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Biological Viability Assessment Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act:

Pacific Northwest

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U.S. DEPARTMENT OF COMMERCE

National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northwest Fisheries Science Center

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Biological Viability Assessment Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Pacific Northwest

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Erratum

An earlier version of this report contained an error in the data for Elwha River steelhead (Puget Sound DPS) that resulted in incorrectly classifying hatchery-origin fish as natural-origin fish. This error has been corrected, resulting in changes to the Elwha River portions of Figures 95 and 96 and Tables 54 and 56, and some associated textual changes that reference these items. In addition, a productivity trend graph for Carbon River steelhead (Puget Sound DPS) was inadvertently omitted from the original report, and has now been added to Figure 96.

We thank John Mahan for bringing the Elwha River data error to our attention.

Author Contributions

Katie Barnas: Data management, habitat.
Lisa Crozier: Climate, environment.
Monica Diaz:[*] Data management.
Michael Ford: Editor and project manager.
Elizabeth Holmes: Status and trends analyses and graphs.
Damon Holzer: Maps.
Chris E. Jordan: Interior Columbia River basin.
Martin Liermann: Ozette Lake.
James Myers: Lower Columbia River, Willamette River, and Puget Sound.
Mindy Rowse: Puget Sound.
Laurie Weitkamp: Oregon coast.
Margaret Williams:[*] Interior Columbia River basin.

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Abstract

In the Pacific Northwest, there are currently 18 evolutionarily significant units (ESUs) or distinct population segments (DPSes) of Pacific salmon and steelhead listed as threatened or endangered under the Endangered Species Act of 1973 (ESA). 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. NMFS is conducting such a review in 2020–21 (USOFR 2019). The NMFS West Coast Region (WCR) is responsible for the five-year review process for Pacific salmon and steelhead (*Oncorhynchus* spp.) and for decision-making regarding any proposed changes in listing status. This report provides updated information and analyses on the biological viability of the listed species, focusing primarily on trends and status in abundance, productivity, spatial structure, and diversity. The information in the report will be incorporated into WCR's review, and WCR will make final determinations about whether changes in listing status are or are not warranted, taking into account not only biological information but also ongoing or planned protective efforts and recovery actions.

Several ESUs/DPSes were evaluated to have declining trends in overall status since the last review. Upper Willamette River steelhead (*O. mykiss*) and Chinook salmon (*O. tshawytscha*) were evaluated to have declining viability due to chronically declining abundance and persistent concerns regarding spatial structure and diversity. Snake River sockeye (*O. nerka*) were evaluated to have a declining viability trend, the result of abundance declines combined with very high vulnerability to climate change. In contrast, a few ESUs/DPSes were evaluated to be improving in viability. Lower Columbia River Chinook salmon were evaluated to have an increasing viability trend, the result of natural spawner increases in multiple populations, combined with dramatic improvements in the fraction natural-origin spawners in several populations. Columbia River chum salmon (*O. keta*) also showed marked improvement in abundance for several extant populations, although many historical populations remain extirpated or at extremely low abundance. Puget Sound steelhead also showed some evidence of improving viability, with the reversal of some previous strongly negative trends.

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Introduction

In the Pacific Northwest (PNW), there are currently 18 evolutionarily significant units (ESUs) or distinct population segments (DPSes)[1] of Pacific salmon and steelhead (*Oncorhynchus* spp.) listed as threatened or endangered under the Endangered Species Act of 1973 (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 2016 (WCR 2016). NMFS is again conducting such a review in 2020–21 (USOFR 2019).

The NMFS West Coast Region (WCR) is responsible for the five-year review process for Pacific salmon and steelhead, and for decision-making regarding any proposed changes in listing status. This report provides updated information and analyses on the biological viability of the listed species, focusing primarily on trends and status in abundance, productivity, spatial structure, and diversity. In some cases, the report considers new information available on ESU or population boundaries. Where possible, this review also summarizes current information with respect to recovery goals identified in recovery plans or technical recovery team (TRT) viability documents.

In three prior viability reports that supported the current listings (Good et al. 2005, Hard et al. 2007, NWFSC 2015), each ESU was categorized 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 a fourth report (Oregon Coast coho salmon [*O. kisutch*]; Stout et al. 2012), the three categories were instead referred to as "high," "moderate," and "low" risk, and included narrative definitions for the "high" and "moderate" risk categories (see Stout et al. 2012, p. 114). In this report, we use the "high," "moderate," and "low" risk categories of Stout et al. (2012). In addition, we also note whether the viability of each ESU appears to be unchanged, improving, or declining, even if the magnitude of the change is not sufficient to warrant a move among the three risk categories (Table 1). The information in the report will be incorporated into WCR's review, and WCR will make final determinations about whether changes in listing status are or are not warranted, taking into account not only biological information but also ongoing or planned protective efforts and recovery actions.

Several ESUs/DPSes were evaluated to have a declining trend in overall status since the last review (Table 1). Upper Willamette River steelhead (*O. mykiss*) and Chinook salmon (*O. tshawytscha*) were judged to have declining viability due to chronically declining abundance and persistent concerns regarding spatial structure and diversity. Snake River sockeye (*O. nerka*) were judged to have a declining viability trend, the result of abundance declines combined with very high vulnerability to climate change. In contrast, a few ESUs/DPSes were evaluated to be improving in viability. Lower Columbia River Chinook salmon were

[1] For Pacific salmon, NMFS uses its 1991 ESU policy (NMFS 1991), which states that a population or group of populations will be considered a DPS if it is an ESU. The species *O. mykiss* is under the joint jurisdiction of NMFS and the U.S. Fish and Wildlife Service (USFWS), so in making its listing determinations, NMFS used the 1996 joint DPS policy (USFWS and NMFS 1996) for this species.

Table 1. Summary of current ESA listing status, recent trends, and risk of extinction of Pacific salmon ESUs/DPSes, by species. Click *Chapter* number to go directly to the related section of this report.

Salmon species	ESU/DPS	ESA listing status	Recent viability trend[a]	2020 extinction risk category[b]	Chapter
Chinook	Upper Columbia River spring-run	Endangered	unchanged	high	1
	Snake River spring/summer-run	Threatened	unchanged	moderate-to-high	3
	Snake River fall-run	Threatened	unchanged	moderate-to-low	4
	Upper Willamette River	Threatened	declining	moderate	12
	Lower Columbia River	Threatened	increasing	moderate	8
	Puget Sound	Threatened	unchanged	moderate	14
Coho	Lower Columbia River	Threatened	unchanged	moderate	9
	Oregon Coast	Threatened	unchanged	moderate-to-low	18
Sockeye	Snake River	Endangered	declining	high	5
	Ozette Lake	Threatened	mixed	moderate-to-high	17
Chum	Hood Canal summer-run	Threatened	unchanged	moderate-to-low	16
	Columbia River	Threatened	unchanged	moderate	11
Steelhead	Upper Columbia River	Threatened	unchanged	high	2
	Snake River Basin	Threatened	unchanged	moderate	6
	Middle Columbia River	Threatened	unchanged	moderate	7
	Upper Willamette River	Threatened	declining	moderate-to-high	13
	Lower Columbia River	Threatened	unchanged	moderate	10
	Puget Sound	Threatened	increasing	moderate	15

[a] *Recent viability trend* summarizes the short-term trend in viability for each ESU/DPS since the prior viability report (NWFSC 2015), based on the expert opinion of the chapter author(s) considering all four viable salmonid population (VSP) criteria (abundance, productivity, spatial structure, and diversity; McElhany et al. 2000).

[b] An ESU or DPS with a *high* risk of extinction is at or near a level of abundance, productivity, spatial structure, or diversity that places its persistence in question, such that the risk of extinction is more than 5% in 100 years. The demographics of an ESU/DPS at a high level of risk may be highly uncertain and strongly influenced by stochastic or depensatory processes. Similarly, an ESU/DPS may be at high risk of extinction if it faces clear and present threats (e.g., confinement to a small geographic area, imminent destruction, modification or curtailment of its habitat, or disease epidemic) that are likely to create such imminent demographic risk. An ESU or DPS ESU at *moderate* risk of extinction exhibits a trajectory indicating that it is more likely than not to be at a high level of extinction risk within 30–80 years. An ESU/DPS may be at moderate risk of extinction due to projected threats or declining trends in abundance, productivity, spatial structure, or diversity. *Low* risk = neither moderate nor high risk.

judged to have an increasing viability trend, the result of natural spawner increases in multiple populations, combined with dramatic improvements in the fraction natural-origin spawners in several populations. Columbia River chum (*O. keta*) also showed marked improvement in abundance for several extant populations, although many historical populations remain extirpated or at extremely low abundance. Puget Sound steelhead also showed some evidence of improving viability, with the reversal of some previous strongly negative trends.

Methods

This report includes both a set of common analyses conducted for every ESU, as well as, in some cases, ESU-specific analyses developed by the individual TRTs. Here, we describe only the common set of analyses; see the individual sections for descriptions of the analyses that pertain to specific ESUs. Abundance and productivity were generally analyzed using quantitative methods, while spatial structure and diversity were analyzed qualitatively.

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 are typically 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, in many cases, such a statistical characterization is either not possible or has not been attempted, although many improvements have been made in the last decade (see specific sections for details). The spawning time series summarized here, and references to the methods and sources for their development, are available from NWFSC's Salmon Population Summary database,[2] 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 programs. 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.

[2] https://www.webapps.nwfsc.noaa.gov/apex/f?p=261:HOME::::::

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 three- or five-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 spawner 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. Where there are missing data, it 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

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:

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

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:

$$y_t = \alpha + \beta t + v_t \tag{2}$$

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:

$$\begin{aligned} \bar{y}_t = \bar{y}_{t-1} + \beta_t &= \bar{y}_{t-1} + u + w_t \\ y_t &= \bar{y}_t + v_t \end{aligned} \tag{3}$$

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.

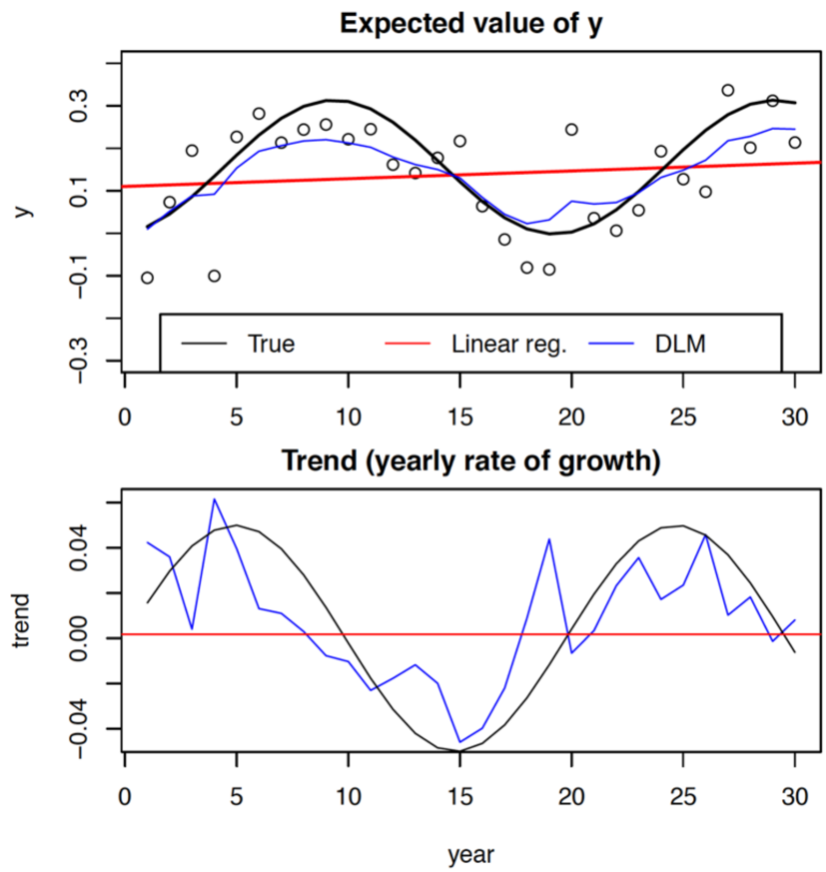

Figure 1. This figure compares a trend analysis using a non-time-varying trend (red line) via linear regression versus a trend analysis using a time-varying trend (blue line). The black line is the true line we are trying to estimate (with the red or blue line), and the dots in the top panel are the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

Multivariate DLMs for analysis of multiple time series from one ESU

A multivariate DLM allows one to estimate time-varying trends using multiple observed time series; in our case, these are populations within ESUs, 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*. The multivariate DLM allowed us to use all spawner count information in the ESU to deal with measurement error in the spawner count data, and, more importantly, to estimate missing spawner count data.

Mathematically, the model being fit is:

$$\begin{bmatrix} \bar{y}_1 \\ \bar{y}_2 \\ \bar{y}_3 \end{bmatrix}_t = \begin{bmatrix} \bar{y}_1 \\ \bar{y}_2 \\ \bar{y}_3 \end{bmatrix}_{t-1} + \begin{bmatrix} u_1 \\ u_2 \\ u_3 \end{bmatrix} + \begin{bmatrix} w_1 \\ w_2 \\ w_3 \end{bmatrix}_t$$
$$\begin{bmatrix} y_1 \\ y_2 \\ y_3 \end{bmatrix}_t = \begin{bmatrix} \bar{y}_1 \\ \bar{y}_2 \\ \bar{y}_3 \end{bmatrix}_t + \begin{bmatrix} v_1 \\ v_2 \\ v_3 \end{bmatrix}_t \tag{4}$$

The u_j are the long-term means 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.
- Unconstrained (unique variances and covariances across all populations).

For R, the following structures were explored:

- Diagonal with unequal variances (no covariance).
- Diagonal with equal variances.

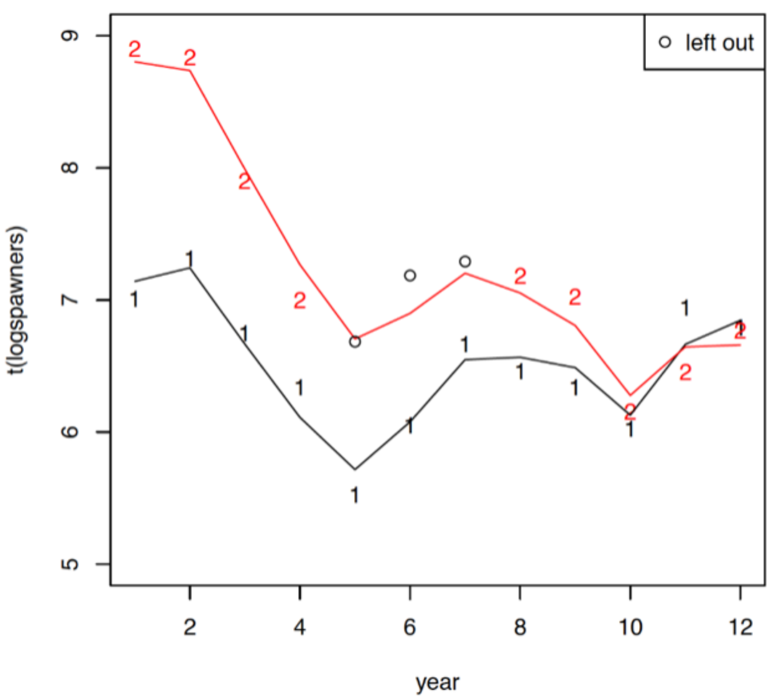

Figure 2. The estimated mean log (spawners) using a multivariate DLM. Information from years when data are available for Time-series 1 is used to inform the estimate for Time-series 2 for the missing years (marked with circles).

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 the Akaike information criterion, 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).

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 smaller 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:

$$\begin{aligned} \bar{z}_t &= \bar{z}_{t-1} + \beta_z + w_t \\ z_t &= \bar{z}_t + v_t \end{aligned} \tag{5}$$

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). Missing values were allowed within the fraction natural-origin time series and would be estimated by the DLM; however, no estimates were used more than one year before the available data or within one year after. For example, if the natural-origin data started in 2001, then the first DLM estimate would be for 2000. This prevented the model from extrapolating too far outside the data.

Summary statistics

The following summary statistics were reported for all ESUs:

- The mean total spawner DLM estimates (from the multivariate DLM fit to the raw total spawners time series in the ESU).
- The mean natural-origin spawner DLM estimates (the total spawner DLM estimate times the fraction natural-origin DLM estimate).
- The raw (original data) total spawners and the raw natural-origin spawner estimates (raw total times fraction natural-origin).

The definition of "spawner" with respect to age varied somewhat across data sources, and depended in some cases upon decisions made by data providers. For Chinook salmon, jacks (males one year younger than the model age) were included as spawners in most cases, but "minijacks" (males two or more years younger than the model age) were never included. Jacks were not included for coho salmon. For steelhead, only anadromous spawners were included.

These metrics are similar to statistics reported in prior viability reports, and provide a common set of relatively simple metrics for comparison across all ESUs/DPSes and populations, and with prior reports. In most cases, there are also ESU/DPS-specific metrics that were developed by technical recovery teams and/or included in recovery plans. Where feasible, these metrics are also reported in the individual ESU/DPS chapters.

15-year trends. A linear regression was fit to 15 years of the mean natural-origin spawner DLM estimates and the slope (trend) reported. The 15-year time period was chosen to remain consistent with prior viability reports, and does not necessarily correspond to any peaks or troughs in the time series.

5-year geometric means. 5-year geometric means $((y_1 y_2 y_3 y_4 y_5)(1/5))$ were computed from the raw total natural spawner and natural-origin spawner DLM estimates. The raw data could have missing values in the calculation, while the DLM estimates would not. For the raw estimates, when there were missing values, the geometric mean was computed only from the non-missing values. For example, if three values were available, $((y_1 y_2 y_3)(1/3))$ was reported.

Average fraction natural-origin. These were computed over five-year time frames 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 DLM estimate at year t divided by the mean total spawner DLM estimate (at year $t-3$ for coho salmon and $t-4$ for all other species). This was plotted for all years of available data.

Harvest. We compiled data on trends in the adult equivalent exploitation rate for each ESU. This information was used to provide some additional context for interpreting abundance trends, similar to the environmental trend information we also report. 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 quantitativly for several ESUs (e.g., NMFS 2001, Ford et al. 2007) and qualitatively for others (NMFS 2004). See specific sections for details.

ESU Boundaries

In its 2015 report, NWFSC (2015) recommended a revision of the steelhead Lower Columbia River DPS and Upper Willamette River DPS boundaries. Specifically, that the Clackamas River winter-run steelhead demographically independent population (DIP), originally included as part of the Lower Columbia River DPS, instead be included in the Upper Willamette River DPS. Genetic research published since 2015 further supports the closer affinity of the Clackamas River winter-run steelhead DIP to Upper Willamette River DPS populations, rather than to Lower Columbia River DPS populations (Winans et al. 2018). We believe that the rationale for revising the placement of the Clackamas River winter-run steelhead DIP originally stated in the 2015 status review is still accurate and appropriate, and does not need further review or revision.

Interior Columbia River Domain Viability Summaries

Upper Columbia River Spring-run Chinook Salmon ESU

Brief description of ESU

The Upper Columbia River 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 (USOFR 2020; Figure 3). The ESU was listed as *Endangered* under the ESA in 1999 (and re-affirmed in 2005, 2012, and 2016).

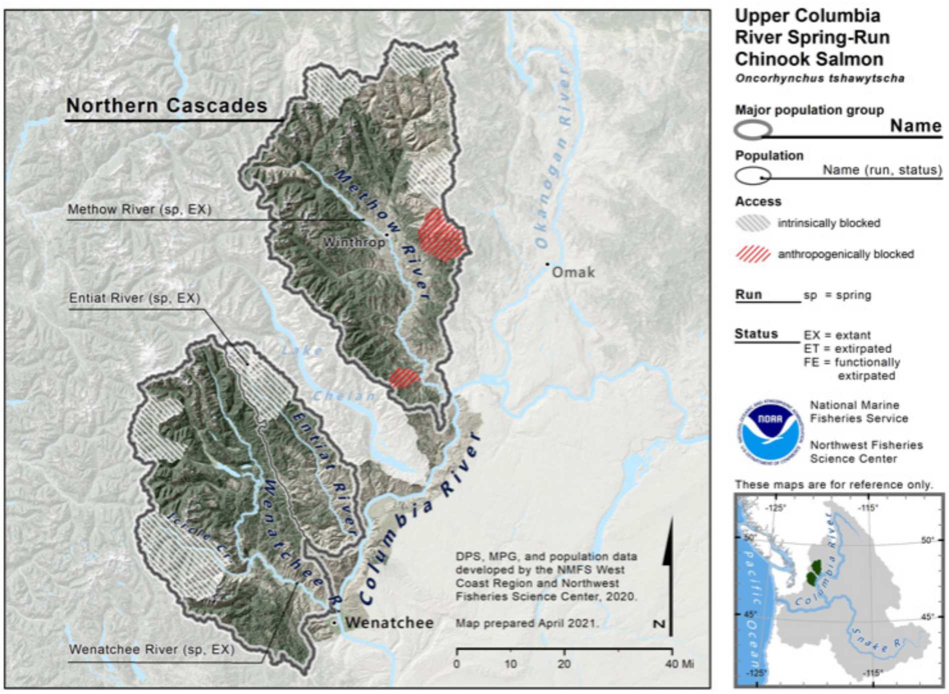

Figure 3. Map of the Upper Columbia River spring-run Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

Summary of previous viability conclusions

2005

In the 2005 review, a slight majority (53%) of the cumulative votes cast by the Biological Review Team (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 River spring-run Chinook salmon 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 viability of the ESU in 2010 was reported in Ford et al. (2011). At that time, the Upper Columbia River spring-run Chinook salmon ESU was not currently meeting the viability criteria (adapted from the Interior Columbia Technical Recovery Team [ICTRT]) in the Upper Columbia Recovery Plan (UCSRB 2007). 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, although the viability of the Upper Columbia River spring-run Chinook salmon ESU had likely improved somewhat since the time of the last BRT status review, the ESU was still clearly at moderate-to-high risk of extinction.

2015

Estimates of natural-origin spawner abundance increased relative to the levels observed in the prior reviews for all three extant populations, and productivities were higher for the Wenatchee and Entiat Rivers, and unchanged for the Methow River (NWFSC 2015). However, abundance and productivity remained well below the viable thresholds called for in UCSRB (2007) for all three populations. Based on the information available for the 2015 review, the risk category for the Upper Columbia River spring-run Chinook salmon ESU remained unchanged from the prior review. Although the viability of the ESU was improved relative to measures available at the time of listing, all three populations remained at high risk.

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 1950s (Mullan et al. 1992). Multiple pass surveys in index areas, complemented by supplemental surveys covering the majority of spawning reaches, have been conducted since the mid-1980s. 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 Wenatchee River downstream of spring-run Chinook salmon spawning areas. The data series for each population have been updated to include return years through 2019. Recent-year estimates of spawner abundance, hatchery- and natural-origin proportions, and age composition were provided by the Washington Department of Fish and Wildlife (WDFW); recruits-per-spawner data for these populations were provided by WDFW and distributed through the Columbia River Basin Coordinated Assessment Data Exchange.[3] Smolt-to-adult return rate data were estimated from PIT-tag detections and distributed by Columbia River DART (Data Access in Real Time).[4,5]

Smolt-to-adult return and recruits-per-spawner rates

Smolt-to-adult return (SAR) estimates (Bonneville Dam to Bonneville Dam) for all three Upper Columbia River spring-run Chinook salmon population data series are generated by Columbia River DART (CBR and Washington 2020) using PIT-tag detections from all release locations within each population basin (Columbia River DART et al. 2020). The indices represent cumulative marine, nearshore, and estuary survival. The SAR series includes estimates for the range of brood years 2002–15 (Figure 4). Over the period of record, the geometric mean SAR for the Entiat and Methow River populations (~3%) represents a low, but reasonable marine survival, with the Wenatchee River SAR of ~1.5% being on the low end, as 2% is roughly a replacement rate. Recruits-per-spawner (R/S) indices are reported as available from StreamNet's Coordinated Assessment Partnership data portal.[6] All populations in the ESU have low (< 1.0) R/S values, implying that the natural replacement rate is not keeping up with all sources of mortality across the life cycle.

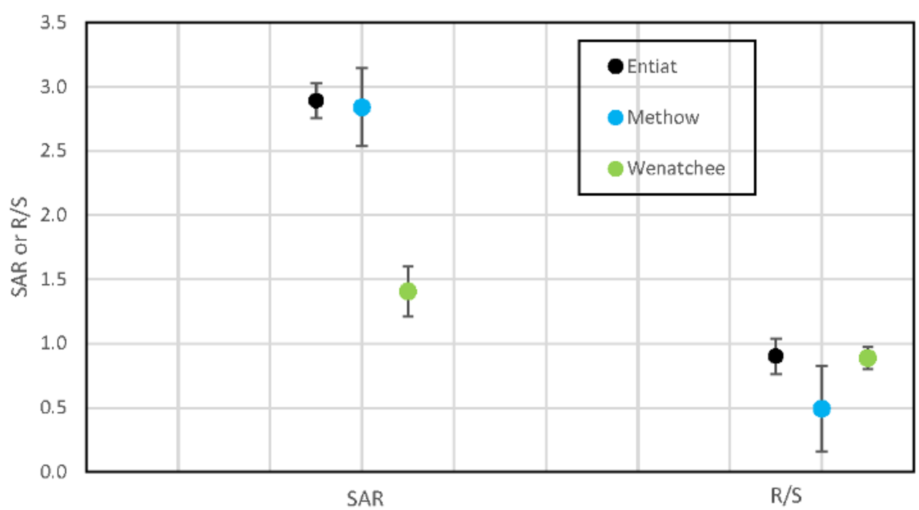

Figure 4. Smolt-to-adult return (SAR) and recruits-per-spawner (R/S) for each of the populations in the ESU. Geometric means of SAR (%) and R/S (fish/fish) are shown for each population, along with the standard error of the estimate (whiskers = ±1 SE). The time period included in the SAR or R/S indices is the past 20 years, depending on data availability.

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[3] https://www.exchangenetwork.net/data-exchange/columbia-river-basin-coordinated-assessment/
[4] http://www.cbr.washington.edu/dart
[5] Columbia River Steelhead and Chinook Natural Origin Spawner Abundance Dataset (1960–2019). Spawner abundance data. Washington Department of Fish and Wildlife and Confederated Tribes of the Colville Reservation. Protocol and methods available: at https://fortress.wa.gov/dfw/score/score/ and https://www.monitoringmethods.org/Protocol/Details/235. Accessed from www.cax.streamnet.org at 22:00 on 26 May 2020 by M. Williams.
[6] https://www.streamnet.org/cap/

Ocean condition indices

Upper Columbia River spring-run 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. 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 1–2 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 River spring-run Chinook salmon 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. 2014b). Indicators of ocean condition are highly correlated with each other, and exhibit strong temporal autocorrelation (Figure 129; Peterson et al. 2019). As a result, when indicators point to conditions that result in poor ocean productivity for salmonid populations, they do so as a suite of indicators, and for runs of "good" or "bad" years (see Habitat chapter). Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019a). Recent (2015–19) ensemble ocean indicator rankings include four of the worst seven years in the past 20, meaning that an entire Chinook salmon generation has been subjected to poor ocean productivity conditions (Figure 129).

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 similar to those used in prior BRT reviews (see Methods) and 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 DPSes to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section. 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 River spring-run Chinook salmon populations showed steep declines beginning in the late 1980s, leading to extremely low abundance levels in the mid-1990s (Figure 5, Table 2). The steep downward trend reflects the extremely low return rates for natural production from the 1990–94 brood years (Figure 6). Estimated 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 1998 BRT risk assessment prior to listing of the

ESU. Using the updated data series for this review, the short-term (five-year) trend in wild spawners has been strongly negative for all three extant populations (Table 2). Longer-term (15-year) trends are also negative for all three populations, although the 95% confidence intervals in each case include 0 (Table 3). 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 and declines after 2010. Average natural-origin returns remain well below ICTRT minimum threshold levels.

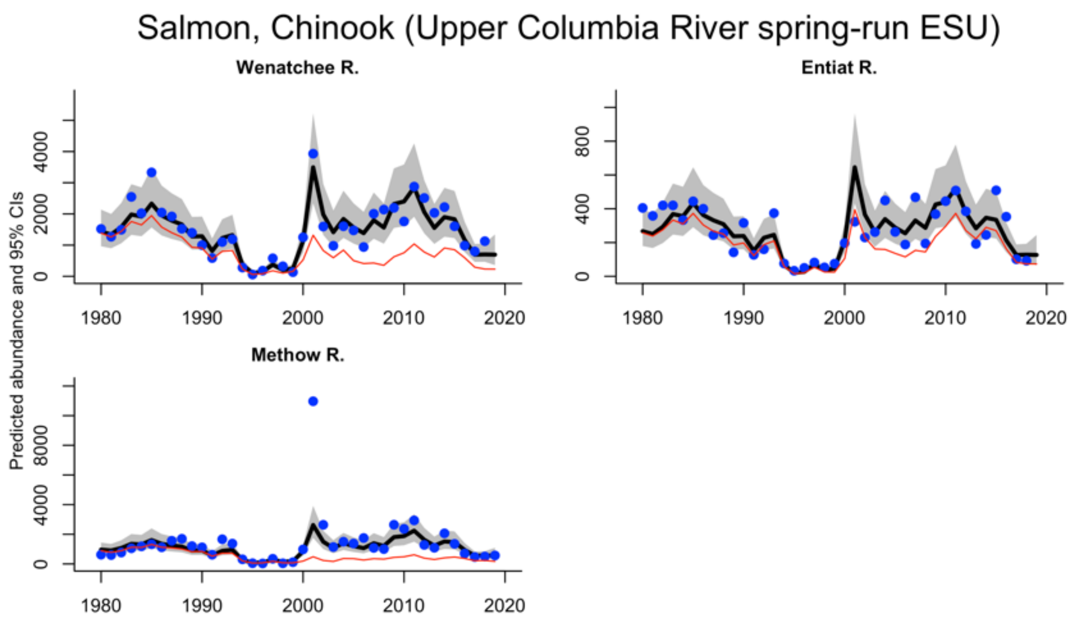

Figure 5. Smoothed trend in estimated total (thick black line, with 95% confidence internal in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

The annual natural return per spawner series for each population directly reflects the patterns in natural-origin abundance, and was only positive during a period of strong population increase (Figure 6). 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. Using the R/S and SAR indicators by population (Figure 4), it is possible to generate an indicator of freshwater productivity (FWPI) as a ratio of R/S and SAR. This quantity can be thought of as an indicator of smolts per spawner, and thus, the overall population productivity in the freshwater environment. FWPI for Upper Columbia River spring-run Chinook salmon populations are low (<100, Figure 7), confirming areas of recovery action focus such as pre-spawn mortality and juvenile rearing habitat condition. The initial risk assessment for this ESU (ICTRT 2007) found that achieving natural-origin abundance and productivity levels above the threshold viability curve corresponding to 5% risk of extinction will require substantial improvements in survival and/or natural production capacity. The long-term population productivity data indicate that this assessment is still valid.

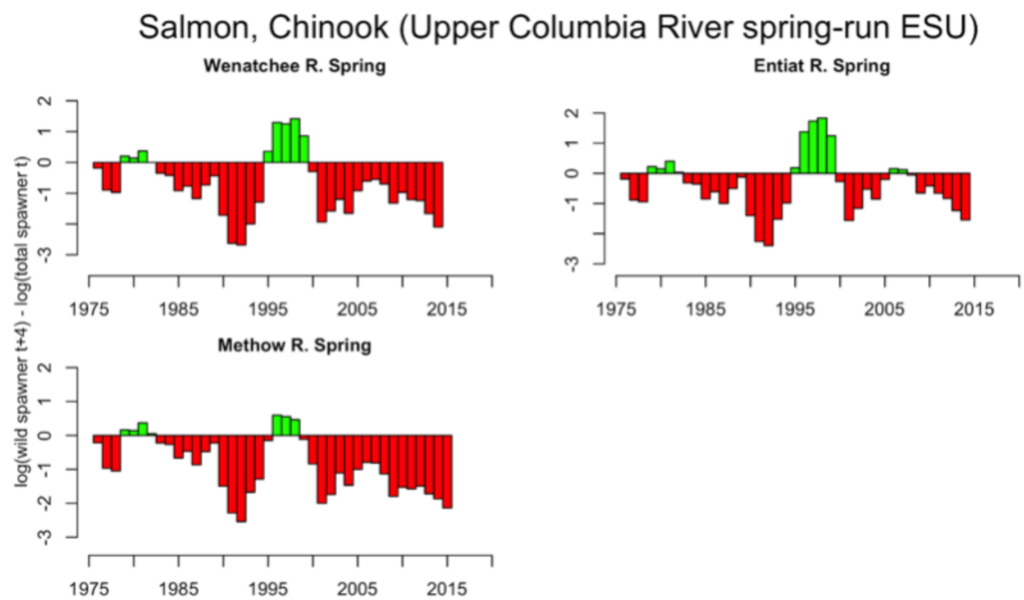

Figure 6. Trends in population productivity, estimated as the log of the smoothed natural-origin spawning abundance in year *t* minus the smoothed natural spawning abundance in year (*t* – 4). Spawning years on x-axis.

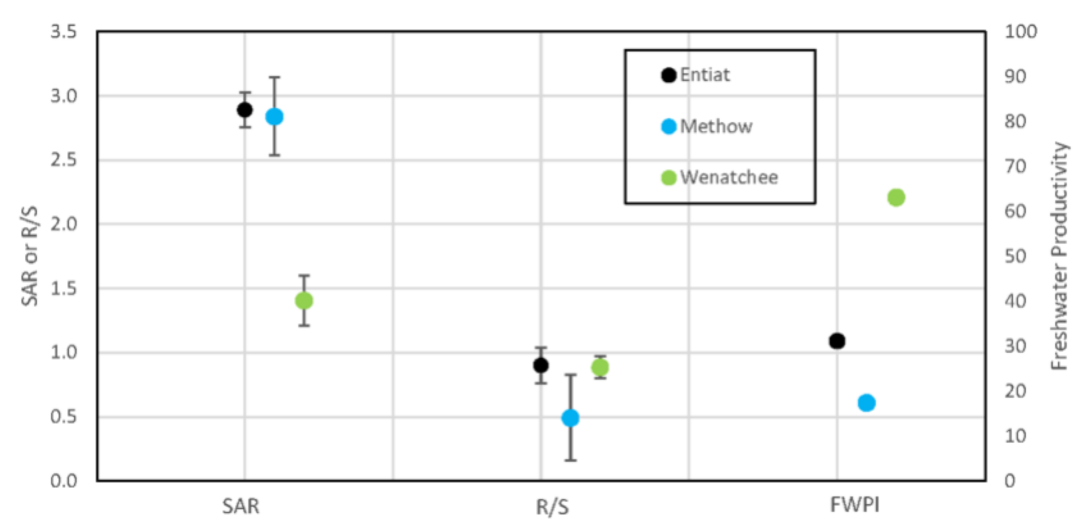

Figure 7. Smolt to adult return (SAR), recruits per spawner (R/S), and freshwater productivity index (FWPI) for each of the populations in the Upper Columbia River spring-run Chinook salmon ESU. Geometric means of SAR and R/S are shown for each population, along with the standard error of the estimate (whiskers represent ±1 SE). The time period included in the SAR or R/S indices is the past 20 years, depending on data.

Table 2. Five-year geometric mean of raw natural spawner counts in the North Cascades major population group (MPG). 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. 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 2 most recent 5-year periods is shown on the far right.

Population	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Wenatchee River SP	380 (735)	99 (192)	668 (1,652)	379 (1,671)	874 (2,247)	443 (1,092)	−49 (−51)
Entiat River SP	153 (179)	37 (56)	148 (280)	129 (278)	256 (333)	137 (202)	−46 (−39)
Methow River SP	726 (867)	44 (75)	292 (2,171)	379 (1,470)	448 (1,820)	232 (659)	−48 (−64)

Table 3. Fifteen-year trends in log natural spawner abundance in the North Cascades MPG, computed from a linear regression applied to the smoothed natural spawner log abundance estimate. Only populations with at least 4 natural spawner estimates (1980–2014) and with at least 2 data points in the first 5 years and last 5 years of the 15-year period are shown.

Population	1990–2005	2004–19
Wenatchee River SP	0.03 (−0.09, 0.14)	−0.03 (−0.09, 0.02)
Entiat River SP	0.03 (−0.09, 0.15)	−0.03 (−0.09, 0.03)
Methow River SP	−0.05 (−0.15, 0.06)	−0.03 (−0.06, 0.01)

Non-treaty harvest

Spring Chinook salmon from the upper Columbia River basin migrate offshore in marine water and where impacts in ocean salmon fisheries are too low to be quantified. The only significant harvest in salmon fisheries occurs in the mainstem Columbia River in tribal and non-tribal fisheries directed at hatchery spring-run Chinook salmon from the Columbia and Willamette Rivers. Exploitation rates have remained relatively low, generally below the target rate of 2% (Figure 8).

Spatial structure and diversity

Abundance and productivity are demographic characteristics of a population that determine its ability to persist into the foreseeable future. Spatial structure and diversity, the other two VSP parameters (McElhany et al. 2000), are characteristics that influence a population's ability to persist and evolve over a much longer time course. Spatial structure and diversity consider a population's 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. Spatial structure and diversity buffer a population against short-term environmental fluctuations and long-term climatic change. Compromised spatial structure and diversity are ultimately expressed as longer-term declines in abundance and productivity.

The proportions of natural-origin contributions to spawning in the Wenatchee and Methow River populations have trended downwards from 1990 to 2008 (Figure 9, Table 4), reflecting the large increase in releases and subsequent returns from the directed supplementation programs in those two drainages (Hillman et al. 2015). Natural-origin fractions increased from 2009 to 2017, reflecting increasing natural-origin abundance trends and changes to hatchery management, before declining again in the last two years. There is currently no direct spring-run hatchery supplementation program in the Entiat River, though the summer-run releases do have the potential to impact the spring run through redd superimposition. Prior to 2011, hatchery-origin spawners in the Entiat River system were predominately strays from Entiat National Fish Hatchery (NFH) releases. The Entiat NFH spring Chinook salmon 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 River (Ford et al. 2015). The nearby Eastbank Hatchery facility is used for rearing the Wenatchee River supplementation stock prior to transfer to the Chiwawa River acclimation pond. It is possible that some of the returns from that program are homing on the Eastbank Hatchery facility and then straying into the Entiat River, the nearest spawning area.

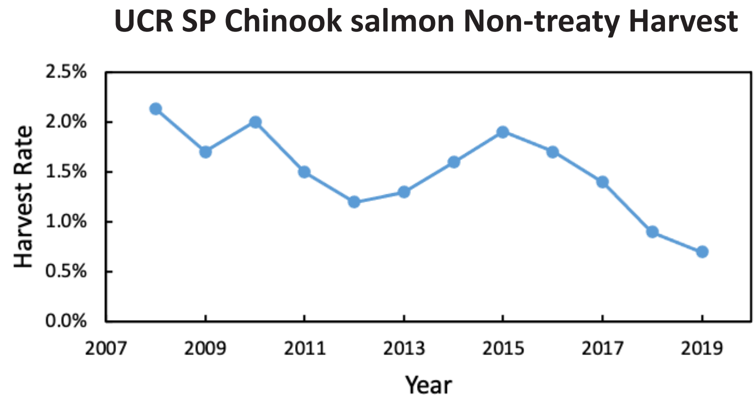

Figure 8. Non-treaty harvest rate for Upper Columbia River spring-run Chinook salmon. Data from the Columbia River Technical Advisory Committee (TAC 2020).

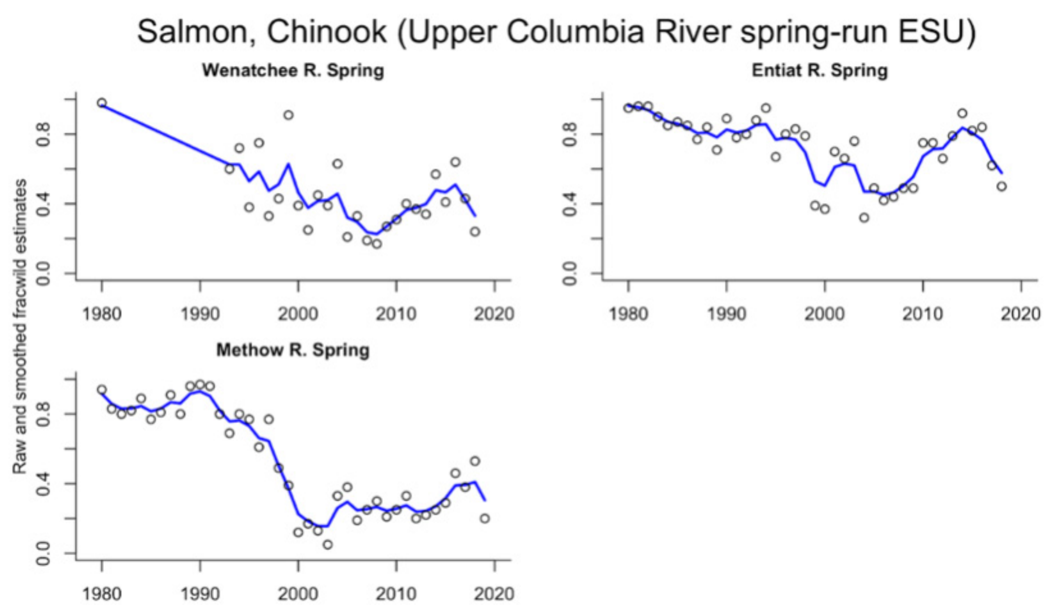

Figure 9. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 4. Five-year mean of fraction natural-origin (sum of all estimates divided by number of estimates).

Population	1995–99	2000–04	2005–09	2010–14	2015–19
Wenatchee River SP	0.56	0.42	0.23	0.40	0.43
Entiat River SP	0.70	0.56	0.47	0.77	0.70
Methow River SP	0.61	0.16	0.27	0.25	0.37

Biological viability relative to recovery goals

NMFS adopted a recovery plan for Upper Columbia River spring-run Chinook salmon and steelhead in 2007 (USOFR 2007b). The plan was developed by the Upper Columbia Salmon Recovery Board (UCSRB) and is available through their website.[7] The 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" (p. 13).

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-run Chinook salmon within each of the extant upper Columbia River populations, measured as eight-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 low 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 low risk score at the population level.

Achieving recovery (delisting) of each ESU via sufficient improvements in 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-run Chinook salmon ESU against the recovery objective. First, "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, second, "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" (p. 119). 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

[7] https://www.ucsrb.org/

Table 5. Upper Columbia River spring-run Chinook salmon ESU population viability status summary. Current abundance and productivity estimates are geometric means (most-recent 10 years for abundance and 20 years for productivity). Standard deviation of annual abundance, standard error, and number of qualifying estimates for productivities in parentheses.

	Abundance/productivity (A/P) metrics				Spatial structure/diversity (SS/D) metrics			
Population	**ICTRT threshold**	**Natural spawning**	**ICTRT productivity**	**Integrated A/P risk**	**Natural processes**	**Diversity risk**	**Integrated SS/D risk**	**Overall risk rating**
Wenatchee River SP	2,000	630 (SD 261)	0.89 (0.09, 17/20)	High	Low	High	High	High
Entiat River SP	500	193 (SD 126)	0.90 (0.14, 19/20)	High	Moderate	High	High	High
Methow River SP	2,000	323 (SD 251)	0.49 (0.33, 19/20)	High	Low	High	High	High

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 UCSRB plan adopted an alternative approach for addressing the limited number of populations in the ESU—a 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" (p. 116). Specific criteria included in the UCSRB plan reflect a combination of the specific criteria recommended by the ICTRT (ICTRT 2007) and in the earlier Quantitative Scientific Report 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-run Chinook salmon population includes a single historical major spawning area.

Recovery update

The UCSRB plan calls for meeting or exceeding the same basic spatial structure and diversity criteria adopted from the ICTRT viability report for recovery.

Overall abundance and productivity (A/P) remains rated at high risk for the each of the three extant populations in this MPG (Table 5). The ten-year geometric mean abundance of adult natural-origin spawners has not changed by more than 25% relative to the levels reported in the 2015 status update. Natural-origin escapements still remains well below the corresponding ICTRT thresholds for all populations. 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 5). 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 the 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 2007).

Based on the combined ratings for A/P and SS/D, all three of the extant populations of Upper Columbia River spring-run Chinook salmon remain rated at high overall risk (Table 5).

Updated biological risk summary

Current estimates of natural-origin spawner abundance decreased substantially relative to the levels observed in the prior review for all three extant populations. Productivities also continued to be very low, and both abundance and productivity remained well below the viable thresholds called for in the Upper Columbia Salmon 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—in particular, a recent run of poor ocean condition years. 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 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.

Given the high degree of year-to-year variability in life stage survivals and the time lags resulting from the five-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 (Zabel and Jordan 2020, Chapter 6). Based on the information available for this review, the Upper Columbia River spring-run Chinook salmon ESU remains at high risk, with viability largely unchanged from the prior review (NWFSC 2015).

Upper Columbia River Steelhead DPS

Brief description of DPS

The Upper Columbia River steelhead DPS includes all naturally spawned anadromous steelhead (*Oncorhynchus mykiss*) populations below natural and manmade impassable barriers in streams in the Columbia River basin upstream from the Yakima River, Washington, to the U.S.–Canada border, as well as six artificial propagation programs (USOFR 2020; Figure 10). The Upper Columbia River steelhead DPS was originally listed under the ESA in 1997; it is currently designated as *Threatened*.

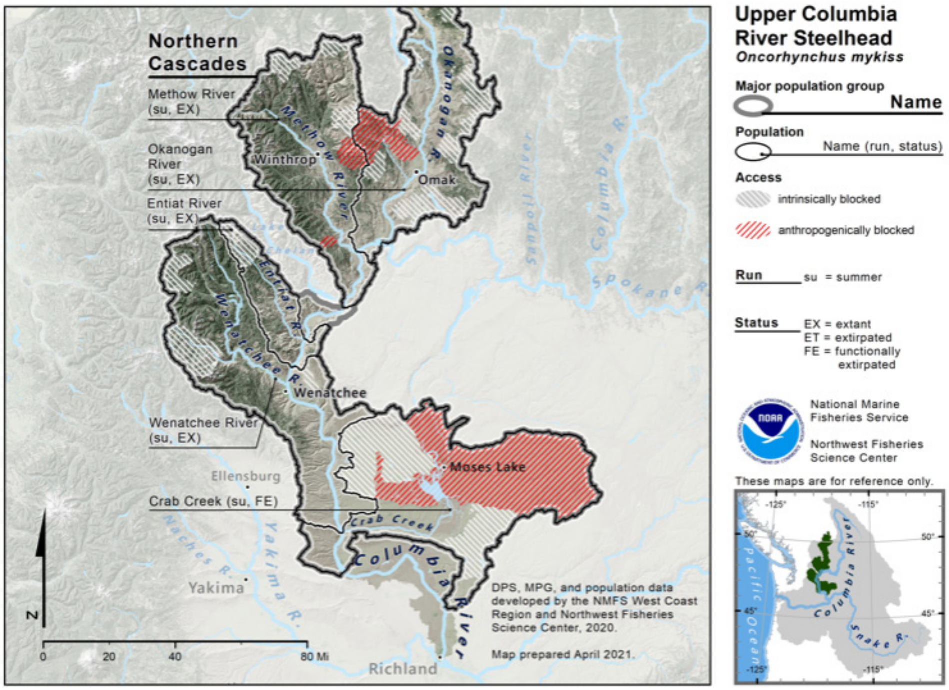

Figure 10. Map of the Upper Columbia River steelhead DPS's spawning and rearing areas, illustrating populations and major population groups.

NMFS has defined steelhead DPSes to include only the anadromous members of this species (USOFR 2005b). Our approach to assessing the current viability of a steelhead DPS is based on evaluating information on the abundance, productivity, spatial structure, and diversity of the anadromous component of the species (Good et al. 2005, USOFR 2020). Many steelhead populations along the U.S. West Coast 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 DPSes (Good et al. 2005, USOFR 2020). 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 viability of the anadromous form.

Summary of previous viability 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 Snake River Basin and Middle Columbia 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 River 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 in the years prior to the review were probably primarily the result of several years of relatively good natural survival in the ocean and in tributary habitats. Tributary habitat actions called for in the Upper Columbia Salmon 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.

2015

Based on the review in 2015, upper Columbia River steelhead populations were determined to have increased relative to the low levels observed in the 1990s, but natural-origin abundance and productivity remained well below viability thresholds for three out of the four populations (NWFSC 2015). The viability 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 viability remains unchanged from the prior review, remaining at high risk driven by low abundance 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 River basin DPSes, the populations differ in the relative changes in survival or limiting capacities that could lead to viable ratings. The required improvement to improve the abundance/productivity estimates for Upper Columbia River steelhead DPS populations is at the high end of the range for all listed interior populations.

Description of new data available for this review

The 2015 NWFSC status review (NWFSC 2015) evaluated the viability of the Upper Columbia River steelhead DPS based on data series through cycle year 2013–14 for each of the four extant populations, along with sampling information collected at Priest Rapids Dam (for the aggregate return to the upper Columbia River basin) and Wells Dam (for the Methow and Okanogan River populations combined). Estimates generated using that methodology are currently available through the 2018–19 cycle years for each population. Spawning escapement estimates are based on a run reconstruction model incorporating annual dam counts, the 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 Washington Department of Fish and Wildlife (WDFW) regional staff (incorporating information from the Colville Tribal Fish and Wildlife Department) and are available through the StreamNet Coordinated Assessment Partnership website.[8] 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[9] 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 (Waterhouse et al. 2020). Comparisons of the results from the updated approach with the methods used in prior years indicate that they generally produce compatible estimates for a given year. Preliminary results are included in this assessment, in parallel to the ongoing data collection-based population assessments, with the understanding that ongoing methodological and data evaluations will result in a single approach to annual population enumeration in the future.

Smolt-to-adult return and recruits per spawner rates

Smolt-to-adult return (SAR) estimates (Bonneville to Bonneville) for all four Upper Columbia River steelhead population data series are generated by the Columbia River Data Access in Real Time (CBR and UW 2020) project using PIT-tag detections from all release locations within each population basin (CBR and UW 2020). The indices represent cumulative marine, nearshore, and estuary survival. The SAR series includes estimates for the range of brood years 2002–15 (Figure 11). Over the period of record, the geometric mean SAR for the Entiat, Methow, and Okanogan River populations (~3%) represents a low, but reasonable marine survival (2% is generally considered a minimal replacement rate), with the Wenatchee SAR of ~5% being a robust rate for a stable population. Recruits per spawner (R/S) indices are reported as available from the StreamNet data portal (StreamNet 2020). All populations in the ESU have low (<1.0) R/S values, implying that the natural replacement rate is not keeping up with all sources of mortality across the life cycle.

[8] https://www.streamnet.org/cap/
[9] PIT = passive integrated transponder.

Ocean condition indices

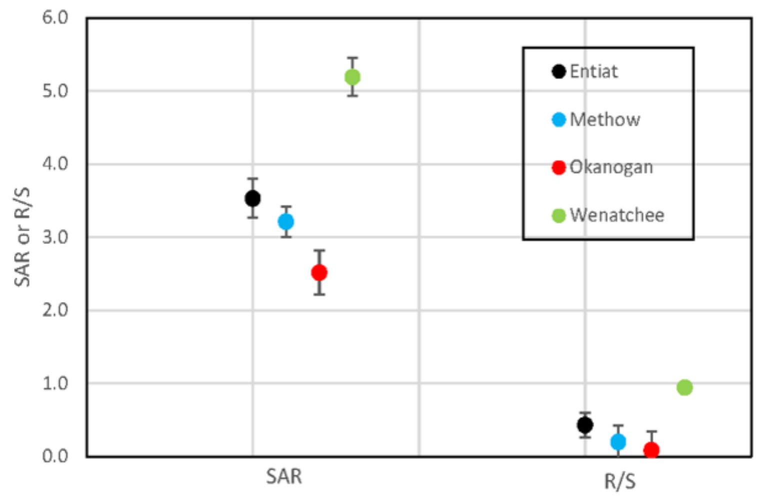

Figure 11. Smolt-to-adult return and recruits per spawner for each of the populations in the ESU. Geometric means of SAR and R/S are shown for each population, along with the standard error of the estimate (whiskers represent ±1 SE). The time period included in the SAR or R/S indices is the past 20 years, depending on data availability.

Juvenile steelhead are more pelagic than salmon, heading off the continental shelf soon after entering the ocean in the spring (Burgner 1992). Steelhead migrate seasonally across the North Pacific Ocean, moving to the north and west in spring and to the south and east, across the entire Pacific, from autumn through winter (Atcheson et al. 2012). Thus, steelhead ocean survival may be impacted by different factors than salmon. In fact, recent work has shown that steelhead population groupings from geographic regions have unique smolt survival trends that appear to be driven by factors affecting them early in their ocean residence, despite steelhead smolts generally being larger than Pacific salmon smolts when they enter the ocean and all making wide-ranging, off-the-continental-shelf migrations, rather than remaining more coastal, as Pacific salmon smolts tend to do (Kendall et al. 2017).

Aggregate annual returns of Columbia River steelhead 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). Work is ongoing to relate indices of ocean condition to steelhead populations up and down the U.S. West Coast. Steelhead marine survival seems to be related to ocean surface temperature in the first summer of ocean entry, and populations respond similarly to spatial patterns of ocean conditions at a rough grain of 250 km between ocean entry points (Kendall et al. 2017). Therefore, broad spatial patterns of ocean conditions may not capture the finer spatial scale of response that steelhead seem to exhibit.

Indicators of ocean condition are highly correlated with each other, and exhibit strong temporal autocorrelation (Figure 129; Peterson et al. 2019). As a result, when indicators point to conditions that result in poor ocean productivity for salmonid populations, they do so as a suite of indicators, and for runs of "good" or "bad" years (see Habitat chapter). Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019a). Recent (2015–19) ensemble ocean indicator rankings include four of the worst seven years in the past 20, meaning that an entire salmon or steelhead generation could have been subjected to poor ocean productivity conditions (Figure 129).

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 ESA Status Reviews, as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this DPS. The standard Status Review metrics were consistently done across all ESUs and DPSes to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section. 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 (five-year geometric mean) of total and natural-origin spawner abundance have declined dramatically, erasing gains observed over the past two decades for all four populations (Figure 12, Table 6). Recent declines are persistent and large enough to result in small, but negative, 15-year trends in abundance for all four populations (Table 7). Updated spawner estimation methods show a strong concordance with existing methods (Figure 13), which is extremely encouraging as the estimation process based on detecting tags from a run-at-large tagging program is a very robust approach to monitoring across the DPS. Annual brood-year R/S estimates have been well below replacement in recent years for all four populations. The R/S estimates summarized in Figure 14 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. All populations are consistently exhibiting natural production rates well below replacement, and natural production has also declined consistently, resulting in an increasing fraction of hatchery fish on the spawning grounds each year.

The required improvement to increase abundance and productivity for Upper Columbia River steelhead populations is at the high end of the range for all listed interior populations. Using the R/S and SAR indicators by population (Figure 15), it is possible to generate an indicator of freshwater productivity (FWPI) as a ratio of R/S and SAR. This quantity can be thought of as an indicator of smolts per spawner, and thus, the overall population productivity in the freshwater environment. FWPI for Upper Columbia River steelhead populations is very low (<100, Figure 15), confirming areas of recovery action focus such as pre-spawn mortality and juvenile rearing habitat condition, as well as mainstem migratory impacts as the SAR are based on Bonneville to Bonneville tag detections and the R/S are based on spawning-ground recruits. The initial ICTRT assessment of abundance/productivity gaps resulted in a pattern similar to that indicated by the long-term productivity metrics (ICTRT 2007). The Wenatchee River population has somewhat higher productivity (SAR, R/S, and FWPI) than the remaining populations in the DPS, but still falls into a high-risk category due to the recent downward trend in both abundance and productivity.

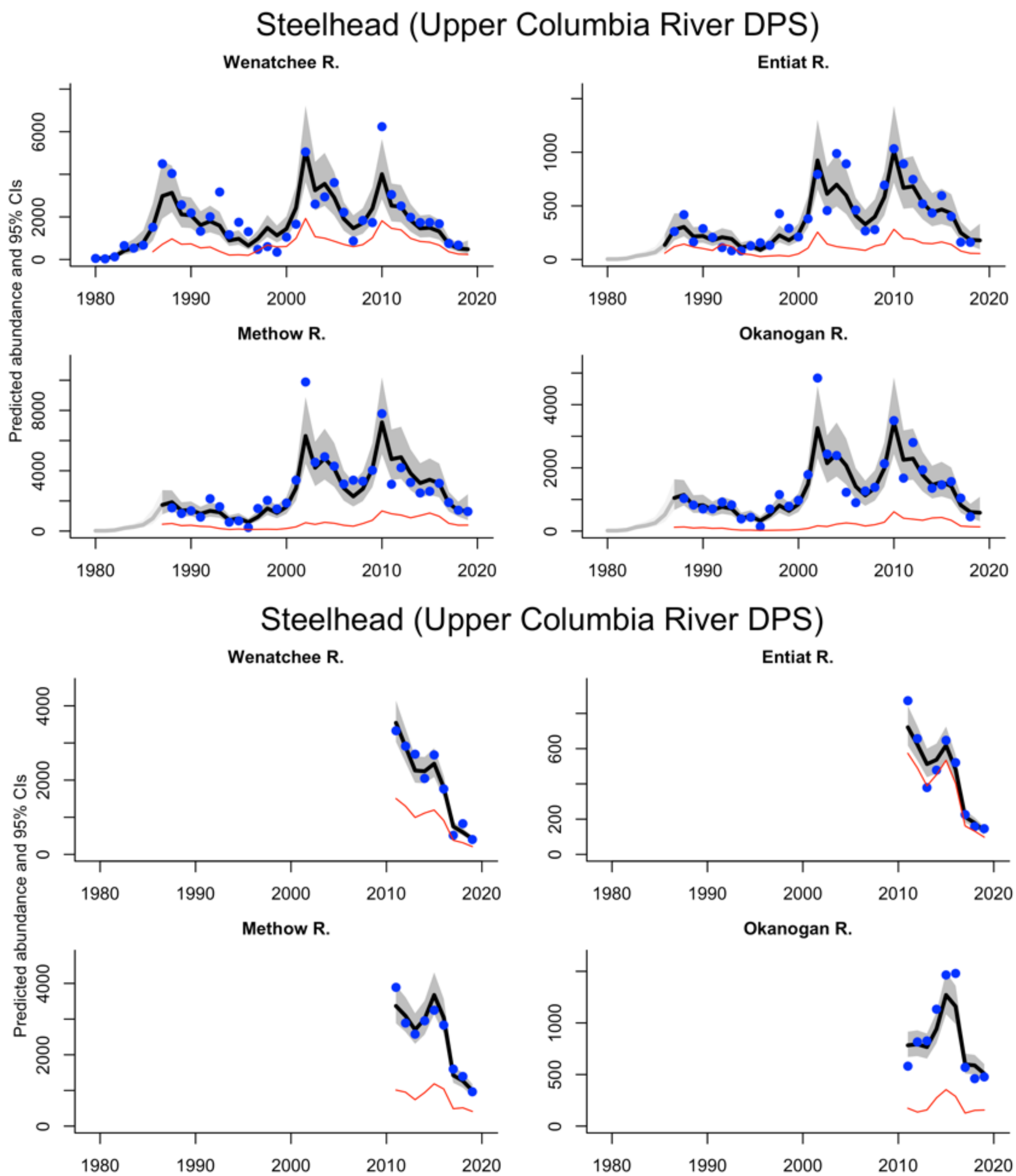

Figure 12. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot. Upper panel is the traditionally generated spawner abundance time series for each population. Lower panel is population estimates based on PIT-tag detections within each population watershed relative to tagging the aggregate upper Columbia River run at large.

Table 6. Five-year geometric mean of raw natural spawner counts in the North Cascades MPG. 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. 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 2 most recent 5-year periods is shown on the far right.

Population	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Wenatchee River	527 (1,847)	264 (741)	774 (2,319)	684 (1,857)	1,497 (2,774)	554 (1,104)	−63 (−60)
Entiat River	68 (134)	38 (201)	110 (491)	102 (462)	203 (688)	92 (280)	−55 (−59)
Methow River	210 (1,206)	97 (937)	435 (4,255)	523 (3,599)	829 (3,833)	595 (1,954)	−28 (−49)
Okanogan River	65 (678)	25 (526)	124 (2,178)	184 (1,328)	404 (2,122)	223 (1,020)	−45 (−52)

Table 7. Fifteen-year trends in log natural spawner abundance in the North Cascades MPG, computed from a linear regression applied to the smoothed natural spawner log abundance estimate. Only populations with at least 4 natural spawner estimates (1980–2014) and with at least 2 data points in the first 5 years and last 5 years of the 15-year period are shown.

Population	1990–2005	2004–19
Wenatchee River	0.06 (0.01, 0.12)	−0.10 (−0.15, −0.06)
Entiat River	0.11 (0.05, 0.17)	−0.06 (−0.11, −0.01)
Methow River	0.11 (0.06, 0.17)	−0.05 (−0.10, −0.01)
Okanogan River	0.10 (0.05, 0.16)	−0.06 (−0.11, −0.02)

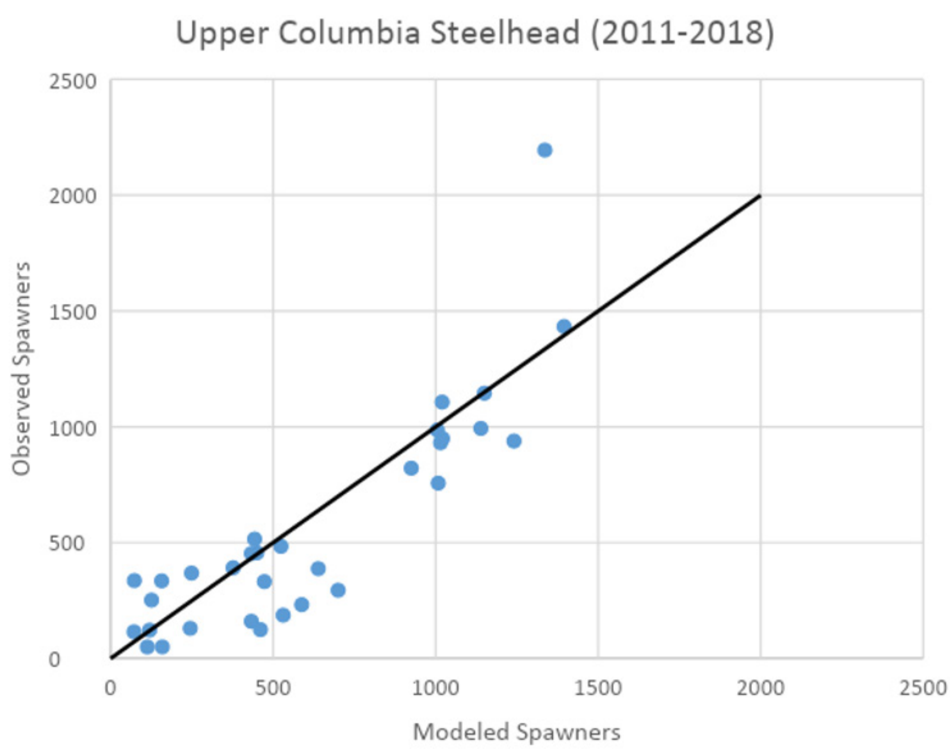

Figure 13. Estimated spawners relative to observed spawners across all 4 populations for all years where both measures exist. Black line represents the expected 1:1 relationship if the run-at-large, tagging-based, model-estimated population measures match the current redd survey-based measures.

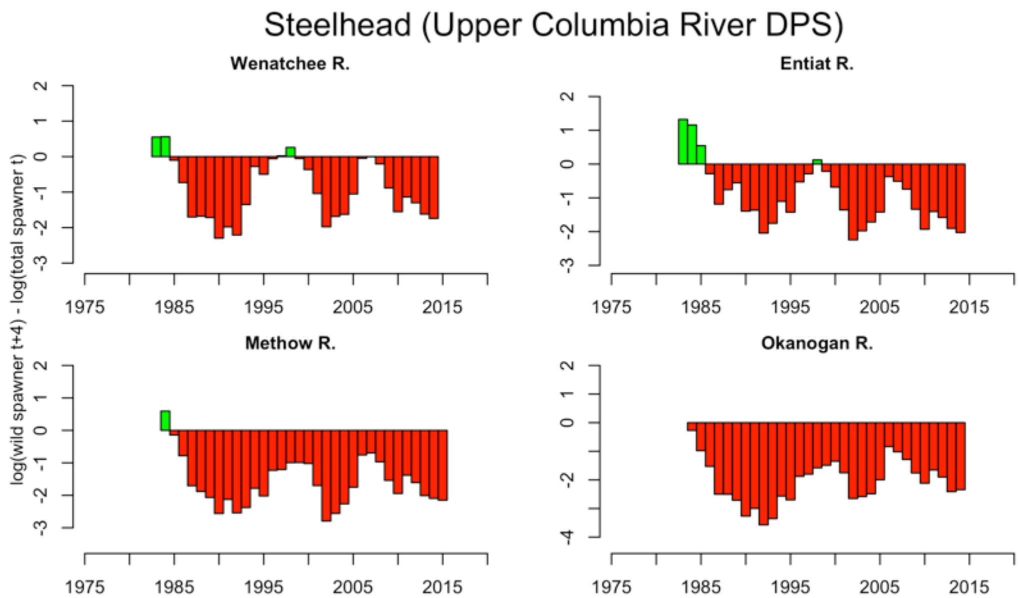

Figure 14. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year (*t* – 4). Spawning years on x-axis.

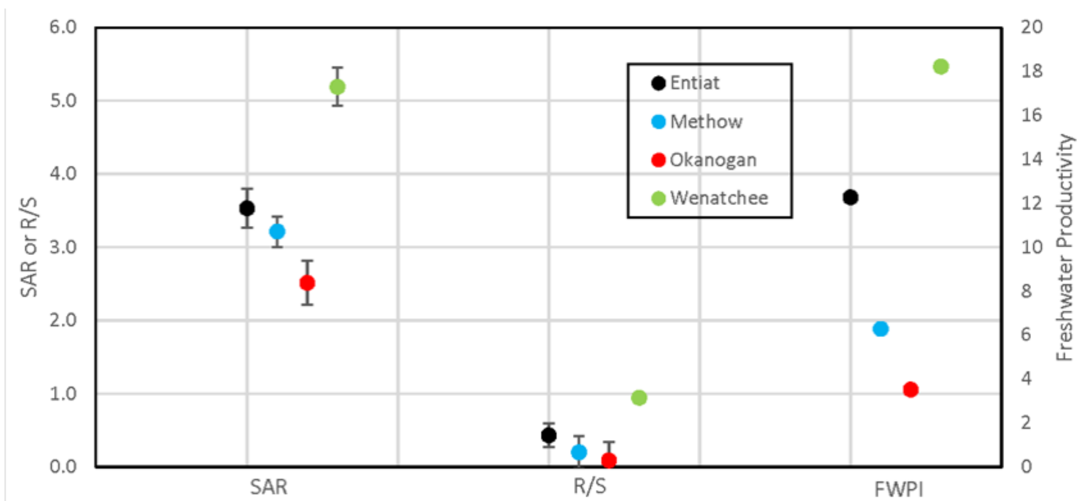

Figure 15. Smolt to Adult Return, Recruits per Spawner, and Freshwater Productivity Index (FWPI) for each of the populations in the DPS. Geometric means of SAR and R/S are shown for each population, along with the standard error of the estimate (whiskers represent +/- one standard error). The time period included in the SAR or R/S indices is the past 20 years, depending on data availability. The FWPI is constructed as a ratio of the geomean R/S and SAR, and can be thought of as a measure of smolts per spawner.

Spatial structure and diversity

With the exception of the Okanogan population, the upper Columbia River steelhead 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 steelhead DPS. More recent studies within the Wenatchee River basin have found differences between samples from the Peshastin River, believed to be relatively isolated from hatchery spawning, and those from other reaches in the basin. 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 are underway based on sampling in the Wenatchee River, as well as other Upper Columbia River steelhead DPS tributaries, and should allow for future analyses of current genetic structure and any impacts of changing hatchery release practices.

Hatchery-origin returns continue to constitute a high fraction of total spawners in natural spawning areas for this DPS (Table 8). The estimated proportion of natural-origin spawners has increased consistently since the late 1990s for all four populations (Figure 16). Natural-origin proportions were 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-tag detections, hatchery-origin spawners in the Entiat River include stray hatchery returns from releases into the Wenatchee River (Hillman et al. 2015). Reproductive effectiveness studies on the Wenatchee River population indicate that hatchery–hatchery matings have dramatically reduced fitness compared to matings among natural-origin fish (Ford et al. 2016), reinforcing the weight the ICTRT placed on the presence of hatchery-origin spawners in the SS/D metric.

Table 8. Five-year mean of fraction natural-origin (sum of all estimates divided by the number of estimates).

Population	**1995–99**	**2000–04**	**2005–09**	**2010–14**	**2015–19**
Wenatchee River	0.41	0.34	0.38	0.56	0.50
Entiat River	0.21	0.24	0.24	0.30	0.33
Methow River	0.13	0.11	0.15	0.24	0.31
Okanogan River	0.05	0.06	0.14	0.21	0.24

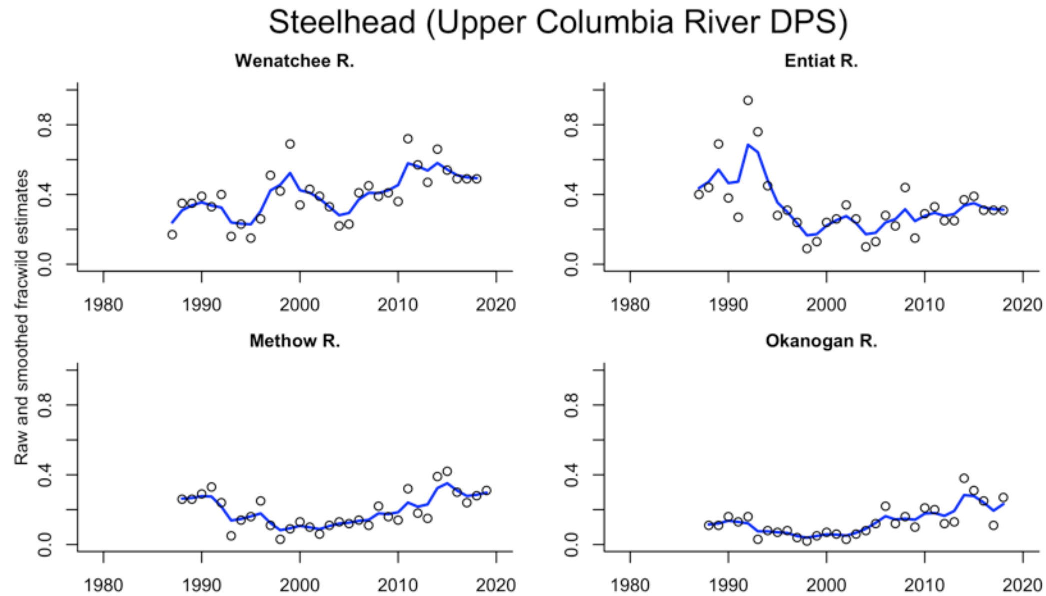

Figure 16. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

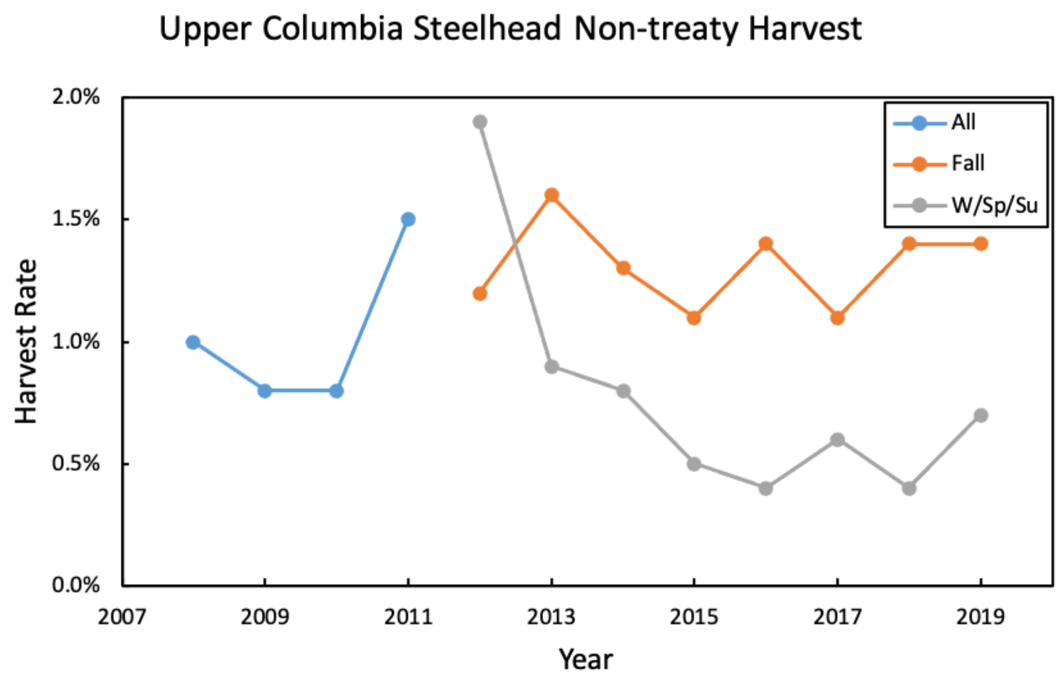

Figure 17. Harvest rates for non-treaty Upper Columbia River steelhead. As of 2012, reporting is generated across two management periods, Fall (orange line) and Winter/Spring/Summer (gray line). Prior to 2012, harvest rate reporting was across all of the calendar year (blue line; TAC 2020).

Non-treaty harvest

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-treaty commercial fisheries was prohibited, and in the mid-1980s, tributary recreational fisheries in Washington adopted mark-selective regulations. There is incidental mortality associated with mark-selective recreational fisheries. Sport fisheries targeting hatchery-run steelhead occur in the mainstem Columbia River and in several upper Columbia River tributaries. In recent years, upper Columbia River exploitation rates have been stable at around 1.5%. As of 2012, rates are estimated over two management intervals per year, Fall and Winter/Spring/Summer (Figure 17).

Biological viability relative to recovery goals

NMFS adopted a recovery plan for Upper Columbia River spring-run Chinook salmon and steelhead in 2007 (USOFR 2007b). The plan was developed by the Upper Columbia Salmon Recovery Board (UCSRB) and is available through their website.

Achieving recovery (delisting) of each ESU via sufficient improvement in abundance, productivity, spatial structure, and diversity is the longer-term goal of the UCSRB plan. The plan includes specific quantitative criteria expressed relative to population viability curves (ICTRT 2007). It includes two quantitative criteria for assessing the viability of the steelhead DPS against the recovery objective: First, "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, second, "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" (p. 121). 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 and Okanogan River 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 DPS. The UCSRB plan adopted an alternative approach for addressing the limited number of populations in the DPS—5% or less risk of extinction for at least three of the four extant populations.

The 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" (p. 117). Specific criteria included in the 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. In the Entiat River, natural spawning should be observed in both historical major spawning areas, with a distribution criteria similar to the Methow River. In the Okanogan River, natural spawning should be observed within the two major spawning areas and within at least two of the five minor spawning areas, with a numerical distribution similar to the Methow River, across the minor spawning areas. 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 low 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 low risk score at the population level.

Table 9. Viability assessments for extant Upper Columbia River 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.

	Abundance/productivity (A/P) metrics				**Spatial structure/diversity (SS/D) metrics**			
Population	**ICTRT threshold**	**Natural spawning**	**ICTRT productivity**	**Integrated A/P risk**	**Natural processes**	**Diversity risk**	**Integrated SS/D risk**	**Overall risk rating**
Wenatchee River	1,000	931 (SD 667)	0.95 (0.06, 13/20)	Moderate	Low	High	High	High
Entiat River	500	140 (SD 89)	0.433 (0.17, 20/20)	High	Moderate	High	High	High
Methow River	1,000	703 (SD 297)	0.20 (0.22, 12/20)	High	Low	High	High	High
Okanogan River	750	297 (SD 189)	0.09 (0.25, 19/20)	High	High	High	High	High

Updated biological risk summary

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 toward 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. Tributary habitat actions called for in the Upper Columbia Salmon 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.

The most recent estimates (five-year geometric mean) of total and natural-origin spawner abundance have declined since the last report, largely erasing gains observed over the past two decades for all four populations (Figure 12, Table 6). Recent declines are persistent and large enough to result in small, but negative 15-year trends in abundance for all four populations (Table 7). The abundance and productivity viability rating for the Wenatchee River exceeds the minimum threshold for 5% extinction risk. The overall Upper Columbia River steelhead DPS viability remains largely unchanged from the prior review, and the DPS is at high risk driven by low abundance and productivity relative to viability objectives and diversity concerns (Tables 9 and 10).

Table 10. Upper Columbia River steelhead DPS population risk ratings integrated across the four VSP parameters. Viability key: HV = highly viable, V = viable, M = maintained, HR = high risk (does not meet viability criteria).

		SS/D risk			
		Very low	**Low**	**Moderate**	**High**
A/P risk	**Very low (<1%)**	HV	HV	V	M
	Low (1–5%)	V	V	V	M
	Moderate (6–25%)	M	M	M	HR (Wenatchee River)
	High (>25%)	HR	HR	HR	HR (Entiat, Methow, Okanogan Rivers)

Snake River Spring/Summer-run Chinook Salmon ESU

Brief description of ESU

The Snake River spring/summer-run Chinook salmon ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon, Grande Ronde, Imnaha, and Salmon River sub-basins, as well as in fifteen artificial propagation programs (USOFR 2020; Figure 18). The ESU was first listed under the ESA in 1992, and the listing was reaffirmed in 2005 and 2012.

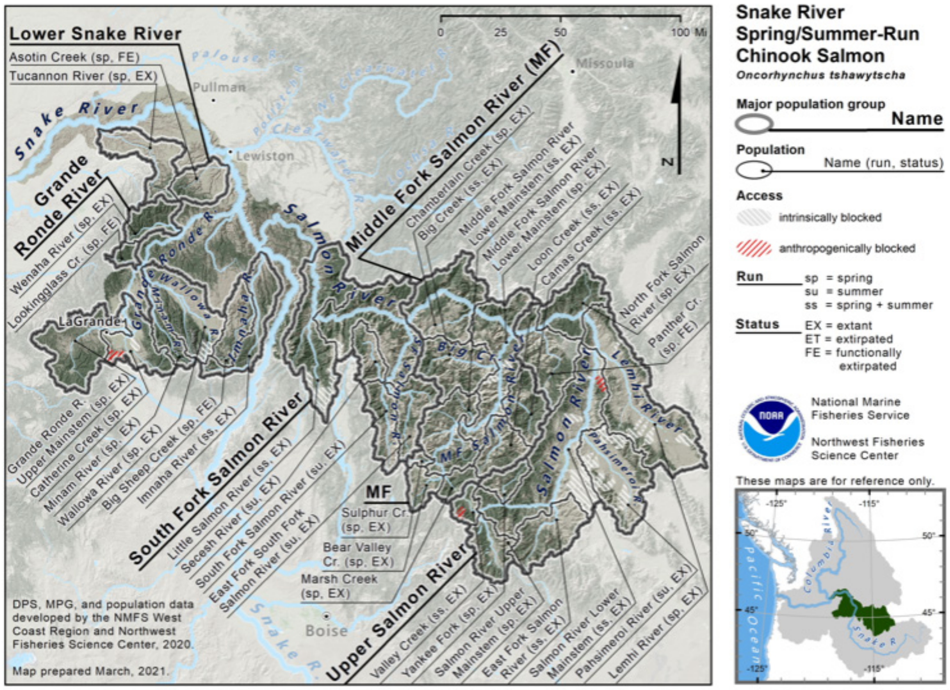

Figure 18. The Snake River spring/summer-run Chinook salmon ESU' spawning and rearing areas, illustrating populations and major population groups.

Summary of previous viability conclusions

2005

The 2005 BRT report evaluated the viability of Snake River spring/summer-run Chinook salmon 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 had 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, that abundance should be measured over at least an eight-year period, and that, by this measure, the then-recent abundance levels across the ESU fell 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 River basin was viewed as a positive action.

2010

Ford et al. (2011) concluded that population-level viability 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.

2015

NWFSC (2015) concluded that the majority of populations in the Snake River spring/summer-run Chinook salmon ESU remained at high overall risk. Natural-origin abundance had increased over the levels reported in the prior review for most populations in this ESU, but 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, and two populations decreased in abundance and increased in productivity. Spatial structure ratings remained unchanged from the prior reviews, with low or moderate risk levels for the majority of populations in the ESU.

Description of new data available for this review

The previous ESA status review (NWFSC 2015) 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 2018 or 2019 return year for populations across all of the MPGs in the Snake River spring/summer-run Chinook salmon ESU. Data and analyses used in this assessment were obtained primarily from state and tribal fisheries agencies. Oregon Department of Fish and Wildlife (ODFW), WDFW, and Idaho Department of Fish and Game (IDFG) updated annual estimates of spawning escapement, hatchery/natural spawner fractions, and age composition for most populations, often incorporating data generated by regional projects conducted by the Nez Perce, Umatilla, and Shoshone–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 Department of Fisheries Resources Management [NPT]). A major advance since the data compilation efforts leading to the 2015 ESA 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 viability indicators, including spawning abundance, hatchery/natural proportions, and age structure through the Coordinated Assessment Partnership.

Efforts to refine and document the estimates for individual populations have continued. In most cases, updates to estimated escapements or hatchery/natural 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 Shoshone–Bannock Tribes Fish and Wildlife Department (SBT) into population-level spawner estimates for the Yankee Fork and Panther Creek.

PIT-tag detection-based population abundance estimates for populations above Lower Granite Dam have been generated for return years 2010–19 based on a state–space patch occupancy model (the Dam Adult Branch Occupancy Model, DABOM) that partitions the natural-origin run at large passing Lower Granite Dam into 28 population groups (IPTDSW 2020). By combining parentage-based-tagging (PBT) identification of phenotypically unmarked hatchery-origin fish, PIT-tag-based escapement data can be used to more robustly estimate natural-origin population abundance. This approach adds valuable information to the robust population estimation process that has been in place for decades based on redd and weir counts. For example, by incorporating data from sex markers and scale ageing, estimates of sex ratio (Figure 19) and age structure can be made for each of the patches in the population estimation model.

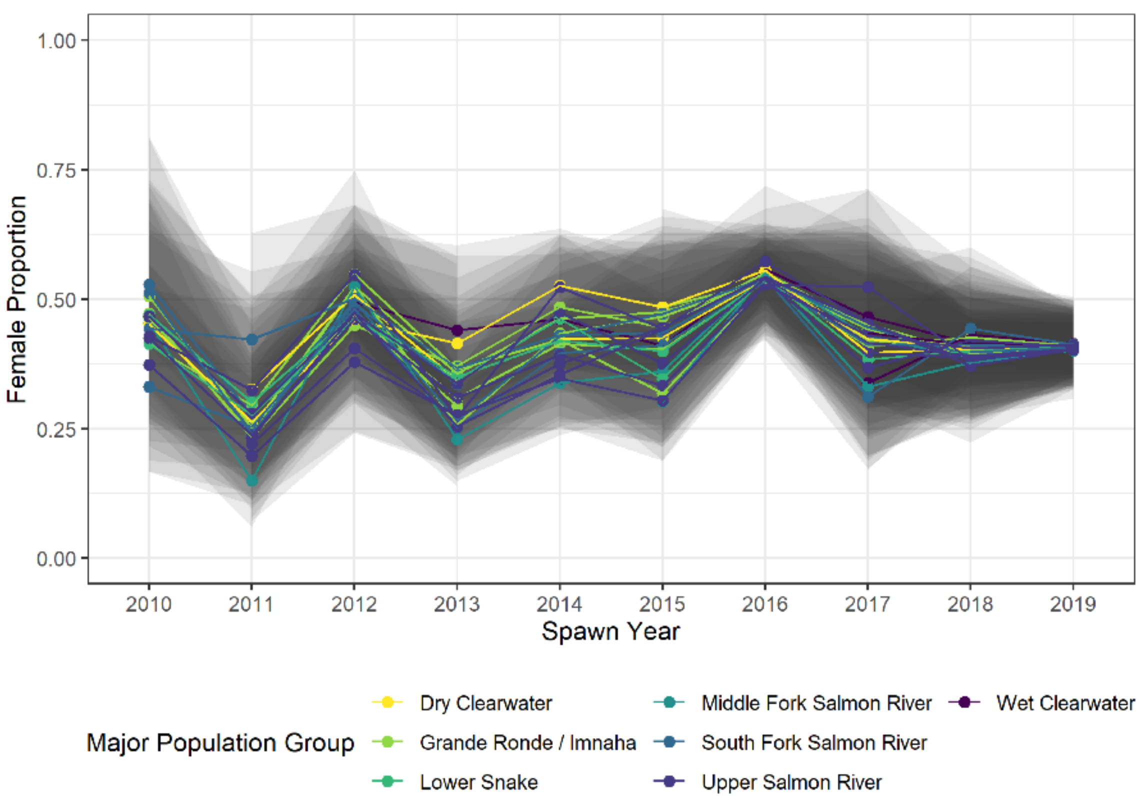

Figure 19. Natural-origin spring/summer-run Chinook salmon female proportion (95% confidence intervals shown in gray) by year and population (grouped by MPG), as estimated from the state–space patch occupancy model, DABOM. Note that the figure includes non-listed population groups from the Clearwater River basin, reproduced directly from IPTDSW 2020.

Population genetic structure

Sampling of adult Snake River spring/summer-run Chinook salmon at Lower Granite Dam, and subsequent detections of PIT tags in ICTRT population spawning areas, has allowed the development of a large genetic data set based on SNP markers (IPTDSW 2020). A neighbor-joining tree (Figure 20) created from PIT-tagged adults over the spawning years SY 2010–19, when combined with reference genetic stock identification (GSI) baseline samples across the ESU, confirms most of the expected population structure, with the notable exception of the samples from fish spawning in the Little Salmon River grouping genetically with the upper South Fork Clearwater River samples (IPTDSW 2020).

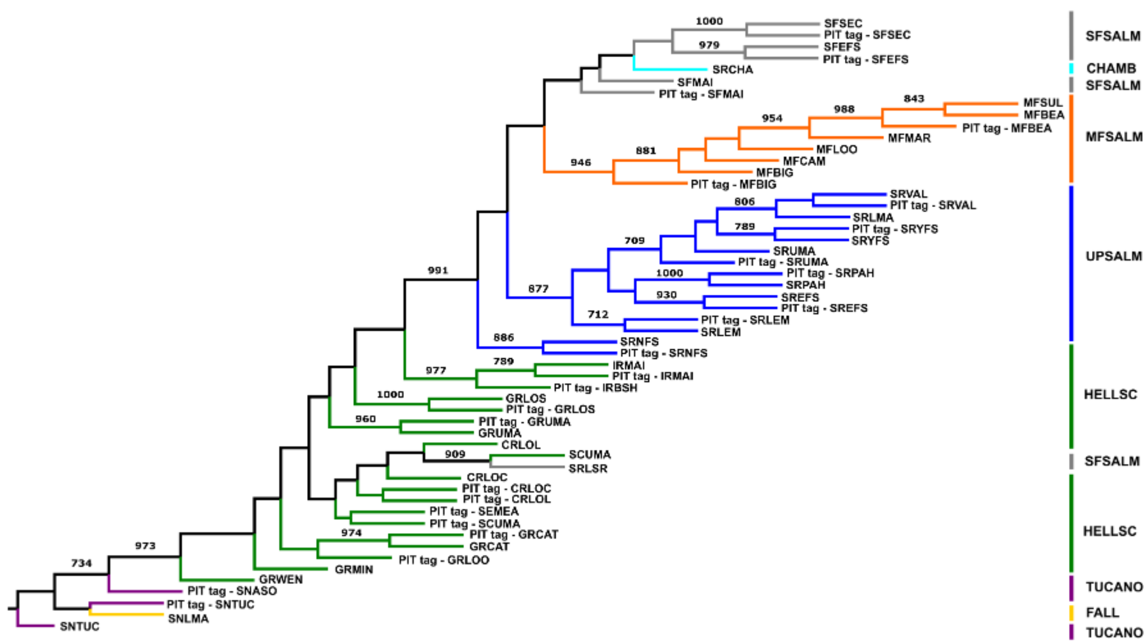

Figure 20. A neighbor-joining tree for Snake River Chinook salmon populations included in the GSI baseline (version 3.1), and collections of PIT-tagged returning adults for SY 2010–19, based on chord distance (Cavalii-Sforza and Edwards 1967). Bootstrap support greater than 70% based on 1,000 replicates is reported. Figure reproduced from IPTDSW (2020).

Spatial structure

ICTRT criteria for evaluating spatial structure within populations are based on observing evidence of spawning usage across defined spawning areas within populations, with an emphasis on historically relatively large contiguous reaches (major and minor spawning areas). Redd surveys were conducted by co-managers (NPT, SBT, ODFW, WDFW, and IDFG), and the geolocated redds were aligned with ICTRT-identified major and minor spawning areas over the 2015–19 run years (Felts et al. 2020). Monitoring occurred in 29 major spawning areas, 13 of which were rated as occupied, while monitoring of eight minor spawning areas resulted in only one being rated as occupied. However, non-occupied ratings of major and minor spawning areas do not equate to only no spawning, as an "unoccupied" rating can also result from patchy spawning not distributed across the entire reach.

Ocean condition indices

Snake River spring/summer-run Chinook salmon are a component of the Columbia River run that is believed to occupy mid-shelf waters during the early ocean life-history phase. Aggregate annual returns of Columbia River spring Chinook salmon are correlated with a range of ocean condition indices, including measures of broad scale physical conditions, local biological indicators, and local physical factors (Figure 129; Peterson et al. 2014a). Several indicators, either individually or in combination, correlate well with spring Chinook salmon adult returns with a lag of 1–2 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 Snake River spring/summer-run Chinook salmon ESU's SAR index generally corresponds to the composite rankings across ocean indicators available for early ocean years starting in the late 1990s (Peterson et al. 2019). Indicators of ocean condition are highly correlated with each other, and exhibit strong temporal autocorrelation (Figure 129; Peterson et al. 2019). As a result, when indicators point to conditions that result in poor ocean productivity for salmonid populations, they do so as a suite of indicators, and for runs of "good" or "bad" years (see Habitat chapter). Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2020). Recent (2015–19) ensemble ocean indicators rankings include four of the worst seven years in the past 20, meaning that an entire Chinook salmon generation has been subjected to poor ocean productivity conditions.

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 ESA status reviews, as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The viability review metrics were done consistently across all ESUs and DPSes to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section. 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 27 Snake River spring/summer-run Chinook salmon ESU populations are summarized in five-year increments (Table 11) and are illustrated in Figure 21. The most recent five-year geometric mean abundance estimates for 26 of the 27 populations are lower than the corresponding estimates for the previous five-year period by varying degrees; the estimate for the 27th population was a slight increase from a very low abundance in the prior five-year period. The ESU abundance data show a consistent and marked pattern of declining population size, with the recent five-year abundance levels for the 27 populations declining by an average of 55%. Medium-term (15-year) population trends in total spawner abundance were positive over the period 1990–2005 for all of the population natural-origin abundance series, and are all declining over the more recent time interval (2004–19; Table 12, Figure 21). The consistent and sharp declines for all populations in the ESU are concerning, as the abundances for some populations are approaching similar levels to those of the early 1990s when the ESU was listed.

Table 11. Five-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 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 were used to compute the geometric mean. Percent change between the 2 most-recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Tucannon River	Lower Snake	230 (314)	34 (84)	226 (398)	276 (403)	285 (422)	47 (185)	–84 (–56)
Wenaha River	Grande Ronde/Imnaha	71 (305)	164 (186)	612 (638)	354 (364)	507 (698)	383 (529)	–24 (–24)
Lostine River	Grande Ronde/Imnaha	82 (159)	105 (108)	398 (711)	340 (899)	1024 (2,807)	366 (925)	–64 (–67)
Minam River	Grande Ronde/Imnaha	110 (284)	162 (166)	541 (552)	449 (460)	684 (765)	375 (401)	–45 (–48)
Catherine Creek	Grande Ronde/Imnaha	0 (102)	59 (59)	124 (256)	71 (209)	430 (890)	85 (237)	–80 (–73)
Grande Ronde River Upper Mainstem	Grande Ronde/Imnaha	33 (96)	32 (32)	54 (103)	22 (109)	155 (906)	51 (218)	–67 (–76)
Imnaha River Mainstem	Grande Ronde/Imnaha	214 (551)	270 (536)	938 (2,142)	286 (1,308)	685 (2,055)	352 (866)	–49 (–58)
South Fork Salmon River Mainstem	South Fork Salmon River	690 (1,089)	344 (602)	968 (1,540)	628 (1,128)	913 (1,184)	160 (497)	–82 (–58)
Secesh River	South Fork Salmon River	(n/a)	187 (206)	997 (1,028)	435 (459)	1043 (1,064)	468 (489)	–55 (–54)
East Fork South Fork Salmon River	South Fork Salmon River	116 (116)	49 (50)	369 (487)	129 (308)	709 (1,147)	359 (629)	–49 (–45)
Chamberlain Creek	Middle Fork Salmon River	121 (121)	35 (35)	468 (468)	198 (198)	454 (454)	228 (228)	–50 (–50)
Middle Fork Salmon River Lower Mainstem	Middle Fork Salmon River	(n/a)	(n/a)	28 (28)	4 (4)	4 (4)	5 (5)	25 (25)
Big Creek	Middle Fork Salmon River	76 (76)	29 (29)	302 (302)	121 (121)	270 (270)	99 (99)	–63 (–63)
Camas Creek	Middle Fork Salmon River	20 (20)	13 (13)	115 (115)	43 (43)	42 (42)	42 (42)	0 (0)
Loon Creek	Middle Fork Salmon River	25 (25)	21 (21)	225 (225)	54 (54)	65 (65)	31 (31)	–52 (–52)
Middle Fork Salmon River Upper Mainstem	Middle Fork Salmon River	(n/a)	13 (13)	140 (140)	52 (52)	104 (104)	58 (58)	–44 (–44)
Sulphur Creek	Middle Fork Salmon River	59 (59)	21 (21)	55 (55)	49 (49)	112 (112)	32 (32)	–71 (–71)
Marsh Creek	Middle Fork Salmon River	102 (102)	99 (99)	285 (285)	126 (126)	563 (563)	197 (197)	–65 (–65)
Bear Valley Creek	Middle Fork Salmon River	177 (177)	95 (95)	662 (662)	305 (305)	777 (777)	236 (236)	–70 (–70)
North Fork Salmon River	Upper Salmon River	22 (22)	8 (8)	112 (112)	59 (59)	129 (129)	41 (41)	–68 (–68)
Lemhi River	Upper Salmon River	51 (51)	51 (51)	198 (198)	86 (86)	262 (262)	238 (238)	–9 (–9)
Salmon River Lower Mainstem	Upper Salmon River	63 (63)	41 (41)	239 (239)	99 (99)	137 (137)	37 (37)	–73 (–73)
Pahsimeroi River	Upper Salmon River	22 (73)	45 (73)	173 (343)	209 (275)	360 (387)	132 (283)	–63 (–27)
East Fork Salmon River	Upper Salmon River	69 (108)	34 (46)	442 (442)	224 (224)	602 (602)	138 (138)	–77 (–77)
Yankee Fork	Upper Salmon River	16 (16)	6 (6)	60 (60)	25 (120)	169 (623)	22 (24)	–87 (–96)
Salmon River Upper Mainstem	Upper Salmon River	227 (275)	68 (86)	671 (1,100)	326 (566)	628 (898)	170 (509)	–73 (–43)
Valley Creek	Upper Salmon River	26 (26)	26 (26)	109 (109)	85 (85)	192 (192)	67 (67)	–65 (–65)

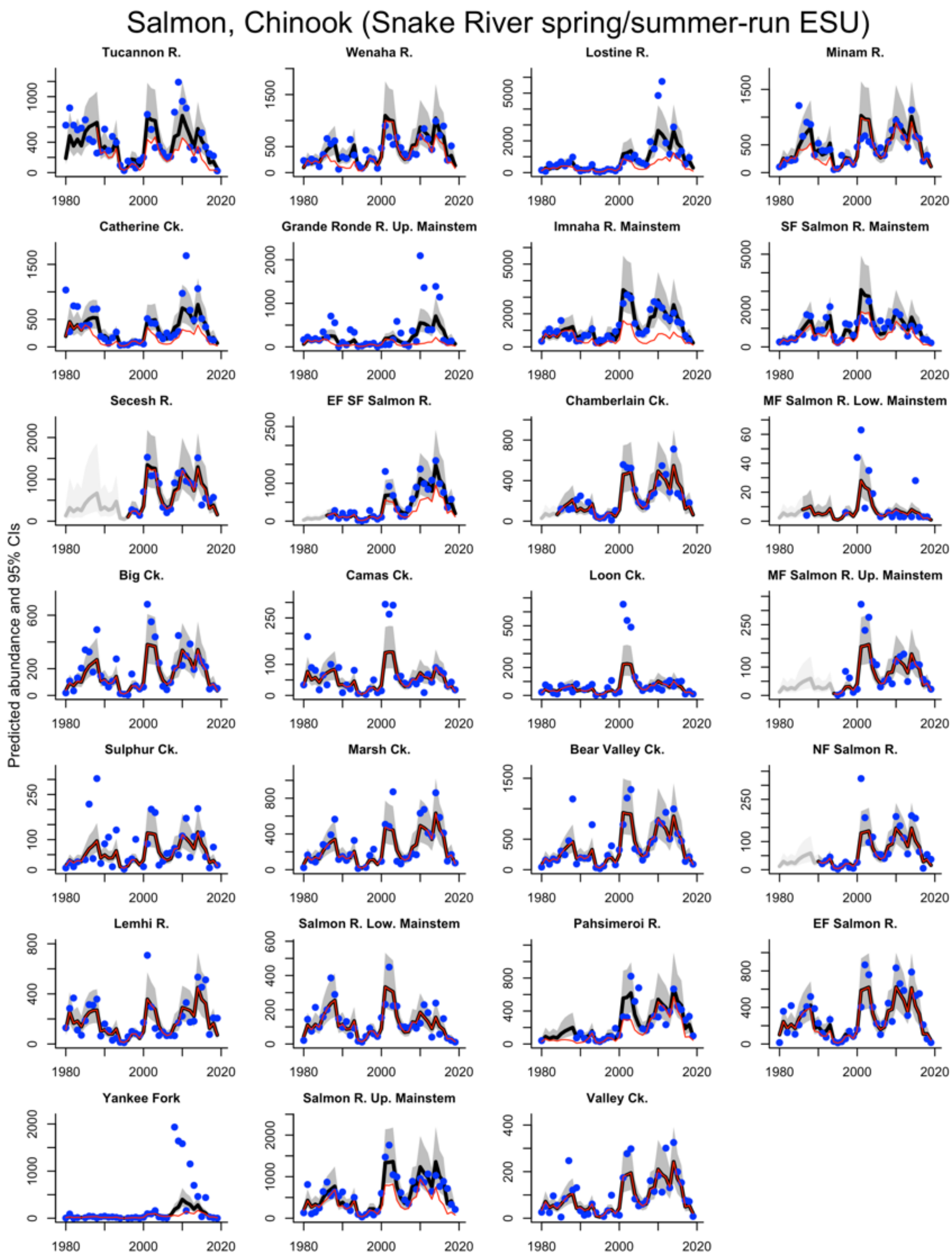

Figure 21. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

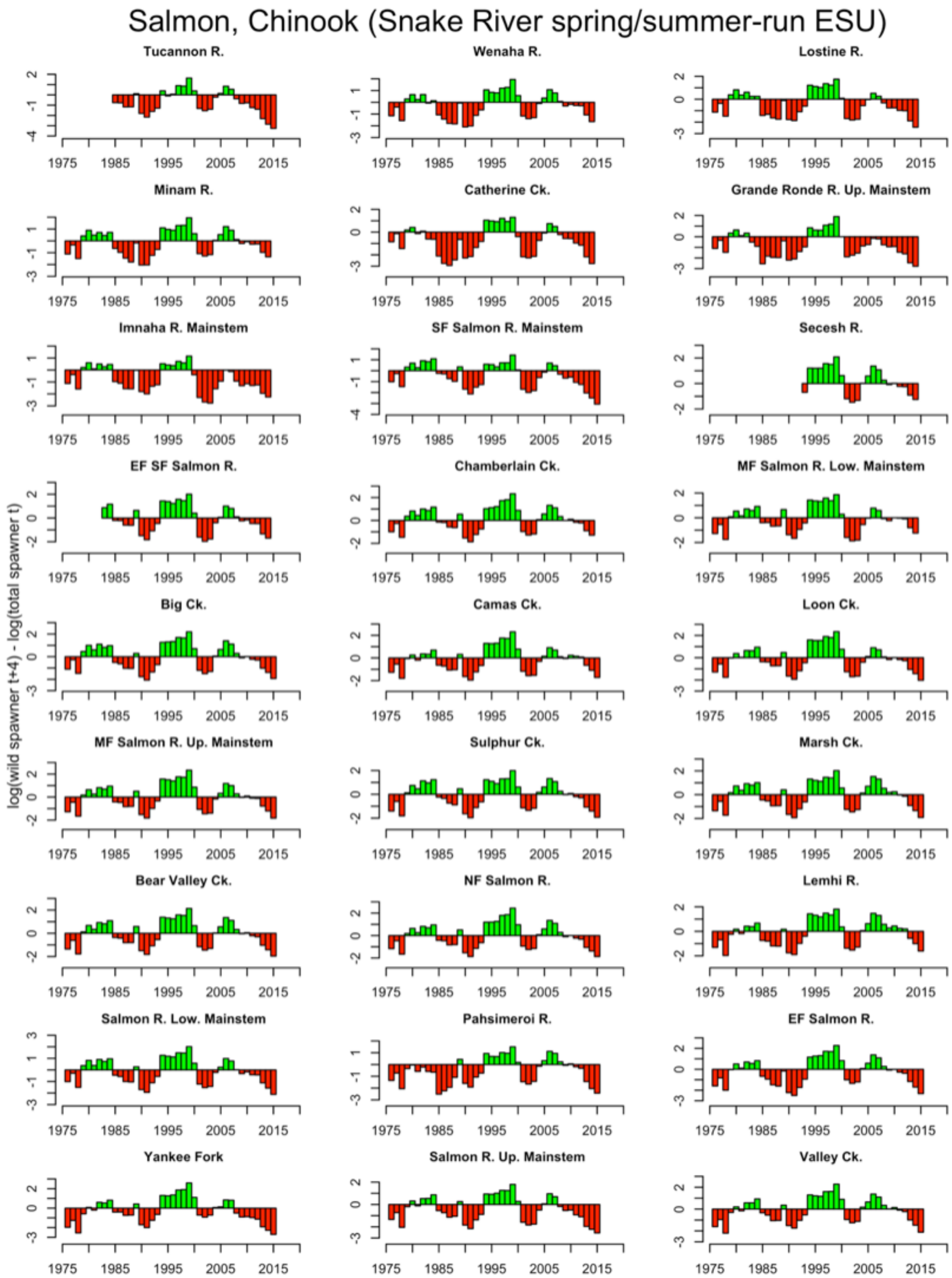

Figure 22. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year $(t-4)$. Spawning years on x-axis.

Table 12. Fifteen-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 wild spawner estimates and with at least 2 data points (actual data, not estimates) in the first 5 years and last 5 years of the 15-year ranges are shown.

Population	**MPG**	**1990–2005**	**2004–19**
Tucannon River	Lower Snake	0.04 (–0.07, 0.15)	–0.13 (–0.23, –0.04)
Wenaha River	Grande Ronde/Imnaha	0.17 (0.08, 0.26)	–0.04 (–0.11, 0.02)
Lostine River	Grande Ronde/Imnaha	0.12 (0.03, 0.21)	0.00 (–0.09, 0.09)
Minam River	Grande Ronde/Imnaha	0.12 (0.03, 0.21)	–0.03 (–0.10, 0.03)
Catherine Creek	Grande Ronde/Imnaha	0.11 (0.03, 0.20)	–0.01 (–0.12, 0.10)
Grande Ronde River Upper Mainstem	Grande Ronde/Imnaha	0.08 (–0.01, 0.17)	0.01 (–0.06, 0.09)
Imnaha River Mainstem	Grande Ronde/Imnaha	0.10 (0.01, 0.19)	–0.02 (–0.09, 0.05)
South Fork Salmon River Mainstem	South Fork Salmon River	0.07 (–0.03, 0.16)	–0.12 (–0.21, –0.03)
Secesh River	South Fork Salmon River	—	–0.02 (–0.09, 0.05)
East Fork South Fork Salmon River	South Fork Salmon River	0.11 (0.00, 0.21)	0.05 (–0.03, 0.13)
Chamberlain Creek	Middle Fork Salmon River	0.11 (0.00, 0.21)	–0.02 (–0.09, 0.05)
Middle Fork Salmon River Lower Mainstem	Middle Fork Salmon River	—	–0.08 (–0.14, –0.02)
Big Creek	Middle Fork Salmon River	0.10 (–0.01, 0.21)	–0.03 (–0.11, 0.04)
Camas Creek	Middle Fork Salmon River	0.11 (0.00, 0.22)	–0.03 (–0.09, 0.03)
Loon Creek	Middle Fork Salmon River	0.14 (0.03, 0.25)	–0.08 (–0.14, –0.01)
Middle Fork Salmon River Upper Mainstem	Middle Fork Salmon River	—	–0.03 (–0.10, 0.04)
Sulphur Creek	Middle Fork Salmon River	0.07 (–0.03, 0.17)	–0.03 (–0.11, 0.04)
Marsh Creek	Middle Fork Salmon River	0.08 (–0.02, 0.19)	0.01 (–0.07, 0.09)
Bear Valley Creek	Middle Fork Salmon River	0.11 (0.01, 0.21)	–0.03 (–0.11, 0.04)
North Fork Salmon River	Upper Salmon River	0.13 (0.02, 0.24)	–0.03 (–0.11, 0.04)
Lemhi River	Upper Salmon River	0.09 (–0.02, 0.19)	0.04 (–0.03, 0.11)
Salmon River Lower Mainstem	Upper Salmon River	0.08 (–0.02, 0.19)	–0.09 (–0.16, –0.02)
Pahsimeroi River	Upper Salmon River	0.16 (0.07, 0.25)	–0.02 (–0.12, 0.07)
East Fork Salmon River	Upper Salmon River	0.14 (0.03, 0.26)	–0.05 (–0.14, 0.03)
Yankee Fork	Upper Salmon River	0.14 (0.02, 0.25)	–0.02 (–0.11, 0.06)
Salmon River Upper Mainstem	Upper Salmon River	0.08 (–0.02, 0.18)	–0.06 (–0.14, 0.02)
Valley Creek	Upper Salmon River	0.12 (0.01, 0.22)	–0.03 (–0.10, 0.05)

Smolt-to-adult return survival estimates (SARs) are generated by the Columbia River Data Access in Real Time (CBR and UW 2020) project using PIT-tag detections from all release locations within each population basin (CBR and UW 2020). The SAR indices represent cumulative marine, nearshore, and estuary survival. Figure 23 shows the geometric mean of R/S and SAR indices for the stocks available across five MPGs in the ESU. In general, these broad-brush descriptors of population processes indicate relatively poor ocean survival for the Salmon River MPGs and relatively poor freshwater productivity for the Grande Ronde/Imnaha and Lower Snake MPGs. Using the R/S and SAR indicators by population, it is possible to generate an indicator of freshwater productivity (FWPI) as a ratio of R/S and SAR. This quantity can be thought of as an indicator of smolts per spawner, and thus, the overall population productivity in the freshwater environment. An FWPI score of >100 should indicate healthy freshwater productivity (roughly 100 smolts per female). The initial assessment by the ICTRT (2007) identified significant abundance/productivity gaps for this ESU. In general, populations within the Grande Ronde/Imnaha and Lower Snake MPGs are still showing the lowest productivity.

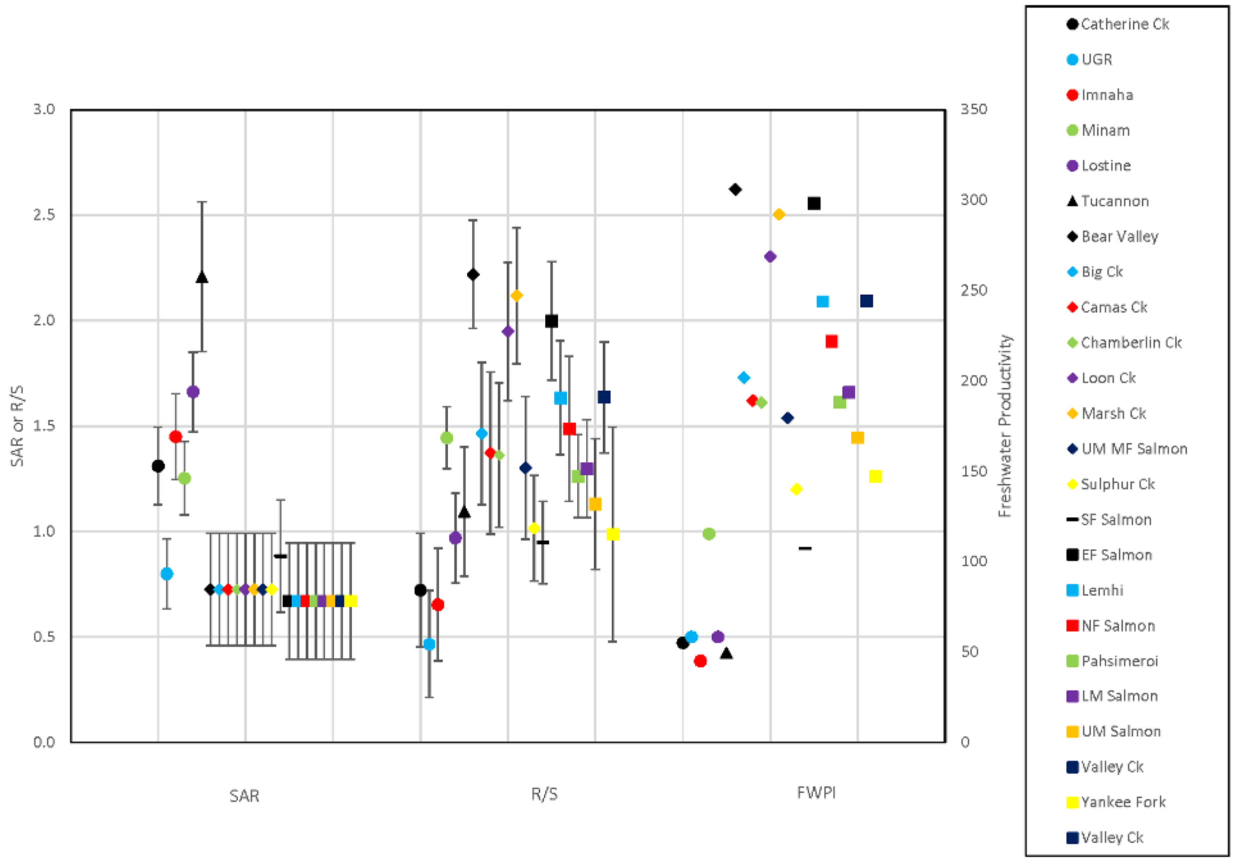

Figure 23. Smolt-to-adult return, recruits per spawner, and freshwater productivity index (FWPI) for each of the populations in the ESU. Geometric means of SAR and R/S are shown for each population, along with the standard error of the estimate (whiskers represent ±1 SE). The time period included in the SAR or R/S indices is the past 20 years, depending on data availability. The FWPI is constructed as a ratio of the geomean R/S and SAR, and can be thought of as a measure of smolts per spawner.

Non-treaty harvest

Harvest impacts on the spring component of this ESU are essentially the same as those on the Upper Columbia River (Figure 24). 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 occurs in the Snake River basin on specific populations within the ESU. Snake River summer Chinook salmon share the ocean distribution patterns of the upper basin spring runs and are only subject to significant harvest in the mainstem Columbia River. Harvest of summer Chinook salmon 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-run Chinook salmon ESU.

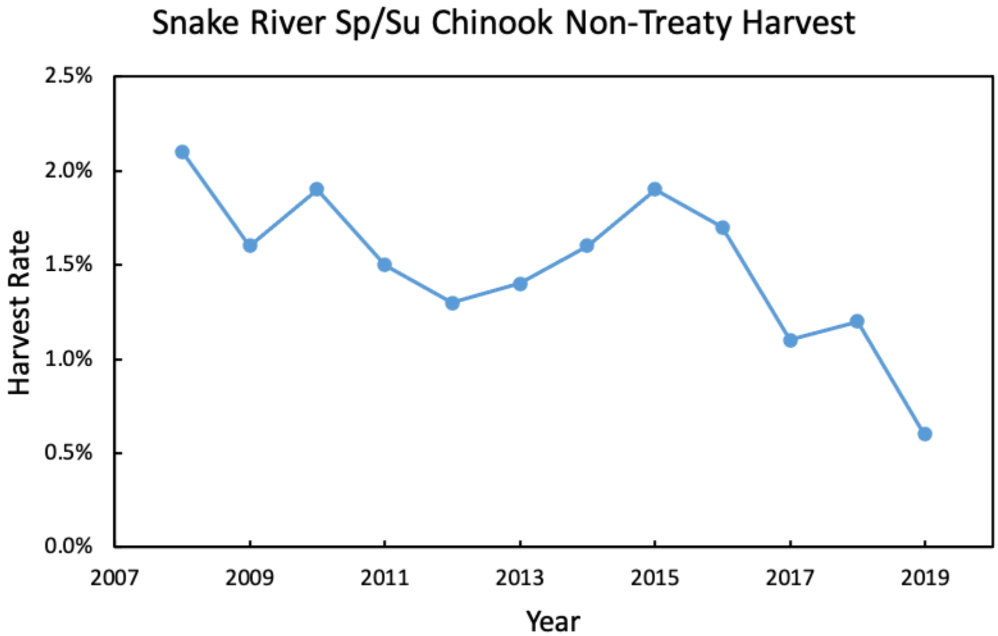

Figure 24. Non-treaty exploitation rates for Snake River spring/summer-run Chinook salmon in the mainstem Columbia River fisheries. Data from the Columbia River Technical Advisory Team (TAC 2015, 2020).

Spatial structure and diversity

Current estimates of spatial structure and diversity ratings for Snake River spring/summer-run Chinook salmon populations are summarized in Table 14. 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 25, Table 13). 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 broodstock 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. 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 and has an ongoing integrated hatchery program that incorporates natural-origin broodstock.

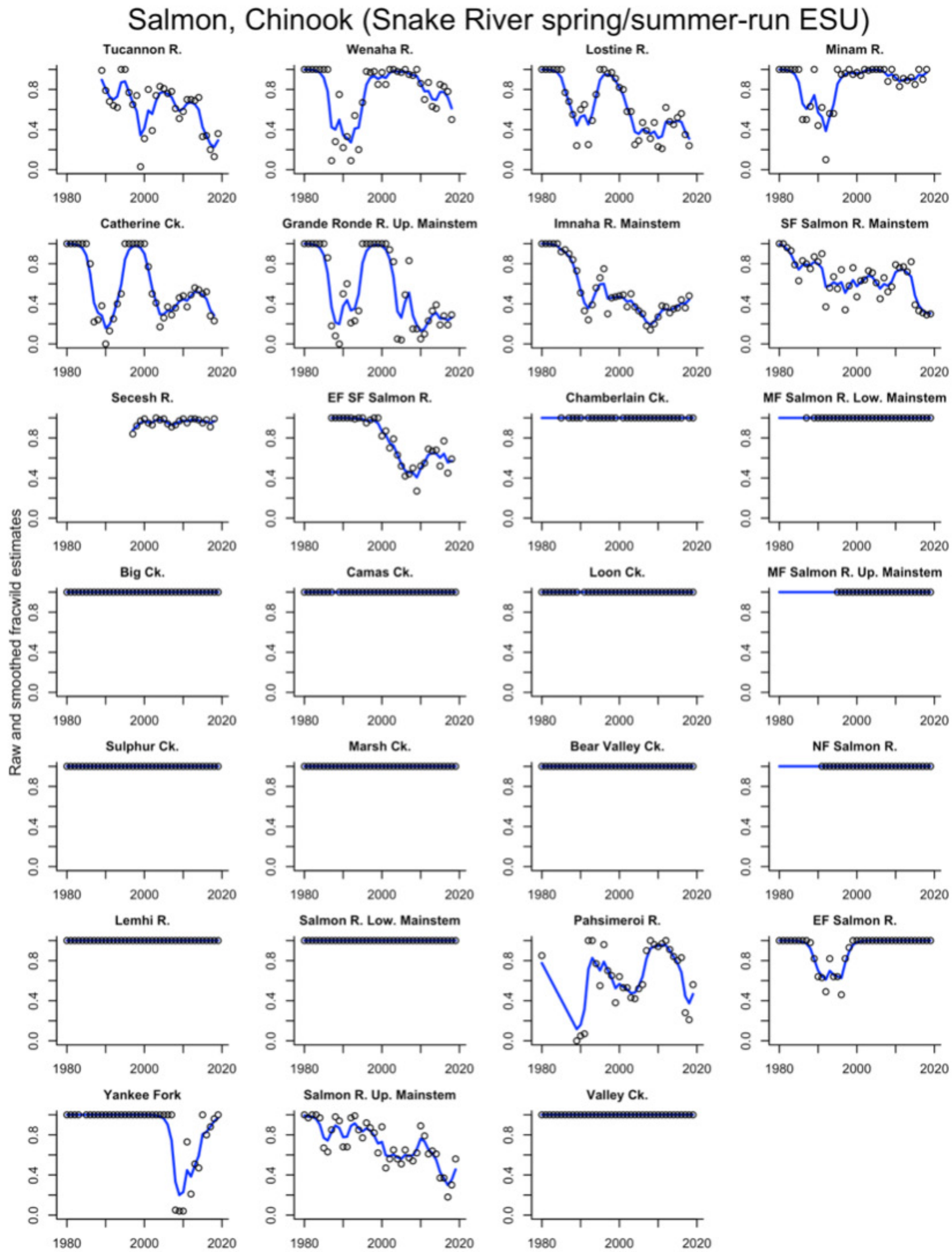

Figure 25. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 13. Five-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates).

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Tucannon River	Lower Snake	0.64	0.61	0.69	0.68	0.27
Wenaha River	Grande Ronde/Imnaha	0.89	0.96	0.97	0.73	0.74
Lostine River	Grande Ronde/Imnaha	0.97	0.61	0.39	0.40	0.42
Minam River	Grande Ronde/Imnaha	0.97	0.98	0.98	0.89	0.94
Catherine Creek	Grande Ronde/Imnaha	1.00	0.57	0.35	0.49	0.38
Grande Ronde River Upper Mainstem	Grande Ronde/Imnaha	1.00	0.76	0.33	0.22	0.24
Imnaha River Mainstem	Grande Ronde/Imnaha	0.53	0.44	0.23	0.34	0.41
South Fork Salmon River Mainstem	South Fork Salmon River	0.59	0.64	0.56	0.77	0.32
Secesh River	South Fork Salmon River	0.91	0.97	0.95	0.98	0.96
East Fork South Fork Salmon River	South Fork Salmon River	0.99	0.76	0.43	0.62	0.58
Chamberlain Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Middle Fork Salmon River Lower Mainstem	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Big Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Camas Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Loon Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Middle Fork Salmon River Upper Mainstem	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Sulphur Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Marsh Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
Bear Valley Creek	Middle Fork Salmon River	1.00	1.00	1.00	1.00	1.00
North Fork Salmon River	Upper Salmon River	1.00	1.00	1.00	1.00	1.00
Lemhi River	Upper Salmon River	1.00	1.00	1.00	1.00	1.00
Salmon River Lower Mainstem	Upper Salmon River	1.00	1.00	1.00	1.00	1.00
Pahsimeroi River	Upper Salmon River	0.65	0.51	0.79	0.93	0.54
East Fork Salmon River	Upper Salmon River	0.77	1.00	1.00	1.00	1.00
Yankee Fork	Upper Salmon River	1.00	1.00	0.52	0.39	0.93
Salmon River Upper Mainstem	Upper Salmon River	0.80	0.62	0.58	0.71	0.36
Valley Creek	Upper Salmon River	1.00	1.00	1.00	1.00	1.00

The single current extant population in the Lower Snake 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 and Dedloff 2015). Hatchery proportions for populations in the Middle Fork Salmon River 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 River MPG populations have ongoing hatchery programs. Hatchery proportions for 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. Integrated hatchery programs are now being implemented in parallel to ongoing production (segregated) operations in the South Fork and East Fork of the South Fork Salmon River facilities. 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. Both of the hatchery facilities in this MPG are operating parallel integrated and segregated programs. 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 populations are regulated by managing passage at Sawtooth weir, located on the mainstem Salmon River near the downstream extent of spawning. Releases of any-origin fish (integrated/segregated) have occurred above the weirs at both the Upper Salmon River and Pahsimeroi facilities to meet escapement goals due to recent low returns. Clearly, the proportion of hatchery-origin spawners (pHOS) in these populations will be impacted, but operating agreement balances the risks associated with introgression with depensation at very low run sizes. Hatchery proportions within the Yankee Fork population have increased substantially in recent years, reflecting returns from a large-scale supplementation effort conducted by SBT. In some recent years, the program has augmented ongoing smolt releases with adult plants in the Yankee Fork and egg boxes in Panther Creek, when there are surplus returns from the Sawtooth Hatchery program in the Upper Salmon River (Gregory and Wood 2013, Denny and Blackadar 2015). Recent efforts to evaluate the origin of the Panther Creek spawning population have shown a mixture of potentially long-term occupants (close to the Middle Fork Snake River) and clear hatchery-origin stocks (South Fork Snake River, Rapid River, Upper Salmon River, and Pahsimeroi River).

Biological viability relative to recovery goals

The ICTRT identified 27 extant and four extirpated populations of Snake River spring/summer-run Chinook salmon that historically used the accessible tributary and upper mainstem habitats within the Snake River drainages (ICTRT 2003). The populations are aggregated into five extant 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 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 Salmon River 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. In 2017, NOAA Fisheries completed a recovery plan for Snake River spring/summer-run Chinook salmon and Snake River Basin steelhead (NMFS 2017c).

Recovery plan criteria

The recovery criteria are hierarchical in nature, with ESU/DPS-level criteria being based on the viability of natural-origin Chinook salmon assessed at the population level. The population-level assessments are based on a set of metrics designed to evaluate risk across the four VSP elements: abundance, productivity, spatial structure, and diversity (McElhany et al. 2000). The recovery plans adopt the ICTRT approach for comparing estimates of current natural-origin abundance (measured as a ten-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. The recovery plans also apply the ICTRT criteria (metrics and example risk thresholds) for assessing the spatial structure and diversity risks based on current information representing each specific population.

The ICTRT recommended 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. Recovery plans use the MPG scenarios and also suggest that at least one population in a viable MPG should meet criteria for "highly viable." Within any particular MPG, there may be several specific combinations of populations that could satisfy these criteria. The recovery plans outline example scenarios that would satisfy the criteria for all extant MPGs. In each case, the remaining populations in an MPG should be at or above "maintained" status.

Lower Snake MPG

This MPG historically contained two populations, and one, Asotin Creek, is currently considered extirpated. The recovery plan basic criteria would call for both populations being restored to viable status. The recovery plan recommends the priority of restoring the Tucannon River to highly viable status, and then evaluating the potential for reintroducing production in Asotin Creek as recovery progresses.

Grande Ronde/Imnaha MPG

This MPG had eight historical populations, two of which are currently considered functionally extirpated. The basic recovery plan criteria call for a minimum of four populations at viable or highly viable status. The potential scenario would include viable populations in the Imnaha River (run timing), the Lostine/Wallowa Rivers (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 Salmon River MPG

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

Middle Fork Salmon River MPG

The recovery plan 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 base example recovery scenario includes Chamberlain Creek (geographic position), Big Creek (large size category), Bear Valley Creek, Marsh Creek, and either Loon Creek or Camas Creek.

Upper Salmon River MPG

This MPG included nine historical populations, one of which, Panther Creek, is considered functionally extirpated. The base example recovery scenario for this MPG includes the Pahsimeroi River (summer-run Chinook salmon life history); the Lemhi River and Upper Salmon Mainstem (very large size category); East Fork Salmon River (large size category); and Valley Creek. The continued and building presence of a spawning population in Panther Creek argues for its role in recovery scenarios to be reconsidered.

Updated biological risk summary

The majority of populations in the Snake River spring/summer-run Chinook salmon ESU remained at high overall risk, with three populations (Minam River, Bear Valley, and Marsh Creek) improving to an overall rating of "maintained" due to an increase in abundance/productivity when measured over a 10–20 year period (Table 14). However, natural-origin abundance has generally decreased over the levels reported in the prior review for most populations in this ESU, in many cases sharply. Relatively low ocean survivals in recent years are likely a major factor in recent abundance patterns. All but three populations in this ESU remained at high risk for abundance and productivity.

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 Grande Ronde River Upper Mainstem, Lemhi River, and Middle Fork Salmon River Lower 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, but it appears likely that these programs reduce risk of extinction in the short term.

In summary, while there have been improvements in abundance/productivity in several populations relative to the time of listing, the majority of populations experienced sharp declines in abundance in the recent five-year period, primarily due to variation in ocean survival. If ocean survival rates remain low, the ESU's viability will clearly become much more tenuous. If survivals improve in the near term, however, it is likely the populations could rebound quickly. Overall, at this time we conclude that the Snake River spring/summer-run Chinook salmon ESU continues to be at moderate-to-high risk.

Table 14. Snake River spring/summer-run Chinook salmon population status relative to ICTRT viability criteria, grouped by MPG. Natural spawning abundance: most recent 10-yr geometric mean (range). ICTRT productivity: 20-yr 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. Populations with no abundance and productivity data are given a default High A/P Risk rating.

	Abundance/productivity (A/P) metrics				Spatial structure/diversity (SS/D) metrics			
Population	**ICTRT threshold**	**Natural spawning**	**ICTRT productivity**	**Integrated A/P risk**	**Natural processes**	**Diversity risk**	**Integrated SS/D risk**	**Overall risk rating**
Lower Snake MPG								
Tucannon River	750	116 (SD 205)	1.09 (0.31, 17/20)	High	Low	Moderate	Moderate	High
Grande Ronde/Imnaha MPG								
Wenaha River	750	437 (SD 191)	1.21 (0.16, 15/20)	High	Low	Moderate	Moderate	High
Lostine River	1,000	654 (SD 400)	0.97 (0.21, 18/20)	High	Low	Moderate	Moderate	High
Minam River	750	544 (SD 256)	1.44 (0.15, 15/20)	Moderate	Low	Moderate	Moderate	Maintained
Catherine Creek	1,000	200 (SD 207)	0.76 (0.27, 20/20)	High	Moderate	Moderate	Moderate	High
Grande Ronde River Upper Mainstem	1,000	80 (SD 157)	0.47 (0.25, 20/20)	High	High	Moderate	High	High
Imnaha River Mainstem	750	513 (SD 214)	0.65 (0.27, 14/20)	High	Low	Moderate	Moderate	High
South Fork Salmon River MPG								
South Fork Salmon River Mainstem	1,000	381 (SD 514)	0.96 (0.20, 12/20)	High	Low	Moderate	Moderate	High
Secesh River	750	472 (SD 396)	—	High	Low	Low	Low	High
East Fork South Fork Salmon River	1,000	483 (SD 265)	—	High	Low	Low	Low	High
Little Salmon River	750	*Insufficient data*	—	—	Low	Low	Low	High
Middle Fork Salmon River MPG								
Chamberlain Creek	750	342 (SD 171)	1.36 (0.34, 17/20)	High	Low	Low	Low	High
Middle Fork Salmon River Lower Mainstem	1,000	163 (SD 114)	1.47 (0.34, 20/20)	High	Very Low	Moderate	Moderate	High
Big Creek	500	45 (SD 37)	1.95 (0.33, 13/20)	High	Low	Moderate	Moderate	High
CamasCreek	500	42 (SD 27)	1.37 (0.42, 17/20)	High	Low	Moderate	Moderate	High
LoonCreek	500	*Insufficient data*	*Insufficient data*	—	Moderate	Moderate	Moderate	High
Middle Fork Salmon River Upper Mainstem	750	71 (SD 43)	1.30 (0.34, 17/20)	High	Low	Moderate	Moderate	High
Sulphur Creek	500	67 (SD 65)	1.02 (0.25, 13/20)	High	Low	Moderate	Moderate	High
Marsh Creek	500	333 (SD 262)	2.11 (0.32, 7/20)	Moderate	Low	Low	Low	Maintained
Bear Valley Creek	750	428 (SD 327)	2.22 (0.26, 13/20)	Moderate	Very Low	Low	Low	Maintained

Table 14 (continued). Snake River spring/summer-run Chinook salmon population status relative to ICTRT viability criteria, grouped by MPG.

Population	Abundance/productivity (A/P) metrics				Spatial structure/diversity (SS/D) metrics			
	ICTRT threshold	Natural spawning	ICTRT productivity	Integrated A/P risk	Natural processes	Diversity risk	Integrated SS/D risk	Overall risk rating
Upper Salmon River MPG								
North Fork Salmon River	2,000	71 (SD 87)	1.30 (0.23, 20/20)	High	Low	Low	Low	High
Lemhi River	1,000	326 (SD 270)	1.13 (0.31, 18/20)	High	Low	Low	Low	High
Salmon River Lower Mainstem	1,000	218 (SD 168)	1.26 (0.20 20/20)	High	Moderate	High	High	High
Pahsimeroi River	2,000	250 (SD 159)	1.63 (0.28, 19/20)	High	High	High	High	High
East Fork Salmon River	500	113 (SD 100)	1.63 (0.26, 17/20)	High	Low	Moderate	Moderate	High
Yankee Fork	1,000	288 (SD 291)	2.00 (0.28, 17/20)	High	Low	High	high	High
Salmon River Upper Mainstem	500	62 (SD 139)	0.99 (0.51, 17/20)	High	Moderate	High	High	High
Valley Creek	500	*Insufficient data*	*Insufficient data*	—	Low	Low	Low	High
Panther Creek	750	*Insufficient data*	*Insufficient data*	—	—	—	—	*See text*

Snake River Fall-run Chinook Salmon ESU

Brief description of ESU

This ESU includes naturally spawned fall-run Chinook salmon originating from the mainstem Snake River below Hells Canyon Dam and from the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River sub-basins (Figure 26). It also includes fall-run Chinook salmon from the following artificial propagation programs: the Lyons Ferry Fish Hatchery Program, the Fall Chinook Acclimation Project, the Nez Perce Tribal Hatchery Program, and the Idaho Power hatchery program (USOFR 2005a). Fish passage is blocked at Hells Canyon Dam (RM 247), the lowest of three impassable dams that form the Hells Canyon Complex. Historically, natural production from this ESU was mainly from spawning in the mainstem of the Snake River upstream of the Hells Canyon 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). There was a single historical population (the Middle Snake River 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 (Connor et al. 2019). 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. The ICTRT identified five major spawning areas (MaSAs) within the currently available Lower Snake River population: Upper Hells Canyon MaSA (Hells Canyon Dam on Snake River downstream to confluence with Salmon River); Lower Hells Canyon MaSA (Snake River from Salmon River confluence downstream to Lower Granite Dam pool); Clearwater River MaSA; Grande Ronde River MaSA; and Tucannon River MaSA. A major spawning area is defined as a system of one or more branches containing sufficient habitat to support at least 500 spawners.

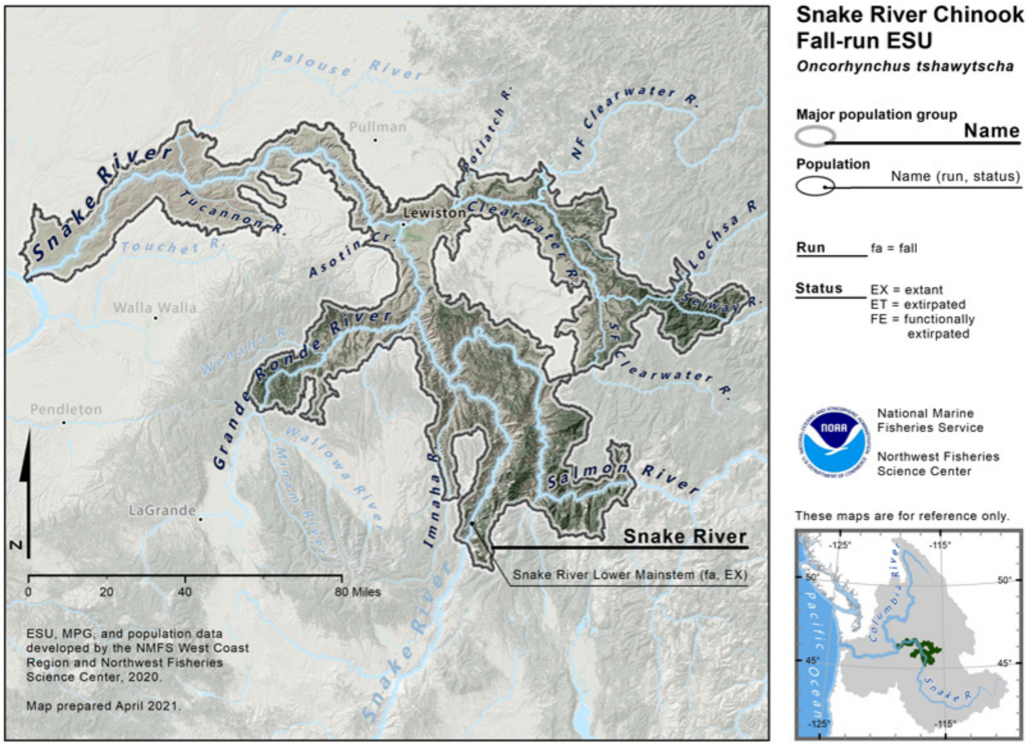

Figure 26. Map of the Snake River fall-run Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

Summary of previous viability conclusions

2005

The 2005 BRT review (Good et al. 2005) included an assessment of Snake River fall-run 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…" (p. 163). 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-run 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 by 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-run Chinook salmon 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-run 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-run Chinook salmon was the only one remaining from an historical ESU that also included large mainstem populations upstream of the current location of the Hells Canyon Complex. The increases in natural-origin abundance were encouraging and largely the result of the supplementation program initiated in 1998. 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.

2015

NWFSC (2015) concluded that the status of Snake River fall-run Chinook salmon had clearly improved compared to the time of listing and compared to prior status reviews. The single extant population in the ESU was meeting the criteria for a rating of "viable" developed by the ICTRT, but the ESU as a whole was not meeting the recovery goals described in the recovery plan for the species, which requires either the single population to be "highly viable with high certainty" and/or the establishment of a second viable population above the Hells Canyon Complex (NMFS 2017b).

Description of new data available for this review

Spawner abundance, productivity, and proportion natural-origin estimates for the Lower Snake River population are based on counts and sampling of adult and jack (<57 cm) fall-run Chinook salmon at Lower Granite Dam (Young et al. 2020). 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-run Chinook salmon are used to define a sampling strategy across the duration of the run that will also achieve hatchery broodstock objectives 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.

Estimates of natural-origin returns are made by subtracting estimated hatchery-origin returns from the total run estimates using hatchery marks and tags and comparative PBT (Young et al. 2020). Since 2015, all fall-run Chinook salmon in the Snake River basin that were spawned in a hatchery were genetically analyzed for subsequent offspring origin analysis. The PBT program allows for a comprehensive assessment of hatchery contributions and, therefore, a more direct assessment of natural returns, including spawner composition on spawning grounds. Hatchery-origin returns were also monitored for juvenile release site fidelity for spawning through radio tracking to evaluate the impacts of supplementation on the natural population (Cleary et al. 2018).

Redd surveys have been conducted annually since 1991. Shallow-water (<3-m) redds are surveyed by small unmanned aircraft systems, when allowed, based on statistical sampling developed by Groves et al. (2016) and Arnsberg et al. (2020b). Deepwater redds are surveyed by underwater camera (Tiffan and Perry 2020).

Sampling methods and statistical procedures used in generating the estimated escapements have improved substantially over the past 15 to 20 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). 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 WDFW Snake River laboratory (Milks and Oakerman 2018).

The U.S. Geological Survey (USGS) Western Fisheries Research Center has developed a two-stage state-space life-cycle model for naturally produced fall Chinook salmon in the Snake River basin (Tiffan and Perry 2020). The model has been used to assess proposed actions for the Columbia River systems biological opinion (NMFS 2020).

Some upper Hells Canyon hatchery releases have been moved to the Salmon River. Fish are beginning to return and any changes in natural production or the pHOS in the Upper Hells Canyon MaSA will require at least one full brood cycle.

The recovery plan was completed in 2017 and outlines the following three potential recovery scenarios:

A. Achieve “highly viable” status for the extant Lower Snake River population and “viable” status for the currently extirpated Middle Snake River population.
B. Achieve “highly viable” status for the Lower Snake River population.
C. Achieve “highly viable” status for the Lower Snake River population with the creation of a Natural Production Emphasis Area (NPEA).

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 adult abundance and productivity at the population level. Evaluations were done using both a set of metrics corresponding to those used in prior BRT reviews, as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU (ICTRT 2007). The relatively simple BRT-level metrics were done consistently across all ESUs and DPSes to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the TRT and Recovery Plan Criteria section. 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 for this assessment are available through NWFSC's Salmon Population Summary database.

Prior to the early 1980s, returns of Snake River fall-run Chinook salmon were likely predominately natural-origin (Bugert et al. 1995). Natural-origin return levels declined substantially following the completion of the three-dam Hells Canyon Complex (1959–67), which completely blocked access to major production areas above Hells Canyon Dam, and the construction of the lower Snake River dams (1962–75). Based on extrapolations from sampling at Ice Harbor Dam (1977–90), the Lyons Ferry Hatchery (1987–present), and at Lower Granite Dam (1990–present), hatchery strays made up an increasing proportion of returns at Lower Granite Dam (the uppermost Snake River mainstem dam) through the 1980s (Bugert and Hopley 1989, Bugert et al. 1990). Strays from out-planting Priest Rapids hatchery-origin fall-run Chinook salmon (an out-of-ESU stock from the mid-Columbia River) and Snake River fall-run 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. The initiation of the supplementation program in 1998 increased returns allowed to naturally spawn.

Since supplementation returns began, naturally spawning fall-run Chinook salmon in the lower Snake River have included both returns originating from naturally spawning parents and from returning hatchery releases. Hatchery-origin fall-run Chinook salmon escaping upstream above Lower Granite Dam to spawn naturally are now predominantly returns from Idaho Power Company, Nez Perce Tribal Hatchery, and Fall Chinook Acclimation Project supplementation program juvenile releases in reaches above Lower Granite Dam and from releases at Lyons Ferry Hatchery that have dispersed upstream. These fish are part of the listed ESU.

Supplementation and other measures since listing led to large increases in natural-origin returns, gradually at first and then, in 2013, adult spawner abundance reached over 20,000 fish (Figure 27, Table 15). From 2012–15, natural-origin returns were over 10,000 adults. Spawner abundance has declined since 2016 to 4,998 adult natural-origin spawners in 2019 (Figure 27). In 2018, natural-origin spawner abundance was 4,916, a quarter of the return in 2013. This appears as a high negative percent change in the five-year geometric mean (Table 15), but, when looking at the trend in longer time frames, across more than one brood cycle, it shows an increase in the ten-year geometric mean relative to the last status

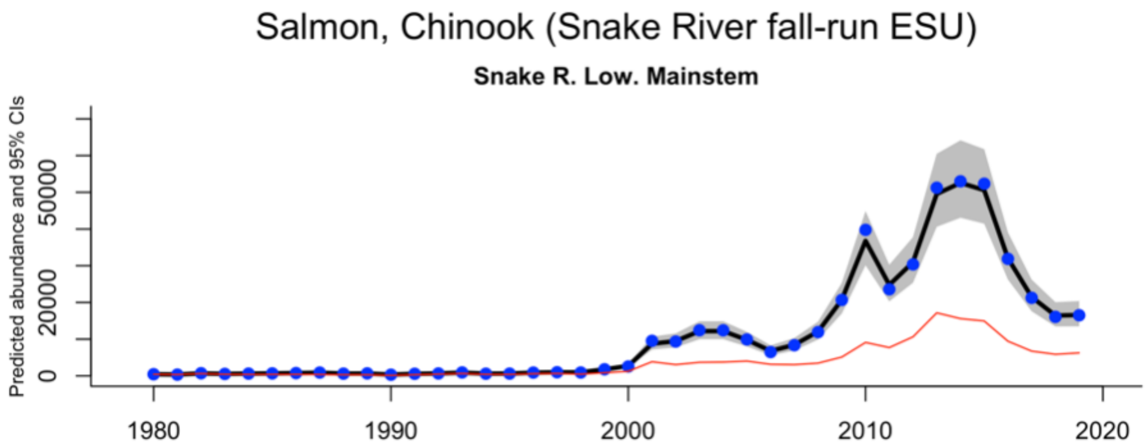

Figure 27. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

Table 15. Five-year geometric mean of raw natural spawner counts. This is the raw total spawner count times the fraction natural estimate, if available. In parentheses is the 5-year geometric mean of raw total spawner counts, 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 2 most recent 5-year periods is shown on the far right.

Population	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Lower Snake River FA	333 (581)	548 (980)	3,014 (8,398)	3,645 (10,581)	11,254 (37,812)	7,252 (22,141)	−36 (−41)

Table 16. Fifteen-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 1980–2014 and with at least 2 data points in the first 5 years and last 5 years of the 15-year period are shown.

Population	1990–2005	2004–2019
Lower Snake River FA	0.22 (0.17, 0.27)	0.05 (−0.01, 0.11)

review, and a near-zero population change for the 15-year trend in abundance (Table 16). The geometric mean natural adult abundance for the most recent ten years (2010–19) is 9,034 (0.15 SE), higher than the ten-year geomean reported in the most recent status review (6,418, 0.19 SE, 2005–14; NWFSC 2015). While the population has not been able to maintain the higher returns it achieved in 2010 and 2013–15, it has maintained at or above the ICTRT defined Minimum Abundance Threshold (3,000) during climate challenges in the ocean and rivers.

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 (ICTRT 2007). Snake River fall-run Chinook salmon have been above the ICTRT defined minimum abundance threshold since 2001. Productivity, as seen in broodyear returns-per-spawner, has been below replacement (1:1) in recent years, and a longer-term, 20-year geometric mean raw productivity is 0.63 (Figure 28)—likely an underestimate of intrinsic productivity. While below-replacement returns are concerning, the long-term (15-year) abundance trend is stable and the population remains well above the minimum abundance threshold set by the ICTRT. Return rates for broodyears 1995–2000 generally exceeded replacement. Spawner-to-spawner ratios for broodyears 2001–03 were below replacement, cycling above replacement for just one year in 2006, and have been below replacement since 2010. In accordance with the ICTRT methods, survival at all life stages is accounted for by calculating productivity at the spawning ground. This includes ocean, downstream and upstream passage, and freshwater survivals.

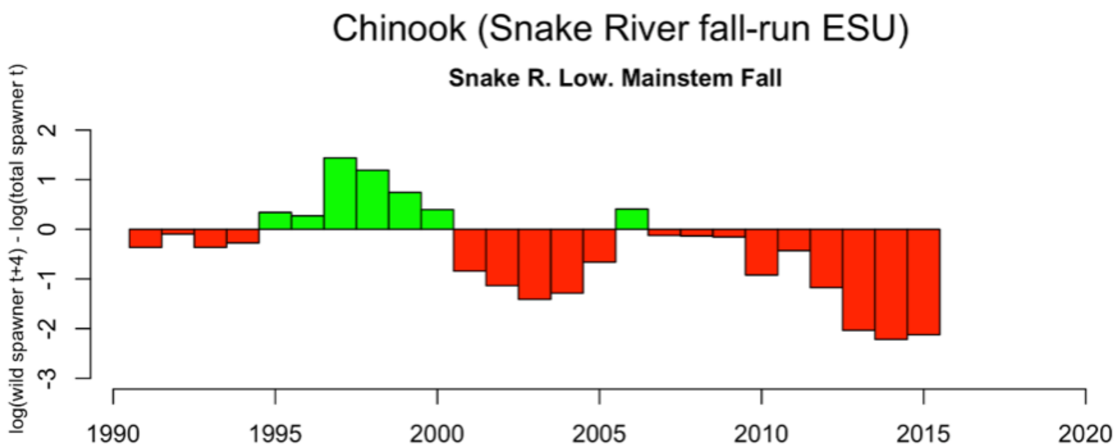

Figure 28. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year (*t* – 4). Spawning years on x-axis.

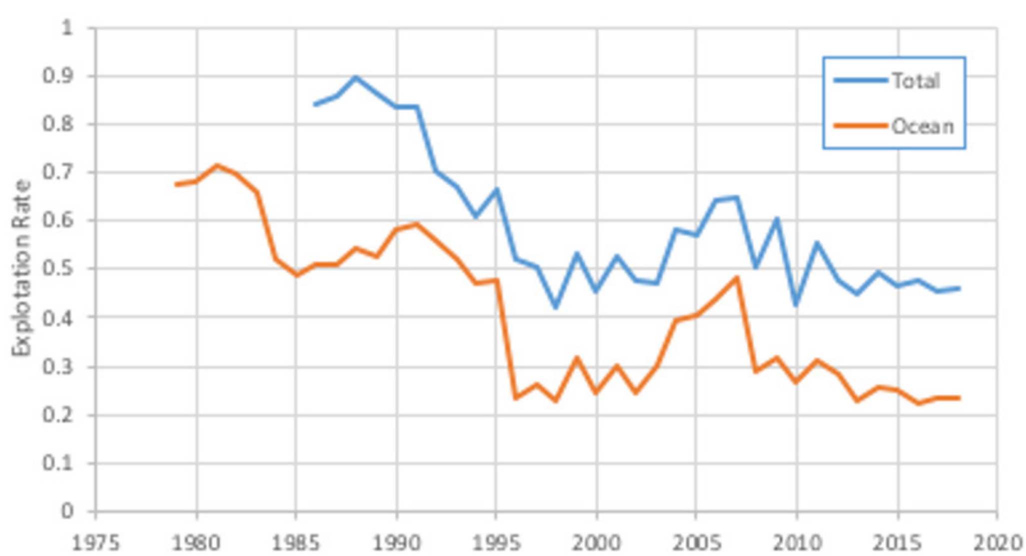

Figure 29. Total exploitation rate for Snake River fall-run Chinook salmon. Data for marine exploitation rates from the Chinook Technical Committee model (Calibration 1503) and for in-river harvest rates from the Columbia River Technical Advisory Committee (TAC 2019 model calibration, old base period).

Snake River fall-run Chinook salmon have a very broad ocean distribution and have been taken in ocean salmon fisheries from Central California through southeastern 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 29). Total exploitation rate has been relatively stable, in the range of 40–50%, since the mid-1990s.

Spatial structure and diversity

The extant Snake River fall-run Chinook salmon 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 Snake River population is the 96-km Upper Hells Canyon reach, extending upriver from the confluence of the Snake and Salmon Rivers 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 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 support fall Chinook salmon spawning in the lower reaches. In addition, there is some historical evidence for production of late spawning Chinook salmon in spatially isolated reaches in upriver tributaries to each of these systems.

Historical records and geomorphic assessments support the historical existence of a fifth MaSA comprising 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, and 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 salmon may have existed in the lower mainstem of the South Fork Salmon River (e.g., Connor et al. 2016). 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.

Annual redd surveys show that fall Chinook salmon spawning occurs in all five of the historical MaSAs (Arnsberg et al. 2020a). PBT has allowed for spawning-ground sampling for parentage analysis. Fidelity studies have indicated there is spawner dispersal within the population from different release sites (Cleary et al. 2018).

The fraction of natural-origin fish on the spawning grounds has remained relatively stable for the last ten years, with five-year means of 31% (2010–14) and 33% (2015–19; Figure 30, Table 17).

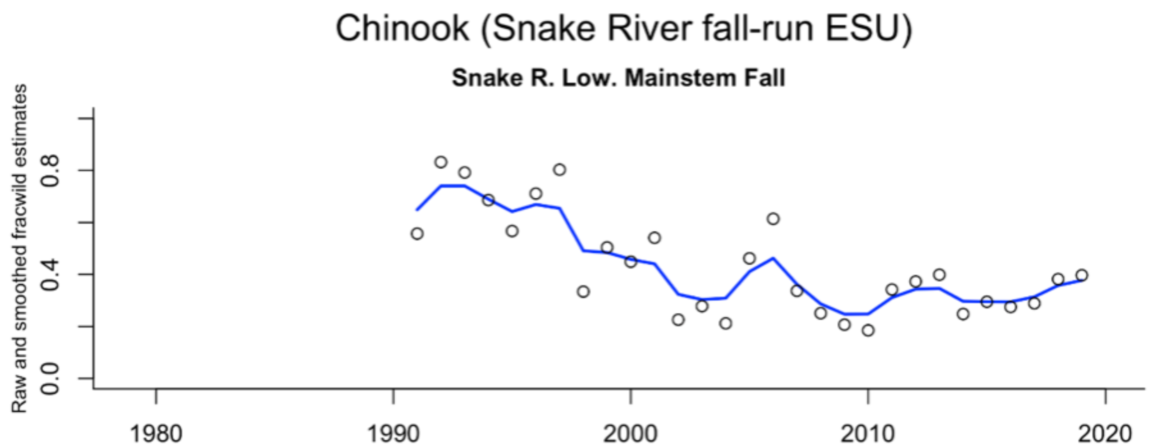

Figure 30. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 17. Five-year mean of fraction natural-origin fish in the population (sum of all estimates divided by the number of estimates).

Population	1995–99	2000–04	2005–09	2010–14	2015–19
Lower Snake River FA	0.58	0.34	0.37	0.31	0.33

Biological viability relative to recovery goals

Consistent with the ICTRT's line of reasoning, the recovery plan contains three recovery scenarios, each consistent with the basic set of viability objectives developed by the ICTRT and each representing a potential pathway to achieving low risk for the ESU. Scenario A would achieve ESU viability with two populations, while Scenarios B and C describe alternative approaches for achieving viability with the single extant Lower Snake River population. Each scenario includes specific criteria and potential metrics for measuring viability characteristics (NMFS 2017b). The scenarios are summarized briefly below.

Scenario A: Two populations, one highly viable and the other viable. This scenario would achieve ESU viability by: a) improving the extant Lower Snake River population to "highly viable" status, and b) reestablishing the extirpated Middle Snake River population above the Hells Canyon Complex to viable status. It reflects a simple, modified application of the ICTRT's general MPG-level viability criteria (which would require that both historical populations achieve "highly viable" status). NMFS determined that this variation on the ICTRT's general criteria was appropriate given the spatial and life-history diversity of the extant Lower Snake River population, and in recognition of the complexities involved in reestablishing the extirpated Middle Snake River population.

Scenario B: A single population measured in the aggregate. Scenario B is based on an alternative application of the ICTRT's criteria. It would achieve ESU viability by improving the status of the extant Lower Snake River population to "highly viable" with a high degree of certainty. VSP characteristics would be evaluated in the aggregate (i.e., population-wide), across all natural-origin adult spawners. The requirement for a high degree of certainty that the population is highly viable would reduce the inherent increased risk associated with a single-population ESU. The spatial complexity and associated ability to support life-history diversity of the Lower Snake River population provide opportunities to achieve the basic ICTRT viability objectives for protection against demographic and catastrophic risk, and to provide for expression of diversity and within-population adaptation to environmental variation.

Scenario C: A single population with natural production emphasis areas. Like Scenario B, Scenario C would achieve ESU viability by achieving high confidence of “highly viable” status for the Lower Snake River population. In this scenario, however, rather than evaluating population status in the aggregate, as under Scenario B, population status would be evaluated based on having a substantial amount of natural production for the ESU come from one or two of the five major spawning areas. These NPEAs would be managed to have a low percentage of hatchery-origin spawners and to support significant levels of natural-origin spawners (other major spawning areas could have higher acceptable levels of hatchery-origin spawners). The NPEAs would make it possible to directly evaluate the productivity of the natural population and ensure that a substantial proportion of the population is subject to natural selection rather than hatchery processes.

While the ten-year geometric mean natural-origin abundance level has been high—8,920 natural-origin spawners (2010–19) relative to the >4,200 natural-origin spawners for the single-population scenario (B) which the population is closer to meeting—the abundance/productivity margin is insufficient to rate as “very low risk” given the uncertainty-buffering requirement under the single-population viability scenario; the most recent 20-year geometric mean raw productivity is 0.63 and the recovery plan calls for ≥1.7 (NMFS 2017b). As a result, the Lower Snake River population is rated at “low risk” (Table 18), rather than “very low risk,” for abundance and productivity.

In terms of spatial structure and diversity, the Lower Snake River Chinook salmon population is rated at “low risk” for Goal A (allowing natural rates and levels of spatially mediated processes), as the population shows regular dispersal into all five available spawning areas. It is rated “moderate risk” for Goal B (maintaining natural levels of variation), resulting in an overall spatial structure and diversity rating of “moderate risk” (Table 18). In particular, the rating reflects the relatively high proportion of within-population hatchery spawners (70%) in all major spawning areas, which does not meet the requirements of either single-population recovery plan strategy (B or C).

Table 18. Lower Snake River fall-run Chinook salmon population risk ratings integrated across the four VSP parameters. Viability key: HV = highly viable, V = viable, M = maintained, HR = high risk (does not meet viability criteria).

		SS/D risk			
		Very low	**Low**	**Moderate**	**High**
A/P risk	**Very low (<1%)**	HV	HV	V	M
	Low (1–5%)	V	V	V (Lower Snake River)	M
	Moderate (6–25%)	M	M	M	HR
	High (>25%)	HR	HR	HR	HR

Updated biological risk summary

Overall population viability for the Snake River fall-run Chinook salmon ESU is determined based on a combination of ratings for current abundance and productivity and combined spatial structure diversity. The current risk rating for the Lower Snake River population is “viable” (Table 18). The single-population delisting options provided in the draft Snake River fall-run Chinook salmon recovery plan would require the population to meet or exceed minimum requirements for “highly viable” with a high degree of certainty. The current rating is based on evaluating current status against the recovery plan criteria for the single, aggregate population scenarios (Scenarios B or C).

To achieve “highly viable” status with a high degree of certainty, the SS/D 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).

Overall, the status of Snake River fall-run Chinook salmon has clearly improved compared to the time of listing. 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 Complex (NMFS 2017b). The Snake River fall-run Chinook salmon ESU therefore is considered to be at a moderate-to-low risk of extinction, with viability largely unchanged from the prior review.

Snake River Sockeye Salmon ESU

Brief description of ESU

The Snake River sockeye salmon ESU includes all naturally spawned anadromous and residual sockeye salmon originating from the Snake River basin, as well as sockeye salmon from the Redfish Lake Captive Broodstock Program and the Snake River Sockeye Salmon Hatchery Program (USOFR 2005a, 2020; Figure 31). This ESU was first listed as endangered under the ESA in 1991; the listing was reaffirmed in 2005, 2012, and 2016.

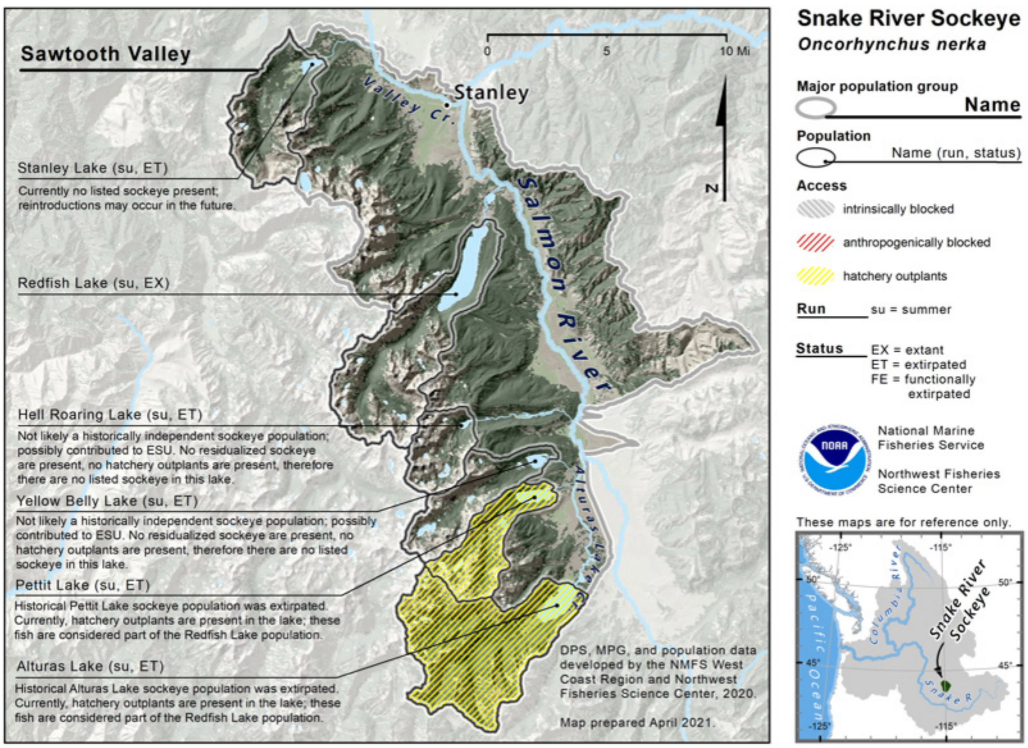

Figure 31. Map of the Snake River sockeye salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

Summary of previous viability 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…" (p. 421), 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 broodstock-based hatchery program, but natural production levels of anadromous returns remained extremely low for this ESU. 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, Idaho, lakes that were candidates for sockeye restoration. The availability of increased numbers of adults supported 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. 2008). 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, 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.

2015

NWFSC (2015) concluded that the Snake River Sockeye ESU remained at extremely high risk ,although there had 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. They concluded that there was no basis for changing the ESU ratings assigned in prior reviews, but that the trend in status appears to be positive.

Description of new data available for this review

Estimates of annual returns are now available through 2019. Adult returns in 2014 were the highest since the current captive brood-based program began, with a total of 1,579 counted back to the Stanley Basin. The majority of the adults captured in recent years were trapped 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. In 2015, conditions during migration led to high mortality within the hydrosystem and an emergency transport of fish from Lower Granite Dam.

Juvenile outmigrant survivals from release to Lower Granite Dam have been highly variable. High in-basin mortality in smolts released in 2015–17 was found to be due to water chemistry shock between the Springfield Hatchery water and the water of Redfish Lake. By 2018, acclimation studies showed that one week at the intermediate-hardness water at Sawtooth Hatchery was sufficient transition from Springfield Hatchery to Redfish Lake

(Johnson et al. 2019). Juvenile survival from Lower Granite Dam to Bonneville Dam from 2008–13 ranged from 40% to 57% (NMFS 2016). Recent years had both higher highs and lower lows. Highs in 2014 and 2018 were at 71% and 64% respectively, but 2015–17 survivals ranged from 12% to 37%, the same years with high mortality at release (Widener et al. 2018).

Upstream adult passage survivals from Bonneville Dam to Lower Granite Dam averaged 60% survival from 2014 to 2018, excluding 2015 when survival was less than 4% due to high temperatures during the migration period. Adult survivals from Lower Granite Dam to the Stanley Basin averaged 56% for 2014–18, excluding 2015 when they dropped to 14%. Temperatures during the adult upstream migration in 2015 were unusually high due to low snowpack coupled with extremely high air temperatures. This resulted in warm water in the major tributaries and led to an almost complete collapse of the run between Bonneville and Lower Granite Dams (NMFS 2016). These losses would have affected the SARs for SY 2010 and SY 2011. 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 and phenotypic and behavioral characteristics, Snake River sockeye salmon are particularly susceptible to high summer temperatures during their adult migration (Crozier et al. 2008, Crozier et al. 2020). The conditions in 2015 are expected to become less rare as climate change progresses.

Abundance and productivity

Adult returns of sockeye salmon to the Sawtooth Basin crashed in 2015, and natural returns have remained low (Table 19, Figure 32). The low returns of fish collected at the Redfish Lake and Sawtooth Hatchery weirs have limited anadromous releases into Redfish Lake to 311 anadromous hatchery fish in 2016 (Figure 33). No natural anadromous fish have been released since 2014, as they are required to be spawned in the captive broodstock program under NMFS Section 10 Permit 1454. Captive adult releases have continued to support spawning in Redfish Lake. Smolt-to-adult return rates suggest that volitional spawning within Redfish Lake appears to be important to the success of the Snake River sockeye salmon captive broodstock-based hatchery program (Kozfkay et al. 2019).

Table 19. Five-year geometric mean of raw natural return counts. In parentheses, 5-year geometric mean of raw total return counts is shown. 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 2 most-recent 5-year periods is shown on the far right.

Population	2000–04	2005–09	2010–14	2015–19	% change
Snake River sockeye	4 (26)	9 (33)	137 (699)	16 (113)	–89 (–84)

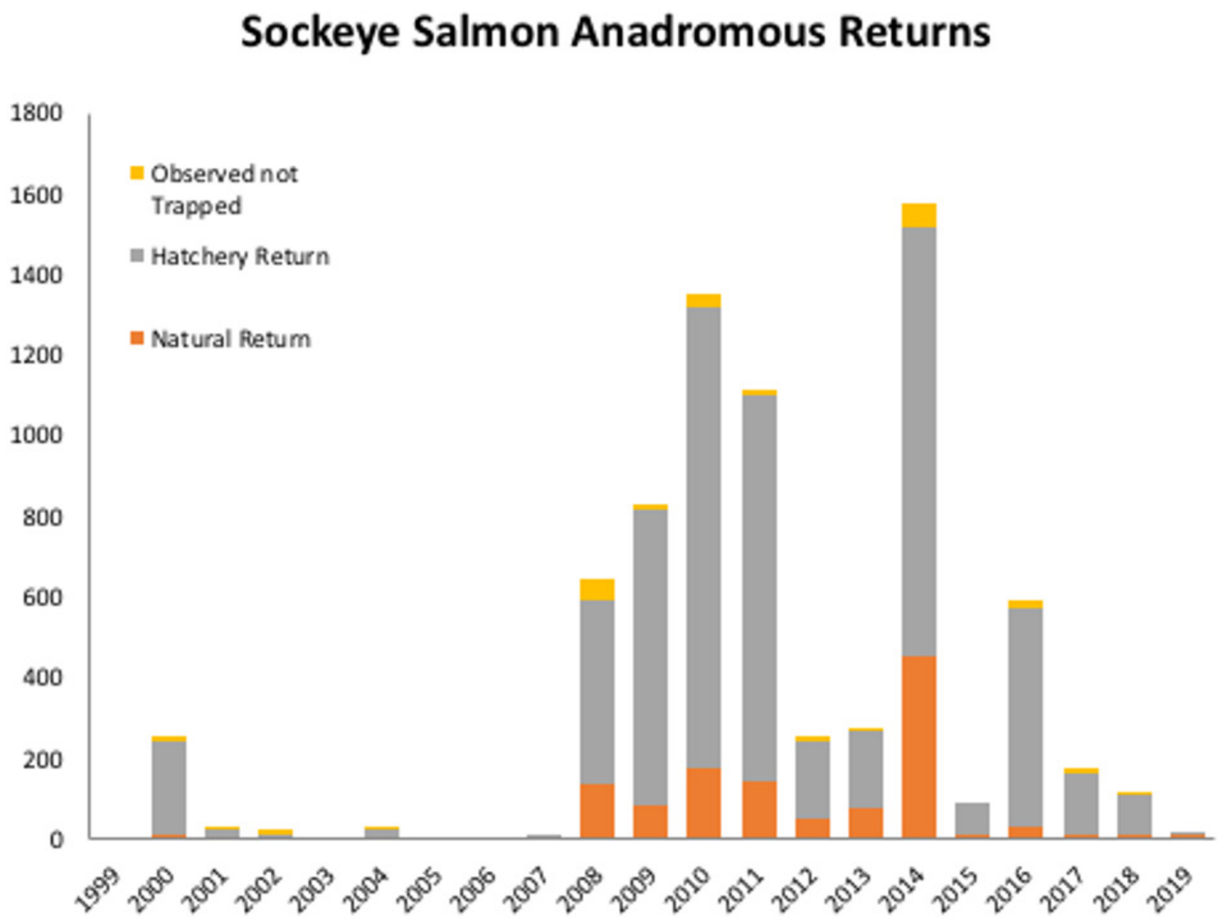

Figure 32. Snake River sockeye salmon anadromous returns, 1999–2019 (figure from Johnson et al. 2020).

In 2015, low snowpack, coupled with extremely high air temperatures throughout the interior Columbia River basin, resulted in warm water in the major tributaries to the lower Snake and Columbia Rivers. Temperatures in the mainstem Columbia River were the highest recorded from roughly mid-June to mid-July. Adult sockeye salmon, which normally migrate during this period, sustained heavy losses in the Columbia River and tributaries, with losses in the mainstem migration corridor exceeding 95% between Bonneville and Lower Granite Dams (NMFS 2016).

With low sockeye salmon returns to the Stanley Basin, the hatchery program remains in its initial phase with a priority on genetic conservation and building sufficient returns to support sustained outplanting (NMFS 2015).

Since discontinuing the presmolt program due to relatively poor smolt-to-adult return rates, direct smolt plants in the lower section of Redfish Lake Creek and in the Salmon River (Sawtooth Hatchery weir) have been increased, ranging from 423,103 to 882,386 per year in the most recent five-year period (2015–19; Figure 34). Survival at planting has improved with acclimation at Sawtooth Hatchery between Springfield Hatchery and release into Redfish Lake Creek (Johnson et al. 2019).

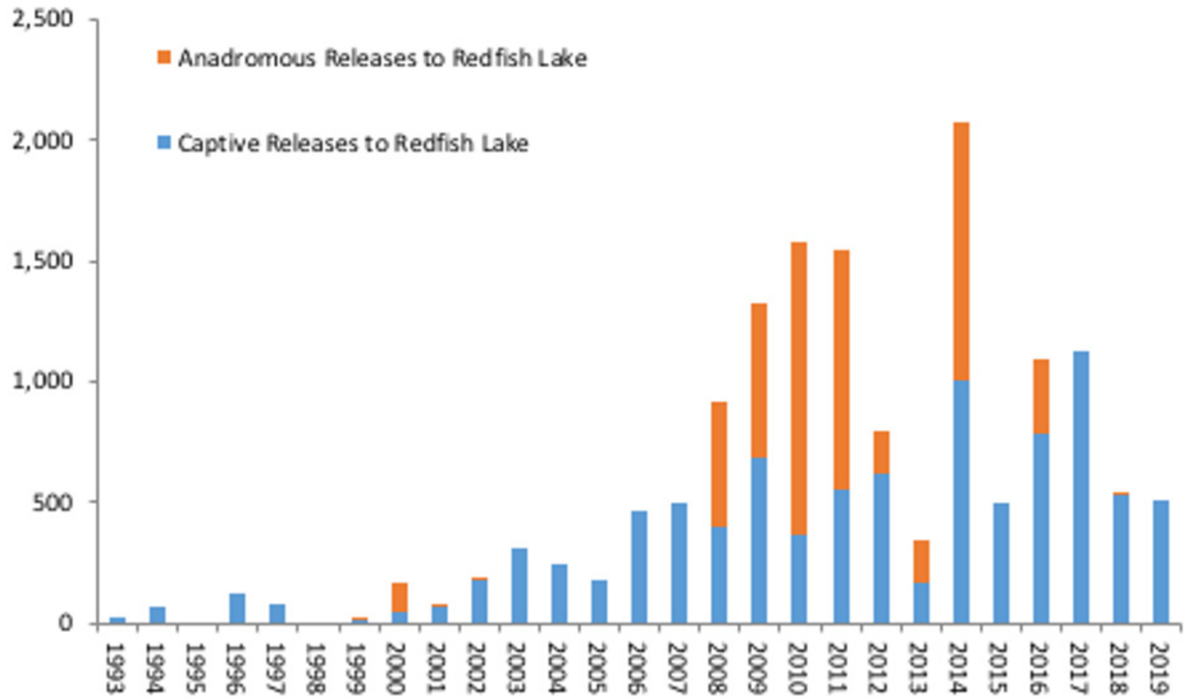

Figure 33. Adult releases into Redfish Lake of anadromous and captive fish (figure from Johnson et al. 2020).

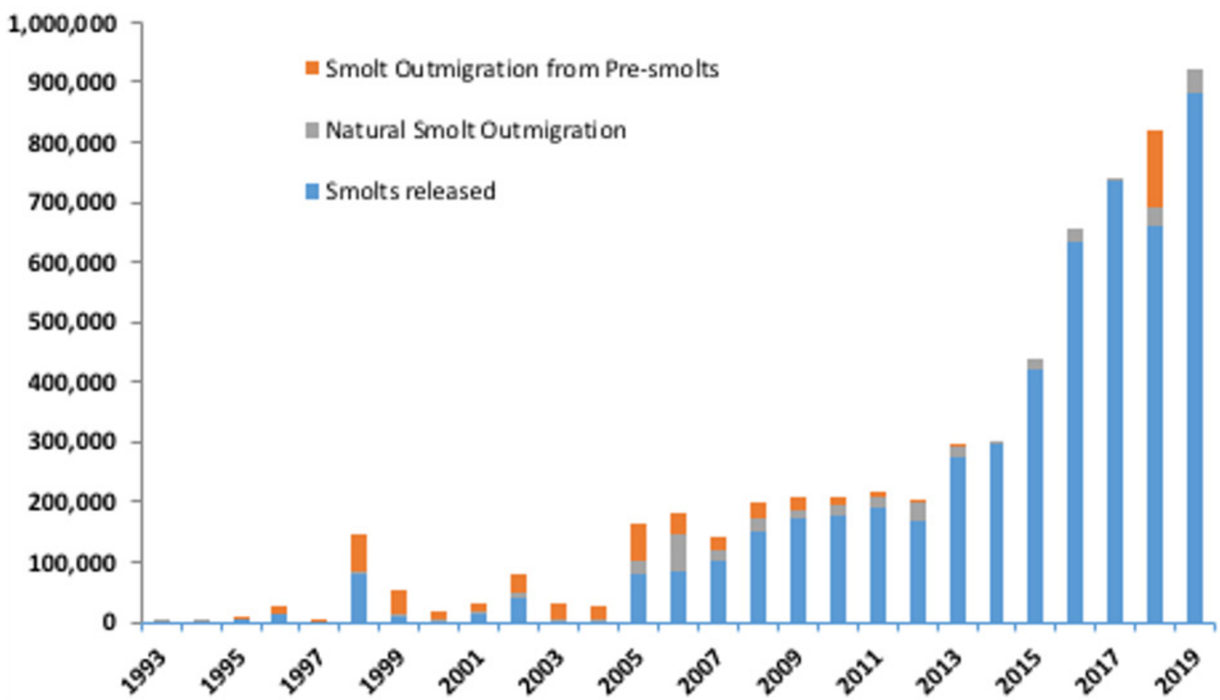

Figure 34. Estimated annual numbers of sockeye salmon smolt outmigrants from the Stanley Basin. This includes all hatchery smolt releases, known outmigrants originating from hatchery presmolt outplants, and estimates of unmarked juveniles migrating from Redfish, Alturas, and Petit Lakes combined (figure from Johnson et al. 2020).

Unmarked juvenile sockeye salmon emigrating from the three lake systems have averaged approximately 22,523 over the most-recent five years, ranging from over 38,886 in 2019 to a low of 5,488 in 2017 (Figure 35). A number of sources could be contributing to the outmigration of unmarked juveniles, including prior years' adults passed into Redfish Lake, captive broodstock adult outplants, egg box outplants, or natural production from residual spawners (Kozfkay et al. 2019).

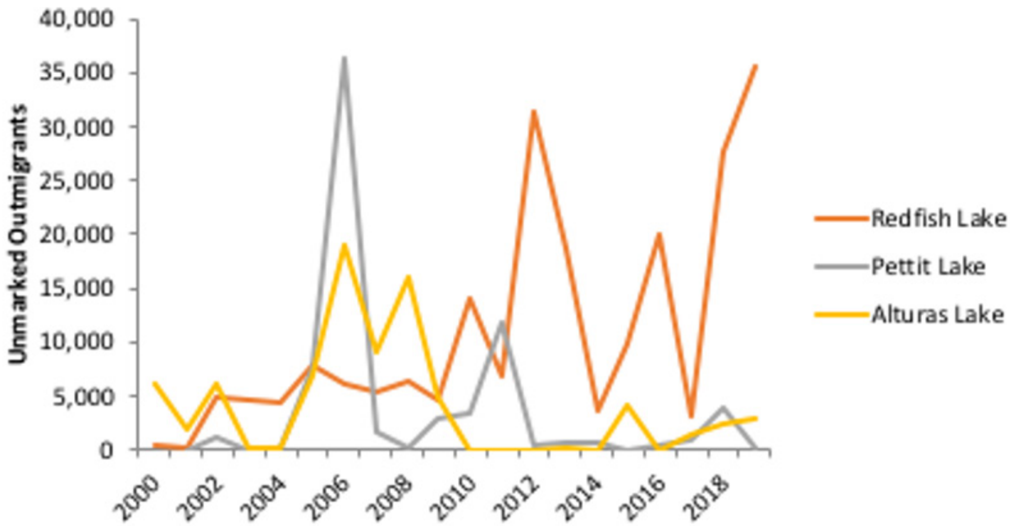

Figure 35. Estimates of unmarked juveniles migrating from Redfish, Alturas, and Petit Lakes (figure from Johnson et al. 2020).

Natural production occurring within Redfish Lake had the highest overall survival rates from the smolt-to-adult life stage, despite having lower emigration survival from the Sawtooth Valley basin to Lower Granite Dam (Johnson et al. 2019). Increases in smolt abundances have not led to increases in natural adult returns (Kozfkay et al. 2019).

Annual basin-to-basin estimates of SAR rates through broodyear 2014 (returns completed in 2019) have been generated for Snake River sockeye salmon through a combination of PBT and a length-at-age key for fish that assign as unknown. Natural production from Redfish Lake SARs averaged 0.41% for the five most-recent brood years (2010–14) that have completed returns, with a ten-year average of 0.86%. Natural production from Pettit and Alturas Lakes, from anadromous and captive releases, averaged 0.86% for the five most-recent broodyear returns, and 0.53% for the ten most-recent. Hatchery production smolts averaged 0.30% and 0.43% for Oxbow Reservoir smolts and 0.08% and 0.21% for Sawtooth Valley smolts in five- and ten-year averages respectively (Johnson et al. 2020). There are two brood years of Springfield Hatchery smolt releases completed now, with no adult returns due to the water chemistry acclimation issues.

The Lower Granite Dam SARs reflect aggregate return rates across two major downstream migration routes: in-river passage and downstream transport to below Bonneville Dam. 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 and 69% for 2012–18 (Widener 2019). The median estimates of juvenile passage survivals for the McNary-to-Bonneville Dam reach (1998–2003, 2006–10) were 0.54 and (2012–18) 0.62, which should be interpreted with caution due to small sample sizes and associated low detection probabilities for many of the individual year estimates. The median estimated survival from Lower Granite Dam to Bonneville Dam for the period 2012–18 was 0.47 (Widener 2019).

Estimated survival in 2019 of Snake River sockeye salmon (hatchery- and natural-origin combined) from the tailrace of Lower Granite Dam to the tailrace of Bonneville Dam was 43.4% (95% CI: 37.7–49.9%). Estimated survival in 2019 of Columbia River sockeye salmon (hatchery- and natural-origin combined) from the tailrace of Rock Island Dam to the tailrace of Bonneville Dam was 73.7% (44.7%–121.5%). Both estimates were above their respective long-term averages of 40.7% and 50.6% (Zabel 2019).

Sockeye transported through the hydrosystem have a much lower adult survival than run-of-the-river. Adult migration through the Columbia River reach was half the observed survival for those transported as juveniles than not (0.30 vs. 0.59; Crozier et al. 2020). Fallback occurs at much higher rates in sockeye than other salmon, and has been a significant predictor of sockeye survival, slowing travel and increasing thermal exposure (Crozier et al. 2018). No transported sockeye survived upstream migration to Lower Granite Dam in 2015.

Sockeye salmon returning to Redfish Lake in Idaho's Sawtooth Valley travel a greater distance from the sea, 1,448 km (900 mi), to a higher elevation (1,996 m [6,500 ft]), than any other sockeye salmon population. They are the southernmost population of sockeye salmon in the world. Adult upstream migration takes place during midsummer, exposing the salmon to altered climate conditions such as higher temperatures and lower flows. Adult upstream passage survivals through the mainstem Columbia River to the mouth of the Snake River are assumed to be relatively high during normal conditions based on inferences from estimates of upstream passage for upper Columbia River sockeye salmon (Johnson et al. 2019). 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 PIT-tag detections at multiple dams also indicate relatively low direct passage mortality upstream to Lower Granite Dam (Johnson et al. 2019). Conditions during the 2015 adult migration led to a loss of 95% of the run between Bonneville Dam and Lower Granite Dam (NMFS 2016).

While conditions in 2015, with very warm water temperatures in the migration corridor and low flows, were uncommonly rare historically, they are expected to become more common in the future as climate change progresses (Crozier et al. 2020). This ESU's reliance on captive broodstock production inhibits any natural evolutionary changes in run timing.

Harvest

Ocean fisheries do not significantly impact Snake River sockeye salmon. 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 36).

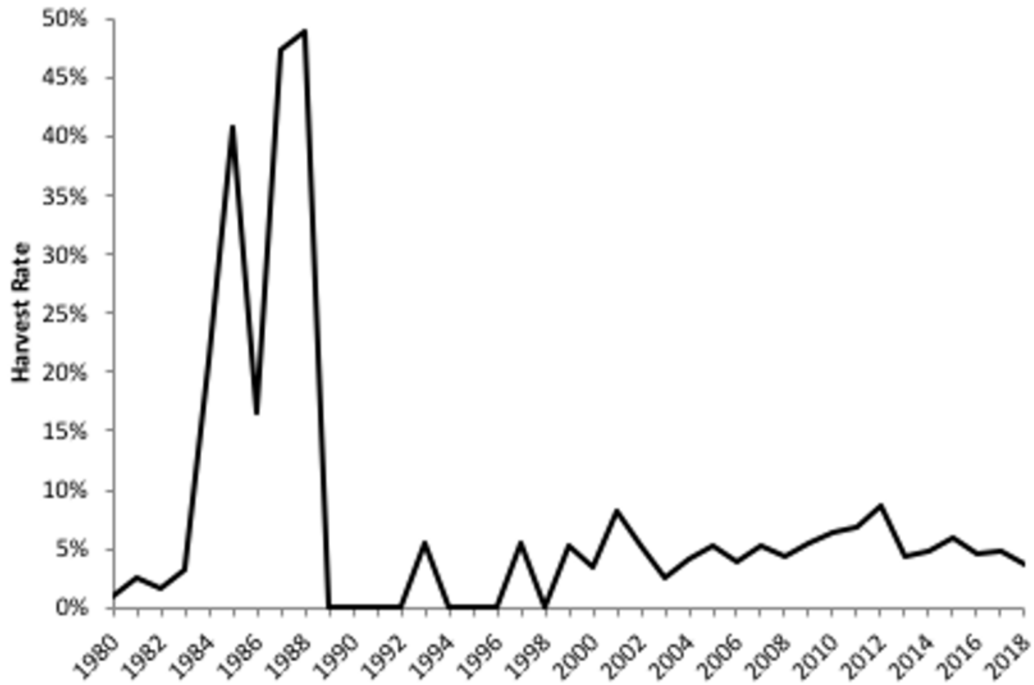

Figure 36. Exploitation rates on Snake River sockeye salmon. Data from the Columbia River Joint Staff Report (2019).

Spatial structure and diversity

There is evidence that the historical Snake River sockeye salmon ESU supported a range of life-history patterns, with spawning populations present in several of the small lakes in the Stanley Basin (NMFS 2015). 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 Fishhook Creek (NMFS 2015). Historical accounts indicate that Alturas Lake Creek supported an early timed riverine, and may have also contained lake shoal spawners (NMFS 2015).

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 reintroduction into Redfish Lake, the initial target for restoring natural production (NMFS 2015). 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 viability 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 salmon remains limited to extremely low levels in Redfish Lake, one of five Sawtooth Valley lakes believed to have historically supported production, with a few thousand outmigrants each year from Pettit and Alturas Lakes. 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 broodstock-based hatchery program.

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 salmon restoration. The availability of increased numbers of adults and juveniles has supported direct evaluation of lake habitat rearing potential, juvenile downstream passage survivals, and adult upstream survivals. Although the captive broodstock 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 salmon reduces the risk of immediate loss, but levels of naturally produced sockeye salmon returns remain extremely low and at high risk from climate change.

Updated biological risk summary

In terms of natural production, the Snake River sockeye salmon 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. Current climate change modeling supports the “extremely high risk” rating with the potential for extirpation in the near future (Crozier et al. 2020). The viability of the Snake River sockeye salmon ESU therefore has likely declined since the time of the prior review, and the extinction risk category remains “high.”

Snake River Basin Steelhead DPS

Brief description of DPS

The Snake River Basin 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 southeastern Washington, northeastern Oregon, and Idaho, as well as several hatchery programs (Figure 37; USOFR 2020). Snake River Basin steelhead are classified as summer-run based on their adult run timing patterns. Much of the freshwater habitat used by Snake River Basin steelhead for spawning and rearing is warmer and drier than that associated with other steelhead DPSes. Snake River Basin steelhead spawn and rear as juveniles across a wide range of freshwater temperature/precipitation regimes.

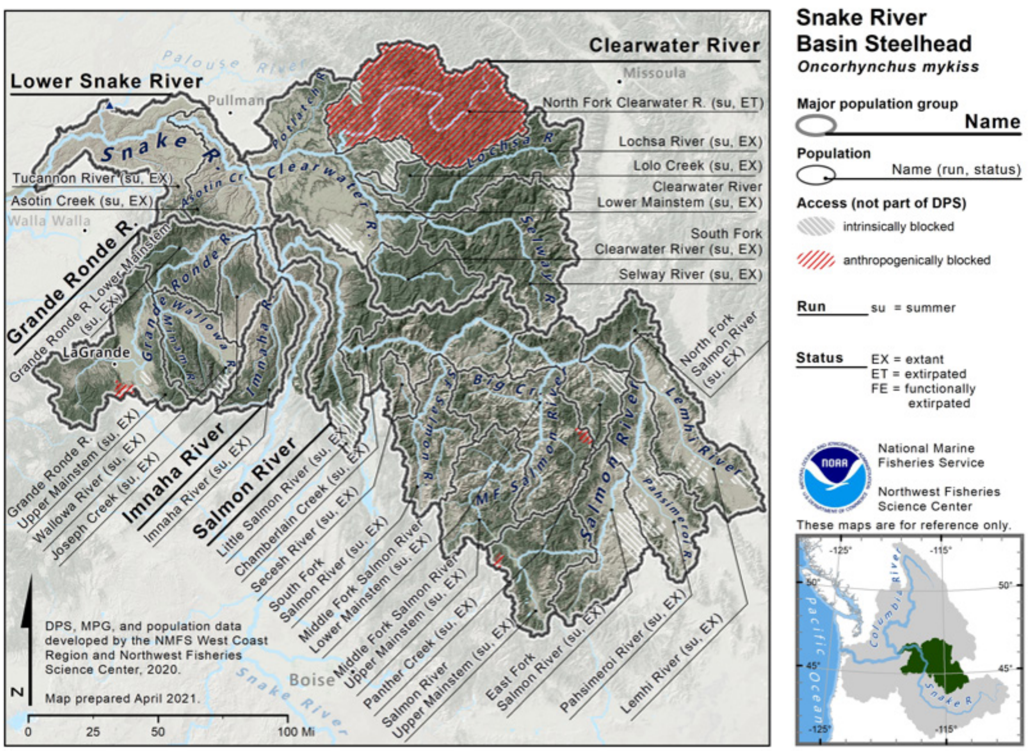

Figure 37. Snake River Basin steelhead DPS spawning and rearing areas, illustrating populations and major population groups.

NMFS has defined DPSes of steelhead to include only the anadromous members of this species (USOFR 2005b). Our approach to assessing the current viability of a steelhead DPS is based on evaluating information about the abundance, productivity, spatial structure, and diversity of the anadromous component of this species (Good et al. 2005). Many steelhead populations along the U.S. West Coast 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 DPSes (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 viability of the anadromous form.

Summary of previous viability 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 viability of B-run populations was specifically cited as a particular concern. The BRT identified 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 of 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" (p. 300). 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 viability 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.

2015

In the last status review (NWFSC 2015), four out of the five MPGs were not meeting the specific objectives in the draft recovery plan based on the updated viability information available for the review, and the viability of many individual populations remained uncertain. The Grande Ronde River Upper Mainstem MPG was 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 River populations were recommended to improve future assessments. The additional monitoring programs instituted in the early 2000s to gain better information on natural-origin abundance and related factors had significantly improved our ability to assess viability at a more detailed level. The new information 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 indicated that differences in A/P 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 were updated. A great deal of uncertainty still remaind regarding the relative proportion of hatchery fish in natural spawning areas near major hatchery release sites within individual populations. Overall, the information analyzed for the 2015 review did not indicate a change in biological risk status from prior reviews.

Description of new data available for this review

In the past, adult abundance data series for the Snake River Basin 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 River populations (Joseph Creek and Grande Ronde River Upper Mainstem), 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 (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 ten different stock groups. The second project generates estimates of the escapement at the population or watershed level for 21 groups with a mixture model (DABOM) based on PIT-tag detections from a network of locations across the DPS. All three data sets are presented, generally as three separate panels for each figure. Since the mixture model-based estimate has only recently been operationalized, most of the resulting time series are too short to be used to generate long-term indices of abundance and productivity (e.g., 15-year trends). It is also important to note that the standardized methods of evaluating abundance and productivity that are applied across all ESUs/DPSes (see Methods) are slightly different from the metrics established by the ICTRT—the primary difference being the time base for estimating abundance (five and 15 years versus ten years) and the productivity measure.

Ocean condition indices

Juvenile steelhead are more pelagic than salmon, heading off the continental shelf soon after entering the ocean in the spring (Burgner 1992). Steelhead migrate seasonally across the North Pacific Ocean, moving to the north and west in spring and to the south and east, across the entire Pacific, from autumn through winter (Atcheson et al. 2012). Thus, steelhead ocean survival may be impacted by different factors than salmon. In fact, recent work has shown that steelhead population groupings from geographic regions have unique smolt survival trends that appear to be driven by factors affecting them early in their ocean residence, despite steelhead smolts generally a) being larger than Pacific salmon smolts when they enter the ocean, and b) making wide-ranging, off-the-continental-shelf migrations, rather than remaining more coastal, as Pacific salmon smolts tend to do (Kendall et al. 2017).

Aggregate annual returns of Upper Columbia River spring-run Chinook salmon 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). Work is ongoing to relate indices of ocean condition to steelhead populations up and down the U.S. West Coast. Steelhead marine survival seems to be related to ocean surface temperature in the first summer of ocean entry, and populations respond similarly to spatial patterns of ocean conditions at a rough grain of 250 km between ocean entry points (Kendall et al. 2017). Therefore, broad spatial patterns of ocean conditions may not capture the finer spatial scale of response that steelhead seem to exhibit.

Indicators of ocean condition are highly correlated with each other, and exhibit strong temporal autocorrelation (Figure 129). As a result, when indicators point to conditions that result in poor ocean productivity for salmonid populations, they do so as a suite of indicators, and for runs of "good" or "bad" years (see Habitat chapter). Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019a). Recent (2015–19) ensemble ocean indicator rankings include four of the worst seven years in the past 20, meaning that an entire salmon or steelhead generation could have been subjected to poor ocean productivity conditions.

Genetic diversity

IDFG has compiled an updated assessment of genetic relationship based on 5,967 samples taken from 62 locations within the DPS (Ackerman et al. 2014). The results generally support the MPG structure derived by the ICTRT and identify clear population-level structure within monophyletic clades in the Salmon River and Clearwater River groups (Hargrove et al. 2021; Figure 38). The upper Salmon River genetic structure has been evaluated further in recent work from IDFG (Powell and Campbell 2020). Differentiation among samples from the Grande Ronde River Upper Mainstem and Lower Snake River MPGs is 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. In addition, stock definitions based on genetic markers (GSI) from 2009–19 returns were used to reconstruct individual stock trajectories from the aggregate LGR counts back to 1985 (Lawry et al. 2020).

Abundance and productivity

Evaluations were done using both a set of metrics corresponding to those used in prior 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 DPSes to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the recovery evaluation section. Population estimates for the time series available for this assessment are archived and available through NWFSC's Salmonid Population Summary database.

The five-year geometric mean abundance estimates for the populations in this DPS all show significant declines in the recent past (Figure 39, Table 20). Each of the populations decreased by roughly 50% in the past five-year period, resulting in a near-zero population change in the past 15 years (Table 21) for the three populations with sufficiently long data time series. Hatchery-origin spawner estimates for these populations continued to be low (Table 22).

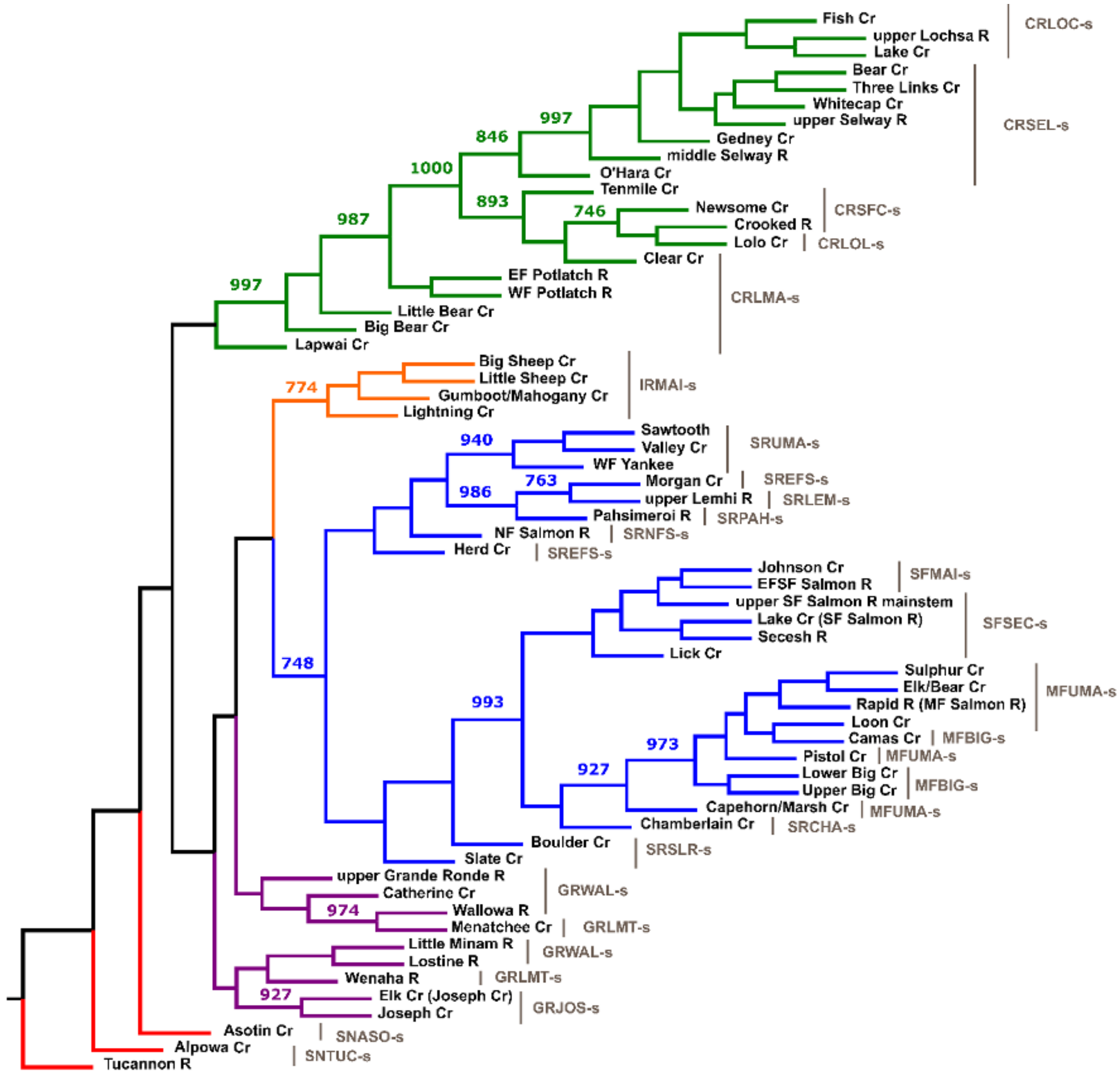

Figure 38. Genetic relationships of steelhead collected from locations across the Snake River basin. The tree is based on Nei's genetic distance. Numbers along branches show number of bootstraps out of 1,000 replicates that support the grouping. Only support greater than 70% is shown. Reproduced from Hargrove et al. (2021).

Populations in the Snake River Basin steelhead DPS exhibited similar temporal patterns in broodyear returns per spawner, oscillating with a rough period of ten years (Figure 40). Return rates for broodyears 1995–99 generally exceeded replacement (1:1). Spawner-to-spawner ratios for broodyears 2001–03 were generally well below replacement for many populations, cycling above replacement during 2005–10, and strongly below replacement since 2010. Broodyear 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.

Results from the genetic stock composition monitoring at Lower Granite Dam (beginning with the 2008–09 cycle year) and the systematic PIT-tag program are providing finer-scale geographic estimates of steelhead returns by region of origin. The GSI-based approach is currently able to break out the aggregate natural returns at Lower Granite Dam into ten

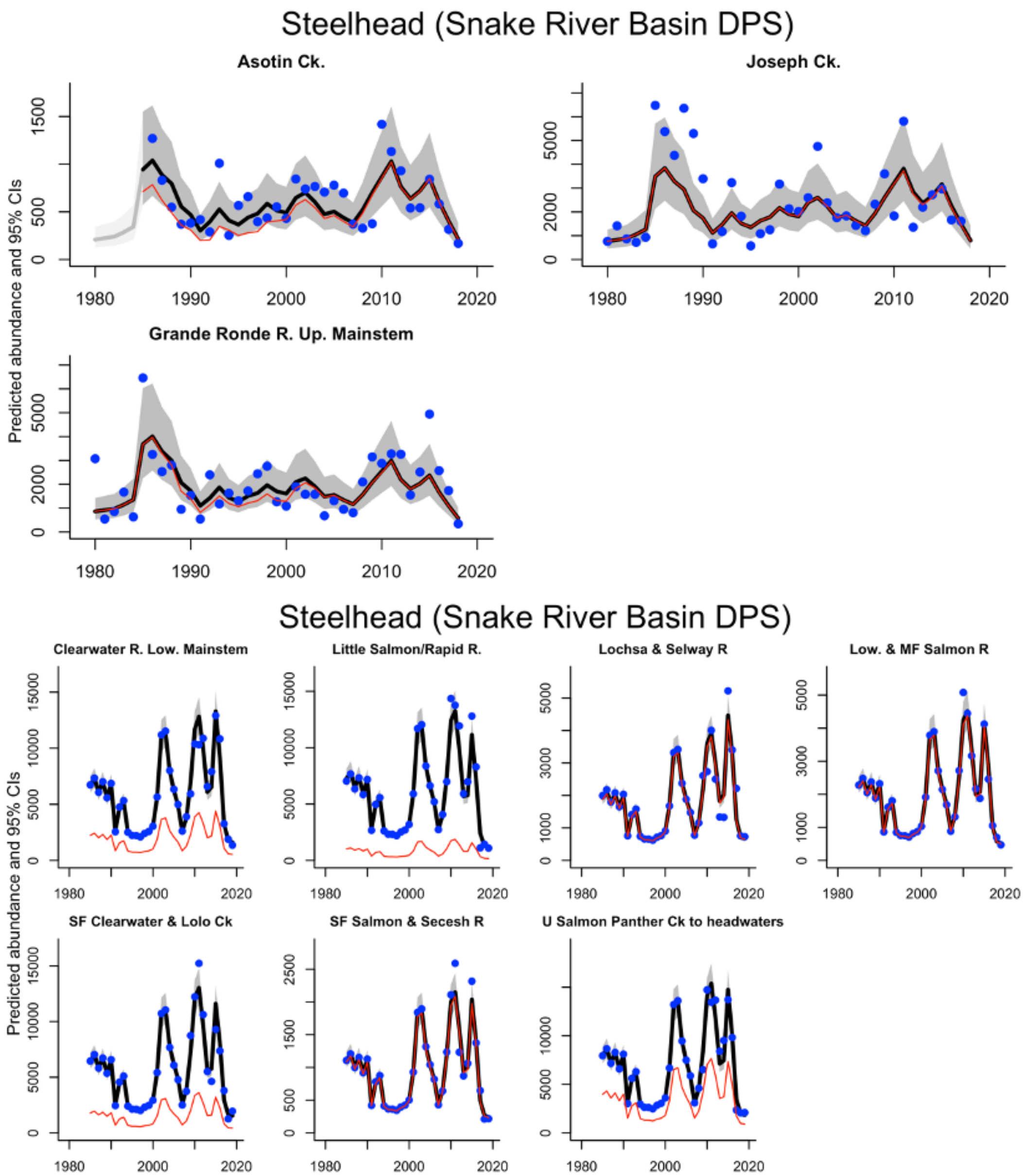

Figure 39. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot. Figure continues on next page. This page, top: Long-term dataset from weir and redd surveys. Bottom: Super-population groups from GSI-based run partitioning of the run-at-large over Lower Granite Dam.

stock reporting groups. The year-to-year patterns in aggregate Snake River basin stocks of wild summer steelhead also show a steep recent decline (Figure 41). Stocks definitions based on genetic markers (2009–19 returns) were used to reconstruct individual stock trajectories from the aggregate LGR counts back to 1985 (Lawry et al. 2020).

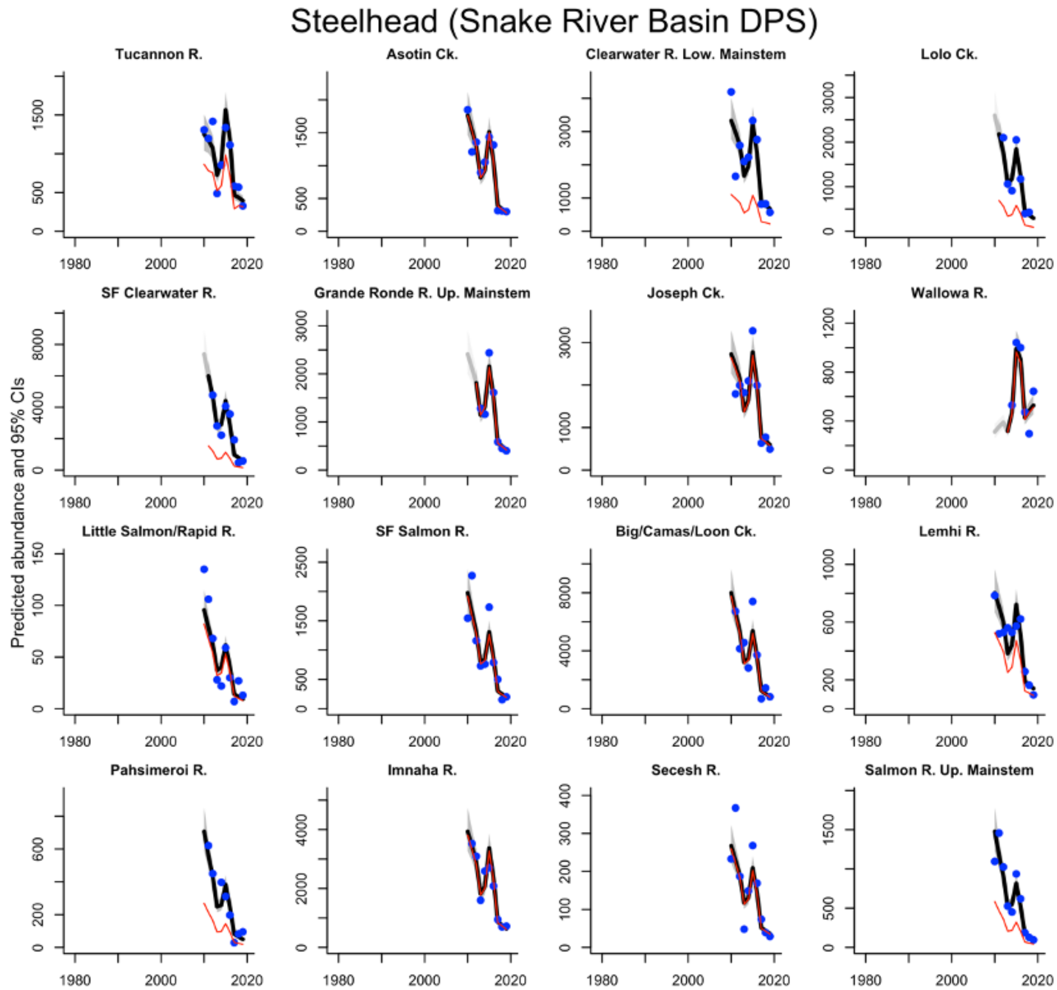

Figure 39 (continued). This page: PIT-tag-based population estimation method based on mixture model and tag detection network across the DPS.

Run reconstructions of ten genetically identified stocks from the aggregate Snake River Basin steelhead natural-origin run show similar patterns of productivity through time (1985–2019, Figure 42), including the low productivity (ln R/S < 0) expected from the declining stock abundance (Lawry et al. 2020).

As noted above, results from the genetic stock composition monitoring at Lower Granite Dam beginning with the 2008–09 cycle year and the systematic PIT-tag program are providing finer-scale geographic estimates of steelhead returns by region of origin. The GSI-based approach is currently able to break out the aggregate natural returns at Lower Granite Dam into ten stock reporting groups (Figure 38). Five of those groupings likely have negligible or very low hatchery contributions (Figure 44). 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 Basin steelhead 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 cutoff (78 cm; Ackerman et al. 2014).

Table 20. Five-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. 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 2 most-recent 5-year periods is shown on the far right. Upper rows: long-term dataset from weir and redd surveys. Middle rows (shaded): super-population groups from GSI-based run partitioning of the run-at-large over Lower Granite Dam. Lower rows: PIT-tag-based population estimation method based on mixture model and tag detection network across the DPS.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Asotin Creek	Lower Snake River	249 (411)	331 (513)	611 (680)	438 (477)	841 (847)	400 (402)	–52 (–53)
Joseph Creek	Grande Ronde River	1,728 (1,728)	1,394 (1,394)	2,533 (2,533)	1,926 (1,926)	2,380 (2,439)	1,936 (1,996)	–19 (–18)
Grande Ronde River Upper Mainstem	Grande Ronde River	1,029 (1,307)	1,443 (1805)	1,165 (1284)	1,453 (1,459)	2,572 (2,604)	1,639 (1,655)	–36 (–36)
Clearwater River Lower Mainstem	Clearwater River	1,351 (4,069)	763 (2,298)	2,352 (7,084)	1,504 (4,531)	3,004 (9,048)	1,364 (4,110)	–55 (–55)
Lochsa and Selway Rivers	Clearwater River	1,170 (1,206)	660 (681)	2,037 (2,100)	1,410 (1,453)	2,109 (2,175)	1,796 (1,852)	–15 (–15)
South Fork Clearwater River and Lolo Creek	Clearwater River	1,082 (3,898)	611 (2,202)	1,885 (6,788)	1,314 (4,732)	2,421 (8,719)	1,011 (3,640)	–58 (–58)
Little Salmon/Rapid River	Salmon River	599 (4,251)	338 (2,400)	1,044 (7,403)	683 (4,847)	1,403 (9,947)	400 (2,840)	–71 (–71)
South Fork Salmon and Secesh Rivers	Salmon River	648 (668)	366 (377)	1,129 (1,164)	757 (780)	1,397 (1,440)	604 (623)	–57 (–57)
Lower and Middle Fork Salmon River	Salmon River	1,334 (1,375)	753 (777)	2,323 (2,395)	1,578 (1,627)	3,014 (3,107)	1,246 (1,284)	–59 (–59)
Upper Salmon River and Panther Creek to headwaters	Salmon River	2,393 (4,805)	1,351 (2,713)	4,165 (8,364)	2,625 (5,272)	5,814 (11,674)	2,112 (4,240)	–64 (–64)
Tucannon River	Lower Snake River	—	—	—	—	679 (985)	460 (695)	–32 (–29)
Asotin Creek	Lower Snake River	—	—	—	—	1,224 (1,234)	558 (561)	–54 (–55)
Clearwater River Lower Mainstem	Clearwater River	—	—	—	—	805 (2,426)	428 (1,289)	–47 (–47)
Lolo Creek	Clearwater River	—	—	—	—	402 (1,268)	253 (799)	–37 (–37)
South Fork Clearwater River	Clearwater River	—	—	—	—	800 (3,100)	388 (1,502)	–52 (–52)
Grande Ronde River Upper Mainstem	Grande Ronde River	—	—	—	—	1,213 (1,220)	832 (838)	–31 (–31)
Joseph Creek	Grande Ronde River	—	—	—	—	1,866 (1,924)	1,063 (1,096)	–43 (–43)
Wallowa River	Grande Ronde River	—	—	—	—	—	605 (623)	—
Imnaha River	Imnaha River	—	—	—	—	2,516 (2,594)	1,181 (1,217)	–53 (–53)
Little Salmon/Rapid River	Salmon River	—	—	—	—	49 (57)	18 (21)	–63 (–63)
South Fork Salmon River	Salmon River	—	—	—	—	1,142 (1,177)	449 (463)	–61 (–61)
Big/Camas/Loon Creeks	Salmon River	—	—	—	—	4,219 (4,350)	1,807 (1,863)	–57 (–57)
Lemhi River (SU)	Salmon River	—	—	—	—	379 (577)	177 (270)	–53 (–53)
Pahsimeroi River	Salmon River	—	—	—	—	183 (481)	41 (107)	–78 (–78)
Secesh River	Salmon River	—	—	—	—	158 (163)	80 (82)	–49 (–50)
Upper Salmon River	Salmon River	—	—	—	—	327 (828)	105 (266)	–68 (–68)

Table 21. Fifteen-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	2004–19
Asotin Creek	Lower Snake River	0.06 (0.04, 0.08)	0.00 (–0.05, 0.06)
Joseph Creek	Grande Ronde River	0.03 (0.01, 0.05)	–0.01 (–0.06, 0.05)
Grande Ronde River Upper Mainstem	Grande Ronde River	0.03 (0.01, 0.05)	–0.02 (–0.07, 0.04)

Table 22. Five-year mean of fraction natural (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range. Upper rows: long-term dataset from weir and redd surveys. Middle rows (shaded): super-population groups from GSI-based run partitioning of the run-at-large over Lower Granite Dam. Lower rows: PIT-tag-based population estimation method based on mixture model and tag detection network across the DPS.

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Asotin Creek	Lower Snake River	0.65	0.90	0.92	0.99	1.00
Joseph Creek	Grande Ronde River	1.00	1.00	1.00	0.98	0.97
Grande Ronde River Upper Mainstem	Grande Ronde River	0.80	0.91	1.00	0.99	0.99
Clearwater River Lower Mainstem	Clearwater River	0.33	0.33	0.33	0.33	0.33
Lochsa and Selway Rivers	Clearwater River	0.97	0.97	0.97	0.97	0.97
South Fork Clearwater River and Lolo Creek	Clearwater River	0.28	0.28	0.28	0.28	0.28
Little Salmon/Rapid River	Salmon River	0.14	0.14	0.14	0.14	0.14
South Fork Salmon and Secesh Rivers	Salmon River	0.80	0.91	1.00	0.99	0.99
Lower and Middle Fork Salmon River	Salmon River	0.97	0.97	0.97	0.97	0.97
Upper Salmon River and Panther Creek to headwaters	Salmon River	0.50	0.50	0.50	0.50	0.50
Tucannon River	Lower Snake River	—	—	—	0.69	0.68
Asotin Creek	Lower Snake River	—	—	—	0.99	1.00
Clearwater River Lower Mainstem	Clearwater River	—	—	—	0.33	0.33
Lolo Creek	Clearwater River	—	—	—	0.32	0.32
South Fork Clearwater River	Clearwater River	—	—	—	0.26	0.26
Grande Ronde River Upper Mainstem	Grande Ronde River	—	—	—	1.00	0.99
Joseph Creek	Grande Ronde River	—	—	—	0.97	0.97
Wallowa River	Grande Ronde River	—	—	—	0.97	0.97
Imnaha River	Imnaha River	—	—	—	0.97	0.97
Little Salmon/Rapid River	Salmon River	—	—	—	0.86	0.86
South Fork Salmon River	Salmon River	—	—	—	0.97	0.97
Big/Camas/Loon Creeks	Salmon River	—	—	—	0.97	0.97
Lemhi River (SU)	Salmon River	—	—	—	0.66	0.66
Pahsimeroi River	Salmon River	—	—	—	0.38	0.38
Secesh River	Salmon River	—	—	—	0.97	0.97
Upper Salmon River	Salmon River	—	—	—	0.40	0.40

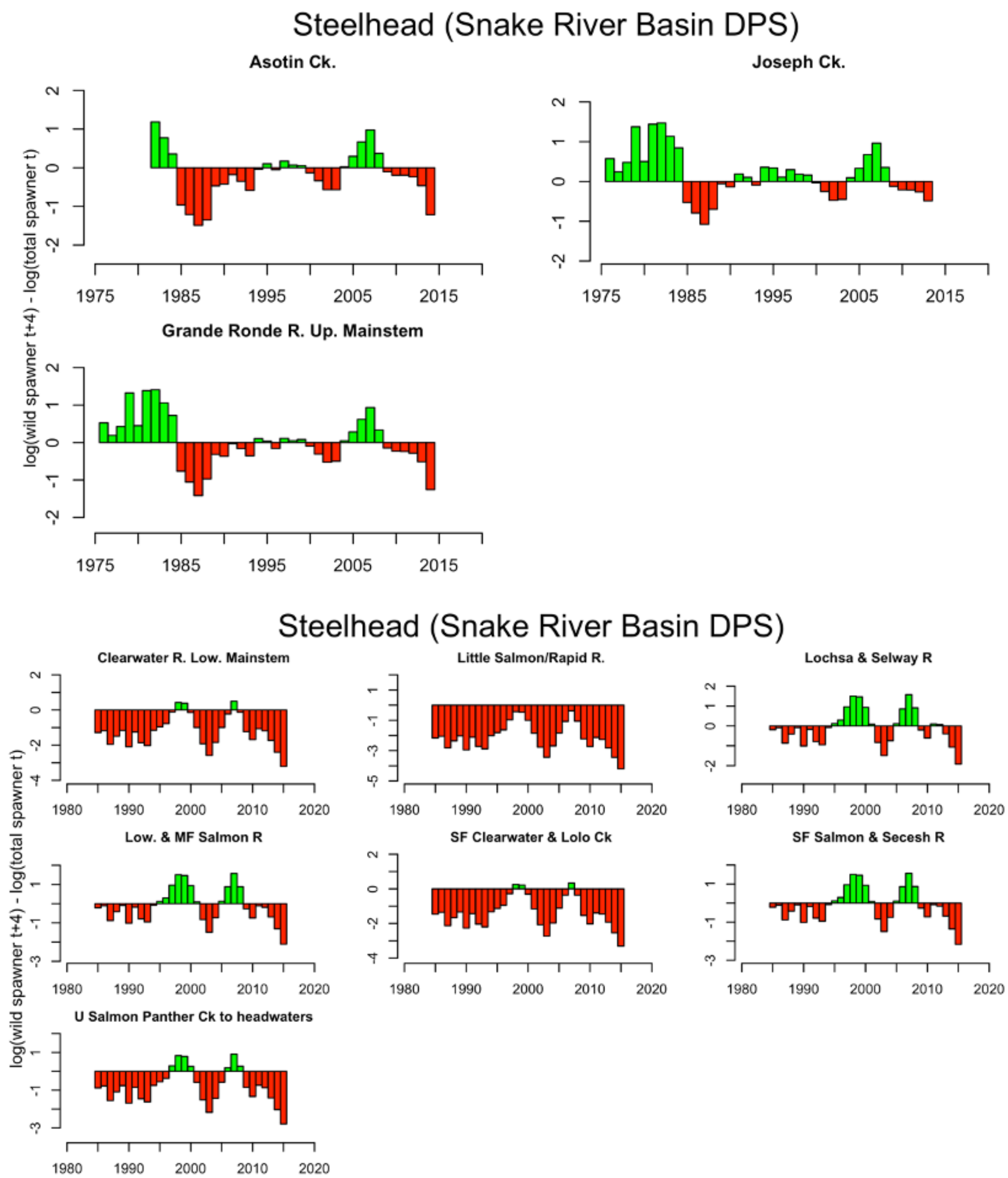

Figure 40. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year ($t - 4$). Spawning years on x-axis. Figure continues on next page. This page, top: Long-term dataset from weir and redd surveys. Bottom: Super-population groups from GSI-based run partitioning of the run-at-large over Lower Granite Dam. Lower panel, PIT tag based population estimation method based on mixture model and tag detection network across the DPS.

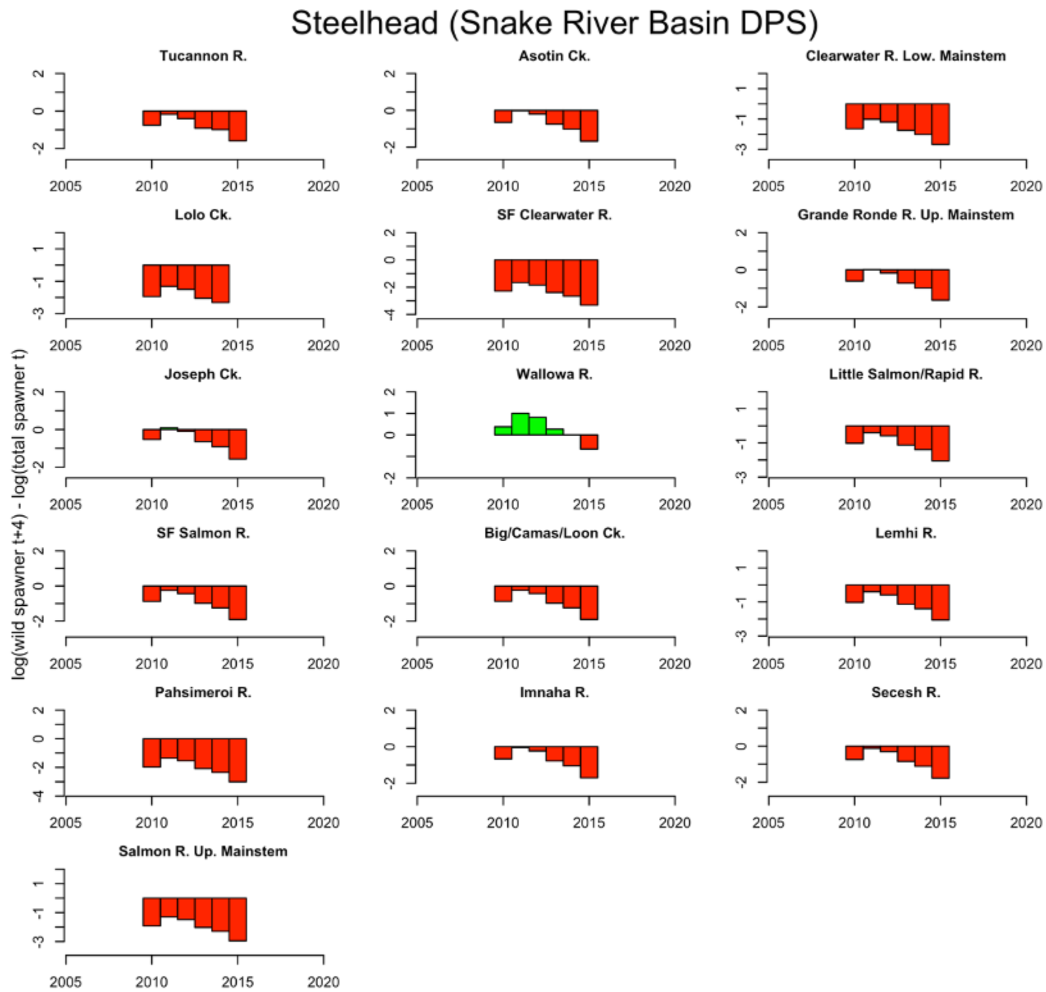

Figure 40 (continued). Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year ($t-4$). This page: PIT-tag-based population estimation method based on mixture model and tag detection network across the DPS.

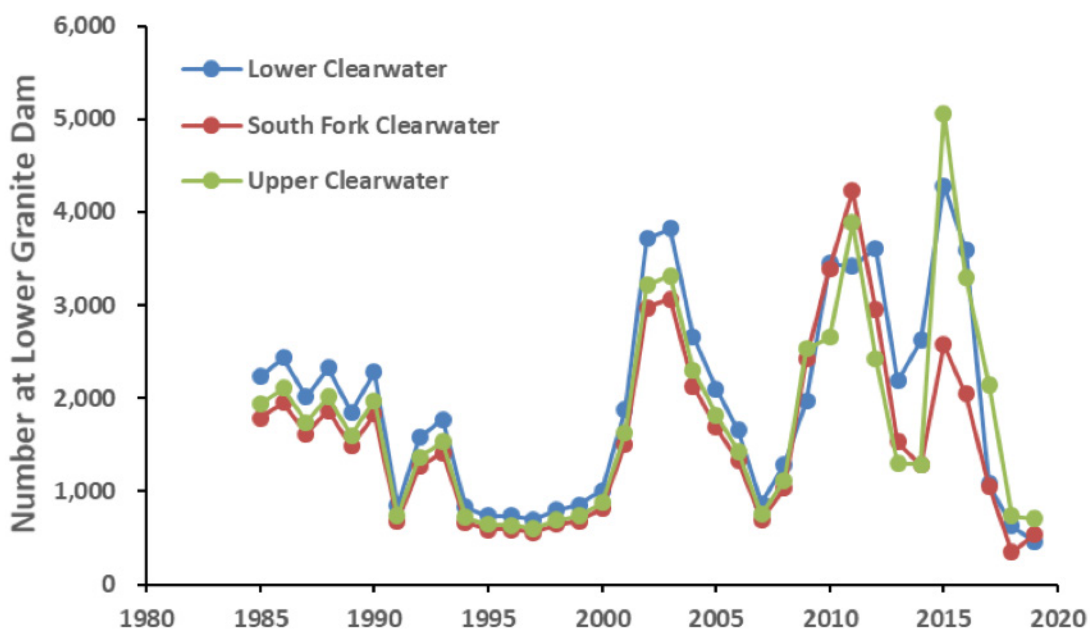

Figure 41. Estimated returns of natural-origin steelhead at Lower Granite Dam by spawning year, 1985–2019. Broken out by Clearwater River (this page) and Salmon River (next page) stocks. Figures from Lawry et al. 2020.

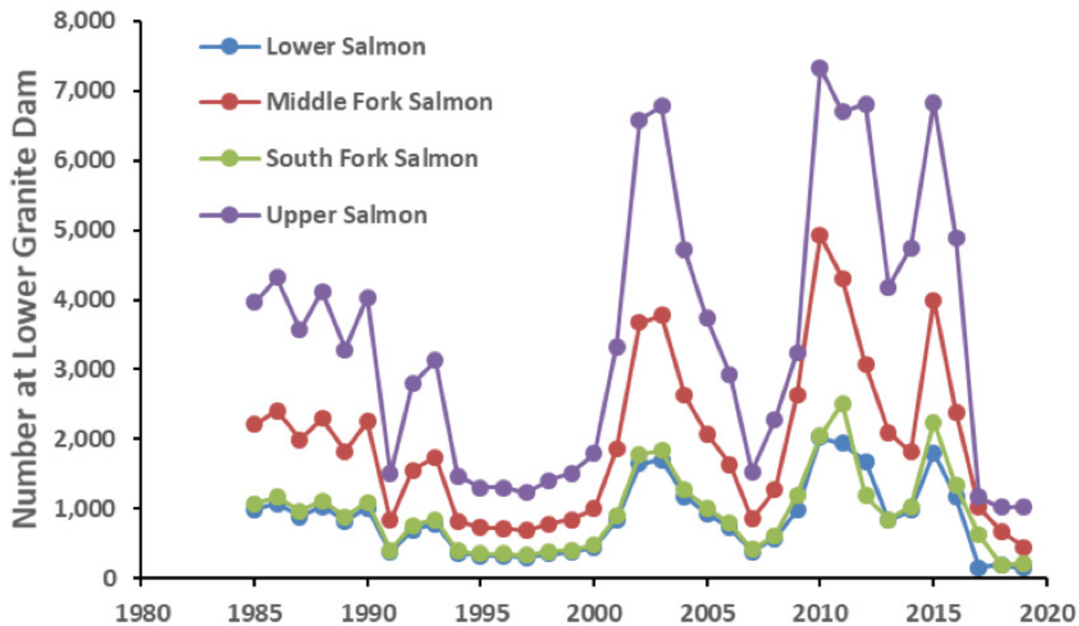

Figure 41 (continued). Estimated returns of natural-origin steelhead at Lower Granite Dam by spawning year, 1985–2019.

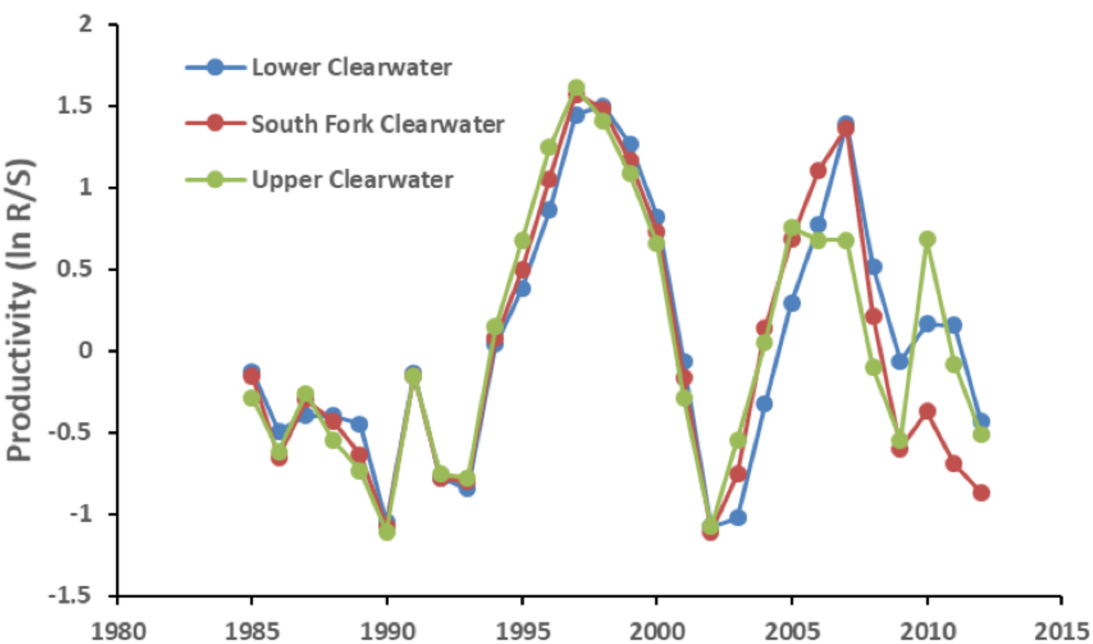

Figure 42. Snake River Basin natural-origin steelhead aggregate stock productivity (ln R/S). This page: Stocks from the Clearwater River MPG. Next page: Stocks from the Salmon River MPG. Run reconstruction for stocks is based on recruits and spawners at Lower Granite Dam. Stocks were identified by a recent PBT effort (2009–19), and extended back to 1985 based on consistent age and stock proportion composition of the run at large. Figures from Lawry et al. 2020.

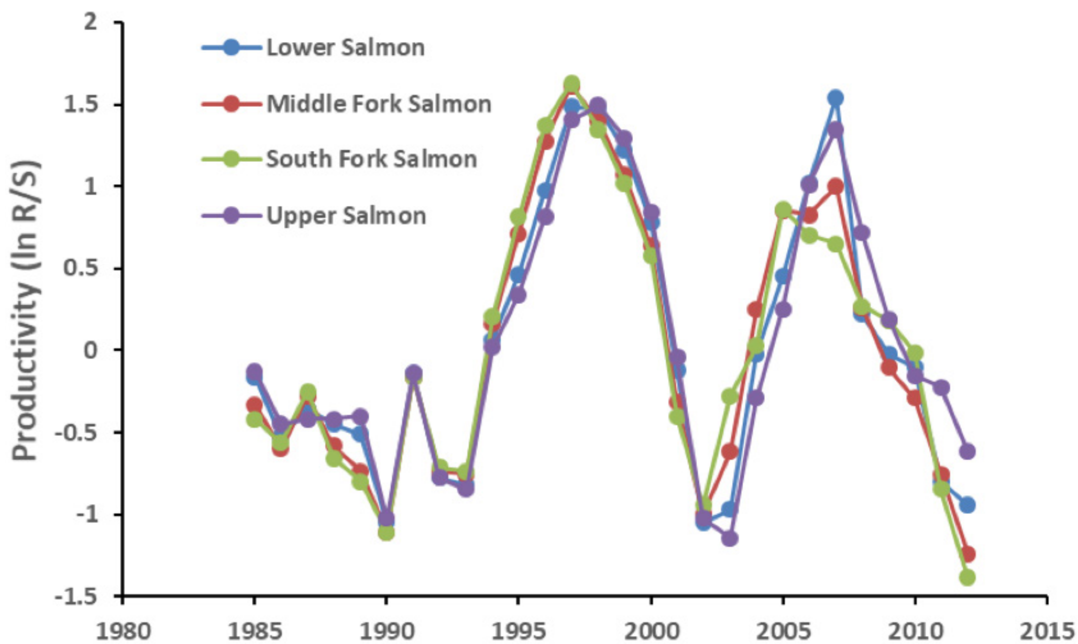

Figure 42 (continued). Snake River Basin natural-origin steelhead aggregate stock productivity (ln R/S).

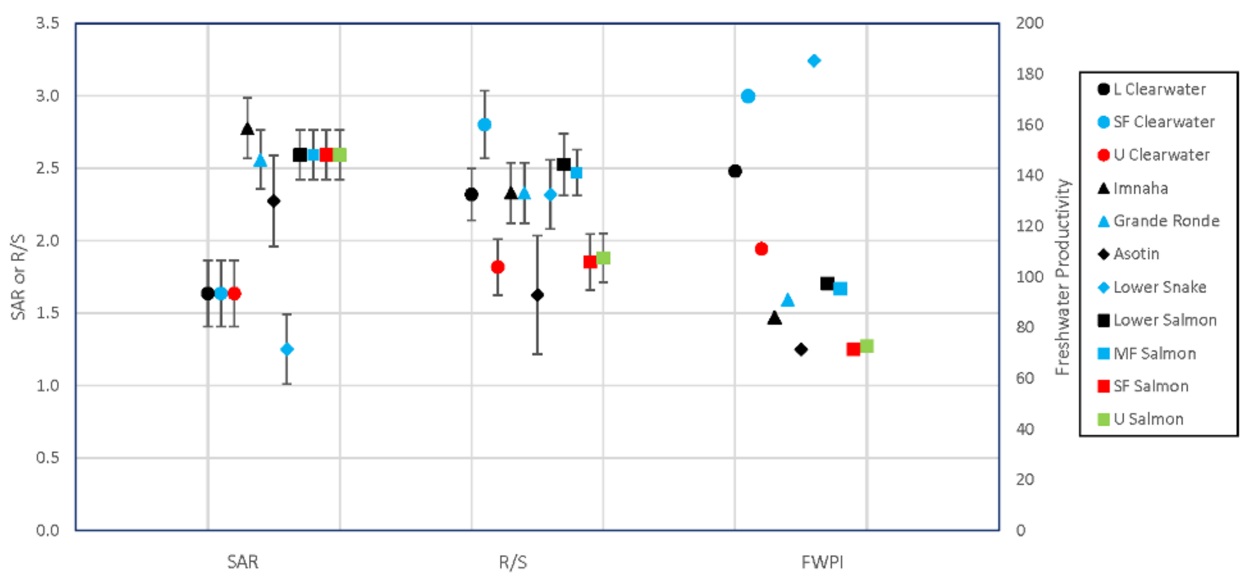

Figure 43. Smolt to Adult Return, Recruits per Spawner, and Freshwater Productivity Index (FWPI) for each of the populations in the ESU. Geometric means of SAR and R/S are shown for each population, along with the standard error of the estimate (whiskers represent +/- one standard error). The time period included in the SAR or R/S indices is the past 20 years, depending on data availability. The FWPI is constructed as a ratio of the geomean R/S and SAR, and can be thought of as a measure of smolts per spawner.

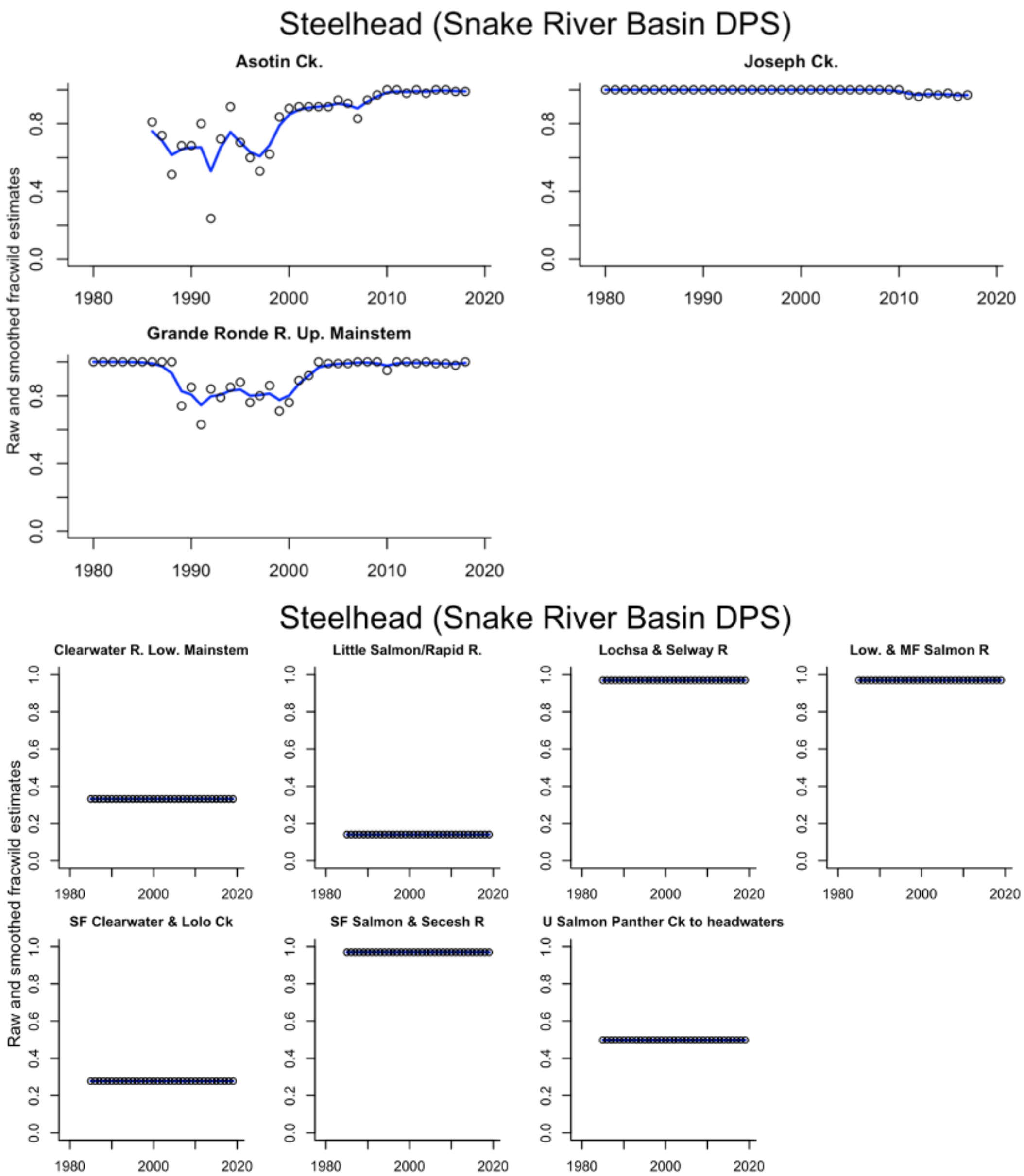

Figure 44. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates. Figure continues on next page. This page, top: Long-term dataset from weir and redd surveys. Bottom: Super-population groups from GSI-based run partitioning of the run-at-large over Lower Granite Dam.

SARs are generated by the Columbia River Data Access in Real Time (CBR and UW 2020) project using PIT-tag detections from all release locations within each population basin (CBR and UW 2020). The SAR indices represent cumulative marine, nearshore, and estuary survival. Figure 43 shows the geometric mean of R/S and SAR indices for the stocks available across four MPGs in the DPS. In general, these broad-brush descriptors of population processes indicate relatively robust long-term behavior of the populations. Using the R/S and SAR indicators by population, it is possible to generate an indicator of freshwater productivity (FWPI) as a ratio of R/S and SAR. This quantity can be thought of as an indicator of smolts per spawner, and thus, the overall population productivity in the freshwater

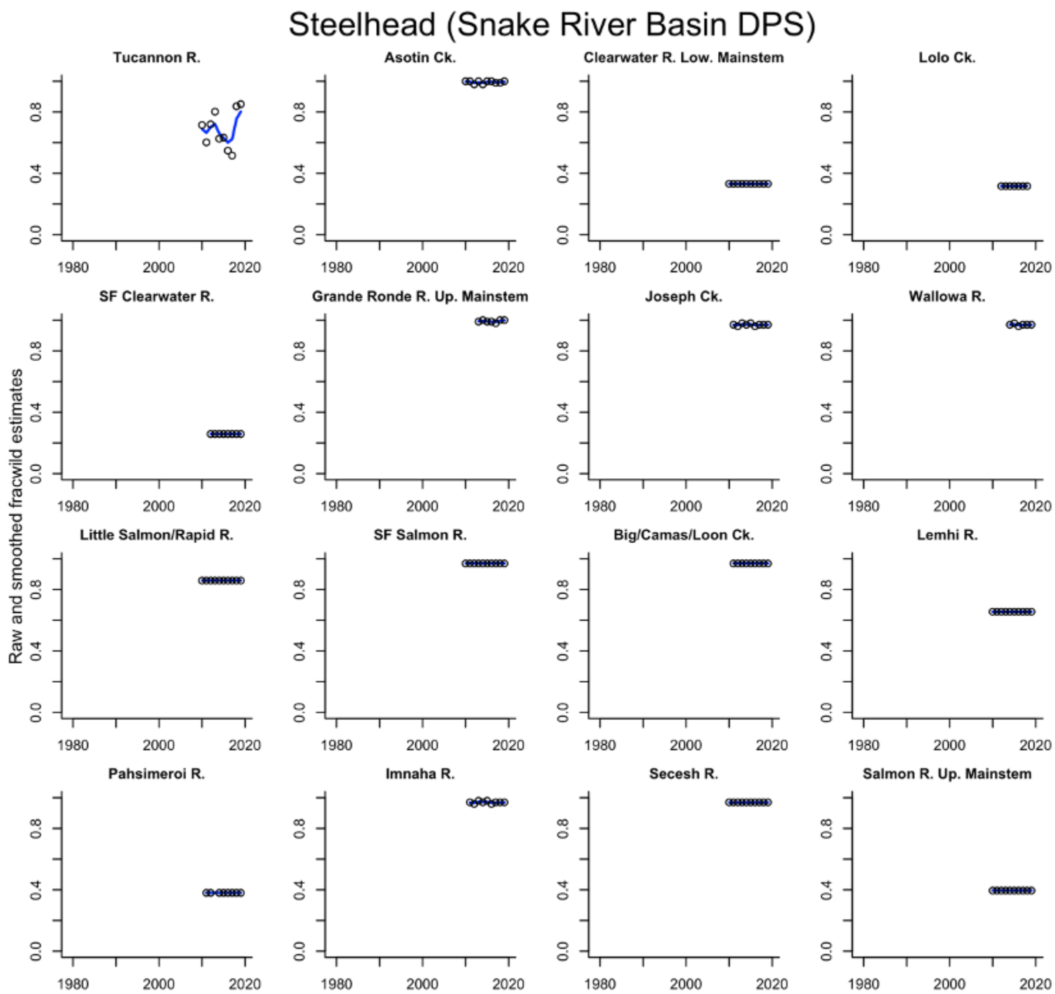

Figure 44 (continued). Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. This page: PIT-tag-based population estimation method based on mixture model and tag detection network across the DPS.

environment. An FWPI score of >100 should indicate baseline freshwater productivity (roughly 100 smolts per female). Clearwater River and Lower Snake River populations are performing at below-replacement levels of SAR (< 2%). The initial assessment by ICTRT (2007) identified no-to-low A/P gaps for this DPS. In general, the ICTRT assessment agrees with the long-term productivity metrics (SAR, R/S, FWPI), with the exception of some lower ocean survivals indicated for the Clearwater River and Lower Snake River populations.

Non-treaty harvest

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 predominately spend one year at sea, return to freshwater before the end of August, and are generally associated with low-to-mid-elevation streams throughout the interior Columbia River basin. B-run steelhead are larger, with most individuals returning after two years in the ocean and entering freshwater after August.

Both the age (size) and run-timing criteria are not absolute, with the A and B designations being the modes of two clearly overlapping distributions (Copeland et al. 2017). The A-run is believed to occur throughout the middle and upper Columbia River and the Snake River basins. The B-run is believed to occur naturally only in the Snake River Basin DPS, contributing, in varying proportions, to the Clearwater River, Middle Fork Salmon River, and South Fork Salmon River populations. The late return timing is most prevalent in the following populations: Lolo Creek, Lochsa and Selway Rivers, South Fork Clearwater River, South Fork Salmon and Secesh Rivers, and Lower and Middle Fork Salmon River. The size criteria (FL > 78 cm) also distinguishes the majority of returns to these populations, though to a lesser degree in the two Middle Fork populations (Bowersox et al. 2019).

Steelhead were historically taken in tribal and non-treaty gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and its tributaries. In the 1970s, retention of steelhead in non-treaty commercial fisheries was prohibited, and in the mid-1980s, tributary recreational fisheries adopted mark-selective regulations. Steelhead are still harvested in tribal fisheries, in mainstem recreational fisheries, and there is incidental mortality associated with mark-selective recreational fisheries. Incidental take on A-run and B-run Snake River Basin steelhead is less than the allowed 2%, with the W/SP/SU management block generally less than 1% and the FA management block exhibiting roughly a 1.5% rate (TAC 2020).

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 risk ratings for all of the Snake River 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.

ICTRT criteria for evaluating spatial structure within populations are based on observing evidence of spawning usage across defined spawning areas within populations, with an 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 due 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). Conducting 11,848 stream surveys in the Clearwater River and Salmon River MPGs (1997–2019), IDFG detected juvenile (age-1 parr) steelhead in 6,487 surveys representing 97 of the 112 spawning areas (major and minor) accessible by spawning adults. Based on this information, spatial structure ratings for Snake River Basin steelhead populations were maintained at the levels assigned in the original ICTRT assessment.

Biological viability relative to recovery goals

The ICTRT identified 24 extant populations within this DPS, organized into five major population groups (ICTRT 2003). They 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 MPGs with extant populations are Lower Snake River (two populations), Clearwater River (five extant populations, one extirpated), Grande Ronde River (four populations), Imnaha River (one population), and Salmon River (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. A DPS-wide recovery plan was completed in 2017, containing management unit plans 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 (NMFS 2017c).

The recovery criteria are hierarchical in nature, with DPS-level criteria being based on the viability of natural-origin steelhead assessed at the population level. 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 (McElhany et al. 2000). The ICTRT approach calls for comparing estimates of current natural-origin abundance (measured as a ten-year geometric mean of natural-origin spawners) and productivity (estimate of return per spawner at low-to-moderate parent spawning abundance) against pre-defined viability curves. In addition, the ICTRT developed a set of specific criteria (metrics and example risk thresholds) for assessing the SS/D risks based on current information representing each specific population.

Snake River Basin steelhead DPS: NOAA recovery plan scenario

The recovery plan 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). The remaining populations also must achieve at least “maintained” status. 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.

Lower Snake River MPG

Both populations (Tucannon River and Asotin Creek) in this MPG are targeted for "maintained" status, as an aggregate stock, with Asotin Creek meeting the criteria for "viable," based on the population-specific abundance and productivity data.

Population-level abundance datasets are not available for the entirety of either of the two populations in this MPG; however, a data series for a large subarea within the Asotin Creek population is available (Table 20). Based on recent-year PIT-tag detections and the Lower Granite Dam genetic stock composition monitoring, Asotin Creek is receiving substantial inputs of adult returns from the Tucannon River and potentially other areas (both natural- and hatchery-origin) in the lower Snake River region. 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 (Bumgarner and Dedloff 2015). One contributing factor is an apparent high overshoot rate of returning adults past their natal stream.

The ICTRT rated both populations at "moderate risk" for the integrated spatial structure and diversity criteria. This 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 and geologic/hydrologic setting. No additional or updated information is available for this review.

Clearwater River MPG

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

The GSI-based run reconstructions allow a partitioning of the MPG into three stocks: Clearwater River Lower Mainstem, South Fork Clearwater River, and the aggregate returns to the watersheds of the Upper and Middle Fork Clearwater River (including Lolo Creek and the Lochsa and Selway Rivers). The assignment of these stocks is reasonably strong, but the TRT populations do not map precisely to monophyletic clades (e.g., Lolo Creek falls within the South Fork Clearwater River clade, as does one tributary of Clearwater River Lower Mainstem). Nonetheless, the overall topology allows a robust A/P assessment based on the resultant run reconstruction. The Clearwater River Lower Mainstem and South Fork Clearwater River aggregate stocks are stable from the last viability review at a "very low risk" designation, both due to high long-term abundance and productivity. The SS/D evaluations remain unchanged from the last viability review ("low" and "moderate," respectively), resulting in "highly viable" and "viable" designations for these stocks. The aggregate Upper and Middle Fork Clearwater River stock, functionally consisting of the Lolo Creek and Lochsa and Selway River populations rates as "moderate risk" for abundance and productivity, with a relatively robust and stable abundance though declining productivity. Based on the previous spatial structure and diversity ratings, the overall risk for the aggregate stock is "maintained."

Based on the updated risk assessments, the Clearwater River MPG does meet the ICTRT criteria for a viable MPG. Although the population-specific PIT-tag-based population trajectories have yet to be run for a sufficiently long period, they will have been by the next viability review, and will be necessary to confirm the MPG status—especially considering the recent sharp downturns of the populations across the DPS.

Grande Ronde River MPG

Improvements in natural production are planned for all four populations in this MPG. Given their current viability, it is expected that Joseph Creek and the Grande Ronde River Upper Mainstem populations are the most likely to satisfy the MPG-level requirement for one "highly viable" and one "viable" population. The average abundance levels have decreased from the prior review period and the productivity remains high; though declining from the past review period, A/P risk ratings still fall in the "low" to "very low" region of the 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 River populations (Copeland et al. 2015a). While the relatively high misclassification rates associated with this group precluded developing reliable direct estimates of annual escapements for use in this review, the results indicate that the estimated returns to Joseph Creek and the upper Grande Ronde River would account for the majority of the aggregate Grande Ronde River run. The Wallowa River and Lower Grande Ronde populations are given a "high" A/P risk rating, reflecting the lack of population-specific data and the overall downward trends of populations in the DPS. More specific data on annual returns would be needed to assign updated specific abundance and productivity ratings to these two populations.

The combined spatial structure and diversity metric for all four populations in this MPG remains unchanged from the last review. The Grande Ronde River steelhead MPG is rated as "maintained" status. Both the Joseph Creek and Grande Ronde River Upper Mainstem populations meet the criteria for "viable," and the remaining two populations are provisionally rated as "high risk" based on the limited abundance and productivity data.

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 as "maintained" in the prior review, based on "moderate" A/P and SS/D ratings. The Imnaha River constitutes one of the stock groups identified in the Lower Granite Dam GSI program, allowing an extrapolated time series for this population to be generated based on the aggregate Lower Granite Dam returns. The projected population data indicate that the Imnaha River population is performing at a "very low" risk level over the recent past. 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 River 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 River 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. Additional information on the relative distribution of hatchery spawners could change the current diversity risk rating.

Salmon River MPG

This relatively large MPG includes 12 extant populations. The recovery plan identifies six populations to prioritize for “viable” status across this MPG. The recovery scenario is consistent with the ICTRT recommendations and includes two middle fork populations, the South Fork Salmon 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 run timing, and achieving a distribution of viable populations across the geographical extent of the MPG—specifically, that beyond the priority populations, all remaining populations would be maintained (<25% risk) with sufficient abundance, productivity, spatial structure, and diversity to provide for ecological functions and to preserve options for species recovery.

Estimates of natural-origin abundance based on GSI and run reconstruction from the aggregate Lower Granite Dam returns are available for four population subgroups within this MPG: Lower Salmon River (one population), the South Fork stock group (two populations), the Middle Fork stock group (three populations), and the Upper Salmon River stock group (six populations). These groupings and the resultant run reconstructions provide robust information to make viability assessment assignments, but including the population-specific PIT-tag-based population trajectories, when they have been run for a sufficiently long period, will be critical to confirm any population, MPG, and DPS risk designations.

In prior reviews, the three Middle Fork Salmon River and the two South Fork Salmon River populations were each assigned “moderate” A/P risk ratings based on the aggregate abundance time series. Based on the updated genetic stock composition run reconstruction returns and productivity, these two stock groups should remain at a “moderate” demographic risk category.

The Little Salmon River population is identified as a distinct single-population group within the current GSI mixture analyses. The recent ten-year geometric mean natural-origin returns at Lower Granite Dam allocated to this stock group, and the productivity based on the run reconstruction, indicate that this population is at “very low” demographic risk. However, the potential for hatchery spawner contributions into natural areas is high, therefore, the resultant productivity for this population based on adult recruit to total spawner estimates should be further evaluated.

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). While the population delineation within the group does not align precisely with the TRT designations, the MPG is monophyletic and distinct from other Salmon River stock groups. More recent information on Panther Creek shows that this population also clusters with the upper Salmon River populations (Vu et al. 2015). The aggregate A/P risk is "moderate" for this stock group, based on low long-term abundance but relatively high productivity. 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.

Updated biological risk summary

Based on the updated viability information available for this review, all five MPGs are not meeting the specific objectives in the draft recovery plan, and the viability of many individual populations remains uncertain (Table 23). The Clearwater River and Grande Ronde River MPGs are rated as "maintained," but more specific data on spawning abundance and the relative contribution of hatchery spawners for the Lower Grande Ronde and Wallowa River populations would improve future assessments, as would population-specific demographics in the upper Clearwater River stock group. The additional monitoring programs instituted in the early 2000s to gain better information on natural-origin abundance and related factors have significantly improved our ability to assess viability 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. However, 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 viability review indicates that the Snake River Basin steelhead DPS remains at "moderate" risk of extinction, with viability largely unchanged from the prior review.

Of particular note, the updated, population-level abundance estimates have made very clear the recent (last five years) sharp declines that are extremely worrisome, were they to continue. The viability metrics used in these analyses (standardized PNW-wide and ICTRT) are intentionally based on long time periods (10–20-year geometric means), to buffer against the rapid swings in abundance that salmon and steelhead populations are known to exhibit. While these filtering approaches intentionally result in muted responses to rapid abundance change, they also can lag in raising concerns about dramatic change in population status. Rapid response metrics, or metrics that are more keyed to system-wide synchronous behavior of population productivity, may be appropriate in these situations.

Table 23. Summary of viability relative to the ICTRT viability criteria, grouped by MPG. Natural spawning = most-recent 10-yr geometric mean (range). ICTRT productivity = 20-yr geometric mean for parent escapements below 75% of population threshold. Current A/P estimates are geometric means. Range in annual abundance, standard error, and number of qualifying estimates for productivities in parentheses. Populations with no abundance and productivity data are given a default "high" A/P risk rating.

	Abundance/productivity (A/P) metrics				Spatial structure/diversity (SS/D) metrics			
Population	**ICTRT threshold**	**Natural spawning**	**ICTRT productivity**	**Integrated A/P risk**	**Natural processes**	**Diversity risk**	**Integrated SS/D risk**	**Overall risk rating**
Tucannon River	1,000	n/a	n/a	High	Low	Moderate	Moderate	High
Lower Snake River (Tucannon River and Asotin Creek)	1,500	750 (SD 751)	2.52 (0.21, 12/20)	Moderate	Low	Moderate	Moderate	Maintained
Asotin Creek	500	574 (SD 389)	1.63 (0.41, 3/20)	Low	Low	Moderate	Moderate	Viable
Lower Grande Ronde River	1,000	n/a	n/a	High	Low	Moderate	Moderate	High
Joseph Creek	500	2,327 (SD 1,291)	1.21 (0.14, 0/20)	Low	Very Low	Low	Low	Viable
Grande Ronde River Upper Mainstem	1,500	2,192 (SD 1,227)	2.01 (0.35, 6/20)	Very Low	Very Low	Moderate	Moderate	Viable
Wallowa River	1,000	n/a	n/a	High	Very Low	Low	Low	High
Imnaha River	1,000	1,811 (SD 1,151)	2.36 (0.21, 9/20)	Very Low	Very Low	Moderate	Moderate	Viable
Clearwater River Lower Mainstem	1,500	2,026 (SD 1,382)	2.32 (0.18, 9/20)	Very Low	Very Low	Low	Low	Highly Viable
South Fork Clearwater River	1,000	1,564 (SD 1,275)	2.80 (0.23, 8/20)	Very Low	Low	Moderate	Moderate	Viable
Lolo Creek	500	1,946 (SD 1,426)	1.82 (0.19, 15/20)	Moderate	Low	Moderate	Moderate	Maintained
Selway River	1,000			Moderate	Very Low	Low	Low	Maintained
Lochsa River	1,000			Moderate	Very Low	Low	Low	Maintained
Little Salmon River	500	750 (SD 751)	2.53 (0.21, 12/20)	Very Low	Low	Moderate	Moderate	Viable
South Fork Salmon River	1,000	919 (SD 816)	1.85 (0.19, 15/20)	Moderate	Very Low	Low	Low	Maintained
Secesh River	500			Moderate	Low	Low	Low	Maintained
Chamberlain Creek	500	1,937 (SD 1,566)	2.47 (0.15, 10/20)	Moderate	Low	Low	Low	Maintained
Lower Middle Fork Salmon River	1,000			Moderate	Very Low	Low	Low	Maintained
Upper Middle Fork Salmon River	1,000			Moderate	Very Low	Low	Low	Maintained
Panther Creek	500	3,502 (SD 2,562)	1.88 (0.17, 16/20)	Moderate	High	Moderate	High	High
North Fork Salmon River	500			Moderate	Low	Moderate	Moderate	Maintained
Lemhi River	1,000			Moderate	Low	Moderate	Moderate	Maintained
Pahsimeroi River	1,000			Moderate	Moderate	Moderate	Moderate	Maintained
East Fork Salmon River	1,000			Moderate	Very Low	Moderate	Moderate	Maintained
Salmon River Upper Mainstem	1,000			Moderate	Very Low	Moderate	Moderate	Maintained

Middle Columbia River Steelhead DPS

Brief description of DPS

The Middle Columbia River steelhead DPS includes all naturally spawning populations of steelhead (*O. mykiss*) spawning in 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 (USOFR 2020; Figure 45). NMFS listed the Middle Columbia River steelhead DPS as threatened in 1999, with that listing designation being affirmed in 2006, 2012, and 2016.

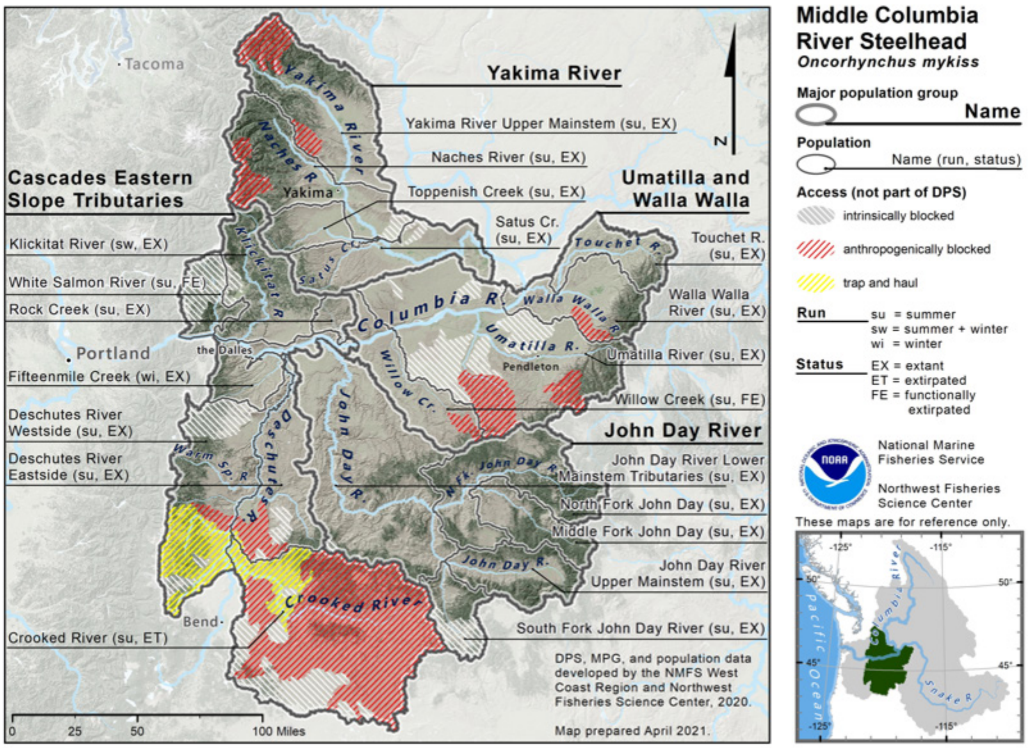

Figure 45. Map of the Middle Columbia River steelhead DPS's spawning and rearing areas, illustrating populations and major population groups.

NMFS has defined DPSes of steelhead to include only the anadromous members of this species (USOFR 2005b). Our approach to assessing the current viability of a steelhead DPS is based on evaluating information on the abundance, productivity, spatial structure, and diversity of the anadromous component of this species (Good et al. 2005, USOFR 2005b). Many steelhead populations along the U.S. West Coast 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 DPSes (Good et al. 2005, USOFR 2005b). 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 viability of the anadromous form.

Summary of previous viability conclusions

2005

Results of a BRT review of the viability of the Middle Columbia River 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 River 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 steelhead are believed to be very common throughout this DPS. The BRT assumed that the presence of resident steelhead 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 Middle Columbia River steelhead DPS was not currently meeting the viability criteria in the 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.

2015

For the 2015 review, there were improvements in the viability ratings for some of the component populations, but the Middle Columbia River steelhead DPS was not meeting the viability criteria described in the recovery plan (NWFSC 2015). In addition, several of the factors cited by the 2010 BRT remained as concerns or key uncertainties. Natural-origin returns to the majority of populations in two of the four MPGs in this DPS had increased modestly relative to the levels reported in the previous five-year review. Abundance estimates for two of three populations with sufficient data in the remaining two MPGs (Cascades Eastern Slope Tributaries and Umatilla/Walla Walla) were marginally lower. Natural-origin spawning estimates were highly variable relative to minimum abundance thresholds across the populations in the DPS. Three of the four MPGs in this DPS included at least one population rated at "low risk" for abundance and productivity. In general, the majority of population-level viability ratings remained unchanged from prior reviews for each MPG within the DPS.

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 17 long-term data series considered in prior status reviews. In addition, data are now available for the habitat in the White Salmon River recently made accessible by the removal of Condit Dam. Spawning surveys conducted by the Yakima/Klickitat Fisheries Project (YKFP) for return years 2012–19 show low numbers, but consistent use of the watershed by spawning steelhead (Zendt 2020). The consistent occupancy of previously extirpated populations (e.g., White Salmon River) and the documentation of use in other, previously marginalized populations (e.g., Rock Creek) warrant further evaluation with respect to these populations' potential role in the DPS recovery strategy.

Abundance estimates for the Yakima MPG populations continue to be based on steelhead counts at Prosser Dam, on the mainstem Yakima River—below all four of the populations in this MPG. Population-specific abundance estimates 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 Creeks. Population-specific estimates of the 2012–14 broodyear escapements were generated from a three-year radio-tagging study. In addition, two other methods were applied over the duration of that study, a GSI approach and a PIT-tag-based tracking program. Preliminary results suggest that the PIT-tag-based approach, which involves proportional tagging at Prosser Dam combined with strategically placed upstream arrays, would be a viable long-term strategy, as has been demonstrated in the Upper Columbia River and Snake River Basin steelhead DPSes. The continued expansion of the network of PIT-tag detection sites within the Yakima River basin is the necessary infrastructure for a robust adult, and potentially juvenile, monitoring program to address many key questions of abundance and productivity, as well as spatial structure and diversity.

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, Washington, 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 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 populations along the U.S. West Coast co-occur with conspecific populations of resident rainbow trout. Previous NWFSC status reviews (e.g., Ford et al. 2011, NWFSC 2015) 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, USOFR 2005b). 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 viability 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 the Middle Columbia River steelhead 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 and Reeves 2000). A recent study of natural-origin steelhead kelts in the Yakima River basin, comparing isotope 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). Satus Creek had the lowest sampled proportions of maternal resident patterns (<8% of samples in 2011 and 2012). The highest proportions were for fish that were assigned to the lower Yakima River basin (38% and 17%). Toppenish Creek and the 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. Despite the documented contribution of resident *O. mykiss* to the anadromous populations across the Middle Columbia River steelhead DPS, there is no evidence that these contributions alone could fully support sufficient productivity to make the populations viable.

Population, MPG, and DPS structure

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. In 2013, 1,325 PIT-tagged fish produced in the John Day River basin were detected passing above Bonneville Dam; 13% of those tagged fish directly migrated into the John Day River based on detections at lower mainstem John Day River arrays. A relatively high proportion (71%) of the adults detected at Bonneville Dam continued upriver past the John Day Dam 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 Dam eventually entered the natal basin. Recent rates of John Day River adult “overshoot” are similar: 44/68 or 65% in 2018, 70/113 or 62% in 2019, and 42/72 or 58% in 2020. High rates of overshooting were also indicated for some other Middle Columbia River 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 (Richins and Skalski 2018).

Genetic analyses of juvenile *O. mykiss* sampled in the Rock Creek drainage indicate a relatively high similarity to the Snake River Basin 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 viability. Matala (2012) also suggests that analysis of archival samples would provide insights into whether historical genetic patterns for this and other Middle Columbia River populations also reflect high exchange rates with the Snake River Basin 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, 2014b, Bare et al. 2015). Spawner abundance estimates generated or extrapolated from spatially balanced 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 John Day River 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 River natural spawning areas have declined substantially in recent years, with the declines being negatively correlated with the proportion of Snake River outmigrants that are barge transported (Banks et al. 2013, Bare et al. 2015). As in prior years, hatchery-origin spawners were concentrated in the lower John Day MPG tributaries.

Genetic sampling data from specific reaches in the John Day River 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 rainbow trout production. There is evidence of cutthroat/steelhead hybridization in Belshaw and Reynolds Creeks that could be contributing to their relative genetic distinctiveness.

Fifteenmile Creek life-history patterns

Fifteenmile Creek is one of two extant natural-origin populations at the western edge of the Middle Columbia River steelhead DPS. Steelhead runs in the downstream neighboring DPS (Lower Columbia River) are generally winter-run. ODFW had classified the Fifteenmile Creek population as winter-run prior to recent PIT-tag studies. Returning natural-origin steelhead PIT-tagged as juveniles in the mainstem Fifteenmile Creek watershed exhibit a summer-timed return pattern, similar to other populations in the Middle Columbia River steelhead DPS (Poxon et al. 2014). The Fifteenmile Creek population includes some smaller tributaries downstream of the Fifteenmile Creek drainage. It is possible that 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 and recruit-per-spawner rates

Smolt-to-adult return survival estimates (SARs) and recruits per spawner (R/S) are available through StreamNet's Coordinated Assessment Partnership data portal[10] (StreamNet 2020) and the Columbia River Data Access in Real Time (CBR and Washington 2020) project using PIT-tag detections from all release locations within each population basin (Columbia River DART et al. 2020). The metrics are based on mainstem hydrosystem tag detections, most commonly Bonneville-to-Bonneville for Middle and Upper Columbia River popualtions. The indices represent cumulative marine, nearshore, and estuary survival (SAR, expressed as a percentage of the smolts returning as adults) and whole life-cycle survival (R/S, expressed as a ratio of adults returning relative to their parents as spawners). SAR and R/S metrics are available for Fifteenmile Creek, Umatilla River, and the aggregate of the five John Day River populations. In general, these broad-brush survival metrics indicate relatively robust population processes for the select stocks, though the low R/S value for Umatilla River and the low SAR value for Fifteenmile Creek could be indicative of ecological limitations to population productivity (e.g., an SAR of 2% is accepted as the minimum rate for a population to be replacing itself).

Ocean condition indices

Juvenile steelhead are more pelagic than salmon, heading off the continental shelf soon after entering the ocean in the spring (Burgner et al. 1992). Steelhead migrate seasonally across the North Pacific Ocean, moving to the north and west in spring and to the south and east, across the entire Pacific, from autumn through winter (Atcheson et al. 2012). Thus, steelhead ocean survival may be impacted by different factors than salmon. In fact, recent work has shown that steelhead population groupings from geographic regions have unique smolt survival trends that appear to be driven by factors affecting them early in their ocean residence, despite steelhead smolts generally a) being larger than Pacific salmon smolts when they enter the ocean, and b) making wide-ranging, off-the-continental-shelf migrations, rather than remaining more coastal, as Pacific salmon smolts tend to do (Kendal et al. 2017).

[10] https://www.streamnet.org/cap/

Aggregate annual returns of Columbia River spring-run Chinook salmon 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). Work is ongoing to relate indices of ocean condition to steelhead populations up and down the U.S. West Coast. Steelhead marine survival seems to be related to ocean surface temperature in the first summer of ocean entry, and populations respond similarly to spatial patterns of ocean conditions at a rough grain of 250 km between ocean entry points (Kendal et al. 2017). Therefore, broad spatial patterns of ocean conditions may not capture the finer spatial scale of response that steelhead seem to exhibit.

Indicators of ocean condition are highly correlated with each other, and exhibit strong temporal autocorrelation (Figure 129; Peterson et al. 2019). As a result, when indicators point to conditions that result in poor ocean productivity for salmonid populations, they do so as a suite of indicators, and for runs of "good" or "bad" years (see Habitat chapter). Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019a). Recent (2015–19) ensemble ocean indicators rankings include four of the worst seven years in the past 20, meaning that an entire salmon or steelhead generation could have been subjected to poor ocean productivity conditions.

Abundance and productivity

Evaluations were done using both a set of metrics corresponding to those used in prior viability reviews as well as a set corresponding to the specific viability criteria based on ICTRT recommendations for this ESU. The viability review level metrics were consistently done across all ESUs and DPSes to facilitate comparisons across domains. Assessments using the ICTRT metrics are described in the recovery evaluation section below.

Abundance data series are available for all five extant populations in the Cascades Eastern Slope Tributaries MPG. Spawner abundance estimates for the most recent five years decreased relative to the prior review for all five populations (Table 24). The 15-year trend in natural-origin spawners was strongly negative for the Deschutes River Eastside population, and essentially zero for the Fifteenmile Creek and Deschutes River Westside runs (Table 25). 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).

Total escapement and natural-origin escapements declined relative to the prior five-year review for all five of the John Day MPG populations (Table 24). Only two of the five populations in this group had a positive 15-year trend in natural-origin abundance (Table 25), driven largely by peak returns in the early 2000s, despite the strong declines over the most recent five-year period (Figure 46).

Five-year geometric mean natural-origin and total abundance estimates for each of the four populations in the Yakima MPG also decreased sharply relative to the prior review (Table 24). All four populations in this group have exhibited increases since the early 1990s, with similar peak return years as other DPS populations, but, given recent declines, the 15-year trend for all populations was essentially zero (Figure 46, Table 24).

Table 24. Five-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 2 most-recent 5-year periods is shown on the far right.

Population	**MPG**	**1990–94**	**1995–99**	**2000–04**	**2005–09**	**2010–14**	**2015–19**	**% change**
Klickitat River	Cascades Eastern Slope Tributaries	—	—	—	1,622 (1,622)	1,358 (1,358)	1,573 (1,573)	16 (16)
Fifteenmile Creek	Cascades Eastern Slope Tributaries	(405)	(396)	(941)	(264)	430 (497)	278 (289)	–35 (–42)
Deschutes River Westside	Cascades Eastern Slope Tributaries	249 (324)	226 (341)	742 (951)	478 (579)	731 (781)	415 (432)	–43 (–45)
Deschutes River Eastside	Cascades Eastern Slope Tributaries	618 (773)	693 (1,440)	3,823 (4,849)	1,419 (1,712)	963 (1,123)	292 (340)	–70 (–70)
Rock Creek	Cascades Eastern Slope Tributaries	—	—	—	113 (113)	396 (396)	266 (266)	–33 (–33)
John Day River Lower Mainstem Tributaries	John Day River	1,021 (1,248)	968 (1,017)	3,479 (4,052)	1,024 (1,382)	2,017 (2,300)	1,006 (1,038)	–50 (–55)
North Fork John Day River	John Day River	1,248 (793)	1,142 (1,200)	2,247 (2,514)	1,488 (1,618)	2,822 (2,879)	910 (914)	–68 (–68)
Middle Fork John Day River	John Day River	1,306 (1,225)	545 (572)	1,229 (1,375)	634 (689)	4,767 (4,864)	2,388 (2,397)	–50 (–51)
South Fork John Day River	John Day River	450 (402)	135 (142)	493 (551)	586 (637)	1,148 (1,171)	776 (779)	–32 (–33)
John Day River Upper Mainstem	John Day River	991 (1,029)	350 (368)	695 (777)	471 (512)	1,086 (1,108)	458 (460)	–58 (–58)
Satus Creek	Yakima River	339 (377)	266 (300)	641 (652)	806 (829)	1,575 (1,608)	650 (656)	–59 (–59)
Toppenish Creek	Yakima River	102 (113)	135 (153)	695 (706)	467 (481)	570 (583)	232 (235)	–59 (–60)
Naches River	Yakima River	281 (313)	260 (294)	854 (868)	823 (846)	1,879 (1,923)	913 (921)	–51 (–52)
Yakima River Upper Mainstem	Yakima River	54 (56)	49 (50)	145 (149)	155 (157)	389 (410)	329 (337)	–15 (–18)
Umatilla River	Umatilla/Walla Walla	1,070 (1,346)	925 (1,664)	2,355 (3,324)	1,946 (2,517)	3,101 (3,687)	2,451 (2,877)	–21 (–22)
Walla Walla River	Umatilla/Walla Walla	995 (995)	516 (522)	957 (997)	711 (733)	1,016 (1,035)	500 (583)	–51 (–44)
Touchet River	Umatilla/Walla Walla	392 (438)	343 (396)	357 (388)	337 (446)	397 (501)	162 (214)	–59 (–57)

Total spawning escapements have decreased in the most recent brood cycle for all three populations in the Umatilla/Walla Walla MPG as well (Table 24). The 15-year trend in natural-origin abundance was positive for the Umatilla River population and slightly negative for Touchet River (Table 25, Figure 46), though the trends are shallow. Population productivity was cyclical, with most populations following a similar pattern of growth and decline (Figure 47).

Using the R/S and SAR indicators by population, it is possible to generate an indicator of freshwater productivity (FWPI) as a ratio of R/S and SAR (Figure 48). This quantity can be thought of as an indicator of smolts per spawner, and thus, the overall population productivity in the freshwater environment. Broodyear 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. FWPI for the Middle Columbia River steelhead populations for which this indicator can be constructed indicate relatively moderate freshwater productivity, with the majority of the populations below 100, a conservative

Table 25. Fifteen-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. Blank cells in the table indicate insuffient data to calculate the trend metric.

Population	MPG	1990–2005	2004–2019
Klickitat River	Cascades Eastern Slope Tributaries	—	0.00 (–0.04, 0.03)
Fifteenmile Creek	Cascades Eastern Slope Tributaries	0.06 (0.01, 0.10)	–0.03 (–0.07, 0.01)
Deschutes River Westside	Cascades Eastern Slope Tributaries	0.08 (0.03, 0.13)	–0.04 (–0.09, 0.00)
Deschutes River Eastside	Cascades Eastern Slope Tributaries	0.10 (0.05, 0.15)	–0.15 (–0.19, –0.11)
Rock Creek	Cascades Eastern Slope Tributaries	—	—
John Day River Lower Mainstem Tributaries	John Day River	0.05 (0.00, 0.10)	–0.03 (–0.08, 0.02)
North Fork John Day River	John Day River	0.08 (0.04, 0.12)	–0.04 (–0.10, 0.01)
Middle Fork John Day River	John Day River	0.01 (–0.03, 0.06)	0.09 (0.02, 0.15)
South Fork John Day River	John Day River	0.04 (–0.02, 0.09)	0.04 (–0.01, 0.09)
John Day River Upper Mainstem	John Day River	–0.01 (–0.06, 0.04)	–0.01 (–0.07, 0.04)
Satus Creek	Yakima River	0.07 (0.02, 0.11)	–0.01 (–0.07, 0.04)
Toppenish Creek	Yakima River	0.14 (0.09, 0.19)	–0.05 (–0.10, –0.01)
Naches River	Yakima River	0.09 (0.05, 0.14)	0.01 (–0.04, 0.06)
Yakima River Upper Mainstem	Yakima River	0.09 (0.05, 0.14)	0.06 (0.01, 0.11)
Umatilla River	Umatilla/Walla Walla	0.07 (0.03, 0.11)	0.00 (–0.05, 0.04)
Walla Walla River	Umatilla/Walla Walla	0.00 (–0.04, 0.05)	–0.03 (–0.08, 0.01)
Touchet River	Umatilla/Walla Walla	0.02 (–0.01, 0.06)	–0.06 (–0.10, –0.01)

estimate of 100 smolts per spawner (Figure 48). The relatively high SAR estimates for the aggregate run to the John Day River, and moderate R/S rates, result in low estimates of freshwater productivity. Low freshwater productivity does point to areas of recovery action focus such as pre-spawn mortality and juvenile rearing habitat condition. Mainstem migratory impacts such as the SAR are based on Bonneville-to-Bonneville tag detections, and the R/S rates are based on spawning ground recruits. The initial assessment of abundance and productivity gaps for the Middle Columbia River steelhead populations indicated a diversity of conditions, but generally fewer than were found for the other interior Columbia River basin listed DPSes (ICTRT 2007). Nonetheless, long-term productivity metrics, where produced, indicate the potential for needed improvements to reduce demographic risk factors.

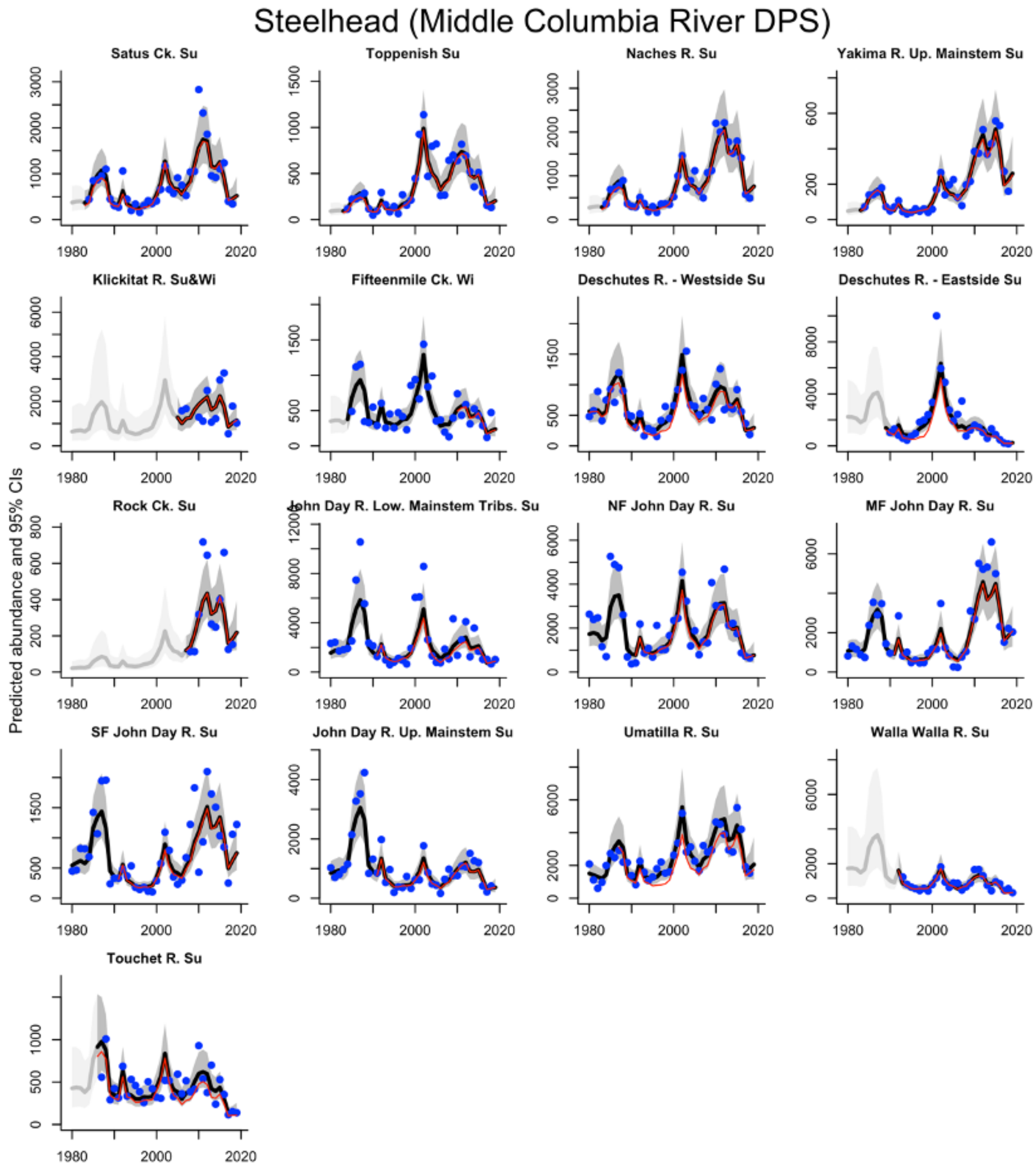

Figure 46. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

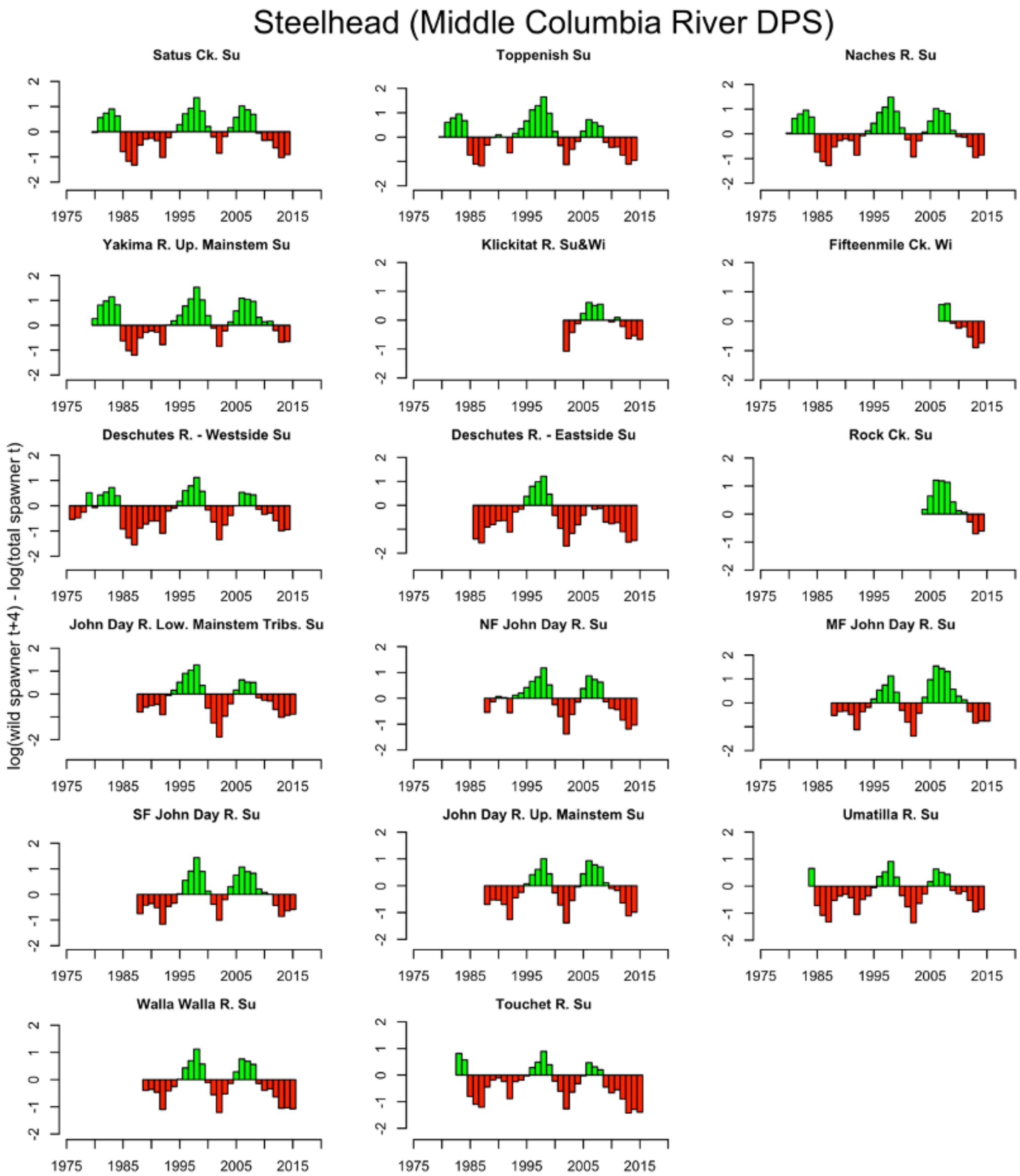

Figure 47. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year $(t-4)$. Spawning years on x-axis.

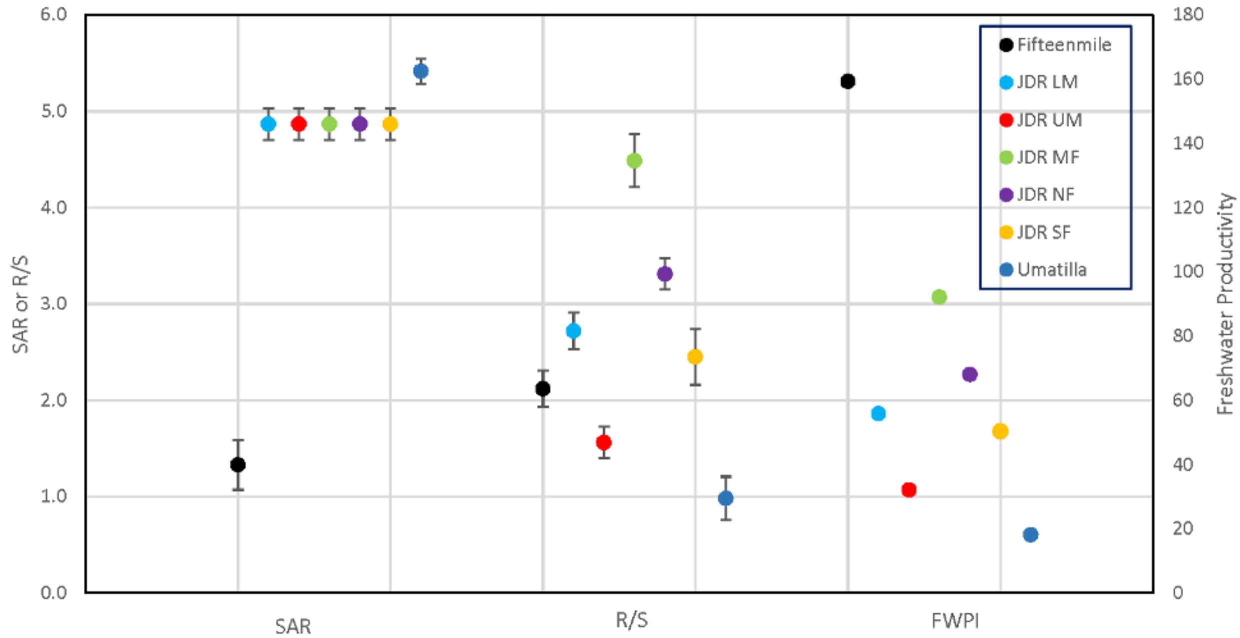

Figure 48. Smolt-to-adult return, recruits per spawner, and freshwater productivity index (FWPI) for each of the populations in the DPS. Geometric means of SAR and R/S are shown for each population, along with the standard error of the estimate (whiskers represent ±1 SE). The time period included in the SAR or R/S indices is the past 20 years, depending on data availability. The FWPI is constructed as a ratio of the geomean R/S and SAR, and can be thought of as a measure of smolts per spawner.

Non-treaty harvest

Steelhead were historically taken in tribal and non-tribal gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and its 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 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 middle Columbia River tributaries (Figure 49, lower panel).

Few winter-run fish migrate above Bonneville Dam, and winter-run steelhead are in the mainstem river at a time when there is generally little or no fishing occurring. The Klickitat River steelhead population within the Middle Columbia River steelhead DPS has a winter-run component, although anadromous production is dominated by summer-run timing. The ICTRT classified Fifteenmile Creek, another Middle Columbia River steelhead 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-treaty fishery impact rates for the natural component are in the range of 0.5% (Figure 49, upper panel).

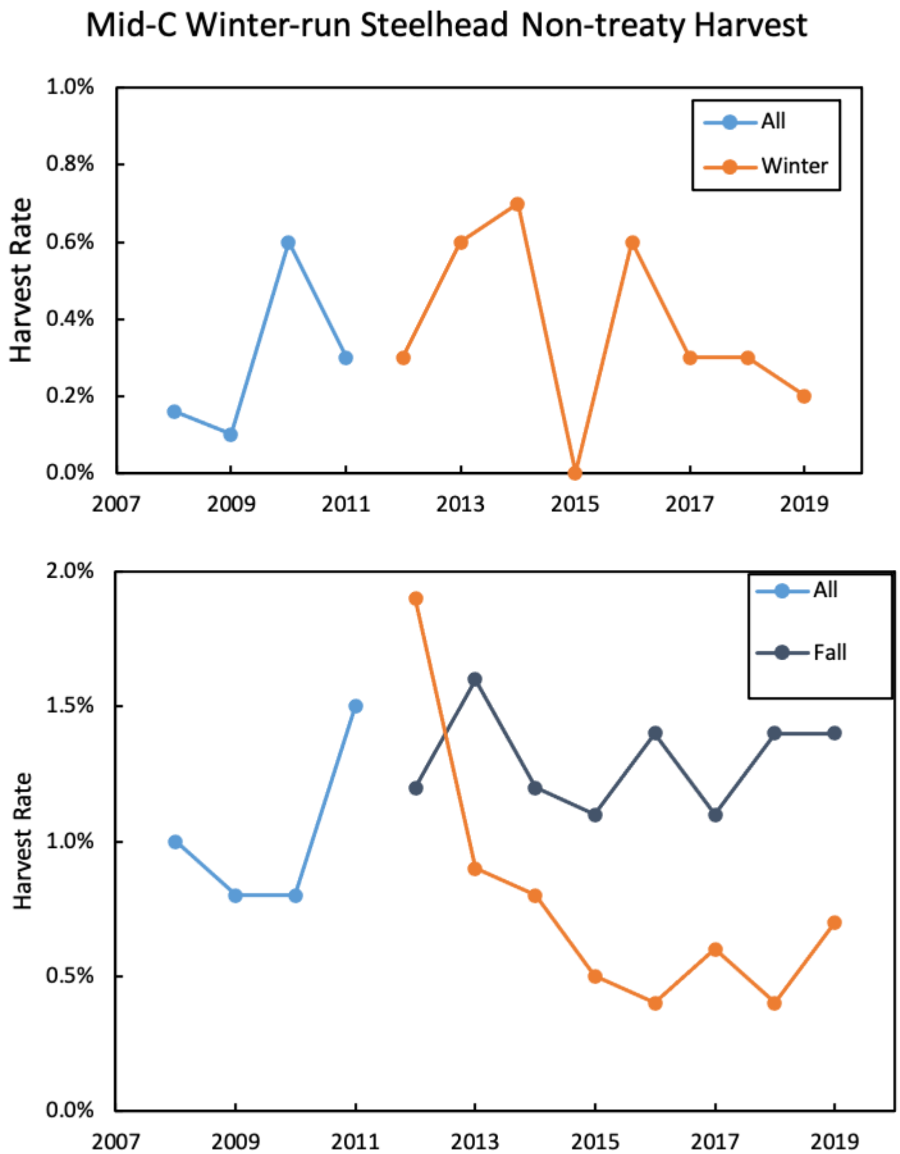

Figure 49. Non-treaty harvest impacts on natural winter- (upper panel) and summer-run (lower panel) steelhead from the Middle Columbia River steelhead DPS. As of 2012, harvest management reporting is broken into two periods, FA and W/SP/SU, where previously reporting was done by full calendar year (TAC 2020).

Spatial structure and diversity

Updated information on spawner and juvenile rearing distribution does not support a change in spatial structure status for Middle Columbia River steelhead DPS populations, though the newly re-established run in the White Salmon River and the developing time series of population data from the Klickitat River and Rock Creek do warrant consideration in the DPS recovery plan. Viability 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 Tributaries MPG, Fifteenmile 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 supports the "low risk" rating for spatial structure, and suggests that the current "moderate" rating for within-population diversity may improve as additional years of data accumulate. The current diversity risk rating of "moderate" was largely based on uncertainty about the effects of the ongoing hatchery program in the basin. Initial results indicate that the separation in time and space between hatchery- and natural-origin spawners has been effective in minimizing introgression. Indices for both spatial structure and diversity risk for the Deschutes River Westside population remain at "moderate risk." The spatial structure rating is due to the loss of natural production from above Pelton Dam/Round Butte, Oregon. The Deschutes River Eastside 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 degradation and potential genetic impacts resulting from chronic and widespread hatchery straying from out-of-basin stocks. The most recent five-year average proportion for natural spawners in the Deschutes River Westside population continues to increase (Table 26). Specific information on spawner distribution and composition for Rock Creek, the other extant population in this MPG, has become available since the prior review. Spawning in this historically small population appears to be dominated by out-of-basin, natural-origin strays.

Table 26. Five-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	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Klickitat River	Cascades Eastern Slope Tributaries	—	—	1.00	1.00	1.00
Fifteenmile Creek	Cascades Eastern Slope Tributaries	—	—	—	0.96	0.96
Deschutes River Westside	Cascades Eastern Slope Tributaries	0.67	0.78	0.83	0.94	0.96
Deschutes River Eastside	Cascades Eastern Slope Tributaries	0.51	0.79	0.84	0.86	0.86
Rock Creek	Cascades Eastern Slope Tributaries	—	—	1.00	1.00	1.00
John Day River Lower Mainstem Tributaries	John Day River	0.95	0.86	0.74	0.88	0.97
North Fork John Day River	John Day River	0.95	0.89	0.92	0.98	1.00
Middle Fork John Day River	John Day River	0.95	0.89	0.92	0.98	1.00
South Fork John Day River	John Day River	0.95	0.89	0.92	0.98	1.00
John Day River Upper Mainstem	John Day River	0.95	0.89	0.92	0.98	1.00
Satus Creek	Yakima River	0.89	0.98	0.97	0.98	1.00
Toppenish Creek	Yakima River	0.88	0.98	0.97	0.98	0.99
Naches River	Yakima River	0.89	0.98	0.97	0.98	1.00
Yakima River Upper Mainstem	Yakima River	0.98	0.97	0.99	0.95	0.99
Umatilla River	Umatilla/Walla Walla	0.56	0.71	0.77	0.84	0.85
Walla Walla River	Umatilla/Walla Walla	0.99	0.96	0.97	0.98	0.87
Touchet River	Umatilla/Walla Walla	0.87	0.92	0.76	0.79	0.76

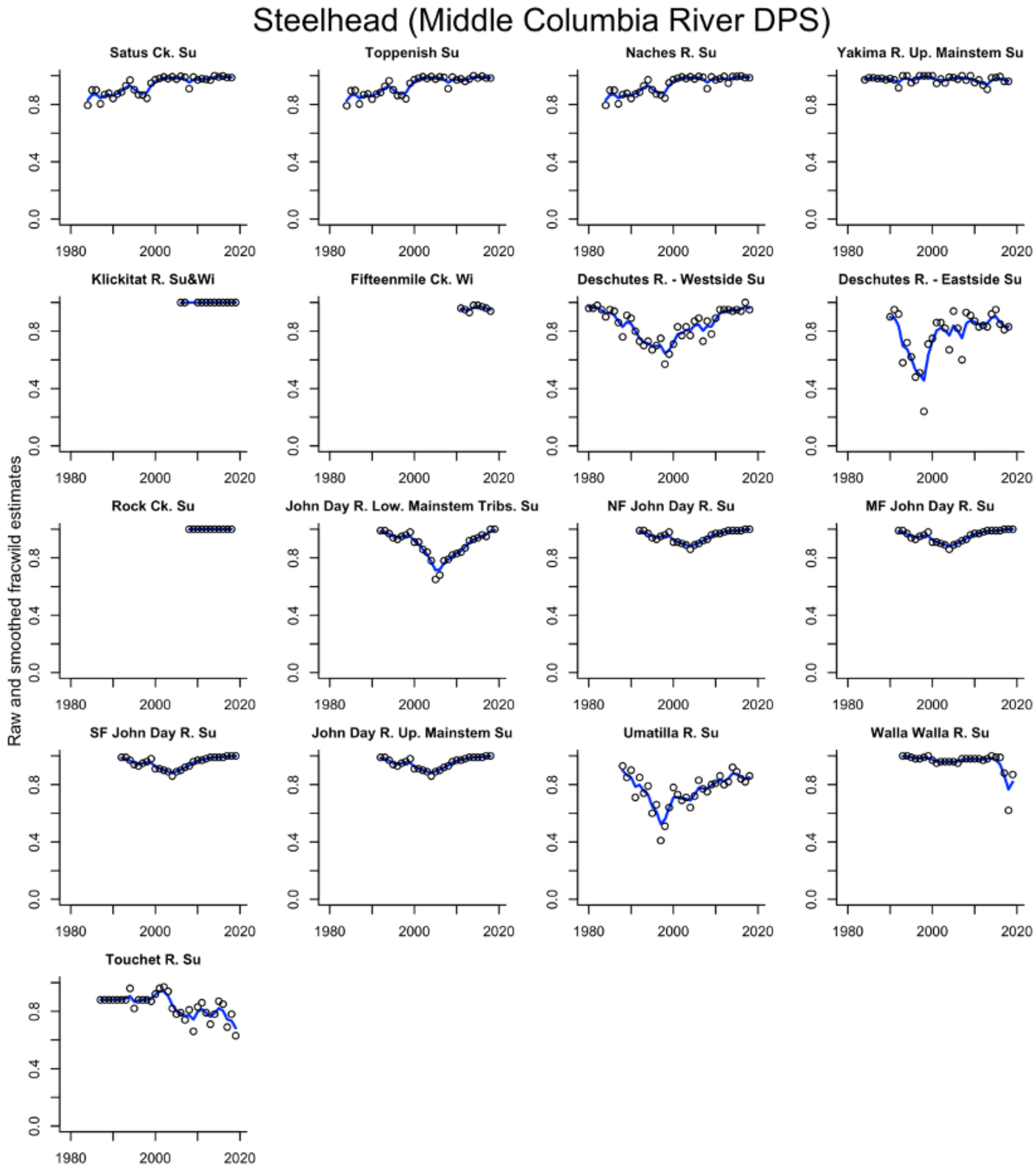

Figure 50. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

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. Total hatchery proportions have increased slightly from the prior review.

The spatial structure for all five populations in the John Day 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 populations have declined considerably in recent years (Figure 50).

Three of the four populations in the Yakima MPG remain at "low risk" for spatial structure impacts based on results from recent radio-tag and PIT-tag studies. Distribution across spawning areas within the fourth population, the Yakima River Upper Mainstem, 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 River basin) remain at prior levels. There are no within-basin hatchery steelhead releases in the Yakima River, and outside-source strays remain at low levels.

Biological viability relative to recovery goals

Recovery strategies outlined in the recovery plan (NMFS 2009a) and its management unit components are targeted on achieving, at a minimum, the ICTRT biological viability criteria requiring 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" (p. 3-2). The recovery plan recognizes that, at the MPG level, there may be several specific combinations of population viability ratings 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" MPG status. The recovery plan recognizes that the management unit plans incorporate a range of objectives that go beyond the minimum biological viability required for delisting.

The ICTRT recovery criteria are hierarchical in nature, with ESU/DPS-level criteria being based on the viability of natural-origin steelhead assessed at the population level (ICTRT 2007).

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 (McElhany et al. 2000). The ICTRT approach calls for comparing estimates of current natural-origin abundance (measured as a ten-year geometric mean of natural-origin spawners) and productivity (estimate of recruit 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 the particular risk threshold of a 5% risk of extinction over a 100-year period.

Middle Columbia River steelhead DPS: NOAA recovery plan scenario

The Middle Columbia River steelhead 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.

Cascades Eastern Slope Tributaries MPG

The Klickitat River, Fifteenmile Creek, and both the Deschutes River 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% risk level). MPG viability could be further bolstered if reintroduction of steelhead into the Crooked River succeeds and if the White Salmon River population successfully recolonizes its historical habitat following the upcoming removal of Condit Dam. The ICTRT originally classified the Fifteenmile Creek population as winter-run. Based on the recent information provided by ODFW described above, that designation should be provisionally changed to summer-run.

John Day River MPG

The John Day River Lower Mainstem Tributaries, North Fork John Day River, and either the Middle Fork John Day River or John Day River Upper Mainstem 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 Yakima River Upper Mainstem. 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. Umatilla River is the only large population, and therefore needs to be viable. In addition, either the Walla Walla River or Touchet River population also needs to be viable.

Overall viability ratings for the populations in the Middle Columbia River steelhead DPS remain generally unchanged from the prior five-year review (Table 27).

Updated biological risk summary

There has been functionally no change in the viability ratings for the component populations, and the Middle Columbia River steelhead DPS does not currently meet the viability criteria described in the Middle Columbia River steelhead recovery plan. In addition, several of the factors cited by the 2005 BRT remain as concerns or key uncertainties. While recent (five-year) returns are declining across all populations, the declines are from relatively high returns in the previous five-to-ten year interval, so the longer-term risk metrics that are meant to buffer against short-period changes in abundance and productivity remain unchanged. Natural-origin spawning estimates are highly variable relative to minimum abundance thresholds across the populations in the DPS. Two of the four MPGs in this DPS include at least one population rated at "low" or "very low" risk for abundance and

Table 27. Summary of Middle Columbia River steelhead DPS viability relative to the ICTRT viability criteria, grouped by MPG. Natural spawning = most-recent 10-yr geometric mean (range). ICTRT productivity = 20-yr geometric mean for parent escapements below 75% of population threshold. Current A/P estimates are geometric means. Range in annual abundance, standard error, and number of qualifying estimates for productivities in parentheses.

	Abundance/productivity (A/P) metrics				Spatial structure/diversity (SS/D) metrics			
Population	**ICTRT threshold**	**Natural spawning**	**ICTRT productivity**	**Integrated A/P risk**	**Natural processes**	**Diversity risk**	**Integrated SS/D risk**	**Overall risk rating**
Klickitat River	1,000	1,462 (SD 919)	1.07 (0.12 8/20)	Moderate	Low	Moderate	Moderate	Maintained
Fifteenmile Creek	500	378 (SD 170)	2.12 (0.19 8/20)	Moderate	Very Low	Low	Low	Maintained
Deschutes River Westside	1,500 (1,000)	538 (SD 306)	1.10 (0.15 18/20)	High	Low	Moderate	Moderate	High
Deschutes River Eastside	1,000	604 (SD 453)	1.75 (0.29 7/20)	Moderate	Low	Moderate	Moderate	Maintained
Rock Creek	500	298 (SD 232)	—	High	Moderate	Moderate	Moderate	High
Crooked River (extirpated)	2,000	—	—	—	—	—	—	Extirpated
White Salmon River (extirpated)	500	—	—	—	—	—	—	Extirpated (recolonizing)
John Day River Lower Mainstem Tributaries	2,250	1,424 (SD 1,026)	2.72 (0.19 12/20)	Moderate	Very Low	Moderate	Moderate	Maintained
North Fork John Day River	1,000	1,852 (SD 1,343)	3.31 (0.16 2/20)	Very Low	Very Low	Low	Low	Highly Viable
Middle Fork John Day River	1,000	3,371 (SD 1,811)	4.49 (0.27 8/20)	Very Low	Low	Moderate	Moderate	Viable
South Fork John Day River	500	943 (SD 552)	2.45 (0.29 10/20)	Very-Low	Very Low	Moderate	Moderate	Viable
John Day River Upper Mainstem	1,000	738 (SD 418)	1.56 (0.16 14/20)	Moderate	Very Low	Moderate	Moderate	Maintained
Satus Creek	1,000 (500)	1,064 (SD 777)	1.92 (0.30 3/20)	Low	Low	Moderate	Moderate	Viable
Toppenish Creek	500	407 (SD 231)	3.35 (0.23 9/20)	Moderate	Low	Moderate	Moderate	Maintained
Naches River	1,500	1,340 (SD 601)	2.00 (0.23 6/20)	Moderate	Low	Moderate	Moderate	Maintained
Yakima River Upper Mainstem	1,500	346 (SD 129)	1.73 (0.15 20/20)	Moderate	Moderate	High	High	High
Umatilla River	1,500	2,747 (SD 1,108)	0.98 (0.27 6/20)	Moderate	Moderate	Moderate	Moderate	Maintained
Walla Walla River	1,000	713 (SD 511)	1.79 (0.18 8/20)	Moderate	Moderate	Moderate	Moderate	Maintained
Touchet River	1,000	253 (SD 222)	0.91 (0.09 19/20)	High	Low	Moderate	Moderate	High

productivity, while the other two MPGs remain in the "moderate" to "high" risk range (Table 27). Updated information indicates that stray levels into the John Day River populations have decreased in recent years. Out-of-basin hatchery stray proportions, although reduced, remain high in spawning reaches within the Deschutes River basin and the Umatilla, Walla Walla, and Touchet River populations. Overall, the Middle Columbia River steelhead DPS remains at "moderate" risk of extinction, with viability unchanged from the prior review.

Lower Columbia River Domain Viability 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 51; USOFR 2020), 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—nine spring-run, 21 fall-run, and two late fall-run—organized in six MPGs based on run timing and ecological region.

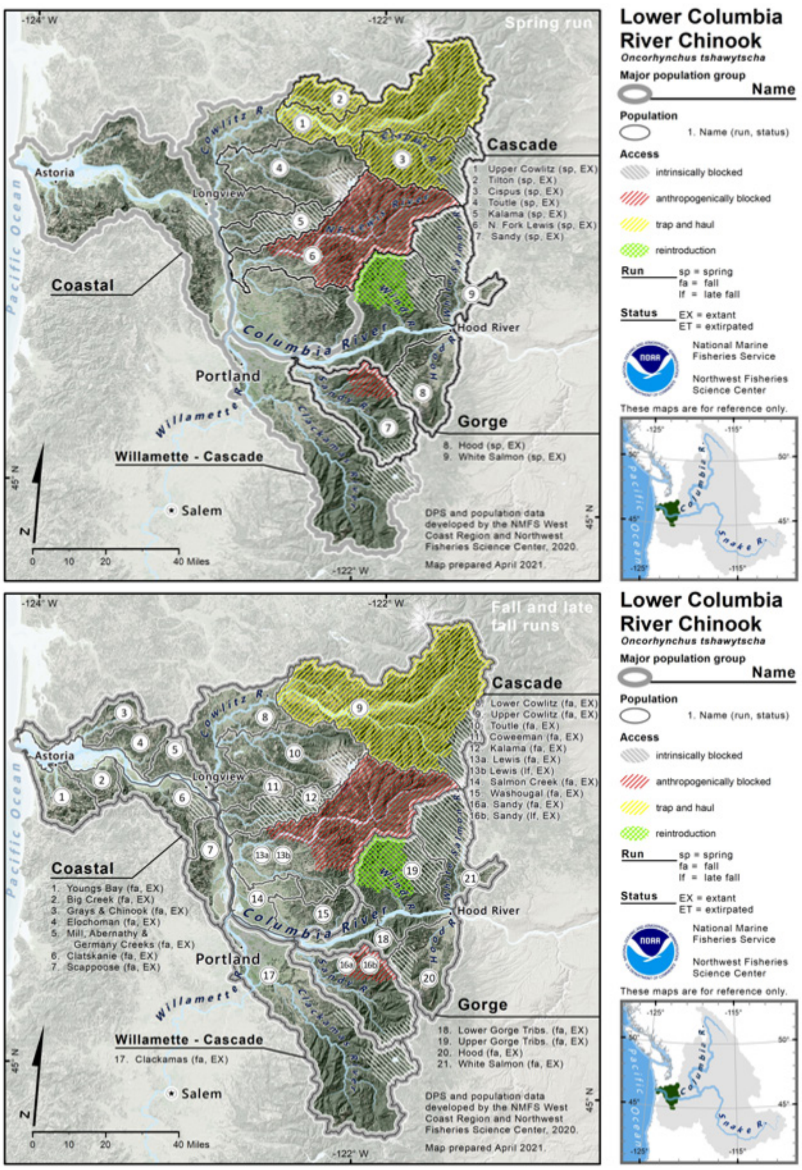

Figure 51. Maps of the Lower Columbia River Chinook salmon ESU's spawning and rearing areas, illustrating basins where demographically independent populations (DIPs) and major population groups (MPGs) are located. Several watersheds contain or historically contained both spring-run DIPs (top map) and fall and late-fall DIPs (bottom map). Areas that are accessible (green), accessible only via trap and haul programs (yellow), or blocked (cross-hatched) are indicated accordingly.

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 the risk factors identified in the original 1998 review. The Willamette–Lower Columbia River Technical Recovery Team (WLC-TRT) estimated that eight-to-ten historical populations in this ESU had been extirpated, the majority of them spring-run populations. The 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 experienced abundance increases in the years leading up to the 2005 status review, as had occurred in other regions.

2010

Ford et al. (2011) noted that three status evaluations of Lower Columbia River Chinook salmon, 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 (fall-run) populations were considered "very high risk" except one, which was considered "high risk." Later modeling conducted in association with tule harvest management suggested that three of the populations (Coweeman River, Lewis River, and Washougal River) 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 per 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 (historical) spawning habitat by dams. Trap-and-haul projects to allow access to historical spawning habitat had been initiated in the Cowlitz River and Lewis River systems, but in 2010 these were not effective enough to produce self-sustaining populations, primarily because of poor downstream juvenile collection. Dams were removed on the Sandy River and Hood River; however, these dams only impeded passage, they did not block it. At the time of the review, the benefits of these actions had not yet been expressed in adult returns. The Sandy River spring-run Chinook salmon population was considered at "moderate" risk and was the only spring-run Chinook salmon population not considered extirpated or nearly so. The Hood River population contained an out-of-ESU hatchery stock. The two late-fall populations, Lewis River and Sandy River, 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 River) 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 prior BRT status review in 2005.

2015

The NWFSC (2015) analysis of the biological risk status of the Lower Columbia River Chinook salmon ESU indicated little change since the prior status review, although there were 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 (Dornbusch 2013), there had been an overall improvement in the status of a number of fall-run populations, although most were still far from the recovery plan goals.

In the 2015 review, improved fall-run VSP scores reflected both changes in biological status and improved monitoring. Spring-run Chinook salmon populations in this ESU were generally unchanged; most populations were at “high” or “very high” 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 had an average of over 1,000 natural-origin spawners and was at “moderate” risk; this appeared partly to be a result of the removal of Marmot Dam in the Sandy River, which eliminated migrational delays and holding injuries that were occurring at the dam’s fish ladder. Further, the removal of a 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 in most cases the downstream juvenile collection efficiencies were 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 were transported into the Tilton River basin and it was not clear if there were any spring-run Chinook salmon remaining in the Toutle River basin. The removal of Condit Dam on the White Salmon River provided 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). Alternatively, spring-run Chinook salmon returning to the Hood River were largely the result of introductions of Deschutes River spring-run origin (the identified but not listed Middle Columbia River spring-run Chinook salmon ESU) and provide no benefit to the status of the ESU. However, some lower Columbia River-origin spring-run Chinook salmon had also been detected in the Hood River, and may have contributed to future native abundance in the river.

The majority of the populations in this ESU remained at “high” risk, with low natural-origin abundance levels. Hatchery contributions remained high for a number of populations, and it is likely that many returning unmarked adults were the progeny of hatchery-origin parents. Where large hatchery programs operated, it is also likely that mismarked hatchery-origin fish contributed to natural spawner counts. While overall hatchery production had been reduced slightly, hatchery-produced fish still represented a majority of fish returning to the ESU. The continued release of out-of-ESU stocks, including Upriver Bright, Rogue River (SAB) fall run, Upper Willamette River spring-run, Carson Hatchery spring-run, and Deschutes River spring-run, remained a concern. Relatively high harvest rates were a potential concern, especially for most spring-run and low abundance fall-run populations. Although there had been a number of notable efforts to restore migratory access to areas upstream of dams, until efforts to improve juvenile passage systems bear fruition, the review concluded that it was unlikely that there would 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 improved and/or provided access, but allowed the restoration of hydrological processes that may have improved downstream habitat conditions. Continued human population growth, land development, and habitat degradation, in combination with the potential effects of climate change, likely presented a continuous negative influence. In addition, coastal ocean conditions at the time of the 2015 review suggested that the recent outmigrant year classes would 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.

Description of new data available for this review

For the current evaluation, data were available for many populations through 2018 or 2019, with some of the datasets going back as far as 1968. This status review benefits from expanded spawner surveys begun after the 2010 review, especially in regard to abundance time series and hatchery contribution to the naturally spawning adults. Presently, there is some level of monitoring for all Chinook salmon populations except those that are functionally extinct (Rawding and Rodgers 2013). 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 and Sampling (OASIS) project. A generalized random tesselation stratified (GRTS-based) spawning ground survey has been conducted in the Coastal stratum by ODFW since 2012. Improvements in spawner census methodologies have unfortunately resulted in the need to "restart" some time series to ensure data compatibility. Methodologies include expansions of index reach redd counts, tributary weir counts, mark/recapture surveys, and hatchery trap, dam trap, and dam ladder counts. Mass marking of hatchery-reared Chinook salmon has become the norm, providing better information on natural-origin recruit (NOR) abundance (instead of the previous method of coded wire tag expansion), allowing mark-selective fisheries (reducing harvest impacts on natural-origin adults and reducing the number of hatchery-origin fish) and facilitating broodstock protocols in hatcheries and natural-origin spawner (NOS) selection at weirs and other facilities. Data time series are available for most populations, although 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 Rivers (2 DIPs combined), Kalama River, North Fork Lewis River, and Sandy River populations. Of these, only the Sandy River population appears to be sustaining natural-origin abundance at near-recovery levels. The most-recent five-year geomean abundance for the Sandy River was 3,359, which represents an 89% increase over 2010–14 (Table 28). The removal of Marmot Dam on the Sandy River in 2007, in conjunction with efforts to reduce the contribution of hatchery-origin fish, has facilitated the improved abundance of spring-run Chinook salmon in that basin, an impressive result given the poor ocean conditions experienced during this last review period. All of the spring-run populations except Sandy River exhibited a recent uniform decline, possibly related to climatic and oceanic conditions (Tables 28 and 29, Figure 52). Elsewhere in this MPG natural-origin abundances for spring-run Chinook salmon were very low, with negative trends. For the Upper Cowlitz/Cispus Rivers, Kalama River, and North Fork Lewis River populations, hatchery returns currently constitute the vast majority of fish returning to the river. In the Upper Cowlitz River, hatchery-origin fish are transported around the dams, whereas in the Kalama and Lewis Rivers, hatchery fish are intercepted at Lower Kalama River Falls and Merwin Dam, respectively. Current programs on the Cowlitz and Lewis Rivers to pass returning adults above and collect and transport migrating juveniles downstream around high-head dams have not attained sufficient efficiencies for the populations to sustain themselves, although considerable progress has been made in recent years (Rubenson et al. 2019, PacifiCorp 2020). Reintroduction efforts have not yet begun to reestablish spring-run Chinook salmon in the Tilton River DIP. WDFW does not recognize the continued existence of the Toutle River spring-run DIP, and adult spawner surveys are not undertaken (WDFW et al. 1993). The Kalama River spring-run hatchery program is run as a segregated program, and returning hatchery-origin adults are excluded from upriver spawning habitat; however, the Kalama River natural-origin spring-run abundance continues to be critically low, with strongly negative long- and short-term trends (Table 28). The spring run in the North Fork Lewis River includes fish naturally spawning below Merwin Dam and fish returning to the Merwin Trap for transportation above Swift Dam (the uppermost dam). In summary: in this MPG, only the Sandy River Chinook salmon DIP has attained moderate abundance levels; three other DIPs 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 historical spring-run DIPs in this MPG are likely at extremely low abundances (Table 28). 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 and spawner surveys have been limited (LCFRB 2020). Native spring-run Chinook salmon in the Hood River declined to critically low levels in the late 1980s and may have

Table 28. Five-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 2 most-recent 5-year periods is shown on the far right. SP = spring-run, FA = fall-run, LFR = late fall-run.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Upper Cowlitz/Cispus Rivers SP	Spring-run Cascade	—	—	—	—	—	171 (5,435)	—
Kalama River SP	Spring-run Cascade	(121)	(127)	(337)	57 (405)	82 (82)	43 (43)	–48 (–48)
North Fork Lewis River SP	Spring-run Cascade	(1,127)	(308)	(556)	(130)	(145)	(112)	(–23)
Sandy River SP	Spring-run Cascade	—	—	—	—	1,778 (2,000)	3,359 (3,667)	89 (83)
Big White Salmon River SP	Spring-run Gorge	—	—	—	—	18 (138)	8 (50)	–56 (–64)
Grays River Tule FA	Fall-run Coastal	(53)	(81)	(214)	83 (188)	79 (448)	228 (579)	189 (29)
Youngs Bay FA	Fall-run Coastal	—	—	—	—	201 (5,105)	145 (1,635)	–28 (–68)
Big Creek FA	Fall-run Coastal	—	—	—	—	0 (1,389)	0 (2,206)	(59)
Elochoman River/ Skamokawa Tule FA	Fall-run Coastal	(530)	(661)	(2771)	(778)	91 (612)	95 (238)	4 (–61)
Clatskanie River FA	Fall-run Coastal	—	—	27 (273)	13 (91)	8 (82)	3 (76)	–62 (–7)
Mill/Abernathy/Germany Creeks Tule FA	Fall-run Coastal	(1,160)	(602)	(2,416)	(727)	67 (688)	28 (151)	–58 (–78)
Lower Cowlitz River Tule FA	Fall-run Cascade	(2,492)	(1,827)	(5,818)	(2,367)	2,562 (3,711)	3,208 (4,161)	25 (12)
Coweeman River Tule FA	Fall-run Cascade	(877)	(796)	(805)	(526)	683 (840)	543 (595)	–20 (–29)
Toutle River Tule FA	Fall-run Cascade	(211)	(788)	(4,689)	(1,826)	330 (1,290)	280 (514)	–15 (–60)
Upper Cowlitz River Tule FA	Fall-run Cascade		(42)	(724)	(2,485)	2,646 (7,779)	1,761 (2,188)	–33 (–72)
Kalama River Tule FA	Fall-run Cascade	(2,714)	(4,192)	(6,911)	(6,156)	540 (7,529)	2,142 (3,808)	297 (–49)
Lewis River Tule FA	Fall-run Cascade	—	(1,423)	(3,487)	(1,599)	1,521 (2,256)	2,003 (3,637)	32 (61)
Clackamas River FA	Fall-run Cascade	—	—	—	—	144 (292)	236 (366)	64 (25)
Sandy River FA	Fall-run Cascade	—	—	—	—	(1,176)	(2,074)	(76)
Washougal River Tule FA	Fall-run Cascade	(2,932)	(3,227)	(4,391)	(2,355)	609 (2,486)	914 (1,643)	50 (–34)
Lower Gorge Tributaries Tule FA	Fall-run Gorge	—	(1,822)	(1,157)	(941)	928 (1,048)	4,528 (4,708)	388 (349)
Upper Gorge Tributaries Tule FA	Fall-run Gorge	—	(277)	(916)	(621)	561 (1,563)	537 (999)	–4 (–36)
Big White Salmon River Tule FA	Fall-run Gorge	(127)	(151)	(2,129)	(939)	759 (962)	283 (502)	–63 (–48)
Lewis River Bright LFR	Late fall-run Cascade	(8,353)	(6,647)	(11,694)	(5,758)	11,671 (11,671)	8,725 (8,725)	–25 (–25)
Sandy River Bright LFR	Late fall-run Cascade	852 (3,594)	815 (3,440)	555 (2,340)	1,097 (4,629)	—	—	—

been completely supplanted by introduced Deschutes River spring-run Chinook salmon, an out-of-ESU hatchery population. With the removal of Powerdale Dam, it has not been possible to estimate the abundance of returning adults with any certainty. Earlier reports of unmarked spring-run Chinook salmon returning to the Hood River (NWFSC 2015) may suggest the persistence of some native fish, but there is no verification of this. The last estimate of natural abundance, 18 adults, was in 2017. There is considerable uncertainty whether this MPG persists, and whether the low abundances observed represent native natural-origin abundances.

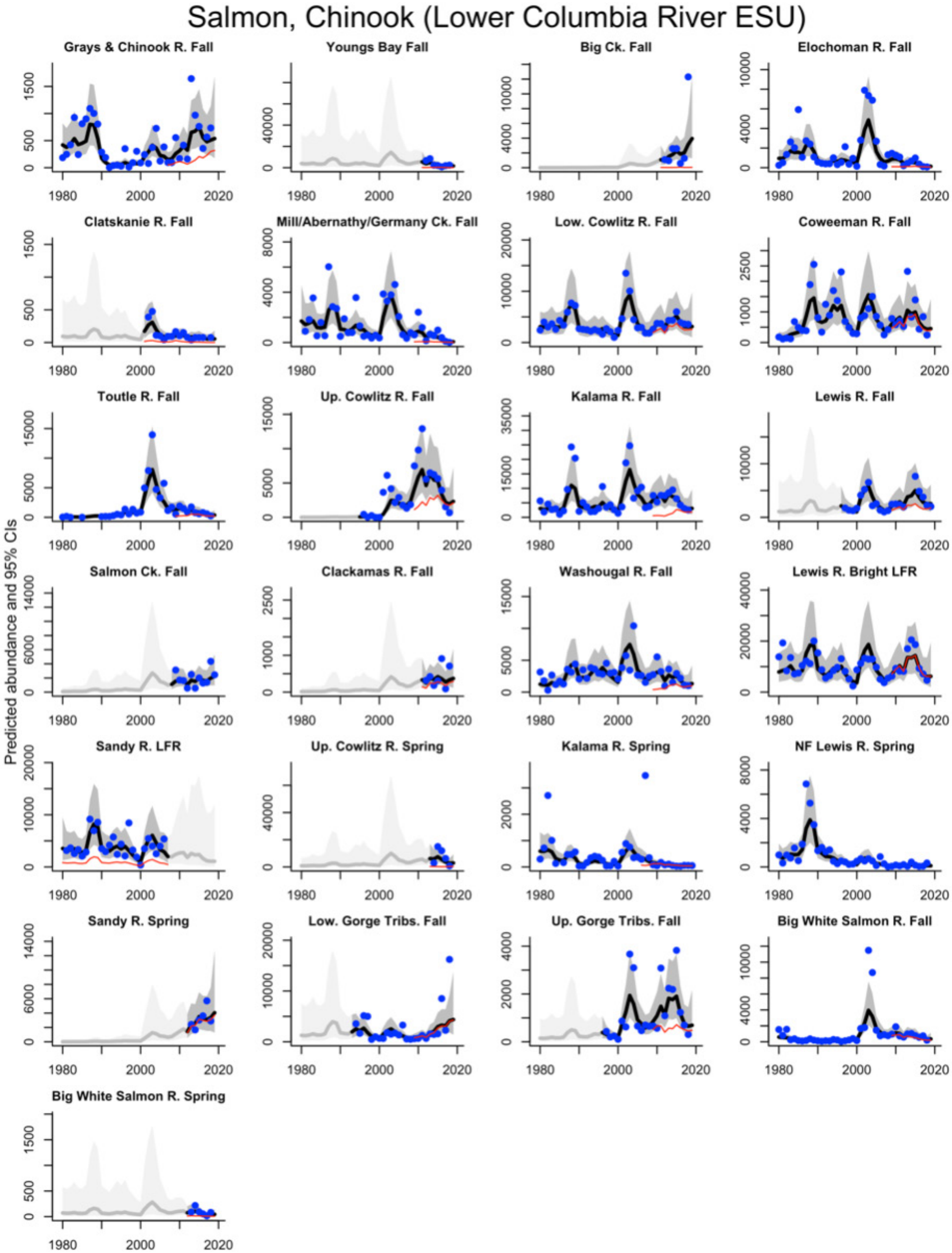

Figure 52. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

Fall-run Coastal 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 30). The abundance of naturally produced adults is low to very low for all populations (Figure 52), with the confounding effects of the first-generation progeny of naturally spawning hatchery fish increasing the uncertainty in any conclusions regarding productivity. Only in the Grays River Tule DIP was there a considerable increase in five-year and longer-term abundance, from 79 to 228 (Tables 28 and 29), although hatchery-origin fish still constitute the majority of natural spawners (Table 30). The Elochoman River/Skamokawa Tule population was largely stable, with a five-year geomean abundance of 95. Of the remaining populations, downward trends were observed in the Youngs Bay, Clatskanie River, and Mill/Abernathy/Germany Creeks Tule populations, all of which have low abundances. Spawning surveys for Youngs Bay and Big Creek are incomplete. Big Creek surveys are not done every year, and returns are dominated by returns to the hatchery. Presently, unmarked fall-run Chinook salmon are passed over the Big Creek weir to spawn naturally in the upper basin, as there is limited spawning habitat below the weir; the most recent estimate for natural-origin spawners was 118 in 2018. The Clatskanie River surveys are strongly influenced by large numbers of hatchery-origin fish being attracted to Plympton Creek, whereas the mainstem Clatskanie River has a few natural-origin spawners (>10), but almost no hatchery fish (Table 28). In surveys conducted in both 2012 and 2013 (the last on record), no Chinook salmon were observed in Scappoose Creek. Overall productivity estimates were negative, except for the Grays River Tule DIP (Figure 53).

Fall-run Cascade MPG

The majority of the populations in this DIP have exhibited stable or slightly positive natural-origin abundance trends. Six of the nine populations exhibited positive short-term trends (Table 28). Natural-origin spawner abundances were 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 (Table 30). The Lower Cowlitz River Tule DIP had the highest five-year abundance (3,208), a 25% increase over the previous period (Table 28); interestingly, the proportion of hatchery-origin spawners in this DIP was relatively low (29.0%), especially given the large hatchery program present (Gleizes et al. 2014). Annual variability in the proportion of hatchery-origin spawners is very high in the Clackamas River (Figure 58), 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 Tule DIP) suggest that the trap-and-haul program at Mayfield Dam has been successful (Serl and Morrill 2010, Rubenson et al. 2019). Overall, most of the fall-run populations in this MPG are improving, even approaching recovery levels in some cases, 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 NMFS's lower Columbia River recovery plan (Dornbusch 2013).

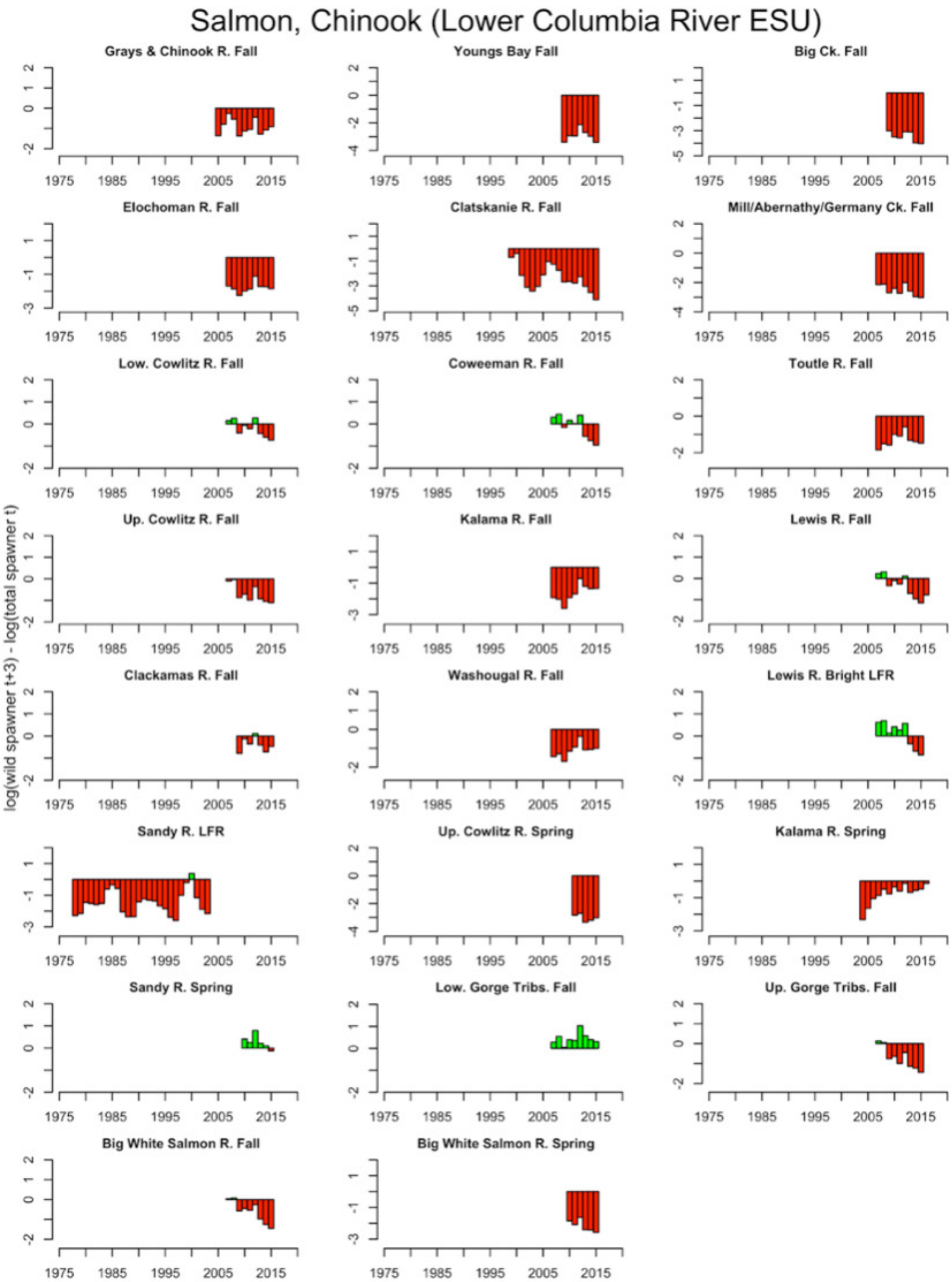

Figure 53. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year ($t - 4$). Spawning years on x-axis.

Table 29. Fifteen-year trends (slope) in log natural spawner abundance computed from a linear regression applied to the smoothed natural spawner log abundance estimate vs. year. In parentheses are the upper and lower 95% CIs. Only populations with at least 4 wild spawner estimates and with at least 2 data points in the first 5 years and last 5 years of the 15-year ranges are shown.

Population	MPG	1990–2005	2004–2019
Kalama River SP	Spring-run Cascade	—	–0.05 (–0.09, –0.01)
Grays River Tule FA	Fall-run Coastal	—	0.12 (0.08, 0.15)
Clatskanie River FA	Fall-run Coastal	—	–0.16 (–0.23, –0.09)
Sandy River Bright LFR	Late fall-run Cascade	0.01 (–0.05, 0.07)	—

Table 30. Five-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates) for Lower Columbia River Chinook salmon ESU populations. A value only in parentheses means that a total spawner count was available but no or only one estimate of natural spawners available. Blanks mean no estimate available in that 5-year range.

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Upper Cowlitz/Cispus Rivers SP	Spring-run Cascade	—	—	—	0.08	0.06
Kalama River SP	Spring-run Cascade	—	—	—	1.00	1.00
North Fork Lewis River SP	Spring-run Cascade	—	—	—	—	—
Sandy River SP	Spring-run Cascade	—	—	—	0.89	0.92
Big White Salmon River SP	Spring-run Gorge	—	—	—	0.13	0.18
Grays River Tule FA	Fall-run Coastal	—	—	0.36	0.22	0.43
Youngs Bay FA	Fall-run Coastal	—	—	—	0.04	0.14
Big Creek FA	Fall-run Coastal	—			0.03	0.04
Elochoman River/ Skamokawa Tule FA	Fall-run Coastal	—	—	—	0.17	0.45
Clatskanie River FA	Fall-run Coastal	—	0.10	0.19	0.09	0.05
Mill/Abernathy/Germany Creeks Tule FA	Fall-run Coastal	—	—	—	0.11	0.22
Lower Cowlitz River Tule FA	Fall-run Cascade	—	—	—	0.70	0.77
Coweeman River Tule FA	Fall-run Cascade	—	—	—	0.82	0.91
Toutle River Tule FA	Fall-run Cascade	—	—	—	0.31	0.55
Upper Cowlitz River Tule FA	Fall-run Cascade	—	—	—	0.35	0.82
Kalama River Tule FA	Fall-run Cascade	—	—	—	0.08	0.57
Lewis River Tule FA	Fall-run Cascade	—	—	—	0.67	0.56
Clackamas River FA	Fall-run Cascade	—	—	—	0.60	0.68
Sandy River FA	Fall-run Cascade	—	—	—	—	—
Washougal River Tule FA	Fall-run Cascade	—	—	—	0.30	0.58
Lower Gorge Tributaries Tule FA	Fall-run Gorge	—	—	—	0.89	0.96
Upper Gorge Tributaries Tule FA	Fall-run Gorge	—	—	—	0.40	0.58
Big White Salmon River Tule FA	Fall-run Gorge	—	—	—	0.80	0.57
Lewis River Bright LFR	Late fall-run Cascade	—	—	—	1.00	1.00
Sandy River Bright LFR	Late fall-run Cascade	0.24	0.24	0.24	—	—

Fall-run Gorge MPG

Many of the populations in this MPG have limited spawning habitat available, either because of inundation of historical habitat in the upper gorge or the loss of access. Natural-origin returns for most populations are in the hundreds of fish, with decreases in abundance noted for those populations for which we have abundance estimates (Figure 52). The removal of Condit Dam in 2011 has restored access to spawning habitat for both fall- and spring-run Chinook salmon; fall-run (tule) Chinook salmon appear to be reestablishing themselves, while spring-run recolonization has been very limited (LCFRB 2020). Recent five-year geomean for the Big White Salmon River was 282, a 63% decline in abundance (Table 28). Chinook salmon estimates on the Oregon side of the Fall-run Gorge MPG have been attempted in the Hood River; however, GRTS criteria have not been met and population estimates are not available after 2007. Escapement to the other smaller tributaries is thought to be very low, and hatchery contribution high. Resolution of the temporal distribution of fall- and late fall-run Chinook salmon in the Sandy River is needed to determine the overall risk of this MPG.

Late fall-run Cascade MPG

The Lewis River Bright DIP in this MPG is likely the most viable in this ESU. The Lewis River Bright DIP has the largest natural-origin abundance in the ESU (8,725), and although the short-term abundance trend is negative, there is a stable long-term trend (Table 28). Merwin Dam, on the North Fork Lewis River, limits the amount of available spawning habitat for late fall-run Chinook salmon, but also controls flows and temperatures. The Sandy River late fall run is no longer directly monitored; the most recent estimate was 373 spawners in 2010 (Takata 2011). Instead, abundance estimates for Sandy River fall-run and late fall-run Chinook salmon are combined by ODFW into a single Sandy River fall-run data series, which increased during the recent review period (five-year geomean = 2,074, a 76% increase). Both the Lewis River and Sandy River late fall runs maintain their abundances without supplementation. Although there is some uncertainty in the status of Sandy River late fall-run Chinook salmon, this MPG appears to be at a relatively low risk.

Harvest

Lower Columbia River Chinook salmon include populations representing three distinct life-history components: spring-run, fall-run, and late fall-run Chinook salmon. These different components are subject to different in-river fisheries (mainstem and tributary) because of differences in river entry timing, but share relatively similar ocean distributions. Harvest rates for populations with different run timings share similar exploitation rate patterns, but differ in absolute harvest rates. With a run timing, tributary-specific harvest rates may differ. All populations saw a drop in exploitation rates in the early 1990s in response to decreases in abundance. There has been a modest increase since then (Figure 54). Ocean fishery impact rates have been relatively stable in the past few years, with the exception of the bright (late fall) component of the ESU.

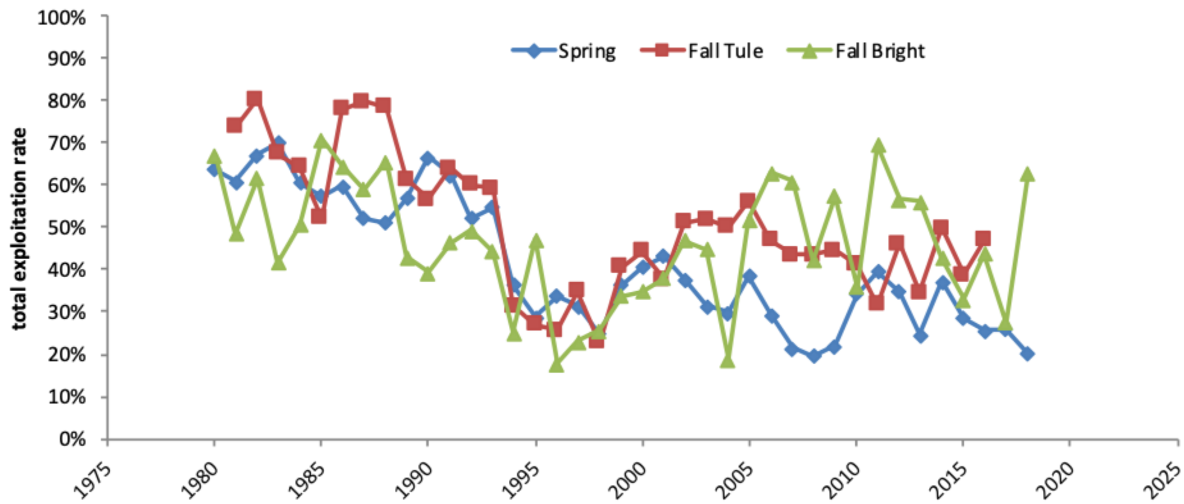

Figure 54. Total exploitation rates on the three components of the Lower Columbia River Chinook salmon ESU. Data for tule (fall-run) Chinook salmon from exploitation rate analysis of aggregate tule stock made up of tag codes from the Big Creek, Cowlitz River, Kalama River, and Washougal River hatcheries. Data for late fall-run Chinook salmon from the CTC exploitation rate analysis for Lewis River Bright late fall-run. Data for spring-run Chinook salmon from CTC model calibration 1503 for Willamette River spring-run Chinook salmon for ocean impacts and TAC run reconstruction data for in-river impacts, using an aggregate of Cowlitz River, Kalama River, Lewis River, and Sandy River spring-run Chinook salmon (ODFW and WDFW 2020a,b).

Spatial structure and diversity

Hatcheries

During the interim since the 2015 status review update, there have been a number of changes in both the quality and quantity of hatchery production in the lower Columbia River. Foremost among these is a reduction in the production of fall-run Chinook salmon from Mitchell Act hatcheries below Bonneville Dam (NMFS 2017a)—specifically, a reduction in or termination of fall-run Chinook salmon programs in Big Creek, Deep River, Kalama River, and Washougal River. These reductions in fall-run Chinook salmon releases in the Coastal (Figure 55) and Cascade strata (Figure 56) have been offset by increases in fall-run Chinook salmon in the Gorge stratum (Figure 57). Additionally, broodstock sources for Mitchell Act hatcheries are required to come from within-stratum/MPG sources by 2022 (NMFS 2017a).

The Hatchery Science Review Group (HSRG 2009) identified the use of out-of-basin stocks in Select Area Fishery Evaluation (SAFE) areas in the lower Columbia River as a concern, especially in light of the high level of straying onto nearby spawning grounds. Approximately 400,000 out-of-ESU Rogue River bright (RRB) 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 55). In the past, naturally produced juvenile Rogue River Chinook salmon and RRB × 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 RRB fall-run Chinook salmon have

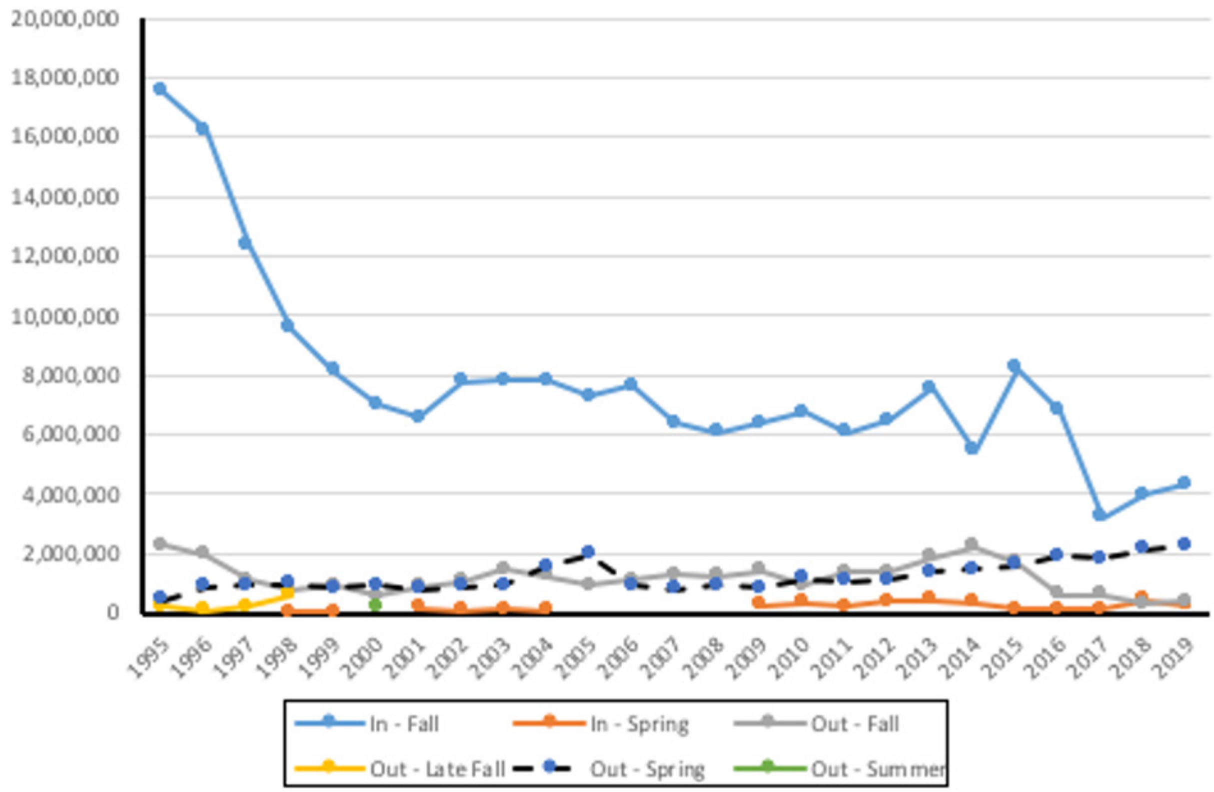

Figure 55. Annual releases of Chinook salmon juveniles into the Coastal stratum of the Lower Columbia River Chinook salmon ESU, 1995–2019. *In* and *Out* indicate whether the source of the release originally came from within or outside of the ESU. Releases of fish weighing less than 2.5 g were removed. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

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 in the Coastal stratum, but may pose a risk to natural-origin juveniles due to competition and predation. There is also the potential for the incidental take or hooking mortality of natural-origin fish in the targeted fisheries in the SAFE zone. 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.

Releases of Chinook salmon into the Cascade stratum have been relatively stable in recent years. There have been some reductions in the number of fall-run Chinook salmon in an effort to decrease the contribution of hatchery-origin fish to naturally spawning adults. Spring-run Chinook salmon production has continued, in part, due to the inaccessibility of historical spring-run spawning and rearing habitat. The termination of the non-native late fall-run Chinook salmon below Bonneville Dam has decreased the risk of introgression between native natural- and hatchery-origin fish.

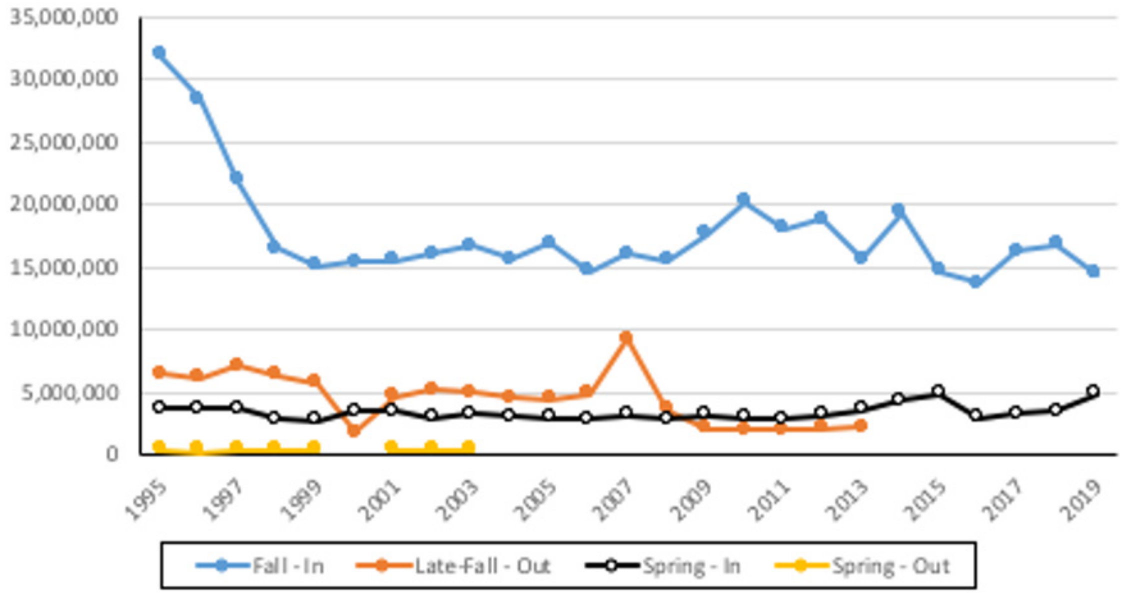

Figure 56. Annual releases of Chinook salmon juveniles into the Cascade stratum of the Lower Columbia River Chinook salmon ESU, 1995–2019. *In* and *Out* indicate whether the source of the release originally came from within or outside of the ESU. Late fall *Out* releases were primarily of upriver bright stocks, and spring *Out* releases primarily represented sources from the Upper Willamette River ESU. Releases of fish weighing less than 2.5 g were removed. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

Hatchery production in the Gorge stratum has focused on the production of fall-run Chinook salmon from Spring Creek National Fish Hatchery (NFH), which decreased during the 2015–19 period. The release of several million non-native upriver (late fall) brights has continued, as has the release of non-native Carson Hatchery spring-run Chinook salmon. It was noted that large numbers of feral and hatchery-origin upriver brights were observed spawning in the Big White Salmon River (LCFRB 2020). Similarly, late fall-run Chinook salmon (upriver brights) are also observed spawning in large numbers (1,000+) below Bonneville Dam, near Ives Island. These large feral populations are a diversity risk to native fall-run populations.

Reductions in the potential influence of hatchery-origin fish in lower Columbia River tributaries are also being implemented via the operation of weirs in a number of basins in both Washington and Oregon in order to remove hatchery-origin Chinook salmon before they can spawn naturally (Whitman et al. 2017, Wilson et al. 2019). At the same time, the release of non-native (out-of-ESU) hatchery-origin fish continues in a number of locales. Overall, the potential risk from hatchery operations to diversity has diminished somewhat during this period.

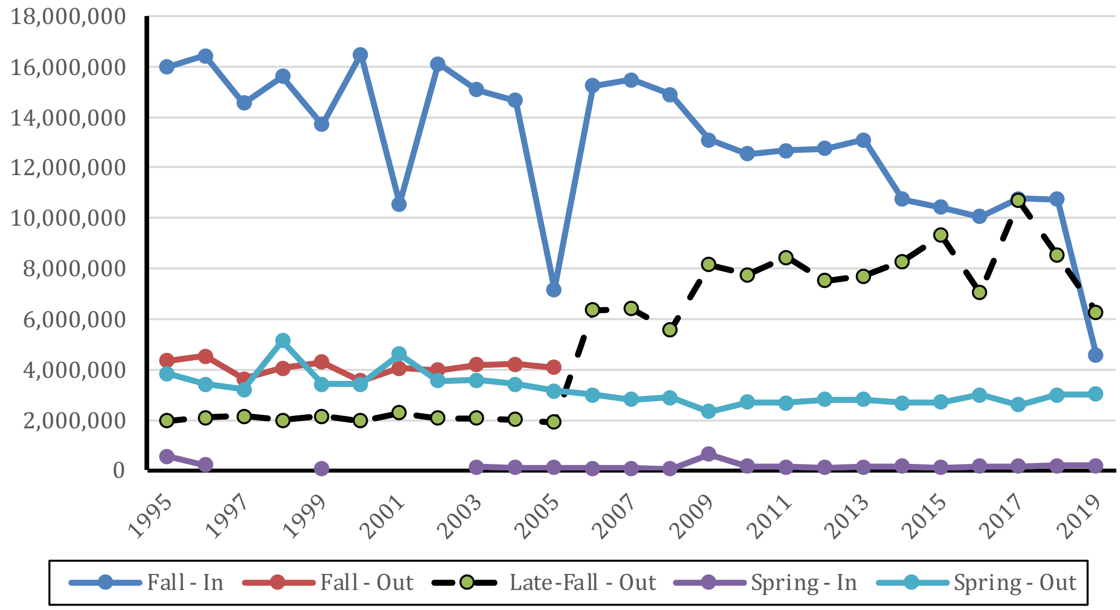

Figure 57. Annual releases of Chinook salmon juveniles into the Gorge stratum of the Lower Columbia River Chinook salmon ESU, 1995–2019. *In* and *Out* indicate whether the source of the release originally came from within or outside of the ESU. Late fall *Out* releases were primarily of upriver bright stocks, and spring *Out* releases were primarily sources from Carson Hatchery, Klickitat Hatchery, and the Deschutes River (Klickitat River spring-run Chinook salmon are included in the identified but not listed Middle Columbia River spring-run Chinook salmon ESU and released into the Gorge stratum). Releases of fish weighing less than 2.5 g were removed. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

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 and Morrill 2010), with collection efficiencies averaging 28.8% for Chinook salmon during 2006–09. More recently, the installation of a new collection structure at Cowlitz Falls Dam appears to provide improved collection efficiency and survival: 78.7% fish passage survival for Chinook salmon in 2019 (Rubenson et al. 2019). Adult returns in 2020 and 2021 will be the first opportunity to assess the benefit of these improvements. In addition, the collection of juvenile fall-run Chinook salmon from the Tilton River at Mayfield Dam appears to be relatively 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. The sediment retention structure (SRS) remains an impediment to fish passage in the North Fork Toutle River. Additionally, the existing Toutle Fish Collection Facility is limited in its capacity to attract fish for transport (LCFRB 2020). On the Hood River, Powerdale Dam was removed in 2010, and while this dam previously allowed fish passage, removal of the dam is thought to have eliminated passage delays and injuries. Condit Dam, on the White Salmon River, was removed in 2011, providing access to previously inaccessible habitat. Spawner surveys of the White Salmon River indicate that both hatchery-origin and unmarked

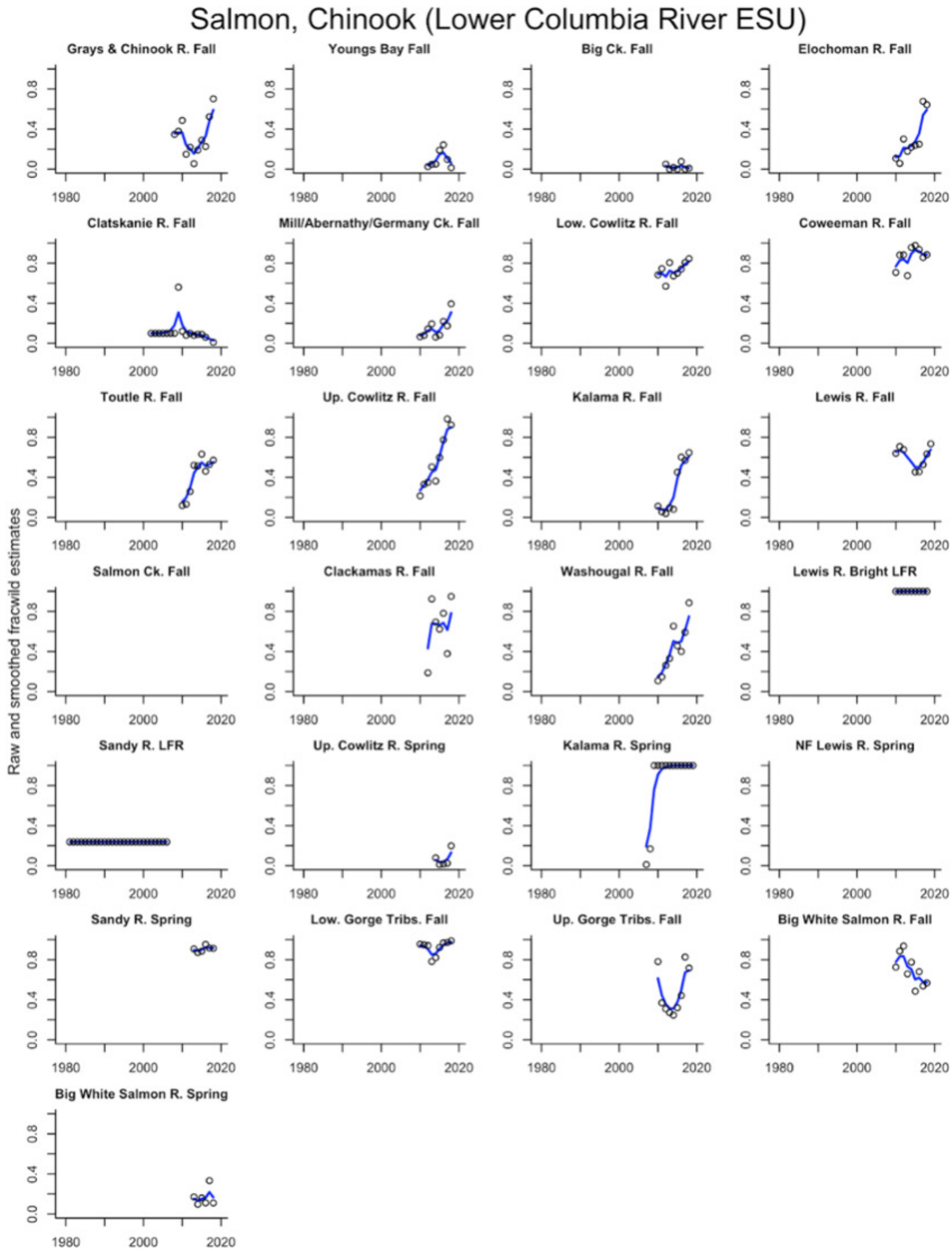

Figure 58. Smoothed trend in the estimated fraction of the natural spawning Lower Columbia River Chinook salmon population consisting of fish of natural origin. Points show the annual raw estimates, where available.

(presumed natural-origin) Chinook salmon are colonizing the newly accessible habitat (LCFRB 2020). 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). Few adults have been available for passage, and juvenile passage efficiencies were initially poor for Chinook salmon, but recent modifications to the collector at Swift Dam have shown improvements in efficiency (PacifiCorp 2020).

Once passage actions are undertaken, it may still take several years for the benefits to become evident. For example, the removal of Marmot Dam in 2007 and the Little Sandy River diversion dam in 2008 have clearly demonstrated improvement in the abundance of spring-run Chinook salmon returning to the Sandy River during this most recent period. Still, several programs continue to improve their operations and may achieve fish collection efficiencies suitable to support sustainable populations in previously inaccessible habitat sometime in the near future (5–10 years). In addition to these large-scale efforts, there have been a number of recovery actions throughout the ESU to remove or improve thousands of sub-standard culverts and other small-scale passage barriers, as well as breaching dikes to provide access to juvenile habitat.

Although the spatial structure contribution to Lower Columbia River Chinook salmon ESU viability has improved during the current review period (2015–19), effective access to upstream habitat in the Cowlitz and Lewis River basins remains the major limitation.

Biological status relative to recovery goals

Of the 32 DIPs in this ESU, seven are at or near the recovery viability goals (Table 31) set in the recovery plan (Dornbusch 2013). The seven DIPs included one spring-run, five fall-run, and one late fall-run DIP. Six of these seven DIPs were located in the Cascade stratum; most of the populations in the Coastal and Gorge strata are doing rather poorly. Many of the remaining populations still require substantial improvements in abundance to reach their viability goals. 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 (Dornbusch 2013). Of greater concern was the large number of DIPs (ten) that either had no abundance information (presumed near zero) or exist at very low abundances. All of the Fall-run Coastal and Fall-run Gorge MPG populations (except the Lower Gorge Tributaries Tule DIP) 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 at or near zero, while others may only persist through hatchery supplementation. The Fall-run Cascade MPG contains a number of populations above or near their recovery goals, while the Late fall-run Cascade MPG may be near viability—although there is some uncertainty in the abundance estimates for the Sandy-River Bright late fall-run DIP.

Some populations met the hatchery contribution criteria for primary or contributing populations established by the HSRG (2009) during the 2015–19 period, although other populations did not meet the criteria but did improve in the proportion of natural-origin spawners. Among these were the Coweeman River Tule fall, Lewis River Bright late fall, and Lewis River Tule fall runs. No criteria were established for stabilizing populations. Thus, only one MPG may have met its viability goals, with most other MPGs far from theirs.

Table 31. Current 5-year geometric mean of raw natural-origin spawner abundances and recovery targets (Dornbusch 2013) for Lower Columbia River Chinook salmon demographically independent populations (DIPs). Numbers in parentheses represent total (hatchery- and natural-origin) spawners. Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > x < 50%, yellow = 50% > x < 100%), green = >100%.

		Abundance	
Stratum	**Population**	**2015–19**	**Target**
Coastal	Grays River Tule FA (WA)	228	1,000
	Youngs Bay FA (OR)	145	505
	Big Creek FA (OR)	0	577
	Elochoman River/Skamokawa Tule FA (WA)	95	1,500
	Clatskanie River FA (OR)	3	1,277
	Mill/Abernathy/Germany Creeks Tule FA (WA)	28	900
	Scappoose Creek FA (OR)	n/a	1,222
Cascade	Upper Cowlitz/Cispus Rivers SP (WA)	171	1,800
	Kalama River SP (WA)	43	300
	North Fork Lewis River SP (WA)	(112)	1,500
	Sandy River SP (OR)	3,359	1,230
	Toutle River SP (WA)	n/a	1,100
	Cispus River SP (WA)	n/a	1,800
	Tilton River SP (WA)	n/a	100
	Lower Cowlitz River Tule FA (WA)	3,208	3,000
	Coweeman River Tule FA (WA)	543	900
	Toutle River Tule FA (WA)	280	4,000
	Upper Cowlitz River Tule FA (WA)	1,761	n/a
	Kalama River Tule FA (WA)	2,142	500
	Lewis River Tule FA (WA)	2,003	1,500
	Clackamas River FA (OR)	236	1,551
	Sandy River FA (OR)	(2,074)	1,031
	Washougal River Tule FA (WA)	914	1,200
	Salmon Creek FA (WA)	n/a	n/a
	Lewis River Bright LFR (WA)	8,725	7,300
	Sandy River Bright LFR (OR)	n/a	3,561
Gorge	Big White Salmon River SP (WA)	8	500
	Hood River SP (OR)	n/a	1,493
	Lower Gorge Tributaries Tule FA (WA & OR)	4,528	1,200
	Upper Gorge Tributaries Tule FA (WA & OR)	537	1,200
	Big White Salmon River Tule FA (WA)	283	500
	Hood River FA (OR)	n/a	1,245

Updated biological risk summary

Overall, there has been modest change since the last status review in the biological status of Chinook salmon populations in the Lower Columbia River Chinook salmon ESU (NWFSC 2015), although some populations did exhibit marked improvements (Figure 52). Increases in abundance were noted in about half of the fall-run populations, and in 75% of the spring-run populations for which data were available. Decreases in hatchery contribution were also noted for several populations. Relative to baseline VSP levels identified in the recovery plan (Dornbusch 2013), there has been an overall improvement in the status of a number of fall-run populations (Table 28), although most are still far from the recovery plan goals.

Improved fall-run status reflects both changes in biological status and improved monitoring. Spring-run Chinook salmon populations in this ESU are generally unchanged; most of the populations are at a "high" or "very high" 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 a five-year average of 3,359, nearly double the previous five-year average. This appears to be due, in part, to the removal of Marmot Dam (eliminating migrational delays and passage injuries) and the diversion dam on the Little Sandy River (restoring access and flow to historical habitat). Elsewhere in the ESU, 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. 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 the status of spring-run Chinook salmon in the Toutle River basin remains unclear. The removal of Condit Dam on the White Salmon River has provided an opportunity for the reestablishment of naturally spawning fall- and spring-run populations with volitional access to historical spawning grounds. The status of spring-run Chinook salmon in the Hood River is unclear; with the removal of Powerdale Dam, there is minimal monitoring in the basin and the abundance and genetic composition of returning spring-run Chinook salmon is unknown. It remains to be determined if any native spring-run Chinook salmon remain, or if they have been supplanted by those from the Deschutes River (Middle Columbia River spring-run Chinook salmon ESU).

Many 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 upriver bright fall-run, RRB fall-run, upper Willamette River spring-run, Carson Hatchery spring-run, and Deschutes River spring-run, remains a concern. Harvest rates are a potential concern, especially for low-abundance tule fall-run populations. There have been a number of notable efforts to restore migratory access to areas upstream of dams, but 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 (Condit Dam, Marmot Dam, and Powerdale Dam) not only improve/provide access, but 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. Finally, although many of the populations in this ESU are at "high" risk, it is important to note that poor ocean and freshwater conditions existed during the 2015–19 period and, despite these conditions, the status of a number of populations improved, some remarkably so (Grays River Tule, Lower Cowlitz River Tule, and Kalama River Tule fall runs). Overall, we conclude that the viability of the Lower Columbia River Chinook salmon ESU has increased somewhat since the last status review, although the ESU remains at "moderate" risk of extinction.

Lower Columbia River Coho Salmon ESU

Brief description of ESU

Lower Columbia River coho salmon were identified as an ESU and listed as threatened in 2005. The listing included a redelineation to incorporate tributaries in the Coastal major population group (MPG) in southwestern Washington. This ESU includes all naturally spawned populations 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 59; USOFR 2020). Myers et al. (2006) identified three MPGs (Coastal, Cascade, and Gorge), containing a total of 24 demographically independent populations (DIPs), in the Lower Columbia River coho salmon ESU.

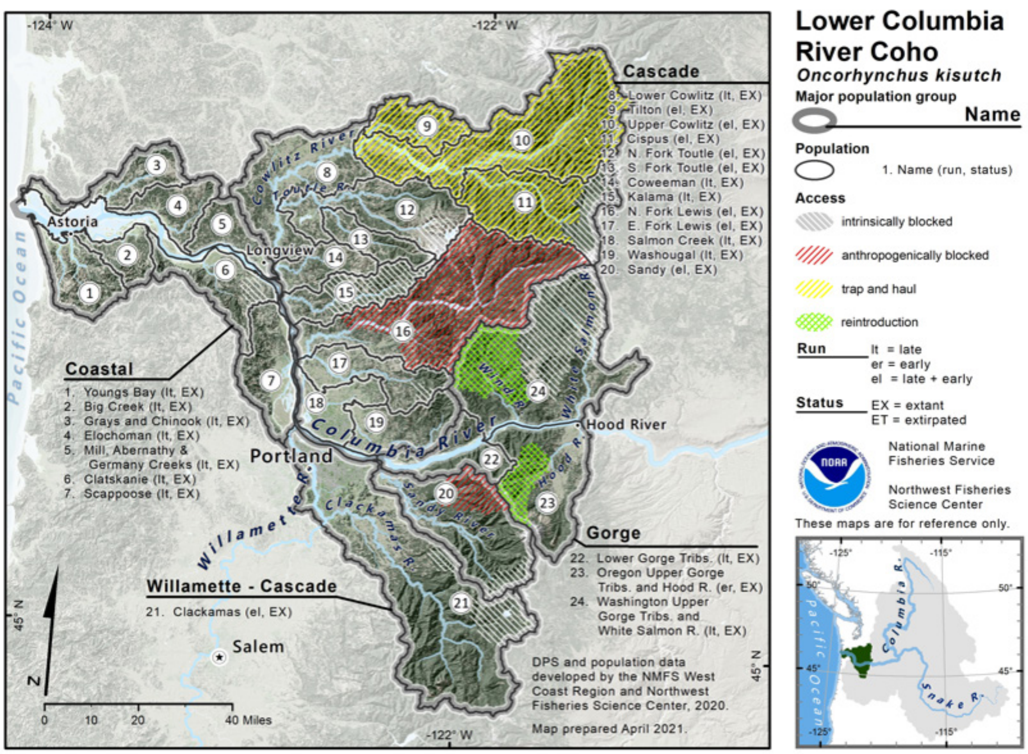

Figure 59. Map of the Lower Columbia River coho salmon ESU's spawning and rearing areas, illustrating demographically independent populations (DIPs) and major population groups (MPGs). Areas that are accessible (green), accessible only via trap-and-haul programs (yellow), or blocked (cross-hatched) are indicated accordingly.

Summary of previous status conclusions

2005

NMFS reviewed the status of the Lower Columbia River coho salmon ESU in 1996, 2001, and again 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 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 for "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 endangered. 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 evaluation resulted in a "threatened" determination in 2005.

2010

Ford et al. (2011) noted that three status evaluations of Lower Columbia River coho salmon, 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 "very high risk." The remaining three (Sandy, Clackamas, and Scappoose Rivers) were considered to be at "high" to "moderate" risk. All of the populations to the north of the Columbia River (in Washington State) were considered "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 thought to occur in these populations, it was not clear that any were truly 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 prior BRT status review in 2005.

In 2010, the ESU Boundaries Review Group (see the ESU Boundaries section in Ford et al. 2011) undertook a reevaluation of the boundary between all lower and middle Columbia River ESUs and DPSes. The review's conclusions emphasized the transitional nature of the boundary between the lower and the middle Columbia River ESUs. The original Lower Columbia River coho salmon ESU boundary was assigned based largely on extrapolation from information about the boundaries for Chinook salmon 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)" (p. 28). To date, this recommendation has not been officially implemented; therefore, the current status review will utilize preexisting ESU boundaries.

2015

The 2015 status review reported improvements in coho salmon abundance, productivity, spatial structure, and diversity; however, this appeared mostly to be due to the improved level of monitoring (more complete accounting) rather than a true change in status over time (NWFSC 2015). In the absence of specific abundance and diversity data, previous status reviews had concluded that hatchery-origin fish dominated many of the coho populations in the Lower Columbia River coho salmon ESU and that there was little natural productivity. Recovery efforts likely also contributed to the observed increases in natural production, but in the absence of longer-term datasets, it was not possible to parse out these effects. Populations with longer-term data sets exhibited either stable or slightly positive abundance trends. Juvenile passage facilities at Cowlitz Falls, Merwin Dam (Lewis River), and North Fork Dam (Clackamas River) were being improved and had the potential to provide upstream populations with better access to high-quality habitat. These and other recovery efforts were thought to likely improve the status of a number of coho salmon DIPs; abundances, however, remained at low levels and the majority of the DIPs were at "moderate" or "high" risk. For the lower Columbia River region, land development and increasing human population pressures continued to degrade habitat, especially in lowland areas. Although populations in this ESU were generally improved, especially in the 2013–14 and 2014–15 return years, poor ocean conditions foreshadowed declines in the upcoming return years. This ESU was considered to be at "moderate" risk of extinction.

Description of new data available for this review

Efforts to standardize and expand monitoring efforts have resulted in abundance time series for a number of populations in this ESU. 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 limited monitoring in the Youngs Bay and Washington Upper Gorge Tributaries 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 and 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.

Abundance and productivity

Coastal MPG

Both short- and long-term trends for almost all coho salmon populations were negative during the 2015–19 review period for six of the seven Coastal MPG populations that were analyzed (Table 32). Only the Mill/Abernathy/Germany DIP abundance was stable, with a five-year geomean of 685. Negative trends were heavily biased by the strong adult return in 2014 and the poor return in 2015. In the absence of data from these two markedly different years, the trends would likely be largely flat (Figure 60). Average natural-origin abundances were in the hundreds of fish, with the exception of the Youngs Bay and likely Big Creek DIPs, which are not monitored except at the hatchery racks. Given the propensity of coho salmon to spawn in smaller tributaries and the year-long freshwater residence of juveniles, the poor freshwater conditions during this period likely affected coho salmon in the Coastal MPG more than in the larger rivers of the Cascade MPG.

Cascade MPG

As with the Coastal MPG, coho salmon populations in the Cascade MPG experienced a marked decline in abundance following the "boom" year of 2014. The five-year geometric means for these populations were in the high hundreds to low thousands, with the exception of the Kalama River and Washougal River DIPs. Population trends were strongly negative, with the exception of the small Kalama River and Salmon Creek populations. The Salmon Creek DIP experienced a slight decline in five-year geometric abundance (4% decline), but maintains a relatively high absolute abundance for a relatively small basin with a five-year geomean of 1,546 (Table 32). Population trends were certainly affected by the very poor spawner counts in 2015. Longer-term, 15-year average trends were largely stable (not significantly different from zero), except for the Upper Cowlitz/Cispus Rivers group (two combined DIPs), which was slightly negative.

The Clackamas River was one of the two populations identified in the original 1996 status review that appeared to be self-sustaining natural populations. While recent returns of unmarked fish to the Clackamas River have shown a marked decline since the 2014–15 record return year, when 10,670 spawners were counted, the 21% decline is one of the smallest in the MPG. The long-term (15-year) trend for this population is slightly positive (Table 33), and the current five-year geomean of 2,889 is still the largest abundance in the ESU (Table 32). Improvements in juvenile downstream passage at dams on the Clackamas River may have counterbalanced the poor environmental conditions. Improvements in juvenile collection at Cowlitz Falls have occurred too recently to be reflected in natural spawner numbers. The six populations in the Cowlitz River basin account for the majority of naturally spawning coho salmon in the MPG, with the Lower Cowlitz River late coho salmon DIP five-year geomean of 2,622. In the Cowlitz River basin, those coho salmon populations that relied on dam passage programs (Upper Cowlitz/Cispus Rivers and Tilton River) exhibited a greater decline relative to those populations located below the high-head dams (Lower Cowlitz River, North and South Fork Toutle rivers, and Coweeman River). The

Table 32. Five-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 1 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 2 most recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Grays/Chinook Rivers (late)	Coastal	—	—	—	—	412 (1,644)	212 (843)	–49 (–49)
Youngs Bay (late)	Coastal	—	—	(41)	(12)	(18)	(14)	(–22)
Big Creek (late)	Coastal	—	—	(117)	(249)	(251)	(122)	(–51)
Elochoman River (late)	Coastal	—	—	—	—	738 (1,478)	558 (874)	–24 (–41)
Clatskanie River (late)	Coastal	—	—	335 (364)	745 (771)	1,262 (1,343)	199 (286)	–84 (–79)
Mill/Abernathy/Germany Creeks (late)	Coastal	—	—	—	—	684 (766)	685 (767)	0 (0)
Scappoose River (late)	Coastal	—	—	502 (535)	464 (469)	717 (717)	448 (454)	–38 (–37)
Lower Cowlitz River (late)	Cascade	—	—	—	—	5,243 (5,934)	2,622 (3,102)	–50 (–48)
Coweeman River (late)	Cascade	—	—	—	—	3,185 (3,502)	1,987 (2,241)	–38 (–36)
North Fork Toutle River (early & late)	Cascade	—	—	—	—	1,480 (2,174)	819 (1,502)	–45 (–31)
South Fork Toutle River (early & late)	Cascade	—	—	—	—	2,199 (2,605)	1,075 (1,407)	–51 (–46)
Upper Cowlitz/Cispus Rivers (early & late)	Cascade	—	0 (6,090)	4,065 (37,862)	5,119 (20,256)	1,093 (13,886)	631 (4,370)	–42 (–69)
Tilton River (early & late)	Cascade	—	1,756 (3,451)	967 (13,414)	995 (3,573)	2,362 (6,773)	1,932 (4,187)	–18 (–38)
Kalama River (late)	Cascade	—	—	—	—	15 (328)	43 (180)	187 (–45)
North Fork Lewis River (early & late)	Cascade	—	—	—	—	1,350 (2,954)	1,275 (6,692)	–6 (127)
East Fork Lewis River (early & late)	Cascade	—	—	—	—	1,850 (2,126)	686 (1,041)	–63 (–51)
Salmon Creek (late)	Cascade	—	—	—	—	1,614 (1,654)	1,546 (1,648)	–4 (0)
Clackamas River (early & late)	Cascade	1,816 (2,787)	502 (768)	2,891 (4,497)	2,995 (5,118)	3,645 (4,174)	2,889 (3,226)	–21 (–23)
Sandy River (early & late)	Cascade	—	—	—	1,094 (1,170)	1,708 (1,851)	854 (889)	–50 (–52)
Washougal River (late)	Cascade	—	—	—	—	478 (763)	174 (694)	–64 (–9)
Hood River (early)	Gorge	—	—	—	273 (471)	183 (751)	29 (64)	–84 (–91)
Washington Upper Gorge Tributaries/White Salmon River (late)	Gorge	—	—	—	53 (72)	39 (53)	45 (60)	15 (13)

populations above dams on the Cowlitz River, the North Fork Lewis River DIP, the Kalama River DIP, and the Washougal River DIP also include large numbers of hatchery-origin spawners, in excess of 70% of the total population. Otherwise, the proportion of hatchery-origin spawners in most populations is generally less than 30%.

Within the recent five-year review period, improvements in ocean and freshwater conditions likely influenced the upturn in abundance (Figure 60) and resulted in positive productivity estimates for a number of populations (Figure 61).

This MPG contains most of the ESU's large river basins and hosts the majority of the ESU's abundance.

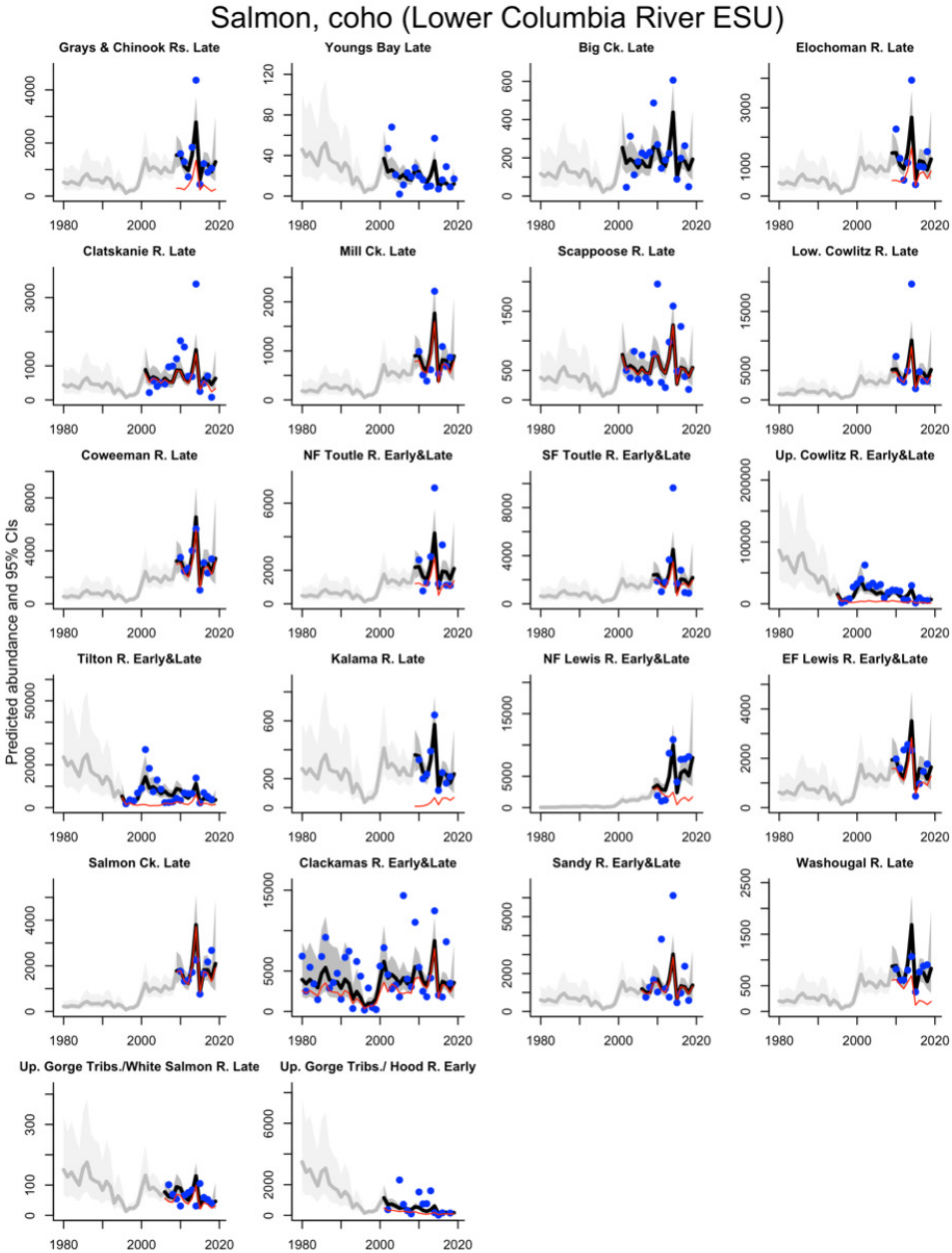

Figure 60. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

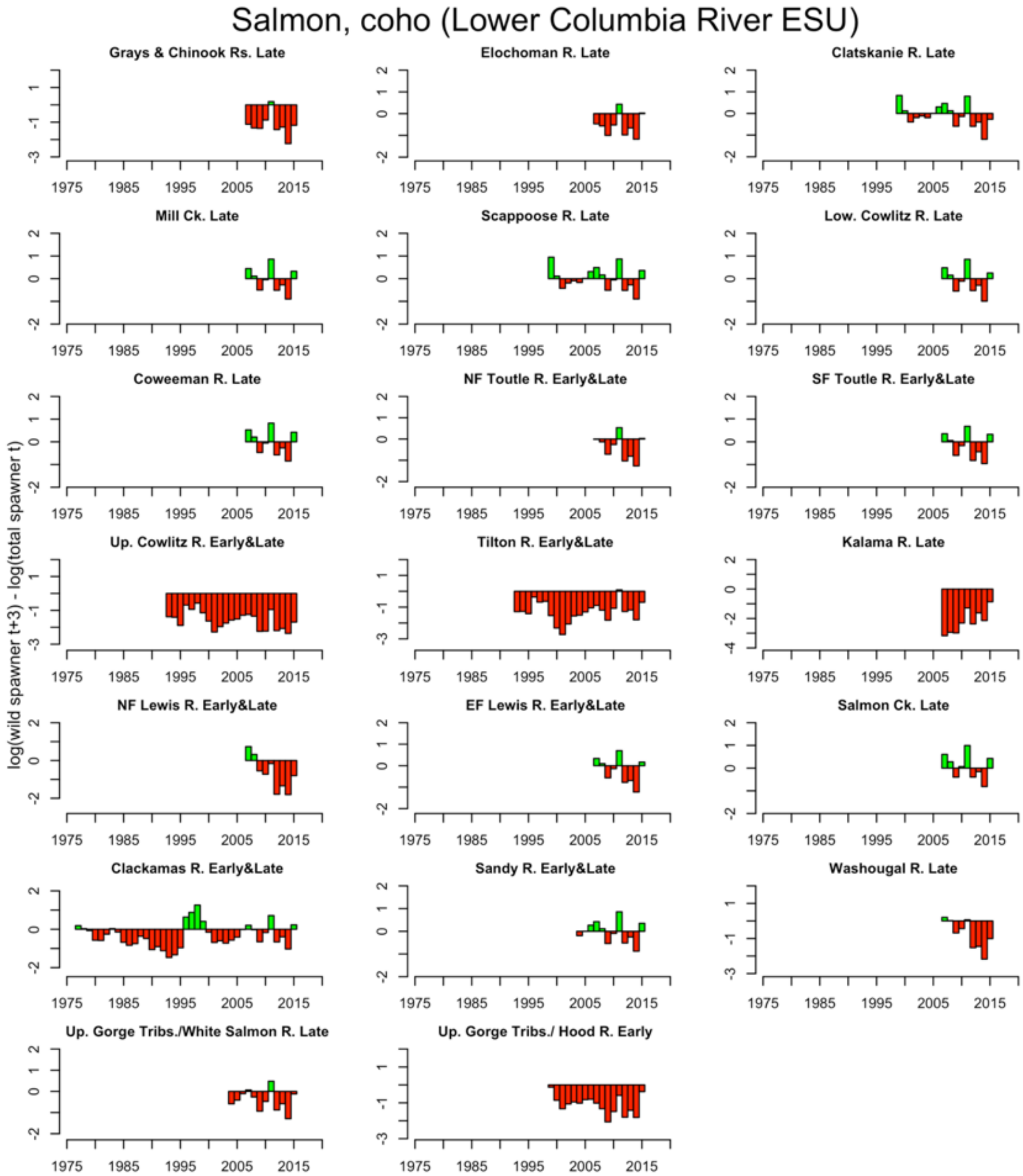

Figure 61. Trends in Lower Columbia River coho salmon population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year ($t - 3$).

Gorge MPG

Natural-origin abundances in this MPG are low; the two populations available (Hood River, and Washington Upper Gorge Tributaries/White Salmon River) both had geomeans of less than 50 (Table 32). Hatchery-origin fish contribute a large proportion of the total number of spawners, most notably in the Hood River. The trend was strongly negative in the Hood River and slightly positive in the White Salmon River. With the exception of the Hood and 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. Monitoring has been limited in the smaller tributaries in Gorge strata, and although insufficient data were available for statistical analysis, it is important to note that natural-origin coho salmon were observed.

Table 33. Fifteen-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	1990–94	1995–99
Clatskanie River (late)	Coastal	—	–0.04 (–0.09, 0.02)
Scappoose River (late)	Coastal	—	0.00 (–0.05, 0.05)
Upper Cowlitz/Cispus Rivers (early & late)	Cascade	—	–0.10 (–0.15, –0.05)
Tilton River (early & late)	Cascade	—	0.00 (–0.06, 0.05)
Clackamas River (early & late)	Cascade	0.04 (–0.05, 0.13)	0.02 (–0.03, 0.07)
Sandy River (early & late)	Cascade	—	0.01 (–0.03, 0.06)
Hood River (early)	Gorge	—	–0.09 (–0.14, –0.04)
Washington Upper Gorge Tributaries/White Salmon River (late)	Gorge	—	–0.04 (–0.09, 0.01)

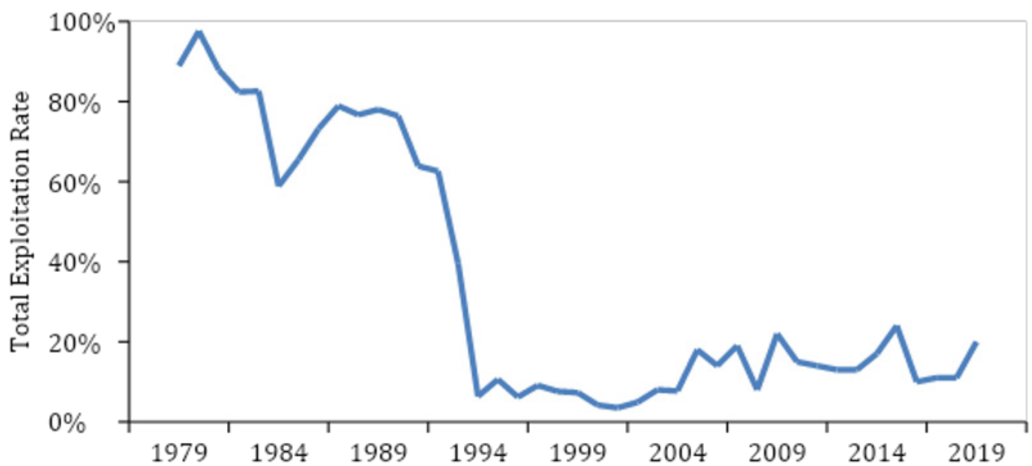

Figure 62. Total exploitation rate on natural Lower Columbia River coho salmon. Data (2005–19) from Table 34 of ODFW and WDFW (2020b).

Other populations

Not included in this ESU are coho salmon that migrate above Willamette Falls; 7,464 natural-origin adult coho salmon were counted at the falls in 2019. Coho have not been planted in the upper Willamette River basin since 1996, and it is believed that these fish are the progeny of lower Columbia River-origin coho salmon (Myers et al. 2006, Keefer et al. 2018). Coho salmon spawning mostly takes place in the westside tributaries to the upper Willamette River, primarily the Tualatin River. We have also not included coho salmon migrating upstream of The Dalles Dam or in the Klickitat River; these are almost entirely the progeny of fish introduced into middle and upper Columbia and Snake River tributaries from lower Columbia River hatchery populations. In 2019, 38,742 adult coho salmon were counted at The Dalles Dam, including both hatchery-origin releases in the interior Columbia River basin and the progeny of naturally spawning fish. In both cases, these fish are spawning outside of the historical boundaries of the Lower Columbia River coho salmon ESU. Historically, coho salmon populations existed above The Dalles Dam, but were extirpated during the last century.

Harvest

Lower Columbia River coho salmon are part of the Oregon Production Index (OPI), 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 of Vancouver Island (WCVI). Canadian coho salmon fisheries were severely restricted in the 1990s to protect upper Fraser River coho salmon, and have remained so ever since. Ocean fisheries off California were closed to coho salmon retention in 1993 and have remained closed ever since. Ocean fisheries for coho salmon off of Oregon and Washington were dramatically reduced in 1993 in response to the depressed status of Oregon coast natural coho and subsequent listing, and moved to mark-selective fishing beginning in 1999. Lower Columbia River coho salmon 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 62). In addition, freshwater fisheries impacts on naturally produced coho salmon have been markedly reduced through the implementation of mark-selective fisheries. More recently, NMFS ESA guidance for the harvest of lower Columbia River natural (LCN) coho salmon in marine and mainstem Columbia River fisheries is based on a matrix describing parent escapement levels for multiple populations and the observed Columbia River OPI smolt-to-jack survival rate. For example, based on this matrix, the total allowable marine and mainstem Columbia River exploitation rate for LCN coho salmon in 2019 fisheries would be no more than 23.0% (PFMC 2019).

Spatial structure and diversity

Hatcheries

Hatchery releases have remained relatively steady at 10–17 million since the 2005 BRT report, with approximately 14 million coho salmon juveniles released in 2019. Many of the populations in the ESU contain a substantial number of hatchery-origin spawners. Production has been shifted into localized areas (e.g., Youngs Bay, Big Creek, and Deep Creek) in order to reduce the influence of hatchery fish in other nearby populations (Scappoose and Clatskanie Rivers; Figure 64). There were no spawner surveys conducted in the Youngs Bay or Big Creek DIPs, but it can be assumed that the proportion of natural spawners is very low. Hatchery influence is also relatively high in the Grays River, with a recent decline in fraction natural (Table 34, Figure 63). The influence of hatchery programs on naturally spawning fish has been reduced in a number of basins with the removal of marked adults at weirs, but other basins indicate an increase in the proportion of hatchery fish spawning naturally (Table 34), perhaps as a result of increased hatchery releases (Figure 64). Mass marking of hatchery-released fish, in conjunction with expanded coho salmon spawning surveys, has provided more accurate estimates of hatchery straying.

Integrated hatchery programs have been developed in a number of basins to limit the loss of genetic diversity. The integrated program in the Cowlitz River was developed for reintroductions into the upper Cowlitz River basin. Large-scale releases of these hatchery-origin coho salmon adults into the upper Cowlitz, Cispus, and Tilton Rivers were used to recolonize

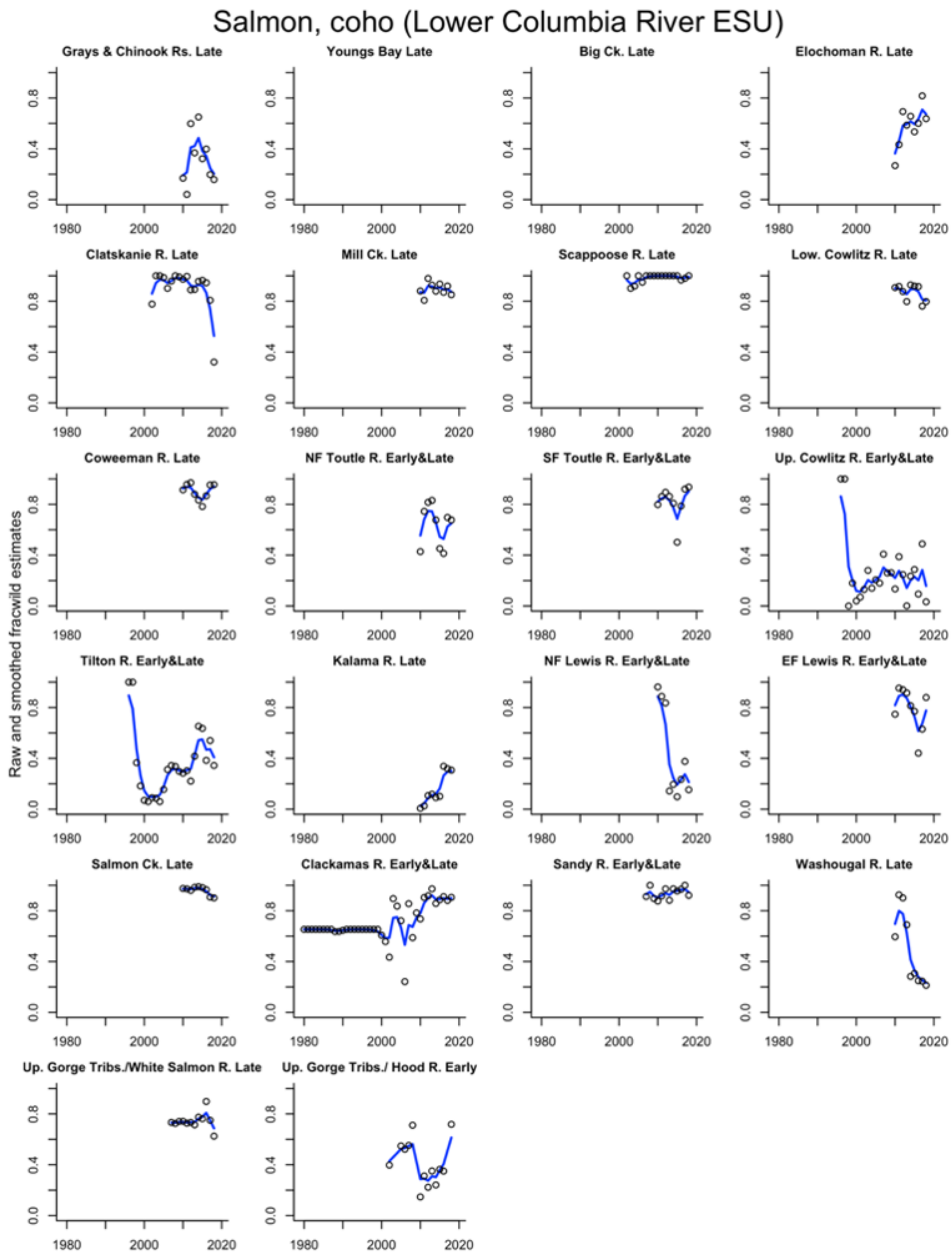

Figure 63. Smoothed trend in the estimated fraction of the naturally spawning Lower Columbia River coho salmon population consisting of fish of natural origin. Points show the annual raw estimates, where available.

stream habitat above the mainstem dams. A segregated program exists for coho salmon releases into the lower Cowlitz River. Overall, juvenile releases into the Cowlitz River basin were reduced some 10 years ago, but have been fairly steady since then (Figure 65). A large integrated program for Type N coho salmon has been ongoing in the Lewis River for over a decade, while the Type S (early) coho salmon program in the Lewis River is operated as a segregated program. Both early- and late-run hatchery-origin coho salmon are transported above Swift Dam in the Lewis River to reestablish production in headwater areas (PacifiCorp 2020).

Table 34. Five-year mean of fraction natural Lower Columbia River coho salmon spawners (sum of all estimates divided by number of estimates). Blanks mean no estimate available in that 5-year range.

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Grays/Chinook Rivers (late)	Coastal	-	-	-	0.37	0.27
Elochoman River (late)	Coastal	-	-	-	0.53	0.65
Clatskanie River (late)	Coastal	-	0.93	0.97	0.94	0.76
Mill/Abernathy/Germany Creeks (late)	Coastal	-	-	-	0.89	0.89
Scappoose River (late)	Coastal	-	0.94	0.99	1.00	0.99
Lower Cowlitz River (late)	Cascade	-	-	-	0.88	0.85
Coweeman River (late)	Cascade	-	-	-	0.91	0.89
North Fork Toutle River (early & late)	Cascade	-	-	-	0.70	0.56
South Fork Toutle River (early & late)	Cascade	-	-	-	0.84	0.79
Upper Cowlitz/Cispus Rivers (early & late)	Cascade	0.73	0.13	0.26	0.20	0.23
Tilton River (early & late)	Cascade	0.64	0.07	0.29	0.38	0.48
Kalama River (late)	Cascade	-	-	-	0.07	0.27
North Fork Lewis River (early & late)	Cascade	-	-	-	0.60	0.22
East Fork Lewis River (early & late)	Cascade	-	-	-	0.87	0.68
Salmon Creek (late)	Cascade	-	-	-	0.98	0.94
Clackamas River (early & late)	Cascade	0.65	0.67	0.64	0.88	0.90
Sandy River (early & late)	Cascade	-	-	0.94	0.92	0.96
Washougal River (late)	Cascade	-	-	-	0.68	0.25
Hood River (early)	Gorge	-	0.40	0.58	0.25	0.48
Washington Upper Gorge Tributaries/White Salmon River (late)	Gorge	-	-	0.73	0.74	0.76

Other hatchery programs in the Cascade MPG have releases less than 500,000; most operate as integrated programs, except for the Kalama River Hatchery. Hatchery-origin spawners contribute to escapement in a number of basins, substantially so in some basins, while the Salmon Creek, Clackamas River, and Sandy River populations have hatchery-origin spawner rates of less than 10% (Table 34).

Releases into the Gorge MPG have remained fairly steady at slightly over 3 million annually (Figure 66). Natural production in this MPG is limited, and the influence of hatchery-origin fish on the spawning grounds remains higher than in other regions (Table 34).

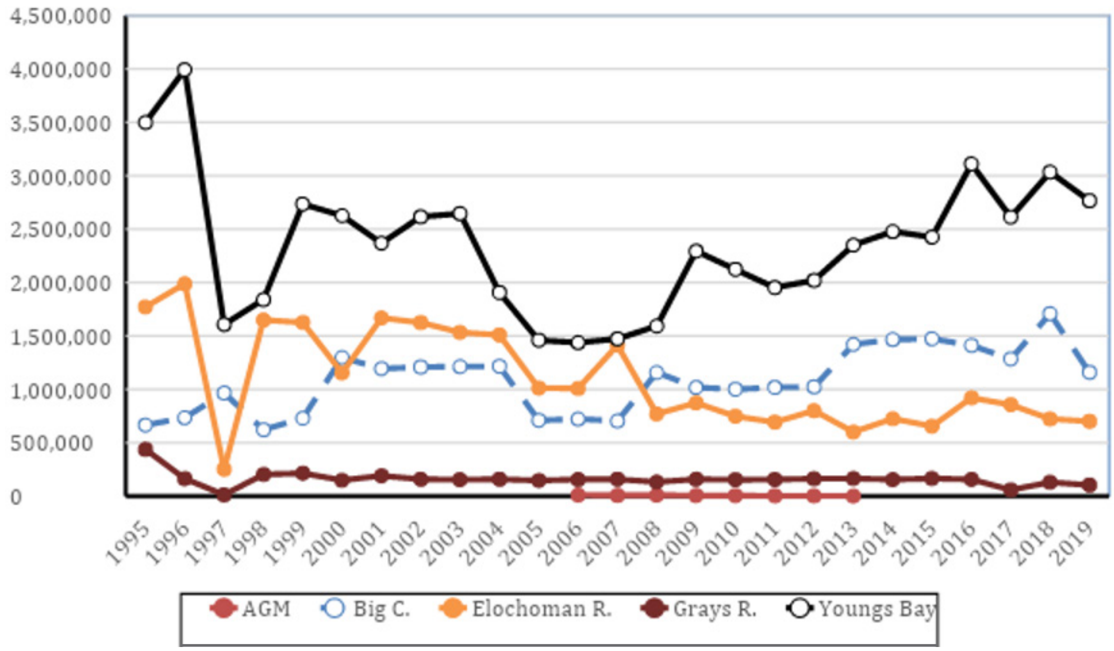

Figure 64. Annual releases of juvenile coho salmon into the Coastal MPG of the Lower Columbia River coho salmon ESU, 1995–2019. All releases were from sources within the ESU. AGM = Abernathy/Germany/Mill Creeks. Releases of fish weighing <2.5 g are not included. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

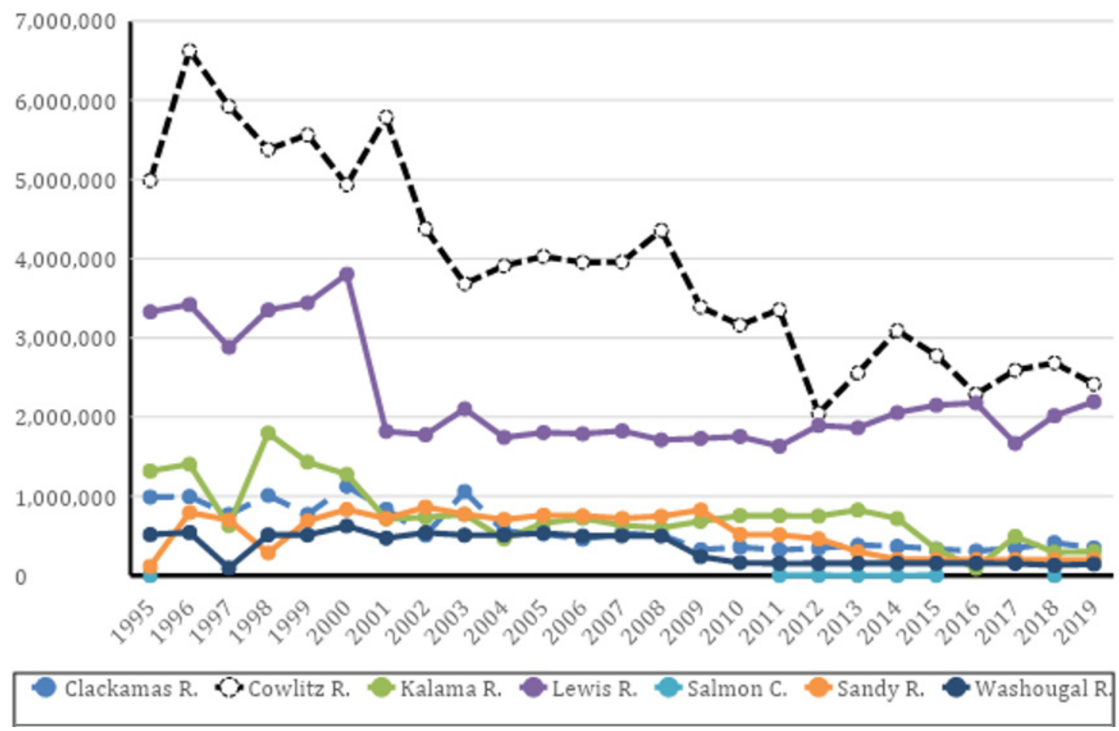

Figure 65. Annual releases of juvenile coho salmon into the Cascade MPG of the Lower Columbia River coho salmon ESU, 1995–2019. All releases were from sources within the ESU. Releases of fish weighing <2.5 g are not included. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

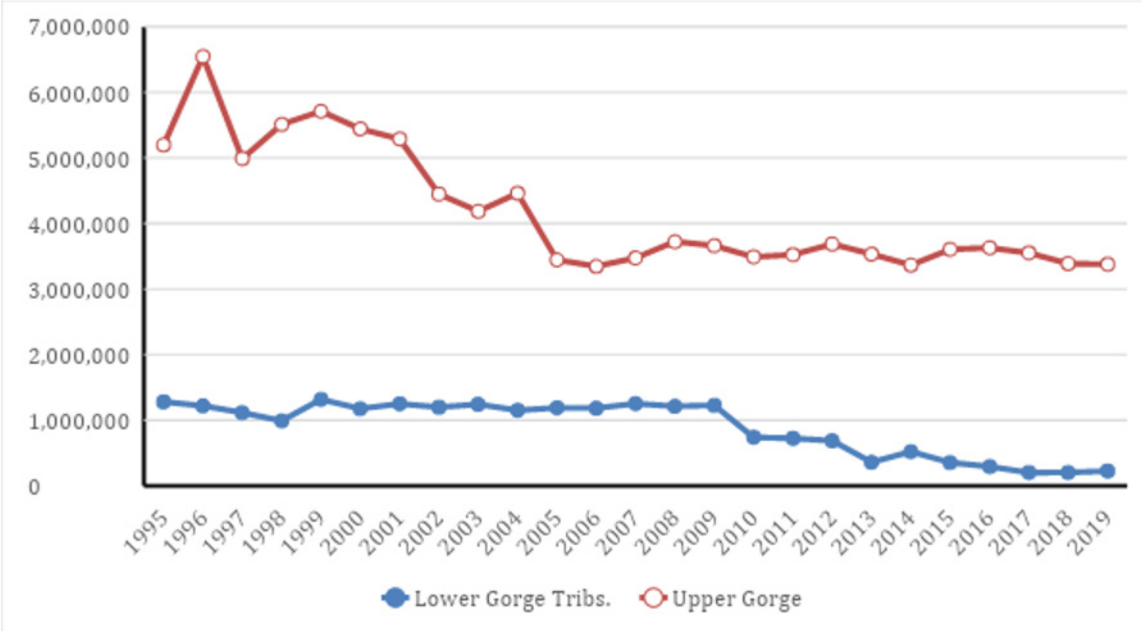

Figure 66. Annual releases of juvenile coho salmon into the Gorge MPG of the Lower Columbia River coho salmon ESU, 1995–2019. Upper Gorge releases include those from the Little White Salmon NFH and Klickitat Hatchery. All releases were from sources within the ESU. Releases of fish weighing <2.5 g are not included. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

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, its removal is thought to eliminate passage delays and injuries. Condit Dam, on the White Salmon River, was removed in 2011, providing access to previously inaccessible habitat. Current monitoring is limited, but screw trap results indicate that coho salmon are successfully spawning in the White Salmon River (Jezorek and Hardiman 2018). 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 were initially poor, but have improved considerably, with the 2019 juvenile collection rate estimated at 64% (PacifiCorp and PUDCC 2020). Nearly 150,000 juvenile coho salmon were produced and collected from the upper North Fork Lewis River. Similarly, efforts to provide downstream juvenile passage at the Cowlitz Dam complex collection sites began in the 1990s, and since that time there have been a number of modifications in the facilities at Cowlitz Falls Dam. Juvenile collection efficiency for coho salmon at the Cowlitz Falls facility in 2019 was 90.4% (Rubenson et al. 2019). Coho salmon from the Tilton River are collected separately at Mayfield Dam. A trap-and-haul program also currently maintains access to the North Toutle River above the SRS, with 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; however, there are limited release sites and only a portion of the upper watershed is accessible. Fish access to the upper Clackamas River basin continues to improve, with recent (2019) estimates for fish guidance efficiency of 94.1% at the North Fork Dam (Ackerman and Pyper 2020). Improvements in juvenile collection on the Clackamas River at Portland General Electric projects, with nearly 200,000 juvenile coho salmon collected annually, are likely to result in increased abundances in the future under more productive ocean conditions. On a more general basis, there have been a number of actions throughout the ESU to remove or improve culverts and other small-scale passage barriers.

There have been incremental improvements in spatial structure during this review period, but poor ocean and freshwater conditions have been such as to mask any benefits from these activities. Similarly, fish passage at culverts has improved, with 132 km (79 mi) of stream habitat being opened up in Washington State alone since 2015 (LCFRB 2020), but a large number of small-scale fish barriers still remain to be upgraded or removed.

Biological status relative to recovery goals

In contrast to the previous status review update, which occurred at a time of near-record returns for several populations, the ESU's abundance has declined during the last five years. Only six of the 23 populations for which we have data appear to be above their recovery goals (Table 35). This includes the Youngs Bay and Big Creek DIPs, which have very low recovery goals, and the Tilton River and Salmon Creek DIPs, which were not assigned goals but have relatively high abundances. Of the remaining DIPs in the ESU, three are at 50–99% of their recovery goals, seven are at 10–50% of their recovery goals, and seven are at <10% of their recovery goals (this includes the Lower Gorge DIP, for which there are no data, but it is assumed that the abundance is low). Hatchery production has been relatively stable, and the proportion of hatchery-origin fish on the spawning grounds has increased for some populations and decreased for others. The transition from segregated hatchery programs to integrated local broodstock programs should reduce the risks from domestication and non-native introgression. Spatial structure has improved incrementally, with improved passage programs at several major dams.

Updated biological risk summary

Overall abundance trends for the Lower Columbia River coho salmon ESU are generally negative. Natural spawner and total abundances have decreased in almost all DIPs (Figure 60), and Coastal and Gorge MPG populations are all at low levels, with significant numbers of hatchery-origin coho salmon on the spawning grounds. Improvements in spatial structure and diversity have been slight, and overshadowed by declines in abundance and productivity. In light of the poor ocean and freshwater conditions that occurred during much of this recent review period, it should be noted that some of the populations exhibited resilience and only experienced relatively small declines in abundance (Figure 60). Some populations were exhibiting positive productivity trends during the last year of review, representing the return of the progeny from the 2016 adult return (Figure 61). For individual populations, the risk of extinction spans the full range, from "low" to "very high." Overall, the Lower Columbia River coho salmon ESU remains at "moderate" risk, and viability is largely unchanged from the prior status review.

Table 35. Current 5-year geometric mean of raw natural-origin spawner abundances and recovery targets (Dornbusch 2013) for Lower Columbia River coho salmon demographically independent populations (DIPs). Numbers in parentheses represent total (hatchery- and natural-origin) spawners. Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > *x* < 50%, yellow = 50% > *x* < 100%), green = >100%.

		Abundance	
Stratum	**Population**	**2015–19**	**Target**
Coastal	Grays/Chinook River (WA)	685	1,800
	Youngs Bay (OR)	448	3,208
	Big Creek (OR)	2,622	3,700
	Elochoman/Skamokawa (WA)	1,987	1,200
	Clatskanie River (OR)	819	1,900
	Mill/Abernathy/Germany Creeks (WA)	1,075	1,900
	Scappoose Creek (OR)	631	2,000
Cascade	Lower Cowlitz River (WA)	n/a	2,000
	Coweeman River (WA)	1,932	n/a
	North Fork Toutle River (WA)	43	500
	South Fork Toutle River (WA)	1,275	500
	Upper Cowlitz River (WA)	686	2,000
	Cispus River (WA)	1,546	n/a
	Tilton River (WA)	2,889	11,232
	Kalama River (WA)	854	5,685
	North Fork Lewis River (WA)	174	1,500
	East Fork Lewis River Tule (WA)	n/a	1,900
	Salmon Creek (WA)	45	1,900
	Clackamas River (OR)	29	5,162
	Sandy River FA (OR)	(2,074)	1,031
	Washougal River Tule FA (WA)	914	1,200
Gorge	Lower Gorge Tributaries Tule FA (WA & OR)	4,528	1,200
	Upper Gorge Tributaries Tule FA (WA & OR)	537	1,200
	Hood River SP (OR)	n/a	1,493

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 lower Willamette and Hood rivers, Oregon (inclusive), as well as multiple artificial propagation programs (USOFR 2020). Myers et al. (2006) identified 23 DIPs, including six summer-run and 17 winter-run steelhead populations (Figure 67).

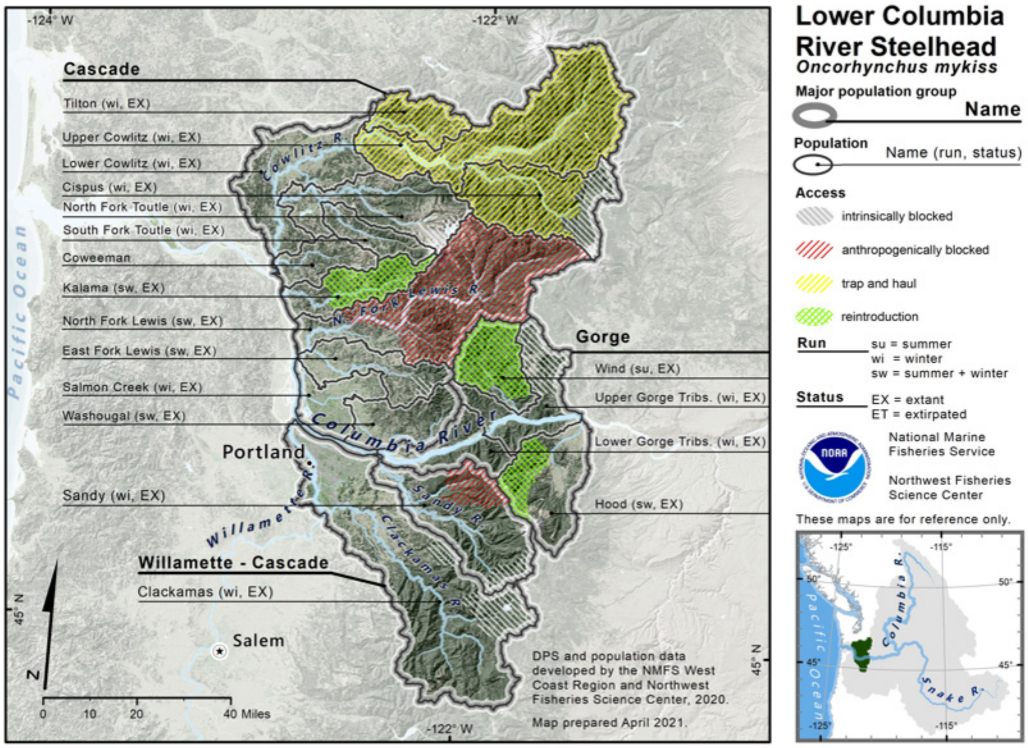

Figure 67. Map of 23 winter and summer-run steelhead demographically independent populations (DIPs) in the Lower Columbia River steelhead DPS. The DPS is separated into two MPGs: Cascade and Gorge. Areas that are accessible (green), accessible only via trap-and-haul programs (yellow), or blocked (cross-hatched), are indicated accordingly.

Summary of previous status conclusions

2005

In 2005, a large majority (73%) of the BRT votes for this DPS 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 Lower Columbia River 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 DPS was currently at “high” risk of extinction. Of the 26 historical populations in the DPS, 17 were considered “high” or “very high” risk. Populations in the upper Lewis, Cowlitz, and White Salmon River 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 River systems, but these had 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.

2015

The 2015 status review update (NWFSC 2015) found that the majority of winter-run steelhead DIPs in this DPS continued to persist, but at low abundances. Hatchery interactions remained a concern in select basins, but the overall situation had somewhat improved compared to prior reviews. Summer-run steelhead DIPs were similarly stable, albeit at low abundance levels. The decline in the Wind River summer-run DIP was a source of concern, given that this population had been considered one of the healthiest of the summer runs in the DPS. Passage programs in the Cowlitz and Lewis River basins were noted to have the “potential” to provide considerable improvements in abundance and spatial structure, but had not produced self-sustaining populations. Low-abundance winter-run returns to the upper Cowlitz River were considered anomalous, related more to a) the development of an integrated hatchery broodstock, and b) 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 offered the opportunity for substantial improvements in the winter-run steelhead population and an opportunity to reestablish summer-run steelhead, but juvenile collection efficiencies were not sufficient to establish viable populations. Habitat degradation continued to be a concern for most populations. Even with modest improvements in the status of several winter-run DIPs, none of the populations was evaluated to be at fully viable status, and similarly none of the MPGs met the criteria for viability. The DPS continued to be at “moderate” risk.

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 adults 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 of total spawner abundance. Where tributaries contained dams or other collection/passage structures, 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 to provide census data; whereas, in the case of Powerdale Dam, limited surveys provide a partial picture of population status for winter-run steelhead, with weirs on the East Fork Hood River and Neal Creek (West Fork Hood River) providing information on summer-run abundance (Simpson 2020).

Abundance and productivity

Cascade MPG (winter run)

This MPG includes native winter-run steelhead in 14 DIPs from the Cowlitz River to the Washougal River, inclusive (Figure 67); however, abundances are only available for ten of them. Of the ten, seven exhibited an increase in five-year geometric means, two exhibited a decrease, and one remains unchanged (Table 36). There is some uncertainty in these abundances, given that six of the ten datasets do not distinguish between natural- and hatchery-origin spawners. The North Fork Toutle River is currently maintained as a natural steelhead gene bank by WDFW (NMFS 2017a), and it may be assumed that the majority of spawners are of natural origin. For most populations, total abundances and natural-origin abundances (where available) have remained low, averaging in the hundreds of fish. Notable exceptions to this were the Sandy and Clackamas River winter-run steelhead populations. The Sandy River winter-run steelhead population experienced a 186% increase in abundance, with a current five-year geomean of 3,615 (Table 36), while maintaining low levels of hatchery-origin steelhead on the spawning grounds (Figure 68). The Clackamas River winter steelhead run was stable, with a five-year geomean of 2,819 (Table 36). Hatchery fish are removed from these rivers at hatchery weirs, in stream weirs, and at North Fork Dam (Whitman et al. 2017). Comparisons of geometric means, however, do not reflect the variation within review periods. There is a strong cyclical pattern in most populations, with a peak in abundance in 2014 and 2015. For most winter-run populations in this MPG, the trend within the 2015–19 period is strongly negative as expressed in annual productivity estimates (Figure 69). There is some concern that this downward trend may be indicative of something more systemic than short-term freshwater or oceanic conditions.

Cascade MPG (summer run)

There are four summer-run steelhead DIPs in the Cascade MPG: Kalama River, North Fork Lewis River, East Fork Lewis River, and Washougal River (Figure 67). Of these, the latter two populations have exhibited declines in abundance, while Kalama River has exhibited abundance increase. Abundance estimates did not distinguish between hatchery- and natural-origin spawners, so there is some uncertainty in the applicability of these trends to the natural population. Summer-run steelhead programs (using non-native Skamania Hatchery-origin broodstock) have been ongoing in both the Kalama and Washougal River basins. The East Fork Lewis River is currently maintained as a natural steelhead gene bank by WDFW (NMFS 2017a), and it may be assumed that the majority of spawners (five-year geomean of 650) are predominately of natural origin. As with the Cascade winter-run steelhead DIPs, there has been considerable annual variability in abundance during the 2015–19 interval, and the current within-census period trend is strongly downward. The North Fork Lewis River is blocked by a series of impassable dams, and, although a trap-and-haul program has been initiated, 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. The recovery

Table 36. Five-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 1 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 2 most recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Coweeman River (winter)	Cascade	(436)	(218)	(458)	(470)	(443)	(528)	(19)
North Fork Toutle River (winter)	Cascade	—	—	—	(449)	(295)	(409)	(39)
South Fork Toutle River (winter)	Cascade	(928)	(344)	(725)	(521)	(432)	(660)	(53)
Upper Cowlitz River (winter)	Cascade	—	(82)	(1,242)	(1,273)	168 (458)	199 (443)	18 (–3)
Tilton River (winter)	Cascade	—	—	(975)	(343)	268 (268)	241 (309)	–10 (15)
Kalama River (winter)	Cascade	(931)	(654)	(1,443)	(1,219)	(866)	(618)	(–29)
East Fork Lewis River (winter)	Cascade	(85)	(214)	(525)	(453)	(356)	(613)	(72)
Clackamas River (winter)	Cascade	1,594 (2,189)	487 (733)	1,371 (1,817)	1,186 (1,599)	2,827 (2,954)	2,819 (3,066)	0 (4)
Sandy River (winter)	Cascade	—	—	—	—	1,263 (1,376)	3,615 (3,858)	186 (180)
Washougal River (winter)	Cascade	(132)	(182)	(479)	(504)	(328)	(427)	(30)
Kalama River (summer)	Cascade	(1,060)	(454)	(382)	(338)	(519)	(561)	(8)
East Fork Lewis River (summer)	Cascade	—	(170)	(402)	(539)	(849)	(650)	(–23)
Washougal River (summer)	Cascade	(220)	(131)	(282)	(612)	(712)	(644)	(–10)
Wind River (winter)	Gorge	—	—	(33)	(16)	(17)	(9)	(–47)
Hood River (winter)	Gorge	—	—	—	—	311 (900)	650 (1,108)	109 (23)
Wind River (summer)	Gorge	(563)	(454)	592 (598)	651 (655)	724 (727)	627 (631)	–13 (–13)

of summer-run steelhead in the Elwha River, apparently by resident *O. mykiss*, suggests a summer run could be reestablished. Although the changes in five-year abundances are not substantial, recent negative trends are of concern. Whether this is a portent of long-term changing oceanic conditions is not clear, but is of some concern regardless of its cause.

Gorge MPG (winter run)

This MPG contains three DIPs: Lower Gorge, Upper Gorge (Wind River), and Hood River. In both the Lower and Upper Gorge, population surveys for winter steelhead are very limited. Abundance levels appear to be improving in the Hood River, with a 109% increase in abundance over the previous review period and a five-year geomean of 651 (Table 36). The development of an integrated hatchery program, in addition to improved access following the removal of Powerdale Dam, may have facilitated the improvement in the Hood River winter run.

Gorge MPG (summer run)

Wind River and Hood River are the two DIPs in the summer run of this MPG. Hood River summer-run steelhead monitoring has been problematic since the removal of Powerdale Dam. Adult abundance in the Wind River has declined since the last review and is trending downward (Table 36, Figure 68). Recent five-year abundance for Wind River summer-run,

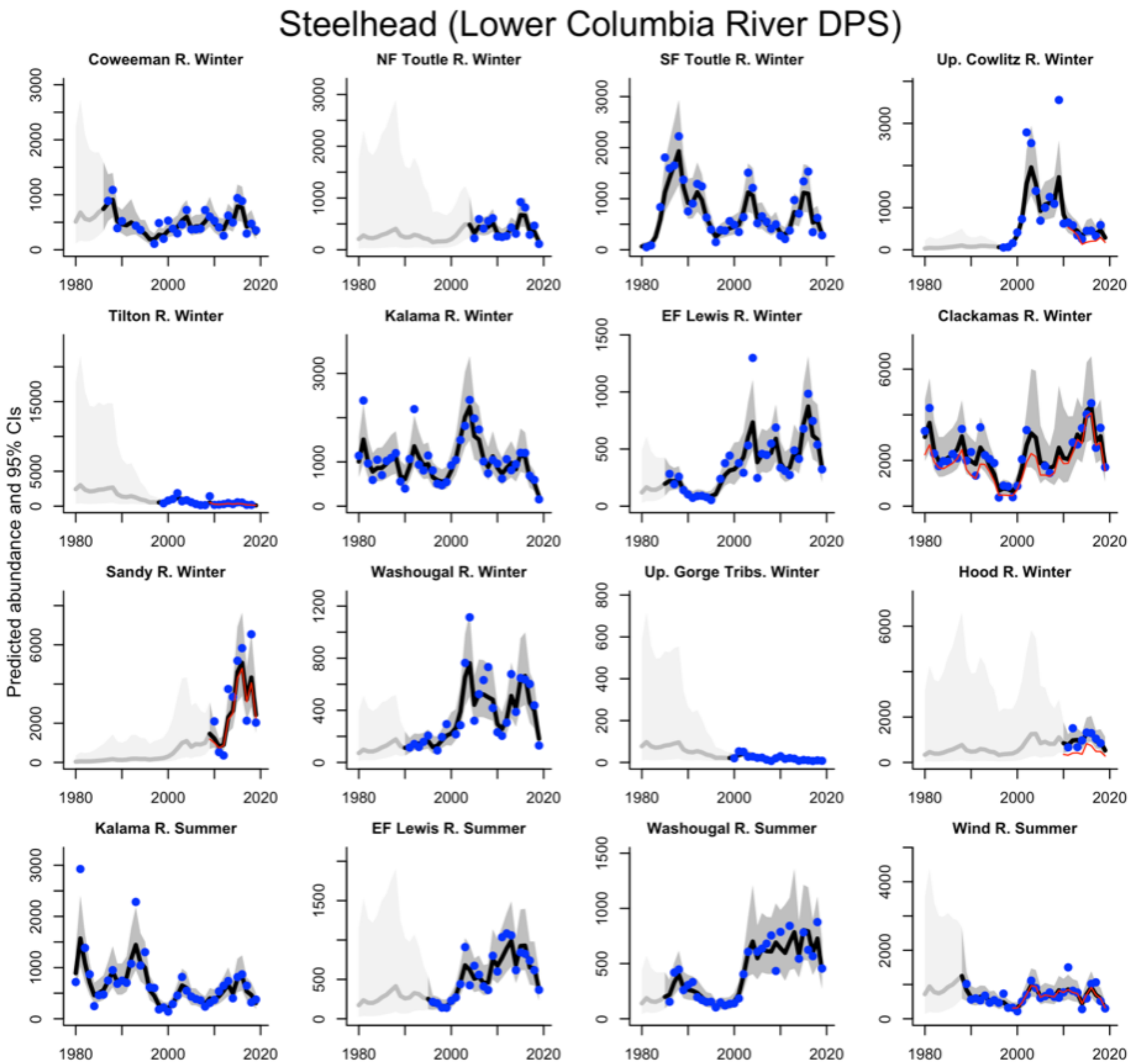

Figure 68. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends the smoothed estimate may be influenced by earlier data points not included in the plot.

a designated natural steelhead gene bank, is 627, a 13% decline from the 2010–14 average (Table 36, Figure 68). The long-term (2005–19) abundance trend for the Wind River is a 2% annual decline (Table 37). Given the presence of only two summer-run DIPs in this MPG and the recent downward trend, the overall status of the MPG is uncertain.

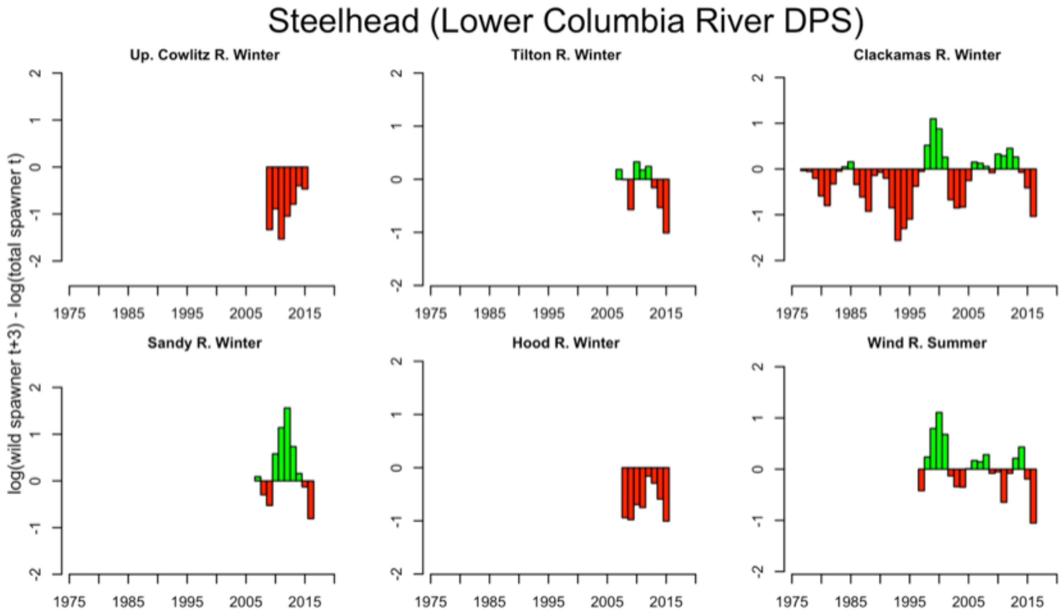

Figure 69. Trends in productivity for demographically independent populations in the Lower Columbia River steelhead DPS, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year ($t-4$).

Table 37. Fifteen-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**	**2004–19**
Clackamas River (winter)	Cascade	0.01 (–0.07, 0.09)	0.06 (0.02, 0.09)
Wind River (summer)	Gorge	—	–0.02 (–0.05, 0.01)

Harvest

Steelhead from this DPS 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 timeframe and primarily affect natural winter steelhead. Winter steelhead harvest in the mainstem from 2015–19 averaged 96 fish annually, primarily from unclipped releases, with an annual rate of 0.3% (ODFW and WDFW 2020a). Similarly, mortalities for unclipped summer-run fish in the lower Columbia River mainstem fisheries averaged 49 fish/year, with mortality rates for unclipped summer-run steelhead of 0.5% in fisheries below Bonneville Dam and 0.01% in the Bonneville Pool. 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 (ODFW and WDFW 2020a).

Recreational harvest of marked hatchery-origin steelhead occurs in most basins and it is likely that non-retention hooking mortality affects most populations in the Lower Columbia River DPS, but that the encounter rate and total mortality are relatively minor.

Hatcheries

Total steelhead hatchery releases in the Lower Columbia River steelhead DPS have decreased slightly since the last status review, declining from an average annual release (summer- and winter-run) of 3 million smolts annually to 2.75 million (Figures 70–72). Some populations continue to have relatively high fractions of hatchery-origin spawners while others have relatively few (Table 38), though data for many populations is not available. WDFW is currently developing a new methodology to assess the hatchery contribution to naturally spawning steelhead. In addition, the North Fork Toutle River, East Fork Lewis River, and Wind River have been established by WDFW as natural gene banks. One of the major changes in hatchery operations was the elimination of the out-of-DPS steelhead broodstock programs in the Kalama River. Previously, out-of-DPS releases were terminated in the Cowlitz and East Fork Lewis Rivers (NWFSC 2015). Out-of-DPS releases continue in the Clackamas River, Sandy River, South Fork Toutle River, and Washougal River with the release of Skamania Hatchery summer-run steelhead. 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.

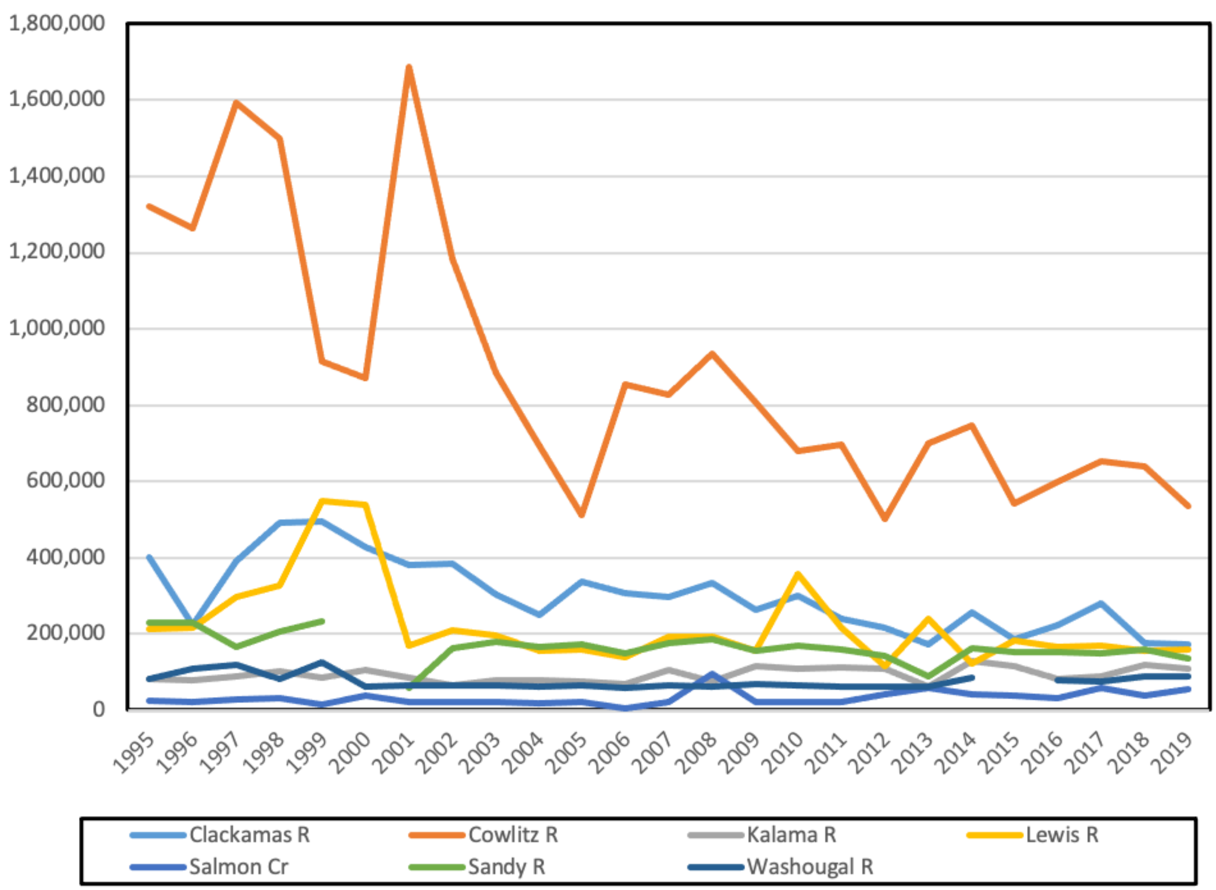

Figure 70. Annual releases of winter-run juvenile steelhead into the Cascade stratum of the Lower Columbia River steelhead DPS, 1995–2019. Data from the Regional Mark Information System (https://www.rmpc.org, March 2020).

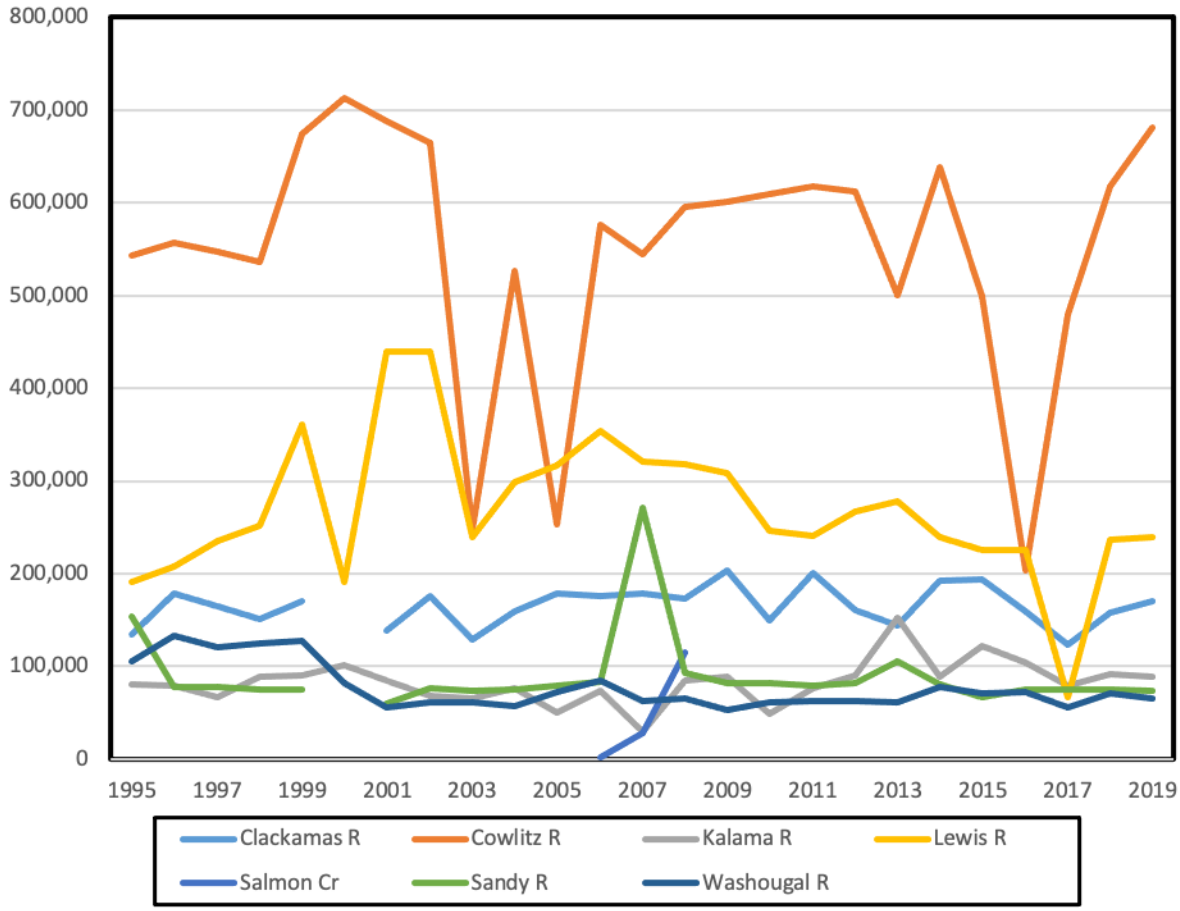

Figure 71. Annual releases of summer-run juvenile steelhead into the Cascade stratum of the Lower Columbia River steelhead DPS, 1995–2019. Data from the Regional Mark Information System (https://www.rmpc.org, March 2020).

There are a number of methods employed to further reduce the incidence of hatchery-origin fish spawning naturally. 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. In some cases, Kalama River hatchery-origin summer-run steelhead are able to ascend the falls and avoid being captured and removed at the fish ladder. In addition, mark-selective recreational fisheries remove some number of hatchery-origin fish from the rivers. Over the years, these actions have incrementally reduced the pHOS from many populations.

In the winter-run Cascade MPG, hatchery releases have remained fairly consistent, with the majority of releases in the Cowlitz River basin (Figure 70). Releases of winter-run steelhead by specific programs into Cowlitz River tributaries have been combined, with distinct programs in the Coweeman, South Fork Toutle, North Fork Toutle, Lower Cowlitz, Tilton, and Upper Cowlitz and Cispus Rivers. Recent changes in hatchery operations—from isolated programs with non-native broodstocks to programs with locally sourced broodstock that continue to integrate natural-origin fish into the broodstock—represent a major effort to decrease the domestication risk from hatchery programs.

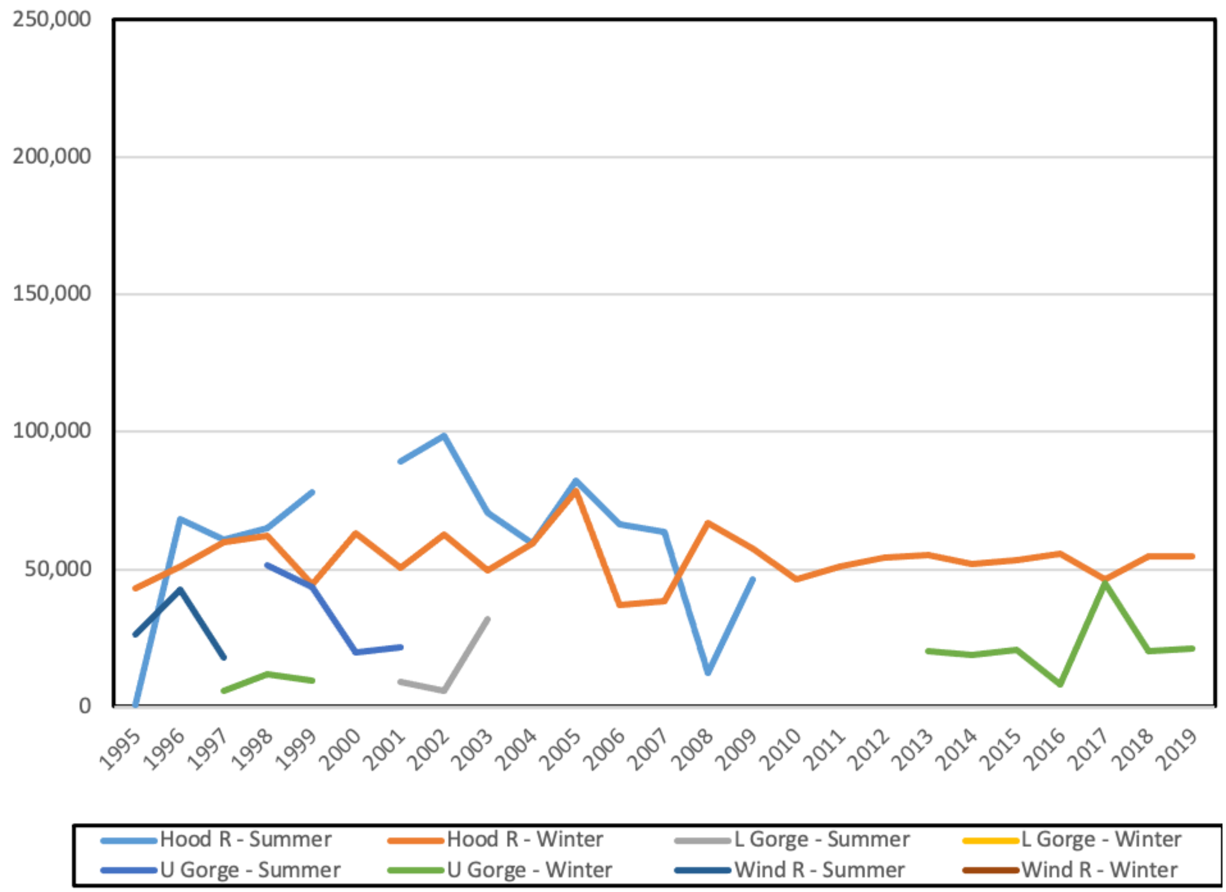

Figure 72. Annual releases of juvenile steelhead into the Gorge stratum of the Lower Columbia River steelhead DPS, by run timing, 1995–2019. Data from the Regional Mark Information System (https://www.rmpc.org, March 2020).

Hatchery releases in the summer-run Cascade MPG have remained fairly steady at 1.3 million fish annually (Figure 71). The majority of these releases are from hatcheries using the out-of-DPS Skamania Hatchery summer-run broodstock, with the exception of the Kalama River integrated summer-run steelhead program. In addition, many of the basins where these fish are released did not historically have summer-run populations, most notably the Cowlitz River basin populations, and the Clackamas and Sandy River DIPs (Myers et al. 2006). The potential effects of these summer-run releases into non-native waters through introgression and competition have been discussed in a number of studies (Kostow and Zhou 2006, Johnson et al. 2018).

There have been limited releases of steelhead into the winter- and summer-run Gorge MPGs (Figure 72), with the predominant program being the integrated winter-run steelhead program in the Hood River. The Wind River, the other major tributary in this area, is designated as a natural gene bank, with minimal hatchery influence.

Table 38. Five-year mean of fraction natural Lower Columbia River steelhead spawners (sum of all estimates divided by the number of estimates), 1995–2019. Blanks (—) indicate that no estimate was available in that 5-year range.

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Upper Cowlitz River (winter)	Cascade	—	—	—	0.70	0.49
Tilton River (winter)	Cascade	—	—	—	1.00	0.79
Clackamas River (winter)	Cascade	0.67	0.76	0.75	0.96	0.92
Sandy River (winter)	Cascade	—	—	—	0.92	0.94
Hood River (winter)	Gorge	—	—	—	0.37	0.61
Wind River (summer)	Gorge	—	—	—	—	—

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 River 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 Cowlitz Falls Dam, most recently in 2017, to improve collection efficiency (Serl and Morrill 2010, Serl et al. 2014, Rubenson et al. 2019). In a recent study, fish collection efficiency (FCE) for steelhead juveniles averaged 81.3%, with fish passage survival being 83.3% for the fish collected (Rubenson et al. 2019). The collection of steelhead kelts remains another area where further improvement is needed. Trap-and-haul operations began on the Lewis River in 2012 for winter-run steelhead, reestablishing access to historically occupied habitat above Swift Dam (RKM 77.1). In the North Fork Lewis River for 2019, the FCE for steelhead collected at the Swift Reservoir Floating Collector was estimated at 27% (PacifiCorp and PUDCC 2020). In the Clackamas River, fish guidance efficiencies for steelhead juveniles at the North Fork Dam were estimated in 2018 at 94.3% under non-spill conditions and 71.7% under spill (Ackerman and Pyper 2019). Juvenile collection in the Clackamas River occurs at River Mill and North Fork Dams, with a combined project collection efficiency of over 95% in 2016–18 (Ackerman and Pyper 2020).

Environmental variability may make it difficult to assess the effects of changes in spatial structure, except through longer-term datasets. These changes 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. 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. Finally, there has been a trap-and-haul operation at the SRS on the North Fork Toutle River since 1989. Transportation above the SRS is limited to two small tributaries, and only a small proportion of the upper basin is utilized (LCFRB 2020). In addition, there have been numerous recovery actions throughout the ESU to remove or improve the thousands of culverts and other small-scale passage barriers.

Biological status relative to recovery goals

Of the 23 DIPs in the Lower Columbia River steelhead DPS, ten are nominally at or above the goals set in the recovery plan (Dornbusch 2013); however, it should be noted that many of these abundance estimates do not distinguish between natural- and hatchery-origin spawners. Notable is the winter-run DIP in the Sandy River, designated a "primary population" in the recovery plan, which is well above its recovery target. Six other primary populations are just above their recovery goals, and, as mentioned above, many of the abundance estimates do not distinguish between hatchery- and natural-origin adults. One population, the Wind River summer-run DIP, is at >50% of its recovery goal, with the remaining 12 DIPs at. Of those 12, abundance data for six DIPs were insufficient for statistical analysis, but presumed to be of low abundance, with one DIP being part of the Upper Cowlitz/Cispus Rivers combined dataset. Both summer- and winter-run MPGs in the Gorge were well below recovery goals. Although the situation in the Cascade stratum is better, improvements in fish passage/collection need to be realized in the Upper Cowlitz, North Fork Toutle, and North Fork Lewis Rivers to achieve recovery goals.

There have been improvements in diversity through hatchery reform, especially the elimination of non-native Chambers Creek winter-run broodstock and some Skamania Hatchery-origin broodstock. Population-specific data on hatchery contribution to the naturally spawning populations is not available for most DIPs, and diversity criteria cannot be properly evaluated without them. Spatial structure remains a concern, especially for those populations that rely on adult trap-and-haul programs and juvenile downstream passage structures for sustainability.

Updated biological risk summary

The majority of winter-run steelhead DIPs in this DPS continue to persist at low abundance levels (hundreds of fish), with the exception of the Clackamas and Sandy River DIPs, which have abundances in the low 1,000s. Although the five-year geometric abundance means are near recovery plan goals for many populations, the recent trends are negative. Summer-run steelhead DIPs were similarly stable, but also at low abundance levels. Summer-run DIPs in the Kalama, East Fork Lewis, and Washougal River DIPs are near their recovery plan goals; however, it is unclear how hatchery-origin fish contribute to this abundance. 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. It is not clear whether the declines observed represent a short-term oceanic cycle, longer-term climatic change, or other systematic issues. While other species in the Lower Columbia River steelhead DPS have a coastal-oriented distribution, steelhead are wide-ranging, and it is more difficult to predict the effects of changes in ocean productivity. Alternatively, most steelhead juveniles remain in freshwater for two years prior to emigration, making them more susceptible to climatic changes in temperature and precipitation.

Spatial structure and abundances are limited due to migrational blockages in the Cowlitz and Lewis River basins. The efficiency of adult passage and juvenile collection programs remain an issue. Recent studies indicate that there have been improvements in juvenile collection efficiency in the Cowlitz River, but these have not been reflected yet in adult abundance.

Table 39. Current 5-year geometric mean of raw natural-origin spawner abundances and recovery targets (Dornbusch 2013) for Lower Columbia River steelhead demographically independent populations (DIPs). Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > x < 50%, yellow = 50% > x < 100%, green = >100%. Numbers in parentheses represent total (hatchery and natural-origin) spawners; * = high uncertainty about whether they are meeting their recovery targets.

		Abundance	
Stratum	**Population**	**2015–19**	**Target**
Cascade	Coweeman River (WA) W	(528)*	500
	NF Toutle River (WA) W	(409)*	600
	SF Toutle River (WA) W	(660)*	600
	Upper Cowlitz River (WA) W	199	500
	Lower Cowlitz River (WA) W	n/a	400
	Cispus River (WA) W	n/a	500
	Tilton River (WA) W	241	200
	Kalama River (WA) W	(618)*	600
	NF Lewis River (WA) W	n/a	400
	EF Lewis River (WA) W	(613)*	500
	Salmon Cr (WA) W	n/a	n/a
	Clackamas River (OR) W	2,819	10,671
	Sandy River (OR) W	3,615	1,519
	Washougal River (WA) W	(427)*	350
	Kalama River (WA) Su	(560)*	500
	NF Lewis River (WA) Su	n/a	500
	EF Lewis River (WA) Su	(650)*	500
	Washougal River (WA) Su	(644)*	500
Gorge	Upper Gorge (Wind R) (WA) W	(9)	n/a
	Lower Gorge (WA & OR) W	n/a	300
	Hood River (OR) W	650	2,079
	Wind River (WA) Su	627	1,000
	Hood River (OR) Su	n/a	2,008

The juvenile collection facilities at North Fork Dam in the Clackamas River appear to be successful enough to support increases in abundance. Hatchery interactions remain a concern in select basins, but the overall situation is somewhat improved compared to prior reviews. It is not possible to determine the risk status of this DPS given the uncertainty in abundance estimates for nearly half of the DIPs. Additionally, nearly all of the DIPs for which there are abundance data exhibited negative abundance trends in 2018 and 2019.

Although a number of DIPs exhibited increases in their five-year geometric means, others still remain depressed, and neither the winter- nor summer-run MPGs are near viability in the Gorge. Overall, the Lower Columbia River steelhead DPS is therefore considered to be at "moderate" risk, and the viability is largely unchanged from the prior review.

Columbia River Chum Salmon ESU

Brief description of ESU

This ESU includes all naturally spawned populations of chum salmon (*O. keta*) in the Columbia River and its tributaries in Washington and Oregon, as well as several artificial propagation programs (Figure 73; USOFR 2020). This ESU is divided into three MPGs with a total of 17 demographically independent populations (DIPs).

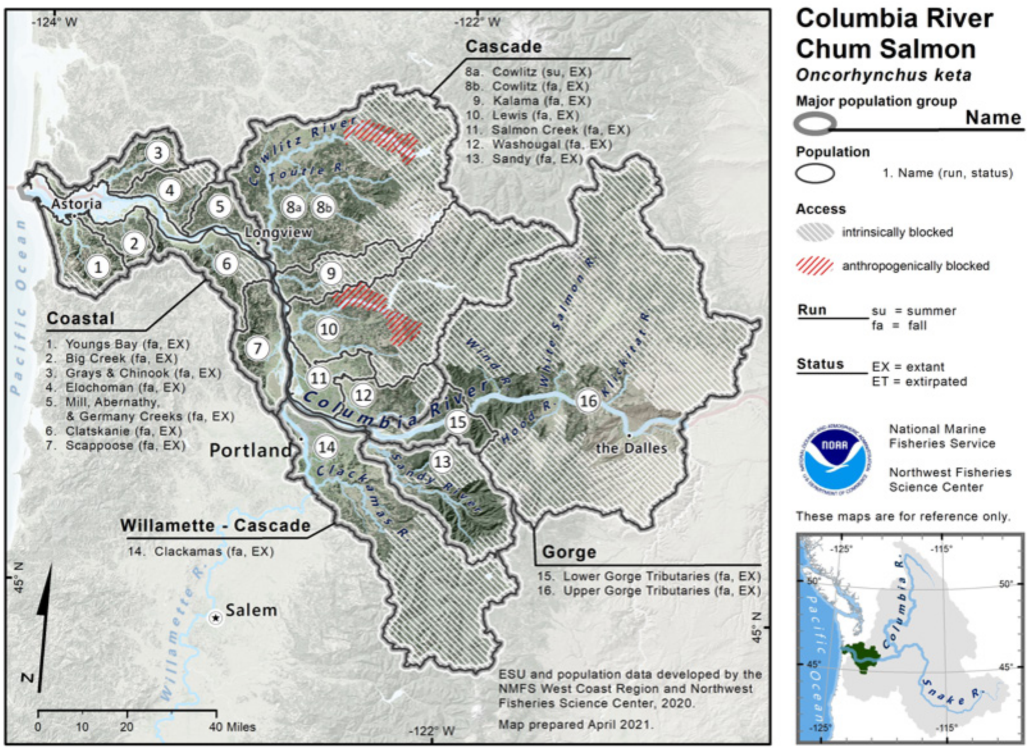

Figure 73. Map of the Columbia River chum salmon ESU's spawning and rearing areas, illustrating all 17 demographically independent populations (DIPs) and the three major population groups (MPGs). Note that Population 8, Cowlitz River, contains two DIPs, a fall and a summer run.

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 had previously identified (Johnson et al. 1997) 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 the 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 it was not determined whether this represented a temporary climate-driven improvement or the beginning of a long-term reestablishment of populations.

2010

Ford et al. (2011) concluded that the vast majority (14 out of 17) chum populations remained extirpated or nearly so. The Grays/Chinook Rivers and Lower Gorge populations exhibited a sharp increase in abundance 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 Rivers showed similar increases in the early 2000s followed by declines, suggesting the increase in chum salmon abundance 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 previous BRT status review in 2005.

2015

NWFSC (2015) found that the majority of the populations in this ESU were at “high” to “very high” risk, with very low abundances. These populations were at risk of extirpation due to demographic stochasticity and Allee effects. One population, Grays/Chinook Rivers, was estimated to be at “low” risk, with spawner abundances in the thousands and demonstrating a recent positive trend. The Washougal River and Lower Gorge populations maintained moderate numbers of spawners and appeared 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. At that time, the potential prospect of poor ocean conditions for the near future was considered a major risk that would put further pressure on these chum salmon populations.

Freshwater habitat conditions were thought to be negatively influencing the 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, continued to be a threat to most chum populations due to projected increases in the population of the greater Vancouver (WA)–Portland (OR) area and the lower Columbia River overall (Metro 2014). The overall viability of this ESU was considered to be relatively unchanged since 2010, and the modest improvements in some populations did not warrant a change in risk category, especially given the uncertainty regarding climatic effects in the near future. This ESU therefore remained at “moderate” to “high” risk.

Description of new data available for this review

In general, most tributaries are surveyed on foot, although chum salmon observations may be incidental to surveys focusing on Chinook or coho salmon (especially late-run 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. As part of its chum salmon restoration program, ODFW monitors fry production in a number of Coastal MPG streams. In general, except where substantial numbers of chum salmon exist, there is limited directed data collection.

Abundance and productivity

Coastal MPG

Grays River

Surveys for chum salmon are regularly conducted in the Grays River. Spawner abundances have exhibited a cyclical pattern, with abundances declining to a few thousand fish in 2013 and 2014, and then peaking in 2016 at a record 30,408 (Figure 74). The current five-year abundance geomean was 10,674, a 70% increase over the previous period (Table 40). Further, productivity estimates for the last review period have been generally positive (Figure 75), as have long-term trends (Table 41). The majority of the returning chum salmon have been naturally produced, 95% on average in 2015–19 (Table 42). The Grays River maintains its position as a stronghold in the MPG and the ESU, with positive short- and long-term trends, despite poor ocean conditions.

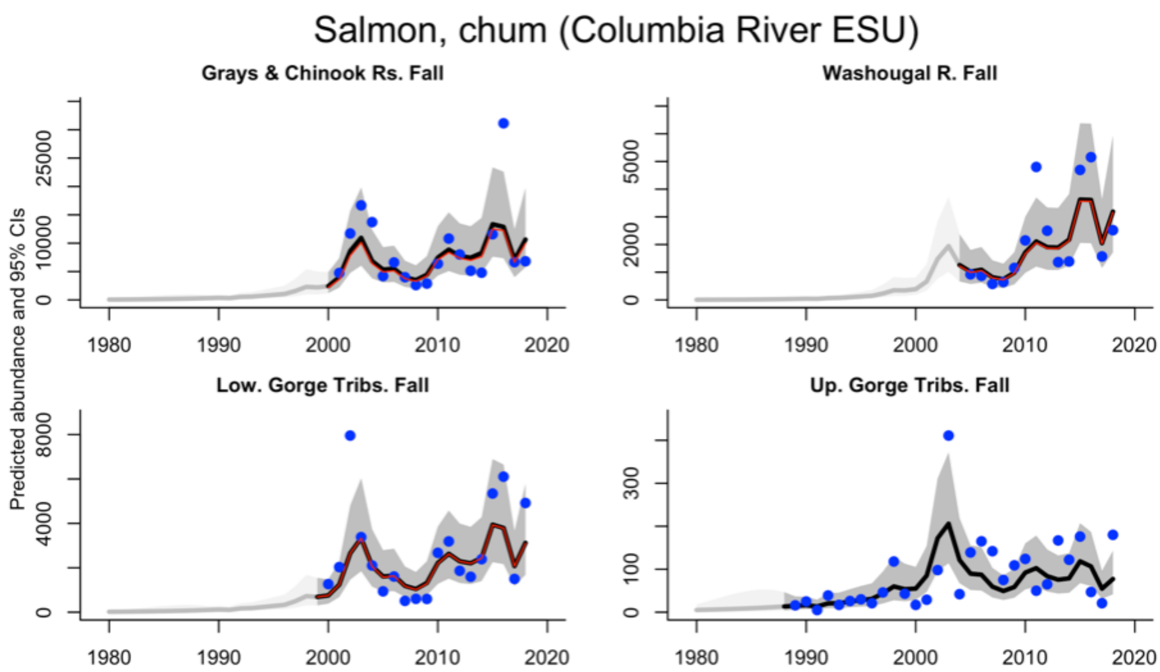

Figure 74. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot. Lower Gorge Tributaries include mainstem Columbia River spawning aggregates (Ives Island, Horsetail Falls, etc.). Upper Gorge Tributaries is based on the Bonneville Dam count, although many chum salmon counted upstream are known to have fallen back and spawned below Bonneville Dam.

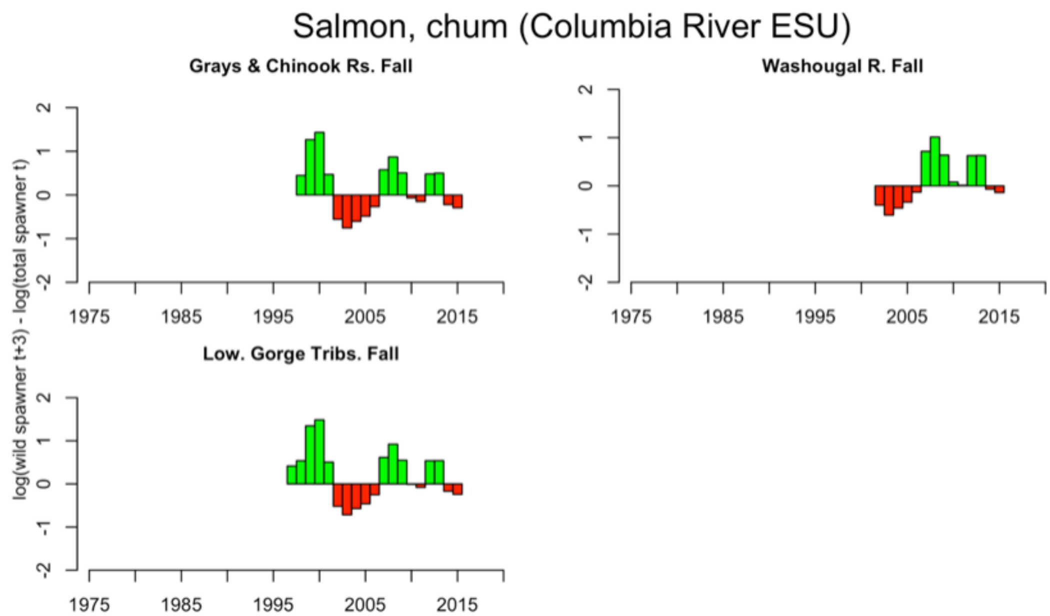

Figure 75. Trends in Lower Columbia River chum salmon demographically independent population (DIP) productivity, 2000–15, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year (*t* – 4).

Table 40. Five-year geometric mean of natural-origin spawner (NOS) counts for Lower Columbia River chum salmon. The Upper Gorge abundance is the dam count at Bonneville Dam and not a spawner estimate. 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 2 most recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Grays/Chinook Rivers FA	Coastal	—	—	9,770 (10,616)	3,589 (3,838)	6,285 (6,709)	10,674 (11,310)	70 (69)
Washougal River FA	Cascade	—	—	—	1,004 (808)	(2,176)	2,703 (3,127)	(44)
Lower Gorge Tributaries FA	Gorge	—	—	2,707 (2,707)	754 (773)	2,263 (2,272)	3,925 (3,938)	73 (73)
Upper Gorge Tributaries FA	Gorge	(18)	(43)	(61)	(122)	(96)	(75)	(–22)

Other Coastal Range DIPs

Adult chum salmon are intermittently observed in very low numbers in most tributaries other than the Grays River or Big Creek, but insufficient data are available for meaningful statistical analysis. Supplementation and reintroduction efforts using surplus hatchery broodstock are underway in a number of tributaries in this MPG, and outmigrating fry have been observed. Most notably, the return of nearly 1,000 unmarked adults to Big Creek in 2020 is likely a result of the hatchery reintroduction program in that basin. The origin of adult chum salmon intermittently observed in other tributaries in this MPG—whether strays from larger populations, supplementation/reintroduction efforts, or locally produced—is uncertain.

Table 41. Fifteen-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	2004–19
Grays/Chinook Rivers FA	Coastal	—	0.07 (0.04, 0.11)
Washougal River FA	Cascade	—	0.11 (0.07, 0.14)
Lower Gorge Tributaries FA	Gorge	—	0.07 (0.04, 0.11)

Cascade MPG

Washougal River

The Washougal River chum salmon DIP includes two spawning aggregates in the mainstem Columbia River just upstream of the I-205 bridge in areas influenced by groundwater seeps (Myers et al. 2006). Population abundance has fluctuated considerably, likely following changes in ocean conditions, with stronger returns in 2015 and 2016 and a decline in 2017–18 (Figure 74). The five-year abundance geomean for 2015–19 was 2,703 (Table 40), with productivity being positive for this period (Figure 75). As with many of the other chum salmon populations, Washougal River chum salmon experience highly variable return rates, but the overall long-term abundance trend has been strongly positive at 11% (Table 41).

Other Cascade Range DIPs

There are reports of chum salmon in a number of tributaries, although systematic surveys are not undertaken. Chum salmon have also been collected at a number of hatcheries and weirs throughout this MPG, but only in very limited numbers. 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-run chum salmon in the Cowlitz River, primarily at the Cowlitz Salmon Hatchery trap.

Gorge MPG

Lower Gorge Tributaries

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 the St. Cloud area along the Washington shoreline. Recent abundances are, on average, improved since the last status review, with peak returns of 5,345 in 2015 and 6,103 in 2016 (Figure 74), compared with the recent five-year abundance geomean of 3,925 (Table 40). The overall medium-term trend since 2005 is positive, 7% (Table 41), with a 73% increase in the recent five-year abundance geomean (Table 40).

Upper Gorge Tributaries

There are no dedicated surveys for chum salmon adults in the upper Gorge MPG; estimates are based on chum salmon migrating past Bonneville Dam to the upper Gorge population area. The chum salmon adult geometric average for 2015–18 was 75, with a 2019 count of 316 (data from University of Washington,[11] July 2020). Interpretation of the Bonneville Dam counts is somewhat problematic given the large naturally spawning chum salmon aggregations just below the dam. In addition, spawning above Bonneville Dam is thought to be very limited due to the loss of historical spawning areas currently inundated in the Bonneville Pool; however, chum salmon fry have been observed at the Bonneville Dam juvenile monitoring facility.

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 0.3% in 2018 (ODFW and WDFW 2020a). 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 during their first spring (Figure 76). 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 collected at the Big Creek weir are transferred to adjacent tributaries (e.g., Bear Creek), although the natural return was very low until 2020. 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. ODFW operates an egg box program in Coastal MPG tributaries; fry production is monitored, as is adult return in these small tributaries. Analysis of adult returns suggest that hatchery production represents a small proportion of adult returns (Table 42).

[11] http://www.cbr.washington.edu/

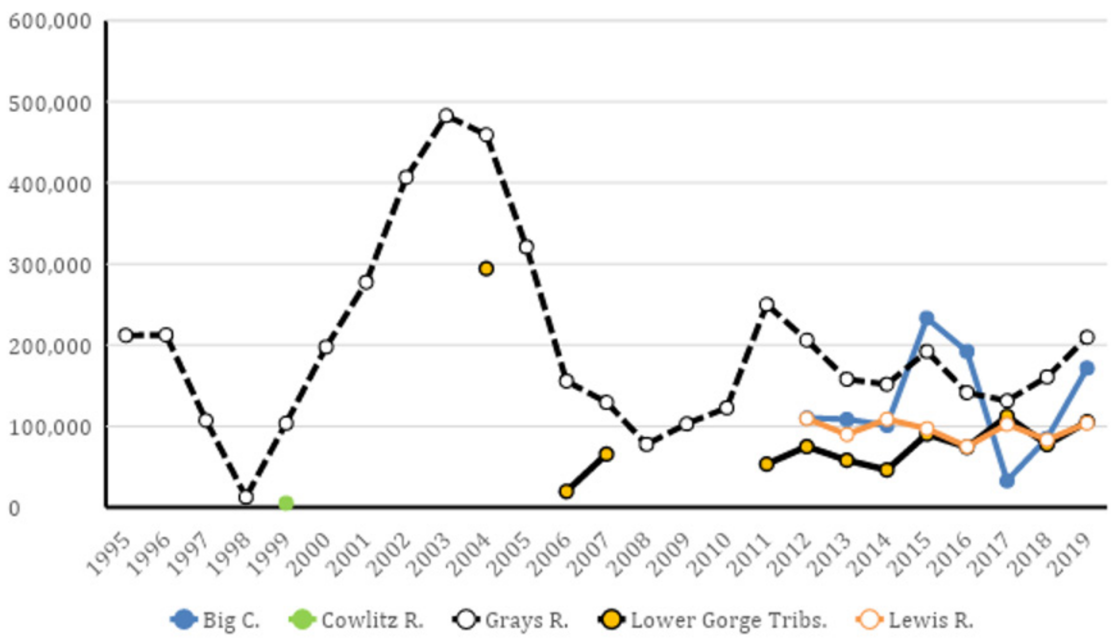

Figure 76. Releases of juvenile chum salmon into the lower Columbia River. All releases were from sources originating from within the ESU. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

Table 42. Five-year mean of fraction natural-origin spawner (sum of all estimates divided by the number of estimates) in lower Columbia River chum salmon populations. Blanks (—) indicate that no estimate was available in that 5-year range.

Population	MPG	1996–2000	2001–05	2006–10	2011–15	2016–20
Grays/Chinook Rivers FA	Coastal	—	0.92	0.95	0.93	0.95
Washougal River FA	Cascade	—	0.98	0.97	—	0.99
Lower Gorge Tributaries FA	Gorge	1	0.99	0.98	1.00	1.00
Upper Gorge Tributaries FA	Gorge	—	—	—	—	—

Spatial structure

Chum salmon generally spawn in the mainstem Columbia River (in areas of groundwater seeps) and the lower reaches of both large and small tributaries, with the exception of the Cowlitz River (Myers et al. 2006). In contrast to other species, mainstem dams have less of an effect on chum salmon distribution; rather, it is smaller, stream-scale blockages that limit chum access to spawning habitat. Upland development can also affect the quality of spawning habitat by disrupting the groundwater upwelling that chum prefer. In addition, juvenile habitat has been curtailed through dikes and revetments that block access to riparian areas that are normally inundated in the spring. Loss of lower river and estuary habitat probably limits the ability of chum salmon to expand and recolonize historical habitat. Presently, detectable numbers of chum salmon persist in only four of the 17 DIPs, a fraction of their historical range.

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. A total of three of 17 populations exceed the recovery goals established in the recovery plan (Dornbusch 2013). The remaining populations have unknown abundances, although it is reasonable to assume that the abundances are very low and unlikely to be more than 10% of the established recovery goals. Although the Big Creek DIP is currently supported by a hatchery supplementation program, natural-origin returns have been very low. Even with the improvements observed during the last five years, the majority of DIPs in this ESU remain at a “very high” risk level. With so many primary DIPs at near-zero abundance, none of the MPGs could be considered viable.

Updated biological risk summary

It is notable that during this most recent review period, the three populations (Grays River, Washougal, and Lower Gorge DIPS) improved markedly in abundance. Improvements in productivity were observed in almost every year during the 2015–19 interval (Figure 74). This is somewhat surprising, given that the majority of chum salmon emigrate to the ocean as subyearlings after only a few weeks, and one would expect the poor ocean conditions to have a strong negative influence on the survival of juveniles (as with many of the other ESUs in this region). In contrast to the three DIPs, the remaining populations in this ESU have not exhibited any detectable improvement in status. Abundances for these populations are assumed to be at or near zero, and straying from nearby healthy populations does not seems sufficient to reestablish self-sustaining populations (Table 43). It may be that the chum salmon life-history strategy of emigrating post-emergence en masse (possibly as a predator swamping mechanism) requires a critical number of spawners to be effective.

Of the risk factors considered, 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. Recent studies also suggest that a freshwater parasite, *Ceratonova shasta*, may be limiting the survival of juvenile chum salmon (WDFW 2019). The prevalence of this parasite may increase with warmer water temperatures from flow modification or climatic change. 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 improvements in some populations do not warrant a change in risk category, especially given the uncertainty regarding climatic effects in the near future. The Lower Columbia River chum salmon ESU therefore remains at "moderate" risk of extinction, and the viability is largely unchanged from the prior review.

Table 43. Current five-year geometric mean of raw natural-origin spawner abundances and recovery targets (Dornbusch 2013) for Lower Columbia River chum salmon demographically independent populations (DIPs). Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > x < 50%, yellow = 50% > x < 100%, green = >100%. Numbers in parentheses represent total (hatchery and natural-origin) spawners.

		Abundance	
Stratum	**Population**	**2015–19**	**Target**
Coast	Youngs Bay FA (OR)	n/a	<500
	Grays/Chinook River FA (WA)	10,027	1,600
	Big Creek FA (OR)	n/a	<500
	Elochoman/Skamokawa FA (WA)	n/a	1,300
	Clatskanie River FA (OR)	n/a	1,000
	Mill/Abernathy/Germany Creeks (WA)	n/a	1,300
	Scappoose Creek (OR)	n/a	1,000
Cascade	Cowlitz River SU (WA)	n/a	900
	Cowlitz River FA (WA)	n/a	900
	Kalama River FA (WA)	n/a	900
	Lewis River FA (WA)	n/a	1,300
	Salmon Creek FA (WA)	n/a	n/a
	Clackamas River FA (OR)	n/a	500
	Sandy River FA (OR)	n/a	1,000
	Washougal River (WA)	3,003	1,300
Gorge	Lower Gorge FA (WA & OR)	3,124	2,000
	Upper Gorge FA (WA & OR)	(85)	900

Upper Willamette River Chinook Salmon ESU

Brief description of ESU

The ESU includes all naturally spawning populations of spring-run Chinook salmon in the Clackamas River, the Willamette River (and its tributaries) above Willamette Falls, Oregon, and several artificial propagation programs (Figure 77; USOFR 2020). Seven demographically independent populations (DIPs) were identified by the TRT (Myers et al. 2006).

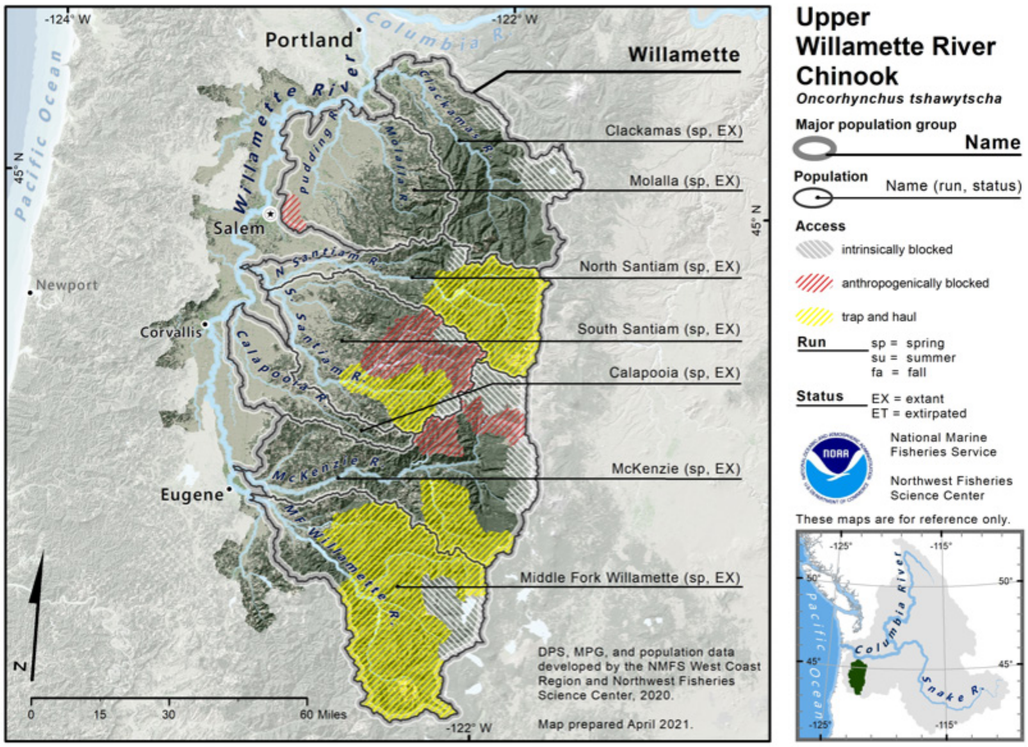

Figure 77. Map of the seven demographically independent populations (DIPs) within the Upper Willamette River Chinook salmon ESU. Areas that are accessible (green), accessible only via trap-and-haul programs (yellow), or blocked (cross-hatched), are indicated accordingly.

Summary of previous status conclusions

2005

NMFS initially reviewed the status of the Upper Willamette River Chinook salmon ESU in 1998 (Myers et al. 1998) and in an update that same year (NMFS 1999a). In the latter update, the BRT was concerned about the few remaining naturally spawning populations of spring-run Chinook salmon in the ESU, and the high proportion of hatchery fish in the remaining runs. The 1998 BRT noted that the ODFW was able to identify only one remaining naturally reproducing population in this ESU, McKenzie River spring-run Chinook salmon.[12] The 1998 BRT was concerned about severe declines in short-term abundance that occurred

[12] In 1998, the Clackamas River also contained a naturally spawning spring-run Chinook salmon population, but ODFW did not recommend the inclusion of the Clackamas River in the ESU at that time.

throughout the ESU, and that the McKenzie River population had declined precipitously, indicating that it might 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 past 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 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 Willamette–Lower Columbia Technical Recovery Team (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 Upper Willamette River Chinook salmon had been conducted since the previous status review 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 "very high risk." The remaining two (Clackamas and McKenzie Rivers) were considered to be at "moderate" to "low" risk. The new data reviewed in 2010 also highlighted the substantial risks associated with prespawning mortality. Although the recovery plans that were being developed at that time targeted key limiting factors for future actions (specifically passage at high-head dams), in 2010 there had not yet been any significant on-the-ground actions 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 previous BRT status review in 2005.

2015

The 2015 status review update (NWFSC 2015) determined that, although overall abundance of natural-origin spawners was somewhat lower than in 2010, the risk status remained unchanged. Adult abundance data were limited in some cases. Of the seven DIPs in this ESU, two—the Molalla and Calapooia Rivers—are not surveyed to any extent, and a third, the Clackamas River, is represented by a dam count of returning adults, although some spawning ground surveys were available. The remaining four populations were comprehensively surveyed in a systematic manner. Data collected included: dam counts (prespawning counts), multiple spawner and redd counts, and carcass recoveries. Prespawn mortality was considerable in some years and reduced the relevancy of adult dam counts, underscoring the importance of on-the-ground surveys. In 2015, major dams continued to reduce or eliminate adult access to well over 50% of the historical upstream spawning habitat in the Clackamas, North Santiam, South Santiam, McKenzie, and Middle Fork Willamette Rivers. Furthermore, the results from genetic pedigree studies indicated variable but low reproductive success for Chinook salmon reintroduced above the dams (<1:1 spawner-to-recruit ratio). Juvenile passage and survival data at the dams are limited and highly variable. Only at Foster Dam (South Santiam River DIP) and Fall Creek Dam (Middle Fork Willamette River DIP) were spawner replacement levels achieved in some years.

The apparent decline in the status of the McKenzie River DIP was 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 high-quality habitat above Cougar Dam (South Fork McKenzie River) was still limited by poor downstream juvenile passage. Additionally, the installation of a temperature control structure in Cougar Dam in 2005 was thought to benefit downstream spawning and rearing success. For the 2010–15 period, natural-origin returns to the Clackamas River remained flat, despite adults having access to much of their historical spawning habitat. Although adults returning to the Calapooia and Molalla River basins are not impeded by dams, habitat conditions (primarily summer temperatures) are such that the productivity of these systems was thought to be very low (near zero). Natural-origin spawners in the Middle Fork Willamette River consisted almost entirely of adults returning to Fall Creek. The abundance of returning natural-origin adults to the Fall Creek Dam numbered in the hundreds; however, marginal habitat conditions resulted in high prespawn mortality and reduced the number of spawners to a half or a third of that number. Finally, improvements were noted in the North and South Santiam River DIPs. This 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-origin spawners in the North Santiam River are still confined to below Detroit Dam and subject to relatively high prespawn mortality rates in accessible reaches below the Big Cliff/Detroit Dam complex, potentially related to high levels of total dissolved gas.

Although there was an overall decrease in the VSP status of the ESU since the previous review (Ford et al. 2011), the magnitude of this change was not sufficient to suggest a change in risk category. Climatic conditions at the time of the review (drought and warm ocean waters) and the prospect of long-term climatic change, in conjunction with the inability of many populations to access historical headwater spawning and rearing areas, were considered major near-and long-term risks to this ESU.

Description of new data available for this review

Through 2017, ODFW conducted comprehensive spawner surveys (redds and carcasses) both below and above dams in the North Santiam, South Santiam, McKenzie, and Middle Fork Willamette Rivers. In the McKenzie River basin, comprehensive surveys were done in 2018 and 2019. Only partial surveys could be done in 2020 because of forest fires (surveys were done on the mainstem below Leaburg, Horse Creek, and the restoration area in the South Fork McKenzie River). Wild carcasses were sampled in these areas. In the North Santiam River basin, comprehensive surveys were done above the dams in 2018 and 2019. Sections below the dams were surveyed only for peak redd counts and distribution. In the South Santiam River basin, comprehensive surveys were done above the dams in 2018–20. Sections below the dams were surveyed only for peak redd counts and distribution, with carcasses sampled in 2020. Comprehensive surveys were done in the Clackamas and Molalla River basins from 2015–19 (surveys could not be done in 2020 because of forest fires). In the Middle Fork Willamette River basin, surveys were done for peak redd counts in 2018, and collaborators at OSU did some surveys in the basin in 2020. Collaborators at OSU also surveyed the Middle Fork Willamette River in 2020.

Direct adult counts are also made at Willamette Falls, Bennett Dam, and Minto Fish Facility (North Santiam River), Foster Fish Facility (South Santiam River), Leaburg and Cougar Dams and the McKenzie Hatchery (McKenzie River), and Fall Creek Dam and Dexter Fish Facility (Middle Fork Willamette River). Intermittent spawner surveys have been conducted in the Molalla and Calapooia Rivers, but are insufficient to estimate population abundance. Beginning in 2018, there has been a transition in the methodology and extent of adult spawner surveys. The U.S. Army Corps of Engineers (USACE), which had funded much of the previous survey work, transitioned from ODFW to private contractors, limiting the extent of those surveys. In 2018 and 2019, parallel spawner survey efforts were undertaken by ODFW and Environmental Assessment Services (EAS; NAI 2019). Comparison of results from 2018 indicated considerable disagreement in spawner abundances in the North Santiam, South Santiam, and McKenzie Rivers. For example, in 2018, North and South Santiam River redd counts by EAS were 57.8% higher than ODFW counts, while in the McKenzie River, ODFW redd counts were 49.1% higher than EAS. Surveys in Fall Creek were interrupted by forest fires, although the number of fish passed upstream at Fall Creek Dam is known. After 2019, spawner surveys below dams in the North and South Santiam Rivers, previously funded by USACE, were terminated, and ODFW continued with surveys, although on a more limited scale. The existence of contrasting estimates and the changes in established methodology for spawner data from 2018 and 2019 make interpretation of abundances from foot and boat surveys more difficult. For our analysis, we considered only the ODFW time series estimates in order to preserve survey methodology consistency. Adult estimates from ladder counts at Willamette Falls, the Bennett dams, Leaburg Dam, and fish collection facilities and hatcheries continue to be collected, and provide a consistent reference point for interpreting spawner surveys.

Genetic pedigree studies of adults returning to tributary dams in the Upper Willamette River Chinook salmon ESU have been ongoing at Detroit Dam (North Santiam River), Foster Dam (South Santiam River), Cougar Dam (McKenzie River), and Fall Creek Dam (Middle Fork Willamette River; Banks et al. 2014a, Evans et al. 2016, O'Malley and Bohn 2017). 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. Collection of tissues for genetic analyses is ongoing at adult collection facilities associated with trap-and-haul programs at high-head dams, and from natural fish collected during spawner surveys; however, not all tissue samples will be genetically analyzed each year. Archiving tissue samples further delays any assessment of reproductive success.

Overall, the development of long-term abundance data for four of the DIPs in the Upper Willamette River Chinook salmon ESU provides considerable insight into population abundance and productivity. Stream surveys and carcass recoveries allow the estimation of the contribution of hatchery-origin fish to the naturally spawning population. There is some uncertainty in estimates for 2018 and 2019, and future monitoring will likely be further constrained by limited resources.

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 the first cohort of mass-marked hatchery-reared fish, was it possible to inventory naturally produced fish with any accuracy. Cohorts returning from 2015–19 were strongly influenced by warmer-than-normal and less-productive ocean conditions, in addition to warmer- and drier-than-normal freshwater conditions. The five-year average abundance geomean for 2015–19 was 6,916 natural-origin (unmarked) adults, a 31% decrease from the previous period (Table 44). While there was a substantial downward trend in total and natural-origin spring-run abundance at Willamette Falls (Figure 78), there were some indications of improving abundance in 2019 and 2020. Improvements in abundance corresponded with improved ocean and freshwater conditions, as well as changes in pinniped predation. In recent years, counts of spring-run Chinook salmon at Willamette Falls have been influenced by pinniped

Table 44. Five-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 2 most recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Willamette Falls SP	Willamette	(42,031)	(27,817)	21,833 (68,324)	8,482 (26,529)	9,975 (40,236)	6,916 (32,189)	–31 (–20)
Clackamas River SP	Willamette	1,291 (3,961)	466 (1,430)	2,110 (3,920)	1,482 (1,906)	1,894 (2,013)	3,617 (3,722)	91 (85)
North Santiam River SP	Willamette	—	—	—	333 (1,064)	401 (1,584)	354 (1,424)	–12 (–10)
South Santiam River SP	Willamette	—	—	—	416 (1,281)	613 (1,685)	337 (1,856)	–45 (–10)
McKenzie River SP	Willamette	—	—	—	1,794 (2,856)	1,479 (2,750)	1,664 (2,916)	13 (6)
Middle Fork Willamette River SP	Willamette	—	—	—	—	92 (1,209)	20 (407)	–78 (–66)

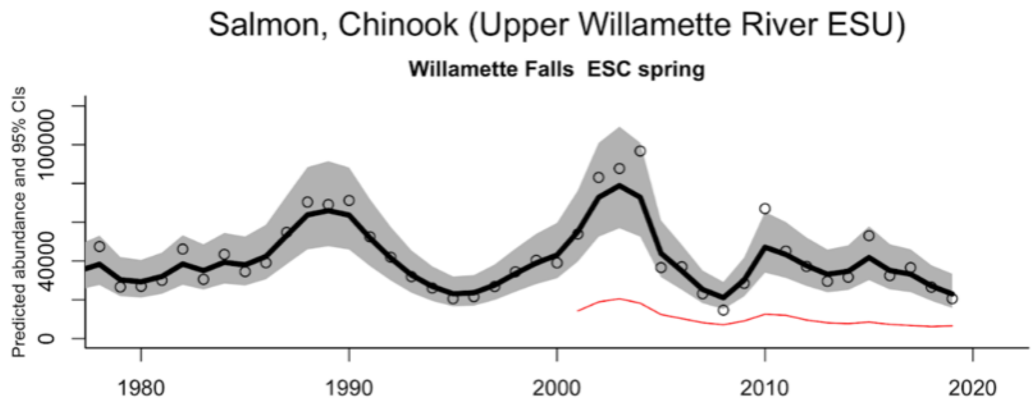

Figure 78. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

Table 45. Fifteen-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 and with at least 2 data points in the first 5 years and last 5 years of the 15-year period are shown.

Population	MPG	1990–2005	2004–2019
Willamette Falls SP	Willamette	—	−0.04 (−0.06, −0.01)
Clackamas River SP	Willamette	0.05 (−0.03, 0.13)	0.06 (0.03, 0.09)
North Santiam River SP	Willamette	—	−0.01 (−0.03, 0.02)
South Santiam River SP	Willamette	—	−0.03 (−0.08, 0.02)
McKenzie River SP	Willamette	—	−0.02 (−0.05, 0.00)

predation at the base of the falls. For the return years 2014–18, pinnipeds were estimated to consume 6–10% of the unmarked Chinook salmon escapement; however, in 2019, when a pinniped removal program was initiated, the rate dropped to approximately 4% (Steingass et al. 2019). Over the last 15 years, the long-term trend for natural-origin returns was −4% (Table 45), suggesting an overall decline in those populations above Willamette Falls.

Clackamas River

Returning spring-run Chinook salmon are enumerated at North Fork Dam, and outmigrating juveniles are collected and counted at River Mill Dam. In contrast to the other populations in this ESU, the recent five-year trend and recent productivity are both positive (Table 44, Figures 79 and 80). The most recent five-year abundance geomean is 3,617, a 91% increase over the 2010–14 period (Table 44). The long-term trend for this population is 6% (Table 45). Improvements in adult returns are likely associated with improvements in the juvenile collection facilities installed by Portland General Electric (PGE) at their dam complex (River Mill, Faraday, and North Fork Dams). A new adult sorting facility was completed in 2013, eliminating the need to handle fish (David et al. 2016). Recent habitat

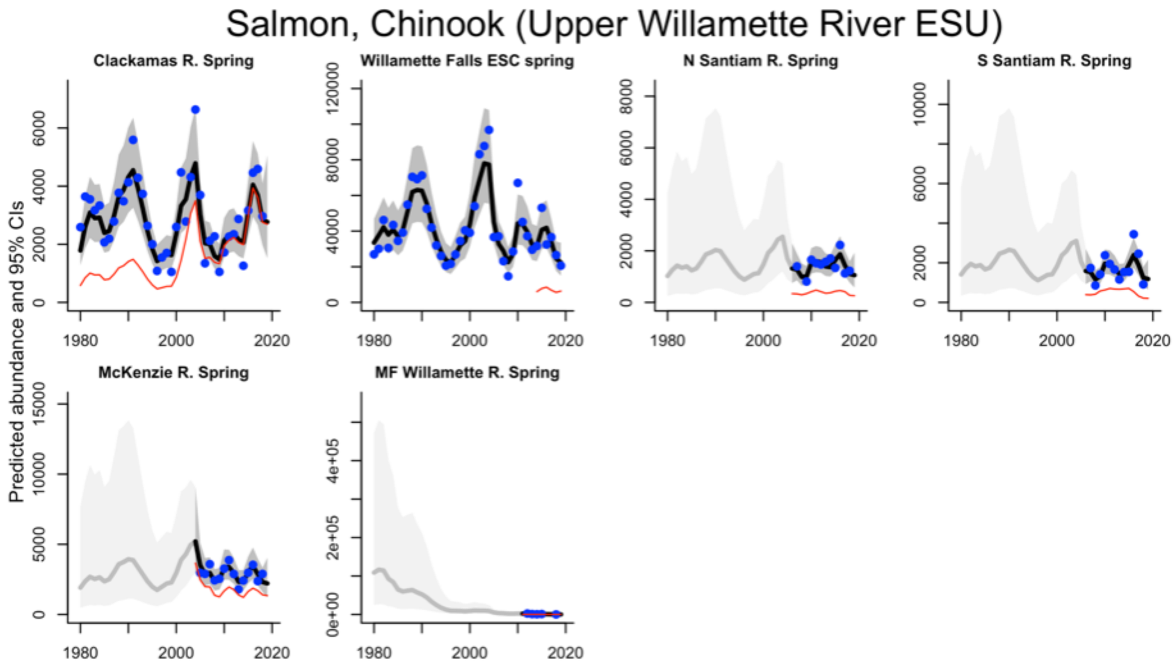

Figure 79. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

restoration projects have been completed in the upper basin. Spawning in the upper Clackamas River is now occurring farther upstream and redds are more widely distributed. Juvenile fish guidance efficiencies for Chinook salmon were recently estimated at 85.5% (Ackerman and Pyper 2020). Given the resilience in adult abundance during the past period of relatively poor ocean conditions, it is expected that the current positive trend will continue into the next five-year review period with improving ocean conditions. Further, the Clackamas River enters the Willamette River below Willamette Falls, and conditions in the mainstem upper Willamette River, above Willamette Falls, may be limiting the other populations in this ESU which share an adult and juvenile migratory corridor.

Molalla River

Chinook salmon surveys were carried out from 2015–19. Low abundances (<100 redds) were observed. A radio tagging study found that only two of the 300 returning Chinook salmon adults tagged at Willamette Falls were detected entering the Molalla River (Jepson et al. 2015). Coded-wire tags from juvenile releases in the Molalla River were recovered in the Molalla River in 2016 (Sharpe et al. 2017). Abundance information is limited to anecdotal reports, recreational catch reports, and recent surveys, all of which are insufficient to provide a useful estimate of abundance; however, it is reasonable to assume that the abundance of natural-origin Chinook salmon is very low.

North Santiam River

Adult natural-origin returns to the North Santiam River, as measured at Bennett Dam and through redd and carcass surveys, have exhibited a decrease in abundance (Figure 79) and have strongly negative productivity (Figure 80). The five-year average abundance geomean for natural-origin spawners was 354, a 12% decrease from the previous period (Table 44). Estimates of NORs at Bennett Dam from 2015–19 ranged from 573 to 1,059 (geometric mean of 849), suggesting either considerable prespawning mortality or an undercount of spawners. Prespawn mortality varies considerably from year to year; in 2015 during an exceptionally warm dry summer, prespawn mortality was estimated to be 63% in the North Santiam River below Detroit Dam, but only 3% in 2016 (Sharpe et al. 2017). 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 Dam are released above the fish barrier to spawn in the North Santiam River reach between Minto and Big Cliff Dams. While this "sanctuary" reach is populated with unmarked adult Chinook salmon, temperature and dissolved gas conditions may contribute to elevated prespawning mortality levels (Sharpe et al. 2018). Further, conditions in the Minto Dam to Big Cliff Dam reach make accurate spawner surveys difficult.

South Santiam River

Spring-run Chinook salmon adults returning to the South Santiam River are monitored via redd counts and carcass recoveries. In addition, direct counts of returning adults are made at the fish collection facility at Foster Dam, where only natural-origin fish are passed above the dam. For the current period, the five-year spawner abundance geomean for the entire South Santiam River was 337, a 45% decrease from 2010–14 (Table 44, Figure 79). The Foster Dam counts, which represent fish migrating to the upper South Santiam River, had a geomean of 305 for this same period; however, this does not account for prespawn mortality or fallbacks. The long-term trend (2015–19) for South Santiam River natural-origin Chinook salmon has been negative, –3% (Table 45). Attempts to improve upstream adult collection with a new adult facility and downstream juvenile passage with a weir gate at Foster Dam have encountered operational difficulties, and instead resulted in decreased adult attraction to the adult collection facility and increased juvenile mortality passing through the weir gate.

Calapooia River

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, and large-scale releases were last made in 1997, although small numbers of fry (<50-mm) were released through 2008. None of the fish that were radio-tagged at Willamette Falls in 2012–14 were detected entering the Calapooia River (Jepson et al. 2015). A few adult Chinook salmon were observed in snorkel surveys in 2012, but it is unclear if they successfully spawned. Since 2012, neither juvenile nor adult Chinook salmon have been observed in annual snorkel surveys in the Calapooia River. Based on the limited information available, it would appear the Calapooia River Chinook salmon population is at a critically low level, at or near zero.

McKenzie River

Within the recent review period, the average natural-origin abundance in the McKenzie River has increased by 13%, to a five-year geomean of 1,664 (Table 44, Figure 79). This improvement in abundance marks a reversal of long-term declines. Still, the long-term trend in abundance (2015–19) is –2% (Table 45). The McKenzie River has been a bellwether for natural production in the upper Willamette River basin, with the majority of historical spawning habitat still accessible. Natural-origin spawners represent the majority of spawners, 57% (Table 46), especially in the upper reaches (NAI 2020). Genetic pedigree-based estimates of cohort replacement rate for the 2007 and 2008 broodyears from hatchery adults released above Cougar Dam were both below replacement, 0.41 and 0.31 respectively (Banks et al. 2014a). Juvenile tagging studies indicate that total survival through Cougar Reservoir and Dam has been poor (Beeman et al. 2013). Currently, multiple options for structural or operational juvenile downstream passage are being investigated, with actualization of a passage strategy still some years off. Additional passage and survival data for juvenile Chinook salmon would help evaluate different passage operations. Habitat restoration efforts by the U.S. Forest Service below Cougar Dam on the South Fork McKenzie River were recently completed, representing a major effort to enhance the floodplain; however, it may be some years before the full measure of success for this effort can be evaluated. Redd counts in the restoration area dramatically increased in 2018 and 2019.

Middle Fork Willamette River and Fall Creek

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. Natural-origin spawner abundance represents redds surveyed below Dexter Dam and above Fall Creek Dam. During the 2015–19 review period, the geomean dropped to 20, a 78% decrease in abundance. Natural-origin spawners (Figure 79) are limited to spawning in the mainstem Middle Fork Willamette River below Dexter Dam, below Fall Creek Dam and Little Fall Creek, where conditions were especially poor during 2015–19, and above Fall Creek Dam, where the majority of natural-origin fish return (Sharpe et al. 2017). Prespawn mortality rates are generally very high, often near 100% below Fall Creek Dam, and only marginally better above Fall Creek Dam (Sharpe et al. 2017, NAI 2019, 2020). Productivity estimates are strongly negative (Figure 80). In addition, the Jones Fire in the Fall Creek watershed in 2017 likely had immediate and long-term effects on fish survival in the basin. Similarly, areas burned in the Willamette River basin in 2019 and 2020 will suffer from the loss of riparian habitat and the deposition of sediment and ash from denuded hillsides. Accessible habitat in the Middle Fork Willamette River is very limited and, until effective upstream and downstream passage past the dams is developed, it is unlikely that abundance will improve markedly.

Table 46. Five-year mean of fraction natural-origin Chinook salmon spawning naturally in the Upper Willamette River Chinook salmon ESU (sum of all estimates divided by the number of estimates). Blanks (—) mean no estimate available in that 5-year range.

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Willamette Falls SP	Willamette	—	0.24	0.30	0.24	0.22
Clackamas River SP	Willamette	0.33	0.58	0.79	0.94	0.97
North Santiam River SP	Willamette	—	—	0.33	0.26	0.26
South Santiam River SP	Willamette	—	—	0.39	0.40	0.21
McKenzie River SP	Willamette	—	—	0.64	0.55	0.57
Middle Fork Willamette River SP	Willamette	—	—	—	0.08	0.07

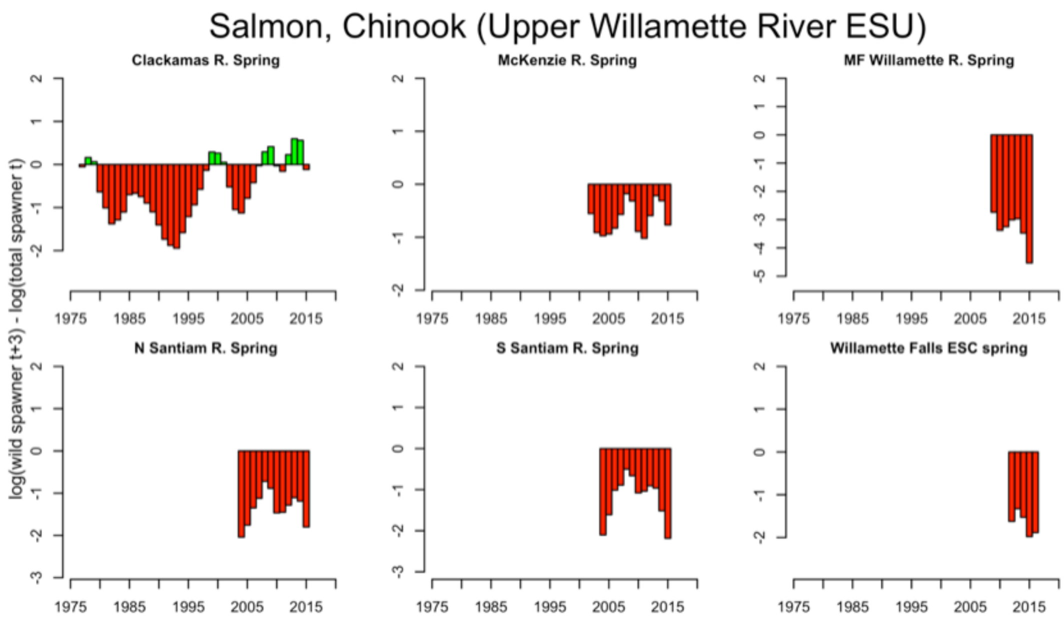

Figure 80. Trends in Willamette Falls counts and population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year ($t-4$).

Harvest

Upper Willamette River spring-run Chinook salmon 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. Ocean fisheries, with the exception of 2016, have been low (Figure 81). The majority of harvest in freshwater fisheries is mark-selective, and harvest rates for naturally produced fish would be considerably less (Figure 82). The Fishery Management Plan for the Willamette River sets the maximum freshwater mortality rate for naturally produced Chinook salmon at 15% (ODFW and WDFW 2020a). Illegal take of unmarked fish is thought to be low.

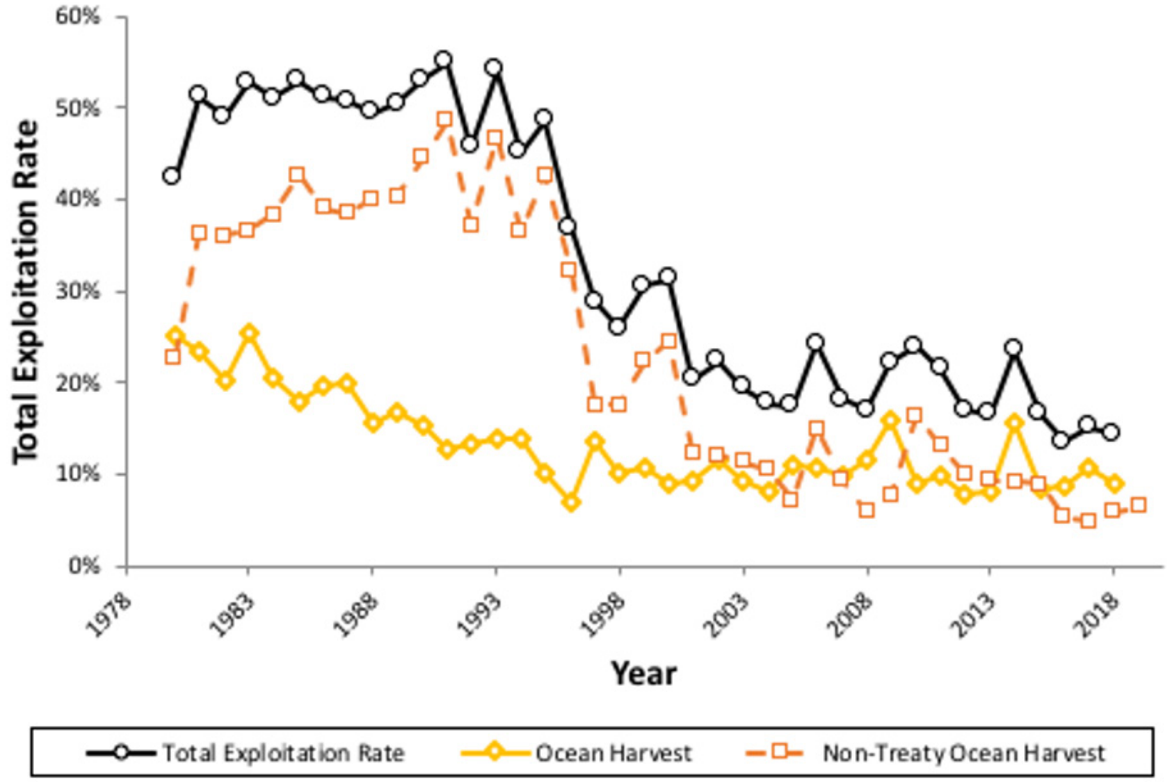

Figure 81. Ocean harvest, terminal harvest, and escapement rates for spring-run Upper Willamette River Chinook salmon, based on coded-wire tag recoveries (PSC 2019). Ocean harvest rates for hatchery and unmarked naturally produced fish are assumed to be comparable; terminal fisheries have been mark-selective since 2001, and unmarked fish mortality rates will be considerably lower: hooking mortality in the Willamette River is assumed to be 12.2% (ODFW and WDFW 2020a).

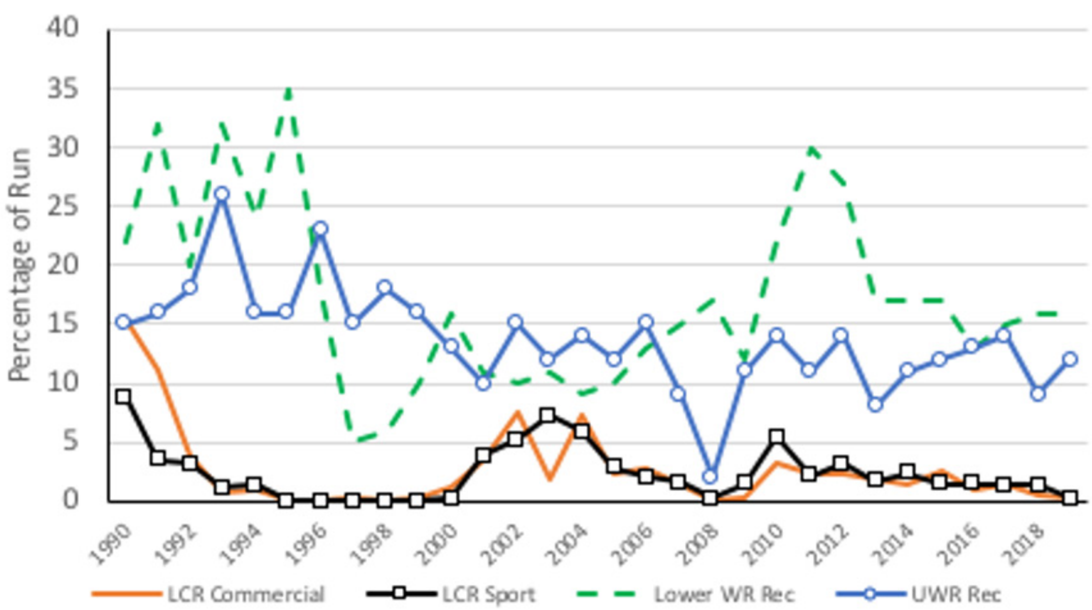

Figure 82. Breakdown of terminal fisheries for Upper Willamette River (UWR) Chinook salmon. Lower Willamette River (below Willamette Falls) and UWR recreational (Rec) fisheries are mark-selective and reflect retention of clipped fish and encounter/hooking mortalities of unmarked fish; hooking mortality rates for the Willamette River are estimated at 12.2% (ODFW and WDFW 2020a).

Spatial structure and diversity

For Upper Willamette River Chinook salmon, diversity concerns include interaction and introgression with hatchery-origin Chinook salmon. Johnson and Friesen (2014) examined the genetic diversity and structure of natural- and hatchery-origin Upper Willamette River spring-run Chinook salmon and found that, while hatchery populations were most similar to natural Chinook salmon from the same basin, they tended to present greater allelic richness. It is not clear whether this is due to the small effective population size of naturally spawning populations, or the legacy of interhatchery transfers between basins. There have been a number of changes in hatchery operations 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 83). 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 westside tributary that does not support a Chinook salmon population, have been made in an effort to maintain a harvestable hatchery return, but reduce hatchery × 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 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, 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 improved for one population by improving broodstock collection and reducing the size of its integrated harvest program. (p. 46)

Recent improvements at the Cougar Dam (2010), Minto Dam (2012), Foster Dam (2014), and Fall Creek Dam (2019) fish collection facilities offer the potential for collecting more hatchery-origin adults and removing them from the naturally spawning component of the populations. Increased collection efficiency has been observed at the Cougar and Minto Dam facilities, while the recently completed Foster Dam facility appears to require further modifications. In concert with improvements in collection efficiency, the number of hatchery fish released has decreased in most basins where there is natural spawning, with increased releases in westside tributaries (Figure 83). In general, the influence of hatchery-origin Chinook salmon on the spawning grounds has shown a slight improvement, with the exception of the South Santiam River, where fish collection at the new facility has been poor (Keefer et al. 2018) leaving more hatchery-origin fish to spawn below Foster Dam (Table 46).

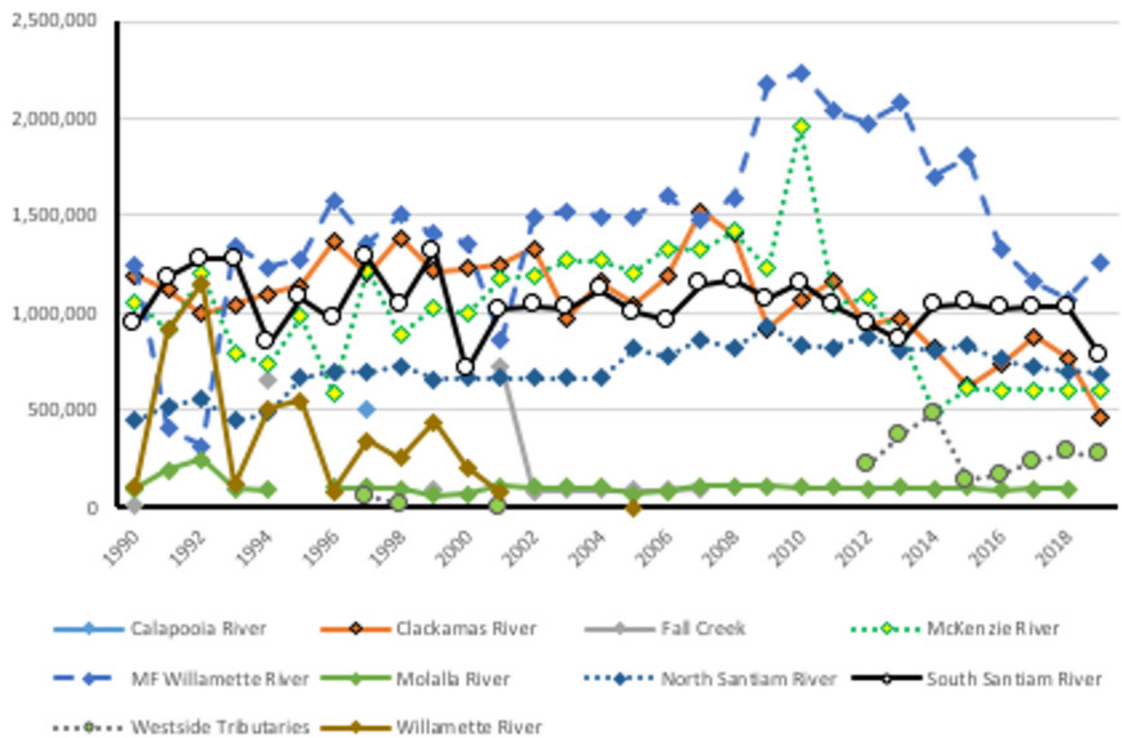

Figure 83. Hatchery releases of juvenile spring-run Chinook salmon into basins of the Upper Willamette River Chinook salmon ESU, 1990–2019. Data for 2019 may be incomplete. Releases of juveniles weighing <2.5 g were not included. Releases into the Row and Coast Fork Rivers were combined under Westside Tributaries. Data from the Regional Mark Information System (https://www.rmpc.org, June 2020).

More recently, NMFS finalized a biological opinion (BiOp) on hatchery operations in the upper Willamette River basin and recommended a number of changes to minimize the potential influence of hatchery-origin fish on natural-origin Chinook salmon and steelhead (NMFS 2019c). Through the BiOp and the individual hatchery genetic and management plans (HGMPs), hatcheries in the upper Willamette River have reduced releases of spring-run Chinook salmon in the McKenzie and North Santiam Rivers, while shifting production to other basins (Figure 83). In addition, the BiOp calls for further action in the McKenzie River to further reduce the number of hatchery fish spawning naturally.

Spatial structure issues remain a major concern in the Willamette River basin. Major dams block volitional passage to historical Chinook salmon habitat in five of the seven DIPs in the ESU. In most cases, effective passage programs are limited by low collection rates for emigrating juveniles. Recent improvements in the juvenile collection at the Clackamas River dams (River Mill and North Fork Dams) resulted in a 91.7% collection efficiency in 2018–19 (Ackerman and Pyper 2020). The improved juvenile collection facility captured 195,123 Chinook salmon juveniles in 2018 and 71,370 in 2019. Juvenile fish passage is also provided at Foster Dam, South Santiam River, where fish can move downstream via the turbines or spill. Recent efforts to improve juvenile passage with a fish weir unfortunately resulted in higher passage mortalities under high pool conditions (Liss et al. 2020). Effective juvenile fish passage is also provided through Fall Creek Dam (Middle Fork Willamette River DIP) via operational drawdown of the reservoir in the fall (Murphy et al. 2019). The reservoir drawdown also had the added benefit of removing non-native species that are potential predators of juvenile salmonids. Juvenile passage in the South Fork McKenzie and North Santiam Rivers has been done on an experimental basis; juvenile collection and passage survival have not been sufficient to sustain naturally spawning Chinook salmon above the dams. Interim measures to improve passage have been proposed for these dams and for dams in the Middle Fork Willamette River, but have yet to be initiated or evaluated. Longer-term structural or operational passage solutions are still being developed. Similarly, passage solutions at the Carmen–Smith Hydroelectric Project on the upper McKenzie River are also in the planning stage.

Table 47. Current 5-year geometric mean of raw natural-origin spawner abundances and one recovery scenario presented in the recovery plan (NMFS 2011) for Upper Willamette River spring-run Chinook salmon demographically independent populations (DIPs). Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > x < 50%, yellow = 50% > x < 100%, green = >100%.

		Abundance	
MPG	**Population**	**2015–19**	**Target**
Willamette	Clackamas River SP	3,617	2,317
	Molalla River SP	n/a	696
	North Santiam River SP	354	5,400
	South Santiam River SP	337	3,100
	Calapooia River SP	n/a	590
	McKenzie River SP	1,664	8,376
	Middle Fork Willamette River SP	20	5,820

Biological status relative to recovery goals

Abundance levels for all but one of the seven DIPs in this ESU remain well below their recovery goals. The Clackamas River DIP currently exceeds its abundance recovery goal and its pHOS goal (<10% hatchery-origin fish). Alternatively, the Calapooia River may be functionally extinct, and the Molalla River remains critically low (there is considerable uncertainty in the level of natural production in the Molalla River). Abundances in the North and South Santiam Rivers have declined since the last review, with natural-origin abundances in the low hundreds of fish. The Middle Fork Willamette River is at a very low abundance, even with the inclusion of natural-origin spring-run Chinook salmon spawning in Fall Creek. While returns to Fall Creek Dam number in the low hundreds, prespawn mortality rates are very high in the basin; however, the Fall Creek program does provide valuable information on juvenile fish passage through operational drawdown. With the exception of the Clackamas River, the proportions of natural-origin spawners in the remainder of the ESU are well below those identified in the recovery goals.

While the Clackamas River appears to be able to sustain above recovery goal abundances, even during relatively poor ocean and freshwater conditions, the remainder of the ESU is well short of its recovery goals.

Updated biological risk summary

Access to historical spawning and rearing areas is restricted by high-head dams in five of the historically most-productive tributaries. Only in the Clackamas River does the current system of adult trap-and-haul and juvenile collection appear to be effective enough to sustain a naturally spawning population (although current juvenile passage efficiencies are still below NMFS criteria). In the McKenzie River, the spring-run Chinook salmon population appears to be relatively stable, having reversed a short-term downward abundance trend that was of concern during the last review. The McKenzie River remains well below its recovery goal, despite having volitional access to much of its historical spawning habitat. The North and South Santiam River DIPs both experienced declines in abundance. Much of the accessible habitat for these populations is relatively poor, and under the warmer and drier conditions experienced during this review period, both juvenile and adult survivals were likely disproportionately affected. Further, water conditions at the adult collection facility at Foster Dam failed to attract adults for transportation to the upper South Santiam River, resulting in more fish spawning below Foster Dam in less desirable habitat. The Middle Fork Willamette River is limited to spawning below Dexter Dam, where conditions all but preclude successful spawning. Under current conditions, Fall Creek is likely near its capacity of several hundred fish. The Calapooia and Molalla Rivers are constrained by habitat conditions, and natural reproduction is likely extremely low. Demographic risks remain "high" or "very high" for most populations, except the Clackamas and McKenzie Rivers, which are at "low" and "low-to-moderate" risk, respectively. The Clackamas River spring-run Chinook salmon population maintains a low pHOS through the removal of all marked hatchery-origin adults at North Fork Dam. Elsewhere, hatchery-origin fish comprise the majority or, in the case of the McKenzie River, nearly half of the naturally spawning population. Diversity risks continue to be a concern.

Spatial structure, specifically access to historical spawning habitat, continues to be a concern. In the absence of effective passage programs, spawners in the North Santiam, Middle Fork Willamette, and to a lesser extent South Santiam and McKenzie Rivers will continue to be confined to more lowland reaches where land development, water temperatures, and water quality may be limiting. Pre-spawning mortality levels are generally high in the lower tributary reaches where water temperatures and fish densities are generally the highest. Areas immediately downstream of high-head dams may also be subject to high levels of total dissolved gas (TDG). The continued placement of natural-origin Chinook salmon and steelhead above the barrier dam at the Minto fish facility and into a short reach immediately below the Detroit/Big Cliff Dam complex is a concern. While this program does limit hatchery-origin introgression and supports local adaptation, 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 dates for establishing effective passage above USACE high-head dams (Big Cliff/Detroit, Green Peter, Cougar, Dexter/Lookout Point, and Hills Creek) in the Willamette River basin are well behind the those established in a 2008 BiOp (NMFS 2008), with current timetables extending well into the 2020s. In addition, passage at the Carmen–Smith Hydroelectric Project on the McKenzie River is still in development. Climate change modeling predicts that in the absence of passage to colder headwater areas, some populations would be at a high risk of extinction by 2040 (Myers et al. 2018). Restoration of access to upper watersheds remains a key element in risk reduction for this ESU. A second spatial structure concern is the availability of juvenile rearing habitat in side-channel or off-channel habitat. River channelization and shoreline development have constrained habitat in the lower tributary reaches and Willamette River mainstem, in turn limiting the potential for fry and subyearling "movers" emigrating to the estuary (Schroeder et al. 2016).

Overall, there has likely been a declining trend in the viability of the Upper Willamette River Chinook salmon ESU since the last review. The magnitude of this change is not sufficient to suggest a change in risk category, however, so the Upper Willamette River Chinook salmon ESU remains at "moderate" risk of extinction.

Upper Willamette River Steelhead DPS

Brief description of ESU

The DPS includes all naturally produced 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 84; USOFR 2020). Also present in this DPS are non-native "early" winter-run steelhead, which predominately spawn in westside tributaries, and non-native summer-run steelhead that spawn throughout the eastside tributaries of the Willamette River basin (Myers et al. 2006). 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.

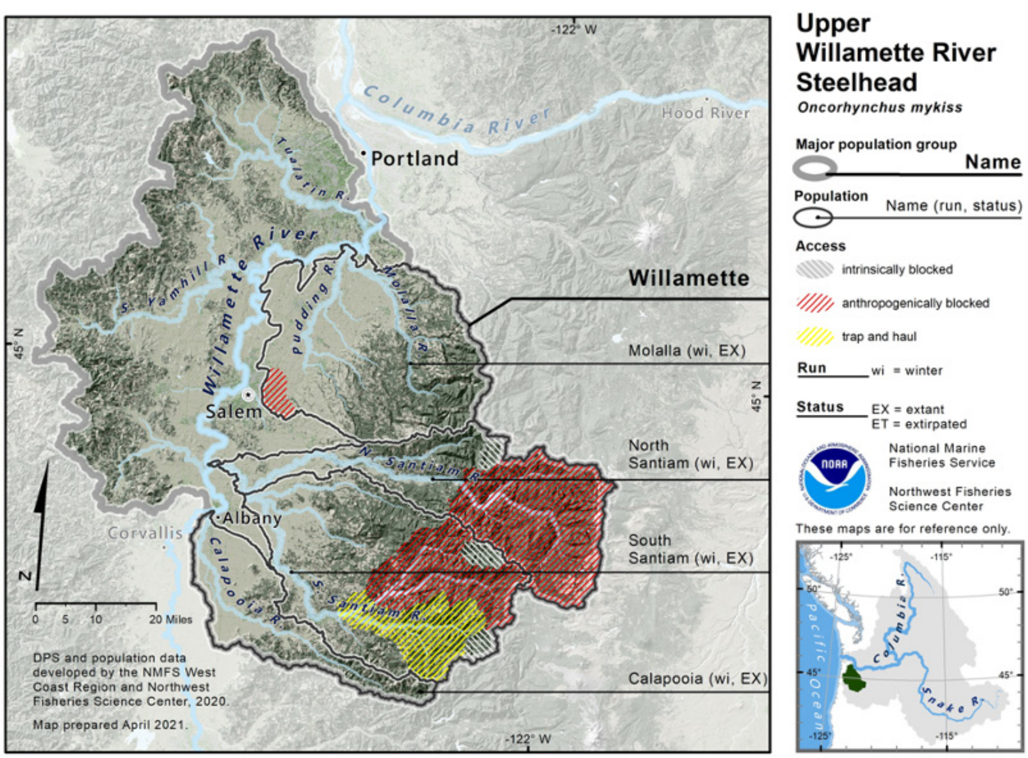

Figure 84. Map of the four demographically independent populations (DIPs) in the Upper Willamette River steelhead DPS.

Summary of previous status conclusions

2005

NMFS initially reviewed the status of the Upper Willamette River steelhead DPS in 1996 (Busby et al. 1996), with an update in 1999 (NMFS 1999b). In the 1999 review, the BRT noted several concerns for this DPS, 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 DPS 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 DPS were assigned to the "likely to become endangered" category, with small minorities in the "in danger of extinction" and "not likely to become endangered" categories. The BRT did not identify any extreme risks for this DPS, 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 River 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 River Chinook salmon 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 had reduced some risks, but non-native summer steelhead hatchery releases were still a concern. Human population expansion within the Willamette River 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 previous BRT status review in 2005.

2015

The declines in abundance noted during the previous review continued through the period 2010–15 (NWFSC 2015). There was considerable uncertainty in many of the abundance estimates, except for perhaps the tributary dam counts. Radio-tagging studies indicate that a considerable proportion of winter steelhead ascending Willamette Falls do not enter the basins that were identified for this DPS; these fish may be non-native early-winter steelhead that appear to have colonized the western tributaries, misidentified summer steelhead, or late-winter steelhead that have colonized tributaries not historically part of the DPS (Jepson et al. 2015, Johnson et al. 2021). More definitive genetic monitoring of steelhead ascending Willamette Falls, in tandem with radio tagging, would be required to provide an estimate of population abundance, as well as a total abundance for the DPS.

The release of non-native summer-run steelhead continued to be a concern. Genetic analysis suggested that there is introgression among native late-winter steelhead and summer-run steelhead (Van Doornik et al. 2015). Accessibility to historical spawning habitat was still limited, especially in the North Santiam River. Much of the accessible habitat in the Molalla, Calapooia, and lower reaches of the North and South Santiam Rivers was considered degraded and under continued development pressure. Although habitat restoration efforts were underway, the time scale for restoring functional habitat did not address the more immediate risks to the DPS.

Description of new data available for this review

Abundance and life history data for steelhead in the Upper Willamette River steelhead DPS are very limited. Consistent redd counts are available for some index reaches, primarily in Thomas and Crabtree Creeks, but these do not provide population-level indicators of abundance. Specific research projects have been undertaken to estimate steelhead spawning abundance and distribution (Mapes et al. 2017), but only in specific basins and for a limited number of years. Adult counts were also available from observations at Willamette Falls, Bennett Dam, the Minto Dam fish facility (North Santiam River), and Foster Dam (South Santiam River). While steelhead counts at Willamette Falls provide a DPS-wide estimate of abundance, there is some uncertainty in distinguishing native late-winter steelhead from non-native early-winter steelhead and unmarked non-native summer steelhead (Johnson et al. 2018, Weigel et al. 2019). Counts of steelhead in eastside tributaries provide more population-specific information on abundance.

Abundance and productivity

Willamette Falls

Winter steelhead counts at Willamette Falls provide a complete count of fish returning to the DPS. In the last five years, counts of steelhead at Willamette Falls experienced a marked decrease, with a record low count in 2017 of 822 (Figure 87). During the 2016–17 return year, pinniped predation at Willamette Falls became a concern. Increases in the pinniped population at the falls, in conjunction with low steelhead return, resulted in an estimated 25% predation rate on winter steelhead (Steingass et al. 2019). With the initiation of pinniped control measures in 2019 and improvements in the steelhead run size, predation levels fell to an estimated 8% in 2019 (Steingass et al. 2019). Overall, there was a 59% decrease in the geometric average for 2015–19 relative to 2010–14 (Table 48). Abundances at Willamette Falls appear to have recovered since the 2017 low, with a recent (unofficial) count of 5,510 winter-run steelhead.

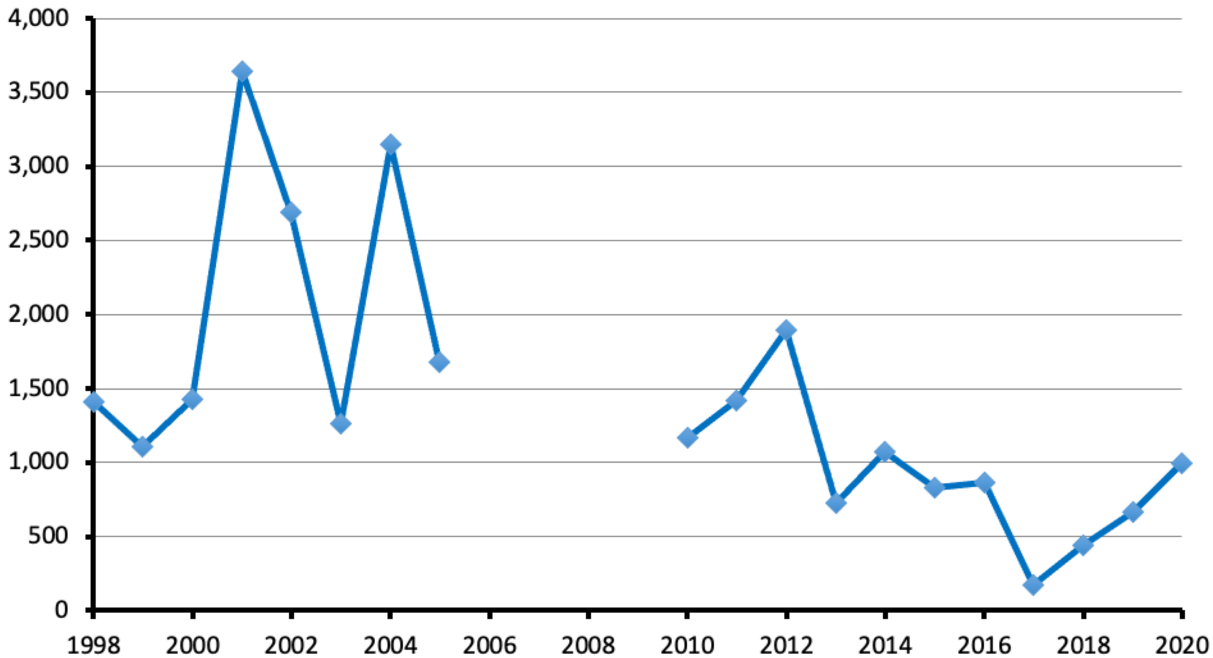

Figure 85. Winter-run steelhead counts at Upper and Lower Bennett Dams in the North Santiam River, 1998–2020. Some steelhead spawning may occur below the dams. Data available: myodfw.com/upper-and-lower-bennett-dams-fish-counts (August 2021).

Molalla River

Population abundance estimates based on spawner (redd) surveys are only available for the Molalla River and associated tributaries (Pudding River, Abiqua Creek) through 2018. 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. Proportional allocation of Willamette Falls may be informative; however, comparisons using radio-tagged steelhead results (Jepson et al. 2013, 2014, 2015) suggest that the proportional assignment may overestimate abundance. In either case, there is considerable uncertainty in the abundance estimates.

North Santiam River and Bennett Dam

Late-winter steelhead spawn throughout the North Santiam River 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 Dams (Figure 85). 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 the dams and some survey data are available for downstream of the dams, 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 prespawn mortality is thought to be low for winter steelhead, and the contribution of non-native early-winter-run fish above the dams is also thought to be low (Johnson et al. 2018). Further, it should be noted that Johnson et al. (2021) found that over half of the unmarked juvenile steelhead sampled below Big Cliff Dam were genetically assigned as non-native early-winter steelhead. The five-year geometric mean (2015–19) for the Bennett Dam counts is 514. Sharpe and Mapes (2017) found substantial differences in abundance estimates for winter-run steelhead in the North Santiam River using index surveys, mark/recapture with radio-tagged steelhead, and the Bennett Dam counts. In light of the uncertainty in abundance estimates for this population, the calculation of short- and long-term trends would convey an unjustified precision. In general, there has been long-term decline in the abundance of this population.

South Santiam River and Foster Dam

Survey data (index redd counts) are 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. In 2016 and 2017, there was a systematic monitoring of the South Santiam River (Mapes et al. 2017). Winter steelhead abundance was estimated at 1,480 ± 721 in 2016, and 157 ± 60 in 2017 (the record low year). Further, Mapes et al. (2017) reported that there were considerable differences between their abundance estimates for South Santiam River tributaries and those generated using the existing index reach-based approach. Therefore, longer time series are less meaningful, in that abundance estimates before 2009 were developed using index surveys to allocate Willamette Falls counts. Finally, Foster Dam counts reflect only a portion of the overall abundance, and the proportion of winter steelhead ascending the ladder can vary from year to year depending on water conditions. Overall, index counts and Foster Dam counts reflect the general trend of winter steelhead counted at Willamette Falls.

Calapooia River

There is a nearly complete, consistent time series for index reach redd counts in the Calapooia River dating back to 1985 (Figure 86). While there is not an expansion available from index reaches to population spawner abundance, available estimates of winter steelhead redds-per-mile demonstrate considerable resiliency. Results for 2015 and 2016 generally reflect good ocean and freshwater conditions. As with the other DIPs in this DPS, measures of escapement were extremely low in 2018 and 2019, and likely in 2017 as well. The improvement in index counts for 2020 suggests reasonable underlying productivity. By comparison, radio-tag mark/recapture estimates for 2013, 2014, and 2015 were 127, 204, and 126, respectively (Jepson et al. 2015). While no quantitative estimate of population abundance is possible, it would appear that the Calapooia River, on average, supports several hundred spawners.

Harvest

There is no consumptive fishery for winter steelhead in the Upper Willamette River. Winter-run steelhead in the Columbia River fishery are intercepted at a low rate, 0.2% (ODFW and WDFW 2020). Similarly, 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 Willamette River recreational fishery are thought to be low. Tribal fisheries occur above Bonneville Dam and do not impact Upper Willamette River steelhead.

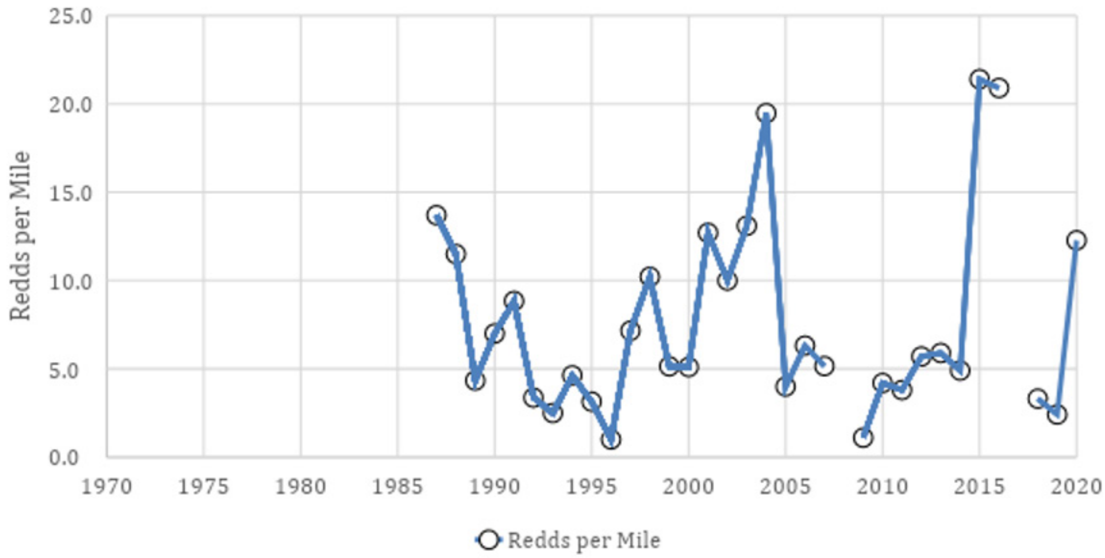

Figure 86. Calapooia River index reach estimates of winter steelhead redds per mile, from ground surveys conducted from 1985–2020.

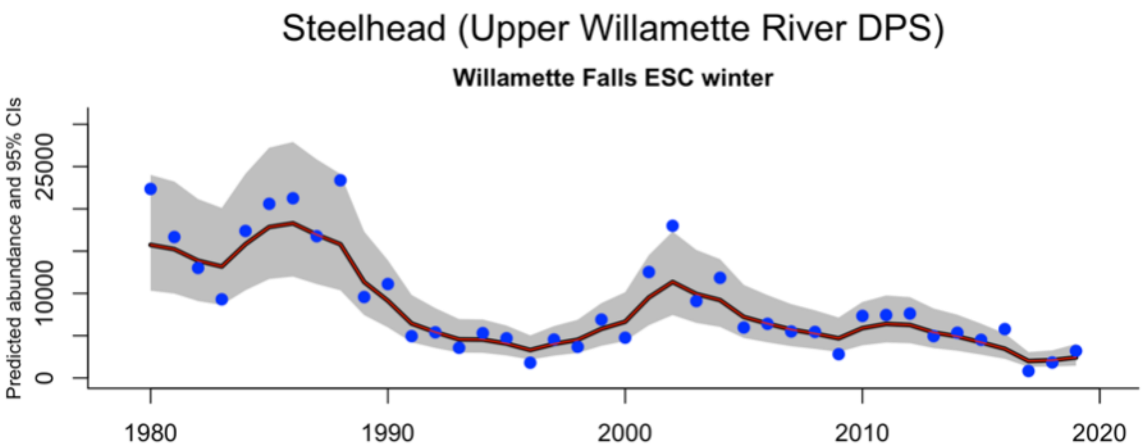

Figure 87. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

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 for the entire Upper Willamette River DPS (including the McKenzie and Middle Fork Willamette Rivers) have decreased slightly, to 500,000 (2015–19; Figure 88). Still, the legacy of previous hatchery-origin releases persists in the upper Willamette River.

Table 48. Five-year geometric mean of raw natural spawner counts for the Upper Willamette River steelhead DPS. Willamette Falls counts represent counts of prespawning winter steelhead, and include an unknown number of non-native early-winter-run steelhead. Population estimates (1990–2009) were calculated using proportional assignment of Willamette Falls counts. 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 2 most recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Willamette Falls W	Cascade	(5,619)	(3,961)	(10,293)	(5,028)	(6,431)	(2,628)	(–59)
Calapooia River W	Cascade	149 (149)	219 (219)	406 (406)	214 (214)	—	—	—
Molalla River W	Cascade	1,182 (1,462)	726 (798)	1,924 (1,924)	1,357 (1,357)	—	—	—
North Santiam River W	Cascade	2,495 (2,928)	1,953 (2,388)	3,333 (3,423)	2,500 (2,500)	—	—	—
South Santiam River W	Cascade	1,940 (1,940)	1,277 (1,277)	2,440 (2,440)	1,594 (1,594)	—	—	—

A recent genetic study by Johnson et al. (2021) evaluated the level of colonization by non-native stocks and introgression between non-native summer-run steelhead and non-native early-winter-run steelhead with native late-winter-run steelhead. This work expanded upon the findings of earlier work by Johnson et al. (2013) and Van Doornik et al. (2015), but collected and analyzed juvenile *O. mykiss*. Johnson et al. (2021) identified westside tributaries as being largely occupied by non-native early-winter-run steelhead originating from releases by Big Creek Hatchery (Lower Columbia River, Southwest Washington steelhead DPS) beginning in the 1920s. With the exception of the lower North Santiam River, native late-winter steelhead are still predominant in eastside tributaries that drain the Cascades north of the McKenzie River. Areas above dams in the North and South Santiam Rivers and in the Calapooia River appear to have little influence from non-native introductions. Below dams in the North Santiam River, pure non-native summer-run and a non-native Big Creek winter-run steelhead were detected, as were hybrids between non-native and native steelhead. Below dams of the South Santiam River, introgression from introduced steelhead was higher than in the North Santiam, with 12% of the juveniles identified as summer-run × native winter-run hybrids and 14% identified as hybrids of non-native early-winter × native late-winter steelhead. In the Molalla River, the predominant genotype was native winter-run steelhead (40%), but a substantial number of hybrids between the native and non-native steelhead were detected. The presence of pure and hybrid summer-run steelhead in the Molalla River is surprising, because summer run steelhead have not been released in this basin since 1998. The establishment of feral non-native summer and winter runs of steelhead poses a genetic risk to the native populations. In addition, the presence of hatchery-reared and feral hatchery-origin fish may affect the growth and survival of juvenile late-winter steelhead.

The exclusion of steelhead from headwater reaches in the North and South Santiam Rivers continues to be the primary spatial structure concern. Although the historical distribution of steelhead is not precisely known, Mattson (1948) and Wallis (1963) indicate that the majority of steelhead and salmon spawning occurred above the current site of Detroit Dam in the North Santiam River. Similarly, in the South Santiam River, while steelhead have access to habitat above Foster Dam, the Middle Santiam River is blocked by Green Peter Dam. Conditions in the South Santiam River above Foster Reservoir may be limiting, due to high (>20°C) summer

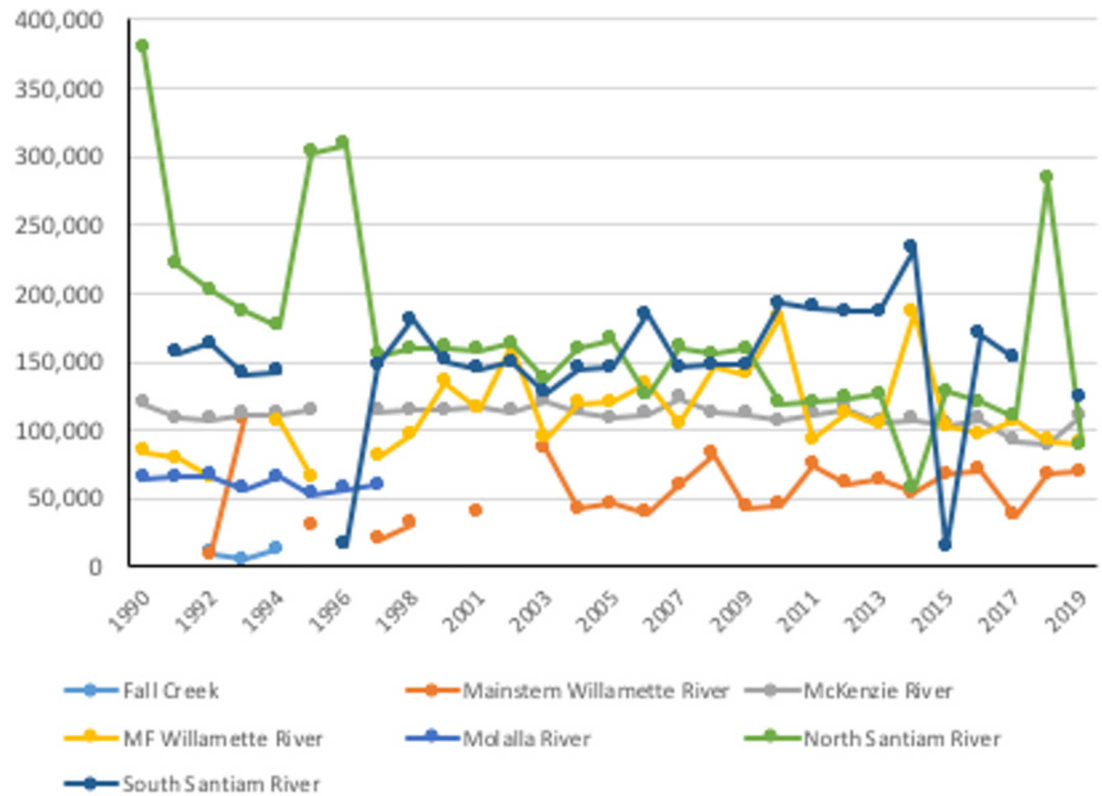

Figure 88. Annual releases of hatchery-origin (Skamania stock) summer-run steelhead into Willamette River tributaries, by sub-basin. Releases of fish <2.5 g are not included. All releases are considered to be out-of-DPS in origin. Data from the Regional Mark Information System (https://www.rmpc.org, April 2020).

prespawning holding temperatures, and poor incubation and rearing habitat conditions (the river is prone to scour during flood episodes). For example, 2010 was a poor year, with scouring floods during incubation. Alternatively, historical habitat (Quartzville Creek and the Middle Santiam River) above Green Peter Dam may provide better spawning and rearing habitat than the upper South Santiam River; previous surveys suggest that the Middle Santiam River and its tributary, Quartzville Creek, were historically preferred steelhead spawning habitat (Parkhurst 1950, Wagner et al. 1963). Efforts to provide passage for steelhead in the North Santiam River are still at the planning stage, and little effort has been allocated to providing passage at Green Peter Dam. Foster Dam provides volitional downstream passage, but juvenile and kelt survivals need to improve further to meet passage criteria. Smaller-scale upstream and downstream passage issues exist throughout the DPS, related in part to water withdrawal structures. While some of these have been addressed, others remain.

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 adult winter steelhead do not experience the same deleterious water temperatures as the spring-run Chinook salmon, and prespawn mortalities are not likely to be significant. Although the recent magnitude of these declines is relatively moderate, continued declines would be a

Table 49. Current 5-year geometric mean of raw natural-origin abundances and recovery scenario targets presented in the recovery plan (NMFS 2011) for Upper Willamette River steelhead demographically independent populations (DIPs). Willamette Falls count includes non-native early-winter-run steelhead, and therefore represents an upper limit to total abundance. No tributary abundance estimates are available and the approximate total DPS abundance is represented by the Willamette Falls count. This total abundance is compared to the sum of the individual DIP targets. Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > *x* < 50%, yellow = 50% > *x* < 100%, green = >100%.

		Abundance	
MPG	**Population**	**2015–19**	**Target**
Cascade	Willamette Falls Winter Count	2,628	n/a
	Molalla River W	n/a	3,000
	North Santiam River W	n/wa	8,358
	South Santiam River W	n/a	3,913
	Calapooia River W	n/a	498
	Total:	2,628	15,769

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 River 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 the 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 North and South Santiam Rivers may be influencing the viability of native steelhead. Overall, none of the populations in the DPS are meeting their recovery goals (Table 49).

Updated biological risk summary

Overall, the Upper Willamette River steelhead DPS continued to decline in abundance. Although the most recent counts at Willamette Falls and the Bennett Dams in 2019 and 2020 suggest a rebound from the record 2017 lows, it should be noted that current "highs" are equivalent to past lows. Uncertainty in adult counts at Willamette Falls are a concern, given that the counts represent an upper bound on DPS abundance. Radio-tagging studies suggest that a considerable proportion of "winter" steelhead ascending Willamette Falls do not enter the tributaries that are considered part of this DPS; these fish may be non-native early-winter steelhead that appear to have colonized the western tributaries, misidentified summer steelhead, late-winter steelhead that have colonized tributaries not historically part of the DPS, or hybrids between native and non-native steelhead. 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.

Introgression by non-native summer-run steelhead continues to be a concern. Genetic analysis suggests that there is introgression among native late-winter steelhead and summer-run steelhead (Van Doornik et al. 2015, Johnson et al. 2018, 2021). Accessibility to historical spawning habitat is still limited, especially in the North Santiam River. Efforts to provide juvenile downstream passage at Detroit Dam are well behind the proscribed timetable (NMFS 2008), and passage at Green Peter Dam has not yet entered the planning stage. Much of the accessible habitat in the Molalla, Calapooia, and the lower reaches of the 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. While the viability of the ESU appears to be declining, the recent uptick in abundance may provide a short-term demographic buffer. Furthermore, increased monitoring is necessary to provide quantitative verification of sustainability for most of the populations. In the absence of substantial changes in accessibility to high-quality habitat, the DPS will remain at "moderate-to-high" risk. Overall, the Upper Willamette River steelhead DPS is therefore at "moderate-to-high" risk, with a declining viability trend.

Oregon and Washington Coast Domain Viability 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, 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 89; USOFR 2020). 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 five geographic regions, or major population groups (MPGs), identified by the Puget Sound Technical Recovery Team (PSTRT 2002) based on similarities in hydrographic, biogeographic, and geologic characteristics of the Puget Sound basin.

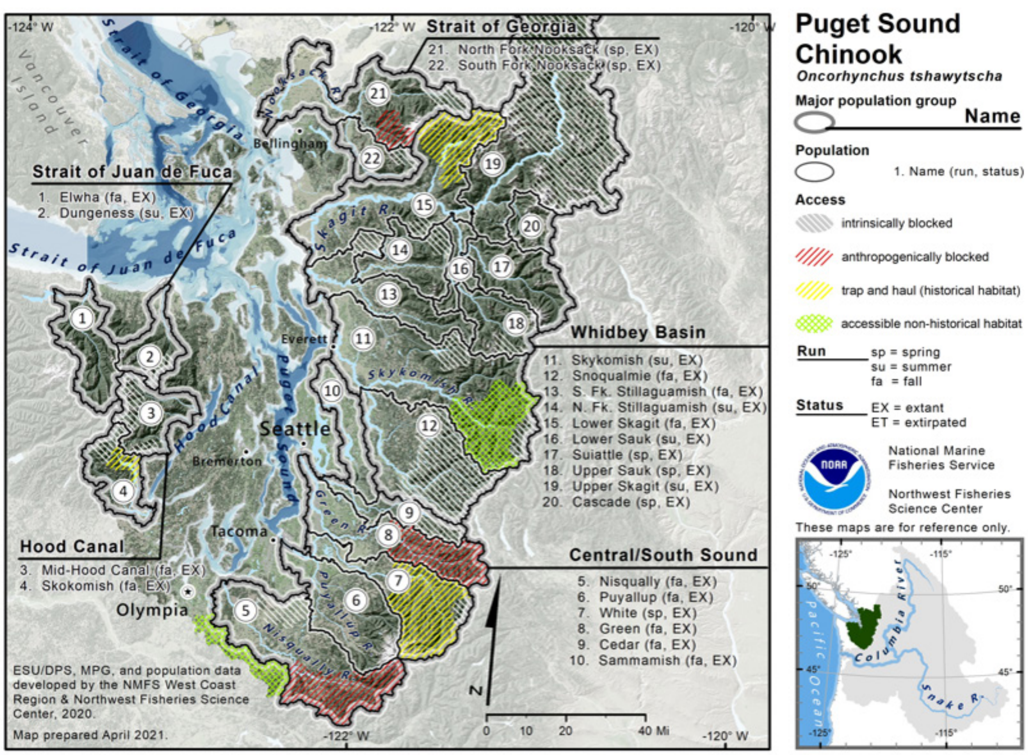

Figure 89. Map of the Puget Sound Chinook salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

Summary of previous status conclusions

2005

In the 2005 review (Good et al. 2005), the BRT concluded that, overall, the status of naturally spawning populations of Puget Sound Chinook salmon had 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 those 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 salmon 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 previous BRT status review in 2005.

2015

NWFSC (2015) concluded that all Puget Sound Chinook salmon populations were still well below the TRT minimum planning range for recovery escapement levels. Most populations were also consistently below the spawner–recruit levels identified by the TRT as consistent with recovery. Across the ESU, most populations further declined in abundance since the 2011 status review, and indeed, this decline had been persistent over the previous seven-to-ten years. Productivity remained low in most populations. Hatchery-origin spawners were present in high fractions in most populations outside the Skagit River watershed, and in many watersheds the fraction of natural-origin spawner abundances had declined over time. The original Puget Sound Chinook salmon recovery plan watershed chapters were completed in 2005 (available: www.fisheries.noaa.gov/resource/document/recovery-plan-puget-sound-chinook-salmon), and habitat monitoring and adaptive management planning documents were completed in 2014 (available: www.psp.wa.gov/salmon-recovery-overview.php), and, along with a series of three- and four-year work plans, these documents identify the habitat improvement projects planned and completed by the 16 watershed programs with the intention to help progress for Chinook salmon recovery in the ESU. There has been considerable variation in efforts amongst watersheds and their plans, but generally, the efforts have been consistent and progressive through challenging funding cycles. In addition, a number of the individual watersheds had begun the process to update their original recovery plan chapters. It is expected that these habitat improvement projects will take years or decades to produce significant improvements in natural population viability parameters. Overall, the 2015 review concluded that new information on abundance, productivity, spatial structure and diversity since the prior review did not indicate a change in the biological risk category.

Description of new data available for this review

This status report incorporates "best available" Chinook salmon population data through 2018, with data for some populations also available through 2019. Spawning abundance data were obtained from WDFW and the Puget Sound tribes as a result of a request for data in the Federal Register, and from individual comanager biologists and staff. Updates for abundance, age, and hatchery contribution data varied from population to population, and were obtained from multiple sources, including the annual postseason harvest reports provided by WDFW and the Puget Sound Treaty Indian Tribes (PSTIT; WDFW and PSTIT 2015, 2016, 2017, 2018, 2019, 2020) and from WDFW's Salmonid Stock Inventory (SaSI), and additional state hatchery data were queried from WDFW's FishBooks database and provided by WDFW staff. Tribal hatchery data were also provided by individual tribal staff. Where data sources conflicted, data were confirmed as much as possible, through collaborative discussions with both tribal and state co-managers. It is important to note that data collection and analysis methodologies for both hatchery and natural spawner abundances have changed in some watersheds/populations over the course of the time series analyzed. This 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 2015 status review, this report also incorporates updates and corrections made in past escapement, age, and hatchery contribution data for many of the populations. These corrections typically have been made by individual tribal and/or state co-managers. 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 continue to meet and collaborate with co-managers regarding the development of a single dataset, but it 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 in the very near future. Please see Acknowledgments for a list of individuals who helped to improve and validate the dataset used for this analysis.

Abundance and productivity

Abundance of the 22 extant natural spawning populations in the Puget Sound Chinook salmon ESU varies considerably between populations. Trends in abundance for individual populations are shown in Figure 90. The populations are grouped into five MPGs: Strait of Georgia, Whidbey Basin, Central/South Sound, Hood Canal, and Strait of Juan de Fuca (Figure 89; Ruckelshaus et al. 2006). The early run timing populations are North and

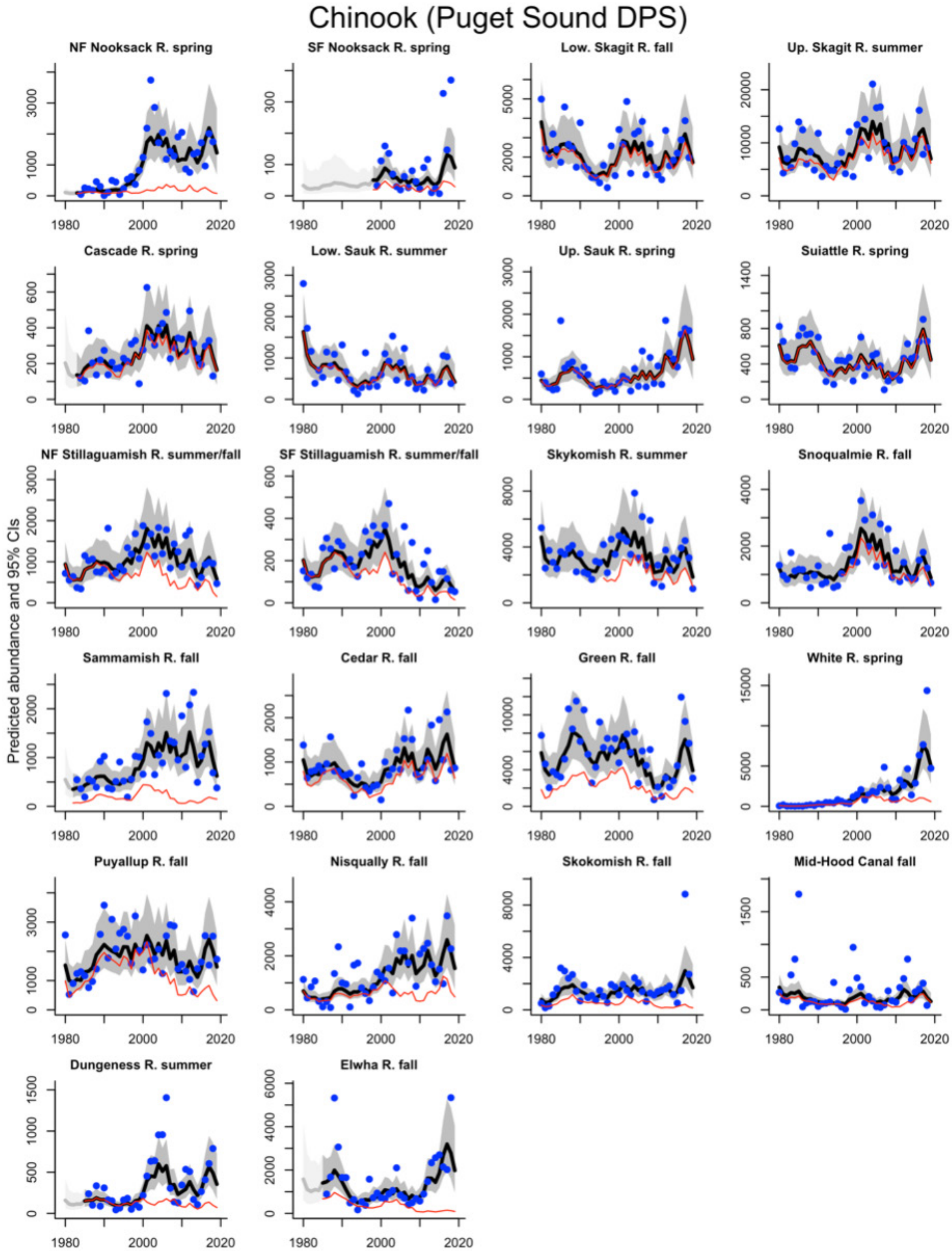

Figure 90. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

South Fork Nooksack Rivers (Strait of Georgia); Cascade, Upper Sauk, and Suiattle Rivers (Whidbey Basin); and White River (Central/South Sound). Summer runs exist in Upper Skagit, Lower Sauk, North and South Fork Stillaguamish, and Skykomish Rivers (Whidbey Basin) and in Dungeness River (Strait of Juan de Fuca). All other populations are fall runs. Newer genetics data have clarified that the two Stillaguamish River populations overlap in spawn timing and distribution, with both summer and fall populations spawning in both forks of the Stillaguamish River (WDFW and PSTIT 2020).

Total abundance in the ESU over the entire time series shows that individual populations have varied in increasing or decreasing abundance. Several populations (North and South Fork Nooksack, Sammamish, Green, White, Puyallup, Nisqually, Skokomish, Dungeness, and Elwha Rivers) are dominated by hatchery returns. Generally, many populations experienced increases in total abundance during the years 2000–08, and more recently in 2015–17, but general declines during 2009–14, and a downturn again in the two most-recent years, 2017–18 (Figure 90). Abundance across the Puget Sound Chinook salmon ESU has generally increased since the last status review, with only two of the 22 populations (Cascade River and North and South Fork Stillaguamish Rivers) showing a negative percentage change in the five-year geometric mean natural-origin spawner abundances since the prior status review (Table 50). Fifteen of the remaining 20 populations with positive percentage changes since the prior status review have relatively low natural spawning abundances (<1,000 fish), so some of 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 natural-origin spawner abundance were computed over two time periods (1990–2005 and 2004–19) for each Puget Sound Chinook salmon population (Table 51). Trends were negative for four of the populations in the earlier period, and for 16 of the 22 populations in the later period. Thus, there is a general decline in natural-origin spawner abundance across all MPGs in the most-recent fifteen years. Upper Sauk and Suiattle Rivers (Whidbey Basin MPG), Nisqually River (Central/South Sound MPG), and Mid-Hood Canal (Hood Canal MPG) are the only populations with positive trends, though Mid-Hood Canal has an extremely low population size. Further, no change in trend between the two time periods was detected in South Fork Nooksack River (Strait of Georgia MPG) or Green and Nisqually Rivers (Central/South MPG). The average trend across the ESU for 1990–2005 was 0.03. The average trends for the MPGs are: Strait of Georgia, 0.03; Whidbey Basin, 0.04; Central/South Sound, 0.04; Hood Canal, 0.03; and Strait of Juan de Fuca, 0.01. The average trend across the ESU for 2004–19 was –0.02. The average trends for the MPGs are: Strait of Georgia, –0.02; Whidbey Basin, –0.02; Central/South Sound, –0.02; Hood Canal, –0.02; and Strait of Juan de Fuca, –0.08 (Table 51). The previous status review (NWFSC 2015) concluded that there were widespread negative trends for the total ESU, despite variable escapements and trends for individual populations. The addition of the data to 2018 now shows even more substantially either flat or negative trends for the entire ESU in natural-origin Chinook salmon spawner population abundances.

Table 50. Five-year geometric mean of raw natural-origin spawner counts. This is the raw total spawner estimate times the fraction natural-origin estimate, if available. In parentheses, 5-year geometric means of raw total spawner estimates (i.e., hatchery and natural) are shown. A value only in parentheses means that a total spawner estimate was available but no (or only one) estimate of natural-origin spawners was available. The geometric mean was computed as the product of estimates 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 2 most recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
North Fork Nooksack River SP	Strait of Georgia	51 (102)	95 (471)	229 (2,186)	275 (1,536)	136 (1,205)	137 (1,553)	1 (29)
South Fork Nooksack River SP	Strait of Georgia	—	—	44 (87)	22 (41)	13 (35)	42 (106)	223 (203)
Lower Skagit River FA	Whidbey Basin	1,332 (1,474)	971 (1,035)	2,531 (2,774)	1,916 (2,228)	1,416 (1,541)	2,130 (2,640)	50 (71)
Upper Skagit River SU	Whidbey Basin	3,970 (5,603)	5,641 (6,185)	10,723 (12,410)	8,785 (10,525)	7,072 (7,457)	9,568 (10,521)	35 (41)
Cascade River SP	Whidbey Basin	151 (188)	209 (213)	340 (371)	302 (342)	298 (317)	185 (223)	–38 (–30)
Lower Sauk River SU	Whidbey Basin	384 (409)	403 (429)	820 (846)	543 (569)	376 (416)	635 (649)	69 (56)
Upper Sauk River SP	Whidbey Basin	404 (408)	265 (267)	427 (427)	506 (518)	854 (880)	1,318 (1,330)	54 (51)
Suiattle River SP	Whidbey Basin	288 (302)	378 (382)	402 (415)	258 (261)	376 (378)	640 (657)	70 (74)
North Fork Stillaguamish River SU/FA	Whidbey Basin	731 (913)	677 (1,177)	1,089 (1,553)	493 (1,262)	417 (996)	302 (762)	–28 (–23)
South Fork Stillaguamish River SU/FA	Whidbey Basin	148 (185)	176 (305)	196 (280)	51(131)	34 (68)	37 (96)	9 (41)
Skykomish River SU	Whidbey Basin	(2,398)	1,497 (3,331)	2,377 (4,849)	2,568 (3,378)	1,689 (2,462)	1,736 (2,806)	3 (14)
Snoqualmie River FA	Whidbey Basin	(963)	1,427 (1,279)	2,036 (2,477)	1,308 (1,621)	839 (1,082)	856 (1,146)	2 (6)
Sammamish River FA	Central/South Sound	197 (576)	149 (564)	336 (1,031)	171 (1,278)	82 (1,289)	126 (879)	54 (–32)
Cedar River FA	Central/South Sound	385 (562)	276 (497)	379 (646)	1,017 (1,249)	699 (914)	889 (1,253)	27 (37)
Green River FA	Central/South Sound	2,697 (5,420)	3,856 (7,274)	2,800 (6,542)	1,305 (3,149)	785 (2,109)	1,822 (6,373)	132 (202)
White River SP	Central/South Sound	269 (378)	242 (616)	1,159 (1,461)	839 (2,099)	652 (2,161)	895 (6,244)	37 (189)
Puyallup River FA	Central/South Sound	2,146 (2,547)	2,034 (2,348)	1,378 (1,794)	1,006 (2,054)	450 (1,134)	577 (1,942)	28 (71)
Nisqually River FA	Central/South Sound	610 (781)	577 (723)	689 (1,296)	551 (1,899)	481 (1,823)	766 (1,841)	59 (1)
Skokomish River FA	Hood Canal	505 (993)	478 (1,233)	479 (1,556)	500 (1,216)	136 (1,485)	265 (2,074)	95 (40)
Mid-Hood Canal FA	Hood Canal	94 (120)	78 (103)	169 (217)	47 (88)	80 (295)	196 (222)	145 (–25)
Dungeness River SU	Strait of Juan de Fuca	117 (117)	104 (104)	99 (520)	151 (374)	66 (279)	114 (476)	73 (71)
Elwha River FA	Strait of Juan de Fuca	428 (673)	275 (735)	491 (995)	140 (605)	71 (1,349)	134 (2,810)	89 (108)

Productivity in the Puget Sound Chinook salmon ESU has been variable across the time period (1980–2018). Figure 91 shows trends in productivity as estimated by the log of the smoothed natural-origin spawning abundance in year t minus the smoothed natural-origin spawning abundance in year (t – 4). Data below zero indicate that natural-origin spawners failed to replace themselves, although in many cases total spawning abundance was maintained through hatchery supplementation (compare red and black lines in Figure 90). Across the Puget Sound Chinook salmon ESU, ten of 22 Puget Sound populations show natural productivity below replacement in nearly all years since the mid-1980s. These include the North and South Fork Nooksack Rivers (Strait of Georgia MPG), North and

Table 51. Fifteen-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 spawner estimates are shown and with at least 2 data points in the first 5 years and last 5 years of the 15-year periods. Lower and upper bounds of the 95% confidence intervals of the estimates are in parentheses.

Population	MPG	1990–2005	2004–19
North Fork Nooksack River SP	Strait of Georgia	0.07 (0.04, 0.10)	–0.03 (–0.07, 0.00)
South Fork Nooksack River SP	Strait of Georgia	–0.01 (–0.03, 0.01)	–0.01 (–0.05, 0.03)
Lower Skagit River FA	Whidbey Basin	0.05 (0.02, 0.08)	0.00 (–0.03, 0.03)
Upper Skagit River SU	Whidbey Basin	0.07 (0.04, 0.10)	–0.01 (–0.04, 0.02)
Cascade River SP	Whidbey Basin	0.06 (0.04, 0.09)	–0.03 (–0.06, –0.01)
Lower Sauk River SU	Whidbey Basin	0.04 (0.01, 0.08)	–0.01 (–0.05, 0.02)
Upper Sauk River SP	Whidbey Basin	0.01 (–0.02, 0.05)	0.07 (0.04, 0.10)
Suiattle River SP	Whidbey Basin	0.01 (–0.01, 0.03)	0.05 (0.01, 0.08)
North Fork Stillaguamish River SU/FA	Whidbey Basin	0.02 (–0.01, 0.05)	–0.06 (–0.10, –0.02)
South Fork Stillaguamish River SU/FA	Whidbey Basin	–0.01 (–0.04, 0.02)	–0.08 (–0.13, –0.03)
Skykomish River SU	Whidbey Basin	0.05 (0.01, 0.09)	–0.05 (–0.08, –0.02)
Snoqualmie River FA	Whidbey Basin	0.08 (0.05, 0.12)	–0.05 (–0.08, –0.02)
Sammamish River FA	Central/South Sound	0.06 (0.02, 0.10)	–0.04 (–0.10, 0.02)
Cedar River FA	Central/South Sound	0.02 (–0.03, 0.07)	0.00 (–0.02, 0.03)
Green River FA	Central/South Sound	–0.01 (–0.05, 0.02)	–0.01 (–0.06, 0.03)
White River SP	Central/South Sound	0.15 (0.11, 0.18)	–0.02 (–0.05, 0.01)
Puyallup River FA	Central/South Sound	–0.01 (–0.03, 0.01)	–0.06 (–0.10, –0.03)
Nisqually River FA	Central/South Sound	0.03 (0.00, 0.05)	0.03 (–0.02, 0.07)
Skokomish River FA	Hood Canal	0.02 (–0.02, 0.05)	–0.09 (–0.14, –0.03)
Mid-Hood Canal FA	Hood Canal	0.04 (0.01, 0.07)	0.06 (0.00, 0.11)
Dungeness River SU	Strait of Juan de Fuca	0.01 (–0.02, 0.03)	–0.04 (–0.08, –0.01)
Elwha River FA	Strait of Juan de Fuca	0.00 (–0.04, 0.04)	–0.11 (–0.17, –0.04)

South Fork Stillaguamish and Skykomish Rivers (Whidbey Basin MPG), Sammamish, Green, and Puyallup Rivers (Central/South Sound MPG), Skokomish River (Hood Canal MPG), and Elwha River (Strait of Juan de Fuca MPG). Productivity in the Whidbey Basin MPG populations was above zero in the mid-to-late 1990s, with the exception of the Skykomish and North and South Fork Stillaguamish River populations. The White River population in the Central/South Sound MPG was above replacement from the early 1980s to 2001, but has dropped in productivity consistently since the late 1980s. In recent years, only five populations have had productivities above zero. These are Lower and Upper Skagit, Lower and Upper Sauk, and Suiattle Rivers in the Whidbey Basin MPG. This is consistent with, and continues the decline reported in, the 2015 status review (NWFSC 2015).

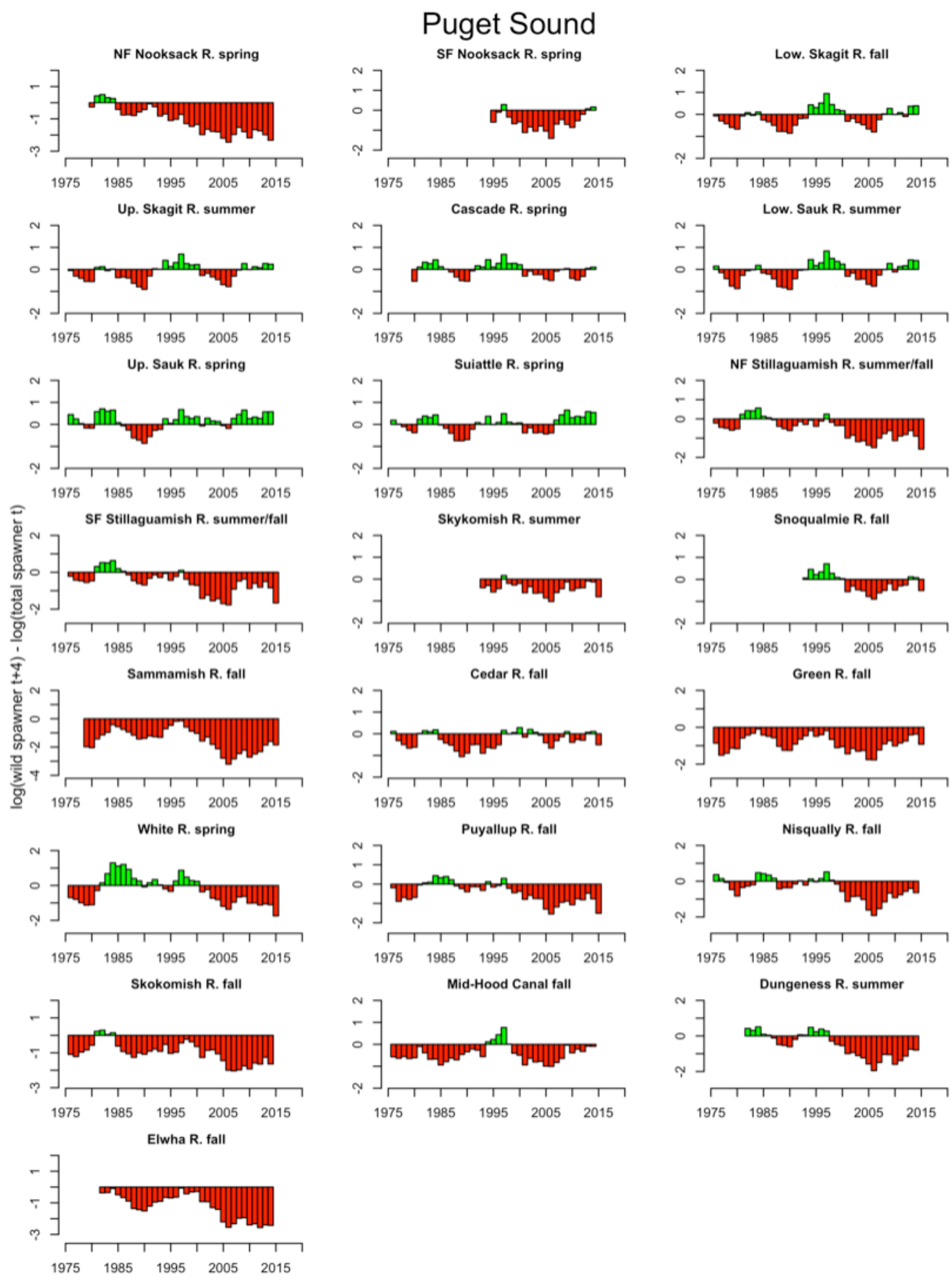

Figure 91. Trends in population productivity, estimated as the log of the smoothed natural-origin spawning abundance in year t minus the smoothed natural-origin spawning abundance in year $(t-4)$.

Harvest

Puget Sound Chinook salmon are harvested in ocean salmon fisheries, in Puget Sound fisheries, and in terminal fisheries in the rivers. They migrate to the north, so for most Puget Sound Chinook salmon populations, nearly all of the ocean fishery impacts occur in Canada and Alaska, where they are subject to the U.S.–Canada Pacific Salmon Treaty (PST). Some populations are also harvested at lower rates in the coastal fisheries off Washington and Oregon. Fisheries within Puget Sound are managed by the state and tribal co-managers under a resource management plan. Fishery impact rates vary considerably among MPGs within Puget Sound, primarily because of different terminal-area management and variable exploitation rates in the Canadian and Alaskan fisheries. For populations in the Hood Canal (Skokomish River) and Central/South Sound MPGs (Nisqually, White, Puyallup, and Green Rivers), substantial terminal-area fisheries are directed at hatchery fish that are produced largely to support tribal and recreational fisheries. For populations in the Whidbey Basin (Skokomish, Stillaguamish, and Skagit Rivers) and Strait of Georgia MPGs (Nooksack River), harvest in the northern fisheries accounts for a large portion of the exploitation.

Chinook salmon populations in Puget Sound generally show a similar pattern: declining exploitation rates in the 1990s, and relatively stable-to-increasing exploitation rates since then (Figure 92). This is primarily a result of Canadian interceptions of Puget Sound Chinook salmon off the West Coast of Vancouver Island (WCVI). During the 1990s, Canada sharply reduced WCVI fisheries in response to depressed domestic 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

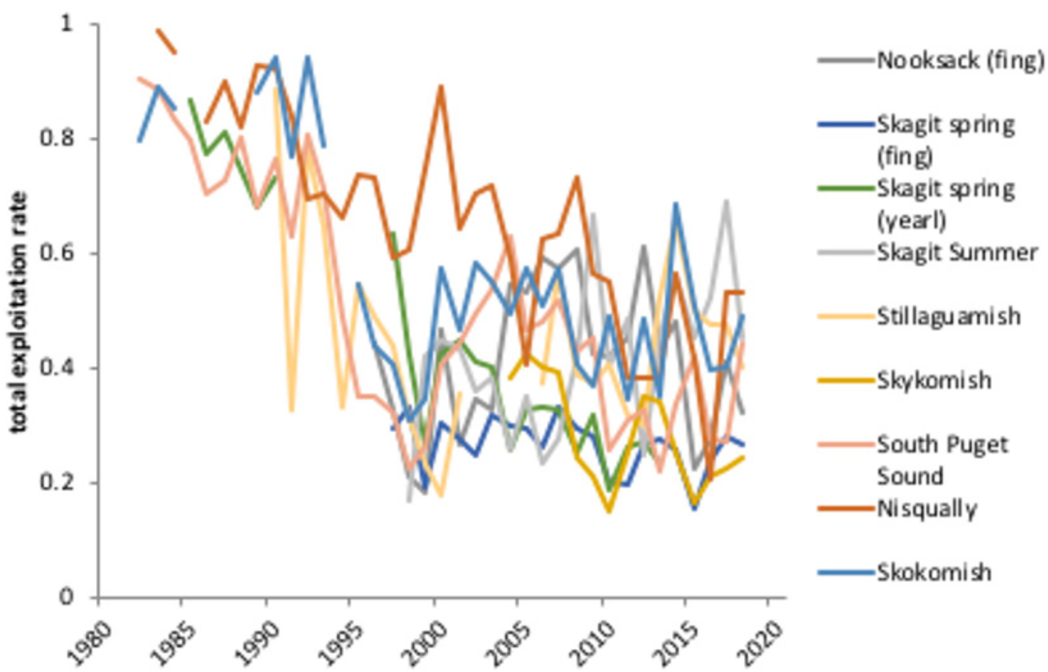

Figure 92. Coded-wire tag-based exploitation rates for Chinook indicator stocks in Puget Sound. From Chinook Technical Committee 2020 Exploitation Rate Analysis, modified to account for mark-selective fisheries in Puget Sound (J. Carey, Pacific Salmon Commission, personal commmunication).

on Puget Sound stocks. A notable exception to this pattern is the North Puget Sound region (Nooksack, Skagit, and Stillaguamish Rivers), as 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 Chinook salmon agreement under the PST, which took effect in 2009, included 30% reductions in Chinook catch ceilings off WCVI and 15% reductions in southeast Alaska. The PST was revised again in 2018, and a new ten-year agreement (2019–28) now specifies further reductions in these catch ceilings at low abundances. Since the 1999 PST Chinook agreement, an abundance-based Chinook management regime, under which fisheries are classified as either aggregate abundance-based management (AABM) or individual stock-based management (ISBM) regimes, has been in place. AABM fisheries constrain catch to a numerical limit computed from either a pre-season forecast or an in-season estimate of abundance; ISBM fisheries constrain annual impacts, within the fisheries of a jurisdiction, for a naturally spawning Chinook salmon stock or stock group (PSC 2020). Goals of the new management regime include an abundance-based framework and the ability to respond to significant changes in the productivity of Chinook salmon stocks, both to preserve the biological diversity of the Chinook salmon resource and to contribute to the restoration of depressed stocks (PSC 2020).

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- vs. hatchery-origin spawners on the spawning grounds.

We can see a declining trend in the proportion of natural-origin spawners across the ESU starting approximately in 1990 and extending through the present (2018). Figure 93 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 2018 time period are the six Skagit River populations. The Skykomish, Snoqualmie, and Cedar River populations had a lower proportion of natural-origin spawners in the late 1990s, but they have rebounded and stayed between 60–90% since the early 2000s. All other populations vary considerably across the whole time period. A number of populations (North and South Fork Nooksack, North and South Fork Stillaguamish, Skykomish, Snoqualmie, White, Puyallup, Nisqually, Skokomish, Dungeness, and Elwha Rivers) show recent declining trends in the fraction natural-origin estimates.

Evidence of the decline in fraction natural-origin spawner abundance is also shown in Table 52. It is important to note that the 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 River populations. With ongoing hatchery programs in the Stillaguamish and Snohomish Rivers, there has been a decrease in five-year mean fraction natural-origin in the last two time periods (2010–14 and 2015–19), particularly in the Stillaguamish River. Note: the fraction natural-origin estimates prior to mass hatchery marking (pre-1997 and 2002–05) in the Skykomish and Snoqualmie Rivers population data have been removed due to concerns by tribal co-managers regarding data quality.

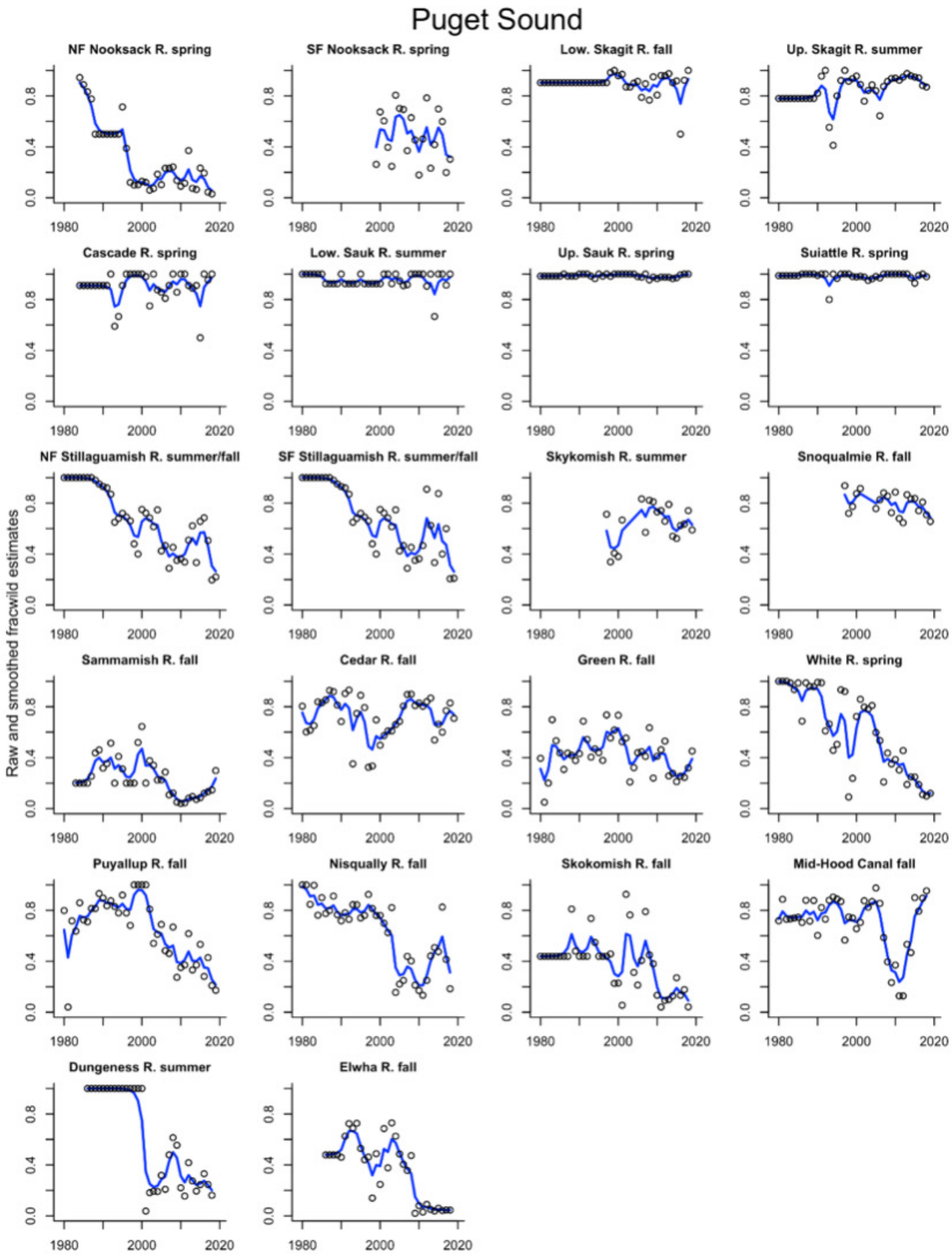

Figure 93. Smoothed trend in estimated fraction of natural-origin spawner abundances (blue line), and annual raw fraction of natural estimates (points).

Table 52. Five-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates).

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
North Fork Nooksack River SP	Strait of Georgia	0.28	0.11	0.19	0.14	0.13
South Fork Nooksack River SP	Strait of Georgia	0.26	0.55	0.57	0.42	0.45
Lower Skagit River FA	Whidbey Basin	0.94	0.91	0.86	0.92	0.84
Upper Skagit River SU	Whidbey Basin	0.91	0.87	0.84	0.95	0.91
Cascade River SP	Whidbey Basin	0.98	0.92	0.89	0.94	0.86
Lower Sauk River SU	Whidbey Basin	0.94	0.97	0.95	0.91	0.98
Upper Sauk River SP	Whidbey Basin	0.99	1.00	0.98	0.97	0.99
Suiattle River SP	Whidbey Basin	0.99	0.97	0.99	0.99	0.97
North Fork Stillaguamish River SU/FA	Whidbey Basin	0.59	0.70	0.40	0.43	0.45
South Fork Stillaguamish River SU/FA	Whidbey Basin	0.59	0.70	0.40	0.54	0.46
Skykomish River SU	Whidbey Basin	0.49	0.52	0.76	0.69	0.62
Snoqualmie River FA	Whidbey Basin	0.81	0.89	0.81	0.78	0.75
Sammamish River FA	Central/South Sound	0.29	0.36	0.16	0.07	0.16
Cedar River FA	Central/South Sound	0.61	0.59	0.82	0.78	0.71
Green River FA	Central/South Sound	0.55	0.47	0.43	0.39	0.30
White River SP	Central/South Sound	0.54	0.79	0.43	0.32	0.15
Puyallup River FA	Central/South Sound	0.88	0.79	0.52	0.41	0.32
Nisqually River FA	Central/South Sound	0.80	0.61	0.30	0.30	0.47
Skokomish River FA	Hood Canal	0.40	0.46	0.45	0.10	0.16
Mid-Hood Canal FA	Hood Canal	0.76	0.79	0.61	0.33	0.89
Dungeness River SU	Strait of Juan de Fuca	1.00	0.32	0.43	0.25	0.25
Elwha River FA	Strait of Juan de Fuca	0.41	0.53	0.35	0.06	0.05

However, the average five-year mean fraction natural-origin estimates for the entire Whidbey Basin MPG remain relatively consistent across all time periods. The Strait of Georgia MPG (North and South Fork Nooksack Rivers) has had increased hatchery influence since the late 1990s and across all time periods. The South Fork Nooksack River population has had extremely small natural fish returns through 2015, but has had increased numbers of natural-origin spawners in the last three years relative to increased supplementation program efforts conducted at Skookum Hatchery (WDFW and PSTIT 2020). This population is at high risk of extinction. The Central/South Sound MPG has had decreasing fraction natural-origin estimates in the Sammamish, Green, White, and Puyallup Rivers populations, and increases in the Cedar population in the three most-recent five-year time periods (2005–09, 2010–14, 2015–19; Figure 93, Table 52). The Nisqually River population data here represent the total volitional escapement, but in the three most-recent years, a supplementation program has been instituted trucking hatchery fish upstream for release on the spawning grounds. This is an effort to supplement natural spawning. In the Hood Canal and Strait of Juan de Fuca MPGs, three of four populations had declining five-year mean fraction natural-origin estimates of fish returns to the spawning grounds. Skokomish River had a slight increase in the most recent five-year time period, but still a very low fraction natural-origin for the population. This population is heavily impacted by the George Adams Salmon Hatchery program. The Mid-Hood Canal population had a higher five-year mean fraction natural estimate in the most recent time period (2014–19) because the hatchery supplementation program was ended in the Hamma Hamma River in 2015. Some supplementation fish continued to return through 2019; however, the population has not proven to be self-sustaining and viable, and recent returns have been very low (Susewind 2020). Genetics data show this population highly correlated to the George Adams Salmon Hatchery and Green River stocks that have been used. State managers conclude from the long-term supplementation program and the genetics composition that if there was an independent population of Chinook salmon that utilized the Mid-Hood Canal streams, then it is most certainly extinct at this point in time. Thus, considering populations by MPG, Whidbey Basin is the only MPG with a consistently high fraction natural-origin spawner abundance, in six of 10 populations. All other MPGs have either variable or declining spawning populations that have high proportions of hatchery-origin spawners.

Biological status relative to recovery goals

The PSTRT 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 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 2–4 Chinook salmon populations in each of the five MPGs in the ESU achieve viability; that at least one population from each major genetic and life-history group historically present within each MPG is viable; and that the populations that do not meet the viability criteria for all four VSP parameters are sustained in order to provide ecological functions and preserve options for ESU recovery. Additional criteria describe habitat conditions that are needed to support viable salmonid populations.

In the Puget Sound Chinook salmon ESU, multiple populations were designated in some river systems. Generally speaking, the data available at that time (including both genetic and spawner abundance) led the TRT to identify some populations based on geographic location of spawning areas. Over the past 15+ years, co-managers have vastly increased the amount and quality of both spawner abundance by area and genetic structure. In a number of river systems, populations that were thought to be geographically isolated have been documented to stray more widely than previously thought (Table 53). Hence we identify these below and give a brief description of each concern relative to management and listing/delisting status. In two cases, Mid-Hood Canal and Sammamish/Cedar Rivers, state and tribal co-managers submitted letters to NOAA to consider a formal change in the population identification (Muckleshoot Indian Tribe 2020, Point No Point Treaty Council 2020, Susewind 2020).

Table 53. Population designation issues in Puget Sound Chinook salmon ESU populations for NWFSC 5-year status review.

ESU	Population	Issue
Puget Sound Chinook salmon	Mid-Hood canal	Did the three streams over which this population is designated historically support Chinook salmon?
	North and South Fork Stillaguamish River	Difference is run timing, not geography.
	Sammamish and Cedar Rivers	Was this river capable of supporting a self-sustaining population?
	North and South Fork Nooksack River	Difference is run timing, not geography.
	Puyallup and White Rivers	Difference is run timing, not geography.
	Lake Cushman "adfluvial" Chinook salmon	Not currently considered a native remnant or viable independent population. Any reason to update in light of Brenkman 2017, recent genetic samples from Tacoma Power, and new passage facilities in operation?

Mid-Hood Canal

Washington state and tribal fishery co-managers have submitted updated information with a) a request from WDFW managers for the Mid-Hood Canal Chinook salmon population to be considered as "not an independent population" and as extinct, and b) a request from Point No Point Treaty Council managers to abstain from such a decision until an ongoing habitat assessment and consideration of a "reintroduction/supplementation program of an appropriate locally adapted natural spawning population" is complete. This would likely be a spring-run Chinook salmon population, rather than the current fall-run population. Both state and tribal co-managers consider that all of the spawning "aggregations" that currently exist in the Duckabush, Dosewallips, and Hamma Hamma Rivers are not sustainable populations nor independent genetically from the current-day Skokomish River population, which is heavily supplemented with Green River fish propagated at George Adams Salmon Hatchery.

In Ruckelshaus et al. (2006), the TRT offered several alternative population structures for the Mid-Hood Canal population from three separate populations (adopted) to Mid-Hood Canal as a subpopulation of a larger Hood Canal population. In the end, the TRT determined that based on historical accounts of Chinook salmon presence, combined with the lack of substantive artificial production (at the time) and the location of the small systems from the nearest major populations (Skokomish and Dungeness Rivers), there was a high likelihood that an independent population was historically present in the Mid-Hood Canal systems, in aggregate. (See pages 54–57 in Ruckelshaus et al. 2006).

The co-managers provide evidence of spawner abundances and genetic analyses that indicate that: a) the 20-year-long supplementations program, ended in 2015, has not produced a sustainable naturally producing Chinook salmon population, and 2) genetic analyses of the existing Mid-Hood Canal Chinook salmon population indicate genetic similarity to the Skokomish River population, including the Green River stock that are used for the George Adams Salmon Hatchery program (Susewind 2020). The viability parameters of spatial structure and diversity are also discussed above. Recent low spawner abundances, despite increased fraction natural estimates in 2018 and 2019, indicate the population cannot be sustained without supplementation. WDFW managers do mention their intention to continue to support habitat restoration and a second attempt to reintroduce Chinook salmon, but of a different broodstock than the previously used Green River/George Adams/Hoodsport stock. The WDFW managers express concern that this population has listing status as "necessary to achieve recovery" for the ESU (i.e., 2–4 populations per MPG; Susewind 2020). However, the tribal managers express concern that any potential change in listing status should more broadly consider harvest, habitat, and genetic diversity parameters that they believe have not yet been adequately identified. In particular, they suggest attempting to reintroduce a locally adapted spring-run population if an ongoing analysis of habitat information is determined to support such a population (Point No Point Treaty Council 2020).

Sammamish River

Tribal fishery managers (Muckleshoot Indian Tribe) have also submitted updated information and a request for the Sammamish River Chinook salmon population to be considered as "not an independent population" and as not distinct from the Cedar River population. They provide information that any of the spawning "aggregations" that previously existed in the Sammamish River and Issaquah Creek watersheds have been heavily populated by hatchery strays for many years, and that current spawners cannot be differentiated from Issaquah Salmon Hatchery or the Cedar River population. Productivity of natural-origin spawner returns indicates that a natural population is not able to persist without the hatchery influence primarily due to poor spawning and rearing habitat in the Sammamish River and its tributaries (Muckleshoot Indian Tribe 2020).

Stillaguamish, Nooksack, White, and Puyallup Rivers

State and tribal co-managers have done considerable genetic mark–recapture work in the North and South Fork Stillaguamish Rivers Chinook salmon populations over the past decade. They have determined that the previously identified "North Fork" and "South Fork" populations are not in fact geographically isolated as determined in Ruckelshaus et al. (2006). Their data do confirm the presence of two populations of Chinook salmon distinguished by genetic characteristics that are expressed in run timing—summer and fall (WDFW and PSTIT 2020). However, the two populations have been determined to overlap in spawn timing and distribution so that both populations spawn in both the North and the South Fork Stillaguamish Rivers. Escapement is still currently estimated for the geographic units rather than for the individual populations (WDFW and PSTIT 2020).

A similar situation exists in the North and South Fork Nooksack Rivers populations, and also in the Puyallup and White Rivers. The TRT previously considered the North Fork and South Fork two populations (both early run-types) that are geographically isolated, but more concerted efforts to obtain spawner abundance data and carcass samples for genetic analyses have allowed more accurate delineation of the genetic makeup of each spawning aggregate. The White River was determined to have an early-run Chinook salmon population that is distinct from the late-run population that remained lower in the system (i.e., in the White River below the diversion dam and in the Puyallup River). Co-managers have done substantial work to obtain more detailed and consistent spawner abundance data and to be able to determine the different components of the populations and their spawning locations. In all cases, data 10–15 years back in time have been re-analyzed and re-tabulated relative to our past data reviews. The new information typically is also used to inform recovery efforts, as well as fishery management decisions, for these populations, though somewhat differently than previously designated by the TRT in Ruckelshaus et al. (2006).

While these new data do now exist, they do not change considerations of listing status in this status review document. We acknowledge the co-managers' request to reassess and revise the Mid-Hood Canal Chinook salmon population status determination, including differences in opinions for possible solutions. We also commend the co-managers' efforts and successes to develop better abundance, productivity, genetic diversity, and spatial structure data. A possible approach to addressing these issues would be to convene a technical team and possibly consider the revision of the Puget Sound Chinook salmon recovery plan. Otherwise, review of the overall plan and ESU recovery goals is necessary, particularly as regards consideration of the spatial structure and diversity viability parameters, and specifically relative to the number of populations in each MPG necessary for recovery. Also, revision of the associated watershed chapters is necessary to describe updated population and habitat information, as well as strategies and actions necessary to achieve Chinook salmon recovery in all MPGs of the Puget Sound Chinook salmon ESU.

Updated biological risk summary

All Puget Sound Chinook salmon populations continue to remain well below the TRT planning ranges for recovery escapement levels. Most populations also remain consistently below the spawner–recruit levels identified by the TRT as necessary for recovery. Across the ESU, most populations have increased somewhat in abundance since the last status review in 2016, but have small negative trends over the past 15 years. Productivity remains low in most populations. Hatchery-origin spawners are present in high fractions in most populations outside the Skagit River watershed, and in many watersheds, the fraction of spawner abundances that are natural-origin have declined over time. Habitat protection, restoration, and rebuilding programs in all watersheds have improved stream and estuary conditions despite record numbers of humans moving into the Puget Sound region in the past two decades. Biannual four-year work plans document the many completed 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 improvements 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, and since the last review, the Puget Sound Partnership has completed this task; however, the program is still not fully functional, neither for providing assessment of watershed habitat restoration/recovery programs, nor for fully integrating the essentially discrete habitat, harvest, and hatchery programs. A recent white paper produced by the Salmon Science Advisory Group of the Puget Sound Partnership concludes that there has been "a general inability of monitoring to link restoration, changes in habitat conditions, and fish response at large-scales" (Puget Sound Partnership 2021). A number of watershed groups are in the process of updating their recovery plan chapters, and this includes prioritizing and updating recovery strategies and actions as well as assessing prior accomplishments. Overall, the Puget Sound Chinook salmon ESU remains at "moderate" risk of extinction, and viability is largely unchanged from the prior review.

Puget Sound Steelhead DPS

Brief description of DPS

This report covers the Puget Sound steelhead DPS. Steelhead are the anadromous form of *O. mykiss*. This DPS includes 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 94; USOFR 2020). The PSTRT 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.

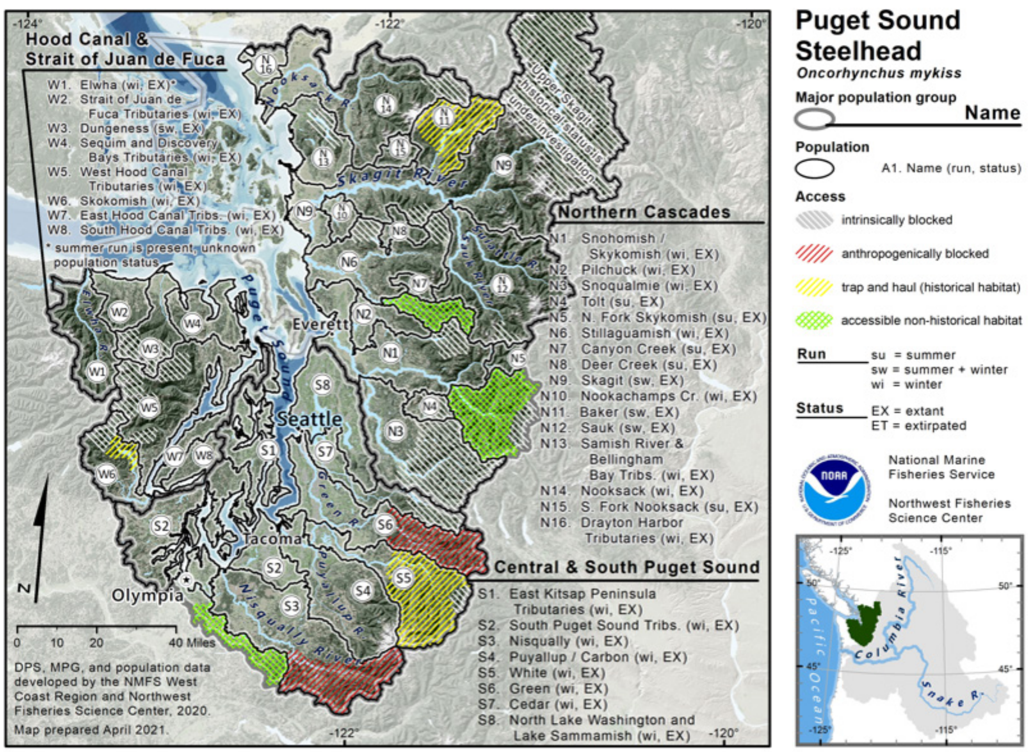

Figure 94. Map of the Puget Sound Steelhead DPS's spawning and rearing areas, identifying 32 demographically independent populations (DIPs) within 3 major population groups (MPGs). The 3 steelhead MPGs are Northern Cascades, Central & South Puget Sound, and Hood Canal & Strait of Juan de Fuca. Areas where dams block anadromous access to historical habitat is marked in red cross-hatching; and areas where historical habitat is accessible via trap and haul programs is marked in yellow cross-hatching. Areas where the laddering of falls has provided access to non-historical habitat is marked in green cross-hatching. Finally, historically inaccessible portions of watersheds are marked in grey and white cross-hatching.

Summary of previous status conclusions

2007

The initial review of this DPS—then designated the Puget Sound steelhead ESU—by the 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 (USOFR 1996). 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 the 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 the 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 (USOFR 2007a); 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 3–10% 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 the Hood Canal & Strait of Juan de Fuca 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 (USOFR 2013); the agency updated its determination of the listing status of the DPS on 14 April 2014 (USOFR 2014).

In 2013, the PSTRT finalized its analyses of Puget Sound steelhead data available through 2011, identifying 32 demographically independent populations (DIPs) and three MPGs within the DPS (Myers et al. 2015) and developing 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- and winter-run populations. Steelhead populations throughout the DPS showed evidence of diminished abundance, productivity, spatial structure, and diversity when compared with available historical evidence for the states of each of these VSP parameters.

2015

The 2015 status review concluded that the biological risks faced by the Puget Sound steelhead DPS had not substantively changed since the listing in 2007, nor since the 2010 status review. In a parallel risk assessment process, the PSTRT 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). Review of abundance trends indicated some minor increases in spawner abundance or improving productivity over the 2–3 years prior to the review; however, most of these improvements were small, and abundance and productivity throughout the DPS remained at levels of concern for demographic risk. Recent increases in abundance that were observed in a few populations were within the range of variability observed in the past several years. Trends in abundance of natural spawners remained predominantly negative. Particular aspects of spatial structure and diversity, including limited availability of suitable habitat, were likely to be limiting the viability of most Puget Sound steelhead populations. Reduced harvest and declining hatchery production of both summer- and winter-run steelhead in the DPS were determined to have decreased those risks to natural spawners.

It was noted that the harvest levels for steelhead in Puget Sound were at very low levels, and that further reductions were not possible or would not significantly improve natural escapement. At the time of the review, environmental trends were not favorable to Puget Sound juvenile and adult steelhead survival and the long term effects of these conditions were forecasted to negatively affect abundance into the near future. Specifically, the exceptionally warm marine waters in 2014 and 2015 and the warm stream temperatures and low summer stream flows observed during 2015 were unfavorable for marine and freshwater survival. These and other environmental indicators pointed to continued warming ocean temperatures, fragmentation or degradation of 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 were expected to constrain any rebound in VSP parameters for Puget Sound steelhead in the near term.

Description of new data available for this review

This report considers population data available through 2019 (where available) to review the current status of Puget Sound steelhead. These data were provided by state and tribal co-managers, including the WDFW and its Salmon and Steelhead Stock Inventory and Salmonscape databases and its district area biologists; Washington tribal biologists; and Northwest Indian Fisheries Commission (NWIFC) biologists. In addition, there have been a number of genetic studies related to Puget Sound steelhead population structure and hatchery-origin steelhead introgression. In most cases, these studies have focused on the influence of non-native hatchery releases on naturally spawning populations (Warheit 2014, Winans et al. 2017, Larson et al. 2018). Results from a number of ongoing studies focusing on juvenile survival in the Salish Sea have been published since the last review. WDFW also produced a report (Cram et al. 2018) that reviewed many of the viability factors discussed in the status review updates. The report focuses on assessing DPS viability with a subset of the DPS for which updated population data are available. Only 20 populations or population groups had sufficient data for statistical analysis. Using the VSP approach, the assessment included a basic analysis of abundance and trend data (abundance and productivity), in addition to spatial structure and diversity, but does not attempt to replicate the population viability analyses (PVAs) conducted in the previous status review update (NWFSC 2015). Additional analyses of Puget Sound steelhead population demographics, distribution, and habitat are provided by the Puget Sound steelhead recovery plan (NMFS 2019b).

Abundance and productivity

The long-term abundance of adult steelhead returning to many Puget Sound rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s; however, in the nearer term, there has been a relative improvement in abundance and productivity (Figures 95 and 96). Of the 20 datasets analyzed, abundance trends were available for seven of the eight winter-run DIPs in the Hood Canal & Strait of Juan de Fuca MPG; for five of the eight winter-run DIPs in the Central & South Puget Sound MPG; and for seven of the 11 winter-run DIPs, but only one of the five summer-run DIPs, in the Northern Cascades MPG (Table 54). One-third of the populations lack monitoring and abundance data; in most cases it is likely that abundances are very low. The data submitted only included natural-origin spawners, therefore statistical analyses for natural spawners and total spawners were identical.

Hood Canal & Strait of Juan de Fuca MPG

In general, populations in this MPG experienced an increase in abundance during the 2015–19 period. The five-year geomean for the Elwha River DIP increased to 1,241 winter-run steelhead, an 82% increase over the 2010–14 period. The five-year geomean for natural-origin spawners rose to 358, a 6% increase. In addition, summer-run steelhead have been observed in the upper Elwha River, with recent counts in the low hundreds of returning adults (Pess et al. 2020). Rather than a recolonization, these fish appear to be reanadromized *O. mykiss* from summer-run steelhead originally isolated behind the Elwha and Glines Canyon Dams. Although a summer run may also persist as residents or at very low abundances elsewhere in the MPG, the Elwha River population is the only extant summer run identified; and while precise data on this “population” are lacking, it represents a considerable contribution to the DPS. The Skokomish River winter-run steelhead DIP exhibited a five-year geomean abundance of 958, an 80% increase over the previous five-year period, and represents the second largest DIP in this MPG (Table 54). Further, both the long-term trend, 10% (Table 55), and recent productivity (Figure 96) are both strongly positive. The Dungeness River summer- and winter-runs DIP abundance was estimated at 408; however, this represented a 21% decrease over the previous period (Table 54). Longer-term trends could not be calculated, but the current abundance level

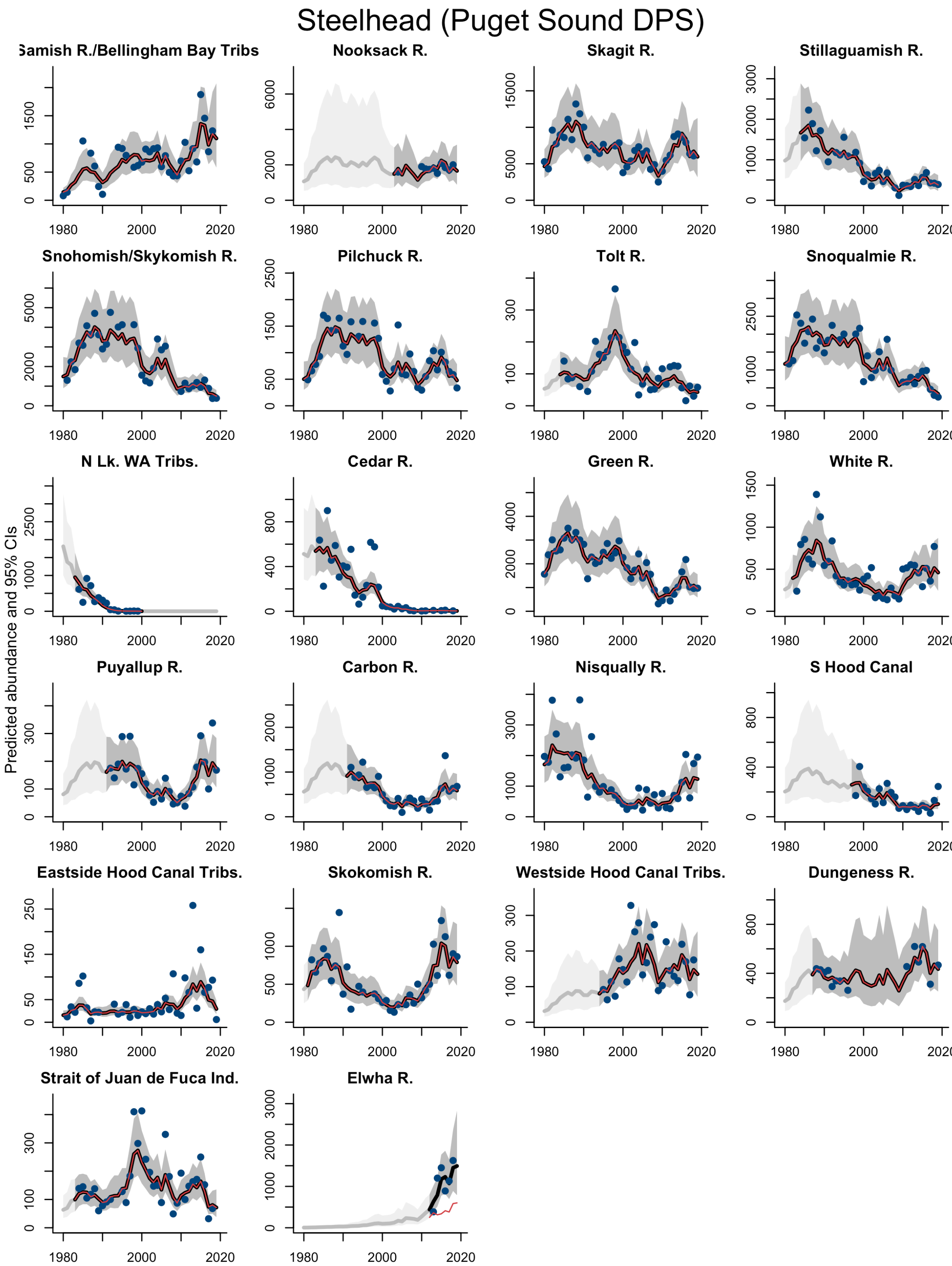

Figure 95. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot. *Note*: For this DPS, all abundance data, except for Elwha River, are only for natural-origin spawners.

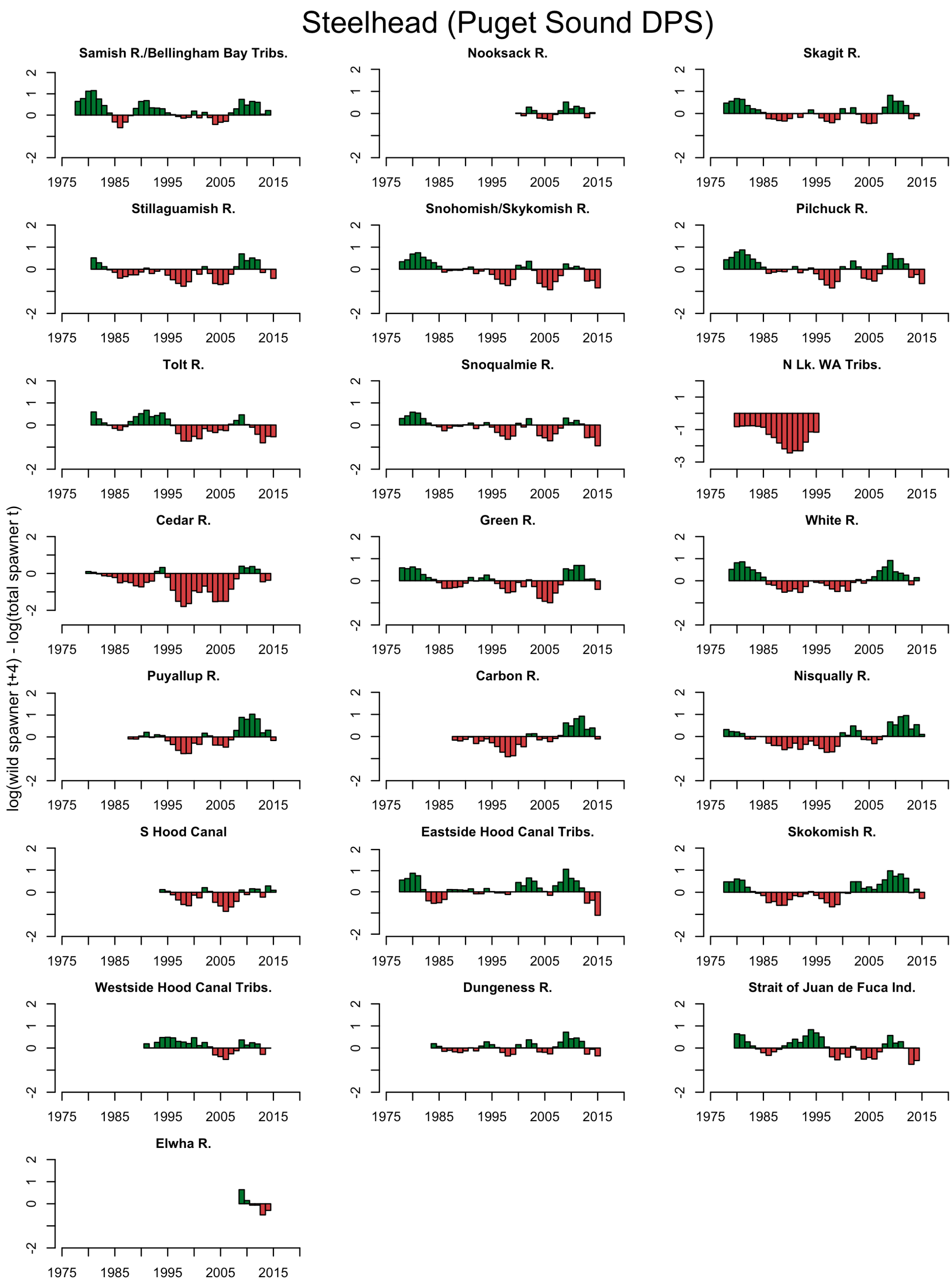

Figure 96. Trends in population productivity of Puget Sound steelhead, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year $(t-4)$.

Table 54. Five-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 none or only one estimate of natural spawners was available. A single value not in parentheses means that the fraction natural was 1.0 and thus, the total count was the same as the natural-origin count. 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. Key: HCSJF = Hood Canal & Strait of Juan de Fuca MPG; NC = Northern Cascades MPG; CSPS = Central & South Puget Sound MPG.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
South Hood Canal W	HCJSF	—	263	176	145	69	91	32
Eastside Hood Canal Tributaries W	HCJSF	27	21	25	37	60	54	–10
Skokomish River W	HCJSF	385	359	205	320	533	938	76
Westside Hood Canal Tributaries W	HCJSF	—	97	208	167	138	150	9
Dungeness River SU/W	HCJSF	356	—	—	—	517	448	–13
Strait of Juan de Fuca Independent Tributaries W	HCJSF	89	191	212	118	151	95	–37
Elwha River W	HCJSF	—	—	—	—	338 (680)	358 (1,241)	6 (82)
Samish River/Bellingham Bay Tributaries W	NC	316	717	852	535	748	1,305	74
Nooksack River W	NC	—	—	—	—	1,745	1,906	9
Skagit River SU/W	NC	7,202	7,656	5,419	4,677	6,391	7,181	12
Stillaguamish River W	NC	1,078	1,166	550	327	386	487	26
Snohomish/Skykomish Rivers W	NC	3,629	3,687	1,718	2,942	975	690	–29
Pilchuck River W	NC	1,225	1,465	604	597	626	638	2
Snoqualmie River W	NC	1,831	2,056	1,020	1,250	706	500	–29
Tolt River SU	NC	112	212	119	70	108	40	–63
North Lake Washington Tributaries W	CSPS	60	4	—	—	—	—	—
Cedar River W	CSPS	241	295	37	12	4	6	50
Green River W	CSPS	2,062	2,585	1,885	1,045	662	1,289	95
White River W	CSPS	524	311	301	173	514	451	–12
Puyallup River W	CSPS	167	196	93	72	85	201	136
Carbon River W	CSPS	969	800	335	246	290	735	153
Nisqually River W	CSPS	1,200	754	409	446	477	1,368	187

is an improvement over estimates from the 1990s. The remaining populations consist of assemblages of small tributaries with abundances of less than 250 individuals. The three Hood Canal winter-run populations (South Hood Canal, Eastside Hood Canal Tributaries, and Westside Hood Canal Tributaries) all experienced increases in abundance from nine to 55% (Table 54), but remain at relatively low population abundances. The Strait of Juan de Fuca Independent Tributaries winter-run DIP abundance fell below 100 for its five-year geomean, a 37% decrease over the previous period. Finally, no information was available for the Sequim and Discovery Bay Tributaries winter-run DIP. Based on previous monitoring in Snow Creek, a small tributary in this DIP, overall abundance is not likely more than 100. Overall, this MPG exhibited an increase in abundance related to the expansion of steelhead spawning in the Elwha River and general improvements among populations in Hood Canal. Total abundance, however, was still low-to-moderate.

Northern Cascades MPG

Within the Northern Cascades MPG over the last five years, there has been considerable variability in the performance of individual basins. The winter-run populations in the Samish River and Bellingham Bay tributaries exhibited a 74% increase in the five-year geometric mean abundance, with an average 1,305 natural-origin spawners for the present review period (Table 54). Additionally, this estimate is an underestimate as it does not include tributaries to Bellingham Bay. Winter-run DIPs in the Nooksack and Skagit River basins exhibited slight increases in their average five-year abundances, although within the 2015–19 period a negative trend in abundances is evident in the Skagit River populations (Figure 95). The Skagit River dataset, which represents five DIPs and may include some summer steelhead redd counts, contains the majority of steelhead estimated in this MPG, with a geomean of 7,181 (Table 54).[13] The Stillaguamish River winter-run DIP exhibited a moderate increase in its five-year geomean abundance of 26% (Table 54), although the longer-term trend for this population from abundance levels in the 1980s is strongly downward (Figure 95). Further, the Stillaguamish River abundance estimate is based on an index rather than a total estimate. DIPs in the Snohomish River basin were stable or negative. The Pilchuck River winter-run DIP experienced a 2% increase in five-year geomean, while both the Snohomish/Skykomish and Snoqualmie River winter-run DIPs both exhibited 29% decreases in recent five-year geomean abundances. Long-term (15-year) trends (2005–19) were also significantly negative, with 8% and 6% annual declines for the Snohomish/Skykomish and Stillaguamish River DIPs, respectively (Table 55).

The Tolt River summer-run DIP, the only summer run (of five in this MPG) for which there was a long-term dataset, experienced a 63% decline in five-year abundance during the 2015–19 period. In addition, there has been a negative 4% trend in abundance since 2005. The current five-year geomean for the Tolt River DIP is only 40 spawners, although this represents redds found in between RMs 3.3 and 7.8. No data were provided for the Drayton Harbor Tributaries winter-run DIP, not for four summer-run DIPs (South Fork Nooksack River, Deer Creek, Canyon Creek, and North Fork Skykomish River). It is assumed that these populations persist, but at very low abundances.

This MPG represents the majority of the abundance for the entire Puget Sound steelhead DPS, with a total abundance (based on five-year geomeans) of over 10,000 natural spawners. Several populations have abundances over 1,000, and others over 250; however, over a third of the populations are not sufficiently monitored to develop population abundance estimates and likely have very low numbers of spawners. Except for the Samish River/Bellingham Bay Tributaries DIP and perhaps the Nooksack and Skagit Rivers, productivity (based on adult:adult ratios) for most populations was negative (in contrast to the five-year geomean trends), suggesting a downward trend into the near future (Figure 96).

[13] Of the five DIPs in the Skagit River, the abundance estimate only includes the Skagit and Sauk Rivers and the Nookachamps Creek DIPs; estimates are not available for the Baker (near zero) or the Cascade River DIPs.

Table 55. Fifteen-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 and with at least 2 data points in the first 5 years and last 5 years of the 15-year period are shown.

Population	MPG	1990–2005	2004–19
South Hood Canal W	HCJSF	—	−0.05 (−0.08, −0.02)
Eastside Hood Canal Tributaries W	HCJSF	0.01 (0.00, 0.03)	0.04 (0.00, 0.08)
Skokomish River W	HCJSF	−0.06 (−0.07, −0.05)	0.10 (0.08, 0.13)
Westside Hood Canal Tributaries W	HCJSF	—	−0.02 (−0.04, 0.00)
Strait of Juan de Fuca Independent Tributaries W	HCJSF	0.05 (0.01, 0.08)	−0.04 (−0.07, −0.01)
Samish River/Bellingham Bay Tributaries W	NC	0.04 (0.02, 0.07)	0.05 (0.02, 0.08)
Skagit River SU/W	NC	−0.03 (−0.04, −0.01)	0.02 (0.00, 0.07)
Stillaguamish River W	NC	−0.07 (−0.09, −0.04)	0.00 (−0.03, 0.03)
Snohomish/Skykomish Rivers W	NC	−0.05 (−0.07, −0.03)	−0.09 (−0.11, −0.06)
Pilchuck River W	NC	−0.06 (−0.08, −0.03)	0.00 (−0.03, 0.02)
Snoqualmie River W	NC	−0.04 (−0.06, −0.03)	−0.07 (−0.09, −0.04)
Tolt River SU	NC	0.00 (−0.04, 0.04)	−0.04 (−0.06, −0.02)
Cedar River W	CSPS	−0.19 (−0.23, −0.14)	−0.11 (−0.16, −0.06)
Green River W	CSPS	−0.03 (−0.05, −0.01)	−0.01 (−0.06, 0.03)
White River W	CSPS	−0.07 (−0.08, −0.06)	0.07 (0.05, 0.09)
Puyallup River W	CSPS	−0.06 (−0.08, −0.04)	0.07 (0.04, 0.11)
Carbon River W	CSPS	−0.10 (−0.12, 0.08)	0.07 (0.04, 0.09)
Nisqually River W	CSPS	−0.10 (−0.11, −0.08)	0.08 (0.04, 0.11)

Central & South Puget Sound MPG

Steelhead populations in the Central and South Puget Sound MPG exhibited strongly positive increases in their five-year abundances. Four populations represent the major basins in this MPG: Green River, Puyallup River, White River, and Nisqually River winter-run DIPs exhibited 94–187% increases in five-year abundances (Table 54). Long-term (15-year) trends for three of these populations—White, Puyallup, and Nisqually—were positive, with annual growth rates of 6–8%, while the Green River DIP long-term trend remained stable at 0% (Table 55). Abundances for the White and Puyallup River winter-run DIPs remain in the low hundreds and continue to be at some demographic risk, although estimates include counts from only portions of the DIPs. Further, abundances for the Puyallup/Carbon River DIP include data series for the Puyallup and Carbon Rivers that could not be combined due to differences in survey protocols. Recent productivity for these four populations has been predominately positive (Figure 96). Two DIPs in the Lake Washington watershed, North Lake Washington Tributaries and Cedar River, had adult abundances near zero, based on fish ladder counts (Chittenden Locks) and Landsburg Dam (Cedar River) and redd counts; however, large numbers of resident *O. mykiss* are found in the Cedar River (Cram et al. 2018). Lastly, no information was available for the East Kitsap Peninsula Tributaries and South Puget Sound Tributaries DIPs. It is assumed that these populations persist, but at very low levels. Total abundance for this MPG is still in the low thousands of fish.

Harvest

Harvest of Puget Sound steelhead is limited to terminal tribal net fisheries and recreational fisheries. In response to declining abundance throughout the 1990s, harvest rates were curtailed in 2003, with "wild" harvest rates reduced to below 10% (NMFS 2018). Recreational fisheries are mark-selective for hatchery stocks, but some natural-origin steelhead are encountered, with a proportion of those fish subject to hooking mortality and noncompliance. Hatchery steelhead production for harvest is primarily of Chambers Creek winter-run stock (South Puget Sound) and Skamania Hatchery summer-run stock, both of which have been selected for an earlier run timing than natural stocks to minimize fishery interactions. In tribal net fisheries, most indirect 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.

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 10–40%, 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 (Figure 97). In 2018, NMFS approved a resource management plan (RMP) for the Skagit Basin that allowed for the directed take of ESA-listed steelhead through both net fisheries and the catch-and-release recreational fishery (NMFS 2018). Under this plan, harvest rates would be based on overall escapement.

Other mortality factors

Steelhead juveniles emigrating from tributaries draining to the Salish Sea face a host of potential predators. Pearson et al. (2015) reviewed information related to avian and marine mammal species that may have influenced the decline in Puget Sound steelhead populations. Increases in the abundance of marine mammals, such as harbor seals (*Phoca vitulina*), or avian predators may be related to decreased juvenile steelhead survival (Moore et al. 2015); tag studies estimated the survival of wild steelhead from tributaries to the Pacific Ocean at only 16%. Berejikian et al. (2016) further implicated harbor seals as a factor in the poor survival of steelhead smolts in the Salish Sea. A mitigating factor in survival appears to be structures, such as the Hood Canal Floating Bridge, that delay migration and make steelhead smolts more susceptible to predation (Moore et al. 2010, 2013). Genetic fitness, tributary-specific freshwater effects, and distance traveled in the Salish Sea from tributary to ocean appear to be factors influencing survival (Moore and Berejikian 2017). In addition, the introduction of freshwater piscivorous species—e.g., largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), walleye (*Sander vitreus*), etc.—may be limiting steelhead viability. Continued increases in the populations of predator species in conjunction with declines in other forage fishes may further reduce the survival of emigrating juvenile steelhead.

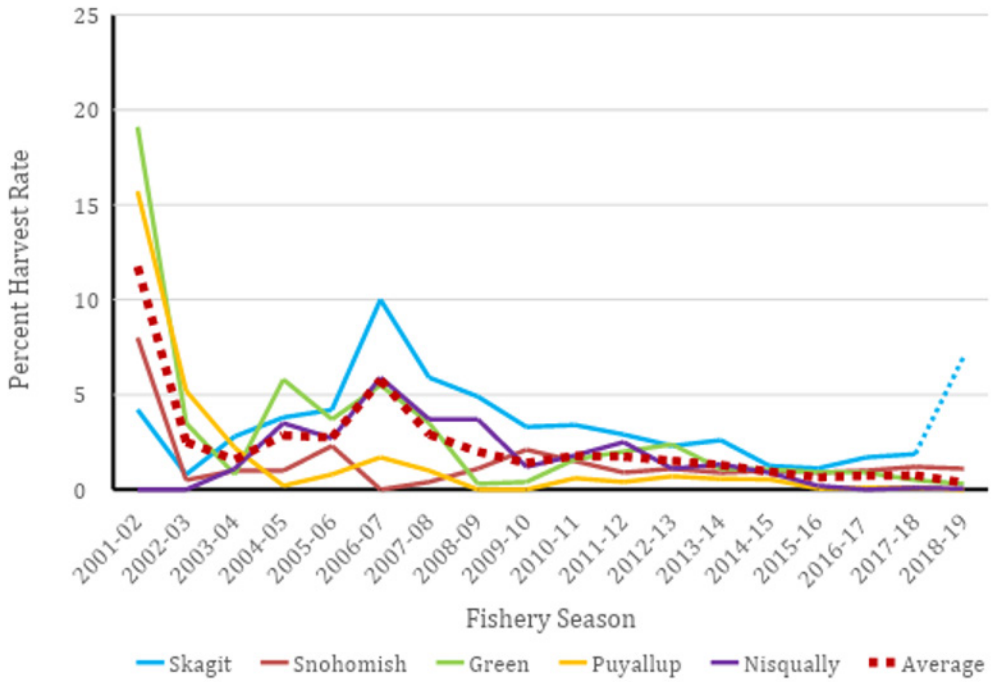

Figure 97. Tribal and non-tribal terminal harvest rate percentages on natural-origin steelhead for five index steelhead populations in Puget Sound, 2001–19. Dotted blue line = post-Skagit RMP harvest rates, a sliding scale regime based on pre-season terminal escapement estimates to the Skagit River. Dotted red line = average harvest rates across the five populations through 2017, and excludes Skagit River for 2018 and 2019. Data from WDFW and PSIT (2016, 2017, 2018, 2019).

Spatial structure and diversity

Abundance and productivity are demographic characteristics of a population that determine its ability to persist into the foreseeable future. Spatial structure and diversity, the other two VSP parameters (McElhany et al. 2000), are characteristics that influence a population's ability to persist and evolve over a much longer time course. Spatial structure and diversity consider a population's 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. Spatial structure and diversity buffer a population against short-term environmental fluctuations and long-term climatic change. Compromised spatial structure and diversity are ultimately expressed as longer-term declines in abundance and productivity.

Diversity can be measured through a variety of life-history trait metrics, for example: age structure, run timing, spawning. It is difficult, however, to interpret the significance of changes in life-history traits under changing environmental conditions. Indeed, the responsiveness of life-history traits to environmental change may be a measure of adequate diversity. It is also unclear if the apparent loss of a phenotype is merely an example of plasticity, rather than the loss of the underlying genetic diversity. One of the few quantifiable risks to diversity is the loss of locally adapted traits through introgression by non-native or domesticated hatchery-origin fish.

Abundance information provided for this update only included natural-origin spawners, so we were unable to calculate the contribution of hatchery-origin fish on the spawning grounds. Moreover, information on the proportion of hatchery-origin spawners (pHOS) is rarely obtained in steelhead spawning surveys due to the near absence of carcasses (to identify hatchery marks). In those basins where hatchery production continues, the magnitude and origin of hatchery releases provides one indicator of the potential risks to diversity.

The recovery plan for Puget Sound steelhead (NMFS 2019b) recognizes that production of hatchery fish of both run types—winter- and summer-run—has posed a considerable risk to diversity in natural steelhead in the Puget Sound steelhead DPS. Because of the origin and aspects of the propagation history of these fish in Puget Sound, the TRT (Hard et al. 2015) considered continued hatchery production of steelhead to represent a major threat to the diversity VSP component for the DPS. Historically, the majority of winter-run broodstocks produced in hatcheries in the DPS were derived from the Chambers Creek stock (southern Puget Sound), which is considered highly domesticated and has been selected repeatedly for early spawn timing for decades, a trait known to be heritable in salmonids (the natural population is now extinct); alternatively, summer-run hatchery broodstock are derived from the long-running Skamania Hatchery stock from the lower Columbia River basin (i.e., out-of-DPS-origin). In response to the risk of introgression between native steelhead populations and hatchery-origin, there has been a general decrease in the overall production from several hatcheries. In addition, Chambers Creek releases were discontinued in the Elwha and Skagit River basins during the last five-year period. Chambers Creek programs continue in the Dungeness, Nooksack, Stillaguamish, Snohomish, and Skykomish River basins. Integrated hatchery programs have emerged in place of many of the Chambers Creek programs; these programs incorporate naturally produced returning adults as broodstocks. Programs are currently underway in the Elwha, Green, and White Rivers. It is planned to discontinue the release of Skamania Hatchery-origin summer-run steelhead in the near future from the three programs currently operating. Additionally, there are plans to develop an integrated hatchery broodstock using local summer-run steelhead; currently the plan is to use unmarked summer-run fish returning to the South Fork Skykomish River. The genetic status of naturally returning summer-run fish to the Skykomish River is currently being studied, and there will likely be new information forthcoming to better understand the level of legacy introgression by Skamania Hatchery summer-run steelhead.

Overall, the risk posed by hatchery programs to naturally spawning populations has decreased during the last five years with reductions in production (especially with non-local programs) and the establishment of locally sourced broodstock (Figures 98–101). Unfortunately, whereas competition and predation by hatchery-origin fish can be readily diminished, it is unclear how long it will take to remove the genetic legacy of introgression by natural selection.

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). There were a number of events that occurred in Puget Sound during the last review period that affected steelhead habitat. While the 2014 completion of the Elwha and Glines Canyon Dam removals occurred during the previous period, the response of steelhead to this action is still being evaluated. It is clear, however, that steelhead are accessing much of this newly available habitat (Pess et al. 2020). Passage operations have begun on the North Fork Skokomish River to reintroduce steelhead above Cushman Dam; although juvenile collection efficiency is still relatively low, further improvements are anticipated. Similarly, improvements in the adult fish collection facility at Mud Mountain Dam (White River) are near completion, with the expectation that improvements in adult survival will facilitate better utilization of habitat above the dam (NMFS 2014). The July 2020 removal of the diversion dam on the Middle Fork Nooksack Dam and of the Pilchuck River Diversion Dam will provide access to important headwater spawning and rearing habitats. Similarly, the proposed modification of Howard Hanson Dam for upstream fish passage and downstream juvenile collection (NMFS 2019a) in the longer term will allow winter steelhead to return to historical headwater habitat in the Green River. It has been hypothesized that summer-run steelhead may have been residualized above Howard Hanson Dam (Myers et al. 2015); restoring access could restore such a run. The effects of these two projects on abundances will not be evident for some time. Four of the top six steelhead populations identified by Cram et al. (2018) as having habitat blocked by major dams are in the process of having passage restored or improved. While fish passage/collection operations are currently underway in the Baker River for sockeye, coho, and Chinook salmon, returning steelhead are not currently transferred above the Baker River dams. In addition, projects focusing on smaller-scale improvements in habitat quality and accessibility are ongoing. Some 8,000 culverts that block steelhead habitat have been identified in Puget Sound (NMFS 2019b), with plans to address these blockages being extended over many years. Small-scale improvements in habitat, restoration of riparian habitat, and reconnecting side- or off-channel habitats will allow better access to habitat types and niche diversification. While there have been some significant improvements in spatial structure, it is recognized that land development, loss of riparian and forest habitat, loss of wetlands, and demands on water allocation all continue to degrade the quantity and quality of available fish habitat.

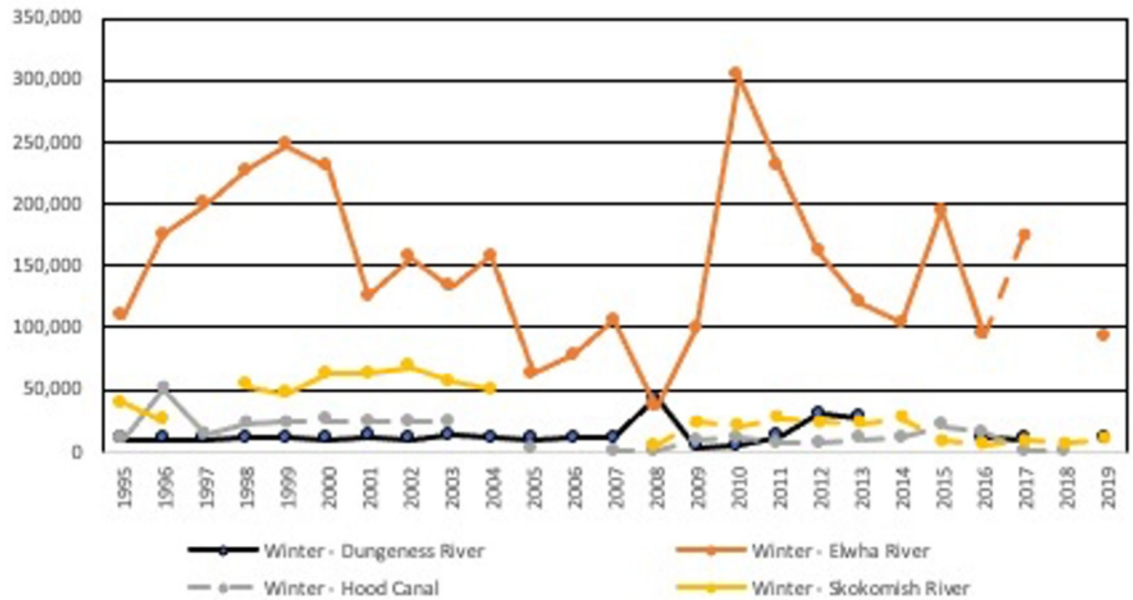

Figure 98. Hatchery releases of winter-run steelhead into rivers of the Hood Canal & Strait of Juan de Fuca MPG, 1995–2019. Dashed portions of lines indicate that some or all of the fish released were native to that population. In addition, releases of fish weighing <2.0 g were not included. Data from the Regional Mark Information System (https://www.rmpc.org, June 2020).

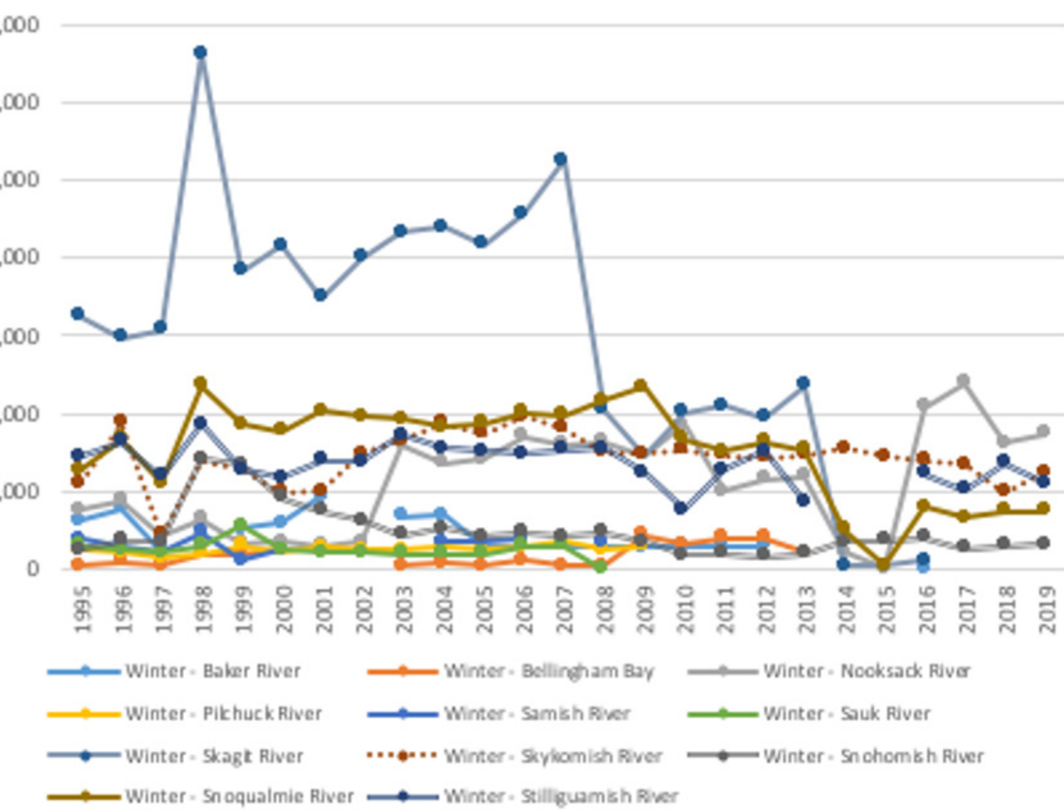

Figure 99. Releases of hatchery-reared winter-run steelhead into rivers in the North Cascades MPG. The majority of these releases are of hatchery stocks not native to the receiving watersheds. In addition, releases of fish weighing less than 2.0 grams were not included. Data from the Regional Mark Information System (https://www.rmpc.org, June 2020).

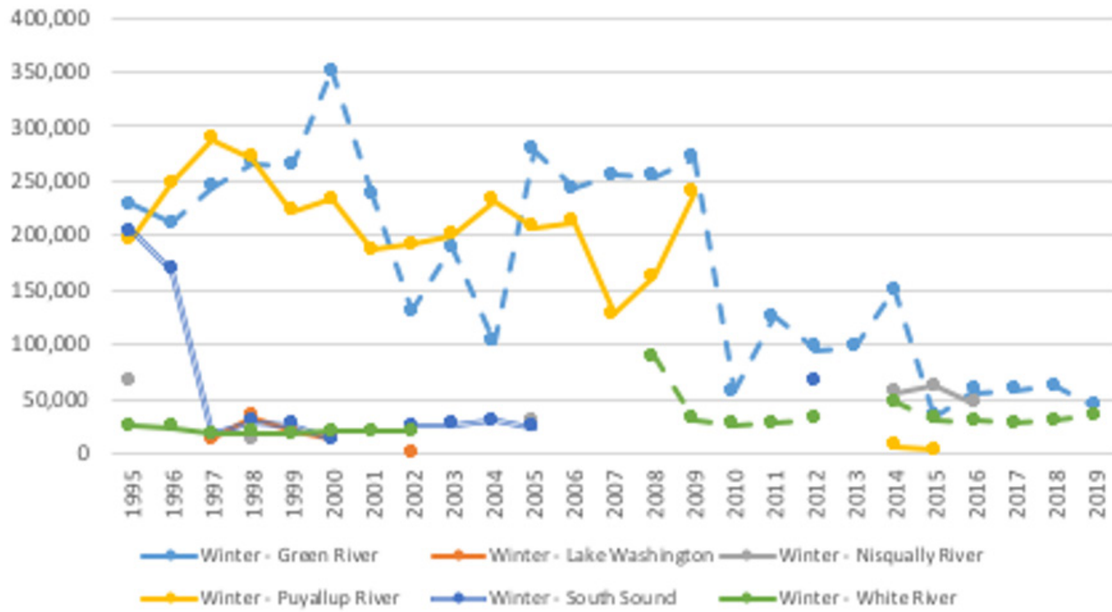

Figure 100. Hatchery releases of winter-run steelhead into rivers of the Central & South Puget Sound MPG, 1995–2019. Dashed portions of lines indicate that some or all of the fish released were native to that population. In addition, releases of fish weighing <2.0 g were not included. Data from the Regional Mark Information System (https://www.rmpc.org, June 2020).

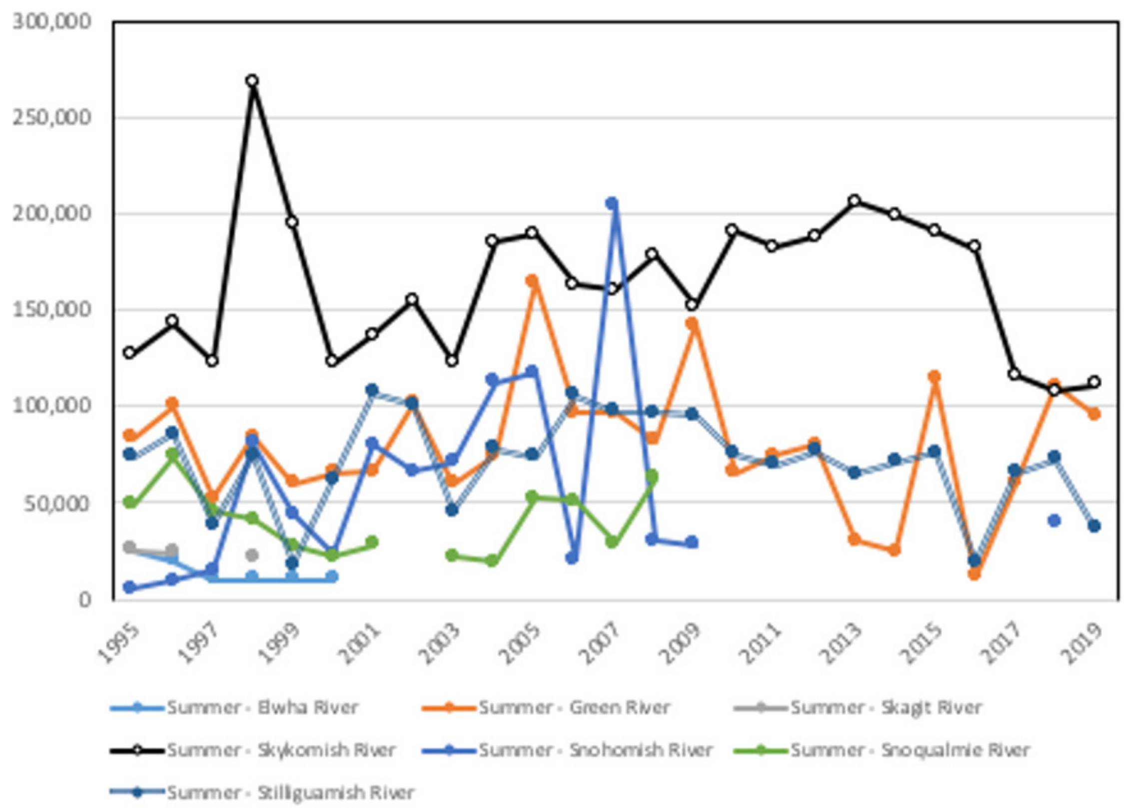

Figure 101. Hatchery releases of summer-run steelhead into Puget Sound DPS rivers from 1995 to 2019. All releases are from hatchery stocks that were founded by out-of-DPS Skamania Hatchery summer-run steelhead. Intermittent releases into other rivers are not shown. In addition, releases of fish weighing less than 2.0 grams were not included. Data from Regional Mark Information System (RMIS: https://www.rmis.org/) accessed 3 June 2020.

Biological status relative to recovery goals

The Puget Sound Steelhead Recovery Team was established by NMFS and convened in March 2014 to develop a recovery plan for the Puget Sound steelhead DPS. This recovery plan was finalized in December 2019 (NMFS 2019b). Recovery targets were calculated using a two-tiered approach adjusting for years of low and high productivity. Abundance information is unavailable for approximately one-third of the DIPs, disproportionately so for summer-run populations. In most cases where no information is available, it is assumed that abundances are very low. Some population abundance estimates are only representative of part of the population (index reaches, etc.). Where recent five-year abundance information is available, 30% (6/20) are at less than 10% of their high productivity recovery targets (lower abundance target); 65% (13/20) are between 10% and 50% of their targets; and 5% (1/20) are between 50% and 100% (Table 56). Although most populations for which data are available experienced an improvement in abundance during the last five years (Figure 95), significant increases in abundance are necessary for all populations to reach even the high productivity (low abundance) recovery targets. A key element to achieving recovery is recovering a representative number of both winter- and summer-run steelhead populations, and the restoration of viable summer-run DIPs would appear to be a long-term endeavor. Alternatively, the relatively rapid reestablishment of summer-run steelhead in the Elwha River does provide a model for potentially reanadromizing summer-run steelhead residualized behind impassable dams. Another diversity element factored into achieving recovery is the proportion of hatchery-origin fish that spawn naturally. Currently, the standard for the proportion of non-native hatchery-origin fish on natural spawning grounds is 5%.

Updated biological risk summary

Consideration of the above analyses indicates that the viability of the Puget Sound steelhead DPS has improved somewhat since the PSTRT 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). Increases in spawner abundance were observed in a number of populations over the last five years (Figure 95). These improvements were disproportionately found within the Central & South Puget Sound and the Hood Canal & Strait of Juan de Fuca MPGs, primarily among smaller populations. The apparent reversal of strongly negative trends among winter-run populations in the White, Nisqually, and Skokomish Rivers abated somewhat the demographic risks facing those populations. Certainly, improvement in the status of the Elwha River steelhead (both winter- and summer-run) following the removal of the Elwha dams reduced the demographic and diversity risk for the DIP and the MPG. Improvements in abundance were not as widely observed in the Northern Cascades MPG. Foremost among the declines were summer- and winter-run populations in the Snohomish River basin. These populations figure prominently as sources of abundance for the MPG and DPS. Additionally, the decline in the Tolt River summer-run steelhead population was especially of concern given that it is the only population for which we have abundance estimates. The demographic and diversity risks to the Tolt River summer-run DIP are very high. In fact, all summer-run steelhead populations in the Northern Cascades MPG are likely at a very high demographic risk. In spite of improvements in some areas, most populations are still at relatively low abundance levels, with about a third of the DIPs unmonitored and presumably at very low levels.

Table 56. Recent (2015–19) 5-year geometric mean of raw natural spawner counts for Puget Sound steelhead populations and population groups compared with Puget Sound steelhead recovery plan high and low productivity recovery targets (NMFS 2019b). Asterisks indicate that the abundance is only a partial population estimate. Superscript *1s* ([1]) indicate that these populations have a combined target. Abundance is compared to the high-productivity individual DIP targets. Colors indicate the relative proportion of the recovery target currently obtained: red = <10%, orange = 10% > *x* < 50%, yellow = 50% > *x* < 100%, green = >100%.

		Abundance		
			Target	
MPG	**Population**	**2015–19**	**High productivity**	**Low productivity**
HCSJF	South Hood Canal	91	2,100	7,100
HCSJF	Eastside Hood Canal Tributaries	93	1,800	6,200
HCSJF	Skokomish River	958	2,200	7,300
HCSJF	Westside Hood Canal Tributaries	150	2,500	8,400
HCSJF	Dungeness River	408	1,200	4,100
HCSJF	Strait of Juan de Fuca Independent Tributaries	95	1,000	3,300
HCSJF	Elwha River	358	2,619	2,619
HCSJF	Sequim and Discovery Bay Tributaries	n/a	500	1,700
NC	Samish River/Bellingham Bay Tributaries	1,305*	1,800	6,100
NC	Nooksack River	1,906	6,500	21,700
NC	Skagit River	7,181.*[1]	15,000	15,000
NC	Stillaguamish River	487	7,000	23,400
NC	Snohomish/Skykomish Rivers	690	6,100	20,600
NC	Pilchuck River	638	2,500	8,200
NC	Snoqualmie River	500	3,400	11,400
NC	Tolt River (SU)	40	300	1,200
NC	Drayton Harbor Tributaries	n/a	1,100	3,700
NC	South Fork Nooksack River (SU)	n/a	400	1,300
NC	Sauk River	[1]	15,000	15,000
NC	Nookachamps River	[1]	15,000	15,000
NC	Baker River	[1]	15,000	15,000
NC	Canyon Creek (SU)	n/a	100	400
NC	Deer Creek (SU)	n/a	700	2,300
NC	North Fork Skykomish River (SU)	n/a	200	500
CSPS	North Lake Washington Tributaries	n/a	4,800	16,000
CSPS	Cedar River	n/a	1,200	4,000
CSPS	Green River	1,282	5,600	18,700
CSPS	White River	130	3,600	12,000
CSPS	Puyallup/Carbon Rivers	136	4,500	15,100
CSPS	Nisqually River	1,368	6,100	20,500
CSPS	East Kitsap Tributaries	n/a	2,600	8,700
CSPS	South Sound Tributaries	n/a	6,300	21,200

Continued limits on harvest will facilitate population rebuilding during "good" (high escapement) years and buffer against demographic risks under "bad" (low escapement) years. Artificial propagation programs have undergone major changes in both the quantity and quality of hatchery fish produced. The proposed termination of the non-native Skamania Hatchery-origin summer-run steelhead programs represents a major effort to reduce introgression, although the genetic legacy of past hatchery releases remains to be determined. The release of the domesticated Chambers Creek hatchery-origin winter steelhead continues in a limited number of basins. More importantly, integrated programs with locally sourced broodstocks have been established to assist in recovery. Risks to diversity from hatchery programs continue, but at a reduced level. Furthermore, self-sustaining natural populations of winter-run steelhead persist throughout the DPS, albeit at low abundances, and with a very limited risk of interaction with hatchery-origin steelhead. Overall, the status of summer-run steelhead populations, or the summer-run component of summer/winter populations, remains somewhat precarious. Information is absent for many populations, and, with the possible exception of the Elwha River, the remaining populations have critically low abundances and/or varying levels of genetic introgression by out-of-DPS sources. There are a number of planned, ongoing, and completed events that will likely benefit steelhead populations in the future, but have not yet effected changes in adult abundance. Among these are the removal of the diversion dam on the Middle Fork Nooksack River, passage improvements at Mud Mountain Dam, the ongoing passage program in the North Fork Skokomish River, and the planned passage program at Howard Hansen Dam. Dam removal in the Elwha River and the resurgence of the endemic winter and summer steelhead runs have underscored the benefits of restoring passage. The Elwha River scenario is perhaps somewhat unique in that upstream habitat is in pristine condition, and smolts emigrate into the Strait of Juan de Fuca, not Puget Sound or Hood Canal. Improvements in spatial structure can only be effective if done in concert with necessary improvements in habitat. Habitat restoration efforts are ongoing, but land development and habitat degradation, concurrent with increasing human population in the Puget Sound corridor, may result in a continuing net loss of habitat. Overall, recovery efforts in conjunction with improved ocean and climatic conditions have resulted in an increasing viability trend for the Puget Sound steelhead DPS, although the extinction risk remains "moderate."

Hood Canal Summer-run Chum Salmon ESU

Brief description of ESU

This 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 102; USOFR 2020). The Puget Sound Technical Recovery Team identified two independent populations for 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).

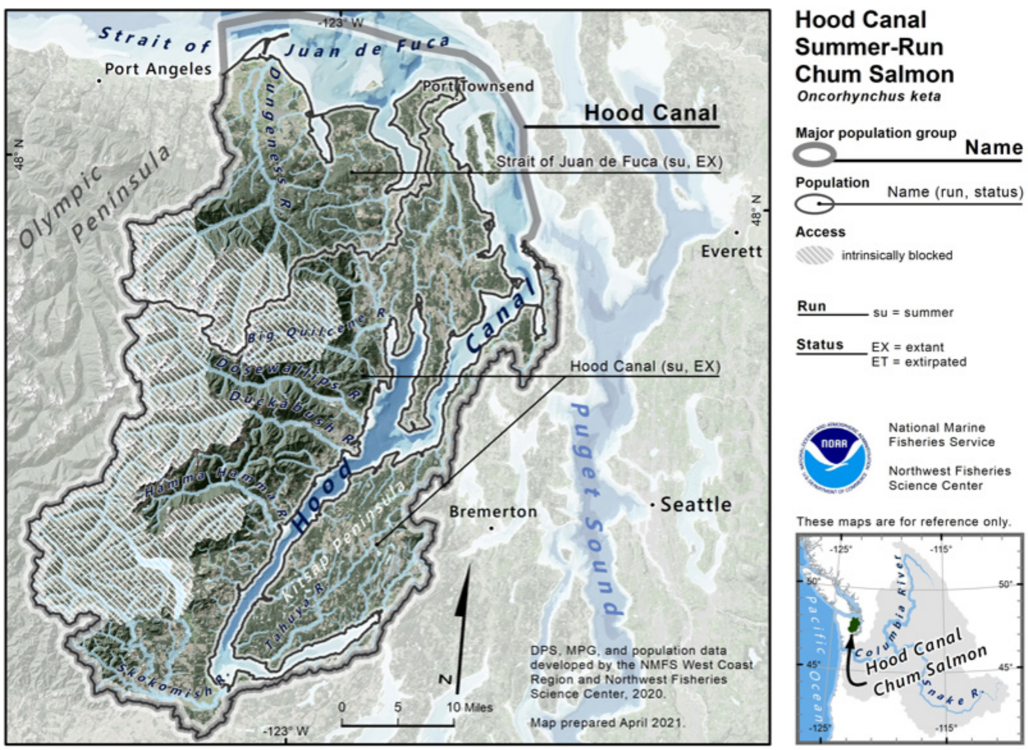

Figure 102. Map of the Hood Canal summer-run chum salmon ESU's spawning and rearing areas, illustrating populations and major population groups.

Summary of previous status conclusions

2005

At the time of the 2005 status review (Good et al. 2005), the PSTRT 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 five years. While spawning abundances had remained relatively high compared to the low levels in the early 1990s, productivity had decreased significantly, being lower for broodyears 2002–06 than for any previous five-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 had been extended further upstream as abundance increased. Overall, however, the new information considered in 2010 did not indicate a change in the biological risk category since the time of the previous BRT status review in 2005.

2015

NWFSC (2015) identified that natural-origin spawner abundance had increased since ESA listing, and spawning abundance targets in both populations had been met in some years. Productivity rates had increased in the prior five years, and had been greater than replacement rates in the past two years for both populations. However, productivity of individual spawning aggregates showed that only two of eight aggregates had viable performance. Spatial structure and diversity viability parameters for each population had increased and nearly met the viability criteria. Despite substantive gains toward meeting viability criteria in the Strait of Juan de Fuca and the Hood Canal summer-run chum salmon populations, the ESU did not meet all of the recovery criteria for population viability.

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 for each fishery management area; data are available from 1974 through 2019. Catch data are available only through 2018. The Point No Point Treaty Tribes (PNPTT) and WDFW completed a five-year review of the Summer Chum Salmon Conservation Initiative for the period 2005–13 (PNPTT and WDFW 2014) detailing all data listed above, and also providing some corrections to previous estimates. The co-managers are currently in the process of completing a new five-year review for the period 2014–19, but the results are not yet published. Adult return data for 2019 were still preliminary. The new review will also include an updated viability analysis. Data presented here in this document include exactly the same data (through 2019). Estimates of age composition for each stream or natural spawning aggregation are also available for the newer period, 2015–19. A genetic stock identification and assessment program was conducted for the 2005–19 time period, and extensive collection of data (DNA, scales, lengths, otoliths, sex, and abundance) was conducted. Mark recoveries of otoliths and adipose fin-clipped returning adults were conducted primarily on the spawning grounds, allowing estimation of level of straying for the SOS fish to other drainages, and estimation of total returns of both natural- and supplementation-origin fish. Supplementation programs were begun in 1992, prior to which all adult summer chum salmon returns to the Strait of Juan de Fuca and Hood Canal 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 supplementation-origin chum salmon contributed to total adult returns. Estimates of the proportions of hatchery fish on the spawning grounds are available from 1974 through 2019 for the Strait of Juan de Fuca and Hood Canal populations (S. Bass, PNPTT, and M. Downen, WDFW, personal communications). 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 8.4–62.8%, with a range of 5.8–40.2% in the Hood Canal population. The hatchery contribution also generally decreased as supplementation programs were terminated as planned (PNPTT and WDFW 2014). All were ended by 2011 in the Strait of Juan de Fuca population, and by 2017 in the Hood Canal population. 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 Strait of Juan de Fuca and the Hood Canal populations, and are shown from 1980 through 2019 in this review (Figure 103). Smoothed trends in estimated total and natural population spawning abundances for both populations have generally increased over the 1980 to 2017 time period. Shorter-term trends, specifically from 2002–16 for the Strait of Juan de Fuca population and from 2003–17 for the Hood Canal population, have coincided with the supplementation programs. The co-managers' 2018 assessment (Lestelle et al. 2018) provides evidence that increased abundances have been sustained at a level higher than during the period of listing. However, since 2016, abundances for both populations have sharply decreased. This began in 2017 for the Strait of Juan de Fuca population and in 2018 for the Hood Canal population. This newest information is important in considering summer-run chum salmon abundance and productivity trends, and the co-managers theorize it to be related to Pacific Decadal Oscillation (PDO) effects on ocean conditions (Lestelle et al. 2018, Lestelle 2020).

Average escapements (geometric means) for five-year intervals beginning in 1990 show estimates of trends over the intervals for both natural-origin spawners (NOS) and total (NOS + SOS) spawners (Table 57). The Strait of Juan de Fuca population had a 29% decrease in abundance of natural-origin (43% decrease in total) spawners in the most recent five-year time period (2015–19) vs. the 2010–14 period. The Hood Canal population had a 46% increase in abundance of natural-origin (40% increase in total) spawners in the same period. Spawner abundances in both populations were lowest throughout the 1990s, but increased in the early 2000s and had been sustained through 2016.

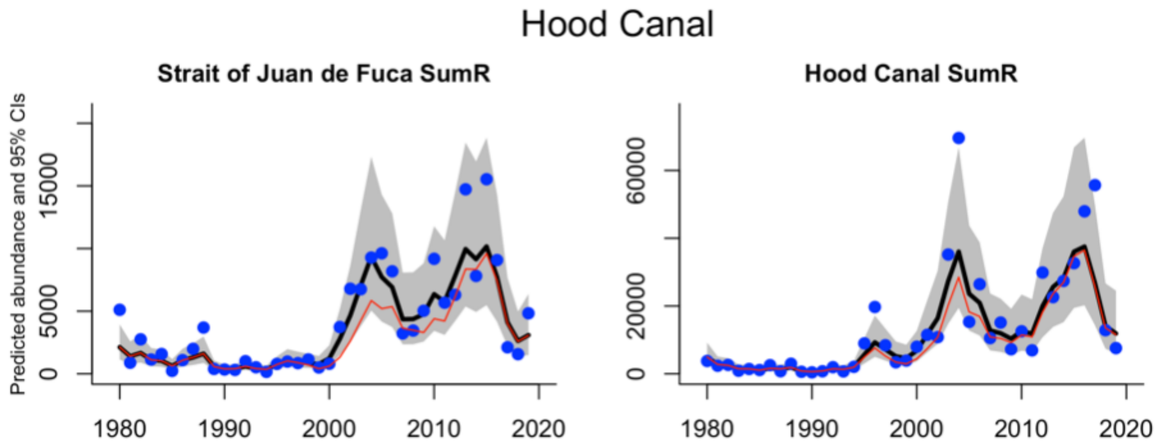

Figure 103. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

Table 57. Fifteen-year trends in log natural-origin spawner abundance computed from a linear regression applied to the smoothed natural-origin spawner log abundance estimate versus year. Only populations with at least 4 natural-origin spawner estimates from 1990 to 2019 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	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Strait of Juan de Fuca SU	Hood Canal	386 (386)	628 (822)	2,195 (4,178)	4,023 (5,353)	6,417 (8,231)	4,580 (4,684)	−29 (−43)
Hood Canal SU	Hood Canal	979 (979)	5,170 (7,223)	13,115 (18,928)	11,284 (13,605)	16,085 (17,419)	23,533 (24,314)	46 (40)

Fifteen-year trends in log natural-origin spawner abundance were computed over two time periods (1990–2005 and 2004–19) from a linear regression model applied to the smoothed natural spawner log abundance estimate over annual return years. Trends were strongly positive in the two populations in the first time period, but abundance trends for both populations have decreased to close to zero in the most recent 15-year period (Table 57).

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 decreased over the past three-to-four years, but had been above replacement rates in five prior years. Productivity rates have varied above and below replacement rates over the entire time period (Figure 104). 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.

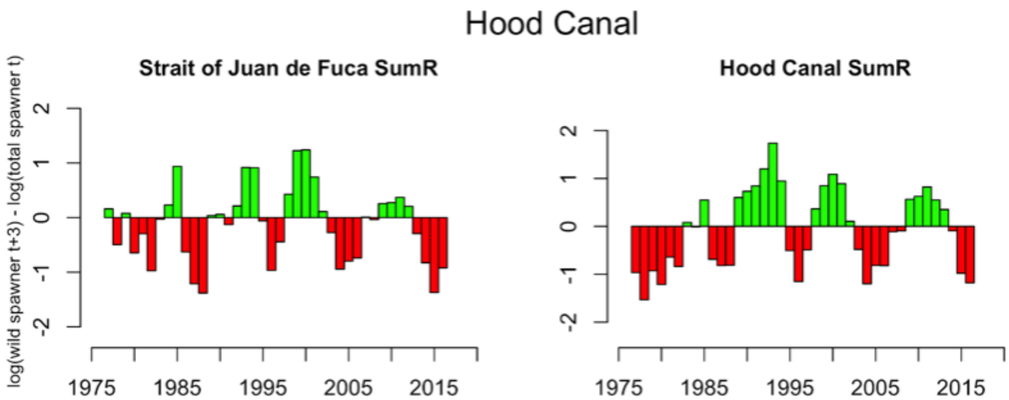

Figure 104. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year (*t* – 4).

Table 58. Fifteen-year trends in log natural spawner abundance computed from a linear regression applied to the smoothed natural spawner log abundance estimate versus year. Only populations with at least 4 natural spawner estimates from 1990 to 2019 and with at least 2 data points in the first 5 years and last 5 years of the 15-year period are shown.

Population	MPG	1990–2005	2004–2019
Strait of Juan de Fuca SU	Hood Canal	0.17 (0.11, 0.23)	0.00 (–0.05, 0.05)
Hood Canal SU	Hood Canal	0.22 (0.17, 0.27)	0.02 (–0.03, 0.08)

Harvest

There are no directed fisheries on Hood Canal summer-run chum salmon. 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 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 50–70%, 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 salmon, and harvest rates fell dramatically (Figure 105). The co-managers 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 exceeded historic levels. Under the BCR, harvest rates have declined to less than 2% for the Strait of Juan de Fuca population and to about 3–15% for Hood Canal summer chum salmon (Lestelle et al. 2018). 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 and 2007–09 in Hood Canal fisheries. From 2000 through 2018, the harvest rate for the ESU has averaged about 7% (data from S. Bass, personal communication).

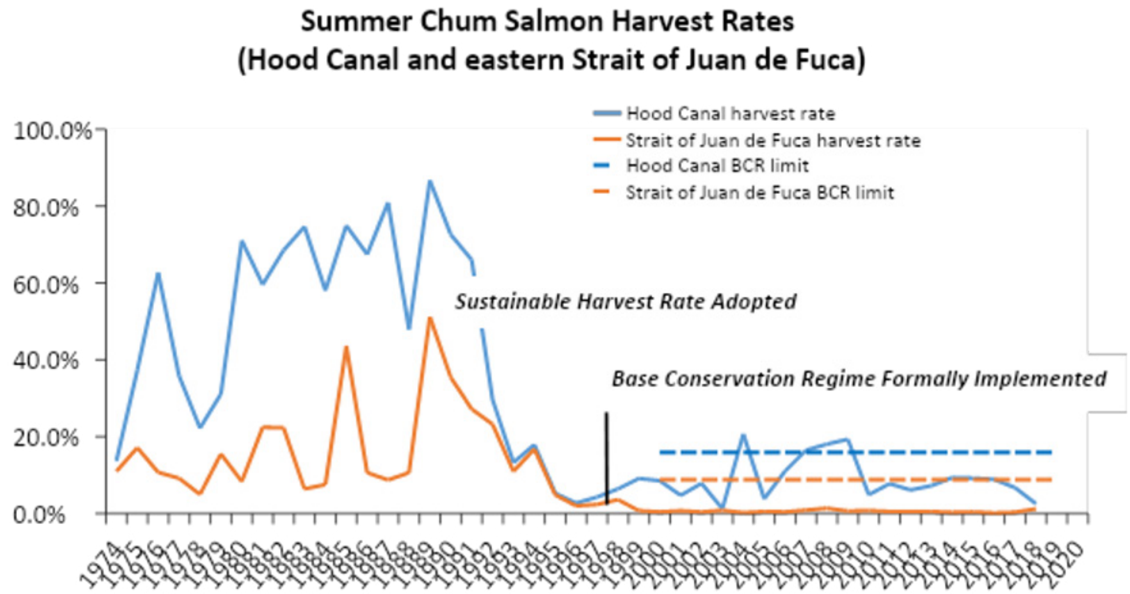

Figure 105. Total exploitation rate on the Hood Canal summer-run chum salmon ESU. Data from co-manager run reconstruction (1974–2018, data from S. Bass, personal communication; figure from PNPTT and WDFW 2014, with updated data).

Spatial structure and diversity

Spatial structure and diversity measures for the Hood Canal summer-run chum salmon recovery program include the reintroduction and sustaining of natural-origin spawning in multiple small streams where summer chum salmon spawning aggregates had been extirpated. A supplementation program was initiated in 1992 to meet this objective, and supportive habitat protection and restoration projects have been conducted in many of the streams as well. The first 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). Previously extirpated spawning subpopulations were reintroduced and have now rebounded in Chimacum Creek in the Strait of Juan de Fuca population. Reintroductions in Big Beef Creek and Tahuya River in the Hood Canal population have not been quite as successful, and it does not appear that the rebounding natural-origin production will be sustained (Lestelle et al. 2018). Two other streams, Dewatto and Skokomish Rivers, which had been deemed extirpated by the PSTRT at the time of their analysis, and which have not had reintroduction efforts, have seen subpopulations rebound substantially in recent years. This follows completion of habitat restoration projects in the lower portion of the Skokomish River and estuary. Habitat on the Dewatto River remains largely intact (Lestelle et al. 2018).

One measure of spatial structure and diversity parameters is related to the proportion of NOS vs. SOS on the spawning grounds. All returning summer-run chum salmon spawners were natural in both populations until fish from the supplementation program began to return to spawn in 1995 (Figure 106). 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 Creeks)

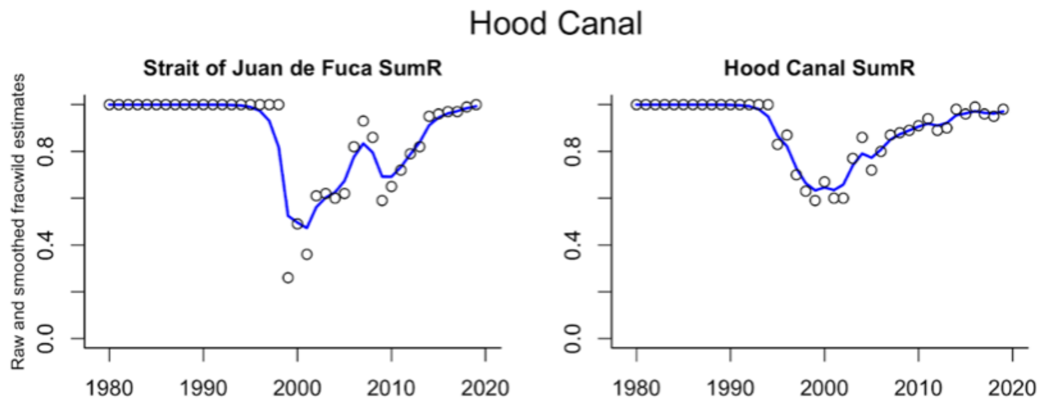

Figure 106. Proportion of each summer-run chum salmon population comprising natural-origin spawners. Line shows a smoothed trend and points show the annual estimates.

Table 59. Five-year mean of fraction natural-origin spawners (sum of all estimates divided by the number of estimates).

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Strait of Juan de Fuca SU	Hood Canal	0.85	0.54	0.76	0.79	0.98
Hood Canal SU	Hood Canal	0.72	0.70	0.83	0.92	0.97

were phased in between 1992 and 2003. As program goals were met, all programs have now been terminated. Lilliwaup Creek did not meet the production targets (e.g., broodstock collections and release numbers) in some earlier years and also had a lack of focused habitat protection/restoration efforts, so supplementation was continued in Lilliwaup Creek through broodyear 2017 (Lestelle et al. 2018). As SOS fish returns have phased out, there has been a gradual return to predominantly natural-origin spawners for both populations (Figure 106, Table 59). For the Hood Canal population, SOS fish returned to the Dewatto and Tahuya Rivers and to Lilliwaup Creek through 2018. The Strait of Juan de Fuca population shows estimates of nearly 100% NOS since 2015 (Figure 106).

Biological status relative to recovery goals

The PSTRT defined the abundance and productivity viability criteria for the Hood Canal summer-run chum salmon ESU using two different PVAs. The first assumed density independence at low population sizes and a replacement growth factor of 1:1. The second method—a viability and risk assessment procedure, or VRAP—assumed 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 state and tribal co-managers have done so several times in recent years. Co-managers conducted a new viability analysis using VRAP in 2017, and again in 2020 with data through return year 2019 (as described above; Lestelle et al. 2014, 2018, Lestelle 2020). These update the analysis conducted by the PSTRT and presented in Sands et al. (2009). The potential impacts of shifts in decadal-scale ocean (i.e., the PDO) and climate regimes on summer chum salmon performance and potential limits to recovery were also considered in the 2017 analysis, and the most recent years (2017–19) of reduced abundances and productivities during a warm PDO ocean cycle which started in 2014 provide more proof for this theory (Lestelle et al. 2014, 2018, Lestelle 2020). Observations of spawner abundances and productivities over the next few years will be necessary to adequately evaluate what the effects of a warm phase of the PDO will require for summer chum salmon management and recovery goals.

The PSTRT VRAP analysis concluded that 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 were attained in the Strait of Juan de Fuca population in 2013 and 2015, and in the Hood Canal population in five years since recovery implementation efforts began (2012, 2014–17; NWFSC unpublished data). Productivity for the Strait of Juan de Fuca population was greater than 1:1 during broodyears 2007–12, and for the Hood Canal population during broodyears 2009–13, though productivity has varied for both populations over the entire time period (Figure 104).

The PSTRT 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 were, for Strait of Juan de Fuca, 4,500 adults (given intrinsic productivity of 5 and capacity of 3,300) and, for Hood Canal, 18,300 adults (given intrinsic productivity of 5 and capacity of 13,500). Results of the co-managers' updated VRAP analysis in 2017 (through broodyear 2013) showed that the minimum abundance viability threshold with zero harvest for Strait of Juan de Fuca population had again increased over prior analyses (2009 and 2014) to 6,300, for an intrinsic productivity of 6 and capacity of 4,600. For the Hood Canal population, the minimum abundance viability threshold in the 2017 analysis had decreased to 4,800 over the prior analyses, with an intrinsic productivity of 8 and capacity of 3,600, with zero exploitation rate. These changes in the viability thresholds were considered to be due to longer datasets that produced more precise estimates of coefficient of variation (CV), particularly for the Hood Canal population, which had one outlier data point—a very high return from the 2000 broodyear (3.5× as high as the next-highest return) that was never repeated. The CV for the Hood Canal population was considerably lower in the more recent analysis also because there are ten spawning aggregations, as opposed to only three in the Strait of Juan de Fuca population (Lestelle et al. 2018). Results of 2017 VRAP analyses also suggested the Hood Canal population would be considered to be at negligible risk of extinction considering current biological performance, provided that the exploitation rate remains very low. The Strait of Juan de Fuca population had a much higher risk of extinction, even with a zero exploitation rate (Lestelle et al. 2018). As noted above, since 2017, both populations have experienced much lower returns, and a 2020 update of the VRAP analysis resulted in considerably reduced population performance under a changing ocean climate. The analysis, with the addition of the 2017–19 return data and put in the context of warm (broodyears 1979–98 and 2014–16) and cool (broodyears 1999–2013) PDO ocean climate cycles, indicates that the Hood Canal population does still exceed the 5% threshold risk curve during both warm and cool PDO regimes with a 0% exploitation rate. At the 30% exploitation rate and during the years of the cool PDO regime, the Hood Canal population greatly exceeded the 5% threshold. The Strait of Juan de Fuca population did exceed the 5% extinction risk threshold with a 0% exploitation rate during the cool ocean regime, but has a very high risk of extinction in the warm PDO time periods even with a 0% exploitation rate (Lestelle 2020).

Table 60. Seven Hood Canal summer-run chum salmon ESU ecological diversity groups proposed by the PSTRT, by geographic region and associated spawning aggregation (from Sands et al. 2009).

MPG	Proposed ecological diversity group	Spawning aggregation(s): Extant* and extinct**
Strait of Juan de Fuca	Dungeness	Dungeness River (unknown status)
	Sequim–Admiralty	Jimmycomelately Creek* Salmon Creek* Snow Creek* Chimacum Creek**
Hood Canal	Toandos	(unknown)
	Quilcene	Big Quilcene River* Little Quilcene River*
	Mid-West Hood Canal	Dosewallips River* Duckabush River*
	West Kitsap	Big Beef Creek** Seabeck Creek** Stavis Creek** Anderson Creek** Dewatto River** Tahuya River** Mission Creek** Union River*
	Lower-West Hood Canal	Hamma Hamma River* Lilliwaup Creek* Skokomish River*

In addition, analyses of individual spawning aggregations indicate that four of six extant spawning aggregations in the Hood Canal population are at relatively high risk of extinction (see Lestelle 2020, Figure 13). The Quilcene spawning aggregation has much higher performance than any of the others, which holds under both warm and cool PDO regimes. Dosewallips, Duckabush, Union, and Hamma Hamma River performances are still viable at the 5% risk threshold under the cool PDO ocean regime, even with 30% exploitation rates, but none of these spawning aggregates are viable under the warm PDO regime, even with zero exploitation rates. These results indicate the importance of the Quilcene spawning aggregation 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. In the Strait of Juan de Fuca population, both Salmon/Snow and Jimmycomelately Creek spawning aggregations are viable with a 30% exploitation rate under the cool PDO regime, but neither a re viable under the warm PDO regime (Lestelle 2020).

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 60). 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 (NMFS 2007, Sands et al. 2009).

Co-managers revisited these criteria and provided guidance in Lestelle et al. (2018). Some new genetics data have prompted these suggestions, with a primary focus including evidence that straying appears to occur along shorelines and into adjacent/neighboring streams, but does not appear to occur across Hood Canal. Hence they suggest that the West Kitsap geographic unit be split into two units, East Hood Canal and South Hood Canal. One "robust" spawning aggregate would be required in each geographic unit. They define the term "robust" and suggest that the most likely spawning aggregates to meet these needs—given geographic location, current available habitat, and spawner abundances—would be Big Beef Creek and Dewatto River in East and South Hood Canal units, respectively (Lestelle et al. 2018). These spawning aggregates would likely need to have re-introduction efforts in order to succeed. It is important to note that this definition of spatial structure and diversity is considered more in terms of habitat capacity and quality than our measure above of genetic diversity or natural- and supplementation-origin abundance. Also relevant in this distinction is that the concept of maintaining high-quality habitat (i.e., promoting protection and completing restoration actions) in these streams is considered to have substantial importance to build resilience of the individual spawning aggregates of the two summer chum salmon populations through time periods of poor ocean productivity. See Lestelle et al. (2018) for a complete discussion of these concepts and analyses. A second spatial structure/diversity suggestion, for the Strait of Juan de Fuca population, was that this population would have less risk of extinction if there were a robust spawning aggregate in the Dungeness River. Regardless of lack of documented historic abundances, they believe this could be achieved with reintroduction and habitat restoration techniques employed in other streams in the ESU. They present data and link observations of re-established spawning, particularly in the Skokomish River, and suggest that large-scale estuarine habitat restoration efforts have contributed to the increases in spawner abundance (Lestelle et al. 2018).

Criteria for spatial structure/diversity were nearly met for Strait of Juan de Fuca and Hood Canal summer-run chum salmon populations until the recent spawner return abundance downturns that started in 2017. As of 2018, the Quilcene, Dosewallips, Duckabush, Union, and Hamma Hamma River subpopulations met the TRT criteria for viability (and also the co-managers' definition of "robust"). That is, spawning aggregates were present and persistent within five of the six major ecological diversity groups identified by the PSTRT (Table 60). Two subpopulations previously considered extirpated (Skokomish and Dewatto Rivers) also rebounded with spawning aggregations despite not having reintroduction projects. An exception to the TRT criteria regarding distance between spawning aggregations is in East Hood Canal (West Kitsap). Spawning abundance in Big Beef Creek has remained consistently low (zero or near-zero for the last three years), and the Tahuya River spawning aggregate has not sustained adequate natural production after supplementation efforts ended. These two streams are about 60 km apart; thus, an additional spawning aggregation is needed, and is most likely achievable in the Dewatto River with additional reintroduction and habitat actions (Lestelle et al. 2018). Habitat restoration has been completed in both Big Beef Creek and Tahuya River as well, and reintroductions of supplementation-origin stock from the Union River are proposed by co-managers (Lestelle et al. 2018). The Lilliwaup Creek spawning aggregate was supplemented through 2017, and supplementation-origin fish were still returning in 2019. There is still considerable uncertainty regarding the historical or current presence of a spawning aggregation in the Dungeness River (PNPTT and WDFW 2014); however, as mentioned above, the co-managers believe that establishing an additional spawning aggregate in the Dungeness River would substantially improve the risk of extinction for the Strait of Juan de Fuca population (Lestelle et al. 2018).

The co-managers' assessment using VRAP was also used for each of the eight extant spawning aggregations to estimate habitat goals. Results led to the recommendation that habitat restoration and protection actions 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 (PNPTT and WDFW 2014). An in-depth discussion of the rationale is presented in both the co-manager status review (PNPTT and WDFW 2014) and in the guidance document (Lestelle et al. 2018).

The Hood Canal Coordinating Council (HCCC) prepared the recovery plan for Strait of Juan de Fuca and Hood Canal 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 salmon 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-run chum salmon ESU. They are currently working with NOAA on this revision.

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 had increased at the time of the last review (NWFSC 2015), but has been down for the last three years for the Hood Canal population, and for the last four years for the Strait of Juan de Fuca population. Productivity of individual spawning aggregates shows that only two of eight aggregates have viable performance. Spatial structure and diversity viability parameters, as originally determined by the TRT, have improved, and nearly meet the viability criteria for both populations. Despite substantive gains toward meeting viability criteria in the Strait of Juan de Fuca and Hood Canal summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time. Overall, the Hood Canal summer-run chum salmon ESU therefore remains at "moderate" risk of extinction, with viability largely unchanged from the prior review.

Ozette Lake Sockeye Salmon ESU

Brief description of ESU

The ESU includes all naturally spawned aggregations of sockeye salmon in Ozette Lake and streams and tributaries flowing into Ozette Lake, Washington (Figure 107; USOFR 2020). The ESU also includes fish originating from one artificial propagation program: the Umbrella Creek/Big River sockeye salmon hatchery program. The Puget Sound Technical Recovery Team (PSTRT) considers the Ozette Lake sockeye salmon ESU to comprise 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.

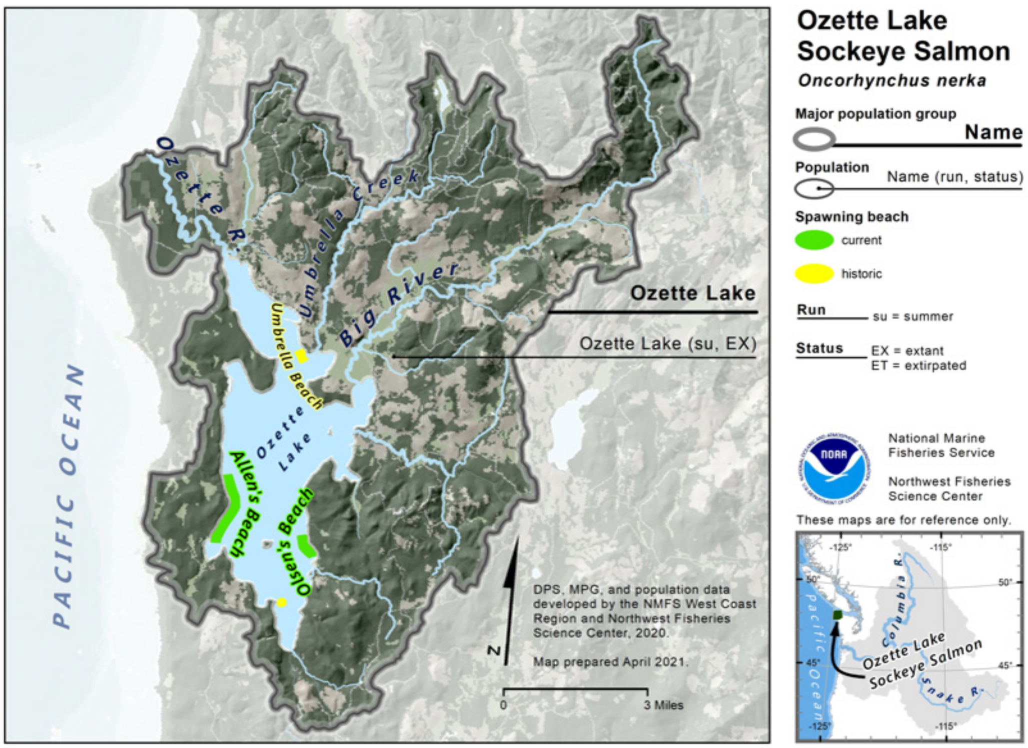

Figure 107. Map of the Ozette Lake sockeye salmon ESU's spawning and rearing areas.

Summary of previous status conclusions

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 were very uncertain, hampering efforts to assess tends and status in the VSP criteria of abundance, spatial structure, and diversity. They recommended that the threatened status remain unchanged.

Ford et al. (2011) concluded that estimates of population abundance for Ozette Lake sockeye salmon 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 when harvest on these stocks was plentiful. The review noted that assessment methods needed to improve in order to evaluate the status of this population/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 previous BRT status review in 2005.

In the 2015 status review update, NWFSC (2015) found that little had changed since the previous update. Abundance estimates were still very low relative to the abundance recovery goal, and insufficient information was available to effectively assess the status of the beach-spawning component of the population.

Description of new data available for this review

Run size estimates based on expanded weir counts have been extended from 2013 to 2019 (M. J. Haggerty, Mike Haggerty Consulting, personal communication). In addition, some of the previous data were updated to improve consistency across the data series. In 2012, no estimate was generated due to a hard drive failure. For 2013 through 2016, estimates were based on a river-spanning weir with a video chamber (Haggerty 2014), while estimates for 2017–19 were based on an ARIS imaging sonar operated in conjunction with a partial weir (Denton 2018, MFM 2020; Haggerty, personal communication). For both methods, portions of the run were not enumerated due to in-river conditions and technical problems. Estimates were expanded to account for these missing days using estimated average run timing from years with more complete data (Haggerty 2014). In addition, a detection rate estimate was applied to account for fish missed during review of the imagery. Data from the Umbrella Creek hatchery program were also updated to 2019 (Hinton and Cooke 2019, MFM 2020). This included estimates of total returns, hatchery fraction, age composition, and the broodstock take. Beach-spawner survey data—including live and dead counts, survey date ranges, and the number of surveys—were also updated (Hinton and Cooke 2019, MFM 2020).

A spawning abundance series from 1977 to 2019 was constructed from a number of different sources (Figure 108, Table 61). As a whole, the source data were very uncertain, with modest improvements in the last decade. From 1977 to 1995, the median expansion estimate from 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 Ozette Lake sockeye hatchery and genetic management plan (MFM 2000), when available. In 1988, counting was conducted for only three 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 2003, values from Haggerty et al. (2009), Table 3.6, were used. From 2004 to 2011, estimates were available from Haggerty (2014). Finally, from 2013 through 2019, data were provided by M. J. Haggerty (personal communication). In 2004, the run size estimate was roughly half of the estimated return to Umbrella Creek. In addition, it was noted that technical issues with the video weir likely led to an underestimate. We therefore treated the estimate as missing, along with 2012. Both years were then filled in using a regression between the Umbrella Creek and Ozette Lake run sizes, where both were available. Because the percentage of natural-origin fish could not be reliably estimated from 1987 to 1999, we used naturally spawning fish to describe abundance except where noted.

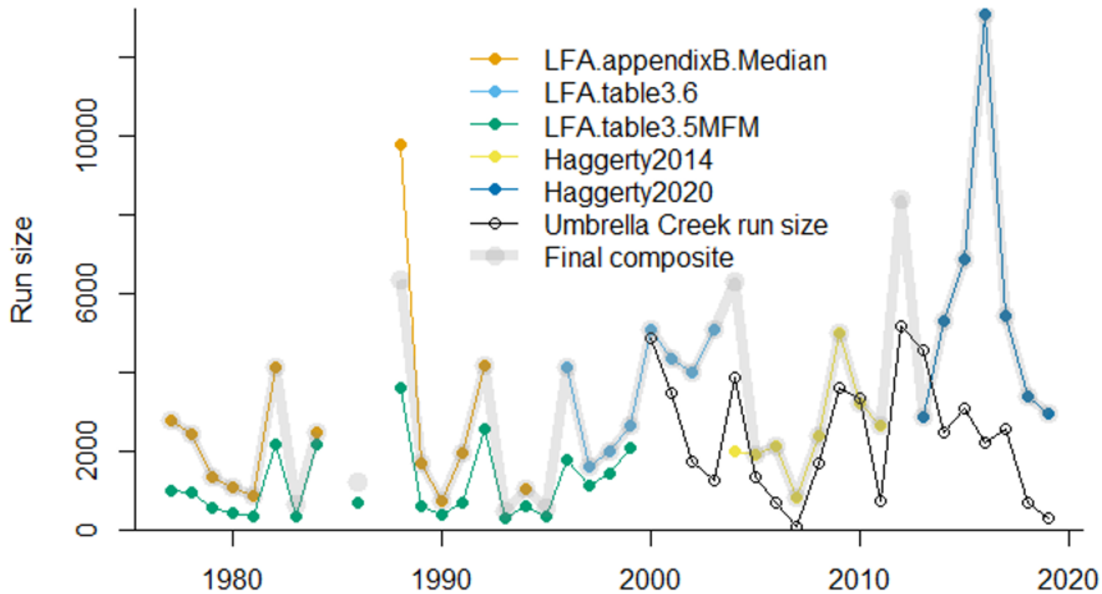

Figure 108. The different sources of data used to construct the natural spawners estimate for Ozette Lake. The thick, light gray line represents the final composite estimate. *LFA* refers to the limiting factors analysis (Haggerty et al. 2009), *Haggerty2014* refers to Haggerty (2014), and *Haggerty2020* refers to data provided by M. J. Haggerty (personal communication).

Table 61. Ozette Lake and Umbrella Creek sockeye salmon abundance.

Year	Spawners	Broodstock	Natural spawners	Fraction natural	Umbrella spawners	Umbrella hatchery-origin
1977	2,752	0	2,752	1.000		
1978	2,398	0	2,398	1.000		
1979	1,335	0	1,335	1.000		
1980	1,054	0	1,054	1.000		
1981	858	0	858	1.000		
1982	4,131	0	4,131	1.000		
1983	617	14	603	1.000		
1984	2,474	27	2,447	1.000		
1985		40		1.000		
1986	1,217	43	1,174	1.000		
1987		123		1.000		
1988	6,341	193	6,148			
1989	1,677	6	1,671			
1990	732	33	699			
1991	1,955	175	1,780			
1992	4,167	109	4,058			
1993	470	32	438			
1994	1,018	54	964			
1995	553	94	459			
1996	4,131	200	3,931			
1997	1,609	263	1,346			
1998	1,970	88	1,882			

Table 61 (continued). Ozette Lake and Umbrella Creek sockeye salmon abundance.

Year	Spawners	Broodstock	Natural spawners	Fraction natural	Umbrella spawners	Umbrella hatchery-origin
1999	2,649	29	2,620			
2000	5,064	213	4,851	0.676	4,842	1,640
2001	4,315	238	4,077	0.971	3,458	124
2002	3,990	170	3,820	0.934	1,718	262
2003	5,075	199	4,876	0.970	1,256	153
2004	6,298	218	6,080	0.938	3,875	389
2005	1,908	187	1,721	0.900	1,321	190
2006	2,135	60	2,075	0.935	686	140
2007	793	45	748	0.991	49	7
2008	2,389	238	2,151	0.906	1,664	225
2009	4,988	219	4,769	0.885	3,611	574
2010	3,220	234	2,986	0.916	3,327	270
2011	2,625	168	2,457	0.910	740	237
2012	8,373	167	8,206	0.678	5,152	2,698
2013	2,859	209	2,650	0.803	4,550	564
2014	5,282	185	5,097	0.987	2,478	67
2015	6,846	208	6,638	0.947	3,053	362
2016	13,073	244	12,829	0.994	2,208	82
2017	5,427	162	5,265	0.978	2,537	118
2018	3,375	111	3,264	0.883	688	396
2019	2,922	28	2,894		304	

Natural spawners were calculated by subtracting the effective catch from the total run size (Figure 109). The effective catch is the number of fish that were removed from the natural spawning population due to harvest (1977–82) or broodstock take (1983–present). Until 2000, all broodstock was taken from beaches (Haggerty et al. 2009, Appendix C). From 2000 on, the broodstock was taken from Umbrella Creek (MFM 2015, Hinton and Cooke 2016, 2019, MFM 2020).

All natural spawners were of natural origin from the first hatchery release in 1983 until 1987. Between 1988 and 1999 there were no reliable data on percent hatchery-origin (Haggerty et al. 2009). Percent natural-origin for this period was therefore designated as unknown. For 2000 to 2019, data were available in MFM (2015) and the Ozette Lake sockeye salmon resource management plans (Hinton and Cooke 2016, 2019, MFM 2020). We calculate percent natural origin as 1 minus the Umbrella Creek run size ($N_{Umb,y}$) times the Umbrella Creek hatchery fraction ($pHOS_{Umb,y}$) divided by total run size entering the lake (N_{tot}):

$$pNOS_{tot,y} = 1 - \frac{N_{Umb,y} \times pHOS_{Umb,y}}{N_{tot,y}}$$

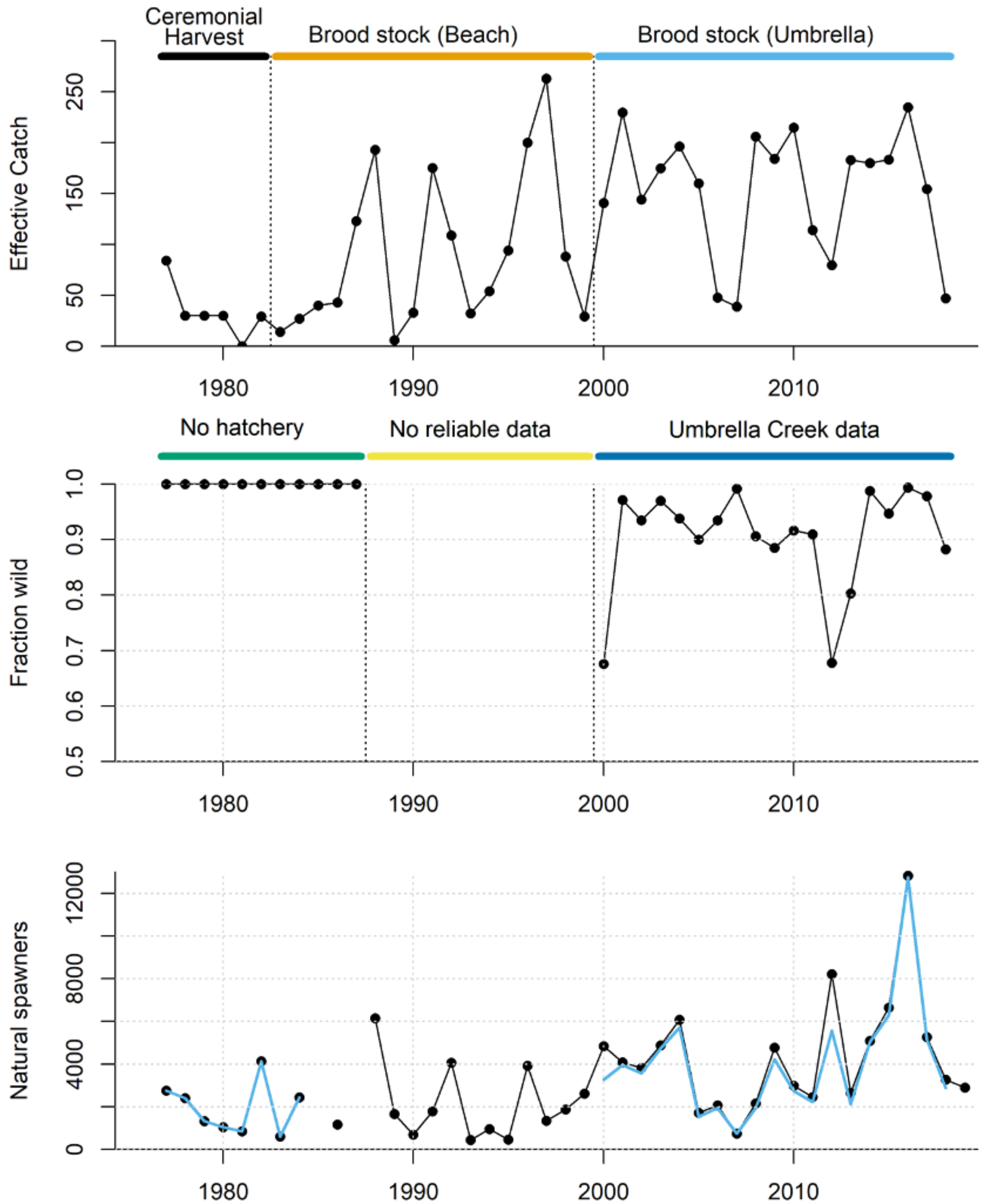

Figure 109. Top: Effective catch over time. Middle: Fraction of natural spawners that are natural-origin. Bottom: Black line = natural spawners, blue line = natural-origin natural spawners.

There are a number of issues with this approach, including: a) that we know there are additional hatchery fish in other tributaries, b) that mortality between entry to the lake and entry to Umbrella Creek is not accounted for, and c) that for many years, estimates are very uncertain for Umbrella Creek and for fish entering the lake. Both a) and b) will tend to produce estimates that are biased high.

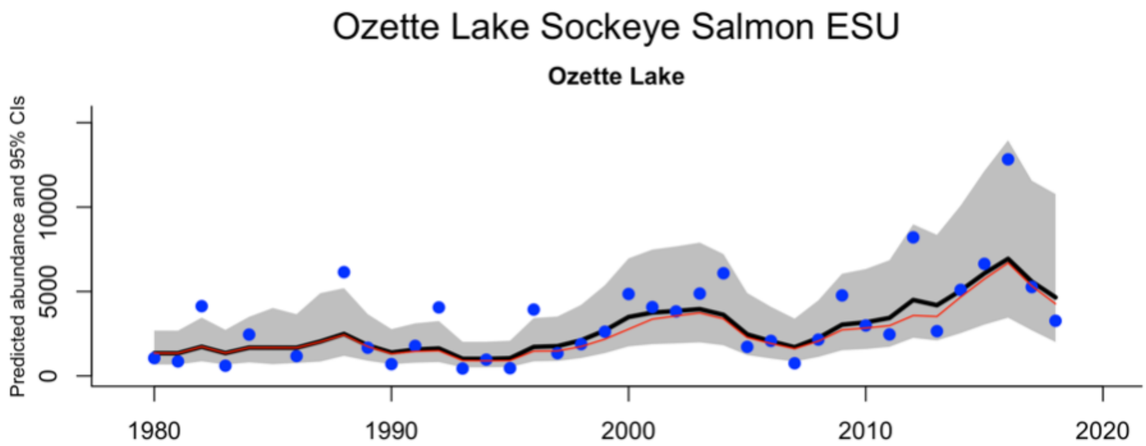

Figure 110. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

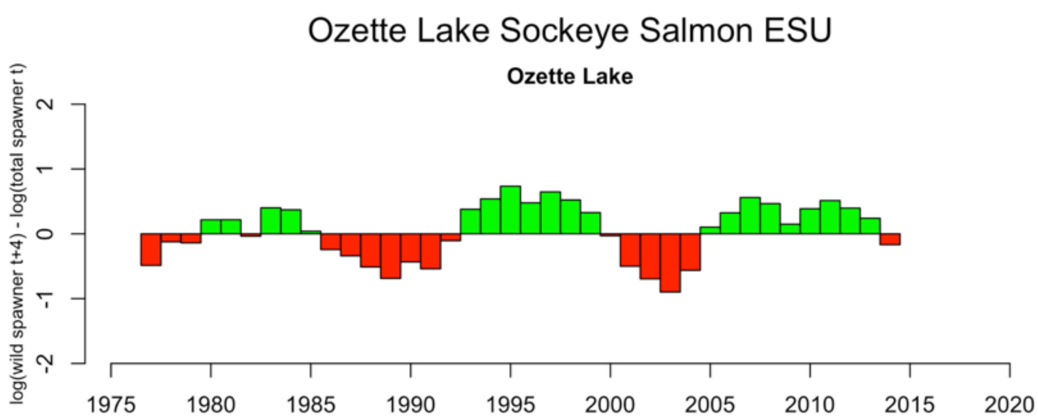

Figure 111. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year t minus the smoothed natural spawning abundance in year $(t-4)$.

Abundance and productivity

For the period from 1977 to 2019, the estimated natural spawners ranged from 438 to 12,829, well below the 31,250–121,000 viable population range set in the Ozette Lake sockeye salmon recovery plan (Figure 109; NMFS 2009b). There remains little evidence of a strong trend in the raw (Figure 109) or smoothed (Figure 110) abundance series—over the full range of years, or more recently—since the last status review (NWFSC 2015). However, the geometric mean of abundance from 2015 to 2019 was higher than the previous five-year geometric mean (Table 62), and the trend over the last 15 years has been positive (Table 63). There is some evidence of the dominant four-year age of return in the abundance series (Figure 112), with the 1980 brood cycle line surpassing the other lines in the late 80s and maintaining this higher level for most four-year cycles since. Estimated productivity, calculated as the abundance in year t divided by the abundance in year $(t-4)$, has shifted between negative and positive values with a suggestive 10–20-year cycle in both the raw (Figure 112) and smoothed

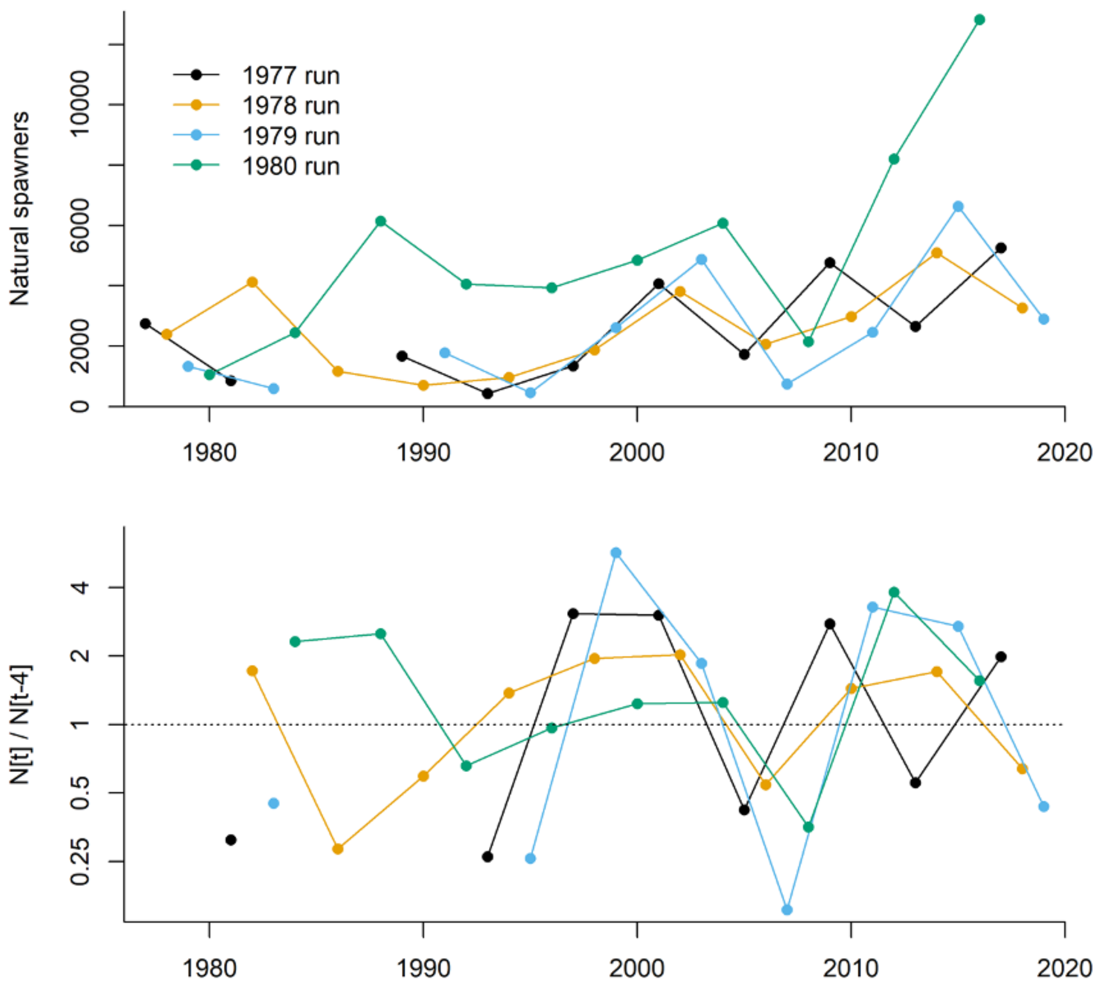

Figure 112. Top: Natural spawners vs. year, with lines connecting 4-year brood cycle lines—e.g., the 1977 brood cycle line includes the years 1977, 1981, 1985, etc. Bottom: Productivity vs. year, where productivity is calculated as natural spawners in year *t* divided by the natural spawners in year ($t-4$). Notice the *y*-axis is on the log scale in the lower panel.

(Figure 111) data. While estimated productivity has recently been positive, we may be entering a negative phase (Figure 111). 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 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 percent hatchery-origin was not estimated until 2000. From 2000–18, the estimates ranged from 3–58%, with a mean of 18%. To date, correcting for percent hatchery-origin has not qualitatively changed the trends in abundance (Figure 109). However, because the Umbrella Creek population is a large component of the total population (averaging 54% over the last decade), 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 61). Therefore, precise estimates of natural-origin spawners depend on good estimates of percent hatchery-origin fish.

Table 62. Five-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. 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 2 most-recent 5-year periods is shown on the far right.

Population	MPG	1990–94	1995–99	2000–04	2005–09	2010–14	2015–19	% change
Ozette Lake	Ozette Lake	(1,163)	(1,643)	4,162 (4,678)	1,789 (1,939)	3,253 (3,820)	5,873 (6,185)	81 (62)

Table 63. Fifteen-year trends (slope) in log total spawner abundance computed from a linear regression applied to the smoothed total spawner log abundance estimate versus year. In parentheses are the upper and lower 95% CIs.

Population	MPG	1990–2005	2004–2019
Ozette Lake	Ozette Lake	0.09 (0.05, 0.12)	0.08 (0.04, 0.11)

Ocean fisheries do not significantly impact Ozette Lake sockeye salmon. Both Ozette Lake and the Ozette River, which connects the lake with the ocean, are closed to salmon fishing (with the exception of a winter steelhead fishery in the Ozette River). The steelhead fishery ends in February, and sockeye salmon do not begin entering the lake until April.

Spatial structure and diversity

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, Kemmerich (1939) noted that from 1923–26, "…most of the spawning seemed to be along the lake shore in suitable places and especially at the mouths of the several creeks" (p. 1). Beach spawning at stream mouths was last observed on Umbrella Beach in broodyear 1978 (Dlugokenski et al. 1981). Contemporary beach surveys are concentrated primarily on Allen's and Olsen's Beaches, but less frequent, more extensive surveys have not observed beach spawning elsewhere for over a decade. Spawning on the upper beach (in shallower water) has also declined, likely resulting from increased shoreline vegetation. Ritchie and Bourgeois (2009) used aerial photography to determine that more than half of the open beach area in 1953 was vegetated by 2003. In the two remaining areas where beach spawning is currently observed, Olsen's and Allen's Beaches, over two-thirds of the open beach was covered with vegetation during the same period. They hypothesize that the new vegetation is caused by increased fine sediment deposition resulting from forest harvest in the tributary watersheds, and changes in the timing of water-level fluctuations.

Estimates of total beach spawner abundance are not available. However, rough minimum estimates have been constructed (Figure 113) using early broodstock collection efforts, some sporadic intentional surveys, and, more recently, methodical surveys using visual and imaging sonar based counts (Haggerty et al. 2009, Haggerty 2013, 2014, Hinton and Cooke 2016, 2019, MFM 2020). Estimating beach spawners as the difference between fish entering the lake and fish entering Umbrella Creek is problematic due to substantial uncertainty in both estimates, mortality between entry into the lake and Umbrella Creek, and additional tributary spawners that are not accounted for (Figure 108). While the available data for beach spawners do not allow for estimates of total abundance, there is, for example, strong evidence that there were very few beach spawners in 2005–10 (Figure 113).

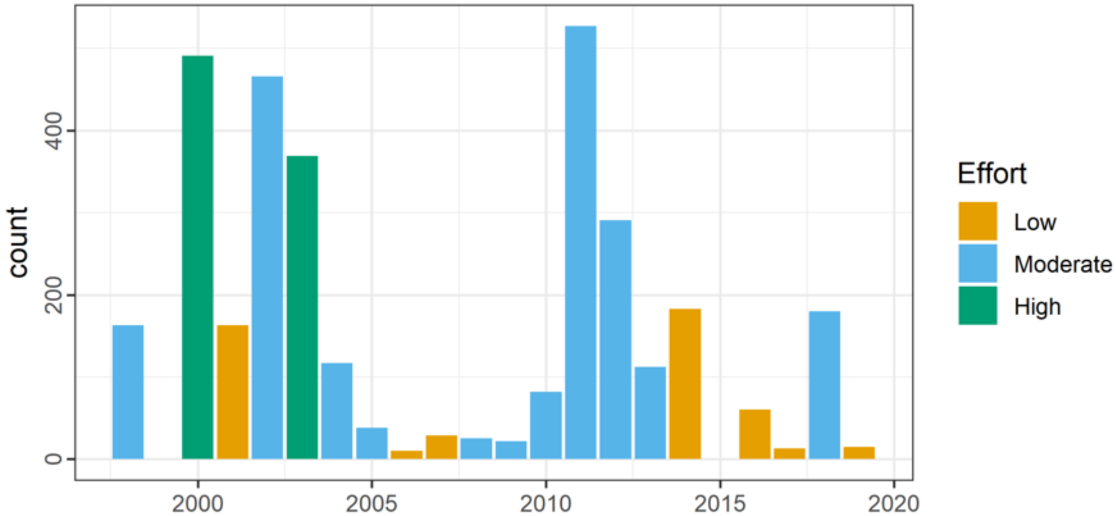

Figure 113. Number of live and dead fish observed in beach counts. These are not estimates of total beach spawners. Points are colored by approximate effort (i.e., number and quality of surveys).

Extensive spawner surveys in the 70s, prior to the hatchery program, found no spawning in the tributaries (Haggerty et al. 2009). 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 that tributary spawning was not significant (Haggerty et al. 2009). After initiation of a hatchery program in the 1980s, spawning aggregations in Umbrella Creek and Big River increased in size through the 1990s. The run size for Umbrella Creek from 2000–11 averaged 48% of the Ozette Lake total run size, and, for some years, made up well over 90% (Figure 108, Table 61).

Historic and current run timing are not well characterized for the beach spawning aggregates. However, beach spawning survey data from 1978 (Dlugokenski et al. 1981) suggested a more extended run timing than is currently observed. Spawners were observed on Allen's Beach in late February, while surveys in the last decade have not extended past January (latest survey date since 2000 was 15 January). However, this earlier end date may be due at least in part to limited effort in recent years. During the 1978 season, peak run timing at Olsen's Beach occurred a month earlier than at Allen's Beach. This difference has not been observed in more recent surveys.

A number of studies have suggested varying degrees of genetic differentiation between spawning locations and between the four cohort lineages (Hershberger et al. 1982, Gustafson et al. 1997, Crewson et al. 2001, Hawkins 2004). In samples from both 1995 (Gustafson et al. 1997) and 2000 (Crewson et al. 2001), there was some evidence for genetic differences between spawners on Olsen's and Allen's Beaches. The two beaches are on opposite sides of the south end of the lake, separated by more than 3 km. Samples from the Umbrella Creek aggregate were closer to the Olsen's Beach spawners, where most of the broodstock was taken (Crewson et al. 2001). There were also observed differences between the four cohort lineages (Gustafson et al. 1997, Crewson et al. 2001, Hawkins 2004), suggesting that the dominant four-year cycle has maintained some genetic differentiation. Two analyses that compared kokanee and sockeye salmon found large genetic differences (Crewson et al. 2001, Hawkins 2004). For a summary of these genetic studies, see Haggerty et al. (2009).

The estimated fraction of hatchery-origin fish returning to Ozette Lake has averaged only 6% in recent years (2000–18). However, the large contribution of the hatchery-supplemented tributary aggregations to the population as a whole allows for larger total hatchery fractions when Umbrella Creek hatchery fraction is high. For example, in 2012, over half (52%) of the estimated 5,152 fish returning to Umbrella Creek were designated as hatchery-origin.

Biological status relative to recovery goals

The proposed criteria for the VSP parameters set in the recovery plan for Ozette Lake sockeye salmon (NMFS 2009b) are:

- *Abundance*: "…should range in abundance between 31,250 and 121,000 adult spawners, over a number of years" (p. 25).
- *Productivity*: "…the population growth rate would have to be stable or increasing" (p. 25).
- *Spatial structure*: "…have multiple spawning aggregations along the lake beaches, which are the known historical spawning areas. The certainty that the population achieves a viable condition would be further increased if spawning aggregations in one or more tributaries to the lake were also established" (p. 25).
- *Diversity*: "…one or more persistent spawning aggregations from each major genetic and life history group historically present within that population. A viable population of sockeye in Lake Ozette also would maintain the historical genetic diversity and distinctness between anadromous sockeye salmon and kokanee salmon in Lake Ozette" (p. 25).

As there is only one population within this ESU, all of the above criteria must be met for that population for the ESU to be considered viable; see, for example, McElhany et al. 2000: "Some populations should exceed VSP guidelines…. This guideline is particularly relevant if an ESU consists of a single population" (p. 126), and McElhany et al. 2000: "A population must meet all of the viable population guidelines to be considered viable with respect to [population size]" (p. 13). Because, in this case, the ESU is a single population, all viable population guidelines must be met for the ESU to be considered viable.

Current status of abundance and productivity

There are sufficient data to determine that the total Ozette Lake abundance is well below the desired lower bound, although the population has increased since the last review and over the past 15 years (Tables 62 and 63). Over the last few decades, productivity for the total Ozette Lake population has exhibited a 10–20-year cyclical pattern alternating between negative and positive values. Average rates over the last five- and 15-year periods have been slightly positive, although we may be entering a negative phase.

Current status of spatial structure and diversity

Defining a historical baseline and assessing the current state of the spatial structure and diversity of the population is difficult due to a paucity of data. In particular, without estimates of abundance for the beach spawning aggregates, it is difficult to assess the degree to which the existing spatial structure is robust to demographic variability. This is especially important since both the abundance and distribution of the beach spawners has declined to a small percentage of historical levels. While abundance estimates for beach spawners are not available, there is relatively strong evidence for a substantial decline during the mid-to-late 2000s, when very few spawners were observed with moderate levels of survey effort (Figure 113). There is also some indication that run timing may have changed since the 1970s.

Currently, 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 while maintaining a genetic reservoir initially established with beach-spawning fish. The addition of the tributary aggregate may have increased or stabilized overall abundance, although this is not yet confirmed by the abundance trends.

Straying of tributary fish into the beach spawning locations may pose a threat to the beach-spawning aggregate, since the tributary-spawning aggregate appears to be much larger than the beach-spawning aggregate for some years. 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 0.5–0.8% (MFM 2015). In addition, 1) there is some evidence that tributary and beach spawning aggregates coexisted in the past, 2) the source of the hatchery program was Ozette Lake fish, 3) the hatchery broodstock is currently naturally spawning tributary fish, 4) there is little evidence of resource limitations in the lake for rearing, and 5) the level of 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. In recent years, very few carcasses have been recovered, making it difficult to detect any potential changes in hatchery fraction (only one carcass recovered from 2015–18; Hinton and Cooke 2019, MFM 2020).

Assessment of the Ozette Lake sockeye salmon population status is substantially hampered by gaps in our knowledge of population abundance and structure. In particular, because the beach-spawning aggregate is considered the core group of interest for recovery (NMFS 2009b) and its abundance has not been estimated, it is difficult to fully assess the population status.

Recommendations:

- Develop and implement a method for enumerating the beach-spawning aggregate. This is a difficult task, but is essential for evaluating the status of the population.
- In order to characterize any spatial or temporal changes in beach spawning, include occasional surveys that are more spatially extensive and capture the beginning and end of the spawn-timing distribution.
- Improve estimates of total population size. The current method of enumeration, imaging sonar, was only recently adopted. Further experimentation with placement and additional resources devoted to review of imagery and analysis will be necessary to ensure reliable estimates.

- 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.
- 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.
- Improve understanding of genetic structure within the population through additional genetic analysis.
- Resurvey beach spawning habitat to assess status and trends in availability.
- If beach habitat restoration is planned, consider increased spawner and habitat surveys at those sites before and after to allow for monitoring of restoration effectiveness.

Updated biological risk summary

Based on an evolving understanding of both the status and the uncertainty in the status of the Ozette Lake sockeye salmon beach-spawning aggregates, we believe the biological risk for Ozette Lake sockeye salmon has increased somewhat compared to prior reviews, largely due to a clearer understanding of the poor condition of the beach-spawning aggregates. Extinction risk is determined by our best prediction of the demographic probability of extinction and the uncertainty in that prediction. More uncertainty will result in a higher risk. In the case of Ozette Lake sockeye salmon, this uncertainty contributes substantially to our evaluation of extinction risk. Stated otherwise, due to substantial uncertainty in the historic and current abundance and structure of the population, it is not possible to rule out further decline in the VSP parameters over the next couple decades, which would increase overall risk. Our perception of risk appears to be higher because:

1. For the last four decades, the abundance of Ozette Lake sockeye salmon natural adult spawners ranged from 438–12,829, well below the lower viability threshold of 31,250–121,000 established in the 2009 NMFS TRT report and the Ozette Lake recovery plan.
2. Over the last few decades, estimated productivity for the total Ozette Lake population has alternated between positive and negative periods. While estimated average productivity over the most recent five- and 15-year periods has been positive, we may now be entering another negative phase. This cyclical pattern in productivity makes it difficult to interpret historical and to predict future trends in productivity, and may increase risk due to the potential for sustained periods of negative productivity.
3. There is accumulating evidence of a sustained reduction in abundance and distribution of beach spawners, aggravating the conditions originally identified by the PSTRT that "…the limited distribution of Ozette Lake sockeye spawners [at that time] put the ESU at high risk" (pp. 3–4).
4. Critical gaps in our knowledge of the beach-spawning aggregates prevent any quantitative assessment of abundance or trends of this portion of the ESU that are considered critical for recovery.
5. The Ozette Lake sockeye salmon ESU consists of a single population, which by itself increases risk of extinction because of limited demographic diversity and redundancy.

Overall, the Ozette Lake sockeye salmon ESU therefore has mixed viability trends, and is likely at "moderate-to-high" risk of extinction.

Oregon Coast Coho Salmon ESU

Brief description of ESU

The Oregon Coast coho salmon ESU consists of coho salmon populations on the Oregon coast from Cape Blanco to the mouth of the Columbia River (Figure 114; USOFR 2020). 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 winter lake rearing by juvenile coho salmon in several large lake systems.

Figure 114. Map of the Oregon Coast coho salmon spawning and rearing areas, illustrating populations and major population groups.

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 been 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, Umpqua, and Mid-South Coast) identified by ODFW for Oregon Coast coho salmon, in addition to the lakes complex confirmed 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 three (Lakes) and 29 (Mid Coast) populations (Figure 114).

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 2001–03, 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 salmon 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)—threats 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.

2015

The Oregon Coast coho salmon ESU was included in the five-year assessment of listed salmonid DPSes in 2015 (NWFSC 2015). The 2015 update included updates of existing data time series through return year 2014. These included spawner abundances, exploitation rates, estimates of the proportion of natural spawners, and marine survival. It also included an updated assessment of the Decision Support System (DSS; see next section). The 2015 update described many positive trends in population abundance and escapement, due in large part to the largest returns of natural coho to the ESU in 2011 and 2014 (both over 338,000 fish) since the 1970s, paired with low harvest rates. The DSS scores for population, stratum, and ESU persistence and sustainability also generally improved. The ESU persistence score (0.73) indicated high certainty that the ESU would persist, or not go extinct over a 100-year period, including the ability to survive prolonged periods of adverse environmental conditions. The ESU sustainability score (0.23) indicated low-to-moderate certainty that the ESU was sustainable, or that it could maintain its genetic legacy and long-term adaptive potential for the foreseeable future. Both scores were higher than scores generated by previous runs of the DSS. However, the 2015 update cautioned that the positive trends were unlikely to continue into the near future, starting with juveniles entering the ocean in 2014, due to the formation of "the Blob". It further advised waiting to see how populations fared during the expected downturn before reconsidering the status of Oregon Coast coho salmon.

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, 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.

Here, we provide scores from three evaluations of the DSS (Table 64) 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 first DSS assessment we provide here was conducted as part of the 2012 BRT evaluation, which included data through the 2009 return year (Stout et al. 2012). The second DSS run was conducted by M. Lewis (ODFW), and used data through the 2014 return year (Lewis 2015). The third assessment was also conducted by Lewis, using data through the 2019 return year (Lewis 2020). Scores provided here for the 2012 evaluation were calculated by Lewis and differ slightly from those found in Stout et al. (2012) due to changes in GIS coverage (which changes fifth-field watershed boundaries), and other issues with the 2012 assessment identified in Lewis (2015). These changes allow direct comparison of the three 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; these models have not been rerun. Part of the rationale for not updating this parameter is that the relative vulnerabilities of populations assessed by the population viability assessment (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 analyses were conducted for either the 2015 or this review. Third, the ESU-level criteria for diversity (ED-1, ED-2, and ED-3) have also not been updated since the DSS was originally evaluated because they relied on professional judgment (Wainwright et al. 2008). Increases in abundance and productivity across all strata observed in the previous assessment suggest ESU diversity has not decreased. Accordingly, the DSS results provided here for all three assessments reflect the original values for PP-2 and the ESU-level diversity criteria, but PF-1 (i.e., habitat quality) is no longer included in calculations of the whole ESU sustainability and persistence scores (Lewis 2015, 2020).

Table 64. Population scores for Oregon Coast coho salmon DSS criteria for assessments conducted in 2012 (using data through 2009 return year), 2015 (data through 2014 return year), and 2020 (data through 2019 return year). Criteria: 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 distribution (>4 fish/mi in half of watersheds), PD-4 = juvenile distribution (pools with >1 fish). See Wainwright et al. (2008, 2014) for additional details. Also included are the Population Persistence (PP), Population Sustainability (PS), and Stratum Sustainability (SS) scores, and the "minimum level of desired status" of each population under the Oregon Coast coho salmon conservation plan (P = pass, F = fail).

		PP-1			PP-2			PD-1			PD-2			PD-3			PD-4			PP			PS			SS			CP
Stratum	**Population**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**2012**	**2015**	**2020**	**status***
North Coast	Necanicum	0.95	0.89	0.63	0.30	0.68	0.45	0.01	0.02	0.02	0.35	0.92	0.89	0.82	0.92	0.80	0.97	0.71	0.36	-0.24	-0.21	-0.26	-0.14	-0.10	-0.16				F
North Coast	Nehalem	0.80	0.99	1.00	0.81	0.83	0.54	0.83	0.87	0.89	0.66	0.79	0.90	0.45	0.53	0.27	0.51	0.78	0.77	0.84	0.91	0.76	0.67	0.78	0.63				P
North Coast	Tillamook Bay	0.90	0.95	0.95	0.42	0.76	0.66	0.12	0.14	0.15	0.42	0.79	0.94	0.23	0.61	0.47	0.64	0.85	0.99	0.55	0.68	0.65	0.35	0.50	0.51				P
North Coast	Nestucca	0.82	0.95	0.92	0.38	0.43	0.41	0.14	0.16	0.16	0.92	0.86	1.00	0.20	0.52	0.49	0.92	0.50	0.13	0.57	0.63	0.61	0.44	0.45	0.41				P
																										0.39	0.47	0.46	
Mid Coast	Salmon	–0.51	–0.81	–0.83	–0.94	–0.71	0.43	–1.00	–1.00	–1.00	–1.00	0.92	0.72	0.64	0.57	0.35	1.00	1.00	1.00	-1.00	-1.00	-1.00	-1.00	-1.00	-1.00				F
Mid Coast	Siletz	0.91	1.00	0.83	0.11	0.86	0.84	0.08	0.10	0.11	0.67	0.93	0.99	0.51	0.90	0.97	0.93	1.00	1.00	0.36	0.81	0.76	0.35	0.58	0.58				P
Mid Coast	Yaquina	0.97	0.89	0.97	0.44	0.93	0.92	0.30	0.33	0.35	0.69	0.93	0.94	0.84	0.95	0.93	1.00	1.00	1.00	0.65	0.85	0.87	0.60	0.73	0.74				P
Mid Coast	Beaver	0.97	0.99	–0.10	0.93	1.00	1.00	0.03	0.04	0.04	0.86	1.00	0.99	1.00	1.00	1.00	1.00	1.00	1.00	0.80	0.82	0.17	0.53	0.56	0.24				P
Mid Coast	Alsea	0.63	0.86	0.59	0.02	0.68	0.84	0.18	0.20	0.22	0.97	0.98	1.00	0.45	0.85	0.92	0.83	1.00	1.00	0.28	0.81	0.76	0.33	0.64	0.64				P
Mid Coast	Siuslaw	0.89	0.77	0.52	0.07	0.81	0.82	0.98	1.00	1.00	0.81	0.91	1.00	0.53	0.72	0.79	0.68	0.89	0.98	0.38	0.85	0.71	0.49	0.85	0.80				P
																										0.42	0.61	0.61	
Lakes	Siltcoos Lake	0.81	0.88	0.09	1.00	1.00	1.00	0.45	0.49	0.452	0.99	0.99	0.93	1.00	1.00	1.00	1.00	1.00	1.00	0.92	0.95	0.42	0.83	0.85	0.53				P
Lakes	Tahkenitch Lake	0.69	0.84	0.58	1.00	1.00	1.00	0.24	0.26	0.242	0.95	0.99	0.96	1.00	1.00	1.00	1.00	1.00	1.00	0.78	0.82	0.72	0.66	0.70	0.64				P
Lakes	Tenmile Lake	0.96	0.78	0.84	1.00	1.00	1.00	0.91	0.97	0.811	0.98	0.98	1.00	1.00	1.00	1.00	–0.36	–0.23	0.63	0.98	0.90	0.93	0.20	0.34	0.87				P
																										0.66	0.70	0.64	
Umpqua	Lower Umpqua	0.68	0.76	0.89	0.73	0.84	0.81	0.80	0.85	0.88	0.42	0.93	0.99	0.78	0.85	0.77	0.61	0.85	0.96	0.74	0.81	0.85	0.65	0.84	0.87				P
Umpqua	Middle Umpqua	0.73	0.66	0.74	0.22	0.48	0.19	0.26	0.28	0.29	0.35	0.99	1.00	0.25	0.39	0.09	0.22	0.66	0.91	0.45	0.61	0.43	0.31	0.53	0.38				P
Umpqua	North Umpqua	–0.96	–0.50	0.38	0.50	0.89	0.86	–0.69	–0.64	–0.59	–0.96	0.13	0.34	–0.52	–0.42	–0.48	–0.66	–0.64	–0.50	-0.95	-0.30	0.52	-0.95	-0.57	-0.41				F
Umpqua	South Umpqua	0.92	0.61	0.07	0.64	0.82	0.29	0.21	0.24	0.24	0.50	0.53	–0.04	0.06	0.28	0.06	0.14	0.35	0.64	0.80	0.75	0.26	0.33	0.45	0.14				P
																										0.32	0.49	0.26	
Mid-South Coast	Coos	0.92	0.91	0.76	0.58	0.91	0.80	1.00	1.00	1.00	0.94	0.97	0.98	0.73	0.88	0.64	0.85	0.70	0.89	0.75	0.89	0.80	0.80	0.87	0.82				P
Mid-South Coast	Coquille	0.96	0.92	0.67	0.84	0.91	0.80	1.00	1.00	1.00	0.98	0.96	1.00	0.68	0.78	0.62	0.80	0.93	0.80	0.91	0.93	0.79	0.87	0.91	0.80				P
Mid-South Coast	Floras	0.99	0.88	0.88	–0.46	0.14	0.37	0.12	0.15	0.14	0.81	1.00	1.00	0.21	0.74	0.73	1.00	1.00	1.00	-0.21	0.43	0.61	-0.10	0.45	0.52				P
Mid-South Coast	Sixes	0.52	0.76	0.82	–0.25	–0.35	–0.31	–0.96	–0.96	–0.95	0.17	0.74	1.00	–0.66	–0.42	–0.66	–0.42	0.17	0.27	-1.00	-1.00	-1.00	-1.00	-1.00	-1.00				F
																										0.35	0.66	0.66	
	Mean score:	0.69	0.71	0.58	0.40	0.66	0.65	0.24	0.26	0.26	0.55	0.87	0.88	0.49	0.65	0.56	0.60	0.69	0.75	0.35	0.52	0.45	0.25	0.40	0.36				

*CP = Oregon Coast coho salmon conservation plan. Populations must have positive PS scores to pass the "minimum level of desired status."

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 2019. These include spawner abundances, exploitation rates, estimates of the proportion of natural spawners, marine survival, and an updated assessment of the DSS through return year 2019. New to this assessment are natural spawner data for dependent populations in the North Coast and Mid Coast strata, which were not included in previous assessments.

The 2015 update provided new marine survival estimates for natural Oregon Coast coho salmon from the life cycle monitoring (LCM) sites (Suring et al. 2012, 2015). These rates are estimated from the number of smolts passing downstream through smolt traps and subsequent numbers of jacks and adults returning 0.5–1.5 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).

The 2015 update used marine survival data from six LCM sites: Nehalem (North Coast), Siletz, Yaquina, and Alsea (Mid Coast), Umpqua (Umpqua), and Coos (Mid-South Coast). Since the 2015 update, the Nehalem LCM site has been terminated. However, Suring (2017) demonstrated that coastwide marine survival estimates that rely on only five sites were highly correlated to the full six-site survival index ($r^2 = 0.99$) without bias. Accordingly, the LCM marine survival time series provided here (Figure 115) is based on the five LCM sites (Suring, unpublished).

Abundance and productivity

Prior to 1940, recruitment of adults to the Oregon Coast coho salmon ESU is estimated to have averaged about 800,000 fish, ranging from 400,000–2,000,000. 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 60s, with peak exploitation rates between 80–90% 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 1955–93, spawner escapements were in the range of 50,000 even as recruitment ranged up to 800,000 fish (Stout et al. 2012).

Caldwell and Cramer (2015) argue that these historic estimates of recruit and spawner abundances may be too high due to methodological changes in how spawners and especially harvest rates were estimated. They propose using the recruitment time series for the 1950s–1980s developed by Lawson (1992), which indicates recruitment during 1950–90 that is substantially below current historic 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 60s and 70s (vs. 350,000), and 100,000 in the 80s (vs. 150,000). Use of the Lawson recruitment time series indicates the timing of the large decrease in abundance occurred before 1950, not afterwards as presently assumed.

Regardless, Oregon Coast coho salmon reached extremely low levels in the late 1990s, when 22,800–47,400 spawners were estimated across the entire ESU (Figures 116 and 117, Table 65). Since that time, management actions by ODFW to dramatically decrease harvest and hatchery production, paired with favorable ocean conditions, have contributed to a strong rebound of Oregon Coast coho salmon (Falcy and Suring 2018). High spawning escapements in 2011 and 2014 (over 338,000 natural spawners) likely resulted in the reestablishment of many of the natural processes associated with salmon populations, although these levels are still below those seen as recently as the mid-1970s (although see discussion below). Marine survival appears to be the principal driver of variation in abundance, and poor ocean conditions since 2014 have again resulted in lower spawner abundances, but not as low as those observed in the late 1990s.

The spawner abundance within the Oregon Coast coho salmon ESU varies by time and population. The large populations (abundances >6,000 spawners since 2015) include Nehalem, Tillamook Bay, Alsea, Siuslaw, Lower Umpqua, Coos, and Coquille (Figure 117, Table 65). The total abundance of spawners within the ESU generally increased between 1999 and 2014, before dropping in 2015 and remaining low (Figure 116). The 2014 Oregon Coast coho salmon return (355,600 natural and hatchery spawners) was the highest since at least the 1950s (2011 was the second highest, with 352,200; ODFW 2015), while the 2015 return (56,000 fish) was the lowest since the late 1990s. Most independent and dependent populations show synchronously high abundances in 2002–03, 2009–11, and 2014, and low abundances in 2007, 2012–13, and now 2015–19 (Figure 117), indicating the overriding importance of marine survival to returns of Oregon Coast coho salmon.

While marine survival is important for Oregon Coast coho salmon (Falcy and Suring 2018), so is high-quality freshwater habitat for juvenile rearing, adult spawning, and egg incubation. These are also habitats that humans control and influence through their actions across the landscape. Of particular importance to Oregon Coast coho salmon BRTs and TRTs (Wainwright et al. 2008, Stout et al. 2012) are the need for American beaver (*Castor canadensis*) within the ESU. Beavers are a keystone species that has wide-ranging impacts on stream ecosystems, because their dams create pools that serve as high-quality habitat for a number of plant, invertebrate, and vertebrate species, including juvenile coho salmon. However, widespread historical removal of beavers has resulted in beaver populations that are a small fraction of their historical abundance (Pollock et al. 2003, 2015). Loss of high-quality beaver-associated habitat has been identified as limiting the production of Oregon Coast coho salmon (see review in Stout et al. 2012).

Five-year geometric mean natural raw spawner abundances increased from 17–7,228 per population in the 1990–94 time period to 189–23,741 for the 2010–14 time period, the highest in the time series (Table 65). Populations decreased during the most recent period (2015–19), to 67–6,740. All populations exhibited a substantial decrease in the geometric mean abundance between the previous five-year period (2010–14) and the current one (2015–19), ranging from –55% (Siletz) to –75% (Tenmile Lake; Table 65).

A similar pattern is observed with 15-year trends in log natural spawner abundances (Table 66): all were positive during the 1990–2005 period, and most were near zero or slightly negative during the 2004–19 period. Furthermore, trends during the earlier 15-year interval (1990–2004) were steeper and no confidence intervals overlapped zero, while the

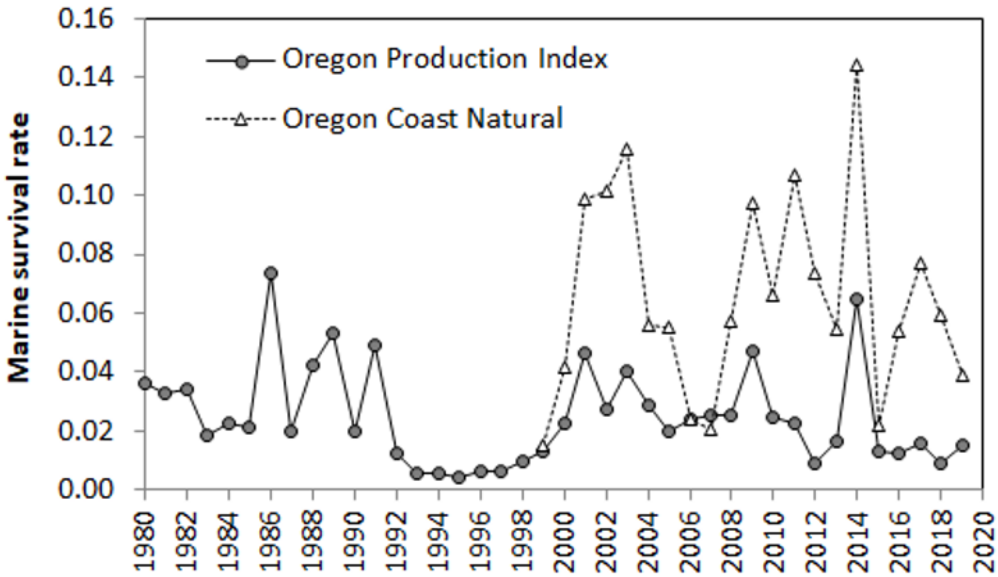

Figure 115. Marine survival rates for Oregon Production Index hatchery-produced coho salmon, 1980–2019, and Oregon Coast natural coho salmon from life cycle monitoring sites, 1999–2019. Data from Suring et al. 2015, PFMC 2020, and Suring (unpublished).

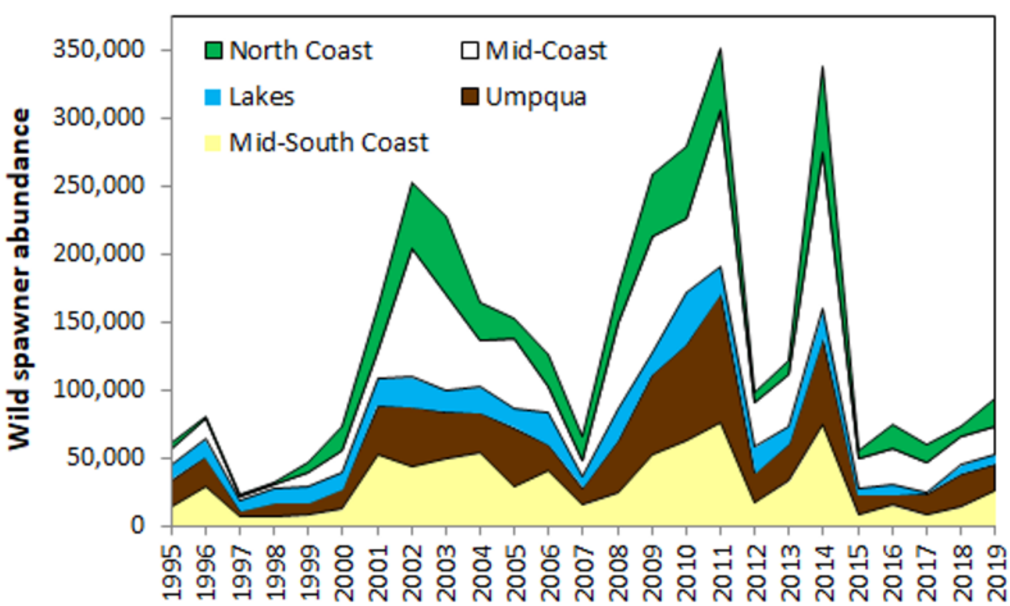

Figure 116. Estimated abundance of natural spawners in the five strata for the Oregon Coast coho salmon ESU, 1995–2019.

recent trends for all populations are relatively flat and the confidence intervals for all but six populations include zero (Table 66). The six populations that don't include zero have negative trends and are located in the Lakes (Siltcoos, Tahkenitch, and Tenmile Lakes) or Mid-South Coast (Coos, Floras, and Sixes) strata.

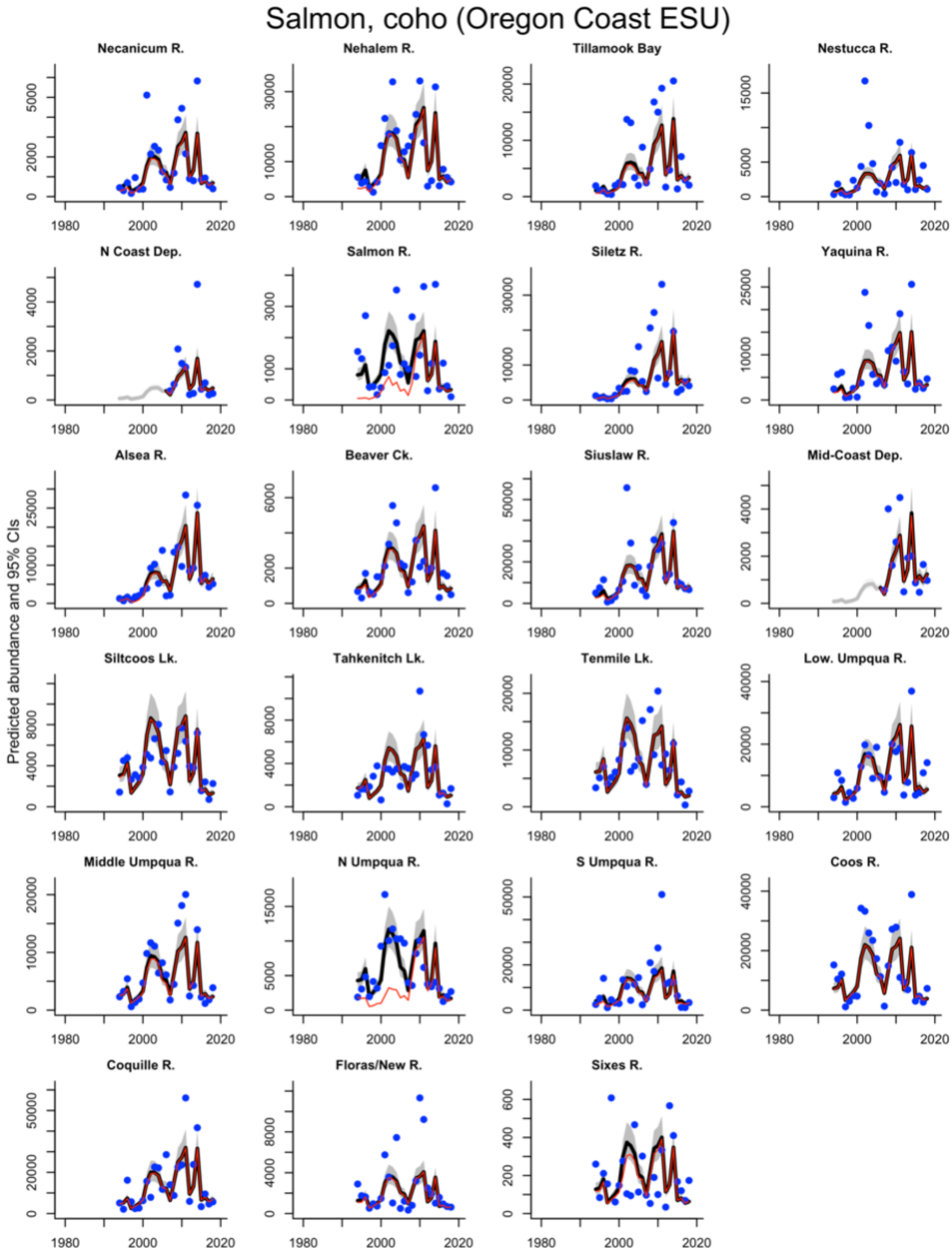

Figure 117. Smoothed trend in estimated total (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot.

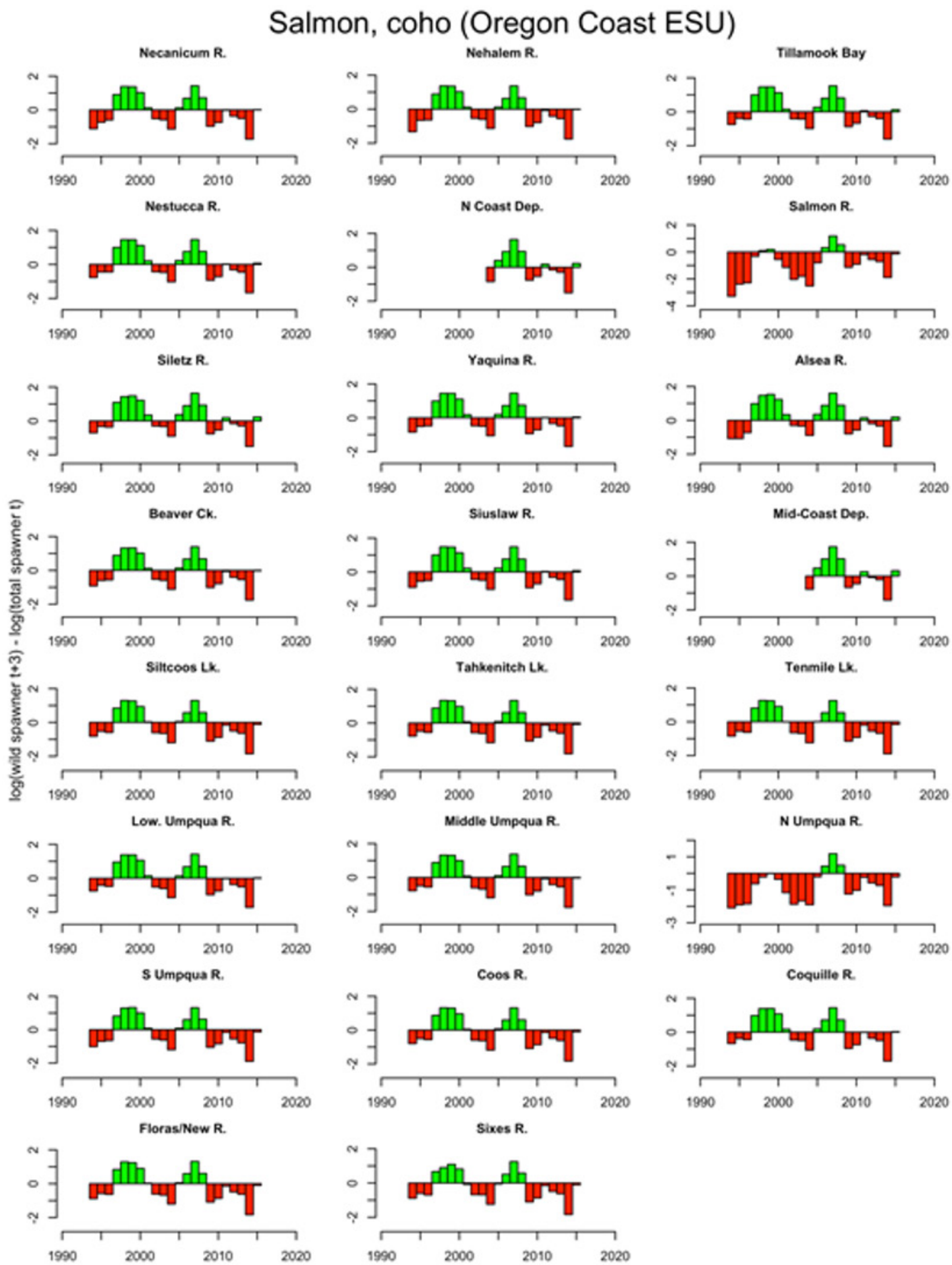

Figure 118. Trends in population productivity, estimated as the log of the smoothed natural spawning abundance in year *t* minus the smoothed natural spawning abundance in year ($t-3$).

Patterns of natural spawner abundances, including short- and long-term trends for dependent populations in the North and Mid Coasts, were similar to larger independent populations (Figure 117). Short-term trends declined by –59% and –51%, respectively, between the two five-year periods (2010–14 and 2015–19; Table 65). Long-term trends were slightly positive (0.02 and 0.05, respectively) during the 2004–19 period (Table 66), although the 15-year trend confidence intervals included zero. Spawner-to-spawner

Table 65. Five-year geometric mean of raw natural-origin 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-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 2 most-recent 5-year periods is shown on the far right. *MPGs*: NC = North Coast, MC = Mid-Coast, L = Lakes, U = Umpqua, MSC = Mid-South Coast.

Population	**MPG**	**1990–94**	**1995–99**	**2000–04**	**2005–09**	**2010–14**	**2015–19**	**% change**
Necanicum	NC	281 (468)	271 (412)	1,794 (1,897)	1,097 (1,175)	2,079 (2,094)	637 (645)	–69 (–69)
Nehalem	NC	2,474 (7,471)	1,355 (2,934)	20,139 (20,469)	14,485 (15,091)	11,523 (11,647)	4,812 (4,849)	–58 (–58)
Tillamook Bay	NC	425 (938)	590 (829)	4,507 (5,015)	5,009 (5,117)	8,437 (8,601)	2,748 (2,755)	–67 (–68)
Nestucca	NC	352 (412)	594 (678)	5,269 (5,394)	1,318 (1,327)	2,764 (2,812)	1,860 (1,860)	–33 (–34)
North Cost Dependent	NC	—	—	—	790 (794)	883 (890)	359 (360)	–59 (–60)
Salmon	MC	17 (267)	44 (645)	272 (1,186)	259 (1,136)	1,450 (1,463)	357 (372)	–75 (–75)
Siletz	MC	493 (930)	427 (597)	3,765 (4,278)	9,628 (10,024)	10,689 (10,697)	3,447 (3,447)	–68 (–68)
Yaquina	MC	546 (658)	1,639 (1,978)	5,486 (5,561)	5,624 (5,817)	9,863 (9,863)	3,199 (3,223)	–68 (–67)
Beaver	MC	347 (347)	655 (767)	2,938 (3,069)	1,637 (1,665)	2,618 (2,618)	812 (813)	–69 (–69)
Alsea	MC	1,235 (1,851)	527 (1,300)	5,556 (5,800)	6,486 (6,510)	14,090 (14,099)	5,616 (5,616)	–60 (–60)
Siuslaw	MC	3,175 (4,554)	2,324 (3,032)	15,762 (15,781)	11,355 (11,625)	21,675 (21,913)	8,179 (8,179)	–62 (–63)
Mid-Coast Dependent	MC	—	—	—	1,509 (1,535)	1,830 (1,860)	890 (900)	–51 (–52)
Siltcoos Lake	L	1,517 (1,568)	3,430 (3,468)	5,451 (5,481)	3,699 (3,702)	5,545 (5,550)	1,571 (1,571)	–72 (–72)
Tahkenitch Lake	L	841 (843)	2,176 (2,206)	2,439 (2,445)	2,851 (2,868)	5,509 (5,513)	885 (889)	–84 (–84)
Tenmile Lake	L	2,616 (2,632)	5,420 (5,420)	8,918 (8,931)	9,547 (9,562)	9,986 (10,008)	1,684 (1,684)	–83 (–83)
Lower Umpqua	U	2,904 (2,976)	4,197 (4,390)	11,348 (11,758)	10,180 (10,944)	12,862 (12,874)	7,082 (7,096)	–45 (–45)
Middle Umpqua	U	2,857 (3,039)	1,828 (1,935)	7,907 (8,265)	5,239 (5,689)	8,797 (8,804)	2,062 (2,062)	–77 (–77)
North Umpqua	U	900 (2,650)	939 (3,276)	2,729 (11,356)	2,946 (6,503)	4,552 (5,018)	1,976 (2,135)	–57 (–57)
South Umpqua	U	1,633 (2,295)	3,125 (4,151)	6,876 (7,272)	8,670 (9,163)	18,237 (19,055)	1,977 (2,326)	–89 (–88)
Coos	MSC	7,228 (8,150)	4,572 (4,597)	19,936 (20,077)	10,048 (10,116)	15,029 (15,053)	4,071 (4,071)	–73 (–73)
Coquille	MSC	3,934 (4,165)	4,117 (4,169)	12,692 (13,099)	15,598 (15,629)	23,800 (23,867)	5,386 (5,386)	–77 (–77)
Floras/New	MSC	—	898 (1,009)	2,868 (2,978)	863 (883)	3,489 (3,489)	898 (898)	–74 (–74)
Sixes	MSC	103 (111)	146 (159)	134 (180)	118 (127)	189 (192)	125 (125)	–34 (–35)

ratios show the same cycle of positive and negative stanzas displayed by independent populations (Figure 118). Given that small populations are more likely to “wink out” than large populations due to stochastic processes, these patterns suggest small dependent populations on the Oregon Coast were not unduly impacted by unfavorable ocean conditions and respond much like their larger neighbors.

Abundance and productivity are captured by three criteria in the DSS (Table 64): population productivity (PP-1, productivity [natural return ratio] at low abundance), probability of persistence (PP-2, based on population viability models which have not been updated since 2008), and critical abundance (PP-3, peak spawner density during lowest years). Scores for population productivity (PP-1) increased in half the populations (11 of 21) between runs in 2012 and 2015. Between 2015 and 2020 DSS runs, scores increased in seven populations, stayed constant in two, and the rest declined (Table 64). The average score across all populations increased from 0.69 in 2012 to 0.71 in 2015, and then declined to 0.58 in 2020. The number of populations with a score of at least 0.3 (i.e., moderate-to-high certainty that population production at low abundance is sufficient to withstand an extended period of adverse environmental conditions) was 19 in both 2012 and 2015, but decreased to 17 populations in 2020.

Table 66. Fifteen-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 2019 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	2004–19
Necanicum	North Coast	0.16 (0.11, 0.21)	–0.04 (–0.12, 0.03)
Nehalem	North Coast	0.19 (0.14, 0.24)	–0.06 (–0.14, 0.01)
Tillamook Bay	North Coast	0.21 (0.17, 0.26)	–0.01 (–0.09, 0.06)
Nestucca	North Coast	0.14 (0.09, 0.19)	–0.03 (–0.11, 0.04)
North Coast Dependent	North Coast	—	0.02 (–0.06, 0.09)
Salmon	Mid-Coast	0.24 (0.18, 0.31)	0.01 (–0.10, 0.11)
Siletz	Mid-Coast	0.21 (0.17, 0.26)	0.02 (–0.05, 0.10)
Yaquina	Mid-Coast	0.18 (0.12, 0.23)	–0.04 (–0.12, 0.04)
Beaver	Mid-Coast	0.13 (0.08, 0.19)	–0.06 (–0.14, 0.01)
Alsea	Mid-Coast	0.17 (0.11, 0.24)	0.01 (–0.07, 0.08)
Siuslaw	Mid-Coast	0.16 (0.11, 0.21)	–0.03 (–0.11, 0.04)
Mid-Coast Dependent	Mid-Coast	—	0.05 (–0.03, 0.13)
Siltcoos Lake	Lakes	0.11 (0.06, 0.15)	–0.10 (–0.17, –0.02)
Tahkenitch Lake	Lakes	0.12 (0.07, 0.17)	–0.08 (–0.16, –0.01)
Tenmile Lake	Lakes	0.12 (0.07, 0.16)	–0.11 (–0.19, –0.04)
Lower Umpqua	Umpqua	0.12 (0.07, 0.17)	–0.05 (–0.12, 0.03)
Middle Umpqua	Umpqua	0.11 (0.06, 0.16)	–0.06 (–0.14, 0.01)
North Umpqua	Umpqua	0.07 (0.02, 0.13)	–0.02 (–0.12, 0.08)
South Umpqua	Umpqua	0.16 (0.10, 0.21)	–0.08 (–0.16, 0.00)
Coos	Mid-South Coast	0.11 (0.07, 0.16)	–0.09 (–0.17, –0.01)
Coquille	Mid-South Coast	0.14 (0.09, 0.19)	–0.05 (–0.12, 0.03)
Floras/New	Mid-South Coast	—	–0.09 (–0.17, –0.01)
Sixes	Mid-South Coast	0.06 (0.02, 0.11)	–0.08 (–0.16, –0.01)

Scores for PP-3 (critical abundance) increased or remained stable at 1.0 between the 2012 and 2015 assessments for all populations except the Sixes River, and mean scores increased from 0.40 to 0.66 (Table 64). Between the 2015 and 2020 DSS runs, however, PP-3 scores decreased in half the populations (12 of 21), although the mean score across all populations in 2020 (0.65) was only 0.01 lower than the mean 2015 (0.66) score. The number of populations with a score of at least 0.3 for PP-3 (i.e., moderate-to-high certainty that population abundance is maintained above levels where small-population demographic risks are likely to occur) went from 15 in 2012 to 18 in both 2015 and 2020.

The overall population persistence (PP) scores (based on PP-1 and PP-3) for individual populations from the most recent run were positive (i.e., with varying certainty the population was persistent) for all but three populations (Necanicum, Salmon, and Sixes). This is an improvement over previous runs, when four populations had negative scores (Necanicum, Salmon, Sixes, and North Umpqua). However, mean PP scores in 2020 were slightly lower (0.45) than in 2015 (0.52), but higher than mean scores in 2012 (0.35; Lewis 2020).

Marine survival has been highly variable over the last four decades (Figure 115). Marine survival rates for the OPI are estimated from hatchery coho salmon from the Columbia River and the 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 Coast natural (OCN) coho are available from ODFW's life cycle monitoring sites starting with the 1999 return (Suring et al. 2012, 2015, Suring, unpublished). As described above, the OCN marine survival time series provided here is based on only five LCM sites, because one LCM site was terminated since the 2015 update (Suring, unpublished).

In general, marine survival of OCN coho salmon is roughly twice as high as OPI coho, but also shows different trends (Figure 115). Mean survival in 1999–2019 was 6.6% for OCN compared to 2.5% for OPI, although in some years the rates are quite similar (e.g., 1999, 2006, 2007, and 2015). The trends for both times series increased between return years 1999 and 2014, due to low marine survival prior to 2001 and extremely high marine survival for fish returning in 2014 (14.5% for OCN, 6.5% for OPI). Marine survival rates for both OCN and OPI coho returning in 2015 plummeted to 2.2% and 1.3%, respectively. OPI has remained below 1.6% since then, while OCN rebounded somewhat in 2017 (7.7%) before dropping again to 3.9% in 2019. Compared to marine survival rates during the earlier assessments, OCN marine survival rates averaged over the most recent five years (5.0% in 2015–19) are substantially lower than in 2010–14 (8.9%), which were the most recent five years used in the 2015 assessment (NWFSC 2015). However, current OCN rates are similar to the five-year time period before that (2005–09, 5.1%), which were the most recent years used by Stout et al. (2012). By contrast, mean OPI marine survival rates in the most recent five years (1.3%) are less than half the rates during the two previous five-year time periods (both 2.8%).

Harvest

OCN coho salmon are part of the OPI, 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 salmon retention in 1993 and have remained closed ever since. Ocean fisheries for coho salmon 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 8–30%. Overall exploitation rates regularly exceeded 60% in the 1980s, but have remained below 20% since 1993 (Figure 119). As discussed above, Caldwell and Cramer (2015) argue that harvest rates on Oregon coho salmon were overestimated by OPI during the 1950s and underestimated in the 1980s and 1990s. This does not affect the low harvest rates beginning in 1993.

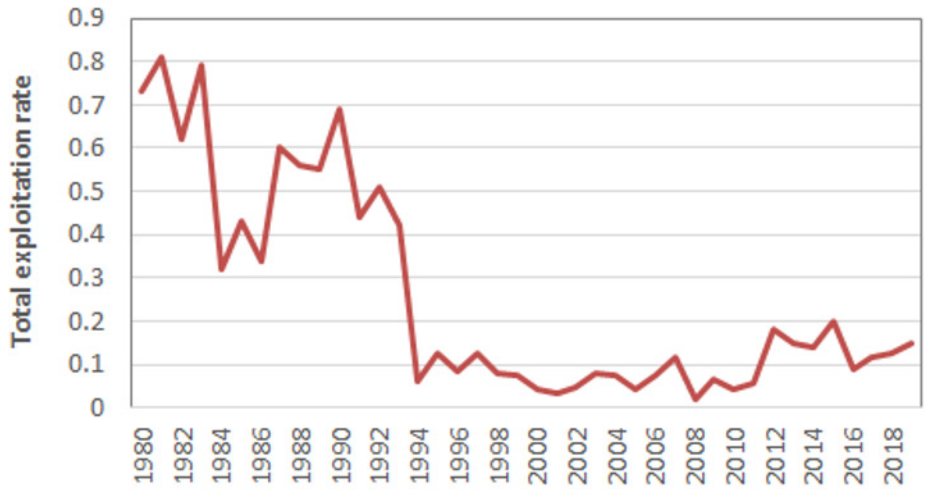

Figure 119. Total marine and freshwater exploitation rates on Oregon coast natural coho salmon. Data from ODFW and PFMC (2020).

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 are similar to previous assessments, or improved in some cases (e.g., reduced hatchery influence). Evidence for spatial structure and diversity is provided indirectly by several criteria in the DSS, as well directly from patterns of spawner abundance and productivity across the geographic range of the ESU.

In the DSS, spatial structure and diversity are evaluated at the population level, with assessments of spawner abundance (PD-1), hatchery influence (PD-2), and spawner (PD-3) and juvenile (PD-4) distributions within watersheds, all of which have been updated in the DSS (Table 64). Spatial structure and diversity were also evaluated at the ESU level, with assessments of genetic diversity (ED-1), phenotypic and habitat diversity (ED-2), and existence of small populations (ED-3); these criteria were evaluated using professional judgment and have not been updated since Wainwright et al. (2008).

For PD-1 (sufficient abundance to avoid genetic risks), between the 2012 and 2015 DSS runs, population scores either increased (17 populations) or remained constant (three populations). Compared to the 2015 run, PD-1 scores from the 2020 run resulted in eight populations with increased scores, nine with no change, and four with decreased scores. Across all populations, mean scores increased from 0.24 in 2012 to 0.26 in 2015, and remained unchanged (0.26) in 2020. The number of populations with PD-1 scores exceeding 0.3 (i.e., moderate-to-high certainty that populations have sufficient spawners to prevent loss of genetic variation) increased from seven in the 2012 assessment to eight in both 2015 and 2020 (Table 64).

The DSS criterion for artificial influence (PD-2) assesses the proportion of naturally produced fish over two generations or six years. Scores for this factor increase every time the DSS is run in response to reduced hatchery production in the ESU. Average scores have increased from 0.55 in 2012, to 0.87 in 2015, to 0.88 in 2020. In the most recent assessment, only two populations (North and South Umpqua) failed to have either high or complete certainty that hatchery influence does not adversely affect natural populations (scores >0.70). Furthermore, trends in the proportion of natural spawners (Figure 120) that are not already at 1.0 are all upwards; these consistently high values are perhaps the highest of any ESU reviewed here. The State of Oregon made an unprecedented effort to reduce hatchery influence in natural Oregon Coast coho salmon populations by greatly reducing the production of hatchery coho salmon along the coast. The result of this action is that all but two independent populations in the entire ESU currently have a five-year average of >95% of natural spawners (Table 67). The sole exceptions are the North and South Umpqua populations. Hatchery production in North Umpqua was terminated in the late 1990s, and PD-2 scores have increased from –0.96 in the 2012 run to 0.34 in the most recent run. South Umpqua has been at or above 0.50 in previous DSS runs, but scored –0.04 in the 2020 run due to anomalously high contribution of hatchery fish in 2016 (Figure 120).

Scores for the two population-level distribution metrics, PD-3 (spawners) and PD-4 (juveniles), both increased between the 2012 and 2015 assessments, indicating improved dispersal of both adults and juveniles across the landscape. The scores from the 2020 assessment were mixed: mean scores for the distribution of spawners decreased, while the score for juveniles continued to increase. Accordingly, spawner distribution (PD-3) scores were 0.49, 0.65, and 0.56 in 2012, 2015, and 2020, respectively. Juvenile distribution (PD-4) scores were 0.60, 0.69, and 0.75 in 2012, 2015, and 2020, respectively. The number of populations with scores of at least 0.3 (i.e., moderate-to-high certainty that historically occupied watersheds in the population's range had spawners and juveniles occupying the available habitat) has increased from 14 and 16 populations for spawners and juveniles, respectively, in 2012, to 17 and 19 populations, respectively, in 2020 (Table 64).

Population sustainability (PS) scores, which are based on PP-1 through PP-3 and PD-1 through PD-4, were positive for all populations in the most recent model run (i.e., they had a varying certainty of being sustainable) except Necanicum, Salmon, North Umpqua, and Sixes, which had negative scores. These same four populations had negative PS scores in the 2015 run, but a fifth population (Floras/New) also received a negative PS score in the 2012 run. Like the trends in PP scores, mean PS scores in 2020 (0.36) were intermediate between those in 2015 (0.40) and 2012 (0.25; Lewis 2020).

The spatial structure of coho salmon populations within the ESU can also be inferred from population-specific spawner abundances (Figure 117) and productivity (Figure 118). In particular, there is no geographic area or stratum within the ESU that appears to have considerably lower abundances or to be less productive than other areas or strata and therefore might serve as a "population sink." Furthermore, if the factors driving 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). Abundance and productivity trends for dependent populations in the North Coast and Mid Coast strata show the same patterns and trends as independent populations, consistent with this premise.

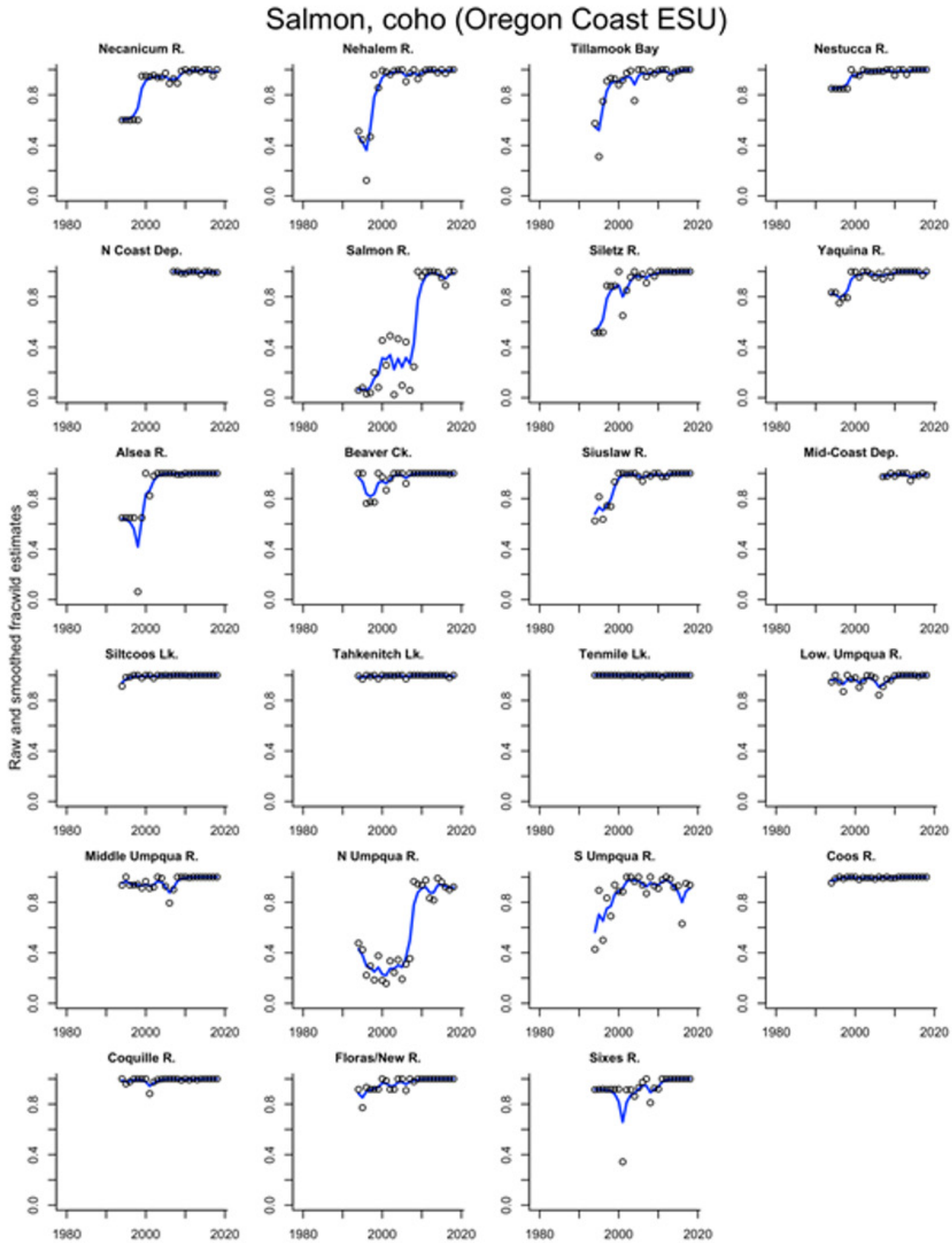

Figure 120. Smoothed trend in the estimated fraction of the natural spawning population consisting of fish of natural origin. Points show the annual raw estimates.

Table 67. Five-year mean of fraction natural (sum of all estimates divided by the number of estimates). Blanks mean no estimate available in that 5-year range.

Population	MPG	1995–99	2000–04	2005–09	2010–14	2015–19
Necanicum	North Coast	0.67	0.95	0.93	0.99	0.99
Nehalem	North Coast	0.57	0.98	0.96	0.99	0.99
Tillamook Bay	North Coast	0.77	0.90	0.98	0.98	1.00
Nestucca	North Coast	0.88	0.98	0.99	0.98	1.00
North Coast Dependent	North Coast	—	—	1.00	0.99	1.00
Salmon	Mid-Coast	0.09	0.34	0.37	0.99	0.96
Siletz	Mid-Coast	0.74	0.89	0.96	1.00	1.00
Yaquina	Mid-Coast	0.83	0.99	0.97	1.00	0.99
Beaver	Mid-Coast	0.86	0.96	0.98	1.00	1.00
Alsea	Mid-Coast	0.53	0.96	1.00	1.00	1.00
Siuslaw	Mid-Coast	0.77	1.00	0.98	0.99	1.00
Mid-Coast Dependent	Mid-Coast	—	—	0.98	0.98	0.99
Siltcoos Lake	Lakes	0.99	0.99	1.00	1.00	1.00
Tahkenitch Lake	Lakes	0.99	1.00	0.99	1.00	1.00
Tenmile Lake	Lakes	1.00	1.00	1.00	1.00	1.00
Lower Umpqua	Umpqua	0.96	0.97	0.93	1.00	1.00
Middle Umpqua	Umpqua	0.95	0.96	0.92	1.00	1.00
North Umpqua	Umpqua	0.30	0.25	0.55	0.91	0.93
South Umpqua	Umpqua	0.77	0.95	0.95	0.96	0.86
Coos	Mid-South Coast	0.99	0.99	0.99	1.00	1.00
Coquille	Mid-South Coast	0.99	0.97	1.00	1.00	1.00
Floras/New	Mid-South Coast	0.89	0.96	0.98	1.00	1.00
Sixes	Mid-South Coast	0.92	0.79	0.93	0.98	1.00

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 2019) are intermediate between the two previous assessments for both ESU persistence and ESU sustainability. The most recent EP value is 0.60 (high certainty the ESU is likely to persist), which is in between the values for 2015 (0.73, high certainty) and 2012 (0.44, moderate certainty). For ES, the current value is 0.24 (low-to-moderate certainty the ESU is sustainable), which is also between the 2015 (0.29, moderate certainty) and 2012 (0.23, low-to-moderate certainty) values.

The decrease in EP and ES values and overall decline in abundance and productivity indicate that Oregon Coast coho salmon were clearly impacted by unfavorable ocean conditions due to marine heat waves. However, Oregon Coast coho salmon fared surprisingly well compared to many other ESUs assessed here, and showed remarkable ability to avoid the extremely low abundances and marine survival rates observed in the late 1990s during the previous extended downturn.

Federal recovery plan

The final federal recovery plan for Oregon Coast coho salmon was released in December 2016 (NMFS 2016). The overriding theme of the plan is "to protect and restore the freshwater and estuarine rearing habitats that support juvenile survival and overall productivity" (p. S-1) so that the ESU is sustainable and persistent and no longer needs federal protection under the ESA. The plan states that the federal government will remove Oregon Coast coho salmon from ESA listing when it determines that:

1. The species has achieved a biological status consistent with recovery—i.e., when the best available information indicates it has sufficient abundance, population growth rate, population spatial structure, and diversity to meet its biological recovery goal.
2. The factors that led to ESA listing have been reduced or eliminated to the point where federal protection under the ESA is no longer needed, and there is reasonable certainty that the relevant regulatory mechanisms are adequate to protect Oregon Coast coho salmon sustainability.

The biological status of the ESU is 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 artificial influence (PD-2). In the 2020 run of the DSS, the majority of populations within each stratum had moderate-to-high certainty the population was sustainable (i.e., PS >0.30; Table 64). At the stratum level, stratum sustainability (SS) scores show that all strata had positive scores (low-to-high certainty the stratum is sustainable). Three strata (Mid Coast, Lakes, and Mid-South Coast) had high certainty the strata were sustainable, one (North Coast) had moderate certainty, and only one stratum (Umpqua) had low-to-moderate certainty (Table 64). The current DSS scores (described above, Lewis 2020) shows that there is low-to-moderate certainty the ESU is sustainable.

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" (p. 3, ODFW 2007). The plan relies on a combination of existing regulatory programs and effective long-term participation in non-regulatory conservation work to achieve desired status. The OCCCP defines the desired status for the ESU, which is evaluated using six measurable 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 passes 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 the previous sentence) must maintain or improve upon their current scores" (p. 2, ODFW 2007, Appendix II). The latest iteration of the DSS (using data through return year 2019) indicates that four independent populations do not meet this criterion (population sustainability <0.0; Lewis 2020). These populations are Necanicum, Salmon, North Umpqua, and Sixes; these same four populations were the only populations to have PS scores below 0.0 in the previous run of the DSS using comparable data (Lewis 2020).

It should be noted, however, that three of these populations are unlikely to have PS scores greater than 0.0 unless the PVAs (the criteria for PP-2) are updated and the outcomes improve substantially. The original PP-2 scores for the Necanicum, Salmon, and Sixes populations were –0.44, –1.0, and –1.0, respectively, when the DSS was originally run (Wainwright et al. 2008); this prevents these populations from attaining sustainability scores >0.0 regardless of improvements for other DSS metrics. A new PVA assessment using a single model and shorter time periods (1990–2019 or 1999–2019) indicates that the risk of extinction for these populations is substantially improved (ODFW 2021), suggesting that updating PP-2 criteria with the full suite of models may result in improved PP-2 scores.

The conclusions of the draft 12-year assessment of the OCCCP (ODFW 2021) are favorable, but more work is needed to improve freshwater productivity to reach broad-sense recovery goals. Specifically, the conclusions state that:

> "After record low spawner abundances in the 1990s, the biological performance of the OC Coho ESU has continued to improve. Low ocean survival in the late 1990s was effectively the end of a period of low realized ocean survival (i.e., including high rates of harvest) that lasted at least a quarter century. Substantial reductions in harvest, coupled with improved conditions for ocean survival, have resulted in recent returns that include some of the highest in decades, and updated population viability modeling indicates that most populations have low risks of extinction over the next 100 years and substantial improvements in populations at most risk (i.e.,

Salmon; Sixes). Updated metrics for the DSS have also remained generally favorable for the ESU and most Independent Populations despite challenging conditions for both freshwater and ocean survival over the past several years.

"A predominant role of ocean survival in recent increases in abundance does not imply an insignificant role of density dependent freshwater mortality in the regulation of OC Coho spawner abundance. Continued improvements in freshwater productivity through habitat protection, restoration, and management will be necessary to consistently achieve the OCCCP's measureable criteria and to provide substantial ecological and societal benefits. Aside from harvest management, few actions are available to directly address fluctuations in ocean survival. However, continued efforts to address freshwater limiting factors will enhance resiliency of OC coho populations under fluctuating ocean conditions and a changing climate" (p. 83, ODFW 2021).

Updated biological risk summary

As stated above, the draft 12-year assessment of the OCCCP (ODFW 2021) highlights favorable improvements for Oregon Coast coho salmon overall, consistent with our assessment. It also notes the strong role that ocean conditions play on adult returns to the ESU, including recent low abundances associated with strong marine heatwaves (see Habitat chapter). The assessment also highlights the need for continued improvements to freshwater productivity to achieve broad-sense desired status, especially given the expected challenges posed by climate and ocean change.

The latest ESU scores for persistence (high certainty of ESU persistence) and sustainability (low-to-moderate certainty of ESU sustainability) also demonstrate that the biological status of the ESU has decreased slightly since the 2015 review (high certainty of persistence, moderate certainty of sustainability), which covered a period of favorable ocean conditions and high marine survival rates. However, current ESU scores have improved relative to the 2012 assessment (moderate certainty of persistence, low-to-moderate certainty of sustainability). This improvement occurred despite similar or better abundances and marine survival rates during the earlier period, suggesting continued benefits due to management decisions to reduce both harvest and hatchery releases.

Despite these somewhat optimistic results for Oregon Coast coho salmon, however, it is unclear what the future will bring. A recent assessment of the vulnerability of ESA-listed salmonid "species" to climate change indicated that Oregon Coast coho salmon had high overall vulnerability, high biological sensitivity and climate exposure, and only moderate adaptive capacity (Crozier et al. 2019a). Because young coho salmon spend a full year in freshwater before ocean entry, the juvenile freshwater stage is considered to be highly vulnerable. They also scored high in sensitivity at the marine stage due to expected changes due to ocean acidification. These results are consistent with the climate change assessment by Wainwright and Weitkamp (2013), which indicated that Oregon Coast coho salmon will likely be negatively affected by climate change at all stages of the life cycle. Overall, the Oregon Coast coho salmon ESU is therefore at "moderate-to-low" risk of extinction, with viability largely unchanged from the prior review.

Recent Trends in Marine and Terrestrial 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, habitat restoration and degradation, hatchery production) and variation in environmental conditions in both freshwater and marine environments. The increasing trends in natural spawners seen for some ESUs and DPSes 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 and 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).

Relatively 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 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 ,led to subsequent declines in abundance in many ESUs and DPSes.

This chapter summarizes what is known about marine and terrestrial conditions to provide environmental context when examining the viability assessments included in this report. Of primary interest are the recent climatic conditions that have existed over the past 15–20 years, i.e., three to four generations of the Pacific salmonids that are being considered in these assessments.

Observed Environmental Conditions

Precipitation and surface air temperature

A strong and persistent warming trend and large year-to-year variations in precipitation are among the most notable features of western U.S. climate in recent decades (Figure 121). For the Pacific Northwest, water year 2015 stands out as the warmest year on record.

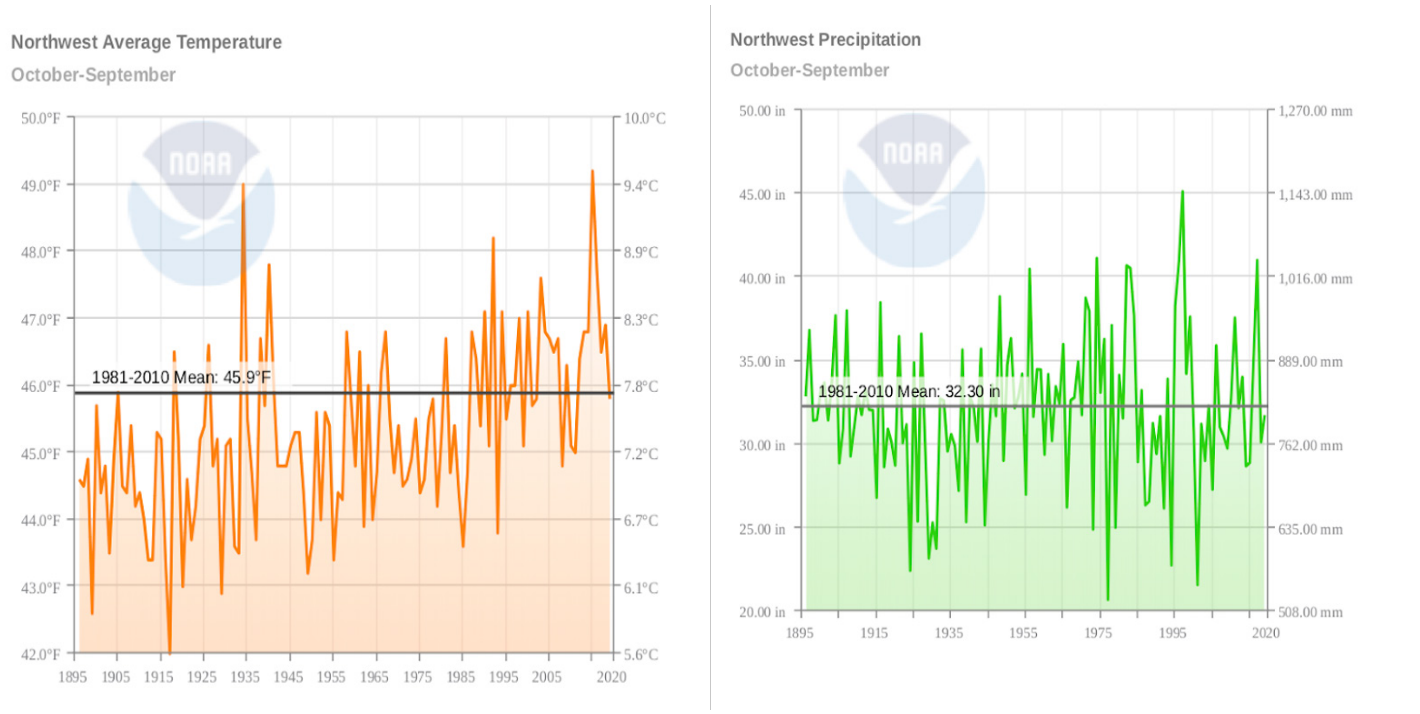

Figure 121. Water year (Oct–Sep) surface air temperature (left) and precipitation (right) for the Pacific Northwest (OR, WA, and ID combined). In each panel, the historical average for 1981–2010 is shown with a black horizontal line. These figures show U.S. Climate Division data and were created at https://www.ncdc.noaa.gov/cag/regional/time-series in October 2021.

Stream flow

A broad-brush overview of water year streamflow variations in western states is provided in Figure 122, where stream gage data indicate substantially more low-flow than high-flow years from 2000–19. Columbia River basin streamflows were below average from 2001–05, 2007–10, and 2015–16, and much above average in water years 2011–12 and 2017–18.

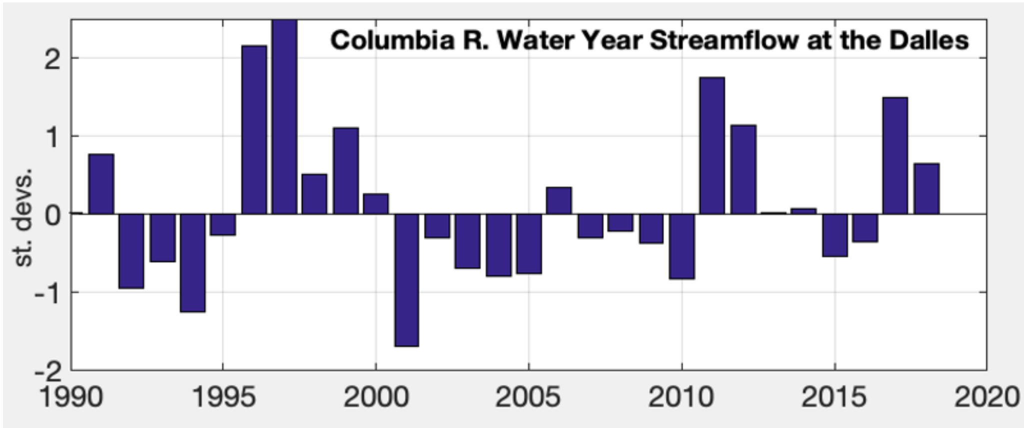

Figure 122. Water year streamflow anomalies (normalized with respect to the 1981–2010 mean and standard deviation) for the Columbia River at The Dalles. Data for this figure were downloaded from https://waterdata.usgs.gov/nwis (October 2021).

Last five years: Annual anomalies from the recent past

Over the past century, temperatures have risen steadily, while precipitation remains highly variable. Warmer temperatures intensify the hydrological cycle within the atmosphere, causing more intense storm events (Warner et al. 2015). Within snow-dominated watersheds, warmer winters and springs reduce snow accumulation and hasten snowmelt. Reduced snowpack causes an earlier and smaller freshet in spring. Reduced snowpack also can lead to lower minimum flows and higher stream temperatures in summer (USGCRP 2018). Projections of climate change in the western United States (USGCRP 2018) indicate that both of these trends are likely to continue. Summer precipitation is projected to decline, exacerbating low flows and high stream temperatures in the western United States.

Winter conditions affect most salmon (i.e., all populations other than steelhead and winter-run Chinook salmon) during the egg and early rearing stages, which may be disturbed and relocated during flood events. Migrating smolts typically benefit from higher flows (Faulkner et al. 2018, Notch et al. 2020), although the impacts on migrating adults varies across populations. Summer conditions affect juveniles rearing in streams (especially steelhead, coho, and yearling Chinook salmon), and adults migrating, holding, or spawning over the summer (many Chinook salmon runs, Columbia and Snake River sockeye salmon, and summer-run steelhead).

A recent assessment of exposure to climate change across the U.S. West Coast region (Crozier et al. 2019a) found that, by the 2040s, average stream temperatures are likely to increase by over two standard deviations across most of the region, and that either flooding (southern domains) or loss of snowmelt (northern domains) was also very likely to change dramatically in most ESUs. Here we put these projected changes within the context of recent conditions (2015–19) by expressing four metrics (summer stream temperature, low flow, high flow, and snowpack) in terms of standard deviations from the recent historical mean (1998–2014). Although they are currently anomalous years, they are likely to represent average conditions in the near future.

To facilitate interpretation of salmon dynamics within individual ESUs, Harvey et al. (2018) averaged environmental conditions across many measurement stations within each of six ecoregions, from the interior Columbia River basin, to the Washington coast, to southern California (Figure 123). We have re-analyzed these results to consider the last five years (2015–19) specifically in relation to the mean and standard deviation of the previous 15 years (1998–14). Deviations for each year (Y_t) were calculated from the raw value (X_t) as $Y_t = (X_t - X_\mu) / X_{SD}$ for each region, where X_μ and X_{SD} were the mean and standard deviation, respectively, over the 1998–2014 period.

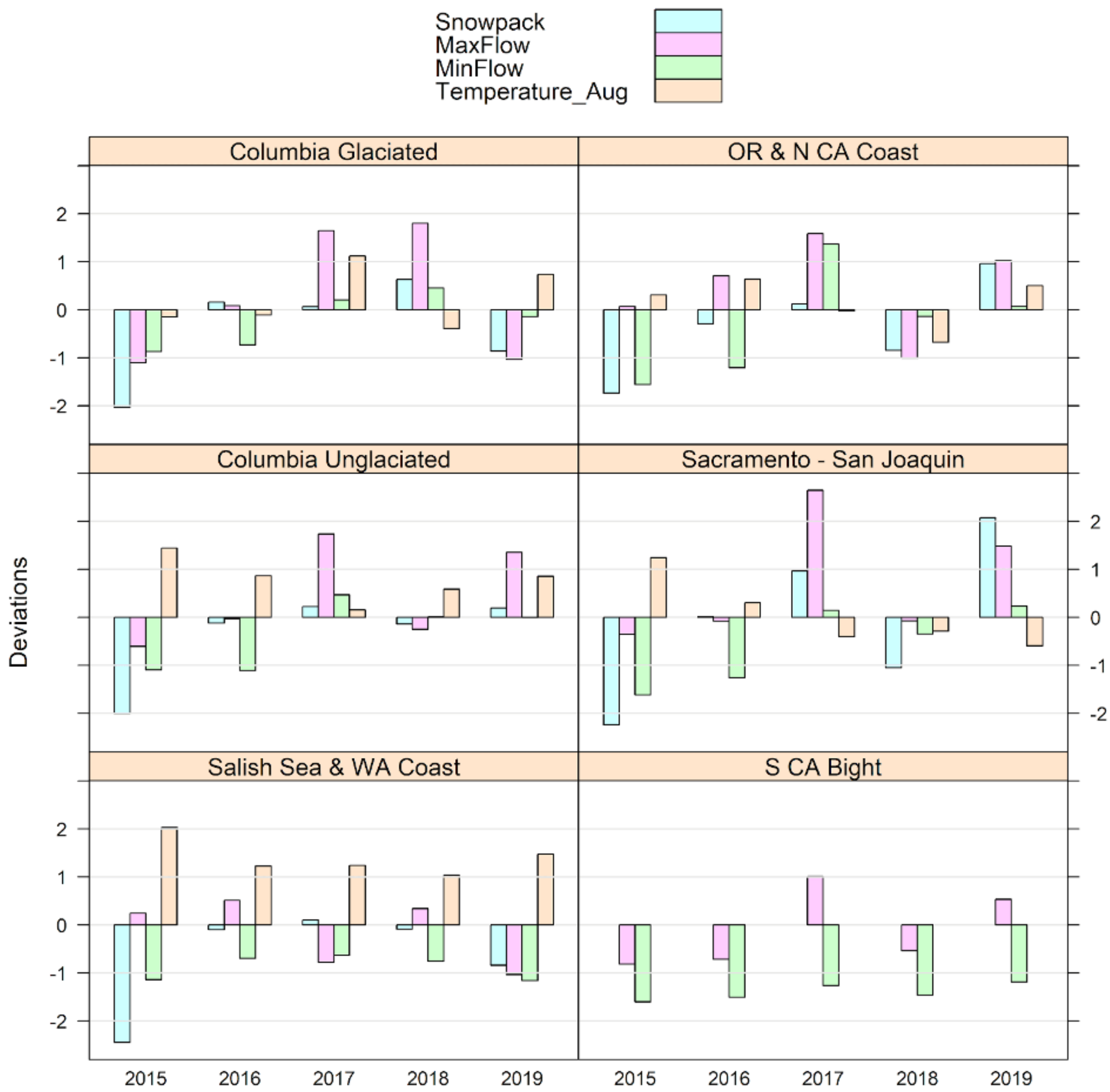

Figure 123. (this page) Deviations from the 1998–2014 baseline period in selected ecoregions in the maximum 1-day flow event per year (MaxFlow), the minimum 7-day flow event per year (MinFlow), snowpack on 1 Apr, and mean Aug stream temperature. (next page) Map of freshwater ecoregions within which conditions were averaged, courtesy of Harvey et al. (2018).

In 2015, the combination of below-average precipitation and record-high surface air temperature brought record-low springtime snowpack to much of the west, leading to what has been called "the western snow drought." The diminished snowpack and high surface temperatures combined with low springtime precipitation yielded especially low runoff to western watersheds in spring and early summer 2015. Temperatures returned to near-normal in much of Washington and Idaho in August (the month shown in Figure 123), but then spiked again in the fall of 2015. Unusually low flows and warm stream temperatures in spring/summer 2015 caused widespread problems for salmon throughout the western United States.

Figure 123 (continued).

In 2016, minimum flows continued to show long-term drought effects, especially in California and the Columbia Unglaciated ecoregion, but other indices were transitioning to more favorable high flows of 2017 in most regions.

Two ecoregions stood out in showing strongly anomalous conditions in all five years: summer temperatures were above average (>1 SD) in the Salish Sea region, and minimum flows were below average (>1 SD) in So Cal Bight throughout the period of this status review.

Particularly notable climate impacts on salmon occurred during the 2015 heatwave in the Pacific Northwest. Using life cycle models of coho salmon in coastal streams in Washington, Ohlberger et al. (2018) found that juvenile production has been limited historically by low-flow periods. In their projections of coho salmon production under future flow scenarios, negative population impacts followed reductions in the mean and increasing variability in annual summer low flows. Other studies (e.g., Larsen and Woelfle-Erskine 2018) found that juvenile coho salmon preferentially select pools with more groundwater intrusion, which stabilizes streams during low-flow periods. Thus, drawdown of coastal aquifers would directly affect potential habitat for these endangered salmon.

In summer 2015, the Columbia River exhibited record temperatures (23°C in the forebay of Bonneville Dam, and 27.5°C in the McNary Dam fish ladder) that affected summer-run salmon populations, especially sockeye and later-migrating components of the spring/summer Chinook run. Only 9% of Snake River sockeye salmon that were detected at Bonneville Dam with PIT tags survived the first stretch of their migration and were detected at McNary Dam (Figure 124). The individuals that survived were those who happened to miss the hottest days in the river (Crozier et al. 2020). Similar high sockeye prespawning mortality due to elevated river temperatures (>19°C) was also observed in the Fraser River basin (MacDonald et al. 2019).

Most of the spring/summer Chinook salmon run reached cooler upriver tributaries before the worst of the heatwave, but mainstem survival of later-migrating populations, such as the Pahsimeroi and South Fork Salmon Rivers, was only 61%, the lowest observed since 2004. Snake River fall-run Chinook salmon hit record high temperatures later in the year, and data from PIT-tagged fish showed that they also exhibited their lowest average apparent survival from Bonneville Dam to Ice Harbor Dam (65%). Interestingly, summer-run steelhead migrate during peak summer temperatures, but they have adapted to move into cool tributaries when temperatures in the mainstem become stressful, and they did not show increased mortality between Bonneville Dam and McNary Dam in 2015 (Crozier et al. 2020, Siegel and Crozier in preparation).

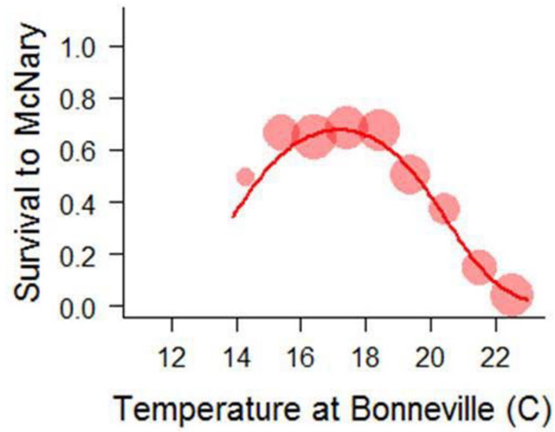

Figure 124. Apparent survival of Snake River sockeye salmon from Bonneville Dam to McNary Dam as a function of the temperature experienced on the day of passage at Bonneville Dam. Survival was assessed using PIT-tag detections at or upstream of McNary Dam. Courtesy of Crozier et al. (2018), Fig. 7.

Ocean conditions

Surface temperatures in the northeastern Pacific Ocean were notably cooler than average from 1999–2002 and again from 2006 through the summer of 2013. They were warmer than normal from 2003–05, and at record highs for much of the period from fall 2013–19 (Figure 125).

For the California Current region, surface temperatures reached record high levels from 2014–16, with 2015 being the single warmest year in the historical record (Jacox et al. 2018). The extreme ocean temperatures for the northeastern Pacific Ocean and the California Current were associated with a small number of persistent wind and weather patterns, some of which have been related to climate conditions in the tropical Pacific Ocean (Di Lorenzo and Mantua 2016, Jacox et al. 2018).

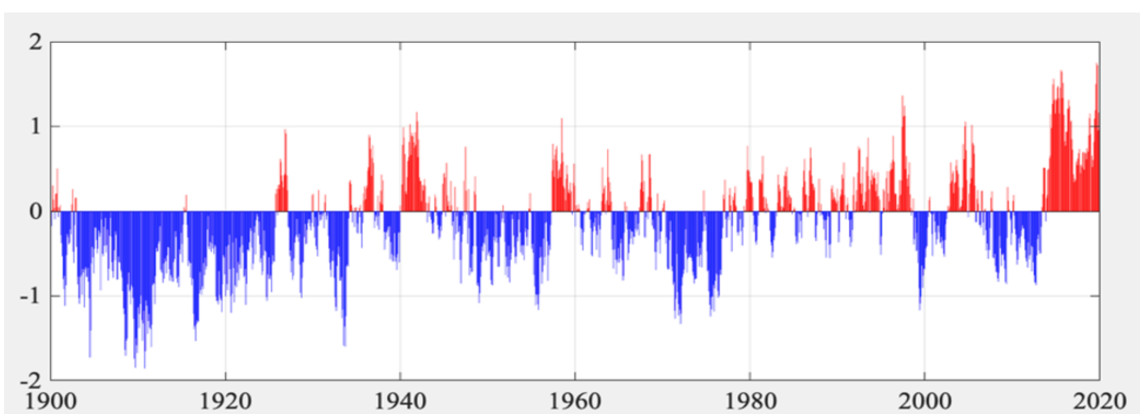

Figure 125. Monthly average sea surface temperature anomaly time series (in °C) for the northeastern Pacific Arc pattern defined by Johnstone and Mantua (2014).

Biological response to marine conditions since 2014

A number of reports provide overviews of recent physical and biological conditions in regions of the northeastern Pacific Ocean that Pacific salmon may occupy during their marine residence period:

- California Cooperative Oceanic Fisheries Investigations' (CalCOFI) State of the California Current (Thompson et al. 2019).
- The Integrated Ecosystem Assessment's California Current Ecosystem Status Report (Harvey et al. 2021).
- Fisheries and Oceans Canada's State of the Physical, Biological and Selected Fishery Resources of Pacific Canadian Marine Ecosystems (Boldt et al. 2019).
- Alaska Fisheries Science Center's Ecosystem Status Reports for the Gulf of Alaska (Zador et al. 2019), the Eastern Bering Sea (Siddon and Zador 2019), and the Aleutian Islands (Zador and Ortiz 2018).
- The Southwest Fisheries Science Center's Coastal Pelagic Survey reports (Stierhoff et al. 2020).

In all cases, the reports show a dramatic biological response at all trophic levels—from primary producers to marine mammals and seabirds—to the marine heatwaves that have spread across the northeastern Pacific Ocean since 2014 and continued into 2020 (and perhaps beyond). These ecosystem changes have had large effects (both positive and negative) on Pacific salmon returns around the Pacific Rim, not just listed species on the U.S. West Coast. How each listed salmon population was impacted by recent anomalous conditions described in this chapter is beyond the scope of this report, and would include many blank sections. This is because each population enters the ocean at different times, locations, and at different sizes (e.g., as subyearling or yearling smolts), even for populations originating within a single river basin (Weitkamp et al. 2015). Where fish from each population go while in the ocean also varies by run timing, natal origin, and species (Beamish 2018); some populations move rapidly offshore (e.g., steelhead), others move northwards along the continental shelf (e.g., Columbia River spring-run Chinook, sockeye, chum, and some coho salmon), while still other populations remain in local waters (many fall-run Chinook, non-Columbia River spring-run Chinook, and some coho). Furthermore, while we have a general understanding of where salmon are during their first and last summers in marine waters, where they go during the one to several years in between is poorly understood. Taken together, these huge variations in salmon marine ecology result in each population being subjected to a unique and poorly documented suite of experiences while in the marine environment, which determines when, where, and how mortality occurs. This poor understanding is especially true for listed populations, because they are like needles in a proverbial haystack. Finding more than one or two individuals from listed populations during their ocean residency—let alone finding enough for robust statistical analyses—is nearly impossible. Consequently, the following section highlights how unusual the entire ecosystem has been—from the lowest to the highest trophic levels, including large salmon populations—to provide a flavor of the likely impacts that listed populations experienced while in marine waters during recent years. Here, we provide brief summaries of the biological trends described by these reports and a few other sources, with an emphasis on findings that are pertinent to salmon survival. Unless otherwise noted, the information comes from the above report series.

Overall, the marine heat wave in 2014–16 had the most drastic impact on marine ecosystems in 2015, with lingering effects into 2016 and 2017. Conditions had somewhat returned to "normal" in 2018, but another marine heat wave in 2019 again set off a series of marine ecosystem changes across the North Pacific. One reason for lingering effects of ecosystem response is due to biological lags. These lags result from species impacts at larval or juvenile stages, which are typically most sensitive to extreme temperatures or changes in food supply. It is only once these species grow to adult size or recruit into fisheries that the impact of the heatwave becomes apparent. For example, most marine mortality for juvenile salmon is thought to occur in the first weeks or months of ocean residence. However, whether marine survival was exceptionally high or low is not known until salmon return as adults, 1–5 years after ocean entry. Biological lags also impact upper trophic levels, such as whales and seabirds, which can take longer to recover from adult mortality and poor body condition.

Primary production

Perhaps the most dramatic change to primary producers was the largest bloom of the diatom *Pseudo-nitzschia* ever recorded in 2015 (McCabe et al. 2016, Bates et al. 2018). The bloom stretched from Southern California to the Aleutian Islands in Alaska, had some of the highest concentrations of cells ever recorded, and was particularly long-lasting. *Pseudo-nitzschia* can produce domoic acid, a neurotoxin that causes amnesic shellfish poisoning, which is potentially fatal in mammals (including humans) and seabirds. In marine food webs, filter-feeding molluscs (primarily bivalves) and planktivorous fishes such as Pacific sardine (*Sardinops sagax caerulea*) and northern anchovy (*Engraulis mordax*) consume *Pseudo-nitzschia*, and species that consume contaminated shellfish and fish become sick or die (McCabe et al. 2016, Bates et al. 2018).

The 2015 bloom caused high domoic acid levels in many commercially and recreationally important species, including Pacific razor clams (*Siliqua patula*), mussels (*Mytilus* spp.) and other bivalves, anchovy and sardines, and benthic scavengers (Dungeness and red rock crab [*Cancer magister and C. productus*]). Transfer of domoic acid to higher trophic levels caused the stranding or death of hundreds of seabirds and marine mammals in 2015 and early 2016, and likely contributed to the large whale unusual mortality event in the Gulf of Alaska in 2015 (Bates et al. 2018).

While subsequent *Pseudo-nitzschia* blooms have not been as extensive as the 2015 bloom, they have continued to cause delays, closures, and restrictions for both razor clam and Dungeness crab fisheries in California, Oregon, and Washington. Southern Oregon and Northern California are particularly prone to elevated domoic acid levels in clams and crabs that exceed permissible levels for human health (20 mg/kg tissue).

Other notable primary-producer-related events include a harmful algal bloom of *Noctiluca* and *Heterosigma* in the Salish Sea (Puget Sound and the Strait of Georgia) in 2018, after a three-year absence. There were also more harmful algal blooms in 2018 than in the previous three years in the Strait of Georgia. In the Gulf of Alaska, phytoplankton blooms were earlier and in higher concentrations in 2017–18 than in the warm years of 2014–16. Surface nutrient concentrations were some of the lowest on record in 2019 across the Gulf of Alaska, which, paired with elevated water temperatures, affected the offshore phytoplankton community, oceanic food webs, and oxygen levels and biogeochemistry.

Lower trophic levels: Copepods, krill, jellyfish, and pyrosomes

Throughout most of the northeastern Pacific Ocean, the marine heatwaves had profound effects on the animals at the base of the food web. Summer copepod communities are normally dominated by cold-water (i.e., lipid-rich) species, but during the heatwaves, northern species were largely or completely absent and warm-water (lipid-poor) species dominated. Not only were southern species abundant, but novel communities were observed in many areas. On the Newport Hydrographic Line (lat 44.6°N), for example, 14 species of copepods that had never been observed were documented, originating both offshore and from southern waters (Peterson et al. 2017). Other changes on the Newport line during the initial heatwave included reduced biomass of copepods and krill, and high abundances of gelatinous organisms such as larvaceans and doliolids (both types of pelagic tunicates). Similar abrupt changes in copepods, krill, and gelatinous organisms were observed from Southern California to the Gulf of Alaska.

To characterize this shift in biomass between major functional groups, Galbraith and Young (2019) developed a "crunchy" versus "squishy" index. The index is the ratio of zooplankton with hard chitinous exoskeletons with high protein and lipid (i.e., crunchy) to zooplankton with hydrostatic skeletons, mainly gelatinous animals with high water content and low nutritional value (i.e., squishy). They show a very high squishy biomass in most areas of British Columbia in 2014–19, peaking in most areas in 2015. Furthermore, Galbraith and Young (2019) expect that years with high squishy index equate to poor survival for juvenile fish and seabirds, which have higher survival when prey quality is high (i.e., crunchier).

The marine heatwave also negatively affected krill growth rates, abundance, and species composition from California to central Alaska. For example, krill were absent from the Steward (Alaska) line during 2014–17, but high in early fall of 2018. Krill length, used to indicate growth, was poor in 2014–16 but increased in 2018 on the Trinidad Head line (lat 41.1°N) in Northern California. Morgan et al. (2019) cautioned that the perceived absence of krill in some areas (Brodeur et al. 2019) was due to changes in depth distribution, rather than absence, because early larval stages were present. In general, most copepod and krill communities returned to more "normal" conditions in 2018.

Jellyfish communities also exhibited dramatic changes from California to Alaska. In the California Current, Pacific sea nettle (*Chrysaora fuscescens*) is the dominant species near shore. However, starting in 2015, there was a dramatic drop in the abundance of sea nettles and concurrent increase in water jellyfish (*Aequorea* spp.) and egg-yolk jellyfish (*Phacellophora camtschatica*; Morgan et al. 2019). These changes to the jellyfish community continued until 2017. In 2019 in the Gulf of Alaska, Zador et al. (2019) reported the highest-ever catches of Northern sea nettle (*Chrysaora melanaster*) in bottom trawls. This species was also extremely abundant in surface trawls in winter 2019 as far south as lat 52°N, hundreds of kilometers from shore (Pakhomov et al. 2019).

Finally, 2017 should be considered the Year of the Pyrosome in the northeastern Pacific Ocean, because of the enormous biomass of the pelagic colonial tunicate, *Pyrosoma atlanticum*, present throughout the region (Brodeur et al. 2018, Miller et al. 2019). Pyrosomes are common in warm open ocean waters throughout the tropics, but are rare north of Southern California.

Pyrosomes showed a clear increase in abundance from south to north during 2013–18 within the California Current system. At the peak of their distribution in 2017, they were everywhere in truly staggering quantities: from Southern California to the northern Gulf of Alaska at densities of up to 200,000 kg/km^3. By 2019, pyrosomes were effectively absent in waters from Oregon northwards, although they were still present in California.

The ecosystem effects of the pyrosome explosion are unknown, but are expected to be large due to their biomass and widespread distribution (Miller et al. 2019). Pyrosomes have low nutrient content, making them a low-quality, high-fiber prey. Despite this, they were observed in the diets of dozens of species, from sea urchins (Echinoidea) and other demersal invertebrates to rockfishes (*Sebastes* spp.) and other commercial fishes, juvenile and adult Pacific salmon, and fin whales (*Balaenoptera physatus*; Brodeur et al. 2018).

Forage fish and squid

Like lower trophic levels, the abundance and species composition of forage fish and squid have been highly variable since 2014. One species that has apparently expanded its range and abundance is the California market squid (*Doryteuthis opalescens*). Throughout the California Current, squid have been increasing in abundance to the point that substantial commercial fisheries for California market squid have been occurring in Washington and Oregon waters since 2016, reaching the highest commercial catches ever recorded in Oregon in 2020 (>7 million tons). Squid catches have also steadily increased during juvenile salmon surveys off the Washington and Oregon coasts (Morgan et al. 2019).

Other species that have increased in recent years in the California Current include Pacific pompano (*Peprilus simillimus*), adult anchovy (*Engraulis mordax*), some species of lanternfishes (Myctophidae), and both jack (*Trachurus symmetricus*) and Pacific mackerel (*Scomber japonicus*), with sardine and anchovy increases especially prominent in central and southern areas. Species with marked declines include Pacific hake (*Merluccius productus*), juvenile sardine and anchovy, Pacific herring (*Clupea pallasii*), and juvenile salmon (especially in 2017 in the northern California Current). Juvenile rockfish were extremely abundant in the northern California Current and as far north as British Columbia in 2016–18 (Chandler et al. 2017, Boldt et al. 2019, Morgan et al. 2019).

The increase in northern anchovy has been particularly strong in Central and Southern California, where it serves as high-quality prey for many species. Adult anchovy were high in 2018 and the highest ever in 2019 in Central California, and larval anchovies were also the highest in the CalCOFI time series in 2019. While breeding common murres (*Uria aalge*) and rhinoceros auklets (*Cerorhinca monocerata*) were apparently unable to take advantage of plentiful anchovy, California sea lions (*Zalophus californianus*) on the Channel Islands did, resulting in very high counts, weights, and growth rates of California sea lion pups in 2018. Humpback whales (*Megaptera novaeangliae*) were also observed congregating near shore along Central California in 2013–19 while feeding on anchovy schools. High consumption of anchovy by maturing Pacific salmon has been associated with thiamine deficiencies, which negatively impacts the survival of offspring (N. Mantua, SWFSC, unpublished data).

One of the more impressive increases in abundance has been anadromous American shad (*Alosa sapidissima*), an exotic species that was introduced to the U.S. West Coast in the 1800s. Counts of shad over Bonneville Dam, the lowest mainstem dam on the Columbia River, reached 6.0 million fish in 2018, the highest ever, but were even higher in 2019 (7.4 million fish). Shad counts in 2020 at Bonneville Dam declined slightly to 6.2 million fish.

Farther north, the biomass of Pacific herring increased in the Strait of Georgia from approximately 2010–18, with mixed trends in other parts of British Columbia. Northern anchovy have been abundant in the Salish Sea (collectively the Strait of Georgia and Puget Sound) since 2016, consistent with increased abundances in years following elevated coastal temperatures (Duguid et al. 2019). Eulachon (*Thaleichthys pacificus*), which have been declining throughout their range, increased in abundance in the Strait of Georgia in both 2015 and 2018, and in the Columbia River in 2014. Juvenile salmon of all species except chum have also been below average off the west coast of Vancouver Island, while chum salmon have been abundant. The catch of juvenile salmon in 2017 in two widely separated surveys targeting juvenile salmon was the lowest in their respective time series. Catches in Icy Strait (Alaska), which normally consists of juvenile pink, chum, and sockeye salmon, and off the Washington/Oregon coast (spring Chinook and coho salmon), were both extremely low. These surveys are used to forecast adult returns, and therefore predicted poor returns in future years, some of which have transpired (e.g., the extremely low Columbia River spring Chinook salmon return in 2019).

In Alaskan waters, capelin (*Mallotus villosus*) and sand lance (*Ammodytes personatus*) appear to have declined, because they have been low or absent in seabird diets that normally contain them since 2014 and 2015, respectively. Capelin also had decreased abundances in acoustic surveys. By contrast, Pacific herring in the eastern Gulf of Alaska and eastern Bering Sea were above long-term means in recent years.

Salmon survival/returns

The abundance of Pacific salmon populations from California to Alaska, like other guilds or trophic levels described in this section, has shown dramatic changes since 2015. While some populations (especially in northern areas) have returned at record high abundances, others have dropped to new lows. The following summary of recent North American Pacific salmon returns provides context for listed salmon populations reviewed in the previous chapters. Specifically, it demonstrates that unusually high or low returns are not restricted to any one region, species, or production type (hatchery or natural), but were continent-wide. For example, recent low steelhead returns to the Columbia River basin parallel extremely low steelhead returns to the Fraser River basin. In many cases, trends of listed species mirror those of hatchery or mixed (hatchery + natural) populations, indicating the critical role that recent unusual environmental conditions have had on North American Pacific salmon. Unless noted, these abundances come from PFMC (2020) and the Pacific Salmon Commission, Columbia River DART, and Alaska Department of Fish and Game websites.[14]

[14] https://www.psc.org/, http://www.cbr.washington.edu/dart, and https://adfg.alaska.gov/.

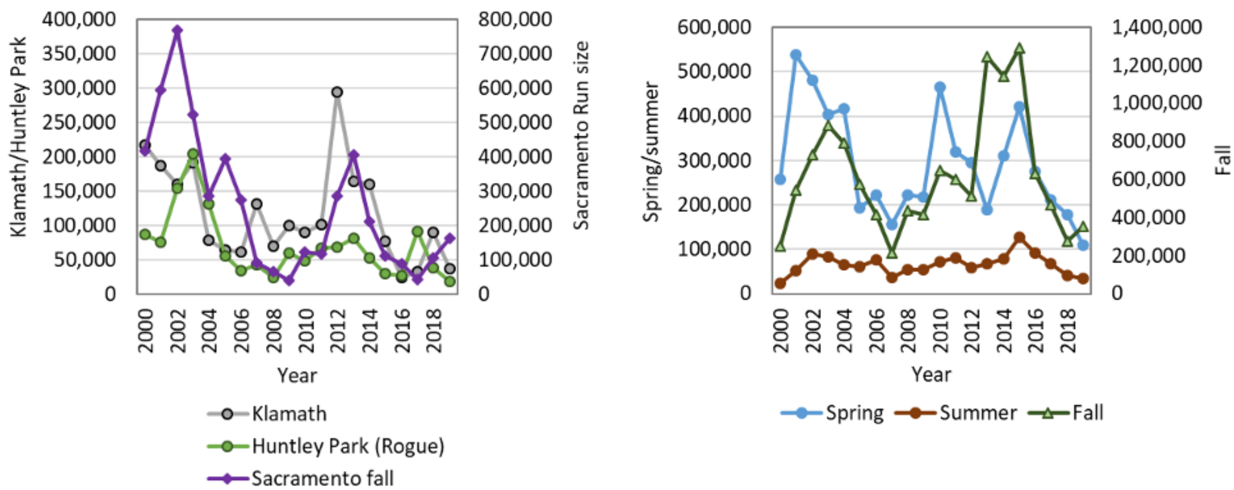

Figure 126. (left) Total escapement of adult Sacramento River fall-run Chinook salmon, total in-river run size of Klamath River fall-run Chinook, and counts of natural fall-run Chinook at Huntley Park (Rogue River). (right) Total in-river run size for Columbia River spring-, summer-, and fall-run Chinook salmon, 2000–19. Data from Council (2020).

The abundance of southern Chinook salmon stocks (Sacramento, Klamath, and Rogue Rivers) has been at very low levels in at least some years since 2014, to the point that several stocks have been declared "overfished" under management regulations. Sacramento River fall-run Chinook salmon have seen large swings in escapement (hatchery + natural), from a high of 406,000 in 2013, steadily declining to 90,000 in 2016, and reaching a low of 43,000 in 2017 (Figure 126). Escapement in 2018 increased to 102,000 and increased again with the 2019 return to 162,000 fish. Total run size of Klamath River Chinook salmon shows a slightly different pattern, with high in river run size in 2012 (295,000), declining to 24,000 in 2016, rebounding to 91,000 in 2018, but declining again to 37,000 in 2019. Indices for south-migrating Oregon coast Chinook salmon (Gold Ray Dam [Rogue River] and Winchester Dam [Umpqua River] counts) also show a steady decline from 2015 to 2019. Peak spawner indices for north-migrating Oregon coast Chinook salmon were highest in 2015 (247 adults/mile) and steadily declined to 2019 (64 adults/mile).

Chinook salmon in the Columbia River have generally been declining since 2015, with details dependent on the year and run (Figure 126). For example, the minimum return of spring Chinook salmon to the Columbia River basin has steadily declined from 2015 (420,000) to 2019 (110,000), one of the lowest levels since the 1990s. Run size for Columbia River summer Chinook salmon has also seen a steady decline from a minimum of 127,000 fish in 2015 to 35,000 in 2019. The minimum run size for Columbia fall Chinook exceeded 1 million fish during 2013–15 and dropped to 275,000 in 2018 (the lowest since 2007), rebounding slightly to 256,000 in 2019.

In Oregon, Washington, and southern British Columbia, several species show consistent patterns, suggesting a common marine cause. For example, coho salmon returns were extremely low in 2015 from the Oregon coast to the Salish Sea, some of the lowest levels on record. The small body size of many of these adults suggested poor feeding conditions during the last summer in marine waters. Steelhead returns were extremely low in 2017 and 2018 in the same

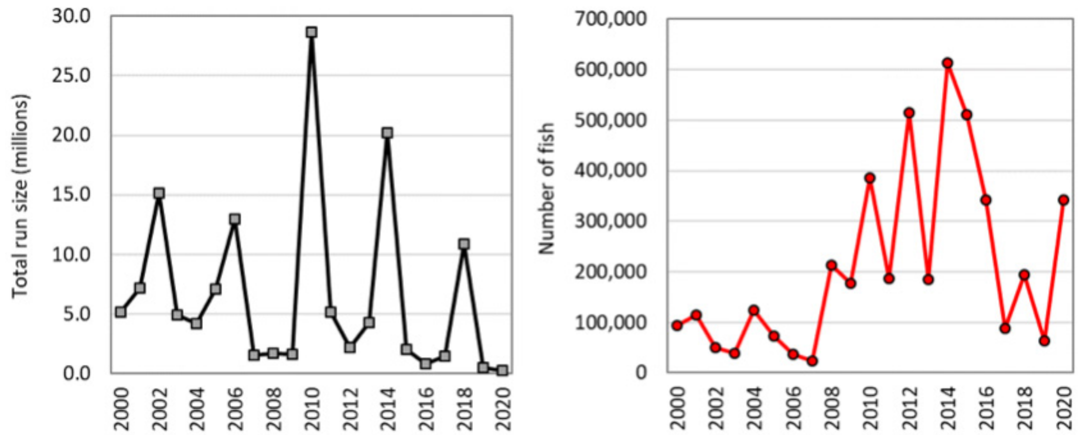

Figure 127. Total run size of Fraser River sockeye salmon (left) and Bonneville Dam counts for Columbia River sockeye salmon (right), 2000–20. Data from https://www.psc.org/ and https://www.fpc.org/fpc_homepage.php.

areas, to the point that Thompson and Chilcotin River (Fraser River, British Columbia) steelhead were petitioned for emergency protection under the Canadian Species At Risk Act (Neilson and Taylor 2018). One species in the region that increased in abundance was chum salmon starting in 2016, perhaps in part due to their reliance on gelatinous prey, which were abundant.

Both Fraser and Columbia River sockeye salmon incurred huge in-river mortalities to returning adults in 2015 due to elevated river temperatures. The following year (2016), Fraser River sockeye salmon had the smallest return on record (total run of <1 million fish), but was even lower in 2019 (500,000 fish), as progeny of the 2015 year class returned as adults (Figure 127). The 2020 Fraser River sockeye salmon return was even lower, at less than 300,000 fish. In contrast, Columbia River sockeye salmon returns were relatively high in 2016 (326,000), below 90,000 in both 2017 and 2019, but reached 341,000 in 2020.

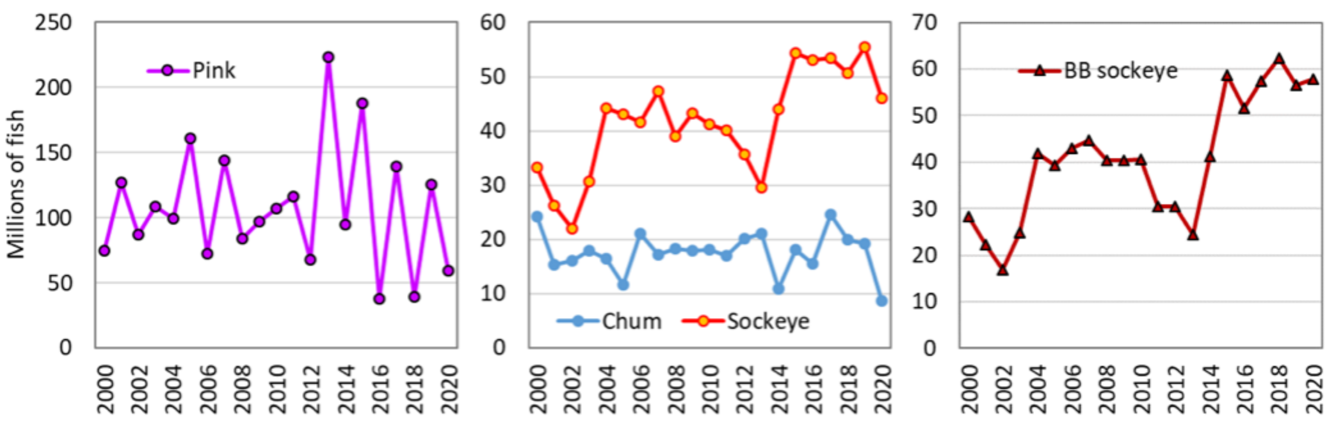

Figure 128. Alaska statewide harvest of pink salmon (left), chum and sockeye salmon (center), and total in-shore run size of Bristol Bay sockeye salmon (right), 2000–20. Note differences in scales on graphs. Data from ADFG and Salomone et al. (2019).

In Alaska, there was a strong east–west gradient in run size, with western Alaska generally having exceptionally high salmon returns, while central and southeastern Alaska saw declines. Perhaps most impressive has been the annual catch of Bristol Bay sockeye salmon during 2015–19, which were among the eight highest years since 1975, including the second (2019) and third highest (2018); see Figure 128 (Salomone et al. 2019). Similarly, 2017 was the highest statewide catch of chum salmon on record, due in part to record Prince William Sound catches. By contrast, pink salmon catches have had both high and low abundances, with 2015 and 2017 having extremely high, and 2016 and 2018 extremely low, catches. 2018 saw the lowest pink salmon return to southeastern Alaska since 1980 (Figure 128).

Other apex fishes

Reports from northern waters suggest changes to the abundance of several apex fishes. Anderson and Workman (2019) report the return of spiny dogfish (*Squalus acanthias*) to west coast Vancouver Island surveys in 2018 after an absence of four years. Arrowtooth flounder (*Atheresthes stomas*) were increasing off Vancouver Island, an increase that extends to the eastern and western Gulf of Alaska and the eastern Bering Sea, after an initial decline due to the heatwave. Sablefish (*Anoplopoma fimbria*) have also been increasing in Alaskan waters. Landings of Pacific hake were extremely high in 2017 and 2018 (Berger et al. 2019).

One species that has not fared well in northern waters is Pacific cod (*Gadus macrocephalus*). The biomass was extremely high in 2015, then crashed. The 2017 Gulf of Alaska survey yielded the lowest biomass of Pacific cod in the time series, down more than 80% since 2013, resulting in the reduction or closure of many Alaskan cod fisheries in 2018. Apex fish predator biomass in the western Aleutian Islands was also the lowest in 2018 since the time series began in 1991, due to declines in both Pacific cod and arrowtooth flounder.

Concurrent with these changes in the abundance of resident fish was the dramatic northward change in the spatial distributions of many fishes and some invertebrates in both 2015 and 2019 in response to warm water. Notable observations included subtropical opah (*Lampris* spp.), billfish (Istiophoriformes), dorado (*Coryphaena hippurus*), and yellowtail jack (*Seriola lalandi*) caught off the Oregon and Washington coasts in both years (E. Schindler, ODFW, personal communication), finescale triggerfish (*Balistes polylepis*) and Louvar fish (*Luvarus imperialis*) off Vancouver Island, and ocean sunfish (*Mola mola*), albacore tuna (*Thunnus alalunga*), Pacific bonito (*Sarda lineolata*), and thresher (*Alopias vulpinus*) and blue sharks (*Prionace glauca*) in Alaska in 2015. There were also tropical sea snakes (Hydrophidae) seen in southern California in 2015, and an invasion of pelagic red crabs (*Pleuroncodes planipes*) that covered beaches in Southern California in 2015 and made it as far north as Newport, Oregon, in 2016.

Seabird productivity

Seabirds consume forage fish that are present at predictable locations and times. Their ability to successfully feed and fledge their chicks (or themselves) is therefore a valuable indicator of the abundance and diversity of forage fish. Measures of chick success have varied widely over the last five years, and depend on the birds' mode of foraging. For example, in the

Semidi Islands archipelago (west of Kodiak, Alaska), surface-feeding black legged kittiwakes (*Rissa tridactyla*) had chick failure in 2019, while diving seabirds on the island had good success in the same season. In general, across reported species and locations, chick success was low in 2015 and 2016, rebounded in 2017 and 2018, and declined again in 2019.

There were also several massive seabird die-offs in response to the 2014–16 northeastern Pacific Ocean marine heatwave. In winter 2014–15, there was a massive die-off of Cassin's auklet (*Ptychoramphus aleuticus*) from northern California to northern Washington. It is estimated that 50,000–100,000 birds died (Coastal Observation and Seabird Survey Team[15]). These birds largely consume krill, and the late Bill Peterson speculated that the warm water prevented the krill from reaching surface waters where the auklets could feed on them (B. Peterson, NWFSC, personal communication). A rigorous analysis suggests that reduced energy content of zooplankton, paired with congregations of birds in a narrow coldwater band along the coast, were to blame for the die-off (Jones et al. 2018).

Another species to suffer a massive die-off was common murres (*Uria aalge*). An estimated 1 million common murres died between summer 2015 and spring 2016. The mortality event affected birds from California to Alaska. Most birds were severely emaciated and, so far, no evidence for anything other than starvation has been found to explain this mass mortality (Piatt et al. 2020). Many colonies also suffered reproductive failure in 2016–17, and another large common murre mortality event occurred along the Washington–Oregon coasts in fall 2019.

Marine mammals

In the California Current, the most obvious impact to marine mammals was the widespread starvation of California sea lion (*Zalophus californianus*) pups in early 2015, resulting in nearly 1,500 malnourished and sick sea lion pups found along California beaches. Strandings in 2015 were the most extreme in the 2013–16 California sea lion unusual mortality event.

Poor feeding conditions in Southern California in 2015 also led to a dramatic increase in the number of California sea lions farther north that summer, especially in the Columbia River, where they fed on returning adult salmon. While the number of California and Steller (*Eumetopias jubatus*) sea lions in the Columbia River (at Bonneville Dam) in the spring has declined since the peak in 2015, the number of Steller sea lions observed at Bonneville Dam and Willamette Falls has been increasing in the fall (Wright et al. 2014, Tidwell et al. 2019). A new Steller sea lion rookery has been established on the northern Washington coast (Carroll Island/Sea Lion Rock complex), with over 100 pups born there in 2015 (Muto et al. 2020)—which, along with a rookery off the north Oregon Coast, are likely sources of increased Steller sea lions in the Columbia River.

Since 2015, there have been two large whale unusual mortality events. The first event occurred in the western Gulf of Alaska and British Columbia in 2015–16 (Savage 2017). A total of 52 whales were reported dead, consisting of 17 fin (*Balaenoptera physalus*) and 34 humpback whales (*Megaptera novaeangliae*). A definitive cause of death could

[15] https://coasst.org/

not be determined, but the event was generally blamed on anomalous physical and biological shifts in the marine environment. The second event was declared for gray whales (*Eschrichtius robustus*) in 2019. This event only affected a single species, but over 250 whales were reported stranded from Mexico to Alaska.

Using Ocean Indicators to Predict Salmon Returns

NWFSC's Annual Salmon Forecast[16] provides annual summaries of ocean indicators based on large-scale physical, regional-scale physical, and local-scale biological data that occur in the year of ocean entry for salmon smolts. This initial summer of ocean life is when most marine mortality is believed to occur; therefore, the indicators describe the conditions experienced by salmon during a critical period. Annual values (columns in Figure 129) for each indicator (rows) have been compared to subsequent Columbia River coho and Chinook salmon returns one and two years later, respectively. The qualitative "stoplight table" rates each indicator in terms of its good, bad, or neutral relative impact on salmon marine survival. Under this system, the best year for that indicator receives a rank of one, while the worst year receives the rank equal to the number of years in the time series (with 2019, 22 years). The stoplight chart thus provides predictions of adult salmon returns one (coho) and two (Chinook) years in advance based on ocean conditions experienced during the first summer of ocean life.

Ocean basin-scale physical indicators include the Pacific Decadal Oscillation (PDO), and the Oceanic Niño Index (ONI), while regional-scale physical indicators include surface- and deep-water salinity and temperature measured on the Newport Hydrographic Line (NHL). Local biological indicators include the composition and abundance of copepods and ichthyoplankton along the NHL, as well as the density of juvenile Chinook and coho salmon caught off the Washington–Oregon coasts during systematic surveys in June. Although the indicators were selected to reflect conditions important for juvenile Chinook and coho salmon survival, they also work for rockfish, sablefish and sardine recruitment (Peterson et al. 2014a), indicating they capture important variability in conditions in the northern California Current.

In general, years that are favorable for salmon survival are characterized by physical conditions that include cold water along the U.S. West Coast before or after the spring outmigration, no El Niño events at the equator, cold and salty water locally (on the NHL), and an early onset of upwelling. Biological conditions that are favorable for salmon survival include lots of lipid-rich (northern) copepods and abundant salmon prey, and high densities of juvenile Chinook and coho salmon caught in the June survey off the Washington–Oregon coasts.

The pattern of indicators over the last 24 years demonstrates that ocean conditions vary from periods when most stoplight chart cells are green (indicating favorable conditions for survival) to periods when most cells are red (indicating unfavorable conditions for survival; Figure 129). For example, the years 1999–02 were largely favorable for salmon survival, 2003–06 were largely unfavorable, etc. While not all cells in the same year are uniformly green (good), red (poor), or yellow (intermediate), in general they show similar patterns. Ocean entry year 1998 was particularly unfavorable, due to the extremely large 1997–98 El Niño event. By contrast, 2008 was the most favorable for salmon survival.

[16] https://www.fisheries.noaa.gov/west-coast/science-data/ocean-ecosystem-indicators-pacific-salmon-marine-survival-northern

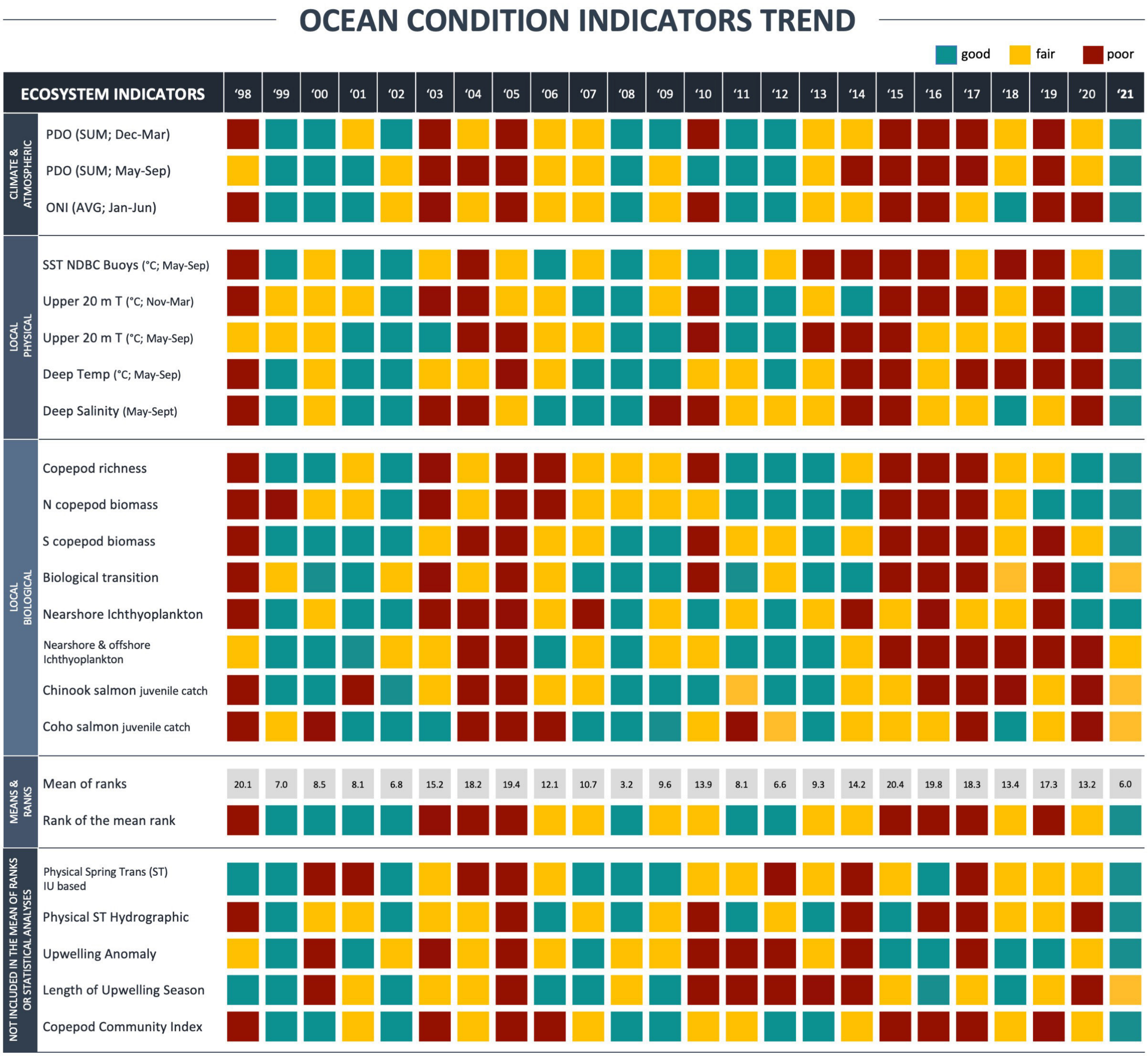

Figure 129. The "stoplight" chart of ocean indicators used to predict salmon returns. Rank scores derived from ocean ecosystem indicator data are color-coded to reflect ocean conditions for salmon growth and survival (green = good, yellow = intermediate, red = poor). The *Mean of ranks* is mean rank scores across all indicators, and provides an overall summary for each year (Peterson et al. 2014a,b).

For purposes of this document, the stoplight chart indicates that, beginning with ocean entry year 2015, conditions in the northern California Current have been mostly unfavorable for ocean survival. Exceptions include 2018 and 2020, which were average, and 2021, which was better than average. The mean rank of indicators during 2015–19 contained four of the seven worst years in the 22-year time series, including the worst overall rank in 2015.

These unfavorable conditions correspond to adult Chinook salmon that returned in 2017–21, and to adult coho salmon that returned in 2016–20. Clearly, both physical and biological ocean conditions have been unfavorable for juvenile salmon survival in the northern California Current, and, by extension, to other areas of the California Current as well.

Climate Vulnerability Assessment

As trends progress toward warmer oceans and streams, more extreme winter flood events, summer low flows, loss of snowpack in the mountains, and ocean acidification, salmon face increasing challenges. Though all salmon share some similarities in their vulnerabilities, particular life-history types, geographic locations, and histories of anthropogenic stress cause some ESUs/DPSes to be especially vulnerable to climate change. The climate vulnerability assessment of Pacific salmon and steelhead (Crozier et al. 2019b) was conducted to characterize this variability.

Crozier et al. (2019b) analyzed the exposure factors in freshwater and the ocean, sensitivity factors by life stage and population characteristics, and adaptive capacity for all listed ESUs/DPSes in the West Coast Region. Using a methodology developed for all NOAA science centers, an expert panel scored each ESU/DPS for each factor, and applied a logic rule to rank overall exposure, sensitivity, and vulnerability. Crozier et al. (2019b) provided a discussion of the scores for each ESU/DPS, which we have attached as a supplement to this document for greater detail on expected impacts.

Here we briefly summarize the main results of that analysis for the northern part of the West Coast Region, as shown in Figure 130. Interior Columbia River sockeye and spring Chinook salmon, as well as Willamette River Chinook salmon, ranked in the highest risk category. A combination of juvenile and adult life-history characteristics put these ESUs at higher risk. Specifically, their prolonged freshwater rearing, migration, and holding during warm summer months increased their exposure to rising summer stream temperatures and loss of snowpack, which affects their migration. The adaptive capacity of these populations is limited by their relatively low population viability due to anthropogenic impacts (especially endangered Snake River sockeye salmon). These particular ESUs face analogous pressures to those at the southern edge of the range, which are more classically assumed to be the most vulnerable to climate change (all Central Valley Chinook salmon ESUs, and the two southernmost coho salmon ESUs were also ranked as highly vulnerable).

Interior and southern steelhead face similar exposures to climate factors as interior and southern Chinook and coho salmon, but they have demonstrated greater adaptive capacity and flexibility in life histories, which reduced their rankings somewhat. Coastal and Puget Sound

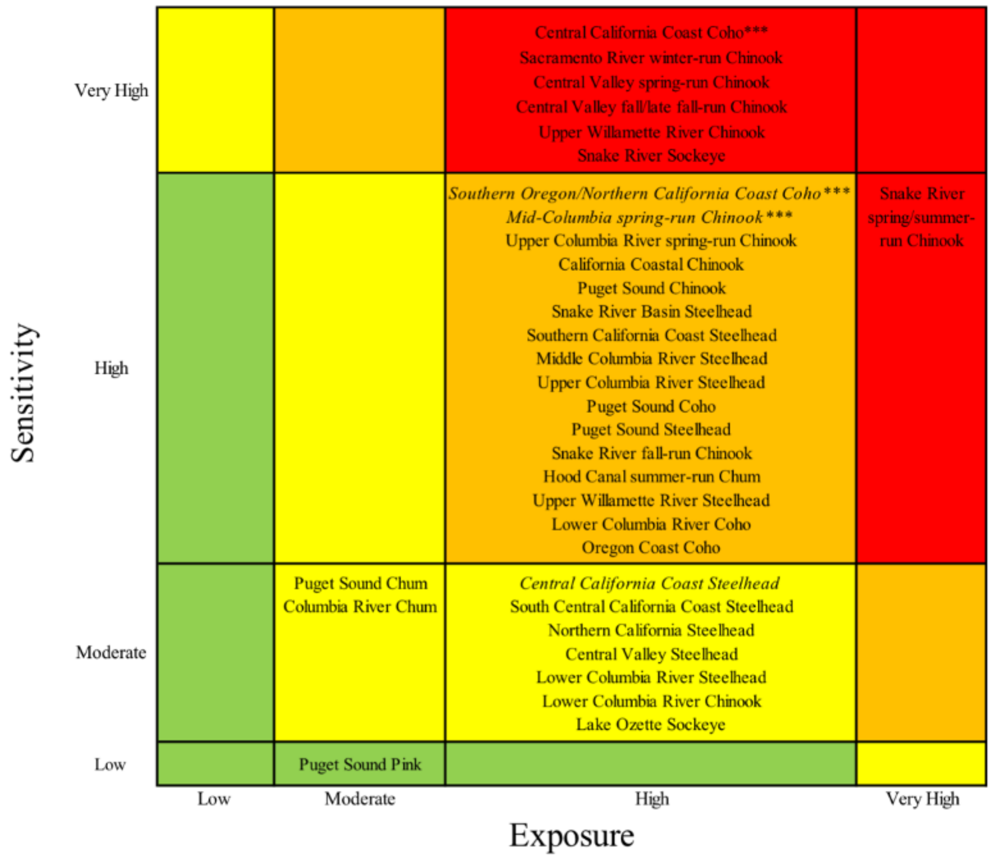

Figure 130. Climate vulnerability categories for each ESU/DPS were a product of sensitivity and exposure scores: red indicates very high vulnerability, orange high, yellow moderate, and green low. Uncertainty in final ranks was represented with a bootstrap analysis. Borderline ESUs/DPSes were those that placed in a higher rank in at least 25% of resampled data. Borderline sensitivity ranks are shown in italics, and borderline exposure ranks indicated with asterisks (*). All other cumulative vulnerability ranks were considered likely. Figure courtesy of Crozier et al. (2019b).

Chinook and coho salmon and steelhead, and Snake River fall Chinook salmon, scored relatively high in adaptive capacity, largely due to their greater flexibility in migration and spawn timing. Nonetheless, some of this diversity within these ESUs/DPSes may be lost due to climate change.

Pink and chum salmon generally appeared to have lower vulnerability because of their short freshwater residence, and their early life stages may be more resilient to shifts in ocean ecosystems (e.g., chum salmon tend to consume a larger fraction of gelatinous prey, which may become more abundant with climate change). However, information on the sensitivity of survival during the marine stage to ocean acidification and other potential ecosystem changes was highly uncertain. Thus, these relative rankings primarily reflect sensitivities during freshwater life stages. To see the scores in all sensitivity and exposure factors and a narrative describing particular risks faced by each ESU/DPS, see Crozier et al. (2019b), Appendix S3, included in this document as Supplement 1.

Assessment of Freshwater Restoration

Restoration and protection of habitat is widely applied as a long-term strategy for meeting salmon conservation and recovery goals, redressing "...present or threatened destruction, modification, or curtailment of species' habitat or range"(Section 4(a)(1)(A), ESA, Listing Factor D). Over the last 20+ years, habitat restoration projects have been planned and executed to improve freshwater rearing, spawning, and migrating habitat for salmonids. Recovery plans specify the management actions needed to meet recovery and habitat restoration goals for each Evolutionarily Significant Unit or Distinct Population Segment. To tie management actions to ecological needs, we compiled habitat project and ecological needs data to provide a way to show, over time, how projects are addressing habitat across the broad scale of Pacific Northwest salmon recovery.

We read each ESU/DPS recovery plan for mentions of habitat concerns (e.g., limiting factors, impaired habitat, etc.). Habitat concerns described as "primary" or "major" within a population were assumed to capture the most pertinent problems. The recovery plans cover 275 populations nested within 18 ESUs/DPSes, and were finalized from 2000 to the present. The habitat concerns in recovery plans were translated into a standardized form using the ecological concerns data dictionary, developed to classify degraded salmon habitat in the Pacific Northwest (Hamm 2012). The ecological concerns data dictionary standardizes commonly used terms to describe habitat degradation—e.g., limiting factors, problems, concerns, impacts, etc. The data dictionary enabled conversion of static text documents into a standardized, queryable database of ecological concerns compatible with Geographical Information Systems (GIS).

The Pacific Northwest Salmon Habitat Project database (PNSHP)[17] served as the source for completed habitat restoration projects (Katz et al. 2007). A restoration project was defined as any action involving physical changes to freshwater or estuarine habitat. All restoration project records contain a type and subtype as well as a spatial location, enabling sampling at any scale using GIS. The queries included projects completed in the 27 years since the first Pacific salmon ESA listing (1992–2018).

Restoration projects were spatially queried for each population within an ESU/DPS, then compared to ecological concerns gathered from recovery plans. Comparing restoration to ecological need at the population scale allows the calculation of both the percentage of projects matching or failing to match the ecological concerns, and the number of ecological concerns that are addressed by at least one project. We created a metric, the Salmon Habitat Assessment and Project Evaluator (SHAPE), which is calculated as the proportion of ecological concerns addressed, minus the proportion of projects without any match within an assessment unit.

In this hypothetical example population (Table 68), the recovery plan calls out seven of the nine ecological concerns as problems. Six of the seven ecological concern (EC) categories have one or more projects addressing them, so are considered a match (YES answers). One recovery plan EC category, Water Quality, has no restoration projects addressing it. Three restoration projects address Habitat Quantity, though Habitat Quantity was not called out in the recovery plan as a major ecological concern.

[17] https://www.webapps.nwfsc.noaa.gov/apex/f?p=409:13::::::

Table 68. Example population comparison of recovery plan ecological concerns and projects that target those types of ecological concern.

	Habitat quantity	Injury & mortality	Food	Riparian condition	Peripheral & transitional habitats	Channel structure & form	Sediment conditions	Water quality	Water quantity
Recovery plan	NO	YES	NO	YES	YES	YES	YES	YES	YES
Project	YES (3 projects)	YES (1)	NO	YES (8)	YES (10)	YES (5)	YES (15)	NO	YES (3)

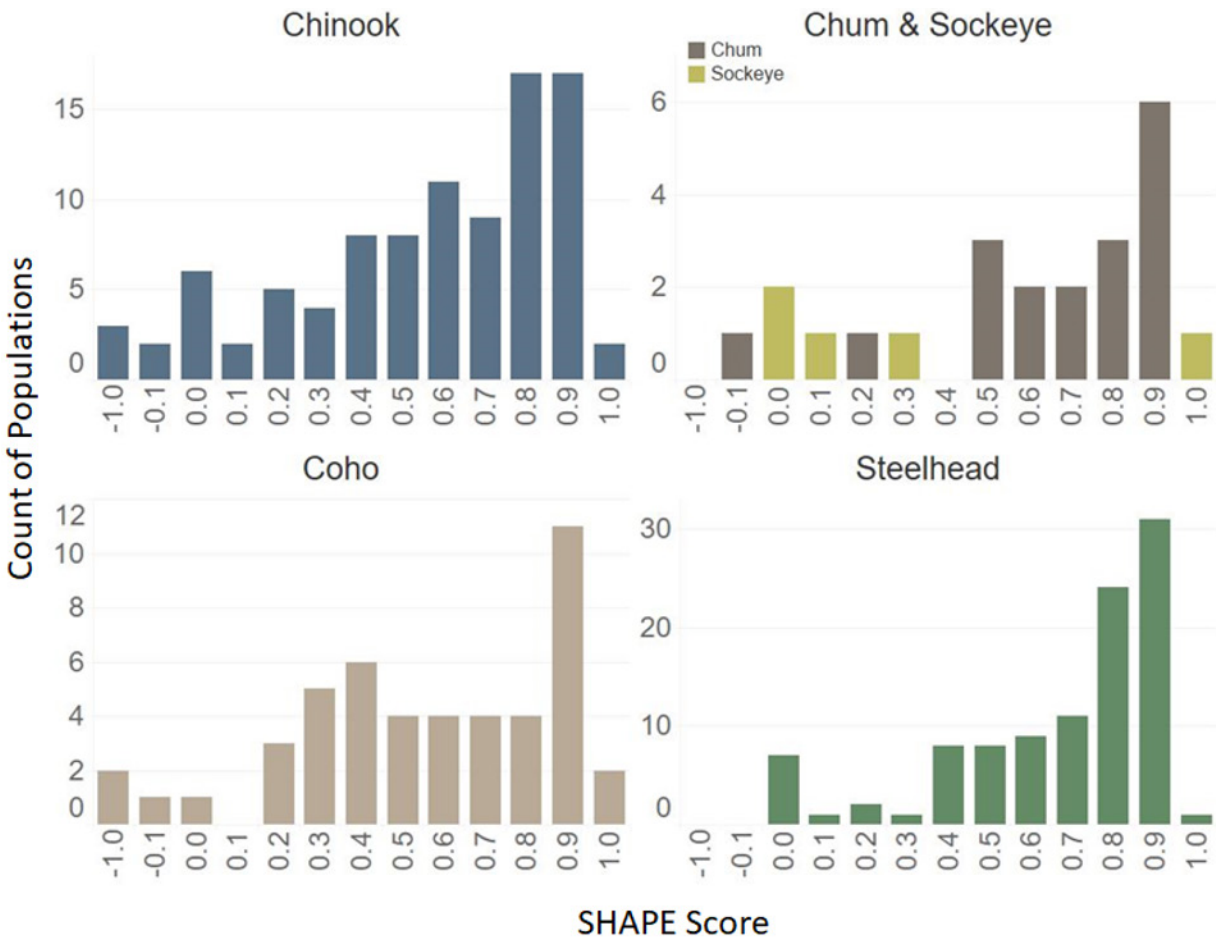

Figure 131. Distribution of population SHAPE scores, by species.

Using our metric (SHAPE = (ECs Addressed / Total ECs) – (Projects not addressing ECs / Total Projects)), the example population in Table 68 scores 0.785 (SHAPE = (6 / 7) – (3 / 42) = 0.785). Population scores range from 1 (meaning all ECss called out in the recovery plan have one or more restoration projects that target them) to –1 (total mismatch, e.g., ECs have no restoration projects that treat them, or there are no ECs, but are projects). If there are no restoration projects within a population spatial unit, then the SHAPE score defaults to 0.

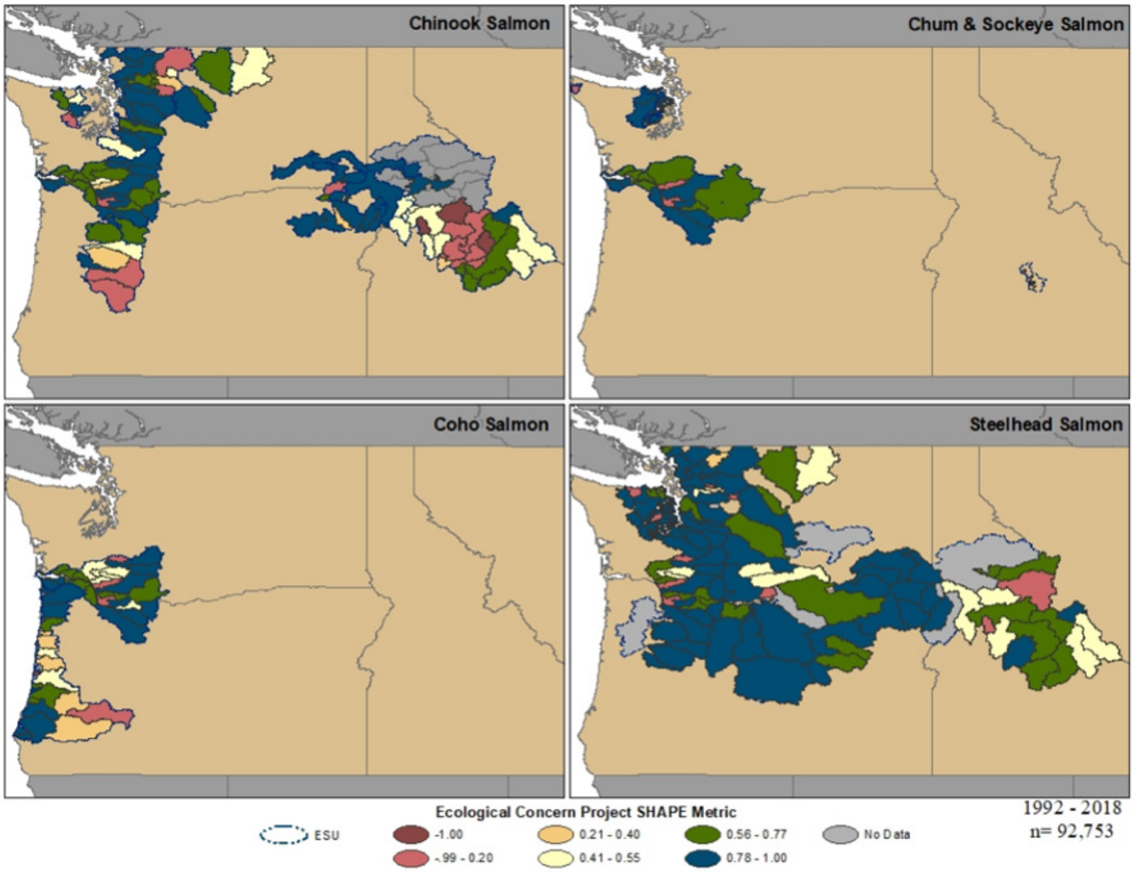

Figure 132. Spatial distribution of SHAPE score across all populations.

Populations with scores closer to 1 have better agreement between types of ecological concern and the projects that have been completed. However, a high score does not signify that all restoration has been completed. Populations with scores below 0.5 (colored in red, orange, and yellow in Figure 132) identify places to focus effort on ecological concerns not yet treated by projects, or project types implemented that do not address ECs identified in the recovery plan. In aggregate, steelhead populations had the highest SHAPE scores of any species, signaling a good match between habitat need and project types implemented (Figures 131 and 132).

The SHAPE metric is designed as a tool to help inform decision-making and restoration project planning. This approach does not take into account socioeconomic factors which also influence project type choice and project placement. Moving forward, SHAPE scores can be used to show progress, in five-year increments, toward alignment between restoration and habitat needs for salmon.

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Supplement 1: Climate Vulnerability Analysis

Crozier et al. (2019b) recently analyzed the vulnerability of all Pacific salmon and steelhead ESUs and DPSes to climate change. Links to this open access paper and its supplement, providing summaries for each ESU/DPS, are provided here:

Main paper: https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0217711

ESU/DPS summaries: https://doi.org/10.1371/journal.pone.0217711.s003

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