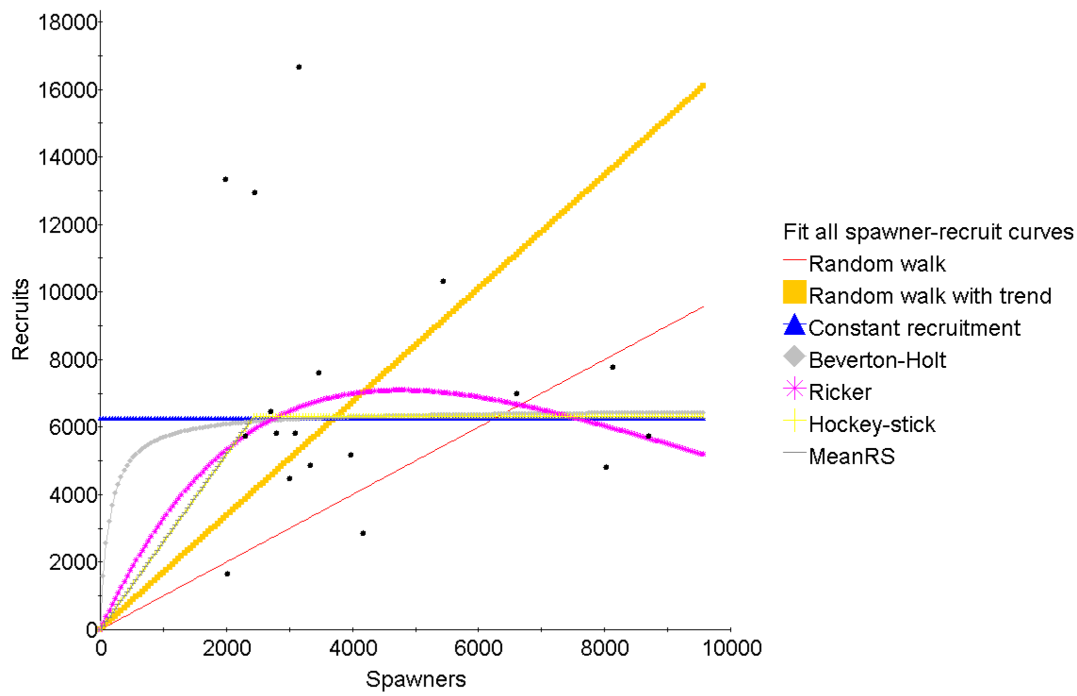

Figure 18: Sandy River late-fall chinook salmon pre-harvest recruitment functions.

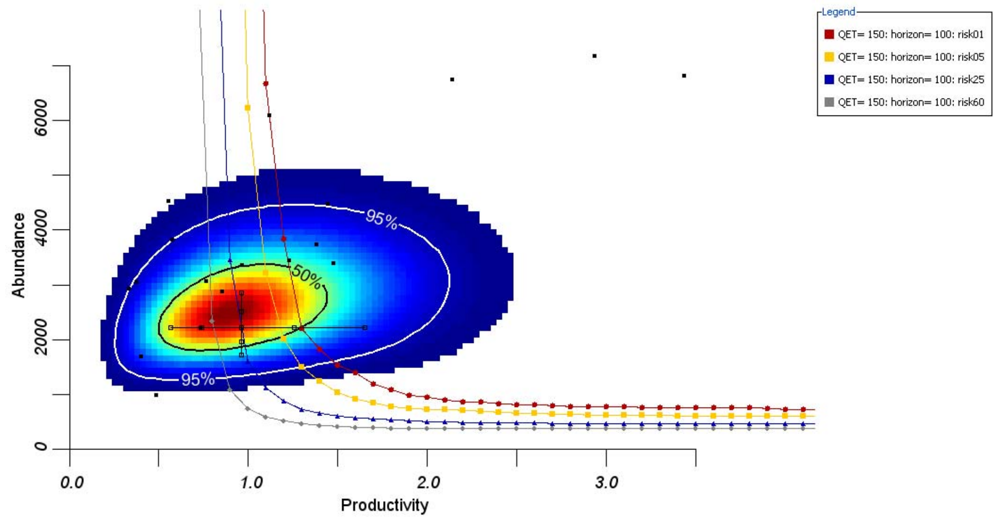

Figure 19: Sandy River late fall chinook escapement viability curves. Measurement error assumptions were: abundance ±40%; hatchery fraction ±70%; age structure shape parameter 20; catch abundance ±40%. CRT = 150.

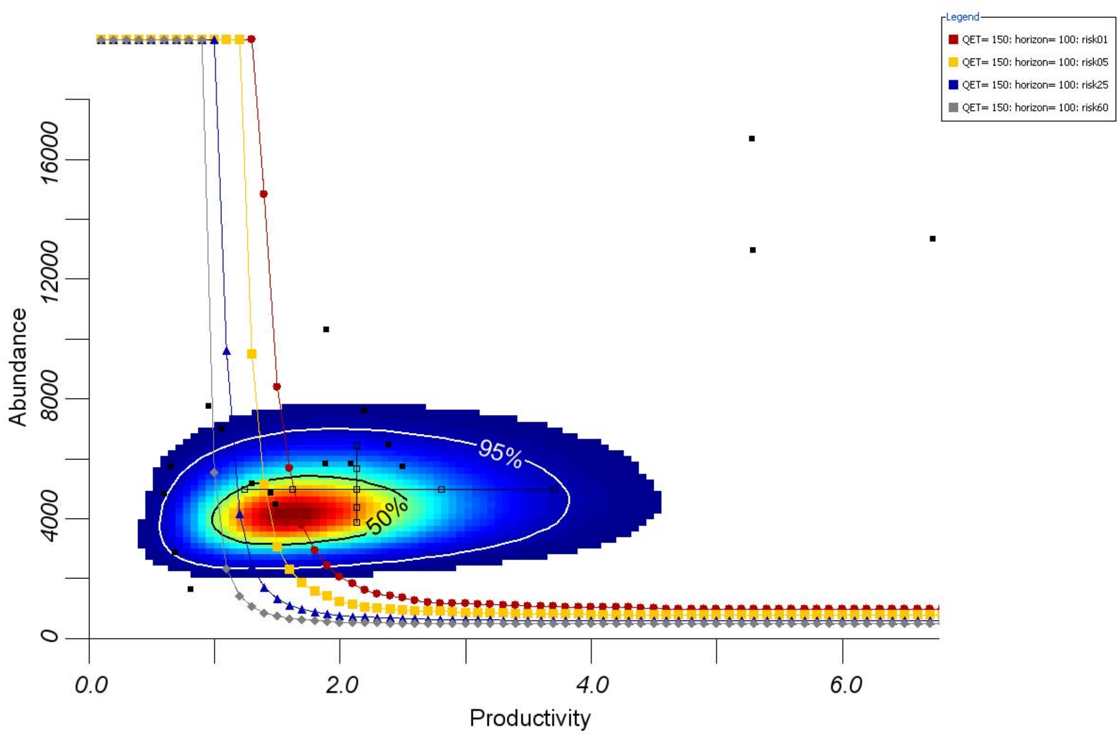

Figure 20: Sandy River late fall chinook pre-harvest viability curves. Measurement error assumptions were: abundance ±40%; hatchery fraction ±70%; age structure shape parameter 20; catch abundance ±40% (Assumes future harvest rate of 25%). CRT = 150.

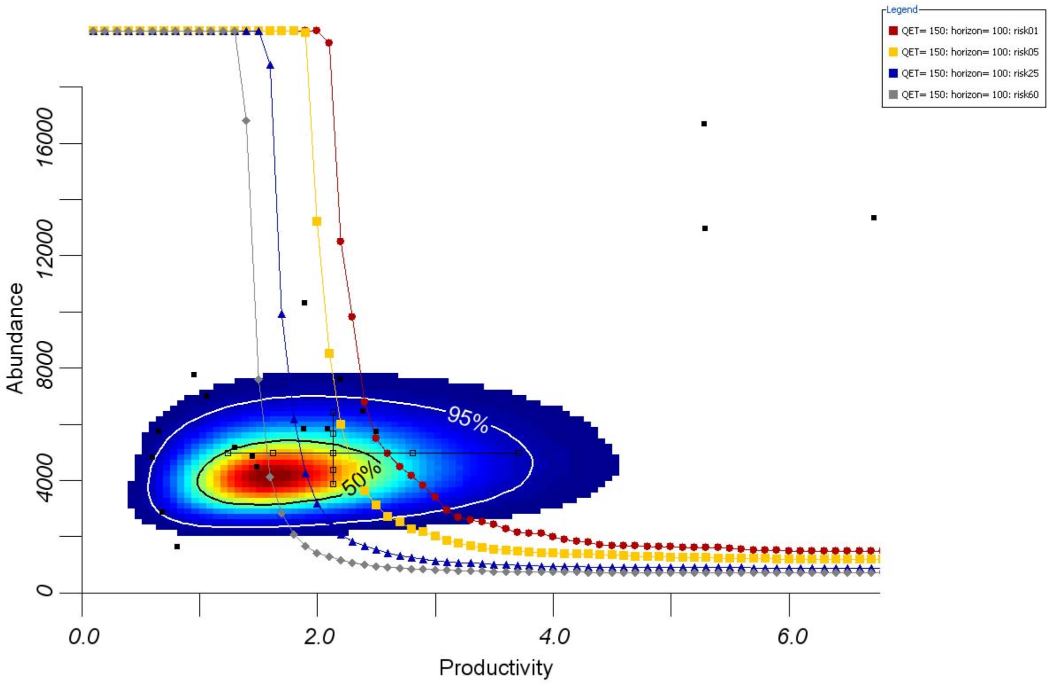

Figure 21: Sandy River late fall chinook pre-harvest viability curves. Measurement error assumptions were: abundance ±40%; hatchery fraction ±70%; age structure shape parameter 20; catch abundance ±40% (Assumes future harvest rate of 50%). CRT = 150.

Table 6: Sandy River late fall chinook summary statistics. The geometric mean natural origin spawner abundance (highlighted) is in the "very low risk" viability criteria category. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1981-2004	1990-2004	1981-2004	1990-2004
Length of Time Series	24	15	24	15
Geometric Mean Natural Origin Spawner Abundance	3085 (2337-4074)	2771 (1868-4110)	NA	NA)
Geometric Mean Recruit Abundance	3505 (2727-4504)	2887 (1917-4347)	6268 (4770-8235)	4708 (3171-6991)
Lambda	0.997 (0.857-1.16)	0.982 (0.827-1.167)	1.135 (0.938-1.373)	1.088 (0.902-1.311)
Trend in Log Abundance	0.983 (0.945-1.024)	0.971 (0.885-1.066)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.94 (0.669-1.321)	0.807 (0.534-1.218)	1.681 (1.174-2.407)	1.316 (0.882-1.962)
Geometric Mean Recruits per Spawner (broods < median spawner abudance)	1.448 (0.898-2.333)	1.063 (0.459-2.463)	2.595 (1.535-4.385)	1.682 (0.763-3.707)
Average Hatchery Fraction	0.05	0.05	NA	NA
Average Harvest Rate	0.4268	0.3771	NA	NA
CAPM median extinction risk probability (5^{th}-95^{th} percentiles)	NA	NA	0.000 (0.000-0.000)	NA
PopCycle extinction risk	NA	NA	<0.01	NA

Table 7: Escapement recruitment parameter estimates and relative AIC values for Sandy River late fall chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	0 (0-0)	0 (0-0)	0.67 (0.54-0.97)	9.1
Random walk with trend	0.94 (0.73-1.3)	0 (0-0)	0.66 (0.55-1)	10.9
Constant recruitment	0 (0-0)	3505 (2883-4420)	0.49 (0.41-0.74)	0
Beverton-Holt	19.3 (3.86->50)	3705 (3023-5337)	0.49 (0.4-0.74)	2
Ricker	2.25 (1.32-3.81)	3987 (3409-6539)	0.49 (0.42-0.78)	2.2
Hockey-stick	1.46 (3.25->50)	3566 (2887-4432)	0.49 (0.4-0.74)	1.4
MeanRS	1.45 (1.04-1.99)	3505 (2882-4217)	0.25 (0.11-0.37)	

Table 8: Pre-harvest recruitment parameter estimates and relative AIC values for Sandy River late fall chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	0 (0-0)	0 (0-0)	0.87 (0.71-1.25)	15.7
Random walk with trend	1.68 (1.28-2.39)	0 (0-0)	0.7 (0.58-1.06)	9.9
Constant recruitment	0 (0-0)	6271 (5077-8100)	0.53 (0.44-0.8)	0
Beverton-Holt	>50 (4.83->50)	6535 (5468-10985)	0.53 (0.44-0.8)	2
Ricker	4.07 (2.25-7.36)	7097 (6046-13237)	0.54 (0.46-0.86)	2.1
Hockey-stick	2.66 (3.97->50)	6505 (5078-8166)	0.52 (0.44-0.81)	1.5
MeanRS	2.59 (1.82-3.68)	6268 (5091-7681)	0.29 (0.13-0.45)	

Table 9: Sandy River late fall chinook CAPM risk category and viability curve results.

Risk Category	Viability Curves			CAPM
	Escapement	Pre-harvest (harvest rate 25%)	Pre-harvest (harvest rate 50%)	
Probability the population is not in "Extirpated or nearly so" category	0.737	0.927	0.657	1.000
Probability the population is above "Moderate risk of extinction" category	0.601	0.865	0.487	1.000
Probability the population is above "Viable" category	0.413	0.748	0.282	1.000
Probability the population is above "Very low risk of extinction" category	0.309	0.613	0.157	0.993

A&P – Sandy River spring Run

A time series of abundance sufficient for quantitative analysis is available for the Sandy River spring run population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 22 to Figure 28 and in Table 10 and Table 13. The total number of spawners in the population has been in the low thousands in recent years, but on average at least half of the fish in some years are estimated to be of hatchery origin. . However, the data suggest general upward population trend that most likely reflects the fact that up until the 1970s spring chinook passage upstream of Marmot Dam was severely restricted due to water diversions that dewatering of the migration channel. The pre-harvest viability curve analysis, PopCycle and the CAPM modeling are in general agreement that the population is not likely to be viable but is in a high to moderate risk category. The escapement viability curve suggests that a population experiencing the pattern of harvest that occurred over the observed time period would not be sustainable in the long term. The long term geometric mean of natural origin spawners for the population is around 300 fish (Table 10), which is in the "extirpated or nearly so" minimum abundance threshold category, but using only the most recent years data, the population would be in the viable category.

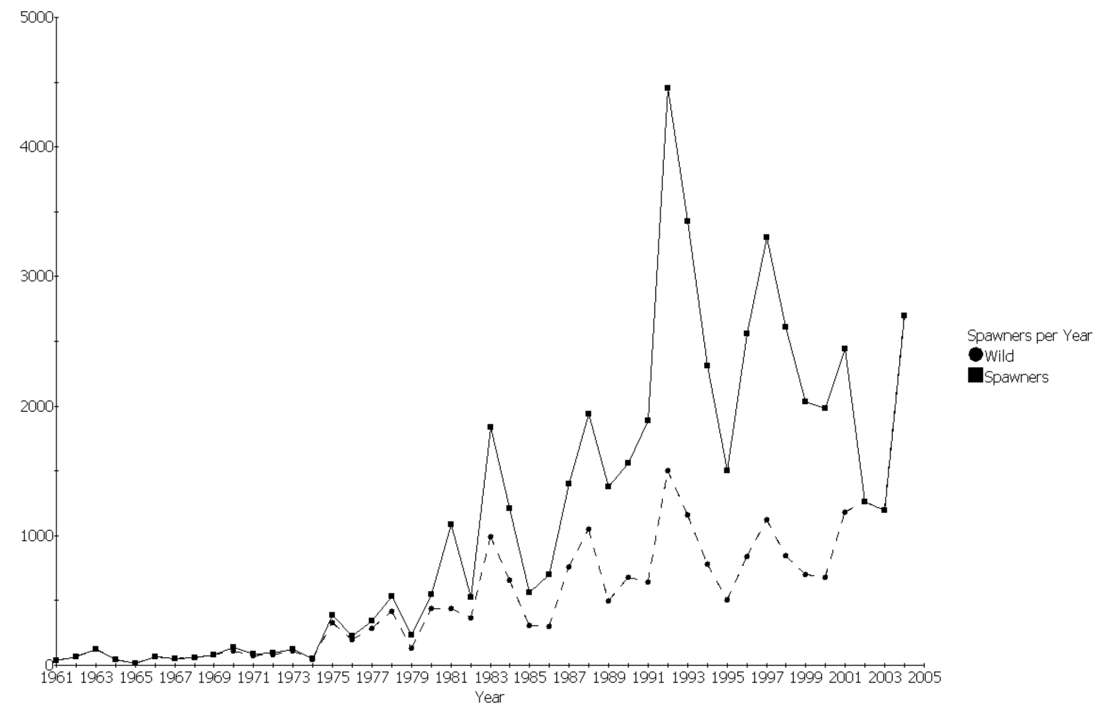

Figure 22: Sandy River spring-run chinook salmon abundance.

Figure 23: Sandy River spring-run chinook salmon hatchery fraction.

Figure 24: Sandy River spring-run chinook salmon harvest rate.

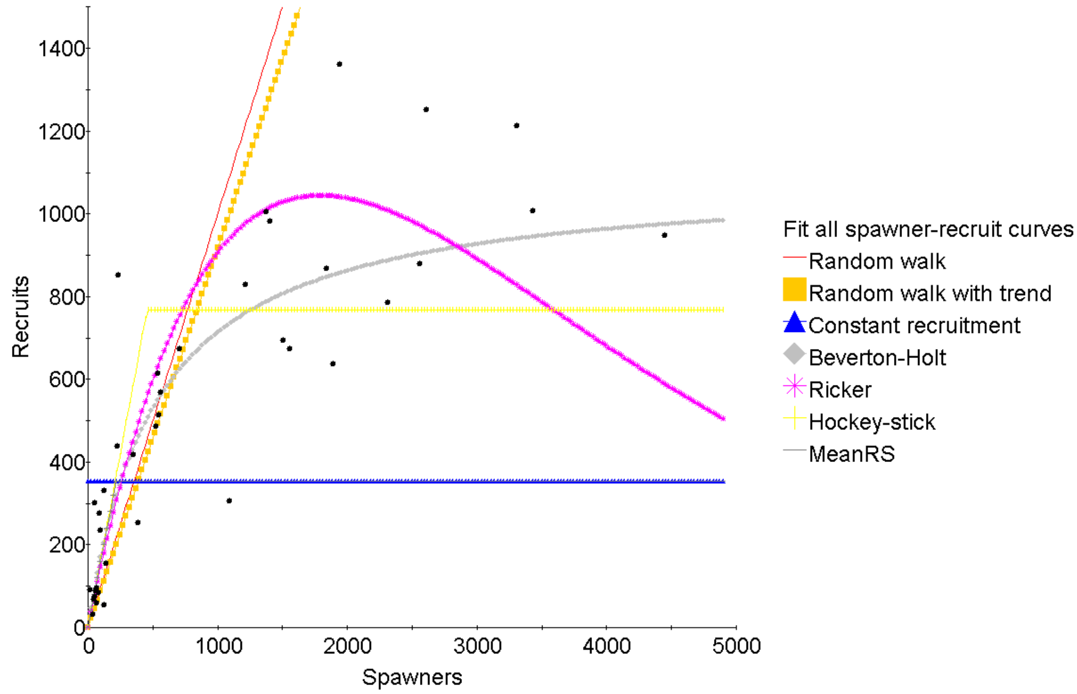

Figure 25: Sandy River spring-run chinook salmon escapement recruitment functions.

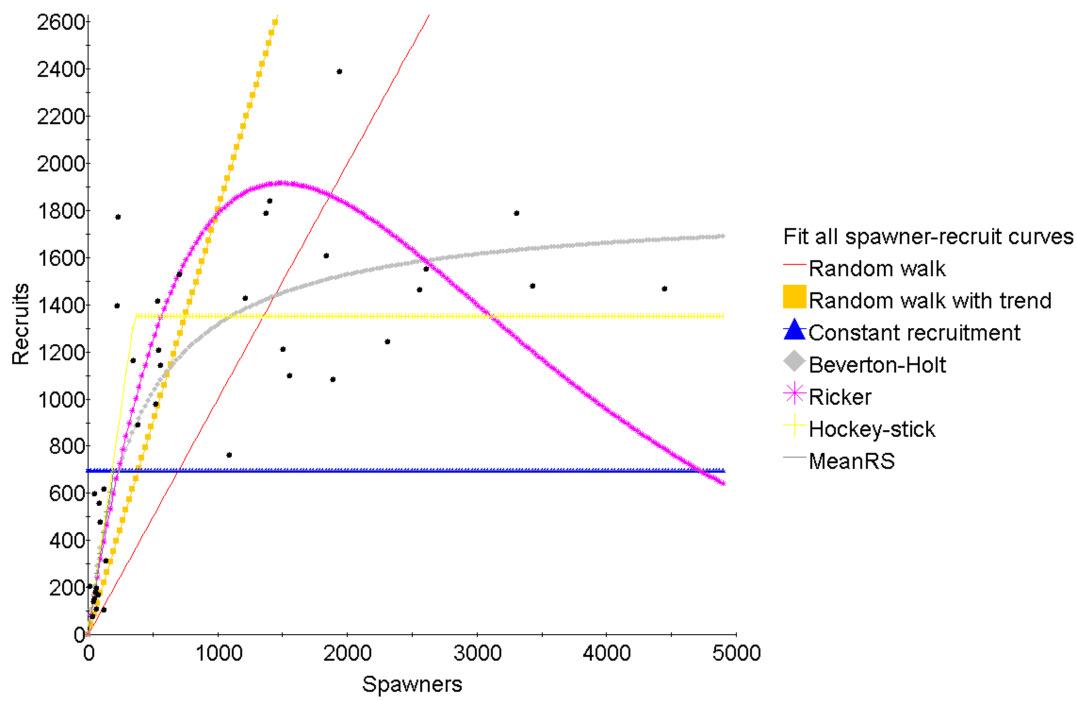

Figure 26: Sandy River spring-run chinook salmon pre-harvest recruitment functions.

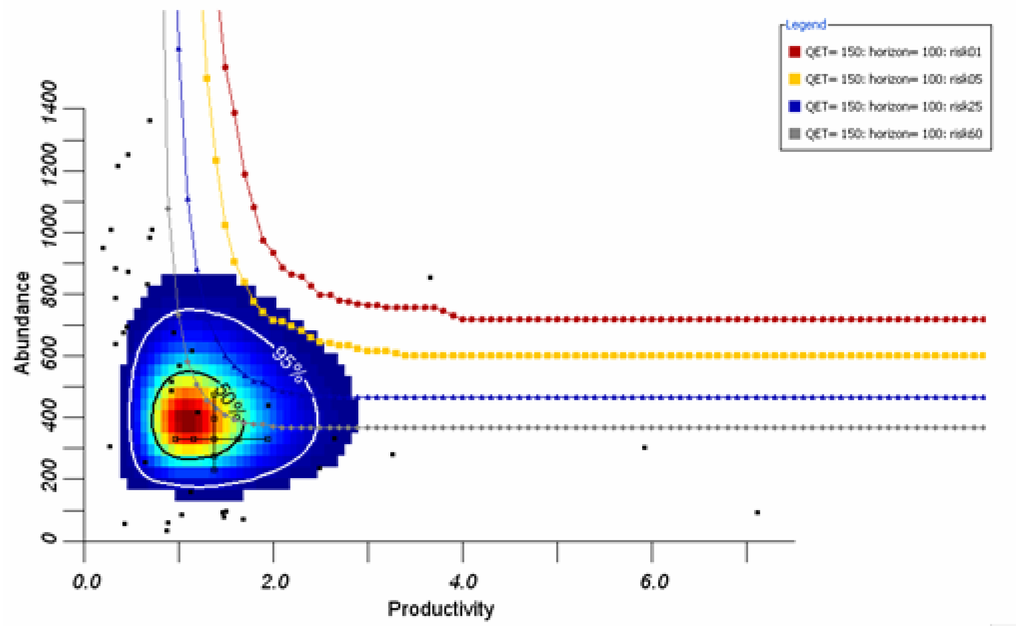

Figure 27: Sandy River spring-run chinook salmon escapement viability curve. Measurement error assumptions were: abundance ±40%; hatchery fraction ±40%; age structure shape parameter 20; catch abundance ±30%. CRT = 150.

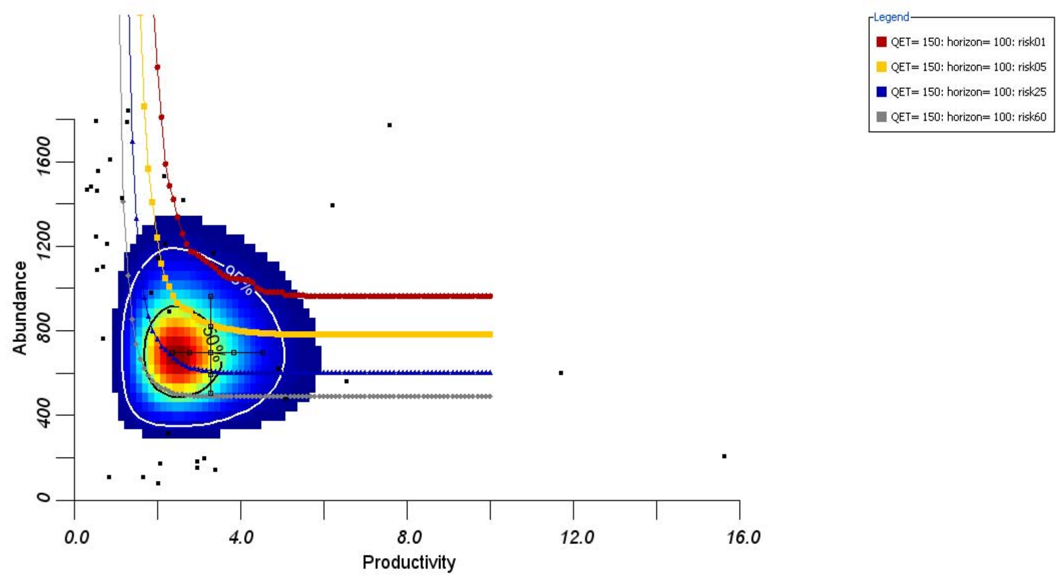

Figure 28: Sandy River spring chinook pre-harvest viability curve. Measurement error assumptions were: abundance ±40%; hatchery fraction ±40%; age structure shape parameter 20; catch abundance ±30%. (Assumes future harvest rate of 25%.) CRT = 150.

Table 10: Sandy River spring chinook summary statistics. The geometric mean natural origin spawner abundance (highlighted) is in the "extirpated or nearly so" viability criteria category for the total time series, but in the "viable" category using only recent year data. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1961-2004	1990-2004	1961-2004	1990-2004
Length of Time Series	44	15	44	15
Geometric Mean Natural Origin Spawner Abundance	297 (202-438)	959 (759-1212)	NA	NA
Geometric Mean Recruit Abundance	355 (251-502)	874 (722-1059)	697 (502-968)	1359 (1193-1548)
Lambda	0.961 (0.853-1.083)	0.834 (0.657-1.059)	1.111 (0.957-1.289)	0.901 (0.725-1.119)
Trend in Log Abundance	1.093 (1.079-1.108)	1.047 (0.997-1.1)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.915 (0.692-1.209)	0.354 (0.292-0.429)	3.332 (2.463-4.508)	0.55 (0.451-0.671)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.535 (1.13-2.084)	0.407 (0.271-0.613)	3.332 (2.463-4.508)	0.688 (0.451-1.05)
Average Hatchery Fraction	0.323	0.515	NA	NA
Average Harvest Rate	0.476	0.376	NA	NA
CAPM median extinction risk probability (5th and 95th percentiles in parenthesis)	NA	NA	0.090 (0.005-0.435)	NA
PopCycle Extinction Risk	NA	NA	0.8	NA

Table 11: Escapement recruitment parameter estimates and relative AIC values for Sandy River spring chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	0 (0-0)	0 (0-0)	0.84 (0.72-1.06)	34.1
Random walk with trend	0.92 (0.74-1.19)	0 (0-0)	0.84 (0.72-1.07)	35.7
Constant recruitment	0 (0-0)	354 (273-493)	1.04 (0.9-1.33)	52.4
Beverton-Holt	2.06 (1.59-2.77)	1092 (832-1578)	0.51 (0.45-0.66)	0
Ricker	1.58 (1.29-1.97)	1044 (899-1360)	0.56 (0.49-0.72)	6.5
Hockey-stick	1.69 (1.32-2.26)	769 (616-1049)	0.55 (0.48-0.72)	6.1
MeanRS	1.63 (1.25-2.14)	355 (267-468)	0.41 (0.26-0.55)	81.9

Table 12: Preharvest recruitment parameter estimates and relative AIC values for Sandy River spring chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	0 (0-0)	0 (0-0)	1.1 (0.94-1.39)	54.3
Random walk with trend	1.8 (1.43-2.4)	0 (0-0)	0.94 (0.81-1.2)	43.7
Constant recruitment	0 (0-0)	697 (547-947)	0.99 (0.85-1.26)	47.7
Beverton-Holt	4.71 (3.63-6.48)	1825 (1423-2497)	0.52 (0.45-0.67)	0.2
Ricker	3.49 (2.86-4.37)	1915 (1664-2372)	0.55 (0.48-0.72)	5.5
Hockey-stick	3.77 (3.03-4.8)	1352 (1131-1718)	0.51 (0.45-0.67)	0
MeanRS	3.54 (2.73-4.62)	697 (533-898)	0.36 (0.2-0.53)	88.2

Table 13: Sandy River spring chinook CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.302	0.858	0.978
Probability the population is above "Moderate risk of extinction" category	0.070	0.595	0.858
Probability the population is above "Viable" category	0.004	0.164	0.297
Probability the population is above "Very low risk of extinction" category	0.000	0.018	0.075

A&P – Hood Spring Run

The 2005 BRT report describe the Hood River spring run as "extirpated or nearly so" and the 2005 Native Fish Status report describes the population as "extinct." A hatchery population with out-of-ESU brood stock is currently in the watershed, but native fish are not considered to be present.

A&P – Criterion Summary

For the abundance and productivity criterion, the most probable risk category for all but two of these populations is high (Figure 29). The exceptions are most probable classifications of 'moderate risk' for the Sandy River spring chinook populations and 'low risk' for the Sandy River late fall chinook. Although the shape of the diamonds in Figure 29 suggest there is considerable uncertainty as to the status classification of these two Sandy populations, even the most optimistic interpretation would place only one population in the viable category. Conversely, the lower tail of the diamonds for these two populations both drop into the 'high risk' category. From the perspective of this viability criterion LCR chinook in Oregon are clearly at high risk.

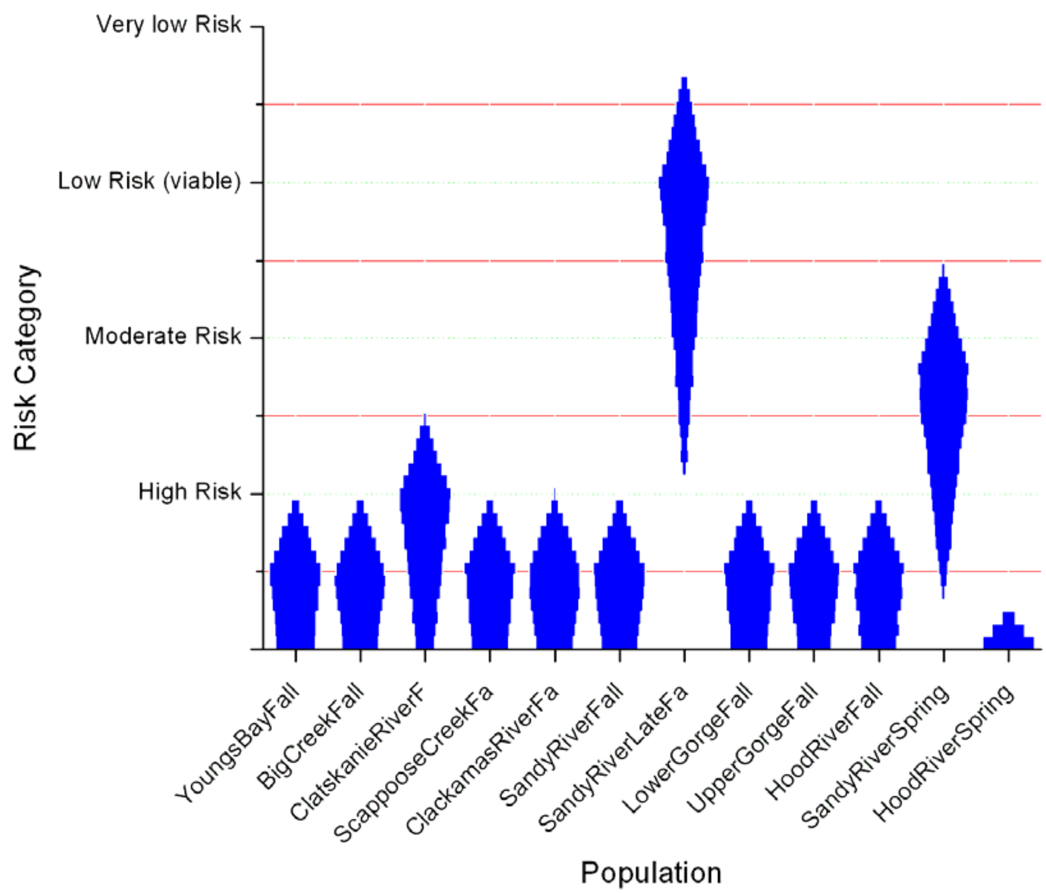

Figure 29: Lower Columbia River chinook salmon risk status summary based on evaluation of abundance and productivity only.

III. Spatial Structure

SS – Youngs Bay

Even under historical conditions, the distribution of fall chinook in this basin was limited. Most tributary streams remain accessible to anadromous fish, particularly in the mainstem areas that were historically suitable for fall chinook (Figure 1)(ODFW 2005). Small areas of marginal habitat for fall chinook are no longer accessible or utilized above a hatchery weir on the NF Klaskanine and in several small valley floor tributaries. ODFW (2005) estimates that 13% of the historical fall chinook habitat is no longer accessible. Habitat degradation in the basin has reduced the spatial distribution of suitable habitats for fall chinook. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for weighted historical productivity of suitable habitats and effects of habitat degradation on currently accessible habitats.

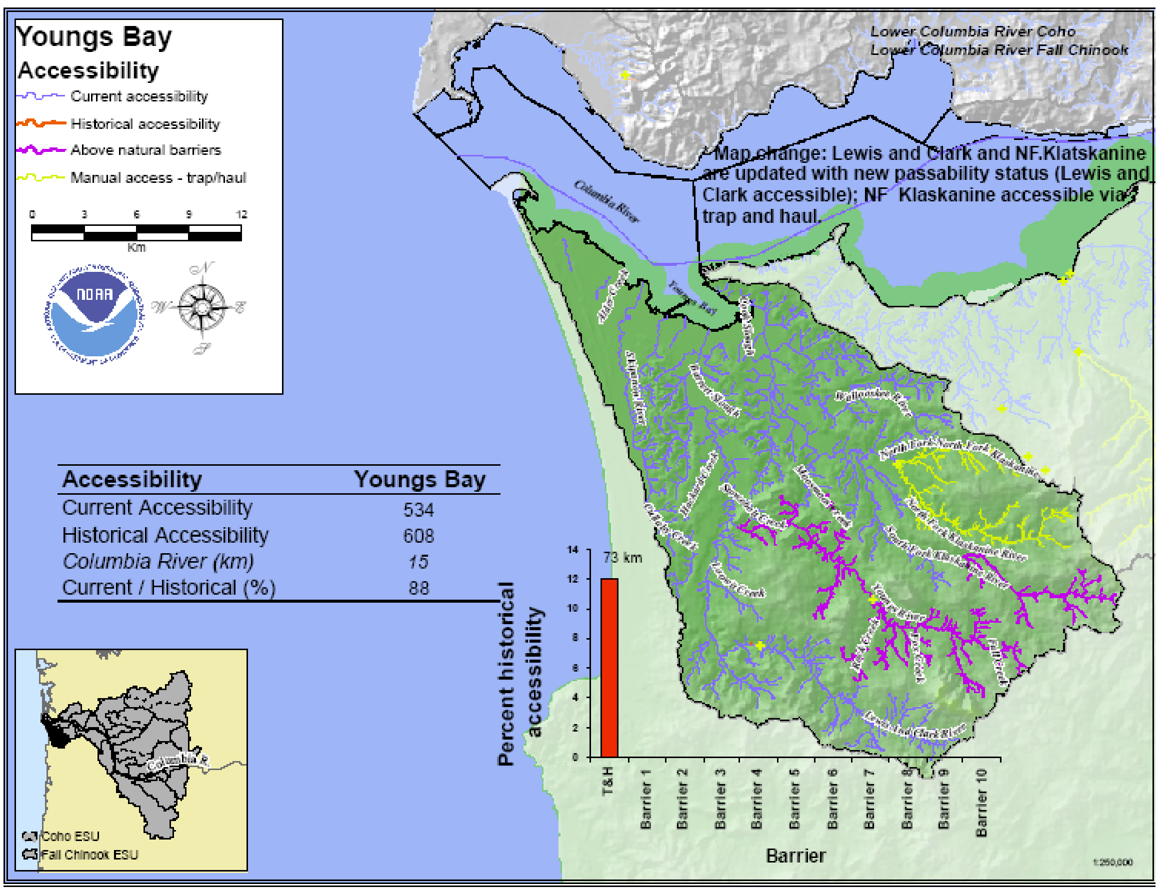

Figure 30: Youngs Bay fall-run chinook salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Big Creek

Even under historical conditions, the distribution of fall chinook in this basin was largely limited to lower mainstem reaches. Most areas that were historically suitable for fall chinook are currently accessible (Figure 31) (ODFW 2005). Hatchery barriers limit access to portions of Gnat Creek but these areas were not productive fall chinook habitats. Habitat degradation in the basin has reduced the spatial distribution of suitable habitats for fall chinook. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for weighted historical productivity of suitable habitats and effects of habitat degradation on currently accessible habitats.

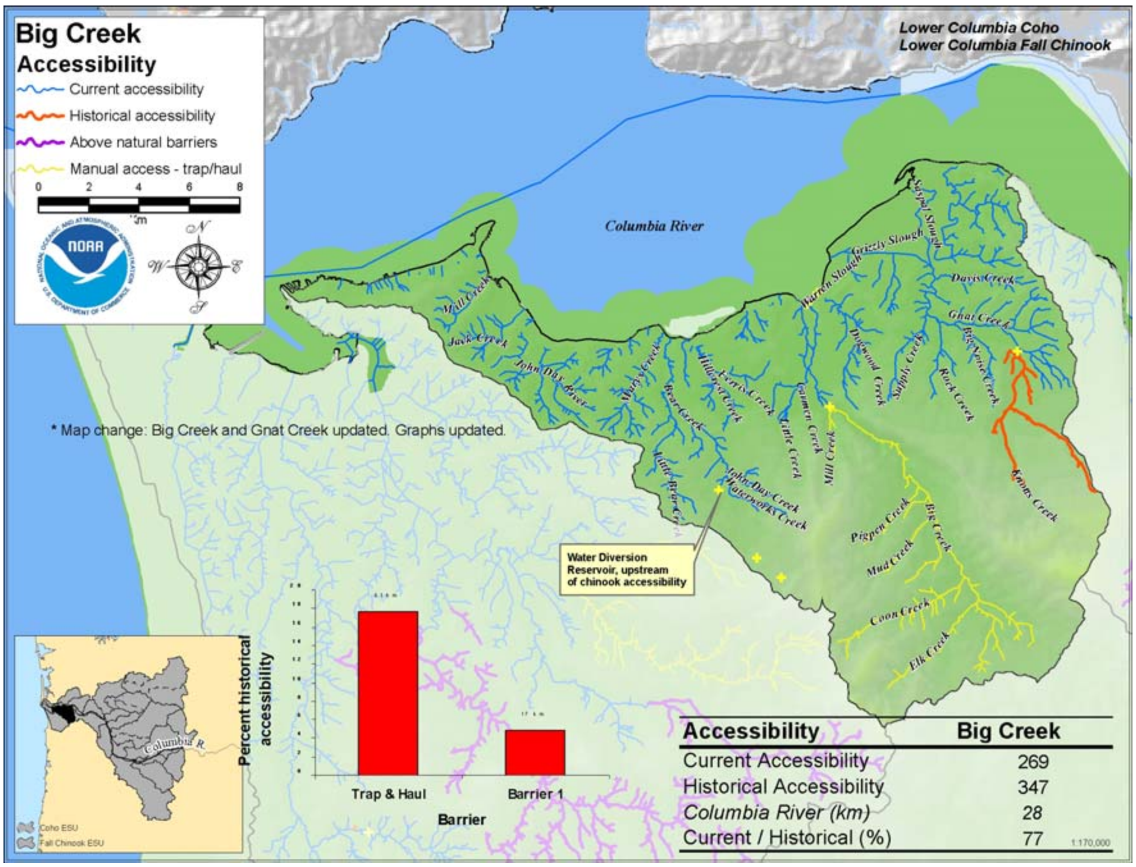

Figure 31: Big Creek fall-run chinook salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Clatskanie

Even under historical conditions, the distribution of fall chinook in this basin was largely limited to lower mainstem reaches. All mainstem areas that were historically suitable for fall chinook are currently accessible (Figure 32)(ODFW 2005). Anadromous access to some smaller streams has been lost but these areas were not productive fall chinook habitats. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for the limited area of suitable habitat and effects of habitat degradation on currently accessible habitats.

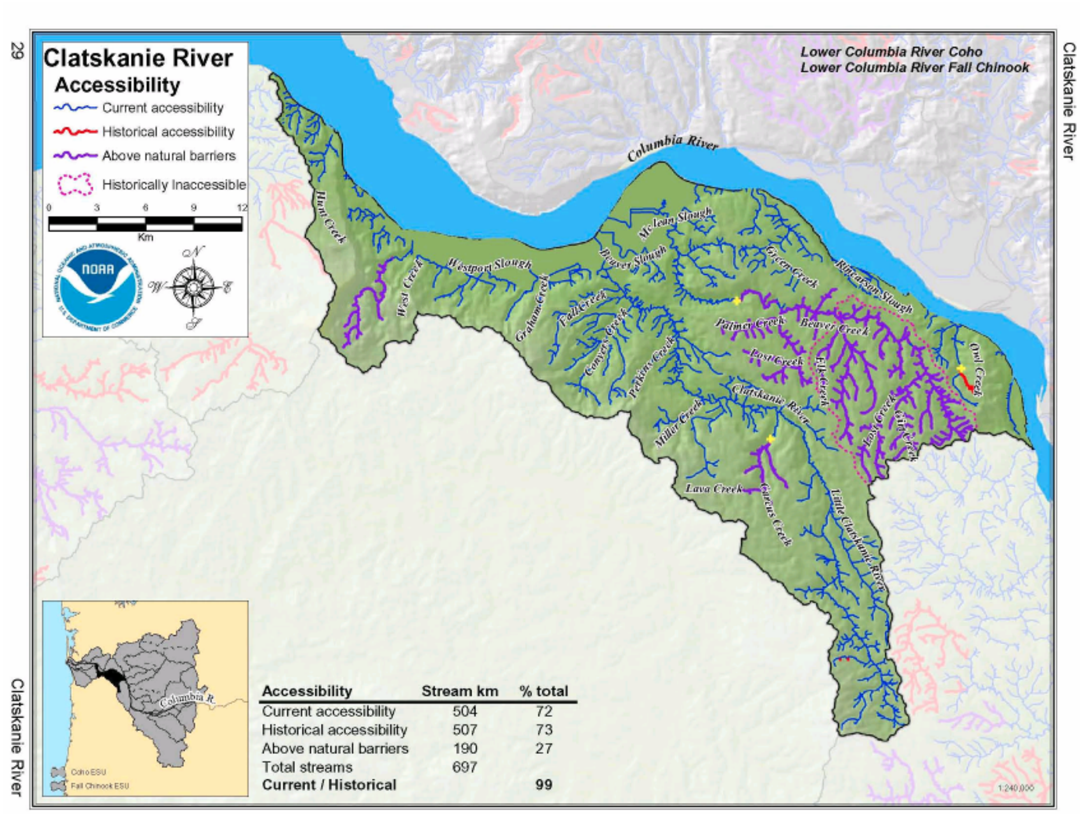

Figure 32: Clatskanie fall chinook current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Scappoose

Even under historical conditions, the distribution of fall chinook in this basin was largely limited to lower mainstem reaches. All mainstem areas that were historically suitable for fall chinook are currently accessible (Figure 33)(ODFW 2005). Anadromous access to some smaller streams has been lost but these areas were not productive fall chinook habitats. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for the limited area of suitable habitat and effects of habitat degradation on currently accessible habitats.

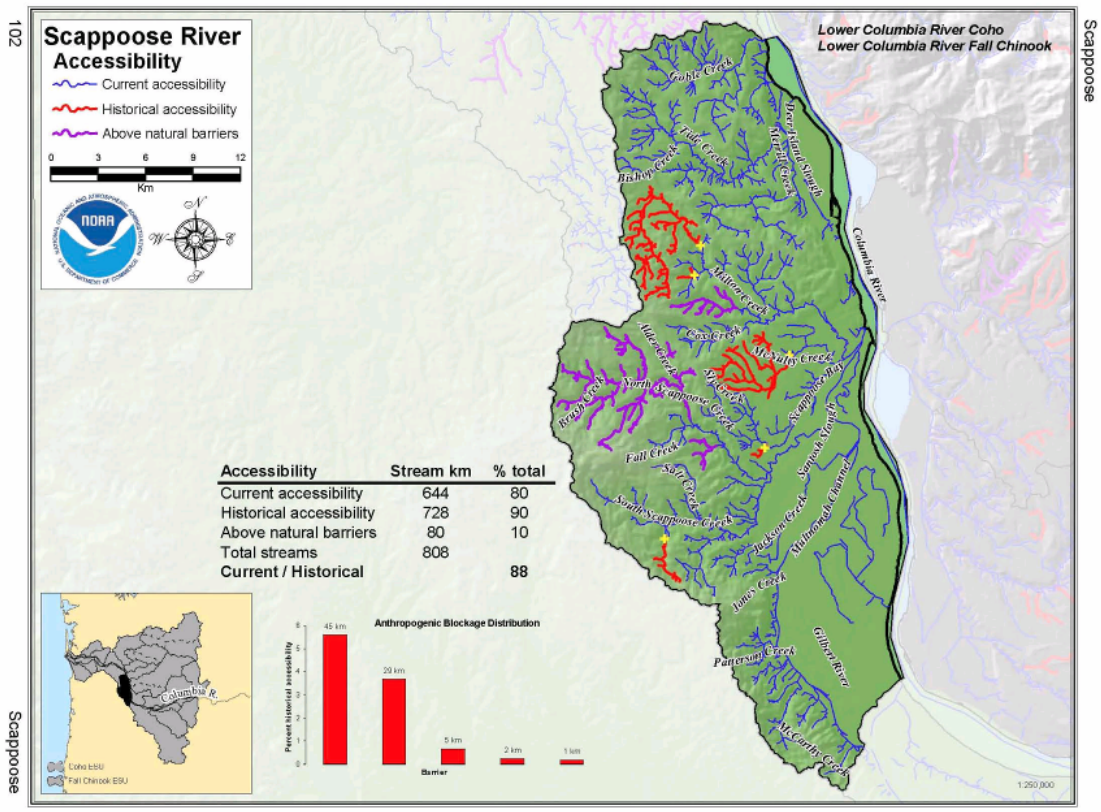

Figure 33: Scappoose Creek fall-run chinook salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Clackamas

Historical fall chinook production areas were limited to the lower mainstem and portions of the mainstem tributaries. All mainstem areas that were historically suitable for fall chinook are currently accessible (Figure 34) (ODFW 2005). Access to some smaller streams in the basin has been lost, but these areas were not productive fall chinook habitats. Habitat degradation in the basin has reduced the spatial distribution of suitable habitats for fall chinook. Habitat changes in the Willamette and Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for weighted historical productivity of suitable habitats and effects of habitat degradation on currently accessible habitats.

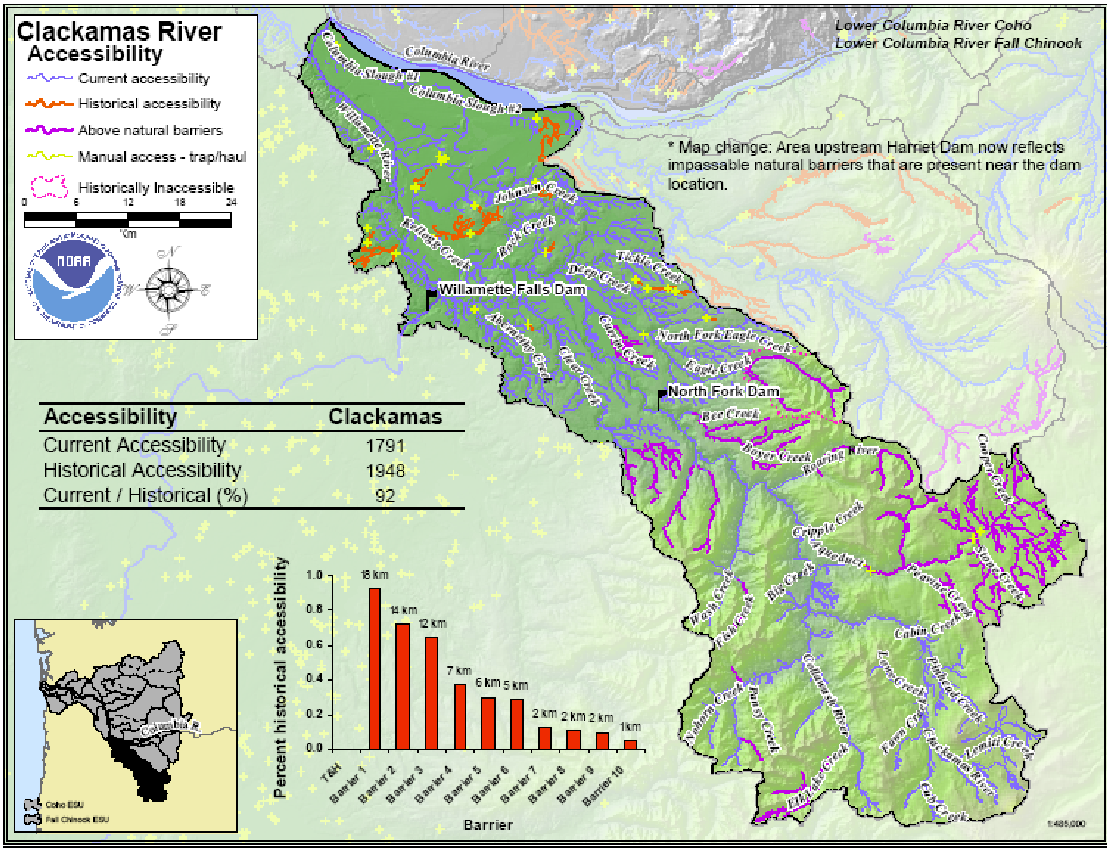

Figure 34: Clackamas fall chinook current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Sandy

Historical fall chinook production areas were limited to the lower mainstem and portions of the mainstem tributaries. Most of the core production area remains accessible (Figure 35). Portions of the historical distribution in the Bull Run River are blocked by a dam. Habitat quality remains adequate to support spawning throughout a significant portion of the accessible range. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for weighted historical productivity of suitable habitats and effects of habitat degradation on currently accessible habitats. Although a significant amount of historically *accessible* habitat is no longer accessible, the majority of habitat historically *used* (because of habitat preference) is still available, so scores were adjusted upward from the base accessibility score.

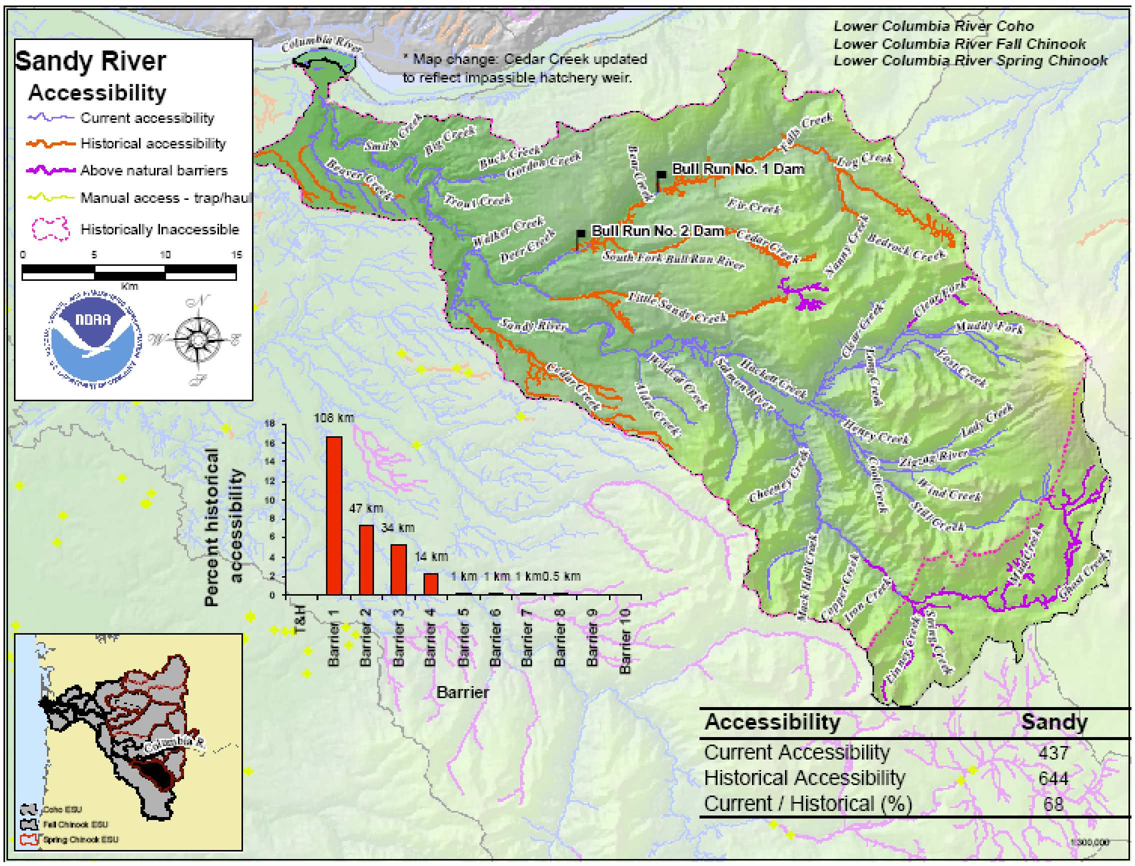

Figure 35: Sandy River fall and spring chinook and coho current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Lower Gorge Tributaries

Most of the small Columbia River gorge streams between the Sandy River and Eagle Creek remain accessible to anadromous fish but habitat availability is limited by the topography (ODFW 2005), specifically impassable waterfalls (Figure 36). Significant historical chinook production was likely limited to low gradient reaches in the lower portions of these streams (ODFW 2005). Significant chinook production occurs in nearby locations of the mainstem Columbia River and in some Washington tributaries. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores. Other local habitat alternations and development have likely reduced habitat quality in some streams. Access scores were modified for the limited area of suitable habitat and effects of habitat degradation on currently accessible habitats.

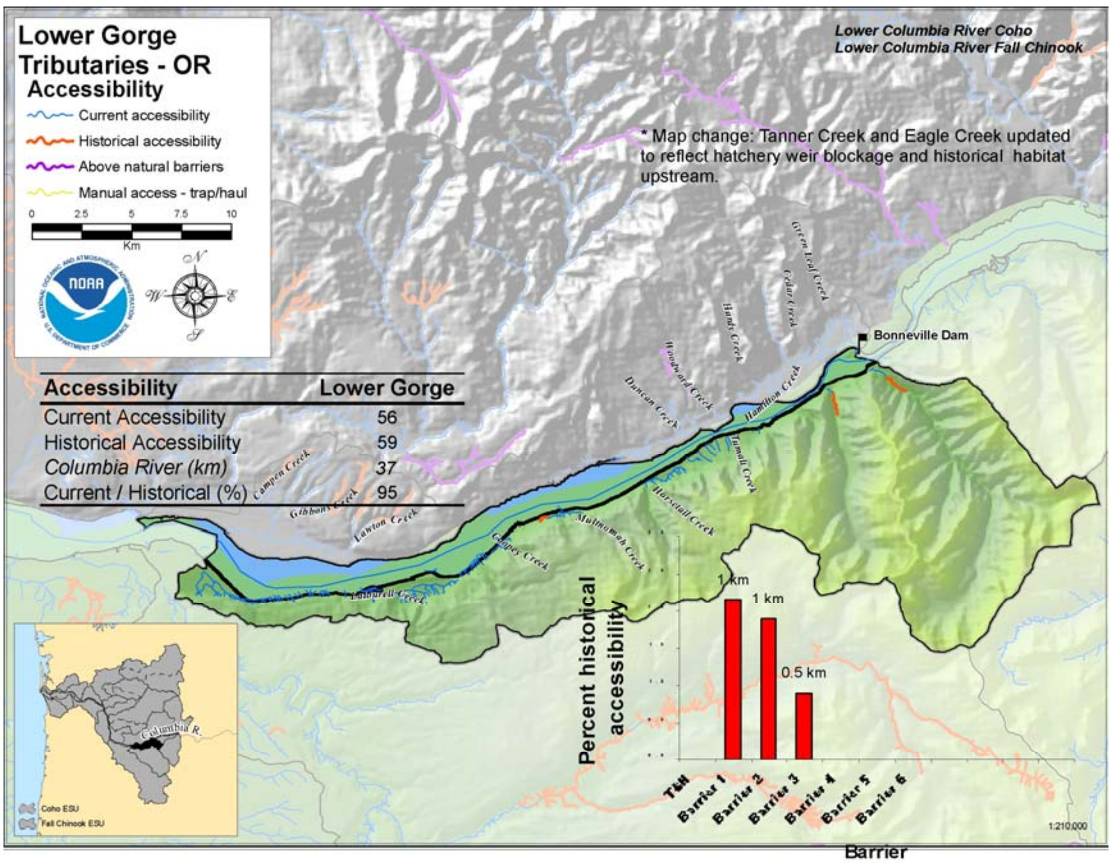

Figure 36: Lower Gorge fall chinook current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Upper Gorge Tributaries

The small Columbia River gorge streams upstream from Eagle Creek remain largely accessible but habitat is limited to the lower portions of these streams by topography and portions of the lower reaches have been inundated by the Bonneville Dam reservoir (Figure 37). Other local habitat alternations and development have likely reduced habitat quality in some streams. Access scores were modified for the limited area of suitable habitat and effects of habitat degradation on currently accessible habitats.

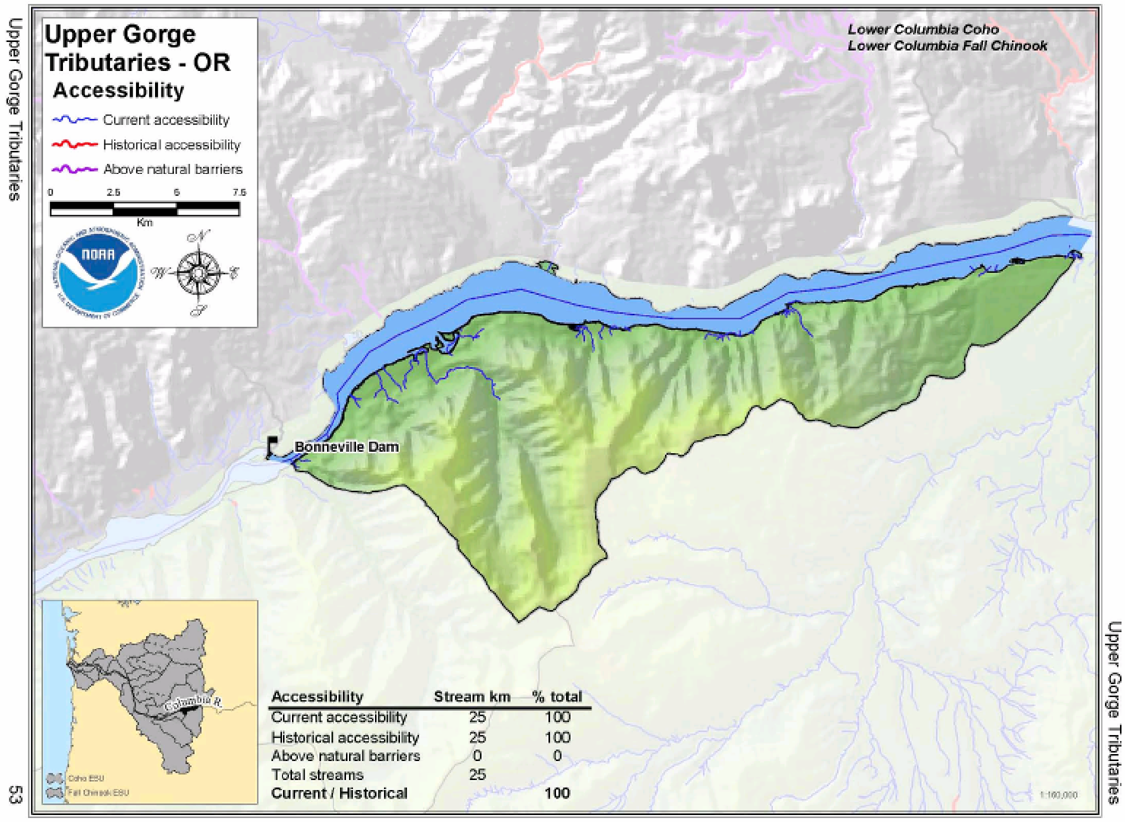

Figure 37: Upper Gorge fall-run chinook salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Hood River

Historical fall chinook production areas were limited to the lower mainstem and portions of the mainstem tributaries. All mainstem areas that were historically suitable for fall chinook are currently accessible (Figure 38)(ODFW 2005). Access to some smaller streams in the basin has been lost but these areas were not productive fall chinook habitats. Habitat degradation in the basin has reduced the spatial distribution of suitable habitats for fall chinook. Portions of the lower reaches have been inundated by the Bonneville Dam reservoir. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on fall chinook salmon and contributed to adjustments to the spatial structure scores.

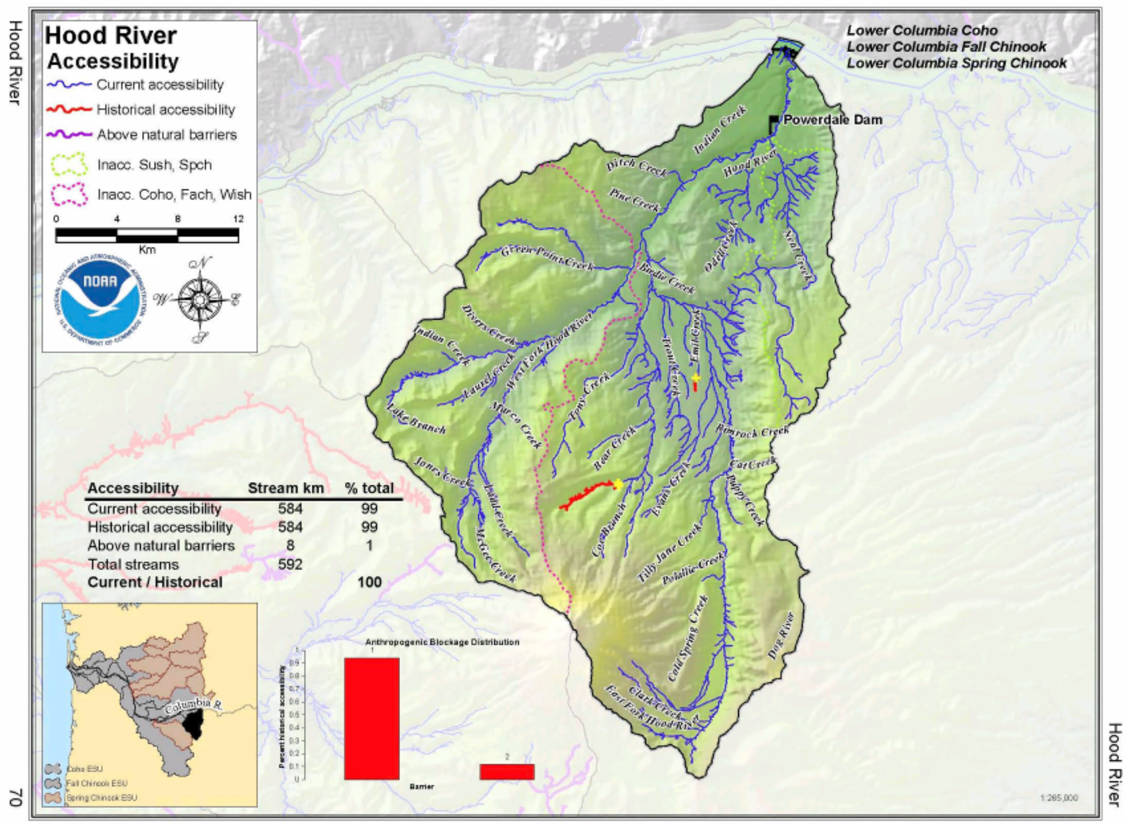

Figure 38: Hood River fall-run chinook and spring-run chinook salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict access (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Hood River (Spring)

Virtually the entire habitat accessible to spring chinook in the Hood River remains accessible today (Figure 38) (ODFW 2005). Blockages are limited to only a few headwater reaches and these streams do not represent significant historical spring chinook production areas. Habitat in this basin was likely not productive for spring chinook prior to development. The native spring chinook run was extirpated and reintroduction attempts are currently underway. Access scores were modified for the effects of habitat limitations in areas of accessible habitat. Habitat declines in the estuary were not factored into spring chinook spatial structure scores because of their life history.

SS – Sandy River (Spring)

Portions of the historical spring chinook range in the Sandy River have been blocked by dam construction in the Bull Run and Little Sandy watersheds (Figure 35). ODFW (2005) estimates that 16% of the historical chinook habitat is no longer accessible. Large areas of productive high quality habitat remain accessible to spring chinook in the remainder of the basin, particularly in the forested upper basin. Production areas are distributed among several tributaries, all of which are in Mt. Hood drainages.

SS – Criterion Summary

Populations in Sandy basin have experienced more than a 30% loss of the habitat historically accessible to chinook due to anthropogenic blockages, primarily dams on the Bull Run River (Figure 39). For the Big Creek and Scappoose Creek populations this loss is approximately 13%. For the other basins, the percent loss has been less than 10%. SS scores for each population were adjusted, where applicable, on the basis of two factors: 1) the suitability/quality of the blocked habitat with respect to chinook production and 2) the degree to which the remaining accessible habitat has been degraded from historical conditions. The adjustments and final SS scores for each population are presented in Table 15.

For the SS criterion the most probable risk category for a majority of the populations was ‘low’ as evidenced by the SS rating in Table 15 and illustrated by the placement of the widest portion of the diamonds in Figure 40. However, these diamonds also show that there is considerable assessment uncertainty. As the top and bottom of the diamond symbols illustrate, it is possible (but not probable) that all of the populations could fall into the ‘low risk’ category. Conversely, it is also possible that all populations could fall into the ‘moderate risk’ category. However, forced to make a most probable call on the overall picture for LCR chinook in Oregon with respect to this criterion we would pick the ‘low risk’ category.

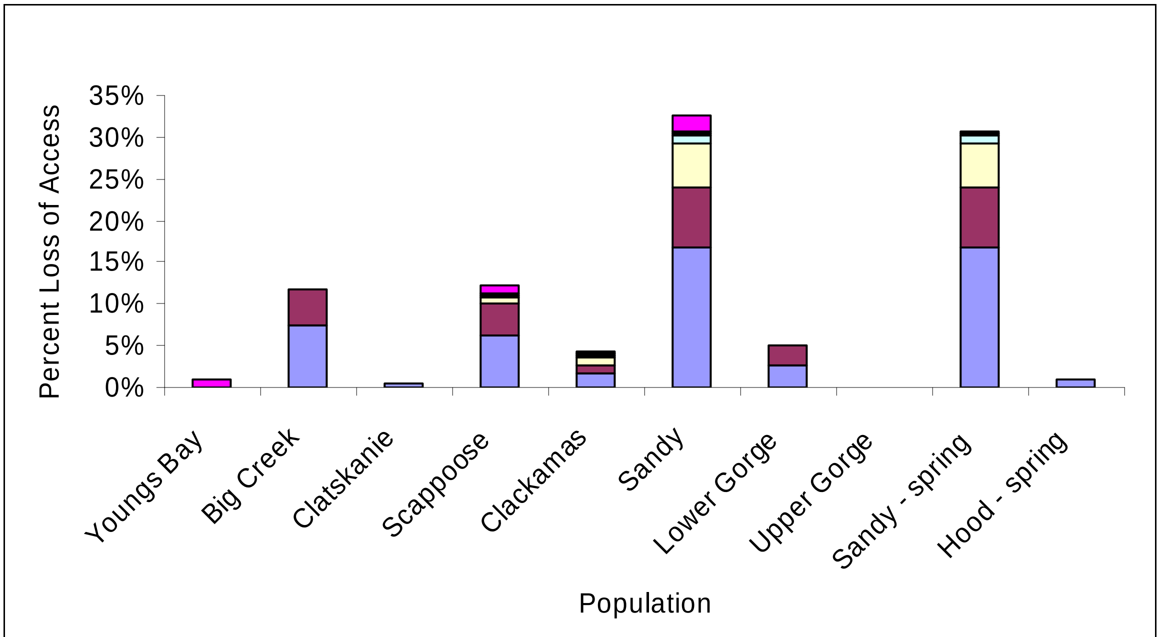

Figure 39: Percent loss in LCR spring and fall chinook accessibility due to anthropogenic blockages (based on Maher et al. 2005 with update by Sheer 2007). Each color represents a blockage ordered from largest to smallest (bottom-up). The topmost blockages, for example the blue segment of the Sandy bar, are a collection of many smaller blockages. The bar graph has been updated to reflect the removal of the largest blockage in Big Creek, still shown in the Atlas maps. Note that the pool of smaller blockages can be greater than larger single blockages.

Table 14: Spatial structure persistence category scores for LCR chinook populations.

Population	Base Access Score	Adjustment for Large Single Blockage	Adjusted Access Score	SS Rating*	Confidence in SS rating
Youngs Bay Fall	4	No	4	3	Low
Big Creek Fall	3	No	3	2.5	Low
Clatskanie Fall	4	No	4	3	Low
Scappoose Creek Fall	3	No	3	2.5	Low
Clackamas Fall	4	No	4	3	Low
Sandy River Fall	2	Yes	1.5	3	Low
Sandy River Late Fall	2	Yes	1.5	2	Low
Lower Gorge Tributaries Fall	3	No	3	2.5	Low
Upper Gorge Tributaries Fall	4	No	4	2.5	Low
Hood River Fall	4	No	4	3	Low
Sandy River spring	2	Yes	1.5	1.75	Low
Hood River spring	4	No	4	3	Low

* SS Rating considers Access Score, Historical Use Distribution, and Habitat Degradation.

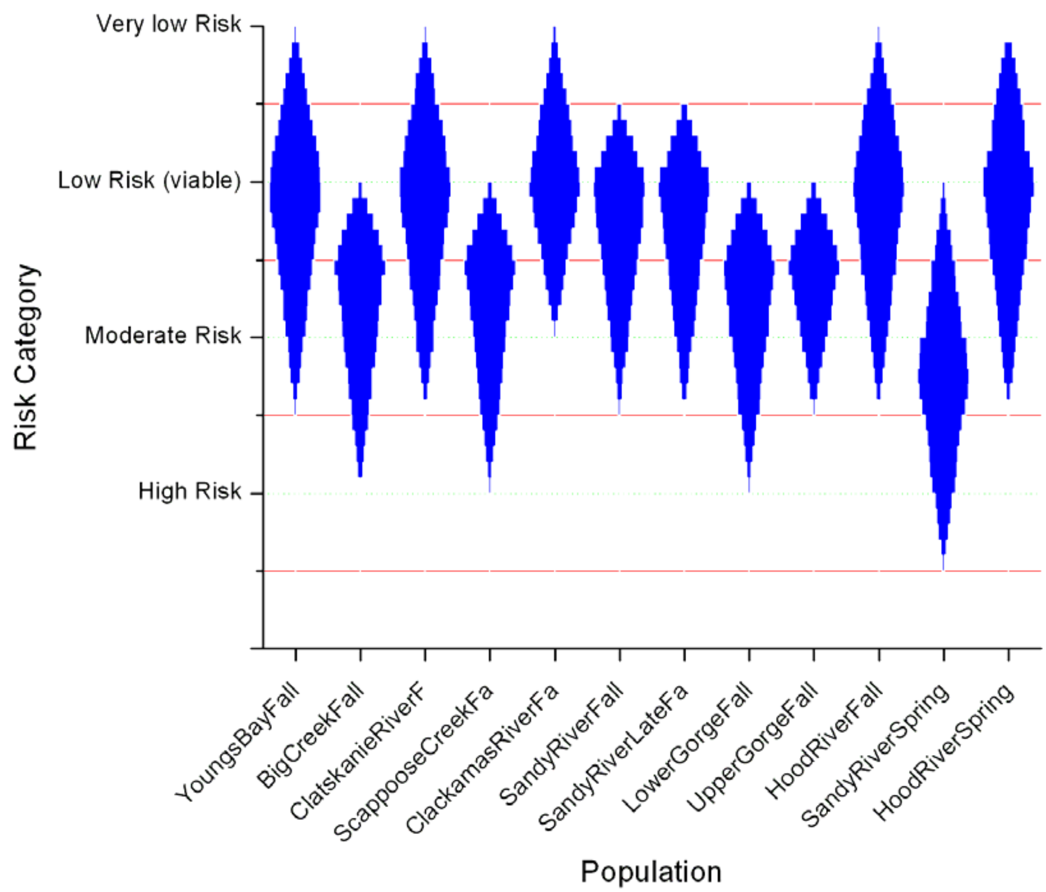

Figure 40: Lower Columbia River chinook salmon risk status summary based on the evaluation of spatial structure.

IV. Diversity

DV – Background and Overview

Of the Pacific salmon, chinook salmon exhibit arguably the most diverse and complex life-history strategies. Healey (1986) described 16 age categories for chinook salmon, 7 total ages with 3 possible freshwater ages. Two generalized freshwater life-history types were initially described by Gilbert (1912): stream-type chinook salmon reside in freshwater for a year or more following emergence, whereas ocean-type chinook salmon migrate to the ocean within their first year. Healey (1983, 1991) has promoted the use of broader definitions for ocean type and stream type to describe two distinct races of chinook salmon. Using Healey's definition, chinook salmon native to the Lower Columbia and Upper Willamette Rivers are considered to be ocean type (Myers et al. 1998). Below this stream/ocean level of diversity, run timing and geographic distribution are the most prominent life history characters used to distinguish populations. Of the five recognized run times, only three are currently observed in the Lower Columbia River: spring, fall, and late fall (it is possible that a winter run existed in the Sandy River Basin, but was extirpated). Each of these run timings is associated with a suite life history characters related to spawning site selection, age at emigration, and age at maturation.

The fall run is currently predominant in the Lower Columbia River, although historically, spring-run fish may have been as numerous as the fall run, if not more so. Fall-run fish return to the river in mid-August and spawn within a few weeks (WDF et al. 1993, Kostow 1995). These fall-run chinook salmon are often called tules and are distinguished by their dark-skin coloration and advanced state of maturation at the time of freshwater entry. Tule fall-run chinook salmon populations historically spawned in tributaries from the mouth of the Columbia River to the White Salmon and Hood Rivers and possibly farther upstream. It is also likely that fish spawned in the mainstem Columbia River above the confluence with the Willamette River. A later returning component of the fall run exists in the Lewis and Sandy Rivers (WDF et al. 1993, Kostow 1995, Marshall et al. 1995). Because of the longer time interval between freshwater entry and spawning, Lewis River and Sandy River late-fall-run chinook salmon are less mature at freshwater entry than tule fall chinook salmon at river entry and are commonly termed lower river "Brights" (Marshall et al. 1995). Confusingly, there are presently a number of other non-native fall-run chinook salmon in the Lower Columbia River that are also generally referred to as brights or "up river brights". Hatchery records and genetic analysis indicate that these fish are the descendants of introduced fall-run chinook salmon from the Rogue River (Oregon coast) and the Upper Columbia River (Priest Rapids Hatchery). With the exception of the late fall-run chinook salmon in the Lewis and Sandy Rivers we know of no information to indicate that this life-history form was historically present anywhere else in the ESU.

The majority of naturally produced fall-run chinook salmon from the Lower Columbia and Lower Willamette Rivers emigrate to the marine environment as subyearlings (Reimers and Loeffel 1967, Howell et al. 1985, Hymer et al. 1992, Olsen et al. 1992, WDF et al. 1993), although much of the current information is confounded by the inclusion of a large number of hatchery reared fish.

Historically, adult fish migrations (especially spring run migrations) were synchronized with periods of high rainfall or snowmelt to provide access to upper reaches of most tributaries where fish would hold until spawning (Fulton 1968, Olsen et al. 1992, WDF et al. 1993). The relationship between flow and run timing was recognized by early fishery biologists: “Another peculiarity in connection with the habits of this species [spring run chinook salmon] of salmon is that they will not enter any stream which is not fed by snow water . . .” (ODF 1900). Fall-run chinook salmon generally spawn in the lower reaches of larger rivers and are less dependent on flow, although early autumn rains and a drop in water temperature often provide a cue for movements to spawning areas.

Marine CWT recoveries for Lower Columbia River stocks tend to occur off the British Columbia and Washington coasts, with a small proportion of tags recovered from Alaska (Myers et al. 1998). With the exception of fish populations not native the ESU (i.e. Rogue River fall-run and Carson National Fish Hatchery (NFH) spring-run chinook salmon) and to a lesser extent the late-fall run chinook salmon there is little variation in the distribution of ocean recoveries.

DV – Youngs Bay Fall Run

Life History Traits **–** There is little information on the life history traits of fall-run chinook salmon spawning in tributaries to Young's Bay. Spawner surveys conducted in late September and early October (a timing associated with "tule" fall-run fish), have observed spawners and redds (Theis and Melcher 1995, (Takata 2005)). This spawn timing is similar to other populations in adjacent Lower Columbia River DIPs. Estimation of spawn timing is complicated by the presence of Rogue River late-fall chinook salmon released from Youngs Bay net pens and late-fall fish from coastal chinook salmon populations. Takata (2005) reported that the majority of spawning in the North Fork and South Fork Klaskanine River were Rogue River stock. Score = 3.0

Effective Population Size **–** Abundance estimates for this DIP have been based on single peak count surveys. Counts have varied from zero to several hundred fish, the majority of which are thought to be of hatchery origin. Score = 2-3

Hatchery Impacts

Hatchery Domestication (PNI) **–** There is no hatchery program in the Youngs Bay DIP that releases fall-run chinook salmon that originate from the Coastal Stratum. ODFW (2003) estimated that in the 1990s over 90% of the naturally spawning fish in this stratum were of hatchery origin. Due to the non-local origin of most hatchery fish, hatchery effects were calculated using the hatchery introgression metric. Score = NA

Hatchery Introgression **–** Hatchery programs for select area fisheries in Youngs Bay have focused on the release of Upper Willamette River spring run and Rogue River late-fall run chinook salmon. Hatcheries in adjacent watershed release a mixture of stocks, for example the Big Creek hatchery broodstock was founded with fish from the Spring Creek NFH (Gorge Stratum). Estimates of hatchery contribution to natural escapement ranges from 50-91% (ODFW 2003, Goodson 2005). Goodson (2005) suggests 90% of the fall-run chinook salmon present were Rogue River (aka Select Area Bright) fish, although it is unclear how run timing differences might limit genetic introgression. Score = 0.5.

Synthetic Approach – There is a very low genetic similarity between the fish released into this DIP and the local naturally-spawning fish. Additionally, the proportion of hatchery fish spawning naturally is very high (pHOS >> 0.50). Diversity persistence score = 0.0.

Anthropogenic Mortality – Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be relatively high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in-river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2.0.

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 3 - 4

***Overall Score* = 1.0.** The large proportion of out-of-ESU and out-of-stratum hatchery fish and the extremely low numbers of potentially native fish observed spawning strongly influenced the score. Previously: 2004 TRT 0.96; 2004 ODFW Fail < 4 criteria meet

DV – Big Creek Fall Run

Life History Traits **–** Run timing and age structure information is available for Big Creek chinook salmon, unfortunately in the absence of a pre-hatchery baseline it is difficult to identify any changes in life history diversity. Currently, fish begin freshwater entry in late August and September with spawning taking place from late-September through early/mid October (Howell et al. 1985, Olsen et al. 1992). Scale analysis indicates that the majority of the fish return as 3 and 4-year olds, with a fair number of 2-year-old males (jacks) and a limited number of 5-year-old fish (Olsen et al. 1992). These life history characteristics are similar to other fall-run chinook salmon in the Coastal stratum. Score = 4.0

Effective Population Size – Goodson (2005) and Theis and Melcher (1995) estimate that the spawning escapement to Big Creek and other streams in the DIP numbers in the thousands of fish, although most are thought to be of hatchery origin. Score = 3-4

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** The Big Creek hatchery was established in 1941 using locally returning fish as broodstock. Since 1941, 8 different stocks of fall-run chinook salmon have been released from this hatchery in addition to a number of spring-run chinook salmon (primarily from the Upper Willamette River ESU). Over 200 million fall-run chinook salmon have been released into the Big Creek Basin. For several years, releases of Rogue River bright fall-run chinook salmon were made from the Big Creek Hatchery, but were terminated because of concerns regarding the straying of these non-native fish into basins throughout the Lower Columbia River. A weir placed in the river for the collection of broodstock blocks access to much of the basin. Passage provided above the weir has been intermittent. Given existing conditions, it is unlikely that the naturally spawning fall-run chinook salmon in this basin are self-sustaining or independent. Genetically, the Big Creek Hatchery population most closely resembles fall-run chinook salmon from the Spring Creek NFH (Gorge fall-run stratum) from which it is descended. It is unclear to what degree these Spring Creek fish could have adapted to local conditions. Recently releases from Big Creek hatchery have been reduced from 10 million to 5-6 million. In 2003, 16,785 chinook returned to the hatchery rack.
>
> PNI $\leq$ 0.1. , Fitness = 0.45 Score = 1.0
>
> *Hatchery Introgression* **–** The PNI metric (#2) was utilized to account for hatchery effects Score = NA
>
> *Synthetic Approach* – Although there is a moderate genetic similarity between the fish released into this DIP and the local naturally-spawning fish, the proportion of hatchery fish spawning naturally is very high (Ph >> 0.50). Diversity persistence score = 1.0.

Anthropogenic Mortality **–** Mortality: Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be rather high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2.0**.**

Habitat Diversity – Although the quality of habitat may be severely degraded the proportion of accessible stream size reflects historical conditions, while much of the elevation diversity has been lost. Score = 3/1.

***Overall Score* = 1.0**. The large proportion of out-of-ESU and out-of-stratum hatchery fish and the extremely low numbers of potentially native fish observed spawning strongly influenced the score. Previously: 2004 TRT 0.96; 2004 ODFW Fail < 4 criteria meet.

DV – Clatskanie River Fall Run

Life History Traits – Naturally spawning fall-run chinook salmon still occur in these streams; however, the majority of these fish appear to be first generation hatchery strays (Theis and Melcher 1995). Merrill (1957) observed chinook salmon spawning just above the tidewater (Rkm 6) during October (at the time of the first survey, October 17th, there were already 7 carcasses on site). Genetic analysis of fall-run fish from these streams is not available; however, based on the marked hatchery strays recovered and geographic proximity it is likely that there would be a strong similarity to stocks released from the Big Creek hatchery and other local facilities. Score = NA

Effective Population Size **–** Index spawner surveys estimate fish density at several hundred fish per mile, which would expand to a few thousand for the whole DIP (Goodson 2005). Score = 3-4

Hatchery Impacts

> *Hatchery Domestication (PNI)* – There is no hatchery program currently operating in this DIP. Goodson (2005) reports >50% of spawning escapement is of hatchery origin, many of which originate from Big Creek (233/240 CWTs) and Elochoman (3/240 CWTs) hatchery programs (Takata 2005). PNI and fitness estimates calculated assuming that hatchery contribution has been at least this high since the initiation of the Big Creek hatchery program. PNI ≤ 0.5. Fitness = 0.75**.** Score = 2.0
>
> *Hatchery Introgression* **–** The majority of hatchery stray fall-run chinook salmon in this DIP originated from the Big Creek Hatchery (BCH) program. Although BCH fish are closely related to Spring Creek NFH fish (Gorge Strata), we have used the PNI calculated to estimate hatchery effects. Score = NA
>
> *Synthetic Approach* – The Big Creek fall-run chinook salmon that represent the majority of naturally spawning hatchery fish are probably moderately genetic similarity between the fish released into this DIP and the local naturally-spawning fish, the proportion of hatchery fish spawning naturally is high (Ph = 0.50). Diversity persistence score = 2.0.

Anthropogenic Mortality **–** Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be relatively high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in-river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2.0**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 4/4.

***Overall Score* = 1.5.** The influence of stray hatchery fish from out-of-basin programs was a major consideration estimating a diversity score. The absence of a hatchery program directly releasing fish into the DIP may provide some opportunity for local adaptation.

Previously: 2004 TRT estimate 1.31; 2004 ODFW Fail < 4 criteria met.

DV – Scappoose Creek Fall Run

Life History Traits – There is little information on historical or current life history traits or genetic characteristics. Spawner surveys have been done intermittently and give little indication of run size or trends in abundance. Parkhurst et al. (1950) observed 60-70 spawning chinook salmon on the 8th of October 1945. Spawner surveys are currently carried out in late September and early October. Score = NA

Effective Population Size – Willis (1960) estimated that the run of chinook salmon in Scappoose Creek averaged 100 fish. Goodson (2005) does not present any abundance information for this DIP, but does state that chinook salmon are present. Abundance is presumed to be low, even considering the presence of hatchery strays. Score = 1-2

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** There is no hatchery program in this DIP; however, there are a number of large fall-run chinook salmon hatcheries in nearby basins (fore example: Cowlitz Salmon Hatchery, Kalama Falls/Fallert Creek Hatchery, Lewis River Hatchery). In the absence of carcass recoveries, specific hatchery influence cannot be established. Score = NA
>
> *Hatchery Introgression* **–** Goodson (2005) does not present any quantitative estimate of the hatchery contribution to escapement, and simply states that the hatchery influence is "excessive". Score = 2.0
>
> *Synthetic Approach*– The majority of hatchery fish that are likely to stray into this DIP probably have a low level of genetic similarity. The proportion of hatchery fish spawning naturally is unknown, but thought to be high (0.75>Ph>0.30). Diversity persistence score = 1.0.

Anthropogenic Mortality **–** Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be relatively high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in-river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2.0**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 4/3

***Overall Score* = 1.5.** Small population size and the influence of a relatively large contribution by hatchery origin fish influenced this score. Due to the poor quantity and quality of information available this score should be considered an interim estimate. Previously: 2004 TRT estimate 1.18; 2004 ODFW Fail < 4 criteria meet

DV – Clackamas River Fall Run

Life History Traits **–** Fall-run chinook salmon were native to the lower Willamette River and its principal tributary, the Clackamas River, and likely other tributaries below Willamette Falls. A tule fall-run existed in the lower Clackamas River until the 1930s (Parkhurst et al. 1950, Gleeson 1972). Dimick and Merryfield (1945) reported that these fish entered the Willamette River in September and October and spawned soon after entering the Clackamas River. Murtagh et al. (1992) indicate that historical records suggest that fall-run chinook salmon may have spawned from September to November. There is little current information available on life history traits, in part because of the inability to distinguish between fall-run and late-spawning spring run chinook salmon. Score = NA.

Effective Population Size **–** Recent spawning escapement estimates indicate that less than 100 fall-run chinook salmon spawn in the lower Clackamas River. Additionally, it is not clear if the existing population is sustainable. Score = 1-2

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** There is currently no hatchery program for fall run fish in the Clackamas or lower Willamette River. Fall-run chinook salmon from Lower Columbia River hatchery stocks were released from 1952 to 1981 to reestablish the run. Hatchery releases of fall chinook salmon last occurred in the 1980s allowing the existing population as least five generations to adapt to local conditions. Presently, the run appears to be maintained through natural reproduction, ODFW (1998) estimated that there were few if any fall-run hatchery fish spawning in the Clackamas River. Score = NA.
>
> *Hatchery Introgression* **–** With the termination of fall run releases into the Clackamas and Willamette River, the level of hatchery influence is thought to be low. There is some potential for interbreeding between spring and fall-run fish in the lower Clackamas River. Score = 3.0.
>
> *Synthetic Approach* – There are no releases of hatchery fall-run fish into the Clackamas River, although a number of spring-run chinook salmon are recovered in the lower river. Genetic similarity between the hatchery fish in this DIP and the local naturally-spawning fish, the proportion of hatchery fish spawning naturally is low or very low. While the number of stray fish may be low, the population of naturally-spawning fish is also very low. The relative proportion of hatchery fish could be high, perhaps in the range of 25% to 50%. Diversity persistence score = 0-2.

Anthropogenic Mortality – Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be rather high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River and lower Willamette River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2.0.

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 4/4.

***Overall Score* = 2.0.** Small effective population size is the primary concern for this DIP, continued low escapements may result is a substantial genetic bottleneck. 2004 TRT estimate 1.34; 2004 ODFW Fail < 4 criteria meet

DV – Sandy River Fall Run

Life History Traits **–** There is considerable debate regarding the historical presence of early (tule) fall-run chinook salmon in the Sandy River. Howell et al. (1985) and Olsen et al. (1992) indicate that although tule fall run have not been stocked since 1977, early spawning fall-run chinook salmon established from those releases and/or strays from current releases continue to spawn below Marmot Dam. Score = NA

Effective Population Size **–** Surveys of "early" fall-run fish in the Sandy River Basin have been intermittent, but it is likely that on average one to a few hundred fish spawn in the basin each year (Theis and Melcher 1995). Score = 2.0

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** There is currently no hatchery program for fall-run chinook salmon in this DIP. It has been suggested that this is a feral population, founded from releases of LCR fall-run hatchery fish from 1930s to the 1970s. Score = NA
>
> *Hatchery Introgression* **–** Uncertainty regarding the origin of fall-run chinook salmon in the Sandy River complicates estimates of out-of-stratum introgression. Few carcasses are recovered and information on the origin of spawning fish is unavailable Score = 2.0
>
> *Synthetic Approach* – Hatchery fall-run chinook salmon have not been released into this basin for some time – it is unclear whether the fish presently spawning are native or feral. Fall-run (early) fish currently straying into this basin are likely to have a level of genetic similarity relative to naturally-spawning fish, the proportion of hatchery fish spawning naturally is very high ($0.10 < Ph < 0.30$). Diversity persistence score = 2.0.

Anthropogenic Mortality **–** Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be rather high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River and lower Willamette River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = NA**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 4/3.

***Overall Score* = 1.0.** Although the effective population size of this population is relatively low, it does appear to be self-sustaining with little hatchery introgression. The origin of this population remains to be clarified. Previously: 2004 TRT 1.16; 2004 ODFW not rated/introduced.

DV – Lower Gorge Fall Run

Life History Traits **–** There is some historical information available for Lower Gorge tributaries. Evermann and Meek (1898) observed "considerable numbers" of chinook in Eagle and Tanner Creeks. Bowers (1902) reported that chinook salmon had entered Eagle and Tanner Creeks by 18 September 1901. Currently, there are fall-run chinook salmon that spawn off of Ives Island, in the mainstem Columbia River below Bonneville Dam (Van Der Naald et al. 2001). These fish appear to have a typical fall-run spawn timing (late September and October). Score = NA.

Effective Population Size – Lower Gorge tributaries are only intermittently surveyed, returns to the hatcheries number in the thousands, but the origin of many of these broodstocks is uncertain. Several hundred full-run fish spawn in the Ives Island vicinity. Score = 2.0

Hatchery Impacts

Hatchery Domestication (*PNI*) – Populations in the Lower Gorge tributaries are likely heavily influenced by hatchery fish straying from Bonneville Hatchery and Spring Creek NFH. In 2003, some 2,852 fall-run fish returned to the Bonneville Hatchery, this was in addition to the 21,297 Upriver Bright fall-run chinook salmon that returned to the hatchery. Spring Creek NFH fall-run returns normally range from 5,000 to 15,000 fish. In addition, there are a number of other hatchery programs that release both Lower Columbia River fall run and URB fall run fish. Although no estimate is available it is likely that the hatchery contribution to natural spawning escapement is over 50%.2.

PNI ≤ 0.1. Fitness = 0.45. Score = NA

Synthetic Approach – Fall-run hatchery fish straying into this area could be from either local tule hatchery programs or upriver bright programs. There is likely a low or very low level of genetic similarity between the fish released into this DIP and the local naturally-spawning fish, the proportion of hatchery fish spawning naturally is very high (Ph >> 0.50). Diversity persistence score = 0.0.

Hatchery Introgression – Several million URB fall-run fish are released into the mainstem Columbia River near Bonneville Dam. Although there is some temporal separation in spawn timing, there is potential for interbreeding. It is not known the degree to which URB fish stray and spawn in Lower Gorge tributaries, although there is a sizable aggregation (several hundred fish) that spawn off of Ives Island. Score = NA

Anthropogenic Mortality – Harvest impacts on fall-run chinook salmon in the Lower Columbia River, have been and continue to be rather high. Recent total harvest for LRH stocks was 47.4% (1999-2002), with nearly half of that taking place in inshore and in river fisheries, where there is some potential for gear-related selection (especially with gillnets). Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2.0**.**

Habitat Diversity – The proportion and character (elevation and stream size) of accessible habitat is somewhat reduced from historical conditions. Score = 4/3

***Overall Score* = 2.0.** There are a number of potential factors that could negatively influence diversity; unfortunately, there are few estimates available to quantify the effects of these factors. This evaluation focused on the Oregon side of the DIP. Previously: 2004 TRT 0.83, 2004 ODFW fail, 4-5 criteria met – combined with Hood River and Upper Gorge Tributaries

DV – Upper Gorge Fall Run

Life History Traits – There is some information available for Upper Gorge tributary chinook salmon, most of which comes from the Washington side of this DIP. Chinook salmon were observed migrating up the Big White Salmon River on 4 September 1896. Hatchery records from the Wind River Hatchery (1928-1938) indicate that eggs were collected from early September to mid-October, with a peak in late September. There is little information on the existing fall-run chinook salmon life history characteristics. Score = NA

Effective Population Size **–** Tributaries in the Upper Gorge are only intermittently surveyed. Observed fish counts range from 0 to a few hundred fish. Score = 1-2

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** Populations in the Upper Gorge tributaries are likely heavily influenced by hatchery fish straying from Bonneville Hatchery, Little White Salmon NFH, and Spring Creek NFH. In 2003, some 2,852 fall-run fish returned to the Bonneville Hatchery, this was in addition to the 21,297 Upriver Bright fall-run chinook salmon that returned to the hatchery. Spring Creek NFH fall-run returns normally range from 5,000 to 15,000 fish. In addition, there are a number of other hatchery programs that release both Lower Columbia River fall run and URB fall run fish. Although no estimate is available it is likely that the hatchery contribution to natural spawning escapement is well over 50%.
>
> PNI ≤ 0.1, Fitness = 0.45 Score = 1.0
>
> *Hatchery Introgression* **–** Several million URB fall-run fish are released into the mainstem Columbia River near Bonneville Dam. Although there is some temporal separation in spawn timing, there is potential for interbreeding. URB fish are known to spawn in tributaries on the Washington side of this DIP, and it is likely that they do likewise on the Oregon side. Score = 2.0
>
> *Synthetic Approach* – Fall-run hatchery fish straying into this area could be from either local tule hatchery programs (Spring Creek NFH) or upriver bright programs. There is likely a low or very low level of genetic similarity between the fish released into this DIP and the local naturally-spawning fish, the proportion of hatchery fish spawning naturally is very high (Ph >> 0.50). Diversity persistence score = 0.0.

Anthropogenic Mortality **–** Fish returning to the Upper Gorge tributaries are subject to both ocean and in-river fisheries. Total harvest rate averaged 66% (1999-2002), with approximately half of the catch being from net fisheries. Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2-3**.**

Habitat Diversity **–** Habitat diversity in this DIP has been most strongly affected by the filling of the Bonneville Pool and the loss of much of the spawning rearing habitat for fall-run chinook salmon. Currently, the habitat model is being modified to account for this loss. Score = 3/2

***Overall Score* = 1.0.** There are a number of potential factors that could negatively influence diversity, unfortunately there are few estimates available to quantify the effects of these factors.

Previously: 2004 TRT 0.83, 2004 ODFW fail, 4-5 criteria met – combined with Hood River and Upper Gorge Tributaries.

DV – Hood River Fall Run

Life History Traits **–** Direct Measures: No information available. Score = NA.

Effective Population Size **–** Based on counts at Powerdale Dam (RKm 6), the average escapement for the past 13 years has been 26 fish. Since some spawning habitat exists below the dam, it is possible that the escapement is somewhat higher. Score = 1.0
Hatchery Impacts

Hatchery Domestication (PNI) **–** There is no hatchery program in the Hood River basin for fall-run chinook salmon. Score = NA

Hatchery Introgression **–** Estimates of the hatchery-origin fish contribution to escapement varies considerably from year to year, but on average represents 12% of the run. Since this estimate is based on visual detection of adipose fin marks it is likely that the actual percentage is somewhat higher. Score = 2.0

Synthetic Approach – Hatchery fish straying into the Hood River are probably upriver bright fall-run chinook salmon, although it is possible that some Spring Creek fish also stray into the Hood River. There is likely a very low level of genetic similarity between the fish released into this DIP and the local naturally-spawning fish. The proportion of hatchery fish spawning naturally is relatively low ($0.10<Ph<0.30$)). Diversity persistence score = 1.0.

Anthropogenic Mortality **–** Fish returning to the Upper Gorge tributaries and Hood River are subject to both ocean and in-river fisheries. Total harvest rate averaged 66% (1999-2002); with approximately have of the catch being from net fisheries. Habitat changes in the estuary and mainstem Columbia River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 1.0**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 4/4

***Overall Score* = 0.5.** Small Ne, hatchery impacts, and high harvest rates all contribute to a poor diversity score for this DIP. Previously: 2004 TRT 1.24, 2004 ODFW fail, 4-5 criteria met – combined with Hood River and Upper Gorge Tributaries

DV – Sandy River Late Fall Run

Life History Traits **–** Late-fall chinook salmon return in September and October and spawn from late-November to February (Howell et al. 1985). Late-fall fish also appear to mature at an older age than early-run fish, with the majority of fish maturing at 4 or 5 years of age (Fulop 2000). There are reports of a winter-run in the Sandy River, although Kostow (1995) suggests that they have been extirpated. It is also possible that the winter-run chinook salmon observed are the "tail-end" of the late returning fall-run fish. Late returning bright fish in the Lewis River have been observed spawning as late as April. Late-fall run fish appear to emigrate as subyearlings. Little is know about the distribution of outmigration timing within the first year. Score = 3.0.

Effective Population Size – This population varies from several hundred to a few thousand. The average abundance for the last 30 years has been over 900 fish (Goodson 2005). There have been a number of years when abundance has declined to below 100 fish. The run of late-returning fall run fish may have historically been over 5,000 fish. Surveys during 2003/2004 resulted in a peak count of 281 fish, 54% of the 10-year average (Takata 2005). Score = 2-3.

Hatchery Impacts

Hatchery Domestication (PNI) – There has been no artificial supplementation of the late-returning fall run. Genetic analysis indicates a strong association between Lewis and Sandy River late-returning fall-run chinook salmon, and these two populations cluster with other Lower Columbia River populations. Score = NA

Hatchery Introgression – There is no hatchery program for late-fall run chinook salmon. Although there is a spring-run program in the Sandy River Basin and fall-run programs in neighboring basins there is little chance of introgression due to differences in run and spawn timing. Score = 4.0

Synthetic Approach – There is no hatchery program in the Sandy River for late-fall run chinook salmon. Hatchery strays are likely to be local tule fall run fish with a low level of genetic similarity relative to the local naturally-spawning fish. Additionally, the proportion of hatchery fish spawning naturally is very low ($Ph < 0.05$).

Diversity persistence score = 4.0.

Anthropogenic Mortality **–** Late-run fall chinook salmon are captured in many of the same ocean fisheries as their early fall-run counterparts. Overall, inshore sport and net harvest impacts are somewhat less for late-fall run fish. From 1999-2002, the average harvest rate for late-fall run fish was 30.7%, using Lewis River fish as a proxy. Habitat changes in the estuary and mainstem Columbia River and lower Sandy River may also have had an influence on juvenile outmigration strategies due to the loss of specific habitats or a reduction in the capacity of existing habitat. Score = 2-3**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions.

Score = 3/3.

Overall Score = 3. Recurring low abundance bottlenecks and the potential for habitat-influenced changes in life history categories were considered to be major factors influencing the diversity score. Previously: 2004 TRT 1.68, 2004 ODFW fail, 4-5 criteria met.

DV – Sandy River Spring Run

Life History Traits **–** Hatchery records indicate that Sandy River spring-run chinook spawned from July to September (ODF 1903). Recent observation indicates that adult spring-run chinook return to the freshwater from May to August and spawn from September to October (Olsen et al. 1992, ODFW 2003). This change in spawn timing is thought to be related to introductions of Upper Willamette River spring-run hatchery fish. Score = 2.0.

Effective Population Size **–** The Sandy River historically had a very large run of spring run chinook salmon. Run size for the Sandy River Basin may have been in excess of 12,000 fish (Mattson 1955). Goodson (2005) estimated the 28-year average abundance at 1,579 fish. Score = 3.0.

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** Hatchery programs have produced spring-run chinook salmon in the Sandy River Basin since the early 1900s. A number of out-of-basin sources have been integrated into the hatchery broodstock (especially from the Upper Willamette River). Hatchery fish that are now being released are externally marked and will be intercepted at Marmot Dam when they return (ODFW 1998). Hatchery fish are not allowed to pass above Marmot Dam (Rkm 43), although examination of otoliths from "unmarked" fish indicated that nearly 20% of the fish being passed over were of hatchery origin (Goodson 2005). Below Marmot Dam, over half of the naturally spawning fish were of hatchery origin, although it is not known how successful these spring-run fish were in the lower river. ODFW is currently replacing the existing Upper Willamette River derived spring-run chinook salmon with naturally produced spring-run adults returning to Marmot Dam. Genetic analysis of naturally spawning fish from the Sandy River suggested that the Sandy River population was genetically intermediate between Upper Willamette River populations and Lower Columbia River spring-run populations. Furthermore, there was little genetic resemblance between the spring-run and late "bright" fall-run fish in the Sandy River Basin. In other Lower Columbia River and coastal basins there is a tendency for different run times in a basin to have evolved from a common source. The Sandy River Basin would be a deviation from this pattern. Microsatellite DNA data indicated that the Sandy River spring-run was genetically distinguishable for the Clackamas Hatchery spring-run broodstock; however, the degree of differentiation was much less than that between spring runs in the Sandy and Yakima Rivers. Bentzen et al. (1998) concluded that although some interbreeding between the Upper Willamette River and Sandy River stocks had occurred, the Sandy River population still retained some of its original genetic characteristics. PNI $\leq$ 0.65 (above dam), 0.25 (below dam), Fitness = 0.85 (above dam), Score = 2.5

> *Hatchery Introgression* **–** Introductions of Upper Willamette River spring-run chinook salmon increased considerably during the 1960s and 1970s. Releases of hatchery fish in the upper Sandy River (above Marmot Dam) have been terminated, it is unclear to what degree the introduction of Willamette River fish into the Sandy River basin has left a genetic legacy of non-local life history characters. Score = 2.0.

Synthetic Approach – The current Sandy River spring-run hatchery broodstock was recently derived from naturally-spawning native spring run fish. There is likely a moderate level of genetic similarity between the fish released into this DIP and the local naturally-spawning fish. Although a higher level of similarity is normally applied, because of the legacy of non-native Upper Willamette spring run the level was held at "moderate.", the proportion of hatchery fish spawning naturally is low (0.10<Ph <0.30). Diversity persistence score = 4.0.

Anthropogenic Mortality **–** Harvest rates for Sandy River spring-run chinook salmon are thought to be similar to Upper Willamette River spring run populations (ODFW 2003). For the period 1999-2002 the harvest rate averaged 40.7%, with a small proportion of that occurring in in-river net fisheries. As with other ocean-type populations, changes in habitat conditions in the Sandy River and mainstem Columbia river and estuary may have an impact on juvenile life histories. Score = 3-4.

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score = 3/3.

***Overall Score* = 2.5.** Habitat changes and the legacy of non-local hatchery introductions most dramatically affected the diversity score. Previously: 2004 TRT estimate 1.64; 2004 ODFW fail, 4-5 criteria met.

DV – Hood River Spring Run

Spring-run chinook salmon in the Hood River are believed to have been extirpated (Kostow 1995, Kostow et al. 2000). Fish from a number of different hatcheries have been released into the Hood River Basin to reestablish a spring run. From 1985 to 1992, over one million fish were released into the Basin from the Carson NFH and the ODFW Looking glass Hatchery (ODFW Stock #81, a Carson NFH derivative). Currently, fish from the Round Butte Hatchery (Deschutes River, Middle Columbia River Spring-Run ESU) are being released into the Hood River Basin as part of a reintroduction program. Fish from the Round Butte introductions and their descendants are not considered part of the Lower Columbia River ESU, and although there appears to be some natural production it is still uncertain if the existing population is sustainable. The existing spring-run population is thought to be wholly derived from Deschutes River spring-run chinook salmon. The existing spring-run is not considered part of the ESU and was not evaluated.

Overall Score = 0.0.

DV – Criterion Summary

With the exception of populations in the Sandy and Clatskanie Rivers, it is possible that most populations in Oregon's portion of this chinook ESU have been either lost or depressed to levels that are currently undetectable. This loss of genetic resources and high incidence of hatchery strays in many of these basins are the primary reasons that 10 of the 12 populations scored so low and fall into a most probable risk category of 'moderate' or 'high' (Figure 41). Only the late fall and spring chinook populations in the Sandy meet the viable threshold, and just barely so. Because of the uncertainty associated with the population ratings for the DV criterion, the possibility exists that all except one of the populations fall into the 'high risk' category, as illustrated by the placement of the lower portion of the diamonds in Figure 41. In light of these results, we conclude that the most probable DV risk classification for Oregon's LCR chinook populations is 'high'.

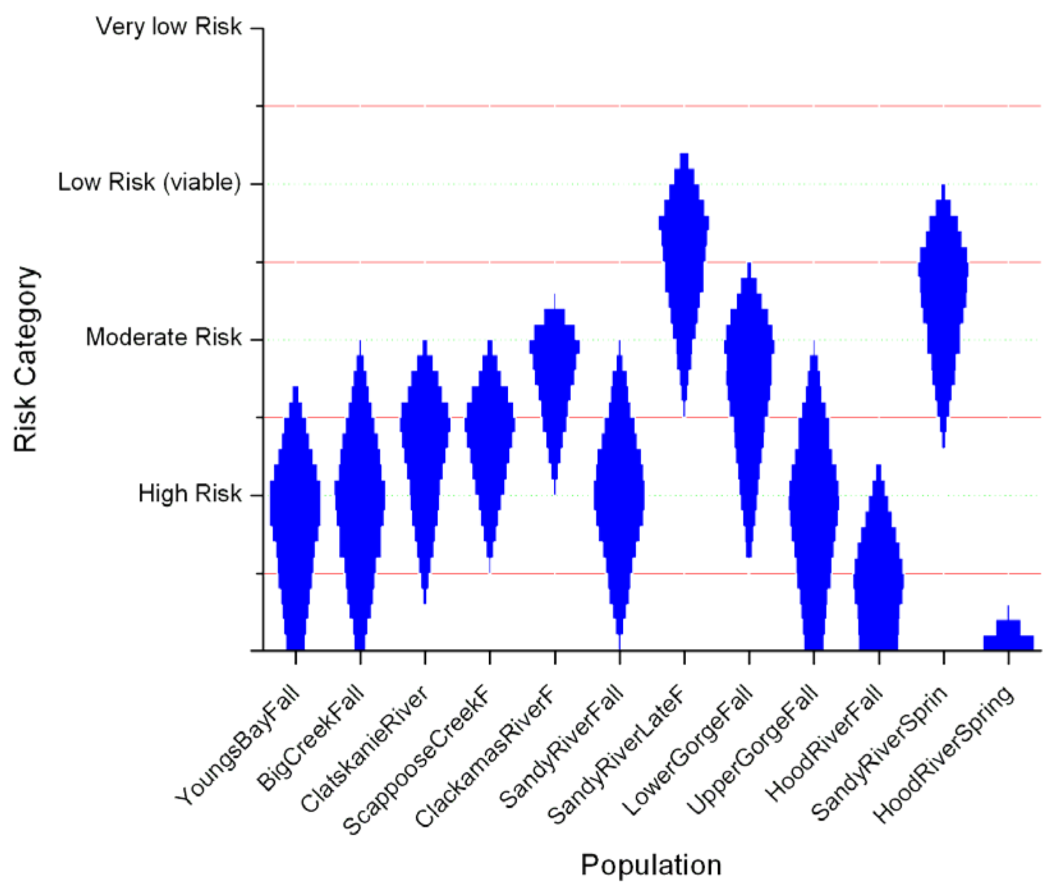

Figure 41: Lower Columbia River chinook salmon risk summary based on the evaluation of diversity only.

V. Summary of Population Results

When the three criteria scores were combined for all the populations, the results indicated that the risk of extinction for LCR chinook in Oregon's portion of the ESU is high (Figure 42 and Figure 43). On a population by population basis, a most probable classification of moderate was obtained for only two populations. Ten of the populations were clearly in the high risk category. In addition, their 'high risk' classification was made with considerable certainty as evidenced by the relatively shortened aspect of the diamonds representing population status. Overall, these chinook populations can be characterized as having a high risk of extinction.

Although a final ESU score is not possible without an assessment of Washington chinook populations using the same methodology, we expect that the overall finding would be similar our results for the Oregon populations. In all likelihood the extinction risk for the combined LCR chinook ESU is high.

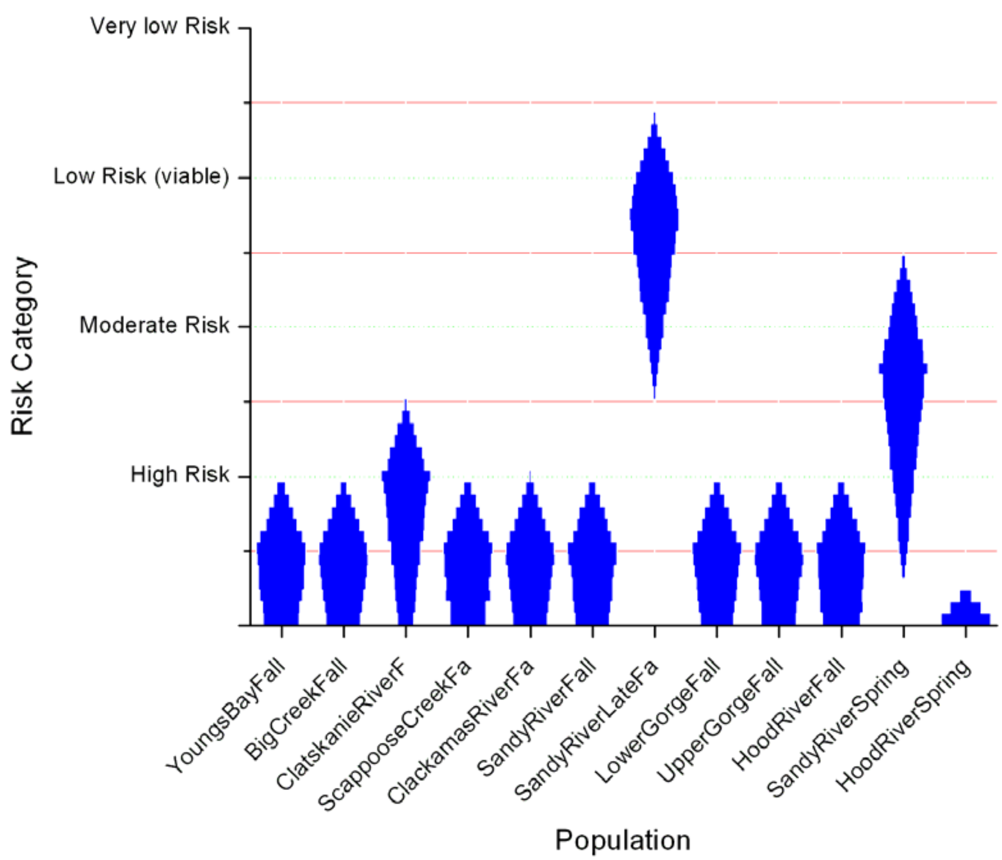

Figure 42: Oregon Lower Columbia River populations status summaries.

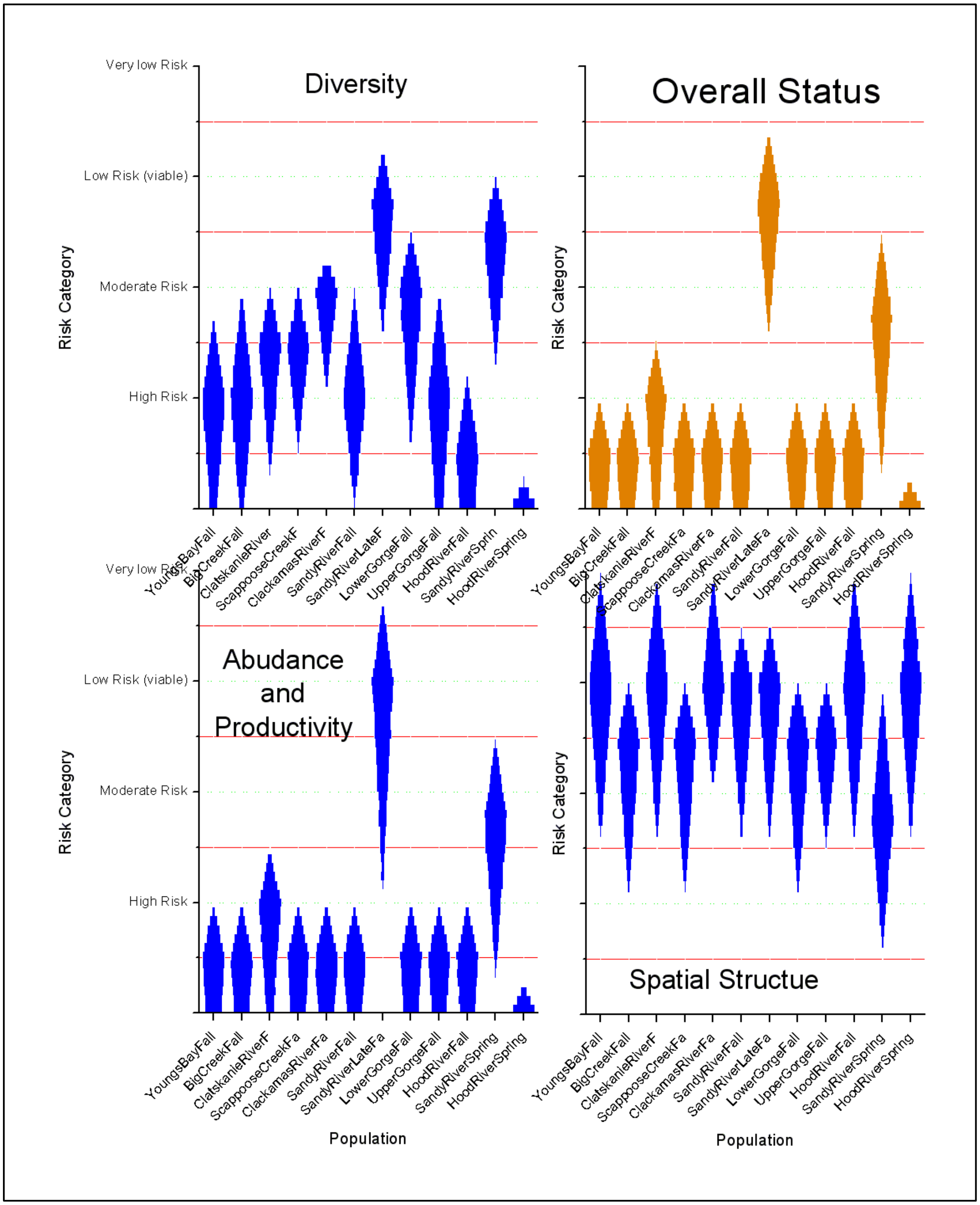

Figure 43: Oregon Lower Columbia River chinook salmon status graphs and overall summary.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 3: Columbia River Chum

September 2007

Paul McElhany[1], Mark Chilcote[2], James Myers[1], Ray Beamesderfer[3]

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

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I. ESU Overview and Historical Range

Based on TRT analysis, the Oregon portion of the CR chum ESU historically contained 8 populations (Figure 1). Historically, over a million chum returned in some years to the Columbia River (McElhany 2005). Recently only a few hundred to a few thousand chum have returned each year to the Columbia, mainly to the Washington side of the Columbia (McElhany 2005). The chum in Washington occur primarily in Grays River, in areas immediately below Bonneville Dam and, to a lesser extent, under the I-205 bridge near Vancouver. All of the historical Oregon side populations are considered extirpated or nearly so. Because of the near universal lack of chum in Oregon, this section on the chum ESU differs somewhat from the sections describing other ESUs in this report. Rather than a population-by-population analysis, we provide a brief description of chum abundance, spatial structure and diversity, followed by a summary of population status.

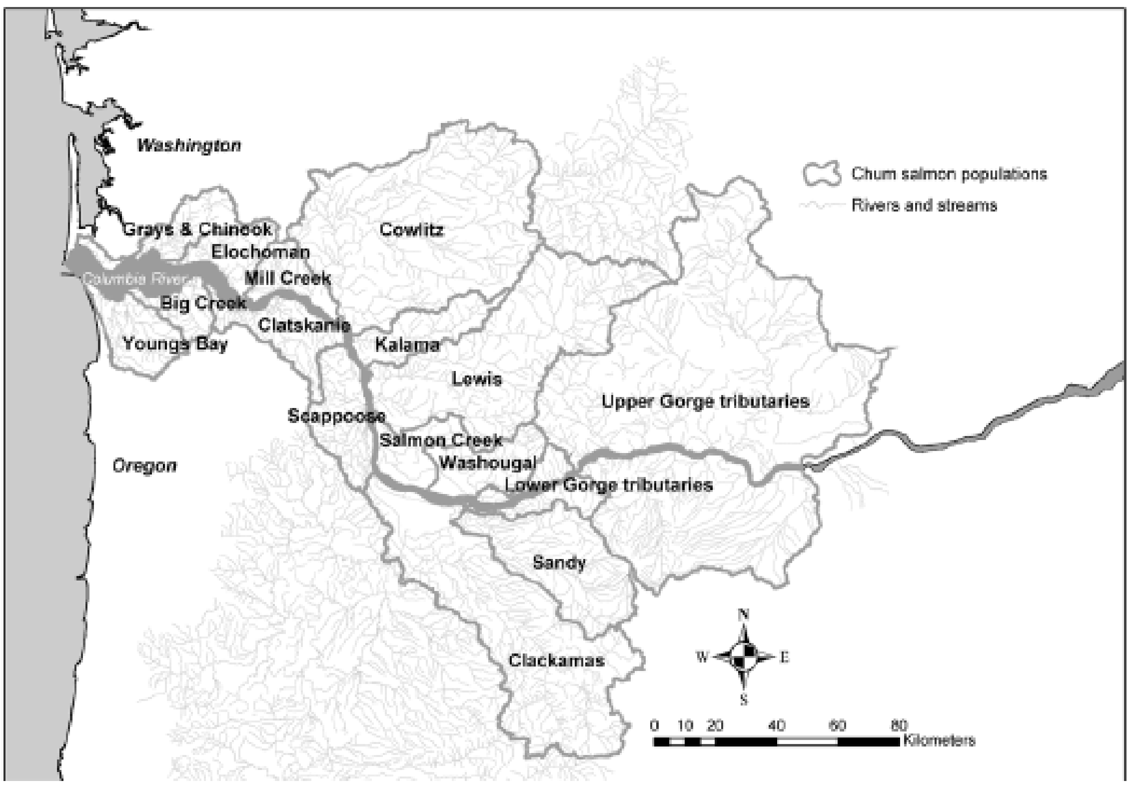

Figure 1: Map of Lower Columbia River fall chinook salmon populations.

II. Abundance and Productivity

There have been few surveys focused on Columbia chum in Oregon. However, chum are seen occasionally in Oregon and chum may be intercepted at hatchery weirs or at dam passage faculties (e.g. North Fork dam on the Clackamas River or Powerdale dam in the Hood River). In 2000, ODFW did conduct a survey focused on chum (Figure 2). Out of 30 sites surveyed, only one chum was observed (Muldoon et al. 2001).

A time series of returns is available for chum trapped at the Big Creek hatchery weir (Figure 3). Except for 2006, only a handful of fish have shown up at the facility each year and in some years no fish have appeared. It is unclear if the fish observed at the Big Creek weir were produced in Oregon or whether they are strays from the naturally producing population at Grays River across the Columbia in Washington. In 1999, a chum hatchery program was initiated in Grays River, so an unknown fraction of the fish observed in 2003-2006 are likely of hatchery origin.

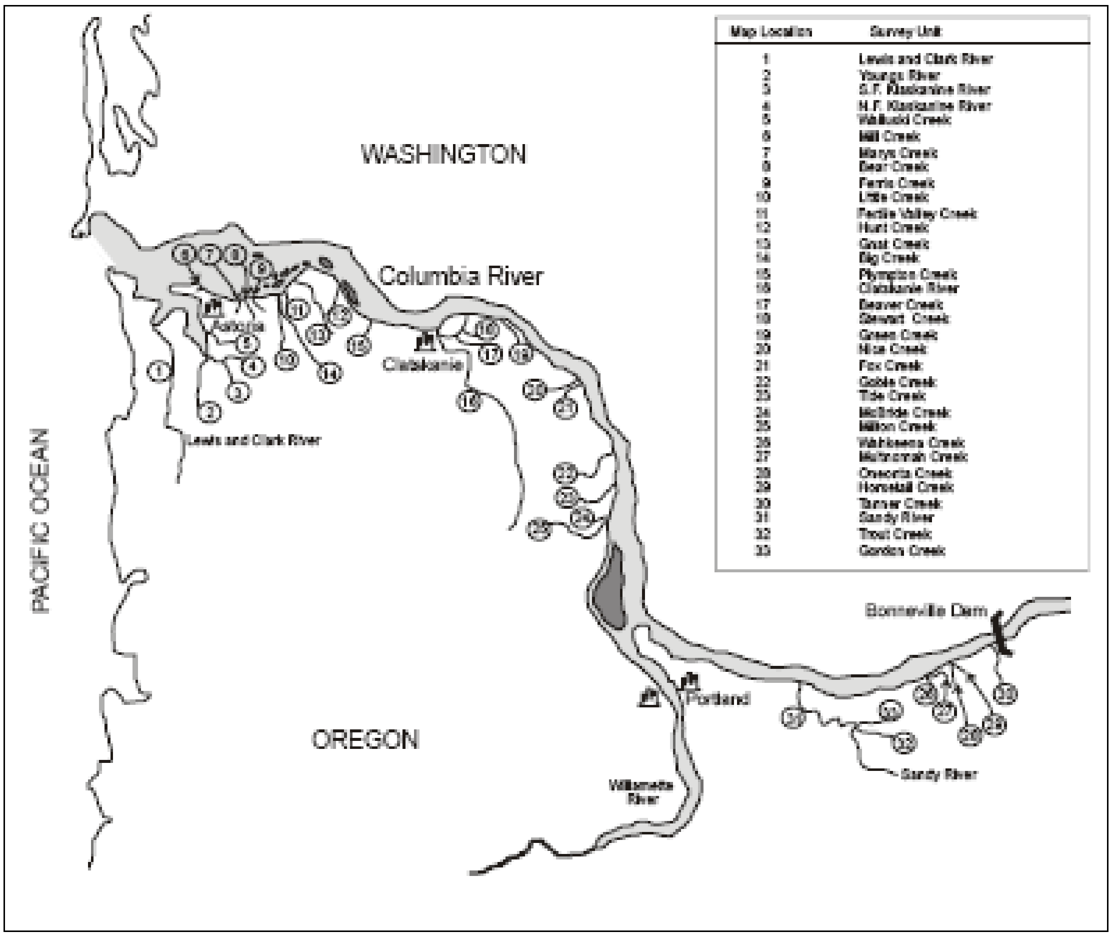

Figure 2: Locations of Oregon Department of Fish and Wildlife 2000 Columbia River stream survey sites (ODFW 2003).

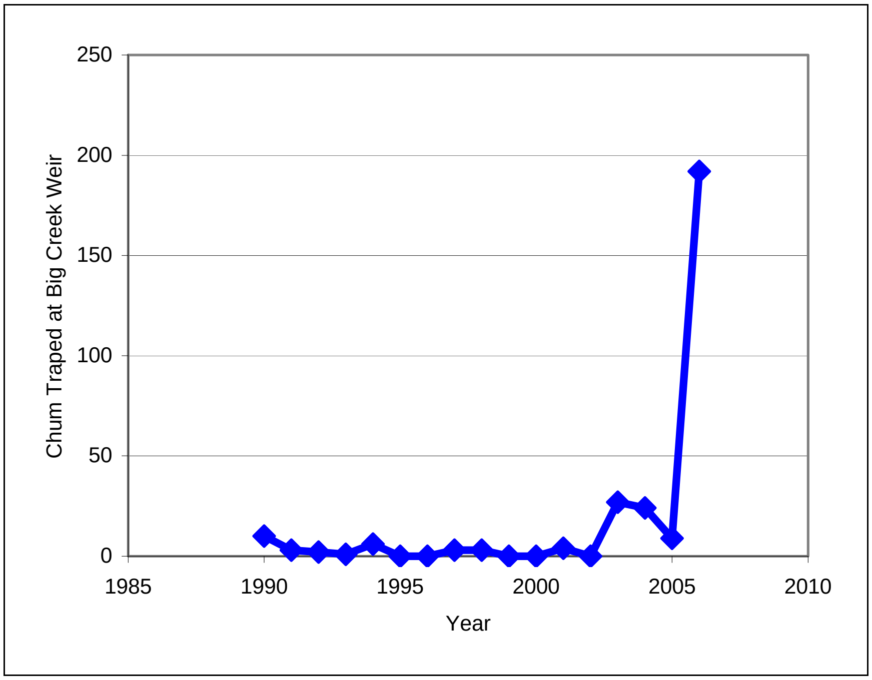

Figure 3: Chum trapped at Big Creek weir (MacIntosh, pers. com. May 15, 2007).

The Lower Columbia Gorge population spans the Columbia, with area in both Oregon and Washington. A survey of chum spawning in the lower gorge population immediately below Bonneville dam has been conducted since 1999 (Figure 4). The majority of the spawning occurs in Washington, but some spawning occurs in Oregon side in the mainstem Columbia near McCord Creek (Figure 5) and Multnomah Falls. These are currently the only documented locations in Oregon with chum redds over multiple years of which we are aware. In 2005, 33 live adult chum were observed in the Multnomah Falls area (Fish Passage Center).

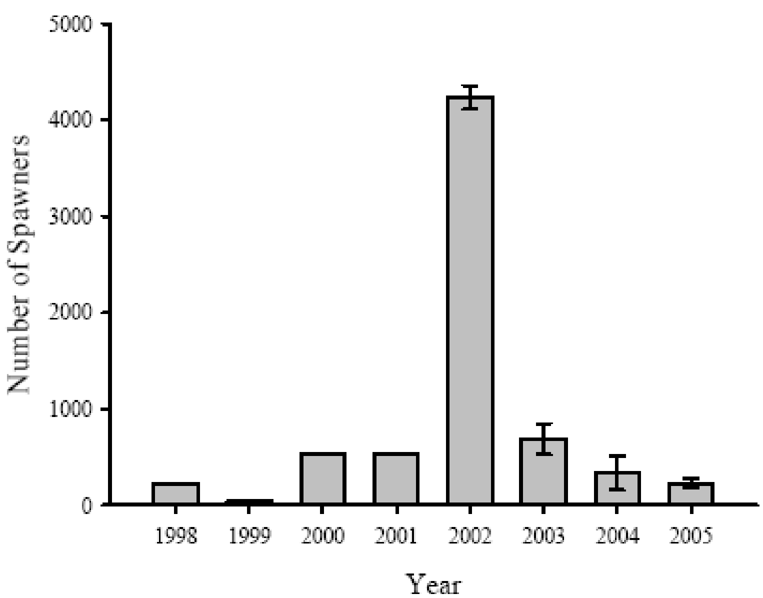

Figure 4: Estimated chum salmon spawner abundance in the Pierce/Ives Island complex below Bonneville Dam (Tomaro et al. 2007).

Figure 5: Chum salmon redd locations below Bonneville Dam in 2005 (Tomaro et al. 2007).

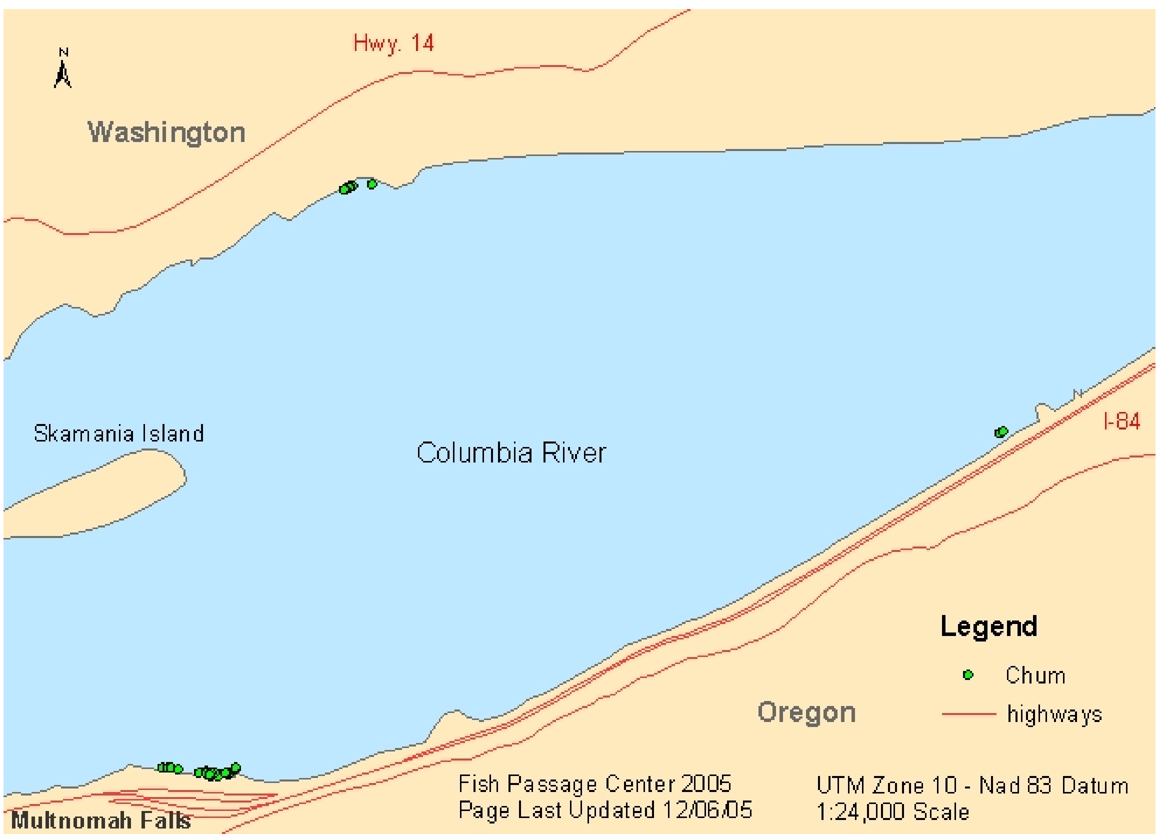

Figure 6: Chum salmon redd locations near Multnomah Falls in 2005 (from Fish Passage Center).

There was historically a chum population between what is now the Bonneville Dam and Celillo Falls (now The Dalles dam). In most years, chum salmon are observed in the ladders at Bonneville Dam (Figure 7). It is not know whether these fish successfully spawn above the dam and if so, what fraction spawn on the Oregon side of the Columbia River. These fish may be strays from the below-Bonneville area that do not successfully spawn above Bonneville. Some fraction may also fall back over Bonneville Dam.

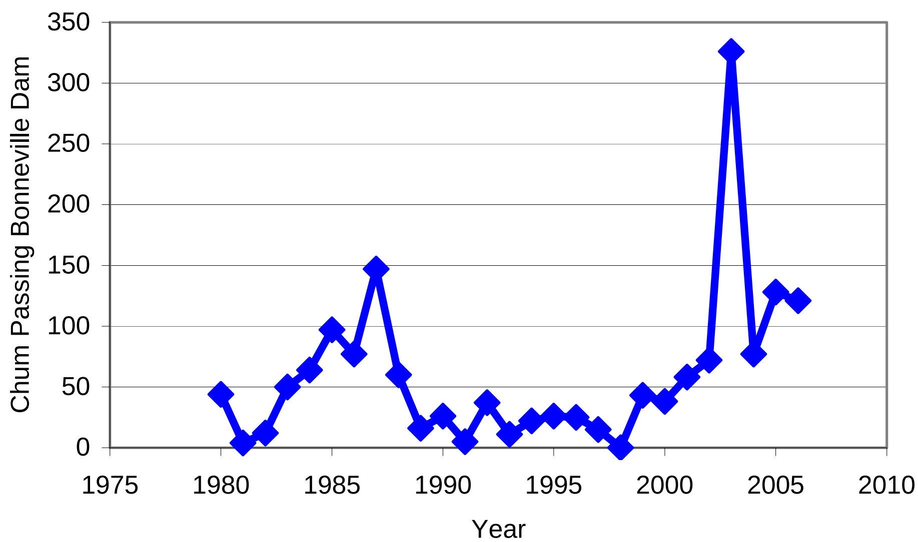

Figure 7: Counts of chum salmon passing Bonneville Dam (Fish Passage Center database http://www.fpc.org/fpc_homepage.html).

III. Spatial Structure

Our knowledge of historical CR chum spatial distribution is incomplete. Chum primarily spawned in the Columbia mainstem and lower tributary reaches and seem to prefer microhabitats with hyporeic flow (Rawding, pers. com.). Maps of current and historical accessibility for chum are available (Maher et al. 2005), but they do not consider microhabitat needs and they do not explore habitat quality. Much of the human population in the region lives in the low elevation, low gradient environment historically used by chum, so we suspect there has been substantial impact on potential spatial structure for chum. Since there are currently few, if any, chum in many of the historical populations, understanding potential spatial structure is important for recovery planning, but is not really necessary for an accurate assessment of population viability.

IV. Diversity

With so few fish, Oregon chum populations have undergone a significant population bottleneck, with likely genetic consequences. Until recently, there have been few hatchery origin chum in the Columbia. In 1999, a hatchery program was initiated in Grays River (McElhany 2005). Fish from this program may stray into Oregon, with potential domestication effects. Give the population bottleneck, maintaining (or establishing) appropriate diversity will likely be a concern when considering how to recovery CR chum populations.

V. Summary

A few chum show up at fish counting facilities and it is likely that some low level, intermittent spawning of chum has gone undetected in Oregon streams. Recent genetic analysis of Washington chum suggests that very small remnant populations may have persisted in the Lower Columbia even when there have been no consistent observations of fish (Small et al. 2006). However, it is clear that all of the Oregon chum populations are in the very high risk category (i.e., extirpated or nearly so). We therefore conclude that the Oregon portion of the CR chum ESU is also at very high risk of extinction.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 4: Lower Columbia Coho

September 2007

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

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I. ESU Overview and Historical Range

The Lower Columbia River (LCR) Coho ESU includes 25 populations that historically existed in the Columbia River basin from the Hood River downstream (Figure 1). The boundaries for this ESU do not extend into upper Willamette portion of the LCR basin, because Willamette Falls (near Portland) was a natural barrier to fall migrating salmonids such as coho salmon.

In general, wild coho in the Columbia basin have been in decline for the last 75 years. The number of wild coho returning to the Columbia River historically was at least 600,000 fish (Chapman, 1986). As recently as 1996, the total return of wild fish may have been as few as 400 fish (Chilcote, 1999). Coinciding with this decline in total abundance has been a reduction in the number of functioning wild populations. All Columbia basin populations upstream of Hood River were extirpated nearly 50 years ago. Of the 25 historical populations that comprised the LCR ESU, only in the Clackamas and Sandy Rivers, is there direct evidence that coho production is not reproductively dependent on the spawning of stray hatchery fish. However, in the last 5 years there has been an increase in the abundance of wild coho in Clackamas and Sandy, plus a re-appearance of moderate numbers wild coho in the Scappoose and Clatskanie basins after a 10-year period in the 1990s when they were largely absent. Additionally, there have been efforts to reestablish coho salmon in the upper Columbia and Snake rivers.

Against this backdrop, we have performed the following status assessment of the eight coho populations that occur within Oregon's portion of the LCR ESU. They include: Youngs Bay, Big Creek, Clatskanie River, Scappoose Creek, Clackamas River, Sandy River, Lower Gorge and Hood River/Upper Gorge. Our assessment consists of three components, each of which evaluates one of the viability criteria (i.e., abundance and productivity, spatial structure, and diversity). This is then followed by a synthesis section where we pool the results from these criteria evaluations into a status rating for each population. Finally, we present an interpretation of the population results in terms of the overall status of Oregon's LCR coho populations and the LCR ESU as a whole.

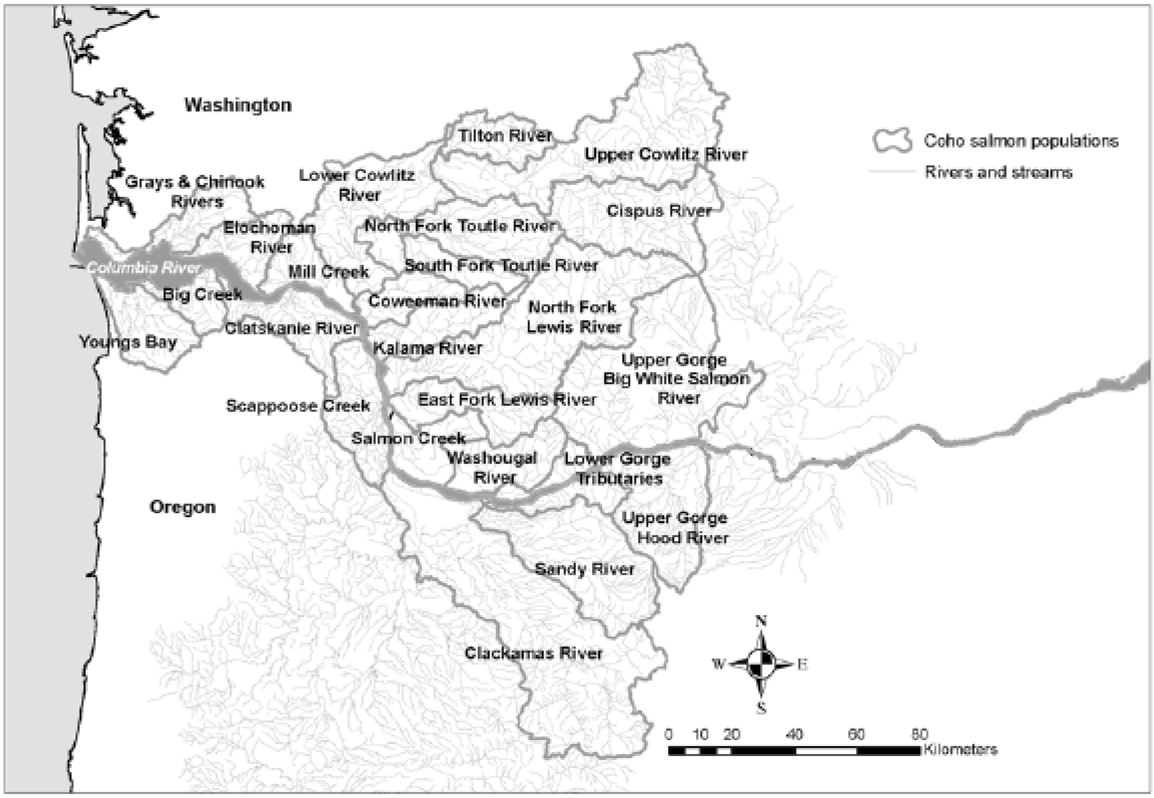


Figure 1: Map of Lower Columbia River coho salmon populations.

II. Abundance and Productivity

A&P – Youngs Bay

ODFW has conducted a peak count of live and dead adult coho at an index site in Youngs Bay since 1949 (Figure 2). The count does not distinguish between hatchery and naturally produced fish and it is not appropriate to conduct a time series analysis with these data. However, the data do indicate that the population has been at low abundance and during the 1990s there were years with no observed coho.

Starting in 2002, a stratified random sample survey has been conducted (Suring et al. 2006), allowing estimation of population size (Figure 3) and hatchery fraction (Figure 4). The random sample estimates abundance for the Astoria population group, which includes both the Youngs Bay and Big Creek populations used in our analysis. The random survey indicates that the number of natural origin spawners is small, with a geometric mean of about 200 fish, which is in the 'extirpated or nearly so' minimum abundance threshold category. The population is dominated by hatchery fish, with on average at least 80% of the coho of hatchery origin. Random survey results show that both the Youngs Bay and Big Creek portions of the Astoria population group have high proportions of hatchery fish. Taken together, these data indicate little, if any natural productivity of coho in the Youngs Bay population and we consider the population most likely in the 'extirpated or nearly so' or 'high risk' category. The Oregon Native Fish Status report (ODFW 2005) listed this population as "fail" for abundance and "fail" for productivity.

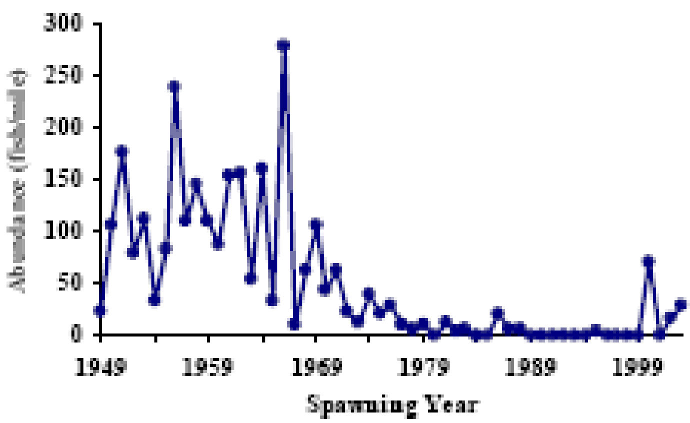


Figure 2: Peak counts of live and dead fish in an index reach in the Youngs Bay coho salmon population (reproduced from (ODFW 2005).

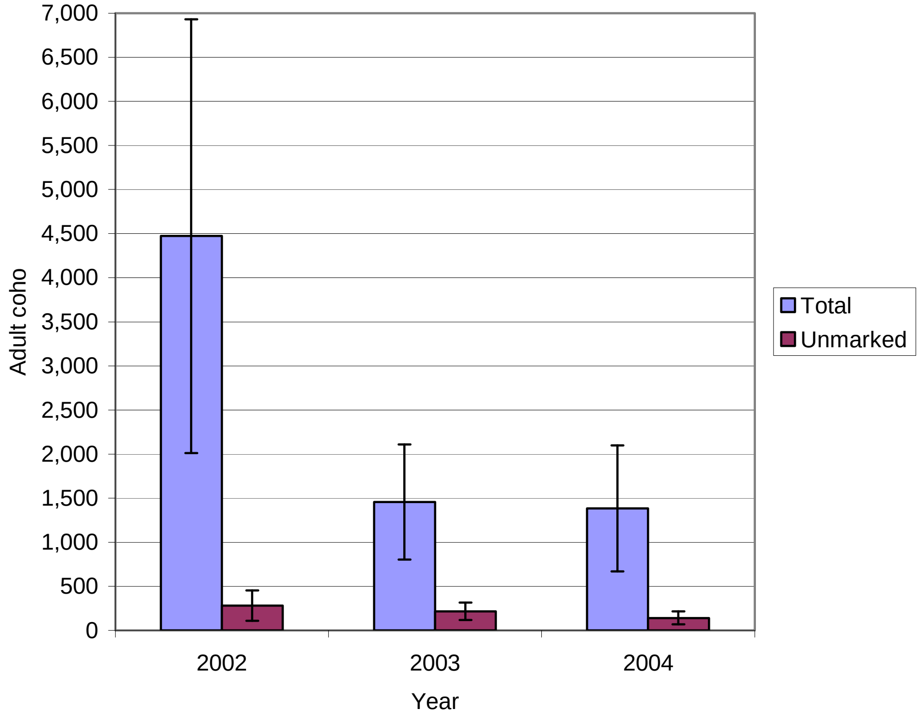


Figure 3: Abundance estimates of adult coho salmon in Astoria population group (Suring et al. 2006). The 'Total' bars show the estimated total adult coho salmon abundance. The 'Unmarked' bars indicate potential natural origin fish (some unmarked fish are likely of hatchery origin). The error bars are 95% confidence intervals.

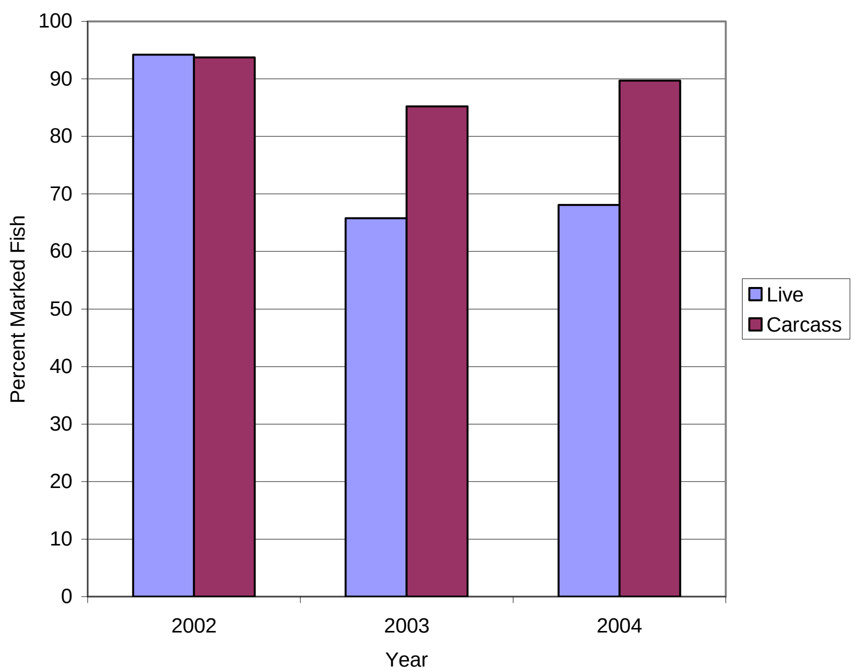


Figure 4: Percent of hatchery marked fish in the Astoria population group (Youngs Bay and Big Creek populations in this document) based on observations of either live fish or carcasses (Suring et al. 2006). Values are adjusted for mark rates of local hatchery releases.

A&P – Big Creek

ODFW has conducted a peak count of live and dead adult coho at an index site in Big Creek since 1950 (Figure 5). The count does not distinguish between hatchery and naturally produced fish and it is not appropriate to conduct a time series analysis with these data. However, the data do indicate that the population has been at low abundance and in many years there were no observed coho.

Starting in 2002, a stratified random sample survey has been conducted (Suring et al. 2006), allowing estimation of population size (Figure 3) and hatchery fraction (Figure 4). The random sample estimates abundance for the Astoria population group, which includes both the Youngs Bay and Big Creek populations used in our analysis. The random survey indicates that the number of natural origin spawners for Youngs Bay and Big creek combined is small, with a geometric mean of about 200 fish, which is in the 'extirpated or nearly so' minimum abundance threshold category. The population is dominated by hatchery fish, with on average at least 80% of the coho of hatchery origin. Random survey results show that both the Youngs Bay and Big Creek portions of the Astoria population group have high proportions of hatchery fish. Taken together, these data indicate little, if any natural productivity of coho in the Big Creek population and we consider the population most likely in the 'extirpated or nearly so' or 'high risk' category. The Oregon Native Fish Status report (ODFW 2005) listed this population as "fail" for abundance and "fail" for productivity.

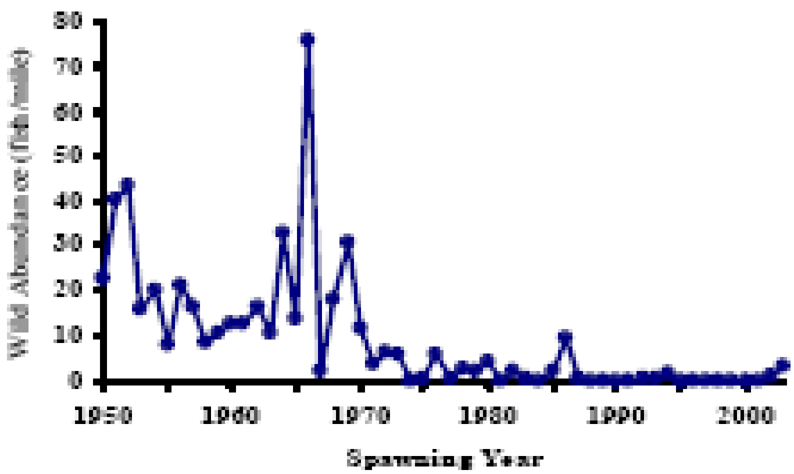


Figure 5: Peak counts of live and dead fish in an index reach in the Big Creek coho salmon population (reproduced from ODFW 2005).

A&P – Clatskanie River

ODFW has conducted a peak count of live and dead adult coho at an index site in the Clatskanie since 1949 (Figure 5). The count does not distinguish between hatchery and naturally produced fish and it is not appropriate to conduct a time series analysis with these data. However, the data do indicate that the population has been at low abundance and in many years there were no observed adult coho (although juveniles were observed in subsequent years – indicating that a small number of adults were present). Starting in 2002, a stratified random sample survey has been conducted (Suring et al. 2006), allowing estimation of population size (Figure 7) and hatchery fraction (Figure 8). The random survey indicates that the number of natural origin spawners for the Clatskanie population is small, with a three year geometric mean of 286 fish, which is in the 'extirpated or nearly so' minimum abundance threshold category. The hatchery fraction data are highly variable, ranging from 80% hatchery fish to 0% hatchery fish, depending on the year. The temporal variability is likely a reflection of the spatial hatchery fraction pattern combined with the particulars of the sampling protocol (Suring et al. 2006). The streams in the western portion of the population area are dominated by hatchery fish, whereas the Clatskanie River itself, in the eastern portion of the population area, appears to be free of hatchery fish. Because there are some returning adults and there do not appear to be many hatchery fish in most of the population area, there is likely some natural production in the Clatskanie. However, the population is currently small and likely dropped to double or single digits in the recent past. Therefore, we consider the population as most likely in the 'high risk' category' but with substantial possibility it is in the 'extirpated or nearly so' category. The Oregon Native Fish Status report (ODFW 2005) listed this population as "fail" for abundance and "fail" for productivity.

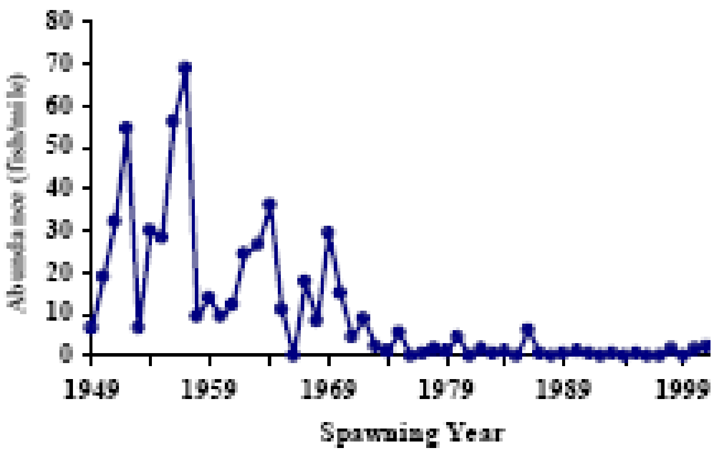


Figure 6: Peak counts of live and dead fish in an index reach in the Clatskanie River coho salmon population (reproduced from ODFW 2005).

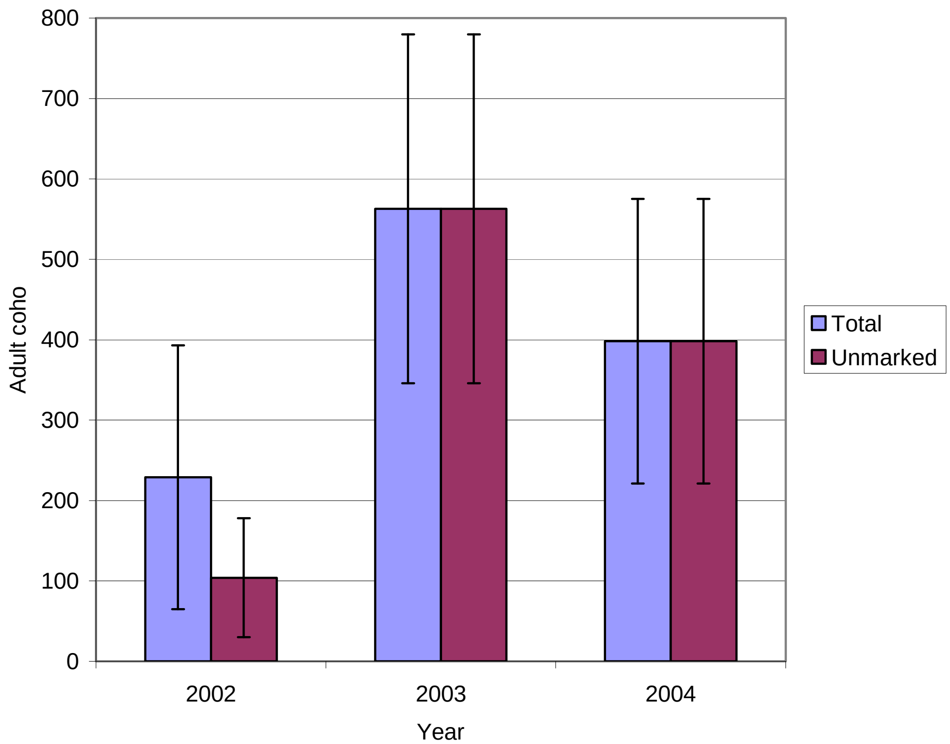


Figure 7: Abundance estimates of adult coho salmon in Clatskanie population (Suring et al. 2006). The 'Total' bars show the estimated total adult coho salmon abundance. The 'Unmarked' bars indicate potential natural origin fish (some unmarked fish are likely of hatchery origin). The error bars are 95% confidence intervals.

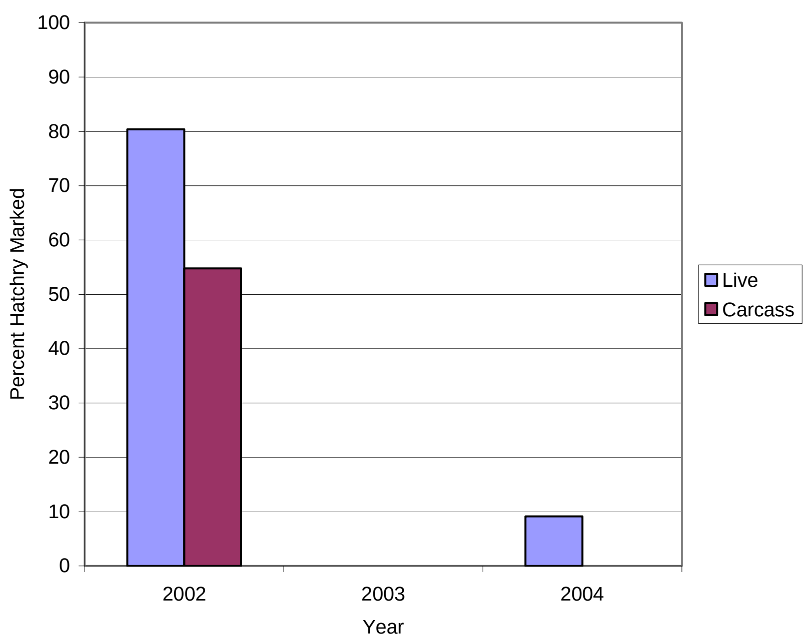


Figure 8: Percent of hatchery marked fish in the Clatskanie population group based on observations of either live fish or carcasses (Suring et al. 2006). Values are adjusted for mark rates of local hatchery releases.

A&P – Scappoose Creek

ODFW has conducted a peak count of live and dead adult coho at an index site in the Scappoose since 1950 (Figure 9). The count does not distinguish between hatchery and naturally produced fish and it is not appropriate to conduct a time series analysis with these data. However, the data do indicate that the population has been at low abundance and in many years there were no observed adult coho. Starting in 2002, a stratified random sample survey has been conducted (Suring et al. 2006), allowing estimation of population size (Figure 10) and hatchery fraction (Figure 11). The random survey indicates that the number of natural origin spawners for the Scappoose population is relatively small, with a three year geometric mean of 470 fish, which is in the 'extirpated or nearly so' minimum abundance threshold category, but approaching the 'high risk' category. The hatchery fraction data indicate that there are currently few hatchery fish in this population. Because there are several hundred returning adults and there do not appear to be many hatchery fish in the population, there is likely some natural production of coho in the Scappoose. However, the population is currently small and likely dropped to double or single digits in the recent past. Therefore, we consider the population as most likely in the 'high risk' category but with a possibility it is in the 'extirpated or nearly so' category. The Oregon Native Fish Status report (ODFW 2005) listed this population as "fail" for abundance and "fail" for productivity.

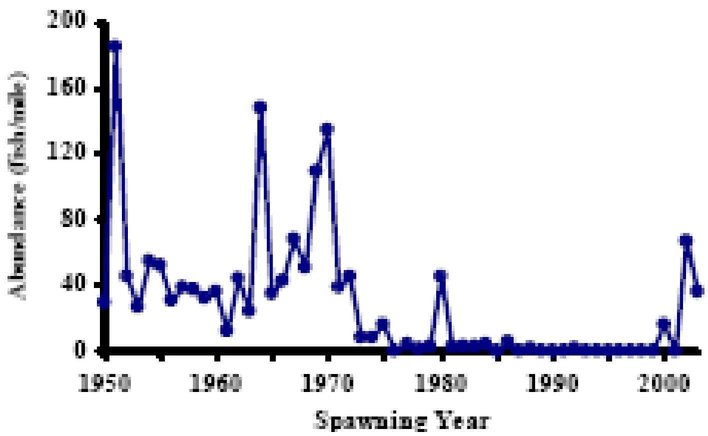


Figure 9: Peak counts of live and dead fish in an index reach in the Scappoose coho salmon population (reproduced from ODFW 2005).

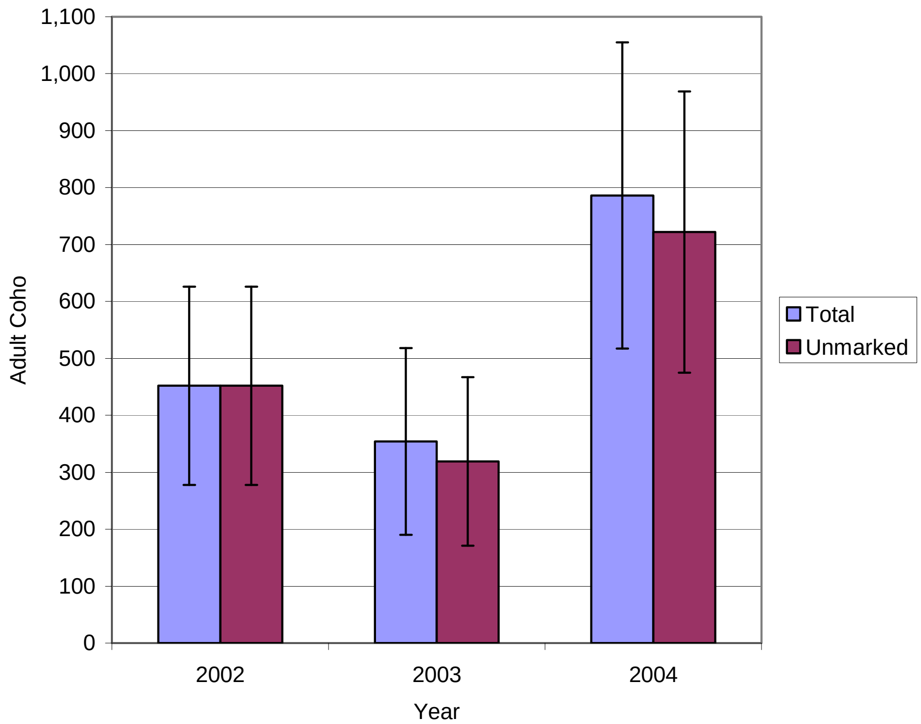


Figure 10: Abundance estimates of adult coho salmon in Scappoose population (Suring et al. 2006). The 'Total' bars show the estimated total adult coho salmon abundance. The 'Unmarked' bars indicate potential natural origin fish (some unmarked fish are likely of hatchery origin). The error bars are 95% confidence intervals.

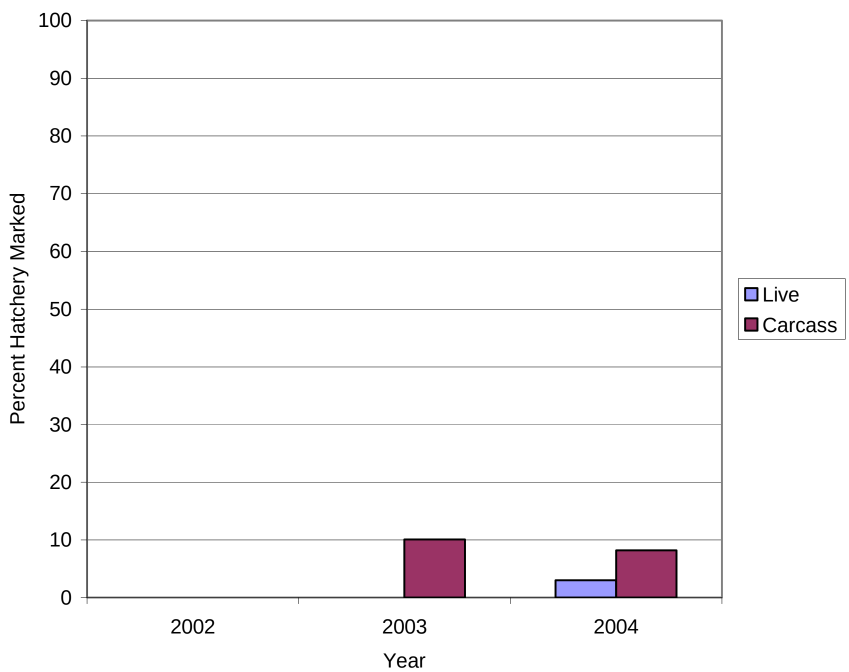


Figure 11: Percent of hatchery marked fish in the Scappoose population group based on observations of either live fish or carcasses (Suring et al. 2006). Values are adjusted for mark rates of local hatchery releases.

A&P – Clackamas River

A time series of abundance sufficient for quantitative analysis is available for the Clackamas population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 12 to Figure 20 and in Table 1 to Table 4. The population long-term geometric mean is about 1,700 natural origin spawners, which is in the high risk minimum abundance threshold category (Table 1). (Note: Coho have the highest minimum abundance thresholds because of high variability and a discrete age structure that does not provide temporal buffering of risk.) Because coho have discrete three year generations, it is useful to look at the abundance patterns for individual cohorts (Figure 13). The data show that cohort A (ending in 2005) is likely at greater risk than the other two cohorts because it has a lower average abundance. The average recent hatchery fraction is estimated at about 25%, making it difficult to obtain a precise estimate of population productivity. The pre-harvest viability curve analysis, the CAPM modeling and the PopCycle model all suggest that the population is currently viable, and perhaps in the very low risk category. The escapement viability curve suggests that the population continued to experience a pattern of harvest similar to the available time series (average impact rate of 73%) would most likely be in the 'extirpated or nearly so' risk category. However, this analysis included years when the fishing mortality was in excess of 80% and therefore incorporates a larger reduction in life history survival than the 25% fishery impact rates that are expected in the future. The Oregon Native Fish Status report (ODFW 2005), which divided the Clackamas River coho into 'early' and 'late' populations, classified both as "passing " interim criteria for abundance and productivity. Based on our evaluation, we conclude that this population is most likely in the low risk category, for the abundance and productivity criterion.

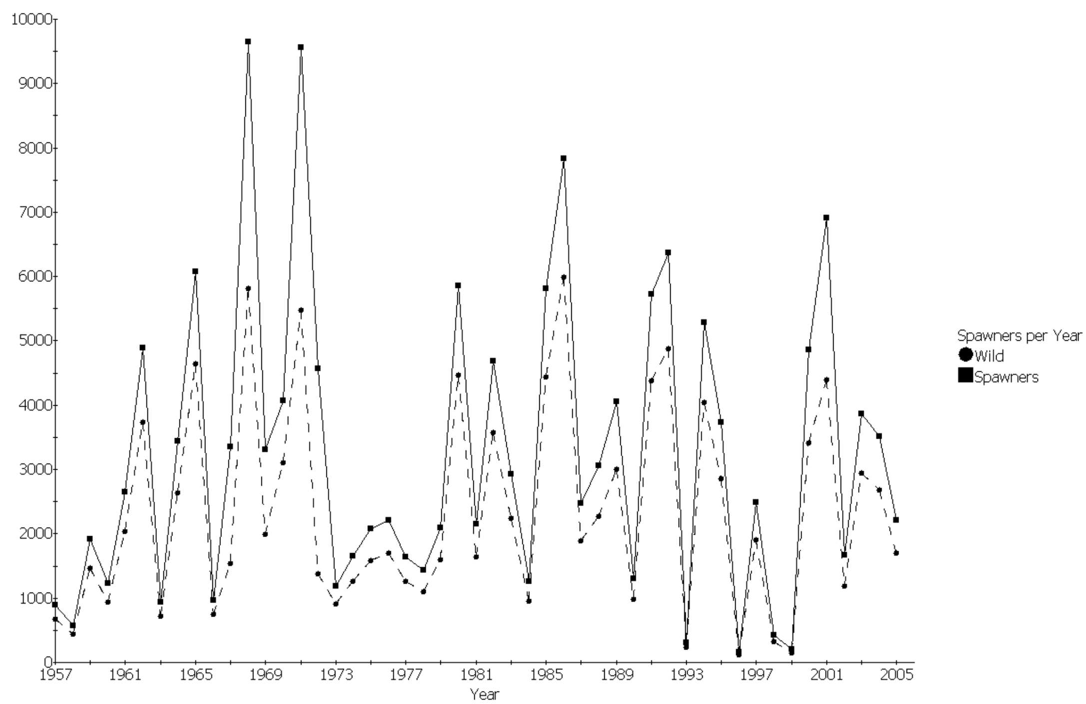
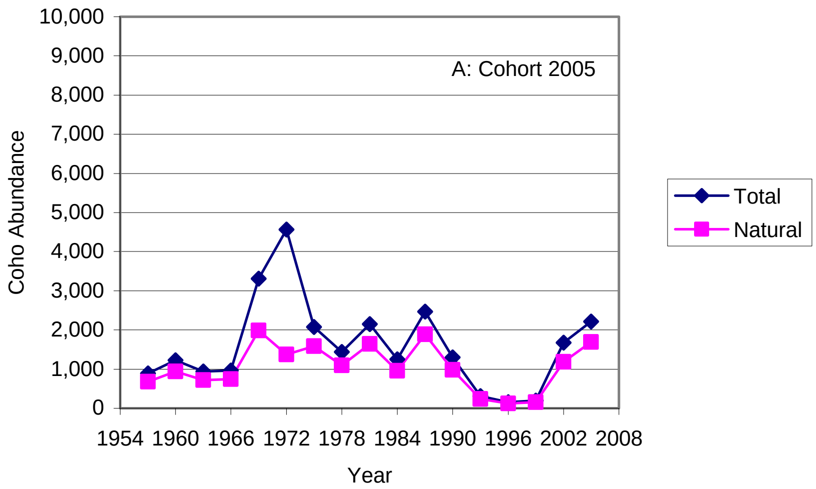
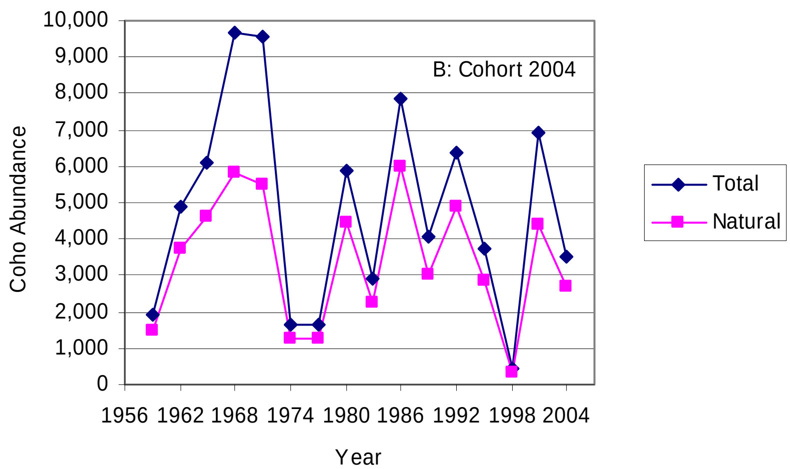


Figure 12: Clackamas River coho salmon abundance.





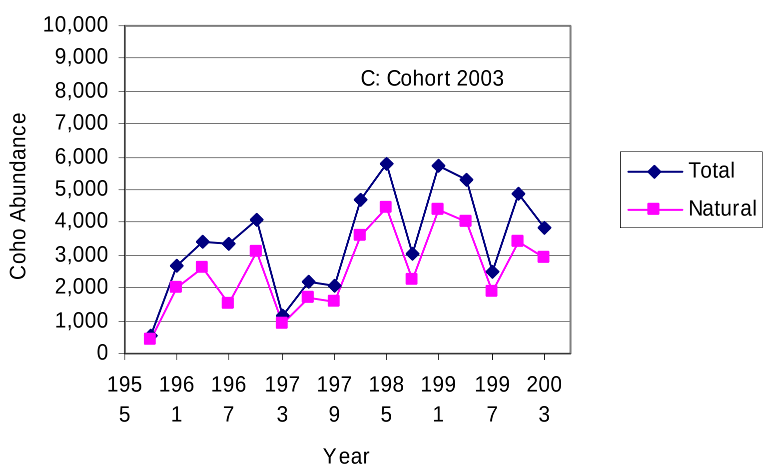


Figure 13: Clackamas River coho salmon abundance by cohort. The geometric mean natural origin abundance for cohort A is 828; for cohort B it is 2,211; and for cohort C it is 2,772.

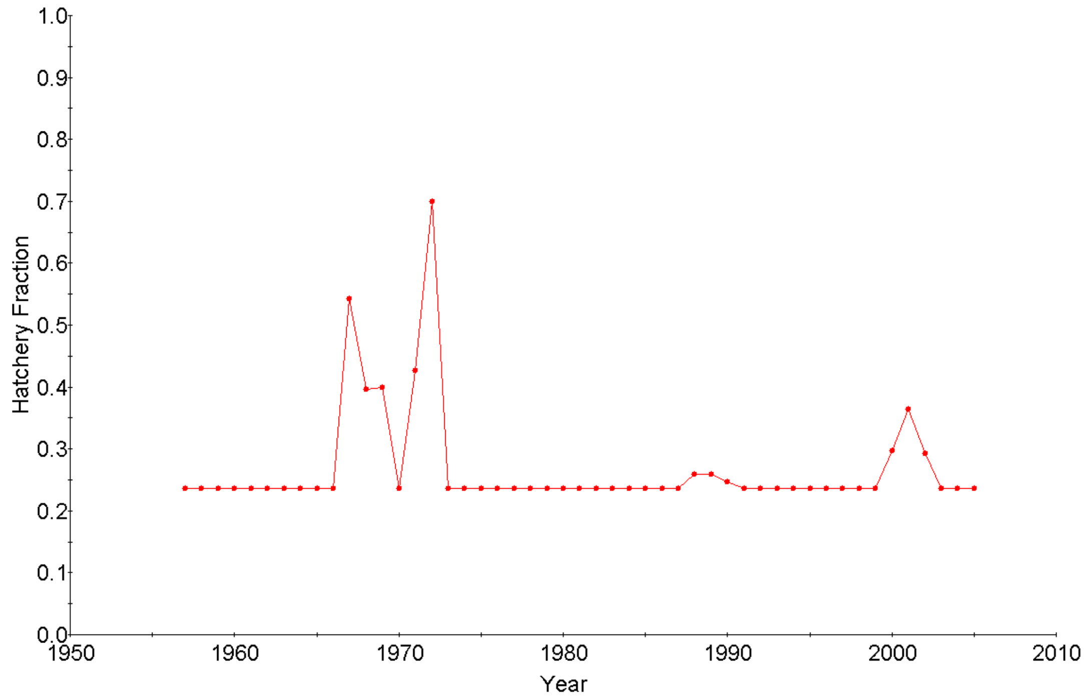


Figure 14: Clackamas River coho salmon hatchery fraction.

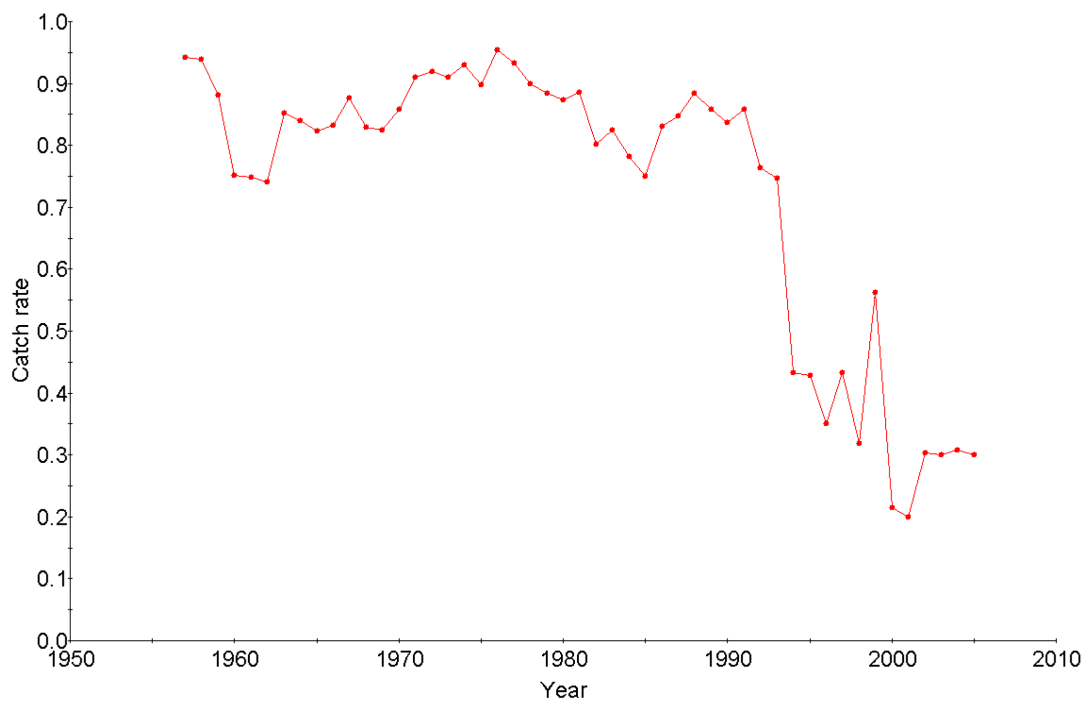


Figure 15: Clackamas River coho harvest rate.

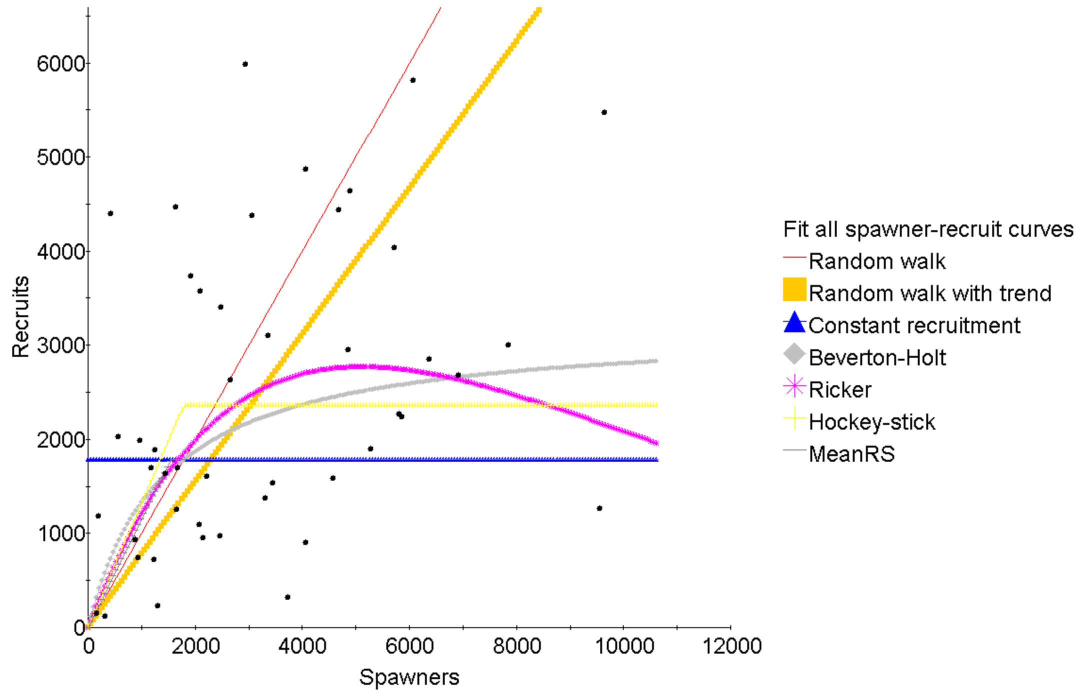


Figure 16: Clackamas River coho salmon escapement recruitment functions.

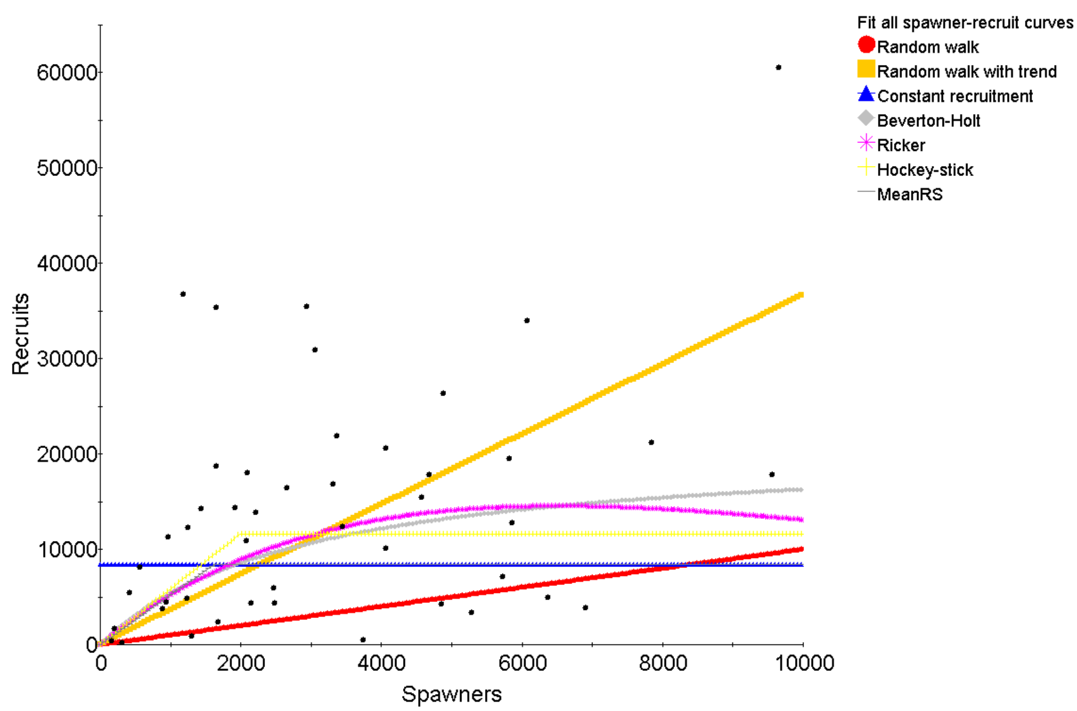


Figure 17: Clackamas River coho salmon pre-harvest recruitment functions.

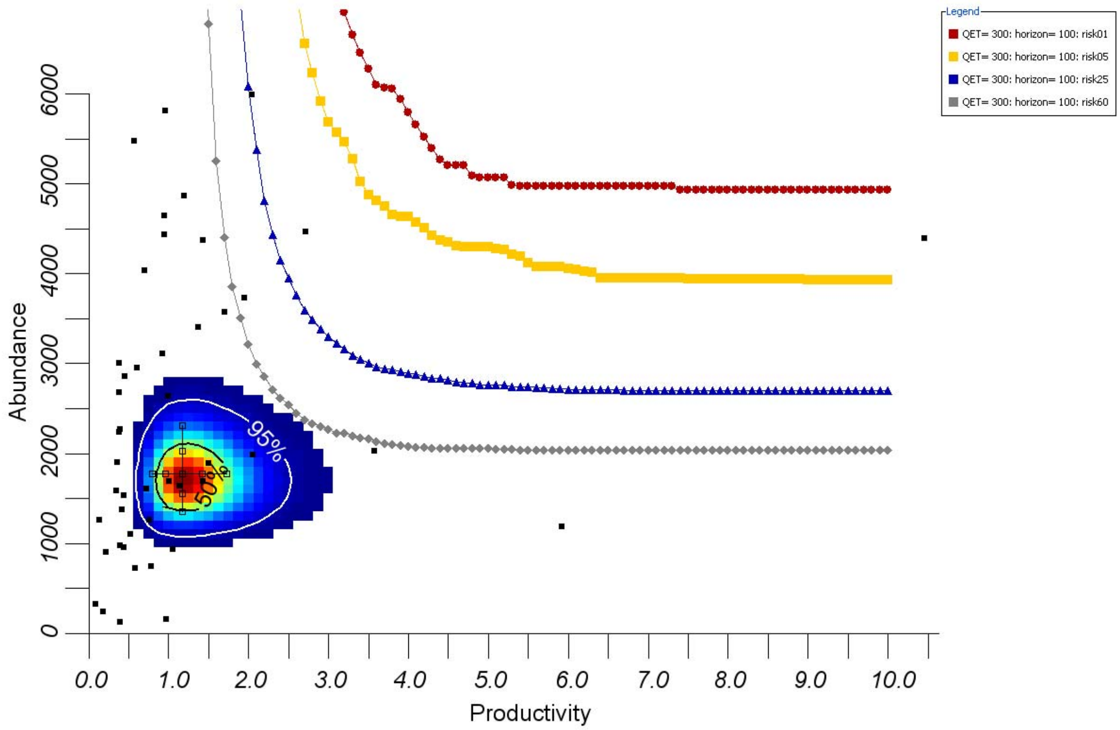


Figure 18: Clackamas River coho salmon escapement viability curve.

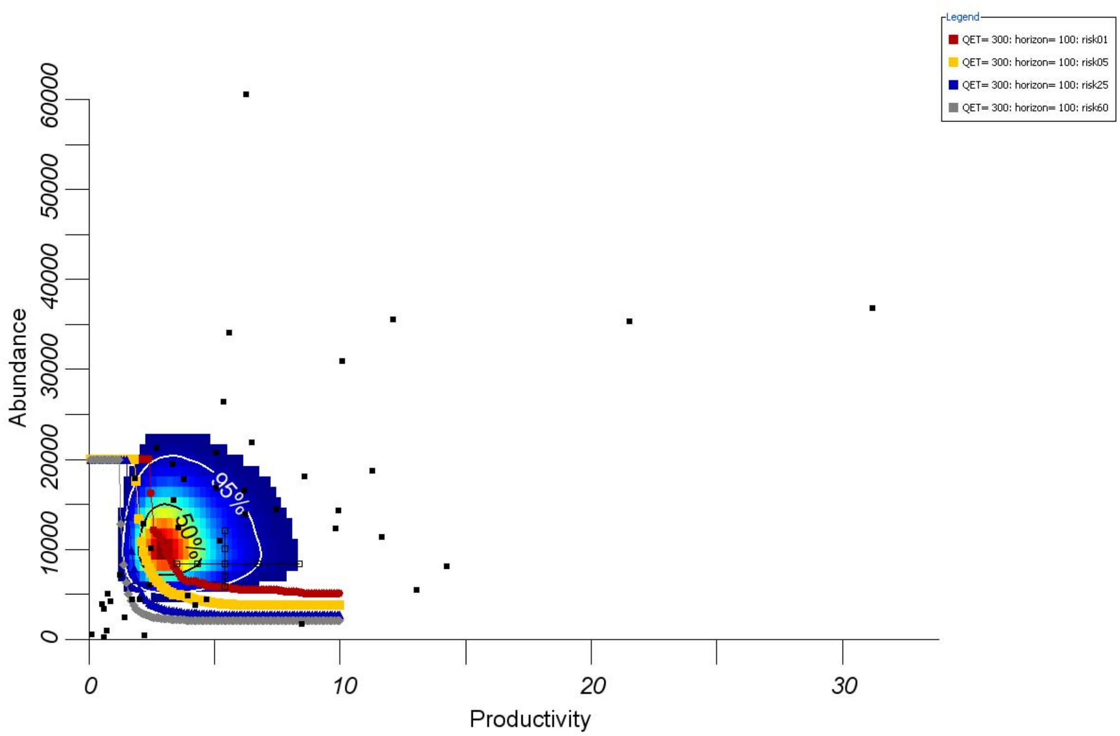


Figure 19: Clackamas River coho salmon pre-harvest viability curve showing all data points.

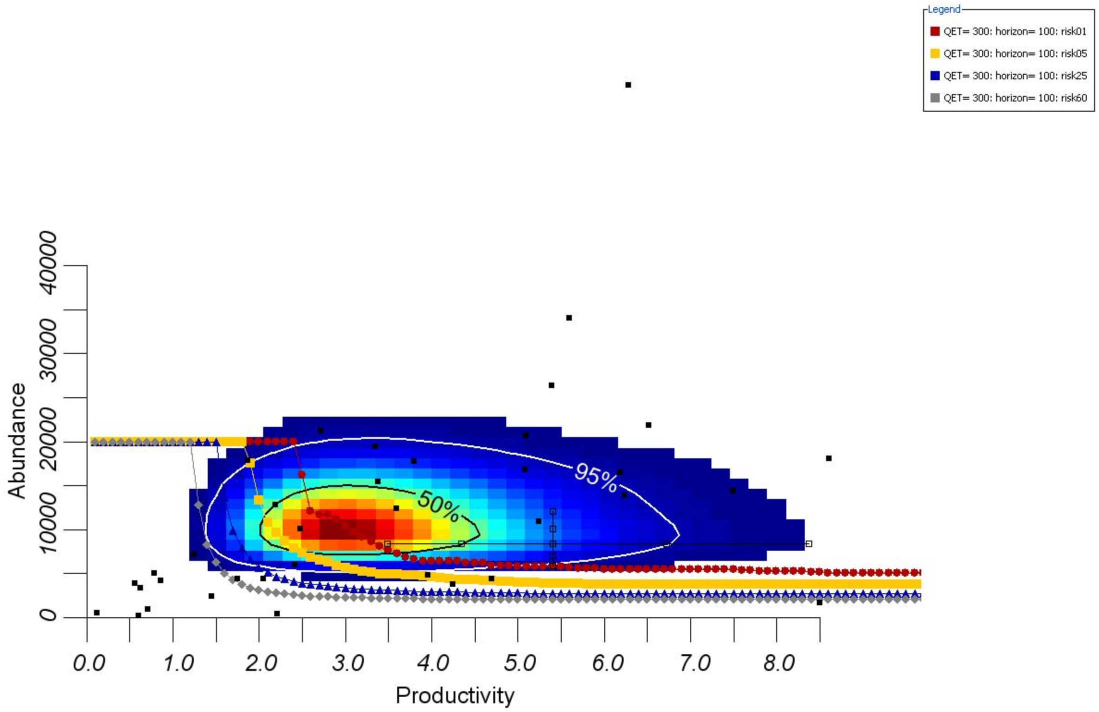


Figure 20: Clackamas River coho pre-harvest viability curve cropped to show details (graph does not include all original data points).

Table 1: Clackamas River coho summary statistics. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	**Recent Years**	**Total Series**	**Recent Years**
Time Series Period	1957-2005	1990-2005	1957-2005	1990-2005
Length of Time Series	49	16	49	16
Geometric Mean Natural Origin Spawner Abundance	1693 (1302-2202)	1,368 (696-2,688)	NA	NA
Geometric Mean Recruit Abundance	1785 (1362-2339)	1164 (527-2574)	8448 (5830-12244)	1937 (949-3955)
Lambda	0.913 (0.821-1.014)	0.886 (0.524-1.499)	1.513 (1.231-1.859)	0.988 (0.614-1.589)
Trend in Log Abundance	1.0 (0.981-1.018)	1.017 (0.874-1.183)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.778 (0.592-1.021)	0.718 (0.378-1.572)	3.681 (2.652-5.108)	1.195 (0.58-2.463)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.149 (0.77-1.713)	1.289 (0.549-3.043)	5.186 (3.315-8.112)	2.223 (0.756-6.54)
Average Hatchery Fraction	0.269	0.252	NA	NA
Average Harvest Rate	0.728	0.460	NA	NA
CAPM median extinction risk probability (5^{th}-95^{th} percentiles)	NA	NA	0.000 (0.000-0.115)	NA
PopCycle extinction risk	NA	NA	0.03	NA

Table 2: Escapement recruitment parameter estimates and relative AIC values for Clackamas River coho. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.94 (0.81-1.15)	15.6
Random walk with trend	0.78 (0.63-1)	NA	0.91 (0.79-1.13)	14.2
Constant recruitment	NA	1783 (1452-2286)	0.9 (0.78-1.12)	13.4
Beverton-Holt	2.26 (1.27-6.84)	3210 (2139-6222)	0.76 (0.67-0.96)	0
Ricker	1.47 (0.98-2.06)	2771 (2339-5249)	0.78 (0.69-0.99)	2
Hockey-stick	1.32 (1.01-5.08)	2364 (1703-3124)	0.79 (0.7-1)	3.3
MeanRS	1.15 (0.85-1.57)	1785 (1428-2211)	0.64 (0.4-0.88)	14.5

Table 3: Pre-harvest recruitment parameter estimates and relative AIC values for Clackamas River coho. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (pre-harvest relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.7 (1.47-2.08)	44
Random walk with trend	3.68 (2.86-4.98)	NA	1.09 (0.95-1.36)	5.2
Constant recruitment	NA	8457 (6387-12028)	1.24 (1.08-1.54)	16.6
Beverton-Holt	7.23 (5.51-16.86)	21530 (11889-24206)	1.02 (0.9-1.28)	1.2
Ricker	6.11 (4.14-9.65)	14383 (11330-23408)	1.03 (0.9-1.29)	1.5
Hockey-stick	5.88 (4.05-11.25)	11650 (8833-18311)	1.01 (0.89-1.28)	0
MeanRS	5.19 (3.67-7.29)	8448 (6175-11298)	1.05 (0.62-1.48)	3.1

Table 4: Clackamas River coho CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in 'extirpated or nearly so' category	0.001	0.999	1.000
Probability the population is above 'Moderate risk of extinction' category	0.000	0.987	0.995
Probability the population is above 'Viable' category	0.000	0.922	0.863
Probability the population is above 'Very low risk of extinction' category	0.000	0.692	0.637

A&P – Sandy River

A time series of abundance sufficient for quantitative analysis is available for the Sandy population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 21 to Figure 29 and in Table 5 to Table 8. The population long-term geometric mean is about 650 natural origin spawners, which is in the ‘extirpated or nearly so’ minimum abundance threshold category (Table 5). (Note: Coho have the highest minimum abundance thresholds because of high variability and a discrete age structure that does not provide temporal buffering of risk.) Because coho have discrete three year generations, it is useful to look at the abundance patterns for individual cohorts (Figure 22). The data show that cohort A (ending in 2005) is likely at greater risk than the other two cohorts because it has a lower average abundance. The pre-harvest viability curve analysis suggests that the population is most likely in the high risk category. The CAPM and PopCycle modeling both suggest that the population is most likely in the moderate risk category. The escapement viability curve suggests that if the population continued to experience the pattern of harvest that occurred over the available time series (average harvest rates = 71%), it would be in the ‘extirpated or nearly so’ risk category. The Oregon Native Fish Status report (ODFW 2005) listed the Sandy coho population as a “pass” for abundance and a “pass” for productivity.

Taken together, the data suggest the population is most likely in the high risk category for the abundance and productivity criterion.

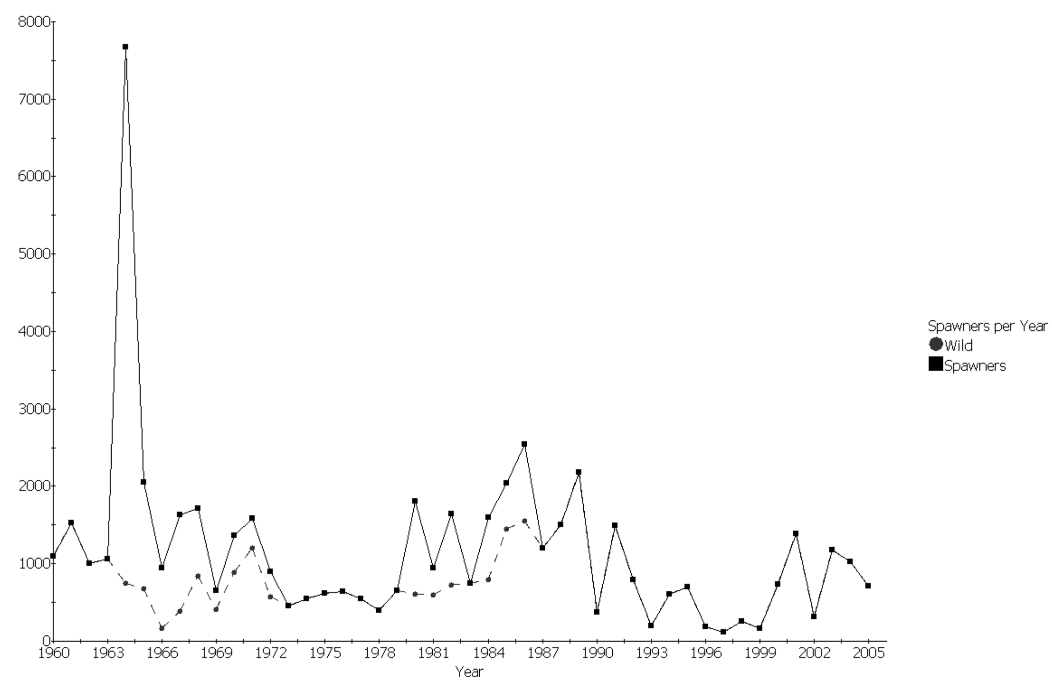
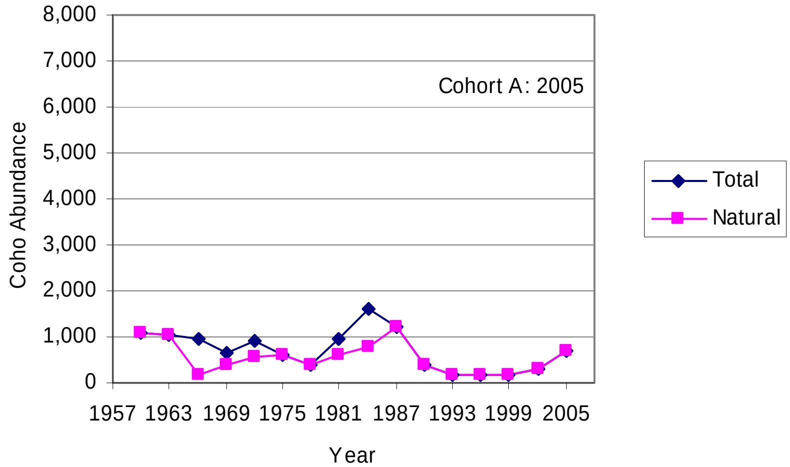
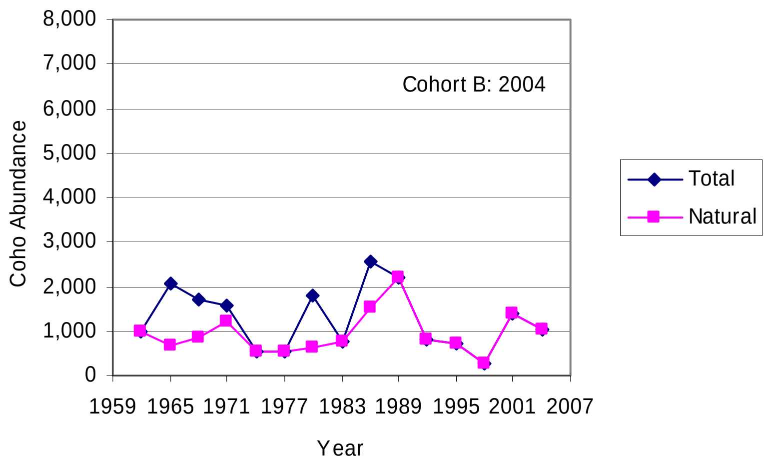


Figure 21: Sandy River coho salmon abundance at Marmot Dam.





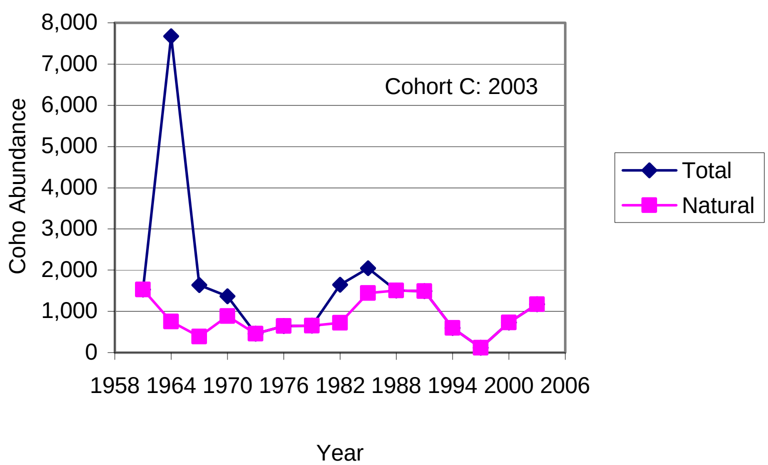


Figure 22: Sandy River coho abundance by cohort. The geometric mean natural origin abundance for cohort A is 451; for cohort B it is 738; and for cohort C it is 833.

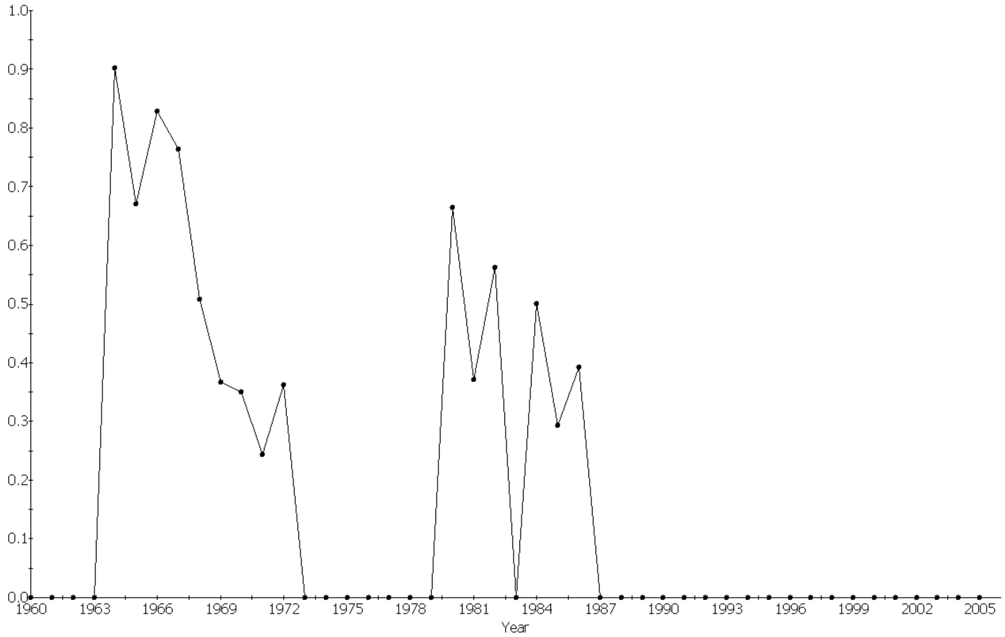


Figure 23: Sandy River coho salmon hatchery fraction.

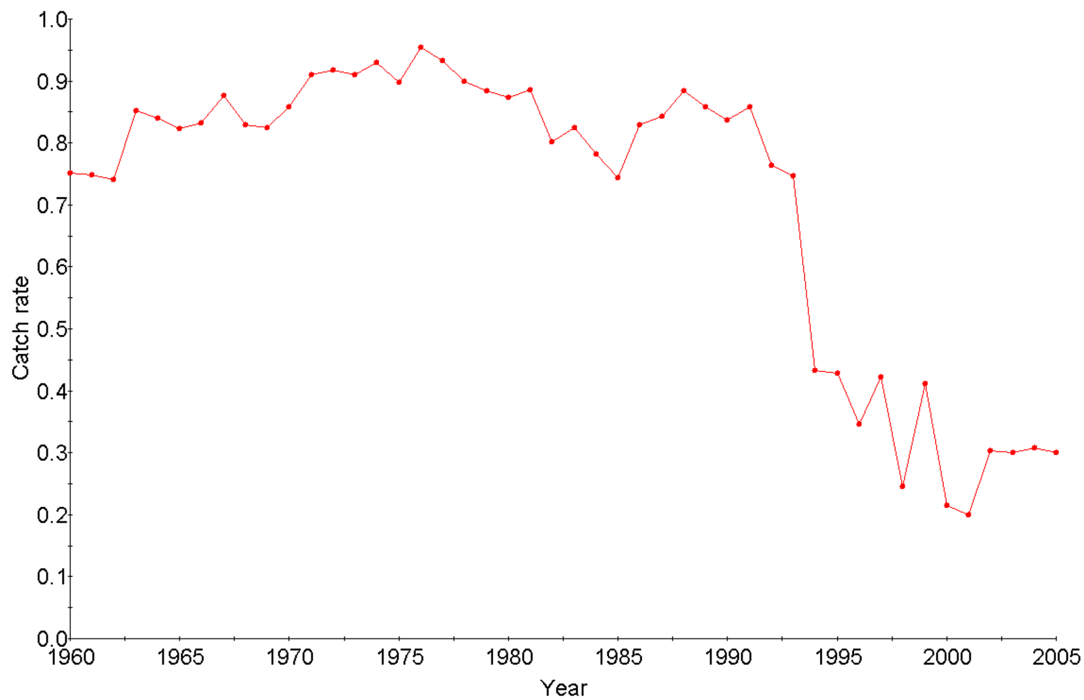


Figure 24: Sandy River coho salmon harvest rate.

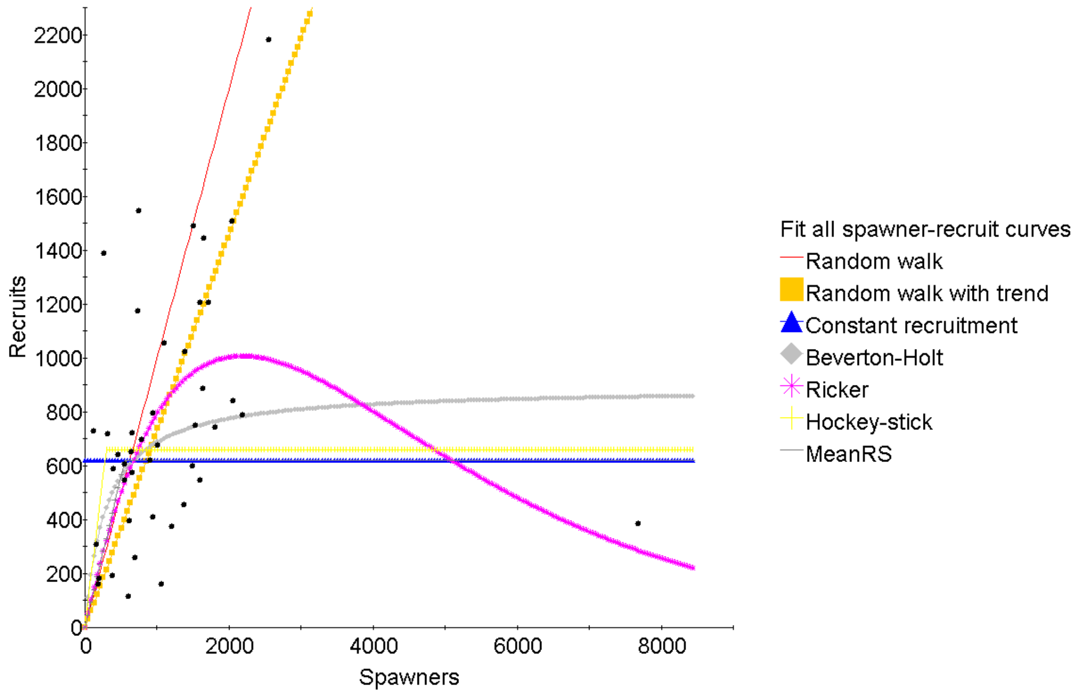


Figure 25: Sandy River coho salmon pre-harvest recruitment functions.

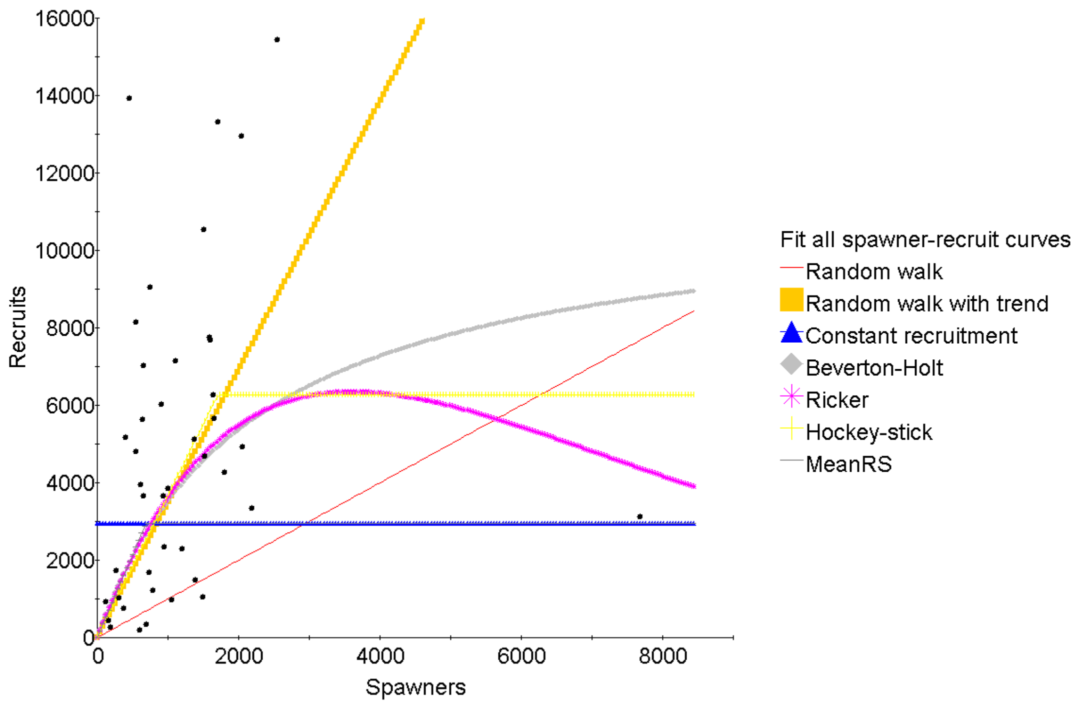


Figure 26: Sandy River coho salmon pre-harvest recruitment functions.

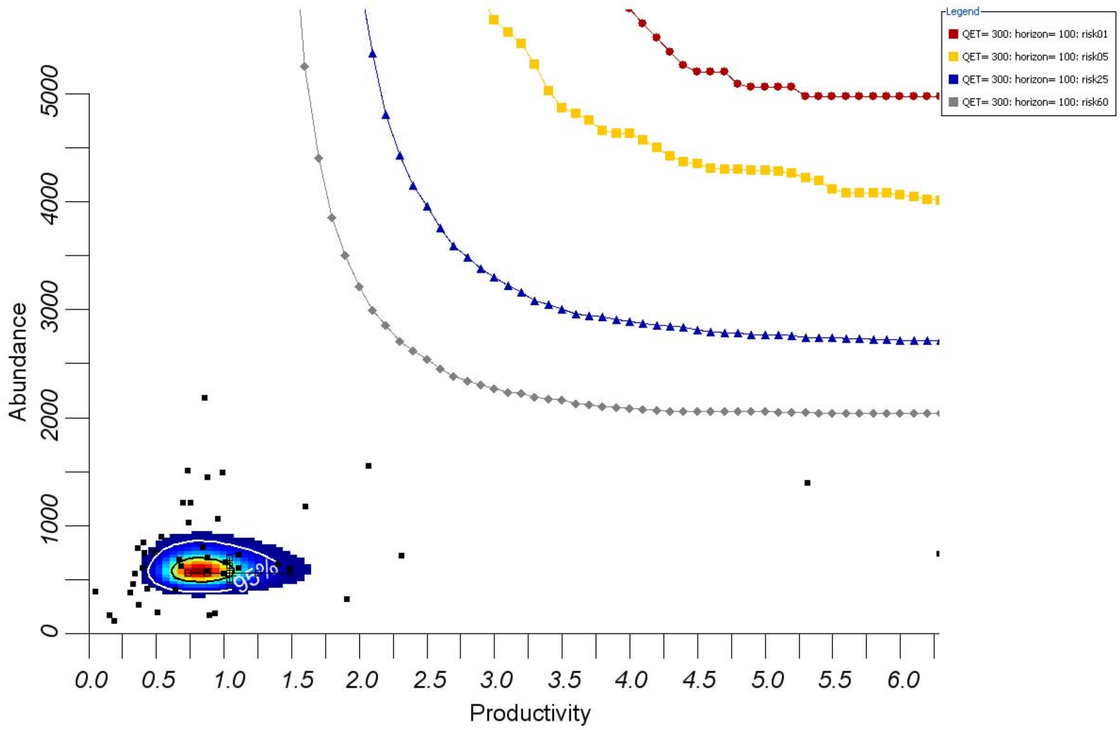


Figure 27: Sandy River coho salmon escapement viability curve.

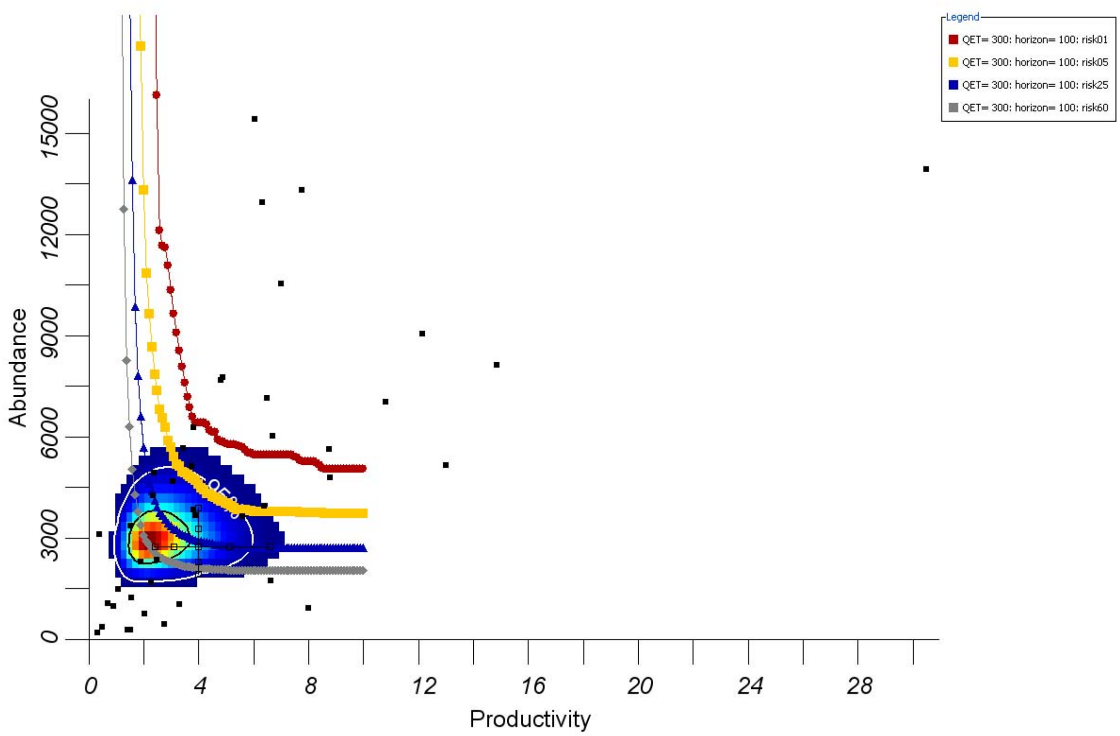


Figure 28: Sandy River coho salmon pre-harvest viability curve showing all data points.

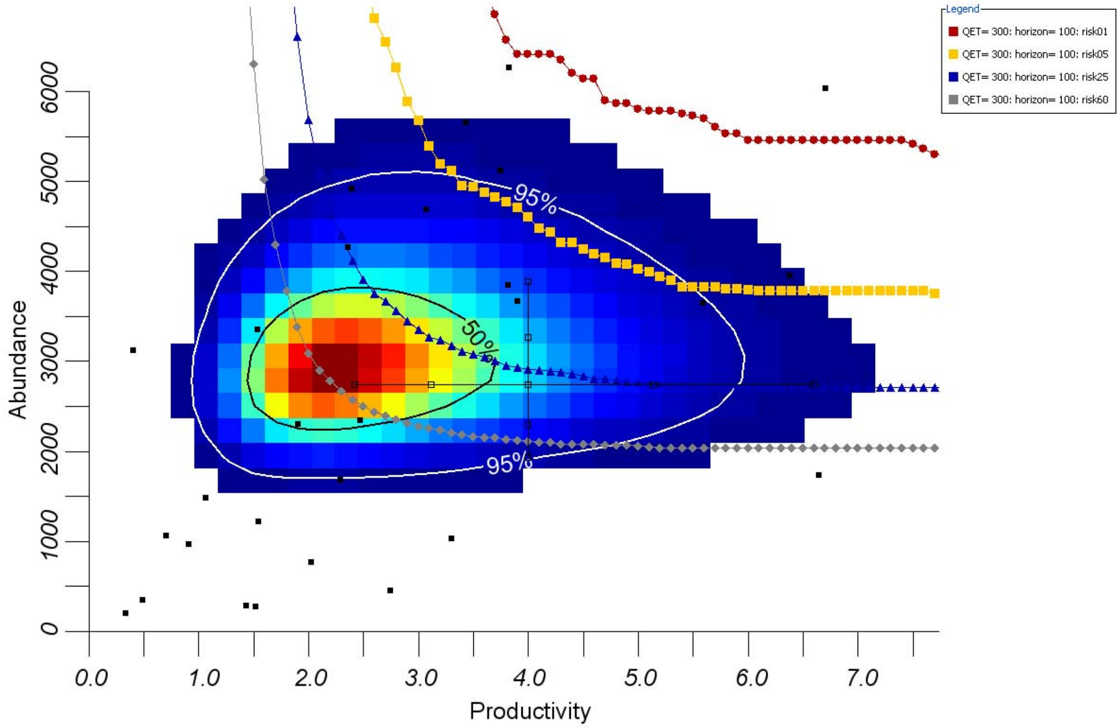


Figure 29: Sandy River coho pre-harvest viability curve cropped to show detail. (Not all the original data are shown.)

Table 5: Sandy Coho summary statistics. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	**Recent Years**	**Total Series**	**Recent Years**
Time Series Period	1960-2005	1990-2005	1960-2005	1990 – 2005
Length of Time Series	46	16	46	16
Geometric Mean Natural Origin Spawner Abundance	647 (529-790)	482 (311-748)	647 (529-790)	482 (311-748)
Geometric Mean Recruit Abundance	620 (504-763)	434 (262-721)	2939 (2062-4189)	699 (443-1104)
Lambda	0.884 (0.753-1.038)	1.01 (0.547-1.865)	1.487 (1.176-1.88)	1.122 (0.607-2.072)
Trend in Log Abundance	0.993 (0.977-1.008)	1.029 (0.934-1.134)	0.993 (0.977-1.008)	1.029 (0.934-1.134)
Geometric Mean Recruits per Spawner (all broods)	0.729 (0.562-0.948)	1.053 (0.567-1.953)	3.458 (2.548-4.694)	1.695 (0.97-2.963)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.118 (0.793-1.577)	1.369 (0.512-3.658)	4.259 (2.593-6.995)	2.274 (0.987-5.239)
Average Hatchery Fraction	0.169	0.000	0.169	0.000
Average Harvest Rate	0.710	0.445	0.710	0.445
CAPM median extinction risk probability (5^{th}-95^{th} percentiles)	NA	NA	0.180 (0.005 -0.520)	NA
PopCycle extinction risk	NA	NA	0.31	NA

Table 6: Escapement recruitment parameter estimates and relative AIC values for Sandy coho. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (pre-harvest relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.9 (0.77-1.11)	27.9
Random walk with trend	0.73 (0.6-0.93)	NA	0.84 (0.73-1.06)	24.2
Constant recruitment	NA	620 (525-750)	0.67 (0.58-0.84)	4.5
Beverton-Holt	3.02 (1.8-16.41)	890 (619-1261)	0.62 (0.55-0.79)	0
Ricker	1.25 (0.97-1.6)	1007 (849-1443)	0.64 (0.57-0.83)	3.6
Hockey-stick	2.23 (1.58-18.88)	658 (534-787)	0.65 (0.58-0.83)	3.8
MeanRS	1.12 (0.86-1.46)	620 (522-732)	0.46 (0.27-0.64)	28.7

Table 7: Pre-harvest recruitment parameter estimates and relative AIC values for Sandy coho. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.58 (1.36-1.96)	42.1
Random walk with trend	3.46 (2.74-4.59)	NA	0.98 (0.85-1.23)	3.1
Constant recruitment	NA	2941 (2249-4098)	1.14 (0.99-1.43)	15.8
Beverton-Holt	5.12 (3.64-9.85)	11289 (5476-23164)	0.94 (0.82-1.19)	1.2
Ricker	4.78 (3.28-6.52)	6346 (4843-22083)	0.93 (0.82-1.18)	0
Hockey-stick	3.68 (2.91-5.63)	6257 (3945-21576)	0.93 (0.82-1.19)	0.8
MeanRS	4.26 (2.89-6.19)	2939 (2199-3885)	0.96 (0.6-1.33)	6.7

Table 8: Sandy coho CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in 'extirpated or nearly so' category	0.000	0.727	0.982
Probability the population is above 'Moderate risk of extinction' category	0.000	0.310	0.562
Probability the population is above 'Viable' category	0.000	0.028	0.180
Probability the population is above 'Very low risk of extinction' category	0.000	0.001	0.063

A&P – Lower Gorge Tributaries

The Lower Gorge coho population spans the Columbia, with a portion of the population area in Washington. In this evaluation, we will just consider the Oregon side. There is limited data for population abundance and productivity for Lower Gorge coho (on either side of the Columbia). However, these data are confounded by a very high proportion of unmarked hatchery fish present in natural spawning populations. Because data collection has been sporadic and the presence of hatchery fish can only be resolved by reading scales sampled from spawned out fish, it is difficult to confirm whether a self-sustaining natural population exists. We assume that the population is most similar to the Upper Gorge/Hood River population, except that the expected abundance is lower due to the relatively smaller amount of available spawning and rearing habitat (see spatial structure section). We consider the lower gorge population in the 'extirpated or nearly so' or 'high risk' category.

A&P – Hood River/Upper Gorge Tributaries

There are two primary sources of abundance information for the Hood River/Upper Gorge coho population, neither of which is sufficient for a quantitative time series analysis. One source of information is the coho count at Powerdale dam and river mile 4.5 on the Hood River (Olsen 2004). A time series is available for 1992 to 2004 (Figure 30) and hatchery fraction information is also available (Figure 31). The Powerdale data indicate very few natural origin spawners and a high fraction of hatchery origin fish in the population. If we assume that in 1993 there was actually one fish (rather than zero), the geometric mean for natural origin fish over the time series is estimated at 12 fish. This time series is somewhat in contrast with the stratified random survey of coho abundance conducted 2002-2004 (Suring et al. 2006) (Figure 32). Because of the large number of unmarked hatchery fish in this section of the lower Columbia River and the limited collection of scales from adults (to estimate the hatchery fraction), we have a difficult time interpreting the significance of the these results. However, it is clear that a very large number of hatchery fish stray into the both the upper and lower gorge coho habitats. At this point, we consider the Powerdale counts to be a more reliable index of the status of the population, however, a more extensive understanding of the abundance and hatchery fraction for this population is required. Based primarily on the assessment of low abundance and high hatchery fraction at Powerdale, we conclude that the population is likely in the 'extirpated or near so' or 'high risk' categories. The Oregon Native Fish Status report (ODFW 2005) listed this population as "fail" for abundance and "fail" for productivity, also based on the Powerdale index.

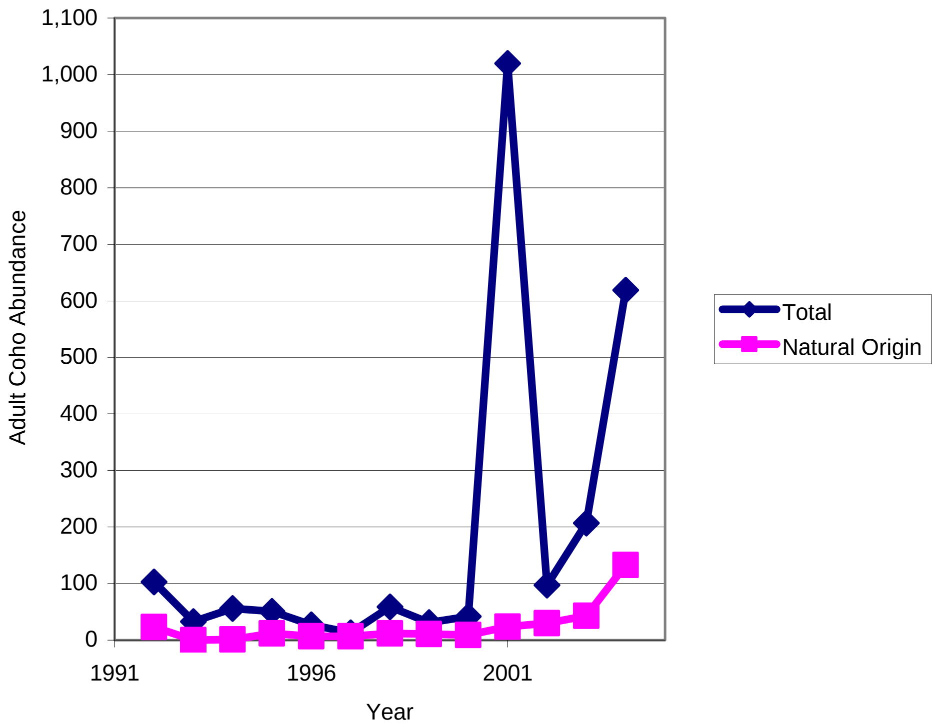


Figure 30: Counts of adult coho salmon (jacks and 3-year-old fish) at Powerdale Dam in the Hood River (Olsen 2004).

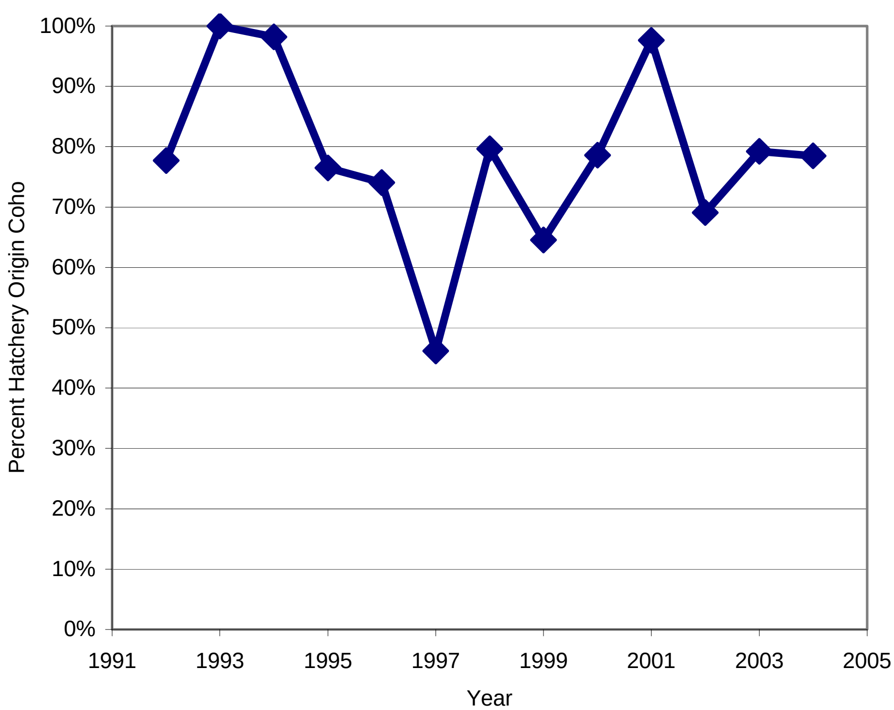


Figure 31: Fraction of hatchery origin spawners at Powerdale Dam in Hood River (Olsen 2004).

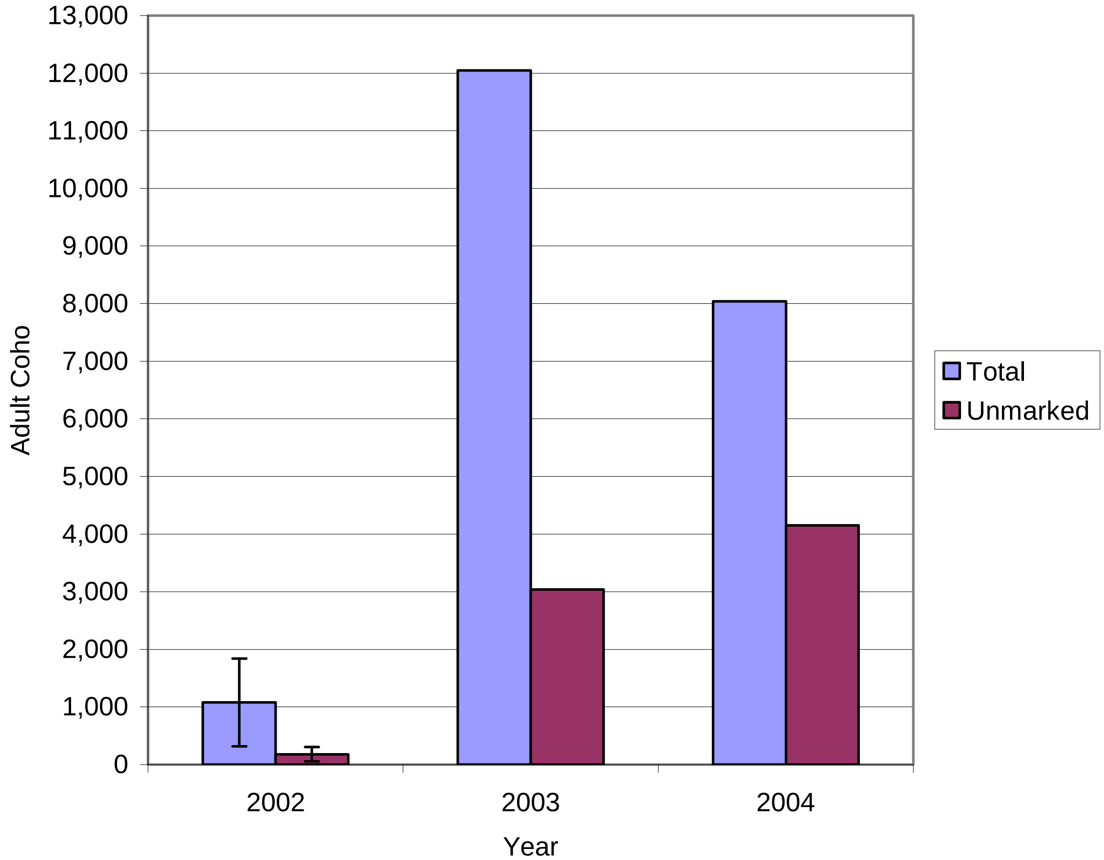


Figure 32: Abundance estimates of adult coho in Upper Gorge and Hood River population (Suring et al. 2006). The 'Total' bars show the estimated total adult coho abundance. The 'Unmarked' bars indicate potential natural origin fish. Many unmarked fish are likely of hatchery origin, so the hatchery fraction is likely even higher than suggested by this graph. The error bars are 95% confidence intervals (only available for 2002).

A&P – Criterion Summary

For the abundance and productivity criterion, the most probable risk category for most of these populations is high or very high (Figure 33). Only one population, the Clackamas, is most probably in the low risk category. The Sandy population is most likely in the high risk category, but the range of possible risk categories is from very high risk to viable. Although there is considerable uncertainty about these 'most probable' classifications, as reflected by the shape of the diamonds (Figure 33), under even the most optimistic interpretation no more than two of the eight populations could possibly fall into the viable classification. From the perspective of this viability criterion, LCR coho populations in Oregon are at high risk.

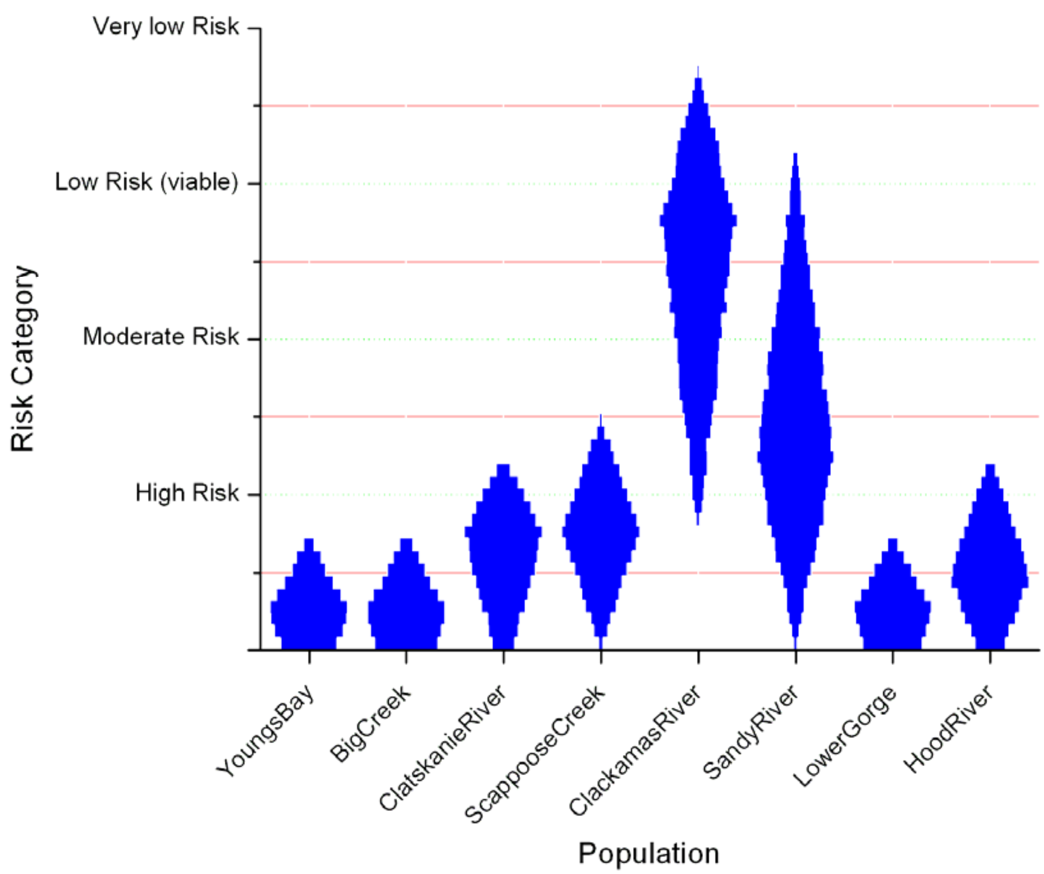


Figure 33: Lower Columbia River coho salmon risk status summary based on evaluation of abundance and productivity only.

III. Spatial Structure

SS – Youngs Bay

Youngs Bay streams including the Skipanon, Lewis and Clark, Klaskanine, and Wallooskee rivers provide an estimated 200 km of usable coho habitat (ODFW 2005) and 563 km of accessible streams (includes higher order streams) (Maher et al. 2005)(Figure 34). Most historical areas remain accessible to anadromous fish (ODFW 2005). A fish ladder provides passage at a Municipal water diversion on the upper Lewis and Clark mainstem. Coho are also trapped and released above hatchery diversion structures on the North Fork Klaskanine. Some loss of accessibility has occurred in higher order tributary streams which were not significant historical coho production areas. Spatial structure has likely been reduced by habitat degradation, particularly in valley floor habitats of the lower basin. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on coho salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for effects of habitat degradation on currently accessible habitats.

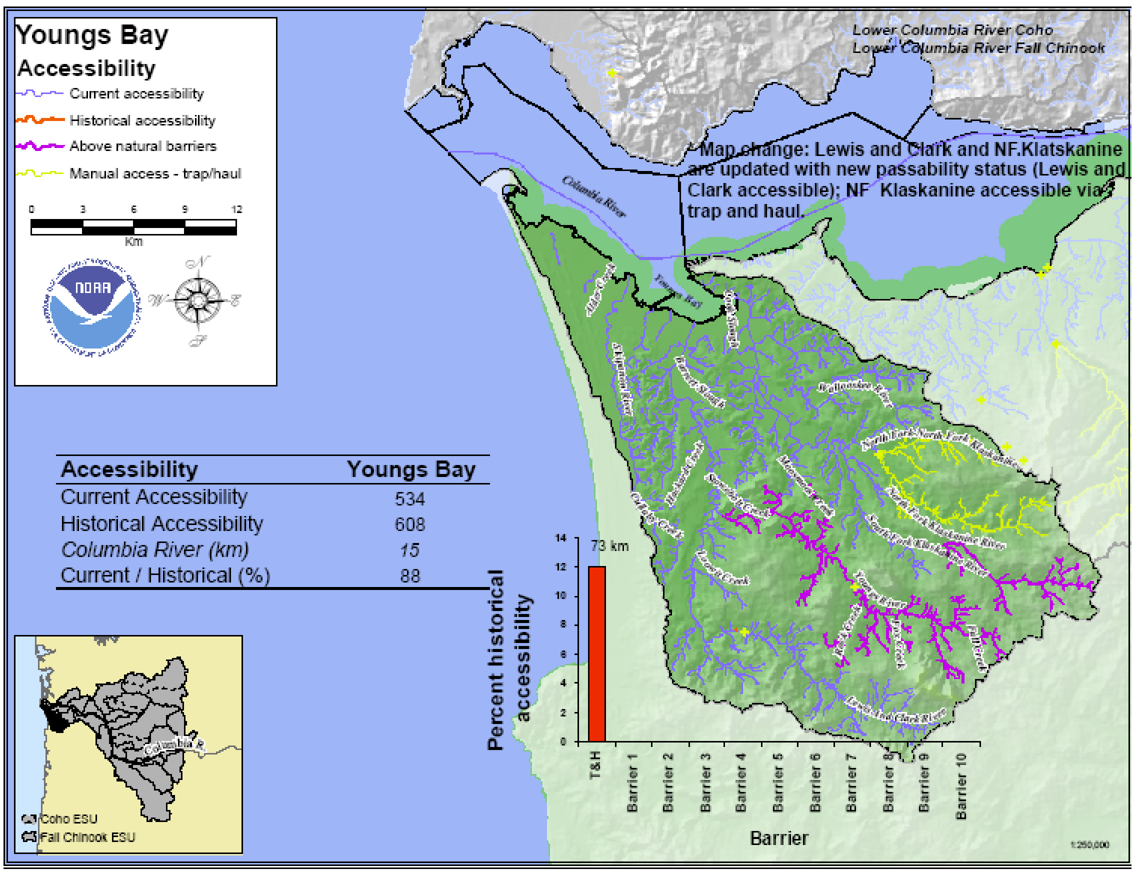


Figure 34 Youngs Bay coho salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Big Creek

Big Creek subbasin streams including the John Day River, Bear Creek, Big Creek, and Gnat Creek historically provided an estimated 180 km of usable coho habitat (ODFW 2005) and historically 352 km of accessible streams (includes higher order streams) (Maher et al. 2005) (Figure 35). Most usable areas (96%) and historically accessible stream km (88%) remain accessible to anadromous fish (ODFW 2005, Maher et al 2005). Hatchery barriers previously limited access to upper Big Creek but since the 2001-2002 return year, all unmarked adult coho returns have been passed upstream of the hatchery weir to utilize the available habitat upstream. A hatchery diversion in upper Gnat Creek blocks coho passage to approximately 6 km of historical habitat but the blocked area is marginal coho habitat. Some loss of accessibility has also occurred in higher order tributary streams which were not significant historical coho production areas. Spatial structure has likely been reduced by habitat degradation, particularly in valley floor habitats of the lower basin. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on coho salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for effects of habitat degradation on currently accessible habitats (-0.5).

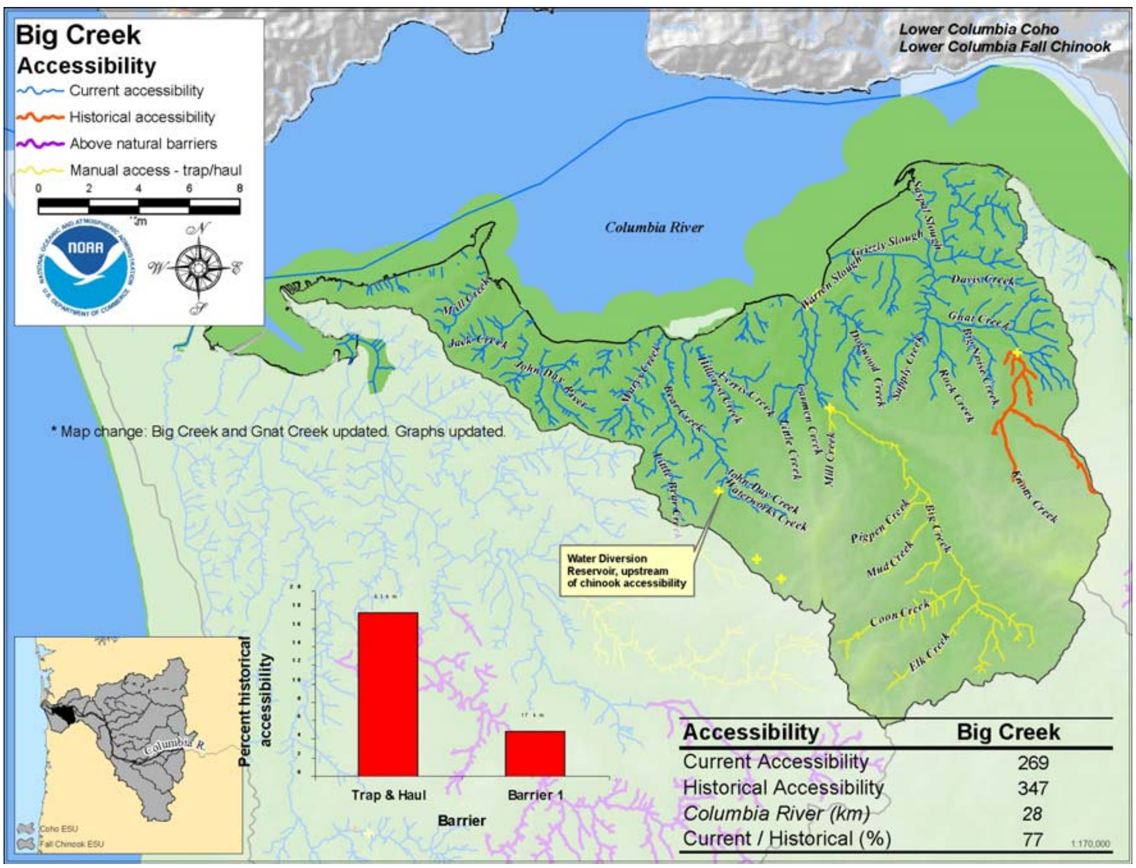


Figure 35 Big Creek coho salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Clatskanie River

Clatskanie subbasin streams, including the Clatskanie River and Beaver Creek, historically provided an estimated 135 km of usable coho habitat (ODFW 2005) and 507 km of accessible streams (includes higher order streams) (Maher et al. 2005) (Figure 36). Most usable areas (92%) and accessible stream km (99%) remain accessible to anadromous fish (ODFW 2005, Maher et al. 2005). Some loss of accessibility has occurred in higher order tributary streams which were not significant historical coho production areas. Spatial structure has likely been reduced by habitat degradation, particularly in valley floor habitats of the lower basin. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on coho salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for effects of habitat degradation on currently accessible habitats (-1).

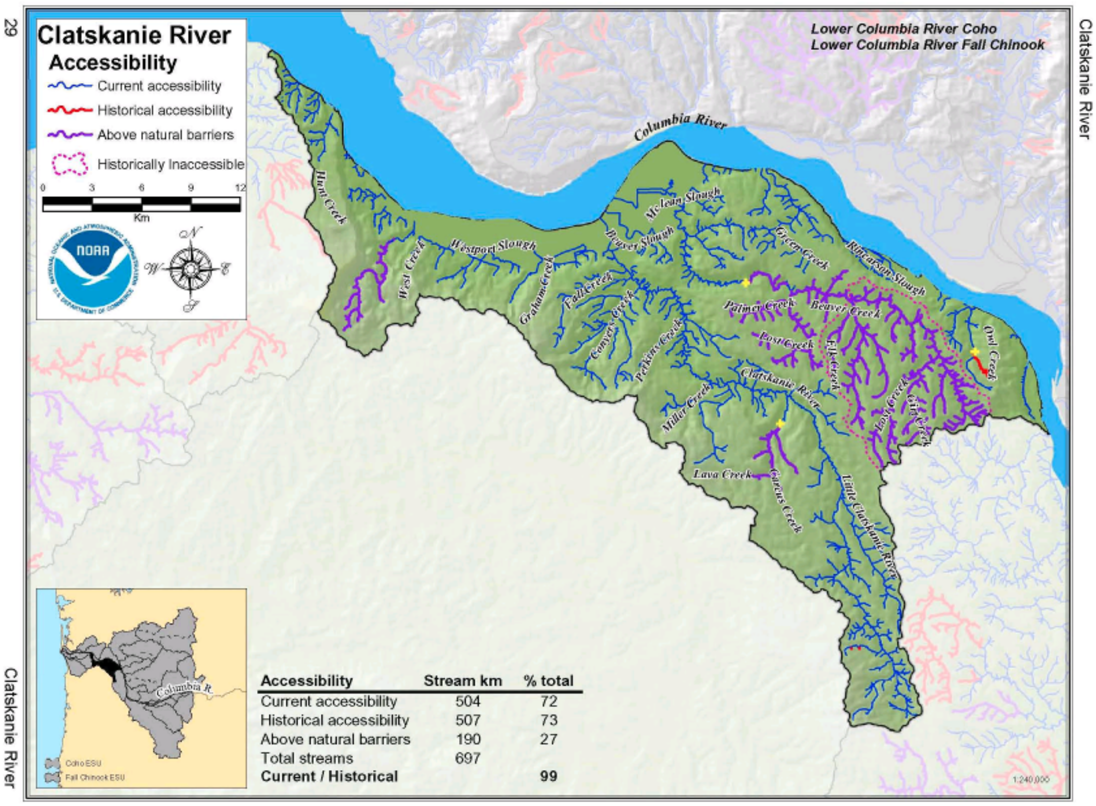


Figure 36 Clatskanie River coho salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Scappoose Creek

The Scappoose subbasin includes a series of small streams including Goble, Tide, Milton, and Scappoose creeks. This area historically provided an estimated 162 km of usable coho habitat (ODFW 2005) and 343 km of accessible streams (includes higher order streams) (Maher et al. 2005) (Figure 37). Most usable areas (92%) and accessible stream km (92%) remain accessible to anadromous fish (ODFW 2005, Maher et al. 2005). Some loss of accessibility has occurred in higher order tributary streams which were not significant historical coho production areas. Spatial structure has likely been reduced by habitat degradation, particularly in valley floor habitats of the lower basin. Habitat changes in the Columbia mainstem and estuary would likely have a significant effect on coho salmon and contributed to adjustments to the spatial structure scores. Access scores were modified for effects of habitat degradation on currently accessible habitats (-0.5).

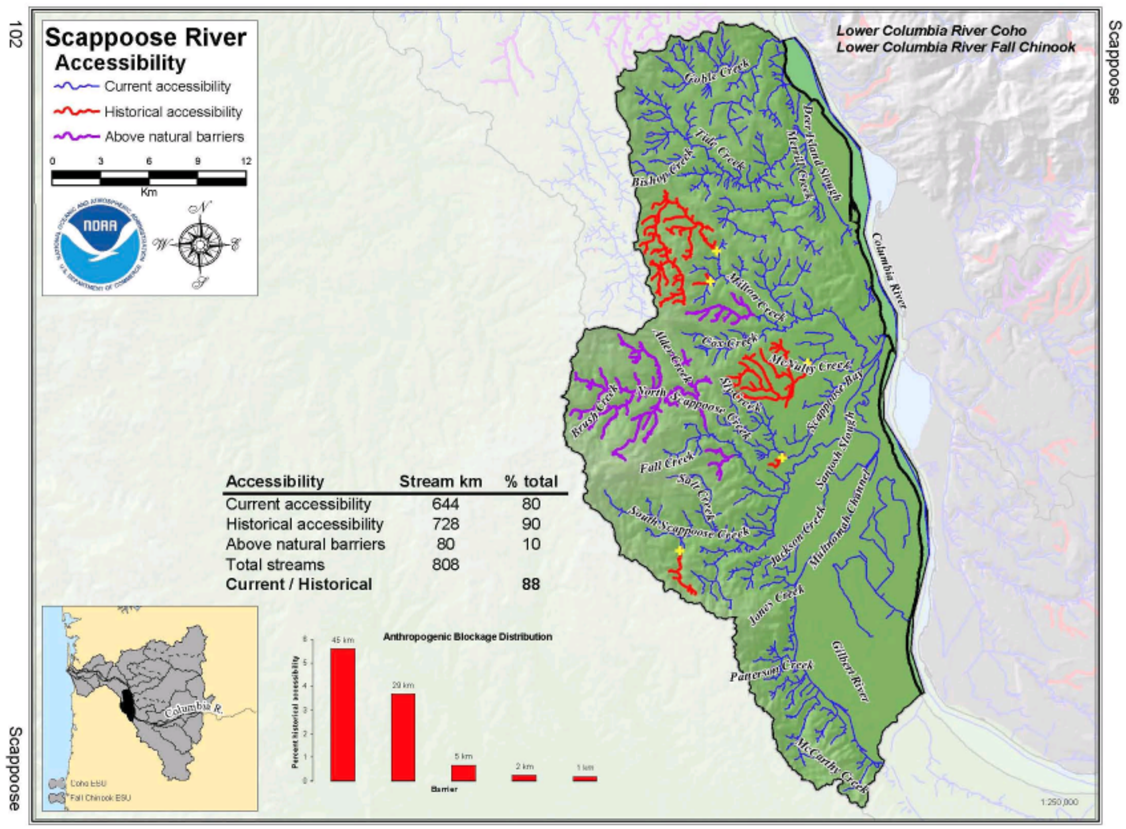


Figure 37 Scappoose Creek coho salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Clackamas River

This area historically provided an estimated 385 km of usable coho habitat (ODFW 2005) and 1,884 km of accessible streams, including higher-order streams (Maher et al. 2005) (Figure 38). Virtually all usable areas (97%) and accessible stream km (96%) remain accessible to anadromous fish (ODFW 2005, Maher et al. 2005). Losses of accessibility are limited to higher order tributary streams, primarily due to watershed development in the lower basin. The upper Clackamas basin contains over half of the historically-suitable habitat for coho and most of that habitat is of high quality today. However, spatial structure has been reduced by significant habitat degradation in lower basin tributaries (e.g., Johnson and Kellogg Creeks). The watershed score was reduced (-0.5) to address a likely loss in spatial diversity related to habitat degradation in the low elevation streams.

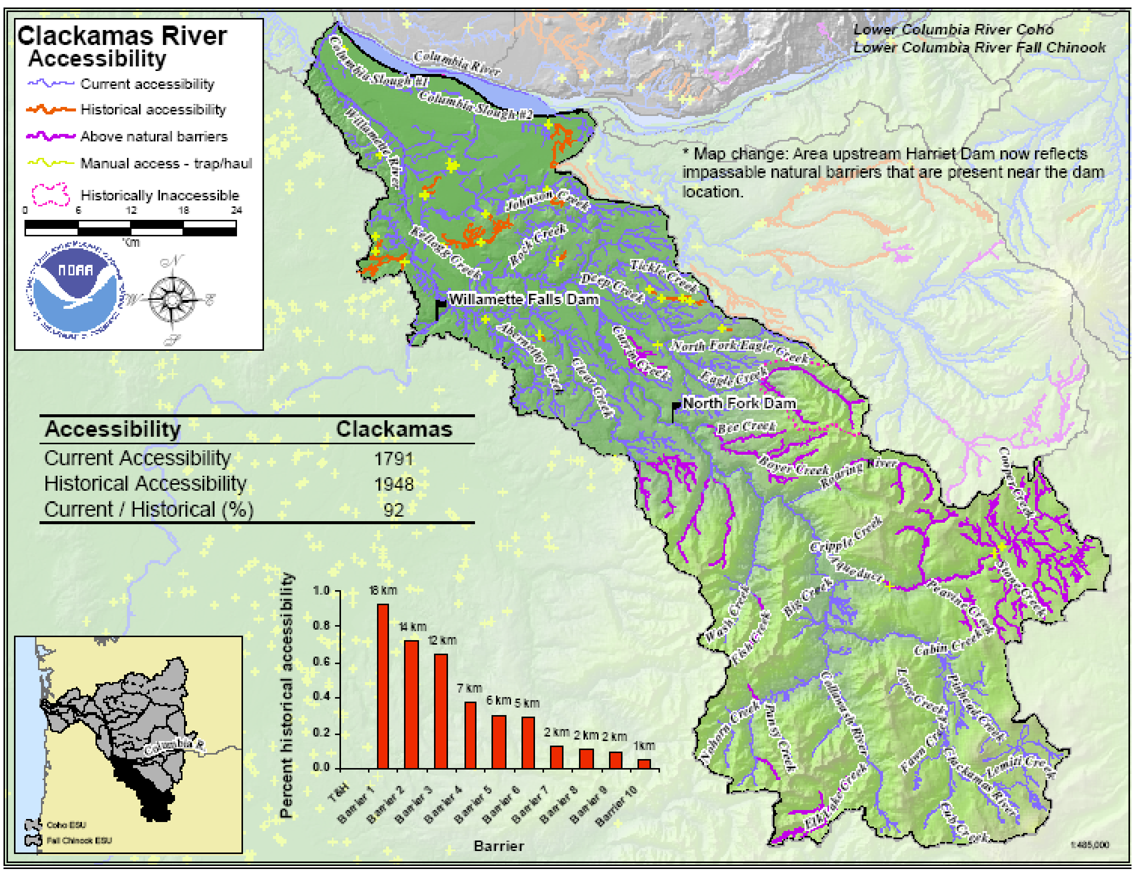


Figure 38: Clackamas coho salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Sandy River

This area historically provided an estimated 264 km of usable coho habitat (ODFW 2005) and 649 km of accessible streams (includes higher order streams) (Maher et al. 2005) (Figure 39). Significant portions (10%) of the historically used coho habitat in the Sandy River have been blocked by dam construction in the Bull Run and Little Sandy watersheds (ODFW 2005). A hatchery weir on Cedar Creek also blocks passage into the upper portions of that tributary. Blocked areas were likely productive habitats for coho. In the remainder of the basin, accessible areas are represented by productive high quality habitat, particularly in the forested upper basin.

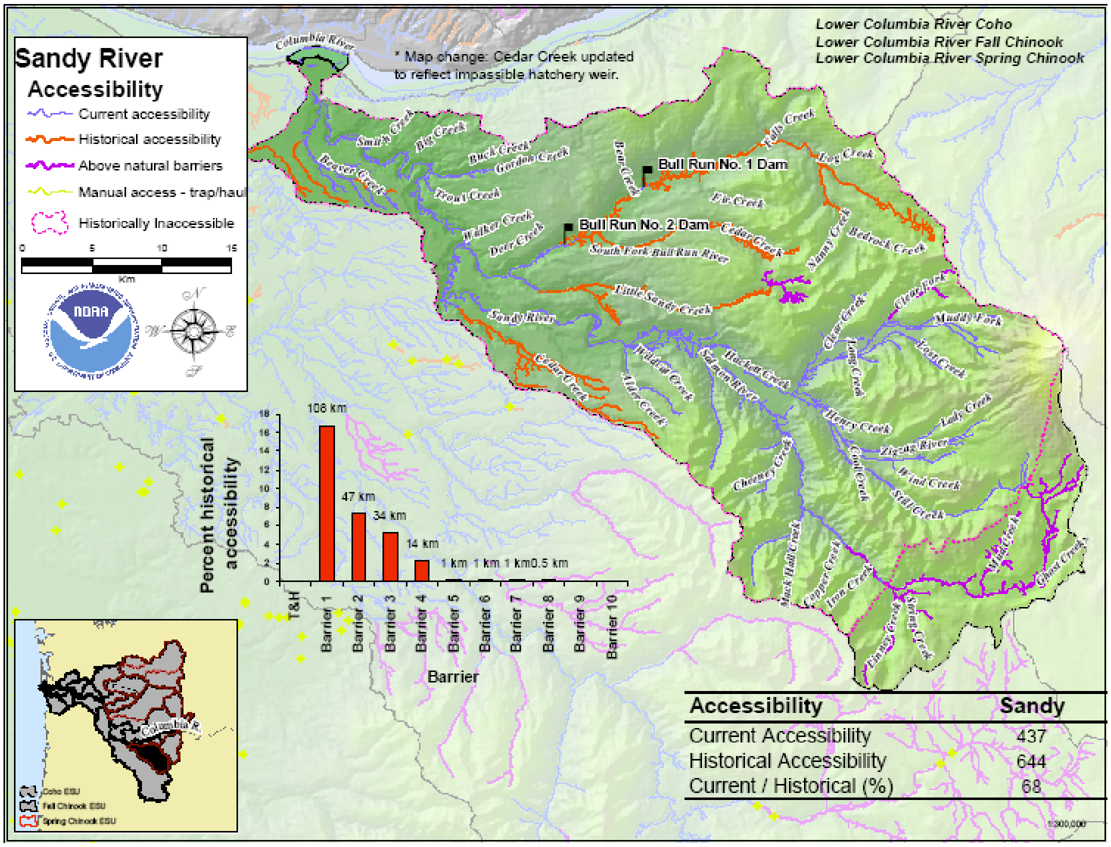


Figure 39: Sandy River coho salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Lower Gorge Tributaries

Most of the small Columbia River gorge streams between the Sandy River and Eagle Creek remain largely accessible to coho (ODFW 2005) (Figure 40). Habitat availability is limited to the lower portions of these streams by topography. Hatchery weirs block coho access to small portions of Tanner and Eagle Creeks. However, because the historical total kilometers of accessible stream is also small for this population, these blockage represent a significant reduction in the percent of historical habitat. The watershed score was reduced (-0.5) to address a likely loss in spatial diversity related to habitat degradation.

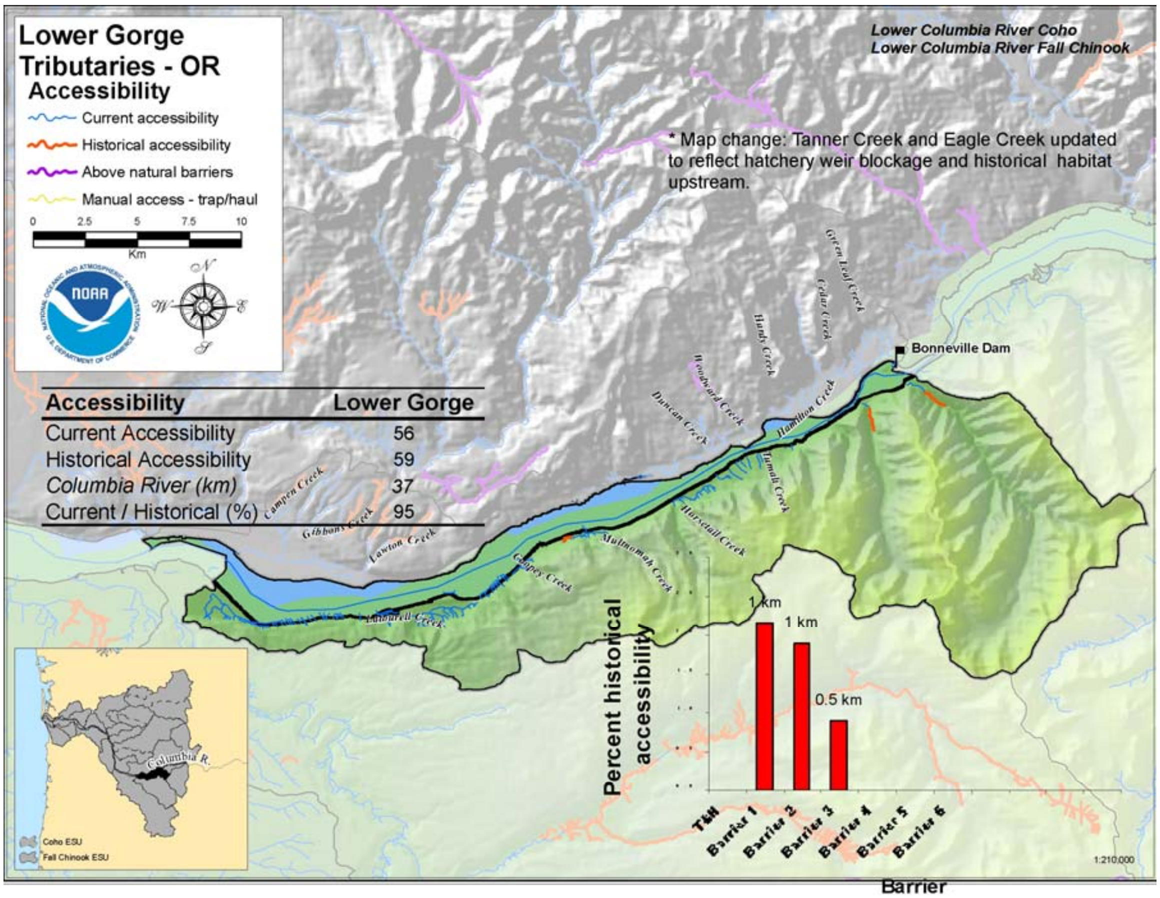


Figure 40: Lower Gorge coho salmon current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Hood River/Upper Gorge Tributaries

This area historically provided an estimated 130 km of usable coho habitat (ODFW 2005) and 609 km of accessible streams (includes higher order streams) (Maher et al. 2005) (Figures 41 and 42). Virtually all usable areas (97%) and accessible stream km (99%) remain accessible to anadromous fish (ODFW 2005, Maher et al. 2005). Blockages are limited to only a few headwater reaches and these streams do not represent significant historical coho production areas. Declines in habitat quality in lower elevations streams of the basin have likely reduced the spatial structure of coho production in the basin. The small Columbia River gorge streams upstream from Eagle Creek remain largely accessible to coho. The amount of habitat is limited to the lower portions of these streams by topography and portions of the lower reaches have been inundated by the Bonneville Dam reservoir. Other local habitat alternations and development have likely reduced habitat quality in some streams. The limited distribution of coho in the basin warrants a downward adjustment to the spatial score. (-1)

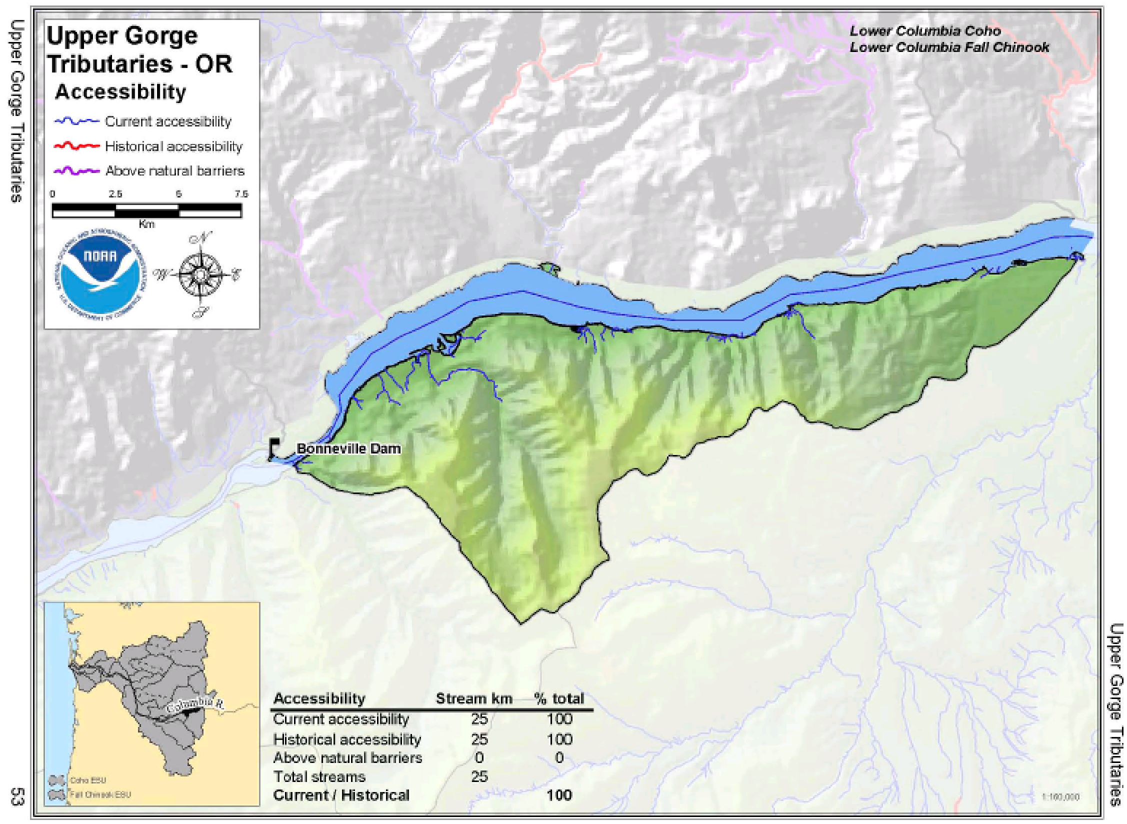


Figure 41 Upper Gorge coho salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use. The Upper Gorge area and Hood River are combined into a single coho salmon population.

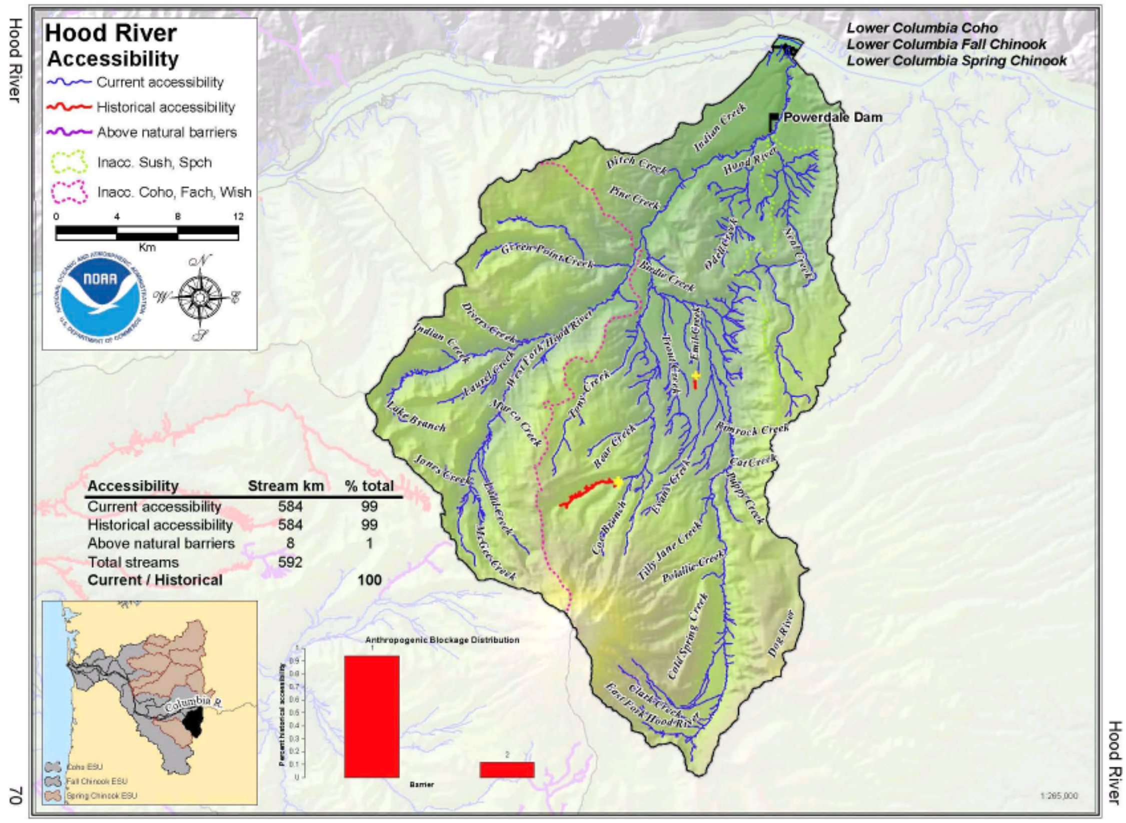


Figure 42 Hood River coho salmon current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Section 1), these graphs depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use. The Upper Gorge area and Hood River are combined into a single coho population.

SS – Criterion Summary

The Sandy River has experienced more than 30 % loss of habitat historically assessable to coho due to anthropogenic blockages and Big Creek and Scappoose Creeks have experienced more than 10% loss (Figure 43). For the other basins, the percent loss has been less than 5%. SS scores for each population were adjusted, where applicable, on the basis of two factors: 1) the suitability/quality of the blocked habitat with respect to coho production and 2) the degree to which the remaining accessible habitat has been degraded from historical conditions. The adjustments and final SS scores for each population are presented in Table 9. Additional details on SS scoring methodology used are provided in Section 1 of this report.

The net assessment of the spatial structure criterion for each population is represented by the diamonds in Figure 44. As described in Section 1 of this report, these diamonds were constructed on the basis of the most likely high, low and mode score for each criterion. The mode score (widest portion of the diamonds in Figure 44) corresponds with the SS rating for each population (Table 9). High and low values (corresponding with the tops and bottoms of the diamonds in Figure 44) were subjectively determined on the basis that the confidence in the accuracy of the SS rating was low for all populations (Table 9). Because of this low confidence, the upper and lower bounds on the SS rating represented a possible score interval that was relatively large. As a result, while the widest portion of the diamonds were at or greater than threshold for low risk category for most of the populations, the lower portion of all the diamonds extended downward into the moderate risk, (non-viable) category.

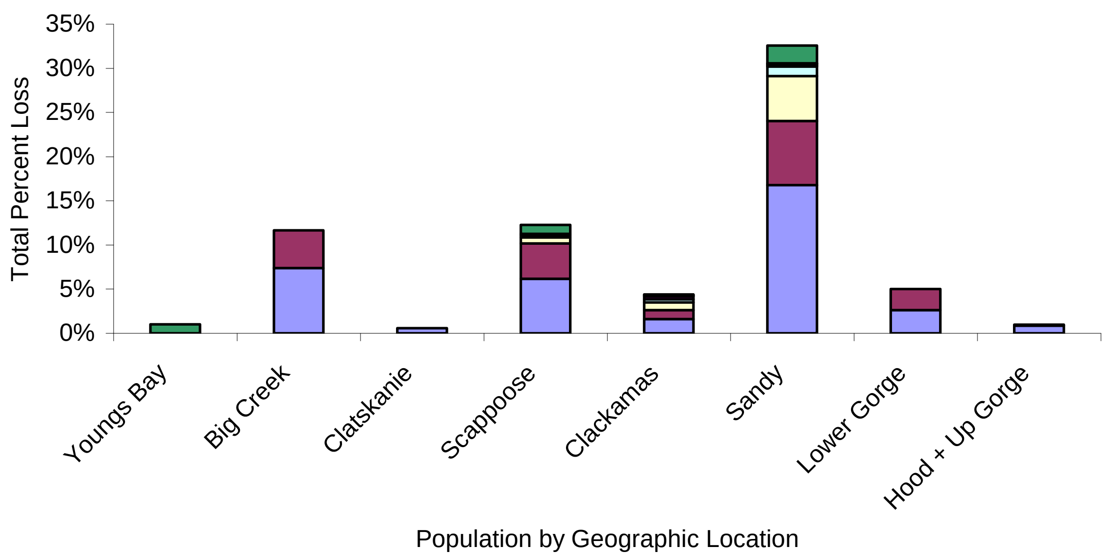


Figure 43: Summary of percent loss in access due to anthropogenic blockages (based on Maher et al. 2004). The total height of the bar indicates total loss. The individual colors represent amount lost by individual blockages. The individual blockages are stacked from largest on the bottom to smallest on the top. These percentage estimates are based on most recent (2007) barrier information that differs from the Maher et al. figures as described in the accessibility map figure legends.

Table 9: Spatial structure persistence category scores for LCR coho populations.

Population	Base Access Score	Adjustment for Large Single Blockage	Adjusted Access Score	SS Rating Considering: Access Score, Historical Use Distribution, and Habitat Degradation	Confidence in SS rating
Youngs Bay	4	No	4	3	Low
Big Creek	3	No	3	2.5	Low
Clatskanie	4	No	4	3	Low
Scappoose	3	No	3	2.5	Low
Clackamas	4	No	3	3	Low
Sandy	2	Yes	1.5	1.5	Low
Lower Gorge Tributaries	3	No	3	2.5	Low
Hood River	4	No	4	3	Low

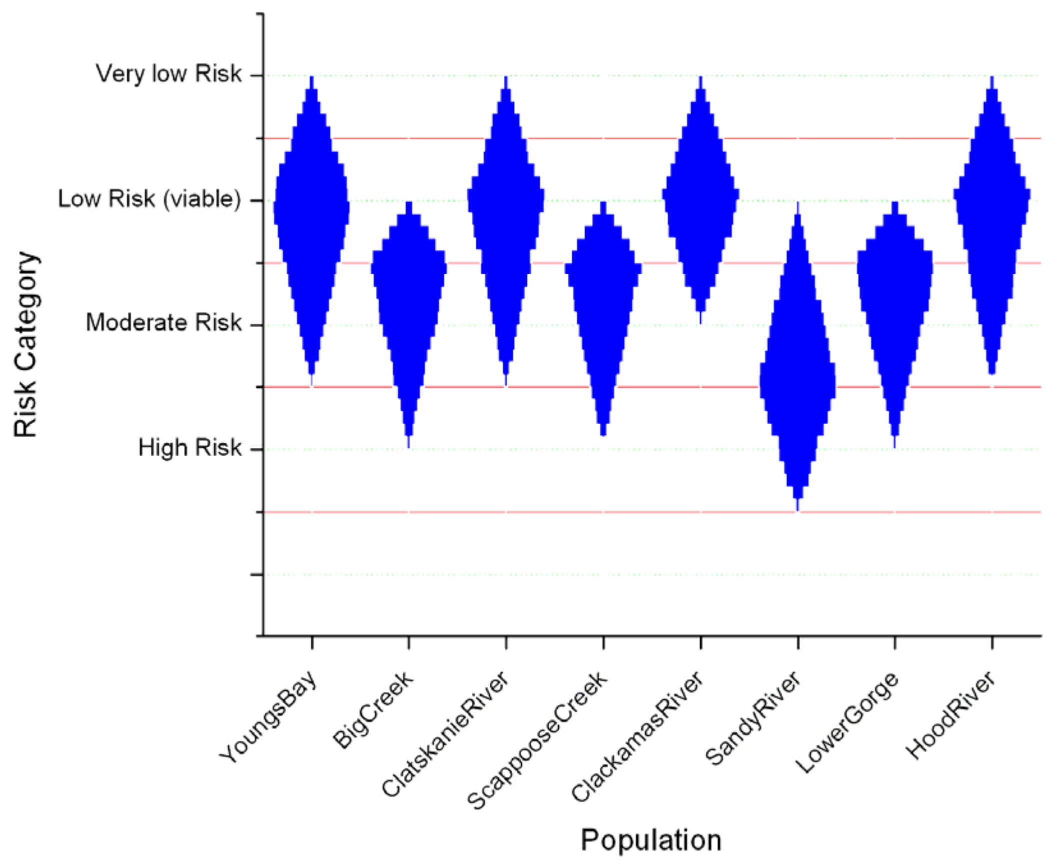


Figure 44 Lower Columbia River coho salmon risk status summary based on the evaluation of spatial structure only.

IV. Diversity Overview

Coho salmon in the Lower Columbia River ESU display one of two major life history types. Early returning, or Type S (for south turning), coho salmon return to freshwater from August to October and spawn from October to November. Coded-wire tagged Type S hatchery fish are predominately recovered off of the Oregon Coast, to the south of the Columbia River, approximately 40% of recoveries (Weitkamp et al. 1995, Weitkamp et al. 2001). The other life history type, late-returning or Type N (north turning) coho salmon, return to freshwater from October through November or December and spawn primarily from November through February, with some fish spawning through to March (WDF et al. 1951). Type N coho salmon have an ocean migration that is predominately north of the mouth of the Columbia River. Differences in ocean migration have been the focus of management strategies to provide fisheries opportunities for certain coastal areas. Ecologically, the run-timing associated with each of these run types is probably more important. It is thought that early returning coho salmon migrate to headwater areas and late-returning fish migrate to the reaches of larger rivers or into smaller stream streams and creeks along the Columbia River (analogous to spring and fall-run chinook salmon). Additionally, coho salmon historically migrating to areas above Bonneville Dam were thought to be early run fish. There does not appear to be much variation in age at emigration to the ocean or in age at maturation. Columbia River coho salmon smolt during their second spring and return to freshwater after one or two years in the ocean. One ocean fish are predominately males (jacks). Analysis of coho salmon scales from adults captured in the Columbia River fishery in 1914, also revealed the presence of two-year old smolts (Marr 1943), although these were thought to have originated from rivers in the Upper Columbia and Snake River Basins.

Genetic analysis of coho populations provides only limited information on population distinctiveness. In the absence of historical baselines for populations and in light of the extensive nature of hatchery transfers, it is difficult to distinguish natural from anthropogenic genetic patterns. While the genetic variability patterns within the Lower Columbia River ESU have been disrupted, substantial differences still exist between the Lower Columbia and Coastal ESUs. These between ESU differences are useful in detecting the legacy of hatchery transfers across ESU boundaries.

As described in the Introduction of this report (Section 1), the diversity criterion rating for each population was based on the evaluation five diversity elements: 1) Life History Traits, 2) Effective Population Size, 3) Impact of Hatchery Fish, 4) Anthropogenic Mortality and 5) Habitat Diversity). Scores for each of these elements were determined and then combined into a single overall diversity category score for each population. A presentation of these results, population by population, follows next.

DV – Youngs Bay

Life History Traits **–** There are insufficient data to evaluate this diversity element for Youngs Bay coho. However, in light of the likelihood that this population became extirpated in the 1990s, the life history traits of the original wild population have been lost. Therefore, we conclude the persistence score for this diversity element should be zero. Those traits currently expressed by the Youngs Bay population most likely originate from the hatchery strays that now predominate the spawning population. Score = 0.0

Effective Population Size **–** Recent surveys have observed low numbers of natural-origin spawners actual abundance may near 50. Score = –0.5

Hatchery Impacts

> *Hatchery Domestication Index* **–** The Klaskanine Hatchery has been in operation since 1911. A number of coho salmon stocks have been imported into hatchery (because of the introduction of numerous stocks with different propagation histories, the PNI estimates may be somewhat higher). Recent surveys estimate the pHOR at 77.3% (2000-2003), although prior to this it is likely to have been nearer 90%. There is no record of pNOB for the hatchery, but unmarked fish are not "intentionally" included in the broodstock. Genetic analysis of Youngs Bay coho salmon indicate a similarity to other LCR coho salmon populations; however, given the magnitude of hatchery introductions it is unknown if this similarity is related to the natural or hatchery-related factors. PNI ≤ 0.1, Fitness = 0.25. Score = 0.5
>
> *Hatchery Introgression* **–** The vast majority of hatchery-origin strays are from coho released from net pens in Youngs Bay (nearly all of these come from Eagle Creek or other upstream Columbia River hatcheries--Sandy River Hatchery, and Oxbow Hatchery (only 563 tagged coho were recovered since 1990). Score = NA
>
> *Synthetic Approach* – A large number of coho salmon juveniles have been released annually into Youngs Bay and its tributaries for several decades. In general, the majority of these fish originate from outside of the Coastal stratum. Recent estimates indicate that over 75% of the spawning coho salmon observed are of hatchery origin (Ph>0.75) with a low or very low genetic similarity between wild and hatchery fish. Diversity persistence score = 0.0.

Anthropogenic Mortality **–** Although the target of this fishery is earlier returning hatchery fish, it is possible the impact rates on the later returning naturally produced fish are higher than then the 25% estimated for most other LCR coho populations. In addition, the existing fishery exerts a very strong selection against the early portion of the return. Prior to the 1990s the harvest rate was higher, perhaps up to 90%. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 1.0.

Habitat Diversity – The habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. Loss of estuary habitat types has been substantial since the mid-1800s. The diversity scores were adjusted downward to reflect this (indicated as a "-" score). Score = 2.0.

Youngs Bay Coho Overall Diversity Score = 0.5.

DV – Big Creek

Life History Traits – There are insufficient data to evaluate this diversity element for Big Creek coho. However, it is likely that this population became extirpated in the 1990s resulting in the loss of the life history traits of the original wild population. Therefore, we conclude the persistence score for this diversity element should be zero. Those traits currently expressed by the Big Creek population most likely originate from the hatchery fish produced at Big Creek hatchery. Score = 0.0.

Effective Population Size – Recent surveys have observed low numbers of natural-origin spawners (zero in some years), actual abundance may have averaged between 50 and 100.Score = 0.5.

Hatchery Impacts

> *Hatchery Domestication (PNI)* – The Big Creek Hatchery has been in operation since 1938. A substantial number of coho salmon have been released into the Big Creek watershed. Big Creek Hatchery does not include unmarked (wild) fish into its broodstock (pNOB= 0), while the pHOR in the Youngs Bay/Big Creek watershed averaged 90% hatchery fish. Genetic analysis of the hatchery broodstock indicates that it is closely related to other LCR coho hatchery stocks. In the last ten years, unmarked coho salmon have been passed over the hatchery weir on Big Creek. This has restored access to a considerable portion of the watershed and created an "all-natural" spawning area above the weir. Returns have numbered a few hundred fish in the last few years. Because of the relatively short duration of this program to date and the long term predominance of hatchery fish in the system, the PNI score was adjusted only slightly to reflect recent conditions. *PNI ≤ 0.2, Fitness = 0.45*
>
> *Hatchery Introgression* – The vast majority of hatchery-origin strays are from the local Big Creek Hatchery, although a few other within ESU strays have been observed (nearly all hatchery origin coho salmon are marked, but few have origin-source tags).
>
> *Synthetic Approach* – The Big Creek Hatchery has released a stock of mixed locally-derived and introduced coho salmon for several decades. Few if any wild (unmarked) fish are included in the broodstock and the proportion of hatchery fish spawning naturally has consistently been near 50% ($0.30<Ph<0.75$) with a low to very low genetic similarity between wild and hatchery fish. Diversity persistence score = 0.5.

Anthropogenic Mortality – Nearby Tongue Point and Blind Slough commercial fisheries potentially have significant impacts on this population. Although the targets of these fisheries are earlier returning hatchery fish, it is possible the impact rates on the naturally produced fish are higher than the 25% estimated for most other LCR populations. In addition, the existing fishery exerts a strong selection against the early portion of the return. Fishery impact rates in the range of 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0.

Habitat Diversity – The habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. Loss of estuary habitat types has been substantial since the mid-1800s. The diversity scores were adjusted downward to reflect this. Score = 2.0.

Big Creek Coho Overall Diversity Score = 1.0.

DV – Clatskanie River

Life History Traits – The paucity of data for this population make the evaluation of this diversity element difficult. However, this population likely went through a severe bottleneck during the 1990s and may have in fact become extirpated. Recent spawning surveys show an increasing number of naturally produced spawners and a relatively low proportion of hatchery fish. The spawn timing of these natural fish appears to be during the November to January time-frame which may be similar to that of the historical coho populations in this region of the lower Columbia. Score = 2.0.

Effective Population Size – Recent surveys have observed low numbers of natural-origin spawners (zero in some years during the 1990s), estimated wild spawner abundance = 74-217 (2002-2004). Score = 2.0.

Hatchery Impacts

Hatchery Domestication (PNI) – The Gnat Creek Hatchery has intermittently released coho salmon. The proportion of hatchery-origin fish has fluctuated considerably, depending, in part, on the intensity of hatchery operations. Genetic analysis of the hatchery broodstock indicates that it is closely related to other LCR coho hatchery stocks. Given the limited level of genetic sampling for this population, it is not possible to discern more population specific information.

PNI ≤ NA, hatchery program intermittent – stray metric used

Hatchery Introgression – The majority of hatchery-origin strays are from local hatcheries producing within ESU coho salmon. Recent stray rates have fluctuated (0 to 67%, average 28.6%). Score = 2.0.

Synthetic Approach – Hatchery coho salmon have not been recently released into the Clatskanie River; however, the proportion of naturally-spawning hatchery fish remains high ($0.10<Ph<0.35$). It is likely that these fish come from nearby hatchery programs (in both Oregon and Washington). Genetic similarity between wild and hatchery-origin fish is presumed to be low. Diversity persistence score =2.0.

Anthropogenic Mortality – Mainstem Columbia and ocean fisheries exert a moderate impact on this population, probably in the range of a 20% to 35% mortality rate. However, the timing of the Columbia River fisheries are thought to select against those portions of the population that return during what was historically the middle of the run timing. In addition, fishery impact rates in the range of 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0.

Habitat Diversity – The habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. Loss of estuary habitat types has been substantial since the mid-1800s. The diversity scores were adjusted downward to reflect this (indicated as a "-" score). Score = 2.5.

Clatskanie River Coho Overall Diversity Score = 2.0.

DV – Scappoose Creek

Life History Traits – The paucity of data for this population make the evaluation of this diversity element difficult. However, this population likely went through a severe bottleneck during the 1990s and may have in fact become extirpated. Recent spawning surveys show an increasing number of naturally produced spawners and a relatively low proportion of hatchery fish. The spawn timing of these natural fish appears to be during the November to January time-frame which may be similar to that of the historical coho populations in this region of the lower Columbia. Score = 2.0.

Effective Population Size – Scappoose Creek has been surveyed for spawning coho salmon since the late 1940s. Early surveys provide only a rough estimate of total abundance, but it is likely that, on average, over a hundred natural-origin coho salmon return to the basin. Score = 2.0.

Hatchery Impacts

> *Hatchery Domestication (PNI)* – There is no hatchery in the Scappoose Creek Basin. Furthermore, there have been relatively few introductions of coho salmon. During the 1980s, there were widespread releases of coho salmon pre-smolts and surplus hatchery adults, although the survival and spawning success of these fish is thought to have been fairly low. Genetic analysis of natural spawners suggests that this population is somewhat distinct form other populations (potentially because of the minimal hatchery influence or small Ne or both). Score = NA.
>
> *Hatchery Introgression* – The proportion of hatchery-origin fish recovered on the spawning grounds is generally low (<10%). It is probable that most of these hatchery fish are from within the ESU. Score = 2.0.
>
> *Synthetic Approach* – There is no hatchery program in Scappoose Creek, nor has there been one in the past. Additionally, hatchery releases have been limited and intermittent. The proportion of hatchery fish spawning naturally is thought to be low (0.10<Ph), although surveys and carcasses recoveries have been limited. It is likely that many of the hatchery fish originate from the large Washington hatchery programs immediately across the Columbia River. Diversity persistence score =2.0 – 3.0.

Anthropogenic Mortality – Mainstem Columbia and ocean fisheries exert a moderate impact on this population, probably in the range of a 20% to 35% impact rate. However, the timing of the Columbia River fisheries are thought to select against those portions of the population that return during what was historically the middle of the run timing. In addition, fishery impact rates in the range of 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0. .

Habitat Diversity – The habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. Loss of estuary habitat types has been substantial since the mid-1800s. The diversity scores were adjusted downward to reflect this (indicated as a "-" score). *Diversity.* Score = 2.0.

Scappoose Creek Coho Overall Diversity Score = 2.0.

DV – Clackamas River

Life History Traits – Although this coho population is one of the two in the LCR that is known to have persisted through the poor marine survival period of the 1990s, it was at very low levels during this period and may have experienced the effects of a genetic bottleneck, In addition, the run timing seems to be in a state of flux. The unimodal timing of the early 1960s, shifted to more protracted and bimodal timing by the 1980s. It is not clear if this change was brought on by natural processes impacting the wild population, introduction of a coho stock with earlier run timing in the late 1960s, or selective pressures due to Columbia fisheries or all three. In recent, years it appears the run timing may be returning to a more unimodal pattern more typical of the early 1960s. Score = 3.0.

Effective Population Size – Surveys indicate that several hundred unmarked coho salmon spawned in the Lower Clackamas River from 2002 to 2004, in addition to the several hundred to a few thousand unmarked coho that are passed above the North Fork Dam. It should be noted that the coho run size probably underwent bottlenecks in the mid-1970s and mid-1990s. Further habitat conditions in the lower Clackamas River and associated tributaries (including Johnson and Kellogg Creeks) are generally poor, suggesting that many of these "unmarked" spawners are not the result of natural production, but may be hatchery-origin fish. Score = 3.0.

Hatchery Impacts

Hatchery Domestication (PNI) – The Eagle Creek NFH releases early run coho salmon, and has received a number of transfers from other hatcheries within the ESU. Genetically the Eagle Creek NFH is somewhat similar to the earlier portion of the wild fish returing to the Clackamas River. The Eagle Creek NFH broodstock was founded in 1958 by fish from the Sandy River Hatchery, but has received introductions from a number of other LCR hatcheries. Wild fish are not included in the hatchery broodstock. With the 100% fin marking of all hatchery coho releases in the 1990s, it became evident that hatchery fish (presumably from Eagle Creek hatchery) only rarely entered the Faraday fish ladder in an attempt to stray into the Clackamas basin upstream of North Fork Dam. In recent years those few stray hatchery fish that entered the fish handling Faraday fish handling facility have been removed from the basin, creating a "hatchery-free" zone in the upper basin. However, from 2000-2002 hatchery fish derived from the local wild population were passed upstream of the dams in an effort to supplement the production. Downstream of North Fork Dam, hatchery strays are commonly observed spawning with wild fish. The basin-wide proportion of hatchery strays varies annually, but in recent years it has averaged 0.28. A rough average of 50% was used in the PNI. Hatcheries do not include unmarked "wild" fish into the broodstock. Average hatchery strays (50% below, 5% above) = 25%. *The isolate nature of Eagle River NFH suggests that using the stray metric might be more appropriate. Score = NA.*

Hatchery Introgression – The vast majority of hatchery-origin strays are from the Eagle Creek Hatchery, although a few other within ESU strays have been observed (nearly all hatchery-origin coho salmon are marked, but few have origin-source tags).

The stray metric was used, with an average stray rate of 25% and adjusted for mostly local hatchery broodstock. Score = 2.0.

Synthetic Approach – With the exception of transplants of adult hatchery made in the 1960s and a “conservation hatchery” program in the 1990s, most of the fish spawning above North Fork Dam have been wild fish. In recent years, the few hatchery fish that attempted to migrate past North Fork Dam, have been removed at the fish sorting facility. The hatchery contribution to the naturally-spawning early run is thought to be relatively low (Ph<0.10).. The early-returning coho salmon hatchery program (Eagle Creek NFH) has incorporated a coho from a number of sources including locally from the Clackamas River (although they do not presently include unmarked broodstock). Diversity persistence score =2.0 – 3.0.

Anthropogenic Mortality – Mainstem Columbia and ocean fisheries exert a moderate impact on this population, probably in the range of 20% to 35% impact rate. However, the timing of the Columbia River fisheries are thought to select against those portions of the population that return during what was historically the middle of the run timing. In addition, fishery impact rates in the range of a 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0.

Habitat Diversity – The habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. The loss of estuary habitat types and mainstem and side channel riparian habitat has been substantial since the mid-1800s. The migratory and juvenile rearing areas include the urbanized portions of the lower Willamette River and Multnomah Channel and Sauvie Island. The diversity scores were adjusted downward to reflect this (indicated as a “-” score). Score = 2.0.

Clackamas River Coho Overall Diversity Score = 2.75.

DV – Sandy River

Life History Traits – Although this coho population is one of the two in the LCR that is known to have persisted through the poor marine survival period of the 1990s, it was at very low levels during this period and may have experienced the effects of a genetic bottleneck, Historical information on run and spawn timing from early in the 1900s is available from hatchery and fisheries records. Comparative information from fish counts made at Marmot Dam and spawning survey information collected from the 2002-2006 suggest that no large changes in life history traits have occurred. Score = 3.0.

Effective Population Size – Spawner abundance estimates are available for Sandy River coho salmon from 1960. The harmonic mean abundance for this period was 499. Historical estimates of abundance suggest that between 10 and 20 thousand coho normally returned to the Sandy River. Score = 3.0.

Hatchery Impacts

Hatchery Domestication (PNI) **–** The impact of hatchery fish in Sandy River Basin is broken into two distinct regions, the watershed above and below Marmot Dam. The area downstream of Marmot Dam represents 10% of the natural coho production area, the remaining 90% is upstream of the dam. The proportion of hatchery fish below Marmot is high, > 80% most years, while upstream of the dam hatchery fish typically represent less than 5% of the spawning population. The basinwide proportion of hatchery fish in recent years has been less than 0.10. Accessible habitat below Marmot Dam contains a mixture of hatchery and natural-origin fish, and accessible habitat above Marmot Dam contains unmarked "wild" fish. The watershed below Marmot Dam accounts for less than 20% of the currently accessible habitat, hatchery contribution varies and carcass recovery is low, estimated pHOR ≥ 75% and the pNOB ≤ 5%. The Sandy River Hatchery has been in operation since 1953, with relatively few introductions from out-of-basin. However, wild fish have not been routinely added to the hatchery broodstock. Genetic analysis does not indicate any strong divergence from other Lower Columbia River populations, or any similarity to coho salmon from other ESUs. *PNI =1.0 (above dam), PNI=0.1 (below dam), 18 generations. Score = 2.0.*

Hatchery Introgression – HOR fish from the Sandy River Hatchery were considered part of the population and their effect was considered in the PNI metric. Out of basin strays are generally rare. Score = 3-4.

Synthetic Approach – The Sandy River Basin is contains two distinct regions relative to the influence of hatchery-origin fish. Since 1999, hatchery-origin fish have been blocked from migrating past the Marmot Dam trap, while the area below the Dam contains a very high proportion of hatchery origin fish (nearly 80%). The area downstream of Marmot Dam represents 10% of the natural coho production area, the remaining 90% is upstream of the dam. The basinwide proportion of hatchery fish in recent years has been less than 0.10. The Sandy River Hatchery has been in operation since 1953, with relatively few introductions from out-of-basin; however, wild (unmarked) fish have not been routinely added to the hatchery broodstock. Genetic similarity is thought to be low to moderate. Diversity persistence score =3.0.

Anthropogenic Mortality – Mainstem Columbia and ocean fisheries exert a moderate impact on this population, probably in the range of a 20% to 35% impact rate. However, the timing of the Columbia River fisheries are thought to select against those portions of the population that return during what was historically in the later portion of the run timing. In addition, fishery impact rates in the range of 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0

Habitat Diversity – The habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. The loss of estuary habitat types and mainstem and side channel riparian habitat has been substantial since the mid-1800s. The diversity scores were adjusted downward to reflect this (indicated as a "-" score).

Score = 1.5.

Sandy River Coho Overall Diversity Score = 2.5.

DV – Lower Gorge Tributaries

Life History Traits – Streams on the Oregon side of the Lower Columbia River Gorge contain relatively little accessible spawning habitat. Historically, there was little effort made to survey these streams, but it appears that late-run coho salmon occupied the habitat. There are insufficient data to evaluate this diversity element for this population of coho. However, it is likely this population became extirpated in the 1990s resulting in the loss of the life history traits of the original wild population. Therefore, we conclude the persistence score for this diversity element should be zero. Those traits currently expressed by this population most likely originate from the hatchery strays from the Bonneville hatchery complex that now predominate the spawning population.

Score = 0.0.

Effective Population Size – Abundance estimates for Oregon side of the Lower Columbia River Gorge population are based on only 5% of the accessible habitat. The estimated average abundance of the naturally produced fish in this population is at critically low levels, N < 50. Additionally, this limited number of spawners is spread across a number of smaller tributaries. Score = 0.5.

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** Tributaries in the Lower Columbia River Gorge population contain a high proportion of hatchery strays. These hatchery fish originated from broodstock of multiple origins, from both within and outside of the gorge stratum. No wild fish are incorporated into the broodstock. The proportion of hatchery coho on the spawning grounds in recent years has been in excess of 0.80. Score = 0.0 Tributaries in the Lower Columbia River Gorge population contain a high proportion of hatchery strays (pHOR ≥ 80%) probably from one of a number of Bonneville complex hatcheries (all of which have highly varied broodstock sources). There is little information available on the pNOB for these hatcheries, but based on the relative proportion of unmarked fish in the overall population pNOB ≤ 10%. PNI =0.1 with an estimated 20 generations. Fitness loss near 65%. Score = 1.0.
>
> *Hatchery Introgression* – Given the variety of broodstock sources used in hatcheries that have influenced this population it is possible to evaluate hatchery influence using either the PNI metric or the within ESU stray metric. In either case the diversity score would indicate a high degree of risk. Stray Rate Metric = 1 (if used in place of the PNI metric)
>
> *Synthetic Approach* – The Lower Gorge Tributaries are thought to be heavily influenced by large releases of hatchery coho salmon from Bonneville Hatchery on the Oregon side and a number of hatcheries on the Washington side. The broodstock for these hatcheries are generally of mixed-stock origin from basins within the Lower Columbia River. Estimates of hatchery-origin contribution to spawning escapement are in excess of 75% (Ph,0.75). Diversity persistence score = 0.0.

Anthropogenic Mortality – Mainstem Columbia and ocean fisheries exert a moderate impact on this population, probably in the range of a 20% to 35% impact rate. However, the timing of the Columbia River fisheries are thought to select against those portions of the population that return during what was historically in the later portion of the run timing. In addition, fishery impact rates in the range of 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0.

Habitat Diversity – The total amount and diversity of habitat available to the natural coho population in this region is extremely limited, even in its native state. Therefore, the net score was downgraded to reflect this fact. In addition,the habitat diversity index scores derived from the worksheet do not include habitat in the Columbia River estuary. The loss of estuary habitat types and mainstem and side channel riparian habitat has been substantial since the mid-1800s. The diversity scores were adjusted downward to reflect this effect as well. Score = 0.5.

Lower Gorge Tributaries Coho Overall Diversity Score = 0.5.

DV – Hood River/Upper Gorge Tributaries

Life History Traits – Coho salmon exist in this population at a very depressed level of abundance. Historical and present-day information is very limited, and primarily concerns run and spawn timing. Coho salmon in the short, low lying, Gorge tributaries appear to exhibit a late-run timing, while fish entering the Hood River Basin may represent an early-run timed run. There are insufficient data to evaluate this diversity element for this population of coho. However, it is possible that wild coho were extirpated during 1990s, causing the loss of the life history traits of the original wild population. Therefore, we conclude the persistence score for this diversity element should be zero. Those traits currently expressed by this population most likely originate from the hatchery strays from the Bonneville hatchery complex that now predominate the spawning population. Score = 0.0.

Effective Population Size – Abundance estimates for Oregon side of the Upper Columbia River Gorge population are based on only 5% of the accessible habitat. The estimated average abundance of NORs in the Gorge tributaries is at a low level, N < 50. Additionally, this limited number of spawners is spread across a number of smaller tributaries. Fish counts at Powerdale Dam, on the Hood River, indicate that the coho run has averaged below 50 fish in the last 15 years. Score = 1.0.

Hatchery Impacts

> *Hatchery Domestication (PNI)* **–** Tributaries in the Upper Columbia River Gorge population contain a high proportion of hatchery fish (pHOR ≥ 80%) that are likely strays from the Bonneville hatchery complex. These hatchery stocks were developed from a number of sources both within and outside of the stratum. Further, wild fish are not used as a portion of the hatchery broodstock. The proportion of hatchery coho on the spawning grounds in recent years has been in excess of 0.80. Score = 0.0. There is little information available on the pNOB for these hatcheries, but based on the relative proportion of unmarked fish in the overall population pNOB ≤ 10%.
>
> *PNI =1.0 with an estimated 20 generations. Fitness loss near 65%. Score = 1.0.*
>
> *Hatchery Introgression* – Stray hatchery fish come from a variety of sources. Local hatcheries contain broodstocks that have been strongly influenced by a number of out-of-basin sources. Calculation of hatchery effects could be done either using the PNI metric or the within ESU metric.
>
> *Stray Rate Metric = 1 (if the PNI metric is not used).*
>
> *Synthetic Approach* – As with the Lower Gorge Tributaries, spawning aggregations in the Upper Gorge Tributaries are thought to be heavily influenced by large releases of hatchery coho salmon from Bonneville Hatchery on the Oregon side and a number of hatcheries on the Washington side. The broodstock for these hatcheries are generally of mixed-stock origin from basins within the Lower Columbia River. Estimates of hatchery-origin contribution to spawning escapement are in excess of 75% (Ph,0.75). Diversity persistence score = 0.0.

Anthropogenic Mortality – Mainstem Columbia and ocean fisheries exert a moderate impact on this population, probably in the range of a 20% to 35% impact rate. However, the timing of the Columbia River fisheries are thought to select against those portions of the population that return during what was historically in the later portion of the run timing. In addition, fishery impact rates in the range of 75% to 90% were experienced by this population from the 1950s to the early 1990s. It is unknown what the legacy of this impact has been on the genetic character of the populations. Score = 2.0..

Habitat Diversity – Much of the spawning habitat for coho salmon in the Upper Gorge DIP was flooded with the filling of the Bonneville Pool. Within the Hood River basin, the historically highest quality coho habitat has been adversely impacted by agricultural and urban development. Score = 1.0.

Hood River/Upper Gorge Tributaries Coho Overall Diversity Score = 1.0.

DV – Criterion Summary

With the exception of the Clackamas and Sandy populations, it is likely that most of the wild LCR coho populations were effectively extirpated in the 1990s. Therefore, the genetic diversity of the original wild populations was nearly lost. Although naturally produced fish have reappeared in recent years (particularly the Scappoose and Clatskanie basins), their lineage is unclear. In the case of the Youngs Bay, Big Creek, Lower and Upper Gorge populations, the current situation where 80%+ of the natural spawners are stray hatchery fish, makes the re-establishment of a self-sustaining, locally adapted wild population unlikely in the future. Better prospects are evident for the Clatskanie and Scappoose populations where the incidence of stray hatchery fish is much lower. The net assessment of the diversity criterion for each population is represented by the diamonds in Figure 48. As described in the Introduction (Section 1) of this report, these diamonds were constructed on the basis of the most likely high, low and mode score for each criterion. The mode score (widest portion of the diamonds in Figure 48) corresponds with the DV rating for each population. High and low values (corresponding with the tops and bottoms of the diamonds in Figure 48) were subjectively determined on the basis that the confidence in the accuracy of the DV rating was low for all populations. The Youngs Bay, Big Creek, and both Gorge Tributaries population most likely fall into the high risk category for this criterion (Figure 48). The most probable classification for the remaining populations is the moderate risk category, although both the Sandy and Clackamas populations are nearly in the low risk category.

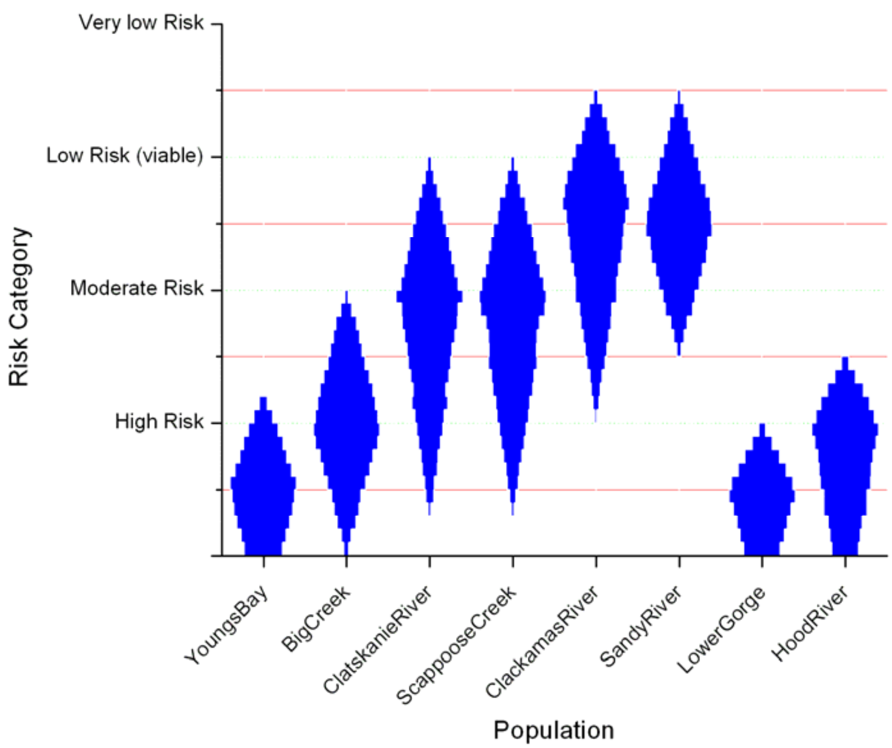


Figure 45: Lower Columbia River coho risk summary based on the evaluation of diversity only.

V. Summary of Population Results

The Clackamas is the only population in Oregon's portion of this ESU that is most likely in the viable category (Figure 46 and Figure 47). The Sandy has population is most likely in the high risk category, but the range of possible risk categories is from very high risk to viable. The remaining populations are clearly in the high or very high risk categories. Even though both the Clatskanie and Scappoose populations show encouraging signs in recent years, the risk of extinction for coho in Oregon's portion of the lower Columbia remains high.

The status of Washington populations is still under assessment; however there is no evidence that self-sustaining populations of wild coho survived the poor marine survival period of the 1990s. When the condition of coho populations on both sides of the Columbia is considered together, the picture is even bleaker. Only one population in the entire ESU—the Clackamas—is approaching viability. It is apparent that no viable populations exist in either the Coast or Gorge stratum. Although a final ESU score is not possible until the assessment of Washington coho populations is complete, we expect that the final score to place this ESU in the high risk category.

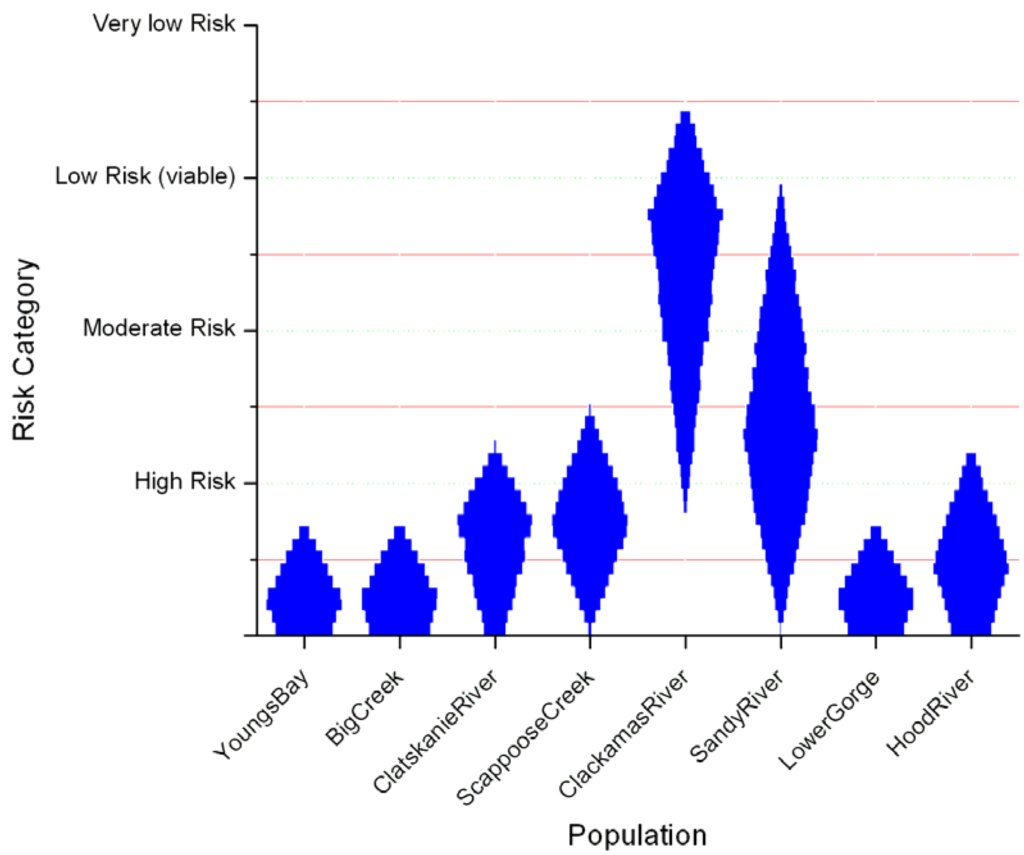


Figure 46: Oregon LCR coho population status summaries based on minimum attribute score method.

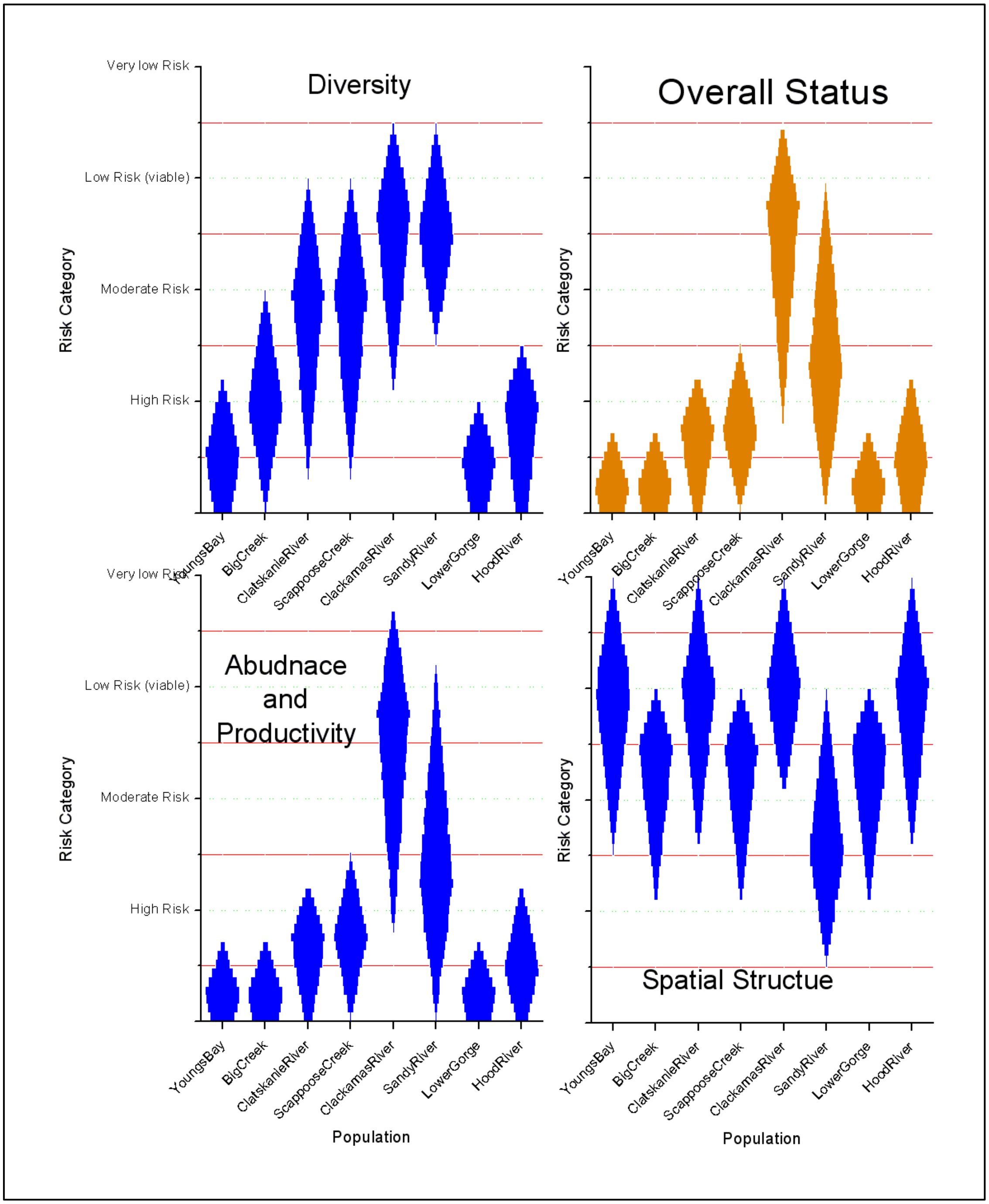


Figure 47: Oregon Lower Columbia River coho salmon status graphs and overall summary.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 5: Lower Columbia Steelhead

September 2007

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

Table of Contents

I. ESU Overview and Historical Range

Five populations of winter steelhead and one population of summer steelhead exist in Oregon's portion of the LCR ESU (Figures 1 and 2). Two populations belong to the cascade winter stratum (Clackamas and Sandy); three populations represent the winter steelhead Gorge stratum Lower Gorge, Upper Gorge, and Hood River. The two Gorge populations exist in both Oregon and Washington. In addition, the sole summer steelhead population for this ESU in Oregon occurs in Hood River (Gorge summer steelhead stratum) (Myers et al. 2006, McElhany et al. 2003).

In general, wild steelhead in the Lower Columbia basin, although depressed from historical levels, are thought to exist in most of their historical range. Unlike coho and chinook, all historical populations of steelhead are believed to be extant. However, up until recent years the presence of naturally spawning hatchery fish in most populations has been high.

The presentation of our assessment begins with three sections, each of which evaluates one of the viability criteria (i.e., abundance and productivity, spatial structure, and diversity). The methods are described in Part 1 of this report. This is then followed by a synthesis section where we pool the results from these criteria evaluations into a status rating for each population. We end our presentation with an interpretation of the population results in terms of the overall status of Oregon's LCR steelhead populations.

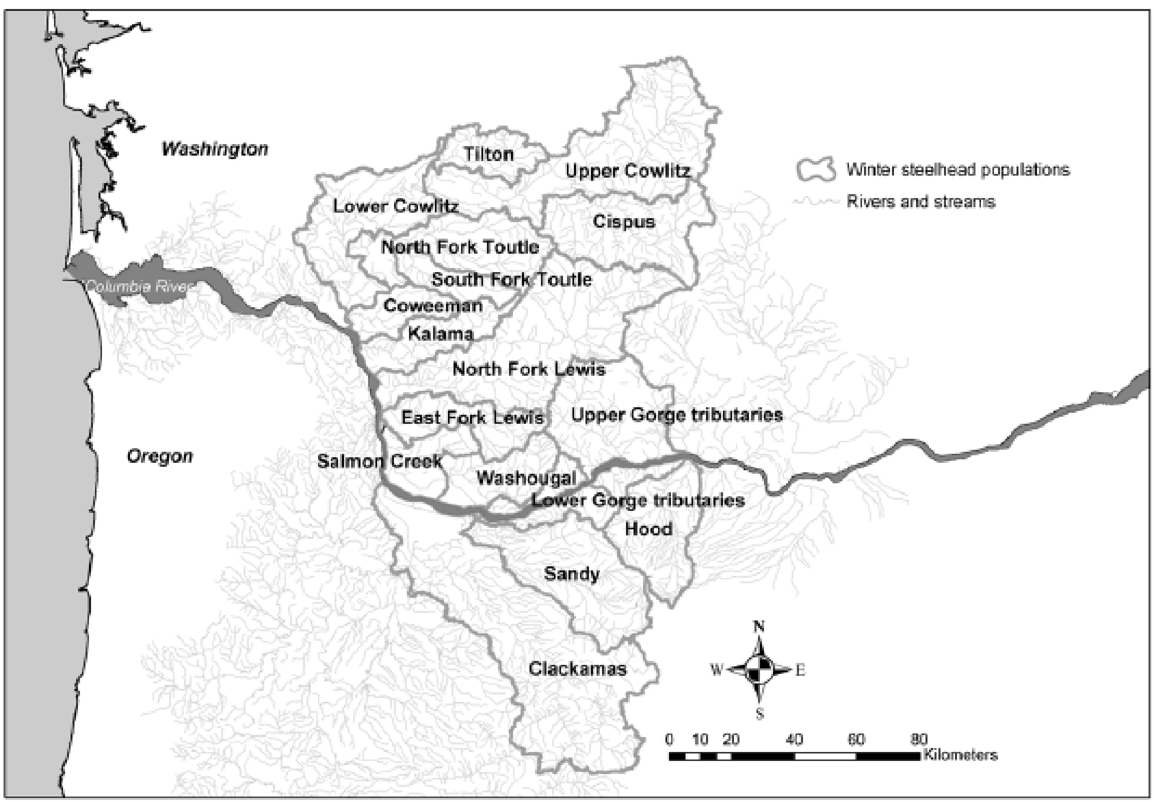


Figure 1: Map of Lower Columbia River winter steelhead populations (Myers et al. 2006).

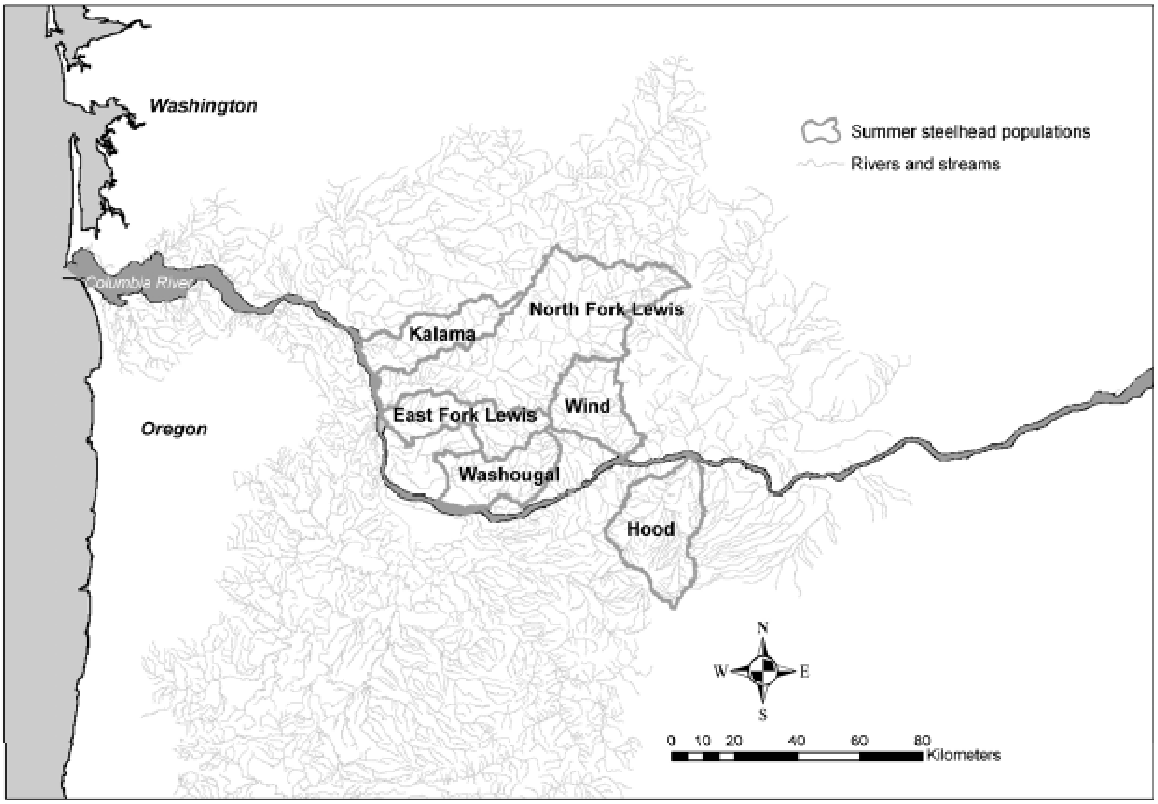


Figure 2: Map of Lower Columbia River summer steelhead populations (Myers et al. 2006).

II. Abundance and Productivity

A&P – Clackamas Winter Steelhead

A time series of abundance sufficient for quantitative analysis is available for the Clackamas River winter steelhead population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 3 to Figure 9 and in Table 1 to Table 4. The population long-term geometric mean is about 1,800 natural origin spawners, which is in the very low risk minimum abundance threshold category (Table 1). The average recent hatchery fraction is estimated at about 25%, making it difficult to obtain a precise estimate of population productivity for wild fish only. The pre-harvest viability curve analysis, the CAPM modeling, and PopCycle all suggest that the population is currently at low risk (viable) or at very low risk. The escapement viability curve suggests that a population experiencing the pattern of harvest that occurred over the available time series, when the average fishery mortality rate averaged 42%, would most likely be in the high to moderate risk category. The Oregon Native Fish Status report (ODFW 2005) listed the Clackamas River winter steelhead population as a “pass” for abundance and a “pass” for productivity.

Although the quantitative analysis of recent time series suggests that this population may be viable, the future impacts of human population growth and climate change add a degree of uncertainty to this result. Therefore, we conclude that the population is most likely in the low risk (viable) category, but with the possibility of being in either the very low risk or the high risk categories.

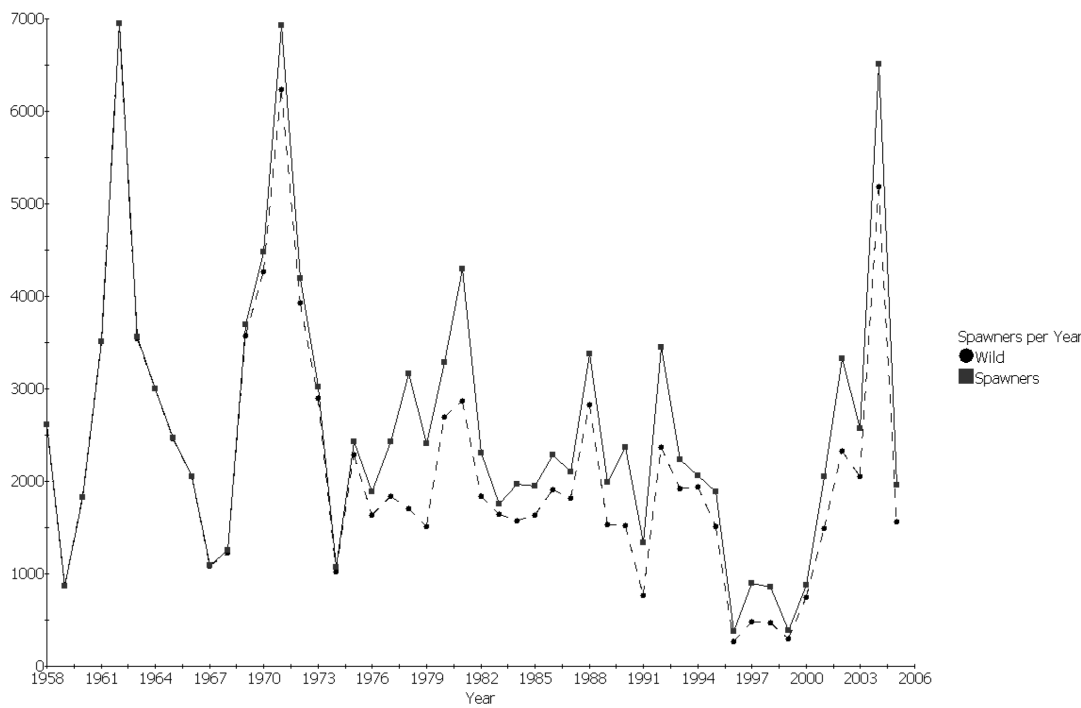


Figure 3: Clackamas River winter steelhead abundance.

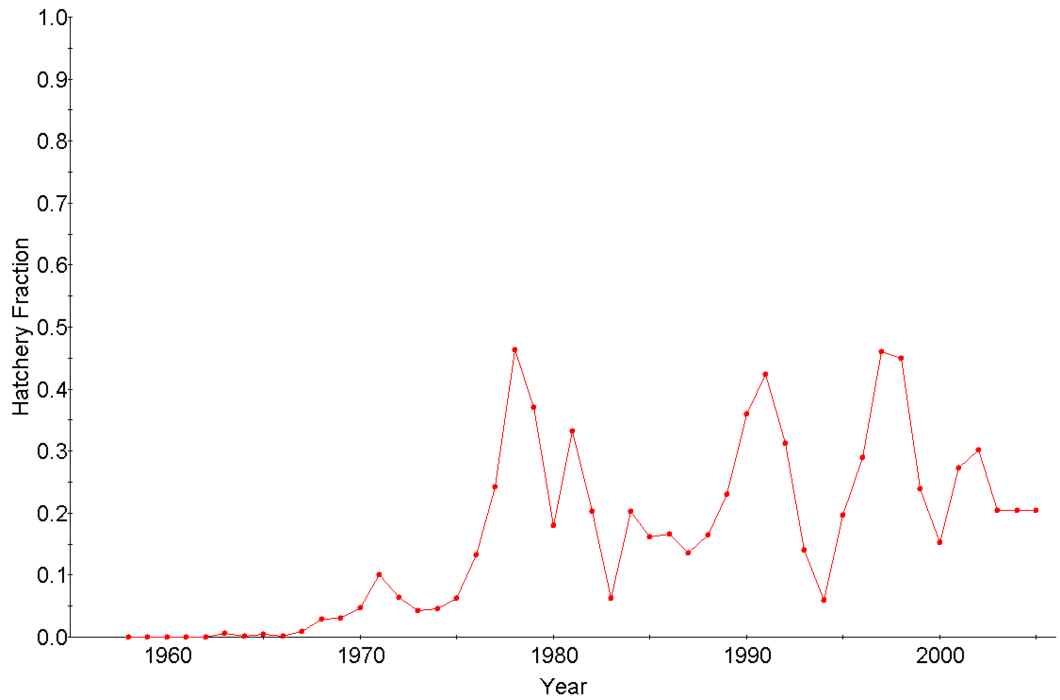


Figure 4: Clackamas River winter steelhead hatchery fraction.

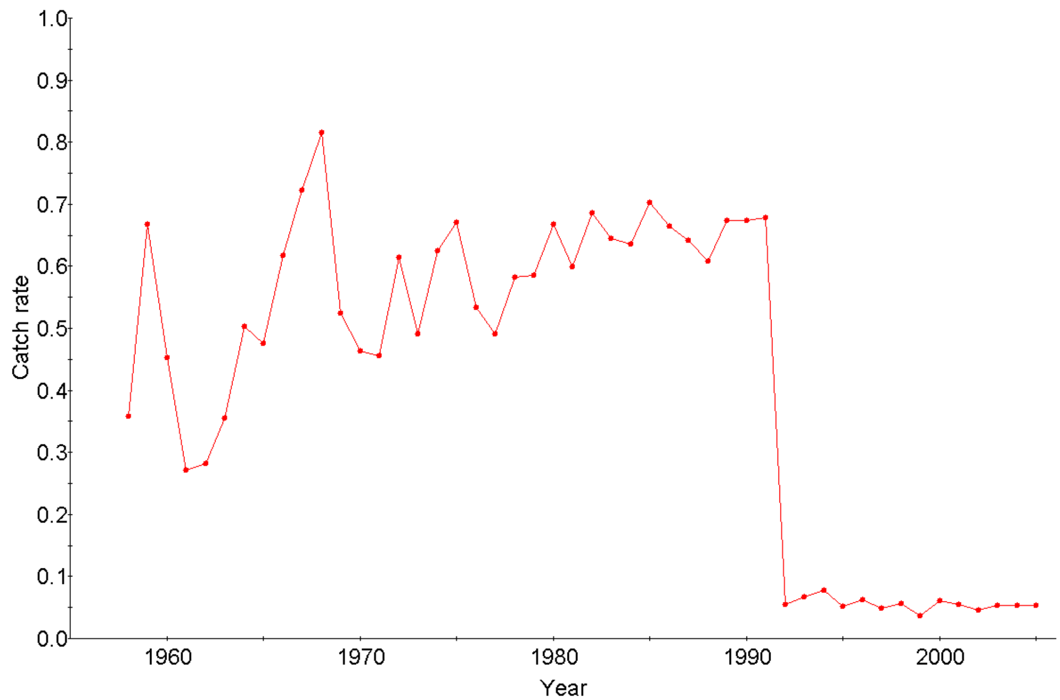


Figure 5: Clackamas River winter steelhead harvest rate.

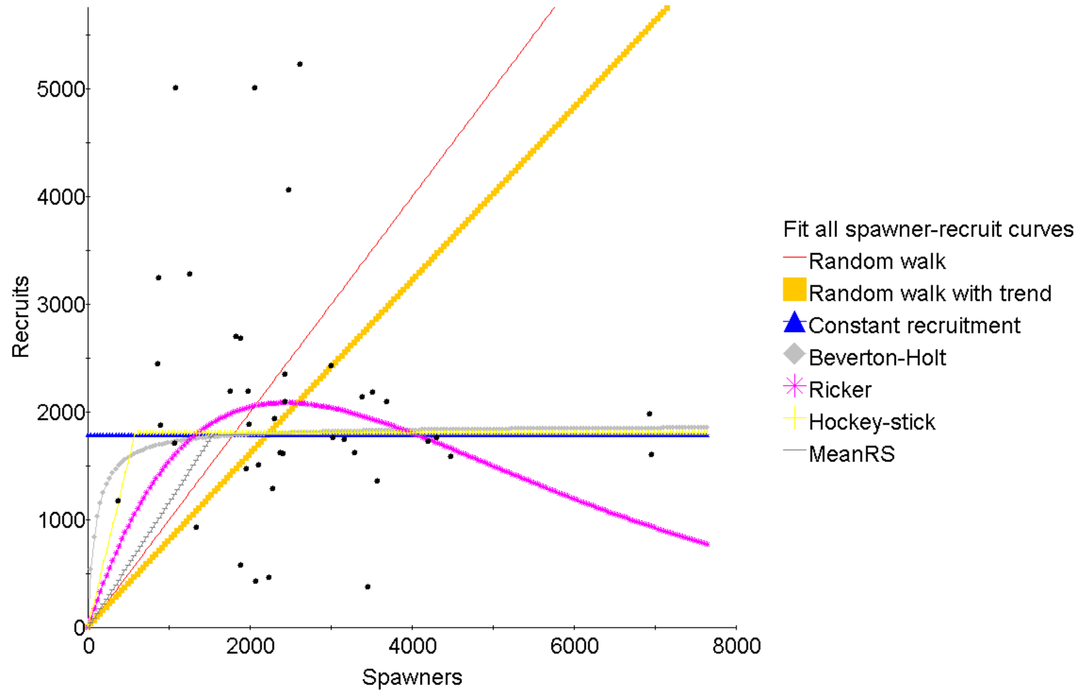


Figure 6: Clackamas River winter steelhead escapement recruitment functions.

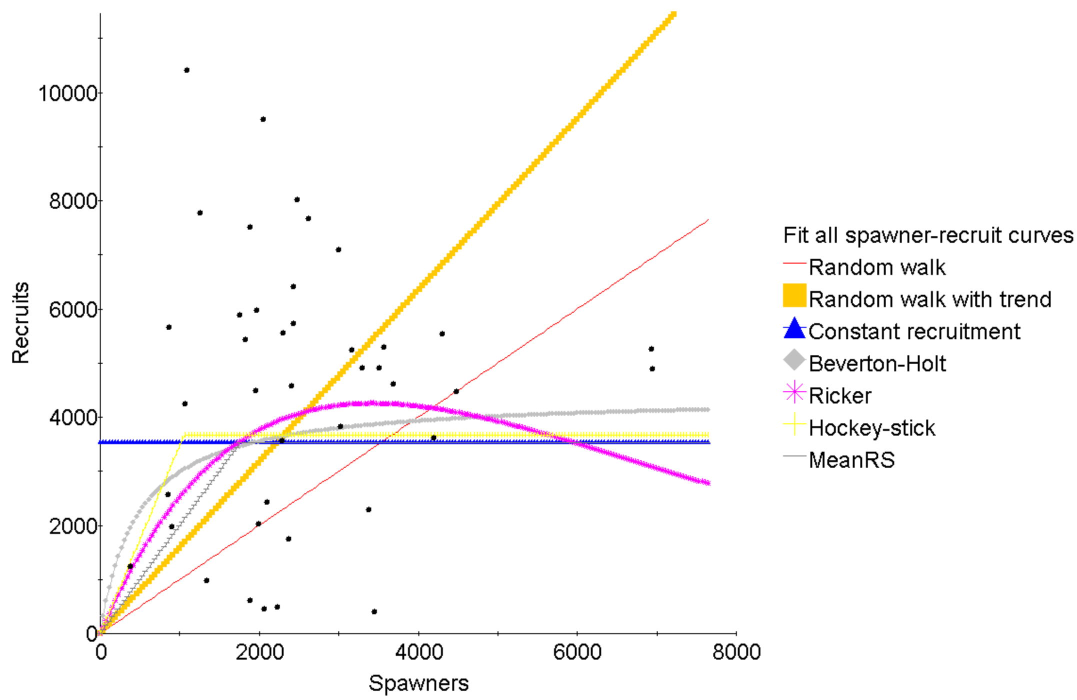


Figure 7: Clackamas River winter steelhead pre-harvest recruitment functions.

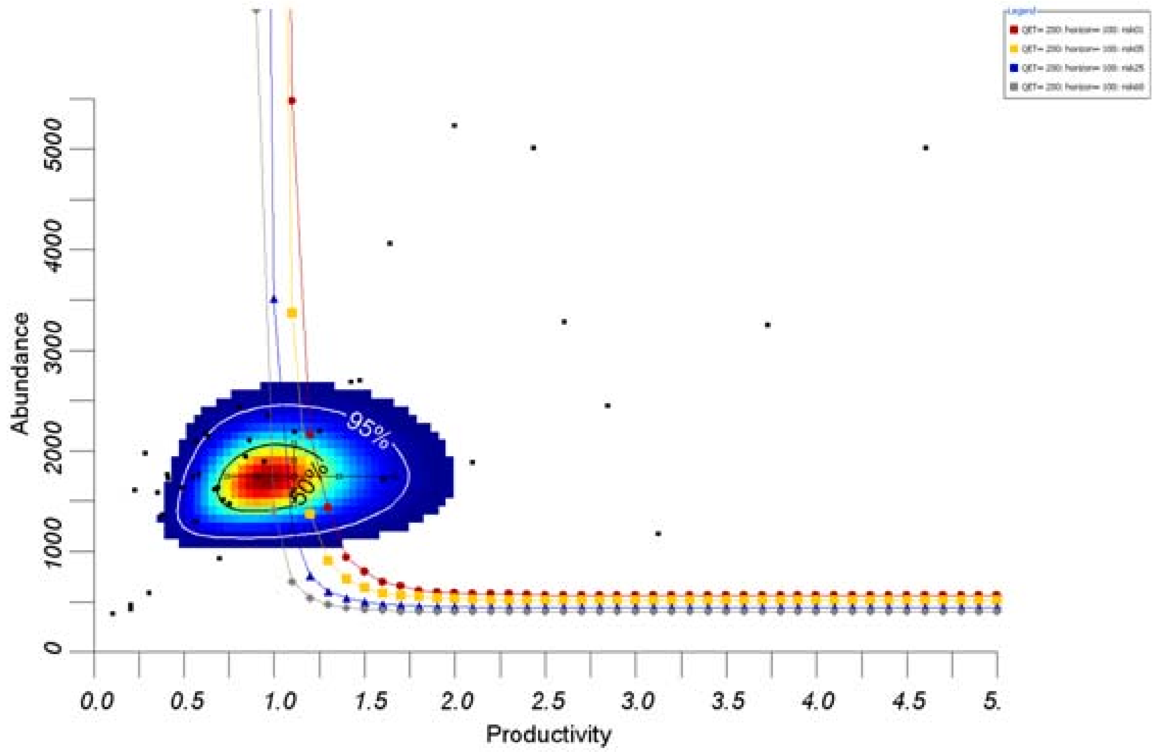


Figure 8: Clackamas River winter steelhead escapement viability curves.

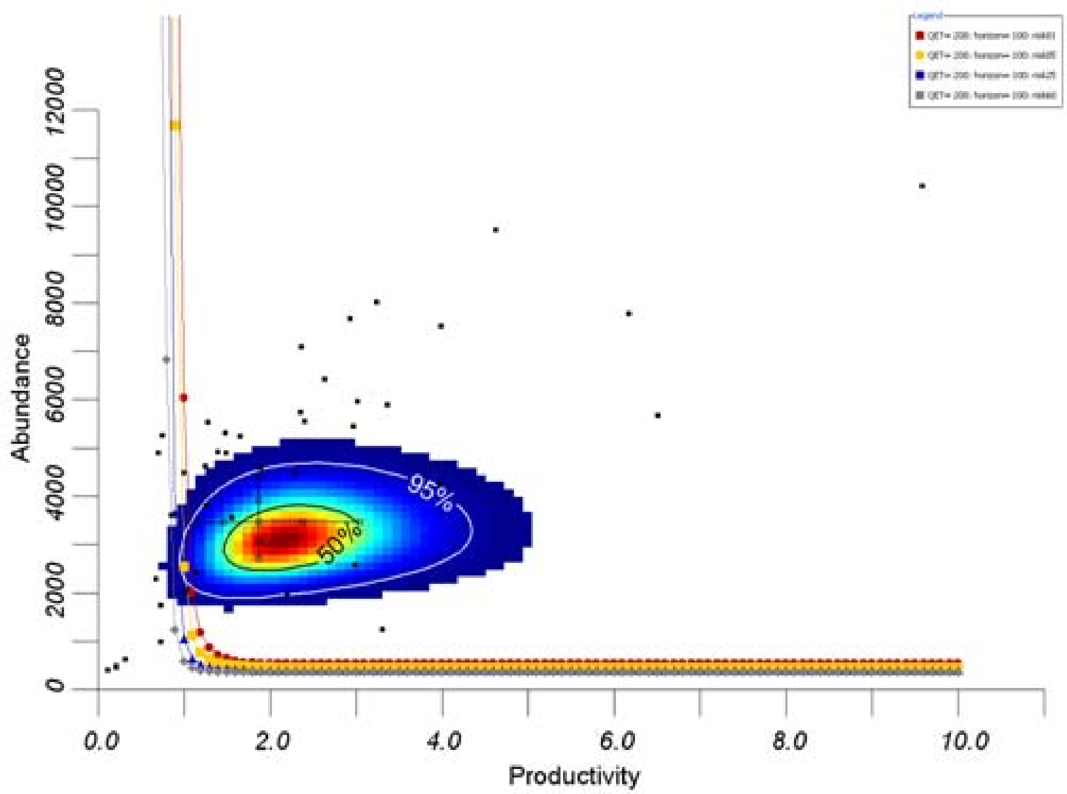


Figure 9: Clackamas River winter steelhead pre-harvest viability curves.

Table 1: Clackamas River winter steelhead summary statistics. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1958-2005	1990-2005	1958-2005	1990-2005
Length of Time Series	48	16	48	16
Geometric Mean Natural Origin Spawner Abundance	1793 (1469-2189)	1168 (750-1818)	1793 (1469-2189)	1168 (750-1818)
Geometric Mean Recruit Abundance	1793 (1488-2160)	892 (521-1525)	3536 (2711-4614)	943 (551-1613)
Lambda	0.964 (0.851-1.091)	0.976 (0.432-2.205)	1.101 (0.953-1.272)	0.96 (0.413-2.228)
Trend in Log Abundance	0.98 (0.967-0.993)	1.03 (0.934-1.137)	0.98 (0.967-0.993)	1.03 (0.934-1.137)
Geometric Mean Recruits per Spawner (all broods)	0.804 (0.613-1.054)	0.617 (0.238-1.603)	1.585 (1.177-2.134)	0.652 (0.251-1.695)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.177 (0.769-1.801)	1.321 (0.378-4.618)	1.985 (1.19-3.312)	1.393 (0.399-4.869)
Average Hatchery Fraction	0.162	0.267	0.162	0.267
Average Harvest Rate	0.421	0.133	0.421	0.133
CAPM median extinction risk probability (5th and 95th percentiles in parentheses)	NA	NA	0.000 (0.000-0.030)	NA
PopCycle extinction risk	NA	NA	0.02	NA

Table 2: Escapement recruitment parameter estimates and relative AIC values for Clackamas winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.88 (0.75-1.09)	31.4
Random walk with trend	0.8 (0.65-1.03)	NA	0.85 (0.74-1.07)	30.7
Constant recruitment	NA	1793 (1543-2114)	0.58 (0.5-0.74)	0
Beverton-Holt	>20 (5.17->20)	1880 (1651-2407)	0.59 (0.51-0.74)	2.4
Ricker	2.32 (1.62-3.47)	2084 (1816-2608)	0.63 (0.55-0.81)	8.4
Hockey-stick	3.1 (2.61->20)	1810 (1544-2133)	0.58 (0.51-0.74)	1.5
MeanRS	1.16 (0.84-1.57)	1793 (1539-2075)	0.44 (0.25-0.62)	31.1

Table 3: Pre-harvest recruitment parameter estimates and relative AIC values for Clackamas River winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted.

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.04 (0.89-1.3)	16.2
Random walk with trend	1.58 (1.27-2.09)	NA	0.93 (0.81-1.18)	9.2
Constant recruitment	NA	3539 (2867-4533)	0.83 (0.72-1.05)	0
Beverton-Holt	9.27 (3.2-18.86)	4403 (3387-8163)	0.82 (0.71-1.04)	0.8
Ricker	3.38 (1.83-5.44)	4254 (3598-12145)	0.84 (0.74-1.09)	2.8
Hockey-stick	3.48 (2.96-18.83)	3676 (2912-4657)	0.81 (0.72-1.05)	0.2
MeanRS	2 (1.37-2.87)	3536 (2830-4337)	0.74 (0.38-1.07)	6.6

Table 4: Clackamas River winter steelhead CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.558	0.999	1.000
Probability the population is above "Moderate risk of extinction" category	0.431	0.998	0.993
Probability the population is above "Viable" category	0.295	0.995	0.617
Probability the population is above "Very low risk of extinction" category	0.220	0.994	0.363

A&P – Sandy Winter

A time series of abundance sufficient for quantitative analysis is available for the Sandy winter steelhead population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 10 to Figure 16 and in Table 5 to Table 8. The population long-term geometric mean is about 850 natural origin spawners, which is in the viable minimum abundance threshold category (Table 5). However, the population shows very low productivity.

The pre-harvest viability curve analysis, PopCycle, and the CAPM modeling all suggest that the population is currently at very high risk, falling into the "extirpated or nearly so" category. The escapement viability curve suggests that if the population continued experiencing the pattern of harvest that occurred over the available time series (average fisheries mortality rate = 0.39), it would most likely be in the extirpated or nearly so risk category. Over much of the time series, the population has had a relatively high fraction of hatchery origin spawners, making estimation of the true productivity problematic (Figure 11). The Oregon Native Fish Status report (ODFW 2005) listed the Sandy winter steelhead population as a "pass" for abundance and a "fail" for productivity.

Considering the available information, we estimate the population most likely in the high risk category or nearly extirpated.

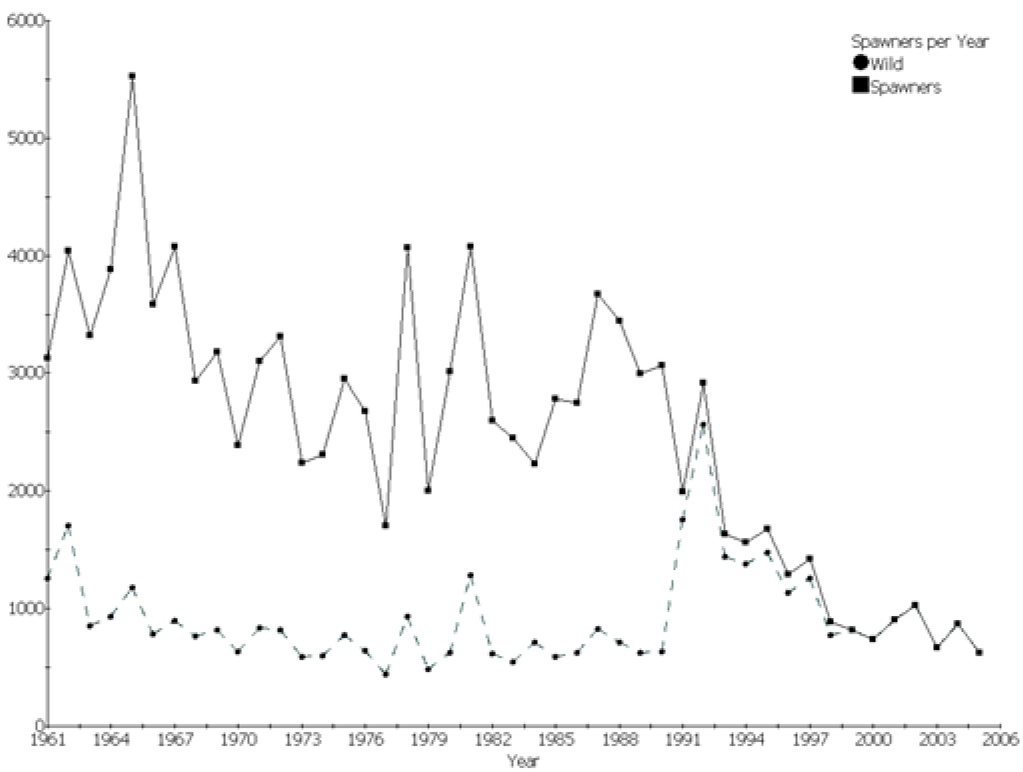


Figure 10: Sandy River winter steelhead abundance.

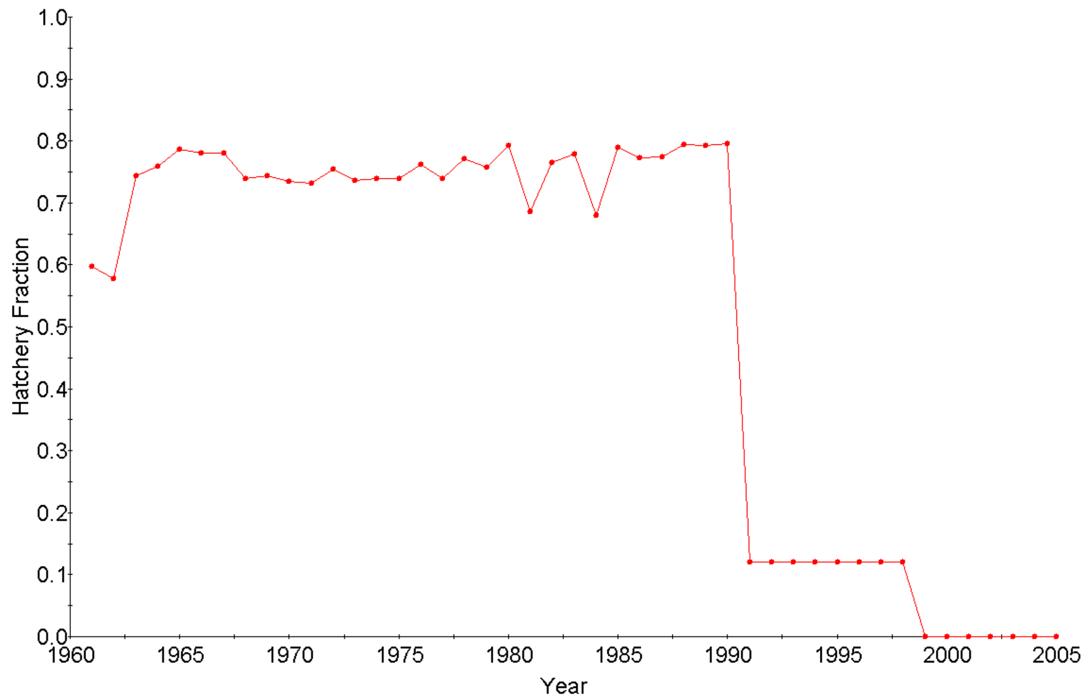


Figure 11: Sandy River winter steelhead hatchery fraction.

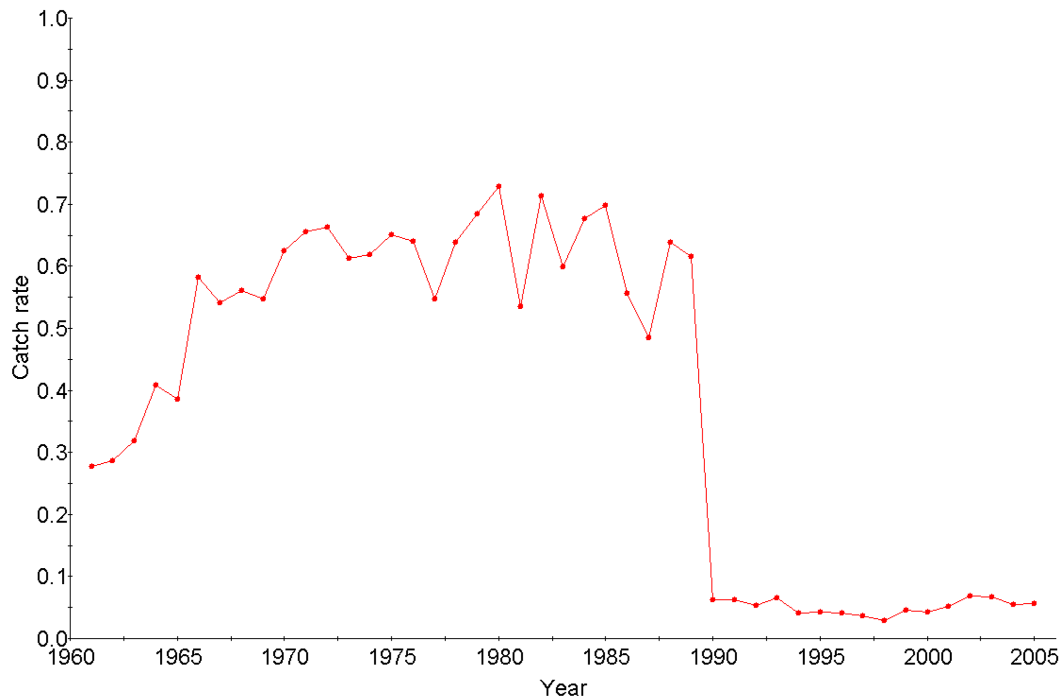


Figure 12: Sandy River winter steelhead harvest rate.

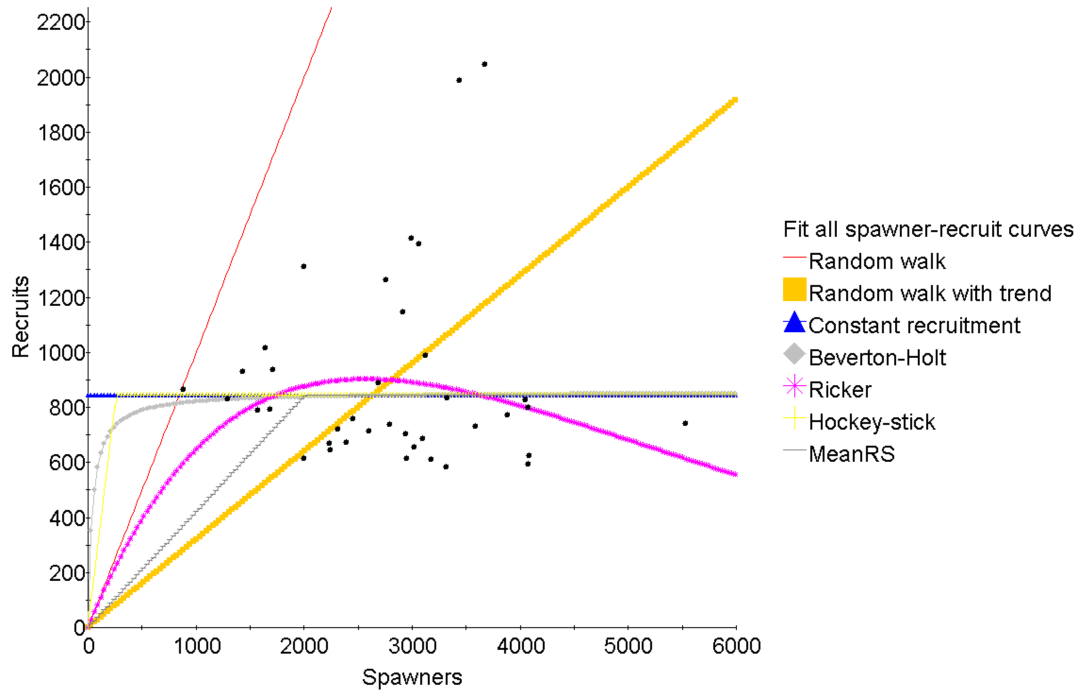


Figure 13: Sandy River winter steelhead escapement recruitment functions.

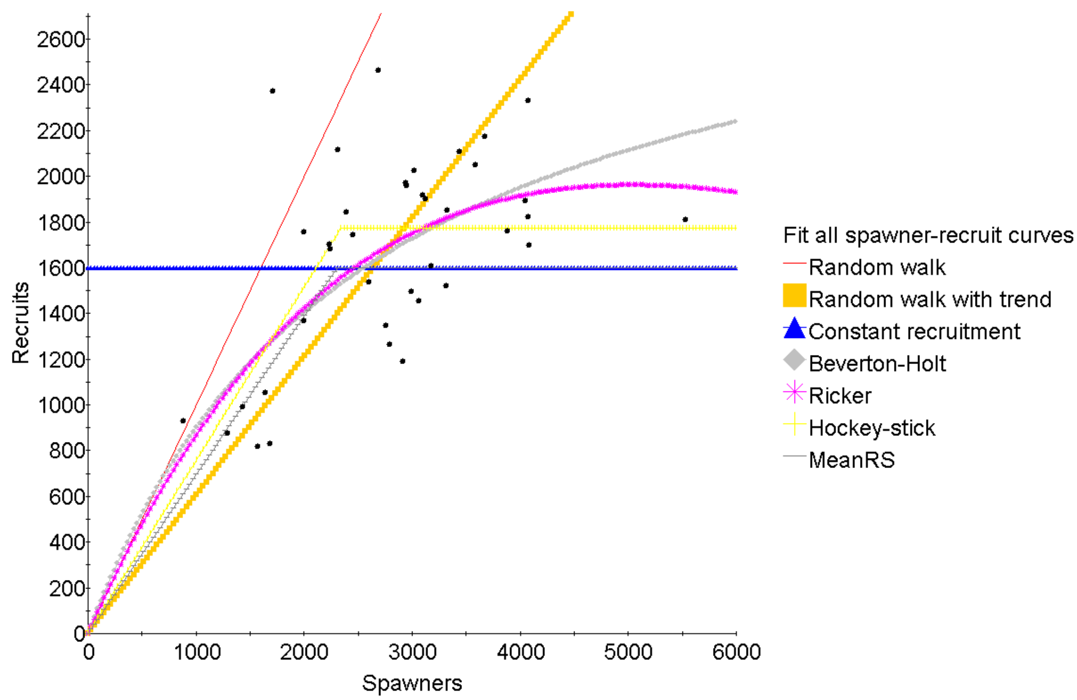


Figure 14: Sandy River winter steelhead pre-harvest recruitment functions.

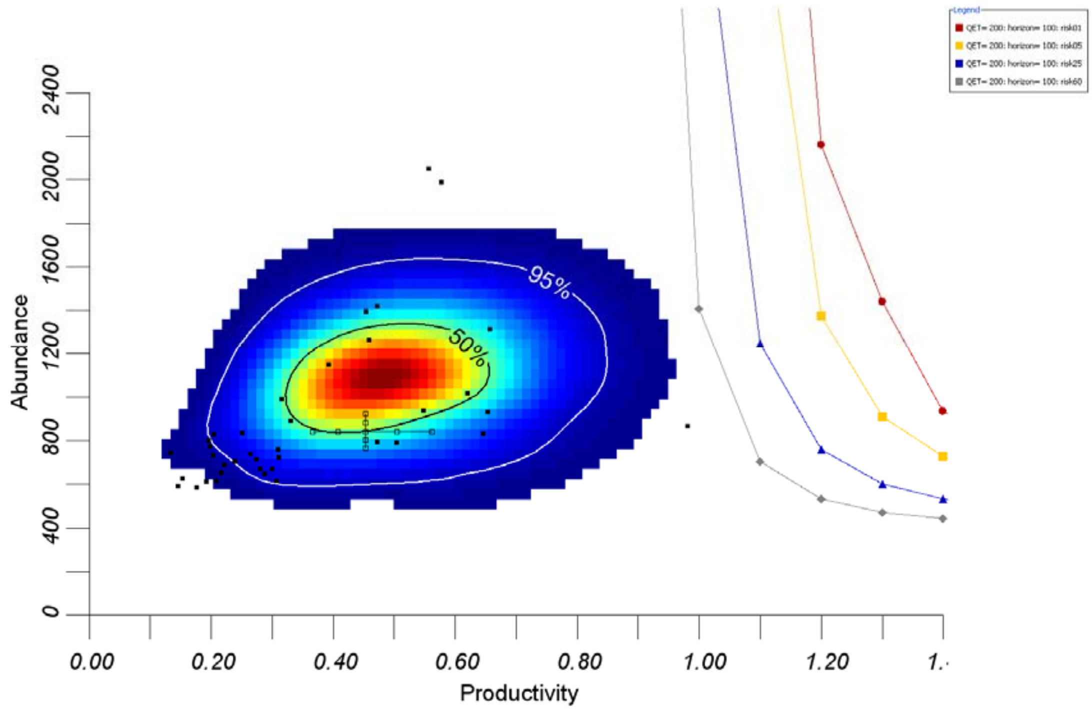


Figure 15: Sandy River winter steelhead escapement viability curve.

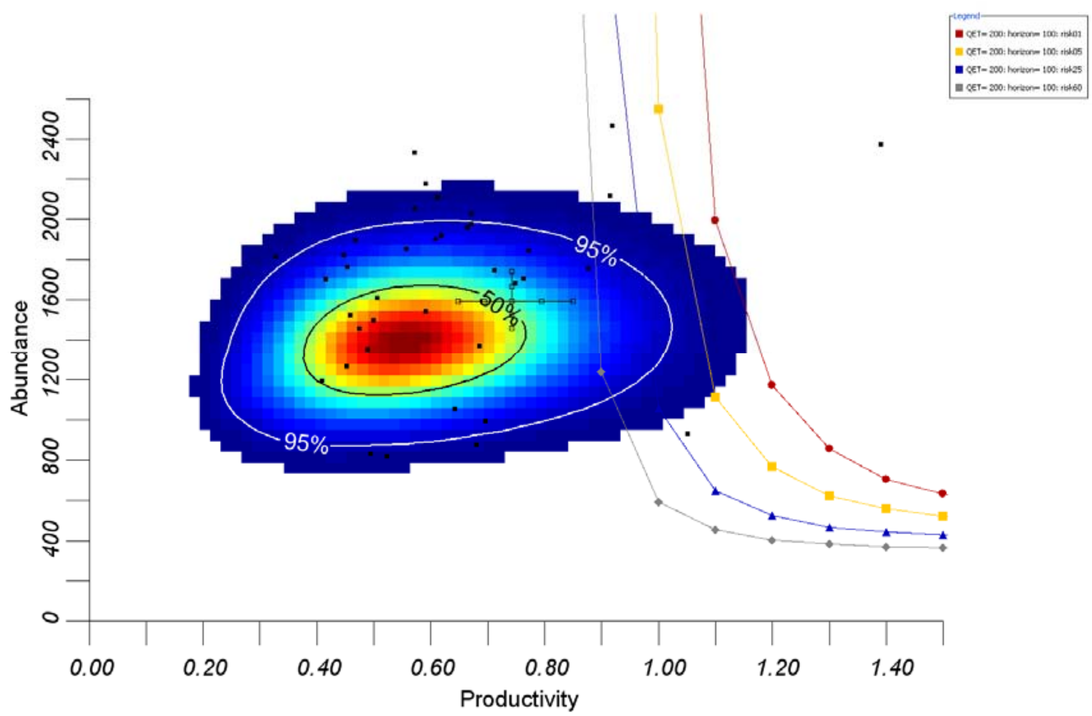


Figure 16: Sandy River winter steelhead pre-harvest viability curve.

Table 5: Sandy River winter steelhead summary statistics. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1961-2005	1990-2005	1961-2005	1990-2005
Length of Time Series	45	16	45	16
Geometric Mean Natural Origin Spawner Abundance	849 (759-949)	1040 (838-1290)	849 (759-949)	1040 (838-1290)
Geometric Mean Recruit Abundance	845 (762-937)	988 (838-1165)	1600 (1451-1765)	1036 (881-1218)
Lambda	0.798 (0.72-0.884)	0.923 (0.794-1.072)	0.906 (0.873-0.941)	0.933 (0.793-1.097)
Trend in Log Abundance	1.002 (0.994-1.011)	0.95 (0.914-0.987)	1.002 (0.994-1.011)	0.95 (0.914-0.987)
Geometric Mean Recruits per Spawner (all broods)	0.32 (0.272-0.376)	0.578 (0.469 -0.713)	0.606 (0.551-0.666)	0.606 (0.488-0.752)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	0.439 (0.349-0.553)	0.676 (0.547-0.836)	0.744 (0.643-0.861)	0.715 (0.573-0.892)
Average Hatchery Fraction	0.519	0.110	0.519	0.110
Average Harvest Rate	0.385	0.051	0.385	0.051
CAPM median extinction risk probability (5th and 95th percentiles in parentheses)	NA	NA	0.910 (0.345-1.000)	NA
PopCycle extinction risk	NA	NA	0.97	NA

Table 6: Escapement recruitment parameter estimates and relative AIC values for Sandy River winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.24 (1.06-1.56)	103
Random walk with trend	0.32 (0.28-0.37)	NA	0.49 (0.42-0.63)	34.3
Constant recruitment	NA	845 (776-924)	0.31 (0.27-0.4)	0
Beverton-Holt	>20 (2.96->20)	858 (801-993)	0.31 (0.27-0.4)	2.1
Ricker	0.95 (0.72-1.3)	902 (837-1020)	0.32 (0.28-0.42)	4.5
Hockey-stick	3.25 (1.65-19.01)	845 (776-923)	0.31 (0.27-0.4)	2
MeanRS	0.42 (0.36-0.49)	845 (779-920)	0.13 (0.08-0.17)	189.7

Table 7: Pre-harvest recruitment parameter estimates and relative AIC values for Sandy River winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.58 (0.49-0.72)	71.9
Random walk with trend	0.61 (0.56-0.66)	NA	0.28 (0.24-0.36)	20.1
Constant recruitment	NA	1600 (1477-1743)	0.29 (0.25-0.38)	22.7
Beverton-Holt	1.26 (0.91-1.99)	3189 (2346-5013)	0.22 (0.19-0.28)	1.7
Ricker	1.06 (0.84-1.27)	1962 (1761-2618)	0.21 (0.19-0.28)	0.6
Hockey-stick	0.76 (0.68-0.86)	1772 (1657-1927)	0.21 (0.19-0.28)	0
MeanRS	0.69 (0.62-0.78)	1600 (1478-1724)	0.05 (0.03-0.07)	811.9

Table 8: Sandy River winter steelhead CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.000	0.051	0.113
Probability the population is above "Moderate risk of extinction" category	0.000	0.019	0.058
Probability the population is above "Viable" category	0.000	0.006	0.005
Probability the population is above "Very low risk of extinction" category	0.000	0.002	0.000

A&P – Lower Gorge Winter

A time series of abundance is not available for the Lower Gorge winter steelhead population. In the native fish report, ODFW treated the Lower and Upper Gorge as a single 'Gorge' population. They assumed that the single Gorge population was similar to the Hood River winter steelhead population and gave it a 'pass' for both abundance and productivity. We assume that the Lower Gorge population is most similar to the Sandy River population, only at lower abundance because there is less available habitat—although, unlike the Sandy the occurrence of naturally spawning hatchery fish has likely been much less of a factor because the nearest steelhead smolt release sites are the Sandy basin and the Hood River. However, given the lack of information and the adverse condition of the Sandy population (and to a lesser extent the Hood population), we believe the Lower Gorge winter steelhead population most likely qualifies for the moderate risk category.

A&P – Upper Gorge Winter

A time series of abundance is not available for the Upper Gorge winter steelhead population. In the native fish report, ODFW treated the lower and Upper Gorge as a single ‘Gorge’ population. They then assumed that this single Gorge population was similar to the Hood winter steelhead population and gave it a ‘pass’ for both abundance and productivity. We assume that the Upper Gorge population is most similar to the Hood winter population (see below), only at lower abundance because there is less available habitat. We therefore consider the Upper Gorge winter steelhead to be most likely in the moderate category, but with some possibility of being in either the viable or nearly extirpated categories.

A&P – Hood Winter

A short time series of abundance starting in 1992 is available for the Hood winter steelhead population based on counts at Powerdale Dam (see appendix B). Descriptive graphs and viability analysis results are provided in Figure 17 to Figure 21 and in Table 9 to Table 11. The population long-term geometric mean is about 400 natural origin spawners, which is in the moderate risk minimum abundance threshold category. The time series is too short for a viability curve, CAPM, or PopCycle analysis. The time series is also probably too short for a meaningful recruit per spawner evaluation (only 7 data points), but the graphs and statistics are presented below for consideration. Three of the recruit-per-spawner estimates are below replacement and four are above. The data contain little information on the relationship between recruits and spawners (Table 10 and Table 11). Based on this scant information, we consider the population most likely in the moderate risk category, but with considerable uncertainty. The Oregon Native Fish Status report (ODFW 2005) listed the Sandy winter steelhead population as a "pass" for abundance and a "pass" for productivity.

Figure 17: Hood River winter steelhead abundance at Powerdale Dam.

Figure 18: Hood River winter steelhead hatchery fraction.

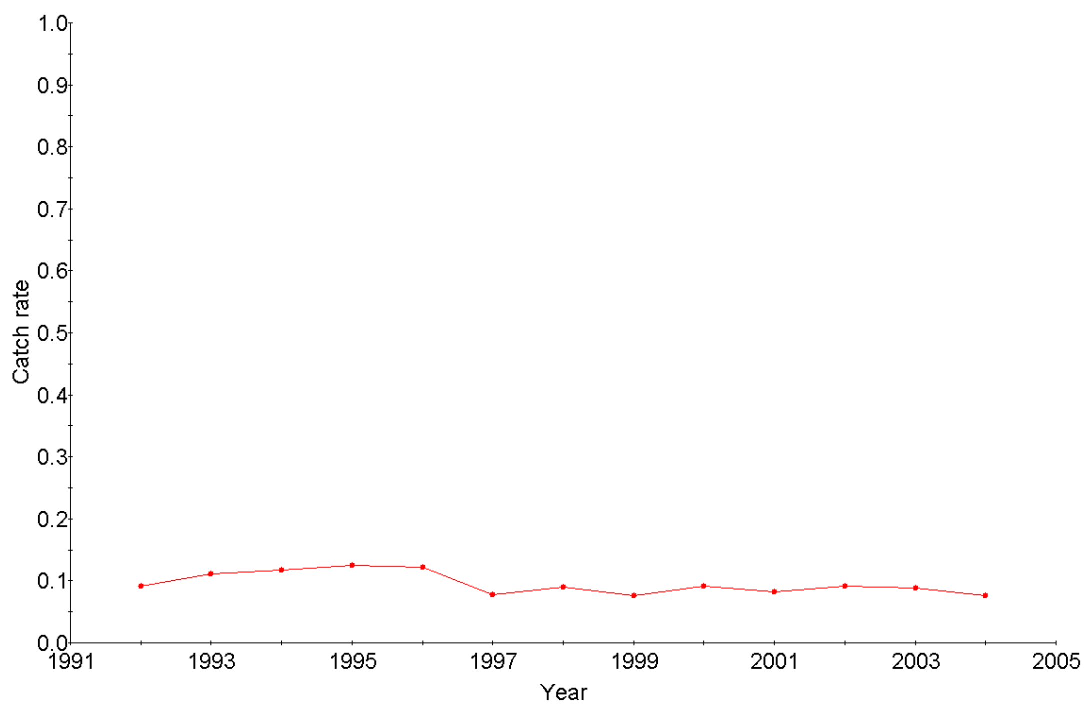


Figure 19: Hood River winter steelhead harvest rate.

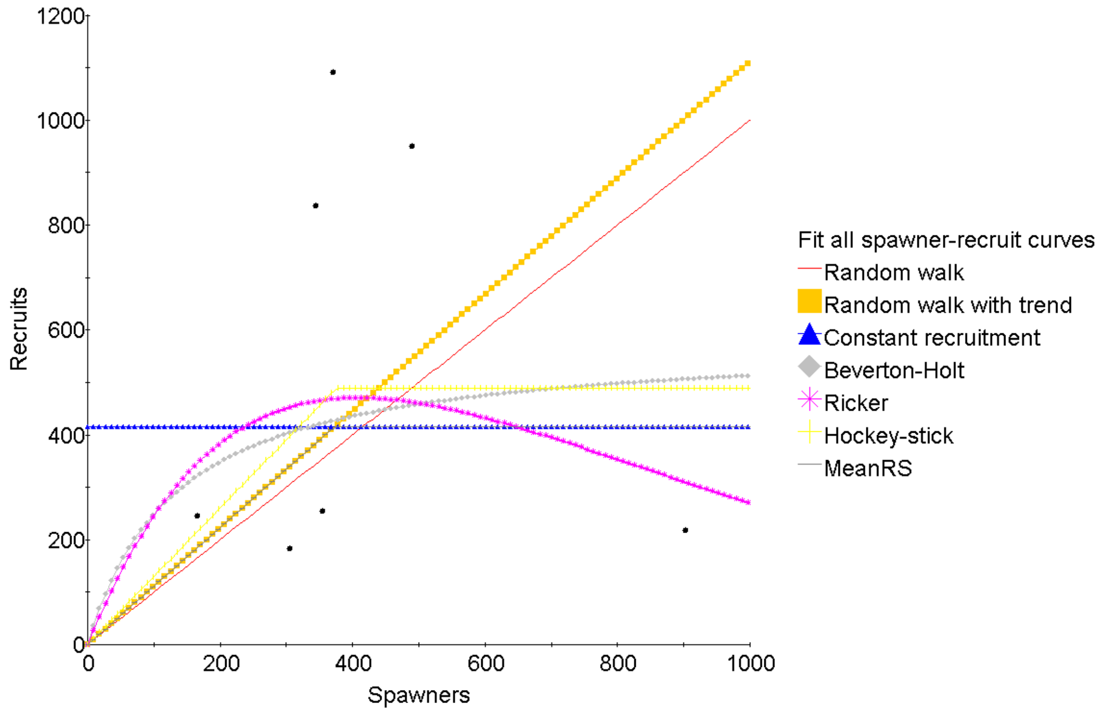


Figure 20: Hood River winter steelhead escapement recruitment functions.

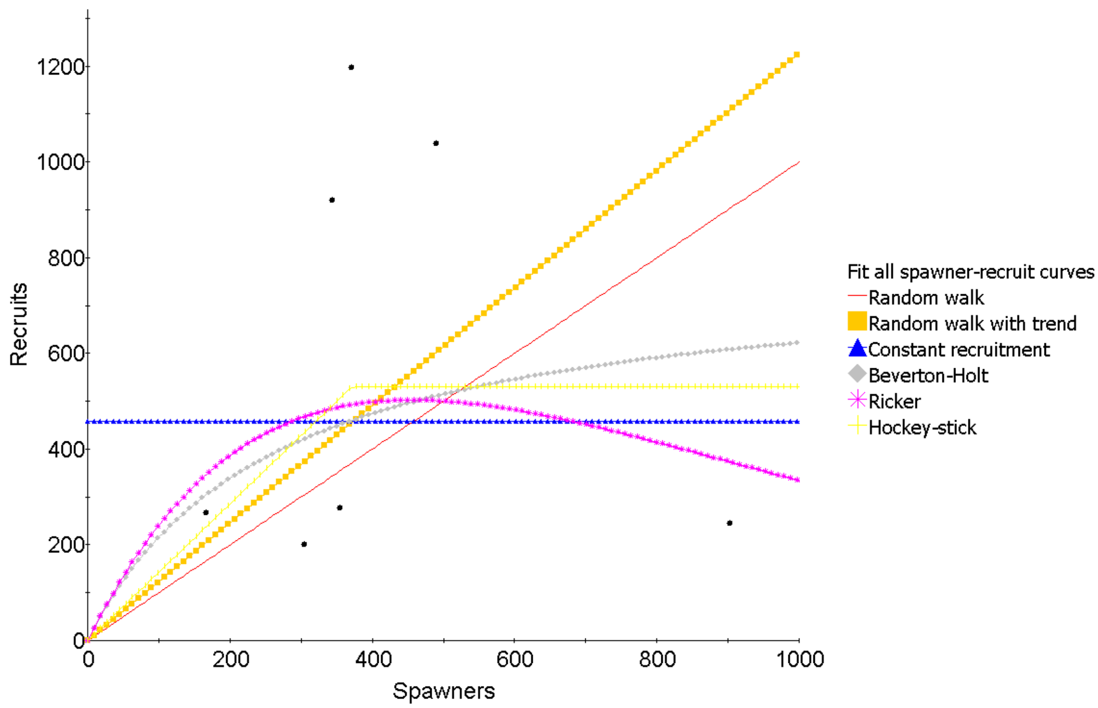


Figure 21: Hood River winter steelhead pre-harvest recruitment functions.

Table 9: Hood River winter steelhead summary statistics.

Statistic	Total Series	Total Series
Time Series Period	1992-2004	1992-2004
Length of Time Series	13	13
Geometric Mean Natural Origin Spawner Abundance	395 (269-581)	NA
Geometric Mean Recruit Abundance	416 (201-861)	457 (221-945)
Lambda	0.985	1.007
Trend in Log Abundance	1.083 (0.987-1.19)	NA
Geometric Mean Recruits per Spawner (all broods)	1.115 (0.486-2.558)	1.224 (0.537-2.792)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.292 (0.671-2.487)	1.413 (0.733-2.724)
Average Hatchery Fraction	0.3228	NA
Average Harvest Rate	0.0953	NA
SPMPC extinction risk (boot strap intervals are ± 10%)	NA	NA

Table 10: Escapement recruitment parameter estimates and relative AIC values for Hood River winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.84 (0.62-1.71)	0
Random walk with trend	1.11 (0.68-3.29)	NA	0.83 (0.67-2.04)	1.9
Constant recruitment	NA	417 (264-1054)	0.73 (0.58-1.84)	0
Beverton-Holt	4.42 (0.84-16.77)	564 (379-17937)	0.71 (0.61-1.96)	1.7
Ricker	3.05 (0.75-9.97)	471 (475-18534)	0.65 (0.62-1.98)	0.5
Hockey-stick	1.33 (0.77-17.75)	502 (310-18066)	0.67 (0.61-1.96)	0.8
MeanRS	1.12 (0.69-1.79)	416 (268-651)	0.52 (0.16-0.64)	2.5

Table 11: Pre-harvest recruitment parameter estimates and relative AIC values for Hood River winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC < 2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.85 (0.63-1.75)	0.2
Random walk with trend	1.23 (0.75-3.87)	NA	0.83 (0.66-2.09)	1.8
Constant recruitment	NA	459 (291-1162)	0.73 (0.58-1.84)	0
Beverton-Holt	4.46 (1.09-18)	658 (383-8425)	0.71 (0.59-1.86)	1.7
Ricker	3.31 (0.88-13.43)	516 (483-9070)	0.66 (0.6-1.98)	0.6
Hockey-stick	1.39 (0.92->20)	536 (318-8435)	0.67 (0.6-1.91)	0.8
MeanRS	1.22 (0.75-1.97)	457 (295-713)	0.51 (0.16-0.63)	2.6

A&P – Hood Summer

A short time series of abundance starting in 1993 is available for the Hood River summer steelhead population based on counts at Powerdale Dam (see appendix B). Descriptive graphs and viability analysis results are provided in Figure 22 to Figure 26 and in Table 12 to Table 14. The population long-term geometric mean is about 200 natural origin spawners, which is in the nearly extirpated or high risk minimum abundance threshold category (Table 12). The time series is too short for a viability curve, CAPM, or PopCycle analysis. The time series is also probably too short for a meaningful recruit per spawner evaluation (only 7 data points), but the graphs and statistics are presented below for consideration. Six of the seven recruit per spawner estimates are below replacement, suggesting low productivity. The data contain little information on the relationship between recruits and spawners (Table 13 and Table 14). Based on this scant information, we consider the population most likely in the nearly extirpated or high risk category, but with considerable uncertainty. The Oregon Native Fish Status report (ODFW 2005) listed the Hood River summer steelhead population as a "fail" for abundance and a "fail" for productivity.

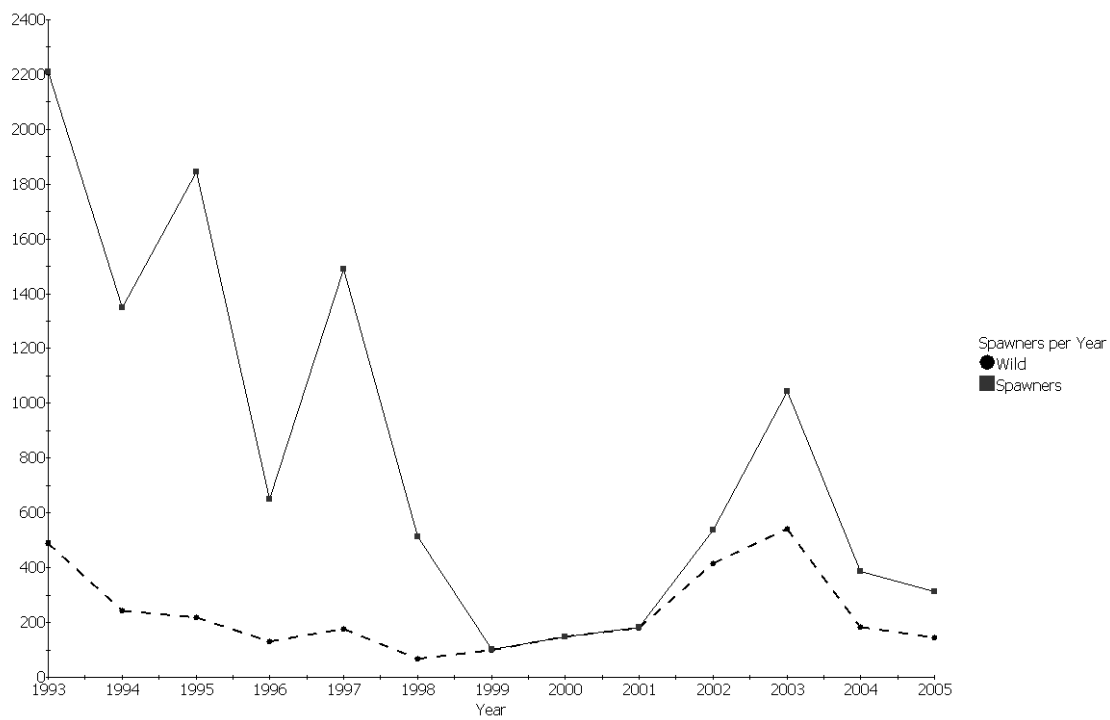


Figure 22: Hood River summer steelhead abundance.

Figure 23: Hood River summer steelhead hatchery fraction.

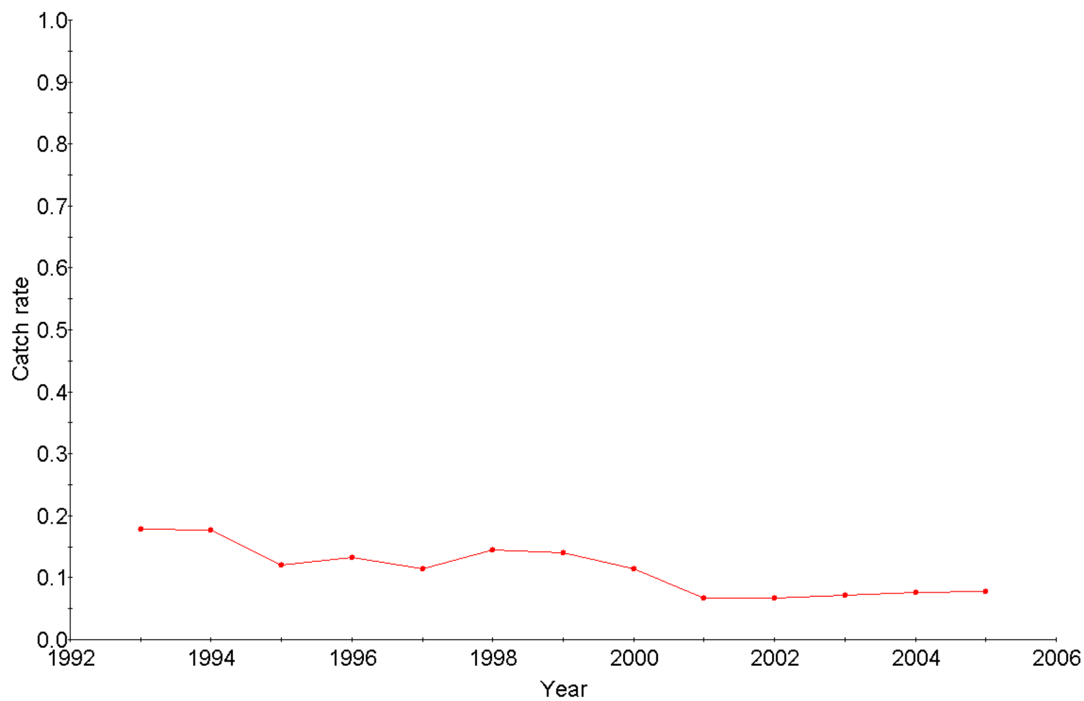


Figure 24: Hood River summer steelhead harvest rate.

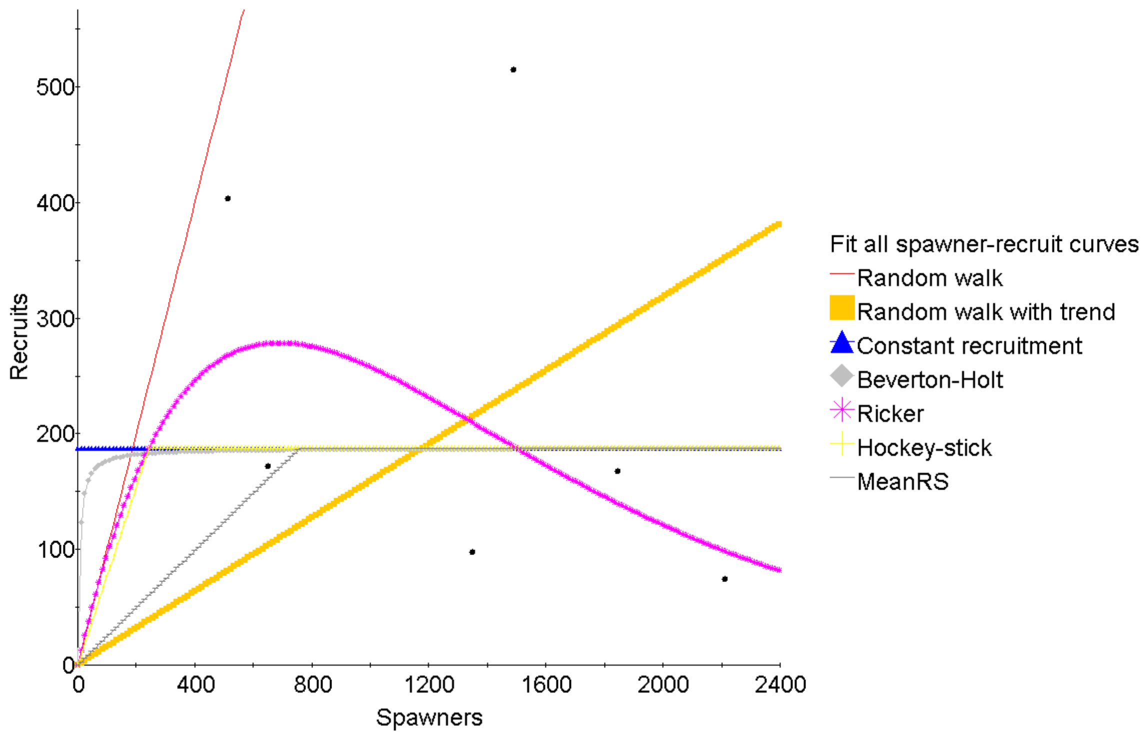


Figure 25: Hood River summer steelhead escapement recruitment functions.

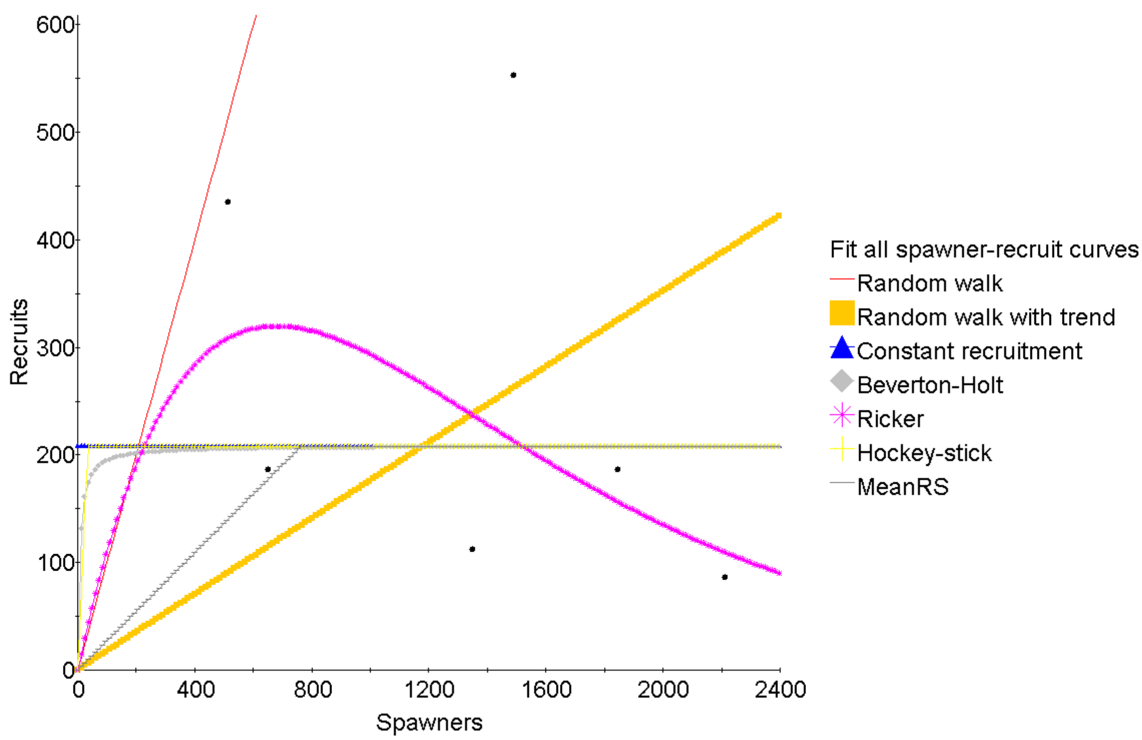


Figure 26: Hood River summer steelhead pre-harvest recruitment functions.

Table 12: Hood River summer steelhead summary statistics

Statistic	Escapement	Pre-harvest
Time Series Period	1993-2005	1993-2005
Length of Time Series	13	13
Geometric Mean Natural Origin Spawner Abundance	195 (135-283)	NA
Geometric Mean Recruit Abundance	188 (84-419)	208 (96-450)
Lambda	0.811 (0.046-14.325)	0.821 (0.049-13.745)
Trend in Log Abundance	0.995 (0.898-1.104)	0.995 (0.898-1.104)
Geometric Mean Recruits per Spawner (all broods)	NA	NA
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	NA	0.528
Average Hatchery Fraction	NA	0.114
Average Harvest Rate	NA	NA

Table 13: Escapement recruitment parameter estimates and relative AIC values for Hood River summer steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	2.13 (1.49-2.89)	11.4
Random walk with trend	0.16 (0.12-0.95)	NA	1.06 (0.86-2.62)	5.1
Constant recruitment	NA	187 (116-594)	0.7 (0.56-2.08)	0
Beverton-Holt	>30 (1.15->30)	188 (117-762)	0.7 (0.57-2.05)	2
Ricker	1.1 (0.25-23.71)	279 (249-3564)	0.59 (0.6-2.47)	0
Hockey-stick	0.77 (1.19->30)	188 (116-683)	0.7 (0.56-2.02)	2
MeanRS	0.25 (0.11-0.55)	188 (118-300)	0.72 (0.13-1.03)	3.5

Table 14: Pre-harvest recruitment parameter estimates and relative AIC values for Hood River summer steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	2.02 (1.43-2.87)	11.3
Random walk with trend	0.18 (0.12-1)	NA	1.03 (0.83-2.58)	5.2
Constant recruitment	NA	209 (130-603)	0.67 (0.53-1.96)	0
Beverton-Holt	>30 (1.1->30)	209 (133-832)	0.67 (0.54-2)	2
Ricker	1.28 (0.26-23.3)	320 (255-4475)	0.57 (0.56-2.47)	0.1
Hockey-stick	6.2 (0.87->30)	208 (131-942)	0.67 (0.54-2.03)	2
MeanRS	0.27 (0.13-0.59)	208 (134-327)	0.67 (0.12-0.95)	3.6

A&P – Criterion Summary

For the abundance and productivity criterion, the most probable risk category for all but two of these populations is high (Figure 27). The exceptions are most probable classifications of 'moderate risk' for the Hood winter-run population and 'low risk' for the Clackamas population. Although the shape of the diamonds in Figure 27 suggest there is considerable uncertainty as to the status classification of these two populations. From the perspective of this viability criterion LCR steelhead in Oregon are clearly at high risk.

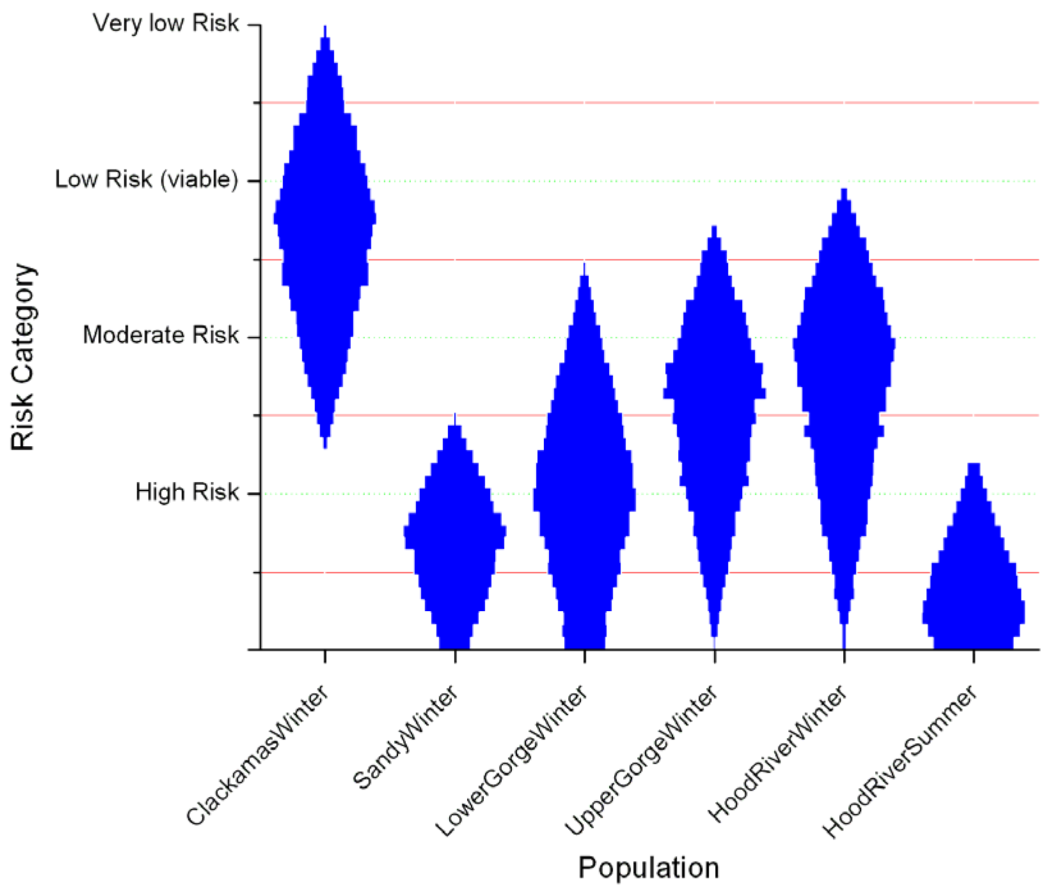


Figure 27: Lower Columbia River steelhead risk status summary based on evaluation of abundance and productivity only.

III. Spatial Structure

SS – Clackamas

Virtually the entire habitat historically accessible to winter steelhead in the Clackamas River remains accessible today (Figure 28) (ODFW 2005). Losses of accessibility are limited to higher order tributary streams, primarily due to watershed development in the lower basin. The upper Clackamas basin contains most of the historically-productive habitat for steelhead and most of that habitat is of high quality today. Spatial structure has likely been reduced by habitat degradation in lower basin tributaries. The watershed score was reduced to address a likely loss in spatial diversity related to habitat degradation in the low elevation streams. Habitat declines in the Willamette and Columbia mainstem and estuary were not factored into steelhead spatial structure scores because these habitats are much less important to the life history of lower Columbia River winter steelhead than for species.

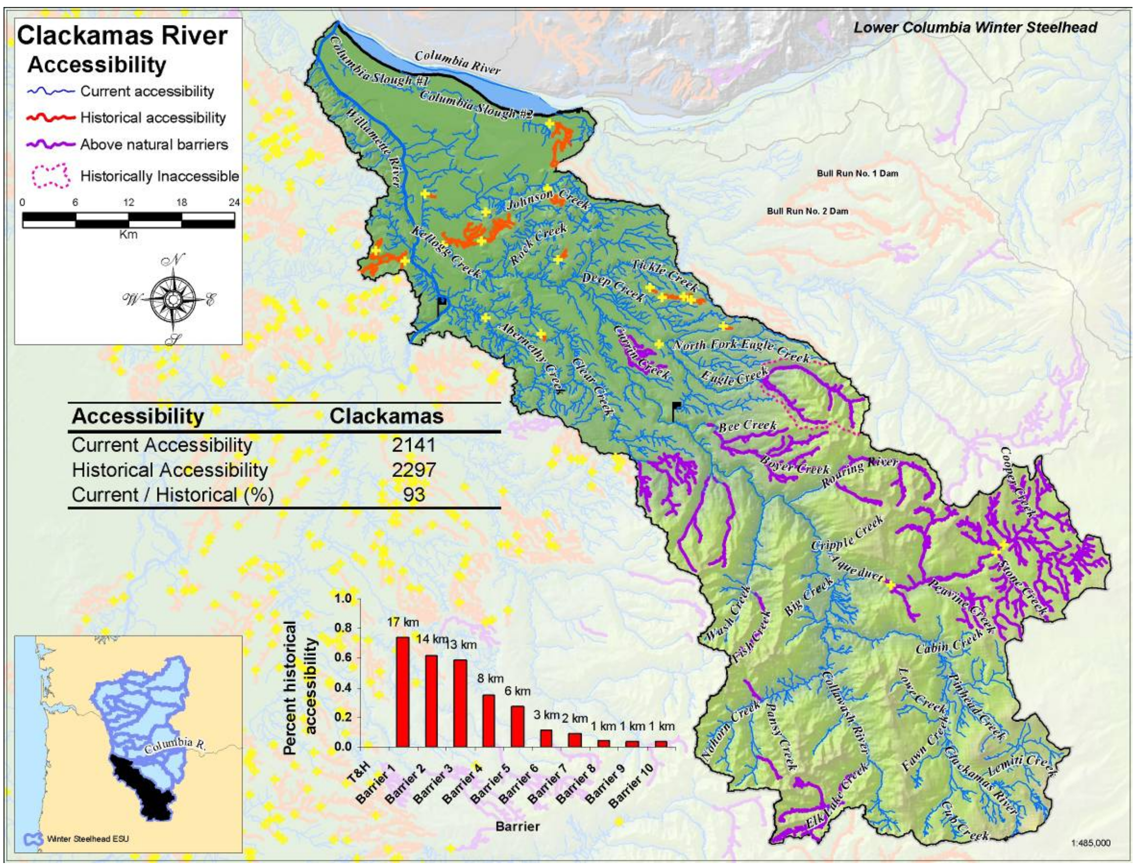


Figure 28: Clackamas River winter steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Sandy

Significant portions of the historical winter steelhead in the Sandy River have been blocked by dam construction in the Bull Run and Little Sandy watersheds (Figure 29) (ODFW 2005). Blocked areas were productive habitats for steelhead. Large areas of productive high quality habitat remain accessible to steelhead in the remainder of the basin, particularly in the forested upper basin. A distribution adjustment was warranted because the remaining habitat is largely concentrated in watersheds directly fed by Mt Hood. No further modification is warranted because of the remaining wide distribution of productive steelhead habitats. Habitat declines in the estuary were not factored into steelhead spatial structure scores.

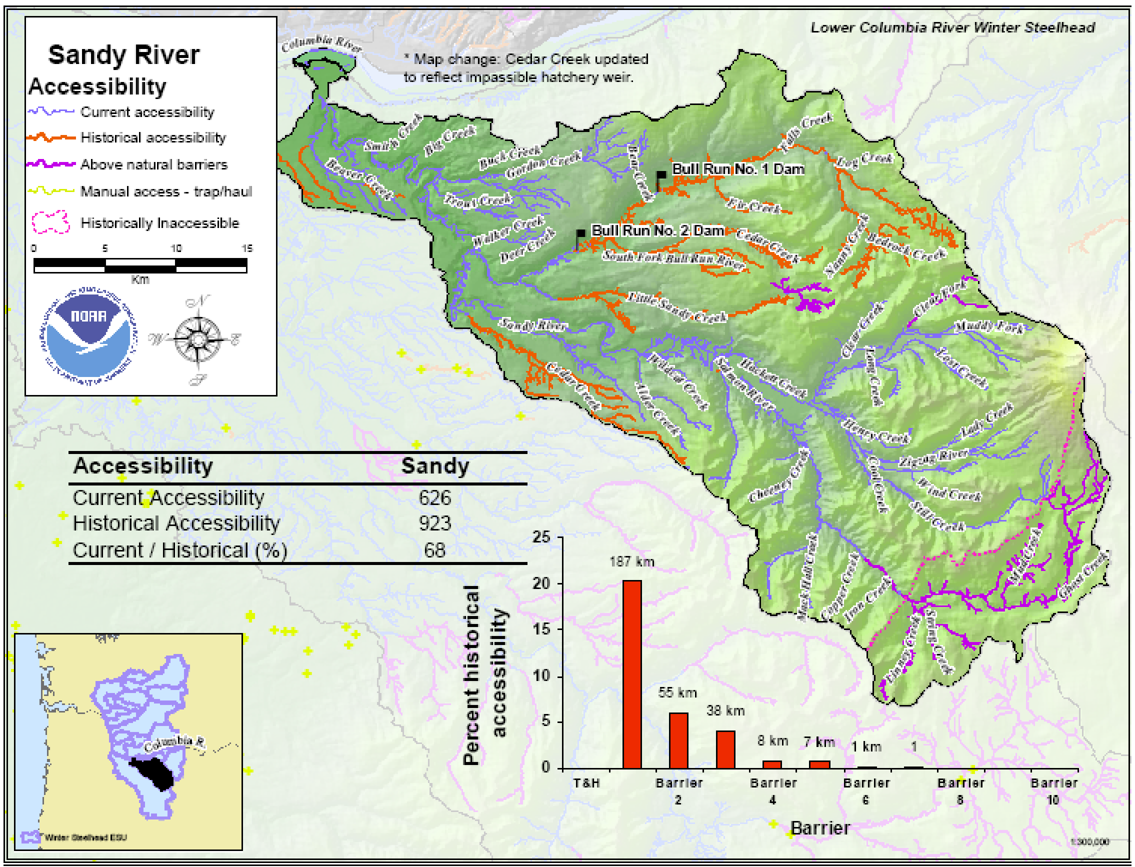


Figure 29: Sandy River winter steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Lower Gorge Tributaries

Most of the small Columbia River Gorge streams between the Sandy River and Eagle Creek remain largely accessible to steelhead (Figure 30) (ODFW 2005). Habitat availability is limited to the lower portions of these streams by topography. A hatchery weir blocks small sections of Tanner and Eagle Creek but this is a significant percentage of the historical habitat in this small Lower Gorge watershed. A further modification is warranted by habitat alterations and development which has likely reduced local habitat quality in some streams.

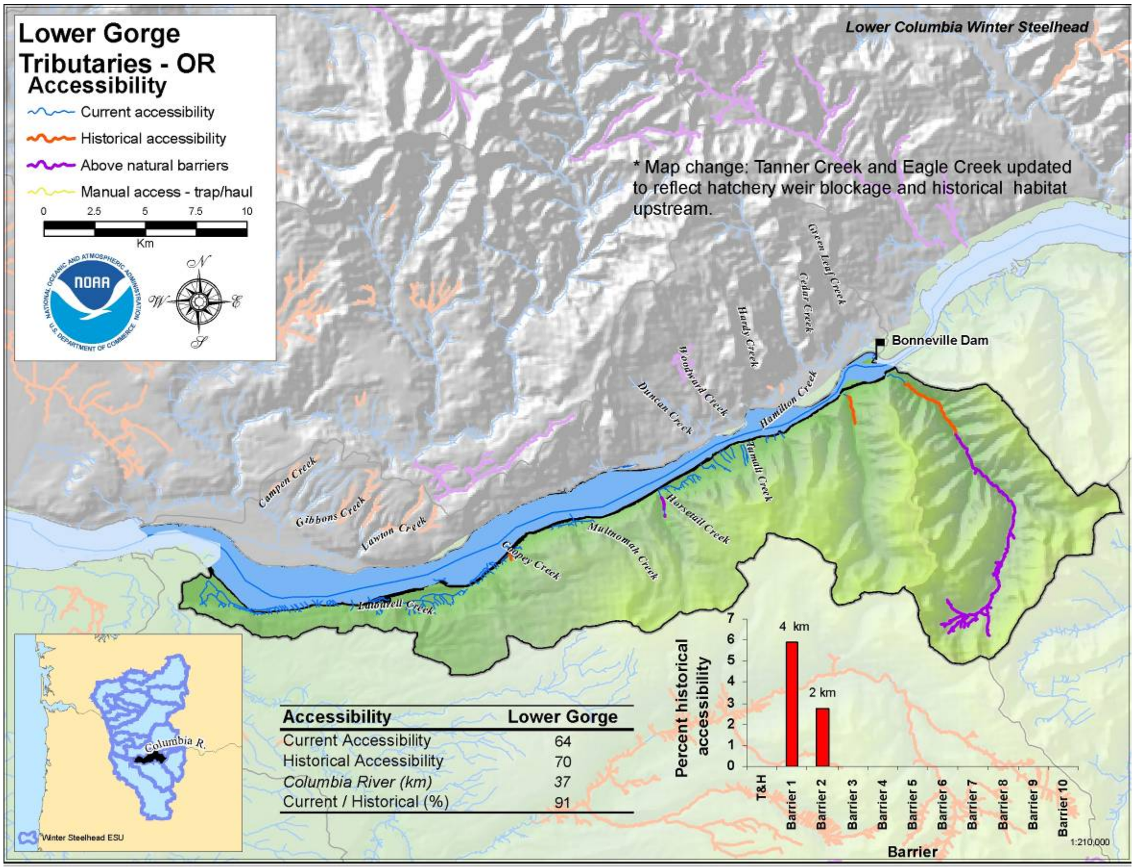
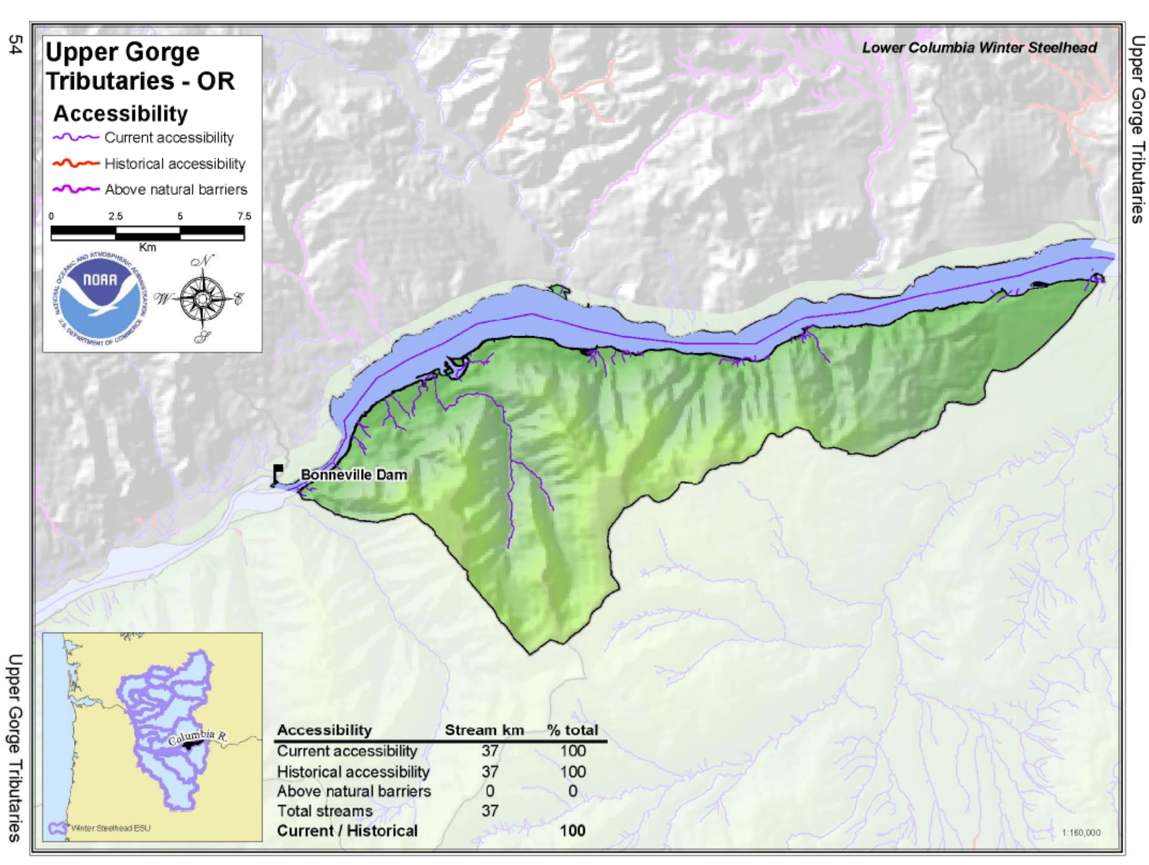


Figure 30: Lower Gorge winter steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Upper Gorge Tributaries

The small Columbia River Gorge streams upstream from Eagle Creek remain largely accessible to steelhead (Figure 31). The amount of habitat is limited to the lower portions of these streams by topography and portions of the lower reaches have been inundated by the Bonneville Dam reservoir. Other local habitat alterations and development have likely reduced habitat quality in some streams.



Accessibility	Stream km	% total
Current accessibility	37	100
Historical accessibility	37	100
Above natural barriers	0	0
Total streams	37	
Current / Historical		**100**

Figure 31: Upper Gorge winter steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Hood River

Virtually the entire habitat historically accessible to winter steelhead in the Hood River remains accessible today (Figure 32) (ODFW 2005). Blockages are limited to only a few headwater reaches and these streams do not represent significant historical steelhead production areas. Declines in habitat quality in lower elevations streams of the basin have likely reduced the spatial structure of steelhead production in the basin.

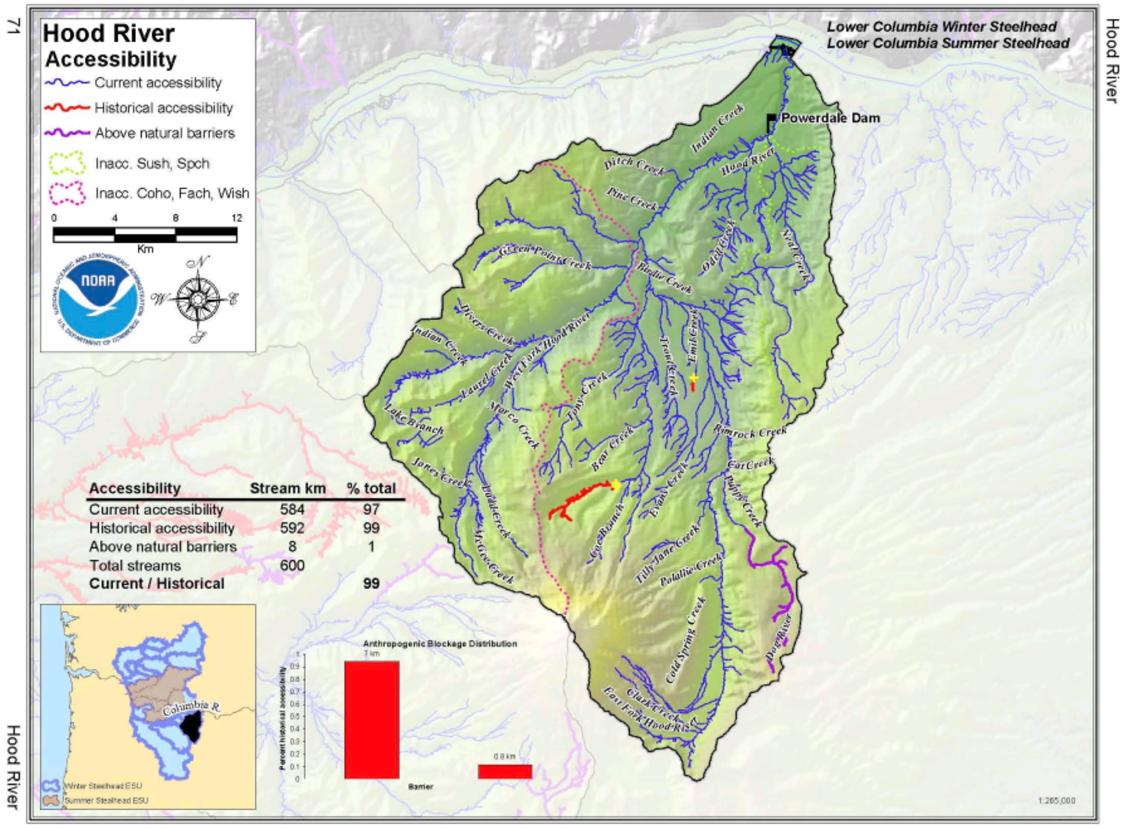


Figure 32: Hood River winter and summer steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Hood River (Summer)

Nearly the entire historical habitat remains accessible to summer steelhead, although significant production areas are largely limited to the West Fork (Figure 32). However, the limited distribution of summer steelhead in the basin warrants a downward adjustment to the spatial score.

SS – Criterion Summary

Steelhead in the Sandy basin have experienced a greater than 30% loss of the habitat historically accessible to steelhead due to anthropogenic blockages, primarily dams on the Bull Run River (Figure 33). For the remainder of the populations, less than 5% of historically accessible habitat has been lost. SS scores for each population were adjusted, where applicable, on the basis of two primary factors: 1) the suitability/quality of the blocked habitat with respect to steelhead production; and 2) the degree to which the remaining accessible habitat has been degraded from historical conditions. The adjustments and final SS scores for each population are presented in

Table 15.

For the SS criterion the most probable risk category for a majority of the populations was 'low' as evidenced by the SS rating in

Table 15 and illustrated by the placement of the widest portion of the diamonds in Figure 34- the Sandy population, with a most probable rating of “moderate risk” being the exception. However, these diamonds also show that there is a substantial level of uncertainty associated with the scoring. For example, as illustrated by the placement of the lower portion of the diamond symbols it is possible (but not probable) that all of the populations could fall into the ‘low risk’ category (Figure 37). However, the most probable call on the overall picture for LCR steelhead in Oregon with respect to this criterion would be the ‘low risk’ category.

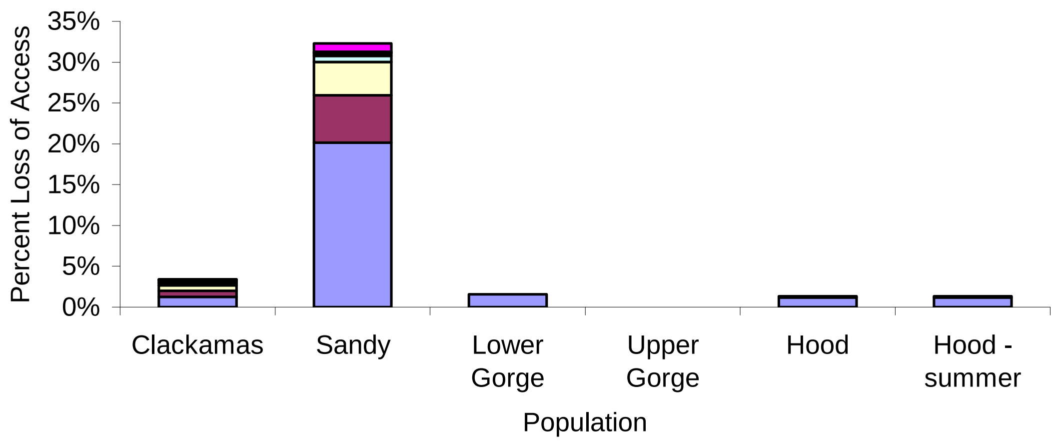


Figure 33: Percent loss in LCR winter and summer steelhead accessibility due to anthropogenic blockages (based on Maher et al. 2005 except as noted). Each color represents a blockage ordered from largest to smallest (bottom-up). The topmost blockages, for example the very top segment of the Sandy bar, represent a collection of many smaller blockages. Note that the pool of smaller blockages can be greater than larger single blockages. These percentage estimates are based on most recent (2007) barrier information that differs from the Maher et al. figures as described in the accessibility map figure legends.

Table 15: Spatial structure scores for LCR steelhead.

Population	Base Access Score	Adjustment for Large Single Blockage	Adjusted Access Score	SS Rating*	Confidence in SS rating
Clackamas	4	N	4	3	M
Sandy	2	Y	1.5	1.5	M
Lower Gorge Tributaries	4	N	4	3	L
Upper Gorge Tributaries	4	N	4	3	L
Hood River	4	N	4	3	M
Hood River – summer	4	N	4	3	L

* Considers Access Score, Historical Use Distribution, and Habitat Degradation.

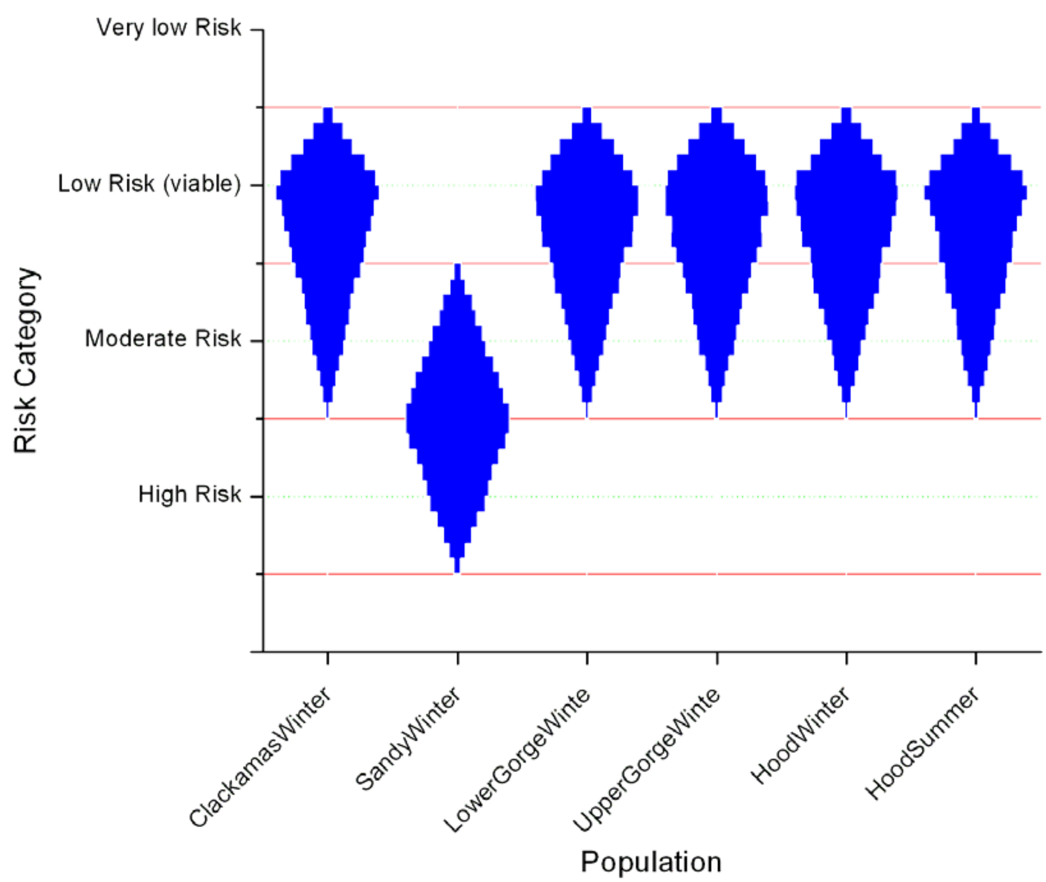


Figure 34: Lower Columbia River steelhead risk status summary based on evaluation of spatial structure only.

IV. Diversity

DV – Background and Overview

Two major life history types of steelhead were historically, and are presently, found in the Lower Columbia River: the summer run and winter run. The life histories of summer- and winter-run steelhead have considerable overlap. Both rear in freshwater for 1 to 4 years prior to smoltification, select similar habitats for freshwater rearing, and spend 1 to 4 years in the ocean. However, substantial differences separate these races at the time of adult freshwater entry, degree of sexual maturity at entry, spawning time, and frequency of repeat spawning.

In the Lower Columbia River, most wild steelhead are 4 to 6 years of age at first spawning, 50 to 91 cm in length, and 2 to 8 kg in weight. However, they can attain ages of 9 years old and reach lengths of over 100 cm (12 kg) (Busby et al. 1996). Steelhead may spawn more than once, although the frequency of repeat spawners is currently relatively low (<10%). At least 9 different initial and 13 different repeat age classes have been identified for Lower Columbia River steelhead (Leider et al. 1986).

Each year, the majority of naturally produced Lower Columbia River summer steelhead return to freshwater primarily between May and October. These fish are sexually immature upon return to their natal streams. The fish subsequently spawn between January and June, with peak spawning between late February and early April (Leider et al. 1986, WDFW unpublished data). The repeat spawner rate is about 5.9% for wild summer steelhead (Hulett et al. 1993). In contrast, wild winter steelhead enter freshwater as sexually mature fish between December and May. Spawning occurs between February and June, with peak spawning time in late April and early May, almost two months later than wild summer steelhead (Leider et al. 1986 and WDFW unpublished data). The repeat spawner rate for wild winter steelhead is 8.1% on the Kalama River; double that of wild summer steelhead (Hulett et al. 1993).

On average, there is a 2-month difference in peak spawning time between winter- and summer-run steelhead, although there is probably certainly some temporal overlap in the spawning distribution (Busby et al. 1996). Within the same watershed winter and summer steelhead maintain a high degree of reproductive isolation by spawning in geographically distinct areas. Hatchery introductions, especially with non-native steelhead, and modifications to barrier falls are a potential source for the breaking down of historical reproductive barriers and the erosion of locally adapted genotypes.

The tendency for summer-run steelhead to return to specific areas above barrier falls may require a higher level of homing fidelity than exhibited by chinook salmon or winter-run steelhead. This fidelity may have important consequences in the rate of development or specificity of locally-adapted traits.

Phelps et al. (1997) examined the relationship between coastal summer and winter steelhead populations. In their genetic analysis, the summer and winter runs within the genetic diversity units (GDUs) were more closely related to each other than to collections from other GDUs, indicating that the run-timing characteristics evolved from a single evolutionary source within each basin. A similar relationship has been observed between spring and fall-run chinook salmon in coastal watersheds in Washington, Oregon, and California, including the Lower Columbia River (Myers et al. 1998). This relationship provides a framework for evaluating the genetic effects of hatchery transfers on target populations.

DV – Clackamas River Winter Run

Life History Traits – Abernethy (1886) reported that steelhead entered the river from December 1st to February 15th. Currently, Clackamas River winter steelhead enter the river from February through May and spawn from May to June (Murtagh et al. 1992). Olsen et al. (1992) reported that prior to the introduction of early-winter (Big Creek) steelhead, passage at North Fork Dam peaked in May. The majority of steelhead return at 4 years of age, with a repeat spawning incidence of 11% (Chilcote 2001). The apparent change in run timing may be due to a number of factors – further investigation is needed. Score = NA

Effective Population Size – In recent years the abundance of returning adults to North Fork Dam has been several hundred to a few thousand, although the long-term average is approximately 450. Score = 3.0

Hatchery Impacts

> *Hatchery Domestication* – There are three hatchery stocks of steelhead released into the Clackamas River, early-winter (introduced), late-winter (native), and summer run (introduced). Since 1999, only unmarked steelhead have been allowed above North Fork Dam, although prior to that the hatchery contribution was about 25% of the run. The ODFW Clackamas Hatchery currently rears a winter run broodstock (122W) developed from unmarked fish at North Fork Dam. In 2003, 18 females and 32 males were spawned (including 25 unmarked fish) at the Clackamas Hatchery for the "wild" broodstock. Score = NA.
>
> *Hatchery Introgression* – There are a number of hatchery programs that release steelhead into the Clackamas River Basin; however only the Clackamas Hatchery winter steelhead (#122) derived from late returning "native" spawners is considered part of the ESU (SHAGG 2003). The Big Creek Hatchery stock of winter steelhead return to the Clackamas River earlier, October to early March, than the native winter steelhead, February to June (Murtagh et al. 1992). Furthermore, the peak spawning period for Big Creek derived fish is January to early March compared with May and June for native Clackamas River winter steelhead
>
> The introduction of early-winter and summer steelhead from outside of the basin may have influenced the diversity of the native late-winter run, although differences in run timing probably limit the degree of introgression. Chilcote (2001) estimated that competition between summer and winter-run steelhead probably reduced the productivity of the winter run population, but it is not know if there has been any effect on life history diversity. Score = 2-3.
>
> *Synthetic Approach* – The situation in the Clackamas is somewhat complex given that two (Skamania-derived summer run and Big Creek-derived winter run) of the three runs of steelhead released into the basin are not native. The locally-derived late-winter steelhead hatchery broodstock program is relatively small. Currently, hatchery fish are removed at North Fork Dam, although prior to 2002 summer run fish were released into the Upper Clackamas River. The proportion of hatchery-origin fish is on the spawning grounds (lower river only) is presently 25%, although in past years it

has been much higher (0.10<Ph <0.30). On average the genetic similarity between hatchery- and naturally-produced is very low. Diversity persistence score = 1.0 – 2.0.

Anthropogenic Mortality – Harvest rates on "unmarked" winter steelhead are thought to be relatively low (<5.0%). From 1917 to 1939, passage at Faraday Dam (North Fork Dam) was blocked after the fish ladder washed out in a flood, prior to this, passage was somewhat restricted. After 1939, much of the watershed was naturally recolonized by steelhead. It is not know how habitat degradation in the lower Clackamas River and lower mainstem Willamette River and its tributaries (Kellogg and Johnson Creeks) may have influenced life history characters. Score =2-3.

Habitat Diversity – Although the quality of habitat may be severely degraded the proportion of there has been little change in the size distribution of the Clackamas and its tributaries. There has been a marked loss in the elevation complexity of the basin. Score = 2/4.

***Overall Score* = 2.5.** There may have been a change in life history characters with the blocked passage at Faraday Dam for 20 years. Effective population size is moderate, with several low abundance years.

Previously: 2004 TRT 1.58; 2004 ODFW pass, all criteria meet.

DV – Sandy River Winter-Run Steelhead

Life History Traits **–** Winter and summer steelhead are present in the Sandy River Basin, although only winter steelhead are thought to be native (Kostow 1995). Steelhead spawning operations on the Salmon River, a Sandy River tributary, collected eggs from March 20 to May 27, 1901 (ODF 1903), a spawn timing similar to present-day native steelhead in the Sandy River, March to early May (Olsen et al. 1992). Current age structure, 63% 4-year-old and 23 % 5-year-old spawners, does not appear to be divergent form other populations in this stratum (Chilcote 2001). Little available information; no known changes. Score = NA.

Effective Population Size – Historically, winter steelhead escapement may have been in excess of 20,000 fish (Mattson 1955). Loss of spawning habitat in the Bull Run and Little Sandy River Basins in combination with the effects of dams on the mainstem Sandy River reduced the run to 4,400 in 1954. More recently, the estimated "wild" escapement of hatchery fish over Marmot Dam (RKm 43) was 851 in 1997, although there was considerable difficulty in distinguishing between wild and hatchery derived winter steelhead (Chilcote 1997). Score = 2-3.

Hatchery Impacts

Hatchery Domestication – Winter steelhead have been propagated in the Sandy River Basin since 1900 (Wallis 1963). There have been three winter steelhead stocks released in the Sandy. Initially, returning adults were intercepted for use as broodstock. Beginning in 1960, Big Creek winter steelhead were introduced into the Sandy River (Wallis 1963). Recently, there has been a phase out from the release of the Big Creek stock (ODFW#013) in favor of the locally derived Sandy River broodstock (ODFW#011W). In 2003, 81 unmarked fish were collected at the Marmot trap (approximately 50% spawners used for the wild broodstock). Hatchery fish constituted nearly 40% of the winter steelhead passing over Marmot Dam in 1997 (Chilcote 1997). However, the frequency of hatchery fish arriving at Marmot Dam has also declined in recent years. In addition, the removal of all marked (hatchery) fish at the Marmot Trap beginning in 1999 prevented hatchery-origin fish from accessing the primary steelhead production areas upstream of the dam. Therefore, the percentage of hatchery fish spawning upstream of Marmot Dam since 1999, has effectively been zero (see Appendix B).Releases of summer steelhead (Skamania Hatchery stock) began in 1976, and spawning escapement to Sandy River currently averages 2,000 fish (Chilcote 1997). Additionally, there are plans to remove several dams on the Bull Run that may provide additional spawning and rearing habitat to a tributary that once produced significant numbers of steelhead (Mattson 1955). *PNI ≤ 1.00 (current) 6 years, 0.25 (historical) 80 years, Fitness = 0.60.* Score = 1.5.

Hatchery Introgression – For a number of years, non-local Big Creek steelhead and Skamania summer steelhead have been released in to the Sandy River. Big Creek releases have been terminated, but still continue for Skamania Hatchery Fish. Due to differences in spawn timing it is not know to what extent the early-winter and summer runs have interbreed with the local population. Competition effects are likely to continue between released summer run and local winter run juveniles. Score = NA

Synthetic Approach – The hatchery situation in the Sandy River is currently in a transitional state. The release of early-winter run steelhead (Big Creek Hatchery) has recently been terminated in preference to a locally-derived late winter run. In addition, summer-run steelhead (Skamania Hatchery) have been released into the basin since 1976. Hatchery (marked) steelhead have been removed at the Marmot Dam trap since 1999. There is likely little steelhead spawning in the lower portion of the river (below Marmot Dam); therefore the effective stray rate is near 0. (Ph <0.05). Naturally-spawning hatchery fish would include both out-of-ESU summer run fish (potentially including feral summer run fish) and locally-derived winter run fish with an overall low genetic similarity between hatchery and wild populations. In consideration of the duration of past hatchery releases throughout the basin the score was reduced by 1. Diversity persistence score = 4.0 - 1.0 = 3.0.

Anthropogenic Mortality **–** Prior to 1991, harvest rates for Sandy River winter steelhead averaged 40%, but with the initiation of selective fisheries this rate dropped to 4% for unmarked fish (Chilcote 2001). Changes in mainstem and estuary habitat may have had an influence on life history diversity – although it is not possible to quantify this effect. Score = NA**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion of accessible stream size reflects historical conditions, while much of the elevation diversity has been lost. Score (Order/Elevation) = 2/3.

Overall Score = 2.0. The long-term effects of the steelhead hatchery program may have had considerable influence on diversity. Also, there are a number of effects (e.g., habitat degradation and harvest) that may have influenced diversity but the information on these processes is limited.

Previously: 2004 TRT 1.56; 2004 ODFW fail, 4-5 criteria met.

DV – Lower and Upper Gorge[1] Tributaries Winter-Run Steelhead

Life History Traits **–** The only information available is from the Washington side of the DIP, Hamilton Creek and Wind River winter run steelhead. River entry begins in December and extends to early May, with spawning occurring from March to early June (SaSI 2003). There is no historical information on steelhead from either side of the Columbia River. Score = NA.

Effective Population Size – Information on the escapement to these DIPs is largely unknown. Some survey work has been undertaken, but on an inconsistent basis. In general, escapement in each of the DIP likely numbers in the tens or low hundreds of fish. Score = 1-2.

Hatchery Impacts

Hatchery Domestication – There have been a number of hatchery releases into these DIPs, although the persistence of these releases is unknown. Although no estimate could be generated, the effect is thought to be significant. Score = NA.

Hatchery Introgression – There is little information on out-of-stratum or out-of-ESU introductions or strays. While large numbers of summer steelhead migrate through these DIPs bound for the Mid and Upper Columbia River and Snake Rivers it is unlikely that any would stray into the small tributaries along the Oregon side of these DIPs. Score = NA.

Synthetic Approach (Lower Gorge) – There is very little available information on the influence of hatchery-origin fish on spawning aggregations within this population. Historically there have been a number of releases from various hatcheries, but there are currently no winter run being released. Large numbers of predominately summer run steelhead migrate past the small tributaries on the Oregon side of this DIP, but it is unlikely that they would be diverted into these small systems. While the number of hatchery fish naturally spawning may be low, the overall abundance in this DIP is also probably low. As a percentage hatchery fish may be significant (Ph>0.10) and the genetic similarity low to very low. Diversity persistence score = 2.0 or 3.0.

Synthetic Approach (Upper Gorge) – There is very little available information on the influence of hatchery-origin fish on spawning aggregations within this population. Historically there have been a number of releases from various hatcheries, but there are currently no winter run being released from the Oregon side of this DIP (although on the Washington side, there are large releases into the Wind River. Large numbers of predominately summer run steelhead migrate past the small tributaries on the Oregon side of this DIP, but it is unlikely that they would be diverted into these small systems. ODFW suggests that this DIP may be similar to the Hood River winter run DIP. While the number of hatchery fish naturally spawning may be low, the overall

---

[1] In light of the paucity of information on these two DIPs, the evaluations have been combined. As more specific information becomes available, it will be useful to evaluate these DIPs independently.

abundance in this DIP is also probably low. As a percentage hatchery fish may be significant (Ph>0.10) and the genetic similarity low to very low. Diversity persistence score = 2.0 or 3.0.

Anthropogenic Mortality **–** Prior to 1991, harvest rates winter steelhead were about 20%, but with the initiation of selective fisheries this rate should have dropped to 4% or less for unmarked fish (Chilcote 2001). Harvest and habitat effects are likely, but have not been quantified. Spring run net fisheries may have incidentally captured returning winter steelhead, potentially at a high rate. Score = 3-4**.**

Habitat Diversity **–** Habitat diversity estimates were not made for these DIPs.
Score (Order/Elevation) = NA

***Overall Score* = 1.5.** Low effective population size and the effects of the hydro-operation have likely influenced these DIPs. Additionally habitat degradation instream and in the migratory/rearing corridors may also have influenced life history diversity.

Previously: 2004 TRT 0.94 (LG) and 0.86 (UG); 2004 ODFW pass, all criteria met.

DV – Hood River Winter Run

Life History Traits **–** Based on observed run timing at Powerdale Dam, the "native" winter steelhead return from March to late June (Olsen et al. 1994). Chilcote (2001) estimated that 60% of the fish returned at Age 4 and 25% at Age 5. Score = NA.

Effective Population Size – Escapement has ranged from a few hundred to nearly a thousand fish with varying levels of hatchery fish contributing to escapement (Goodson 2005). Score = 2-3.

Hatchery Impacts

> *Hatchery Domestication* – Hatchery winter steelhead (ODFW Big Creek Hatchery #13) were released into the Hood River Basin since 1962. Genetic analysis by Schreck et al. (1986) indicated that the Hood River Hatchery broodstock was similar to Eagle Creek NFH broodstock (Big Creek influenced). The program was terminated following the development of a local winter steelhead broodstock (ODFW #50W) in 1991. The winter steelhead #50W broodstock was established using unmarked returning steelhead, although it is possible that some naturally produced Big Creek origin fish were incorporated (as well as unmarked fish from other basins or hatcheries). Hatchery broodstock have been derived from a mix of returning marked fish and unmarked fish captured from the river – unmarked fish have contributed from 50 – 100% of broodstock in any given year. Genetically, the present-day Hood River and Big Creek winter steelhead are quite distinct from one another (Kostow et al. 2000). It is not known to what extent non-native hatchery introductions and habitat degradation have altered life history trait expression. For 2000-2004, the average contribution of hatchery fish to natural escapement was 39% (Goodson 2005). PNI ≤ 0.6, Fitness > 0.90. Score = 3-4.
>
> *Hatchery Introgression* – The introduction of Big Creek winter steelhead may have resulted in the loss of local adaptation. Recent genetic analysis suggests that the legacy of these introductions has been minimal. Score = NA.
>
> *Synthetic Approach* – Both winter and summer run steelhead are released into the Hood River Basin. In 1991 a locally derived winter run hatchery broodstock was developed for the Hood River, prior to that Big Creek Hatchery early-winter run steelhead were released. Recent information from fish passed over Powerdale Dam in the lower Hood River suggest nearly 50% of the run is of hatchery origin (0.75 > Ph > 0.30). Approximately half of the broodstock used in the hatchery are naturally produced. Diversity persistence score = 1.0 - 2.0.

Anthropogenic Mortality **–** Chilcote (2001) estimated that the average harvest rate from 1995-2000 for unmarked "wild" fish was approximately 14%. Changes in river conditions in the Hood River Basin and in the migratory and rearing corridors in the mainstem and estuary may also have affected life history diversity, but to an unknown extent. Score = NA**.**

Habitat Diversity **–** Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score (Order/Elevation) = 1/4

Overall Score = 2.5. Effective population size was a primary concern; however, hatchery effects and habitat degradation are largely unknown but probably significant factors.

Previously: 2004 TRT 1.81; 2004 ODFW pass, all criteria met.

DV – Hood River Summer-Run Steelhead

Life History Traits **–** Steelhead enter the river from May to early November (Olsen et al. 1994). Chilcote (2001) estimated that 56% of the run consists of Age 4 fish, and 29% of Age 5 fish. Score = NA.

Effective Population Size – Native summer steelhead escapement was 181 in 1997, and may have been as low as 80 in 1998 (Chilcote 1997). Since that time abundance has averaged a few hundred fish, 293 for 2000 to 2004 (Goodson 2005). Score = 2.

Hatchery Impacts

Hatchery Domestication – A local summer steelhead broodstock (ODFW #50W) was established in 1998, using unmarked returning summer steelhead. Skamania Hatchery derived summer steelhead (ODFW #24) have been released in the basin for a number of years, and it is possible that unmarked (naturally produced) Skamania summer steelhead were incorporated into the broodstock (Kostow et al. 2000). From 1993 to 1998, unmarked summer steelhead accounted for only 16.1% of the summer steelhead passed over Powerdale Dam (Goodson 2005). Beginning in 1997, however, releases in the upper basin were terminated and marked summer steelhead are prevented from migrating past Powerdale Dam (Rkm 6.4). With the development of a locally-base broodstock, the percentage of hatchery-origin fish allowed past Powerdale Dam has increased to 58% of escapement in 2004. Unmarked fish are used as broodstock for the current hatchery program (50W). There is no genetic analysis available for Hood River summer steelhead. *PNI ≤ 0.85, Fitness > 0.90*. Score = 3-4.

Hatchery Introgression – It is unclear to what extent previous releases of Skamania Hatchery summer steelhead may have influenced the genetic and phenotypic diversity of the local population. Future genetic studies may provide some insight into this effect. Score = NA.

Synthetic Approach – Both winter and summer run steelhead are released into the Hood River Basin. In 1998 a locally derived summer run hatchery broodstock was developed for the Hood River, prior to that Skamania Hatchery summer run steelhead were released. Recent information from fish passed over Powerdale Dam in the lower Hood River suggest nearly 50% of the run is of hatchery origin (0.75>Ph>0.30). Currently, unmarked fish are used as broodstock—high to moderate.

Diversity persistence score = 2.0 - 3.0.

Anthropogenic Mortality **–** Chilcote (2001) estimated that the average harvest rate from 1995-2000 for unmarked "wild" fish was approximately 10%. Changes in river conditions in the Hood River Basin and in the migratory and rearing corridors in the mainstem and estuary may also have affected life history diversity, but to an unknown extent. Score = 3-4.

Habitat Diversity – Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions.

Score (Order/Elevation) = 3/4.

***Overall Score* = 2.0.** Of the factors that could be evaluated, effective population size had the strongest downward effect on diversity. Past hatchery introductions and habitat degradation effects were also thought to be significant, but could not be evaluated. Previously: 2004 TRT 1.26; 2004 ODFW fail, 4-5 criteria met.

DV – Criterion Summary

With the exception of the Gorge populations, there is empirical evidence that all of the historical populations in Oregon's portion of this DIP are extant. Loss of genetic resources due to small population size during the 1990s and high incidence of hatchery strays are the primary reasons that the majority of the populations had a most probable risk classification 'moderate' or 'high' (Figure 35). Only the winter steelhead populations in the Clackamas and Hood basins met the viable threshold, and just barely so. Because of the uncertainty associated with the population ratings for the DV criterion, the possibility exists that three of the six populations fall into the 'high risk' category, as illustrated by the placement of the lower portion of the diamonds in Figure 38. However, overall we believe the most probable DV risk classification for Oregon's LCR steelhead populations is 'moderate'.

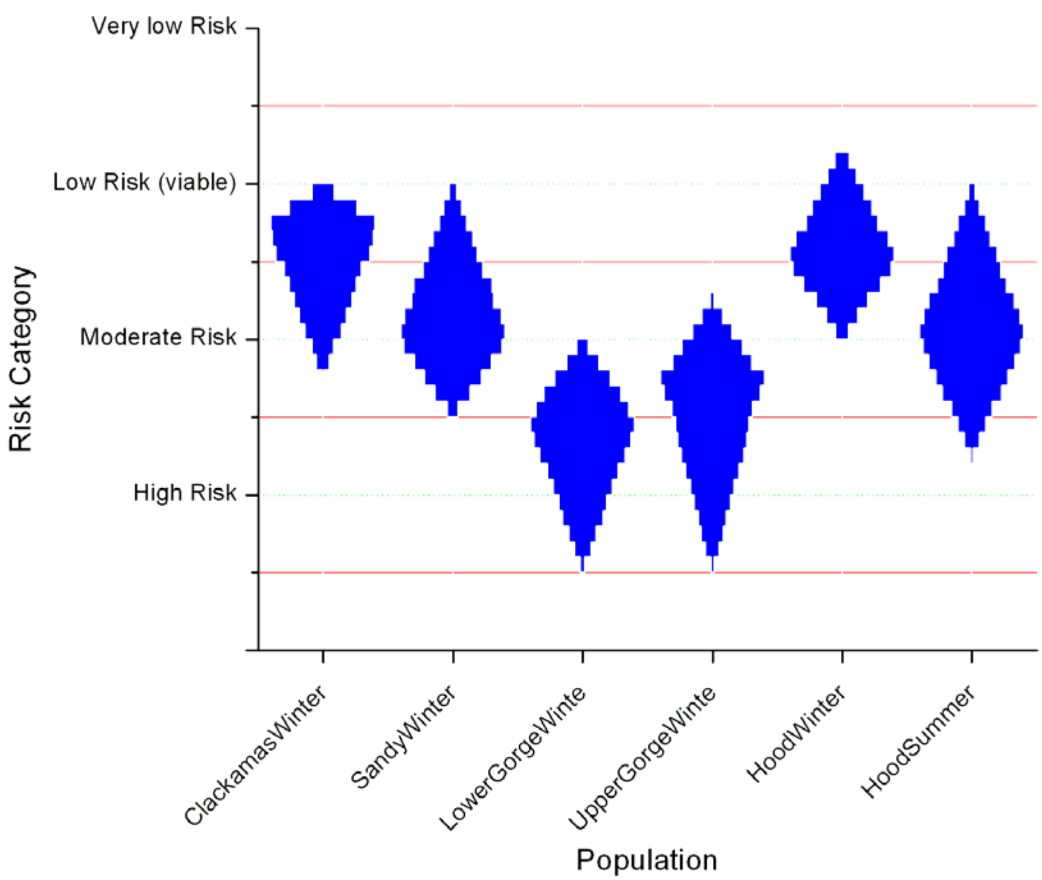


Figure 35: Lower Columbia River steelhead risk status summary based on evaluation of diversity only.

V. Summary of Population Results

The result we obtained when the scores for all three population criteria were combined was that the risk of extinction for LCR steelhead in Oregon's portion of this DIP was high. Results using the minimum distribution method illustrated by Figure 36 and Figure 37 support this conclusion. A most probable classification for the Clackamas population is low risk. Three of the six populations were clearly in the high risk category. The uncertainty associated with these scores was considerable, as evidenced by the relatively stretched aspect of the diamonds for Hood winter and two Gorge populations. However, we conclude that the most probable risk classification for Oregon's LCR steelhead is 'moderate'.

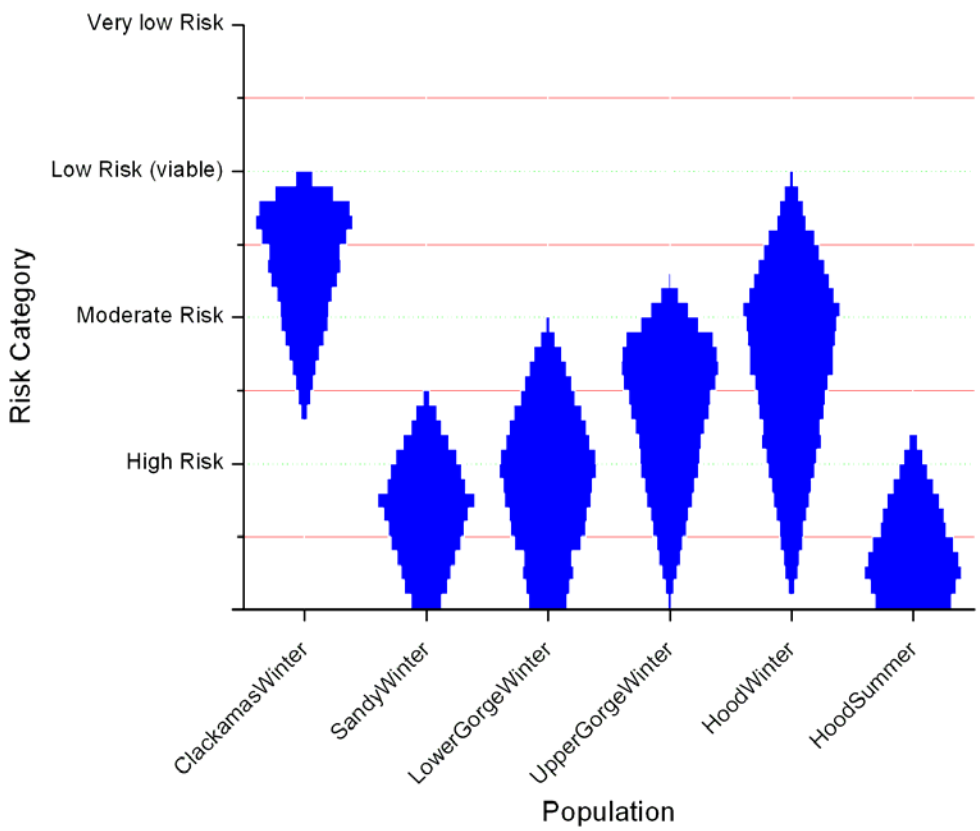


Figure 36: Oregon Lower Columbia River steelhead population status summaries based on minimum distribution method.

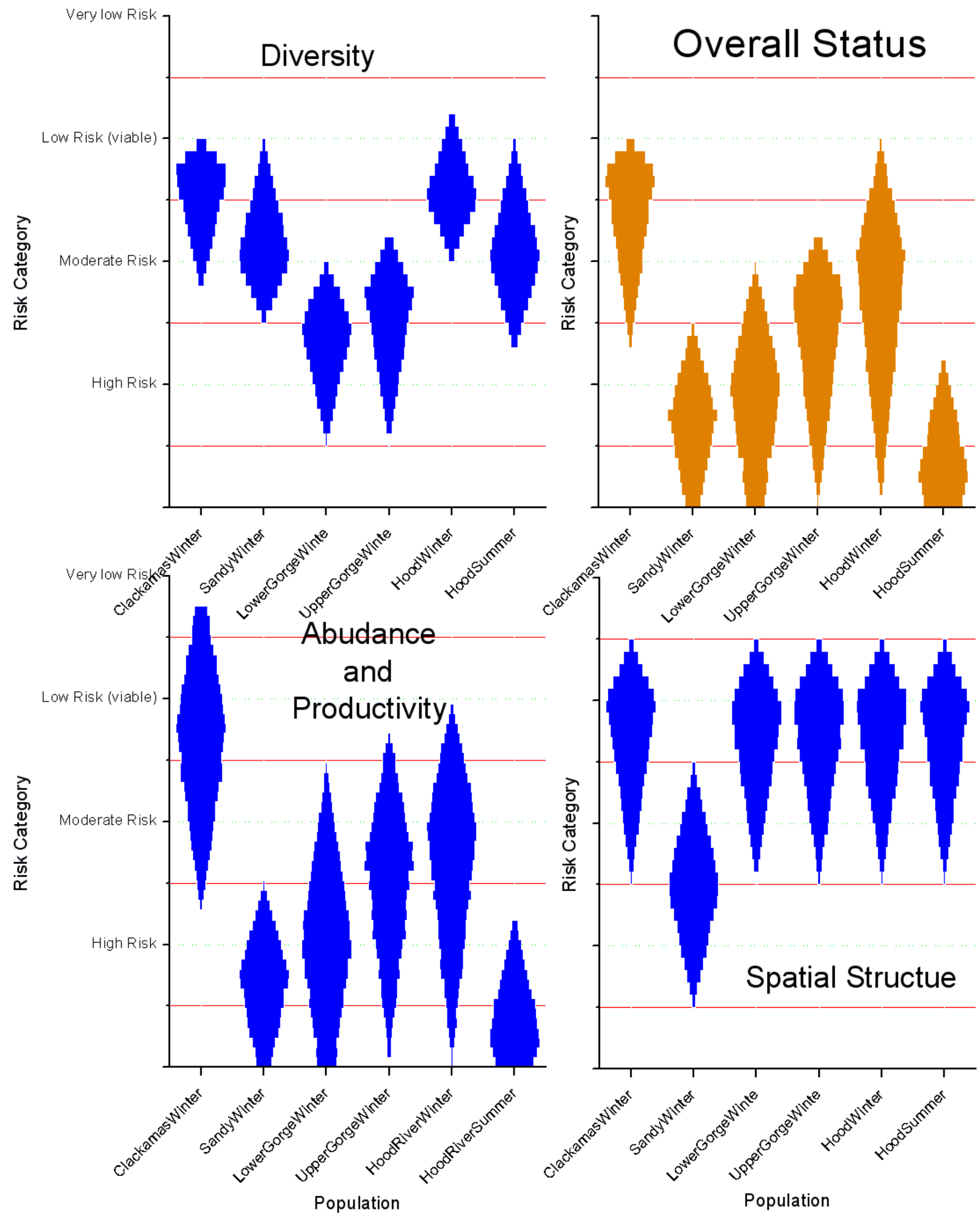


Figure 37: Oregon Lower Columbia River steelhead status graphs of each attribute and the overall summary.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 6: Upper Willamette Chinook

September 2007

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

Table of Contents

I. ESU Overview and Historical Range

The UW chinook ESU consists of seven populations as shown in **Error! Reference source not found.**. All the populations in the ESU are in a single stratum since they share a similar life history pattern (spring run) and a single ecozone (McElhany et al. 2003, Myers et al. 2006).

Spring chinook in the Willamette basin are extremely depressed. Historically, the spring run of chinook may have exceeded 300,000 fish (Myers et al. 2003). However, not only is the current ESU abundance of wild fish less than 10,000 fish, but only in two locations (McKenzie and Clackamas) does significant natural production occur. This ESU has been adversely impacted by the degradation and loss of spawning and rearing habitat associated with hydropower development as well as by interactions with the large number of natural spawning hatchery fish. Further, only in recent years has it been possible to separately identify hatchery and wild fish, thereby making the assessment of natural spring chinook populations feasible.

The presentation of our assessment begins with three sections, each of which evaluates one of the viability criteria (i.e., abundance/productivity, spatial structure, and diversity). This is then followed by a synthesis section where we pool the results from these criteria evaluations into a status rating for each population. The methods are described in Part 1 of this report. We end our presentation with an interpretation of the population results in terms of the overall status of this ESU.

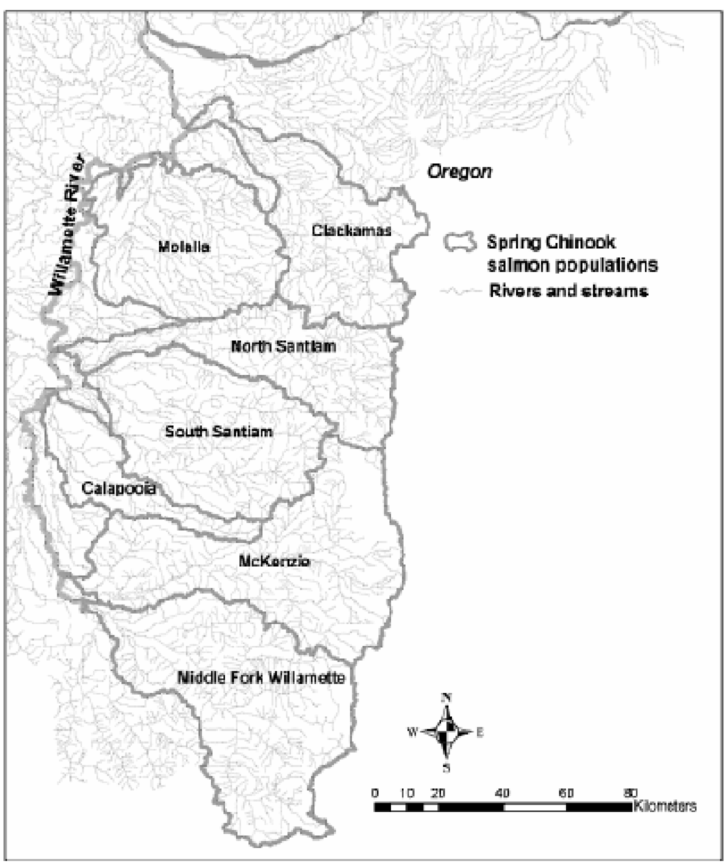


Figure 1: Map of populations in the Upper Willamette chinook ESU.

II. Abundance and Productivity

A&P – Clackamas

A time series of abundance sufficient for quantitative analysis is available for the Clackamas spring run population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 2 to Figure 8 and in **Error! Reference source not found.** to Table 4. The population long-term geometric mean is about 900 natural origin spawners, which is in the moderate risk minimum abundance threshold category (**Error! Reference source not found.**). The impact of fisheries on this population has resulted in an average mortality rate of 35% in recent years. However, there is considerable uncertainty in these mortality rate estimates. Therefore estimates of pre-harvest population productivity, which incorporates these fishery impact rates, are also likely to be imprecise. The pre-harvest viability curve analysis, the CAPM modeling and the PopCycle modeling all suggest that the population is currently viable. The escapement viability curve suggests that a population experiencing the pattern of harvest that occurred over the available time series would most likely be in the moderate risk category. One characteristic of all spring chinook salmon populations we assessed is that there appears to be a high rate of pre-spawning mortality which is an increased risk factor (the effective abundance is lower than estimated by spawner counts). For the Clackamas it has been estimated about 20% of the females die before spawning (Figure 9). The Oregon Native Fish Status report (ODFW 2005) listed the Clackamas spring chinook population as a “pass” for abundance and a “fail” for productivity.

Although there is considerable uncertainty in the analysis of this population for the A&P criterion, we conclude the most probable classification for this population under the A&P criterion is the low extinction risk category.

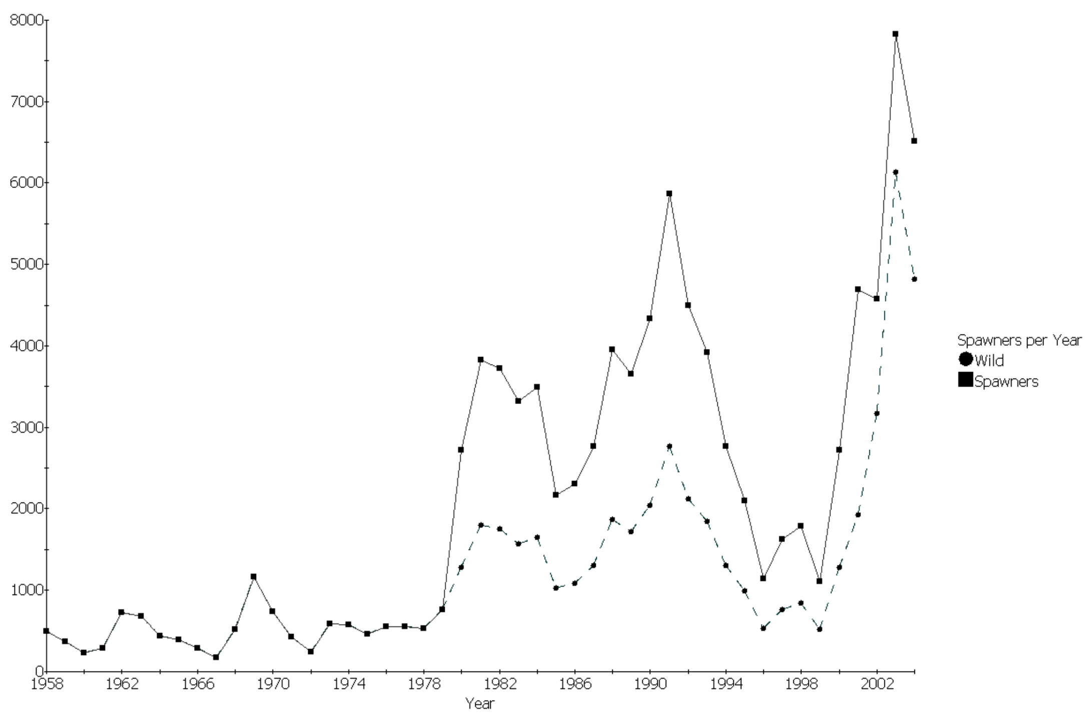


Figure 2: Clackamas River spring chinook abundance.

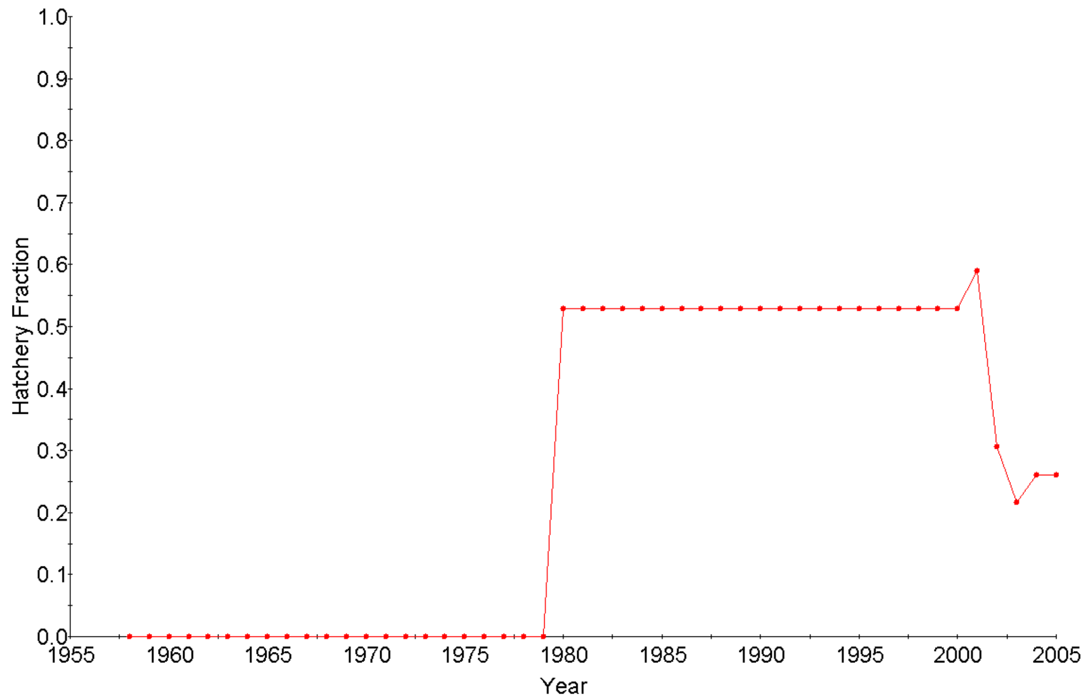


Figure 3: Clackamas River spring chinook hatchery fraction.

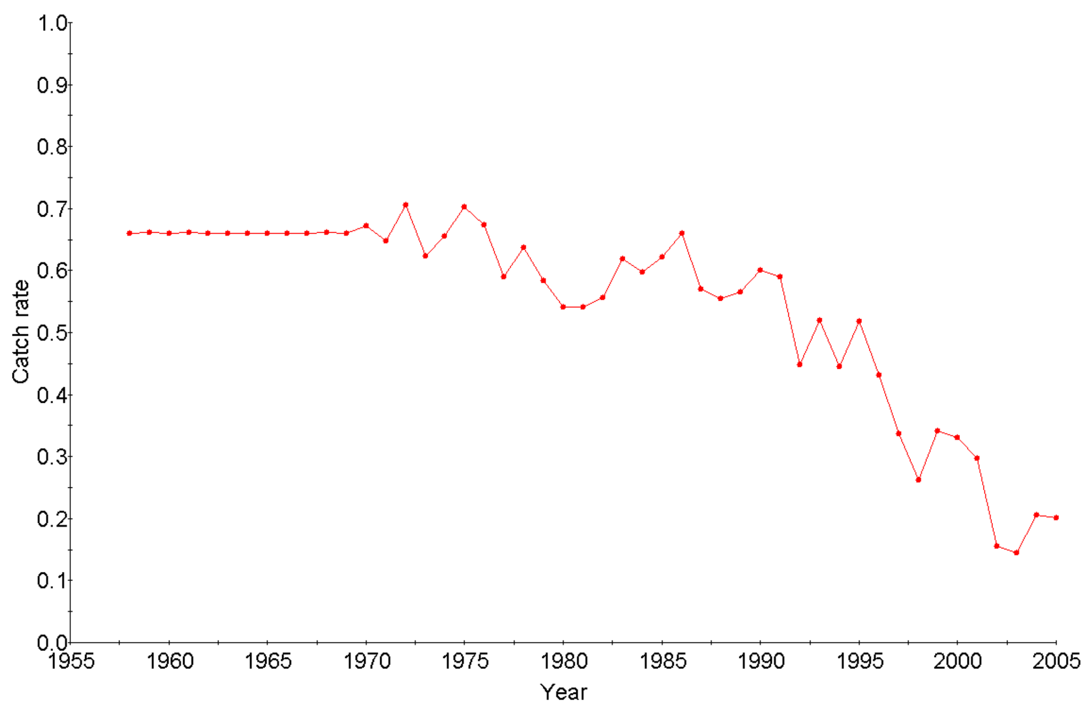


Figure 4: Clackamas River spring chinook harvest rate.

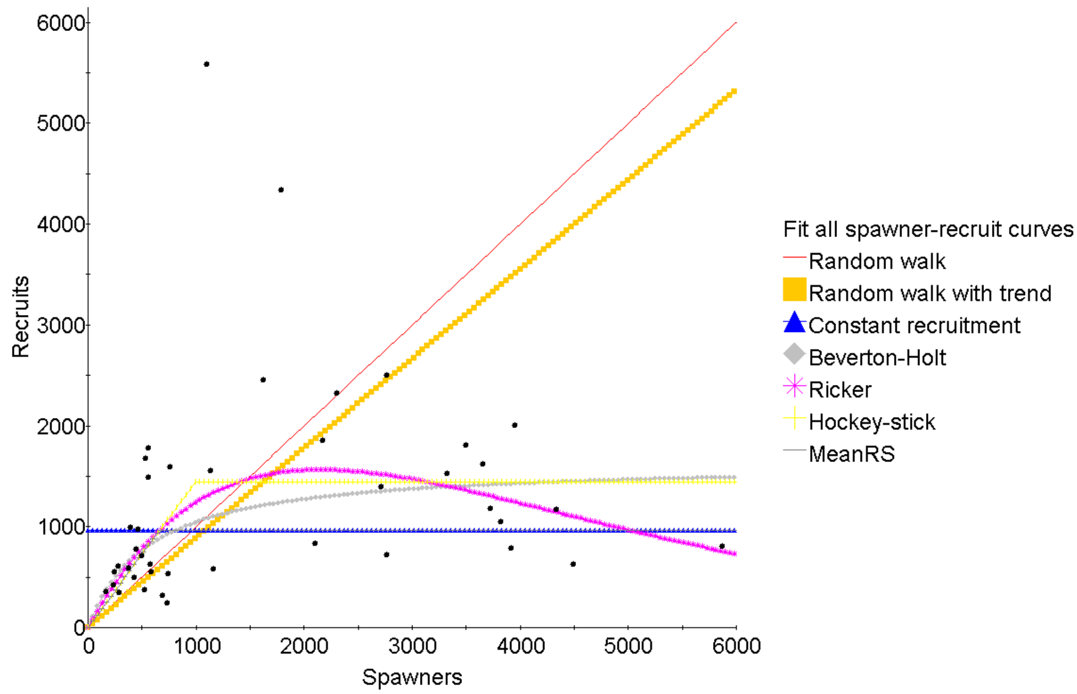


Figure 5: Clackamas River spring chinook escapement recruitment functions.

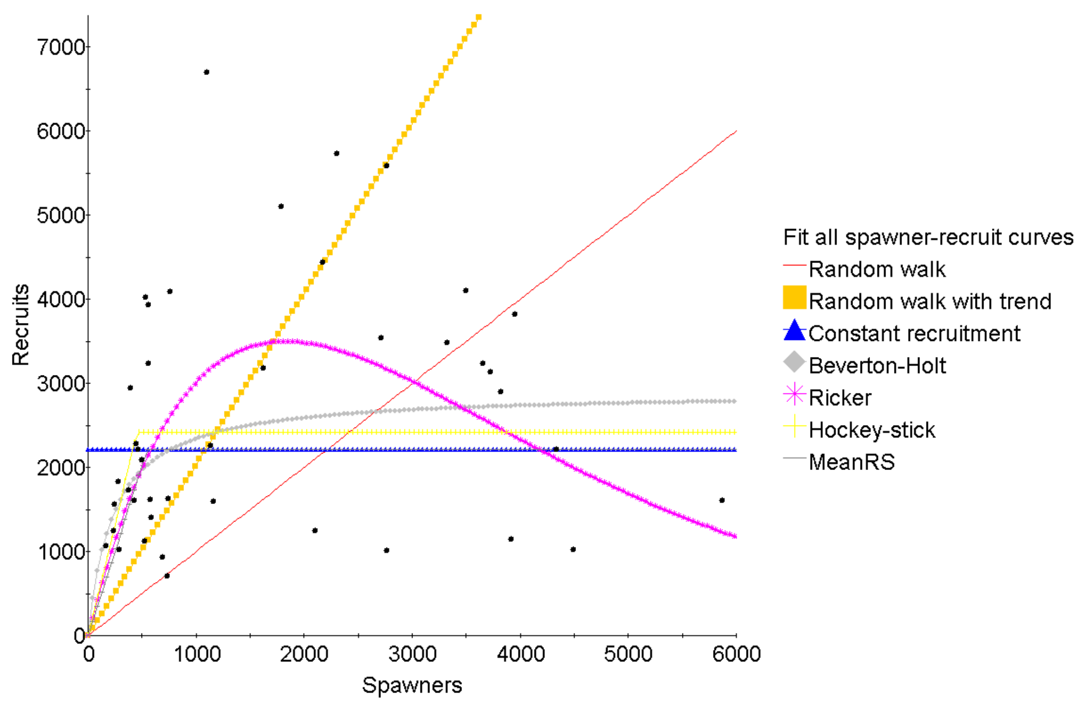


Figure 6: Clackamas River spring chinook pre-harvest recruitment functions.

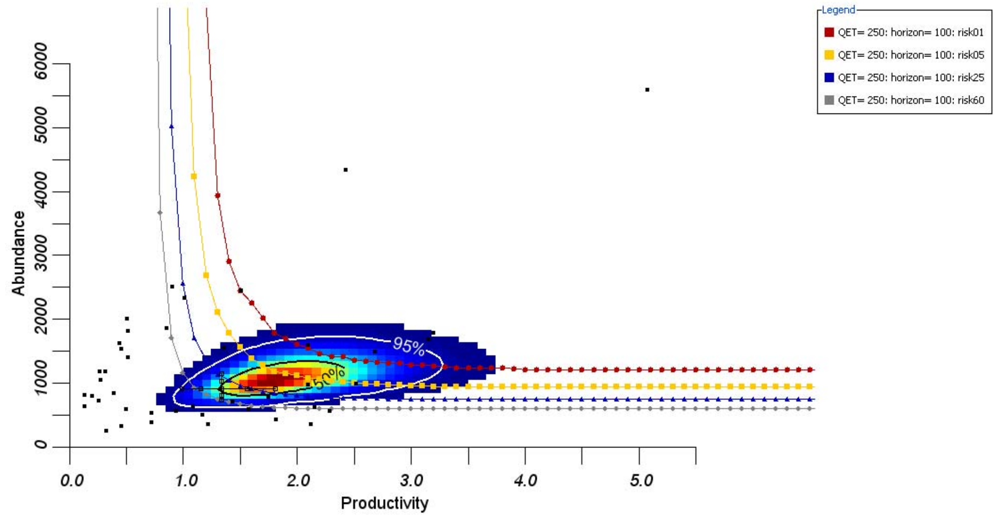


Figure 7: Clackamas River spring chinook escapement viability curve.

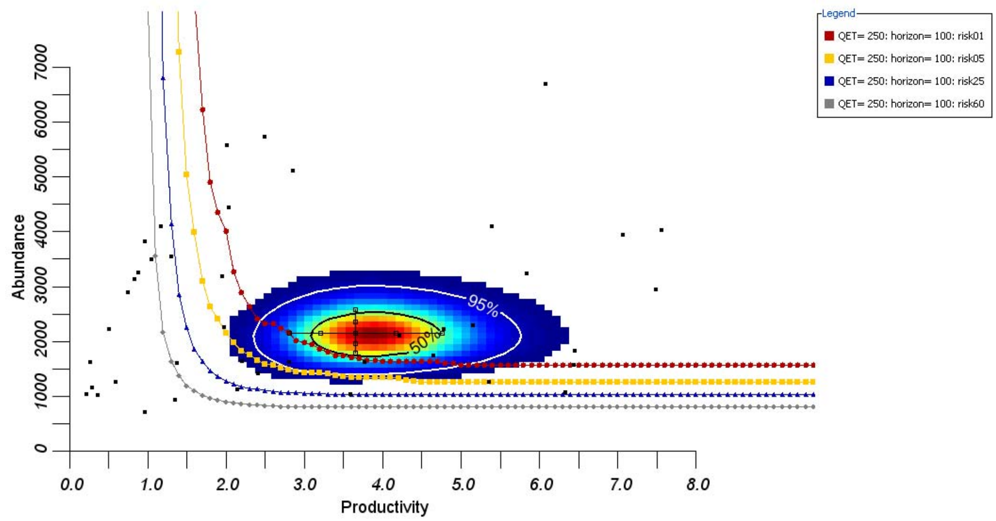


Figure 8: Clackamas River spring chinook pre-harvest viability curve.

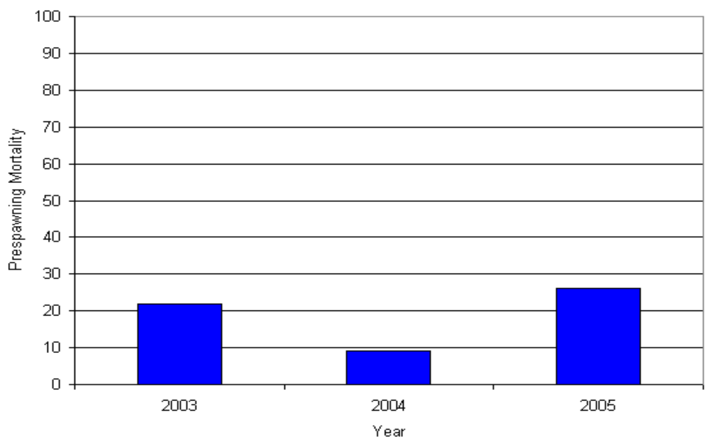


Figure 9: Spring chinook pre-spawning mortality in the Clackamas based on carcass surveys of the fraction of female fish that died prior to spawning (Schroeder et al. 2005).

Table 1: Clackamas River spring chinook summary statistics. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	**Recent Years**	**Total Series**	**Recent Years**
Time Series Period	1958-2005	1990-2005	1958-2005	1990-2005
Length of Time Series	48	16	48	16
Geometric Mean Natural Origin Spawner Abundance	902 (713-1141)	1656 (1122-2443)	NA	NA
Geometric Mean Recruit Abundance	968 (775-1210)	1385 (790-2428)	2216 (1848-2657)	2048 (1266-3313)
Lambda	0.967 (0.849-1.102)	0.902 (0.422-1.929)	1.151 (0.995-1.331)	0.958 (0.487-1.886)
Trend in Log Abundance	1.044 (1.033-1.055)	1.048 (0.965-1.139)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.888 (0.667-1.182)	0.555 (0.221-1.395)	3.8 (2.95-4.897)	0.82 (0.359-1.874)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.462 (1.102-1.94)	1.174 (0.365-3.782)	1.044 (1.033-1.055)	1.566 (0.528-4.644)
Average Hatchery Fraction	0.266	0.466	NA	NA
Average Harvest Rate	0.543	0.364	NA	NA
CAPM median extinction risk probability (5th and 95th percentiles in parentheses)	NA	NA	0.000 (0.000-0.025)	NA
PopCycle extinction risk	NA	NA	0.02	NA

Table 2: Escapement recruitment parameter estimates and relative AIC values for Clackamas spring chinook. The 95% probability intervals on parameters are shown in parentheses. The "best" approximating model (relative AIC=0) is shown in bright green. Models nearly indistinguishable from best (relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.91 (0.78-1.13)	36.4
Random walk with trend	0.89 (0.71-1.16)	NA	0.91 (0.79-1.14)	37.7
Constant recruitment	NA	968 (815-1185)	0.71 (0.61-0.89)	16.6
Beverton-Holt	2.9 (1.98-8.07)	1634 (1140-2301)	0.59 (0.53-0.77)	4.5
Ricker	1.98 (1.59-2.53)	1564 (1369-1900)	0.57 (0.5-0.72)	0
Hockey-stick	1.45 (1.22-2.23)	1446 (1080-1839)	0.59 (0.52-0.77)	4.4
MeanRS	1.46 (1.17-1.79)	968 (811-1164)	0.39 (0.24-0.55)	43.8

Table 3: Pre-harvest recruitment parameter estimates and relative AIC values for Clackamas spring chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.21 (1.04-1.5)	66.7
Random walk with trend	2.03 (1.61-2.72)	NA	0.98 (0.85-1.24)	51.2
Constant recruitment	NA	2217 (1920-2609)	0.58 (0.5-0.72)	6.1
Beverton-Holt	12.19 (7.75-27.39)	2901 (2315-3647)	0.53 (0.47-0.68)	1.7
Ricker	5.2 (4.27-6.52)	3496 (3102-4111)	0.52 (0.46-0.67)	0
Hockey-stick	5.32 (4.14-26.21)	2422 (1999-2891)	0.54 (0.48-0.7)	3
MeanRS	4.02 (3.26-4.88)	2216 (1918-2567)	0.3 (0.21-0.39)	55.9

Table 4: Clackamas spring chinook CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in 'Extirpated or nearly so' category	0.971	1.000	1.000
Probability the population is above 'Moderate risk of extinction' category	0.843	1.000	1.000
Probability the population is above 'Viable' category	0.475	0.996	0.983
Probability the population is above 'Very low risk of extinction' category	0.106	0.895	0.818

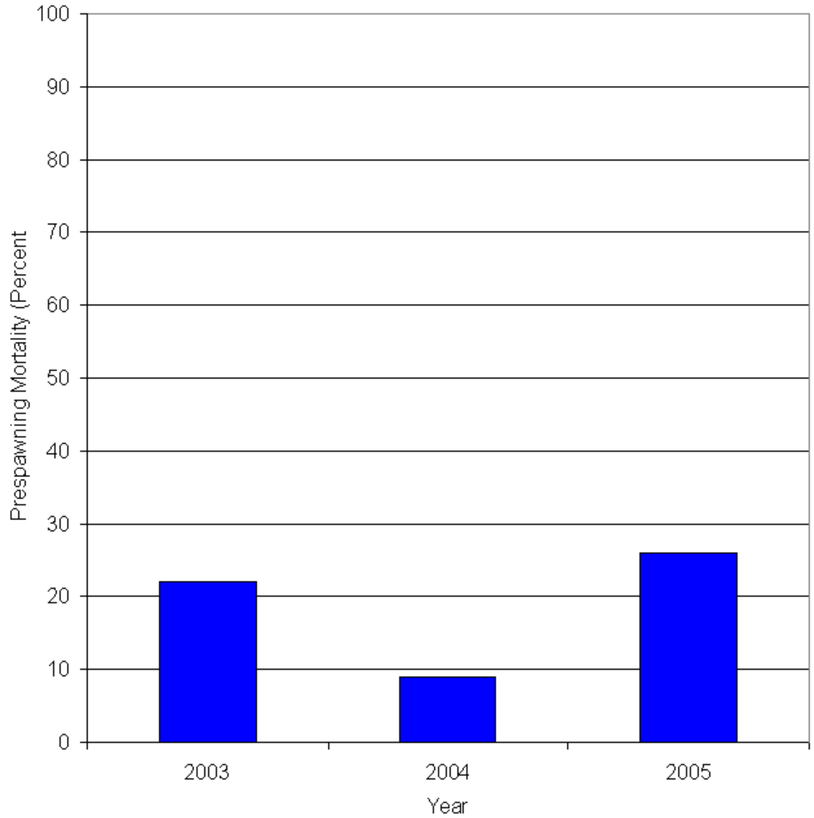


Figure 10: Estimated pre-spawning mortality of spring chinook in the Clackamas River upstream of North Fork Dam. Based on carcass survey (Schroeder et al. 2005).

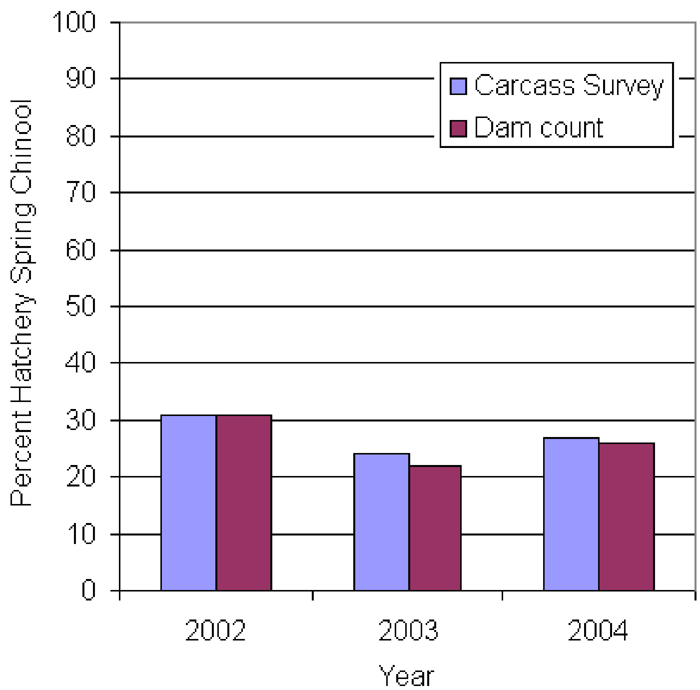
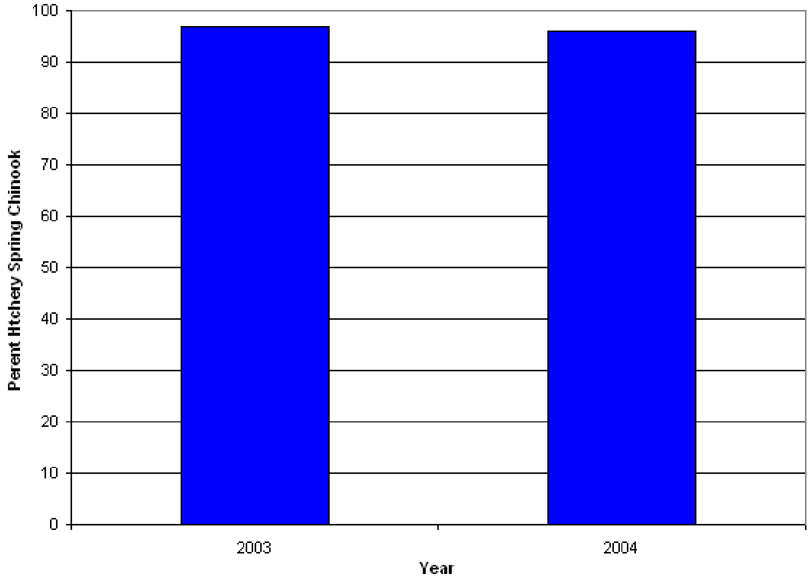


Figure 11: Percent of hatchery origin spring chinook spawners in the Clackamas River upstream of North Fork Dam base on two different estimation methods (Schroeder et al. 2005).

A&P – Molalla

Recent spawning surveys indicate a relatively low density of spawning in the Molalla (Figure 12). Of those fish returning, nearly all are of hatchery origin



(

Figure 13). Pre-spawning mortality in 2003 in the Molalla was estimated at 69% (9 of 13 female carcasses recovered still contained eggs and therefore indicated pre-spawning mortality). Taken together, these data indicate little, if any, natural production of spring chinook in the Molalla. Based on this evidence, this population under the A&P criterion is most likely at very high extinction risk. The Oregon Native Fish Status report (ODFW 2005) listed the Molalla spring chinook population as a "fail" for abundance and a "fail" for productivity.

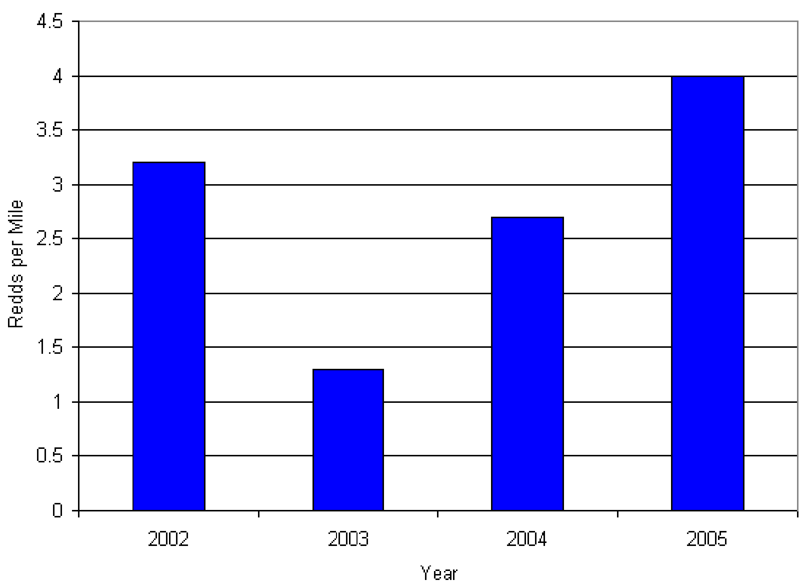


Figure 12: Spring chinook redds per mile in Molalla River surveys (Schroeder et al. 2005).

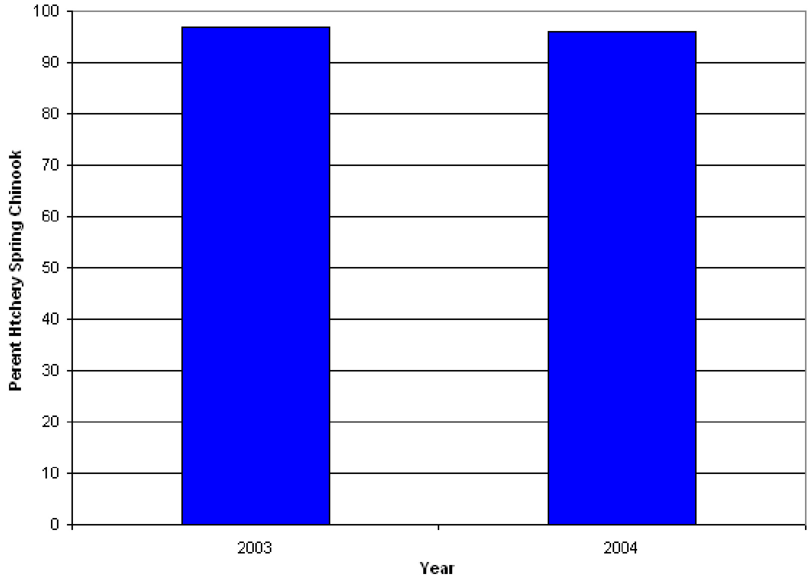


Figure 13: Percent hatchery origin spring chinook spawners in the Molalla River (Schroeder et al. 2005).

A&P – North Santiam

Recent redd survey results for the North Santiam are show in Figure 14 and Table 5. These indicate a relatively low redd density in this population. Of the fish that return nearly all are of hatchery origin (Figure 15). In addition there is a high estimated pre-spawning mortality (Figure 16). Although the pre-spawning mortality estimates are not considered very precise, it appears that more than half the females that return to the river die before spawning. Taken together, these data indicate little, if any, natural production of spring chinook in the North Santiam. Based on this evidence, this population under the A&P criterion is most likely at very high extinction risk. The Oregon Native Fish Status report (ODFW 2005) listed the North Santiam spring chinook population as a “fail” for abundance and a “fail” for productivity.

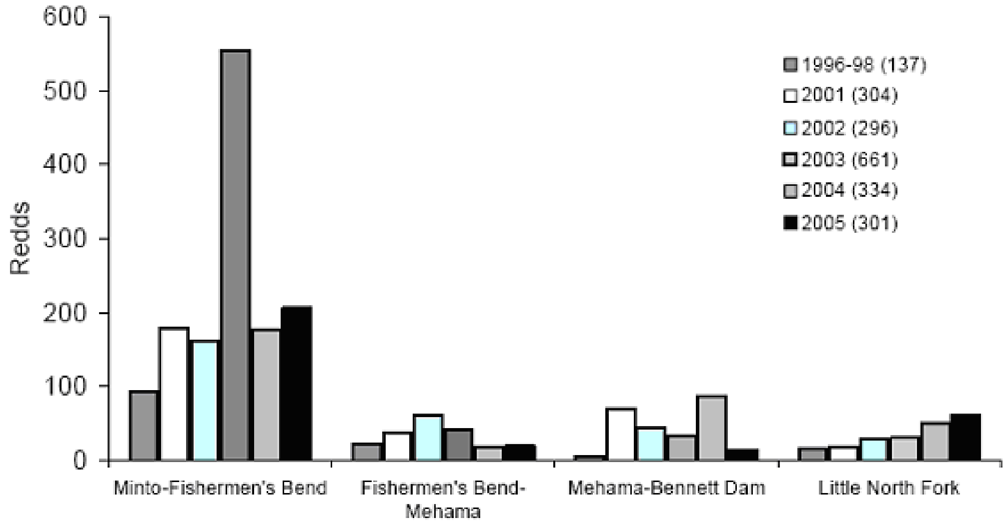


Figure 8. Spring Chinook salmon redds counted in four areas of the North Santiam basin upstream of Bennett dams, 1996–1998 average and 2001–2005. Total redds counted in the basin are in parentheses in the legend.

Figure 14: Number of Redds counted in sections of the North Santiam River. Copied from Schroeder et al. (2005).

Table 5: Redds per mile in sections of the North Santiam River. Copied from Schroeder et al. (2005).

Table 8. Summary of spawning surveys for spring Chinook salmon in the North Santiam River, 2005, and comparison to redd densities in 1996–2004. Spawning in areas downstream of Stayton may include some fall Chinook.

Survey section	Length (mi)	Counts		Redds/mi									
		Carcass	Redd	2005	2004	2003	2002	2001	2000	1999	1998	1997	1996
Minto–Fishermen's Bend	10.0	145	206	20.6	17.7	55.5	16.2	17.9	23.0[a]	15.6	11.8	8.5	7.8
Fishermen's Bend–Mehama	6.5	26	20	3.1	2.8	6.5	9.4	5.7	5.8	3.1	4.3	2.5	3.5
Mehama–Stayton Is.	7.0	23	14	2.0	12.6	4.7	6.1	10.0	**b**	--	0.6	0.9	1.0
Stayton Is.–Stayton	3.3	33	24	7.3	7.9	3.6	3.0	6.7	**b**	--	10.0	3.6	2.0
Stayton–Greens Bridge	13.7	7	4	0.3	0.2	0.1	0.4	0.1	--	0.0	0.4	1.1	0.1
Greens Br.–mouth	3.0	3	0	0.0	0.0	1.7	4.7	--	--	--	4.7	9.7	--
Little North Santiam	17.0	73	61	3.6[f]	3.0[e]	1.8[d]	1.8[c]	1.1[a]	1.3[a]	1.0	2.2[a]	0.6[a]	0.0

[a] *Corrected number.*
[b] *Data was recorded for Mehama–Stayton and density was 0.9 redds/mi.*
[c] *400 unclipped adult spring Chinook were released on August 20 and 30, September 5 and 6, 2002.*
[d] *268 unclipped adult spring Chinook were released in June (25th), July (9th,15th,22nd), August (25th), and September (2nd,4th).*
[e] *377 unclipped adult spring Chinook were released on July 9, August 19 and 27, and September 9.*
[f] *329 unclipped adult spring Chinook were released on July 27, August 30, and September 2, 6, 9, and 12.*

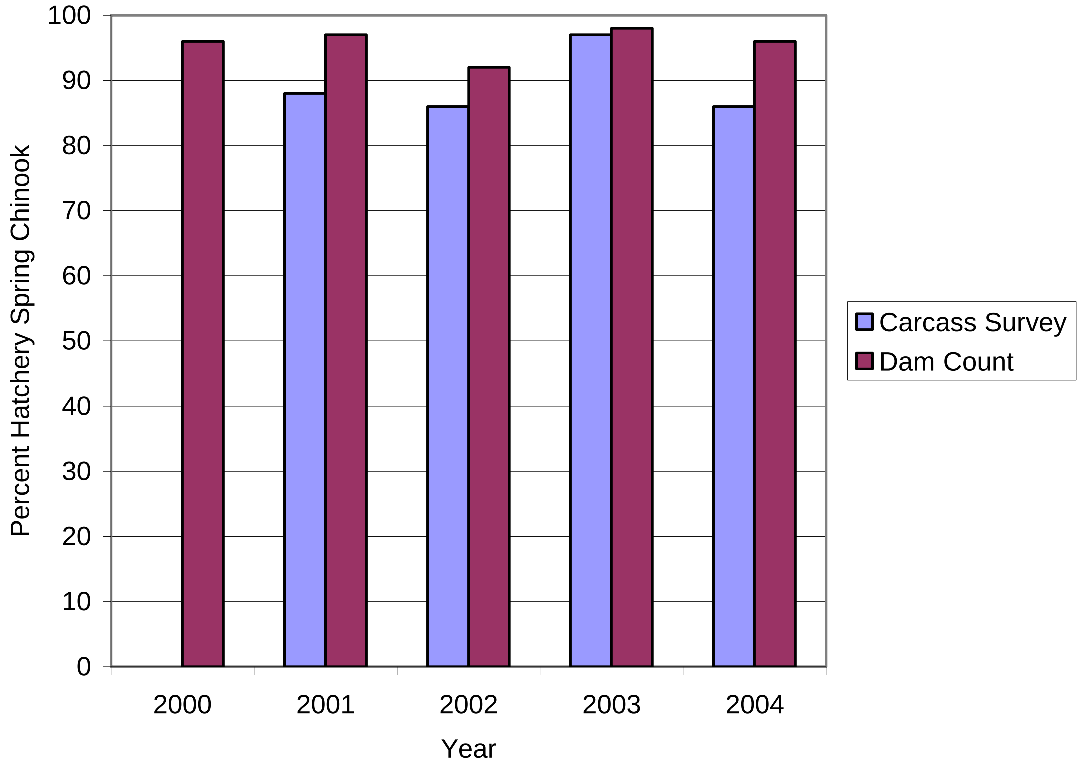


Figure 15: Percent of spring chinook spawners of hatchery origin in the North Santiam. The carcass survey is the region Minto to Bennet Dam, including Little North Santiam. The dam count is Bennet dam trap (Schroeder et al. 2005).

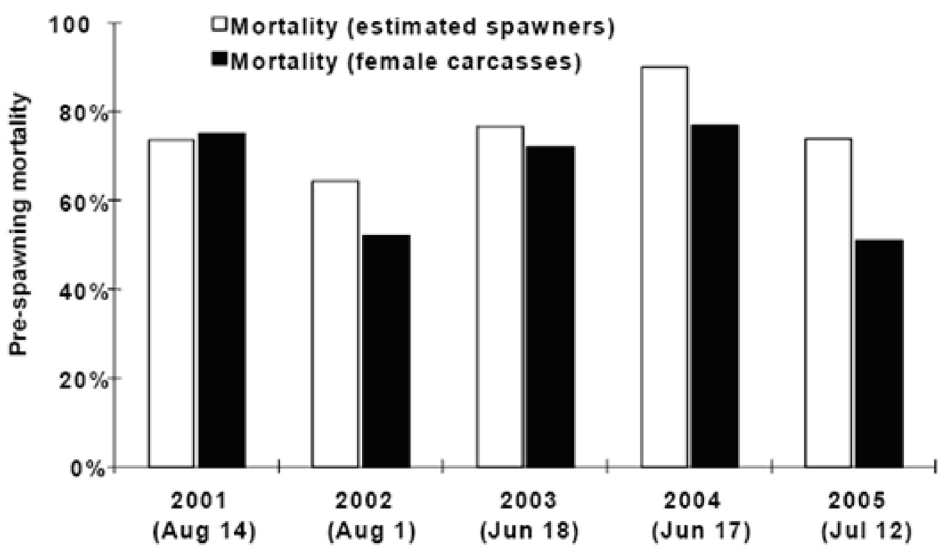


Figure 16: Pre-spawning mortality estimates for the North Santiam River based on two different estimation methods. Copied from Figure 17 in Schroeder et al. (2005).

A&P – South Santiam

Recent redd survey results for the South Santiam are show in Figure 14 and Table 6. These indicate a relatively low redd density for most of the system, but the abundance is higher than in the North Santiam. However, of the fish that return nearly all are of hatchery origin (Figure 18). In addition, estimates for pre-spawning mortality were quite high (Figure 19), although levels in the South Santiam appear lower than in the North Santiam. Taken together, particularly when considering the hatchery fraction, these data indicate little, if any, natural production of spring chinook in the South Santiam. Based on this evidence, this population under the A&P criterion is most likely at very high extinction risk. The Oregon Native Fish Status report (ODFW 2005) listed the South Santiam spring chinook population as a "fail" for abundance and a "fail" for productivity.

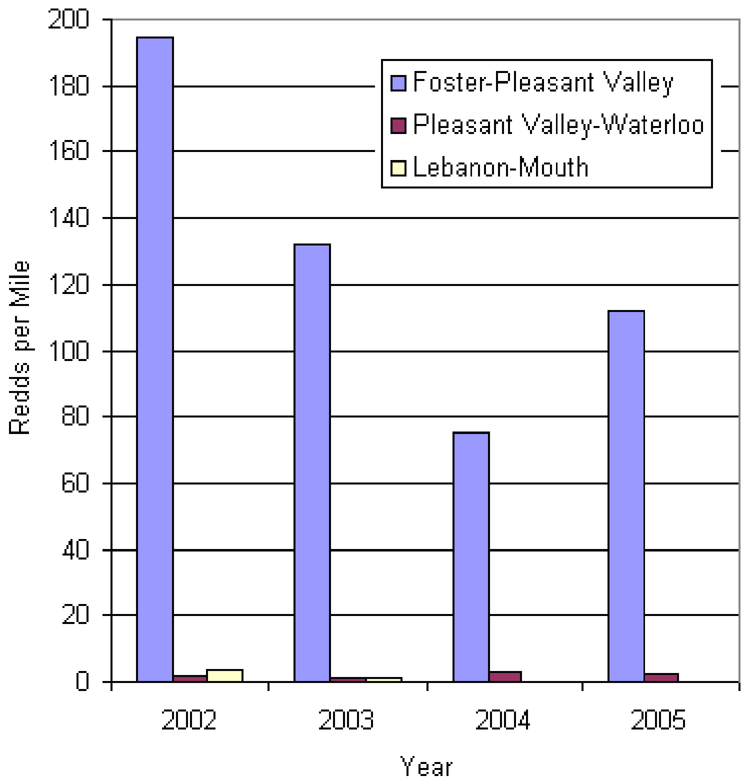


Figure 17: Redds per mile of spring chinook in sections of the South Santiam River. Lengths of the sections are Foster-Pleasant Valley = 4.5 miles, Pleasant Valley-Waterloo = 10.5 miles, and Lebanon-Mouth = 20 miles.

Table 6: Table showing spawning survey results for South Santiam spring chinook. Copied from Schroeder et al. (2005).

Table 14. Summary of Chinook salmon spawning surveys in the Middle Fork Willamette, South Santiam, and Molalla basins, 2005.

River, section	Length (mi)	Carcasses: Non fin-clipped[a]	Carcasses: Fin-clipped	Redds	Redds/mi 2005	2004	2003	2002	1998
Middle Fork Willamette									
Dexter–Jasper	9.0	8	37	9	1.0	1.0	1.5	7.1	1.1
Fall Creek (above reservoir)	16.0	12	c	130	8.1	12.9	6.1	12.9	--
South Santiam									
Foster–Pleasant Valley	4.5	124	401	507	112.7	75.1	132.0	194.4	36.0
Pleasant Valley–Waterloo	10.5	14	68	23	2.2	3.3	1.5	1.8	1.8
Lebanon–mouth	20.0	1	6	--	--	0.2	1.0	3.4	2.9
Molalla									
Horse Cr–Pine Cr[b]	6.2	4	19	25	4.0	2.7	1.3	3.2	--

[a] *Otoliths have not yet been read to determine the proportion of wild and hatchery fish.*
[b] *A segment of the Haybarn Cr–Trout Cr section of which we surveyed 16.1, 11.5, and 16.3 mi in 2004, 2003, and 2002, respectively.*
[c] *No fin-clipped fish were processed.*

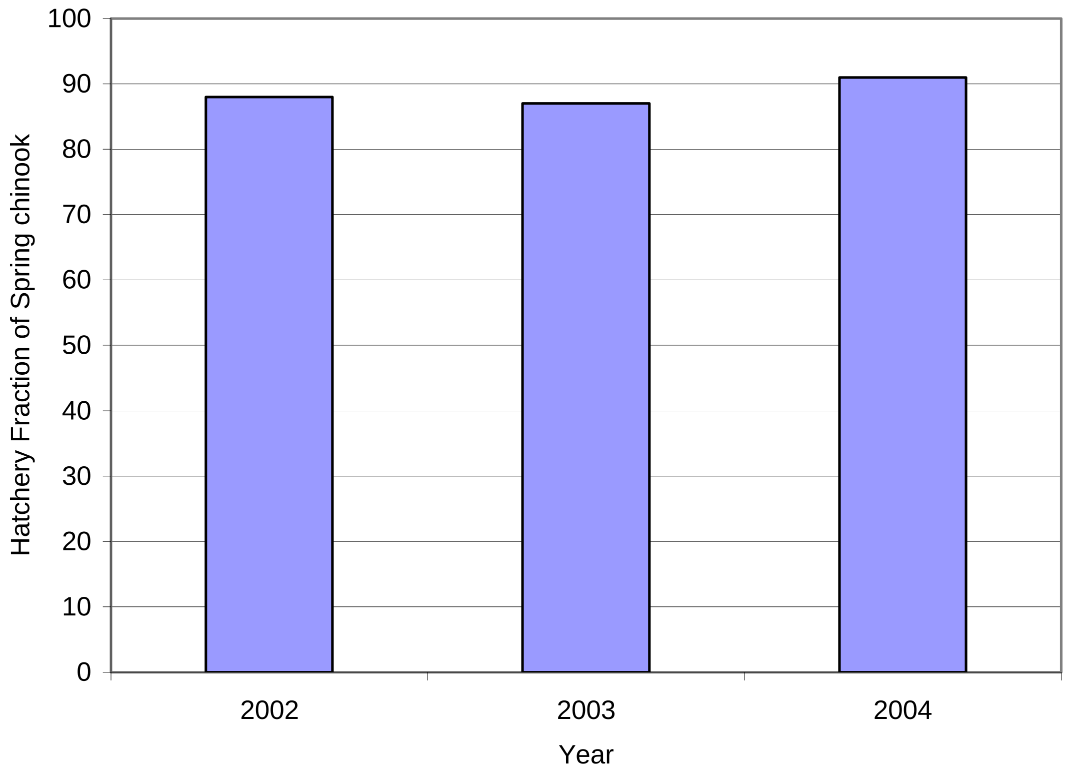


Figure 18: Percent of spring chinook spawners of hatchery origin in the South Santiam (Schroeder et al. 2005). Based on carcass recoveries in the area from Foster to Waterloo.

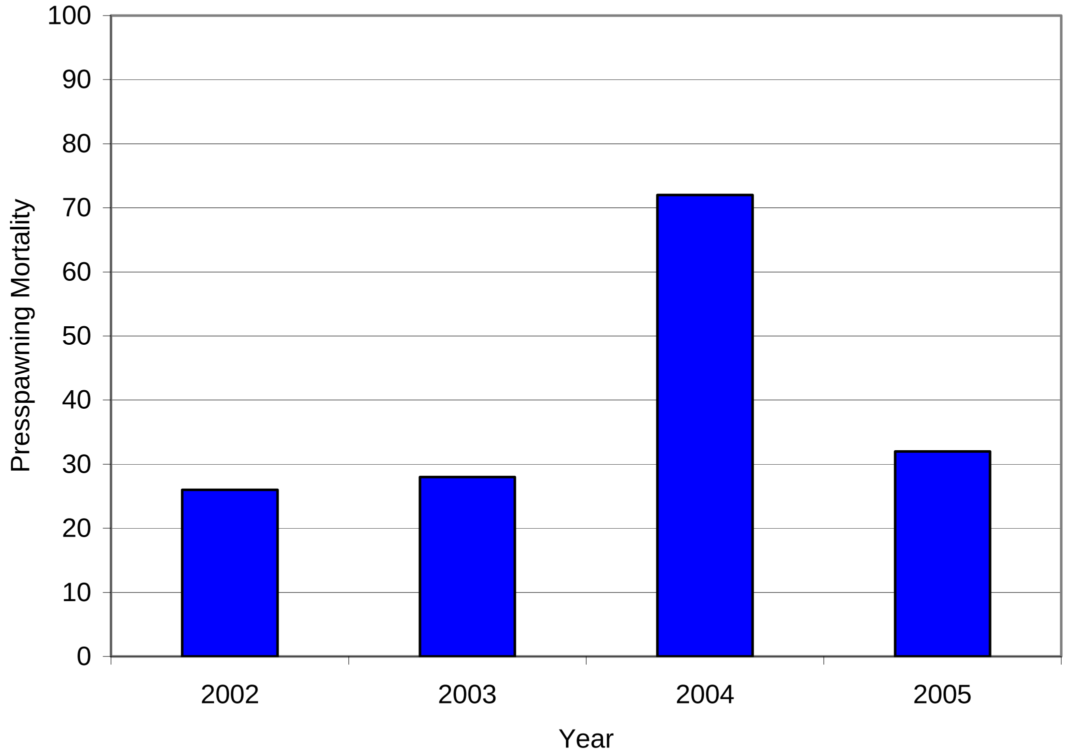


Figure 19: Pre-spawning mortality estimates for the South Santiam River (Schroeder et al. 2005).

A&P – Calapooia

Spring chinook surveys were conducted in 2002 and 2003, with the finding of 16 redds in 2002 and 2 redds in 2003 (Schroeder et al. 2005). In 2003, about 200 adult hatchery origin spring chinook were released into the Calapooia (Schroeder et al. 2003). These hatchery fish are likely responsible for producing the 2 redds observed. Of 48 carcasses surveyed in 2003, 43 (90%) were fin clipped as hatchery fish; the origin of the other 5 fish was unknown, as not all hatchery origin fish are clearly fin clipped (Schroeder et al. 2003). A survey of 27 female carcasses in the Calapooia in 2003 found 100% pre-spawning mortality (Schroeder and Kenaston 2004). The data indicate there is little or no natural production of spring chinook in the Calapooia and we considered the population to be extirpated or nearly so. The Oregon Native Fish Status report (ODFW 2005) listed the Calapooia spring chinook population as a “fail” for abundance and a “fail” for productivity.

A&P – McKenzie

A time series of abundance sufficient for quantitative analysis is available for the Clackamas spring run population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 20 to Figure 26 and in Table 7 to Table 10. The population long-term geometric mean natural origin spawners is relatively high (>1,500), which is in the very low risk minimum abundance threshold category (**Error! Reference source not found.**). The proportion of hatchery fish in recent years has averaged 35%, making it difficult to obtain a precise estimate of population productivity for wild fish. The pre-harvest viability curve analysis suggests that the population is most likely in the high to moderate risk category. The CAPM and PopCycle modeling suggests that the population is most likely in the moderate risk category, with a CRT risk estimates of 11% and 8% in 100 years, respectively. The escapement viability curve suggests that a population experiencing the pattern of harvest that occurred over the available time series (average mortality rate = 0.44) would be in high or very high risk category. There is considerable uncertainty about the level of pre-spawning mortality in the basin, but it may be significant (Figure 27). The Oregon Native Fish Status report (ODFW 2005) listed the North Santiam spring chinook population as a "pass" for abundance and a "pass" for productivity.

Taken together, the data suggest that with respect to the A&P criterion the most probable classification for this population is the moderate extinction risk category. However, given the uncertainty associated with the analysis, there is a small possibility that the risk classification could be very high or very low.

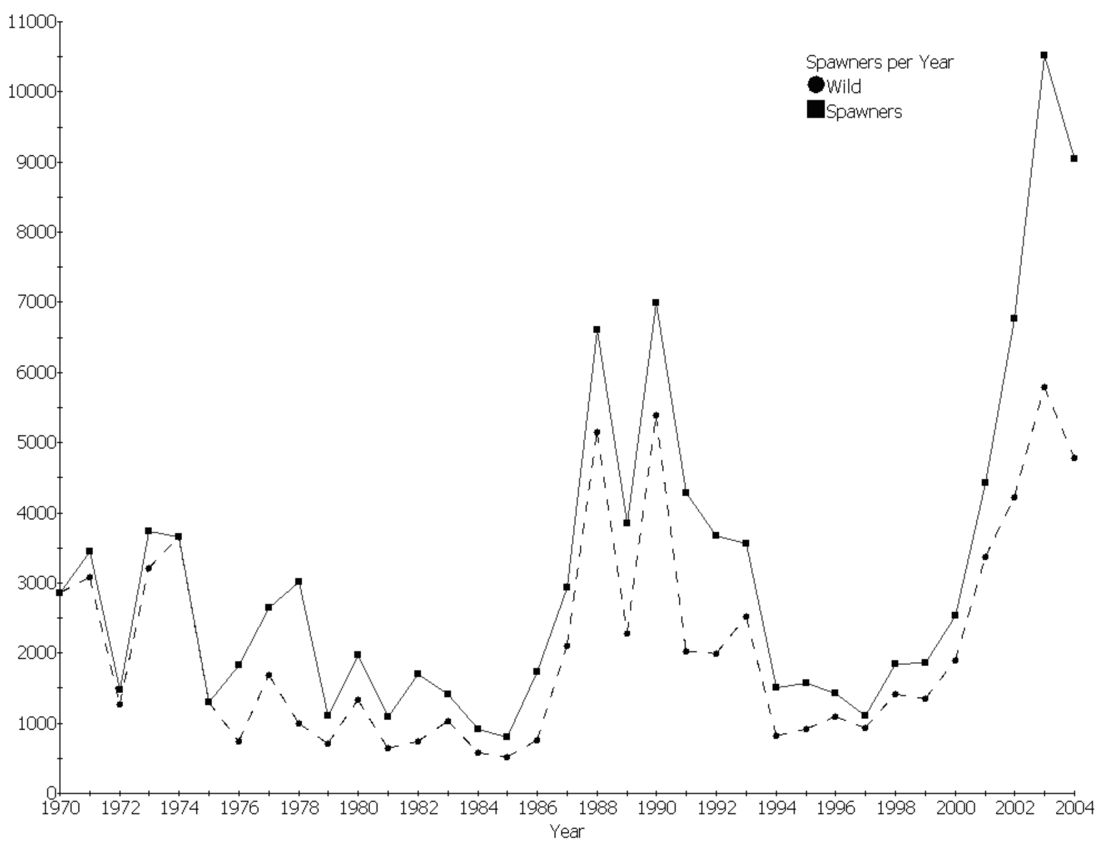


Figure 20: McKenzie spring chinook abundance.

Figure 21: McKenzie spring chinook hatchery fraction.

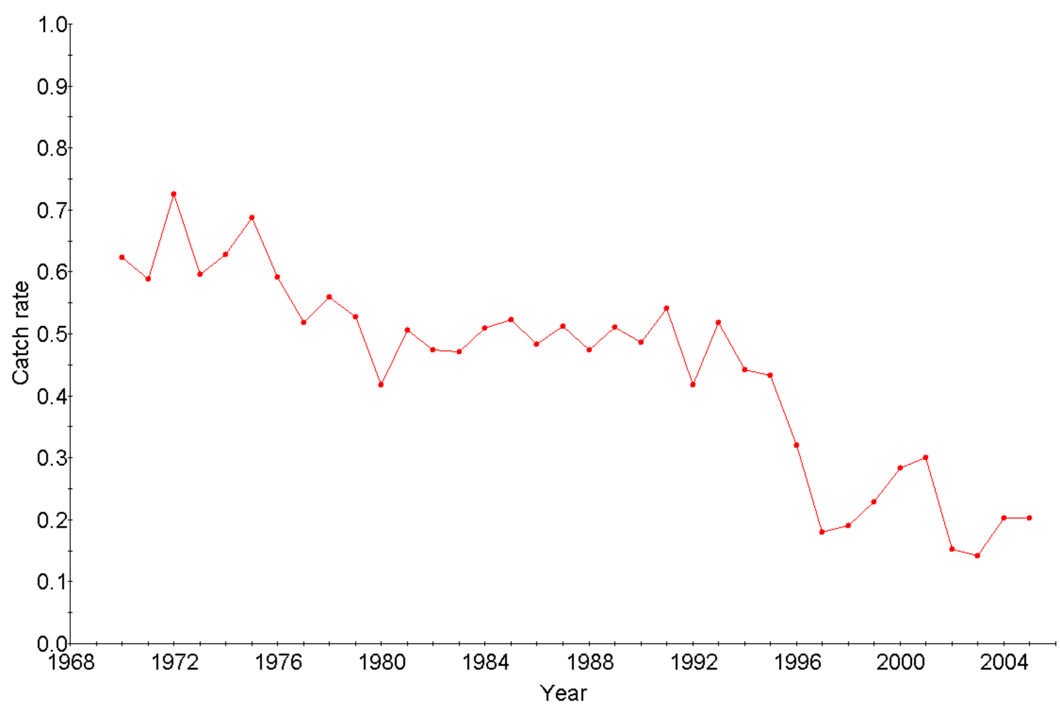


Figure 22: McKenzie spring chinook harvest rate

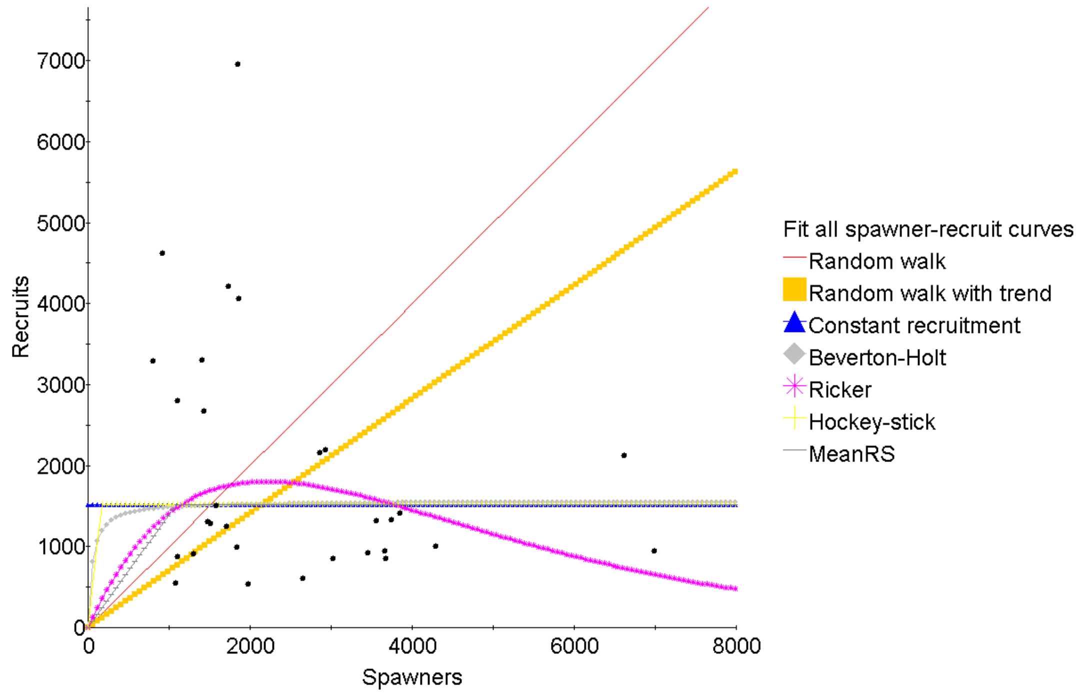


Figure 23: McKenzie spring chinook escapement recruitment functions.

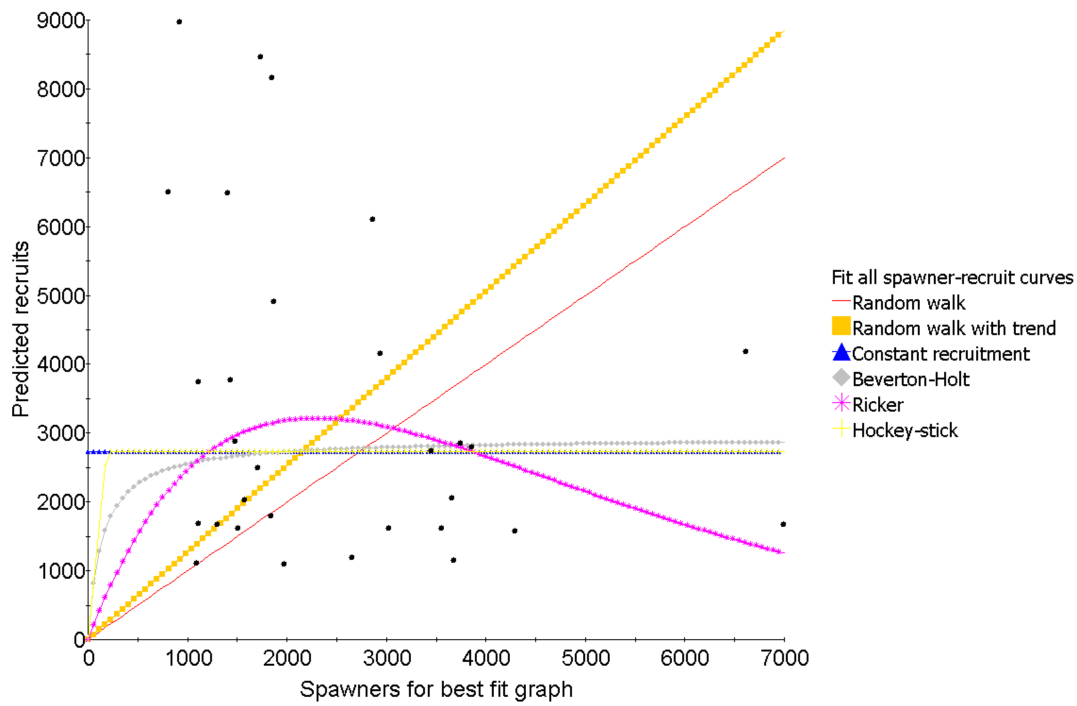


Figure 24: McKenzie spring chinook pre-harvest recruitment functions.

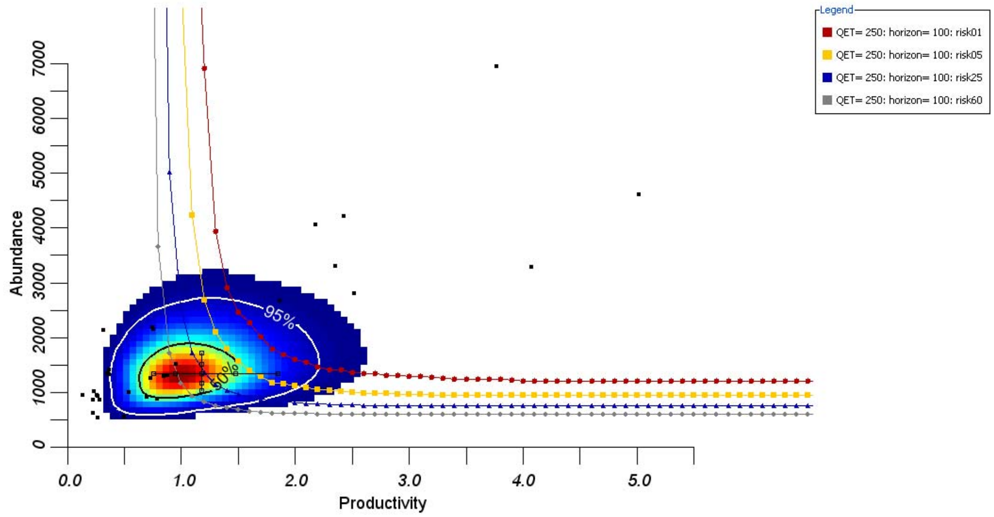


Figure 25: McKenzie spring chinook escapement viability curve.

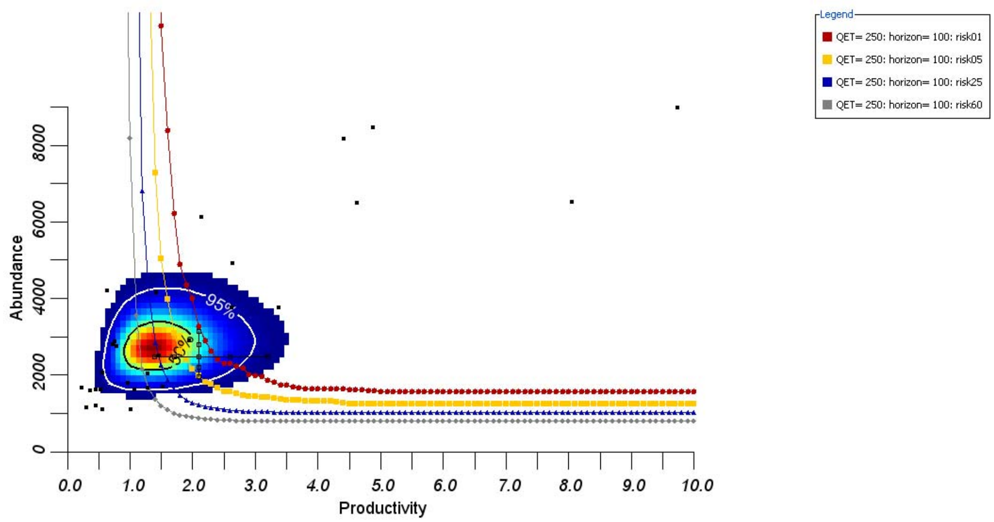


Figure 26: McKenzie spring chinook pre-harvest viability curve.

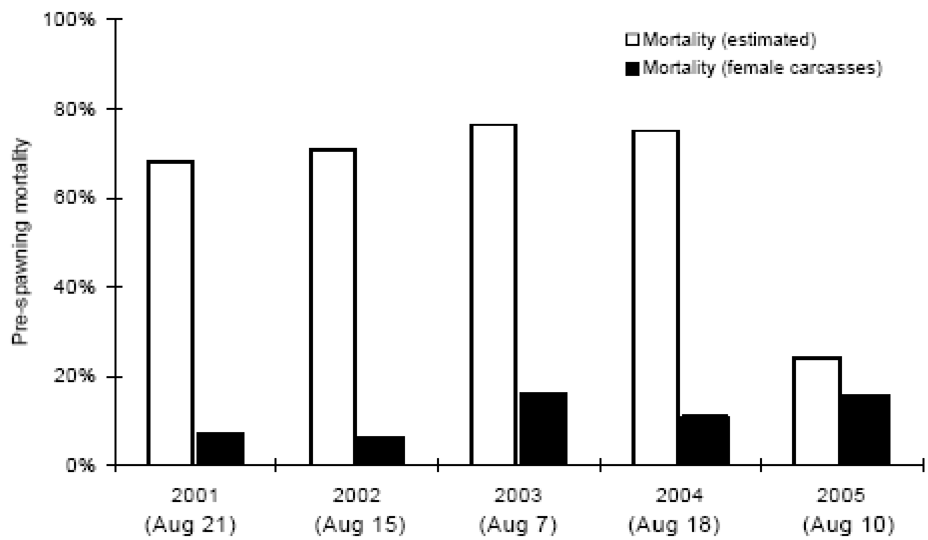


Figure 19. Pre-spawning mortality (%) of Chinook salmon in the McKenzie Basin estimated from recovery of female carcasses that died before spawning, and from number of successful spawners upstream of Leaburg Dam (from redd counts) to number of potential spawners (from counts of fish passing Leaburg Dam), 2001–2005. Starting dates of carcass surveys for each year are given in parentheses.

Figure 27: Estimates of pre-spawning mortality in the McKenzie River based two different methods. Copied from Schoerder et al. 2005. Schoerder et al. express more confidence in the carcass survey than the dam count method, but the exact reason for the discrepancy is unresolved.

Table 7: McKenzie spring chinook summary statistics. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1970-2005	1990-2005	1970-2005	1990-2005
Length of Time Series	36	16	36	16
Geometric Mean Natural Origin Spawner Abundance	1655 (1305-2099)	2104 (1484-2983)	NA	NA
Geometric Mean Recruit Abundance	1521 (1182-1957)	1835 (1113-3026)	2730 (2142-3479)	2491 (1586-3912)
Lambda	0.927 (0.761-1.129)	0.944 (0.517-1.722)	1.041 (0.858-1.264)	0.992 (0.549-1.793)
Trend in Log Abundance	1.017 (0.994-1.04)	1.047 (0.972-1.126)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.705 (0.485-1.024)	0.782 (0.339-1.802)	2.223 (1.47-3.362)	1.061 (0.488-2.307)
Geometric Mean Recruits per Spawner (broods < median spawner abudance)	1.307 (0.848-2.016)	1.775 (0.969-3.25)	1.017 (0.994-1.04)	2.289 (1.283-4.082)
Average Hatchery Fraction	0.318	0.329	NA	NA
Average Harvest Rate	0.444	0.315	NA	NA
CAPM median extinction risk probability (5th and 95th percentiles in parenthesis)	NA	NA	0.125 (0.030-0.355)	NA
PopCycle extinction risk	NA	NA	0.08	NA

Table 8: Escapement recruitment parameter estimates and relative AIC values for McKenzie spring chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.04 (0.88-1.36)	25.1
Random walk with trend	0.7 (0.54-1)	NA	0.98 (0.84-1.32)	23.6
Constant recruitment	NA	1521 (1255-1922)	0.66 (0.57-0.88)	0
Beverton-Holt	29.76 (5.38-28.87)	1568 (1301-2115)	0.67 (0.57-0.9)	2.4
Ricker	2.22 (1.47-3.7)	1803 (1512-2462)	0.7 (0.61-0.95)	4.9
Hockey-stick	9.3 (2.79-28.6)	1521 (1245-1915)	0.66 (0.57-0.89)	2
MeanRS	1.4 (1.02-1.95)	1521 (1247-1859)	0.49 (0.31-0.64)	13

Table 9: Pre-harvest recruitment parameter estimates and relative AIC values for McKenzie spring chinook. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.98 (0.82-1.27)	23.6
Random walk with trend	1.26 (0.96-1.78)	NA	0.95 (0.81-1.26)	23.8
Constant recruitment	NA	2733 (2262-3410)	0.64 (0.55-0.85)	0
Beverton-Holt	29.96 (7.05-29.05)	2842 (2400-3923)	0.65 (0.56-0.87)	2.7
Ricker	3.81 (2.53-6.22)	3218 (2731-4359)	0.68 (0.59-0.93)	5.6
Hockey-stick	6.24 (4-28.59)	2729 (2251-3403)	0.64 (0.54-0.85)	2
MeanRS	2.41 (1.76-3.31)	2730 (2259-3318)	0.46 (0.3-0.59)	15.5

Table 10: McKenzie spring chinook CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in 'Extirpated or nearly so' category	0.656	0.804	0.997
Probability the population is above 'Moderate risk of extinction' category	0.428	0.606	0.835
Probability the population is above 'Viable' category	0.193	0.333	0.103
Probability the population is above 'Very low risk of extinction' category	0.062	0.125	0.002

A&P – Middle Fork Willamette

Recent redd survey results for the Middle Fork Willamette River are show in Figure 28. These indicate a relatively low redd density in this population. Of the fish that return nearly all are of hatchery origin (Figure 29). In addition there is a high estimated pre-spawning mortality (Figure 30). Although the pre-spawning mortality estimates are not considered very precise, it appears that over 80% the females that return to the river die before spawning; second only to the Calapooia population for the highest spring chinook pre-spawn mortality in the Willamette. Taken together, these data indicate little, if any, natural production of spring chinook in the Middle Fork Willamette. Based on this evidence, this population under the A&P criterion is most likely at very high extinction risk. The Oregon Native Fish Status report (ODFW 2005) listed the "Upper Willamette" spring chinook population (contains the Middle Fork population plus Mosby Creek) as a "fail" for abundance and a "fail" for productivity.

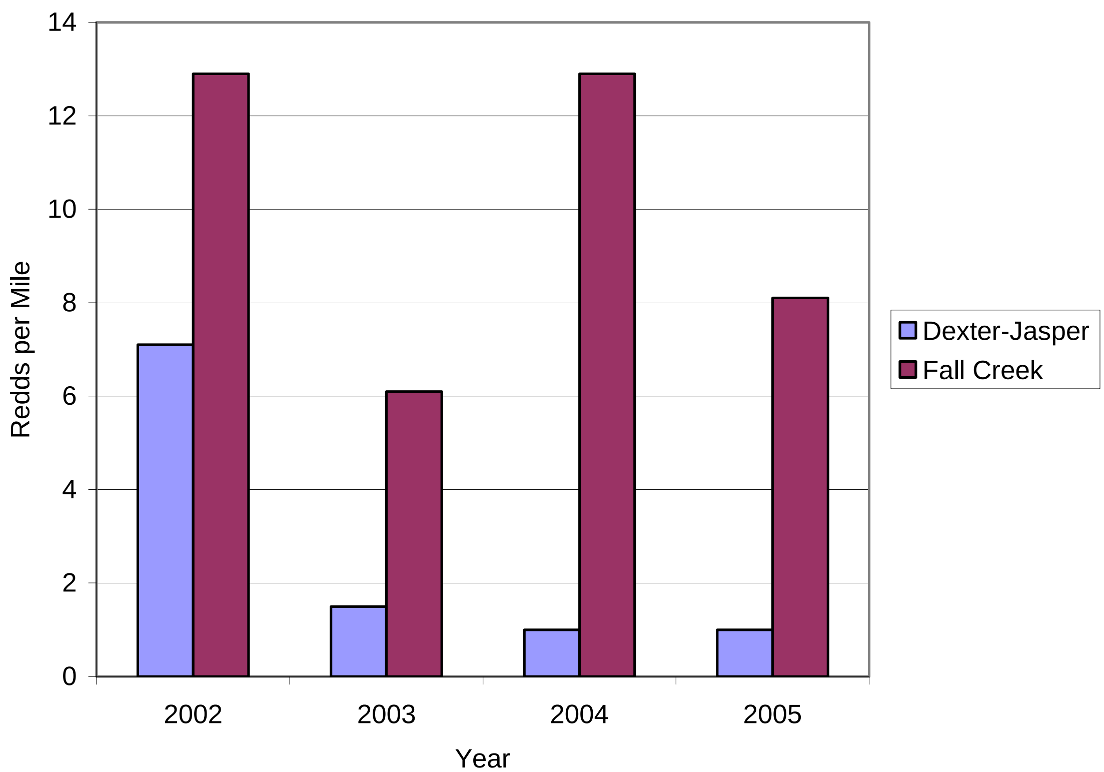


Figure 28: Redds per mile of spring chinook in sections of the Middle Fork Willamette (Schoeder et al. 2005). The Dexter-Jasper survey was 9.0 miles and the Fall Creek survey was 16 miles.

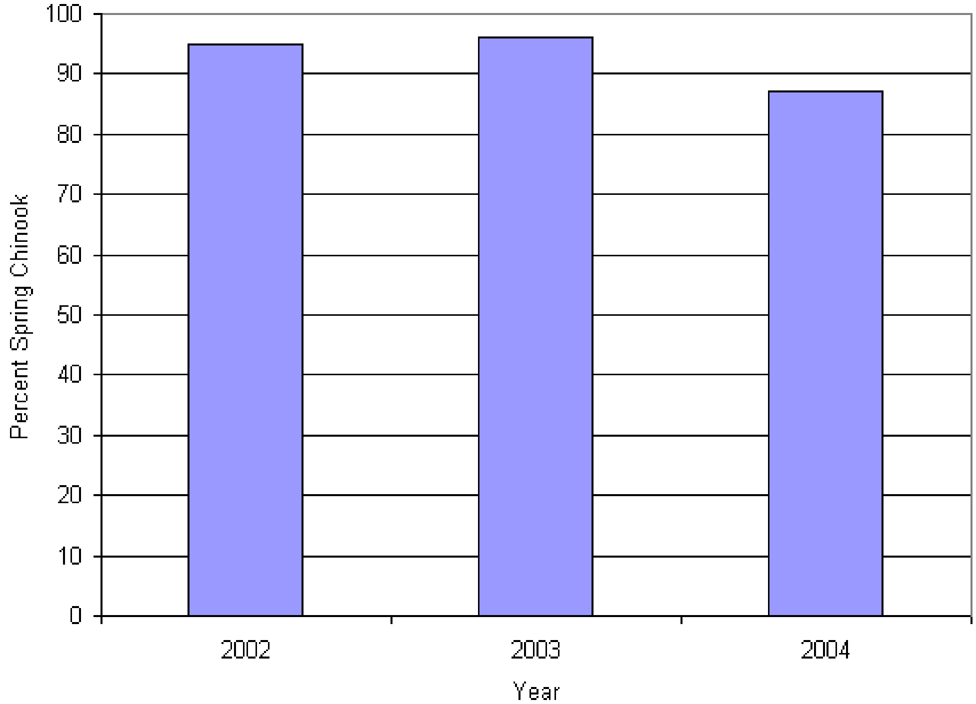


Figure 29: Percent of spring chinook spawners of hatchery origin in the Middle Fork Willamette between Dexter and Jasper and Fall Creek (Schroeder et al. 2005).

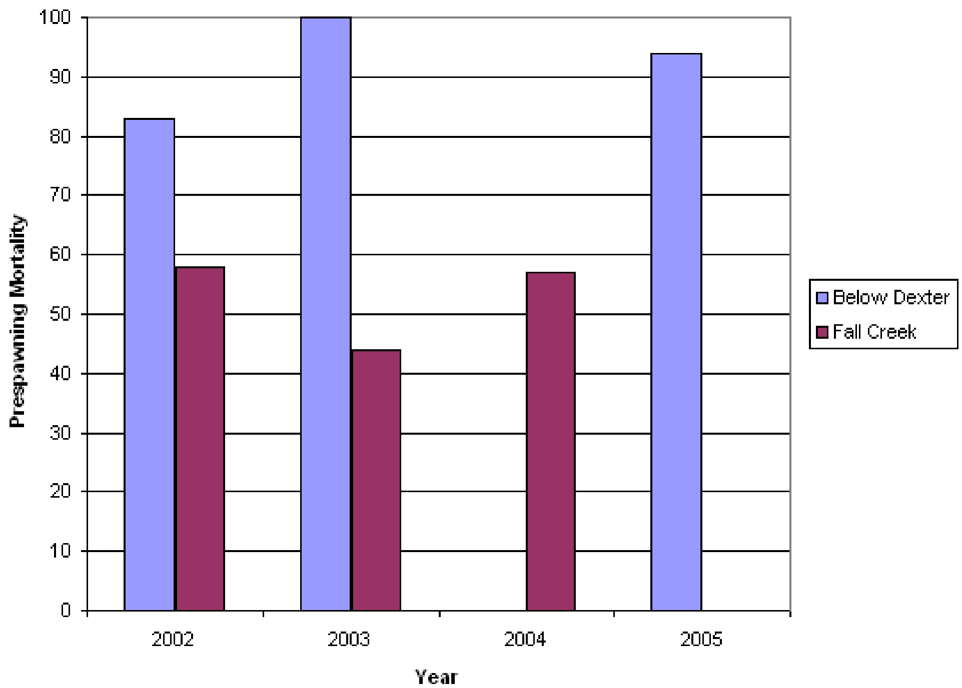


Figure 30: Pre-spawning mortality estimates for spring chinook in the Middle Fork Willamette (Schroeder et al. 2005).

A&P – Criterion Summary

The abundance and productivity status evaluation results are shown in Figure 31. The Molalla, North Santiam, South Santiam, Calapooia and Middle Fork Willamette populations are all considered at very high risk or nearly extirpated. Lengthy time series of abundance for these populations are not available, but recent survey data suggest low numbers of redds, an extremely high proportion of hatchery fish (i.e., very few wild fish) and unsustainably high pre-spawning mortality rates. Based on these findings we conclude that very little natural production is taking place for these populations. In contrast there is evidence that natural production of spring chinook is occurring for the McKenzie and Clackamas populations.

In terms of the quantitative classifications for the abundance and productivity criterion, the most probable risk category for all but two of these populations was relatively certain and very high as illustrated by the diamonds in Figure 31. The exceptions are most probable classifications of 'low risk' for the Clackamas population and 'moderate risk' McKenzie population. However, for these two populations there is considerable amount of uncertainty in these conclusions as illustrated in Figure 31 by the height of the diamond symbols. It is possible (but not probable) that the conservation risk for these populations may be very low or high. However, regardless of this uncertainty, the UW ESU as a whole most likely belongs in the high risk category for this criterion. Five of the seven populations are at very high risk and the most probable risk classifications for the remaining two are 'low' and 'moderate'.

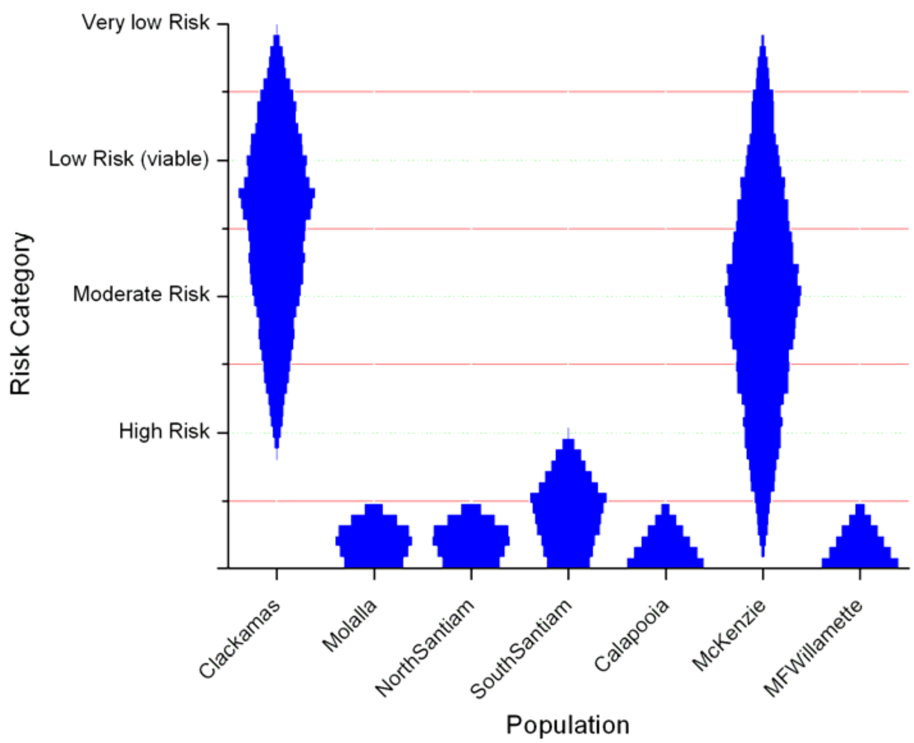


Figure 31: Upper Willamette spring chinook risk status summary based on evaluation of abundance and productivity only.

III. Spatial Structure

SS – Clackamas

Virtually the entire habitat accessible to spring chinook in the Clackamas River remains accessible today (Figure 32)(ODFW 2005). The upper Clackamas basin contains the historically-productive habitat for spring chinook and most of that habitat is of high quality today. Little spring chinook production was likely from lower basin streams where development has been extensive. A portion of the historical rearing habitat for spring chinook has been inundated by construction of three Clackamas mainstem dams – the significance of related effects on spatial diversity is unclear because reservoirs now provide significant over-winter habitat. The watershed score was reduced to address a likely loss in spatial diversity related to habitat declines in lower Clackamas, Willamette and Columbia mainstems and the estuary which may have affected the fall migrant life history pattern of this species.

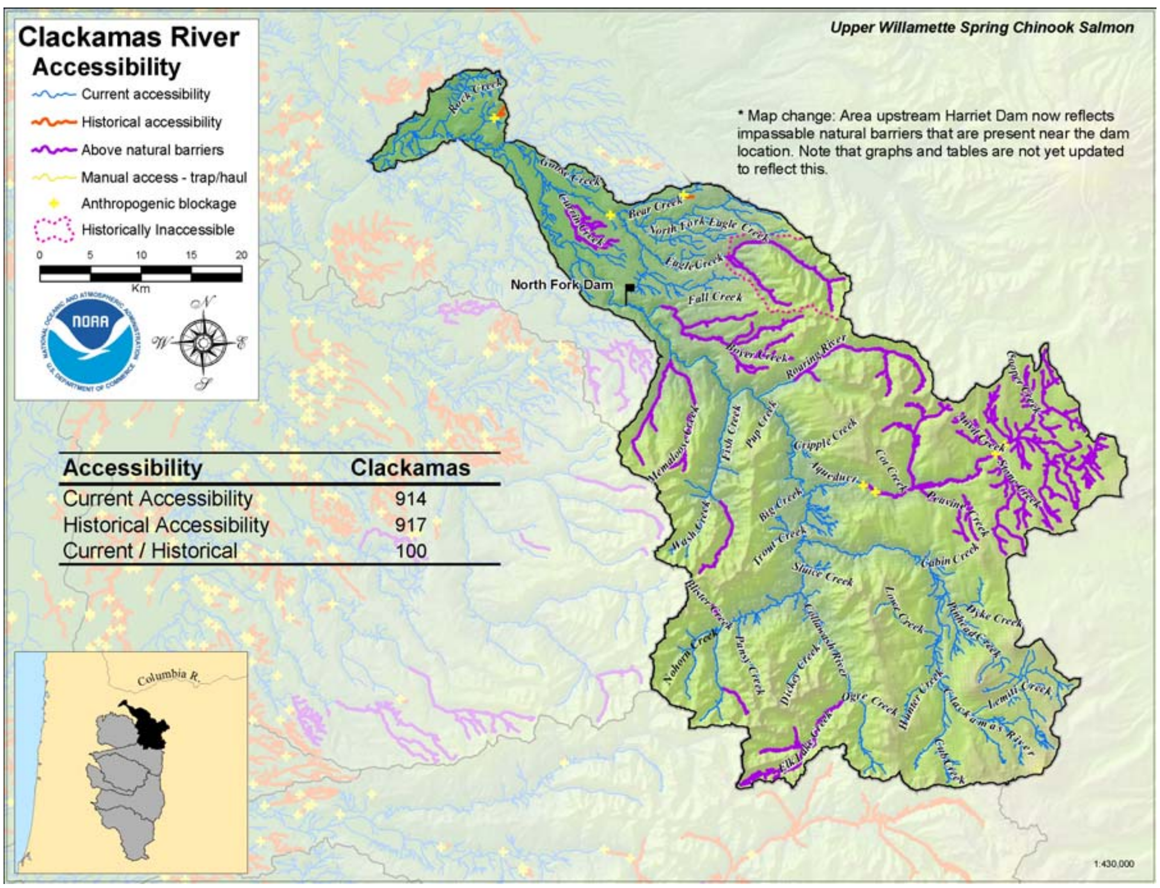


Figure 32: Clackamas River spring chinook current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Molalla

Land use and road building has limited access of anadromous fish to many higher order tributaries in the Molalla system but no large mainstem fish barriers are present. On a stream mile basis this impairment is significant (Figure 33). However, historical spring chinook spawning and rearing areas were limited to mainstem areas that remain over 95% accessible (ODFW 2005). Habitat degradation due to land use has reduced water quality and the availability of suitable spawning habitat for spring chinook in the Molalla River. The combined effects of high accessibility in historically suitable habitats and habitat quality degradation in the sub-basin and downstream, result in a modified risk score.

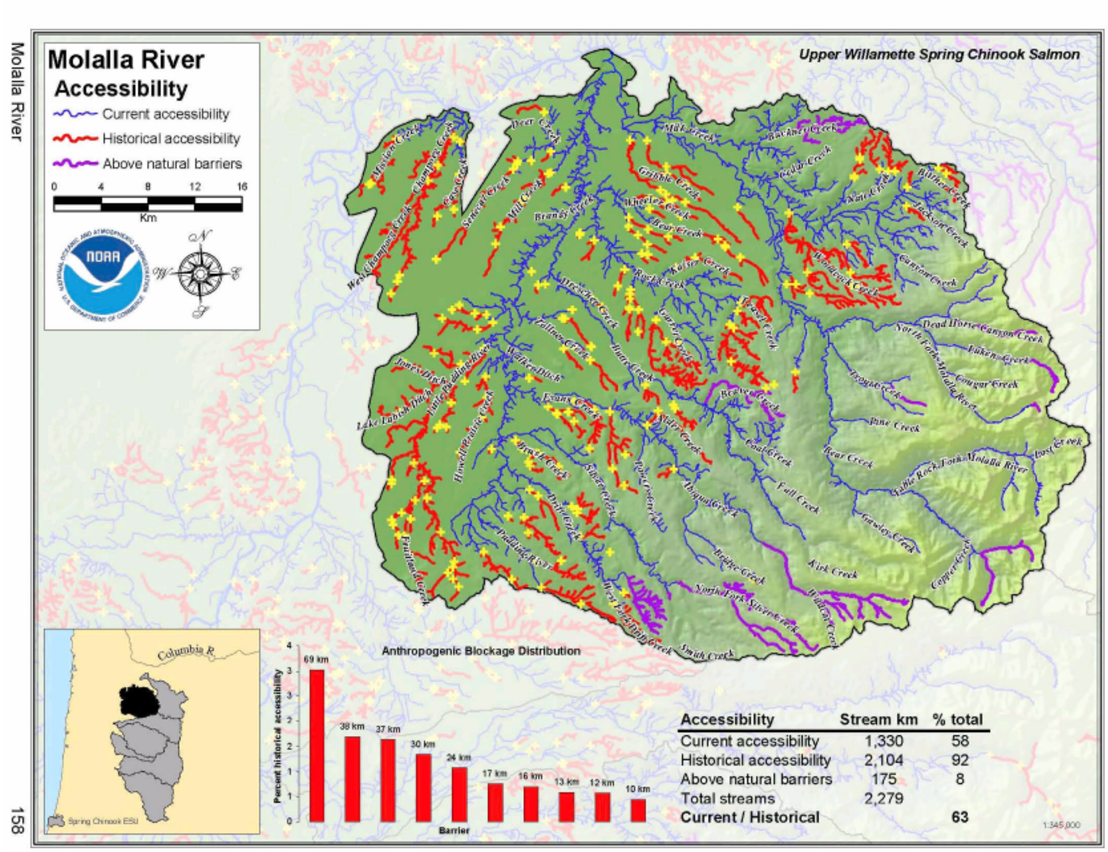


Figure 33: Molalla River spring chinook current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – North Santiam

Access to large portions of the historically productive spring chinook habitat has been blocked by the Detroit Reservoir (Figure 34). ODFW estimates that 42% of the historically-suitable for spring chinook is now inaccessible (ODFW 2005). Historically this area was the primary spring chinook production area for the North Santiam because the habitat is of such high quality. Much of the remaining accessible habitat is not well suited for spring chinook although some favorable reaches may still be found in the Little North Santiam River.

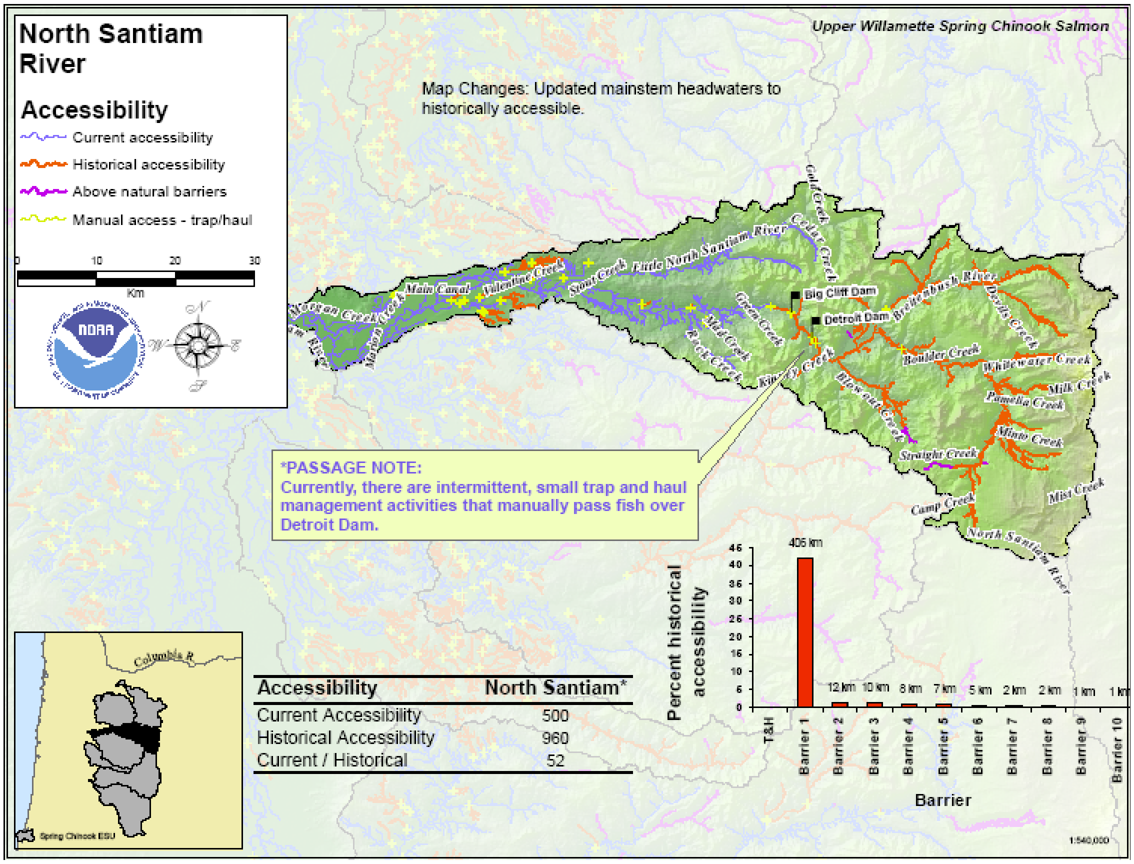


Figure 34: North Santiam River spring chinook current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – South Santiam

Access to large portions of the historically-productive spring chinook habitat have been blocked by Foster and Green Peter Dams, though there is currently and experimental trap and haul program at Foster Dam (Figure 35). ODFW estimates that 40% of the historically-suitable for spring chinook is now inaccessible (ODFW 2005). Like the North Santiam these blocked areas contained some of the best spring chinook habitat in the basin. ODFW (2005) estimates that historically 70% of the spring chinook production from this system originated from this now inaccessible portion of the watershed. The remaining habitat is not well suited for spring chinook. The watershed score for spatial structure was further reduced to account for relative poor habitat suitability in the remaining accessible habitat and in the Willamette and Columbia mainstems and the estuary.

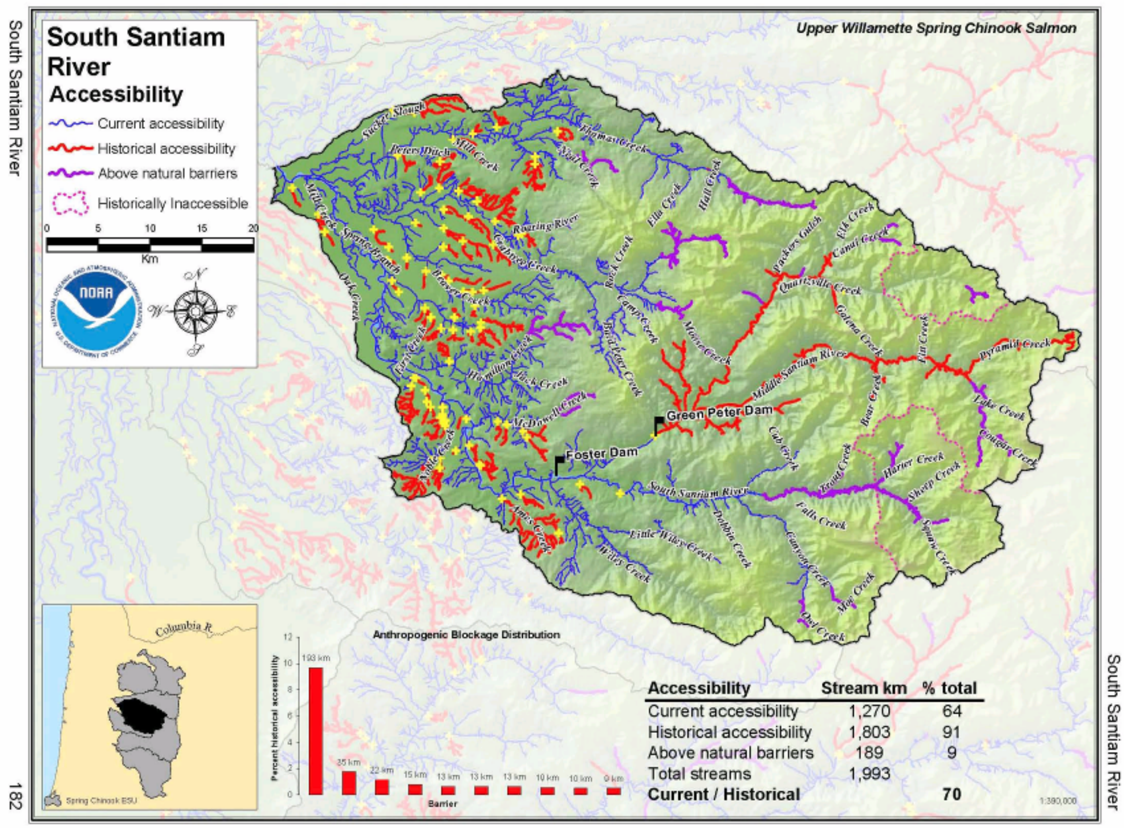


Figure 35: South Santiam River spring chinook current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Calapooia

Over half of the stream length historically accessible to spring chinook in the Calapooia is currently blocked (Figure 36). In addition, habitat degradation has substantially reduced the quality of remaining accessible habitat, making spatial structure a substantial source of risk in the Calapooia.

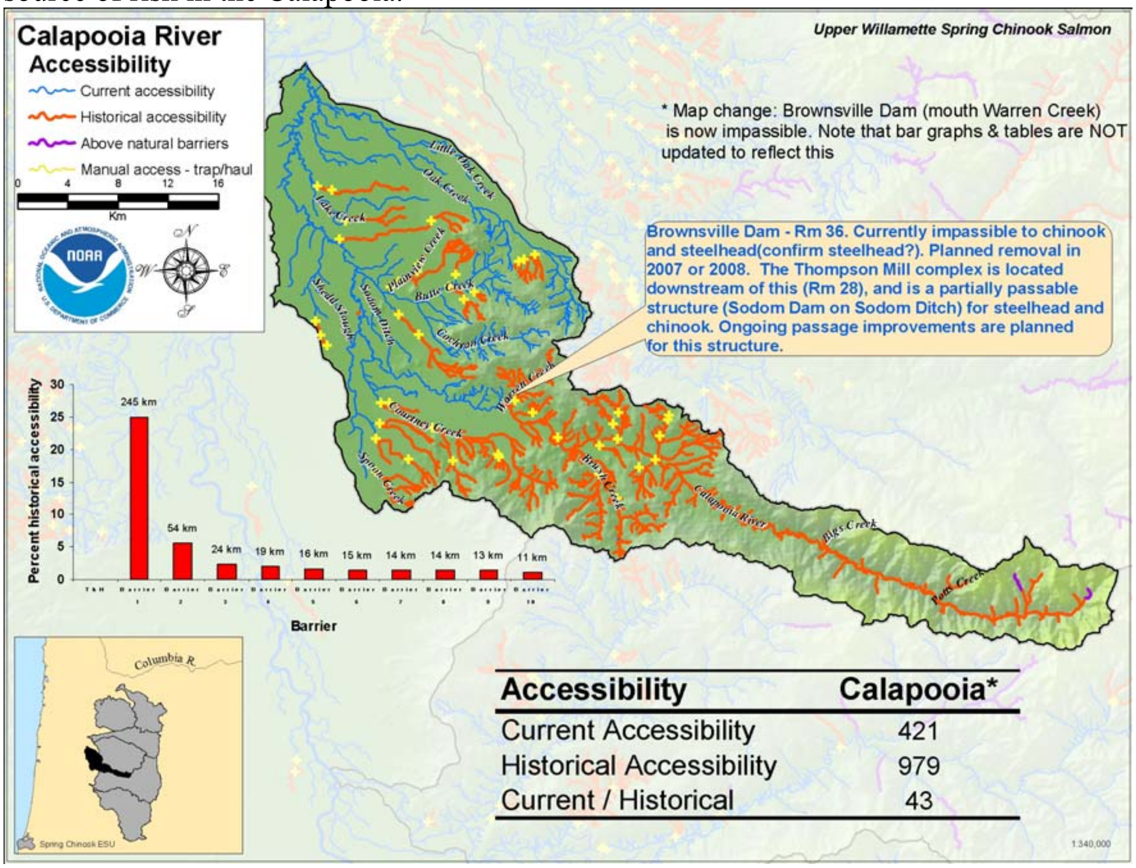


Figure 36: Calapooia River spring chinook current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use. (NOTE: The Brownsville Dam is not considered a barrier for steelhead.)

SS – McKenzie

Most of the historical spring chinook habitat in the McKenzie River remains accessible today (Figure 37) and this system supports the largest extant spring chinook population upstream of Willamette Falls (ODFW 2005). Historical habitats have been blocked on McKenzie River tributaries by the Cougar and Blue River dams. ODFW (2005) estimates that 16% of the historical habitat has been blocked on a stream mile basis and the accessibility analysis including higher order streams estimates a 25% loss (Maher et al. 2005). High quality habitats remain accessible in other parts of the system. The watershed score for spatial structure was reduced to account for losses in historically-significant rearing habitat in the upper Willamette mainstem.

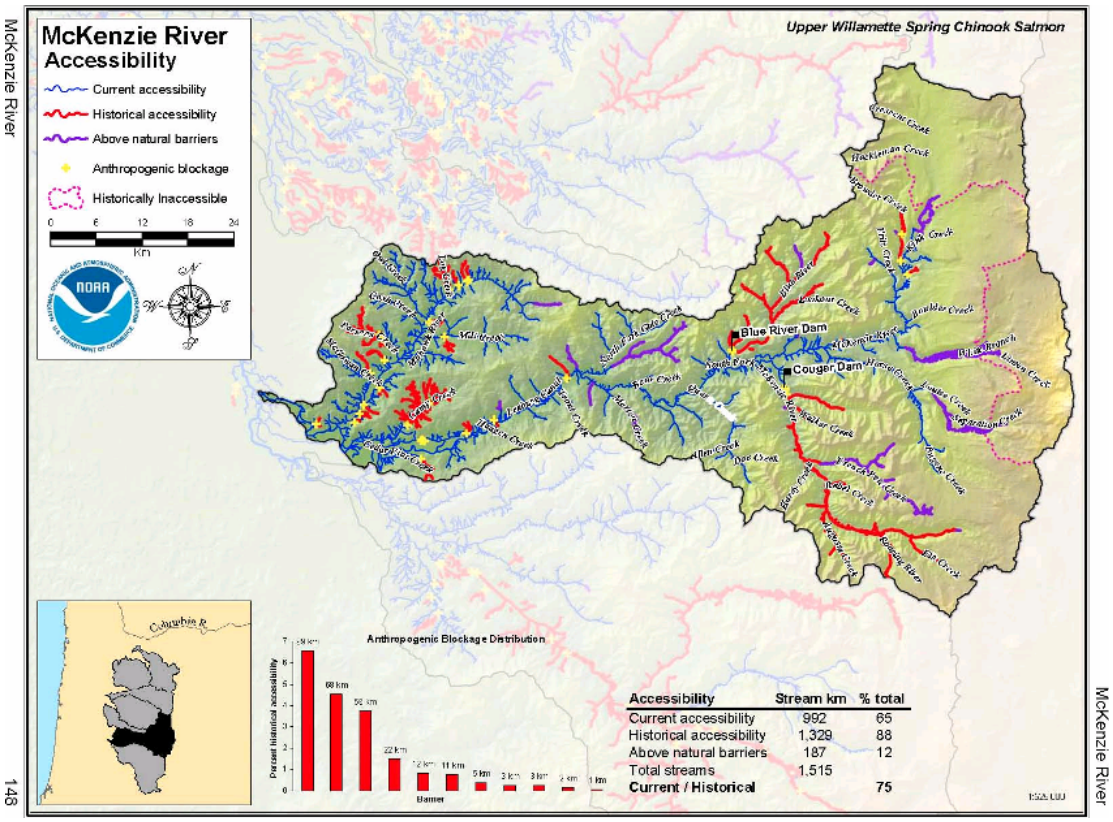


Figure 37: McKenzie River spring chinook current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Middle Fork Willamette

The majority of the historical spring chinook habitat in the Middle Fork Willamette has been blocked by dams (Figure 38). ODFW (2005) estimates that 57% of the historical habitat is no longer accessible, and that this habitat accounted for an even greater portion of the historical production. The remaining accessible habitats are not well suited to spring chinook production.

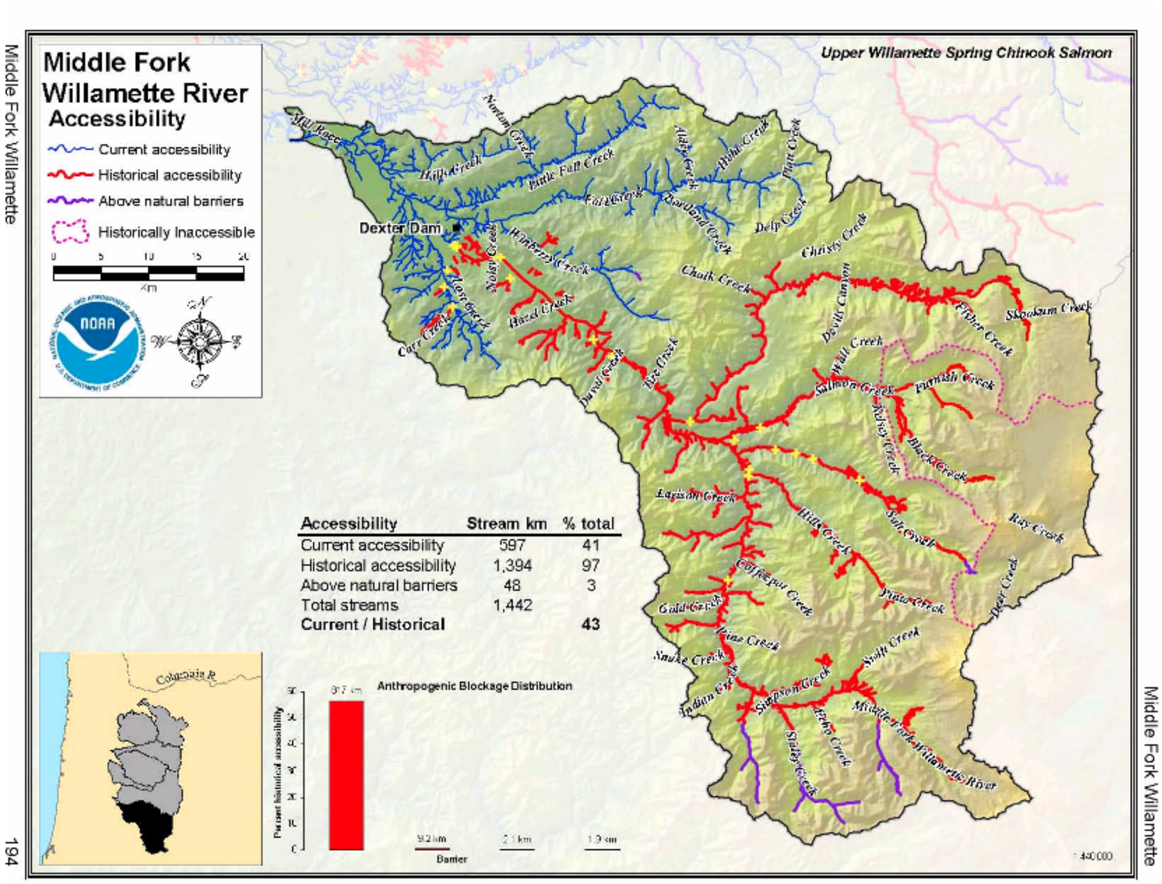


Figure 38: Middle Fork Willamette River spring chinook current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Criterion Summary

Except for the Clackamas population, the percentage of historically accessible habitat lost due to human activities (primarily dam construction) exceeds 25% for each population within this ESU (Figure 39). In the case of populations in the North Santiam Calapooia and Middle Fork Willamette, habitat loss has been particularly high (around 50%).

SS scores for each population were adjusted, where applicable, on the basis of two factors: 1) the suitability/quality of the blocked habitat with respect to chinook production and 2) the degree to which the remaining accessible habitat has been degraded from historical conditions. The adjustments and final SS scores for each population are presented in Table 11. For the SS criterion the most probable risk category for a majority of the populations was 'high' or 'very high' as evidenced by the SS rating in Table 11 and illustrated by the placement of the widest portion of the diamonds in Figure 40. The remaining three populations have a most probable risk classification of 'low' risk. However, when the uncertainty associated with these rating is considered, only one population (Clackamas) is clearly in the 'low' risk category. The other two populations (Molalla and McKenzie) the three populations may in fact belong in the 'moderate' risk category. Given the wide range among the populations in terms of scores for this criterion, it is difficult to draw conclusions as to an overall ESU rating. However, we conclude the most probable ESU risk classification for the SS criterion would be 'high'.

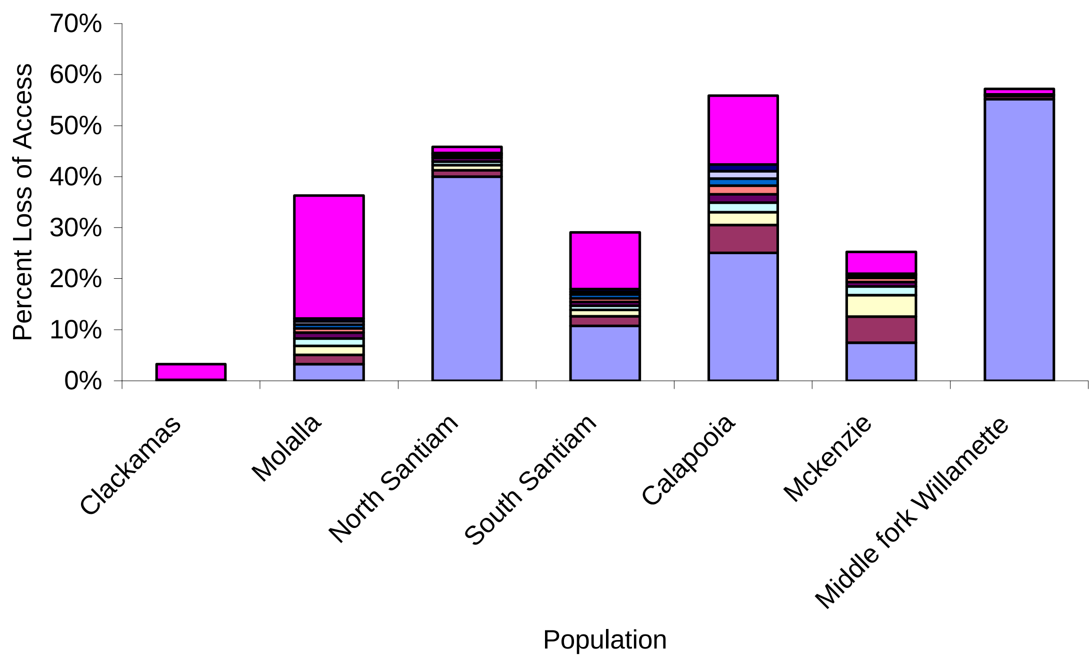


Figure 39: Percent loss in Upper Willamette spring chinook accessibility due to anthropogenic blockages (based on Maher et al. 2005). Each color represents a blockage ordered from largest to smallest (bottom-up). The top most blockages, for example the pink segment of the Calapooia bar are a collection of many smaller blockages. Note that the pool of smaller blockages can be greater than larger single blockages. These percentages are based on current (2007) accessibility estimates and may differ from the access maps above as described in the map figure legends.

Table 11: Spatial structure persistence category scores for UW chinook populations.

Population	Base Access Score	Adjustment for Large Single Blockage	Adjusted Access Score	SS Rating Considering: Access Score, Historical Use Distribution, and Habitat Degradation	Confidence in SS rating
Clackamas	4	no	4	3.5	M
Molalla	2	no	2	2.5	L
North Santiam	1	yes	0.5	0.5	H
South Santiam	2	no	2	1	M
Calapooia	1	yes	0.5	0.5	M
McKenzie	3	no	3	2.5	M
Middle Fork Willamette	1	yes	0.5	0.5	M

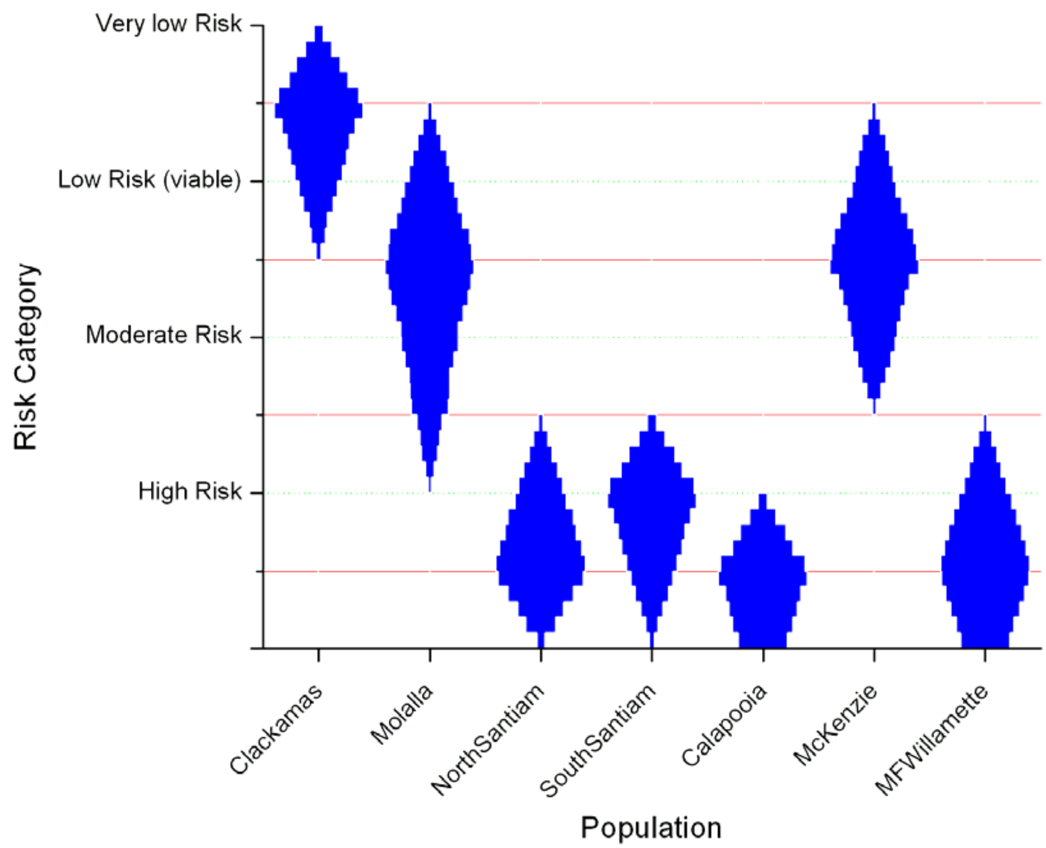


Figure 40: Upper Willamette spring chinook risk status summary based on evaluation of spatial structure only.

IV. Diversity

DV – Background and Overview

Historically, the Willamette River Basin provided sufficient spawning and rearing habitat for large numbers of spring-run chinook salmon. The predominant tributaries to the Willamette River that historically supported spring-run chinook salmon include the Molalla (RKm 58), Calapooia (RKm 192), Santiam (RKm 174), McKenzie (RKm 282), and Middle Fork Willamette Rivers (RKm 301)—all drain the Cascade Range to the east (Mattson 1948, Nicholas 1995). There are no direct estimates of the size of the chinook salmon runs in the Willamette River Basin prior to the 1940s (Table 8). Wilkes (1845) estimated that the fishery at Willamette Falls could yield up to 800 barrels (122,000 kg) of salmon. Collins (1892) reported that 16,874 salmon (303,732 kg) were shipped to Portland from the Willamette Falls fishery in April and May 1889. This estimate would not include tribal harvest or harvest that was shipped to markets other than Portland. McKernan and Mattson (1950) presented anecdotal information that the Native American fishery at Willamette Falls may have yielded 908,000 kg of salmon (454,000 fish @ 9.08 kg). Mattson (1948) estimated that the spring chinook salmon run in the 1920s may have been five times the existing run size of 55,000 fish (in 1947) or 275,000 fish, based on egg collections at salmon hatcheries. In general, it is likely that the Willamette River Basin historically supported a run of several hundred thousand fish.

Prior to the laddering of Willamette Falls, passage by returning adult salmonids (RKm 37) was only possible during the winter and spring high-flow periods. The early run timing of Willamette River spring-run chinook salmon relative to other Lower Columbia River spring-run populations is viewed as an adaptation to flow conditions at Willamette Falls. chinook salmon begin appearing in the Lower Willamette River in February, but the majority of the run ascends Willamette Falls in April and May, with a peak in mid May. Wilkes (1845) reported that the salmon run over the falls peaked in late May. Low flows during the summer and autumn months prevented fall-run salmon from accessing the Upper Willamette River Basin. Since the Willamette Valley was not glaciated during the last epoch (McPhail and Lindsey 1970), the reproductive isolation provided by the falls probably has been uninterrupted for a considerable time period. Willamette Falls may have been formed by the receding floodwaters of the Bretz Floods (12,000–15,000 years before present) (Nigro 2001). This isolation has provided the potential for significant local adaptation relative to other Columbia River population.

DV – Clackamas River Spring-Run Chinook Salmon

Life History Traits – Barin (1886) observed a run of chinook salmon that "commences in March or April, sometimes even in February." Additionally, from 1890 to 1903 spring run fish were spawned at the Clackamas Hatchery from mid-July to late August (Willis et al. 1995). Currently, the majority of spawning takes place from September through early October (Willis et al. 1995). Clackamas River spring-run chinook mature primarily at 4 years old (62% of the run) and at 5 years old (35% of the run) (Howell et al. 1985). Score = 2.

Effective Population Size – Historically, the Clackamas River supported a large population of spring-run chinook salmon; however, the construction of the Cazadero Dam in 1904 (RKm 43) and River Mill Dam in 1911 (RKm 37) limited migratory access to the majority of the historical spawning habitat for the spring run. In 1917, the fish ladder at Cazadero Dam was destroyed by floodwaters, eliminating fish passage to the upper basin (ODFW 1992). The average annual dam count (River Mill or North Fork Dam) from 1952-59 was 461 (Murtagh et al. 1992). Adult counts over North Fork Dam rose from 592 in 1979 to 2,122 in 1980 (Murtagh et al. 1992). Passage over North Fork Dam has averaged over 2,000 fish annually over the last 30 years. Additionally, several thousand spring-run chinook return to the Clackamas Hatchery each year. Score = 2.

Hatchery Impacts

Hatchery Domestication – Hatchery production of spring-run chinook salmon in the basin continued using broodstock captured at the Cazadero and River Mill Dams (Willis et al. 1995). Transfers of Upper Willamette River hatchery stocks (primarily the McKenzie River Hatchery) began in 1913, and between 1913 and 1959 over 21.3 million eggs were transferred to the Clackamas River Basin (Wallis 1961, 1962, 1963). Furthermore, a large proportion of the transfers occurred during the late 1920s and early 1930s to supplement the failure of the runs in the Clackamas River Basin at that time (Leach 1932). In 1942 spring-run chinook salmon propagation programs in the Clackamas River Basin were discontinued.

Artificial propagation activities were restarted in 1956 using eggs from a number of upper Willamette River hatchery stocks. The program released approximately 600,000 smolts annually through 1985. In 1976, the ODFW Clackamas Hatchery (located below River Mill Dam) began releasing spring-run chinook salmon (Willamette River hatchery broodstocks were used, since it was believed that the returns from the local population was too small to meet the needs of the hatchery (Murtagh et al. 1992)). Increases in adult returns over the North Fork Dam, and increases in redd counts above the North Fork Reservoir corresponded to the initial return of adults to the hatchery in 1980 (ODFW 1992, Willis et al. 1995). The Clackamas Hatchery predominately uses fish returning to the hatchery rack. Recent changes management policy by ODFW include releasing hatchery fish farther downstream and mass marking all hatchery releases to allow the removal of hatchery fish ascending the North Fork Dam. Prior to mass marking, it was estimated that over 75% of the fish spawning above the North Fork Dam were hatchery origin. Despite passing only unclipped fish in 2002 and 2003, studies have found that 24-30% of the spawners above North Fork Dam were hatchery-origin fish (Goodson 2005).

Genetic analysis by NMFS of naturally produced fish from the upper Clackamas River indicated that this stock was similar to hatchery stocks from the Upper Willamette River Basin (Myers et al. 1998, see Appendix A). This finding agrees with an earlier comparison of naturally produced fish from the Collawash River (a tributary to the upper Clackamas River) and upper Willamette River hatchery stocks (Schreck et al. 1986). This strongly suggests that fish introduced from the upper Willamette River have significantly interbreed into, if not overwhelmed, spring-run fish native to the Clackamas River Basin, and obscured any genetic differences that exist prior to hatchery transfers.

PNI ≤ 0.10 , Fitness = 0.65. (This scoring is problematical – issues include whether to consider the Upper Willamette origin of this broodstock as an introduction from out of basin. Also, the stock being introduced had already been used in other hatcheries for many generations.) Score = 1.5.

Hatchery Introgression **–** There is some uncertainty regarding the historical relationship between the spring-run chinook salmon above Willamette Falls and those in the Clackamas River. It is not clear if the genetic and phenotypic similarity between populations in the Upper Willamette River and Clackamas River is the result of massive hatchery transfers or a historical relationship. Score = NA.

Synthetic Approach – The hatchery propagation of Clackamas River chinook salmon began in the 1800s with the construction of the first hatchery in the Columbia River Basin. In recent years, hatchery operations have been marked by the importation of millions of spring-run chinook salmon eggs from the upper tributaries of the Willamette River, (above Willamette Falls). Estimates of hatchery contribution to the spawning escapement (base on passage at North Fork Dam) have historically been well above 75%, but currently between 30-50% (Goodson et al. 2005). Juveniles released into the Clackamas River have come from local adult hatchery returns and importation from other Upper Willamette River hatcheries. Genetic similarity is considered to be low, based on the lack of inclusion of “wild” (unmarked) spawners and imported eggs from outside of the basin. Diversity persistence score = 1.0.

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = 2-3**.**

Habitat Diversity – Changes to the distribution of gradients and river size has been relatively minor, although this does not consider changes in habitat quality, especially in the lower Clackamas River. Score (Order/Elevation) = 3/3.

Overall Score = 2.0. Direct changes in life history and hatchery effects were the primary concerns for this population, although many effects (especially habitat degradation) could not be accurately measured, but may also be important.

Previously: 2004 TRT 1.31, 2004 ODFW fail, 4-5 of the criteria met.

DV – Molalla River Spring-Run Chinook Salmon

Life History Traits – Craig and Townsend (1946) collected a number of subyearling juveniles moving downstream from the Molalla River. Score = NA

Effective Population Size – The Molalla River is located just above Willamette Falls and 50 Km from the mouth of the Willamette River. By 1903, the abundance of chinook salmon in the Molalla River had already decreased dramatically (ODF 1903). Surveys in 1940 and 1941 recorded 882 and 993 spring-run chinook salmon present, respectively (Parkhurst et al. 1950). Mattson (1948) estimated the run size to be 500 in 1947. Efforts are currently underway to reestablish natural production in the Molalla River Basin using other upper Willamette River spring-run populations, primarily North Santiam, Middle Fork, and McKenzie River hatchery stocks. Analysis of carcasses from the 2002 run indicated that only 2% (2) of the fish were naturally-produced of the 102 carcasses examined (Lindsey 2003). Natural productivity appears to be very low (Goodson 2005). Score = 1-2.

Hatchery Impacts

> *Hatchery Domestication* **–** There is no hatchery program in the Molalla River, although a large number of spring-run chinook salmon have been introduced from other Upper Willamette River populations. No genetic analysis is available for this population. Score = 1-2.
>
> *Hatchery Introgression* **–** Given the preponderance of non-local hatchery-origin fish in this DIP, use of this metric was considered more appropriate than using the PNI. The diversity score was adjusted to reflect the fact that hatchery introductions have come from the same stratum. Score = 1-2.
>
> *Synthetic Approach* – There is no hatchery program in the Molalla River Basin; however, a large number of Upper Willamette River spring-run chinook salmon from other hatchery programs in the ESU have been released. Analysis of carcasses suggests that a very large proportion (Ph>0.75) of the spawning adults are of hatchery origin (Lindsey 2003, Goodson et al. 2005). The genetic similarity between hatchery fish released (all from outside of the basin) and wild (unmarked) fish is thought to be low. Diversity persistence score = 0.0

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = 2-3.

Habitat Diversity – Although the quality of habitat may be severely degraded the proportion of accessible stream size reflects historical conditions, while much of the elevation diversity has been lost. Although not currently part of the model, considerable changes in the character of the mainstem Willamette River (i.e., loss of side channel habitat and channel braiding). Score (*Order/Elevation*) = 3/3.

Overall Score = 1.0. The small population size of this population and the high proportion of non-local hatchery fish on the spawning grounds were primary sources of concern. Habitat degradation and its effect(s) on life history traits may also be important, but are presently difficult to quantify.

Previously: 2004 TRT 0.64, 2004 ODFW fail, < 4 criteria met.

DV – North Santiam River Spring-Run Chinook Salmon

Life History Traits – Hatchery records from early in the 1900s indicates that spawning began in late August and continued until early October, with spawning currently occurring slightly later (OSHS 1925, Willis et al. 1995). North Santiam River spring-run chinook salmon mature primarily at 5 years old (55%) and 4 years old (41%). Alteration in the temperature and rate of discharge from the Dams has probably had a significant impact on the survival of eggs deposited below the dam. Changes in the temperature regime have resulted in accelerated embryonic development rates and premature emergence. Cramer et al. (1996) reports chinook salmon fry in the North Santiam River moving downstream in late November, in contrast to normal emergence in February or March (Craig and Townsend 1946). Score = 2.

Effective Population Size – The estimated run size for the entire North Santiam River Basin was 2,830 in 1947 (Mattson 1948). The naturally-produced component of the run in 2002 was estimated at 592 fish. Recent estimates of pre-spawning mortality have been high (>50%). Redd counts in recent years, 2000-2004, have be well below 100 redds (Goodson 2005). Score = 1-2.

Hatchery Impacts

Hatchery Domestication **–** The Oregon Fish Commission began egg-taking operations in 1911 when adults were captured below the confluence of the North Santiam and Breitenbush Rivers, and below where most of the natural spawning areas (except for the Little North Santiam River). The largest egg collection was 13,200,000 in 1934 (this would correspond to 4125 females @ 3200 eggs/female (Wallis 1963)). Between 1911 and 1960, the overwhelming majority of hatchery fish released into the North Santiam basin have come from adults captured from within the watershed, other introduction have come from the South Santiam, McKenzie, and Willamette River Hatcheries (Willis 1963). Analysis of carcasses sampled above Bennett Dam, indicated that only 4, 2, and 8% of the spawners in 2000, 2001, and 2002 (respectively) were naturally produced (Lindsey 2003). On average, the Marion Forks Hatchery collects a small number (< 5%) of natural origin fish to include in the broodstocks.

Genetic analysis of naturally produced juveniles from the North Santiam River indicated that the naturally produced fish were most closely related (although still significantly distinct (P >0.05) from other naturally- and hatchery-produced spring-run chinook from the Upper Willamette and Clackamas Rivers (NMFS 1998). PNI ≤ 0.10. Fitness = 0.35. Score = 1.0.

Hatchery Introgression **–** Although fish have been introduced from other basins in the Upper Willamette River, hatchery effects/introgression effects were considered in the indirect effects criteria. Score = NA.

Synthetic Approach – A hatchery program has operated in the North Santiam River for nearly 100 years. The influence of hatchery fish became more pronounced with the construction of Detroit Dam, and the loss of the majority of the natural spawning grounds. Currently, hatchery fish account for approximately 90% of the natural spawners (Ph>0.75)—due in part to low natural productivity and a high incidence of pre-spawning mortality. Additionally, the hatchery incorporates a very low number of unmarked fish as broodstock. Diversity persistence score = 0.5.

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has likely reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = 2-3.

Habitat Diversity – Habitat diversity loss is most severe for this DIP due to the loss of higher elevation spawning areas. Score *(Order/Elevation)* = 3/1

Overall Score = 1.0. Apparent changes in life history characteristics, a small naturally-spawning component and the potential for hatchery domestication were primarily concerns. There were additional factors that could not be quantified for lack of information.

Previously: 2004 TRT 1.00, 2004 ODFW fail, <4 criteria met.

DV – South Santiam River Spring-Run Chinook Salmon

Life History Traits – South Santiam River spring-run chinook salmon mature predominately at 4 years-old (62%) and 5 years-old (34%) (Smith et al. 1987). There does not appear to have been much change in the spawn timing for fish in this DIP, with spawning occurring from August to late September and early October (OSHS 1925, Willis 1960, Wevers et al. 1992). Score = NA.

Effective Population Size – Escapement to the South Santiam River was estimated to be 1,300 in 1947 (Mattson 1948). ODFW (1995) considered that the naturally-spawning populations in the South Santiam River were "probably extinct". In 1998, there were 166 spring-run chinook salmon redds observed in the South Fork; however it was presumed that these are the progeny of hatchery produced spring-run (Lindsay et al. 1999). In 2002, it was estimated that 14% (227) of the spring run sampled below Foster Dam consisted of naturally-produced fish, in addition to 444 fish, 58% of the total, passed above Foster Dam. Currently, surveys count an average of 100 redds each year. Score = 2-3.

Hatchery Impacts

Hatchery Domestication **–** Wallis (1961) suggested that because of poor husbandry practices, releases from the South Santiam Hatchery did not significantly contribute to escapements (the hatchery may have mined returning naturally produced adults each year). In recent years the proportion of naturally-spawning fish that are of hatchery origin has been over 80% (Goodson 2005). In 2003, over 6,000 spring-run fish were collected at the South Santiam Hatchery, the contribution of natural-origin fish to the broodstock is thought to be small (<5%).

No genetic analyses are available for South Santiam River spring-run chinook salmon.

PNI $\leq$ 0.10. Fitness = 0.60. Score = 1.5.

Hatchery Introgression **–** Fall-run chinook salmon are also present in the Santiam River Basin, but the spring-run and fall-run chinook salmon are thought to be spatially and temporally separated on the spawning grounds. Score = NA.

Synthetic Approach – The South Santiam Hatchery has been producing spring-run chinook salmon since 1925. Wallis (1961) concluded that hatchery contributed little to escapements during the first decades of its operation. Currently, a large proportion of returning adults are of hatchery origin (Ph>0.75). The genetic similarity between hatchery fish released and wild (unmarked) fish is thought to be low due to the low proportion of unmarked fish included as broodstock. Diversity persistence score = 0.5.

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = **2-3.**

Habitat Diversity – Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score *(Order/Elevation)* = 4/3.

Overall Score =1.5. The large numbers of hatchery fish relative to natural-origin fish were a major concern. Additional concerns included small effective population size and habitat mediated changes in diversity (although it was difficult to quantify the later).

Previously: 2004 TRT 1.09, 2004 ODFW fail < 4 criteria met.

DV – Calapooia River Spring-Run Chinook Salmon

Life History Traits – No information available. Score = NA

Effective Population Size – A small run of spring chinook salmon historically existed in the Calapooia River. Parkhurst et al. (1950) reported that the run size in 1941 was approximately 200 adults, while Mattson (1948) estimated the run at 30 adults in 1947. ODFW (1995) considered the run in the Calapooia to be extinct, with limited future production potential. Goodson (2005) estimates that this population is extremely small (<50). Score = 1.

Hatchery Impacts

> *Hatchery Domestication* **–** It is believed the overwhelming majority of fish spawning in the Calapooia are of hatchery origin (introduced from other Upper Willamette River hatcheries) (Goodson 2005). The majority of the Upper Willamette River hatchery broodstocks have been under culture for extended periods (>15 generations).
>
> PNI estimate not used. Score = NA.
>
> *Hatchery Introgression* **–** Given the preponderance of non-local hatchery-origin fish in this DIP, use of this metric was considered more appropriate than using the PNI. The diversity score was adjusted to reflect the fact that hatchery introduction came from the same stratum. Score = 1-2.
>
> *Synthetic Approach* – There is no hatchery program in the Calapooia River Basin; however, a large number of Upper Willamette River spring-run chinook salmon (both juveniles and surplus adults) from other hatchery programs in the ESU have been released. Very few redds are observed in the river, and it is thought that natural productivity is very low. The genetic similarity between hatchery fish released (all from outside of the basin) and wild (unmarked) fish is thought to be low. Diversity persistence score = 0.0.

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = 2-3.

Habitat Diversity – Although the quality of habitat may be severely degraded the proportion and character (elevation and stream size) of accessible habitat reflects historical conditions. Score *(Order/Elevation)* = ¾.

Overall Score =1.0. Small population size (the population was considered extirpated by ODFW) and the preponderance of non-local hatchery fish were primary concerns. Other facts may also be important, but sufficient information is not presently available to quantify these effects.

Previously: 2004 TRT 0.70 , 2004 ODFW fail, <4 criteria met.

DV – McKenzie River Spring Run Chinook Salmon

Life History Traits – ODF (1903) surveyed much of the M'Kenzie [sic]. In their report they state, "It has been generally reported by settlers and those living along the river that salmon can be seen spawning during the months of August and September all along the river, but principally from Leaburg post office up to its source." Currently, spring-run chinook salmon ascend Leaburg Dam in two modes, one between May and early July and the other in late August and September. Recent analysis indicates that the majority of fish mature as 5 year-olds (56%) with 44% of the fish maturing as 4 year olds (Lindsey et al. 1997). Score = NA.

Effective Population Size – The 30-year average count of natural-origin fish at Leaburg Dam has been 1,980 (Goodson 2005); however, recent counts have been as high as 4,070 (2004). Score = 3-4.

Hatchery Impacts

Hatchery Domestication **–** The McKenzie River Hatchery has been in operation for nearly 100 years. During the early years of operation, attempts were made to collect the entire run via a weir at the mouth of the McKenzie River. Husbandry limitations probably minimized the influence of hatchery-origin fish during the early years. Currently, a large number of adipose-clipped, hatchery-origin, adults are prevented from accessing spawning grounds above Leaburg Dam. Analysis of otolith marked fish indicated that 67% (2001) and 55% (2002) of the spawned-out carcasses above Leaburg Dam were naturally-produced (Lindsey 2003). Overall, it is estimated that the hatchery contribution to escapement is approximately 35% (Goodson 2005), although the inclusion of natural-origin fish into the hatchery broodstock is thought to be low.

Genetic analysis of juveniles from the McKenzie River indicated that the naturally produced fish were most closely related other naturally- and hatchery-produced spring-run chinook from the Upper Willamette and Clackamas Rivers (NMFS 1998, see Genetics Appendix). There is very little apparent straying based on the recoveries of CWT fish released from the McKenzie River Hatchery, with more than 97% of all freshwater recoveries occurring in the McKenzie River Basin. PNI $\leq$ 0.2, Fitness = 0.55. Score = 1.5.

Hatchery Introgression **–** Relatively few out-of-basin strays are recovered in the McKenzie River. Score = 4.

Synthetic Approach – Of the populations in the UWR chinook ESU, the McKenzie probably has the lowest level of hatchery fish on the spawning grounds. This is due, in part, to the removal of marked hatchery-origin fish at Leaburg Dam and the "relatively" high productivity of the McKenzie Basin. Recent estimates suggest that the hatchery contribution to escapement is 35% (Goodson 2005). In general, there have been few transfers of UWR fish from other rivers into the McKenzie Basin. The McKenzie Hatchery, however, includes few unmarked fish into its broodstock. Diversity persistence score = 1.5

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = **2-3.**

Habitat Diversity – The proportion and character (elevation and stream size) of accessible habitat reflects is similar to historical conditions, although the loss of higher elevation habitat is considerable. Score *(Order/Elevation)* = 3/2.

Overall Score =1.5. Of the effects that could be quantified, the long term presence of the McKenzie River Hatchery program was thought to be significant. Changes in life history due to the altered thermal regime or changes in the juvenile migratory corridor and downstream rearing habitat could not be estimated due to lack of information.

Previously: 2004 TRT 1.79, 2004 ODFW estimate fail, 4-5 criteria met.

DV – Middle Fork Willamette River Spring-Run Chinook Salmon

Life History Traits – Studies of juvenile emigration from the Middle Fork Willamette River in 1941 indicated that downstream migration occurred on a more or less continuous basis from March through the autumn (Craig and Townsend 1946). Natural production is currently limited and it is not possible to accurately estimate the existing juvenile and adult life history strategies. Currently, hatchery spawning takes place from early September and into early October (Willis et al. 1995). Score = NA

Effective Population Size – There were spawning aggregations in Fall Creek, Salmon Creek, North Fork Middle Willamette River, mainstem Middle Fork Willamette River, and Salt Creek (Mattson 1948, Parkhurst et al. 1950). Collectively, these areas would likely have produced tens of thousands of fish. Based on records from the Willamette River Hatchery (Dexter Ponds) (1911-present), the largest egg collection of 11,389,000 in 1918 (Wallis 1962) would correspond to 3,559 females (@ 3,200 eggs/female). Although Parkhurst et al. (1950) estimated the Fall Creek Basin could support several thousand salmon, by 1938 the run had already been severely depleted. In 1947, the run had dwindled to an estimated 60 fish (Mattson 1948). Construction of the Fall Creek Dam (1965) included fish passage facilities, but passage is only possible during high flow years (Connolly et al. 1992). Recent estimates suggest escapement averages a few hundred fish, depending primarily on what is re-released from hatchery returns. Fewer than 100 redds are normally counted (Firman et al. 2004, Firman et al. 2005). Score= 2-3.

Hatchery impacts

Hatchery Domestication **–** ODFW (1995) concluded that the native spring-run population was extinct, although some natural production, presumably by hatchery origin adults still occurs. Of the 260 carcasses examined from the Middle Fork Willamette River (including Fall Creek), 11 (4%) were estimated to have been naturally produced (Lindsey 2003). In 2003, 7,340 spring run chinook salmon returned to the Willamette Hatchery, very few if any of there are likely to have been naturally produced. Of the 1,525 fish analyzed at the Willamette Hatchery, only 4 fish were unmarked (Firman et al. 2004). The Willamette Hatchery has been in operation since 1911, and has exchanged broodstock with other Upper Willamette River hatcheries throughout much of this period (Wallis 1962). *PNI ≤ 0.1, Fitness = 0.30.* Score = 1.5.

Hatchery Introgression **–** Of the 46 CWTs recovered from the spawning grounds, 1 came from the McKenzie River, 1 came from a release of Middle Fork Willamette stock released into Youngs Bay, and 44 came from the Willamette River Hatchery (Firman et al. 2004). Score = 4.

Synthetic Approach – Although historically the Middle Fork Willamette River was a major contributor to the UWR ESU. Currently there is little natural production in this basin, due to the construction of Dexter Dam and Dorena Dam (Row River). The Willamette Hatchery has been propagating spring-run chinook salmon since 1911 and currently releases 1,600,000 yearlings (2006). For the 2002-2004 return years the proportion of hatchery fish naturally spawning ranged fro 72 to 96% (Ph>0.75). The inclusion of unmarked fish into the hatchery broodstock is likely less than 5%.

Furthermore, the hatchery has imported large numbers of fish from other UWR hatcheries. Diversity persistence score = 0.0

Anthropogenic Mortality – Total harvest for catch years 1999-2002, averaged 40.7% for Upper Willamette River populations. Due to the initiation of selective sport fisheries, the harvest impact on unmarked fish is somewhat less than this average. Changes in river conditions in the Clackamas River, Lower Willamette River, and Columbia River and estuary have likely had an effect on juvenile life history diversity. Specifically, the loss of juvenile rearing areas has reduced the contribution of subyearling migrants to the population (Craig and Townsend 1946, Mattson 1962). Score = **2-3.**

Habitat Diversity – The diversity of habitat in this DIP has been highly modified, especially in the relative loss of higher elevation habitats. Score *(Order/Elevation)* = 3/1

Overall Score = 1.0. The small size of the naturally-produced population (the population was considered extirpated by ODFW) and the preponderance of hatchery fish (even though they potentially represent local sources) were primary concerns. The shift in available spawning habitat from higher elevation streams to habitat below the dams was also a concern.

Previously: 2004 TRT 1.21, 2004 ODFW fail, meets <4 of the criteria

DV – Criterion Summary

With respect to the diversity criterion, populations in this ESU were classified into either the 'moderate' or 'high' risk categories (Figure 41) In addition, as the short profile of the diamonds symbols in Figure 41 illustrate, these DV ratings were made with a higher relative degree of certainty than for other criteria (Figures 31 and 40). The loss of genetic resources because of small population sizes, loss of historically accessible habitat and the high incidence of hatchery strays are the primary factors that resulted in the DV criterion population ratings.

The DV ratings and associated uncertainty result in only one population, the Clackamas, being placed into the 'moderate' risk category with confidence. As the diamond symbols in Figure 41 illustrate, the remaining populations are clearly in the 'high' risk category or are borderline between the 'moderate' and 'high' risk classification. Given these results, we conclude the most probable DV criterion risk classification for this ESU is 'high'.

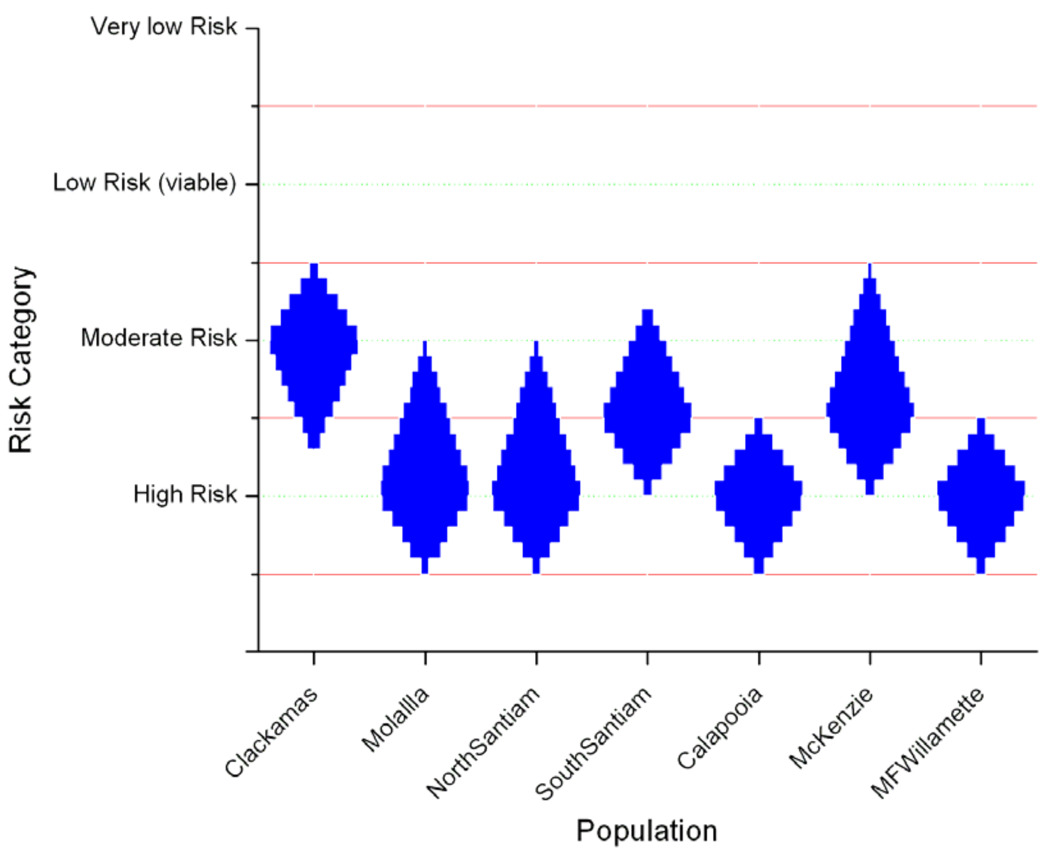


Figure 41: Upper Willamette spring chinook risk status summary based on evaluation of diversity only.

V. Summary of Population Results

The result we obtained when the scores for all three population criteria were combined was that the risk of extinction for UW chinook is high (Figure 42 and Figure 43). The Clackamas population exhibited the lowest extinction risk, being most likely in the 'low' risk category. Five of the seven populations were clearly in the high risk category. In addition, their 'high risk' classification was made with considerable certainty as evidenced by the relatively shortened aspect of the diamonds representing population status. Overall, these chinook populations and therefore the ESU can be characterized as having a high risk of extinction.

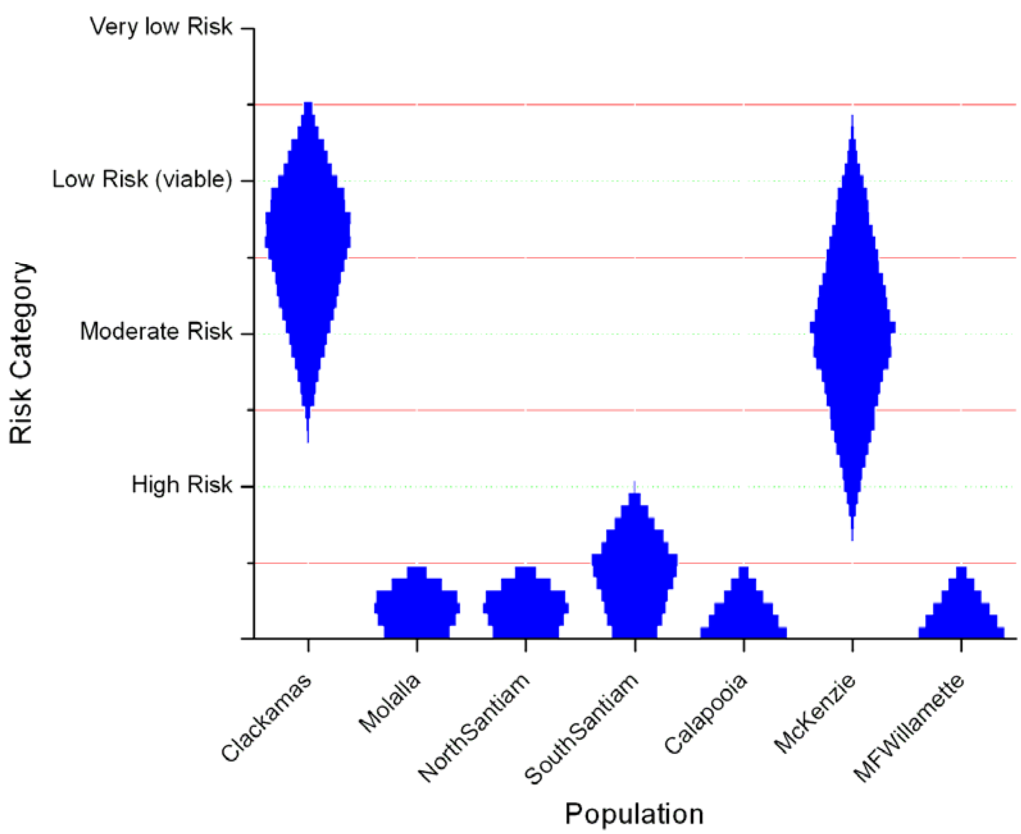


Figure 42: Upper Willamette spring chinook population status summaries based on minimum score method.

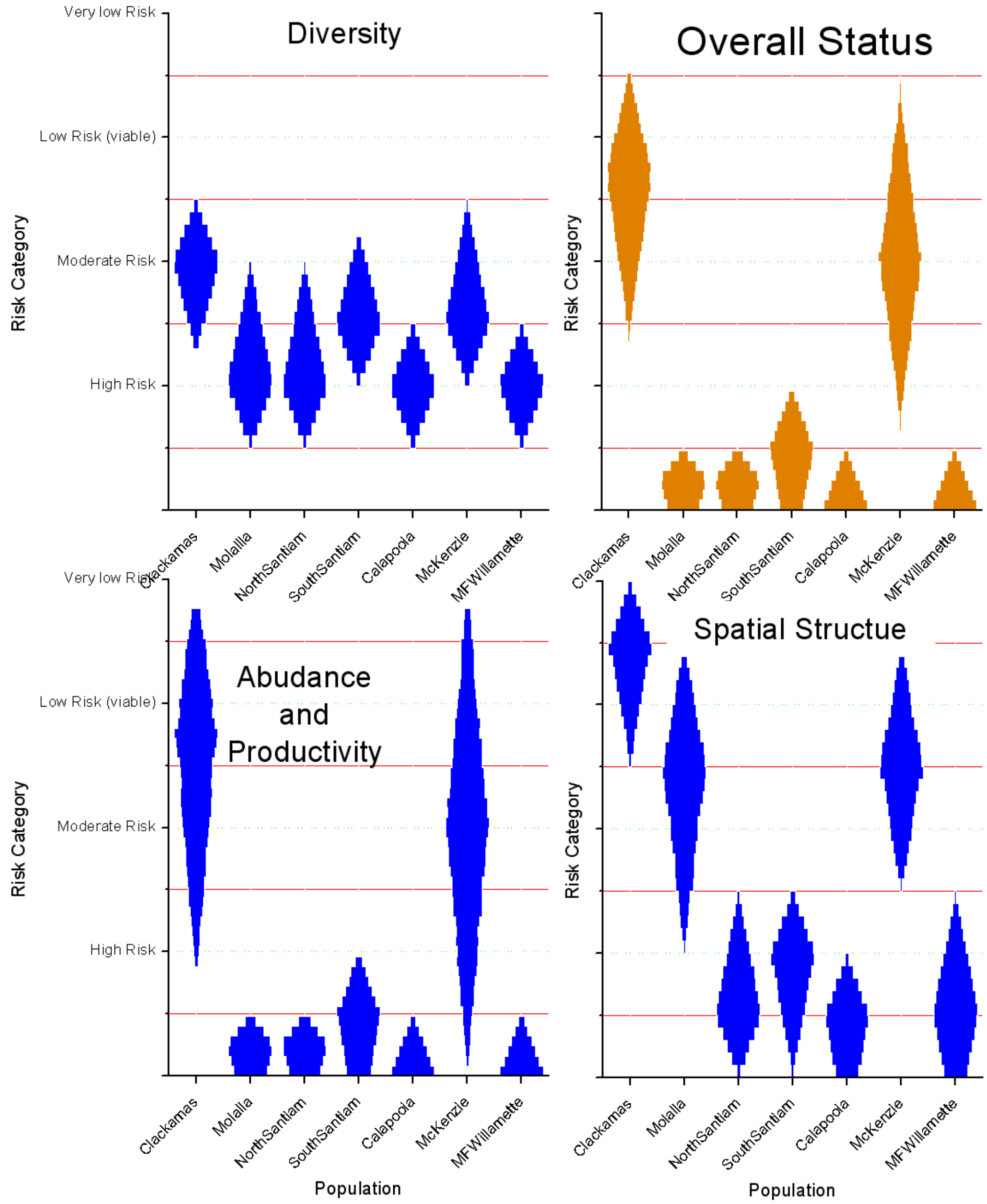


Figure 43: Upper Willamette steelhead status graphs of each attribute and the overall summary.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Part 7: Upper Willamette Steelhead

September 2007

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I. ESU Overview and Historical Range

The Upper Willamette Steelhead ESU consists of four populations occupying watersheds as shown in Figure 1. The four populations in the ESU are the Molalla, North Santiam, South Santiam, and Calapooia. The West Side Tributaries represent an area of intermittent use by steelhead, which may be important for ESU recovery, but is not considered to have historically been an independent population (Myers et al. 2006). The population structure described here differs from the population structure reported in the Oregon Native Fish Status report (ODFW 2005). In its report, ODFW identified four populations on the west side of the Willamette Valley and segregated the South Santiam into upper and lower watersheds.

Steelhead in this ESU are depressed from historical levels, although to a much less extent than spring Chinook in the Willamette basin. Further, all of the historical populations remain extant with moderate numbers of wild steelhead produced each year. However these populations have been adversely impacted by the alteration and loss of spawning and rearing habitat associated with hydropower development. Hatchery reared winter steelhead are no longer released into any of the UW steelhead populations. However, introduced hatchery summer steelhead still occur in the North and South Santiam basins and also migrate via the mainstem Willamette River to the McKenzie River basin.

A time series of abundance is available for all four populations in the ESU (Appendix B). However, spawner abundance estimates, with the exception of the upper South Santiam, are based entirely on spawning surveys conducted for a small portion of the steelhead habitat. The results from these surveys are then expanded for the entire watershed to obtain an estimate for population abundance. As a consequence there is considerable uncertainty concerning the accuracy of these abundance estimates

The presentation of our assessment begins with three sections, each of which evaluates one of the viability criteria (i.e., abundance/productivity, spatial structure, and diversity). This is then followed by a synthesis section where we pool the results from these criteria evaluations into a status rating for each population. We end our presentation with an interpretation of the population results in terms of the overall status of this ESU. The methods are described in Part 1 of this report.

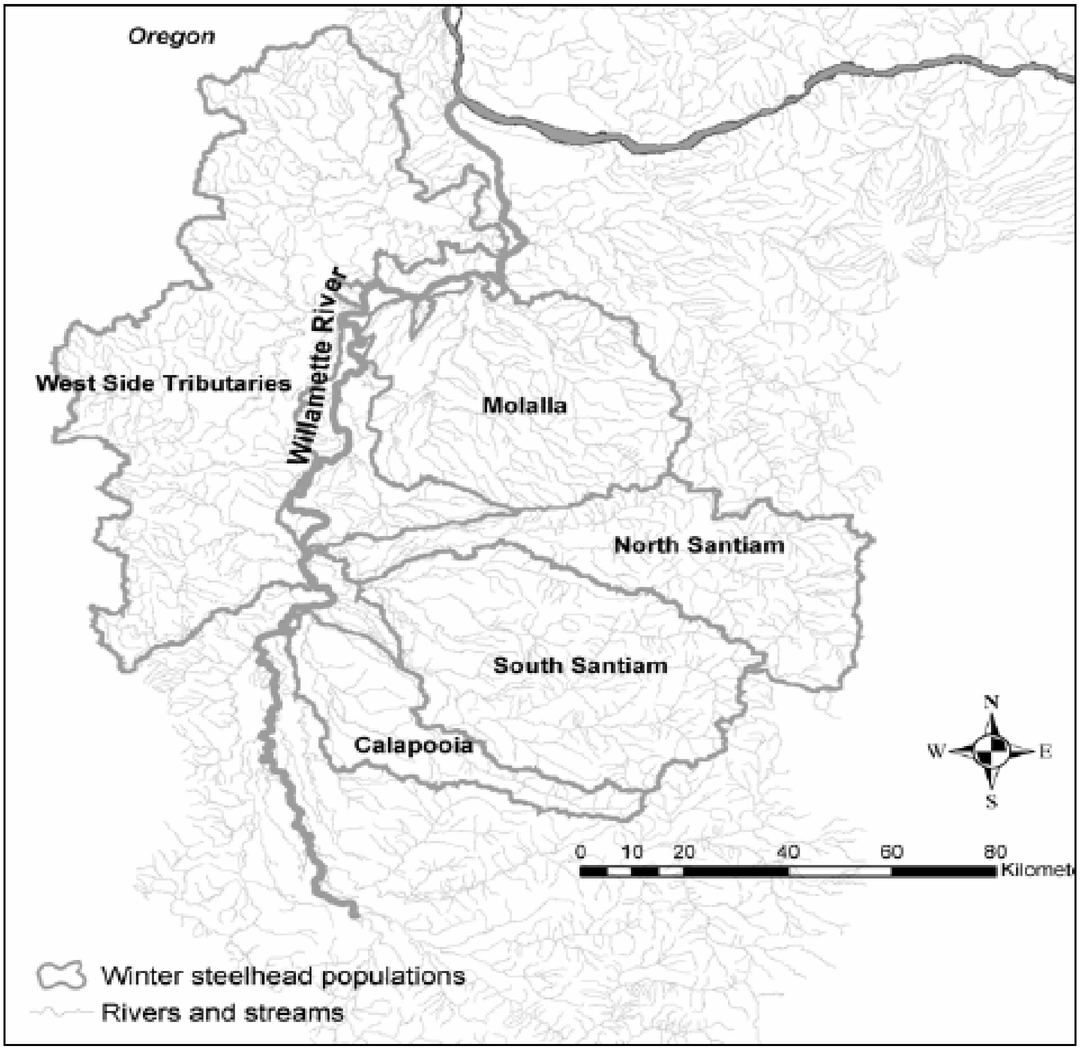

Figure 1: Map of populations in Upper Willamette winter steelhead ESU. The West Site Tributaries are considered an "intermittent use" area, not an independent population.

II. Abundance and Productivity

A&P – Molalla

A time series of abundance sufficient for quantitative analysis is available for the Molalla population (Appendix B). Descriptive graphs and viability analysis results are provided in Figure 2 to Figure 9 and in Table 1 and Table 4. The population is relatively large, with a long-term geometric mean natural origin spawner of 1,233 and a recent geometric mean of 937. These values are in the viable to very low risk minimum abundance threshold (MAT) category.

The modeling results reflect the uncertainty in the input data and therefore in the population status. The pre-harvest viability curve analyses suggest that the population is probably viable if harvest levels remain at current rates (average post-1990 fishery mortality rate = 0.10). The escapement viability curves suggest that the harvest pattern observed over the course of the time series which included a period of time when the fishery mortality rate was 0.23 is not likely to be sustainable by the population. Largely because of the high amount of measurement error in the input data, the "blobs" describing the current population status are relatively large and span all of the viability curve risk categories.

The CAPM analysis indicates that the population is very likely not viable and the predicted CRT risk probability over 100 years is around 24%. The PopCycle model suggests a CRT risk probability of around 21%. Overall, we estimate that the population is most likely in the moderate risk category based on abundance and productivity data, but the range of possibility spans the entire spectrum from very low risk to very high risk. The Oregon Native Fish Status report (ODFW 2005) gave this population a "pass" for abundance and productivity.

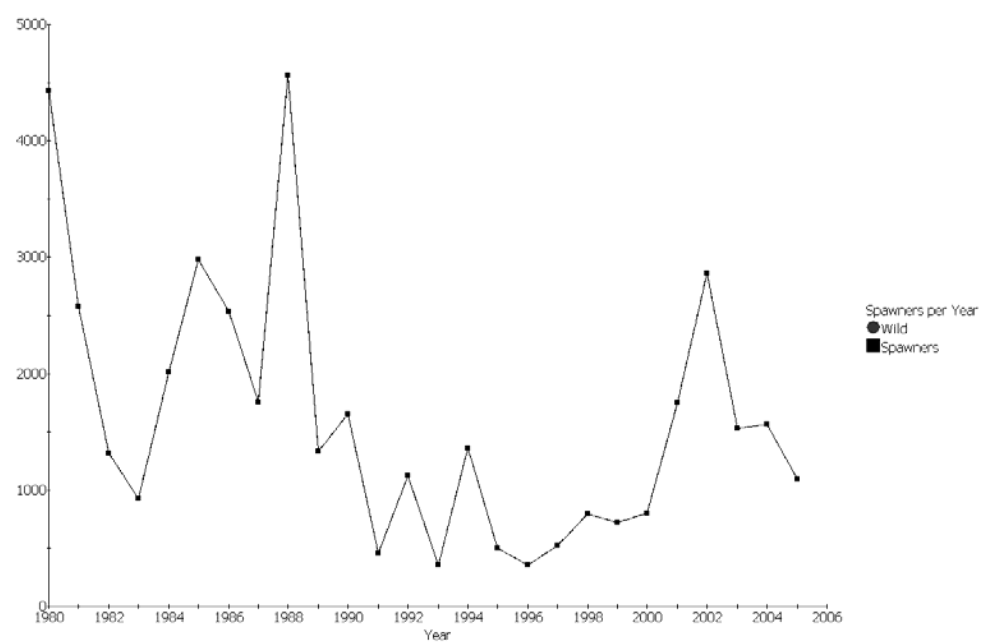

Figure 2: Molalla River winter steelhead abundance.

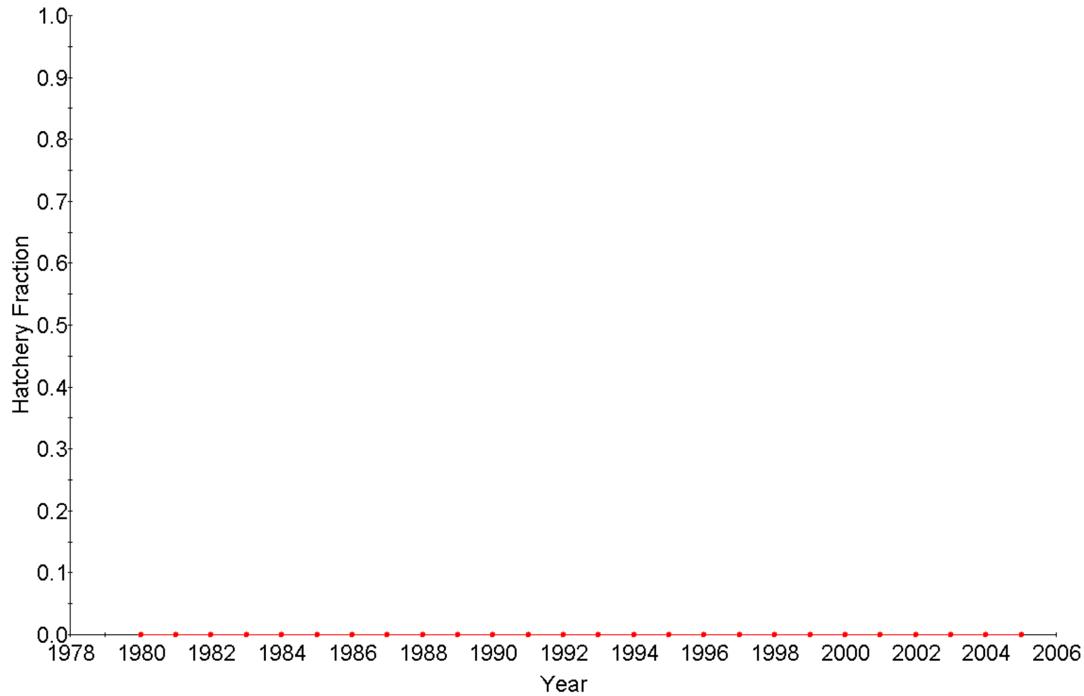

Figure 3: Molalla River winter steelhead hatchery fraction.

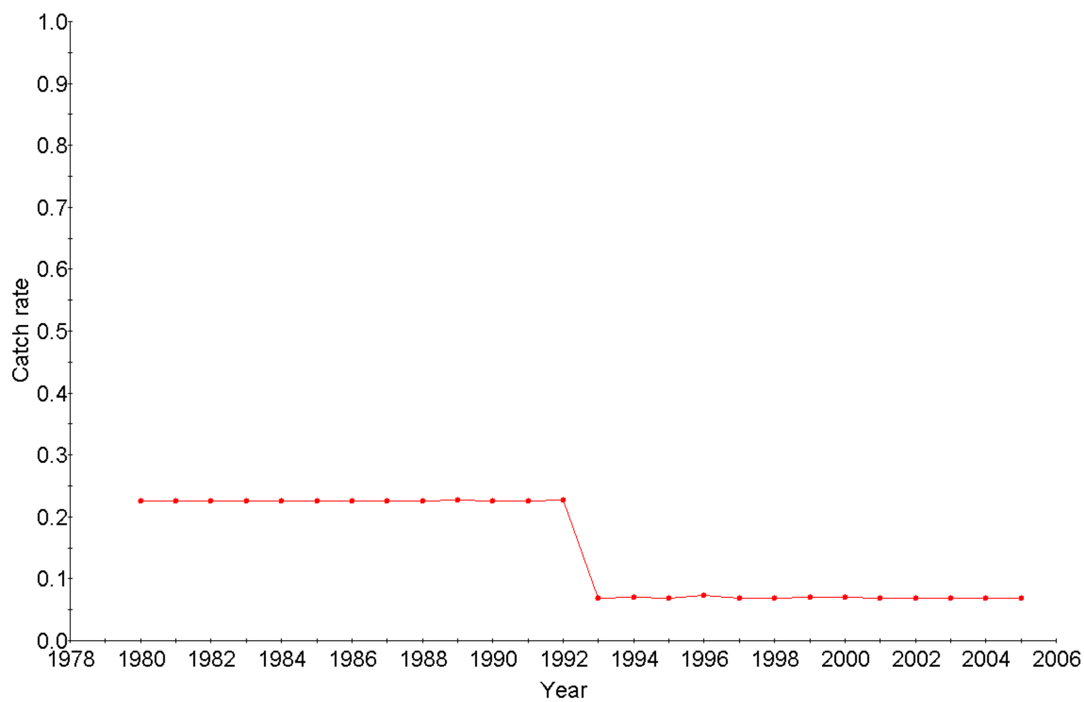

Figure 4: Molalla River winter steelhead harvest rate.

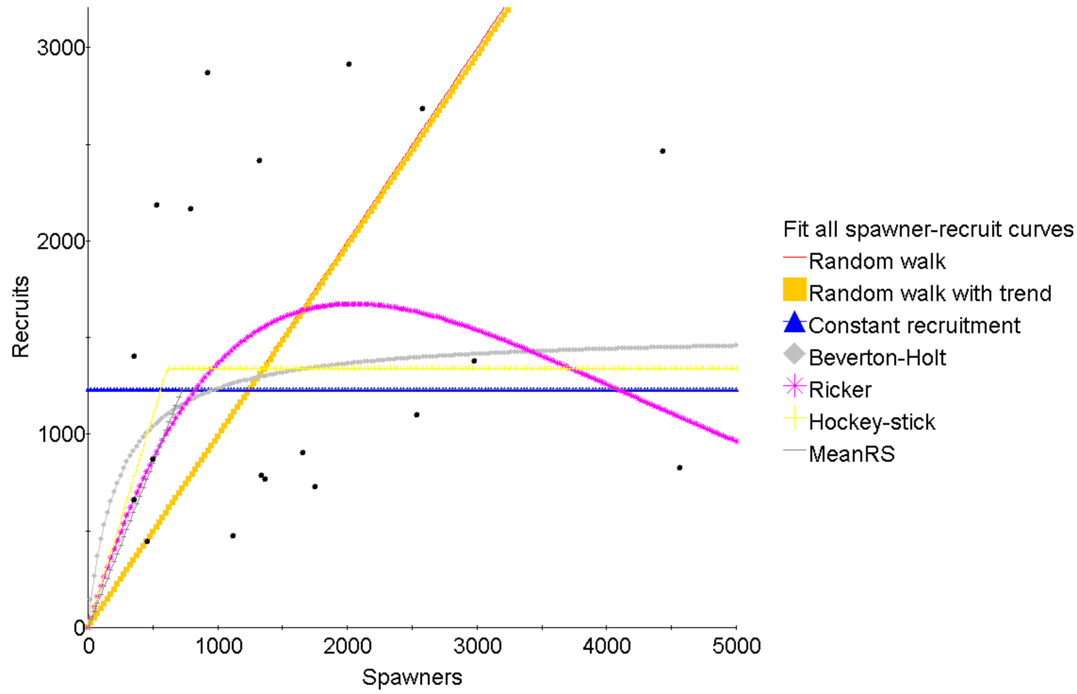

Figure 5: Molalla River winter steelhead escapement recruitment functions.

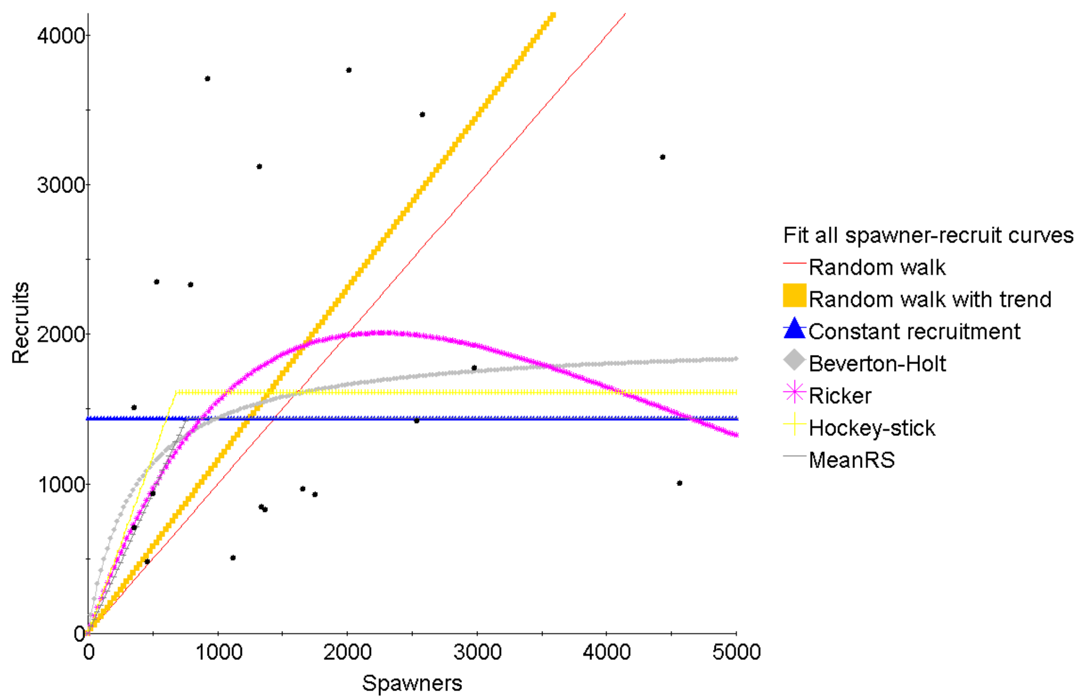

Figure 6: Molalla River winter steelhead pre-harvest recruitment functions.

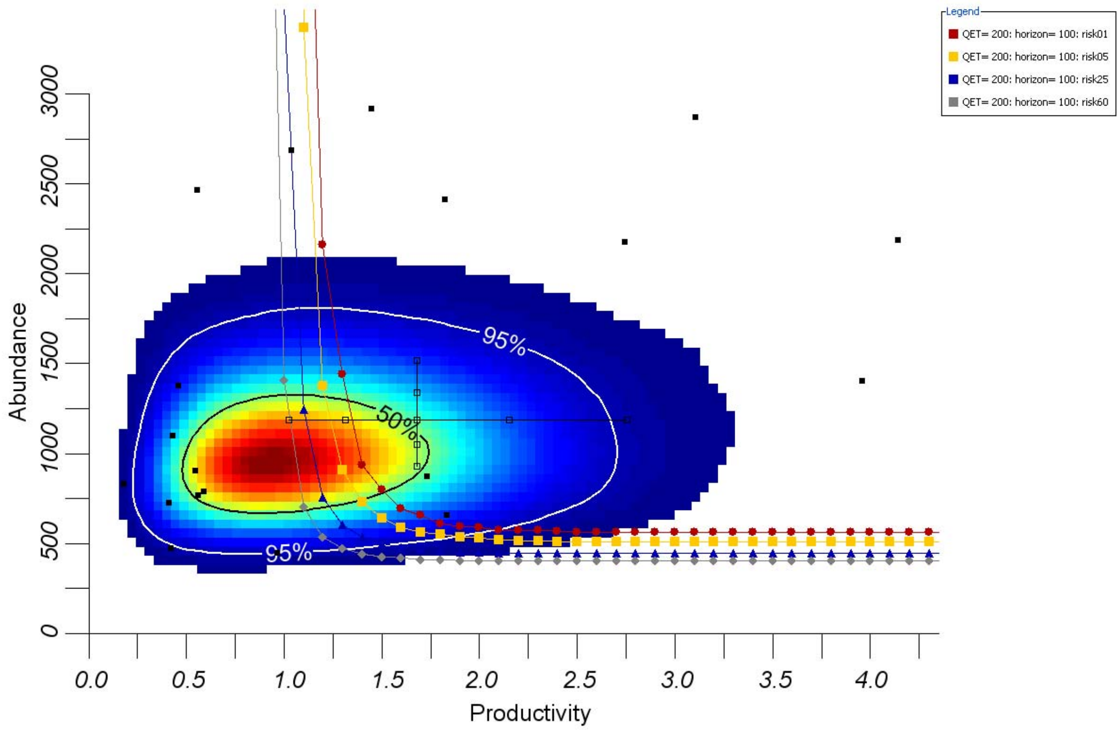

Figure 7: Molalla River winter steelhead escapement viability curves.

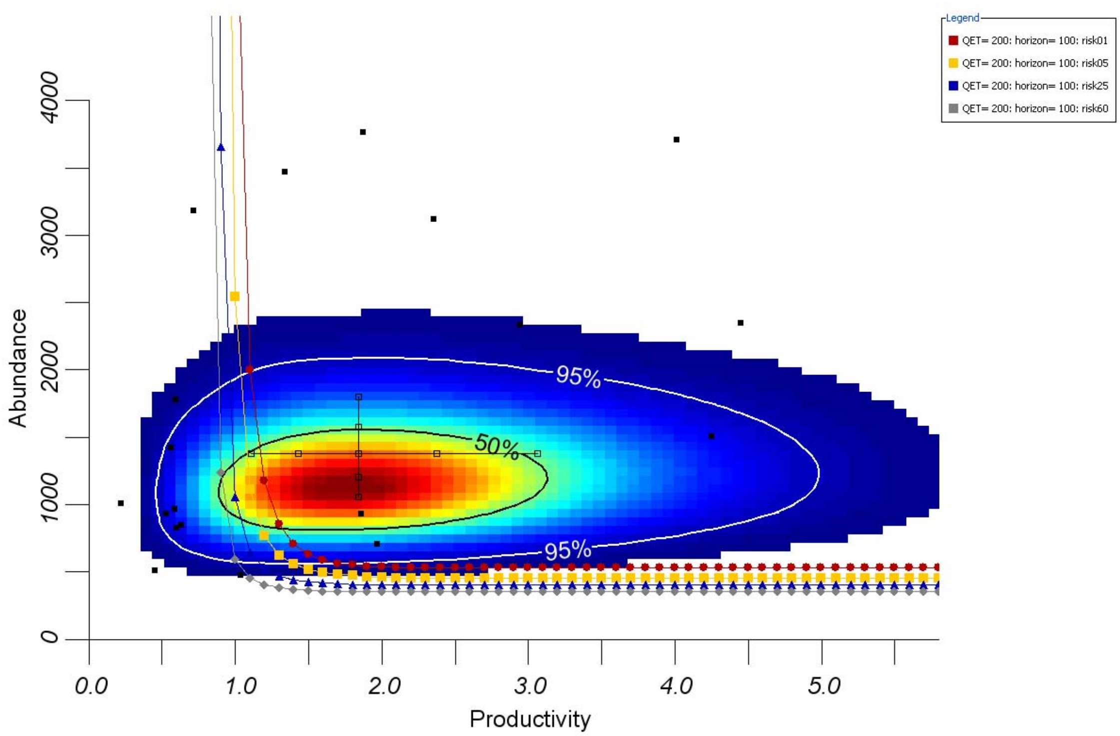

Figure 8: Molalla River winter steelhead pre-harvest viability curves.

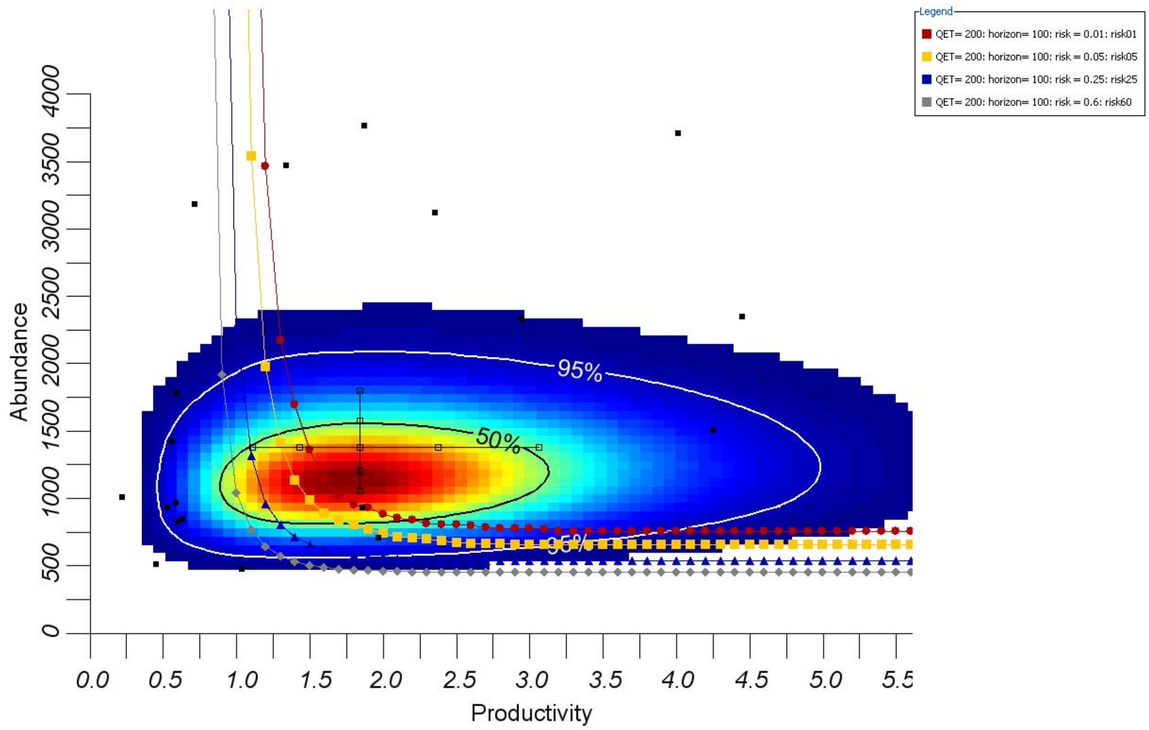

Figure 9: Molalla River winter steelhead pre-harvest viability curves.

Table 1: Molalla River winter steelhead summary statistics. The geometric mean natural origin spawner abundance (highlighted) is in the "viable" to "very low risk" viability criteria category. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1980-2005	1990-2005	1980-2005	1990-2005
Length of Time Series	26	16	26	16
Geometric Mean Natural Origin Spawner Abundance	1273 (952-1703)	914 (655-1275)	NA	NA
Geometric Mean Recruit Abundance	1233 (911-1669)	937 (595-1475)	1440 (1036-2001)	1006 (639-1584)
Lambda	0.988 (0.79-1.235)	1.058 (0.698-1.602)	1.016 (0.813-1.27)	1.066 (0.69-1.647)
Trend in Log Abundance	0.966 (0.931-1.002)	1.059 (0.989-1.132)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.985 (0.64-1.517)	1.378 (0.704-2.699)	1.15 (0.753-1.757)	1.48 (0.756-2.899)
Geometric Mean Recruits per Spawner (broods < median spawner abundance)	1.695 (0.97-2.963)	2.275 (1.268-4.081)	1.889 (1.064-3.353)	2.443 (1.361-4.384)
Average Hatchery Fraction	0	0	NA	NA
Average Harvest Rate	0.147	0.098	NA	NA
CAPM median extinction risk probability (5th and 95th percentiles in parenthesis)	NA	NA	0.240 (0.135-0.480)	NA
PopCycle extinction risk	NA	NA	0.21	NA

Table 2: Escapement recruitment parameter estimates and relative AIC values for Molalla winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.87 (0.71-1.24)	11.4
Random walk with trend	0.99 (0.72-1.51)	NA	0.87 (0.73-1.29)	13.4
Constant recruitment	NA	1232 (979-1643)	0.61 (0.51-0.91)	0
Beverton-Holt	6.5 (3-28.14)	1528 (1090-2335)	0.59 (0.5-0.9)	0.9
Ricker	2.23 (1.32-3.66)	1674 (1382-3172)	0.63 (0.54-0.99)	3.1
Hockey-stick	2.19 (2.2->30)	1339 (983-1682)	0.59 (0.51-0.9)	0.6
MeanRS	1.69 (1.14-2.46)	1233 (981-1554)	0.37 (0.24-0.47)	15

Table 3: Pre-harvest recruitment parameter estimates and relative AIC values for Molalla winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.87 (0.7-1.23)	8.1
Random walk with trend	1.15 (0.85-1.74)	NA	0.85 (0.71-1.26)	9.6
Constant recruitment	NA	1441 (1121-1965)	0.66 (0.55-0.99)	0
Beverton-Holt	5.34 (2.45->30)	1970 (1284-3610)	0.64 (0.54-0.96)	0.1
Ricker	2.41 (1.34-4.01)	2007 (1646-5365)	0.65 (0.56-1.04)	1.6
Hockey-stick	2.37 (2.21->30)	1610 (1129-2078)	0.63 (0.55-0.98)	0
MeanRS	1.89 (1.26-2.77)	1439 (1125-1850)	0.42 (0.27-0.54)	10.7

Table 4: Molalla River winter steelhead CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.682	0.922	0.982
Probability the population is above "Moderate risk of extinction" category	0.613	0.898	0.528
Probability the population is above "Viable" category	0.531	0.855	0.002
Probability the population is above "Very low risk of extinction" category	0.450	0.814	0.000

A&P – North Santiam

A time series of abundance sufficient for quantitative analysis is available for the North Santiam population (Appendix B). Descriptive graphs and viability analysis results are provided beginning with Figure 9 and in Table 5 and Table 8. The population is relatively large, with a long-term geometric mean natural origin spawner population of 2,722 and a recent geometric mean of 2,109 (Table 5). These values are in the very low risk minimum abundance threshold (MAT) category.

The modeling results reflect the uncertainty in the input data and therefore in the population status. The pre-harvest viability curve analyses suggest that the population is probably viable if harvest levels remain low. The escapement viability curves suggest that the harvest pattern observed over the course of the time series is likely to be sustainable. Largely because of the high amount of measurement error in the input data, the “blobs” describing the current population status are relatively large and span all of the viability curve risk categories.

The CAPM analysis indicates that the population is viable as evidenced by a median value for predicted CRT probabilities of 0.005. The PopCycle analysis also suggests a low risk (<0.01). We estimate that the population is most likely in the viable category, but the range of possibility spans the entire spectrum from very low risk to very high risk. The Oregon Native Fish Status report (ODFW 2005) gave this population a “pass” for abundance and productivity.

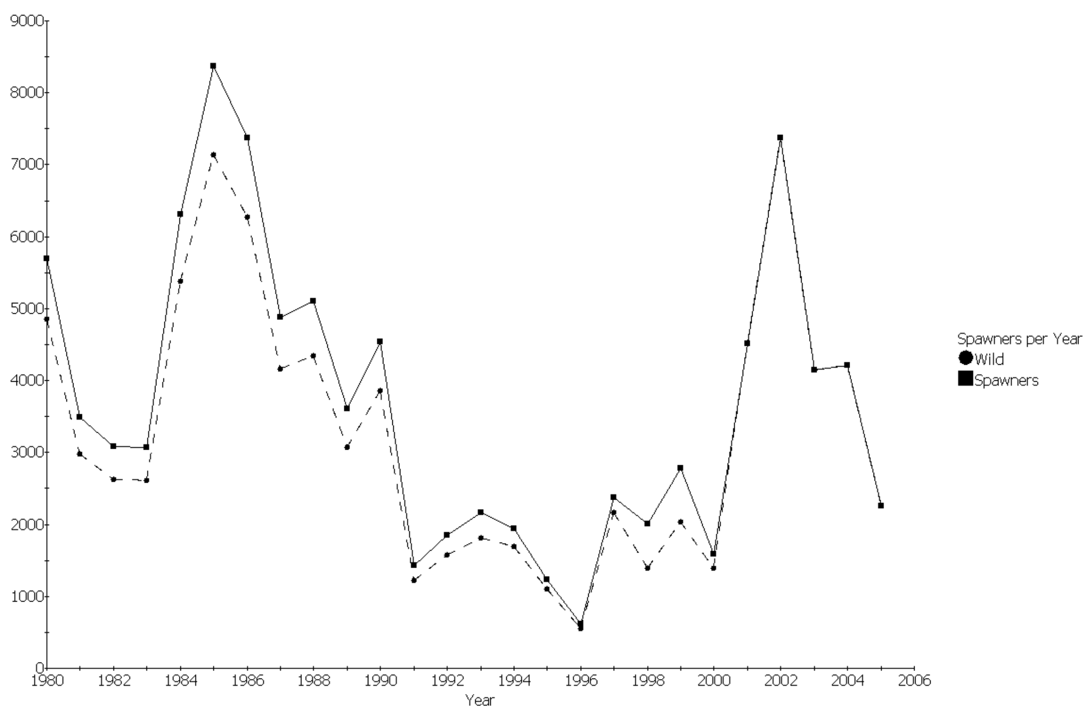

Figure 10: North Santiam River winter steelhead abundance.

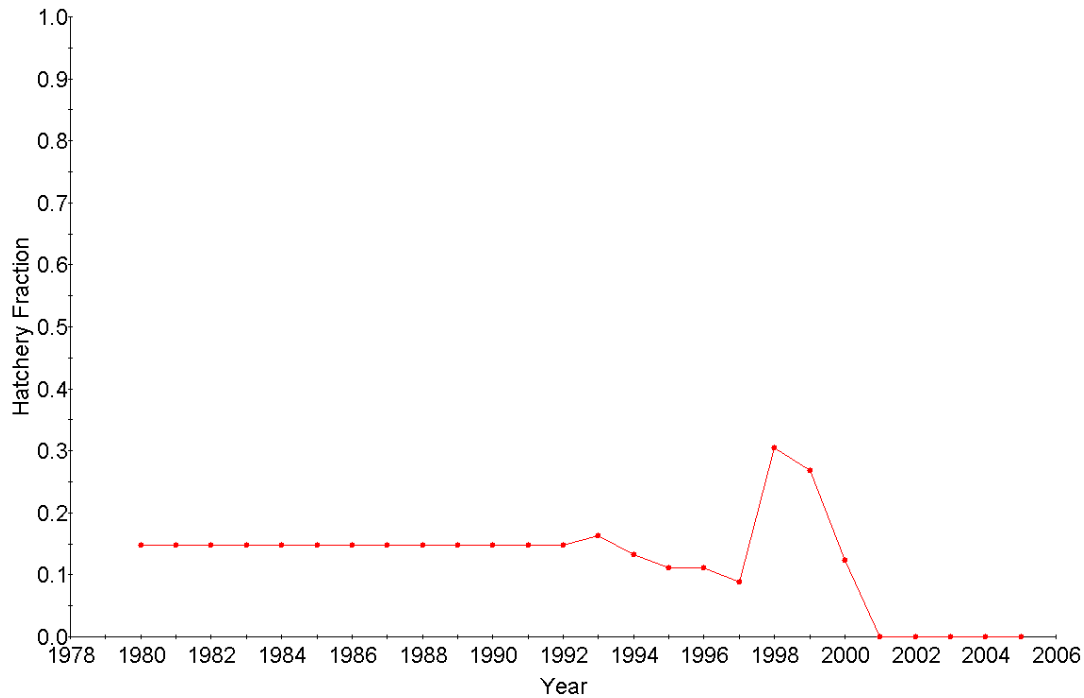

Figure 11: North Santiam River winter steelhead hatchery fraction.

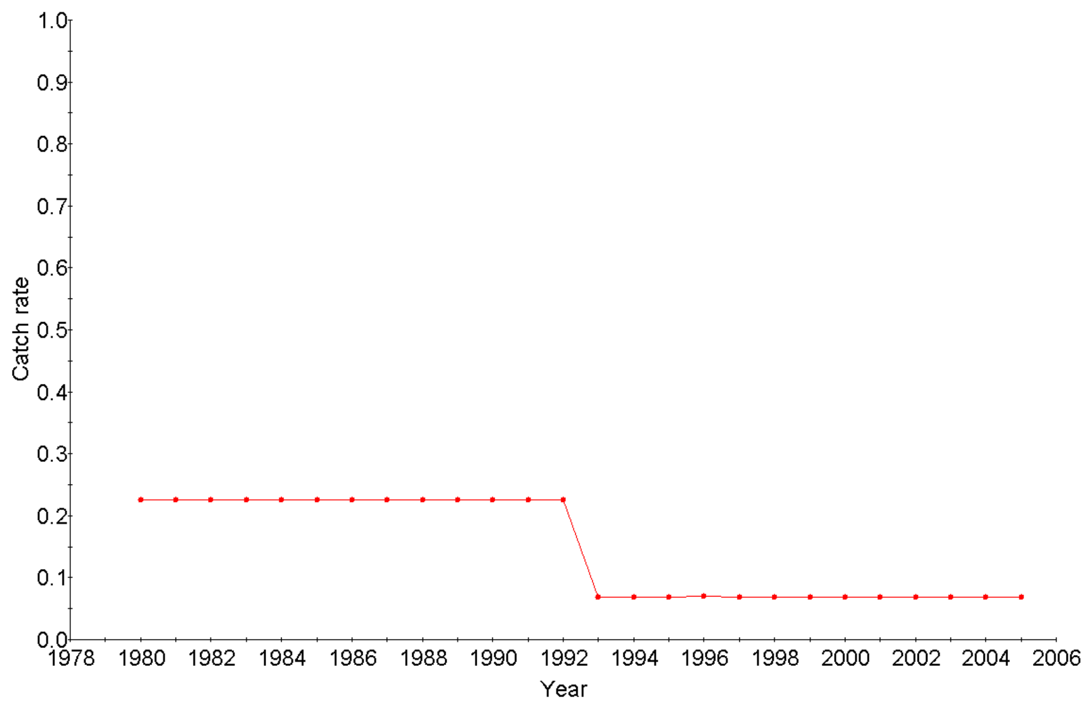

Figure 12: North Santiam River winter steelhead harvest rate.

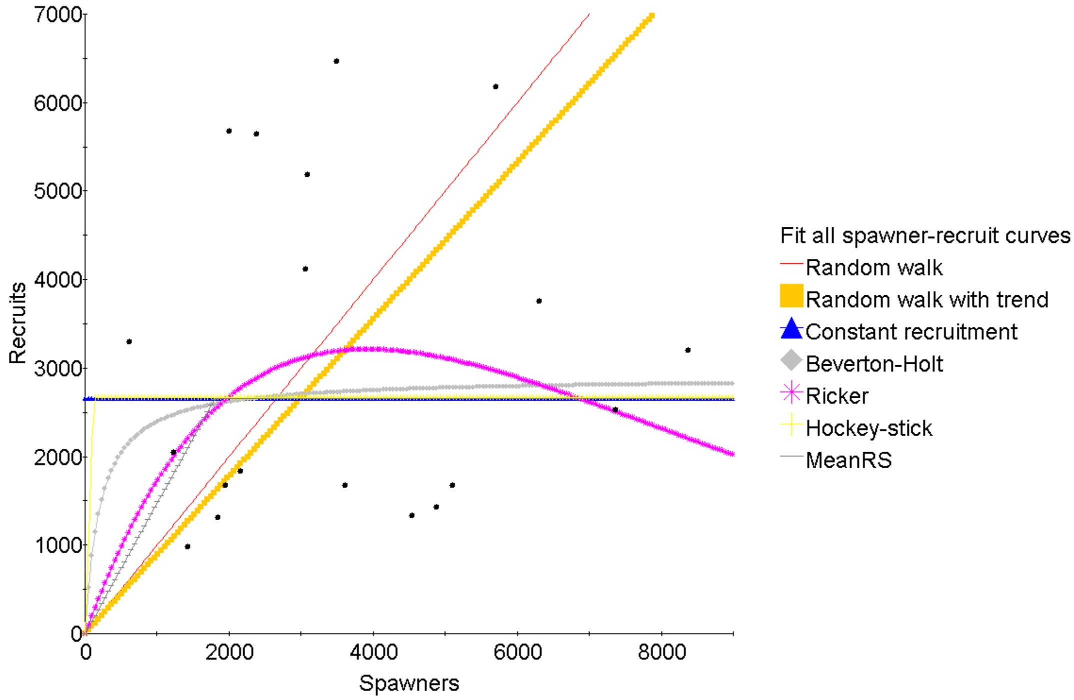

Figure 13: North Santiam River winter steelhead escapement recruitment functions.

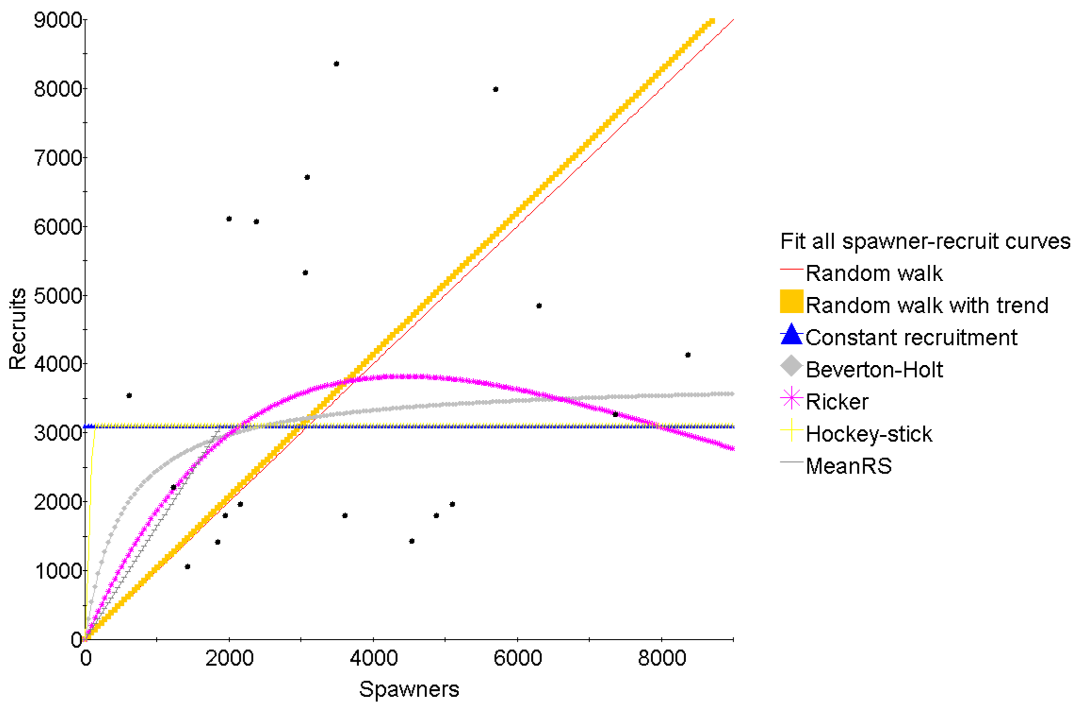

Figure 14: North Santiam River winter steelhead pre-harvest recruitment functions.

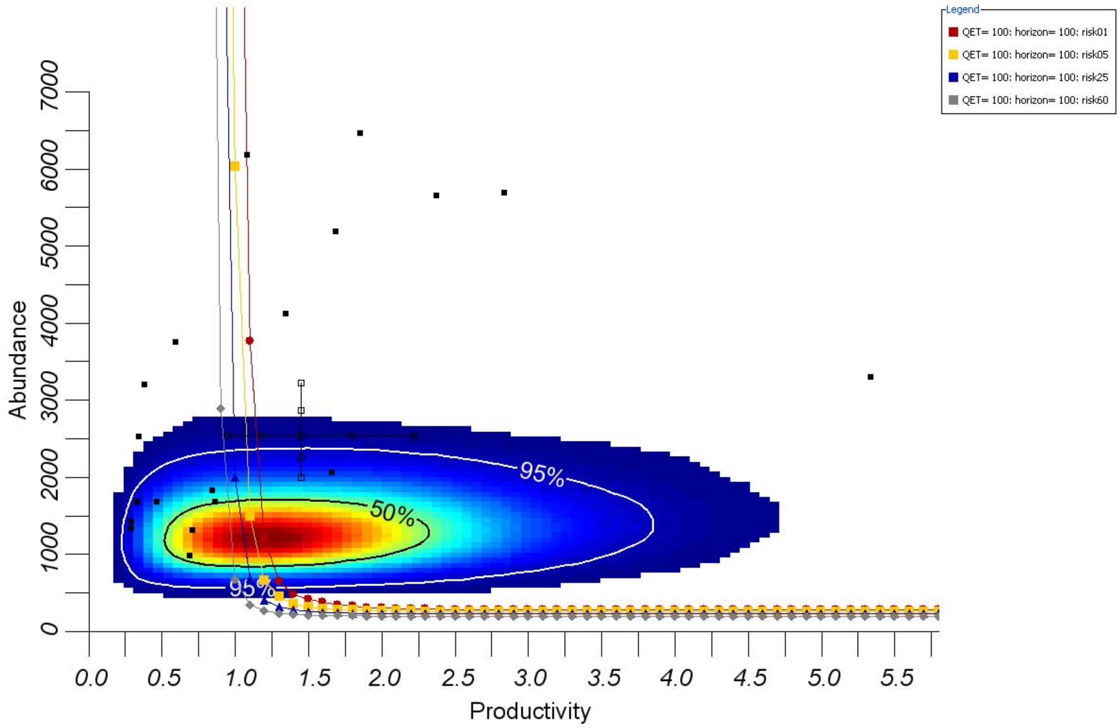

Figure 15: North Santiam River winter steelhead escapement viability curves.

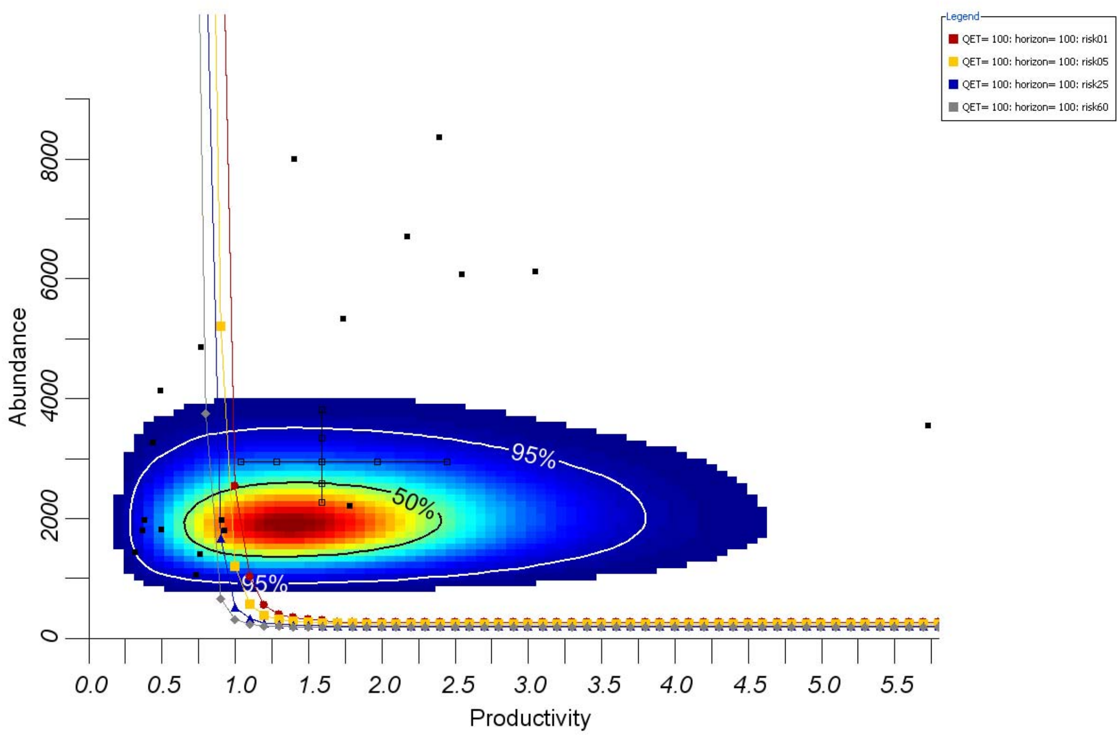

Figure 16: North Santiam River winter steelhead pre-harvest viability curves.

Table 5: North Santiam Winter Steelhead summary statistics. The geometric mean natural origin spawner abundance (highlighted) is in the "very low risk" viability criteria category. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1980-2005	1990-2005	1980-2005	1990-2005
Length of Time Series	26	16	26	16
Geometric Mean Natural Origin Spawner Abundance	2722 (2098-3531)	2109 (1485-2994)	NA	NA
Geometric Mean Recruit Abundance	2662 (1984-3571)	2187 (1341-3567)	3100 (2259-4256)	2350 (1441-3832)
Lambda	0.983 (0.786-1.231)	1.035 (0.705-1.519)	1.011 (0.81-1.262)	1.043 (0.696-1.562)
Trend in Log Abundance	0.98 (0.946-1.014)	1.065 (0.993-1.142)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.886 (0.59-1.331)	1.226 (0.619-2.429)	1.032 (0.692-1.539)	1.317 (0.665-2.609)
Geometric Mean Recruits per Spawner (broods < median spawner abudance)	1.474 (0.911-2.383)	1.368 (0.657-2.848)	1.642 (1.012-2.666)	1.47 (0.706-3.059)
Average Hatchery Fraction	0.124	0.109	NA	NA
Average Harvest Rate	0.147	0.098	NA	NA
CAPM median extinction risk probability (5th and 95th percentiles in parenthesis)	NA	NA	0.005 (0.000-0.075)	NA
PopCycle extinction risk	NA	NA	0.02	NA

Table 6: Escapement recruitment parameter estimates and relative AIC values for North Santiam winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.83 (0.68-1.18)	10.8
Random walk with trend	0.89 (0.66-1.33)	NA	0.82 (0.68-1.22)	12.4
Constant recruitment	NA	2658 (2120-3503)	0.59 (0.49-0.88)	0
Beverton-Holt	13.82 (3.19->30)	2898 (2298-4606)	0.59 (0.49-0.89)	1.8
Ricker	2.23 (1.24-4.06)	3213 (2716-5765)	0.62 (0.53-0.97)	3.7
Hockey-stick	23.56 (3.63->30)	2664 (2128-3521)	0.59 (0.49-0.88)	2
MeanRS	1.47 (1.07-2.07)	2662 (2130-3330)	0.43 (0.27-0.54)	13.5

Table 7: Pre-harvest recruitment parameter estimates and relative AIC values for North Santiam winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.81 (0.66-1.15)	6.9
Random walk with trend	1.03 (0.77-1.52)	NA	0.81 (0.67-1.2)	8.8
Constant recruitment	NA	3097 (2431-4192)	0.64 (0.53-0.95)	0
Beverton-Holt	6.96 (2.75->30)	3783 (2684-6085)	0.63 (0.53-0.94)	1.4
Ricker	2.34 (1.3-4.42)	3817 (3209-7548)	0.65 (0.56-1.01)	2.7
Hockey-stick	>30 (3.01->30)	3100 (2434-4290)	0.64 (0.53-0.96)	2
MeanRS	1.64 (1.18-2.3)	3100 (2430-3964)	0.47 (0.31-0.59)	10.4

Table 8: North Santiam winter steelhead SPMPC risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.782	0.851	1.000
Probability the population is above "Moderate risk of extinction" category	0.746	0.815	0.998
Probability the population is above "Viable" category	0.708	0.784	0.913
Probability the population is above "Very low risk of extinction" category	0.666	0.741	0.603

A&P – South Santiam

A time series of abundance sufficient for quantitative analysis is available for the South Santiam population (Appendix B). Descriptive graphs and viability analysis results are provided beginning with Figure 17 and in Table 9 and Table 12. The population is relatively large, with a long-term geometric mean natural origin spawner of 2,727 and a recent geometric mean of 2,302 (Table 9). These values are in the very low risk minimum abundance threshold (MAT) category.

The modeling results reflect the uncertainty in the input data and therefore in the population status. The pre-harvest viability curve analyses suggest that the population is probably viable if harvest levels remain low. The escapement viability curves suggest that the harvest pattern observed over the course of the time series is likely sustainable. Largely because of the high amount of measurement error in the input data, the “blobs” describing the current population status are relatively large and span all of the viability curve risk categories. This suggests caution in risk conclusions.

The CAPM analysis indicates that the population is viable as evidenced by a median value for predicted CRT probabilities of 0.005. The PopCycle analysis also suggests a very low risk (<0.01). We estimate that the population is most likely in the viable category, but the range of possibility spans the entire spectrum from very low risk to very high risk. The Oregon Native Fish Status report (ODFW 2005) gave this population a “pass” for abundance and productivity.

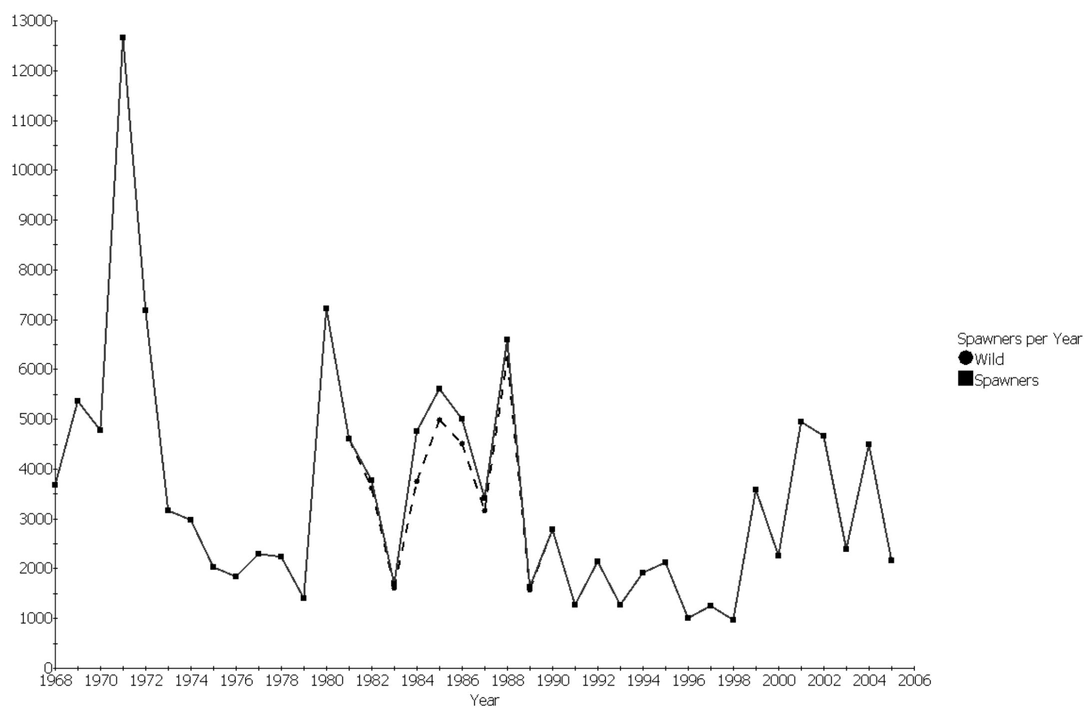

Figure 17: South Santiam Winter steelhead Abundance.

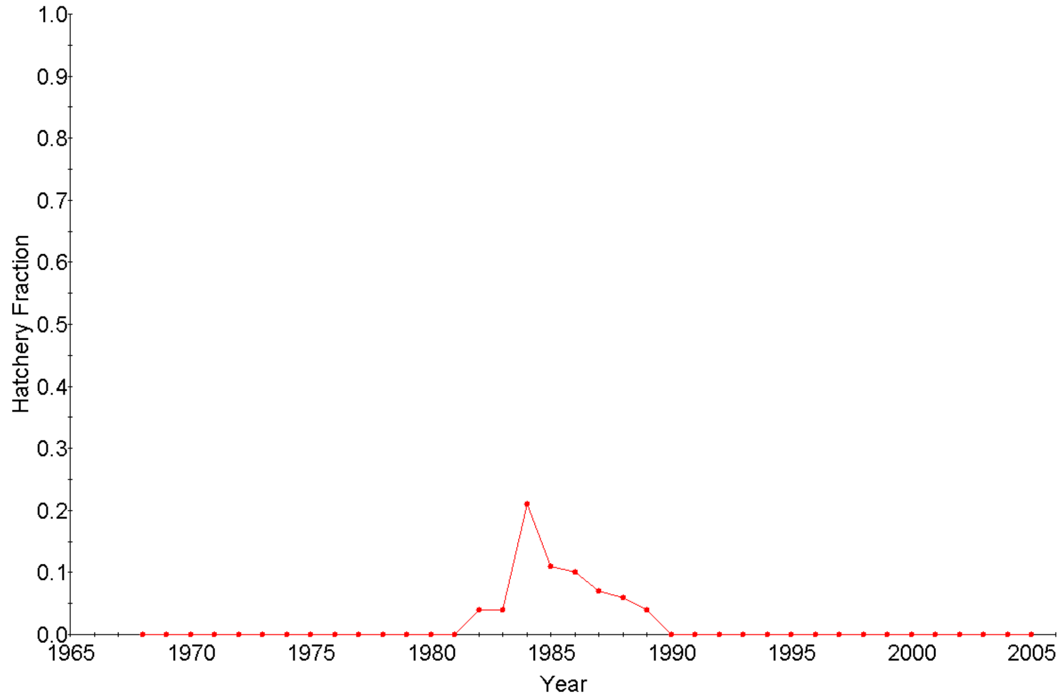

Figure 18: South Santiam River winter steelhead hatchery fraction.

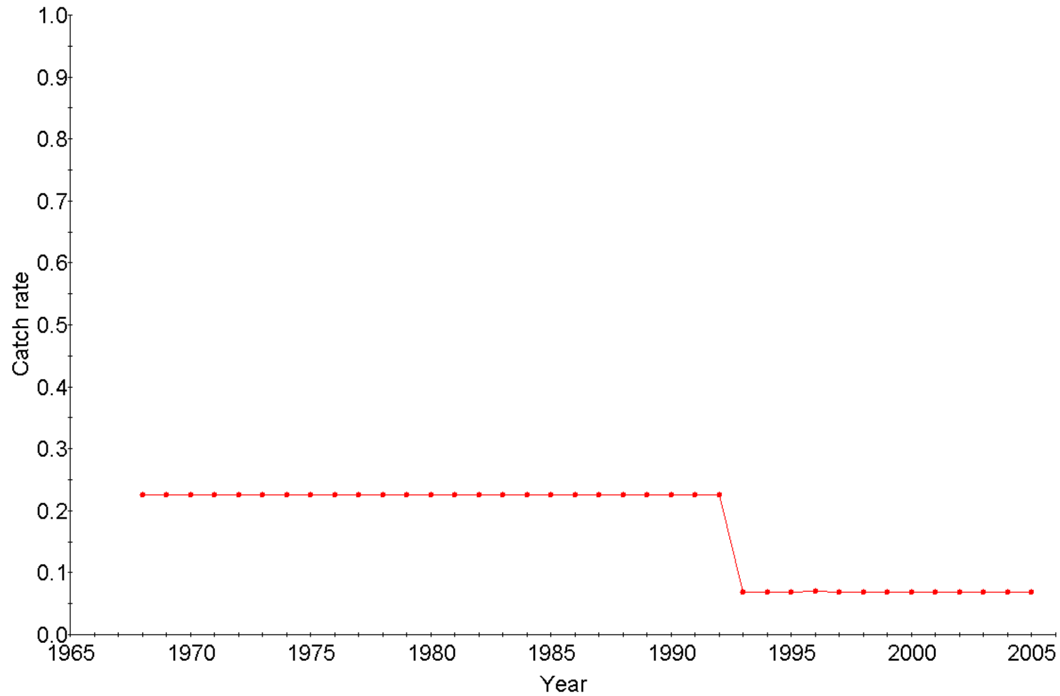

Figure 19: South Santiam River winter steelhead harvest rate.

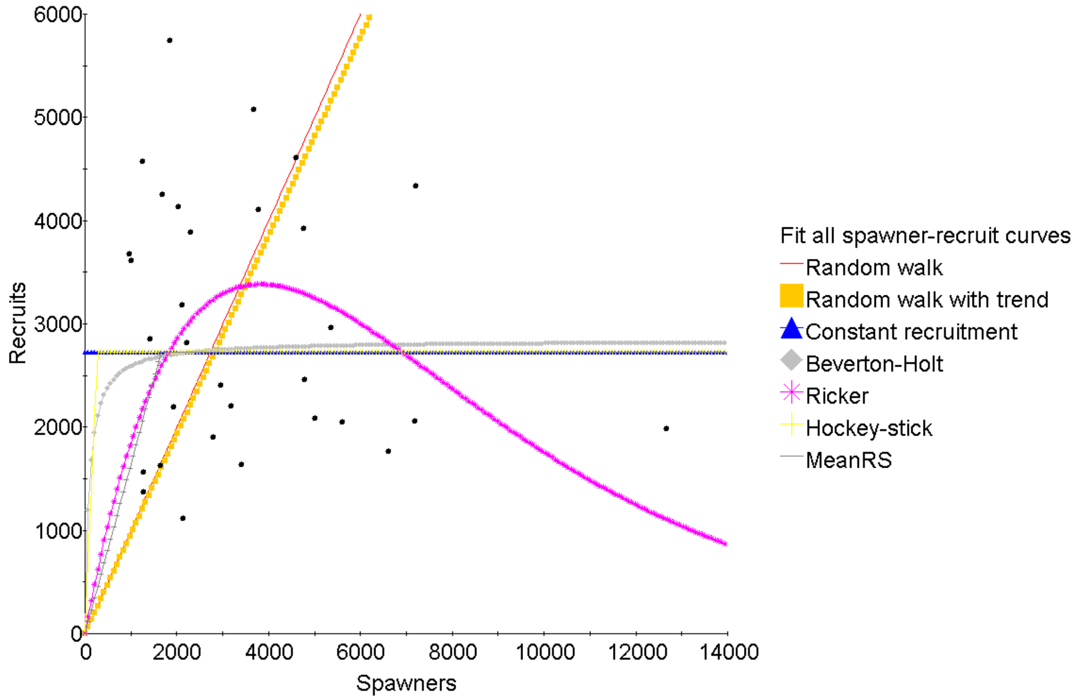

Figure 20: South Santiam River winter steelhead escapement recruitment functions.

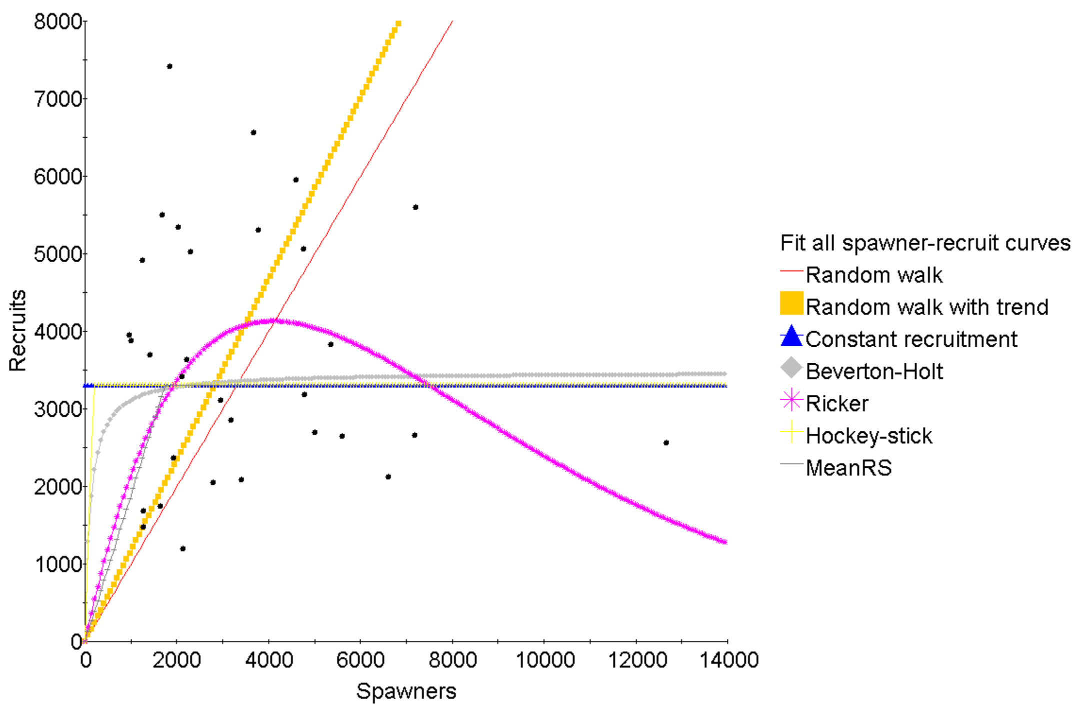

Figure 21: South Santiam River winter steelhead pre-harvest recruitment functions.

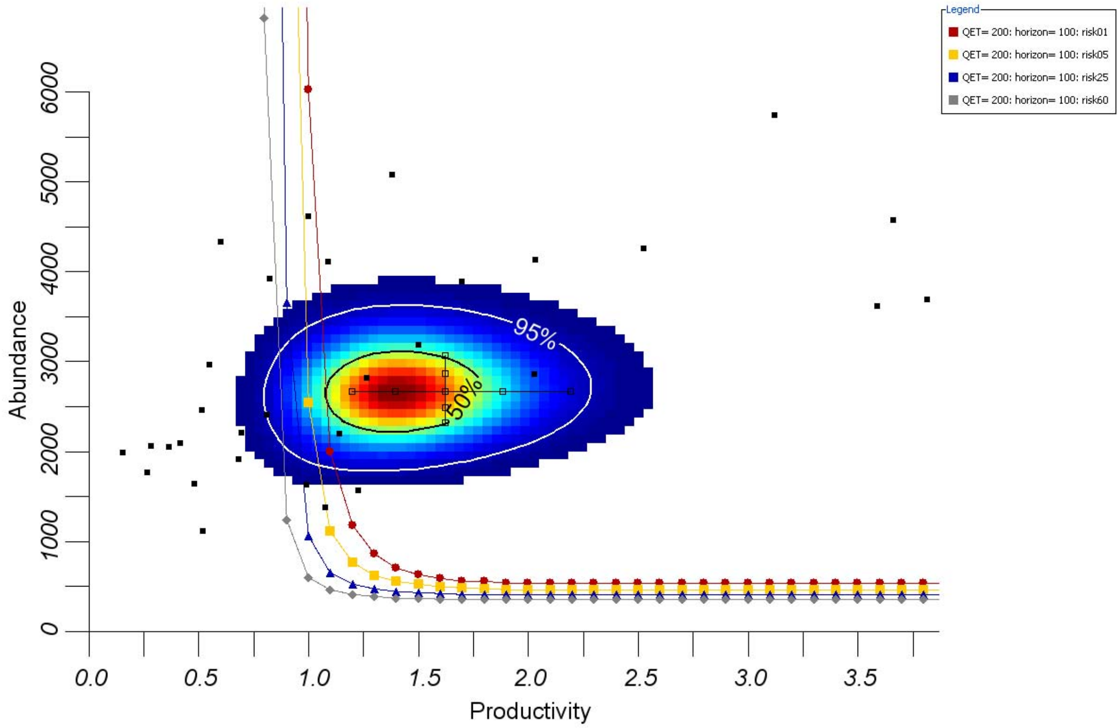

Figure 22: South Santiam River winter steelhead escapement viability curves.

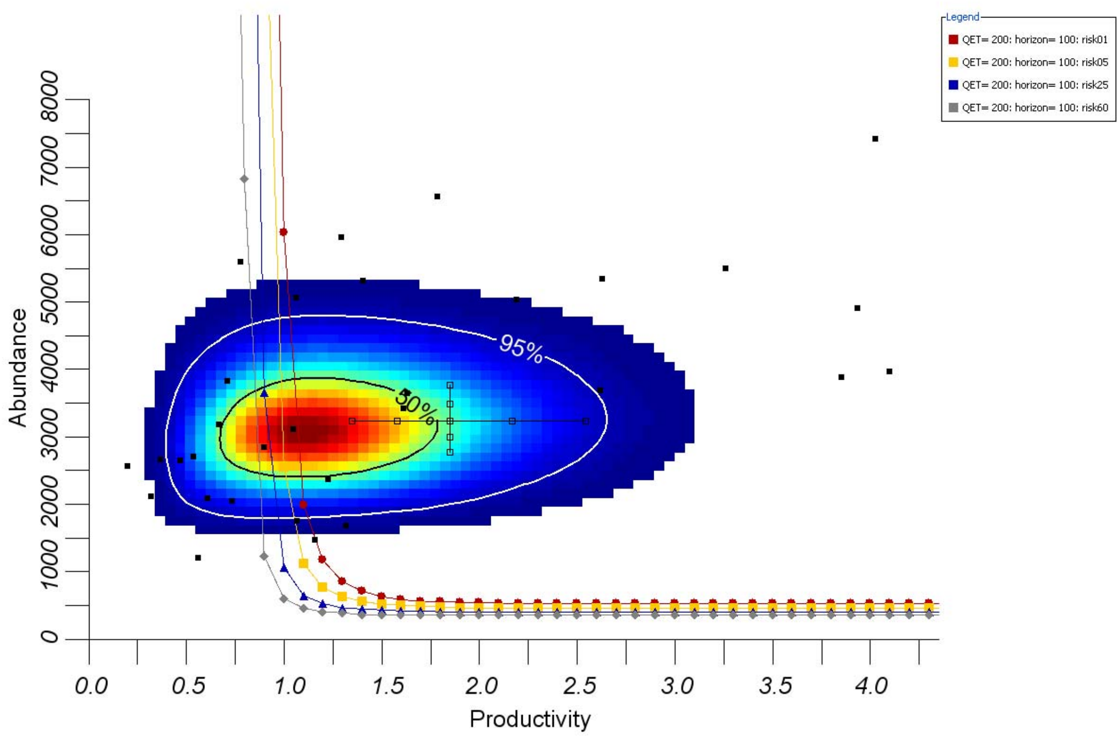

Figure 23: South Santiam River winter steelhead pre-harvest viability curves.

Table 9: South Santiam Winter Steelhead summary statistics. The geometric mean natural origin spawner abundance (highlighted) is in the "very low risk" viability criteria category. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	**Recent Years**	**Total Series**	**Recent Years**
Time Series Period	1968-2005	1990-2005	1968-2005	1990-2005
Length of Time Series	38	16	38	16
Geometric Mean Natural Origin Spawner Abundance	2862 (2350-3486)	2149 (1618-2853)	NA	NA
Geometric Mean Recruit Abundance	2727 (2328-3194)	2320 (1584-3399)	3309 (2786-3930)	2492 (1701-3651)
Lambda	0.976 (0.855-1.114)	1.052 (0.773-1.43)	1.014 (0.892-1.152)	1.06 (0.764-1.471)
Trend in Log Abundance	0.981 (0.965-0.998)	1.054 (0.997-1.115)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	0.962 (0.714-1.295)	1.509 (0.854-2.666)	1.167 (0.873-1.559)	1.621 (0.917-2.864)
Geometric Mean Recruits per Spawner (broods < median spawner abudance)	1.643 (1.191-2.266)	1.666 (0.908-3.055)	NA	NA
Average Hatchery Fraction	0.018	0	NA	NA
Average Harvest Rate	0.172	0.098	0.172	0.098
CAPM median extinction risk probability (5th and 95th percentiles in parenthesis)	NA	NA	0.005 (0.000-0.030)	NA
PopCycle extinction risk	NA	NA	0.02	NA

Table 10: Escapement recruitment parameter estimates and relative AIC values for South Santiam winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.8 (0.67-1.03)	37.4
Random walk with trend	0.96 (0.77-1.27)	NA	0.8 (0.68-1.06)	39.3
Constant recruitment	NA	2727 (2400-3140)	0.42 (0.36-0.56)	0
Beverton-Holt	>30 (7.01->30)	2839 (2540-3507)	0.42 (0.37-0.57)	2.3
Ricker	2.41 (1.86-3.18)	3381 (2988-4079)	0.47 (0.41-0.64)	8.2
Hockey-stick	10.21 (3.59->30)	2725 (2399-3132)	0.42 (0.36-0.56)	2
MeanRS	1.64 (1.29-2.09)	2727 (2401-3087)	0.22 (0.15-0.27)	71

Table 11: Pre-harvest recruitment parameter estimates and relative AIC values for North Santiam winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	0.79 (0.67-1.02)	31.5
Random walk with trend	1.17 (0.93-1.52)	NA	0.78 (0.66-1.03)	32.3
Constant recruitment	NA	3308 (2882-3863)	0.46 (0.39-0.61)	0
Beverton-Holt	>30 (6.18->30)	3477 (3105-4578)	0.46 (0.39-0.61)	1.9
Ricker	2.74 (2.1-3.66)	4130 (3632-5076)	0.49 (0.42-0.66)	5.4
Hockey-stick	18.13 (3.77->30)	3307 (2884-3856)	0.46 (0.39-0.61)	2
MeanRS	1.89 (1.47-2.43)	3309 (2884-3784)	0.24 (0.17-0.31)	59.3

Table 12: North Santiam winter steelhead CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.988	0.835	1.000
Probability the population is above "Moderate risk of extinction" category	0.975	0.794	0.997
Probability the population is above "Viable" category	0.956	0.751	0.960
Probability the population is above "Very low risk of extinction" category	0.913	0.678	0.637

A&P – Calapooia

A time series of abundance sufficient for quantitative analysis is available for the Calapooia population (Appendix B). Descriptive graphs and viability analysis results are provided beginning with Figure 24 and in Table 13 and Table 16. The population is small, with a long-term geometric mean natural origin spawner of 458 and a recent geometric mean of 339 (Table 9). These values are in the moderate risk minimum abundance threshold (MAT) category.

The modeling results reflect the uncertainty in the input data and therefore in the population status. The pre-harvest viability curve analyses suggest that the population is probably viable if harvest levels remain low. The escapement viability curves suggest that the harvest pattern observed over the course of the time series is likely sustainable. Largely because of the high amount of measurement error in the input data, the "blobs" describing the current population status are relatively large and span all of the viability curve risk categories. This suggests caution in risk conclusions.

The PopCycle modeling and the CAPM analysis indicates that the population is not viable and the predicted quasi-extinction probability over 100 years is 20% for PopCycle, and around 22% for CAPM. We estimate that the population is most likely in in the "moderate risk" category, but the range of possibility spans the entire spectrum from very low risk to very high risk. The Oregon Native Fish Status report (ODFW 2005) gave this population a "pass" for abundance and productivity.

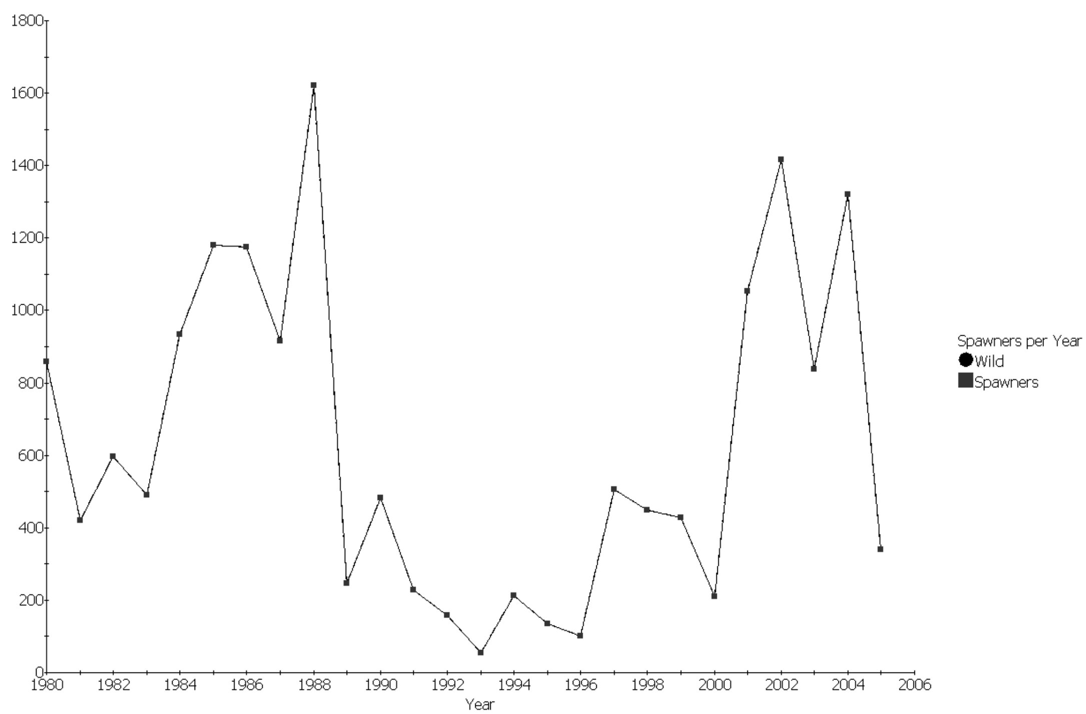

Figure 24: Calapooia River winter steelhead abundance.

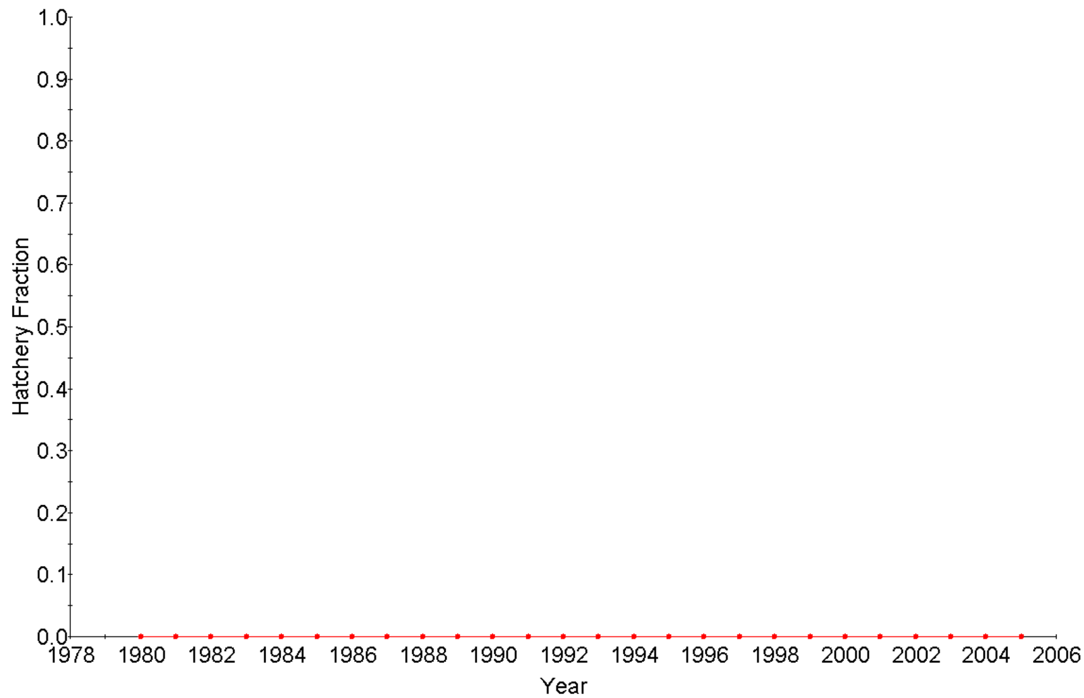

Figure 25: Calapooia River winter steelhead hatchery fraction.

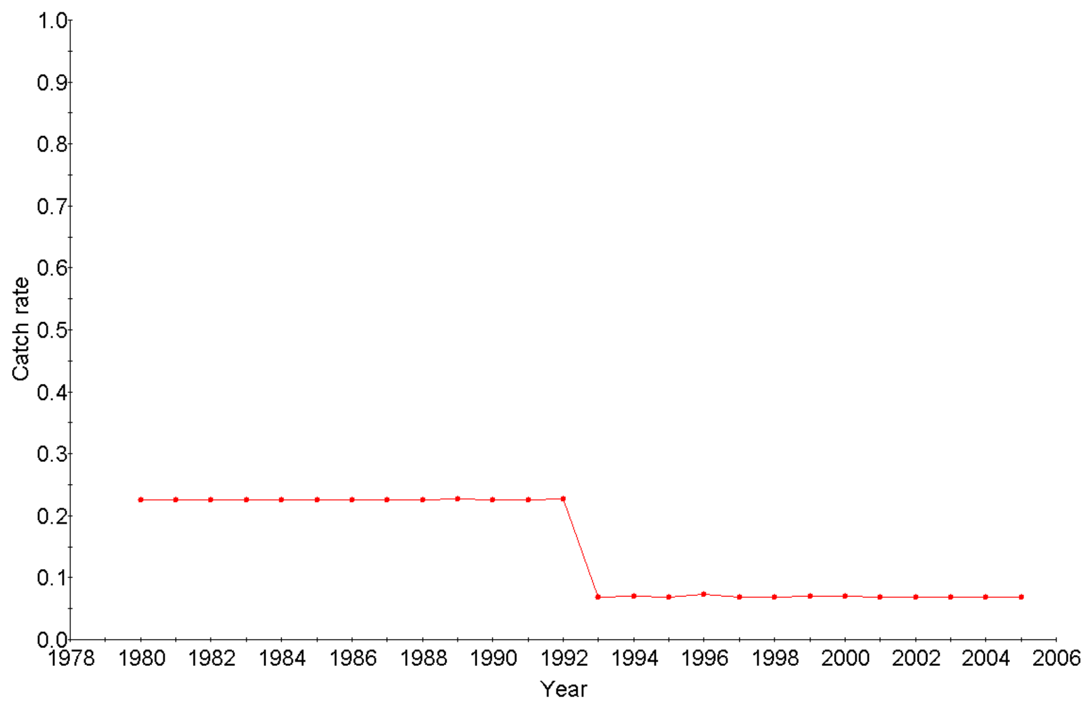

Figure 26: Calapooia River winter steelhead harvest rate.

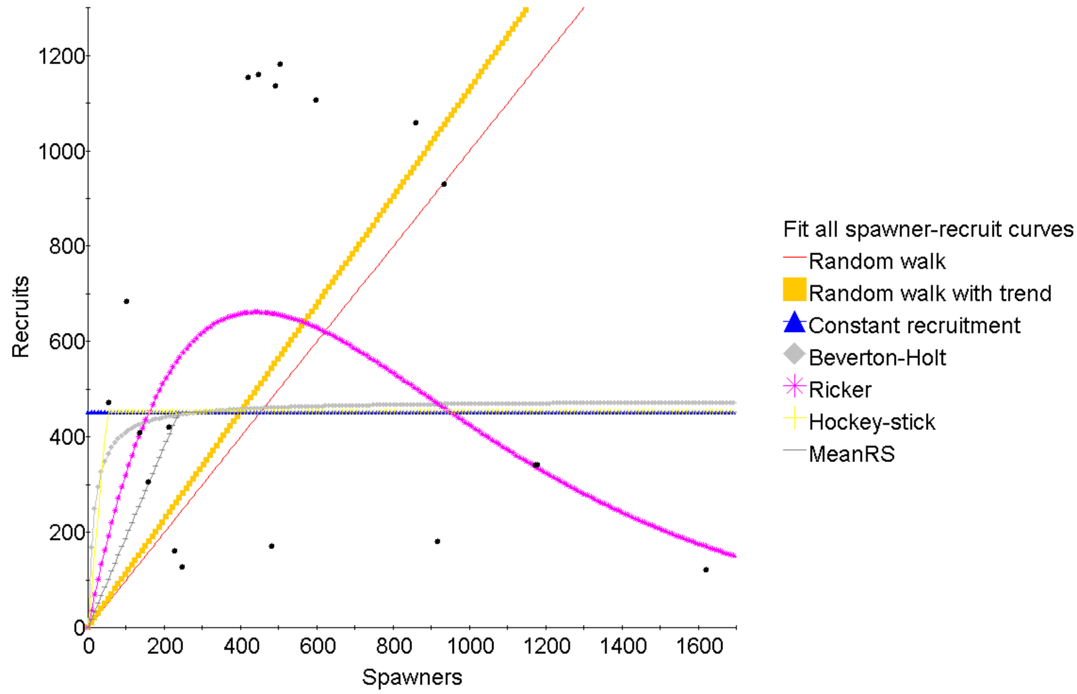

Figure 27: Calapooia River winter steelhead escapement recruitment functions.

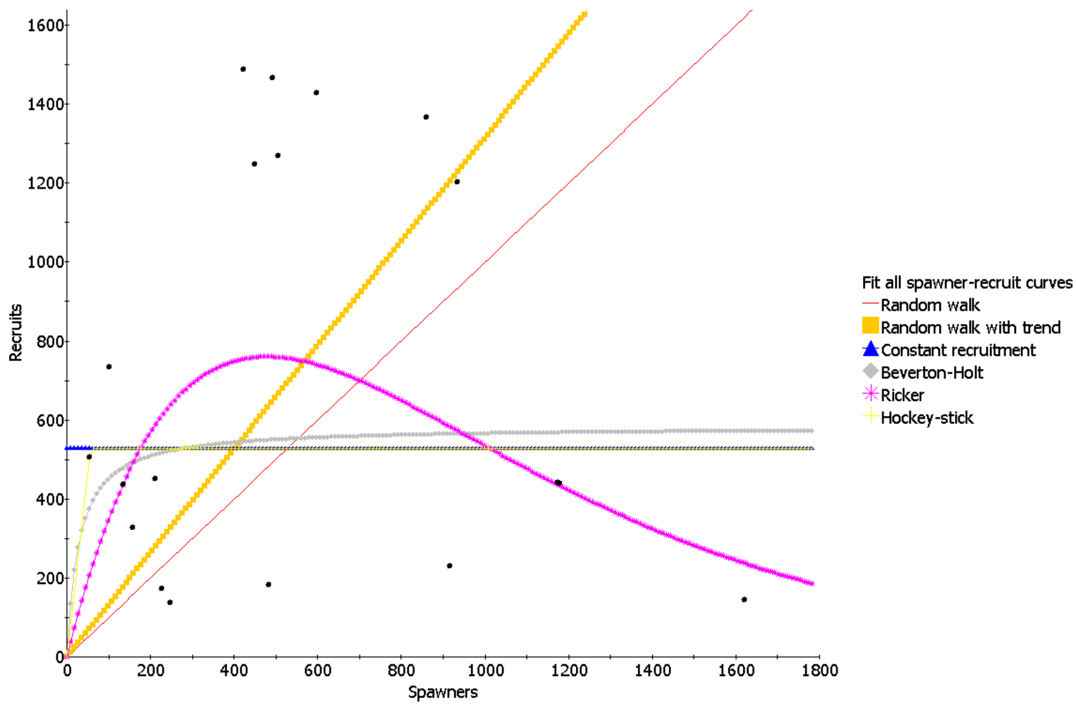

Figure 28: Calapooia River winter steelhead pre-harvest recruitment functions.

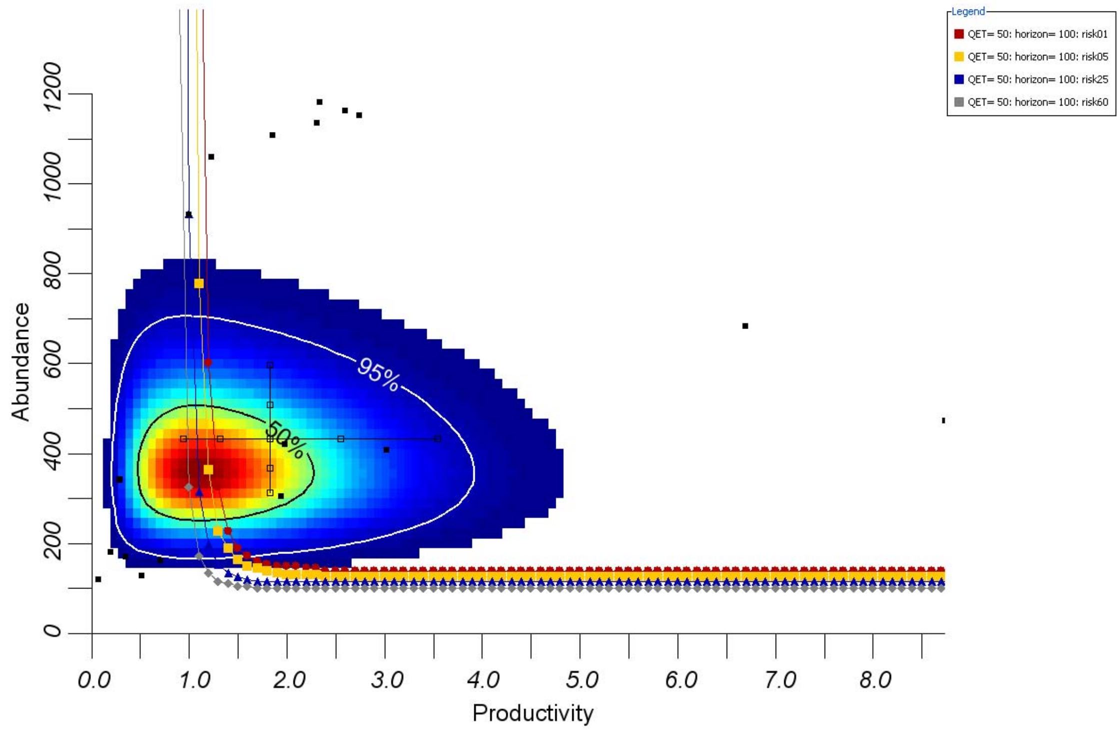

Figure 29: Calapooia River winter steelhead escapement viability curves.

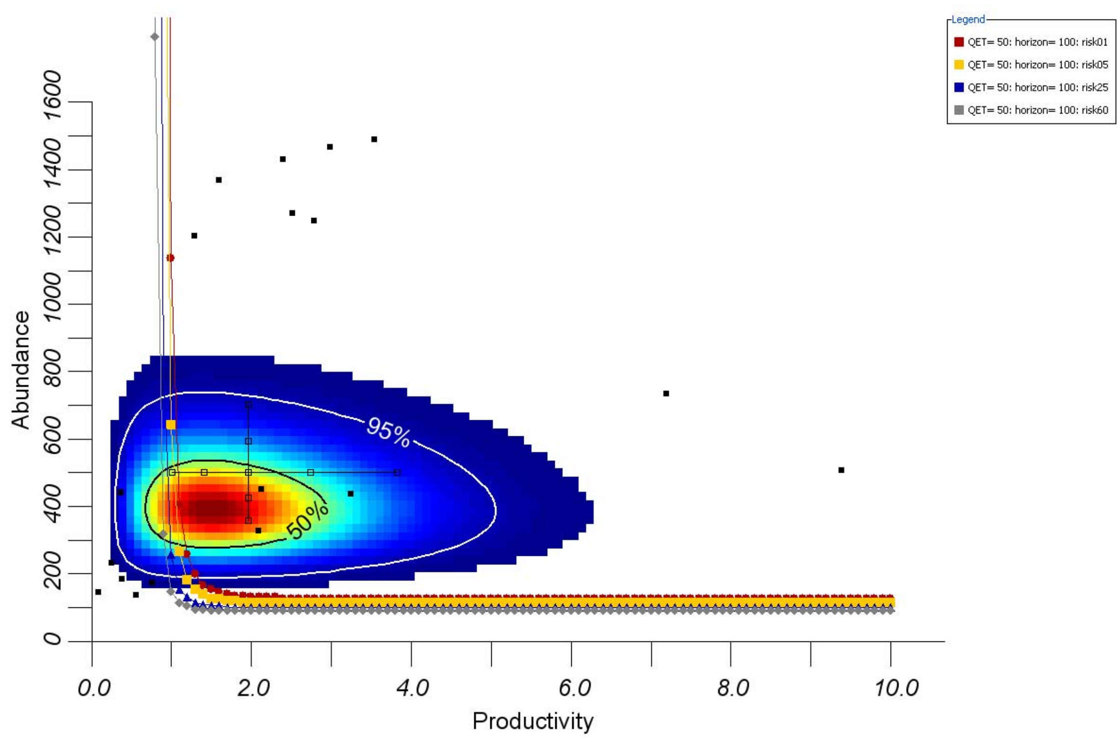

Figure 30: Calapooia River winter steelhead pre-harvest viability curves.

Table 13: Calapooia Winter Steelhead summary statistics. The geometric mean natural origin spawner abundance (highlighted in yellow) is in the "moderate risk" viability criteria category. The 95% confidence intervals are shown in parentheses.

Statistic	Escapement		Pre-harvest	
	Total Series	Recent Years	Total Series	Recent Years
Time Series Period	1980-2005	1990-2005	1980-2005	1990-2005
Length of Time Series	26	16	26	16
Geometric Mean Natural Origin Spawner Abundance	458 (319-657)	339 (206-560)	NA	NA
Geometric Mean Recruit Abundance	453 (304-675)	441 (253-769)	529 (350-799)	474 (272-826)
Lambda	1.023 (0.743-1.409)	1.128 (0.959-1.328)	1.053 (0.772-1.436)	1.136 (0.941-1.372)
Trend in Log Abundance	0.987 (0.94-1.037)	1.13 (1.035-1.235)	NA	NA
Geometric Mean Recruits per Spawner (all broods)	1.126 (0.617-2.055)	2.163 (1.007-4.646)	1.315 (0.731-2.365)	2.324 (1.082-4.99)
Geometric Mean Recruits per Spawner (broods < median spawner abudance)	1.905 (0.901-4.024)	2.799 (1.069-7.329)	2.084 (0.981-4.43)	3.007 (1.149-7.872)
Average Hatchery Fraction	0.000	0.000	NA	NA
Average Harvest Rate	0.148	0.099	NA	NA
CAPM median extinction risk probability (5th and 95th percentiles in parenthesis)	NA	NA	0.22	NA
PopCycle extinction risk	NA	NA	0.20	NA

Table 14: Escapement recruitment parameter estimates and relative AIC values for Calapooia winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.22 (0.99-1.73)	14.3
Random walk with trend	1.13 (0.75-2.09)	NA	1.21 (1.01-1.81)	16.1
Constant recruitment	NA	453 (338-670)	0.8 (0.67-1.19)	0.5
Beverton-Holt	>30 (4.53->30)	477 (370-886)	0.8 (0.68-1.21)	2.5
Ricker	4.05 (2.44-8.07)	661 (527-1103)	0.76 (0.64-1.21)	0
Hockey-stick	9.01 (4.46->30)	452 (337-676)	0.8 (0.67-1.2)	2.5
MeanRS	1.9 (1.13-3.17)	453 (332-613)	0.85 (0.53-1.08)	7.6

Table 15: Pre-harvest recruitment parameter estimates and relative AIC values for Calapooia winter steelhead. The 95% probability intervals on parameters are shown in parentheses. The model that is the "best" approximation (i.e., relative AIC = 0) is shown in bright green. Models that nearly indistinguishable from best (i.e., relative AIC <2) are shown in darker green. Models that are possible, but less likely, contenders as best (i.e., 2 < relative AIC < 10) are shown in yellow. Models that are very unlikely to be the best approximating model (i.e., relative AIC > 10) are not highlighted (i.e., white background).

Model	Productivity	Capacity	Variance	Relative AIC
Random walk	NA	NA	1.22 (0.99-1.73)	12.8
Random walk with trend	1.31 (0.88-2.4)	NA	1.18 (0.99-1.76)	13.8
Constant recruitment	NA	529 (392-802)	0.83 (0.69-1.23)	0.4
Beverton-Holt	25.19 (4.2-28.6)	574 (438-1169)	0.83 (0.7-1.26)	2.3
Ricker	4.33 (2.55-9.17)	756 (607-1348)	0.78 (0.67-1.26)	0
Hockey-stick	9.38 (4.21-28.65)	533 (390-817)	0.84 (0.7-1.25)	2.4
MeanRS	2.08 (1.22-3.5)	529 (384-721)	0.87 (0.53-1.11)	6.8

Table 16: Calapooia winter steelhead CAPM risk category and viability curve results.

Risk Category	Viability Curves		CAPM
	Escapement	Pre-harvest	
Probability the population is not in "Extirpated or nearly so" category	0.744	0.895	0.997
Probability the population is above "Moderate risk of extinction" category	0.692	0.880	0.817
Probability the population is above "Viable" category	0.630	0.849	0.072
Probability the population is above "Very low risk of extinction" category	0.590	0.824	0.003

A&P – Criterion Summary

The most probable risk classification was 'moderate' risk for the Molalla and Calapooia populations and 'low' risk for the North and South Santiam populations. However, as illustrated in Figure 31 by the tall aspect of the diamond symbols, the evaluation results for UW steelhead populations reflect a high degree of uncertainty. Because of this assessment uncertainty, the possible (not probable) risk classifications range from very low to very high for all four populations. In light of this and the most probable classification results, we conclude that overall, these population results place the ESU in the 'moderate' risk category with respect to the A&P criterion.

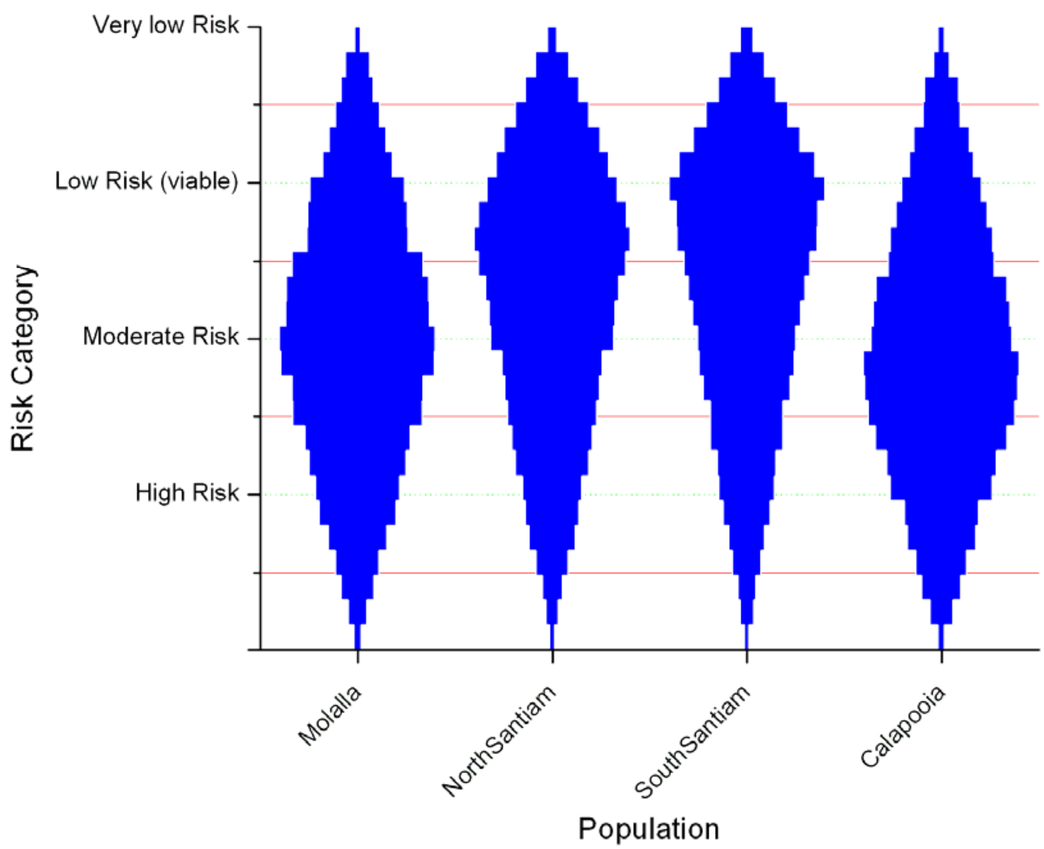

Figure 31: Graph of abundance and productivity risk estimate for Upper Willamette Steelhead.

III. Spatial Structure

SS – Molalla

Land use and road building has limited access of anadromous fish to many higher order tributaries in the Molalla and Pudding rivers but no large mainstem fish barriers are present. On a stream mile basis this impairment is significant (Figure 32). However, small high order streams that comprise most of the blocked area were not highly productive winter steelhead habitats. ODFW (2005) reports that virtually all of the historically significant steelhead habitat remains accessible. Habitat degradation due to land use has reduced water quality and the availability of suitable rearing habitat for steelhead in the Molalla River.

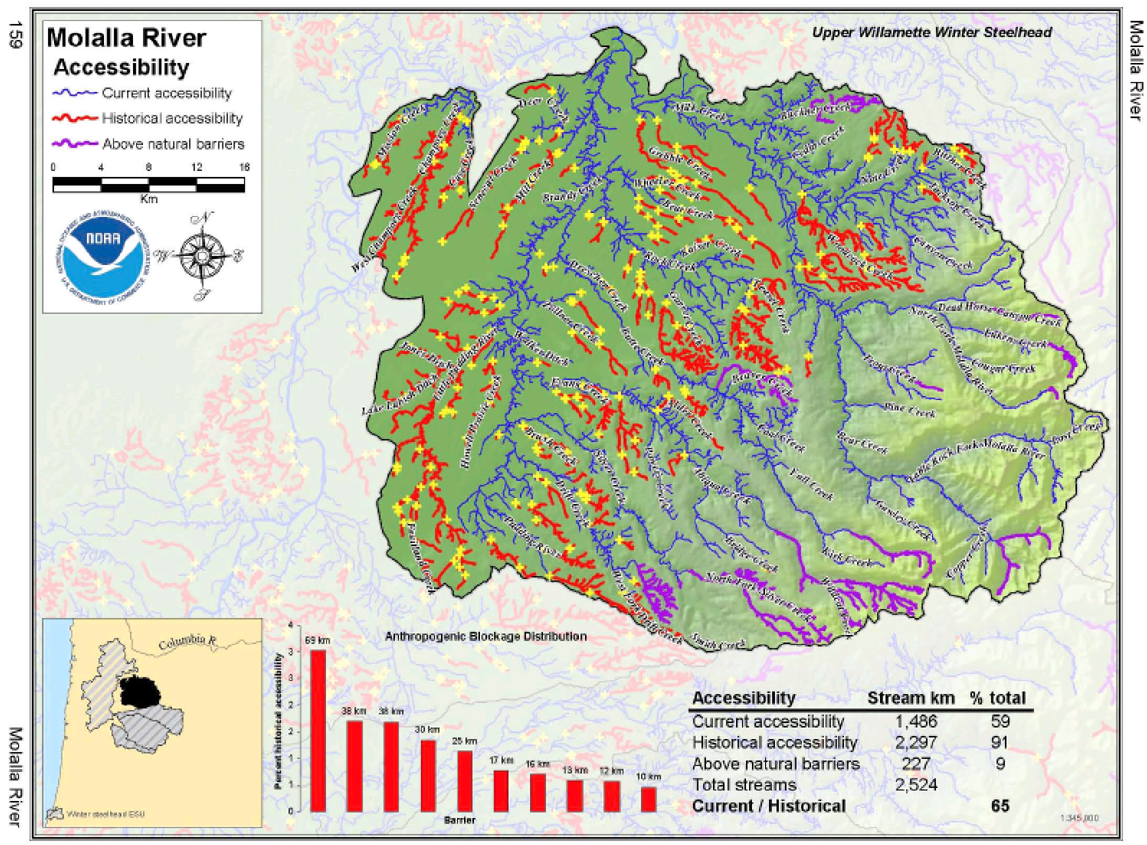

Figure 32: Molalla River winter steelhead current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – North Santiam

Access to large portions of historically productive steelhead habitat has been blocked by Detroit Reservoir (Figure 33). ODFW estimates that 46% of the historically suitable habitat for steelhead is now inaccessible (ODFW 2005).The blocked areas historically included some of the most productive habitats in this system although productive habitat remains in the Little North Santiam River. The watershed score for spatial structure was further reduced to account for habitat declines in the remaining accessible habitat.

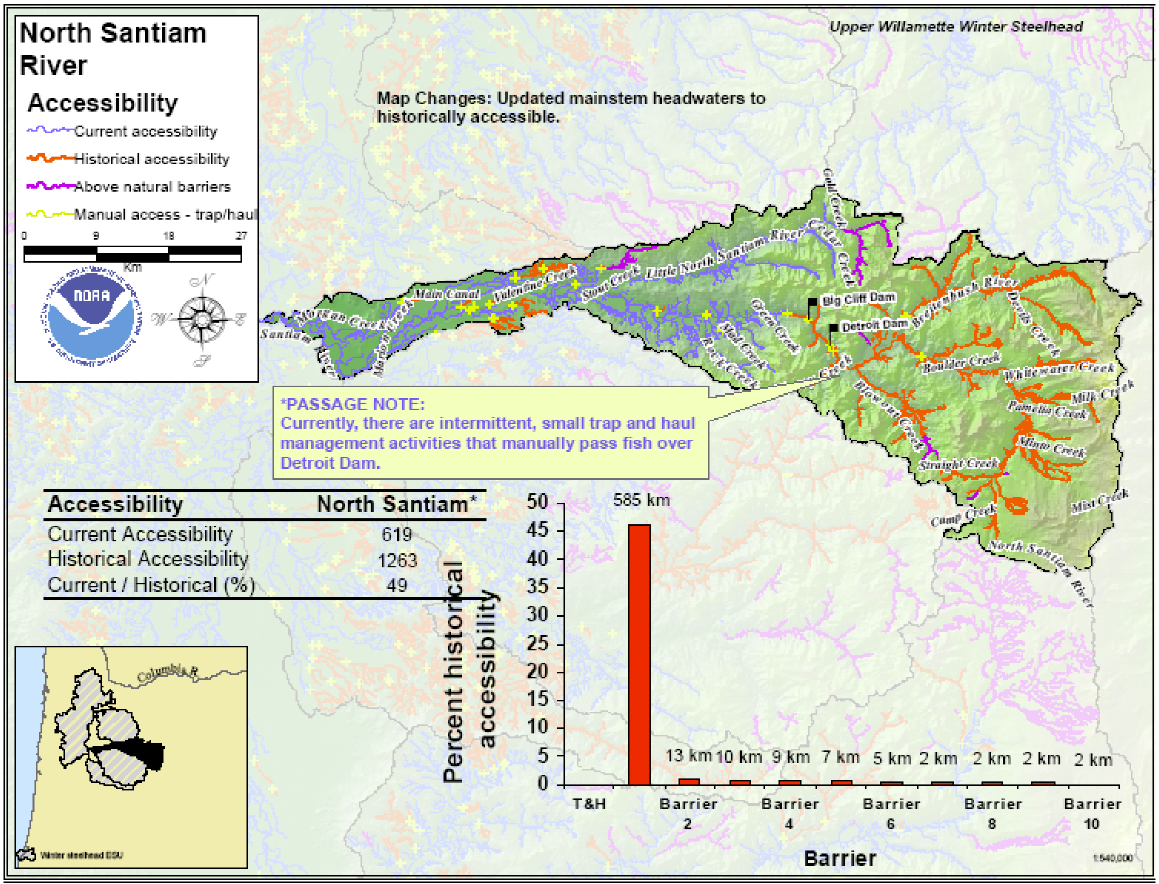

Figure 33: North Santiam River winter steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – South Santiam

Access to the upper South Santiam has been blocked by Foster and Green Peter Dams although significant steelhead habitat remains in other portions of this system (Figure 34). In the case of Foster Dam, a trap and haul program is currently moving fish upstream of this blockage. There is no passage of steelhead above Green Peter Dam and so the historical production area upstream of this dam is no longer accessible. ODFW (2005) estimates that 17% of the historically suitable habitat for steelhead is now inaccessible. Access has also been impaired in the upper reaches of many small low-elevation tributaries although these areas likely did not historically support high densities of steelhead. Habitat degradation due to land use and flow regulation has reduced water quality and the availability of suitable rearing habitat for steelhead in the South Santiam River.

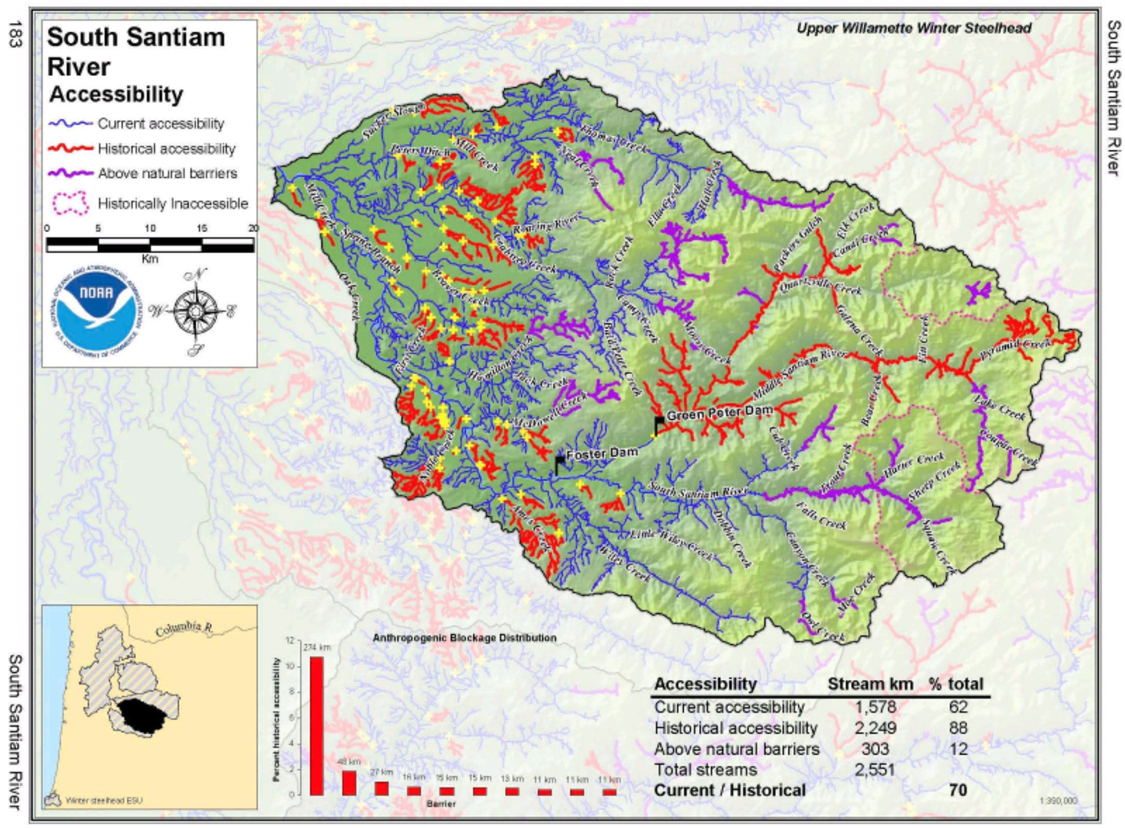

Figure 34: South Santiam River winter steelhead current and historical accessibility (from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use.

SS – Calapooia

Steelhead returning to the Calapooia basin do not have accessibility to potential production areas that they had historically. (Figure 35). In addition, habitat degradation has substantially reduced the spatial distribution of suitable steelhead habitat within the accessible area. However, some of the blocked habitat may not have been historically used by winter steelhead.

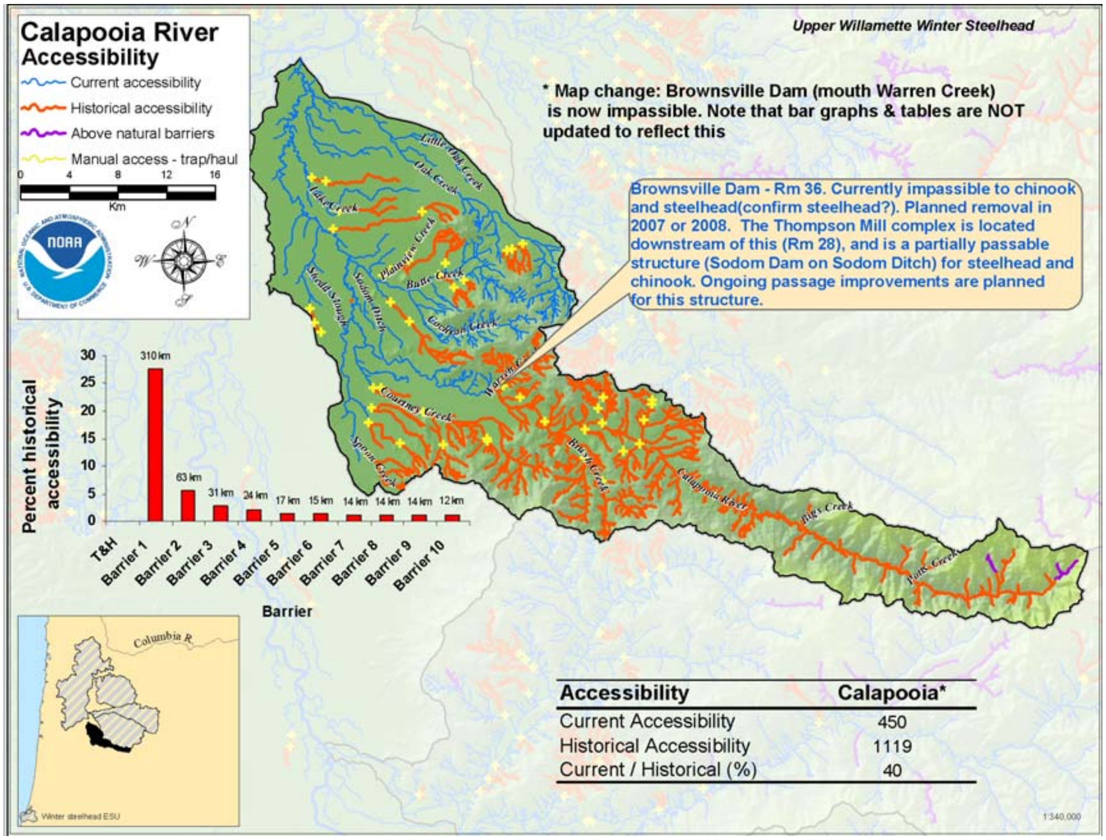

Figure 35: Calapooia River winter steelhead current and historical accessibility (updated by Sheer 2007 from Maher et al. 2005). As described in the Introduction (Part 1), these maps depict *access* (i.e., where fish could swim) and not necessarily habitat that fish would use. NOTE: This map incorrectly indicates that steelhead are blocked by Brownsville Dam on the mainstem Calapooia. Although the dam is a barrier for spring Chinook, it is generally considered passable by steelhead.

SS – Criterion Summary

The percentage of historically accessible habitat lost due to human activities exceeds 30% for all of the populations within this ESU (Figure 36). SS scores for each population were adjusted, where applicable, on the basis of two factors: 1) the suitability/quality of the blocked habitat with respect to Chinook production and 2) the degree to which the remaining accessible habitat has been degraded from historical conditions. The adjustments and final SS scores for each population are presented in Table 18.

For the SS criterion the most probable risk category for three of the four populations is either 'moderate' or 'high' (Figure 37). Only the Molalla population received a most probable risk classification of 'low'. Although there is a degree of uncertainty associated with these scores, overall we conclude that the most probable risk classification for these populations (and ESU) with respect to the SS criterion is 'moderate'.

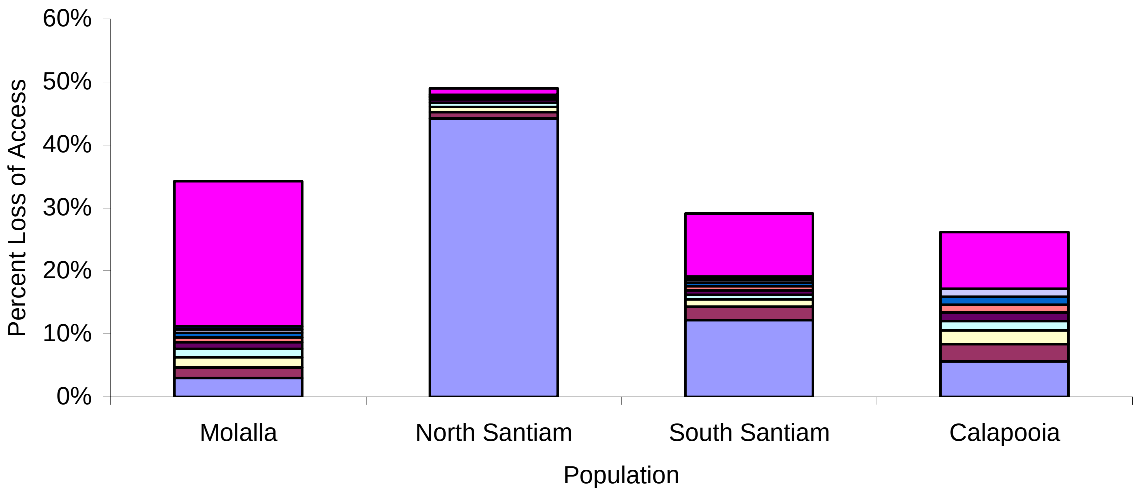

Figure 36: Percent loss in Upper Willamette winter steelhead accessibility due to anthropogenic blockages (based on Maher et al. 2005). Each color represents a blockage ordered from largest to smallest (bottom-up). The topmost blockages (i.e., the pink segment of the Calapooia bar) represent a collection of many smaller blockages. Note that in the Upper Willamette winter steelhead some of these pools of smaller blockages represent a larger percent loss of access than the largest blockage in that same population. The figure considers Brownsville Dam in the Calapooia passable for steelhead (i.e., it does NOT match the map in Figure 35).

Table 17: UW steelhead spatial structure scores.

Population	Base Access Score	Adjustment for Large Single Blockage	Adjusted Access Score	SS Rating Considering: Access Score, Historical Use Distribution, and Habitat Degradation	Confidence in SS rating
Molalla	2	no	2	3	M
North Santiam	1	yes	0.5	1	M
South Santiam	2	no	2	2	L
Calapooia	1	no	1	1	M

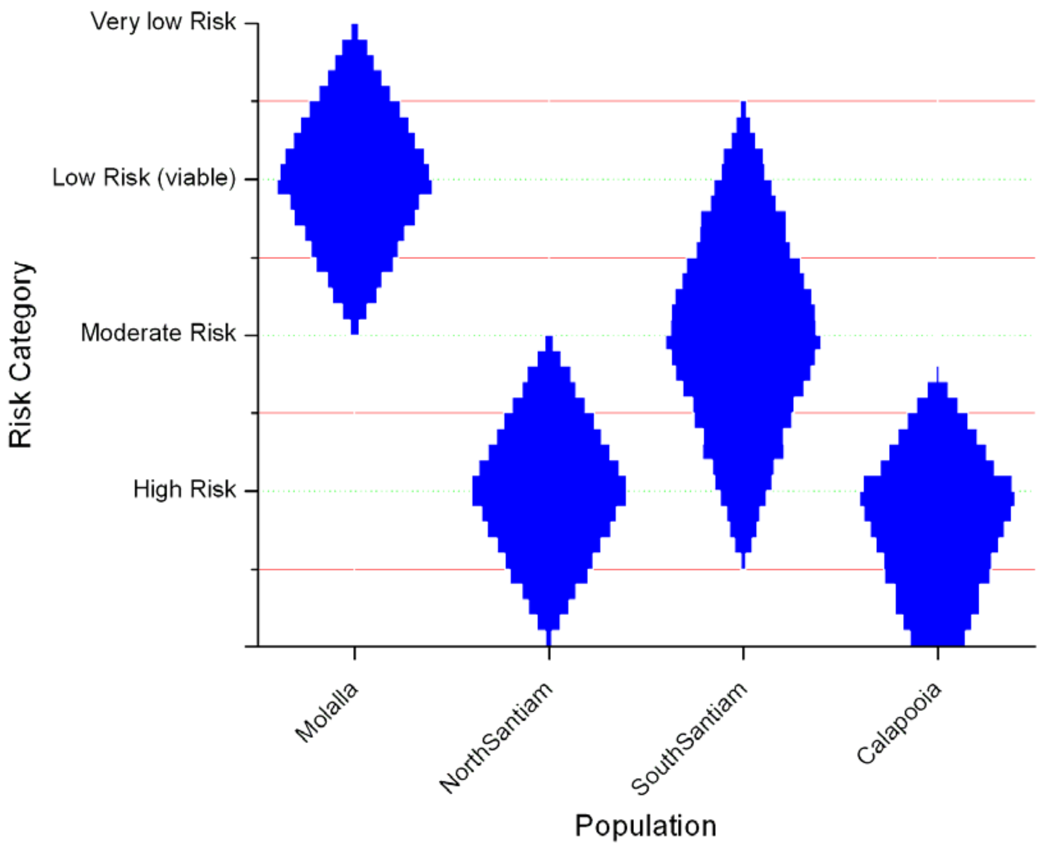

Figure 37: Summary of spatial structure risk scores for Upper Willamette steelhead.

IV. Diversity

DV – Background and Overview

Late-run winter steelhead are considered the only native run in the Upper Willamette River ESU. The same flow conditions at Willamette Falls (RKm 37) that only permitted access to spring-run chinook salmon also provided an isolating mechanism for this run time. Howell et al. (1985), however, reported that the peak passage time at Willamette Falls for "wild" winter steelhead is in April. Redd counts for late-run winter steelhead in the Willamette River Basin are conducted in May (Howell et al. 1985). ODFW currently uses February 15th to discriminate between native and non-native Big Creek (early-run) winter steelhead at Willamette Falls (Kostow 1995). Recent analyses of returning steelhead adults indicate that Upper Willamette River late-winter steelhead mature at four different ages: age 4 (48%); age 5 (41%); age 6 (10%); and age 7 (6%).

It is generally agreed that steelhead did not historically emigrate farther upstream than the Calapooia River (Fulton 1970). Since the Willamette Falls were laddered in the early 1900s, hatchery stocks of summer and early-run winter steelhead have also been introduced into the Upper Willamette River from other ESUs. In 1982, it was estimated that 15% of the late-run winter steelhead ascending Willamette Falls were of hatchery origin (Howell et a. 1985). Counts of native late-run steelhead moving past Willamette Falls had a 5-year geometric mean abundance of just over 3,000 fish (data through 1997) (ODFW 1998). All of the hatchery programs for steelhead were discontinued in the late 1990s, except for summer steelhead programs in the South Santiam, McKenzie, and Middle Fork Willamette River, where winter steelhead are not native.

The predominant tributaries to the Willamette River that historically supported steelhead include the Molalla (RKm 58), Calapooia (RKm 192), Santiam (RKm 174)—all drain the Cascades to the east (Mattson 1948, Nicholas 1995). The status of *O. mykiss* in basins that drain the Coastal Range is the subject of considerable debate. Although anadromous *O. mykiss* may occur in the Westside tributaries, it is generally thought that these are the progeny of introduced Lower Columbia River steelhead, or representative of sporadic occupation by native late-run steelhead. In this document and in the review of historical populations (Myers et al. 2003) spawning aggregations in the Westside tributaries are not considered demographically independent populations.

DV – Molalla River Winter-Run Steelhead

Life History Traits **–** Winter steelhead ascend Willamette Falls from December through May, with a peak in March and April (Firman et al. 2005). Although Big Creek (non-native) early-winter steelhead are no longer released in the Upper Willamette River, the presence of feral early-run fish may influence the characterization of late-winter run life history traits. Given the similarity in life history characteristics between early and late-winter steelhead it is difficult to identify whether there has been a change in late-winter life history characteristics or whether early-winter fish have been misclassified as late-winter fish. Score = NA

Effective Population Size **–** Recent escapement estimates for Molalla River steelhead are in the low thousands of fish (Goodson 2005). In general, several hundred fish returned annually to the Molalla River, except in the mid-1990s when escapement was below 100. Additionally, earlier escapement estimate did not distinguish between natural and hatchery-origin fish. Score = 3.

Hatchery Impacts

Hatchery Domestication – Releases of hatchery-origin late-winter fish were suspended in the late 1990s. Historically, hatchery production may have represented a substantial fraction of production. Genetic analyses indicate a close genetic affinity between winter steelhead populations in the Santiam, Molalla (North Fork), and Calapooia Rivers. Steelhead that are the progeny of summer-run and early winter-run steelhead are genetically distinct from presumptive native steelhead. Differences in spawn timing among these run-times may limit (but not eliminate) the potential for interbreeding. Score = NA.

Hatchery Introgression – The Molalla River has received introductions of three distinct runs of steelhead: native late-run winter steelhead, introduced early-run steelhead (from the Lower Columbia River), and introduced Skamania Hatchery summer-run steelhead (Chilcote 1997). Releases of the early-run steelhead into the Molalla River were discontinued in 1997 (Chilcote 1997), although some natural production of early-run winter steelhead may still occur. Overall, hatchery contribution to escapement has been near 40%, although currently it is near 0%. Score = 2-3.

Synthetic Approach – Hatchery releases into the Molalla River were discontinued in the late 1990s. Prior to that time, there were releases of non-native early-winter steelhead (Big Creek Hatchery) and summer steelhead (Skamania Hatchery), as well as late-winter steelhead from the North Santiam River. It is unclear to what extent these non-native releases have influenced the genetic diversity of the Molalla river steelhead. Currently the only strays into the Mollala River are likely from summer steelhead programs in the McKenzie and Santiam Rivers. Currently, $Ph<0.05$ although past hatchery introductions may have had an effect, especially when wild abundance was very low (<100) in the 1990s. Diversity persistence score = 3.0 - 4.0.

Anthropogenic Mortality **–** Historically, harvest rates for Molalla River steelhead has been near 20% (Chilcote 2001). With the recent introduction of selective fisheries this rate has fallen below 5%. Habitat changes in the Molalla River, Lower Willamette River, and mainstem Columbia River, may have influenced the expression of life history traits, especially juvenile traits. Score = 3.

Habitat Diversity **–** Historically, harvest rates for Molalla River steelhead has been near 20% (Chilcote 2001). With the recent introduction of selective fisheries this rate has fallen below 5%. Habitat changes in the Molalla River, Lower Willamette River, and mainstem Columbia River, may have influenced the expression of life history traits, especially juvenile traits. Score = ND/3.

Overall Score = 2.0. Many of the diversity concerns for this population are related to the legacy effects of hatchery releases from past years. There was considerable uncertainty in estimating these metrics. Additionally, habitat effects are largely unknown. Previously: 2004 TRT 1.51, 2004 ODFW Pass.

DV – North Santiam River Winter-Run Steelhead DIP Diversity Evaluation

Life History Traits **–** Winter steelhead ascend Willamette Falls from December through May, with a peak in March and April (Firman et al. 2005). Passage at Bennett Dam (North Santiam) normally peaks in April (Firman et al. 2005). Score = NA.

Effective Population Size **–** Overall, in recent years the escapement to the North Santiam River has included over 1,000 fish. Score = 3.

Hatchery Impacts

Hatchery Domestication – Surveys done in 1940 estimated that the run of steelhead was at least 2,000 fish (Parkhurst et al. 1950). Parkhurst also reports that larger runs of steelhead existed in the Breitenbush, Little North Santiam, and Marion Fork Rivers. Native steelhead were artificially propagated at the North Santiam Hatchery beginning in 1930, when a record 2,860,500 eggs (686 females x 4170 eggs/female) were taken (Wallis 1963). Production was somewhat intermittent during the 1940s. Attempts to capture all returning steelhead were unsuccessful due to the frequency and magnitude of spring floods (Wallis 1963). With the construction of Detroit Dam, the contribution of naturally-produced fish to escapement declined considerably. The release of hatchery propagated late-run winter steelhead was discontinued in 1998 (NMFS 1999). Recent escapements (through 1994) have averaged 1,800 fish, although the contribution of hatchery-origin fish was unknown (Busby et al. 1996).

Genetic analyses indicate a close affinity between winter steelhead populations in the Santiam, Molalla (North Fork), and Calapooia Rivers. Steelhead that are the progeny of summer-run and early winter-run steelhead are genetically distinct from presumptive native steelhead. Differences in spawn timing among these run-times may limit (but not eliminate) the potential for interbreeding.

PNI $\leq$ 0.84 (40 years hatchery production, PNI=0.80, 10 years no production) Fitness = 0.85. Score = 3.

Hatchery Introgression – Some summer steelhead are recovered in the North Santiam, and the effect of these fish on the native winter-run steelhead is unknown. Score = NA.

Synthetic Approach – Hatchery releases into the North Santiam River were discontinued in 1999. Prior to that time, there were releases of locally derived late-winter steelhead beginning in the1920s. Additionally, some summer run fish (Skamania Hatchery) are released in the North Santiam and South Santiam rivers. Currently, Ph<0.05 although past hatchery introductions may have had an effect. Diversity persistence score = 3.0 - 4.0.

Anthropogenic Mortality **–** Historically, harvest rates for North Santiam River steelhead has been near 20% (Chilcote 2001). With the recent introduction of selective fisheries this rate has fallen below 5%. Habitat changes in the Santiam River (especially thermal and flow conditions below Detroit Dam), Lower Willamette River, and mainstem Columbia River, may have influenced the expression of life history traits, especially juvenile traits. Score = 2-3**.**

Habitat Diversity **–** Habitat diversity loss is most severe for this DIP due to the loss of higher elevation spawning areas. Stream order was not determined. *(Order/Elevation)* Score = ND/1

Overall Score = 2.0. Major changes in habitat were thought to have had a significant effect on life history diversity. Other effects, such as the legacy of hatchery operations are difficult to estimate. Previously: 2004 TRT 1.46 , 2004 ODFW fail, 5 criteria met

DV – South Santiam Winter-Run Steelhead

Life History Traits **–** Winter steelhead are spawned at the South Santiam Hatchery during late April and May (Howell et al. 1985). The majority of returning adults are 2-ocean fish (84%), 3-ocean fish (16%) (Howell et al. 1985). Score = NA

Effective Population Size **–** ODFW considers the late-run winter steelhead in the South Santiam River to be one population, although Foster Dam may influence the distribution of spawners in the river (Chilcote 1997). Natural spawners above and below Foster Dam are monitored as distinct units and appear to be demographically independent. Currently, the combined escapement to the South Santiam is a few thousand fish, 2296 (2000-2004), but during the mid-1990s the average near 1,000 (Goodson 2005). Score = 3.

Hatchery Impacts

Hatchery Domestication – Native late-run winter steelhead and introduced Skamania Hatchery summer-run are both present in the south Santiam River. Hatchery releases of winter steelhead have not occurred in this basin since 1989, and the proportion of hatchery-reared fish that currently spawn naturally in the South Santiam River is believed to be less than 5% (Chilcote 1997), although prior to 1989 it was over 40% (Goodson 2005). Hatchery operations began in 1926, and in 1940 a record 3,335,000 eggs were taken from 800 females (Wallis 1961). The run size at this time was probably much larger because it was not possible to install the weir in the river until much of the run had already moved far upstream (Wallis 1961).

Genetic analyses indicate close genetic affinity between winter steelhead populations in the Santiam, Molalla (North Fork), and Calapooia Rivers. Steelhead that are the progeny of summer-run and early winter-run steelhead are genetically distinct from presumptive native steelhead. Differences in spawn timing among these run-times may limit (but not eliminate) the potential for interbreeding. PNI ≤ 0.84 (40 years hatchery production, PNI=0.80, 10 years no production). Fitness = 0.85. Score = 3.

Hatchery Introgression – Large numbers of summer-run steelhead (Skamania Hatchery stock, out-of-ESU) are released into the South Santiam River. In 2003, 11,493 summer steelhead returned to the South Santiam Hatchery. Although differences in spawn timing may limit the potential for genetic introgression, it is unclear how competition between summer and winter steelhead juveniles or adults may influence the expression of life history traits. Score = NA.

Synthetic Approach – Hatchery releases of locally-derived late-winter steelhead into the South Santiam River were discontinued in 1989. Currently, over 100,000 summer run fish (Skamania Hatchery-origin) are released from the South Santiam. Winter steelhead that arrive at Foster Dam are transported above the dam, although summer steelhead are not. This effectively creates two zones in the South Santiam River, below Foster Dam where summer and winter steelhead com-mingle and above Foster Dam where only naturally-produced (unmarked) fish are allowed. Currently, $Ph < 0.05$ above Foster Dam, but likely $0.10 < Ph < 0.30$. Diversity persistence score = 3.0.

Anthropogenic Mortality **–** Historically, harvest rates for South Santiam River steelhead has been near 20% (Chilcote 2001). With the recent introduction of selective fisheries this rate has fallen below 5%. Habitat changes in the Santiam River (especially thermal and flow conditions below Detroit Dam), Lower Willamette River, and mainstem Columbia River, may have influenced the expression of life history traits, especially juvenile traits. Score = 3.0**.**

Habitat Diversity **–** Habitat diversity loss is most moderate for this DIP due to the loss of higher elevation spawning areas. Stream order was not determined. Score (*Order/Elevation*) = ND/3.

Overall Score = 2.0. The legacy of hatchery operations in combination with the continued release of summer-run steelhead presented notable risks. Additional concerns included the loss of habitat diversity.

Previously: 2004 TRT 1.59; 2004 ODFW Fail, 5 criteria met.

DV – Calapooia River Winter-Run Steelhead

Life History Traits **–** No information available. Score = NA.

Effective Population Size **–** Willis et al. (1960) reported that both live and dead steelhead were observed in the Calapooia River on 12 May 1958, in addition to 427 redds.

In 1993, spawner density estimates for the Calapooia River were at a record low, 1.8 spawners per mile (Chilcote 1997). The average escapement of late-run winter steelhead to the Calapooia River reached critically low levels during the mid-1990s (1993-1997) with returns of 61 fish (ODFW 1998). In the last four years escapement has reached several hundred fish (427) (Goodson 2005). Score = 1-2.

Hatchery Impacts

> *Hatchery Domestication* – There is no hatchery program on the Calapooia River. Chilcote (1997) estimates that hatchery fish (predominately strays from other Upper Willamette River DIPs) constitute less than 5% of escapement.
>
> Genetic analysis indicated a close affinity between winter-run steelhead in the Calapooia River and native late-run winter steelhead in the Santiam and Molalla basins. Score = NA
>
> *Hatchery Introgression* – The incidence of stray hatchery fish, summer-run steelhead, or winter-run steelhead from other basins in the Upper Willamette River is thought to be low, although given the low escapement even a few fish could have a significant influence on the population. Score = 3-4.
>
> *Synthetic Approach* – There are currently no hatchery releases of steelhead into the Calapooia River. The proportion of hatchery fish on the natural spawning grounds is thought to be low ($Ph<0.05$) although the genetic similarity would be very low. Diversity persistence score = 3.0 - 4.0.

Anthropogenic Mortality **–** Historically, harvest rates for Calapooia River steelhead have been low, near 10% (Chilcote 2001). With the recent introduction of selective fisheries this rate has fallen below 5%. Habitat changes in the Calapooia, Lower Willamette River, and mainstem Columbia River may have influenced the expression of life history traits, especially juvenile traits. Score = 3-4**.**

Habitat Diversity **–** Habitat diversity loss is most moderate for this DIP due to the loss of higher elevation spawning areas. Stream order was not determined. Score*(Order/Elevation)* = 3-4.

Overall Score = 1.5. Small population size appears to be the greatest threat to diversity. Abundance is low enough that genetic drift, introgression with non-local fish, and selection could dramatically influence genetic variation in this population.

Previously: 2004 TRT 1.78; 2004 ODFW Pass.

DV – Criterion Summary

With respect to the diversity criterion evaluation, populations in this ESU were all classified into the 'moderate' risk category (Figure 38); although, in the case of the Calapooia population, a classification of 'high' risk may be an equally appropriate determination. The loss of genetic resources because of small population sizes and loss of historically accessible habitat are the primary factors that resulted in the DV criterion population ratings.

The uncertainty associated with these population scores for the DV criterion was relatively small. Given this result and the individual populations scores themselves, we conclude that the most probable risk classification for these populations (and ESU) with respect to the SS criterion is 'moderate'.

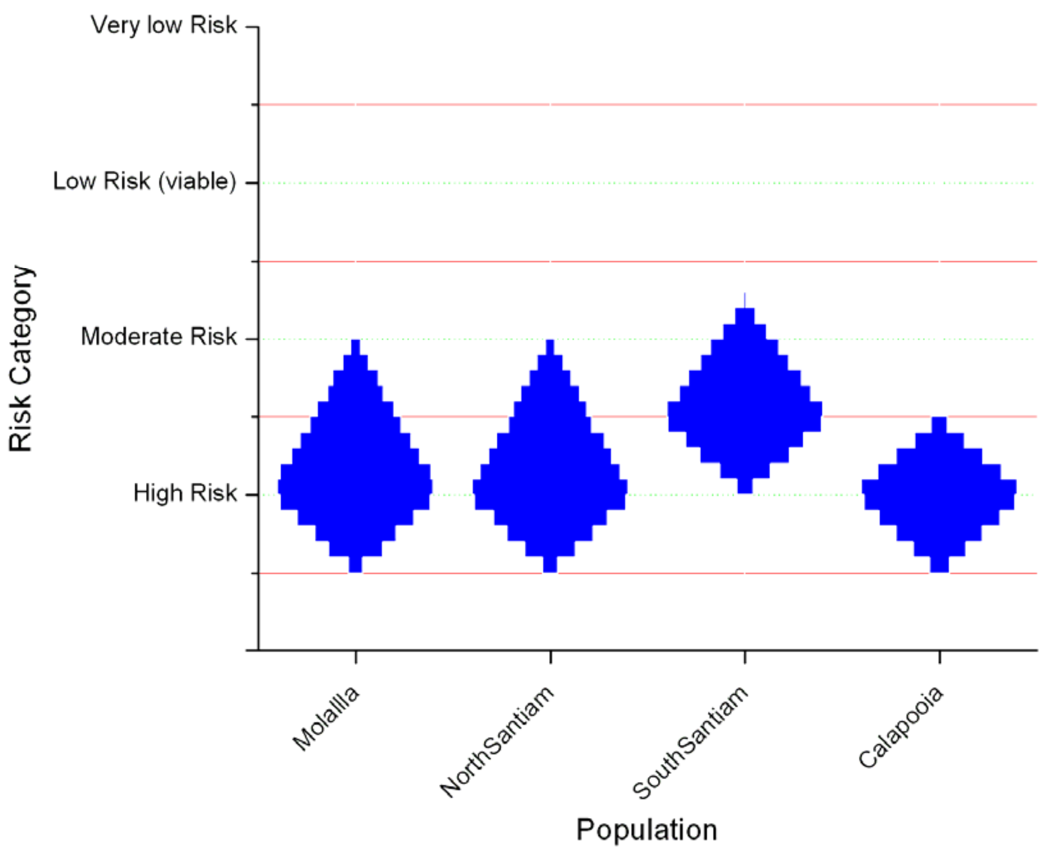

Figure 38: Summary of diversity evaluation for Upper Willamette steelhead populations.

V. Summary of Population Results

When the scores for all three population criteria were combined, we concluded that the most likely risk of extinction for all UW steelhead populations is moderate (Figures 39 and 40). However, there is considerable uncertainty in these population risk estimates. Based on this analysis, we conclude that the overall extinction risk for the UW steelhead ESU is moderate.

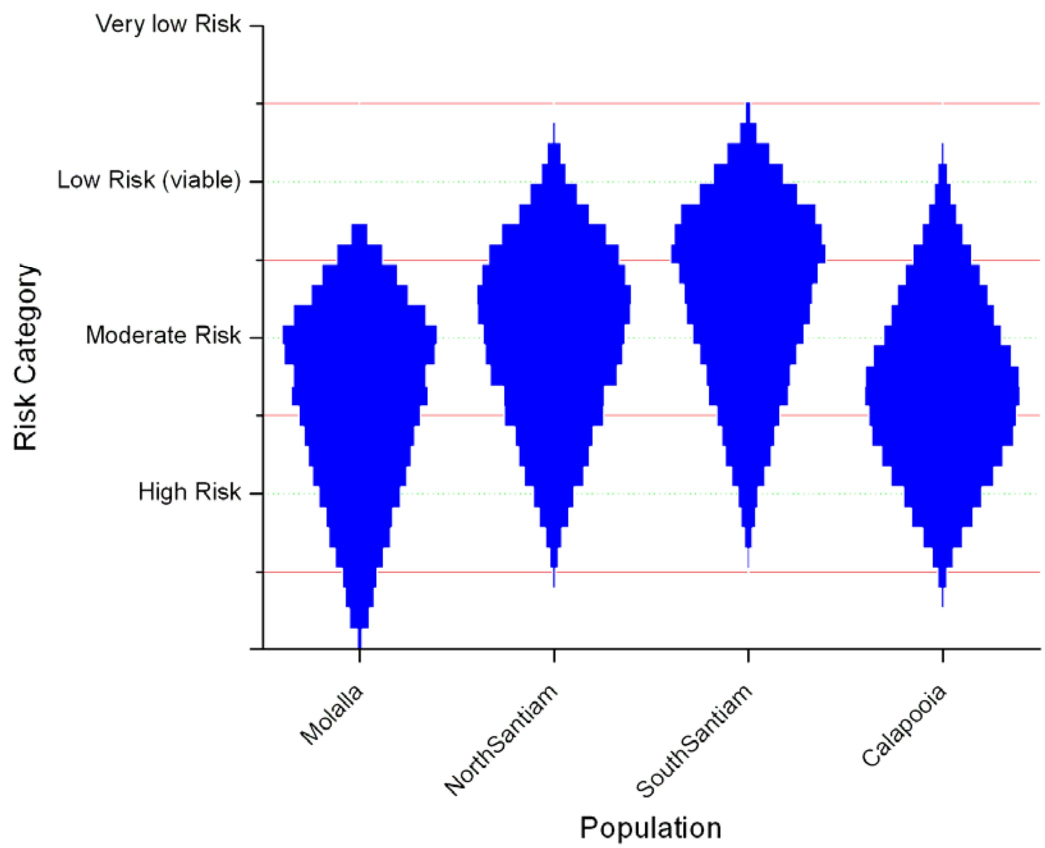

Figure 39: Overall population status assessment for Upper Willamette steelhead using the minimum distribution approach.

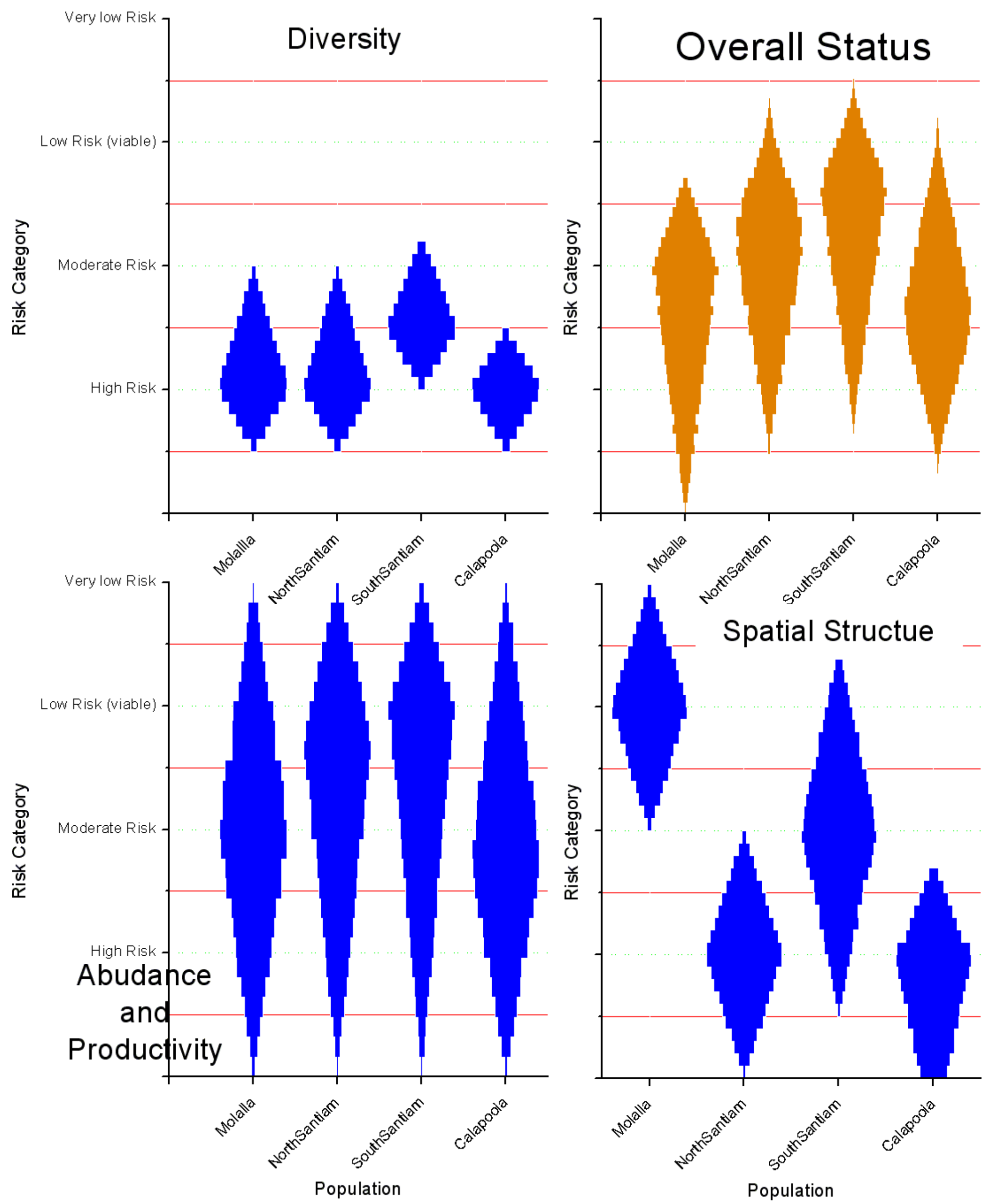

Figure 40: Upper Willamette steelhead status graphs of each attribute and the overall summary.

Literature Cited

Myers, J. M., C. Busack, D. Rawding, A. R. Marshall, D. J. Teel, D. M. Van Doornik, and M. T. Maher. 2006. Historical population structure of Pacific salmonids in the Willamette River and lower Columbia River basins. NOAA, Seattle.

ODFW. 2005. 2005 Oregon native fish status report. ODFW, Salem, OR.

Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Appendix A: Random Multinomial Finite Sampling Method

September 2007

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

Background

Note: This method is used to include uncertainty in our estimate of the population attribute weights for calculating overall population risk status and for adding error to age structure parameters in the viability curves.

In simulation modeling we often encounter parameters that are a vector of fractions partitioning some whole. For example, we may have a parameter that describes the fraction of fish that spawn at a given age, with 20% at age 3, 70% at age 4, and 10% at age 5. The fractions need to sum to 100% and are therefore clearly not independent. Other examples are weights on some linear function where the weight totals must sum to one or the fraction of the habitat that is in specific categories. These fraction vectors are often estimated with uncertainty and for Monte Carlo simulations we need to randomly generate new vectors that sum to one and have a controlled distribution around the point estimates.

If we treat the fractions as probabilities, the vectors describe a multi-nominal distribution (e.g., the probability of age 3 is 0.2, the probability of age 4 is 0.7, and the probability of age 5 is 0.1). To obtain a random vector with the appropriate properties we apply a finite sampling approach, which can be described with a dart board analogy.

Assume that the point estimate vector is as described in the pie chart of Figure 1A. Assume this pie chart is a dart board. We can throw a finite number of darts at the board (say, 20), which will give us a situation like Figure 1B. The darts are thrown randomly at the board and must all land on the board. We can then calculate the fraction of the 20 darts that land in each wedge of the pie, which gives us a new random vector (shown as new pie chart in Figure 1C). If we repeat this process many times, on average the fraction of darts in each wedge will equal the original point estimate vector. However, any particular throw of 20 darts will likely vary from the original, giving us the random noise that we need.

We can control the amount of variation in the distribution by changing the number of darts that we throw each time. If we throw only 20 darts there is likely to be a fair bit of variation between the point estimate vector and any particular random vector. However, if we throw 200 darts each time (Figure 1D), each random vector will be relatively close to the original point estimate (Figure 1E). Thus, we can control the amount of variation in our random draws by adjusting what we call the “shape parameter” because it affects the shape of the generator output distribution. If we throw an infinite number of darts, we always get an original point estimate vector. We can see the effect of changing this shape parameter by looking at a cumulative frequency plot (Figure 2). As the shape parameter decreases, the range of the random generator distribution increases. This relationship is also illustrated in Figure 3.

This method has the advantage of simultaneously changing all the parameters of the vector, retaining the constraint that they sum to one. A feature of the approach is that the distribution of the random generator for any particular fraction is a function of the value of that fraction. For example, the range of the distribution if the point estimate is 10% for a particular category will be different than the range of a category with a point estimate of

50% (Figure 4). This makes sense if we consider that the distributions are constrained – values cannot be less than zero or greater than one, so point estimates that are near these boundaries will have different distributions from those of point estimates that are not near the boundaries. Another feature of the method is that fractions that have a point estimate of zero will have a value of zero for all random vectors (the width of the pie wedge is infinitely small and no darts can land there). This will not be a problem for most applications.

One limitation of the approach is that the output distribution from the random generator is discrete, rather than continuous. Because we are throwing a finite number of darts (say N), the values in the output random vectors will all be fractions of N (i.e., x/N, where is x an integer and $0 \leq x \leq N$). If, for example, the shape parameter is 20, then there are only 21 possible values in the output vectors, which are 0.00, 0.05, 0.10, 0.15, 0.20, …0.90, 0.95, 1.00 (i.e., values in 5% increments). If the shape parameter was 200, there are 201 possible output values and the output becomes more continuous. The possible values are 0.000, 0.005. 0.010, 0.015, … 0.995, 1.000 (i.e., values in 0.5% increments). This discrete output feature is responsible for the "stair step" appearance of the cumulative frequency graphs (Figures 2 and 3). For many applications, the fact that the generator produces only discrete values will not be a problem, but it is useful to be aware of this effect, especially when using shape parameter values less than 20.

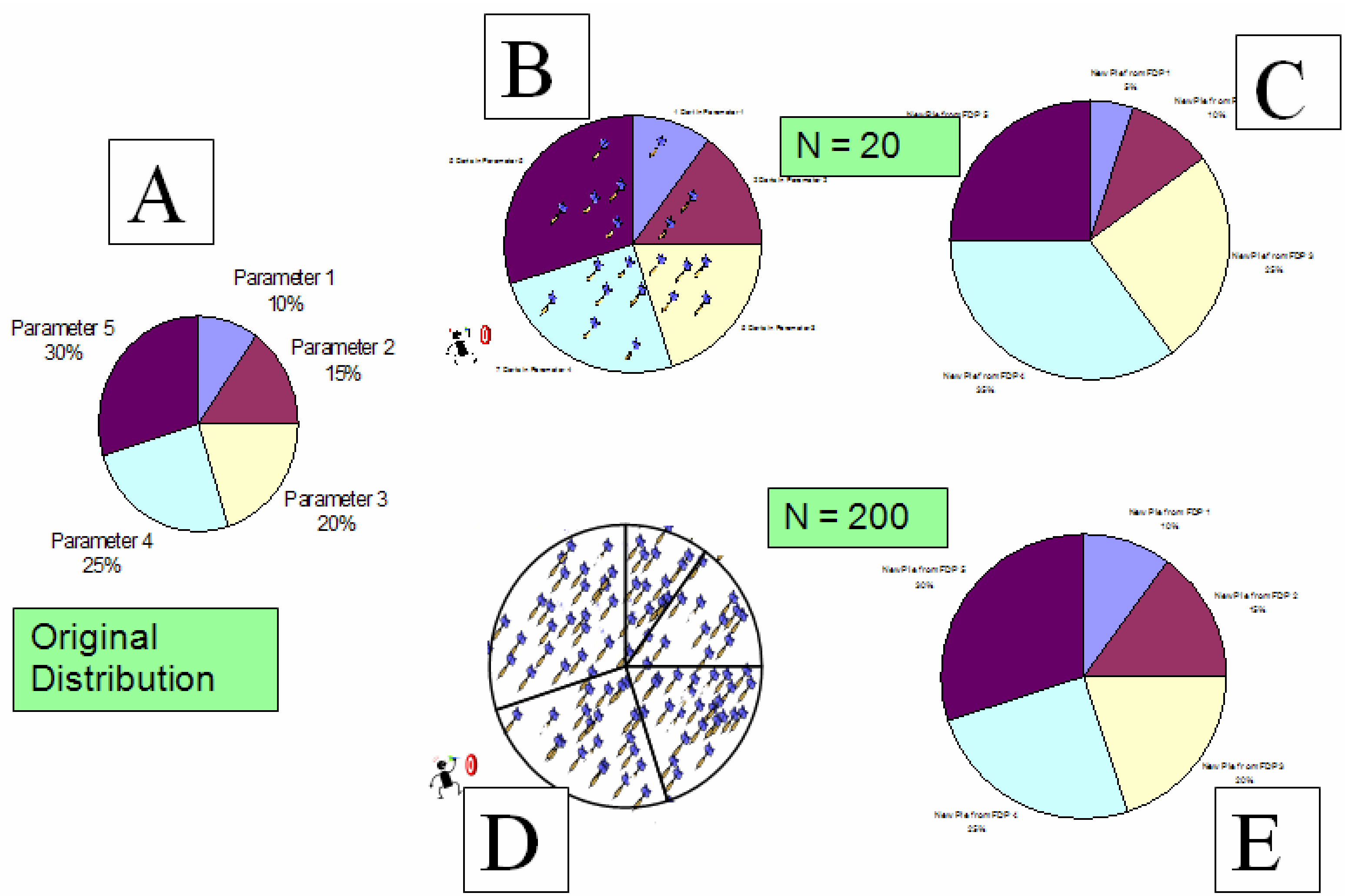

Figure 1: Illustration of the multi-nominal finite random sampling method using a dart board analogy. Pie chart A represents the point estimate fraction vector (a.k.a. dart board). Chart B shows a random sample of 20 darts, with chart C showing the resulting fraction of darts in each wedge, representing the new random fraction vector. Chart D shows a random sample of 200 darts, with chart E showing the resulting fraction of darts in each wedge, representing a new random fraction vector.

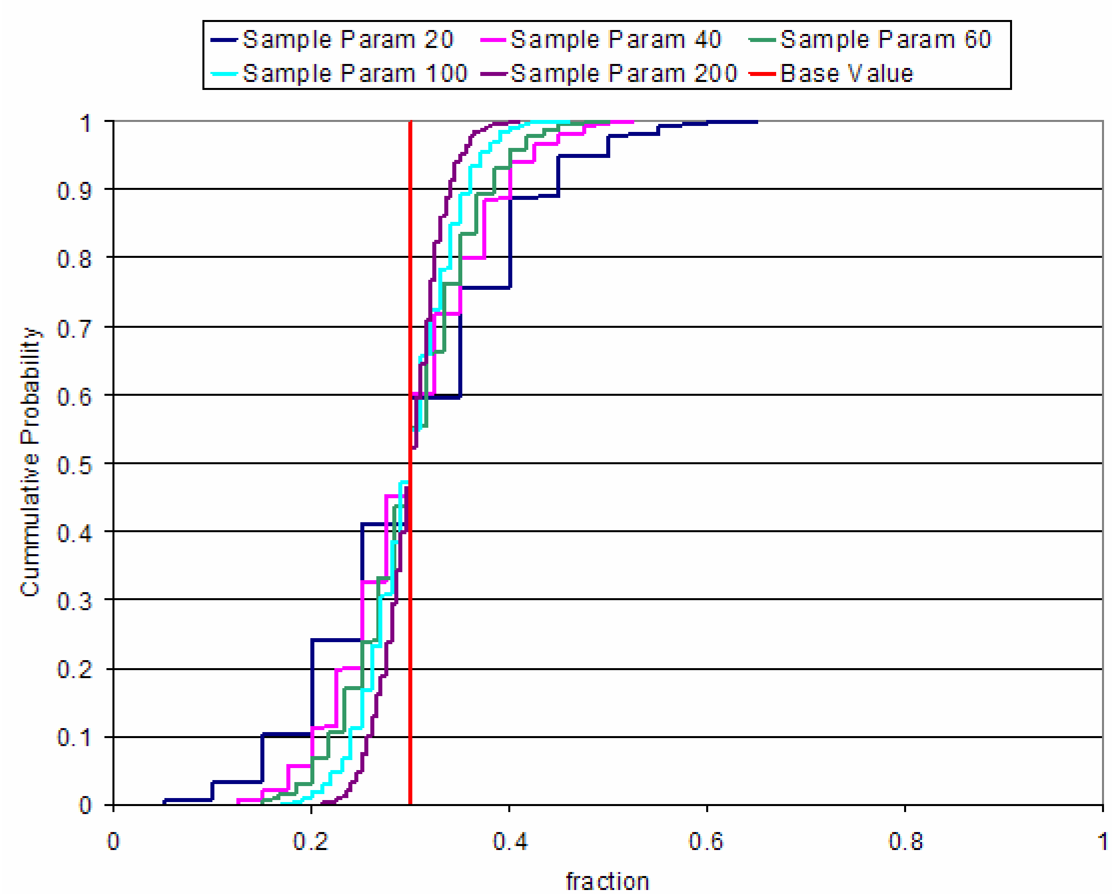

Figure 2: Cumulative frequency distribution of random samples is the point estimate ("Base Value") of 0.3. The different lines indicate distributions with different shape parameters ("Sample Param").

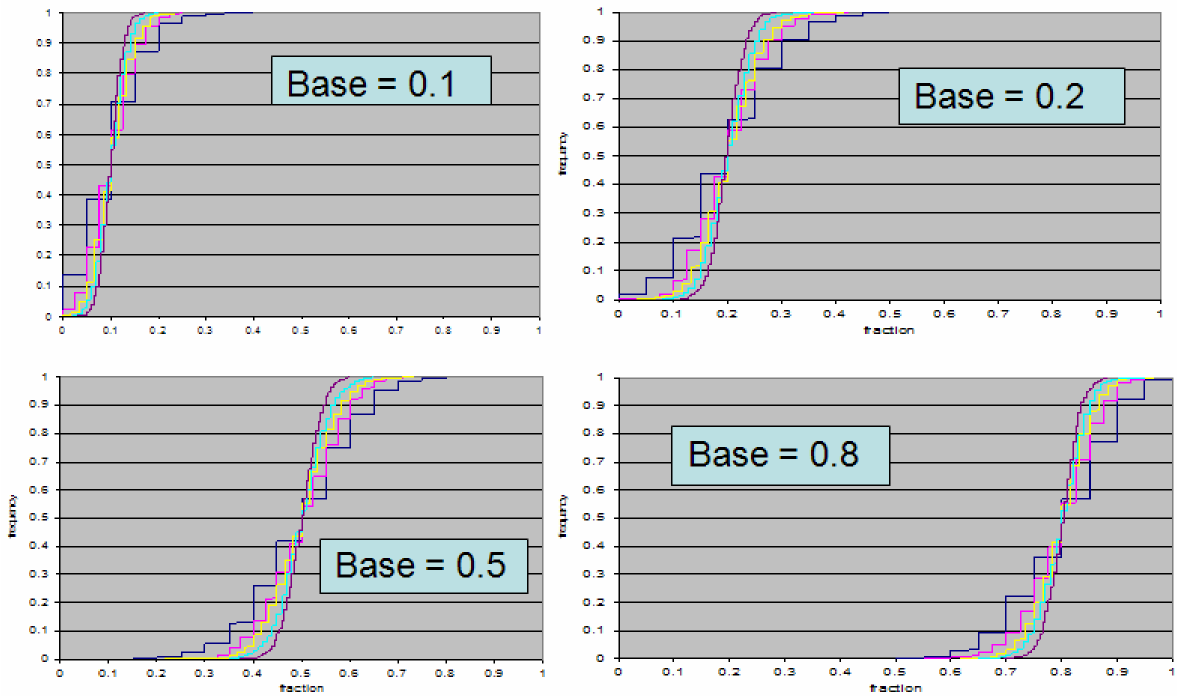

Figure 3: Cumulative frequency distribution of multi-nominal finite random sampling output for different point estimate fractions ("Base"). The different curves are for different shape parameters (values of 20, 40, 60, 100, and 200).

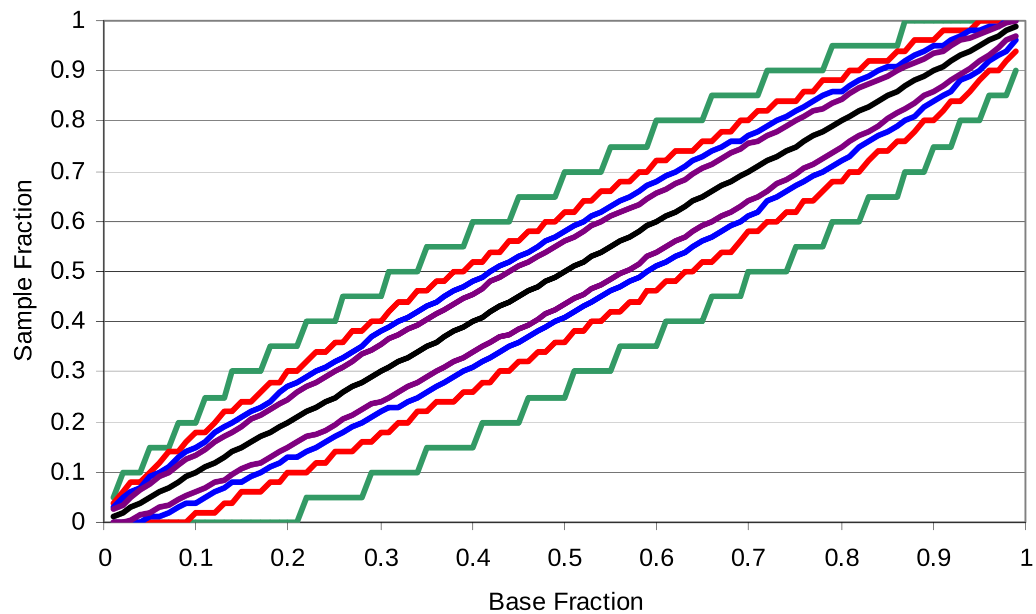

Figure 4: 90% probability bands for multi-nominal finite random sampling output. The black line indicates an infinite shape parameter (the output always equals the point estimate). The purple, blue, red and green lines represent shape parameters of 200, 100, 50, and 20, respectively. To interpret the figure, pick a base fraction (point estimate) on the x-axis then look at the range between the curves on the y-axis at that point. 90% of the output values from the random generator would be within this range.

Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Appendix B:
Oregon Abundance Time Series

September 2007

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

Background

The collection of data from populations of salmon and steelhead in these ESUs has been neither comprehensive nor consistent. Data is entirely missing for a significant number of populations. For those populations where data is available, the nature of this data and the methods to collect it were often dissimilar. For example, steelhead abundance for Sandy River populations was based on counts of fish passing Marmot Dam. In contrast, steelhead spawner abundance estimates for the Calapooia River were based on redd density estimates in short stream survey sections extrapolated for all of the estimated stream kilometers of spawning habitat in the basin.

The purpose of Appendix B is to describe these different methodologies and document the resulting estimates of spawners and pre-harvest abundance of wild fish for each population. Additional information on age composition, fishery catch rates, and the proportion of the spawning population that were hatchery fish are also presented. These data represent the basic information from which the viability metrics for abundance, productivity, and (to a lesser extent) diversity were generated.

The raw data sets for these populations were frequently incomplete. Sometimes survey data covering a year or more was missing. In other instances, harvest rate estimates for a particular fishery were unavailable. A variety of methods were developed to fill in these data gaps in order to assemble a full data set for use in analyses as reported here. There is no recognized “correct” method for accomplishing this; a range of alternative methods could be used to generate the numbers needed to approximate the missing data.

The methods presented here to develop a full data set for each population represent only one of the available alternatives. The goal was to achieve a reasonable balance between extracting too much out of less than ideal data sets versus discarding usable information because it didn’t conform to rigid data protocols. In the case of chum salmon, however, there were no observations in any Oregon population from which to develop data sets. We were therefore unable to perform a quantitative evaluation of this species, other than by making inferences using data sets from the Washington side of the Columbia River where two populations are still known to exist.

Population Data Descriptions

1. Fall Chinook (Late) – Sandy

The abundance data for this population is based upon spawning survey observations conducted from 1984 to present. Both peak redd and fish counts are obtained in these surveys, but in the opinion of ODFW biologists the redd count data were more reliable. Following methodology developed by ODFW, the peak redd count was multiplied by an expansion factor of 2.5 to estimate total season spawners for the survey section. A fish per km density estimate was then determined by dividing the number of spawners by the length of the survey section, which was approximately 16 km. This spawner density was then expanded for the total 67 linear kilometers of spawning habitat for fall chinook in the Sandy basin to yield annual estimates of total spawner abundance for the population (Table 1). The number of stream kilometers utilized by fall chinook within this basin was based on information provided by Maher et al. (2005).

Spawner survey data were missing from 1981 to 1984 and from 1990. To fill in the data for these missing years, we used the observed relationship between sport fishery catch estimates and spawner abundance estimates in years when data were available for both. It was found that for the 15-year period after 1984, 75% of the variation in spawner abundance estimates could be associated with variations in the sport fishery catch estimates. This relationship was then used, along with catch estimates for years without spawner data, to estimate what the spawner abundance might have been in 1981 to 1984 and 1990.

Although hatchery fall chinook are found in this basin, they belong to the Tule type of fall chinook that spawn earlier than the late Sandy fall chinook population. Occurrence of hatchery strays during the time when the wild population spawns has been rare. However, occasionally one of the carcass samples taken during the spawning surveys is found to contain a CWT indicating it was of hatchery origin. Therefore, a low stray of 5% (or 95% wild fraction) was assumed for the population.

Sandy River late fall chinook are caught in ocean fisheries, Columbia River mainstem fisheries, and tributary sport fisheries. The impact of ocean fisheries varies depending on how many years a chinook stays at sea before returning. For example, 3-year-olds get exposed to one season of fishing, 6-year-olds to three seasons of fishing. We used the estimated impact rate on 4-year-old adults (the predominant age category) as an average to represent ocean fishery impacts. Most of these impact estimates came from a report that included data for wild North Lewis River fall chinook in Washington (Daignerault et al. 2003). Sandy late fall chinook have similar timing and age composition as wild North Fork Lewis fall chinook. It was therefore assumed that the ocean distribution and fishery impacts on these two populations would be similar.

Columbia River fishery impact estimates provided by Daignerault et al. (2003) were also used in this data summary, except for the years after 1993 when impact rates specific to the Sandy population, as presented in the FMEP prepared by ODFW, were used. Tributary fishery impact rates were estimated from annual sport catch estimates provided by ODFW. From 2002 to the present, regulations that require the release of all unmarked chinook have been in effect for the Sandy basin. This change effectively lowered the impact of sport fisheries as only the mortality associated with post-release mortality of fish that were caught was now a factor. We assumed this regulation change effectively reduced sport fishery impacts to 10% of their former level. The overall impact of the three fisheries was estimated as:

1 – [(1-OceanHR)*(1-ColmHR)*(1-TribHR)]

Age composition of spawning adults was based on scales collected and read from Sandy River fall chinook by ODFW from 1998 to 2004. For the purposes of these analyses, fish observed as Age 2 were not included in the summary and the proportions for the remaining ages were adjusted so they would equal 1.00. Age 2 fish are largely jacks and comprise a small portion of the return. Inclusion of jacks is in the total return estimate and can cause analytical problems because they are less susceptible to fisheries, particularly the Columbia River gillnet fishery.

Table 1: Basic data set developed for Sandy River fall chinook.

Spawn Year	Total Spawners	Wild Fraction	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning			
					Age3	Age4	Age5	Age6
1981	2998	0.95	0.492	2904	0.143	0.694	0.157	0.006
1982	3472	0.95	0.498	3442	0.143	0.694	0.157	0.006
1983	2447	0.95	0.482	2278	0.143	0.694	0.157	0.006
1984	3157	0.95	0.491	3049	0.143	0.694	0.157	0.006
1985	1983	0.95	0.446	1594	0.143	0.694	0.157	0.006
1986	2703	0.95	0.630	4596	0.143	0.694	0.157	0.006
1987	8702	0.95	0.352	4735	0.143	0.694	0.157	0.006
1988	6610	0.95	0.640	11743	0.143	0.694	0.157	0.006
1989	8129	0.95	0.443	6476	0.143	0.694	0.157	0.006
1990	3340	0.95	0.364	1908	0.143	0.694	0.157	0.006
1991	2792	0.95	0.511	2915	0.143	0.694	0.157	0.006
1992	3976	0.95	0.442	3145	0.143	0.694	0.157	0.006
1993	5446	0.95	0.399	3612	0.143	0.694	0.157	0.006
1994	2299	0.95	0.397	1516	0.143	0.694	0.157	0.006
1995	4163	0.95	0.397	2745	0.143	0.694	0.157	0.006
1996	2013	0.95	0.397	1327	0.143	0.694	0.157	0.006
1997	8021	0.95	0.397	5289	0.143	0.694	0.157	0.006
1998	3088	0.95	0.397	2036	0.143	0.694	0.157	0.006
1999	1796	0.95	0.397	1184	0.143	0.694	0.157	0.006
2000	345	0.95	0.397	228	0.143	0.694	0.157	0.006
2001	3335	0.95	0.397	2199	0.143	0.694	0.157	0.006
2002	5189	0.95	0.196	1268	0.143	0.694	0.157	0.006
2003	3793	0.95	0.196	927	0.143	0.694	0.157	0.006
2004	2397	0.95	0.196	586	0.143	0.694	0.157	0.006
2005	5681	0.95	0.196	1319	0.143	0.694	0.157	0.006
2006	9934	0.95	0.196	2306	0.143	0.694	0.157	0.006

2. Fall Chinook (Tule) – Clatskanie

Peak counts of spawning fall chinook in a 3.2 km survey section of the Clatskanie River was the source of raw data for building the data set for this population. Annual peak spawner counts were converted into an estimated season count by multiplying by a correction factor of 1.7. Using these converted numbers, a spawner density (spawners per stream km) was estimated for each year. An estimate for spawner abundance for the entire population was obtained by multiplying these annual spawner densities times the total number kilometers of fall chinook spawning habitat (Table2). We used the Maher et al. (2005) estimate of 16 km spawning habitat for these expansions.

In recent years the proportion of stray hatchery fish into this basin appears low, as evidenced by relatively rare recoveries of CWT hatchery fish during spawning surveys. We assumed 15% of the spawners were hatchery strays from 1970 to present and 0% were hatchery strays prior to 1970, when the likelihood of stray hatchery fish was assumed to be lower because sources of hatchery fish were more distant and less numerous.

The primary fishery impacts on the Clatskanie population have been the ocean fishery and the Columbia River mainstem fishery. Sport catch of fall chinook within the Clatskanie basin is relatively minor and was not included in our calculations. Fishery impact rates from 1986 to present were estimated based on CWT recovery data for Tule fall chinook released from nearby Big Creek Hatchery as provide by Mark Lewis (ODFW). It was assumed that these rates were similar to those experienced by the Clatskanie population. Prior to 1986, Cowlitz Tule fall chinook were used as a proxy to estimate fishery impacts. Measured impact rates for ocean and Columbia fisheries Cowlitz River fall chinook were available for 1980-83 and 1964-68. For years during this period with no data, the ocean impact rates were estimated as either the 1964-68 average or 1980-83 average depending on which dates were chronologically nearest to year for which data was missing. For the Columbia River impacts, a relationship between the number of season fishing days set for the commercial gillnet season between August 20 and September 20 and the subsequent fishery impact rate was relied upon. This relationship, first described by Cramer and Vigg (1999), was able to explain 76% of the variation in Columbia River impact rates on Cowlitz fall chinook on the basis of the number of days the fishing season was open between August 20 and September 20. Cumulative fishery impacts were calculated as:

1- [(1-OceanHR)*(1-ColumbiaHR)].

Age composition of Clatskanie Tule fall chinook was determined from ODFW CWT data from fall chinook returning to nearby Big Creek hatchery. Annual estimates of age composition using these CWT data (excluding Age 2 jacks) was averaged for the time period (1986 to 2002) to yield the average age composition recorded in Table 2.

Finally, a preliminary run reconstruction and calculation of recruits per spawner yielded unrealistically high values for the years 1953,1958-61, 1989, 1992-93, and 2000. This was most likely caused by the observation of only a single spawner, poor survey conditions resulting in an underestimate, or other unknown factors. Therefore, to make these data more compatible with the limits of fall chinook life history and recruitment rates – the peak spawner count for each of these years was increased until the R/S value was less than 50. A value of 50 recruits per spawner was assumed to be the upper limit on the reproductive rate of naturally reproducing fall chinook. In most cases this manipulation required increasing the observed peak count by only 1 to 2 spawners. In addition to this adjustment, there was also a minimum value of 50, placed on the spawner estimate. For example, if the spawner estimate in a particular year was 12, then a value of 50 substituted. The logic behind this change was that values less than the CRT level (which is 50 for this population) would seem unlikely if this population is continuing to persist. We assume repeated spawner levels less than CRT would likely lead to population extinction which has not occurred. We assume then that escapement estimates less than 50 are more likely an outcome of measurement error rather than true spawner abundance.

Table 2: Basic data set developed for Clatskanie Tule fall chinook.

Spawn Year	Total Spawners	Wild Fraction	Overall Fishery Mortality	Total Wild Catch	*Proportion by Age at Spawning*			
					Age3	Age4	Age5	*Age6*
1952	219	1.00	0.924	2673	0.211	0.540	0.250	*0.000*
1953	50	1.00	0.924	610	0.211	0.540	0.250	*0.000*
1954	50	1.00	0.924	610	0.211	0.540	0.250	*0.000*
1955	50	1.00	0.924	610	0.211	0.540	0.250	*0.000*
1956	152	1.00	0.892	1257	0.211	0.540	0.250	*0.000*
1957	50	1.00	0.860	308	0.211	0.540	0.250	*0.000*
1958	50	1.00	0.780	178	0.211	0.540	0.250	*0.000*
1959	50	1.00	0.796	196	0.211	0.540	0.250	*0.000*
1960	50	1.00	0.717	126	0.211	0.540	0.250	*0.000*
1961	50	1.00	0.765	162	0.211	0.540	0.250	*0.000*
1962	152	1.00	0.828	733	0.211	0.540	0.250	*0.000*
1963	379	1.00	0.828	1831	0.211	0.540	0.250	*0.000*
1964	260	1.00	0.804	1065	0.211	0.540	0.250	*0.000*
1965	50	1.00	0.850	284	0.211	0.540	0.250	*0.000*
1966	523	1.00	0.823	2425	0.211	0.540	0.250	*0.000*
1967	76	1.00	0.900	684	0.211	0.540	0.250	*0.000*
1968	50	1.00	0.850	283	0.211	0.540	0.250	*0.000*
1969	124	1.00	0.835	626	0.211	0.540	0.250	*0.000*
1970	67	0.85	0.881	423	0.211	0.540	0.250	*0.000*
1971	50	0.85	0.804	174	0.211	0.540	0.250	*0.000*
1972	62	0.85	0.696	120	0.211	0.540	0.250	*0.000*
1973	161	0.85	0.865	881	0.211	0.540	0.250	*0.000*
1974	87	0.85	0.711	182	0.211	0.540	0.250	*0.000*
1975	186	0.85	0.835	798	0.211	0.540	0.250	*0.000*
1976	186	0.85	0.773	538	0.211	0.540	0.250	*0.000*
1977	87	0.85	0.711	182	0.211	0.540	0.250	*0.000*
1978	50	0.85	0.727	113	0.211	0.540	0.250	*0.000*
1979	198	0.85	0.727	449	0.211	0.540	0.250	*0.000*
1980	322	0.85	0.588	392	0.211	0.540	0.250	*0.000*
1981	248	0.85	0.599	315	0.211	0.540	0.250	*0.000*
1982	459	0.85	0.638	687	0.211	0.540	0.250	*0.000*

1983	161	0.85	0.549	167	0.211	0.540	0.250	*0.000*
1984	50	0.85	0.560	54	0.211	0.540	0.250	*0.000*
1985	161	0.85	0.514	145	0.211	0.540	0.250	*0.000*
1986	161	0.85	0.683	295	0.211	0.540	0.250	*0.000*
1987	337	0.85	0.676	598	0.211	0.540	0.250	*0.000*
1988	707	0.85	0.678	1266	0.211	0.540	0.250	*0.000*
1989	397	0.85	0.594	494	0.211	0.540	0.250	*0.000*
1990	174	0.85	0.388	94	0.211	0.540	0.250	*0.000*
1991	50	0.85	0.601	64	0.211	0.540	0.250	*0.000*
1992	50	0.85	0.616	68	0.211	0.540	0.250	*0.000*
1993	50	0.85	0.585	60	0.211	0.540	0.250	*0.000*
1994	59	0.85	0.442	40	0.211	0.540	0.250	*0.000*
1995	84	0.85	0.327	35	0.211	0.540	0.250	*0.000*
1996	464	0.85	0.381	243	0.211	0.540	0.250	*0.000*
1997	67	0.85	0.337	29	0.211	0.540	0.250	*0.000*
1998	149	0.85	0.143	21	0.211	0.540	0.250	*0.000*
1999	124	0.85	0.241	33	0.211	0.540	0.250	*0.000*
2000	50	0.85	0.345	22	0.211	0.540	0.250	*0.000*
2001	50	0.85	0.382	26	0.211	0.540	0.250	*0.000*
2002	388	0.85	0.470	293	0.211	0.540	0.250	*0.000*
2003	472	0.85	0.457	337	0.211	0.540	0.250	*0.000*
2004	74	0.85	0.423	46	0.211	0.540	0.250	*0.000*
2005	211	0.85	0.423	131	0.211	0.540	0.250	*0.000*
2006	*126*	*0.85*	*0.423*	*78*	*0.211*	*0.540*	*0.250*	*0.000*

3. Spring Chinook – Sandy

The basic information used to estimate the abundance of spring chinook in the Sandy basin were the counts of upstream migrating adults as they passed Marmot Dam on the Sandy River. These counts represented at least 90% of the entire run, as very little of spawning and rearing habitat for spring chinook occurs downstream of Marmot Dam. Although spring chinook have been counted at Marmot Dam since 1951, the data collected through 1960 is thought to be unreliable for a variety of reasons. Primarily the issue is that the number of fish counted is much lower than the number caught within the basin for these early years. In some cases, the unadjusted data suggest an 80% tributary fishery impact rate. It is highly unlikely a fishery could generate these levels of impact. However, this may also be an artifact of extremely high in-river mortalities associated with unfavorable water conditions for summer holding prior to migration past Marmot Dam. To avoid these complications and reduce uncertainty we choose to only use data collected from 1961 to present (Table 3).

Spring chinook were not counted at Marmot Dam from 1971 to 1976 and only a partial count was made in 1983. In addition, the recorded count for 1964 of 660 fish was thought to be an erroneous overestimate of the return. Based on a regression between sport catch and dam counts, annual estimates of sport catch within the Sandy basin for 1964, 1971-76, and 1983 were used to estimate dam counts for these years. This regression was developed from those years with both dam count and catch data during the period 1961 to 2001. From this regression it appeared that 82% of the variation in Marmot Dam counts could be explained by the observed variations in annual sport catch estimates.

A substantial number of hatchery fish are known to return to the Sandy basin. The first hatchery spring chinook returned in 1970. The program size was gradually increased from 50,000 fish in the mid-1970, to nearly 500,000 fish by the end of the 1990s.

However, only in recent years were direct measurements of the hatchery fraction possible via inspection of returning adults for fin clips. Prior to 2001 only a small portion or none of the hatchery release was fin clipped before they were released as smolts. Therefore, from 1961 to 2001 hatchery fish could not be visually counted separately from wild fish.

To estimate the proportion of hatchery fish for this earlier period a simple relationship was developed between the number of hatchery smolts released during the recent years when all fish were fin clipped (and could be identified as hatchery fish when they returned) and the proportion of hatchery fish observed at Marmot Dam. Based on this relationship, the average number of wild smolts produced in those years was estimated. Using this average number of wild smolts, and assuming that this was a rough estimate of wild smolt production in previous years, the ratio between wild smolts and number of hatchery smolts released for each year prior to 2002 was determined. A record of the number of hatchery smolts released is available for all years. The estimated annual ratios hatchery to wild smolts were assumed to represent the ratio of hatchery and wild adults in the corresponding return years.

It is also notable that beginning in 2002, all hatchery fish arriving at the Marmot Dam counting facility were removed from the trap and not passed upstream. Therefore, although at least 50% of the fish trapped at Marmot Dam were hatchery fish, wild fish comprised essentially 100% of the natural spawning population upstream of Marmot Dam (Table 3).

Sandy spring chinook salmon are caught in ocean fisheries, Columbia River mainstem fisheries, and in-river sport fisheries. The estimated ocean impact rates were assumed to be the same as those reported by Beamesderfer (1999) for Willamette River spring chinook. The mainstem Columbia fishery impacts reported by ODFW in their FMEP for spring chinook were used to represent the mortality caused by this fishery. Finally, annual sport catch estimates (from catch cards) for the Sandy were used to estimate impacts of the tributary fishery. However, the ODFW reported sport catch estimates were adjusted downward 32% to ensure they were not overestimates of the impact. From various locations in the Willamette basin both statistical creel programs and catch card estimates of sport catch have been made in at least four different years (ODFW, unpublished data). It is assumed that the creel estimates of catch are more accurate than the catch card estimates. Across all of the locations and years compared, the creel estimate of catch averaged 68% of the catch card estimate. This result was the basis of adjustments made to the catch card data estimates for the Sandy spring chinook fishery.

From 2002 to present only fin clipped chinook could be kept by sport anglers within the Sandy basin. Therefore, the only impact of the sport fishery on wild spring chinook was catch and release mortality. It was assumed that 15% of all sport caught and released wild spring chinook died later from stress. This rate was applied to the average sport catch impact rate in years *before* the catch and release regulations went into effect to estimate an average mortality impact of the sport fishery during the recent years.

The overall impact of the ocean, Columbia, and tributary fishery impacts fisheries was estimated as:

1 – [(1-OceanHR)*(1-ColmHR)*(1-TribHR)].

Age composition of Sandy spring chinook was determined from scale samples obtained from fishery and carcass recovery sampling. Age 2 fish were excluded from data sets.

Table 3: Basic data set developed for Sandy spring chinook.

Spawn Year	Hatch Fish at Dam	Hatch Fish Passed	Total Spawners	Wild Fraction	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning			
							Age3	Age4	Age5	Age6
1961	0	0	37	1.000	0.539	43	0.00	0.60	0.39	0.01
1962	0	0	65	1.000	0.450	53	0.00	0.60	0.39	0.01
1963	0	0	124	1.000	0.462	107	0.00	0.60	0.39	0.01
1964	0	0	41	1.000	0.502	41	0.00	0.60	0.39	0.01
1965	0	0	13	1.000	0.747	38	0.00	0.60	0.39	0.01
1966	0	0	63	1.000	0.441	50	0.00	0.60	0.39	0.01
1967	0	0	51	1.000	0.497	50	0.00	0.60	0.39	0.01
1968	0	0	61	1.000	0.441	48	0.00	0.60	0.39	0.01
1969	0	0	81	1.000	0.562	104	0.00	0.60	0.39	0.01
1970	26	26	137	0.808	0.525	122	0.00	0.60	0.39	0.01
1971	13	13	85	0.850	0.502	72	0.00	0.60	0.39	0.01
1972	14	14	94	0.850	0.502	81	0.00	0.60	0.39	0.01
1973	19	19	125	0.850	0.502	108	0.00	0.60	0.39	0.01
1974	8	8	51	0.850	0.502	43	0.00	0.60	0.39	0.01

1975	58	58	386	0.850	0.502	331	0.00	0.60	0.39	0.01
1976	24	24	224	0.891	0.502	201	0.00	0.60	0.39	0.01
1977	62	62	346	0.821	0.520	308	0.00	0.60	0.39	0.01
1978	123	123	535	0.770	0.373	245	0.00	0.60	0.39	0.01
1979	102	102	233	0.561	0.729	352	0.00	0.60	0.39	0.01
1980	108	108	548	0.803	0.708	1064	0.00	0.60	0.39	0.01
1981	649	649	1089	0.404	0.643	792	0.00	0.60	0.39	0.01
1982	155	155	522	0.703	0.646	670	0.00	0.60	0.39	0.01
1983	845	845	1837	0.540	0.502	1000	0.00	0.60	0.39	0.01
1984	557	557	1211	0.540	0.551	803	0.00	0.60	0.39	0.01
1985	258	258	561	0.541	0.639	536	0.00	0.60	0.39	0.01
1986	403	403	702	0.426	0.524	329	0.00	0.60	0.39	0.01
1987	643	643	1401	0.541	0.492	734	0.00	0.60	0.39	0.01
1988	892	892	1940	0.540	0.421	762	0.00	0.60	0.39	0.01
1989	881	881	1376	0.360	0.405	336	0.00	0.60	0.39	0.01
1990	877	877	1557	0.437	0.579	934	0.00	0.60	0.39	0.01
1991	1249	1249	1888	0.339	0.532	726	0.00	0.60	0.39	0.01
1992	2947	2947	4451	0.338	0.412	1052	0.00	0.60	0.39	0.01
1993	2268	2268	3429	0.338	0.464	1007	0.00	0.60	0.39	0.01
1994	1526	1526	2309	0.339	0.362	445	0.00	0.60	0.39	0.01
1995	1002	1002	1503	0.333	0.440	393	0.00	0.60	0.39	0.01
1996	1723	1723	2561	0.327	0.386	526	0.00	0.60	0.39	0.01
1997	2185	2185	3304	0.339	0.314	513	0.00	0.60	0.39	0.01
1998	1769	1769	2612	0.323	0.326	409	0.00	0.60	0.39	0.01
1999	1360	1336	2032	0.343	0.436	538	0.00	0.60	0.39	0.01
2000	1323	1309	1986	0.341	0.411	473	0.00	0.60	0.39	0.01
2001	2312	1262	2445	0.484	0.390	756	0.00	0.60	0.39	0.01
2002	3039	0	1262	1.000	0.194	303	0.00	0.60	0.39	0.01
2003	2683	0	1197	1.000	0.194	288	0.00	0.60	0.39	0.01
2004	2587	0	2698	1.000	0.194	648	0.00	0.60	0.39	0.01
2005	2131	0	1653	1.000	0.194	397	0.00	0.60	0.39	0.01

4. Winter Steelhead – Clackamas

Winter steelhead were counted as they pass North Fork Dam on the Clackamas River. While the majority of the winter steelhead production is believed to be upstream from this counting location, a significant amount of steelhead habitat also exists in the portion of the basins downstream from North Fork Dam. Based upon estimates by ODFW, 40% of the production area occurs in this lower portion of the basin.

The number of total spawners for this population is based on the counts of winter steelhead at NF Dam, expanded for the production area downstream of the dam by dividing the dam count by 0.60 (Table4). As stated previously, 40% of the production of wild steelhead is thought to occur in the lower basin. This number had to be adjusted somewhat in those earlier years when a consumptive fishery was permitted on winter steelhead upstream of NF Dam. In other words, not all fish that were counted at NF Dam in those years survived to spawn.

In addition, the estimate of naturally spawning hatchery fish (which is included in the total spawner estimate) had to be adjusted to account for the hatchery fish that were removed from the counting facility at NF Dam and prevented from continuing upstream, plus the number of hatchery fish that returned to Eagle Creek Hatchery and were removed from the natural spawning population.

The identification of hatchery and wild fish in recent years was reasonably straightforward as all returning hatchery fish were identifiable by fin clip marks previously applied juvenile hatchery steelhead during hatchery rearing phase of their life history.

Estimation of hatchery and wild fish proportions prior to 1995 was more difficult because returning hatchery fish were not fin clipped. An alternate approach based on run-timing differences between wild and hatchery fish was used to make these estimates for the earlier time period.

It was found from the timing of counts of returning winter steelhead at NF Dam that prior to the first return of hatchery steelhead in 1968, less than 1% of the run passed NF Dam before March 31. However, the predominate hatchery stock used up until 1999 had a run and spawn timing that was characteristically 1 to 3 months earlier than the wild fish. It was found that from 1995 to 1999 when all returning hatchery fish were also fin clipped that the proportion of hatchery fish as estimated by the ratio of fin clips and the proportion of hatchery fish estimated by the ratio of the NF Dam fish count before March 31 and the count after March 31 was nearly the same. For these five years, 99% of the variation in the proportion of hatchery fish as determined by fin clip data, could be related to the proportion of the total run that migrated past NF Dam prior to March 31.

Based on this temporal relationship, annual winter steelhead counts at NF Dam from 1968 to 1994 were divided into an early and late portion, based on the March 31 sorting date. The early proportion was then assumed to represent the proportion of hatchery fish in that year's particular return.

Fishery impacts on this winter steelhead population occur primarily within the Clackamas basin. Catch card estimates for the Clackamas winter steelhead sport fishery, adjusted to reduce likely bias, were used to estimate the total catch. The bias adjustment consisted of multiplying all catch card estimates by 0.63. This reduction adjustment was based on data from other steelhead fisheries where catch estimates from both statistical creel surveys and catch cards were available. In these comparisons, the creel survey estimate, considered to be the more accurate of the two methods was consistently smaller. The 0.63 adjustment factor used here was based on 10 years of creel survey and catch card data collected for the winter steelhead fishery in the Alsea River. A regression of these data resulted in a significant relationship between the two (R2 = 0.87) however, with a slope of 0.63. In other words, a catch card estimate of 100, corresponded with a creel survey catch estimate of 63.

From these adjusted estimates of catch and estimates of spawner escapement, fishery impact rates were calculated. For hatchery and wild fish these rates were equal until 1992 when catch and release regulations were imposed wild fish. This regulation, in effect to the present, reduced the mortality rate on wild fish to the incidental mortality associated with the handling and stress of being caught and released. In preparing the mortality data shown in Table4 for wild fish, we assumed that 10% of those fish caught subsequently died post-release. We estimated the proportion of the wild run that was initially caught from our estimates of harvest rate on hatchery fish (for which catch and release regulations were not in effect).

Age composition data based on the analysis of scales sampled from sport steelhead fishery in Clackamas River from 1984 to 1991.

Table 4: Basic data set developed for Clackamas River winter steelhead.

Spawn Year	Total Spawners	Wild Fraction	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1958	2616	1.000	0.358	1459	0.005	0.510	0.398	0.083	0.004
1959	870	1.000	0.667	1745	0.005	0.510	0.398	0.083	0.004
1960	1829	1.000	0.453	1514	0.005	0.510	0.398	0.083	0.004
1961	3512	1.000	0.272	1312	0.005	0.510	0.398	0.083	0.004
1962	6949	1.000	0.283	2735	0.005	0.510	0.398	0.083	0.004
1963	3564	0.994	0.356	1955	0.005	0.510	0.398	0.083	0.004
1964	2999	0.999	0.503	3038	0.005	0.510	0.398	0.083	0.004
1965	2473	0.995	0.476	2235	0.005	0.510	0.398	0.083	0.004
1966	2056	0.998	0.618	3320	0.005	0.510	0.398	0.083	0.004
1967	1087	0.991	0.723	2809	0.005	0.510	0.398	0.083	0.004
1968	1259	0.971	0.815	5401	0.005	0.510	0.398	0.083	0.004
1969	3690	0.969	0.524	3935	0.005	0.510	0.398	0.083	0.004
1970	4476	0.952	0.463	3675	0.005	0.510	0.398	0.083	0.004
1971	6930	0.899	0.456	5212	0.005	0.510	0.398	0.083	0.004
1972	4197	0.936	0.615	6273	0.005	0.510	0.398	0.083	0.004
1973	3023	0.957	0.490	2781	0.005	0.510	0.398	0.083	0.004
1974	1069	0.955	0.625	1701	0.005	0.510	0.398	0.083	0.004
1975	2432	0.938	0.671	4647	0.005	0.510	0.398	0.083	0.004
1976	1883	0.867	0.533	1862	0.005	0.510	0.398	0.083	0.004
1977	2433	0.757	0.491	1778	0.005	0.510	0.398	0.083	0.004
1978	3166	0.537	0.582	2368	0.005	0.510	0.398	0.083	0.004
1979	2408	0.629	0.585	2132	0.005	0.510	0.398	0.083	0.004

1980	3290	0.820	0.668	5425	0.005	0.510	0.398	0.083	0.004
1981	4297	0.667	0.600	4294	0.005	0.510	0.398	0.083	0.004
1982	2304	0.797	0.686	4009	0.005	0.510	0.398	0.083	0.004
1983	1751	0.938	0.644	2974	0.005	0.510	0.398	0.083	0.004
1984	1973	0.797	0.635	2741	0.005	0.510	0.398	0.083	0.004
1985	1952	0.838	0.702	3863	0.005	0.510	0.398	0.083	0.004
1986	2282	0.834	0.664	3763	0.005	0.510	0.398	0.083	0.004
1987	2100	0.864	0.642	3254	0.005	0.510	0.398	0.083	0.004
1988	3378	0.836	0.608	4381	0.005	0.510	0.398	0.083	0.004
1989	1993	0.770	0.673	3165	0.005	0.510	0.398	0.083	0.004
1990	2369	0.641	0.673	3126	0.005	0.510	0.398	0.083	0.004
1991	1334	0.576	0.678	1620	0.005	0.510	0.398	0.083	0.004
1992	3452	0.687	0.055	139	0.005	0.510	0.398	0.083	0.004
1993	2230	0.859	0.067	139	0.005	0.510	0.398	0.083	0.004
1994	2064	0.940	0.078	165	0.005	0.510	0.398	0.083	0.004
1995	1886	0.803	0.052	82	0.005	0.510	0.398	0.083	0.004
1996	376	0.711	0.064	18	0.005	0.510	0.398	0.083	0.004
1997	896	0.539	0.049	25	0.005	0.510	0.398	0.083	0.004
1998	859	0.551	0.055	28	0.005	0.510	0.398	0.083	0.004
1999	388	0.760	0.037	11	0.005	0.510	0.398	0.083	0.004
2000	879	0.848	0.061	48	0.005	0.510	0.398	0.083	0.004
2001	2048	0.727	0.055	86	0.005	0.510	0.398	0.083	0.004
2002	3330	0.698	0.046	111	0.005	0.510	0.398	0.083	0.004
2003	2574	0.796	0.054	117	0.005	0.510	0.398	0.083	0.004
2004	6509	0.796	0.054	295	0.005	0.510	0.398	0.083	0.004
2005	1959	0.796	0.054	89	0.005	0.510	0.398	0.083	0.004

5. Winter Steelhead – Sandy

Total spawner abundance estimates for Sandy winter steelhead were derived from counts of steelhead passing Marmot Dam. Although there is some steelhead habitat in the basin downstream from Marmot Dam approximately 85% of the steelhead production area is upstream. For the purposes of this summary, population data is only meant to represent that portion of the basin upstream of Marmot Dam. No adjustment was made to add the 15% additional production believed to originate in the downstream portion of the watershed.

Complete counts of winter steelhead for the spawning years 1971 through 1977 and in 1983 were not available (Table5). To replace these missing data, values were generated from catch card estimates of sport catch in the same years in the following manner. A regression of sport catch and Marmot Dam counts of steelhead was made for those years when both data were available. From this relationship, which was found to have an R^2 value of 0.63 (n = 25), approximate numbers of winter steelhead for those years when no data were collected were estimated.

From 1999 to present, returning hatchery fish were indefinable because they all had been fin clipped prior to their release as smolts. Therefore, the calculation of the wild fraction in the spawning populations was relatively straightforward. However, prior to 1999, estimating the fraction of wild fish in the natural spawning population (the other portion being hatchery fish) was more difficult. To estimate the proportion of hatchery fish for this earlier period, we developed a method using the annual number of smolts released into the basin and the location of these releases.

Prior to 1989, the majority of hatchery smolts were released upstream of Marmot Dam. However, starting in 1989 the release sites were all moved downstream to reduce the number of hatchery fish homing to the upper portion of the basin. From 2000 to 2003 years the proportion of the run reaching Marmot Dam of hatchery origin averaged 0.12. It should also be noted that during this time the fishing regulations permitted the keeping of only hatchery fish and any wild fish that were caught had to be released. During this period of downstream smolt releases, hatchery and wild determinations were only made after 1999. Therefore, to estimate the fraction of hatchery fish between 1999 and 1991 (1991 being the primary adult return year for the 1989 smolt release), the average of the 2000 – 03 period was used. It should also be noted that in Table5, the fraction of wild fish is reported as being 1.000 for all years after 1998. This reflects the fact that those hatchery fish that arrived at Marmot Dam were removed during the counting procedures and prevented from continuing upstream.

Prior to 1989, hatchery smolts were released upstream of Marmot Dam and there were no differential harvest regulations on wild and hatchery fish. Scale samples obtained from Sandy steelhead caught in the sport fishery from 1984 to 1989 were analyzed and classified as either hatchery or wild fish. From these data hatchery proportions were determined. The average release of hatchery steelhead smolts for this period was related to the average proportion of hatchery fish observed during this same time frame. From these data a rough approximation of the number of wild smolts was calculated. Using this average estimate of wild smolts as a fixed number and comparing this to the number hatchery smolts released in each year prior to 1984, annual ratios of wild to hatchery smolts were generated. The proportion of adult hatchery fish was assumed to be the same as the proportion of hatchery smolts estimated two years previously (the most common ocean residence period for adults was 2 years). In this manner the proportion of hatchery fish in the spawning population (and fraction of wild fish) was estimated for the period from 1983 to 1961.

Fishery impacts on this winter steelhead population occur primarily within the Sandy basin. Catch card estimates for the Sandy winter steelhead sport fishery, adjusted to reduce likely bias, were used to estimate the total catch. The bias adjustment consisted of multiplying all catch card estimates by 0.63. This reduction adjustment was based on data from other steelhead fisheries where catch estimates from both statistical creel surveys and catch cards were available (see discussion on this topic in the previous Clackamas winter steelhead section).

From these adjusted estimates of catch and estimates of spawner escapement, fishery impact rates were calculated. For hatchery and wild fish these rates were equal until 1990 when catch and release regulations were imposed wild fish. This regulation, in effect to the present, reduced the mortality rate on wild fish to the incidental mortality associated with the handling and stress of being caught and released. In preparing the mortality data shown in Table5 for wild fish, we assumed that 10% of those fish caught subsequently died post-release. We estimated the proportion of the wild run that was initially caught from our estimates of harvest rate on hatchery fish (for which catch and release regulations were not in effect).

Age composition data based on the analysis of scales sampled from sport steelhead fishery in Sandy River from 1984 to 1991.

Table 5: Basic data set developed for Sandy River winter steelhead.

Spawn Year	Wild Fish at Dam	Wild Fish Passed[a]	Total Spawners	Wild Frac	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1961	3124	0.402	0.277	482	0.002	0.495	0.406	0.091	0.005
1962	4045	0.422	0.287	686	0.002	0.495	0.406	0.091	0.005
1963	3325	0.256	0.319	399	0.002	0.495	0.406	0.091	0.005
1964	3880	0.241	0.408	644	0.002	0.495	0.406	0.091	0.005
1965	5529	0.213	0.386	740	0.002	0.495	0.406	0.091	0.005
1966	3584	0.219	0.582	1093	0.002	0.495	0.406	0.091	0.005
1967	4076	0.220	0.541	1058	0.002	0.495	0.406	0.091	0.005
1968	2938	0.261	0.561	978	0.002	0.495	0.406	0.091	0.005
1969	3176	0.256	0.547	983	0.002	0.495	0.406	0.091	0.005
1970	2390	0.265	0.625	1057	0.002	0.495	0.406	0.091	0.005
1971	3100	0.269	0.656	1589	0.002	0.495	0.406	0.091	0.005
1972	3312	0.246	0.662	1601	0.002	0.495	0.406	0.091	0.005
1973	2243	0.263	0.613	934	0.002	0.495	0.406	0.091	0.005
1974	2311	0.260	0.618	973	0.002	0.495	0.406	0.091	0.005
1975	2951	0.261	0.651	1439	0.002	0.495	0.406	0.091	0.005
1976	2683	0.238	0.640	1136	0.002	0.495	0.406	0.091	0.005
1977	1705	0.260	0.548	537	0.002	0.495	0.406	0.091	0.005
1978	4071	0.228	0.638	1636	0.002	0.495	0.406	0.091	0.005
1979	2000	0.242	0.684	1047	0.002	0.495	0.406	0.091	0.005

1980	3015	0.207	0.730	1682	0.002	0.495	0.406	0.091	0.005
1981	4078	0.314	0.536	1477	0.002	0.495	0.406	0.091	0.005
1982	2600	0.235	0.714	1525	0.002	0.495	0.406	0.091	0.005
1983	2449	0.221	0.600	811	0.002	0.495	0.406	0.091	0.005
1984	2232	0.320	0.677	1496	0.002	0.495	0.406	0.091	0.005
1985	2787	0.211	0.699	1365	0.002	0.495	0.406	0.091	0.005
1986	2752	0.227	0.557	783	0.002	0.495	0.406	0.091	0.005
1987	3675	0.225	0.485	780	0.002	0.495	0.406	0.091	0.005
1988	3440	0.206	0.638	1250	0.002	0.495	0.406	0.091	0.005
1989	2993	0.208	0.617	1001	0.002	0.495	0.406	0.091	0.005
1990	3065	0.205	0.063	42	0.002	0.495	0.406	0.091	0.005
1991	1995	0.879	0.063	117	0.002	0.495	0.406	0.091	0.005
1992	2916	0.879	0.053	144	0.002	0.495	0.406	0.091	0.005
1993	1636	0.879	0.065	100	0.002	0.495	0.406	0.091	0.005
1994	1567	0.879	0.041	59	0.002	0.495	0.406	0.091	0.005
1995	1680	0.879	0.042	65	0.002	0.495	0.406	0.091	0.005
1996	1287	0.879	0.042	49	0.002	0.495	0.406	0.091	0.005
1997	1426	0.879	0.036	47	0.002	0.495	0.406	0.091	0.005
1998	883	0.879	0.029	23	0.002	0.495	0.406	0.091	0.005
1999	816	1.000	0.046	39	0.002	0.495	0.406	0.091	0.005
2000	741	1.000	0.043	33	0.002	0.495	0.406	0.091	0.005
2001	902	1.000	0.053	50	0.002	0.495	0.406	0.091	0.005
2002	1031	1.000	0.069	76	0.002	0.495	0.406	0.091	0.005
2003	671	1.000	0.067	48	0.002	0.495	0.406	0.091	0.005
2004	871	1.000	0.055	51	0.002	0.495	0.406	0.091	0.005
2005	626	1.000	0.055	37	0.002	0.495	0.406	0.091	0.005

6. Winter Steelhead – Hood River

The primary data source for Hood River steelhead is obtained at the fish handling facility at Powerdale Dam, near the mouth of the basin. At this facility all steelhead are counted, hatchery and wild determinations made, and scales taken from each fish for subsequent age determination. The results of this data collection effort are summarized in Table6.

Hood River steelhead are caught in both mainstem gillnet fisheries and sport fisheries in the Hood River downstream of Powerdale Dam. From 1997 to 2003, the sport catch was estimated from statistical creel surveys. The primary target of these fisheries is hatchery fish. From these creel surveys the number of hatchery fish caught was estimated. Using this number and the count of hatchery fish upstream at Powerdale Dam it was possible to estimate a harvest rate for hatchery steelhead. However, for wild steelhead the impact rate is much lower because the angling regulations required that all wild steelhead that are caught be released and not kept. It was assumed that there was a 10% mortality rate for caught and released wild steelhead. Therefore, the mortality impact rate of this sport fishery on wild fish was 0.10 times the rate estimated for hatchery fish.

Prior to 1997 there were no statistical creel surveys to estimate catch in the Hood River. For this earlier period, we used the catch card estimates for the Hood River winter steelhead fishery, adjusted downward to account for the overestimation bias of these data. The 0.47 adjustment factor applied to the catch card data for this purpose was derived from observations between 1997 and 2003. In these years, the statistical creel estimate of catch averaged 0.47 of the catch card estimate for the same period.

A portion of the Hood River winter steelhead return is also caught in mainstem Columbia gillnet fishery. Although the impact rate of this fishery is thought to be low, there is some uncertainty as what this level actually is. For the purposes of this exercise we assumed the average fishery related mortality rate on wild fish 0.05. The overall impact rate of both the mainstem and Hood River fisheries on returning adults was calculated as: 1-[(1 – ColumbiaHR) * (1 – HoodHR)].

Table 6: Basic data set for Hood River Winter Steelhead.

Spawn Year	Wild Fish at Dam	Wild Fish Passed[a]	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
							Age3	Age4	Age5	Age6	Age7
1992	688	618	902	0.685	0.082	62	0.020	0.662	0.290	0.028	0.000
1993	402	345	355	0.972	0.096	43	0.103	0.478	0.375	0.045	0.000
1994	378	300	305	0.984	0.096	40	0.028	0.724	0.243	0.005	0.000
1995	203	161	166	0.970	0.102	23	0.156	0.585	0.231	0.023	0.005
1996	275	210	371	0.566	0.094	29	0.107	0.682	0.188	0.023	0.000
1997	284	238	490	0.486	0.064	20	0.045	0.722	0.202	0.031	0.000
1998	221	182	344	0.529	0.075	18	0.066	0.644	0.279	0.011	0.000
1999	297	256	443	0.578	0.065	21	0.214	0.543	0.207	0.036	0.000
2000	912	865	1089	0.794	0.087	87	0.010	0.896	0.091	0.003	0.000
2001	1008	878	1534	0.572	0.073	79	0.028	0.681	0.274	0.017	0.000
2002	1024	950	1633	0.582	0.085	95	0.035	0.609	0.333	0.023	0.000
2003	719	654	1066	0.614	0.080	63	0.025	0.604	0.329	0.041	0.000
2004	582	507	1077	0.471	0.068	42	0.046	0.646	0.272	0.036	0.000

[a] In each year a portion of the wild return was removed to be used for hatchery program broodstock. Therefore, the number of wild fish passed upstream was less than the number that arrived at the dam.

7. Summer Steelhead – Hood River

The methods used to obtain and summarize data for Hood River summer steelhead were essentially the same as for Hood River winter steelhead described in the previous section. At the Powerdale Dam fish handling facility, all summer steelhead were counted, hatchery and wild determinations made, and scales taken from each fish for subsequent age determination. The results of this data collection effort are summarized in Table7.

Hood River steelhead are caught in both mainstem Columbia gillnet fishery and the sport fishery in the Hood River downstream of Powerdale Dam. From 1997 to 2003, the sport catch was estimated from statistical creel surveys. The primary target of this fishery is hatchery fish. The number of hatchery fish caught was estimated from these creel surveys. Using this number and the hatchery fish count upstream at Powerdale Dam it was possible to estimate a harvest rate for hatchery steelhead. However, for wild steelhead the impact rate is much lower because angling regulations required that all wild steelhead that are caught be released and not kept. It was assumed that there was a 10% mortality rate for caught and released wild steelhead. Therefore, the mortality impact rate of this sport fishery on wild fish was 0.10 times the rate estimated for hatchery fish.

Prior to 1997 there were no statistical creel surveys to estimate catch in the Hood River. For this earlier we used the catch card estimates for the Hood River winter steelhead fishery, adjusted downward to account for the overestimation bias of these data. The 0.46 adjustment factor applied to the catch card data for this purpose was derived from observations between 1997 and 2003. In these years, the statistical creel estimate of catch averaged 0.46 of the catch card estimate for the same period.

A substantial portion of the overall fishery impact on Hood River summer is due to mainstem Columbia River gillnet fisheries, especially prior to 2001. The estimated impact rates of these fisheries on wild summer steelhead were based primarily on analyses provided by ODFW and WDFW (2000).

The overall impact rate of both the mainstem and Hood River fisheries on returning adults was calculated as: 1-[(1 – ColumbiaHR) * (1 – HoodHR)]

Table 7: Basic data set for Hood River summer steelhead.

Spawn Year	Wild Fish at Dam	Wild Fish Passed[a]	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning[b]				
							Age3	Age4	Age5	Age6	Age7
1993	489	489	2211	0.221	0.179	106	0.000	0.065	0.668	0.265	0.002
1994	243	243	1348	0.180	0.175	52	0.000	0.052	0.495	0.406	0.048
1995	218	218	1845	0.118	0.122	30	0.000	0.025	0.441	0.478	0.055
1996	131	131	650	0.202	0.135	20	0.000	0.118	0.656	0.218	0.008
1997	178	178	1491	0.119	0.116	23	0.000	0.049	0.744	0.195	0.012
1998	78	65	513	0.127	0.120	11	0.000	0.118	0.628	0.254	0.000
1999	129	98	102	0.961	0.111	16	0.000	0.139	0.620	0.241	0.000
2000	180	147	149	0.987	0.096	19	0.000	0.166	0.647	0.180	0.006
2001	207	180	181	0.994	0.059	13	0.000	0.128	0.545	0.310	0.016
2002	476	415	539	0.770	0.058	30	0.000	0.166	0.740	0.086	0.008
2003	620	542	1042	0.520	0.064	42	0.000	0.121	0.517	0.337	0.026
2004	219	183	388	0.472	0.063	15	0.000	0.186	0.503	0.299	0.013
2005	180	143	311	0.460	0.062	12	0.000	0.111	0.600	0.272	0.016

[a] Starting with the 1997-98 return In each year a portion of the wild return was removed to be used for hatchery program broodstock. Therefore, the number of wild fish passed upstream was less than the number that arrived at the dam.
[b] Note that for summer steelhead scales are collected in the summer/fall time period, 6 to 12 months before spawning takes place and therefore ages determined from reading these scales were advanced one year to be standardized to the year of spawning not the year of return. For example, a summer steelhead that is determined from scales taken in July to be 4 years old, is closer to being 5-years old when it spawns the following April.

8. Coho – Clackamas

Coho are counted as they pass North Fork Dam on the Clackamas River. While the majority of the coho production is believed to be upstream from this counting location, a significant amount of coho habitat also exists in the portion of the basins downstream from North Fork Dam. Based upon estimates by ODFW, 40% of the production area occurs in this lower portion of the basin.

The number of total wild spawners for this population is based on the counts of wild coho at NF Dam, expanded for the production area downstream of the dam by dividing the dam count by 0.60 (Table8). Estimating hatchery spawner abundance was more complicated. Upstream of NF Dam, the incidence of hatchery coho in most years was thought to be very low. This conclusion was based on the very low number of fin-clipped hatchery fish observed at NF Dam counting facility (<2% of the run) in recent years. Since all hatchery coho in the lower Columbia basin had been fin-clipped prior to their release as smolts during this period, we are reasonably confident that the proportion of hatchery strays upstream of NF Dam has been low.

However, there were three times since 1957 when this has not been the case. From 1967 to 1971 a substantial number of excess hatchery fish returning to various lower Columbia hatchery facilities were transported to the basin upstream of NF Dam and released. For most of these years the number of transported hatchery fish outnumbered the count of wild fish passing NF Dam.

In 1988 -90 and again in 2000 – 02, hatchery fish from an experimental program using Clackamas wild fish as parental broodstock returned to the upper basin. In most years these hatchery fish represented less than 15% of the total spawners upstream of NF Dam.

The proportion of hatchery fish downstream of NF Dam was not been measured until recent years when extensive spawning surveys have been conducted. The results from these recent surveys document an average of proportion of hatchery fish of 0.52. These hatchery fish are most likely from the large hatchery program at Eagle Creek Hatchery in the lower basin, which has been producing coho for a long period of time. Therefore, we assumed the proportion of hatchery fish observed in recent years approximated the proportion of hatchery fish in most years since 1957. Using this assumption we were able to estimate the number of hatchery spawners from the estimated number of wild fish in the lower basin each year and the assumption that they represented 1 – 52% = 48% of the natural spawning population each year.

Wild Clackamas coho are caught primarily in ocean and Columbia River fisheries. The estimation of the impact rates for the Columbia River fisheries are complicated by the variable nature of both the run timing of natural produced coho returning to the Clackamas basin and the variable timing of the fisheries themselves. Shifts in both have occurred over the period these data. The estimates of overall fishery impacts (ocean and Columbia River) provided here are preliminary estimates prepared by ODFW and will likely change with future data and analyses.

Table 8: Basic data set for Clackamas River coho.

Spawn Year	Total Wild Fish Count[a]	Wild Fish Spawners	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Age Proportion	
							Age2	Age3
1957	678	678	887	0.764	0.942	11065	0.000 [b]	1.000
1958	433	433	567	0.764	0.940	6738	0.000	1.000
1959	1464	1464	1918	0.764	0.882	10900	0.000	1.000
1960	938	938	1228	0.764	0.751	2829	0.000	1.000
1961	2029	2029	2657	0.764	0.749	6056	0.000	1.000
1962	3731	3731	4886	0.764	0.740	10642	0.000	1.000
1963	718	718	941	0.764	0.852	4146	0.000	1.000
1964	2631	2631	3445	0.764	0.840	13817	0.000	1.000
1965	4640	4640	6076	0.764	0.824	21705	0.000	1.000
1966	739	739	968	0.764	0.833	3679	0.000	1.000
1967	1534	1534	3358	0.457	0.876	10851	0.000	1.000
1968	5816	5816	9646	0.603	0.829	28217	0.000	1.000
1969	1988	1988	3305	0.601	0.824	9324	0.000	1.000
1970	3104	3104	4065	0.764	0.858	18781	0.000	1.000
1971	5477	5477	9557	0.573	0.910	55114	0.000	1.000
1972	1372	1372	4570	0.300	0.918	15441	0.000	1.000
1973	900	900	1179	0.764	0.911	9192	0.000	1.000
1974	1261	1261	1652	0.764	0.929	16588	0.000	1.000
1975	1586	1586	2077	0.764	0.897	13858	0.000	1.000
1976	1694	1694	2218	0.764	0.954	35096	0.000	1.000
1977	1254	1254	1643	0.764	0.933	17433	0.000	1.000
1978	1096	1096	1436	0.764	0.899	9804	0.000	1.000
1979	1602	1602	2097	0.764	0.884	12229	0.000	1.000
1980	4469	4469	5852	0.764	0.874	30888	0.000	1.000
1981	1638	1638	2145	0.764	0.885	12667	0.000	1.000
1982	3574	3574	4681	0.764	0.802	14479	0.000	1.000
1983	2239	2239	2932	0.764	0.825	10572	0.000	1.000
1984	956	956	1252	0.764	0.782	3440	0.000	1.000
1985	4583	4438	5812	0.764	0.745	13354	0.000	1.000
1986	6086	5986	7839	0.764	0.829	29533	0.000	1.000
1987	1941	1886	2470	0.764	0.843	10436	0.000	1.000
1988	2267	2267	3060	0.741	0.884	17214	0.000	1.000
1989	3006	3006	4056	0.741	0.859	18248	0.000	1.000
1990	979	979	1300	0.753	0.836	4997	0.000	1.000
1991	4372	4372	5726	0.764	0.859	26545	0.000	1.000
1992	4866	4866	6373	0.764	0.764	15785	0.000	1.000
1993	235	235	308	0.764	0.747	695	0.000	1.000
1994	4036	4036	5286	0.764	0.433	3080	0.000	1.000
1995	2852	2852	3735	0.764	0.428	2137	0.000	1.000
1996	122	120	158	0.764	0.347	65	0.000	1.000
1997	1977	1896	2482	0.764	0.422	1444	0.000	1.000
1998	461	321	420	0.764	0.246	150	0.000	1.000
1999	283	153	200	0.764	0.410	197	0.000	1.000
2000	3406	3406	4855	0.702	0.215	934	0.000	1.000

2001	4392	4392	6909	0.636	0.200	1095	0.000	1.000
2002	1184	1184	1673	0.708	0.303	515	0.000	1.000
2003	2947	2947	3859	0.764	0.300	1263	0.000	1.000
2004	2681	2681	3511	0.764	0.308	1196	0.000	1.000
2005	1694	1694	2218	0.764	0.300	726	0.000	1.000

[a] In certain years a portion of the wild return was removed at the dam to be used for hatchery program broodstock. Therefore, the number of wild fish that spawned naturally was less than returned to the basin in these years.

[b] Although a variable number of age 2 jacks were observed in most years – they were not consistently counted. Since 2 year old coho are thought to be a minor contribution to the reproductive characteristics of coho populations, no attempt was made to quantify their abundance or their pre-harvest abundance.

9. Coho – Sandy River

Total spawner abundance estimates for Sandy coho were derived from counts of fish passing Marmot Dam. Although there is coho habitat in the basin downstream from Marmot Dam, most is upstream. For the purposes of this summary, population data is only meant to represent that portion of the basin upstream of Marmot Dam. No adjustment was made to add the 15% additional production believed to originate in the downstream portion of the watershed.

Complete counts of coho for the spawning years 1970 through 1977 and in 1983 were not available (Table9). To replace these missing data, values were generated from counts of wild coho observed at NF Dam on the Clackamas. A regression of Marmot and NF dam counts of coho for those years when both data were collected generated a $R^2 = 0.53$. Using this relationship, annual counts of wild fish counted at NF dam were used to predict the return of wild coho to the Sandy for those years where Marmot counts were not available.

The incidence of hatchery coho upstream of Marmot Dam in the majority of years was thought to be very low. This conclusion was based on the very low number of fin-clipped hatchery fish observed at Marmot Dam counting facility (<2% of the run) in recent years. Since all hatchery coho in the lower Columbia basin, and in particular those released into the lower Sandy basin from Cedar Creek Hatchery, had been fin-clipped prior to their release as smolts during this period, we are reasonably confident that the proportion of natural hatchery strays upstream of Marmot Dam has been low.

However, from 1964 to 1972 and again from 1980 to 1986 a substantial number of excess hatchery fish returning to Cedar Creek Hatchery and other lower Columbia hatchery facilities were transported to the basin upstream of Marmot Dam and released. When compared to the number of wild fish passing Marmot dam in these years, it was evident more than 50% of the natural spawning population were hatchery fish (Table9).

Wild Sandy coho are caught primarily in ocean and Columbia River fisheries. The estimation of the impact rates for the Columbia River fisheries are complicated by the variable nature of the fishery timing over the years since the early 1960s. The estimates of overall fishery impacts (ocean and Columbia River) provided here are preliminary estimates prepared by ODFW and will likely change with future data and analyses.

Table 9: Basic data set for Sandy River coho.

Spawn Year	Wild Spawners	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Age Proportion	
						Age2	Age3
1960	1102	1102	1.000	0.751	3323	0.000[a]	1.000
1961	1525	1525	1.000	0.749	4553	0.000	1.000
1962	1006	1006	1.000	0.740	2869	0.000	1.000
1963	1056	1056	1.000	0.852	6095	0.000	1.000
1964	749	7674	0.098	0.840	3934	0.000	1.000
1965	677	2053	0.330	0.824	3167	0.000	1.000
1966	162	947	0.171	0.833	806	0.000	1.000
1967	386	1636	0.236	0.876	2730	0.000	1.000
1968	841	1713	0.491	0.829	4081	0.000	1.000

1969	411	649	0.633	0.824	1928	0.000	1.000
1970	888	1368	0.649	0.858	5374	0.000	1.000
1971	1205	1591	0.757	0.910	12123	0.000	1.000
1972	573	900	0.637	0.918	6450	0.000	1.000
1973	457	457	1.000	0.911	4667	0.000	1.000
1974	548	548	1.000	0.929	7204	0.000	1.000
1975	619	619	1.000	0.897	5412	0.000	1.000
1976	642	642	1.000	0.954	13295	0.000	1.000
1977	546	546	1.000	0.933	7590	0.000	1.000
1978	397	397	1.000	0.899	3552	0.000	1.000
1979	652	652	1.000	0.884	4979	0.000	1.000
1980	606	1806	0.336	0.874	4189	0.000	1.000
1981	591	939	0.629	0.885	4569	0.000	1.000
1982	722	1648	0.438	0.802	2925	0.000	1.000
1983	745	745	1.000	0.825	3520	0.000	1.000
1984	798	1598	0.499	0.782	2871	0.000	1.000
1985	1445	2045	0.707	0.745	4211	0.000	1.000
1986	1546	2546	0.607	0.829	7502	0.000	1.000
1987	1205	1205	1.000	0.843	6479	0.000	1.000
1988	1506	1506	1.000	0.884	11438	0.000	1.000
1989	2182	2182	1.000	0.859	13246	0.000	1.000
1990	376	376	1.000	0.836	1920	0.000	1.000
1991	1491	1491	1.000	0.859	9052	0.000	1.000
1992	790	790	1.000	0.764	2562	0.000	1.000
1993	193	193	1.000	0.747	570	0.000	1.000
1994	601	601	1.000	0.433	459	0.000	1.000
1995	697	697	1.000	0.428	522	0.000	1.000
1996	181	181	1.000	0.347	96	0.000	1.000
1997	116	116	1.000	0.422	85	0.000	1.000
1998	261	261	1.000	0.246	85	0.000	1.000
1999	162	162	1.000	0.410	113	0.000	1.000
2000	730	730	1.000	0.215	200	0.000	1.000
2001	1388	1388	1.000	0.200	346	0.000	1.000
2002	310	310	1.000	0.303	135	0.000	1.000
2003	1173	1173	1.000	0.300	503	0.000	1.000
2004	1025	1025	1.000	0.308	457	0.000	1.000
2005	717	717	1.000	0.300	307	0.000	1.000

[a] Although a variable number of age 2 jacks were observed in most years – they were not consistently counted. Since 2 year old coho are thought to be a minor contribution to the reproductive characteristics of coho populations, no attempt was made to quantify their abundance or their pre-harvest abundance.

10. Spring Chinook – Clackamas

Spring chinook are counted as they pass North Fork Dam on the Clackamas River. While the majority of the spring chinook production occurs upstream from this counting location, 22% of the spring chinook habitat is population is thought to utilize the basin downstream of NF Dam based on data provided by Maher et al. (2005). Therefore, the number of spring chinook for the entire population was estimated by dividing the count at NF Dam by 0.78.

Only since 2002 has it been possible to visually discriminate between hatchery and wild fish as they passed NF Dam. During this period all fin clipped fish (hatchery fish) were removed from the ladder and prevented from passing upstream. Therefore, only unmarked spring chinook were present in the upper basin. However, otoliths obtained from spring chinook carcasses sampled upstream of NF Dam in 2002 and 2003 were analyzed by ODFW. Twenty six percent of the fish sampled in these years were found to have growth patterns that indicated they were hatchery fish. Therefore, the count of hatchery and wild fish at NF Dam (which used fin marks to distinguish hatchery from wild fish) was adjusted to account for this significant portion of unmarked hatchery fish.

From 1980 to 2001, the separate counts of hatchery and wild fish were not available. For the purposes of this data summary the fraction of wild fish was assumed to be equal to the proportion of wild fish estimated from 2002 to 2003 as they were counted arriving at the NF Dam (not after hatchery fish were sorted out and only unmarked fish passed upstream).

Clackamas spring chinook are caught in ocean, Columbia River, lower Willamette, and lower Clackamas fisheries. The overall fishery impact rate associated with these fisheries shown in Table10 was provided by ODFW. The age data reported here (Table10) is an average of annual data collected from Willamette basin spring chinook sampled by ODFW.

Table 10: Basic data set for Clackamas spring chinook.

Spawn Year	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1958	495	1.000	0.661	964	0.00	0.60	0.39	0.01	0.00
1959	372	1.000	0.661	725	0.00	0.60	0.39	0.01	0.00
1960	232	1.000	0.661	451	0.00	0.60	0.39	0.01	0.00
1961	285	1.000	0.661	556	0.00	0.60	0.39	0.01	0.00
1962	730	1.000	0.661	1420	0.00	0.60	0.39	0.01	0.00
1963	685	1.000	0.661	1333	0.00	0.60	0.39	0.01	0.00
1964	443	1.000	0.661	862	0.00	0.60	0.39	0.01	0.00
1965	393	1.000	0.661	765	0.00	0.60	0.39	0.01	0.00
1966	283	1.000	0.661	551	0.00	0.60	0.39	0.01	0.00
1967	168	1.000	0.661	326	0.00	0.60	0.39	0.01	0.00
1968	522	1.000	0.661	1018	0.00	0.60	0.39	0.01	0.00
1969	1164	1.000	0.660	2262	0.00	0.60	0.39	0.01	0.00
1970	737	1.000	0.672	1508	0.00	0.60	0.39	0.01	0.00
1971	426	1.000	0.648	785	0.00	0.60	0.39	0.01	0.00
1972	243	1.000	0.706	585	0.00	0.60	0.39	0.01	0.00

1973	584	1.000	0.624	968	0.00	0.60	0.39	0.01	0.00
1974	576	1.000	0.656	1098	0.00	0.60	0.39	0.01	0.00
1975	463	1.000	0.702	1092	0.00	0.60	0.39	0.01	0.00
1976	554	1.000	0.674	1146	0.00	0.60	0.39	0.01	0.00
1977	557	1.000	0.590	802	0.00	0.60	0.39	0.01	0.00
1978	532	1.000	0.637	935	0.00	0.60	0.39	0.01	0.00
1979	758	1.000	0.584	1062	0.00	0.60	0.39	0.01	0.00
1980	2716	0.471	0.541	1505	0.00	0.60	0.39	0.01	0.00
1981	3823	0.471	0.541	2118	0.00	0.60	0.39	0.01	0.00
1982	3725	0.471	0.557	2207	0.00	0.60	0.39	0.01	0.00
1983	3325	0.471	0.619	2547	0.00	0.60	0.39	0.01	0.00
1984	3498	0.471	0.598	2447	0.00	0.60	0.39	0.01	0.00
1985	2168	0.471	0.622	1682	0.00	0.60	0.39	0.01	0.00
1986	2300	0.471	0.660	2106	0.00	0.60	0.39	0.01	0.00
1987	2764	0.471	0.570	1723	0.00	0.60	0.39	0.01	0.00
1988	3954	0.471	0.555	2317	0.00	0.60	0.39	0.01	0.00
1989	3652	0.471	0.565	2235	0.00	0.60	0.39	0.01	0.00
1990	4337	0.471	0.600	3068	0.00	0.60	0.39	0.01	0.00
1991	5866	0.471	0.591	3985	0.00	0.60	0.39	0.01	0.00
1992	4495	0.471	0.448	1720	0.00	0.60	0.39	0.01	0.00
1993	3916	0.471	0.520	2000	0.00	0.60	0.39	0.01	0.00
1994	2766	0.471	0.445	1043	0.00	0.60	0.39	0.01	0.00
1995	2098	0.471	0.519	1065	0.00	0.60	0.39	0.01	0.00
1996	1137	0.471	0.431	406	0.00	0.60	0.39	0.01	0.00
1997	1622	0.471	0.338	389	0.00	0.60	0.39	0.01	0.00
1998	1786	0.471	0.263	300	0.00	0.60	0.39	0.01	0.00
1999	1101	0.471	0.342	269	0.00	0.60	0.39	0.01	0.00
2000	2724	0.471	0.331	635	0.00	0.60	0.39	0.01	0.00
2001	4694	0.410	0.298	817	0.00	0.60	0.39	0.01	0.00
2002	4572	0.693	0.155	580	0.00	0.60	0.39	0.01	0.00
2003	7828	0.784	0.145	1038	0.00	0.60	0.39	0.01	0.00
2004	6516	0.739	0.205	1244	0.00	0.60	0.39	0.01	0.00
2005	3689	0.739	0.201	685	0.00	0.60	0.39	0.01	0.00

[a] Although a minor number of age 3 jacks were observed in most years – they were not consistently counted. Since 3 year old chinook are thought to be a minor contribution to the reproductive characteristics of chinook populations, no attempt was made to quantify their abundance or their pre-harvest abundance.

11. Spring Chinook – McKenzie

The source of data used to estimate abundance of McKenzie spring chinook were counts of migrating adults passing Leaburg Dam as reported by Firman et al. (2005). Counts of jacks (age 3, precocious males) are not included in these data. Most of the spawning and rearing habitat for this population is located upstream from this counting location.

Wild and hatchery fish have both been substantial portions of the natural spawning population upstream of Leaburg Dam since 1976. Estimates of the wild fraction from 1994 to present were taken from the 2001 FMEP prepared by ODFW or Firman et al. (2005). Prior to 1994, specific wild fraction estimates were not available. For the purposes of generating data for this recovery planning effort, the wild fraction for this earlier time period was estimated from a regression between the number of hatchery fish recovered at the McKenzie Hatchery trap and the estimate of hatchery fish passing Leaburg Dam from 1994 to 2005. It was found that 77% of the variation in the estimated number of hatchery chinook passing Leaburg Dam between 1994 and 2005 could be associated with the number of fish trapped at McKenzie Hatchery. Since the number of fish trapped at McKenzie hatchery has been recorded since 1970, it was then possible to use these numbers to approximate the likely number of hatchery fish that passed Leaburg Dam from 1970 to 1993 and thereby obtain wild fraction estimates.

McKenzie spring chinook are caught in ocean, Columbia River, lower Willamette, and McKenzie River fisheries. The overall fishery impact rate associated with these fisheries shown in Table11 was calculated from the following: HRoverall = 1 – [(1-OceanHR)*(1-ColumbiaHR)*(1-WillamHR)*(1-McKenzieHR)]. The 2001 FMEP prepared by ODFW was the primary source of the fishery impact data for all fisheries except the McKenzie River fishery. In this case, the impact rate was determined by dividing the combined count of all chinook at Leaburg Dam and the McKenzie Hatchery trap for each year into an adjusted sport catch estimated based on ODFW punch-card data records. The ODFW reported sport catch estimates were adjusted downward 32% to ensure they were not overestimates of impact. From various locations in the Willamette basin both statistical creel programs and catch card estimates of sport catch have been made in at least four different years (ODFW, unpublished data). It is assumed that the creel estimates of catch are more accurate than the catch card estimates. Across all of the locations and years compared, the creel estimate of catch averaged 68% of the catch card estimate.

Finally, from 1995 onward angling regulations required the release of any fish caught without a fin clip mark. This regulation was intended to focus the fishing mortality on hatchery fish and significantly reduce the impact on wild fish. The estimated impact of these catch and release impacts on wild fish was assumed to be 10% of the average catch rate for the period in the McKenzie prior to 1995. This was based on the assumption that the interception rate on wild fish was relatively unchanged from previous years and that the delayed mortality of caught and released fish was 10%.

The age data reported here for McKenzie spring chinook were based on annual scale samples collected by ODFW from returning adult spring chinook and subsequent age analyses (Table11).

Table 11: Basic data set for McKenzie spring chinook.

Spawn Year	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3[a]	Age4	Age5	Age6	Age7
1970	2857	0.997	0.623	4705	0.00	0.51	0.48	0.01	0.00
1971	3451	0.893	0.588	4400	0.00	0.61	0.38	0.01	0.00
1972	1478	0.855	0.726	3353	0.00	0.35	0.65	0.00	0.00
1973	3742	0.859	0.597	4755	0.00	0.45	0.53	0.02	0.00
1974	3657	1.000	0.629	6193	0.00	0.42	0.55	0.03	0.00
1975	1300	1.000	0.687	2857	0.00	0.50	0.47	0.03	0.00
1976	1833	0.402	0.592	1069	0.00	0.64	0.35	0.01	0.00
1977	2650	0.634	0.518	1807	0.00	0.49	0.49	0.01	0.00
1978	3020	0.331	0.560	1272	0.00	0.49	0.50	0.01	0.00
1979	1107	0.634	0.527	781	0.00	0.40	0.59	0.01	0.00
1980	1972	0.671	0.417	947	0.00	0.50	0.47	0.03	0.00
1981	1087	0.584	0.506	650	0.00	0.48	0.50	0.01	0.00
1982	1706	0.432	0.475	666	0.00	0.59	0.40	0.01	0.00
1983	1405	0.729	0.471	913	0.00	0.60	0.40	0.01	0.00
1984	921	0.634	0.509	606	0.00	0.56	0.43	0.01	0.00
1985	808	0.634	0.522	560	0.00	0.60	0.39	0.01	0.00
1986	1736	0.432	0.484	702	0.00	0.71	0.29	0.01	0.00
1987	2933	0.714	0.512	2199	0.00	0.68	0.32	0.01	0.00
1988	6613	0.779	0.474	4647	0.00	0.63	0.36	0.01	0.00
1989	3852	0.590	0.511	2372	0.00	0.41	0.58	0.01	0.00
1990	6988	0.772	0.486	5100	0.00	0.56	0.43	0.01	0.00
1991	4287	0.473	0.541	2395	0.00	0.40	0.57	0.02	0.00
1992	3679	0.539	0.417	1421	0.00	0.32	0.67	0.02	0.00
1993	3554	0.709	0.518	2710	0.00	0.39	0.59	0.02	0.00
1994	1507	0.540	0.442	645	0.00	0.48	0.50	0.01	0.00
1995	1577	0.580	0.433	697	0.00	0.39	0.59	0.02	0.00
1996	1432	0.760	0.319	511	0.00	0.40	0.59	0.01	0.00
1997	1110	0.840	0.179	204	0.00	0.56	0.43	0.01	0.00
1998	1848	0.760	0.190	329	0.00	0.43	0.56	0.01	0.00
1999	1862	0.720	0.228	397	0.00	0.50	0.49	0.01	0.00
2000	2533	0.749	0.284	751	0.00	0.55	0.44	0.01	0.00
2001	4428	0.760	0.301	1446	0.00	0.53	0.47	0.00	0.00
2002	6774	0.623	0.152	759	0.00	0.76	0.24	0.01	0.00
2003	10524	0.550	0.142	960	0.00	0.35	0.64	0.00	0.00
2004	9043	0.529	0.203	1220	0.00	0.56	0.42	0.01	0.00
2005	3061	0.832	0.203	649	0.00	0.56	0.42	0.01	0.00

[a] Although a minor number of age 3 jacks were observed in most years (1% to 3% of the total return) – they were not consistently counted. Since 3 year old chinook are thought to be a minor contribution to the reproductive characteristics of chinook populations, no attempt was made to quantify their abundance or their pre-harvest abundance

12. Winter Steelhead – Molalla

The abundance of winter steelhead in the Molalla basin (Table12) was based on spawning survey data, adjusted so that the combined count of all steelhead populations in the Willamette steelhead ESU did not exceed the count of wild winter steelhead estimated to have passed Willamette falls. The methodology will be described in some detail for the Molalla population. For other populations, since the approach is basically the same, the reader will be referred back to the Molalla population methodology described in the following paragraphs.

Spawning surveys were conducted in the Molalla basin in most years from 1980 to 2001. The peak count of steelhead redds observed in these surveys was converted to fish per stream kilometer by multiplying the redd count by 1.35 to convert the data so that it was expressed as the number of spawners. This number was then divided by the length of survey to obtain a fish per kilometer spawner density estimate. These annual density estimates were then expanded by the 240 stream kilometers of total steelhead habitat reported by Maher (2005) for the Molalla basin.

Spawning survey data were missing for 1986 and 1987 as well as from 2002 to 2005. To fill-in these missing years of data, a regression between redds per kilometer and the count of wild winter steelhead at Willamette Falls was developed. From this relationship, the Willamette Falls count could be used to approximate Molalla steelhead redd densities for 1986-87 and 2002-05. These densities were then converted to total spawner estimates as described for the other years.

With the exception of the Upper South Santiam, similar spawning survey data sets and expansion to total spawner population estimate was the case for all other populations in the ESU (i.e., North Santiam, South Santiam, and Calapooia). However, it was noted that when all of these individual population estimates were added together, there were a number of years when this combined estimate was substantially greater or sometimes less than the count of wild winter steelhead at Willamette Falls.

To clear up this data inconsistency, a simple adjustment procedure was used, based upon the assumption that the Willamette Falls count was more accurate for the ESU, than the combined count of estimates for individual populations based on spawning surveys. The adjustment procedure involved selecting a multiplication factor that would bring the combined annual spawner estimate based on the spawning survey data into line with the total count of wild winter steelhead at Willamette Falls for each corresponding year. This correction factor was then applied to all individual population data sets, essentially standardized the population estimates such that their new combined value would match the count at Willamette Falls for each year.

Although hatchery winter steelhead have been released into the Molalla basin in past years, this program was terminated in the late 1990s. Because the particular stock of fish used in this basin had a spawn timing that was 2 months earlier than that of the wild population and the spawning surveys focused on the time period when the wild fish spawned, it is unlikely any of the redds counted during these surveys were produced by hatchery fish. We have therefore have assigned a wild fraction for this population of 1.00 in all years. However, it should be acknowledged that is not entirely accurate because an unrecorded number of hatchery fish most likely spawned naturally within the basin during part of the years covered by these data.

Steelhead from this population were caught in fisheries conducted in the Columbia, Willamette, and Molalla Rivers. The impact rates presented in this data summary are from ODFW's FMEP on Willamette steelhead. The major reduction in fishery related mortality that occurred in 1993 was caused by the switch to angling regulations that permit the retention of only fin-clipped, hatchery fish. Unclipped steelhead were assumed to wild and if caught were required to be released. A 10% percent post-release mortality was assumed for caught and released steelhead. It was assumed that the catch rate (not kill rate) of wild fish from 1993 to present was the same as for the period prior to 1993, when the catch and release regulations on wild fish were not in place.

The age composition data presented in Table12 is from scale reading analyses of scales that were sampled from wild North Santiam steelhead in the 1980s. There were insufficient scales obtained from the Molalla population during this period to make similar analyses. However, it was assumed that since both populations were from the same ESU and adjacent to each other within the Willamette basin that the age structure of the Molalla population was probably quite similar to that of the North Santiam.

Table 12: Basic data set for Molalla winter steelhead.

Spawn Year	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1980	4435	1.00	0.23	1294	0.000	0.481	0.412	0.101	0.006
1981	2583	1.00	0.23	753	0.000	0.481	0.412	0.101	0.006
1982	1322	1.00	0.23	385	0.000	0.481	0.412	0.101	0.006
1983	924	1.00	0.23	269	0.000	0.481	0.412	0.101	0.006
1984	2013	1.00	0.23	587	0.000	0.481	0.412	0.101	0.006
1985	2983	1.00	0.23	870	0.000	0.481	0.412	0.101	0.006
1986	2539	1.00	0.23	741	0.000	0.481	0.412	0.101	0.006
1987	1755	1.00	0.23	512	0.000	0.481	0.412	0.101	0.006
1988	4566	1.00	0.23	1332	0.000	0.481	0.412	0.101	0.006
1989	1334	1.00	0.23	389	0.000	0.481	0.412	0.101	0.006
1990	1654	1.00	0.23	482	0.000	0.481	0.412	0.101	0.006
1991	460	1.00	0.23	134	0.000	0.481	0.412	0.101	0.006
1992	1119	1.00	0.23	326	0.000	0.481	0.412	0.101	0.006
1993	359	1.00	0.07	27	0.000	0.481	0.412	0.101	0.006
1994	1366	1.00	0.07	101	0.000	0.481	0.412	0.101	0.006
1995	501	1.00	0.07	37	0.000	0.481	0.412	0.101	0.006
1996	355	1.00	0.07	26	0.000	0.481	0.412	0.101	0.006
1997	528	1.00	0.07	39	0.000	0.481	0.412	0.101	0.006
1998	792	1.00	0.07	59	0.000	0.481	0.412	0.101	0.006
1999	718	1.00	0.07	53	0.000	0.481	0.412	0.101	0.006
2000	800	1.00	0.07	59	0.000	0.481	0.412	0.101	0.006
2001	1752	1.00	0.07	130	0.000	0.481	0.412	0.101	0.006
2002	2865	1.00	0.07	212	0.000	0.481	0.412	0.101	0.006
2003	1532	1.00	0.07	114	0.000	0.481	0.412	0.101	0.006
2004	1570	1.00	0.07	116	0.000	0.481	0.412	0.101	0.006
2005	1093	1.00	0.07	81	0.000	0.481	0.412	0.101	0.006

13. Winter Steelhead – North Santiam

The abundance of winter steelhead in the North Santiam basin (Table13) was based on spawning survey data, adjusted so that the combined count of all steelhead populations in the Willamette steelhead ESU did not exceed the count of wild winter steelhead estimated to have passed Willamette Falls. See the Molalla winter steelhead population section for a more detailed description of this methodology.

Spawning survey data for this basin was missing for quite few of the years. When the missing data was represented by a single year, 1984, 1986, 1990, and 1996 an approximate value was filled in by taking the average of the year before and after the missing data point. When the missing data was for a string of two or more years, in this case 1980-82 and 1999-00, the fill-in values were obtained from a regression of known data point with a paired data set for the Calapooia population. From this relationship and the redd densities observed in the Calapooia, redd density values for the North Santiam were generated for the missing data years.

Until the 2001 return, hatchery winter steelhead were present within the North Santiam basin. Because this particular hatchery stock was derived from the later spawning wild fish, the spawn timing was similar. This meant that the redd counts made during spawning surveys likely included some that were produced by hatchery fish. Therefore, the estimate of total spawner abundance had to be split between hatchery and wild fish to accommodate this situation. This was done using data obtained from 1993 to 2000 when it was possible to identify hatchery and wild fish in fishery recoveries and counting locations on the basis of the presence or absence of a fin clip. The average fraction of wild fish observed for this time period was applied to previous years as a means to estimate the wild fraction for this earlier time period.

Steelhead from this population were caught in fisheries conducted in three the locations: the Columbia, Willamette, and North Santiam Rivers. The impact rates presented in this data summary are from ODFW's FMEP on Willamette steelhead. The major reduction in fishery associated mortality that occurred in 1993 was caused by the switch to angling regulations that permit the retention of only fin-clipped, hatchery fish. Unclipped steelhead were assumed to wild and if caught were required to be released. A 10% percent post-release mortality was assumed for caught and released steelhead. It was assumed that the catch rate (not kill rate) of wild fish from 1993 to present was the same as for the period prior to 1993, when the catch and release regulations on wild fish were not in place. The age composition data presented in Table13 is from the scale reading analyses of scales that were sampled from wild North Santiam steelhead in the 1980s.

Table 13: Basic data set for North Santiam winter steelhead.

Spawn Year	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1980	5700	0.852	0.23	1416	0.000	0.481	0.412	0.101	0.006
1981	3491	0.852	0.23	868	0.000	0.481	0.412	0.101	0.006
1982	3081	0.852	0.23	766	0.000	0.481	0.412	0.101	0.006
1983	3066	0.852	0.23	762	0.000	0.481	0.412	0.101	0.006
1984	6307	0.852	0.23	1567	0.000	0.481	0.412	0.101	0.006
1985	8375	0.852	0.23	2081	0.000	0.481	0.412	0.101	0.006

1986	7368	0.852	0.23	1831	0.000	0.481	0.412	0.101	0.006
1987	4876	0.852	0.23	1212	0.000	0.481	0.412	0.101	0.006
1988	5104	0.852	0.23	1268	0.000	0.481	0.412	0.101	0.006
1989	3604	0.852	0.23	896	0.000	0.481	0.412	0.101	0.006
1990	4534	0.852	0.23	1127	0.000	0.481	0.412	0.101	0.006
1991	1428	0.852	0.23	355	0.000	0.481	0.412	0.101	0.006
1992	1847	0.852	0.23	459	0.000	0.481	0.412	0.101	0.006
1993	2160	0.837	0.07	134	0.000	0.481	0.412	0.101	0.006
1994	1944	0.868	0.07	125	0.000	0.481	0.412	0.101	0.006
1995	1236	0.889	0.07	81	0.000	0.481	0.412	0.101	0.006
1996	618	0.889	0.07	41	0.000	0.481	0.412	0.101	0.006
1997	2379	0.911	0.07	161	0.000	0.481	0.412	0.101	0.006
1998	2006	0.695	0.07	103	0.000	0.481	0.412	0.101	0.006
1999	2781	0.732	0.07	151	0.000	0.481	0.412	0.101	0.006
2000	1593	0.876	0.07	103	0.000	0.481	0.412	0.101	0.006
2001	4507	1.000	0.07	334	0.000	0.481	0.412	0.101	0.006
2002	7368	1.000	0.07	546	0.000	0.481	0.412	0.101	0.006
2003	4151	1.000	0.07	308	0.000	0.481	0.412	0.101	0.006
2004	4217	1.000	0.07	313	0.000	0.481	0.412	0.101	0.006
2005	2251	1.000	0.07	167	0.000	0.481	0.412	0.101	0.006

14. Winter Steelhead – South Santiam

The abundance of winter steelhead in the South Santiam basin was based on two methods. For the area downstream of Foster Dam (approximately ½ of the basin's steelhead habitat) spawning survey data was used, adjusted so that the combined count of all steelhead populations in the Willamette steelhead ESU did not exceed the count of wild winter steelhead estimated to have passed Willamette falls. See the Molalla winter steelhead population section for a more detailed description of this methodology.

Counts of winter steelhead at Foster Dam were used to estimate spawner abundance for the upper portion of the basin. Numbers from both areas (and methods) were combined to obtain the total spawner data presented in Table14.

The data set of winter steelhead counts at Foster Dam start in 1968, however the spawner survey data for the lower portion of the basin (downstream of Foster Dam) do not start until 1980. To approximate the number of spawners in the lower basin between 1968 and 1980, a relationship was developed between the Foster Dam counts and spawner abundance estimates for the basin downstream of Foster Dam derived from the spawning survey methodology.

Using this relationship, the Foster Dam counts were used to approximate the lower basin spawner escapement. It should be noted that Green Peter Dam (upstream of Foster Dam) was still passing wild steelhead during this earlier period. However, the steelhead return above Green Peter went extinct in the late 1970s. Therefore, to make the Foster Dam counts used for the prediction regression (post-1980) comparable to the Foster Dam counts in the 1970s, it was necessary to subtract out the number of steelhead counted passing Green Peter Dam.

Finally, it should be noted that spawning surveys in the lower section of the South Santiam were not conducted every year. The years with missing data were the same as the case for the North Santiam. These missing data points were filled in following the same procedure as described for the North Santiam.

With the exception of a period during the 1980s, there has been no hatchery winter steelhead program in the South Santiam. The wild fraction among the spawning population was essentially 1.00 in all years except during this period in the 1980s. During this period the wild fraction was computed as the total spawner estimate minus the hatchery fish counted at Foster Dam, divided by the total spawner estimate.

Steelhead from this population are caught in fisheries conducted in three the locations: the Columbia, Willamette, and South Santiam Rivers. The impact rates presented in this data summary are from ODFW's FMEP on Willamette steelhead. The major reduction in fishery associated mortality that occurred in 1993 was caused by the switch to angling regulations that permit the retention of only fin-clipped, hatchery fish. Unclipped steelhead were assumed to wild and if caught were required to be released. A 10% percent post-release mortality was assumed for caught and released steelhead. It was assumed that the catch rate (not kill rate) of wild fish from 1993 to present was the same as for the period prior to 1993, when the catch and release regulations on wild fish were not in place.

The age composition data presented in Table14 is from the scale reading analyses of scales that were sampled from wild North Santiam steelhead in the 1980s. There were insufficient scales obtained from the South Santiam population during this period to make similar analyses. However, it was assumed that since both populations were from the same ESU and adjacent to each other within the Willamette basin that the age structure of the South Santiam population was probably quite similar to that of the North Santiam.

Table 14: Basic data set for North Santiam winter steelhead.

Spawn Year	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1968	3674	1.00	0.23	1072	0.000	0.481	0.412	0.101	0.006
1969	5367	1.00	0.23	1565	0.000	0.481	0.412	0.101	0.006
1970	4777	1.00	0.23	1393	0.000	0.481	0.412	0.101	0.006
1971	12667	1.00	0.23	3694	0.000	0.481	0.412	0.101	0.006
1972	7191	1.00	0.23	2097	0.000	0.481	0.412	0.101	0.006
1973	3172	1.00	0.23	925	0.000	0.481	0.412	0.101	0.006
1974	2966	1.00	0.23	865	0.000	0.481	0.412	0.101	0.006
1975	2032	1.00	0.23	593	0.000	0.481	0.412	0.101	0.006
1976	1840	1.00	0.23	537	0.000	0.481	0.412	0.101	0.006
1977	2291	1.00	0.23	668	0.000	0.481	0.412	0.101	0.006
1978	2227	1.00	0.23	650	0.000	0.481	0.412	0.101	0.006
1979	1408	1.00	0.23	411	0.000	0.481	0.412	0.101	0.006
1980	7213	1.00	0.23	2104	0.000	0.481	0.412	0.101	0.006
1981	4600	1.00	0.23	1342	0.000	0.481	0.412	0.101	0.006
1982	3772	0.96	0.23	1052	0.000	0.481	0.412	0.101	0.006
1983	1686	0.96	0.23	473	0.000	0.481	0.412	0.101	0.006
1984	4756	0.79	0.23	1097	0.000	0.481	0.412	0.101	0.006
1985	5600	0.89	0.23	1450	0.000	0.481	0.412	0.101	0.006
1986	5005	0.90	0.23	1318	0.000	0.481	0.412	0.101	0.006
1987	3408	0.93	0.23	920	0.000	0.481	0.412	0.101	0.006
1988	6604	0.94	0.23	1803	0.000	0.481	0.412	0.101	0.006
1989	1636	0.96	0.23	459	0.000	0.481	0.412	0.101	0.006
1990	2786	1.00	0.23	810	0.000	0.481	0.412	0.101	0.006
1991	1275	1.00	0.23	372	0.000	0.481	0.412	0.101	0.006
1992	2144	1.00	0.23	625	0.000	0.481	0.412	0.101	0.006
1993	1275	1.00	0.07	94	0.000	0.481	0.412	0.101	0.006
1994	1923	1.00	0.07	143	0.000	0.481	0.412	0.101	0.006
1995	2118	1.00	0.07	157	0.000	0.481	0.412	0.101	0.006
1996	1006	1.00	0.07	75	0.000	0.481	0.412	0.101	0.006
1997	1248	1.00	0.07	92	0.000	0.481	0.412	0.101	0.006
1998	967	1.00	0.07	72	0.000	0.481	0.412	0.101	0.006
1999	3580	1.00	0.07	265	0.000	0.481	0.412	0.101	0.006
2000	2256	1.00	0.07	167	0.000	0.481	0.412	0.101	0.006
2001	4951	1.00	0.07	367	0.000	0.481	0.412	0.101	0.006
2002	4663	1.00	0.07	345	0.000	0.481	0.412	0.101	0.006
2003	2384	1.00	0.07	176	0.000	0.481	0.412	0.101	0.006
2004	4487	1.00	0.07	333	0.000	0.481	0.412	0.101	0.006
2005	2155	1.00	0.07	160	0.000	0.481	0.412	0.101	0.006

15. Winter Steelhead – Calapooia

The abundance of winter steelhead in the Calapooia basin (Table15) was based on spawning survey data, adjusted so that the combined count of all steelhead populations in the Willamette steelhead ESU did not exceed the count of wild winter steelhead estimated to have passed Willamette falls. See the Molalla winter steelhead population section for a more detailed description of this methodology.

Spawning survey data for this basin was missing for 1984, 1986, 1990, 1996, and 1999. An approximate value for these single data points was filled in by averaging the redds per kilometer values for year before and after the year for which there were no data.

Hatchery steelhead have never been released in the Calapooia basin and the strays from other hatchery programs have never been observed. Therefore, the fraction of wild fish for this population was assumed to 1.00 in all years.

Steelhead from this population are caught in fisheries conducted in three the locations: the Columbia, Willamette, and Calapooia Rivers. The impact rates presented in this data summary are from ODFW's FMEP on Willamette steelhead. The major reduction in fishery associated mortality that occurred in 1993 was caused by the switch to angling regulations that permit the retention of only fin-clipped, hatchery fish. Unclipped steelhead were assumed to wild and if caught were required to be released. A 10% percent post-release mortality was assumed for caught and released steelhead. It was assumed that the catch rate (not kill rate) of wild fish from 1993 to present was the same as for the period prior to 1993, when the catch and release regulations on wild fish were not in place.

The age composition data presented in Table15 is from the scale reading analyses of scales that were sampled from wild North Santiam steelhead in the 1980s. There were insufficient scales obtained from the Calapooia population during this period to make a similar analyses. However, it was assumed that since both populations were from the same ESU, the age structure of the Calapooia population was similar to that of the North Santiam.

Table 15: Basic data set for Calapooia winter steelhead.

Spawn Year	Total Spawners	Wild Frac	Overall Fishery Mortality	Total Wild Catch	Proportion by Age at Spawning				
					Age3	Age4	Age5	Age6	Age7
1980	859	1.00	0.23	251	0.000	0.481	0.412	0.101	0.006
1981	421	1.00	0.23	123	0.000	0.481	0.412	0.101	0.006
1982	597	1.00	0.23	174	0.000	0.481	0.412	0.101	0.006
1983	491	1.00	0.23	143	0.000	0.481	0.412	0.101	0.006
1984	933	1.00	0.23	272	0.000	0.481	0.412	0.101	0.006
1985	1179	1.00	0.23	344	0.000	0.481	0.412	0.101	0.006
1986	1174	1.00	0.23	342	0.000	0.481	0.412	0.101	0.006
1987	916	1.00	0.23	267	0.000	0.481	0.412	0.101	0.006
1988	1620	1.00	0.23	472	0.000	0.481	0.412	0.101	0.006
1989	246	1.00	0.23	72	0.000	0.481	0.412	0.101	0.006
1990	482	1.00	0.23	141	0.000	0.481	0.412	0.101	0.006
1991	227	1.00	0.23	66	0.000	0.481	0.412	0.101	0.006
1992	157	1.00	0.23	46	0.000	0.481	0.412	0.101	0.006

1993	54	1.00	0.07	4	0.000	0.481	0.412	0.101	0.006
1994	212	1.00	0.07	16	0.000	0.481	0.412	0.101	0.006
1995	135	1.00	0.07	10	0.000	0.481	0.412	0.101	0.006
1996	102	1.00	0.07	8	0.000	0.481	0.412	0.101	0.006
1997	505	1.00	0.07	37	0.000	0.481	0.412	0.101	0.006
1998	448	1.00	0.07	33	0.000	0.481	0.412	0.101	0.006
1999	428	1.00	0.07	32	0.000	0.481	0.412	0.101	0.006
2000	211	1.00	0.07	16	0.000	0.481	0.412	0.101	0.006
2001	1052	1.00	0.07	78	0.000	0.481	0.412	0.101	0.006
2002	1417	1.00	0.07	105	0.000	0.481	0.412	0.101	0.006
2003	838	1.00	0.07	62	0.000	0.481	0.412	0.101	0.006
2004	1319	1.00	0.07	98	0.000	0.481	0.412	0.101	0.006
2005	339	1.00	0.07	25	0.000	0.481	0.412	0.101	0.006

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Appendix C: Washington Abundance Time Series

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Overview

The following time series were used to estimate population variance as part of the viability curve analyses. The time series were compiled from a number of sources, which are described at immediately before the data. In general, the abundance numbers come from the SASSI reports available on the internet on September 21, 2006. The other information is generally based on datasets used for the 2005 BRT status reports (Good et al. 2005). Interpolation and extrapolation were used to fill in missing data as noted. The data in these time series should generally be considered to contain substantial measurement error.

Lower Columbia Chinook ESU

Coweeman Fall Chinook

Table 1: Coweeman fall chinook. Abundance time series 1974-2000 from BRT data. Hatchery fraction, harvest rate, and age structure from BRT data through 2000 except the years 1974-1979 for which only the number of spawners are available.

Year	Spawners	Frac Wild	Catch	Age1	Age2	Age3	Age4	Age5	Regime
1974	156	1	411.1651	0	0.008144	0.193196	0.658828	0.139831	1
1975	296	1	766.7287	0	0.011446	0.221415	0.575154	0.191984	1
1976	528	1	1234.452	0	0.029573	0.258948	0.539344	0.172135	1
1977	337	1	822.0096	0	0.020239	0.281036	0.554216	0.14451	1
1978	243	1	632.1514	0	0.026512	0.163535	0.661671	0.148281	1
1979	344	1	808.9804	0	0.028628	0.205728	0.557856	0.207788	1
1980	180	1	467.6704	0	0.034147	0.319117	0.539718	0.107017	1
1981	116	1	354.7915	0	0.085859	0.08511	0.671753	0.157278	1
1982	149	1	310.8108	0	0.072795	0.136401	0.682311	0.108494	1
1983	122	1	184.1893	0	0	0.295695	0.704305	0	1
1984	683	1	1469.031	0	0.108752	0.060821	0.627884	0.202544	1
1985	491	1	495.6582	0	0.054283	0.181791	0.632135	0.13179	1
1986	396	1	732.641	0	0.107388	0.193396	0.392989	0.306228	1
1987	386	1	946.4771	0	0.131326	0.133325	0.39403	0.34132	1
1988	1890	1	5459.079	0	0.019518	0.239038	0.688146	0.053298	1
1989	2549	1	4227.767	0	0.011611	0.049362	0.361827	0.577199	1
1990	812	1	1087.904	0	0.044531	0.142168	0.308381	0.504921	1
1991	340	1	420.2416	0	0	0.190374	0.336362	0.473265	1
1992	1247	1	915.0649	0	0.015098	0.055277	0.713301	0.216324	1
1993	890	1	1850.743	0	0.022074	0.19323	0.502596	0.2821	1
1994	1695	1	812.6174	0	0.038167	0.219079	0.475003	0.267752	1
1995	1368	1	315.5211	0	0.021048	0.259963	0.46541	0.253579	1
1996	2305	1	769.5067	0	0.001519	0.144222	0.596439	0.257819	1
1997	689	1	379.7632	0	0	0.005677	0.606143	0.38818	1
1998	491	1	589.9693	0	0.006505	0.039854	0.318893	0.634749	1
1999	299	1	593.0077	0	0.010587	0.175345	0.240682	0.573387	1
2000	290	1	501.5362	0	0.007107	0.069033	0.849781	0.07408	1
2001	802	1	722.6773	0	0.012279	0.140503	0.585346	0.261872	1
2002	877	1	1810.272	0	0.003315	0.159369	0.563227	0.274088	1
2003	1106	1	5195.55	0	0.001241	0.051171	0.284635	0.662953	1
2004	1503	1	2771.614	0	0.00879	0.049703	0.594502	0.347005	1

Cowlitz Fall Chinook

Table 2: Cowlitz fall chinook. Abundance time series 1964 -1985 from BRT report data; 1986-2002 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. For 2001-2002, hatchery fraction is the average of 1999-2000 and age structure is the average of 1994-2000. The harvest rate for 1999-2002 is the average of the corresponding years for Coweeman fall chinook.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	Regime
1964	3312	-99	-99	-99	-99	-99	-99	-99	1
1965	5707	-99	-99	-99	-99	-99	-99	-99	1
1966	4782	-99	-99	-99	-99	-99	-99	-99	1
1967	5487	-99	-99	-99	-99	-99	-99	-99	1
1968	2303	-99	-99	-99	-99	-99	-99	-99	1
1969	4260	-99	-99	-99	-99	-99	-99	-99	1
1970	9706	-99	-99	-99	-99	-99	-99	-99	1
1971	22758	-99	-99	-99	-99	-99	-99	-99	1
1972	21027	-99	-99	-99	-99	-99	-99	-99	1
1973	8390	-99	-99	-99	-99	-99	-99	-99	1
1974	7566	-99	-99	-99	-99	-99	-99	-99	1
1975	4766	-99	-99	-99	-99	-99	-99	-99	1
1976	3726	-99	-99	-99	-99	-99	-99	-99	1
1977	5837	-99	-99	-99	-99	-99	-99	-99	1
1978	3192	-99	-99	-99	-99	-99	-99	-99	1
1979	8253	-99	-99	-99	-99	-99	-99	-99	1
1980	1793	0.261	-99	0	0.331	0.177	0.256	0.229	1
1981	3213	0.261	-99	0	0.331	0.177	0.256	0.229	1
1982	2100	0.261	1598	0	0.331	0.177	0.256	0.229	1
1983	2463	0.261	1209	0	0.331	0.177	0.256	0.229	1
1984	1737	0.261	947	0	0.331	0.177	0.256	0.229	1
1985	2229	0.261	1394	0	0.331	0.177	0.256	0.229	1
1986	6390	0.261	2717	0	0.331	0.177	0.256	0.229	1
1987	7990	0.261	2149	0	0.331	0.177	0.256	0.229	1
1988	7375	0.261	2410	0	0.331	0.177	0.256	0.229	1
1989	2750	0.261	1842	0	0.331	0.177	0.256	0.229	1
1990	2680	0.261	857	0	0.331	0.177	0.256	0.229	1
1991	2683	0.12	322	0	0.331	0.177	0.256	0.229	1
1992	2374	0.12	224	0	0.331	0.177	0.256	0.229	1
1993	2634	0.062	293	0	1	0	0	0	1
1994	2351	0.189	19	0	0.247	0	0	0.753	1
1995	1707	0.131	159	0	0.285	0.687	0	0	1
1996	2724	0.577	310	0	0.092	0	0.703	0.205	1
1997	2160	0.715	366	0	0	0.013	0.703	0.285	1
1998	1045	0.367	243	0	0.085	0.244	0.643	0	1
1999	2700	0.156	227	0	0.212	0.311	0	0.477	1
2000	5013	0.097	422	0	0.727	0.158	0	0.115	1
2001	14427	0.1265	1213	0	0.318	0.174	0.232	0.272	1
2002	10329	0.1265	869	0	0.318	0.174	0.232	0.272	1

East Fork Lewis Fall Chinook

Table 3: East Fork Lewis fall chinook. Abundance time series 1980-1985 from BRT report data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. For 2001-2003, hatchery fraction is the average of 1998-2000 and age structure is the average from 1984-2000. Harvest rate for 1999-2003 is the average of the corresponding years for Coweeman fall chinook.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	Regime
1980	484	1	-99	0	0.083	0.205	0.485	0.227	1
1981	403	1	-99	0	0.083	0.205	0.485	0.227	1
1982	318	1	390	0	0.083	0.205	0.485	0.227	1
1983	307	1	260	0	0.083	0.205	0.485	0.227	1
1984	184	1	194	0	0.071	0.089	0.768	0.071	1
1985	600	1	357	0	0.174	0.211	0.462	0.153	1
1986	445	1	492	0	0.127	0.394	0.412	0.068	1
1987	157	1	637	0	0.141	0.244	0.449	0.167	1
1988	476	1	534	0	0.101	0.143	0.584	0.172	1
1989	591	0.78	288	0	0.055	0	0.39	0.555	1
1990	342	1	157	0	0.039	0.163	0.264	0.534	1
1991	230	1	231	0	0.08	0.32	0.32	0.28	1
1992	202	1	206	0	0.056	0.157	0.694	0.093	1
1993	156	1	140	0	0.071	0.238	0.488	0.202	1
1994	395	1	80	0	0.247	0.065	0.521	0.167	1
1995	100	1	291	0	0.099	0.162	0.27	0.468	1
1996	167	1	90	0	0.012	0.189	0.692	0.107	1
1997	184	1	141	0	0	0.022	0.62	0.359	1
1998	52	1	103	0	0.055	0.491	0.236	0.218	1
1999	109	1	71	0	0.027	0.45	0.423	0.099	1
2000	323	1	211	0	0.056	0.149	0.646	0.149	1
2001	530	1	347	0	0.095	0.178	0.49	0.237	1
2002	1375	1	899	0	0.095	0.178	0.49	0.237	1
2003	727	1	476	0	0.095	0.178	0.49	0.237	1

Elochoman Fall Chinook

Table 4: Elochoman fall chinook. Abundance time series 1964-1985 from BRT data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. For 2001-2003 age structure is the average from 1991-2000, hatchery fraction is the average for 1998-2000, and harvest is the average for 1998-2000 Coweeman fall chinook adjusted for the Elochoman fall chinook proportion of wild catch.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5
1964	95	-99	-99	-99	-99	-99	-99	-99
1965	191	-99	-99	-99	-99	-99	-99	-99
1966	155	-99	-99	-99	-99	-99	-99	-99
1967	347	-99	-99	-99	-99	-99	-99	-99
1968	756	-99	-99	-99	-99	-99	-99	-99
1969	301	-99	-99	-99	-99	-99	-99	-99
1970	455	-99	-99	-99	-99	-99	-99	-99
1971	367	-99	-99	-99	-99	-99	-99	-99
1972	108	-99	-99	-99	-99	-99	-99	-99
1973	500	-99	-99	-99	-99	-99	-99	-99
1974	245	-99	-99	-99	-99	-99	-99	-99
1975	220	-99	-99	-99	-99	-99	-99	-99
1976	1682	-99	-99	-99	-99	-99	-99	-99
1977	568	-99	-99	-99	-99	-99	-99	-99
1978	1846	-99	-99	-99	-99	-99	-99	-99
1979	1478	-99	-99	-99	-99	-99	-99	-99
1980	64	0.415	-99	0	0.139	0.467	0.348	0.042
1981	138	0.415	-99	0	0.139	0.467	0.348	0.042
1982	340	0.415	389	0	0.139	0.467	0.348	0.042
1983	1016	0.415	439	0	0.139	0.467	0.348	0.042
1984	294	0.415	355	0	0.139	0.467	0.348	0.042
1985	464	0.415	480	0	0.139	0.467	0.348	0.042
1986	915	0.415	1343	0	0.139	0.467	0.348	0.042
1987	2458	0.415	1231	0	0.139	0.467	0.348	0.042
1988	1370	0.415	1284	0	0.139	0.467	0.348	0.042
1989	122	0.415	649	0	0.139	0.467	0.348	0.042
1990	174	0.415	201	0	0.139	0.467	0.348	0.042
1991	196	0.092	123	0	1	0	0	0
1992	190	1	224	0	0	0.1	0.9	0
1993	288	0.778	423	0	0.063	0.08	0.839	0.018
1994	521	0.982	22	0	0.026	0.821	0.146	0.007
1995	156	0.5	57	0	0.077	0.244	0.551	0.128
1996	533	0.655	78	0	0.072	0.693	0.212	0.023
1997	1875	0.107	98	0	0	1	0	0
1998	228	0.25	104	0	0.088	0.614	0.246	0.053
1999	718	0.251	529	0	0.061	0.739	0	0.2
2000	196	0.617	145	0	0	0.38	0.587	0.033
2001	2354	0.373	1736	0	0.0698	0.7136	0.1752	0.0415
2002	7581	0.414	5589	0	0.0698	0.7136	0.1752	0.0415
2003	6820	0.468	5028	0	0.0698	0.7136	0.1752	0.0415

Grays Fall Chinook

Table 5: Grays fall chinook. Abundance time series 1964-1985 from BRT data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. For 2001-2003, age structure is the average for 1991-2000 and the catch rate is the average for 1998-2000.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	Regime
1964	92	-99	-99	-99	-99	-99	-99	-99	1
1965	136	-99	-99	-99	-99	-99	-99	-99	1
1966	127	-99	-99	-99	-99	-99	-99	-99	1
1967	137	-99	-99	-99	-99	-99	-99	-99	1
1968	338	-99	-99	-99	-99	-99	-99	-99	1
1969	129	-99	-99	-99	-99	-99	-99	-99	1
1970	359	-99	-99	-99	-99	-99	-99	-99	1
1971	622	-99	-99	-99	-99	-99	-99	-99	1
1972	674	-99	-99	-99	-99	-99	-99	-99	1
1973	503	-99	-99	-99	-99	-99	-99	-99	1
1974	624	-99	-99	-99	-99	-99	-99	-99	1
1975	706	-99	-99	-99	-99	-99	-99	-99	1
1976	1144	-99	-99	-99	-99	-99	-99	-99	1
1977	1495	-99	-99	-99	-99	-99	-99	-99	1
1978	2685	-99	-99	-99	-99	-99	-99	-99	1
1979	1206	-99	-99	-99	-99	-99	-99	-99	1
1980	197	0.652	-99	0	0.048	0.271	0.566	0.115	1
1981	351	0.652	-99	0	0.048	0.271	0.566	0.115	1
1982	422	0.652	909	0	0.048	0.271	0.566	0.115	1
1983	927	0.652	796	0	0.048	0.271	0.566	0.115	1
1984	340	0.652	591	0	0.048	0.271	0.566	0.115	1
1985	838	0.652	976	0	0.048	0.271	0.566	0.115	1
1986	1047	0.652	1961	0	0.048	0.271	0.566	0.115	1
1987	1113	0.652	1428	0	0.048	0.271	0.566	0.115	1
1988	1010	0.652	1309	0	0.048	0.271	0.566	0.115	1
1989	813	0.652	835	0	0.048	0.271	0.566	0.115	1
1990	287	0.652	351	0	0.048	0.271	0.566	0.115	1
1991	200	0.935	100	0	0.064	0.385	0.316	0.235	1
1992	4	1	71	0	0	0	1	0	1
1993	43	1	84	0	0.07	0.372	0.535	0.023	1
1994	47	1	8	0	0	0.255	0.745	0	1
1995	29	1	17	0	0	0	0.517	0.483	1
1996	365	0.479	24	0	0.08	0.646	0.194	0.08	1
1997	14	0.643	39	0	0.222	0.222	0.556	0	1
1998	93	0.409	19	0	0	0.789	0	0.211	1
1999	303	0.508	201	0	0	0	1	0	1
2000	97	0.959	64	0	0.043	0.043	0.796	0.118	1
2001	251	0.625	166	0	0.044	0.353	0.501	0.102	1
2002	82	0.625	54	0	0.044	0.353	0.501	0.102	1
2003	387	0.625	256	0	0.044	0.353	0.501	0.102	1

Kalama Fall Chinook

Table 6: Kalama fall chinook. Abundance time series 1964-1985 from BRT data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. For 2001-2003, the hatchery fraction is the average for 1998-2000 and age structure is the average for 1992-2000. Harvest rate for 1999-2000 is the average of Coweeman fall chinook for 1999-2003.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	Regime
1964	4942	-99	-99	-99	-99	-99	-99	-99	1
1965	5559	-99	-99	-99	-99	-99	-99	-99	1
1966	2739	-99	-99	-99	-99	-99	-99	-99	1
1967	3308	-99	-99	-99	-99	-99	-99	-99	1
1968	2893	-99	-99	-99	-99	-99	-99	-99	1
1969	2381	-99	-99	-99	-99	-99	-99	-99	1
1970	2976	-99	-99	-99	-99	-99	-99	-99	1
1971	3165	-99	-99	-99	-99	-99	-99	-99	1
1972	3465	-99	-99	-99	-99	-99	-99	-99	1
1973	6262	-99	-99	-99	-99	-99	-99	-99	1
1974	12834	-99	-99	-99	-99	-99	-99	-99	1
1975	18123	-99	-99	-99	-99	-99	-99	-99	1
1976	8352	-99	-99	-99	-99	-99	-99	-99	1
1977	6549	-99	-99	-99	-99	-99	-99	-99	1
1978	3711	-99	-99	-99	-99	-99	-99	-99	1
1979	2731	-99	-99	-99	-99	-99	-99	-99	1
1980	5850	0.503	-99	0	0.024	0.337	0.399	0.24	1
1981	1917	0.503	-99	0	0.024	0.337	0.399	0.24	1
1982	4595	0.503	4291	0	0.024	0.337	0.399	0.24	1
1983	2722	0.503	2948	0	0.024	0.337	0.399	0.24	1
1984	3043	0.503	1897	0	0.024	0.337	0.399	0.24	1
1985	1259	0.503	3308	0	0.024	0.337	0.399	0.24	1
1986	2601	0.503	11757	0	0.024	0.337	0.399	0.24	1
1987	9651	0.503	13688	0	0.024	0.337	0.399	0.24	1
1988	24549	0.503	18587	0	0.024	0.337	0.399	0.24	1
1989	20495	0.503	13116	0	0.024	0.337	0.399	0.24	1
1990	2157	0.503	5478	0	0.024	0.337	0.399	0.24	1
1991	5152	0.541	2026	0	0.024	0.228	0.628	0.12	1
1992	3683	0.475	2343	0	0.051	0.411	0.418	0.12	1
1993	1961	0.887	3911	0	0.011	0.097	0.764	0.128	1
1994	2190	0.731	184	0	0.106	0.691	0.096	0.106	1
1995	3094	0.686	630	0	0.024	0.272	0.592	0.113	1
1996	10676	0.443	1017	0	0	0.42	0.491	0.089	1
1997	3548	0.398	1128	0	0.006	0	0.72	0.274	1
1998	4355	0.691	550	0	0.012	0.545	0.282	0.161	1
1999	2655	0.031	459	0	0	0	0	1	1
2000	1420	0.187	245	0	0	0.707	0	0.293	1
2001	3714	0.303	642	0	0.018	0.355	0.42	0.208	1
2002	18952	0.303	3275	0	0.018	0.355	0.42	0.208	1
2003	24782	0.303	4282	0	0.018	0.355	0.42	0.208	1

Lewis River Late Fall Chinook (brights)

Table 7: Lewis River late fall chinook (brights). Abundance time series 1980-1985 from BRT data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure are from BRT data through 2000. Hatchery fraction and wild fraction for 2001-2003 is the average for 1998-2000. Harvest rate for 1980-1981 is the harvest rate for 1982, and for 1999-2003 is the average for 1996-1998. Age structure for 2001-2003 is the average for 1980-1990.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	Regime
1980	14918	0.915	22337	0	0.08	0.135	0.385	0.401	1
1981	21275	0.915	31855	0	0.08	0.135	0.385	0.401	1
1982	9206	0.915	13784	0	0.08	0.135	0.385	0.401	1
1983	14755	0.915	9088	0	0.08	0.135	0.385	0.401	1
1984	8078	0.915	6150	0	0.08	0.135	0.385	0.401	1
1985	9474	0.915	11785	0	0.08	0.135	0.385	0.401	1
1986	11983	0.915	16587	0	0.08	0.135	0.385	0.401	1
1987	12935	0.915	21977	0	0.08	0.135	0.385	0.401	1
1988	12052	0.915	18507	0	0.08	0.135	0.385	0.401	1
1989	12199	0.915	11856	0	0.08	0.135	0.385	0.401	1
1990	17506	0.915	8969	0	0.08	0.135	0.385	0.401	1
1991	9066	0.971	8286	0	0.061	0.134	0.291	0.514	1
1992	6307	0.899	8187	0	0.231	0.062	0.398	0.31	1
1993	7025	0.922	4480	0	0.09	0.283	0.109	0.519	1
1994	9936	0.87	2527	0	0.154	0.135	0.641	0.07	1
1995	9715	1	11408	0	0.031	0.085	0.247	0.637	1
1996	14166	0.911	3095	0	0.005	0.082	0.534	0.379	1
1997	8670	0.942	3973	0	0.007	0.026	0.459	0.507	1
1998	5935	0.876	3045	0	0.043	0.067	0.18	0.709	1
1999	3184	0.767	1559	0	0.069	0.299	0.392	0.24	1
2000	9820	0.895	3851	0	0.106	0.174	0.596	0.125	1
2001	15000	0.846	6369	0	0.08	0.135	0.385	0.401	1
2002	17954	0.836	7753	0	0.08	0.135	0.385	0.401	1
2003	21049	0.859	8745	0	0.08	0.135	0.385	0.401	1

Mill Creek Fall Chinook

Table 8: Mill Creek fall chinook. Abundance time series 1980-1985 from BRT data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. Hatchery fraction for 2001-2003 is the average for 1998-2000. Harvest rate for 1999-2003 is the average of corresponding years for Coweeman fall chinook, adjusted for Mill Creek fall chinook wild catch rates for 2001-2003. Age structure for 2001-2003 is averaged from 1991-2000.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	Regime
1980	516	0.494	-99	0	0.077	0.399	0.434	0.09	1
1982	1367	0.483	-99	0	0.077	0.399	0.434	0.09	1
1982	2750	0.5	3049	0	0.077	0.399	0.434	0.09	1
1983	3725	0.511	2465	0	0.077	0.399	0.434	0.09	1
1984	614	0.519	1449	0	0.077	0.399	0.434	0.09	1
1985	1815	0.526	1675	0	0.077	0.399	0.434	0.09	1
1986	979	0.486	4622	0	0.077	0.399	0.434	0.09	1
1987	6168	0.586	4400	0	0.077	0.399	0.434	0.09	1
1988	3133	0.689	4983	0	0.077	0.399	0.434	0.09	1
1989	2792	0.692	3131	0	0.077	0.399	0.434	0.09	1
1990	620	0.632	1380	0	0.077	0.399	0.434	0.09	1
1991	2017	0.851	914	0	0.075	0.743	0.176	0.006	1
1992	839	0.473	1286	0	0.091	0.768	0.088	0.053	1
1993	885	0.711	2128	0	0.072	0.291	0.623	0.014	1
1994	3854	0.402	168	0	0.017	0.205	0.636	0.142	1
1995	1395	0.512	188	0	0.028	0.444	0.301	0.227	1
1996	593	0.543	116	0	0.307	0.224	0.457	0.012	1
1997	603	0.227	111	0	0.007	0.328	0.496	0.168	1
1998	368	0.598	119	0	0.059	0.086	0.659	0.195	1
1999	575	0.694	237	0	0	0.612	0.343	0.045	1
2000	409	0.584	169	0	0.111	0.288	0.56	0.041	1
2001	4024	0.625333	1662	0	0.05628	0.40235	0.441195	0.100174	1
2002	2481	0.625333	1024	0	0.05628	0.40235	0.441195	0.100174	1
2003	3810	0.625333	1573	0	0.05628	0.40235	0.441195	0.100174	1

Washougal Fall Chinook

Table 9: Washougal fall chinook. Abundance time series 1964-1985 from BRT data, 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2000. For 2001-2003, hatchery fraction is the average for 1998-2000 and age structure is the average for 1991-2000. Harvest rate for 1999-2003 is the average of the corresponding years for Coweeman fall chinook.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	Regime
1964	230	-99	-99	-99	-99	-99	-99	-99	1
1965	206	-99	-99	-99	-99	-99	-99	-99	1
1966	290	-99	-99	-99	-99	-99	-99	-99	1
1967	170	-99	-99	-99	-99	-99	-99	-99	1
1968	153	-99	-99	-99	-99	-99	-99	-99	1
1969	70	-99	-99	-99	-99	-99	-99	-99	1
1970	85	-99	-99	-99	-99	-99	-99	-99	1
1971	1700	-99	-99	-99	-99	-99	-99	-99	1
1972	1300	-99	-99	-99	-99	-99	-99	-99	1
1973	203	-99	-99	-99	-99	-99	-99	-99	1
1974	2977	-99	-99	-99	-99	-99	-99	-99	1
1975	982	-99	-99	-99	-99	-99	-99	-99	1
1976	3037	-99	-99	-99	-99	-99	-99	-99	1
1977	1652	-99	-99	-99	-99	-99	-99	-99	1
1978	593	-99	-99	-99	-99	-99	-99	-99	1
1979	2388	-99	-99	-99	-99	-99	-99	-99	1
1980	3437	0.455	-99	0	0.097	0.248	0.552	0.103	1
1981	1841	0.455	-99	0	0.097	0.248	0.552	0.103	1
1982	330	0.455	1772	0	0.097	0.248	0.552	0.103	1
1983	2677	0.455	1502	0	0.097	0.248	0.552	0.103	1
1984	1217	0.455	1061	0	0.097	0.248	0.552	0.103	1
1985	1983	0.455	1528	0	0.097	0.248	0.552	0.103	1
1986	1589	0.455	3321	0	0.097	0.248	0.552	0.103	1
1987	3625	0.455	2928	0	0.097	0.248	0.552	0.103	1
1988	3328	0.455	3449	0	0.097	0.248	0.552	0.103	1
1989	4578	0.455	2786	0	0.097	0.248	0.552	0.103	1
1990	2205	0.455	1669	0	0.097	0.248	0.552	0.103	1
1991	3673	0.472	1776	0	0.103	0.445	0.452	0	1
1992	2399	0.762	2361	0	0.129	0.051	0.82	0	1
1993	3924	0.516	3537	0	0.043	0	0.625	0.331	1
1994	3888	0.704	341	0	0.096	0.059	0.845	0	1
1995	3063	0.393	387	0	0.078	0.548	0	0.374	1
1996	2921	0.166	215	0	0.206	0	0.513	0.28	1
1997	4669	0.116	249	0	0.258	0.087	0.655	0	1
1998	2971	0.239	190	0	0	1	0	0	1
1999	3129	0.683	2026	0	0.011	0.228	0.715	0.046	1
2000	2155	0.701	1395	0	0.044	0.059	0.897	0	1
2001	3901	0.541	2526	0	0.097292	0.225279	0.517067	0.097348	1
2002	6050	0.541	3918	0	0.097292	0.225279	0.517067	0.097348	1
2003	3444	0.541	2230	0	0.097292	0.225279	0.517067	0.097348	1

Wind Fall Chinook

Table 10: Wind fall chinook. Abundance time series 1964-1985 from BRT data; 1986-2003 from SASSI. Harvest rate and age structure from BRT through 2000 and hatchery fraction through 1979. For 2001-2003, age structure is the average of 1991-2001. Hatchery fraction for 1980-2003 is the average of the corresponding years for Coweeman fall chinook catch data. Harvest rate for 2001-2003 is the average for 1998-2000.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	Regime
1964	783	-99	-99	-99	-99	-99	-99	-99	1
1965	105	-99	-99	-99	-99	-99	-99	-99	1
1966	964	-99	-99	-99	-99	-99	-99	-99	1
1967	274	-99	-99	-99	-99	-99	-99	-99	1
1968	267	-99	-99	-99	-99	-99	-99	-99	1
1969	29	-99	-99	-99	-99	-99	-99	-99	1
1970	54	-99	-99	-99	-99	-99	-99	-99	1
1971	1845	-99	-99	-99	-99	-99	-99	-99	1
1972	1235	-99	-99	-99	-99	-99	-99	-99	1
1973	487	-99	-99	-99	-99	-99	-99	-99	1
1974	610	-99	-99	-99	-99	-99	-99	-99	1
1975	574	-99	-99	-99	-99	-99	-99	-99	1
1976	646	-99	-99	-99	-99	-99	-99	-99	1
1977	971	-99	-99	-99	-99	-99	-99	-99	1
1978	1527	-99	-99	-99	-99	-99	-99	-99	1
1979	946	-99	-99	-99	-99	-99	-99	-99	1
1980	401	1	1042	0	0.033	0.467	0.458	0.042	1
1981	256	1	783	0	0.033	0.467	0.458	0.042	1
1982	365	1	761	0	0.033	0.467	0.458	0.042	1
1983	495	1	747	0	0.033	0.467	0.458	0.042	1
1984	134	1	288	0	0.033	0.467	0.458	0.042	1
1985	170	1	172	0	0.033	0.467	0.458	0.042	1
1986	422	1	781	0	0.033	0.467	0.458	0.042	1
1987	776	1	1903	0	0.033	0.467	0.458	0.042	1
1988	1206	1	3483	0	0.033	0.467	0.458	0.042	1
1989	112	1	186	0	0.033	0.467	0.458	0.042	1
1990	11	1	15	0	0.033	0.467	0.458	0.042	1
1991	58	1	72	0	0.103	0.759	0.138	0	1
1992	54	1	40	0	0	0.5	0.5	0	1
1993	0	-99	0	-99	-99	-99	-99	-99	1
1994	11	1	5	0	0	0.727	0.273	0	1
1995	4	1	1	0	0	0.75	0.25	0	1
1996	166	1	55	0	0	0.729	0.271	0	1
1997	148	1	82	0	0	0.264	0.669	0.068	1
1998	213	1	256	0	0.052	0.188	0.667	0.094	1
1999	126	0.33	250	0	0	0	1	0	1
2000	14	1	24	0	0.143	0.286	0.357	0.214	1
2001	444	0.777	400	0	0.024	0.36	0.574	0.042	1
2002	375	0.777	774	0	0.024	0.36	0.574	0.042	1
2003	1574	0.777	7394	0	0.024	0.36	0.574	0.042	1

Cowlitz Spring Chinook

Table 11: Cowlitz spring chinook. Abundance time series 1980-1985 from BRT data, and 1986-2003 from SASSI. Hatchery fraction and age structure from BRT through 2002. For 2003, age structure, hatchery fraction, and catch rate are the average of 2000-2001. Harvest rates are missing from the original dataset and have been set to 0.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	Regime
1980	197	1	0	0	0	0.32	0.35	0.33	1
1981	1116	1	0	0	0	0.32	0.35	0.33	1
1982	279	1	0	0	0	0.32	0.35	0.33	1
1983	95	1	0	0	0	0.32	0.35	0.33	1
1984	161	1	0	0	0	0.32	0.35	0.33	1
1985	261	1	0	0	0	0.32	0.35	0.33	1
1986	959	1	0	0	0	0.32	0.35	0.33	1
1987	90	1	0	0	0	0.32	0.35	0.33	1
1988	221	1	0	0	0	0.32	0.35	0.33	1
1989	684	1	0	0	0	0.32	0.35	0.33	1
1990	320	1	0	0	0	0.32	0.35	0.33	1
1991	284	1	0	0	0	0.32	0.35	0.33	1
1992	279	1	0	0	0	0.32	0.35	0.33	1
1993	236	1	0	0	0	0.32	0.35	0.33	1
1994	167	1	0	0	0	0.32	0.35	0.33	1
1995	347	1	0	0	0	0.32	0.35	0.33	1
1996	36	1	0	0	0	0.32	0.35	0.33	1
1997	455	1	0	0	0	0.32	0.35	0.33	1
1998	356	1	0	0	0	0.32	0.35	0.33	1
1999	285	1	0	0	0	0.32	0.35	0.33	1
2000	266	1	0	0	0	0.32	0.35	0.33	1
2001	347	1	0	0	0	0.32	0.35	0.33	1
2002	419	1	0	0	0	0.32	0.35	0.33	1
2003	1937	1	0	0	0	0.32	0.35	0.33	1

Kalama Spring Chinook

Table 12: Kalama spring chinook. Abundance time series 1980-1985 from BRT data, and 1986-2003 from SASSI. Hatchery fraction, harvest rates, and age structure are missing for the entire data set, and have been replaced with values for Cowlitz spring chinook.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	Regime
1980	340	1	0	0	0	0.32	0.35	0.33	1
1981	848	1	0	0	0	0.32	0.35	0.33	1
1982	2892	1	0	0	0	0.32	0.35	0.33	1
1983	1150	1	0	0	0	0.32	0.35	0.33	1
1984	134	1	0	0	0	0.32	0.35	0.33	1
1985	0	1	0	0	0	0.32	0.35	0.33	1
1986	181	1	0	0	0	0.32	0.35	0.33	1
1987	527	1	0	0	0	0.32	0.35	0.33	1
1988	496	1	0	0	0	0.32	0.35	0.33	1
1989	584	1	0	0	0	0.32	0.35	0.33	1
1990	34	1	0	0	0	0.32	0.35	0.33	1
1991	34	1	0	0	0	0.32	0.35	0.33	1
1992	198	1	0	0	0	0.32	0.35	0.33	1
1993	348	1	0	0	0	0.32	0.35	0.33	1
1994	408	1	0	0	0	0.32	0.35	0.33	1
1995	392	1	0	0	0	0.32	0.35	0.33	1
1996	272	1	0	0	0	0.32	0.35	0.33	1
1997	45	1	0	0	0	0.32	0.35	0.33	1
1998	46	1	0	0	0	0.32	0.35	0.33	1
1999	244	1	0	0	0	0.32	0.35	0.33	1
2000	34	1	0	0	0	0.32	0.35	0.33	1
2001	578	1	0	0	0	0.32	0.35	0.33	1
2002	898	1	0	0	0	0.32	0.35	0.33	1
2003	766	1	0	0	0	0.32	0.35	0.33	1

Lewis Spring Chinook

Table 13: Lewis spring chinook. Abundance time series 1980-1985 from BRT data, and 1986-2003 from SASSI. Hatchery fraction, harvest rates and age structure are missing for the entire data set, and have been replaced with values for Cowlitz spring chinook.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	Regime
1980	1002	1	0	0	0	0.32	0.35	0.33	1
1981	345	1	0	0	0	0.32	0.35	0.33	1
1982	1081	1	0	0	0	0.32	0.35	0.33	1
1983	801	1	0	0	0	0.32	0.35	0.33	1
1984	1653	1	0	0	0	0.32	0.35	0.33	1
1985	530	1	0	0	0	0.32	0.35	0.33	1
1986	1875	1	0	0	0	0.32	0.35	0.33	1
1987	6850	1	0	0	0	0.32	0.35	0.33	1
1988	5267	1	0	0	0	0.32	0.35	0.33	1
1989	3594	1	0	0	0	0.32	0.35	0.33	1
1990	1419	1	0	0	0	0.32	0.35	0.33	1
1991	1632	1	0	0	0	0.32	0.35	0.33	1
1992	1328	0.973	0	0	0	0.32	0.35	0.33	1
1993	1518	0.941	0	0	0	0.32	0.35	0.33	1
1994	478	0.99	0	0	0	0.32	0.35	0.33	1
1995	279	0.996	0	0	0	0.32	0.35	0.33	1
1996	504	1	0	0	0	0.32	0.35	0.33	1
1997	417	1	0	0	0	0.32	0.35	0.33	1
1998	213	1	0	0	0	0.32	0.35	0.33	1
1999	270	1	0	0	0	0.32	0.35	0.33	1
2000	475	1	0	0	0	0.32	0.35	0.33	1
2001	669	1	0	0	0	0.32	0.35	0.33	1
2002	487	1	0	0	0	0.32	0.35	0.33	1
2003	679	1	0	0	0	0.32	0.35	0.33	1

LCR Steelhead

East Fork Lewis Summer Steelhead

Table 14: East Fork Lewis summer steelhead. Abundance time series 1996-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	age7	Regime
1996	197	0.51	6	0	0	0.11	0.619	0.197	0.074	0	1
1997	141	0.48	6	0	0	0.087	0.62	0.193	0.087	0.013	1
1998	139	0.58	4	0	0.006	0.146	0.605	0.177	0.055	0.011	1
1999	229	0.6	4	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2000	271	0.8	7	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2001	440	0.7	8	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2002	910	0.84	13	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2003	425	0.8	27	0	0.006	0.146	0.605	0.177	0.055	0.011	1

Kalama Summer Steelhead

Table 15: Kalama summer steelhead. Abundance time series 1977-1985 from BRT data, and 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	age7	Regime
1977	1469	0.273	633	0	0.011	0.149	0.557	0.138	0.136	0.01	1
1978	4554	0.223	1079	0	0.009	0.272	0.592	0.08	0.044	0.002	1
1979	2604	0.186	832	0	0.026	0.238	0.539	0.124	0.045	0.027	1
1980	2647	0.271	844	0	0.017	0.256	0.561	0.109	0.049	0.008	1
1981	11524	0.254	2978	0	0	0.169	0.571	0.222	0.034	0.004	1
1982	13686	0.101	1075	0	0.003	0.147	0.61	0.211	0.014	0.015	1
1983	5274	0.165	1621	0	0	0.09	0.682	0.196	0.021	0.011	1
1984	1155	0.214	738	0	0.009	0.199	0.545	0.191	0.037	0.019	1
1985	1567	0.294	854	0	0.008	0.171	0.677	0.09	0.054	0	1
1986	473	0.163	799	0	0	0.186	0.563	0.186	0.043	0.022	1
1987	748	0.138	148	0	0	0.111	0.624	0.142	0.099	0.025	1
1988	950	0.302	217	0	0.005	0.111	0.682	0.168	0.03	0.005	1
1989	684	0.203	90	0	0.022	0.148	0.584	0.24	0.006	0	1
1990	745	0.446	74	0	0	0.163	0.569	0.226	0.042	0	1
1991	704	0.405	16	0	0	0.063	0.695	0.147	0.084	0.011	1
1992	1075	0.404	5	0	0.005	0.163	0.589	0.203	0.03	0.01	1
1993	2283	0.318	204	0	0	0.046	0.698	0.175	0.074	0.008	1
1994	1041	0.271	72	0	0	0.099	0.511	0.302	0.073	0.015	1
1995	1302	0.428	9	0	0	0.082	0.624	0.175	0.087	0.033	1
1996	614	0.348	15	0	0	0.11	0.619	0.197	0.074	0	1
1997	598	0.2	38	0	0	0.087	0.62	0.193	0.087	0.013	1
1998	205	0.27	2	0	0.006	0.146	0.605	0.177	0.055	0.011	1
1999	220	0.541	70	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2000	140	0.824	107	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2001	286	0.846	81	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2002	454	0.712	89	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2003	817	0.5	145	0	0.006	0.146	0.605	0.177	0.055	0.011	1

Washougal Summer Steelhead

Table 16: Washougal summer steelhead. Abundance time series 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	age7	Regime
1977	1469	0.273	633	0	0.011	0.149	0.557	0.138	0.136	0.01	1
1978	4554	0.223	1079	0	0.009	0.272	0.592	0.08	0.044	0.002	1
1979	2604	0.186	832	0	0.026	0.238	0.539	0.124	0.045	0.027	1
1980	2647	0.271	844	0	0.017	0.256	0.561	0.109	0.049	0.008	1
1981	11524	0.254	2978	0	0	0.169	0.571	0.222	0.034	0.004	1
1982	13686	0.101	1075	0	0.003	0.147	0.61	0.211	0.014	0.015	1
1983	5274	0.165	1621	0	0	0.09	0.682	0.196	0.021	0.011	1
1984	1155	0.214	738	0	0.009	0.199	0.545	0.191	0.037	0.019	1
1985	1567	0.294	854	0	0.008	0.171	0.677	0.09	0.054	0	1
1986	473	0.163	799	0	0	0.186	0.563	0.186	0.043	0.022	1
1987	748	0.138	148	0	0	0.111	0.624	0.142	0.099	0.025	1
1988	950	0.302	217	0	0.005	0.111	0.682	0.168	0.03	0.005	1
1989	684	0.203	90	0	0.022	0.148	0.584	0.24	0.006	0	1
1990	745	0.446	74	0	0	0.163	0.569	0.226	0.042	0	1
1991	704	0.405	16	0	0	0.063	0.695	0.147	0.084	0.011	1
1992	1075	0.404	5	0	0.005	0.163	0.589	0.203	0.03	0.01	1
1993	2283	0.318	204	0	0	0.046	0.698	0.175	0.074	0.008	1
1994	1041	0.271	72	0	0	0.099	0.511	0.302	0.073	0.015	1
1995	1302	0.428	9	0	0	0.082	0.624	0.175	0.087	0.033	1
1996	614	0.348	15	0	0	0.11	0.619	0.197	0.074	0	1
1997	598	0.2	38	0	0	0.087	0.62	0.193	0.087	0.013	1
1998	205	0.27	2	0	0.006	0.146	0.605	0.177	0.055	0.011	1
1999	220	0.541	70	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2000	140	0.824	107	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2001	286	0.846	81	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2002	454	0.712	89	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2003	817	0.5	145	0	0.006	0.146	0.605	0.177	0.055	0.011	1

Wind Summer Steelhead

Table 17: Wind summer steelhead. Abundance time series 1989-2004 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT data through 2003. Hatchery fraction for 2004 is the average for 2001-2003, harvest rate the average for 2001-2003, and age structure the average for 1998-2003.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	age7	Regime
1989	1016	0.66	212	0	0.022	0.148	0.584	0.24	0.006	0	1
1990	561	0.82	103	0	0	0.163	0.569	0.226	0.042	0	1
1991	596	0.74	74	0	0	0.063	0.695	0.147	0.084	0.011	1
1992	535	0.65	96	0	0.005	0.163	0.589	0.203	0.03	0.01	1
1993	677	0.94	107	0	0	0.046	0.698	0.175	0.074	0.008	1
1994	468	0.76	58	0	0	0.099	0.511	0.302	0.073	0.015	1
1995	543	0.76	54	0	0	0.082	0.624	0.175	0.087	0.033	1
1996	466	0.9	49	0	0	0.11	0.619	0.197	0.074	0	1
1997	734	0.81	74	0	0	0.087	0.62	0.193	0.087	0.013	1
1998	320	0.84	23	0	0.006	0.146	0.605	0.177	0.055	0.011	1
1999	323	0.84	22	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2000	193	0.96	16	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2001	416	0.98	32	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2002	669	0.99	41	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2003	1067	0.99	59	0	0.006	0.146	0.605	0.177	0.055	0.011	1
2004	816	0.99	52	0	0.006	0.146	0.605	0.177	0.055	0.011	1

Coweeman Winter Steelhead

Table 18: Coweeman winter steelhead. Abundance time series 1987-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT through 2002. For 1996-97 and 2003, age structure, hatchery fraction, and catch rates are averaged from the previous three years (1993-1995 and 2000-2002, respectively).

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	age7	Regime
1987	889	0.5	178	0	0.006	0.076	0.414	0.443	0.061	0	1
1988	1088	0.5	218	0	0.004	0.02	0.561	0.386	0.029	0	1
1989	392	0.5	78	0	0.005	0.093	0.588	0.286	0.028	0	1
1990	522	0.5	104	0	0	0.005	0.464	0.475	0.055	0	1
1991	-99	-99	-99	-99	-99	-99	-99	-99	-99	-99	1
1992	-99	-99	-99	-99	-99	-99	-99	-99	-99	-99	1
1993	438	0.5	9	0	0	0.049	0.324	0.551	0.074	0.002	1
1994	362	0.5	7	0	0	0.037	0.723	0.202	0.038	0.001	1
1995	68	0.5	5	0	0	0.027	0.562	0.375	0.035	0.001	1
1996	44	0.5	1	0	0	0.0423	0.5088	0.3915	0.0559	0.0015	1
1997	108	0.5	2	0	0	0.0423	0.5088	0.3915	0.0559	0.0015	1
1998	486	0.5	6	0	0.002	0.068	0.513	0.37	0.044	0.003	1
1999	198	0.5	3	0	0.002	0.068	0.513	0.37	0.044	0.003	1
2000	530	0.5	6	0	0.002	0.068	0.513	0.37	0.044	0.003	1
2001	384	0.5	6	0	0.002	0.068	0.513	0.37	0.044	0.003	1
2002	298	0.5	4	0	0.002	0.068	0.513	0.37	0.044	0.003	1
2003	460	0.5	3	0	0.002	0.068	0.513	0.37	0.044	0.003	1

East Fork Lewis Winter Steelhead

Table 19: East Fork Lewis winter steelhead. Abundance time series 1985-1996 from BRT data, and 1997-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT except for 1985-2003, for which harvest rates for Kalama winter steelhead are used and adjusted for the East Fork Lewis winter steelhead proportion of wild catch. No data are available for 1995-1996, so these years have been excluded from the analysis.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	Regime
1985	282	1	393	0	0	0.026	0.527	0.399	0.048	1
1986	192	1	121	0	0	0.026	0.527	0.399	0.048	1
1987	258	1	52	0	0	0.026	0.527	0.399	0.048	1
1988	140	1	57	0	0	0.026	0.527	0.399	0.048	1
1989	102	1	105	0	0	0.026	0.527	0.399	0.048	1
1990	72	1	77	0	0	0.026	0.527	0.399	0.048	1
1991	88	1	2	0	0	0.026	0.527	0.399	0.048	1
1992	90	1	1	0	0	0.026	0.527	0.399	0.048	1
1993	78	1	6	0	0	0.026	0.527	0.399	0.048	1
1994	53	1	1	0	0	0.026	0.527	0.399	0.048	1
1995	-99	-99	-99	-99	-99	-99	-99	-99	-99	1
1996	-99	-99	-99	-99	-99	-99	-99	-99	-99	1
1997	192	1	12	0	0	0.026	0.527	0.399	0.048	1
1998	420	1	23	0	0	0.026	0.527	0.399	0.048	1
1999	476	1	40	0	0	0.026	0.527	0.399	0.048	1
2000	-99	1	11	0	0	0.026	0.527	0.399	0.048	1
2001	328	1	16	0	0	0.026	0.527	0.399	0.048	1
2002	316	1	12	0	0	0.026	0.527	0.399	0.048	1
2003	624	1	40	0	0	0.026	0.527	0.399	0.048	1

Kalama Winter Steelhead

Table 20: Kalama winter steelhead. Abundance time series 1977-1985 from BRT data, and 1986-2003 from SASSI. Hatchery fraction, harvest rate, and age structure from BRT through 2002. For 2003, hatchery fraction, catch rate, and age structure are the averages for 2000-2002.

Year	Spawners	Frac Wild	Catch	age1	age2	age3	age4	age5	age6	Regime
1977	946	0.818	1229	0	0.004	0.176	0.441	0.236	0.108	0.035
1978	1615	0.43	1114	0	0.003	0.118	0.483	0.358	0.034	0.005
1979	521	0.713	647	0	0.003	0.056	0.524	0.367	0.051	0
1980	1347	0.761	1067	0	0.001	0.063	0.644	0.264	0.027	0.001
1981	2770	0.776	2162	0	0	0.073	0.44	0.424	0.059	0.005
1982	1109	0.784	1719	0	0	0.056	0.427	0.466	0.045	0.006
1983	874	0.609	1020	0	0	0.062	0.327	0.553	0.058	0
1984	2007	0.47	959	0	0.007	0.134	0.564	0.244	0.051	0
1985	1066	0.592	1487	0	0.008	0.121	0.453	0.41	0.008	0
1986	1021	0.363	643	0	0	0.113	0.534	0.299	0.049	0.006
1987	1091	0.547	218	0	0.006	0.076	0.414	0.443	0.061	0
1988	1199	0.505	486	0	0.004	0.02	0.561	0.386	0.029	0
1989	556	0.65	571	0	0.005	0.093	0.588	0.286	0.028	0
1990	396	0.471	424	0	0	0.005	0.464	0.475	0.055	0
1991	1065	0.744	26	0	0	0.04	0.427	0.485	0.047	0
1992	2193	0.693	15	0	0	0.025	0.652	0.285	0.038	0
1993	937	0.73	75	0	0	0.049	0.324	0.551	0.074	0.002
1994	806	0.792	13	0	0	0.037	0.723	0.202	0.038	0.001
1995	1144	0.783	53	0	0	0.027	0.562	0.375	0.035	0.001
1996	806	0.452	48	0	0	0.027	0.622	0.327	0.02	0.004
1997	507	0.9	33	0	0	0.047	0.602	0.333	0.018	0
1998	472	1	28	0	0.002	0.068	0.513	0.37	0.044	0.003
1999	544	1	46	0	0.002	0.068	0.513	0.37	0.044	0.003
2000	921	1	99	0	0.002	0.068	0.513	0.37	0.044	0.003
2001	1042	1	51	0	0.002	0.068	0.513	0.37	0.044	0.003
2002	1495	1	59	0	0.002	0.068	0.513	0.37	0.044	0.003
2003	1815	1	117	0	0.002	0.068	0.513	0.37	0.044	0.003

North Fork Toutle Winter Steelhead

Table 21: North Fork Toutle winter steelhead. Abundance time series 1989-2003 from SASSI data. Hatchery fraction, harvest rate, and age structure from BRT through 2002. For 2003, hatchery fraction, harvest rate, and age structure are the averages for 2000-2002.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	age6	age7	Regime
1989	18	1	0	0	0.018	0	0.596	0.351	0.035	0	1
1990	36	1	0	0	0	0.222	0.444	0.333	0	0	1
1991	108	1	1	0	0	0	0.739	0.261	0	0	1
1992	322	1	3	0	0	0.266	0.557	0.177	0	0	1
1993	165	1	2	0	0	0.047	0.647	0.273	0.033	0	1
1994	90	1	1	0	0	0.048	0.238	0.571	0.131	0.012	1
1995	175	1	2	0	0	0.163	0.612	0.224	0	0	1
1996	251	1	3	0	0	0.164	0.673	0.164	0	0	1
1997	183	1	2	0	0	0.044	0.681	0.212	0.062	0	1
1998	149	1	1	0	0	0.034	0.68	0.258	0.028	0	1
1999	133	1	1	0	0	0.008	0.672	0.297	0.023	0	1
2000	238	1	2	0	0.002	0.09	0.595	0.284	0.028	0.001	1
2001	185	1	2	0	0.002	0.09	0.595	0.284	0.028	0.001	1
2002	328	1	3	0	0.002	0.09	0.595	0.284	0.028	0.001	1
2003	410	1	4	0	0.002	0.09	0.595	0.284	0.028	0.001	1

Washougal Winter Steelhead

Table 22: Washougal winter steelhead. Abundance time series 1991-2003 from SASSI except for 1996 and 2000 for which escapement data are not available and no analysis possible. Hatchery fraction, harvest rate, and age structure from BRT data except for the years 1995-2003, for which harvest rate and age structure re missing.

Year	Spawners	FracWild	Catch	age1	age2	age3	age4	age5	age6	Regime
1991	114	1	-99	0	0	0.026	0.527	0.399	0.048	1
1992	142	1	-99	0	0	0.026	0.527	0.399	0.048	1
1993	118	1	-99	0	0	0.026	0.527	0.399	0.048	1
1994	158	1	-99	0	0	0.026	0.527	0.399	0.048	1
1995	206	1	-99	0	0	0.026	0.527	0.399	0.048	1
1996	-99	-99	-99	-99	-99	-99	-99	-99	-99	1
1997	92	-99	-99	-99	-99	-99	-99	-99	-99	1
1998	195	-99	-99	-99	-99	-99	-99	-99	-99	1
1999	294	-99	-99	-99	-99	-99	-99	-99	-99	1
2000	-99	-99	-99	-99	-99	-99	-99	-99	-99	1
2001	216	-99	-99	-99	-99	-99	-99	-99	-99	1
2002	286	-99	-99	-99	-99	-99	-99	-99	-99	1
2003	764	-99	-99	-99	-99	-99	-99	-99	-99	1

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Appendix D: Viability Curve Analysis

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Background

This appendix describes modifications in the viability curve analysis methods of the TRT viability report that were made in conducting the Oregon WLC population status evaluations, based on the inclusion of new data from Washington populations and from the refinement of thinking on some topics. For a complete description of the viability curve analysis methodology, see the TRT viability report (McElhany et al. 2006).

Variance

Accurate estimates of recruitment variability (variance) are difficult to obtain, especially at the population level. As one solution to this problem we have calculated species level variance estimates by averaging all of the individual population variance estimates for each species. In addition to the variance estimate, we have also estimated temporal autocorrelation in the same manner (i.e., an average of values obtained for all populations of each species). Autocorrelation is the tendency for annual differences between observed and model predicted recruitment to be somewhat correlated from one year to the next. (This tendency for streaks of “good years” and “bad years” might be caused, for example, by shifts in marine productivity.)

In the draft viability report, we relied on averaging information from only Oregon WLC populations. We have now included Washington LCR populations in the average and the viability curves used in this report are based on the new variance estimates in Table 1. We have also changed the variance and autocorrelations estimation methods so that they are based on residuals from fitting the MeanRS model rather than directly curve-fitting a hockey-stick function (see viability report). Including data from Washington populations had a much greater effect on average variance than the change to using MeanRS residuals. Finally, we calculated a “generic WLC salmon” variance and autocorrelation based on the average of the species averages. The steelhead variance estimates are substantially below those of the other species and there was concern that this may be an artifact of the relatively short time series. Therefore, we conducted the steelhead viability assessment using both the steelhead specific and the generic salmon variance estimates.

Table 1: Variance and autocorrelation based on MeanRS method. chinook, coho and steelhead estimates are based on average of Oregon and Washington WLC populations.

Species	Variance	Correlation (Lag1)	Correlation (Lag 2)
Chinook	0.863	0.346	0.172
Chum	0.809	0.000	0.000
Coho	1.005	0.292	0.027
Steelhead	0.435	0.518	0.280
Generic WLC Salmon	0.778	0.292	0.114

QET and CRT

The forward projection model is used to develop the viability curve tests for the probability that a population will drop to a Critical Risk Threshold (CRT). The CRT describes an abundance level below which the population will be at highly elevated extinction risk because of processes not considered in the extinct risk model (e.g., demographic stochasticity).

In the viability report and previous analyses, we referred to similar thresholds as 'quasi-extinction thresholds' (QET). However, this term suggested to some that we were modeling a level below which a population would experience certain extinction. This is not the case – the lower threshold (the CRT or QET) is simply a region with greatly increased probability of extinction, but until the population is actually down to having only members of a single gender, recovery is possible. Because of the limitations of extinction risk models, using lower thresholds rather than zero fish is a common practice in conservation biology, but setting the actual value is always challenging.

In this analysis, the CRT is a function of the watershed size and we have partitioned Oregon WLC populations into small, medium, and large size categories. In Table 2 we reproduce the summary CRT table from the viability report.

Table 2: Thresholds for Oregon WLC populations copied from the TRT viability report (McElhany et al. 2006). The fish per spawning km associated with the threshold is shown in parentheses rounded to nearest km. The stream km combines the "Spawning and rearing" and "Previous/Historical" categories from the ODFW fish distribution data summarized in the WLC habitat atlas (Maher et al. 2005). This may represent an overestimate of the historical spawning habitat because it is likely that not all stream km categorized as "Previous/Historical" was spawning habitat (i.e., some may be "Migratory and rearing" habitat). Stream km for some chum populations is not available (N/A). (McElhany et al. 2006a) *Note: The CRT column is labeled "QET" in the TRT viability report.

ESU	Life History	Population	Stream (Km)	Size Category	CRT*
Lower Columbia Chinook	Fall	Big Creek	16	Small	50 (3)
		Clackamas River	61	Medium	150 (2)
		Clatskanie River	16	Small	50 (3)
		Lower Gorge Tributaries	10	Small	50 (5)
		Upper Gorge Tributaries	2	Small	50 (25)
		Hood River	39	Small	50 (1)
		Sandy River	75	Medium	150 (2)
		Scappoose Creek	7	Small	50 (7)
		Youngs Bay Tributaries	35	Small	50 (1)
	Spring	Hood River	75	Medium	150 (2)
		Sandy River	125	Medium	150 (1)
Lower Columbia Chum		Big Creek	71	Medium	200 (3)
		Clackamas River	N/A	N/A	N/A
		Clatskanie River	4	Small	100 (25)
		Lower Gorge Tributaries	N/A	N/A	N/A

		Upper Gorge Tributaries	N/A	N/A	N/A
		Hood River	N/A	N/A	N/A
		Sandy River	N/A	N/A	N/A
		Scappoose Creek	N/A	N/A	N/A
		Youngs Bay Tributaries	91	Medium	200 (2)
Lower Columbia Coho		Big Creek	78	Small	100 (1)
		Clackamas River	465	Large	300 (1)
		Clatskanie River	105	Medium	200 (2)
		Lower Gorge Tributaries	14	Small	100 (7)
		Sandy River	247	Large	300 (1)
		Scappoose Creek	125	Medium	200 (2)
		Youngs Bay Tributaries	94	Small	100 (1)
		Hood River	119	Medium	200 (2)
Lower Columbia Steelhead	Summer	Hood River	131	Medium	100 (1)
	Winter	Clackamas River	492	Large	200 (0)
		Lower Gorge Tributaries	14	Small	50 (4)
		Upper Gorge Tributaries	12	Small	50 (4)
		Hood River	154	Medium	100 (1)
		Sandy River	348	Large	200 (1)
Upper Willamette Chinook	Spring	Calapooia River	59	Medium	150 (3)
		Clackamas River	182	Large	250 (1)
		McKenzie River	244	Large	250 (1)
		Molalla River	104	Medium	150 (1)
		North Santiam River	129	Medium	150 (1)
		South Santiam River	190	Large	250 (1)
		Middle Fork Willamette R.	272	Large	250 (1)
Upper Willamette Steelhead	Winter	Calapooia River	91	Small	50 (1)
		Molalla River	240	Large	200 (1)
		North Santiam River	198	Medium	100 (1)
		South Santiam River	323	Large	200 (1)

Harvest Rate and Measurement Error Assumptions

For the pre-harvest viability curves, we must also make assumptions about future harvest. We assumed that future harvests would be similar to that observed in recent years. Harvest rate assumptions are copied from the viability report and shown in Table 3. The viability curve analysis also requires assumptions about the measurement error of input parameters, which are copied from the viability report and shown in Table 4. The values in these tables did not change from the TRT viability report, but because these parameters are important for the viability curve analysis, they are repeated here.

Table 3: Future harvest rate assumptions for Oregon WLC populations based on approximations of current harvest rates (McElhany et al. 2006a).

ESU	Harvest Rate
LCR Fall Chinook	50%
LCR Spring Chinook	25%
CR Chum	5%
LCR Coho	25%
LCR Steelhead	10%
UW Chinook	25%
UW Steelhead	10%

Table 4: Measurement error assumptions for viability curve analysis input parameters for Oregon WLC populations. Modified from (McElhany et al. 2006a). Age composition is the shape parameter from finite multi-nominal sampling (See Appendix A).

ESU	Life History	Population	Data Collection Method	Spawner Abundance	Hatchery Proportion	Age Composition	Fishery Impact
Chinook	Fall (tule)	Clatskanie	Spawning Surveys	±40%	±70%	20	±40%
	Late Fall (bright)	Sandy	Spawning Surveys	±40%	±20%	20	±40%
	Spring	Sandy River	Spawning Surveys	±40%	±40%	20	±30%
Lower Columbia Coho		Big Creek*	Spawning Surveys	±50%	±40%	500	±50%
		Clackamas	Dam Passage Counts	±20%	±20%	500	±50%
		Clatskanie*	Spawning Surveys	±50%	±40%	500	±50%
		Sandy River	Dam Passage Counts	±20%	±20%	500	±50%
		Scappoose River*	Spawning Surveys	±50%	±40%	500	±50%
		Youngs Bay*	Spawning Surveys	±50%	±40%	500	±50%
Lower Columbia Steelhead	Summer	Hood River*	Dam Passage Counts	±20%	±20%	50	±40%
	Winter	Clackamas	Dam Passage Counts	±20%	±20%	20	±40%
		Hood River*	Dam Passage Counts	±20%	±20%	100	±40%
		Sandy River	Dam Passage Counts	±20%	±20%	20	±40%
Upper Willamette Chinook	Spring	Calapooia*	Spawning Surveys	±40%	±40%	100	±30%
		Clackamas	Dam Passage Counts	±20%	±20%	20	±30%
		McKenzie	Spawning Surveys (partial dam count)	±40%	±40%	20	±30%
		Molalla*	Spawning Surveys	±40%	±40%	20	±30%
Upper Willamette Steelhead	Winter	Calapooia	Spawning Surveys	±70%	±60%	20	±40%
		Molalla	Spawning Surveys	±70%	±60%	20	±40%
		N. Santiam	Spawning Surveys	±70%	±60%	20	±40%
		S. Santiam (Lower)	Spawning Surveys	±70%	±60%	20	±40%
		S. Santiam (Upper)	Trap and Handle	±5%	±5%	20	±40%

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Appendix E: Conservation Assessment and Planning Model (CAPM)

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Background

The following describes CAPM (Conservation Assessment and Planning Model) a population viability model that was developed to assist salmonid conservation and recovery planning in Oregon. The model's primary outputs are forecast probabilities of population extinction. Each forecast is performed under a specific set of assumptions concerning key variables such as reproductive rate, habitat capacity, environmental variability, critical population abundance, proportion of hatchery fish, and fishery related mortality rates. Values for these variables (and others contained within the model structure) can be set to represent current conditions for the population or they can be set to reflect alternate conditions that are expected to occur in response to the implementation of specific recovery strategies. Therefore, modeling results can provide insight into the likelihood of population extinction should conditions remain unchanged in the future and also the likelihood of population extinction should these conditions change in response to implementation of successful recovery strategies.

A wide variety of viability models have been used by conservation biologists to estimate the vulnerability of populations to extinction (Shaffer 1981, 1990; Murphy et al. 1990; Nickelson and Lawson 1998). CAPM represents yet another approach to estimating population viability. It was generally based on methodology described by Burgman et al. (1993) and Morris and Doak (2002). However, CAPM also draws on original methodologies described here for the first time here.

In general, CAPM forecasts the probability of population extinction by simulating wild spawner abundance over a future time period of 100 years. Depending on the average life age of the species, this requires the simulation of 20 to 33 cycles of spawners and subsequent recruits (100 years). CAPM relies on spawner-recruit functions to accomplish this. These functions predict recruits (offspring) from two variables: 1) the number of parents (spawners), and 2) an independent environmental index of cyclic variations in freshwater and marine survival. SNEG, an index of high elevation maximum snow depths, was used as the environmental survival variable. Although, several other survival related indices were considered for this purpose (e.g., PDO, OPI, and PNI), SNEG, when evaluated across all species, appeared to have the greatest power to explain observed variations in population recruitment.

As is characteristic of all population viability models, CAPM attempts to mimic the stochastic behavior of population recruitment as it occurs in nature. Without this stochastic component added to the model, recruitment functions will always yield the same value for recruits produced for each input value for spawner abundance. So for example, from a spawner escapement 500 fish, a specific recruitment function might predict 800 recruits would be produced. More importantly, each time a spawner abundance of 500 was seeded into the recruitment function, the forecast number of subsequent recruits would always be exactly 800. However, real fish populations don't behave this way. For example, the repetition of a 500 fish spawner escapement for say 10 years in a row, would most likely result in 10 different values for the number of recruits produced. These recruit abundance values may average 800 fish, but random and unknown variations in annual survival could easily produce a range in the annual recruit number from 400 to 1200. Therefore, the inclusion of a stochastic component to CAPM, ensured that recruitment functions would produce a range of values from each spawner abundance level rather than the same answer over and over. It was assumed that inclusion of this stochastic element would produce a more accurate model of real populations and their vulnerability to extinction.

Although the stochastic component is not unique among population viability assessment (PVA) models, there are several features of CAPM that are perhaps atypical. The first is the use of an independent environmental variable (SNEG) within the recruitment function. This was done to obtain a more accurate mathematical description of the biological recruitment process observed for each population. Secondly, rather than using only one recruitment model to simulate population recruitment, CAPM uses three. It was assumed that in doing so the adverse consequences of case by case inaccuracies of data fits to a particular recruitment function could be reduced. Thirdly, a probability of extinction was calculated for each set of recruitment function parameters estimated via the bootstrap process (description to follow). Therefore, CAPM results for each run of the model consist of many estimates of probability extinction for each population. The range, median and distribution of these extinction probabilities is used to help gauge model results in terms of uncertainty with respect to how well the shape of the recruitment curves fit observed population data.

Key topics discussed in this summary of CAPM are: 1) the population recruitment function, 2) fitting population recruitment curves, 3) addition of stochastic effects, 4) assumptions about future conditions, 5) program mechanics, and 6) model output.

Population Recruitment Function

As stated earlier, three equations were used to simulate population recruitment. The first of these was based on the Beverton-Holt recruitment model (Beverton-Holt 1957). The second function was the Ski recruitment model. This is a previously undescribed recruitment curve that is similar to the Beverton-Holt (BH) model. However the Ski (SK) curve builds to maximum recruit capacity more rapidly than the BH model for each increment in spawner abundance, from mid-range spawner levels upward (Figure D1). The Crowbar model, also a previously undescribed recruitment curve, was the third function used by CAPM. The Crowbar (CB) function is similar to the SK function however it has a unique feature in that reproductive rates at very low spawner abundance levels decline rather than increase (Figure D2). As a consequence maximum recruits per spawner typically occur at higher spawner abundance levels than for either the BH or SK curves. Essentially the CB curve has built-in depensation, while the BH and SK curves do not. Mathematically these three recruitment functions are described by the following equations:

Beverton-Holt:	$R_t = (a * S_t) / (1 + (a / b * S_t))$	(1)
Ski:	$R_t = b * (1 - \exp(- (a / b * S_t)))$	(2)
Crowbar:	$R_t = b * \exp((- b * \exp(- 1)) / (a * S_t))$	(3)

Where R_t = the total number of adults produced from the spawners of a particular year (t), S_t = number of spawners in year t, a = maximum recruits per spawner, b = and capacity of habitat expressed as maximum possible recruits.

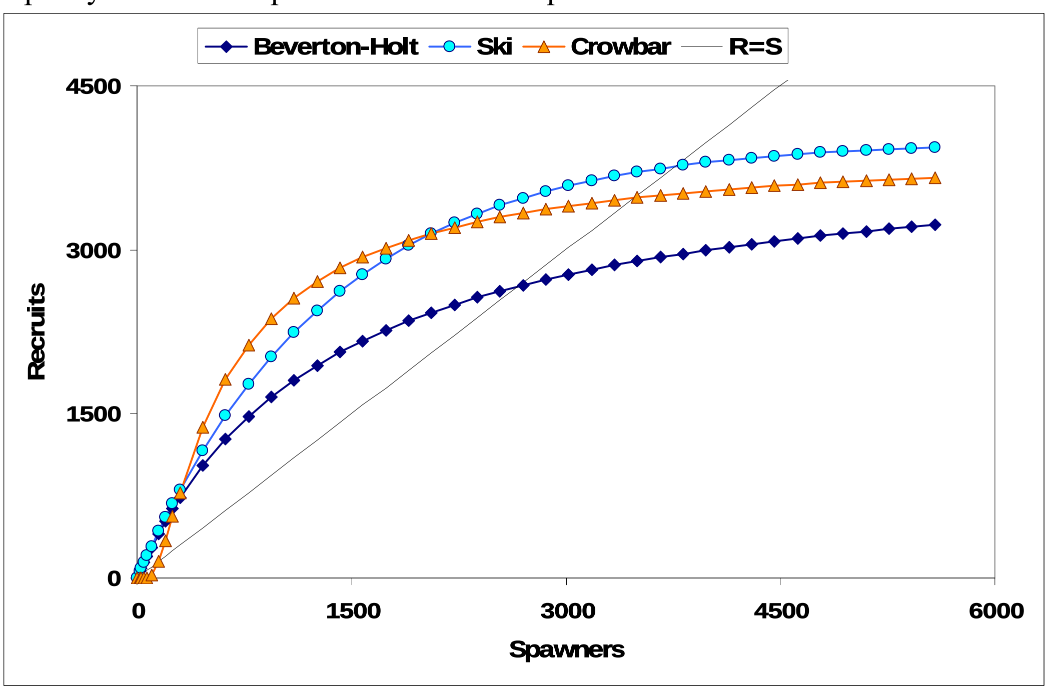

Figure 1: Recruitment curves for three spawner-recruit functions used within CAPM for an example where maximum recruits per spawner (a) equals 3.0 and maximum habitat capacity (b) equals 4,000.

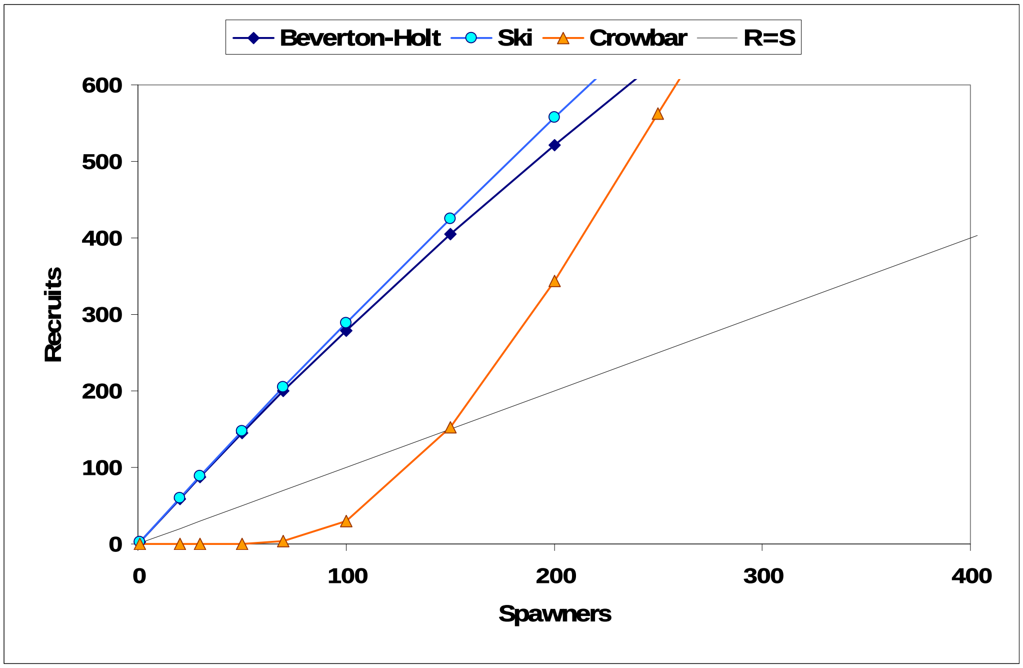

Figure 2: Recruitment curves for three spawner-recruit functions used within CAPM for an example where maximum recruits per spawner (a) equals 3.0 and maximum habitat capacity (b) equals 4,000, (same as Figure D1, but expanded to better compare functions at low spawner abundance levels).

As noted earlier, the CAPM recruitment functions included an environmental variable, SNEG. The inclusion of this variable changed the recruitment relationship from the 2-dimenisional curve of Figure D1, to a 3-dimensional recruitment surface as illustrated in Figure D3. Mathematically the inclusion of this second variable modifies the recruitment Equations 1, 2, and 3 to the following:

BH: $R_t = (a * S_t) / (1 + (a / b * S_t)) * (\exp(c * SNEG_{t+m}))$ (4)

SK: $R_t = b * (1 - \exp(- (a / b * S_t))) * (\exp(c * SNEG_{t+m}))$ (5)

CB: $R_t = b * \exp((- b * \exp(- 1)) / (a * S_t)) * (\exp(c * SNEG_{t+m}))$ (6)

Where R_t, S_t, a, b are as defined in previously (Equations 1, 2, and 3) and c = the parameter for the snow index $SNEG_{t+m}$ for the year t+m, where t = spawner brood year, and m = a modification number that best aligns the index to the recruitment performance of the species and region. The value for m ranges from +2 to -2.

The snow index (SNEG) was derived from the maximum annual snow depths observed at Mt. Rainier and Crater Lake National Park sample sites from 1945 to present. These annual average snow depths were then converted so they were expressed as a deviation from the 1945 to 2007 average. A negative value in the resulting data set meant the maximum depth was less than the 1945-2007 average; a positive value meant the snow depth measurement was greater than the 1945 to 2007 average. The SNEG index was calculated as the 7-year moving average of these data. Therefore the 1948 SNEG year was the average of annual snow deviations from 1945 to 1951, the 1949 SNEG year the average of annual data from 1946 to 1952, and so forth until the last index point for the 2004 SNEG year which was the average of the years 2001 to 2007. As illustrated in Figure 4 these 7-year averages show both a cyclic pattern and a downward slope since the 1950s.

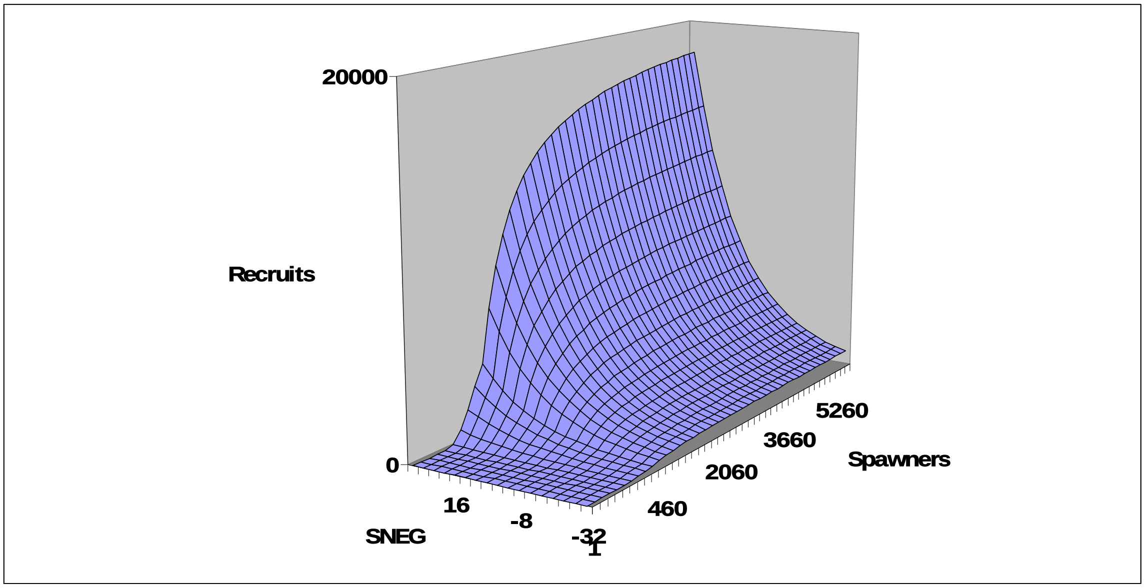

Figure 3: Illustration of Crowbar recruitment function with SNEG variable added (a = 3.0; b = 4,000; and c =0.05).

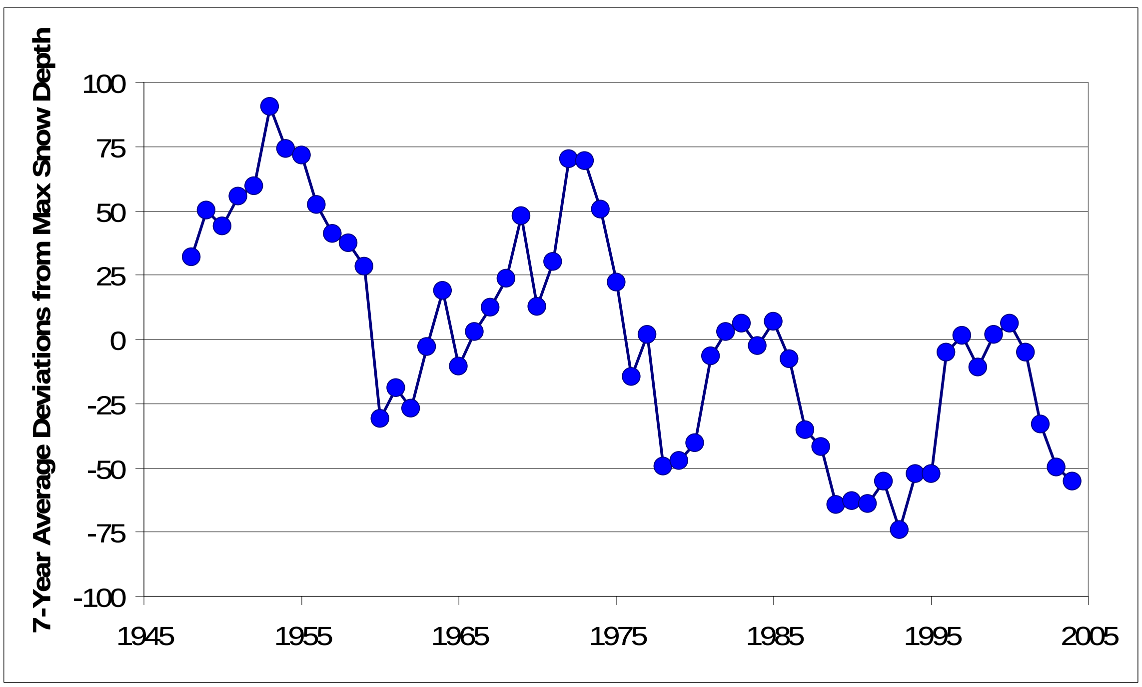

Figure 4: SNEG 7-year moving average index of annual maximum snow depths, expressed as deviations (cm) from 1945 to 2007 mean maximum snow depth; snow data are from annual measurements averaged for two high elevation monitoring stations, Mt. Rainier and Crater Lake.

Fitting Population Recruitment Curves

Fitting the recruitment curves to the observed data was a two step process. First a baseline data set of spawners, recruits, and SNEG index values was constructed. Because the SNEG index was not population specific, it was easily obtained following methods previously described. However, spawner and recruit data sets were more difficult to develop. For many populations, there are either no data or too few years of data to perform a recruitment analysis. Where sufficient data exist there were often data gaps of unknown hatchery fish fractions that had to be resolved in order to build a useable data set. The steps involved were specific to each population and are reported, along with the resulting data sets, in Appendix B. However, there several common and important elements to this data set building process that should be highlighted.

One of these important elements is that “spawners” are defined as the total of both wild and naturally spawning hatchery fish. When hatchery fish occur in the data base they are not given a reproductive success “discount” to correct for their likely reproductive inefficiency compared to wild fish. This discounting step was not taken for several reasons. First, it is not clear how much discount to apply. Second, such discounts may not appropriately account for the full impact of naturally spawning hatchery fish on subsequent population productivity. For example, in those studies showing large reproductive differences in reproductive success between naturally spawning hatchery fish and wild fish, a sizable portion of the naturally produced smolts were offspring of hatchery spawners. However, the marine survival of those natural smolts having hatchery parentage is typically less than those from wild parents. Therefore, there is likely a density dependent effect of hatchery spawners on smolt production, that can not be accounted for by applying a simple discount to hatchery spawners proportional full life history reproductive differences between hatchery and wild spawners.

Another important feature of the recruitment curve fitting was the estimation of brood year specific recruits from spawner escapement data. The number of recruits produced by each brood year of spawners was estimated by reconstructing each production group using the following relationship:

$$R_t = \sum [(A_j * S_{t+j})/(1 - F_{t+j})] \qquad (3)$$

where R_t represents the number of naturally produced (wild) recruits by fish that spawned in year t, A_j is the proportion of fish having age j at spawning (j = 2, 3, 4, 5, 6, 7), S_{t+j} is the number of wild spawners in year t + j, and F_{t+j} is the cumulative fishing mortality rate for the return of fish that spawned in year t + j.

The second step of fitting the recruitment curves was estimating the parameters for each equation and capturing the uncertainty associated with these estimates. Recruitment equation parameters were estimated via multivariate non-linear regression using the DataFit software developed by Oakdale Engineering (Oakdale, Pennsylvania). The DataFit software parameter estimation algorithm is based on the Levenberg-Marquardt method described by Marquardt (1963). Because the errors were assumed to be log-normally distributed, the form of the recruitment equations upon which the regression analyses were performed were modified to the lognormal form as follows, where ϵ_t represents the lognormal error term.

BH: $Ln(R_t) = Ln(a) + Ln(S_t) - Ln(1 + (a / b * S_t)) + (c * SNEG_{t+m}) + \epsilon_t$ (7)
SK: $Ln(R_t) = Ln(b) + Ln(1 - exp(- (a / b * S_t))) + (c * SNEG_{t+m}) + \epsilon_t$ (8)
CB: $Ln(R_t) = Ln(b) + ((- b * exp(- 1)) / (a * S_t)) + (c * SNEG_{t+m}) + \epsilon_t$ (9)

The regression input data as used for CAPM consisted of a table with the first two columns containing the annual values for the predictor variables S (spawner abundance) and SNEG (snow index) and the third column the corresponding values for the response variable Ln(R) (natural log of recruits). Each row of the data table represented the observations associated with one brood year.

The procedure for estimating equation parameters and the associated standard deviation of the residuals entailed more than performing a single DataFit-based regression analysis of a population's data set. Instead, a Monte Carlo bootstrapping procedure was used to repeatedly sample the population data set. A regression analysis was then performed on each data set sample using the same DataFit-based nonlinear regression routine. This meant that for every bootstrap sample an estimate of recruitment equation parameters and associated standard deviations were generated for all three recruitment curves (BH, SK, and CB). Therefore, if 500 bootstrap samples were drawn, 500 parameter and standard deviation estimates for each of the three recruitment equations would be generated. The primary purpose of this extended bootstrap procedure was to better understand the range and magnitude of possible errors in estimating recruitment equation parameters.

Simulating Population Recruitment – Addition of Stochastic Effects

An important element of simulating population recruitment within CAPM was the inclusion of random variation in the recruitment process. This element was intended to represent the effect of natural variations in annual recruitment and be consistent with the assumption that for real populations, the recruitment process is not a simple, unwavering deterministic process. To accomplish this, the error term (ϵ_t) in equations 7, 8, and 9 was replaced with a number (dev_t) randomly drawn from a normal distribution having a mean of zero and standard deviation equal to the regression standard deviation. Each time one of these equations was used within CAPM to simulate population recruitment; a new random number was drawn and used to calculate a new value for dev_t. This randomly fluctuating component of the recruitment equation was the primary source of stochasticity for CAPM population abundance simulations and ultimately estimates of extinction risk.

Simulating Population Recruitment – Snow Data

Each estimate of population recruitment within CAPM requires a value for SNEG. Therefore, for the population simulation portion of this model it was necessary to generate 100-year sequences of the SNEG index in manner that was consistent with observed SNEG values of the past Figure 4D. The method implemented within CAPM to do this begins by randomly selecting 120 snow depth values from a normal distribution having a mean equal to the 1945-2007 maximum snow depth value (442 cm) and a standard deviation from the same period (101 cm). For most model runs each pool of random numbers was adjusted downward to make the starting snow conditions equal to the average snow depth conditions of the last 30 years (1977 to 2007). This adjustment entailed subtracting the difference between 1977-2007 snow average and the 1945-2007 snow average (37cm) from each random number.

In addition, CAPM was also capable of performing model runs that assumed a downward long-term trend in maximum snow depth. To implement this capability within a model run, each randomly picked snow depth value was adjusted downward by subtracting: (the slope of the downward trend expressed as change in snow depth per year) x (year number in model run sequence -1). For example, in year 2 of the simulation, if the change rate was expected to be -0.5cm per year and then the snow depth value would be adjusted downward by -0.5cm. In year 3, the adjustment would be – 1.0 cm, for year 4, -1.5cm, and so forth.

Once the string of randomly selected snow depths had been selected and adjusted, they were converted to be deviations by subtracting from each, the 1945 to 2007 mean snow depth (37cm). Model run values for SNEG were then calculated as the moving 7-year averages of this string of annual snow depth deviations. The same string of SNEG values were used for each bootstrap sample and associated parameter estimates.

Assumptions about Future Conditions

Like all viability models, CAPM is build around assumptions concerning the future conditions a population will likely experience. Since model runs are meant to simulate a future time period lasting 100 years, these future condition assumptions are usually critical to extinction probabilities forecast by the model. As reported here, CAPM results were generally based on the assumption that future conditions would approximate those experienced by populations from 1977 to 2007. The primary way for doing this was as previously described, constructing the simulation values for the SNEG index such that they would be representative of the 1977 to 2007 observations.

However, with respect to harvest and naturally spawning hatchery fish, the simulated future conditions were not always intended to represent those observed from 1977 to 2007. For fishery harvest the reason for this was that fishery impacts prior to the 1990s were higher than those of the most years and those anticipated in the future. For example, Lower Columbia coho experienced cumulative harvest rates of 75% to 90% prior to 1990. Since 1990, these rates have been reduced and it is unlikely that for wild fish they will exceed 25% in the future. The assumed fishery impact rates used for all model runs are consistent for those presented earlier in Table 8 of this report.

The second departure from past conditions is the way hatchery fish were treated. In many populations the presence of hatchery fish on the spawning grounds has been a significant, yet highly variable feature. Most evidence suggests that naturally spawning hatchery fish tend to lower the overall natural reproductive rate for mixed populations of hatchery and wild fish compared to populations comprised only of wild fish.

However, in terms of natural offspring produced, hatchery fish may also make a substantial contribution, especially when they represent more than 50% of the natural spawning population.

The net effect of increased spawners and decreased reproductive rate as a result of naturally spawning hatchery fish is difficult to evaluate. However, for the purposes of understanding extinction risk and recovery potential, the key question is whether the wild fish would able to sustain themselves without 'reproductive support', should this 'support' indeed be a net positive benefit. As self-sustainability is the key question, we have chosen to make two conservative assumptions about future conditions as it relates to hatchery fish. First, that the recruitment parameters estimated during the period when hatchery fish were present in the past, are assumed to be representative of the wild fish in the population. Second, only wild fish are assumed to be present in the future and as a consequence their persistence dependent only on their ability to be self-sustaining. In this way the results obtained from the modeling exercise reflect our best estimate of the potential of the wild fish to maintain themselves in the future.

In reality, it is expected that substantial improvements in the natural reproductive rate will occur in many populations because the proportion of naturally spawning hatchery fish has been greatly reduced in recent years. For example, winter steelhead populations in the upper Willamette ESU. Also, it is possible that in places where naturally spawning hatchery will continue to occur (e.g., coho in the Youngs Bay streams), a portion of the naturally produced fish will be dependent on the reproductive contribution of stray hatchery fish.

However, the key question to be addressed by the viability modeling effort was: "if we assume the productive capacity of the population observed over the past 30 years is representative of wild fish, would such a population be self-sustaining in the future with no hatchery fish present." This is a conservative way to ask the question because it assumes any possible negative impact of past interactions with hatchery fish will continue and yet the future contribution of stray hatchery fish to the production of naturally produced offspring would be eliminated.

Program Mechanics

The CAPM is a program written in Visual Basic linked to nonlinear regression fitting algorithms provided by DataFit software. An Excel spreadsheet provides the user interface for data input, model run set up and display of the results. Starting with a population data set, CAPM proceeds through a series of calculations to produce multiple estimates of extinction probability. As stated earlier, the result is a distribution of extinction probabilities rather than a single estimate. The key steps in this series of calculations and the order in which they occur for the populations examined in this report are outlined in the following description.

After the raw population data and model conditions are entered, the program generates multiple estimates of the parameters for each of the three recruitment equations via the bootstrap process. For viability estimates 300 bootstrap samples were drawn as described earlier. Each of these bootstrap samples was fit to the three recruitment equations using the DataFit software. The net result was 900 parameter sets (i.e., 300 samples x 3 recruitment equations). For each parameter set, CAPM simulated a future 100 year sequence of population abundance. The starting abundance for each of these 100 year simulations was set to equal the value for recruitment equation parameter for capacity, b, as described in Equations 7, 8, and 9. These 100-year simulations were repeated 500 times for each parameter set. The number of 100-year simulations that were found to incur an 'extinction event', was then divided by 500 to obtain a probability of extinction for the parameter sample. This process was repeated until probabilities of extinction probabilities were obtained for all parameter sets. This modeling protocol was computer intensive and time consuming as it essentially required the simulation of 45 million years of population growth and abundance (i.e., 500 repetitions of a 100 year simulation period, for each of 900 parameter sets).

The definition of an 'extinction event' was the occurrence of an average spawner abundance that was less than a population-specific Critical Risk Threshold (CRT). Conceptually, the CRT was defined as the level below which recruitment processes and the likelihood of population rebound was judged to be uncertain and potentially unlikely.

The period of years used for calculating the average abundance to test against the CRT was equal to the average age of spawners in the population. So, for example, for coho a 3-year average was used and for summer steelhead a 5-year average was used. In summary, the detection of an 'extinction event' occurred if within the 100-year simulation of spawner abundance numbers, there was a sequence of years equal to the average age of the species (e.g., 3 years for coho) with an average abundance less than the CRT value.

The process of simulation recruitment under the CAPM also involved an assumption that at extremely low spawner abundance levels recruitment would totally fail. In other words, no recruits would be produced. This reproductive fail point was set at 20% of a population's CRT. So for example, the reproductive fail point for Clackamas coho was assumed to be 60, which is 20% of the CRT value of 300. Therefore, within the CAPM population simulations, zero recruits would be predicted for any simulated spawner escapement of Clackamas coho less than 60 fish.

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Viability Status of Oregon Salmon and Steelhead Populations in the Willamette and Lower Columbia Basins

Appendix F: PopCycle Model Description

September 2007

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Prepared for
Oregon Department of Fish and Wildlife and
National Marine Fisheries Service

PopCycle Model Description

PopCycle is a simple stochastic salmon stock recruitment model for analysis of population viability. The model estimates annual spawner numbers over a 100-year period for a prescribed number of iterations (Figure 1). The model is initialized with recent population size and subsequent numbers are calculated using a stochastic stock-recruitment function described by input parameters. Recruits are estimated as an ocean adult cohort. Annual numbers of fish from this cohort are apportioned among years based on an input age schedule. The model includes optional inputs to apply fishing rates in each year to calculate harvest and fishery effects on population dynamics. Optional inputs are also included for analysis of demographic effects of natural spawning by hatchery fish based on inputs for hatchery releases, release to adult survival, and rates of natural spawning by hatchery fish. Risks were expressed based on probabilities of future spawning escapement less than prescribed threshold values. The model is built in Microsoft Excel using Visual Basic. A simple interface page facilitates model use and review of results. Descriptions of derivation and application of model variables and inputs follow.

Conservation risks

This analysis estimates population viability based quasi-extinction and critical risk thresholds. A quasi-extinction threshold (QET) is defined as a population size where functional extinction occurs due to the effects of small population processes (McElhany et al. 2006). The model assumes that extinction occurs if the average annual population size over a generation (g) falls below this threshold at any point in a modeled trajectory. Quasi-extinction risk is thus estimated as the proportion of all iterations where the moving generational average spawner number falls below the QET at any point in each 100 year simulation. Estimated risks are compared to benchmark values of 60% 25%, 5%, and 1% risk levels identified by the Willamette/Lower Columbia Technical Recovery Team (McElhany et al. 2006) as corresponding to high, moderate, low, and very low extinction risks.

The analysis also considers risks of falling below a conservation risk threshold (CRT) that is greater than the assumed quasi-extinction level. The CRT level might be considered analogous to a point where a population is threatened with falling to lower levels where the risk of extinction becomes significant. For the purposes of this analysis, CRT is defined as a level where diversity is eroded and population resilience may be lost. CRT may be considered to be the risk of being threatened with becoming endangered with quasi-extinction.

Population-specific estimates of extinction risks and improvement scalars were based on QET values of 50 for all populations and CRT values ranging from 50 to 300 depending on species and the size of the basin inhabited by a population (McElhany et al. 2006). While there is an extensive amount of literature on the relationships among extinction risk, persistence time, population abundance, and level of variation in demographic parameters, there are no simple generic abundance levels that can be identified as viable (McElhany et al. 2000). Because empirical data on actual extinction and conservation risk levels is lacking, QET and CRT values were based on theoretical numbers identified in the literature based on genetic risks. Effective population sizes between 50 to 500 have been identified as levels which theoretically minimize risks of inbreeding depression and losses of genetic diversity, respectively (Franklin 1980, Soule 1980, Thompson 1991, Allendorf et al. 1997). Effective population size assumes balanced sex ratios and random mating. Benchmark values in this analysis assume approximately equivalent effects of differences between effective and census population sizes, and the multi-year generation structure of salmon (Waples 1990, 2004; Lindley et al. 2007). Relatively low QET values are supported by recent observations of salmon rebounds from very low numbers (e.g., Oregon lower Columbia River coho: ODFW 2005 and Washington lower Columbia winter steelhead: D. Rawding, WDFW, unpublished) and apparently-sustainable small population sizes of salmon in other regions (e.g., King Salmon River Chinook population in Alaska: McPherson et al. 2003).

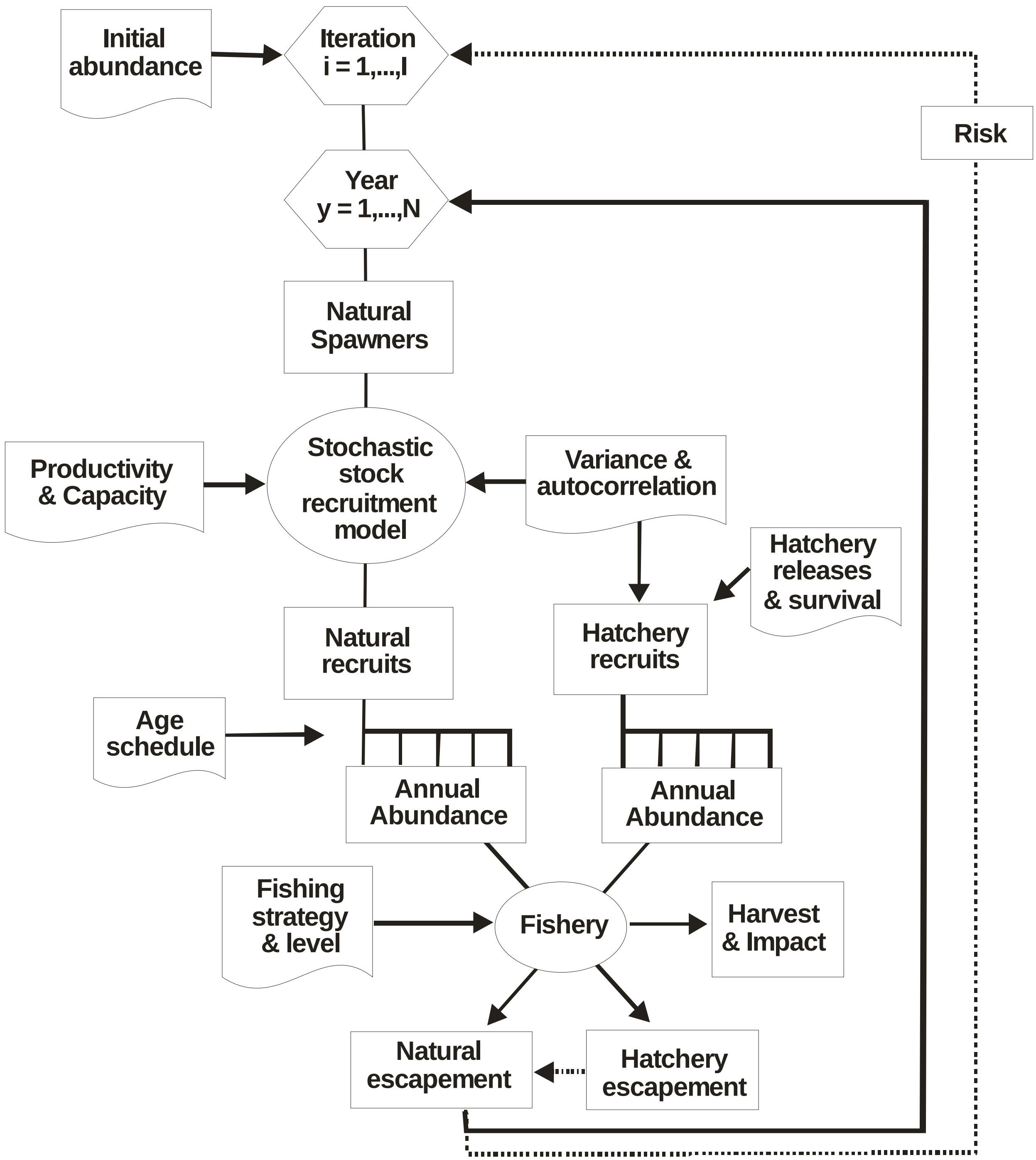


Figure 1: Model algorithm.

Stock-Recruitment Function

The model stock recruitment function can be based on either the hockey stick, Beverton-Holt, or Ricker functional forms.

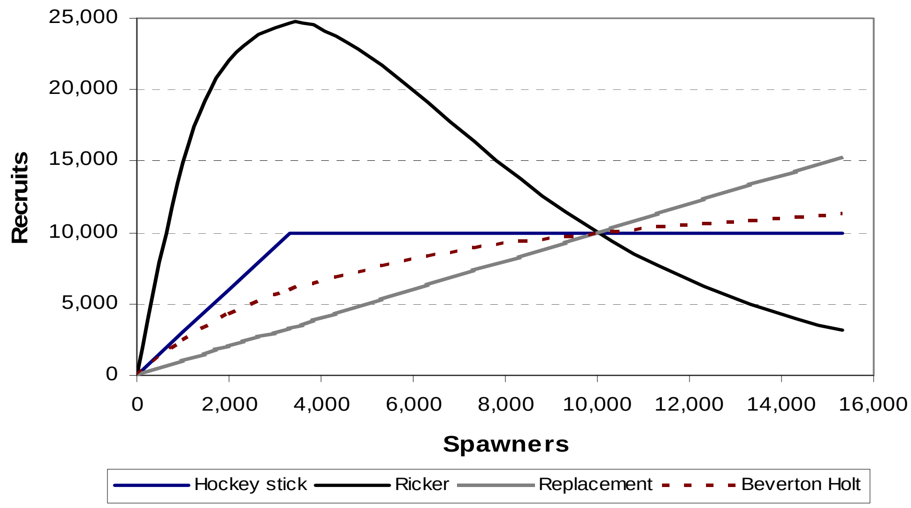


Figure 2: Example stock-recruitment curves based on a productivity parameter of 3 recruits per spawner (maximum observed at low numbers) and an equilibrium population size of 10,000.

The Hockey-Stick form of the relationship is:

$$R_y = (S_y)(p)(e^{\varepsilon}) \text{ when } (S_y)(p) < N_{eq}$$

$$R_y = (N_{eq})(e^{\varepsilon}) \text{ when } (S_y)(p) \geq N_{eq}$$

where

R_y = recruits,
S_y = spawners,
p = parameter for productivity (average recruits per spawner at spawner numbers under full seeding levels),
N_{eq} = parameter for equilibrium abundance,
e = exponent, and
ε = normally-distributed error term ~ $N(0, \sigma^2)$

The Beverton-Holt form of the relationship is:

$$R_y = \{a\, S_y / [1 + (S_y(a - 1)/ N_{eq})]\}\, e^{\varepsilon}$$

where

R_y = recruits,
S_y = spawners,
a = productivity parameter (maximum recruits per spawner at low abundance),
N_{eq} = parameter for equilibrium abundance,
e = exponent, and
ε = normally-distributed error term ~ $N(0, \sigma^2)$.

The Ricker form of the relationship is:

$$R_y = S\ e^{\alpha\ [1-(S/Neq)] + \varepsilon}$$

where

- R_y = recruits,
- S_y = spawners,
- α = Ricker productivity parameter (maximum recruits per spawner at low abundance),
- N_{eq} = parameter for equilibrium abundance,
- e = exponent, and
- ε = normally-distributed error term.

Population-specific assessments of risk and improvement scalars were based on the best available data for each population. Population-specific stock-recruitment parameters were used where available. Parameters were based on a hockey stick formulation and the mean RS approach identified by McElhany et al. (2006). This approach defines the equilibrium abundance based on the median pre-harvest recruitment level observed in the historical data time series.

The productivity parameter was based on the geometric mean of recruits per spawner for spawning escapements less than the median value in the data set. Pre-harvest stock-recruitment data was used to estimate intrinsic population parameters to account for significant and well documented changes in harvest patterns over time.

Population parameters were inferred from habitat conditions in many cases where population-specific stock recruitment data were unavailable. Habitat inferences were generally based on the Ecosystem Diagnosis and Treatment Model (LCFRB 2005). EDT results are in the form of Beverton-Holt function parameters. Note that MeanRS and Beverton-Holt equilibrium and productivity parameters are related but not directly comparable. Where specific population data were lacking, representative values were used consistent with the assumed population status based on other anecdotal information.

Analyses were based on initial population sizes equal to the average equilibrium abundance as specified with the corresponding stock recruitment parameter (N_{eq}). Equilibrium rather than recent abundance levels were used to provide estimates of representative long-term risks and avoid confounding effects of large annual fluctuations in spawner escapements in recent years.

For instance, viability estimates based on record low escapements during poor El Niño conditions of the late 1990s would have resulted in different results than would have been calculated from recent high returns associated with a post-El Niño transition to more favorable ocean conditions.

Additional sensitivity analyses were conducted to examine the effect of initial abundance on risks, particularly including near-term risks.

Stock-Recruitment Variance

The stochastic simulation model incorporated variability about the stock-recruitment function to describe annual variation in fish numbers and productivity due to the effects of variable freshwater and marine survival patterns (as well as measurement error in stock assessments). This variance is modeled as a lognormal distribution (e^{ε}) where ε is normally distributed with a mean of 0 and a variance of σ_z^2 (Peterman 1981).

The model allows for simulation of autocorrelation in stock-recruitment variance as follows:

$$Z_t = \text{Ø}\, Z_{t-1} + \varepsilon_t, \quad \varepsilon_t \sim N(0, \sigma_e^2)$$

where

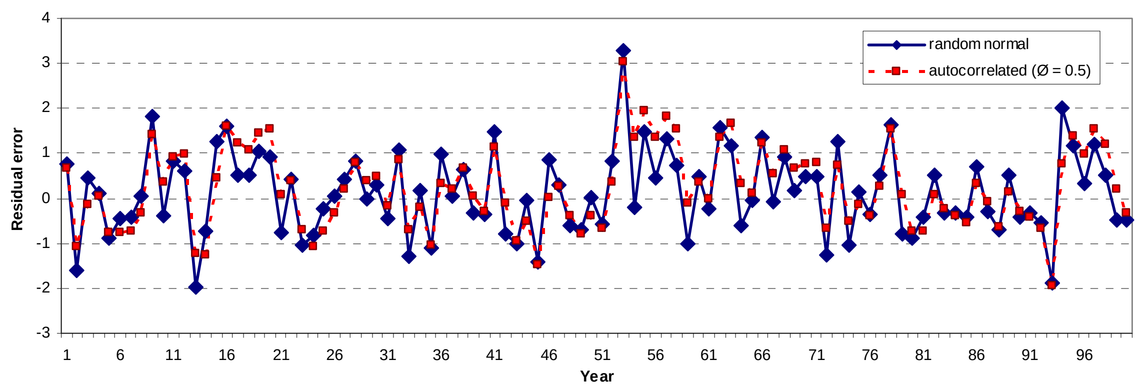
Z_t =	autocorrelation residual,
Ø =	lag autoregression coefficient,
ε_t =	autocorrelation error, and
σ_e^2 =	autocorrelation error variance.

The autocorrelation error variance (σ_e^2) is related to the stock-recruitment error variance (σ_z^2) with the lag autoregression coefficient:

$$\sigma_e^2 = \sigma_z^2\,(1 - \text{Ø}^2)$$

Model simulations using the autocorrelated residual options were seeded in the first year with a randomly generated value from $N(0, \sigma_z^2)$.

Variance and autocorrelation in population-specific risk analyses were generally based on species values reported by McElhany et al. (2006), except where good population-specific estimates were available for long term datasets.



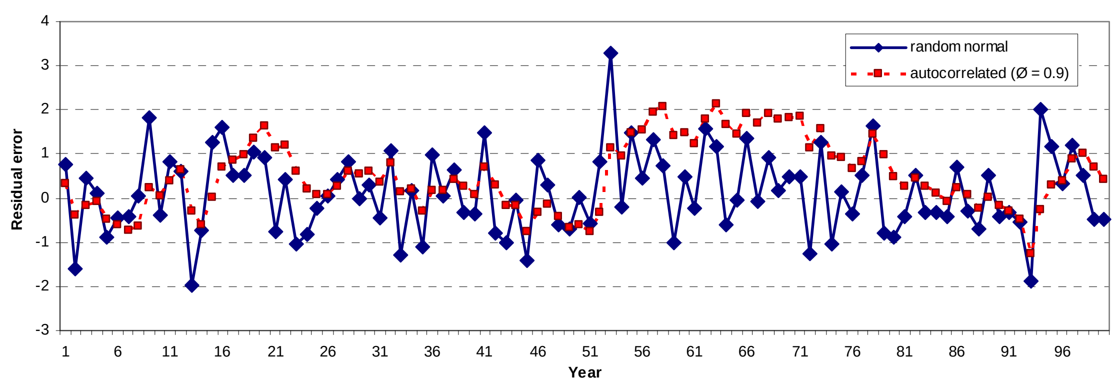


Figure 3. Examples of autocorrelation effect on randomly generated error patterns (σ_z^2 = 1).

Depensation & Recruitment Failure Thresholds

The model provides options to limit recruitment at low spawner numbers consistent with depensatory effects of stock substructure and small population processes. Options include 1) progressively reducing productivity at spawner numbers below a specified recruitment depensation threshold (RDT) and/or 2) setting recruitment to zero at spawner numbers below a specified recruitment failure threshold (RFT):

$$R' = R * (1 - \mathrm{Exp}((\mathrm{Log}(1 - 0.95) / (RDT - 1)) * S)) \text{ when } S > RFT$$
$$R' = 0 \text{ when } S < RFT$$

where

R' = Number of adult recruits after depensation applied,
R = Number of adult recruits estimated from stock-recruitment function,
S = spawners, and
RDT = Recruitment depensation threshold (spawner number).

Population-specific analyses were based on a RFT of 50 and a recruitment depensation threshold equal to the CRT.

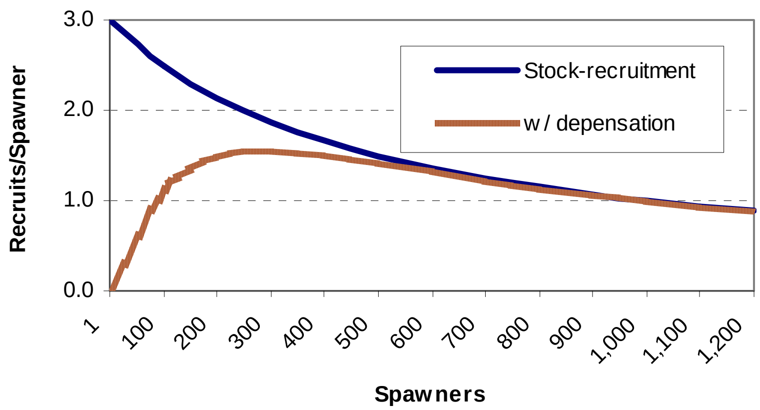


Figure 4: Example of depensation function effect on recruits per spawner at low spawner numbers based on a Beverton-Holt function (a = 3.0, Neq =1,000, γ =500).

Production Trend

The model includes an optional input to allow average productivity to be annually incremented upward or downward so that effects of trends in habitat conditions might be considered:

$$R'' = R' (1 + t)^y$$

where

R' = Number of adult recruits after depensation applied, and
t = proportional annual change in productivity.

McElhany et al. (2006) assumed a median annual decline of ln(y) = 0.995 to future simulations based on a precautionary expectation of declining snow packs, survival indices, and climate change. Population-specific analyses included in this analysis assumed a long-term trend equivalent to a 20% reduction in net productivity over 100 years.

Improvement Scalar

The model includes an optional scalar which is used to estimate the effects of incremental improvements in realized recruitment on quasi-extinction risks:

$$R^* = R'' (1 + C/100)$$

where

C = Improvement scalar (%), and
R* = Number of adult recruits after application of the improvement scalar.

Note that application of an improvement scalar results in a proportion increase in equilibrium population size and productivity at spawner numbers less than the equilibrium value (Figure 5). Population-specific improvement scalars will be used in future applications to represent increments needed to reach prescribed risk levels (1%, 5%, 25%) relative to a baseline at the time of the original ESA listing.

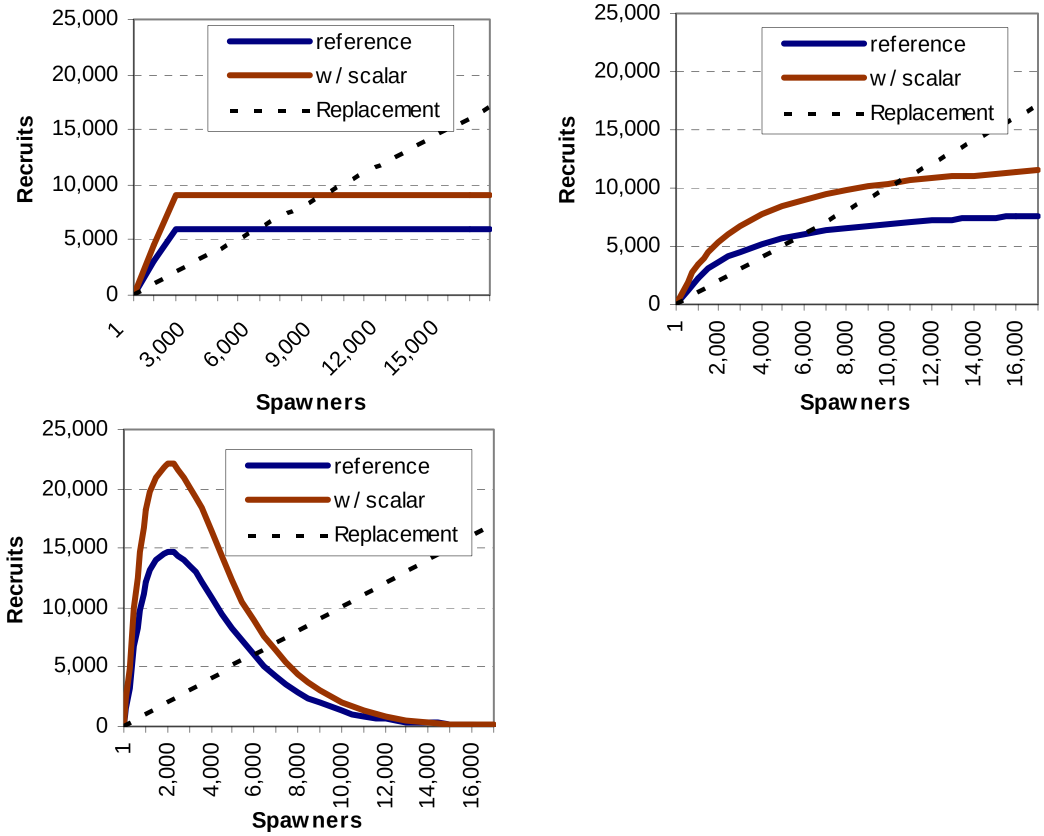


Figure 5. Example of effects of improvement scalar (50%) on hockey-stick, Beverton Holt, and Ricker stock-recruitment relationships based on an equilibrium abundance of 6,000 and a productivity parameter of 3 recruits per spawner.

Annual Abundance

Numbers of naturally-produced fish ($N_{.y}$) destined to return to freshwater in each year are estimated from a progressive series of recruitment cohorts based on a specified age composition:

$$N_{.y} = \Sigma N_{xy}$$

$$N_{xy} = R^*_{y-x} m_x$$

where

N_{xy} = Number of mature naturally-produced adults of age x destined to return to freshwater in year y, and

m_x = Proportion of adult cohort produced by brood year spawners that returns to freshwater in year x

Species-specific age schedules were based on unpublished WDFW data for fall Chinook (1980-2004 lower river tule returns) and average values estimated for other species in McElhany et al. (2006). McElhany et al. (2006) numbers were revised to include jack proportions for coho (age 2) based on Clackamas and Sandy River data and spring Chinook (age 3) based on McKenzie, Clackamas, and Sandy River data. Jacks were included to reflect their genetic contributions to effective population sizes.

Table 1: Average spawner age composition based on escapement data available for Willamette and lower Columbia salmon populations (McElhany et al. 2006 and WDFW unpublished).

Species	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Generation (yrs)
Coho	0	0.05	0.95	0	0	0	0	3
Spring chinook	0	0	0.05	0.54	0.40	0.01	0	4
Fall chinook	0	0.06	0.42	0.46	0.06	0.00	0	4
Chum	0	0	0.41	0.57	0.02	0	0	4
Steelhead	0	0	0.01	0.45	0.42	0.11	0.01	5

Hatchery Fish

The model includes option inputs for modeling co-occurring natural and hatchery populations. Number of hatchery-produced fish ($H_{.y}$) destined to return to freshwater in each year is estimated based on input juvenile release numbers (J), release-to-adult survival rates (SAR), and age composition (m_x):

$$H_{.y} = \Sigma H_{xy}$$

$$H_{xy} = (J)(SAR)(e^{\varepsilon})(m_x)$$

where

H_{xy} = Number of mature hatchery-produced adults of age x destined to return to freshwater in year y

Note that the model incorporates random normal variation in hatchery survival rates among release cohorts using a scalar based on natural productivity derived from the stock-recruitment variance. Thus, hatchery and natural numbers varied in strict tandem. The corresponding assumption would be that variation in hatchery and wild production was highly correlated due to common effects of freshwater and marine factors. Hatchery fish were not modeled in this risk analysis.

Fisheries & Harvest

Annual numbers are subject to optional fishing rates. This option is useful for adjusting future projections for changes in fisheries and evaluating the effects of alternative fishing strategies and levels. Fishery impact is defined in the model in terms of the adult equivalent number of fish that die as a result of direct and indirect fishery effects:

$$IN_y = N_{.y}\, fN_y \quad \text{and } IH_y = H_{.y}\, fH_y$$

where

- IN_y = fishery impact in number of naturally-produced fish,
- fN_y = fishery impact mortality rate on naturally produced fish including harvested catch and catch-release mortality where applicable,
- IH_y = Fishery impact in number of hatchery-produced fish, and
- fH_y = fishery impact mortality rate including harvested catch and other mortality where applicable.

Estimates of population-specific risks were based on pre-harvest stock-recruitment parameters calculated using fishery harvest rates representative of current conditions: 25% for coho, 25% for spring Chinook, 50% for fall Chinook, 50% for late fall Chinook, 5% for chum, and 10% for steelhead. Rates include ocean and freshwater fisheries and represent management practices in years prior to listing (intended to reflect conditions that led to status at the time of listing). Note that conservation measures implemented since listing have further reduced fishing rates from historical levels.

Spawning Escapement

Estimates of natural spawning escapement (S_y) include naturally-produced fish that survive fisheries plus a proportion of the hatchery escapement that spawns naturally decremented by the relative spawning success of a hatchery fish:

$$S_y = SN_y + SH_y$$

$$SN_y = (N_{.y} - IN_y)$$

$$SH_y = (H_{.y} - IH_y)\, q\, \tau$$

where

- SN_y = Naturally-produced spawners in year y,
- SH_y = Hatchery-produced natural spawners in year y,
- q = proportion of hatchery escapement that spawns naturally, and
- τ = spawning success of a naturally-spawning hatchery fish relative to that of a naturally-produced spawner.

The model also tracks the proportion of natural influence by hatchery fish (pNI):

$$pNI_y = SH_y / S_y$$

Note that the relative fitness of a hatchery spawner is applied only to first generation hatchery spawners and continuing hatchery fitness effects in subsequent generations are to be represented in model applications by changes in stock-recruitment parameters.

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