Revised Viability Criteria for Salmon and Steelhead in the Willamette and Lower Columbia Basins

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Willamette/Lower Columbia Technical Recovery Team
And
Oregon Department of Fish and Wildlife

Paul McElhany (NMFS, WLC-TRT), Craig Busack (WDFW, WLC-TRT), Mark Chilcote (ODFW), Steve Kolmes (University of Portland, WLC-TRT), Bruce McIntosh (ODFW), Jim Myers (NMFS, WLC-TRT), Dan Rawding (WDFW, WLC-TRT), Ashley Steel (NMFS, WLC-TRT), Cleve Steward (Steward and Associates, WLC-TRT), David Ward (ODFW), Tim Whitesel (USFWS, WLC-TRT), Chuck Willis (USACE, WLC-TRT)

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Part 1: Introduction

Report Overview

This report, developed jointly by the Willamette/Lower Columbia Technical Recovery Teams (WLC-TRT) and the Oregon Department of Fish and Wildlife (ODFW), consists of three parts: Part 1, which includes this overview, provides some basic definitions and concepts; Part 2 contains recommendations for viability criteria for Willamette and Lower Columbia salmon and steelhead; and Part 3 is an analysis of current extinction risk status for Oregon Lower Columbia River (LCR) coho populations. The current status evaluation of coho is provided as both a "test run" of the viability criteria, and as useful information for coho recovery planning. Evaluation of the current status for other WLC populations is planned for the future.

In 2003, the WLC-TRT released a report describing recommended viability criteria for salmon and steelhead Evolutionarily Significant Units (ESUs) in the WLC (McElhany et al. 2003). The viability portion of this report provides a revision of the 2003 criteria. The WLC-TRT, in collaboration with ODFW, undertook this revision to improve the criteria by incorporating new analyses by the WLC-TRT, other TRTs, state agencies, and others. In addition, the Lower Columbia Fish Recovery Board (LCFRB) applied the 2003 criteria in developing a recovery plan for the Washington portion of the LCR ESUs (ref) and this application suggested several modifications to the criteria.

Although written as a standalone document, this report heavily references the 2003 viability report. An understanding of the 2003 report will help in understanding this report because 1) to avoid redundancy, the rationale for many of the criteria from the 2003 report is not repeated here and 2) some sections of this report focus on why changes have been made from the 2003 report.

Although the criteria developed in this report should apply equally well to both Oregon and Washington populations, the viability criteria examples and the LCR coho current status assessment focus on Oregon populations. This is because Washington and Oregon are at different points in the recovery planning process. Washington has already completed an interim recovery plan that contains goals and current status assessments based largely on the 2003 WLC-TRT viability report (ref). Oregon is currently developing a recovery plan for WLC ESUs and is therefore in position to make immediate use of updated viability criteria and current status evaluations. Consequently, ODFW has been engaged in providing the most recent data for these analyses. Updating goals and status evaluations for Washington populations will likely be accomplished through the Lower Columbia Fish Recovery Board's (LCFRB) recovery plan revision process (ref).

It should also be noted that with respect to LCR coho, ODFW has been managing Oregon populations as a State Endangered Species since their listing by the Oregon Fish and Wildlife Commission in 1999. The recovery prescriptions and recovery criteria that resulted from this listing are contained within a separate State of Oregon management plan. This State plan is due for a 5-year review and update in 2006. It is the intent of ODFW to utilize as much as possible the WLC-TRT viability criteria document and analyses developed here as the basis for this updating.

Definitions

To understand the scope and focus of this report, it is useful to start with some definitions. Some of these terms were defined in the 2003 report and we are providing clarification or modification here; other terms were not explicitly defined in the 2003 report. These definitions are intended to be consistent with current NMFS definitions and policy.

<u>Viability criteria</u> – Viability criteria are the primary focus of Part 1 of this report. Viability criteria describe biological or physical performance conditions that when met indicate a population or ESU is not likely to go extinct. Viability criteria have two components: a ***metric***, which is the parameter measured, and a ***threshold***, which is the value of the metric above which a population or ESU is considered viable. For reasons described below, viability criteria focus on the biological performance of the fish as the primary indicator of extinction risk. The framework for the viability criteria follows the Viable Salmonid Population report (VSP, McElhany et al. 2000). Viability criteria are intended to inform delisting criteria and therefore focus on metrics that could be used in evaluations at some future point in time.

<u>Delisting criteria</u> – The Endangered Species Act (ESA) requires that recovery plans for listed species contain "measurable and objective criteria" that when met would result in the removal of the species from the endangered species list. To be removed from the list, a species must no longer be in danger of or threatened with extinction. Court rulings and NMFS policy indicate that delisting criteria must include both ***biological criteria*** and ***listing factor criteria*** that address the threats to a species (i.e., the listing factors in ESA section 4[a][1]). The viability criteria relate most directly to the biological delisting criteria; however, they are not synonymous. NMFS establishes delisting criteria based on both science and policy considerations. For instance, science can identify the best metrics for assessing extinction risk and thresholds of those metrics associated with a given level of risk, but setting the acceptable level of risk for purposes of the ESA is a policy decision.

<u>Listing factor (threats) criteria</u> –Delisting criteria must include both biological criteria and criteria that address the threats to a species, organized under the following five listing factors in section 4(a)(1) of the ESA:

A. the present or threatened destruction, modification, or curtailment of [a species] habitat or range;

B. over-utilization for commercial, recreational, scientific or educational purposes

C. disease or predation;

D. the inadequacy of existing regulatory mechanisms;

E. other natural or manmade factors affecting its continued existence.

This report does not provide a complete exploration of listing factor criteria. However, we do consider linkages between viability criteria and listing factor criteria and provide some recommendations for listing factor criteria, particularly with regard to habitat.

<u>Risk standards</u> – In developing viability criteria, it is necessary to define a level of "acceptable risk" to inform setting thresholds. Since viability criteria are intended to inform delisting criteria, thresholds need to relate to some standard for acceptable risk at the ESU scale. However, there is currently no quantitative definition of acceptable risk at the ESU scale under the ESA, and

evaluating ESU risk quantitatively is problematic anyway. The viability criteria follow the VSP framework, which partitions the ESU into component populations. At the population scale, NMFS has given policy guidance that for Pacific salmon and steelhead a population with >95% persistence probability be considered "viable" – at least for initial exploration (policy ref). Although this provides some guidance, it is recognized that there is no simple relationship between a population level risk standard and ESU-level acceptable risk.

Broad-sense recovery goals – The recommendations in this report are focused on thresholds related to ESA delisting. Other, broad-sense recovery goals may be developed by recovery planners that are consistent with ESA delisting but are designed to go beyond delisting to achieve other legislative mandates, treaty obligations, or cultural and social values. Development of such goals is outside the scope of this report.

Current ESU status evaluation – A current ESU status evaluation is an assessment of the current extinction risk for populations and ESUs. Like viability criteria, current status evaluation relies on *metrics and thresholds.* However, viability criteria (as defined above) differ in an important way from current status evaluations. Current status evaluations are based on the information that is currently available on the ESU in question, whereas viability criteria are necessarily more speculative and describe metrics and thresholds for data that have not yet been collected. The viability criteria can be considered "prospective" and the current status analysis "retrospective". This distinction is discussed in more detail below.

Hatchery policy – In 2005, NOAA published a policy in the Federal Register clarifying the role of hatchery production in risk assessments (ref). As currently being applied, the policy states that a non-listed ESU must be naturally self-sustaining and must be able to persist without input of hatchery-produced fish. This standard is used in the viability criteria and current status evaluations in this report.

Recovery strategies and actions – NOAA asked TRTs to recommend viability criteria and to evaluate current population and ESU status. Recovery strategies and actions will be developed in other recovery planning forums and not by TRTs. However, where recovery strategy issues seem obviously to flow from TRT analyses we occasionally discuss those issues in this report. Recovery strategies and actions are not a purpose of this report, however, and this is not a comprehensive treatment of recovery strategy issues.

Monitoring Programs – A rigorous research, monitoring, and adaptive management framework is essential in ESA recovery plans. Research and monitoring helps to ensure that appropriate data are collected and evaluated to assess the biological status of ESUs, status of threats to ESUs, effectiveness of recovery actions, and overall progress toward recovery. Adaptive management ensures that recovery actions are adjusted based on results of research and monitoring, so that plans will be more effective and efficient both biologically and economically. This TRT report does not include a comprehensive monitoring strategy; however, to the extent that it is useful and practical, given the primary purpose of this report, we discuss monitoring issues that flow from our analyses.

ESU scenario – The viability criteria described in this report allow for some flexibility in which populations will be targeted for a particular recovery level to achieve a viable ESU. An ESU scenario is an explicit description of which populations in an ESU are targeted for a given recovery level. Developing an ESU scenario requires both biological and policy considerations and will be undertaken in other recovery planning forums.

Conceptual Issues

Limits to viability criteria – Evaluation of population and ESU status--now or in the future-- should utilize all available, relevant information. When defining viability criteria, however, it is impossible to know exactly what information will be available in the future (since it depends on what monitoring is implemented and on specific environmental conditions). Therefore, it is unrealistic to expect that viability metrics and thresholds developed today will be the only determinants of species status in the future. In addition to uncertainty about what information will be available in the future, there will also likely be advances in assessment methods. Despite these uncertainties, these viability criteria can: 1) give a sense of the order of magnitude of improvement required for populations and ESUs; 2) provide guidance on what to monitor to evaluate extinction risk; and 3) provide information for prioritization among populations and risk factors.

Limits of risk assessment – The technical challenge inherent in the viability criteria question is substantial in that it requires identifying conditions at the threshold between threatened and not threatened. Most evaluation techniques, including quantitative population viability analysis (PVA), are relatively reliable at determining when a population is clearly at risk or clearly not at risk, but are unstable in determining the status of populations on the cusp (ref). This basic instability suggests that we should include an explicit consideration of uncertainty in setting thresholds.

Viability criteria vs. current status methods – The methods in this report used for the viability criteria and the current status evaluation of LCR coho are similar but not identical. Where data were available, we applied the viability criteria to the coho populations as one informative source of information on population status. In addition to the evaluation of viability criteria metrics, we evaluated other quantitative and qualitative information about population status. This is in keeping with our approach that an actual population evaluation should not simply consider the *a priori* viability criteria, but should consider *any* relevant information.

Part 2: Viability Criteria

Viability Overview

Criteria Framework – Following the approach in the 2003 viability report and the VSP report, criteria were developed based on a hierarchical framework (Figure 1). The ESU is partitioned into demographically independent populations (*sensu* VSP) and the populations are then grouped into strata (a.k.a. "Major Population Groups" (ref)) that share similar environments, life-history characteristics, and geographic proximity. The status of individual populations is estimated by examining a number of population attributes. The status of each stratum is determined by considering the status of each of its member populations; the status of the ESU as a whole is determined by considering the status of each of its strata. The populations and strata for WLC ESUs used in this report are defined in Myers et al. (2005). Copies of population and strata boundary maps from Myers et al. (2005) are provided in Appendix A of this report.

In the VSP report, the four population-level attributes were: 1) abundance; 2) growth rate/productivity; 3) spatial structure; and 4) diversity. In the 2003 report, the WLC-TRT used five attributes: 1) abundance and productivity; 2) Juvenile out-migrant (JOM) productivity; 3) diversity; 4) habitat; and 5) spatial structure. In this report, we use three population-level viability attributes: 1) abundance and productivity; 2) spatial structure; and 3) diversity. We combine abundance and productivity into a single attribute (as we did in 2003) rather than separate them as in the VSP report because abundance and productivity are so interlinked in how they affect extinction risk that they need to be considered simultaneously. In this report, we consider JOM productivity as a subset of abundance and productivity and do not follow the 2003 approach of designating it a separate attribute. The habitat criteria described in the 2003 report are now included as part of our discussion of listing factors criteria.
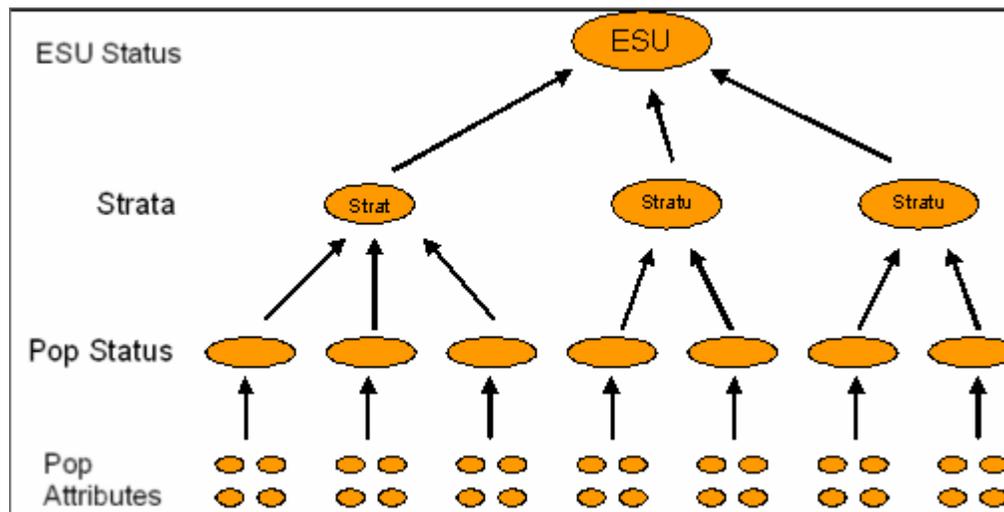

Figure 1 Diagram of hierarchical viability criteria.

ESU- Level Criteria

The TRTs were asked to provide criteria that would be informative for ESA delisting decisions. This requires an explicit or implicit definition of the term "threatened species" as used in the ESA. In the 2003 report, we defined a viable ESU as one that is unlikely to be at risk of

extinction, or as one with a high probability of persistence. These are very qualitative definitions but it is our intent that the criteria we have associated with a viable ESU describe an ESU that is not a "threatened species" as the term is used in the ESA. There is currently no quantitative definition of a threatened species (e.g., X risk of extinction in Y years) and there is no quantitative risk level associated with our ESU viability criteria.

In describing a viable ESU, the 2003 viability report ESU level criterion stated that:

> "1. Every stratum (life history and ecological zone combination) that historically existed should have a high probability of persistence."

The strata represent major diversity units within the ESU and provide a substantial buffer against the negative effects of environmental variation, catastrophic events, and loss of genetic variation (discussed in 2003 viability report). The TRT considered that loss of any particular strata would significantly increase the extinction risk to the ESU. The TRT continues to support the view that loss of any stratum is a significant reduction in the resilience of the ESU.

One reason for restoring all strata to a high persistence level discussed in the 2003 report has received increased support in the past few years. Maintaining diversity provides a buffer against the uncertain future presented by global climate change. Recent studies are beginning to detail the possible extent of climate change effects on salmon habitat (ref). These studies argue strongly that the landscape of the Northwest will undergo profound changes and because each of the strata will likely respond differently (and still unpredictably) to these changes, it is prudent to plan for maintaining all strata.

However, restoring all historical strata to a "high persistence" level may prove extremely difficult and policy makers have had to explore the continuum of ESU-level risk associated with some strata not at the high persistence level. Such cases should be evaluated on a case-by-case basis. One case that has already arisen in applying the 2003 criteria involves recovery of the gorge stratum (LCFRB ref). Because of passage problems at Bonneville Dam and the flooding of habitat by the Bonneville pool, recovery of the upper gorge strata to high persistence probability for some ESUs will likely be very challenging and as a consequence, it is uncertain if a "high persistence" stratum, as defined by the TRT criteria, can be re-established. In evaluating this particular case, the TRT concluded that the ESU-level risk in not having all strata would clearly be higher, but that the increased risk would be reduced by the fact that 1) if the goals of the plan were achieved, although the strata would not meet the TRT's criteria for high probability of persistence, they would be improved in status from their current condition; 2) the gorge stratum and cascade stratum are relatively similar as compared to the cascade vs. coast stratum so the buffering effect of diversity is not as great; 3) the cascade stratum is targeted for "very" high persistence (above minimal TRT strata criteria) to help buffer the ESU; and 4) options for recovery of the stratum are preserved in case future conditions or analyses require high stratum persistence for ESU viability.

Strata Level Criteria

To define a "high persistence" stratum, as used in the ESU-level criteria, the 2003 viability report provides the following criteria:

> "1. Individual populations within a stratum should have persistence probabilities consistent with a high probability of strata persistence.

2. Within a stratum, the populations restored/maintained at viable status or above should be selected to:

a. Allow for normative metapopulation processes, including the viability of "core" populations, which are defined as the historically most productive populations.

b. Allow for normative evolutionary processes, including the retention of the genetic diversity represented in relatively unmodified historical gene pools.

c. Minimize susceptibility to catastrophic events."

The first criterion is then developed into a quasi-quantitative framework for determining an adequate persistence probability for each individual population. The approach is based on defining the persistence probability of individual populations on a qualitative 0-4 scale, then assessing stratum risk by averaging the population "scores." The extinction risk associated with each of the categories is shown in Table 1 and the stratum thresholds are shown in Table 2. In addition to meeting the stratum average threshold, the 2003 criteria required that a high persistence stratum have at least two of the population in category 3 ("viable") or greater.

Table 1 Population persistence probabilities associated with persistence categories (copied from 2003 viability report).

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

Table 2 Population persistence category averages associated with stratum criteria (copied from 2003 viability report).

Stratum Persistence Probability Category	Average of Population Risks
Low persistence	Average < 2
Moderate persistence	2 <= Average < 2.25; at least two populations >= 3
High persistence	Average >= 2.25; at least two populations >= 3

The origin and motivation for this approach are provided in the 2003 viability report and are not repeated here. We continue to find the approach reasonable. It explicitly recognizes that population risk is a continuum and there may be many combinations of population status that could result in a high persistence stratum. The actual level of risk associated with the threshold was based on qualitative professional judgment, as we did not consider enough data existed to parameterize the metapopulation model needed to provide a quantitatively derived threshold. Since the 2003 viability report, several other TRTs have developed strata level criteria (ref.). Although these teams have considered and conducted initial explorations of the metapopulation models, they have ultimately also relied on the expert judgment approach. Although the exact criteria differ among TRTs, we consider the qualitative level of risk for the strata criteria among TRTs to be similar.

There are some concerns, however, about the implied precision of a threshold that goes out two decimal places (e.g., 2.25). Because we use a qualitative population score (0-4) to describe a qualitative criteria (high stratum persistence), there is little quantitative precision in the criteria. One way to remove the misperception that the threshold is precise is to rescale the problem. The

0-4 scale is arbitrarily based on the level of precision that we think is provided by the population level risk assessments; perhaps we should change the scale such that the stratum risk levels are identical, but the actual threshold does not include so many decimal places (e.g., a threshold of 60 rather than 2.25 might sound better even if it means the same thing). This is an issue of presentation rather than substance, but since presentation matters, we have explored the issue in Appendix B. Despite the potential attractiveness of rescaling, we conclude that it is preferable to keep the original 0-4 scale and the 2.25 threshold and then emphasize the actual level of precision (or lack thereof) associated with the threshold.

Whereas the first stratum-level criterion addresses *how many* populations need to be viable, the second stratum-level criterion addresses *which* of the populations need to be viable. The 2003 viability report provides a list of populations considered "core" and "legacy," but provides no quantitative guidelines for this second criterion. The 2003 report relies on case-by-case consideration of proposed strata-level scenarios and we support continuing that approach. In developing a recovery plan for the Washington portion of the LCR ESUs, the LCFRB developed a scenario that seemed to satisfy this criterion.

ESU-Level Recovery Strategies

The 2003 viability report recommended two ESU-level recovery strategies:

"1. Until all ESU viability criteria have been achieved, no population should be allowed to deteriorate in its probability of persistence.

2. High levels of recovery should be attempted in more populations than identified in the strata viability criteria because not all attempts will be successful."

Although the strata-level criteria allow that not all populations must be viable for the ESU as a whole to be at low risk, these two recovery strategies provide some important cautions about "writing off" populations early in the recovery planning process. We continue to support these recommendations.

Population-Level Status

To apply the strata-level criteria we need to integrate the assessment of individual population attributes into a 0-4 persistence category "score" for each population. In the 2003 viability report, this was accomplished by a method that evaluated each population level attribute on a 0-4 scale, and then estimated overall population status as a weighted average of the individual attributes. The weighting gives twice as much influence to the abundance and productivity score as to the other attributes.

This is a simple, straightforward way to integrate the population attributes, but it does not directly incorporate uncertainty into the evaluation. The Oregon Coast Coho TRT has been developing an intriguing fuzzy logic decision support system for integrating multiple population attributes; however, we have not yet sufficiently explored this option and so continue to use the simple averaging approach. To incorporate uncertainty into the averaging approach for the LCR coho current status evaluation (Part 3), we used expert opinion to define probability distributions for the individual attribute scores, and then took a weighted averaged of the distributions to obtain an overall population score probability distribution.

Population Abundance and Productivity Criteria

Background and Introduction

In the 2003 viability report, we provided the following guidelines for abundance and productivity criteria:

ADULT POPULATION PRODUCTIVITY AND ABUNDANCE CRITERIA GUIDELINES
1. In general, viable populations should demonstrate a combination of population growth rate, productivity, and abundance that produces an acceptable probability of population persistence. Various approaches for evaluating population productivity and abundance combinations may be acceptable, but must meet reasonable standards of statistical rigor.
2. A population with a non-negative growth rate and an average abundance approximately equivalent to estimated historical average abundance should be considered to be in the highest persistence category. The estimate of historical abundance should be credible, the estimate of current abundance should be averaged over several generations, and the growth rate should be estimated with an adequate level of statistical confidence. This criterion takes precedence over criterion 1.

We continue to support these general guidelines. The guidelines recognize that a variety of approaches may be taken in evaluating abundance and productivity, and several methods were discussed in the 2003 report. In this update to the 2003 report, we more fully explore the types of analyses and metrics that are useful for estimating a population's probability of persistence with reasonable statistical rigor, and provide guidance on when to employ a particular approach.

It is useful to provide some clarification on the relationship between criterion 1 and 2. The first criterion implies that we can model extinction risk as a function of abundance and productivity and set viability thresholds based on that modeling. The second criterion was developed with the recognition that model predictions are uncertain, but we are reasonably confident that the historical populations were viable. Therefore, regardless of any model predictions, a population performing at historical levels would be considered viable. This criteria approach is dependent on our population definitions and the assumption about historical viability.

The fundamental data used to evaluate population abundance and productivity is a time series of abundance (e.g., Figure 2). Additional information is often required to evaluate the time series and relate any information in the time series into an assessment of population extinction risk. This additional information may include data on the fraction of hatchery origin spawners (e.g., Figure 3), the population harvest rate (e.g., Figure 4), population age structure, etc. The 2003 report discussed using such information to evaluate population abundance and productivity through simple rules of thumb, a population growth rate approach (PCC), recruits per spawner analyses, and multi-lifestage modeling. These approaches exhibit a range of data requirements. At one extreme are rules-of-thumb, which often require minimal information on raw abundance, and at the other extreme are multi-lifestage models, which often require detailed information on life-stage specific density-dependent survival. In general, the models with greater data requirements will provide more precise (and hopefully more accurate) estimates of risk, assuming that they are adequately parameterized. However, data are often lacking, and in the 2003 report we recommended that the population change criteria (PCC) approach be used as a default because it has relatively minimal data requirements and the potential biases of the method are reasonably well understood. A more comprehensive approach to viability criteria would

consider the entire range of data types available and craft metrics that extract the most information on risk status from each data type.

In this report, we replace the approach of using PCC as a default with a more generalized viability curve approach. We introduced the concept of a "viability curve" in our 2003 report. A viability curve is described by a combination of population abundance and productivity that produce the same extinction risk (e.g., 5% risk in 100 years). This can be shown as an extinction risk iso-cline on a graph plotting population abundance against population productivity (Figure 5). Populations with an abundance and productivity above and to the right of the viability curve are considered "viable" whereas those below and to the left are considered "not viable." This has proven a useful framework for considering abundance and productivity viability criteria because it emphasizes the interaction between abundance and productivity while highlighting the importance of a population's productivity as a viability predictor.

Applying the viability curve approach requires two separate but closely related analyses. The first analysis describes the functional relationship between abundance, productivity, and extinction risk (i.e., drawing the curve). The second, related, analysis determines the best metric for evaluating a given population relative to the viability curve. Much of the discussion that follows involves data-dependent variations on these two analyses. In general, we have developed "idealized" viability curves for each population, and focus uncertainty in the analysis on the estimation of where a population is relative to the curve. We are not recommending a single method for drawing the viability curve and evaluating status relative to the curve. Rather, we recommend using and comparing multiple viability curve methods for any real population assessment because the best analysis is likely to be data dependent and cannot be entirely specified *a priori*. For use as initial goals, we have provided "benchmark" curves and evaluation methods. Although these benchmark goals and methods are useful for providing guidance, it is important to remember that they do not provide a complete answer to the question of whether a population has demonstrated adequate abundance and productivity. Thoroughly evaluating whether a population has demonstrated adequate abundance and productivity will likely require population-specific analyses (see the LCR coho evaluation in Part 3 as an example).
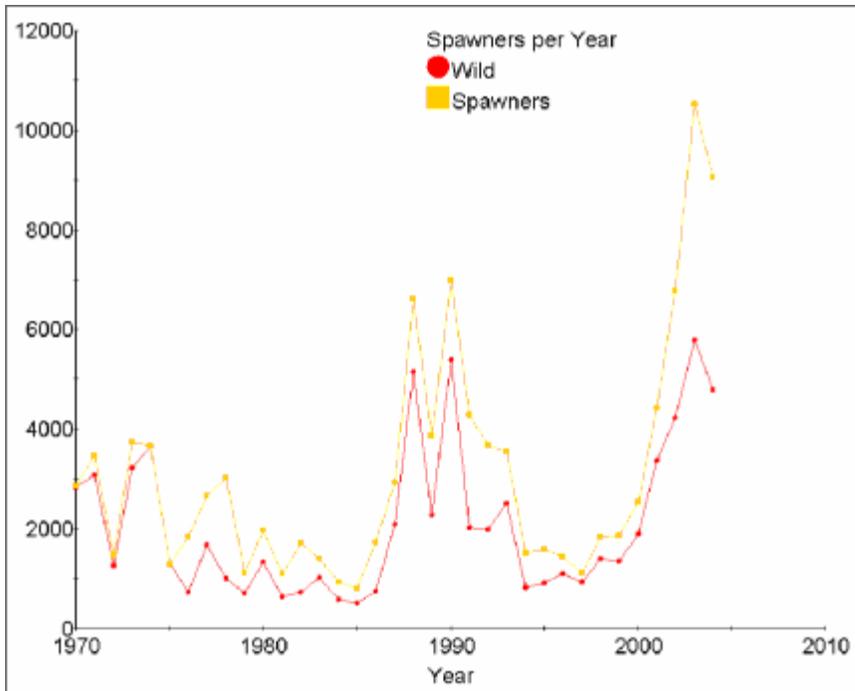
Figure 2 Time series of abundance for McKenzie River Spring Chinook (see Appendix C for data source). The red line indicates natural Oregon spawners and the yellow line indicates hatchery origin spawners.
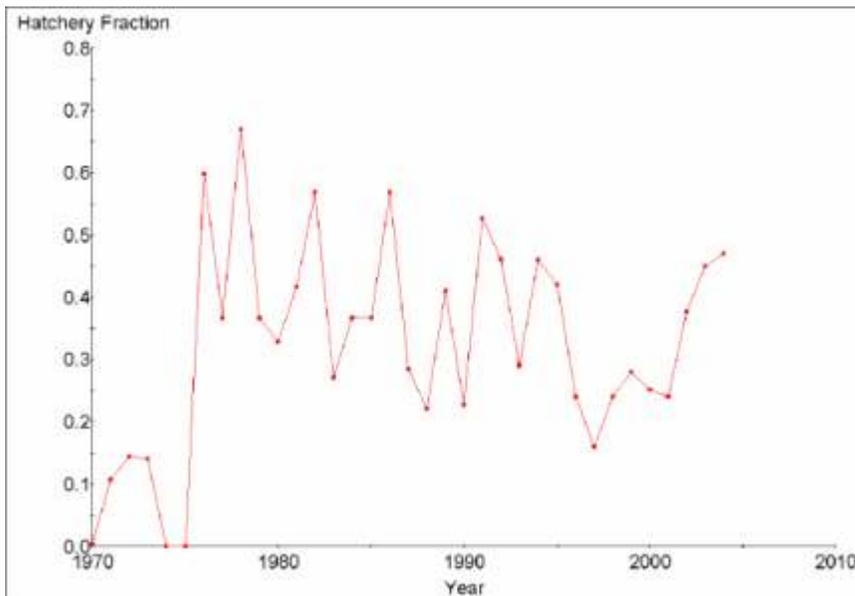
Figure 3 Time series of the fraction of hatchery origin spawners in the McKenzie River Spring Chinook population (see Appendix C for data source).
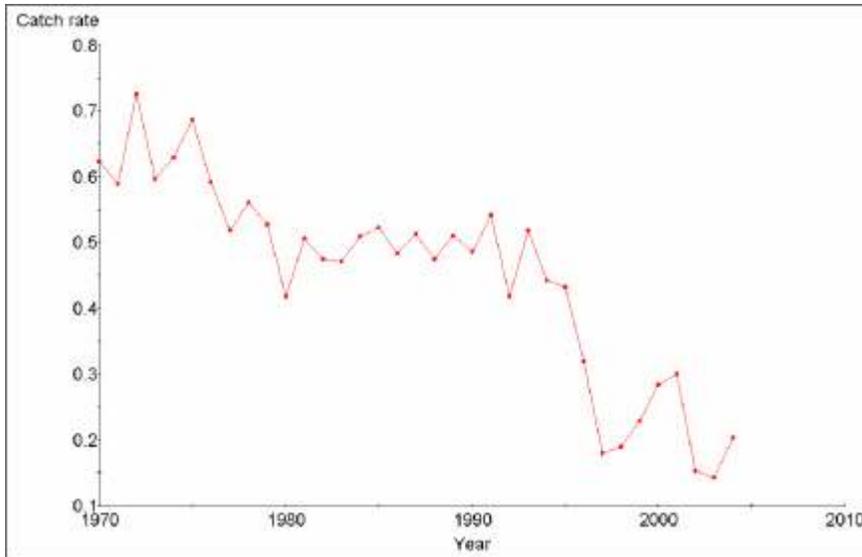
Figure 4 Time series of harvest rate on natural origin McKenzie River Spring Chinook (see Appendix C for data source).
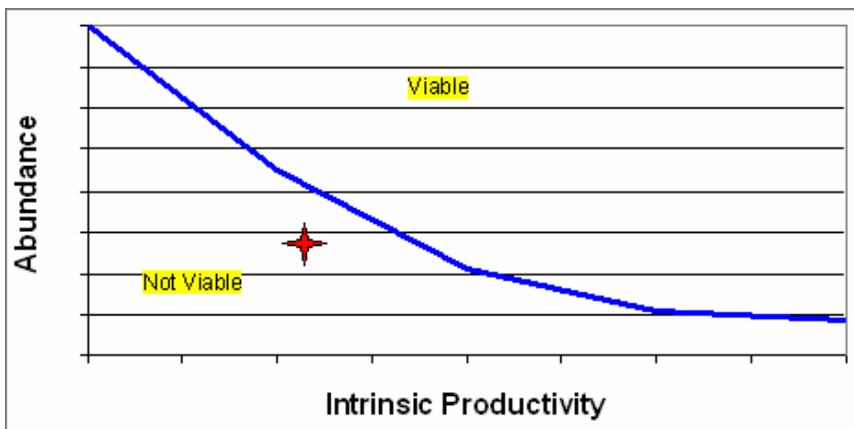
Figure 5 Conceptual illustration of the viability curve approach. All the combinations of abundance and productivity on the curve have the same extinction risk. The star indicates a population with a non-viable combination of productivity and abundance.

Drawing the Viability Curve

To draw the curve, we start by identifying an appropriate population dynamics model, which is used to predict population extinction risk. To draw a viability curve, we perform many different runs of the extinction risk model where only capacity and productivity vary (i.e., all other parameters are held constant), then systematically explore combinations of productivity and capacity to identify combinations that have a specified extinction risk.

Drawing the viability curve requires a model of extinction risk and there are a number of factors to consider in developing the model. At a minimum, the model must include parameters for abundance and productivity (i.e., the axis of the viability curve graph), but the model must also consider variability in productivity, any density dependent relationships, the appropriate definition of "extinction" and a host of other issues. Below, we describe some of these issues and how they have been incorporated into our curve development. Also, in Appendix D, we present sensitivity analyses exploring curve development. Much of the analyses done with viability

curves have been done using the computer program SPAZ (Salmon Population Analysis Zprogram) developed by Payne and McElhany (ref) available on the internet at …

Recruitment Function

The recruitment function describes how many offspring ("recruits") are produced by a group of spawners. It is often appropriate to use a density dependent function, in which the number of recruits per spawner decreases as the total number of spawners increase. Generically, a recruitment function can be written as

$$R = f(a,b,S) * \varepsilon_p,$$ eq.1

where R = recruits, f is some recruitment function, a is intrinsic productivity, b is a term related to capacity, S is spawners, and ε_p is a random variable representing process error. If age structure is included in the equation, it becomes

$$R_t = \sum_{i=1}^{\max Age} m_i \left(f(a,b,S_{t-i}) * \varepsilon_p \right),$$ eq. 2

where m_i is the fraction of fish that return at age i and max Age is the maximum age of return.

Density dependent recruitment functions commonly used in fisheries biology include the Beverton-Holt, Ricker and Hockey Stick (see Figure 6). The intrinsic productivity describes the number of recruits that spawner is likely to produce if there are very few other spawners (i.e., no density effects). This is the "productivity" parameter of the viability curve graph. The capacity term is the average maximum number of recruits that can be produced, no matter how many spawners are present. In our viability curve graphs, we have NOT presented capacity on the "abundance" axis, but rather a related term, the predicted equilibrium abundance (N_{eq}). The N_{eq} is the expected long-term average number of recruits to the population. It is a function of both the population's capacity and productivity and the specific type of recruitment function. A population's N_{eq} will generally be less than a population's capacity because of density effects (except for the Hockey Stick function where $N_{eq} = b$). As a population's productivity increases, N_{eq} gets closer to the capacity. We present N_{eq} in the viability curve graphs rather than capacity because the N_{eq} more closely represents the number of fish that are expected to be observed in the population.

The process variance term ε_p in equation 1 describes how much variability there is in the recruitment relationship. For example, 100 spawners do not always produce the same number of recruits; there is some variability. The amount and pattern of this variability is an important determinant of a population's extinction risk. In general, populations with highly variable productivity are at higher extinction risk because there is an increased chance that a population will decline to low abundance (i.e., have low recruitment). Issues surrounding variance can be complex and this topic is addressed further in a following section.

It is important to note that drawing the viability curve does not in itself require estimation of any specific population's abundance or productivity. The curve is a generic construct that is made by estimating the extinction risk associated with all possible combinations of abundance and productivity, then identifying the combinations with the same acceptable extinction risk. (The actual algorithm used is a bit more computationally efficient than this simple description, but this is conceptually what happens.)

Although population abundance and productivity estimates are not needed to generate the viability curve, analysis of existing empirical time series is important for determining which recruitment function is appropriate and to estimate the pattern of population variability. In addition, analysis of existing data is important for estimating some of the other parameters, such as QET (see below).

We recommend that a thorough evaluation consider multiple recruitment functions, as the most appropriate model may be population or data specific. However, after exploring a number of recruitment functions (Appendix D), we propose that viability curves using the Hockey Stick model will be generally informative, particularly in combination with the meanRS evaluation method described below. An analysis comparing the fit of different recruitment models to available data indicated generally poor precision in parameter estimates for all models and little distinction in the quality of fit to different models (Payne and McElhany, in prep). For example, the data for McKenzie spring Chinook in Figure 6 do not provide strong support for *any* of the recruitment models. Since analysis of the existing time series provides no compelling empirical reason to select a particular function, we have opted to use a relatively simple model that has the basic features we expect in a density-dependent recruitment relationship – namely that higher productivity populations are more "resilient" (i.e., tend to increase if perturbed to low abundance) and that there is some maximum number of recruits supported by the environment. Although we are not describing the Hockey Stick model as a "default," we have used it in the current status evaluations and provide benchmark viability curves for all WLC species based on this recruitment function.

There has been some debate among the TRTs and elsewhere over which recruitment model is most "conservative" or "precautionary." In many ways, this seems like an ill formed question – all of the recruitment models have parameter combinations that would indicate a very robust population and all have combinations that would indicate a population with a high extinction risk. Rather than ask which recruitment function is most or least precautionary, it seems more appropriate to ask about the level of precaution associated with an entire criteria package (i.e., metric and threshold). We believe that our approach to drawing the viability curve and the method for incorporating uncertainty into the assessment of a population's status relative to the curve provide the information needed to make an educated decision in selecting more or less precautionary criteria.
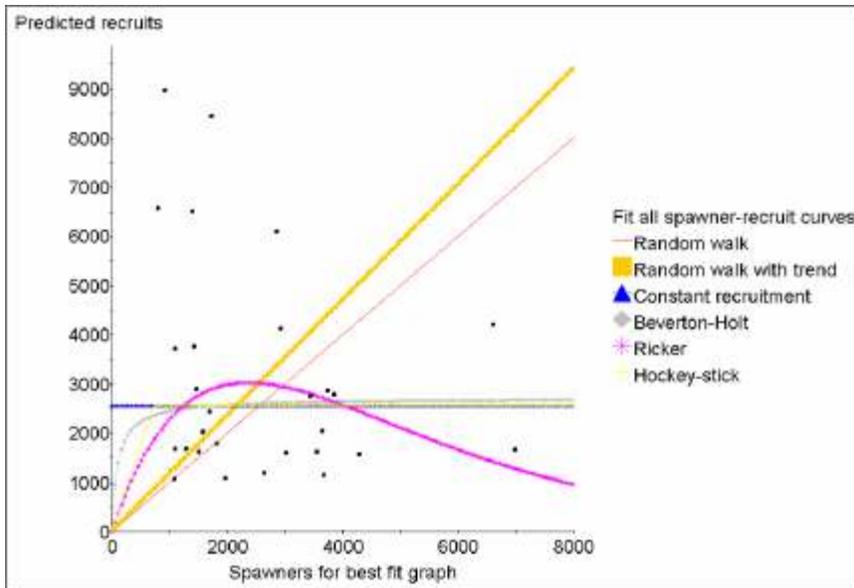
Figure 6 Recruitment functions fit to McKenzire River spring Chinook data. Analysis based on preharvest recruitment (see Appendix C for data sources).

Initial Population Size

The initial population size affects a population's extinction risk. All else being equal, a population with a smaller initial population size has a higher extinction risk than a population with a large initial population size. For the purposes of creating the viability curve, we set the initial population size at the N_{eq} associated with a given abundance and productivity combination. Thus, populations have no initial tendency to increase or decrease, but could do so because of population variability.

Process Error

A key issue in drawing the viability curve is how to model and estimate process error (equation 1). Since process error is generally a multiplicative process (e.g., the product of many small survival probabilities) it is typically modeled as being lognormal and expressed as e^x, were x is distributed $N(0, \sigma^2)$. Process errors can be treated as either independent or as temporally autocorrelated. If the errors are independent, there is no tendency to have "good" or "bad" streaks – any year is as likely to be above or below average recruitment as any other. If the errors are autocorrelated, bad years (lower than average recruitment) would tend to be followed by more bad years and good years followed by good. Note that these are just tendencies; even with autocorrelation sometimes bad years will be followed by good and vice versa. Because of the higher possibility for sequential years of poor recruitment, adding autocorrelation will generally increase extinction risk (all else being equal).

Patterns of marine survival are likely to be highly temporally autocorrelated, resulting in high autocorrelation in recruitment (Figure 7). Periods of favorable or unfavorable conditions for salmon are referred to as "regimes" and seem to occur on decadal scales, though the pattern is far from regular. Autocorrelation is included by setting ε_p as a random variable of the form e^x, with x normally distributed with zero mean, variance σ^2, and temporal correlation matrix G.

Variance can be estimated by fitting equation 1 to available salmon time series. We used a Bayesian approach which fits a, b and σ^2 simultaneously (Payne and McElhany, in prep). We estimated the correlation matrix by calculating the correlation of the residuals from the fitted model at different temporal lags. This produces a correllogram plotting correlation against lag, which can be converted into a correlation matrix (Figure 8).

Because variance and autocorrelation estimates from individual populations tend to have low precision, estimates averaged across multiple populations may be more accurate. Table 3 shows the pooled (averaged) variance estimates for each species based on analysis of Oregon WLC populations. These pooled estimates were used to generate the benchmark curves. We only used up to second order (2-year) lag correlations because at longer lags the number of available data points for estimating correlation declines such that the correlation estimate becomes very uncertain.

For the autocorrelated model, the y-axis of the viability curve graph is still the equilibrium abundance of the deterministic recruitment function and the initial size for the extinction risk modeling is still set as this equilibrium abundance. The viability curve is conceptualized to represent the extinction risk under "standard" or long-term average conditions – the a and b parameters are the long-term average since the process error is set with a median of zero.

The challenges in estimating variance and autocorrelation are substantial because of potential problems such as 1) the data sets are relatively short, which greatly reduces precision and prevents detection of autocorrelation with long time lags, 2) there is often a great deal of measurement error which cannot be separated from process error, 3) assumptions in the estimation of recruits, such as using a fixed age structure, may lead to errors in the estimation of autocorrelation. We have taken a relatively simple approach to estimating variance and autocorrelation - further analysis on this topic would be useful, but our estimates provide initial values for the benchmark curves.
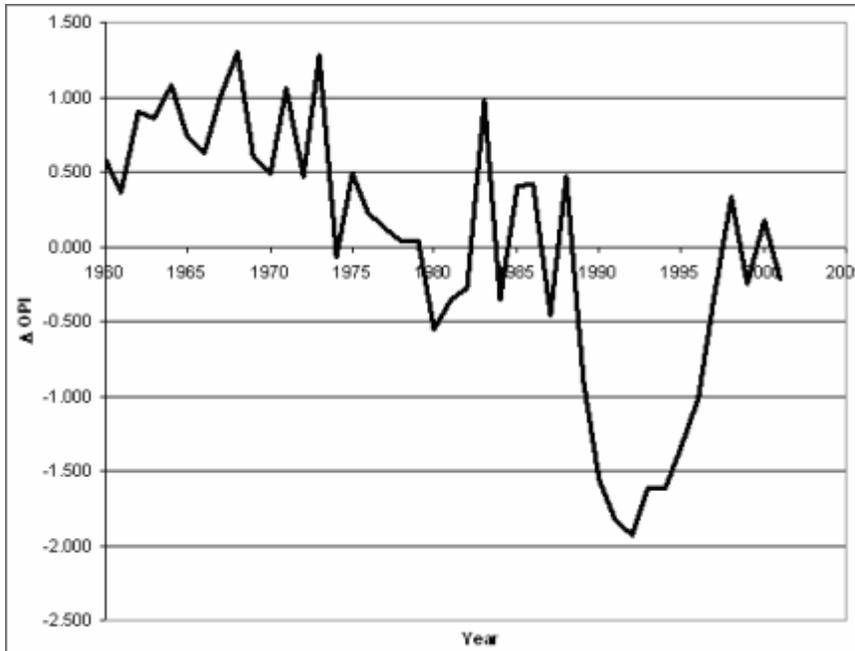
Figure 7 Deviations in the Oregon Production Index (OPI) from mean conditions (log scale). The OPI is and index of Oregon coast coho marine survival. The index shows multi-year periods of higher than average survivals (1960s) and multi-year periods of lower than average survivals (1990s) [ref].
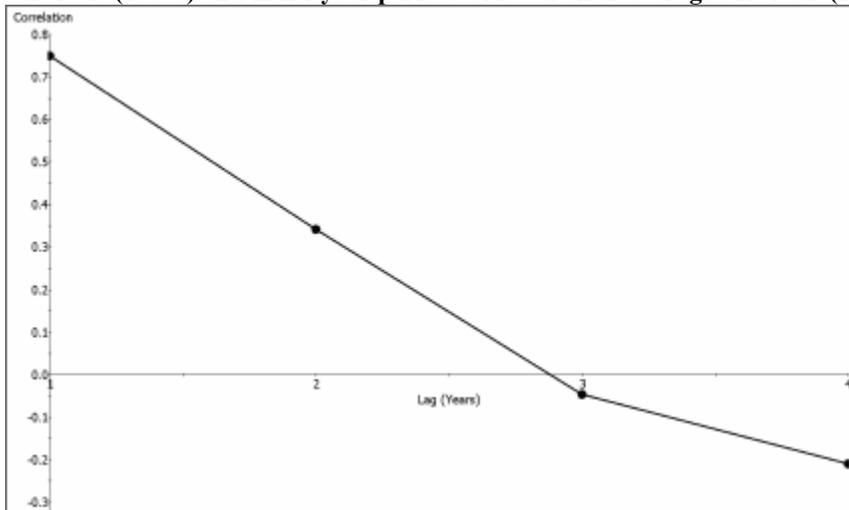
Figure 8 Correllogram for McKenzie River Spring Chinook. Based on residuals from Hockey Stick recruitment function and pre-harvest recruitment.

Table 3 Hockey Stick variance and autocorrelation. Chinook, coho and steelhead estimates are based on average of Oregon WLC populations. Chum variance based on average of Grays River and Lower Gorge populations from WLC-TRT viability report (McElhany et al., 2003, Appendix G). Chum autocorrelation based on average of Chinook, coho and steelhead values.

Species	Variance	Correlation (Lag1)	Correlation (Lag 2)
Chum	1.050	0.467	0.215
Chinook	0.614	0.451	0.180
Coho	1.050	0.429	0.154
Steelhead	1.208	0.548	0.311

Long-Term Trend

There are some indications of long-term downward trend in recruitment conditions. These indications include declining snow packs, declining survival indices (e.g., OPI) and projections of climate change. To accurately reflect expected future conditions, it is precautionary to include these downward trends. These could be included in equation 1 by setting the mean of the process error to something other than zero,

$$\varepsilon_p = e^x, x \approx N(y, \sigma^2) \qquad\qquad \text{eq. 3}$$

where the annual median rate of decline in recruitment is $\ln(y)$. (The average rate is $\ln(y - \sigma^2/2)$.

A perpetual long-term trend would ultimately lead to inevitable extinction. We are not hypothesizing a perpetual downward trend, but are interested in exploring the consequences of a downward trend over a relatively short 100 year time span. An analysis of a declining index of snow pack which is correlated with salmon survival (Appendix C), indicates a median annual decline of $\ln(y) = 0.995$. We have NOT included this downward trend in the benchmark curves. Including this trend would shift the viability curves up and to the right.

Age Distribution

Setting the viability curve requires an estimate of spawner age structure. For the viability curves we used estimates of the average age structure pooled at the same species scale as the variance estimates. These age structure estimates are shown in Table 4. For the benchmark curves, steelhead were assumed to be semmelparous, though they actually show some repeat spawning (average <10%).

Table 4 Average fraction of fish at each age that return to spawn for species in the Oregon WLC.

Species	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7
Chum	0	0	0.41	0.57	0.02	---	---
Chinook	0	0	0	0.572	0.42	0.008	---
Coho	0	0	1	---	---	---	---
Steelhead	0	0	0.007	0.453	0.422	0.112	0.006

Depensation

At very low abundance numbers, populations may experience a decrease in reproductive success because of factors such as the inability to efficiently find mates, random demographic effects (the variation in individual reproduction become important), changes in predator-prey interactions, and other "Allee" effects. Such depensatory effects are difficult to detect statistically with available data, but it is precautionary to include depensatory processes in creating the viability curve. In developing the viability curve, we have modeled depensation in terms of a simple reproductive failure threshold (RFT). If the population of spawners in any particular year drops below the RFT, the number of recruits from those spawners is set at zero. If the number of spawners in a year is below the RFT, the population is not necessarily extinct because it could be rescued by fish that are still in the ocean that will return in the next few years.

Many of the processes that can drive depensation are a function of both the absolute abundance of the population and the population density on the landscape. At very small population size, populations are likely to be at risk from some processes (e.g., demographic stochasticity) no

matter how densely they are packed in a watershed. However, for some processes, such as the likelihood of finding a mate, the risk is a function of how widely distributed the fish are. Because of the need to consider both absolute and density dependent processes, we have set the RFT for the benchmark curves based on binning populations into watershed size categories. This allows setting the RFT with some absolute bounds (both maximum and minimum) while also taking into consideration population density in a non-linear way. The watershed size categories are shown in Table 5, RFT values associated with each size category are shown in Table 6, and the values for specific Oregon WLC populations are shown in Table 7. The size categories (Table 5) are species specific to reflect the requirements for different species to sustain different size populations. The values for RFT associated with each size category in Table 6 are based largely on analysis by Chilcote, which estimates RFT values for WLC populations on a fish per km basis (Appendix E). The species specific differences in RFT values are based on density differences observed in relatively healthy populations. It is assumed that if healthy populations of a particular species tend to occur at a higher density the RFT for that species will occur at a higher density. This is a largely untested assumption and illustrates just some of the uncertainty associated with these thresholds. Although we believe these thresholds provide reasonable values for the benchmark curves, it is important to explore sensitivity to these values and test these assumptions during any population evaluation (Appendix D). Our estimate of relative density for healthy populations by species follows the general pattern of steelhead < chinook < coho < chum.

Extinction Threshold

Generating a viability curve requires defining the conditions where a model trajectory is considered "extinct". Because of concern about depensatory processes and uncertainty about how both the populations and the models perform at very low population size, we typically model populations to a "quasi-extinction threshold" (QET). Ecological and demographic risk processes not captured in the simple recruitment function model are likely to come into play at abundances below the QET. An extinction event is more than a single year reproductive failure and we have set QET as a threshold abundance averaged over a population's mean generation time. Like the RFT, processes that affect QET are likely to be a function of both absolute abundance and of how the population is spread out on the landscape, so we have set QET using the same size category approach as setting RFT. Based on an analysis by Chilcote (Appendix E), we have estimated the QET for the benchmark curves at the same values as the RFT (Tables 4 and 5). If the average annual population size over a generation falls below this threshold at any point in a modeled trajectory, the population is considered extinct. All of the caveats and concerns about uncertainty associated with the RFT thresholds also apply to the QET values. Based on the new analyses by Chilcote, the RFT and QET values differ from the 2003 PCC analysis, which used a value of 50. It is tempting to conclude that since the new QETs are higher the criteria are more precautionary. However, the model used in 2003 (PCC) is different from the model in these benchmark curves, making a direct comparison problematic.

Table 5 Watershed size categories based on historical spawning stream km.

Species	Watershed Size Category		
	Small	**Medium**	**Large**
Chum	<50	50-100	>100
Chinook	<50	50-150	>150
Coho	<100	100-200	>200
Steelhead	<100	100-200	>200

Table 6 Modeling and Criteria Thresholds. Stream km is based on ODFW database. In extinction risk modeling, id a population drops below the reproductive failure threshold (RFT) in a single year, the reproductive success for that year only is assumed to zero. In the extinction risk modeling, the average annual population size over a sequential period equal to the length of one generation drops below the quasi-extinction threshold (QET) at any point during a simulation trajectory, the population is considered extinct.

Species	Size Category	RFT & QET
Chum	Small	100
	Medium	200
	Large	300
Chinook	Small	50
	Medium	150
	Large	250
Coho	Small	100
	Medium	200
	Large	300
Steelhead	Small	50
	Medium	100
	Large	200

Table 7 Thresholds for Oregon WLC populations. The number of fish per spawning km associated with the threshold is shown in parentheses rounded to nearest km. The stream km is a combination of the "Spawning and rearing" plus "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).

ESU	Life History	Population	Stream (Km)	Size Category	RFT & QET
Lower Columbia Chinook	Fall	Big Creek	16	Small	50 (3)
		Clackamas	61	Medium	150 (2)
		Clatskanie	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 River	7	Small	50 (7)
		Youngs Bay	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	N/A	N/A	N/A
		Clatskanie	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 River	N/A	N/A	N/A
		Youngs Bay	91	Medium	200 (2)
Lower Columbia Coho		Big Creek	78	Small	100 (1)
		Clackamas	465	Large	300 (1)
		Clatskanie	105	Medium	200 (2)
		Lower Gorge Tributaries	14	Small	100 (7)
		Sandy River	247	Large	300 (1)
		Scappoose River	125	Medium	200 (2)
		Youngs Bay	94	Small	100 (1)
		Hood River	119	Medium	200 (2)
Lower Columbia Steelhead	Summer	Hood River	131	Medium	100 (1)
	Winter	Clackamas	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	59	Medium	150 (3)
		Clackamas	182	Large	250 (1)
		McKenzie	244	Large	250 (1)
		Molalla	104	Medium	150 (1)
		North Santiam	129	Medium	150 (1)
		South Santiam	190	Large	250 (1)
		Middle Fork Willamette	272	Large	250 (1)
Upper Willamette Steelhead	Winter	Calapooia	91	Small	50 (1)
		Molalla	240	Large	200 (1)
		North Santiam	198	Medium	100 (1)
		South Santiam	323	Large	200 (1)

Harvest

Deciding how to include harvest in the analysis is challenging. TRT criteria have tended to be set based only on escapement, where escapement is the number of fish that have returned to the spawning grounds after experiencing all sources of mortality. This has the advantage of treating harvest the same as other sources of mortality. However, harvest is somewhat different in that it can be changed quickly and explicitly in response to fish performance. Consequently, escapement can become decoupled from fish performance in the entire pre-harvest portion of the lifecycle. This can lead to misleading status assessments if only the post-harvest (escapement) data are used. A classic example of where this has occurred is in Oregon coho, where escapement remained relatively steady but harvest rates dropped from >90% to <20%. The near constant escapement masked a serious decline in productivity (i.e., populations went below replacement.)

To obtain a clearer picture of how fish are performing, we developed both escapement (i.e., post-harvest) and pre-harvest viability curves. For the escapement curves, we based the abundance axis of the benchmark viability curves on fish returning to the spawning grounds, and drew the viability curve without any additional harvest (all harvest occurs prior to escapement). For the pre-harvest curves, we based the abundance axis of the benchmark viability curves on pre-harvest recruitment values, and drew the viability curve assuming a particular harvest strategy in the future. A pre-harvest assessment of population status would be based on estimates of pre-harvest productivity and abundance. A disadvantage of the pre-harvest approach is the requirement to assume a particular harvest strategy for 100 years into the future. However, the alternative is to base the assessment of population status only on potentially misleading escapement estimates. In examining current (2005) population status, this problem is likely to be particularly acute because harvest rates have varied greatly over the recent past. By looking at both escapement and pre-harvest analyses, we expect to obtain a more accurate assessment.

Since there is uncertainty about what harvest strategies will be employed in the future, in Appendix D we explore criteria sensitivity to this parameter. For the benchmark curves, we assumed that future harvest will be a similar to current harvest rates (Table 8). For the benchmarks, we have modeled harvest as a simple fraction of the pre-harvest recruits, but a more complex strategy could also be evaluated. Since projected harvest rates are likely to be population specific, a complete population evaluation should probably explore more future harvest rate assumptions that the average values used for the benchmarks.

Table 8 Future harvest rate assumptions for Oregon WLC populations based on approximations of current harvest rates.

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%

Acceptable Extinction Risk

A single viability curve shows abundance and productivity combinations that have a single extinction risk where extinction risk is defined as the probability of dropping below the QET in a given amount of time. To use the viability curve as a management threshold, the curve must define an "acceptable risk" level. Determining the acceptable level of risk is ultimately a policy decision. Initial guidance from NMFS defined a viable population as one with an extinction risk of less than 5% in 100 years. For our strata evaluation approach, populations are evaluated on a 0-4 scale and the threshold risks associated with that scale are 1%, 5%, 25% and 40% in 100 years. The benchmark curves have been developed for each of these thresholds (Figure 9).
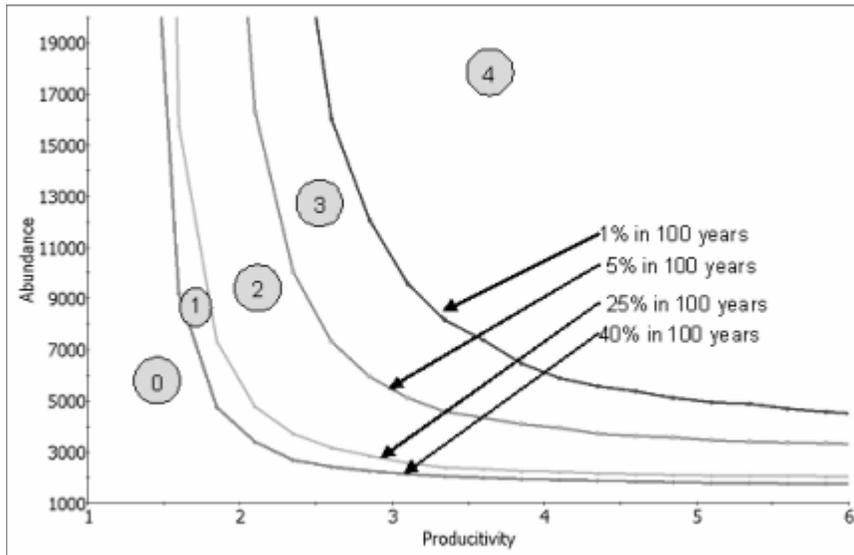
Figure 9 Viability curves showing relationship between risk levels and population persistence categories (example based on Chinook curve). Each of the curves indicates a different risk level. The numbers in circles are the persistence categories associated with each region of the chart (i.e., the area 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).

Benchmark Curves
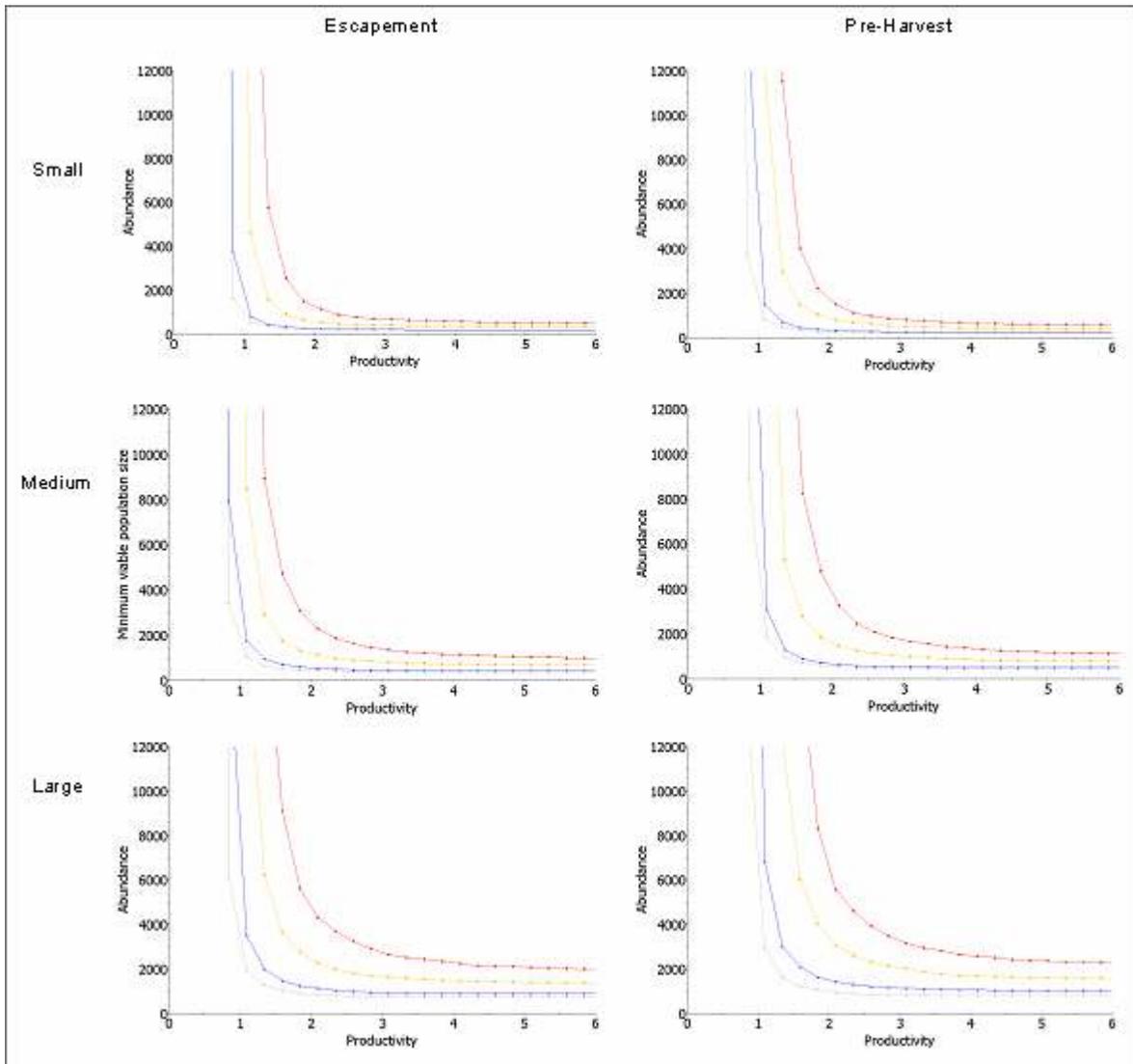
**Figure 10 Steelhead**
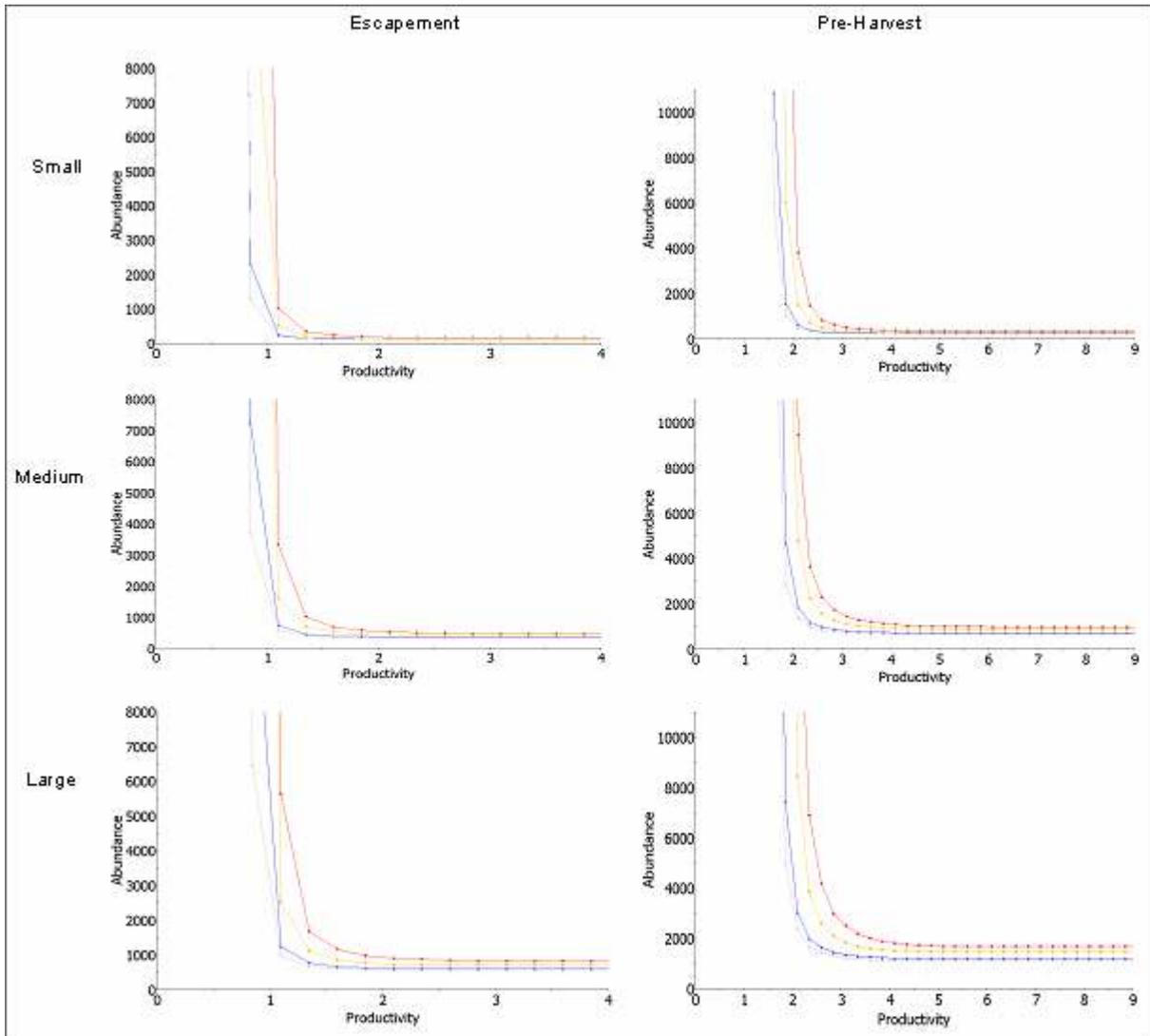
**Figure 11 Fall Chinook**
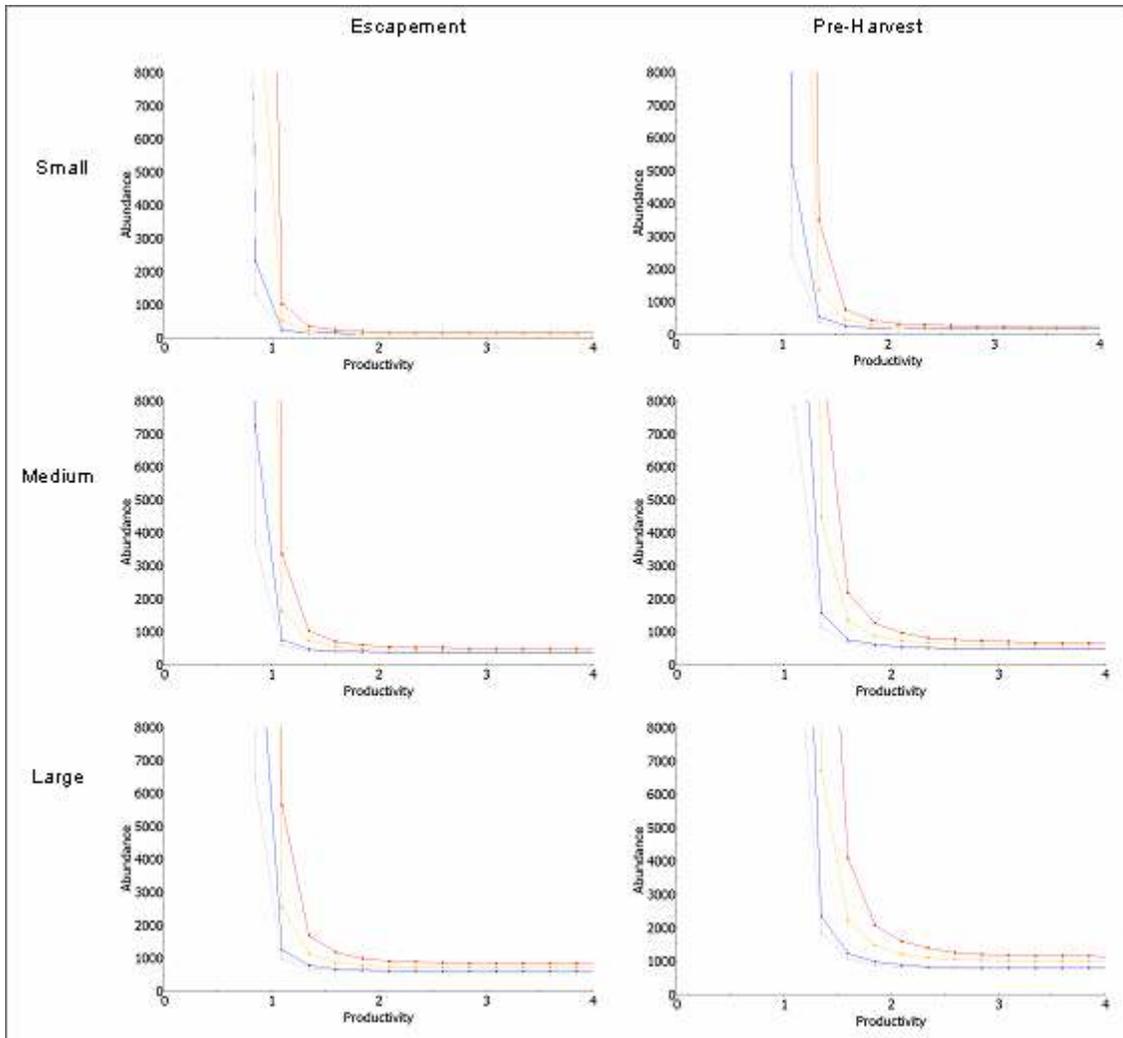
Figure 12 Spring Chinook
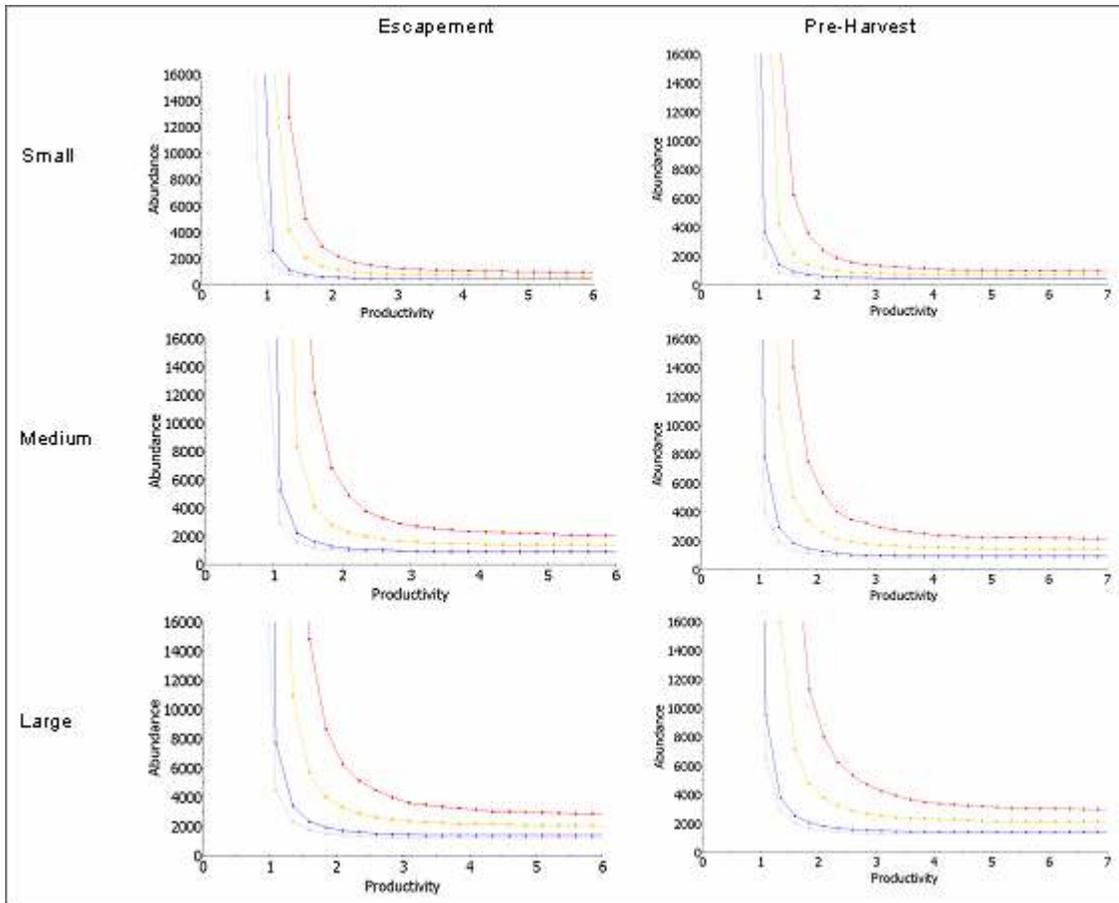
Figure 13 Chum
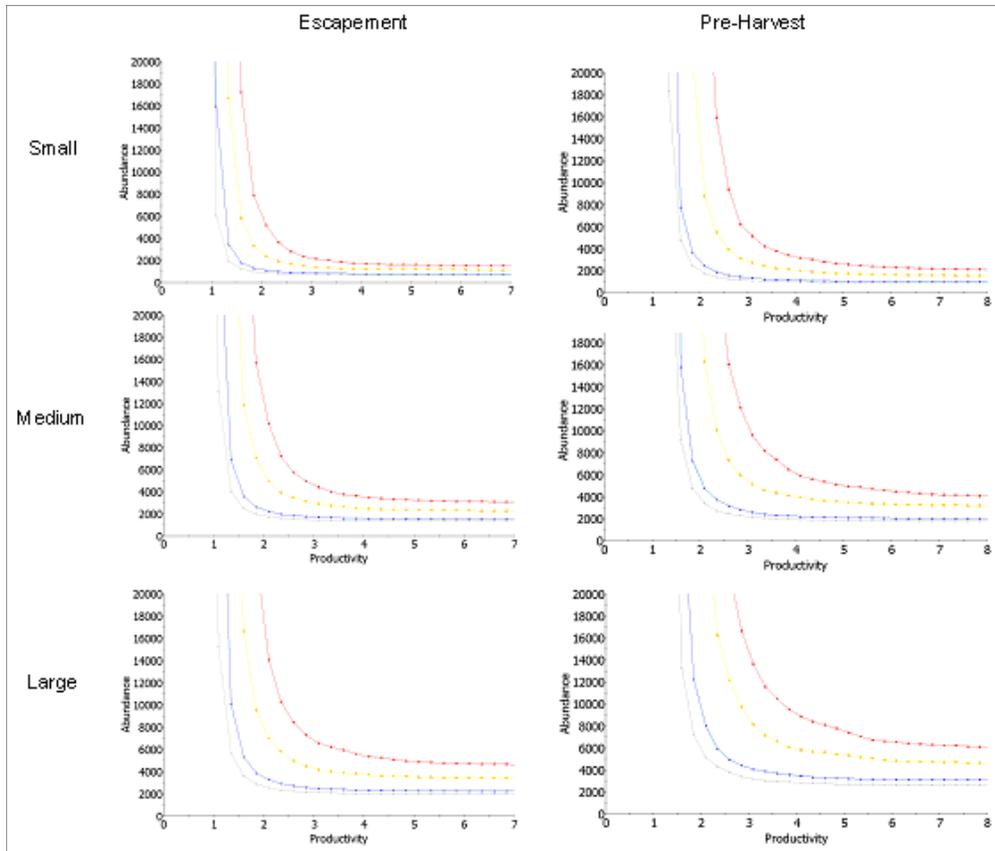
Figure 14 Coho

Measuring Status Relative to the Curve

Measuring the status of a population relative to a viability curve requires determining the appropriate metric and an evaluation of the uncertainty in the estimation of the metric. Uncertainty in the metric must consider both estimation error and measurement error. In this context, estimation error refers to error in calculating a population metric from finite number of data points and measurement error refers to uncertainty in the actual data points. In addition to including uncertainty, an assessment of population status needs to consider patterns of marine survival. All estimates of productivity are based on natural productivity (i.e., Hatchery origin fish contribute to spawners, but not recruits). As noted above, our recruitment is based on both post-harvest (escapement) and pre-harvest estimates.

Choosing a Metric

It makes intuitive sense that in estimating a population's abundance and productivity for comparison to a particular viability curve we should fit a time series of abundance to the same recruitment function used to generate the curve (i.e., "matching recruitment functions"). For example, if a viability curve was generated with a Beverton-Holt recruitment function, it makes sense to fit the available time series to a Beverton-Holt recruitment function. In general, this is the approach we have taken. However, there is some usefulness in looking at other measures of abundance and productivity, since fitting messy data to a curve can lead to potential biases. As example alternative metrics, abundance (pre-harvest) could be measured as a simple arithmetic or geometric mean and productivity could be measured as mean recruits per spawner over either

the entire time series or over only the low spawner abundance years. The IC-TRT has developed a flow chart for deciding what metric to use based on characteristics of the data (ref).

We use a "meanRS" approach as the benchmark method for comparison to the benchmark curves. This method uses the geometric mean recruitment (pre-harvest) over the entire data set as the abundance metric. The productivity estimate is the geometric mean recruits per spawner, where the number of spawners is less than the median number of spawners (i.e., the lowest half of the spawner values). These metrics have a relatively intuitive relationship to the Hockey Stick function used for benchmark viability curves and may not have some of the biases associated with fitting recruitment functions. The average abundance should relate to the "equilibrium" abundance ceiling of the Hockey Stick function (and benchmark curve axis). The geometric mean recruits per spawner for low spawner abundances ($S < S_{median}$) should relate to the productivity parameter of the Hockey Stick, which is a constant recruits per spawner value for all spawner values below the ceiling. The method is less likely to overestimate the intrinsic productivity of the population than curve fitting because there is no extrapolation down to recruits per spawner at one spawner. The meanRS method estimates productivity over the range of spawner values actually observed and since this range is above one spawner may exhibit some density dependence – thus the estimate is likely to be relatively precautionary. The IC-TRT has explored a metric similar to meanRS, though we have taken a somewhat different approach to addressing uncertainty.

Uncertainty

In general, we will need to know how much confidence we have in where a population is relative to the viability curve before making a particular management decision (e.g., delisting). To display how sure we are about where a population is relative to the viability curve, we can draw "probability contours" on the viability curve graph. Management thresholds, which would serve as a basis for management decisions, can be set based on the probability that a population is above the curve and visually evaluated by determining if the appropriate probability contour is above the curve (Figure 15). For example, if we want to be 95% sure that a population is above the viability curve, we would examine the 95% contour for a population. In examining current population status, we present a continuous color probability surface map overlaid on the viability curve with contours drawn at 50%, and 95% probability. The technical challenge is appropriately drawing these probability contours. We have taken into consideration both estimation error and measurement error. It is important to note that large measurement error can make the contours very large, increasing the chance that a population will "fail" the viability test. The size of a probability contour can generally be reduced by collecting better quality data.
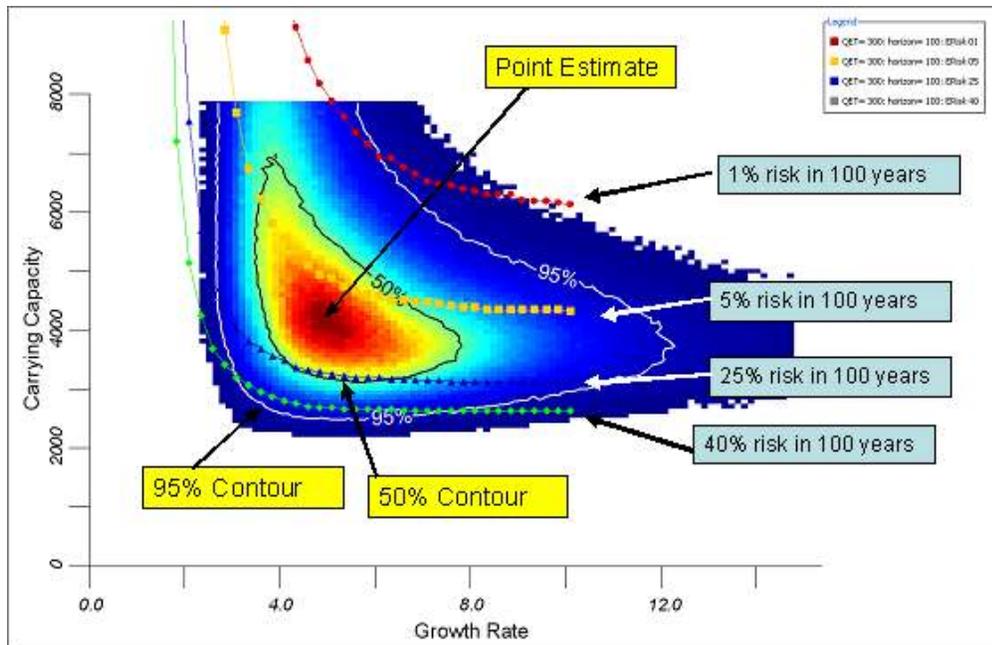
Figure 15 Example of current status contours combined with viability curves. In this example, the point estimate of the population indicates a persistence category of 2 (i.e., between 25% and 5% viability curves). To ensure at least a 50% chance that the population exceeds a given viability curve we 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). To ensure at least a 95% chance that the population exceeds a given viability curve we would examine the 95% contour, which in this example suggests the population is in persistence category 0 (the bottom of the 95% contour is below the 40% viability curve).

Estimation Error

Even if all the values in a time series (e.g., spawner abundance) were measured with perfect accuracy, there would still be uncertainty associated with our estimates of abundance and productivity because we are using data from relatively short time series (e.g., 20 years). We refer to this uncertainty as estimation error. In fitting recruitment curves, we estimate parameters using a Bayesian approach, which yields probability distributions for the model parameters (see Figure 16 and Figure 17). We can look at the joint posterior probability distributions for the productivity and equilibrium abundance estimate to obtain a 2-dimensional look at uncertainty for comparison to the viability curve (see Figure 18) when fitting recruitment curves.

To evaluate and display estimation error for the meanRS method, we have explored both bootstrap and parametric statistic approaches. For the bootstrap method, we generate hundreds of thousands of new datasets the same length as the original data sets by re-sampling the annual abundance and recruitment pairs from the original data set with replacement. We then calculate the test statistics (i.e., geomean recruits and geomean R/S for $S < S_{median}$) and show the resulting estimates as a two dimensional surface plot (see Figure 19). With the parametric statistic approach, we estimate the standard error about the mean recruits and R/S (see Figure 19). We assume that recruits and R/S are lognormally distributed so the resulting error bars are asymmetrical on the natural scale. Two standard errors is approximately equal to a 95% confidence interval (assuming the lognormal distribution function is valid). We use the standard error rather than the standard deviation because the parameters on the axis of the viability curve are the mean behavior of the population. Uncertainty about the mean is captured by the standard

error, whereas the standard deviation describes the uncertainty about the individual data points. If the lognormal assumption is valid and the sample size is large enough, the bootstrap and standard error approaches should yield similar results. For the benchmark metric, we use the bootstrap method because it produces a clearer sense of the parameter estimates in two dimensions and its interpretation does not depend on the assumption of any particular distribution of the data.
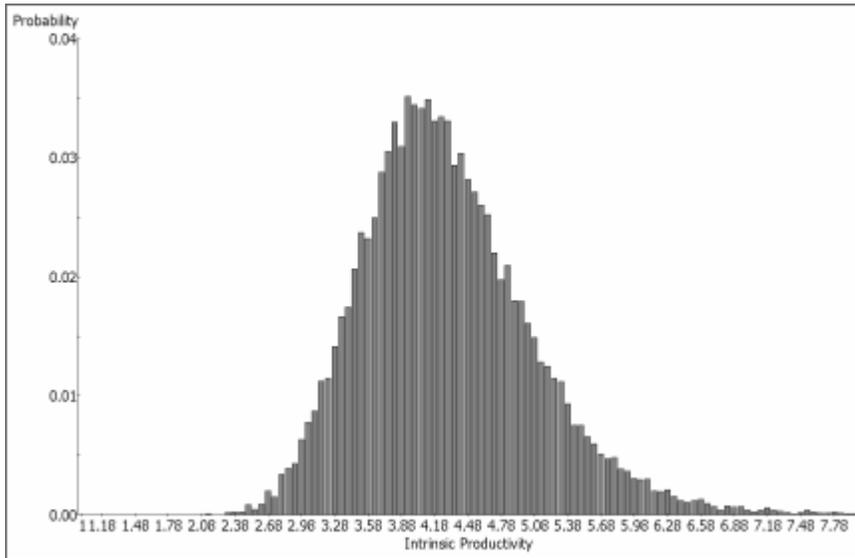
Figure 16 Posterior probability distribution of productivity for the Beverton-Holt model applied to Sandy River spring Chinook. (Note that this data set has a higher correspondence to the hypothesized recruitment function than most other Oregon WLC populations – see appendix D for more typical results.)
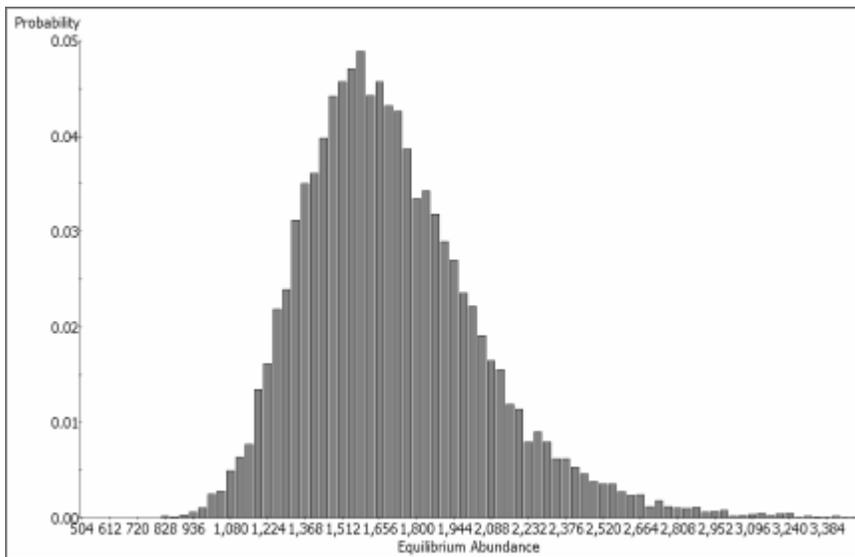
Figure 17 Posterior probability distribution of equilibrium abundance for the Beverton-Holt model applied to Sandy River spring Chinook. (Note that this data set has a higher correspondence to the hypothesized recruitment function than most other Oregon WLC populations – see appendix D for more typical results.)
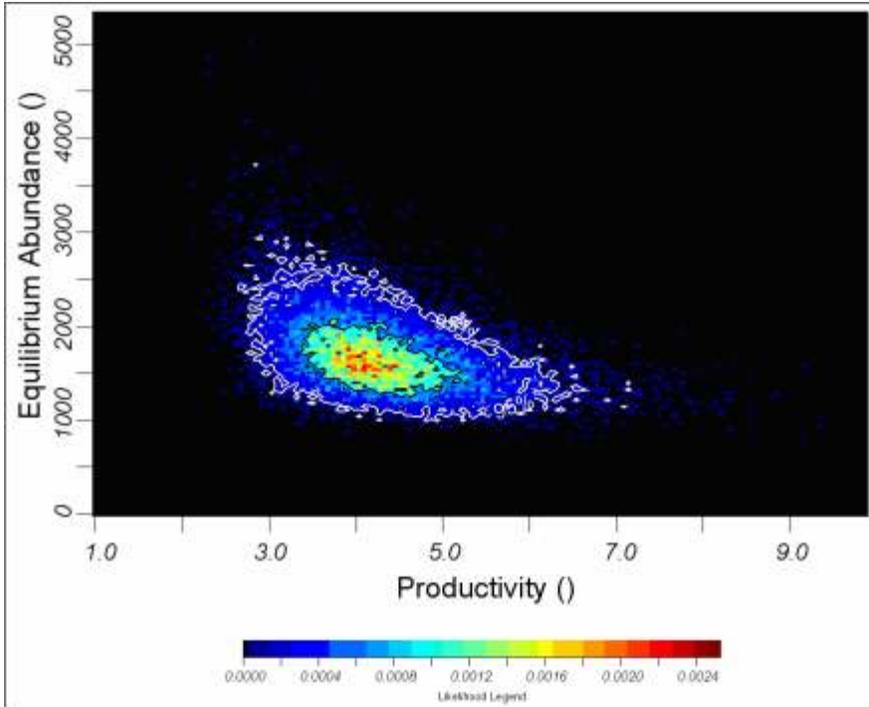
Figure 18 Joint posterior probability distribution of productivity and equilibrium abundance for the Beverton-Holt model applied to Sandy River spring Chinook. (Note that this data set has a higher correspondence to the hypothesized recruitment function than most other Oregon WLC populations – see appendix D for more typical results.)
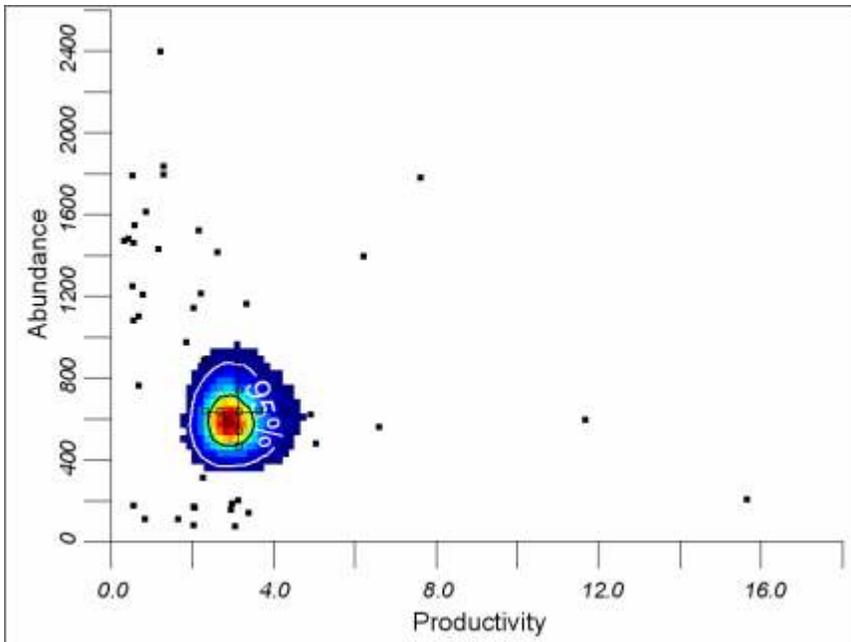
**Figure 19 MeanRS method applied to Sandy spring Chinook. The black dots are the actual data points from the Sandy population. The color contour shows the joint distributions of the bootstrap means. The black lines show two standard errors about the means, with an open square at one standard error.**

Measurement Error

All parameters used to calculate a population's abundance and productivity (e.g., spawner counts) are estimated with uncertainty, which is often considerable. It is important to include this uncertainty in estimating where a population is relative to the viability curve. Although the issue has been discussed by the TRTs, this measurement error has generally not been explicitly included in viability criteria. We used a Monte Carlo approach to including measurement error in the probability contours. The basic approach was to first estimate probability distributions describing the likely values of the input parameters. We then did many random draws from these distributions, creating hundreds or thousands of "plausible data sets". We then estimated the abundance and productivity from each of the plausible data sets (via curve fitting or bootstrapping the meanRS metrics) and treated the resulting distribution of abundance and productivity estimates as part of the probability contour for comparison to the viability curve.

The input parameters for which we estimated distributions are:

- Spawner abundance

- Fraction of hatchery origin spawners

- Relative reproductive success of hatchery origin spawners

- Catch (the number of additional natural origin spawners that would have returned if there had not been a harvest – necessary to estimate pre-harvest recruits)

- Age distribution

Adding measurement error to the assessment can greatly add to our evaluation of the uncertainty in the parameter estimates (Figure 20). Estimating the uncertainty around all of these parameters is challenging and as a first approximation for a current population status evaluation, we have simply used professional judgment to describe the measurement error distributions (Table 9 and Table 10). An analysis of some of the uncertainty surrounding hatchery fraction estimates is presented in Appendix G.

Adding measurement error can result not just in an expansion of the region of uncertainty around the point estimate, but an actual shift in mean value for the prediction. This phenomenon can be observed in many of the meanRS graphs by comparing the red area of the contour plot (most likely values) to the geometric mean with standard error bars (Figure 23-Figure 46). The point estimate of a non-linear model *without* variability in the input parameters is not necessarily the same as the point estimate of the same non-linear model *with* variability in the input parameters (i.e., Jensen's Inequality [ref]). The meanRS approach is based on a geometric mean, which is a non-linear model and Jensen's Inequality applies. This potential shift in the mean value is another reason it is important to consider all sources of input error and not rely on the simple point estimates.
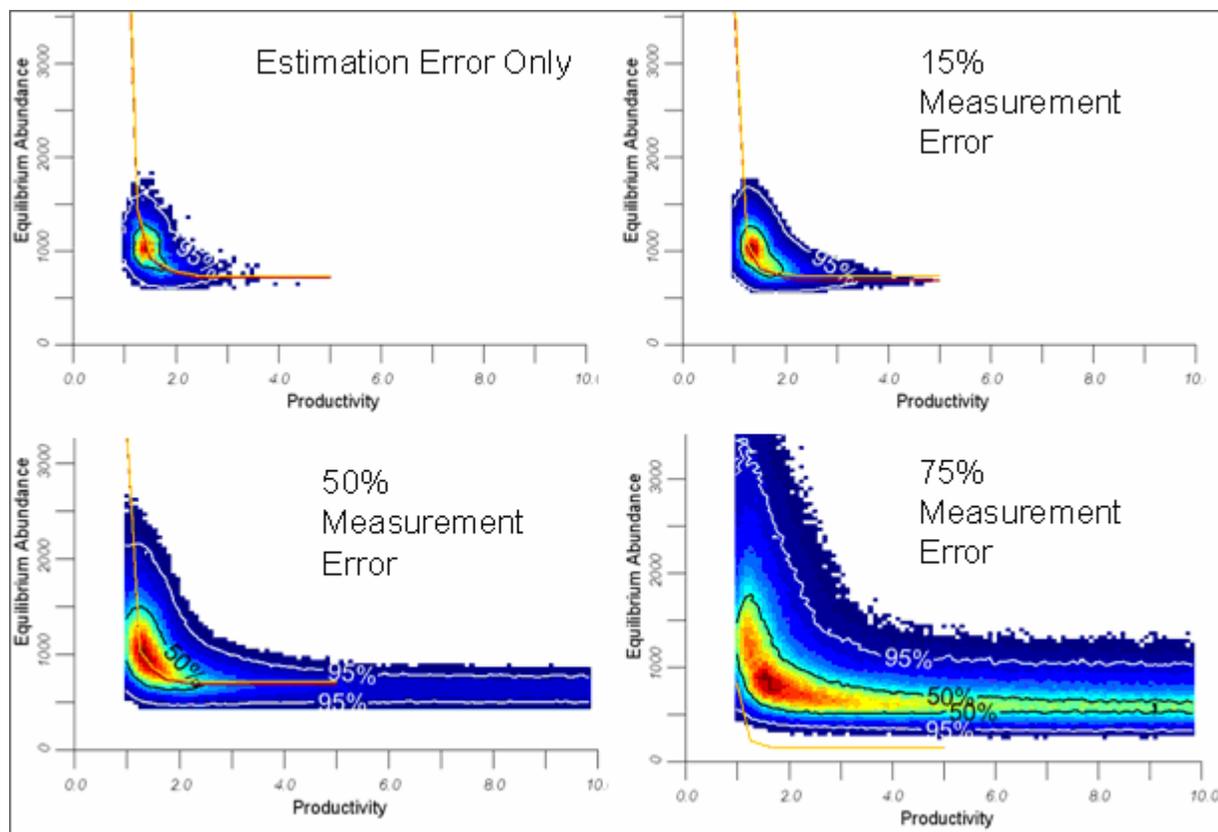
Figure 20 Example of including measurement error in assessment of productivity and equilibrium abundance. Based on Hockey Stick fit for Sandy spring Chinook population. The measurement error assumed an independent uniform distribution for all the parameters listed above.

Table 9 Estimates of measurement errors associated with different types of data and collection methods for Oregon WLC salmon and steelhead species. Measurement error is assumed to follow a uniform distribution with a range plus or minus a percent of the point estimate. Measurement error estimates are rough approximations provided by Mark Chilcote (see Appendix F). Note that age structure cannot be modeled with a simple uniform age distribution of error because all the age classes must add to one. Consequently, we use a multinomial sampling approach that approximates the uniform distributions in this table.

Data Element	Species	Data collection method		
		Spawning Surveys	Dam Passage Counts	Trap and Handle
Spawner Abundance	Steelhead	±70%	±20%	±5%
	Chinook	±40%	±20%	±5%
	Coho	±50%	±20%	±5%
Hatchery Proportion	Steelhead	±60%	±20%	±5%
	Spring Chinook	±40%	±20%	±5%
	Fall Chinook	±70%	±50%	±40%
	Coho	±40%	±20%	±20%
Age Composition	Steelhead	±40%	±40%	±40%
	Chinook	±40%	±40%	±40%
	Coho	±5%	±5%	±5%
Fishery Impact	Steelhead	±40%	±40%	±40%
	Spring Chinook	±30%	±30%	±30%
	Fall Chinook	±40%	±40%	±40%
	Coho	±50%	±50%	±50%

Table 10 Estimates of measurement error associated with different data types for specific Oregon WLC salmon and steelhead populations. The measurement error estimates for each data type and collection method by species are shown in Table 9. The collection methods listed are the current methods for each population. Measurement error is assumed to follow a uniform distribution with a range plus or minus a percent of the point estimate. The table shows only those populations for which time series data were available for comparison to viability curves. For the majority of populations, we do not have sufficient data for any quantitative comparison to viability curves. Populations denoted with "*" have some data but the data are not sufficient for productivity estimates. Note that the Hood River steelhead age structure error estimates are lower than those suggested by Table 9 because the unique method used for those populations is considered relatively precise.

ESU	Life History	Population	Data Collection Method	Spawner Abundance	Hatchery Proportion	Age Composition	Fishery Impact
Chinook	Spring	Sandy River	Spawning Surveys	±40%	±40%	±40%	±30%
Lower Columbia Coho		Big Creek*	Spawning Surveys	±50%	±40%	±5%	±50%
		Clackamas	Dam Passage Counts	±20%	±20%	±5%	±50%
		Clatskanie*	Spawning Surveys	±50%	±40%	±5%	±50%
		Sandy River	Dam Passage Counts	±20%	±20%	±5%	±50%
		Scappoose River*	Spawning Surveys	±50%	±40%	±5%	±50%
		Youngs Bay*	Spawning Surveys	±50%	±40%	±5%	±50%
Lower Columbia Steelhead	Summer	Hood River*	Dam Passage Counts	±20%	±20%	±10%	±40%
	Winter	Clackamas	Dam Passage Counts	±20%	±20%	±40%	±40%
		Hood River*	Dam Passage Counts	±20%	±20%	±10%	±40%
		Sandy River	Dam Passage Counts	±20%	±20%	±40%	±40%
Upper Willamette Chinook	Spring	Calapooia*	Spawning Surveys	±40%	±40%	±40%	±30%
		Clackamas	Dam Passage Counts	±20%	±20%	±40%	±30%
		McKenzie	Spawning Surveys (partial dam count)	±40%	±40%	±40%	±30%
		Molalla*	Spawning Surveys	±40%	±40%	±40%	±30%
Upper Willamette Steelhead	Winter	Calapooia	Spawning Surveys	±70%	±60%	±40%	±40%
		Molalla	Spawning Surveys	±70%	±60%	±40%	±40%
		N. Santiam	Spawning Surveys	±70%	±60%	±40%	±40%
		S. Santiam (Lower)	Spawning Surveys	±70%	±60%	±40%	±40%
		S. Santiam (Upper)	Trap and Handle	±5%	±5%	±40%	±40%

We are not recommending these measurement error distributions above as benchmark values – the amount of measurement error will obviously be data dependent and should be evaluated at the time of any population assessment. The values in these tables are only initial estimates for current population evaluations. It is important to note that the amount of error assumed for

currently available data sets is often quite high. For example, Figure 21 illustrates potential variation in spawner time series.
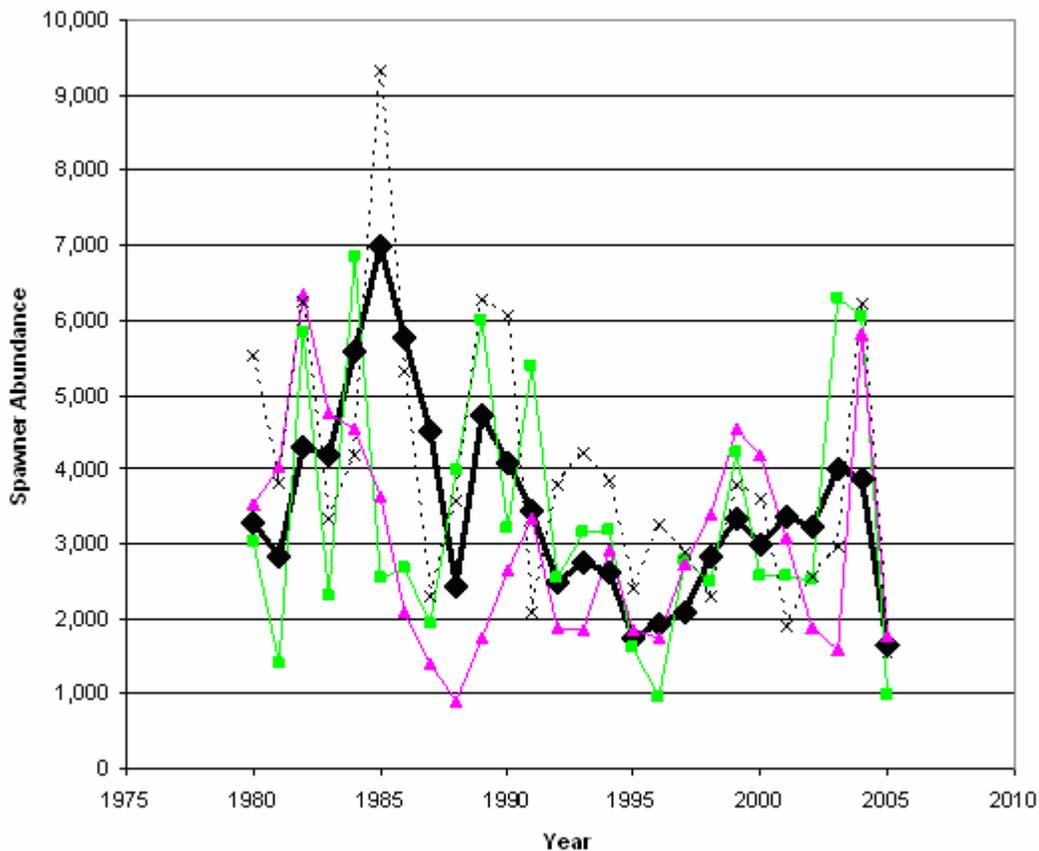
Figure 21 Potential time series for North Santiam steelhead. The black line is the point estimate time series. The other curves show equally plausible time series for this population based on random draws from a uniform distribution ± 70% of the point estimate spawner count, which the error rate associated with the survey method used for steelhead in the Upper Willamette.

Hatchery Production

In keeping with our definition of a viable population, we are assessing the natural productivity and abundance of a population relative to the viability curve. Hatchery origin fish in the wild can contribute to spawners, but they do not count as recruits. When hatchery origin and natural origin fish spawn together on the spawning ground, we need to make some estimate of the reproductive success of hatchery origin fish relative to natural origin fish. In some cases, hatchery origin spawners have been shown to have a lower reproductive success than natural origin spawners, presumably because of domestication effects. However, there is little empirical data on the relative reproductive success of hatchery origin spawners; the success is highly population specific and it is expected to change over time in response to evolutionary processes. Because of this uncertainty, we have taken the precautionary approach in our benchmark metric of assuming that hatchery-origin fish have the same reproductive success as natural origin fish for our current population evaluations. In the future, this should be evaluated on a case-by-case basis. Note that the assumption of equal reproductive success is a precautionary assumption in the context of

estimating productivity, but it may not be a precautionary assumption in the context of evaluating population diversity (see diversity section).

Hatchery fish can have a complex impact on productivity estimates. In those studies where the reproductive success of naturally spawning hatchery fish has been evaluated, the egg to smolt survival of hatchery offspring under natural conditions can be sufficient to create a large number of juveniles. These hatchery offspring have the potential to compete for food and space with offspring of wild spawners and thereby reduce the overall survival of the wild population. Therefore, even if there is no genetic interaction between the hatchery and wild spawners, the impact of hatchery fish on the overall natural recruitment may be considerable. From a strictly numerical standpoint, hatchery offspring may reduce (via competition) the fraction of wild offspring that survive to become smolts. However, it is also thought that the conversion of naturally produced smolts to returning adults is higher for offspring of wild fish than for offspring of hatchery fish. Effectively, then, offspring of hatchery fish "tie up" the limited freshwater habitat and then, once they reach the ocean, survive poorly. The net result is to reduce the efficiency with which a basin produces fish. Therefore, any adjustments to standardize for the effect of naturally spawning hatchery spawners must incorporate both the issue of differential reproductive success and density dependent effects on juvenile rearing and survival. Simply adjusting the number of hatchery spawners downward prior to analysis so they are expressed in terms of wild fish equivalent units has the potential to confuse the observation of the true density dependent recruitment performance of the combined population of wild and hatchery spawners. This is another reason we chose not to incorporate any ad hoc adjustments to the reproductive effectiveness of hatchery fish in our analyses.

Marine Survival

The viability curve is constructed to represent long-term "average" conditions (including the existence of any marine survival generated autocorrelation). Any particular short-term data set may not reflect the expected long-term average behavior of the population. In particular, marine survival patterns, which are expect to change on decadal scale dynamics, can greatly confuse an assessment of a population's long-term average behavior. It is therefore necessary to "standardize" a particular short-term time series to long-term average marine survival conditions. This can be done by standardizing the estimate of recruits before fitting the recruitment function:

$$R_t = R_t * \frac{1}{\phi_t} \quad , \qquad\qquad\qquad\qquad \text{eq. 3}$$

where ϕ_t is a fraction indicating how much marine survival in any given recruitment cohort deviates from the long average marine survival (i.e., ObservedMarineSurvival$_t$ / AverageMarineSurvival). Estimating ϕ is extremely challenging and often surrogates are required. Surrogates suggested for WLC populations include the Oregon production index (OPI), the Pacific decadal oscillation (PDO), SNEG (a snow index developed by Chilcote, ref) (Figure 22), near shore sea surface temperature (SST), and estimates of marine survival from hatchery index stocks. Since ϕ is a ratio, the point estimate of the ratio of any of the indices (observedAnnual/average) will be the point estimate of ϕ. However, the uncertainty around how well any of the indices relate to marine survival will vary greatly. Because of the advantages in reducing the size of the abundance and productivity probability contour, we recommend that monitoring programs be designed to provide good estimates of marine survival. In particular, having at least one population in each stratum in which marine survival is directly measured via

smolt counting should significantly improve the estimates. Because none of the marine survival indices we have considered so far seem very precise, we did not use any marine survival standardization for the analysis of current status. Understanding marine survival rates should be a high research priority because it has a potentially large impact on viability assessment.
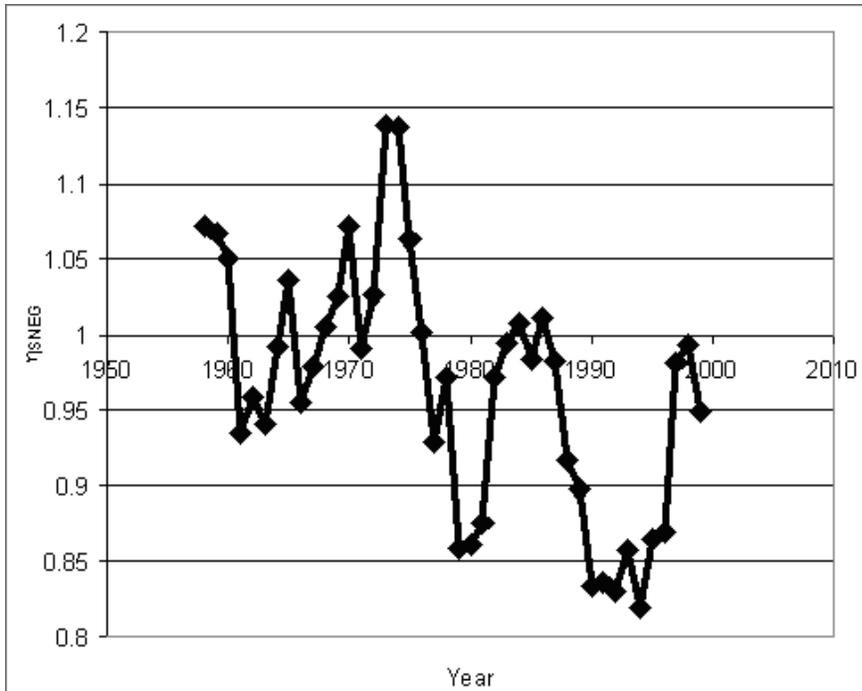

Figure 22 Annual SNEG cascade snow index relative to long term average.

Current Population Status

This section of the report does not provide a complete status evaluation of the WLC populations. Rather, this section provides an application of the benchmark curves and metrics to current Oregon population data as part of an evaluation of both the method and current population status. A complete population status evaluation for Oregon LCR coho is provided in Part 3 of this report. Evaluation of these graphs is a component of that evaluation but, as noted elsewhere, a complete evaluation should consider all available information at the time of the evaluation, not just the limited subset of metrics that can be defined *a priori*. The following curves are based on the parameters defined above and on the data described in Appendix B. The standard error bars do not include measurement error. As noted in the introduction to this report, analysis of Washington populations will likely occur in the future.
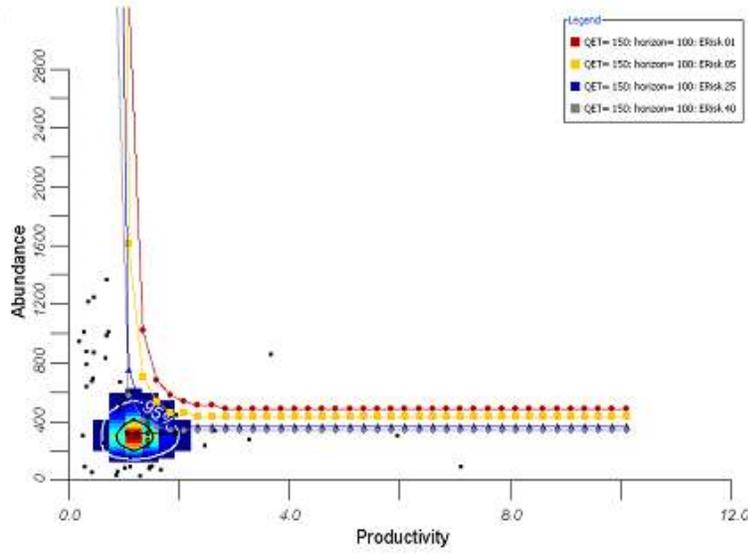
Figure 23 Sandy spring Chinook escapement.
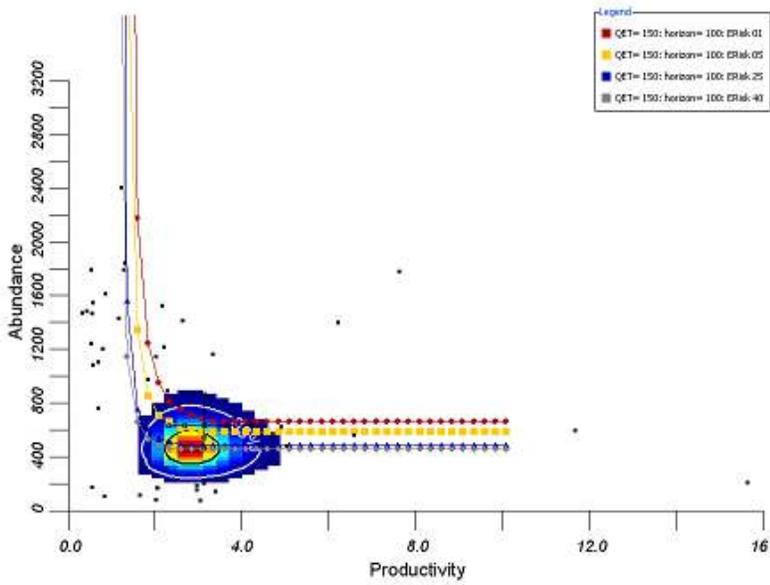
Figure 24 Sandy Spring chinook pre-harvest.
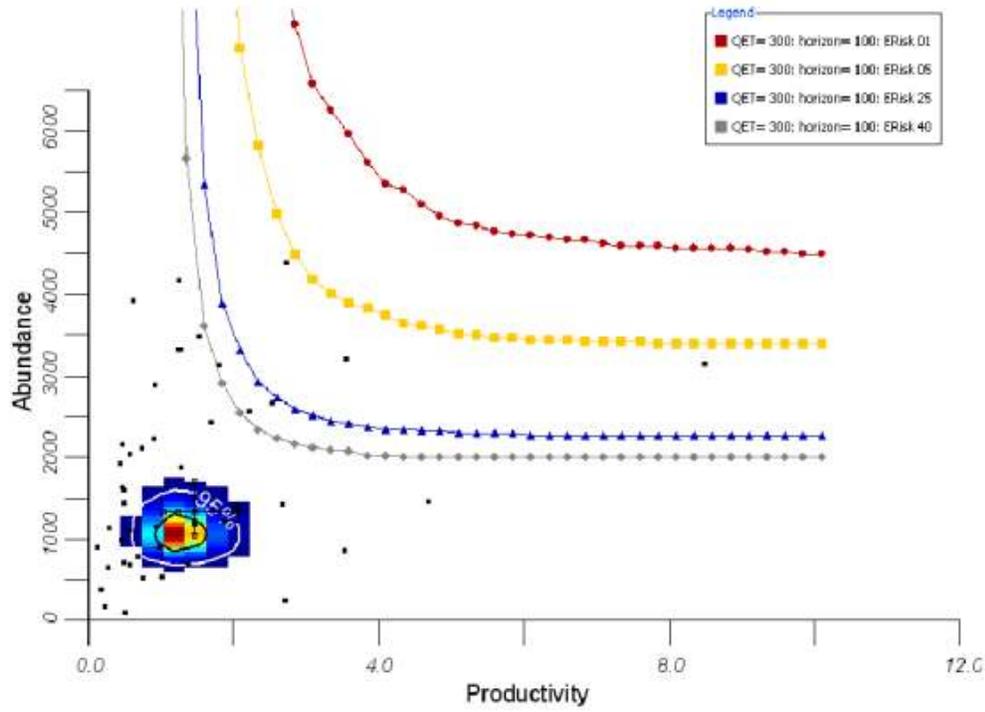
Figure 25 Clackamas coho escapement.
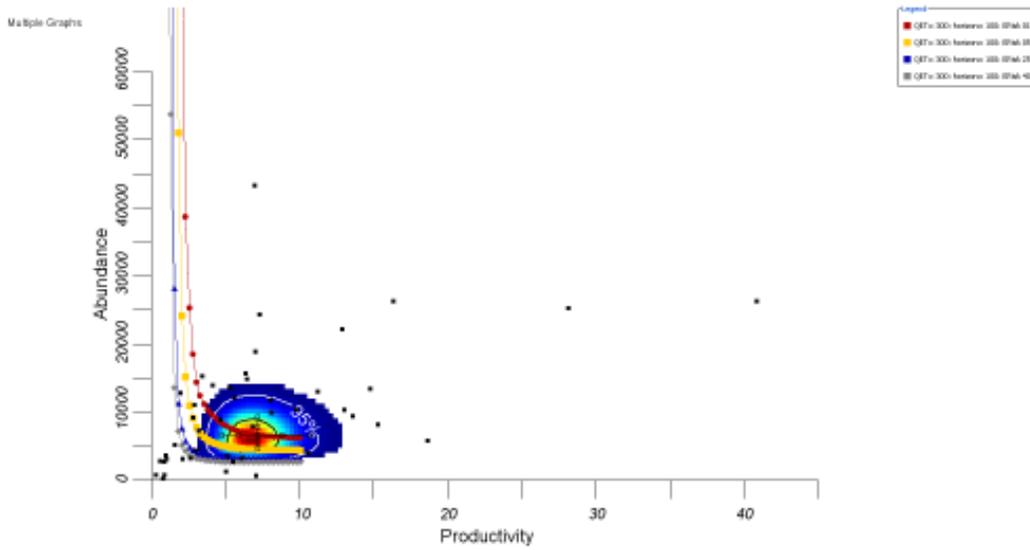
Figure 26 Clackamas coho pre-harvest.
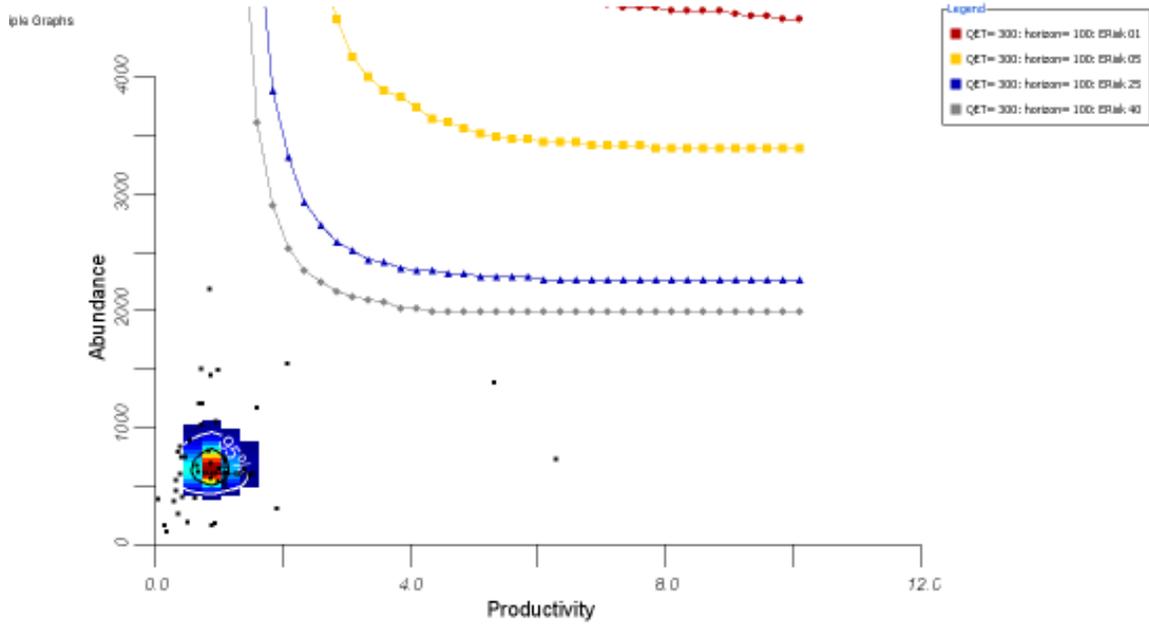
Figure 27 Sandy coho escapement.
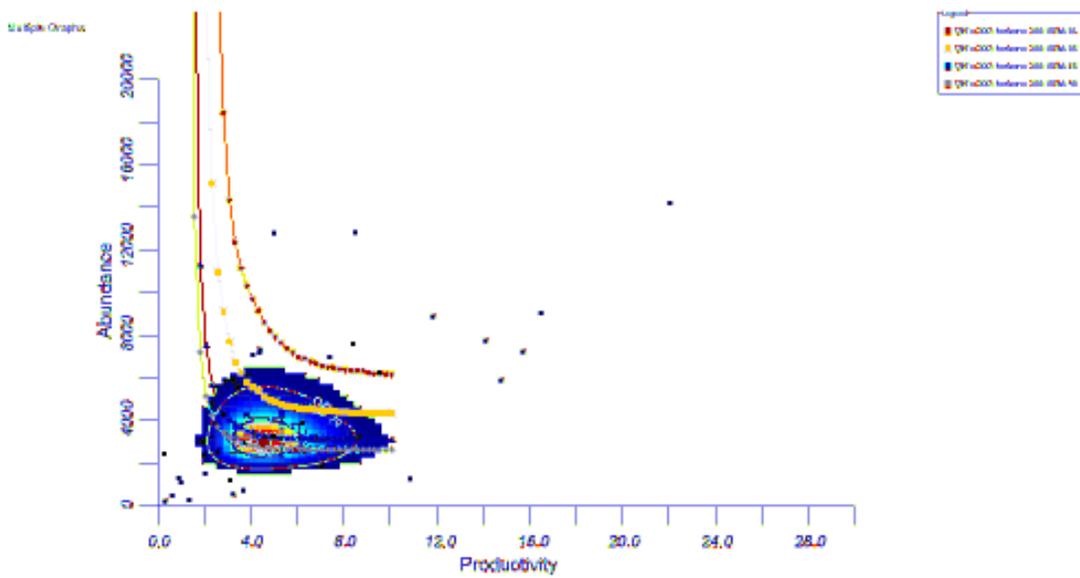
Figure 28 Sandy coho pre-harvest.
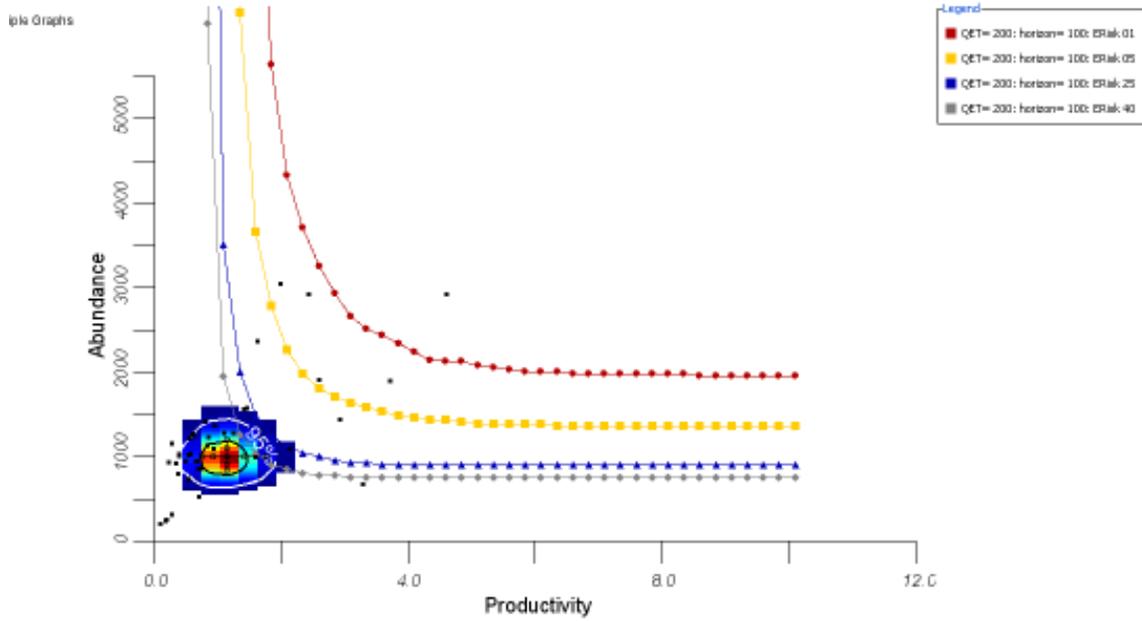
Figure 29 Clackamas steelhead escapement.
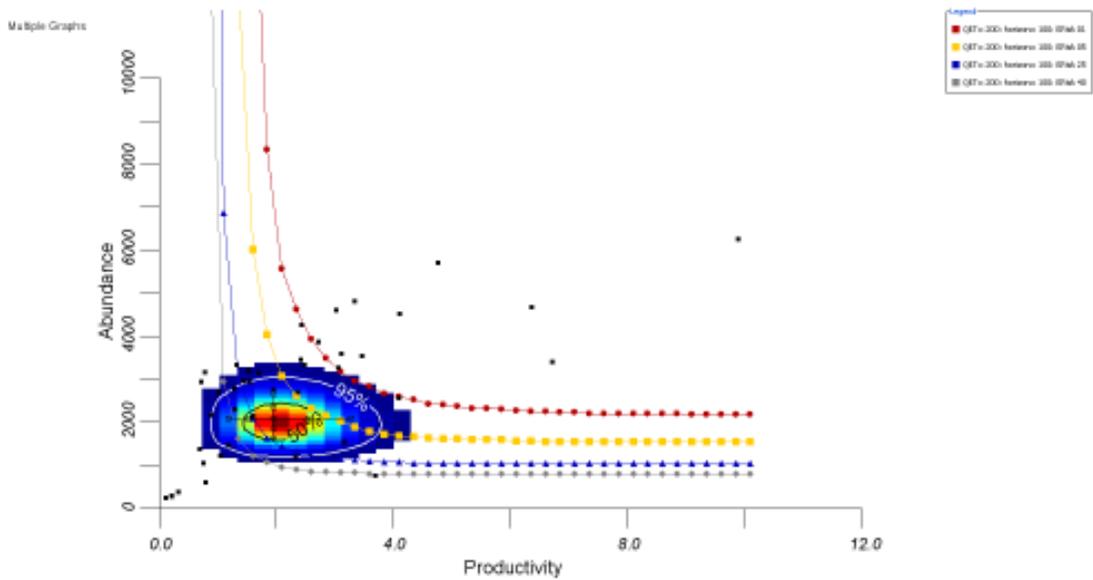
Figure 30 Clackamas steelhead pre-harvest.
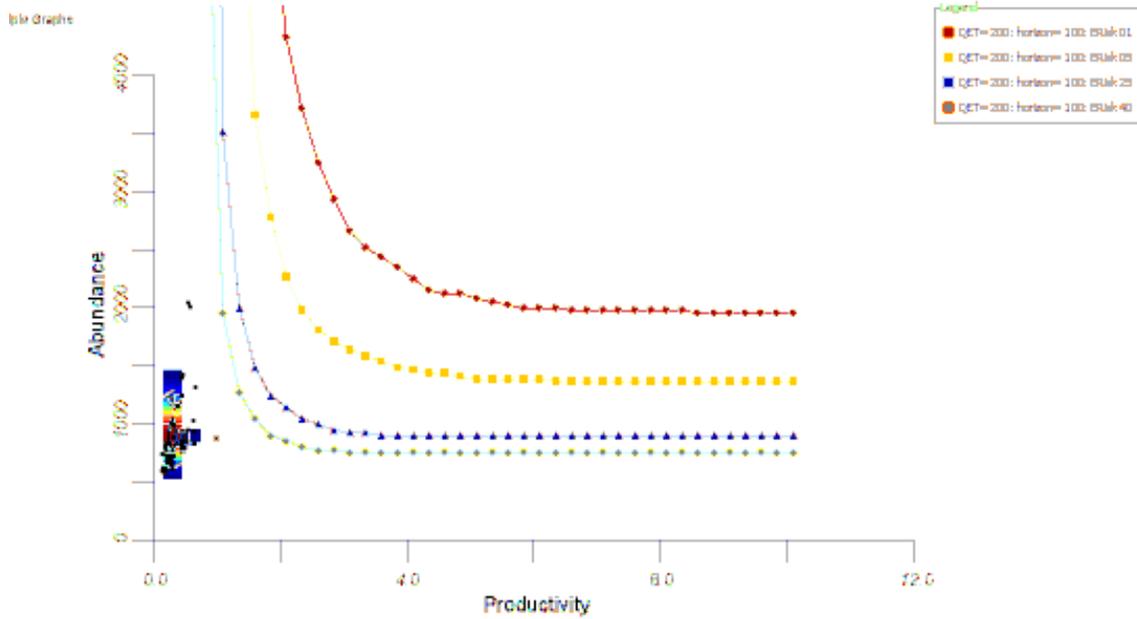
Figure 31 Sandy steelhead escapement.
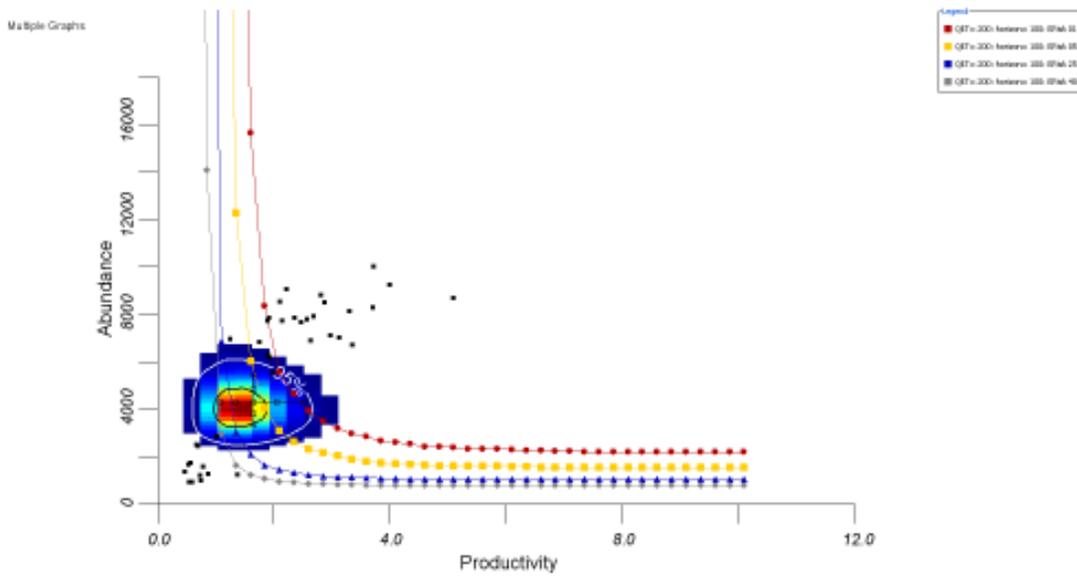
Figure 32 Sandy steelhead pre-harvest.
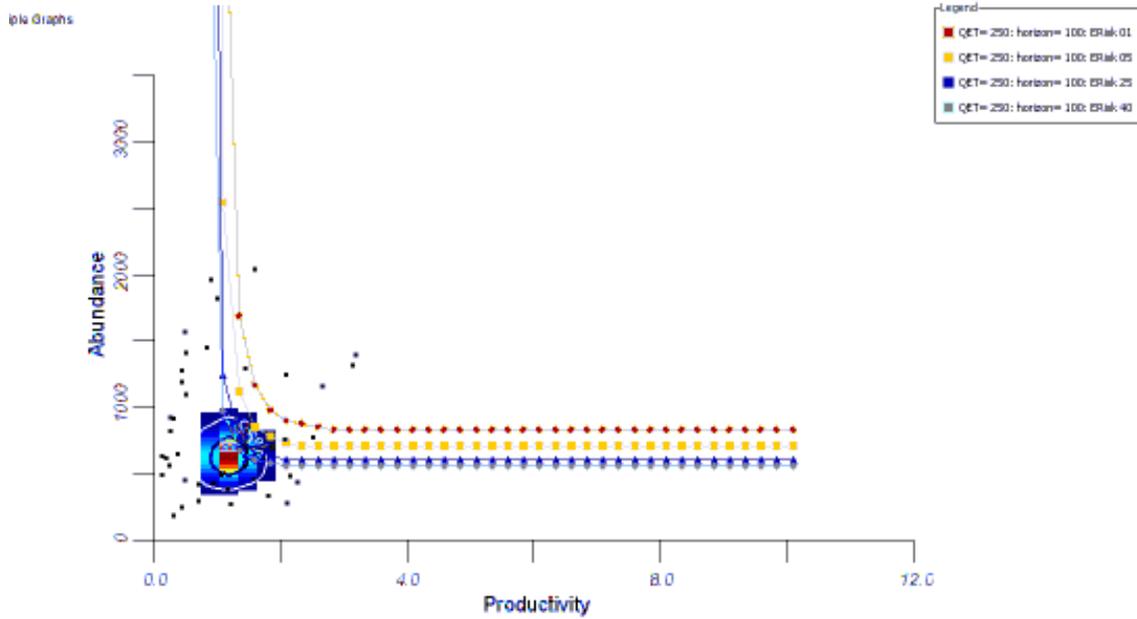
Figure 33 Clackamas spring Chinook escapement.
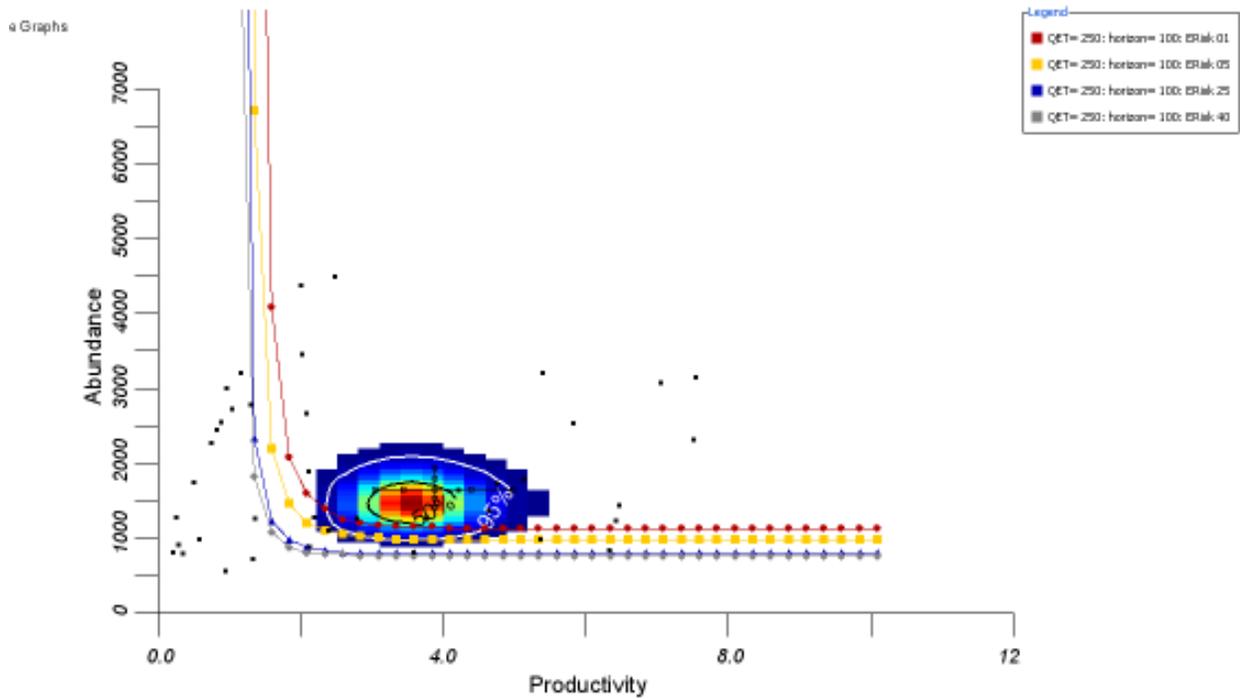
Figure 34 Clackamas spring Chinook pre-harvest.
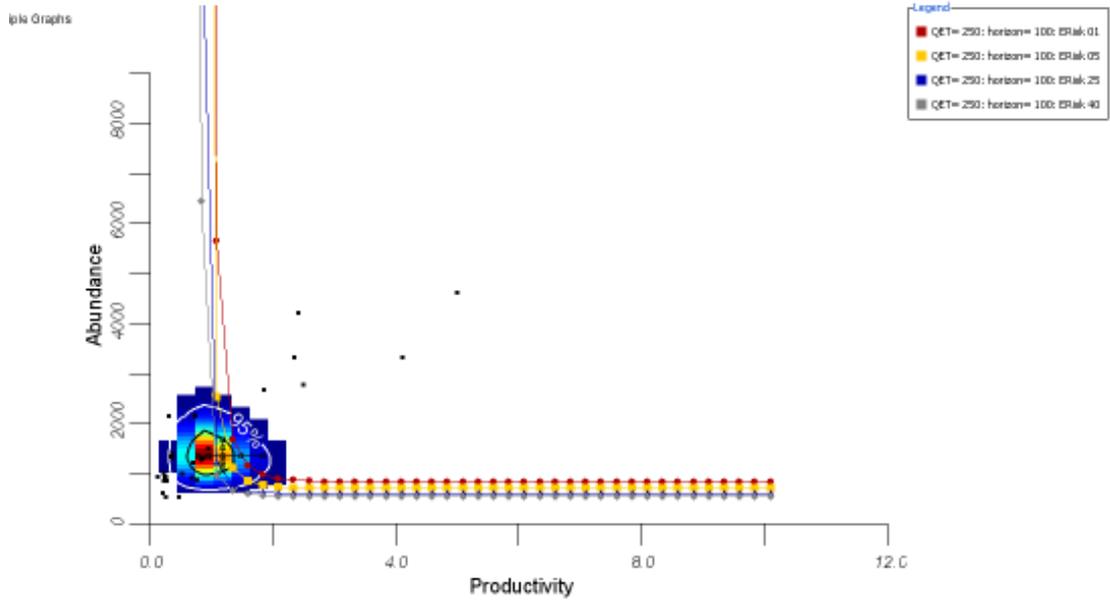
Figure 35 McKenzie spring Chinook escapement.
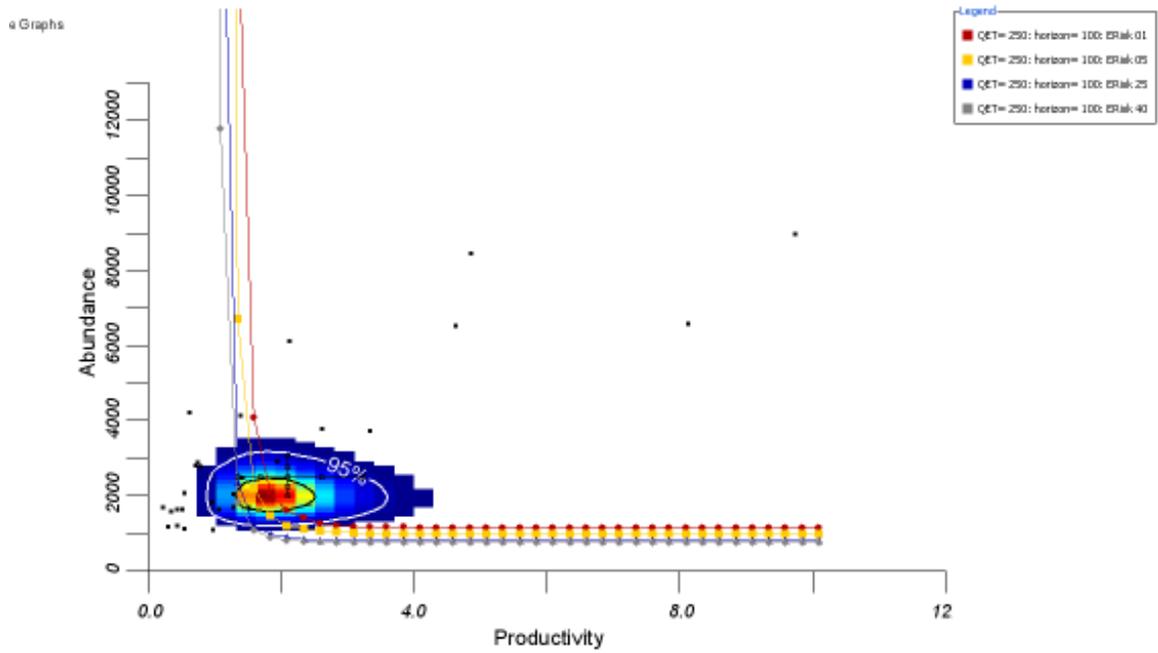
Figure 36 McKenzie spring chinook pre-harvest.
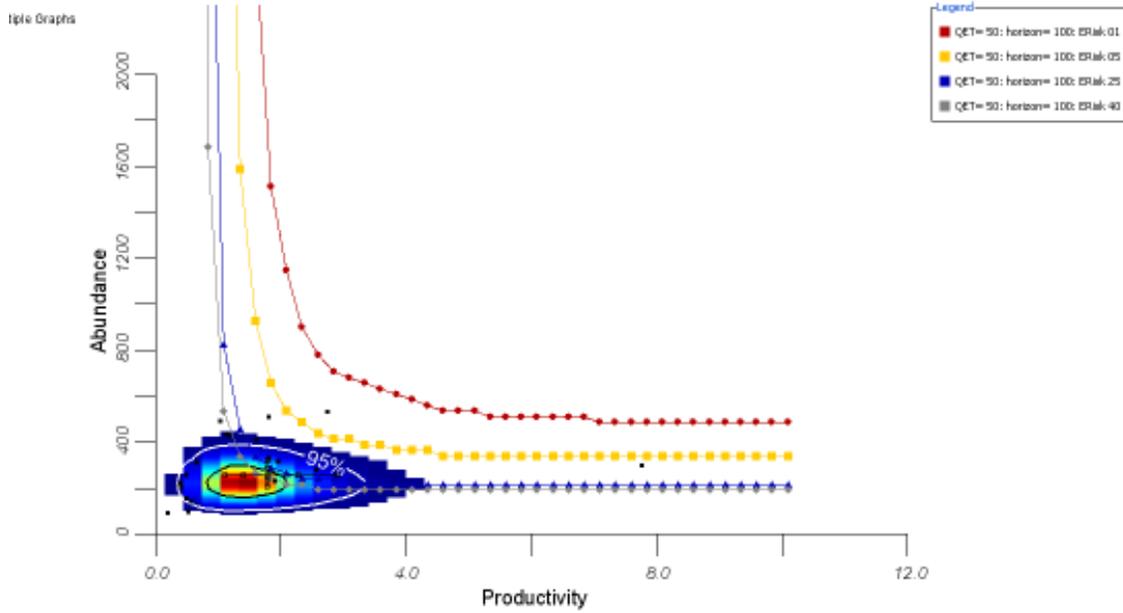
Figure 37 Calapooia steelhead escapement.
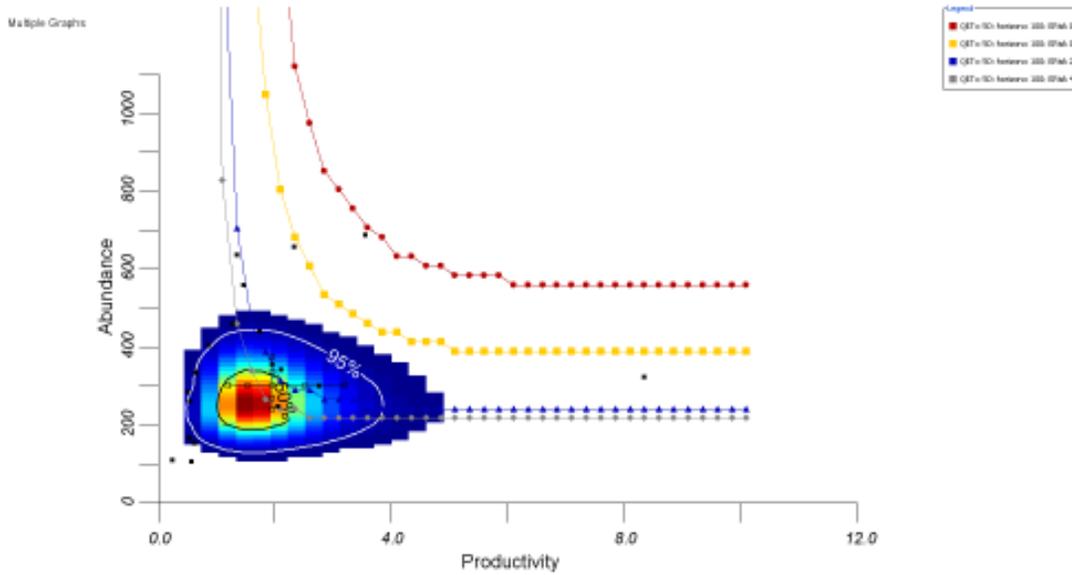
Figure 38 Calapooia steelhead pre-harvest.
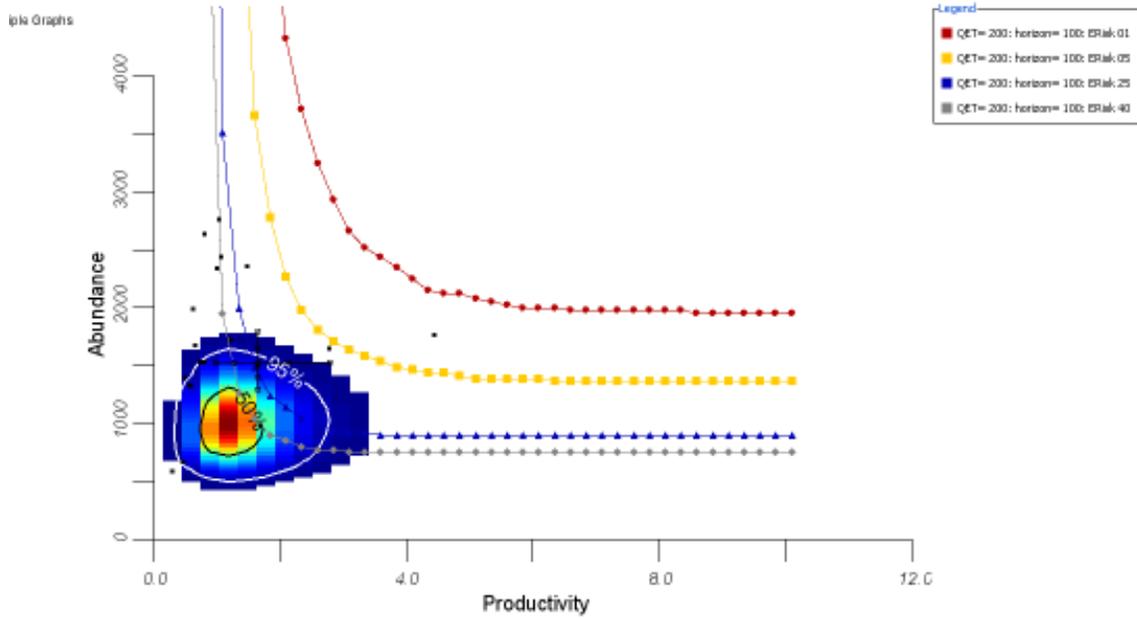
Figure 39 Molalla steelhead escapement.
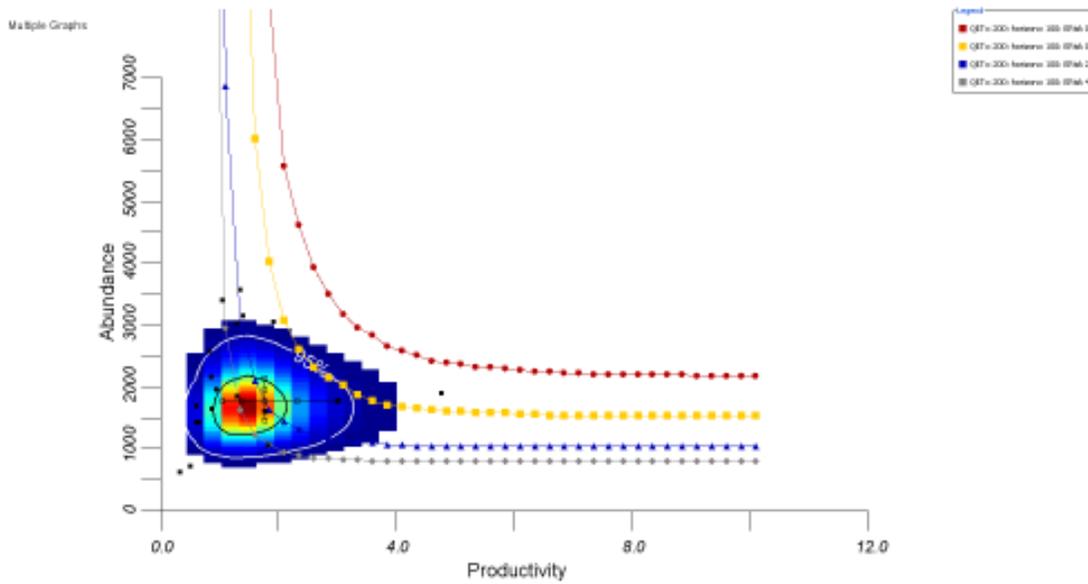
Figure 40 Molalla steelhead pre-harvest.
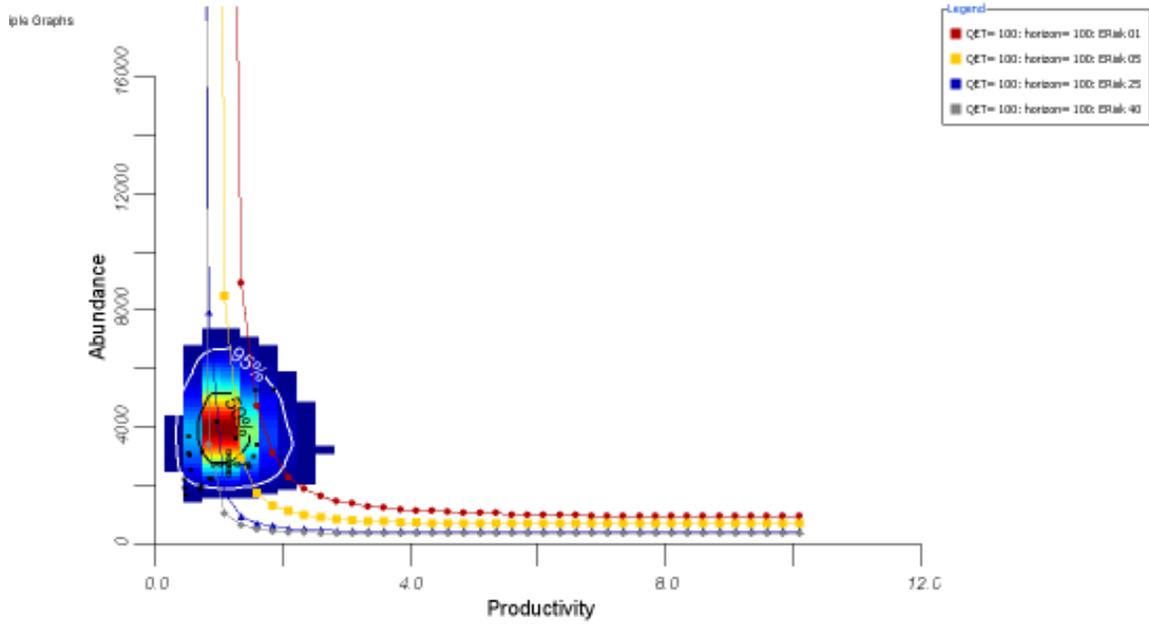
Figure 41 North Santiam steelhead escapement.
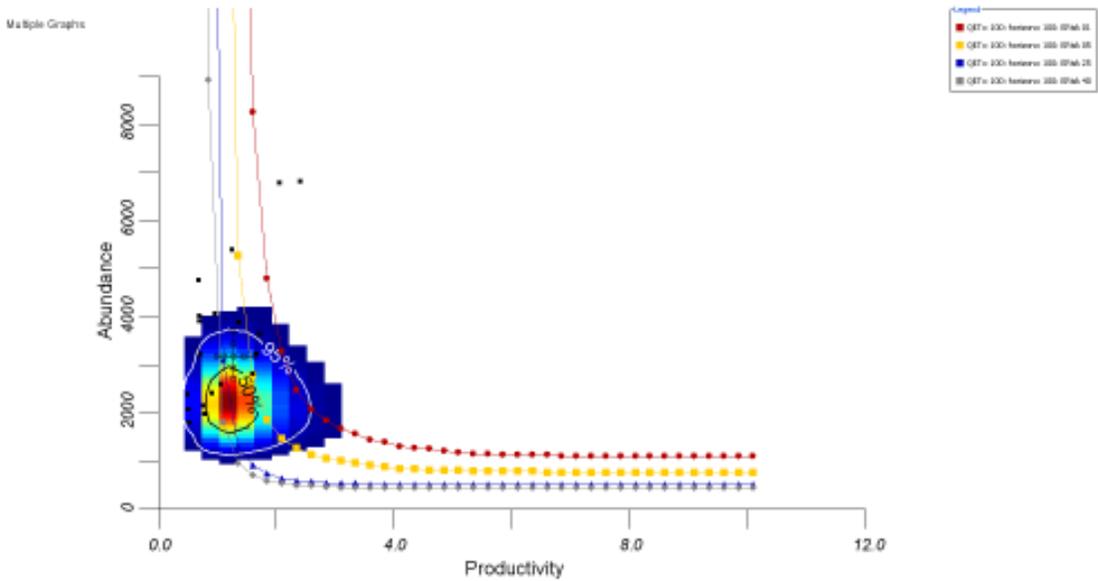
Figure 42 North Santiam steelhead pre-harvest.
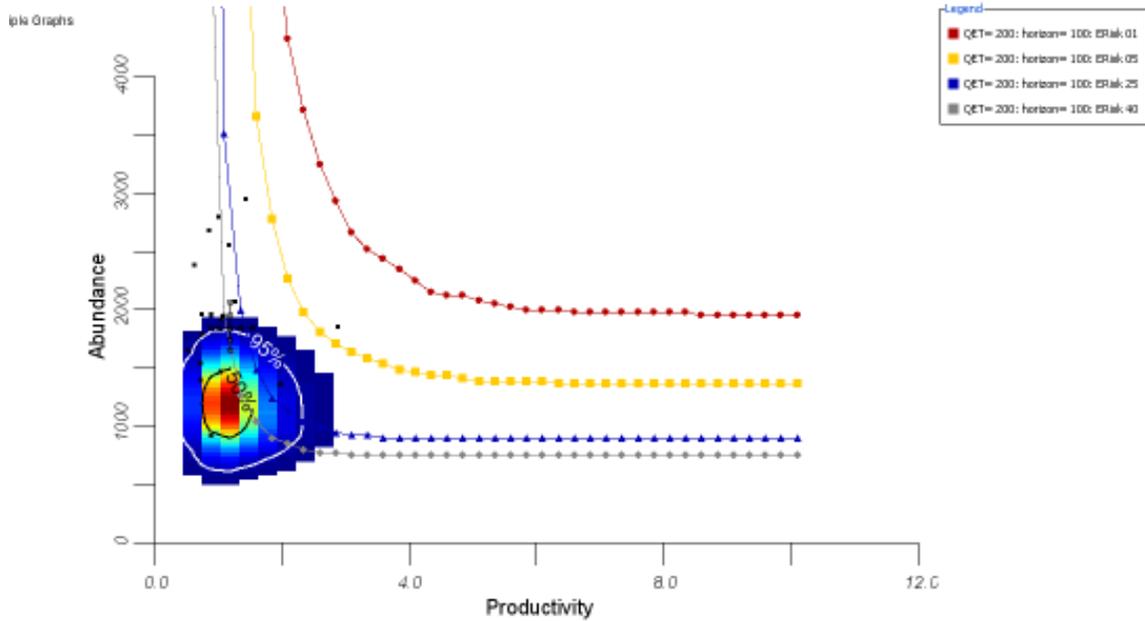
Figure 43 South Santiam (lower) steelhead escapement.
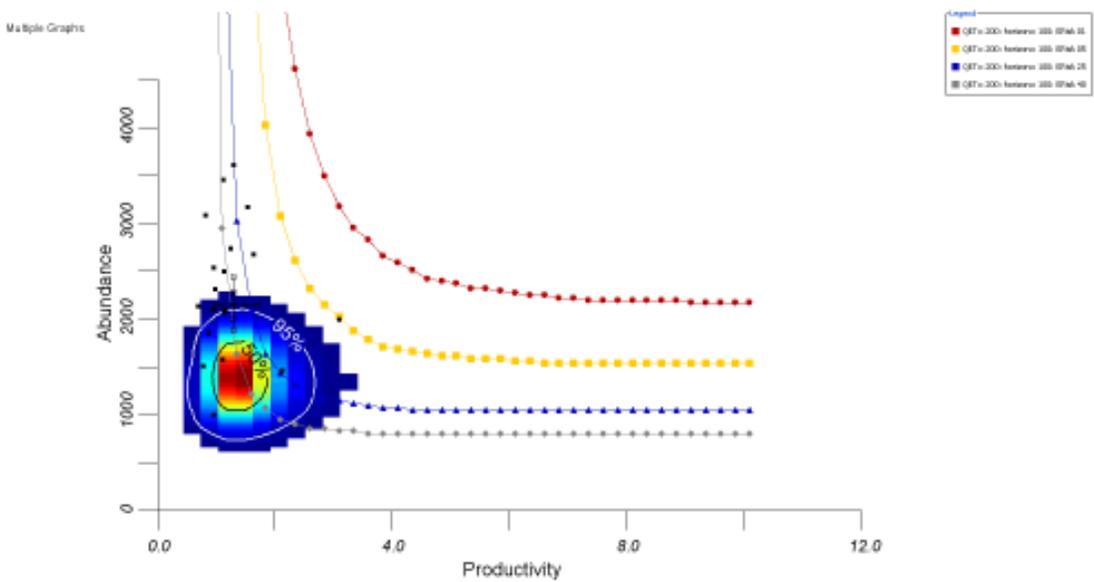
Figure 44 South Santiam (upper) steelhead pre-harvest.
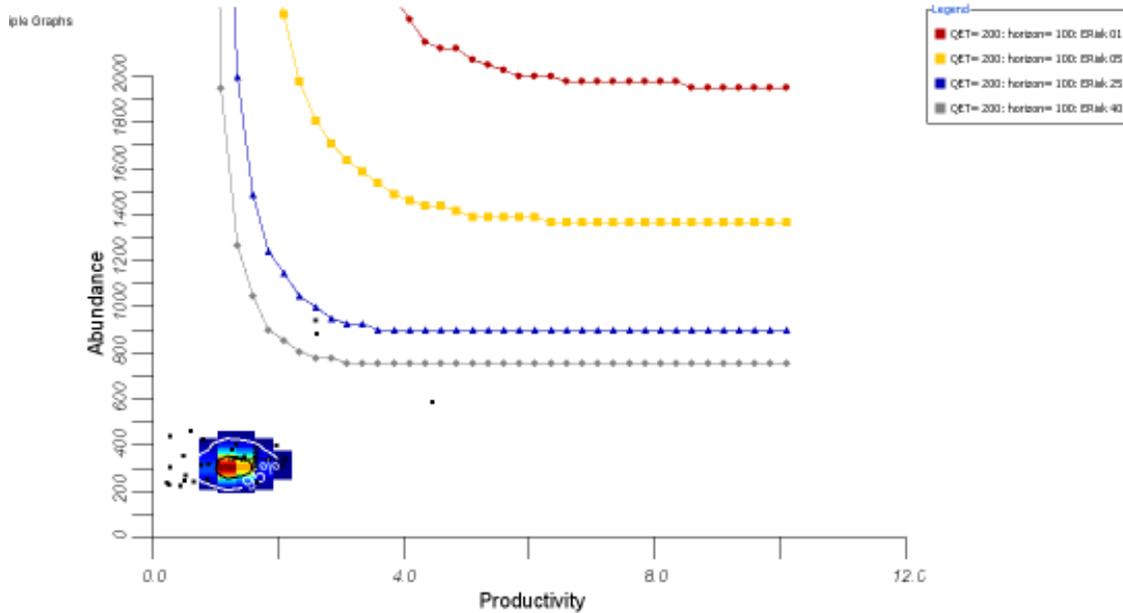
Figure 45 South Santiam (upper) steelhead escapement.
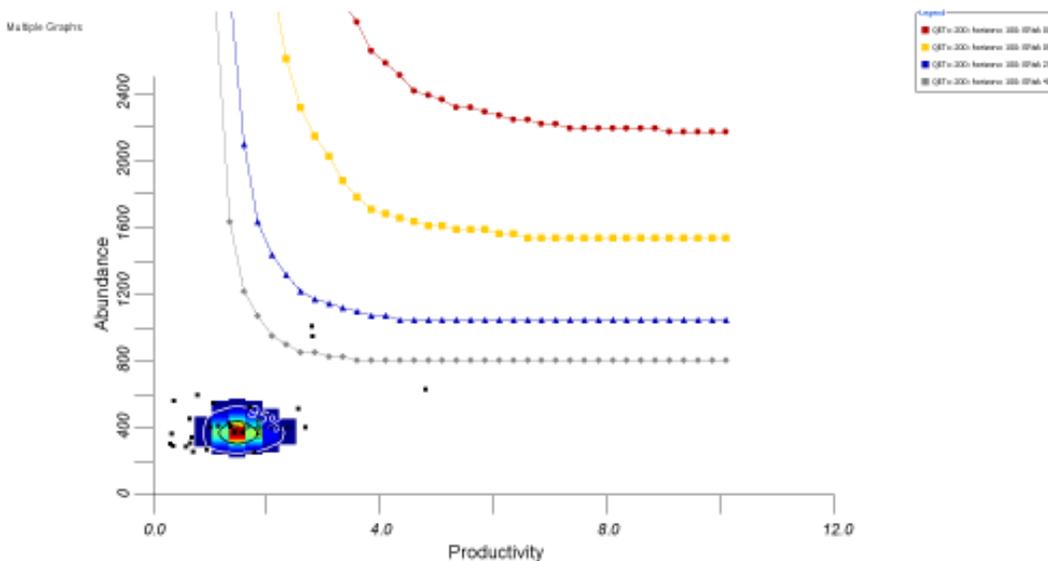
Figure 46 South Santiam (upper) steelhead pre-harvest.

Populations with Limited Data

Drawing a target viability curve for a population requires little population specific information. As noted above, no assessment of a population's current abundance and productivity is required. An assumption of population variability is needed, but as we are producing pooled ESU wide estimates from all population with sufficient data, these pooled estimates can be applied to populations within an ESU for which specific information are lacking. The RFT and QET are population specific, but the only information required is an estimate of stream miles for spawning. These estimates are available in the WLC for all populations (including Washington) except some Oregon chum populations. As a consequence of the low population specific

information requirements, we can produce viability curves for use as criteria for nearly all of the WLC populations.

Although we can generate viability curves for all populations, we cannot provide a complete quantitative assessment of where a population is relative to the curve without a time series of spawner abundance, fraction of hatchery origin spawners, the harvest rate of natural origin fish, population age structure and (ideally) an annual index of marine survival. We generally need in excess of 15 years of data to have even moderate precision in the status estimate. If for whatever reason these data are lacking or of poor quality for estimating productivity and abundance, it may still be possible to provide a qualitative or rough quantitative approximation of the current status of a population. It is imperative to maintain an adequate assessment (and communication) of the uncertainty associated with such *ad hoc* methods. Where data are sparse, it may be necessary to recognize that no assessment of population status relative to the curve is possible. In fact, this is the situation for evaluating the current status of most of the Oregon WLC populations

The methods for conducting this qualitative approximation will be data dependent and we cannot provide exhaustive recommendations. For some populations, an approximation based on extrapolation from neighboring populations with similar habitat and adequate data may be adequate. Often, there are a few years of population abundance data even if there are insufficient data to estimate productivity. This at least allows some approximation of where a population is relative to one of the axes of the viability curve graph. In the current status evaluation section of this report, we provide an application of some of these approximations.

In general, we encourage the collection of high quality data that are adequate to evaluate where a population is relative to the viability curve. This is particularly true for populations targeted to achieve a high viability status, where managers will desire greater certainty of population status before making management decisions.

Population Change Criteria and the Viability Curve

The population change criteria proposed in our 2003 report can be expressed in terms of a viability curve. The model used to define the curve is based on Hockey Stick recruitment function applied to a 4-year running sum of abundance rather than age-structured recruitment. The approach uses the projected growth rate of the population from its current abundance to a target abundance as a measure of productivity. The method describes one way to get from the current status (where only abundance need be known) to a point above the viability curve (Figure 47). The PCC approach has the advantage of providing a specific abundance and growth rate target for a population, but does not include the flexibility of assessment allowed with a generalized viability curve approach. In addition, the growth rate of a population may be an overly precautionary estimate of productivity (see discussions in 2003 viability report). However, there may be some management advantages to including the PCC targets in goal discussions because they are relatively concrete and, if met, would generally indicate a viable population consistent with the viability curve approach. PCC targets for Oregon WLC populations and additional discussion of the method are provided in Appendix F. PCC targets for Washington LCR populations are included as part of the Washington interim recovery plan (ref).
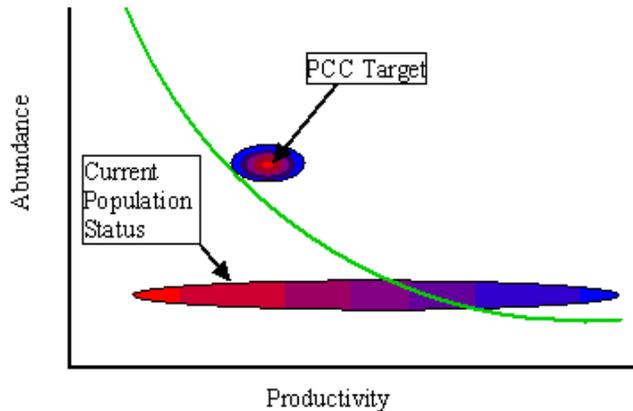
Figure 47 Diagram showing relationship of PCC targets and viability curves. The current population shows a condition where the abundance is known with relative confidence but the productivity is not (However, productivity is assumed to be relatively low because the PCC method should only be applied to populations considered at risk). The PCC target is the abundance and productivity associated with obtaining the PCC target growth rate from current abundance with some measure of confidence.

Minimum Abundance Threshold (MAT)

The viability curve describes a relationship between abundance, productivity, and extinction risk based on specific assumptions about recruitment and variability. There are biological and ecological factors that affect the relationship between abundance and extinction risk that are not addressed in the viability curve models. These factors include genetic issues (see Table # in Diversity section), ecosystem function (e.g., marine derived nutrients), catastrophic risks, and others. Consequently, we propose a minimum average abundance threshold that would apply regardless of where a population falls relative to the viability curve. In addition to considering factors not addressed by the viability curve, a minimum abundance criterion provides a direct measure of whether the population has been in a low abundance range where depensatory processes can operate. The viability curve includes consideration of depensation though the RFT and the QET, but an additional metric evaluating depensation risks that does not depend on the entire suite of assumptions in the viability curve will increase confidence in the evaluation.

Considering all of these factors, a number of minimum size thresholds for salmon have been proposed. Largely because of data limitations, the California TRTs have relied on a "rule-of-thumb" approach for setting viability criteria (Table 11). These criteria are based on fairly generic conservation biology principles rather than salmon population specific assessments. The California criteria are drawn from both the IUCN criteria and from Allendorph et al.(ref). The California criteria include abundance and trend; our focus here is on abundance.

Table 11 California TRT viability criteria rules of thumb (Lindley et al., in review)

Table 1. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids. Overall risk is determined by the highest risk score for any category.

	Risk of Extinction		
Criterion	High	Moderate	Low
Extinction risk from PVA	> 20% within 20 years	> 5% within 100 years	< 5% within 100 years
	– or any ONE of –	– or any ONE of –	– or any ONE of –
Population size	$N_e \leq 50$	$50 < N_e \leq 500$	$N_e > 500$
	–or–	–or–	–or–
	$N \leq 250$	$250 < N \leq 2500$	$N > 2500$
Population decline	Precipitous decline[a]	Chronic decline or depression[b]	No decline apparent or probable
Catastrophe, rate and effect[c]	Order of magnitude decline within one generation	Smaller but significant decline[d]	not apparent
Hatchery influence[e]	High	Moderate	Low

[a] Decline within last two generations to annual run size ≤ 500 spawners. Historically small but stable population not included.

[b] Run size has declined to ≤ 500, but now stable.

[c] Catastrophes occuring within the last 10 years.

[d] Decline < 90% but biologically significant.

[e] See Figure 1 for assessing hatchery impacts.

The IC-TRT has also included minimum population size criteria (Table 12), which they have overlaid onto viability curve graphs (e.g., Figure 48). The IC-TRT minimum size criteria are based largely on genetic and spatial distribution concerns.

Table 12 Minimum population size criteria from IC-TRT viability draft July 2005.

Table 1. Minimum abundance thresholds by species and historical population size (spawning area) for Interior Columbia Basin stream type chinook and steelhead population (Table 3). Median weighted area and corresponding spawners per km (calculated as ratio with corresponding threshold) provided for populations in each size category (see attachment B).

Population Size Category	Stream Type Chinook (Upper Columbia Spr, Snake Spr/Sum ESUs)			Steelhead (Upper Columbia, Middle Columbia & Snake River ESUs)		
	Threshold	Median Weighted Area (m X 10,000)	Spawners per KM (weighted)	Threshold	Median Weighted Area (m X 10,000)	Spawners per KM (weighted)
Basic	500	20	25.0	500	63	4.9
Intermediate	750	42	18.0	1,000	302	3.3
Large	1,000	77	13.0	1,500	627	2.4
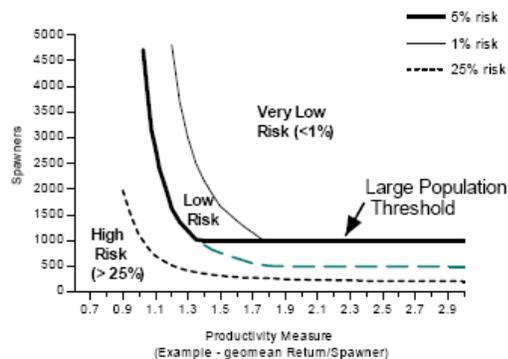
Figure 48 Viability curve showing minimum size threshold from IC-TRT viability report July 2005 (page 12).

In our 2003 viability report, we provided minimum population size criteria associated with the PCC targets. These sizes are updated in Appendix B of this report and listed here in Table 13. These criteria are based on concerns about estimating population growth rates at low initial abundances.

Table 13 Minimum sizes from PCC approach (2003). PCC analyses have not yet been completed for coho.

Species	Minimum Size (4-year average)
Chinook	1,400
Chum	1,100
Coho	N/A
Steelhead	600

Based on consideration of all these criteria recommendations, and, more importantly, the justification for the various criteria, we recommend the MAT values shown in Table 14 measured as a geometric mean over the recent time series. This criterion is in addition to--NOT in place of--the viability curve criteria. We have not specified the required length of a "recent time series" or the level of confidence required that a population really is averaging over the MAT values. These important considerations should be evaluated on a case-by-case basis. We caution against relying on the point estimate of the geometric mean, especially if the time series is very short (<20 years) or highly variable. The MAT values are species and size specific following the same basic logic discussed in the RFT and QET sections.

The size thresholds in Table 14 are based largely on the viability curve analysis. The abundance ranges are the equilibrium abundance of the viability curves where they tend to asymptote at high productivity (e.g., productivity >6), with the constraint that the minimum size for category 3 is 500 spawners and the minimum size for category 4 is twice the minimum size for category 3. The minimum size of 500 for category 3 was applied to the small and medium Chinook populations and to the small steelhead populations. In general we can see how a population is doing relative to the MAT values by examining the population contours on the y-axis of the appropriate viability curve. However, the MAT values are not necessarily identical to the viability curves in persistence categories 3 and 4, so both analyses are necessary.

Table 14 A viable population needs to have a geometric mean spawner population size greater that the mean abundance threshold (MAT), where a population's geometric mean spawning size is measured over a long time period (e.g., 20 years) with acceptable confidence (e.g., >95% confidence interval.).

Species	Size Category	Minimum Abundance Threshold (MAT)				
		Persistence Category 0	Persistence Category 1	Persistence Category 2	Persistence Category 3	Persistence Category 4
Chum	Small	0-400	400-500	500-700	700-1,400	>1,400

	Medium	0-900	900-1,000	1,000-1,400	1,400-2,800	>2,800
	Large	0-1,100	1,100-1,500	1,500-2,000	2,000-4,000	>4,000
Chinook	Small	0-100	100-200	200-500	500-1,000	>1,000
	Medium	0-300	300-350	350-500	500-1,000	>1,000
	Large	0-550	550-600	600-700	700-1,400	>1,400
Coho	Small	0-700	700-800	800-1,100	1,100-2,200	>2,200
	Medium	0-1,300	1,300-1,500	1,500-2,200	2,200-4,400	>4,400
	Large	0-2,000	2,000-2,300	2,300-3,400	3,400-6,800	>6,800
Steelhead	Small	0-200	200-300	300-400	500-1,000	>1,000
	Medium	0-400	400-500	500-700	700-1,400	>1,400
	Large	0-800	800-1,000	1,000-1,400	1,400-2,800	>2,800

Combining Abundance and Productivity Metrics

The benchmark escapement viability curves, the pre-harvest viability curves and the MAT analyses all provide evaluations on the 0-4 persistence category scale. The simplest approach to obtaining an overall abundance and productivity score for a population is to average these scores. Although this may be satisfactory in many cases, in other cases, additional information beyond these metrics may be evaluated or there may be some population specific reason to not weigh the metrics equally. Therefore, we recommend calculating the average of the viability curves and MAT scores, but not consider this the sole determinant of a population's abundance and productivity score for the overall population synthesis. A population specific evaluation will be required.

Diversity Criteria

The diversity criteria section is divided into two main sub-sections. The first sub-section, Diversity Overview, provides a discussion of the importance of diversity as an indicator of population viability and a discussion of the utility and challenges with various types of diversity measures. The second sub-section, Diversity Metrics and Thresholds, describes the TRT recommended viability criteria.

Diversity Overview

The establishment of criteria for each of the viability elements provides a measure of the status of a population and perhaps more importantly provides guidance for recovery actions to restore and/or preserve those populations. The inclusion of diversity criteria helps ensure the preservation of the underlying genetic resources necessary for a population to fully exploit existing ecological opportunities, adapt to future environmental changes, or simply maintain a sustainable status. The emphasis must be on preservation, because once lost genetic variation is effectively gone forever (Riddell 1993). Riddell (1993) presented 10 principles for conserving diversity, primarily through the conservation of distinct reproductive groups. The focus of Riddell's strategy was to "manage Pacific salmon from the premise that localized spawning populations are genetically different, and valuable to the long term production of this resource." Populations and subpopulations (demes) were viewed as standard units for preserving diversity. The conservation of diversity could be achieved by "maximizing the spatial and temporal distribution of demes, …maintaining populations with unique genetic traits or, genetic traits of importance, [or] maintaining populations occupying atypical habitats or expressing unusual phenotypic traits." Similarly, the viability documents from all of the TRTs have addressed the issue of interpopulation diversity, ensuring that populations representative of major life history

strategies are preserved. For the WLC-TRT the creation of life history strata provided a mechanism for maintaining course-scale ESU diversity by directly identifying life-history strategies (i.e., fall run/spring run) or by indirectly identifying populations that have adapted to ecologically distinct geographic regions. Interpopulation diversity is relatively easy to identify (in part because the major strategies can be described in general terms) and because it is relatively easy to correlate these life history strategies with major ecological regions. The primary mechanism for the conservation of interpopulation diversity is the identification of specific populations to preserve or recover; however, what is still lacking is a means to monitor and evaluate the diversity "health" of the individual populations (i.e., *intra*population diversity).

Previous TRT documents (e.g., McElhany et al. (2003) and Riddell (1993)) are less specific regarding within population life history diversity, primarily because our understanding of the ecological nuances of fine-scale diversity is far from complete. Specifically, these documents do not--*and cannot*--definitively state which traits are most important and how much within population diversity is enough for now or in the future. We also ignore for the moment the problem of genetic load and excessive diversity, primarily because under natural conditions it is unlikely to be encountered, especially in depressed populations. Whereas major life history strategies (complexes of life history characters) generally occur at the population level and thus provide clear units (populations) to quantify and monitor, intrapopulation diversity exists throughout the population and not necessarily in distinct subpopulation packets. Major life history strategies are expressed every generation, in contrast to intrapopulation diversity, which entwines an underlying genetic propensity with specific environmental conditions, specific habitats, or both. Even more problematic is the requirement to express a complex sequence of life history characters to successfully complete a life-history trajectory. For example, fry or subyearling emigrants must undergo physiological changes to allow them to adjust to estuarine or marine conditions, and they also need to exhibit behavioral modifications (i.e., schooling) at the appropriate time. Because we are only able to observe fish during a fraction of their life, it may not be possible to identify the specific diversity characters that are present or absent. We can only assess successful life history trajectories retrospectively from returning adults (primarily through scale analysis). Although the guidelines put forth by Riddell (1993) can be applied on an intrapopulation basis, the problem of identifying distinct population units to conserve is problematical. Within a population, local spawning aggregations do exist; however, by definition these subpopulation units experience considerable gene flow within a population and it is not expected that substantial differences in life history characters would evolve in subpopulations. This is not to diminish the importance of these within population units. There are a number of populations that consist of multiple smaller tributaries (especially in the Lower Columbia River coastal tributaries and the Gorge tributaries), these tributaries serve important demographic and diversity functions. Consideration of the status of within population units, including minor tributaries to larger rivers, has been included in the spatial structure criteria to minimize the "double counting" of conditions.

It has been argued that diversity criteria are unnecessary in evaluating population viability simply because a population could not reach abundance and productivity thresholds if diversity were significantly reduced. While there is some validity to this argument, it should be underscored that population diversity is important for long-term resilience and adaptability. Relatively short-term (e.g., 5- to 10-year) observations of abundance and productivity alone are unlikely to be sufficient for the identification of a population's long-term risk of extinction because of inadequate diversity. Depending on the variability in climate, many traits may not be

expressed during this time interval. The establishment of diversity criteria provides the necessary mechanism for preserving a population's genetic resources during the recovery process, thereby increasing the likelihood of establishing or maintaining sustainable populations into the foreseeable future and beyond.

Determining what to measure and how to measure it

If the development of diversity criteria is problematic, it is in part due to our lack of understanding of the expression of individual life history traits (the genetic and environmental effects) and the degree of correlation between those traits. In most cases we are relying on the direct measurement of what we presume to be important life history traits. Current monitoring efforts for most populations include the measurement of a limited number of life history characters at one life history stage. While much of the work has focused on returning or spawning adults, otolith and scale growth ring analyses are useful in describing earlier life stages. Also, sampling returning adults provides a biased sample, because only the successful strategies are represented and not the full spectrum of life history traits expressed. As with the other viability criteria, any diversity criteria will have to be based initially on those characters for which a current database exists. It is possible that future monitoring efforts may focus on altogether different traits that are more indicative of population fitness. The establishment of diversity criteria will ultimately depend on identifying meaningful measures of diversity. We can either directly measure diversity, by observing the variation in specific life history traits, or indirectly, by measuring factors that may influence the rate of local adaptation or the random effects of population size. Additionally, genetic analysis can be used to indirectly measure diversity, or more correctly measure the characteristics of neutral genetic markers within and among populations. While direct measures focus on life-history traits that are the product of local adaptation (natural selection), diversity can also be indirectly measured in terms of the magnitude and specificity of selective forces that influenced the expression of important life history traits. Whether natural or manmade, selective forces may produce differential survival among fish exhibiting different life history traits. The identification of these selective forces could provide a useful surrogate when the direct observation of life history traits is not possible. Of these methods, the direct measure of life history traits is the most intuitive and the most widely used.

Direct Measures of Diversity

In listing potential life history traits to measure it is useful to attempt to distinguish between major life history strategies and phenotypic traits. The Interior Columbia TRT (IC-TRT) lists a number of major traits in their Viability Criteria document (July 2005) for the populations in the Lower Columbia River domain, most of which would correspond loosely to the run timing strata in the Lower Columbia River. Additionally, the IC-TRT considered each of the juvenile emigration patterns (i.e., fry, fingerling (spring), fingerling (fall), and yearling (second spring)) as major life history strategies. Having identified strategies and traits of interest it is necessary to develop adequate means of quantifying these traits as part of an overall measure of diversity. The measure of change in diversity is somewhat dependent on the character being assessed. For example, juvenile emigration normally occurs during specific temporal windows (as listed above). Use of these emigration windows provides a course scale measure of diversity, while the proportion of juveniles in each temporal window and the mode and range of each window constitutes a finer scale measure. As with most measures of biological traits there is some

uncertainty in identifying whether statistically significant changes in traits are biologically important, or whether statistically non-significant differences are biologically important.

Life History Strategies

1. Timing of return to fresh water
    a. Run time (e.g., fall vs. spring)
    b. Variation within a specific run time
2. Age at maturation
3. Spawn timing
4. Outmigration timing
    a. Distribution to downstream or upstream rearing habitat
    b. Specific nursery habitat utilization
5. Smoltification timing
    a. Entrance to marine environment
    b. Duration of residence in intertidal or Columbia River plume areas
6. Developmental rate
7. Egg size
8. Fecundity
9. Freshwater distribution
10. Ocean distribution
11. Size at maturation
12. Timing of ascension to the natal stream

For most of these traits one would want to know a number of parameters: the mean, mode, and range of occurrences. Additionally, measures of life history characters need to be correlated with freshwater and climatic conditions. Changes in both major and minor traits by themselves are not necessarily indicative of changes in diversity. In many cases, phenotypic responses to environmental fluctuations are indicative of a "healthy" population. Of more concern would be the absence of a correlation in between variation in environmental parameters and changes in life history characters.

Direct measures can be very useful. Often we associate specific population traits with local freshwater conditions (hydrology, water temperature, and distance to the ocean). While general geographic or ecological conditions may dictate the general nature of many of these traits, there is still considerable variation in trait expression for most populations both intra- and interannually. Much of this diversity is responsive to annual variation in climatic or ecological conditions, and only in hindsight can the question whether a population has sufficient diversity to be responsive to annual changes be evaluated. For example, whether a fry, spring subyearling, fall subyearling, or yearling outmigrant strategy is more successful in Chinook salmon populations west of the Cascades depends in large part on rainfall and stream temperature patterns for the Spring and Summer in the specific basin. For a population, the potential costs of bet-hedging with multiple life history strategies needs to be balanced with adaptation to *present* environmental conditions. For example, early returning fall-run Chinook salmon might encounter high water temperatures during a low water year that would delay the return to their spawning grounds until conditions are suitable (although there is a limit in how long they can hold). Ultimately, the relative fitness of each life-history type to the spawning population is the benchmark by which the population evaluates the "success" of a trajectory. One can be fairly

confident in asserting that multiple outmigrant strategies are necessary or "desired", but it would be riskier for recovery entities or fish populations to attempt to establish quotas on each temporal window. Simply verifying that a population is still capable of expressing a specific life history type *may* be sufficient if one assumes that a population will express a specific trait when the environmental cues are present. Similarly, monitoring variability rather than some mean or mode in traits, such as spawn timing, may be the more useful measure of some diversity traits.

The issue of which traits are "most" important is problematical. Depending on conditions, it may be that a different trait is limiting population recovery in different years. While monitoring the "usual suspects" may in the end help us understand the relationship between variability in specific traits and environmental fluctuations, it may not be immediately useful to the manager in the field or the recovery planner. Additionally, changes in some unmonitored traits may create a cascade of changes in other life history traits. For example, the decrease in the size of coho salmon in Puget Sound may result in smaller egg sizes (as fish compensate for decreases in fecundity); in turn, smaller egg sizes may alter the time and size of emergence, which may alter the ability of juvenile coho salmon to establish feeding territories, or the timing and size of their downstream migration, etc. Without a clear sense of the interrelatedness of traits, monitoring the changes in one character may not explain changes in the overall fitness of a population. Alternatively, some traits may buffer the loss of diversity in other traits and effectively mask underlying deterioration in genetic diversity. The selection of which traits to monitor may depend on historical precedent for that population and the professional opinion of local biologists.

While directly monitoring life history traits is an important task in understanding how a fish or fish population relates to its environment, it many not provide the "litmus test" of viability that is desired. Certainly, long term changes in life history traits may be indicative of underlying genetic changes, although these types of changes are usually in response to long term environmental changes or long term anthropogenic selection (including deliberate or incidental artificial propagation effects or harvest). More subtle diversity changes may occur through the loss of genes influencing life history traits that are only rarely expressed. Therefore, direct measurement of diversity is most powerful when done over an extended period of years with a high likelihood of including multiple environmental conditions and when potential causal factors (e.g., environmental conditions or harvest regimes) are monitored in tandem.

Indirect Measures of Diversity

An alternative to directly measuring the life history traits of interest is to measure the external factors or processes that influence their underlying genetic components or their phenotypic expression.

Effective Population Size

There are a number of indirect measures of diversity. Effective population size (*Ne*) is one example of an indirect measure that can be related to genetic and phenotypic diversity. *Ne* represents the effective number of spawners that contribute to the next generation, rather than simply the census number of spawners observed on the spawning grounds. Reductions in *Ne* below 100 and 500 individuals can result in a loss of genetic diversity over the short- and long term, respectively. Under most conditions the *Ne* criterion is somewhat redundant as a viability goal, since the VSP abundance goal for a viable population is likely to be much higher than *Ne*-

based abundance level (to achieve and *Ne* of 100 the census population would have to be 200-400 adults -- McElhany et al. 2000). Monitoring *Ne* is useful in defining how far *below* VSP a population is. Additionally, even short term (e.g., a single generation) decreases in *Ne* can have long term negative effects on a population's genetic resources. Calculating *Ne* is relatively straightforward (see Hartl 2000), if the underlying population conditions are known. Effective size is also included in other proposed measures of diversity (i.e., the AHA/Fitfish Hatchery-Domestication model).

Spatial Distribution

The ICR TRT has chosen two other indirect factors as measures of diversity: the source and magnitude of strays and distribution of spawners across distinct ecological areas. Spawner distribution measures are intended to cover both Spatial Structure and Diversity Criteria. If fish spawn and rear in a variety of freshwater habitats in a basin, the population, as a whole, will be buffered against year-to-year environmental variations. Additionally, different initial incubation and rearing conditions may set juveniles from different spawning aggregations on different life-history trajectories providing further life history diversity. Over time, these spawning aggregations have become, or will become, locally adapted to each of these diverse environments. The monitoring requirements for spawner spatial structure monitoring; however, seem fairly intensive, especially for species like coho salmon that are difficult to observe, and for which there is a limited historical baseline. Alternatively, one could assess the diversity of accessible habitat as a reasonable proxy for life history diversity and its underlying genetic diversity. Habitats that reflect historical diversity should ultimately restore and maintain historical levels of life-history diversity.

Hatcheries

The discussion of hatchery programs in this section will focus on the effects of hatchery-origin fish on diversity. Hatcheries affect diversity largely through the process of domestication and the introgression of non-adapted hatchery transplants. Additionally, hatchery propagation may produce non-genetic effects on the expression of life history traits via non-natural rearing regimes (i.e., non-natural size and time of release); however, the long-term consequences of these effects on diversity are not well understood. Issues related to competition, predation, or disease transmittal by hatchery-origin fish, while important to overall population viability, are not considered in this section, but will be addressed in the discussion of population productivity

Hatchery influence, whether it results from the inclusion of naturally-produced fish into hatchery broodstocks or the "straying" of hatchery-origin fish onto the natural spawning grounds, is presented as a measure of population diversity in the All-H Hatchery Analyzer (AHA) model (HRTDG 2005) and in the Interior Columbia TRT's Viability document. This model estimates the degree to which hatchery/domestication influences erode local adaptation, delay the rate of adaptation, or create a genetic load of recessive deleterious alleles, etc. It should be remembered that hatchery/domestication influences are one part of a multi-function process, where the effects (domestication, inbreeding, natural and artificial selection) are multiplicative rather additive over time. Because the effects of domestication are retained in a fish's genome, it is not possible to eliminate them simply by removing the fish from the hatchery environment. Instead, natural selective forces must reestablish the frequency of locally adapted genes (a process that can easily be protracted over many generations, depending on conditions).

The AHA model tries to simulate many of the effects of hatchery operations, and in doing so provides a useful indirect measure of the impact on diversity. Beyond domestication, the introduction of non-local genotypes (not addressed in the AHA model) is likely to have an erosive effect on locally-adapted life history traits and overall population fitness. Objective measures of the effect of non-local genotypes on sustainability are not available. In general, the geographic distance, similarity in environmental conditions, and similarity in life history traits between introduced and local populations will have to be considered. Overall, given the scale of hatchery production, the low level of natural production, and the quality of much of the spawning habitat in the Lower Columbia and Upper Willamette rivers, the diversity aspects of hatchery introgression and competition must be included as a potential major factor influencing overall population viability.

Genetic Markers

Genetic analysis can be a valuable tool for measuring diversity. Most commonly, allozyme, microsatellite DNA, or mitochondrial DNA analyses have been used to describe inter- and intrapopulation genetic variability. It should be stressed that these techniques detect variation in neutral genetic markers, and are not directly representative of genes coding for life history diversity or local adaptation. This does not preclude the development of future techniques that may yield more direct measures of population fitness. Genetic patterns or changes in neutral markers, however, can be indicative of the factors affecting the breeding structure of a population, including the relationship of spawning aggregations within a population. Population bottlenecks (small Ne presently or sometime in the past), estimates of Ne, detection of hybridization, hatchery introgression, or inbreeding are commonly detected through changes in neutral markers and are likely to have an associated effect on the expression of life history traits. In general, fitness traits are sensitive to inbreeding. The utility of genetic analysis is somewhat limited due to the absence of historical baselines. The present-day genetic status of most populations in the Lower Columbia and Upper Willamette rivers has been, with a few notable exceptions, largely compromised by non-indigenous hatchery introductions and/or reductions in population abundance to critically low levels. Nevertheless, continued monitoring of genetic changes within and among populations can reveal changes in the genetic characteristics of a population or ESU. In the short term, as recovery actions begin to reestablish natural processes (e.g., local adaptation) it may be unclear if the genetic changes detected are indicative of a positive change in the population.

Artificial and Natural Selection

Our discussions of Viability Criteria always seem to drift toward ever more complex and inclusive models. As with the present iterations of the habitat model, we seem to be very good at quantifying what is bad rather than establishing what is good, or especially what is best. This is not meant to discourage present attempts to evaluate diversity, habitat or spatial structure, but to remind us that it may be better in the near term to be satisfied with simple (and slightly imperfect) models. In this case the simple model is a general description of diversity and its role in improved fitness through local adaptation. If we cannot describe in a predictive manner optimal life history characteristics, we can attempt to ensure that the conditions for a population to adapt to environmental changes and ecological opportunities are preserved or restored. As always, it is challenging to describe universally desirable conditions, but we do have a fairly

extensive list of what *not* to do. Many activities impose an artificial selective force and generally reduce fitness in the natural environment. Such activities include:

1. Harvest regimes that select for run timing or size
2. Size-based or temporal upstream or downstream migration barriers
3. Disruption of juvenile rearing habitats with large-scale hatchery releases
4. Allowing naturally-spawning population abundances to fall to critically low levels
5. "Mining" a sub-component of a natural run for hatchery production (e.g., selectively collecting some temporal portion of the run, larger/older fish, etc.)
6. Use of release strategies that encourage natural spawning of hatchery origin fish (i.e., net pen releases)
7. Release of non-native fish or other invasive species
8. Use of artificial transport (e.g., "trap and haul") that may select (temporally or otherwise) certain fish to access natural spawning grounds
9. Anthropogenic changes in environmental conditions (water impoundments or withdrawals, global climatic changes, etc)

In assessing the effects of artificial selection we estimate the degree to which a population's life-history diversity has been modified relative to its historical locally-adapted state. This locally-adapted state, like the local habitat itself, is not static but dynamic. Adaptation to short-term climatic cycles is likely to be part of the existing range of life-history traits, while long term climatic changes may require changes in the suites of phenotypes for many traits. These long-term changes have generally occurred at a pace slow enough for trait selection to occur without a dramatic decline in fitness. Anthropogenic-driven habitat changes generally occur at a rate many times faster than natural rates of change (other than catastrophic events), and the new habitat conditions are generally, but not always, less productive than historical conditions. These dramatic habitat changes can exert a strong selective force on the expression of life history traits and have a profound effect on population diversity. As part of the indirect measures of diversity we have included changes in habitat: habitat diversity, habitat quantity and habitat quality. It is important to draw a distinction between how habitat changes influence diversity and how these changes influence productivity and abundance (which is covered in the Productivity and Abundance metrics).

Diversity Metrics and Thresholds
Given the uncertainties in trying to quantify diversity and the unlikely prospect of finding one "unified field theory" model to use as a metric, we are proposing using four different approaches to evaluate diversity. Where possible, emphasis should be given to direct measures of diversity, with indirect measures used to supplement the evaluation of diversity. These include:

Direct Measures
      1. Life history expression and responsiveness
      Indirect measures
          1. Habitat diversity and occupancy
          2. Hatchery introgression and domestication
   3. Evaluation of adaptation

Direct Measure: Life History Expression and Responsiveness

Direct measures of diversity, e.g., the monitoring of life-history traits, require the monitoring of important fitness related phenotypes. The selection of which phenotypes to measure was discussed at length earlier. There are several approaches to the evaluation of diversity and life-history trait expression; these consist of static and dynamic observations. Under a static system, existing phenotypes can be compared to historical observations for same population or to phenotypes observed in historical or existing "viable" populations in ecologically similar basins. Historical information is useful but not often available, and there is some uncertainty in assigning basin ecotype. Under the dynamic system, long-term monitoring of life history traits is combined with the monitoring of environmental conditions. In contrast to other viability metrics, the dynamic approach assumes that the stationarity assumptions are violated.

Static measures of diversity can be readily applied to most populations, but should be used with some caution in that they require the development of generalized life history diversity profiles, an expectation of life history trait expression. Further, one would need to rely on more than a single year's data to acquire a useful estimate of life history diversity. Diversity scores would be evaluated based on the deficit or surplus of life history types relative to the historical or ecotype baseline. For example, the time of peak spawning for many Upper Willamette River spring run Chinook salmon populations has shifted from mid-August to mid-September. Some of this change may be strictly environmental, but it would appear that much of this change has been genetic. Similarly, may streams that historically-contained late-returning coho salmon that spawned from December through February now contain early-run coho salmon that spawn from late-October through November (Myers et al. 2006). Changes of this magnitude may have had a strong influence on overall population fitness, and would merit a low diversity score.

The dynamic metric attempts to evaluate diversity from a functional perspective. It focuses on phenotypic responses as a proxy for genotypic changes and variation. A variety of life history characteristics are monitored over time. Changes in the occurrence, mean, mode, and variation of these characters are monitored for shifts or constrictions in expression. Additionally, changes in life history characters are compared with conditions in freshwater, estuarine, and saltwater habitats. Changes in life history characters that correspond to habitat/climate changes are generally considered positive indicators of diversity, while an absence of adaptive changes in life history characters in the face of environmental shifts or changes in life history characters in the absence of any environmental perturbation is of more concern, as is adaptation to an artificial environment.

In reviewing the existing diversity information for a population it will be necessary to discriminate between those changes in life history traits that appear to be responses to environmental cues and those life history changes or the absence of life-history changes that do not appear appropriate for existing conditions. Where possible, it will be important to identify life history characters that are absent, based on either historical information or populations existing under similar ecological conditions. This is certainly one area where more sophisticated models are needed, but the biological understanding to run the models is limiting.

Box 1 Change in Spawning Time Relative to Temperature
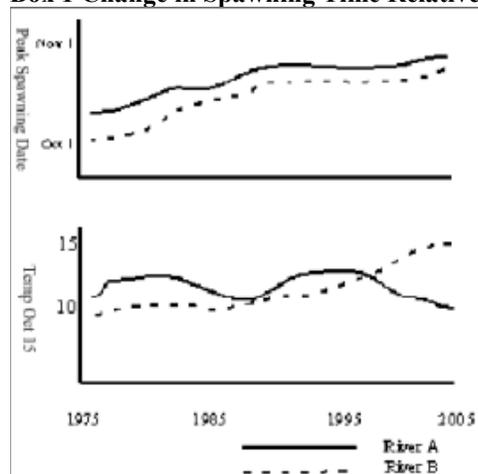
In Box 1 of the populations in the two rivers, the population in River A exhibits a delay in peak spawning without apparent environmental stimuli. The population in River B also shows a delay in spawning, but most likely due to increases in river temperature. The River A population may be undergoing selection for a later spawning time due to other (non-environmental) causes that may result in decreased fitness.

The evaluation of both static and dynamic diversity will require a careful weighting of the relative importance of each trait to the fitness of the population and the magnitude of change in each trait. It is possible that the scoring rubric would differ between populations depending, in part, on the information gathered for each population. Without a substantial database it may not be possible adequately develop such a system. In lieu of attempting to present a prototype we are presenting, as an interim measure, the criteria developed by the ICBTRT (2005) for life history strategies and phenotypic characters. This measure is predicated on the presence or absence of an "expected" life-history variation in a population (e.g., the presence of subyearling spring migrants).

Table 15
Table 16 Preliminary criteria describing risk levels associated with major life history strategies and change in phenotypic characteristics (from ICRTRT 2005).

	Risk Level (Viability Score)			
Factor	**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 historic pathways present, but significant reduction in variability or substantial change in relative distribution.	Permanent loss of major pathway.
Reduction in 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.

Indirect Measures of Population Life History Diversity: Diversity of available habitat types

This metric is based on the close relationship between life history diversity and habitat diversity. This relationship is based on the long-term selection by the environment on life history traits, commonly referred to as the process of local adaptation. Local adaptation can take a number of forms; in stable homogeneous environments a limited number of traits may be expressed, while in variable complex environments a broad suite of traits might be expressed by a population over time. Heterogeneous habitats provide for a larger number of possible life history trajectories than do homogeneous habitats. Multiple life history trajectories buffer the effects of climatic anomalies and other types of environmental variation. Where there has been a reduction in habitat complexity from historical conditions, populations may be at a greater risk due to the absence of alternative life histories. This is not to say that there is anything inherently lacking in naturally homogeneous habitat. Lastly, the ability of a population to exploit the multitude of habitat types in a basin, river, or estuary dramatically increases the productivity and sustainability of that population. It should be noted that the present iteration of this evaluation includes only the natal basin(s) for independent populations. Migratory corridors and juvenile rearing habitat beyond the limits of the population boundaries need to be included to provide a more complete evaluation of habitat diversity. The occupancy measure provides a functional measure of diversity. Overall, this first metric takes into account the potentially stabilizing effect of a diverse habitat, and evaluates the degree to which a population's diversity allows it exploit the habitat. Lastly, this metric has the added benefit of being readily quantifiable.

Habitat diversity is related to both genetic and life history diversity in salmonids at several scales. Waples et al. (2001) have demonstrated a strong correlation between regional ecological diversity experienced by a population and the number of genetic and life history groups within that population for chum, Chinook, and steelhead salmon. Across populations within an ESU, there have been documented differences in migration pattern, age at spawning, and spawn timing that are correlated with hydrologic regime (Beechie et al., In Press). Relationships between freshwater habitat diversity and genetic diversity within a population have been identified by Hendry et al. (1998).

> While the elements of ecological diversity most important for preserving life history and genetic diversity are unknown, we can make use of some general indicators. There are several guiding principles: 1) habitats with differing spatial structure (gradient, width) require life history adaptations in terms of spawn timing, spawning location, and early freshwater migrations, 2) individuals within a population that occupy habitats of differing distance to the sea may display a range of migration timing, 3) variations in the natural flow regime or alterations to the natural flow regime are likely to lead to differences in size at migration and migration timing (Steel 1999), and 4) variations in water temperature pattern are likely to lead to variations in life history pattern related to egg emergence and development timing (Iwamoto 1982, Linley 1993). We have strong evidence that in parts of the Willamette River basin with radically altered thermal regimes, salmon life history patterns have been altered and fitness has been reduced (ACOE 2000).

While there are no known quantitative thresholds at which a loss in ecological diversity reduces genetic and life history diversity, we can make some estimates. The two habitat distribution elements that we can most easily measure and relate to potential genetic and life history diversity are stream order and elevation. Stream order serves as a proxy for stream size. Elevation indexes hydrologic zone, migration distance, and gradient. Other elements of habitat distribution that

were considered included gradient, habitat quality, biological diversity, and presence of cool water springs. At this point, we are unable to use these measures due to a lack of existing data. The two habitat quality elements of ecological diversity that are most easily measured and most likely related to life history adaptations are flow and temperature.

To quantify reductions in diversity, we compare the current and historical distribution of fishes and the natural and downstream flow and water temperature patterns. The change in ecological diversity with respect to the distribution of habitat across different elevation categories can be calculated as follows:

where n = the number of $\Delta ED_{elevation} = \sum_i \left| p_{Hi} - p_{Ci} \right| / n$ elevation categories, i, $p_{Hi} =$ the proportion of the historical occupancy in elevation category i and $p_{Ci} =$ the proportion of the current occupancy range in elevation category i.

The change in ecological diversity related to stream order, $\Delta ED_{streamorder}$, can be calculated in the same manner with categories defined as stream orders 1-7 and all larger streams lumped together as an eighth category. Note that this scoring metric will not detect total magnitude of habitat loss as that would be covered under spatial structure. Also note that this metric can be calculated for nearly every population because it relies only on very basic stream data and current and historical fish distributions.

A 0-4 rating follows for both stream order and elevation:

$\Delta ED_{elevation}$ **or** $\Delta ED_{streamorder}$	**Ranking**
<0.02	4
0.02-0.06	3
0.06-0.10	2
0.10-0.015	1
>0.15	0

An example of metric calculations under two scenarios is provided in Box 1. Definitions of elevation categories by species, calculation details, and current rankings for populations in Willamette Lower Columbia ESUs are provided in Appendix A.
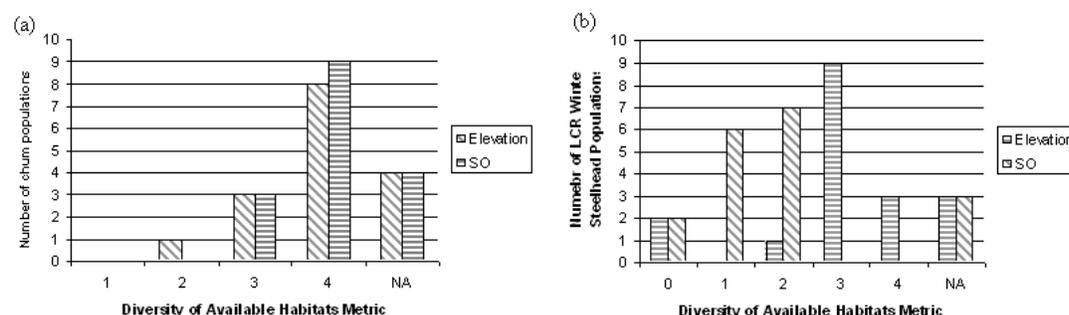
Figure 49 Distribution of the Diversity of Available Habitats metric for Lower Columbia chum and winter steelhead.

A summary of the diversity of available habitat rankings by elevation and by stream order for Chinook and for chum is provided in Figure 49. Both metrics display a range of values with most chum populations seeing little change in the diversity of available habitats and many winter

steelhead seeing changes in the diversity of available habitats particularly with respect to stream order. Figure 50 displays the correlation between the diversity of available habitats metrics by stream order and by elevation. The low correlation between these metrics suggests that they are picking up different kinds of changes in the distribution of habitats, that they are not double-counting habitat loss, and that retaining both metrics provides useful information.
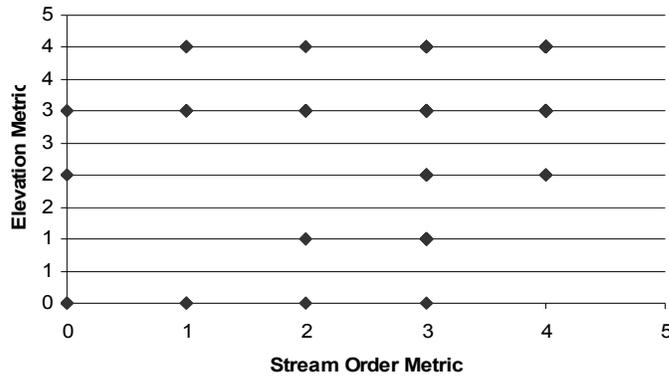
Figure 50 Correlation between the diversity of available habitats metric calculated using elevation and calculated using stream order over all species for all populations for which both metrics could be calculated. The correlation coefficient is 0.43.

Flow and temperature ratings can be developed in a similar manner. Because there is not always historical data and we expect changes in stream flow and temperature due to climate change and natural climate cycles, we may be able to compare upstream and downstream patterns rather than current and historic patterns. Average monthly flow and average monthly stream temperature could be calculated both above and below any water management facilities. Metrics, ΔED_{flow} and $\Delta ED_{temperature}$, could be calculated as above for N = the 12 months of the year. The same 0-4 rating system can be used as a placeholder until real data is tested. A disadvantage of this scheme is that small-scale changes in water temperature pattern cannot be detected.

The average of the rankings for all 4 ΔED ratings summarizes the ecological diversity component of the diversity ranking for each population. In cases where there are no dams, only $\Delta ED_{elevation}$ and $\Delta ED_{streamorder}$ would be averaged.

Implication thoughts: With this strategy, fish could not get a good score with major modifications to flow and temperature but, perhaps they could reach VSP if everything else was perfect. Also, a population could get a good score for flow and temp even under climate change scenarios. We assume that if they are not adapting to climate change then their abundance and productivity ratings would show those declines. The flow and temperature metrics here provide a buffer against quick anthropogenic change that may not be otherwise be immediately detectable in population performance.

Box 2

Example 1a One large dam that blocks all higher elevation habitats

Elevation Range	Historical KM	Current KM	Historical Percent	Current Percent	Absolute Difference
0-500	567	555	0.23	0.36	0.13
500-1000	877	860	0.36	0.56	0.20
1000-1500	412	40	0.17	0.03	0.14
1500-2000	589	0	0.24	0.00	0.24
2000-2500	25	0	0.01	0.00	0.01
Total	**2470**	**1455**			**0.14**

Example 1b Multiple Small Blockages

Elevation Range	Historical KM	Current KM	Historical Percent	Current Percent	Absolute Difference
0-500	567	301	0.23	0.19	0.03
500-1000	877	765	0.36	0.50	0.14
1000-1500	412	218	0.17	0.14	0.03
1500-2000	589	235	0.24	0.15	0.09
2000-2500	25	25	0.01	0.02	0.01
Total	**2470**	**1544**			**0.06**
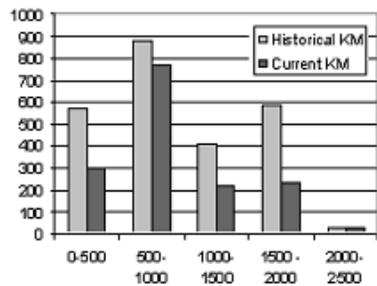
Example 1a (Dam Blockage)   Example 1b (Multiple Small Blockages)
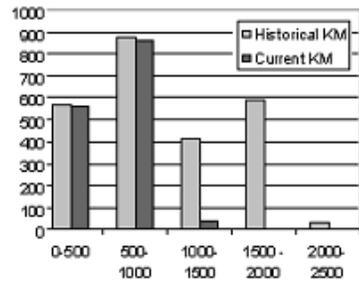
Hatchery Introgression and Domestication

There is a growing body of knowledge indicating that fish reared under artificial conditions (i.e., hatcheries) for one or more generations are less adapted to natural conditions than their natural-origin counterparts. Whether this loss is due to maladapted learned behaviors, the loss of genetic variation for life history traits, or selection for traits adapted to the hatchery environment rather than the natural environment is unknown.

The potential for a loss in fitness is more pronounced when the founding broodstock or subsequent introductions into the hatchery are not locally-adapted to the watershed. Models are currently being developed to assess the effects of hatchery fish interbreeding and competing with natural fish in the wild, as well as the effects of natural fish being taken into the hatchery for broodstock purposes. The FITFISH Model (HSRTG 2005) estimates the consequences of hatchery domestication and introgression on the viability of naturally-produced populations. The model relies primarily on the Proportion of Natural Influence (PNI), the relationship between the percent of hatchery origin natural spawners (pHOS) and the percent natural origin broodstock in the hatchery (pNOB), to determine the rate of domestication. In general, the greater the proportion of natural-origin fish utilized as broodstock in the hatchery the slower the rate of hatchery domestication for the broodstock overall. Similarly, lowering the proportion of hatchery-origin fish that spawn in the wild decreases the frequency of "domesticated" genes entering the natural population's gene pool. PNI is calculated as pNOB/(pHOS + pNOB), with values near 1 indicating low levels of domestication and low values (near 0) indicating a strong domestication effect (Box 3a). In addition to the potentially negative effects of hatchery programs, it also provides for the positive supplementation effects of hatchery releases, although these benefits would be considered in the abundance criteria.

The FITFISH model can also be used as a tool to identify actions that may moderate the effects of hatchery fish by modifying the input variables. As with the Habitat Diversity and Occupancy Metric, this metric is readily quantifiable, although there is considerable uncertainty in a number of the parameters used to derive the model. As our knowledge of the effects of hatchery domestication on fitness improves, there will be further opportunities to refine this approach. In the interim, the FITFISH model provides a basis for estimating the effects of hatchery operations, in addition to modeling effects of changes in those operations. Resource managers should not view the result of this model separately, but consider all of the available population's information available.

Box 3a 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.
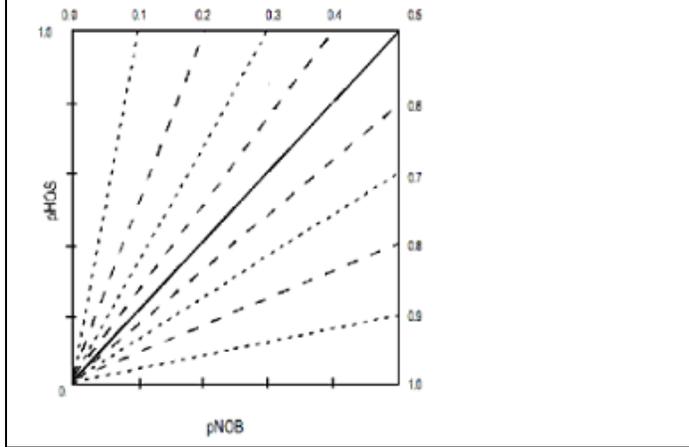
The relative influence of PNI on diversity (expressed as a change in fitness) increases substantially as hatchery operations progress for several generations (Box 3b). In general, the longer fish stay in a hatchery environment rather than in a natural environment, the greater the change in fitness (most agree it is a negative change). Incorporating natural origin fish into hatchery broodstocks, while minimizing hatchery strays onto the natural spawning grounds provides the highest PNI. While using natural-origin fish as broodstock is desirable, it should be noted that "mining" natural populations can result in other diversity concerns: direct selection of temporal portions of the run, or severe reductions in the effective size of the natural spawning population.

Within a single generation, the rate of change depends primarily on the duration of rearing (as a proportion of the life cycle and the importance of the life history stage to natural survival) and the rearing protocol utilized in the hatchery. Box 3b provides estimates of the effect of hatchery rearing and the PNI for the population on fitness over time (Busack et al. 2005). The lower 95% confidence interval bound was selected to give an ecologically conservative estimate of domestication effects. Thus the longer domestication selection affects a fish and its descendants the less well adapted to natural conditions the population is likely to be.

Box 3b. 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)

Lower 2.5 % Bound on Fitness over Time
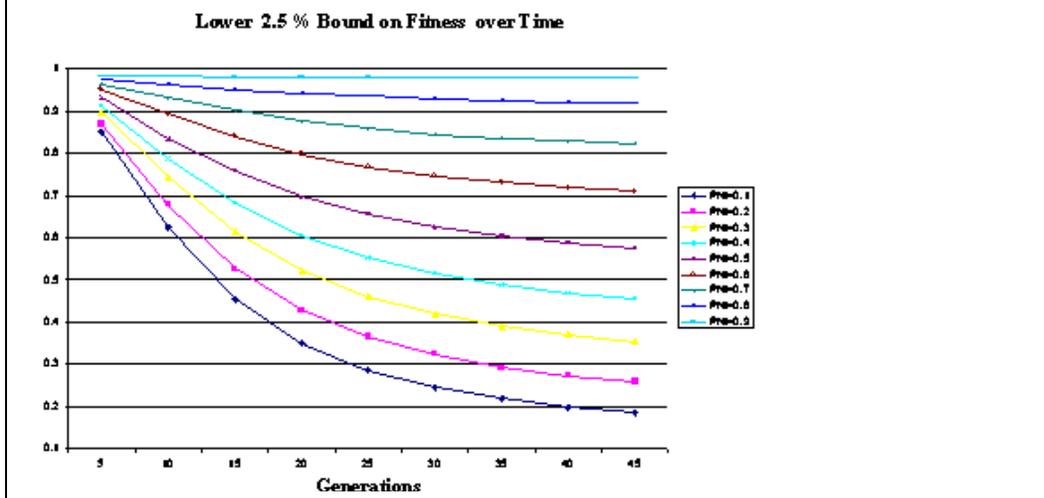
The decline in fitness, predicted or observed, presents a substantial challenge for recovery planners. Decreased fitness will lower productivity with predictable consequences for the viability of the population. Restoring fitness (local adaptation) could be a protracted process, depending on whether hatchery effects can be removed or the influence of hatchery rearing reduced substantially (by placing a larger portion of the population in the wild and/or making hatchery conditions more similar to natural conditions or reducing the duration of hatchery rearing). Furthermore, the more extreme the loss in fitness the more likely that some genetic variation (locally adapted alleles) will be lost completely, making a return to pre-hatchery conditions very difficult.

Box 3c. Loss of fitness over time (from Box 3b) and diversity score for populations affected by artificial propagation programs.

Percent Fitness Loss	Diversity Score
0.0 -2.5	4
2.5 – 5.0	3.5
5.0 – 10.0	3.0
10.0 – 15.0	2.5
15.0 – 25.0	2.0
25.0 – 45.0	1.5
45.0 – 65.0	1.0
65.0 – 85.0	0.5
> 85.0	0

For those populations where it is possible to estimate a long term PNI value, the loss in fitness (from 3b) can be converted to a Diversity Score for domestication (Table 3c). Few populations have been sufficiently monitored to provide accurate estimates of PNI. Additionally, the relatively recent advent of mass marking requires that estimates of hatchery stray rate and NOB introgression be used. For pre-1940 programs many researchers have assumed that there was a minimal return of hatchery fish and hatcheries were simply mining the natural populations (this may be especially true of steelhead and coho programs). While most salmon and steelhead hatchery broodstocks have been founded and maintained from a variety of sources, in general,

these transfers have been among populations within the same Stratum. For many of these populations the PNI metric may be useful.

In computing the PNI measure it may be necessary to make a number of assumptions. For many hatcheries that are operated as "isolated" programs the PNI would be near 0. In using Box 3b, it is assumed that isolated programs experience some genetic "leakage" and a PNI of 0.1 is assumed. Prior to the initiation of mass-marking programs estimates pHOR were based on the expansion of a few coded-wire-tag recoveries from the spawning grounds. Hatchery estimates of pNOB are seldom available. Where necessary, pNOB can be estimated to have a maximum value of 1-pHOR (unless the hatchery is sited near the natural spawning grounds or if the hatchery trap is in the migratory corridor pNOB is likely to be less than 1-pHOR). Additionally, where hatchery programs have been dramatically modified or terminated there can be a considerable variation in the PNI over time. PNI estimates can be combined using a number of methods with very different results (see Box 3d). Presently, there is very little information to predict the recovery rate for a population's PNI. The rate will depend of the intensity of selection in the natural environment and the "intactness" of genetic variation in the population. For example, ODFW suspended winter-run steelhead artificial production in the Upper Willamette River in the late 1990s. Monitoring over the next several years should provide valuable information on the recovery rate for local adaptation. This recovery might be reflected in increasing productivity if environmental changes can be adjusted for.

Box 3d. Change in PNI over time using arithmetic and harmonic averages. PNI was estimated for a hypothetical population that initially had a large scale production hatchery with a high proportion of hatchery strays (PNI < 0.15) followed by the termination of the hatchery program (PNI becomes 1.0). For the 20 year hatchery cycle, the PNI for 5 years was repeated. The recovery of the PNI is estimated over time using arithmetic (A) or harmonic (H) means.

PNI					Hatchery Free Years	PNI
0.05	0.05	0.12	0.06	0.07	0 yrs post hatchery	0.069 (A)
5 yrs of hatchery prod. as above					0 yrs post hatchery	0.062 (H)
5 yrs of hatchery prod. as above					12 yrs post hatchery	0.726 (A)
5 yrs of hatchery prod. as above					12 yrs post hatchery	0.186 (H)
5 yrs of hatchery prod. as above					25 yrs post-hatchery	0.844 (A)
5 yrs of hatchery prod. as above					25 yrs post-hatchery	0.287 (H)
20 yrs of hatchery prod. as above					12 yrs post hatchery	0.418 (A)
20 yrs of hatchery prod. as above					12 yrs post hatchery	0.097 (H)
20 yrs of hatchery prod. as above					25 yrs post-hatchery	0.586 (A)
20 yrs of hatchery prod. as above					25 yrs post-hatchery	0.131 (H)

Mechanisms for Adaptation
Diversity provides a population with the ability to maximize the exploitation of ecological niches (as described in the Habitat Diversity Metric), and to adapt to environmental changes that may occur in the future. Being unable to predict how a population will respond to future changes, one may instead be able to assess whether the processes that will enable the population to adapt are functioning or at least not encumbered. This metric addresses a vast array of diversity elements not included in the hatchery/domestication criteria. Simply, this metric includes those factors that influence the underlying genetic variation in a population. It includes both natural and

anthropogenic selection, changes in neutral genetic markers, and effective population size. One should consider the relative significance of potential factors, including (but not limited to):

1. Selection effects due to anthropogenic selection
   a. Harvest size selection (gill net or size catch limits)
   b. Harvest temporal effects on emigration or return migration (openings or closings)
   c. Selective passage upstream or downstream, temporal or physical
   d. Non-random selection of hatchery broodstock
   e. Other anthropogenic effects
2. Selection effects due to environmental modification
   a. Low water conditions delaying migrations
   b. Thermal barriers to migrations
   c. Temporal physical barriers (diversions, etc)
3. Reductions in effective population size (Ne)
   a. Small Ne (<50, 100, 250)
   b. Alteration in age structure (preponderance of single age classes or loss of age class)
   c. Shifts in sex ratio
4. Changes in genetic markers
   a. Violation of Hardy-Weinburg
   b. Significant change in gene frequencies
   c. Loss of rare alleles

Each of these factors can be readily quantified individually; but it is more difficult to objectively relate the above changes to viability scores (risk of extinction), or to combine these factors into a single score. Additionally, various anthropogenic effects could be additive, multiplicative, or even potentially negate the diversity effects of one another. For the near term, professional judgment will be needed to derive a single score for this metric based on the magnitude and potential impact of each of the factors.

Effective population size, Ne, the number of fish that actually contribute to subsequent generations, is dependent on a number of factors. Changes in census size, sex ratio and variation in family size can be related to a number of human related effects. For example, hatchery supplementation can result in the disproportinate representation of some families that were reared as juveniles in the hatchery over their naturally reared counterparts (although later differential survival may even this out some). Past hatchery practices of using a single large male to fertilize eggs from several females also dramatically decreased the effective population size. Catastophic events can create several redutions in Ne that have lasting negative diversity effects. Luckily, in those species that spawn at multiple age classes, this effect is greatly diminished. While Ne can be computed specifically for each population, one can approximate the value vary from one-fourth to one-half of the census spawning population.

Within the Diversity Score, anthropogenic mortality includes the selective effects of a variety of non-natural sources. Harvest related mortality is one component of this score, especially mortality from fishing gear that is selective for size (i.e., gill nets). Fishing seasons can exert temporal selection on a run of salmon or steelhead. Alterations in flow or passage can also also result in the selection for or against specific run times. For example, seasonal spill over dams can

improve downstream smolt survival for a portion of the outmigration. Alternatively, seasonal water withdrawls can influence the surival of outmigrants or over-summering juveniles. Summer Chinook salmon in the Yakima River were thought to have been extirpated by high water teperatures encountered by returning adults due to irrigation withdrawls.

Box 4a: Examples of diversity component scores. Component scores would be combined via a geometric mean. For this example N = 2Ne.

Effective Population Size	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

Anthropogenic Mortality Rate (%)[2]					
> 95%	x				
80%-95%		x			
45%-80%			x		
20%-45%				x	
< 20%					x

[1] Within population stray rate is accounted for in the RAMP model
[2] 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).

Local adaptation can also be affected by the introgression of non-native spawners into naturally-spawning populations. Historically, some level of interbreeding certainly existed between populations; however, the establishment of hatchery populations using non-native broodstock sources and the disruption of migratory paths due to habitat degradation has likely increased the degree of gene flow between populations by several fold. The more distant the relationship between a population and the source population of the straying fish, the less likely that the straying fish will contribute genes that are beneficial to the receiving population. Thus, stray fish from a population within the ESU, but from a different strata, are likely to have a smaller deleterious effect than fish straying in from outside of the ESU. Alternatively, the more distinct the stray fish are from the receiving population the less likely they are to interbreed with local fish (because of differences in spawn timing, body morphology, or other behavioral characteristics). It should be underscored that there are other potential effects from straying fish not directly related to the diversity criteria.

Box 4b: 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:
$(.67) * (2 \text{ [w/i ESU@20\%]}) + (.33) * (3 \text{ [out of ESU@10\%]}) = 2.3.$
Remember that the stray rate is based on the proportion of effective (spawning) non-local fish.

Diversity Criteria

In the end, each of the diversity metrics needs to be combined into one score. The diversity metrics are not completely independent measures, but roughly capture different aspects of a population's diversity. Current metrics are not sufficiently quantitative to stand up to statistical analysis nor produce a meaningful numbers of significant digits, but will initially rely on some qualitative index of uncertainty to facilitate consolidating the metrics, possibly through a weighting process. At this stage, with the diversity criteria in a very rough form, most of the emphasis is on identifying the right parameters to evaluate. It may initially require the use of expert opinion to produce a single diversity value. In the interim, these guidelines can also be used to direct monitoring efforts and data collection prior to the development of more objective system(s) of evaluation. Better models will come later, but not until we can better identify which of the thousands of variables is more important to population sustainability.

In Part 3, we provide an example of how the diversity criteria would be applied to LCR coho populations. The paucity of information for many populations requires that we rely on expert opinion to score a number of the metrics and apply weights to each score based on the quality of the underlying information used to arrive at that score.

Spatial Structure Criteria

Due to time constraints, we have not yet revised the spatial structure criteria from the 2003 report. The 2003 report criteria describe general principles more than specific metrics. We continue to support and develop metrics that are consistent with the criteria of the 2003 report.

Threats Criteria

Overview

NOAA Fisheries has developed a multi-part approach to making delisting decisions for Pacific salmon (Figure 51). This framework provides for a biological viability assessment focusing on the VSP population parameters (abundance, productivity, spatial structure and diversity) and a threats assessment focusing on environmental factors responsible for the ESU's threatened status. These threat focused criteria have been referred to as "threats criteria", "listing factor criteria" and "limiting factors criteria". The previous sections of this report have focused primarily on the biological viability component of this process. Although it is not a comprehensive look at the threats assessment part of the process, in this section we provide some discussion and criteria recommendations involving the interaction between these two parts of the decision framework. In particular we focus on the concept of stationarity and its importance in developing viability criteria.
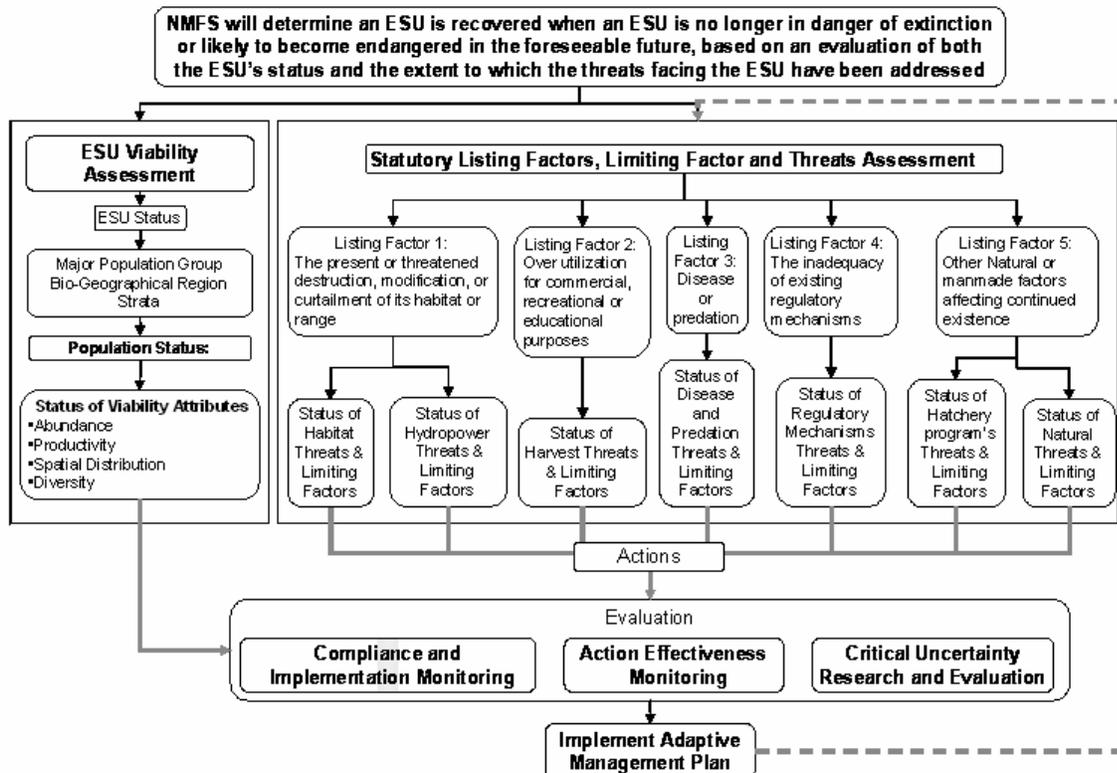

Figure 51 NOAA Fisheries framework for delisting decisions for Pacific salmonids (draft 8 January 2006).

Stationarity

The purpose of viability criteria is to describe metrics and threshold conditions that indicate when a population or ESU is viable. In general, we expect estimates of a population's abundance and productivity to be one of our best indicators of a population's viability. Abundance and productivity measures should integrate the salmon's entire experience over the entire life-cycle. Therefore, if there is a problem in any aspect of the salmon environment, it should ultimately be reflected in the performance of the salmon population itself. For example, any habitat problems, or over harvest, or negative hatchery interactions should show up in abundance and productivity metrics.

However, abundance and productivity metrics have some important limitations. Assessing a population's abundance and productivity can only provide information about how the population has performed in the recent past. To evaluate whether a population is viable, we need to make some predictions about how the population will behave in the future. Frequently, we make the assumption of "stationarity" – that is, the population will behave in the future similar to the way it has in the recent past. This assumption is predicated on the assumption that all the factors that affect abundance and productivity (e.g., habitat processes, harvest patterns, etc.) will continue in the future the way they have in the recent past. It is possible for a population to meet all of the abundance and productivity criteria yet still be considered at risk of extinction if the stationarity assumption is likely to be violated.

From an abundance and productivity metric perspective, we need environmental criteria to provide a metric for evaluating the stationarity assumption. The spatial structure and diversity criteria can also be viewed in terms of providing tests of the stationary assumption. For example spatial structure criteria provide tests of assumptions about catastrophic risk (a violation of stationarity) and diversity criteria provide tests of assumptions about changing evolutionary trajectories (another stationarity violation). In looking at the interaction between viability and threats criteria, we can ask whether environmental conditions in the future are likely to be significantly different than they have been in the recent past. This is crucial information for a complete risk evaluation and it cannot be obtained by simply looking at abundance and productivity. Table 17 illustrates how a prediction of future habitat condition interacts with the abundance and productivity metrics. A combination of an acceptable abundance and productivity metric and a predicted decline in environmental condition (* in the table) could result in a determination that a population is not viable, depending on the magnitude (and certainty) of the predicted decline. Theoretically, under extremely limited circumstances, a population with an unacceptable abundance and productivity metric but a predicted improvement in habitat (** in the table) might be considered viable, but this would be nearly impossible to demonstrate with any level of confidence and we would consider such a population at high risk of extinction.

Environmental condition as assessed by fish performance over, say, the last 20 years may not reflect fish response to future habitat conditions for several reasons. Long time lags in changes to environmental conditions may make it impossible to predict the future by just looking at fish. For example, it may take a long, but predictable, time for sediment to accumulate to some critical level in a particular reach. Ignoring such an important piece of available information would lead to an incomplete viability assessment. Another example might be the long-term trends in habitat created by global climate change that are unlikely to be reflected in current measures of abundance and productivity. In addition to these long time lag events, a short-term (e.g., 20 year) assessment of abundance and productivity may not be good predictor of the future habitat

conditions because they are too short to include rare events. Rare events, such as catastrophic mud slides or invasions by exotic species, may be generally predictable, but not reflected in short-term measures of abundance and productivity. Finally, short-term measures of abundance and productivity may not be reflective of future environmental conditions if management actions affecting habitat, harvest, hydro operations, etc. are predicted to change. As an extreme example, a population could have excellent abundance and productivity performance, but if it is known that an obstruction will be built at the mouth of the river in the following year, we should conclude that the population is at high risk.

Table 17 Conceptual relationship between predicted environmental condition, abundance and productivity metrics and population viability.

		Predicted Environmental Condition		
		Will Improve	Will Not Change	Will Decline
Abundance and Productivity Metric	Acceptable	Viable	Viable	Not Viable*
	Not Acceptable	Not Viable**	Not Viable	Not Viable

Habitat Criteria

Habitat criteria are a subset of threats criteria focusing on watershed condition. In setting viability and threats criteria, we are looking for a relatively small number of metrics that provide a clear indication of a population's extinction risk. If we use abundance and productivity criteria as a primary direct measure of population performance in the recent past, we can use habitat criteria as one of our primary tests of the stationarity assumption. The stationarity assumption test is an evaluation of what the habitat will be like in the future. There are three categories of metrics that we might look at in predicting future habitat condition: 1) habitat trends, 2) catastrophic risks, and 3) habitat management/land use.

Evaluating trends in habitat condition (e.g., stream temperature, flow dynamics) can provide a test of the abundance and productivity stationarity assumption. If habitat condition has been declining in the recent past, it may be reasonable to assume that it will continue declining at a similar rate in the future. In using habitat condition trend metrics, it is important to consider both the *magnitude* and the *certainty* of any decline. Because trend in habitat condition can be defined in strictly biological/physical terms, there is a reasonable understanding of which habitat attributes are important for salmon, and the problem is amenable to fairly straight forward statistics, we have some hope of developing concrete metrics. However, as discussed in the habitat trend criteria section below, a host of issues remain.

Populations may be subject to predictable catastrophic risks that are not captured in an assessment of habitat trend or in abundance and productivity metrics. Such risks might come from forest fires, volcanic eruptions, toxic spills, global climate change, potential exotic species introductions, etc. While it may be difficult or impossible to frame specific *a priori* criteria for evaluating the risk from potential catastrophic events, a thorough evaluation of such risks should be completed during recovery planning, monitored over time, and re-evaluated at the time of viability assessment.

Shifts in habitat management and land use are probably the most difficult (and arguably most important) stationarity assumption to evaluate. Since landscape management priories can shift quickly based on political, social, technological and economic changes, it is difficult to decide what assumptions to use in predicting future habitat management and land use activities. These

changes cannot necessarily be predicted based on trends in habitat condition. Despite the difficulty in prediction, estimating future habitat management and land use activities is important to consider in evaluating risk because the activities have such a profound impact on salmon habitat. Providing specific *a priori* metrics and thresholds for determining if a population is at risk because of likely future changes in habitat management or land use is not possible (at least not in the scope of this report). However, a thorough evaluation of such risks should be completed at the time of viability assessment. The evaluation will not be strictly biological as it must consider all the factors (e.g., social, economic, legal) that go into the prediction.

Habitat Trend Criteria

General habitat attributes that can be relatively directly related to salmon performance and therefore are good candidates for habitat trend metrics are shown in Table 17. This table presents broad categories, not specific metrics. In developing specific metrics, it is important to take into consideration the natural spatial and temporal variability of the attributes. For example, an observed increase in fine sediment over time could be a useful indicator of future population extinction risk. How do we measure the change in fine sediment? There is natural temporal and spatial variation in how fine sediment is deposited within a single river and different rivers have different spatial and temporal patterns. One approach is to develop "benchmarks" that can be used to craft watershed specific measures relevant to salmon. For example, if it is known that fine sediment loads greater than X cause mortality in salmon fry, the benchmark X can be a useful reference point in creating a custom metric for a particular population. The best metric might be something like the fraction of reaches that exceed the benchmark at a given time of year, or the number of days a year that the benchmark is exceeded in a particular reach. Although the benchmark may represent a standard applicable throughout the domain, a specific metric based on the benchmark would be developed for a specific watershed based on the watershed's particular dynamics, monitoring feasibility, etc. Benchmarks that represent critical (e.g., lethal) thresholds may be particularly useful, since increasing trends in either the area in a watershed or time the watershed exceeds the benchmark is likely to be informative about future habitat condition with regard to salmon. In a section below, we more fully develop an example of benchmarks for a particular habitat attribute (maximum temperature) and describe in a general sense how it can be converted into watershed specific metrics.

The general habitat attributes that are included in Table 18 reflect instream habitat conditions and watershed processes (e.g., flow dynamics) as these will integrate the effects of landscape and human activities. In practice, it may be useful to look at surrogates for these attributes, but it is difficult (and perhaps detrimental) to specify exactly what surrogates may be appropriate for the future. For example, there is a correlation between the percent of impervious surface in a watershed and sediment levels (ref.), so that the current percent impervious surface may be a good surrogate for sediment (i.e., trend in impervious surface might be a good metric for a current status evaluation). However, changes in land use practices may decouple this relationship, but we are reasonable confident that there will always be a relationship between fish population status and sediment, so we focus on the primary metric of sediment.

Table 18: List of habitat attributes to be considered in the formulation of habitat criteria/guidelines (modified slightly from NMFS 1996).

Streamflow
Water Temperature
Sediment

Dissolve Oxygen
Chemical Contaminants
Nutrient Deficiency
Physical Barriers (fish passage)
Habitat Attributes
Pools
Large Woody Debris
Substrate
Off-Channel Refugia
Interactions with Exotics
Watershed, Stream Channel, Estuarine Conditions
Streambank stability
Floodplain Connectivity
Tidal Flat Connectivity
Channel Width/Depth Ratio
Disturbance History
Riparian Area Condition
Drainage Network Increase

Once an appropriate habitat attribute metric has been defined, it is necessary to determine a statistic for evaluating a trend in the metric and finally to associate thresholds in that trend metric with a specific extinction risk (criteria = metric & threshold). One simple approach is to use a basic linear regression, which is probably a reasonable default. However, natural temporal autocorrelation could lead to erroneous conclusions if using a simple linear regression and must be considered. (As an illustration of the complexities introduced by temporal autocorrelation, consider the adjustments needed in the abundance and productivity criteria because of autocorrelation in marine survival.) Identifying a trend threshold associated with a specific extinction risk is particularly challenging. How steep of a habitat decline is associated with a high risk population? How much statistical confidence is appropriate? How do the habitat trend metrics interact to give an overall indication of population risk? We are not providing answers to these questions in the current report, but note that these issues will need to be address during a viability assessment.

Given the above discussion, the steps for development and application of habitat trend viability criteria might be:

1. Identify benchmarks for specific habitat attributes that are based on the attribute categories in Table 18.
2. Describe watershed/population specific metrics using the benchmarks.
3. Monitor the metrics over time.
4. Statistically evaluate the temporal trend in the metrics, paying attention to potential autocorrelation patterns.
5. Decide the extinction risk associated with the trend. (In general, we have too little information to set thresholds *a priori*.)

In applying this approach, it would be necessary to prioritize attributes for benchmark development and monitoring. It would likely be impractical to attempt trend monitoring of all the attributes in Table 18. Highest priority for benchmark development and monitoring should focus on the attributes considered most limiting for each population. Identifying the most limiting features is an important part of the recovery planning process but is it outside the scope of this report.

Implementation Goals

In addition to habitat trend criteria it will be useful for a recovery plan to set goals for habitat conditions that are considered needed to support local population and ESU recovery. These could be referred to as "implementation goals" A complete discussion of developing this type of goal is beyond the scope of this report. However, it is useful to note that many of the steps involved in the development of a habitat trend viability criteria are similar to what is needed for the development of implementation goals. The "benchmark values" can be important reference points for setting habitat recovery goals. Developing a population specific metric to monitor for habitat trend requires a careful consideration of the natural spatial and temporal dynamics of the watershed – the same information necessary for developing implementation goals. In fact, in some cases, the same basic metrics could be used for goals and viability - specific targets for the metrics could be set as habitat planning goals whereas the viability criteria are concerned with trends in these metrics. For example, an implementation goal for a watershed might be to reduce fine sediment below the threshold in 75% of the reaches, and the viability criteria might focus on the temporal trend in the fraction of reaches that exceed the benchmark.

An Example: Maximum Stream Temperature

The development of habitat stationarity criteria and implementation goals can be explored by considering one habitat attribute (maximum temperature) as an example. This example illustrates both the challenges and (most importantly) feasibility of identifying appropriate metrics. One of the challenges is deciding how to distinguish between viability criteria, habitat trend criteria, implementation goals etc. We have made recommendations for these distinctions below, but recognize that the exact language (and regulatory framework) will depend on management decisions.

Maximum temperature tolerances are a particularly accessible metric of considerable regional significance (Richter and Kolmes, 2005). In many streams that once were inhabited by large salmon runs, temperature regimes are now inhospitable. An important factor in the recovery of salmonid populations is the restoration of temperature regimes (Poole et al., 2001a, b, 2004). Salmon recovery poses an enormous challenge due to competing societal priorities and regional population growth (Lackey, 2003; Lichatowich, 1999) and projections concerning climatic change that will alter the hydrology of the Pacific Northwest (Mote et al., 2003)

Human activities can affect thermal regimes by simplifying the physical structure of aquatic systems, thereby eliminating natural thermal buffers and insulators (Poole and Berman, 2001). Clearing and developing land, dredging or straightening streams, grazing and other land-use activities influence temperature regimes by altering factors external to the stream, and the amount of water flowing in the stream (Poole et al., 2001a, b). These activities often directly or indirectly simplify the structure of stream channels or riparian zones, as has occurred in the lower Willamette River, Oregon (Sedell and Froggatt, 1984). This type of channel simplification can potentially increase temporal variability and decrease fine-scale spatial variability in stream temperature, both of which may have negative consequences for salmonids (Poole et al., 2001b, Poole and Berman, 2001). Removing riparian vegetation in small streams, where shading is important, can increase daily variation in stream temperature (Beschta, 1997), and empirical manipulation of streams reveals complex relationships between maxima, minima, and mean temperatures and different temperature drivers (Johnson, 2004). For streams where groundwater buffers temperature, change in groundwater temperature or flow dynamics can alter the seasonal

availability of cold water, increasing seasonal variation in water temperature (Poole and Berman, 2001). Altered flow regimes with concomitant effects including temperature changes, and regional expressions of global climate change, have influenced chinook and steelhead migration over the last 50 years (Brannon et al., 2004).Water temperature is an indicator of habitat quality, acting as an integrator of what is happening in a watershed (Poole et al., 2001b).

Thermal refugia are important in maintaining salmonid populations because salmonids may be exposed to stressful or lethal temperatures for part of the day when daily variation in stream temperature is high. Small-scale thermal refugia provide important habitat for salmonids during periods of warmer water temperatures (Berman and Quinn, 1991; Ebersol, 2002; Gibson, 1966; Kaya et al., 1977; Torgerson et al., 1999), and changes in temperature extremes, or mean temperature, can result in loss of the refugia and therefore the salmonids. At peak summertime temperatures, only a patchwork of fish habitat in some streams may be cool enough (Cavallo, 1997; Kaya et al., 1977). Loss of riparian vegetation, the elimination of beaver populations, removal of large woody debris, channel simplification, reduced groundwater discharge due to changes in upland vegetation or urbanization, water withdrawals, and other human activities cause the loss of the fine-scale spatial distribution of appropriate thermal habitats upon which salmon rely (Poole et al., 2001b). This can cause fish to migrate greater distances to find appropriate habitats or not find them at all.

In the same way, seasonal variation in temperature can create thermal barriers to salmonid immigration and emigration. Human activities can increase the coarse scale temporal variation of streams, exposing salmonids to extremes beyond the normal range of variation and resulting in habitat fragmentation and elimination of large, well-connected tracts of high-quality thermal habitat. This habitat fragmentation has been shown to degrade both fish population structure and persistence (Dunham and Reiman, 1999; Poole and Berman, 2001; Poole et al., 2001b; Reiman and Dunham, 2000).

In a recent U.S. Environmental Protection Agency (EPA) document entitled EPA Region 10 Guidance for Pacific Northwest State and Tribal Water Quality Standards (EPA 910-B-03-002), (Environmental Protection Agency, 2003), the EPA recommended a multifaceted approach for state and tribal temperature standards to support native salmonids. This approach includes the adoption of: (1) new or revised numeric water quality temperature standards to protect salmonids at each life stage, and a set of "uses", including spawning, egg incubation, fry emergence, juvenile rearing, smoltification, and migration. It also recommended (2) criteria that focus on summer maximum temperature conditions, related to human activities, that are the greatest water temperature concern in the Pacific Northwest (see also Lackey, 2003). These would include criteria to protect temperature-sensitive salmonid uses at times of the year in spring-early summer or late summer-fall where additional protection is required. Provisions were also made (3) to protect water temperatures that are currently colder than the numeric criteria, to protect the last strongholds of ESA - listed salmonids. Finally provisions were made (4) to protect salmonids from thermal plume impacts, to prevent instantaneous lethal temperatures, thermal shock, migration blockage, and other adverse impacts on sensitive life stages.

The EPA Guidance (EPA, 2003) provides for case-by-case EPA reviews of situations where the numeric criteria are unachievable or inappropriate, for example in cases where a use-attainability analysis indicates that high natural background temperatures might prevent criteria from being achieved.

The EPA (2003) recommends that temperature-limit criteria be based upon upper optimal physiological temperature preferences known to support requisite biological processes of recognized salmonid life-history stages. Fish-habitat relationships indicate a critical need for criteria to help direct human activities so that habitat conditions are prevented from continuing to deteriorate in a region where climatic cycles can temporarily mask underlying processes (Anderson, 1998; Beschta et al., 1987; Brannon et al., 2004; Chao et al., 2000; Hicks, 2000; Independent Science Group, 1996; Lackey, 2003; McCullough, 1999; McCullough et al., 2001; Mote et al., 2003; PNSHIWG, 1998; Poole and Berman, 2001; Poole et al., 2001a; Sedell and Froggatt, 1984).

We recommend a habitat trend criterion based on habitat trend (i.e., stationarity) for maximum temperature that uses the EPA upper optimal temperature values as benchmarks:

> For a viable population, there should be no biologically meaningful increase in the spatial extent or temporal frequency of salmon habitat in the watershed approaching or exceeding the 7-DAM (7-day average of the maximum daily temperature) maxima benchmarks in Table 19. A biologically meaningful increase will be a function of the spatial distribution of the high temperatures relative to salmon distribution, the magnitude of any trend, and the statistical confidence in the trend.

This is consistent with EPA's call (2003) for protection of high quality habitat whose water is currently colder than numeric temperature criteria. EPA (2003) believes that the thermally optimal waters that do exist will be crucial for the survival of ESA-listed salmonids, and that their additional warming would jeopardize the potential to control water temperatures in warmer habitats downstream. This requirement of thermal nondeterioration is a challenging target in the face of regional population growth (Lackey, 2003) and climate change projections (Mote et al., 2003). This criterion leaves intentionally vague the definition of a *meaningful* increase, referring to the spatial distribution of the high temperatures relative to salmon distribution, the magnitude of any trend, and the statistical confidence in the trend. In general, it is challenging to detect trends in short term data sets for highly variable (and autocorrelated) parameters, such as stream temperature. For example, twenty years of data or more may be necessary to avoid confounding by temperature oscillations driven by the Pacific Decadal Oscillation (PDO) (Anderson, 1998, Chao et al., 2000), which has considerable effects on climate in the Pacific Northwest.

In addition to the temperature trend viability criteria, we recommend as reference thresholds for implementation goals the benchmarks in Table 19. If a watershed exceeded these thresholds in a spatial/temporal pattern that affected salmon, recovery planning would need to propose ways to fix the thermal regime regardless of whether the direction of change is non-deteriorating.

Table 19 Upper Optimal Temperature Criteria/Guidelines

Life Stage	7-Day-Average Maximum Daily Temperatures
Spawning and incubation	13°C (55°F)
Core Juvenile rearing	16°C (61°F)
Non-Core Juvenile Rearing	18°C (64°F)
Adult migration[1]	18°C (64°F)

[1] EPA makes special provision for a 20°C limit with additional natural thermal regime narrative in cases where historical thermal limits can be shown to have exceeded 18°C, along with provisions for protection and restoration of the natural thermal regime including cold water refugia where possible.

| Smoltification except steelhead | 16°C (61°F) |
| Steelhead smoltification at fourth-level HUC[2] watershed | 14°C (57°F)[3] |

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[2] HUC = hydrologic unit code.

For all these goals and criteria, the significant challenge of defining the spatiotemporal range over which they should be applied remains (Richter and Kolmes, 2005). Those spaces occupied by threatened and endangered salmonids need to be regulated at the times of year that sensitive life stages are present, and defining the bodies of water involved and the times to apply the standards requires additional consideration and research. The complex life histories of salmonids, the variety of habitats used by their different life stages, and the spatially and temporally dynamic nature of the habitats involved, make this an enormous scientific undertaking (Brannon et al., 2004; Hicks, 2000; McCullough, 1999; McCullough et al., 2001; Poole and Berman, 2001; Poole et al., 2001a). Salmonid populations have evolved to intimately fit into the intricate thermal regimes of the Northwest (Brannon et al., 2004) making an understanding of the species-habitat relationships crucial in the face of accelerating regional (Lackey, 2003) and global (Mote et al., 2003) environmental changes, as well as natural periodic sources of variability (Anderson, 1998; Chao et al., 2000). Laboratory studies cannot fully substitute for field data, because of difficulties in replicating acclimation conditions, food availability, social interactions including territoriality, diurnal physiochemical periodicity, and the complexities of microhabitats accessible to fish in nature (Poole et al., 2001). Historic thermal regimes are often poorly understood, making evolutionary interpretations required for salmonid recovery efforts (Brannon et al., 2004) an even greater, although no less important, challenge (Poole et al., 2001).

Using use-attainability analysis, as prescribed by the existing Clean Water Act, there is no obligation to attempt to achieve unattainable conditions (in this case, to apply the temperature criteria). Use-attainability analysis can be brought into play in situations where numeric criteria cannot be achieved, and under limited circumstances (EPA, 2003) a State or Tribe can adopt a different use for that water, and temperature criteria sufficient to protect that new use. EPA (2003) indicated that the new use should be "the most protective salmonid use that is attainable," and that all uses attained since 1975 must be protected. A new use is considered "compromised" or "degraded", and may only be in effect for part of the year (e.g., summer) and "unqualified, healthy salmonid use may be attainable other times of year and therefore may be the appropriate use then" (EPA, 2003). Factors that might preclude attainment include dams and other hydrologic modifications that cannot be operated in compliance with numeric temperature standards, as well as pollution that would cause more environmental damage to remediate than to leave in its existing state (EPA, 2003). Appropriate applications of numeric criteria in conjunction with use-attainability analysis is discussed in EPA (2003) section VI.1.C, and thermal heterogeneity is discussed in Ebersol (2002). Evidence for nonattainability would include the anthropogenic factors involved (EPA, 2003) the density, size, and duration of thermal refugia (Ebersol, 2002); and data pertaining to the physiological consequence for the existing salmonid population.

Part 3: Current Status of Oregon LCR coho populations

LCR Coho Abundance and Productivity

Overview

The LCR coho ESU contains eight populations in Oregon (Figure 52). Of these, only two have sufficient data for a viability curve analysis; Clackamas and Sandy. Spawning surveys, particularly those conducted in 2002, 2003 and 2004 provide some information on the other Oregon populations in LCR. We start by presenting the time series data for Clackamas and Sandy, then the viability curve analysis. Next we discuss data for the other populations which are largely based upon spawning surveys. Finally we present the abundance and productivity summaries for the populations.
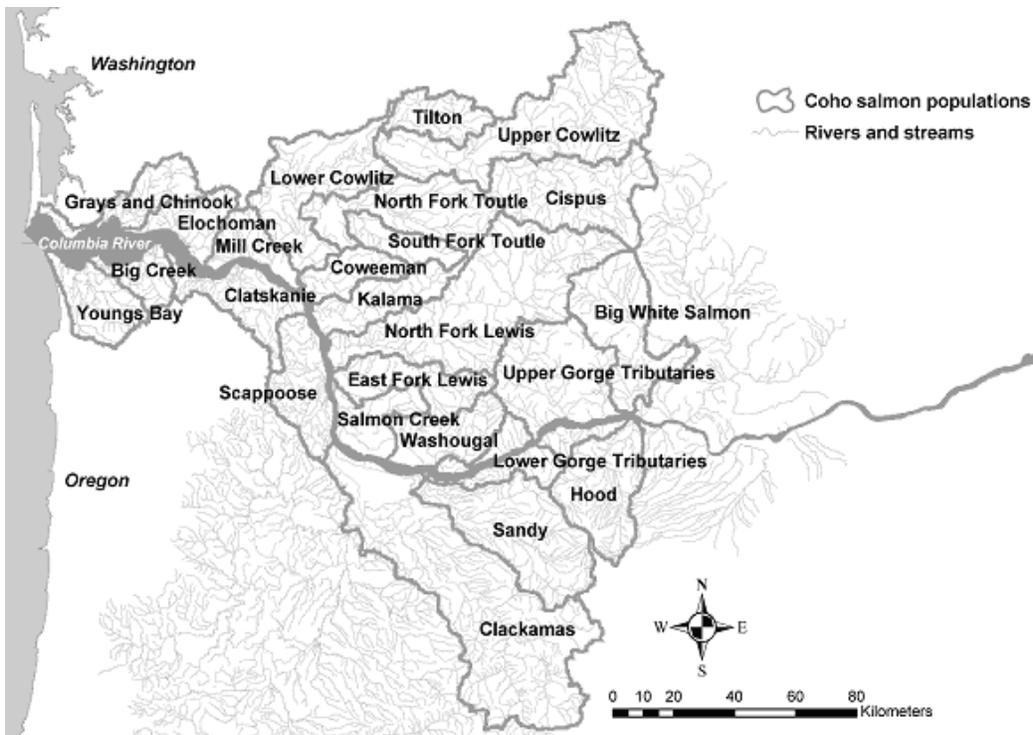
Figure 52 Map of LCR coho populations.

Time Series

In Figure 53 - Figure 62 we present time series information on spawner abundance, hatchery fraction, harvest rates, and recruitment for the Clackamas and Sandy populations.
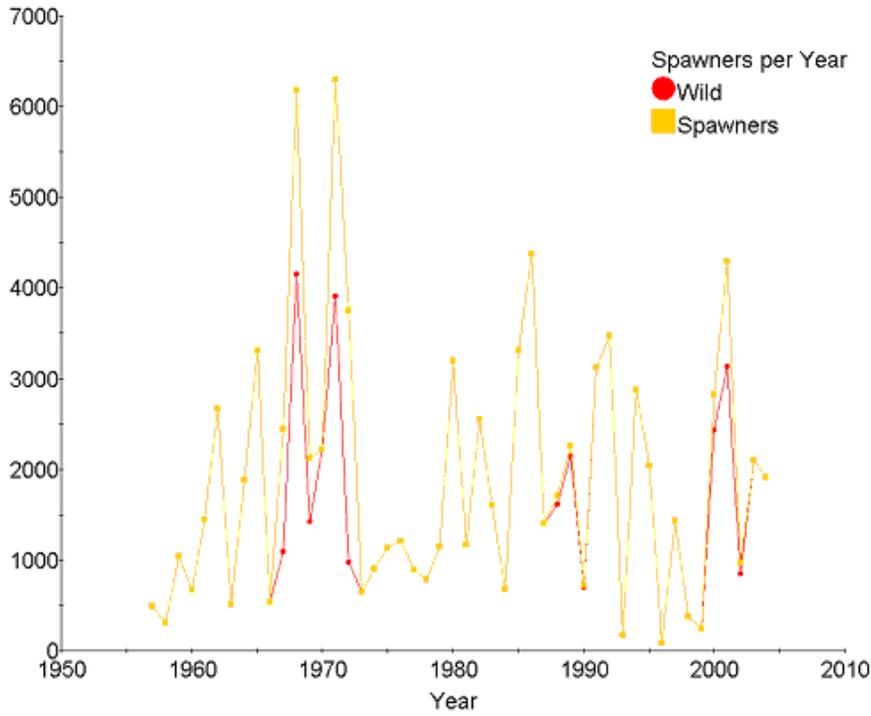
Figure 53 Clackamas coho abundance.
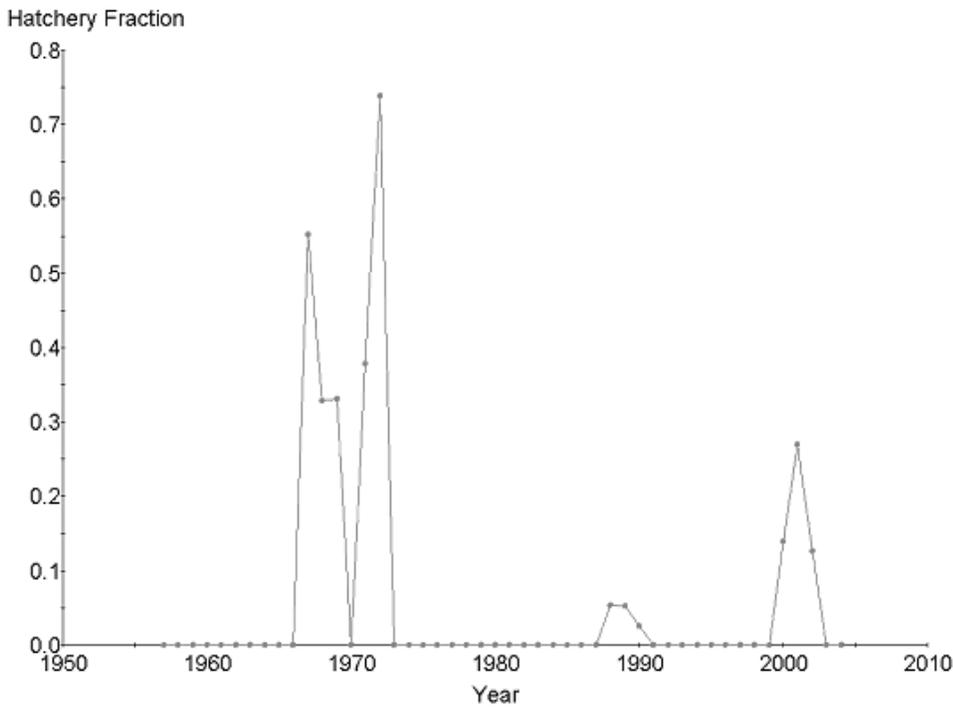
Figure 54 Clackamas coho fraction of hatchery origin spawners.
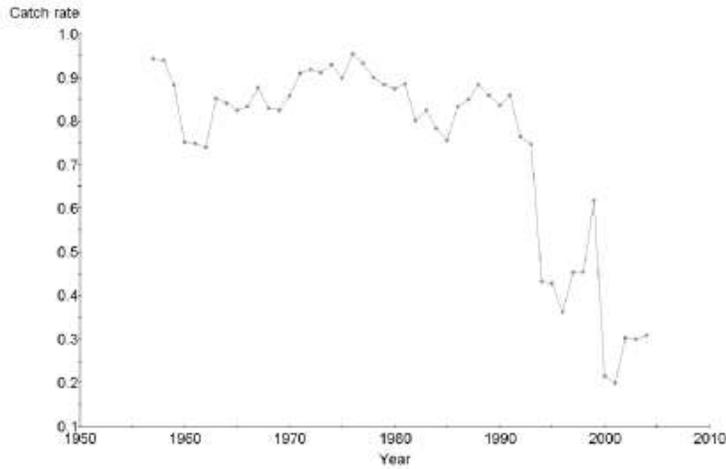
Figure 55 Clackamas coho harvest rate.
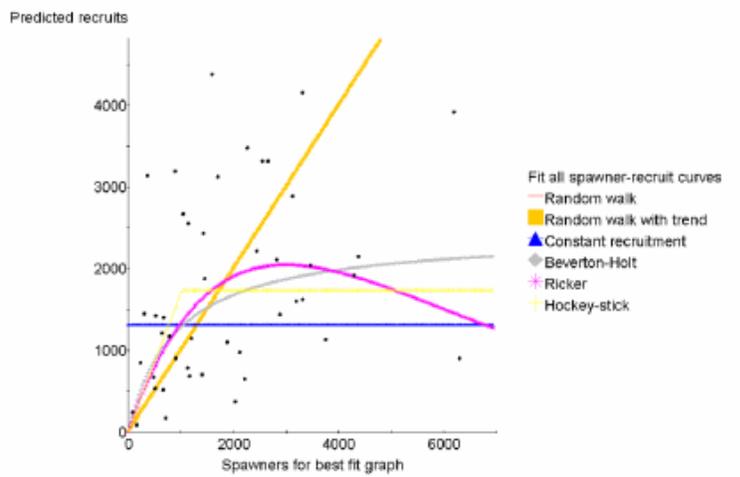
Figure 56 Clackamas coho post-harvest (escapement) recruitment functions.
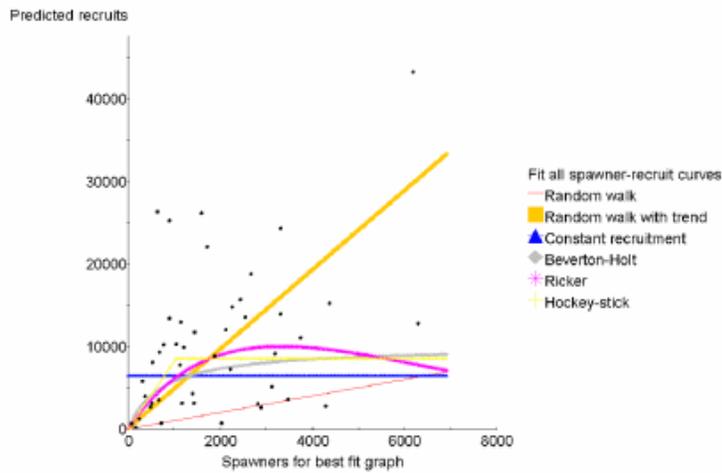

**Figure 57 Clackamas coho pre-harvest recruitment functions.**
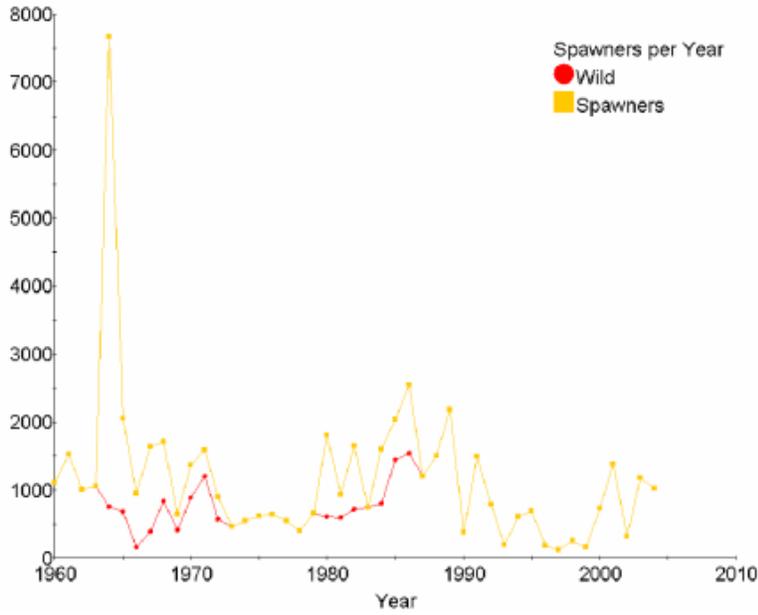
Figure 58 Sandy coho abundance.
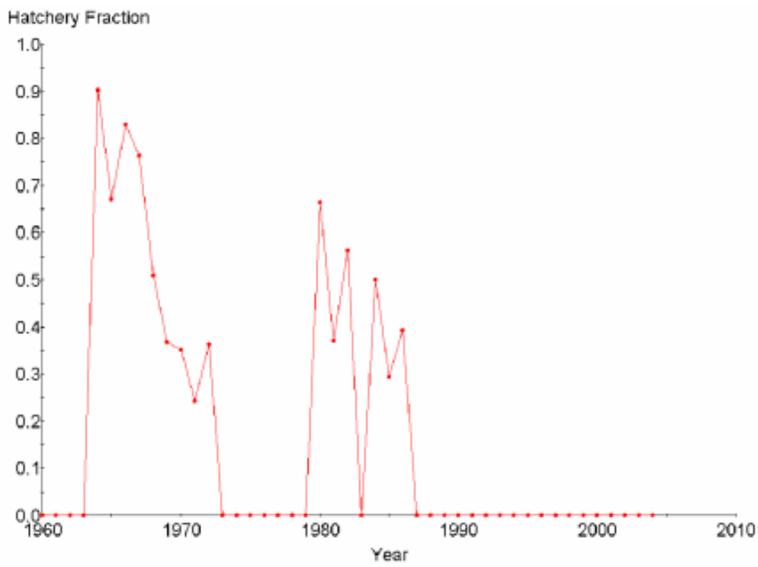
Figure 59 Sandy coho fraction of hatchery origin spawners.
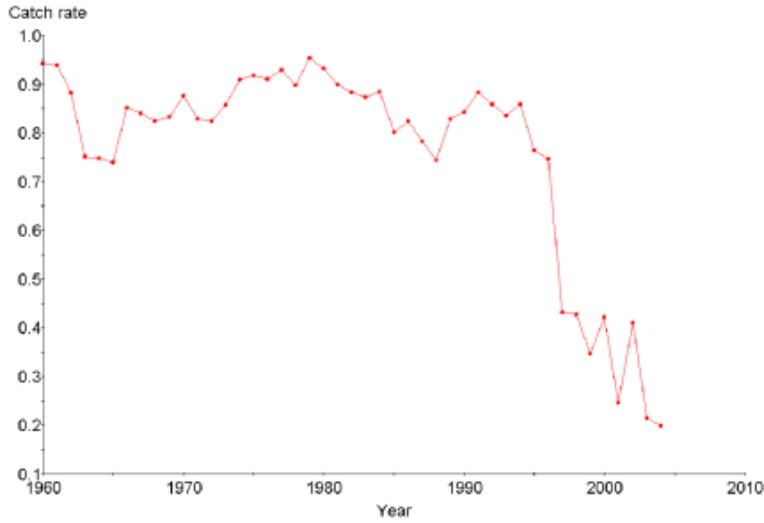
Figure 60 Sandy coho harvest rate.
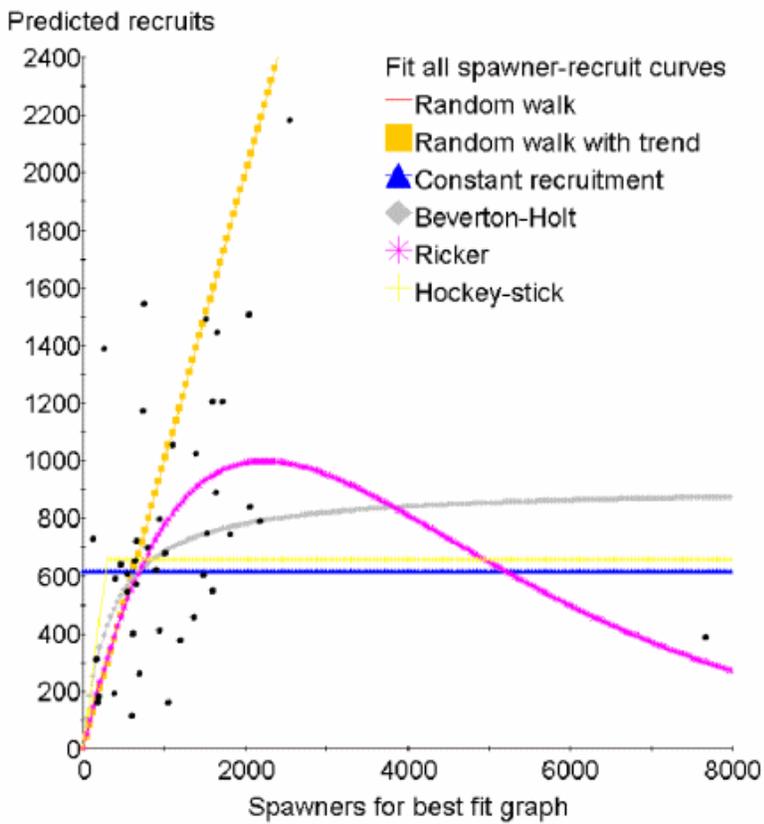
**Figure 61 Sandy coho post-harvest (escapement) recruitment functions.**
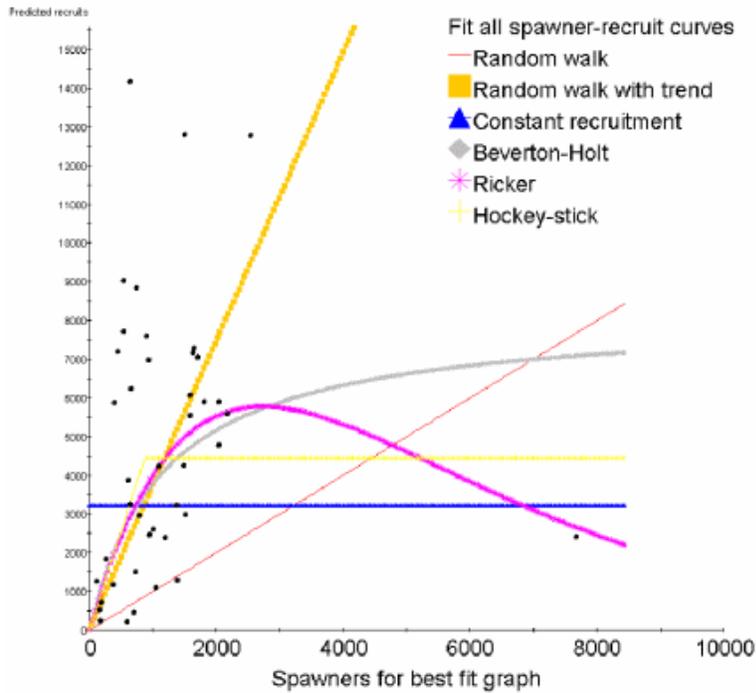
Figure 62 Sandy coho pre-harvest recruitment functions.

Viability Curves

Figure 63 - Figure 66 show the benchmark viability curve analysis for Clackamas and Sandy coho based on the data in Figure 53 - Figure 62.
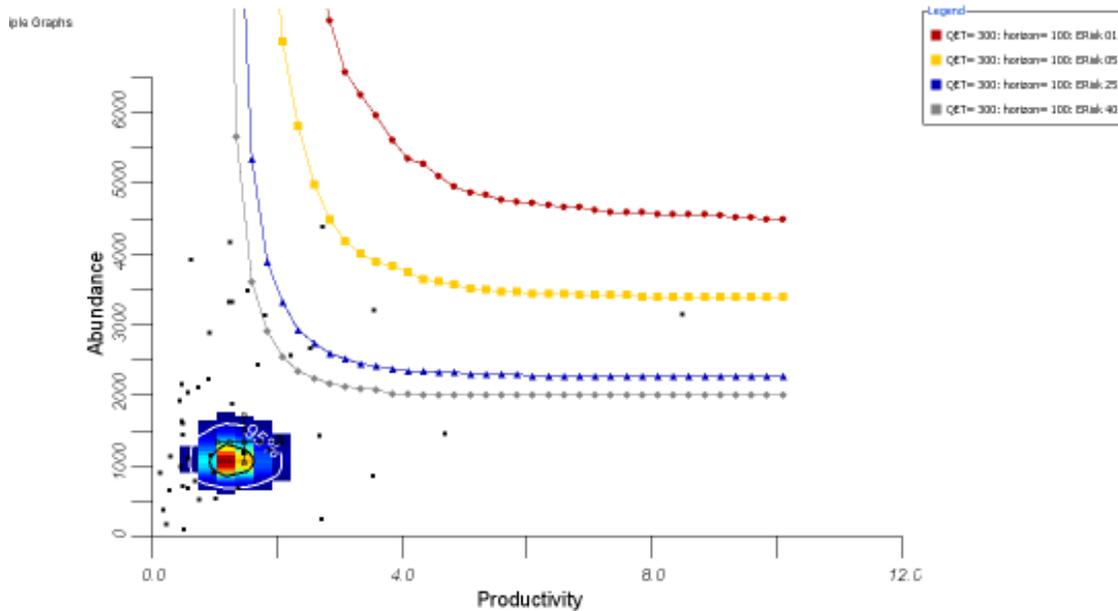

**Figure 63 Clackamas coho escapement benchmark viability curves.**
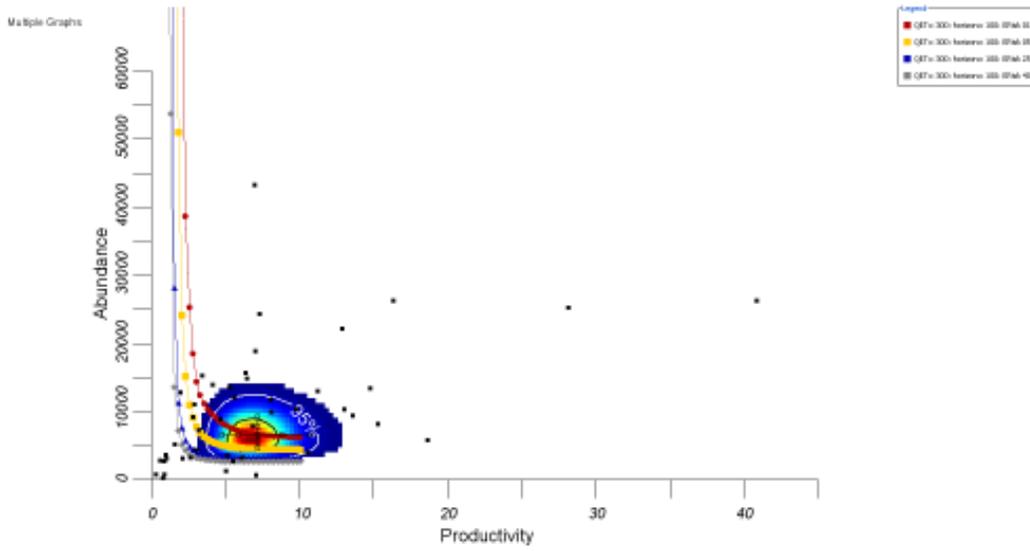
**Figure 64 Clackamas coho pre-harvest benchmark viability curves.**
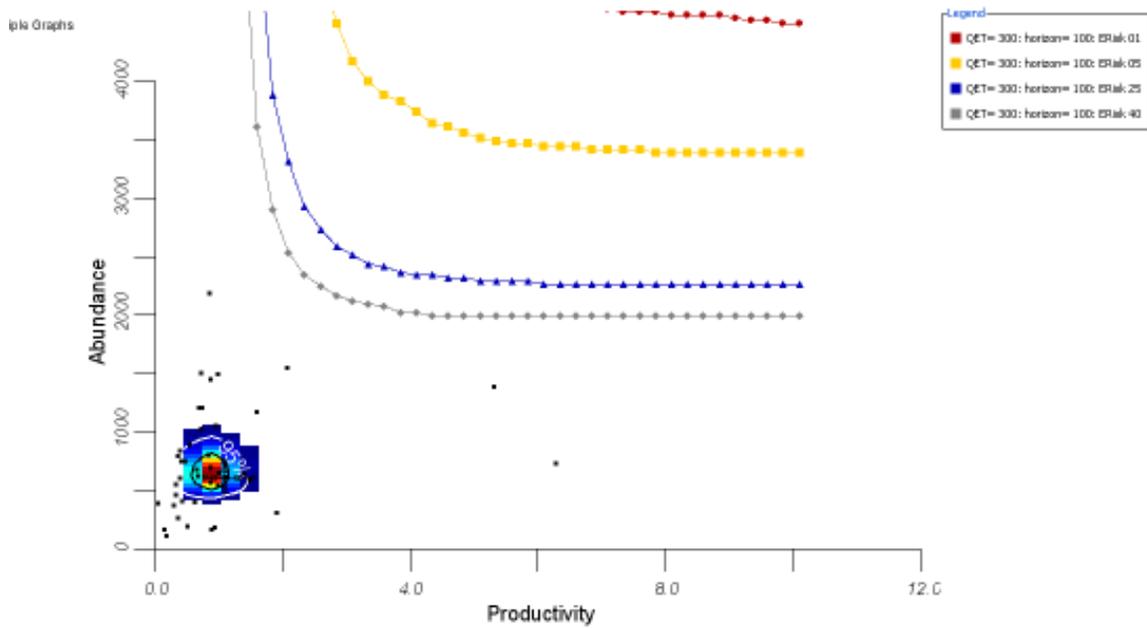
**Figure 65 Sandy coho escapement benchmark viability curves.**
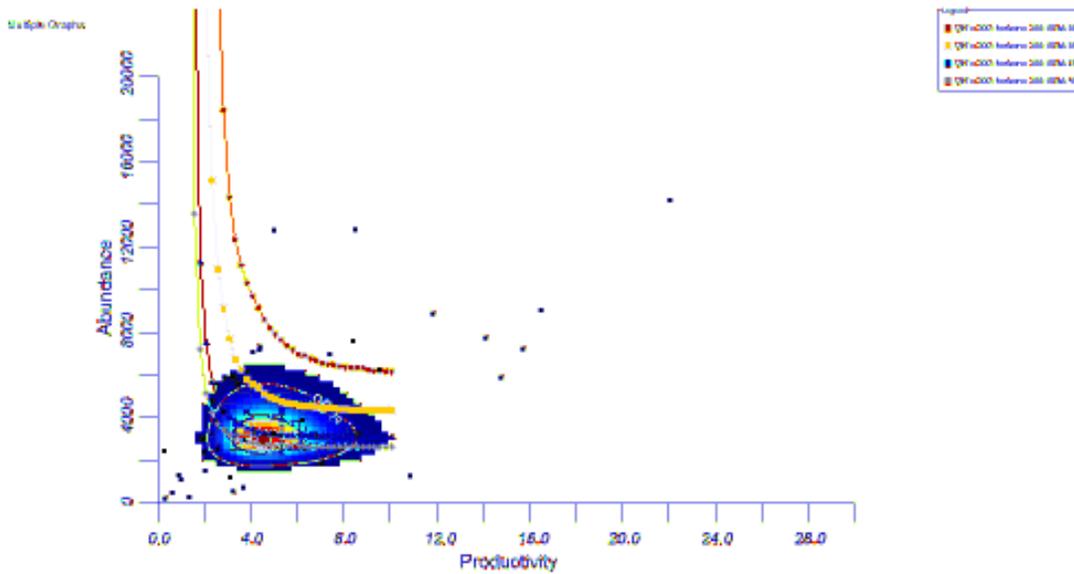
Figure 66 Sandy coho pre-harvest benchmark viability curves.

Spawner Surveys

Spawner surveys spanning several decades have been conducted in the Oregon LCR tributaries
down stream of the Willamette confluence with the Columbia. These data are summarized in the
BRT status review (2003), copied here as Figure 67 - Figure 70. These were non-random
surveys, where the methodology and effort tended to vary from year to year. Therefore, these
data are probably not very precise and treating them as an actual time series is probably
inappropriate. However, the surveys do provide some information. In the 1990s, it appears that
many of these populations were extirpated or nearly so. In some years the surveys found no
natural origin adults. Since these were not complete censuses, it would be inappropriate to
conclude that there were no fish. However, it does seem that abundance was *very* low, if not
zero.
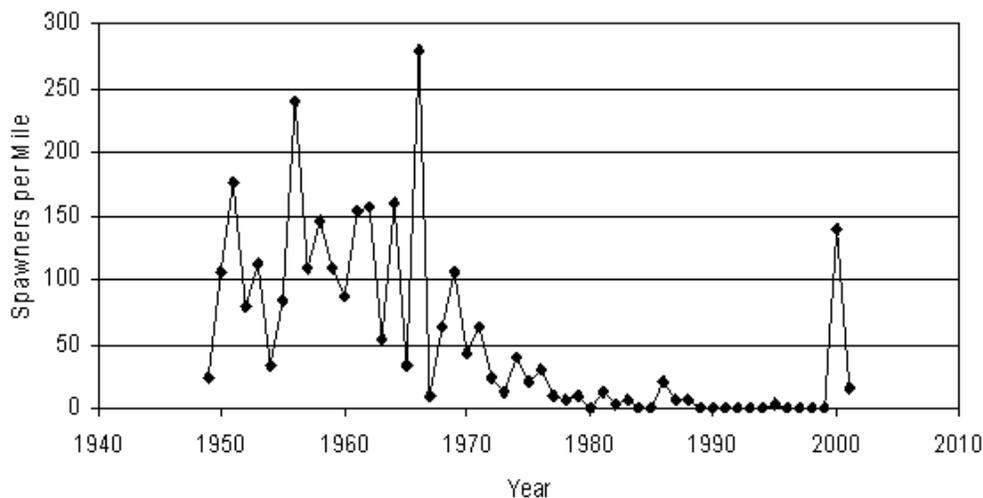
**Figure 67 Youngs Bay coho salmon spawners per mile, 1949–2001 (copied from BRT 2003).**
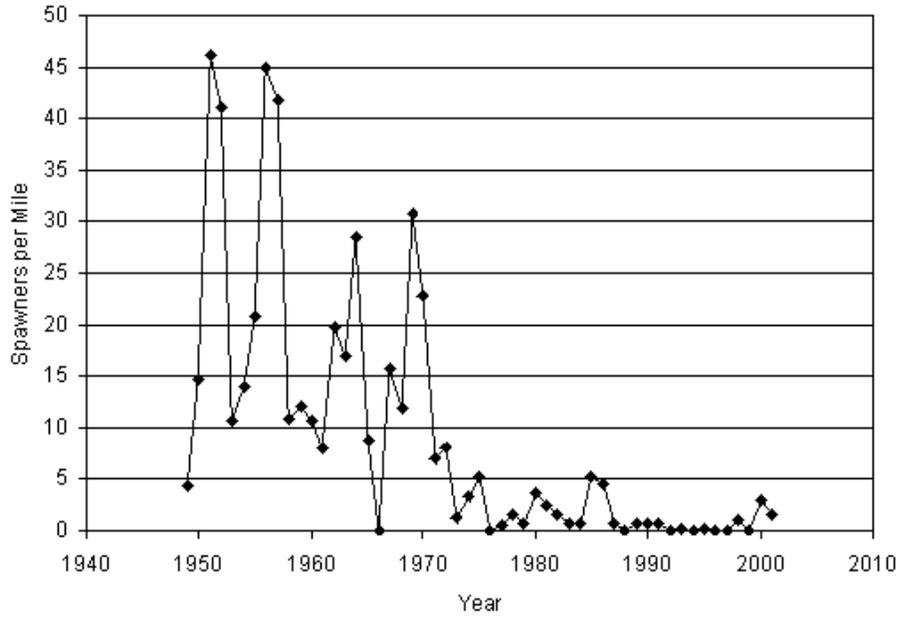
Figure 68 Big Creek coho salmon spawners per mile, 1949–2001(copied from BRT 2003).
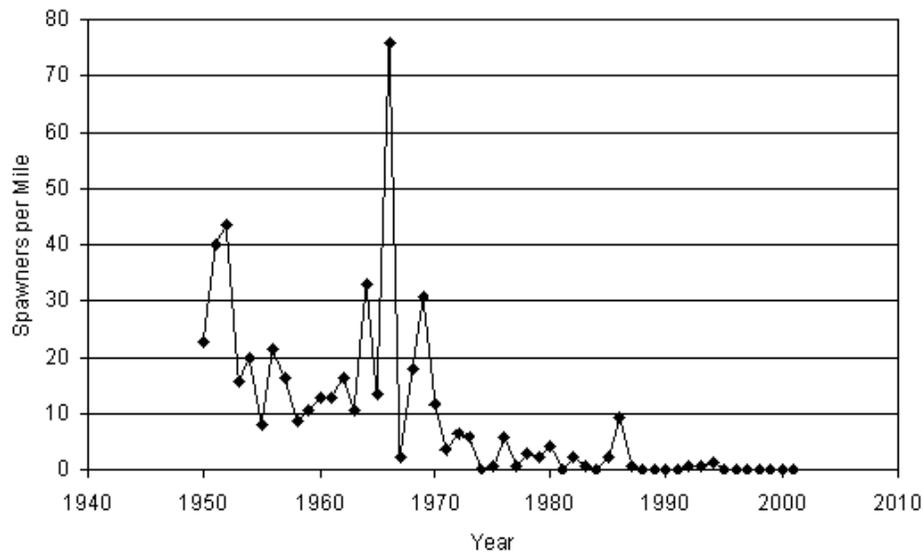

Figure 69 Clatskanie River coho salmon spawners per mile, 1949–2001(copied from BRT 2003).
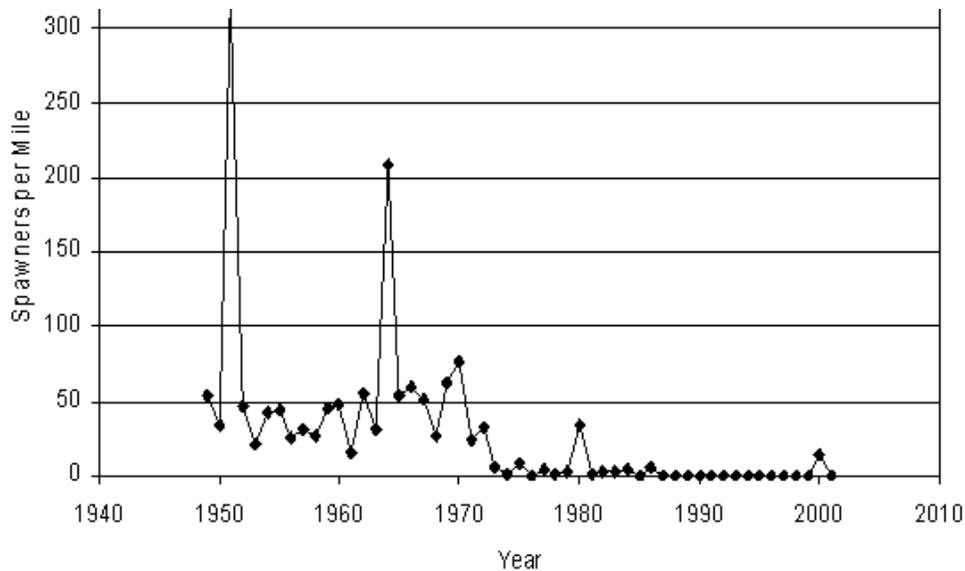
Figure 70 Scappoose River spawners per mile, 1949–2001(copied from BRT 2003).

Beginning in 2002, ODFW initiated stratified random spawner surveys for Oregon LCR coho populations. These expanded surveys provide more reliable abundance estimates than the earlier efforts. However, only three years of data are available to date. Figure 71 - Figure 73 show the survey escapement and hatchery fraction estimates. The survey units do not exactly match the population units we are using for this analysis. The Astoria population complex contains both the Youngs Bay and Big Creek populations. The Bonneville population complex contains both the Lower Gorge and Hood River populations. The Clackamas and Sandy surveys do not include the entire populations for 2002).

There are a number of important caveats about these 2002-2004 survey data. In the Sandy, only areas below Marmot dam were surveyed in 2002 whereas in 2003 the entire basin was surveyed. This largely explains the differences in hatchery fraction in the two years – 100% hatchery in 2002 below the dam and 5% in 2003 in the entire basin. Also the estimate of 'natural' fish for the Bonneville population the last several years (>3,000 fish) is highly suspect. There are technical difficulties in distinguishing natural origin and hatchery origin spawners because hatchery coho released in the interior Columbia are not marked and scale analysis for these gorge populations may be unreliable (Chilcote, pers. com.). It is also not plausible that the ~20km of available habitat in the gorge would produce a natural return three times that of the Clackamas, which has over 400km of habitat. In addition, smolt trapping data indicate that the Hood River, the largest system in this area, rarely produces more than 500 wild smolts per year – so it is not producing anywhere near 3,000 natural origin returns. Based on this information, we hypothesize that there are likely at most a few hundred naturally produced coho in the Bonneville population complex area.
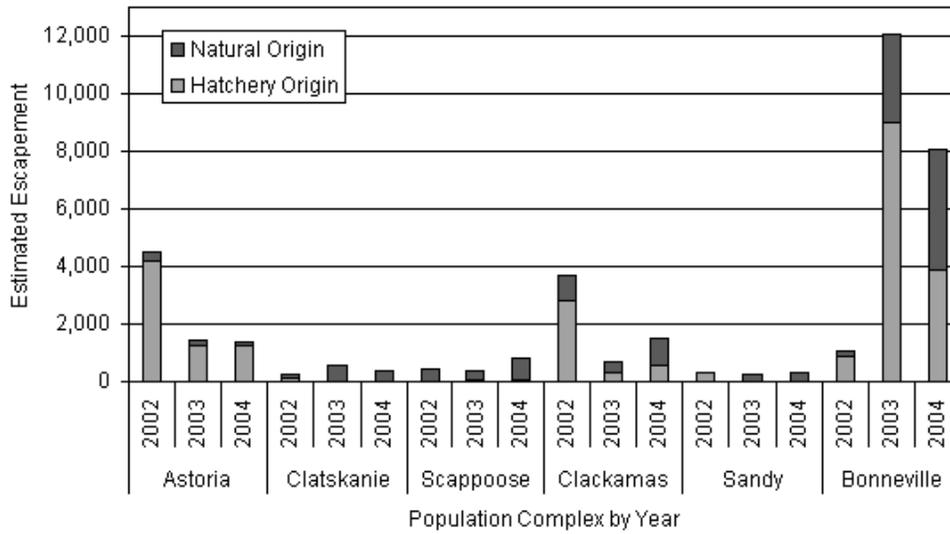
Figure 71 Escapement estimates for Oregon LCR coho population complexes.
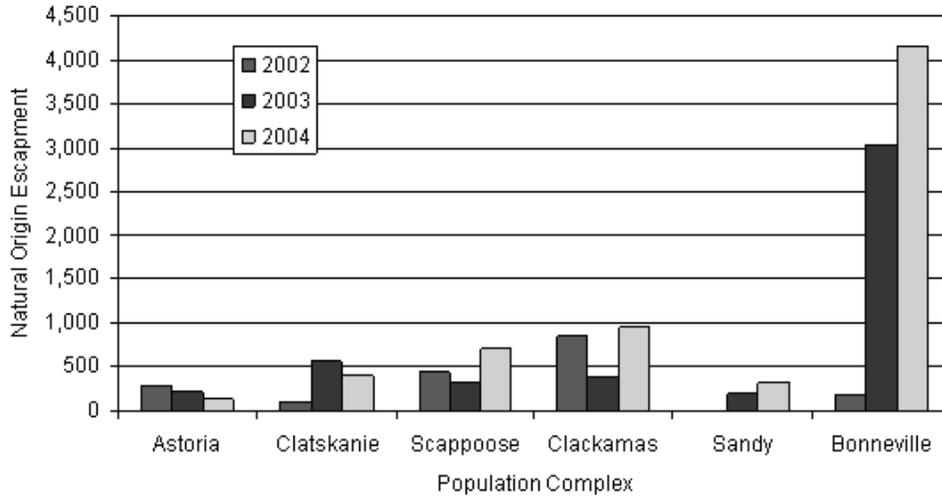
Figure 72 Natural origin escapement for Oregon LCR coho population complexes.
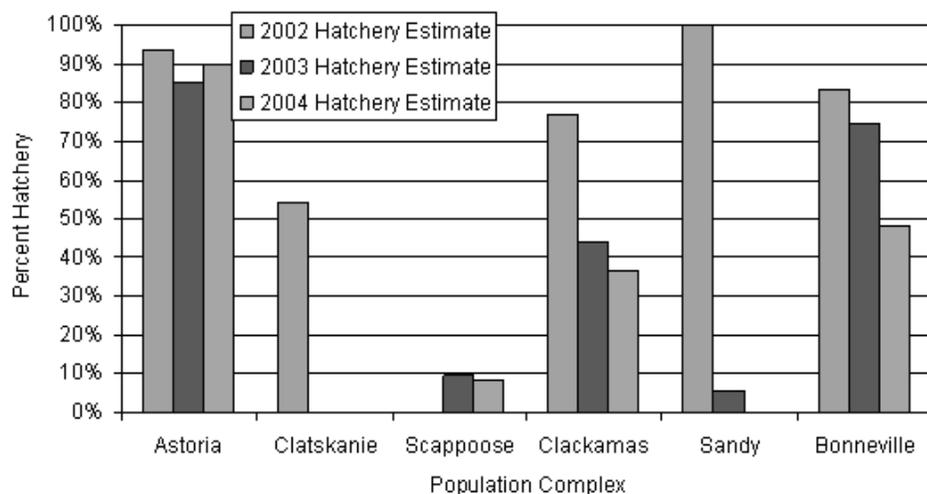
Figure 73 Estimated fraction of hatchery origin spawners for Oregon LCR coho population complexes.

Abundance and Productivity Summary

Population abundance and productivity is evaluated on the 0-4 persistence category scale described in the viability criteria section (Table 20). The point estimates for the Clackamas and Sandy are based primarily on the benchmark viability curve analysis. Estimates for the other populations are based primarily on the 2002-2004 survey data. These data provide an estimate of the geometric mean abundance (over only 3 years) which was compared to the MAT criteria. The evaluation of abundance was also informed by the recognition that populations were even lower in the 1990s.

The productivity estimates of the survey-only populations were informed by the fraction of hatchery origin fish in the population. If a population consists primarily of natural origin fish, it is reasonable to assume that there is some natural productivity in the watershed. However, if the spawners in a population are primarily of hatchery origin, the population could be sustained solely by hatchery production and have little, if any, natural production. If is difficult to estimate the exact natural productivity of a population with a significant fraction of hatchery origin fish, but the presence of a large fraction of hatchery spawners certainly calls into question whether the population is self-sustaining.

There is considerable uncertainty in relating the available data to extinction risk. Figure 74 provides a summary of the abundance and productivity scores for the Oregon LCR coho populations that includes an indication of the uncertainty associated with the scores. This distribution of uncertainty is based on both the probability contours of the viability curve assessment and on professional judgment.

Table 20 Oregon LCR Abundance and productivity Summary.

Population	Viability Curve (escapement)	Viability Curve (pre-harvest)	Average Natural Origin Spawners	Productivity Estimate	Overall Point Estimate
Youngs Bay	---	---	0 (<200 spawners)	0 (~90% hatchery)	0
Big Creek	---	---	0 (<200 spawners	0 (~90% hatchery)	0
Clatskanie	---	---	0 (~285 spawners)	1 (~20% hatchery)	0.5
Scappoose	---	---	0 (~470 spawners)	1 (~5% hatchery)	1
Clackamas	0	3.5	0 (~1,000 spawners)	(use viability curve)	2.5
Sandy	0	1.5	0 (~700 spawners)	(use viability curve)	1
Lower Gorge	---	---	0 (few spawners)	0 (~70% hatchery)	0

| Hood | --- | --- | 0 (<500) spawners) | 0 (>70% hatchery) | 0 |
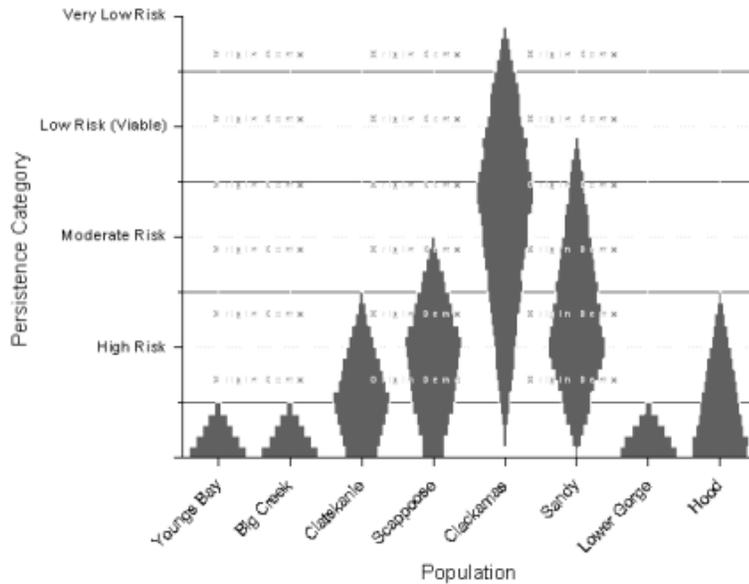
Figure 74 Summary of abundance and productivity persistence category estimates for Oregon LCR coho populations.


LCR Coho Spatial Structure

Overview

The TRT has not completely revised viability criteria for spatial structure, so the metrics used in this LCR coho current status evaluation are preliminary and incomplete. However, they do address two of the key spatial structure issues: 1) total quantity of available habitat; and 2) spatial distribution of accessible habitat. We have primarily based the evaluation on maps of accessible habitat developed in the Oregon WLC habitat atlas (Maher et al., 2004). The coho accessibility maps for LCR populations are copied here in Figure 75 - Figure 83. 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, 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.
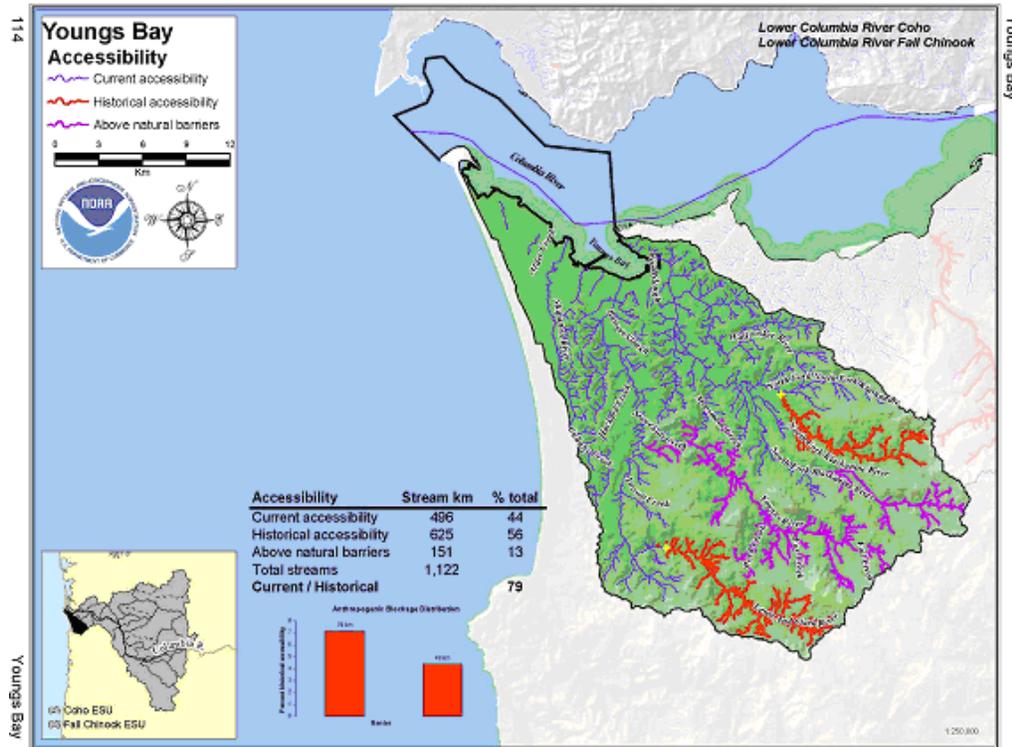
Figure 75 Youngs Bay coho current and historical accessibility (from Maher et al., 2005).
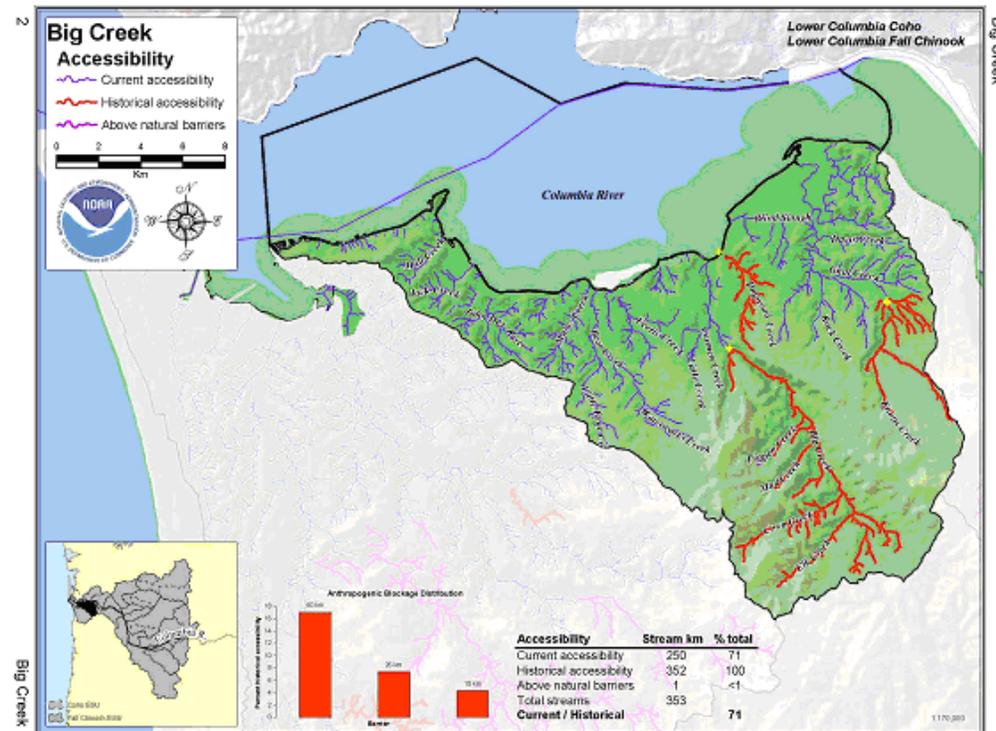
Figure 76 Big Creek coho current and historical accessibility (from Maher et al. 2005). Note that the large blockage on Big Creek shown on this map has been removed since the data base used for this map was developed and the habitat in Big Creek proper is now accessible.
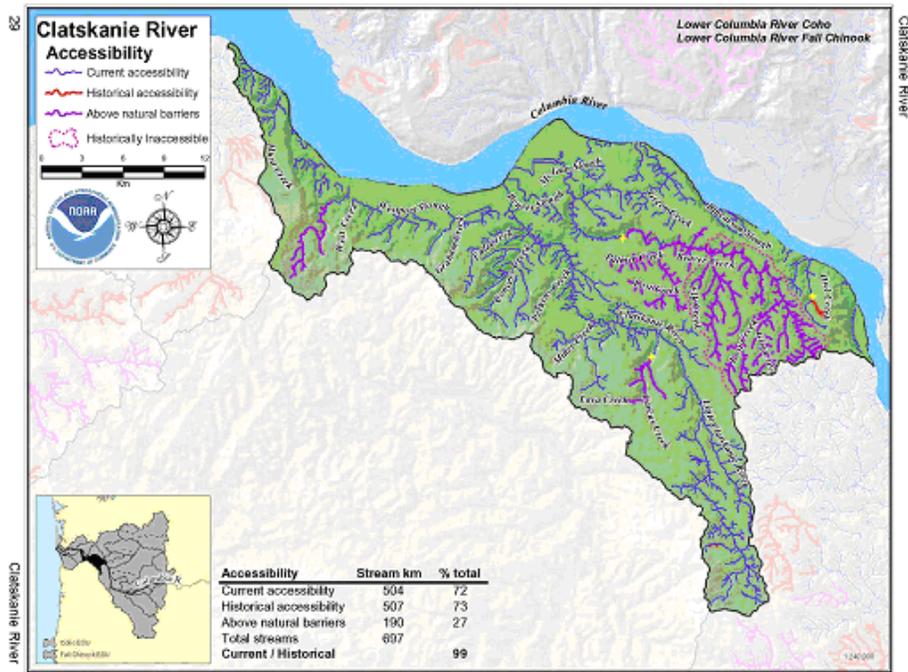
Figure 77 Clatskanie coho current and historical accessibility (from Maher et al. 2005).
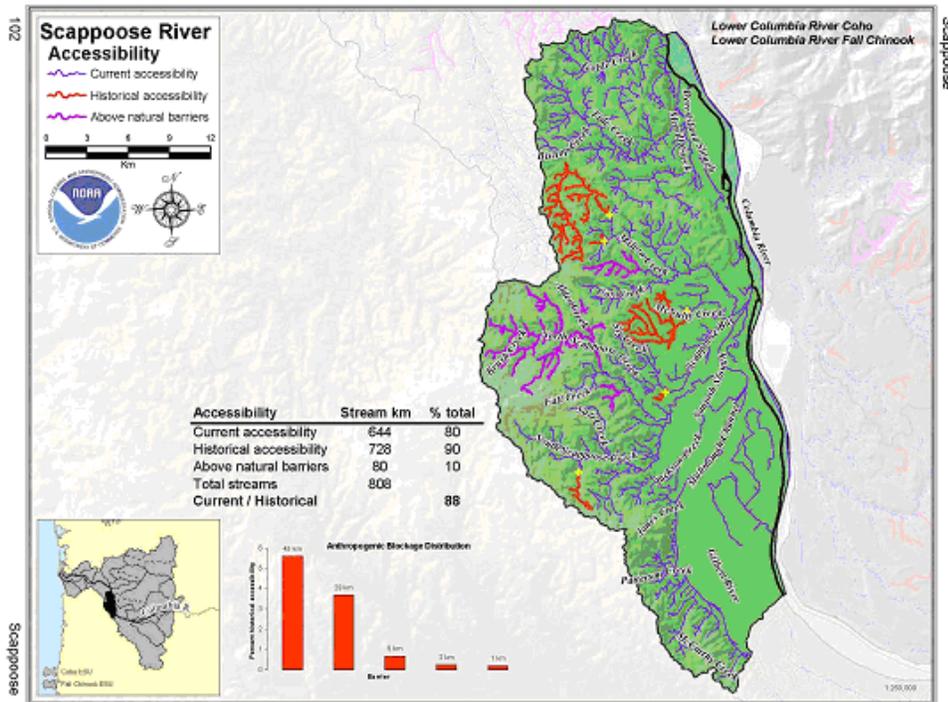
Figure 78 Scappoose coho current and historical accessibility (from Maher et al. 2005).
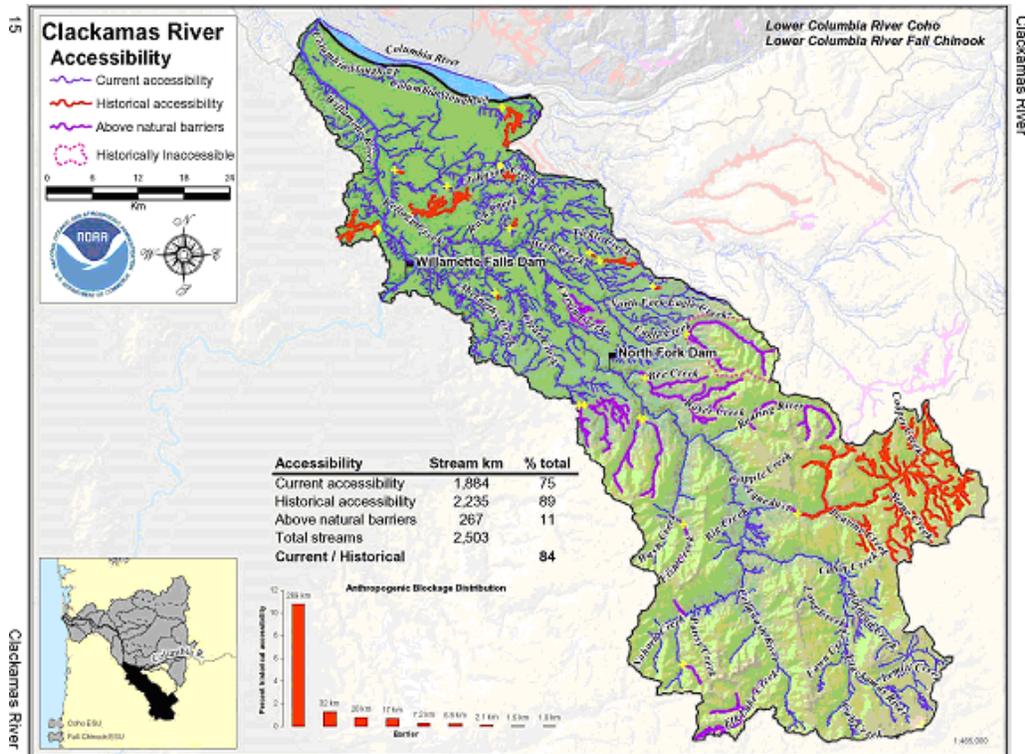
**Figure 79 Clackamas coho current and historical accessibility (from Maher et al. 2005).**
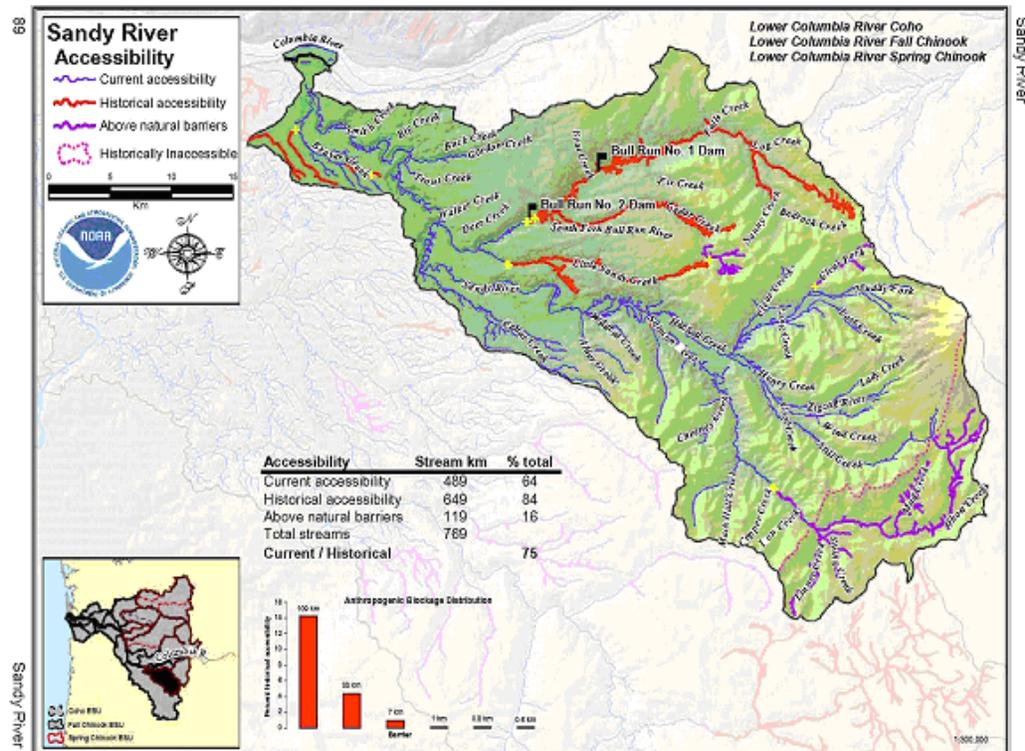
**Figure 80 Sandy coho current and historical accessibility (from Maher et al. 2005).**
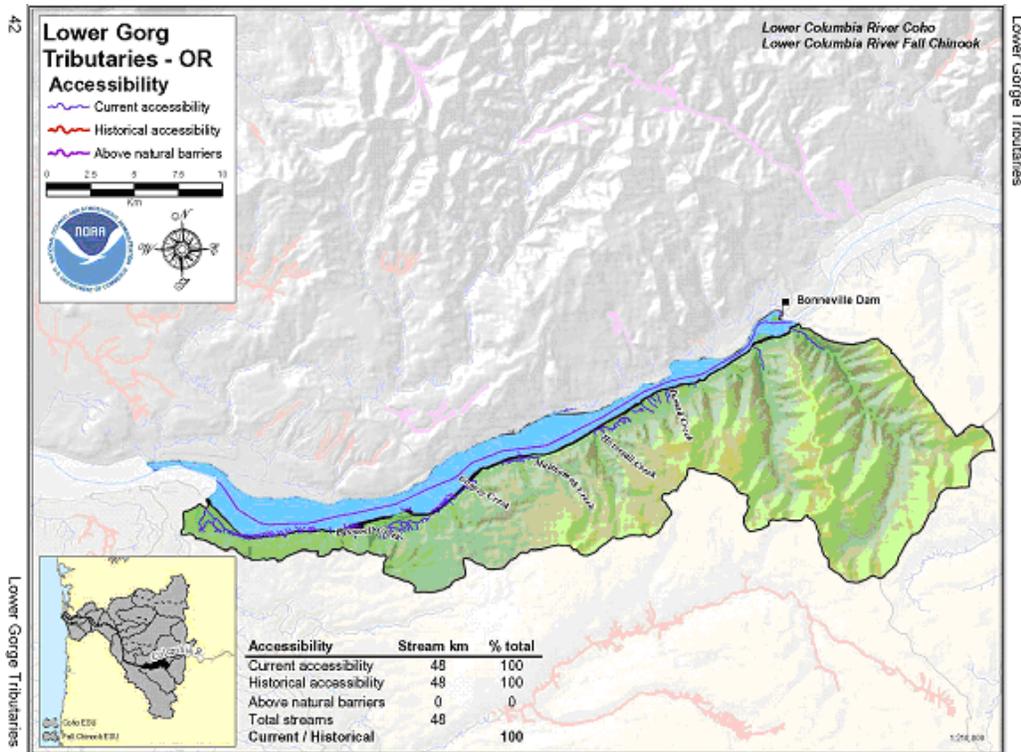
**Figure 81 Oregon Lower Gorge coho current and historical accessibility (from Maher et al. 2005).**
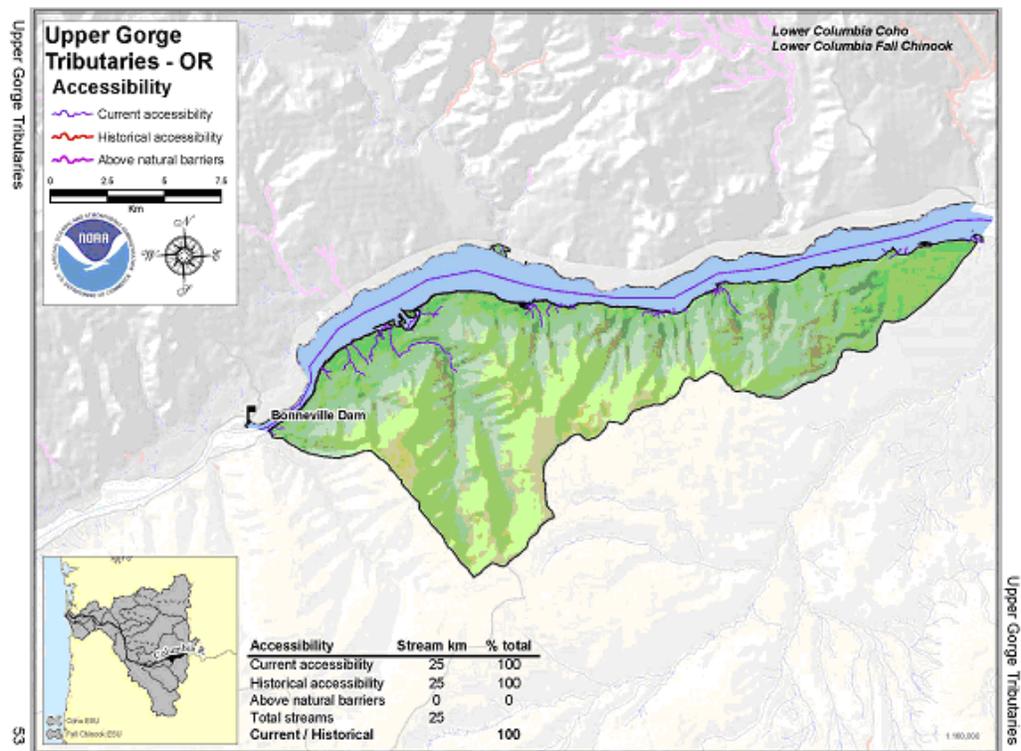
Figure 82 Upper Gorge coho current and historical accessibility (combine with Hood River) (from Maher et al. 2005).
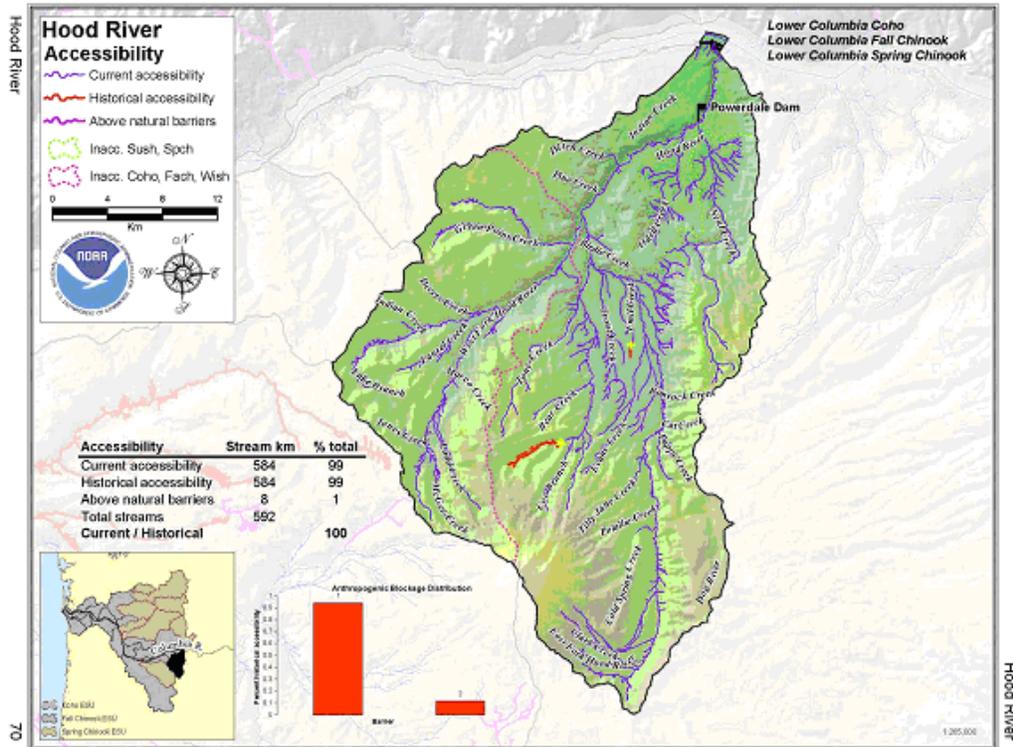
Figure 83 Hood River coho current and historical accessibility (combine with upper gorge). (from Maher et al. 2005).

Spatial Structure Metrics

One 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 the eggs in one basket." We developed metric and threshold guidelines that are a function of both the amount of historically accessible habitat and the size of the watershed (Table 21). Historical accessibility seems 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.

Table 21 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. We hypothesize 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 propose the following guideline:

> If 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 apply a qualitative evaluation based on consideration of the actual access maps.

Analysis of Oregon LCR populations

Data for Oregon LCR coho populations are summarized in Figure 84. Appling the thresholds in Table 21 and the reduction for populations with large single blockages produces the results in Table 23.
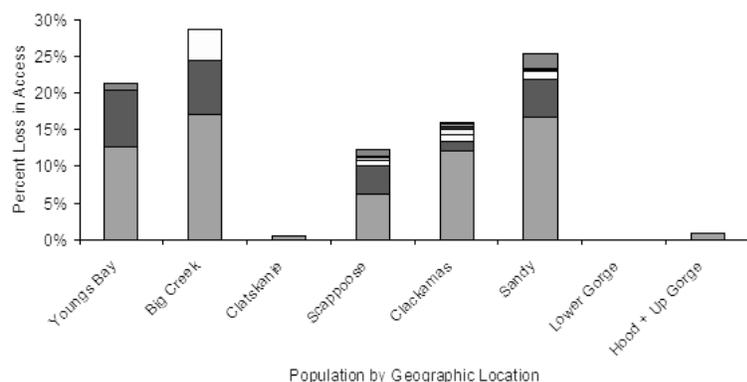
Figure 84 Percent loss in access due to anthropogenic blockages (based on Maher et al. 2004). The total height of the bar indicates total loss. The individual colors represent amount lost by individual blockages. The individual blockages are stacked from largest on the bottom to smallest on the top (i.e., blue is the largest). The green band indicates a pooling of very small blockages. (Note that the Big Creek data in this figure reflects the recent removal of a blockage that is inaccurately shown to still exist in the map of Figure 76).

Table 22 Results of accessibility analysis from to Oregon LCR coho.

Population	Quantity Score	Distribution Adjustment	Overall Access Score	Modified Point Estimate
Youngs Bay	2	Yes	1.5	1
Big Creek	3	No	3	2.5
Clatskanie	4	No	4	3.5
Scappoose	3	No	3	2.5
Clackamas	3	No	3	2
Sandy	3	Yes	2.5	2
Lower Gorge	4	No	4	1.5
Hood	4	No	4	1.5

The results from Table 22 are not translated directly into spatial structure scores because there are a number of issues not addressed in the simple access metrics. In the Sandy, the total loss in access is near the category 2 threshold and the single greatest loss, the Bull Run watershed, is

hypothesized to contain a relatively large fraction of the quality habitat in the basin. Therefore, we reduce the overall watershed persistence category from 2.5 to 2. The Lower Gorge and Hood populations also require some adjustment. The method used for comparing current and historical access did not effectively evaluate changes along the mainstem of the Columbia. In particular, it did not evaluate the loss in habitat that has resulted from the flooding of the Bonneville Pool. Since habitat along the mainstem and under the Bonneville pool was likely significant for these populations, losses of that habitat are also considered significant and we reduce the watershed spatial structure score to 2 for the Lower Gorge and Hood River populations. Because reduction in usable rearing habitat in the lower Clackamas as a result of urbanization has degraded the population spatial structure, the watershed score was reduced to 2.5. Finally, all populations of LCR coho are at increased risk from spatial structure degradation because of simplification and removal of habitat in Columbia River estuary (see "At Rivers End" report, 2004). Because of this risk, all of the watershed scores were reduced by 0.5 for the final population modified point estimates (Table 22).

There is considerable uncertainty in relating spatial structure to extinction risk. Figure 85 provides a summary of the spatial structure scores for the Oregon LCR coho populations that includes and indication of the uncertainty associated with the scores. This distribution of uncertainty is based on professional judgment.
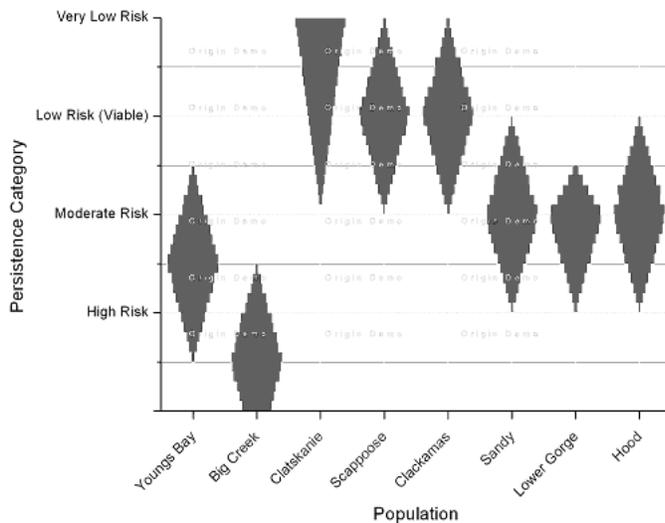
Figure 85 Summary of spatial structure scores for LCR populations. The shape of the bars indicates the relative level of confidence in the persistence score.

Diversity

The following population descriptions and diversity estimates are presented to illustrate how the diversity criteria are implemented, rather than to provide definitive diversity scores. In most cases, because of the paucity of information, expert opinion has been relied upon to incorporate factors that have had an effect on the overall diversity score. The TRT strongly encourages more extensive monitoring throughout the ESU with a focus on life history characteristics at critical stages in the life cycle. Considerable uncertainty will be associated with the diversity estimates until more definitive information is available.

Population Diversity Scores

Youngs Bay Coho Salmon

Direct Measures: Spawning information is available for coho salmon returning to the Klaskanine Hatchery in the early 1960s, mid-October to mid-December. Given the mixture of stocks--from both within and outside of the ESU--that have been introduced, it is unclear if this information is representative of the historical population. Recent surveys suggest a somewhat earlier distribution, especially compared to natural-spawning fish in nearby LCR basins.

1. *Diversity Score = NA, but change in spawn timing may be indicative of continued hatchery influence. Historically these coho salmon may have been late-run fish; the early-run timing suggests a non-native hatchery influence.*

   Indirect Measures: The Klaskanine Hatchery has been in operation since 1911. A number of different stocks of coho salmon have been imported into hatchery (it should be noted that 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.

2. *PNI ≤ 0.1, Fitness = 0.25, Diversity Score = 0.5*

   Recent surveys have observed low numbers of natural-origin spawners (zero in some years during the 1990s), actual abundance may near 50. It is not know to what degree naturally spawning hatchery-origin fish contribute to the productivity.

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 0 to 1*

   Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

   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).

5. *Most strays accounted for in PNI index.*

   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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 3 – 1 =2.*

Final Diversity Score = 0.5 (2004 TRT estimate 0.67)

Emphasis was placed on the effective population size and hatchery influence estimates. Ne and hatchery risks were roughly additive. Adjusted habitat diversity provides an additional negative factor to the diversity score.

Big Creek Coho Salmon

Direct Measures: There is some information on the run and spawn timing of coho salmon intercepted at the hatchery in Big Creek and Gnat Creek. A protracted run timing was observed from mid-September through February, with spawning observed from late-November to February. Current surveys are limited by the low numbers of natural spawners, but it appears that the only the early portion of the historical spawning interval is represented.

1. *Diversity Score = NA, but change in life history traits may be indicative of diversity loss.*

   Indirect Measures: 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.

2. *PNI ≤ 0.2, Fitness = 0.45, Diversity Score = 1.0.*

   Recent surveys have observed low numbers of natural-origin spawners (zero in some years), actual abundance may have averaged between 50 and 100. Some consideration might be given to the potential contribution of the hatchery broodstock to the genetic resources of the population (because that the Big Creek Hatchery broodstock was establish with local fish); however, given the long duration of propagation the genetic integrity of the broodstock might not be well adapted to local natural conditions.

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 1 or 2*

   Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

   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).

5. *Most strays accounted for in PNI index.*

   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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 3 – 1 =2.*

Final Diversity Score = 1.0 (2004 TRT estimate 0.76)

Emphasis was placed on the effective population size and hatchery influence estimates. Ne and hatchery risks were roughly additive. Adjusted habitat diversity provides an additional negative factor to the diversity score.

Clatskanie River Coho Salmon

Direct Measures: Information on the run and spawn timing of coho salmon is available for fish intercepted at the hatcheries on Big Creek and Gnat Creek. During the 1960s a protracted run timing was observed from mid-September through February, with spawning observed from late-November to February. Currently, coho spawning in the Clatskanie River is still representative of a late-run life history.

1. *Diversity Score = NA, but run timing does not appear to have diverged from the 1960s. All of the coastal tributaries may have historically only contained late-run coho salmon.*

    Indirect Measures: 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.

2. *PNI ≤ NA, hatchery program intermittent – stray metric used*

    Recent surveys have observed low numbers of natural-origin spawners (zero in some years during the 1990s), estimated NOR abundance = 74 – 217 (2002-2004).

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 2(recent escapement)*

    Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

    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%).

5. *Stray Rate Metric = 2*

    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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 4 – 1.5 =2.5.*

> Final Diversity Score = 1.5 (2004 TRT estimate 0.71)

Current Ne and hatchery metrics were "relatively" good compared to adjacent populations; however, because of the likelihood of past Ne bottlenecks and hatchery introductions the scores were adjusted downward for past effects. Adjusted habitat diversity provides an additional negative factor to the diversity score.

Scappoose Creek Coho Salmon

Direct Measures: There is little information on life history traits for Scappoose Creek coho salmon. Spawner surveys during the 1940s and 1950s suggested a late-run timing, while current surveys indicate the continued expression of this trait.

1. *Diversity Score = NA, little information to base a score on, but life history trait expression may be stable.*

   Indirect Measures: 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 presmolts 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).

2. *PNI ≤ not scored. Stray metric used.*

   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.

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 2*

   Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

   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.

5. *Stray Rate Metric = 3-4*

   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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 4 - 2 = 2.*

   Final Diversity Score = 2.0 (2004 TRT estimate 0.76)

The Ne estimate was the strongest factor used in the Diversity Score, some weight was placed on the low survey counts during the 1990s. Adjusted habitat diversity provides an additional negative factor to the diversity score.

Clackamas River Coho Salmon

Direct Measures: There has been considerable interest in the apparent bimodality of coho salmon returning the Clackamas River. It is unclear whether the apparent separation early and late-returning coho salmon is the result of harvest or simply the constriction of two naturally overlapping run times.

1. *Diversity Score = NA, but change in life history traits may be indicative of diversity loss.*

Indirect Measures: 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 similar to Clackamas River natural-origin late-returning coho salmon. 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. During most years, hatchery fish are removed at the North Fork Dam creating a "hatchery-free" zone in the upper basin, but hatchery strays can be found in the Eagle Creek drainage and the lower Clackamas River. The proportion of hatchery strays varies from year to year, but a rough average of 50% was used in the PNI. Hatcheries do not include unmarked "wild" fish into the broodstock.

2. *PNI ≤ 0.1, Fitness = 0.5, Diversity Score = 1.0*

   Surveys indicate that several hundred unmarked coho salmon spawned in the Lower Clackamas River from 2002 to 2004, in addition 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.

3. *Current Effective Population (range N = 2Ne or N = 4e), Diversity score = 3*

   Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect. There is little information on what selective effects the upstream and down stream passage programs at North Fork Dam have had on Clackamas coho salmon.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

   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).

5. *Most strays accounted for in PNI index.*

   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 Multinoma Channel and Sauvie Island. The diversity scores were adjusted downward to reflect this (indicated as a "-" score).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 4 – 2 =2.*

Final Diversity Score = 2.5 (2004 TRT estimate 1.89)

   Emphasis was placed on the effective population size. Current population size would not appear to present a high risk to diversity, although past bottlenecks may have had an effect on diversity (harmonic mean = 799 for 1958-2004). It was unclear if the bimodality in run timing was indicative of a major shift in life history traits (if so the Diversity Score would be decreased further). Although hatchery effects were estimated to have strongly reduced the relative fitness of the hatchery fish, it appears that the occurrence of hatchery fish on the

spawning grounds may be localized in the basin,. Due to the "local" nature of the hatchery broodstock, the PNI index was used instead of the hatchery stray metric. Ne and hatchery risks were roughly additive. Adjusted habitat diversity provides an additional negative factor to the diversity score.

Sandy River Coho Salmon

Direct Measures: "Historical" information on run and spawn timing from early in the 1900s is available from hatchery and fisheries records. Fish passage at Marmot Dam and information from the 2002-2004 stratified surveys provide fairly accurate estimates of current life history traits. In general, for those life history traits for which we have information there appear to be no substantial changes.

1. *Diversity Score = 3 expression may be stable.*

   Indirect Measures: The Sandy River Basin is currently managed as two distinct regions. 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 $\geq$ 75% and the pNOB $\leq$ 5%. The Sandy River Hatchery has been in operation since 1953, with relatively few introductions from out-of-basin. Genetic analysis does not indicate any strong divergence from other Lower Columbia River populations, or any similarity to coho salmon from other ESUs.

2. *PNI =1.0 (above dam), PNI=0.1 (below dam)*

   Abundance estimates are available for Sandy River coho salmon from 1960, with a harmonic mean of 499. Historical estimates of abundance suggest that between 10 and 20 thousand coho normally returned to the Sandy River.

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 2*

   Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

   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.

5. *Stray Rate Metric = 3-4*

   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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 3 – 1.5 =1.5.*

Final Diversity Score = 2.5 (2004 TRT estimate 1.68)

Emphasis was placed on the effective population size and the limited degree of introgression between hatchery and wild fish (albeit primarily for the last 10 years). Life history traits may not have been strongly. Due to the "local" nature of the hatchery broodstock, the PNI index was

primarily used instead of the hatchery stray metric. Ne and hatchery risks were roughly additive. Adjusted habitat diversity provides an additional negative factor to the diversity score.

Lower Columbia River Gorge Coho Salmon

Direct Measures: 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. Recent surveys of the "Bonneville" population by ODFW, which includes the Lower Columbia River Gorge, Upper Columbia River Gorge, and Hood River Basin, have observed both early- and late-run coho salmon. Given the high rate of hatchery straying (> 80%), it is difficult to identify current life history characteristic for NORs.

1. *Diversity Score = NA.*

   Indirect Measures: Tributaries in the Lower Columbia River Gorge population contain a high proportion of hatchery strays (pHOR ≥ 80%) that may have originated from one of a 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%.

2. *PNI =0.1 with an estimated 20 generations. Fitness loss near 65%, Diversity Score =1*

   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 NOR in this population is at critically low levels, N < 50. Additionally, this limited number of spawners is spread across a number of smaller tributaries.

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 0.5*

   Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

   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.

5. *Stray Rate Metric = 1 (if used in place of the PNI metric)*

   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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 2 – 1 = 1*

Final Diversity Score = 0.5 (2004 TRT estimate 0.81)

Critically low Ne and the high degree of hatchery influence on the spawning ground, especially by early-run coho hatchery stocks suggest that diversity risks to this population are very high.

Upper Columbia River Gorge and Hood River Coho Salmon

Direct Measures: 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. Hatchery influence, currently and in the past, <u>may</u> have had an effect on life history traits.

1. *Diversity Score = NA – considerable uncertainty in historical and present day LH traits.*

    Tributaries in the Upper Columbia River Gorge population contain a high proportion of hatchery strays (pHOR $\geq$ 80%) that likely originated from 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 $\leq$ 10%.

2. *PNI =1.0 with an estimated 20 generations. Fitness loss near 65%, Diversity Score =1*

    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 < 100. 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.

3. *Effective Population (range N = 2Ne or N = 4e), Diversity score = 1.5*

    Harvest effects are *relatively* low (20%) for natural origin fish and take place in coastal fisheries that may not exert a selective effect.

4. *Anthropogenic mortality Diversity Score = NA – not all effects measurable.*

    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

5. *Stray Rate Metric = 1 (if the PNI metric is not used)*

    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).

6. *Habitat Diversity Index (from worksheet) minus adjustment: Diversity Score = 4 – 1 =3.*

    Final Diversity Score = 1.0 (2004 TRT estimate 0.80 (Upper Gorge) and 1.10 (Hood River)

Small Ne and the large influence of hatchery-origin spawners on the natural population were the primary factors considered. There is considerable uncertainty regarding the origin of unmarked fish observed on the spawning grounds.

Diversity Summary

Point estimate diversity scores are summarized in Table 23. There is considerable uncertainty in relating diversity to extinction risk. Figure 86 provides a summary of the diversity scores for the Oregon LCR coho populations that includes an indication of the uncertainty associated with the

scores. The scores in Table 23 are the central values (mode) for the distributions. This distribution of uncertainty is based on professional judgment.

Table 23 Diversity scores for Oregon Lower Columbia River coho salmon populations based on criteria presented in this document and criteria developed by ODFW 2005 and WLC TRT 2004. TRT evaluations were based on a 0 to 4 scale (high to low risk of extinction).

Population	TRT 2006	ODFW 2005	TRT 2004
Youngs Bay	0.5	Fail*	0.67
Big Creek	1.5	Fail	0.76
Clatskanie River	1.5	Fail	0.71
Scappoose Creek	1.5	Fail	0.76
Clackamas River	2.0	Pass	1.89
Sandy River	2.0	Pass (Ne ?)**	1.68
Lower Columbia River Gorge Tributaries	0.5	Fail	0.81
Hood River (HR) and Oregon Upper Columbia River Gorge Tributaries (UCRG)	1.0	Fail	0.80 (UCRG) 1.10 (HR)

* Failed one or more of the diversity metrics (Ne, % hatchery stray)
** Slightly below Ne threshold, but passed other metrics.
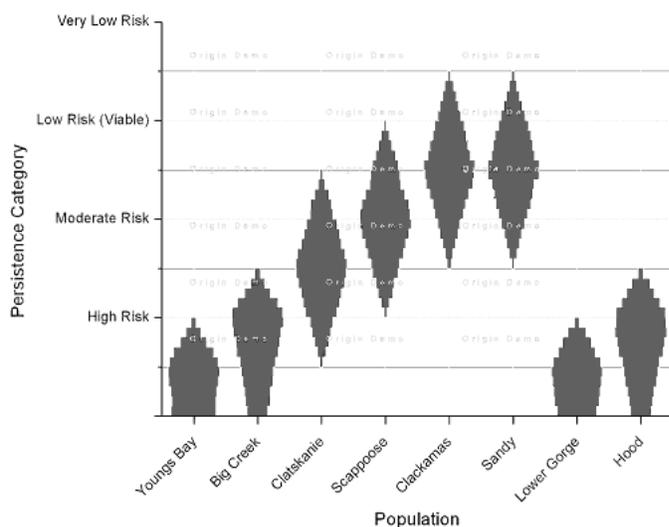
Figure 86 Summary of diversity risk evaluation for Oregon LCR coho populations.

Population Synthesis and Summary

Following the viability criteria methods, the approach used to integrate abundance, spatial structure and abundance is a simple weighted average, where abundance and productivity are weighted twice as much as the combined influence of spatial structure and diversity. We calculated the average over the certainty distributions to get the estimates in Figure 87. Figure 88 shows how abundance and productivity, spatial structure and diversity contribute to the risk evaluation. None of the Oregon populations is in a clear low risk category (Clackamas is closest). Previous analyses of the Washington populations (e.g., TRT report, BRT report, LCFRB) indicate that the Washington populations are also at high risk. Therefore, we conclude that all of the LCR coho strata (i.e., Coast, Cascade and Gorge) are at significant risk and the ESU as a whole is at significant risk (i.e., not viable).
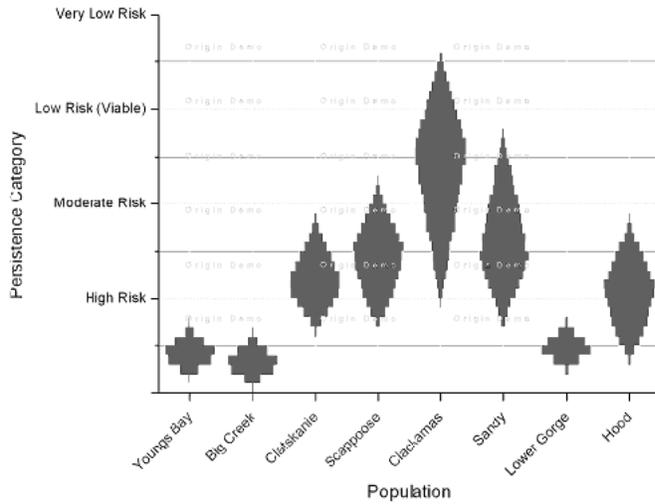
Figure 87 Overall summary of population status for Oregon LCR coho populations.
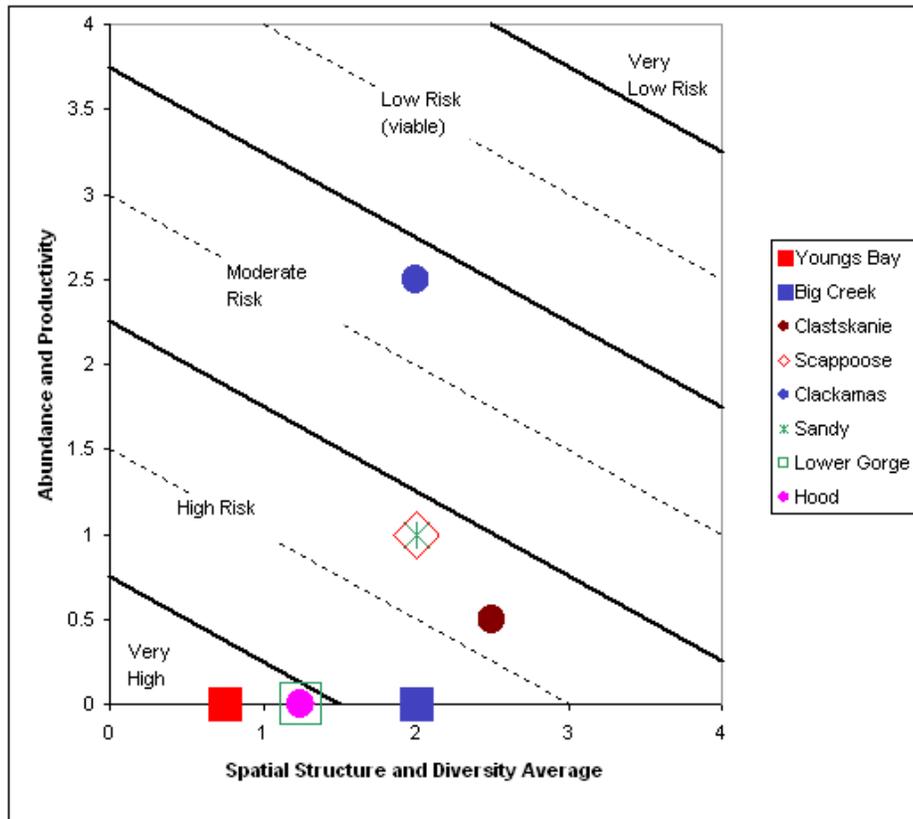

Figure 88 Oregon LCR coho population status point estimates as a function of abundance and productivity and the average of spatial structure and diversity scores. Populations up and to the right are at low risk, and populations down and to the left are high risk. The diagonal lines show overall population risk categories.

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Appendix A: Population and Strata Boundaries

Jim Myers

1. Lower Columbia River Chinook Salmon ESU
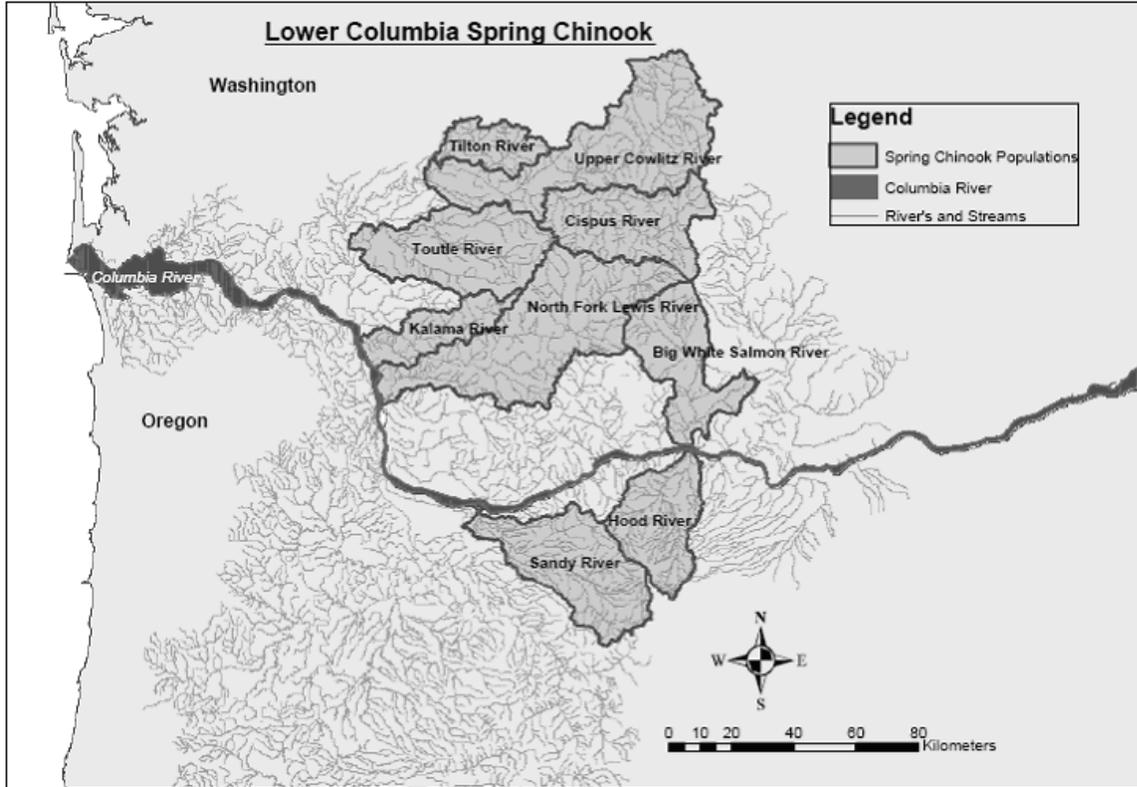
Table A-1 Lower Columbia River Chinook Salmon ESU – Spring Run DIPs

Strata	Populations
Cascade Tributaries Spring Run	Tilton River
	Upper Cowlitz River
	Cispus River
	Toutle River
	Kalama River
	North Fork Lewis River
	Sandy River
Gorge Tributaries Spring Run	Big White Salmon River
	Hood River
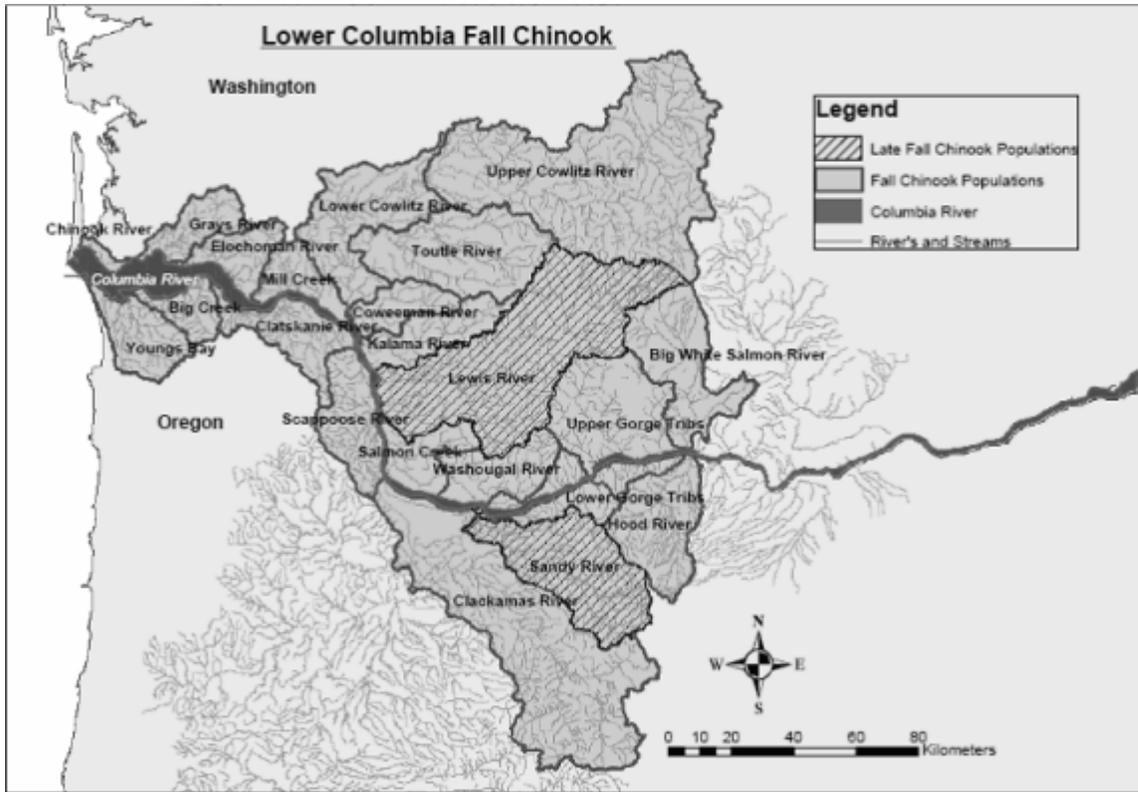
Table A-2 Lower Columbia River Chinook Salmon ESU – Fall Run DIPs

| Strata	Populations
Coastal Tributaries Fall Run	Grays River Youngs Bay Big Creek Elochoman River Clatskanie River Mill Creek Scappoose Creek
Cascade Tributaries Fall Run	Lower Cowlitz River Upper Cowlitz River Toutle River Coweeman River Kalama River Lewis River Salmon Creek Clackamas River Washougal River Sandy River
Cascade Tributaries Late-Fall Run	Lewis River Sandy River
Gorge Tributaries Fall Run	Lower Columbia River Gorge Upper Columbia River Gorge Big White Salmon River Hood River

2. Lower Columbia River Coho Salmon ESU
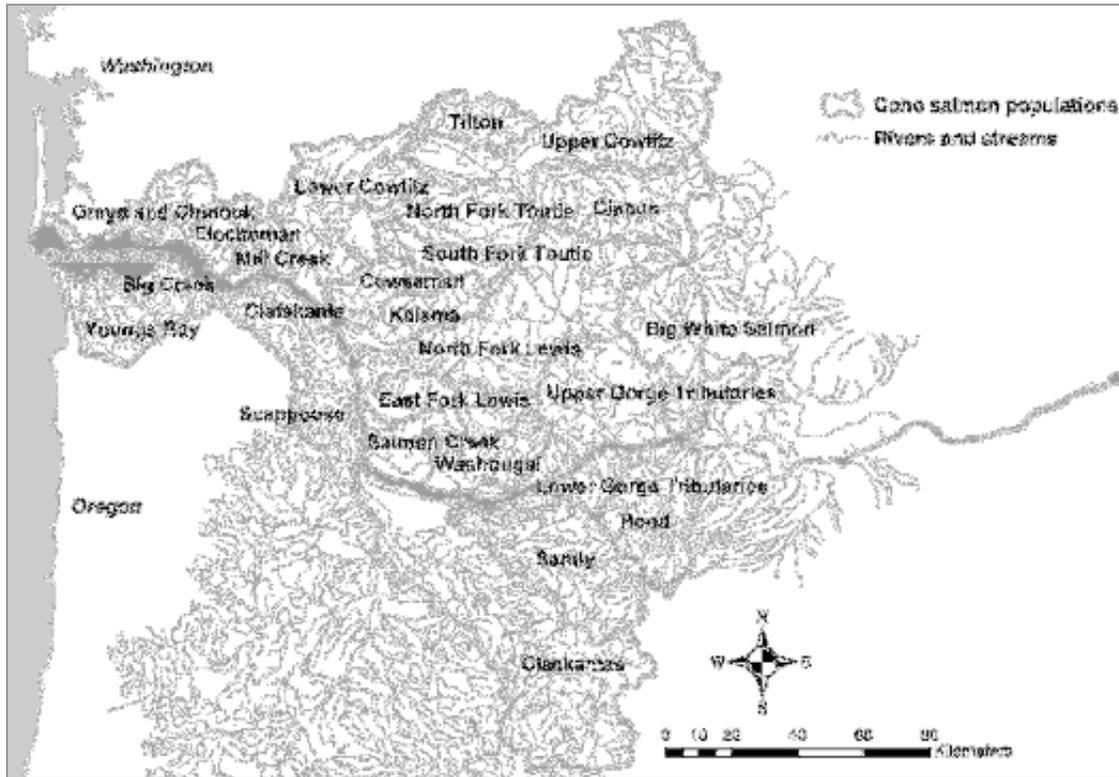
Table A-3 Lower Columbia River Coho Salmon ESU – DIPs

| Strata	Populations
Coastal Tributaries	Grays River Youngs Bay Big Creek Elochoman River Clatskanie River Mill Creek Scappoose Creek
Cascade Tributaries	Lower Cowlitz River Upper Cowlitz River Tilton River Cispus River North Fork Toutle River South Fork Toutle River Coweeman River Kalama River North Fork Lewis River East Fork Lewis River Salmon Creek Clackamas River Washougal River Sandy River
Gorge Tributaries Fall Run	Lower Columbia River Gorge Big White Salmon River and Washington Upper Columbia River Gorge Tributaries Hood River and Oregon Upper Columbia River Gorge Tributaries

3. Lower Columbia River Chum Salmon ESU
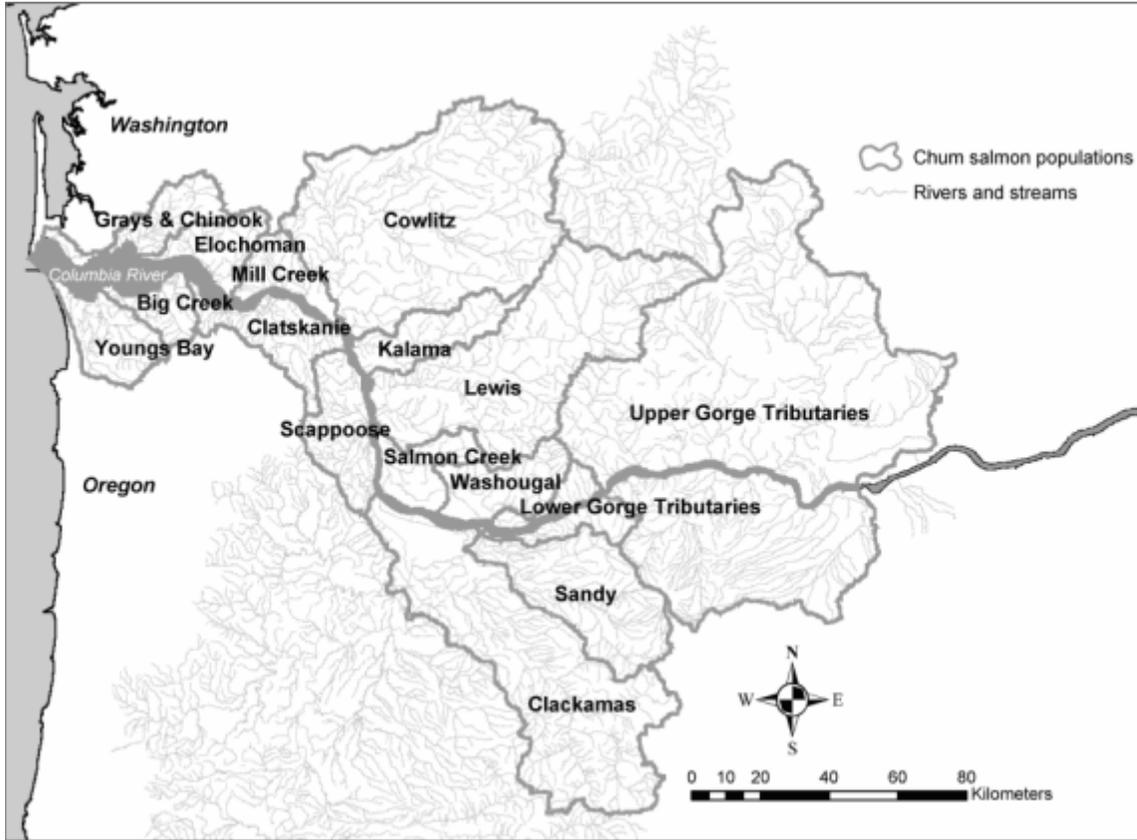
Table A-4 Lower Columbia River Chum Salmon ESU – DIPs

Strata	Populations
Coastal Tributaries	Grays River
	Youngs Bay
	Big Creek
	Elochoman River
	Clatskanie River
	Mill Creek
	Scappoose Creek
Cascade Tributaries	Cowlitz River Fall Run
	Cowlitz River Summer Run
	Kalama River
	Lewis River
	Salmon Creek
	Clackamas River
	Washougal River
	Sandy River
Gorge Tributaries	Lower Columbia River Gorge
	Upper Columbia River Gorge

4. Lower Columbia River Steelhead ESU
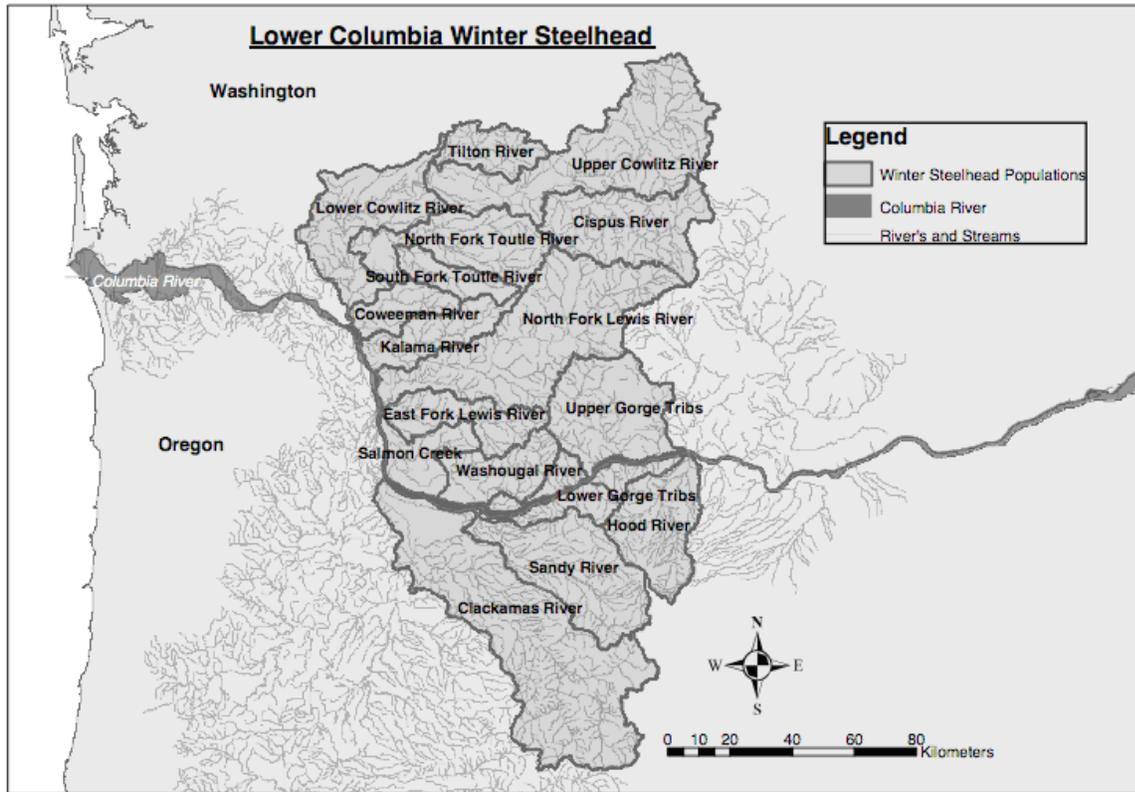
Table A-5 Lower Columbia River Steelhead Salmon ESU – Winter Run DIPs

Strata	Populations
Cascade Tributaries – Winter Run	Lower Cowlitz River
	Upper Cowlitz River
	Tilton River
	Cispus River
	North Fork Toutle River
	South Fork Toutle River
	Coweeman River
	Kalama River
	North Fork Lewis River
	East Fork Lewis River
	Salmon Creek
	Clackamas River
	Washougal River
	Sandy River
Gorge Tributaries Winter Run	Lower Columbia River Gorge
	Upper Columbia River Gorge Tributaries
	Hood River
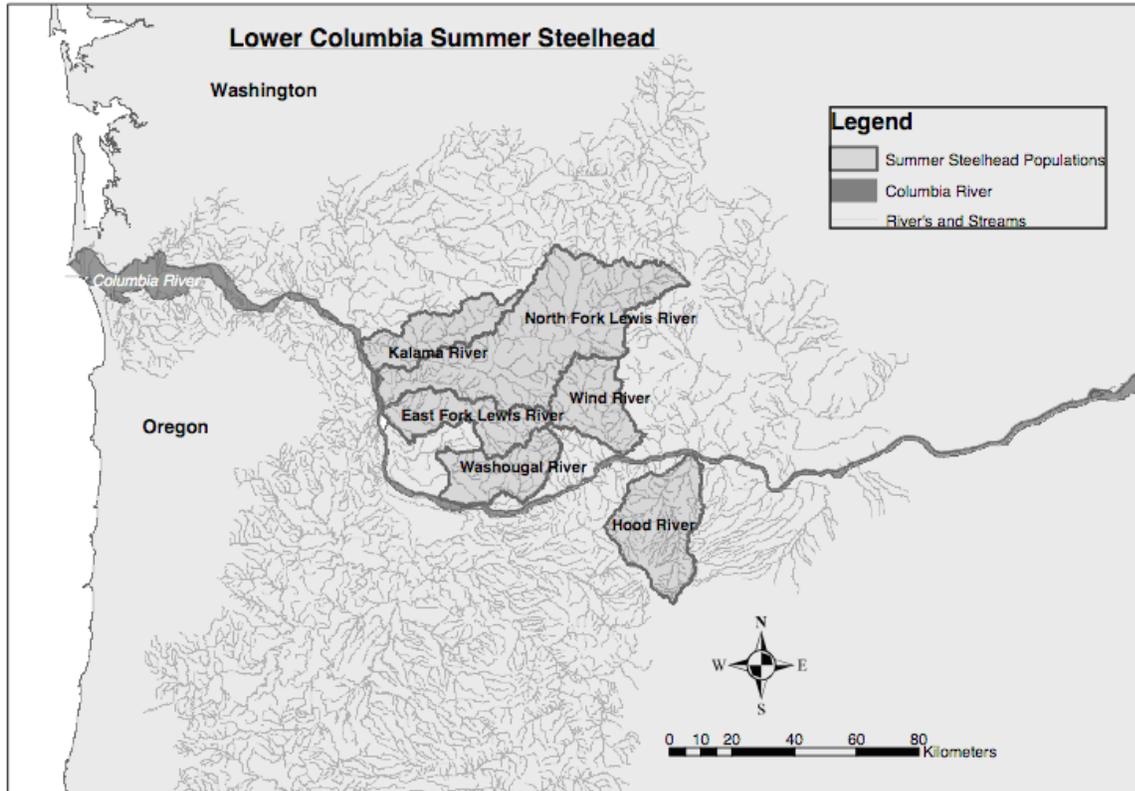
Table A-6 Lower Columbia River Steelhead Salmon ESU – Summer Run DIPs

| Strata	Populations
Cascade Tributaries – Summer Run	Kalama River
	North Fork Lewis River
	East Fork Lewis River
	Salmon Creek
	Washougal River
Gorge Tributaries Summer Run	Wind River
	Hood River

5. Upper Willamette River Chinook Salmon ESU
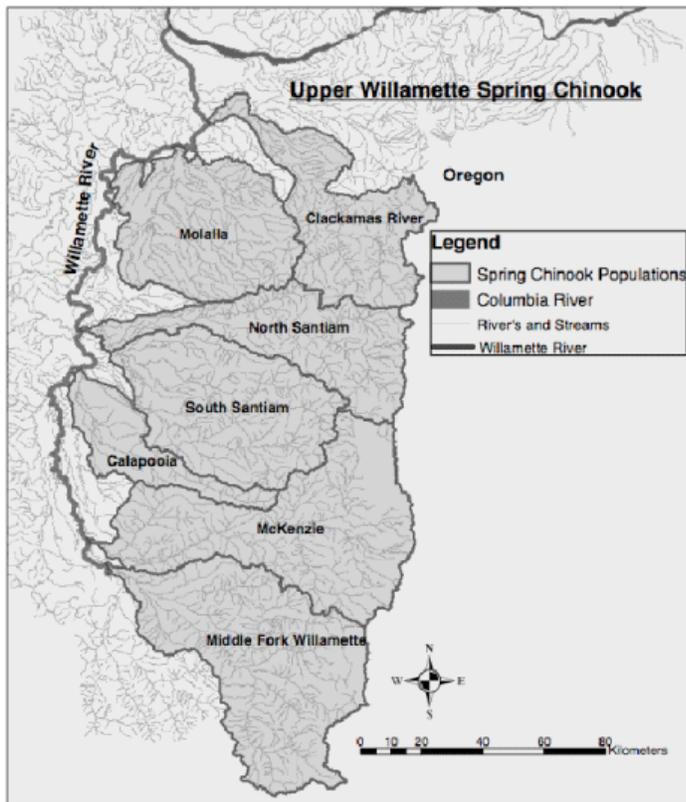
Table A-7 Upper Willamette River Chinook Salmon ESU – Spring Run DIPs

Strata	Populations
Cascade Tributaries – Spring Run	Clackamas River
	Molalla River
	North Santiam River
	South Santiam River
	Calapooia River
	McKenzie River
	Middle Fork Willamette River

6. Upper Willamette River Steelhead ESU
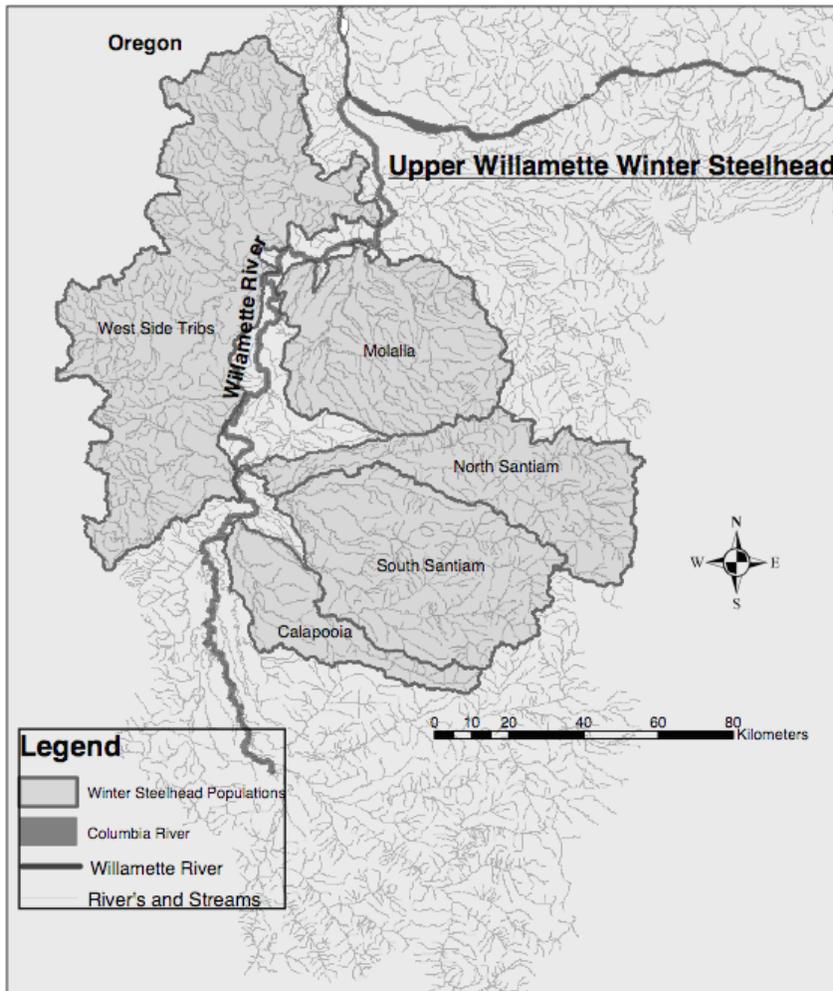
Table A-8 Upper Willamette River Steelhead ESU – Winter Run DIPs

Strata	Populations
Cascade Tributaries – Winter Run	Molalla River North Santiam River South Santiam River Calapooia River
Coastal Range Tributaries	*West Side Tributaries – Not identified as a DIP, but may provide important rearing/holding habitat.*

Appendix B: Stratum Threshold Considerations

Paul McElhany

This appendix explores the stratum threshold with the assumption that the actual threshold is correct (or at least constant), but we ask if there is an easier way to explain the criteria that does not imply excessive precision. One of the first things to note is the non-linearity of the population persistence categories. The population persistence categories associated with a given population persistence probability and the stratum threshold are shown in Figure B-1. Since the threshold is a simple arithmetic average of the persistence categories, the persistence categories act as a weight in averaging the actual persistence probabilities. The weighting scale results in low and moderate persistence probabilities (e.g., <85%) contributing relatively little to long-term strata persistence. This seems an appropriate weighting; higher risk populations contribute some to strata persistence by providing connectivity between stronger populations and there is some probability that they will be able to maintain themselves. However, there is a relatively high likelihood that they will go extinct and should therefore not be given much weight in strata persistence criteria. The weight also indirectly incorporates uncertainty – assessment methods (and criteria) are generally good at identifying populations clearly not at risk (e.g., >>95% probability is very certain), but much less so at intermediate risk levels (e.g., 50% probability is very uncertain). Thus the weighting places a higher value on more certain criteria and less value on uncertain criteria.
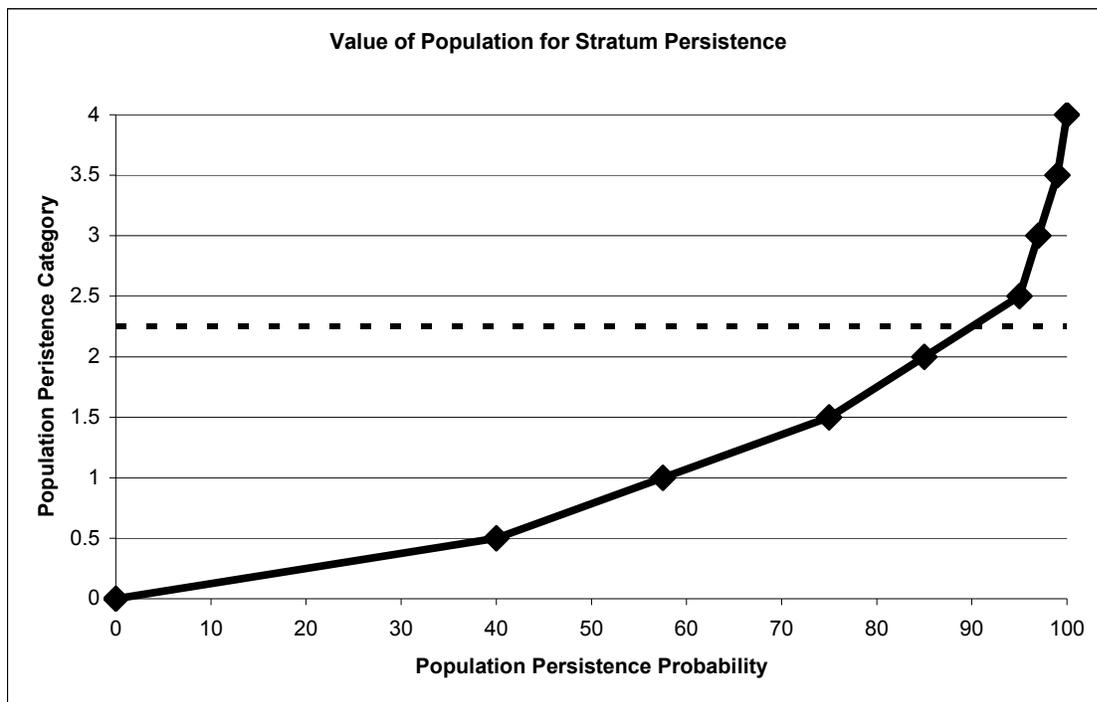

Figure B-1 Relationship of population persistence categories and population persistence probability.

It is important to note that although we have confidence in a general non-linear weighting, we do not have a quantitative analysis of either the weighting or the thresholds. Both amount to an

expert judgment rule-of-thumb, which makes it apparent why we do not want to imply excessive precision.

In trying to simplify the criteria, we explored using a weighting function directly relying on the population persistence probabilities. Then, rather than use the 0-4 category scale, population risk could just be considered on 0-100% probability scale and the stratum score taken as a weighted average on that scale. The threshold could then be set on the 0-100 scale. Figure B-2 fits an exponential curve to the weights previously used by the TRT. This curve does not capture a key feature of the original weights – the large increase in value assigned to populations as they exceed 95% probability relative to probabilities below 95%. We believe that the original weighting more accurately reflects the high value of very low risk populations. We could describe a multi-part function to include this feature, but then the procedure gets more complex and defeats our goal of simplification. As a consequence, we believe it is actually clearer to stick with the original qualitative 0-4 scoring system and 2.25 threshold, which easily allows for a complex weighting system, than to apply a weighting function directly to the population persistence probabilities.
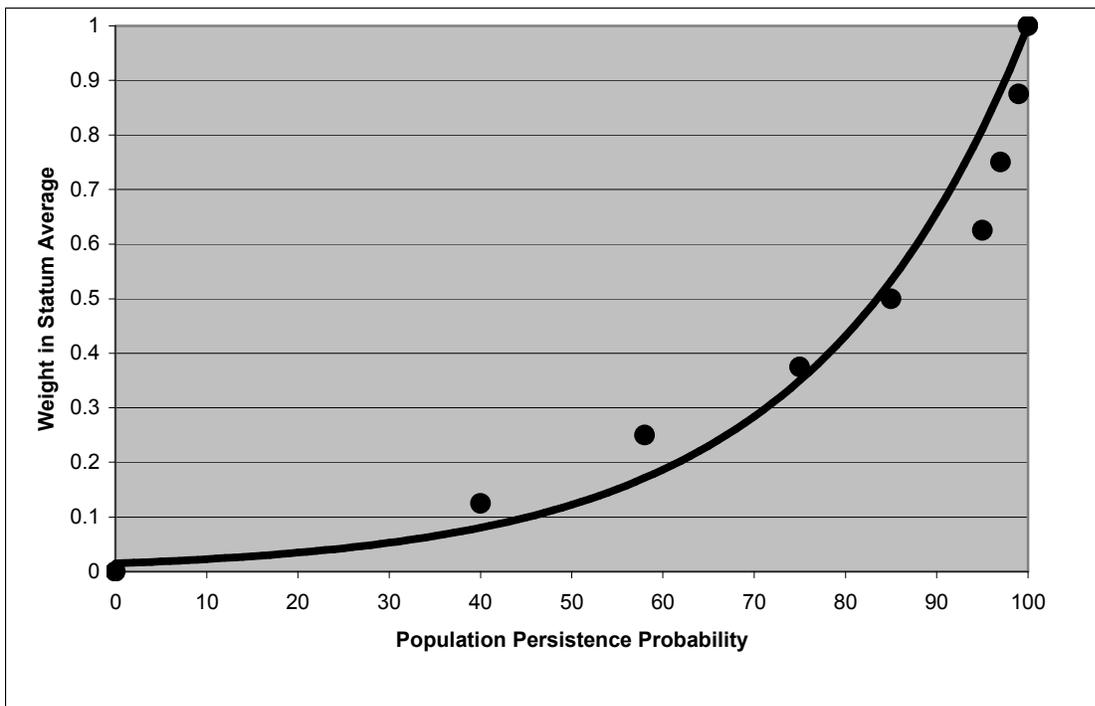

Figure B-2 Fit of an exponential curve to the weights implied by the population persistence categories.

Appendix C: Oregon WLC Abundance and Productivity Data

Mark Chilcote, Paul McElhany, John Payne, and Ejiro Akporobaro

Overview

Time series data on population abundance, harvest rates, hatchery fraction and age structure were provided by Mark Chilcote based primarily on information collected by ODFW. Documentation on these data sets is currently incomplete, but many of the sources are referenced in the NOAA ESU status review (BRT 2003). In this appendix, we present graphs of data on Oregon WLC spring chinook and coho populations. We plan to post electronic data sets with documentation on the TRT web site as soon as possible for Oregon chinook (including fall chinook), coho, and steelhead.

Population Data

Clackamas Spring Chinook
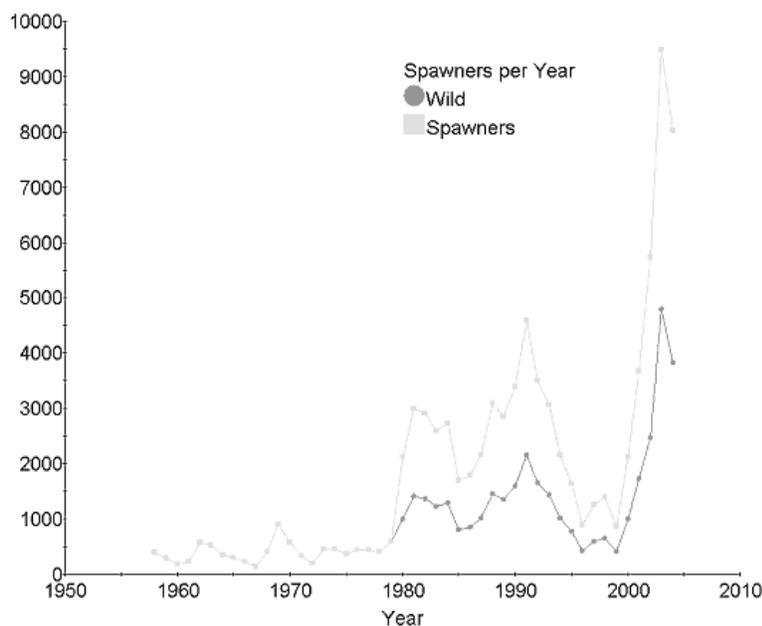
Figure C-3 Clackamas Spring Chinook Abundance
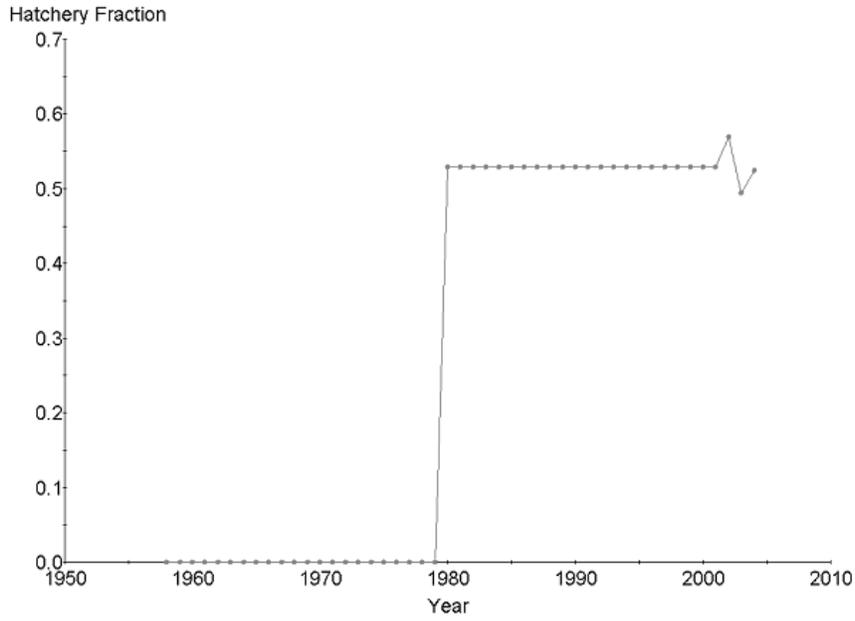
Figure C-4 Clackamas Spring Chinook Hatchery Fraction
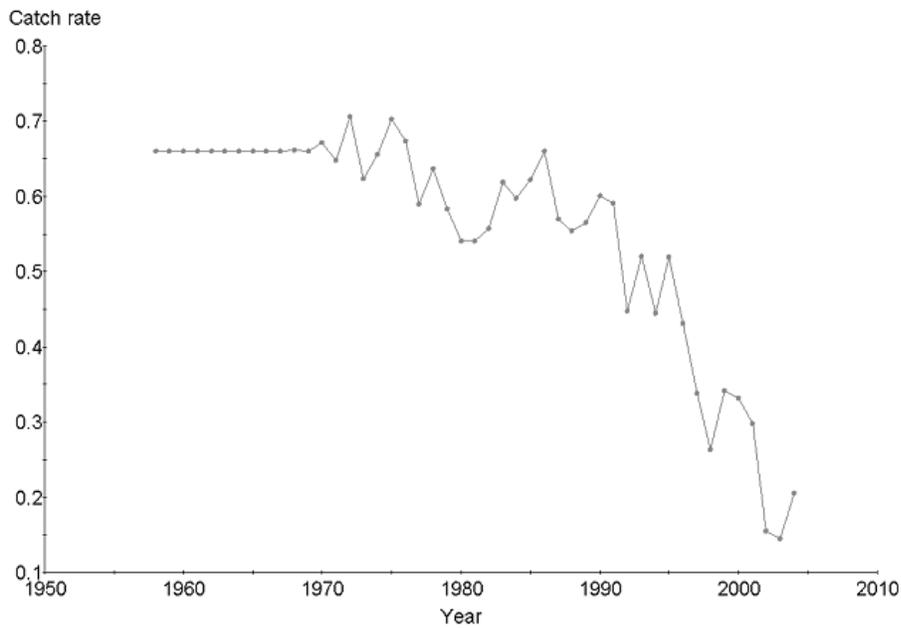
Figure C-5 Clackamas Spring Chinook Harvest
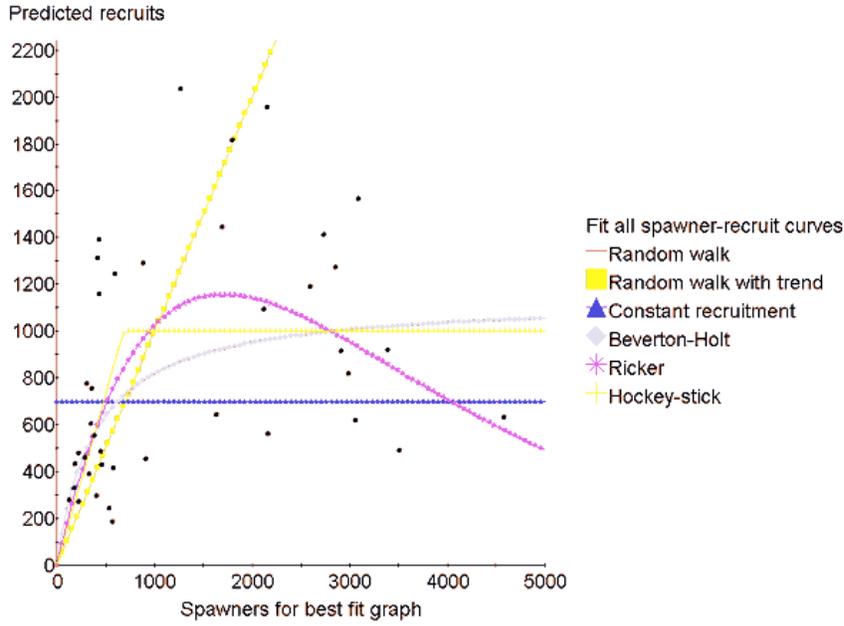
Figure C-6 Clackamas Spring Chinook post-harvest recruitment
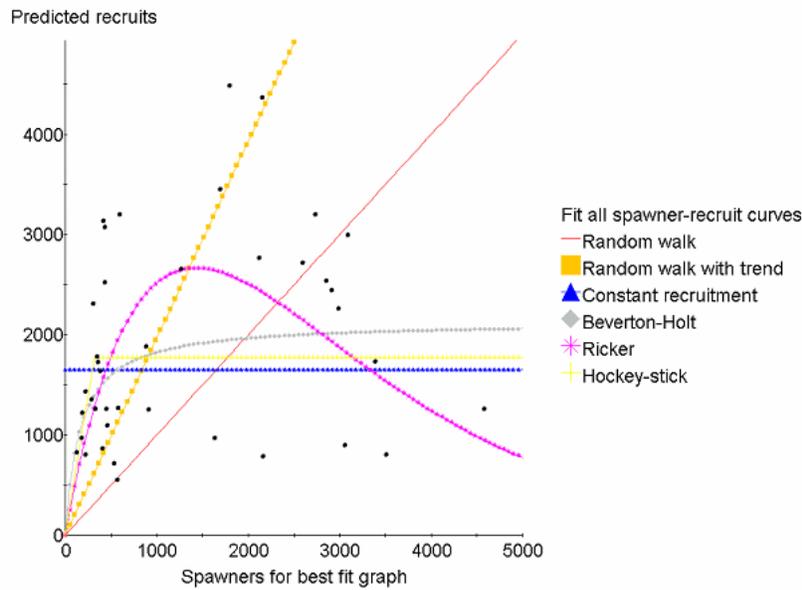
Figure C-7 Clackamas Spring Chinook pre-harvest recruitment
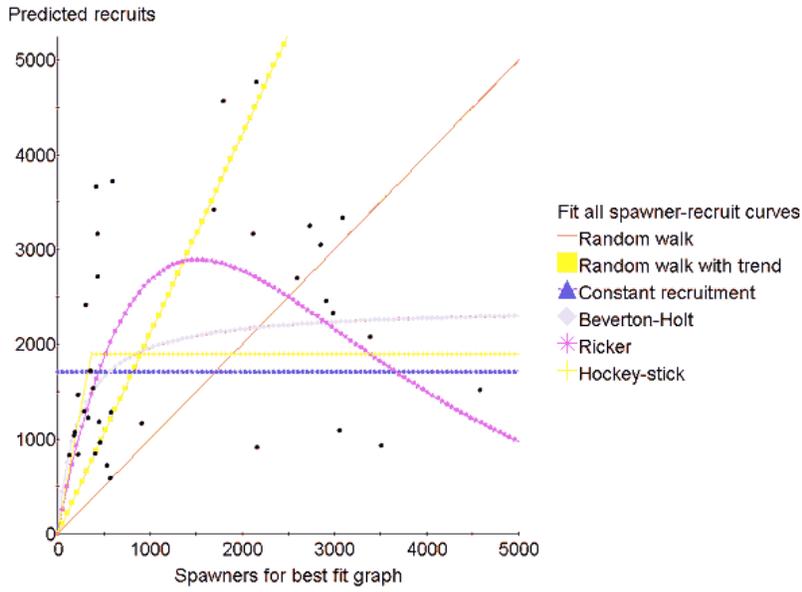
Figure C-8 Clackamas Spring Chinook pre-harvest mar. std. recruitment

McKenzie Spring Chinook
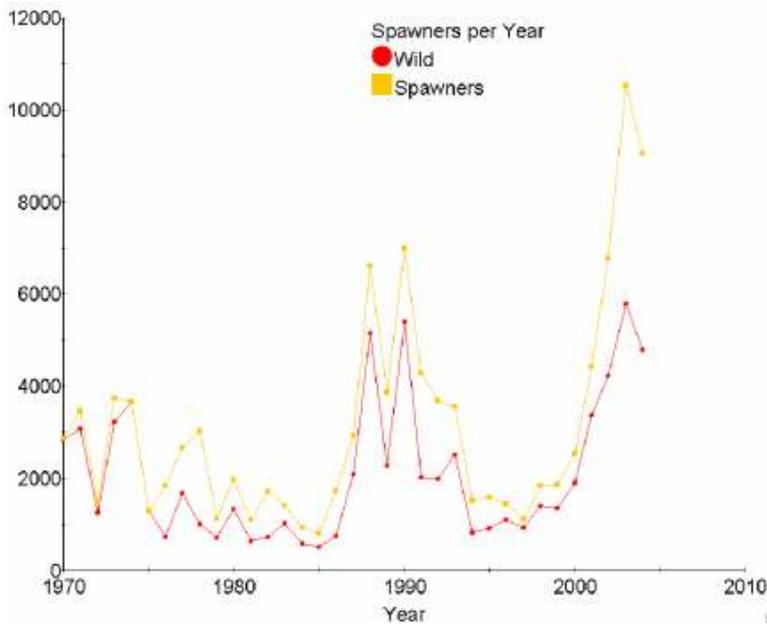
Figure C-9 McKenzie Spring Chinook Abundance
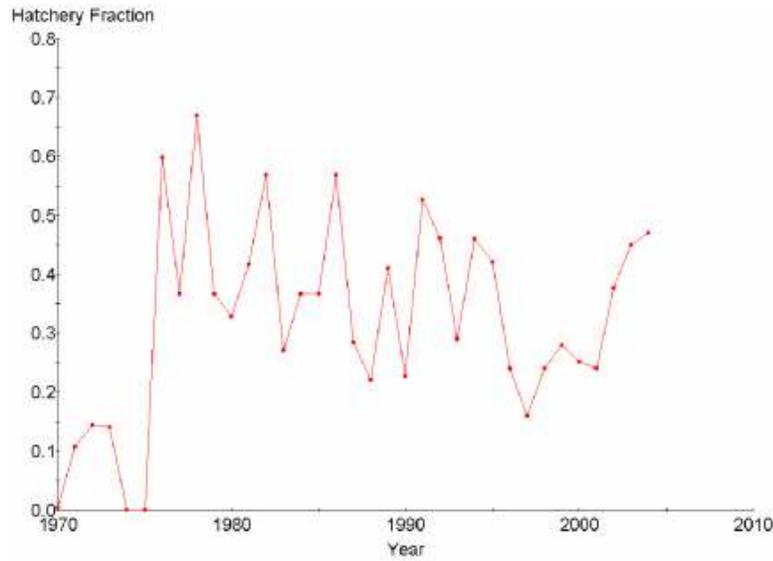
Figure C-10 McKenzie Spring Chinook Hatchery Fraction
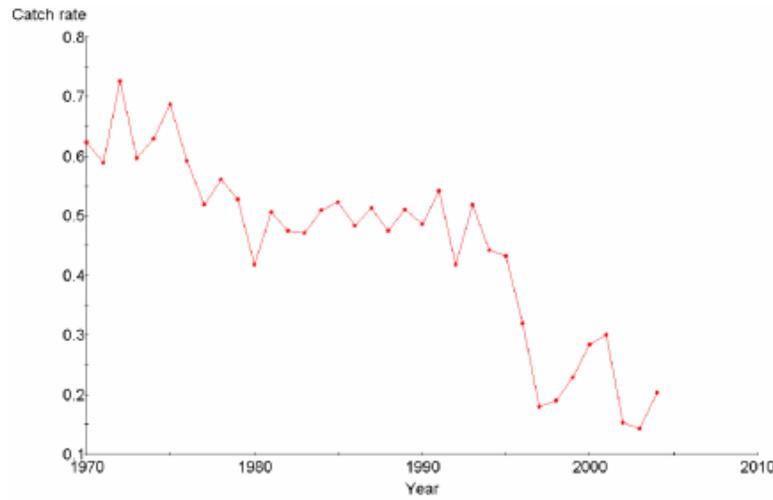
Figure C-11 McKenzie Spring Chinook Harvest Rate
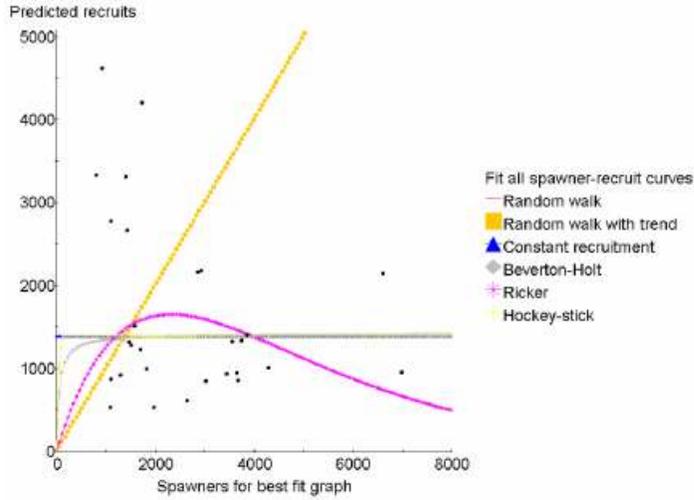
Figure C-12 McKenzie Spring Chinook post-harvest recruitment
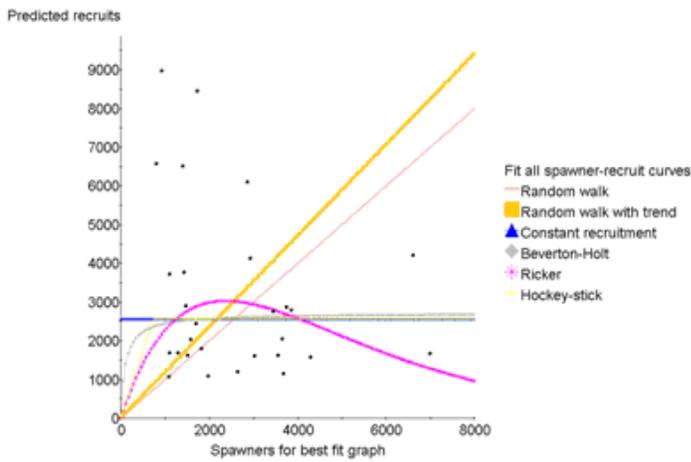
Figure C-13 McKenzie Spring Chinook pre-harvest recruitment
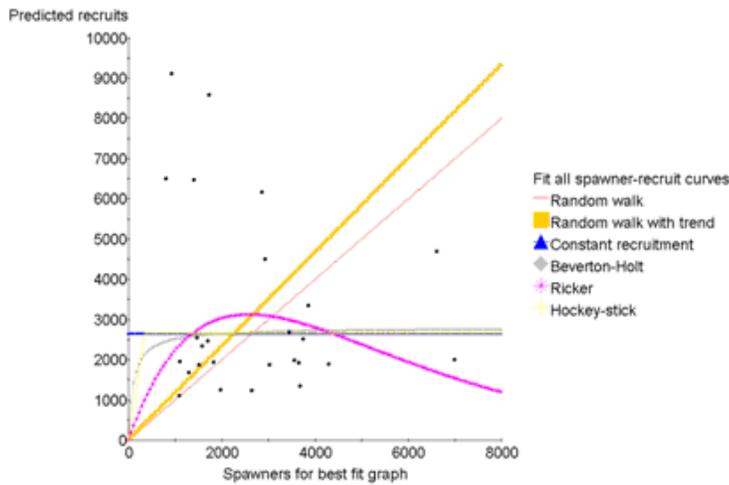
Figure C-14 McKenzie Spring Chinook pre-harvest mar. std. recruitment

Sandy Spring Chinook
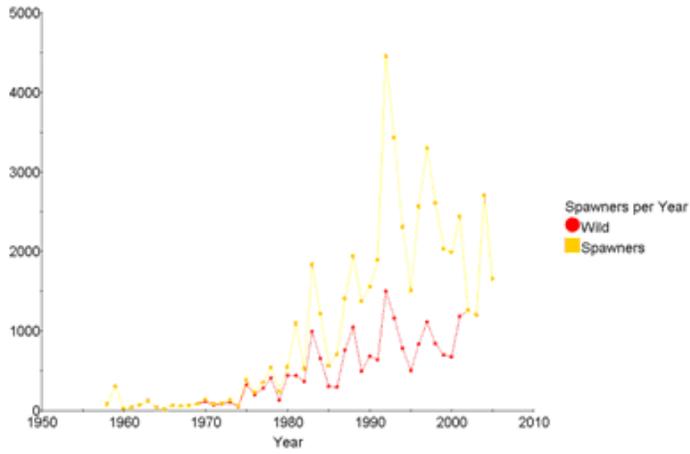
Figure C-15 Sandy Spring Chinook Abundance
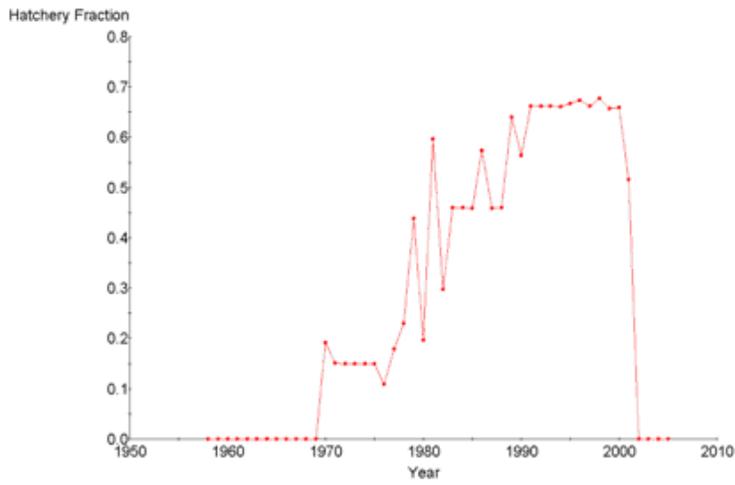
Figure C-16 Sandy Spring Chinook Hatchery Fraction
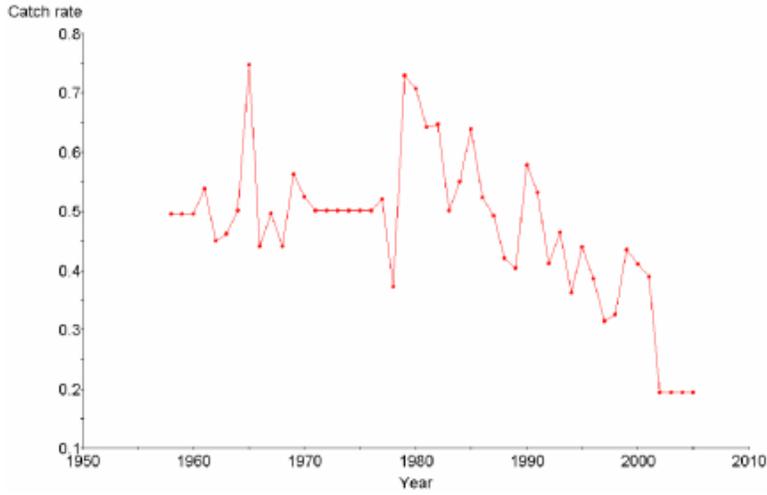
Figure C-17 Sandy Spring Chinook Harvest Rate
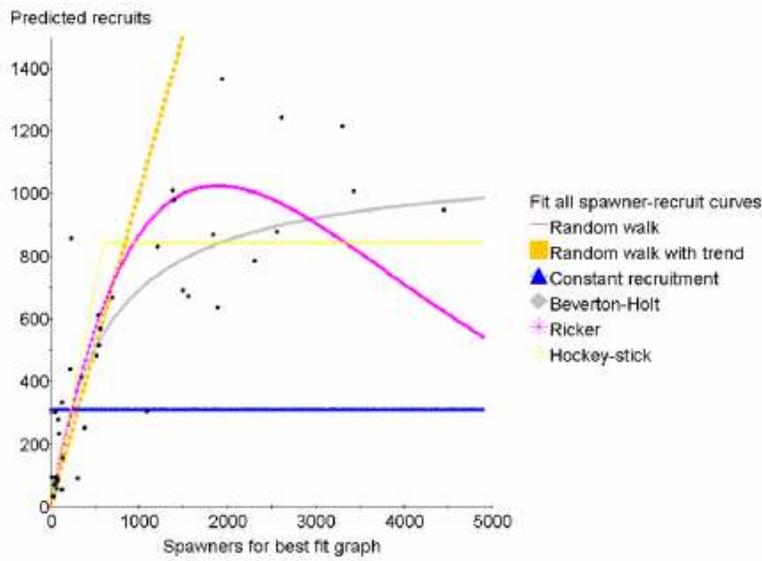
Figure C-18 Sandy Spring Chinook post-harvest recruitment
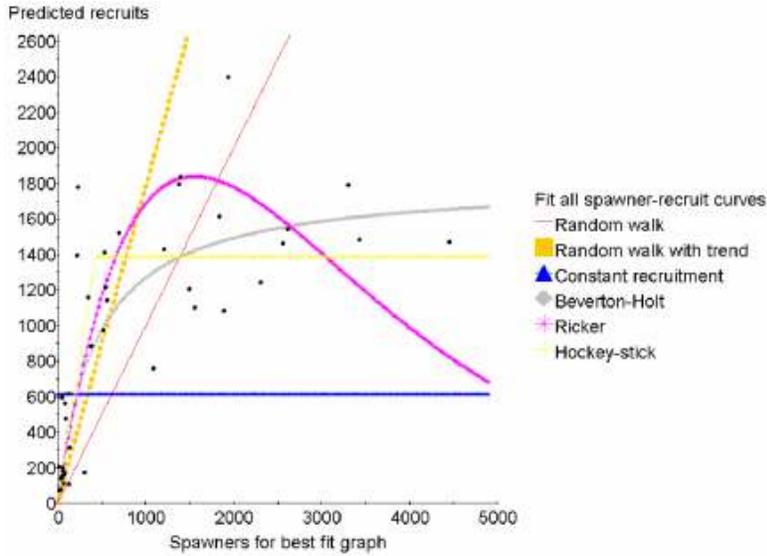
Figure C-19 Sandy Spring Chinook pre-harvest recruitment
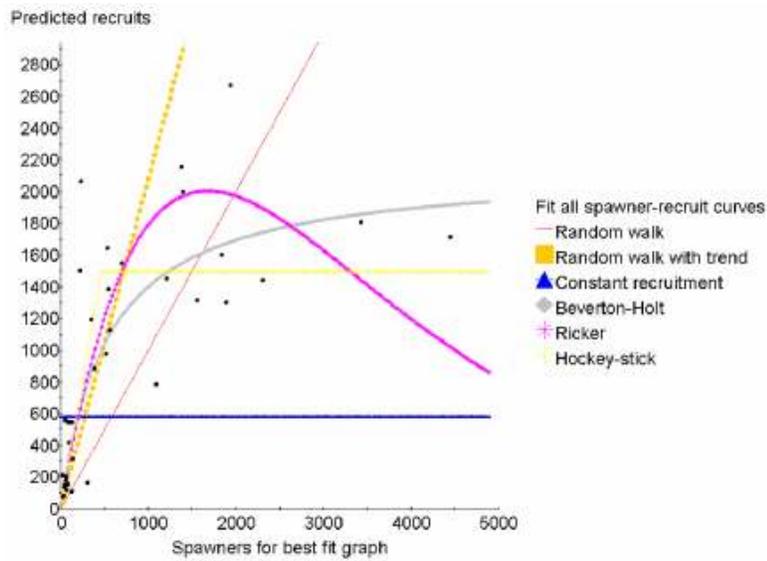
Figure C-20 Sandy Spring Chinook pre-harvest mar. std. recruitment

Clackamas Spring Chinook

Clackamas Coho
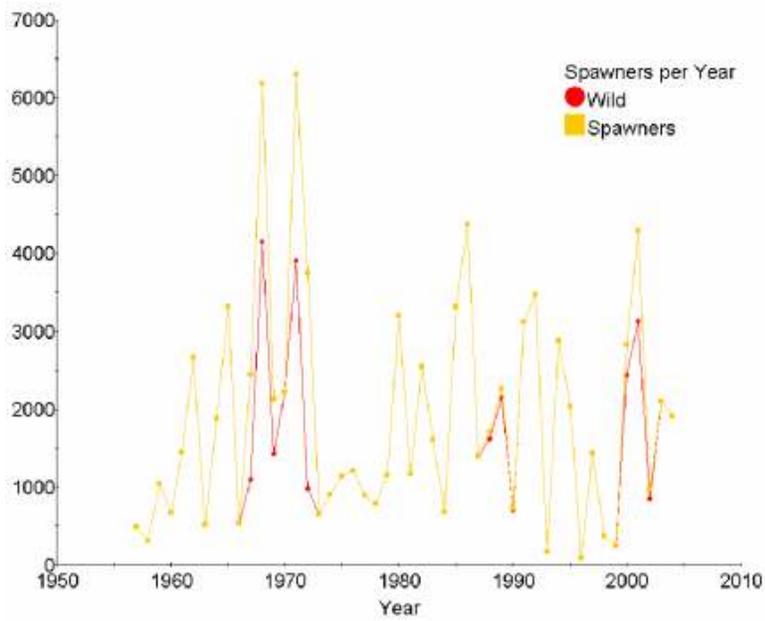
Figure C-21 Clackamas Coho Abundance
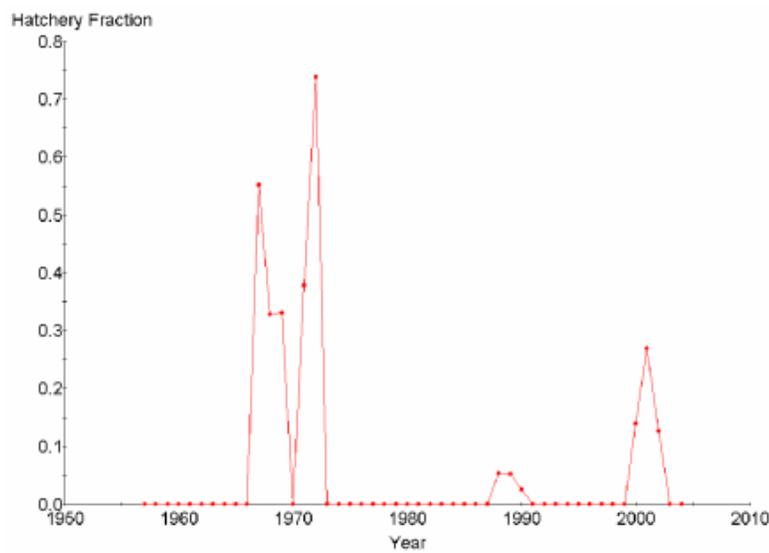
Figure C-22 Clackamas Coho Hatchery Fraction

Figure C-23 Clackamas Coho Harvest Rate
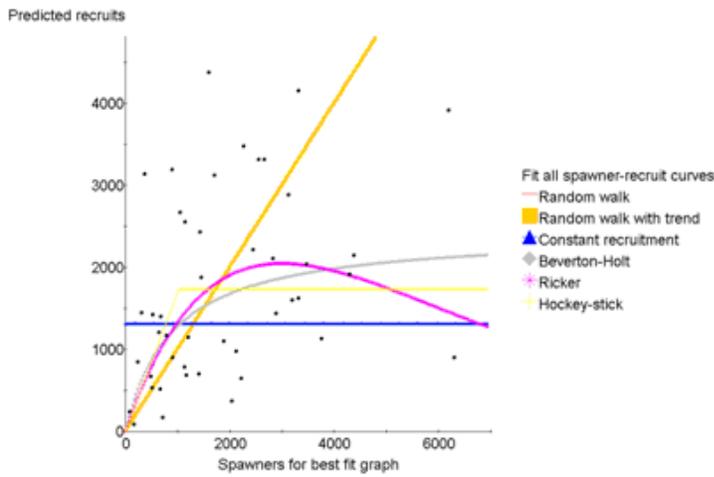
Figure C-24 Clackamas Coho Post-Harvest Recruitment
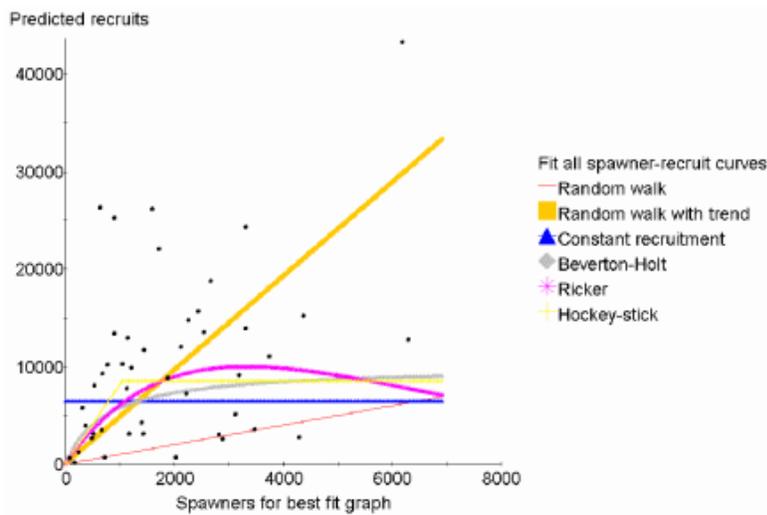
Figure C-25 Clackamas Coho Pre-Harvest Recruitment
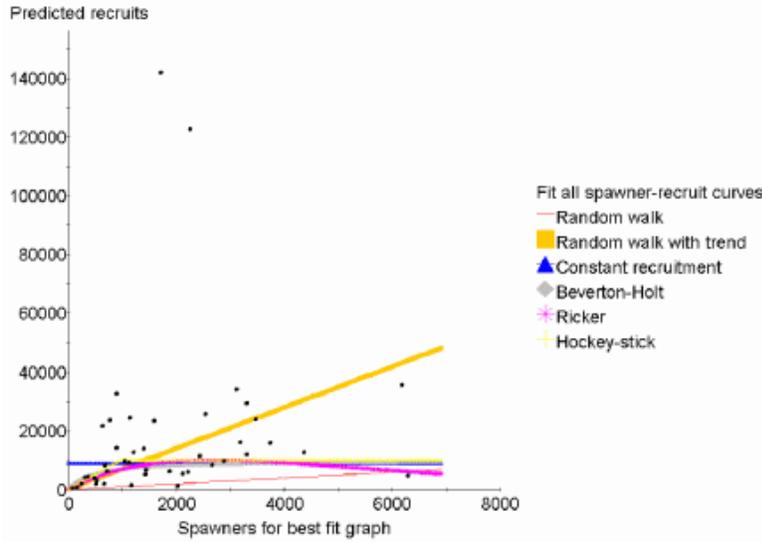
Figure C-26 Clackamas Coho Pre-Harvest mar. std. Recruitment

Sandy Coho
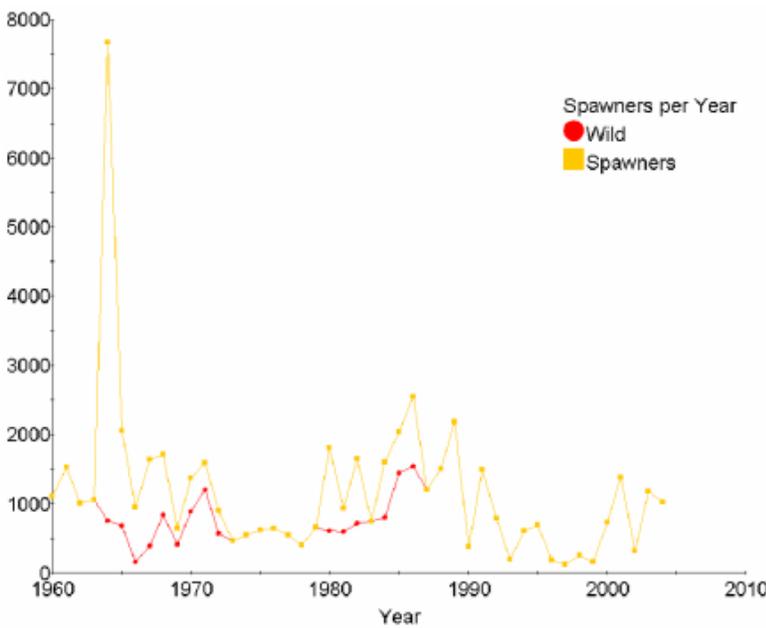
Figure C-27 Sandy Coho Abundance
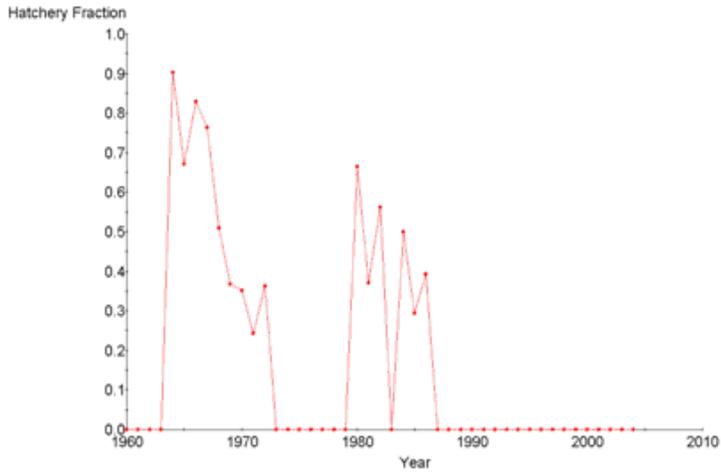
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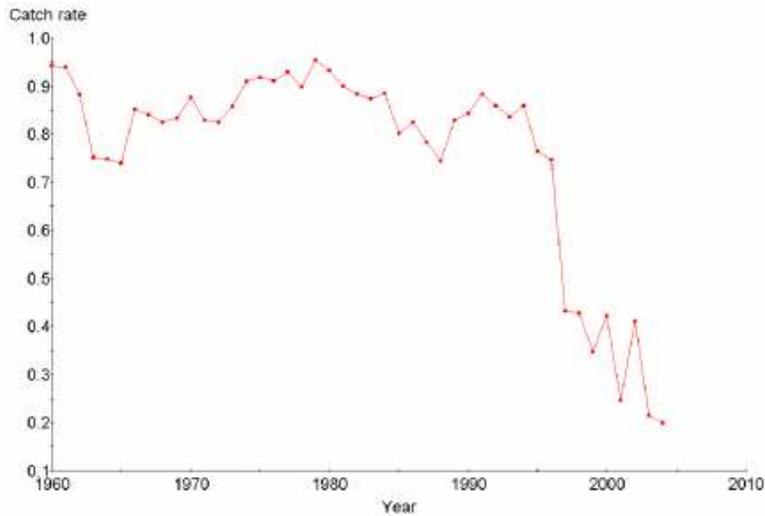
Figure C-29 Sandy Coho Harvest Rate
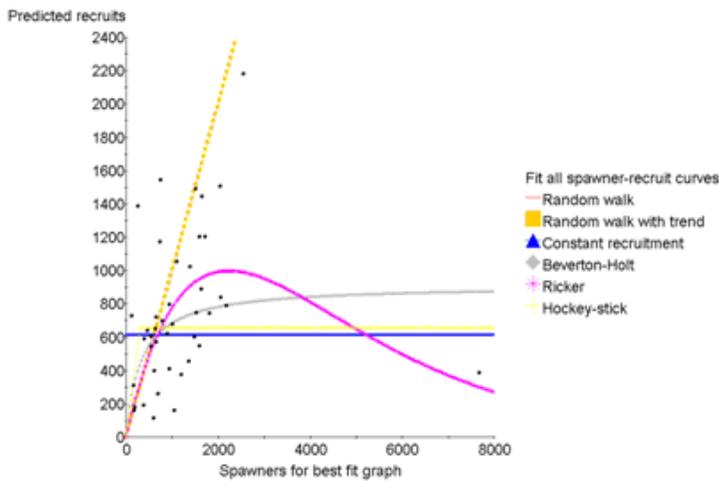
Figure C-30 Sandy Coho Post-Harvest Recruitment
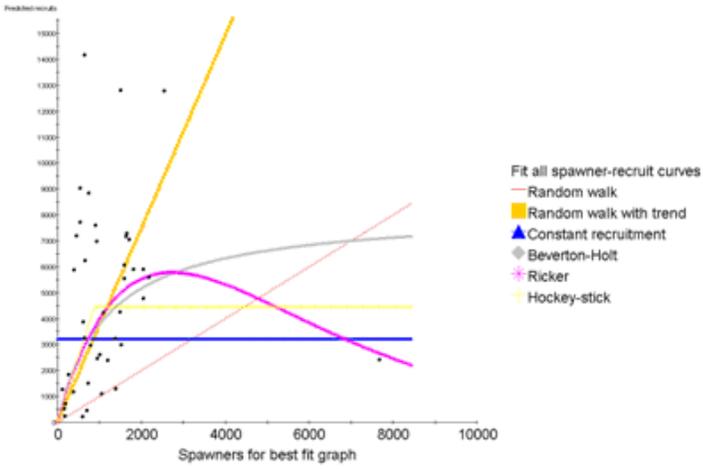
Figure C-31 Sandy Coho Pre-Harvest Recruitment
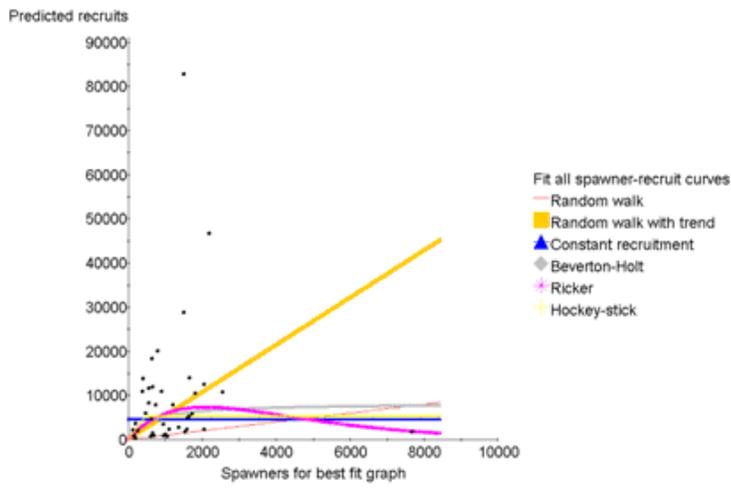
Figure C-32 Sandy Coho Pre-Harvest mar. std. Recruitment

Appendix D: Viability Curve Sensitivity Analysis

John Payne, Ejiro Akporobaro, and Paul McElhany

In the body of this report, we describe a method for comparing a "viability curve" with a set of probability contours centered on an estimate of the current population status.

A **viability curve** shows combinations of abundance and productivity that give a population a certain extinction risk. A viability curve is created by simulating many population trajectories and summarizing information about the extinction risk associated with each set of parameter values. The position of a viability curve is determined by 1) a group of parameters that describe the generation-to-generation population growth function, and 2) parameters that describe policy options, such as an acceptable risk of extinction and a time horizon.

The **probability contours** describe uncertainty about the current status of the population. The position and shape of the contours are determined by 1) *measurement error* in the data, in quantities such as annual spawner counts and estimated fraction of hatchery fish, and 2) *parameter estimation error*, i.e., error that comes from estimating unknown quantities from the data. This type of error is influenced by our parameter estimation technique.

The plots in the body of the report give a quick visual way to assess the current status of the population in relation to estimated extinction risk, but do not fully reflect the many dimensions of uncertainty around the viability curves and current status estimates. This appendix is a demonstration of the effects of the parameters that have the largest influence on viability curves and status contours. We use a single data set (Clackamas Spring Chinook) and the same default parameter set used in the body of the report, and then vary influential parameters one at a time to gauge their influence.

Note:

- The stock-recruit function affects *both the viability curve and the probability contours* (section 1).

- Data characteristics (section 2), assumptions about catch (section 3), measurement error (section 4), marine survival index (section 5), and the relative reproductive success of hatchery fish (section 6) affect *only the probability contours* around our estimate of current population status.

- Variance (section 7), autocorrelation (section 8), generation time (section 9), QET and RFT (section 10), and the policy decisions of acceptable risk (section 11) and fishing mortality (section 12) affect *only the viability curve*.

1. Effect of Recruitment Function
The recruitment function defines the generation-to-generation relationship and as such is at the heart of both the viability curve simulations and the estimation of error, and the choice of recruitment functions can have strong effects. We found (Payne and McElhany, in prep.) that very simple, unrealistic functions such as a Constant Recruitment model (recruits = R*, where R* is the average recruitment) often fit the data better than 3-parameter models traditionally used in fisheries, such as the Ricker, Beverton-Holt and Hockey Stick models. Most of the data showed evidence of density dependence, but we found no statistical basis for preferring one of

the 3-parameter curves over another. Further, the parameters from the curves are not directly comparable; for instance, carrying capacity has a different meaning in a Ricker model than in a Beverton-Holt model, where it is rarely reached.

In the following example, the current status probability contours were generated by using a Bayesian MCMC routine to fit a recruitment curve to the data. In each case, the function fitted to the data was used to create the viability curve. The comparison shows the relative similarity of the Beverton-Holt and Hockey Stick models, which tend to fit the data similarly, and produce similar projections. The Ricker model has a descending limb at high spawner densities, which has two effects: 1) it tends to fit the data better (although there is no reason to think it is truly a better model); and 2) it generates a (debatably) pathological extinction risk curve, where higher productivity is associated with higher extinction risk, because high spawner numbers move the recruitment toward the lower end of the descending limb of the curve. Both effects are visible in this example.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-1: Hockey Stick model
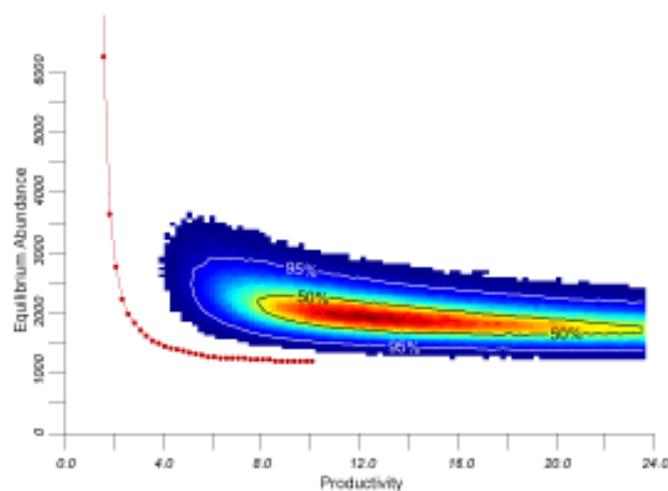
Parameter	Value
Population	Clackamas Spring Chinook
Model	Beverton - Holt
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-2: Beverton-Holt model
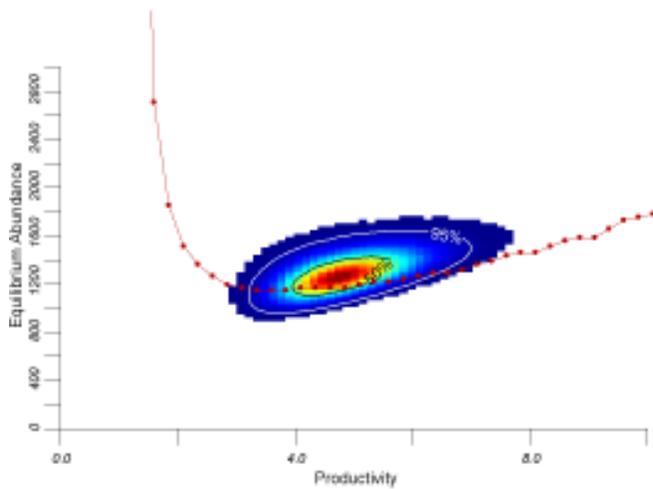
Parameter	Value
Population	Clackamas Spring Chinook
Model	Ricker
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-3: Ricker model

2. How characteristics of the data affect the current status contours

The probability contours around the estimate of current population status represent our uncertainty about the exact values of population size and productivity. Therefore, the shape of the contours depends on the estimation method we use, and on the variability of the data. In this section, we have fitted a Hockey Stick model to three data sets.

In the first example (Figure D-4, Sandy Spring Chinook), the data fit reasonably well. The second example (Figure D-5, Clackamas Coho) demonstrates that productivity and abundance estimates are correlated. The data are fit fairly well by a curve with a carrying capacity around 7500 fish and productivity of around 8.0 recruits/spawner. However, the extension to higher abundances along the y-axis shows that a curve with lower productivity (around 5) and much higher carrying capacity (up to 50,000) fits the data almost as well. A common problem, particularly in populations with large or unknown hatchery influence, is that a lack of data at low spawner densities often results in a poorly-defined upper limit for our estimate of productivity (Figure D-6, McKenzie Spring Chinook).
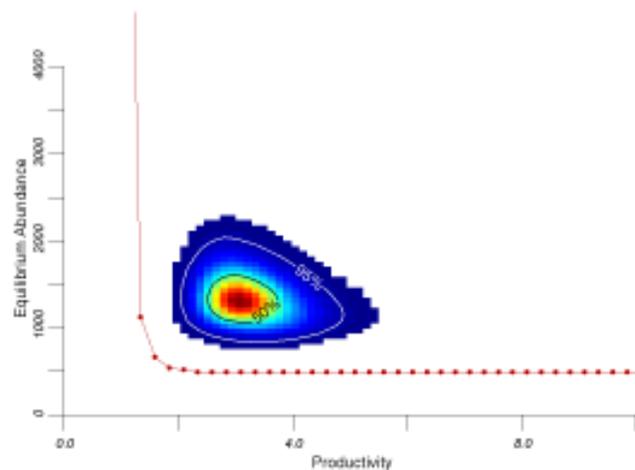
Parameter	Value
Population	Sandy Spring Chinook
Model	Hockey – Stick
RFT	150
QET	300
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	SNEG
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-4: A data set that fits reasonably well.
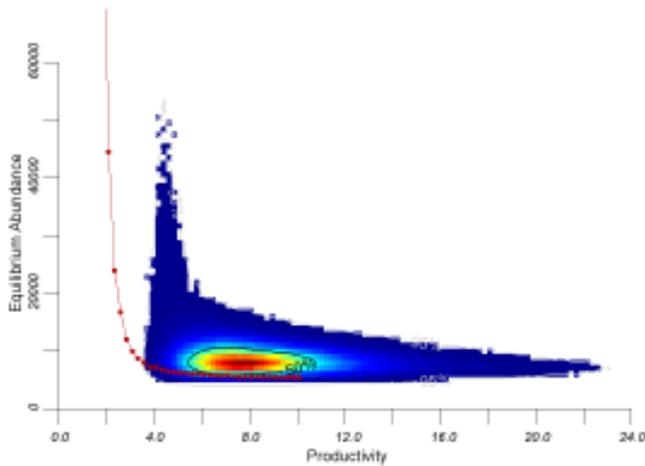
Parameter	Value
Population	Clackamas Coho
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-5: A data set showing correlation between productivity and carrying capacity estimates.
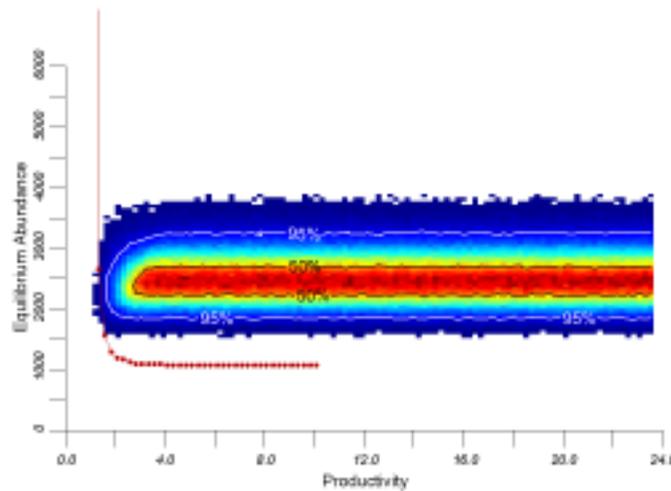
Parameter	Value
Population	McKenzie Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-6: A data set with an undefined upper limit to productivity, caused by lack of information at low spawner abundances.

3. Effect of pre-catch vs. post-catch

Catch distorts our estimates of the population's *potential* productivity and abundance, because adults that survive and return to spawn, but are caught in terminal fisheries, are not counted as recruits. Figure D-7 shows a pre-catch estimate, in which we have calculated productivity and abundance as if those spawners had returned. Figure D-8 shows the population's *actual* (realized) productivity and abundance, estimated from post-catch (i.e., observed) recruits. Both plots are important for policy decisions.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	None
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-7 Pre-catch (i.e., potential, without catch) productivity and abundance.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	None
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-8 Post-catch (i.e., observed) productivity and abundance.

4. Effect of measurement error

Currently available salmon data sets contain large amounts of measurement error in spawner counts, hatchery proportion, age composition of spawners, and catch. To explore the effects of measurement error, we simulated 500 new data sets for each level of error by adding stochastic, uniformly-distributed error to our Clackamas Spring Chinook data set. We fitted a Hockey Stick curve to each of the simulated data sets using an MCMC process and saved some of the posterior distribution points from each run. The resulting über-posterior distribution can be interpreted as an indication of the effect of measurement error on our parameter estimates. Figure D-9 shows the original data set without measurement error. Figure D-10 shows measurement error of ±10% in spawner counts, hatchery proportion, and catch; the age distribution of spawners was drawn from a multinomial distribution with a mean of the observed distribution and a sample size of 50. Figure D-11 shows errors of 20%, with a smaller sample size (20) for age data (which gives more variation from the observed data).

Surprisingly, although a general increase in uncertainty is observed (the contours spread out) with greater measurement errors, even the addition of substantial error does not change our

parameter estimates by much. We interpret this as an indication that the data are already noisy, as is evident in the stock recruitment function (Figure D-12).

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	None
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-9 Estimated population status from the original data set, without added error.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±10%; ±10%; 50; ±10%; ±10%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-10 - 10% measurement error, with a sample size of 50 for age.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±50%; ±50%; 20; ±50%; ±50%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-11 - 20% measurement error, with a sample size of 20 for age.
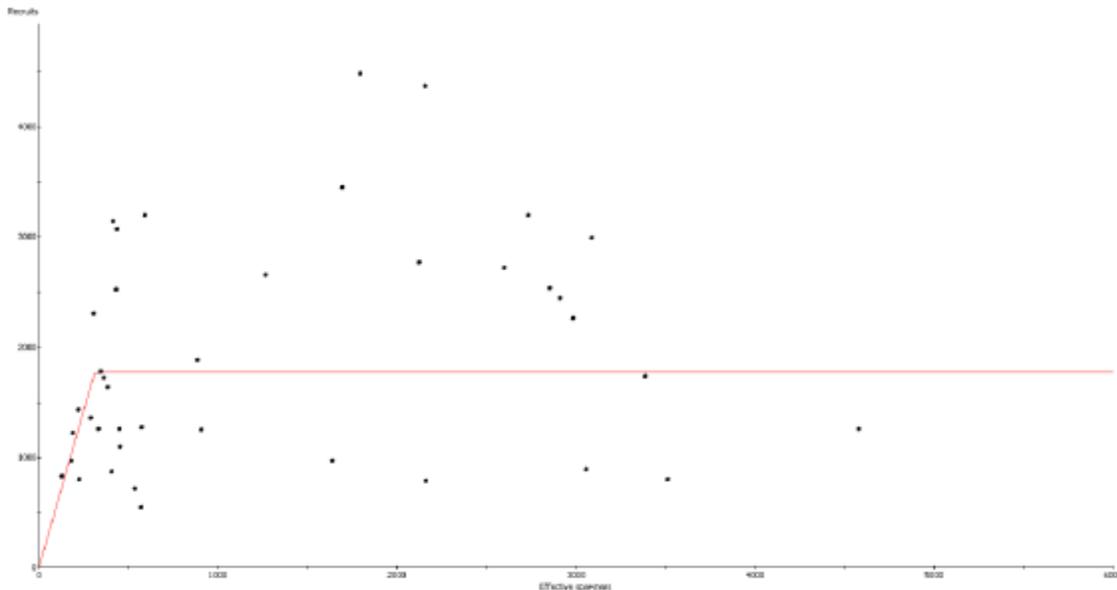
Figure D-12 Spawners (x-axis) vs. recruits (y-axis), for the data set used in Figure D-9 through Figure D-11. The curve is a Hockey Stick function.

5. Effect of marine survival index

There is evidence that ocean conditions cause strong variation in year-to-year survival of anadromous salmonids, and that for many species the transition to the marine environment is a time of high mortality. Experience shows that adjusting recruitment for constant smolt-to-adult survival can make the stock-recruitment relationship much tighter. Unfortunately, smolt-to-adult survival data are rarely collected, and researchers have tried to identify environmental indices that can be used as a proxy for survival data when survival data are not available. To date, most of the correlations found have been weak or unreliable. One environmental index that has been used by Oregon researchers is the SNEG index of snow depth (Figure D-13). The index has been scaled to have a mean of 1 and we used it to standardize recruitment for average conditions (Figure D-14). Applying the SNEG index changes our estimates of recruitment, and therefore also influences the shape of the probability contours for the current population status (Figure D-15 and Figure D-16).
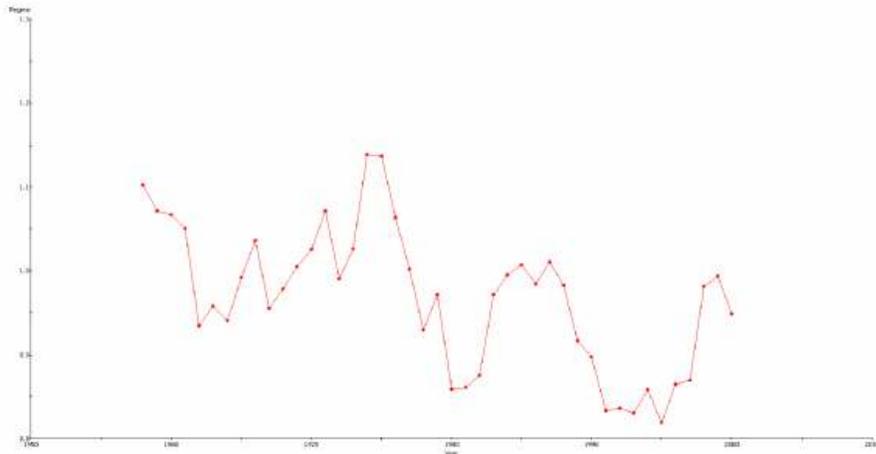
Figure D-13 SNEG index of snow depth from 1960 - 2000.
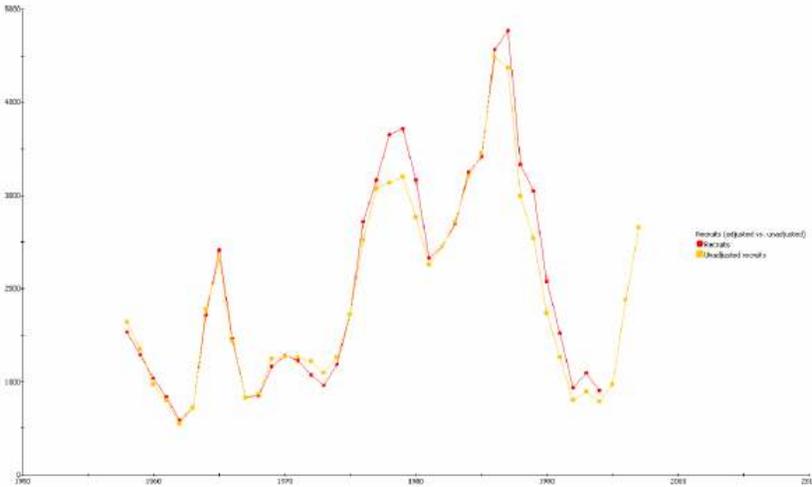
Figure D-14 Recruitment with and without adjustment for marine survival.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20% ; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-15 No marine index adjustment.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	SNEG
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-16 Population status, with recruits adjusted for SNEG index.

6. Effect of assumptions about the relative reproductive success of hatchery fish

Hatchery and wild fish must be separated in our analyses if we are to understand the productivity and abundance of wild salmon populations. To this end, the data sets we use contain an estimate of the number of hatchery fish that spawn on the spawning grounds with wild fish. However, it is not known how much these hatchery fish contribute to recruitment in following years because a critical parameter, the reproductive success of the hatchery fish relative to wild fish, is not well understood. A few studies have suggested values substantially less than 1, and we demonstrate values of 0 (no success) to 0.5 (half as successful as wild fish) to 1 (same as wild fish) in Figure D-17 through Figure D-19. Whether relative fitness has a strong effect on an individual data set depends on the fraction of spawners that are hatchery fish, and the pattern of recruitment. This data set shows little change; however, it is common for the same range of relative fitness to make a difference of up to 3x in productivity estimates when the fraction of hatchery origin fish is consistently large.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years
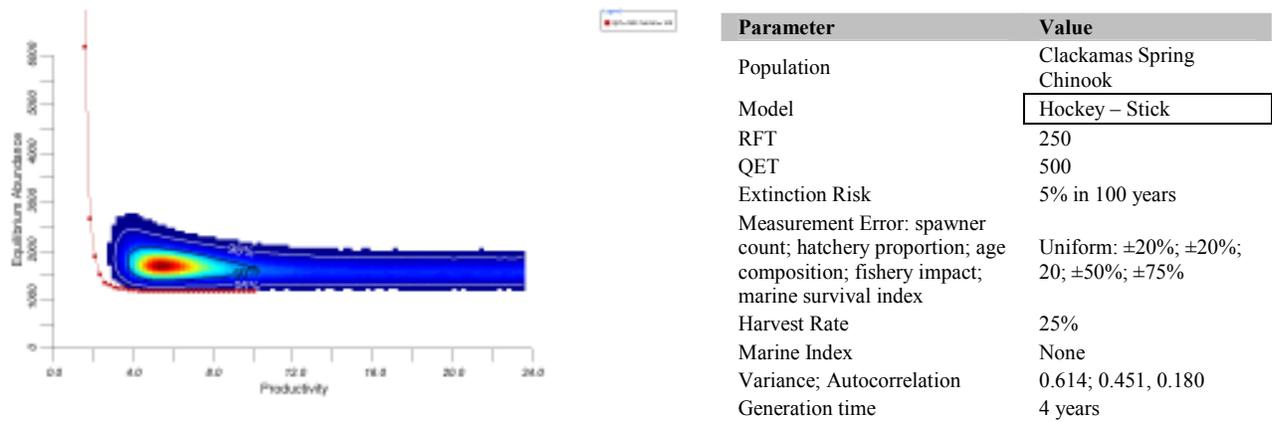
Figure D-17 Relative hatchery success = 0 (no reproduction by hatchery fish).

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years
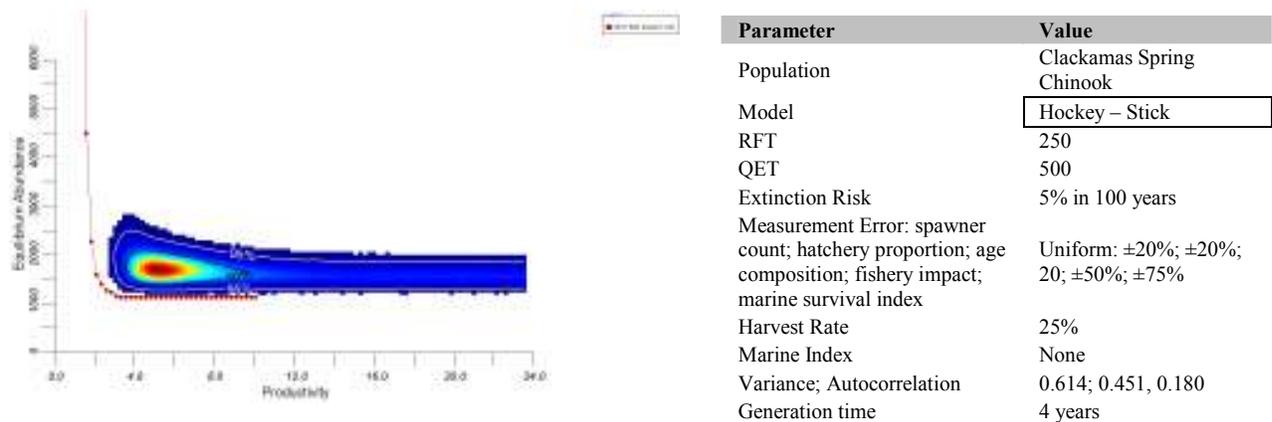
Figure D-18 Relative reproductive success of hatchery fish = 0.5 (half as successful as wild fish).

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years
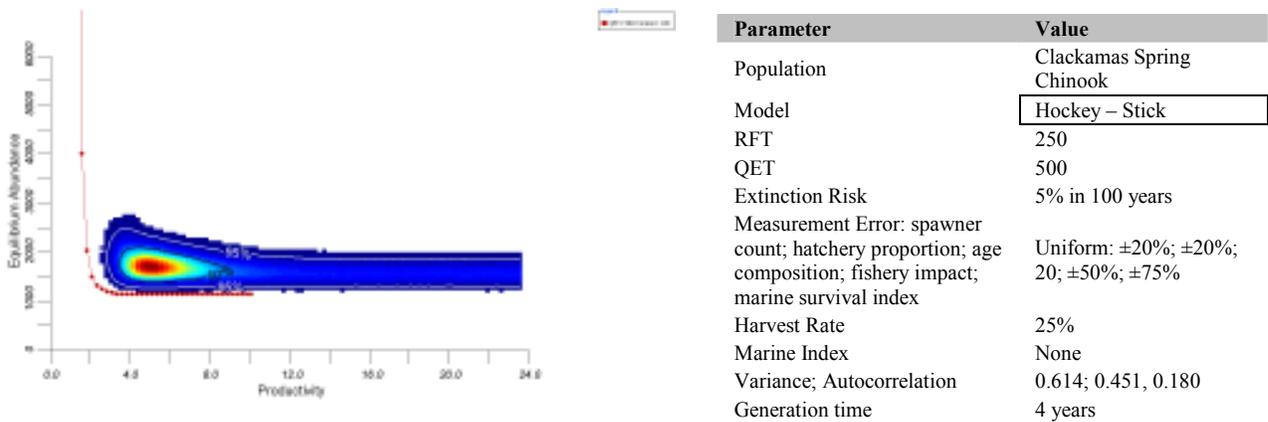
Figure D-19 Relative reproductive success of hatchery fish = 1 (the same as wild fish).

7. Effect of variance on position of the viability curve

To create a viability curve, population growth trajectories are simulated by adding stochastic error to a deterministic stock-recruitment function. In the body of this report, we estimate the stochastic error as the residual variance from fitted stock-recruit curves, pooled across populations. This process probably overestimates the variance, since it includes both observation and process error, and the magnitude of the variance affects simulations. On average, higher variance makes simulated trajectories more variable and therefore more likely to go extinct, which is the pattern seen in Figure D-20. However, for a population that is on average declining, high variance may reduce extinction risk by increasing the likelihood of a rare high productivity year.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.3, 0.6, 1.2, 3; 0.451, 0.180
Generation time	4 years
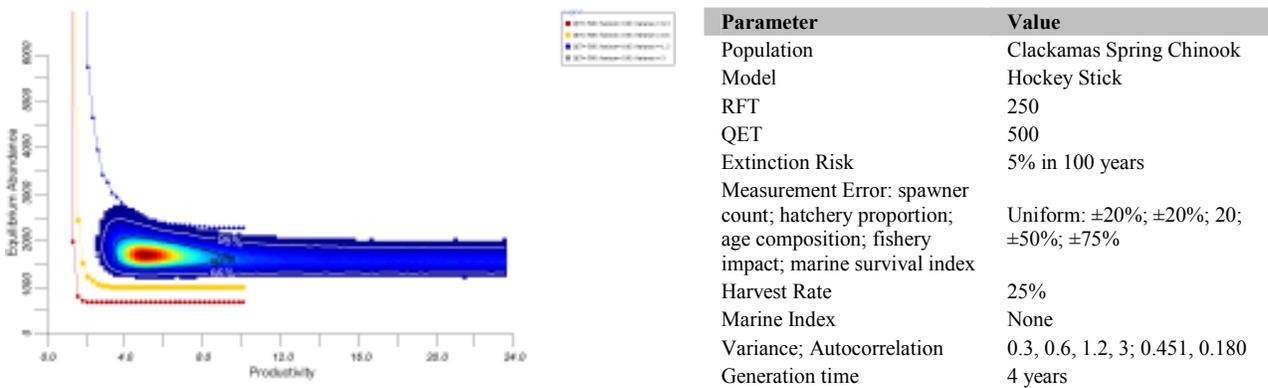
Figure D-20 Viability curves estimated with variance from 0.3 (approximately half of the minimum value used in the report) to 3 (about twice the maximum used in the report). A population with a high variance must be larger than a population with low variance, to have the same risk of extinction.

8. Effect of autocorrelation on viability curves

In nature, population growth rates tend to be autocorrelated; that is, successive years are more similar to each other than would be expected by chance. This tends to cause swings of bad and good years that last longer than they would otherwise. We roughly estimated autocorrelation by calculating it at 1- and 2-year lags for all of the populations and then using the mean values of (0.451, 0.180) for the simulations. The effect on simulated trajectories is similar to what you would see with a higher variance; it tends to require a population to have a higher size and productivity to be viable (Figure D-21).

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180 And 0.614; None
Generation time	4 years

Figure D-21 Viability curves with and without autocorrelation.

9. Effect of generation time on viability curves

We define extinction as having occurred when the population stays below the quasi-extinction threshold (QET) for at least one generation. In the body of the report, we used a generation time of 3 years for coho, 4 years for chinook, and 5 for steelhead, based on the mean age of reproduction of each species, averaged over populations. Longer generation times are slightly more resistant to extinction (Figure D-22).

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	3, 4 , 5 years

Figure D-22 A chinook population, compared to viability curves with generation times of 3, 4 and 5.

10. Effect of QET and RFT

The quasi-extinction threshold (QET) is defined as the threshold below which the population is considered extinct. The principle reason for *not* using a threshold of zero (actual extinction) is that populations tend to experience many problems at small sizes that are not well understood or easily quantified and act stochastically, including genetic problems such as inbreeding depression; demographic problems such as unequal sex ratios; and social problems described under the general category of Allee effects. A reproductive failure threshold (RFT) is defined as a threshold below which no reproduction occurs. It can be defined separately from QET (Figure D-23), but we have set RFT = QET in the body of this report, as it is easier to interpret the results. Both thresholds have strong effects on the location of the viability curve, and the higher of the two tends to determine the location of the curve, as can be seen by comparing Figure D-23 and Figure D-24.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250, 500, 1000
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-23 Reproductive failure threshold (RFT) set at one half, equal, and twice the value of the quasi-extinction threshold (QET).

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	1, 50, 100, 500, 1000
QET	1, 50, 100, 500, 1000
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-24 Reproductive failure threshold and quasi-extinction threshold equal, and set at 1, 5, 10, 100, and 1000.

11. Effect of acceptable extinction risk

Viability curves have the same risk of extinction at every point. Policy makers must decide which viability curve to compare the current population status to. We present curves for several different risk levels (Figure D-25). The choice of time horizon, not shown here, is also a policy decision.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	1%, 5%, 25%, 40% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	25%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-25 Viability curves for 4 different levels of acceptable risk (1, 5, 25, and 40 percent).

12. Effect of fishing mortality

The future catch rate assumed has a very strong effect on population trajectories, and hence on the position of the viability curve (Figure D-26). We applied fishing mortality by subtracting a fixed proportion of recruits from the recruitment each year, using mortalities of 0, 20, 50, and 75%. Historical fishing mortality rates were as high as or higher than 75% for many stocks, and most present rates are much lower, although few go much lower than 20%.

Parameter	Value
Population	Clackamas Spring Chinook
Model	Hockey – Stick
RFT	250
QET	500
Extinction Risk	5% in 100 years
Measurement Error: spawner count; hatchery proportion; age composition; fishery impact; marine survival index	Uniform: ±20%; ±20%; 20; ±50%; ±75%
Harvest Rate	0%, 20%, 50%, 75%
Marine Index	None
Variance; Autocorrelation	0.614; 0.451, 0.180
Generation time	4 years

Figure D-26 The effect on viability curves of applying fishing mortality to simulated population growth trajectories. The curves show mortality of 0, 20, 50, and 75%; obviously a much larger population is required in order to support high mortality from fishing. The time horizon of 100 years is probably influential here as well; one would expect that with mortality rates as high as 75%, the population might not be viable in the longer term.

Appendix E: Thresholds for Quasi-extinction and Depensation

Mark Chilcote

As discussed in the main body of this report, modeling population viability requires that an extinction threshold be set. One option would be to set this value at zero. To do so would require that the viability model be able to accurately simulate population recruitment at extremely low spawner abundance levels. However, the recruitment process at such low levels is poorly understood. It is poorly understood because for most populations very few low abundance data points are available from which recruitment rates can be examined. Therefore under these circumstances, picking "near zero" recruitment rates to be used in a population viability model simulations becomes more of a guessing game than an exercise based upon empirical data. The WLC TRT wished to avoid this situation. Therefore, the group decided to established test thresholds at greater than zero levels. This non-zero extinction threshold for the purposes of viability modeling is often referred to as the quasi-extinction threshold (QET).

Development of QET values for WLC salmon and steelhead populations was in part based on the literature concerning the phenomenon of depensation at low spawner abundance. Empirical observation of low spawner density recruitment in WLC was also used to develop QET values. Both of these elements are discussed in the following paragraphs.

Depensation

Some have suggested that by inspecting spawner–recruit data sets it may be possible to identify a threshold spawner abundance, below which the underlying recruitment performance of a population starts to fail because spawners can not find mates or for other reasons that are not understood (Myers 2001). This breakdown of the recruitment function at extremely low spawner densities is commonly referred to as depensation. The spawner density corresponding to the onset of depensation could be used as a reference point for setting QET values, the underlying concept being that if the population level drops to a level less than this assumed depensation point (threshold) then recruitment performance may begin to fail. Should this occur, then the population abundance could continue downward over the course of the next 2 to 5 generations until the population becomes truly extinct (zero fish). This scenario has been referred to by some as the "extinction vortex."

However, the statistical challenge of demonstrating the spawner density level at which depensation begins is daunting (Shelton and Healey 1999). Barrowman (2000), examining recruitment data for coho populations primarily in British Columbia, suggested that depensation may become a factor for coho populations when spawner densities are less than 1 female per kilometer of stream length. Assuming a 50/50 sex ratio, this equates to a spawner density of 2.0 fish per kilometer. Chilcote (1999) observed that several populations of coho in the Lower Columbia River seemed to lose their innate resilience (expressed in terms of recruits per spawner) at low spawner densities of 3.9 fish per mile (2.3 fish per kilometer) and essentially became extinct in the 1990s. These results seem to support the conclusions of Barrowman (2000).

For coho, therefore, it appears that for spawner densities less than 2.0 fish per kilometer, one might conclude that the production of recruits is progressively less than recruitment curve

predictions as a result of depensatory processes. Less clear is whether 2.0 fish per kilometer is an accurate value for the onset of depensation for other related species such as steelhead and chinook. However, based upon anecdotal information spawner densities for steelhead and spring chinook tend to be naturally less than for coho. Therefore, it is possible that the onset of depensation for these species may be at densities less than 2.0 fish per kilometer. However, lacking specific information that supported this view, a spawner density of 2.0 fish per kilometer as the threshold for the onset of depensatory processes may be the best one to generically apply to the species evaluated in this report.

<u>Empirical Evidence</u>
For most populations, spawner densities less than 2.0 fish per kilometer have not been observed and therefore the verification of a supposed depensation process is difficult. However, when data from multiple populations was pooled, a sufficient number of points were generated to perform a rough look for evidence of depensation. This analysis was possible for only coho and steelhead.

A plot of natural log recruits per spawner on spawner density for Clackamas and Sandy coho data generally yielded values greater than replacement when the examination was restricted to those years when the observed spawner density was less than 16.0 fish per kilometer (Figure E-1).
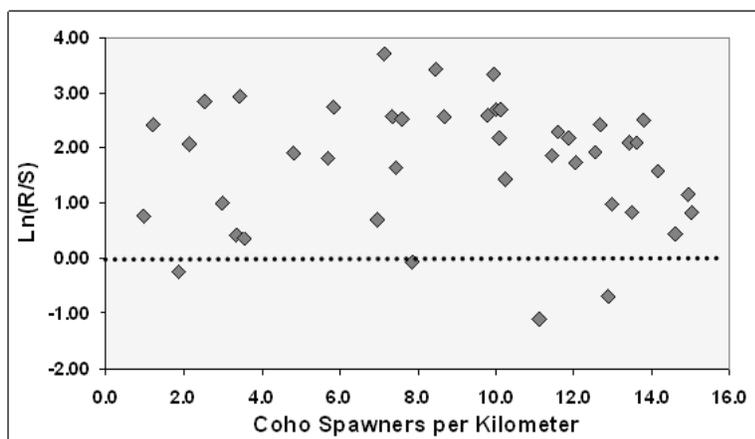
Figure E-1 Plot of recruits produced per spawner for those observations when spawner densities were less than 16 fish per kilometer in the Clackamas and Sandy coho populations spawning in the period from 1958 to 2001.

Although data points do not arrange themselves in any clear pattern, visually it appears that the recruits per spawner values seem to level out or even decline at the lower spawner densities. This is possibly evidence of depensation. In contrast, similar comparisons using steelhead data do not suggest any leveling out or decline at low spawner density (Figure E-2). There appears to be a broadly defined increase in recruitment rates as the density of spawners decrease. Even at quite low densities this relationship seems to hold.
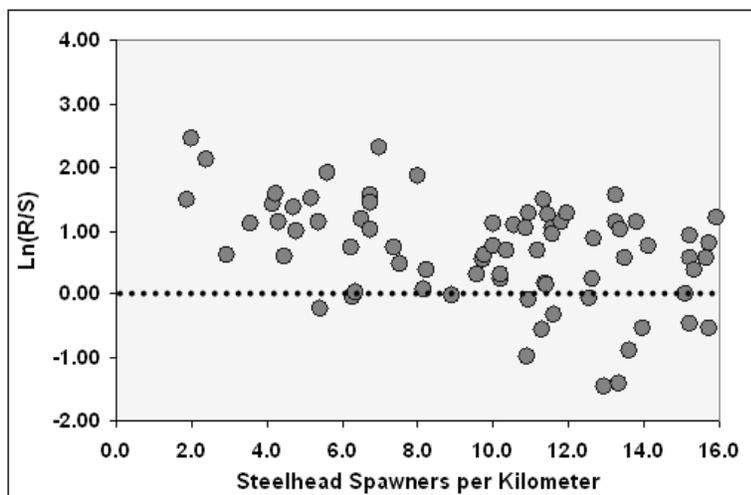
Figure E-2 Plot of recruits produced per spawner for those observations when spawner densities were less than 16 fish per kilometer for LCR and Willamette steelhead populations, 1958 to 1999 brood years.

The steelhead populations also differ from the coho populations in that the Ln(R/S) values in general are less for steelhead. This probably reflects some of the basic biological differences between these two species. Even for relatively healthy populations, coho generally occur at higher spawner densities than do steelhead.

Although it is difficult to generalize from these empirical recruitment data, one could claim that an ad hoc averaging of the coho and steelhead recruitment performance gives a result that is somewhat supportive the 2.0 fish per kilometer threshold for the onset of the depensation processes.

Setting the QET

It appeared impossible to verify a recruitment depensation threshold for all species, let alone knowing how strongly this depensation process intensifies as spawner densities decline from the threshold level to zero. Such knowledge, were it available, would have been extremely helpful in determining the low spawner density at which the population fails to replace its self and thereby defining the extinction "tip in" point. However, this degree of fine tuning for the purposes of viability modeling and forecasting the probability of extinction was not possible.

Therefore, an alternate approach was taken. The logic for this was as follows. Spawner densities less than approximately 1.0 fish per kilometer have almost never been observed. As a result, the possibility that recruitment performance at such low levels is inadequate to sustain the population (R/S < 1.0) can not be eliminated. In fact, even complete reproductive failure (R/S = 0.0) at such low spawner densities is a possibility that exits. Given this uncertainty and the desire to manage this uncertainty by erring on the side conservation, a decision was made to assume that recruitment performance for spawner densities of 1.0 fish per kilometer or less would be essentially zero (no recruits). Further that for viability modeling purposes and to be consistent with this operating assumption, the QET was also set at this same spawner density (1.0 fish per kilometer).

Putting this QET value in the context of the population viability modeling described in this report, an extinction event was defined as the occurrence of a string of annual population

abundance forecasts that when averaged over a period of a single generation (e.g., for coho, 3 years) had a value less than the QET.

Appendix F: Measurement Error in Oregon WLC Salmon Data

Mark Chilcote

Note: This analysis is intended to provide initial "ball park" estimates of measurement error and it expected that these values will be refined through evaluation of empirical data and considering a broader sample of expert opinion.

1A. Spawner abundance - Spawning survey methodology

Steelhead = ± 70% of population abundance estimate. Some serious problems with the typical data in that it is often collected during one of the worst times of year to observe fish and redds (winter/early spring) and steelhead typically spawn over a long time period in a low density fashion across much of a basin's habitat. Therefore, expanding redds per mile or fish per mile data for an entire basin is a sketchy exercise. Also, keep in mind I'm trying to capture the where range is for 90% of the points, in reality there are some years where because of floods or the unavailability of folks to sufficient to survey with the necessary frequency, there will be a real flyer estimate (or no data at all). However, in general I am assuming these fliers will fall into the other 10% outside of the 90% zone.

Chinook = ±40% of population abundance estimate. A little better here because the surveys are done during a better time of year to actually observe fish. Also, chinook tend to have a more concentrated and restricted spawner distribution compared to steelhead and therefore the expansion to a basin estimate is a little less of a reach.

Coho = ±50% of population abundance estimate. Coho are also more concentrated in their spawner distribution compared to steelhead (although not as much as for chinook). However, spawning surveys are done in a bad time of year for observations November – January.

1B. Spawner abundance – typical dam count methodology

Steelhead, Chinook, and Coho = ±20% of estimated dam passage. Although we tend to think of dam counts as nose counts, there are problems in that some fish may "fallback" after passage and exit the basin, or pass a second time and become double counted. Also, in some locations – particularly for years past, traps and/or ladders were not run 24hours a day. Often a count was made for part of the day or part of the hour and then the number expanded for the entire period. In other words, not all the "noses" were actually counted.

1C. Spawner abundance – trap and handle type of counting procedure.

Steelhead, Chinook, and Coho = ±5% of the population estimate. This is the situation where each fish that passes a facility is trapped and individually handled before being passed upstream. The Hood River trap facility at Powerdale Dam is an example of this. Assuming everyone can count, the estimates from this method are about as accurate as we can get – however, there still is the fish "fallback" problem that could introduce error.

2A. Hatchery/wild proportions - Spawning survey methodology

Steelhead = ±60% of the point estimate for proportion of hatchery fish. Because steelhead carcasses are very rarely encountered in typical surveys (low spawner density and they

usually get washed away), estimates of hatchery/wild fish is either on the basis of visual identification of missing adipose fins on resting, live fish (not an easy task) or inference from the ratio hatchery to wild fish caught in local sport fisheries (also not very easy to get). Then there is the same problem as for the abundance data of how to expand these estimates for an entire basin. The good news is that all hatchery steelhead have been fin clipped for a long time, back to the early 1990s, so there is at least several years of data which usually can be examined. Also, the 1-year old smolt life history is usually quite rare in most wild populations, whereas nearly universal for hatchery fish. Therefore, scale reading provides another means to backup the hatchery/wild estimates – and also has allowed fairly reliable estimates of hatchery wild ratios back into the 1970s via reading of collected scale samples.

Spring Chinook = ±40% of the point estimate for proportion of hatchery fish. The advantage with both Chinook and coho is that generally it possible to recover a good number of carcasses from which to inspect for hatchery fin clips. Although, there is still the expansion problem of these data for the entire population. This can be tricky if there is a local hatchery that tends to concentrate naturally spawning hatchery fish and the sample of carcasses is not stratified in a way to weigh the basin-wide expansion on the basis of wild distribution as opposed to a simple expansion of the observed mark rate (which will be biased upward because most of the samples will probably come from the vicinity of the hatchery where the carcasses will be most abundant). However, these problems are surmountable and given the summer/fall spawn timing and more restricted distribution, hatchery/wild estimates are feasible IF the hatchery fish have been fin clipped. Unfortunately, it has only been in the last 3 or 4 years that hatchery spring chinook have been 100% fin clipped. Further, since hatchery and wild reared chinook generally spend equal periods of time in freshwater (unlike steelhead) it is much more difficult to separate the two via scale analyses. So the bad news is that until fairly recently the proportion of hatchery fish in natural spawning populations has been nearly impossible to measure.

Fall Chinook = ±70% of the point estimate for proportion of hatchery fish. For fall Chinook, most of the hatchery production is not fin marked prior to release which is a problem. Further, the freshwater residence time between wild and hatchery fish is essentially the same and therefore it is not easy separate the two on the basis of scale analysis. Typically, the frequency of hatchery fish in a natural population is made by expanding the mark rate for the hatchery release. For example, the observation of one fin-clipped fall chinook belonging to production group where only 5% of the release was marked, would expanded such that 19 of the unmarked fish also observed would be classified as hatchery fish. Such expansion obviously has serious statistical problems.

Coho = ±40% of the point estimate for proportion of hatchery fish. Basically the same positives and negatives as with spring chinook, although hatchery coho have been 100% marked for a longer time (since the late 1990s) – plus scale growth patterns are often different enough between hatchery and wild fish that the two can be discriminated on the this basis as well.

2B. Hatchery/wild proportions - typical dam count methodology

Steelhead, Spring Chinook, and Coho = ±20% of point estimate for proportion of hatchery fish. Assuming that these data come from a period when fish are fin-marked, then the primary issue here is one of expanding hatchery proportions to the full 24 hour day, in cases

where counts are not made continuously. Also, there is a possible issue of differential "fallback" rates between hatchery and wild fish that could bias the proportion estimate.

Fall Chinook = ±50% of the point estimate. Although more, if not all of the spawning population is observed – the low mark rate for hatchery fish means that there often must be a significant expansion made to estimate unmarked hatchery fish. This introduces considerable error and will be unavoidable until a mass marking of hatchery fall chinook is accomplished.

2B. Hatchery/wild proportions - trap and handle type of counting procedure.

Steelhead and Spring Chinook = ±5% of point estimate for proportion of hatchery fish. Again assuming the hatchery fish are 100% marked the primary source of error in this scenario is the fallback issue and mis-marked or unmarked hatchery fish (generally <5%).

Coho = ±20% of point estimate for proportion of hatchery fish. Generally the same situation as for steelhead and spring chinook, however a proportion of the hatchery coho production released into the upper and mid-Columbia is not fin marked. These fish appear to stray at high rates in the vicinity of their source hatcheries near Bonneville dam and further upstream into the Hood River in particular. The potential for these unmarked hatchery strays also exists for areas downstream of Bonneville as well.

Fall Chinook = ±40% of point estimate for proportion of hatchery fish. Again, the primary problem here is that so few of the hatchery fish are fin-marked prior to release. Therefore, even if each fish is handled and fin marks recorded an expansion is still necessary and this is likely a significant source of error.

3. Age composition

Steelhead and Chinook = ±40% of the proportion of any of the possible ages at maturity estimated for the population. For example, if the age composition of steelhead spawners was determined to be 0.40 4-year olds, 0.50 5-year olds and 0.10 6-year olds, then the possible range would be 0.40±40%, 0.50±40%, and 0.10±40%. The errors come primarily from the fact that age data, typically determined via scale analyses, is based upon a sub-sample of the population. (The exception to this is data for Hood River steelhead, for which scales are taken from every fish passed upstream and an age is determined). The other source of error is the scale analyses its self, although this is the lesser of the two problems.

Coho = ±5% of the proportion of any of the possible ages at maturity estimated for the population. Coho have a simplified life history, they are either 2-year olds (jacks) or 3-year olds. Size alone is often used to classify a coho return into an age category. Although it is possible to back these age determinations up with scale analyses, the size differences are usually quite large and readily observable. In fact this is usually so evident that coho are classified as being either jacks or "adults" (3-year olds) when they are counted or observed during spawning surveys. Bottom line, age composition for coho is not a big estimation problem. Although biologically it important in than essentially 100% of the jacks are males.

4. Fishery impacts and catch

Steelhead = ±40% of estimated number of fish that die as a result of fisheries (either direct mortality or post-releases mortality). The impact of sport fisheries on wild steelhead is considered low since 1992 when catch and release regulations were imposed on wild fish

(only fin-marked hatchery fish can be kept). The primary source of mortality is post-release mortality of those wild fish that are caught. The uncertainty in this post-release mortality combines with the uncertainty in how many wild fish are actually hooked and released to yield a less than accurate picture of net fishery impacts. The number of wild steelhead caught each year is typically not monitored and instead average catch rates observed prior to the implementation of catch and release regulations are typically applied to recent times. In addition, steelhead in the Columbia are subjected to commercial and tribal gillnet fisheries. These fisheries kill wild and hatchery fish at equal rates (they are not selective). It is believed the net impact of these fisheries is low, however considerable uncertainty does exist in the impact rate measurements.

Spring Chinook = ±30% of estimated number of fish that die as result of fisheries (either direct mortality or post-releases mortality). Like steelhead, sport caught spring chinook can not be kept by anglers and must be released. However, this is relatively recent regulation that has been implemented in the last few years with the advent of 100% marking of hatchery spring chinook smolts. Spring chinook are also encountered by commercial fisheries in the Columbia and the ocean. However, the impact rate of these fisheries is actively managed and monitoring is facilitated by the recovery of coded wire tagged fish in both of these fisheries.

Fall Chinook = ±40% of estimated number of fish that die as result of fisheries. Ocean fisheries probably have the greatest impact on fall chinook populations. However, the problems of estimating these rates for wild fall chinook from the lower Columbia are substantial. First, most of the fishery impact information is referenced to hatchery fish, which are marked with coded wire tags. However, given that the proportion of this marking is low, the fraction of the catch that is naturally produced and originating from the lower Columbia is nearly impossible to detect with any confidence.

Coho = ±50% of estimated number of fish that die as result of fisheries. Because of their relative rarity, the number of wild coho from the Columbia basin that are caught in sport and commercial fisheries is poorly understood. Coho impacts for hatchery reared fish are readily available, however the timing and susceptibility of wild coho relative to these hatchery stocks is still under investigation. Therefore, the confidence in estimates of fishery impacts on wild coho in the Columbia basin is low.

The ranges of errors described above are summarized in Table F-1.

Table F-1 Summary of possible range of errors in the measurement of key data elements for the assessment of lower Columbia and Willamette salmon and steelhead.

Data Element	Species	Data collection method		
		Spawning Surveys	Dam Passage Counts	Trap and Handle
Spawner	Steelhead	±70%	±20%	±5%
Abundance	Chinook	±40%	±20%	±5%
	Coho	±50%	±20%	±5%
Hatchery	Steelhead	±60%	±20%	±5%
Proportion	Spring Chinook	±40%	±20%	±5%
	Fall Chinook	±70%	±50%	±40%
	Coho	±40%	±20%	±20%
Age	Steelhead	±40%	±40%	±40%
Composition	Chinook	±40%	±40%	±40%
	Coho	±5%	±5%	±5%
Fishery Impact	Steelhead	±40%	±40%	±40%

Spring Chinook	±30%	±30%	±30%
Fall Chinook	±40%	±40%	±40%
Coho	±50%	±50%	±50%

Appendix G: Hatchery Fraction Estimation Error

Paul McElhany

For some species, such as fall chinook, a very small fraction (e.g., 5%) of hatchery spawners are tagged at the hatchery and population level hatchery fraction estimates are made based on the recovery of only a few fish. This can lead to considerable uncertainty in the estimate of the fraction of hatchery origin fish. This appendix explores the probability distribution of hatchery fraction using the current sampling schemes.

The method of estimating the probability distribution of the hatchery fraction takes a Bayesian approach. We take a two stage approach, first estimating the probability distribution for the fraction of hatchery fish in the sample, then estimating the probability distribution for the fraction of hatchery fish in the total population, based on the probability distribution of the sample.

We first calculate the probability of obtaining the observed number of tags from a hypothetical population of Y hatchery fish. This is a binomial probability where the "probability of success" is the fraction of fish of the age class that were tagged at the hatchery; the "number of trials" is Y, the hypothetical population size; and the "number of successes" is the number of observed tags. This probability is calculated for all possible hatchery fish population sizes. The possible hatchery fish population size ranges from a minimum of the number of tags observed (there is a remote chance that the tagged fish are the only hatchery fish in the population) to a maximum of all the fish in the sample.

In the language of Bayesian statistics, these binomial probabilities are "the probability of the data given the hypothesis." What we need is the Bayesian posterior probability, which is "the probability of the hypothesis given the data." That is, the binomial gives the probability of observing Z tags given Y hatchery fish and we need the probability of Y hatchery fish given that Z tags are observed. To calculate the posterior probability, we assume a uniform prior distribution between the number of observed tags and the total number of fish in the sample. The posterior probability for a particular Y is then found by dividing the probability of observing Z tags given Y hatchery fish by the sum of the probabilities of observing Z tags over all possible Ys. This produces the probability distribution for the faction of hatchery fish in the subset of the population sampled for hatchery tags.

We take a similar approach to estimating the probability distribution of the fraction of hatchery fish in the total population. We find the binomial probability that there are Y hatchery fish in the sample given H hatchery fish in the population. This is then multiplied by the probability (calculated in the previous step) that there are Y hatchery fish in the sample. The probabilities for a given H are summed across Ys to give the probability of the Y distribution given H hatchery fish. To get the posterior distribution (i.e., the probability of H hatchery fish given the distribution of Ys) we divide the probability of the Y distribution given H hatchery fish in the population by the sum of all probabilities of the Y distribution given H hatchery fish.

This approach requires the following data:

- Total population size (N)
- Number of fish in the population sampled for hatchery tags (X)

- Number of tags observed (Z)
- Fraction of fish tagged at hatchery (mark rate)

If the fraction of hatchery fish marked varies every year, we need to deal with the age structure, which gets very messy, but the same basic approach can be applied. In the case where the hatchery mark rate varies, we need the fraction of hatchery fish marked each year and the age structure (ideally of both the hatchery tagged fish and the sample as a whole, as these may differ given the small number of tags recovered).

Sample results for a "typical" Fall chinook population are shown in Figures 1-3. The total population size is 500 spawners, the number of spawners sampled for hatchery tags is 100 and the tag rate at the hatchery is 5% of releases. The different curves show the probability that the population contains a given fraction of hatchery origin spawners if 0, 1, 2, 3, 4, or 5 tagged fish are recovered in the sample of 100. The point estimate fraction of hatchery origin fish for the different number of recoveries are 0%, 20%, 40%, 60%, 80%, and 100%, respectively. This is a difference of 20% in the hatchery fraction estimate based on a recovery difference of only a single fish! Data sets often report only these point estimates. The probability curves show that there is considerable uncertainty.
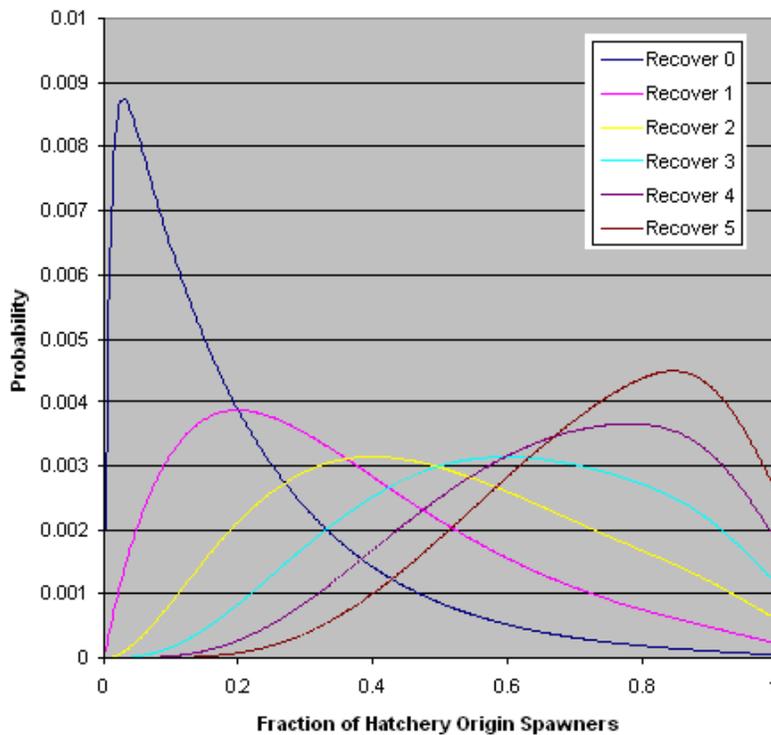
Appendix H: Population Change Criteria

Paul McElhany and John Payne

The population change criteria (PCC) approach was developed as part of the 2003 viability report as a precautionary way to set abundance and productivity targets for populations with little data. The PCC approach is more through discussed in the 2003 report, including an entire appendix discussing concerns about the approach. However, it is useful to provide a brief summary, some updated analyses and a discussion on PCC in the context of the revised criteria.

Method

At a minimum, estimating extinction risk with population viability modeling (PVA) requires estimating a population's abundance, "resilience" (or productivity) and variability, then assuming that these parameter values will continue into the future (the stationarity assumption), and finally projecting forward with these parameters to determine if the population is likely go extinct or drop below some quasi-extinction threshold QET). Estimating a population's resilience, or tendency to return to higher abundance if perturbed to low abundance, is key to understanding extinction risk.

The PCC approach uses a population's growth rate as a precautionary measure of the population's resilience. The basic assumption is that if a population is perturbed to low abundance that it is likely to the return to higher abundance at, on average, the same rate at which the population has been observed to grow in the recent past.

Although the extinction risk model used in PCC could be easily applied to estimate current extinction risk, the approach has been developed explicitly to set future population targets. To set these future targets, we ask the following question: How much will a population need to grow in a given amount of time to have a growth rate and final abundance so that it will have an acceptable extinction risk? To address this question, we need a forward projection model and some parameter estimates. The forward projection model uses a simple stochastic Hockey Stick recruitment function, where populations grow, with some variation, at a fixed average growth rate up to a carrying capacity. The parameter estimation procedure is illustrated in Figure H-1 and Table H-1.

Table H-1 PCC parameters

Parameter	Description
Variance	Variance is estimated from historical time series. To obtain a more robust estimate, we pool variance estimates for all populations of a given species in the domain.
Productivity ("Resilience")	The productivity associated with a give target is estimated as the growth rate for a population growing from its current population size to the target populations size IN A FIXED AMOUNT OF TIME.
Initial Abundance	The initial abundance for the forward projection model is set as the target abundance.
Capacity	The population ceiling is set as the target abundance
QET	The QET is set at 50 spawners/ year base on concerns about loss of genetic diversity and demographic stochasticity.
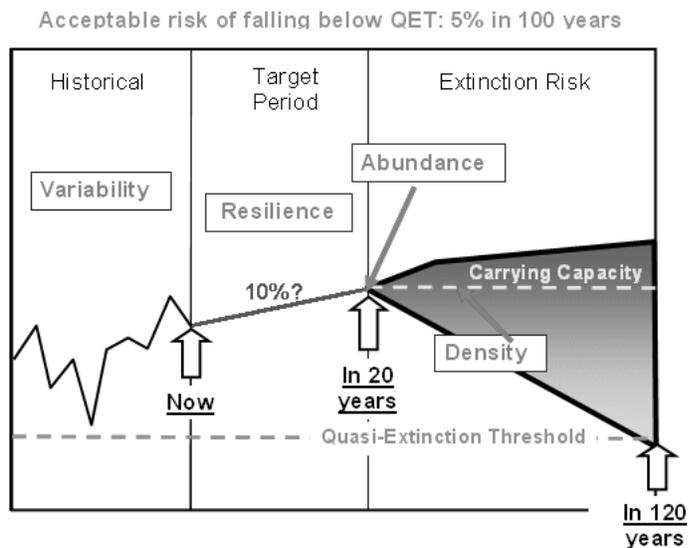
Figure H-1 PCC parameter estimation.

The target abundance/growth rate is found through a search process -by trying a bunch of different abundance values and determining which one just yields an acceptable extinction risk – not too high, not too low.

Relation to Viability Curve Approach

The viability curve criteria (VCC) approach discussed in the body of this report has the potential for addressing density dependence in a more sophisticated way. However, if a viability curve is drawn using a Hockey Stick recruitment function and if the productivity metric used to evaluate a population relative to the curve is average recruits per spawner (without consideration of density dependence), the underlying conceptual model would be basically identical to PCC. Both the interior Columbia recovery criteria and some of ODFW's current status evaluations in the WLC explore viability curve applications along these lines. There are advantages and disadvantages to expressing the same criteria approach as either PCC or VCC.

PCC and small populations

The PCC approach is generally not appropriate if a population is currently extremely small or extremely large. We developed a modified approach to address the situation of very small populations. If a population currently has less than 150 spawners we set the PCC target based on a current population of 150 spawners. This has the effect of setting a floor on the target abundance.

Big Caveats

There are many assumptions and caveats associated with the PCC approach (see the 2003 viability report.) Two of the big ones involve hatchery production and autocorrelation in marine survival. We provide ways to address these issues, but the results presented below assume no hatchery fish contributing to spawning and average marine survival rates over the observation period. In interpreting these results, it is important to understand the implication of these assumptions.

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Oregon PCC results

We have estimated pooled variance for WLC chinook, steelhead, and chum, but have not yet completed coho (Table H-2)

Table H-2

Species	Variance	Degrees of Freedom
steelhead	0.04	35
chum	0.07	19
chinook	0.09	51

Using these variance estimates and the methods described in the 2003 report produces the 20 year targets in Figure H-2. Applying this to the current abundance estimates in the 2005 BRT report yields the Oregon population targets in Figure H-3:Figure H-5. These targets are four year spawner averages in 20 years. A key thing to note is that most chinook and chum populations are current estimated to be below the 150 lower bound (or natural abundance is unknown, but assumed low). As a consequence, most population targets are the low abundance defaults. A similar result is expected to hold for coho, as only two populations currently have very many fish.
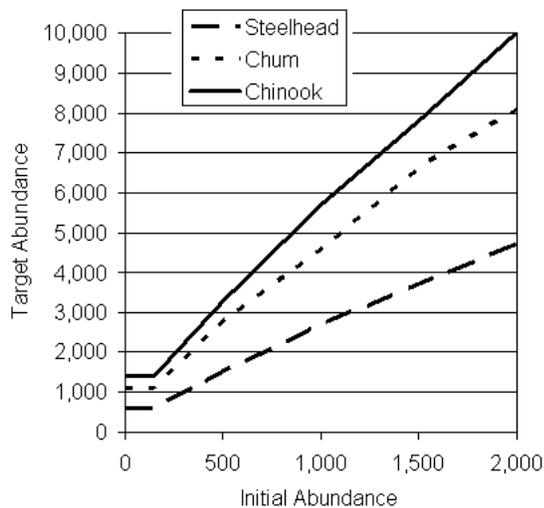
Figure H-2 PCC targets for 4 year average spawners in 20 years.
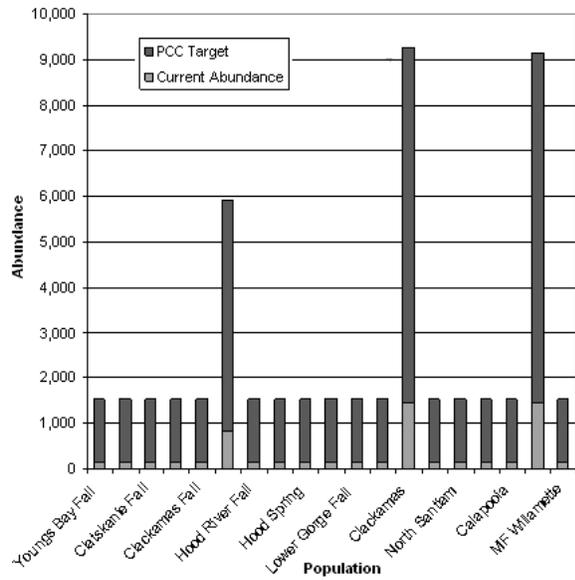
Figure H-3: Chinook
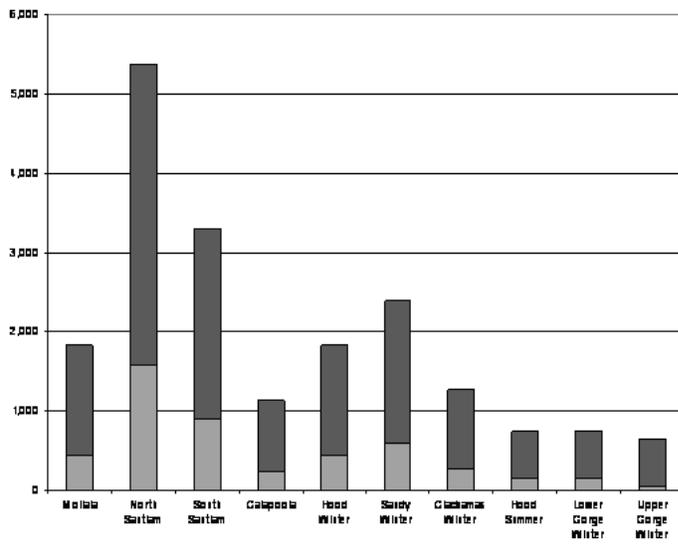
Figure H-4: Steelhead

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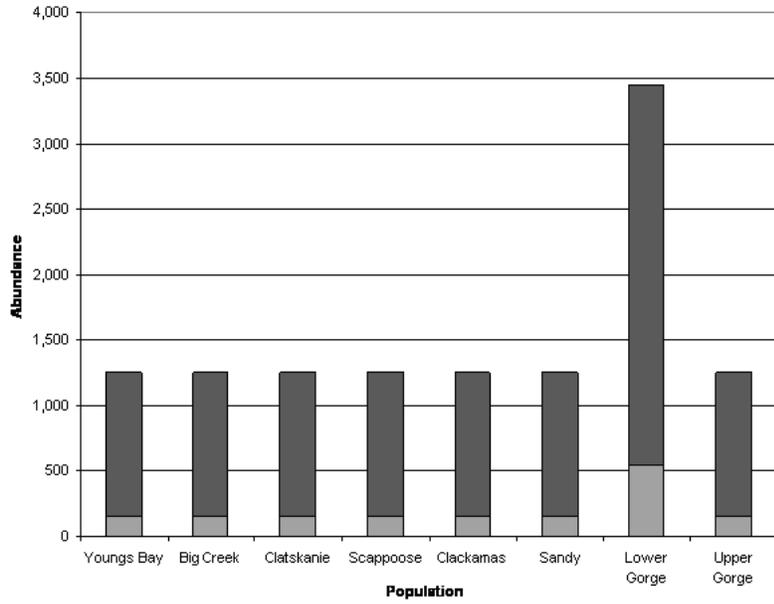
Figure H-5: Chum

Appendix I: Calculating metrics on the diversity of available habitats

Mindi B. Sheer and E. Ashley Steel

Metrics of diversity relating to habitat and occupancy were calculated for populations in the Lower Columbia and Willamette ESUs using the best available stream reach variables (stream order, elevation). Two metrics of ecological diversity were calculated, one using stream order and the other using elevation. The following equation was used to calculate diversity of both stream order and elevation:

$$\Delta ED_{elevation} = \frac{\sum_i (P_{Hi} - P_{Ci})}{n}$$

n = number of elevation (or stream order) categories

P_{Hi} = proportion of the historical potential range in elevation category i

P_{Ci} = proportion of the current potential range in elevation category i

The potential historical range was described as currently accessible streams plus historically accessible streams by population or species. For example, when calculating the number of kilometers of 1st order streams for chum in the Sandy river, the "historic occupancy proportion" (P_{Hi}) of 1st order streams (current + historic) was calculated by dividing this sum by the total number of kilometers of 1st order streams in the entire watershed (that fell within the boundaries of the chum population). The difference was calculated by subtracting the proportion of currently accessible 1st order streams (P_{Ci}) from the historic proportion. This process was repeated for elevation.

Stream Order and Elevation

Stream length within each ESU/population extent was summarized as either historically accessible or currently accessible using information from natural and manmade barriers (Steel and Sheer 2003; Sheer and Steel, in review). Variables describing aquatic habitat (stream order and stream elevation) were categorized and summarized for the historical and currently accessible areas.

Strahler's stream order for every stream reach was previously calculated using a generated hydrologic network (Steel and Sheer 2003). Stream orders $\geq 7^{th}$ order were grouped together.

Elevation was estimated for the midpoint of every stream reach within the study area (average reach length = 70 m) using a 10 m Digital Elevation Model (DEM) binned into elevation bands. We used species-specific elevation ranges for chum and all other species, so two DEMs were used to assign binned elevations to midpoints, depending on population (Table I-1). The smaller elevation bands were used for chum because of the limited elevation extent of their range.

Table I-1 Elevation categories of streams by species range.

Chum		Chinook, steelhead, coho	
Elevation (m)	*Category*	*Elevation (m)*	*Category*

0 - 60	1	0 - 150	1
60 - 120	2	150 - 300	2
120 - 180	3	300 - 450	3
180 - 240	4	450 - 600	4
240 - 300	5	600 - 750	5
300 - 360	6	750 - 900	6

Diversity Values

Diversity values were calculated for both stream order and elevation where possible, then converted to ranks for interpretation (Table I-2 and Table I-3).

Table I-2 Conversion values from above equation to a 0-4 scoring system for the diversity of available habitats.

Score	Rank
< 0.02	4
≥0.02 - 0.06	3
≥0.06 - 0.10	2
≥0.10 - 0.15	1
≥0.15	0

In some cases it was not possible to calculate differences in types of habitat due to lack of data either in the accessible area, or in the inaccessible area for the particular population. Notable areas where this occurred are populations such as the Upper Cowlitz, Tilton, Cispus, and Upper Gorge Tributaries. These are indicated with "NA" in Table I-3. For a few other cases, data for the particular feature (stream order, elevation, or both) was missing for the population; these are indicated with a "ND" in Table I-3. It may be possible to obtain data for some of these populations in future analyses. Due to time limitations, calculations for coho were done using the early 2005 population boundaries. Values for the Big White Salmon, Hood River, and Upper Gorge Tributaries will need to be updated and recalculated to represent for the two new combined coho populations ("Oregon upper gorge tributaries & Hood River", "Washington upper gorge tributaries & Big White Salmon River").

Table I-3 Diversity values for Willamette-Lower Columbia ESUs. O Metric is the diversity of available habitats metric calculated using stream order and E Metric is the diversity of available habitats metric calculated using elevation.

7. *Species*	8. *ESU*	9. *Population*	10. *Order*	11. *O Metric*	12. *Elevation*	13. *E Metric*
14. BRIGHTS	15. LCR	16. Lewis Riv	17. 0.02	18. 4	19. 0.08	20. 2
21. BRIGHTS	22. LCR	23. Sandy Riv	24. 0.03	25. 3	26. 0.05	27. 3
28. CHUM	29. LCR	30. Big Creek	31. 0.04	32. 3	33. 0.05	34. 3
35. CHUM	36. LCR	37. Clackamas	38. 0.01	39. 4	40. 0.01	41. 4
42. CHUM	43. LCR	44. Clatskani	45. NA	46. NA	47. NA	48. NA
49. CHUM	50. LCR	51. Cowlitz R	52. 0.01	53. 4	54. 0.02	55. 3
56. CHUM	57. LCR	58. Elochoman	59. 0.01	60. 4	61. 0.00	62. 4

63. CHUM	64. LCR	65. Grays & C	66. NA	67. NA	68. NA	69. NA
70. CHUM	71. LCR	72. Kalama Ri	73. 0.03	74. 3	75. 0.00	76. 4
77. CHUM	78. LCR	79. Lewis Riv	80. 0.02	81. 4	82. 0.03	83. 3
84. CHUM	85. LCR	86. LowerGorge	87. 0.00	88. 4	89. 0.00	90. 4
91. CHUM	92. LCR	93. Mill Cree	94. NA	95. NA	96. NA	97. NA
98. CHUM	99. LCR	100. Salmon Cr	101. 0.01	102. 4	103. 0.01	104. 4
105. CHUM	106. LCR	107. Sandy Riv	108. NA	109. NA	110. NA	111. NA
112. CHUM	113. LCR	114. Scappoose	115. 0.01	116. 4	117. 0.01	118. 4
119. CHUM	120. LCR	121. UpperGorge	122. 0.01	123. 4	124. 0.00	125. 4
126. CHUM	127. LCR	128. Washougal	129. 0.02	130. 3	131. 0.07	132. 2
133. CHUM	134. LCR	135. Youngs Ba	136. 0.00	137. 4	138. 0.00	139. 4
140. COHO	141. LCR	142. Big Creek	143. 0.03	144. 3	145. 0.10	146. 1
147. COHO	148. LCR	149. Big White	150. 0.28	151. 0	152. 0.31	153. 0
154. COHO	155. LCR	156. Cispus	157. NA	158. NA	159. NA	160. NA
161. COHO	162. LCR	163. Clackamas	164. 0.01	165. 4	166. 0.01	167. 4
168. COHO	169. LCR	170. Clatskani	171. 0.00	172. 4	173. 0.00	174. 4
175. COHO	176. LCR	177. Coweeman	178. 0.02	179. 3	180. 0.02	181. 3
182. COHO	183. LCR	184. EF Lewis	185. 0.03	186. 3	187. 0.01	188. 4
189. COHO	190. LCR	191. Elochoman	192. 0.01	193. 4	194. 0.05	195. 3
196. COHO	197. LCR	198. Grays & C	199. 0.00	200. 4	201. 0.00	202. 4
203. COHO	204. LCR	205. Hood Rive	206. 0.01	207. 4	208. 0.01	209. 4
210. COHO	211. LCR	212. Kalama Ri	213. 0.03	214. 3	215. 0.03	216. 3
217. COHO	218. LCR	219. Lower Cow	220. 0.01	221. 4	222. 0.02	223. 3
224. COHO	225. LCR	226. LowerGorge	227. 0.02	228. 3	229. 0.08	230. 2
231. COHO	232. LCR	233. Mill Cree	234. 0.01	235. 4	236. 0.02	237. 3
238. COHO	239. LCR	240. NF Lewis	241. 0.04	242. 3	243. 0.14	244. 1
245. COHO	246. LCR	247. NF Toutle	248. 0.02	249. 3	250. 0.05	251. 3
252. COHO	253. LCR	254. Salmon Cr	255. 0.01	256. 4	257. 0.03	258. 3
259. COHO	260. LCR	261. Sandy Riv	262. 0.03	263. 3	264. 0.05	265. 3
266. COHO	267. LCR	268. Scappoose	269. 0.01	270. 4	271. 0.05	272. 3
273. COHO	274. LCR	275. SF Toutle	276. 0.01	277. 4	278. 0.01	279. 4
280. COHO	281. LCR	282. Tilton Riv	283. NA	284. NA	285. NA	286. NA
287. COHO	288. LCR	289. Upper Cow	290. NA	291. NA	292. NA	293. NA
294. COHO	295. LCR	296. UpperGorge	297. NA	298. NA	299. NA	300. NA
301. COHO	302. LCR	303. Washougal	304. 0.06	305. 2	306. 0.12	307. 1
308. COHO	309. LCR	310. Youngs Ba	311. 0.01	312. 4	313. 0.03	314. 3
315. FACH	316. LCR	317. Big Creek	318. 0.03	319. 3	320. 0.10	321. 1
322. FACH	323. LCR	324. Big White	325. 0.14	326. 1	327. 0.23	328. 0

329. FACH	330. LCR	331. Chinook	332. 0.00	333. 4	334. 0.00	335. 4
336. FACH	337. LCR	338. Clackamas	339. 0.01	340. 4	341. 0.00	342. 4
343. FACH	344. LCR	345. Clatskani	346. 0.00	347. 4	348. 0.00	349. 4
350. FACH	351. LCR	352. Coweeman	353. ND	354. ND	355. ND	356. ND
357. FACH	358. LCR	359. Elochoman	360. ND	361. ND	362. ND	363. ND
364. FACH	365. LCR	366. Grays	367. NA	368. NA	369. NA	370. NA
371. FACH	372. LCR	373. Hood	374. 0.01	375. 4	376. 0.01	377. 4
378. FACH	379. LCR	380. Kalama	381. 0.02	382. 3	383. 0.03	384. 3
385. FACH	386. LCR	387. Lewis	388. 0.02	389. 4	390. 0.06	391. 2
392. FACH	393. LCR	394. Lower Cow	395. 0.00	396. 4	397. 0.02	398. 3
399. FACH	400. LCR	401. LowerGorge	402. 0.02	403. 3	404. 0.08	405. 2
406. FACH	407. LCR	408. Mill Cree	409. 0.01	410. 4	411. 0.02	412. 3
413. FACH	414. LCR	415. Salmon Cr	416. ND	417. ND	418. ND	419. ND
420. FACH	421. LCR	422. Sandy	423. 0.02	424. 4	425. 0.04	426. 3
427. FACH	428. LCR	429. Scappoose	430. 0.01	431. 4	432. 0.05	433. 3
434. FACH	435. LCR	436. Toutle	437. 0.01	438. 4	439. 0.02	440. 3
441. FACH	442. LCR	443. Upper Cow	444. NA	445. NA	446. NA	447. NA
448. FACH	449. LCR	450. UpperGorge	451. NA	452. NA	453. NA	454. NA
455. FACH	456. LCR	457. Washougal	458. 0.03	459. 3	460. 0.12	461. 1
462. FACH	463. LCR	464. Youngs	465. 0.00	466. 4	467. 0.03	468. 3
469. SPCH	470. LCR	471. Big White	472. 0.28	473. 0	474. 0.31	475. 0
476. SPCH	477. LCR	478. Cispus	479. NA	480. NA	481. NA	482. NA
483. SPCH	484. LCR	485. Hood	486. 0.02	487. 3	488. 0.02	489. 3
490. SPCH	491. LCR	492. Kalama	493. 0.02	494. 4	495. 0.01	496. 4
497. SPCH	498. LCR	499. NF Lewis_	500. 0.04	501. 3	502. 0.14	503. 1
504. SPCH	505. LCR	506. Sandy	507. 0.03	508. 3	509. 0.05	510. 3
511. SPCH	512. LCR	513. Tilton	514. ND	515. ND	516. ND	517. ND
518. SPCH	519. LCR	520. Toutle	521. 0.01	522. 4	523. 0.03	524. 3
525. SPCH	526. LCR	527. Upper Cow	528. NA	529. NA	530. NA	531. NA
532. SPCHW	533. Will	534. Calapooia	535. 0.04	536. 3	537. 0.01	538. 4
539. SPCHW	540. Will	541. Clackamas	542. 0.03	543. 3	544. 0.02	545. 3
546. SPCHW	547. Will	548. McKenzie_	549. 0.03	550. 3	551. 0.06	552. 2
553. SPCHW	554. Will	555. M.F. Willamett	556. 0.04	557. 3	558. 0.12	559. 1
560. SPCHW	561. Will	562. Molalla	563. 0.05	564. 3	565. 0.02	566. 3
567. SPCHW	568. Will	569. North San	570. 0.03	571. 3	572. 0.13	573. 1
574. SPCHW	575. Will	576. South San	577. 0.01	578. 4	579. 0.04	580. 3
581. SUSH	582. LCR	583. EF Lewis	584. 0.03	585. 3	586. 0.01	587. 4
588. SUSH	589. LCR	590. Hood Rive	591. 0.02	592. 3	593. 0.01	594. 4

595. SUSH	596. LCR	597. Kalama Ri	598. 0.02	599. 4	600. 0.01	601. 4
602. SUSH	603. LCR	604. NF Lewis	605. 0.04	606. 3	607. 0.15	608. 0
609. SUSH	610. LCR	611. Washougal	612. 0.05	613. 3	614. 0.10	615. 1
616. SUSH	617. LCR	618. Wind Rive	619. 0.02	620. 4	621. 0.03	622. 3
623. WISH	624. LCR	625. Cispus	626. NA	627. NA	628. NA	629. NA
630. WISH	631. LCR	632. Clackamas	633. 0.09	634. 2	635. 0.01	636. 4
637. WISH	638. LCR	639. Coweeman	640. 0.06	641. 2	642. 0.02	643. 3
644. WISH	645. LCR	646. EF Lewis	647. 0.16	648. 0	649. 0.06	650. 2
651. WISH	652. LCR	653. Hood Rive	654. 0.11	655. 1	656. 0.01	657. 4
658. WISH	659. LCR	660. Kalama Ri	661. 0.14	662. 1	663. 0.04	664. 3
665. WISH	666. LCR	667. Lower Cow	668. 0.10	669. 1	670. 0.02	671. 3
672. WISH	673. LCR	674. LowerGorge	675. 0.09	676. 2	677. 0.04	678. 3
679. WISH	680. LCR	681. NF Lewis	682. 0.06	683. 2	684. 0.15	685. 0
686. WISH	687. LCR	688. NF Toutle	689. 0.06	690. 2	691. 0.04	692. 3
693. WISH	694. LCR	695. Salmon Cr	696. 0.15	697. 0	698. 0.05	699. 3
700. WISH	701. LCR	702. Sandy Riv	703. 0.06	704. 2	705. 0.04	706. 3
707. WISH	708. LCR	709. SF Toutle	710. 0.13	711. 1	712. 0.01	713. 4
714. WISH	715. LCR	716. Tilton	717. NA	718. NA	719. NA	720. NA
721. WISH	722. LCR	723. Upper Cow	724. 0.12	725. 1	726. 0.16	727. 0
728. WISH	729. LCR	730. UpperGorge	731. NA	732. NA	733. NA	734. NA
735. WISH	736. LCR	737. Washougal	738. 0.08	739. 2	740. 0.02	741. 3
742. WISH	743. LCR	744. Wind Rive	745. 0.12	746. 1	747. 0.04	748. 3
749. WISHW	750. Will	751. Calapooia	752. ND	753. ND	754. 0.02	755. 3
756. WISHW	757. Will	758. Molalla	759. ND	760. ND	761. 0.02	762. 3
763. WISHW	764. Will	765. North San	766. ND	767. ND	768. 0.14	769. 1
770. WISHW	771. Will	772. South San	773. ND	774. ND	775. 0.04	776. 3
777. WISHW	778. Will	779. West Side	780. ND	781. ND	782. 0.03	783. 3

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