STATUS REVIEW UPDATE FOR PACIFIC SALMON AND STEELHEAD LISTED UNDER THE ENDANGERED SPECIES ACT: PACIFIC NORTHWEST

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NORTHWEST FISHERIES SCIENCE CENTER

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INTRODUCTION AND SUMMARY OF CONCLUSIONS

In the Pacific Northwest, there are currently 17 distinct population segments (DPS) or evolutionarily significant units (ESUs)[1] of Pacific salmon and steelhead listed as threatened or endangered under the Endangered Species Act (ESA) (Table 1). The ESA requires that the National Marine Fisheries Service (NMFS) review the status of listed species under its authority at least every five years and determine whether any species should be removed from the list or have its listing status changed. The most recent such review for ESA listed salmon in the Pacific Northwest occurred in 2011, and did not result in any changes in ESA listing status (Ford *et al.* 2011)[2]. NMFS is again conducting such a review in 2015/16[3].

The NMFS West Coast Region is responsible for the 5-year review process and decision-making regarding proposed changes in listing status. This report provides updated information and analyses on the biological status of the listed species, focusing on 1) information on ESU boundaries, and 2) trends and status in abundance, productivity, spatial structure and diversity. Where possible, this review also summarizes current information with respect to recovery goals identified in recovery plans or Technical Recovery Team viability documents.

In two of the three formal status reviews that supported the current listings (Good *et al.* 2005; Hard *et al.* 2007) the Biological Review Team (BRT) categorized each ESU as either "in danger of extinction", "likely to become endangered" or "not likely to become endangered", based on the ESU's abundance, productivity, spatial structure and diversity. In the third status review (Oregon Coast coho salmon; (Stout *et al.* 2012), the three categories were instead referred to as "high", "moderate" and "low" risk, and included narrative and probability of extinction definitions for the "high" and "moderate" risk categories (see p. 114 of Stout *et al*. 2012). In this report, for each listed ESU, we summarize whether there is new biological information to indicate that an ESU is likely to have moved from one of the three biological risk categories to another. In addition, we also note whether each ESU appears to be stable, improving, or declining in risk status, whether or not such changes warrant a change in category (Table 1). The information in the report will be incorporated into the Region's review, and the Region will make final determinations about any proposed changes in listing status, taking into account not only biological information but also ongoing or planned protective efforts and recovery actions.

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[1] For Pacific salmon, NMFS uses its 1991 ESU policy, that states that a population or group of populations will be considered a Distinct Population Segment if it is an Evolutionarily Significant Unit. The species *O. mykiss* is under the joint jurisdiction of the NMFS and the Fish and Wildlife Service, so in making its listing determinations NMFS used the 1996 Joint FWS-NMFS DPS policy for this species. Throughout this document ESU and DPS are used interchangeably.
[2] http://www.westcoast.fisheries.noaa.gov/publications/frn/2011/76fr50448.pdf
[3] http://www.westcoast.fisheries.noaa.gov/publications/frn/2015/80fr6695.pdf

Table 1 – Summary of current ESA listing status, recent trends and summary of conclusions

Species	ESU/DPS	2010 risk category[1]	ESA listing status	Recent risk trend[2]	Change in risk category[1]?
Chinook	Upper Columbia spring	In danger of extinction	Endangered	Stable	No
	Snake River spring/summer	Likely to become endangered	Threatened	Stable	No
	Snake River fall	Likely to become endangered	Threatened	Improving	No
	Upper Willamette spring	Likely to become endangered	Threatened	Declining	No
	Lower Columbia	Likely to become endangered	Threatened	Stable/Improving	No
	Puget Sound	Likely to become endangered	Threatened	Stable/Declining	No
Coho	Lower Columbia	In danger of extinction	Threatened	Stable/Improving	No
	Oregon Coast	Moderate risk	Threatened	Improving	Possibly
Sockeye	Snake River	In danger of extinction	Endangered	Improving	No
	Lake Ozette	Likely to become endangered	Threatened	Stable	No
Chum	Hood Canal summer	Likely to become endangered	Threatened	Improving	No
	Columbia River	Likely to become endangered	Threatened	Stable	No
Steelhead	Upper Columbia	In danger of extinction	Threatened	Improving	No
	Snake River	Likely to become endangered	Threatened	Stable/Improving	No
	Middle Columbia	Likely to become endangered	Threatened	Stable/Improving	No
	Upper Willamette	Likely to become endangered	Threatened	Declining	No
	Lower Columbia	Likely to become endangered	Threatened	Stable	No
	Puget Sound	Likely to become endangered	Threatened	Stable	No

[1]Risk category reflects the assessment of ESU/DPS viability summarized in the prior status review (Ford *et al.* 2011). These risk categories do not include an evaluation of the ESA Sec. 4(a)(1) listing factors, and thus do not represent a conclusion regarding ESA listing status.

[2]Recent risk trend summarizes the overall trends in risk status for each ESU/DPS since the prior status review, in the judgement of the chapter author considering all four VSP criteria (abundance, productivity, spatial structure and diversity).

METHODS

This report includes both a set of common analyses conducted for each ESU as well as in some cases ESU-specific analyses developed by the individual technical recovery teams (TRTs). Here, we describe only the common set of analyses; see the individual sections for a description of the analyses that pertain to specific ESUs.

Spawning abundance and trends – All of the Pacific Northwest TRTs spent considerable time and effort developing spawning abundance data for the populations they identified within ESUs. In almost all cases these estimates are derived from state, tribal or federal monitoring programs. The raw information upon which the spawning abundance estimates were developed consists of numerous types of data including redd counts, dam counts, carcass surveys, information on pre-spawning mortality, and distribution within populations, which the TRTs used to develop estimates of natural origin spawning abundance. It is important to recognize that spawning abundance estimates and related information such as the fraction of spawners that are natural origin are not in most cases 'facts' that are known with certainty. Rather, they typically are estimates based on a variety of sources of information, some known with greater precision or accuracy than others. Ideally, these estimates would be characterized by a good understanding of the degree of variation due to measurement error. However, for the most part such a statistical characterization is either not possible or has not been attempted. The spawning time series summarized here and references to the methods and sources for their development are available from the Northwest Fisheries Science Center's Salmon Population Summary database[4] and are also discussed in the ESU-specific chapters.

COMMON METRICS

Multivariate dynamic linear modeling (DLM) was used to estimate population-specific mean trends in each ESU from the log of total spawner counts. The result is an estimate of the mean or smoothed total spawner counts, from which summary statistics regarding trends were computed. We focus exclusively on fish spawning in nature, but often these naturally spawning populations include some numbers of hatchery-origin fish, either as part of a deliberate supplementation effort or due to straying from hatchery populations. For the rest of this report, a "natural-origin" or "wild" fish refers to a fish whose parents spawned naturally, and a "hatchery-origin" fish refers to a fish whose parents were spawned in a hatchery, regardless of prior generation origin.

In order to estimate the trend of natural-origin spawners in populations that also include hatchery-origin spawners, a univariate DLM was applied to the logit of the fraction natural-origin estimate to produce a smoothed proportion natural-origin time series. This was used to produce an estimate of the mean natural-origin spawners for years when fraction natural-origin estimates were unavailable.

The mean or smoothed total spawner count is similar (in concept) to a 3- or 5-year geometric mean; the goal is the same—to produce an estimate that smooths over single year variation. Such variation arises from observation error in the spawning counts and also from peaks and troughs in spawners

[4] https://www.webapps.nwfsc.noaa.gov/apex/f?p=238:home:0.

numbers due to the life-history of salmonids or environmental variation. The multivariate DLM approach has a number of advantages. Most importantly it is a statistical model for which maximum-likelihood diagnostics, model selection criteria, and confidence intervals are available. It is a time-series model, which addresses temporal autocorrelation in the data. It deals with missing data and provides an estimate for the missing year with appropriately wider confidence intervals. And lastly, it allows us to use information across all populations within an ESU to estimate the level of year-to-year variation in the mean spawner count—the process variance—and allows us to estimate the year-to-year covariance, which is often high, across populations within an ESU. The latter improves estimation of missing values because populations with data in one year help inform the values for populations with missing data that year.

DYNAMIC LINEAR MODELING FOR TIME-VARYING TREND ESTIMATION

Dynamic linear models (DLMs) are similar to linear regression models with a yearly trend. Like a classic trend analysis using linear regression, the goal is to estimate the mean spawner count at *x*, where *x* is year (time). Linear regression models, however, use a time-constant yearly trend (which appears as the regression line versus time) while DLMs allow the trend to be time-varying.

In mathematical terms this means that the classic linear regression of log spawners (y) against year treats the trend (β) or yearly growth in the mean spawner count as a constant and fits the following model:

$$\begin{aligned} \bar{y}_t &= \bar{y}_{t-1} + \beta \\ y_t &= \bar{y}_t + v_t \end{aligned}$$

where y_t are the observations, $\bar{y}_t$ is the mean of y_t and v_t are normal-distributed errors. The mean spawner count in year t is the mean spawner count in year $t - 1$ plus the constant trend value β. Normally, we write this model in classic linear regression form as

$$y_t = \alpha + \beta t + v_t$$

with the mean of y_t equal to $\alpha + \beta t$. A DLM, in contrast, allows us to fit a model with a time-varying β. Specifically, the following model

$$\begin{aligned} \bar{y}_t = \bar{y}_{t-1} + \beta_t &= \bar{y}_{t-1} + u + w_t \\ y_t &= \bar{y}_t + v_t \end{aligned}$$

The time-varying β is modeled as $u + w_t$, where w_t is a normally distributed random variable.

Figure 1 shows example spawner data where a time-varying sinusoidal β (yearly growth rate) was used to generate counts (the circles) using the DLM model above. The black line in the top panel of Figure 1 shows the true mean *y*. The red line shows the estimate from a linear regression of *y* against year with a non-time-varying β. The blue line shows the estimate from a DLM where the β is allowed to vary in time. The bottom panel shows the estimate of β compared to the true sinusoidal β that generated the data. This illustrates the power of DLM when the objective is to estimate a time-varying trend.

MULTIVARIATE DLMS FOR ANALYSIS OF MULTIPLE TIME SERIES FROM ONE ESU

A multivariate DLM allows one to estimate time-varying trends using a multiple observed time series, in our case populations within ESU, where parameter sharing is allowed across the time series. Specifically, one can constrain the variances to be the same across time series and to allow covariance across time series. The latter allows information from time series with data in year *t* to help inform the estimate of mean *y* for time series that have no data in year *t*.

Mathematically, the model being fit is

$$\begin{bmatrix}\bar{y}_1\\ \bar{y}_2\\ \bar{y}_3\end{bmatrix}_t = \begin{bmatrix}\bar{y}_1\\ \bar{y}_2\\ \bar{y}_3\end{bmatrix}_{t-1} + \begin{bmatrix}u_1\\ u_2\\ u_3\end{bmatrix} + \begin{bmatrix}w_1\\ w_2\\ w_3\end{bmatrix}_t$$

$$\begin{bmatrix}y_1\\ y_2\\ y_3\end{bmatrix}_t = \begin{bmatrix}\bar{y}_1\\ \bar{y}_2\\ \bar{y}_3\end{bmatrix}_t + \begin{bmatrix}v_1\\ v_2\\ v_3\end{bmatrix}_t$$

The u_j are the long-term mean of $\beta_{j,t}$. The trend at year *t* is $\beta_{j,t} = u_j + w_{j,t}$. The w_t and v_t are error terms drawn from a multivariate normal distribution with variance-covariance matrix **Q** and **R** respectively. The structure of **Q** and **R** allows one to specify different types of parameter constraints (for example equal variances across populations).

MODEL SELECTION

Model selection was used to select the structure of **Q** and **R**. The following structures were explored for **Q**: diagonal with unequal variances (no covariance across populations in terms of good and bad years and populations allowed to have different year-to-year variability), diagonal with equal variances (no covariance across populations and populations constrained to have the same year-to-year variability), one variance and one covariance across all populations, equal variances and covariances across similar run timings in a population, and unconstrained (unique variances and covariances across all populations). For **R** the following structures were explored: diagonal with unequal variances (no covariance) and diagonal with equal variances. The **R** represents the residual non-time-dependent error and was assumed not to covary across populations (**Q** and **R** cannot both have covariance terms in the DLM due to identifiability constraints). Across the majority of ESUs, model selection gave the most data support (quantified with AICc) to a **Q** with one variance and one covariance across all populations in an ESU and an **R**, the residual variance-covariance matrix, with one variance across populations. Because **Q** has covariance terms, estimates of mean spawner numbers can be provided for populations with missing data because the data from other populations helps inform the estimates (Figure 2 shows an example).

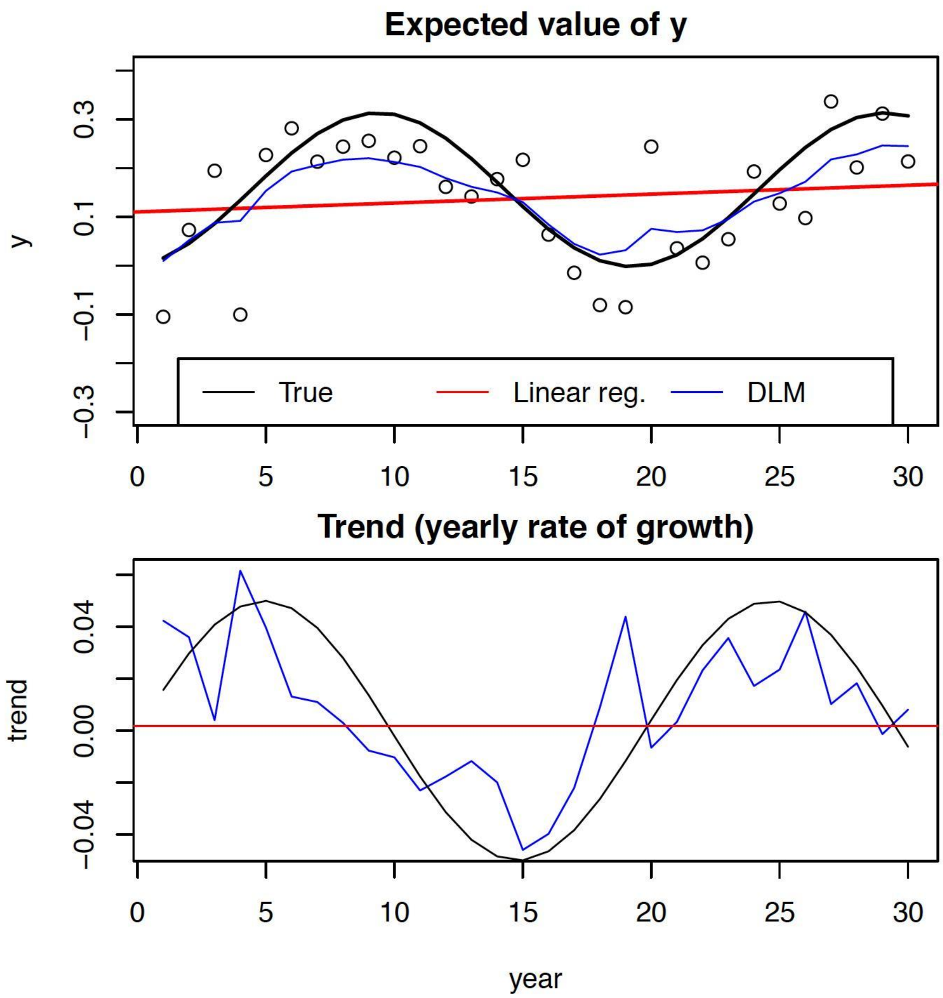


Figure 1 -- This figure compares a trend analysis using a non-time-varying trend (red line) via linear regression versus a trend analysis using a time-varying trend (blue line). The black line is the true line we are trying to estimate (with the red or blue line) and the dots in the top panel are the observations of the black line.

CODE TO FIT A MULTIVARIATE DLM

The MARSS R package was used to fit multivariate DLMs to the log-spawner counts (or indices in some cases). The package handles missing data entered as NAs for missing years. The following example code fits 2 time-series via a multivariate DLM using the MARSS R package:

```
library(MARSS)

logspawners = log(matrix(c(

  1106, 1503, 853, 566, 251, 424, 783, 639, 566, 413, 1035, 890,

  7348, 6880, 2699, 1096, NA, NA, NA, 1318, 1127, 472, 637, 869

  ), 2,12, byrow=TRUE))

model=list(

  Q="equalvarcov",

  R="diagonal and equal",

  U="unequal")

fit=MARSS(logspawners, model=model)
```

NATURAL-ORIGIN SPAWNER ESTIMATES

For some populations, there were estimates of the fraction of total natural spawners that were of natural-origin. However, for many populations, these data were noisy and had many missing years. In addition, the number of years with fraction natural-origin information was often shorter than the years with total spawner counts. To estimate a mean natural-origin spawner estimate, similar to the mean total spawner estimate, the mean total spawner estimate was multiplied by a smoothed estimate of the fraction natural-origin. The smoothed estimate was produced by fitting a univariate DLM to the logit $z_t = \log(f/(1-f))$ of the fraction natural-origin estimates with a time-varying β. Specifically, the following model was fit:

$$\begin{aligned} \bar{z}_t &= \bar{z}_{t-1} + \beta_z + w_t \\ z_t &= \bar{z}_t + v_t \end{aligned}$$

The mean natural-origin spawner estimate at time t was then $\bar{y}_t \exp(\bar{z}_t)/(\exp(\bar{z}_t)+1)$. Each time series of fraction natural-origin from each population was fit independently (no covariance assumed across populations).

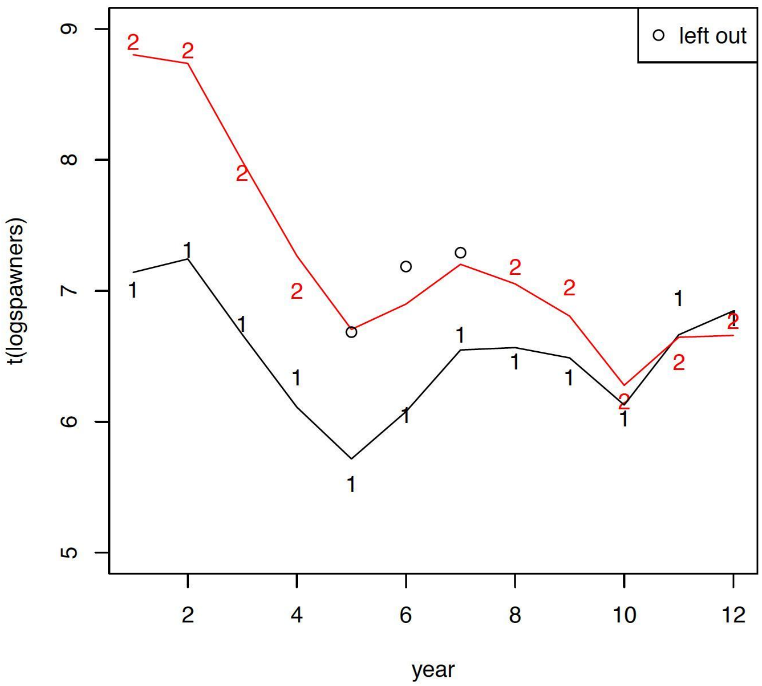


Figure 2 -- The estimated mean log (spawners) using a multivariate DLM. Notice that the information from the years when data are available for time-series 1 are used to inform the estimate for time-series 2 for the missing years (marked with a circle).

SUMMARY STATISTICS

The following summary statistics were reported for the mean total spawner estimates, the mean natural origin spawner estimates, and the raw total and natural origin spawner estimates. These are similar to statistics reported in prior status reviews.

15-year trends. A linear regression was fit to 15 years of the mean natural origin spawner estimate and the slope (trend) reported.

5-year geometric means. 5-year geometric means were computed from the raw total and natural origin spawner estimates, which may have missing values. When there were missing values, the geometric mean was computed only from the non-missing values. For example, if 3 values were available, $(y_1 y_2 y_3)^{(1/3)}$ was reported.

Average fraction natural origin. These were computed from the raw estimates of fraction natural-origin.

Productivity metric. Because age of return data were not consistently available across all ESUs and populations, a generic productivity metric was computed as the mean natural-origin spawner estimate at year *t* divided by the mean total spawner estimate at year *t* – 3 for coho salmon and *t* – 4 for all other species.

Harvest. We compiled data on trends in the adult equivalent exploitation rate for each ESU. It is important to note that magnitude and trend of an exploitation rate cannot be interpreted uncritically as a trend in level of risk from harvest. Analyses relating exploitation rate to extinction risk or recovery probability have been conducted in a quantitative way for several ESUs (e.g., NMFS 2001; Ford *et al.* 2007; NWFSC 2010) and qualitatively for others (NMFS 2004). See specific sections for details.

ESU BOUNDARIES

The ESA allows listing of species, subspecies and distinct population segments (DPS) of vertebrates. The ESA as amended in 1978, however, provides no specific guidance for determining what constitutes a DPS. Waples (1991) developed the concept of an Evolutionarily Significant Unit (ESU) for identifying DPS for Pacific salmon. This concept was adopted by NMFS in applying the ESA to anadromous salmonid species (NMFS 1991). The NMFS policy stipulates that a salmon population or group of populations is considered a DPS if it represents an ESU of the biological species. An ESU is defined as a population or group of populations that 1) is substantially reproductively isolated from conspecific populations, and 2) represents an important component in the evolutionary legacy of the species.

In 2006 NMFS changed its practice of applying the ESU policy to steelhead populations, and instead applied the joint USFWS-NMFS DPS definition in determining species of steelhead for listing consideration (71 FR 834, 5 January 2006). This change was initiated because steelhead are jointly administered with USFWS, and USFWS does not use the ESU policy in its listing decisions (71 FR 834, 5 January 2006). Under the joint USFWS and NMFS DPS policy, a group of organisms is a DPS if it is both "discrete" and "significant" from other such populations. Evidence of discreteness can include being ''markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, and behavioral factors," and evidence of significance includes persistence in an unusual or unique ecological setting, evidence that a group's extinction would result in a significant gap in the range of the taxon, or markedly different genetic characteristics from other populations (see DPS Policy; 61 FR 4722 for details). NMFS has concluded that under the DPS policy, resident and anadromous forms of steelhead are discrete (and hence are different DPS), whereas biological review teams have generally concluded that resident and anadromous steelhead within a common stream are part of the same ESU if there is no physical barrier to interbreeding (see Good *et al.* 2005 for an extensive discussion of this issue).

Information that can be useful in determining the degree of reproductive isolation includes incidence of straying, rates of recolonization, degree of genetic differentiation, and the existence of barriers to migration. Insight into evolutionary significance or discreteness can be provided by data on genetic and life history characteristics, habitat differences, and the effects of stocks transfers or supplementation efforts on historical patterns of diversity.

Life history characteristics that have been useful in establishing ESU or DPS boundaries include juvenile emigration and adult return timing, age structure, ocean migration patterns, and body size and morphology, and reproductive traits (i.e., egg size). Population genetic structure can be very informative for estimating the degree of reproductive isolation among populations. Similarly, mark/recapture studies provide information on the level of inter-population migration, although straying does not necessarily always result in successful genetic introgression if stray fish do not breed or breed as effectively as fish from the local population.

Habitat and ecological information has been extensively used to establish ESU and DPS boundaries, especially where there is little population specific information available. Given the high level of homing fidelity exhibited by salmonids and the associated degree of local adaptation in life history traits, habitat characteristics become a useful proxy for potential differences in life history traits. Similarly, biogeographic boundaries and the distribution and ESU structure of similar species have been used where information on the species in question is lacking.

In initially defining the structure of ESUs and DPSs, the BRTs analyzed a variety of different data types of varying quality. At the time, the BRTs recognized that ESU boundaries would not necessarily be discrete, but rather in some cases a transitional zone covering one or more basins might exist at the interface between ESUs. In some cases, especially where there was not an obvious geographic feature to rely on and in the absence of biological or genetic data, there was some degree of uncertainty in the identification of ESU boundaries. Population-specific information was frequently limited and in some cases natural populations in the apparent transitional zone had been extirpated or modified by the transfer of fish between basins. Ultimately, the BRTs have used the best available information to assign transitional populations into ESUs with the understanding that if additional information became available the decisions regarding the boundaries could be revisited.

The majority of the ESUs and DPSs for Pacific salmon and steelhead were initially defined in the late 1990s as part of the coast-wide status review process undertaken by the NMFS. In the intervening 15 years, the most marked change in population monitoring has arguably been in the analysis of additional genetic variation. The majority of the genetics information available to the original BRTs in the 1990s was developed using starch-gel electrophoresis of allozymes. The utilization of DNA microsatellite and single-nucleotide polymorphisms (SNPs) technology in fisheries during the last 20 years has provided a wealth of additional genetic information. Overall, these techniques have provided a finer level of discrimination than was possible with allozymes. Furthermore, since the initial listings there have been extensive monitoring efforts throughout the West Coast, many of which include genetic analysis. Thus the quality and quantity of genetic information available to address the issue of ESU and DPS delineation has improved considerably since the time of the original ESA listings.

For a number of populations, monitoring efforts over the last 20 years have also expanded the existing databases on abundance, spawn timing, and migratory patterns, and this information has also been informative for understanding population structure. Additionally, the mass marking of hatchery-origin juveniles has improved the quality of the data collected, especially regarding the life history data of naturally-produced fish.

Ford *et al*. (2011) summarized information potentially justifying reconsideration of boundaries for Puget Sound and Washington Coast ESUs of coho salmon, Lower Columbia River Chinook Salmon and

Middle Columbia River Chinook Salmon Spring Run ESUs, and Lower Columbia River and Middle Columbia River steelhead DPSs.

This review considers new information regarding the boundary between the Lower Columbia River steelhead DPS and the Upper Willamette River Steelhead DPS. Specifically, we review new information that may help clarify the placement of the native winter run steelhead in the Clackamas River, a tributary to the lower Willamette River (Figure 3). This new information includes a genetic analyses based on DNA data (microsatellites and single-nucleotide polymorphisms; SNPs) whereas the original status reviews (Busby *et al.* 1996; Myers *et al.* 2006) examined protein-based allozyme data. More importantly, the recent DNA studies also include samples representing more steelhead populations, including the Clackamas River winter run, which was not well represented in the earlier allozyme datasets. Currently, the native steelhead in the Willamette River below Willamette Falls are included in the lower Columbia River DPS (Busby *et al*. 1996). These include winter run steelhead in the Clackamas River basin (whose confluence is just below the falls), that are considered a demographically independent population (DIP) within the Lower Columbia River DPS (Myers *et al.* 2006).

A number of other boundary issues have been raised, primarily regarding the extension of DPS/ESU boundaries beyond the estimated historical range. These include: colonization by Lower Columbia River coho salmon ESU fish into the Upper Willamette Basin and upstream of the Dalles Dam, or colonization by late-winter steelhead upstream of the Calapooia River in the Upper Willamette River. In these cases there is little doubt regarding the origin of the fish; however, the classification of these fish and their spawning habitat is regarded as a policy question rather than a biological one and is not considered here.

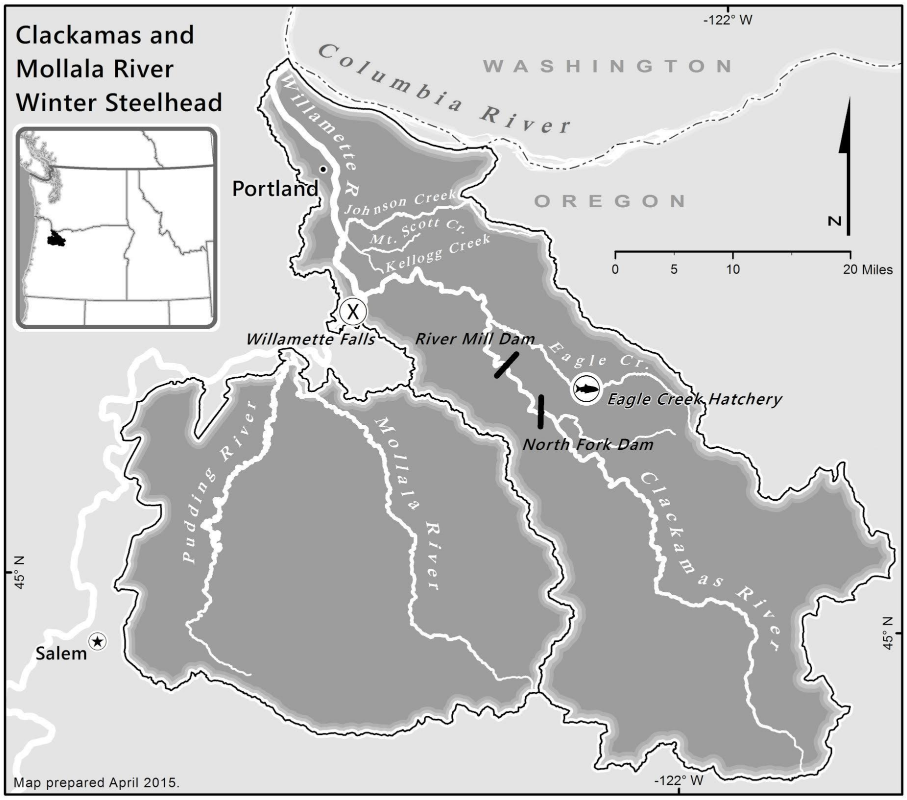


Figure 3 – Map of the Lower Willamette River

INFORMATION RELATED TO THE ORIGINAL DELINEATION OF THE LOWER COLUMBIA RIVER AND THE UPPER WILLAMETTE RIVER STEELHEAD DPS

The first coast-wide steelhead BRT (Busby *et al*. 1996) reviewed biological and geographic information on steelhead populations to identify DPSs (then ESUs) in Washington, Idaho, Oregon and California. Busby *et al*. (1996) reviewed previous genetic studies (primarily based on allozymes) and also compiled and analyzed a data set consisting of 42 allozyme loci in 108 steelhead population samples ranging from California to northern Washington. The Busby *et al*. (1996) analysis confirmed earlier findings (Allendorf 1975; Utter & Allendorf 1977; Okazaki 1984; Schreck *et al.* 1986; Reisenbichler *et al.* 1992) that the region's steelhead populations consist of distinct coastal and inland genetic lineages. In the Columbia River, the inland and coastal genetic lineages are separated near the Cascade Crest. Busby *et al*. (1996) identified the Middle Columbia River, Upper Columbia River, and Snake River DPSs within the inland lineage and the Lower Columbia River and Upper Willamette River DPSs within the coastal genetic lineage. Both winter and summer run steelhead populations are native to the Lower Columbia River and included in that DPS, whereas in the Upper

Willamette River only late winter run steelhead were considered native and included in the Upper Willamette River DPS.

Two of the steelhead allozyme studies reviewed by Busby *et al*. (1996) included population data from the Lower Columbia and Willamette Rivers. Both Schreck *et al*. (1986) and Reisenbichler *et al*. (1992) used cluster analyses to depict population groupings and found that steelhead in the Upper Willamette River, above Willamette Falls formed a genetic group distinct from Lower Columbia River populations. The study by Schreck *et al*. (1986) included samples of winter run steelhead from Eagle Creek National Fish Hatchery, which is located on Eagle Creek, a tributary to the Clackamas River. Eagle Creek NFH has propagated Big Creek stock from the Lower Columbia River and Clackamas River stock, however the stock origins of the majority of steelhead hatchery releases in Eagle Creek prior to 1989 are unknown (Myers *et al*. 2006). Schreck *et al*. (1986) analyzed data from two Eagle Creek NFH stocks, identified in their report as Big Creek and Native. In their allozyme analysis, both the Big Creek and Native samples from Eagle Creek NFH were genetically most similar to the Lower Columbia River populations, forming a separate sub-group different from Upper Willamette River late-winter run samples.

Subsequent to the Busby et al. (1996) review, genetic relationships among steelhead populations in the Willamette River and lower Columbia River basins were examined as part of a study of historical population structure of the region's salmon and steelhead (Myers et al. 2006). Myers et al. (2006) analyzed a Washington Department of Fish and Wildlife (WDFW) allozyme dataset focusing on the Lower Columbia River and computed genetic distances between each pair of populations. Within the Lower Columbia River DPS, the distance values and a dendrogram based on them revealed little genetic differentiation that aligned with geographic relationships. The dataset included a single sample of Clackamas River winter run steelhead, which clustered separately from other lower river populations. Myers et al. (1996) also analyzed a NWFSC allozyme data set, which included population data from both the Upper Willamette River and Lower Columbia River DPSs. The NWFSC data did not include samples from Willamette River steelhead below Willamette Falls. In a dendrogram based on the NWFSC allozyme data all of the native winter run populations from the Upper Willamette River clustered separately from Lower Columbia River summer and winter run steelhead, a finding consistent with previous studies indicating genetic differentiation between the two DPSs.

In addition to genetic differences, the previous reviews examined differences in run and spawn timing between winter run steelhead in the Lower Columbia River and Upper Willamette River. Although adult timing patterns differ among lower Columbia River populations, winter runs are considered "early", primarily entering freshwater beginning in October with peak spawning occurring in winter (Howell *et al.* 1985). Native winter run populations in the upper Willamette River are considered "late winter" and enter freshwater beginning in February with peak spawning occurring in the spring. Steelhead in the Clackamas River are late winter run type with peak spawning in May and June (Oregon Department of Fish and Wildlife (ODFW) 1990; Murtagh *et al.* 1992). Stone (1878) also noted that steelhead in the Clackamas River, especially those in the upper basin, peak in May but may spawn into the late spring and early summer.

In the original BRT status review (Busby *et al*. 1996), the boundary between the Lower Columbia River DPS and Upper Willamette River DPS was identified as Willamette Falls on the Willamette

River (rkm 43). The location of the DPS boundary was based on two factors. First, the allozyme data of Schreck *et al*. (1986) showed a genetic affinity of steelhead in the Willamette River below the falls with populations in the Lower Columbia River. Second, under historic flow conditions Willamette Falls was only passable during high river flows in winter and spring and therefore may have been an isolating mechanism for Upper Willamette River steelhead. The seasonal flow patterns permitted the basin's winter run steelhead to ascend the falls and access upriver spawning areas beginning in late March or April (Dimick & Merryfield 1945). However, the falls provided a migration barrier to adults returning in other seasons, including summer steelhead. Willamette Falls was also a historic barrier to adult coho and fall Chinook salmon and was identified as an ESU boundary for spring Chinook salmon (Myers *et al*. 1998).

NEW GENETIC INFORMATION ON LOWER COLUMBIA RIVER AND UPPER WILLAMETTE RIVER STEELHEAD

Recent steelhead DNA studies have provided new information on population genetic structure in the Columbia River Basin. Blankenship *et al*. (2011) surveyed genetic variation at 13 microsatellite DNA loci in 226 sample collections from throughout the Columbia and Snake rivers. In their analyses, inland and coastal genetic lineages were distinct and within the coastal lineage population aggregates were generally concordant with DPS configurations. Native late winter run populations in the Upper Willamette River clustered separately from Lower Columbia River winter and summer run populations. Samples of introduced (from the Lower Columbia River) summer and early winter steelhead in the Upper Willamette River were assigned as part of the Lower Columbia River genetic aggregate. Blankenship *et al*. (2011) included samples from Eagle Creek NFH and also from Eagle Creek natural origin steelhead. In addition, they analyzed genetic samples from winter run steelhead collected at the Clackamas River North Fork Dam and also a Clackamas River sample of unknown run time and origin (hatchery or wild). Similar to the earlier allozyme study of Schreck *et al*. (1986), the Eagle Creek NFH sample clustered with Lower Columbia River populations in the microsatellite DNA analysis. However, the natural-origin Eagle Creek samples and both Clackamas River samples were more similar to the Upper Willamette River late winter run genetic aggregate.

Matala *et al*. (2014) conducted the first geographically broad examination of steelhead genetic population structure in the Columbia and Snake rivers using SNPs. They used a set of 158 putatively neutral SNPs (i.e., SNPs not under natural selection) to analyze genetic relationships among populations in the coastal steelhead lineage. Their study included nine population samples in the Lower Columbia River, four in the Clackamas River, and six from the Upper Willamette River. The Clackamas River samples included both winter run and introduced Skamania summer run stock. The Upper Willamette River samples were from native winter run populations in eastside tributaries and winter run steelhead from presumptive introduced populations in westside tributaries. Matala et al. (2014) found that population relationships depicted using the 158 SNPs aligned with DPS designations with native winter run steelhead, with eastside Upper Willamette River tributaries forming a group distinct from Lower Columbia River populations. In their analysis, populations in Upper Willamette River westside tributaries clustered with Lower Columbia River samples, providing further support for the hypothesis that these populations originated from introduction of Lower Columbia River fish. Similar to the microsatellite results of Blankenship *et al*. (2011), Matala *et al*. (2014) found that the Clackamas River and naturally spawning Eagle Creek winter run samples clustered with Upper Willamette River native winter run steelhead and not with the Lower Columbia River samples.

Recently, Van Doornik *et al*. (2015) studied steelhead population genetic structure in the Willamette River. The study employed 15 microsatellite DNA loci and included several new samples of both the river's native and introduced populations. Samples from earlier studies were also examined in the study, including the Clackamas River samples analyzed by Blankenship *et al*. (2011). Van Doornik *et al*. (2015) identified three major Willamette River population groups consisting of 1) introduced summer run populations, 2) introduced early winter run and western tributary populations and 3) native late winter run populations in eastern tributaries. A sample of the Eagle Creek NFH early winter run population was included in the second early winter group, while samples of Clackamas River and Eagle Creek naturally produced steelhead were included in the third, native late winter run genetic group. Van Doornik *et al*. (2015) noted that their data also suggested some introgression by the introduced early winter run into the wild Clackamas River late winter populations.

For the current review, 13 microsatellite DNA loci were compiled from the Blankenship *et al*. (2011) and Van Doornik *et al*. (2015) studies to further examine whether Clackamas River late winter run steelhead align with populations in the Lower Columbia River or the native Willamette River genetic population group. Data were from 15 populations in the Lower Columbia River, three Clackamas River late winter populations, and six native populations from the Willamette River from above Willamette Falls (Table 2). For some locations, samples taken from multiple years were pooled.

Table 2 – Collection information for samples used to analyze genetic relationships among steelhead samples in the Lower Columbia River and Willamette River. Included were samples of natural spawning populations and the Cowlitz Hatchery late winter run population, which is part of the Lower Columbia River DPS. DPS abbreviations are LCR, Lower Columbia River and UWR, Upper Willamette River. Microsatellite DNA data for Lower Columbia River populations are from Blankenship *et al.* (2011). Microsatellite DNA data for Clackamas and Willamette River populations are from Van Doornik *et al.* (2015).

Sampling Location	DPS	Run type	Collection year	Sample Size
Lower Columbia River				
Cowlitz River Hatchery	LCR	Late Winter	2008	96
Cowlitz River, Barrier Dam	LCR	Winter	2005	143
Cowlitz River tributaries	LCR	Winter	2008-2009	59
Coweeman River	LCR	Winter	2006	138
Green River	LCR	Winter	2006	97
North Fork Toutle River	LCR	Winter	2005	99
South Fork Toutle River	LCR	Winter	2005-2007	73
Kalama River	LCR	Summer	2005	100
Kalama River Trap	LCR	Winter	2005	47
North Fork Lewis River, Cedar Trap	LCR	Winter	2005	60
North Fork Lewis River, Merwin Dam	LCR	Winter	2005	98
East Fork Lewis River	LCR	Winter	2005-2006	77
Sandy River, Marmot Dam	LCR	Winter	2005	98
Washougal River	LCR	Winter	2005-2006	71
Hood River, Powerdale Dam	LCR	Winter	2006	99
Willamette River				

Sampling Location	DPS	Run type	Collection year	Sample Size
Clackamas River	LCR	Late winter	2000	41
Clackamas River, Eagle Creek	LCR	Late winter	2000	63
Clackamas River, North Fork Dam	LCR	Late winter	2005	42
North Fork Molalla River	UWR	Late winter	1996	49
North Santiam River, Bennett Dam	UWR	Late winter	2005	45
South Santiam River, Foster Dam	UWR	Late winter	2005	49
South Santiam River, Foster Dam	UWR	Late winter	2009	50
South Santiam River, Wiley Creek	UWR	Late winter	1997	28
Calapooia River	UWR	Late winter	1997	36

Population genetic structure was assessed with the analytical methodologies used in the recent Willamette River steelhead study conducted by Van Doornik *et al.* (2015). Details on the methods used for the following analyses are provided in that study. Genetic diversity among samples was examined by computing pair-wise F_{ST} values using the program GenAlEx (Peakall and Smouse 2006). The critical value used to test for significance between pair-wise F_{ST} values ($P = 0.008$) was corrected for multiple tests (Narum 2006). F_{ST} values were significantly different from each other for all pairs of samples, except for Foster Dam (2009) with North Santiam ($P = 0.073$), and for North Fork Mollalla with Wiley Creek ($P = 0.022$). The average F_{ST} value in comparisons of Clackamas River samples with Upper Willamette River samples was 0.023 (Table GENX2). Comparisons of Clackamas River samples with samples in the Lower Columbia River DPS averaged 0.032. These average F_{ST} values suggest that Clackamas River steelhead are more genetically differentiated from Lower Columbia River steelhead than they are from Upper Willamette River fish.

Table 3 -- Average pairwise F_{ST} values for 24 steelhead populations in the Lower Columbia River and Willamette River.

Comparison	Average F_{ST}
All samples	0.030
Within Willamette River (including Clackamas River)	0.019
Within Lower Columbia River	0.016
Willamette River (including Clackamas River) vs Lower Columbia River	0.045
Clackamas River vs other Willamette River	0.023
Clackamas River vs Lower Columbia River	0.032

Genetic population structure was examined by estimating Cavalli-Sforza and Edwards (1967) chord distances among samples over 1,000 bootstrap replicates using PHYLIP (Felsenstein 2005). The resulting distance values were then used to construct a consensus neighbor-joining tree (Figure 4). In addition, a principal coordinates analysis was conducted based upon pairwise F_{ST} values (Figure 5). Consistent with previous DNA analyses (Blakenship *et al.* 2011, Matala *et al.* 2014, Van Doornik *et* *al*. 2015), both figures depict two main clusters comprised of Lower Columbia River and Willamette River samples. Clackamas River samples clustered with upper Willamette River samples.

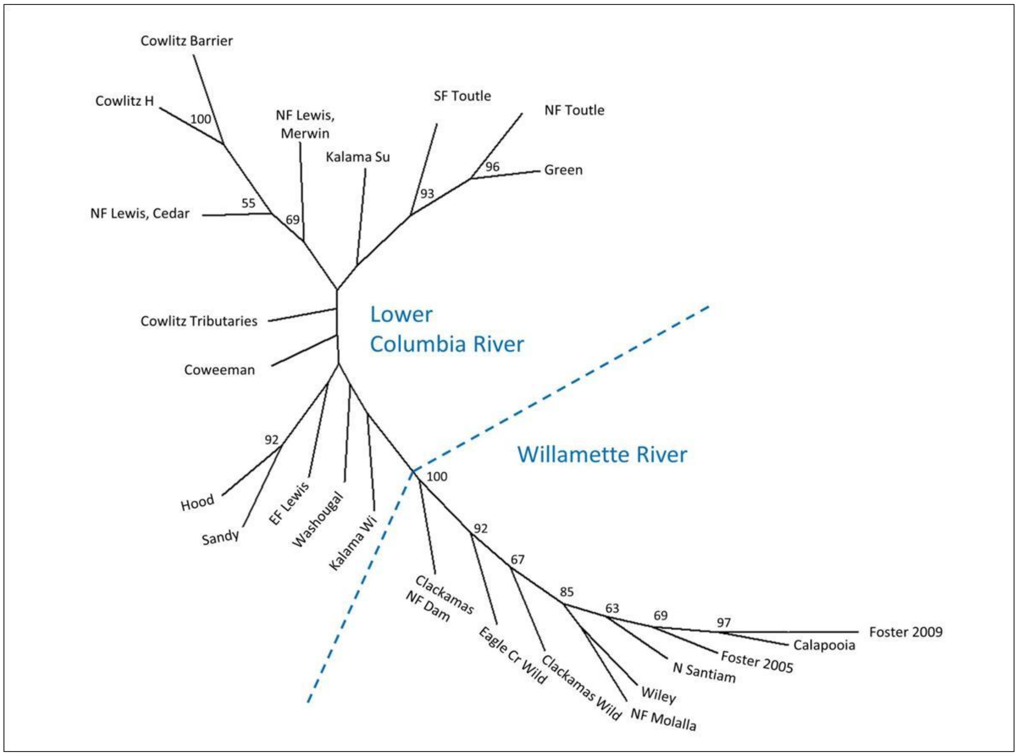


Figure 4 -- Consensus neighbor-joining tree of Cavalli-Sforza and Edwards (1967) chord distances for lower Columbia River and Willamette River steelhead samples. Bootstrap values are show at nodes with >50% consensus. Populations with "wi" or no notation are winter run, summer run populations are notated with "su". "H" – hatchery-origin.

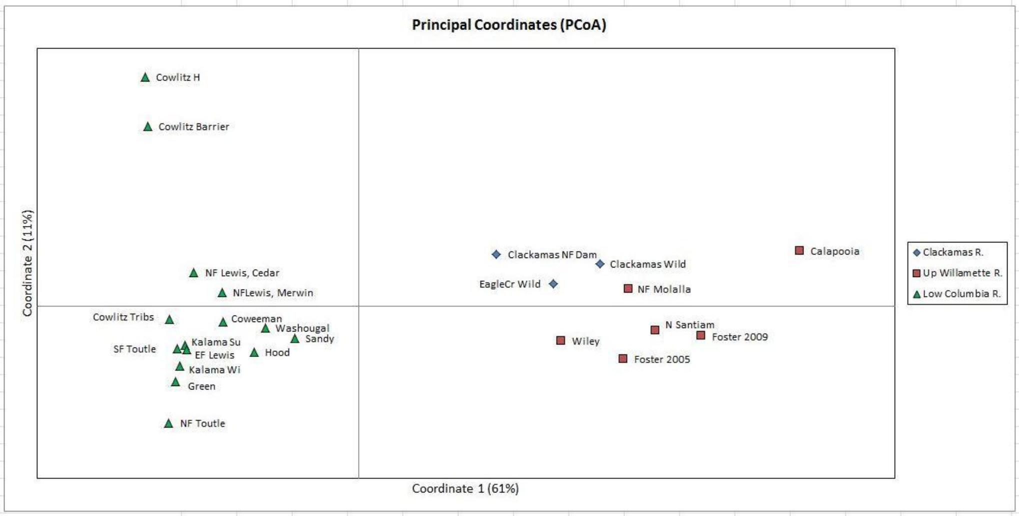
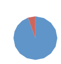


Figure 5 -- Principal components plot of pairwise F_{ST} values among lower Columbia River and Willamette River steelhead samples. Populations with "wi" or no notation are winter run, summer run populations are notated with "su". "H" – hatchery-origin.

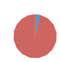
A Bayesian clustering analysis implemented in the program STRUCTURE (Falush *et al.* 2003) was used to infer the number of populations or population groups present in the compiled microsatellite dataset. In this analysis population membership of each individual fish sample is not identified a priori. Using the methods and parameters described by Van Doornik *et al*. (2015) these analyses revealed that two population groups were most likely (i.e., had the greatest value of the metric ΔK). Each of the 24 population samples was then evaluated for proportional membership in the two population groups. Lower Columbia River samples predominately belonged to the first group with membership coefficients ranging from 0.58 to 0.95 (Table 4). Willamette River samples, including those from the Clackamas River primarily belonged to the second group with membership coefficients from 0.75 to 0.96.

Table 4 -- Population group membership values for lower Columbia River and Willamette River steelhead samples.

Sampling Location	Run Type	Membership Coefficient		Chart
		PopGroup1	PopGroup2	
LOWER COLUMBIA RIVER				
Cowlitz River Hatchery	Late Winter	0.945	0.055	
Cowlitz River, Barrier Dam	Winter	0.913	0.087	

Sampling Location	Run Type	Membership Coefficient PopGroup1	PopGroup2	Chart
Cowlitz River tributaries	Winter	0.843	0.157	
Coweeman River	Winter	0.781	0.219	
Green River	Winter	0.845	0.155	
North Fork Toutle River	Winter	0.903	0.097	
South Fork Toutle River	Winter	0.830	0.170	
Kalama River	Summer	0.821	0.179	
Kalama River Trap	Winter	0.751	0.249	
North Fork Lewis River, Cedar Trap	Winter	0.828	0.172	
North Fork Lewis River, Merwin Dam	Winter	0.790	0.210	
East Fork Lewis River	Winter	0.745	0.255	
Sandy River, Marmot Dam	Winter	0.601	0.399	
Washougal River	Winter	0.673	0.327	
Hood River, Powerdale Dam	Winter	0.576	0.424	

WILLAMETTE RIVER

Sampling Location	Run Type	Membership Coefficient PopGroup1	PopGroup2	Chart
Clackamas River	Late winter	0.119	0.881	
Clackamas River, Eagle Creek	Late winter	0.120	0.880	
Clackamas River, North Fork Dam	Late winter	0.254	0.746	
North Fork Molalla River	Late winter	0.108	0.893	
North Santiam River, Bennett Dam	Late winter	0.070	0.930	
South Santiam River, Foster Dam, 2005	Late winter	0.103	0.897	
South Santiam River, Foster Dam, 2009	Late winter	0.059	0.942	
South Santiam River, Wiley Creek	Late winter	0.103	0.897	
Calapooia River	Late winter	0.037	0.963	

CONCLUSIONS

The review of recent DNA studies presented here, as well as the genetic analysis conducted for this report indicate that winter run steelhead in the Clackamas River are genetically more similar to native winter run steelhead in the Upper Willamette River than to steelhead in the Lower Columbia River. At the time of the original coast-wide status review (Busby *et al.* 1996) allozyme data existed for only a single putative native Clackamas River winter run population from Eagle Creek NFH. Analysis of that sample suggested that Clackamas River steelhead were genetically aligned with Lower Columbia River populations. It is possible that overlap in adult return times may have resulted in interbreeding of the steelhead stocks cultured at Eagle Creek NFH, including the Big Creek stock that was imported from the Lower Columbia River. If so, that may explain the affinity of that earlier genetic sample with those from the Lower Columbia River. Van Doornik *et al*. (2015) pointed out that the microsatellite DNA data, as evidenced in the STRUCTURE analysis of population group membership, also suggest that Clackamas River winter run steelhead may have experienced some level of introgression from Lower Columbia River stocks. That observation is supported here, where membership coefficients to the Lower Columbia River population group were somewhat greater for Clackamas River samples than for upper Willamette River samples. Overall, the new genetic information indicates that the boundary of the Lower Columbia River DPS and Upper Willamette River DPS should be revised. In addition, a review of the boundary would benefit from the collection of genetic data from any winter run steelhead populations in the Willamette River below Willamette Falls that have not previously been sampled. For example, natural spawning steelhead populations were historically present in Johnson and Mount Scott creeks (Myers *et al*. 2006).

INTERIOR COLUMBIA RIVER DOMAIN STATUS SUMMARIES

UPPER COLUMBIA RIVER SPRING-RUN CHINOOK SALMON ESU

BRIEF DESCRIPTION OF ESU

The Upper Columbia Spring-Run Chinook salmon ESU includes naturally spawning spring-run Chinook salmon in the major tributaries entering the Columbia River upstream of Rock Island Dam and associated hatchery programs (70FR37160; Figure 6). The ESU was listed as Endangered under the ESA in 1998 (affirmed in 2005 and 2012).

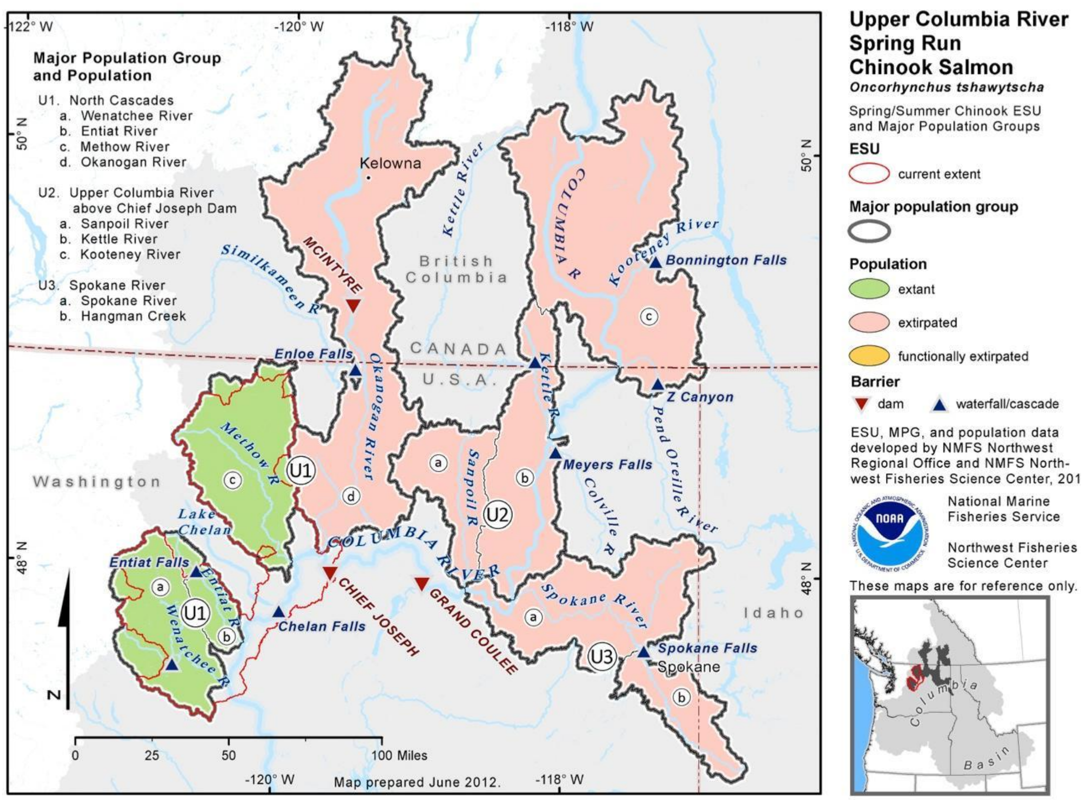


Figure 6 – Map of the Upper Columbia River Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

In the 2005 review, a slight majority (53%) of the cumulative votes cast by the BRT members placed this ESU in the "in danger of extinction" category, with the next category, "likely to become endangered", receiving a substantial number of votes as well (45%) (Good *et al.* 2005). The 2005 BRT review noted that Upper Columbia Spring Chinook populations had "rebounded somewhat from the critically low levels" observed in the 1998 review. Although the BRT considered this an encouraging sign, they noted that the increase was largely driven by returns in the two most recent spawning years available at the time of the review. The BRT ratings were also influenced by the fact that two out of the three extant populations in this ESU were subject to extreme hatchery intervention measures in response to the extreme downturn in returns during the 1990s. Good *et al*. (2005) stated that these measures were "...a strong indication of the ongoing risks to this ESU, although the associated hatchery programs may ultimately play a role in helping to restore naturally self-sustaining populations."

2010

The status of the ESU in 2010 was reported in Ford *et al*. (2011). At that time, the Upper Columbia Spring Chinook ESU was not currently meeting the viability criteria (adapted from the ICTRT) in the Upper Columbia Recovery Plan. Increases in natural origin abundance relative to the extremely low spawning levels observed in the mid-1990s were encouraging; however, average productivity levels remained extremely low. Overall, the report concluded that the viability of the Upper Columbia Spring Chinook salmon ESU had likely improved somewhat since the time of the last BRT status review, but the ESU was still clearly at moderate-to-high risk of extinction.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Annual abundance estimates for each of the extant populations in this ESU are generated based on expansions from redd surveys and carcass sampling. Index area redd counts have been conducted in these river systems since the late 1950's. Multiple pass surveys in index areas complemented by supplemental surveys covering the majority of spawning reaches have been conducted since the mid 1980's. For more recent years, estimates of annual returns to the Wenatchee River population also reflect counts and sampling data obtained at a trap at the Tumwater Dam on the mainstem river downstream of spring Chinook spawning areas. The data series for each population has been updated to include return years 2009 to 2014. Recent year estimates of spawner abundance, hatchery and natural origin proportions and age composition were provided by the Washington Department of Fish and Wildlife and are available through the WDFW SCoRE website[5].

Smolt Production

Natural production of spring Chinook salmon from the Chiwawa River tributary to the Wenatchee River has been monitored since 1991 (Hillman *et al.* 2015). Smolt traps at the mouth of the Chiwawa River and in the downstream Wenatchee River mainstem allow for generating annual estimates of total smolt production resulting from spawning in the Chiwawa River. Most of the smolts leaving the Wenatchee River from production in the Chiwawa River emigrate as yearlings in the spring of their second year of life. A portion of Chiwawa River production moves downstream in the summer and

[5] https://fortress.wa.gov/dfw/score/score/species/chinook.jsp?species=Chinook#spawning

fall and overwinters in the mainstem Wenatchee River before emigrating in the spring. Analyses done in support of a life cycle model for Wenatchee Spring Chinook indicate that the proportion of presmolts emigrating downstream for extended rearing and overwintering increases substantially with density (Jorgensen *et al.* 2013a). Smolt production from the Chiwawa River has increased since the early 1990s, with peak production occurring in 2001 and 2002 (Figure 7).

Smolt to Adult Return Rates

The ICTRT current productivity metric incorporates an adjustment for annual smolt to adult return rate (SAR) estimates to reduce the impact of short-term climate variability (ICTRT 2007b). The SAR index used for all three Upper Columbia River Spring Chinook salmon population data series uses natural origin smolt-to-adult estimates derived from smolt and adult monitoring of production from the Chiwawa River along with a longer data series of smolt to adult return survival estimates for Leavenworth Hatchery releases. The indices represent cumulative out of basin survivals (downstream passage, ocean life stages, upstream passage including harvest escapement rates). The SAR series has been updated to include estimates through the 2009 brood year (Figure 8). SAR estimates for the 2006-2008 brood outmigrants were at the high end of the range for the whole series, but below the peak SAR levels observed in the early 2000s. The aggregate Upper Columbia SAR series showed similar patterns to SARs for other Interior Columbia River ESUs; relatively low survivals in the early 1990s brood years followed by peaks in the late 1990s and late 2000's. The large year to year fluctuations in marine survival reflected in these series makes it difficult to detect potential changes in abundance that might result from recent actions to improve survival or capacity.

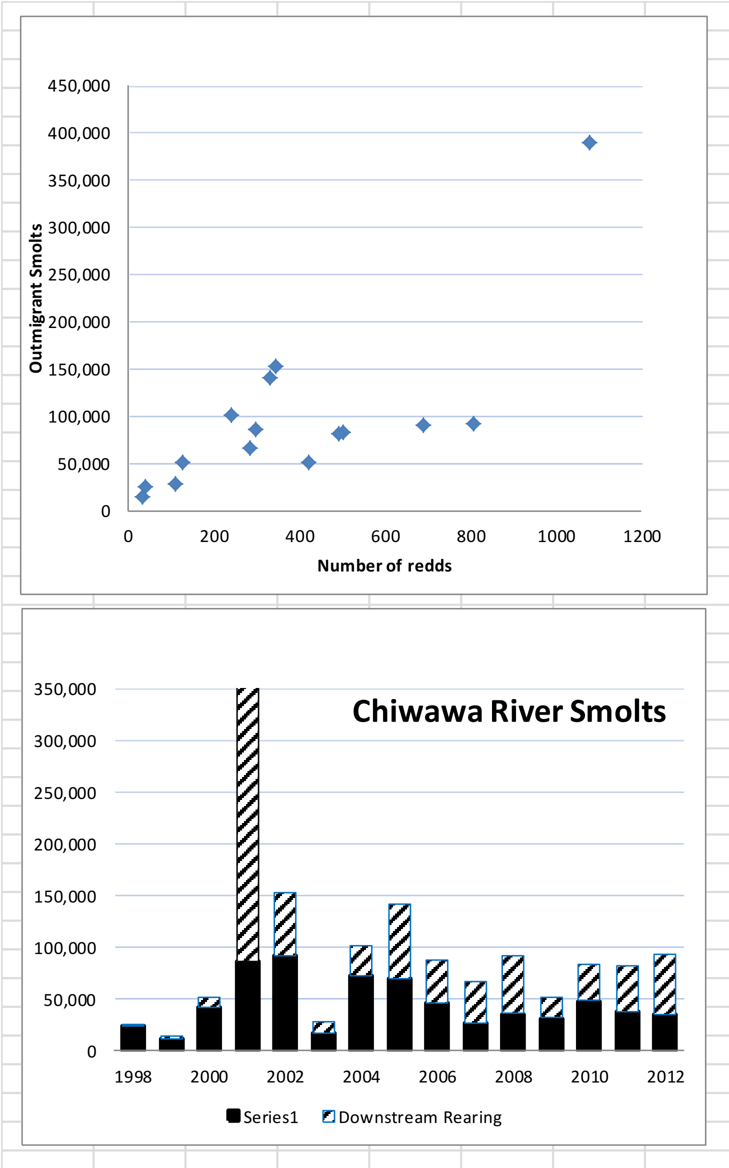


Figure 7 - Chiwawa River natural smolt production. Top: number of smolts produced vs. parent brood year redd counts. Bottom: number of smolts produced that are natal rearing (black bars) and downstream rearing (striped bars) components by brood year (Hillman *et al.* 2014).

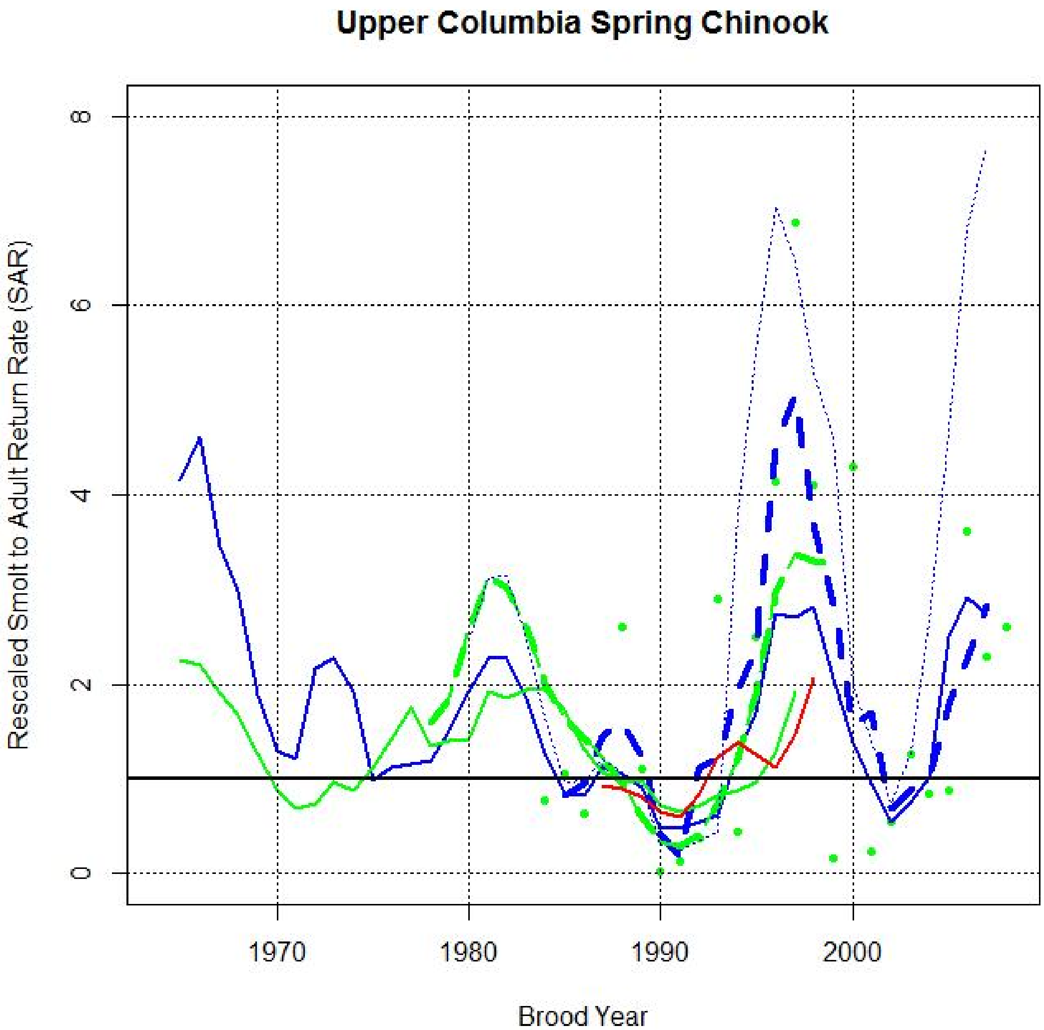


Figure 8 - Upper-Columbia River natural-origin spring Chinook salmon aggregate smolt-to-adult return rates (blue points and heavy dashed line) estimated as brood year ratios of smolt outmigrants to returning adults. Aggregate SARs for other Interior Columbia basin ESUs and DPSs provided for comparison. Snake River aggregate Spring/Summer Chinook (solid blue), Snake River aggregate natural origin steelhead (dashed green), Tuccannon spring Chinook (dotted blue). Upper Columbia steelhead (green dashed line), Mid-Columbia steelhead (red line). Each SAR series is rescaled by dividing annual values by the corresponding series mean to faclilitate relative comparison. Lines are three year moving averages.

Ocean Condition Indices

Upper Columbia spring Chinook salmon are a component of the Columbia River spring Chinook run that is believed to occupy mid-shelf waters during the early ocean life history phase (see Environmental Trends section below). Aggregate annual returns of Columbia River Spring Chinook are correlated with a range of ocean condition indices including measures of broad scale physical conditions, local biological indicators, and local physical factors (Peterson *et al.* 2014a). Several indicators, either individually or in combination, correlate well with spring Chinook salmon adult returns with a lag of 1 to 2 years. However, for each specific indicator or combination, there are anomalous years that fall outside of the apparent relationships. Work is continuing to further understand the relationships among physical and biological 'drivers' and annual levels of ocean survival for salmonid species in the ocean environment. After accounting for age at return at time of ocean entry, the annual pattern in the Upper Columbia spring Chinook ESU SAR index generally corresponds to the composite rankings across ocean indicators available for early ocean years starting in the late 1990s (Peterson *et al*. 2014).

Multiple Population Analyses

The 2009 FCRPS Adaptive Management and Implementation Plan called for more detailed metapopulation analyses that could be used to help identify populations particularly vulnerable to extinction due to isolation as well as to understand commonalities and differences in year to year variations among populations (Fullerton *et al.* 2013). Preliminary results indicate that the three extant Upper Columbia spring Chinook salmon populations are relatively distinct and isolated from other populations, both in terms of genetics/dispersal characteristics as well as in patterns of annual abundance. In the multiple population abundance trend analysis, all three Upper Columbia River populations showed a strong correlation with a particular pattern that was not identified with populations from other regions; a general increase from the late 1950s through the mid 1980s followed by an abrupt decline and a subsequent slow increase (Jorgensen *et al.* 2013b). More effort will be needed to understand the drivers for this pattern and the implications for future environmental influence.

ABUNDANCE AND PRODUCTIVITY

Updated data series on spawner abundance, age structure and hatchery/natural proportions were used to generate current assessments of abundance and productivity at the population level. Evaluations were done using both a set of metrics corresponding to those used in prior Biological Review Team (BRT) reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The BRT level metrics were consistently done across all ESUs and DPSs to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section below. The ICTRT abundance and productivity metrics are measured over longer time frames to dampen the effects of annual variations and they use annual natural origin age composition to calculate brood year recruitment when sampling levels meet regional fishery agency criteria.

Annual spawning escapements for all three of the extant Upper Columbia spring Chinook populations showed steep declines beginning in the late 1980s, leading to extremely low abundance levels in the mid-1990s (Figure 9, Table 5). The steep downward trend reflects the extremely low return rates for natural production from the 1990-94 brood years (Figure 10). Brood year replacement rates were consistently below 1.0 even at low parent spawner levels throughout the 1990s. Steeply declining trends across indices of total spawner abundance were a major consideration in the 1997 BRT risk assessment prior to listing of the ESU. Updating the data series to include 2009-2014, the short-term (e.g., 15 year) trend in wild spawners has been neutral for the Wenatchee population and positive for the Entiat and Methow populations (Table 6). In general, both total and natural origin escapements for all three populations increased sharply from 1999 through 2002 and have shown substantial year to year variations in the years following, with peaks around 2001 and 2010. Average natural origin returns remain well below ICTRT minimum threshold levels.

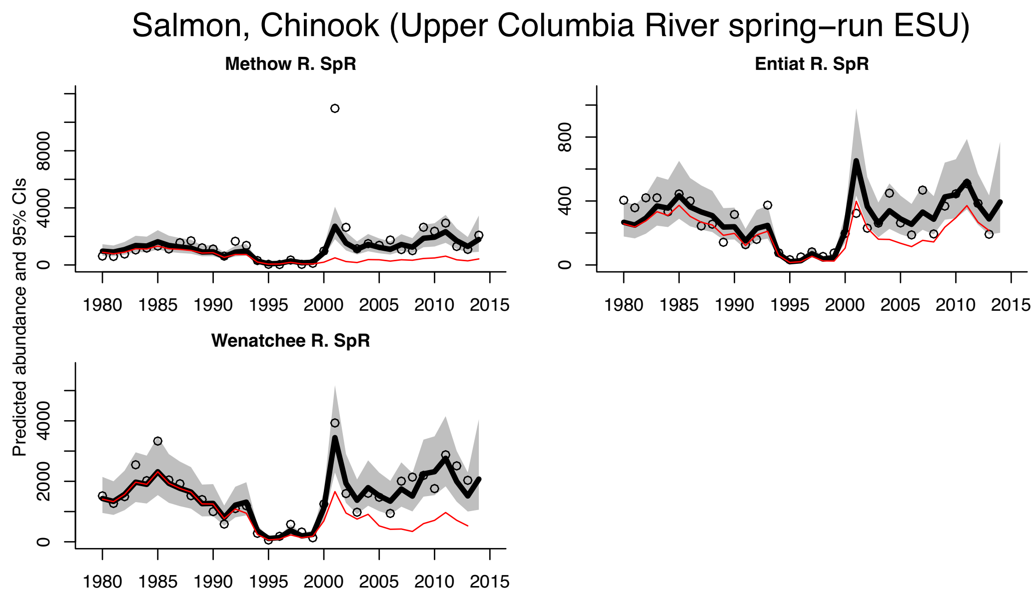


Figure 9 -- Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

The annual return per spawner series for each population directly reflects the patterns in natural origin aboundance (Figure 10). Brood year escapements with positive return per spawner values are associated with those years leading up to the peaks in natural origin spawner returns in each series.

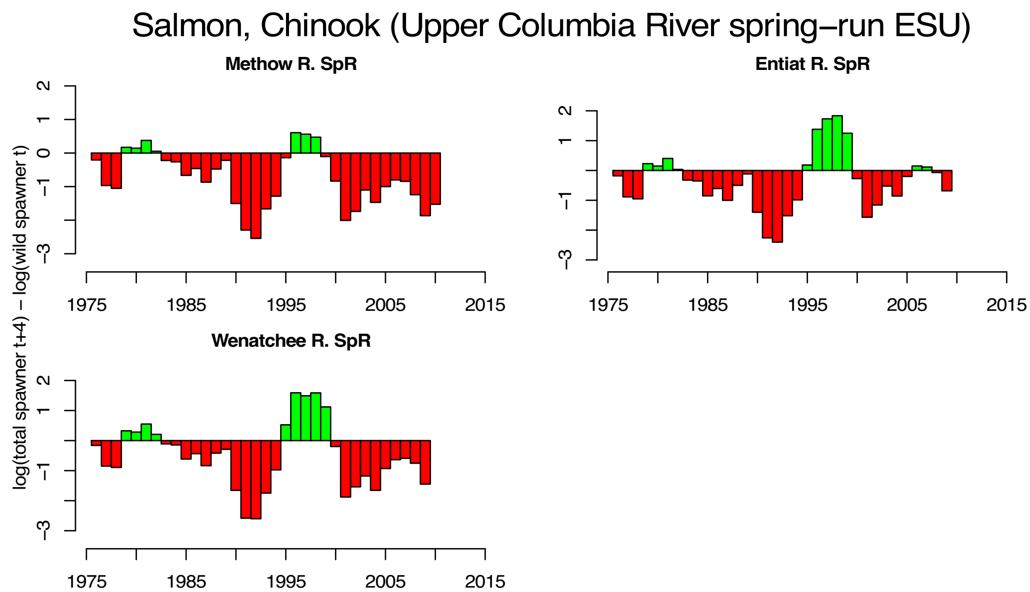


Figure 10 – Trends in population productivity, estimated as the log of the smoothed natural origin spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4). Spawning years on x-axis.

Table 5 -- 5-year geometric mean of raw natural spawner counts. This is the raw total spawner count times the fraction natural estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners available. The geometric mean was computed as the product of counts raised to the power of reciprocal the number of counts available (2 to 5). A minimum of 2 values was used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Methow R. SpR	Up. Columbia/East Slope Cascades	722 (867)	44 (75)	292 (2171)	379 (1470)	425 (1828)	12 (24)
Entiat R. SpR	Up. Columbia/East Slope Cascades	153 (179)	37 (56)	148 (280)	129 (278)	265 (360)	105 (29)
Wenatchee R. SpR	Up. Columbia/East Slope Cascades	621 (735)	120 (192)	860 (1652)	385 (1671)	785 (2254)	104 (35)

Table 6 -- 15-year trends in log natural spawner abundance computed from a linear regression applied to the smoothed natural spawner log abundance estimate. Only populations with at least 4 natural spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Methow R. SpR	Up. Columbia/East Slope Cascades	-0.05 (-0.15, 0.06)	0.07 (0.02, 0.12)
Entiat R. SpR	Up. Columbia/East Slope Cascades	0.03 (-0.09, 0.15)	0.08 (0.01, 0.14)
Wenatchee R. SpR	Up. Columbia/East Slope Cascades	0.02 (-0.1, 0.14)	0.01 (-0.05, 0.07)

Smolt Production

Natural production of spring Chinook salmon from the Chiwawa River tributary to the Wenatchee River has been monitored since 1991 (Hillman *et al.* 2015). Smolt traps at the mouth of the Chiwawa River and in the downstream Wenatchee River mainstem allow for generating annual estimates of total smolt production resulting from spawning in the Chiwawa River. Most of the smolts leaving the Wenatchee River from production in the Chiwawa River emigrate as yearlings in the spring of their second year. A portion of Chiwawa River production moves downstream in the summer and fall and overwinters in the mainstem Wenatchee River before emigrating in the spring (Figure 7). Smolt production from two other Wenatchee River tributaries has been monitored for shorter periods; Nason Creek (2012 starting year) and White River (2012 starting year). Both series show some indication of density dependent effects at higher parent spawning levels (Hillman *et al.*, 2014)

Harvest

Spring Chinook salmon from the upper Columbia basin migrate offshore in marine water and where impacts in ocean salmon fisheries are too low to be quantified. The only significant harvest occurs in the mainstem Columbia River in tribal and non-tribal fisheries directed at hatchery spring Chinook salmon from the Columbia and Willamette Rivers. Exploitation rates have remained relatively low, generally below 10%, though they have been increasing in recent years (Figure 11). The increases have resulted from increased allowable harvest rates under the abundance driven sliding scale harvest rate strategy guiding annual management in response to continued large returns of hatchery spring Chinook to the Columbia River Basin.

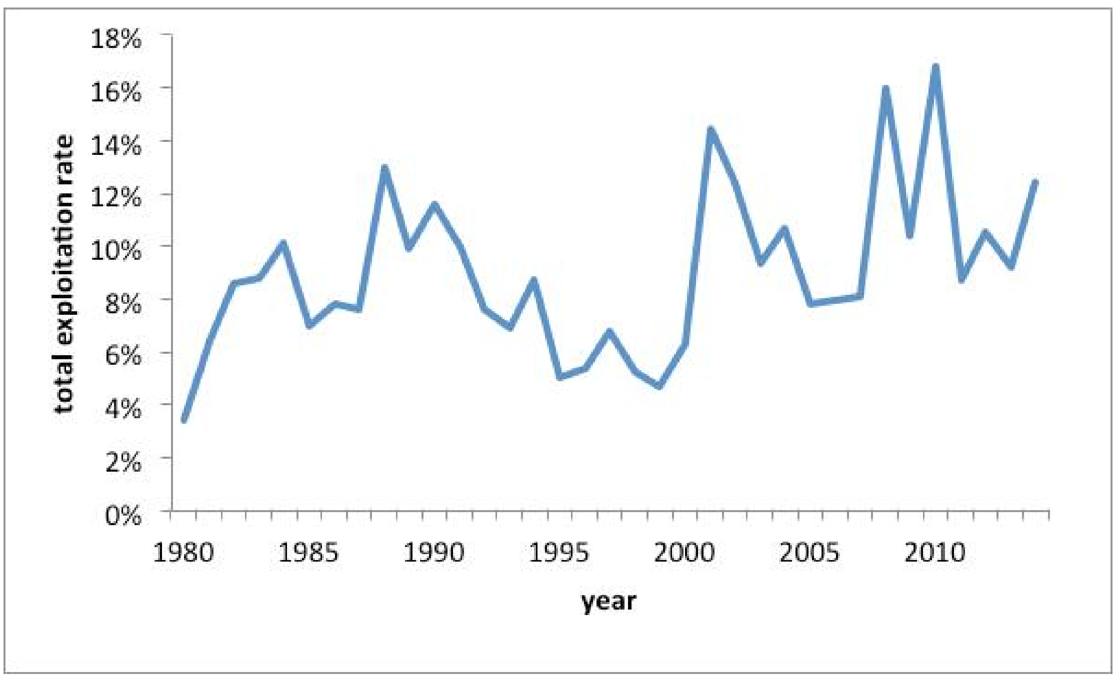


Figure 11 -- Total exploitation rate for upper Columbia River spring Chinook salmon. Data from the Columbia River Technical Advisory Committee (TAC 2015).

SPATIAL STRUCTURE AND DIVERSITY

The proportions of natural origin contributions to spawning in the Wenatchee and Methow populations have trended downwards since 1990 (Figure 12, Table 7), reflecting the large increase in releases and subsequent returns from the directed supplementation programs in those two drainages (Hillman *et al.* 2015). There is no direct hatchery supplementation program in the Entiat River. Hatchery-origin spawners in the Entiat River system are predominately strays from Entiat NFH releases. The Entiat NFH spring Chinook release program was discontinued in 2007, and the upward trend in proportional natural origin since then can be attributed to that closure. In recent years, hatchery supplementation returns from the adjacent Wenatchee River program have also strayed into the Entiat (Ford *et al.* 2015). The nearby Eastbank Hatchery facility is used for rearing the Wenatchee River supplementation stock prior to transfer to the Chiwawa acclimation pond. It is possible that some of the returns from that program are homing on the Eastbank facility and then straying into the Entiat River, the nearest spawning area.

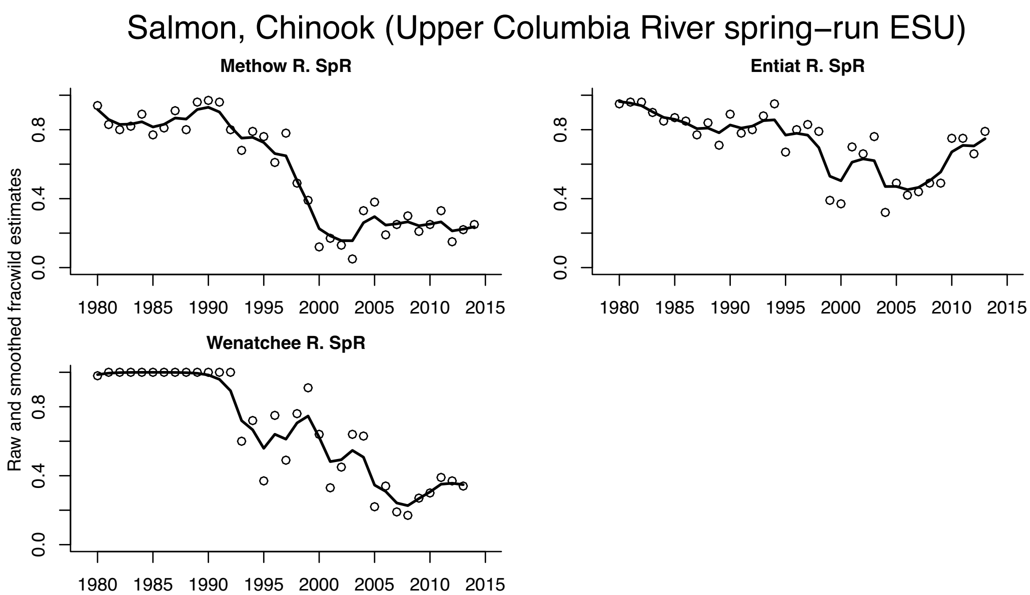


Figure 12 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 7 -- 5-year mean of fraction natural origin (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Methow R. SpR	0.84	0.61	0.16	0.27	0.24
Entiat R. SpR	0.86	0.70	0.56	0.47	0.74
Wenatchee R. SpR	0.86	0.66	0.54	0.24	0.35

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

NOAA Fisheries (National Marine Fisheries Service adopted a recovery plan for Upper Columbia Spring Chinook and steelhead in 2007 (FR 72 #194. 57303-57307). The Plan was developed by the Upper Columbia Salmon Recovery Board (UCSRB) and is available through their website (http://www.ucsrb.com/). The Upper Columbia Salmon Recovery Plan's overall goal is "...to achieve recovery and delisting of spring Chinook salmon and steelhead by ensuring the long-term persistence of viable populations of naturally produced fish distributed across their native range."

Two incremental levels of recovery objectives are incorporated into the Upper Columbia Salmon Recovery Plan. Increasing natural production sufficiently to upgrade each Upper Columbia River ESU from "endangered" to "threatened" status is stated as an initial objective. The Plan includes three specific quantitative reclassification criteria expressed relative to population viability curves (ICTRT 2007). Abundance and productivity of natural origin spring Chinook salmon within each of the extant Upper Columbia populations, measured as 8-year geometric means (representing approximately two generations), must fall above the viability curve representing the minimum combinations projecting to a 10% risk of extinction over 100 years. In addition, the plan incorporates explicit criteria for spatial structure and diversity adopted from the ICTRT viability report. The mean score for the three metrics representing natural rates and spatially mediated processes should result in a moderate or lower risk in each of the three populations and all threats defined as high risk must be addressed. In addition, the mean score for the eight ICTRT metrics tracking natural levels of variation should result in a moderate or lower risk score at the population level.

Achieving recovery (delisting) of each ESU via sufficient improvement in the abundance, productivity, spatial structure and diversity is the longer-term goal of the UCSRB Plan. The Plan includes two specific quantitative criteria for assessing the status of the Spring Chinook ESU against the recovery objective; "The 12-year geometric mean (representing approximately three generations) of abundance and productivity of naturally produced spring Chinook within the Wenatchee, Entiat and Methow populations must reach a level that would have not less than a 5% extinction-risk (viability) over a 100 year period" and "at a minimum, the Upper Columbia Spring Chinook ESU will maintain at least 4,500 naturally produced spawners and a spawner:spawner ratio greater than 1:1 distributed among the three populations". The minimum number of naturally produced spawners (expressed as 12 year geometric means) should exceed 2,000 each for the Wenatchee and Methow River populations and 500 within the Entiat River. Minimum productivity thresholds were also established in the Plan. The 12-year geometric mean productivity should exceed 1.2 spawners per parent spawner for the two larger populations (Wenatchee and Methow Rivers), and 1.4 for the smaller Entiat River population. The ICTRT had recommended that at least two of the three extant populations be targeted for highly viable status (less than 1% risk of extinction over 100 years) because of the relatively low number of extant populations remaining in the ESU. The UC Plan adopted an alternative approach for addressing the limited number of populations in the ESU – 5% or less risk of extinction for all three extant populations.

The Upper Columbia Salmon Recovery Plan also calls for '… restoring the distribution of naturally produced spring Chinook salmon and steelhead to previously occupied areas where practical; and conserving their genetic and phenotypic diversity." Specific criteria included in the UCSRB Plan reflect a combination of the specific criteria recommended by the ICTRT (ICTRT 2007) and in the earlier QAR effort (Ford *et al.* 2001). The Plan incorporates spatial structure criteria specific to each spring Chinook salmon population. For the Wenatchee River population, the criteria call for observed natural spawning in four of the five major spawning areas as well as in at least one of the minor spawning areas downstream of Tumwater Dam. In the Methow River, natural spawning should be observed in three major spawning areas. In each case, the major spawning areas should include a minimum of 5% of the total return to the system or 20 redds, whichever is greater. The Entiat River Spring Chinook population includes a single historical major spawning area.

The Plan calls for meeting or exceeding the same basic spatial structure and diversity criteria adopted from the ICTRT viability report for recovery as for reclassification (see above).

Table 8 - Upper Columbia spring Chinook salmon ESU population viability status summary. Current abundance and productivity estimates are geometric means. Range in annual abundance, standard error and number of qualifying estimates for productivities in parentheses. Upward arrows: current estimates increased over prior review. Oval: no change since prior review.

Population	Abundance and productivity metrics				Spatial structure and diversity metrics			Overall viability rating
	ICTRT minimum threshold	Natural spawning abundance	ICTRT productivity	Integrated A/P risk	Natural processes risk	Diversity risk	Integrated SS/D risk	
Wenatchee River 2005–2014	2,000	545 ↑ (311-1,030)	0.60 ↑ (0.27, 15/20)	High	Low	High	High	**High risk**
Entiat River 2005–2014	500	166 ↑ (78-354)	0.94 ↑ (0.18, 12/20)	High	Moderate	High	High	**High risk**
Methow River 2005–2014	2,000	379 ↑ (189-929)	0.46 ⬭ (0.31, 16/20)	High	Low	High	High	**High risk**

Overall abundance and productivity (A/P) remains rated at high risk for the each of the three extant populations in this MPG/ESU (Table 8). The 10-year geometric mean abundance of adult natural-origin spawners has increased for each population relative to the levels reported in the 2011 status update, but natural origin escapements remain below the corresponding ICTRT thresholds. The combinations of current abundance and productivity for each population result in a high risk rating when compared to the ICTRT viability curves.

The composite spatial structure/diversity (SS/D) risks for all three of the extant populations in this MPG are rated at high (Table 8). The spatial processes component of the SS/D risk is low for the Wenatchee and Methow river populations and moderate for the Entiat River (due to a loss of production in lower section which increases effective distance to other populations). All three of the extant populations in this MPG are rated at high risk for diversity, driven primarily by chronically high proportions of hatchery-origin spawners in natural spawning areas and lack of genetic diversity among the natural-origin spawners (ICTRT 2008).

Based on the combined ratings for A/P and SS/D, all three of the extant populations of Upper Columbia spring Chinook salmon remain rated at high overall risk (Table 8).

UPDATED BIOLOGICAL RISK SUMMARY

Current estimates of natural origin spawner abundance increased relative to the levels observed in the prior review for all three extant populations, and productivities were higher for the Wenatchee and Entiat and unchanged for the Methow. However abundance and productivy remained well below the viable thresholds called for in the Upper Columbia Recovery Plan for all three populations. Short-term patterns in those indicators appear to be largely driven by year-to year fluctuations in survival rates in areas outside of these watersheds. All three populations continued to be rated at low risk for spatial structure but at high risk for diversity criteria. Large-scale supplementation efforts in the Methow and Wenatchee Rivers are ongoing, intended to counter short-term demographic risks given current average survival levels and the associated year-to-year variability. Under the current recovery plan, habitat protection and restoration actions are being implemented that are directed at key limiting factors. Achieving natural origin abundance and productivity levels above the threshold viability curve corresponding to 5% risk in extinction will require substantial improvements in survival and/or natural production capacity (Figure 13). Given the high degree of year-to-year variability in life stage survivals and the time lags resulting from the 5 year life cycle of the populations, it is not possible to detect incremental gains from habitat actions implemented to date in population level measures of adult abundance or productivity. Efforts are underway to develop life stage specific estimates of performance (survival and capacities) and to use a life cycle model framework to evaluate progress. Based on the information available for this review, the risk category for the Upper Columbia Spring Chinook ESU remains unchanged from the prior review (Ford *et al*. 2011). Although the status of the ESU is improved relative to measures available at the time of listing, all three populations remain at high risk.

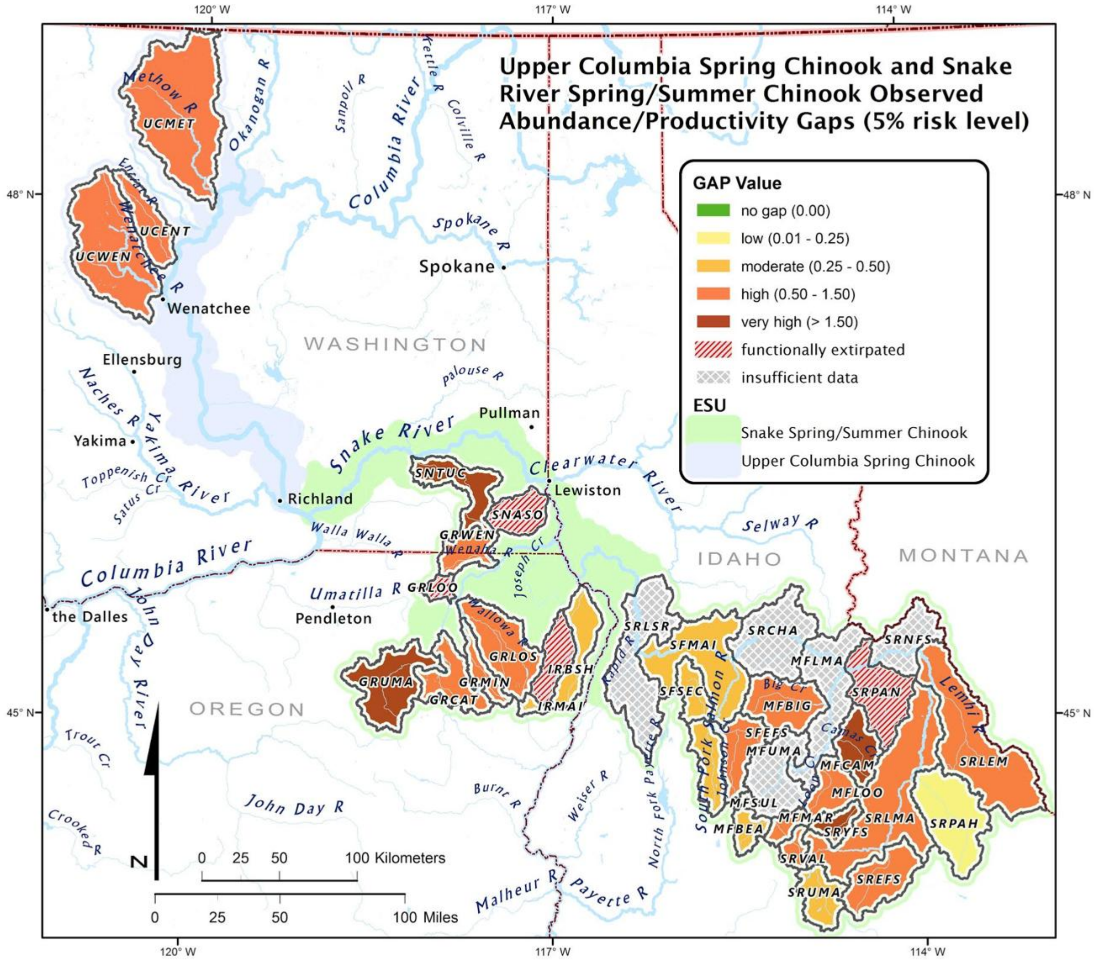


Figure 13 -- Abundance and productivity gaps for Upper Columbia spring Chinook ESU populations (map also includes Snake River Spring/Summer Chinook ESU populations for comparison). Populations with insufficient data to generate gaps are shaded in gray. Gaps are defined as relative improvement in productivity or limiting capacity required for a population to exceed its corresponding 5% risk viability curve (ICTRT 2007).

The Upper Columbia Recovery Plan includes a number of strategies for improving survival in tributary habitats and the mainstem migration corridor along with complementary harvest management and hatchery management regimes. The time frames for implementing actions and for those actions to result in improved survivals vary across strategies. Improved passage survivals relative to conditions prevalent at the time of listing are expected to be relatively immediate. Given the anticipated action implementation schedule and assumptions regarding time lags for realizing target habitat improvements incorporated into the Upper Columbia Recovery Plan, improvements in survival due to changes in habitat conditions are expected accrue over a 10–50 year period.

UPPER COLUMBIA RIVER STEELHEAD DPS

BRIEF DESCRIPTION OF ESU

The Upper Columbia Steelhead DPS includes all naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassable barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the US-Canada border, as well as six artificial propagation programs: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan Rivers), Winthrop NFH, Omak Creek and the Ringold steelhead hatchery programs (Figure 14). The Upper Columbia Steelhead DPS was originally listed under the ESA in 1997; it is currently designated as threatened.

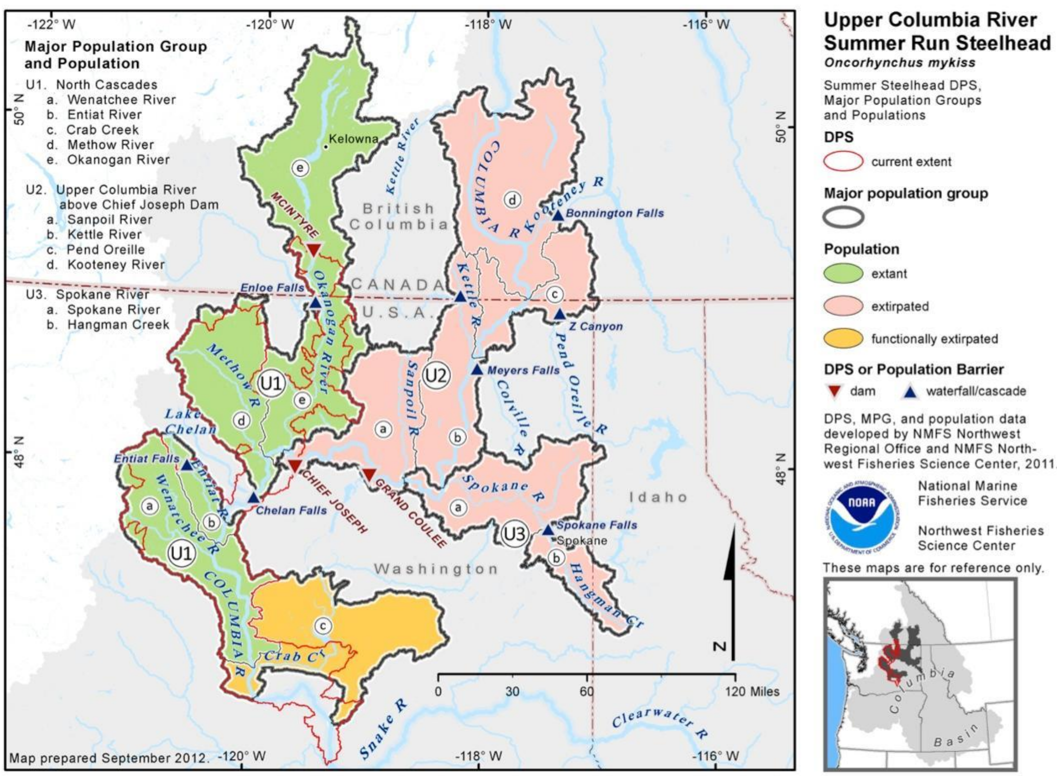


Figure 14 – Map of the Upper steelhead DPS's spawning and rearing areas, illustrating populations and major population groups.

NOAA Fisheries has defined DPSs of steelhead to include only the anadromous members of this species (70 FR 67130). Our approach to assessing the current status of a steelhead DPS is based evaluating information the abundance, productivity, spatial structure and diversity of the anadromous component of the species (Good *et al*. 2005; 70 FR 67130). Many steelhead populations along the West Coast of the U.S. co-occur with conspecific populations of resident rainbow trout. We recognize that there may be situations where reproductive contributions from resident rainbow trout may mitigate short-term extinction risk for some steelhead DPSs (Good *et al*. 2005; 70 FR 67130). We assume that any benefits to an anadromous population resulting from the presence of a conspecific resident form will be reflected in direct measures of the current status of the anadromous form.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The 2005 BRT cited low growth rate/productivity as the most serious risk factor for the Upper Columbia River steelhead DPS (Good *et al*. 2005). In particular, the BRT concluded that the extremely low replacement rate of natural spawners highlighted in the 1998 review continued through the subsequent brood cycle. The 2005 BRT assessment also identified very low natural spawner abundance compared to interim escapement objectives and high levels of hatchery spawners in natural areas as contributing risk factors. The 2005 BRT report did note that the number of naturally produced steelhead returning to spawn within this DPS had increased over the levels reported in the 1998 status review. As with the Mid-Columbia and Snake River DPS reviews, the 2005 BRT recognized that resident *O. mykiss* were associated with anadromous steelhead production areas for this DPS. The review stated that the presence of resident *O. mykiss* was considered a mitigating factor by many of the BRT members in rating extinction risk.

2010

The 2010 status review update reported that Upper Columbia steelhead populations had increased in natural origin abundance in recent years, but productivity levels remained low (Ford *et al*. 2011). The proportions of hatchery origin returns in natural spawning areas remained extremely high across the DPS, especially in the Methow and Okanogan River populations. The modest improvements in natural returns that had been observed the years prior to the review were probably primarily the result of several years of relatively good natural survival in the ocean and tributary habitats. Tributary habitat actions called for in the Upper Columbia Recovery Plan were anticipated to be implemented over the next 25 years and the benefits of some of those actions would require some time to be realized. Overall, the new information considered did not indicate a change in the biological risk category since the time of the last BRT status review.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

The 2011 NWFSC status review (Ford *et. al* 2011) evaluated the status of the Upper Columbia Steelhead DPS based on data series through cycle year 2008/2009 for each of the four extant populations, along with sampling information collected at Priest Rapids Dam for the aggregate return to the Upper Columbia Basin and Wells Dam (Methow and Okanogan populations combined). Estimates generated using that methodology are currently available through the 2013/2014 cycle years for each population. Spawning escapement estimates are based on a run reconstruction model incorporating annual dam counts, results of a three year radio tracking program and estimates of broodstock and fisheries removals in various reaches above Rock Island Dam. Estimates are generated by WDFW regional staff (ncorporating information from the Colville Tribal Fish & Wildlife Department) and are available through the WDFW SCoRE website[6]. An updated approach for estimating population level escapements has been initiated in recent years. That approach uses mark/recapture statistics based on data generated from the combination of systematic PIT tagging of a target proportion of the returns passing Rock Island Dam (below all four population spawning tributaries) and subsequent detections at arrays in each of the tributaries. Preliminary comparisons of the results from the updated approach with the methods used in prior years indicate they generally produce compatible estimates for a given year. It is anticipated that future estimates of annual population level spawning escapements for the Upper Columbia Steelhead DPS will be based on the new methods. After five or more years are available to allow for refinements in the approach and a comparison of results from applying the old and new methodologies under a range of return levels, prior year escapement reconstructions may be revised (A. Murdoch, WDFW, pers. comm).

The SAR index for the Upper Columbia Steelhead DPS series uses natural origin smolt to adult estimates based on gatewell smolt sampling at Rock Island Dam and adult return combined with natural origin adult monitoring at Priest Rapids Dam. The index represent cumulative out of basin survivals - downstream passage, ocean life stages, upstream passage including harvest impact (Figure 16).

ABUNDANCE AND PRODUCTIVITY

Updated data series on spawner abundance, age structure and hatchery wild proportions were used to generate current assessments of abundance and productivity at the population level. Evaluations were done using both a set of metrics corresponding to those used in prior Biological Review Team (BRT) reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The BRT level metrics were consistently done across all ESUs and DPSs to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section below. The ICTRT abundance and productivity metrics are measured over longer time frames to dampen the effects of annual variations and they use annual natural origin age composition to calculate brood year recruitment when sampling levels meet regional fishery agency criteria.

The most recent estimates (5-year geometric mean) of total and natural-origin spawner abundance have increased relative to the prior review for all four populations (Figure 17 ,Table 9). The abundance series for the aggregate return monitored at Priest Rapids Dam (Figure 15) and for all

[6] https://fortress.wa.gov/dfw/score/

four populations generally reflect a common pattern in annual returns for both hatchery and natural origin fish. Although the magnitudes vary among the individual populations, each series shows three peaks in annual returns occurring in the mid-1980s, the early 2000s and 2010/2011. That pattern appears to be largely driven by variations in smolt to adult return rates (Figure 16). In spite of the recent increases, natural-origin returns remain well below target levels.

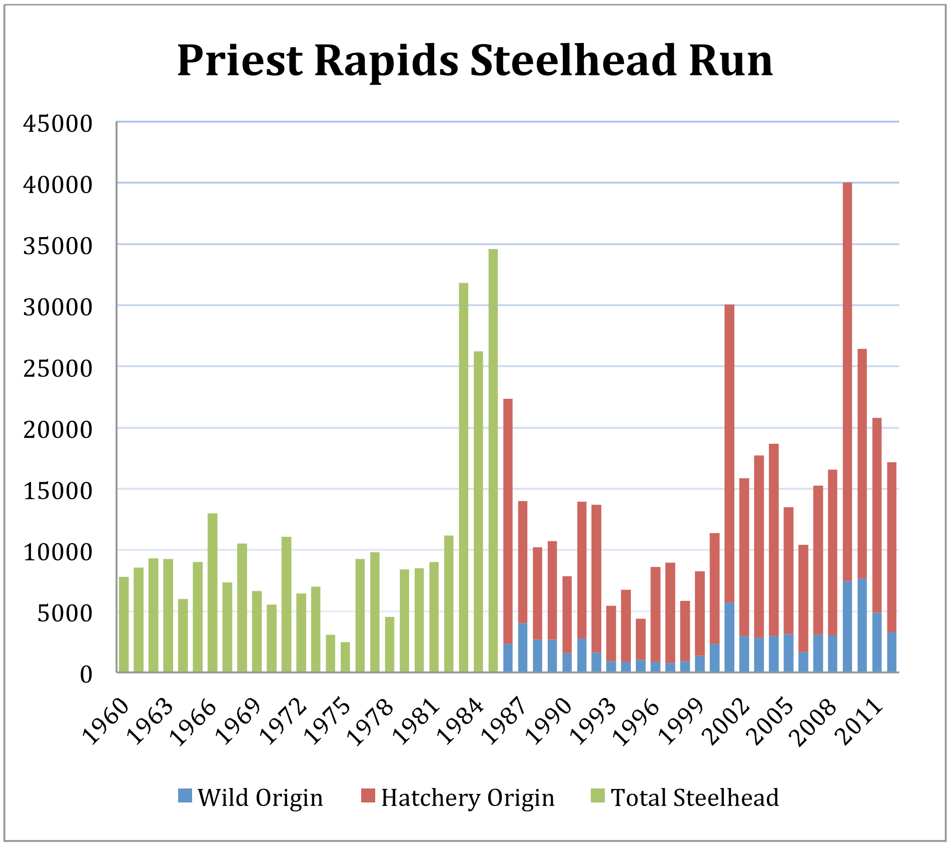


Figure 15 - Estimated passage of steelhead at Priest Rapids Dam based on ladder counts and WDFW trap sampling for run composition. (Brood year = passage year+1) Sampling program initiated in 1986 and are estimates of total (hatchery plus wild) run size. Counts for prior years were not directly sampled to determine hatchery proportions.

Annual brood year return-per-spawner estimates have been well below replacement in recent years for all four populations, with the exception of a few years for the Wenatchee River. The return per spawner estimates summarized in Figure 18 are ratios of the estimated natural origin returns produced from spawners in each brood year, under the assumption that both hatchery and natural origin fish contribute to production as parent spawners. In spite of the fact that each population is consistently exhibiting natural production rates well below replacement, natural production has not declined consistently, but has fluctuated at levels well below recovery objectives. The large numbers of hatchery fish on the spawning grounds each year may be subsidizing spawning at levels well above the current natural carrying capacity of the system.

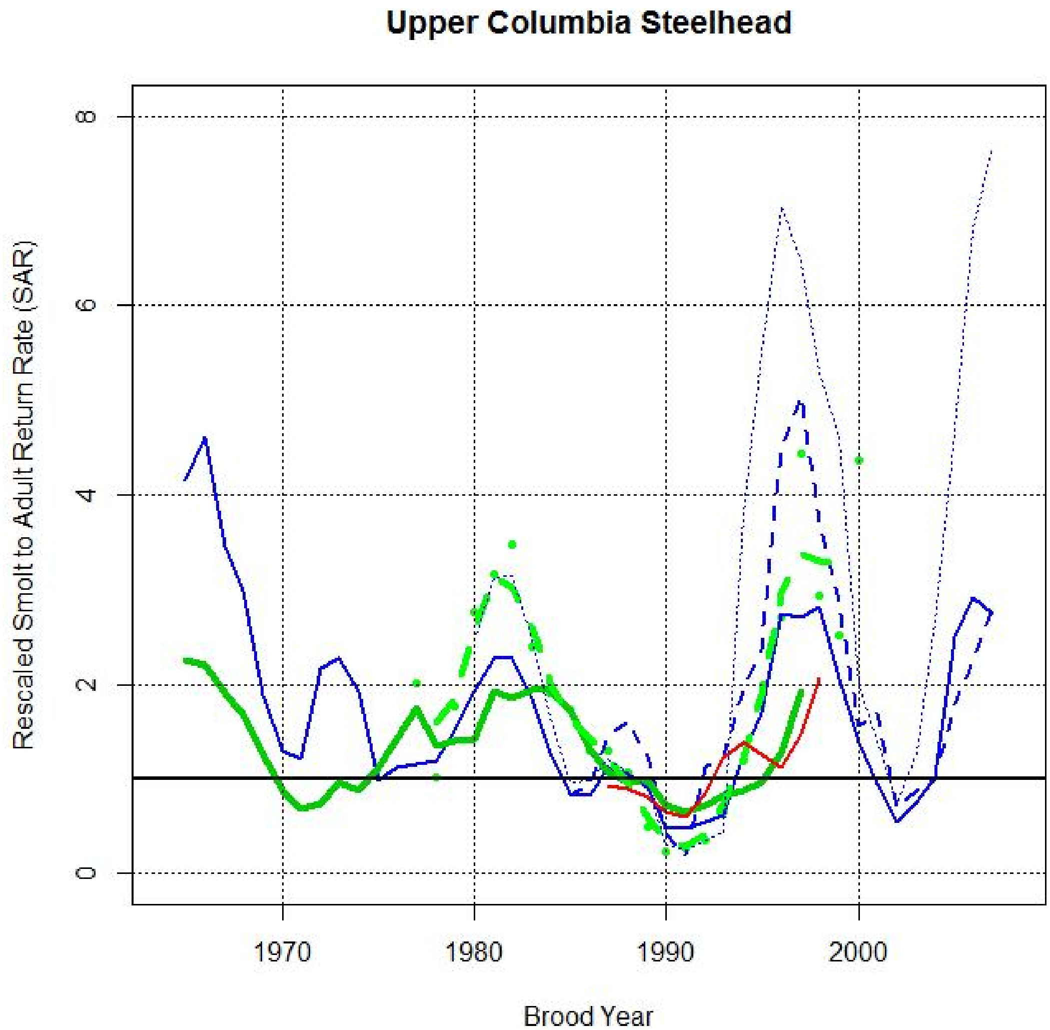


Figure 16 - Upper-Columbia River natural origin steelhead aggregate smolt to adult return rates (green points and heavy dashed line). Aggregate SARs for other Interior Columbia basin ESUs and DPSs provided for comparison. Snake River aggregate Spring/Summer Chinook (solid blue), Snake River aggregate natural origin steelhead (dashed green), Tuccannon spring Chinook salmon (dotted blue), Mid-Columbia steelhead (red line). Each SAR series is rescaled by dividing annual values by the corresponding series mean to faclilitate relative comparison. Lines are three year moving averages.

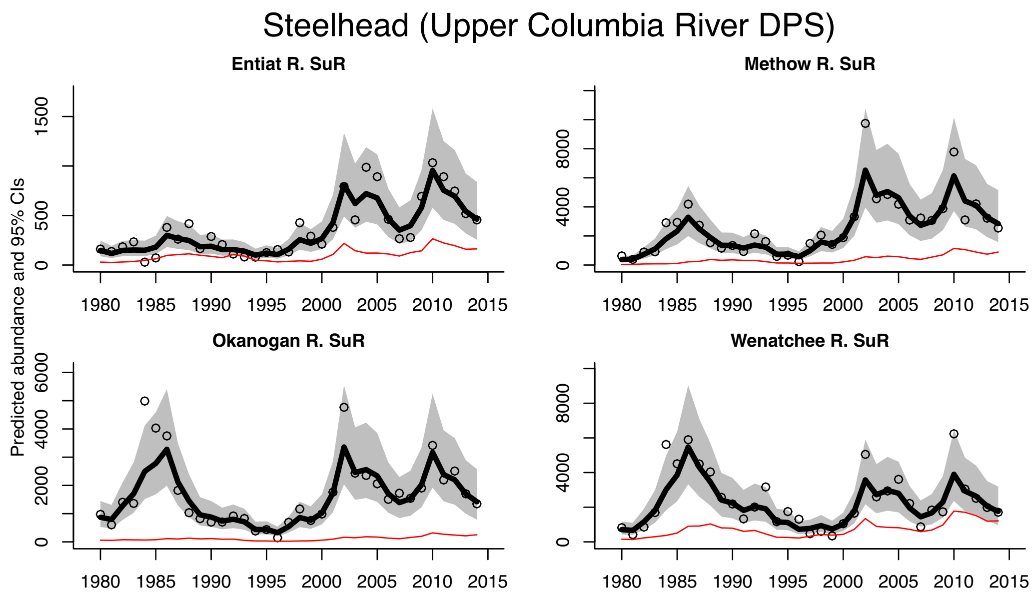


Figure 17 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

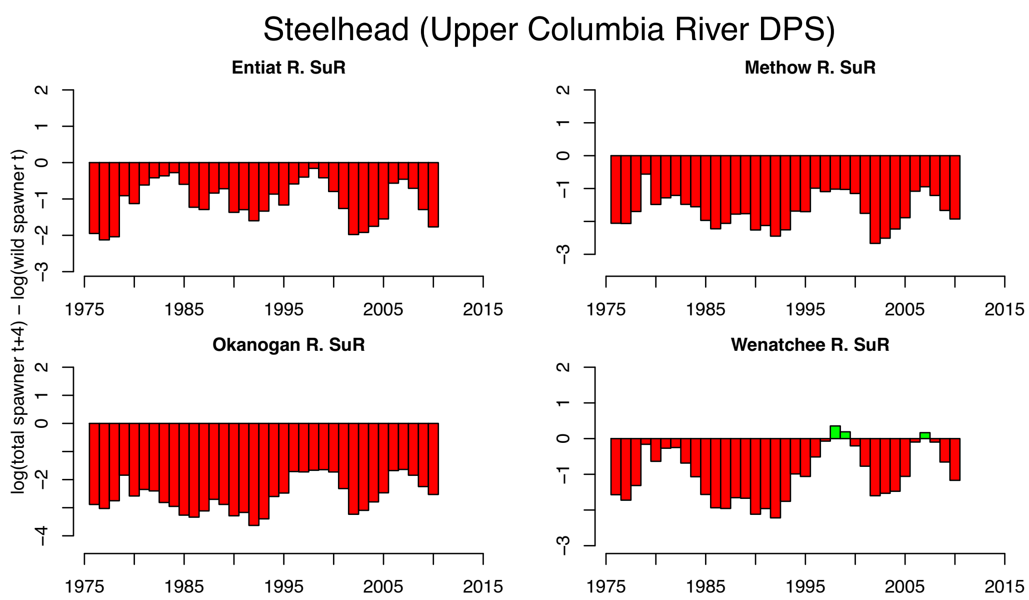


Figure 18 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4). Spawning years on x-axis.

Table 9 --5-year geometric mean of raw natural spawner counts. This is the raw total spawner count times the fraction natural estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values was used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Entiat R. SuR	Up. Columbia/East Slope Cascades	68 (134)	38 (200)	107 (491)	102 (462)	209 (696)	105 (51)
Methow R. SuR	Up. Columbia/East Slope Cascades	274 (1206)	100 (927)	434 (4228)	504 (3463)	841 (3839)	67 (11)
Okanogan R. SuR	Up. Columbia/East Slope Cascades	65 (678)	23 (522)	123 (2163)	144 (1735)	248 (2123)	72 (22)
Wenatchee R. SuR	Up. Columbia/East Slope Cascades	525 (1847)	265 (742)	772 (2318)	678 (1857)	1548 (2767)	128 (49)

Table 10 -- 15-year trends in log wild spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Entiat R. SuR	Up. Columbia/East Slope Cascades	0.04 (-0.02, 0.11)	0.07 (0.02, 0.11)
Methow R. SuR	Up. Columbia/East Slope Cascades	0.06 (-0.01, 0.12)	0.1 (0.06, 0.14)
Okanogan R. SuR	Up. Columbia/East Slope Cascades	0.06 (-0.02, 0.14)	0.1 (0.06, 0.14)
Wenatchee R. SuR	Up. Columbia/East Slope Cascades	0.04 (-0.01, 0.1)	0.07 (0.03, 0.11)

SPATIAL STRUCTURE AND DIVERSITY

With the exception of the Okanogan population, the upper Columbia River populations were rated as low risk for spatial structure. The high risk ratings for diversity are largely driven by high levels of hatchery spawners within natural spawning areas and lack of genetic diversity among the populations. The basic major life history patterns (summer A-run type, tributary and mainstem spawning/rearing patterns, and the presence of resident populations and subpopulations) appear to be present. All of the populations were rated at high risk for current genetic characteristics by the ICTRT. Genetics samples taken in the 1980s indicate little differentiation within populations in the upper Columbia River DPS. More recent studies within the Wenatchee River basin have found differences between samples from the Pashastin River, believed to be relatively isolated from hatchery spawning, and those from other reaches within the Wenatchee. This suggests that there may have been a higher level of within and among population diversity prior to the advent of major hatchery releases (Seamons *et al.* 2012). Genetic studies based on sampling in the Wenatchee as well as other Upper Columbia River steelhead population tributaries are underway and should allow for future analyses of current genetic structure and any impacts of changing hatchery release practices (A. Murdoch, WDFW pers. comm.).

Hatchery-origin returns continue to constitute a high fraction of total spawners in natural spawning areas for this DPS (Table 11). The estimated proportion of natural-origin spawners has increased consistently since the late 1990s for all four populations (Figure 19). Natural-origin proportions were the highest in the Wenatchee River (58%). Although increasing, natural origin proportions in the Methow and Okanogan rivers remained at low levels. There are currently direct releases of hatchery origin juveniles in three of the four populations, the exception being the Entiat River. Based on PIT detections, hatchery origin spawners in the Entiat River include stray hatchery returns from releases into the Wenatchee River (Hillman *et al.* 2015).

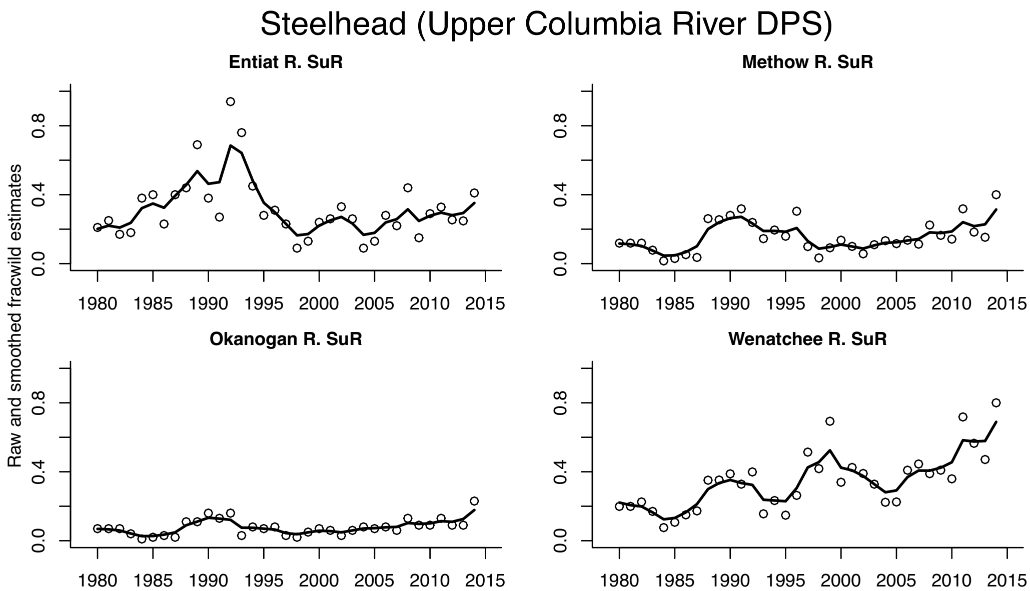


Figure 19 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 11 -- 5-year mean of fraction natural origin (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Entiat R. SuR	0.56	0.21	0.24	0.24	0.31
Methow R. SuR	0.24	0.14	0.11	0.15	0.24
Okanogan R. SuR	0.11	0.05	0.06	0.09	0.13
Wenatchee R. SuR	0.30	0.41	0.34	0.38	0.58

NOAA Fisheries adopted a recovery plan for upper Columbia River spring Chinook salmon and steelhead in 2007 (FR 72 #194, 57303–57307). The plan was developed by the Upper Columbia Salmon Recovery Board (UCSRB) and is available at: http://www.nwr.noaa.gov/Salmon-Recovery-Planning/Recovery-Domains/Interior-Columbia/Upper-Columbia/Upper-Col-Plan.cfm.

Achieving recovery (delisting) of each ESU via sufficient improvement in abundance, productivity, spatial structure, and diversity is the longer term goal of the UC Recovery Plan. The UC Recovery Plan includes specific quantitative criteria expressed relative to population viability curves (ICTRT 2007). It includes two quantitative criteria for assessing the status of the steelhead DPS against the recovery objective: "The 12-year geometric mean (representing approximately three generations) of abundance and productivity of naturally produced steelhead within the Wenatchee, Entiat, and Methow populations must reach a level that would have not less than a 5% extinction-risk (viability) over a 100 year period" and "at a minimum, the Upper Columbia Steelhead DPS will maintain at least 3,000 naturally produced spawners and a spawner:spawner ratio greater than 1:1 distributed among the three populations." The minimum number of naturally produced spawners (expressed as 12-year geometric means) should exceed 1,000 each for the Wenatchee and Methow river populations and 500 each for the Entiat and Okanogan river populations. The plan also established minimum productivity thresholds. These natural spawner abundance criteria replace the interim targets referenced in the 2005 BRT report. The 12-year geometric mean productivity should exceed 1.1 spawners per parent spawner for the two larger populations (Wenatchee and Methow Rivers), and 1.2 for the smaller Entiat River and Okanogan populations.

The ICTRT had recommended that at least two of the four extant populations be targeted for highly viable status (less than 1% risk of extinction over 100 years) because of the relatively low number of extant populations remaining in the ESU. The UC Recovery Plan adopted an alternative approach for addressing the limited number of populations in the ESU—5% or less risk of extinction for at least three of the four extant populations.

The UC Recovery Plan also calls for "… restoring the distribution of naturally produced spring Chinook salmon and steelhead to previously occupied areas where practical, and conserving their genetic and phenotypic diversity." Specific criteria included in the UC Recovery Plan reflect a combination of the criteria recommended by the ICTRT (ICTRT 2007) and an earlier pre-TRT analytical project (Ford *et al*. 2001). The plan incorporates spatial structure criteria specific to each steelhead population. For the Wenatchee River population, the criteria require observed natural spawning in four of the five major spawning areas as well as in at least one of the minor spawning areas downstream of Tumwater Dam. In the Methow River, natural spawning should be observed in three major spawning areas. In each case, the major spawning areas should include a minimum of 5% of the total return to the system or 20 redds, whichever is greater. The Entiat River spring Chinook population includes a single historical major spawning area. The plan incorporates criteria for spatial structure and diversity adopted from the ICTRT viability report. The mean score for the three metrics representing natural rates and spatially mediated processes should result in a moderate or lower risk in each of the three populations and all threats defined as high risk must be addressed. In addition, the mean score for the eight ICTRT metrics tracking natural levels of variation should result in a moderate or lower risk score at the population level.

Table 12 -Viability assessments for extant Upper Columbia Steelhead DPS populations. Natural spawning abundance: most recent 10 year geometric mean (range). ICTRT productivity: 20 year geometric mean for parent escapements below 75% of population threshold. Current abundance and productivity estimates are geometric means. Range in annual abundance, standard error and number of qualifying estimates for productivities in parentheses. Upward arrows: current estimates increased over prior review. Oval: no change, downward arrow indicate estimate has decreased.

	Abundance and productivity metrics				Spatial structure and diversity metrics			
Population	**ICTRT minimum threshold**	**Natural spawning abundance**	**ICTRT productivity**	**Integrated A/P risk**	**Natural processes risk**	**Diversity risk**	**Integrated SS/D risk**	**Overall viability rating**
Wenatchee River 2005–2014	1,000	1,025 ↑ (386-2,235)	1.207 ○ (.021, 3/20)	Low	Low	High	High	**Maintained**
Entiat River 2005–2014	**500**	146 ↑ (59-310)	0.434 ↓ (.22, 12/20)	High	Moderate	High	High	**High risk**
Methow River 2005–2014	1,000	651 ↑ (365-1,105)	0.371 ○ (0.37, 3/20)	High	Low	High	High	**High risk**
Okanogan River 2005–2014	750	189 ↑ (107-310)	0.154 ○ (.275, 6/20)	High	High	High	High	**High risk**

UPDATED BIOLOGICAL RISK SUMMARY

Upper Columbia River steelhead populations have increased relative to the low levels observed in the 1990s, but natural origin abundance and productivity remain well below viability thresholds for three out of the four populations (Table 13). The status of the Wenatchee River steelhead population continued to improve based on the additional years information available for this review. The abundance and productivity viability rating for the Wenatchee River exceeds the minimum threshold for 5% extinction risk. However, the overall DPS status remains unchanged from the prior review, remaining at high risk driven by low abudance and productivity relative to viability objectives and diversity concerns. Application of the criteria for abundance/productivity results in relatively coarse scale ratings for each population. Across Interior Columbia DPSs, the populations differ in the relative changes in survival or limiting capacities that could lead to viable ratings (Figure 20). The required improvement to improve the abundance/productivity estimates for Upper Columbia Steelhead populations is at the high end of the range for all listed Interior populations (Figure 20).

Given the recent changes in hatchery practices in the Wenatchee River and the potential for reduced hatchery contributions or increased spatial separation of hatchery vs. natural origin spawners, it is possible that genetic composition could trend towards patterns consistent with strong natural selection influences in the future. Ongoing genetic sampling and analysis could provide information in the future to determine if the diversity risk is abating. The proportions of hatchery-origin returns in natural spawning areas remain high across the DPS, especially in the Methow and Okanogan river populations. The improvements in natural returns in recent years largely reflect several years of relatively good natural survival in the ocean and tributary habitats. Tributary habitat actions called for in the Upper Columbia Recovery Plan are anticipated to be implemented over the next 25 years and the benefits of some of those actions will require some time to be realized.

Table 13 - Upper Columbia Steelhead DPS Steelhead population viability ratings integrated across the four VSP parameters. Viability key: HV, highly viable; V, viable; M, maintained; and HR, high risk (does not meet viability criteria).

		Spatial structure/diversity risk			
		Very low	Low	Moderate	High
Abundance/ productivity risk	Very low (<1%)	HV	HV	V	M
	Low (1–5%)	V	V	V	M Wenatchee
	Moderate (6–25%)	M	M	M	HR
	High (>25%)	HR	HR	HR	HR Entiat Methow Okanogan

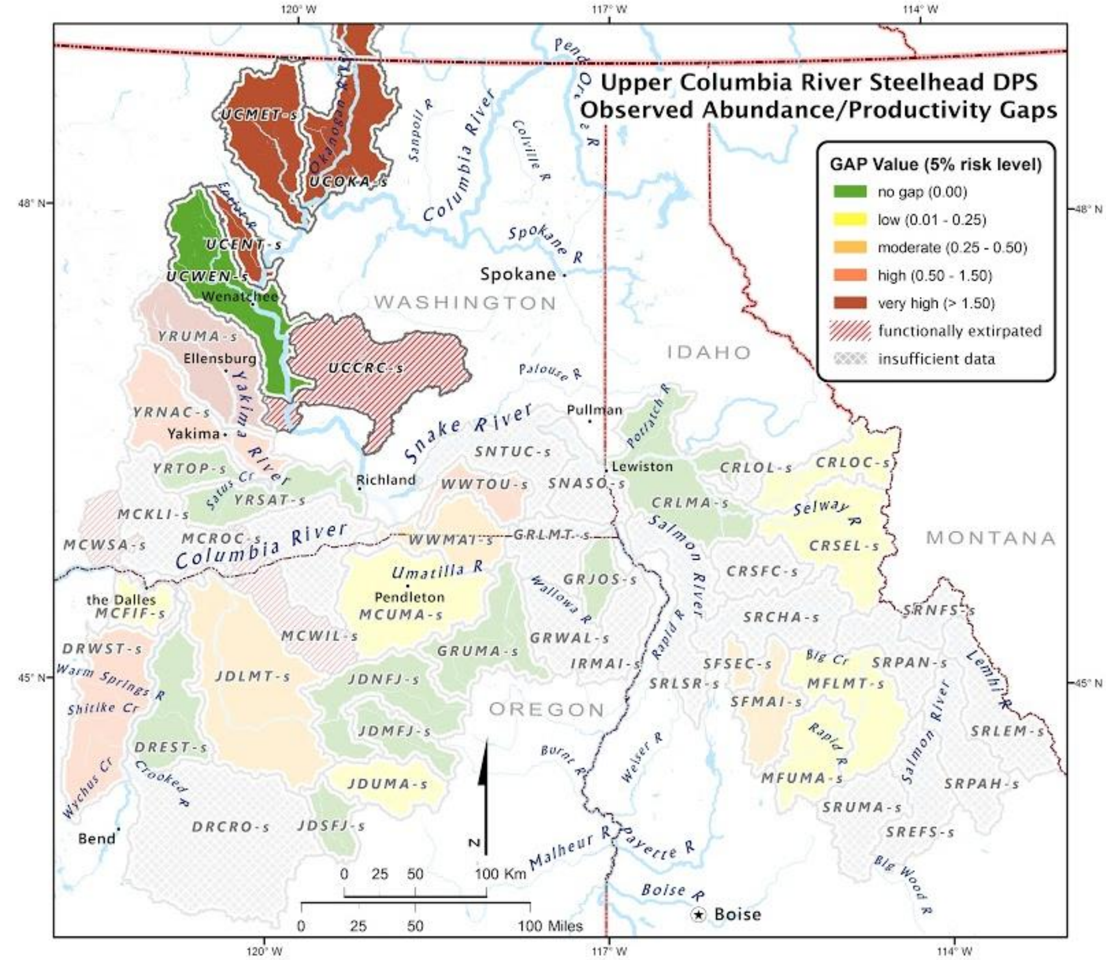


Figure 20 – Upper Columbia steelhead DPS population abundance/productivity gaps. Populations with insufficient data to generate gaps shaded in gray. Gaps are defined as relative improvement in productivity or limiting capacity required for a population to exceed its corresponding 5% risk viability curve (ICTRT, 2007b). Gap estimates for populations in the Mid-Columbia DPS and Snake River DPS provided for comparison (shaded colors).

SNAKE RIVER SPRING/SUMMER-RUN CHINOOK SALMON ESU

BRIEF DESCRIPTION OF ESU

The Snake River Spring-Summer Chinook salmon ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins, as well as fifteen artificial propagation programs (Figure 21). The ESU was first listed under the ESA in 1992, and the listing was reaffirmed in 2005 and 2012.

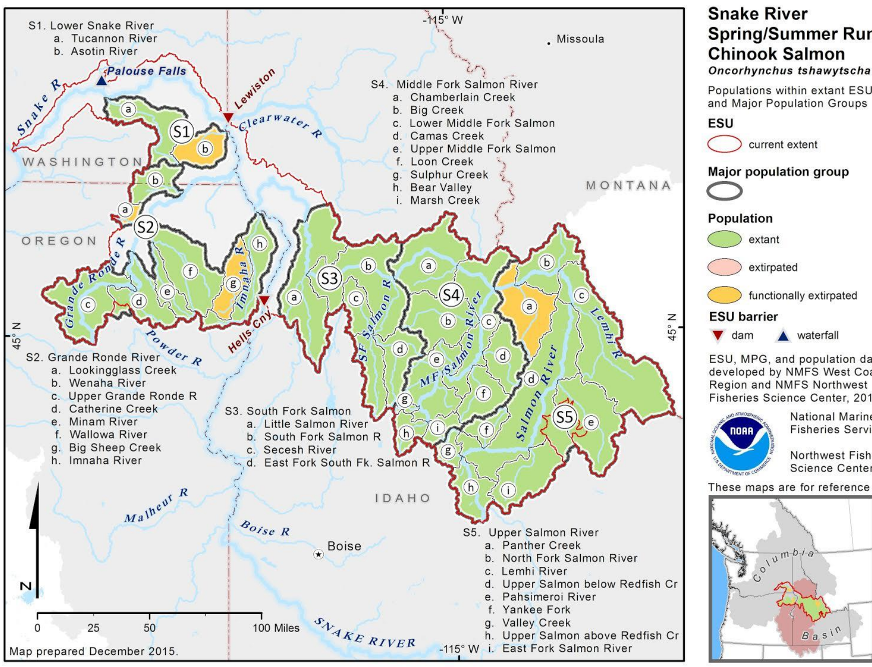


Figure 21 –The Snake River spring/summer-run Chinook salmon ESU' spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The 2005 BRT report evaluated the status of Snake River spring/summer Chinook using data on returns through 2001, with the majority of BRT risk rating points being assigned to the most likely to be endangered category (Good *et al*. 2005). The BRT noted that although there were a number of extant spawning aggregations within this ESU, a substantial number of historical spawning populations have been lost. The most serious risk factor for the ESU was low natural productivity (spawner to spawner return rates) and the associated decline in abundance to extremely low levels relative to historical returns. Large increases in escapement estimates for many (but not all) areas for the 2001 return year were considered encouraging by the BRT. However the BRT also acknowledged that return levels were highly variable and that abundance should be measured over at least an 8 year period and that by this measure the then recent abundance levels across the ESU fall short of interim objectives. The BRT was concerned about the high level of production/mitigation and supplementation hatchery programs across the ESU, noting that these programs represented ongoing risks to natural populations and made it difficult to assess trends in natural productivity and growth rates. The phasing out of the non-native Rapid River-origin hatchery program in the Grande Ronde Basin was viewed as a positive action.

2010

Ford *et al*. (2011) concluded that population level status ratings remained at high risk across all MPGs within the ESU; although natural spawning abundance estimates had increased, all populations remained below minimum natural origin abundance thresholds. Relatively low natural production rates and spawning levels below minimum abundance thresholds remained a major concern across the ESU. The ability of populations to be self-sustaining through normal periods of relatively low ocean survival remained uncertain. Factors cited by the 2005 BRT (Good *et al*. 2005) remained as concerns or key uncertainties for several populations. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the prior BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

The previous BRT review (Ford *et al*. 2011) analyzed spawner abundance data series for most populations in this ESU using expansions from index area redd counts and weir estimates (ICTRT 2010). The current ICTRT data series extends the time period of record through at least the 2013 or 2014 return year for populations across all of the MPGs in the Spring/Summer Chinook ESU. Data and analyses used in this assessment were obtained primarily from state and tribal fisheries agencies. ODFW, WDFW and IDFG updated annual estimates of spawning escapement, hatchery/wild spawner fractions and age composition for most populations, often incorporating data generated by regional projects conducted by the Nez Perce, Umatilla and Shosone Bannock tribal fisheries departments. In several cases the primary source for information on a population was an ongoing tribal sampling program (e.g., the Didson sonar based program in the Secesh River and the mark recapture weir sampling project in Johnson Creek – both conducted by the Nez Perce Tribal Fisheries department). A major advance since the data compilation efforts leading to the 2011 NWFSC status review has been the cooperative efforts of regional fish managers to maintain regionally compatible databases using standardized formats and methods to promote efficiency and access to population level estimates of key status indicators including spawning abundance, hatchery/natural proportions and age structure.

Efforts to refine and document the estimates for individual populations have continued. In most cases, updates to estimated escapements or hatchery/wild spawner proportions for prior years have been relatively minor. Notable additions and changes include incorporation of additional spawner survey and weir count data provided by the Soshone-Bannock Tribal Fisheries Department into population level spawner estimates for the Yankee Fork, updating the data series for the Lemhi River population to account for spawning estimates in the Hayden Creek tributary, and the addition of data series for two additional populations (the Upper and Lower Middle Fork populations). Population level estimates derived from these sources for this assessment are available through the NWFSC Salmon Population Summary database (http://www.nwfsc.noaa.gov/trt/mapsdata.cfm).

Freshwater Production Assessments

Recent analyses of smolt production from Salmon River and Grande Ronde Chinook populations have identified or corroborated relatively strong density dependent growth and mortality effects (Walters *et al.* 2013b; Copeland *et al.* 2014c; ISAB 2015). In addition, new insights into the prevalence and the potential importance of juvenile migration tactics for individual populations have become available (Copeland & Venditti 2009; Copeland *et al.* 2014c). Information from these studies will be discussed in both the Abundance/Productivity and Spatial Structure/Diversity sections.

Relative density had effects on survival during a particular stage (e.g. egg to summer parr) as well as on growth. The effects of density dependence on growth were often correlated with mortality rates during the next life history phase (overwintering). In general, the Snake River studies did not support a density related migration mechanism.

Multiple Population Analyses

The 2009 FCRPS Adaptive Management and Implementation Plan identified a need for more detailed metapopulation analyses that could be used to help identify populations particularly vulnerable to extinction due to isolation as well as to understand commonalities and differences in year to year variations among populations (Fullerton *et al*. 2013; Jorgensen *et al*. 2013). More effort will be needed to develop and implement metapopulation models that can be used to fully accomplish those objectives, but some preliminary insights are available (Fullerton *et al.* 2013). Specifically, results from expanded genetics/dispersal analyses indicate general relationships among populations that would be consistent with ICTRT delineations, however there were some outlier populations or deviations in common patterns. For example, The Lemhi River and the Grande Ronde River are in two MPGs that are geographically separated, but their trends in abundance are more similar than to the other populations assigned to their respective MPGs, in spite of being over 800 stream kms apart. This may be a result of a common response to correlated environmental factors. The populations in the South Fork Salmon MPG showed a high degree of diversity. Two of the populations in this group showed relatively unique patterns in annual trends in abundance that did not correlate well with any other population. Continuing the metapopulation analytical work should, in the future, either further validate or provide a scientific basis for updating the objectives behind the population recovery scenario options recommended by the ICTRT.

Smolt to Adult Return Rates

The ICTRT current productivity metric incorporates an adjustment for annual smolt to adult return rate (SAR) estimates to reduce the impact of short term climate variability (ICTRT 2007). The SAR index used in earlier analyses has been extended using estimates based on the sampling the aggregate natural origin smolt outmigrants and adult returns at Lower Granite Dam. The indices represent cumulative out of basin survivals (downstream passage, ocean life stages, upstream passage including harvest escapement rates). The SAR series derived from estimates of the aggregate wild smolt outmigration and the corresponding adult returns summed by age over the associated return years shows a series of fluctuations that are similar to SAR series for other Columbia Basin ESUs/DPSs (Figure 22). In general, series of relatively high and low years in smolt to return rates were similar among Columbia Basin ESUs/DPSs, although there are some differences. All of the indices showed peaks in SARs for brood years in the early 1980s and the late 2000s, and relatively low survivals in the early 1990s and 2000s.

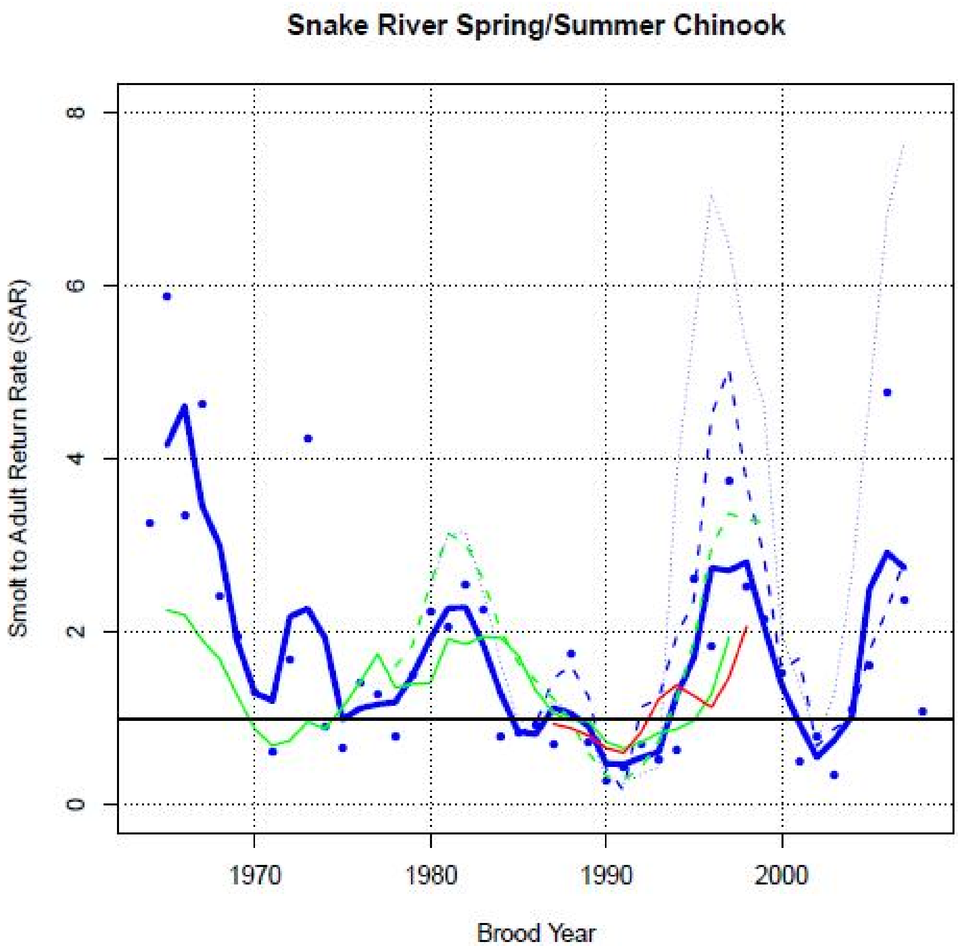


Figure 22 -- Snake River spring/summer Chinook salmon aggregate smolt to adult return rates (blue points and heavy line). Aggregate SARs for other Interior Columbia basin ESUs and DPSs provided for comparison. Snake River aggregate wild steelhead run (solid green), Upper Columbia spring Chinook (blue dashed line), Upper Columbia steelhead (green dashed line) and, Mid-Columbia steelhead (red line). Each SAR series is rescaled by dividing annual values by the corresponding series mean to faclilitate relative comparison.

Ocean Condition Indices

Snake River spring/summer Chinook salmon are part of the Columbia River upriver yearling dominated Chinook run that is believed to occupy mid-shelf waters during the early ocean life history phase (see Environmental Trends section below). The summer components of this ESU are more similar in ocean distribution to the spring Chinook runs than to the runs from the non-listed summer/fall ESU in the mid/upper Columbia. Aggregate annual returns of Columbia River spring Chinook are correlated with a range of ocean condition indices including measures of broad scale physical conditions, local biological indicators, and local physical factors (Peterson *et al*. 2014). Several indicators, either individually or in combination, correlate well with spring Chinook adult returns with a lag of 1 to 2 years. However, for each specific indicator or combination, there are anomalous years that fall outside of the apparent relationships. Work is continuing to further understand the relationships among physical and biological 'drivers' and annual levels of ocean survival for salmonid species in the ocean environment (Peterson *et al*. 2014). After accounting for age at return vs. ocean entry, the annual pattern in the aggregate Snake River Chinook salmon ESU SAR index (Figure 22) generally corresponds to the composite rankings across ocean indicators available for early ocean years starting in the late 1990s (Peterson *et al*. 2014).

Genetic Diversity

Results from two studies of patterns in genetic diversity within and among populations in the Snake spring/summer Chinook salmon ESU have recently been published. Van Doornik *et al*. (2011) analyzed genetic samples from some Salmon River populations, and reported no evidence for significant introgression of hatchery stocks. The study reported on results of analyzing recent samples from locations in Grande Ronde River basin populations that have been subject to past supplementation efforts involving an outside stock (Rapid River). The study was designed to determine if the genetic profiles of naturally produced Chinook salmon juveniles showed evidence of introgression. Samples from four of the populations (Minam River, Wenaha River, Lostine River and the Imnaha River) indicated that within and among population diversity retained their distinctions from the out of basin stocks used to supply prior releases. There were indications of some low level introgression from Rapid River stock in the Wenaha and Minam River samples. Lookingglass Creek and Upper Grande Ronde samples indicated substantial influence of the Rapid River stock. The results for Lookingglass Creek reflect the virtual replacement of the original run by the large scale hatchery program. Van Doornik *et al*. (2013) speculate that the strong Rapid River genetic signal in the Upper Grande Ronde River samples may reflect a combination of factors, including the relatively poor productivity of the natal run under current habitat and environmental conditions along with a greater similarity in habitat characteristics with the areas in which the Rapid River stock originated.

ABUNDANCE AND PRODUCTIVITY

Updated data series on spawner abundance, age structure and hatchery/natural proportions were used to generate current assessments of abundance and productivity at the population level. Evaluations were done using both a set of metrics corresponding to those used in prior Biological Review Team (BRT) reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The BRT level metrics were done consistently across all ESUs and DPSs to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section below. The ICTRT abundance and productivity metrics are measured over longer time frames to dampen the effects of annual variations and they use annual natural origin age composition to calculate brood year recruitment when sampling levels meet agency criteria.

Estimates of the annual abundance of natural origin spawners within each of 26 Snake River Spring Summer Chinook ESU populations are summarized in five year increments Table 14 and are illustrated in Figure 23. Five years reflects the 5 year brood cycles typical of Interior Columbia spring/summer Chinook salmon populations. The most recent five year geometric mean abundance estimates for 25 out of the 26 populations are higher than the corresponding estimates for the previous five year period by varying degrees, the estimate for the 26th population was for no change from a very low abundance in the prior five year period. The recent five year abundance levels for 17 of the populations were more than twice the estimates for the previous five year period. Four of the five populations with the highest relative increases were populations with significant levels of direct hatchery supplementation (Yankee Fork, Catherine Creek, Upper Grande Ronde River and East Fork of the South Fork). Marsh Creek and the Lemhi River had the highest relative increases among populations that were not supplemented by hatchery production. The level of increase for the other populations exhibiting a positive change ranged from 34% to 81%.

Short-term (15 year) population trends in total spawner abundance were positive over the period 1999 to 2014 for 23 of the 26 population natural origin abundance series, but the relative rates of increase for each population were lower than estimates of trend for the prior review period (Table 15). Trends for most populations in the Middle Fork and Upper Salmon MPGS are strongly positive. Two populations in the Middle Fork MPG (Marsh Creek and Loon Creek) along with one (Lemhi River) in the Upper Salmon MPG had relatively flat trends in total abundance since 1995. Short-term trends in total abundance for the South Fork MPG were also positive but at lower levels than in the Middle Fork and Upper Salmon MPGs, with the exception of the relatively strong trend in the East Fork South Fork population. In the Grande Ronde MPG, three of the populations exhibited moderately positive trends, and the remaining three had relatively flat or slightly negative trajectories in total spawning abundance since 1995. The most recent 15 year trend estimate for the single extant population in the Lower Snake MPG, the Tucannon River, had a similar positive trend as in the prior review. The trend in natural origin spawners for three populations were flat (Lower Middle Fork Mainstem) or slightly negative (Camas and Loon Creeks). One population (the Lower Middle Fork Salmon River Mainstem) declined at a rate of 9% per year over the period.

It is important to put the recent average abundance and trend estimates in a longer term context. The short term trends described in this report allow for a detailed assessment of the performance of populations since the steep declines and extremely low spawner levels observed from the early 1970s through the 1990s. Estimates of population level escapement for many of these populations are available going back into the 1950's and 1960s, as are dam counts representing the aggregate returns from all Snake River populations (e.g., Ice Harbor counts beginning in 1962). The historical population specific spawner estimates and the dam count aggregate return estimates all indicate that returns in the late 1950s and early 1960s were generally higher than recent returns, in most cases by a substantial amount (Ford *et al.* 2011).

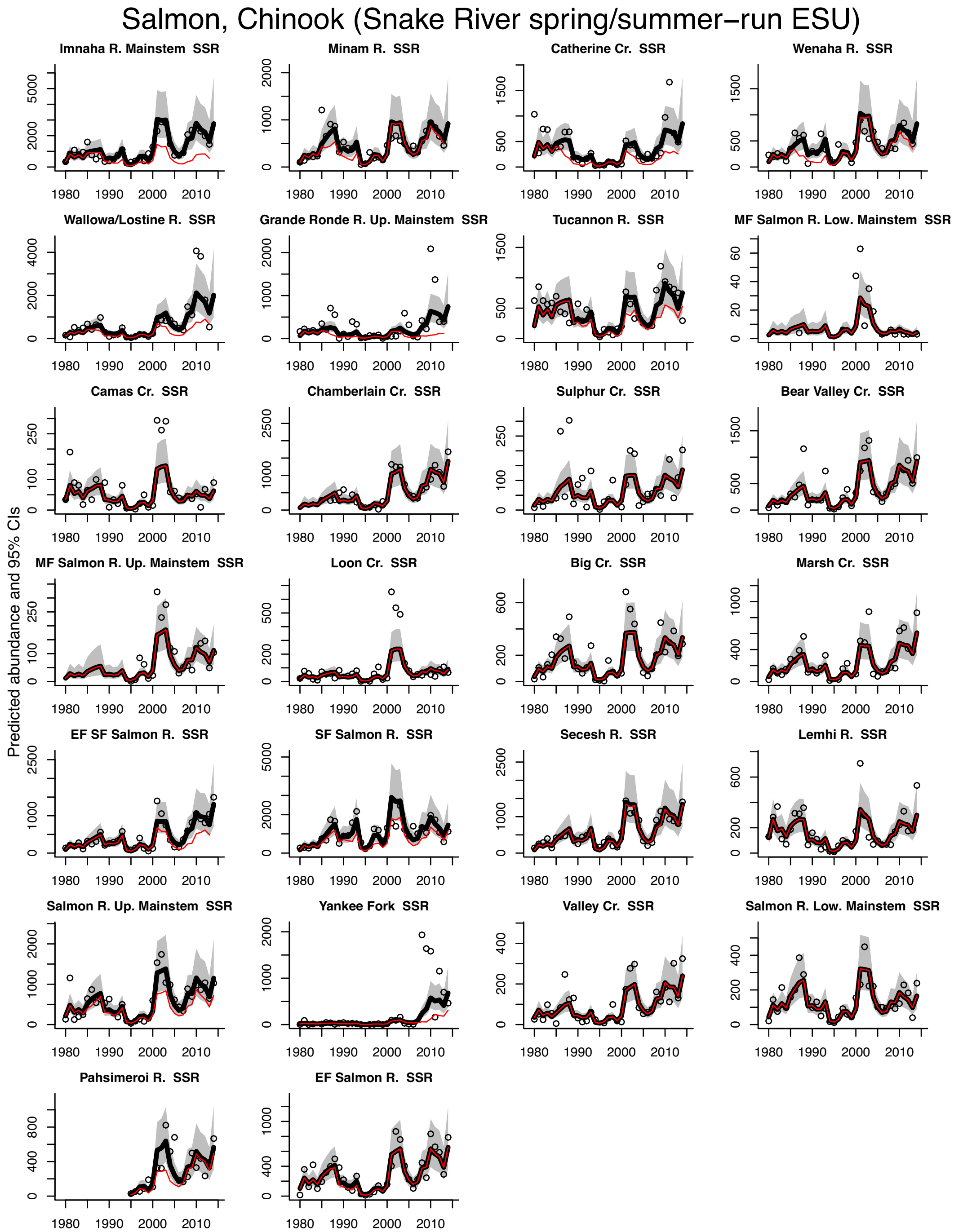


Figure 23 – Smoothed trend in estimated total (thick black line) and natural origin (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

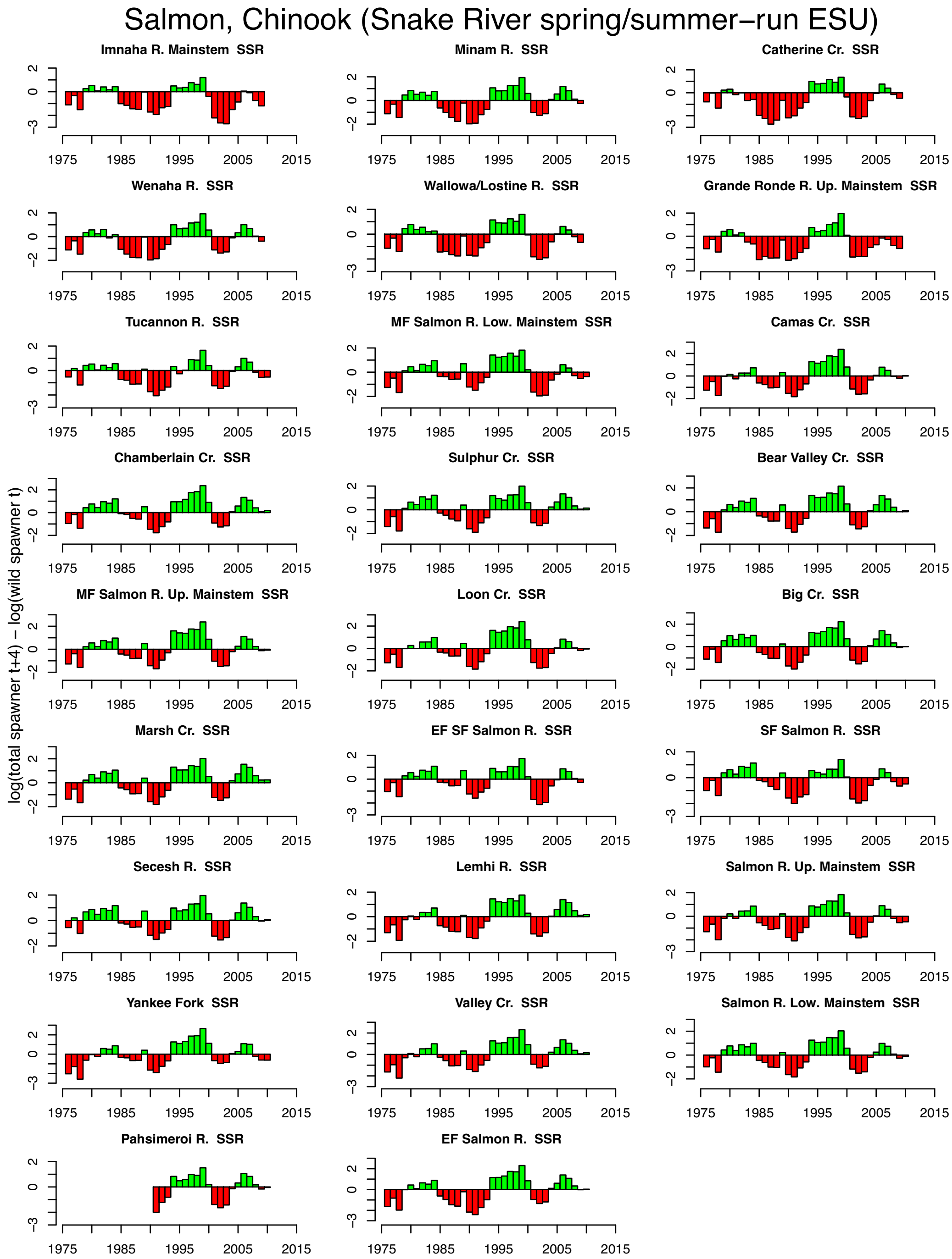


Figure 24 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).). Spawning years on x axis.

Table 14 -- 5-year geometric mean of raw natural origin spawner counts. This is the raw total spawner count times the fraction natural origin estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural origin spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Imnaha R. Mainstem SSR	Grande Ronde/Imnaha	218 (529)	231 (452)	899 (2032)	264 (1196)	699 (2041)	165 (71)
Minam R. SSR	Grande Ronde/Imnaha	110 (284)	162 (166)	541 (552)	449 (460)	619 (698)	38 (52)
Catherine Cr. SSR	Grande Ronde/Imnaha	27 (102)	56 (56)	126 (259)	70 (205)	368 (852)	426 (316)
Wenaha R. SSR	Grande Ronde/Imnaha	71 (305)	164 (186)	612 (638)	354 (364)	488 (643)	38 (77)
Wallowa/Lostine R. SSR	Grande Ronde/Imnaha	82 (159)	101 (104)	317 (619)	246 (729)	809 (1962)	229 (169)
Grande Ronde R. Up. Mainstem SSR	Grande Ronde/Imnaha	33 (96)	31 (32)	55 (105)	26 (141)	114 (816)	338 (479)
Tucannon R. SSR	Low. Snake	230 (314)	34 (84)	226 (398)	273 (400)	409 (678)	50 (70)
MF Salmon R. Low. Mainstem SSR	MF Salmon R.			28 (28)	4 (4)	4 (4)	0 (0)
Camas Cr. SSR	MF Salmon R.	20 (20)	13 (13)	115 (115)	43 (43)	42 (42)	-2 (-2)
Chamberlain Cr. SSR	MF Salmon R.	286 (286)	85 (85)	1107 (1107)	470 (470)	1074 (1074)	129 (129)
Sulphur Cr. SSR	MF Salmon R.	59 (59)	21 (21)	55 (55)	49 (49)	112 (112)	129 (129)
Bear Valley Cr. SSR	MF Salmon R.	177 (177)	95 (95)	662 (662)	319 (319)	776 (776)	143 (143)
MF Salmon R. Up. Mainstem SSR	MF Salmon R.		13 (13)	140 (140)	52 (52)	104 (104)	100 (100)
Loon Cr. SSR	MF Salmon R.	25 (25)	21 (21)	225 (225)	54 (54)	65 (65)	20 (20)
Big Cr. SSR	MF Salmon R.	76 (76)	29 (29)	302 (302)	121 (121)	270 (270)	123 (123)
Marsh Cr. SSR	MF Salmon R.	102 (102)	99 (99)	285 (286)	126 (126)	564 (564)	348 (348)
EF SF Salmon R. SSR	SF Salmon R.	273 (284)	125 (127)	392 (545)	139 (339)	575 (1041)	314 (207)
SF Salmon R. SSR	SF Salmon R.	690 (1089)	344 (602)	968 (1540)	626 (1124)	923 (1194)	47 (6)
Secesh R. SSR	SF Salmon R.	338 (348)	212 (227)	951 (978)	434 (458)	994 (1014)	129 (121)
Lemhi R. SSR	Up. Salmon R.	51 (51)	51 (51)	198 (198)	86 (86)	262 (262)	205 (205)
Salmon R. Up. Mainstem SSR	Up. Salmon R.	227 (275)	67 (85)	675 (1104)	327 (564)	624 (897)	91 (59)
Yankee Fork SSR	Up. Salmon R.	16 (16)	6 (6)	60 (60)	25 (120)	169 (623)	576 (419)
Valley Cr. SSR	Up. Salmon R.	26 (26)	26 (26)	109 (109)	85 (85)	192 (192)	126 (126)
Salmon R. Low. Mainstem SSR	Up. Salmon R.	63 (63)	41 (41)	239 (239)	99 (99)	137 (137)	38 (38)
Pahsimeroi R. SSR	Up. Salmon R.		45 (67)	172 (343)	226 (298)	360 (388)	59 (30)
EF Salmon R. SSR	Up. Salmon R.	68 (107)	34 (46)	442 (442)	224 (224)	594 (594)	165 (165)

Table 15 -- 15-year trends in log natural origin spawner abundance computed from a linear regression applied to the smoothed natural origin spawner log abundance estimate. Only populations with at least 4 natural origin spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Imnaha R. Mainstem SSR	Grande Ronde/Imnaha	0.1 (0.01, 0.18)	0.02 (-0.06, 0.1)
Minam R. SSR	Grande Ronde/Imnaha	0.12 (0.03, 0.2)	0.05 (-0.02, 0.11)
Catherine Cr. SSR	Grande Ronde/Imnaha	0.09 (0.01, 0.17)	0.08 (0, 0.16)
Wenaha R. SSR	Grande Ronde/Imnaha	0.17 (0.08, 0.25)	0.02 (-0.05, 0.09)
Wallowa/Lostine R. SSR	Grande Ronde/Imnaha	0.1 (0.02, 0.18)	0.1 (0.02, 0.17)
Grande Ronde R. Up. Mainstem SSR	Grande Ronde/Imnaha	0.07 (-0.02, 0.16)	0.05 (-0.03, 0.13)
Tucannon R. SSR	Low. Snake	0.04 (-0.07, 0.14)	0.1 (0.03, 0.18)
MF Salmon R. Low. Mainstem SSR	MF Salmon R.		-0.09 (-0.16, -0.02)
Camas Cr. SSR	MF Salmon R.	0.11 (0, 0.22)	-0.01 (-0.08, 0.07)
Chamberlain Cr. SSR	MF Salmon R.	0.1 (0, 0.21)	0.08 (0.01, 0.15)
Sulphur Cr. SSR	MF Salmon R.	0.06 (-0.04, 0.16)	0.07 (0, 0.13)
Bear Valley Cr. SSR	MF Salmon R.	0.11 (0.01, 0.21)	0.06 (-0.01, 0.13)
MF Salmon R. Up. Mainstem SSR	MF Salmon R.		0.03 (-0.04, 0.11)
Loon Cr. SSR	MF Salmon R.	0.14 (0.03, 0.25)	-0.01 (-0.09, 0.07)
Big Cr. SSR	MF Salmon R.	0.1 (-0.01, 0.21)	0.06 (-0.02, 0.13)
Marsh Cr. SSR	MF Salmon R.	0.08 (-0.02, 0.18)	0.08 (0.01, 0.15)
EF SF Salmon R. SSR	SF Salmon R.	0.04 (-0.05, 0.12)	0.06 (-0.02, 0.14)
SF Salmon R. SSR	SF Salmon R.	0.06 (-0.03, 0.15)	0.02 (-0.05, 0.09)
Secesh R. SSR	SF Salmon R.	0.08 (-0.01, 0.17)	0.05 (-0.02, 0.12)
Lemhi R. SSR	Up. Salmon R.	0.08 (-0.01, 0.18)	0.04 (-0.03, 0.11)
Salmon R. Up. Mainstem SSR	Up. Salmon R.	0.08 (-0.02, 0.18)	0.06 (-0.01, 0.13)
Yankee Fork SSR	Up. Salmon R.	0.13 (0.02, 0.25)	0.16 (0.08, 0.23)
Valley Cr. SSR	Up. Salmon R.	0.12 (0.02, 0.22)	0.09 (0.02, 0.15)
Salmon R. Low. Mainstem SSR	Up. Salmon R.	0.08 (-0.02, 0.18)	0 (-0.06, 0.07)
Pahsimeroi R. SSR	Up. Salmon R.		0.11 (0.05, 0.17)
EF Salmon R. SSR	Up. Salmon R.	0.15 (0.03, 0.26)	0.07 (0, 0.15)

Harvest impacts on the spring component of this ESU are essentially the same as those on the Upper Columbia River (Figure 25). Harvest occurs in the lower portion of the mainstem Columbia River. Mainstem Columbia River fisheries represent the majority of harvest impacts on this ESU. In some years additional harvest in the Snake River basin on specific populations within the ESU occurs. Estimates of total exploitation rate including the Snake River basin components are included in the SPS database. Snake River summer Chinook share the ocean distribution patterns of the upper basin spring runs and are only subject to significant harvest in the mainstem Columbia River. The increases in recent years have resulted from increased allowable harvest rates under the abundance driven sliding scale harvest rate strategy guiding annual management in response to continued large returns of hatchery spring Chinook to the Columbia River basin. Harvest of summer Chinook has been more constrained than that of spring Chinook with consequently lower exploitation rates on the summer component of this ESU. However, the overall pattern of exploitation rates calculated by the TAC is nearly identical to that of the Upper Columbia River spring Chinook.

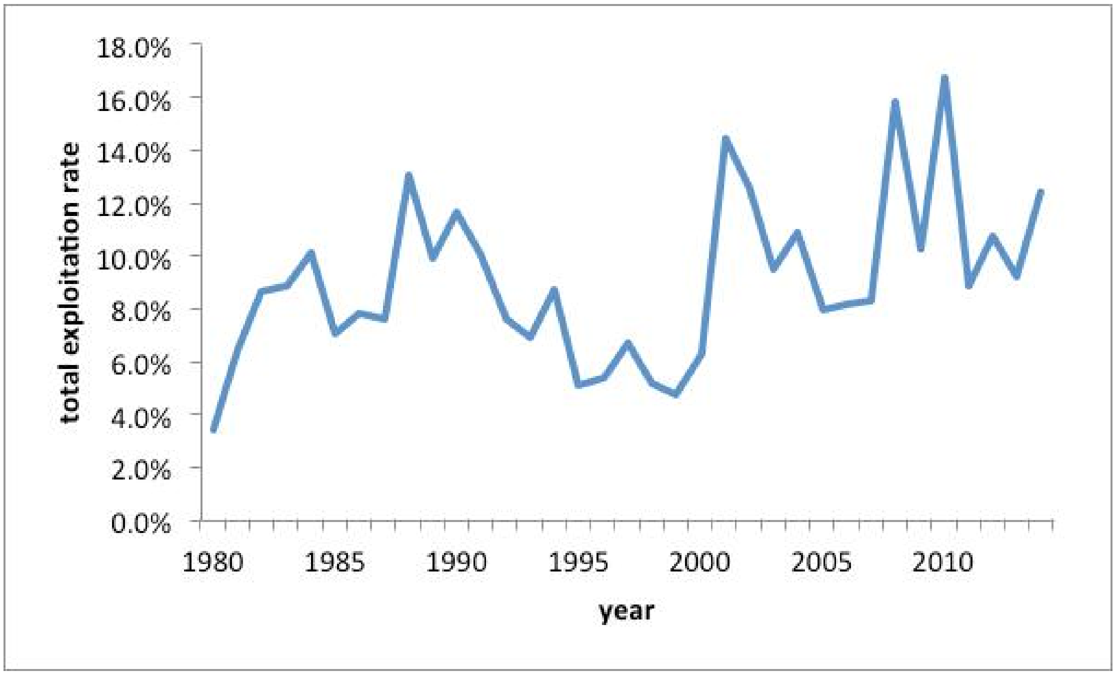


Figure 25 -- Total exploitation rates for Snake River spring/summer Chinook salmon in the mainstem Columbia River fisheries. Data from the Columbia River Technical Advisory Team (TAC 2015).

SPATIAL STRUCTURE AND DIVERSITY

Current estimates of spatial structure and diversity ratings for Snake River Spring/Summer Chinook populations are summarized in Table 17. The ICTRT ratings for spatial structure remain unchanged. Most population abundance estimates are based on redd or weir counts conducted across reaches within or across major spawning areas. Recent survey results are consistent with records for the years analyzed by the ICTRT.

The proportion of hatchery origin spawners within populations varies considerably across MPGs (Figure 26, Table 16). All five extant populations in the Grande Ronde River basin had relatively high hatchery spawner proportions in the 1990s, reflecting the large scale use of out of basin stock (Rapid River) in local releases during that period. Managers transitioned the release programs to incorporate local natural origin brood stock in the mid 1990s. Currently five of the six extant natural population tributaries as well as Lookingglass Creek (with an extripated natal population) have targeted hatchery releases. During that transition, returning hatchery origin fish from the Rapid River releases were actively removed prior to spawning. Returns from natural origin broodstock increased as the specific in-basin programs reached their smolt production objectives. The current local broodstock based hatchery programs in three of the basins are designed to supplement natural spawning while contributing to meeting mitigation objectives (http://www.dfw.state.or.us/fish/HGMP/final.asp#3). Releases into Lookingglass Creek, an extirpated population, are a conventional segregated program. The historical Lookingglass Creek run is believed to have been extirpated as a result of the out of basin hatchery program. The current program uses broodstock that originated from Catherine Creek. The Minam and Wenaha River populations do not have direct supplementation programs. The Imnaha River, an adjacent river basin to the Grande Ronde, is also in this MPG, has an ongoing integrated hatchery program that incorporates natural origin broodstock.

The single current extant population in the Lower Snake River MPG, the Tucannon River, has an ongoing supplementation program, and hatchery returns have constituted about a third of spawning in natural areas in recent years. Mark recapture estimates compared to redd count and carcass recoveries indicate that prespawn mortalities in the Tucannon River have been relatively high in recent years. Efforts are underway to further quantify and to identify potential direct causes (Bumgarner & Dedloff 2015). Hatchery proportions for populations in the Middle Fork Salmon MPG are based on carcass recoveries and remain very low, indicating negligible straying rates as there are no direct release programs in this river basin.

Three of the four South Fork Salmon MPG populations have ongoing hatchery programs. Hatchery proportions for the two of the three populations in the South Fork Salmon River with active hatchery programs decreased marginally in the most recent five year update. The Secesh River continues to show low hatchery proportions reflecting some straying from the programs in the adjacent populations. The ICTRT included a fourth population in the neighboring Little Salmon River drainage in this MPG. This population includes returns from large scale hatchery releases although some of its side tributary spawning areas likely have low hatchery contributions. Direct estimates of natural origin spawners for this population are limited to weir passage counts for the Rapid River tributary.

In the Upper Salmon River MPG, four of the seven populations with sufficient information to directly estimate hatchery contributions had very low hatchery proportions (Lemhi River, East Fork Salmon River, Valley Creek and the Lower Mainstem Salmon River). The most recent five year mean for the Pahsimeroi River was also relatively low. This system is part of the Idaho Supplementation Study and has undergone substantial variation in directed supplementation over recent brood cycles. Two of the other populations in this MPG are the subject of active hatchery release programs as reflected in their respective average spawner proportions. Hatchery contributions to spawning in the bulk of the habitat used by the Upper Salmon River population are regulated by managing passage at Sawtooth weir, located on the mainstem Salmon River near the downstream extent of spawning. Hatchery proportions within the Yankee Fork population have increased substantially in recent years, reflecting returns from a large scale supplementation effort conducted by the Shosonne Bannock tribal fisheries department. In some recent years the program has augmented ongoing smolt releases with adult plants using surplus returns from the Sawtooth Hatchery program in the Upper Salmon River (Gregory & Wood 2013; Denny & Blackadar 2015).

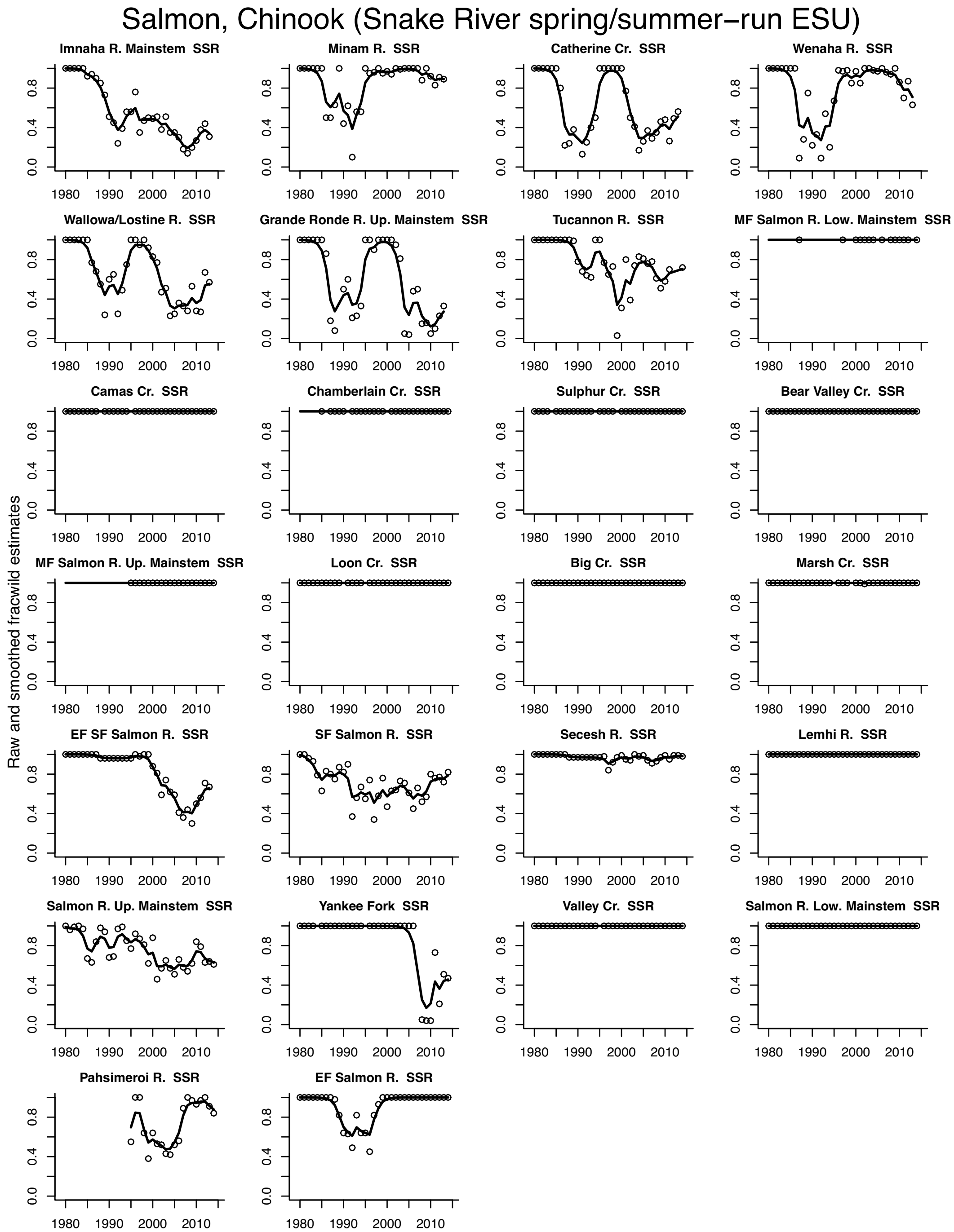


Figure 26 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 16 -- 5-year mean of fraction natural origin spawners (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Imnaha R. Mainstem SSR	0.43	0.53	0.45	0.23	0.35
Minam R. SSR	0.46	0.97	0.98	0.98	0.89
Catherine Cr. SSR	0.32	1.00	0.57	0.35	0.45
Wenaha R. SSR	0.28	0.89	0.96	0.97	0.76
Wallowa/Lostine R. SSR	0.55	0.97	0.56	0.35	0.45
Grande Ronde R. Up. Mainstem SSR	0.37	0.98	0.76	0.27	0.18
Tucannon R. SSR	0.74	0.64	0.61	0.69	0.67
MF Salmon R. Low. Mainstem SSR		1.00	1.00	1.00	1.00
Camas Cr. SSR	1.00	1.00	1.00	1.00	1.00
Chamberlain Cr. SSR	1.00	1.00	1.00	1.00	1.00
Sulphur Cr. SSR	1.00	1.00	1.00	1.00	1.00
Bear Valley Cr. SSR	1.00	1.00	1.00	1.00	1.00
MF Salmon R. Up. Mainstem SSR		1.00	1.00	1.00	1.00
Loon Cr. SSR	1.00	1.00	1.00	1.00	1.00
Big Cr. SSR	1.00	1.00	1.00	1.00	1.00
Marsh Cr. SSR	1.00	1.00	1.00	1.00	1.00
EF SF Salmon R. SSR	0.96	0.99	0.73	0.42	0.61
SF Salmon R. SSR	0.66	0.59	0.64	0.56	0.77
Secesh R. SSR	0.97	0.94	0.97	0.95	0.98
Lemhi R. SSR	1.00	1.00	1.00	1.00	1.00
Salmon R. Up. Mainstem SSR	0.84	0.80	0.63	0.58	0.70
Yankee Fork SSR	1.00	1.00	1.00	0.52	0.39
Valley Cr. SSR	1.00	1.00	1.00	1.00	1.00
Salmon R. Low. Mainstem SSR	1.00	1.00	1.00	1.00	1.00
Pahsimeroi R. SSR		0.71	0.51	0.79	0.93
EF Salmon R. SSR	0.64	0.77	1.00	1.00	1.00

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The ICTRT identified 27 extant and 4 extirpated populations of Snake River Spring/Summer Chinook that historically used the accessible tributary and upper mainstem habitats within the Snake River drainages (ICTRT 2003). The populations are aggregated into five extant Major Population Groupings (MPGs) based on genetic, environmental and life history characteristics. The Lower Snake River MPG includes the Tucannon River and Asotin Creek (extirpated) populations. The Grande Ronde/Imnaha River MPG includes six populations within the Grande Ronde River drainage and two in the Imnaha River. Three populations within the South Fork Salmon River drainage and a fourth in the Little Salmon River form an additional MPG. Chamberlain Creek along with six populations in the Middle Fork drainage constitute the next upstream MPG. The Upper Salmon River MPG includes several major tributary populations along with two mainstem sections also classified as independent populations.

NOAA Fisheries has initiated recovery planning for the Snake River drainage, organized around a subset of management unit plans corresponding to State boundaries. A tributary recovery plan for one of the major management units, the Lower Snake River tributaries within Washington state boundaries, was developed under the auspices of the Lower Snake River Recovery Board and was accepted by NOAA Fisheries in 2005. The LSRB Plan provides recovery criteria, targets and tributary habitat action plans for the two populations of Spring/Summer Chinook in the Lower Snake MPG in addition to the Touchet River (Mid-Columbia Steelhead DPS) and the Washington sections of the Grande Ronde River. Planning efforts are underway for the Oregon and Idaho drainages. Viability criteria recommended by the ICTRT are being used in formulating recovery objectives within each of the management unit planning efforts.

TRT and Recovery Plan Criteria

The recovery plans being synthesized and developed by NOAA Fisheries will incorporate viability criteria recommended by the ICTRT (ICTRT 2007a, b). The ICTRT recovery criteria are hierarchical in nature, with ESU/DPS level criteria being based on the status of natural origin Chinook salmon assessed at the population level. A detailed description of the ICTRT viability criteria and their derivation (ICTRT 2007a) can be found at www.nwfsc.noaa.gov/trt/col/trt_viability.cfm. Under the ICTRT approach, population level assessments are based on a set of metrics designed to evaluate risk across the four viable salmonid population (VSP) elements – abundance, productivity, spatial structure and diversity (McElhany *et al.* 2000). The ICTRT approach calls for comparing estimates of current natural origin abundance (measured as a 10 year geometric mean of natural origin spawners) and productivity (estimate of return per spawner at low to moderate parent spawning abundance) against predefined viability curves. In addition, the ICTRT developed a set of specific criteria (metrics and example risk thresholds) for assessing the spatial structure and diversity risks based on current information representing each specific population. The ICTRT viability criteria are generally expressed relative to particular risk threshold - low risk is defined as less than a 5% risk of extinction over a 100 year period and very low risk as less than a 1% probability over the same time period.

Snake River Spring/Summer Chinook: ICTRT Example Recovery Scenarios

The ICTRT recommends that each extant MPG should include viable populations totaling at least half of the populations historically present, with all major life history groups represented. In addition, the viable populations within an MPG should include proportional representation of large and very large populations historically present. The ICTRT also recommended that at least one population in a viable MPG should meet criteria for Highly Viable (e.g., 1% risk or less). Within any particular MPG, there may be several specific combinations of populations that could satisfy the ICTRT criteria. The ICTRT identified example scenarios that would satisfy the criteria for all extant MPGs (ICTRT 2005). In each case the remaining populations in an MPG should be at or above maintained status.

Lower Snake River MPG: This MPG historically contained two populations, and one, Asotin Creek, is currently considered extirpated. The ICTRT basic criteria would call for both populations being restored to viable status. The ICTRT recommended that recovery planners should give priority to restoring the Tucannon River to highly viable status, and evaluate the potential for reintroducing production in Asotin Creek as recovery planning progresses.

Grande Ronde MPG: This MGP had eight historical populations, two of which are currently considered functionally extirpated. The basic ICTRT criteria call for a minimum of 4 populations at viable or highly viable status. The potential scenario identified by the ICTRT would include viable populations in the Imnaha River (run timing), the Lostine/Wallowa River (large size) and at least one from each of the following pairs: Catherine Creek or Upper Grande Ronde (large size populations); and Minam River or Wenaha River.

South Fork MPG: Two of the four historical populations in this MPG should be restored to viable or highly viable status. The ICTRT recommends that the populations in the South Fork drainages should be given priority relative to meeting MPG viability objectives given the relatively small size and the high level of potential hatchery integration for the Little Salmon River population.

Middle Fork MPG: The ICTRT criteria call for at least five of the nine populations in this MPG to be rated as viable, with at least one demonstrating highly viable status. The ICTRT example recovery scenario included Chamberlain Creek (geographic position), Big Creek (large size category), Bear Valley Creek, Marsh Creek, and either Loon Creek or Camas Creek.

Upper Salmon MPG: This MPG included nine historical populations one of which, Panther Creek, is considered functionally extirpated. The ICTRT example recovery scenario for this MPG includes the Pahsimeroi River (summer Chinook life history); the Lemhi River and Upper Salmon Mainstem (very large size category); East Fork Salmon River (large size category) and Valley Creek.

Table 17 - Snake River spring/summer Chinook populations. Updated status summary vs. draft recovery plan viability objectives. ↑ = improved since prior review. ↓ = Decreased since prior review. ○ = no change. Shaded populations are the most likely combinations within each MPG to be improved to viable status. Current abundance and productivity estimates expressed as geometric means (standard error). .

Population	Abundance/Productivity Metrics				Spatial Structure and Diversity Metrics			Overall Viability Rating
	ICTRT Minimum Threshold	*Natural Spawning Abundance*	*ICTRT Productivity*	*Integrated A/P Risk*	*Natural Processes Risk*	*Diversity Risk*	*Integrated SS/D Risk*	
Lower Snake River MPG								
Tucannon River	750	↑ 267 (.19	↓ .69 (.23)	**High**	Low	Moderate	**Moderate**	High
Asotin Creek	500	*extirpated*						*Extirpated*
Grande Ronde/Imnaha MPG								
Wenaha River	750	↓ 399 (.12)	↑ .93 (.21)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Lostine/Wallowa R.	1,000	↑ 332 (.24)	↑ .98 (.12)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Lookingglass R. (ext)	500	*extirpated*						***extirpated***
Minam R.	750	↑ 475 (.12)	↑ .94 (18)	**High(M)**	Low	Moderate	**Moderate**	**HIGH RISK**
Catherine Creek	1,000	↑ 110 (.31)	↑ .95 (.15)	**High**	Moderate	Moderate	**Moderate**	**HIGH RISK**
Upper Gr. Ronde R.	1,000	↑ 43 (.26)	↑ .59 (.28)	**High**	High	Moderate	**High**	**HIGH RISK**
Imnaha River	750	↑ 328 (.21)	↑ 1.20 (.09)	**High (M)**	Low	Moderate	**Moderate**	**HIGH RISK**
South Fork MPG								
South Fork Mainstem	1,000	↑ 791 (.18)	↓ 1.21 (.20)	**High (M)**	Low	Moderate	**Moderate**	**HIGH RISK**
Secesh River	750	↑ 472 (.18)	○ 1.25 (.20	**High(M)**	Low	Low	**Low**	**HIGH RISK**
East F,/Johnson Cr.	1,000	↑ 208 (.24)	↓ 1.15 (.20)	**High**	Low	Low	**Low**	**HIGH RISK**
Little Salmon River	750	*Insf. data*			Low	Low	**Low**	**HIGH RISK**
Middle Fork MPG								
Chamberlain Creek	750	↑ 641 (.17)	↓ 2.26 (.45)	**Moderate**	Low	Low	**Low**	**Maintained**
Big Creek	1,000	↑ 164 (.23)	↓ 1.10 (.21)	**High**	Very Low	Moderate	**Moderate**	**HIGH RISK**
Loon Creek	500	↓ 54 (.10)	↓ .98 (.40)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Camas Creek	500	↑ 38 (.20)	↓ .80 (.29)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Lower Mainstem MF	500	*Insf. data*	*Insf.data*	-	Moderate	Moderate	**Moderate**	**HIGH RISK**
Upper Mainstem MF	750	↑ 71 (.18)	↓ 0.50 (.72)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Sulphur Creek	500	↑ 67 (.99)	↑ .92 (.26)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Marsh Creek	500	↑ 253 (.27)	↓ 1.21 (.24)	**High**	Low	Low	**Low**	**HIGH RISK**

Bear Valley Creek	750	↑ 474 (.27	↓ 1.37 (.17)	**High(M)**	Very Low	Low	**Low**	**HIGH RISK**
Upper Salmon River MPG								
Salmon Lower Main	2,000	↓ 108 (.18)	↑ 1.18 (.17)	**High**	Low	Low	**Low**	**HIGH RISK**
Salmon Upper Main	1,000	↑ 411 (.14)	↑ 1.22 (.19)	**High (M)**	Low	Low	**Low**	**HIGH RISK**
Pahsimeroi River	1,000	↑ 267 (.16)	↑ 1.37 (.20)	**High (M)**	Moderate	High	**High**	**HIGH RISK**
Lemhi River	2,000	↑ 143 (.23)	↑ 1.30 (.23)	**High**	High	High	**High**	**HIGH RISK**
Valley Creek	500	↑ 121 (.20)	↑ 1.45 (.15)	**High**	Low	Moderate	**Moderate**	**HIGH RISK**
Salmon East Fork	1,000	↑ 347 (.22)	↑ 1.08 (.28)	**High**	Low	High	**high**	**HIGH RISK**
Yankee Fork	500	↑ 44 (.45)	↓ .72 (.39)	**High**	Moderate	High	**High**	**HIGH RISK**
North Fork	500	*Insf. data*	*Insf. data*		Low	Low	**Low**	**HIGH RISK**
Panther Creek (ext)	750	*Insf. data*	*Insf. data*					*Extirpated*

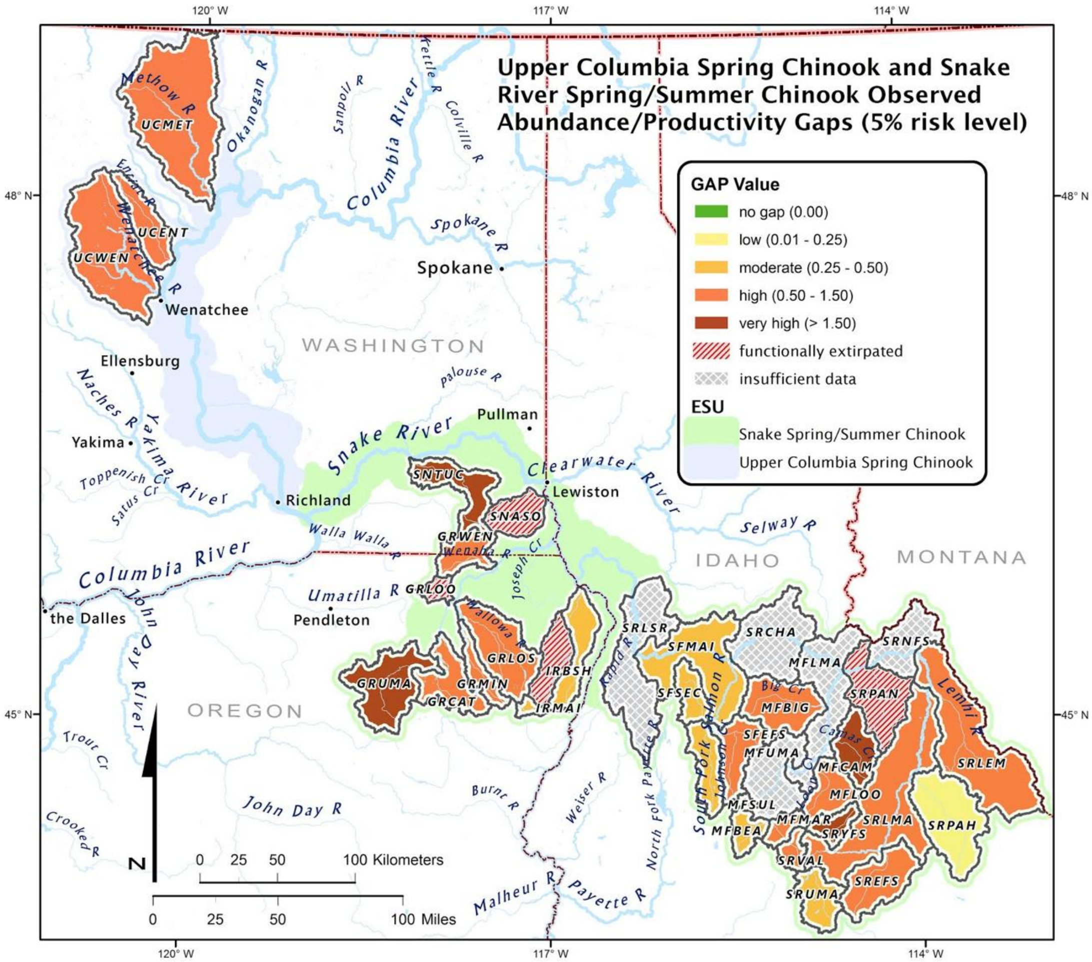


Figure 27 Abundance and productivity gaps for Snake River Spring/Summer Chinook ESU populations (map also includes Upper Columbia Spring Chinook ESU populations for comparison). Populations with insufficient data to generate gaps shaded in gray. Gaps are defined as relative improvement in productivity or limiting capacity required for a population to exceed its corresponding 5% risk viability curve (ICTRT, 2007b).

UPDATED BIOLOGICAL RISK SUMMARY

The majority of populations in the Snake River spring/Summer Chinook salmon ESU remained at high overall risk, with one population (Chamberlain Creek in the Middle Fork MPG) improving to an overall rating of maintained due to an increase in abundance (Table 17). Natural origin abundance has increased over the levels reported in the prior review for most populations in this ESU, although the increases were not substantial enough to change viability ratings. Relatively high ocean survivals in recent years were a major factor in recent abundance patterns. Ten populations increased in both abundance and productivity, seven increased in abundance while their updated productivity estimates decreased, two populations decreased in abundance and increased in productivity. One population, Loon Creek in the Middle Fork MPG, decreased in both abundance and productivity. Although all but one population in this ESU remained at high risk for abundance and productivity, there is a considerable range in the relative improvements to life cycle survivals or limiting life stage capacities required to attain viable status (Figure 27). In general, populations within the South Fork grouping had the lowest gaps among MPGs. The other multiple population MPGs each have a range of relative gap levels.

Spatial structure ratings remain unchanged from the prior reviews, with low or moderate risk levels for the majority of populations in the ESU. Four populations from three MPGs (Catherine Creek and Upper Grande Ronde, Lemhi River and Lower Middle Fork Mainstem) remain at high risk for spatial structure loss. Three of the four extant MPGs in this ESU have populations that are undergoing active supplementation with local broodstock hatchery programs. In most cases those programs evolved from mitigation efforts and include some form of sliding scale management guidelines designed to maximize potential benefits in low abundance years and reduce potential negative impacts at higher spawning levels. Efforts to evaluate key assumptions and impacts are underway for several programs.

While there have been improvements in abundance/productivity in several populations relative to prior reviews, those changes have not been sufficient to warrant a change in ESU status.

SNAKE RIVER FALL-RUN CHINOOK SALMON ESU

BRIEF DESCRIPTION OF ESU

The extant Snake River fall Chinook salmon ESU includes fish spawning in the lower mainstem of the Snake River and the lower reaches of several of the associated major tributaries including the Tucannon, the Grande Ronde, Clearwater, Salmon and Imnaha Rivers (Figure 28). This ESU was originally listed under the ESA in 1992 (most recently reaffirmed in 2005 and 2012). Historically, natural production from this ESU was mainly from spawning in the mainstem of the Snake River upstream of the Hells Canyon Dam complex. The spawning and rearing habitat associated with the current extant population represents approximately 20% of the total historical habitat available to the ESU (Dauble *et al.* 2003). Based on updated information, there was a single historical population (the Middle Snake population) above the current location of Hells Canyon Dam, consisting of two major spawning areas. The primary (largest and most productive) Middle Snake River subpopulation likely spawned within the area of direct aquifer influence described by Connor *et al*. (in preparation). Temperature conditions during spawning and incubation were strongly influenced by water inputs from the aquifer, allowing for earlier emergence timing and rapid growth especially in the reaches upstream of the current Swan Falls Dam site. A single population above Hells Canyon is a revision of the original determination of two populations above Hells Canyon Dam based on historical accounts of spawner distribution and spatial geomorphic considerations (ICTRT 2007). A key factor in that decision was a 56-km gap in suitable spawning habitat reported in Parkhurst (1950). Based on a detailed review of the geomorphic potential in that region, the gap was overestimated and was more likely less than 25 km (Connor *et al*. in prep).

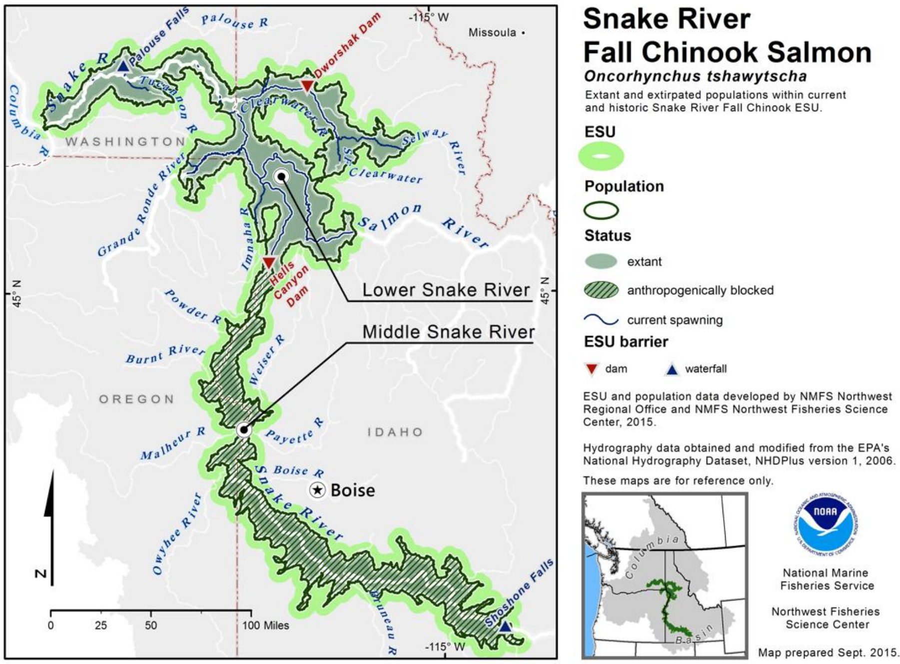


Figure 28 -- Map of the Snake River fall-run Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The 2005 BRT review (Good *et al*. 2005) included an assessment of Snake River fall Chinook salmon based on data for runs through the 2001 return year. A majority of the rating points assigned by individual BRT members fell into the "likely to become endangered" category (60%). The BRT review noted that "…this outcome represented a somewhat more optimistic assessment of the status of this ESU than was the case at the time of the original status review…". Reasons cited for a more optimistic rating included: the number of natural origin spawners in 2001 was well over 1,000 for first time since 1975, management actions had reduced the number of outside origin stray hatchery fish passing to the spawning grounds, the increasing contribution of native Lyons Ferry fish from supplementation programs and the fact that recent natural origin returns had been fluctuating between 500 and 1,000 spawners – somewhat higher than previous levels. The 2005 BRT status ratings for the Snake River fall Chinook salmon ESU were also influenced by concerns that the geometric mean abundance at the time was below 1,000 ("…a very low number for an entire ESU"), and because of the large fraction of hatchery fish on the spawning grounds. Additional concerns cited by the BRT included the fact that a large portion of historical mainstem habitat was inaccessible. Some BRT members were concerned about the possibility that a natural historical buffer between Snake River fall Chinook and other Columbia River ESUs may have existed and that it had been compromised by hatchery straying.

2010

Ford *et al*. (2011) concluded that abundance and productivity estimates for the single remaining population of Snake River Fall Chinook salmon had improved substantially relative to the time of listing. However the current combined estimates of abundance and productivity population still resulted in a moderate risk of extinction of between 5% and 25% in 100 years. The extant population of Snake River fall Chinook was the only remaining from an historical ESU that also included large mainstem populations upstream of the current location of the Hells Canyon Dam complex. The increases in natural origin abundance were encouraging. However, hatchery origin spawner proportions had increased dramatically in the years prior to the review – on average, 78% of the estimated adult spawners were hatchery origin over the most recent brood cycle leading up to the 2010 review. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Spawner abundance, productivity and proportion natural origin estimates for the Lower Mainstem Snake River population are based on counts and sampling at Lower Granite Dam. Separate estimates of the numbers of adult (age 4 and older) and jack (age 3) fall Chinook salmon passing over Lower Granite Dam are derived using ladder counts and the results of sampling a portion of each year's run using a trap associated with the ladder. A portion of the fish sampled at the trap are retained and used as hatchery broodstock. Each year, projected return levels of hatchery- and natural-origin Snake River fall Chinook salmon are used to define a randomized sampling strategy across the duration of the run that will also achieve hatchery broodstock objectives for the Snake River fall Chinook programs and be consistent with impact limits on co-occurring listed steelhead returns. Fish shunted into the trap are measured, sampled for scales to determine age, and examined for marks and/or tags. Fish removed for broodstock are transported to Lyons Ferry and Nez Perce Tribal hatcheries (on alternative days) for holding and spawning. Coded wire tags (CWTs) are read at spawning. The data from trap sampling, including the CWT recovery results, passive integrated transponder (PIT) tag detections and the incidence of adipose clips, are used to construct daily estimates of hatchery proportions in the run.

At present, estimates of natural-origin returns are made by subtracting estimated hatchery-origin returns from the total run estimates (Young *et al*. 2012). In the near future, returns from a Parental Based genetic Tagging (PBT) program will allow for a comprehensive assessment of hatchery contributions and, therefore, a more direct assessment of natural returns.

Sampling methods and statistical procedures used in generating the estimated escapements have improved substantially over the past 10 to 15 years. Beginning with the 2005 return, estimates are available for the total run apportioned into natural and hatchery returns by age (and hatchery-origin) with standard errors and confidence limits (e.g., Young *et al.* 2012). Current estimates of escapement over Lower Granite Dam for return years prior to 2005 were also based on adult dam counts and trap sampling. Methods varied across years and are generally described in annual reports compiled by the Washington Dept. of Fish and Wildlife Snake River laboratory (Milks *et al*. 2014). In the near future, the escapement estimates for 1999-2004 return years will be updated using the new escapement reconstruction framework.

ABUNDANCE AND PRODUCTIVITY

The updated data series described above of spawner abundance, age structure and hatchery/natural proportions were used to generate current assessments of abundance and productivity at the population level. Evaluations were done using both a set of metrics corresponding to those used in prior Biological Review Team (BRT) reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The relatively simple BRT level metrics were done consistently across all ESUs and DPSs to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section below. The ICTRT abundance and productivity metrics are measured over longer time frames to dampen the effects of annual variations and they use annual natural origin age composition to calculate brood year recruitment when sampling levels meet regional fishery agency criteria. Population level estimates derived from these sources for this assessment are available through the NWFSC Salmon Population Summary database (http://www.nwfsc.noaa.gov/trt/mapsdata.cfm).

Prior to the early 1980s, returns of Snake River fall Chinook salmon were likely predominately of natural-origin (Bugert 1995). Natural return levels declined substantially following the completion of the three-dam Hells Canyon Complex (1959-1967), which completely blocked access to major production areas above Hells Canyon Dam, and the construction of the lower Snake River dams (1962-1975). Based on extrapolations from sampling at Ice Harbor Dam (1977-1990), the Lyons Ferry Hatchery (1987-present) and at Lower Granite Dam (1990-present), hatchery strays made up an increasing proportion of returns at the uppermost Snake River mainstem dam through the 1980s (Bugert & Hopley 1989; Bugert *et al.* 1990). Strays from out-planting Priest Rapids hatchery-origin fall Chinook salmon (an out-of-ESU stock from the mid-Columbia) and Snake River fall Chinook salmon from the Lyons Ferry Hatchery program (on-station releases initiated in the mid-1980s) were the dominant contributors. Estimated natural-origin returns reached a low of less than 100 fish in 1990.

In recent years, naturally spawning fall Chinook salmon in the lower Snake River have included both returns originating from naturally spawning parents and from returning hatchery releases. Hatchery-origin fall Chinook salmon escaping upstream above Lower Granite Dam to spawn naturally are now predominantly returns from supplementation program juvenile releases in reaches above Lower Granite Dam and from releases at Lyons Ferry Hatchery that have dispersed upstream. These fish are considered to be part of the listed ESU.

The geometric mean natural adult abundance for the most recent 10 years of annual spawner escapement estimates (2005-2014) is 6,418, with a standard error of 0.19. Natural-origin spawner abundance has increased relative to the levels reported in the most recent status review (Ford *et al.* 2011), driven largely by relatively high escapements in the most recent three years (Table 18).

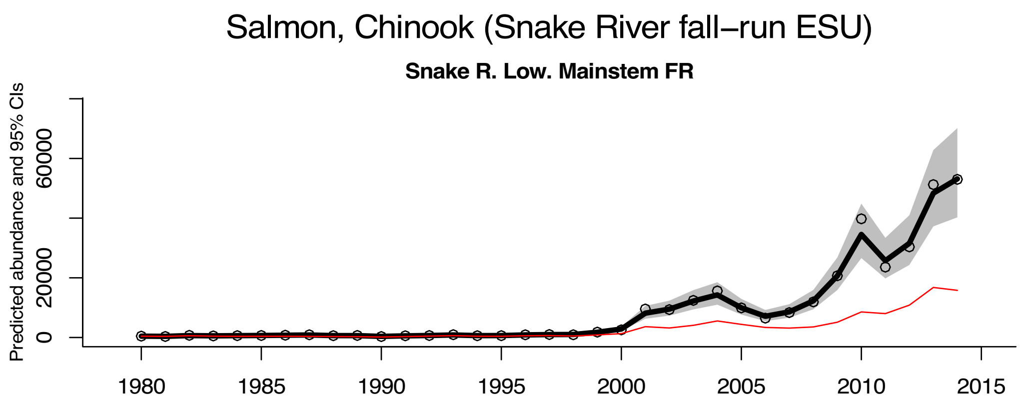


Figure 29 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

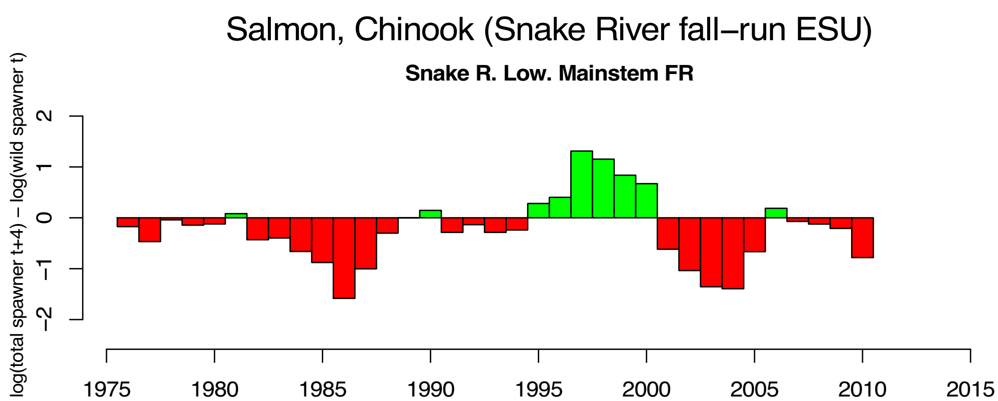


Figure 30 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4). Spawning years on x axis.

Table 18 -- 5-year geometric mean of raw natural spawner counts. This is the raw total spawner count times the fraction natural estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Snake R. Low. Mainstem FR	Snake R.	333 (581)	548 (980)	3049 (8496)	3662 (10581)	11254 (37812)	207 (257)

Table 19 -- 15-year trends in log natural spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Snake R. Low. Mainstem FR	Snake R.	0.22 (0.17, 0.26)	0.15 (0.1, 0.19)

Snake River fall Chinook salmon have a very broad ocean distribution and have been taken in ocean salmon fisheries from central California through southeast Alaska. They are also harvested in-river in tribal and non-tribal fisheries. Historically they were subject to total exploitation rates on the order of 80%. Since they were originally listed in 1992, fishery impacts have been reduced in both ocean and river fisheries (Figure 31). Total exploitation rate has been relatively stable in the range of 40% to 50% since the mid 1990s.

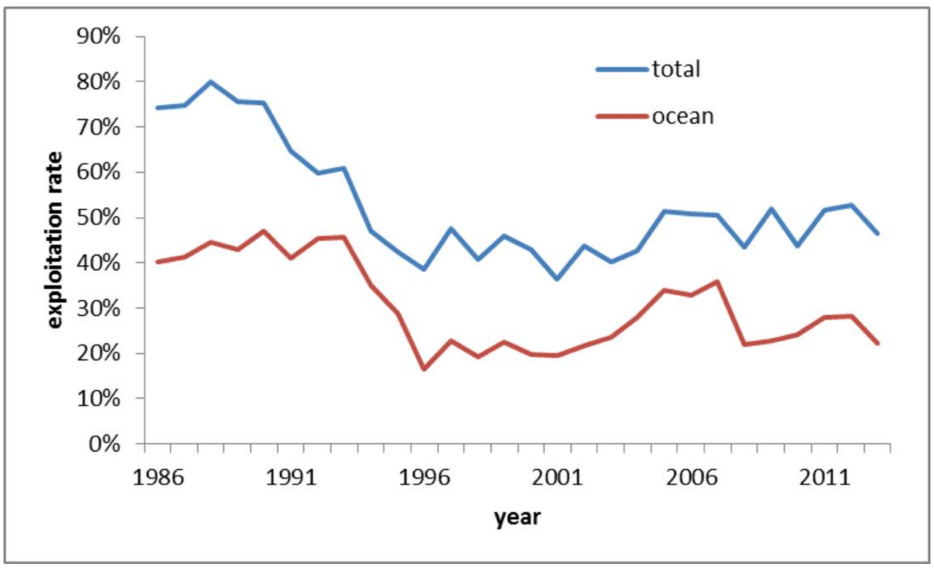


Figure 31 -- Total exploitation rate for Snake River fall Chinook salmon. Data for marine exploitation rates from the Chinook Technical Committee model (Calibration 1503) and for in-river harvest rates from the Columbia River Technical Advisory Committee (TAC 2014, and Robin Ehlke, WDFW, personal communication).

SPATIAL STRUCTURE AND DIVERSITY

The extant Lower Snake River Fall Chinook population consists of a spatially complex set of five historical major spawning areas (ICTRT 2007), each of which consists of a set of relatively discrete spawning patches of varying size (Connor *et al.* 2001; Groves *et al.* 2013). The primary MaSA in the extant Lower River population is the 96-km Upper Hells Canyon Reach, extending upriver from the confluence of the Salmon River to the Hells Canyon Dam site, where the canyon walls narrow and strongly confine the river bed. A second mainstem Snake River MaSA, the Lower Hells Canyon, extends 69 km downstream from the Salmon River confluence to the upper end of the contemporary Lower Granite Dam pool. The lower mainstem reaches of two major tributaries to the mainstem Snake River, the Grande Ronde and the Clearwater Rivers, were also identified by the ICTRT as MaSAs. Both of these river systems currently supports fall Chinook spawning in the lower reaches. In addition, there is some historical evidence for production of late spawning Chinook in spatially isolated reaches in upriver tributaries to each of these systems. Attempts are underway to develop a separate early spawning component into the upper Clearwater River using the South Fork Clearwater weir as a broodstock collection point (Hesse & Johnson 2012).

Historical records and geomorphic assessments support the historical existence of a fifth MaSA comprised of spawning habitats in the Lower Tucannon River and the adjacent inundated mainstem Snake River section associated with Little Goose and Lower Monumental Dams. Several other tributaries of varying size (e.g., the Salmon and Imnaha Rivers, Alpowa and Asotin Creeks) enter the mainstem Snake River within each of the MaSAs defined above. Production in those lower mainstem sections is considered part of the adjoining mainstem MaSA (ICTRT 2007). Similar to the Grande Ronde and Clearwater Rivers, anecdotal accounts suggest that late spawning Chinook may have existed in the lower mainstem of the South Fork Salmon River (e.g., Connor *et al.*, in prep). Historically, some level of fall Chinook salmon spawning may have occurred in the lower Snake River in the reach currently inundated by the Ice Harbor Dam pool (Dauble *et al.* 2003). Spawners using the lowest potential spawning reaches in the Snake River, currently inundated by Ice Harbor Dam, could have been associated with either the Lower Snake River population or a population centered on mainstem Columbia River spawning areas currently inundated by John Day and McNary Dams.

Although annual redd surveys show that fall Chinook spawning occurs in all five of the historical MaSAs, the inability to obtain carcass samples representative of the mainstem MaSAs makes assessment of natural origin spawner distributions difficult. Reconstruction of natural origin spawners based on hatchery expansions and data from homing/dispersal studies on acclimated hatchery releases indicate that four out of the five MaSAs are contributing to naturally produced returns. Carcass samples are obtained in the Tucannon River, expanding the hatchery marked recoveries in that MaSA account for virtually all of the redds, suggesting negligible natural origin returns (Milks & Oakerman 2014).

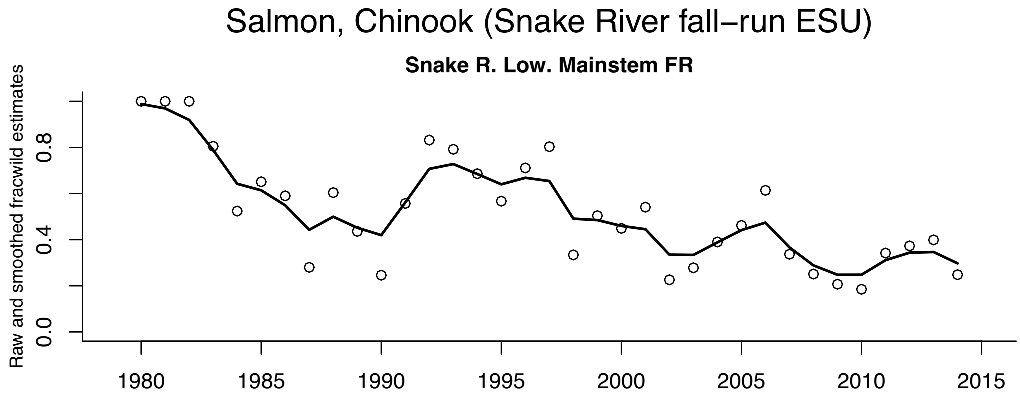


Figure 32 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 20 --5-year mean of fraction natural origin fish in the populaiton (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Snake R. Low. Mainstem FR	0.62	0.58	0.38	0.37	0.31

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Productivity, defined in the ICTRT viability criteria as the expected replacement rate at low to moderate abundance relative to a population's minimum abundance threshold, is a key measure of the potential resilience of a natural population to annual environmentally driven fluctuations in survival. The ICTRT Viability Report (ICTRT 2007) provided a simple method for estimating population productivity based on return-per-spawner estimates for the most recent 20 years. To assure that all sources of mortality are accounted for, the ICTRT recommended that productivities used in Interior Columbia River viability assessments be expressed in terms of returns to the spawning ground. Other management applications express productivities in terms of pre-harvest recruits. Pre-harvest recruit estimates are available for Snake River fall Chinook salmon.

The ICTRT Viability report (2007) also acknowledged that alternative means of assessing productivity at low to moderate spawning abundance may be appropriate or required, especially in cases where total (natural- plus hatchery-origin) spawning levels consistently are at or above the minimum threshold for a particular population. In particular, it anticipated that fitted stock-recruit models might provide a useful alternative for evaluating a population's abundance and productivity relative to specific recovery criteria. The ICTRT recommended that if such an approach was used the 'steepness' parameter (Hilborn & Walters 1992) of the stock-recruit model would be an appropriate index of productivity. Steepness is defined as the expected return-per-spawner at a parent-spawner level of 20% of the predicted equilibrium escapement for a data series. Steepness is derived algebraically from the more basic stock-recruit curve parameters (productivity at the origin and capacity). While the consistently high spawner escapements driven by a combination of natural and hatchery supplementation returns have complicated interpretation of results from the simple R/S method, the increased range in parent escapement estimates has increased the feasibility of using fitted stock-recruit relationships as an alternative approach for estimating production parameters.

Estimates of current productivity for this population were developed using both the simple average R/S method and by fitting stock-recruit functions using maximum likelihood statistical routines (nls routine in the R statistical package). Using the ICTRT simple 20-year R/S method, the current estimate of productivity for this population (1990-2009 brood years) is 1.53 with a standard error of 0.18. Findings using the simple R/S method indicate that there have been years when abundance was high but productivity (R/S) fell below the replacement level (Figure 33), indicating potential influence from density-dependence limitations, poor ocean conditions, or poor migration conditions. This estimate of productivity, however, may be problematic for two reasons: 1) the increasingly small number of years that actually contribute to the productivity estimate means that there is increasing statistical uncertainty surrounding that estimate, and 2) the years contributing to the estimate are now far in the past and may not accurately reflect the true productivity of the current population. Under the simple R/S method, all of the R/S estimates for years after 1999 are excluded from the average due to the high total (hatchery plus wild) escapements in those years. Total escapements for brood years 2010 through 2014 are also well above the minimum threshold levels and will be excluded in calculating productivity using the simple ICTRT method in future assessments.

Expressing productivity as an expected average return-per-spawner from parent-spawner escapements below levels associated with strong density-dependent effects is a key feature of the ICTRT methods for assessing current population performance against viability curves. The ICTRT determined, based on preliminary sensitivity analyses, that estimated productivities derived by fitting stock-recruit relationships to current data series could be compared to a single set of viability curves if those estimates were expressed as steepness (ICTRT 2007).

Four alternative stock-recruit models (Table 21) were fit to the 1991-2010 brood year spawner and return data set for the Lower Mainstem Snake River fall Chinook salmon population: 1) Constant RS - a model that assumed a constant underlying R/S value that is invariant with respect to spawner density, 2) Beverton-Holt RS, 3) Ricker RS, and 4) the Shepard model RS (Shepard 1982), a form that includes a third fitted parameter corresponding to the general shape of the relationship. Each function was fit with and without an annual PDO term to evaluate the potential contribution of year to year variations in ocean conditions. The nls routine in the R statistical package was then used to estimate the parameters of the four stock-recruit models (Table 22). The models were statistically compared using the AICc criteria (AICcmodavg package).

Regardless of whether recruits were measured as returns to the spawning grounds or as pre-harvest recruits, based on a comparison of AICc values the three models incorporating density-dependent terms (Beverton-Holt, Ricker and Shepard) fit the data significantly better than the constant R/S model (Table 22). The estimated equilibrium abundance estimates from the three density-dependent models were each below the recent 10-year geometric mean natural abundance estimate of 6,418. The Beverton Holt model had the lowest (most supported) AICc score, followed by the Shepard function. The fitted relationships for natural log return per spawner vs. parent spawners and the results of bootstrapping to illustrate the potential influence of parameter uncertainty for the Beverton-Hold function are provided in Figure 33. The inset pie chart in the top panel summarizes the proportions of the bootstrap samples that fall into the four possible risk categories. 67% of the samples exceeded the viability curve for Very Low Risk, compared to the recovery plan requirement of 80%. The spawner/recruit plot includes the 1991-2014 recruit and parent spawner pairs, unadjusted and adjusted to reflect the fitted PDO relationship included in the analysis.

Table 21 - Stock-Recruit functions fit to Snake River Fall Chinook brood year 1991-2010 data series.

Model		Equation
Constant RS		$Recruits := \alpha * Spawners * \epsilon^{(0,\sigma)}$
	With PDO	$Recruits := \alpha * e^{c*PDOnorm} * Spawners * \epsilon^{(0,\sigma)}$
Beverton Holt		$Recruits := \dfrac{\alpha * Spawners}{(1 + \frac{a}{b} * Spawners)} * \epsilon^{(0,\sigma)}$
	With PDO	$Recruits := \dfrac{\alpha * e^{(c*PDOnorm)} * Spawners}{(1 + \frac{a}{b} * Spawners)} * \epsilon^{(0,\sigma)}$
Ricker		$Recruits := \alpha * Spawners * e^{(-b*Spawners)} * \epsilon^{(0,\sigma)}$
	With PDO	$Recruits := \alpha * e^{(c*PDOnorm)} * Spawners * e^{(-b*Spawners)} * \epsilon^{(0,\sigma)})$
Shepard		$Recruits := \dfrac{\alpha * Spawners}{(1 + \frac{a}{b} * Spawners)^d} * \epsilon^{(0,\sigma)}$
	With PDO	$Recruits := \frac{\alpha * e^{(c*PDOnorm)} * Spawners}{(1 + \frac{a}{b} * Spawners)^d} * \epsilon^{(0,\sigma)}$

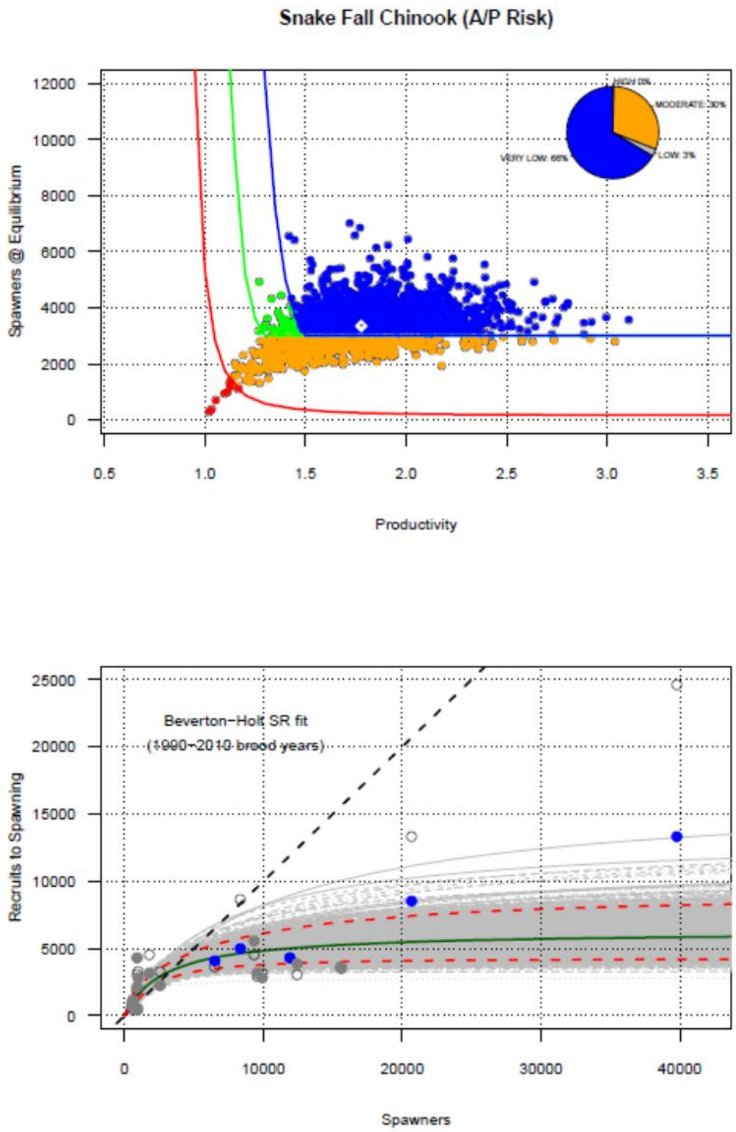


Figure 33 - Beverton Holt stock recruit relationship fitted to broodyears 1991-2010 Snake River Fall Chinook adult escapement estimates. Includes parameter uncertainty generated using the nlsBoot routine in the R statistical package. Top panel: Summary of bootstrap results (2,000 iterations) plotted against Snake Fall Chinook viability curves. Pie chart in upper right corner summarizes the proportions of bootstrap runs vs. ICTRT viability curves (High, Moderate, Low and Very Low risk). Bottom panel: Data points (with and without average fitted PDO multiplier). Black dashed line is 1:1 replacement.

Table 22 - Snake River fall Chinook spawner/recruit function fits. See text for details.

Recruits (Spawners)											
SR Model	Recruits	a	b	c	d	Resid SE	Alpha	steepness	Equil	AICc	AICc diff.
BH	EscwPDO	0.79	6210	-0.0304	NA	0.5383	2.2	1.774	3387	39.4	0
Shepard	EscwPDO	2.094	88	-0.03	0.594	0.5321	8.12	2.173	2395	41.3	1.9
BH	Esc	0.503	8530	NA	NA	0.6475	1.65	1.46	3360	44.8	5.4
Constant	Esc	-0.214	NA	NA	NA	0.8346	0.81	NA	NA	46.5	7.1
Shepard	Esc	1.222	456	NA	0.544	0.6448	3.39	1.699	2265	46.6	7.2
RK	EscwPDO	0.228	0.000057	-0.0238	NA	0.7039	1.26	1.2	3961	50.1	10.7
RK	Esc	0.118	0.000043	NA	NA	0.7454	1.12	1.099	2744	50.4	11
Constant	EscwPDO	-0.215	15280	-0.006	NA	0.8537	0.81	2.305	10812	55.8	16.4
Recruits (Spawners plus Harvest)											
SR Model	Recruits	a	b	c	d	Resid SE	Alpha	steepness	Equil	AICc	AICc diff.
BH	AERUNwPDO	1.229	15280	-0.0247	NA	0.4907	3.42	2.305	10812	35.7	0
Shepard	AERUNwPDO	2.985	22	-0.025	0.483	0.4759	19.8	2.055	9542	36.9	1.2
RK	AERUNwPDO	0.827	0.000049	-0.0196	NA	0.6063	2.29	1.939	16919	44.2	8.5
Constant	AERUNwPDO	0.451	NA	-0.004	NA	0.732	1.57	NA	NA	55.8	20.1

While the 10-year geometric mean natural-origin abundance level has been high, the abundance/productivity margin is insufficient to rate as Very Low Risk given the uncertainty-buffering requirement under the single population viability scenario. The potential that the 'true' underlying abundance and productivity being estimated from the samples falls above the 5% viability curve (with minimum abundance threshold) is greater than 80%. As a result, the Lower Mainstem Snake River fall Chinook salmon population is rated at **Low Risk,** rather than Very Low Risk for abundance and productivity.

The recently released Proposed NMFS Snake River Fall Chinook Recovery Plan (NMFS 2015b) proposes that a single population viability scenario could be possible given the unique spatial complexity of the Lower Mainstem Snake River fall Chinook salmon population if major spawning areas supporting the bulk of natural returns are operating consistent with long-term diversity objectives. Under a single population scenario, the requirements for a sufficient combination of natural abundance and productivity could be based on a combination of total population natural abundance and relatively high production from one or more major spawning areas with relatively low hatchery contributions to spawning. At present (escapements through 2014), given the widespread distribution of hatchery releases and the lack of direct sampling of reach-specific spawner compositions, there is no indication of a strong differential distribution of hatchery returns among major spawning areas.

In terms of spatial structure and diversity, the Lower Mainstem Snake River fall Chinook salmon population was rated at **low risk** for Goal A (allowing natural rates and levels of spatially mediated processes) and **moderate risk** for Goal B (maintaining natural levels of variation) resulting in an overall spatial structure and diversity rating of **Moderate Risk** (Table 23). The moderate risk rating was driven by changes in major life history patterns, shifts in phenotypic traits and high levels of genetic homogeneity in samples from natural-origin returns. In addition, risk associated with indirect factors, specifically the high levels of hatchery spawners in natural spawning areas and the potential for selective pressure imposed by current hydropower operations and cumulative harvest impacts contribute to the current rating level.

UPDATED BIOLOGICAL RISK SUMMARY

Overall population viability for the Lower Mainstem Snake River fall Chinook salmon population is determined based on the combination of ratings for current abundance and productivity and combined spatial structure diversity.

Table 23 -- Lower Mainstem Snake River fall Chinook salmon population risk ratings integrated across the four viable salmonid population (VSP) metrics. ***Viability Key: HV – Highly Viable; V – Viable; M – Maintained; HR – High Risk; Green shaded cells – meets criteria for Highly Viable; Gray shaded cells – does not meet viability criteria (darkest cells are at greatest risk).***

		Spatial Structure/Diversity Risk			
		Very Low	**Low**	**Moderate**	**High**
Abundance/ Productivity Risk	**Very Low** (<1%)	**HV**	**HV**	**V**	**M**
	Low (1-5%)	**V**	**V**	**V** **Lower Main. Snake**	**M**
	Moderate (6 – 25%)	**M**	**M**	**M**	**HR**
	High (>25%)	**HR**	**HR**	**HR**	**HR**

The overall current risk rating for the Lower Mainstem Snake River fall Chinook salmon population is "viable" **(**Table 23**)**. The single population delisting options provided in the draft Snake River Fall Chinook Recovery Plan would require the population to meet or exceed minimum requirements for Highly Viable (green shaded combinations) with a high degree of certainty.

The current rating described above is based on evaluating current status against the criteria for the aggregate population. The overall risk rating is based on a low risk rating for abundance/productivity and a moderate risk rating for spatial structure/diversity. For abundance/productivity, the rating reflects remaining uncertainty that current increases in abundance can be sustained over the long run. The geometric mean natural abundance for the most recent 10 years of annual spawner escapement estimates (2005-2014) is 6,418 fish. Using the ICTRT simple 20-year R/S method, the current estimate of productivity for this population (1990-2009 brood years) is 1.5. Given remaining uncertainty and the current level of variability, the point estimate of current productivity would need to meet or exceed 1.70, which is the present potential metric for the population to be rated at very low risk. While natural-origin spawning levels are above the minimum abundance threshold of 4,200, and estimated productivity is also high, the estimates are not high enough to account for the uncertainty buffer needed to achieve a rating of very low risk.

For spatial structure/diversity, the moderate risk rating was driven by changes in major life history patterns, shifts in phenotypic traits, and high levels of genetic homogeneity in samples from natural-origin returns. In particular, the rating reflects the relatively high proportion of within-population hatchery spawners in all major spawning areas and the lingering effects of previous high levels of out-of-ESU strays. In addition, the potential for selective pressure imposed by current hydropower operations and cumulative harvest impacts contribute to the current rating level.

Given the information available in 2015, an increase in estimated productivity (or a decrease in the year-to-year variability associated with the estimate) would be required, assuming that natural-origin abundance of the single extant Snake River fall Chinook salmon population remains relatively high. An increase in productivity could occur with a further reduction in mortalities across life stages. Such an increase could be generated by actions such as a reduction in harvest impacts (particularly when natural-origin spawner return levels are below the minimum abundance threshold) and/or further improvements in juvenile survivals during downstream migration. It is also possible that survival improvements resulting from actions (e.g., more consistent flow-related conditions affecting spawning and rearing, and increased passage survivals resulting from expanded spill programs) in recent years have increased productivity, but that increase is effectively masked as a result of the relatively high spawning levels in recent years. A third general possibility is that productivity levels may be decreased over time as a result of negative impacts of chronically high hatchery proportions across natural spawning areas. Such a decrease would also be largely masked by the high annual spawning levels.

Diversity: To achieve highly viable status with a high degree of certainty, the spatial structure/diversity rating needs to be low risk. This status assessment used the ICTRT framework for evaluating population-level status in terms of spatial structure and diversity organized around two major goals: maintaining natural patterns for spatially mediated processes and maintaining natural levels of variation (ICTRT 2007).

For a single population scenario, achieving low risk for spatial structure/diversity would require that one or more major spawning areas produce a significant level of natural-origin spawners with low influence by hatchery-origin spawners relative to the other major spawning areas. At present (escapements through 2014), given the widespread distribution of hatchery releases and hatchery-origin returns across the major spawning areas within the population, and the lack of direct sampling of reach-specific spawner compositions, there is no indication of a strong differential distribution of hatchery returns among major spawning areas.

Overall, the status of Snake River fall Chinook salmon has clearly improved compared to the time of listing and compared to prior status reviews. The single extant population in the ESU is currently meeting the criteria for a rating of "viable" developed by the ICTRT, but the ESU as a whole is not meeting the recovery goals described in the recovery plan for the species, which require the single population to be "highly viable with high certainty" and/or will require reintroduction of a viable population above the Hells Canyon Dam complex (NMFS 2015b).

SNAKE RIVER SOCKEYE SALMON ESU

BRIEF DESCRIPTION OF ESU

The ESU includes all anadromous and residual sockeye salmon from the Snake River Basin, Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program (Figure 34). This ESU was first listed as endangered under the ESA in 1991; the listing was reaffirmed in 2005 and 2012.

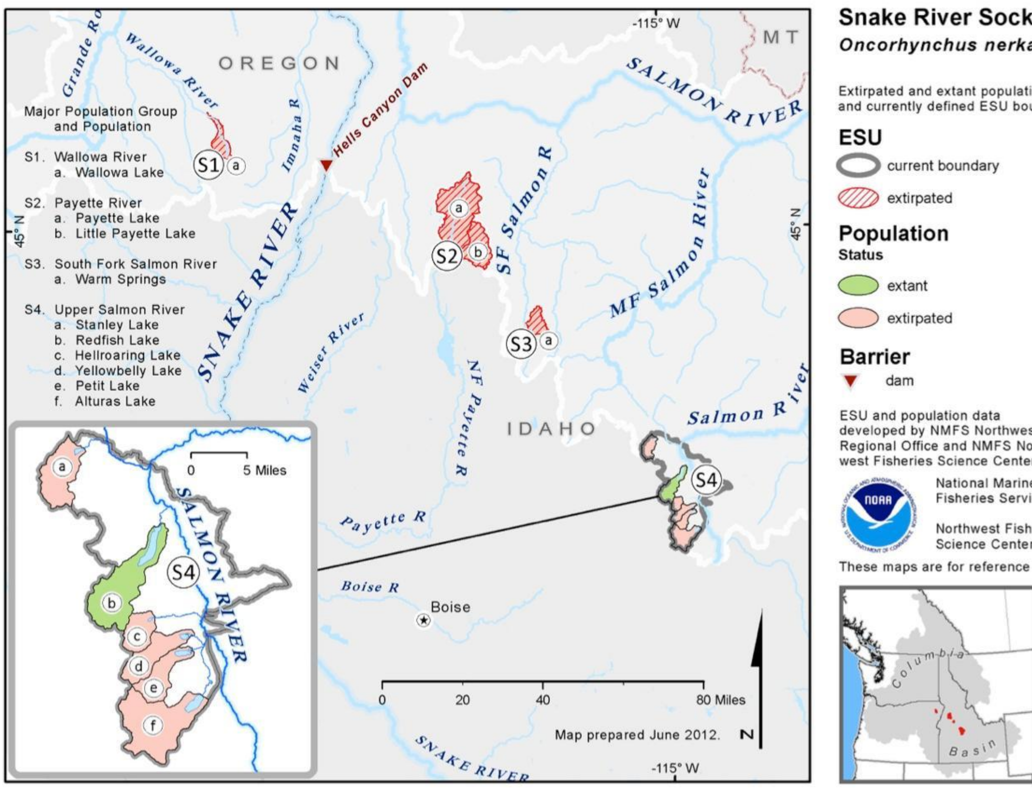


Figure 34 -- Map of the Snake sockeye salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The 2005 BRT assigned the Snake River Sockeye salmon ESU to the "in danger of extinction" category (Good *et al*. 2005). This high risk rating was reflected in the scoring by all members of the BRT. The

BRT rated the ESU at extremely high risk across all four basic risk measures (abundance, productivity, spatial structure and diversity), noting that only 16 naturally produced adults have been counted since 1991. The BRT assessment acknowledged that the emergency captive brood program initiated in 1991 had, "… at least temporarily...rescued this ESU from the brink of extinction…" and that ongoing research had substantially increased biological and environmental information about the ESU.

2010

Ford *et al*. (2011) concluded that substantial progress had been made with the Snake River sockeye salmon captive brood stock based hatchery program, but natural production levels of anadromous returns remained extremely low for this ESU. In then recent years, sufficient numbers of eggs, juveniles and returning hatchery adults had been available from the captive brood program to allow for initiation of efforts to evaluate alternative supplementation strategies in support of re-establishing natural production of anadromous sockeye. Limnological studies and direct experimental releases were being conducted to elucidate production potential in three of the Stanley Basin lakes that were candidates for sockeye restoration. The availability of increased numbers of adults and was supporting direct evaluation of lake habitat rearing potential, juvenile downstream passage survivals and adult upstream survivals. Although the captive brood program had been successful in providing substantial numbers of hatchery produced *O. nerka* for use in supplementation efforts, substantial increases in survival rates across life history stages were needed in order to re-establish sustainable natural production (Hebdon *et al.* 2004; Keefer *et al.* 2008a). The increased abundance of hatchery reared Snake River sockeye salmon reduced the risk of immediate loss, but levels of naturally produced sockeye salmon returns remained extremely low. As a result, overall, Ford *et al*. (2011) concluded that although the risk status of the Snake River sockeye salmon ESU appeared to be on an improving trend in 2010, the new information considered did not indicate a change in the biological risk category since the time of the prior BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Estimates of annual returns are now available through 2014. Adult returns in 2008 and 2009 were the highest since the current captive brood based program began with a total of 650 and 809 adults counted back to the Stanley basin. Approximately two-thirds of the adults captured in each year were taken at the Redfish Lake Creek weir; the remaining adults were captured at the Sawtooth Hatchery weir on the mainstem Salmon River upstream of the Redfish Lake Creek confluence.

At this stage of the recovery efforts for Snake River Sockeye, information on the relative survival rates for rearing and migratory life stages provides valuable insights into the potential for restoring sustainable natural production and the levels of improvement that may be necessary to accomplish production objectives. The recent increases in the availability of hatchery juveniles has allowed for tagging on a sufficient scale to generate relatively precise estimates of both juvenile and adult life stage survivals. Estimates are summarized in the NOAA Snake River Sockeye Recovery Plan.

Juvenile outmigrant survivals from release to Lower Granite Dam have been highly variable, with indications that most mortality is incurred prior to migrants passing the confluence of the North Fork of the Salmon River. Survivals from Lower Granite Dam to below Bonneville Dam reflect two pathways: juveniles collected and transported to below Bonneville Dam and in-river migrants. Juvenile survival from Lower Granite to Bonneville Dam since 2008 has ranged from 40% to 57% (NMFS 2014).

Upstream adult passage survivals from Bonneville Dam to Lower Granite Dam averaged over 70% from 2010-2012, dropping off to 44% in 2013, likely in response to high temperatures during the migration period. Adult survivals from Lower Granite Dam to the Sawtooth Basin also averaged over 70% for 2010-12, dropping off to 33% in 2013. Temperatures during the adult upstream migration in 2015 were unusually high. Preliminary estimates indicated substantial losses in both reaches with only 14% of pit tagged fish detected at Bonneville Dam reaching McNary Dam, the last mainstem Columbia River dam before the Snake River confluence (B. Bellarud, NOAA Fisheries, pers. comm.). Preliminary indications are that survival from McNary to Lower Granite Dam and beyond were also low. The implications of this range in annual survivals for recovery efforts are uncertain and will depend on the relative frequency of passage conditions across future years. Given their particular run timing, phenotypic and behavioral characteristics, Snake River sockeye may be particularly susceptible to high summer temperatures during their adult migration (Crozier *et al.* 2008a).

ABUNDANCE AND PRODUCTIVITY

Adult returns of sockeye salmon to the Sawtooth Basin continued to increase through return year 2014 (Table 24). The higher returns of fish collected at the Redfish Lake and Sawtooth weirs have supported substantial increases in the number of adults released above the Redfish Lake Creek weir (Table 25). Annual adult releases in the most recent five years (2011-2014) have averaged over 1200; almost double the average for the prior 5 year period. The large increases in returning adults in recent years reflect improved downstream and ocean survivals as well as increases in juvenile production since the early 1990s.

Although total sockeye salmon returns to the Sawtooth Basin in recent years have been high enough to allow for some level of spawning in Redfish Lake, the hatchery program remains in its initial phase with a priority on genetic conservation and building sufficient returns to support sustained outplanting (NMFS 2015a).

Table 24 - Adult sockeye salmon returns to Stanley basin sites (P. Kline and C Kozfkay, IDFG pers. comm. March, 2015).

	Redfish Lake Creek**				Sawtooth Fish Hatchery**					Other Traps (LGD, EFSR)					
	Natural Return	Hatchery Return	Alturas L. Natural Return	Total Return***	Natural Return	Alturas L. Natural Return	Hatchery Return	Total Alturas Return	Total Return to SFH	Natural Return to Other Traps	Hatchery Return to Other Traps	Total to Other Traps	Untrapped Fish in the Basin	Total Trapped	Total in Basin
1985	3	0		3	11		0		11					14	14
1986	29	0		29	0				0					29	29
1987	16	0		16	0				0					16	16
1988	1	0		1	0				0					1	1
1989	0	0		0	1				1					1	1
1990	0	0		0	0				0					0	0
1991	4	0		4	0				0					4	4
1992	1	0		1	0				0					1	1
1993	8	0		8					0					8	8
1994	1	0		1					0					1	1
1995	0	0		0					0					0	0
1996	1	0		1					0					1	1
1997	0	0		0					0					0	0
1998	1	0		1					0					1	1
1999	0	0		0			7	0	7					7	7
2000	10	109		119	0	0	124	0	124	0	0	0	14	243	257
2001	4	11		15	0	0	8	0	8	0	0	0	3	23	26
2002	1	6	1	8	4	0	3	1	7	0	0	0	7	15	22
2003	0	2		2	0	0	0	0	0	0	0	0	1	2	3
2004	0	1		1	4	0	18	0	22	0	1	1	3	24	27
2005	2	0		2	0	0	4	0	4	0	0	0	0	6	6
2006	0	0		0	1	0	2	0	3	0	0	0	0	3	3
2007	0	1		1	3	0	0	0	3	0	0	0	0	4	4
2008	48	332	0	380	91	1	126	1	218	0	0	0	50	598	648
2009	75	492	1	568	9	1	239	2	249	0	0	0	16	817	833
2010	141	504	7	652	20	7	621	14	648	3	19	22	33	1322	1355
2011	111	431	0	542	32	2	522	2	556	0	1	1	18	1099	1117
2012	40	67	0	107	12	0	123	0	135	0	0	0	15	242	257
2013	49	173	0	222	30	0	16	0	46	0	2	2	2	270	272
2014*	443	1035	0	1479	10	0	24	0	34	0	3	3	63	1516	1579

*All numbers in 2014 are preliminary as genetic analyses are pending. Likely adjust 10% higher or lower for all categories.

**Some of the fish returning to SFH and RFL are strays from release locations at alternate sites (e.g. adults originating from egg boxes in Alturas retu

***2014 return includes 1 fish of unknown origin (hatchery or wild)

Increased production from the captive brood program has resulted in sufficient release and outplanting levels for initial evaluations of alternative supplementation strategies (Hebdon *et al.* 2004). Hatchery reared pre-smolts have been outplanted into each of the three lakes since the mid-1990s (Table 25). Presmolt outplants using progeny from the Redfish Lake hatchery programs into Redfish, Alturas, and Petit lakes were initiated in the mid-1990s but have not continued in recent years due to relatively poor relative smolt to adult return rates for that particular strategy. Direct smolt plants in the lower section of Redfish Lake Creek and in the Salmon River (Sawtooth weir) have averaged more than 220,000 per year in the most recent five year period (2011-2014).

Unmarked juvenile *O. nerka* emigrating from the three lake systems have averaged approximately 18,500 over the most recent 5 years, ranging from over 30,000 in 2012 to a low of 4,200 in 2014. A number of sources could be contributing to the outmigration of unmarked juveniles including prior years adults passed into Redfish Lake, egg box outplants, natural production from resident spawners or kokanee.

Table 25 – Releases of adults and progeny from Redfish Lake captive brood program into Redfish Lake, Redfish Lake Creek and the Salmon River at or above the Sawtooth Weir (C. Kozfkay, IDFG pers. comm. March, 2015)

	Redfish Lake adult releases					Redfish Lake juvenile releases			Sawtooth Hatchery weir smolts
Release Year	Captive	Anadromous hatchery	Anadromous Naturals	Anadromous (Unknown Origin)	Total	Eyed Eggs	Presmolts	Smolts below RFLC weir	Upper Salmon River Release of Hatchery Reared Smolts
1993	24				24				
1994	63				63		14,119		
1995					0		83,045	3,794	
1996	120				120	105,000	1,932	11,545	
1997	80				80	85,378	152,322		
1998					0		95,248	37,583	44,032
1999	18	3			21		23,886	4,859	4,859
2000	46	114	6		166		48,051	148	
2001	65	10	4		79		83,003	13,915	
2002	177	7	5		189		106,501	38,672	
2003	309				309		59,810		
2004	244				244		79,887		96
2005	176				176		39,870	39,269	39,061
2006	465				465		61,804	46,430	39,622
2007	498				498		62,015	54,582	47,094
2008	396	406	113	52	967		57,093	73,808	76,587
2009	680	637	14	0	1,331		34,561	73,681	99,374
2010	367	1,130	79	0	1,576		31,413	60,498	118,780
2011	558	924	66	0	1,548		50,054	191,048	
2012	622	161	12	0	795		11,354	166,652	
2013	162	150	34	0	346			273,080	
2014	1,098	1,114			2,212			296,389	

Table 26 - Estimated annual numbers of salmon smolt outmigrants from the Stanley basin. This includes hatchery smolt releases, known outmigrants originating from hatchery pre-smolt outplants, and estimates of unmarked juveniles migrating from Redfish, Alturas, and Stanley lakes combined. (C. Kozfkay, IDFG pers. comm. March, 2015)

Year	No. Pre-smolts planted	Estimated outmigration from planted pre-smolts	No. smolts planted	No. eyed eggs planted	Estimated unmarked outmigration	Total estimated outmigration
1993	0	0	0	0	569	569
1994	14,119	0	0	0	1,820	1,820
1995	91,572	823	3,794	0	357	4,974
1996	1,932	14,715	11,545	105,000	923	27,183
1997	255,711	401	0	105,767	304	705
1998	141,871	61,877	81,615	0	2,799	146,291
1999	40,271	38,750	9,718	20,311	3,108	51,576
2000	72,114	12,971	148	65,200	6,602	19,721
2001	106,166	16,595	13,915	0	2,764	33,274
2002	140,410	25,716	38,672	30,924	10,704	75,092
2003	76,788	26,116	0	199,666	4,952	31,068
2004	130,716	22,244	96	49,134	4,643	26,983
2005	72,108	61,474	78,330	51,239	22,135	161,939
2006	107,292	33,401	86,052	184,596	61,312	180,765
2007	82,105	25,848	101,676	51,008	16,023	143,547
2008	85,005	28,269	150,395	67,984	22,240	200,904
2009	59,538	24,852	173,055	75,079	12,429	210,336
2010	65,851	10,505	179,278	59,683	17,533	207,316
2011	50,054	8,904	191,048	42,665	18,788	218,740
2012	11,354	2,373	166,652	0	31,821	200,846
2013	0	31	273,080	0	20,205	293,316
2014	0	0	296,389	0	4,239	300,628

*estimated outmigration is not by broodyear but is by outmigration year.

Annual estimates of an index of SARs have been generated for Snake River sockeye as the estimated number of smolts at Lower Granite Dam in a given year divided into the number of returning adults 2 years later (NWFSC 2009). The median SAR index for the 1998–2006 series of annual estimates was 0.2%, with annual indices ranging from a low of 0.07% to a high of 1.04. SAR estimates for 5 of the 9 years in the series were based on less than 50 adults returning to Lower Granite Dam; therefore these results should be interpreted with caution. Currently available SAR estimates do not include the full effect of the relatively large returns in 2009 and 2010 observed for runs returning to the upper Columbia (Lake Wenatchee and Lake Okanogan) and Snake rivers.

The lower Granite SARs reflect aggregate return rates across two major downstream migration routes: in-river passage and downstream transport to below Bonneville Dam. Estimates of the proportion transported over the 1998 to 2006 outmigration years have ranged from approximately 50% to more than 90%. The median estimated survival of juvenile in-river migrants downriver from Lower Granite Dam through the lower Snake River to McNary Dam on the mainstem Columbia River was 67% for the period 1996–2010, individual year estimates ranged from 28% to 76% (Ferguson 2010). The median estimate of juvenile passage survivals for the McNary Dam to the Bonneville Dam reach (1998–2003, 2006–2010) was 0.54, which should be interpreted with caution due to small sample sizes and associated low detection probabilities for many of the individual year estimates (Ferguson 2010).

Adult upstream passage survivals through the mainstem Columbia River to the mouth of the Snake River are assumed to be relatively high based on inferences from estimates of upstream passage for upper Columbia River sockeye (NWFSC 2008). Comparisons of adult sockeye counts at Ice Harbor and Lower Granite dams indicate direct losses are also low for passage through the lower Snake River. Adult passage survival estimates based on passive integrated transponder (PIT) tag detections at multiple dams also indicate relatively low direct passage mortality upstream to Lower Granite Dam (NMFS 2008).

However, comparisons of the estimated number of adult sockeye salmon at Lower Granite Dam versus returning to the Sawtooth Basin indicate relatively high loss rates through this reach in some years. Keefer *et al*. (2008b) conducted an adult radio tagging study of passage survivals upstream from Lower Granite Dam in 2000 and concluded that high in-river mortalities for Snake River adults could be explained by "… a combination of high migration corridor water temperatures and poor initial fish condition or parasite loads." Keefer *et al*. (2008b) examined current run timing patterns of Snake River sockeye versus records from the early 1960s, concluding that the apparent shift to an earlier run timing in more recent years may reflect increased mortalities for later migrating adults.

HARVEST

Ocean fisheries do not significantly impact Snake River sockeye. Within the mainstem Columbia River, treaty tribal net fisheries and non-tribal fisheries directed at Chinook salmon do incidentally take small numbers of sockeye. Most of the sockeye harvested are from the Upper Columbia River (Canada and Lake Wenatchee), but very small numbers of Snake River sockeye are taken incidental to summer fisheries directed at Chinook salmon. In the 1980s fishery impact rates increased briefly due to directed sockeye fisheries on large runs of Upper Columbia River stocks (Figure 35)

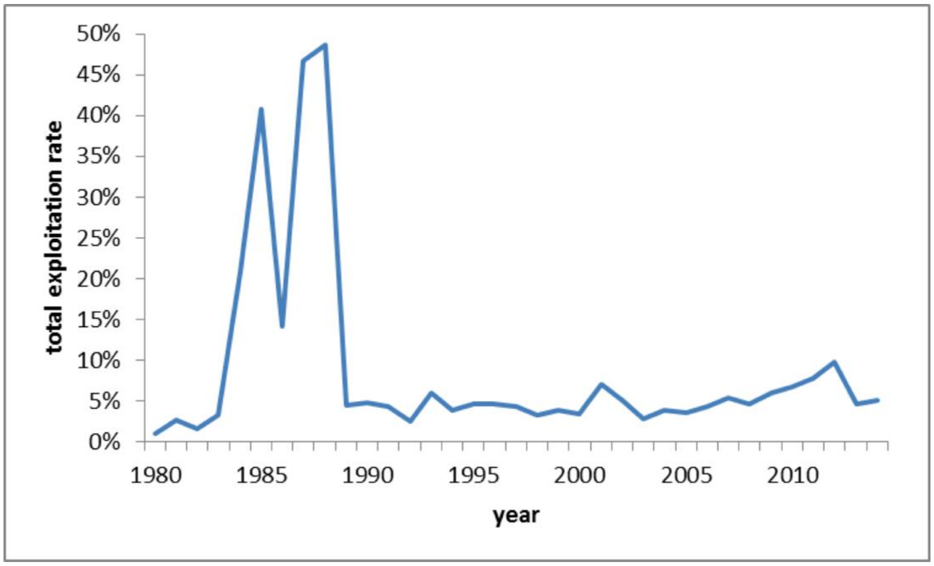


Figure 35 -- Exploitation rates on Snake River sockeye salmon. Data from the Columbia River Joint Staff Report (2015).

SPATIAL STRUCTURE AND DIVERSITY

There is evidence that the historical Snake River Sockeye ESU supported a range of life history patterns, with spawning populations present in several of the small lakes in the Sawtooth Basin (NMFS 2015a). Historical production from Redfish Lake was likely associated with a lake shoal spawning life history pattern although there may have also been some level of spawning in Fish Hook Creek (NMFS 2015a). Historical accounts indicate that Alturas Lake Creek supported an early timed riverine and may have also contained lake shoal spawners (NMFS 2015a).

At present, anadromous returns are dominated by production from the captive spawning component. The ongoing reintroduction program is still in the phase of building sufficient returns to allow for large scale retintroduction into Redfish Lake, the intial target for restoring natural production (NMFS 2015a). Initial releases of adult returns directly into Redfish Lake have been observed spawning in multiple locations along the lake shore as well as in Fishhook Creek (NMFS 2015). There is some evidence of very low levels of early timed returns in some recent years from outmigrating naturally produced Alturas Lake smolts. At this stage of the recovery efforts, the ESU remains rated at High Risk for both spatial structure and diversity.

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Long term recovery objectives for this ESU are framed in terms of natural production. At this point in time, natural production of anadromous Snake River Sockeye remains limited to extremely low levels in Redfish Lake, one of five Sawtooth Valley lakes believed to have historically supported production. As a result, the overall biological status relative to recovery goals is high risk. Substantial progress has been made with the Snake River sockeye salmon captive brood stock based hatchery program. In recent years sufficient numbers of eggs, juveniles, and returning hatchery adults have been available from the captive brood based program to allow for initiation of efforts to evaluate alternative supplementation strategies in support of re-establishing natural production of anadromous sockeye.

Limnological studies and direct experimental releases are being conducted to elucidate production potential in three of the Stanley basin lakes that are candidates for sockeye restoration. The availability of increased numbers of adults and juveniles in recent years is supporting direct evaluation of lake habitat rearing potential, juvenile downstream passage survivals, and adult upstream survivals. Although the captive brood program has been successful in providing substantial numbers of hatchery produced sockeye salmon for use in supplementation efforts, substantial increases in survival rates across life history stages must occur in order to re-establish sustainable natural production (e.g., Hebdon *et al*. 2004, Keefer *et al*. 2008). The increased abundance of hatchery reared Snake River sockeye reduces the risk of immediate loss, but levels of naturally produced sockeye returns remain extremely low.

UPDATED BIOLOGICAL RISK SUMMARY

In terms of natural production, the Snake River Sockeye ESU remains at extremely high risk although there has been substantial progress on the first phase of the proposed recovery approach – developing a hatchery based program to amplify and conserve the stock to facilitate reintroductions. At this stage of the recovery program there is no basis for changing the ESU ratings assigned in prior reviews, but the trend in status appears to be positive.

SNAKE RIVER BASIN STEELHEAD DPS

BRIEF DESCRIPTION OF ESU

The Snake River steelhead DPS includes all naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassable barriers in streams in the Snake River Basin of southeast Washington, northeast Oregon, and Idaho as well as six artificial production programs: the Tucannon River, Dworshak NFH, Lolo Creek, North Fork Clearwater River, East Fork Salmon River, and the Little Sheep Creek/Imnaha River Hatchery steelhead hatchery programs (Figure 36; Federal Register notice 71FR834). Snake River steelhead are classified as summer run based on their adult run timing patterns. Much of the freshwater habitat used by Snake River steelhead for spawning and rearing is warmer and drier than that associated with other steelhead DPSs. Snake River steelhead spawn and rear as juveniles across a wide range of freshwater temperature/precipitation regimes. Fisheries managers classify Columbia River summer run steelhead into two aggregate groups, A-run and B-run, based on ocean age at return, adult size at return and migration timing. A-run steelhead are predominately spend one year at sea and are assumed to be associated with low to mid-elevation streams throughout the Interior Columbia basin. B-run steelhead are larger, with most individuals returning after 2 years in the ocean.

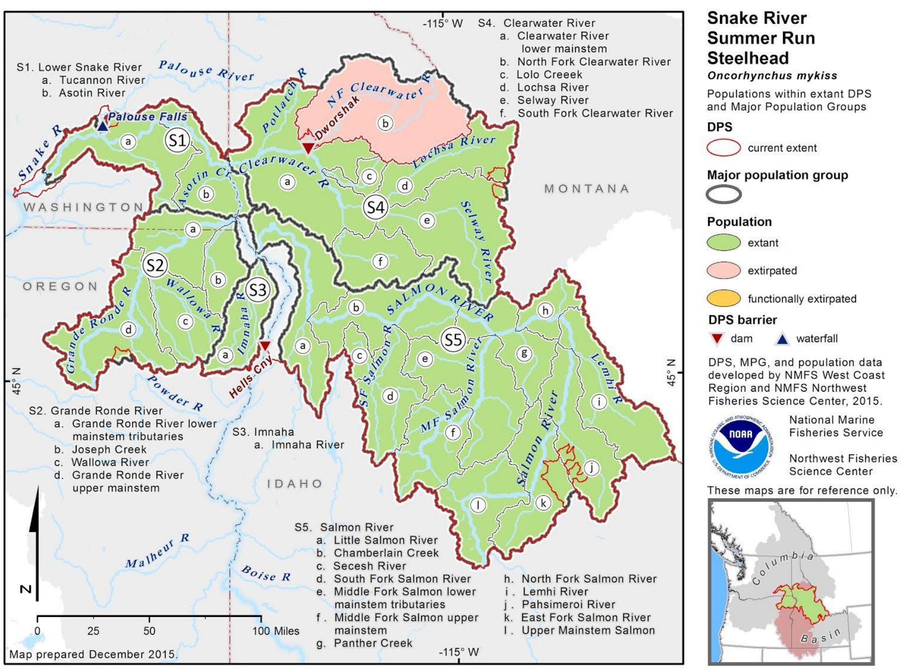


Figure 36 -- Snake River steelhead DPS spawning and rearing areas, illustrating populations and major population groups.

NOAA Fisheries has defined DPSs of steelhead to include only the anadromous members of this species (70 FR 67130). Our approach to assessing the current status of a steelhead DPS is based evaluating information the abundance, productivity, spatial structure and diversity of the anadromous component of this species (Good *et al*. 2005). Many steelhead populations along the West Coast of the U.S. co-occur with conspecific populations of resident rainbow trout. We recognize that there may be situations where reproductive contributions from resident rainbow trout may mitigate short-term extinction risk for some steelhead DPSs (Good *et al*. 2005). We assume that any benefits to an anadromous population resulting from the presence of a conspecific resident form will be reflected in direct measures of the current status of the anadromous form.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The 2005 BRT report highlighted moderate risks across all four primary factors (productivity, natural origin abundance, spatial structure and diversity) for this DPS. A majority (70%) of the risk assessment points assigned by the BRT were allocated to the "likely to become endangered" category. The continued relatively depressed status of B-run populations was specifically cited as a particular concern. The BRT identified that the general lack of direct data on spawning escapements in the individual population tributaries as a key uncertainty, rendering quantitative assessment of viability for the DPS difficult. The BRT also identified the high proportion hatchery fish in the aggregate run over Lower Granite Dam combined with the lack of tributary specific information on relative spawning levels as a second major uncertainty and concern. The BRT cited the upturn in return levels in 2000 and 2001 as evidence that the DPS "…is still capable of responding to favorable environmental conditions." However the report also acknowledged that abundance levels remain well below interim targets for spawning aggregations across the DPS.

2010

Ford *et al*. (2011) concluded that the level of natural production in the two populations with full data series and the Asotin Creek index reaches was encouraging, but the status of most populations in this DPS remained highly uncertain. Population-level natural origin abundance and productivity inferred from aggregate data and juvenile indices indicated that many populations were likely below the minimum combinations defined by the ICTRT viability criteria. A great deal of uncertainty remained regarding the relative proportion of hatchery fish in natural spawning areas near major hatchery release sites. There was little evidence for substantial change in ESU viability relative to the 2005 BRT review. Overall, therefore, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the prior BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

In the past, adult abundance data series for the Snake River Steelhead DPS were limited to a set of aggregate estimates (total, A-run and B-run counted at Lower Granite Dam), estimates for two Grande Ronde populations (Joseph Creek and Upper Grande Ronde River), and index area or weir counts for subsections of several other populations. Obtaining estimates of annual abundance and information on the relative distribution of hatchery spawners for additional populations within the DPS has been a high priority. Two projects based on representative sampling of adult returns at Lower Granite Dam have resulted in estimates of the numbers of natural returns for additional populations or groups of populations for spawning years 2009-14 (QCI 2013; Copeland *et al.* 2015a). One of those approaches, a mixed stock analysis genetics sampling project, is generating estimates of natural origin adults originating from nine different stock groups. A second project generates estimates of the escapement at the population or watershed level for several of the populations in the DPS.

In addition, ODFW has continued to refine sampling methods for the redd count based population estimates on Joseph Creek and the Upper Grande Ronde. A weir based mark/recapture project on Joseph Creek has provided more direct estimates of the number of adult steelhead emigrating into Joseph Creek.

Genetic Diversity

IDFG has compiled an updated assessment of genetic relationships among 66 samples taken from within populations across (Ackerman *et al.* in prep). The results generally support the MPG structure derived by the ICTRT and identified relatively clear population level structure within the Salmon River and Clearwater groups (Figure 37). Differentiation among samples from the Grande Ronde and Lower Snake MPGs are less distinct, indicating the possibility of relatively high rates of exchange among those groups as well as with production from adjacent drainages. At this time it is not possible to determine whether those patterns reflect ongoing, past or periodic exchanges or influences of hatchery fish originating from out of basin stocks.

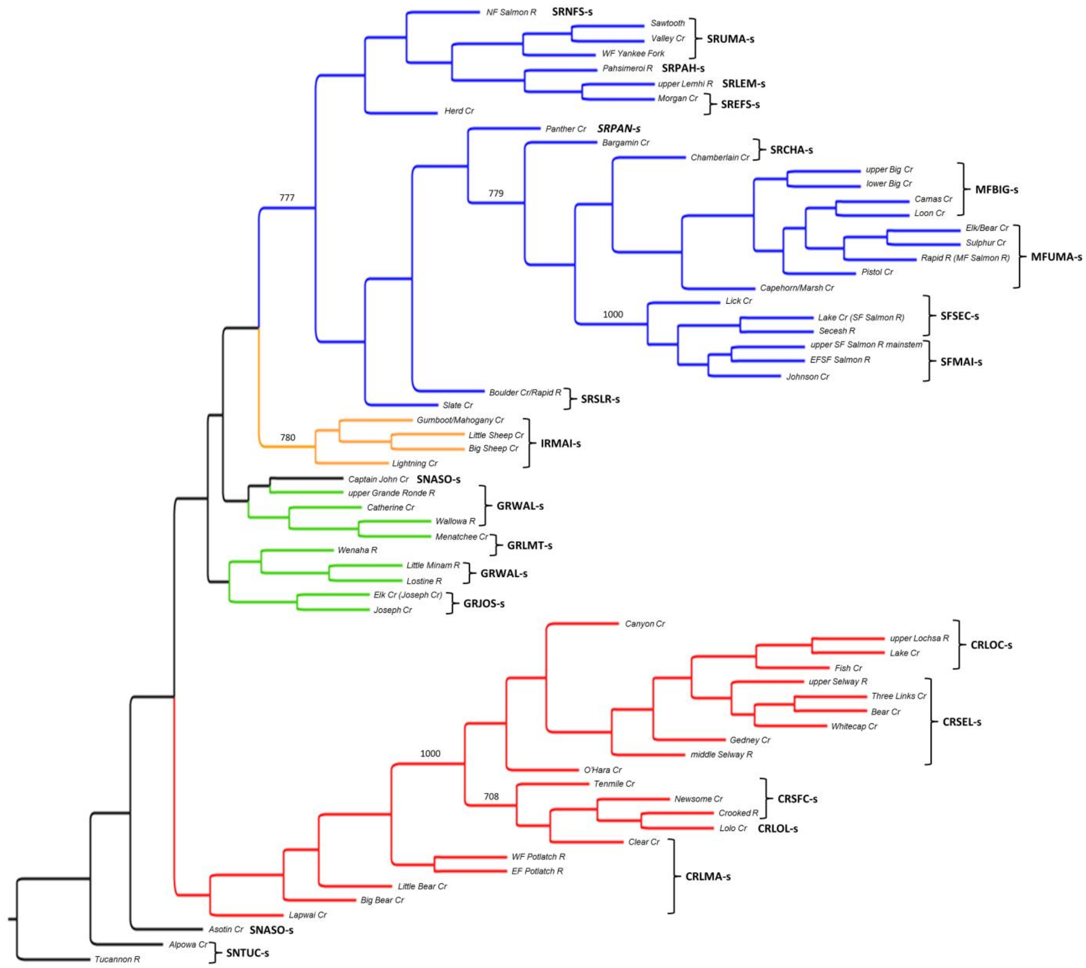


Figure 37 -- From Ackerman *et al.* (in prep) Genetic relationships of steelhead collected from locations across the Snake River basin. The tree is based on Nei's genetic distance and numbers along branches show number of bootstraps out of 1,000 replicates that support the grouping. Only support greater than 70% is shown.

ABUNDANCE AND PRODUCTIVITY

Evaluations were done using both a set of metrics corresponding to those used in prior Biological Review Team (BRT) reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The BRT level metrics were consistenly done across all ESUs and DPSs to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the Recovery evaluation section below. Derived estimates for the two complete population series available for this assessment are archived and available through the NWFSC Salmon Population Summary database (http://www.nwfsc.noaa.gov/trt/mapsdata.cfm).

The most recent five year geometric mean abundance estimates for the two long term data series of direct population estimates (Joseph Creek and Upper Grande Ronde Mainstem) were both increased over the prior review estimates (Table 27). Each of the populations increased an average of 2% per year over the past 15 years (Table 28). Hatchery origin spawner estimates for both populations continued to be low. Both populations are approaching the peak abundance estimates observed in the mid-1980s (Figure 38).

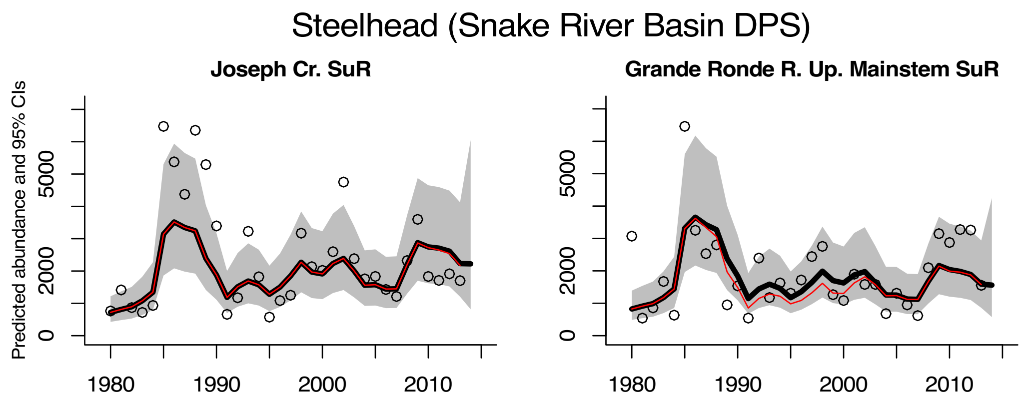


Figure 38 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

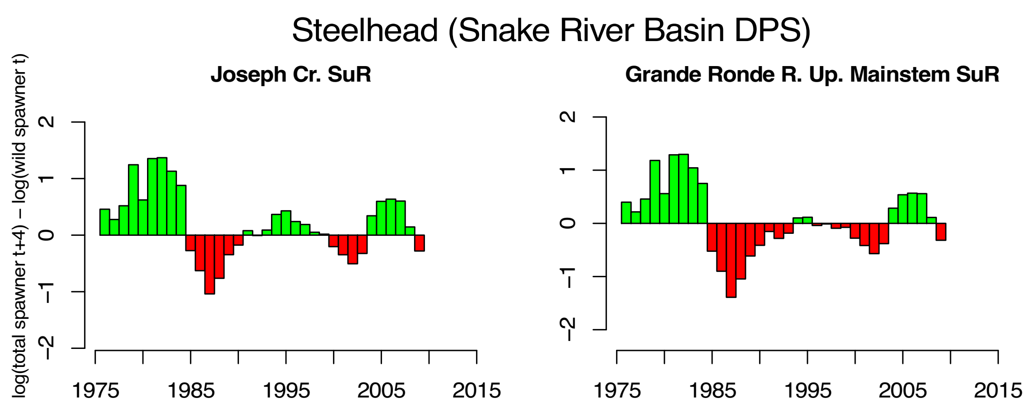


Figure 39 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4). Spawning years on x axis.

Table 27 -- 5-year geometric mean of raw natural spawner counts. This is the raw total spawner count times the fraction natural estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to

compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Joseph Cr. SuR	Grande Ronde R.	1728 (1728)	1394 (1394)	2533 (2533)	1926 (1926)	1747 (1786)	-9 (-7)
Grande Ronde R. Up. Mainstem SuR	Grande Ronde R.	1031 (1307)	1441 (1805)	1164 (1284)	1377 (1384)	2585 (2627)	88 (90)

Table 28 -- 15-year trends in log natural spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Joseph Cr. SuR	Grande Ronde R.	0.02 (0, 0.04)	0.02 (-0.01, 0.04)
Grande Ronde R. Up. Mainstem SuR	Grande Ronde R.	0.02 (0, 0.04)	0.02 (0, 0.04)

Counts of the aggregate runs of natural origin steelhead at Lower Granite Dam (LGR) were also increased relative to the prior review (Figure 40). The 2011-2014 geometric mean count of natural origin A run steelhead at LGR were over twice the corresponding estimate for the prior review, and the updated B run geometric mean was over 50% higher than for the prior review. The hatchery origin steelhead runs to Lower Granite Dam were lower relative to the prior review. As a result the geometric mean estimates of the A and B components of the total run (includes both hatchery and natural origin fish) were down from the prior review (down 7% and 15%, respectively).

The year to year patterns in aggregate Snake Basin and Upper Columbia River runs of wild summer steelhead show similar patterns since 1985 (Figure 40). Both runs declined from peak returns in the mid-1980s, remaining at relatively low levels through the late 1990's. Both runs increased substantially in the early 2000's before dropping and increasing to peak returns in 2010.

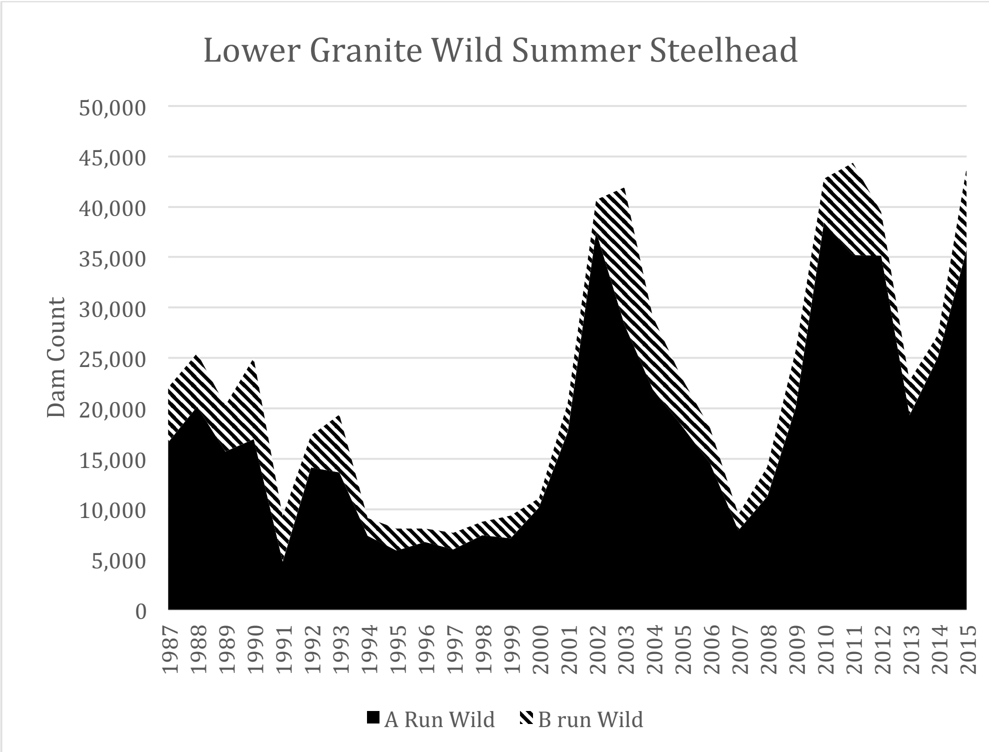


Figure 40- Estimated returns of natural origin steelhead at Lower Granite Dam by spawning year. Broken out by A and B run categories based on B run size criteria (>78 cm).

Smolt to adult return survival estimates (SARs) for the aggregate natural Snake River Steelhead run are available for outmigration years 1964 through 2011 (Figure 41). Year to year variations in SAR represent a major influence on the annual returns of Snake River natural origin steelhead although the pattern is complicated by the fact that multiple broods (predominately ages 3-6) contribute to each particular return year escapement. The relatively high adult returns in the mid-1980s as well as the early and late 2000's correspond to higher average SARs for the corresponding brood years.

Representative SAR series for the aggregate Snake River Steelhead natural origin run show similar general patterns to indices for other Interior Columbia River Basin steelhead DPSs and Chinook ESUs in recent years, indicating that they may be subject to some of the same influences during the smolt to adult phase (Figure 41). The individual series show relative peaks in roughly the same time periods although there are some differences in the timing and relative magnitude of year to year variations.

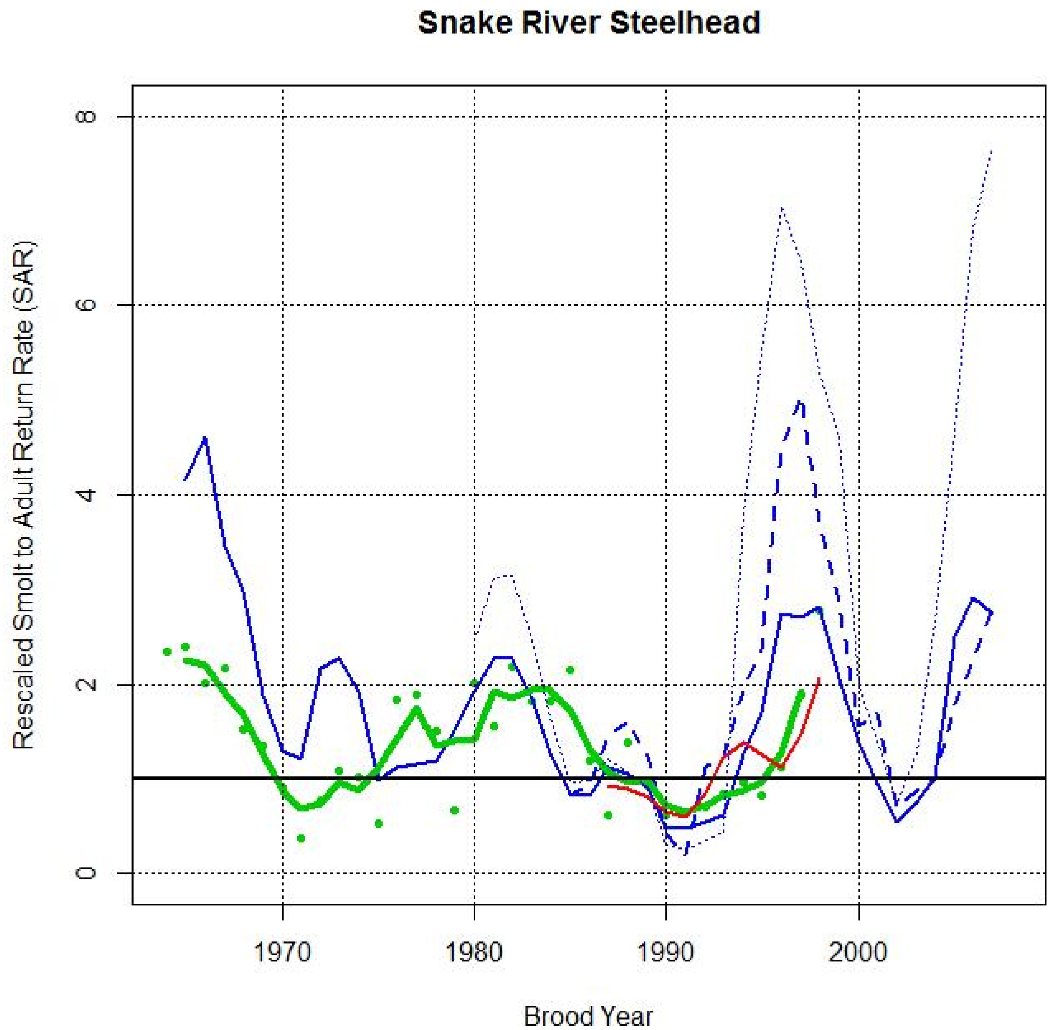


Figure 41 - Snake River natural origin steelhead aggregate smolt to adult return rates (green points and heavy line). Aggregate SARs for other Interior Columbia basin ESUs and DPSs provided for comparison. Snake River aggregate spring/summer Chinook (solid blue), Tuccannon spring Chinook (dotted blue), Upper Columbia spring Chinook (blue dashed line), Upper Columbia steelhead (green dashed line) and, Mid-Columbia steelhead (red line). Each SAR series is rescaled by dividing annual values by the corresponding series mean to faclilitate relative comparison. Lines are three year moving averages.

As noted above, results from the genetic stock composition monitoring at Lower Granite Dam beginning with the 2008-2009 cycle year and the systematic PIT tag program are providing finer scale geographic estimates of steelhead returns by region of origin. The genetic stock identification based approach is currently able to break out the aggregate natural returns at Lower Granite Dam into 10 stock reporting groups (Figure 42). Five of those groupings likely have negligible or very low hatchery contributions (Table 29). Four of those groupings also have a high assignment probability based on baseline sensitivity analyses (Ackerman *et al.* 2014). In addition, the first adult returns that fully reflected the Snake River Steelhead parental based tagging (PBT) program for hatchery fish allowed for generating explicit estimates of adult returns by major hatchery programs beginning with the 1-salt returns in 2011 and 2-salt returns in 2012. In the genetic assignment study, information on each individual presumptive natural origin fish randomly sampled at Lower Granite was used to evaluate the proportions of returns assigned to each stock group that were above and below the B run size criteria cut off (78 cm) (Ackerman *et al.* 2014).

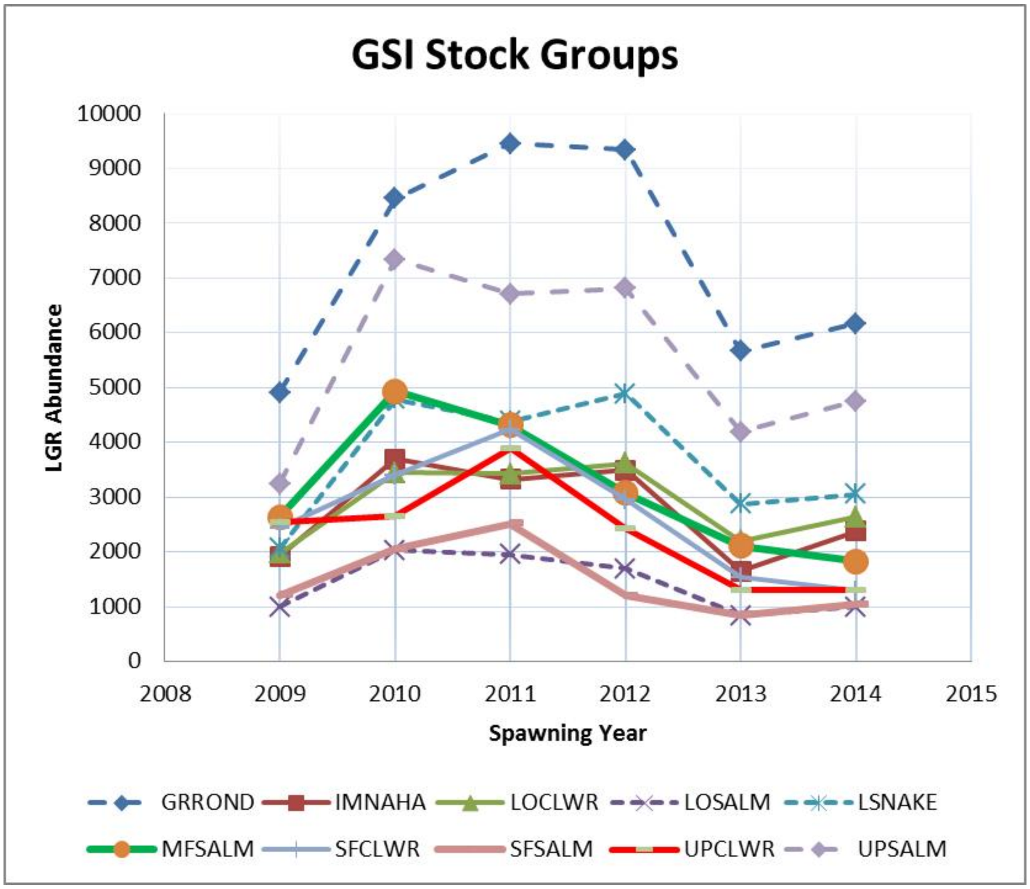


Figure 42 - Snake River steelhead stock group abundance at Lower granite Dam based on Genetic Stock Identification. Solid lines: stock groups with high genetic differentiation, low potential hatchery spawner contributions. Dashed lines: stock groups with either low relative GSI differentiation or high potential for hatchery contributions. From Ackerman *et al.* (2014).

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Table 29 -- Summary of information on potential contributions of hatchery returns to spawning escapements in Snake River Steelhead DPS populations organized by Major Population Group and Genetic stock groups. Hatchery program releases within each population are identified. Direct Estimates column identifies available direct estimates of hatchery spawner contributions in natural areas within populations. 2007 ICTRT review column includes rating and rationale notes regarding within population releases (a- within population releases ; b- no releases but evidence of strays; c-no releases; d-no releases and distrance from mainstem; e-reduction plus distance; f-limited area releases; g: inference from limited weir sampling; h-proximity to major releases). Preliminary run reconstruction hatchery proportions are results from three recent year (2010/11, 2011/12 and 2012/13) studies (Copeland *et al.* 2013, 2014a and 2014b). Tabulated into proportion intervals. Stippled population areas assumed to have negligible hatchery returns.

Major Population Group	Stock Group	TRT Population	Hatchery Program	Direct Estimates	2007 ICTRT Review	Preliminary Run Reconstruction Hatchery Proportions			
						<.25	*.25-.50*	*.50-.75*	*>.75*
Lower Snake MPG	***LOSNK***	Tucannon R.	LFH,Tucannon endemic		High[a]		3		
		Asotin Cr.	none	Low (WDFW Upper Asotin	Mod[b]	1	1	1	
Grande Ronde MPG	***GROND***	Joseph Cr.	none	Low (NPT Joseph Cr. MRC)	Low[c]				
		Upper GR	none		Low[e]				
		Lower GR	Cottonwood		Moderate[f]		1	1	
		Wallowa R.	Wallowa		Low[f]		1		
Imnaha River MPG	***IMNAHA***	Imnaha R.	Big Sheep Creek		>10%[g]	2			
Clearwater MPG	***LOCLW***	Lower Clearwater R.	Dwarshak/Kooskia **	North Fork/Kooskia hatchery weirs Potlatch Cr. samping	Moderate[f]				
		Lolo Cr.	Dworshak		High[a]		1	1	1
		South Fk.	Dworshak		High[a]			1	1
		Lochsa R.	none	Low (IDFG Fish Cr. weir)	Low[c]				
		Selway R.	none		Low[c]				
Salmon River MPG	***SOFK***	Secesh R.	none		Very Low[d]				
		South Fk.	none		Low[c]				
	LOSALM	Little Salmon R.	Multiple	Low (IDFG Rapid R. weir)	High[a]			1	2
	MFKSAL	Upper Middle Fk.	none		Very Low[d]				
		Lower Middle FK.	none		Low[c]				
		Chamberlain Cr.	none		Low[c]				
	UPSAL	North Fk.			High[h]				
		Panther Cr.	past egg box plants	Low (IDFG surveys)	Moderate[a]				
		Pahsimeroi R.			High[a]			2	1
		Lemhi R.			High[a]		1	2	
		East Fk. Salmon R.	East Fk		High[a]			3	
		Upper Salmon R.	multiple		High[a]		1	2	
Hells Canyon tribs	***HELLSC***	Hells Canyon tribs	Oxbow						2

HARVEST

Summer-run steelhead from the upper basin are divided into 2 runs by managers: The A-run, and the B-run. These runs are believed have differences in timing, but managers separate them on the basis of size alone in estimating the size of the runs. The A-run is believed to occur throughout the Middle Columbia, Upper Columbia, and Snake River Basins, while the B-run is believed to occur naturally only in the Snake RiverDPS, contributing in varying proportions, in the Clearwater River, Middle Fork Salmon River, and South Fork Salmon River.

Steelhead were historically taken in tribal and non-tribal gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and in tributaries. In the 1970s, retention of steelhead in non-tribal commercial fisheries was prohibited, and in the mid 1980s, tributary recreational fisheries in Washington adopted mark-selective regulations. Steelhead are still harvested in tribal fisheries, in mainstem recreational fisheries, and there is incidental mortality associated with mark-selective recreation recreational fisheries. The majority of impacts on the summer run occur in tribal gillnet and dip net fisheries targeting Chinook salmon. Because of their larger size, the B-rum fish are more vulnerable to the gillnet gear. Consequently, this component of the summer run experiences higher fishing mortality than the A-run component (Figure 49). In recent years, total exploitation rates on the A-run have been stable at around 5%, while exploitation rates on the B-run have generally been in the range of 15% to 20%. Sport fisheries targeting hatchery run steelhead with incidental impacts on wild returns also occur in the mainstem Columbia River and sections of the Snake, Clearwater and Salmon Rivers.

SPATIAL STRUCTURE AND DIVERSITY

The ICTRT viability criteria adopted in the draft Snake River Management Unit Recovery plans include explicit criteria and metrics for both spatial structure and diversity. With one exception, spatial structure ratings for all of the Snake Basin steelhead populations were low or very low risk given the evidence for distribution of natural production within populations. The exception was Panther Creek, which was given a high risk rating for spatial structure based on the lack of spawning in the upper sections. No new information was provided that would change those ratings.

Updated information is available for two important factors that contribute to rating diversity risk under the ICTRT approach: hatchery spawner fractions and the life history diversity. Updates to the estimated hatchery fractions for the two ongoing long term population specific abundance series are summarized in Figure 43 and Table 30. Hatchery contributions remain relatively low. The first year results of a major effort to better elucidate contributions of individual hatchery programs are now available. At present, direct estimates of hatchery returns based on PBT analysis are available for the run assessed at Lower Granite Dam (Ackerman *et al.* 2015). IDFG is leading the coordinated development of a simple run reconstruction model that uses reach specific harvest and weir removal estimates to generate estimates of hatchery fish escaping to spawn in natural areas for releases within tributary habitats associated with each population (Copeland *et al.* 2013; Copeland *et al.* 2014a; Copeland *et al.* 2014b; Copeland *et al.* 2015a). Preliminary estimates are available for three recent cycle years (Table 29). Given the preliminary nature of these results, the relative proportions are summarized as annual estimates within four general levels from 0 to 1.0.

Information from the GSI assessment sampling provide an opportunity to evaluate the relative contribution of B run returns within each stock group. No populations fell exclusively into the B run size category, although there were clear differences among population groups in the relative contributions of the larger B run life history type. Fish assigned to the UPCLWR, SFSAL and SFCLWR had the highest proportion of B run lengths (median estimates over the five available study year ranging from 49 to 58%). The Middle Fork drainage population aggregate (MFSAL) had an intermediate level of contributions of fish exceeding the B run length threshold, averaging 20%. The remaining populations had low (<10%) or very low (1-2%) contributions from the B run size category.

ICTRT criteria for evaluating spatial structure within populations are based on observing evidence of spawning usage across defined spawning areas within populations, with and emphasis on historically relatively large contiguous reaches (major spawning areas). Evaluating the occupancy of steelhead major spawning areas in the Snake River basin is problematic given the fact that systematic redd surveys are not routinely conducted to to adverse environmental conditions affecting accurate counts. IDFG has recently updated estimates of occupancy for many steelhead populations using juvenile survey data (Copeland *et al.* 2015b). Juvenile surveys in 23 major spawning areas distributed across populations in the Clearwater and Salmon River MPGs met sufficiency criteria for occupancy evaluation – all 23 met minimum ICTRT requirements for full occupancy. The remaining 22 major spawning areas that qualified to provide estimates in some years also showed consistent juvenile steelhead presence consistent with spawning use. Based on this information, spatial structure ratings for Snake River steelhead populations were maintained at the levels assigned in the original ICTRT assessment.

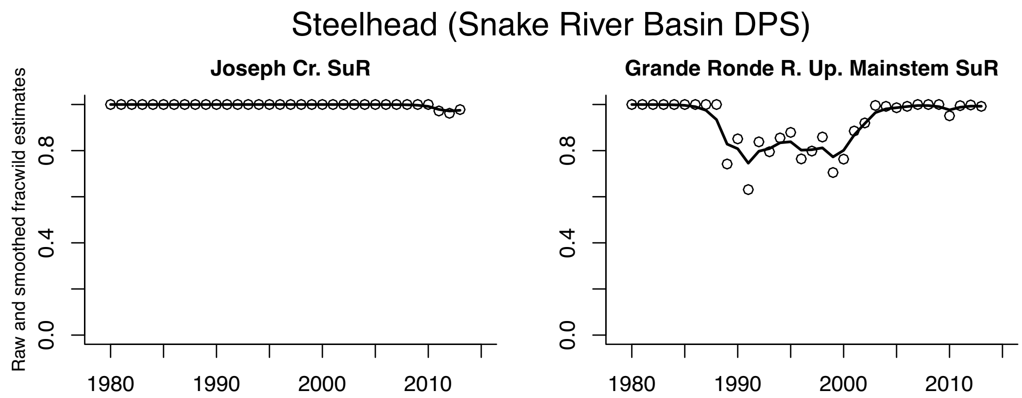

Figure 43 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 30 -- 5-year mean of fraction wild (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Joseph Cr. SuR	1.00	1.00	1.00	1.00	0.98
Grande Ronde R. Up. Mainstem SuR	0.79	0.80	0.91	1.00	0.98

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The Interior Columbia Basin Technical Recovery Team (ICTRT) identified 24 extant populations within this DPS, organized into 5 major population groups (ICTRT 2003). The ICTRT also identified a number of potential historical populations associated with tributary habitat above the Hells Canyon Dam complex on the mainstem Snake River, a barrier to anadromous migration. The five major population groups (MPGs) with extant populations are: the Lower Snake River MPG (2 populations); the Grande Ronde MPG (4 populations); the Imnaha River population/MPG; the Clearwater River MPG (5 extant populations, 1 extirpated); and the Salmon River MPG (12 populations). In addition, the ICTRT concluded that small tributaries entering the mainstem Snake River below Hells Canyon Dam may have historically been part of a larger population with a core area currently cut off from anadromous access. That population would have been part of one of the historical upstream MPGs.

NMFS recovery planning for the Snake River drainage is organized around a subset of management unit plans corresponding to State boundaries. A tributary recovery plan for one of the major management units (MUs), the Lower Snake River tributaries within Washington state boundaries, was developed under the auspices of the Lower Snake River Recovery Board and was accepted by NOAA Fisheries in 2005. The LSRB Plan provides recovery criteria, targets and tributary habitat action plans for the two populations of Spring/Summer Chinook in the Lower Snake MPG along with the Touchet River (Mid-Columbia Steelhead DPS) and the Washington sections of the Grande Ronde River. Draft MU plans have been developed for the Oregon and Idaho drainages, each covering the respective MPGs contained within those states. Viability criteria recommended by the ICTRT were adopted formulating recovery objectives within each of the management unit planning efforts.

The ICTRT recovery criteria are hierarchical in nature, with ESU/DPS level criteria being based on the status of natural origin steelhead assessed at the population level. A detailed description of the ICTRT viability criteria and their derivation (ICTRT 2007) can be found at www.nwfsc.noaa.gov/trt/col/trt_viability.cfm. Under the ICTRT approach, population level assessments are based on a set of metrics designed to evaluate risk across the four viable salmonid population elements – abundance, productivity, spatial structure and diversity (McElany *et al*. 2000). The ICTRT approach calls for comparing estimates of current natural origin abundance (measured as a 10 year geometric mean of natural origin spawners) and productivity (estimate of return per spawner at low to moderate parent spawning abundance) against predefined viability curves. In addition, the ICTRT developed a set of specific criteria (metrics and example risk thresholds) for assessing the spatial structure and diversity risks based on current information representing each specific population. The ICTRT viability criteria are generally expressed relative to a particular risk threshold - low risk is defined as less than a 5% risk of extinction over a 100 year period and very low risk as less than a 1% probability over the same time period.

Table 31 - Summary of available natural origin abundance and productivity estimates for Snake River Steelhead DPS populations. Limited to populations with direct estimates (Joseph Creek and Upper Grande Ronde) or GSI stock groups with low misclassification rates and low estimated or inferred hatchery proportions. ICTRT minimum abndance thresholds summed for stock group aggregates. Methods: Redd Exp – expansion from index area and supplemental redd counts using fish per redd estimates; MRC –mark recapture study ; GSI: run reconstruction based on genetic stock identification estimates from the natural origin run at Lower Granite Dam accounting for estimated harvest and weir removals above Lower Granite Dam (e.g. Copeland *et al*. 2015).

Major Population Group	Stock Group/ Population	ICTRT Minimum Abundance Thresholds		10 Year (2005-2014) Natural Origin Abundance (se)	20 Year (1999-2008) Brood year Intrinsic Productivity (se)	Estimation Method
Grande Ronde	Joseph Creek (pop)	500		1,839 (.09)	1.87 (.20)	Redd Exp/MRC
	Upper Grande Ronde (pop)	1,500		1,649 (.21)	3.15 (.40)	Redd Exp
Clearwater	Lower Clearwater (pop)	1,500		2,099 (0.15)	2.36(.16)	GSI
	Upper Clearwater (stkgrp)	2,000		1,650 (0.17)	2.33 (0.18)	GSI
	Lochsa R.		*1,000*			
	Selway R.		*1,000*			
Salmon	South Fork (stkgrp)	1,500		1,028 (0.17)	1.80 (.148)	GSI
	Secesh R.		*500*			
	South Fork Mainstem R.		*1,000*			
	Middle Fork (stkgrp)	2,500		2,213 (0.16)	2.38 (.104)	GSI
	Upper Middle Fork River		*1,000*			
	Lower Middle Fork River		*1,000*			
	Chamberlain Creek		*500*			
Lower Salmon	Upper Asotin Cr. (subpop)			617 (0.16)	NA	weir est.
	Asotin Creek		*500*			

Snake River Steelhead DPS: NOAA Draft Recovery Plan Scenario

The ICTRT recommends that each extant MPG should include viable populations totaling at least half of the populations historically present, with all major life history groups represented (ICTRT 2007). In addition, the viable populations within an MPG should include proportional representation of large and very large populations historically present. Within any particular MPG, there may be several specific combinations of populations that could satisfy the ICTRT criteria. The Oregon and Idaho Management Unit sections of the draft Snake River Recovery Plan each incorporate specific population restoration and protection scenarios at the MPG level that are consistent with ICTRT recommendations.

Based on the new GSI information for stock groups within this DPS described above, the major life history pattern designations determined by the ICTRT should be updated (Table 32). With one exception, all of the populations assigned by the ICTRT as A run type remain the same. The former B run population designations are revised to reflect the relative proportions of large (<78 cm) adults in the individual stock groups in the genetic assessments of natural origin returns (e.g. Ackerman *et al*.

2014, 2015). The Lower Clearwater population falls into a single population stock group in the genetic analyses, although it has a relatively high potential misclassification rate. The estimated proportion B size class adults to this group is high enough that it provisionally classified as Low B in updating the ICTRT life history pattern assignments.

Table 32 - Updated major life history category designations for Snake River Steelhead DPS populations based on initial results from genetic stock identification studies. Populations designated as A have no or negligible B size category returns in stock group samples. Remaing populations categories reflect relative contribution of fish exceeding B size threshold. (High >40%, Moderate 15 to 40% ,Low <15%).

Major Population Group	Population	2007 ICTRT Major Life History Pattern	Change?	2015 Assessment Updated Major Life History Pattern
Lower Snake River	*Tucannon River*	A		
	Asotin River	A		
Grand Ronde River	*Joseph Cr.*	A		
	Upper Grand Ronde	A		
	Lower Grand Ronde	A		
	Wallowa River	A		
Imnaha River	Imnaha	A		
Clearwater River	Lower Mainstem	A	Provisional	Low B
	South Fork	B	yes	Hi B
	Selway	B	yes	Hi B
	Lochsa	B	yes	Hi B
	LoLo Cr	A/B	yes	Hi B
Salmon River	South Fork	B	yes	Hi B
	Secesh	B	yes	Hi B
	Lower Middle Fk	B	yes	Moderate B
	Upper Middle Fk	B	yes	Moderate B
	North Fk	A		
	Panther Cr	A		
	Pahsimeroi	A		
	Lemhi	A		
	Upper Sal	A		
	Upper Sal (East fk)	A		
	Chamberlain Cr.	A		

Lower Snake River MPG: Both populations (Tucannon River and Asotin Creek) in this MPG are targeted for viable status, with at least one meeting the criteria for highly viable.

Population level abundance data sets are not available for either of the two populations in this MPG. A data series for a large subarea within the Asotin Creek population is available (Table 31). The ICTRT classified Asotin Creek as a Basic population with a minimum abundance threshold of 500 spawners. The recent 10 year geometric mean natural origin spawners for the Upper Asotin Creek sub-area alone exceeds the threshold (500) for the population. Based on recent year PIT tag detections and the Lower Granite genetic stock composition monitoring, Asotin Creek is receiving substantial inputs of adult returns from the Tucannon River and potentially other areas (both natural origin and hatchery) in the lower Snake River region. While the aggregate analyses indicate that total escapement into the Asotin population may include substantial numbers of hatchery origin fish, hatchery fish are currently being removed at several weirs and traps (J. Bumgarner, WDFW pers. comm.). The actual proportional contribution of hatchery spawners to total spawning is not known.

Population level spawner escapement estimates are not available for the Tucannon River population, but indications are that numbers of spawning steelhead in the system are low (e.g. Bumgarner & Dedloff 2013). One contributing factor is an apparent high overshoot rate of returning adults past their natal stream. A portion of the outmigrating natural smolt production from the Tucannon River has been PIT tagged in recent years (Bumgarner & Dedloff 2013). Analysis of returning PIT tagged adults (2005-2012 return years) indicates overshoot rates past the Tucannon River and over Lower Granite Dam (Bumgarner & Dedloff 2013). An average of 12.1% of the run over Ice Harbor are not detected subsequently (loss or spawn in an unknown location). An average of 30.7% of the return over Ice Harbor enter the Tucannon River directly. On average, 59.3% of the returning PIT tagged adults overshoot past the Tucannon River and over Lower Granite Dam. Of those overshootss, 21.2% drop down after overwintering and are subsequently detected in the Tucannon River, resulting in a total of 43.3% into the Tucannon River by both pathways. The remaining 44.6% apparently remain above Lower Granite Dam with an unknown but likely significant portion spawning in Asotin Creek. PIT tagged returns from hatchery releases of endemic and Lyons Ferry stock into the Tucannon River show similar straying proportions.

The ICTRT rated both populations at moderate risk for the integrated spatial structure and diversity criteria. The moderate risk rating was driven by two of the diversity factors – phenotypic patterns and hatchery influence (spawner composition). The risk rating for phenotypic traits reflected uncertainty as to whether traits of the current populations are consistent with the historical patterns or with unaltered reference populations in a similar habitat, geologic, and hydrologic setting. No additional or updated information is available for this review. Hatchery spawners in the Tucannon River still include out of basin Lyons Ferry adults as well as returns from an endemic broodstock program. Recent PIT tag study results indicate that wild spawners is more uniform throughout the Tucannon River, returns from endemic broodstock releases are primarily detected in the upper ½ to ¼ of the system and out of basin detections are centered in the lower ½ to ¼ of the river (Miller *et al.* 2015). It is known that out of basin hatchery stocks do migrate into the Asotin, although the average level of contribution to natural spawning has not been quantified (Copeland *et al*. 2015). As a result, the risk rating for spawner composition remains at moderate for both populations.

The overall population viability ratings for both populations reflect a combination of known condition and uncertainties about key factors, primarily average natural origin abundance and productivity and hatchery influences. Both populations are currently rated at Moderate risk overall, with the possibility that the Tucannon River could be at high risk for abundance and productivity.

More direct estimates of natural origin abundance and hatchery contribution rates for a series of years would be required to change ratings in future assessments.

Grande Ronde MPG: Improvements in natural production are planned for all four populations in this MPG. Given their current status, it is expected that Joseph Creek and the Upper Grande Ronde River populations are the most likely to satisfy the MPG level requirement for one highly viable and one viable population. Although the average abundance levels have dropped from the prior review period, the paired geometric mean natural origin spawner abundance and productivity estimates for both populations exceed the 1% viability curves for their respective size categories (Basic and Large respectively). One of the aggregate natural origin stock groups identified based on genetic sampling at Lower Granite Dam includes all four Grande Ronde populations (e.g., Copeland *et al*. 2015). While, the relatively high misclassification rates associated with this group precluded developing reliable direct estimates of annual escapements for this group for use in this review, the results indicate that the estimated returns to Joseph Creek and the Upper Grande Ronde would account for the majority of the aggregate Grande Ronde run. The ICTRT assigned the Wallowa and Lower Grande Ronde populations a moderate A/P risk rating reflecting the general level of returns of A run steelhead, subarea weir and redd counts. More specific data on annual returns would be needed to assign updated specific abundance and productivity ratings to these two populations.

All four populations in this MPG were assigned Low risk ratings for combined spatial structure and diversity in previous reviews (Ford *et al*. 2011). Preliminary analyses based on the Lower Granite Dam genetic stock identification project, combined with initial brood returns from the parental based tagging program, suggest that hatchery fish may be contributing to spawning in the Lower Grande Ronde and the Wallowa population at significant levels (Copeland *et al*. 2015). More information on the relative distribution and levels of contribution would be useful. At this time, the risk ratings for hatchery contributions to those two programs are increased to moderate.

The Grande Ronde Steelhead MPG is tentatively rated as achieving viable status. One population (Joseph Creek) is Highly Viable, the Upper Grande Ronde population meets the criteria for Viable, and the remaining two populations are provisionally rated as Maintained. Efforts are underway that might lead to population specific abundance and productivity series for those two populations and to a more explicit understanding of the relative distribution of hatchery spawners.

Imnaha River MPG: The Imnaha River population will need to meet highly viable status for this one population MPG to be rated as viable under the basic ICTRT criteria.

The Imnaha River Steelhead population was rated was rated as maintained in the prior review, based on moderate ratings for abundance and productivity (average A run surrogate) and spatial structure/diversity. The Imnaha River constitutes one of the stock groups identified in the Lower Granite genetic stock identification program, although it is one of the stock groups with relatively high misclassification potential (Table 28). For that reason we have not explicitly adopted an extrapolated time series for this population. However, the general results from the genetic stock identification project to date and the two available annual PIT tag based estimates of steelhead returns into the Imnaha River (2011 and 2012 spawning years) suggest that natural production may be exceeding the ICTRT minimum threshold of 1,000 for this population. Information from the PBT hatchery study indicates that the number of hatchery returns from Imnaha River releases that remain available to spawn after harvest and weir removals may be substantial. While it is likely that those returns are concentrated in one section of the population (Big Sheep Creek), the relative distribution of hatchery and natural spawners is uncertain. Estimates of hatchery proportions in the upper end of the mainstem Imnaha are relatively low (Harbeck *et al.* 2015), but there is uncertainty about proportions in the lower mainstem Imnaha River.

Based on the information currently available, the Imnaha steelhead population is not meeting the Highly Viable rating for a single population MPG called for in the draft Snake River Recovery Plan. Achieving a Highly Viable rating would require achieving a Very Low risk rating for abundance and productivity and a Low overall risk rating for spatial structure and diversity. It is possible that additional years information from the PIT tag array project and/or refinements to the genetic stock identification program will result in improved estimates in future reviews. Additional information on the relative distribution of hatchery spawners could change the current diversity risk rating.

Clearwater River MPG: This MPG includes five extant and one extirpated (North Fork Clearwater River) populations. The draft recovery scenario for this MPG calls for recovery of the Lower Clearwater River (large size), along with the Lochsa River and the Selway River.

The previous status reviews rated the Lower Clearwater River population at moderate risk and both the Lochsa and Selway River populations at high risk for abundance and productivity, based on the averages from the aggregate A and B run estimates of Lower Granite Dam returns. Results from the genetic stock composition project support breaking out natural origin returns to the Lochsa and Selway River populations as a specific stock group with relatively low misclassification error. Extrapolating from the four years of direct estimates, the estimated 10 year geometric mean of 1,650 for the stock group falls short of the combined minimum thresholds for the two populations (2,000 = 2 X 1,000). The estimated geometric mean productivity for the stock aggregate is 2.33. Assuming that aggregate estimates generally represent the current status of each component population, each population would rate at moderate A/P risk. The genetic stock composition analysis does support partitioning out Lower Clearwater returns from the aggregate natural return at Lower Granite Dam, but this single population stock group has a higher potential rate of misclassification than the Upper Clearwater River group. Based on the current GSI mixture analysis extrapolation, the combination of recent geometric mean abundance and productivity for Lower Clearwater population exceeds the ICTRT 1% viability curve (minimum abundance threshold of 1,500).

The remaining two populations in the Clearwater MPG (Lolo Creek and the South Fork Clearwater) constitute another stock reporting group in the genetic stock composition analysis. This grouping has relatively high misclassification rates (Table 29). PIT tag arrays have recently been installed in Lolo Creek and the upper South Fork and one year of estimates from the Lower Granite natural origin PIT tag project are available (2012). In that year, an estimated 680 natural origin steelhead escaped into Lolo Creek, 1201 into the upper South Fork Clearwater River (QCI 2013). There are relatively large and consistent hatchery releases into the area, especially within the South Fork Clearwater. The PBT results for the initial year of adult hatchery returns (2012) indicate substantial numbers of hatchery fish are available to spawn after accounting for known removals. It is not possible at this time to generate productivity estimates for this grouping since estimates of the total number of spawners including hatchery fish are not available. For this review, the provisional high risk A/P ratings applied in prior reviews will be carried forward. Additional years information from the genetics stock program combined with refinements in the analysis should allow for updating the provisional ratings in future reviews.

The Locha, Selway and Lower Clearwater River populations were assigned Low combined spatial structure/diversity ratings in prior reviews, while South Fork and Lolo Creek populations were rated moderate risk. The moderate ratings were driven largely by high risk for spawner composition. No additional information is available that would alter the ratings. More specific data on potential hatchery contributions to spawning in Lolo Creek and South Fork is consistent with the high risk ratings assigned to this particular factor in the prior reviews.

Based on the updated risk assessments, the Clearwater MPG does not meet the ICTRT criteria for a viable MPG. Although the more explicit information on natural origin spawner abundance indicates that the Lower Clearwater River, Lochsa River and Selway River populations are improved in overall status relative to prior reviews, the South Fork and Lolo Creek populations do not achieve maintained status due in part to uncertainties regarding productivity and hatchery spawner composition.

Salmon River MPG: This relatively large MPG includes 12 extant populations. The draft Idaho MU Recovery Plan identifies six populations to prioritize for viable status across this MPG. The recovery scenario is consistent with the ICTRT recommendations and includes the two Middle Fork populations (highest B proportions within the MPG), the South Fork River, Chamberlain Creek, Panther Creek and the North Fork Salmon River populations. The proposed scenario for this MPG includes consideration for historical population size, inclusion of populations exhibiting a range of A and B run timing proportions, and achieving a distribution of viable populations across the geographical extent of the MPG.

Estimates of natural origin abundance with relatively low misclassification potentials are available for two population subgroups within this MPG, the Middle Fork stock group (3 populations) and the South Fork stock group (2 populations). The remaining seven populations in the MPG fall into two additional stock groups with relatively high misclassification potential and, in some cases, are associated with substantial hatchery releases.

In prior reviews the three Middle Fork Salmon River and the two South Fork Salmon River populations were each assigned high risk ratings for abundance/productivity based on the aggregate abundance time series for B run steelhead passing Lower Granite Dam. Based on the genetic stock composition study, the 10 year geometric mean escapement above Lower Granite for the two population Middle Fork stock group (2,213) is below the combined minimum thresholds (2,500 = 2 X 1,000+500). The estimated intrinsic productivity for the stock group over the most recent 20 year series was 2.38. Assuming those stock group estimates apply to each component population, the resulting combinations would fall below the population specific minimum abundance thresholds associated with the 5% risk curves but above the 25% viability curve, corresponding to a moderate risk rating. The 10 year geometric mean natural origin escapement estimate for the South Fork Salmon stock group is 1,028, below the sum of the minimum abundance thresholds for the two component populations (500+1,000). The estimated intrinsic productivity (relative to the aggregate thresholds) is 1.88. Under the same assumptions as for the Middle Fork grouping, the updated abundance and productivity ratings for the two South Fork populations would be moderate.

The Little Salmon River population is identified as a distinct single population group within the current GSI mixture analyses, but sensitivity analysis indicates it has a relatively high misclassification rate (Ackerman *et al*. 2014). The recent 10 year geometric mean natural origin returns at Lower Granite dam allocated to this stock group in the GSI assessment is 991 (T. Copeland, IDFG, pers. comm.), which would exceed the minimum threshold of 500 for this Basic sized population. In addition, the potential for hatchery spawner contributions into natural areas is high, therefore it is not possible to calculate productivity for this population based on adult recruit to total spawner estimates.

The remaining populations within the Salmon River MPG fall within a single aggregate stock group in the GSI analysis (North Fork Salmon River, Pahsimeroi River, Lemhi River, East Fork Salmon River and Upper Salmon River). This stock group has relatively high potential for misclassification in the GSI mixture analysis (Ackerman *et al*. 2014). In addition, there are ongoing hatchery releases into habitats associated with most of the populations in this grouping. Preliminary run reconstructions based on PBT estimates of hatchery returns at Lower Granite Dam adjusted for subsequent fishery and hatchery weir removals indicate that substantial numbers of hatchery origin adults escape and are available to spawn in natural areas. The distribution of these potential spawners relative to natural origin adults is not well understood.

Estimates of natural origin steelhead escaping into the Lemhi River population are available for three years (2010-12) based on PIT tag recoveries (QCI 2013). Those estimates range from 428 to 680, all well below the ICTRT minimum threshold of 1,000 spawners for this Intermediate size population. Natural origin abundance estimates are also available for the tributary segment of the Pahsimeroi River population. Only natural origin fish are passed above that weir, and the most recent 10 year geometric mean count (2005-2014) was 90. Large numbers of hatchery steelhead (adipose-clipped smolts) are released below the Pahsimeroi River weir and in the mainstem section of the Upper Salmon between the Pahsimeroi River and the Lemhi River for harvest augmentation under ongoing mitigation programs. Not all returning adults are intercepted in fisheries or captured at hatchery weirs, as a result there are not any current estimates of either the number or proportion of hatchery-origin steelhead that spawn naturally in the mainstem and small tributary habitats that are also part of the Pahsimeroi River steelhead population.

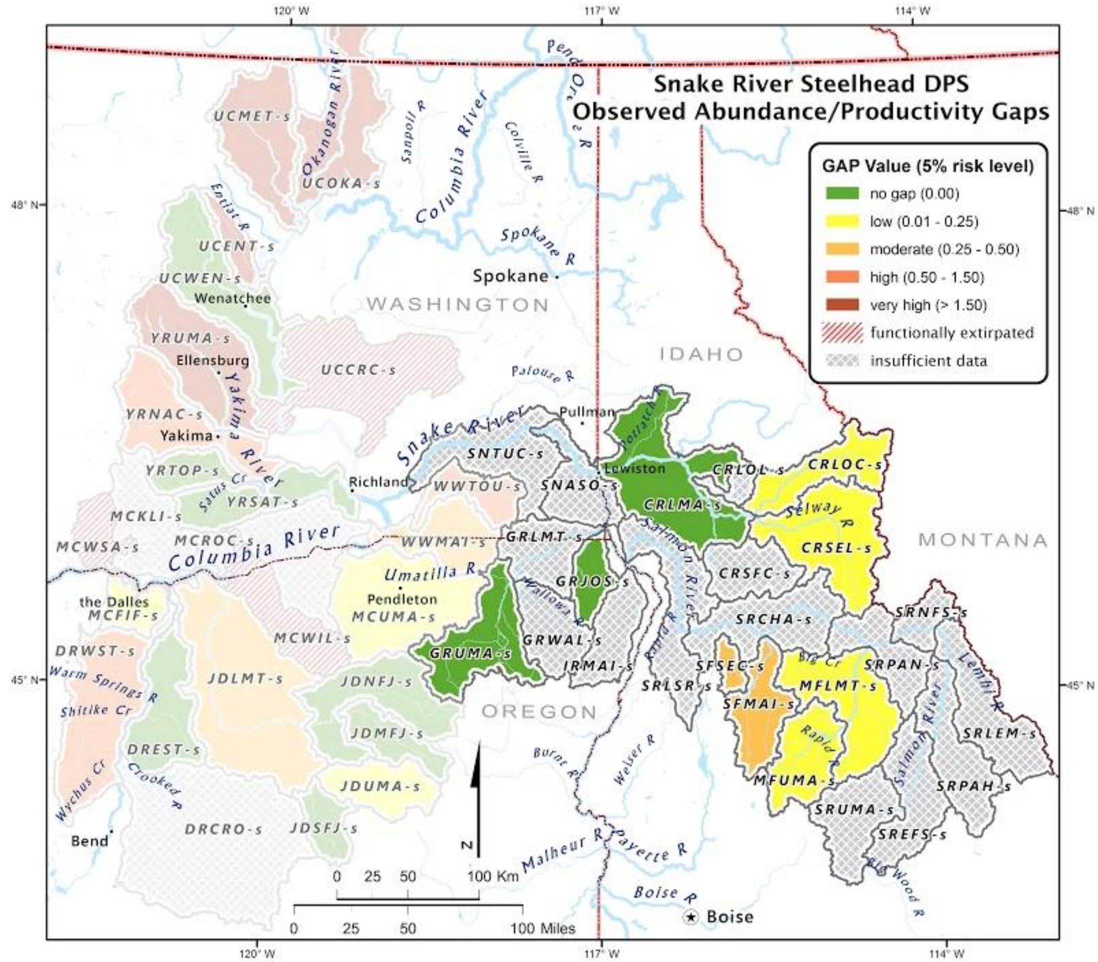

Figure 44. Snake River DPS steelhead population abundance/productivity gaps (bold colors). Populations with insufficient data to generate gaps shaded in gray. Gaps are defined as relative improvement in productivity or limiting capacity required for a population to exceed its corresponding 5% risk viability curve (ICTRT 2007). Gap estimates for populations in the Upper Columbia DPS and Mid-Columbia River DPS provided for comparison (shaded colors).

Table 33 – Summary of status relative to the ICTRT viability criteria. Ratings with ? are based on limited or provisional data series (see text).

Population	Abundance/Productivity Metrics				Spatial Structure and Diversity Metrics			Overall Viability Rating
	ICTRT Minimum Threshold	*Natural Spawning Abundance*	*ICTRT Productivity*	*Integrated A/P Risk*	*Natural Processes Risk*	*Diversity Risk*	*Integrated SS/D Risk*	
Tucannon River	1,000	NA	NA	**High??**	Low	Moderate	**Moderate**	**HIGH RISK??**
Asotin Creek	500		NA	**Moderate?**	Low	Moderate	**Moderate**	**MAINTAINED?** (HIGH RISK??)
Lower Grande Ronde River	1,000	NA	NA		Low	Moderate	**Moderate**	**MAINTAINED?**
Joseph Creek	500	1,839	1.86	**Very Low**	Very Low	Low	**Low**	**HIGHLY VIABLE**
Upper Grande Ronde	**1500**	1,649 (.21)	3.15 (.40)	**Viable** (Moderate)	Very Low	Moderate	**Moderate**	**VIABLE**
Wallowa River	1,000	NA	NA	**High??**	Very Low	Low	**Low**	**Moderate?**
Imnaha River	1,000	NA	NA	**Moderate?**	Very Low	Moderate	**Moderate**	**Moderate?**
Lower Main. Clearwater R.	1,500	2,099 (.15)	2.36(.16)	**Moderate?**	Very Low	Low	**Low**	**MAINTAINED?**
South Fork Clearwater R.	1,000	NA	NA	**High**	Low	Moderate	**Moderate**	**MAINTAINED/HIGH RISK?**
Lolo Creek	500	NA	NA	**High**	Low	Moderate	**Moderate**	
Selway R.	1,000	1,650 (0.17)	2.33 (0.18)	**Moderate?**	Very Low	Low	**Low**	**MAINTAINED?**
Lochsa R.	1,000			**Moderate?**	Very Low	Low	**Low**	
		NA	NA					
Little Salmon R.	500	NA	NA	**Moderate?**	Low	Moderate	**Moderate**	**MAINTAINED?**
South Fork Salmon R.	1,000	1,028 (0.17)	1.80 (.148)	**Moderate?**	Very Low	Low	**Low**	**MAINTAINED?**
Secesh R.	500			**Moderate?**	Low	Low	**Low**	**MAINTAINED?**
Chamberlain Creek	500	2,213 (0.16)	2.38 (.104)	**Moderate?**	Low	Low	**Low**	**MAINTAINED?**
Lower Middle Fork Salmon R.	1,000			**Moderate?**	Very Low	Low	**Low**	**MAINTAINED?**
Upper Middle Fork Salmon R.	1,000			**Moderate?**	Very Low	Low	**Low**	**MAINTAINED?**
Panther Creek	500	NA	NA	**Moderate**	High	Moderate	**High**	**HIGH RISK?**
North Fork Salmon R.	500	NA	NA	**Moderate**	Low	Moderate	**Moderate**	**MAINTAINED?**

Pahsimeroi R.	1,000	NA	NA	**Moderate**	Moderate	Moderate	**Moderate**	**MAINTAINED?**
East Fork Salmon R.	1,000	NA	NA	**Moderate**	Very Low	Moderate	**Moderate**	**MAINTAINED?**
Up Main. Salmon R.	1,000	NA	NA	**Moderate**	Very Low	Moderate	**Moderate**	**MAINTAINED?**

UPDATED BIOLOGICAL RISK SUMMARY

Four out of the five MPGs are not meeting the specific objectives in the draft Recovery Plan based on the updated status information available for this review, and the status of many individual populations remains uncertain (Table 33). The Grande Ronde MPG is tentatively rated as viable, but more specific data on spawning abundance and the relative contribution of hatchery spawners for the Lower Grande Ronde and Wallowa populations would improve future assessments. The additional monitoring programs instituted in the early 2000's to gain better information on natural origin abundance and related factors have significantly improved our ability to assess status at a more detailed level. The new information has resulted in an updated view of the relative abundance of natural origin spawners and life history diversity across the populations in the DPS. The more specific information on the distribution of natural returns among stock groups and populations indicates that differences in abundance/productivity status among populations may be more related to geography or elevation rather than A run vs. B run. Based on these results, the major life history category designations for populations in the DPS have been updated (Table 33). A great deal of uncertainty still remains regarding the relative proportion of hatchery fish in natural spawning areas near major hatchery release sites within individual populations. Overall, the information analyzed for this status review does not indicate a change in biological risk status.

MIDDLE COLUMBIA RIVER STEELHEAD DPS

BRIEF DESCRIPTION OF ESU

The Middle Columbia River steelhead distinct population segment (DPS) includes all naturally spawning populations of steelhead (*Oncorhynchus mykiss*) using tributaries upstream and exclusive of the Wind River (Washington) and the Hood River (Oregon), excluding the Upper Columbia River tributaries (upstream of Priest Rapids Dam) and the Snake River (Figure 45). The Middle Columbia River steelhead DPS was listed as threatened by NOAA Fisheries in 1999, with that listing designation being affirmed in 2006 and 2012.

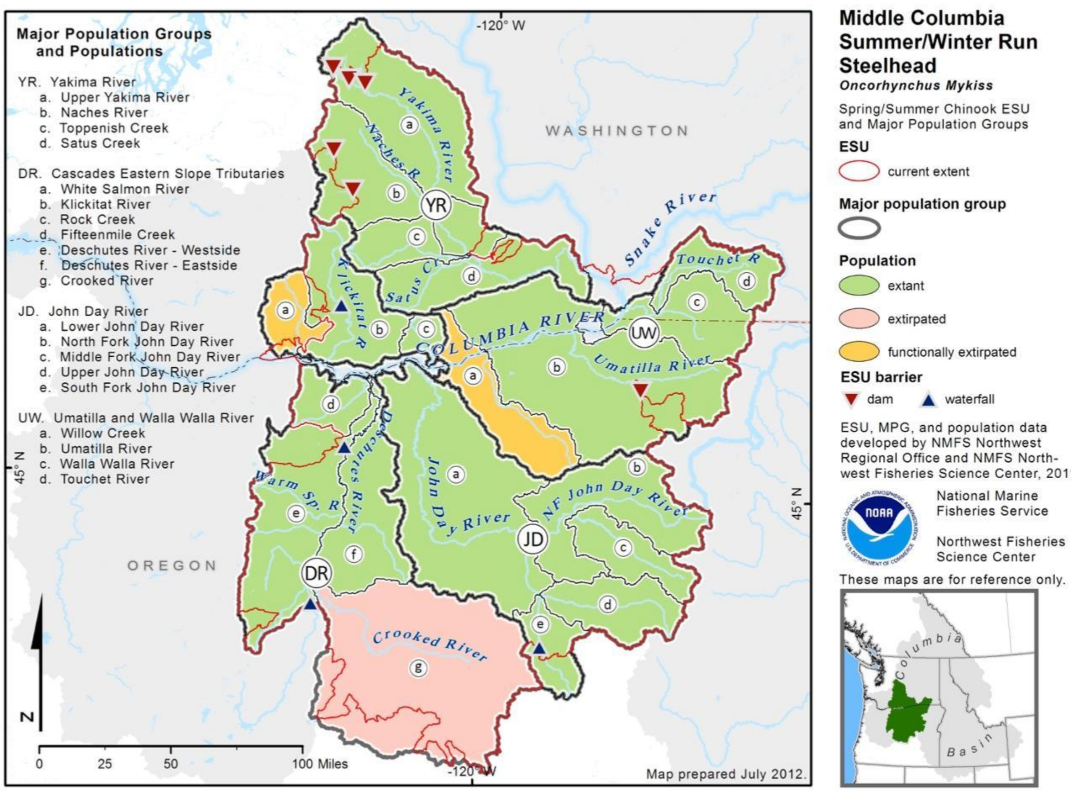

Figure 45 -- Map of the Middle Columbia River steelhead DPS's spawning and rearing areas, illustrating populations and major population groups.

NOAA Fisheries has defined DPSs of steelhead to include only the anadromous members of this species (70 FR 67130). Our approach to assessing the current status of a steelhead DPS is based evaluating information the abundance, productivity, spatial structure and diversity of the anadromous component of this species (Good *et al*. 2005; 70 FR 67130). Many steelhead populations along the West Coast of the U.S. co-occur with conspecific populations of resident rainbow trout. We recognize that there may be situations where reproductive contributions from resident rainbow trout may mitigate short-term extinction risk for some steelhead DPSs (Good *et al*. 2005; 70 FR 67130). We assume that any benefits to an anadromous population resulting from the presence of a conspecific resident form will be reflected in direct measures of the current status of the anadromous form.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

Results of a BRT review of the status of the Middle Columbia Steelhead DPS were summarized in Good *et al*. (2005). A slight majority (51%) of the cumulative scores across the BRT were for assigning this DPS to the "threatened but not endangered" category. The remaining votes (49%) were for the "not likely to become endangered" designation. The BRT noted that this particular DPS was difficult to evaluate. Reasons cited included: the wide range in relative abundance for individual populations across the DPS (e.g., spawning abundance in the John Day and Deschutes basins had been relatively high, while returns to much of the Yakima River drainage had remained relatively low); chronically high levels of hatchery strays into the Deschutes River, and a lack of consistent information on annual spawning escapements in some tributaries (e.g. Klickitat River). In addition, resident *O. mykiss* are believed to be very common throughout this DPS. The BRT assumed that the presence of resident *O. mykiss* below anadromous barriers mitigated extinction risk to the DPS to some extent, but a slight majority of BRT members concluded that significant threats to the anadromous component remained.

2010

Ford *et al*. (2011) concluded that there had been improvements in the viability ratings for some of the component populations, but the Mid-Columbia Steelhead DPS was not currently meeting the viability criteria in the Mid-Columbia Steelhead Recovery Plan. In addition, several of the factors cited by the 2005 BRT (Good *et al.* 2005) remained as concerns or key uncertainties. Natural origin spawning estimates were highly variable relative to minimum abundance thresholds across the populations in the DPS. Updated information indicated that stray levels into at least the Lower John Day River population were also high. Returns to the Yakima River basin and to the Umatilla and Walla Walla Rivers had been higher over the most recent brood cycle while natural origin returns to the John Day River had decreased. Out of basin hatchery stray proportions, although reduced, remained very high in the Deschutes River basin. Overall the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Updated abundance and hatchery contribution estimates have been provided by regional fisheries managers for each of the 15 long term data series considered in prior status reviews. In addition, the two years of mark recapture based escapement estimates of wild and hatchery origin steelhead entering the Klickitat River first reported in the 2011 review have been extended to include 2008-14 returns and the first samples of steelhead adults in the Rock Creek (Yakima; Figure 45) population have been reported (Harvey 2014).

Abundance estimates for the Yakima River populations continue to be based on steelhead counts at Prosser Dam on the mainstem Yakima below all four of the populations in this MPG. Population specific abundance estimates for return years 1985-2009 are based on a run reconstruction allocation method that incorporates average distributions observed in a three year radio tagging study (Frederiksen *et al.* 2014) in the early 1990s along with Roza Dam counts and redd counts in Satus and Toppenish Creek. Population specific estimates of the 2012-2014 brood year escapements were generated from a three year radio-tagging study. In addition, two other methods were applied over the duration of that study, a genetic stock identification approach and a PIT tag based tracking program. Regional biologists are reviewing the results of those efforts. A full analysis of the results from the three year radio tracking study is being completed, including a comparative assessment across methods that could lead to recommendations for a long term monitoring approach (Frederiksen *et al*. 2014). Preliminary results suggest that the PIT tag based approach, which involves proportional tagging at Prosser combined with strategically placed upstream arrays, would be a viable long term strategy.

WDFW regional biologists have updated the methodology used to generate steelhead spawner abundance estimates for the Touchet River. The updated estimates are based on annual redd counts in the mainstem above the town of Dayton and include an adjustment to include spawners in two tributaries entering below that reach (Coppei and Waits Creeks). Age composition and hatchery/natural proportions for spawning in the reach above the Dayton are based on sampling at a mainstem weir at Dayton. Hatchery spawner proportions are adjusted to account for differential removals of hatchery fish at the weir and for the endemic broodstock program (natural returns).

Resident Contributions to Anadromous Production

Many steelhead (*O. mykiss*) populations along the West Coast of co-occur with conspecific populations of resident rainbow trout. Previous NWFSC status reviews (e.g. Ford *et al*. 2011) have recognized that there may be situations where reproductive contributions from resident rainbow trout could mitigate short-term extinction risk for some steelhead DPS populations (Good *et al*. 2005; 70 FR 67130). In general, we assume that any benefits to an anadromous population resulting from the presence of a conspecific resident form will be reflected in direct measures of the current status of the anadromous form. Potential contribution rates of co-occurring resident production to anadromous returns vary considerably among populations as a function of habitat and survival patterns (Satterthwaite *et al.* 2010). In th**e** Middle Columbia DPS, a study in the Deschutes River Basin found no evidence of a significant contribution from the very abundant resident form to anadromous returns (Zimmerman & Reeves 2000). A recent study of natural origin steelhead kelts in the Yakima Basin, comparing chemical patterns in otoliths with water chemistry sampling, found evidence for variable maternal resident contribution rates to andromous returns, with a high degree of variation among natal areas and across years (Courter *et al.* 2013). The Satus River had the lowest sampled proportions of maternal resident patterns (<8% of samples in 2011 and 2012). The highest proportions were for fish that assigned to the Lower Yakima basin (38% and 17%). Toppenish Creek and Naches River were intermediate. The authors note that the ability to discriminate among natal rearing areas in the study could be improved by expanding the number of geochemical markers in the regional water sampling and otolith analyses.

Upstream Passage Losses

The increasing use of PIT tags applied to representative samples from steelhead populations (both natural production and hatchery releases) has identified relatively high loss rates of returning adults from specific populations, either as mortalities or as strays into non-natal basins (http://www.dfw.state.or.us/fish/crp/mid_columbia_river_plan_WASTB_workshop.asp). The following examples are from presentations available at the workshop website. In 2013, 1325 PIT tagged fish produced in the John Day River basin were detected passing above Bonneville Dam and 13% of those tagged fish directly migrated into the John Day River based on detections at Lower John Day mainstem arrays. A relatively high proportion (71%) of the adults detected at Bonneville Dam continued upriver past the John Day and were next detected at McNary Dam. After overwintering, 616 of those fish dropped back and entered the John Day River. Accounting for both the direct and delayed entries, approximately 57% of the returns detected at Bonneville eventually entered the natal basin. High rates of overshooting were also indicated for some other Mid-Columbia steelhead populations. A proportion of the returning adults tagged as juveniles in the Yakima River Basin initially migrated upstream into the Upper Columbia River, although a relatively high proportion did eventually fall back to be detected entering the Yakima River.

Genetic analyses of juvenile *O. mykiss* sampled in the Rock Creek drainage indicate a relatively high similarity to Snake River DPS, suggesting relatively high stray rates from that region into Rock Creek (Matala 2012). Sampling adult spawners in Rock Creek, including conducting PBT based analysis of any hatchery fish, would clarify the current stock status. Matala (2012) also suggests that analysis of archival samples (if any exist) would provide insights into whether historical genetic patterns for this and other Mid-Columbia DPS populations also reflect high exchange rates with Snake River DPS, or whether the current patterns are a relatively recent change.

John Day River Studies

ODFW sampling programs in the John Day River basin continue to provide information on adult spawner abundance, juvenile productivity and genetic structure (Banks *et al.* 2013; Banks *et al.* 2014b; Bare *et al.* 2015). Spawner abundance estimates generated or extrapolated from EMAP sampling in the basin are included in the updated abundance and productivity assessments described above.

Estimates of outmigrant smolt production based on smolt trapping are available for a limited number of years for the Middle Fork and South Fork populations. The patterns in production vs. parent redd counts are consistent with density dependent relationships, although more data pairs for each series will be necessary to derive specific functional relationships.

Proportions of out of basin hatchery steelhead in John Day natural spawning areas have declined substantially in recent years (Figure 51), with the declines being negatively correlated with the proportion of Snake River outmigrants that are barge transported (Banks *et al.* 2013; Banks *et al.* 2014b). As in prior years, hatchery origin spawners were concentrated in the Lower John Day population tributaries.

Genetic sampling data from specific reaches in the John Day basin showed some differentiation, but did not directly correspond to the population structure inferred from geographic separation and dispersal rate assumptions hypothesized by the ICTRT (2003). In most cases there was temporal correlation among samples taken from the same sites over years, but differences among sites were not significant. Exceptions to this general pattern included Indian, Belshaw and Reynolds Creeks. Indian Creek is a reach above a series of cascades and may be dominated by local resident trout production. There is evidence of cutthroat/steelhead hybridization in Belshaw and Reynolds Creeks that could be contributing to their relative genetic distinctiveness.

Yakima Genetics

Results from the analysis of genetic samples taken in subareas across the Yakima River drainage generally support the hierarchical population structure identified by the ICTRT with one significant exception (T. Seamons, pers. comm.). The Naches ICTRT population designation was not fully supported by the genetic data. Collections from the Naches River upstream of the confluence with the Tieton River clustered together, but collections from the lower Naches River tributaries and tributaries to the mainstem Yakima River did not group with upper Naches collections and did not form a group of their own. Instead, these populations appeared to be a mix of Naches and Upper Yakima ancestry, which may reflect the ancestral state or may be a result of more recent natural and anthropogenic influences.

Fifteen Mile Creek Life History patterns

Fifteen Mile Creek is one of two extant natural-origin populations at the western edge of the Mid Columbia Steelhead DPS. Steelhead runs in the downstream neighboring DPS (Lower Columbia River) are generally winter run. ODFW had classified the Fifteen Mile Creek population as winter run prior to recent PIT tag studies. Returning natural origin steelhead PIT tagged as juveniles in the mainstem Fifteen Mile Creek watershed exhibit a summer timed return pattern, similar to other populations in the middle Columbia River DPS (Poxon *et al.* 2014). The Fifteen Mile Creek population includes some smaller tributaries downstream of the Fifteen Mile Creek drainage. It is possible a component of natural production associated with those small streams is winter run. ODFW has observed that genetic analyses might resolve the potential existence of a winter run component.

Smolt to adult return rates

Smolt to adult return survival estimates (SARs) for an average of three representative indices -- Umatilla River, Warm Springs tributary, and the aggregate natural Snake River steelhead run -- are available for outmigration years 1964 through 2011 (Figure 46). Year to year variations in SAR represent a major influence on the annual returns of natural origin steelhead although the pattern is complicated by the fact that multiple broods (predominately ages 3-6) contribute to each particular return year escapement. The relatively high adult returns in the mid-1980s as well as the early and late 2000's correspond to higher average SARs for the corresponding brood years.

Representative SAR series for other Interior Basin ESUs and DPSs show similar general patterns in recent years, indicating that they may be subject to some of the same influences during the smolt to adult phase (Figure 46). Both Chinook series show peaks in roughly the same time periods although there are some differences in the timing and magnitude of year to year variations.

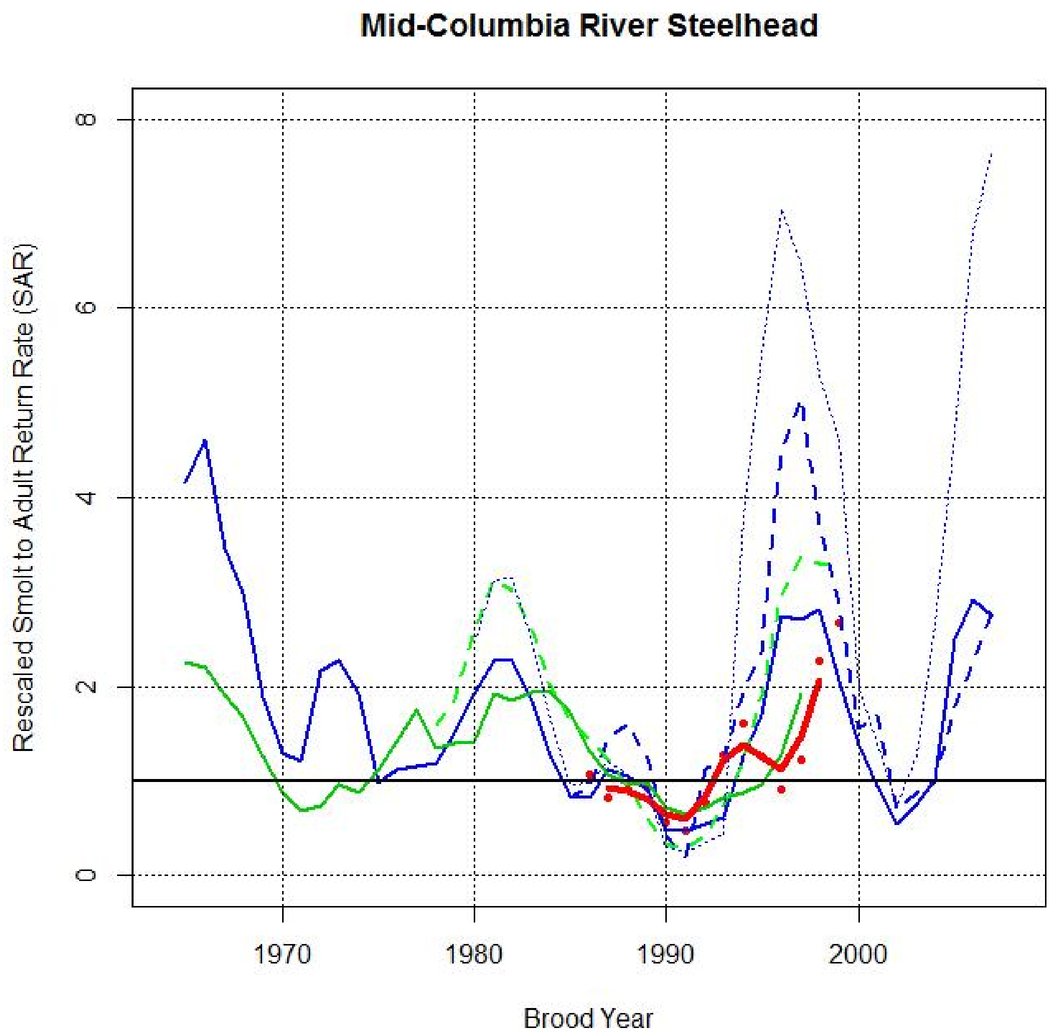

Figure 46 - Mid-Columbia River natural origin steelhead aggregate smolt to adult return rates (red points and heavy line). Aggregate SARs for other Interior Columbia basin ESUs and DPSs provided for comparison. Snake River aggregate spring/summer Chinook salmon (solid blue), Snake River aggregate natural origin steelhead (dashed green), Tuccannon spring Chinook salmon (dotted blue), Upper Columbia spring Chinook salmon (blue dashed line), Upper Columbia steelhead (green dashed line). Each SAR series is rescaled by dividing annual values by the corresponding series mean to faclilitate relative comparison. Lines are three year moving averages.

ABUNDANCE AND PRODUCTIVITY

Evaluations were done using both a set of metrics corresponding to those used in prior Biological Review Team (BRT) reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The BRT level metrics were consistenly done across all ESUs and DPSs to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the Recovery evaluation section below.

Total escapement and natural-origin escapements increased relative to the prior five year review for all five of the John Day populations (Table 34). Four out of the five populations in this group had a positive 15 year trend in natural origin abundance (Table 35) driven largely by peak returns in the early 2000's and the most recent five year period (Figure 47). The Lower Mainstem population was the exception; the recent peak in returns was relatively low compared to prior years and its 15 year trend was slightly negative (Table 35).

Five year geometric mean natural origin and total abundance estimates for each of the four populations in the Yakima River MPG also increased relative to the prior review (Table 34). All four populations in this group have exhibited relatively steady increases since the early 1990s, with similar peak return years as other DPS populations (Figure 47).

Total spawning escapements have increased in the most recent brood cycle for all three populations in the Umatilla-Walla Walla MPG as well, although the proportional increases were on below those for most populations in the John Day and Yakima MPGs (Table 34). The 15 year trend in natural origin abundance was positive for the Umatilla River population and slightly negative for the Touchet River (Table 35, Figure 47). The data series for the Walla Walla River population is relatively short, with no apparent trend since the initial estimates in the mid-1990s.

Abundance data series are available for three of the five extant populations in the East Cascades MPG along with 7 years of estimates for a fourth population (Klickitat River). Spawner abundance estimates for the most recent five years increased relative to the prior review for the Umatilla, Walla Walla and Touchet River populations (Table 34). The 15 year trend in natural origin spawners was positive for the West Side Deschutes population, and negative for the Fifteen Mile and East Side Deschutes runs (Table 35). Based on mark-recapture analysis, the recent five year (2010-14) geometric mean passage of steelhead over Lyle Falls in the Lower Klickitat River has been 1,358 natural origin and 2,726 hatchery fish (Zendt *et al.* 2013). There is evidence that unknown portions of both components fall back after initially ascending through Lyle Falls. There is significant tribal and sport harvest associated with the Klickitat steelhead run, with the sport harvest being targeted on hatchery fishVirtually all tribal harvest occurs below Lyle Falls, and sport harvest is currently recorded as to below or above Lyle Falls. So the Lyle Falls mark recap estimate does represent escapement past the primary fishery harvest – it does not account fall back, hook/release sport fishery mortality or other pre-spawn mortality occurring above the falls (J. Zendt, pers. comm.). Preliminary estimates of escapements into Rock Creek were recently developed, and a high proportion of the observed steelhead in that system were out of basin strays (Harvey 2014).

Populations in all four of the mid-Columbia steelhead MPGs exhibited similar temporal patterns in brood year returns per spawner (Figure 48). Return rates for brood years 1995–1999 generally exceeded replacement (1:1). Spawner to spawner ratios for brood years 2001–2003 were generally well below replacement for many populations. Brood year return rates reflect the combined impacts of year to year patterns in marine life history stages, upstream and downstream passage survivals as well as density dependent effects resulting from capacity or survival limitations on tributary spawning or juvenile rearing habitats.

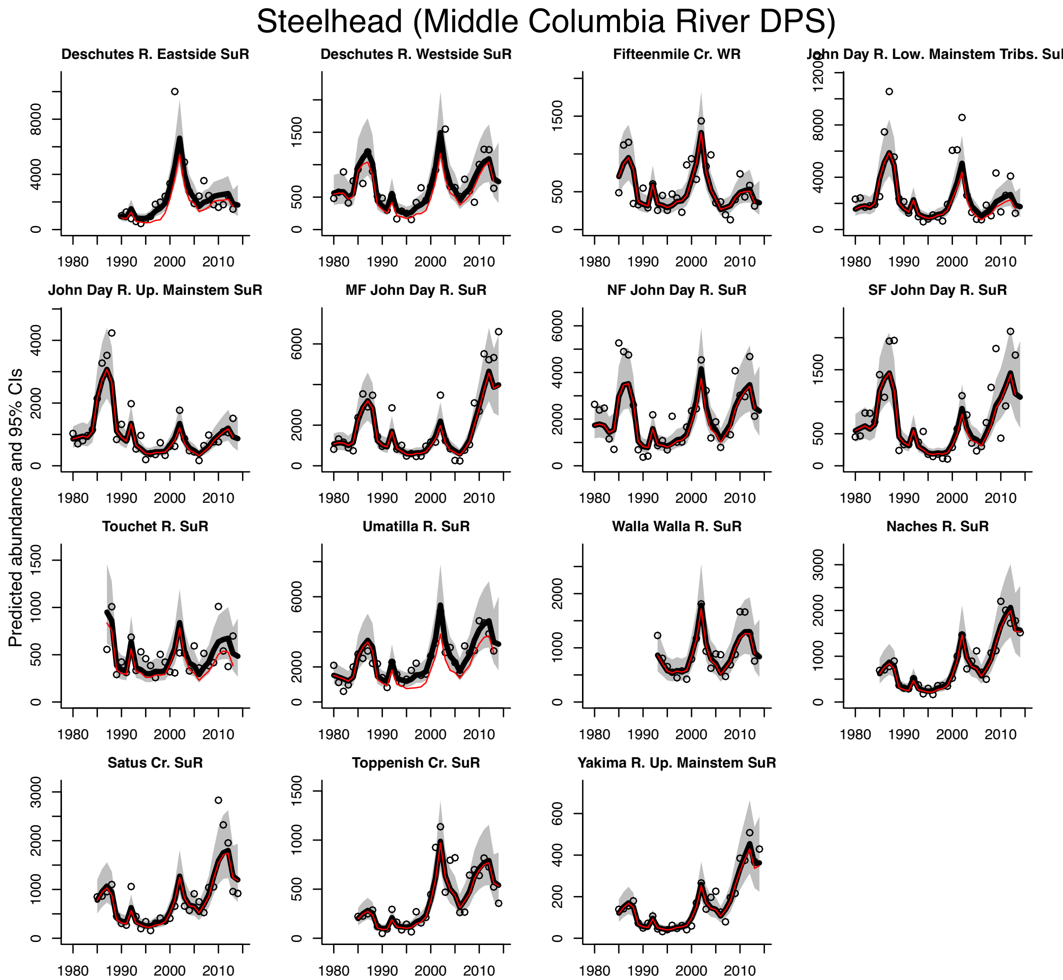

Figure 47 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

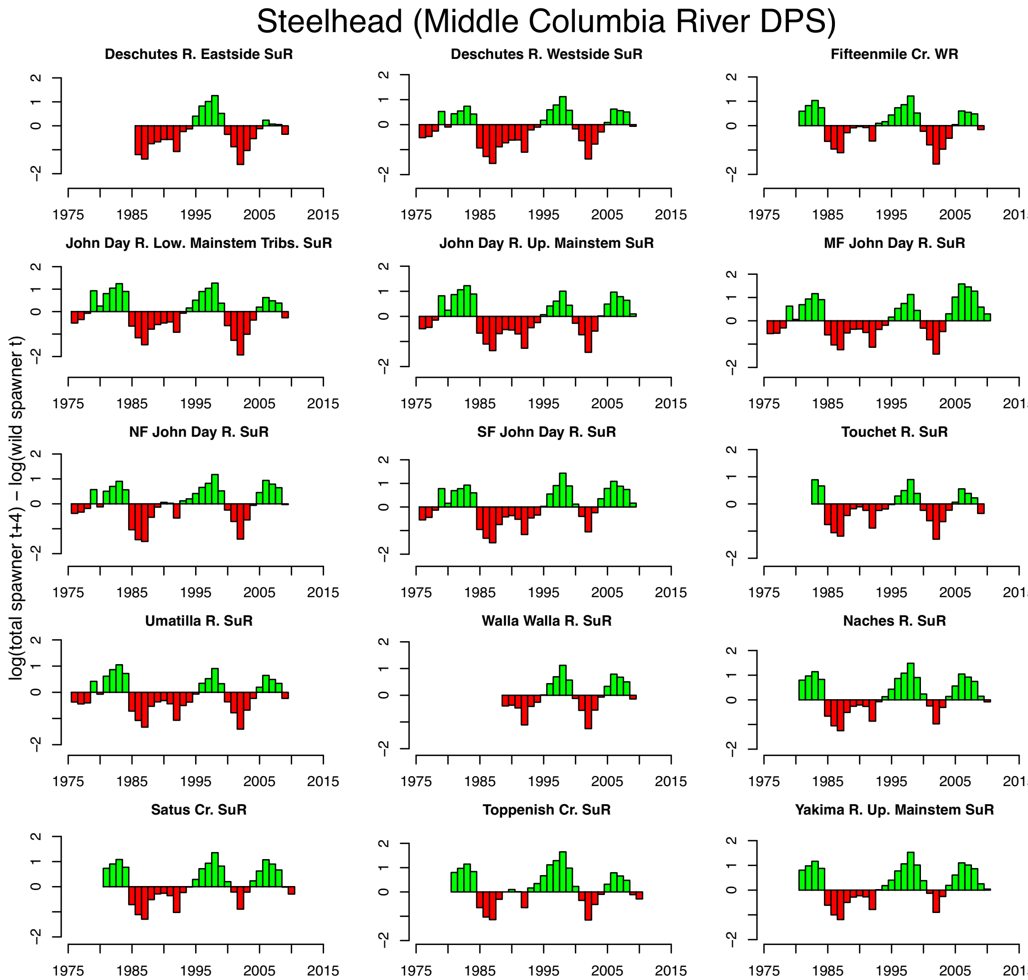

Figure 48 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4). Spawning years on x axis.

Table 34 -- 5-year geometric mean of raw natural spawner counts. This is the raw total spawner count times the fraction wild estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% C
Deschutes R. Eastside SuR	Cascades E. Slope Tribs.	607 (761)	693 (1439)	3823 (4848)	1872 (2354)	1540 (1803)	-18
Deschutes R. Westside SuR	Cascades E. Slope Tribs.	248 (323)	226 (341)	742 (950)	477 (578)	935 (993)	96
Fifteenmile Cr. WR	Cascades E. Slope Tribs.	405 (405)	396 (396)	941 (941)	264 (264)	471 (490)	78
John Day R. Low. Mainstem Tribs. SuR	John Day R.	1235 (1248)	968 (1017)	3487 (4052)	1024 (1382)	1745 (2059)	70
John Day R. Up. Mainstem SuR	John Day R.	1019 (1029)	350 (368)	695 (777)	471 (512)	1050 (1072)	123
MF John Day R. SuR	John Day R.	1210 (1225)	545 (572)	1229 (1375)	634 (689)	4776 (4864)	653
NF John Day R. SuR	John Day R.	785 (793)	1142 (1200)	2247 (2514)	1488 (1618)	3011 (3073)	10[illegible]
SF John Day R. SuR	John Day R.	398 (402)	135 (142)	493 (551)	586 (637)	1077 (1099)	84
Touchet R. SuR	Umatilla/Walla Walla R.	392 (438)	342 (395)	354 (387)	337 (446)	489 (615)	45
Umatilla R. SuR	Umatilla/Walla Walla R.	1068 (1344)	919 (1660)	2341 (3312)	1931 (2498)	3214 (3921)	66
Walla Walla R. SuR	Umatilla/Walla Walla R.	995 (995)	516 (522)	957 (997)	717 (739)	1239 (1274)	73
Naches R. SuR	Yakima R. Group	285 (313)	260 (293)	855 (868)	823 (846)	1775 (1829)	116
Satus Cr. SuR	Yakima R. Group	343 (377)	266 (300)	640 (652)	807 (829)	1585 (1624)	96
Toppenish Cr. SuR	Yakima R. Group	103 (113)	135 (153)	693 (705)	468 (481)	575 (588)	23
Yakima R. Up. Mainstem SuR	Yakima R. Group	55 (56)	49 (50)	145 (149)	155 (157)	390 (410)	152

Table 35 -- 15-year trends in log natural spawner abundance computed from a linear regression applied to the smoothed natural spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Deschutes R. Eastside SuR	Cascades E. Slope Tribs.	0.12 (0.05, 0.18)	-0.02 (-0.07, 0.
Deschutes R. Westside SuR	Cascades E. Slope Tribs.	0.08 (0.03, 0.13)	0.03 (-0.01, 0.
Fifteenmile Cr. WR	Cascades E. Slope Tribs.	0.05 (0.01, 0.1)	-0.05 (-0.09,
John Day R. Low. Mainstem Tribs. SuR	John Day R.	0.03 (-0.02, 0.09)	-0.02 (-0.07, 0.
John Day R. Up. Mainstem SuR	John Day R.	-0.02 (-0.07, 0.03)	0.03 (-0.01, 0.
MF John Day R. SuR	John Day R.	0 (-0.04, 0.05)	0.11 (0.05, 0.
NF John Day R. SuR	John Day R.	0.07 (0.03, 0.11)	0.03 (-0.02, 0.
SF John Day R. SuR	John Day R.	0.03 (-0.02, 0.08)	0.1 (0.06, 0.1
Touchet R. SuR	Umatilla/Walla Walla R.	0.02 (-0.01, 0.06)	0 (-0.04, 0.0
Umatilla R. SuR	Umatilla/Walla Walla R.	0.06 (0.02, 0.11)	0.04 (0, 0.0
Walla Walla R. SuR	Umatilla/Walla Walla R.	0 (-0.05, 0.04)	0.01 (-0.03, 0.
Naches R. SuR	Yakima R. Group	0.1 (0.05, 0.15)	0.08 (0.04, 0.
Satus Cr. SuR	Yakima R. Group	0.07 (0.03, 0.12)	0.07 (0.04, 0.
Toppenish Cr. SuR	Yakima R. Group	0.15 (0.1, 0.19)	0.03 (-0.01, 0.
Yakima R. Up. Mainstem SuR	Yakima R. Group	0.09 (0.04, 0.14)	0.1 (0.06, 0.1

HARVEST

Summer-run steelhead from the upper basin are divided into 2 runs by managers: The A-run, and the B-run. These runs are believed have differences in timing, but managers separate them on the basis of size alone in estimating the size of the runs. The A-run is believed to occur throughout the Middle Columbia, Upper Columbia, and Snake River Basins, while the B-run is believed to occur naturally only in the Snake RiverDPS, contributing in varying proportions, in the Clearwater River, Middle Fork Salmon River, and South Fork Salmon River.

Steelhead were historically taken in tribal and non-tribal gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and in tributaries. In the 1970s, retention of steelhead in non-tribal commercial fisheries was prohibited, and in the mid 1980s, tributary recreational fisheries in Washington adopted mark-selective regulations. Steelhead are still harvested in tribal fisheries, in mainstem recreational fisheries, and there is incidental mortality associated with mark-selective recreation recreational fisheries. The majority of impacts on the summer run occur in tribal gillnet and dip net fisheries targeting Chinook salmon. Sport fisheries targeting hatchery run steelhead occur in the mainstem Columbia River and in several Mid-Columbia River tributaries. In recent years, total mainstem Columbia River exploitation rates on the A-run have been stable at around 5% (Figure 49).

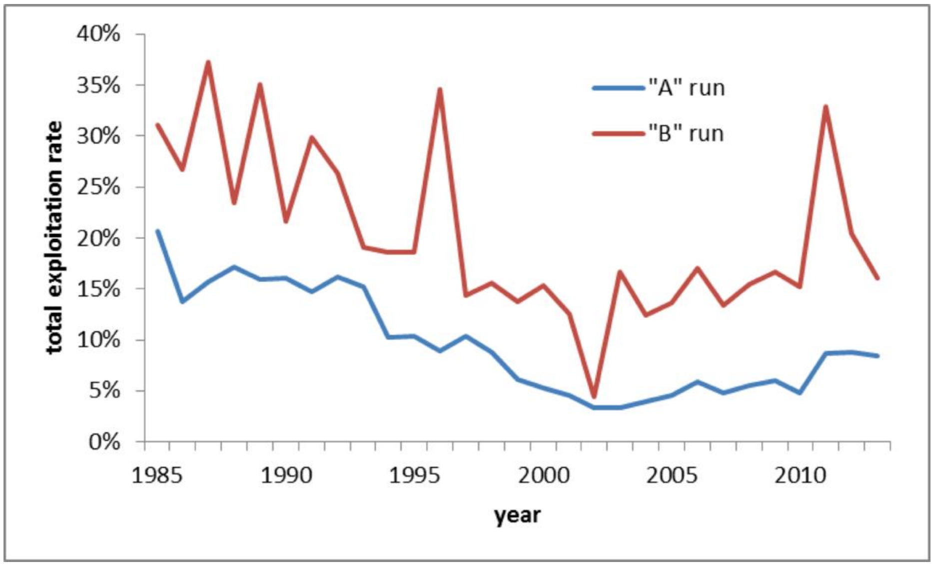

Figure 49 -- Total harvest impacts on natural summer steelhead above Bonneville Dam. Data for 1985-1998 from NMFS biological opinion (Peter Dygert, NMFS, personal communication), and for 1999-2008 from TAC run reconstruction (Joint Staff, 2014).

Few winter-run fish migrate above Bonneville Dam where tribal fisheries occur. In addition, winter-run steelhead are in the mainstem river at a time when there is generally little or no fishing occurring there. The Klickitat River Steelhead population within the Mid-Columbia Steelhead DPS has a winter run component, although anadromous production is dominated by summer run timing. The ICTRT classified Fifteen Mile Creek, another Mid-Columbia DPS population located in the downstream extent of the DPS, as winter run although recent information summarized in this assessment indicates that its core production area exhibits summer run timing. Recreational fisheries in Washington tributaries have been mark selective since the mid-1980s. Because very few of the fish ascend above Bonneville Dam, there was little focus on this run prior to listing. Total non-tribal fishery impact rates for the natural component are only available back to 2001 (Figure 50). In that time period, estimated impact rates have been in the range of 1.5% to 3% except for 2002.

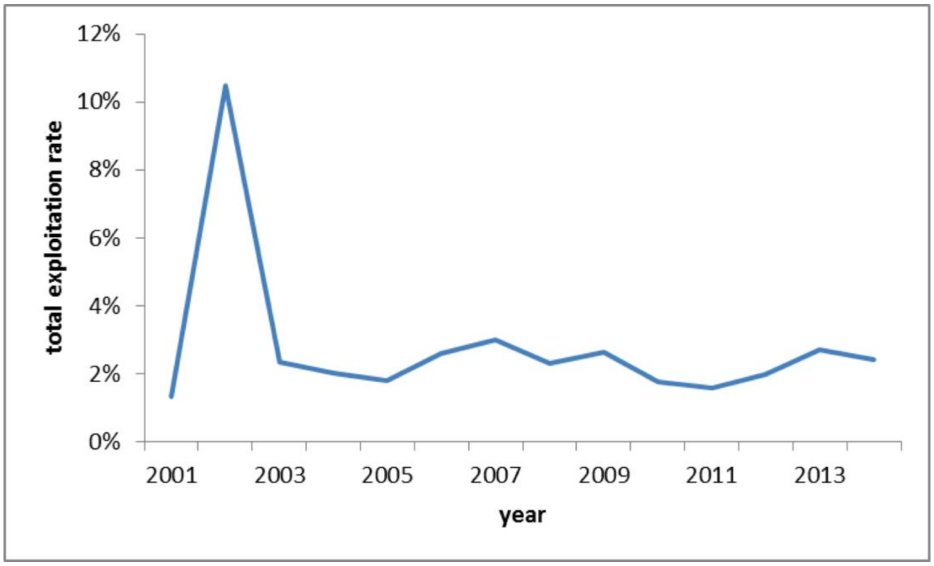

Figure 50 -- Total exploitation rates in non-treaty fisheries on natural winter steelhead from the Columbia Basin. Winter steelhead include the Lower Columbia River ESU, Upper Willamette River ESU, and portions of the Middle Columbia River and Washington Coastal ESUs. Data form TAC run reconstruction (TAC, 2015).

SPATIAL STRUCTURE AND DIVERSITY

Updated information on spawner and juvenile rearing distribution does not support a change in spatial structure status for Mid-Columbia Steelhead DPS populations. Status indicators for within population diversity have changed for some populations, although in most cases the changes have not been sufficient to shift composite risk ratings for a particular population.

In the Cascades Eastern Slope MPG, Fifteen Mile Creek remains rated at low risk for spatial structure and diversity. Spawning distributions mimic inferred historical patterns, life history diversity and phenotypic characteristics are believed to be intact and adult sampling indicates low contributions from straying out of basin hatchery stocks. Additional information obtained from spawner distribution and genetic sampling in the Klickitat River support the low risk rating for spatial structure and suggest that the current moderate rating for within population diversity may improve as additional years data accumulate. The current diversity risk rating of moderate was largely based on uncertainty about effects of the ongoing hatchery program in the basin. Initial results indicate that the separation in time and space between hatchery origin and wild spawners has been effective in minimizing introgression. Indices for both spatial structure and diversity risk for the Westside Deschutes population remain at moderate risk. The spatial structure rating is due to the loss of natural production from above Pelton/Round Butte. The Eastside Deschutes population is rated at low risk for spatial structure. Both populations are rated at moderate risk for diversity based on reductions in life history diversity as a result of habitat degredation and potential genetic impacts resulting from chronic and widespread hatchery straying from out of basin stocks. The most recent five year average proportion wild for spawners in both populations is higher than in the prior review (Table 34). Specific information on spawner distribution and composition for the other extant population in this MPG, Rock Creek, has become available since the prior review. Spawning in this historically small population appears to be dominated by out of basin strays.

The most recent results from spawner surveys and juvenile sampling are consistent with the moderate risk rating assigned to Umatilla/Walla Walla MPG populations in prior reviews, reflecting the contracted range and the existence of gaps among spawning areas within each population. Diversity risk remains at moderate, with no new information indicating increased life history or phenotypic diversity. Prior reviews have also identified concerns regarding the proportions of out of basin hatchery fish contributing to spawning in all three populations, with the highest proportions being observed in the Umatilla and Touchet Rivers. The downward trend in hatchery origin spawners in the Umatilla River has continued. In the Touchet River system, total hatchery proportions have decreased slightly from the prior review, and there has been a substantial shift towards returns from the test endemic stock program (Bumgarner & Dedloff 2015). Five year average out of basin hatchery contribution rates have declined to just below 2% compared to 13% for the 1995-1999 return years.

The spatial structure ratings for all five populations in the John Day River MPG remains rated at low or very low risk based on recent updated spawner distributions. Habitat conditions believed to limit life history and phenotypic diversity remain relatively unchanged. Hatchery proportions estimated for John Day River populations have declined considerably in recent years (Figure 51). In 2012, the estimated hatchery spawner contribution rate into the aggregate five population John Day River natural production areas was 2%, the lowest since the proportional sampling scheme and PIT tag detection arrays were initiated (Banks *et al*. 2014).

Three of the four populations in the Yakima MPG remain at low risk for structure based on results from the recent radio tag and pit tag studies described above. Distribution across spawning areas within the fourth population, the Upper Yakima River, continues to be substantially reduced from inferred historical levels and is rated at moderate. As with the populations in the Umatilla/Walla Walla MPG, risks due to the loss of life history and phenotypic diversity inferred from habitat degradation (including passage impacts within the Yakima Basin) remain at prior levels. There are no within basin hatchery steelhead releases in the Yakima and outside source strays remain at low levels.

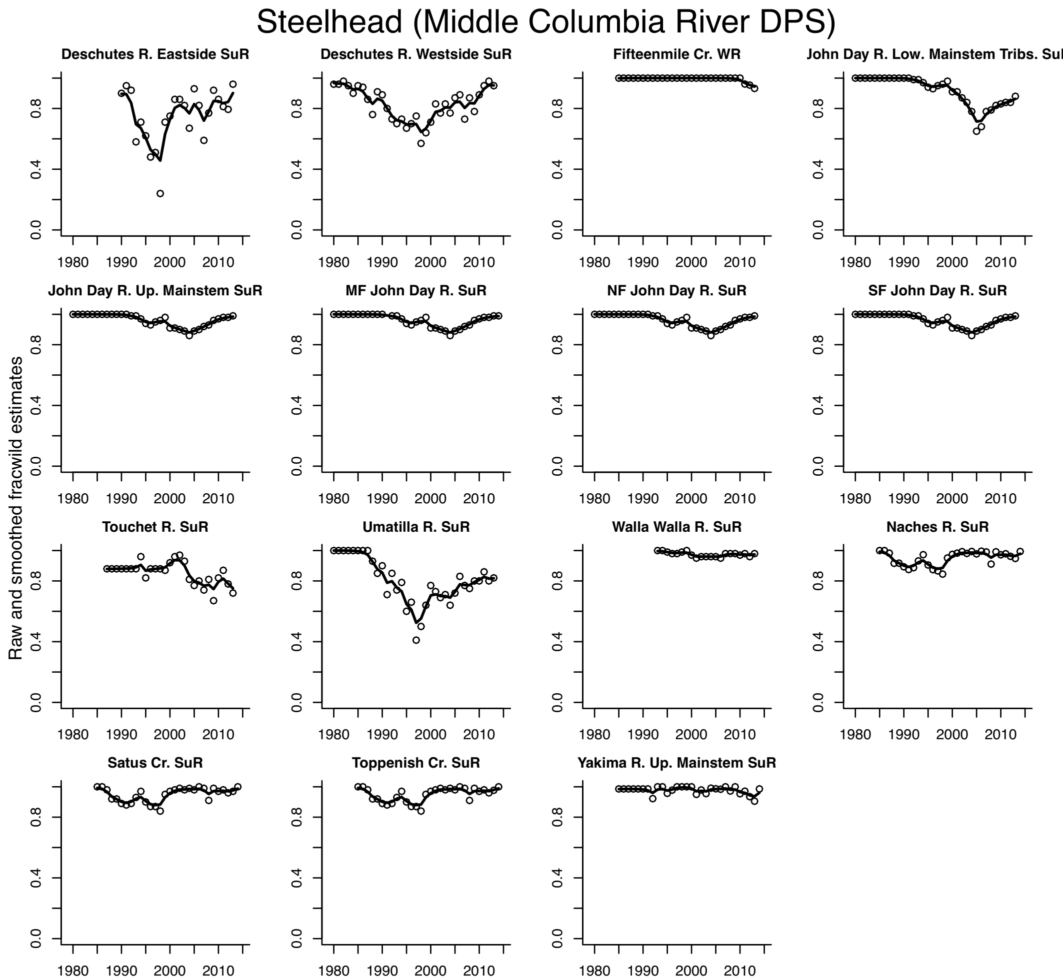

Figure 51 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 36 -- 5-year mean of fraction natural spawners (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Deschutes R. Eastside SuR	0.81	0.51	0.79	0.81	0.86
Deschutes R. Westside SuR	0.77	0.67	0.78	0.83	0.94
Fifteenmile Cr. WR	1.00	1.00	1.00	1.00	0.96
John Day R. Low. Mainstem Tribs. SuR	0.99	0.95	0.86	0.74	0.85
John Day R. Up. Mainstem SuR	0.99	0.95	0.89	0.92	0.98
MF John Day R. SuR	0.99	0.95	0.89	0.92	0.98
NF John Day R. SuR	0.99	0.95	0.89	0.92	0.98
SF John Day R. SuR	0.99	0.95	0.89	0.92	0.98
Touchet R. SuR	0.90	0.87	0.92	0.76	0.80
Umatilla R. SuR	0.80	0.56	0.71	0.77	0.82
Walla Walla R. SuR	1.00	0.99	0.96	0.97	0.97
Naches R. SuR	0.91	0.89	0.99	0.97	0.97
Satus Cr. SuR	0.91	0.89	0.98	0.97	0.98
Toppenish Cr. SuR	0.91	0.89	0.98	0.97	0.98
Yakima R. Up. Mainstem SuR	0.98	0.99	0.97	0.99	0.95

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Recovery strategies outlined in the plan and its management unit components are targeted on achieving, at a minimum, the ICTRT biological viability criteria require that the DPS should "... have all four major population groups at viable (low risk) status with representation of all the major life history strategies present historically, and with the abundance, productivity spatial structure, and diversity attributes required for long-term persistence." The plan recognizes that, at the major population group level, there may be several specific combinations of populations that could satisfy the ICTRT criteria. Each of the management unit plans identifies particular combinations that are the most likely to result in achieving viable major population group status. The recovery plan recognizes that the management unit plans incorporate a range of objectives that go beyond the minimum biological status required for delisting.

The ICTRT recovery criteria are hierarchical in nature, with ESU/DPS level criteria being based on the status of natural-origin steelhead assessed at the population level. A detailed description of the ICTRT viability criteria and their derivation (ICTRT 2007) can be found at www.nwfsc.noaa.gov/trt/col/trt_viability.cfm.

Under the ICTRT approach, population level assessments are based on a set of metrics designed to evaluate risk across the four viable salmonid population elements: A/P, spatial structure, and diversity (McElhany *et al.* 2000). The ICTRT approach calls for comparing estimates of current natural-origin abundance (measured as a 10-year geometric mean of natural-origin spawners) and productivity (estimate of return per spawner at low to moderate parent spawning abundance) against predefined viability curves. In addition, the ICTRT developed a set of specific criteria (metrics and example risk thresholds) for assessing the spatial structure and diversity risks based on current information representing each specific population. The ICTRT viability criteria are generally expressed relative to particular risk threshold—5% risk of extinction over a 100-year period.

The Mid-Columbia Recovery Plan identifies a set of most likely scenarios to meet the ICTRT recommendations for low risk populations at the MPG level. In addition, the management unit plans generally call for achieving moderate risk ratings (maintained status) across the remaining extant populations in each MPG.

John Day River MPG: The lower mainstem John Day River, North Fork John Day River and either the Middle Fork John Day River or upper mainstem John Day River populations should achieve at least viable status. The management unit plan also calls for at least one population to be highly viable, consistent with ICTRT recommendations.

Yakima River MPG: To achieve viable status, two populations should be rated as viable, including at least one of the two classified as large—the Naches River and the upper Yakima River. The remaining two populations should, at a minimum meet the maintained criteria. The management unit plan also calls for at least one population to be highly viable, consistent with ICTRT recommendations.

Umatilla/Walla-Walla MPG: Two populations should meet viability criteria. The management unit plan also calls for at least one population to be highly viable, consistent with ICTRT recommendations. The Umatilla River is the only large population, and therefore needs to be viable. In addition either the Walla Walla River or Touchet River also needs to be viable.

Cascades Eastern Slope MPG: The Klickitat, Fifteen Mile, and both the Deschutes Eastside and Westside populations should reach at least viable status.to meet MPG level viability objectives. The management unit plans also call for at least one population to be highly viable, consistent with ICTRT recommendations. The Rock Creek population should reach maintained status (25% or less risk level). MPG viability could be further bolstered if reintroduction of steelhead into the Crooked River succeeds and if the White Salmon population successfully recolonizes its historical habitat following the upcoming removal of Condit Dam. The ICTRT originally classified the Fifteen Mile Creek population as winter run. Based on the recent information provided by ODFW described above, that designation should be provisionally changed to summer run.

Overall viability ratings for the populations in the Mid-Columbia Steelhead DPS remained generally unchanged from the prior five year review (Table 37). One population, Fifteen Mile Creek, shifted downward from Viable to Maintained status as a result of a decrease in natural origin abundance to below its ICTRT minimum abundance threshold. The Toppenish River (Yakima MPG) dropped in both estimated abundance and productivity but the combination remained above the 5% viability curve and therefore its overall rating remained as Viable. The majority of the populations showed increases in estimates of productivity.

Table 37 – Summary Middle Columbia Steelhead DPS status relative the ICTRT viability criteria, grouped by MPG.

Population	Abundance/Productivity Metrics				Spatial Structure and Diversity Metrics			Overall Viability Rating
	ICTRT Minimum Threshold	*Natural Spawning Abundance*	*ICTRT Productivity*	*Integrated A/P Risk*	*Natural Processes Risk*	*Diversity Risk*	*Integrated SS/D Risk*	
Eastern Cascades MPG								
Fifteen Mile Creek	500	356 (.16)	1.84 (.19)	**Moderate**	Very Low	Low	**Low**	**Maintained**
Deschutes (Westside)	1,500 (1,000)	634 (.13)	1.16 (.15)	**High**	Low	Moderate	**Moderate**	**High Risk**
Deschutes (Eastside)	1,000	1,749 (.05)	2.52 (.24)	**Low**	Low	Moderate	**Moderate**	**Viable**
Klickitat River	1,000			**Moderate??**	Low	Moderate	**Moderate**	**Maintained(?)**
Rock Creek	500				Moderate	Moderate	**Moderate**	**High Risk?**
Crooked River (ext)	*2,000*							***Extirpated***
White Salmon R.(ext)	*500*							***Extirpated.***
Yakima River MPG								
Satus Creek	1,000 (500)	1127 (.17)	1.93 (.12)	**Low**	Low	Moderate	**Moderate**	**Viable**
Toppenish Creek	500	516 (.14)	2.52 (.19)	**Low**	Low	Moderate	**Moderate**	**Viable**
Naches River	1,500	1,244 (.16)	1.83 (.10)	Moderate	Low	Moderate	**Moderate**	**Moderate**
Upper Yakima River	1,500	246 (.18)	1.87 (.10)	**Moderate**	Moderate	High	**High**	**High Risk**
John Day River MPG								
Lower John Day Tribs	2,250	1,270 (.22)	2.67 (.19)	**Moderate**	Very Low	Moderate	**Moderate**	**Maintained**
Middle Fork John Day	1,000	1,736 (.41)	3.66 (.26)	**Low**	Low	Moderate	**Moderate**	**Viable**
North Fork John Day	1,000	1,896 (.19)	2.48 (.23)	**Very Low**	Very Low	Low	**Low**	**Highly Viable**
South Fork John Day	500	697 (.27)	2.01 (.21)	**Low**	Very Low	Moderate	**Moderate**	**Viable**
Upper John Day	1,000	641 (.21)	1.32 (.18)	**Moderate**	Very Low	Moderate	**Moderate**	**Maintained**
Umatilla/Walla Walla MPG								
Umatilla River	1,500	2,379 (.11)	1.20 (.32)	**Moderate**	Moderate	Moderate	**Moderate**	**Maintained**
Walla Walla River	1,000	877 (.13)	1.65 (.11)	**Moderate**	Moderate	Moderate	**Moderate**	**Maintained**
Touchet River	1,000	382 (.12)	1.25 (.11)	**High**	Low	Moderate	**Moderate**	**High Risk**

UPDATED BIOLOGICAL RISK SUMMARY

There have been improvements in the viability ratings for some of the component populations, but the Mid-Columbia River Steelhead DPS is not currently meeting the viability criteria described in the Mid-Columbia Steelhead Recovery Plan. In addition, several of the factors cited by the 2005 BRT remain as concerns or key uncertainties. Natural origin returns to the majority of populations in two of the four MPGs in this DPS increased modestly relative to the levels reported in the previous five year review. Abundance estimates for 2 of 3 populations with sufficient data in the remaining two MPGs (Eastside Cascades and Umatilla/Walla-Walla) were marginally lower. Natural-origin spawning estimates are highly variable relative to minimum abundance thresholds across the populations in the DPS. Three of the four MPGs in this DPS include at least one population rated at low risk for abundance and productivity (Table 37). The survival gaps for the remaining populations are generally smaller than those for the other Interior Columbia Basin listed DPSs (Figure 52). Updated information indicates that stray levels into the John Day River populations have deceased in recent years. Out of basin hatchery stray proportions, although reduced, remain high in spawning reaches within the Deschutes River basin populations. In general, the majority of population level viability ratings remained unchanged from prior reviews for each MPG within the DPS.

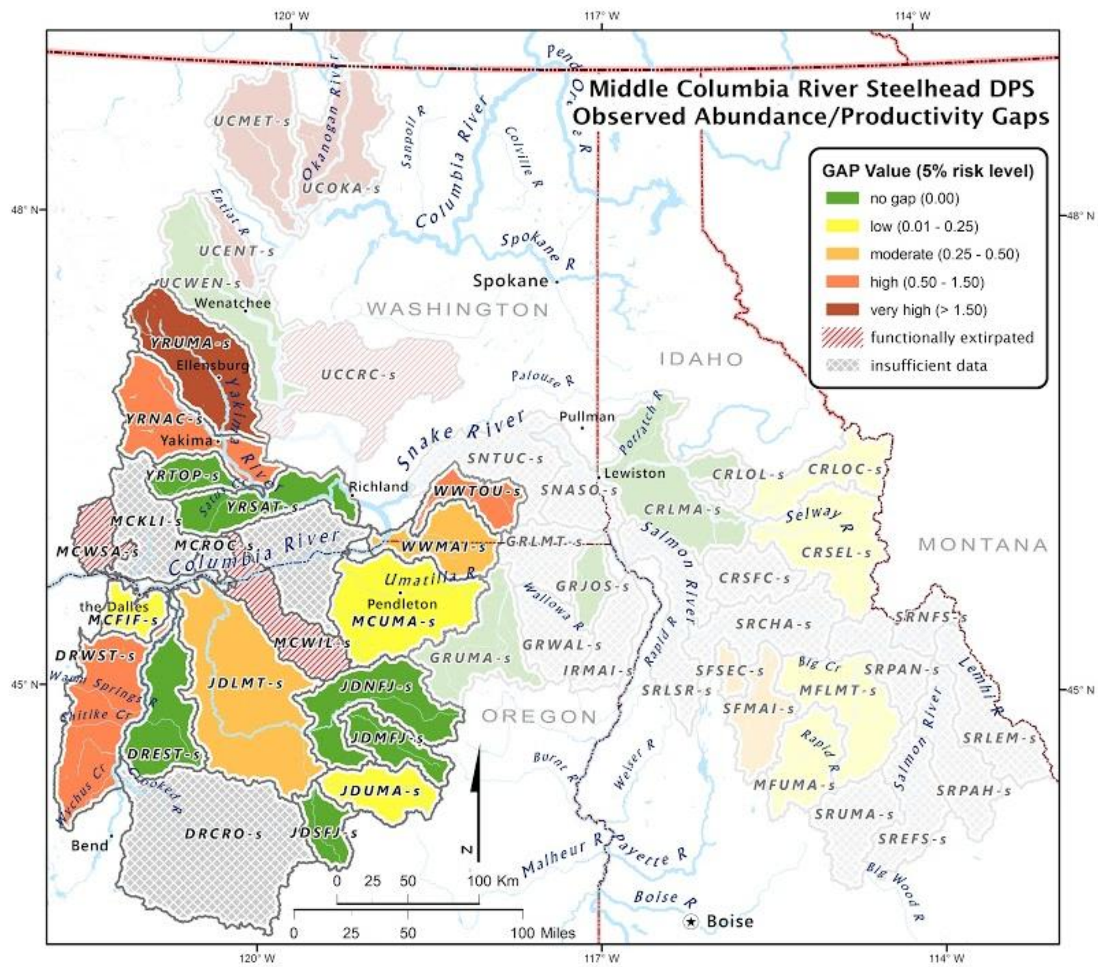

Figure 52 - Mid-Columbia Steelhead population abundance/productivity gaps.). Populations with insufficient data to generate gaps shaded in gray. Gaps are defined as relative improvement in productivity or limiting capacity required for a population to exceed its corresponding 5% risk viability curve (ICTRT, 2007). Gap estimates for populations in the Upper Columbia DPS and Snake River DPS provided for comparison (shaded colors).

LOWER COLUMBIA RIVER DOMAIN STATUS SUMMARIES

LOWER COLUMBIA RIVER CHINOOK SALMON ESU

BRIEF DESCRIPTION OF ESU

The ESU includes all naturally-produced populations of Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon east of the Hood River and the White Salmon River, and includes the Willamette River to Willamette Falls, Oregon (Figure 53), with the exception of spring-run Chinook salmon in the Clackamas River. The ESU spans three distinct ecological regions: Coastal, Cascade, and Gorge. Distinct life-histories (run and spawn timing) within ecological regions in this ESU were identified as major population groups (MPGs). In total, 32 historical demographically-independent populations were identified in this ESU, 9 spring-run, 21 fall-run, and 2 late-fall run, organized in 6 MPGs (based on run timing and ecological region).

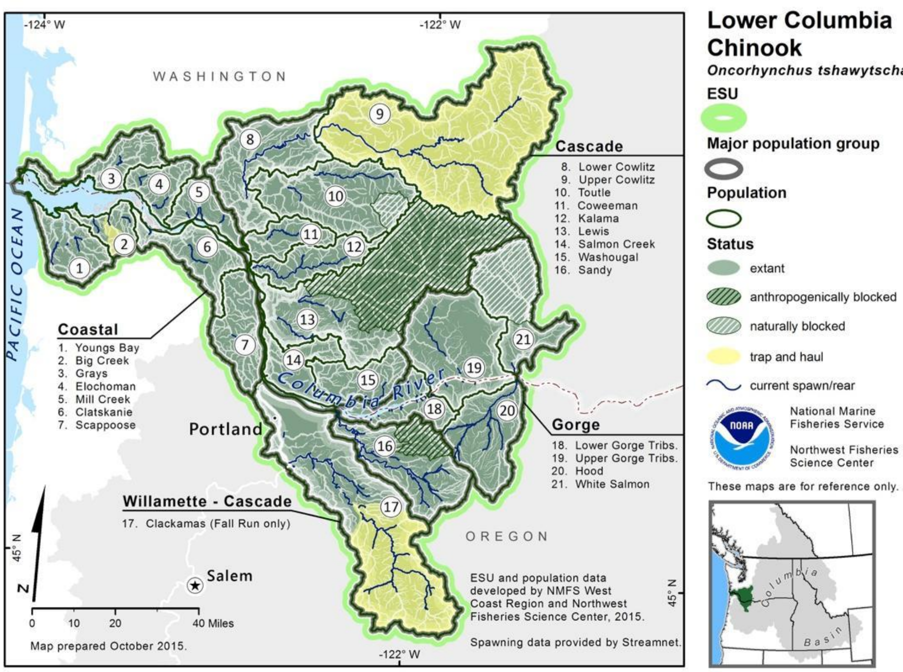

Figure 53 -- Map of the Lower Columbia River Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups. Several watersheds contain or historically contained both fall and spring runs; only the fall-run populations are illustrated here. For some populations access to part or all of their historical spawning habitat is only possible through trap and haul operations (as indicated by textured areas within basins).

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

In the 2005 update, a majority of the BRT votes for the Lower Columbia River Chinook salmon ESU fell in the “likely to become endangered” category, with minorities falling in the “in danger of extinction” and “not likely to become endangered” categories (Good *et al*. 2005). The BRT was still concerned about all of the risk factors identified in the original 1998 review. The WLC-TRT estimated that 8 to 10 historical populations in this ESU had been extirpated, most of them spring-run populations. Near loss of that life history type remained an important BRT concern. Although some natural production appeared to occur in 20 or so populations, only one exceeded an average 1,000 spawners annually. High hatchery production continued to pose genetic and ecological risks to natural populations and to mask their performance. Most populations in this ESU had not seen as pronounced increases in the years leading up to the status review had as occurred in many other geographic areas.

2010

Ford *et al*. (2011) noted that three status evaluations of LCR Chinook status, all based on WLC-TRT criteria, had been conducted since the last BRT status update in 2005. All three evaluations concluded that the ESU was at very high risk of extinction. Of the 32 historical populations in the ESU, 28 were considered extirpated or at very high risk. Based on the recovery plan analyses, all of the tule populations were considered very high risk except one that was considered at high risk. The modeling conducted in association with tule harvest management suggested that three of the populations (Coweeman, Lewis and Washougal) were at a somewhat lower risk. However, even these more optimistic evaluations suggest that the remaining 18 populations were at substantial risk because of very low natural origin spawner abundance (<100/population), high hatchery fraction, habitat degradation and harvest impacts.

Ford *et al*. (2011) noted that spring Chinook populations remained cut-off from access to essential spawning habitat by hydroelectric dams. Projects to allow access had been initiated in the Cowlitz and Lewis systems but in 2010 these were not close to producing self-sustaining populations. Dams were removed on the Sandy River and Hood River; however, it was unclear at the time the review what the benefits of these actions would be. The Sandy River spring Chinook salmon population, was considered at moderate risk and was the only spring Chinook population not considered extirpated or nearly so. The Hood River population contained an out-of-ESU hatchery stock. The two late-fall populations, Lewis and Sandy, were the only populations considered at low or very low risk. They contained relatively few hatchery fish and as of 2010 had maintained high spawner abundances (especially Lewis) since the last BRT evaluation in 2005. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

In the previous Status Review update, the ESU Boundaries Review Group undertook a revaluation of the boundary between all Lower Columbia and mid- Columbia ESUs and DPSs (see Ford *et al*. 2012). The conclusions emphasized the transitional nature of the boundary between the Lower Columbia River ESU and the Mid-Columbia River ESU. After considering new DNA data, the review concludesd: “Given the transitional nature of the Klickitat River Chinook salmon population, it might be reasonable to assign that population to the Lower Columbia River (LCR) Chinook Salmon ESU.” In the absence of an official change in the boundary, however, the Lower Columbia River Chinook salmon

ESU is being evaluated here without considering the Klickitat River. No boundary changes were discussed for the LCR Chinook salmon ESU as a part of this review.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

For the current evaluation, data were available for many populations through 2013 or 2014, with some of the data sets going back as far as 1968. There have been a number of recent efforts to standardize survey methods. Guidance provided by Crawford and Rumsey (2011) emphasized the need for a common set of population parameters that could be used to evaluate VSP criteria across all populations. In 2010 WDFW expanded their efforts to survey Chinook and coho salmon in the Lower Columbia River, specifically focusing on data appropriate for evaluating VSP criteria (Rawding *et al.* 2014). These data include: abundance, proportion hatchery-origin spawners, age and sex. Similar efforts have been undertaken by ODFW to more uniformly undertake spawner surveys across the Oregon Coast and Lower Columbia River through their Oregon Adult Salmonid Inventory & Sampling (OASIS) project. Presently, there is some level of monitoring for all Chinook salmon populations except those that are functionally extinct (Rawding & Rodgers 2013). Methodologies include expansions of index reach redd counts, tributary weir counts, mark/recapture surveys, and hatchery trap, dam trap, and dam ladder counts. Hatchery-origin fish are nearly all adipose clipped with a portion also being coded wire tagged. Full implementation of mass marking fall-run Chinook salmon provides better information on NOR abundance (instead of the previous method of CWT expansion), allows for mark selective fisheries, facilitates broodstock protocols in hatcheries, and NOR spawner selection at weirs and other facilities. For many of the DIPs monitored through these projects the complete data are available for only a few years and there is considerable uncertainty in analyzing data time series across different survey methodologies, especially those data series for years prior to 2010.

ABUNDANCE AND PRODUCTIVITY

SPRING-RUN CASCADE MPG

Of the seven spring-run DIPs in this MPG there are abundance estimates for the Upper Cowlitz/Cispus (2 DIPs combined), Kalama, and Sandy populations. Of these, only the Sandy River spring-run population appears to be a currently self-sustaining population. The Sandy River is also the only spring-run population that exhibited a substantial increase in absolute abundance (Table 38). In contrast, the other spring-run populations in this MPG have very low abundances of natural-origin spawners. For the Upper Cowlitz/Cispus, and Lewis River populations hatchery supplementation currently provides the overwhelming contribution to escapement and some form of direct handling/transportation is necessary to provide access to historical spawning grounds. For the Upper Cowlitz and Lewis rivers, current downstream juvenile passage efficiencies are not sufficient for the populations to sustain themselves. The Kalama River spring-run hatchery program is run as a segregated program and returning HOR adults are excluded from upriver spawning habitat. WDFW does not recognize the continued existence of the Toutle River spring-run DIP, and adult spawner surveys are not undertaken (WDF *et al.* 1993). Recent abundances for the Kalama River spring-run DIP have been critically low, with strongly negative long- and short-term trends (Table 38). The decline in the Kalama River spring-run DIP is somewhat surprising in that returning adults are placed above the Kalama Hatchery and have access to historical spawning habitat (although historically this run was never likely very large). The data series for the North Fork Lewis River reflects fish naturally-spawning below Merwin Dam. This habitat was not historically used by spring-run Chinook salmon, is likely not suitable for spring-run Chinook salmon, and is also heavily used by spawning late-fall run fish; therefore, abundances and trends in this data series were not thought to be informative. Reintroduction efforts have not yet begun to reestablish spring-run Chinook salmon in the Tilton River DIP. In summary, only one DIP has even a low to moderate abundance level, three have very low abundances, and the remaining three have few if any naturally-spawning individuals, although the populations may persist as hatchery stocks in some cases.

SPRING-RUN GORGE MPG

Both of the two spring-run historical DIPs in this MPG are extirpated or nearly so. In the Big White Salmon River, the removal of Condit Dam in 2011 reestablished access to historical spring-run Chinook salmon spawning grounds. Although some spring-run fish have spawned in the basin subsequent to the dam removal, the origin of those fish is not known. Native spring-run Chinook salmon in the Hood River declined to critically low levels in the late 1980s and may have been completely supplanted by introduced Deschutes River spring-run Chinook salmon, an out-of-ESU hatchery population. There have been recent returns of unmarked spring-run Chinook salmon to the Hood River, some of which genetically appear to represent Lower Columbia River populations. The net contribution of these fish is unknown, but if successful they hold some promise for recovering a population relevant to the Lower Columbia River ESU.

COASTAL FALL-RUN MPG

In general, the DIPs in this MPG are dominated by hatchery-origin spawners from one of the many large production hatcheries in the area (Table 40). The abundance of naturally-produced adults appears to be relatively stable although at a very low level, with the confounding effects of the progeny of naturally-spawning hatchery fish increasing the uncertainty in any conclusions regarding productivity. The Clatskanie River surveys are strongly influenced by large numbers of hatchery-origin fish being attracted to Plympton Creek[7], whereas the mainstem Clatskanie River has a few natural-origin spawners, but almost no hatchery-fish. In surveys conduct in both 2012 and 2013, no Chinook salmon were observed in Scappoose Creek.

FALL-RUN CASCADE MPG

The majority of the populations in this DIP have exhibited stable or slightly positive natural origin abundance trends. Natural origin spawners number in the high hundreds to low thousands of fish, with the majority of the fish on the spawning grounds being natural origin, except for the Toutle, Kalama, and Washougal rivers where hatchery programs strongly influence the composition of naturally-spawning fish. Interestingly, the proportion of hatchery-origin spawners in lower Cowlitz River was relatively low, 19.0%, especially given the large hatchery program present (Gleizes *et al.*

[7] There is no hatchery on Plympton Creek, but this small creek consistently attracts returning hatchery-origin adults

2014). Annual variability in the proportion of hatchery-origin spawners is very high in the Clackamas River[8], although only a few years of data are available. Recent improvements in natural adult returns to the Tilton River (part of the Upper Cowlitz River Fall Run DIP) suggest that the trap and haul program at Mayfield Dam has been relatively successful (Serl & Morrill 2010). Overall, this MPG exhibits stable population trends, but at low abundance levels, and while the level of hatchery contribution to naturally spawning adults is relatively better than in other MPGs in this ESU, most populations are still far above the hatchery contribution target of 10% identified in the NMFS Lower Columbia River recovery plan (Dornbush and Sihler 2013).

FALL-RUN GORGE MPG

Many of the populations in this MPG have limited spawning habitat available, either because of inundation or the loss of access[9]. Additionally, the prevalence of returning hatchery-origin fish to spawning grounds presents a considerable threat to diversity, especially the return of non-native upriver bright fall-run Chinook salmon. Natural-origin returns for most populations are in the hundreds of fish. The removal of Condit Dam in 2011 has restored access to spawning habitat for both fall-run and spring-run Chinook salmon; Chinook salmon estimates on the Oregon side of the Gorge MPG are only available for the Hood River, escapement to the other smaller tributaries is thought to be very low and hatchery contribution high.

LATE-FALL RUN MPG

The two populations in this MPG are likely the most viable DIPs in this ESU and both populations are sustained without any hatchery supplementation. The Lewis River late-fall DIP has the largest natural abundance in the ESU and has a strong short-term positive trend (Table 38) and a stable long term trend (Table 39), suggesting a population near capacity. Although the Merwin Dam limits the amount of available spawning habitat, it also controls flows and minimizes hydrological extremes. Additionally, the thermal regime has been altered such that autumn water temperatures are warmer than normal and spring and summer temperatures are likely to be cooler than normal. Changes in temperature regime can alter incubation and emergence timing. The Sandy River late-fall run has not been directly monitored in a number of years; the most recent estimate was 373 spawners in 2010 (Takata 2011). Their somewhat distinct adult return timing and spawning minimize their interception in coastal and in-river fisheries.

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[8] There is no fall-run Chinook salmon hatchery program on the Clackamas River.
[9] Historically, spawning habitat was limited by the steep gradient along the Columbia River throughout the MPG, anthropogenic effects have further constrained the available habitat.

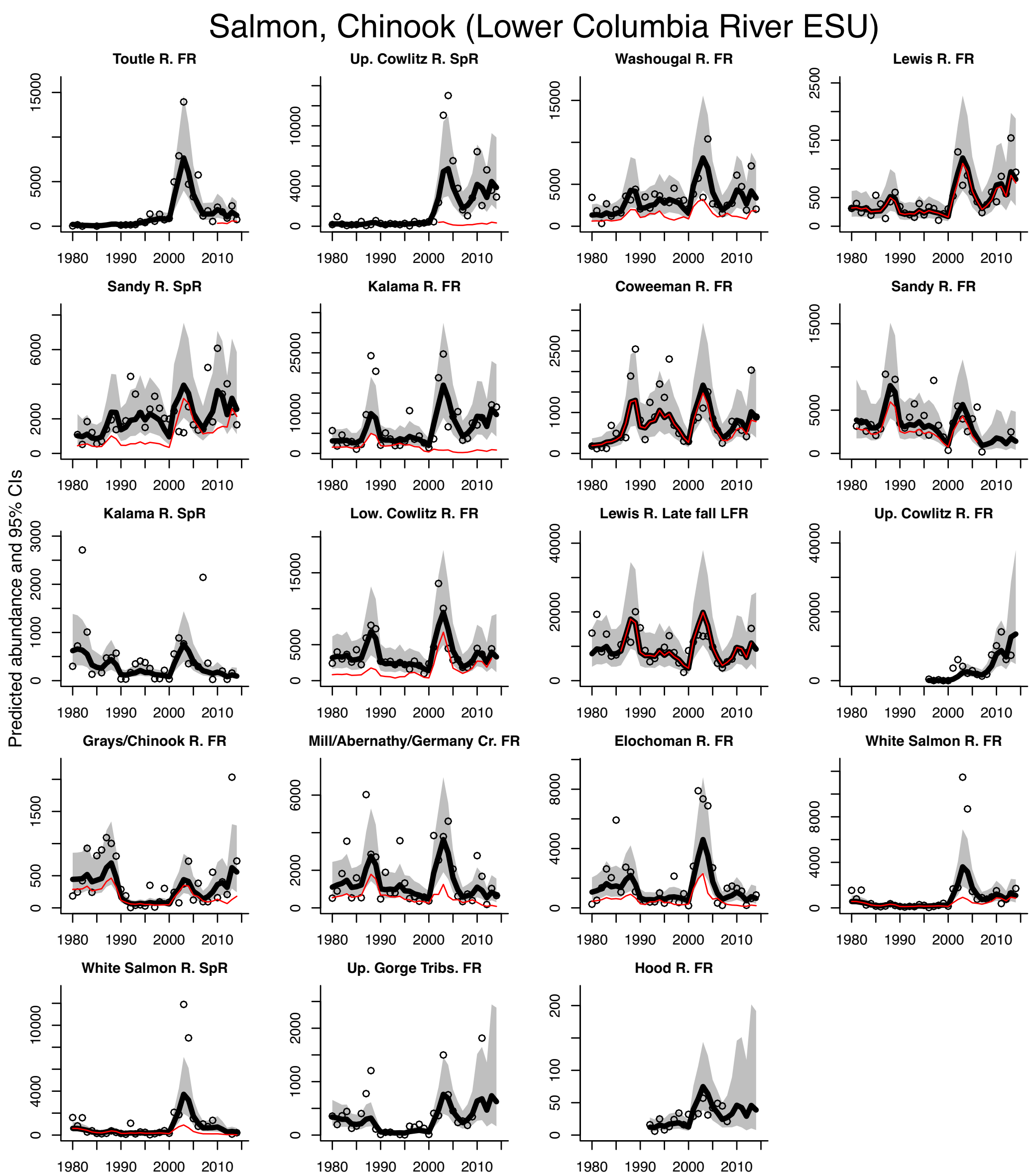

Figure 54 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

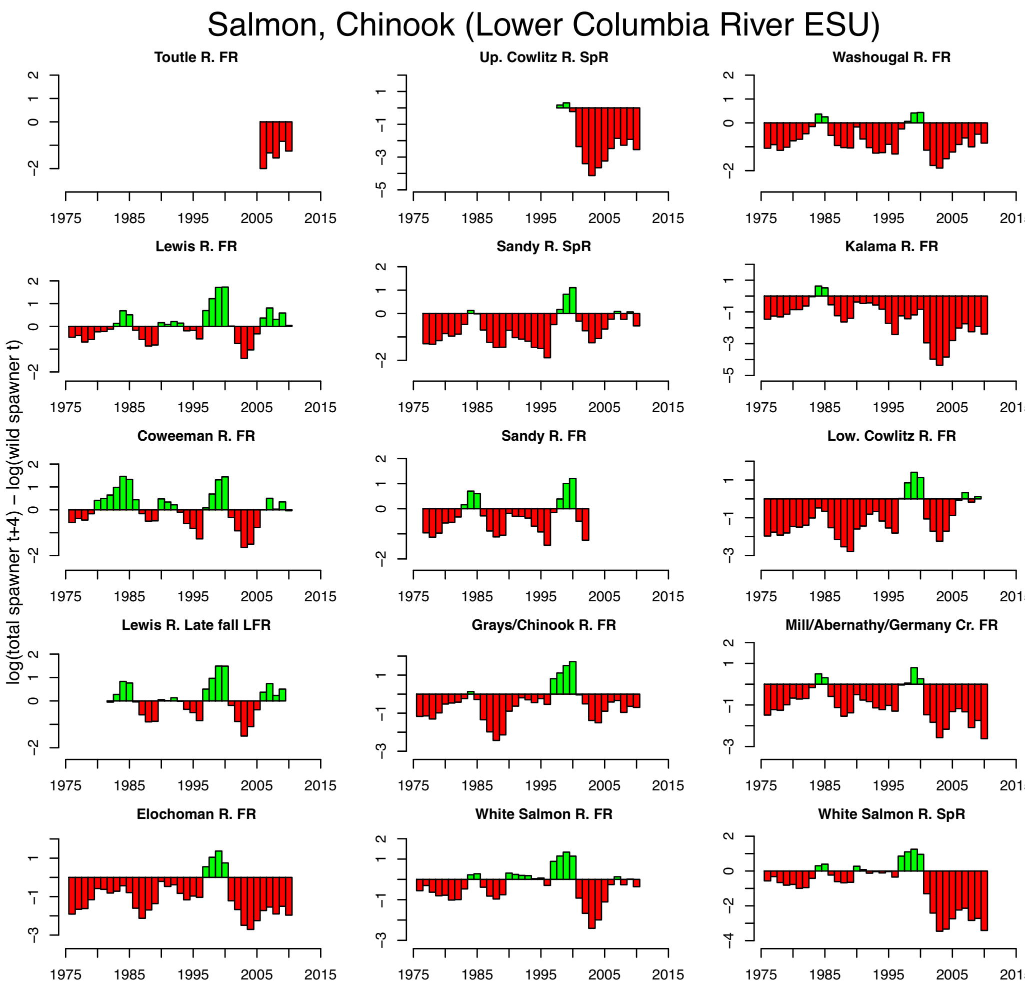

Figure 55 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4). Spawning years on x axis.

Table 38 -- 5-year geometric mean of raw natural-origin spawner (NOS) counts. This is the raw total spawner count times the fraction NOS estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of NOS available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Toutle R. FR	Cascade	(194)	(788)	(4689)	(1826)	374 (1397)	(-23)
Up. Cowlitz R. SpR	Cascade	(206)	(214)	427 (2343)	97 (2602)	279 (3893)	188 (50)
Washougal R. FR	Cascade	1669 (2932)	854 (3227)	1866 (4396)	1002 (2355)	1106 (3813)	10 (62)
Lewis R. FR	Cascade	250 (250)	215 (215)	529 (666)	(424)	732 (788)	(86)
Sandy R. SpR	Cascade	755 (2530)	644 (2322)	1068 (1817)	1388 (1953)	1731 (3201)	25 (64)
Kalama R. FR	Cascade	1654 (2714)	1266 (4192)	356 (6911)	230 (6156)	802 (9304)	249 (51)
Coweeman R. FR	Cascade	877 (877)	796 (796)	721 (805)	380 (526)	624 (770)	64 (46)
Sandy R. FR	Cascade	2732 (3594)	2614 (3440)	1778 (2340)	3518 (1562)		
Kalama R. SpR	Cascade	(121)	(127)	(337)	(295)	(96)	(-67)
Low. Cowlitz R. FR	Cascade	461 (2529)	580 (1827)	2676 (5818)	(2367)	2802 (3760)	(59)
Lewis R. Late fall LFR	Cascade	8353 (8353)	6647 (6647)	11694 (11694)	5758 (5758)	9856 (9856)	71 (71)
Up. Cowlitz R. FR	Cascade		(42)	(724)	(2485)	(8982)	(261)
Grays/Chinook R. FR	Coastal	48 (53)	47 (81)	178 (214)	116 (188)	100 (457)	-14 (143)
Mill/Abernathy/Germany Cr. FR	Coastal	680 (1153)	290 (602)	381 (2292)	293 (658)	90 (893)	-69 (36)
Elochoman R. FR	Coastal	261 (530)	196 (661)	511 (2771)	191 (778)	107 (676)	-44 (-13)
White Salmon R. FR	Gorge	125 (127)	127 (151)	636 (2129)	(939)	780 (980)	(4)
White Salmon R. SpR	Gorge	203 (205)	132 (158)	694 (2324)	(1048)	13 (138)	(-87)
Up. Gorge Tribs. FR	Gorge	(24)	(76)	(289)	(280)		
Hood R. FR	Gorge	(13)	(17)	(35)	(37)		

Table 39 -- 15-year trends in log natural-origin spawner (NOS) abundance computed from a linear regression applied to the smoothed NOS log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Toutle R. FR	Cascade		
Up. Cowlitz R. SpR	Cascade		0.08 (0, 0.15)
Washougal R. FR	Cascade	0.05 (0, 0.09)	-0.02 (-0.07, 0.03)
Lewis R. FR	Cascade	0.1 (0.05, 0.15)	0.05 (-0.01, 0.11)
Sandy R. SpR	Cascade	0.12 (0.06, 0.17)	0.05 (-0.01, 0.12)
Kalama R. FR	Cascade	-0.1 (-0.15, -0.06)	0.01 (-0.05, 0.07)
Coweeman R. FR	Cascade	0.01 (-0.04, 0.06)	0 (-0.06, 0.06)
Sandy R. FR	Cascade	0 (-0.05, 0.05)	
Low. Cowlitz R. FR	Cascade	0.14 (0.06, 0.22)	0.05 (-0.04, 0.14)
Lewis R. Late fall LFR	Cascade	0.04 (-0.01, 0.09)	0 (-0.06, 0.06)
Grays/Chinook R. FR	Coastal	0.09 (0.02, 0.17)	0 (-0.07, 0.07)
Mill/Abernathy/Germany Cr. FR	Coastal	0 (-0.05, 0.05)	-0.12 (-0.18, -0.05)
Elochoman R. FR	Coastal	0.1 (0.02, 0.17)	-0.14 (-0.23, -0.04)
White Salmon R. FR	Gorge	0.13 (0.08, 0.18)	0.08 (0.02, 0.14)
White Salmon R. SpR	Gorge	0.1 (0.04, 0.15)	

HARVEST

Lower Columbia River Chinook salmon include three distinct life-history components: spring-run Chinook, tule fall Chinook, and late fall Chinook. These different components are subject to different in-river fisheries because of differences in river entry timing, but share relatively similar ocean distributions. Life history types share similar patterns, but different absolute exploitation rates. All saw a drop in exploitation rates in the early 1990s with a modest increase since then (Figure 56). Ocean fishery impact rates have been relatively stable in the past few years, with the exception of the bright fall component of the ESU.

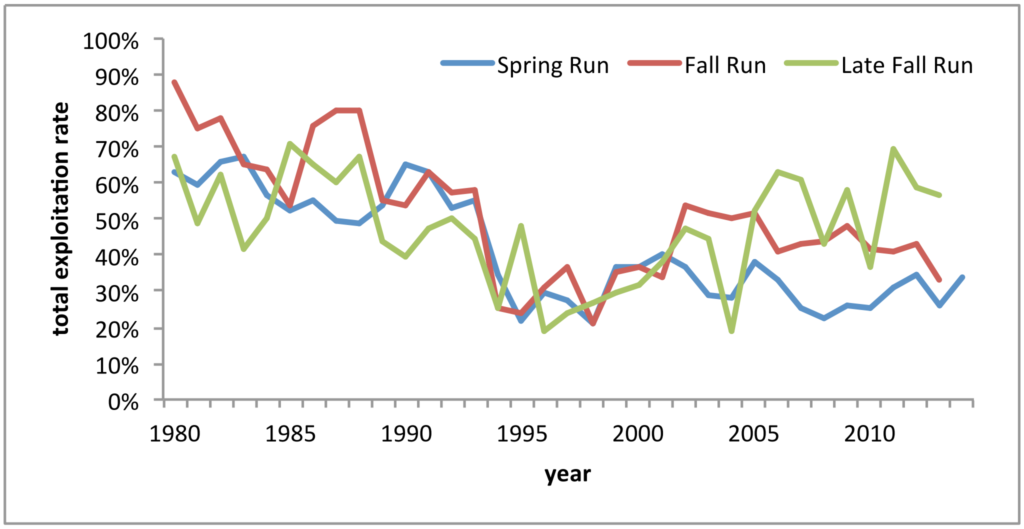

Figure 56 -- Total exploitation rates on the three components of the Lower Columbia River Chinook ESU. Data for tule fall Chinook from exploitation rate analysis of aggregate tule stock made up of tag codes from the Big Creek, Cowlitz, Kalama, and Washougal hatcheries. Data for bright fall Chinook from the CTC exploitation rate analysis (CTC in prep). Data for spring Chinook from CTC model calibration 1503 (CTC in prep) for Willamette River spring Chinook for ocean impacts and TAC run reconstruction data for in-river impacts using an aggregate of of Cowlitz, Kalama, Lewis, and Sandy River spring-run Chinook salmon (Robin Ehlke, WDFW, personal communication).

SPATIAL STRUCTURE AND DIVERSITY

HATCHERIES

A recent review by the HSRG (HSRG 2009) identified 19 hatchery programs, many long-standing, with some hatcheries having been in operation for over 100 years. On average fall-run Chinook salmon programs have released 50 million fish annually, with spring-run and upriver bright (URB) programs releasing a total of 15 million fish annually (Figure 57). As a result of this high level of hatchery production and low levels of natural production, many of the populations contain over 50% hatchery fish among their naturally spawning assemblages (Figure 59, Table 40).

In addition, the release of a number of out-of-ESU stocks continues to be a concern (Willamette River and Interior Columbia River stocks of spring-run Chinook salmon programs and the upriver bright (URB) and Select Area Bright (SAB) programs). Annual production out-of-ESU stocks has been approximately 12.5 million fish (2008-2014). URB releases were transitioned from Bonneville Hatchery to the Little White Salmon NFH beginning in 2010 in order reduce interactions with native tule fall-run Chinook salmon spawning below Bonneville Dam. A study by Smith and Engle (Smith & Engle 2011) found that 4.3 to 15.0% of juveniles in the (Big) White Salmon River were LCR fall-run x URB hybrids, yet no returning hybrid adults were detected. This would suggest that the risks of long-term genetic introgression may be low, but that the short term effect on productivity may be significant.

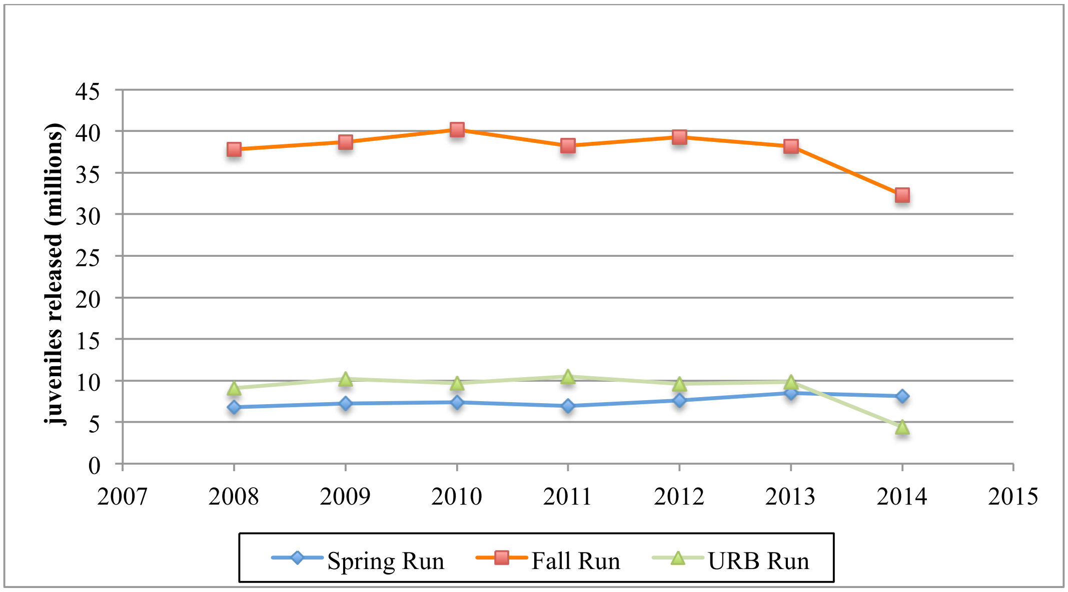

Figure 57 - Annual releases of juvenile Chinook salmon, by run type, in the Lower Columbia River ESU from 2008 to 2014 (2014 data may not be complete). This data does not include releases into the SAFE zone. Data from RMIS (http://www.rmpc.org/ accessed 6 January 2015).

Furthermore, the HSRG (2009) identified the use of out-of-basin stocks in Select Area Fishery Evaluation (SAFE) areas as a concern, especially in light of the high level of straying onto nearby spawning grounds. Approximately 750,000 out-of-ESU Rogue River Bright (RRB[10]) fall-run Chinook salmon are currently being released into Youngs Bay, creating a potential for interaction with natural-origin fall-run juveniles and adults (Figure 58). In the past, naturally produced juvenile Rogue River Chinook salmon and RRB x LCR fall-run Chinook salmon juvenile hybrids have been detected in nearby tributaries on the Washington State side of the Lower Columbia River (Marshall 1997). Naturalized and hatchery-origin Rogue River Bright (aka SAB) fall-run Chinook salmon have also been recovered during spawning surveys in the Grays River (Rawding *et al*. 2014), although many first generation hatchery-origin fish were removed at the weir on the Grays River. Releases of out-of-ESU Upper Willamette River spring-run Chinook salmon into Oregon tributaries near the mouth of the Columbia River may not pose a long-term genetic risk, due to the absence of spring-run spawning habitat; but may pose a risk to natural-origin juveniles due to competition and predation. The continued large scale release of both native and non-native Chinook salmon hatchery stocks into the Youngs Bay and Big Creek DIPs will likely constrain the recovery of these populations, which are currently identified as only “secondary populations” in the recovery plan.

[10] This hatchery stock is also identified by ODFW as 052 Select Area Brights (SAB).

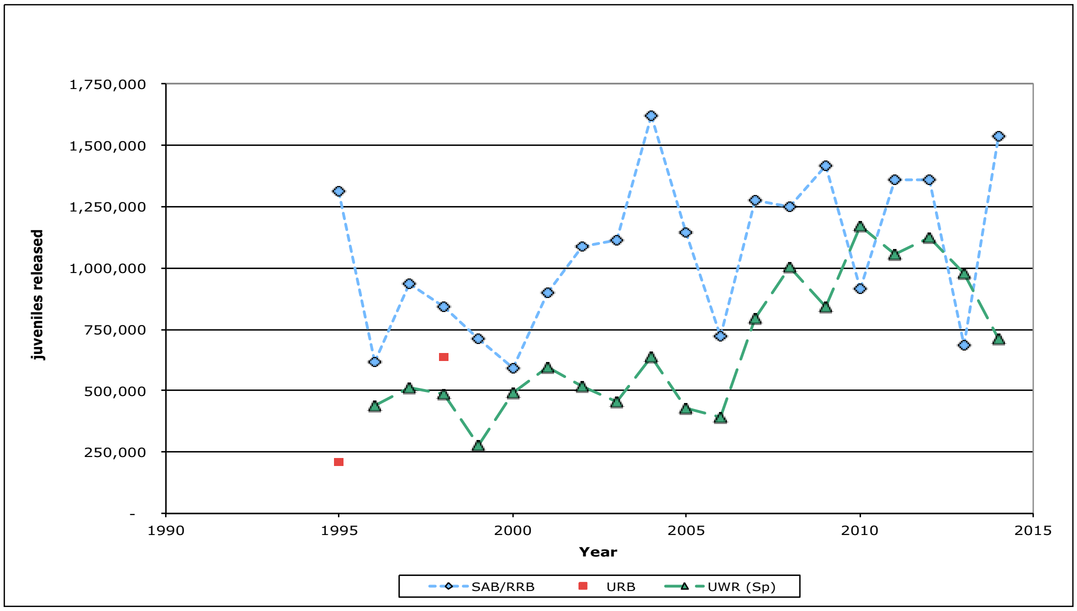

Figure 58 - Releases of out-of-ESU hatchery stocks into the Youngs Bay and Big Creek DIPs from 1995 – 2014. Select-area brights (SAB) are fall-run Chinook salmon from the Rogue River, upriver brights (URB) originated from late-fall runs in the Upper Columbia River, and Upper Willamette River spring-run Chinook salmon (UWR) are the progeny of fish returning to the Upper Willamette River.

SPATIAL STRUCTURE

There have been a number of large-scale efforts to improve accessibility, one of the primary metrics for spatial structure, in this ESU. Passage efforts on the Cowlitz River at Cowlitz Falls began in 1996 for Chinook salmon and other salmonids. There have been a number of structural and operational changes in the collection protocol for out-migrating juveniles (Serl *et al*. 2010), with collection efficiencies averaging 28.8% for Chinook salmon during 2006-2009. At the current fish collection efficiency (FCE) levels for outmigrating juveniles, naturally-spawning Chinook salmon cannot establish sustainable populations (Serl *et al.* 2014); however, further studies and modifications at the Cowlitz Falls facility are continuing in order to improve passage efficiency. Juvenile collection at Mayfield Dam appears to be more relatively more successful with increasing numbers of fall-run Chinook salmon returning in the last few years. Spring-run reintroductions are not planned for the Tilton River. More recently on the Hood River, Powerdale Dam was removed in 2010 and while this dam previously provided fish passage, removal of the dam is thought to eliminate passage delays and injuries. Condit Dam, on the White Salmon River, was removed in 2011 and this provided access to previously inaccessible habitat. Fish passage operations for spring-run Chinook salmon (trap and haul) were begun on the Lewis River in 2012, reestablishing access to historically-occupied habitat above Swift Dam (RKm 77.1), but few adults have been available for passage and juvenile passage efficiencies have been poor. In addition, there have been a number of recovery actions throughout the ESU to remove or improve culverts and other small-scale passage barriers. Many of these actions have occurred too recently to be fully evaluated; however, some data are now available for many actions that occurred prior to 2010, but were not able to be assessed in the previous status review. These include the removal of Marmot Dam in 2007 and the Little Sandy River diversion dam in 2008.

For a number of projects where passage has been restored through dam removal, structural modification, or operational modification it remains to be demonstrated that both adult and juvenile passage survival is sufficient to provide some level of self-sufficiency to upstream population components. If recruit:spawner ratios are well below one, it is unlikely that there is any benefit to population special structure and passage operations may actually represent a net loss in productivity and abundance[11].

There are other ecological benefits to transporting adults above impassable dams even if the process is not sustainable, although these may not directly improve the population's VSP score.

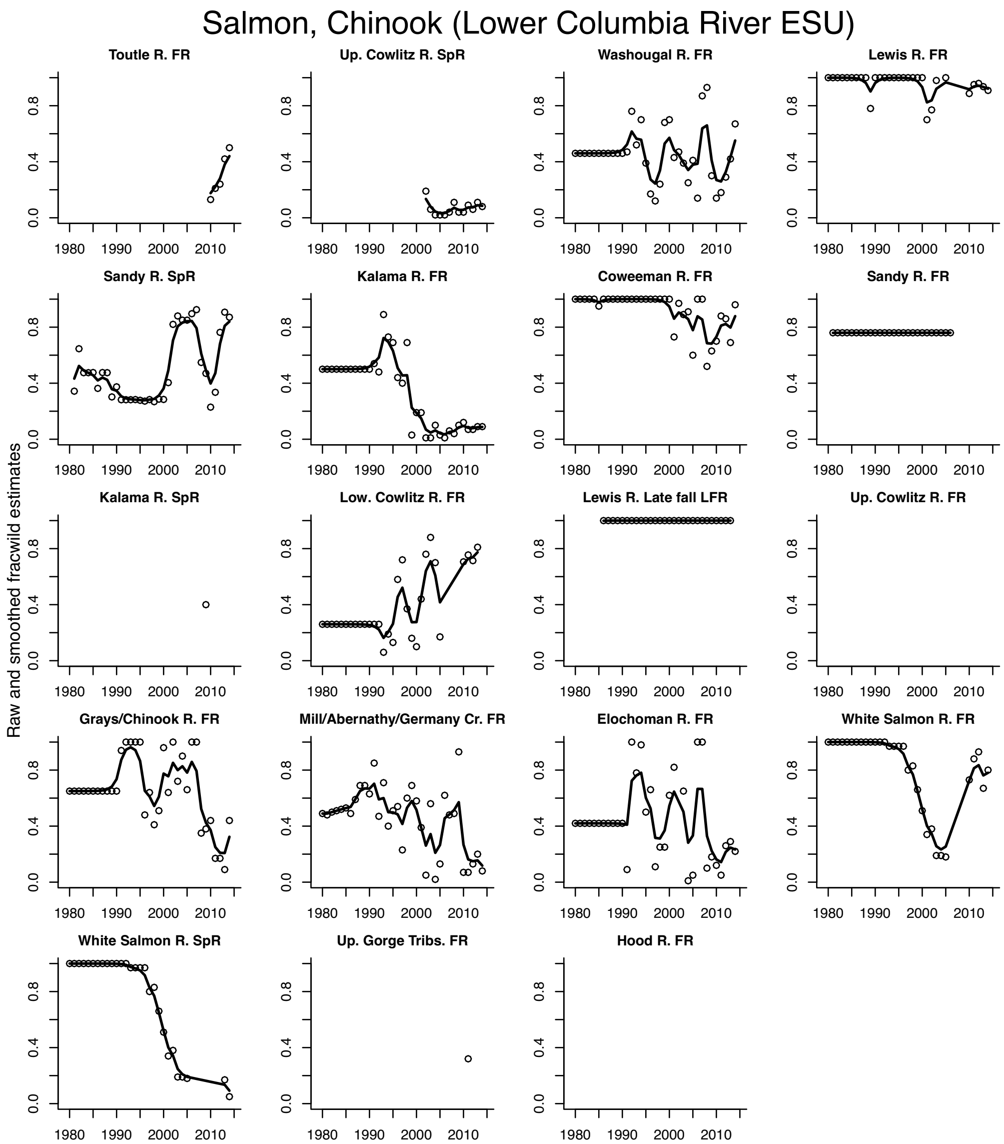

Figure 59 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 40 -- 5-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Toutle R. FR					0.30
Up. Cowlitz R. SpR			0.09	0.05	0.08
Washougal R. FR	0.58	0.32	0.45	0.53	0.34
Lewis R. FR	1.00	1.00	0.86	1.00	0.93
Sandy R. SpR	0.30	0.28	0.65	0.74	0.62
Kalama R. FR	0.63	0.45	0.10	0.05	0.09
Coweeman R. FR	1.00	1.00	0.90	0.75	0.82
Sandy R. FR	0.76	0.76	0.76	0.76	
Kalama R. SpR				0.40	
Low. Cowlitz R. FR	0.21	0.39	0.58	0.17	0.75
Lewis R. Late fall LFR	1.00	1.00	1.00	1.00	1.00
Up. Cowlitz R. FR					
Grays/Chinook R. FR	0.92	0.61	0.84	0.68	0.26
Mill/Abernathy/Germany Cr. FR	0.61	0.51	0.32	0.53	0.11
Elochoman R. FR	0.65	0.35	0.52	0.47	0.19
White Salmon R. FR	0.99	0.85	0.32	0.18	0.80
White Salmon R. SpR	0.99	0.85	0.32	0.18	0.11
Up. Gorge Tribs. FR					0.32
Hood R. FR					

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Of the 32 DIPs in this ESU, only the 2 late-fall run populations (Lewis River and Sandy River) could be considered viable or nearly so; with a few exceptions the remainder of the populations fell far short of their recovery goals in abundance. A total of 7 of 32 populations are at or near their recovery viability goals (Figure 60 Figure 61), although under the recovery plan scenario only two of these populations had scores above 3.0. The remaining populations generally require a higher level of viability and most require substantial improvements to reach their viability goals. Those populations that did meet their recovery goals did so because the goals were set at low, status quo, levels. In addition, the estimated proportion of hatchery-origin spawners was well in excess of the limits set in the recovery plan for many of the primary populations (Dornbush 2013). All of the Coastal and Gorge MPG fall-run populations likely fell within the high to very-high risk categories. Similarly, with the exception of the Sandy River spring-run DIP, all of the spring-run DIPs in the Cascade and Gorge MPGs are at high to very high risk categories, with a number of populations functionally extinct, while others may only persist through hatchery supplementation. The Cascade fall-run MPG contains populations at moderate to high risk, while the Cascade late-fall MPG may be near viability (there is some uncertainty in the abundance estimates for the Sandy-River late-fall DIP).

Few populations met the hatchery contribution criteria for primary or contributing populations established by the HSRG (2009) in the 2010-2014 period, although some populations did improve in the proportion of natural origin spawners. Among these were the Coweeman River fall run, Lewis River late-fall run, and Lewis River fall-runs. No criteria were established for stabilizing populations.

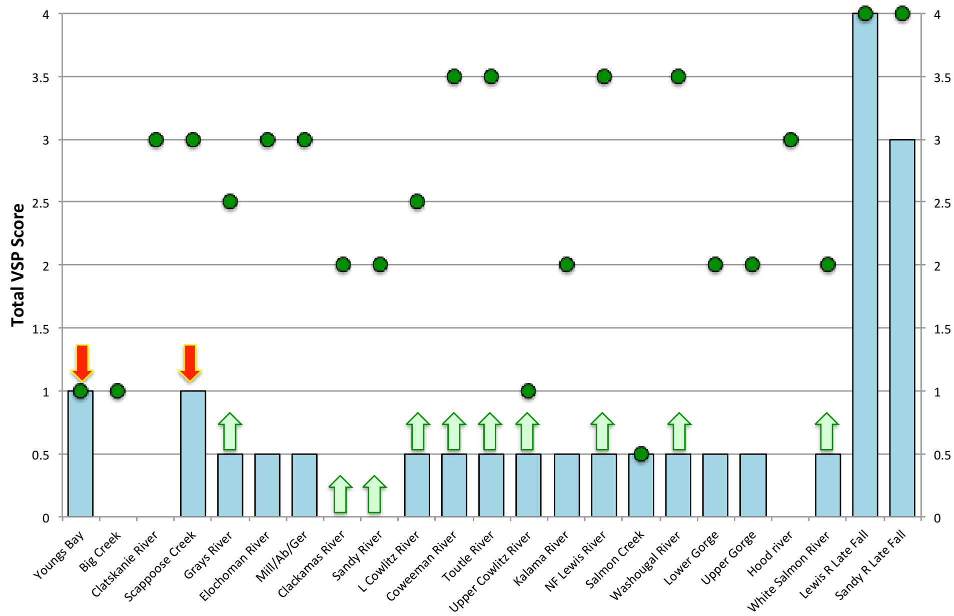

Figure 60 - VSP status of fall-run and late-fall run demographically independent populations in the Lower Columbia River Chinook salmon ESU, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013); green circles indicate the recovery goals. Arrows indicate the general direction, but not the magnitude, of any VSP population score based on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany *et al*. 2006). A VSP score of 3.0 represents a population with a 5% risk of extinction within a 100 year period.

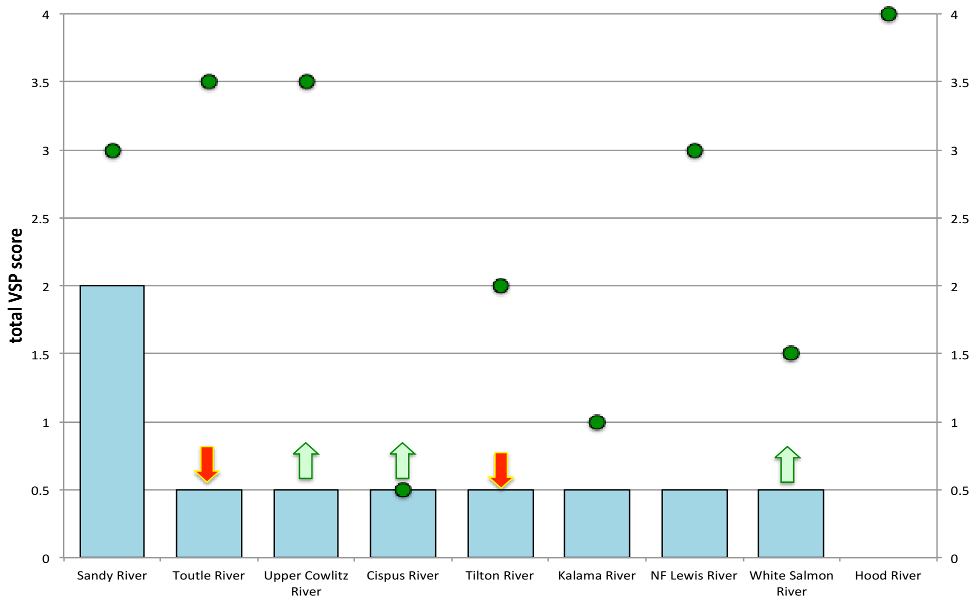

Figure 61 - VSP status of spring-run demographically independent populations in the Lower Columbia River Chinook salmon ESU, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013); green circles indicate the recovery goals. Arrows indicate the direction, but not the magnitude, of the VSP score change based on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany *et al*. 2006). A VSP score of 3.0 represents a population with a 5% risk of extinction within a 100 year period.

UPDATED BIOLOGICAL RISK SUMMARY

Overall, there was little change since the last status review (Ford *et al*. 2011) in the biological status of Chinook salmon populations in the Lower Columbia River ESU, although there are some positive trends. Increases in abundance were noted in about 70% of the fall-run populations and decreases in hatchery contribution were noted for several populations. Relative to baseline VSP levels identified in the Recovery Plan (Dornbush 2013) there has been an overall improvement in the status of a number of fall-run populations (Figure 60, Figure 61), although most are still far from the recovery plan goals.

These improved fall-run VSP scores reflect both changes in biological status and improved monitoring. Spring-run Chinook populations in this ESU are generally unchanged; most of the populations are at a high or very risk due to low abundances and the high proportion of hatchery-origin fish spawning naturally. In contrast, the spring-run Chinook salmon DIP in the Sandy River has an average of over a thousand natural-origin spawners and is at moderate risk. Additionally, the removal of Marmot Dam in the Sandy River eliminated migrational delays and holding injuries that were occurring at the fish ladder. Further, the removal of the diversion dam on the Little Sandy River restored access and flow to historical salmon habitat. Many of the spring-run populations rely upon passage programs at high head dams and downstream juvenile collection efficiencies are still too low to maintain self-sustaining natural runs. While limited numbers of naturally-produced spring run fish return to the Cowlitz and Cispus rivers, no spring-run fish are transported into the Tilton River Basin and it is not clear if there are any spring-run Chinook salmon remaining in the Toutle River Basin. The removal of Condit Dam on the White Salmon River provides an opportunity for the reestablishment of a spring-run population with volitional access to historical spawning grounds (abundance estimates prior to 2012 reflected fish spawning below Condit Dam during the spring run temporal spawning window). Spring-run Chinook salmon in the Hood River are largely of Deschutes River spring-run origin (Middle Columbia River Spring Run ESU) and are provide no benefit to the status of the ESU; however, some Lower Columbia River spring-run Chinook salmon have been detected in the Hood River and their contribution (when sufficiently quantified) may need to be considered during future evaluations.

The majority of the populations in this ESU remain at high risk, with low natural-origin abundance levels. Hatchery contributions remain high for a number of populations, and it is likely that many returning unmarked adults are the progeny of hatchery-origin parents, especially where large hatchery programs operate. While overall hatchery production has been reduced slightly, hatchery-produced fish still represent a majority of fish returning to the ESU. The continued release of out-of-ESU stocks, including URB, Rogue River (SAB) fall run, Upper Willamette River spring run, Carson Hatchery spring run, and Deschutes River spring run, remains a concern. Relatively high harvest rates are a potential concern, especially for most spring-run and low abundance fall-run populations (NMFS 2012). Although there have been a number of notable efforts to restore migratory access to areas upstream of dams, until efforts to improve juvenile passage systems bear fruition, it is unlikely that there will be significant improvements in the status of many spring-run populations. Alternatively, dam removals (i.e. Condit Dam, Marmot Dam, and Powerdale Dam) not only improve/provide access, but the allow the restoration of hydrological processes that may improve downstream habitat conditions. Continued land development and habitat degradation in combination with the potential effects of climate change may present a continuing strong negative influence into the foreseeable future. In addition, coastal ocean conditions would suggest that recent outmigrant year classes will experience below average ocean survival with a corresponding drop in spawner abundance in the near term, depending on the duration and intensity of the existing situation (see Recent trends in marine and terrestrial environments section, below).

LOWER COLUMBIA RIVER COHO SALMON ESU

BRIEF DESCRIPTION OF ESU

Lower Columbia River coho salmon were identified as an ESU[12] and listed as threatened in 2005. This ESU includes all naturally spawned populations[13] of coho salmon in the Columbia River and its tributaries in Washington and Oregon, from the mouth of the Columbia River up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon, as well as multiple artificial propagation programs (Figure 62). Myers *et al* (Myers *et al.* 2006) identified three MPGs (Coastal, Cascade, and Gorge), containing a total of 24 DIPs in the Lower Columbia River coho salmon ESU.

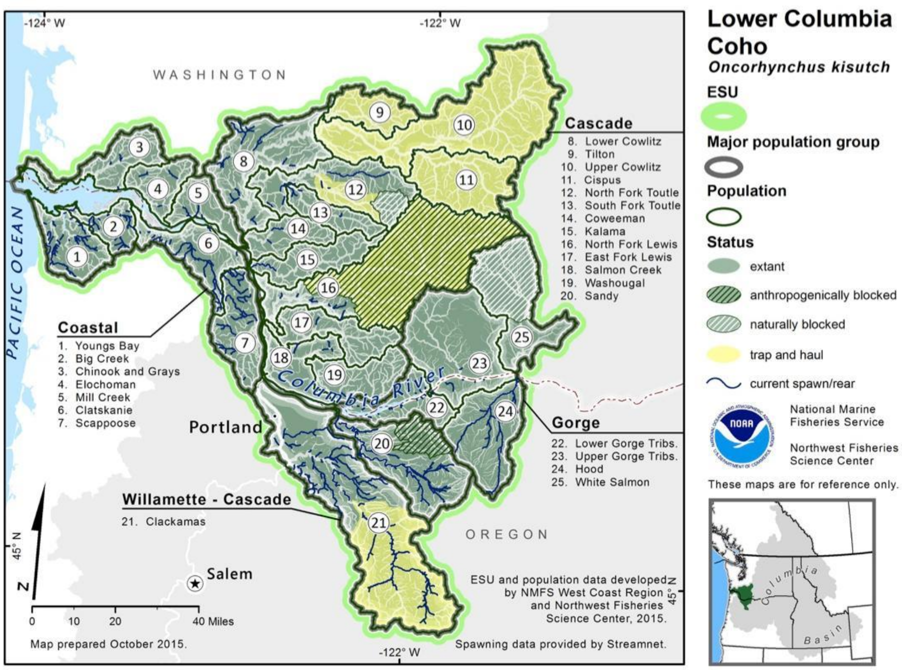

Figure 62 -- Map of the Lower Columbia River coho salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

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[12] The current ESU was redelineated in 2005 to incorporate Washington tributaries in the Coastal MPG.

[13] Two major native life history types are recognized among Lower Columbia River coho salmon populations: Type N or late returning, and Type S or early returning. The life history types differ according to run timing, spawn timing, ocean migration patterns and spawning habitat preference (see Myers *et al*. 2006).

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

NMFS reviewed the status of the Lower Columbia River coho salmon ESU in 1996, again in 2001 and in 2005. In the 2001 review, the BRT was concerned that the vast majority (over 90%) of historical populations in the Lower Columbia River coho salmon ESU appeared to be either extirpated or nearly so. The two populations with any significant production (Sandy and Clackamas rivers) were at appreciable risk because of low abundance, declining trends, and failure to respond after a dramatic reduction in harvest. The large number of hatchery coho salmon in the ESU was also considered an important risk factor. The majority of the 2001 BRT votes were for “at risk of extinction” with a substantial minority “likely to become endangered.” As a result of the 2001 BRT review, the ESU was identified as a “candidate species”, but not listed under the ESA as threatened or endanged. An updated status evaluation was conducted in 2005, also with a majority of BRT votes for “at risk of extinction” and a substantial minority for “likely to become endangered.” This BRT evalutation resulted in a “threatened” determination in 2005.

2010

Ford *et al*. (2011) noted that three status evaluations of LCR coho status, all based on WLC-TRT criteria, had been conducted since the prior BRT status update in 2005. All three evaluations concluded that the ESU was currently at very high risk of extinction. Of the 24 historical populations in the ESU, 21 were considered at very high risk. The remaining three (Sandy, Clackamas and Scappoose) were considered to be at high to moderate risk. All of the Washington side populations were considered at very high risk, although uncertainty was high because of a lack of adult spawner surveys. As was noted in the 2005 BRT evaluation, smolt traps indicated some natural production in Washington populations, though given the high fraction of hatchery origin spawners suspected to occur in these populations it was not clear that any were self-sustaining. Overall, the new information that was considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

In 2010, the ESU Boundaries Review Group (see ESU Boundaries section in Ford *et al*. 2011) undertook a revaluation of the boundary between all lower Columbia and mid- Columbia ESUs and DPSs. The review’s conclusions emphasized the transitional nature of the boundary between the lower Columbia ESUs and the mid-Columbia ESUs. The original Lower Columbia coho salmon ESU boundary was assigned based largely on extrapolation from information about the boundaries for Chinook and steelhead. The ESU Boundaries Review Group concluded, “It is therefore reasonable to assign the Klickitat population to the lower Columbia coho ESU. This would establish a common boundary for Chinook salmon, coho salmon, chum salmon, and steelhead at the Celilo Falls (Dalles Dam).” To date, this recommendation has not been officially implemented; therefore, the current status review will utilize preexisting ESU boundaries.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Since the last status review there have been a number of efforts to standardize monitoring efforts. Guidance provided by Crawford and Rumsey (2011) emphasized the need for a common set of population parameters that could be used to evaluate VSP criteria across all populations. In 2010

WDFW expanded their efforts to survey Chinook and coho salmon in the Lower Columbia River, specifically focusing on data appropriate for evaluating VSP criteria (Rawding *et al*. 2014). Monitoring efforts cover all of the coho salmon populations in the Lower Columbia River, with the exception of the extirpated (but now accessible) White Salmon River and with a low priority on Youngs Bay and Upper Gorge populations (Rawding and Rodgers 2013). These data included: abundance, proportion hatchery-origin spawners, age and sex. Similar efforts have been undertaken by ODFW to more uniformly undertake spawner surveys across the Oregon Coast and Lower Columbia River through their Oregon Adult Salmonid Inventory & Sampling (OASIS) project. Methodologies include expansions of index reach redd counts, tributary weir counts, mark/recapture surveys, and hatchery trap, dam trap, and dam ladder counts.

Many of the WDFW coho population datasets reflect only three or four years of sampling, except for some intensively monitored watersheds (IMW) or where passage structures have allowed for adult counts. With the exception of some smaller tributaries in the Gorge MPG, surveys now include most of the DIPs in this ESU.

ABUNDANCE AND PRODUCTIVITY

COAST RANGE CASCADE MPG

Long-term abundances were generally stable (Table 42), with short term variability most strongly influenced by ocean conditions. Of the populations in the MPG, Scappoose Creek is somewhat distinctive in exhibiting a positive abundance trend and containing few hatchery-origin fish. Similarly, the Clatskanie River coho salmon population maintains moderate numbers of naturally-produced spawners, with proportionately few hatchery-origin spawners[14]. In addition, the initiation of spawner surveys in Washington tributaries indicated the presence of moderate numbers of coho salmon, with total abundances in the hundreds to low thousands of fish, a substantial proportion of which were naturally produced (Figure 63). These new data series for Washington tributaries are too short to calculate meaningful population trends. Oregon tributaries in this MPG have abundances in the hundreds of fish with the majority of the fish being naturally produced. Previously (McElhany *et al.* 2006), these populations were thought to have been dominated by hatchery-origin adults and baseline VSP scores reflected this inferred very high risk status (Figure 69).

WESTERN CASCADE MPG

The coho salmon populations in the Sandy and Clackamas River were the only two populations identified in the original 1996 Status Review that appeared to be self-sustaining natural populations. Abundance trends for these populations also represent the longest complete set of observations for any Lower Columbia River coho salmon populations. With the removal of Marmot Dam in 2008, inventory methods for the Sandy River coho salmon populations have undergone some significant

[14] Unofficial spawner estimates for 2014-15 in the Clatskanie River and Scappoose Creek were 3547 (3126 natural) and 1477 (1477 natural), respectively (*http://oregonstate.edu/dept/ODFW /spawn/reports.htm*).

changes. Recent returns of unmarked fish to the Clackamas River[15] have shown a marked improvement in run size (Table 41), and the unofficial coho count for 2014-2015, 10,670 spawners would be the highest recorded. Natural-origin returns to the Sandy River have remained fairly stable since the initial status review in the mid-1990s, although there appears to be a continued hatchery presence. Hatchery fish are collected at the Cedar Creek weir for the Sandy River Hatchery, with only natural-origin (unmarked) coho salmon passed above. Estimates for the 2014-15 return year indicate a dramatic improvement in escapement, similar to the Clackamas River, with 5,942 natural-origin spawners.[16]

Coho salmon trapped at the Cowlitz hatchery barrier dam are transported to the Upper Cowlitz and Cispus Rivers and Tilton River and have been enumerated since the mid-1990s. There were substantial returns of natural origin coho salmon to the Tilton and Upper Cowlitz/Cispus rivers in 2014[17], and collection efficiencies at Mayfield and Cowlitz Falls are adequate to sustain the populations (>50%). In addition, a large number of hatchery-origin fish from the integrated hatchery program were transported upstream.

Two or three years of abundance data are available for Washington DIPs in this MPG, including estimates of natural and hatchery-origin contribution. Total abundances are in the hundreds to low thousands of fish with little consistent trend across DIPs (Figure 63). In many instances the proportion and absolute number of natural-origin spawners is quite high. For example, surveys in the Lower Cowlitz River indicated that the contribution of hatchery-origin fish to tributary spawners is <10% (Gleizes et al. 2014). Long-term trends for the Washington tributaries are only available for the trap and haul programs on the Upper Cowlitz (Cowlitz and Cispus and Tilton rivers); however, for the shorter term abundance series (2 to 3 years) it was not possible to determine a meaningful trend.

This MPG contains most of the ESU's large river basins and the majority of the ESU's abundance. Where natural origin abundances were available the trends were stable for most populations.

COLUMBIA GORGE MPG

Natural origin abundances in this MPG are low, with hatchery-origin fish contributing a large proportion of the total number of spawners, most notably in the Hood River. With the exception of the Hood and Big White Salmon Rivers, much of the spawning habitat is in small independent tributaries to the Columbia River and in many cases the accessibility is relatively poor. Presently, lack of funding prevents monitoring of coho salmon recolonization in the White Salmon basin. There was no clear trend in the abundance data across populations.

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[15] Clackamas River NOR counts include counts made at the North Fork Dam, spawner surveys in the mainstem Clackamas River below the dam, and fish counted at the hatchery weir on Eagle Creek (below North Fork Dam).
[16] http://oregonstate.edu/dept/ODFW/spawn/pdf%20files/coho/AnnualEstESU2004-2014.pdf
[17] Juvenile fish emigrating from the Tilton River are differentially marked so that returning adults can be distinguished from those originating from the Cispus/Cowlitz rivers.

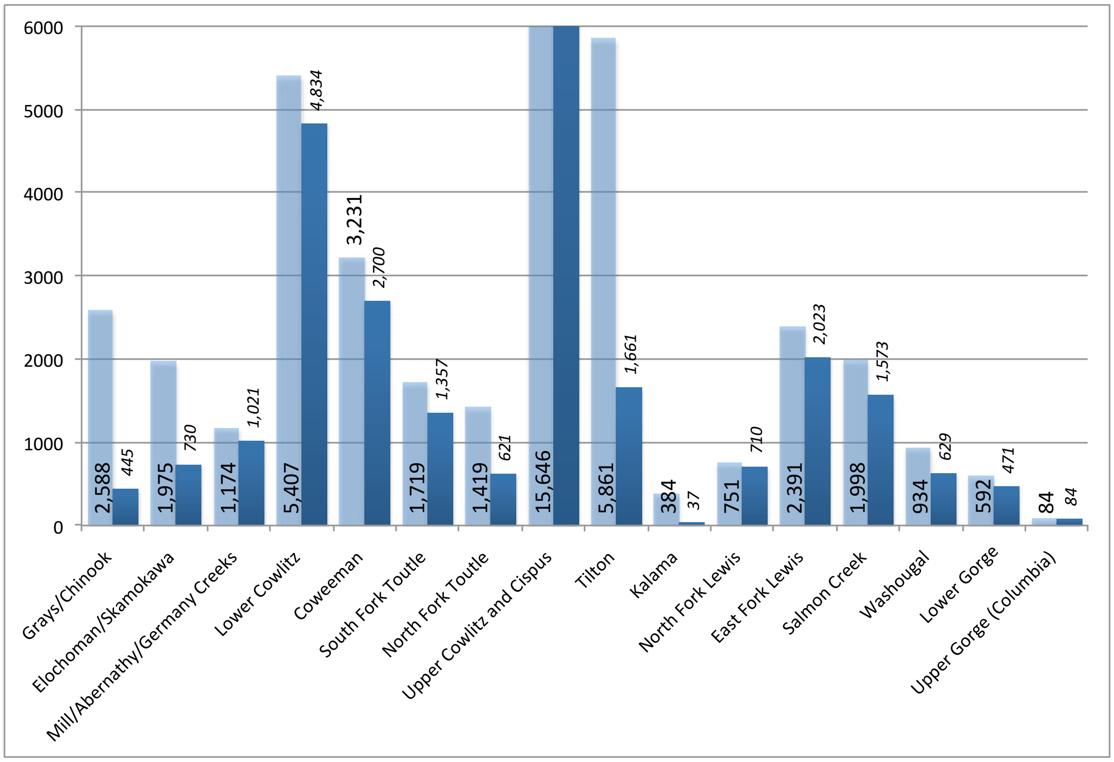

Figure 63. Average coho salmon total spawner abundance (light blue) and natural spawner abundance (dark blue) (2009-2013) for Washington tributaries. For some tributaries data is only available for 2010-2012. Note that the column for Upper Cowlitz and Cispus rivers has been truncated, and the natural spawner abundance is 10,546.

OTHER POPULATIONS

Not included in this review are coho salmon that migrate above Willamette Falls; 18,062 natural-origin coho salmon were counted at the falls in 2014-2015[18]. Coho have not been planted in the Upper Willamette Basin since 1996, and it is believed that these fish are the progeny of Lower Columbia River origin coho salmon spawning in tributaries to the Upper Willamette River, primarily the Tulatin River. We have also not included coho salmon migrating upstream of the Dalles Dam, these are almost entirely the progeny of fish introduced into Mid- and Upper-Columbia and Snake River tributaries from LCR hatchery populations. In 2014, 157,646 coho salmon (adults and jacks) were counted at the Dalles Dam[19], these include both hatchery-origin releases in the Interior Columbia River Basin and the progeny of naturally-spawning fish. In both cases these fish are

[18] Web site accessed on 9 January 2015 from http://www.dfw.state.or.us/fish/fish_counts/willamette/2015/2015_Monthly_sheet.pdf.
[19] Counts are through 31 October 2014, web site accessed on 9 January 2015 from http://www.nwp.usace.army.mil/Missions/Environment/Fish/Counts.aspx

spawning outside of the historical boundaries of the Lower Columbia River ESU. Historically, coho salmon populations existed above the Dalles Dam, but were extirpated during the last century.

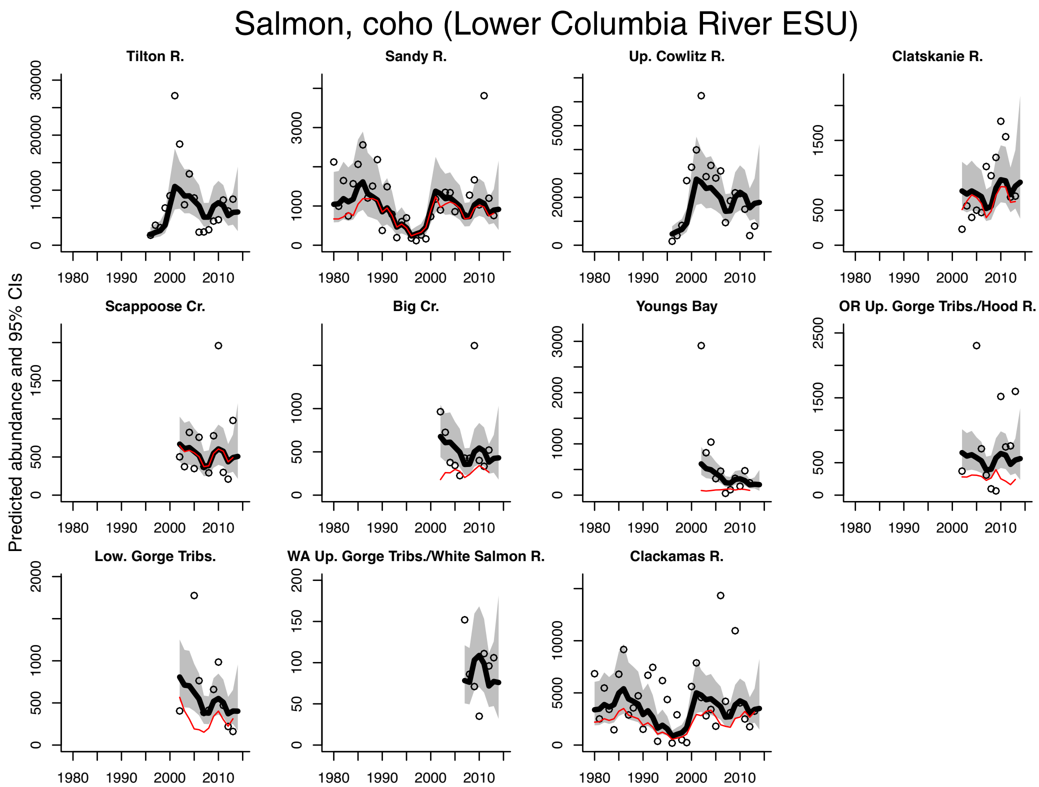

Figure 64 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates. Additonal data was available for other Washington populations, but the time series were too short to estimate trends.

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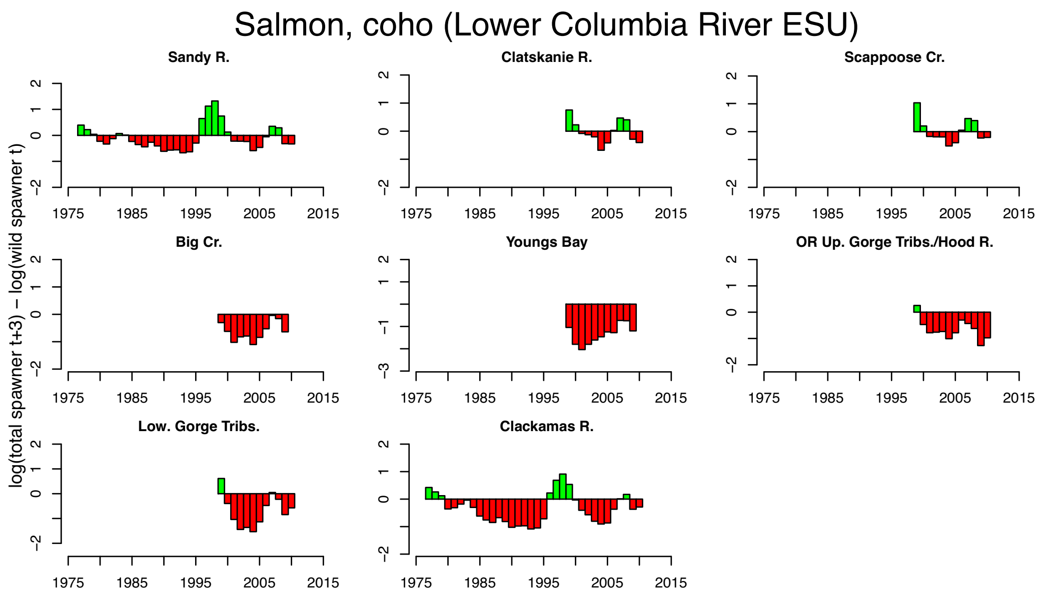

Figure 65 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 3).

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Table 41 -- 5-year geometric mean of raw natural-origin spawner (NOS) counts. This is the raw total spawner count times the fraction NOS estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of NOS available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Tilton R.	Cascade		(3453)	(13370)	(3598)	(6668)	(85)
Sandy R.	Cascade	552 (552)	228 (228)	883 (1068)	1029 (1082)	1250 (1373)	21 (27)
Up. Cowlitz R.	Cascade		(6199)	(37862)	(20189)	(10101)	(-50)
Clatskanie R.	Coastal			286 (372)	664 (801)	903 (1074)	36 (34)
Scappoose Cr.	Coastal			503 (536)	463 (468)	589 (589)	27 (26)
Big Cr.	Coastal			169 (641)	339 (476)	263 (411)	-22 (-14)
Youngs Bay	Coastal			191 (1357)	48 (178)	112 (271)	133 (52)
OR Up. Gorge Tribs./Hood R.	Gorge				205 (317)	297 (1082)	45 (241)
Low. Gorge Tribs.	Gorge				239 (678)	232 (358)	-3 (-47)
WA Up. Gorge Tribs./White Salmon R.	Gorge				(98)	(79)	(-19)
Clackamas R.	Willamette-Cascade	1811 (2787)	499 (768)	2929 (4539)	2942 (5168)	2116 (2755)	-28 (-47)

Table 42 -- 15-year trends in log natural-origin spawner (NOS) abundance computed from a linear regression applied to the smoothed NOS log abundance estimate. Only populations with at least 4 NOS estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Sandy R.	Cascade	0.04 (-0.02, 0.1)	0 (-0.03, 0.03)
Clatskanie R.	Coastal		0.05 (0.02, 0.09)
Scappoose Cr.	Coastal		0.01 (-0.02, 0.04)
Big Cr.	Coastal		0.06 (0.03, 0.1)
Youngs Bay	Coastal		0.03 (0.01, 0.06)
OR Up. Gorge Tribs./Hood R.	Gorge		
Low. Gorge Tribs.	Gorge		
Clackamas R.	Willamette-Cascade	0.06 (-0.01, 0.12)	0.03 (-0.01, 0.06)

HARVEST

Lower Columbia River coho salmon are part of the Oregon Production Index, and are harvested in ocean fisheries primarily off the coasts of Oregon and Washington, with some harvest that historically occurred off of the West Coast Vancouver Island (WCVI). Canadian coho salmon fisheries were severely restricted in the 1990s to protect upper Fraser River coho, and have remained so ever since. Ocean fisheries off California were closed to coho retention in 1993 and have remained closed ever since. Ocean fisheries for coho off of Oregon and Washington were dramatically reduced in 1993 in response depressed status of Oregon Coast natural coho and subsequent listing, and moved to mark-selective fishing beginning in 1999. Lower Columbia River coho benefitted from the more restrictive management of ocean fisheries. Overall exploitation rates regularly exceeded 80% in the 1980s, but have remained below 30% since 1993 (Figure 66). In addition, freshwater fisheries impacts on naturally-produced coho salmon have been markedly reduced through the implementation of mark-selective fisheries. The most recent impact rate for Lower Columbia River coho salmon was 17.1% in 2014 (TAC 2015).

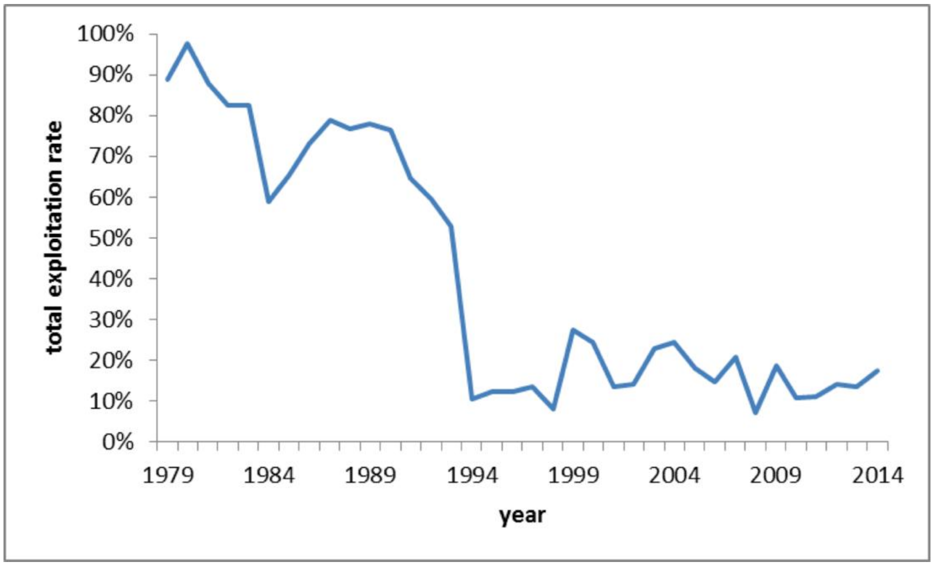

Figure 66 -- Total exploitation rate on lower Columbia River natural coho salmon. Data prior to 2005 from TAC (2014); 2005-2014 from STT 2015.

SPATIAL STRUCTURE AND DIVERSITY

HATCHERIES

Hatchery releases have remained relatively steady at 10–17 million since the 2005 BRT report (Figure 67). The HSRG (2009) reported that overall hatchery production remains relatively high (15.7 million coho released in tributary programs and 2.1 million released in SAFE areas). Most of the populations in the ESU contain a substantial number of hatchery-origin spawners. Recent efforts to shift production into localized areas (e.g., Youngs Bay and Big Creek) in order to reduce the influence of hatchery fish in other nearby populations (e.g., Scappoose and Clatskanie) are considered as in transition at this time. Reductions were also noted in the number of hatchery-origin juvenile coho salmon released into the Sandy River. Mass marking of hatchery-released fish, in conjunction with expanded coho spawning surveys, has provided more accurate estimates of hatchery straying.

Integrated hatchery programs were developed in a number of basins to limit the loss of genetic diversity. The integrated program in the Cowlitz River was recently initiated using predominantly natural-origin broodstock. Large scale releases of hatchery-origin coho salmon adults into the Upper Cowlitz, Cispus, and Tilton rivers is likely partly responsible for the high numbers of returning NORs. An integrated program for Type N coho as been ongoing in the Lewis River for over a decade. Still, the majority of hatchery production is from segregated programs and few populations met the HSRG (2009) criteria for primary or contributing populations.

The HSRG (2009) recommended a number of infrastructure changes to hatcheries to improve the homing and collection of returning hatchery fish. Overall the HSRG (2009) report concludes that changes in hatchery programs alone are unlikely to result in populations achieving their recovery goals without additional changes in harvest (more selective fisheries to remove hatchery-origin fish) and improvements in habitat.

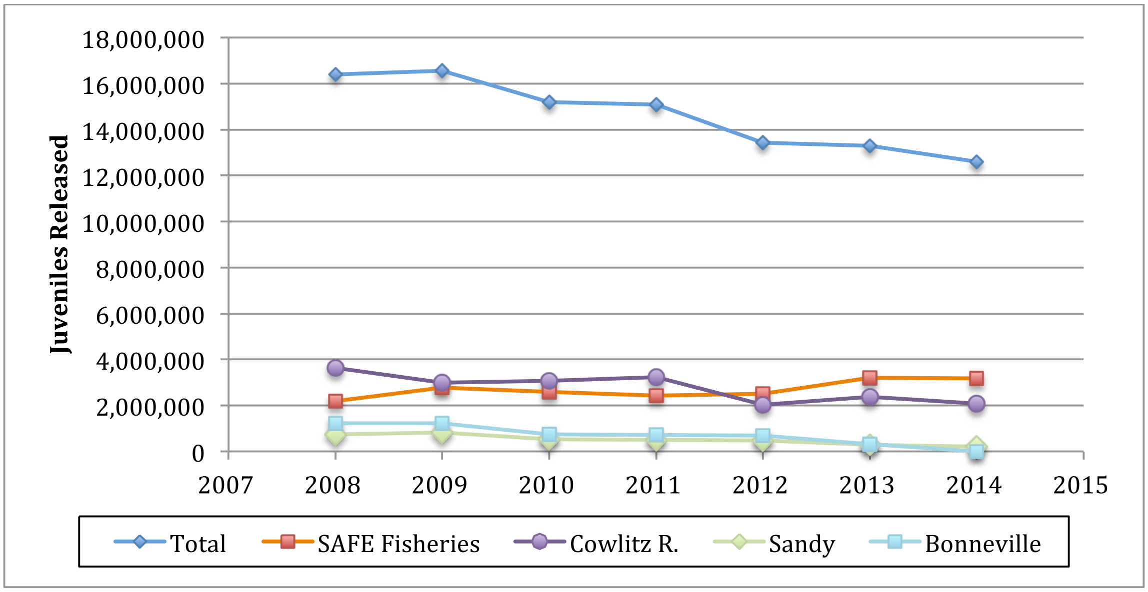

Figure 67. Annual releases of coho salmon juveniles into the LCR ESU from 2008 to 2014 (2014 levels may be incomplete). Individual tributary releases presented are for those programs that exhibited substantial changes from 2008 to 2014. Data from RMPC (http://www.rmpc.org/ accessed January 6, 2015).

Table 43 -- 5-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Tilton R.					
Sandy R.	1.00	1.00	0.87	0.95	0.91
Up. Cowlitz R.					
Clatskanie R.			0.82	0.85	0.85
Scappoose Cr.			0.94	0.99	1.00
Big Cr.			0.33	0.61	0.65
Youngs Bay			0.14	0.37	0.42
OR Up. Gorge Tribs./Hood R.			0.40	0.67	0.31
Low. Gorge Tribs.			0.84	0.40	0.69
WA Up. Gorge Tribs./White Salmon R.					
Clackamas R.	0.65	0.65	0.67	0.62	0.81

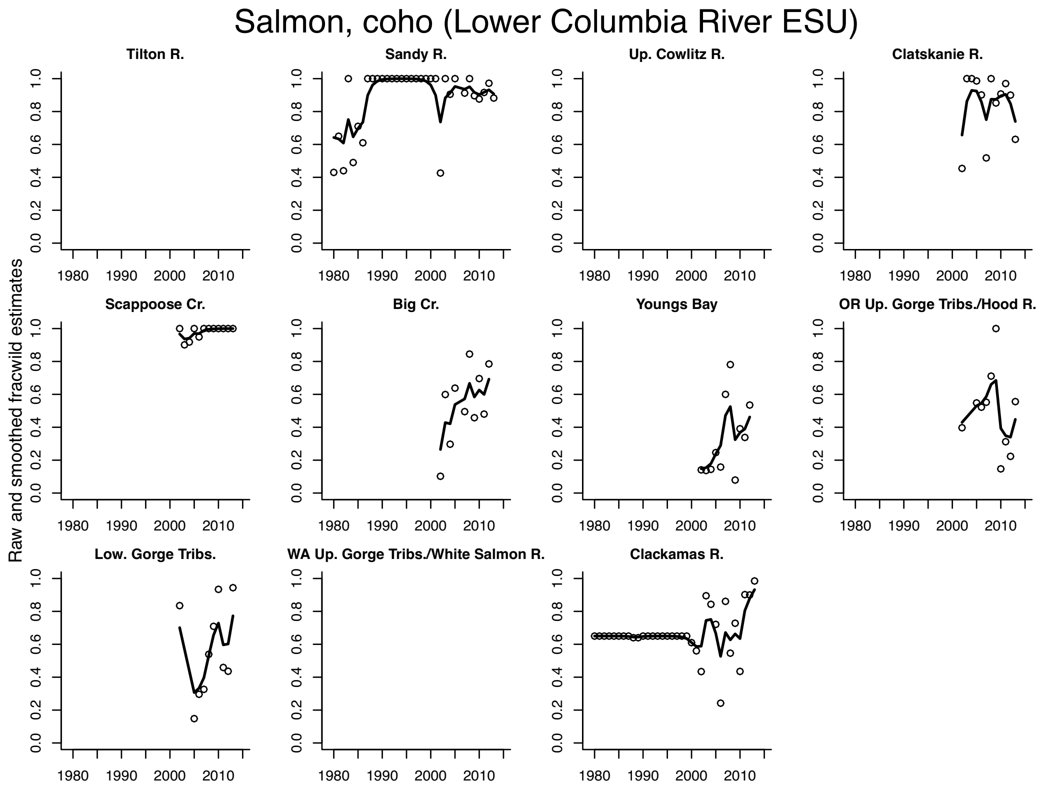

Figure 68 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

SPATIAL STRUCTURE

There have been a number of large-scale efforts to improve accessibility, one of the primary metrics for spatial structure, in this ESU. On the Hood River, Powerdale Dam was removed in 2010 and while this dam previously provided fish passage removal of the dam is thought to eliminate passage delays and injuries. Condit Dam, on the White Salmon River, was removed in 2011 and this provided access to previously inaccessible habitat. Current monitoring efforts do not include coho salmon surveys, so the extent of recolonization is unknown. Fish passage operations (trap and haul) were begun on the Lewis River in 2012, reestablishing access to historically-occupied habitat above Swift Dam (RKm 77.1), juvenile passage efficiencies are still relatively poor. In addition, efforts to provide downstream juvenile passage at the Cowlitz Dam complex collection sites began in the 1990s, and since that time there has been a number of modifications in the facilities and a gradual increase in the numbers of naturally-produced coho salmon adults. Coho salmon returns to the Cispus/Upper Cowlitz basin from 2002-2007 exhibited an overall recruit per spawner ratio of 0.80 (Serl and Morrill 2010). Presently, the trap and haul program for the Upper Cowlitz, Cispus, and Tilton River populations are the only means by which coho salmon can access spawning habitat for these populations. A trap and haul program also currently maintains access to the North Toutle River above the sediment retention structure (SRS), with a coho salmon and steelhead being passed above the dam (Liedtke *et al.* 2013). This SRS transportation program relocates coho salmon into the North Fork Toutle DIP[20]; however, there are limited release sites and only a portion of the upper watershed is accessible. On a more general basis, there have been a number of recovery actions throughout the ESU to remove or improve culverts and other small-scale passage barriers. Many of these actions have occurred too recently to be fully evaluated; however, some data is now available for many actions that occurred prior to 2010, but were not able to be assessed in the previous status review. These include the removal of Marmot Dam in 2007 and the Little Sandy River diversion dam in 2008. Additionally, access to habitat above the Sandy River Hatchery weir on Cedar Creek (Sandy River Basin) was restored in 2010.

For a number of projects where passage has been restored through dam structural modification or operational modification it remains to be demonstrated that both adult and juvenile passage survival is sufficient to provide some level of self-sufficiency to upstream population components. If recruit:spawner ratios are well below one, it is unlikely that there is any benefit to population special structure and passage operations may actually represent a net loss in productivity and abundance.

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The estimated changes in VSP status for coho salmon populations in Figure 69 reflect both improvements in abundance, diversity, and spatial structure and improvements in monitoring. As discussed earlier, previous status reviews lacked adequate quantitative data on abundance and hatchery contribution for a number of populations. During previous status reviews, anecdotal information provided suggested that hatchery-origin fish dominated many of the populations and that natural productivity was very low. Recent surveys provide a more accurate understanding of the status of these populations; however, with only two or three years of data it is not possible to determine whether there has been a true improvement in status, but certainly the contribution of naturally-produced fish is much higher than previously thought.

A total of 6 of 23 populations are at or near their recovery viability goals (Figure 69), although under the recovery plan scenario none of these populations had recovery goals above 2.0 (moderate risk). The remaining populations generally require a higher level of viability and most require substantial improvements to reach their viability goals. In the Coastal MPG, the Scappoose Creek DIP is the only population at moderate risk, with the Clatskanie River DIP at moderate to high risk and the others remain at high risk. Similarly, in the Cascade MPG, the Clackamas River DIP, and the Upper Cowlitz and Cispus DIPs[21] may be in the moderate to low risk categories, with the remainder of the DIPs being at moderate to high risk. All of the populations in the Gorge MPG are likely in the very high risk category.

For most populations the proportion of hatchery origin fish naturally spawning exceeds criteria set for primary and contributing populations. With recent dam removals and the initiation of trap and haul programs there are few major spatial structure limitations; however, smaller migrational barriers, such as culverts limit spatial structure.

[20] North Fork Toutle River coho salmon currently have volitional access only to the Green River, a tributary to the North Fork Toutle.
[21] The Upper Cowlitz River and Cispus River DIPs are currently treated a single demographic unit by WDFW.

Improved monitoring has substantiated the presence of natural-origin coho salmon in a number of populations previously thought to be dominated by hatchery production; however, overall abundance is still relatively low. Furthermore, none of the MPGs meet the criteria for viability. The Lower Columbia River coho salmon ESU most likely remains at the moderate risk category.

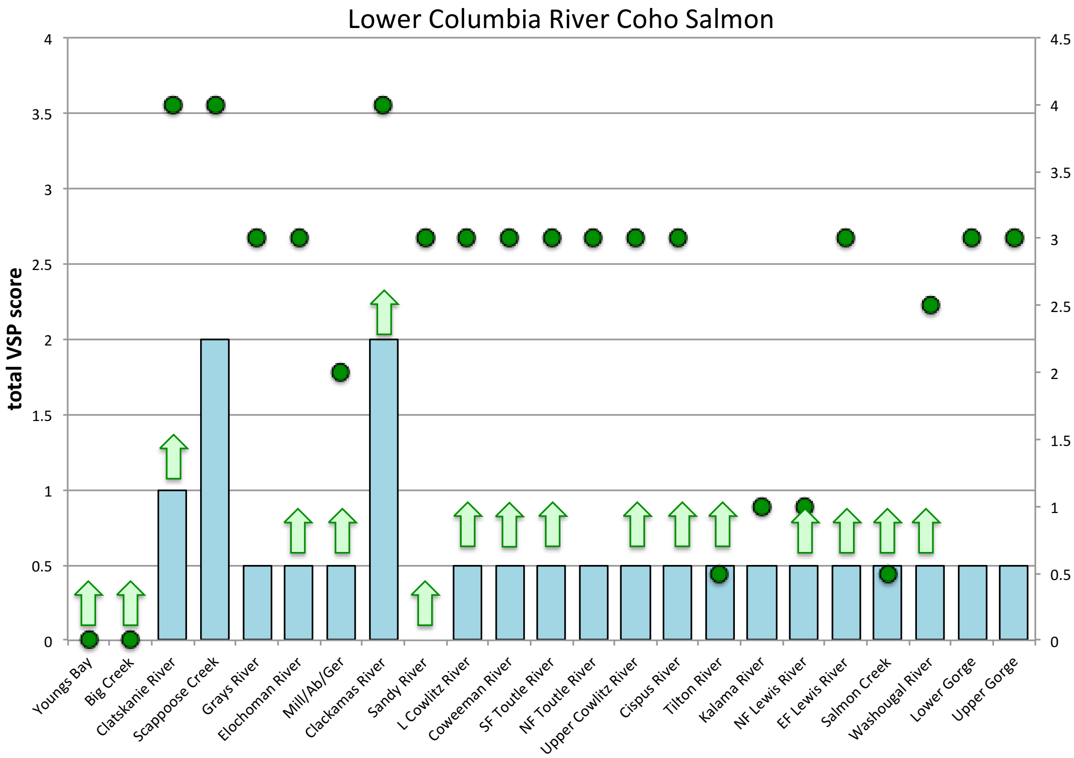

Figure 69. VSP status of demographically independent populations in the Lower Columbia River coho salmon ESU, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013), green circles indicate the recovery goals. Arrows indicate the general direction, but not the magnitude, of any VSP population score based on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany et al. 2006). A VSP score of 3.0 is represents a population with a 5% risk of extinction within a 100 year period.

UPDATED BIOLOGICAL RISK SUMMARY

The status of a number of coho populations have changed since the reviews by McElhany *et al* (McElhany *et al.* 2006), Ford *et al.* (2012) and Dornbush (2013). Changes in abundance and productivity, diversity and spatial structure were generally positive; however, this appears to be mostly due to the improved level of monitoring (and therefore understanding of status) in Washington tributaries rather than a true change in status over time. In the absence of specific abundance and diversity data, earlier status reviews had concluded that hatchery origin fish dominated many of the coho populations in the Lower Columbia River ESU and that there was little

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natural productivity. Recent recovery efforts may have contributed to the observed natural production, but in the absence of longer term data sets it is not possible to parse out these effects. Populations with longer term data sets exhibit stable or slightly positive abundance trends. Some trap and haul programs appear to be operating at or near replacement, although other programs still are far from that threshold and require supplementation with additional hatchery-origin spawners . Initiation of or improvement in the downstream juvenile facilities at Cowlitz Falls, Merwin, and North Fork Dam are likely to further improve the status of the associated upstream populations. While these and other recovery efforts have likely improved the status of a number of coho salmon DIPs, abundances are still at low levels and the majority of the DIPs remain at moderate or high risk. For the Lower Columbia River region land development and increasing human population pressures will likely continue to degrade habitat, especially in lowland areas. Although populations in this ESU have generally improved, especially in the 2013/14 and 2014/15 return years (Figure 69), recent poor ocean conditions suggest that population declines might occur in the upcoming return years (see Environmental trends section below). Regardless, this ESU is still considered to be at moderate risk.

LOWER COLUMBIA RIVER STEELHEAD DPS

BRIEF DESCRIPTION OF DPS

The DPS includes all naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassable barriers in streams and tributaries to the Columbia River between the Cowlitz and Wind Rivers, Washington (inclusive), and the Willamette and Hood Rivers, Oregon (inclusive), as well as multiple artificial propagation programs. Myers *et al*. (2006) identified 23 DIPs, including 6 summer-run steelhead populations and 17 winter-run populations (Figure 70).

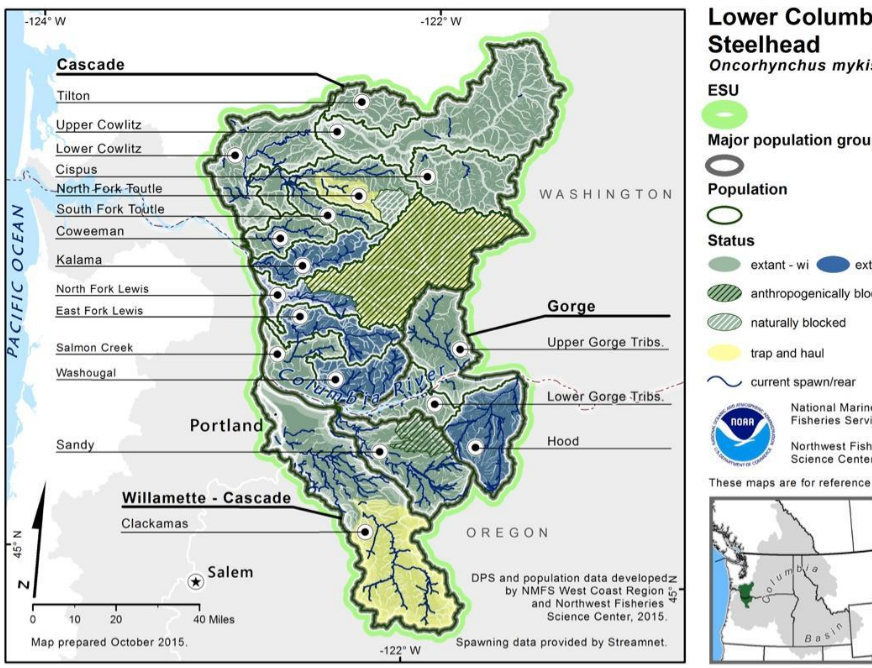

Figure 70 – Map of populations in the Lower Columbia River steelhead DPS.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

In 2005, a large majority (over 73%) of the BRT votes for this ESU fell in the "likely to become endangered" category, with small minorities falling in the "in danger of extinction" and "not likely to become endangered" categories (Good *et al*. 2005). The BRT found moderate risks in all the VSP categories. All of the major risk factors identified by previous BRTs still remained. Most populations were at relatively low abundance, and those with adequate data for modeling were estimated to have a relatively high extinction probability. Some populations, particularly summer run, had higher returns in the most recent years included in the 2005 report (years 2001 and 2002). The WLC-TRT (Myers *et al.* 2006) estimated that at least four historical populations were extirpated. The hatchery contribution to natural spawning remained high in many populations.

2010

Ford *et al*. (2011) summarized three status evaluations of LCR steelhead status, all based on WLC-TRT criteria, which had been conducted since the last BRT status update in 2005. All three evaluations concluded that the ESU was currently at high risk of extinction. Of the 26 historical populations in the ESU, 17 were considered at high or very high risk. Populations in the upper Lewis, Cowlitz and White Salmon watersheds remained cut-off from access to essential spawning habitat by hydroelectric dams. Projects to reestablish access had been initiated in the Cowlitz and Lewis systems but these were not yet produced self-sustaining populations. The populations generally remained at relatively low abundances with low productivity. Overall, the information considered did not indicate a change in the biological risk category since the time of the 2005 BRT status review.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

For most of the populations in this DPS abundance estimates for winter-run steelhead were calculated by expanding redd counts from index and census surveys and for summer-run steelhead a mark-resight survey of adult during prespawn holding is employed (Rawding and Rodgers 2013). In many cases, river conditions limit access and visibility during winter steelhead spawning creating some uncertainty in the expansion to total spawner abundance. Since the last status review there has ben an effort to standardize survey methods and validate redd to adult expansions. Where tributaries contained dams, abundance and hatchery proportions were estimated by direct adult counts, or a combination of redd surveys and dam counts. Weirs were operated in some tributaries to count adults and remove or exclude hatchery-origin adults. Where dams have been removed, (as in the Sandy River) spawner surveys have been expanded on tributaries.

ABUNDANCE AND PRODUCTIVITY

WINTER-RUN WESTERN CASCADE MPG

This MPG includes native winter-run steelhead in 14 DIPs from the Cowlitz River to the Washougal River, inclusive (Figure 70). Abundances have remained fairly stable and, in general, are correlated with cyclical changes in ocean condition (Figure 71). For most populations, total abundances and natural-origin abundances (where available) have remained low, averaging in the hundreds of fish (Table 44). Notable exceptions to this were the Clackamas[22] and Sandy River winter-run steelhead

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[22] For the Clackamas River winter steelhead population, the North Fork Dam count provided the longest available data set for statistical analysis. This data set does not include winter steelhead spawning below the dam (for which we have a shorter time series based on redd count expansions). For 2013 and 2014, total spawners below the dam were 1831 (85% NOR) and 2171 (99% NOR), respectively (Jacobsen *et al*. 2014).

populations, that are exhibiting recent rises in NOR abundance and maintaining low levels of hatchery-origin steelhead on the spawning grounds (Jacobsen *et al.* 2014). Abundances in the Tilton and Upper Cowlitz/Cispus rivers are highly variable, in part because of ongoing changes in collection efficiency of juvenile downstream passage structures as well as the use of natural-origin adults as broodstock in developing an integrated hatchery stock.

SUMMER-RUN CASCADE MPG

There are four summer-run steelhead DIPs in the MPG: Kalama River, North Fork Lewis River, East Fork Lewis River, and Washougal River (Figure 70). Until recently migratory access to the North Fork Lewis River summer-run DIP was blocked by a series of impassable dams, although summer-run are not currently being considered as part of the reintroduction program. There is some uncertainty regarding the status of this population, specifically if residualized *O. mykiss* contain a genetic legacy of the historical population and if they are capable of reinitiating an anadromous life-history. Long and short term trends for the Kalama, East Fork Lewis and Washougal DIPs are positive, absolute abundances have been in the hundreds of fish. The most recent surveys (2014) indicate a drop in abundance for all three DIPs. Whether this is a portent of changing oceanic conditions is not clear, but is of some concern regardless of its cause.

WINTER-RUN GORGE MPG

This MPG contain three DIPs, Lower Gorge, Upper Gorge, and Hood River. In both the Lower and Upper Gorge populations surveys for winter steelhead are very limited. Abundance levels have been low, but relatively stable, in the Hood River. In recent years, spawners from the integrated hatchery program have constituted the majority of the naturally spawning fish.

SUMMER-RUN GORGE MPG

The Wind River and Hood River are the two DIPs in this MPG. Hood River summer-run steelhead have not been monitored since the last status review; efforts are currently underway to provide accurate estimates of fish ascending the West Fork of the Hood River. Adult abundance in the Wind River remains stable, but at a low level (hundreds of fish; Table 44). In addition, there are catch and release fishery for natural summer run steelhead in the Wind and Hood Rivers, but encounter and incidental mortality estimates are not available, but impacts are likely to be relatively low. Given the presence of only two summer-run DIPs in this MPG, the overall status of the MPG is uncertain.

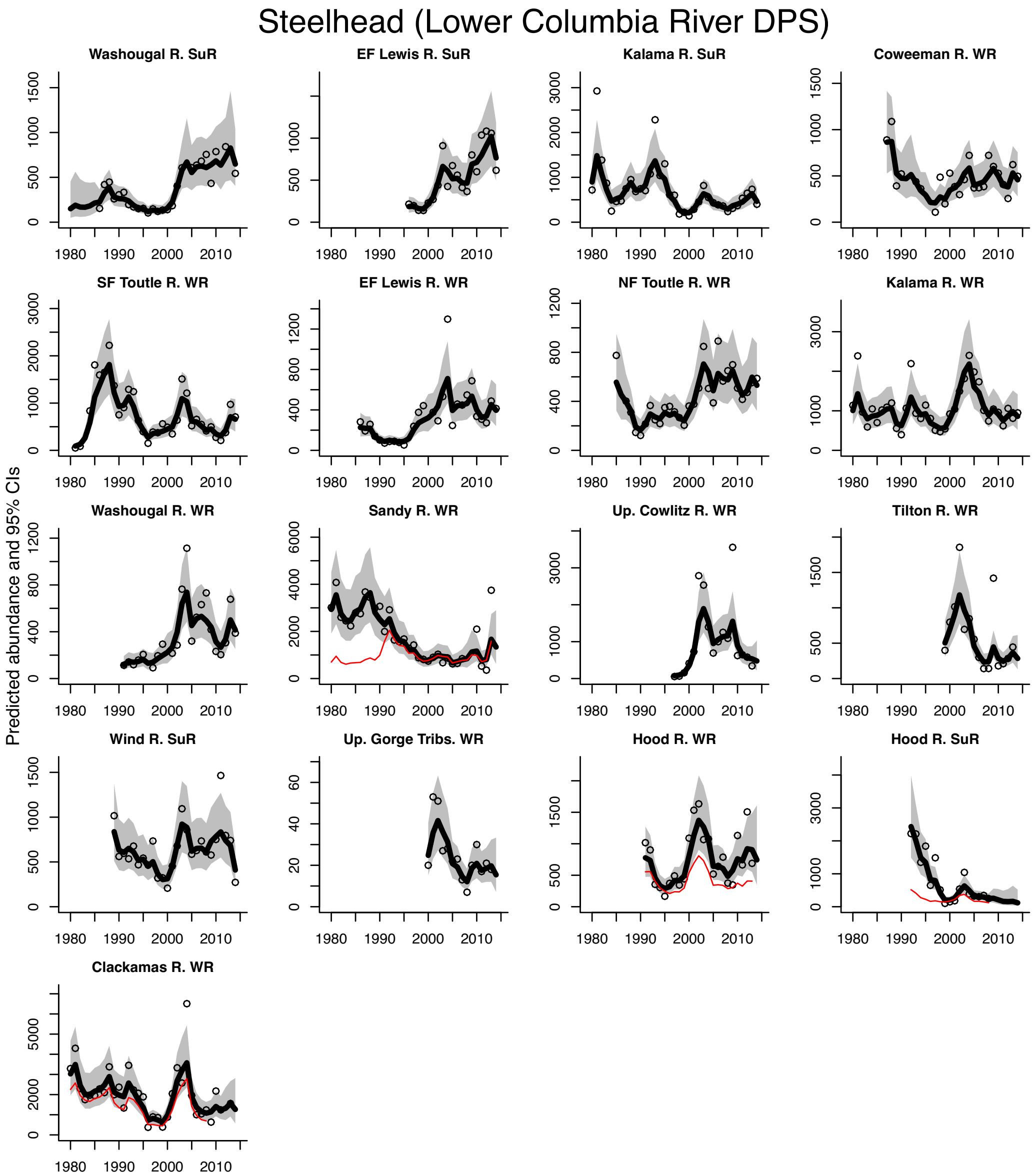

Figure 71 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

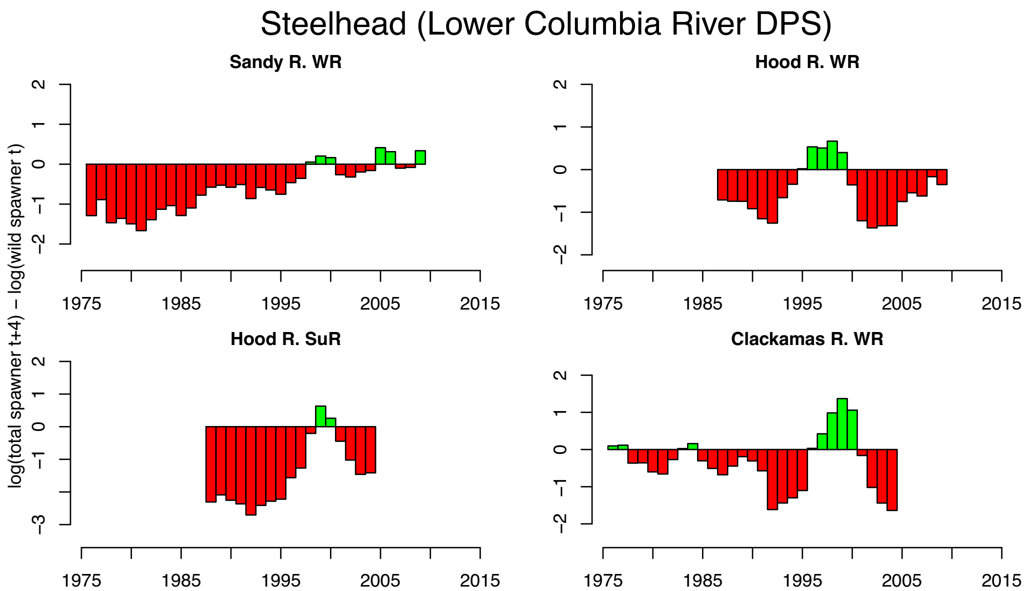

Figure 72 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).

Table 44 -- 5-year geometric mean of raw natural-origin spawner (NOS) counts. This is the raw total spawner count times the fraction NOS estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of NOS available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Washougal R. SuR	Cascade	(220)	(131)	(282)	(612)	(712)	(16)
EF Lewis R. SuR	Cascade		(170)	(402)	(539)	(849)	(58)
Kalama R. SuR	Cascade	(1060)	(454)	(382)	(338)	(518)	(53)
Coweeman R. WR	Cascade	(436)	(218)	(458)	(470)	(443)	(-6)
SF Toutle R. WR	Cascade	(928)	(344)	(725)	(521)	(432)	(-17)
EF Lewis R. WR	Cascade	(85)	(214)	(525)	(453)	(356)	(-21)
NF Toutle R. WR	Cascade	(221)	(293)	(495)	(616)	(504)	(-18)
Kalama R. WR	Cascade	(931)	(654)	(1443)	(1219)	(866)	(-29)
Washougal R. WR	Cascade	(132)	(182)	(479)	(504)	(328)	(-35)
Sandy R. WR	Cascade	1411 (2148)	1058 (1173)	833 (834)	698 (699)	997 (1103)	43 (58)
Up. Cowlitz R. WR	Cascade		(82)	(1242)	(1273)	(532)	(-58)
Tilton R. WR	Cascade			(975)	(343)	(262)	(-24)
Wind R. SuR	Gorge	(563)	(454)	(569)	(625)	(707)	(13)
Up. Gorge Tribs. WR	Gorge			(33)	(16)	(21)	(31)
Hood R. WR	Gorge	457 (561)	206 (341)	751 (1256)	282 (509)	421 (940)	49 (85)
Hood R. SuR	Gorge	386 (1878)	127 (622)	255 (358)	151 (303)		
Clackamas R. WR	Willamette-Cascade	1597 (2189)	486 (733)	1946 (2514)	862 (1100)	(1615)	(47)

Table 45 --15-year trends in log natural-origin spawner (NOS) abundance computed from a linear regression applied to the smoothed NOS log abundance estimate. Only populations with at least 4 NOS estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Sandy R. WR	Cascade	-0.05 (-0.08, -0.02)	0.02 (0, 0.05)
Hood R. WR	Gorge	0.02 (-0.03, 0.07)	-0.04 (-0.07, 0)
Hood R. SuR	Gorge	-0.05 (-0.1, 0)	
Clackamas R. WR	Willamette-Cascade	0.02 (-0.05, 0.09)	

HARVEST

There is no direct harvest of naturally-produced steelhead in the Lower Columbia River DPS, other than the Zone 6 treaty fishery above Bonneville Dam. Steelhead are also intercepted in mainstem and tributary fisheries targeting non-listed hatchery and naturally-produced salmon, and hatchery steelhead. Mark-selective commercial tangle net fisheries in the mainstem Columbia River occur during the winter-spring time frame and primarily affect wild winter steelhead. Impact rates on all winter steelhead are estimated to be less than 2% (U.S. v Oregon biological assessment). Release mortality rates are estimated to be less than 15% during this fishery (TAC or US v OR BA). During the 2014 season an estimated 350 unmarked winter steelhead were encountered with a 20% mortality rate, 70 fish (ODFW & WDFW 2015). Recreational fisheries targeting marked hatchery-origin steelhead encounter natural-origin fish at a relatively high rate, but hooking mortality rates are generally lower than release mortality rates in the commercial fisheries. Estimated mainstem fishery mortality for naturally produced winter-run steelhead has averaged 2.2% (2009-2013) for non-Indian commercial and recreational fisheries (ODFW and WDFW 2015). The impact rate for Lower Columbia River winter steelhead in mainstem fisheries in 2014 was 0.6% (TAC 2015).

See the chapter on Middle Columbia River Steelhead for a discussion of trends in harvest rates for Columbia Basin steelhead.

HATCHERIES

Total steelhead hatchery releases in the Lower Columbia River Steelhead DPS have decreased since the last status review, declining from an total (summer and winter run) release of approximately 3.5 million to 3 million (Figure 73) from 2008 to 2014. Some populations continue to have relatively high fractions of hatchery-origin spawners, whereas others (e.g., Wind) have relatively few hatchery-origin spawners (Table 46, Figure 74). One of the major changes in hatchery operations was the elimination of the out-of-DPS steelhead broodstock programs in the Cowlitz River Basin. The early-winter Chambers Creek program was replaced by an integrated late-winter steelhead program, and Skamania summer-run releases were terminated in the NF Toutle River. Out of DPS releases of Skamania summer-run and Chambers Creek early-winter-run steelhead have also been terminated in the EF Lewis River. Integrated broodstocks have been developed for a number of populations (Cowlitz River, Kalama River, NF Lewis River and Sandy River) and populations in the Wind, East

Fork Lewis, and Green rivers have been designated gene-banks with no further hatchery releases; however, out of DPS stocks continue to be released, primarily early-winter Chambers Creek steelhead and summer-run Skamania steelhead into a number of basins, including the Kalama River, Lewis River, Salmon Creek, and Clackamas River. Where hatcheries maintain multiple stocks of steelhead there continues to be some risk of hybridization between different run times or native and out-of-DPS stocks.

Where adults are handled in census (complete capture) upstream passage programs (e.g., Clackamas River, Cowlitz River, Kalama River winter run, and Lewis River) hatchery-origin fish are often removed from the river or recycled for additional harvest opportunities. Some portion of hatchery-origin fish are also removed in locations with partial capture adult traps (Wind River, Kalama River summer run, Washougal River.), in the some Kalama River hatchery-origin summer-run steelhead are able to ascend the Falls and avoid being captured in the fish ladder.

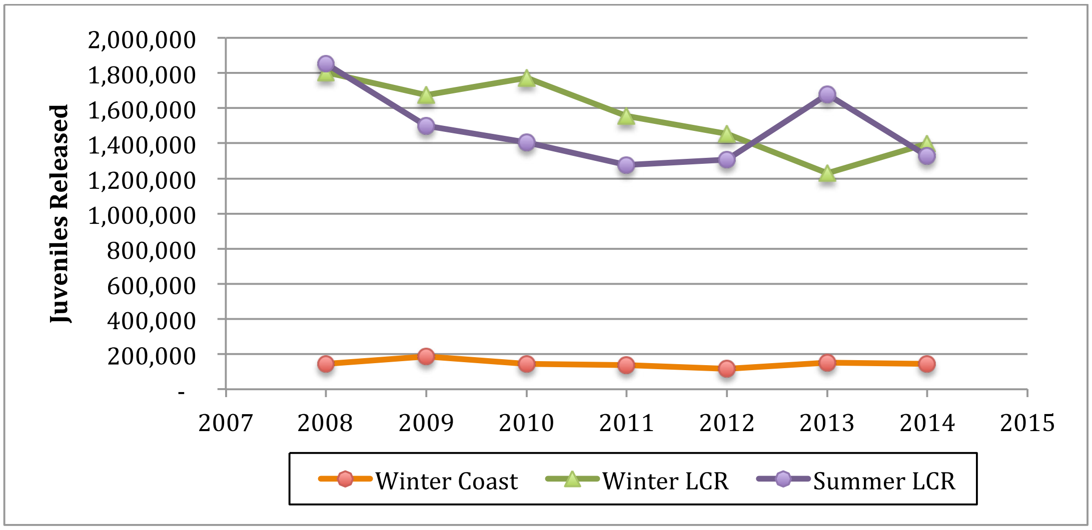

Figure 73 - Annual releases of juvenile steelhead into the Lower Columbia Rive, by run timing, from 2008 to 2014. LCR indicates releases into the Lower Columbia River DPS and Coast represents releases into Columbia River tributaries downstream of the LCR DPS, coastal stratum. Data from RMIS (http://www.rmpc.org/ accessed January 6, 2015).

SPATIAL STRUCTURE

There have been a number of large-scale efforts to improve accessibility (one of the primary metrics for spatial structure) in this ESU. Efforts to provide access to the Upper Cowlitz Basin (Upper Cowlitz, Cispus, and Tilton Rivers) began in 1996 with the initiation of juvenile collection at Cowlitz Falls Dam. There have been a number of structural and operational changes at the dam to improve collection efficiency; however, spawner:adult ratios have achieved replacement in only a few years since the initiation of the program (Serl et al 2010, Serl et al. 2014)[23]. More recently on the Hood River, Powerdale Dam was removed in 2010 and while this dam previously provided for fish passage, removal of the dam is thought to eliminate passage delays and injuries. Trap and haul operations were begun on the Lewis River in 2012 for winter-run steelhead, reestablishing access to historically-occupied habitat above Swift Dam (RKm 77.1). In 2014, 1033 adult winter steelhead (integrated program fish) were transported to the upper Lewis River; however, juvenile collection efficiency is still below target levels. Finally, there has been a trap and haul operation at the sediment retention structure (SRS) on the North Fork Toutle River since 1989. The escapement of winter steelhead to the North Fork Toutle River represents about one-half of the recent (2010-2014)[24] natural-origin abundance in this DIP. Transportation above the SRS is limited to two small tributaries and only a small proportion of the upper basin is utilized. In addition, there have been a number of recovery actions throughout the ESU to remove or improve culverts and other small-scale passage barriers. Many of these actions (including the removal of Condit Dam on the White Salmon River) have occurred too recently to be fully evaluated; however, data is now available for many actions that occurred prior to 2010, but were not able to be assessed in the previous status review. These include the removal of Marmot Dam in 2007 and the Little Sandy River diversion dam in 2008, and Hemlock Dam on Trout Creek (wind River) in 2009. Additionally, beginning in 2010, unmarked steelhead have been passed above the hatchery weir on Cedar Creek, a tributary to the Sandy River.

For a number of projects where passage has been restored through dam removal, structural modification, or operational modification it remains to be demonstrated that both adult and juvenile passage survival is sufficient to provide some level of self-sufficiency to upstream population components. If recruit:spawner ratios are well below one, it is unlikely that there is any benefit to population spatial structure and passage operations may represent a net loss in productivity and abundance.

[23] The juvenile fish passage program is targeting four species: Chinook salmon, coho salmon, cutthroat trout, and winter steelhead. Optimization of fish collection efficiencies (FCEs), especially for subyearling Chinook salmon, often results in declines in other species.
[24] SASI (https://fortress.wa.gov/dfw/score/score/species/population_details.jsp?stockId=6714), accessed July 14, 2015. The Green River, a tributary to the North Fork Toutle River, accounts for the majority of accessible spawning habitat.

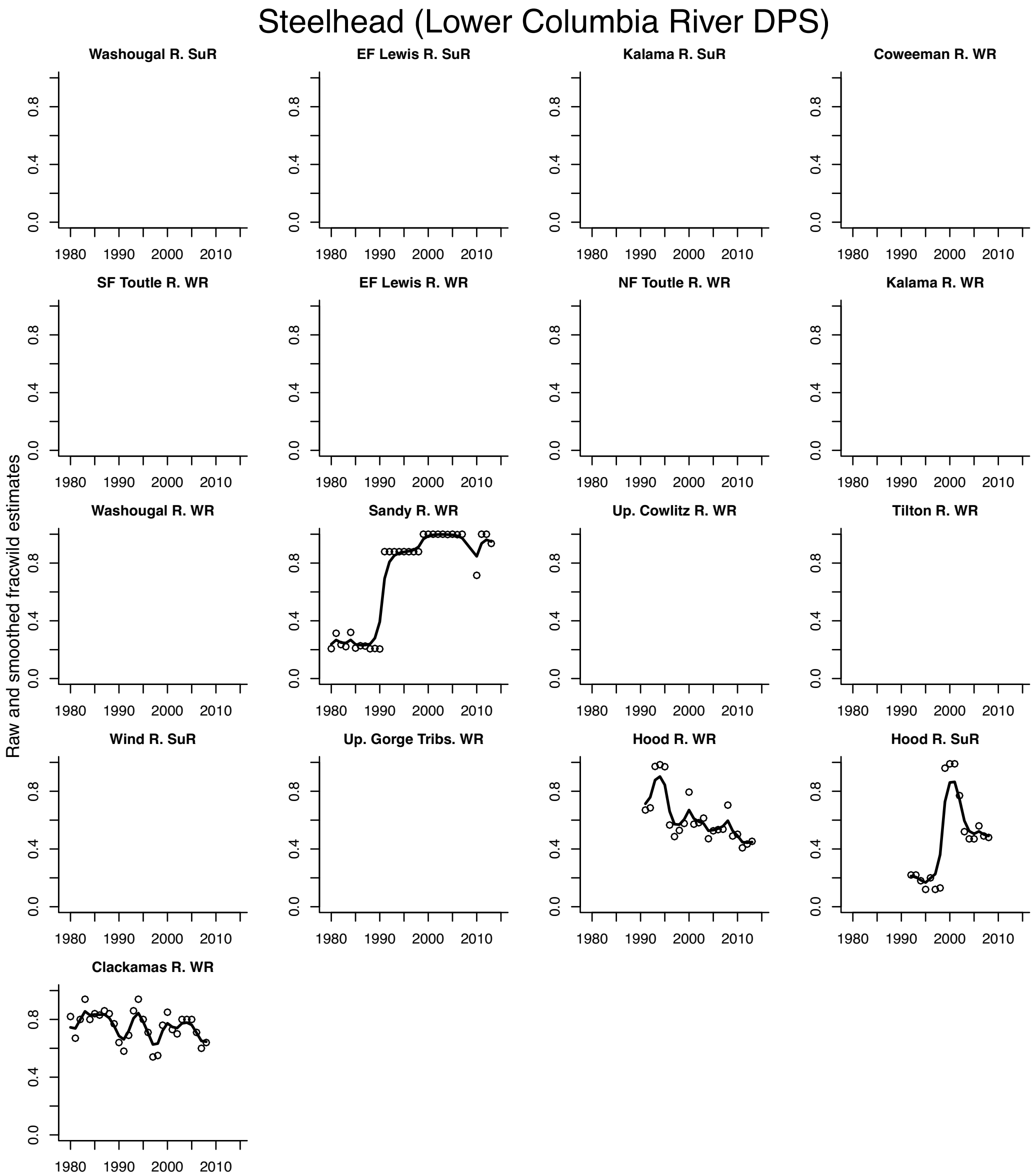

Figure 74 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 46 -- 5-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Coweeman R. WR					
EF Lewis R. SuR					
EF Lewis R. WR					
Kalama R. SuR					
Kalama R. WR					
NF Toutle R. WR					
Sandy R. WR	0.74	0.90	1.00	1.00	0.91
SF Toutle R. WR					
Tilton R. WR					
Up. Cowlitz R. WR					
Washougal R. SuR					
Washougal R. WR					
Hood R. SuR	0.21	0.31	0.75	0.50	
Hood R. WR	0.83	0.63	0.61	0.56	0.45
Up. Gorge Tribs. WR					
Wind R. SuR					
Clackamas R. WR	0.74	0.67	0.78	0.69	

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Overall, the status of DIPs relative to their recovery goals is little changed since the last review. A number of changes noted in Figure 75 and Figure 76, reflect "corrections" in the baseline VSP scores assigned in the recovery plan due to improvements in monitoring and updates in the existing databases. For example, the North Fork Toutle River DIP has maintained an natural origin abundance of a few hundred winter steelhead for the last 10 years, and this would suggest a higher VSP score than the 0.5 estimated in the Recovery Plan (Dornbush and Sihler 2013). Additionally, natural origin abundance in the Tilton River has increased over the last five years and the hatchery contribution has been reduced to near zero. A total of 5 of 22 populations are at or near their recovery viability goals, although under the recovery plan scenario only two of these populations had scores above 2.0. The remaining populations generally require a higher level of viability and most require substantial improvements to reach their viability goals. For the summer-run steelhead DIPs, "improvement" in the Hood River summer-run steelhead was related to correcting the previous assumption that lack of data indicated an absence of fish rather than a lack of monitoring. Summer-run steelhead are present in the Hood River, although monitoring was suspended after Powerdale Dam removed.

While there have been improvements in diversity through hatchery reform, spatial structure is still a concern for some populations that rely on adult trap and haul programs and juvenile downstream passage structures for sustainability (although juvenile passage efficiency has generally been higher for steelhead and coho salmon than Chinook salmon).

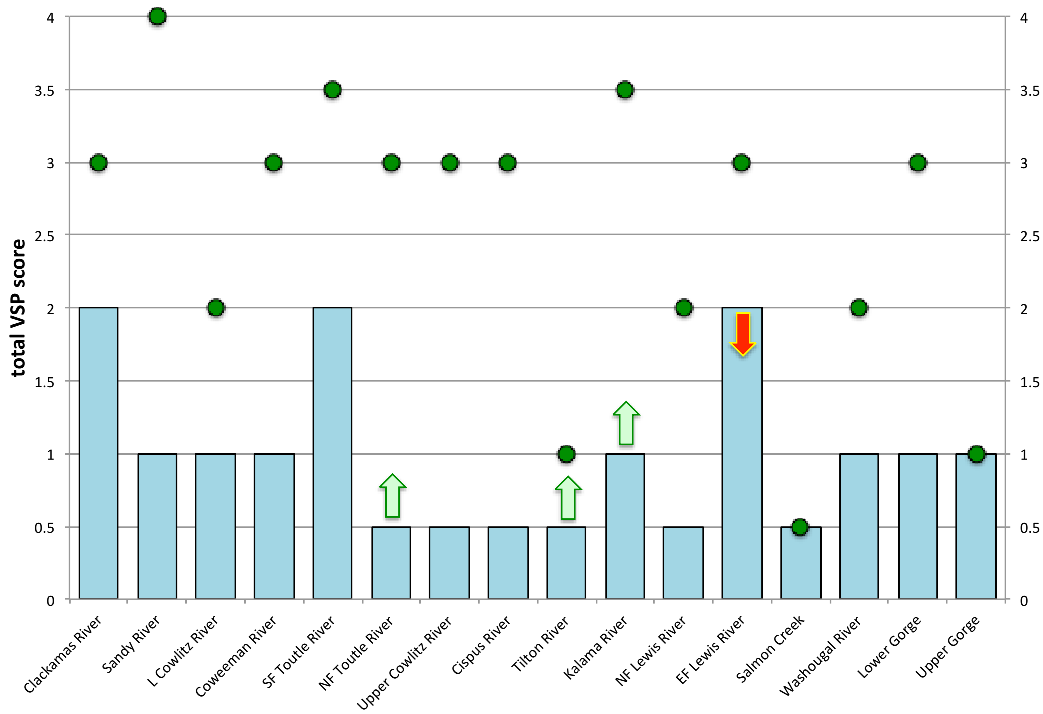

Figure 75. VSP status of winter run demographically independent populations in the Lower Columbia River steelhead DPs, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013), green circles indicate the recovery goals Arrows indicate the general direction, but not the magnitude, of any VSP population score based on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany et al. 2006). A VSP score of 3.0 is represents a population with a 5% risk of extinction within a 100 year period.

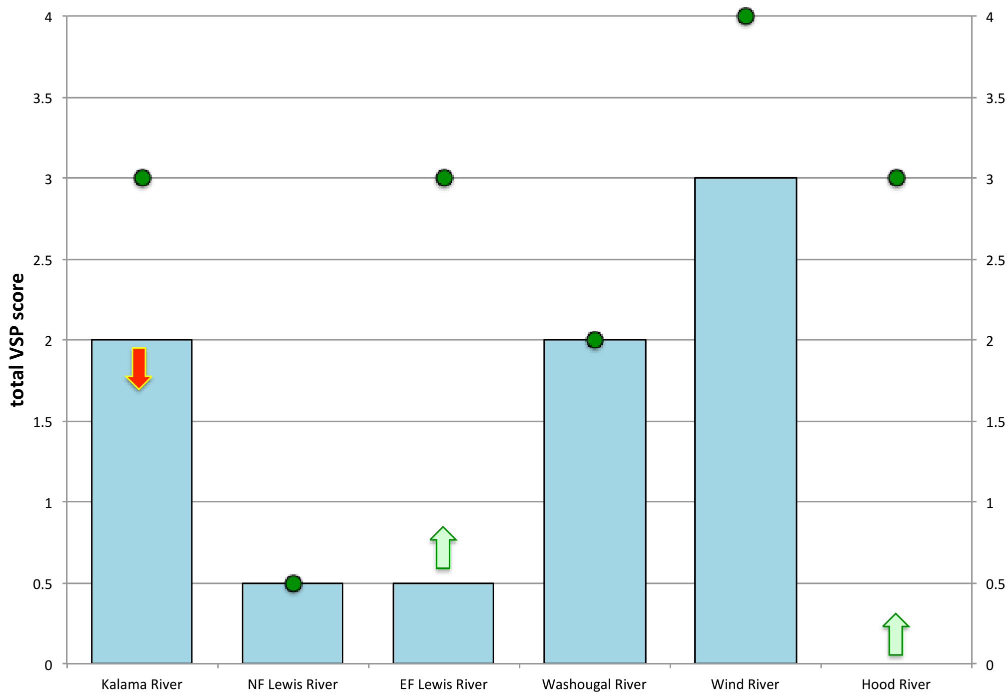

Figure 76. VSP status of summer run demographically independent populations in the Lower Columbia River steelhead DPS, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013), green circles indicate the recovery goals. Arrows indicate the general direction, but not the magnitude, of any VSP population score based on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany et al. 2006). A VSP score of 3.0 is represents a population with a 5% risk of extinction within a 100 year period.

UPDATED BIOLOGICAL RISK SUMMARY

The majority of winter-run steelhead DIPs in this DPS continue to persist at low abundances. Hatchery interactions remain a concern in select basins, but the overall situation is somewhat improved compared to prior reviews. Summer-run steelhead DIPs were similarly stable, but at low abundance levels. The decline in the Wind River summer-run DIP is a source of concern, given that this population has been considered one of the healthiest of the summer-runs; however, the most recent abundance estimates suggest that the decline was a single year aberration. Passage programs in the Cowlitz and Lewis basins have the potential to provide considerable improvements in abundance and spatial structure, but have not produced self-sustaining populations to date. Recent low winter-run returns to the Upper Cowlitz River may be anomalous, related more to the development of an integrated hatchery broodstock and temporary modifications at the Cowlitz Falls Dam to benefit Chinook salmon than to a decline in viability. Efforts to provide passage above North Fork Lewis River dams offer the opportunity for substantial improvements in the winter run steelhead population and the only opportunity to reestablish summer-run steelhead. Habitat degradation continues to be a concern for most populations. Even with modest improvements in the status of several winter-run DIPs, none of the populations appear to be at fully viable status, and similarly none of the MPGs meet the criteria for viability. The DPS therefore continues to be at moderate risk.

COLUMBIA RIVER CHUM SALMON ESU

BRIEF DESCRIPTION OF ESU

The ESU includes all naturally spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon, as well as four artificial propagation programs (Figure 77).

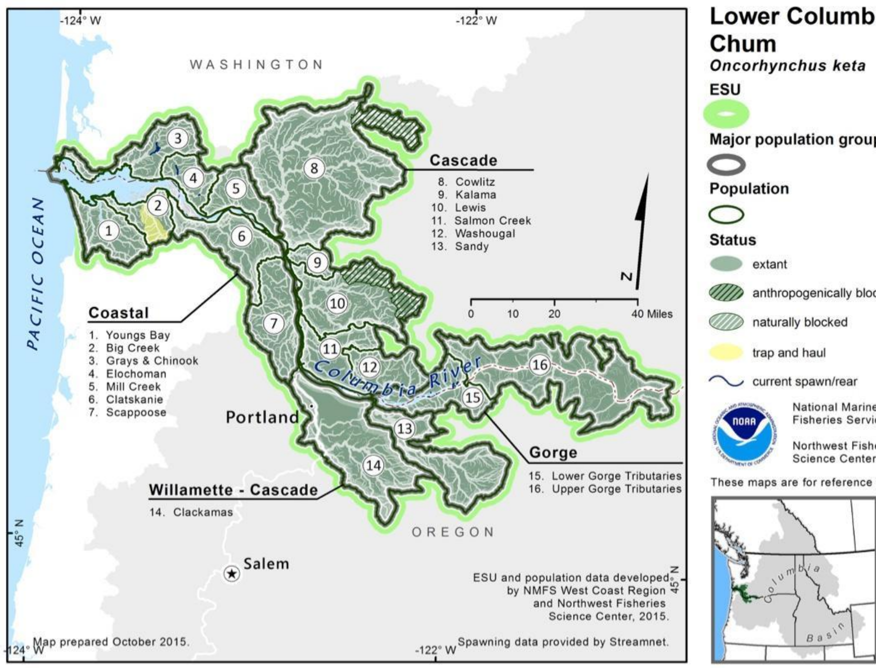

Figure 77 -- Map of the Lower Columbia River chum salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

In the 2005 status review (Good *et al*. 2005), nearly all votes for the Columbia River chum salmon ESU fell in the "likely to become endangered" (63%) or "in danger of extinction" (34%) categories. The BRT had substantial concerns about every VSP element. Most or all risk factors the BRT previously identified remained important concerns. The WLC-TRT estimated that close to 90% of this ESU's historical populations were extinct or nearly so, resulting in loss of much diversity and connectivity between populations. The 2005 BRT was concerned that populations that remained were small, and overall abundance for the ESU was low. The ESU had shown low productivity for many decades. The BRT was encouraged that unofficial reports for 2002 suggested a large increase in abundance in some (perhaps many) locations, but was unclear on the cause of the increase and whether it would be sustaining for multiple years.

2010

Ford *et al*. (2011) concluded that the vast majority (14 out of 17) chum populations remain extirpated or nearly so. The Grays River and Lower Gorge populations showed a sharp increase in 2002, but then declined back to relatively low abundance levels in the range of variation observed over the prior several decades. Chinook and coho populations in the Lower Columbia and Willamette showed similar increases in the early 2000's followed by declines, suggesting the increase in chum was related to ocean conditions. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Most tributaries are surveyed by foot, although chum salmon observations may be incidental to surveys focusing on Chinook or coho salmon. Standardized mark-recapture surveys have been undertaken and population estimates are available for the Grays River, Hamilton Creek, and the mainstem Columbia River. In many other tributaries, potential chum salmon habitat is monitored for the presence of spawners either through directed surveys or indirectly with multispecies surveys providing some coverage for most other populations (Chinook River, Elochoman River, Skamokawa Creek, Mill, Abernathy, and Germany Creeks and the Lewis River). Chum salmon are also enumerated at hatchery traps, tributary weirs, and dam fish passage facilities. WDFW and ODFW has expanded the location and number of salmon spawning surveys, providing some coverage for most populations.

ABUNDANCE AND PRODUCTIVITY

COASTAL RANGE MPG

GRAYS RIVER

Surveys for chum salmon are regularly conducted in the Grays River. Spawner abundances have exhibited a cyclical pattern, with peak levels over 10,000 fish in 2002. Abundances declined to a few thousand fish in 2006-2008, and then peaked 2010-2012 (Figure 78). The majority of the returning chum salmon have been naturally produced, 93.4% on average (2001-14) (Figure 81). The Grays River has maintained its position as a stronghold in the MPG and the ESU, with both positive short and long term trends.

OTHER COASTAL RANGE DIPS

Populations in this MPG other than the Grays River DIP exist at very low abundances and some may be functionally extinct. Adult chum are intermittently observed in very low numbers (<10) in most tributaries other than the Grays River or Big Creek. Returns of adult chum salmon to the Big Creek weir normally number in the tens of fish. In the past these fish were excluded from migrating above the weir, but more recently unmarked fish have been passed above the weir. Supplementation and reintroduction efforts using surplus hatchery broodstock are underway in a number of tributaries in this MPG and outmigrating fry have been observed.

CASCADE RANGE MPG

WASHOUGAL RIVER CHUM SALMON

The 2005 BRT report noted the discovery of two chum spawning aggragates in the mainstem Columbia River just upstream of the I-205 bridge in areas influenced by groundwater seeps. This spawning aggregation is demographically part of the Washougal River DIP and genetically similar to other populations in the Gorge MPG (Myers *et al.* 2006). Population abundance has fluctuated considerably, likely following changes in ocean conditions, with stronger returns in 2002-2004 and 2010-2012 (Figure 78). As with many of the other populations, Washougal River chum salmon experience highly variable return rates, approximately a 5-fold range in the last 15 years. The abundance trend has been stable and potentially slightly positive.

OTHER CASCADE RANGE CHUM SALMON DIPS

There are reports of chum salmon in a number of tributaries, although systematic surveys for chum are not undertaken. In November 2013, two adult chum salmon were observed at the North Fork Dam in the Clackamas River.[25] Chum salmon have also been collected at a number of hatcheries and weirs throughout this MPG, but only in very limited numbers (<10). While the absolute numbers of fish present in many populations are critically low, they may represent important reserves of genetic diversity. Finally, there have been recurring observations of early returning "summer" chum salmon in the Cowlitz River, primarily at the Cowlitz Salmon Hatchery trap.

GORGE MPG

LOWER GORGE CHUM SALMON

This population includes chum salmon returning to Hamilton, Hardy, and Duncan Creeks, as well as those returning to spawn in the Ives Island area of the mainstem Columbia River below Bonneville Dam. Other mainstem Columbia River spawning aggregations include Multnomah and Horsetail Creeks on the Oregon shoreline and in the St. Cloud area along the Washington shoreline. Recent abundances are, on average, somewhat improved since the last status review; however, ocean conditions are likely responsible for this increase (Figure 78). The overall trend since 2000 is

[25] Data provided by Garth Wyatt, Fish Biologist, PGE, 9 December 2013.

negative, with the recent peak in abundance (2010-2011) being considerably lower than the previous peak in 2002 (Table 47).

UPPER GORGE CHUM SALMON

In most years, a small number of chum salmon migrate past Bonneville Dam to the upper Gorge population area; recently (2010-14), Chum salmon adult counts have averaged 105.6±47.7 (SD) (Data from http://www.nwp.usace.army.mil/Missions/Environment/Fish/Counts.aspx accessed 4 March 2015). Spawning above Bonneville is thought to be very limited due to the loss of historical spawning areas now under the Bonneville Pool; however, for the first time chum fry were observed at the Bonneville Dam juvenile monitoring facility in 2010.

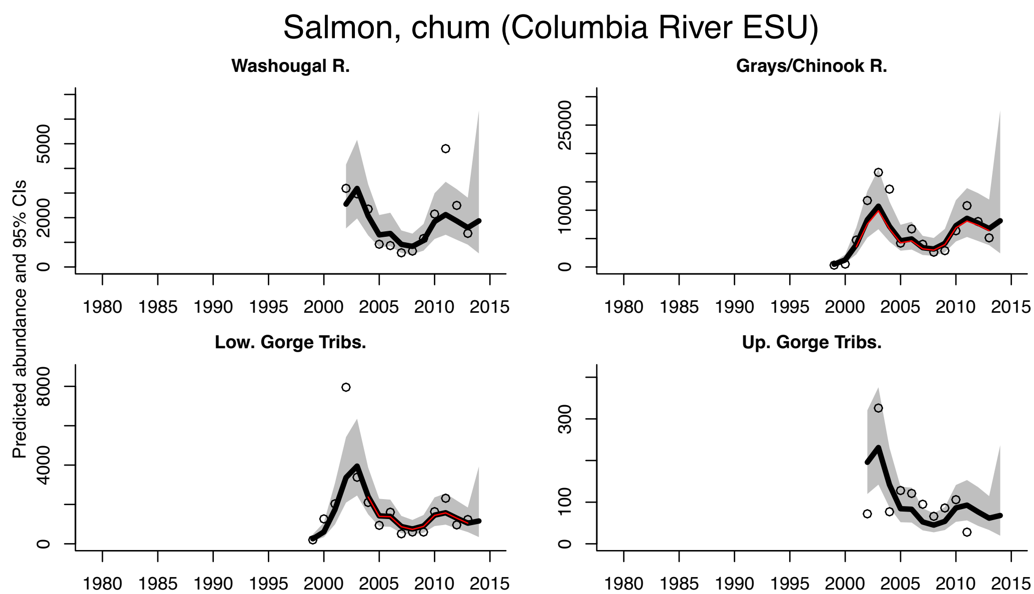

Figure 78 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates. Lower Gorge Tributaries include mainstem Columbia River spawning aggragates (ie. Ives Island, Horsetail Falls, etc.). Upper Gorge Tributaries is based on the Bonneville Dam count, although many Chum salmon counted upstream are know to have fallen back and spawned below Bonneville Dam.

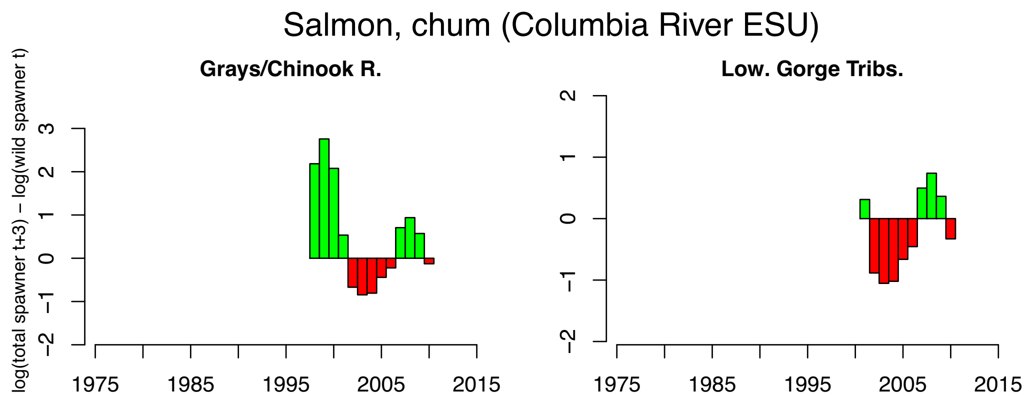

Figure 79 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year t - smoothed natural spawning abundance in year (t – 4).

Table 47 -- 5-year geometric mean of natural-origin spawner (NOS) counts. This is the raw total spawner count times the fraction NOS estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of NOS available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Washougal R.	Cascade	(53)	(117)	(1532)	(1079)	(1854)	(72)
Grays/Chinook R.	Coastal	116 (132)	297 (332)	4570 (4995)	3742 (3998)	7269 (7667)	94 (92)
Low. Gorge Tribs.	Gorge	121 (128)	202 (209)	1985 (2021)	1015 (1034)	1292 (1296)	27 (25)
Up. Gorge Tribs.	Gorge	(8)	(12)	(118)	(62)	(76)	(23)

Table 48 -- 15-year trends in log NOS spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1990 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Grays/Chinook R.	Coastal		0.1 (0.02, 0.18)
Low. Gorge Tribs.	Gorge		

HARVEST

Columbia River chum salmon were historically abundant and subject to substantial harvest until the 1950s (Johnson *et al.* 1997). In recent years there has been no directed harvest of Columbia River chum salmon. Data on the incidental harvest of chum salmon in lower Columbia River gillnet fisheries exist, but escapement data are inadequate to calculate exploitation rates. Incidental commercial landings have been approximately 100 fish per year since 1993 (except 275 fish in 2010), and all recreational fisheries have been closed since 1995. The incidental harvest rate on Columbia River chum salmon was estimated to be 1.9% in 2013 (ODFW and WDFW 2015) and 0.8% in 2014 (TAC 2015). Overall, the exploitation rate has been estimated at below 1% for the last five years.

SPATIAL STRUCTURE AND DIVERSITY

HATCHERIES

There are currently four hatchery programs in the Lower Columbia River releasing juvenile chum salmon: Grays River Hatchery, Big Creek Hatchery, Lewis River Hatchery, and Washougal Hatchery. The Lewis River Hatchery releases fish into the East Fork Lewis River and the Washougal Hatchery releases fish into Duncan Creek. The total annual production from these hatcheries has not exceeded 500,000 fish, with the majority being released as unmarked fish[26] during their first spring (Figure 80). Transfers of Grays River eggs to the Big Creek Hatchery are scheduled to be phased out as production of the Big Creek Hatchery stock is expanded (Homel 2014). Unmarked fish are allowed to spawn naturally above the Big Creek weir, and excess hatchery fish are released into nearby basins to help reestablish naturally-spawning populations. With the exception of the Grays River stock of fish raised at Big Creek Hatchery, all of the hatchery programs in this ESU use integrated stocks developed to supplement natural production.

[26] Fish are not externally marked, but all hatchery fish have otolith thermal marks. Limited number have coded-wire-tags (CWT) or parentage-based tags (PBT).

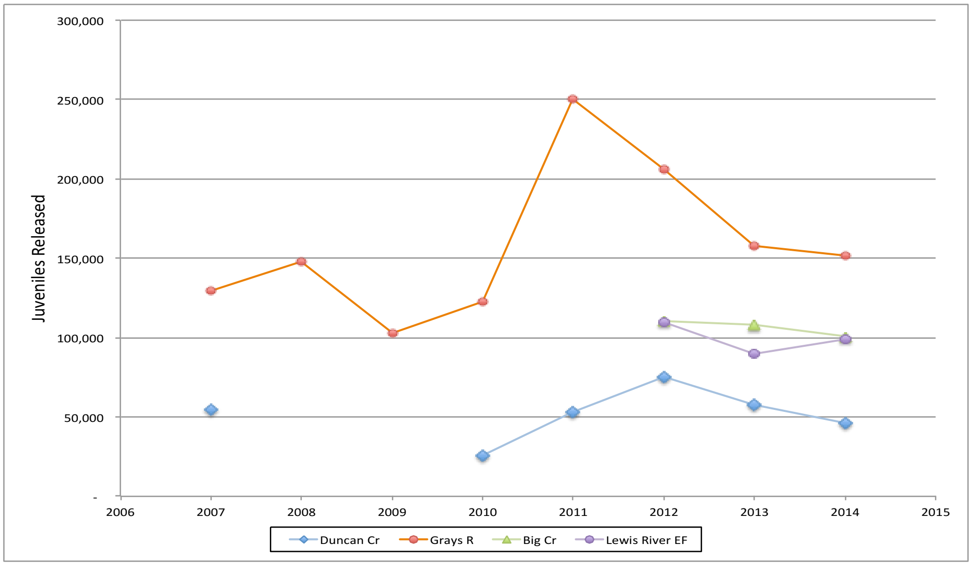

Figure 80. Releases of juvenile chum salmon (O. keta) from hatcheries in the Lower Columbia River. Duncan Creek fish originate from the Washougal Hatchery. Data from RMIS (http://www.rmpc.org/ accessed January 6, 2015) and Tom Hillson, WDFW, 12 December 2015.

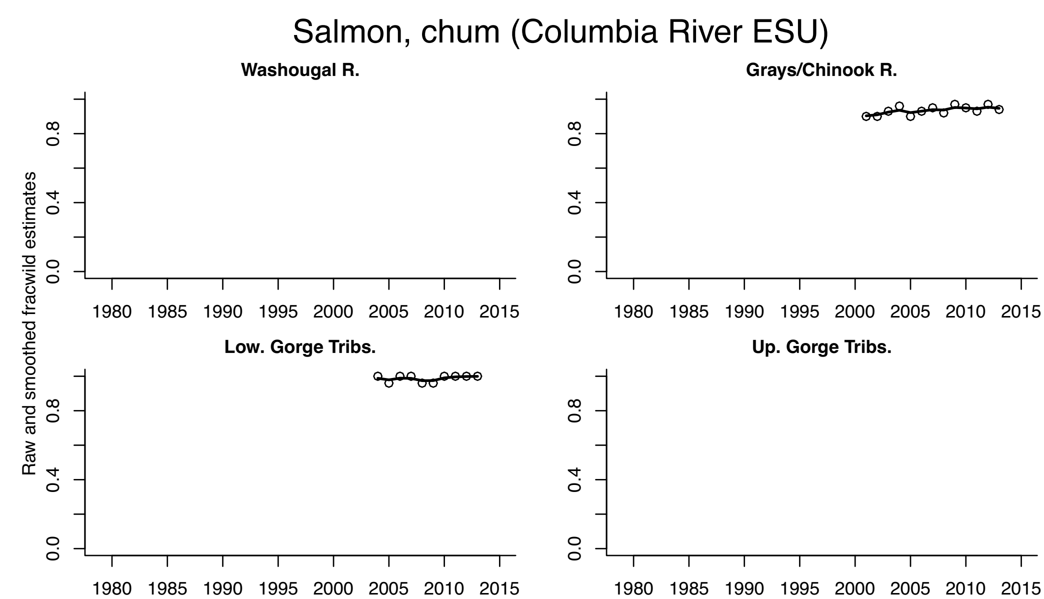

Figure 81 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 49 --5-year mean of fraction natural-origin spawner (NOS) (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Washougal R.					
Grays/Chinook R.			0.92	0.93	0.95
Low. Gorge Tribs.			1.00	0.98	1.00
Up. Gorge Tribs.					

SPATIAL STRUCTURE

There have been a number of large-scale efforts to improve accessibility, one of the primary metrics for spatial structure, in this ESU. On the Hood River, Powerdale Dam was removed in 2010 and while this dam previously provided for fish passage, removal of the dam is thought to eliminate passage delays and injuries. Condit Dam, on the White Salmon River, was removed in 2012 and this provided access to previously inaccessible habitat. Both of these dams were above Bonneville Dam, and at present there are few fish available (122 adults in 2014) to colonize these accessible habitats. Fish passage operations were begun on the Lewis River in 2012 reestablishing access to historically-occupied habitat above Swift Dam (RKm 77.1). Chum salmon are currently not included in the trap and haul program. It is more likely that smaller scale recovery actions throughout the ESU to remove or improve culverts, open dikes, or restore stream connectivity will provide more tangible benefits to chum salmon populations in the near future. For chum salmon, lateral access in the lower reaches of rivers may be more important than providing more access to upper watersheds.

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Overall, the status of most chum salmon populations is unchanged from the baseline VSP scores estimated in the recovery plan (Figure 82). A total of 3 of 17 populations are at or near their recovery viability goals, although under the recovery plan scenario these populations have very low recovery goals of 0. The remaining populations generally require a higher level of viability and most require substantial improvements to reach their viability goals. The status the Big Creek DIP is likely better than was initially described in the recovery plan, and the initiation of a supplementation program will likely improve the VSP status of this population. The Washougal River DIP has exhibited a positive abundance trend over the last ten years. The Grays River DIP has improved in status and may be at or near viable status (low risk). Lastly, population abundance declines in the Lower Gorge DIP since 2010 suggest that the previous 3.0 VSP score may be too high, although this population still remains one of the healthiest in the ESU. Even with the improvements observed during the last five years, the majority of DIPs in this ESU remain at a high or very high risk category and considerable progress remains to be made to achieve the recovery goals.

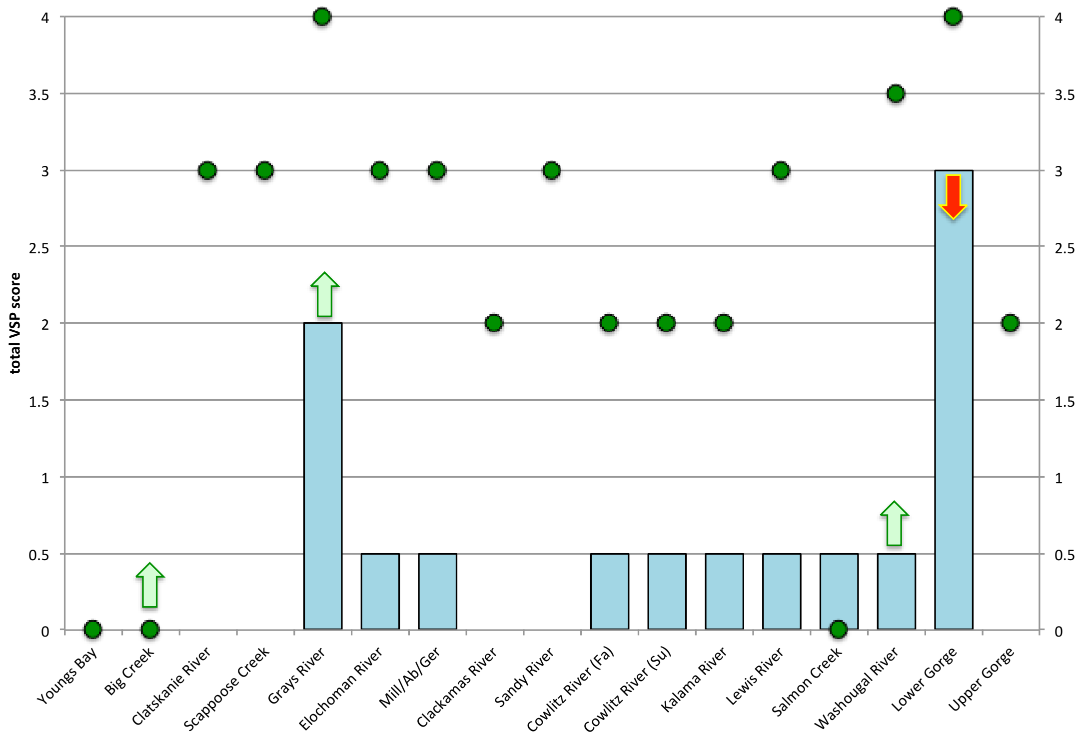

Figure 82. VSP status of demographically independent populations in the Lower Columbia River chum salmon ESU, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013), green circles indicate the recovery goals. Arrows indicate the general direction, but not the magnitude, of any VSP population score based on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany et al. 2006). A VSP score of 3.0 is represents a population with a 5% risk of extinction within a 100 year period.

UPDATED BIOLOGICAL RISK SUMMARY

The majority of the populations in this ESU are at high to very high risk, with very low abundances. These populations are at risk of extirpation due to demographic stochasticity and Allee effects. One population, Grays River, is at low risk, with spawner abundances in the thousands and demonstrating a recent positive trend. The Washougal River and Lower Gorge populations maintain moderate numbers of spawners and appear to be relatively stable. The life history of chum salmon is such that ocean conditions have a strong influence on the survival of emigrating juveniles. The potential prospect of poor ocean conditions for the near future may put further pressure on these chum salmon populations.

Freshwater habitat conditions may be negatively influencing spawning and early rearing success in some basins, and contributing to the overall low productivity of the ESU. Land development, especially in the low gradient reaches that chum salmon prefer, will continue to be a threat to most chum populations due to projected increases in the population of the greater Vancouver-Portland area and the Lower Columbia River overall (Metro 2014). The viability of this ESU is relatively unchanged since the last review and the modest improvements in some populations do not warrant a change in risk category, especially given the uncertainty regarding climatic effects in the near future. This ESU therefore remains at moderate to high risk.

UPPER WILLAMETTE CHINOOK SALMON ESU

BRIEF DESCRIPTION OF ESU

The ESU includes all naturally spawning populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon, as well as several artificial propagation programs (Figure 83).

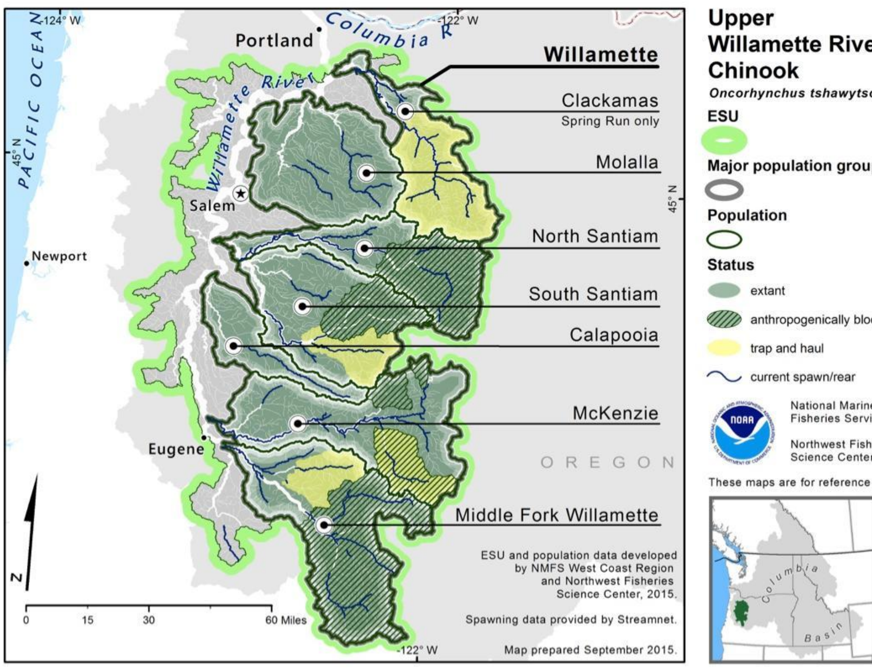

Figure 83 - Map of the Upper Willamette River Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

NMFS reviewed the status of the Upper Willamette River Chinook salmon ESU initially in 1998 (Myers 1998) and updated it that same year (NMFS 1998). In the 1998 update, the BRT noted several concerns for this ESU. The 1998 BRT was concerned about the few remaining populations of spring-run Chinook salmon in the Upper Willamette River ESU, and the high proportion of hatchery fish in the remaining runs. The 1998 BRT noted with concern that the Oregon Department of Fish and Wildlife (ODFW) was able to identify only one remaining naturally reproducing population in this ESU, the spring-run Chinook salmon in the McKenzie River. The 1998 BRT was concerned about severe declines in short-term abundance that occurred throughout the ESU, and that the McKenzie River population had declined precipitously, indicating that it may not be self-sustaining. The 1998 BRT also noted that the potential for interactions between native spring-run and introduced fall-run Chinook salmon had increased relative to historical times due to fall-run Chinook salmon hatchery programs and the laddering of Willamette Falls. The 1998 BRT partially attributed the declines in spring-run Chinook salmon in the Upper Willamette River ESU to the extensive habitat blockages caused by dam construction. A majority of the 1998 BRT concluded that the Upper Willamette River Chinook salmon ESU was likely to become endangered in the foreseeable future. A minority of 1998 BRT members felt that Chinook salmon in this ESU were not presently in danger of extinction, nor were they likely to become so in the foreseeable future.

The 2005 BRT considered updated abundance information, habitat accessibility analyses and the results of preliminary WLC-TRT analyses. These analyses supported previous BRT conclusions that the majority of populations in the ESU were likely extirpated or nearly so and that excessive numbers of hatchery fish and loss of access to historical habitat were important risk factors. The McKenzie River population was the only population identified as potentially self-sustaining and increases in abundance were noted for this population in the most recent returns available at the time (2000 and 2001). However, the BRT was concerned about the long-term potential for this population. The majority (70%) of the 2005 BRT votes fell in the “likely to become endangered” category, with a minority in the “in danger of extinction” and the “not likely to become endangered categories”.

2010

Ford *et al*. (2011) noted that two related status evaluations of UW Chinook had been conducted since the prior BRT status update in 2005. Both evaluations were based on the WLC-TRT viability criteria and both concluded that the ESU was at very high risk of extinction. Of the seven historical populations in the ESU, five were considered at very high risk. The remaining two (Clackamas and McKenzie) were considered to be at moderate to low risk. New data collected since the last BRT report verified the high fraction of hatchery origin fish in all of the populations in the ESU, with even the Clackamas and McKenzie having hatchery fractions above WLC-TRT viability thresholds. The new data reviewed in 2010 also highlighted the substantial risks associated with pre-spawning mortality. Although recovery plans were targeting key limiting factors for future actions, in 2010 there had been no significant on-the-ground-actions since the last BRT report to resolve the lack of access to historical habitat above dams nor had there been substantial actions removing hatchery fish from the spawning grounds. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Comprehensive spawner surveys (redds and carcasses) have been conducted in the North Santiam, South Santiam, McKenzie, and Middle Fork Willamette Rivers by ODFW. Direct adult counts are also made at Willamette Falls, Bennett Dam and Minto Fish Facility (North Santiam), Foster Fish Facility (South Santiam), Leaburg and Cougar dams and the McKenzie Hatchery (McKenzie River), Fall Creek Dam and Dexter Fish Facility (Middle Fork Willamette River). Intermittent spawner surveys have been conducted in the Molalla and Calapooia Rivers. Carcasses are assessed for origin (hatchery/natural) based on external marks and otoliths marks, and females are assessed for the proportion of unspawned eggs.

Genetic pedigree studies of adults returning to tributary dams in the Upper Willamette have been ongoing at Detroit Dam (North Santiam River), Foster Dam (South Santiam River), and Cougar Dam (McKenzie River) (Banks *et al.* 2014a). These studies provide information on the productivity of adults transported above impassable dams, and are critical in evaluating the success of juvenile fish passage systems.

ABUNDANCE AND PRODUCTIVITY

WILLAMETTE FALLS

Chinook salmon counts at Willamette Falls have been undertaken since 1946, when 53,000 Chinook salmon were counted; however, not until 2002 with the return of nearly 100% marked hatchery-reared fish was it possible to inventory naturally-produced fish with any accuracy. Fish returning in 2002 benefitted from very good ocean conditions and the calculated trend since then (nearly -10% annually) is influenced by that peak; in any event, the last five years (2010-2014) have also seen a downward trend in natural origin adult returns, with an overall geometric mean of 9,269 fish (Figure 84, Table 50). In recent years counts of spring-run Chinook salmon at Willamette Falls have been influenced by pinniped predation at the base of the falls. For the 2014 run year, an estimated 453 (±73) unmarked Chinook salmon were consumed primarily by California sea lions and less frequently by Stellar sea lions and Pacific harbor seals (Wright *et al.* 2014)

CLACKAMAS RIVER CHINOOK SALMON

Returning spring-run Chinook salmon are enumerated at North Fork Dam and outmigrating juveniles are collected and counted at River Mill Dam. As with other data series in the Willamette Basin, accurate abundance estimates for natural-origin adults were not possible until 2002, with the return of mass-marked hatchery-origin juveniles. The recent 5-year trend is relatively stable although the abundance is depressed (Figure 84). Portland General Electric (PGE), the operators of the dam complex, have recently installed new juvenile bypass systems at River Mill and North Fork dams, which may dramatically improve the collection efficiency and survival of outmigrating juveniles, especially Chinook salmon subyearlings. While the 2014 return of Chinook salmon, 983 fish, was the lowest since the last review, there is some expectation that the benefits of improved juvenile passage will be detected in the next few years.

MOLALLA RIVER CHINOOK SALMON

Chinook salmon surveys have been carried out intermittently in recent years. Surveys undertaken in 2011 and 2012 found a few adult spring-run Chinook salmon, the majority of which were marked hatchery-origin fish. Additionally, juvenile Chinook salmon were only observed in the North Fork and mainstem Molalla River and in very limited numbers (Bio-Surveys LLC 2012). For the 2012 return year, Jepson *et al.* (2013) estimated that the escapement of marked hatchery-origin and unmarked (presumptive NOR) fish to the Molalla River was 456 fish (95% confidence interval 171-1,315 fish) and 112 (43-285), respectively, by expanding the return of radio-tagged fish. In 2013, the estimated escapement for marked and unmarked fish was 92 (21-502) and 100 (14-537), respectively (Jepson et al. 2014). In 2014, none of the 76 unclipped Chinook salmon radio tagged at Willamette Falls returned to the Molalla River, while only 2 of the 224, clipped radio tagged fish were detected (Jepson *et al.* 2015). An expansion of the radio-tag detections would suggests total escapement to the Molalla River at only 211 fish in 2014. Similarly, a 2014 survey of the Pudding River found low numbers of juvenile Chinook salmon in Abiqua Creek (Bio-Surveys LLC 2014).

NORTH SANTIAM RIVER CHINOOK SALMON

Adult NOR returns to the North Santiam River, as measured at Bennett Dam and through redd and carcass surveys, have exhibited an increase in abundance in contrast to many of the other populations in the ESU and the combined count at Willamette Falls (Figure 84). This may be related to improved fish passage at Bennett Dam, resulting in a decrease in subsequent pre-spawning mortality, or it may be related to temperature-control operations at Detroit Dam that have resulted in a more "normal" incubation temperature regime for Chinook salmon. Estimates of NORs at Bennett Dam from 2001-2005 ranged from 217 to 721, geometric mean of 514. Furthermore, of those fish that passed Bennett Dam from 2001-2005 some 63.2% were estimated to have died prior to spawning. The current 5-year geometric mean of spring-run Chinook salmon ascending Bennett Dam is 1372 (2010-2014), and the observed prespawning mortality during this period was only 30.5% (Table 50)[27]. Spawner abundance, based on redd count, is noticeably less than the Bennett Dam counts, 412 (2010-2014)[28], but exhibits a similar recent positive trend. Genetic analysis of returning adults suggests that there is some contribution to escapement by the progeny of hatchery-origin spawners transported above Detroit Dam. Presently, natural-origin fish that reach the fish handling facilities at Minto are transported above the fish barrier to spawn in the North Santiam reach between Minto and Big Cliff Dam. While this "sanctuary" reach is solely populated with unmarked adult Chinook salmon, temperature and dissolved gas conditions may contribute to elevated prespawning mortality levels.

SOUTH SANTIAM RIVER CHINOOK SALMON

Spring-run Chinook salmon adults returning to the South Santiam River are monitored via redd counts and carcass recoveries in the mainstem South Santiam. Carcass recoveries are used to estimate the proportion of NOR and HOR spawners. In addition, direct counts of returning adults are made at the Foster fish collection facility at Foster Dam, where only NORs are passed above the dam. Foster Dam counts may be biased by conditions at the adult trap below Foster Dam, because not all fish produced upstream of the dam are attracted to the trap. Additionally, some of the NORS that enter the trap may be the offspring of spawners from reaches below the dam.

For the available Foster Dam time series (2007-2014) the abundance of NOR spawners has exhibited a positive trend, although not significantly (due in part to the limited number of years) and ocean conditions during the initial years of the trend may have biased the trend; however, given the overall negative NOR abundance trend at Willamette Falls the South Santiam should be viewed in a more

[27] Table data reflects Bennett Dam counts to 2013.
[28] Differences between the Bennett Dam counts and redd-based spawner estimates suggest that prespawning mortality counts and redd counts and expansions contain considerable uncertainty.

positive light. Prespawning mortality below and above Foster Dam averages 26.3%±5.4% and 33.3%±11.3%, respectively. Above Foster PSM levels may be affected by past adult trap and haul handling protocols. Geometric mean abundance for natural-origin adults in the South Santiam River from 2010-2014 was 575. In addition, it appears that there is a very small number of Chinook salmon in Green Peter Reservoir that exhibit an adfluvial life history (Romer & Monzyk 2014). There fish are most likely the descendants of hatchery-origin fish released in the reservoir over the course of several years. Some juveniles may be able to migrate downstream to Foster Reservoir, although the contribution to the population is likely negligible. While the presence of these fish confirms the continued suitability of the Middle Santiam River above Green Peter for spawning and rearing, adaptation to the adfluvial life history may impact the fitness of the anadromous portion of the population.

It appears that juvenile passage through Foster Dam is sufficiently high to sustain a naturally-spawning aggregation above the Dam, although total abundance is still quite low. Genetic analysis indicates that the replacement rates for the 2007 and 2008 broodyears were 0.96 and 1.16, respectively (O'Malley *et al.* 2014). Efforts are currently underway to improve both adult collection and juvenile downstream passage at Foster Dam. The USACE complete a new adult collection facility at Foster Dam to reduce handling-related injuries and provide adequate holding facilities for adults before release above the dam. Operational and structural modifications to Foster Dam to improve juvenile downstream passage are being studied presently, although it is unclear what form these improvements will take or when they will be accomplished.

CALAPOOIA RIVER CHINOOK SALMON

There has been limited monitoring of spring-run Chinook salmon in the Calapooia River basin, in part due to the low numbers of adults returning to the basin. Supplementation efforts have been terminated, large-scale releases were last made in 1997, although small numbers of fry (<50mm) were released through 2008. None of the fish that were radio-tagged at Willamette Falls in 2012-2014 were detected entering the Calapooia (Jepson et al 2013, 2014, 2015). A few adult Chinook salmon were observed in snorkel surveys in 2012, but it is unclear if they successfully spawned. Based on the limited information available, it would appear the Calapooia River Chinook salmon population is at a critically low level, if not functionally extirpated.

MCKENZIE RIVER CHINOOK SALMON

The status of spring-run Chinook salmon in the McKenzie River is monitored through both dam counts at Leaburg and Cougar dams, and through extensive spawner surveys (redd and carcass counts) throughout the basin. Genetic pedigree analysis of transported adults provides further information on the productivity of stream reaches above Cougar Dam. Numerous long-term abundance and life-history data sets exist for this population. Prior to the initiation of mass-marking for hatchery releases, hatchery contribution to spawning abundance was estimated through scale analysis, so it is possible to estimate NOR abundance prior to the 2002 return year.

Overall, McKenzie River spring-run Chinook salmon natural origin abundance has declined to levels not seen since the time of listing. This decline has occurred despite the restoration of access to spawning habitat in the South Fork McKenzie River above Cougar Dam through a trap and haul program. Genetic pedigree based estimates of cohort replacement rate for the 2007 and 2008 broodyears from hatchery adults released above the dam were both below replacement, 0.41 and 0.31, respectively (Banks *et al.* 2014a). Juvenile tagging studies suggest that total survival through Cougar Reservoir and Dam project has been poor (Beeman *et al.* 2013). While the effort to restore access to spawning habitat above Cougar Dam has resulted in the natural production of juveniles and returning adults, at the current levels for juvenile downstream passage and adult return there appears to be little net improvement in productivity.

Overall, redd counts for the entire McKenzie River have declined over the last five years, suggesting a more systematic limiting factor. Both short-term and long-term trends for the entire population are negative (Table 50 and Table 51).

MIDDLE FORK WILLAMETTE RIVER AND FALL CREEK CHINOOK SALMON

Chinook salmon in the Middle Fork Willamette River are monitored through redd and carcass surveys throughout much of the basin. In addition, fish are enumerated at both the Dexter Trap and at the Fall Creek trap below Fall Creek Dam. Presently, unmarked fish are transported above Fall Creek Dam. From 2006-2014, the pHOS for fish transported above Fall Creek Dam has averaged 4.6% (±1.5%), while predominately marked hatchery fish are transported above Dexter Dam to the North Fork Middle Fork Willamette River and Hills Creek (above Hills Creek Dam). Fish transported above Dexter Dam are part of an experimental program to assess the potential for a sustained trap and haul process around the dams[29]. Although the transported hatchery-origin adults successfully reproduce, in the absence of adequate downstream juvenile fish passage facilities it is unlikely that this program currently provides any substantial direct benefit to population abundance or productivity. Alternatively, the progeny of fish passed above Fall Creek Dam have a much higher likelihood of successful downstream passage via the complete drawdown of Fall Creek Reservoir. Based on returns to Fall Creek Dam, adult-to-adult return rates[30] have averaged 0.97 from 2010-2014). With the exception of spawning reaches above Fall Creek Dam, the remainder of the currently accessible portion of the Middle Willamette Basin, below Dexter Dam and Fall Creek Dam, is subject to conditions that result in a very high prespawning mortality and very poor incubation and juvenile survival. NOR spawners above Fall Creek averaged 138±40 fish from 2002-2014, with a slightly positive long-term trend. Estimates of prespawning mortality can be quite high in some years for the fish transported above Fall Creek Dam[31]. Of the hatchery-origin adults transported above Dexter Dam, prespawning mortalities have been high for fish transported to Hills Creek above Hills Creek Dam (49.3% 2012-14) compared to the North Fork Middle Fork Willamette River (39.0%, 2012-2014). Longer transportation times to Hills Creek are thought to be partially responsible for these differences (Naughton *et al.* 2014).

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[29] As a secondary benefit, the progeny of transported fish provide forage for Bull Trout.
[30] Adult to adult rates calculated as NOR adults returning to Fall Creek Dam divided by the average number of adults (NOR and HOR) passed above Fall Creek Dam four and five years previously.
[31] Prespawning mortality is estimated from recovered carcasses and may be biased depending on the number and timing of surveys, the number of carcasses recovered, and the seasonal river conditions.

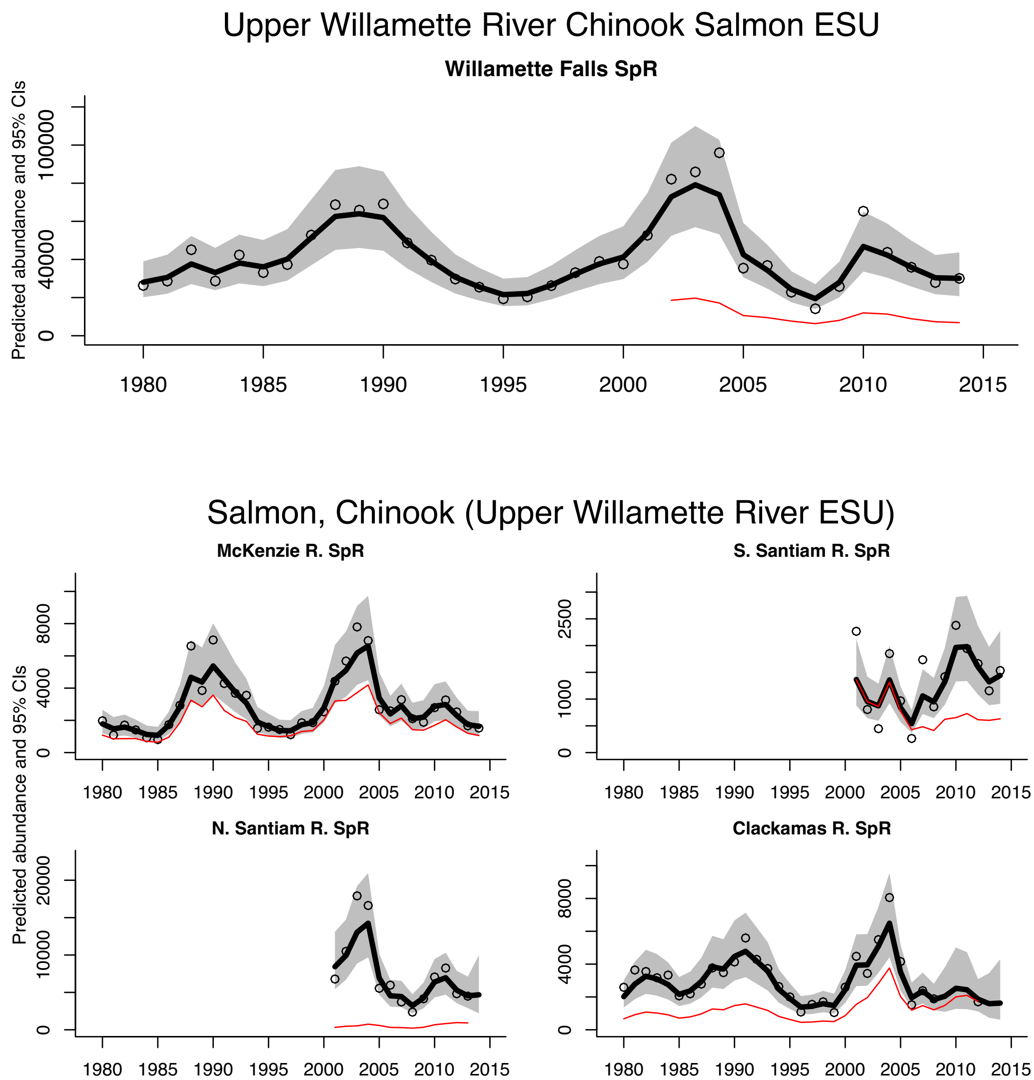
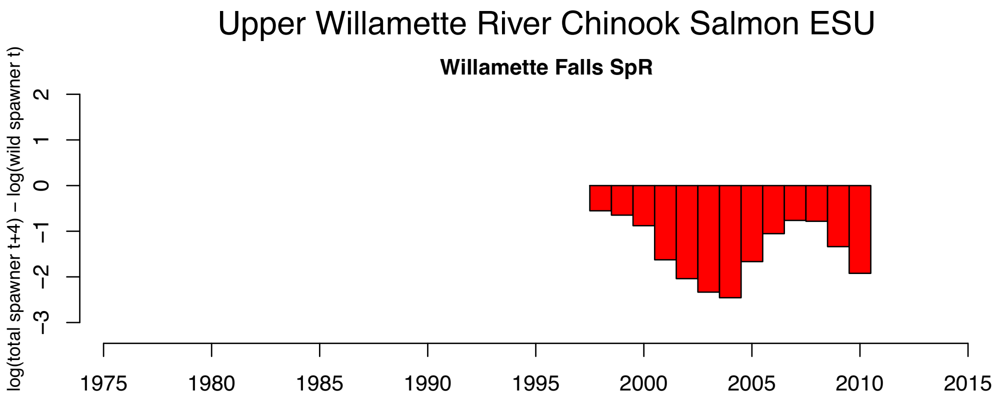


Figure 84. Smoothed trend in estimated total (thick black line) and natural (thin red line) Willamette Falls counts and population spawning abundance. Points show the annual raw spawning abundance estimates. Clackamas River data reflects counts at North Fork Dam. North Santiam River data reflect counts at Upper and Lower Bennett Dam.



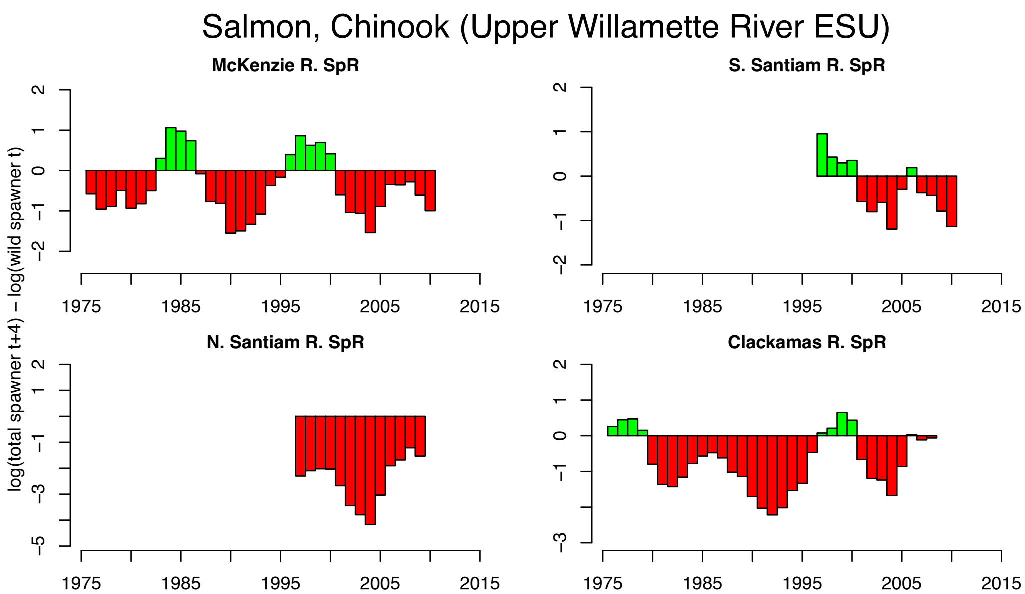


Figure 85. Trends in Willamette Falls counts and population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).

Table 50 -- 5-year geometric mean of raw natural-origin spawner (NOS) counts. This is the raw total spawner count times the fraction NOS estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of NOS available. North Santiam River data reflect counts at Upper and Lower Bennett Dam to 2013. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Willamette Falls SpR	Willamette	(39891)	(26608)	20900 (66906)	7567 (25547)	9269 (38630)	22 (51)

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Clackamas R. SpR	Willamette	1307 (3961)	472 (1430)	2063 (4460)	1381 (2308)		
McKenzie R. SpR	Willamette	2134 (3583)	1118 (1539)	3241 (5100)	1793 (2457)	1446 (2254)	-19 (-8)
N. Santiam R. SpR	Willamette			408 (12064)	290 (4136)	852 (5963)	194 (44)
S. Santiam R. SpR	Willamette			1108 (1108)	450 (883)	575 (1686)	28 (91)

Table 51 -- 15-year trends in log natural-origin spawner (NOS) abundance computed from a linear regression applied to the smoothed NOS log abundance estimate. Only populations with at least 4 NOS estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Willamette Falls WR	Willamette		-0.04 (-0.06, -0.01)

Population	MPG	1990-2005	1999-2014
McKenzie R. SpR	Willamette	0.03 (-0.02, 0.09)	-0.05 (-0.09, -0.01)
S. Santiam R. SpR	Willamette		-0.04 (-0.07, 0)
N. Santiam R. SpR	Willamette		0.08 (0.03, 0.14)
Clackamas R. SpR	Willamette	0.05 (-0.02, 0.13)	

HARVEST

Upper Willamette river spring Chinook are taken in ocean fisheries primarily in Canada and Alaska. They are also taken in lower mainstem Columbia River commercial gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and the Willamette River. These fisheries are directed at hatchery production, but historically could not discriminate between natural and hatchery fish. In the late 1990s ODFW began mass-marking the hatchery production, and recreational fisheries within the Willamette River switched over to retention of only hatchery fish, with mandatory release of unmarked fish. Overall exploitation rates reflect this change in fisheries dropping from the 50%-60% range in the 1980s and early 1990s to around 30% since 2000 (Figure 86), with difference observed in both ocean (Figure 87) and freshwater fisheries (Figure 88). Hooking mortalities are generally estimated at 10%, although river temperatures likely influence this rate. Illegal take of unmarked fish is thought to be low (Figure 86).

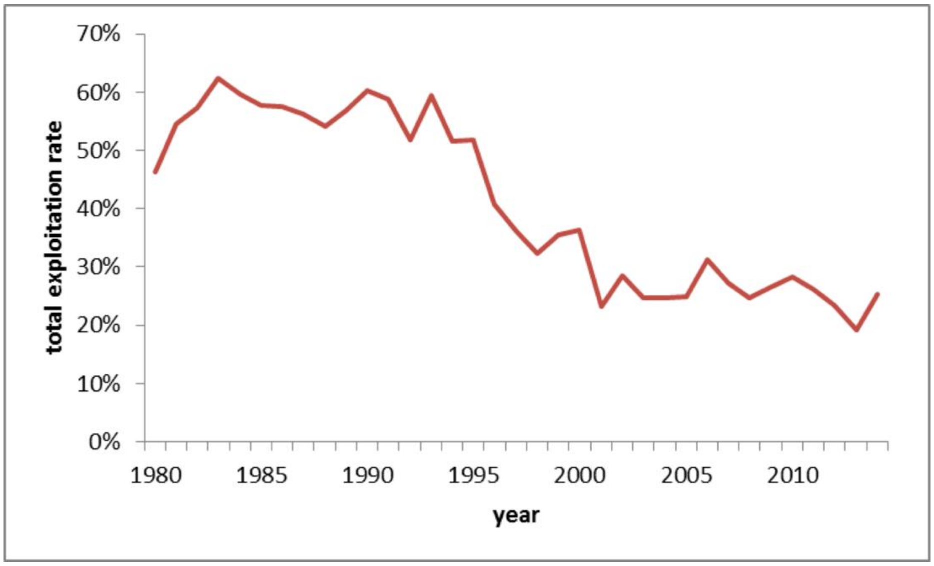


Figure 86. Total exploitation rates on Willamette River Spring Chinook. Data from CTC model calibration 1503 for ocean impacts, from TAC (2010) for inriver impacts from 1980-1997, Chris Kern, ODFW, personal communication for 1998-2009, and Jeff Wisler, ODFW personal communication for 2010-2014.

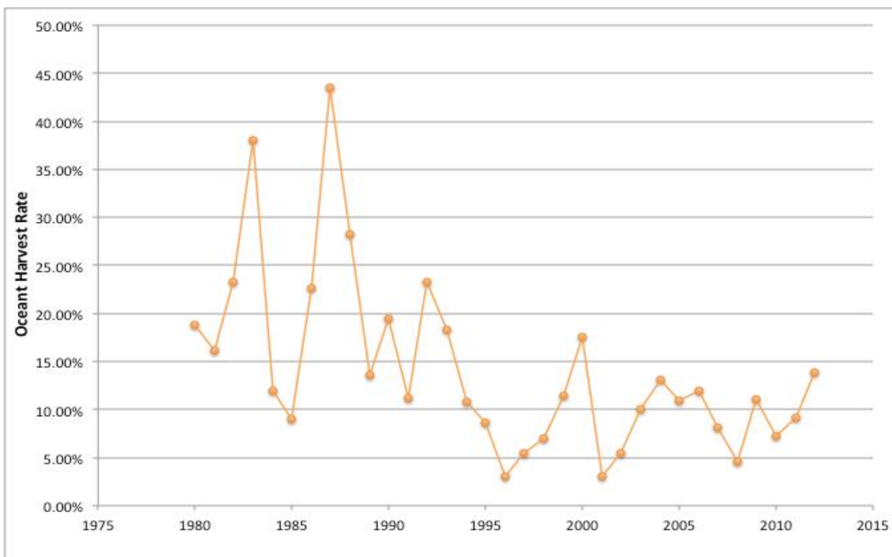


Figure 87. Ocean harvest rates for Upper Willamette River spring-run Chinook salmon based on coded-wire tag recoveries (PSC CTC 2014). Given the non-selective nature of ocean fisheries, harvest rates for hatchery and presumed naturally-produced fish is assumed to be comparable.

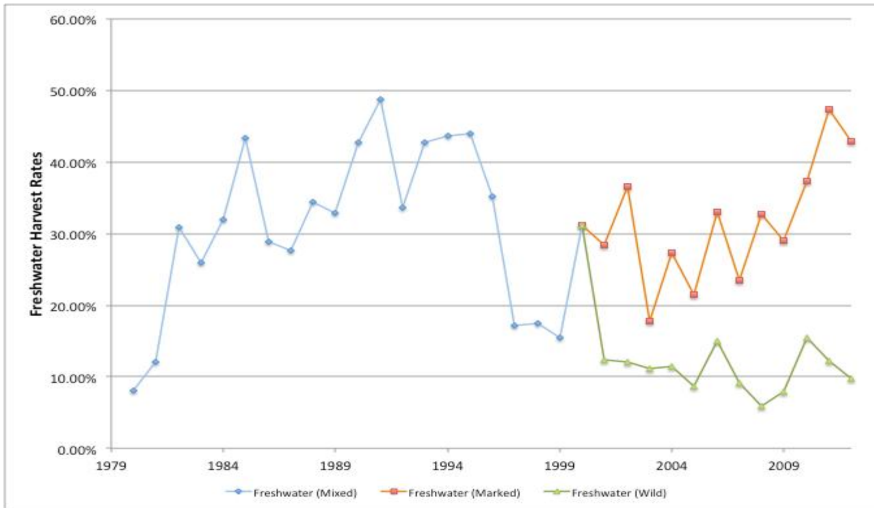


Figure 88. Freshwater harvest rates for marked (hatchery) and unmarked (naturally-produced) Upper Willamette River spring-run Chinook salmon (ODFW 2014, PSC CTC 2014). Harvest rates for both marked and unmarked fish prior to the initiation of selective fisheries in 2001 are thought to be similar. Harvest rates for unmarked fish from 2001-2012 are based on catch and release mortalities using encounter rates.

SPATIAL STRUCTURE AND DIVERSITY

Hatchery production has remained relatively stable since the initial status review (Myers *et al.* 1998). In general, production levels are based on mitigation agreements related to the construction of dams in the Willamette River Basin (Figure 89). There have been a number of operational changes at hatcheries. Mass marking of hatchery-origin Chinook salmon began in 1997, with all returning adults being marked by 2002. Off-station releases within some basins have been curtailed in an effort to limit natural spawning by hatchery-origin fish. Releases of juvenile Chinook salmon into the Coast Fork, a West side tributary that does not support a Chinook salmon population, have been made in an effort to maintain a harvestable hatchery return, but reduce hatchery x natural adult interaction on the natural spawning grounds in Eastside tributaries. Some of these returning adults have returned to their hatchery of origin rather than the Coast Fork release site. A review of hatchery operations by the Hatchery Science Review Group (HSRG) in 2009 identified a number of modifications to improve the status of Chinook salmon. Foremost was an increase in the proportion of naturally-produced fish into the hatchery broodstock; however, in many basins the abundance of naturally-produced Chinook salmon was critically low precluding their use as broodstock (HSRG 2009). Further the HSRG (2009) concluded:

> Options for improving the integrated hatchery programs in this ESU are limited due to the low number of natural-origin fish in the subbasin. This is generally the result of limited access to quality habitat cut off by flood control and hydropower development. Options for improving hatchery programs or achieving conservation goals are limited until this issue is addressed. Contribution to conservation was

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improved for one population by improving broodstock collection and reducing the size of its integrated harvest program. (pg 46)

Recent improvements at the Cougar (2010), Minto (2012) and Foster (2014) fish collection facilities offer the potential for collecting more hatchery origin adults and removing them from the natural-spawning component of the populations. Increased collection efficiency has been observed at the Cougar and Minto facilities, while the recently completed Foster facility appears to require further modifications[32]. Ultimately, these facilities should be able to reduce the pHOS in both the North and South Santiam populations. Plans are being developed for improvements in the facilities at Fall Creek and Dexter Dam.

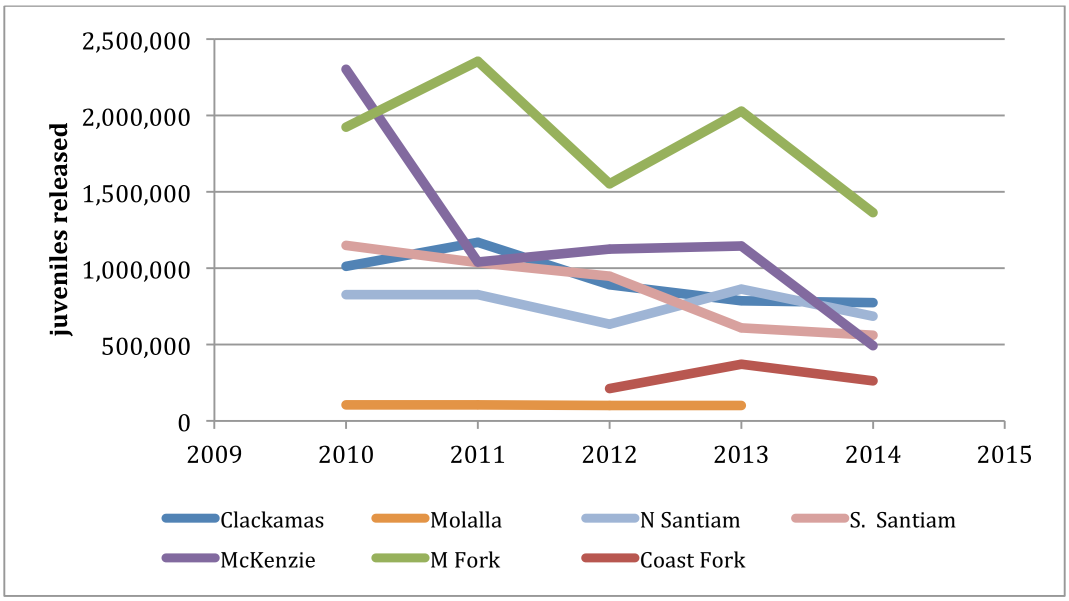

Figure 89. Hatchery releases of juvenile spring-run Chinook salmon into basins of the Upper Willamette Chinook salmon ESU from 2010 to 2014. Data from Regional Mark Processing Center (RMPC.org) accessed 14 December 2014.

[32] C. Sharpe, ODFW. Presentation to Willamette Science Review, 12 February 2015, Portland, Oregon.

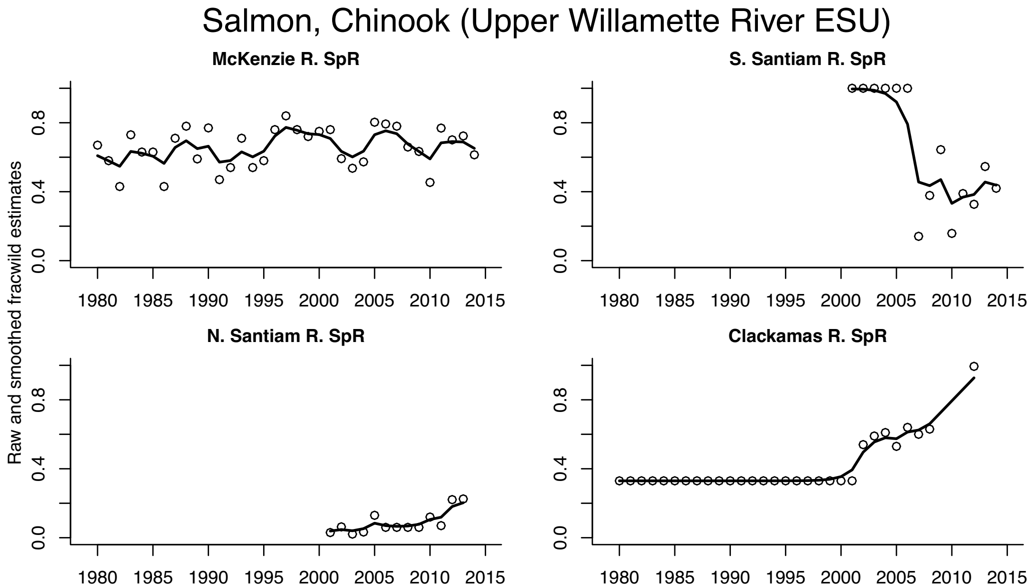

Figure 90 -- Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates. Data for the South Santiam River spring run only reflects fish passed above Foster Dam for 2000 to 2005.

Table 52 -- 5-year mean of fraction natural-origin spawner (NOS) (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range. Data for South Santaim only describes fish passed above Foster Dam.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Willamette Falls SpR			0.24	0.30	0.24

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
McKenzie R. SpR	0.61	0.73	0.64	0.73	0.65
S. Santiam R. SpR			1.00	0.63	0.37
N. Santiam R. SpR			0.04	0.07	0.16
Clackamas R. SpR	0.33	0.33	0.48	0.60	0.99

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Abundance levels for five of the seven DIPs in this ESU remain well below their recovery goals. Of these, the Calapooia River may be functionally extinct and the Molalla River remains critically low (although perhaps only marginally better than the 0 VSP score estimated in the Recovery Plan).

Abundances in the North and South Santiam rivers have risen since the last review, but still range only in the high hundreds of fish. Improvement in the status of the Middle Fork Willamette River relates solely to the return of natural adults to Fall Creek; however, the capacity of the Fall Creek basin alone is insufficient to achieve the recovery goals for the Middle Fork Willamette River DIP. The Fall Creek program also provides valuable information relevant to the use of reservoir draw downs as a method of juvenile downstream passage. The proportion of natural origin spawners improved in the North and South Santiam basins, but was still well below identified recovery goals. The presence of juvenile (subyearling) Chinook salmon in the Molalla River suggests that there is some limited natural production.

The Clackamas and McKenzie Rivers have previously been viewed as natural population strongholds, but have both experienced declines in abundance[33]. Overall, populations appear to be at either moderate or high risk, there has been likely little net change in the VSP score for the ESU since the last review, so the ESU remains at moderate risk.

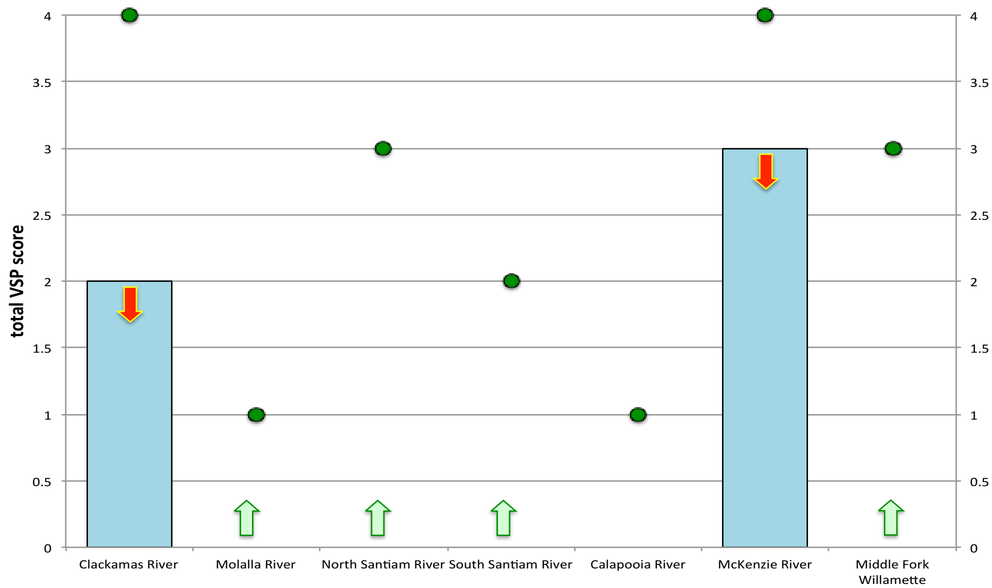

Figure 91. VSP status of demographically independent populations in the Upper Willamette River Chinook salmon ESU, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013), green circles indicate the recovery goals. Arrows indicate the general direction, but not the magnitude, of any VSP population scorebased on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany et al. 2006). A VSP score of 3.0 is represents a population with a 5% risk of extinction within a 100 year period.

[33] Spring-run Chinook salmon counts on the Clackamas River are taken at North Fork Dam, where presently only unmarked fish are passed above. A small percentage of these unmarked fish are of hatchery origin. While there is some spawning below the Dam, it is not clear if any progeny from the downstream redds contribute to escapement.

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UPDATED BIOLOGICAL RISK SUMMARY

In evaluating the status of Upper Willamette River spring-run Chinook salmon there are number of general considerations that affect some or all of the populations. In addition to the prespawning mortalities monitored in the specific population basins, there is a shortfall in abundance between Willamette Falls and East side tributary census points[34] due to prespawning mortality or spawning in the unsurveyed lower reaches of east or west-side tributaries (Jepson *et al.* 2013; Jepson *et al.* 2014) where spawning and incubation conditions are less well-suited to spring-run Chinook salmon. Radio tagging results from 2014 suggest that few fish strayed into west-side tributaries (no detections) and relatively fewer fish were unaccounted for between Willamette Falls and the tributaries, 12.9% of clipped fish and 5.3% of unclipped fish (Jepson *et al*. 2015). Access to historical spawning and rearing areas is restricted by large dams in the four historically most productive tributaries, and in the absence of effective passage programs will continue to be confined to more lowland reaches where land development, water temperatures, and water quality may be limiting. Prespawning mortality levels are generally high in the lower tributary reaches where water temperatures and fish densities[35] are generally the highest. Areas immediately downstream of high head dams may also be subject to high levels of total dissolved gas (TDG). While the relationship between TDG levels and mortality is related to a complex interaction of fish species, age, depth, and history of exposure (Beeman & Maule 2006), the relative risks are quite high in some reaches. For example, natural origin Chinook salmon and steelhead are passed above the barrier dam at the Minto fish facility into a short reach immediately below the Detroit/Big Cliff Dam complex. At certain times of the year, water spilled over Detroit and Big Cliff dams has the potential to produce high levels of TDG, which could affect a significant portion of the incubating embryos, in-stream juveniles, and adults in the basin, although the effect of this impact has not been quantified.

The apparent decline in the status of the McKenzie River DIP in the last 10 years is a source of concern given that this population was previously seen as a stronghold of natural production in the ESU. In contrast to most of the other populations in this ESU, McKenzie River Chinook salmon have access to much of their historical spawning habitat, although access to historically high quality habitat above Cougar Dam (South Fork McKenzie River) is still limited by poor downstream juvenile passage. Additionally, the installation of a temperature control structure in Cougar Dam in 2008 was thought to benefit downstream spawning and rearing success. Similarly, natural-origin returns to the Clackamas River have remained flat, despite adults having access to much of their historical spawning habitat. Although returning adults have access to most of the Calapooia and Molalla basin, habitat conditions are such that the productivity of these systems is very low. Natural-origin spawners in the Middle Fork Willamette River consisted solely of adults returning to Fall Creek. While these fish contribute to the DIP and ESU, at best the contribution will be minor. Finally, improvements were noted in the North and South Santiam DIPs. The increase in abundance in both DIPs was in contrast to the other DIPs and the counts at Willamette Falls. While spring-run Chinook salmon in the South Santiam DIP have access to some of their historical spawning habitat, natural

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[34] Census points include: dams, traps, index reaches, or radio-tracking antennae stations.

[35] Reaches downstream of fish hatcheries contain relatively large numbers of hatchery fish, which may also be more susceptible to prespawning mortality.

origin spawners in the North Santiam are still confined to below Detroit Dam[36] and subject to relatively high prespawning mortality rates.

Although there has likely been an overall decrease in the VSP status of the ESU since the last review (Figure 91), the magnitude of this change in not sufficient to suggest a change in risk category. Given current climatic conditions and the prospect of long-term climatic change, the inability of many populations to access historical headwater spawning and rearing areas may put this ESU at greater risk in the near future.

[36] Some hatchery-origin spawners are currently transported above Detroit Dam; however downstream juvenile survival through existing passage outlets is extremely low and likely would not achieve replacement.

UPPER WILLAMETTE STEELHEAD DPS

BRIEF DESCRIPTION OF ESU

The DPS includes all naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River (Figure 92). The DPS/ESU Boundaries Review Group (see DPS/ESU Boundaries section above) considered new genetic information relating to the relationship between the Clackamas River winter steelhead and steelhead native to the Lower Columbia and Upper Willamette River DPSs. The Review Group concluded that there was sufficient information available for considering reassigning the Clackamas River winter steelhead population to the Upper Willamette River steelhead DPS. In addition, late winter steelhead have been observed in the Willamette River, upstream of its confluence with the Calapooia River, to the McKenzie River and Fall Creek. It is unclear where these fish originated from and whether they constitute sustainable populations outside of the presumed historical boundaries. Further review is necessary before there can be any consideration of redefining the DPS; therefore, the present status evaluation is being conducted based on existing DPS boundaries (Figure 92).

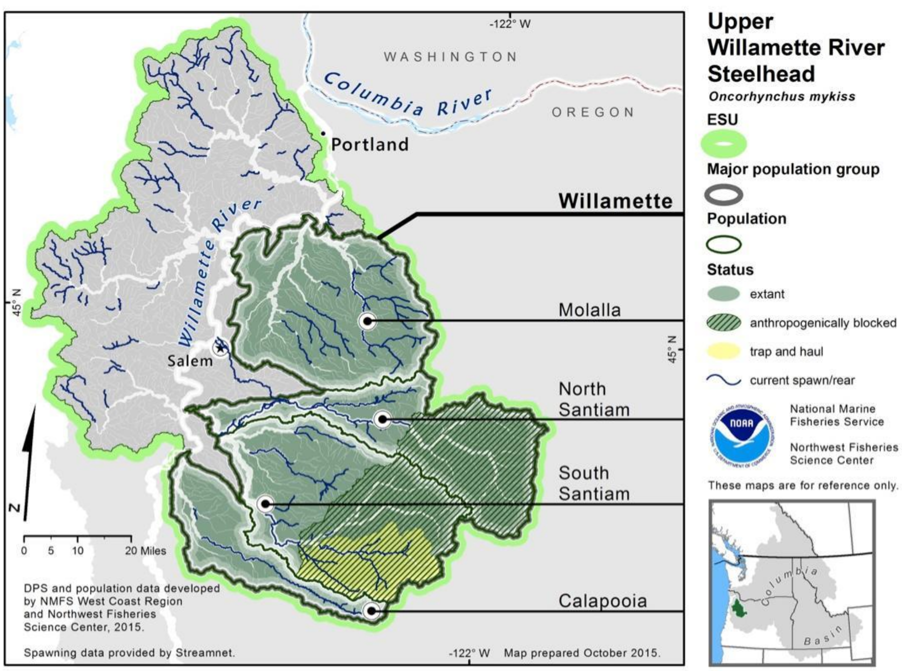

Figure 92 -- Map of the Willamette River winter steelhead DPS's spawning and rearing areas, illustrating populations and major population groups.

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SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

NMFS initially reviewed the status of the Upper Willamette River steelhead ESU in 1996 (Busby *et al*. 1996), with an update in 1999 (NMFS 1999). In the 1999 review, the BRT noted several concerns for this ESU, including relatively low abundance and steep declines since 1988. The previous BRT was also concerned about the potential negative interaction between non-native summer-run steelhead and native winter-run steelhead. The previous BRT considered the loss of access to historical spawning grounds because of dams to be a major risk factor. The 1999 BRT reached a unanimous decision that the Upper Willamette River steelhead ESU was at risk of becoming endangered in the foreseeable future.

In the 2005 status update (Good *et al*. 2005), a majority (over 71%) of the BRT votes for this ESU fell in the "likely to become endangered" category, with small minorities falling in the "in danger of extinction" and "not likely to become endangered" categories. The BRT did not identify any extreme risks for this ESU but found moderate risks in all the VSP categories. On a positive note, the 2005 BRT noted that after a decade in which overall abundance (Willamette Falls count) hovered around the lowest levels on record, adult returns for 2001 and 2002 were up significantly, on par with levels seen in the 1980s. Still, the total abundance was considered small for an entire ESU, resulting in a number of populations that were each at relatively low abundance.

2010

Ford *et al*. (2011) noted that since the 2005 BRT status update, Upper Willamette steelhead initially increased in abundance but subsequently declined to levels observed in the mid-1990s when the DPS was first listed. The DPS appeared to be at lower risk than the Upper Willamette Chinook ESU, but continued to demonstrate the overall low abundance pattern that was of concern during prior reviews. The elimination of winter run hatchery release in the basin reduced some risks, but non-native summer steelhead hatchery releases were still a concern. Human population expansion within the Willamette Basin constituted a significant risk factor for these populations. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

ABUNDANCE AND PRODUCTIVITY

Estimates of steelhead abundance for this DPS were based on redd counts in the North and South Santiam basins. Adult counts were also available from observations at Willamette Falls, Bennett Dam and the Minto Fish Facility (North Santiam River), and Foster Dam (South Santiam River). In addition, results from tracking studies of radio-tagged winter steelhead were expanded to estimate spawner abundances in specific DIPs. Steelhead arriving at Willamette Falls have also been sampled for genetic analysis to determine the relative proportions of native (late winter steelhead) and out-of-DPS (early winter, summer, or summer/winter hybrid steelhead) genotypes represented in the run.

WILLAMETTE FALLS

Winter steelhead counts at Willamette Falls provide a complete count of fish returning to the DPS. In the last 10 years, returns to Willamette Falls have averaged 5,828±98(SE) winter steelhead, of those an average of 3,832±109 returned after February 15th[37]. Of these fish, if one apportions the late-winter fish to the four eastside tributaries that historically supported late-winter steelhead based on the results of the radio-tagging work from 2012-2014 (Jepson *et al*. 2013, 2014, 2015), the 10-year average for returning adults would be an average 3,409 using all winter-run counts and 2,351 using the February 15th demarcation. Analysis of the radio tagging data suggests that the February 15th demarcation at Willamette Falls before or after February 15th does not predict whether winter steelhead are destined for east-side or west-side tributaries. Based on the three years of radio-tag data, an average of 59.3±3.1% (SE) of the winter-run steelhead ascending Willamette Falls enter the four steelhead DIP basins.

Trend analysis using the last 10 years of return data indicates 2.9% annual increase using the post-February 15th data and a slight 0.6% annual increase using the total winter count. Long-term abundance (1971 to present) is negative for both post-February 15th (-3.2%) and total winter-run counts (-3.5%), although the hatchery-origin winter steelhead are included in the counts from 1971 through the 1990s. In recent years counts of winter steelhead at Willamette Falls have been influenced by pinniped predation at the base of the falls. For the 2014 run year, an estimated 906 (±130) early and late winter steelhead were consumed primarily by California sea lions and less frequently by Stellar sea lions and Pacific harbor seals (Wright *et al.* 2014). For 2014, this represent 11-18% of the total winter steelhead run entering the Willamette River (Wright *et al.* 2014). Additionally, the inability to discriminate between early- and late-winter steelhead beyond an expansion of the radio tag work limits the precision of any estimates. In general, overall abundance for the Upper Willamette River winter steelhead DPS remains low with recent trends being stable.

MOLALLA RIVER

Population abundance estimates based on spawner (redd) surveys are only available for the Molalla and associated tributaries (Pudding River, Abiqua Creek) through 2006. These estimates relied on a proportional apportionment of winter-run steelhead counts at Willamette Falls based on index redd counts in the four winter-run steelhead populations. Recent estimates, based on the proportional migration of winter-run steelhead tagged at Willamette Falls (Jepson *et al*. 2013, Jepson *et al*. 2014), indicate that a significantly smaller portion of the steelhead arriving at Willamette Falls are destined for the Molalla River. Based on radio-tag detections and the total winter-run steelhead count at the Falls, the estimated escapement (95% CI) to the Molalla for 2012-2014 was 976 (660-1,406), 903 (651-1,223), and 757 (540-1,042), respectively. As indicated by the broad confidence intervals, these estimates give a only general indicator of steelhead abundance. Previous escapement estimates (1980 to 2006) had a geometric mean of 1237 ranging from 97 to 4658, long term trend show an annual 3.7% decline, although this decline is likely an overestimate due to the inclusion of hatchery

[37] February 15th marked the estimated demarcation between returning non-native early-winter steelhead and native late-winter steelhead.

fish in the early years. Estimated declines (Figure 94) in the Molalla River are based correlations with observed trends in the North and South Santiam Rivers. Given that the Molalla River has no major migrational bariers, limiting factors in the Molalla River are more likely related to habitat degradation. Abundance is likely relatively stable, but at a depressed level.

NORTH SANTIAM RIVER AND BENNETT DAM

Late-winter steelhead spawn throughout the North Santiam Basin except for reaches above the Big Cliff/Detroit Dam complex. Currently, the best measure of steelhead abundance is the count of returning winter-run adults to Upper and Lower Bennett Dam. Recent passage improvements at the dams and an upgraded video counting system have contributed to a higher level of certainty in adult estimates. While there are steelhead spawning below Bennett[38], it is likely that these dam counts approximate the population run size. The Bennett Dam counts may also approximate spawner counts, given that post-dam prespawning mortality is thought to be low for winter steelhead. Unfortunately, steelhead were not counted at Bennett Dam from 2006 to 2010, due to budget constraints. The most recent average count for unmarked (presumed native) winter steelhead (2010-1014) is only 1195 ± 194. Longer term trends 1999-2014 are negative, -5 ±3%. Radio-tagging studies (Jepson et al 2013, 2014, 2015) provided additional estimates of abundance that were similar to the Bennett Dam counts (Figure 93), with an average abundance of 1154.

SOUTH SANTIAM RIVER AND FOSTER DAM

Survey data (index redd counts) is available for a number of tributaries to the South Santiam River; in addition, live counts are available for winter steelhead transported above Foster Dam. Temporal differences in the index reaches surveyed and the conditions under which surveys were undertaken make the standardization of data among tributaries very difficult. For the Foster Dam time series, the most recent 5-year average (2010-2014) has been 304±34, with a negative trend in the abundance over those years (recognizing that the 2010 return reflected good ocean conditions). Longer time series are less meaningful, in that abundance estimates before 2009 were developed using different methodologies. Expanding the radio-tag tracking data (Jepson 2013, 2014, 2015) for 2012-2014 yields South Santiam abundances of 1,226 (875-1,693), 1,134 (853-1,474), and 1,312 (1,010-1,758), respectively. In addition to steelhead spawning in the mainstem South Santiam River, annual spawning surveys of tributaries below Foster Dam (Thomas, Crabtree, and Wiley creeks) indicate the consistent presence of low numbers of spawning steelhead.

[38] Personal Communication: C. Sharpe, ODFW, Sept. 2, 2015.

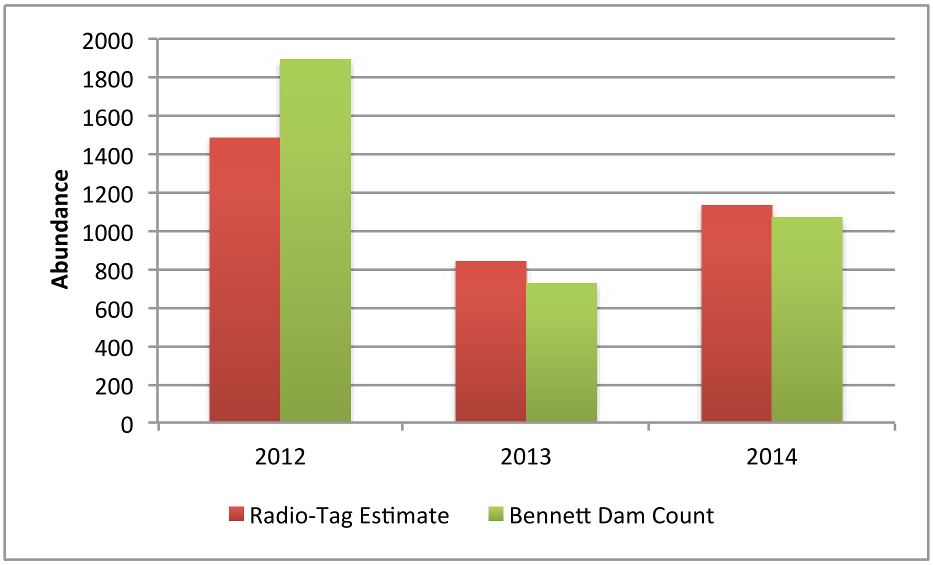
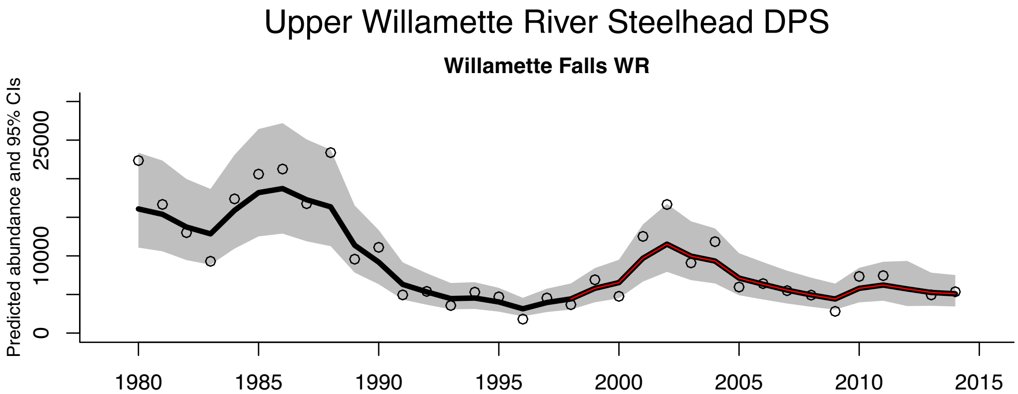


Figure 93. Comparison of North Santiam winter steelhead abundance using radio-tag mark-recapture expansion (Jepson et al. 2013,2014,2015) and Bennett Dam counts.

CALAPOOIA RIVER

There is a nearly complete and consistent time series for index reach redd counts in the Calapooia River dating back to 1985. While there is not an expansion available from index reach to population spawner abundance, the trend in redds/mile is generally negative, although this is due in part to the time series beginning with at the time of good ocean conditions. The redds/mile trend generally reflects good ocean conditions in the late 1980s and early 2000s, in addition to a period of poor ocean conditions in the mid-1990s. Abundance is thought to be rather low, population estimates (95% CI) based on radio tagged winter steelhead (Jepson et al 2013, 2014, 2015) for 2012, 2013, and 2014 are 127 (43-366), 204 (99-408), and 126 (54-289) respectively. These numbers would suggest that abundances have been fairly stable, albeit at a depressed level.



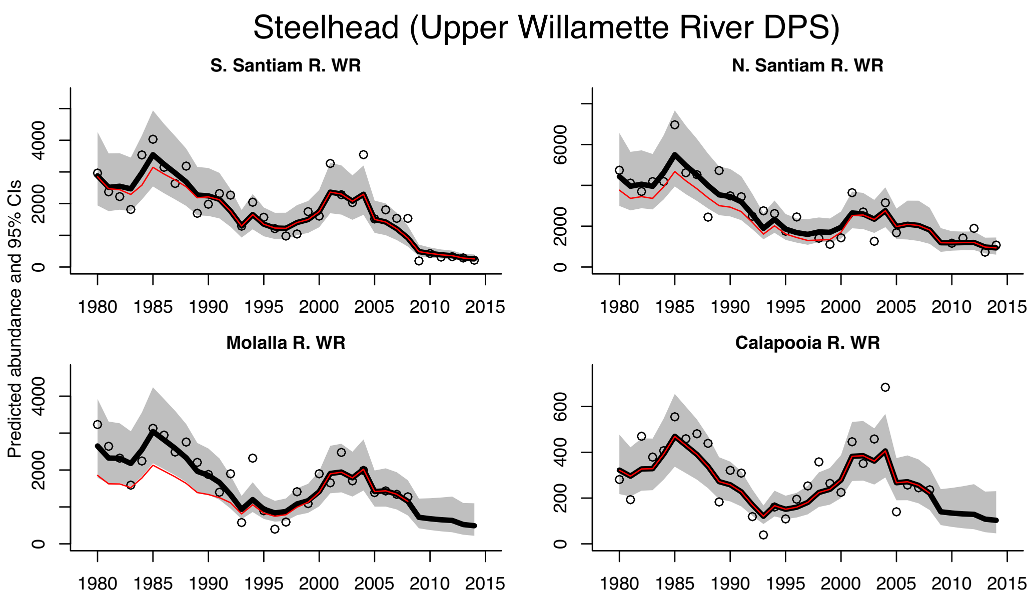
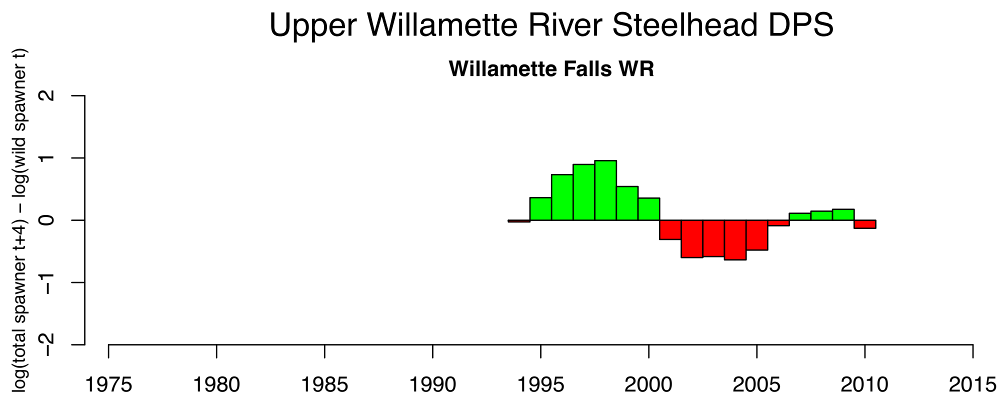


Figure 94 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates. Abundance estimates include both early (non-native) and late-winter (native) steelhead. Abundance estimates for Willamette Falls likely includes a much larger proportion of non-native fish than for the East Side tributary estimates.



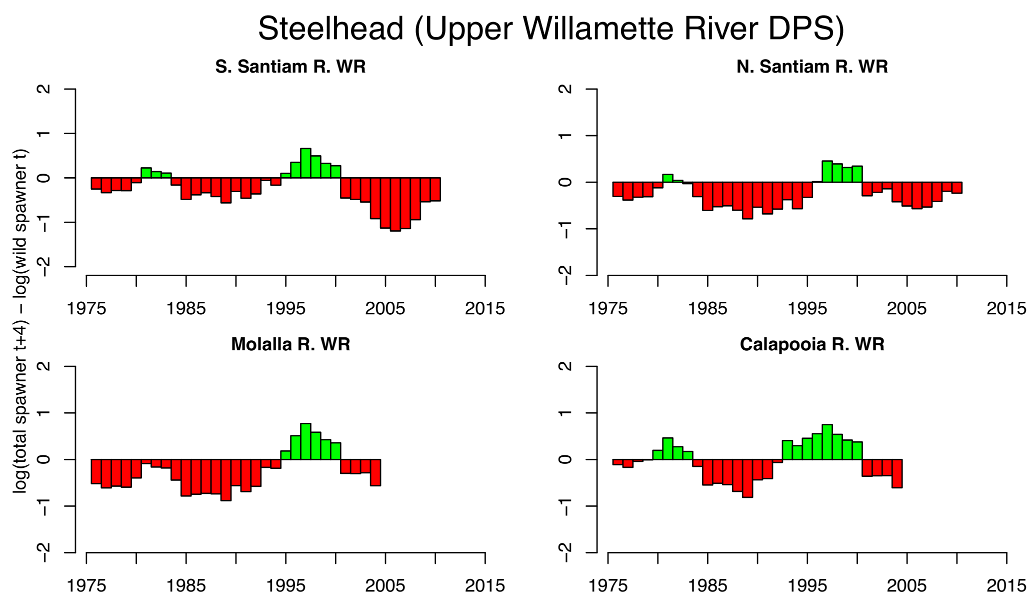


Figure 95 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).

Table 53 -- 5-year geometric mean of raw natural origin spawner counts. This is the raw total spawner count times the fraction natural origin estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural origin spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Willamette Falls WR	Willamette	(5619)	5039 (3961)	10135 (10135)	4926 (4926)	6164 (6164)	25 (25)

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
S. Santiam R. WR	Willamette	1940 (1940)	1277 (1277)	2440 (2440)	1044 (1044)	306 (306)	-71 (-71)
N. Santiam R. WR	Willamette	2494 (2928)	1285 (1611)	2178 (2234)		1195 (1195)	
Molalla R. WR	Willamette	1182 (1462)	726 (798)	1924 (1924)	1357 (1357)		
Calapooia R. WR	Willamette	149 (149)	219 (219)	406 (406)	214 (214)		

Table 54 -- 15-year trends in log natural origin spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 natural origin spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Willamette Falls WR	Willamette		-0.04 (-0.06, -0.01)

Population	MPG	1990-2005	1999-2014
S. Santiam R. WR	Willamette	0.01 (-0.02, 0.04)	-0.16 (-0.2, -0.12)
N. Santiam R. WR	Willamette	0 (-0.03, 0.04)	-0.05 (-0.08, -0.02)
Molalla R. WR	Willamette	0.04 (0.01, 0.07)	
Calapooia R. WR	Willamette	0.05 (0.02, 0.09)	

HARVEST

There is no directed fishery for winter steelhead in the Upper Willamette River. Due to differences in return timing between native winter-run steelhead, introduced hatchery summer-run steelhead, and hatchery spring-run Chinook salmon the encounter rates for winter-run fish in the recreational fishery are thought to be low. Sport fishery mortality rates were estimated at 0-3% (Beamsderfer *et al.* 2011). There is additional incidental mortality in the commercial net fisheries for Chinook salmon and steelhead in the lower Columbia River. Tribal fisheries occur above Bonneville Dam and do not impact Upper Willamette River steelhead.

See the chapter on Middle Columbia River Steelhead for a discussion of trends in harvest rates for Columbia Basin steelhead.

SPATIAL STRUCTURE AND DIVERSITY

Winter-run steelhead hatchery programs were terminated in the late 1990s. Currently, the only steelhead programs in the Upper Willamette River release Skamania Hatchery-origin summer-run steelhead. Annual total releases have been relatively stable at around 600,000 from (2009-2014), although the distribution has changed some with fewer fish being released in the North Santiam and corresponding increases in the South Santiam and Middle Fork Willamette rivers (Figure 96).

There has been some concern regarding the effect of introduced summer-run steelhead on native late-winter steelhead. There is some overlap in the spawn timing for summer- and late-winter steelhead, and genetic analysis has identified approximately 10% of the juvenile steelhead as summer x winter-run hybrids at Willamette Falls and in the Santiam Basin (Johnson *et al.* 2013). Early-winter steelhead, derived from earlier (now discontinued) releases of non-native Big Creek Hatchery steelhead have established themselves in tributaries draining the west side of the Willamette Valley. Based on the results of Johnson *et al.* (2013), approximately 10.5% of the juveniles sampled at Willamette Falls were early- x late-winter steelhead hybrids, with similar proportions detected in the North and South Santiam, 11.1% and 14.8%, respectively. While not directly determining the presence of hybrids, Van Doornik *et al.* (2015) concluded that late-winter (eastern tributary) steelhead had largely maintained their genetic distinctiveness over time. Even in the absence of long-term introgression, there are still concerns that hybridization will decrease the overall productivity of the native population.

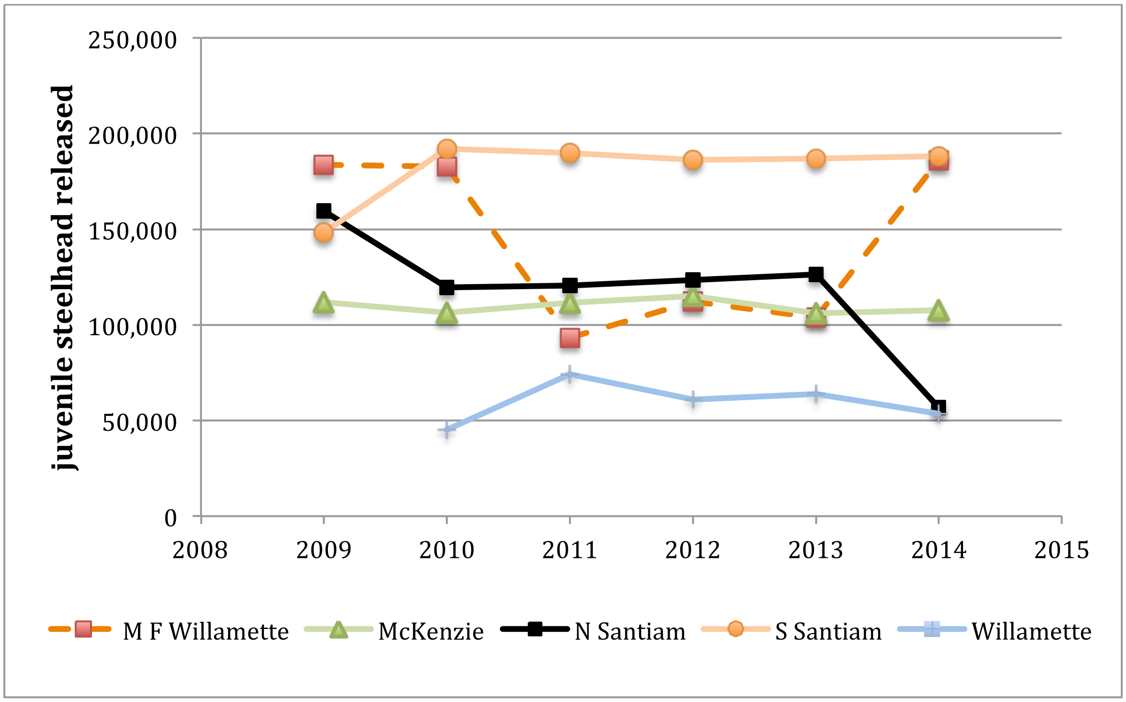
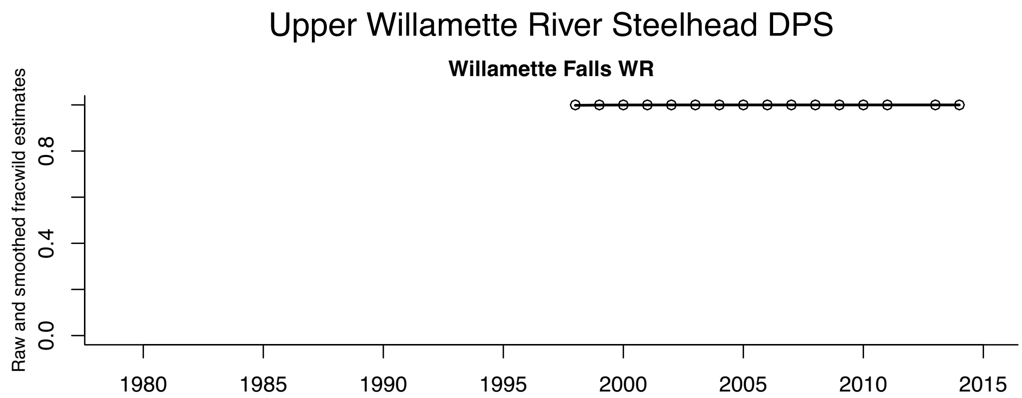


Figure 96 - Annual releases of hatchery-origin (Skamania stock) summer-run steelhead into Willamette River tributaries by subbasin. All of these releases are considered to be out-of-DPS origin. Data from RMIS (http://www.rmpc.org/ accessed January 6, 2015).

The presence of hatchery-reared and feral hatchery-origin fish may also affect the growth and survival of juvenile late-winter steelhead. In the North and South Santiam rivers, juveniles are largely confined below much of their historical spawning and rearing habitat. Releases of large numbers of hatchery-origin summer steelhead may temporarily exceed rearing capacities and displace winter-run juvenile steelhead.



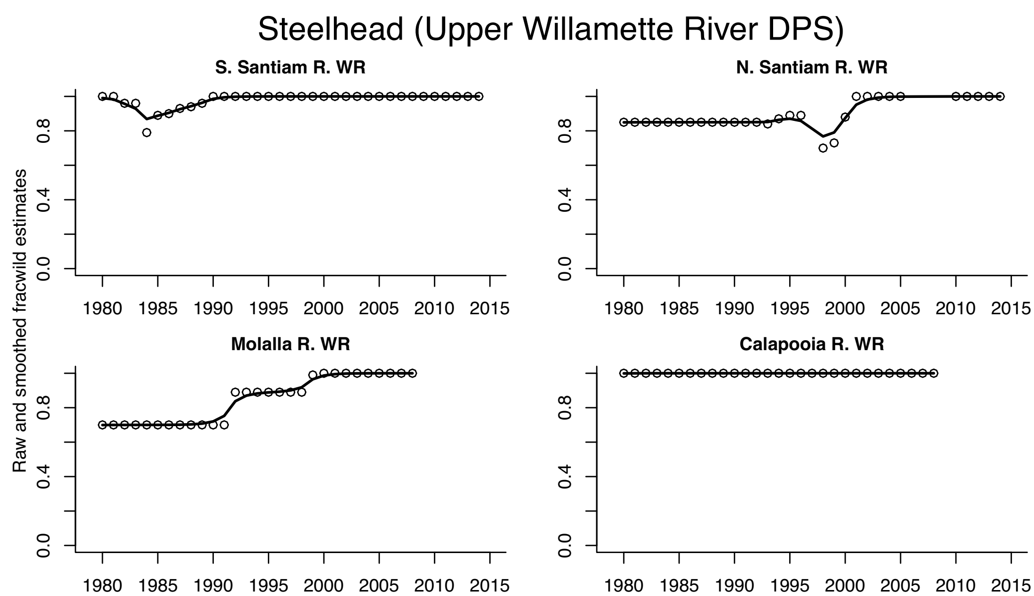


Figure 97 – Fraction natural origin spanwers.

Table 55 -- 5-year mean of fraction wild (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Willamette Falls WR		1.00	1.00	1.00	1.00

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
S. Santiam R. WR	1.00	1.00	1.00	1.00	1.00
N. Santiam R. WR	0.85	0.80	0.98	1.00	1.00
Molalla R. WR	0.81	0.91	1.00	1.00	
Calapooia R. WR	1.00	1.00	1.00	1.00	

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

Populations in this DPS have experienced long-term declines in spawner abundance. The underlying cause(s) of these declines is not well understood. Returning winter steelhead do not experience the same deleterious water temperatures as the spring-run Chinook salmon. Although the recent magnitude of these declines is relatively moderate, continued declines would be a cause for concern. Improvements to Bennett Dam fish passage and operational temperature control at Detroit Dam may be providing some stability in abundance in the North Santiam River DIP. It is unclear if sufficient high quality habitat is available below Detroit Dam to support the population reaching its VSP recovery goal, or if some form of access to the upper watershed is necessary to sustain a "recovered" population. Similarly, the South Santiam Basin may not be able to achieve its recovery goal status without access to historical spawning and rearing habitat above Green Peter Dam (Quartzville Creek and Middle Santiam River) and/or improved juvenile downstream passage at Foster Dam.

While the diversity goals are partially achieved through the closure of winter-run steelhead hatchery programs in the Upper Willamette River, there is some concern that the summer-run steelhead releases in the South Santiam River may be influencing the viability of native steelhead in the North and South Santiam rivers. Overall, none of the populations in the DPS are meeting their recovery goals (Figure 98).

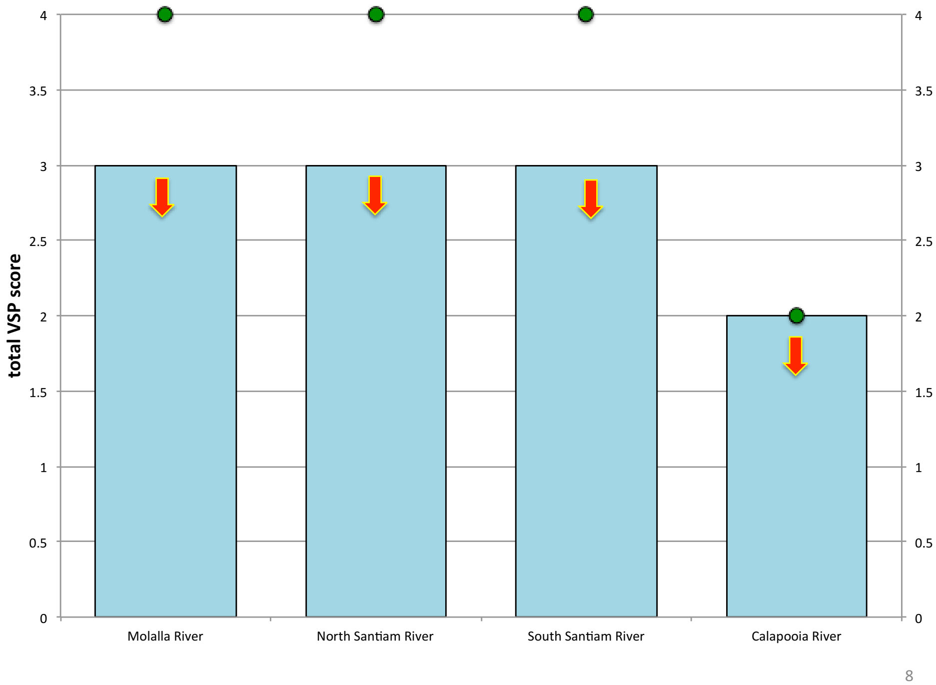


Figure 98 - VSP status of demographically independent populations in the Upper Willamette River Chinook salmon ESU, bars indicate the initial VSP status (as identified in the Recovery Plan-Dornbush and Sihler 2013), green circles indicate the recovery goals. Arrows indicate the general direction, but not the magnitude, of any VSP population scorebased on new data reviewed in this status review update. Arrows reflect the conclusions of the section author; a formal review of VSP scores would require the conviening of a Biological Review Team. Viable Salmon Population scores represent a combined assessment of population abundance and productivity, spatial structure and diversity (McElhany et al. 2006). A VSP score of 3.0 is represents a population with a 5% risk of extinction within a 100 year period.

UPDATED BIOLOGICAL RISK SUMMARY

Overall, the declines in abundance noted during the previous review (Ford *et al*. 2012) continued through the period 2010-2015 (Figure 94). There is considerable uncertainty in many of the abundance estimates, except for perhaps the tributary dam counts. Radio-tagging studies suggest that a considerable proportion of winter steelhead ascending Willamette Falls do not enter the DIPs that constitute this DPS; these fish may be non-native early winter steelhead that appear to have colonize the western tributaries, misidentified summer steelhead, or late-winter steelhead that have colonized tributaries not historically part of the DPS. More definitive genetic monitoring of steelhead ascending Willamette Falls in tandem with radio tagging work needs to be undertaken to estimate the total abundance of the DPS.

The release of non-native summer-run steelhead continues to be a concern. Genetic analysis suggests that there is some level introgression among native late-winter steelhead and summer-run steelhead (Van Doornik *et al.* 2015). Accessibility to historical spawning habitat is still limited, especially in the North Santiam River. Much of the accessible habitat in the Molalla, Calapooia, and lower reaches of North and South Santiam rivers is degraded and under continued development pressure. Although habitat restoration efforts are underway, the time scale for restoring functional habitat is considerable.

OREGON AND WASHINGTON COAST DOMAIN STATUS SUMMARIES

PUGET SOUND CHINOOK SALMON ESU

BRIEF DESCRIPTION OF ESU

The ESU includes all naturally spawning populations of Chinook salmon from rivers and streams flowing into Puget Sound including the Strait of Juan De Fuca from the Elwha River, eastward, including rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington, as well as numerous artificial propagation programs (Figure 99). The Puget Sound Chinook salmon ESU is composed of 31 historically quasi-independent populations, 22 of which are extant (Ruckelshaus *et al.* 2006). The populations are distributed in 5 geographic regions, or major population groups (MPG's), identified by the TRT (PSTRT 2002) based on similarities in hydrographic, biogeographic, and geologic characteristics of the Puget Sound basin.

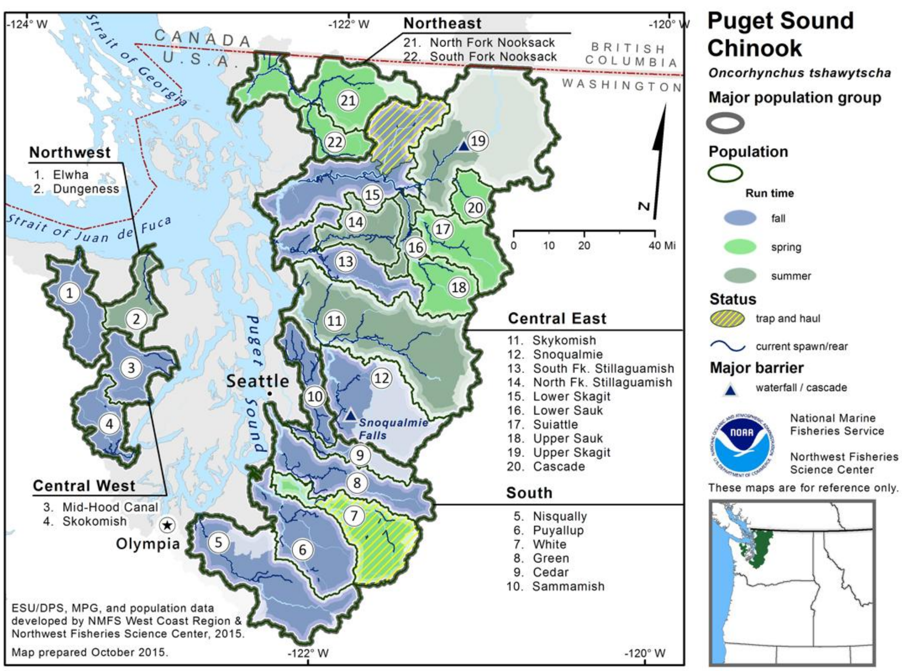


Figure 99 -- Map of the Puget Sound Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

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SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

In the 2005 review (Good *et al*. 2005), the BRT concluded that, overall, the status of natural spawning populations of Puget Sound Chinook salmon was improved relative to the time of the previous status review conducted with data through 1997 (Myers *et al*. 1998). Also, the overall trends in natural spawning escapements for Puget Sound Chinook salmon populations estimated in 2005 remained similar to that presented in the previous status review (data through 1997), with some populations doing marginally better and others worse.

2010

Ford *et al*. (2011) concluded that all Puget Sound Chinook populations were well below the TRT minimum planning range for recovery escapement levels. Most populations were also consistently below the spawner-recruit levels needed for recovery. The exceptions were the Skagit system populations, which tended to have higher status. The Whidbey Basin MPG was also at relatively low risk. The other four MPGs were considered to be at high risk of extinction due to low abundance and productivity values. Their low numbers also contributed to poor spatial distribution of spawners throughout the ESU. Overall, the new information on abundance, productivity, spatial structure and diversity considered in the 2010 review did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

This status report incorporates Chinook salmon population data through 2013, with data for some populations also available through 2014. Spawning abundance data were obtained from WDFW and the Puget Sound tribes as a result of a request for data in the Federal Register. Updates for abundance, age, and hatchery contribution data varied from population to population, and were obtained from the annual postseason harvest reports provided by Washington Department of Fish and Wildlife and the Puget Sound Treaty Indian Tribes, and from the WDFW SaSI database (http://wdfw.wa.gov/conservation/fisheries/sasi/). Additional hatchery data were queried from WDFW statewide hatchery database, both queried online in annual hatchery escapement reports (http://wdfw.wa.gov/hatcheries/escapement/) and data provided by staff. It is important to note that data collection methodologies in both hatcheries and natural spawning locations have changed somewhat over the course of the time series analyzed, which creates some uncertainty and potential bias in the calculations of trends.

This status review focuses on data starting in 1980, although some populations have data going back much further. In addition to including additional recent years of spawning data compared to the 2010 status review, the report also incorporates updates and corrections made in past escapement, age, and hatchery contribution data for many of the populations. These corrections were made by individual tribal and/or state co-managers, and were obtained from periodic management report updates (WDF *et al.* 1993; PSIT & WDFW 2013). These data updates and methods are consistent with both the PSTRT's use for determining population viability, and for prior NOAA status reviews. It is important to note that opinions vary among co-managers regarding data quality, for example, regarding estimates of hatchery contributions to spawning grounds in years prior to mass marking.

We have met with co-managers regarding the development of a single data set, but this data set is not yet fully validated nor agreed upon by all co-managers. We encourage the co-managers to continue this effort and we hope to help resolve the various needs for data management and reporting.

At the root of data discrepancies are concerns regarding data quality, i.e., accuracy and precision of escapement estimates, of estimated hatchery stray rates, and likely measurement error associated with escapement methods and changes in methodology made over time. An assessment of data associated with viability parameters (abundance, productivity, distribution and diversity) was conducted in 2011 to determine data quality, including estimates and protocols (PSEMP 2012). This assessment provided an inventory of all VSP monitoring, including hatchery effectiveness monitoring, data quality and certainty, and data gaps. This assessment represents a good starting point for co-managers to consider building ESU-wide improvements and consistency for monitoring Puget Sound Chinook populations. In addition, the Sentinel Stocks Program, implemented as part of the 2009 Pacific Salmon Treaty Agreement (and funded from the U.S. Letter of Agreement) has more rigorously studied escapement assessments in several Puget Sound watersheds (Stillaguamish River 2007-2015, Green River 2000-2002 and 2010-2012, Snohomish River 2011-2015, and Nooksack River 2009-2015). Results from the Sentinel Stock Program could be used in future efforts to improve escapement estimation protocols in the Puget Sound Chinook salmon ESU (CTC 2015). Results from these studies have been recently summarized and a final report will be available in 2016 (Tom Cooney, pers. comm.).

ABUNDANCE AND PRODUCTIVITY

Abundance of the 22 extant natural spawning populations of Chinook salmon in the Puget Sound ESU has varied considerably between populations. Trends in abundance for individual populations are shown in Figure 100. The populations, grouped by MPG and run timing (early run (ER), late run (LR) or summer run (SuR)), include: Strait of Juan de Fuca MPG (Elwha and Dungeness), Hood Canal MPG (Skokomish and Mid-Hood Canal), Central/South Puget Sound MPG (Sammamish, Cedar, Green, White, Puyallup and Nisqually), Whidbey Basin MPG (Lower Skagit, Upper Skagit, Cascade, Lower Sauk, Upper Sauk, Suiattle, North Fork Stillaguamish, South Fork Stillaguamish, Skykomish, and Snoqualmie), and Strait of Georgia MPG (North Fork Nooksack and South Fork Nooksack) (NMFS 2006). The early run timing populations are North and South Forks Nooksack, Cascade, Upper Sauk, Suiattle, in the northern Puget Sound MPGs and White River population in Central/South Puget Sound. Dungeness is the only summer run, and all other populations are late runs.

Total abundance in the ESU over the entire time series shows that individual populations have varied in increasing or decreasing abundance, with some being dominated by hatchery returns. Generally, many populations experienced an increase in abundance from during the years 2000-2008 and then declining in the last 5 years (Figure 100). Abundance across the Puget Sound ESU has generally decreased since the last status review, with only 6 of 22 populations (Cascade, Cedar, Mid-Hood Canal, Nisqually, Suiattle and Upper Sauk) show a positive % change in the 5-year geometric mean natural-origin spawner abundances since the prior status review (Table 56). However, all 6 of these populations have relatively low natural spawning abundances of < 1000 fish, so these increases represent small changes in total abundance. Given lack of high confidence in survey techniques, particularly with small populations, there remains substantial uncertainty in detecting trends in small populations.

Fifteen-year trends in log wild spawner abundance were computed over two time periods (1990-2005 and 1999-2014) for each Puget Sound Chinook population (Table 57). Trends were negative in the latter period for 17 of the 22 populations but only 2 of the 22 populations (Elwha and Puyallup) in the earlier period. Thus there is a general decline in wild spawner abundance across all MPGs in the recent fifteen years. North Fork Nooksack (Strait of Georgia MPG), Cascade and Upper Sauk (Whidby Basin MPG), Cedar (Central/South MPG) and Dungeness (Strait of Juan de Fuca MPG), are the only populations with positive trends. The Cedar and the Upper Sauk populations are the only 2 populations that show increasing trends between the earlier and later 15-year time periods (Table 57). The average trend across the ESU for the 1990-2005 15-year time period was 0.05. The average trends for the Regions/MPGs are Strait of Georgia (0.05), Whidby Basin (0.04), Central/South Puget Sound (0.06), Hood Canal (0.02), and Strait of Juan de Fuca (0.06). The average trend across the ESU for the later 15-year time period (1999-2014) was -0.02. The average trends for the Regions/MPGs are Strait of Georgia (-0.01), Whidby Basin (-0.02), Central/South Puget Sound (-0.03), Hood Canal (-0.07), and Strait of Juan de Fuca (0.01). While the previous status review in 2010 (Ford *et al*. 2011) concluded there was no obvious trend for the total ESU escapements and trends for individual populations were variable, addition of the data to 2014 now does show widespread negative trends in natural-origin Chinook salmon spawner population abundances.

Chinook salmon productivity in the Puget Sound ESU across the time period (1980-2015) has been variable. Figure 101 shows trends in productivity as estimated by the log of the smoothed natural-origin spawning abundance in year t minus the smoothed natural spawning abundance in year (t-4). Data below zero indicate that natural spawners failed to replace themselves, although in many cases total spawning abundance was maintained through hatchery supplementation (compare red and black lines in Figure 100). Across the Puget Sound ESU, 8 of 22 Puget Sound populations show natural productivity below replacement in all years since the mid-1980's. These include the Skykomish in Whidby Basin MPG, the Skokomish in the Hood Canal MPG, North and South Forks Nooksack in the Strait of Georgia MPG, Dungeness and Elwha in the Strait of Juan de Fuca MPG, and Green and Puyallup in the Central/South Puget Sound MPG. Productivity in the Whidby Basin MPG populations was above zero during much of the 1990's, with the exception of the Skykomish and North Fork Stillaguamish populations. White River population in the Central/South Puget Sound MPG was above replacement from the mid 1980's to early 2000's, but has dropped in productivity consistently since the late 1980's. In recent years, only 8 populations have been above zero. These are Cascade, Lower Sauk, Lower Skagit, Suiattle, Upper Sauk, Upper Skagit in the Whidby Basin MPG, and Mid-Hood Canal and Cedar River in the Hood Canal and Central/South Puget Sound MPG's, respectively. This is consistent with, and continues the decline reported in the 2010 Status Review (Ford *et al*. 2011).

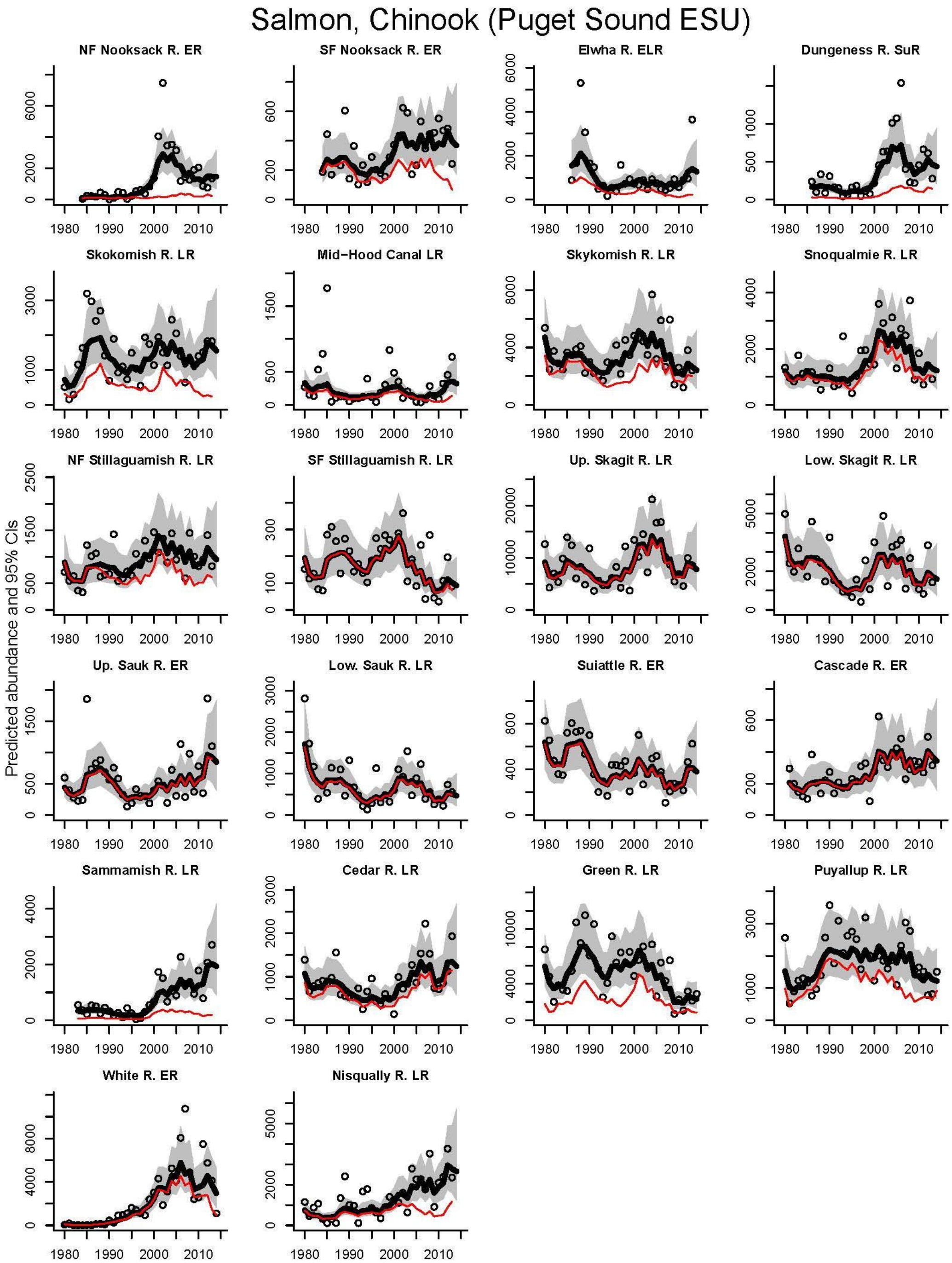

Figure 100 – Smoothed trend in estimated total (thick black line) and natural (thin red line) Puget Sound Chinook salmon ESU individual populations spawning abundance. Points show the annual raw spawning abundance estimates.

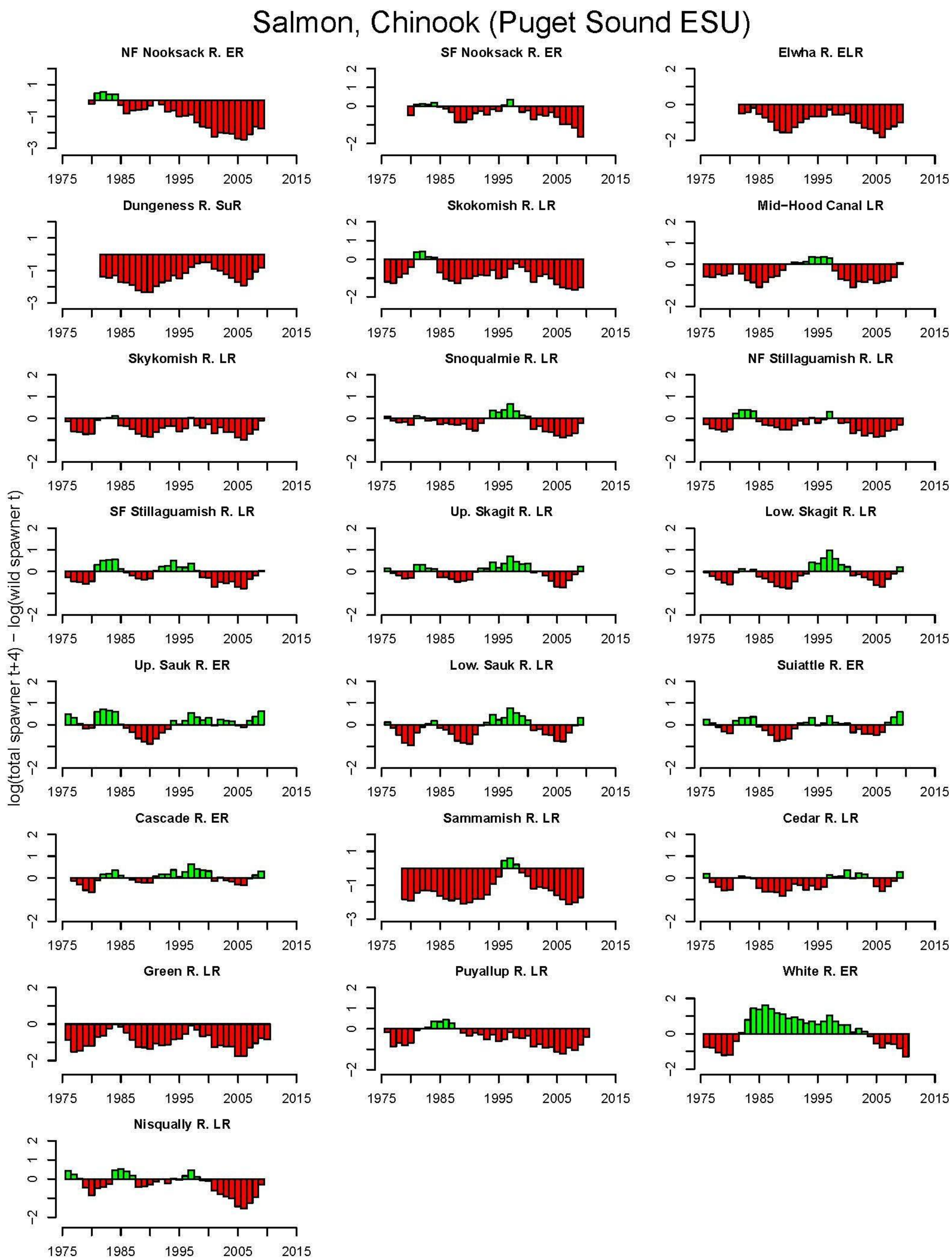

Figure 101 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).

Table 56 -- 5-year geometric mean of raw wild spawner counts. This is the raw total spawner count times the fraction wild estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of wild spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
NF Nooksack R. ER	Strait of Georgia	52 (102)	97 (476)	229 (3476)	277 (1675)	154 (1167)	-44 (-30)
SF Nooksack R. ER	Strait of Georgia	126 (171)	133 (217)	235 (398)	244 (388)	88 (418)	-64 (8)
Elwha R. ELR	SJF	420 (658)	274 (735)	357 (716)	193 (597)	164 (1152)	-15 (93)
Dungeness R. SuR	SJF	20 (117)	18 (104)	71 (527)	162 (508)	119 (477)	-27 (-6)
Skokomish R. LR	Hood Canal	506 (994)	478 (1232)	479 (1556)	500 (1216)	256 (1627)	-49 (34)
Mid-Hood Canal LR	Hood Canal	93 (119)	152 (186)	169 (217)	47 (88)	75 (314)	60 (257)
Skykomish R. LR	Whidbey Basin	1658 (2325)	1494 (3327)	2606 (4842)	2388 (3350)	1693 (2320)	-29 (-31)
Snoqualmie R. LR	Whidbey Basin	873 (1035)	739 (1187)	2161 (2480)	1311 (1965)	885 (1143)	-32 (-42)
NF Stillaguamish R. LR	Whidbey Basin	553 (742)	603 (946)	967 (1225)	550 (984)	574 (976)	4 (-1)
SF Stillaguamish R. LR	Whidbey Basin	150 (150)	241 (241)	219 (219)	101 (102)	71 (87)	-30 (-15)
Up. Skagit R. LR	Whidbey Basin	5389 (5599)	6159 (6267)	12039 (12484)	9975 (10611)	6924 (7194)	-31 (-32)
Low. Skagit R. LR	Whidbey Basin	1417 (1473)	1001 (1041)	2765 (2857)	2118 (2216)	1391 (1446)	-34 (-35)
Up. Sauk R. ER	Whidbey Basin	394 (409)	258 (268)	413 (428)	498 (518)	836 (867)	68 (67)
Low. Sauk R. LR	Whidbey Basin	399 (414)	414 (433)	812 (853)	546 (572)	413 (432)	-24 (-24)
Suiattle R. ER	Whidbey Basin	295 (302)	373 (382)	405 (415)	254 (261)	351 (360)	38 (38)
Cascade R. ER	Whidbey Basin	185 (189)	208 (213)	364 (371)	334 (341)	338 (345)	1 (1)
Sammamish R. LR	Central/South Sound	52 (227)	32 (160)	385 (1040)	289 (1281)	160 (1679)	-45 (31)
Cedar R. LR	Central/South Sound	367 (509)	369 (541)	405 (643)	1043 (1275)	881 (1075)	-16 (-16)
Green R. LR	Central/South Sound	2253 (5331)	2149 (7272)	4099 (6624)	1334 (3187)	897 (2168)	-33 (-32)
Puyallup R. LR	Central/South Sound	2143 (2543)	1611 (2340)	1171 (1687)	795 (2012)	598 (1186)	-25 (-41)
White R. ER	Central/South Sound	565 (645)	1307 (1415)	3128 (3309)	4170 (5301)	1689 (3471)	-59 (-35)
Nisqually R. LR	Central/South Sound	630 (806)	596 (748)	891 (1319)	587 (1963)	701 (2577)	19 (31)

Table 57 -- 15-year trends in log wild spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period. Lower and upper bounds of the 95% confidence intervals of the estimates are in parentheses. LR = late run, ER = early run, SuR = summer run.

Population	MPG	1990-2005	1999-2014
NF Nooksack R. ER	Strait of Georgia	0.07 (0.04, 0.09)	0.04 (0, 0.07)
SF Nooksack R. ER	Strait of Georgia	0.03 (0, 0.06)	-0.06 (-0.1, -0.02)
Elwha R. ELR	SJF	-0.02 (-0.06, 0.02)	-0.06 (-0.1, -0.03)
Dungeness R. SuR	SJF	0.14 (0.08, 0.19)	0.09 (0.03, 0.14)
Skokomish R. LR	Hood Canal	0.02 (-0.01, 0.05)	-0.07 (-0.11, -0.02)
Mid-Hood Canal LR	Hood Canal	0.03 (0, 0.07)	-0.07 (-0.11, -0.02)
Skykomish R. LR	Whidbey Basin	0.03 (0, 0.06)	-0.02 (-0.04, 0.01)
Snoqualmie R. LR	Whidbey Basin	0.09 (0.05, 0.12)	-0.05 (-0.08, -0.03)
NF Stillaguamish R. LR	Whidbey Basin	0.04 (0.02, 0.06)	-0.04 (-0.06, -0.01)
SF Stillaguamish R. LR	Whidbey Basin	0.01 (-0.01, 0.03)	-0.1 (-0.12, -0.08)
Up. Skagit R. LR	Whidbey Basin	0.07 (0.05, 0.09)	-0.03 (-0.06, 0)
Low. Skagit R. LR	Whidbey Basin	0.05 (0.02, 0.09)	-0.03 (-0.06, -0.01)
Up. Sauk R. ER	Whidbey Basin	0.01 (-0.02, 0.04)	0.06 (0.04, 0.08)
Low. Sauk R. LR	Whidbey Basin	0.05 (0.01, 0.08)	-0.04 (-0.07, -0.01)
Suiattle R. ER	Whidbey Basin	0.01 (-0.01, 0.03)	-0.01 (-0.04, 0.01)
Cascade R. ER	Whidbey Basin	0.06 (0.04, 0.08)	0.01 (-0.01, 0.03)
Sammamish R. LR	Central/South Sound	0.17 (0.11, 0.23)	-0.02 (-0.06, 0.02)
Cedar R. LR	Central/South Sound	0.03 (0, 0.06)	0.07 (0.05, 0.1)
Green R. LR	Central/South Sound	0.02 (-0.02, 0.06)	-0.12 (-0.16, -0.09)
Puyallup R. LR	Central/South Sound	-0.03 (-0.05, -0.02)	-0.06 (-0.08, -0.03)
White R. ER	Central/South Sound	0.19 (0.17, 0.21)	-0.03 (-0.08, 0.01)
Nisqually R. LR	Central/South Sound	0.05 (0.03, 0.06)	-0.01 (-0.05, 0.03)

HARVEST

Puget Sound Chinook are harvested in ocean salmon fisheries, in Puget Sound fisheries, and in terminal fisheries in the rivers. They migrate to the north, so nearly all of the ocean fishery impacts occur in Canada and Alaska where they are subject to the Pacific Salmon Treaty. Fisheries within Puget Sound are managed by the state and tribal co-managers under a resource management plan. Fishery impact rates vary widely among regions within Puget Sound primarily because of different terminal area management. Hood Canal and South Sound stocks support relatively intense terminal area fisheries directed at hatchery fish produced largely to support tribal and recreational fisheries.

Stocks from most regions within Puget Sound show a similar pattern of declining exploitation rates in the 1990s and increasing exploitation rates since then (Figure 102). This is primarily a result of Canadian interceptions of Puget Sound Chinook off the West Coast of Vancouver Island (WCVI). During the 1990s Canada sharply reduced fisheries off WCVI in response to depressed stocks. Since then, WCVI stock status has improved somewhat and Canadian managers have changed the temporal pattern of fishing to avoid WCVI stocks. This has resulted in increased impacts on Puget Sound stocks. The notable exception to this pattern is the North Puget Sound region. These stocks migrate through the Strait of Georgia. Canadian stocks in the Strait of Georgia have not recovered and most fisheries in Canadian inside waters for Chinook and coho salmon have been shut down. The most recent Pacific Salmon Treaty Chinook agreement took effect in 2009 that includes 30% reductions in Chinook catch ceilings off WCVI, and 15% reductions in southeast Alaska.

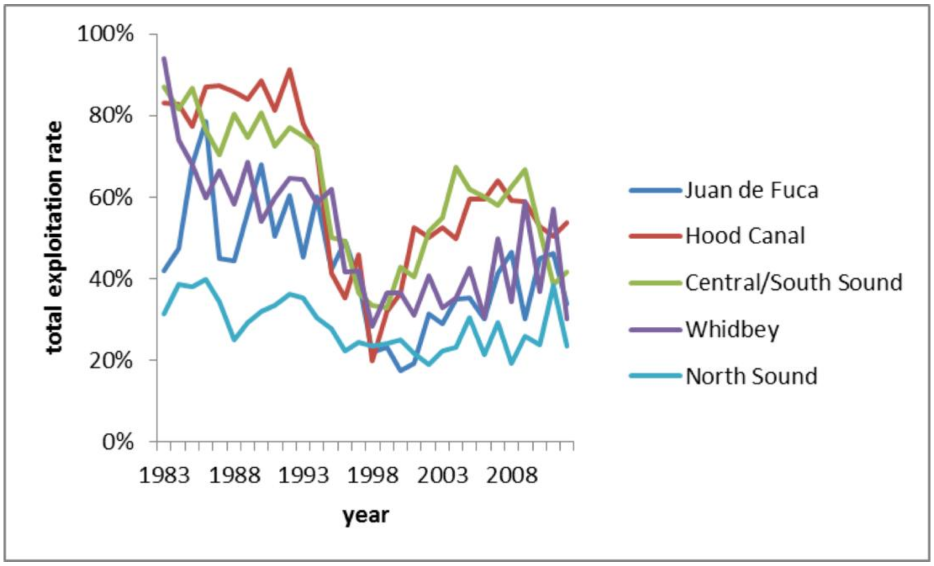

Figure 102 -- Total exploitation rates on natural Puget Sound Chinook salmon by major population group. Data from Fishery Regulation Analysis Model validation runs prepared for the 2014 State of Salmon in Watersheds biennial report of the Governor's Salmon Recovery Office to the Washington State Legislature (Angelika Hagen-Breaux, WDFW personal communication).

SPATIAL STRUCTURE AND DIVERSITY

Measures of spatial structure and diversity can give some indication of the resilience of a population to sustain itself. Spatial structure can be measured in various ways, but here we assess the proportion of natural-origin spawners (wild fish) vs. hatchery-origin spawners on the spawning grounds.

We can see a declining trend in the proportion of natural-origin spawners across the ESU during the entire time period 1990-2014. Figure 103 shows the smoothed trends in the estimated fraction of the natural spawning populations that consist of natural–origin spawners. The populations with the highest fractions of natural-origin spawners across the entire 1980 to 2014 time period are the 6 Skagit populations, and also South Fork Stillaguamish population in the Whidby Basin MPG. All other populations vary considerably across the whole time period, and 12 (North Fork Stillaguamish, Snoqualmie, Mid Hood Canal, Skokomish, North Fork Nooksack, South Fork Nooksack, Elwha, Nisqually, Puyallup, Sammamish and White) show declining trends in the fraction wild estimates. Skykomish, Dungeness, Cedar are the only populations which show more recent trends of increasing fraction natural-origin spawner abundances.

Evidence of the decline in fraction wild spawner abundance is also shown in Table 58. It is important to note that quality of hatchery contribution data in the earlier time periods, prior to mass marking programs, may be poor, so the long-term trends may lack accuracy in the earlier years. In the Whidbey Basin MPG, the fraction natural origin abundance has been consistently high in the six Skagit populations, and despite ongoing hatchery programs in the Stillaguamish and Snohomish rivers, the average 5-year mean fraction wild estimates for that MPG remains consistent across all time periods. The Strait of Georgia MPG (North and South Forks Nooksack) has had increasing hatchery influence, particularly in the recent 5-year time period. The South Fork Nooksack population had an extremely small wild fish return in 2013, and this population is at high risk of extinction. The Central/South Puget Sound MPG had varied fraction wild estimates in the Cedar and Green, and increases in recent years in Cedar, Nisqually, and Puyallup populations, but decreasing 5-year mean fraction wild estimates in the recent (2010-2014) time period in the Sammamish and White populations (Figure 103, Table 58). In the Hood Canal and Strait of Juan de Fuca MPGs, all 4 populations had declining 5-year mean fraction wild estimates of fish returns to the spawning grounds. Thus, considering populations by MPG, the Whidbey Basin MPG is the only MPG with consistently high fraction natural-origin spawner abundance, in 6 of 10 populations. All other MPG's have either variable or declining spawning populations that have high proportions of hatchery-origin spawners.

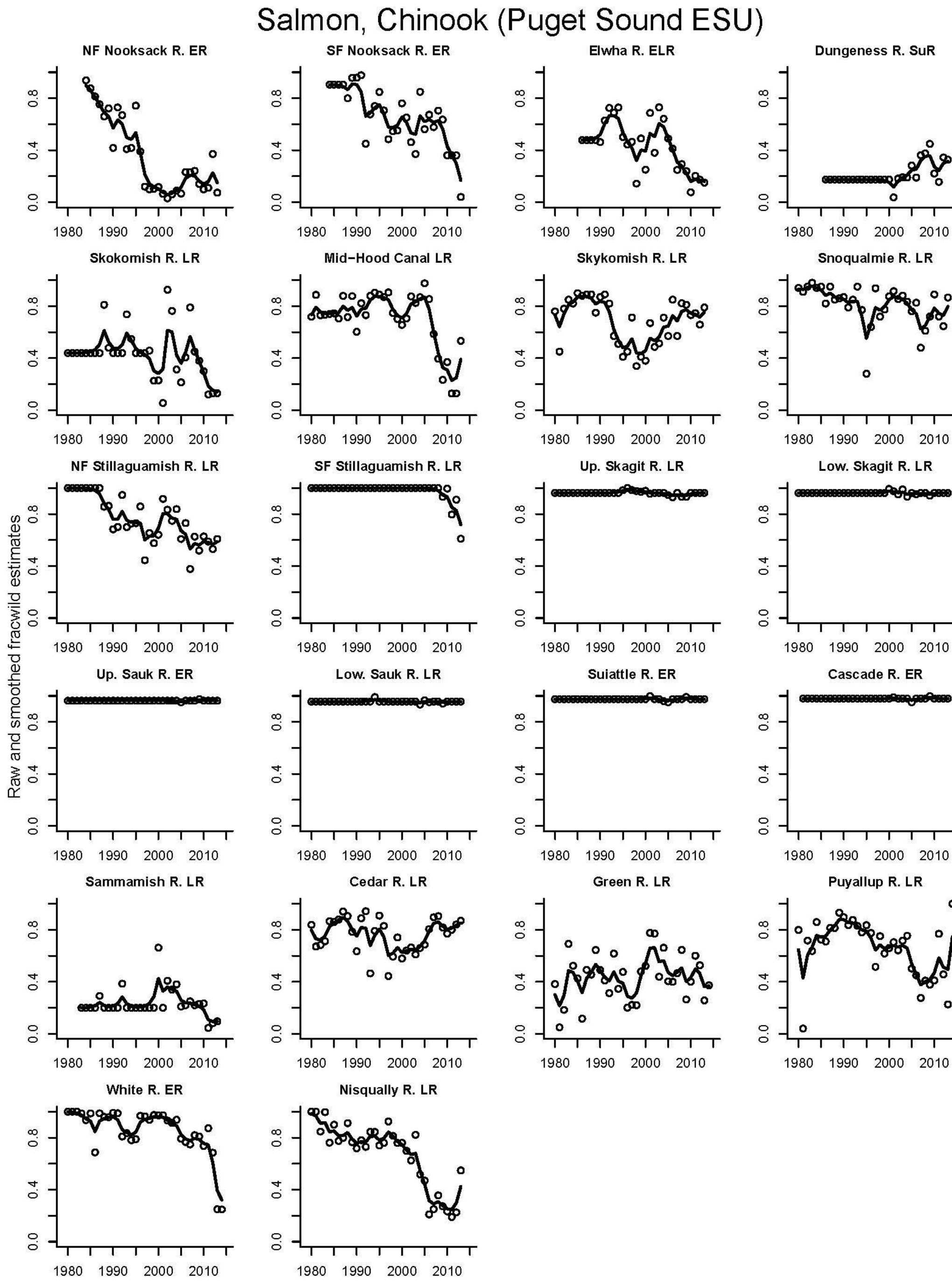

Figure 103 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 58 -- 5-year mean of fraction natural origin spawners (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
NF Nooksack R. ER	0.53	0.29	0.07	0.18	0.16
SF Nooksack R. ER	0.76	0.63	0.62	0.63	0.28
Elwha R. ELR	0.65	0.41	0.54	0.34	0.15
Dungeness R. SuR	0.17	0.17	0.16	0.33	0.26
Skokomish R. LR	0.52	0.40	0.46	0.45	0.17
Mid-Hood Canal LR	0.79	0.82	0.79	0.61	0.29
Skykomish R. LR	0.73	0.46	0.55	0.72	0.73
Snoqualmie R. LR	0.85	0.67	0.87	0.68	0.78
NF Stillaguamish R. LR	0.75	0.65	0.80	0.57	0.59
SF Stillaguamish R. LR	1.00	1.00	1.00	0.99	0.83
Up. Skagit R. LR	0.96	0.98	0.96	0.94	0.96
Low. Skagit R. LR	0.96	0.96	0.97	0.96	0.96
Up. Sauk R. ER	0.96	0.96	0.96	0.96	0.96
Low. Sauk R. LR	0.96	0.96	0.95	0.95	0.96
Suiattle R. ER	0.98	0.98	0.98	0.97	0.98
Cascade R. ER	0.98	0.98	0.98	0.98	0.98
Sammamish R. LR	0.24	0.20	0.40	0.23	0.11
Cedar R. LR	0.74	0.70	0.63	0.82	0.82
Green R. LR	0.44	0.32	0.63	0.44	0.43
Puyallup R. LR	0.84	0.70	0.70	0.40	0.57
White R. ER	0.88	0.93	0.95	0.79	0.56
Nisqually R. LR	0.78	0.80	0.68	0.31	0.30

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The Puget Sound TRT provided viability criteria for each population based on historical information and models with which they developed planning ranges for spawner abundance and productivity (PSTRT 2002). They also specified spatial structure and diversity criteria characteristic of low risk populations. The planning ranges are based on estimates of salmon abundance that can be supported by properly functioning habitat at both low productivity and high productivity. They also recommended ESU-level criteria including: the viability status of all populations in the ESU is improved from current conditions, that 2 to 4 Chinook populations in each of the 5 MPG's within the ESU achieve viability, at least 1 population is viable from each major genetic and life history group historically present within each of the 5 MPGs, and that the populations that do not meet the viability criteria for all 4 VSP parameters are sustained in order to provide ecological functions and preserve options for ESU recovery. Additional criteria described habitat conditions that are needed to support viable salmonid populations.

UPDATED BIOLOGICAL RISK SUMMARY

All Puget Sound Chinook salmon populations are still well below the TRT planning ranges for recovery escapement levels. Most populations are also consistently below the spawner-recruit levels identified by the TRT as consistent with recovery. Across the ESU, most populations have declined in abundance since the last status review in 2011, and indeed, this decline has been persistent over the past 7 to 10 years. Productivity remains low in most populations. Hatchery-origin spawners are present in high fractions in most populations outside the Skagit watershed, and in many watersheds the fraction of spawner abundances that are natural-origin have declined over time. Habitat monitoring and adaptive management planning efforts to develop monitoring plans was undertaken in all individual watersheds of Puget Sound in 2014. Watershed documents can be found on the Puget Sound Partnership website (http://www.psp.wa.gov/SR_threeyearworkplan.php). These reports and prior annual three-year workplans document the many habitat actions that were initially identified in the Puget Sound Chinook salmon recovery plan. The expected benefits will take years or decades to produce significant improvement in natural population viability parameters. Development of a monitoring and adaptive management program was required by NMFS in the 2007 Supplement to the Shared Strategy Recovery Plan, but this program is, as yet, not fully functional for providing assessment of watershed habitat restoration/recovery programs, nor of properly integrating the essentially discrete habitat, harvest and hatchery programs. Overall, new information on abundance, productivity, spatial structure and diversity since the 2010 review does not indicate a change in the biological risk category since the time of the last BRT status review.

PUGET SOUND STEELHEAD DPS

BRIEF DESCRIPTION OF DPS

This report covers the Distinct Population Segment (DPS) of Puget Sound steelhead (*Oncorhynchus mykiss*). These fish are the anadromous form of *O. mykiss* that occur in rivers, below natural barriers to migration, in northwestern Washington State that drain to Puget Sound, Hood Canal, and the Strait of Juan de Fuca between the U.S./Canada border and the Elwha River, inclusive (Figure 104). The Puget Sound Steelhead Technical Recovery Team (TRT) considered genetic and life-history information from steelhead on the Olympic Peninsula and Washington coast and concluded that there was no compelling evidence to alter the DPS boundary described above.

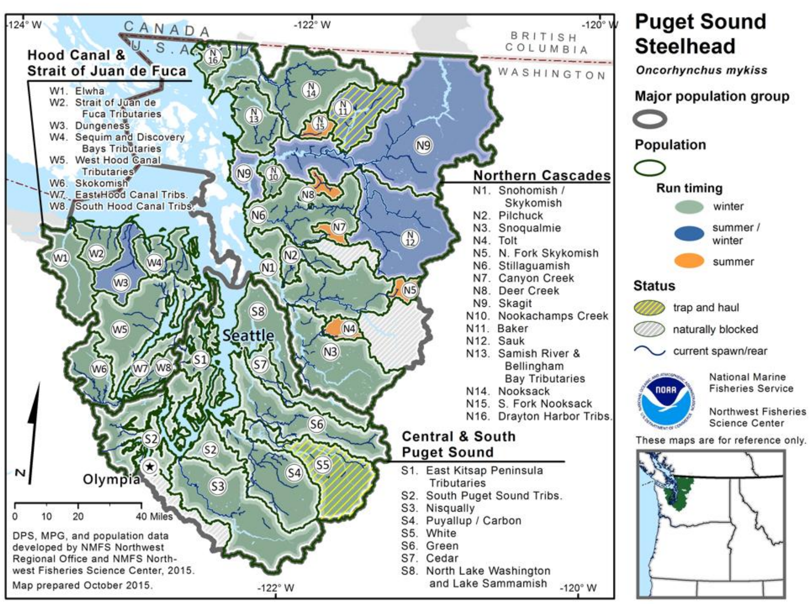

Figure 104 -- Map of the Puget Sound Steelhead DPS's spawning and rearing areas, identifying 32 demographically independent populations (DIPs) within 3 major population groups (MPGs). The 3 steelhead MPGs are Northern Cascades, Central & South Puget Sound, and Hood Canal & Strait of Juan de Fuca.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The initial review of this DPS—then called the Puget Sound Steelhead Evolutionarily Significant Unit (ESU)—by a Biological Review Team (BRT) was completed in 1996 as part of a coastwide status review conducted in response to two listing petitions received by NOAA that identified other potentially imperiled steelhead populations in 1993 and 1994 (Busby *et al*. 1996). Subsequent to that BRT review, NOAA issued a determination that listing of Puget Sound steelhead was not warranted (61 FR 41451). In response to a petition to list Puget Sound steelhead received in September 2004, a newly convened BRT completed its report summarizing the status of the Puget Sound Steelhead DPS in June 2007 (Hard *et al*. 2007). The BRT considered the major risk factors facing Puget Sound steelhead to be widespread declines in abundance and productivity for most natural populations in the DPS (including those in Skagit and Snohomish rivers, previously considered strongholds for steelhead in Puget Sound); the low abundance of all summer-run populations; and continued releases of out-of-DPS hatchery fish from Skamania River-derived summer-run and highly domesticated Chambers Creek-derived winter-run stocks. Most of the populations in the DPS are small, and recent declines in abundance of natural fish have persisted despite widespread reductions in harvest of natural steelhead in the DPS since the mid-1990s. After considering these and other factors such as reduced complexity of spatial structure, evidence for minor contribution of resident *O. mykiss* to anadromous abundance and productivity, and persistently low marine survival of steelhead from Puget Sound, the BRT concluded that steelhead in the DPS were likely to become at risk of extinction throughout all or a significant portion of their range in the foreseeable future, but were not currently in danger of extinction. Subsequent to the BRT's review, NMFS issued its final determination to list the Puget Sound Steelhead DPS as a threatened species under the ESA on 11 May 2007 (72 FR 26722); the effective date of the listing was 11 June 2007.

2010

The 2010 review of the listed Puget Sound Steelhead DPS concluded that its status had not changed substantially since the 2007 listing (Ford *et al.* 2011). Most populations within the DPS were showing continued downward trends in estimated abundance, a few sharply so, and evidence for low productivity was evident throughout the DPS. For all but a few populations, population growth rates were declining on the order of 3 to 10% annually, and extinction risk for most populations over the foreseeable future was estimated to be moderate to high, especially for those in the Central & South Puget Sound and Hood Canal & Strait of Juan de Fuca major population groups (MPGs). The major risk factors facing Puget Sound steelhead had also not changed substantively since listing. Following the 2010 status review, NMFS proposed critical habitat for Puget Sound steelhead on 14 January 2013 (78 FR 2726); the agency updated its determination of the listing status of the DPS on 14 April 2014 (79 FR 20802).

In 2013, the Puget Sound Steelhead TRT finalized its analyses of Puget Sound steelhead data available through 2011 to identify 32 demographically independent populations (DIPs) and 3 MPGs within the DPS (Myers *et al.* 2015) and develop viability criteria for the DPS (Hard *et al.* 2015). In its viability report, the TRT concluded that the threatened Puget Sound Steelhead DPS is not currently viable. The TRT found that low population viability is widespread throughout the DPS, across all three MPGs, and includes both summer-run and winter-run populations. Steelhead populations throughout the DPS showed evidence of diminished abundance, productivity, diversity, and spatial structure when compared with available historical evidence for the states of each of these salmonid population (VSP) parameters.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

This report considers data available through 2014 (where available) to review the current status of Puget Sound steelhead. These data were provided by state and tribal comanagers, which included the Washington Department of Fish and Wildlife, including its Salmon and Steelhead Stock Inventory and Salmonscape databases, and its district area biologists; Washington tribal biologists; and Northwest Indian Fisheries Commission biologists. The report focuses on assessing viability of a subset (n = 22) of the 32 populations in the DPS identified by the TRT for which updated demographic data are available. The viability assessment incorporates basic analyses of abundance and trend, followed by a set of population viability analyses (PVAs) for several of the DIPs and MPGs within the DPS. It also considers the TRT's analyses of steelhead abundance, productivity, diversity and spatial structure in Puget Sound and briefly compares them to the analyses of status developed in the current report.

ABUNDANCE AND PRODUCTIVITY

Abundance of adult steelhead returning to nearly all Puget Sound rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s. Trends in abundance for 22 of the 32 DIPs in the DPS are depicted in Figure 105. The plots depict trends for 8 of the 16 winter-run (WR) and summer-run (SuR) DIPs in the Northern Cascades MPG (Nooksack River WR, Samish River and Bellingham Bay Tributaries WR, Skagit River WR, Pilchuck River WR, Snohomish River/Skykomish River WR, Snoqualmie River WR, Stillaguamish River WR, and Tolt River SuR), 6 of the 8 winter-run DIPs in the Central & South Puget Sound MPG (Cedar River WR, Green River WR, Nisqually River WR, North Lake Washington and Lake Sammamish WR, Puyallup River/Carbon River WR, and White River WR), and 8 of the 8 winter-run DIPs in the Hood Canal & Strait of Juan de Fuca MPG (Dungeness River WR, East Hood Canal Tributaries WR, Elwha River WR, Sequim and Discovery Bay Tributaries WR, Skokomish River WR, South Hood Canal Tributaries WR, Strait of Juan de Fuca Tributaries WR, and West Hood Canal Tributaries WR). Data are available for only one summer-run DIP, the Tolt River in the Northern Cascades MPG.

Most of the analyses that follow use metrics described in the "Methods: Common Metrics" section of this report (p. 8). The plots in Figure 105 show smoothed trends for both natural and total (natural + hatchery) escapements over the time series. Total abundance of steelhead in these populations has shown a generally declining trend over much of the DPS. Since 1980, only half of the 22 populations show evidence of a neutral or increasing trend, and most of these are in the Hood Canal & Strait of Juan de Fuca MPG. Several of the neutral trends are influenced by low estimated abundance in the early 1980s; nearly half of the 8 populations showing neutral trends since 1980 show declining trends between the late 1980s-early 1990s and about 2009, when increasing trends are often apparent. The patterns are nearly identical for trends in natural-origin abundance (Figure 105).

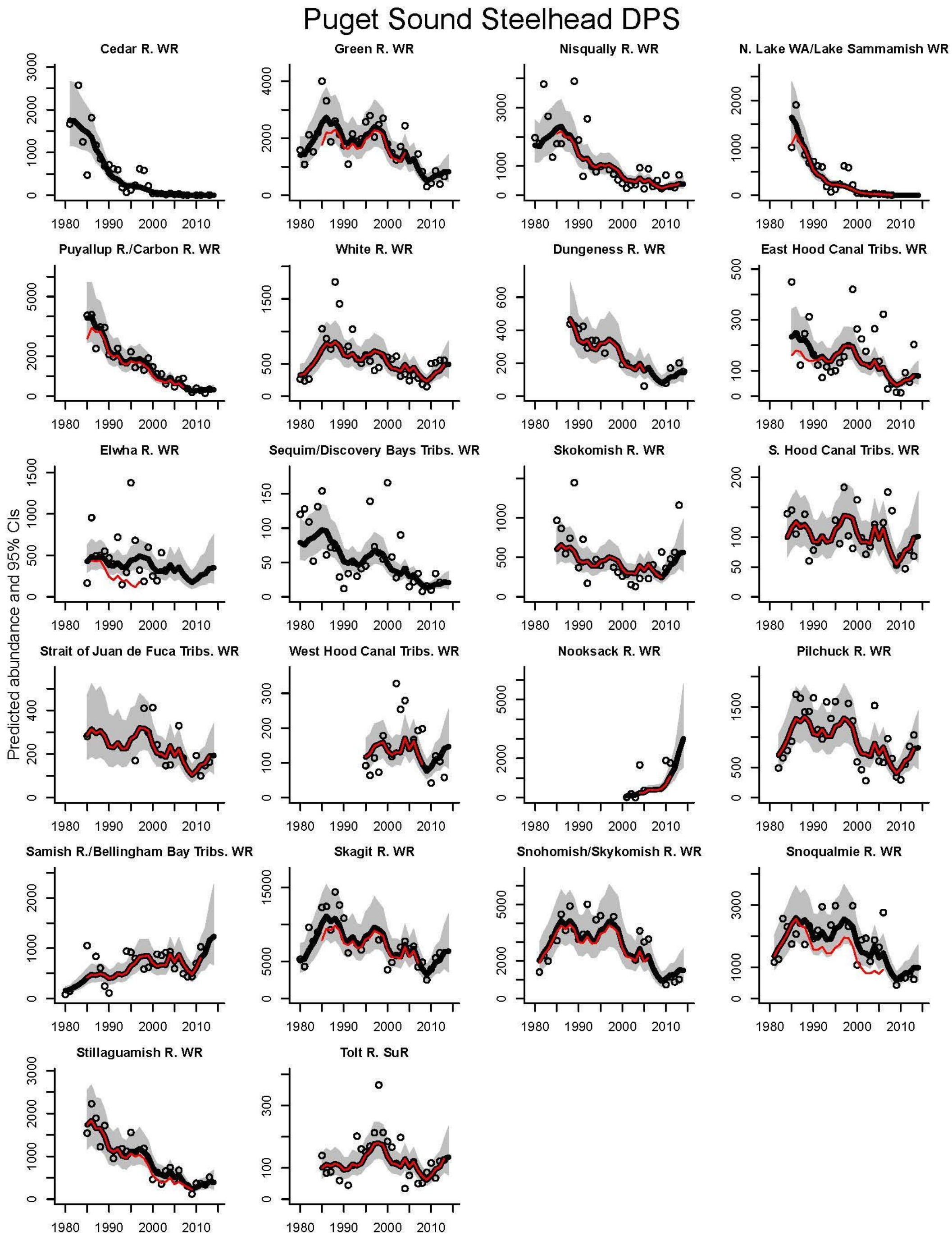

Figure 105 -- Smoothed trends in estimated total (thick black line) and natural (thin red line) population spawning abundance of Puget Sound steelhead. Points show the annual raw spawning abundance estimates. Grey bands depict the 95% confidence intervals around the estimates. WR, winter run; SuR, summer run.

In general, broad patterns of steelhead abundance across the Puget Sound DPS are similar to those summarized in the last status review that considered data through 2009 (Ford *et al.* 2011). Smoothed trends in abundance indicate modest increases since 2009 for 13 of the 22 DIPs (Samish River and Bellingham Bay Tributaries WR, Pilchuck River WR, White River WR, Skokomish River WR, Strait of Juan de Fuca Tributaries WR, Skagit River WR, Green River WR, West Hood Canal Tributaries WR, and Nooksack River WR, with East Hood Canal Tributaries WR, Dungeness River WR, Elwha River WR, and Tolt River SuR also showing early signs of an upward trend). However, several of these upward trends are not statistically different from neutral, and most populations remain small. In 2005-2009, the geometric mean abundance was fewer than 250 adults for 8 of 19 populations evaluated (Table 59). One of these populations was in the Northern Cascades MPG, three (including Cedar River WR) were in the Central & South Puget Sound MPG, and four were in the Hood Canal & Strait of Juan de Fuca MPG. Eleven of the populations across the DPS had fewer than 500 spawners annually. During this period, the patterns for natural spawners were nearly identical.

In the intervening five years, some significant changes in abundance (total and natural) are evident for a few populations. Inspection of geometric means of total spawner abundance from 2010 to 2014 indicates that 9 of 20 populations evaluated had geometric mean abundances fewer than 250 adults and 12 of 20 had fewer than 500 adults (Table 59). The largest populations are Nooksack River WR, Samish River and Bellingham Bay Tributaries WR, Skagit River WR, Snohomish River/Skykomish River WR, and Snoqualmie River WR in the Northern Cascades MPG; Green River WR and White River WR in the Central & South Puget Sound MPG; and Skokomish River WR in the Hood Canal & Strait of Juan de Fuca MPG. The smallest populations (those with fewer than about 100 spawners annually) are Tolt River SuR in the Northern Cascades MPG; Cedar River WR and North Lake Washington and Lake Sammamish WR in the Central & South Puget Sound MPG; and Dungeness River WR, East Hood Canal Tributaries WR, Elwha River WR, Sequim and Discovery Bay Tributaries WR, and South Hood Canal Tributaries WR in the Hood Canal & Strait of Juan de Fuca MPG. Between the two most recent five-year periods (2005-2009 and 2010-2014), the geometric mean of estimated abundance increased by an average of 5.4%. For seven populations in the Northern Cascades MPG, the increase was 3%; for five populations in the Central & South Puget Sound MPG, the increase was 10%; and for six populations in the Hood Canal & Strait of Juan de Fuca MPG, the increase was 4.5% (Table 59).

Inspection of the change in geometric means of raw natural-origin spawner abundance over the same periods shows evidence of additional increases but there is information for only a few populations. For eight populations across the DPS, the geometric mean of estimated natural abundance has increased by an average of 21.6%. For three populations in the Northern Cascades MPG, the increase was 35.0%; for two populations in the Central & South Puget Sound MPG, the increase was 77.0%; and for three populations in the Hood Canal & Strait of Juan de Fuca MPG, the decrease was 21.6% (Table 59). Across the DPS, the trends in abundance have shown a pattern of initial increase, followed by a decade-long decline, and most recently have shown evidence of a slight increase for some populations. Several populations show continued declines. Over all the populations evaluated, the proportional changes in five-year geometric mean abundance of smoothed total spawners were a mix of positive and negative between the first two five-year periods (1990-1994 and 1995-1999)(Figure 106). Between the five-year periods 1995-1999 and 2005-2009, most populations showed appreciable declines in abundance, ranging between 10 and nearly 100% (Figure 106). Between the most recent two five-year periods (2005-2009 and 2010-2014), several populations showed increases in abundance between 10 and 100%, but about half have remained in decline (Figure 106). Natural spawners showed nearly identical patterns of change in abundance to the total

Table 59 -- 5-year geometric mean of raw natural spawner counts for Puget Sound steelhead. This is the raw total spawner count times the fraction natural estimate, if available. In parentheses, the 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners was available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values was used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right. MPG, major population group; WR, winter run; SuR, summer run.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Cedar R. WR	Central & South PS	(321)	(298)	(37)	(12)	(4)	(-67)
Green R. WR	Central & South PS	1566 (1730)	2379 (2505)	1618 (1693)	(716)	(552)	(-23)
Nisqually R. WR	Central & South PS	1201 (1208)	759 (759)	413 (413)	375 (375)	442 (442)	18 (18)
N. Lake WA/Lake Sammamish WR	Central & South PS	321 (321)	298 (298)	37 (37)	12 (12)		
Puyallup R./Carbon R. WR	Central & South PS	1860 (1954)	1523 (1660)	907 (1000)	641 (476)	(277)	(-42)
White R. WR	Central & South PS	696 (696)	519 (519)	466 (466)	225 (225)	531 (531)	136 (136)
Dungeness R. WR	Hood Canal & SJF	356 (356)		182 (186)		(141)	
East Hood Canal Tribs. WR	Hood Canal & SJF	110 (110)	176 (176)	202 (202)	62 (62)	60 (60)	-3 (-3)
Elwha R. WR	Hood Canal & SJF	206 (358)	127 (508)	(303)			
Sequim/Discovery Bays Tribs. WR	Hood Canal & SJF	(30)	(69)	(63)	(17)	(19)	(12)
Skokomish R. WR	Hood Canal & SJF	503 (385)	359 (359)	259 (205)	351 (351)	(580)	(65)
S. Hood Canal Tribs. WR	Hood Canal & SJF	89 (89)	111 (111)	103 (103)	113 (113)	64 (64)	-43 (-43)
Strait of Juan de Fuca Tribs. WR	Hood Canal & SJF		275 (275)	212 (212)	244 (244)	147 (147)	-40 (-40)
West Hood Canal Tribs. WR	Hood Canal & SJF		97 (97)	210 (210)	174 (149)	(74)	(-50)
Nooksack R. WR	Northern Cascades			(80)		1779 (1834)	
Pilchuck R. WR	Northern Cascades	1300 (1300)	1465 (1465)	604 (604)	597 (597)	614 (614)	3 (3)
Samish R./Bellingham Bay Tribs. WR	Northern Cascades	316 (316)	717 (717)	852 (852)	534 (534)	846 (846)	58 (58)
Skagit R. WR	Northern Cascades	7189 (7650)	7656 (8059)	5424 (5675)	5547 (4767)	(5123)	(7)
Snohomish/Skykomish R. WR	Northern Cascades	3634 (3877)	4141 (4382)	2562 (2711)	2945 (3084)	(930)	(-70)
Snoqualmie R. WR	Northern Cascades	1832 (2328)	2060 (2739)	856 (1544)	1396 (1249)	(680)	(-46)
Stillaguamish R. WR	Northern Cascades	1078 (1078)	1024 (1166)	401 (550)	259 (327)	(392)	(20)
Tolt R. SuR	Northern Cascades	112 (112)	212 (212)	119 (119)	73 (73)	105 (105)	44 (44)

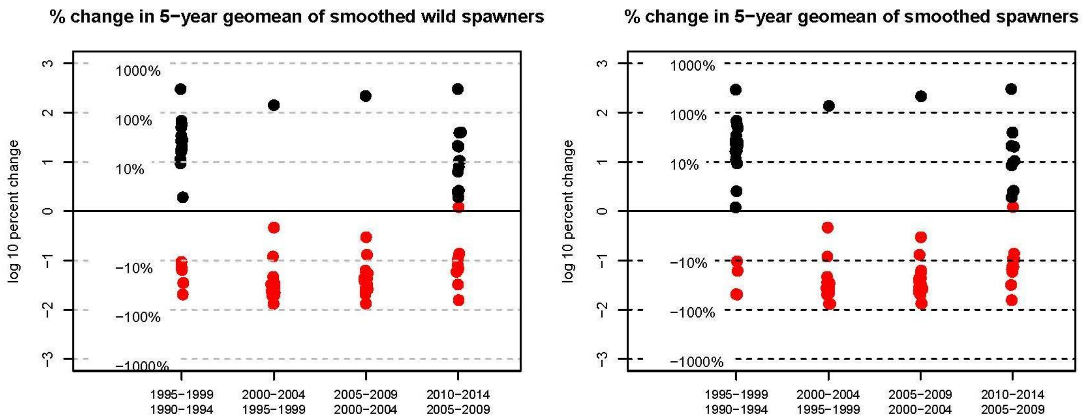

Figure 106 -- Percent changes (and log percent changes) in the geometric means of smoothed spawner abundance across Puget Sound steelhead populations between four consecutive five-year periods (1990-1994 to 1995-1999, 1995-1999 to 2000-2004, 2000-2004 to 2005-2009, and 2005-2009 to 2010-2014). Red dots indicate negative changes, and black dots indicate positive changes Left panel, total spawners; right panel, natural spawners.

spawners over these periods (Figure 106). Figure 107 illustrates the unimodal frequency distribution of these changes in geometric mean abundance for natural spawners; most of the changes across populations are in the range of -20% to +40% between consecutive periods. Long-term (15-year) trends in natural spawners are predominantly negative (Figure **108**).

Linear regressions of smoothed log natural spawner abundance applied to steelhead DIPs over two 15-year time series (16 DIPs in 1990-2005 and 8 DIPs in 1999-2014) indicate that declining slopes of trends in abundance are pervasive in the first period and common in the latter period (Table 60). For the 1990-2005 period, the average regression slope across the DPS was -0.04. For the seven populations in the Northern Cascades MPG, the average slope was -0.02; for the six populations in the Central & South Puget Sound MPG, the average slope was -0.03; and for the three populations in the Hood Canal & Strait of Juan de Fuca MPG, the average slope was -0.02. All but 6 of the 16 declining slopes were significantly ($P < 0.05$) less than zero. There was only one significantly ($p < 0.05$) positive trend (for Samish River/Bellingham Bay WR).

For the 1999-2014 period, the average regression slope across the DPS was -0.03. There was a little less variability in the abundance trends in the later period, but for both periods many trends were negative, and there were no significantly ($p < 0.05$) positive trends. For the three populations in the Northern Cascades MPG, the average slope was less than -0.01; for the two populations (White River WR and Nisqually River WR) illustrated in the Central & South Puget Sound MPG, the average slope was between -0.03 and -0.04; and for the three populations in the Hood Canal & Strait of Juan de Fuca MPG, the average slope was -0.04. Three of the seven declining trends were significantly less than zero. No population showed a significant change in slope between periods (Table 60)). A comparison with the analyses of abundance trends from the previous status review (Ford *et al.* 2011) shows no clear evidence that abundance is increasing; declining or neutral trends remain common across the

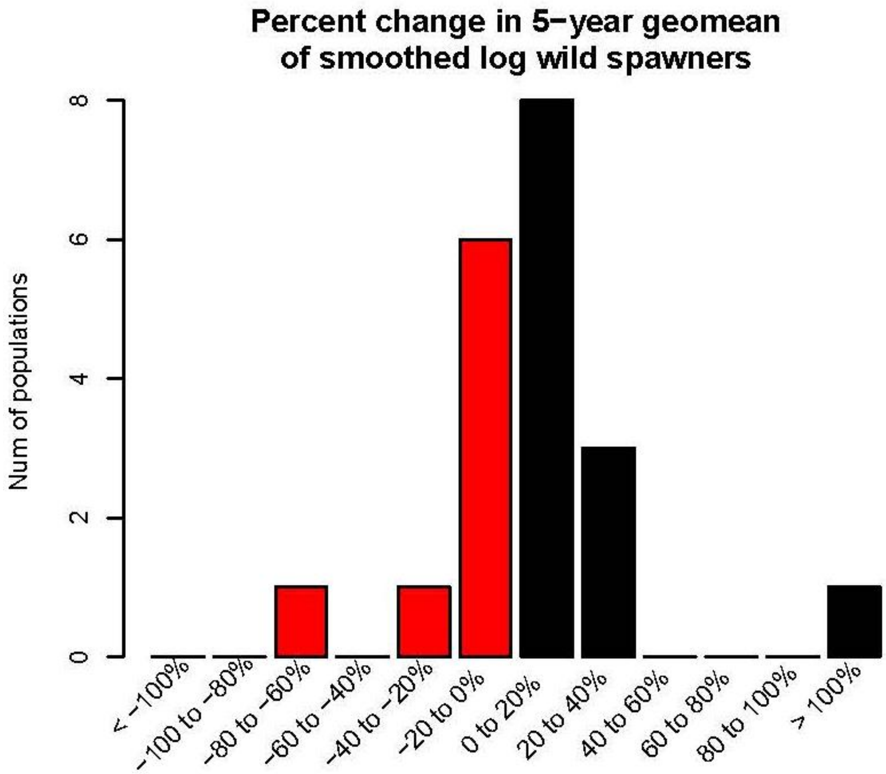

Figure 107 – Frequency distribution of percent changes in five-year geomeans of smoothed log natural steelhead spawners. Red bars indicate negative changes; black bars indicate positive changes.

DPS. Furthermore, in general, steelhead abundance across the DPS remains well below levels needed to sustain natural production into the future.

Steelhead productivity in Puget Sound has been temporally variable for most populations since the mid-1980s. Figure 109 depicts the trends in productivity, estimated as the log of smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance four years earlier, for 19 steelhead DIPs. Natural productivity measured this way is more or less equivalent to the intrinsic rate of natural increase, r, and it has been well below replacement for most of this period for at least eight of these DIPs. These include, in the Northern Cascades MPG: Stillaguamish River WR and Snoqualmie River WR (and, to a lesser extent, Skagit River WR and Green River WR); in the Central & South Puget Sound MPG: North Lake Washington and Lake Sammamish WR, Puyallup River/Carbon River WR, and Nisqually River WR; and in the Hood Canal & Strait of Juan de Fuca MPG: East Hood Canal Tributaries WR, Dungeness River WR, and Elwha River WR. For the other populations, productivity has fluctuated around replacement, but most have been predominantly below replacement since about 2000. That said, some populations are showing signs of productivity that has been above replacement since about 2009; these include Tolt River SuR and Pilchuck River WR (see also Nooksack River WR) (Northern Cascades MPG); Nisqually River WR and White River

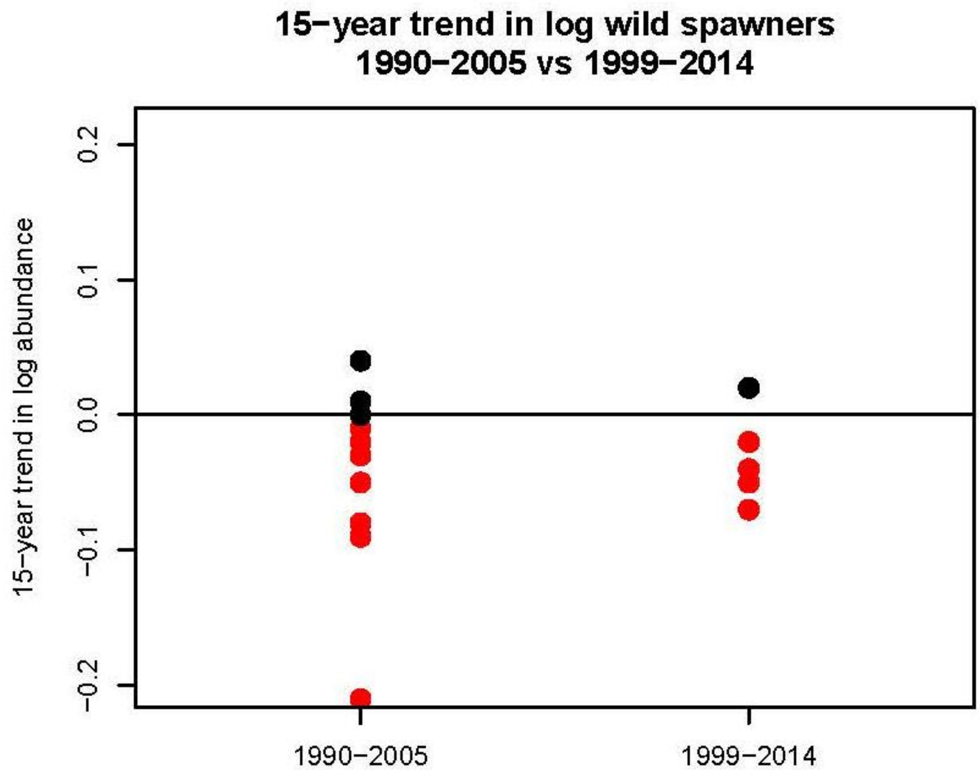

Figure 108 -- Plot of 15-year trend in log abundance of natural steelhead spawners across Puget Sound steelhead populations between two consecutive 15-year periods (1990-2005 and 1999-2014). Red dots indicate negative trends; black dots indicate positive trends.

WR (Central & South Puget Sound MPG); and East Hood Canal Tributaries WR, South Hood Canal Tributaries WR, and Strait of Juan de Fuca Tributaries WR (Hood Canal & Strait of Juan de Fuca MPG) (Figure 109).

Thus, there are some signs of modest improvement in steelhead productivity since the 2011 review, at least for some populations, especially in the Hood Canal & Strait of Juan de Fuca MPG. However, these modest changes must be sustained for a longer period (at least two generations) to lend sufficient confidence to any conclusion that productivity is improving over larger scales across the DPS. Moreover, several populations are still showing dismal productivity, especially those in the Central & South Puget Sound MPG, and two major DIPs in the Hood Canal & Strait of Juan de Fuca MPG—Dungeness River WR and Elwha River WR—are exhibiting this same pattern (Figure 109).

Collectively, there is no clear evidence to suggest that increases in abundance of spawners and improvements in productivity across the DPS since the last review was conducted are sufficient to support a change in conclusion about demographic risk to steelhead viability. The recent increases in abundance that have been observed for a few populations have been modest and within the range of variability observed in the past several years. Trends in abundance, especially for natural spawners, remain predominantly negative or flat over the time series examined. The recent upward estimates

Table 60 -- 15-year trends in log natural spawner abundance for Puget Sound steelhead, computed from a linear regression applied to the smoothed natural spawner log abundance estimate. Only populations with at least 4 natural spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and the last 5 years of the 15-year period. Lower and upper bounds of the 95% confidence intervals of the estimates are in parentheses. MPG, major population group; WR, winter run; SuR, summer run.

Population	MPG	1990-2005	1999-2014
Green R. WR	Central & South PS	-0.02 (-0.05, 0)	
Nisqually R. WR	Central & South PS	-0.08 (-0.09, -0.06)	-0.05 (-0.08, -0.03)
N. Lake WA/Lake Sammamish WR	Central & South PS	-0.21 (-0.24, -0.19)	
Puyallup R./Carbon R. WR	Central & South PS	-0.09 (-0.1, -0.07)	
White R. WR	Central & South PS	-0.03 (-0.05, -0.02)	-0.02 (-0.05, 0.01)
Dungeness R. WR	Hood Canal & SJF	-0.05 (-0.07, -0.04)	
East Hood Canal Tribs. WR	Hood Canal & SJF	-0.01 (-0.03, 0.01)	-0.07 (-0.1, -0.04)
Elwha R. WR	Hood Canal & SJF		
Skokomish R. WR	Hood Canal & SJF	-0.02 (-0.04, -0.01)	
S. Hood Canal Tribs. WR	Hood Canal & SJF	0 (-0.01, 0.02)	-0.02 (-0.05, 0)
Strait of Juan de Fuca Tribs. WR	Hood Canal & SJF		-0.04 (-0.06, -0.01)
West Hood Canal Tribs. WR	Hood Canal & SJF		
Pilchuck R. WR	Northern Cascades	-0.03 (-0.05, -0.01)	-0.02 (-0.05, 0)
Samish R./Bellingham Bay Tribs. WR	Northern Cascades	0.04 (0.02, 0.06)	0.02 (-0.01, 0.05)
Skagit R. WR	Northern Cascades	-0.02 (-0.04, 0)	
Snohomish/Skykomish R. WR	Northern Cascades	-0.03 (-0.05, -0.01)	
Snoqualmie R. WR	Northern Cascades	-0.05 (-0.08, -0.03)	
Stillaguamish R. WR	Northern Cascades	-0.08 (-0.11, -0.06)	
Tolt R. SuR	Northern Cascades	0.01 (-0.02, 0.03)	-0.02 (-0.05, 0.01)

of productivity are promising but are limited to a relatively few populations and span only one to a few years when smoothed. Thus, the improving patterns are neither widespread nor sustainable yet.

HARVEST

Puget Sound steelhead are harvested in terminal tribal gillnet fisheries and in recreational fisheries. Fisheries are directed at hatchery stocks, but some harvest of natural origin steelhead occurs incidentally to hatchery-directed fisheries. Winter-run hatchery steelhead production is primarily of Chambers Creek (Deschutes River) stock, which for several generations has been selected for earlier run timing than natural stocks to minimize fishery interactions. Hatchery production of summer-run steelhead is primarily of Skamania River (lower Columbia River Basin) stock, which has been selected for earlier spawn timing than natural summer-run steelhead to minimize interactions on the spawning grounds. In recreational fisheries, retention of wild steelhead is prohibited, so all harvest impacts occur as the result of release mortality and non-compliance. In tribal net fisheries, most fishery impacts occur in fisheries directed at salmon and hatchery steelhead. Some additional impacts occur in pre-terminal fisheries, but these are negligible and data are insufficient to attribute them to individual populations. Consequently, harvest impacts are reported as terminal harvest rates.

Most Puget Sound streams have insufficient catch and escapement data to calculate exploitation rates for natural steelhead. Populations with sufficient data include those in the Skagit, Green, Nisqually,

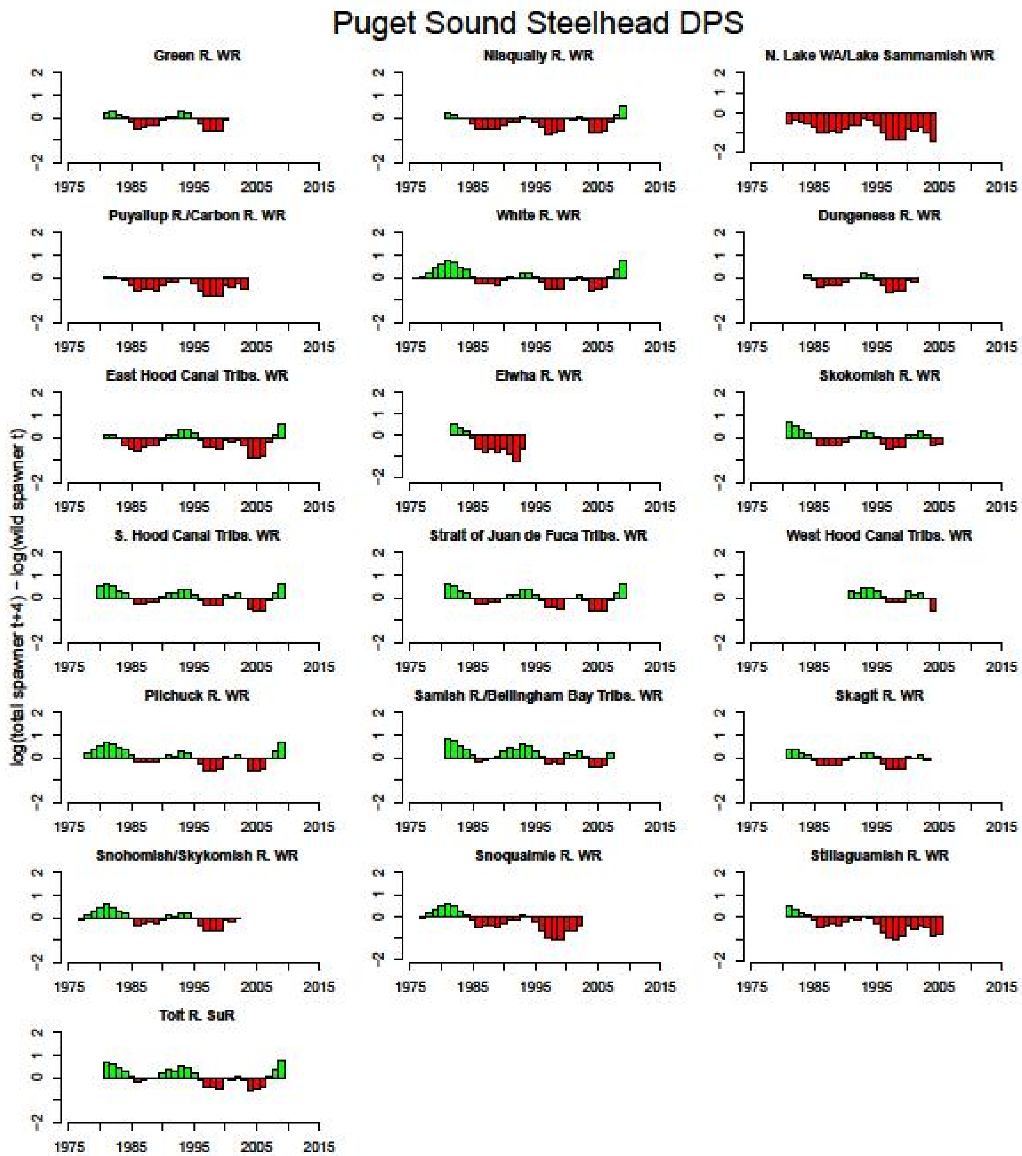

Figure 109 -- Trends in population productivity of Puget Sound steelhead, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year (t – 4). WR, winter run; SuR, summer run.

Puyallup, and Snohomish rivers (Figure **110**). Harvest rates differ widely among the different rivers, but all have declined since the 1970s and 1980s. Harvest rates on natural steelhead during the earlier period averaged between 10% and 40%, with some populations in the central and south parts of Puget Sound, such as the Green and Nisqually river populations, experiencing harvest rates over 60%. In recent years, terminal harvest rates have continued to decline, averaging less than 2% over the last five years. These rates are expected to continue to decline for the near future.

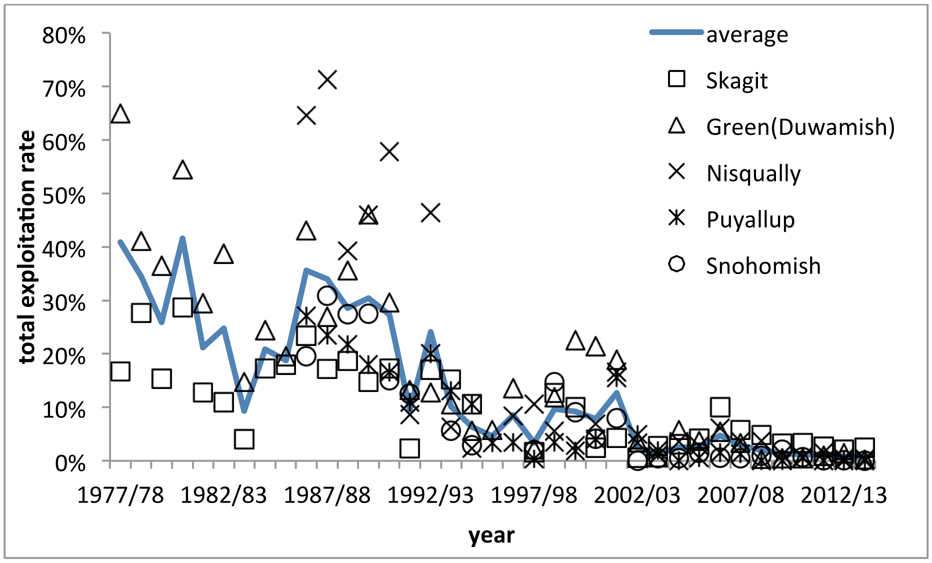

Figure 110 -- Terminal harvest rates on natural steelhead from Puget Sound rivers. Data from the Puget Sound Steelhead Harvest Management Plan, Appendix A (Bob Leland, Washington Department of Fish and Wildlife, personal communication).

STOCHASTIC POPULATION RISK ANALYSES

In addition to the above evaluations of steelhead abundance and productivity in Puget Sound, we conducted two additional analyses of the time series of demographic data available for steelhead populations in the DPS. To assess the uncertainty in future projections of reaching specific low-abundance thresholds for Puget Sound steelhead, we conducted stochastic simulations of quasi-extinction risk for several of the abundance time series given the latest estimates of population growth rate and the Puget Sound Steelhead TRT's estimates of habitat carrying capacity. Most of the time series extended from the early 1980s to about 2013, but some were from the late 1970s to 2014. Those time series with fewer than 15 abundance estimates (e.g., Nooksack River WR) were excluded. The QETs applied here were estimated by the TRT (Hard *et al*. 2015) and are based on a low average of 24 spawners measured over four consecutive years for Snow Creek steelhead, then scaling by the ratio of the estimate of intrinsic potential for the watershed supporting the candidate DIP to that of Snow Creek. The Snow Creek winter-run steelhead population was chosen by the TRT because it is a natural anadromous population with sustained natural production in a relatively stable watershed, and provides accurate estimates of adult escapement, smolt production, and intrinsic potential to serve as a basis for estimating QETs throughout the Puget Sound Steelhead DPS. Table 61 summarizes the output from these simulations. Consistent with the other analyses of abundance and productivity, these simulations provide evidence of widespread declining steelhead productivity in Puget Sound. The average of the geometric mean of the intrinsic rate of increase, r, for

Table 61 -- Summary of stochastic simulations of quasi-extinction risk for Puget Sound steelhead populations. r, geometric mean of the intrinsic rate of natural increase (SD, standard deviation); QET, estimated quasi-extinction threshold abundance; yrs to QET, median no. of years that simulated populations take to reach QET; % below QET, percentage of 1000 simulated populations that drop below QET within 100 years under two growth models (exp,

exponential growth model; d-d, density-dependent growth model with K equal to the capacity abundance identified by Hard *et al.* (2015). A time series of 15 years of abundance was the minimum threshold for these simulations (e.g., Nooksack River WR was therefore not included, as n = 7). South Puget Sound Tributaries WR had no escapement data; therefore, this population is not included.

Population	r	SD(r)	QET	yrs to reach QET	% below QET exp	% below QET d-d
Cedar R. WR	-0.172	0.883	35	17	95.2	97.0
Dungeness R. WR	-0.072	0.392	30	29	88.6	99.1
East Hood Canal Tribs. WR	-0.012	0.805	27	17	42.1	75.5
Elwha R. WR	0.086	1.070	41	7	16.3	22.5
Green R. WR	-0.015	0.434	69	34	39.7	74.9
Nisqually R. WR	-0.032	0.740	55	29	46.4	69.8
North Lake WA and Lake Sammamish WR	-0.257	0.776	36	10	99.8	99.9
Pilchuck R. WR	0.026	0.453	34	35	12.0	87.6
Puyallup R./Carbon R. WR	-0.071	0.457	58	39	77.0	95.4
Samish R. and Bellingham Bay Tribs. WR	0.063	0.747	31	13	12.5	64.4
Sequim/Discovery Bays Tribs. WR	-0.046	0.738	25	10	68.4	90.9
Skagit R. WR	0.002	0.364	157	51	14.1	67.4
Skokomish R. WR	0.008	0.617	50	29	29.2	45.4
Snohomish/Skykomish R. WR	-0.012	0.477	73	35	36.4	87.0
Snoqualmie R. WR	-0.024	0.484	58	35	43.6	79.5
South Hood Canal Tribs. WR	-0.027	0.431	30	18	60.1	76.2
Strait of Juan de Fuca Tribs. WR	-0.039	0.497	26	25	63.0	100.0
Stillaguamish R. WR	-0.046	0.614	67	29	60.5	73.1
Tolt R. SuR	-0.004	0.584	25	18	41.3	100.0
West Hood Canal Tribs. WR	-0.026	0.587	32	9	58.0	74.1
White R. WR	0.024	0.467	64	16	20.1	32.0

21 populations across the DPS is -0.031. The evidence for decline is more pervasive for the populations in the South and Central Puget Sound MPG, where the average estimate of r was -0.074; r for populations in the Northern Cascades MPG averaged about 0.001, and those in the Hood Canal & Strait of Juan de Fuca MPG averaged -0.019. These estimates all had wide variability, however, and estimates for individual populations were generally not distinguishable from zero.

One thousand simulations of each population were run to determine the likelihood that each simulated population would reach its specified quasi-extinction threshold (QET, as identified in Hard *et al.* (2015)). All the simulated populations reached their corresponding QET levels of abundance within a median of 51 years, given current demographic trends and under an assumption of stationarity. The average of the median number of years to reach QET over all 21 populations was 24.0 years. For populations in the Northern Cascades MPG, the average of the median number of years to QET was 30.8 years; for those in the Central & South Puget Sound MPG, the average was 24.8 years; and for the Hood Canal & Strait of Juan de Fuca MPG, the average was 16.4 years. All but six populations reached QET within a median of 30 years; the exceptions were Green River WR (median, 34 years), Pilchuck River WR (median, 35 years), Puyallup/Carbon River WR (median, 39 years), Skagit River WR (median, 51 years), Snohomish/Skykomish River WR (median, 35 years), and Snoqualmie River WR (median, 35 years).

Applying two different population growth models to the simulations indicates that, for most of these populations, the majority of simulated trajectories reached QET within 100 years (the VSP criterion for 95% probability of persistence). One model was a simple density-independent model of exponential population growth fitted to the data; the other was a model that incorporated density dependence through a simple damping function that was applied to the population growth rate as abundance (N) approached estimated carrying capacity, K, through the function $1-(N_{t-1}/K)$, where t is the current time step (year). Both models were implemented with the 'popbio' package in R (R Core Team 2012). K was given as the abundance computed from intrinsic potential estimates by the Puget Sound Steelhead TRT (Hard *et al.* 2015).

The results of these simulations indicate that under both models, most simulated populations would reach their specified QET abundances within 100 years, and generally within 30 years. Under the density-independent growth model, the proportion of simulated populations that reached QET varied from 12.0% to 99.8%; the mean proportion across the 21 populations was 48.8%. The average proportions of populations reaching QET in each of the three MPGs were 31.5% for populations in the Northern Cascades MPG, 58.2% for populations in the Central & South Puget Sound MPG, and 56.6% for populations in the Hood Canal & Strait of Juan de Fuca MPG. Under the exponential growth model, the populations across the DPS with the lowest proportions reaching QET were Samish River and Bellingham Bay Tributaries WR, Pilchuck River WR, and Skagit River WR in the Northern Cascades MPG, White River WR in the Central & South Puget Sound MPG, and Elwha River WR in the Hood Canal & Strait of Juan de Fuca MPG; all had proportions reaching QET lower than about 20%. Populations with the highest proportions reaching QET were North Lake Washington and Lake Sammamish WR and Cedar River WR in the Central & South Puget Sound MPG and Dungeness River WR in the Hood Canal & Strait of Juan de Fuca MPG, all of which had proportions exceeding 80% (Table 61).

Under the density-dependent model, the proportions of each of the populations reaching QET were generally higher, and sometimes considerably higher. The proportions ranged from a low of 22.5% to a high of 100%, and the mean proportion for all 21 populations was 76.7%. The average proportions of populations reaching QET in each of the three MPGs were 79.8% for populations in the Northern Cascades MPG, 73.5% for populations in the Central & South Puget Sound MPG, and 76.9% for populations in the Hood Canal & Strait of Juan de Fuca MPG. Under the density-dependent growth model, the populations across the DPS with the lowest proportions reaching QET were White River WR in the Central & South Puget Sound MPG and Elwha River WR in the Hood Canal & Strait of Juan de Fuca MPG; those with the highest proportions were Cedar River WR and North Lake Washington and Lake Sammamish WR in the Central & South Puget Sound MPG, Dungeness River WR in the Hood Canal & Strait of Juan de Fuca MPG, and Tolt River SuR in the Northern Cascades MPG (Table 61). These analyses support the earlier patterns of demographic trend and indicate that many steelhead populations throughout the DPS have a high risk of declining to levels of abundance at or below their estimated QETs.

Where possible we also applied auto-regressive state space models ('MARSS' package in R; (Holmes *et al.* 2014)) to the abundance time series for Puget Sound steelhead DIPs, to predict probabilities of population abundance reaching particular abundance thresholds (QETs) and to quantify forecast uncertainty for projected future change in abundance over a range of time intervals into the future, again based on observed trends and assuming stationarity of conditions. Point estimates of probability that population abundance would decline to a specified QET level within 100 years ranged from near 0 for Skagit River WR (Northern Cascades MPG) to near 0.9-1.0 for Stillaguamish River WR and Snoqualmie River WR (Northern Cascades MPG); Puyallup River/Carbon River WR (Central & South Puget Sound MPG); and South Hood Canal Tributaries WR and Strait of Juan de Fuca

Tributaries WR (Hood Canal & Strait of Juan de Fuca MPG). Cedar River WR and North Lake Washington and Lake Sammamish WR (Central & South Puget Sound MPG) are already below their QET abundances. However, where estimable the 95% confidence intervals around these estimates were generally wide over the 100-year time horizon. Exceptions to this pattern included Stillaguamish River WR, Snoqualmie River WR, Puyallup/Carbon River WR, and Nisqually River WR, where confidence intervals are narrower and abundances are expected to fall to QET levels in the near future (within a few decades).

One way to illustrate these patterns systematically is through estimates of abundance forecast uncertainty. Figure 111 depicts graphs of these estimates for 19 steelhead populations in Puget Sound. These uncertainty "envelopes" depict regions of high certainty and uncertainty surrounding the population forecasts. The black and white areas represent parameter spaces where rates of population decline over specific time periods are estimated with 95% or higher confidence, with the white region representing rates of decline that are not likely to be exceeded (maximal) and the black region representing minimal expected rates. In the white region, the probability of a specified population decline is ≤ 5%. In the black region, the probability of a specified population decline is ≥ 95%. The grey regions define less certain areas of parameter space between these extremes, with the dark grey region representing the region of highest uncertainty.

Several patterns are evident in these forecast uncertainty envelopes. First, the variability in abundance time series for most populations yields large areas of uncertainty in forecasting future abundance, especially over VSP time scales. Second, some populations may remain stable or have the potential to increase in abundance—as indicated by the lack of a black wedge—at least over the short term. However, it is important to note that while this outcome is feasible, it is generally not likely. Exceptions which we can be confident will not increase in abundance include Snoqualmie River WR and Stillaguamish River WR in the Northern Cascades MPG; Nisqually River WR and Puyallup River/Carbon River WR in the Central & South Puget Sound MPG; and South Hood Canal Tributaries WR and Strait of Juan de Fuca Tributaries WR in the Hood Canal & Strait of Juan de Fuca MPG. Both the Cedar River WR and North Lake Washington and Lake Sammamish WR (Central & South Puget Sound MPG; not shown in Figure 111) also fall into this category, as they are both currently far below their QETs. There are no escapement data for South Puget Sound Tributaries WR, but the trend in catch data (not shown) also suggests that future abundance is expected to remain low there.

Third, high rates of decline are expected over the short term for several of the populations identified above. Even for those populations with large uncertainties associated with abundance forecasts, maximal rates of decline over the short term are expected to be quite high. To take one example, the Pilchuck River WR shows a large area of forecast uncertainty (Figure 111), and it is possible that this population could even increase in abundance (see also Figure 109 and Figure 105). Nevertheless, the rate of decline in abundance that we can be confident ($P > 0.95$) this population will not exceed is still about 99% in less than 20 years. By comparison, the Puyallup River/Carbon River WR population is expected to decline by about 90% in 20 years as well (and 99% in 45 years), but because of the lower variability in the data we can also be confident that this population will decline by nearly that rate within 25-30 years.

When combined with the previous demographic analyses, inspection of forecast uncertainty across the DPS indicates that, for most populations, we remain highly uncertain in predicting whether past trends will continue. However, the predominance of declining trends and the prospect that some

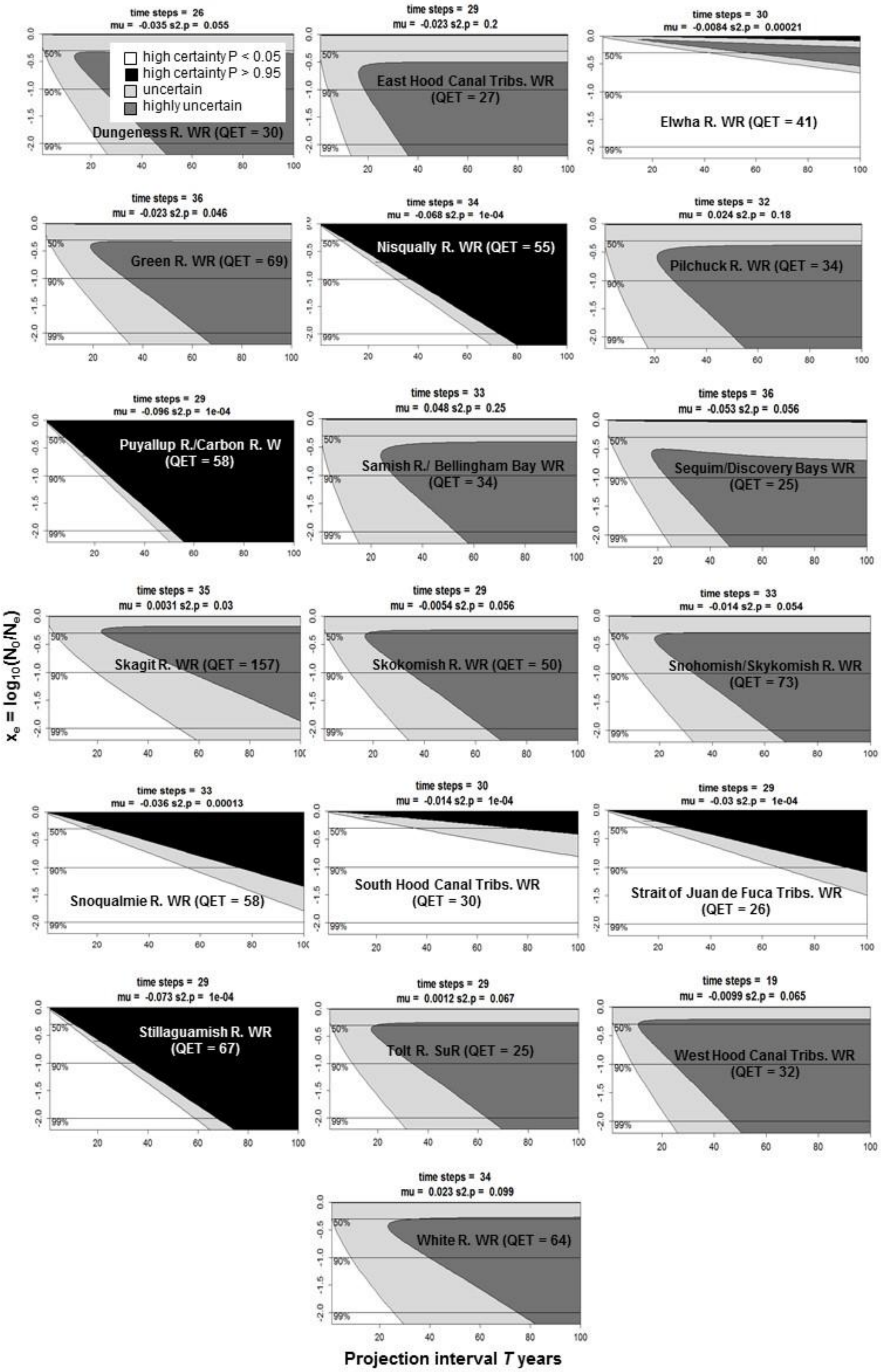
time steps = 26
mu = -0.035 s2.p = 0.055
high certainty P < 0.05
high certainty P > 0.95
uncertain
highly uncertain
Dungeness R. WR (QET = 30)
time steps = 29
mu = -0.023 s2.p = 0.2
East Hood Canal Tribs. WR
(QET = 27)
time steps = 30
mu = -0.0084 s2.p = 0.00021
Elwha R. WR (QET = 41)
time steps = 36
mu = -0.023 s2.p = 0.046
Green R. WR (QET = 69)
time steps = 34
mu = -0.068 s2.p = 1e-04
Nisqually R. WR (QET = 55)
time steps = 32
mu = 0.024 s2.p = 0.18
Pilchuck R. WR (QET = 34)
time steps = 29
mu = -0.096 s2.p = 1e-04
Puyallup R./Carbon R. W
(QET = 58)
time steps = 33
mu = 0.048 s2.p = 0.25
Samish R./ Bellingham Bay WR
(QET = 34)
time steps = 36
mu = -0.053 s2.p = 0.056
Sequim/Discovery Bays WR
(QET = 25)
time steps = 35
mu = 0.0031 s2.p = 0.03
Skagit R. WR (QET = 157)
time steps = 29
mu = -0.0054 s2.p = 0.056
Skokomish R. WR (QET = 50)
time steps = 33
mu = -0.014 s2.p = 0.054
Snohomish/Skykomish R. WR
(QET = 73)
time steps = 33
mu = -0.036 s2.p = 0.00013
Snoqualmie R. WR (QET = 58)
time steps = 30
mu = -0.014 s2.p = 1e-04
South Hood Canal Tribs. WR
(QET = 30)
time steps = 29
mu = -0.03 s2.p = 1e-04
Strait of Juan de Fuca Tribs. WR
(QET = 26)
time steps = 29
mu = -0.073 s2.p = 1e-04
Stillaguamish R. WR
(QET = 67)
time steps = 29
mu = 0.0012 s2.p = 0.067
Tolt R. SuR (QET = 25)
time steps = 19
mu = -0.0099 s2.p = 0.065
West Hood Canal Tribs. WR
(QET = 32)
time steps = 34
mu = 0.023 s2.p = 0.099
White R. WR (QET = 64)
50%
90%
99%
0.0
-0.5
-1.0
-1.5
-2.0
20
40
60
80
100
$x_e = \log_{10}(N_0/N_e)$
Projection interval *T* years

Figure 111 -- Graphs of forecast uncertainty for 19 Puget Sound steelhead populations showing the projected future changes in abundance as a function of time projection in years. Each population's estimate of quasi-extinction threshold (QET) is given; mu is the mean annual change in abundance and s2.p is the process error variance. The ordinate axis is the log ratio of the initial abundance at the start of the forecast to the expected abundance at the end of the projected interval in years. The black and white areas represent parameter spaces where rates of population decline over specific time periods are estimated with 95% or higher confidence, with the white area reflecting maximal expected rates of decline and the black area reflecting minimal expected rates. The gray areas encompass the uncertainty envelope for estimating extinction risk (P < 0.95). Not all populations are amenable to these analyses because of lack of abundance data; additional populations not considered here include Cedar River WR and North Lake WA and Lake Sammamish WR (where current anadromous abundance for each is already well below QET) and South Puget Sound Tributaries WR (for which only catch data are available).

declines could be as steep as 90% from current abundance levels within a decade or two provide little confidence in expecting population viabilities to improve substantially in the short term.

For another assessment, the Puget Sound Steelhead TRT (Hard *et al.* 2015) used a Bayesian Network framework it developed for all four VSP criteria to develop assessments of the influence of abundance and productivity on population viability and then built hierarchical networks to scale up their evaluation of these criteria at the MPG and DPS levels. The DIP-level Bayesian Network for a representative population of winter-run steelhead in the DPS is given in Figure 112. It is informative to compare the TRT's assessment to the more current demographic information for Puget Sound steelhead available here. The Puget Sound Steelhead TRT recently completed its evaluation of several factors that influence the abundance and productivity VSP criteria for steelhead in this DPS (Hard *et al*. 2015). For population abundance, these factors included estimated adult abundance relative to adult capacity, estimated juvenile abundance relative to juvenile capacity, and the probability the population would reach its quasi-extinction threshold abundance within 100 years. For productivity, these factors included the probability the population's mean growth rate was <1 (ideally calculated from a population viability analysis), based on an estimate of smolts per spawner (a measure of productivity in freshwater) and an estimate of adults per smolt (a measure of marine survival); and the estimated frequency of repeat spawners. These factors were discretized into distinct bins by the Bayesian Network model (which more easily permitted integration of quantitative and qualitative information, including expert opinion from TRT members).

Through its analyses of steelhead viability with Bayesian Networks, the TRT found that viability was generally highest for populations in the Northern Cascades MPG. When considered together, most DIPs in the DPS exhibited relatively low probabilities of viability with respect to abundance and productivity (40–50%). Population viabilities with respect to abundance and productivity were higher for a few Northern Cascades DIPs (probabilities of viability approaching 55–60%). Viabilities were generally lower in the Central & South Puget Sound and the Hood Canal and the Strait of Juan de Fuca MPGs; they were lowest in the Central & South Puget Sound MPG, where most populations are at low abundance. The TRT found that the Puyallup River/Carbon River WR DIP had the highest viability in this DPS, based on data available through 2011. Across the DPS, the decline in DIP viability with declining abundance and productivity was nonlinear, and viability was more sensitive to productivity than to abundance (Hard *et al.* 2015).

A sensitivity analysis of the Bayesian Network models found that population viability was generally lowest when abundance and productivity were limiting (probabilities of viability <~30%). This analysis employed a metric called entropy reduction to assess this; entropy reduction is a measure of the variability in model output explained by a model component. In general, most components of the

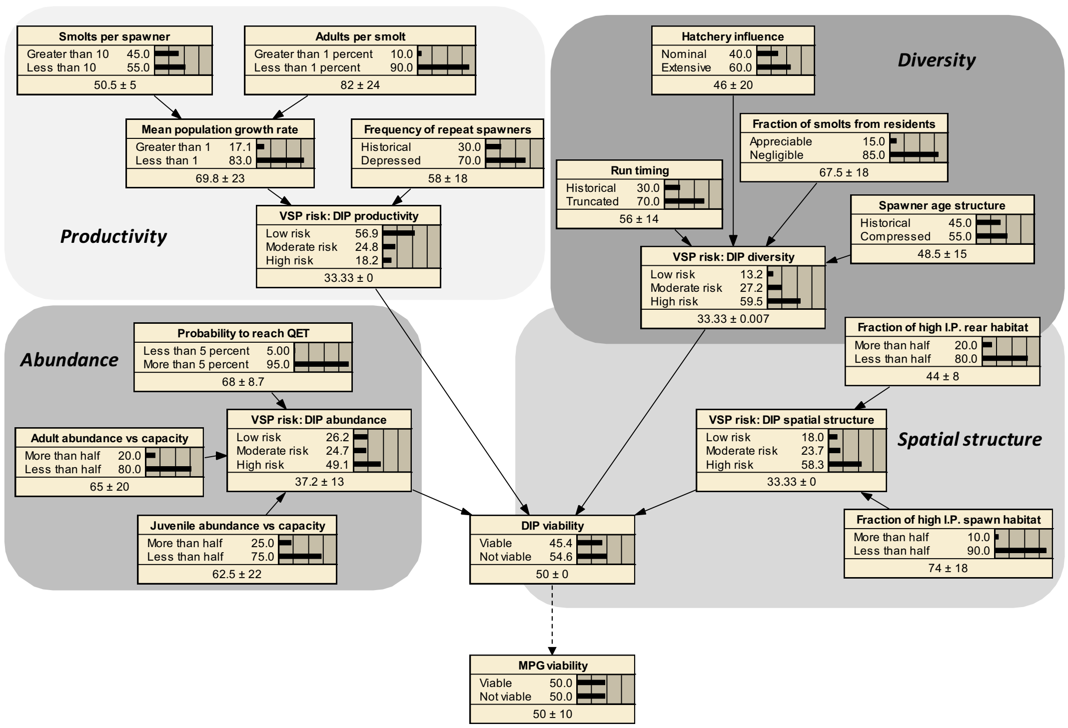

Figure 112 -- A Bayesian Network to characterize the viability of a representative DIP of winter-run steelhead in the Puget Sound DPS. This was the common framework adopted by the Puget Sound Steelhead TRT. The influence of the DIP's abundance on its viability is represented by the "VSP risk: DIP abundance" node (lower left) that incorporates influences of adult abundance, juvenile abundance, and quasi-extinction risk on DIP viability. The influence of the DIP's productivity on its viability is represented by the "VSP risk: DIP productivity" node (upper left) that incorporates influences of population growth rate and frequency of repeat spawning on DIP viability; the node for population growth rate is itself influenced by freshwater survival (smolts per spawner) and marine survival (adults per smolt). The influence of the DIP's diversity on its viability is represented by the "VSP risk: DIP diversity" node (upper right) that incorporates influences of the distribution of run timing, influence of hatchery fish on natural diversity, the adult age distribution, and the proportion of migrant smolts produced by resident adults. The influence of the DIP's spatial structure on its viability is represented by the "VSP risk: DIP spatial structure" node (lower right) that incorporates influences of the fraction of IP habitat occupied by rearing juveniles and the fraction of IP habitat occupied by spawning adults. This network is a subnetwork that then determines the viability of the DIP's corresponding MPG, as indicated by the dashed arrow at bottom; similar subnetworks for the viability of each DIP in the MPG combine to influence the MPG's overall viability. The MPG networks then combine to estimate the overall viability of the DPS. A description of the nodes, the underlying probabilities of viability and a summary of how these were derived for this subnetwork are given in the Puget Sound Steelhead TRT's viability criteria document (Hard *et al.* 2015).

models explained little variation individually, reflecting the high uncertainty associated with most individual effects on viability for specific populations. As one example, for the Samish River and Bellingham Bay Tributaries WR DIP in the Northern Cascades MPG, the sensitivity of viability to abundance (entropy reduction, 6.7%), productivity (entropy reduction, 5.2%), diversity (entropy reduction, 1.3%, primarily a result of hatchery fish influence), and spatial structure (entropy reduction, 1.6%) indicated that its viability is limited more by abundance and productivity than by diversity and spatial structure. For the abundance criterion, the most important contributing factor was adult abundance (entropy reduction, 2.1%), followed by juvenile abundance (entropy reduction, 0.1%) and by the probability of abundance dropping below the specified QET of 31 fish (entropy reduction, < 0.1%). For the productivity criterion, the most important contributing factor was population growth rate (entropy reduction, 2.0%), followed by iteroparity (entropy reduction, 0.3%). Other factors contributing to population viability included marine survival rate (entropy reduction, 1.2%), spatial structure (total entropy reduction for spawning and rearing area occupied, 0.5%), and altered spawn timing (entropy reduction, 0.3%). Hatchery influence and alteration of age structure were considered to have minor influences on viability. The Most Probable Explanation (MPE) for the viability of this DIP was that it is not viable, but there was considerable uncertainty around this MPE: the probability that this DIP could be viable was estimated at 69.2% (Hard *et al.* 2015).

A contrasting example is provided by the Nisqually River WR DIP in the Central & South Puget Sound MPG. The sensitivity of its viability to abundance (entropy reduction, 6.0%), productivity (entropy reduction, 6.9%), diversity (entropy reduction, 1.3%), and spatial structure (entropy reduction, 1.7%) indicated that viability of steelhead in the Nisqually River is limited more by productivity. For the abundance criterion, the most important contributing factor was adult abundance (entropy reduction, 1.2%), followed by the probability of reaching the specified QET (entropy reduction, 0.2%) and then by juvenile abundance (entropy reduction, < 0.1%). For the productivity criterion, the most important contributing factor was population growth rate (entropy reduction, 2.8%), followed by repeat spawning (entropy reduction, 1.4%). Other factors contributing to viability included marine survival rate (entropy reduction, 1.5%), spawn timing (entropy reduction, 0.3%), and spatial structure criteria (total entropy reduction for spawning and rearing area occupied, 0.5%). Hatchery influence and alteration of age structure had minor influences on viability. The MPE for the viability of this DIP was that it is not viable, and there was little uncertainty around the MPE, because the probability that it might be viable was estimated at only 11.9%.

In this case, while the explanatory power of individual components of viability was also low, the collective power of the combined components to explain variation in viability was higher for Nisqually River WR steelhead than for Samish River and Bellingham Bay Tributaries WR steelhead. The primary reason for this was the lower variability around the declining trend in abundance and the greater precision of the low productivity estimate for the former population.

A simplified representation of the results of the Bayesian Network analyses, using an 18-point scale for viability and illustrated in a “stop light” framework, shows that about half (8 of 16) of the DIPs in the Northern Cascades MPG had moderate to high viability scores for abundance, and about half had low scores (Figure 113). In the Central & South Puget Sound MPG, all eight DIPs scored low for abundance. Similarly, in the Hood Canal & Strait of Juan de Fuca MPG, most (6 of 8) DIPs scored low for abundance (Dungeness River WR and Strait of Juan de Fuca Tributaries WR were the exceptions). Nearly all of the 16 DIPs in the Northern Cascades MPG had moderate viability scores for productivity (Stillaguamish River WR was the sole exception with a low score). In the Central & South Puget Sound MPG, 6 of 8 DIPs scored low for productivity, with Green River WR and White River WR receiving moderate scores. In the Hood Canal & Strait of Juan de Fuca MPG, most (5 of 8) DIPs also scored low for productivity, with the exceptions being East Hood Canal Tributaries WR and Skokomish River WR (moderate viability scores) and West Hood Canal Tributaries WR (high viability score).

The TRT’s conclusions are largely concordant with the analyses presented in this report and based on updated data. There is no clear evidence to suggest that patterns of abundance, or trends in

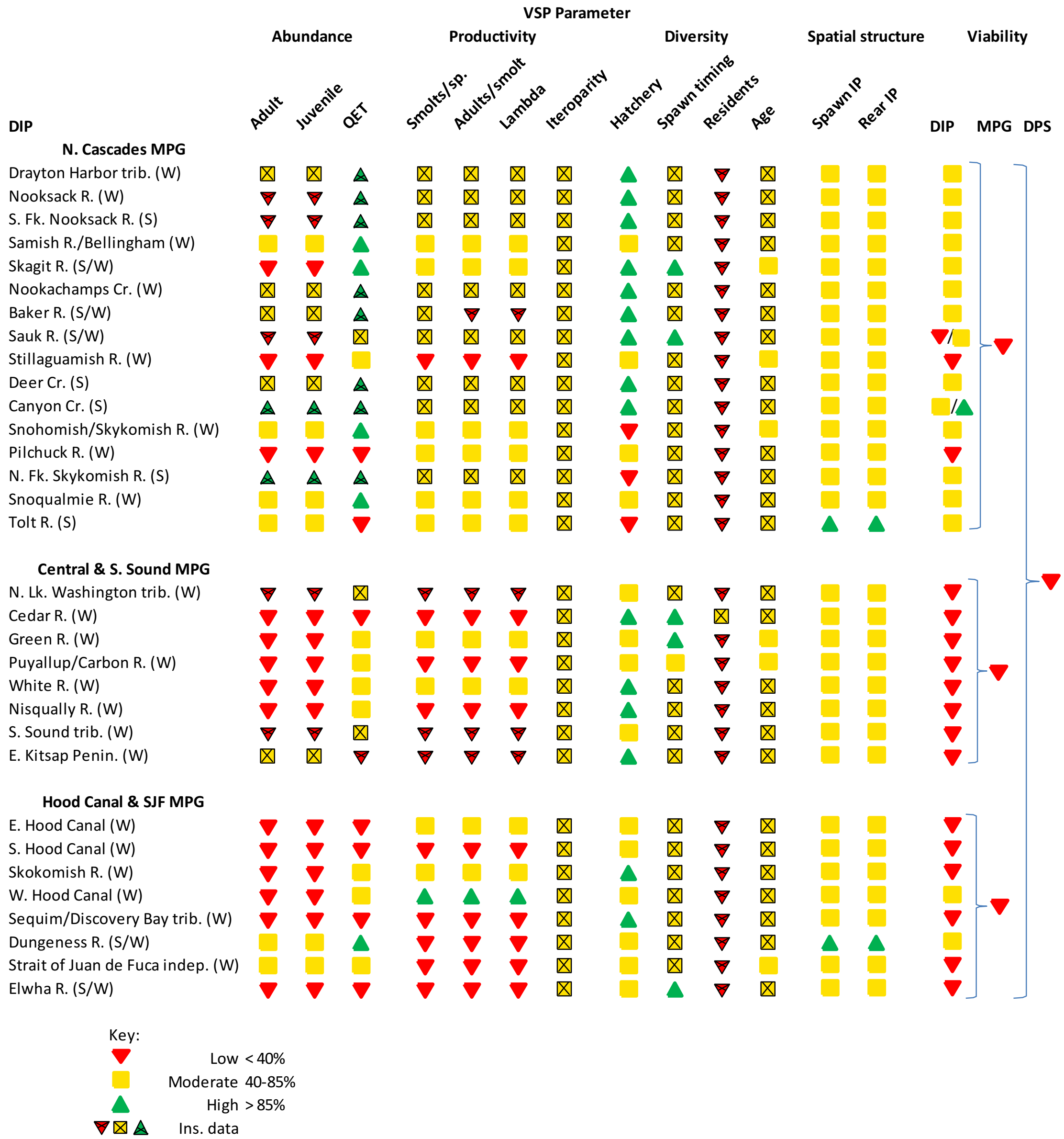

Figure 113 -- Estimates of current viability (low = not viable, moderate = intermediate, high = viable) for the 32 DIPs of Puget Sound steelhead (S = summer run and W = winter run) using the VSP framework developed by the Puget Sound Steelhead TRT. Note that many criteria are supported by insufficient data and in most (but not all) of those cases they were given an intermediate value with respect to influence on viability. See Hard *et al.* (2015) for details.

abundance or productivity, for Puget Sound steelhead have changed appreciably since the TRT's assessment of these factors among populations across the DPS.

SPATIAL STRUCTURE AND DIVERSITY

Abundance and productivity are demographic characteristics of a population that determine its ability to persist into the future. Spatial structure and diversity, the other two VSP parameters, are characteristics that influence a population's ability to sustain its identifying characteristics—such as utilization of habitat, distribution of spawning aggregations, genetic and phenotypic traits, life-history characteristics such as growth rate, frequency and phenology of reproduction (seasonal run and spawn timing), and age structure. Demographic risks due to low abundance and productivity are typically shorter-term considerations for viability. Compromised spatial structure and diversity are often thought to pose risks to viability over the longer-term, so long as short-term demographic risks do not threaten viability.

Since the Puget Sound Steelhead TRT completed its review using data available through 2011, the only new data on spatial structure and diversity that have become available have been estimates of the fraction of hatchery fish on spawning grounds in the most recent years. In this section we first evaluate this information. We then summarize the TRT's more general findings on spatial structure and diversity for Puget Sound steelhead.

Production and release of hatchery steelhead of both run types in Puget Sound has declined in recent years for most areas, and estimates of the fraction of hatchery steelhead spawning natural are low for many rivers. The populations with the highest estimated proportions of hatchery spawners are winter-run populations and include Elwha River WR, Snoqualmie River WR, and Stillaguamish River WR. However, these estimates for the Elwha River WR ceased by the late 1990s. For all populations except the Snoqualmie River WR, the estimated fractions of natural-origin spawners have been higher than 0.7 since 2005 (Table 62). However, it is important to note that the estimate of this fraction is not available for a considerable number of steelhead populations—a dozen in the latest period. For 17 DIPs across the DPS, the five-year average for fraction of natural-origin steelhead spawners exceeded 0.75 during the 2005-2009 period; during the 2010-2014 period, this average was near 1.0 for the 9 populations that could provide an estimate. These values are similar to those estimated for earlier five-year periods back to 1990-1994 (Table 62). The fraction of first-generation hatchery fish on steelhead spawning grounds, which has declined in recent years, is similar to the average since the last status review. The frequency distributions of the fraction of natural fish among steelhead spawners in Puget Sound indicate that this fraction is generally 0.9 or greater for both time periods, but these distributions also show that this fraction could not be estimated for several populations, especially in the 2010-2014 period.

The TRT concluded that production of hatchery fish of both run types—winter run and summer run—has posed considerable risk to diversity in natural steelhead in the Puget Sound DPS. Because of the origin and aspects of the propagation history of these fish in Puget Sound, the TRT considered continued hatchery production of steelhead there to represent a major threat to the diversity VSP component for the DPS. Winter-run fish produced in hatcheries across the DPS are derived from the Chambers Creek stock in southern Puget Sound, which has been selected repeatedly for early spawn timing for decades, a trait known to be heritable in salmonids (the natural population is now extinct); summer-run hatchery fish are derived from the Skamania River stock in the lower Columbia River Basin (i.e., out-of-DPS origin). That said, the Washington Department of Fish and Wildlife has terminated several early-winter steelhead hatchery programs (Skagit River and Soos Creek), reduced smolt release numbers substantially from several others (e.g., Tokul Creek and Wallace Hatchery/

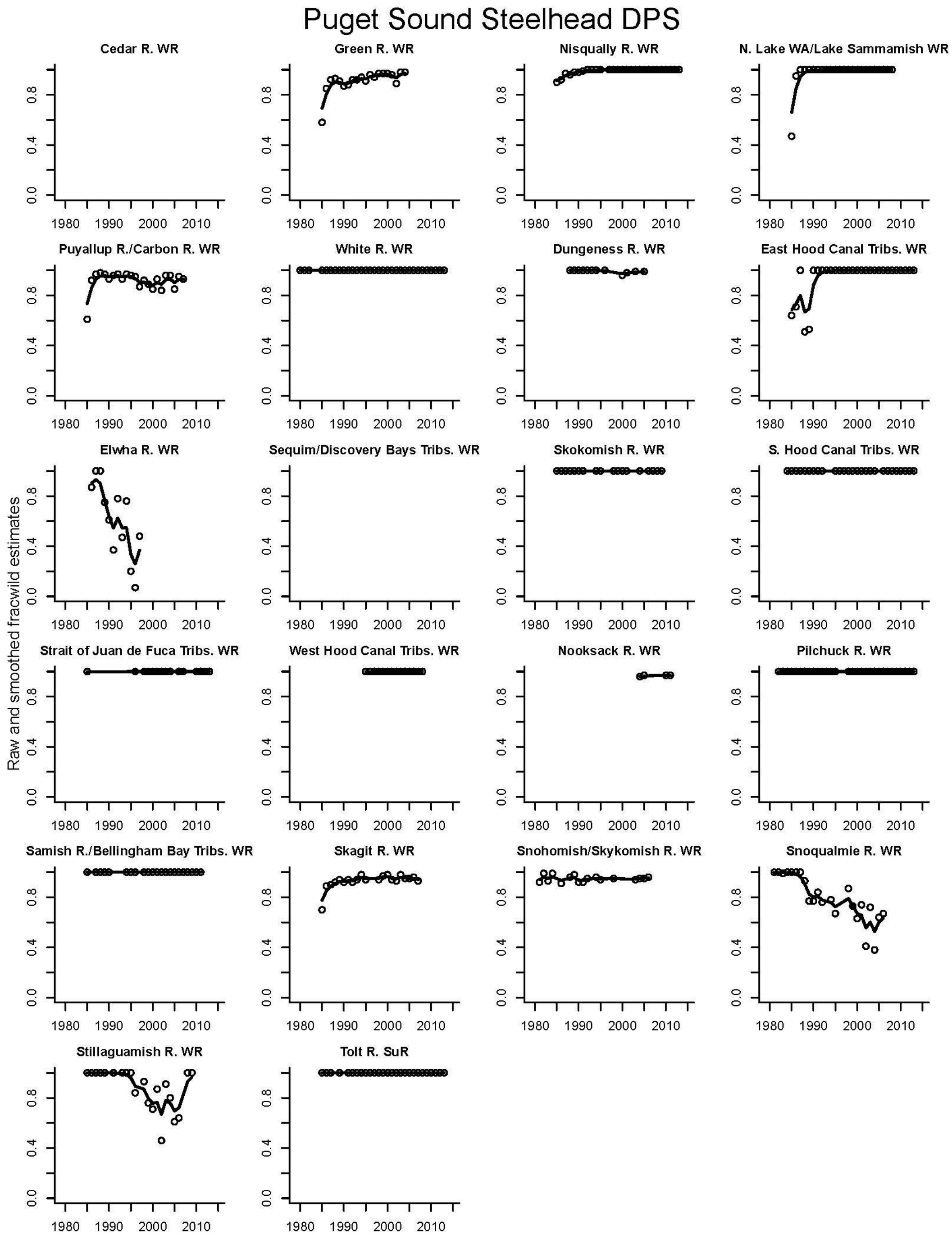

Figure 114 -- Smoothed trends in the estimated fraction of the spawning population consisting of steelhead of natural origin in Puget Sound. Points show the annual raw estimates. WR, winter run; SuR, summer run.

Table 62 -- 5-year mean of fraction natural (sum of all estimates divided by the number of estimates) for Puget Sound steelhead. Blanks indicate that no estimate is available for that 5-year range. WR, winter run; SuR, summer run.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Cedar R. WR					
Green R. WR	0.91	0.95	0.96		
Nisqually R. WR	0.99	1.00	1.00	1.00	1.00
N. Lake WA/Lake Sammamish WR	1.00	1.00	1.00	1.00	
Puyallup R./Carbon R. WR	0.95	0.92	0.91	0.91	
White R. WR	1.00	1.00	1.00	1.00	1.00
Dungeness R. WR	1.00	1.00	0.98	0.99	
East Hood Canal Tribs. WR	1.00	1.00	1.00	1.00	1.00
Elwha R. WR	0.60	0.25			
Sequim/Discovery Bays Tribs. WR					
Skokomish R. WR	1.00	1.00	1.00	1.00	
S. Hood Canal Tribs. WR	1.00	1.00	1.00	1.00	1.00
Strait of Juan de Fuca Tribs. WR		1.00	1.00	1.00	1.00
West Hood Canal Tribs. WR		1.00	1.00	1.00	
Nooksack R. WR			0.96	0.97	0.97
Pilchuck R. WR	1.00	1.00	1.00	1.00	1.00
Samish R./Bellingham Bay Tribs. WR	1.00	1.00	1.00	1.00	1.00
Skagit R. WR	0.94	0.95	0.96	0.95	
Snohomish/Skykomish R. WR	0.94	0.95	0.94	0.96	
Snoqualmie R. WR	0.79	0.76	0.58	0.66	
Stillaguamish R. WR	1.00	0.88	0.75	0.81	
Tolt R. SuR	1.00	1.00	1.00	1.00	1.00

Reiter ponds), ceased off-station early-winter smolt releases altogether, stopped the practice of "recycling" adults trapped at the hatcheries downstream to enhance sport fisheries, and maintained traps open for the entire duration of the early-winter hatchery adult period to remove the fish and reduce straying risks. Unpublished estimates from the Washington Department of Fish and Wildlife suggest that the influence of hatchery in several populations are now low. As a consequence, the risk posed by hatchery steelhead programs in the DPS has declined since the 2011 review.

In a broader context, the Puget Sound Steelhead TRT recently completed its evaluation of factors that influence the diversity and spatial structure VSP criteria for steelhead in this DPS (Hard *et al.* 2015). For diversity, these factors included hatchery fish production, the potential contribution of resident fish to anadromous fish production, and run timing of adult steelhead. Because little quantitative information on these elements of diversity was available for most steelhead populations in the DPS, the TRT used the Bayesian Network framework it developed for all four VSP criteria to develop semi-quantitative or qualitative assessments of the influence of these factors on population viability and then built hierarchical networks to scale up their evaluation of diversity (and the other) criteria at the MPG and DPS levels. The TRT relied primarily on two considerations: 1) the potential influence of hatchery-produced steelhead, most of which are either highly domesticated (Chambers Creek winter run) or out-of-basin source stocks (Skamania River summer run), on wild fish; and 2) evidence for an alteration in natural run timing from historical patterns. More recently, Warheit's (Warheit 2014) report summarizing evidence for introgression of hatchery steelhead from segregated programs into natural steelhead populations in Puget Sound showed a wide range of such effects, with some natural populations nearly completely hatchery-derived.

For spatial structure, the factors the TRT considered for influence on viability included fraction of suitable rearing and spawning habitat occupied by steelhead in the DPS (as measured by intrinsic potential, a measure of historical production or capacity based on the relationship between suitable habitat area and estimates of historical steelhead density).

Quantitative information on spatial structure and connectivity was not available for most steelhead populations in the DPSs the TRT used its Bayesian Network framework to assess the influence of these factors on viability at the population, MPG, and DPS scales. The TRT examined how steelhead DIPs tended to be related according to habitat characteristics, estimating a Gower similarity coefficient that incorporated maximum elevation, current spawnable area, mean bankfull width, mean stream gradient, maximum mean temperature, and presence of permanent snowpack in watersheds harboring the DIPs, and then determined influence on viability for each DIP by estimating occupancy of juvenile and adult steelhead in reaches within distinct habitat classes encompassed within steelhead intrinsic potential area.

The Bayesian Network analyses by the Puget Sound Steelhead TRT indicated that diversity and spatial structure have smaller influences on population viability than do abundance and productivity. Populations throughout the DPS showed strong influences of both abundance and productivity on viability; viability appeared to be especially sensitive to low productivity (Hard *et al.* 2015). Nevertheless, viability did depend on sufficient diversity and spatial structure, and these VSP elements sometimes had substantial effects on viability. This was particularly true if the influence of hatchery fish in a population had been high, if there was evidence that run timing had been altered or compressed, or if adult or juvenile distribution was limited in suitable habitat.

The TRT concluded that population viability with respect to diversity and spatial structure was highest in the Northern Cascades MPG and lowest in the Central & South Puget Sound MPG (Figure 115). Populations in the Northern Cascades MPG tended to show more variability in viability and diversity was generally higher; populations in both the Central & South Puget Sound and the Hood Canal & Strait of Juan de Fuca MPGs, where viabilities were lower, were influenced by depressed diversity (Hard *et al.* 2015). Those populations with higher viabilities did not show a consistently strong influence of diversity. Viability tended to increase with both diversity and spatial structure for populations in the Northern Cascades and Hood Canal & Strait of Juan de Fuca MPG; the pattern was less clear for populations in the Central & South Puget Sound MPG, but there was less variability in spatial structure of steelhead populations in that MPG, and diversity there tended to be lower as well. Populations in the Northern Cascades MPG tended to show more variability in viability and diversity was generally higher; populations in both the Central & South Puget Sound and the Hood Canal & Strait of Juan de Fuca MPGs, where viabilities were lower, were influenced by depressed diversity (Hard *et al.* 2015). In general, variation in diversity, spatial structure, and viability was highest in the Northern Cascades MPG, where the number of populations and occupancy of suitable habitat was highest (Figure 115). Again using an 18-point scale for viability, a simplified framework showed most DIPs have moderate viability scores for diversity, but this largely reflected the lack of reliable quantitative information for most of the factors the TRT considered for this VSP criterion (see Figure 113). Three populations were considered to have depressed diversity due to substantial hatchery influence until recently; these were Snohomish River/Skykomish River WR, Snoqualmie River WR, and Tolt River SuR, located in the Northern Cascades MPG. Others were given a moderate ranking for diversity due to small annual breeding population sizes (and presumably low effective population sizes).

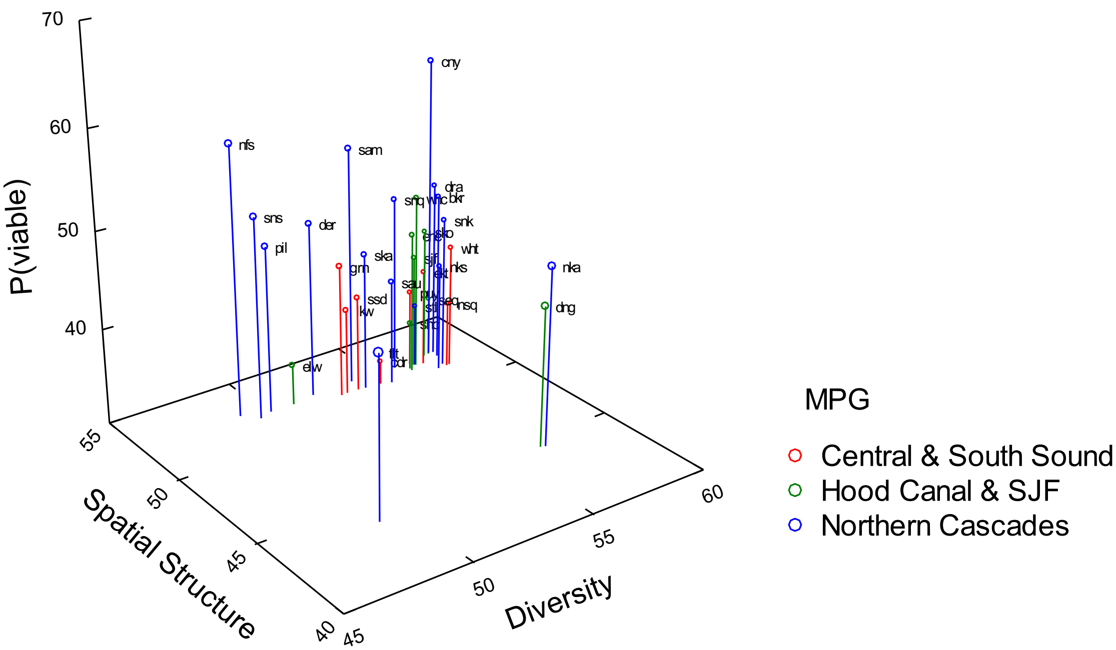

Figure 115 -- Scatter plot of the probabilities of viability for each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of VSP parameter estimates of influence of diversity and spatial structure on viability. Probabilities of viability were computed from the DIP-level Bayesian Networks developed by the Puget Sound Steelhead TRT (Hard *et al.* 2015). Populations from the three MPGs are coded in red (Central & South Puget Sound), green (Hood Canal & Strait of Juan de Fuca), and blue (Northern Cascades). Diversity and spatial structure scores are estimated from intermediate metrics computed by the DIP-level Bayesian Networks. Three-letter DIP codes (WR = winter-run and SuR = summer-run steelhead) are: dra = Drayton Harbor Tributaries WR, nks = Nooksack River WR, sns = South Fork Nooksack River SuR, sam = Samish River and Bellingham Bay Tributaries WR, ska = Skagit River SuR and WR, nka = Nookachamps Creek WR, bkr = Baker River SuR and WR, sau = Sauk River SuR and WR, stl = Stillaguamish River WR, der = Deer Creek SuR, cny = Canyon Creek SuR, snk = Snohomish/Skykomish Rivers WR, pil = Pilchuck River WR, nfs = North Fork Skykomish River SuR, snq = Snoqualmie River WR, tlt = Tolt River SuR, lkw = North Lake Washington and Lake Sammamish WR, cdr = Cedar River WR, grn = Green River WR, puy = Puyallup/Carbon Rivers WR, wht = White River WR, nsq = Nisqually River WR, ssd = South Puget Sound Tributaries WR, ekt = East Kitsap Peninsula Tributaries WR, ehc = East Hood Canal Tributaries WR, shc = South Hood Canal Tributaries WR, sko = Skokomish River WR, whc = West Hood Canal Tributaries WR, seq = Sequim/Discovery Bays Tributaries WR, dng = Dungeness River SuR and WR, sjf = Strait of Juan de Fuca Tributaries WR, and elw = Elwha River WR.

The viability of several populations across the DPS depended on the influence of spatial structure. This was especially true for some populations in the Northern Cascades MPG and in the Hood Canal & Strait of Juan de Fuca MPG. In fact, only the Nookachamps River WR and Tolt River SuR DIPs in the Northern Cascades MPG and the Dungeness River WR in the Hood Canal & Strait of Juan de Fuca MPG showed a relatively small influence of spatial structure on viability. When considered together with productivity, spatial structure had a modest influence on population viability; for DIPs with a high influence of spatial structure on viability, the DIPs with lowest viabilities were influenced heavily by low productivity. For the few DIPs that showed a modest influence of spatial structure on viability, viability tended to be moderate (Hard *et al.* 2015). Using the 18-point scale for viability, nearly all

DIPs had moderate viability scores for spatial structure. The TRT considered just two DIPs—Tolt River SuR and Dungeness River WR—to score highly for this criterion with respect to viability (Figure 113).

Although abundance and productivity are still the major limitations to viability of Puget Sound steelhead populations, particular elements of diversity and spatial structure, including natural spawning by hatchery fish and limited use of suitable habitat, are still contributing to these limitations, with few prospects for substantial improvement in the next few years.

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The Puget Sound Steelhead Recovery Team was established by NOAA Fisheries and convened in March 2014 to develop a Recovery Plan for the Puget Sound Steelhead DPS. This Recovery Plan has not yet been drafted, and draft recovery goals are not yet available for the DPS or its component DIPs (a draft of the Recovery Plan is expected from the Recovery Team in 2016). The Recovery Team is working from the characterization of independent steelhead populations in the Puget Sound DPS and the viability criteria developed for them by the Puget Sound Steelhead TRT (Myers *et al.* 2015, Hard *et al.* 2015) to identify these recovery goals. That being said, the DPS's current status, particularly with respect to abundance and productivity, is considered to be well below the targets needed to achieve delisting and recovery.

UPDATED BIOLOGICAL RISK SUMMARY

Consideration of the above analyses indicates that the biological risks faced by the Puget Sound Steelhead DPS have not substantively changed since the listing in 2007, or since the 2011 status review. Furthermore, the Puget Sound Steelhead TRT recently concluded that the DPS was at very low viability, as were all three of its constituent MPGs, and many of its 32 DIPs (Hard *et al.* 2015). Although the most recent data available indicate some minor increases in spawner abundance or improving productivity over the last 2-3 years, most of these improvements are small and abundance and productivity throughout the DPS remain at levels of concern for demographic risk. Recent increases in abundance that have been observed in a few populations have been within the range of variability observed in the past several years. Trends in abundance of natural spawners remain predominantly negative. Particular aspects of diversity and spatial structure, including limited use of suitable habitat, are still likely to be limiting viability of most Puget Sound steelhead populations. Reduced harvest and declining production of both summer- and winter-run hatchery fish in the DPS have limited those risks to natural spawners in recent years.

In the near term, the outlook for environmental conditions affecting Puget Sound steelhead is not optimistic. While harvest and hatchery production of steelhead in Puget Sound are currently at low levels and are not likely to increase substantially in the foreseeable future, some recent environmental trends not favorable to Puget Sound steelhead survival and production are expected to continue. The exceptionally warm marine waters in 2014 and 2015 and warm stream temperatures observed during 2015 were unfavorable for high marine or freshwater survival. The overall effects of these environmental conditions will not be known until adults return beginning this fall and continuing for the next few years. Nevertheless, a positive pattern in the Pacific Decadal Oscillation, which has been in place since January 2014, is expected to continue, and current El Niño conditions will probably persist through at least the end of 2015 (see chapter on Recent trends in marine and terrestial environments and their likely influence on Pacific salmon in the Pacific Northwest, below). These and other environmental indicators point to continued conditions of warming ocean temperatures, fragmented or degraded freshwater spawning and rearing habitat, reduced snowpack, altered hydrographs producing reduced summer river flows and warmer water, and low marine survival for salmonids in the Salish Sea. These conditions are almost certain to constrain any rebound in VSP parameters for Puget Sound steelhead in the near term.

HOOD CANAL SUMMER-RUN CHUM SALMON ESU

BRIEF DESCRIPTION OF ESU

The ESU includes all naturally spawning populations of summer-run chum salmon in Hood Canal tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington, as well as several artificial propagation programs (Figure 116). The Puget Sound Technical Recovery Team identified two independent populations for the Hood Canal summer chum, one which includes the spawning aggregations from rivers and creeks draining into the Strait of Juan de Fuca, and one which includes spawning aggregations within Hood Canal proper (Sands *et al.* 2009).

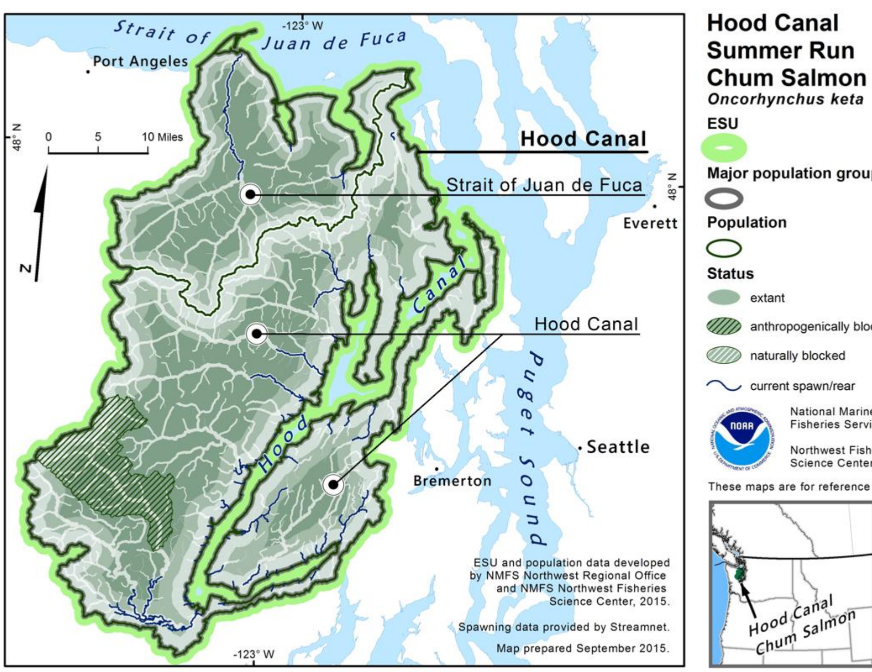

Figure 116 -- Map of the Hood Canal summer-run chum salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

At the time of the 2005 status review (Good *et al*. 2005), the Puget Sound TRT had not yet finalized its population designations or viability criteria for this ESU. Most stocks were showing positive growth rates and increased natural spawning abundance compared to the time of listing. These increases were likely a result of harvest reductions, supplementation programs in some streams, habitat restoration projects in freshwater and nearshore habitats, and possibly improvements in ocean conditions.

2010

Ford *et al*. (2011) noted that the spawning abundance of this ESU had clearly increased since the time of listing, although the abundance for the 2010 review was down from the previous 5-years. While spawning abundances had remained relatively high compared to the low levels in the early 1990's, productivity had decreased significantly, being lower for brood years 2002-2006 than any previous 5-year average since 1971. Diversity had increased from the low values seen in the 1990s due both to the reintroduction of spawning aggregates and the more uniform relative abundance between populations; this was considered to be a good sign for viability in terms of spatial structure and diversity. Spawning survey data showed that the spawning distribution within most streams has been extended further upstream as abundance had increased. Overall, the new information considered in 2010, however, did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Escapement data, total run size, estimated natural-origin spawners (NOS) and supplementation-origin spawners (SOS), age distribution of the natural-origin escapement, and hatchery broodstock take are recorded per spawning aggregation and catch information are available for each fishery management area from 1974 through 2013. The Point No Point Treaty Tribes (PNPTT) and Washington Department of Fish and Wildlife (WDFW) recently completed a five-year review of the Summer Chum Salmon Conservation Initiative for the period 2005 through 2013 (PNPTT and WDFW 2014) which details all data listed above, and also provides some corrections to previous estimates. Estimates of age composition for each stream or natural spawning aggregation are available for the newer period 2005-2013. A genetic stock identification and assessment program was continued through the 2005-2013 time period, and extensive collection of data (DNA, scales, lengths, otoliths, sex, abundance) was conducted. Mark recoveries of otoliths and adipose fin clipped returning adults were conducted primarily on the spawning grounds and allowed estimation of level of straying of the supplementation-origin program fish to other drainages, and estimation of total returns of both natural origin and supplementation origin fish. Supplementation programs were begun in 1992, prior to which all summer chum adult returns to Hood Canal and Strait of Juan de Fuca were natural-origin fish. The first hatchery supplementation-origin adults returned to spawn naturally in 1995, but 2001 was the first year in which large returns of summer chum supplementation-origin fish contributed to total adult returns. Estimates of the proportions of hatchery fish on the spawning grounds are available from 1974 through 2013 for the Hood Canal and Strait of Juan de Fuca populations (PNPTT and WDFW 2014). Hatchery contribution varies greatly among the spawning aggregations within each population. It is generally highest in the Strait of Juan de Fuca population, ranging from 8.4 to 62.8% in the Strait of Juan de Fuca population, and 5.8 to 40.2% in the Hood Canal population. The hatchery contribution also generally decreased as supplementation programs were terminated as planned (PNPTT and WDFW 2014). To estimate run size, state and tribal co-managers apportion catch data out to spawning aggregates based on the location of the fish catch in relation to the spawning tributaries (PNPTT and WDFW 2014).

ABUNDANCE AND PRODUCTIVITY

Estimates of total (NOS + SOS) and natural (NOS) spawning abundances are available from 1974 for both the Hood Canal population and the Strait of Juan de Fuca population, and are shown from 1980 through 2013 in this review (Figure 117). Smoothed trends in estimated total and natural population spawning abundances for both Hood Canal and Strait of Juan de Fuca populations have generally increased over the 1980 to 2014 time period. Shorter-term trends since about 2004, coinciding with the supplementation programs, have seen increased abundances sustained at a level higher than during the period of listing.

Average escapements (geometric means) for 5-year intervals beginning in 1990 show estimates of trends over the intervals for both natural-origin spawners (NOS) and total (NOS + SOS) spawners (Table 63). The Hood Canal population has had a 25% increase in abundance of natural-origin spawners in the most recent 5-year time period over the 2005-2009 time period. The Strait of Juan de Fuca has had a 53% increase in abundance of natural-origin spawners in the most recent 5-year time period. Spawner abundances in both Hood Canal and Strait of Juan de Fuca populations were lowest throughout the 1990's but increased in the early 2000's.

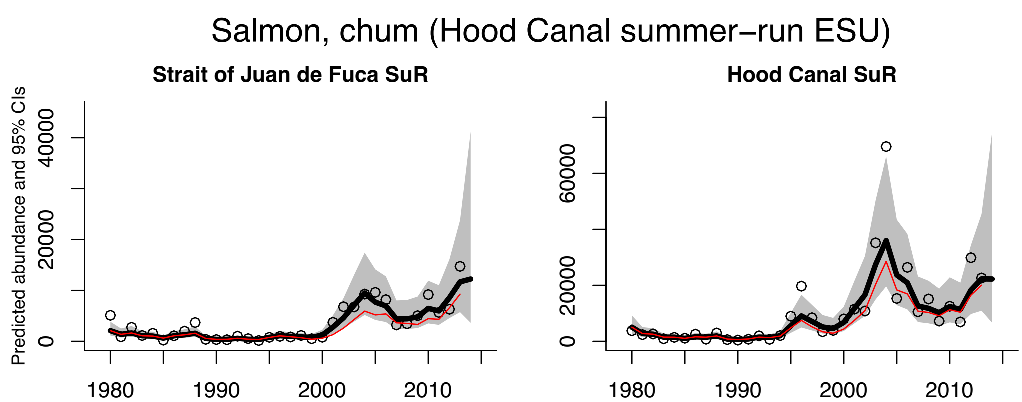

Figure 117 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

Fifteen-year trends in log natural-origin spawner abundance were computed over two time periods (1990 – 2005 and 1999 – 2014) from a linear regression model applied to the smoothed wild spawner log abundance estimate over annual return years. Trends were positive in the two populations in both time periods, although trends were lower in the most recent 15-year period than in the prior period (Table 64).

Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year (t-4), have increasing over the past five years, and have been above replacement rates in the two most recent years. However productivity rates have been varied above and below replacement rates over the entire time period (Figure 118). This is the realized productivity rate, and values below zero indicate that productivity in a given year is estimated to be below replacement rates for returning natural-origin spawners. PNPTT and WDFW (2014) provide a detailed analysis of productivity for the ESU, each population, and by individual spawning aggregation, and report that 3 of the 11 stocks exceeded the co-manager's interim productivity goal of an average of 1.6 R/S over 8 years. They also report that natural-origin recruit/spawner rates have been highly variable in recent brood years, particularly in the Strait of Juan de Fuca population. Only one spawning aggregation (Chimacum) meets the co-manager's interim recovery goal of 1.2 recruits per spawner in 6 of most recent 8 years.

The co-managers' status review indicates that productivity has been relatively low, as measured by recruits per spawner (R/S), in brood years 2003-2006, perhaps reflecting higher densities due to increased abundance. In one spawning aggregate at Big Beef Cr. they have consistently measured low productivity (<1) across all years. Big Beef Creek is a minor contributor to the overall Hood Canal population, but this component is considered essential for recovery (NMFS 2007). Other individual HC stocks also show varying degrees of density dependence (PNPTT and WDFW 2014). The Strait of Juan de Fuca population is not as clearly density dependent, except that there are indications of such in Salmon and Snow creeks in Discovery Bay.

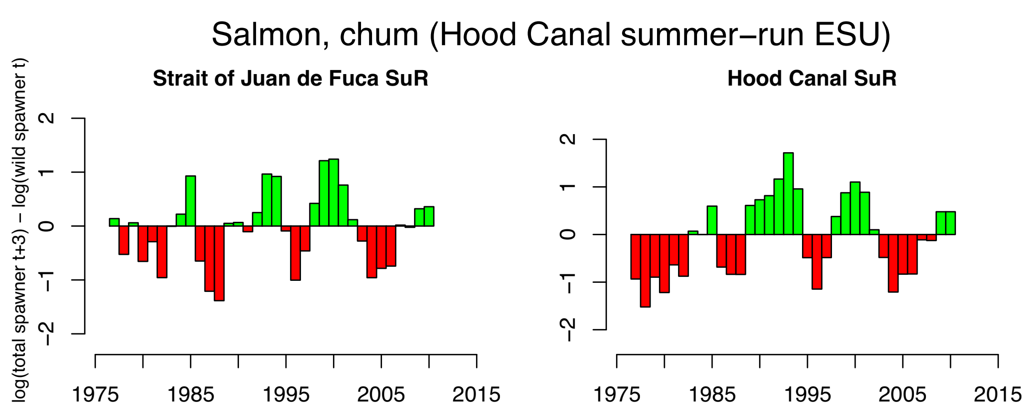

Figure 118 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).

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Table 63 -- 5-year geometric mean of raw wild spawner counts. This is the raw total spawner count times the fraction wild estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of wild spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Strait of Juan de Fuca SuR	Hood Canal	386 (386)	629 (822)	2190 (4178)	4020 (5353)	6169 (8339)	53 (56)
Hood Canal SuR	Hood Canal	979 (979)	5169 (7223)	13145 (18928)	11307 (13605)	14152 (15553)	25 (14)

Table 64 -- 15-year trends in log wild spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Strait of Juan de Fuca SuR	Hood Canal	0.17 (0.11, 0.23)	0.15 (0.08, 0.21)
Hood Canal SuR	Hood Canal	0.22 (0.17, 0.27)	0.07 (0.01, 0.13)

HARVEST

There are no directed fisheries on Hood Canal summer chum. However, they are taken incidentally in fisheries directed at other species in the Strait of Juan de Fuca, in Hood Canal, and in Canada. Because the populations from the Eastern Strait of Juan de Fuca (Dungeness River through Port Townsend Bay) are not subject to fisheries in Hood Canal directed at Chinook and coho salmon, they experience lower overall harvest rates in general. Historically, the the Eastern Strait of Juan de Fuca population experienced harvest rates on the order of 10-30% with rates as high as 50% in individual years. The Hood Canal population was subject to harvest rates that were typically on the order of 50% to 70% with rates in individual years approaching 90% (PNPTT and WDFW 2014).

In response to severely depressed runs of summer-run chum salmon, in the early 1990s, the State of Washington and the Western Washington Treaty Tribes took measures to curb the incidental harvest of summer chum and harvest rates fell dramatically (Figure 119). The co-managers have implemented a Base Conservation Regime (BCR) and continued to constrain harvest impacts as runs have approached or returned to historic levels, leading to escapements that have exceed historic levels. Under the BCR, harvest rates have declined to about 2% to 15% for Hood Canal summer chum and to less than 2% for the Strait of Juan de Fuca summer chum. Harvest rates have been below the BCR harvest rate limits for all years in the Strait of Juan de Fuca fisheries and for all years except 2004 in Hood Canal fisheries. From 2000 through 2013, the harvest rate for the ESU has averaged about 8% (PNPTT and WDFW 2014).

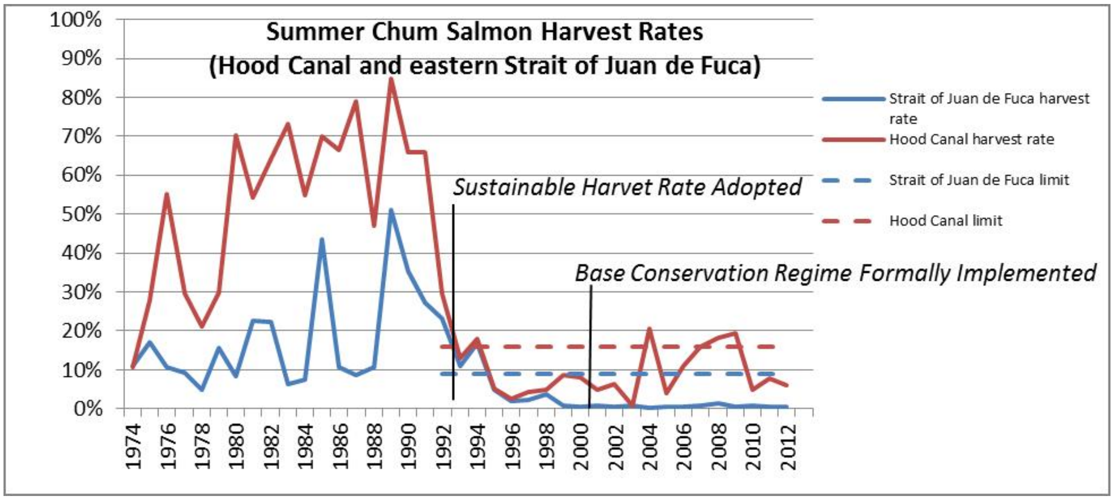

Figure 119 -- Total exploitation rate on the combined Hood Canal/Strait of Juan de Fuca summer chum salmon ESU. Data from WDFW run reconstruction (1974-2011 data from http://wdfw.wa.gov/fishing/salmon/chum/pugetsound/data.html; 2012 and 2013 data from Aaron Dufault, WDFW, personal communication).

SPATIAL STRUCTURE AND DIVERSITY

Spatial structure and diversity measures for the Hood Canal summer chum recovery program include the reintroduction and sustaining of natural-origin spawning in multiple small streams where summer chum spawning aggregates had been extirpated. A supplementation program was initiated in 1992 to meet this objective. The first supplementation-origin spawners (SOS) began to return in 1995, however, it wasn't until 2001 that large numbers of SOS program fish were widely distributed in each population (PNPTT and WDFW 2014). Spatial structure and diversity parameters are measured here in terms of the proportion of natural-origin spawners (NOS) vs. supplementation-origin (SOS) spawners on the spawning grounds. All returning summer chum spawners were wild in both populations until fish from the supplementation program began to return to spawn in 1995 (Figure 120). Supplementation programs were intended to run for a maximum duration of three generations, or 12 years. Programs in the Strait of Juan de Fuca population (Salmon, Jimmycomelately, and Chimacum creeks), and in the Hood Canal population (Big Quilcene, Hamma Hamma, Lilliwaup, Union, Tahuya and Big Beef) were phased in between 1992 and 2003. As program goals were met all programs (except Lilliwaup) had been terminated by 2014. Lilliwaup did not meet the production targets (e.g., broodstock collections and release numbers) in some earlier years and only recently are there indications of recovery so supplementation is ongoing (PNPTT and WDFW 2014). As SOS fish returns have phased out, there has been a gradual return to predominantly natural-origin spawners for both Hood Canal and Strait of Juan de Fuca summer chum populations (Figure 120, Table 65). For the Hood Canal population, SOS fish are still returning to the Tahuya River (through 2018) and to Lilliwaup River. The Strait of Juan de Fuca population shows lower estimates of proportion natural-origin spawners (NOS) primarily because relatively large numbers of SOS from the last program release in 2010 are still returning to Jimmycomelately Creek through 2015 (Figure 120).

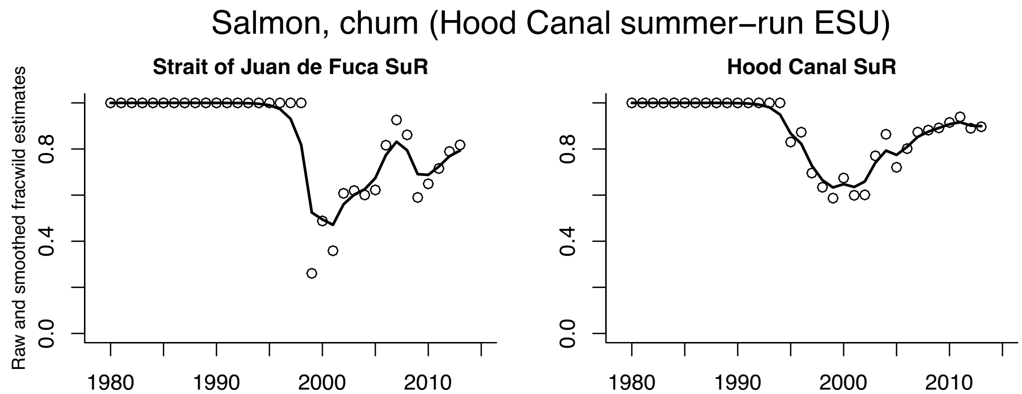

Figure 120 – Proportion of each summer chum population comprised of natural-origin spawners. Line shows a smoothed trend and points show the annual estimates.

Table 65 -- 5-year mean of fraction natural origin spawners (sum of all estimates divided by the number of estimates).

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Strait of Juan de Fuca SuR	1.00	0.85	0.53	0.76	0.74
Hood Canal SuR	1.00	0.72	0.70	0.83	0.91

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The PS TRT defined the abundance and productivity viability criteria for the Hood Canal and Strait of Juan de Fuca summer chum salmon populations using two different population viability analyses (PVAs). One PVA method assumed density independence at low population sizes and replacement growth factor of 1:1. The other method (Viability and Risk Assessment Procedure, or VRAP) assumped density dependence between recruits and spawners and generated a series of spawner-recruit curves based on variable productivities and capacities, and fixed exploitation rates (NMFS 2007; Sands *et al.* 2009). We have not conducted a detailed VRAP assessment for this review, but the co-managers did so recently (Lestelle *et al.* 2014). They conducted a new viability analysis using VRAP and incorporating updated data (as described above), as well as, new data from five additional brood years (2002-2006). This updates the analysis conducted by the PS TRT and presented in Sands et al (2009). The potential impacts of shifts in decadal-scale ocean (i.e., the Pacific Decadal Oscillation) and climate regimes on summer chum performance and potential limits to recovery were also considered (Lestelle *et al*. 2014).

The minimum viability levels assuming density independence were 12,500 for the Strait of Juan de Fuca population and 24,700 for the Hood Canal population. Abundance of natural-origin spawners has clearly increased since listing in 1999 and these targets have been attained in the Strait of Juan de Fuca population in 2013 (2014 data are not included here), and in the Hood Canal population in five years since recovery implementation efforts began, including the two most recent years considered in this analysis (2012, 2013). Productivity for both populations has been greater than 1:1 for the past 2 years considered in this analysis (2012, 2013), though varied over the entire time period (Figure 118).

The PS TRT used VRAP to model viability (defined as ≤5% risk of extinction over 100 years) given specific intrinsic productivity, capacity, and exploitation rates. The resulting minimum spawner escapement numbers for the Strait of Juan de Fuca was 4,500 adults given intrinsic productivity of 5 and capacity of 3,300. For the Hood Canal population, a resulting minimum escapement of 18,300, given intrinsic productivity of 5 and capacity of 13,500. Results of the co-managers' updated assessment with a longer data set indicate changes may be appropriate in these viability thresholds due to more precise estimates of coefficient of variation. The minimum abundance viability threshold with zero harvest for Strait of Juan de Fuca population is 5,700 with intrinsic productivity of 6 and capacity of 5,100. For the Hood Canal population, minimum abundance viability threshold is 8,700 with intrinsic productivity of 8 and capacity of 7000, with zero exploitation rate. Results of VRAP analyses also suggest the Hood Canal population would be considered to be at negligible risk of extinction with current biological performance, provided that the exploitation rate remains very low. The Strait of Juan de Fuca population has a much higher risk of extinction, even with a zero exploitation rate (Lestelle *et al*. 2014). In addition, analyses of individual spawning aggregations indicate that six of the eight extant spawning aggregations in the two populations are at relatively high risk of extinction (see Figure 19 in Lestelle et al. 2014). Quilcene spawning aggregation has much higher performance than any of the other spawning aggregations, and Dosewallips performance is close to being viable at the five percent risk threshold. These results indicate the importance of both the Quilcene (and Dosewallips) spawning aggregations to the total population viability, and the necessity of continuing to evaluate the individual spawning aggregations (including their spatial distribution and diversity) to determine population viability.

The TRT defined viability for spatial structure as the need to maintain spawning aggregations that are well-distributed across the historical range of the populations (Sands *et al*. 2009). Most spawning aggregations need to be within 20 km of adjacent aggregations, and the major spawning aggregations need to be <40 km apart to meet the spatial structure viability parameter. Seven Ecological Diversity Groups were identified in the ESU, and three criteria were used by the TRT in defining recovery for the diversity viability parameter (Table 66). Diversity viability criteria specified that one or more spawning aggregations must be persistent within each of the two to four major ecological diversity groups historically present within the two populations (Sands *et al*. 2009, NMFS 2007).

Table 66 -- Seven ecological diversity groups as proposed by the PSTRT for the Hood Canal Summer Chum ESU by geographic region and associated spawning aggregation. (From Sands et al. 2009).

Geographic Region(population)	Proposed Ecological Diversity Groups	Spawning aggregations: Extant* and extinct**
Eastern Strait of Juan de Fuca	Dungeness	Dungeness R (unknown status)
	Sequim-Admiralty	Jimmycomelately Cr* Salmon Cr* Snow Cr* Chimacum Cr**
Hood Canal	Toandos	Unknown
	Quilcene	Big Quilcene R* Little Quilcene R*
	Mid West Hood Canal	Dosewallips R* Duckabush R*
	West Kitsap	Big Beef Cr** Seabeck Cr** Stavis Cr** Anderson Cr** Dewatto R** Tahuya R** Mission Cr** Union R*
	Lower West Hood Canal	Hamma Hamma R* Lilliwaup Cr* Skokomish R*

Criteria for spatial structure are nearly met for Strait of Juan de Fuca and Hood Canal summer chum populations. One exception is in east Hood Canal where spawning aggregations in Big Beef Creek and Tahuya River are about 60 km apart, thus an additional spawning aggregation would be needed in either Dewatto River or Anderson Creek (PNPTT and WDFW 2014). Spawning aggregations are present and persistent within five of the six major ecological diversity groups identified by the PS TRT (Table 66). There is still considerable uncertainty regarding the historical or current presence of a spawning aggregation in the Dungeness River (PNPTT and WDFW 2014). Though not specified in the viability criteria, the TRT used the Shannon diversity index to quantitatively assess variance in spatial distribution. Higher diversity values indicate a more uniform distribution of the population among spawning aggregations, indicating more robust populations. A recently updated analysis by co-managers (PNPTT and WDFW 2014) indicate that Shannon diversity indices were lower for both summer chum populations during the 1980's and 1990's, but have rebounded since the mid-2000's to levels higher approaching the base period of 1974-1978 (PNPTT and WDFW 2014). This is consistent with the 2010 NOAA five-year status review (Ford *et al.* 2011), the results showing an increase in the spatial structure and diversity of escapement, primarily due to reintroduction efforts in three watersheds and a more uniform relative abundance within each population.

Supplementation programs have been very successful in both increasing natural spawning abundance in 6 of 8 extant streams (Salmon, Big Quilcene, Lilliwaup, Hamma Hamma, Jimmycomelately, and Union) and increasing spatial structure due to reintroducing spawning aggregations to three streams (Big Beef, Tahuya, and Chimacum). The reintroductions have had mixed success, with Chimacum Creek being very successful, but natural-origin production has not yet been sustained in Big Beef Creek and Tahuya River (PNPTT and WDFW 2014). In general, habitat degradation is considered limiting to natural origin production. Habitat preservation and restoration projects in individual watersheds have been implemented concurrently with supplementation programs and have aided in the ability to sustain natural-origin production.

The co-managers assessment using VRAP was also used for each of the 8 extant spawning aggregations in Hood Canal and Strait of Juan de Fuca in order to estimate habitat goals for each spawning aggregation. Results led to the recommendation that habitat restoration and protection actions can then be done strategically to reduce the performance gaps for spawning aggregations projected to be below viability while also balancing the importance of biological diversity, spatial structure, and population abundance and productivity to long-term viability (Lestelle et al. 2014). The co-managers propose using the results of these analyses to develop new criteria and harvest provisions for a "Recovering" regime that would replace the Base Conservation Regime (Lestelle *et al*. 2014). An in depth discussion of the rationale is presented in both the co-manager status review (PNPTC and WDFW 2014) and in the guidance document (Lestelle *et al*. 2014).

The Hood Canal Coordinating Council (HCCC) prepared the recovery plan for Hood Canal and Eastern Strait of Juan de Fuca Summer Chum salmon in cooperation with local counties of the ESU and the co-managers (HCCC 2005). This plan currently guides habitat protection and restoration activities for summer chum recovery. Despite gains in habitat protection and restoration, the co-managers remain concerned that given the pressures of population growth, existing land use management measures through local governments (i.e., shoreline management plans, critical area ordinances, and comprehensive plans) may be compromised or not enforced. The Hood Canal Coordinating Council and co-managers advocate for the development of a strong habitat monitoring and adaptive management program as part of the recovery plan and recommend it be integrated to complement the existing stock assessment, harvest and hatchery management programs. The HCCC and co-managers propose there are sufficient new data and assessments which warrant revision of the current recovery plan, including updating recovery goals, prioritizing future habitat protection and restoration actions, addressing harvest goals, continuing reintroduction efforts, and continuing monitoring and evaluation for the Hood Canal Summer Chum ESU.

UPDATED BIOLOGICAL RISK SUMMARY

Natural-origin spawner abundance has increased since ESA-listing and spawning abundance targets in both populations have been met in some years. Productivity was quite low at the time of the last review (Ford *et al*. 2011), though rates have increased in the last five years, and have been greater than replacement rates in the past two years for both populations. However, productivity of individual spawning aggregates shows only two of eight aggregates have viable performance. Spatial structure and diversity viability parameters for each population have increased and nearly meet the viability criteria. Despite substantive gains towards meeting viability criteria in the Hood Canal and Strait of Juan de Fuca summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time.

LAKE OZETTE SOCKEYE SALMON ESU

The Lake Ozette Sockeye ESU was listed as threatened in 1999. Two subsequent status reviews in 2005 and 2010 determined that this status should be maintained.

BRIEF DESCRIPTION OF ESU

The ESU includes all naturally spawned aggregations of sockeye salmon in Lake Ozette and streams and tributaries flowing into Lake Ozette, Washington (**Figure 121**). The ESU also includes fish originating from two artificial propagation programs: the Umbrella Creek and Big River sockeye hatchery programs. The Puget Sound TRT considers the Lake Ozette sockeye salmon ESU to be composed of one historical population (Currens *et al.* 2009), with substantial sub-structuring of individuals into multiple spawning aggregations. The primary existing spawning aggregations occur in two beach locations (Allen's and Olsen's beaches), and in two tributaries (Umbrella Creek and Big River).

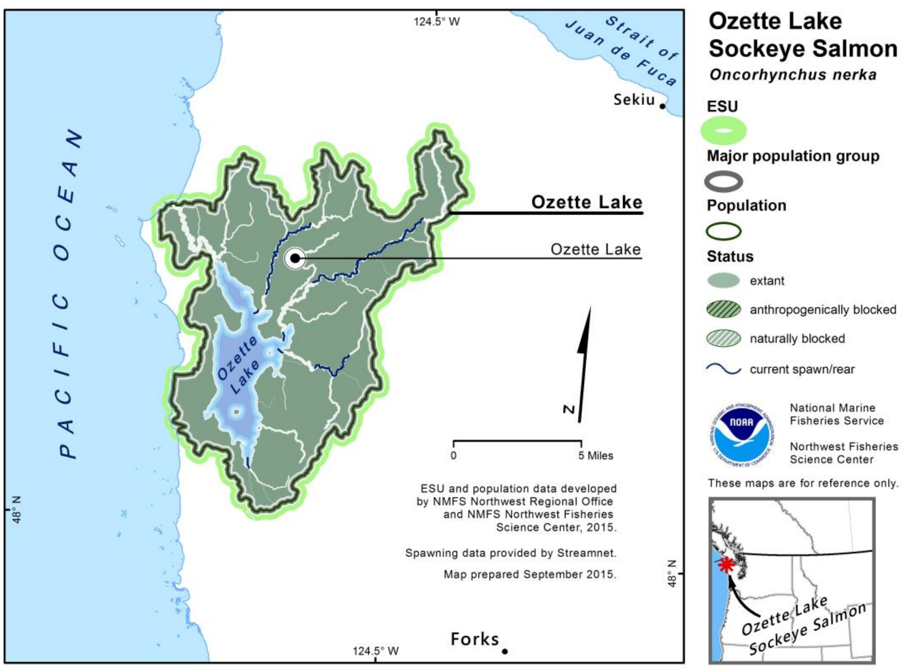

Figure 121 - Map of the Lake Ozette sockeye salmon ESU's spawning and rearing areas.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

Good *et al*. (2005) found little evidence of an increasing trend in population abundance since the listing in 1999 and emphasized that the available data was very uncertain and hampered efforts to assess trends and status in the VSP criteria. They recommended that the threatened status remain unchanged.

2010

Ford *et al*. (2011) concluded that estimates of population abundance for Lake Ozette sockeye remained highly variable and uncertain, making it impossible to detect changes in abundance trends or in productivity. It was clear, though, that population levels remained very low compared to historical levels. The review noted that assessment methods must improve in order to evaluate the status of this ESU and its responses to recovery actions. Overall, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the last BRT status review in 2005.

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

Run size estimates based on expanded weir counts have been extended from 2004 to 2012 in a report prepared for the National Marine Fisheries Service (Haggerty & Makah Fisheries Management 2014). Updated information on the hatchery program, tributary spawners and beach spawners was included in the Lake Ozette Sockeye Hatchery Genetics Management Plan Extension Request supporting tables (MFM 2015). These includes estimates of total run size and broodstock take for Umbrella Creek from 2000 to 2013, and estimates of proportion hatchery origin spawners (pHOS) during the same period for Umbrella Creek, Big River, and Allen's and Olsen's beaches. The NMFS biological opinion on the extension of the Ozette Sockeye Hatchery and Genetic Management Plan (NMFS 2015) included more recent beach spawning data, supplementing what had previously been available in the Limiting Factors Analysis report (Haggerty *et al.* 2009).

A spawning abundance series from 1977 to 2011 was constructed from a number of different sources (**Figure 122**, **Table 67**). As a whole the source data were very uncertain, with modest improvements in precision in recent years. From 1977 to 1995 the median expansion estimate from appendix B of the Lake Ozette Sockeye Limiting Factors Analysis (Haggerty *et al.* 2009, Appendix B) was used. Missing years during this period were filled with estimates based on the regression between these values and estimates from the Lake Ozette sockeye hatchery and genetic management plan (MFM 2000) when available or linear interpolation otherwise. In 1988 counting was conducted for only 3 days resulting in a total of 218 fish. We therefore discounted the high appendix B value (9,770), instead defaulting to the regression method. From 1996 to 1999 values from Table 3.6 of the Lake Ozette Sockeye Limiting Factors Analysis (Haggerty *et al.* 2009) were used. Finally, from 2000-2011 estimates from the Lake Ozette Sockeye Hatchery Genetics Management Plan Extension Request supporting tables (MFM 2015) were used where available with missing years filled in based on the regression between these values and Umbrella creek estimated run size from the same document. Run size estimates for the majority of years are highly uncertain. Estimates for many years were based on just a few days of observations, the weir was not always fish tight, and some data were lost due to failure of the storage media. Because % natural origin fish could not be reliably estimated from 1987 to 1999 we used total natural spawning fish to describe abundance except where noted.

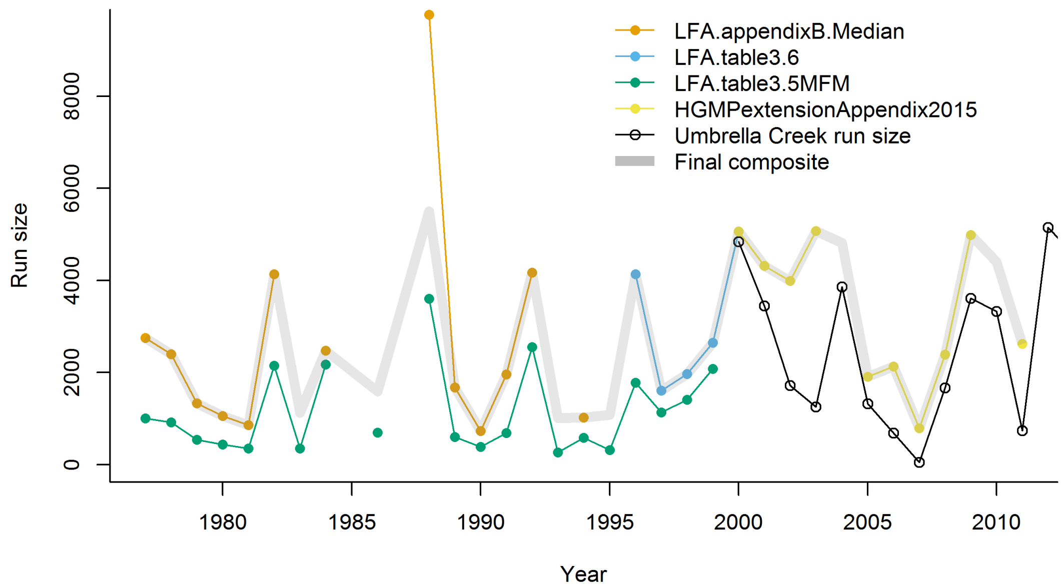

Figure 122 - The different sources of data used to construct annual run size estimates for Lake Ozette. The thick light gray line represents the final composite estimate. LFA refers to Haggerty *et al.* 2009. HGMPextensionAppendix refers to Lake Ozette Sockeye Hatchery Genetics Management Plan Extension Request Appendix 2015.

Table 67 – Lake Ozette and Umbrella Creek sockeye abundance.

Year	Spawners	Brood Stock	Natural spawners	Fraction Wild	Umbrella Spawners	Umbrella Hatchery origin
1977	2752	0	2752	1	NA	0
1978	2398	0	2398	1	NA	0
1979	1335	0	1335	1	NA	0
1980	1054	0	1054	1	NA	0
1981	858	0	858	1	NA	0
1982	4131	0	4131	1	NA	0
1983	1128	14	1114	1	NA	0
1984	2474	27	2447	1	NA	0

1985	2031	40	1991	1	NA	0
1986	1588	43	1545	1	NA	0
1987	3547	123	3424	1	NA	0
1988	5506	193	5313	NA	NA	NA
1989	1677	6	1671	NA	NA	NA
1990	732	33	699	NA	NA	NA
1991	1955	175	1780	NA	NA	NA
1992	4167	109	4058	NA	NA	NA
1993	1016	32	984	NA	NA	NA
1994	1018	54	964	NA	NA	NA
1995	1080	94	986	NA	NA	NA
1996	4131	200	3931	NA	NA	NA
1997	1609	263	1346	NA	NA	NA
1998	1970	88	1882	NA	NA	NA
1999	2649	29	2620	NA	NA	NA
2000	5064	213	4851	0.68	4842	1640
2001	4315	238	4077	0.97	3447	123
2002	3990	170	3820	0.94	1718	262
2003	5075	199	4876	0.97	1256	170
2004	4815	218	4597	0.92	3861	387
2005	1908	187	1721	0.90	1321	190
2006	2135	60	2075	0.94	686	140
2007	786	45	741	1.00	49	7
2008	2389	238	2151	0.91	1664	234
2009	4988	219	4769	0.89	3611	574
2010	4402	234	4168	0.94	3326	270
2011	2625	168	2457	0.93	740	237
2012	NA	167	NA	NA	5152	2698

The percentage of natural spawners that were of natural origin was 100% until 1986, since the first hatchery release was in 1983. Natural spawners for 1986 and 1987 were also assumed to be 100% natural origin since the first tributary spawner was observed in 1988. Between 1988 and 1999 there was no reliable data on percent hatchery origin (Haggerty *et al*. 2009, section 3). Percent wild origin for this period was therefore designated as unknown. For 2000 to 2011, we estimated the percentage of natural spawners as,

$$TotalRunSize - Brookstock - \%UmbrellaMarked(UmbrellaRunSize - Broodstock).$$

Here, $TotalRunSize$ is the estimated number of adults passing the weir, $Brookstock$ is the number of adults taken as brood stock for the hatchery program, $\%UmbrellaMarked$ is the percentage of captured Umbrella Creek fish with an observed hatchery mark, and $UmbrellaRunSize$is the estimated run size into Umbrella Creek based on mark recapture estimates. This estimate assumes that the Umbrella Creek run comprises most of the hatchery fish and that mortality from lake entry to tributary entry is negligible. The beach spawning fish are almost exclusively natural origin; however a smaller tributary spawning aggregation in Big River has a relatively high percent hatchery origin. We did not include these fish in this calculation because there was no estimate of the total number of Big River spawners. The extent to which each of these assumptions is violated is unknown but likely introduces a positive bias in the total number of natural origin spawners.

Natural spawners were calculated by subtracting the effective catch from the total run size (**Figure 123**). The effective catch is the number of fish that were removed from the natural spawning population due to harvest (1977-1982) or broodstock take (1983-present). Until 2000 all broodstock was taken from beaches and therefore predominately wild. From 2000 on, the brood stock was taken from Umbrella Creek and was therefore corrected for hatchery origin.

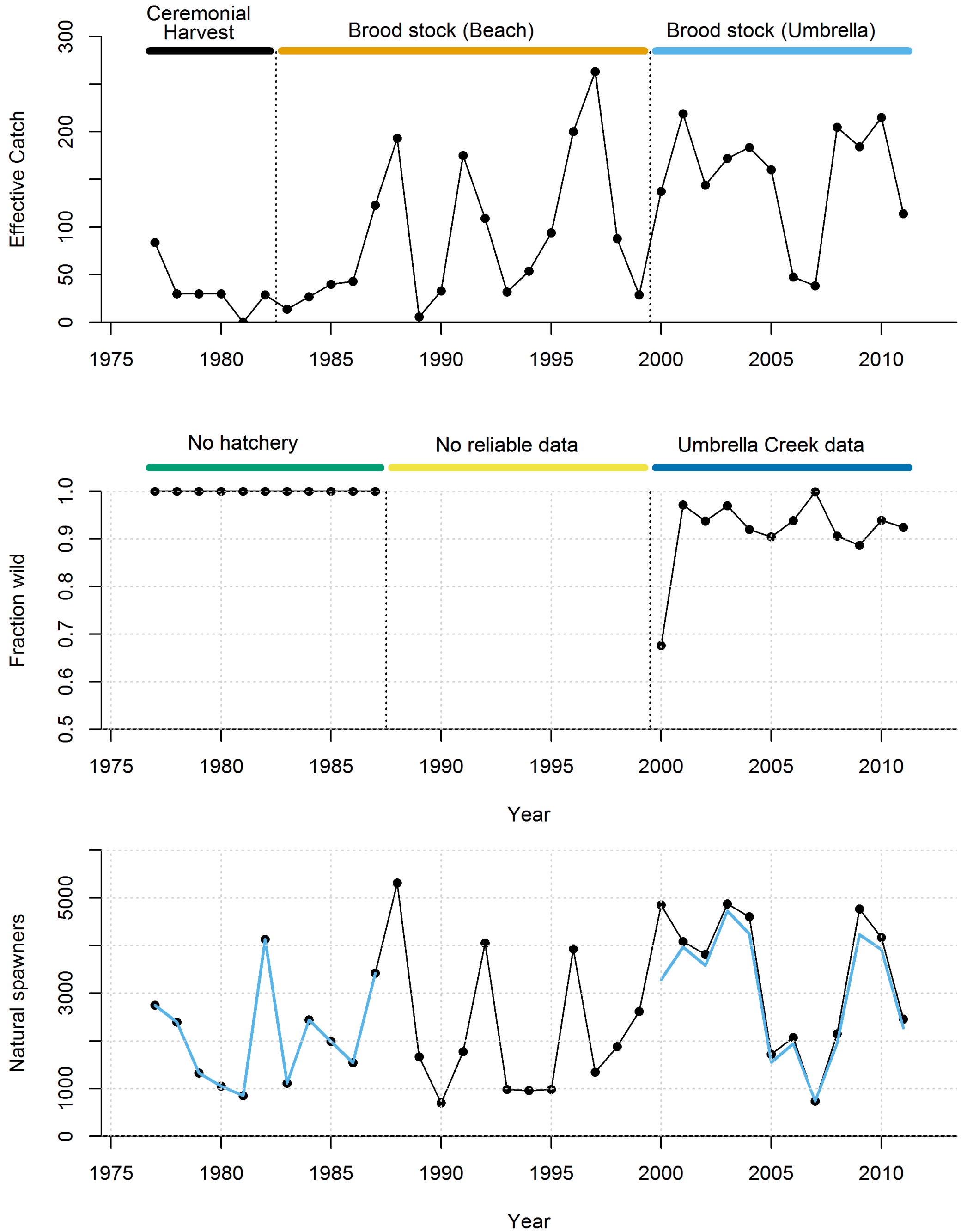

Figure 123 - Panel 1: effective catch over time. Panel 2: Fraction of natural spawners that are wild origin. Panel 3: Black line = natural spawners, blue line = natural origin natural spawners.

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ABUNDANCE AND PRODUCTIVITY

For the period from 1977 to 2011 the estimated natural spawners ranged from 699 to 5,313, well below the 31,250 – 121,000 viable population range proposed in the Lake Ozette sockeye recovery plan (NMFS 2009) (**Figure 123**). There is little evidence of a trend in the raw (**Figure 123**) or smoothed (Figure 124) abundance series over the full range of years or more recently since the last status review (Ford *et al.* 2011). There is some evidence of the dominant 4 year age of return in the abundance series (**Figure 126**), with the 1980 brood cycle line surpassing the other lines in late 80's and maintaining this higher level until 2000. Estimated productivity, calculated as the abundance in year t divided by the abundance in year t-4, fluctuated around 1 with no apparent overall trend but a suggestion of a 10 to 20 year cycle in both the raw (**Figure 126**) and smoothed (Figure 125) data. Given the degree of uncertainty in the abundance estimates, any interpretation of trends of small magnitude or over short time periods is speculative. Apparent patterns may be artifacts of substantial changes to the estimation method over time and/or changes in quantities on which the assumptions are based (e.g. run timing) (Haggerty & Makah Fisheries Management 2014). While hatchery origin fish were known to contribute to the population after the initiation of the Umbrella Creek Hatchery program in the mid 1980s, the % hatchery origin was not estimated until 2000. From 2000-2011 the estimated % hatchery origin ranged from 0% to 32% with a mean of 9%. To date correcting for % hatchery origin has not qualitatively changed the trends in abundance (**Figure 123**). However, because the Umbrella creek population is a large component of the total population (averaging over 50% for the last decade of data), large hatchery origin returns to Umbrella Creek can translate to large hatchery fractions overall. For example, in 2012 over 50% of fish returning to Umbrella Creek were hatchery origin (**Table 67**). Therefore, precise estimates of natural origin spawners depend on good estimates % hatchery origin fish.

Ocean fisheries do not significantly impact Lake Ozette sockeye salmon. Both Lake Ozette and the Ozette River, connecting the lake with the ocean, are closed to salmon fishing.

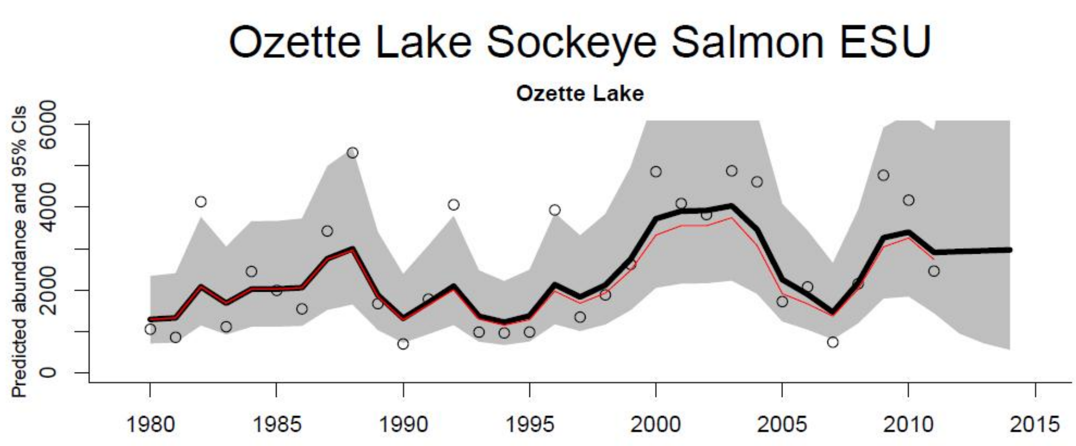

Figure 124 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

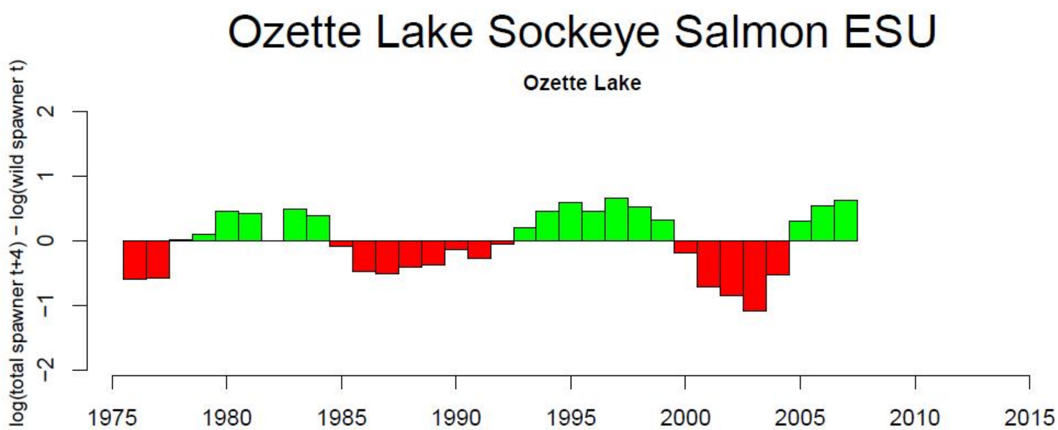

Figure 125 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 4).

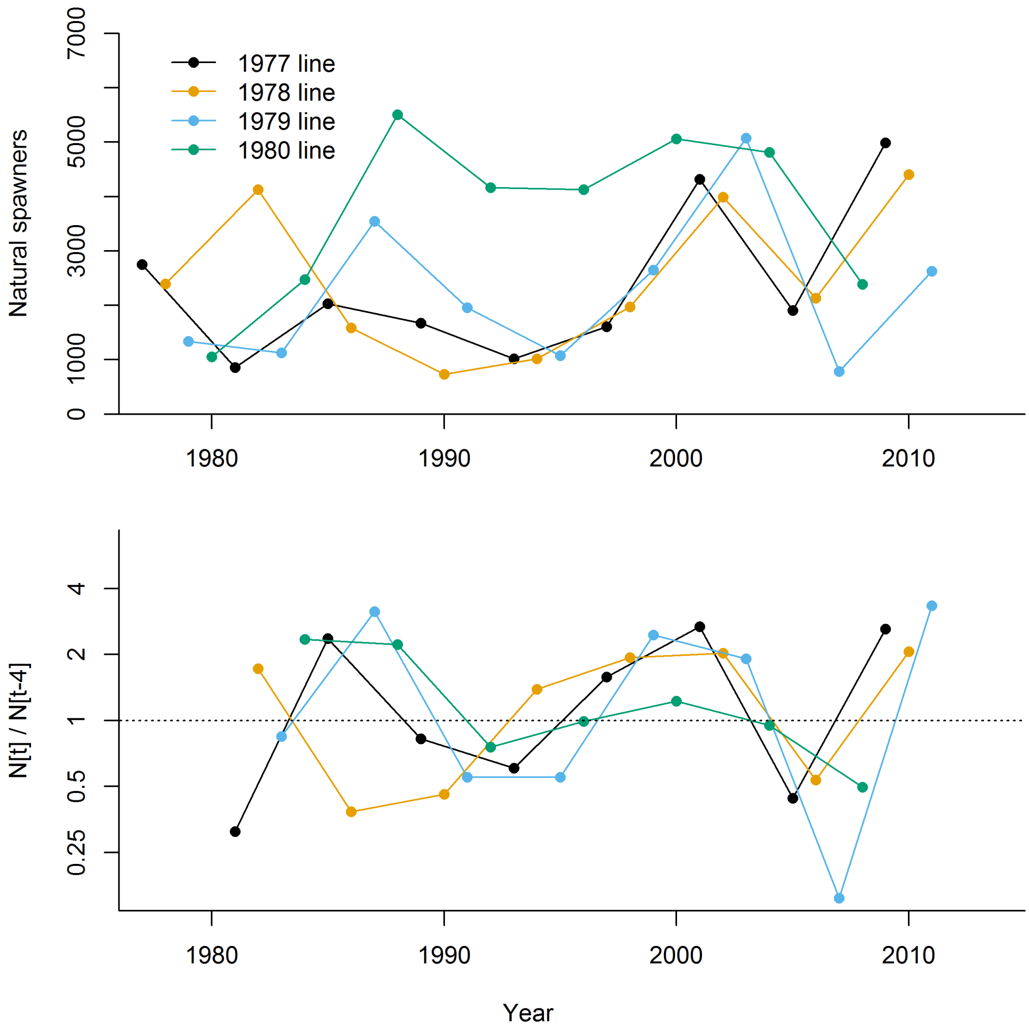

Figure 126 - Upper panel: Natural origin spawners vs year with lines connecting 4 year brood cycle lines. So, for example the 1977 brood cycle line includes the years 1977, 1981, 1985, etc... Lower panel: Productivity vs year where productivity is calculated as natural origin spawners in year t divided by the natural origin spawners in year t-4. Annual brood stock removals were ignored since the levels constituted small proportions of the total run sizes. Notice the y-axis is on the log scale in the lower panel.

SPATIAL STRUCTURE AND DIVERSITY

The current and historical distribution of spawners in the beaches and tributaries is uncertain. Extensive spawner surveys in the 70's, prior to the hatchery program, found no spawning in the tributaries (Haggerty *et al*. 2009, 3.4.3.1.2). The extent to which the tributaries were used prior to this time is uncertain with some attributing part of the decline in the overall population to loss of tributary spawners while others argue tributary spawning was not significant (Haggerty *et al.* 2009, 3.4.3.1.2). The historical geographic extent of beach spawning is not well documented. It is certain, however, that it was more spatially extensive than the current distribution. For example, Umbrella beach historically supported spawning before sediment input from Umbrella Creek covered the suitable substrate. In addition, spawning on the upper beach (in shallower water) has declined in recent years, likely resulting from increased shoreline vegetation.

Starting in the early 1990's, spawning aggregations in Umbrella Creek and Big River increased in size. The average run size for Umbrella Creek from 2000 to 2011 was over half that of the Ozette total run size, and some years were well over 90% (**Table 67**). Estimates of total beach spawner abundance are not available. However rough minimum estimates have been constructed using early brood stock collection efforts, some sporadic intentional surveys and more recently methodical surveys using visual and imaging sonar based counts (NMFS 2015; Haggerty *et al.* 2013 and 2014). While survey methods for beach spawners does not allow for estimates of total abundance, there is strong evidence that from 2005 to 2010 there were very few beach spawners. Since then the observed number of beach spawners has recovered to levels seen before this decline.

The estimated fraction of hatchery origin fish returning to Lake Ozette has been low in recent years (averaging 9% from 2000-2011). However, the large contribution of the hatchery supplemented tributary aggregations to the population as a whole allows for larger hatchery fractions. For example, in 2012, an estimated 2,698 hatchery origin adults returned to Umbrella Creek, which constituted over half of the Umbrella return (MFM 2015).

Tributary spawners appear to have a higher incidence of 3 and 5 year old returns compared to the historic beach spawning dominated population (Haggerty *et al*. 2009). Additional age data currently being collected and analyzed help determine the status and trends in age structure of the different aggregates.

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The proposed Criteria for the VSP parameters set in the recovery plan for Lake Ozette sockeye (NMFS 2009) are:

Abundance: 31,250 – 121,000 spawners, over a number of years

Productivity: Population growth rate stable or increasing

Spatial Structure: Multiple spatially distinct and persistent spawning aggregations across the historical range of the population

Diversity: One or more persistent spawning aggregations from each major genetic and life history group historically present within the population.

There is sufficient data to determine that the abundance is well below the desired lower bound. Over the last few decades, productivity appears to have temporarily trended up and down but overall has remained stable around 1. Defining a historical base line and assessing the current state of the spatial structure and diversity of the population is difficult due to a paucity of data. However, a growing tributary data set (MFM 2015) and improvements in the beach spawner surveys (Haggerty & Makah Fisheries Management 2013, 2014) will likely provide a better understanding of these parameters in the future. It appears that the Umbrella Creek Hatchery program has successfully introduced a tributary spawning aggregate. This has increased the spatial and possibly genetic structure of the population and by increasing the diversity of age at return reduced the consequences of an isolated poor year. Also, the addition of the tributary aggregate has likely increased overall abundance although this is not yet obvious in the abundance trends. However, the beach spawning aggregate is considered the core group of interest for recovery (NMFS 2009), and while it appears that the beach spawning aggregate has recovered from a substantial decline during the mid to late 2000's, the current numbers of beach spawners is well below historic levels and restricted to a subset of historic spawning beaches.

Straying of tributary fish into the beach spawning locations may pose a threat to the beach spawning aggregate given that the tributary spawning aggregate has recently been much larger than the beach spawning aggregate. However, to date there appears to be little exchange between the beach spawning and tributary spawning aggregates. The estimated proportion of beach spawners that are hatchery origin has been very low, with pHOS estimates ranging from 0.5% to 0.8% (MFM 2015). In addition 1) there is evidence that tributary and beach spawning aggregates coexisted in the past 2) the source of the hatchery program was Ozette fish, 3) the hatchery brood stock is currently naturally spawning tributary fish, 4) there is little evidence of resource limitations in the lake for rearing and 5) the level hatchery intervention into the natural sockeye salmon life cycle is minimal (egg boxes producing fry). However, interactions between these two aggregates should continue to be monitored.

Assessment of Lake Ozette Sockeye population status is substantially hampered by gaps in our knowledge of population abundance and structure. While improvements in monitoring over the last decade or two have been welcome, there is still a great deal of uncertainty.

Recommendations:

- Improve estimates of total population size. This is necessary both for developing a population level abundance series and dividing the population into the different aggregates. Efforts should be directed towards adopting the imaging sonar for estimating the total sockeye salmon run entering Lake Ozette. This approach would have a much smaller effect on fish movement relative to the upriver weir placed each year to count sockeye salmon, and can be used in the early part of the run.

- Improve estimates of the tributary spawners. Specifically re-evaluate the mark-recapture methodology for Umbrella Creek estimates and develop a method for estimating run size in the other tributaries.
- Continue efforts to enumerate the beach spawning aggregates with the goal of moving from an index to an abundance estimate. Also an occasional spatially extensive survey would provide a more concrete picture of distribution.
- Develop and implement regular sampling to estimate hatchery fraction and age structure for each of the aggregates. Also investigate alternative approaches for estimating overall hatchery origin and age structure.

UPDATED BIOLOGICAL RISK SUMMARY

Abundance of Lake Ozette sockeye has not changed substantially from the last status review. The quality of data continues to hamper efforts to assess more recent trends and spatial structure and diversity although this situation is improving. Based on this review, there is no evidence to suggest a change in the biological risk category.

OREGON COAST COHO SALMON ESU

BRIEF DESCRIPTION OF ESU

The Oregon Coast coho salmon ESU (OC ESU) consists of coho salmon populations on the Oregon coast from Cape Blanco to the mouth of the Columbia River (Figure 127). The geographic area is physically diverse, and includes numerous rocky headlands and an extensive area with sand dunes. Most rivers within the ESU drain the west slope of the Coast Range, with the exception the Umpqua River, which extends through the Coast Range to drain the Cascade Mountains (Weitkamp *et al.* 1995). While most coho salmon populations within the ESU use stream and riverine habitats, there is extensive lake rearing by juvenile coho salmon in several large lake systems.

The Oregon and Northern California Coasts Technical Recovery Team (ONCCTRT) evaluated the historical population structure of the 56 populations that were likely to have been present historically within the ESU (Lawson *et al.* 2007). This was conducted with a simple conceptual model of population demographics, which classifies populations based on their isolation and persistence. Populations that appeared likely to have been capable of persisting in isolation were classified as independent (21 populations). Small populations in smaller coastal basins may not have be able to maintain themselves continuously for periods as long as hundreds of years without strays from adjacent populations and were classified as dependent populations (36 populations; Lawson *et al*. 2007).

The ONCCTRT used the substantial genetic and biogeographic structure within the ESU to identify biogeographic strata among populations (Lawson *et al*. 2007). These strata represent the genetic and geographic similarities among populations such that preservation of sustainable populations within each stratum will conserve major genetic diversity within the ESU, and spread risks of losing genetic and geographic diversity due to catastrophes (Wainwright *et al.* 2008). The ONCCTRT determined that the four monitoring areas (North Coast, Mid Coast, Mid-South Coast, Umpqua) identified by ODFW for Oregon coast coho salmon, in addition to the lakes complex identified by Ford *et al*. (2004), reflected the geography, ecology and genetics of the landscape (Lawson *et al*. 2007). Accordingly, the five strata each contain between 3 (Lakes) and 29 (Mid Coast) populations (Figure 127).

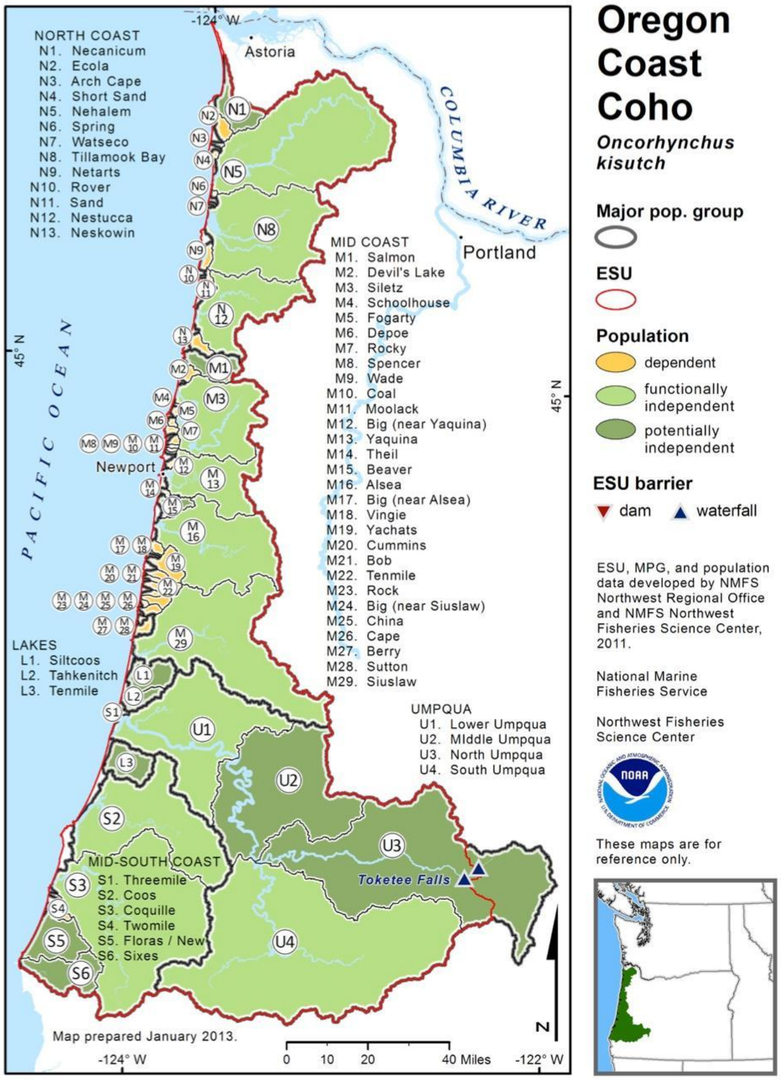

Figure 127 -- Map of the Oregon Coast coho salmon spawning and rearing areas, illustrating populations and major population groups.

SUMMARY OF PREVIOUS STATUS CONCLUSIONS

2005

The 2005 status review conclusions for the ESU as a whole reflected ongoing concerns for the long-term health of this ESU: a majority of BRT opinion was in the "likely to become endangered" category, with a substantial minority falling in the "not likely to become endangered" category (Good *et al*. 2005). Although they considered relatively high returns in 2001 and 2002 to be encouraging, most members thought that the factor responsible for the increases was more likely to be unusually favorable marine productivity conditions rather than improvement in freshwater productivity. The majority of BRT members thought that to have a high degree of confidence that the ESU was healthy, high spawner escapements should be maintained for a number of years and the freshwater habitat should demonstrate the capability of supporting high juvenile production from years of high spawner abundance.

The 2005 status review considered the long-term decline in productivity to be the most serious concern for this ESU. With all directed harvest for these populations eliminated, harvest management (i.e., reducing harvest rates) could no longer compensate for declining productivity. The BRT was concerned that the long-term decline in productivity reflected deteriorating conditions in freshwater habitat and that the ESU would likely experience very serious risks of local extinctions during the next cycle of poor ocean conditions. With the cushion provided by strong returns in 2001–2003, the 2003 BRT had much less concern about short-term risks associated with abundance than did earlier BRTs.

2010

A thorough status review for Oregon Coast coho was conducted by Stout *et al.* (2012) in response to a delisting petition. In that review, the overall assessment of extinction risk to the ESU, taking into account both the demographic risk parameters and an evaluation of threats, indicated considerable uncertainty about its status, with the BRT assessment evenly split between moderate risk and low risk at 47% each, and a small minority of weight (6%) at high risk. This uncertainty was due largely to the difficulty in balancing the clear improvements in some aspects of the ESU's status over the prior approximately 15 years (increased abundance, lower harvest rates, reduced hatchery risks) against persistent threats potentially driving the longer term status of the ESU (habitat degradation, climate change), which probably had not changed over the same time frame and were predicted to degrade in the future. In addition, the BRT noted that accurately predicting the long-term trend of a complex system is inherently difficult, and this also led to some uncertainty in the overall risk assessment.

DECISION SUPPORT SYSTEM FOR OREGON COAST COHO SALMON

The ONCCTRT developed a knowledge-based Decision Support System (DSS) for the Oregon Coast coho salmon ESU (Wainwright *et al*. 2008). The DSS was designed to evaluate the biological sustainability of the entire ESU, where 'biological sustainability' implies that "a population is able to survive prolonged periods of adverse environmental conditions, while maintaining its genetic legacy and long-term adaptive potential" (Wainwright *et al.* 2014) p. 278). The DSS consists of a suite of biological criteria that contribute to ESU sustainability. These criteria were expressed as logical propositions that could be evaluated from empirical data or professional judgment. At the lowest level, propositions were evaluated from data collected at the watershed or population scale, population-scale combinations were aggregated at the stratum scale, and finally to the entire ESU (Wainwright et al. 2008, Wainwright *et al*. 2014).

The DSS uses a diverse array of biological criteria to evaluate ESU biological status. This list includes: watershed- and population-level spawner and juvenile occupancy and distributions, population-specific productivity, probability of persistence (from population viability models), spawner abundance, artificial influence; and ESU-wide genetic and phenotypic diversity (Wainwright *et al*. 2008). Accordingly, the DSS includes specific criteria for most of the categories discussed on the following pages to evaluate the current status of Oregon Coast coho salmon.

We provide scores from two evaluations of the DSS as indicators of whether particular attributes of the ESU have been improving or declining, in addition to values and trends in actual data on population attributes (e.g., spawner abundance, marine survival). The next DSS assessment (after the that of Wainwright *et al.* 2008) was conducted as part of the 2012 BRT evaluation, which included data through the 2009 return year (Stout *et al*. 2012). The most recent DSS run was conducted by Mark Lewis (ODFW), and used data through the 2014 return year (Lewis 2015). Scores provided here for the 2012 evaluation were calculated by M. Lewis and differ slightly from those found in Stout *et al*. (2012) due to changes in GIS coverage (which changes 5th field watershed boundaries), and other issues with the 2012 assessment identified in Lewis (2015). These changes allow direct comparison of the two DSS assessments, which was previously not possible due to methodological differences. Direct comparisons to the original 2008 assessment are presently not possible due to these methodological differences.

In using the DSS to evaluate current levels of ESU persistence and sustainability, it should be noted that three criteria have not been updated since first calculated by Wainwright *et al*. (2008). First, population-level probability of persistence (PP-2) requires results from four population viability models, and has not been updated. Part of the rationale for not updating this parameter is that the relative vulnerabilities of populations assessed by the PVA models are unlikely to change with the addition of a few more years of data (Stout *et al*. 2012). Second, population functionality (PF-1) is based on habitat quantity, and was not updated by Stout *et al*. (2012) because it would have required a major reanalysis of habitat data. Instead, Stout *et al*. (2012) did an analysis of habitat data to look for trends in habitat quality; no such analysis was conducted for this current review. Third, the ESU-level criteria for diversity (ED-1, ED-2, ED-3) have also not been updated since the DSS was originally evaluated because they relied on professional judgment (Wainwright *et al*. 2008); recent increases in abundance and productivity across all strata suggest ESU diversity has not decreased. Accordingly, the DSS results provided here for the 2012 and 2015 assessments reflect the original values for PP-2 and the ESU-level diversity criteria, but PF-1 is no longer included in calculations of the whole ESU sustainability and persistence scores (Lewis 2015).

DESCRIPTION OF NEW DATA AVAILABLE FOR THIS REVIEW

The available data for the Oregon coast coho salmon ESU are mainly updates of existing data time series through return year 2014. These include spawner abundances, exploitation rates, estimates of the proportion of wild spawners, and marine survival and an updated assessment of the DSS (discussed above).

One new data series now available is marine survival estimates for wild Oregon Coast coho salmon from the Life Cycle Monitoring (LCM) sites (Suring *et al.* 2012), E. Suring, ODFW unpubl. Data; Figure 128). These marine survival data come from six LCM sites: Nehalem River (North Coast stratum), Siletz, Yaquina, and Alsea Rivers (Mid Coast stratum), Umpqua (Umpqua Stratum) and Coos River (mid-South Coast stratum) (data from the LCM site on the Trask River was not included due to shorter time series). Marine survival rates provided here are estimated from the number of smolts passing downstream through smolt traps and subsequent number of jacks and adults returning 1 and 2 years later, respectively. The coastwide estimate of marine survival is the average survival from all LCM sites adjusted for harvest (E. Suring, ODFW, unpublished data).

ABUNDANCE AND PRODUCTIVITY

Prior to 1940 recruitment of adults to the Oregon Coast ESU is estimated to have averaged about 800,000 fish, ranging from 400,000 to 2,000,000. After 1940 typical recruitments dropped to about 300,000, peaking at 800,000. Another drop following the ocean regime shift in 1976 led to recruitments in the range of 100,000, with a low of 26,000 in 1997 (Stout *et al*. 2012). Spawner escapement has shown a different pattern due to large changes in harvest management. Prior to 1940, ocean and in-river exploitation rates are estimated to have been about 50%. They rose through the 1950s and 1960s with peak exploitation rates between 80% and 90% in the 1970s. Abundance and harvest started to decline in the 1980s until fisheries were closed in 1993 due to extreme low abundance and poor marine survival. During the period from 1955 to 1993 spawner escapements were in the range of 50,000 even as recruitment ranged up to 800,000 fish. Since then, exploitation rates have been in the range of 10% - 30%, and the abundance of spawners has been much more representative of total recruitment. Once harvest was curtailed, it took several years for recruitment to improve, driven by improving marine survival and also, to some extent, reduction in the release of hatchery fish in the ESU (Buhle *et al.* 2009). Recent high spawning escapements up to 350,000 signal a shift from the management practices of the past six decades, and appear to be resulting in the reestablishment of many of the natural processes associated with salmon populations. However, even with the recent period of favorable marine conditions, the maximum ESU production has not reached the levels seen as recently as the mid-1970s (although see discussion below). Now that harvest has become less of a constraint on spawner escapements it has become evident that marine survival is now the principal driver of interannual and interdecadal variation in abundance.

In a recent report, Caldwell and Cramer (2015) argue that these historic estimates of recruit and spawner abundances may be biased high due to methodological changes, resulting in updated pre-1970s population productivity estimates that are similar to levels observed today. They cite several noteworthy methodological changes in how spawner numbers are estimated during this period, which started as peak counts in high quality index areas during 1950-1971, to area-under-the curve (AUC) estimates of total spawners in stratified random sampling plan reaches beginning in 1990. Caldwell and Cramer (2015) point to studies directly comparing different methodologies (e.g., peak counts vs. AUC), which show that resulting spawner abundances were not directly comparable, and results varied among basins and across time.

Similarly, Caldwell and Cramer (2015) identify several issues with how harvest rates on Oregon Coast Natural coho salmon have been estimated, which have likely over-estimated harvest rates during the 1950s and under-estimated the rates during the 1980s and 1990s. These issues include population-specific variation in ocean distributions not accounted for by the OPI harvest rate and changes in harvest effort among areas over time. Because ocean recruits are calculated directly from harvest rates, over-estimates of harvest rates results in over-estimates in recruitment and therefore productivity, and vice versa (under-estimated harvest results in under-estimated recruits).

Because of these problems, Caldwell and Cramer (2015) propose using a recruitment time series developed by Lawson (Lawson 1992) for the 1950s-1980s because it specifically corrects for these harvest issues. This corrected time series indicates recruitment during 1950-1990 that is substantially below other historical estimates. For example, the corrected time series gives recruitment averaging roughly 300,000 during the 1950s (vs. 400,000 in the currently-used historic reconstruction), 200,000 during the 1960s and 1970s (vs. 350,000), and 100,000 in the 1980s (vs. 150,000). Assuming pre-1940 estimates of Oregon coast coho salmon abundance (800,000) are correct, use of the Lawson recruitment time series indicates the timing of the large decrease in abundance occurred before 1950, not afterwards as has been assumed.

Furthermore, Caldwell and Cramer (2015) advocate that declining productivity during the last half-century is not due exclusively to freshwater habitat degradation, but also reflect management practices of high hatchery releases and harvest rates. They argue that these management practices allowed hatchery fish to dominate naturally-spawning populations, which decreased population productivity. Since the 1990s, greatly reduced harvest rates and almost complete elimination of hatchery fish, has allowed the productivity of Oregon Coast coho to rebound (see also Buhle *et al.* 2009). They note that although ocean survival rates have contributed to the recent rebound in OC ESU, marine survival rates over the last decade are still less than half of what they were during 1970-1976.

The spawner abundance of coho salmon within the Oregon Coast ESU varies by time and population. The large populations (recent abundances $\geq$ 20,000 spawners) include the Coos, Coquille, Nehalem, Siletz, Siuslaw, and South Umpqua Rivers (Figure 130). The total abundance of spawners within the ESU has been generally increasing since 1999, with total abundance exceeding 280,000 spawners in three of the last five years (Figure 129). The 2014 Oregon Coast coho salmon return (355,600 spawners) is the highest since at least the 1950's (2011 is the 2nd highest with 352,200; ODFW 2015). Most independent populations show an overall increasing trend in abundance, with synchronously high abundances in 2002-2003, 2009-2011 and 2014, and low abundances in 2007 and 2009 (Figure 130). This synchrony is evidence for the overriding importance of marine survival to recruitment and escapement of coho salmon in the Oregon Coast ESU.

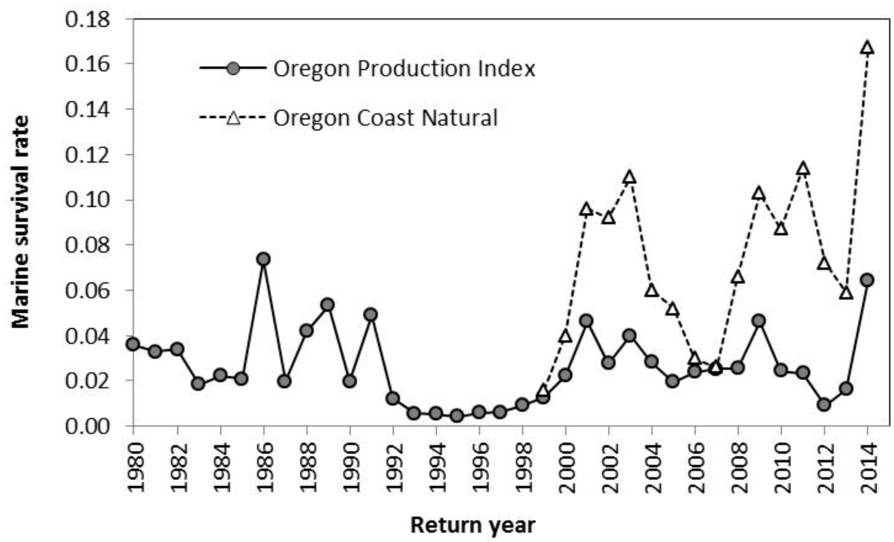

Figure 128 -- Marine survival rates for Oregon Production Index hatchery-produced coho salmon, 1980-2014, and Oregon Coast Natural coho salmon from life cycle monitoring sites, 1999-2014. Data from PFMC 2015, Suring et al. 2012, and E. Suring (ODFW, unpubl. data).

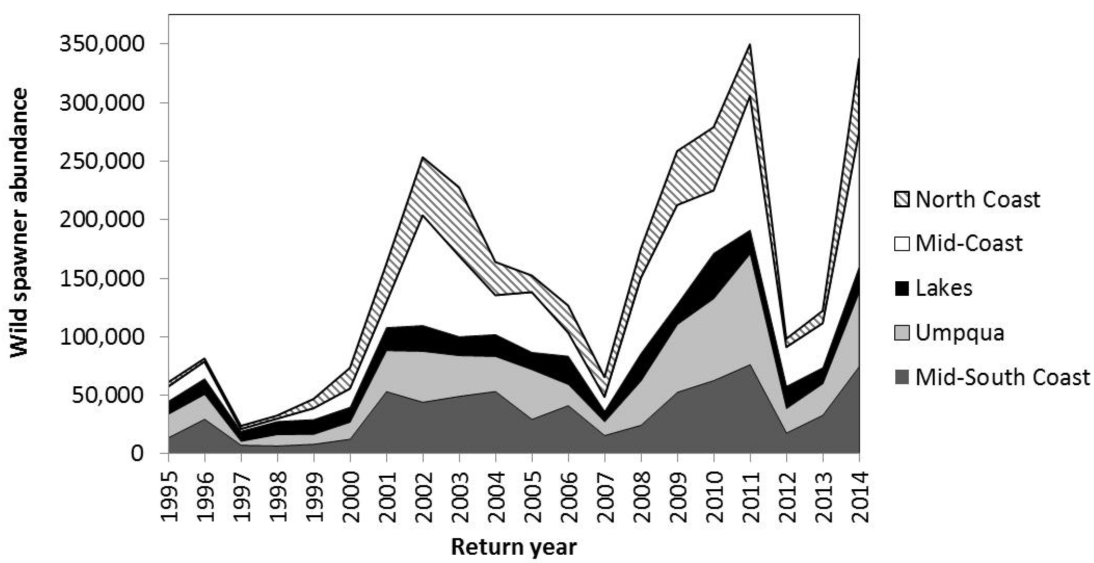

Figure 129 -- . Estimated abundance of wild natural spawners in the five strata for Oregon Coast coho salmon ESU, 1995-2014.

5 year geometric mean wild spawner abundances have increased from 17 – 7228 per population in the 1990 to 1994 time period, to 189 – 23741 for the 2010 to 2014 time period with the highest abundance occurring in the most recent time period (2010-2014). The vast majority of populations (20 of 21) exhibited an increase in the geometric mean abundance between the previous 5 year period (2005-2009) and the current one (2010-2014) (Table 69).

A similar pattern is observed with 15 year trends in log wild spawner abundances: all are positive whether the 1990-2005 or 1999-2014 time period is used. Trends during the earlier 15 year interval (1990-2004) were steeper and no confidence intervals overlapped zero, while the recent trends for all populations are considerably less steep (although all are positive) and the confidence intervals for all but eight populations include zero (Table 70). This increasing abundance trend occurs across all strata in the ESU.

Spawner abundance is captured by two criteria in the DSS, critical abundance (PP-3; mean spawner densities in lower years) and spawner abundance (PD-1; harmonic mean abundance sufficient to avoid genetic risks). Scores for PP-3 increased between the 2012 and 2015 assessments for all populations (with the exception of the Sixes River), and mean scores have increased from 0.40 to 0.66 (Table 68). The number of populations with a score of at least 0.5 for PP-3 (=moderate to high certainty that population abundance is maintained above levels where small-population demographic risks are likely to occur) increased from 10 populations in 2010 to 16 in 2015. For PD-1, scores have either increased (18 populations) or remained constant (3 populations), with no populations showing a decline; across all populations, mean scores increased from 0.24 to 0.26. The number of populations with PD-1 scores exceeding 0.50 (=moderate to high certainty that populations have sufficient spawners to prevent loss of genetic variation) remained constant in the two assessments (6 populations; Table 68).

Marine survival has been highly variable over the last three decades (Figure 128). Marine survival rates for the Oregon Production Index (OPI) are estimated from hatchery coho from the Columbia and Oregon and California coasts. OPI coho are mostly from the Columbia River and subject to in-river as well as marine influences. Marine survival rates for Oregon Coastal natural (OCN) coho are available from ODFW's Life Cycle Monitoring sites starting with the 1999 return (E. Suring, unpubl. data, Suring et al. 2012). In general, marine survival of OCN coho salmon is roughly twice as high (mean survival in 1999-2014 was 0.074) than survival of OPI coho (0.029 during 1999-2014), although in some years the rates are quite similar (e.g., 1999, 2006, 2007). The trends for both times series are increasing since return year 1999 due to low marine survival rates during 1992-1999 and extremely high marine survival for fish returning in 2014 (0.17 for OCN, 0.06 for OPI). However, OPI marine survival during the 1980s (mean = 0.034) is higher than the 1999-2014 period (0.029), and the overall trend is slightly downwards. The two time series are strongly correlated (Pearson correlation $r = 0.76$, $p<0.01$), which suggests a strong common environmental influence on marine survival rates.

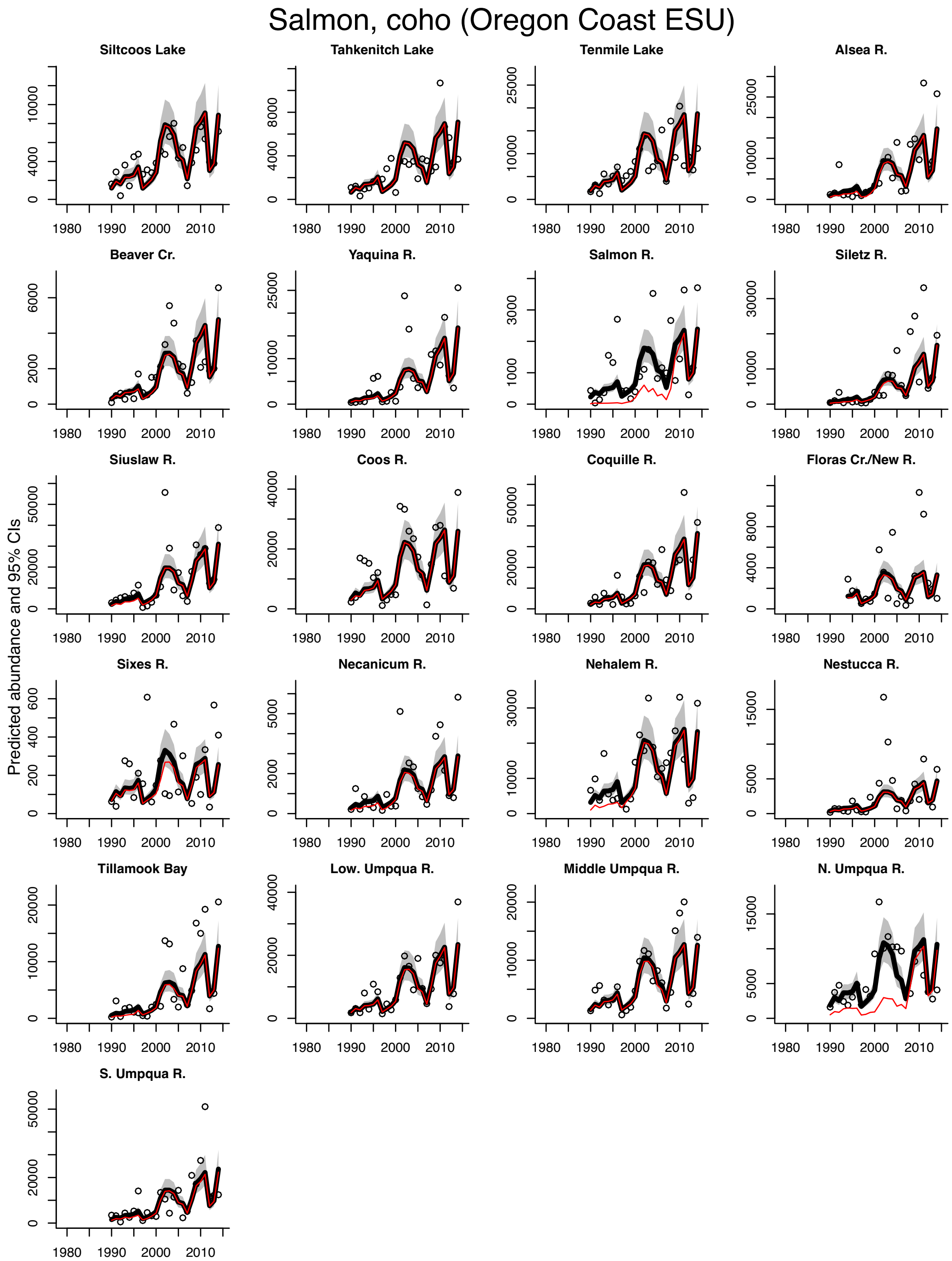

Figure 130 – Smoothed trend in estimated total (thick black line) and natural (thin red line) population spawning abundance. Points show the annual raw spawning abundance estimates.

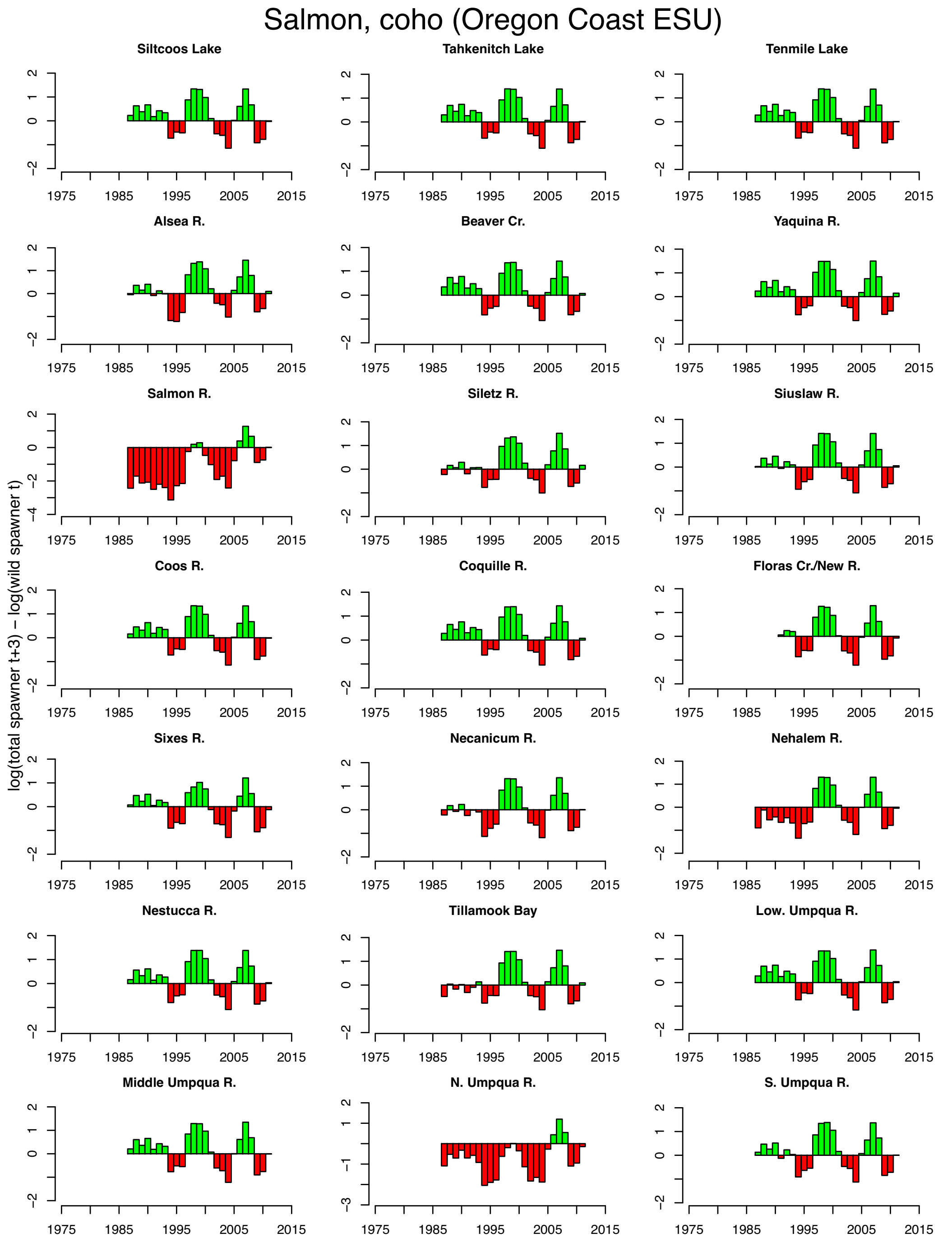

Figure 131 – Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* - smoothed natural spawning abundance in year (*t* – 3).

Table 68 -- Population scores for Oregon Coast coho salmon decision support system criterion for assessments conducted in 2012 (using data through 2009 return year) and 2015 (data through 2014 return year). The criteria are: PP-1—Population productivity (geometric mean of natural return ratio in low years); PP-3—Critical abundance (mean spawner densities in low years); PD-1—Spawner abundance (harmonic mean sufficient to avoid genetic risks); PD-2—Artificial influence (% of hatchery fish on spawning grounds); PD-3—Spawner distributions (>4 fish per mile in half of watersheds); PD-4—Juvenile distributions (pools with ≥ 1 fish). See Wainwright et al. (2008, 2014) for additional details. Also included is the 'minimum level of desired status' of each population under the Oregon Coast Coho Conservation Plan.

		Decision Support System criterion and assessment year												OR Cst Coho
Stratum	Population	PP-1		PP-3		PD-1		PD-2		PD-3		PD-4		Cons Plan
		2012	2015	2012	2015	2012	2015	2012	2015	2012	2015	2012	2015	status*
North Coast	Necanicum	0.95	0.89	0.30	0.68	0.01	0.02	0.35	0.92	0.82	0.92	0.97	0.71	F
North Coast	Nehalem	0.80	0.99	0.81	0.83	0.83	0.87	0.66	0.79	0.45	0.53	0.51	0.78	P
North Coast	Tillamook	0.90	0.95	0.42	0.76	0.12	0.14	0.42	0.79	0.23	0.61	0.64	0.85	P
North Coast	Nestucca	0.82	0.95	0.38	0.43	0.14	0.16	0.92	0.86	0.20	0.52	0.92	0.50	P
Mid Coast	Salmon	-0.51	-0.81	-0.94	-0.71	-1.00	-1.00	-1.00	0.92	0.64	0.57	1.00	1.00	F
Mid Coast	Siletz	0.91	1.00	0.11	0.86	0.08	0.10	0.67	0.93	0.51	0.90	0.93	1.00	P
Mid Coast	Yaquina	0.97	0.89	0.44	0.93	0.30	0.33	0.69	0.93	0.84	0.95	1.00	1.00	P
Mid Coast	Beaver	0.97	0.99	0.93	1.00	0.03	0.04	0.86	1.00	1.00	1.00	1.00	1.00	P
Mid Coast	Alsea	0.63	0.86	0.02	0.68	0.18	0.20	0.97	0.98	0.45	0.85	0.83	1.00	P
Mid Coast	Siuslaw	0.89	0.77	0.07	0.81	0.98	1.00	0.81	0.91	0.53	0.72	0.68	0.89	P
Lakes	Siltcoos	0.81	0.88	1.00	1.00	0.45	0.49	0.99	0.99	1.00	1.00	1.00	1.00	P
Lakes	Tahkenitch	0.69	0.84	1.00	1.00	0.24	0.26	0.95	0.99	1.00	1.00	1.00	1.00	P
Lakes	Tenmile	0.96	0.78	1.00	1.00	0.91	0.97	0.98	0.98	1.00	1.00	-0.36	-0.23	F
Umpqua	Lower Umpqua	0.68	0.76	0.73	0.84	0.80	0.85	0.42	0.93	0.78	0.85	0.61	0.85	P
Umpqua	Middle Umpqua	0.73	0.66	0.22	0.48	0.26	0.28	0.35	0.99	0.25	0.39	0.22	0.66	P
Umpqua	North Umpqua	-0.96	-0.50	0.50	0.89	-0.69	-0.64	-0.96	0.13	-0.52	-0.42	-0.66	-0.64	F
Umpqua	South Umpqua	0.92	0.61	0.64	0.82	0.21	0.24	0.50	0.53	0.06	0.28	0.14	0.35	P
Mid-South Coast	Coos	0.92	0.91	0.58	0.91	1.00	1.00	0.94	0.97	0.73	0.88	0.85	0.70	P
Mid-South Coast	Coquille	0.96	0.92	0.84	0.91	1.00	1.00	0.98	0.96	0.68	0.78	0.80	0.93	P
Mid-South Coast	Floras	0.99	0.88	-0.46	0.14	0.12	0.15	0.81	1.00	0.21	0.74	1.00	1.00	P
Mid-South Coast	Sixes	0.52	0.76	-0.25	-0.35	-0.96	-0.96	0.17	0.74	-0.66	-0.42	-0.42	0.17	F
	Mean score	*0.69*	*0.71*	*0.40*	*0.66*	*0.24*	*0.26*	*0.55*	*0.87*	*0.49*	*0.65*	*0.60*	*0.69*	

*For the Oregon Coast Coho Conservation Plan minimum level of desired status, Necanicum received an "F" due to a negative score for PP-2. This criterion is based on population viability modeling, and has not been updated since 2008 (Wainwright et al. 2008). Tenmile Lake also received a negative score for PD-4 (juvenile distributions), despite having the largest population in the lakes stratum. The negative value was likely due to few pools being sampled for juvenile coho salmon, rather than limited distribution of juveniles.

Table 69 -- 5-year geometric mean of raw natural origin spawner counts. This is the raw total spawner count times the fraction wild estimate, if available. In parentheses, 5-year geometric mean of raw total spawner counts is shown. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural origin spawners available. The geometric mean was computed as the product of counts raised to the power 1 over the number of counts available (2 to 5). A minimum of 2 values were used to compute the geometric mean. Percent change between the most recent two 5-year periods is shown on the far right.

Population	MPG	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014	% Change
Siltcoos Lake	Lakes	1517 (1568)	3433 (3468)	5458 (5481)	3702 (3702)	5550 (5550)	50 (50)
Tahkenitch Lake	Lakes	841 (843)	2175 (2206)	2445 (2445)	2851 (2868)	5502 (5513)	93 (92)
Tenmile Lake	Lakes	2616 (2632)	5420 (5420)	8931 (8931)	9562 (9562)	9988 (10008)	4 (5)
Alsea R.	Mid-Coast	1235 (1851)	525 (1300)	5552 (5800)	6502 (6510)	14104 (14104)	117 (117)
Beaver Cr.	Mid-Coast	347 (347)	654 (767)	2942 (3069)	1637 (1665)	2618 (2618)	60 (57)
Yaquina R.	Mid-Coast	546 (658)	1637 (1978)	5482 (5561)	5629 (5817)	9863 (9863)	75 (70)
Salmon R.	Mid-Coast	17 (267)	44 (645)	263 (1186)	260 (1136)	1448 (1463)	457 (29)
Siletz R.	Mid-Coast	493 (930)	428 (597)	3761 (4278)	9638 (10024)	10697 (10697)	11 (7)
Siuslaw R.	Mid-Coast	3175 (4554)	2323 (3032)	15890 (15890)	11367 (11625)	21648 (21913)	90 (88)
Coos R.	Mid-South Coast	7228 (8150)	4579 (4597)	19956 (20077)	10056 (10116)	15023 (15053)	49 (49)
Coquille R.	Mid-South Coast	3934 (4165)	4118 (4169)	12691 (13099)	15598 (15629)	23741 (23837)	52 (53)
Floras Cr./New R.	Mid-South Coast		898 (1009)	2869 (2978)	863 (883)	3489 (3489)	304 (295)
Sixes R.	Mid-South Coast	103 (111)	147 (159)	133 (180)	118 (127)	189 (192)	60 (51)
Necanicum R.	North Coast	281 (468)	271 (412)	1798 (1897)	1097 (1175)	2077 (2094)	89 (78)
Nehalem R.	North Coast	2474 (7471)	1354 (2934)	20139 (20469)	14507 (15091)	11530 (11647)	-21 (-23)
Nestucca R.	North Coast	352 (412)	595 (678)	5263 (5394)	1319 (1327)	2739 (2790)	108 (110)
Tillamook Bay	North Coast	425 (938)	590 (829)	4503 (5015)	5003 (5117)	8332 (8487)	67 (66)
Low. Umpqua R.	Umpqua	2904 (2976)	4200 (4390)	11326 (11758)	10183 (10944)	12874 (12874)	26 (18)
Middle Umpqua R.	Umpqua	2857 (3039)	1830 (1935)	7912 (8265)	5237 (5689)	8804 (8804)	68 (55)
N. Umpqua R.	Umpqua	900 (2650)	929 (3276)	2724 (11346)	2924 (6488)	4367 (4856)	49 (-25)
S. Umpqua R.	Umpqua	1633 (2295)	3119 (4151)	6866 (7269)	8675 (9106)	18185 (18995)	110 (109)

Table 70 -- 15-year trends in log wild spawner abundance computed from a linear regression applied to the smoothed wild spawner log abundance estimate. Only populations with at least 4 wild spawner estimates from 1980 to 2014 are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year period.

Population	MPG	1990-2005	1999-2014
Siltcoos Lake	Lakes	0.11 (0.06, 0.15)	0.03 (-0.03, 0.09)
Tahkenitch Lake	Lakes	0.12 (0.07, 0.17)	0.04 (-0.01, 0.1)
Tenmile Lake	Lakes	0.12 (0.07, 0.16)	0.04 (-0.02, 0.1)
Alsea R.	Mid-Coast	0.17 (0.11, 0.24)	0.09 (0.02, 0.15)
Beaver Cr.	Mid-Coast	0.13 (0.08, 0.19)	0.07 (0.01, 0.12)
Yaquina R.	Mid-Coast	0.18 (0.12, 0.23)	0.09 (0.03, 0.15)
Salmon R.	Mid-Coast	0.24 (0.18, 0.31)	0.16 (0.08, 0.24)
Siletz R.	Mid-Coast	0.21 (0.17, 0.26)	0.1 (0.05, 0.16)
Siuslaw R.	Mid-Coast	0.16 (0.11, 0.21)	0.06 (0, 0.12)
Coos R.	Mid-South Coast	0.11 (0.07, 0.16)	0.03 (-0.03, 0.09)
Coquille R.	Mid-South Coast	0.14 (0.09, 0.19)	0.06 (0.01, 0.12)
Floras Cr./New R.	Mid-South Coast		0.02 (-0.04, 0.08)
Sixes R.	Mid-South Coast	0.06 (0.02, 0.11)	0.01 (-0.04, 0.07)
Necanicum R.	North Coast	0.16 (0.11, 0.21)	0.05 (-0.01, 0.11)
Nehalem R.	North Coast	0.19 (0.14, 0.24)	0.03 (-0.03, 0.09)
Nestucca R.	North Coast	0.14 (0.09, 0.19)	0.05 (0, 0.11)
Tillamook Bay	North Coast	0.21 (0.17, 0.26)	0.08 (0.03, 0.14)
Low. Umpqua R.	Umpqua	0.12 (0.07, 0.17)	0.05 (-0.01, 0.11)
Middle Umpqua R.	Umpqua	0.11 (0.06, 0.16)	0.04 (-0.02, 0.1)
N. Umpqua R.	Umpqua	0.07 (0.02, 0.13)	0.13 (0.07, 0.19)
S. Umpqua R.	Umpqua	0.16 (0.1, 0.21)	0.06 (0, 0.12)

HARVEST

Oregon coast natural (OCN) coho salmon are part of the Oregon Production Index, and are harvested in ocean fisheries primarily off the coasts of Oregon and Washington. Historically they were also harvested in recreational and commercial troll fisheries from central California to the west coast of Vancouver Island. Canadian coho salmon fisheries were severely restricted in the 1990s to protect upper Fraser River coho, and have remained so ever since. Ocean fisheries off California were closed to coho retention in 1993 and have remained closed ever since. Ocean fisheries for coho off of Oregon and Washington were dramatically reduced in 1993 in response to the depressed status of Oregon Coast natural coho, and ocean fisheries have moved to primarily mark-selective fishing beginning in 1999. The consultation standard for management of ocean fisheries places caps on impact rates that vary with the stock status and have ranged from 8% to 30%. Overall exploitation rates regularly exceeded 60% in the 1980s, but have remained below 20% since 1993 (Figure 132). As discussed above, Caldwell and Cramer (2015) argue that harvest rates on Oregon coho were over-estimated by OPI during the 1950s and under-estimated by the OPI in the 1980s and 1990s. This does not affect the low harvest rates beginning in 1993.

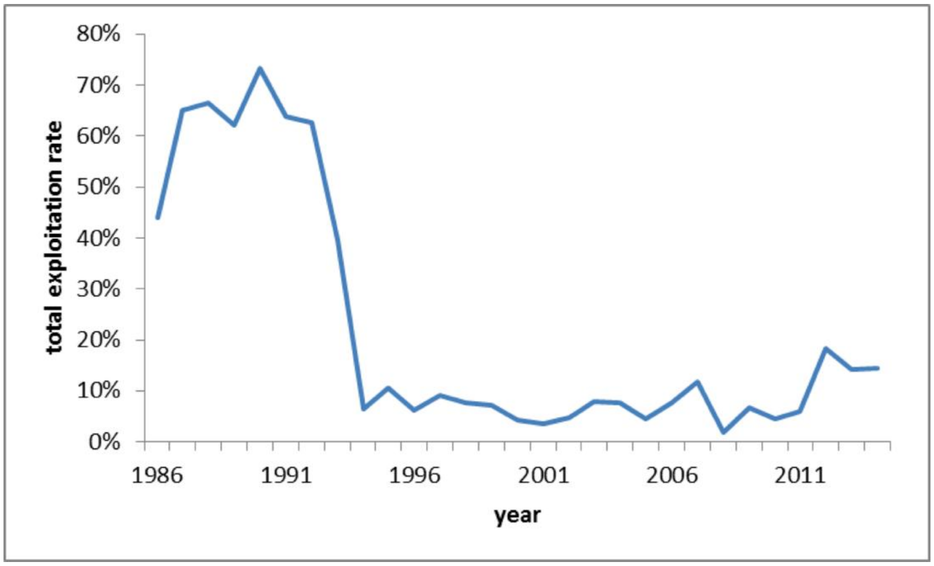

Figure 132 -- Total marine and freshwater exploitation rates on Oregon coast natural coho salmon. Data from FRAM validation runs (STT 2015).

SPATIAL STRUCTURE AND DIVERSITY

Several types of evidence can be used to infer the spatial structure and diversity of coho salmon in this ESU. Taken together, they all indicate that current spatial structure and diversity is similar to previous assessments or improved in some cases (e.g., reduced hatchery influence). Evidence for spatial structure and diversity is provided by several criteria in the DSS, as well directly from trends in spawner abundance and hatchery influence across the geographic range of the ESU.

In the DSS, spatial structure and connectivity are evaluated at the population level with assessments of spawner (PD-3) and juvenile (PD-4) distributions within watersheds, and at the ESU level with an assessment of barriers to migration (ED-1c). Scores for the two population-level distribution metrics, PD-3 and PD-4, both increased between the 2012 and 2015 assessments, indicating improved conditions. The 2012 assessment had mean PD-3 and PD-4 scores of 0.49 and 0.60, respectively, which increased to 0.65 and 0.69, respectively, in the 2015 assessment (Table 68). The number of populations with scores of at least 0.50 (=moderate to high certainty that historically occupied watersheds in the population's range had spawners and juveniles occupying the available habitat) also increased between the two assessments, from 12 to 17 populations (out of 21 populations total) for PD-3 between the 2012 and 2015 assessments, while PD-4 scores of at least 0.50 increased from 16 to 17 populations (Table 68).

The ESU-level assessment of barriers to migration (ED-1c) has not been reevaluated since the original assessment (Wainwright et al. 2008). At that time, it was concluded that there was low certainty that genetic diversity was not compromised by changes in the movements of fish. Increased scores for both spawner (PD-3) and juvenile (PD-4) distributions suggest that it is unlikely that barriers to migration have increased since the original evaluation.

The spatial structure of coho salmon populations within the ESU can also be inferred from population-specific spawner abundances (Figure 130) and productivity (Figure 131). In particular, there is no geographic area or stratum within the ESU that appears to have considerably lower abundances or be less productive than other areas or strata and therefore might serve as a "population sink". Furthermore, if the factors responsible for increasing abundances in independent populations apply equally to dependent populations, then it is unlikely that small populations are being lost at unusually high rates, which is a concern for spatial structure (McElhany et al. 2000).

Criteria for diversity in the DSS evaluated at the population level include spawner abundance sufficient to prevent loss of genetic variation (PD-1) and hatchery influence (PD-2). At the ESU level, diversity is evaluated as genetic diversity (ED-1), and phenotypic and habitat (ED-2) diversity, and loss of small populations (ED-3). As discussed under the Abundance and Productivity section, above, PD-1 scores have increased between the 2012 and 2015 assessments (Table 68).

Scores for PD-2, hatchery influence, had the greatest increase of any metric in the two evaluations, from a mean of 0.55 in 2012 to 0.87 in 2015 (Table 68). In this case, 20 of 21 populations had PD-2 scores exceeding 0.50 (moderate to high certainty that hatchery fish will not have adverse effects on natural populations) with most (15 of 21) having score exceeding 0.90. The population with the lowest PD-2 score (North Umpqua), still showed improvements between the assessments, increasing from -0.96 to +0.13 (Table 68).

While hatchery influence is assessed in the DSS with a truth curve, the direct observation of the consistently upwards trends in the proportion of natural spawners (Figure 133) is straightforward to interpret and perhaps the highest of any ESU reviewed in this report. The State of Oregon made an unprecedented effort to reduce hatchery influence in wild Oregon coast coho salmon populations by greatly reducing the production of hatchery coho salmon along the coast. The result of this action is all but one independent population in the OC ESU currently have a 5 year average of $\geq$98% of wild spawners (Table 71). The sole exception is the North Umpqua, which has greatly reduced hatchery influence compared to previous reviews, but still has a 5 year average of 88% wild spawners. Like the abundance data, this minimal level of hatchery influence occurs across all strata in the OC ESU.

The three metrics for ESU-level diversity (ED-1, ED-2, ED-3) were based on the professional judgment of BRT members during the original assessment (Wainwright *et al*. 2008). They have not been re-evaluated since they were first developed. At that time, the conclusions were that there was low certainty that ESU-level genetic diversity was sufficient for long-term sustainability in the ESU (ED-1a), it was uncertain that human-driven selection was not decreasing genetic diversity (ED-1b), and there was low to moderate certainty that phenotypic diversity was present within the ESU at levels comparable to healthy ESUs or the historical template (Wainwright *et al.* 2008). Observed upward trends in abundance and productivity and downward trends in hatchery influence discussed earlier make decreases in genetic or life history diversity or loss of dependent populations in recent years an unlikely outcome.

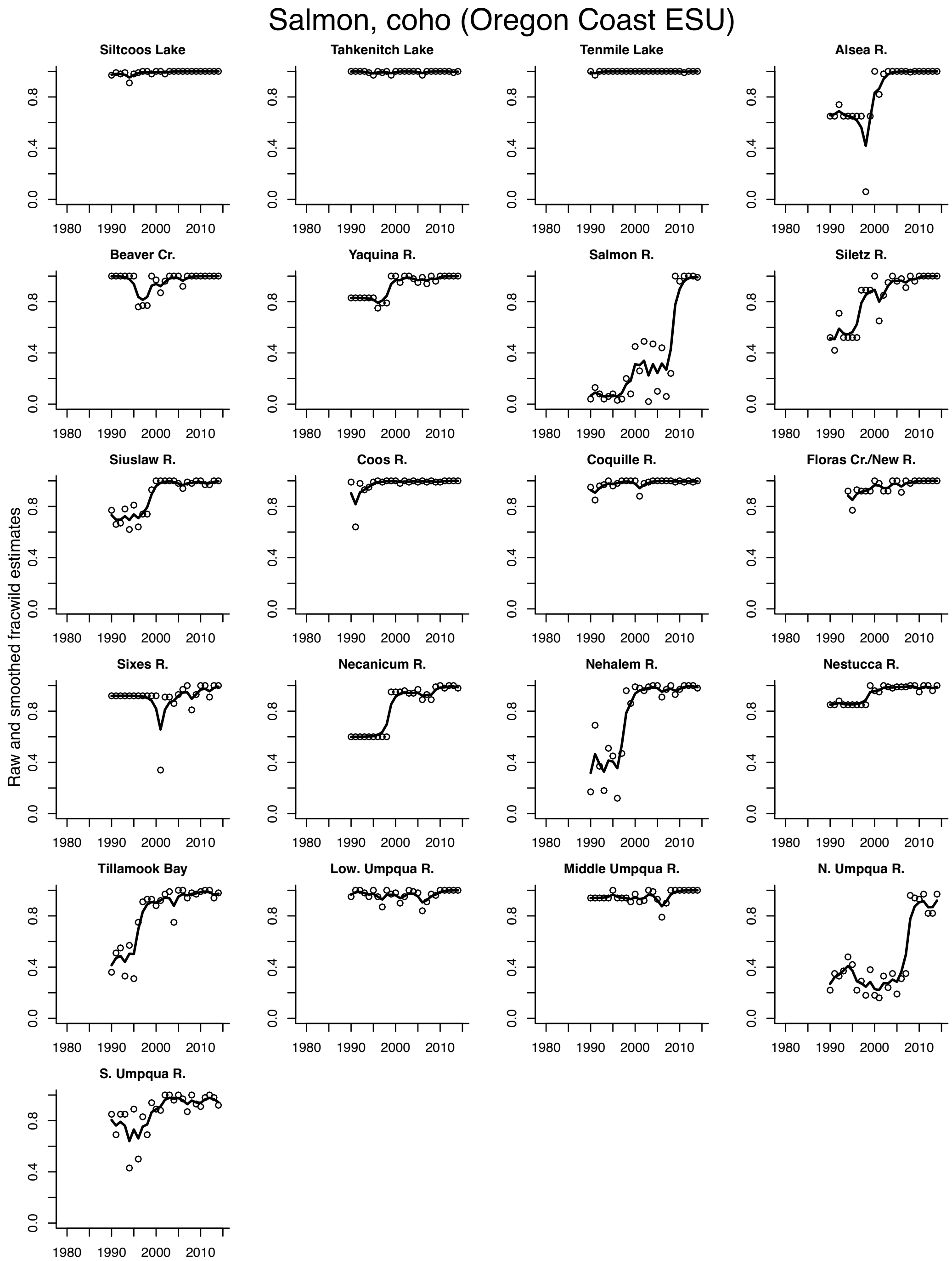

Figure 133 – Smoothed trend in the estimated fraction of the natural spawning population consisting of fish if natural origin. Points show the annual raw estimates.

Table 71 – 5-year mean of fraction wild (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	1990-1994	1995-1999	2000-2004	2005-2009	2010-2014
Siltcoos Lake	0.97	0.99	1.00	1.00	1.00
Tahkenitch Lake	1.00	0.99	1.00	0.99	1.00
Tenmile Lake	0.99	1.00	1.00	1.00	1.00
Alsea R.	0.67	0.53	0.96	1.00	1.00
Beaver Cr.	1.00	0.86	0.96	0.98	1.00
Yaquina R.	0.83	0.83	0.99	0.97	1.00
Salmon R.	0.07	0.09	0.34	0.37	0.99
Siletz R.	0.54	0.74	0.89	0.96	1.00
Siuslaw R.	0.70	0.77	1.00	0.98	0.99
Coos R.	0.90	1.00	0.99	0.99	1.00
Coquille R.	0.95	0.99	0.97	1.00	1.00
Floras Cr./New R.	0.92	0.89	0.96	0.98	1.00
Sixes R.	0.92	0.92	0.79	0.93	0.98
Necanicum R.	0.60	0.67	0.95	0.93	0.99
Nehalem R.	0.38	0.57	0.98	0.96	0.99
Nestucca R.	0.86	0.88	0.98	0.99	0.98
Tillamook Bay	0.46	0.77	0.90	0.98	0.98
Low. Umpqua R.	0.98	0.96	0.96	0.93	1.00
Middle Umpqua R.	0.94	0.95	0.96	0.92	1.00
N. Umpqua R.	0.35	0.30	0.25	0.55	0.90
S. Umpqua R.	0.73	0.77	0.95	0.95	0.96

BIOLOGICAL STATUS RELATIVE TO RECOVERY GOALS

The DSS for Oregon Coast coho salmon was specifically developed to evaluate biological recovery criteria for the entire ESU at two levels, persistence (EP) and sustainability (ES), which imply different levels of risk (Wainwright *et al.* 2008). The persistence analysis evaluates the ability of the ESU to persist (i.e., not go extinct) over a 100-year period without artificial support, including the ability to survive prolonged periods of adverse environmental conditions that may be expected to occur during a 100 year time frame. It is based on population productivity, probability of persistence, and abundance relative to critically low thresholds (Stout *et al.* 2012).

The sustainability analysis evaluates the ability of the ESU to maintain its genetic legacy and long-term adaptive potential for the foreseeable future. Sustainability implies stability of habitat availability and other conditions necessary for the full expression of the population's (or ESU's) life history diversity into the foreseeable future. Criteria used to evaluate population sustainability are objective measures of spawner abundance, artificial influence, spawner and juvenile distributions, and habitat capacity. It also includes ESU-level measures of genetic diversity, phenotypic and habitat diversity, and small populations.

The most recent overall scores from the DSS (using data through return year 2014) show improvement over previous iterations for both ESU persistence and sustainability. The most recent EP value is 0.73 (high certainty the ESU is likely to persist), compared to values of 0.44 for the 2012 assessment (moderate to high certainty the ESU is likely to persist). For ESU sustainability, the current value is 0.29 (moderate certainty the ESU is sustainable), which is also higher than values resulting from previous assessments (0.23, or low to moderate certainty the ESU is sustainable).

Federal Recovery Plan

A proposed federal recovery plan for Oregon Coast coho salmon was released in October 2015 (NMFS 2015c). The aim of the plan is to, "... establish self-sustaining, naturally spawning Oregon Coast coho salmon populations that are sufficiently abundant, productive, and diverse to persist in the long term, defined as the next 100 years. The species needs to be resilient enough to survive catastrophic changes in the environment, including events such climate change and decreases in ocean productivity. Overall, the recovery direction for Oregon Coast coho salmon has a single overriding focus: restoring degraded habitat and the ecosystem processes that affect the habitat. Most recommended actions target the protection and restoration of freshwater and estuarine habitats, especially habitats that support juvenile rearing coho salmon" (NMFS 2015b, p. ES-2). The Plan states that the federal government will remove Oregon Coast coho salmon from ESA listing when it determines that: 1) the species is sufficiently recovered from a biological perspective, and 2) factors that led to listing have been reduced or eliminated to the point where federal protection under the ESA is no longer needed.

The biological status of the ESU will be evaluated by the DSS (described previously), and must meet two criteria: 1) most of the independent populations have to be sustainable in each stratum, and 2) all five strata have to be sustainable for the whole ESU to be sustainable. The DSS elements considered in this assessment include spawner abundance (PD-1), spawner distribution (PD-3), juvenile distribution (PD-4), critical abundance (PP-3), population productivity (PP-1), and aritificial influence (PD-2).

Delisting considers both biological status and the five status of listing factors/threats (status of habitat, over utilization, disease or predation, adequecy of regulatory mechanisms, and other factors). Delisting will be guided by two principles: 1) the biological recovery criteria should provide at least a moderate certainty that the ESU is sustainable, and 2) we need to be reasonably certain that the relevant regulatory mechansims are "adequate" to protect Oregon Coast coho salmon. The current DSS scores (described above, Lewis 2015) show that there is moderate certainty the ESU is sustainable. However, there are concerns for listing factors related to habitat and regulatory mechanisms, as discussed in the recovery plan.

Oregon Recovery Plan

The State of Oregon developed an Oregon Coast Coho Conservation Plan (OCCCP) in 2007 to "ensure the continued viability of the Oregon Coast Coho Evolutionary Significant Unit (ESU) and to achieve a desired status that provides substantial ecological and societal benefits" (ODFW 2007), p. 3). The plan relies on a combination of existing regulatory programs plus effective long-term participation in non-regulatory conservation work to achieve desired status. The OCCCP defines the desired status for the ESU, which will be evaluated using six measureable criteria that pertain to population abundance, persistence, productivity, distribution, diversity, and habitat. The goal of the Conservation Plan will be met when: 1) all independent populations pass the six measurable criteria for independent populations and 2) the aggregate of dependent populations within a biogeographic stratum pass the two measurable criteria for dependent populations.

The plan recognizes that positive improvement may occur before full desired status is achieved. Therefore, the plan defines a *minimum level of desired status* as, "All 21 independent populations pass all the sustainability criteria (as defined by the Oregon/Northern California Coast TRT). A pass is defined as any positive truth value for the individual criteria, a fail is a truth value ≤ 0.0. Populations that currently pass (as defined in previous sentence) must maintain or improve upon their current scores" (ODFW 2007, Appendix II p. 2). The latest iteration of the DSS (using data through return year 2014) indicates that five independent populations do not meet this criterion (Population Sustainability < 0.0) (Lewis 2015). These populations are the Necanicum, Tenmile Lake, Salmon, N. Umpqua, and Sixes River; these same four populations were the only populations to have PS scores below 0.0 in previous runs of the DSS using earlier data (Wainwright *et al*. 2008, Stout *et al*. 2012). Accordingly, the ESU has not yet reached a minimum level of desired status based on these criteria.

UPDATED BIOLOGICAL RISK SUMMARY

Many positive improvements to Oregon Coast coho salmon are described by ODFW (2015), including positive long-term abundance trends and escapement. Increases in ESU scores for persistence and sustainability also clearly indicate the biological status of the ESU is improving, due in large part to management decisions (reduced harvest and hatchery releases) and favorable environmental variation (i.e., high marine survival). However, as Lawson (1993) stated over two decades ago, "The true measure of success for such [stream restoration] projects is the continued survival of the population through subsequent episodes of low abundance" (Lawson 1993, p. 6), when discussing cycles in ocean productivity, habitat restoration, and the productivity of Oregon Coast coho salmon. Lawson (1993) cautioned that variation in ocean productivity can mask the true benefits of stream restoration projects; increased abundances are incorrectly attributed to stream restoration when the increases resulted from high marine survival. Consequently, it is only when marine survival is low that it becomes apparent whether habitat quality and quantity are sufficient to support self-sustaining populations. With marine survival rates expected to decrease for Oregon Coast coho salmon entering the ocean in 2014 (Peterson *et al*. 2014), 2015 and 2016 (see next chapter), it may be advisable to wait to observe how populations fare during this potential downturn before deciding to change their status.

RECENT TRENDS IN MARINE AND TERRESTIAL ENVIRONMENTS AND THEIR LIKELY INFLUENCE ON PACIFIC SALMON IN THE PACIFIC NORTHWEST

INTRODUCTION

The current status of listed Pacific salmon populations is influenced by numerous factors, including human activities (e.g., fishing mortality, hatchery production, habitat restoration and degradation), and natural variation in environmental conditions in both freshwater and marine environments. The increasing trends in natural spawners seen for some DPS at least partially reflect favorable environmental conditions in marine waters of the northern California Current and in freshwater habitats in recent years. It is well established that ocean conditions during the first weeks or months of marine life have a large influence on overall marine survival for salmon (Pearcy 1992; Pearcy & McKinnell 2007). Accordingly, a large portion of the short-term variation in population productivity may be due to ocean conditions, which fluctuate at short time scales. For example, marine survival can vary by over an order of magnitude between years (Lindley *et al.* 2009).

These productive conditions resulted in high freshwater and marine survival rates and subsequent high adult returns for many salmon stocks throughout the Pacific Northwest at various times especially in the the early 2010s. However, changes in ocean and freshwater conditions beginning in early 2014 due to exceptionally warm ocean waters and associated terrestrial impacts, plus a strengthening El Niño event, suggest that this period of high marine survivals will not persist, and salmon returns in the next few years may be considerable lower than those experienced recently.

This chapter summarizes what is known about marine and terrestrial conditions since the development of the "warm blob" in winter, 2013/2014, and their likely influence on salmon productivity in the Pacific Northwest. Although our understanding of how environmental conditions influence salmon survival has greatly increased in recent years (Quinn 2005a; Pearcy & McKinnell 2007; Crozier *et al.* 2008b), it is impossible to predict exactly how the currently anomalous conditions will affect individual salmon populations. It is also not known how long these unfavorable conditions will last.

METHODS

We use a variety of published and unpublished sources to document the current anomalous conditions in both freshwater and marine environments. Given the recent onset of these conditions (late fall 2013), only a few peer-reviewed papers have been published on the phenomena to date. For marine conditions, our primary sources are the NWFSC's Ocean Indicators annual report (Peterson et al. 2014), the State of the California Current Report (CCIEAT 2015), the Fisheries and Oceans Canada report on Pacific marine ecosystems in 2014 (Chandler *et al.* 2015) and Bond et al. (Bond *et al.* 2015). Information on freshwater conditions include NOAA's National Center for Environmental Information (NOAA NCEI), U.S. Geological Survey's National Water Information System, and U.S. Department of Agriculture's Natural Resource Conservation Service (USDA NRCS).

Our intent with this summary is not to provide an exhaustive review of what is known about current conditions, but instead provide an overview, with a particular emphasis on environmental factors that are important to salmon productivity and survival. In many cases, current environmental conditions in marine and freshwater habitats are outside the range of prior observations, therefore their biological effects are difficult to predict. Only in hindsight will we be able to tell how these conditions affected salmon survival.

RESULTS

OBSERVED ENVIRONMENTAL CONDITIONS

Environmental conditions in both fresh and marine waters inhabited by Pacific Northwest salmon are influenced, in large part, by two ocean-basin scale drivers, the Pacific Decadal Oscillation (PDO; Mantua et al. 1997) and the El Niño-Southern Oscillation (ENSO). Starting in late 2013, however, abnormally warm conditions in the Central NE Pacific Ocean known as the "warm blob" (Bond *et al*. 2015) has also had a strong influence on both terrestrial and marine habitats. Here, we briefly describe the features as they affect both marine and terrestrial environments.

The Warm Blob

Marine waters in the North Pacific ocean have been warmer than average since late fall 2013, when the "warm blob" first developed in the central Gulf of Alaska (Bond *et al*. 2015). The warm blob was caused by lower than normal heat loss from the ocean to the atmosphere and of relatively weak mixing of the upper ocean, due to unusually high and persistent sea level pressure. Temperature anomalies of the near-surface (upper ~100 m) waters exceeded 3°C in January 2014, or 4 standard deviations (Freeland & Whitney 2014). These anomalies were the greatest observed in this region and season since at least the 1980s and possibly as early as 1900 (Bond *et al*. 2015).

The region of warm SST anomalies was isolated to offshore waters during winter 2013-14 (Figure 134). It spread into the coastal domain of Alaska and northern British Columbia in May 2014, and then into the nearshore waters of the Pacific Northwest in September 2014, causing rapid increases in SSTs (Chandler *et al*. 2015). For example, surface temperatures recorded at Stonewall Bank (NOAA Buoy 46050; 20 nautical miles west of Newport, Oregon), increased by 5.6°C over a 21 hour period on September 14-15, 2014 (Figure 135), as the warm blob moved ashore (www.ndbc.noaa.gov/). Sea surface temperatures across the NE Pacific have continued to be 1-3°C above average during winter and spring of 2015 (http://polar.ncep.noaa.gov/sst/ophi/).

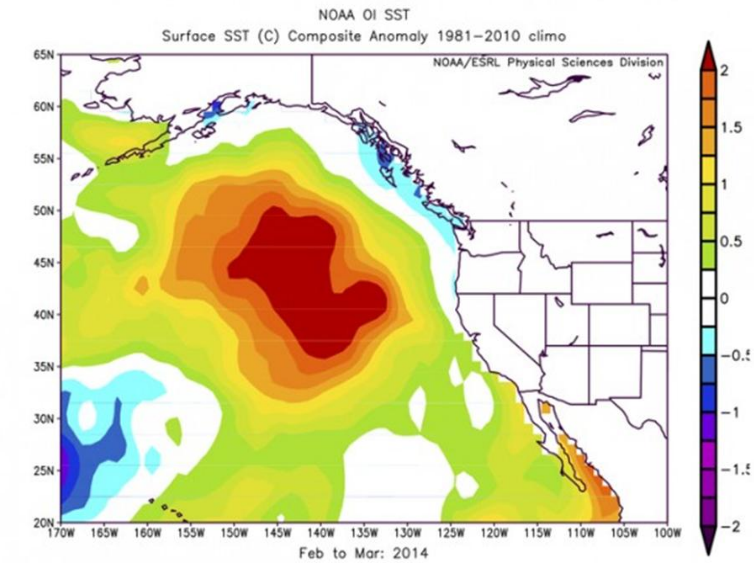

Figure 134 -- Mean sea surface temperature anomalies in the Northeast Pacific Ocean during February and March 2014 showing the warm water associated with the warm blob. (NOAA)

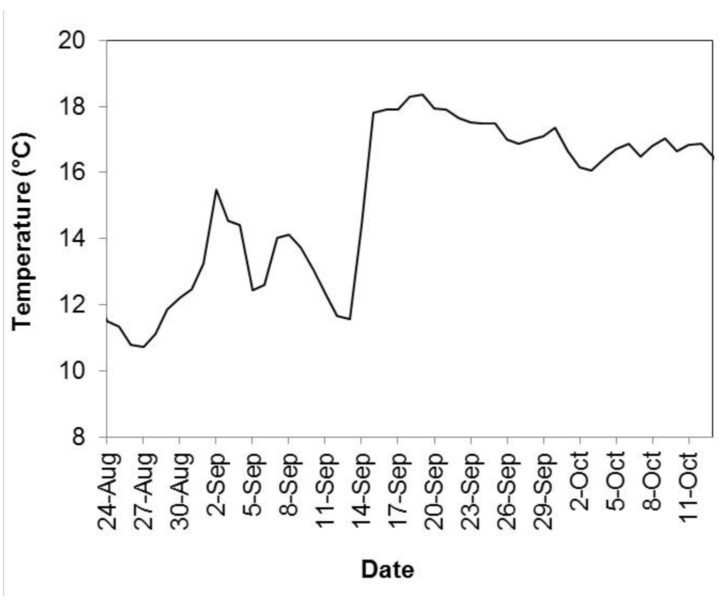

Figure 135 -- Sea surface temperatures recorded at Stonewall Bank (NOAA Buoy 46050; 44°39'22" N 124°31'33" W) on 24 August -12 October 2014, showing the rapid rise in temperature on 13-14 September 2014 as the 'warm blob' moved on shore.

Pacific Decadal Oscillation

The PDO describes the most prominent mode of variability in the North Pacific sea surface temperature field (Mantua *et al.* 1997). Positive values are characterized by warm SSTs along the West Coast of North America and cold SSTs in the central North Pacific, while negative values have the opposite pattern (cold along the coast and warm in the central North Pacific). The PDO also influences freshwater habitats, especially during winter. Positive PDO values are associated with warm and dry PNW winters (especially for the Interior Columbia River Basin (ICRB)) and therefore low snowpack, while negative values are associated with cold wet winters throughout the PNW (high snowpack) (Mantua *et al*. 1997).

Because the PDO is a measure of SSTs and the eastern North Pacific Ocean has been extremely warm, it has been positive since January 2014. It reached the highest monthly levels ever observed during December 2014 (+2.51), and January (+2.45) and February (+2.3) 2015 (Figure 136). As long as marine water remains water along the West Coast, the PDO will remain positive. Current forecasts of global water temperatures (from the NOAA NCEP coupled forecast system model version 2[39]) indicate SSTs along the West Coast will remain 0.5-1°C above average through the period of forecast (Mar-May 2016). If this occurs, the PDO will remain positive at least through spring 2016, or perhaps longer (N. Mantua, NOAA Fisheries, pers. comm.).

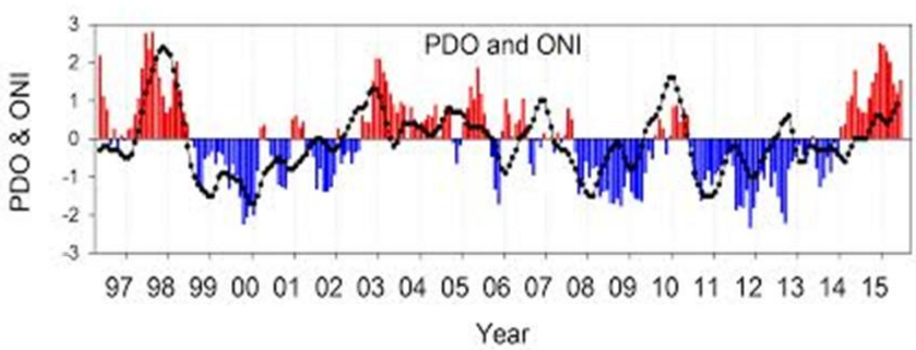

Figure 136 -- Time series of the Pacific Decadal Oscillation (PDO; red and blue vertical bars) and Oceanic El Niño Index (ONI; black line) during 1996-2015. The PDO shifts between positive (warm) to negative (cold) values at roughly decadal scales and has been positive since January 2014, while the ONI has a higher frequency. Figure from B. Peterson (NWFSC).

El Niño-Southern Oscillation

El Niño-Southern Oscillation (ENSO) is a tropical phenomenon that influences climate patterns around the globe. Much like the PDO, the warm phase (El Niño) is characterized by warm SSTs along the West Coast of North America, while negative values (La Niña) produce cold SSTs along the coast. Like the PDO, ENSO also influences terrestrial environments, and Pacific Northwest winter snowpack is low during warm El Niño events and high during cool La Nina years.

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[39] http://www.cpc.ncep.noaa.gov/products/CFSv2/CFSv2seasonal.shtml

The Oceanic Niño Index (ONI) is the three-month running-mean SST departures in the Niño 3.4 region (http://www.cpc.ncep.noaa.gov/). El Niño events are defined as positive ONIs greater than or equal to +0.5°C, while La Niña events have a negative ONI less than or equal to -0.5°C. These thresholds must be exceeded for a period of at least 5 consecutive overlapping 3-month seasons. The ONI first exceeded +0.5°C during the September-October-November period, and has remained above 0.5°C since then. Based on this criterion, a weak El Niño was declared in April 2015.

The current prediction (as of 21 September 2015) is a high probability (≥95%) that El Niño conditions will continue through winter 2015/2016, gradually weakening through spring 2016. How strong this El Niño event will be is difficult to predict. The latest ENSO forecasts point to a strong to very strong El Niño persisting into spring 2016, with some models predicting that this event will be comparable to the exceptional 1997/98 event (http://www.elnino.noaa.gov/).

Freshwater environments

Sea surface temperatures across the Northeast Pacific Ocean are anomalously warm due to persistent high pressure off the PNW coast and weak winds and a lack of upwelling off the Pacific southwest (PSW) coast. This warm water offshore has contributed to above average terrestrial temperatures in the PNW (Bond et al. 2015). Mean air temperatures for Washington, Oregon, and Idaho were the warmest on record for the 24 month period ending in August 2015 (from a 120 year record starting in 1895). These exceptionally warm air temperatures were most pronounced during second half of 2014 (warmest July-December 2014 on record), and the first half of 2015 (warmest January-August 2015), and less extreme during the first half of 2014 (15th warmest during January-June 2014). However, June 2015 was the warmest on record for the three state area, 8°F above the long term average and 2.6°F above the previous warm year. In contrast, precipitation in the Pacific Northwest was slightly above average during 2014, ranking 31st and 32nd wettest during January-June and July-December, respectively. Since January 2015, however, precipitation has been below average and the 8 month period from January to August was the 11th driest on record (http://www.ncdc.noaa.gov/temp-and-precip/climatological-rankings).

The exceptionally warm air during the winter of 2014/2015 and below average precipitation from January-April resulted in anomalously low snow pack conditions in the Olympic and Cascade Mountains, with most areas having less than 25% of average snow pack in April 2015 (compared to the 1981-2010 record). Many areas—especially in the southern Oregon Cascades and Sierra Nevada—that typically have continuous snow coverage during the winter had no measurable snow. Consequently, by June 2015, most basins in Washington, Idaho, Oregon, California and Nevada had 0% of normal snow pack (www.wcc.nrcs.usda.gov/gis/snow.html).

This lack of snowpack and anomalously low precipitation from January to August has had large impacts on river discharge throughout the Pacific Northwest. Stream flow in June 2015 in most small and large Washington and Oregon rivers was below average (waterdata.usgs.gov/or/nwis/sw). During the June, the Columbia River near Quincy, WA (USGS Station 14246900) was flowing at roughly 70% of its normal rate (230 KCFS vs the long term average of 330 KCFS; Figure 137). These low flow rates throughout the Northwest are expected to remain below normal through fall 2015 (www.nwrfc.noaa.gov/ws/).

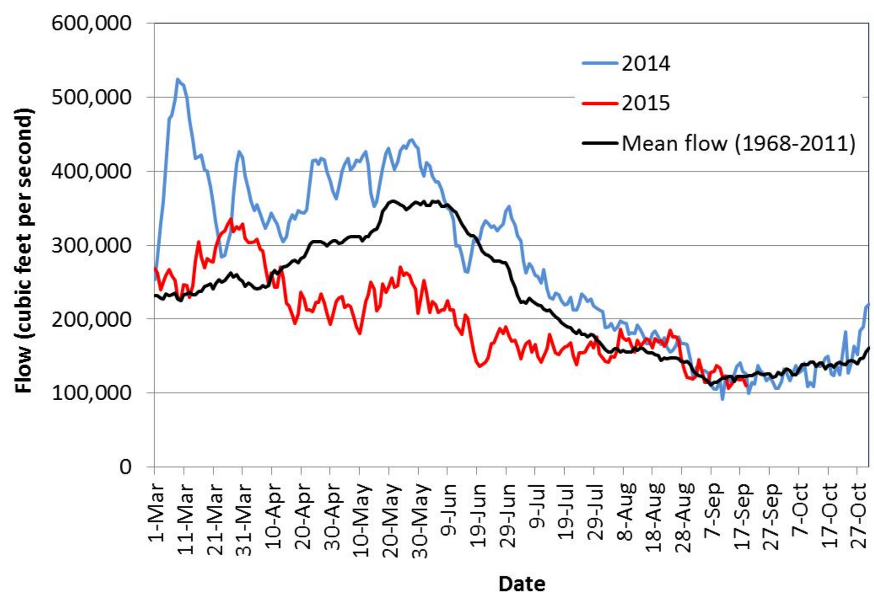


Figure 137 -- Columbia River flow measured near Quincy, WA (USGS Station 14246900) during 2014 and 2015, compared to the long term mean (1968-2011). Data from http://waterdata.usgs.gov/nwis/dv/?site_no=14246900&agency_cd=USGS&referred_module=sw

The combined effects of low flows and high air temperatures are expected to result in higher than normal stream temperatures, although the extent to which this is true is not presently known because most of water temperature time series formerly available from the USGS have been terminated. In June 2015, the Columbia River at the Dalles Dam was 3.6°C above normal (19.1°C vs. 15.3°C), the Willamette River at Portland (USGS Station 14211720) was 5.3°C above average, and the Snake River near Anatone, WA (USGS Station 13334300) was 3.9°C above average (Figure 138) These three stations also recorded temperatures of at least 20°C on 72, 99, and 72 consecutive days, respectively, during the period from May 1 to July 31, 2015 . Maximum water temperatures measured on the Willamette River reached 26.0°C in mid July 2015, which is at or near lethal limits for Pacific salmon (Fagerland *et al.* 195). There have been reports of fish kills of salmon and sturgeon in the Willamette and mainstem Columbia Rivers in late June and July. Cooler air temperatures in July and August allowed stream temperatures to decline, and the Willamette, Columbia and Snake Rivers have been near the long term mean since mid August, 2015 (Figure 138).

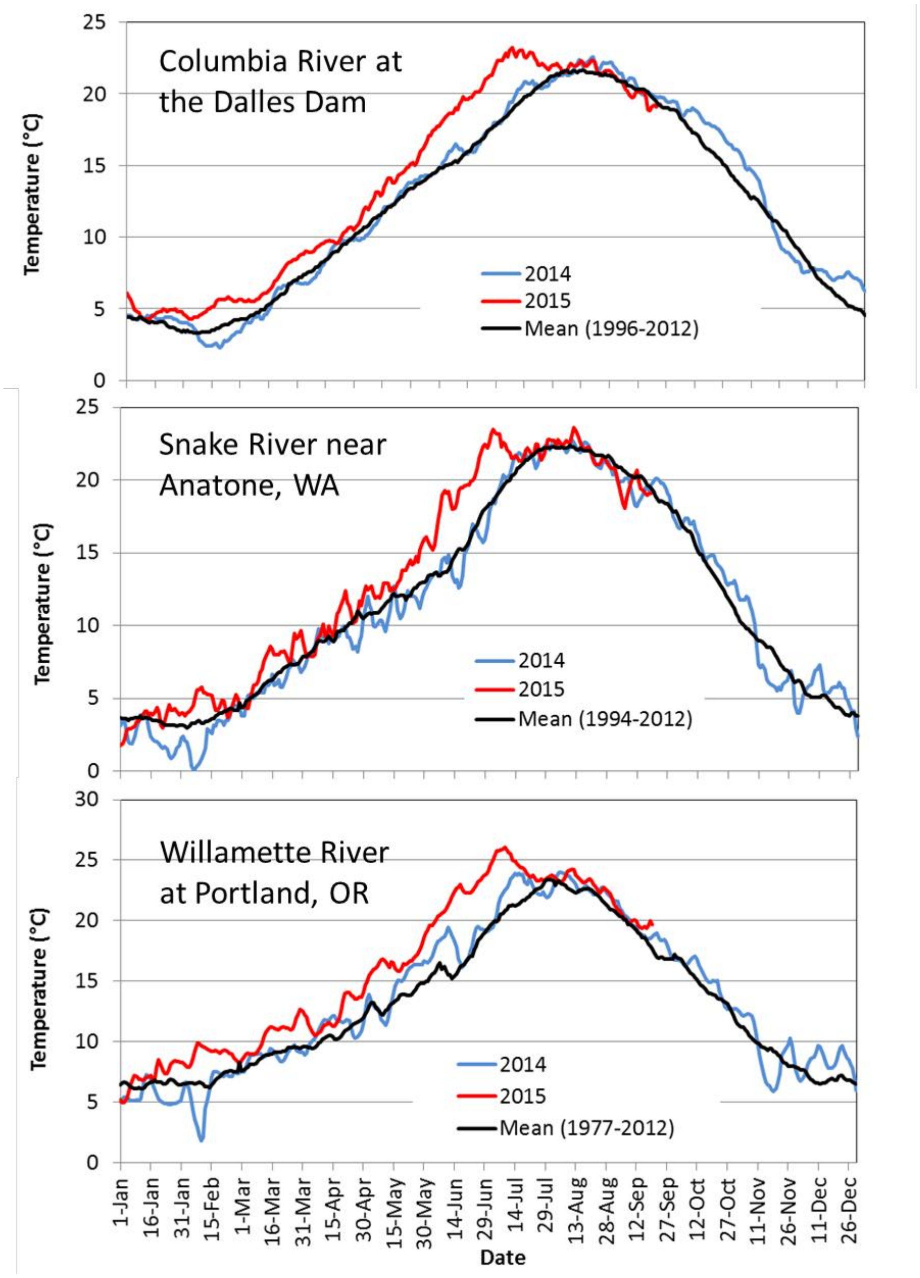


Figure 138 -- Water temperature measured in the Columbia River at the Dalles Dam (USGS Station 14105700; top), Snake River near Anatone, WA (USGS Station13334300; middle) and Willamette River in Portland, OR (USGS Station 14211720; bottom) during 2014 and 2015, compared to the long term mean. Data from USGS National Water Information System (http://waterdata.usgs.gov/).

BIOLOGICAL CONSEQUENCES OF MARINE ENVIRONMENTAL CONDITIONS

Pacific salmon are a cold water species, therefore current elevated temperatures in both freshwater and marine habitats are expected to be detrimental to their growth and survival (Crozier *et al.* 2008b; Wainwright & Weitkamp 2013). In marine environments, however, environmental conditions also have large indirect effects on salmon. This occurs because temperature changes are typically associated with different parcels of water, which come with their own planktonic ecosystem, including salmon prey and predators. In many cases, the influence of these indirect effects are larger than those due directly to physiological effects of changing temperatures (Trudel *et al.* 2002; Beauchamp *et al.* 2007).

Pacific decadal oscillation

As part of the original description of PDO, Mantua *et al*. (1997) demonstrated that changes in the PDO were related to changes in Pacific salmon populations from Alaska to California in an inverse pattern: positive PDO values were associated with high salmon catches in Alaska and low catches in the Columbia River and Washington, Oregon, and California, while negative PDO values had the opposite effect: low salmon catches to the north and high in the south.

Since the original publication, many additional studies have related the phase of the PDO to the dynamics of marine species indicating it describes conditions that are important for survival. For example, species in the Northern California Current that benefit from negative PDO (cool water off the Washington/Oregon coast) include Columbia River salmon and northern copepods, and recruitment of both northern anchovies and Dungeness crab, while species the prosper during positive PDOs include southern copepods and sardines (Peterson & Schwing 2003; Lindegren *et al.* 2013; Shanks 2013). Clearly, the PDO captures important variability in physical environments that drive the productivity of the coastal ecosystem.

El Niño events

The biological effects at higher trophic levels of large El Niño events in the California Current are less predictable and poorly understood than changes in the PDO. This occurs because large El Niño events are relatively infrequent (the last two large events occurred in 1982/83 and 1997/98), and El Niño events are tropical phenomena with variable impacts on extra-tropical systems such as the California Current (Huyer *et al.* 2002). That said, the typical El Niño year impacts in the California Current are similar to those associated with the warm phases of the PDO, and in some extreme cases much more dramatic (like those associated with the extreme 1982/83 and 1997/98 El Niño events).

Several important biological impacts were noted during the last two extreme El Niño events. During both events, there were dramatic increases in poleward flow, elevated temperatures to 200m depth, reduced upwelling and greatly reduced nutrient levels (Pearcy & Schoener 1987; Huyer *et al.* 2002). The biological impact of these conditions resulted in changes throughout the ecosystem. During the 1982/83 event, primary and secondary production was greatly reduced from southern California to Vancouver Island, especially in 1983 (Pearcy and Schoener 1987). During the 1997/98 event, the copepod assemblage along the Newport Hydrographic (NH) line became dominated by southern and offshore species starting in late summer 1997, while normally dominant boreal species had almost completely disappeared; the overall abundance of copepods were also greatly reduced. These changes to the copepod assemblage persisted for roughly a year, although some boreal species did not recover to normal levels until the summer of 1999 (Peterson *et al.* 2002).

Changes were also observed at higher trophic levels during both strong El Niño events. There were unusual sightings of a variety of subtropical (and largely predatory) fishes along the Coast of Oregon, including Dorado (*Coryphaena hippurus*), Yellowtail (*Seriola lalandi*), California barracuda (*Sphyraena argentea*), and striped marlin (*Tetrapturus audux*), many of which were range extensions (Pearcy & Schoener 1987; Pearcy 2002a). The 1997/98 event was also the first time Humboldt squid (*Dosidicus gigas*) had been observed so far north, although it has since been found as far north as

Sitka, Alaska (Wing 2006b; Litz *et al.* 2011). Like the influx of warm water fishes to the Oregon Coast, there was also influx of warm-water cetaceans to Monterey Bay during 1997 and concurrent decline of cold-water cetaceans during the El Niño (Benson *et al.* 2002). Sea birds were also negatively impacted by the 1983 El Niño (Pearcy and Schoener 1987).

The impact of these strong El Niño events on Pacific salmon is highly variable. During the 1982/83 event, the size of both coho and Chinook salmon caught in Oregon and California fisheries were the smallest on record, and survival of adults during summer of 1983 was thought to be extremely low (Pearcy and Schoenes 1987). Marine survival rates for coho salmon that entered the ocean in 1983 and 1984 were also extremely low. However, these impacts were not observed for coho salmon returning to Washington rivers, which have a much more northern migration pattern (Weitkamp & Neely 2002), nor to Chinook salmon with a northern migratory pattern (Pearcy and Schoenes 1987).

In contrast, the 1997/98 El Niño did not appear to have as adverse effects on salmon survival as the 1982/83 event. Marine survival of Oregon Production Index coho salmon was extremely low (0.7%) for fish entering marine waters in 1997, but it was equally low ($\leq$ 0.8%) for fish that entered in 1992-1996. Similarly, counts of adult salmon at Bonneville dam for spring and fall Chinook, coho, and sockeye salmon and steelhead were generally low in 1998 and 1999 (assuming these fish spent 1-2 years in marine waters), but not extremely low (i.e., not the lowest of the series).

As noted above, ocean conditions important for PNW salmon became unusually warm early in 2014, and are currently at or near record warm temperatures for much of the northeast Pacific Ocean. There is an abundance of evidence highlighting impacts on coastal marine ecosystems, including sea bird die offs, range shifts for subtropical fish and plankton, etc. Juvenile salmon entering the coastal ocean in 2015 may have experienced especially poor ocean conditions. The expected impacts of the 2015/16 El Niño include intense winter downwelling, increased northward moving currents, increased upper ocean stratification, and overall reduced productivity. These conditions will likely prime the PNW's coastal ocean for very poor productivity in spring 2016. Combining the expected El Niño effects over the next 6 to 8 months with existing warm ocean conditions will likely lead to poor or perhaps very poor early marine survival for PNW salmon going to sea in spring 2016.

*NWFSC Ocean indicators*The NWFSC has been using of a suite of physical and biological ocean indicators to describe the conditions experienced by juvenile salmon entering marine waters in the Northern California Current. These indicators—both individually and collectively--have been shown to influence juvenile salmon growth and survival (Peterson *et al.* 2014b). While these indicators were selected specifically for juvenile salmon, a recent analysis suggests they capture ecosystem variation important to the recruitment of non-salmonid species, including sablefish, rockfish and sardines (Peterson *et al.* 2014a). They have also been used to predict marine survival for Puget Sound stocks (Zimmerman 2015). These indicators include physical processes or conditions at ocean-basin scales (PDO, ONI), and regional/local scales (water temperature and salinity at surface and depth), and biological conditions (copepod composition, winter ichthyoplankton), as well as actual juvenile salmon abundances in June (Peterson *et al.* 2014b).

The copepod community on the Newport Hydrographic (NH) line has received particular emphasis in the NWFSC indicators because copepods are planktonic and drift with the ocean currents. Therefore, the type of copepods found on the NH line reflects the type of water being transported into the NCC: the presence of subtropical (southern) species off Oregon indicates transport of subtropical water from the south, while subarctic (northern) species indicates transport of coastal, subarctic waters from the north. Southern copepods typically dominate the winter copepod community and northern copepods dominate the summer community, with the "biological spring transition" index defining when it switches from one to the other. Northern copepods have much higher lipid levels than southern copepods, and therefore likely produce food webs that promote high growth and survival in salmon (juvenile salmon don't eat copepods directly) (Peterson *et al.* 2014b).

During winter/spring of 2015, 17 species of copepods were caught within 25 miles of shore on the NH line that had never been observed on the line in 20 years of biweekly sampling (B. Peterson, NWFSC, unpubl. data). These species were all subtropical or pelagic species, suggesting that subtropical offshore water was present on the continental shelf. Unusual copepods were also observed on the NH line during the 1997/98 El Niño, but the observations in 2015 far surpass the 1997/98 El Niño event. The biological transition in spring 2015 was also extremely late (late June), and the abundance of northern copepods was extremely low during summer 2015, suggesting a poor base for the food chain (B. Peterson, NWFSC, pers. comm.). Juvenile coho and Chinook salmon caught off the Washington and Oregon Coast in June 2015 had relatively low condition factor (the ratio of individual fish weight to length), consistent with poor feeding conditions (C. Morgan, Oregon State University, pers. comm.)

State of the California Current Report

Many of the ocean indicators used by NWFSC are also described in the annual State of the California Current Report (SCCR), which is focused on the entire California Current, from the US-Canada border to the US-Mexico border (CCIEAT 2015). The SCCR also describes the current state of additional indicators, including the North Pacific Gyre Oscillation (NPGO), upwelling, dissolved oxygen levels, and ocean acidification, and abundances of forage fish, salmon, groundfish, marine mammals, and seabirds. Notable changes in these indicators during 2014 were a decrease in the NPGO index and weaker than normal downwelling during winter 2014 and a late physical spring transition (when the slope of cumulative upwelling becomes positive) at 45°N. Both the decline in the NPGO and the late timing of the spring transition are associated with reduced productivity.

State of Pacific Canadian marine ecosystems report

Many of the unusual conditions in the California current described above were also present in Canadian waters off the west coast of British Columbia (Chandler *et al*. 2015). This includes reduced nutrient levels in offshore waters, rapid rises in SSTs as the warm water mass moved onshore, and unusually high abundances of southern copepods during summer 2014. At higher trophic levels, catches of smooth pink shrimp *(Pandalus jordani)* off the west coast of Vancouver Island was nearly twice as high as the previous maximum, and estimated herring biomass was higher in 2014 than 2013, although there was a marked absence of Pacific sardine in Canadian waters for a second year in a row (Chandler *et al*. 2015). The warm water also the likely cause for the extremely high diversion rate of sockeye salmon bound for the Fraser River, which returned around the north end of Vancouver Island via Johnstone Strait (vs. around the south end via Strait of Juan de Fuca) at the highest rate ever recorded.

In contrast to unusual conditions observed off the West Coast of British Columbia, conditions within the Strait of Georgia were not particularly unusual. For example, salinity and temperature of water within the Strait of Georgia was fairly typical to other years during most of 2014, the timing of the phytoplankton bloom was also normal, and juvenile salmon survival was comparable to other recent years. One notable difference was that waters of the Strait of Juan de Fuca were warmer than normal in September and October, reflecting the influence of warm coastal waters off Vancouver Island.

EXPECTATIONS FOR PACIFIC SALMON

It is well established that ocean conditions during the first weeks or months of marine life have a large influence on overall marine survival for salmon (Pearcy 1992; Pearcy & McKinnell 2007). Accordingly, where salmon are during the first summer of ocean residence and the conditions they experience has a large impact on their survival. In general, Pacific salmon from the Pacific Northwest can be grouped by their ocean migration patterns: sockeye, chum, and spring Chinook salmon move rapidly north along the continental shelf to Alaskan waters and reside in the Gulf of Alaska for most of their ocean residence, fall Chinook remain in local waters (although their location during winter months is largely unknown), while coho salmon display a hybrid pattern: some move rapidly northern while other remain in local waters during the first summer before moving north to Alaska (Myers *et al.* 1996; Burke *et al.* 2013; Fisher *et al.* 2014). Steelhead have a unique marine migration pattern and move directly offshore and apparently west across the North Pacific Ocean (Myers *et al.* 1996; Hayes *et al.* 2012; Daly *et al.* 2014). There can also be large variation within these general groups, such as the large change in marine distributions for coho salmon from the Washington Coast vs Columbia River and south (Weitkamp and Neely 2002), or the change in Chinook marine distributions for populations north and south of Cape Blanco, Oregon (Nicholas & Hankin 1988; Weitkamp 2010).

These differences in migration patterns paired with heterogeneous ocean conditions have resulted in species and population differences in survival in the past, and will no doubt be important in the future. For example, the 1982/83 El Niño had much more severe impacts on Chinook and coho salmon populations with "southern" distributions, than those with more northern distributions, described previously. Similarly, fall Chinook salmon from the Columbia River that entered the ocean in 2011 returned in record high numbers, while spring Chinook salmon from the same system entering in the same year had low returns (and below predictions). This difference is thought to be due to differences in ocean conditions encountered by the two runs: spring Chinook salmon migrate rapidly to Alaska, where ocean conditions were extremely unproductive in 2011, while fall Chinook remain off the Washington/Oregon coast, where conditions were quite productive.

A reverse situation to 2011 appears to have occurred in spring 2014. Many indicators of ocean conditions in the local waters of the NCC were largely unfavorable for juvenile salmon. In contrast, growth rates for juvenile salmon caught in northern British Columbia and the Gulf of Alaska during summer 2014 were the highest on record (M. Trudel, DFO, pers. comm.; J. Moss, AFSC, pers. comm.). However, market squid (*Doryteuthis opalescens* [formerly *Loligo opalescens*]) have been extremely abundant in surface trawls conducted between San Francisco Bay and Cape Flattery during surveys conducted by the SWFSC and NWFSC in 2014 and 2015. Squid provide a high quality prey for juvenile salmon that are large enough to consume them, due to relatively high energy levels. The exceptionally large return of coho salmon to the Oregon coast and Columbia River in fall 2014 suggest that ocean conditions in the California Current must have been favorable, at least for returning adults. How favorable these ocean conditions were for juvenile salmon that entered the ocean in 2014 won't be known until the adults return in 2015 (coho salmon) or 2016 (Chinook salmon). Initial returns (October 2015) indicate that coho salmon returns to the Columbia River and Oregon coast are well below expectations. Ocean conditions for salmon entering marine waters in 2015 are expected to have been much worse than 2014, but the effects of these potentially disastrous conditions won't be known until 2016 (coho salmon) or 2017 (Chinook salmon), when the adults begin to return.

CONCLUSION

It is clear that current anomalously warm marine and freshwater conditions have been and will continue to be unfavorable for Pacific Northwest salmon. How extreme the effects will be is difficult to predict, although decreased salmon productivity and abundance observed during prior warm periods provide a useful guide. How long the current conditions will last is also unknown, but NOAA's coupled forecast system model (CFS version 2) suggests that the warm conditions associated with the strengthening El Niño will persist at least through spring 2016. The model currently predicts temperature anomalies during the March-April-May 2016 period will exceed 2°C at the equator and 0.5-2°C in the NE Pacific. Unfortunately, longer forecasts are not available.

On a positive note, after previous strong El Niño events (e.g., 1982/83 and 1997/98), there was a rapid transition from warm to cold conditions along the West Coast, which resulted in greatly improved marine survival for Pacific salmon for several years following the El Niño. Whether a similar rapid transition to cold conditions will occur with this El Niño is not known or presently forecast, but is within the realm of possibility.

Pacific salmon are a cold water species: they flourish in cold streams and cold and productive marine ecosystems, such as those present in the early 2010s, resulting in record returns for many ESUs. The exceptionally warm marine waters in 2014 and 2015 (and associated warm-water food webs) and warm stream temperatures observed during 2015 were unfavorable for high marine or freshwater survival. West Coast salmon entering the ocean in 2016 will likely encounter subtropical foodwebs that do not promote high survival. The full impact of these unusual environmental conditions will not be known until adults return beginning this fall and continuing for the next few years.

CLIMATE CHANGE

Climatic conditions affect salmonid abundance, productivity, spatial structure, and diversity through direct and indirect impacts at all life stages (e.g., ISAB 2007; Lindley *et al.* 2007; Crozier *et al.* 2008a; Moyle *et al.* 2013; Wainwright & Weitkamp 2013). Salmon have adapted to a wide variety of climatic conditions in the past, and thus inherently could likely survive substantial climate change at the species level in the absence of other anthropogenic stressors.

Currently, the adaptive ability of these threatened and endangered species is depressed due to reductions in population size, habitat quantity and diversity, and loss of behavioral and genetic variation. Without these natural sources of resilience, systematic changes in local and regional climatic conditions due to anthropogenic global climate change will likely reduce long-term viability and sustainability of populations in many of these ESUs. Adapting to climate change may eventually involve changes in multiple life history traits and/or local distribution, and some populations or life-history variants might die out. Importantly, the character and magnitude of these effects will vary within and among ESUs.

The Intergovernmental Panel on Climate Change (IPCC) and U.S. Global Change Research Program recently published updated assessments of anthropogenic influence on climate, as well as projections of climate change over the next century (IPCC 2013; Melillo *et al.* 2014). Reports from both groups document ever-increasing evidence that recent warming bears the signature of rising concentrations of greenhouse gas emissions.

The U.S. Global Change Research Program report contains regional-focus chapters for the northwest (Snover *et al.* 2013; Mote *et al.* 2014) and southwest U.S. (Garfin *et al.* 2014). These regional reports synthesize information from an extensive literature review, including a broad array of analyses of regional observations and climate change projections. These synthesis reports were the primary source for this West Coast summary. References to the primary literature can be found in those reports. Updates to this summary can be found in annual literature reviews conducted by NOAA-Fisheries (available at http://www.nwfsc.noaa.gov/trt/lcm/freshwater_habitat.cfm).

HISTORICAL CLIMATE TRENDS

Observed historical trends in climate reflect the early influence of greenhouse gas emissions, and often indicate the general direction of future climate change. These observations also reflect natural variability in climate at multiple time scales. Natural variability alternately intensifies and relaxes (or partially reverses) the long-term trends. Attribution of historical trends to anthropogenic factors is most certain at the global scale over time scales of centuries to millennia because at these scales we can better account for natural variability.

Historical records show pronounced warming in both sea-surface and land-based air temperatures. There is moderate certainty that the 30-year average temperature in the Northern Hemisphere is now higher than it has been over the past 1,400 years. In addition, there is high certainty that ocean acidity has increased with a drop in pH of 0.1. Furthermore, glaciers and sea-ice have receded, while sea level has risen (global mean rose 0.19 m over the last century). In recent decades, the frequency of extreme high temperature or heavy precipitation events has increased in many regions. An anthropogenic influence on this shift in frequency is "very likely" (IPCC 2014).

Regional and local trends include the following observations:

- In both the Northwest and Southwest:
    - air temperatures have increased since the late 1800s
    - springtime snow-water equivalent has decreased (since 1950)
    - snowmelt occurs earlier in the year

- In the Southwest, drought over the past 4 years is unprecedented in the historical record and may be the worst in over 1,000 years. This drought has been attributed to a combination of anthropogenic influence on temperature and natural variability in precipitation (Williams *et al.* 2015). Trends in precipitation vary spatially up or down, with no statistically significant trends in precipitation averages or extremes in the Northwest.

- In both the Northwest and Southwest, widespread tree mortality has been observed, wildfires have increased in both frequency and area burned, and insect outbreaks have increased (Garfin *et al.* 2014; Mote *et al.* 2014).

- Historical trends in the California Current are heavily influenced by patterns in wind-driven ocean circulation, which correlates with large-scale climate drivers such as the North Pacific Gyre Oscillation (Peterson *et al.* 2013) and Pacific Decadal Oscillation (Jacox *et al.* 2014). Spatially variable trends in upwelling intensity (Jacox *et al.* 2014) and hypoxia (Peterson *et al.* 2013), and longer trends in atmospheric forcing and sea surface temperature (Johnstone & Mantua 2014) probably reflect natural climate variability to a much greater extent than anthropogenic forcing.

- The pH of California Current has decreased by about 0.1 and by 0.5 in aragonite saturation state since pre-industrial times (Hauri *et al.* 2009). Furthermore, infrastructure in coastal areas is increasingly damaged by erosion and flooding (Garfin *et al.* 2014; Mote *et al.* 2014; Sweet *et al.* 2014).

PROJECTED CLIMATE CHANGES

Trends in warming and ocean acidification are highly likely to continue during the next century (IPCC 2013). Scenarios considered in the IPCC fifth assessment report range from the severely curtailed greenhouse gas emissions of representative concentration pathway (RCP) 2.6 to business as usual in RCP 8.5.

Based on means across global climate models spanning the full breadth of these emissions scenarios, IPCC projected the following ranges across the Northern Hemisphere by 2081-2100:

- Spring snow cover declines of 7-25%
- Glacier recessions of 15-85%
- Sea surface temperature increases of 1.1-3.6°C
- Global sea level increases of 11-38 inches
- Global ocean pH decreases of 38 to 109%, which correspond to a drop in pH of 0.14-0.32.

Regional projections add spatial variability and specificity to these themes. In winter across the west, the highest elevations (e.g. in the Rocky Mountains) will shift from consistent longer (>5 months) snow-dominated winters to a shorter period (3-4 months) of reliable snowfall (Klos *et al.* 2014); lower, more coastal or more southerly watersheds will shift from consistent snowfall over winter to alternating periods of snow and rain ("transitional"); lower elevations or warmer watersheds will lose snowfall completely, and rain-dominated watersheds will experience more intense precipitation events and possible shifts in the timing of the most intense rainfall (e.g., Salathe *et al.* 2014).

By the 2080s, Tohver et al. (2014) anticipate a complete loss of snow-dominated basins in the Cascades and U.S. portion of the Rockies, with only a few "mixed" basins of rain- and snow-fed runoff remaining at the highest elevations. Flooding is projected to increase in basins that experience a mix of snow and rain in winter (Mote *et al.* 2014; Salathe *et al.* 2014; Tohver *et al.* 2014). Erosion and flooding in coastal areas are projected to increase with rising sea levels (Garfin *et al.* 2014; Mote *et al.* 2014; Sweet *et al.* 2014).

Among seasons, the greatest temperature shifts are expected in summer. Warmer summer air temperatures will increase both evaporation and direct radiative heating. When combined with reduced winter water storage, warmer summer air temperatures will lead to lower minimum flows in many watersheds. Higher summer air temperatures will depress minimum flows and raise maximum stream temperatures even if annual precipitation levels do not change (e.g., Sawaske & Freyberg 2014). Summer precipitation also influences summer flows, but projections for precipitation are less certain than for temperature. Coastal weather can differ from region-wide projections due to changes in fog, on-shore winds, or precipitation (Johnstone & Dawson 2010; Potter 2014).

Widespread ecosystem shifts are very likely, and may be abrupt due to disturbances from increasing wildfires, insect outbreaks, droughts, and tree diseases (Garfin *et al.* 2014; Mote *et al.* 2014). Climate projections often favor invasive fish species over native species, with declines exacerbated by the greater vulnerability of native species to existing anthropogenic stressors (Lawrence *et al.* 2012; Lawrence *et al.* 2014; Quiñones & Moyle 2014).

In response to projected changes in both climate and land use practices, estuary dynamics are expected to change as well, with depth and salinity altered by changing sea level, upwelling regimes, and freshwater input (Yang *et al.* 2015). Intense upwelling events can move hypoxic and acidic water into estuaries, especially when freshwater input is reduced (e.g., Columbia River estuary, Roegner *et al.* 2011). Sea level projections differ at local vs. global scales due to local wind and temperature trends and land movement. Specifically, the National Research Council (2012) predicted a lower rise in sea level off the coasts of Washington and Oregon (62 cm) than off the coast of California (92 cm) by 2100.

Higher sea surface temperatures and increased ocean acidity are predicted for marine environments in general (IPCC 2013). However, regional marine impacts will vary, especially in relation to productivity. The California Current is strongly influenced by seasonal upwelling of cool, deep, water that is high in nutrients and low in dissolved oxygen and pH.

Ecological effects of climate change in the California Current are very sensitive to impacts on upwelling intensity, timing, and duration. Projections of how climate change will affect upwelling are highly variable across models, with predicted trends ranging from negative to positive (Bakun 1990; Mote & Mantua 2002; Snyder *et al.* 2003; Diffenbaugh *et al.* 2008; Bakun *et al.* 2010). An analysis of 21 global climate models found that most predicted a slight decrease in upwelling in the California Current, although there is a latitudinal cline in the strength of this effect, with less impact toward the north (Rykaczewski *et al.* 2015).

Much of the near-shore California Current is expected to be corrosive (undersaturated in aragonite) in the top 60 m during all summer months within the next 30 years, and year-round within 60 years (Gruber *et al.* 2012). Thermal stratification and hypoxia are expected to increase (Doney *et al.* 2014).

IMPACTS ON SALMON

Studies examining the effects of long-term climate change to salmon populations have identified a number of common mechanisms by which climate variation is likely to influence salmon sustainability. These include direct effects of temperature such as mortality from heat stress, changes in growth and development rates, and disease resistance. Changes in the flow regime (especially flooding and low flow events) also affect survival and behavior. Expected behavioral responses include shifts in seasonal timing of important life history events, such as the adult migration, spawn timing, fry emergence timing, and the juvenile migration.

Indirect effects on salmon mortality, growth rates and movement behavior are also expected to follow from changes in the freshwater habitat structure and the invertebrate and vertebrate community, which governs food supply and predation risk (Petersen & Kitchell 2001; ISAB 2007; Crozier *et al.* 2008a). Both direct and indirect effects of climate change will vary among Pacific salmon ESUs and among populations in the same ESU. Adaptive change in any salmonid population will depend on the local consequences of climate change as well as ESU-specific characteristics and existing local habitat characteristics.

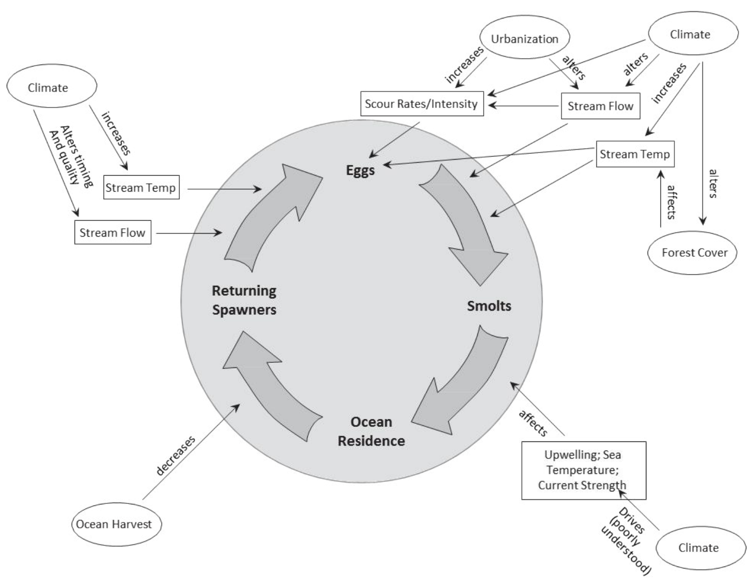
Because climate has such profound effects on survival and fecundity, salmon physiology and behavior are exquisitely adapted to local environmental conditions. These adaptations vary systematically among populations and are exhibited in traits such as age and timing of juvenile and adult migrations, with potential differences in physiology and migration routes (Quinn 2005b). These traits often have a significant plastic (non-genetic) component, which allows them to respond quickly to environmental change. Yet these traits also differ genetically among populations (Carlson & Seamons 2008).

Directional climate change could therefore drive many salmonid populations into a maladaptive state. Such an outcome would likely cause reductions in abundance, productivity, population spatial structure and population diversity. In some cases, this can lead to extirpation if a population cannot adapt quickly enough. In other cases an adaptive solution may not exist because of conflicting pressures within or between life stages.

Climate impacts in one life stage generally affect body size or timing in the next life stage. For this reason, the cumulative life-cycle effects of climate change must be considered to fully appreciate the scope of risk to a given population. Even without interactions among life stages, the sum of impacts in many stages will have cumulative effects on population dynamics.

Climate effects tend to be negative across multiple life stages (Healey 2011; Wade *et al.* 2013; Wainwright & Weitkamp 2013). However, there may be mitigating responses in some ESUs or life stages. Individualistic impacts within and among ESUs will depend on factors such as existing physical and biological heterogeneity, proximity to the limits of physiological tolerance under present climate conditions, and the extent of local climate change.

In many cases, directional climate change exacerbates existing anthropogenic threats. Examples include streams or rivers where stream temperatures are already elevated due to land-use modifications (Battin *et al.* 2007) or where flow is reduced due to water diversions (Walters *et al.* 2013a). In the Columbia River, dams have altered the hydrological regime by causing an earlier and smaller freshet, which is the same type of effect expected from climate change (Naik & Jay 2011b, a). Any of these stressors in combination with one another or with climate impacts will present pressures of much greater concern than they would individually, but they also offer potential solutions.



Conceptual model from (Figure from McClure et al. 2013) shows potential links between anthropogenic climate perturbations and habitat conditions affecting survival of Chinook salmon Oncorhynchus tshawytscha *during each life-stage. Effects of these drivers can be positive or negative, depending on the magnitude and direction of change.*

Changes in winter precipitation will likely affect incubation and/or rearing stages of most populations. Changes in the intensity of cool-season precipitation could influence migration cues for fall and spring adult migrants, such as coho and steelhead. Egg survival rates may suffer from more intense flooding that scours or buries redds.

Changes in hydrological regime, such as a shift from mostly snow to more rain, could drive changes in life history, potentially threatening diversity within an ESU. It is possible that even characteristic life-history traits used to help define the ESU will be threatened. For example, the juvenile freshwater rearing period is very sensitive to temperature, with the yearling life-history strategy used only by populations in cooler watersheds (Beechie *et al.* 2006). Frequency of the yearling life-history type will likely decline as movement downstream into estuaries or near-shore habitat is initiated at younger ages. Implications of this behavioral shift for juvenile survival, ocean migration behavior, and age at maturity are uncertain.

Changes in summer temperature and flow will affect both juvenile and adult stages in some populations, especially those with yearling life histories and summer migration patterns. Juvenile rearing and migration survival is often correlated with these factors (Quinn 2005b; Crozier & Zabel 2006; Crozier *et al.* 2010).

Adults that migrate or hold during peak summer temperatures can experience very high mortality in unusually warm years. For example, in 2015 only 4% of adult Redfish Lake sockeye survived the migration from Bonneville to Lower Granite Dam after confronting temperatures over 22°C in the lower Columbia River. After prolonged exposure to temperatures over 20°C, salmon are especially likely to succumb to diseases that they might otherwise have survived (Materna 2001; Miller *et al.* 2014). They are also more vulnerable to any sort of stress, such as catch-and-release fisheries (Boyd *et al.* 2010).

Changing hydrology and temperature will also affect the timing of smolt migrations and spawning (Crozier & Hutchings 2014; Hayes *et al.* 2014; Otero *et al.* 2014). If smolts migrate at a smaller size because they leave freshwater habitat earlier, they might have lower survival due to size-selective predation (Thompson & Beauchamp 2014). Marine arrival timing is extremely important for smolt-to-adult survival (Scheuerell *et al.* 2009), and has been historically synchronized with the timing and predictability of favorable ocean conditions (Spence & Hall 2010). Given the uncertain effects of climate change on upwelling timing and intensity, impacts on juvenile survival from shifts in migration timing are also difficult to predict.

In some populations, behavior during the early ocean stage is consistent among years, suggesting a genetic rather than a plastic response to environmental conditions (Burke *et al.* 2014, Hassrick et al in press). These populations might change their behavior over time if the fitness landscape changes, but responses will likely be relatively slow and could be dominated by decadal ocean dynamics or productivity outside the California Current (e.g., the Gulf of Alaska for northern migrants).

Other populations show more variable behavior after ocean entry (Weitkamp 2010; Fisher *et al.* 2014), and some show heightened sensitivity to interannual climate variation, such as the El Niño Southern Oscillation. Such variability might increase ESU-level resilience to climate change, assuming some habitats remain highly productive.

Marine migration patterns could also be affected by climate-induced contraction of thermally suitable habitat. Abdul-Aziz *et al*. (2011) modeled changes in summer thermal ranges in the open ocean for Pacific salmon under multiple IPCC warming scenarios. For chum, pink, coho, sockeye and steelhead, they predicted contractions in suitable marine habitat of 30-50% by the 2080s, with an even larger contraction (86-88%) for Chinook salmon under the medium and high emissions scenarios (A1B and A2).

Northward range shifts are a climate response expected in many marine species, including salmon (Cheung *et al.* 2015). However, salmon populations are strongly differentiated in the northward extent of their ocean migration, and hence will likely respond individualistically to widespread changes in sea surface temperature.

In most Pacific salmon species, size at maturation has declined over the past several decades. This trend has been attributed in part to rising sea surface temperatures (Bigler *et al.* 1996; Pyper & Peterman 1999; Morita *et al.* 2005). Mechanisms involved in such responses are likely complex, but appear to reflect a combination of density-dependent processes, including increased competition due to higher salmon abundance in recent years and temperature (Pyper & Peterman 1999). Temperature-related size effects could involve increased metabolic costs at higher temperatures, and/or shifts in spatial distribution in response to ocean conditions. Younger spawners affect population growth rates by exhibiting lower fecundity and reducing the population stability that stems from having multiple age classes reproduce.

Numerous researchers have reported that salmon marine survival is highly variable over time and often correlated with large-scale climate indices (Mueter *et al.* 2002; Mueter *et al.* 2005; Petrosky & Schaller 2010; Litzow *et al.* 2014; Stachura *et al.* 2014; Sydeman *et al.* 2014). For example, Pacific salmon from Washington and Oregon exhibited extremely low marine survival and dramatic population declines during a "warm phase" of the Pacific Decadal Oscillation in the 1980s and 1990s (Levin 2003; Zabel *et al.* 2006). These declines were attributed to low ocean productivity in the warm ocean of that period.

Many fish communities, including key salmon prey and predators, experience changes in abundance and distribution during warm ocean periods (Pearcy 2002b; Wing 2006a; Cheung *et al.* 2009). However, food chain dynamics in the open ocean are flexible and difficult to predict into the future.

The full implications of ocean acidification on salmon are not known at this time. Olfaction and predator-avoidance behavior are negatively affected in some fish species, including pink salmon (Leduc *et al.* 2013; Ou *et al.* 2015). Pink salmon also showed reductions in growth and metabolic capacity under elevated CO_2 conditions (Ou *et al.* 2015). Some high-quality salmon prey (e.g., krill) might be negatively affected by ocean acidification, but there are several possible pathways by which higher trophic levels might compensate for changes at a lower trophic level. From their analysis of multi-trophic responses to ocean acidification, Busch *et al*. (2013) concluded that impacts to salmon could conceivably be positive. However, they emphasized that a better understanding of both direct and indirect feedback loops is necessary before drawing definitive conclusions.

To what extent a future warmer ocean will mimic historic conditions of warm-ocean, low-survival periods is not known. Current indications are that a warmer Pacific Ocean is generally less productive at mid latitudes, and hence likely to be less favorable for salmon.

Analysis of ESU-specific vulnerabilities to climate change by life stage will be available in the near future, upon completion of the *West Coast Salmon Climate Vulnerability Assessment*. Climate effects on one Pacific salmon ESU, the Oregon coastal coho, were recently assessed by Wainwright and Weitkamp (2013). Below we reproduce the extensive list of effects they reported for this ESU; many of these effects will likely be shared by other ESUs.

In summary, both freshwater and marine productivity tend to be lower in warmer years for most populations considered in this status review. These trends suggest that many populations might decline as mean temperature rises. However, the historically high abundance of many southern populations is reason for optimism and warrants considerable effort to restore the natural climate resilience of these species.

Projected climate changes affecting Oregon coho, as reported by Wainwright and Weitkamp (2013). Abbreviations: LWD, large woody debris, – – strongly negative, – negative, ο neutral, + positive, + + strongly positive.

Physical/chemical pattern	Certainty of change	Process affecting Oregon coast coho salmon	Range of effects					Certainty of effect
			– –	–	ο	+	+ +	
Terrestrial habitat								
Warmer, drier summers	Moderate	Increased fires, increased tree stress & disease affect LWD, sediment supplies, riparian zone structure	×	×	×			Low
Reduced snow pack, warmer winters	High	Increased growth of higher elevation forests affect LWD, sediment, riparian zone structure			×	×		Low
Freshwater habitat								
Reduced summer flow*	High	Less accessible summer rearing habitat		×				Moderate
Earlier peak flow*	High	Potential migration timing mismatch	×	×	×			Moderate
Increased floods*	Moderate	Redd disruption, juvenile displacement, sediment dynamics	×	×	×	×		Moderate
Higher summer stream temp	Moderate	Thermal stress, restricted habitat availability, increased susceptibility to disease, parasites, & predators	×	×				Moderate
Higher winter stream temp	Low	Increased fry growth, shorter incubation				×	×	Low
Estuarine habitat								
Higher sea level	High	Reduced availability of wetland habitats	×	×				Moderate
Higher water temperature	Moderate	Thermal stress, increased susceptibility to disease, parasites &predators	×	×				Moderate
Combined effects		Changing ecosystem composition and structure	×	×	×	×	×	Low
Marine habitat								
Higher ocean temperature	High	Thermal stress, shifts in migration, range shifts, susceptibility to disease, parasites, & predators	×	×				Moderate
Intensified upwelling	Moderate	Increased nutrients (food supply), coastal cooling, ecosystem shifts; increased offshore transport			×	×	×	Low
Delayed spring transition	Low	Food timing mismatch with juvenile migrants, ecosystem shifts		×	×			Low
Intensified stratification	Moderate	Reduced food supply, change in habitat structure	×	×				Low
Increased acidity	High	Disruption of food supply, ecosystem shifts	×	×				Moderate
Combined effects		Changing ecosystem composition & structure; food supply & predation		×	×	×	×	Low

* Strong negative effects are for the snow-fed portions of the Umpqua Basin only.

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