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Biological Status of the Washington Coast Chinook Salmon Evolutionarily Significant Unit: Report of the Status Review Team

May 2025

U.S. DEPARTMENT OF COMMERCE

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Northwest Fisheries Science Center

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Biological Status of the Washington Coast Chinook Salmon Evolutionarily Significant Unit: Report of the Status Review Team

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National Oceanic and Atmospheric Administration
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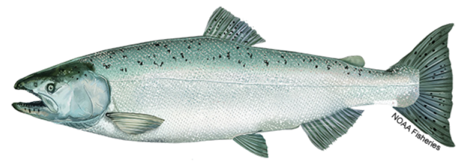
Background

Chinook salmon, also known as king, spring, quinnat, Sacramento, California, or tyee salmon, are the largest of the Pacific salmon. The species distribution historically ranged from the Ventura River in California to Point Hope, Alaska, in North America, and from Hokkaido, Japan, to the Anadyr River, Russia, in northeastern Asia.



Adult Chinook salmon die after they lay their eggs in streams. Young salmon will rear in freshwater systems for a few weeks up to two years, then migrate to the ocean for an additional one or five years before returning to the stream they were born in to spawn. Chinook salmon populations generally either stay close to shore or migrate to the middle of the ocean; coastally oriented populations are subject to considerable fisheries harvest. Populations are commonly categorized according to the timing of their return to freshwater to spawn, with five run-times identified: winter, spring, summer, fall, and late fall. Spring- and summer-run populations return to freshwater several weeks to months in advance of spawning, while fall-, late fall-, and winter-run populations spawn soon after returning to freshwater.

Chinook salmon along the U.S. West Coast have been organized into evolutionarily significant units (ESUs), based on similarities in their life-history characteristics, genetic composition, and the ecology of their rivers. When NOAA Fisheries considers Pacific salmon for listing as threatened or endangered under the Endangered Species Act (ESA), it does so in terms of ESUs . The Washington Coast (WC) Chinook Salmon ESU extends from west of the Elwha River to north of the mouth of the Columbia River. The WC Chinook Salmon ESU includes spring-, summer-, and fall-run fish. Fall-run Chinook salmon can be found in both small and large rivers, while the spring and summer runs are mostly found in larger rivers. In 1999, NOAA Fisheries determined that the ESU did not warrant listing as threatened or endangered under the ESA.

In July 2023, NOAA Fisheries was petitioned by the Center for Biological Diversity and Pacific Rivers ("the Petitioners") to reevaluate the status of WC Chinook salmon. The Petitioners requested that spring-run Chinook salmon in the WC be considered as a separate ESU and listed as threatened or endangered. The Petitioners assert that spring-run Chinook salmon are facing existential threats. Therefore, they requested that, if NOAA Fisheries does not list the spring-run WC Chinook salmon populations as a uniquely threatened or endangered ESU, then the current WC Chinook salmon ESU—which includes spring-, summer-, and fall-run populations—should be listed as threatened or endangered under the ESA. NOAA Fisheries agreed to a reexamination of the status of WC Chinook salmon.

NOAA Fisheries formed a Status Review Team (SRT) to gather and review all available information to complete five specific tasks:

1. Evaluate the current boundaries and makeup of the WC Chinook Salmon ESU.
2. Complete a demographic risk analysis.
3. Review and comment on a threats analysis.
4. Complete the extinction risk analysis, considering demographic risks and threats.
5. If necessary, conduct a separate analysis to evaluate whether the ESU is at moderate or high risk of extinction in a significant portion of its range.

Key Takeaways

The SRT gathered recent information on the WC populations, much of it from federal, state and tribal biologists. Information related to the genetic and life-history characteristics of populations in the WC Chinook Salmon ESU was compared with neighboring ESUs. Recent abundance data were compared with data gathered at the time of the last status review in 1998. Information on the current environmental condition of the land and rivers in the range of the WC Chinook Salmon ESU was also considered, as were predictions of the effects of climate change. After reviewing and discussing the information gathered, the SRT concluded:

- Existing information supports the composition of the existing ESU; spring-, summer-, and fall-runs of Chinook salmon are closely related to one another.
- Information is available for the majority of Chinook salmon populations in the ESU, and most of those populations have exhibited a stable or positive abundance trend in the last 15 years, and generally since the last review.
- Total annual ESU spawning abundance is likely between 40,000 and 50,000 fish.
- Harvest rates have been quite high for most fall-run populations and slightly lower for spring- and summer-run populations, but spawner abundances appear stable, suggesting these populations are reproducing well.
- A number of hatchery programs operate in the ESU, some quite large. Most of them release fish to provide harvest opportunities.
- Harmful hatchery effects have been reduced to some extent through the use of locally derived hatchery broodstocks, reduction or elimination of away-from-hatchery releases, and controlling the use of natural-origin adults as broodstock.
- Habitat conditions are good in Olympic National Park. Outside of the park, where there has been intensive tree cutting, much of the forest and river habitat is slowly recovering. In the southern portion of the ESU, land development has had a greater negative effect on stream and streamside habitats. Climate change is likely to result in summertime river temperatures being too warm, with lower river flows. Winter snowfall will transition to rain in the future, often in the form of major rainfall events. The loss of glaciers in the Olympic Mountains is already proceeding at a rapid rate, and it is likely that these glaciers will be gone completely by the end of this century.

The SRT members unanimously concluded that the WC Chinook Salmon ESU is most likely at a low risk of extinction.

Links used in this section:

- Chinook salmon: https://www.fisheries.noaa.gov/species/chinook-salmon
- Evolutionarily significant units and the Endangered Species Act: https://www.fisheries.noaa.gov/laws-and-policies/glossary-endangered-species-act
- Petitioned: https://www.fisheries.noaa.gov/action/petition-list-washington-coast-chinook-salmon-threatened-or-endangered-under-esa
- Threats that Chinook salmon face: https://www.fisheries.noaa.gov/species/pacific-salmon-and-steelhead/esa-protected-species
- Climate change: https://www.fisheries.noaa.gov/topic/climate-change/understanding-the-impacts

Executive Summary

On 17 July 2023, the Center for Biological Diversity and Pacific Rivers (hereafter, "the Petitioners") submitted a petition to the U.S. Secretary of Commerce to list the spring-run Chinook salmon (*Oncorhynchus tshawytscha*) on the Washington coast (WC) as a threatened or endangered evolutionarily significant unit (ESU) under the Endangered Species Act (ESA) or, alternatively, to list WC Chinook salmon (inclusive of all run types) as a threatened or endangered ESU. The Petitioners also requested the designation of critical habitat concurrent with ESA listing. In 1999, the National Marine Fisheries Service (NMFS) had identified the WC Chinook salmon ESU as comprising populations of spring-, summer-, and fall-run Chinook salmon spawning north of the Columbia River and west of the Elwha River, and had determined that the ESU did not warrant listing as threatened or endangered under the ESA (USOFR 1999b).

In response to the petition, the Northwest Fisheries Science Center (NWFSC) convened a Status Review Team (SRT) in 2024, consisting of scientists from NWFSC and the NMFS West Coast Region (WCR). The SRT reviewed information relevant to the configuration (boundaries) and risk of extinction for this ESU, including the biological and demographic status of Chinook salmon, past and current hatchery operations, watershed habitat conditions, past and present fisheries, land use and associated regulations, and the effects of climate change. In developing their risk analysis, the SRT utilized information from published sources (peer-reviewed articles and agency and tribal reports), written information submitted by state and tribal co-managers, personal knowledge, and traditional ecological knowledge (TEK) provided to the SRT during and after technical meetings with state and tribal organizations. The SRT met several times (virtually) with representatives from the Washington Department of Fish and Wildlife (WDFW), the Northwest Indian Fisheries Commission (NWIFC), and tribal nations within the WC Chinook Salmon ESU or with treaty/management interests within the ESU.

The first task of the SRT was to review the configuration of the ESU as it was originally defined by Myers et al. (1998). Most of the new information related to ESU configuration consisted of genetic studies published since 1998 that included Chinook salmon populations within the WC Chinook Salmon ESU and adjacent basins. The team reviewed this information and concluded that the available data do not support an alternative ESU configuration. Specific to the issue of the distinctiveness of spring-run Chinook salmon raised by the petitioners, the team evaluated the available information and concluded that the spring run in the ESU does not meet either prong of the NMFS ESU policy (Waples 1991). In particular, the team found that the best available information demonstrates that WC spring-run populations are not substantially reproductively isolated from WC fall-run populations, and that WC spring-run populations are not a significant component of the evolutionary legacy of the species as a whole.

Following a review and discussion of the information available, SRT members evaluated the viability of individual Chinook salmon populations in the ESU using the four viable salmonid population (VSP) categories: abundance, productivity, spatial structure, and diversity (McElhany et al. 2000, Hard et al. 2015). Further, SRT members also estimated

the relative effect of the ESA Section 4(a)(1) listing factors for decline: habitat loss and destruction, over-utilization, disease and predation, inadequacy of existing regulatory mechanisms, hatchery effects, and climate change. The threat from each of these factors was rated 1–5 (1 = very low risk/threat, 5 = very high risk/threat). Individual population assessments for VSP parameters and threats (ESA Section 4(a)(1)) provided a basis for assessing the overall risk of extinction to the ESU and subsequent evaluations of risk to significant portions of the range (SPR). Overall for the ESU, the average viability scores ranged from 1.3–1.6, indicating a conclusion of a low-risk status. For threats, average scores by threat ranged from 1.4–2.1, with only climate change having an average above 2.

In quantifying the extinction risk to the ESU, each SRT member allocated ten risk points to three categories: low, medium, and/or high risk. This methodology has been adapted by previous SRTs and provides a measure of overall risk and the certainty underlying that risk assessment. The team unanimously concluded that the WC Chinook Salmon ESU was most likely to be at low risk of extinction, with individual team members placing between six and ten (out of ten) likelihood points in the low-risk category. Across all members, the average risk assignments for each risk category were 8.4 (1.4 standard deviation [SD]) for low risk, 1.6 (1.3 SD) for moderate risk, and 0.0 (0.0 SD) for high risk.

Following an assessment of the ESU risk, it was necessary to determine if there is a significant portion of the range of the ESU that is at a higher risk level than the entire ESU. A "significant portion of the range" is defined in the 2014 Joint U.S. Fish and Wildlife and NOAA SPR policy (USOFR 2014) as:

> A portion of the range of a species is "significant" if the species is not currently endangered or threatened throughout its range, but the portion's contribution to the viability of the species is so important that, without the members in that portion, the species would be in danger of extinction, or likely to become so in the foreseeable future, throughout all of its range.

The SRT discussed multiple scenarios for identifying "portions" based on geography or on the range of different biological (life-history) types. The team approached the SPR question by first looking across the individual populations for areas with higher-than-typical risk. Note that "higher" does not necessarily imply that these are at moderate or high risk; only that they are at higher-than-average risk for the ESU as a whole. The team identified four such areas:

1. The northern coast and the Strait of Juan de Fuca. Many of these populations are characterized by small population sizes, occupy small watersheds, and are under substantial hatchery influence—factors that all indicate that these populations are at higher risk than is typical for the ESU as a whole.
2. Southern coastal areas of the ESU, including Willapa Bay. These areas are characterized by lower-gradient streams that are likely more susceptible to warming temperatures predicted by future climate change. These areas are also largely in private land ownership, with a greater present level of, and future potential for, land development and habitat degradation compared to areas that are protected in Olympic National Park or other public lands.

3. The Upper Chehalis River basin. This basin includes both spring- and fall-run populations of Chinook salmon. In contrast to other basins in the ESU, the Upper Chehalis River drains the lower-elevation Willapa Hills, rather than the Olympic Mountains. As such, this basin is more vulnerable to climate change, especially the spring-run population.
4. Early-returning spring- and summer-run populations were thought to be at a higher risk due to their relatively low abundances, limited headwater spawning habitat, and the vulnerability of these habitats to ongoing and future climate change that decreases summer low flows and increases summer stream temperatures.

A review of the risk scores for each of these portions indicated that for the north coast and Strait of Juan de Fuca, while individual populations were at somewhat higher risk of extinction than the ESU overall, the average VSP score for this region (1.4) was actually lower than the ESU average (1.5). Populations in Willapa Bay did exhibit a higher average VSP risk score (1.8) relative to the entire ESU (1.5); however, the team did not consider that the Willapa River tributaries alone constituted a significant portion, and the risk level was still considered to be low. In the SPR assessment of the early-returning populations portions (areas 3 and 4 above), the team concluded that, although the early run may potentially represent an important part of the ESU, it had an average VSP risk score of 1.6 (between "very low" and "low") and is not currently at moderate or high risk of extinction. The team therefore concluded that the Washington Coast Chinook Salmon ESU is not at risk in a significant portion of its range.

Abbreviations

CFS	cubic feet per second
CMS	cubic meters per second
CTC	Chinook salmon Technical Committee
DIP	demographically independent population
DOC	U.S. Department of Commerce
ESA	U.S. Endangered Species Act
ESU	evolutionarily significant unit
HSRG	Hatchery Science Review Group
IP	intrinsic potential
IPM	integral projection model
MARSS	multivariate autoregressive state-space
NFH	National Fish Hatchery
NMFS	National Marine Fisheries Service (NOAA)
NOAA	National Oceanic and Atmospheric Administration (DOC)
NPS	National Park Service
NWIFC	Northwest Indian Fisheries Commission
NWFSC	Northwest Fisheries Science Center (NMFS)
ONP	Olympic National Park
OP	Olympic Peninsula (Washington)
PST	Pacific Salmon Treaty
PVA	population viability analysis
QET	quasi-extinction threshold
SaSI	salmonid stock inventory
SPR	significant portion of its range
SRT	status review team
USFWS	U.S. Fish and Wildlife Service
VSP	viable salmonid population
WC	Washington coast (i.e., WC Chinook Salmon ESU)
WCR	West Coast Region (NMFS)
WDFW	Washington Department of Fish and Wildlife
WRIA	water resource inventory area

Introduction

On 17 July 2023, the Center for Biological Diversity and Pacific Rivers (hereafter, "the Petitioners") submitted a petition to the U.S. Secretary of Commerce to list the spring-run Chinook salmon (*Oncorhynchus tshawytscha*) on the Washington coast (WC) as a threatened or endangered evolutionarily significant unit (ESU) under the Endangered Species Act (ESA) or, alternatively, to list WC Chinook salmon (inclusive of all run types) as a threatened or endangered ESU. The Petitioners also requested the designation of critical habitat concurrent with ESA listing. In 1999, the National Marine Fisheries Service (NMFS) had identified the WC Chinook salmon ESU as comprising populations of spring-, summer-, and fall-run Chinook salmon spawning north of the Columbia River and west of the Elwha River, and had determined that the ESU did not warrant listing as threatened or endangered under the ESA (USOFR 1999b).

The Petitioners requested that spring-run Chinook salmon in the WC Chinook Salmon ESU be considered a separate ESU and listed as threatened or endangered. The petitioners based their request on recent research into the genomic basis for premature and mature adult migration in salmonids. The Petitioners argue that these studies indicate that WC spring-run Chinook salmon are reproductively isolated and evolutionarily significant, and therefore should be considered a separate ESU based on NMFS policy. The petition includes an overview of new research into the genomic basis for migration timing in salmonids, as well as general biological information about spring-run Chinook salmon in the WC, including their distribution and range, life-history characteristics, and habitat requirements, as well as basin-level population status, trends, and factors contributing to population status. The majority of biological (e.g., life-history, genetics, ecological) and abundance information presented pertains to the spring-run populations, with a very limited review of information related to fall-run populations. In the petition, fall-run population information is usually presented in the context of providing a surrogate for missing information on the spring run. Although the threats analysis is equally relevant to both spring-, summer-, and fall-run populations, the Petitioners assert that spring-run Chinook salmon are facing existential threats, and therefore, if NMFS does not delineate and list WC spring-run Chinook salmon populations as threatened or endangered under the ESA, the current WC Chinook Salmon ESU that includes spring-, summer-, and fall-run populations should be listed as threatened or endangered under the ESA.

On 7 December 2023, NMFS published a 90-Day finding that the petition provided sufficient information to indicate that listing may be warranted (USOFR 2023). In preparation for an evaluation of the petition, the NMFS West Coast Region (WCR) requested that the Northwest Fisheries Science Center (NWFSC) convene a status review team (SRT) to reassess the configuration and status of this ESU.[1]

[1] Letter from Jennifer Quan, WCR Regional Administrator, to Nicolle Hill, NWFSC Deputy Science Director. 5 December 2023. Science Center Support for the Status Review of the Washington Coast Chinook Salmon.

The SRT was convened with five specific tasks:

1. To evaluate the WC Chinook Salmon ESU configuration.
2. To complete a demographic risk analysis.
3. To review and comment on a threats analysis to be compiled by WCR.
4. To complete the extinction risk analysis, considering demographic risks and threats.
5. If necessary, to conduct a significant portion of the range (SPR) analysis, depending on the outcome of Task 4, to evaluate whether the ESU is at moderate or high risk of extinction in a significant portion of its range.

Previous Determinations

In 1999, NMFS identified the WC Chinook Salmon ESU (Figure 1) as comprising coastal populations of spring-, summer-, and fall-run Chinook salmon spawning north of the Columbia River and west of the Elwha River, and determined that the ESU did not warrant listing as threatened or endangered under the ESA (USOFR 1999b). The designation of the ESU and the risk determination were based on an analysis by the Chinook salmon coastwide Status Review Team (SRT; Myers et al. 1998). In their status review, the SRT established the Washington Coast ESU as including Chinook salmon populations:

> …[in] coastal basins north of the mouth of the Columbia River to, but not including, the Elwha River. This ESU includes fall, summer, and spring runs of chinook. These fish exhibit an ocean-type life history (as do all coastal stocks in Washington, Oregon, and California), but their marine distribution and age structure differs considerably from fish in the Puget Sound (ESU 8) and Lower Columbia River (ESU 9) ESUs. Fish in this ESU generally mature at 3-, 4-, and 5-years-old and migrate in a northerly direction to British Columbian and Alaskan coastal waters (Myers et al. 1998, p. xviii).

Subsequent to the identification of the WC Chinook Salmon ESU, the 1998 SRT assessment of extinction risk concluded that:

> Long-term trends for most populations in this ESU have been upward; however, several smaller populations are experiencing sharply downward trends. Fall-run populations are predominant and tended to be at a lower risk than spring or summer runs. Hatchery production is significant in the southern portion of this ESU, whereas the majority of the populations in the northern portion of the ESU have minimal hatchery influence. The SRT unanimously concluded that Chinook salmon in this ESU are not in danger of extinction nor are they likely to become so in the foreseeable future (Myers et al. 1998, p. xxiii).

This document reports on the results of a comprehensive status review of WC Chinook salmon, starting with a review of the ESU as it was defined by Myers et al. (1998). To provide a context for evaluating these populations of Chinook salmon, biological and ecological information for Chinook salmon in adjoining ESUs (e.g., Puget Sound, Oregon Coast) was

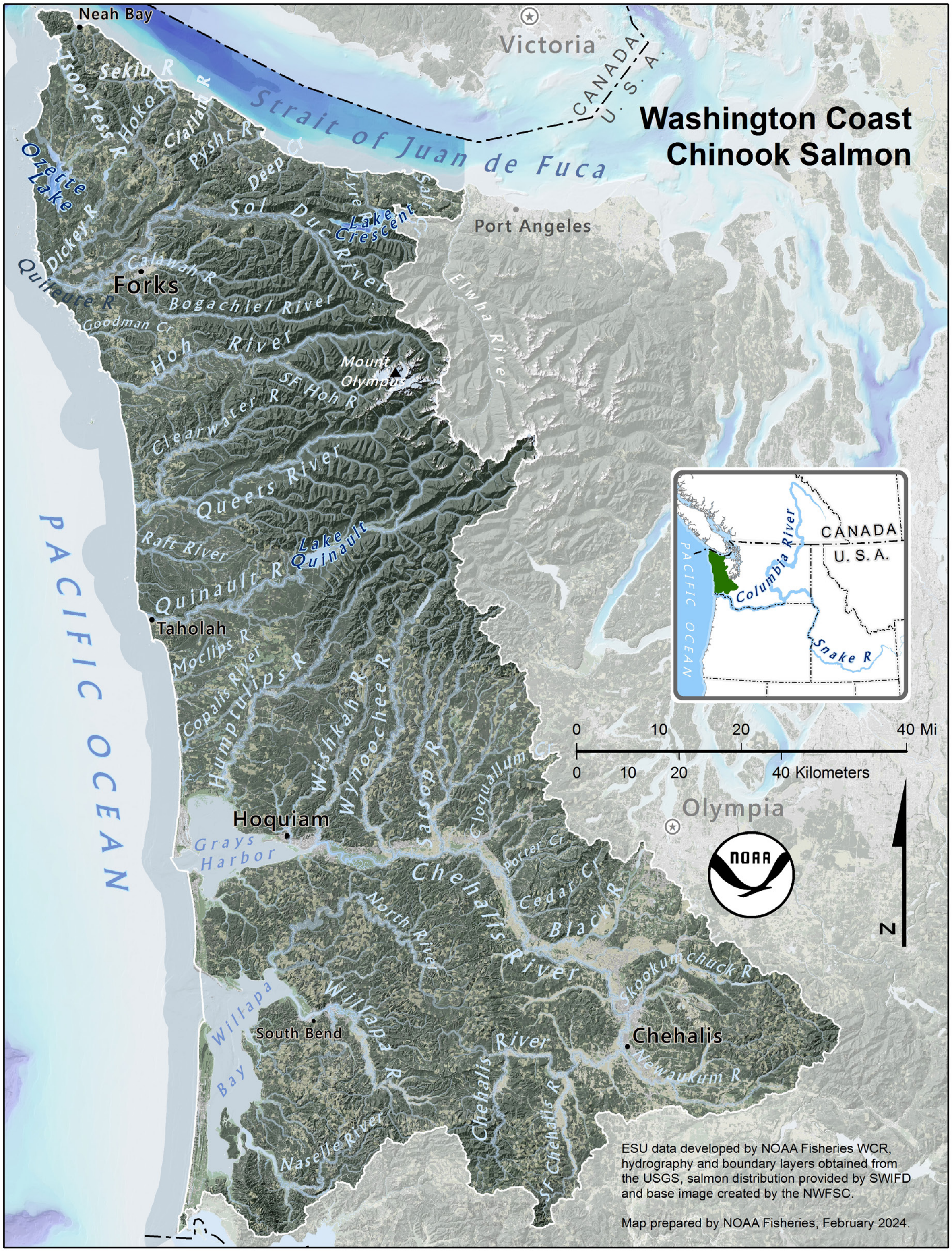


Figure 1. Map showing the freshwater extent of the Washington Coast Chinook Salmon ESU as defined by Myers et al. (1998), including major rivers and geographic features. Base imagery prepared by NWFSC.

also considered. This review thus encompasses, but is not restricted to, the populations identified by the Petitioners. Because the ESA stipulates that listing determinations should be made on the basis of the best scientific information available, NMFS formed a team of scientists with diverse backgrounds in salmon biology to conduct this review. This Status Review Team (SRT) included Katie Barnas, Michael J. Ford, Damon M. Holzer, Jeff Jorgensen, Martin Liermann, Paul Moran, Jim Myers, Andrew O. Shelton, and Laurie Weitkamp (NWFSC), and Shivonne Nesbit (WCR). In addition to the information presented by the petition, information in the scientific literature and written information provided by public comments and by state and tribal managers was also assessed by the SRT. The SRT also considered traditional ecological knowledge (TEK) received during the course of several meetings.

To determine whether a listing under the ESA is warranted, two questions must be addressed:

1. Is the entity in question a "species" as defined by the ESA?
2. If so, is the "species" threatened or endangered in all or a significant portion of its range?

These two questions are addressed in separate sections of this report. Additionally, in this report, NMFS reviews the following potential factors for decline: 1) destruction or modification of habitat, 2) overutilization by humans, 3) disease or predation, 4) inadequacy of existing regulatory mechanisms, and 5) other natural or human factors. If NMFS determines that a listing should be proposed, then NMFS is required by law (ESA Section 4(a)(1)) to identify one or more of these factors as being responsible for the species' threatened or endangered status.

ESU Configuration

The "species" question

As amended in 1978, the ESA allows listing of "distinct population segments" of vertebrates, as well as named species and subspecies. However, the ESA provides no specific guidance for determining what constitutes a distinct population segment. To clarify the issue for Pacific salmon, NMFS published a policy document describing how the agency will apply the definition of "species" in the ESA to anadromous salmonid species (USOFR 1991). A more detailed discussion of this topic appeared in NMFS (1998). The NMFS policy stipulates that a salmon population (or group of populations) will be considered "distinct" for purposes of the ESA if it represents an evolutionarily significant unit (ESU) of the biological species. An ESU is defined as a population that: 1) is substantially reproductively isolated from conspecific populations, and 2) represents an important component of the evolutionary legacy of the species. The term "evolutionary legacy" is used in the sense of "inheritance," that is, something received from the past and carried forward into the future. Specifically, the evolutionary legacy of a species is the genetic variability that is a product of past evolutionary events and that represents the reservoir upon which future evolutionary potential depends. Conservation of these genetic resources should help to ensure that the dynamic process of evolution will not be unduly constrained in the future.

The NMFS policy identifies a number of types of evidence that may be considered in the species determination. For each of the criteria, the policy advocates a holistic approach that considers various types of information as well as their strengths and limitations. Isolation does not have to be absolute, but it must be strong enough to permit evolutionarily important differences to accrue in different population units. Important types of information to consider include natural rates of straying and recolonization, natural barriers, and measurements of genetic differences between populations. Data from genetic analyses can be particularly useful for this criterion because they reflect levels of gene flow that have occurred over evolutionary time scales.

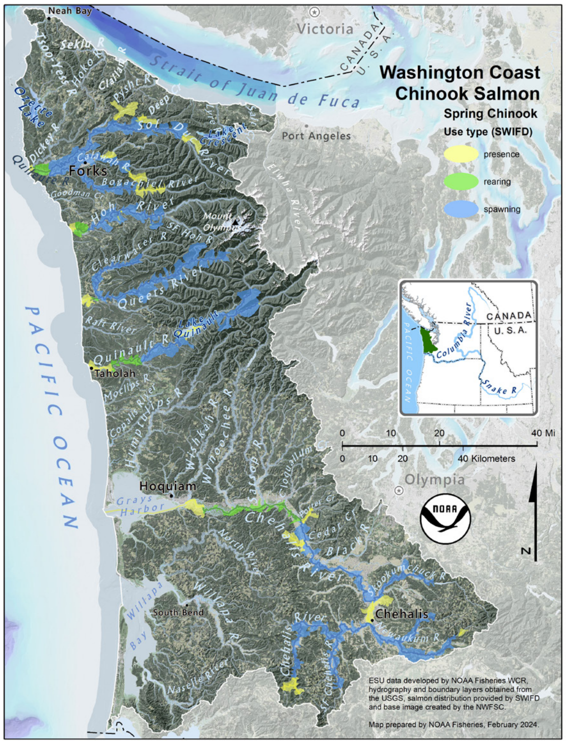
The key question with respect to the second ESU criterion is, if the population became extinct, would this represent a significant loss to the ecological/genetic diversity of the species? Again, a variety of types of information should be considered. Phenotypic and life-history traits such as size, fecundity, migration patterns, and age and time of spawning may reflect local adaptations of evolutionary importance, but interpretation of these traits is complicated by their sensitivity to environmental conditions. Analyses of genetic data provide valuable insight into the process of genetic differentiation among populations, and in some cases may also provide direct information regarding the extent of adaptive differences. Habitat differences suggest the possibility for local adaptations, but do not necessarily prove that such adaptations exist.

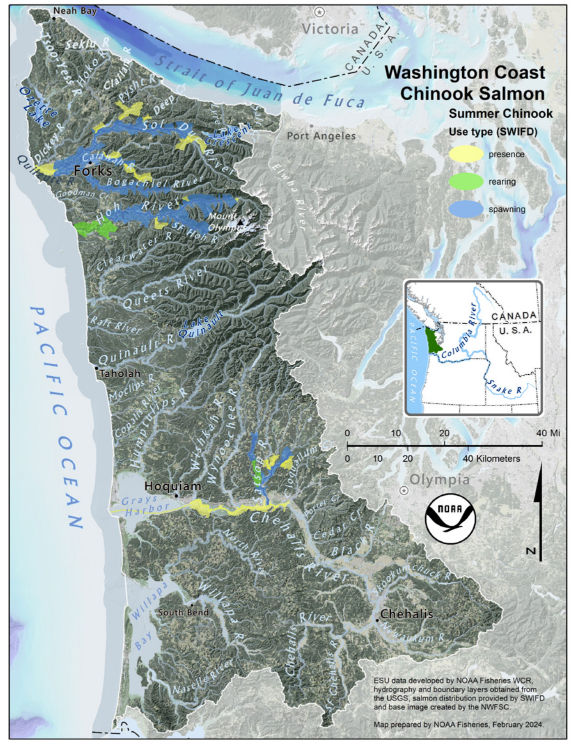
Description of Currently Identified ESU

Freshwater and nearshore distribution and life history

Washington Coast (WC) Chinook Salmon ESU populations express an ocean-type life-history strategy. As defined by Healey (1983), this strategy is characterized by a predominately subyearling juvenile emigration to salt water, and a coastally oriented marine migration pattern. Ocean-type Chinook salmon are found in watersheds throughout the coastal U.S. Pacific Northwest: Puget Sound, the Washington coast, the Columbia River, and the Oregon coast; however, more nuanced differences in run timing, age structure, and ocean distribution between populations belonging to the WC ESU and adjacent ESUs were utilized to establish the existing ESU designations (Myers et al. 1998).

The WC Chinook Salmon ESU includes spring-, summer-, and fall-run timings. Rivers in this region tend to be shorter, with low gradients near the coast. These low-gradient areas are preferred spawning sites for fall-run Chinook salmon, and the fall run predominates in most systems (Figure 2). The relatively small size of most rivers limits the amount of spawning habitat available, and reduces the likelihood of spatial separation of run times, with the possible exception of the Chehalis River, which is physically much larger than any of the other WC basins.





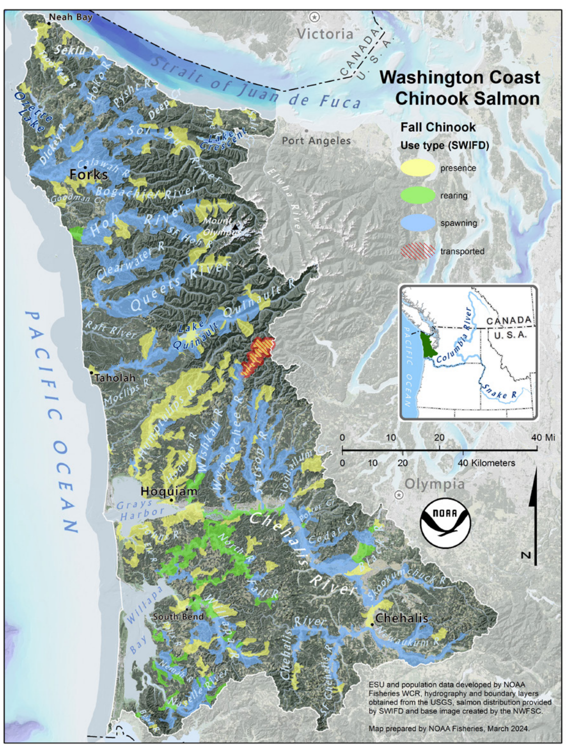


Figure 2. Freshwater habitat usage for major life-history stages of spring-, summer-, and fall-run Washington Coast Chinook salmon. Data are from Statewide Washington Integrated Fish Distribution (SWIFD), https://geo.wa.gov/datasets/wdfw::statewide-washington-integrated-fish-distribution/about

An inventory of existing runs, the state and tribal Salmon and Steelhead Stock Inventory (SASSI; WDF and WWTIT 1993) recognized two spring-run, four summer-run, four spring/summer-run,[2] and 23 fall-run "stocks" on the WC (Table 1, Figure 3). In addition to the populations identified by SASSI, there are earlier reports of spring runs in the Hoko (Water Resource Inventory Area [WRIA] 19) and Wynoochee Rivers (WRIA 22), and fall runs in the Pysht, Sekiu, Deep Creek, East Twin, West Twin, Lyre, and Clallam Rivers (Figure 4, WRIA 19; WDF 1973). There is evidence of a spring run existing in the Wynoochee River (Figure 4; Deschamps 1958); however, it appears to have been extirpated. It is unclear whether some fall-run Chinook salmon in the smaller 19 tributaries were the result of hatchery returns/strays or natural populations. Based on available information, this ESU likely contains 33 extant populations, with potentially 5–10 historical populations that may have been extirpated (WDF 1973, WDF and WWTIT 1993).

Peak spawning periods for spring-, spring/summer, and summer-run populations occur from mid-September to early October, which is somewhat later than in Puget Sound and the Strait of Juan de Fuca (Table 2). Peak river-entry times for spring- and summer-run populations range from April/May to August. Population run-timing designations—spring-, spring/summer-, and summer-run—can be somewhat arbitrary. In general, fish from these "early-returning" runs enter freshwater at an immature stage and hold in the river for an extended period (months) before spawning. Where multiple runs exist in a basin, WDFW and tribal co-managers designate redds constructed prior to 15 October as spring/summer-run Chinook salmon (state and tribal co-managers, personal communication), with redds produced afterward as being from fall-run fish. In contrast, fall-run fish enter freshwater at a more advanced stage of maturation, and spawn soon after. Peak spawning periods for coastal fall runs occur from late October to early December, with a tendency for later spawning in more southerly rivers.

Among the fall-run populations in the WC Chinook Salmon ESU, there is some variability in run and spawn timing. Hoko River fall-run Chinook salmon, the only population in the western Strait of Juan de Fuca, begin spawning in mid-September and continue through mid-November, a somewhat earlier and shorter interval than other WC fall-run populations (Marshall et al. 1995).

Chinook salmon from WC streams emigrate to saltwater predominately as subyearlings. The northern portion of the ESU (north of the Quinault River) contains rivers that drain to productive, albeit small, estuary and coastal rearing areas as rearing habitat. The limited basin size of many coastal watersheds mandates the reliance on extended estuary or coastal rearing by juveniles. The southern rivers of the WC contain numerous large estuary areas, especially in Grays Harbor and Willapa Bay. Furthermore, high summer water temperatures and related low flows may be responsible for early emigration. Chinook salmon from coastal populations (ocean-type) tend to have much larger eggs than inland, predominantly stream-type, populations (Rich 1920,[3] Nicholas and Hankin 1989, Lister 1990). Larger eggs result in larger juveniles and may enable an earlier and more successful emigration to marine rearing habitat (Fowler 1972, Kreeger 1995).

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[2] Some populations identified in SASSI that exhibit an intermediate run timing are given the "spring/summer" classification.

[3] Rich, W. H. 1920. Racial differences in salmon. Unpublished manuscript.

Table 1. Presumptive Chinook salmon populations by Water Resource Inventory Area (WRIA; Phinney and Bucknell 1975, Marshall et al. 1995) and population properties (WDF and WWTIT 1993) from the Salmon and Steelhead Stock Inventory (SASSI). Status denotes the relative health of the populations, production denotes whether the population is supported by natural-origin (wild) or hatchery-origin spawners (cultured), or a mix of the two (composite). See Term definitions for status, stock origin, and production type definitions.

		WDFW Conservation Group	SASSI (WDF and WWTIT 1993)			
WRIA	**River**	**Genetically Distinct Unit**	**Timing**	**Status**	**Origin**	**Production**
19	Hoko R	15 Western Strait Chinook	Fall	Depressed	Native	Composite
20	Tsoo-Yess R	13 North Coast Fall Chinook	Fall	Unknown	Native	Cultured
20	Sol Duc R	14 North Coast Spring Chinook	Spring	Healthy	Non-native	Composite
20	Bogachiel R	14 North Coast Spring Chinook	Summer	Unknown	Native	Wild
20	Sol Duc R	14 North Coast Spring Chinook	Summer	Healthy	Native	Composite
20	Calawah R	14 North Coast Spring Chinook	Summer	Unknown	Native	Wild
20	Bogachiel R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
20	Dickey R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
20	Sol Duc R	13 North Coast Fall Chinook	Fall	Healthy	Native	Composite
20	Calawah R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
20	Hoh R	14 North Coast Spring Chinook	Spr/Su	Healthy	Native	Wild
20	Hoh R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
21	Queets R	14 North Coast Spring Chinook	Spr/Su	Depressed	Native	Wild
21	Clearwater R	14 North Coast Spring Chinook	Spr/Su	Depressed	Native	Wild
21	Queets R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
21	Clearwater R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
21	Quinault R	14 North Coast Spring Chinook	Spr/Su	Depressed	Native	Wild
21	Quinault R	13 North Coast Fall Chinook	Fall	Healthy	Native	Wild
21	Cook Cr	13 North Coast Fall Chinook	Fall	Healthy	Native	Composite
22	Satsop R	14 North Coast Spring Chinook	Summer	Healthy	Mixed	Wild
22	Humptulips R	11 South Coast Fall Chinook	Fall	Healthy	Mixed	Wild
22	Hoquiam R	11 South Coast Fall Chinook	Fall	Healthy	Native	Wild
22	Wishkah R	11 South Coast Fall Chinook	Fall	Healthy	Native	Composite
22	Wynoochee R	11 South Coast Fall Chinook	Fall	Healthy	Native	Wild
22	Satsop R	11 South Coast Fall Chinook	Fall	Healthy	Mixed	Composite
22	Chehalis R	11 South Coast Fall Chinook	Fall	Healthy	Mixed	Wild
22	Johns/Elk Rs	11 South Coast Fall Chinook	Fall	Unknown	Mixed	Wild
23	Chehalis R	12 South Coast Spring Chinook	Spring	Healthy	Native	Wild
24	Willapa Bay	13 North Coast Fall Chinook	Fall	Healthy	Mixed	Composite
24	Palix R	13 North Coast Fall Chinook	Fall	Healthy	Mixed	Composite
24	Nemah R	13 North Coast Fall Chinook	Fall	Healthy	Mixed	Composite
24	Naselle R	13 North Coast Fall Chinook	Fall	Healthy	Mixed	Composite
24	Fall R	13 North Coast Fall Chinook	Early Fall	Depressed	Native	Wild

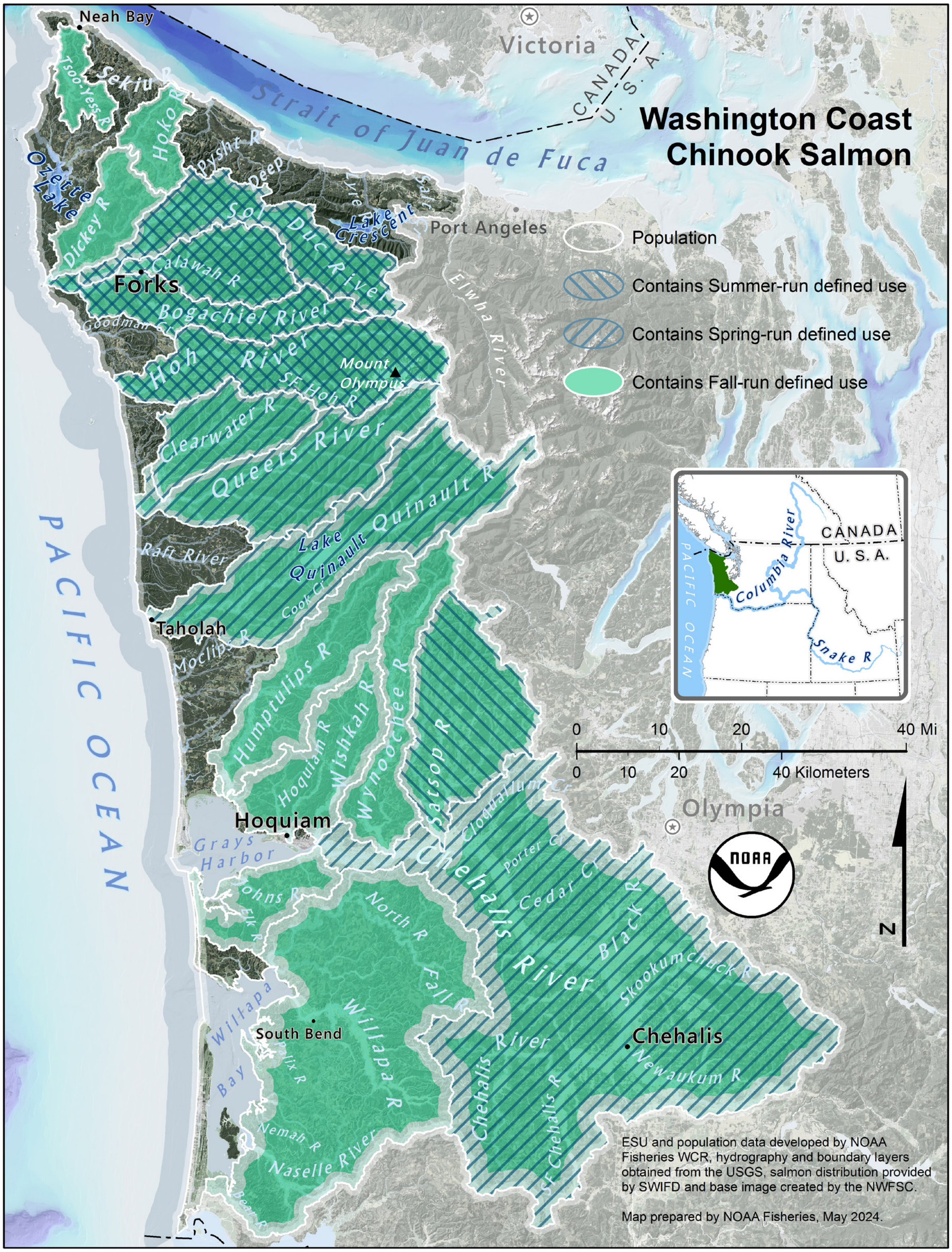

Figure 3. River systems identified as supporting spring-, summer-, and fall-run populations of Chinook salmon within the WC Chinook Salmon ESU (SWIFD).

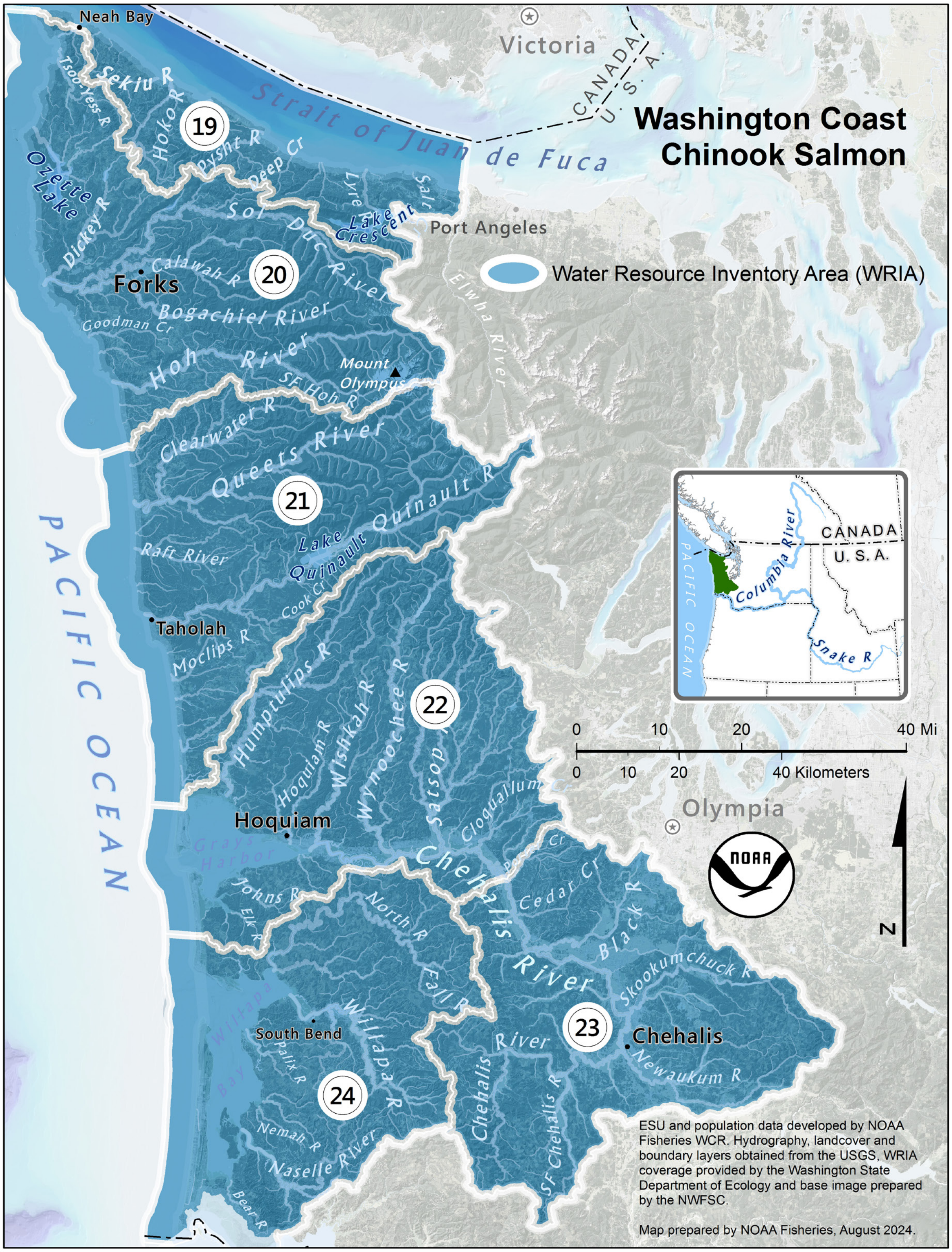

Figure 4. Water Resource Inventory Areas (WRIAs) on the Washington Coast.

Table 2. Washington Coast Chinook Salmon ESU presumptive population freshwater return entry (gray) and spawn (blue) timing, with peak spawning period identified by the letter *P* based on citation(s) noted.

Population	Run	Mar			Apr			May			June			July			Aug			Sept			Oct			Nov			Dec			Citation(s)
Hoko R	Fall																							P								WDF and WWTIT 1993
Tsoo-Yess R	Fall																								P	P						WDF and WWTIT 1993
Sol Duc R	Spring																				P											WDF and WWTIT 1993
Sol Duc R	Fall																									P						WDF and WWTIT 1993
Bogachiel R	Summer																					P										WDF and WWTIT 1993, QTNR 1995
Bogachiel R	Fall																									P						WDF and WWTIT 1993
Calwah R	Summer																					P										WDF and WWTIT 1993
Calwah R	Fall																									P						WDF and WWTIT 1993, QTNR 1995
Dickey R	Fall																															WDF and WWTIT 1993
Hoh R	Summer																					P										WDF and WWTIT 1993, HIT 1995
Hoh R	Fall																									P						WDF and WWTIT 1993
Queets R	Summer																						P									WDF and WWTIT 1993
Queets R	Fall																															WDF and WWTIT 1993
Clearwater R	Summer																						P									WDF and WWTIT 1993
Clearwater R	Fall																															WDF and WWTIT 1993
Quinault R	Spr/Su																															WDF and WWTIT 1993
Quinault R	Fall																															WDF and WWTIT 1993
Chehalis R	Spring																				P											WDF and WWTIT 1993
Chehalis R	Fall																															WDF and WWTIT 1993
Wynoochee R	Spring																				P											WDF and WWTIT 1993
Wynoochee R	Fall																								P							WDF and WWTIT 1993
Satsop R	Summer																					P										WDF and WWTIT 1993
Satsop R	Fall																								P							WDF and WWTIT 1993
Elk R	Fall																								P							WDF and WWTIT 1993
Willapa R	Fall																								P							WDF and WWTIT 1993
North R	Fall																						P									WDF and WWTIT 1993

Coastal Chinook salmon from this region also mature at a later age than stocks from Puget Sound, the Lower Columbia River, and southern Oregon coastal areas (Nicholas and Hankin 1989, Skagit River System Cooperative 1995, Quinault Fisheries Division 1995, WDFW 1995). On average, the most common age classes are four- (28.4%) and five-year-olds (46.4%), with three-and six-year-old and up returns being roughly equally represented at 13.0% and 12.2% respectively (Myers et al. 1998). Age structure varies among populations within the region, and is influenced by a number of factors, including run-timing, basin conditions, and inclusion of hatchery fish in age samples. The numerically large populations of Chinook salmon on smaller coastal rivers may create competition for mates and select for larger (older) male Chinook salmon (Roni and Quinn 1995).

The WC Chinook Salmon ESU is very similar to a conservation unit identified by WDFW (Marshall et al. 1995), the Coastal Major Ancestral Lineage (MAL III), which includes coastal Chinook salmon in five genetic diversity units (GDUs): South Coast Fall, Chehalis Spring, North Coast Fall, North Coast Spring, and Western Strait [of Juan de Fuca]. The Coastal MAL also includes Chinook salmon in the Elwha and Dungeness Rivers, but otherwise corresponds to the ESU. WDFW's MAL units were established using similar criteria (genetic differences, geographic distribution, and life-history characteristics) to those used by NMFS (Busack and Marshall 1995, Marshall et al. 1995).

Term definitions

The following definitions are reproduced from WDF and WWTIT (1993).

Status

Healthy: A stock of fish experiencing production levels consistent with its available habitat and within the natural variations in survival for the stock.

Depressed: A stock of fish whose production is below expected levels based on available habitat and natural variations in survival rates, but above the level where permanent damage to the stock is likely.

Critical: A stock of fish experiencing production levels that are so low that permanent damage to the stock is likely or has already occurred.

Extinct: A stock of fish that is no longer present in its original range, or as a distinct stock elsewhere. Individuals of the same species may be observed in very low numbers, consistent with straying from other stocks.

Unknown: There is insufficient information to rate stock status.

Stock origin

Native: An indigenous stock of fish that has not been substantially impacted by genetic interactions with non-native stocks, or by other factors, and is still present in all or part of its original range. In limited cases, a native stock may also exist outside of its original habitat (e.g., captive broodstock programs).

Non-native: A stock that has become established outside of its original range.

Mixed: A stock whose individuals originated from commingled native and non-native parents, and/or by mating between native and non-native fish (hybridization); or a previously native stock that has undergone substantial genetic alteration.

Unknown: This description is applied to stocks where there is insufficient information to identify stock origin with confidence.

Production type

Wild: A stock that is sustained by natural spawning and rearing in the natural habitat, regardless of parentage (includes native). Equivalent to NOAA's *NOR* (natural-origin recruit.)

Cultured: A stock that depends upon spawning, incubation, hatching, or rearing in a hatchery or other artificial production facility.

Composite: A stock sustained by both wild and artificial production.

Geographic and ecological characteristics

The fidelity with which Chinook salmon return to their natal stream implies a close association between a specific population and its freshwater environment. The selective pressures of different freshwater environments may be responsible for differences in life-history strategies among stocks. Miller (1982) hypothesized that local temperature regimes are the major factor influencing life-history traits. If the boundaries of distinct freshwater habitats coincide with differences in life histories, it would suggest a certain degree of local adaptation, with selective forces limiting introgression. Therefore, identifying distinct freshwater, terrestrial, and climatic regions may be useful in identifying Chinook salmon ESUs. The Environmental Protection Agency (EPA) has established a system of ecoregion designations based on soil type, topography, climate, potential vegetation, and land use (Omernik 1987). Ecoregions and other geographic terrestrial and climatic differences were utilized by Myers et al. (1998) in identifying Chinook salmon ESUs.

With the exception of some headwater regions in the Olympic Mountains, the majority of the WC Chinook Salmon ESU lies within Coast Range Ecoregion #1 (Level III, Omernik 1987). This ecoregion extends from the Olympic Peninsula through the Coast Range proper and down to the Klamath Mountains and the San Francisco Bay area. This region is influenced by medium-to-high rainfall levels due to the interaction between marine weather systems and the mountainous nature of the region. Topographically, the entire ecoregion averages about 500 m (1,600 ft) in elevation, with most peaks in the range under 1,200 m (3,900 ft). These mountains are generally rugged with steep canyons. Between the ocean and the mountains lies a narrow coastal plain composed of sand, silt, and gravel. Tributary streams are relatively short and have steep gradients; therefore, surface runoff is rapid and water storage is relatively short-term during periods of no recharge. These rivers are especially prone to low flows during times of drought. Regional rainfall averages 200–240 cm (79–95 in) per year, but within the ecoregion correlates with the topography. Average annual river flows for most rivers in this region are among the highest found on the U.S. West Coast when adjusted for watershed area. River flows peak during winter rain storms common in December and January, with snowmelt adding to the surface runoff in the spring for those basins with higher-elevation headwaters, providing a second flow peak, and there are long periods when the river flows are maintained at least 50% of peak flow. During July or August there is very little precipitation; this period may expand to 2–3 months every few years. River flows are correspondingly at their lowest and temperatures at their highest during August and September (Omernik and Gallant 1987).

For Chinook salmon populations within the WC Chinook Salmon ESU (Figure 5), the hydrological conditions vary considerably with topography. Rivers along the Strait of Juan de Fuca (western WRIA 19) tend to be shorter, draining the Olympic foothills, and exhibit a rainfall-dominated hydrograph. Rivers draining to the Pacific Ocean, north of the Chehalis River, are a mix of shorter rain-driven rivers and larger rivers with headwaters in the Olympic Mountains, with snow and transitional hydrographs (WRIAs 20 and 21); rivers south of the Chehalis River drain the relatively low-elevation Willapa Hills and exhibit rain-driven hydrographs (WRIAs 22–24). Local geology, land ownership, and other factors influence conditions within each WRIA, and are discussed below.

Western Strait of Juan de Fuca

The western Strait of Juan de Fuca (western portion of WRIA 19) includes the area between the Hoko River and Cape Flattery (Figure 4). The largest watersheds in this group are the Hoko and Sekiu Rivers. In addition, there are numerous small, independent streams that flow northerly, draining the relatively low-elevation foothills of the Olympic Mountains into the Strait of Juan de Fuca. These drainages have steep gradients near their headwaters, but contain many kilometers of moderate-gradient lowland stream channel (WDF 1973). The region is characterized by a cool, maritime climate with annual precipitation increasing as one moves either west or upward in elevation (McHenry and Kowalski-Hagaman 1996). Annual precipitation typically ranges from 200–330 cm (80–130 in) in the headwaters of these streams.

Olympic Peninsula streams in the Twin River formation, which includes all of western WRIA 19, have high natural erosion rates, due to a geology of sedimentary rocks, sandstones, and siltstones, which quickly break down in the heavy rainfall climate (Benda 1993). The stream channels in the region change quickly, due to variations in flow and sediment inputs. The Hoko and Sekiu Rivers have tidal influence that extends upstream for several miles (Light and Herger 1994). The Hoko River is the largest in this area, with approximately 40 km (25 mi) of mainstem and 129 km (80 mi) of tributaries

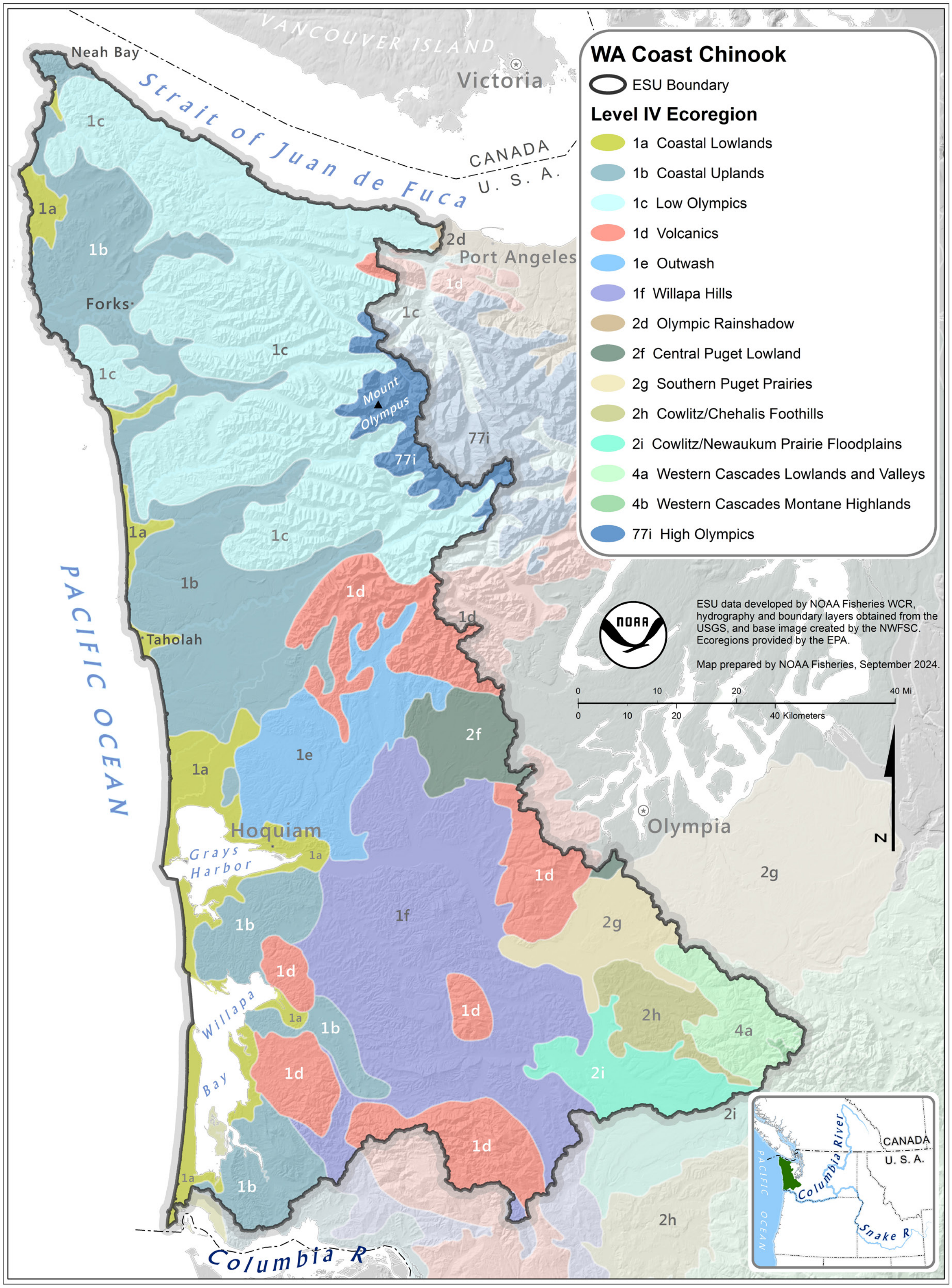

Figure 5. U.S. EPA Level IV Ecoregions within the WC Chinook Salmon ESU. Ecoregions identify areas with shared climatic, geologic, and vegetative characteristics (Omernik and Griffith 2014). Nomenclature for Level IV Ecoregions includes the “1” Level III designation.

(Light and Herger 1994). Although Chinook salmon have been observed in limited numbers spawning in other streams draining to the Strait, the Hoko River contains the only currently recognized Chinook salmon population (Marshall et al. 1995). Historically, other Chinook salmon populations may have been present in the Pysht, Sekiu, Sail, Lyre, and Clallam Rivers (McHenry and Kowalski-Hagaman 1996).

North Pacific Coast

WRIA 20 includes all streams that drain into the Pacific Ocean from Cape Flattery south to, but excluding, Kalaloch Creek (Figure 4). The largest basin in the WRIA is the Quillayute River with its four major sub-basins: the Dickey, Calawah, Bogachiel, and Sol Duc Rivers. Other basins in the WRIA include the Wa'atch, Tsoo-Yess, Ozette, and Hoh River systems, as well as several small, independent streams. Within this WRIA are 569 streams and 2,181 stream km (1,355 stream mi; Light and Herger 1994). Annual rainfall in the basin is the highest in the state, ranging from 200 cm (80 in) near the coast to 610 cm (240 in) in the Olympic Mountains (McHenry and Kowalski-Hagaman 1996). This region is often exposed to high wind and heavy rainstorms, which play important roles in current habitat problems located in disturbed (logged or developed) areas. Much of the headwater drainages for the Quillayute and Hoh Rivers are located in Olympic National Park (Table 3). In these undisturbed areas, temperate rainforests of coniferous old growth are dominated by Sitka spruce (*Picea sitchensis*) in the lowlands and western hemlock (*Tsuga heterophylla*) with silver fir (*Abies amabilis*) in the higher elevations. Bigleaf maple (*Acer macrophyllum*) is also a component of the rainforests. The old-growth conifers can reach up to 60 m (200 ft) in height, and are characterized by somewhat open canopies and low densities.

WRIA 21 comprises the Queets and Quinault River watersheds, along with several smaller drainages. The climate is temperate maritime, with an average annual precipitation of 300–500 cm (120–200 in). Winter storms commonly deliver over 25 cm (10 in) of precipitation in a single event. Similar to other west-side watersheds, the Queets and Quinault River basins lie within a region of temperate rainforest and are dominated by Sitka spruce, red alder (*Anus rubra*), bigleaf maple, western red cedar (*Thuja plicata*), and Douglas fir (*Pseudotsuga mensiesii*) in the lowlands, with western hemlock and silver fir in the higher elevations (Smith and Caldwell 2001). The vast majority of land in the Queets

Table 3. Total watershed areas and the proportion of watershed areas inside the Olympic National Park (ONP) boundaries for the major coastal tributaries in the Olympic Peninsula Steelhead DPS. Data from USGS's PAD database, https://doi.org/10.5066/P9Q9LQ4B.

Basin	Tributary	Total area	Within ONP	% within	Outside ONP	% outside
Quillayute R	Bogachiel R	395.51 km^2	212.09 km^2	54%	183.42 km^2	46%
	Calawah R	351.67 km^2	66.73 km^2	16%	284.94 km^2	84%
	Dickey R	223.53 km^2	0.00 km^2	0%	223.53 km^2	100%
	Sol Duc R	603.45 km^2	194.03 km^2	39%	409.42 km^2	61%
Hoh R	Hoh R	770.97 km^2	445.63 km^2	58%	325.34 km^2	42%
Queets R	Queets R	769.50 km^2	388.92 km^2	51%	380.61 km^2	49%
Quinault R	Quinault R	1,123.48 km^2	567.02 km^2	50%	556.42 km^2	50%

River basin is forested, with ownership dominated by the federal government (ONP, 75%), Quinault tribal lands (13%), the State of Washington (10%), and private timber companies (2%; NWIFC 2020). The Sitka spruce zone extends along the coast and inland up the Queets and Quinault River valleys, while the western hemlock zone dominates from sea level to approximately 610 m (2,000 ft) in elevation. The Pacific silver fir zone exists from 610–1,220 m (2,000–4,000 ft) in elevation, and extends upwards to the subalpine forest in the Olympic Mountains. As in WRIA 20, the upper watersheds of the Queets and Quinault Rivers lie within ONP (Table 3), with much of the remaining area in the Quinault Reservation.

South Pacific Coast: Grays Harbor (WRIAs 22 and 23)

The Grays Harbor region includes the Grays Harbor estuary and watersheds within WRIAs 22 and 23. The geographic area includes the entire Chehalis River drainage and all tributaries to the Chehalis River. Historically, stream flows in the Olympic Peninsula headwater systems were seasonally supplemented by permanent snow and ice cover, but glacial loss has removed this summer flow component (Lestelle 2009). The Chehalis River basin is the second-largest basin in Washington State, second only to the Columbia River. The largest tributaries, in order of average annual discharge, are the Satsop River sub-basin, Wynoochee River, Skookumchuck River, Newaukum River, and Cloquallum Creek. In addition to the Chehalis River, numerous other independent rivers drain into Grays Harbor, such as the Humptulips, Hoquiam, Johns, and Elk Rivers, and several smaller streams. Grays Harbor is about 19 km (12 mi) wide at the widest point, and at high tide covers about 250 km^2 (97 mi^2). A 3 km (2 mi) wide channel connects Grays Harbor to the Pacific Ocean. Two major river basins drain into Grays Harbor. The Chehalis River drains 5,698 km^2 (2,200 mi^2) into the inner harbor, while the Humptulips River basin drains 635 km^2 (245 mi^2) into North Bay. North Bay is relatively undeveloped, while the inner harbor is heavily industrialized. Pulp mills, landfills, sewage treatment plants, and log storage facilities are all located within the inner harbor. Pre-contact, this region was predominantly temperate rain forest, but nearly 81% of the Chehalis River basin has been converted to commercial forest (Hashim 2002).

In total, there are 1,391 streams with 5,396 lineal stream km (3,353 mi) within the two WRIAs. Grays Harbor is a complex of rivers, shorelines, estuaries, beaches, and tidal flats. This area provides important rearing habitat for juvenile salmonids and an interface between the ocean and freshwater for emigrating juvenile salmonids and returning adults. All salmonid species use estuarine and nearshore environments at some time during their life cycle (Sandell et al. 2011).

Critical food fish for salmonids also occupy areas within the harbor. The larval northern anchovy (*Engraulis mordax*) is found in deeper waters of Grays Harbor and serves as food for Chinook and chum (*Oncorhynchus keta*) salmon (Simenstad and Eggers 1981). These authors also suggest that open-water zooplankton levels limit the population of juvenile salmonids in Grays Harbor. Both Pacific herring (*Clupea harangus*) and Pacific sand lance (*Ammodytes hexapterus*) spawning beds are found in this area. Both of these species are important food items for salmonids. Large woody debris (LWD) in the estuary was common prior to logging and settlements, but is now believed to be very low. Estuarine LWD serves as cover for juvenile salmonids (Martin and Dieu 1997). The wood also creates firm substrates in a fine sediment environment, and is used as nurse logs by spruce and cedar.

South Pacific Coast: Willapa Basin (WRIA 24)

In total, there are roughly 745 streams encompassing over 2,366 lineal stream km (1,470 mi) in the Willapa River region (Light and Herger 1994). Annual rainfall in the basin has averaged about 216 cm (85 in), with a range of 112–368 cm (44–145 in) and an average of 8 cm (3 in) of rain per month during the summer (The Willapa Alliance 1998). No streams within the Willapa River basin originate from glaciers; all depend on surface and groundwater inputs. Therefore, precipitation plays an important role in the quantity and quality of salmon habitat. However, Willapa Bay salinity appears to be linked not only to the Willapa River basin drainages, but also to flow from the Columbia and Chehalis River basins (The Willapa Alliance 1998). Many salinity profiles for U.S. West Coast estuaries show a peak in the summer and a low in the winter, but Willapa Bay salinity drops in the late spring, when snowmelt from the Columbia and Chehalis River basins is emptying into the Pacific Ocean. Because the greatest source of freshwater for Willapa Bay is the Columbia River, the Willapa Bay ecosystem depends upon the maintenance of water quality in the Columbia River. The rivers and tributaries in this area provide spawning and rearing habitat for winter steelhead trout (*Oncorhynchus mykiss*) and chum, coho (*O. kisutch*), and Chinook salmon.

Historically, the Willapa Bay area faced disturbances from natural fires about every 200 years. The last great fire was in the early 1400s. Since then, the climate has changed to become wetter and cooler (Pentec Environmental, Inc. 1997). Tree trunks measuring 4–10 ft in diameter attest to the size of the trees at the onset of timber harvest.

Ocean distribution

The ocean distribution of Chinook salmon, especially ocean-type Chinook salmon (Healey 1983), appears to be a heritable trait, with populations from geographic regions exhibiting characteristic coastal ocean migrations (Myers et al. 1998). Ocean distribution similarities among populations represent an important life-history characteristic that can be used to define ESUs. We use published analyses of coded-wire tag (CWT) recoveries from hatchery fish caught in ocean fisheries (Weitkamp 2010, Shelton et al. 2019, 2021) to describe spatial differences among rivers in CWT recoveries and estimated ocean distributions, updating the original analysis in Myers et al. (1998).

From Shelton et al. (2021) we used estimates of ocean distribution for fall-run salmon from WC stocks. This estimated distribution is derived from CWT Chinook salmon released into the Quinault, Quillayute, Humptulips, and Tsoo-Yess Rivers between 1977 and 2009. Spatial areas follow codes from Shelton et al. 2021 (see Figure 6) and span from Monterey, California (MONT), to north southeastern Alaska (NSEAK). Panels show proportional distributions for each stock during the summer and fall seasons (Figure 7). Estimated ocean distributions are derived from a population dynamic model that uses CWT and fishing effort for commercial and recreational fleets from California to Alaska (Shelton et al. 2019, 2021). The WC Chinook salmon stocks are far north-migrating Chinook salmon (CTC 2023), and are found predominantly in British Columbia and Alaska during the summer, with more of them present off Washington and Vancouver Island during the fall in anticipation of the freshwater spawning migration.

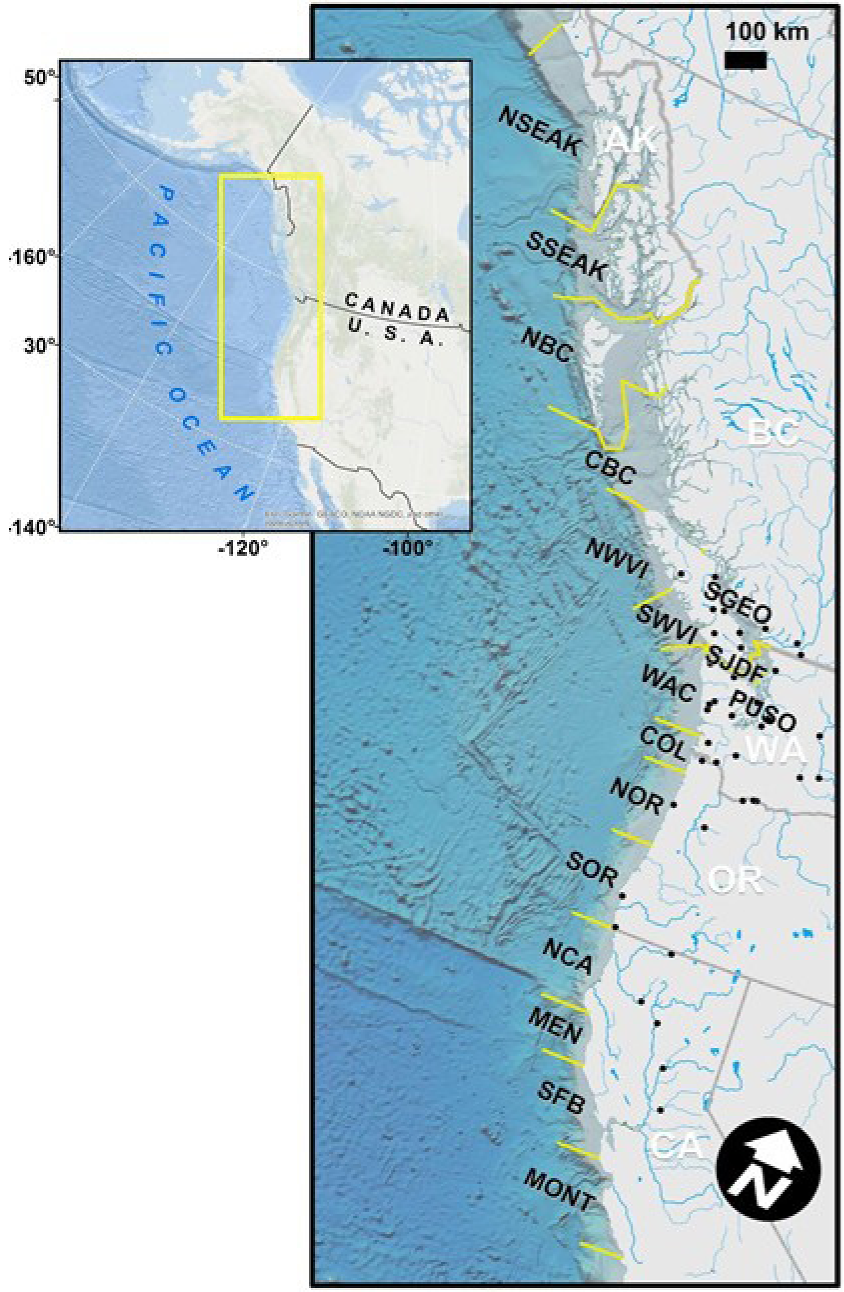

Figure 6. Ocean fishery regions used in describing the recovery of CWT Chinook salmon released from designated hatcheries (black dots). Adapted from Shelton et al. (2021), their Figure 1.

Weitkamp (2010) provides ocean recoveries of additional CWT groups from the WC Chinook Salmon ESU. The main distinction between the methods of Shelton et al. (2019, 2021) and Weitkamp (2010) is that Shelton et al. attempted to account for fisheries effort and season in their estimates of distribution, while Weitkamp summarized mainly the spatial distribution CWT recoveries. They also use different years of CWT recovery data: Weitkamp (2010) used CWT recoveries through 2004. Shelton et al. (2021) used data through 2015. We present Weitkamp (2010) results for the proportion of CWT recovered by state (Alaska, Washington, Oregon, or California) or Province (British Columbia) for four

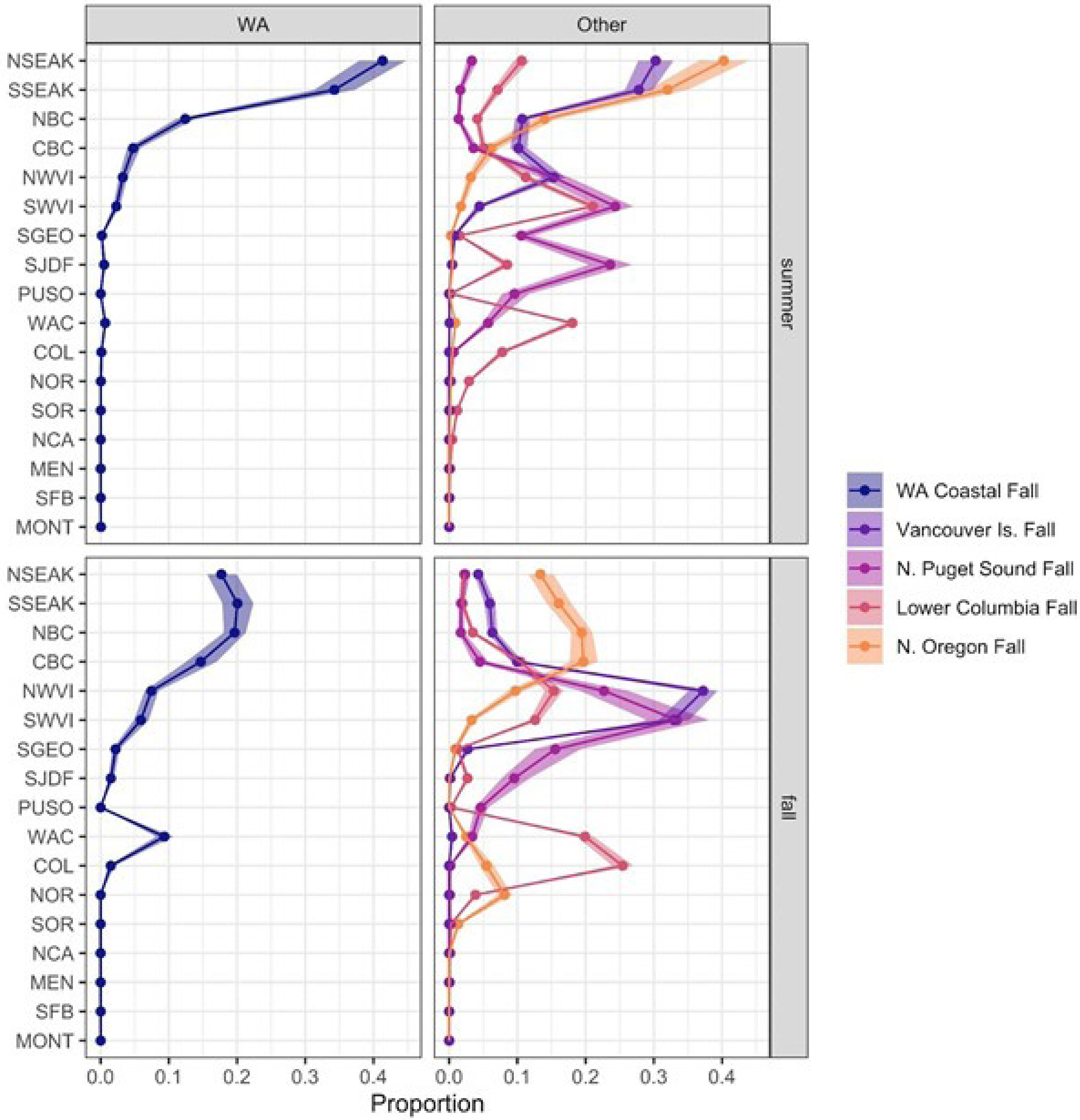

Figure 7. *Left*: Ocean distribution estimates for two fall-run stocks in summer (Jun–Jul) and fall (Aug–Oct) from CWT Chinook released in the Quinault, Quillayute, Humptulips, and Tsoo-Yess rivers (all based on recoveries between 1979 and 2015). The *y*-axis shows spatial areas between Monterey, CA (*MONT*) and north southeastern Alaska (*NSEAK*; see also Figure 6). *Right*: Estimated distributions for 5 other Chinook salmon groups. These are proportional distributions (proportions for each season–origin combination sum to 1) for the among-year average distribution.

stocks (Figure 8). Three stocks—Willapa Bay, Quinault River, and Grays Harbor fall-run—are identical to the stocks used in Shelton et al. (2021; Figure 7), while Sol-Duc River spring-run Chinook salmon were not included in Shelton's analysis. For comparison, patterns of CWT recoveries for adjacent ESUs are presented in Figure 8 as well.

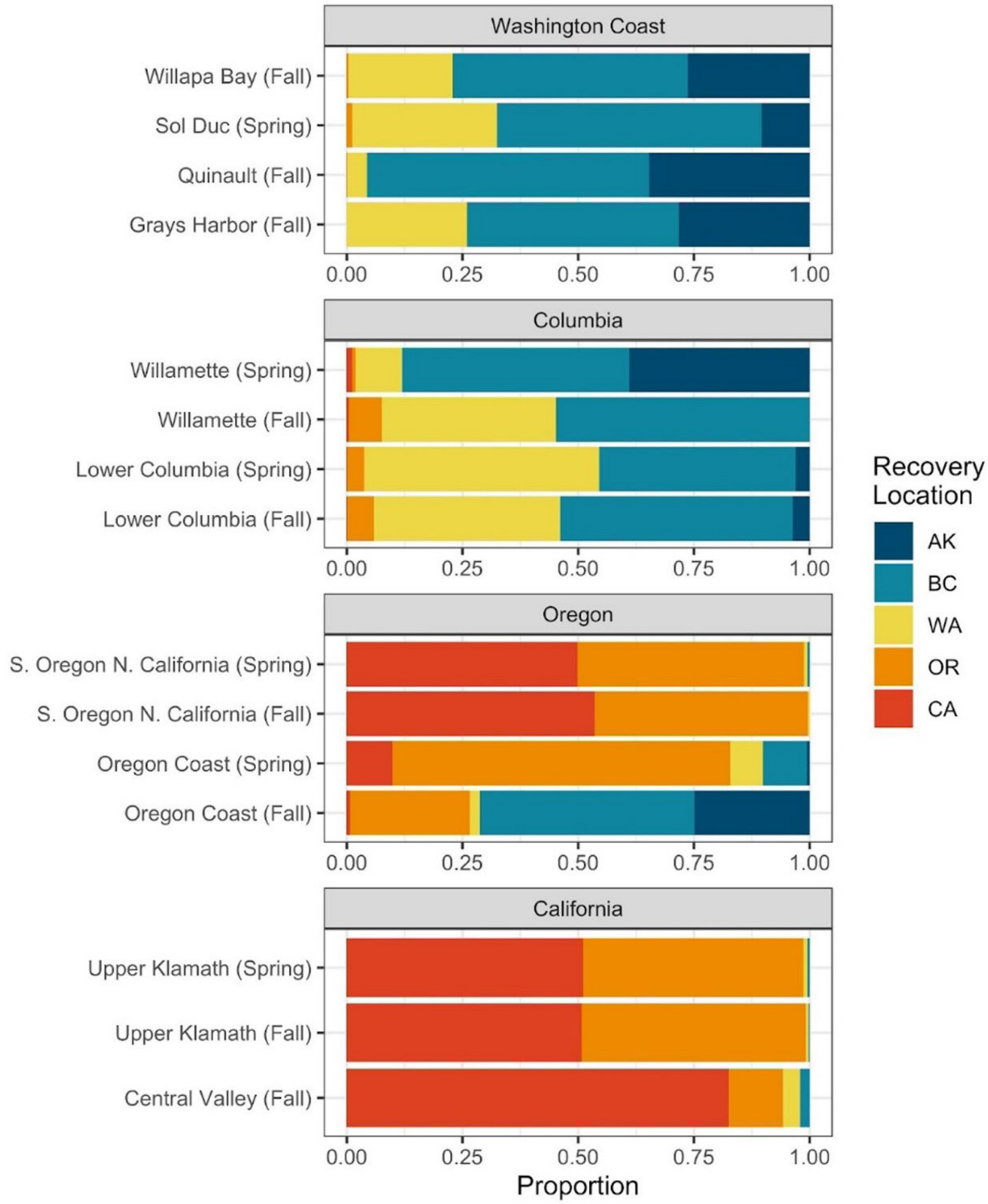

Figure 8. Proportional distribution of ocean CWT recoveries by state/province for WC Chinook salmon as well as other ESUs. Run type is noted along with river of origin. Recoveries of CWT from nearby ESUs (Columbia River, Oregon Coast, California) are shown for comparison. See Weitkamp (2010) for methodological details.

In summary, WC Chinook salmon show a more northerly pattern of ocean distribution, with significant recoveries in southeastern Alaskan fisheries, distinguishing them from adjacent Chinook salmon populations from Vancouver Island, Puget Sound, and the Lower Columbia River. Washington Coast Chinook salmon have northerly distributions similar to Oregon Coastal Chinook salmon, but are rarely caught in Oregon waters.

Genetics

Previous studies

Along with consideration of ecological, spatial distribution, and life-history characteristics, patterns of genetic variation provide useful information for inferring the degree of reproductive isolation among populations. Myers et al. (1998) summarized the genetic studies of Chinook salmon available at that time and conducted analyses. The method of evaluating genetic variation at that time was the electrophoretic separation of protein variants. Their analyses included numerous samples of Chinook salmon from the U.S. West Coast, including nine samples from Washington coastal rivers collected between 1981 and 1993. All of the WC samples were considered to be taken from fall runs, except for one spring-run sample from the Skookumchuck River. Based on a distance matrix constructed from allele frequencies at 31 allozyme loci, the WC samples all grouped together in a cluster, distinct from samples in other river systems (Figure 9). This analysis, along with consideration of ecological, distributional, and life-history information, was used to identify the WC Chinook Salmon ESU (see summary above). The spring-run sample from the WC (Sample 107 in Figure 9) was genetically similar to the nearby WC fall-run samples, and did not cluster with spring-run samples from other areas. This general pattern was seen throughout the range of coastal Chinook salmon, and helped inform the conclusion that run-timing was not a good criterion for ESU configurations in coastal Chinook populations (Waples et al. 2004, 2022, Ford et al. 2021).

Subsequent genetic studies

In the 26 years following the Myers et al. (1998) status review, numerous studies on Chinook salmon genetics have been published. Here, we focus on studies that directly inform our understanding of the identification of the WC Chinook Salmon ESU. These include large-scale studies of Chinook salmon coastwide, similar to those reported by Myers et al. (1998), but utilizing more recently collected samples or different methods of genetic analysis. They also include studies that focus specifically on the genetic basis of run timing in WC Chinook salmon.

Large-scale studies

Waples et al. (2004) analyzed largely the same genetic data as Myers et al. (1998). One of the main conclusions of their paper was that in coastal Chinook salmon ESUs, run timing does not correspond to distinct evolutionary lineages. Beacham et al. (2006) reported on a large coastwide microsatellite dataset for 325 Chinook salmon population samples. Similar to Waples et al. (2004), the authors found that genetic patterns for coastal populations are structured by geography rather than by run timing. The study included four population samples from the WC—Sol Duc River, Quinault River, Hoh River, and Queets River—sampled between 1995 and 1997. All were classified as fall-run, except for the Hoh River sample, which was classified as spring-run. Similar to prior studies, the WC samples, including the spring-run sample, were more genetically similar to each other than they were to samples from other coastal river systems.

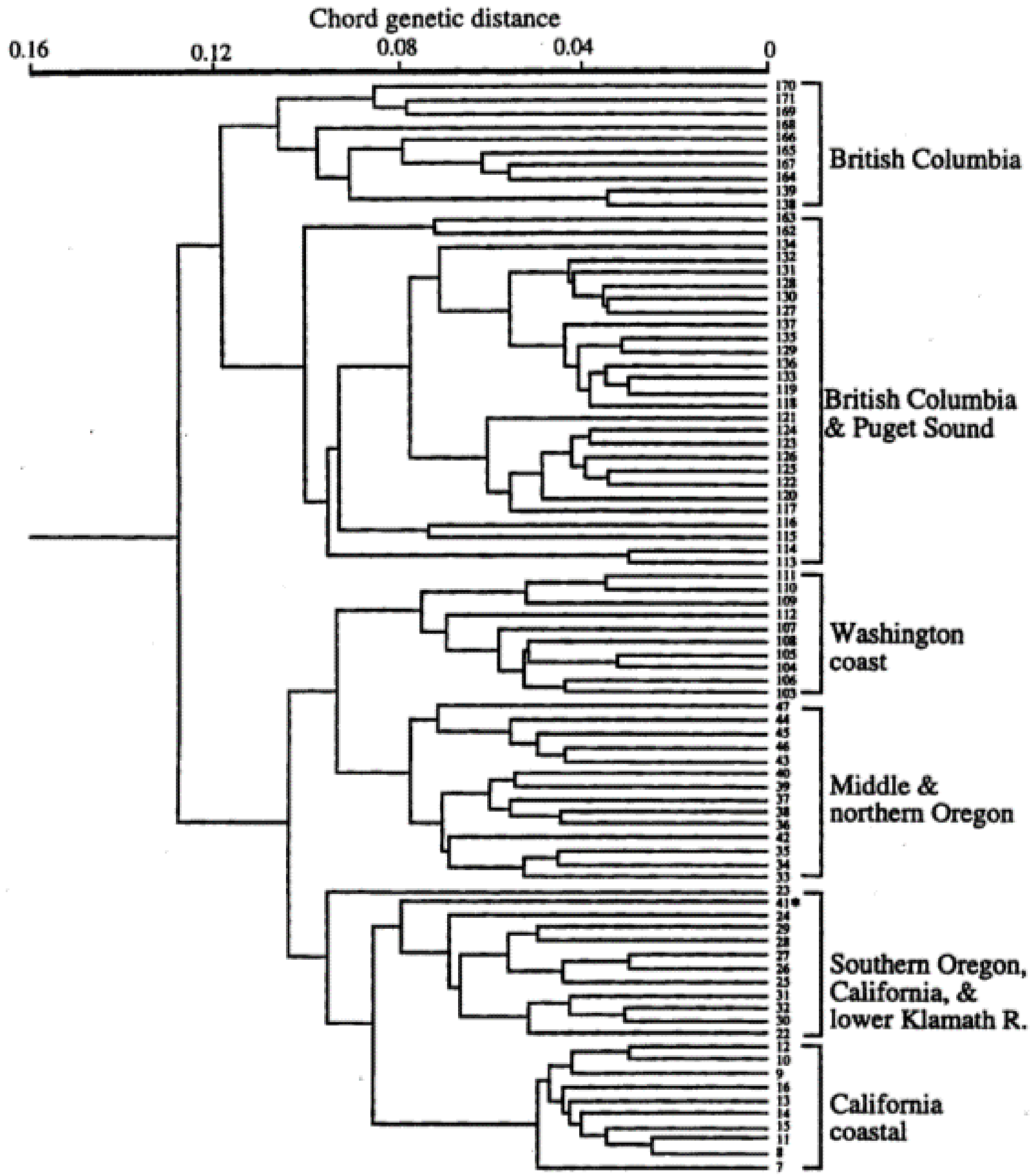

Figure 9. Unweighted pair group method with arithmetic averages (UPGMA) tree of Cavalli-Sforza and Edwards (1967) chord distances based on 31 allozyme loci between 83 composite samples of Chinook salmon. The Washington Coast samples are: 103 Naselle Hatchery fall, 104 Wynoochee River and Hatchery fall, 105 Wishkah River fall, 106 East Fork Satsop River fall, 107 Skookumchuck River spring, 108 Humptulips Hatchery fall, 109 Quinault Hatchery fall, 110 Queets River Hatchery fall, and 111 Hoh River fall. Reproduced from Myers et al. (1998).

Seeb et al. (2007) also reported on a large coastwide microsatellite data set for 110 Chinook salmon population samples. The study contained three WC samples—Queets River fall, Quillayute River fall, and Sol Duc spring—collected between 1995 and 2003. The results are generally similar to earlier studies, except that the Sol Duc sample clustered with samples from the Lower Columbia River separate from the other Washington coast samples. This pattern is likely due to the use of hatchery broodstocks in the Sol Duc River that originated from the Lower Columbia River.

Moran et al. (2013) reported on a coastwide dataset that examined variation at 13 microsatellite loci for 144 Chinook salmon population samples, including many of the same samples previously analyzed by Seeb et al. (2007). Similar to Waples et al. (2004), this study found that genetic patterns for coastal populations were structured by geography and not by run timing. The study included seven samples from the WC, all fall runs except for a Sol Duc River sample, which was spring-run. Like Seeb et al. (2007), the Sol Duc sample clustered with Lower Columbia River samples rather than WC.

Studies focused specifically on the Washington coast

There are several relatively recent population genetic studies that focus specifically on patterns of variation in WC Chinook salmon, including a specific focus genetic variation related to run timing.

Brown et al. (2017) surveyed variation at 286 single nucleotide polymorphism (SNP) loci from ~30 stream/run-time groups within the WC Chinook Salmon ESU, the Lower Columbia River, and numerous rivers in Puget Sound and British Columbia. Sampling dates for the WC were generally in the early 2000s, although a few samples included more recent years. The primary objective of the study was understanding the population structure of the Chehalis River basin, including effects of run timing, but the study is also a valuable source of information for overall ESU structure.

By increasing the number of rivers sampled and utilizing a relatively large number of genetic loci, the results from this study considerably extend those from prior studies. Overall, the patterns of variation are consistent with the existing ESU boundaries (Figure 10). In particular, Chinook salmon from the WC are clearly distinct from other coastal Chinook salmon in the Lower Columbia River, Puget Sound, and British Columbia. The results also support the current boundary in the Strait of Juan de Fuca, with the Hoko River being more closely related to the WC ESU and the Elwha River more closely related to the Puget Sound ESU. The sample from the Sol Duc Hatchery also appears to be intermediate between WC and Lower Columbia River populations, consistent with some prior studies.

The relationships between spring and fall runs are also consistent with prior studies. Spring- and fall-run samples within major areas (WC, Lower Columbia River, Puget Sound, British Columbia) are much more closely related to each other than they are to the same run types in other areas. This pattern extends even within the WC samples; the spring and fall runs from the Hoh River, for example, are more closely related to each other than either one is to the corresponding run in the Chehalis River. Within the Chehalis River basin, the report also found little genetic differentiation between run types from the same tributary. The authors note, however, that there is a potential problem for interpreting their results as they relate to run timing. Because the fish were sampled during the spawning season, the time of entry to freshwater was not known. Instead, the study used proxies, such as spawning date and fish condition, to infer individual run timing. If a significant number of salmon were incorrectly assigned, this would have the effect of obscuring any patterns of genetic differentiation between the run types. In addition, many of the samples were taken a decade or more ago, so might not be informative of current conditions.

Figure 10. Patterns of genetic similarity among Chinook salmon from the Washington Coast, the Lower Columbia River, British Columbia and the Strait of Juan de Fuca, and Puget Sound. This figure reproduces Brown et al. (2017), their Figure 10.

Thompson et al. (2019) explored the problem of potentially inaccurate run-time information by genotyping a subset of the samples used by Brown et al. (2017) for the Chehalis River basin based on a marker in the GREB1L gene region that is associated with run timing in Chinook salmon (reviewed by Waples et al. 2022). The marker was validated for the Chehalis River population using a set of known-timed samples obtained from fisheries near the mouth of the Chehalis River. Using the GREB1L marker as the indicator of run timing, the authors found that many of the salmon previously identified as spring-run based on spawning maturation did not have the spring-run genotype, thus indicating that they might be fall-run salmon previously misidentified as spring-run. Based on the GREB1L marker, spring-run Chinook salmon were only found in the upper portions of the basin, in the Newaukum, Skookumchuck, and Upper Chehalis Rivers. In the Newaukum River, the spring-run salmon were also partially differentiated from the fall-run salmon throughout the genome, indicating some (unquantified) degree of reproductive isolation.

Gilbertson and Bingaman (2023; cf. Gilbertson et al. 2021) used similar genotyping methods to examine variation at the GREB1L region in samples of fry, focusing only on fry samples from the upper portion of the Chehalis River basin. Based on genotypes from several thousand fry sampled between 2020 and 2022, they found that 69–83% were homozygous fall-run, 3–8% were homozygous spring-run, and 13–24% were heterozygous at the GREB1L marker. They calculated a statistic (F_{IS}) that indicated a degree of departure from random mating that amounted to a modest (13–26%) reduction compared to the expectation if the run types were mating in a random manner. They concluded that this degree of interbreeding shows that spring-run fish have lost their unique habitat and are at risk of being outcompeted by the numerically dominant fall-runs.

Spidle (2008) conducted a genetic study to examine the relationship between an introduced spring-run hatchery stock and the native summer-run population in the Sol Duc River. As was discussed above, a non-native spring-run stock originating from the Lower Columbia River was introduced into the Sol Duc River in the 1970s and propagated through the early 2000s. An apparently native summer run was subsequently used as broodstock, and also exists as a natural population in the river. Using samples collected in the early 2000s and genotypes at 13 microsatellite loci, Spidle (2008) found that the Sol Doc spring and summer runs were genetically very similar to each other, and that both were more similar to another WC population (Quillayute River fall-run) than to the Lower Columbia and Dungeness River populations that had contributed to the spring-run broodstock.

Genetics discussion and conclusions

The genetic information at neutral markers (microsatellites, SNPs) that has been collected since the prior 1998 status review continues to support the existing ESUs boundaries. The study by Brown et al. (2017) is particularly informative in this respect, as it includes numerous population samples from the WC and geographically proximate areas. These data fully support the existing southern and northern/eastern boundaries of the WC Chinook Salmon ESU originally designated by Myers et al. (1998; Figure 1).

The team also analyzed the question of whether the spring run within the currently defined WC Chinook Salmon ESU meets the criteria to be considered an ESU in its own right. After carefully considering this question, the team concluded that spring- and fall-run Chinook salmon on the WC are part of the same ESU. This conclusion is consistent with the 1998 review and with several other recent status reviews that have addressed the run-timing question for other Chinook salmon ESUs or steelhead DPSes (Anderson et al. 2018, Pearse et al. 2019, Ford et al. 2021). The reasons for this conclusion are similar to those in the earlier reports, and are summarized below.

For spring-run Chinook salmon along the WC to be an ESU, they would have to meet both of the two criteria in the NMFS ESU policy that are required for a group of salmon populations to be an ESU. These criteria are that, to be an ESU, a population or group of populations must: 1) be substantially reproductively isolated from other such groups, and 2) represent an important component of the evolutionary legacy of the species. The team concluded that the best available genetic information, including the new studies conducted in the decades since the last review in 1998, shows that spring-run WC Chinook salmon meet neither of these criteria.

Reproductive isolation

The available genetic data provide two different types of information related to reproductive isolation. First, patterns of selectively neutral variation (microsatellites, or SNPs that are scattered throughout the genome) can be used to infer the relative degree of reproductive isolation of populations over the course of many generations. Populations that are less isolated from each other, either due to more recent common ancestry or ongoing interbreeding, are genetically more similar to each other at neutral loci than are populations that are more isolated. These patterns can be visualized in the form of population trees like the one reproduced in Figure 10, which illustrates that populations from the WC (of all run timings) are more genetically similar to each other (and thus less reproductively isolated) than they are to populations in the Columbia River or Puget Sound.

The same tree also shows that spring-run fish are more similar to fall-run fish in the same river than they are to spring-run fish in other rivers. This general pattern is seen in many populations of coastal Chinook salmon (Myers et al. 1998, Waples et al. 2004, Moran et al. 2013, Waples et al. 2022). It even holds within the WC Chinook Salmon ESU itself; spring-run fish in the Hoh River, for example, are more similar to fall-run fish in the Hoh River than they are to spring-run fish in the Chehalis River. These patterns are indicative of relative, not absolute, isolation, but do indicate that within a river, spring- and fall-run fish are less isolated from each other than are fish from different rivers within the same ESU.

The SNPs within the GREB1L gene that have been shown to be associated with run timing provide another means of assessing reproductive isolation between spring and fall runs. In cases like the study of Thompson et al. (2019), where these SNPs correlate highly with run timing, the presence of heterozygous genotypes at these loci provides compelling, direct evidence of recent interbreeding between spring- and fall-run fish. Each of the available studies of GREB1L variation within the Chehalis River (there are not yet any equivalent data for other WC rivers) found substantial numbers of heterozygotes, thus concluding that interbreeding between spring- and fall-run fish was common (Thompson et al. 2019, Gilbertson et al. 2021, Gilbertson and Bingaman 2023).

Some of these studies, as well as the petition, hypothesize that this interbreeding is due solely or mostly to recent, anthropogenic changes to habitat, and that historically a high degree of reproductive isolation of spring-run fish in coastal populations must have been a requirement for their existence. The petition argues (e.g., on pp. ii, 9, and 24) that even though the genetic data indicate that there are high levels of interbreeding between spring- and fall-run fish, this interbreeding should be considered a threat that must be prevented rather than a natural characteristic of the system. However, this pattern of interbreeding is consistent with the coastwide patterns of neutral variation that have accumulated over many generations (see above), suggesting that such interbreeding is not just a recent phenomenon (Thompson et al. 2020, Waples et al. 2022). Using a different type of analysis, Thompson et al. (2020) concluded that interbreeding between spring- and fall-run Chinook salmon in the Klamath River had been going on for hundreds of years, for example, and that gene flow between spring- and fall-run fish, rather than being a threat, was important for the maintenance of genetic diversity and population viability. Consistent with this finding, in designating Chinook salmon ESUs coastwide, Myers et al. (1998) found that most coastal river systems do not contain the habitats required for early-run Chinook salmon to persist in isolation from late-run.

Based on these multiple and consistent lines of evidence, the SRT thus concluded by consensus that the best available information indicates that spring- and fall-run Chinook salmon in the WC are not sufficiently reproductively isolated to be considered distinct ESUs.

Evolutionary legacy

Previous status review reports discussed whether early (spring and summer) runs, as defined by GREB1L genotype, are an important component of the evolutionary legacy of the species (Anderson et al. 2018, Pearse et al. 2019, Ford et al. 2021). These reports uniformly concluded that variation in run timing is an important component of diversity, both within salmon species and within many of the currently defined Chinook salmon ESUs (e.g., Ford et al. 2021, pp. 22–23 for a summary). Recent scientific reviews (Ford et al. 2020, Waples et al. 2022) also concluded that intraspecific diversity in run timing is important to overall species viability. In this sense, there is little disagreement among salmon scientists that run timing diversity is important to conserve. However, in coastal Chinook salmon, variation in run timing is present as a polymorphic trait *within* numerous interbreeding populations. Like other polymorphic traits, such as age at maturity, length, or anadromy/residency (for steelhead), run timing does not, in general, map cleanly onto a discrete set of reproductively isolated populations. Instead, for coastal Chinook salmon, the best available data show that run timing exists as variation within closely related groups of populations, or even, in many cases, within a single interbreeding population.

The fact that the genetic basis of run timing has been shown to be largely concentrated in one genomic region does not change this observation, and indeed reinforces it. Variation contained within the GREB1L region may well be an important component of the evolutionary legacy of the species, but this variation does not, in the case of coastal Chinook salmon, exist in reproductively isolated populations. Ignoring the first prong of the ESU policy (isolation) and focusing only on the second (importance), as the petition proposes, would therefore create incoherent ESUs. For example, the studies on GREB1L variation in the Chehalis River show that there is frequent interbreeding of fish with alternative genotypes at GREB1L (Thompson et al. 2019, Gilbertson et al. 2021, Gilbertson and Bingaman 2023). This implies that families will contain siblings with alternative GREB1L genotypes. If the currently defined ESU were to be split on the basis of these genotypes, as the petition proposes, it would result in the nonsensical situation in which full siblings would frequently be different “species” under the ESA.

Even if the very small degree of reproductive isolation between spring- and fall-run Chinook salmon in the same river were considered sufficient to meet the first prong of the ESU policy, the result would not be simply splitting each coastal Chinook salmon ESU into two components, because this would again create incoherent ESUs in which large portions of each ESU would be more closely related to individuals in other ESUs than in the same ESU. Instead, spring and fall runs within each individual river would need to be put into separate ESUs. This extreme case of taxonomic splitting would result in potentially scores of new ESUs, a situation the NMFS ESU policy was explicitly designed to avoid (Waples 1991).

Based on the above considerations, the team concluded that spring-run Chinook salmon spawning in Washington coastal rivers, considered on their own, are not an ESU.

Artificial propagation

Artificial propagation is considered in the identification of the ESU, in that hatchery broodstocks created by importing Chinook salmon from outside of the ESU would not be considered part of the ESU. Further, the interaction of hatchery-origin and native natural-origin Chinook salmon may affect the status of naturally spawning populations from the standpoint of ESU membership and extinction risk.

Supplementing Chinook salmon natural production with hatcheries began in Washington coastal rivers at the turn of the previous century. The first Chehalis River hatchery began operation in 1899, and the Willapa River Hatchery (Trap Creek) began operation in 1900. These hatcheries were built as a response to degraded habitat and declining salmon runs (Little 1898).

Information on historical hatchery operations provides some details on past introductions of non-native Chinook salmon populations into WC Chinook Salmon ESU rivers. Additionally, hatchery reports contain information on the abundance and run timing of local populations, and on the watershed conditions, river conditions, and effects of local fisheries.

Initial hatchery production in the WC was predominantly in the Grays Harbor and Willapa Bay basins, with more northerly hatchery releases beginning in the Quinault River in 1914 (Cobb 1930). Much of the early salmon hatchery production focused on the release of unfed or fed fry (fish < 2 g), and the success of these releases, although numerically large, was likely negligible (Myers et al. 1998). Fry releases from WC hatcheries continued until 1940, when extended hatchery rearing and better hatchery feeds became available (Wendler and Deschamps 1955b). To some extent, early (pre-1940) programs likely mined the rivers of returning naturally produced adults for broodstock, rather than supplementing the natural populations, further depressing natural run sizes. An additional consequence of these early hatchery programs was the importation of Chinook salmon from distant areas, predominantly from other hatcheries on the Lower Columbia River, to supplement coastal hatchery egg collection shortfalls. Out-of-ESU hatchery stocks continued to be imported into the late 20th Century; notably: 1) a Cowlitz (Lower Columbia River ESU) × Umpqua (Oregon Coast ESU) hybrid spring-run was established at the Sol Duc Hatchery, and 2) Lower Columbia River ESU and Puget Sound ESU fall-run hatchery stocks were imported into Grays Harbor and Willapa Bay (Myers et al. 1998). The majority of the out-of-ESU introductions were terminated in the 1980s, but the Sol Duc Hatchery spring-run Chinook salmon program was not discontinued until 2006.

Current hatchery production

Presently, hatcheries in the WC Chinook Salmon ESU are operated by a variety of entities: the U.S. Fish and Wildlife Service (USFWS), WDFW, tribal nations, and nonprofits (Table 4, Appendix A). All of the hatcheries are operated for harvest goals except the Bingham Creek Hatchery, which was identified as a conservation hatchery (Anderson et al. 2020). Hatchery production across the ESU has focused on the release of subyearling fall-run Chinook salmon, with projected annual releases of 14.2 million juveniles. The release target for summer-run Chinook salmon, solely from the Sol Duc Hatchery, is 1.5 million, with recent releases of 840,000 subyearling and 300,000 yearling juveniles. The majority of these releases are marked with a clipped adipose fin, and in most cases a portion of each hatchery release is coded-wire tagged. External marking allows the identification of the origin of fish in fisheries, thus allowing for selective harvest and for broodstock composition goals to be reached. Marking also allows for estimation of the proportion of hatchery fish spawning naturally. Coded-wire tagging allows for area-specific estimates of harvest in ocean and freshwater fisheries, smolt-to-adult survival, and straying (recovery in non-natal basins). Further, the vast majority of off-station releases of juvenile Chinook salmon has been discontinued to allow for better return rates to the hatchery rack and less potential for spawning naturally with native Chinook salmon.

Of the 13 WC Chinook Salmon ESU hatcheries (Figure 11) that release Chinook salmon juveniles, only four were operated as segregated programs (i.e., they do not incorporate unmarked natural-origin broodstock); nine were operated as integrated programs (WDFW 2024). For these integrated hatcheries, WDFW reported that 10–30% of the broodstock utilized were of natural origin (Anderson et al. 2020).

Recent hatchery production in WRIA 19, the Strait of Juan de Fuca, has solely been from the Hoko River Hatchery, with releases averaging 424,000 from 2012–21 (Figure B1). In WRIA 20, the northwestern coastal portion of the ESU, hatchery production has averaged 2.09 million juvenile Chinook salmon, with the majority of fall-run production coming from the Makah National Fish Hatchery (NFH; 1.6 million) and summer-run Chinook salmon coming from the Sol Duc Hatchery (666,000) for 2012–22 (Figure B2). For WRIA 20, hatchery production emphasized releases of fall-run Chinook salmon from the Quinault Lake Hatchery, averaging 760,000 annually, and the Salmon River Fish Culture Facility, Queets River, averaging 195,000 from 2012–21 (Figure B3). Collectively, the hatcheries in WRIAs 22 and 23, Grays Harbor and the Chehalis River, have released an average of 807,000 annually from 2012–21 (Figure B4). Finally, in WRIA 24, releases of fall-run Chinook salmon from the Forks Creek Hatchery, Willapa River, have averaged 1.5 million over the last 10 years, but current annual releases have been reduced to 400,000 (Figure B5). The Nemah and Naselle River hatcheries have collectively averaged annual releases of 5.2 million fall-run Chinook salmon subyearlings; current combined production goals for the two hatcheries are 8.3 million subyearling juveniles (Figure B6). Hatchery production trends over the last 15 years have focused on

Table 4. Chinook salmon hatchery programs in the WC Chinook Salmon ESU.

Hatchery program	WRIA	Operation	Location of juvenile releases	Run	Program goal (egg take)	Planned release	Type of broodstock program
Hoko Falls Hatchery	19	Tribal	Hoko R	Fall	500,000	Hoko R 370,000 Little Hoko R 50,000	Integrated
Educket Creek	20	Tribal	Waatch R	Fall	100,000	Educket Cr 100,000	Integrated (mixed)
Makah NFH[a]	20	FWS/Tribal	Tsoo-Yess R	Fall	2,700,000	Tsoo-Yess R 2,200,000	Integrated (mixed)
Lonesome Creek	20	Tribal	Sol Duc R	Summer	n/a	*transferred to Sol Duc R*	Segregated
Bear Springs	20	Tribal	Sol Duc R	Summer	n/a	*transferred to Sol Duc R*	Segregated
Sol Duc Hatchery	20	WDFW	Quillayute R	Summer	1,500,000	Sol Duc R 1,140,000	Integrated (mixed)
Salmon River FCF[b]	21	Tribal	Queets R	Fall	250,000	Salmon R 200,000	Segregated
Quinault Lake Complex	21	Tribal	Quinault R	Fall	1,500,000	Quinault R 1,300,000	Segregated
Lake Aberdeen Hatchery	22	WDFW	Wynoochee R	Fall	75,000	Van Winkle Cr 50,000	Integrated
Humptulips Hatchery	22	WDFW	Humptulips R	Fall	590,000	Stevens Cr 500,000	Integrated (mixed)
Satsop Spring Hatchery	22	CBFTF[c]	Satsop R	Fall	600,000	Satsop River 300,000	Integrated
Bingham Cr Hatchery	22	WDFW	Satsop R	Fall	*varies*[d]	Satsop R 200,000	Integrated
Forks Creek Hatchery	24	WDFW	Willapa R	Fall	440,000	Fork Cr 400,000	Integrated (mixed)
Naselle Hatchery	24	WDFW	Naselle R	Fall	5,500,000	Naselle R 5,000,000	Integrated (mixed)
Nemah Hatchery	24	WDFW	North Nemah R	Fall	3,700,000	N Nemah R 3,300,000	Segregated

[a] *NFH* = National Fish Hatchery.
[b] *FCF* = Fish Culture Facility.
[c] CBFTF: Chehalis Basin Fisheries Task Force. Integrated Program information from 2024 Future Brood Document Draft, 1108 p. (https://wdfw.wa.gov/sites/default/files/publications/02295/all_alpha_2022_2nd_draft.pdf)
[d] Received from Satsop Spring.

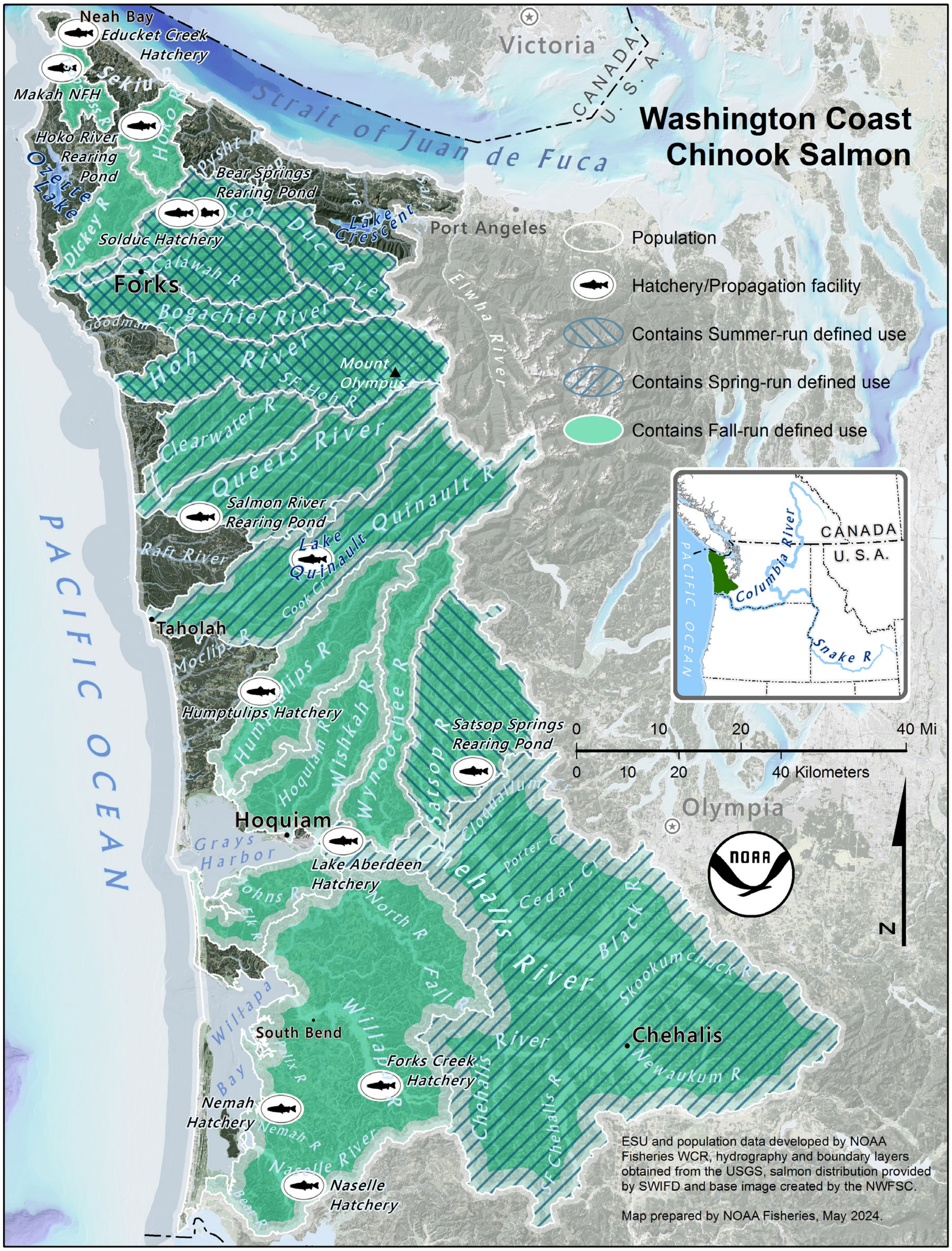

Figure 11. State, tribal, and federal hatchery locations, and related natural runs of Chinook salmon, in the WC Chinook Salmon ESU.

larger fall-run programs that are generally able to collect sufficient numbers of returning adults to meet production goals. Out-of-ESU introductions have been eliminated, and exchanges among hatcheries are limited to within-basin or within-WRIA.

The influence of hatchery-origin adults spawning naturally has been monitored to a limited extent. In general, the proportion of hatchery-origin spawners (pHOS) is higher in rivers in WRIA 24, Willapa Bay (Table 5), which contains the largest hatchery programs in the ESU and degraded natural spawning habitat that limits natural production (Smith 1999b, Smith and Wenger 2001, NWIFC 2020).

ESU designation: Overall conclusion

Previous work by Myers et al. (1998) identified the populations with shared genetic, life-history, and ecological habitat characteristics that would constitute the WC Chinook Salmon ESU.

> Coastal populations spawning north of the Columbia River and west of the Elwha River are included in this ESU. These populations can be distinguished from those in Puget Sound by their older age at maturity and more northerly ocean distribution. Allozyme data also indicates geographical differences between populations from this area and those in Puget Sound, the Columbia River, and the Oregon coast ESUs. Populations within this ESU are ocean-type Chinook salmon and generally mature at ages 3, 4, and 5. Ocean distribution for these fish is more northerly than that for the Puget Sound and Lower Columbia River ESUs. The boundaries of this ESU lie within the Coastal Ecoregion, which is strongly influenced by the marine environment: high precipitation, moderate temperatures, and easy migration access (Myers et al. 1998, pp. 121).

The SRT reviewed the criteria utilized to establish composition of the WC Chinook Salmon ESU and concluded that, since the time of the original status review, subsequent genetic analyses (see Genetics), life-history monitoring, ocean distribution data, and ecological typing have confirmed the distinctness of populations in this ESU.

The SRT also considered the Petitioners' scenario where the ESU would be partitioned into two ESUs based on run timing of spring/summer (early) vs. fall (late), and the underlying genetic polymorphism (Prince et al. 2017, Thompson 2019, Thompson et al. 2020, Waples et al. 2022). In general, populations with different run times in the same basin were more genetically similar than similar run times in different basins. Although early- and late-returning populations are different both biologically and ecologically, the SRT concluded that these differences do not rise to a level that justifies creating distinct ESUs; this comports with the findings of the SRTs for the Olympic Peninsula Steelhead DPS and the Oregon Coast and Southern Oregon/Northern California Coastal Chinook Salmon ESUs (OC and SONCC Status Review Team 2023).

Table 5. Proportion hatchery-origin fall-run adults spawning naturally observed in selected WC Chinook Salmon ESU rivers.

Year	Hoko R	Quinault R	Wishkah R	Humptulips R	Willapa R	Nemah R	Naselle R
1980	—	0.444	—	—	—	—	—
1981	—	0.215	—	—	—	—	—
1982	—	0.234	—	—	—	—	—
1983	—	0.347	—	—	—	—	—
1984	—	0.303	—	—	—	—	—
1985	—	0.287	—	—	—	—	—
1986	—	0.312	0.000	0.192	—	—	—
1987	—	0.465	0.000	0.228	—	—	—
1988	0.058	0.486	0.000	0.020	—	—	—
1989	0.126	0.547	0.000	0.060	—	—	—
1990	0.614	0.507	0.000	0.032	—	—	—
1991	0.856	0.503	0.056	0.041	—	—	—
1992	0.397	0.160	0.121	0.032	—	—	—
1993	0.196	0.467	0.121	0.058	—	—	—
1994	0.325	0.265	0.120	0.042	—	—	—
1995	0.669	0.265	0.120	0.036	—	—	—
1996	0.651	0.138	0.121	0.029	—	—	—
1997	0.538	0.218	0.121	0.057	—	—	—
1998	0.578	0.115	0.121	0.095	—	—	—
1999	0.558	0.166	0.120	0.103	—	—	—
2000	0.598	0.338	0.119	0.088	0.657	0.676	0.841
2001	0.452	0.041	0.120	0.086	0.676	1.000	0.783
2002	0.684	0.732	0.121	0.146	0.750	0.981	0.823
2003	0.593	0.599	0.121	0.127	0.668	0.889	0.825
2004	0.730	—	0.121	0.067	0.825	0.924	0.834
2005	0.744	—	0.121	0.087	0.790	0.945	0.859
2006	0.812	—	0.121	0.235	0.685	0.856	0.858
2007	0.571	—	0.121	0.119	0.746	0.818	0.859
2008	0.875	—	0.121	0.058	0.826	0.916	0.861
2009	0.903	—	0.121	0.149	0.773	0.956	0.854
2010	0.013	—	0.121	0.049	0.685	0.880	0.817
2011	0.156	—	0.121	0.134	0.703	0.929	0.867
2012	0.681	—	0.121	0.124	0.661	0.823	0.915
2013	0.069	—	0.259	0.332	0.771	0.871	0.815
2014	0.042	—	0.094	0.164	0.737	0.989	0.810
2015	0.485	—	0.291	0.296	0.699	0.981	0.685
2016	0.920	—	0.000	0.209	0.808	0.690	0.749
2017	0.526	—	0.071	0.602	0.756	0.895	0.256
2018	0.108	—	0.088	0.427	0.542	0.863	0.553
2019	0.877	—	0.088	0.535	0.609	—	—
2020	0.787	—	0.030	0.285	0.720	—	—
2021	—	—	0.119	0.155	0.675	—	—
2022	—	—	—	0.168	0.586	—	—
2023	—	—	—	0.430	0.702	—	—

Demographics

Historical information

There is minimal historical information available regarding Chinook salmon abundance and distribution in the WC Chinook Salmon ESU. Harvest data provide the earliest quantitative measure of Chinook salmon abundance, and can also include information related to run timing. The first salmon canneries were established in the late 1800s; however, cannery packs do not provide catch information for many tributaries, and even in those tributaries that contained canneries (predominantly in Grays Harbor and Willapa Bay), it is not clear what proportion of the catch was canned, what the harvest rates were, or even if the fish canned came from those basins. Even so, cannery pack data (1898–1928) provide at least a partial picture of Chinook salmon abundance (Table 6), with a maximum pack of 115,593 fish in 1911, a time when hatchery production likely had a minimal positive influence on abundance. Finally, early cannery pack data need to be put in the context of contemporary narratives provided by fisheries managers. Little (1898, pp. 38–39), Washington State Fish Commissioner, stated:

> In several of the main streams of the Willapa Harbor district, not twenty-five per cent as many salmon reach the main spawning grounds in the streams as did some years ago. On quite a number of the streams extensive logging operations have been carried on, and dams have been placed in the streams with no proper means for the ascent of the salmon, and also a great amount of refuse, in times past, have been cast into these streams.
>
> What is true of the Willapa Harbor district is also true of the Gray's Harbor county. Several of the large tributaries of Gray's Harbor, notably the Hoquiam and Wishkah rivers, that ten or twelve years ago teemed with thousands of salmon, at the present time hardly a fish of that character is to be found. This decrease has been caused mainly on account of extensive logging operations carried on in these rivers. Large dams have been built for the purpose of splashing the streams in order to drive the logs to tidewater, and even had these dams, which they have not, proper fishways for the ascent of the fish, the flooding of the streams during the period when the salmon spawn, in its most critical stage, must inevitably drive the fish entirely from the river.

Fisheries catch statistics provide the majority of information related to presence and abundance, but limited salmon surveys were conducted in 1930 in the Grays Harbor and Willapa Bay tributaries. At the time, the Chinook salmon run in the Chehalis River, including Wildcat Creek and Cloquallam Creek [sic], was rated as a "medium run," and Chinook salmon in the Wishkah River, Humptulips River, and Stevens Creek were rated as "scarce" for surveys conducted in mid-September (Pollock 1932).

Later reports provided an overview of fisheries harvests from 1935–70 (Table 7). Of the harvests reported, spring and summer catches (April–July) were noted for the major watersheds: Quillayute, Hoh, Queets, and Quinault Rivers, and Grays Harbor; all basins

Table 6. Cannery pack reported for WC Chinook salmon, 1892–1928 (Cobb 1930). Fish numbers are based on a case consisting of 48 one-pound cans with canning wastage and harvest loss and discard accounting for 50% of catch, with an average fish weight of 10 kg (22 lb; Myers et al. 1998).

	Sol Duc R		Hoh R		Queets R		Quinault R		Grays Harbor		Willapa Bay		
Year	Cases	Fish	Cases	Fish	Cases	Fish	Cases	Fish	Cases	Fish	Cases	Fish	Total
1892	—	—	—	—	—	—	—	—	4,500	19,636	3,000	13,091	32,727
1893	—	—	—	—	—	—	—	—	4,500	19,636	1,700	7,418	27,055
1894	—	—	—	—	—	—	—	—	12,300	53,673	2,700	11,782	65,455
1895	—	—	—	—	—	—	—	—	56	244	4,636	20,230	20,474
1896	—	—	—	—	—	—	—	—	7,816	34,106	4,551	19,859	53,965
1897	—	—	—	—	—	—	—	—	3,100	13,527	8,100	35,345	48,873
1898	—	—	—	—	—	—	—	—	5,100	22,255	5,865	25,593	47,847
1899	—	—	—	—	—	—	—	—	5,000	21,818	5,650	24,655	46,473
1900	—	—	—	—	—	—	—	—	6,700	29,236	6,700	29,236	58,473
1901	—	—	—	—	—	—	—	—	—	—	—	—	—
1902	—	—	—	—	—	—	—	—	4,000	17,455	5,836	25,466	42,921
1903	—	—	—	—	—	—	—	—	—	—	2,300	10,036	10,036
1904	—	—	—	—	—	—	—	—	4,339	18,934	3,000	13,091	32,025
1905	—	—	—	—	—	—	—	—	2,050	8,945	4,650	20,291	29,236
1906	—	—	—	—	—	—	—	—	2,500	10,909	4,000	17,455	28,364
1907	—	—	—	—	—	—	—	—	1,000	4,364	3,530	15,404	19,767
1908	—	—	—	—	—	—	—	—	1,000	4,364	4,017	17,529	21,892
1909	—	—	—	—	—	—	—	—	5,721	24,964	1,455	6,349	31,313
1910	—	—	—	—	—	—	—	—	15,495	67,615	2,923	12,755	80,369
1911	—	—	—	—	—	—	5,000	21,818	15,773	68,828	5,717	24,947	115,593
1912	414	1,807	—	—	750	3,273	—	—	9,060	39,535	6,123	26,719	71,332
1913	206	899	—	—	1,082	4,721	—	—	1,253	5,468	67	292	11,380
1914	237	1,034	—	—	1,175	5,127	51	223	11,899	51,923	2,924	12,759	71,066
1915	388	1,693	—	—	—	—	1,144	4,992	4,219	18,410	3,148	13,737	38,832
1916	—	—	—	—	1,506	6,572	1,365	5,956	12,400	54,109	5,115	22,320	88,957
1917	—	—	372	1,623	713	3,111	309	1,348	12,124	52,905	1,720	7,505	66,493
1918	—	—	60	262	381	1,663	1,497	6,532	8,731	38,099	921	4,019	50,575
1919	—	—	18	79	450	1,964	165	720	4,370	19,069	1,152	5,027	26,858

Table 6 (continued). Cannery pack reported for WC Chinook salmon, 1892–1928.

	Sol Duc R		**Hoh R**		**Queets R**		**Quinault R**		**Grays Harbor**		**Willapa Bay**		
Year	**Cases**	**Fish**	**Cases**	**Fish**	**Cases**	**Fish**	**Cases**	**Fish**	**Cases**	**Fish**	**Cases**	**Fish**	**Total**
1920	—	—	524	2,287	—	—	—	—	337	1,471	62	271	4,028
1921	—	—	366	1,597	—	—	—	—	942	4,111	623	2,719	8,426
1922	—	—	—	—	—	—	150	655	57	249	1,168	5,097	6,000
1923	—	—	—	—	100	436	512	2,234	3,076	13,423	375	1,636	17,729
1924	—	—	—	—	127	554	—	—	35	153	169	737	1,444
1925	—	—	—	—	1,745	7,615	450	1,964	992	4,329	13,500	58,909	72,816
1926	—	—	—	—	592	2,583	—	—	—	—	1,491	6,506	9,089
1927	—	—	—	—	1,420	6,196	—	—	2,076	9,059	2,668	11,642	26,897
1928	—	—	—	—	—	—	—	—	—	—	502	2,191	2,191
1929	—	—	—	—	—	—	—	—	—	—	—	—	—
1930	—	—	—	—	—	—	—	—	—	—	—	—	—
												Mean	**38,527**
												Median	**31,669**

Table 7. In-river harvest of WC Chinook salmon, 1935–70 (Wendler and Deschamps 1955a, Brix and Kolb 1971, WDF 1973).

Year	**Hoko**	**Tsoo-Yess**	**Ozette**	**Quillayute**	**Hoh**	**Queets**	**Queets (spr/sum)**	**Quinault**	**Grays Harbor**	**Chehalis**	**Willapa Bay**	**Total**
1935	—	—	—	313	3	3,071	—	7,372	17,157	967	13,142	42,025
1936	—	—	—	1,700	919	2,979	—	3,385	7,485	147	15,258	31,873
1937	—	10	—	544	2,190	1,342	—	1,579	8,661	—	15,205	29,531
1938	—	22	—	983	1,194	2,667	—	2,991	9,042	243	14,318	31,460
1939	—	1	—	1,983	880	1,007	—	2,638	9,878	597	9,068	26,052
1940	—	31	—	1,530	737	1,198	—	1,583	8,052	557	12,182	25,870
1941	—	—	1	1,186	555	859	—	1,448	6,754	644	6,028	17,475
1942	—	5	—	1,345	396	—	—	2,442	5,545	2,268	9,033	21,034
1943	—	35	—	1,200	473	2,370	—	2,755	5,438	743	9,985	22,999
1944	—	4	—	1,456	413	2,305	—	1,069	5,288	595	6,350	17,480
1945	—	27	—	4,478	1,668	5,520	—	7,324	12,688	1,052	13,291	46,048
1946	—	35	—	2,772	250	4,867	—	1,850	6,155	525	8,384	24,838
1947	—	7	—	1,334	830	2,835	—	643	4,882	371	4,586	15,488

Table 7 (continued). In-river harvest of WC Chinook salmon, 1935–70.

Year	Hoko	Tsoo-Yess	Ozette	Quillayute	Hoh	Queets	Queets (spr/sum)	Quinault	Grays Harbor	Chehalis	Willapa Bay	Total
1948	—	698	491	1,920	1,040	1,698	—	1,719	5,835	5,079	6,823	25,303
1949	—	45	1,876	2,574	710	1,477	—	1,591	4,809	876	7,678	21,636
1950	—	1,851	1,629	1,913	1,398	2,025	—	5,187	7,665	113	7,459	29,240
1951	—	544	1,212	2,128	2,022	9,249	839	1,972	7,987	338	8,045	33,497
1952	422	69	394	3,061	747	4,967	1,355	5,591	9,494	204	10,050	34,999
1953	454	111	431	1,826	2,973	5,577	1,373	2,038	5,968	2,220	6,648	28,246
1954	723	127	823	2,064	830	3,325	467	1,986	4,762	1,884	6,065	22,589
1955	357	63	404	933	4,127	4,626	1,923	1,580	—	1,617	—	13,707
1956	112	15	241	988	2,414	4,409	2,289	3,455	—	1,310	—	12,944
1957	115	66	428	1,309	2,121	3,975	951	2,760	—	1,109	—	11,883
1958	250	53	147	1,859	3,099	4,280	834	3,148	—	1,177	—	14,013
1959	86	70	—	724	1,738	4,853	1,363	2,017	—	995	—	10,483
1960	329	159	—	1,480	1,298	3,627	1,285	2,695	—	813	—	10,401
1961	157	70	3	1,817	1,669	3,074	603	2,097	—	1,090	—	9,977
1962	180	100	—	3,425	1,936	3,222	1,047	2,111	—	1,444	—	12,418
1963	204	140	1	4,128	1,809	3,347	875	3,023	—	1,883	—	14,535
1964	428	127	1	7,085	2,662	4,309	1,210	1,521	—	1,293	—	17,426
1965	308	102	1	5,422	2,594	7,474	267	3,735	—	779	—	20,415
1966	322	39	—	7,388	3,653	9,375	676	3,909	—	827	6,710	32,223
1967	84	24	—	2,940	1,680	3,078	77	2,092	9,984	1,790	13,888	33,770
1968	72	39	—	3,419	1,535	4,159	763	2,068	14,067	1,304	11,043	36,402
1969	107	46	—	3,436	4,212	4,776	788	1,018	13,678	1,108	17,333	44,606
1970	170	78	1	2,417	2,929	6,611	1,079	1,807	14,773		23,213	51,999
Avg.	**257**	**146**	**476**	**2,363**	**1,658**	**3,844**	**1,003**	**2,672**	**8,585**	**1,117**	**10,471**	**24,858**

where early-returning (spring- or summer-run) Chinook salmon populations have been identified. Of the total catch, early-returning harvest was a relatively small proportion (< 10%), except in the Hoh River, where the catch of early-returning (e.g., spring/summer runs) was 26.4% of the total catch (Wendler and Deschamps 1955a). Additionally, early-returning fish were also harvested in Willapa Bay. Wendler and Deschamps (1955) indicate that there was a spring run in the North River; this spring run may correspond to the population that SASSI (WDF et al. 1993) identified as an early fall run of Chinook salmon in the North River. Further, Wendler and Deschamps (1955a) suggest that Chinook salmon catches had declined since the early 1900s. From 1913 to 1934, harvest in Grays Harbor averaged 28,221 Chinook salmon, with a peak catch of 84,647, compared to an average catch of 7,677 from 1935 to 1953, with a peak catch of 17,157 in 1935.[4] While in-river Chinook salmon harvests were stable or declining in various WC rivers during 1935–53, it is not possible to include fish that were caught in the offshore troll fisheries (for which salmon origin is unknown). On average, some 275,000 Chinook salmon were captured in the offshore fishery annually (Wendler and Deschamps 1955a).

In their report, Wendler and Deschamps (1955b) suggest that past and ongoing legacy effects from splash dams and gravel mining in Grays Harbor and Willapa Bay, in addition to municipal and pulp mill wastes, were possibly responsible for some of this decline.

Current Populations and Data Description

Abundance

This section provides a summary of adult Chinook salmon abundance data for the WC Chinook Salmon ESU. The ESU extends from the Pysht River, emptying into the Strait of Juan de Fuca in the north, to Willapa Bay, close to the Oregon border in the south (Figure 4, Table 8). For demographic analysis, populations are defined by both river and run timing, based on the Washington Stock Status Indicator naming system, with spring-, spring/summer-, summer-, and fall-run timings represented in the ESU. Hatchery influence is variable, with 13 hatcheries within the ESU. Here we plot the escapement data, fit a dynamic linear model, and calculate 15-year trends, five-year geometric means, and between-population cross-correlations.

Population definitions

We used escapement data from WDFW's salmon population indicators database[5] and the corresponding population definitions, with two exceptions. We combined two pairs of populations in Willapa Bay (Nemah–Palix and Naselle–Bear) based on basin size, proximity, and to be consistent with groupings in three datasets provided by WDFW and tribes (April 2024).[6] We also eliminated the Cook Creek population from this analysis due to a lack

[4] Fish traps were a major contributor to harvest until they were banned in 1934 (Wendler and Deschamps 1955).
[5] https://fortress.wa.gov/dfw/score/score/
[6] Dataset 1: B. Hoffman, Hoh Tribe and WDFW, personal communication. Age-specific run reconstruction for summer and fall Chinook salmon in the Hoh River, WA, 2000–23. Hoh Tribe collects the great majority of the scales used for aging through extensive sampling of their net fisheries; WDFW collects some scales and

of any recent data. This resulted in a total of 28 populations, with 19 fall populations and nine spring/summer populations with abundance time series data (Table 8), compared with the 33 populations identified in SASSI (WDF and WWTIT 1993). The other publicly available escapement data, provided in the Pacific Fisheries Management Council's (PFMC) blue book, are aggregated to larger population complexes (Table 8).

Run timing

Escapement numbers for the major run-timing groups are based on when redds are built. Redds constructed on or after 15 October are attributed to fall-run spawners, while redds constructed before 15 October are assigned to spring-, spring/summer-, or summer-run spawners, depending on the basin (Hoh Tribe, Quinault Indian Nation, Quileute Nation, and WDFW, personal communication). The early-run naming convention (i.e., spring, spring/summer, or summer) varies between populations (Table 8) and data sources (e.g., the WDFW database vs. the PFMC blue book).

Aggregates

We grouped the populations into three aggregates (Table 8): a north–fall aggregate extending down to and including the Quinault River populations, a south–fall aggregate which includes all Grays Harbor and Willapa Bay populations, and a spring/summer aggregate which includes all populations in the ESU with spring, spring/summer, or summer run timing. These aggregates provide structure for subsequent trend analyses and summaries.

Origin

Our goal was to assess the status of the natural-origin component of the different populations. In general, the risk assessment for an ESU is based on the status of the natural-origin stock, and hatchery-origin salmon are rarely included regardless of the broodstock origin. This required breaking escapement into natural- and hatchery-origin values in basins where hatchery- and natural-origin spawners co-occurred.

There are two summer-run Chinook hatcheries on the Sol Duc River, and 11 fall-run hatchery programs distributed throughout the geographic range of these populations (Figure 11). Therefore, in-river spawners (or escapement) may include hatchery-origin fish. For some populations there are no nearby hatcheries, resulting in very few hatchery-origin spawners in these populations (e.g., Hoh River), while for others, large local hatchery releases can result in substantial numbers of hatchery-origin fish spawning in the river (e.g., Willapa Bay populations).

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processes samples at WDFW aging lab. Dataset 2: T. Jurasin, Quinault Indian Nation, and M. Scharpf, WDFW, personal communication. Natural, hatchery, and total age-specific run size and harvest for the Chehalis River and Humptulips River fall and spring Chinook salmon populations, Grays Harbor, WA, 1970–2023. Dataset 3: B. McClellan and M. Wagner, WDFW, personal communication. Age-specific run reconstructions for fall Chinook salmon, Willapa Bay, WA, 2000–23. Naselle, Nemah, North Smith, and Willapa River tributaries to Willapa Bay.

Table 8. The populations used in the abundance analysis. The SRT aggregates describe the groups used in this analysis, while the PFMC aggregates are the population definitions used in the PFMC blue book. Spawner origin is assigned to one of three categories: *known* = both hatchery- (HOR) and natural-origin (NOR) fish, but the quantity of both is included; *natural* = fish are assumed to be predominantly natural-origin; *mixed* = both hatchery- and natural-origin fish, where the two are combined into a single escapement value. The reasoning behind the origin designations is included in the *Origin details* column.

Population	SRT aggregate	PFMC aggregate	Origin	Origin Details
Hoko fall	north–fall	Hoko sum/fall	known (HOR & NOR)	known origin
Dickey fall	north–fall	Quillayute fall	natural	no fall hatchery, assumed NOR
Sol Duc fall	north–fall	Quillayute fall	natural	no fall hatchery, assumed NOR
Quillayute–Bogachiel fall	north–fall	Quillayute fall	natural	no fall hatchery, assumed NOR
Calawah fall	north–fall	Quillayute fall	natural	no fall hatchery, assumed NOR
Hoh fall	north–fall	Hoh fall	natural	no fall hatchery, assumed NOR
Queets fall	north–fall	Queets fall	mixed	fall hatchery, assumed mixed
Clearwater fall	north–fall	Queets fall	natural	no fall hatchery, assumed NOR
Quinault fall	north–fall	—	mixed	fall hatchery, known before 2003, mixed after
Humptulips fall	south–fall	Grays Harbor fall	known (HOR & NOR)	fall hatchery, known origin
Hoquiam fall	south–fall	Grays Harbor fall	known (HOR & NOR)	known origin
Wishkah fall	south–fall	Grays Harbor fall	known (HOR & NOR)	known origin
Wynoochee fall	south–fall	Grays Harbor fall	natural	assumed NOR based on sum to provided aggregate
Satsop fall	south–fall	Grays Harbor fall	natural	assumed NOR based on sum to provided aggregate
Chehalis fall	south–fall	Grays Harbor fall	natural	assumed NOR based on sum to provided aggregate
North–Smith fall	south–fall	Willapa Bay fall	known (HOR & NOR)	known origin after 2000, unsure about before
Willapa fall	south–fall	Willapa Bay fall	known (HOR & NOR)	known origin after 2000
Nemah–Palix fall	south–fall	Willapa Bay fall	known (HOR & NOR)	known origin after 2000
Naselle–Bear fall	south–fall	Willapa Bay fall	known (HOR & NOR)	known origin after 2000
Sol Duc sum	spr/sum	Quillayute spr/sum	mixed	sum hatchery, assumed mixed
Quillayute–Bogachiel sum	spr/sum	Quillayute spr/sum	mixed	close summer hatchery, assumed mixed
Calawah sum	spr/sum	Quillayute spr/sum	mixed	close summer hatchery, assumed mixed
Hoh spr/sum	spr/sum	Hoh spr/sum	natural	no spring hatchery, assumed NOR
Queets spr/sum	spr/sum	Queets spr/sum	natural	no spring hatchery, assumed NOR
Clearwater spr/sum	spr/sum	Queets spr/sum	natural	no spring hatchery, assumed NOR
Quinault spr/sum	spr/sum	—	natural	no spring hatchery, assumed NOR
Satsop sum	spr/sum	Grays Harbor spr	natural	no spring hatchery, assumed NOR
Chehalis spr	spr/sum	Grays Harbor spr	natural	no spring/summer hatcheries, assumed NOR based on sum to provided aggregate

The escapement database includes two columns that provide information about spawner origin. We used the following rules when determining how to classify escapement data series with respect to the degree of hatchery influence (Table 8):

- Based on WDFW's Salmonid Population Indicators (SPI) Database,[7] for any escapement value where the Escapement Methodology column (SPI) was "Hatchery-origin spawners" or "Natural-origin spawners," we used the classification *hatchery* or *natural* respectively. Examples of hatchery-origin spawners include all years for the Hoko River and 2000 and later for some of the Willapa Bay populations.
- For populations with hatcheries in the basin or nearby where the column Production Type is either *composite* or *natural* and Escapement Methodology does not indicate origin, we assume that the origin is *composite* (i.e., unknown proportion of hatchery spawners). Examples include Quinault fall after 2003, Queets fall, and Quillayute summer populations (i.e., Sol Duc, Quillayute–Bogachiel, Calawah).
- For populations where there is no hatchery in the basin (or nearby) with the same run timing, we assume that the escapement is predominately natural-origin and use the *natural* designation. This would include all spring, spring/summer, and summer populations outside of the Quillayute River basin and fall populations in the Quillayute and Hoh River basins (where there are no fall-run hatcheries).
- For the Chehalis River basin fall-run populations (the Wynoochee, Satsop, and Chehalis Rivers), we assumed that the escapement values were of natural origin. This is because when we combined the escapement values from these populations with the other Chehalis River populations (with known origin, Hoquiam and Wishkah Rivers), this reproduced the total natural-origin escapement data provided by the tribes and WDFW in the run reconstruction data (Jurasin and Scharpf, personal communication, and Hoffman, personal communication).

This resulted in 15 populations where escapement was assumed to be predominantly of natural origin, eight populations where hatchery and natural escapement were counted separately (known), and five populations where escapement included both natural and hatchery fish and separate counts were not available (mixed; Table 9). For the abundance analyses, we used natural escapement where possible (origin = known or natural) and mixed counts of hatchery and natural fish when both natural and hatchery fish were present but no separate counts were available (i.e., populations with origin = mixed in Table 8).

Table 9. The assumed origin for populations in the different aggregates. Key: *known* = both hatchery- and natural-origin fish but included as separate counts; *mixed* = both hatchery- and natural-origin fish but only a combined count is available; *natural* = fish assumed to be predominantly natural-origin.

Origin	North–fall	South–fall	Spring/summer
known	1	7	0
mixed	2	0	3
natural	6	3	6

[7] https://data.wa.gov/dataset/WDFW-Salmonid-Population-Indicators-Database-SPI-M/x25s-cxg8/about_data

Population summaries

To quantify temporal patterns in WC Chinook salmon escapement we follow Ford et al. (2022) and use multivariate dynamic linear modeling (DLM) to estimate population-specific trends. DLMs serve to analyze time series and provide an estimate of the smoothed escapement trend after accounting for observation and process errors (see Ford et al. 2022 and citations therein for details). The model is applied independently to each of the three aggregates. In addition to the estimated escapement series for each population, we calculated total escapement over time for each aggregate (north–fall, south–fall, and spring/summer). The DLMs were developed as Bayesian models using the `Stan` statistical software as implemented in the R computing language (`R` v. 4.2.3, R Core Team 2022; `Rstan` v. 2.26.22, Stan Development Team 2023). We used a single observation variance and single process variance and covariance for all populations (equivalent to the options `R` = *diagonal and equal* and `Q` = *equalvarcov* in the R `MARSS` library).

For each population, we summarized the smoothed escapement estimates from the DLM by calculating five-year geometric means and 15-year linear trends fit to the logged escapement estimates at five-year intervals. Five and 15 years correspond to approximately one and three generations for these populations, since the average age is between four and five years.

We did not calculate a trend if < 10 of the 15 years had observed escapement values. The DLM estimates for the Hoko River did not capture the variability in the observed data during the last decade (Figure 12). Therefore, we did not use the DLM estimates to generate 15-year trends or five-year geometric means for the Hoko River.

The degree to which escapement time series for the different populations tracked each other was characterized by calculating the Pearson correlation coefficients between the logged raw escapement counts. Because periods of missing data varied by population, we used all complete escapement pairs for each population pair (`use` = *pairwise.complete.obs* for the R function `cor`).

Results

Reported and smoothed escapement

The DLM fit the escapement time series fairly well (Figure 12, Figure 13, Figure 14). Most north–fall populations (Figure 12) show relatively stable trends, with the exception of a peak in the late 1980s. In the last decade, Hoko River natural escapement was highly variable relative to the other populations, and the model did a poor job of capturing this variability. The Hoko River escapement, while separated into hatchery- and natural-origin, is likely heavily influenced by the hatchery component and thus does not track the other populations as well. Therefore, we do not report the 15-year trend or five-year geometric mean statistics for the Hoko River, since these are based on the DLM results.

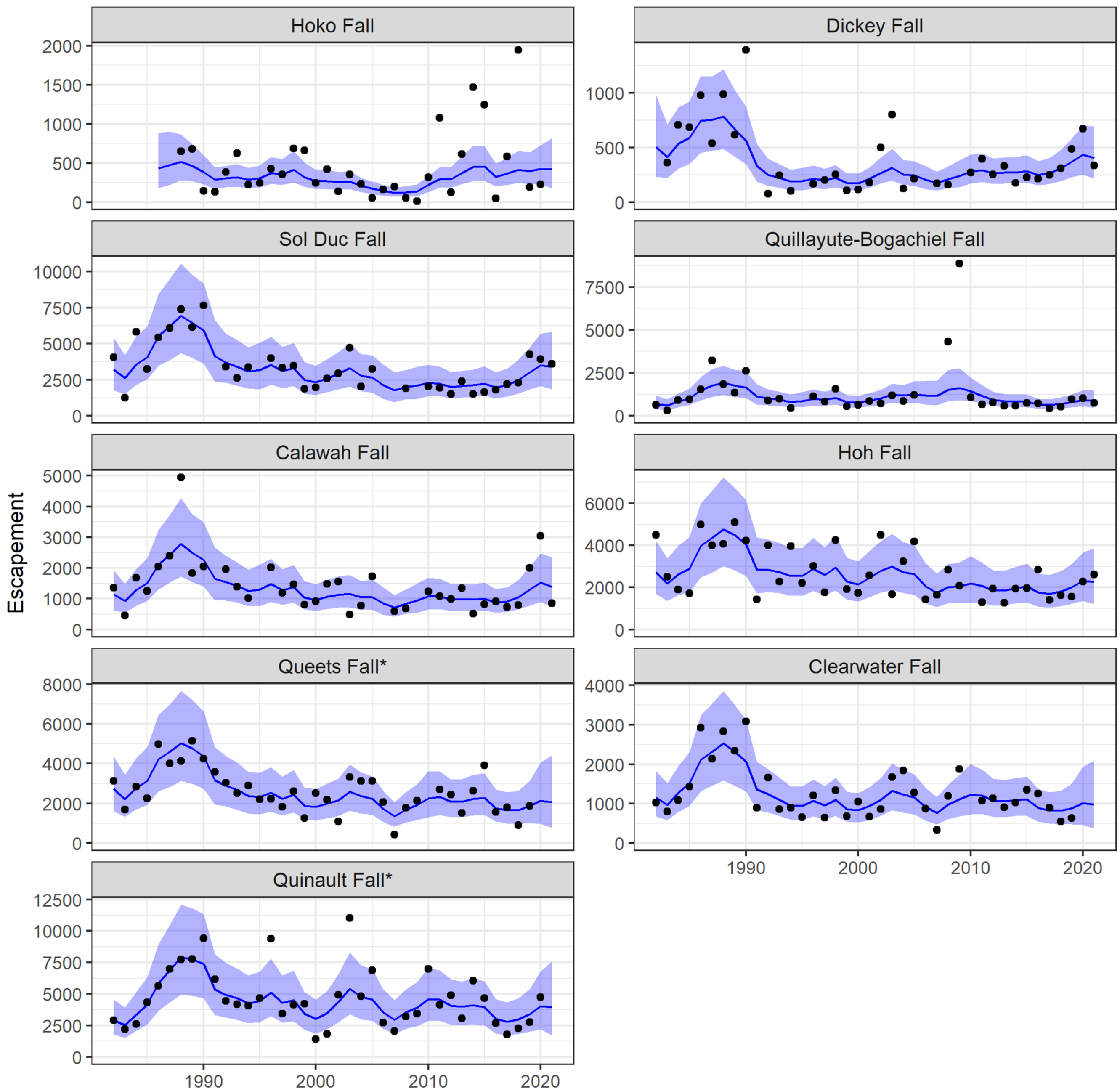

Figure 12. Escapement trends for WC Chinook salmon populations in the north–fall aggregate. Points are observations, blue lines and shaded areas are model predictions of abundance and 95% credible intervals. Starred population names (*) indicate that the escapement may include hatchery-origin spawners. For all other populations, the plotted escapement is natural-origin.

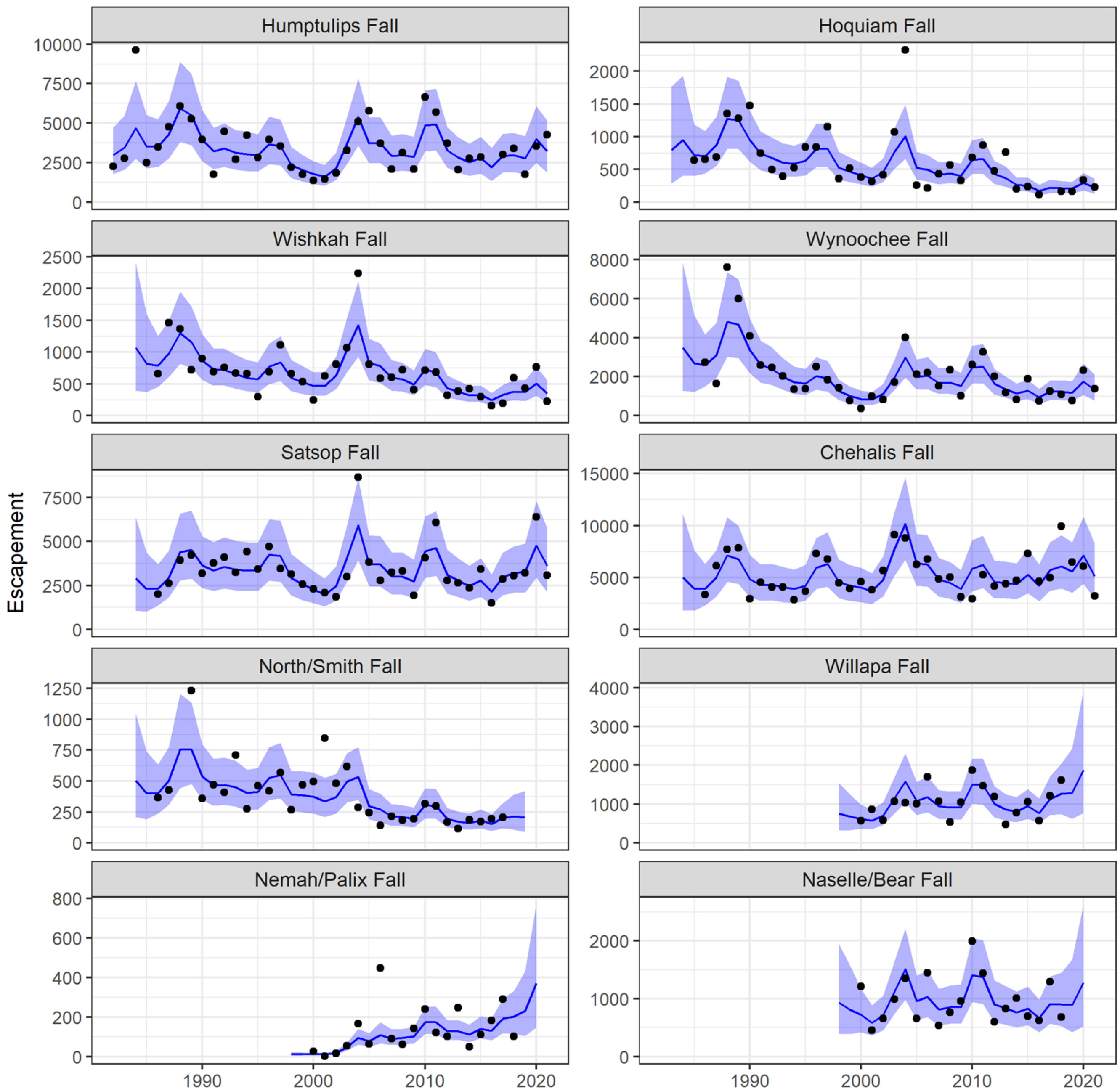

Figure 13. Escapement trends for WC Chinook salmon populations in the south–fall aggregate. Points are observations, blue lines and shaded areas are model predictions of abundance and 95% credible intervals. Starred population names (*) indicate that the escapement may include hatchery-origin spawners. For all other populations, the plotted escapement is natural-origin.

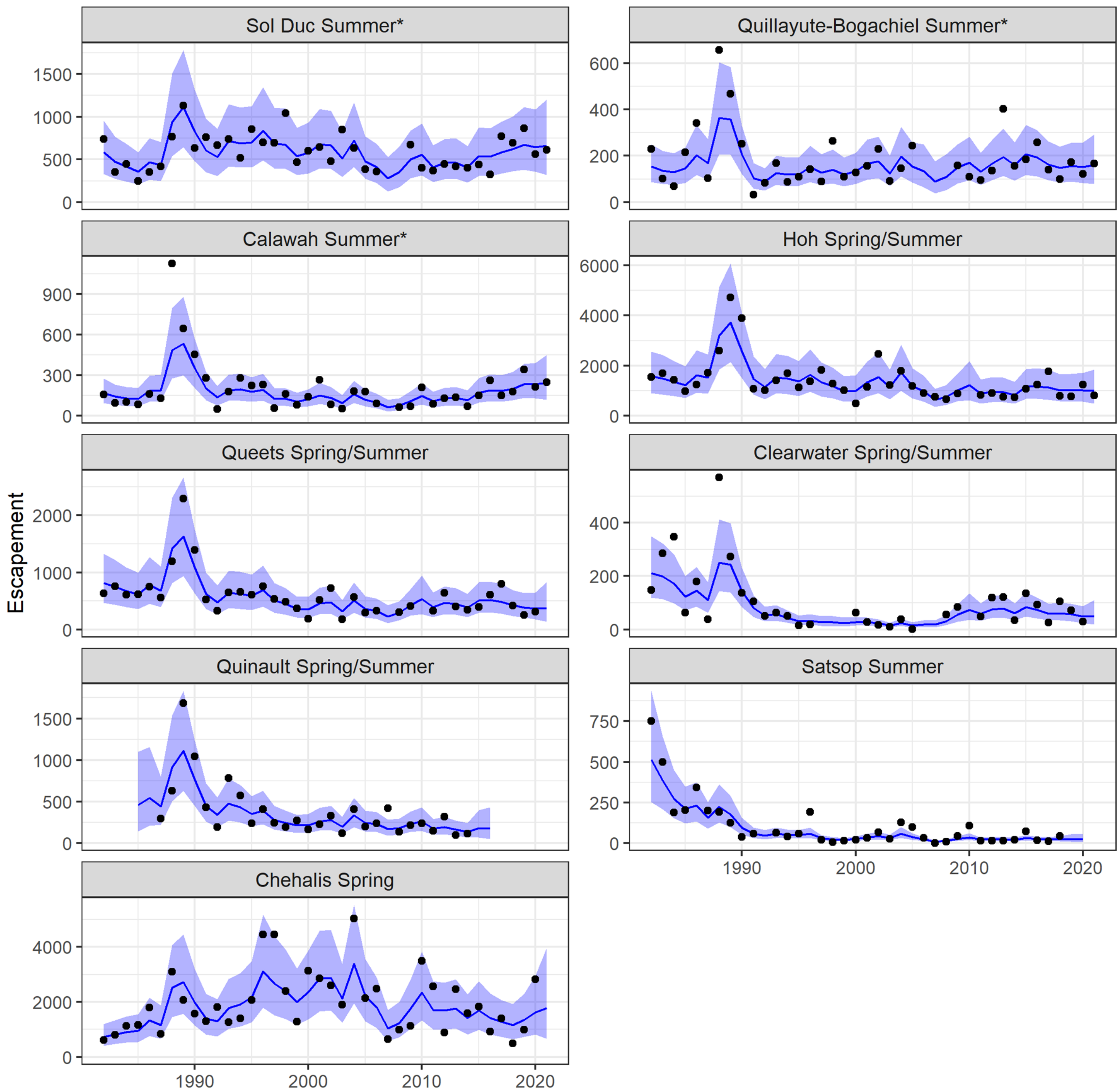

Figure 14. Escapement trends for WC Chinook salmon populations in the spring/summer aggregate. Points are observations, blue lines and shaded areas are model predictions of abundance and 95% credible intervals. Starred population names (*) indicate that the escapement may include hatchery-origin spawners. For all other populations, the plotted escapement is natural-origin.

The south–fall populations (Figure 13) did not display a consistent overall trend, with some populations declining (e.g., Hoquiam), others stable (e.g., Humptulips), and one increasing (Nemah–Palix) . In addition to the peak in the late 1980s, shared with the north–fall populations, many south–fall populations exhibited two additional peaks around 1997 and 2004.

The spring/summer aggregate (Figure 14) natural escapement values tended to be relatively stable over the last few decades, with the exception of a common peak in the late 1980s shared with the fall aggregate populations. There was some evidence for a decline in the Clearwater and Satsop populations from highs in the 1980s.

Total trends for the three aggregates capture some of the same patterns seen in the individual populations (Figure 15), with larger escapements in the late 1990s and less pronounced increases around 2004 and 2010.

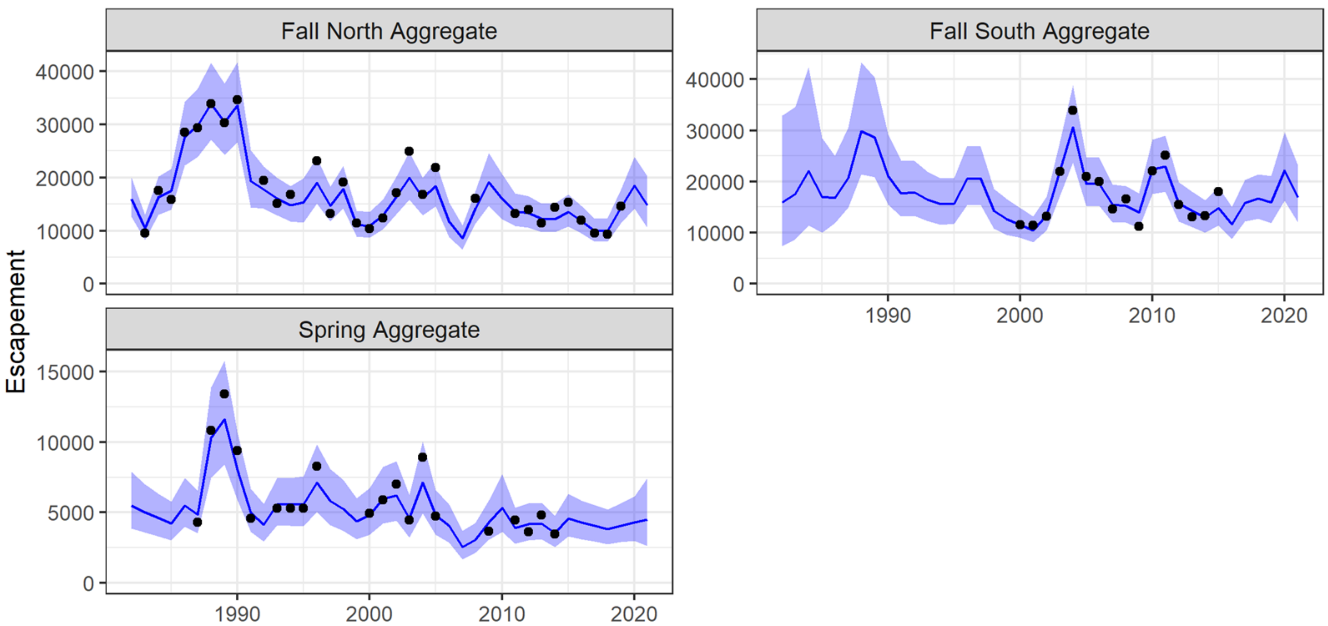

Figure 15. Combined escapement for three WC Chinook salmon aggregates: north–fall, south–fall, and spring/summer. The north–fall aggregate does not include the Hoko River population since the time series model does a poor job of fitting this population. Points are observations, blue lines and shaded areas are model predictions of abundance and 95% confidence intervals. Points are only included for years when escapement is available for all populations in the aggregate.

Between-population cross-correlations

Temporal patterns in escapement shared among populations provide estimates of cross-correlations between the log-transformed escapement series (Figure 16, Figure 17). In general, the northern fall-run populations were positively correlated (Figure 16). Many of the Grays Harbor populations that drain from the north (Humptulips, Hoquiam, Wishkah, and Wynoochee Rivers) also tended to correlate with the north–fall aggregate populations. Notice that the Hoko River population did not correlate with the other northern aggregate

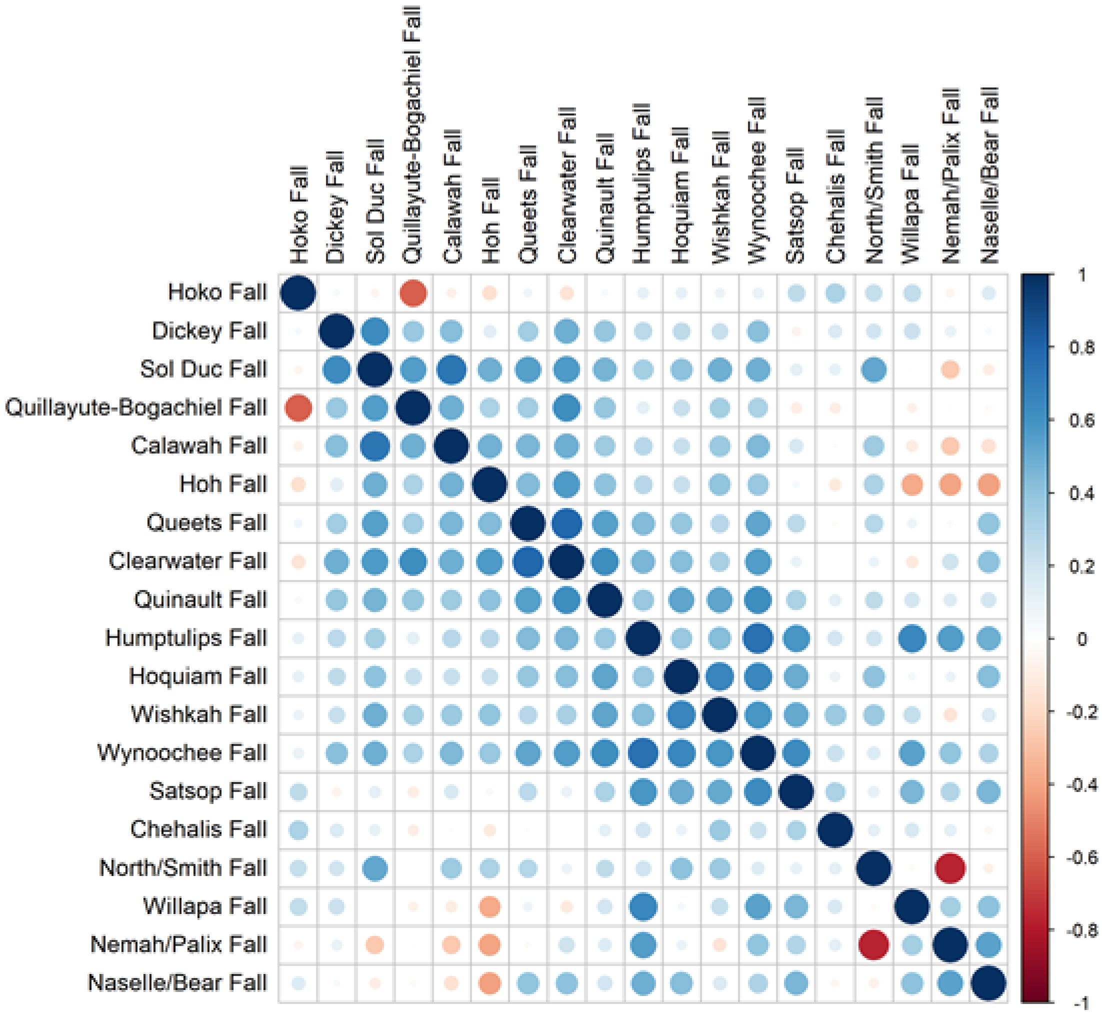

Figure 16. Between-population cross-correlations in escapements for fall populations.

populations. There was also some evidence for correlation between the south–fall aggregate populations, but not as pronounced as for the north–fall aggregate. There was some evidence for cross-correlation between the spring/summer aggregate populations (Figure 17).

15-year trends

The 15-year trends on the logged estimated escapement (from the DLM) tended to fall between –0.10 and 0.10 (Figure 18, Figure 19, Figure 20, Table 10). Values of 0.10 and 0.05 correspond to 10.5% and 5.1% annual rates of increase, respectively. For populations from all three aggregates, trends tended to be lower during the two periods from 1987–2001 and 2002–16. During the most recent 15-year period (2007–21), the trend tended to be flat or slightly increasing, with some exceptions.

5-year geometric means

Five-year geometric means (Table 11) tended to reflect the patterns seen in the trend analysis.

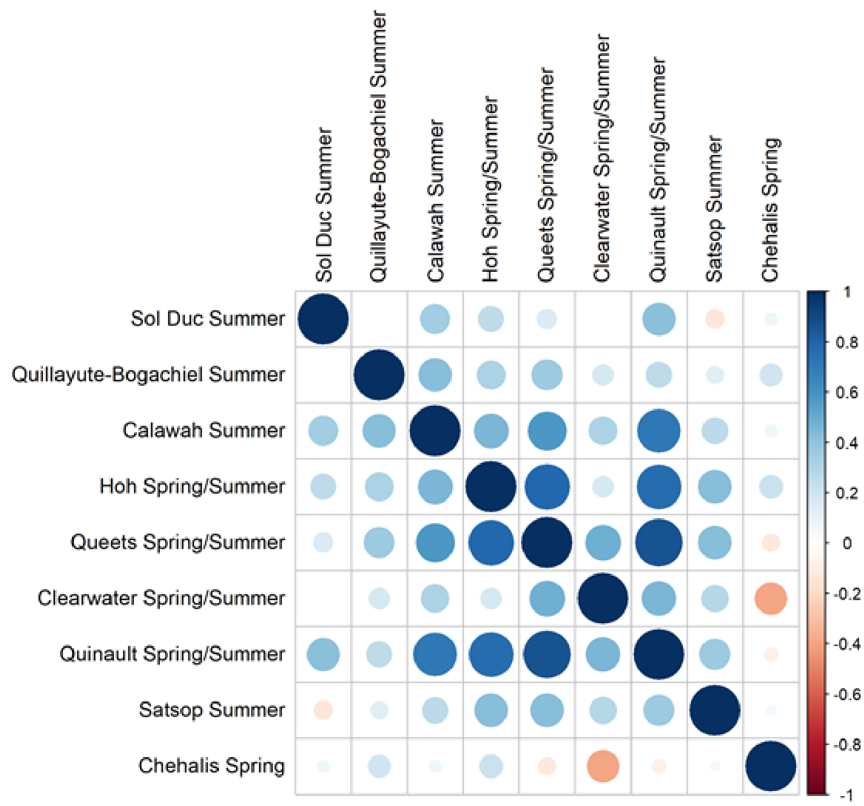

Figure 17. Between-population cross-correlations in escapement for spring/summer populations.

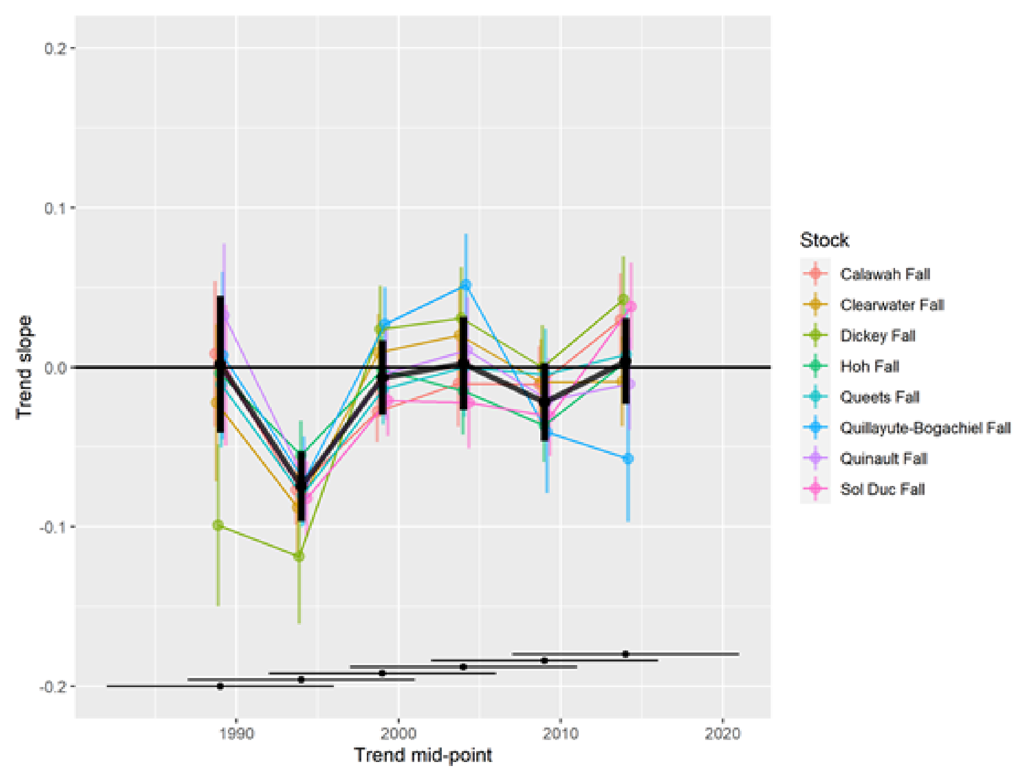

Figure 18. Annual rates of change over 15 years for predicted log escapement for the north–fall aggregate populations and the aggregate total (in black). Vertical bars represent 95% confidence intervals. Offset horizontal bars along the x-axis show the 15-year periods over which the trends are calculated. The Hoko River was not included as the DLM produced unreliable estimates for the last decade.

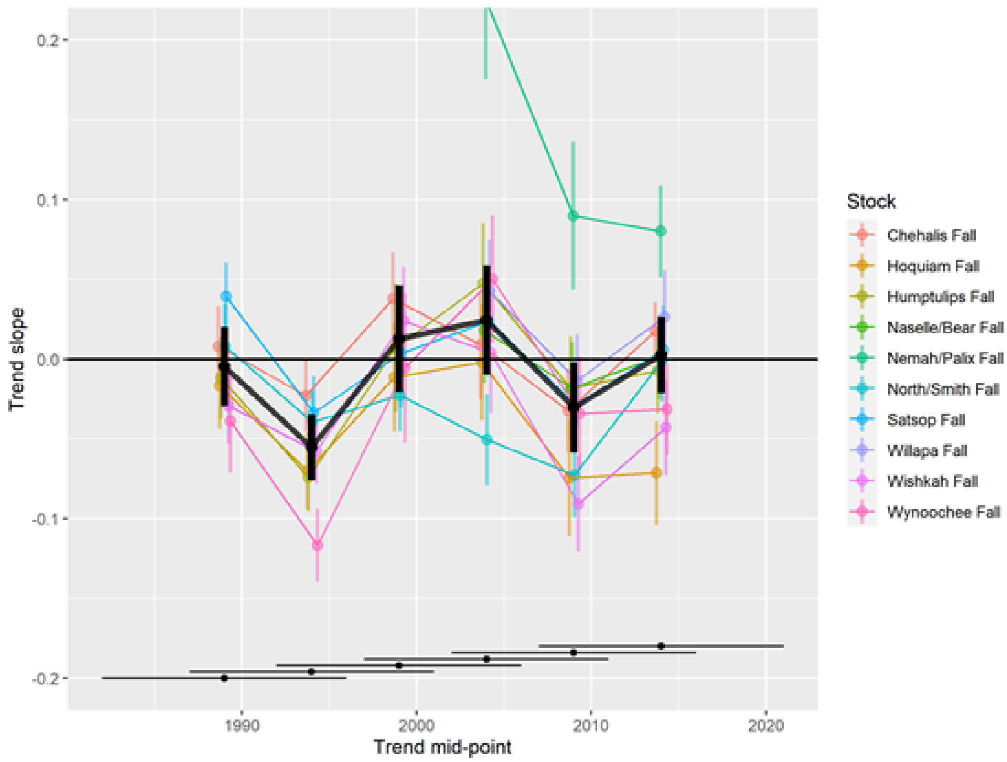

Figure 19. Linear 15-year trends for predicted log escapement for the south–fall aggregate populations and the aggregate total (in black). Vertical bars represent 95% confidence intervals. Offset horizontal bars along the x-axis show the 15-year periods over which the trends are calculated.

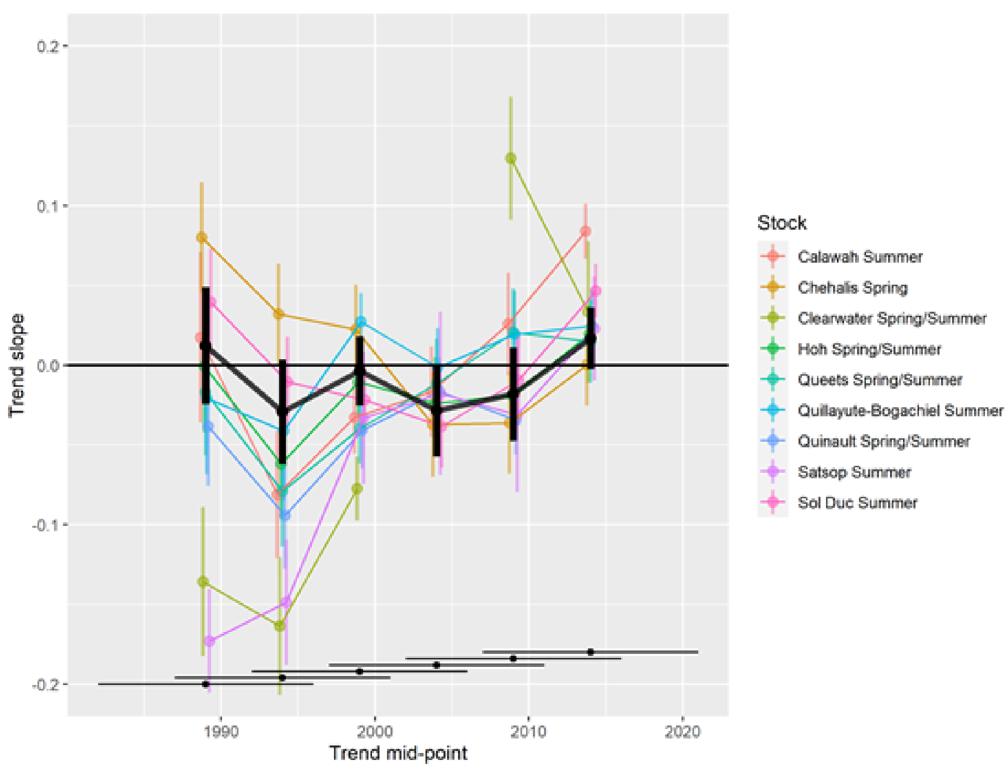

Figure 20. Linear 15-year trends for predicted log escapement for the spring/summer aggregate populations and the aggregate total (in black). Vertical bars represent 95% confidence intervals. Offset horizontal bars along the x-axis show the 15-year periods over which the trends are calculated.

Table 10. Fifteen-year trends (slope) in log Chinook salmon escapement computed from a linear regression applied to the predicted log escapement estimates. In parentheses are the upper and lower 95% confidence interval limits. Slopes estimates were replaced with *n/a* if observed escapement was missing for > 5 yr of any 15 yr period. Key: *f* = fall, *sp* = spring, *su* = summer, *agg* = aggregate.

Population	1982–96	1987–2001	1992–2006	1997–2011	2002–16	2007–21
Calawah f	0.01 (−0.04,0.05)	−0.08 (−0.10,−0.06)	−0.03 (−0.05,−0.01)	−0.01 (−0.04,0.02)	−0.01 (−0.03,0.01)	0.03 (0.00,0.06)
Clearwater f	−0.02 (−0.07,0.03)	−0.09 (−0.12,−0.06)	0.01 (−0.02,0.03)	0.02 (−0.01,0.05)	−0.01 (−0.04,0.02)	−0.01 (−0.04,0.02)
Dickey f	−0.10 (−0.15,−0.05)	−0.12 (−0.16,−0.08)	0.02 (0.00,0.05)	0.03 (0.00,0.06)	0.00 (−0.03,0.03)	0.04 (0.02,0.07)
Hoh f	0.00 (−0.04,0.03)	−0.06 (−0.08,−0.03)	0.00 (−0.02,0.02)	−0.01 (−0.04,0.01)	−0.04 (−0.06,−0.01)	0.00 (−0.02,0.03)
Queets f	−0.01 (−0.05,0.03)	−0.08 (−0.10,−0.06)	−0.01 (−0.04,0.01)	0.00 (−0.03,0.03)	0.00 (−0.03,0.02)	0.01 (−0.02,0.04)
Quillayute–Bogachiel f	0.01 (−0.05,0.06)	−0.07 (−0.10,−0.04)	0.03 (0.00,0.05)	0.05 (0.02,0.08)	−0.04 (−0.08,0.00)	−0.06 (−0.10,−0.02)
Sol Duc f	−0.01 (−0.05,0.04)	−0.08 (−0.11,−0.06)	−0.02 (−0.04,0.00)	−0.02 (−0.05,0.01)	−0.03 (−0.06,−0.01)	0.04 (0.01,0.07)
Quinault f	0.03 (−0.01,0.08)	−0.07 (−0.09,−0.05)	0.00 (−0.03,0.03)	0.01 (−0.02,0.04)	−0.0 2(−0.05,0.01)	−0.01 (−0.04,0.02)
North–fall agg	**0.00** **(−0.04,0.04)**	**−0.07** **(−0.10,−0.05)**	**−0.01** **(−0.03,0.02)**	**0.00** **(−0.03,0.03)**	**−0.02** **(−0.05,0.00)**	**0.00** **(−0.02,0.03)**
Chehalis f	0.01 (−0.02,0.03)	−0.02 (−0.05,0.00)	0.04 (0.01,0.07)	0.01 (−0.02,0.04)	−0.03 (−0.06,−0.01)	0.02 (0.00,0.04)
Hoquiam f	−0.02 (−0.04,0.01)	−0.07 (−0.09,−0.05)	−0.01 (−0.05,0.02)	0.00 (−0.04,0.03)	−0.07 (−0.11,−0.04)	−0.07 (−0.10,−0.04)
Humptulips f	−0.01 (−0.04,0.01)	−0.07 (−0.10,−0.05)	0.01 (−0.03,0.05)	0.05 (0.01,0.09)	−0.02 (−0.05,0.01)	−0.01 (−0.03,0.02)
Naselle–Bear f	n/a	n/a	n/a	0.02 (−0.02,0.05)	−0.02 (−0.05,0.01)	0.00 (−0.02,0.03)
Nemah–Palix f	n/a	n/a	n/a	0.23 (0.18,0.28)	0.09 (0.04,0.14)	0.08 (0.05,0.11)
North–Smith f	0.01 (−0.02,0.03)	−0.04 (−0.06,−0.02)	−0.02 (−0.05,0)	−0.05 (−0.08,−0.02)	−0.07 (−0.10,−0.05)	0.00 (−0.03,0.02)
Satsop f	0.04 (0.02,0.06)	−0.03 (−0.06,−0.01)	0.00 (−0.03,0.04)	0.02 (−0.01,0.06)	−0.03 (−0.06,0)	0.01 (−0.02,0.03)
Willapa f	n/a	n/a	n/a	0.04 (0.01,0.07)	−0.01 (−0.04,0.02)	0.03 (0.00,0.06)
Wishkah f	−0.03 (−0.05,−0.01)	−0.06 (−0.08,−0.04)	0.02 (−0.01,0.06)	0.00 (−0.03,0.04)	−0.09 (−0.12,−0.06)	−0.04 (−0.07,−0.01)
Wynoochee f	−0.04 (−0.07,−0.01)	−0.12 (−0.14,−0.09)	−0.01 (−0.05,0.04)	0.05 (0.01,0.09)	−0.03 (−0.07,0.00)	−0.03 (−0.06,0.00)
South–fall agg	**0.00** **(−0.03,0.02)**	**−0.06** **(−0.08,−0.03)**	**0.01** **(−0.02,0.05)**	**0.02** **(−0.01,0.06)**	**−0.03** **(−0.06,0.00)**	**0.00** **(−0.02,0.03)**
Calawah su	0.02 (−0.04,0.07)	−0.08 (−0.12,−0.04)	−0.03 (−0.06,−0.01)	−0.02 (−0.05,0.01)	0.03 (−0.01,0.06)	0.08 (0.07,0.1)
Chehalis sp	0.08 (0.05,0.11)	0.03 (0.00,0.06)	0.02 (−0.01,0.05)	−0.04 (−0.07,0.00)	−0.04 (−0.07,0.00)	0.00 (−0.03,0.03)
Clearwater sp/su	−0.14 (−0.18,−0.09)	−0.16 (−0.21,−0.12)	−0.08 (−0.10,−0.06)	n/a	0.13 (0.09,0.17)	0.03 (−0.01,0.08)
Hoh sp/su	0.00 (−0.04,0.04)	−0.06 (−0.10,−0.03)	−0.01 (−0.03,0.01)	−0.02 (−0.05,0)	−0.02 (−0.05,0.01)	0.02 (0,0.04)
Queets sp/su	−0.02 (−0.06,0.02)	−0.08 (−0.11,−0.04)	−0.04 (−0.06,−0.02)	−0.01 (−0.04,0.02)	0.02 (−0.01,0.05)	0.01 (−0.01,0.04)

Table 10 (continued). Fifteen-year trends (slope) in log Chinook salmon escapement computed from a linear regression applied to the predicted log escapement estimates.

Population	1982–96	1987–2001	1992–2006	1997–2011	2002–16	2007–21
Quillayute–Bogachiel su	−0.02 (−0.07,0.03)	−0.04 (−0.08,0.00)	0.03 (0.01,0.05)	0.00 (−0.03,0.02)	0.02 (−0.01,0.05)	0.02 (0,0.05)
Quinault sp/su	−0.04 (−0.08,0.00)	−0.09 (−0.13,−0.06)	−0.04 (−0.06,−0.02)	−0.02 (−0.04,0.00)	−0.03 (−0.06,−0.01)	n/a
Satsop su	−0.17 (−0.21,−0.14)	−0.15 (−0.19,−0.11)	−0.03 (−0.07,0.01)	−0.02 (−0.07,0.03)	−0.03 (−0.08,0.02)	0.02 (−0.01,0.06)
Sol Duc su	0.04 (0.01,0.07)	−0.01 (−0.04,0.02)	−0.02 (−0.04,0)	−0.04 (−0.06,−0.01)	−0.01 (−0.04,0.02)	0.05 (0.03,0.06)
Sp/su agg	**0.01 (−0.02,0.05)**	**−0.03 (−0.06,0.00)**	**0.00 (−0.03,0.02)**	**−0.03 (−0.06,0.00)**	**−0.02 (−0.05,0.01)**	**0.02 (0.00,0.04)**

Table 11. Five-year geometric means of smoothed escapement. If > 2 of the 5 yr were missing escapement values, the mean was assigned *n/a*. Key: *f* = fall, *sp* = spring, *su* = summer, *agg* = aggregate.

Population	1982–86	1987–91	1992–96	1997–2001	2002–06	2007–11	2012–16	2017–21
Calawah f	1,298	2,385	1,374	1,082	1,070	976	930	1,210
Clearwater f	1,320	2,170	1,015	877	1,184	1,082	1,037	n/a
Dickey f	563	618	193	180	267	251	261	354
Hoh f	2,793	4,112	2,659	2,340	2,741	2,053	1,830	1,956
Queets f	2,848	4,458	2,500	1,934	2,282	1,903	2,085	n/a
Quillayute–Bogachiel f	850	1,699	872	815	1,186	1,468	759	747
Quinault f	3,479	7,300	4,639	3,445	4,714	3,948	3,719	3,204
Sol Duc f	3,592	6,055	3,286	2,583	2,798	n/a	1,979	2,873
North–fall agg	**16,774**	**28,823**	**16,547**	**13,264**	**16,268**	**13,834**	**12,616**	**13,110**
Chehalis f	n/a	5,487	4,436	4,471	6,841	4,967	4,565	5,879
Hoquiam f	n/a	996	660	495	623	501	283	230
Humptulips f	3,593	4,472	3,224	2,170	3,580	3,584	2,706	3,142
Naselle–Bear f	n/a	n/a	n/a	n/a	1,036	1,028	794	n/a
Nemah–Palix f	n/a	n/a	n/a	n/a	59	121	126	n/a
North–Smith f	n/a	591	450	402	379	239	172	n/a
Satsop f	n/a	3,715	3,569	2,689	3,823	3,467	2,636	3,516
Willapa f	n/a	n/a	n/a	n/a	1,102	1,122	876	n/a
Wishkah f	n/a	986	659	567	906	609	336	382
Wynoochee f	n/a	3,607	1,944	1,114	1,929	1,927	1,252	1,327
South–fall agg	**n/a**	**n/a**	**n/a**	**13,504**	**20,332**	**17,587**	**13,778**	**n/a**
Calawah su	149	323	177	126	119	98	145	219
Chehalis sp	926	1,859	1,973	2,436	2,420	1,552	1,588	1,419
Clearwater sp/su	167	151	43	n/a	19	n/a	74	56
Hoh sp/su	1,449	2,332	1,428	1,159	1,291	886	1,009	1,038
Queets sp/su	724	1,012	600	422	391	n/a	465	405
Quillayute–Bogachiel su	152	216	120	136	155	n/a	183	156
Quinault sp/su	n/a	688	395	243	254	202	n/a	n/a
Satsop su	306	128	53	26	37	20	24	n/a
Sol Duc su	455	743	688	629	546	n/a	479	637
Sp/su agg	**n/a**	**7,480**	**5,508**	**5,209**	**5,236**	**3,703**	**4,149**	**n/a**

Table 12. Selected estimates of Chinook salmon productivity in recruits per spawner, from Dorner et al. (2017, their Table 1), for natural populations along the U.S. West Coast and southern British Columbia. The Washington Coast population is in bold.

River/population group	Region	Broodyears	Recruits/Spawner	
			Median	**Range**
Oregon Coast	Oregon Coast	1979–2008	2.31	0.27–7.76
Lewis R	Lower Columbia	1979–2008	2.26	0.37–15.14
Lyons Ferry	Snake R	1979–2008	5.01	2.37–13.53
Washington Coast	**Washington Coast**	**1979–2008**	**3.05**	**1.09–8.55**
Snohomish R	Puget Sound	1979–2008	2.68	0.73–4.21
Skagit R	Puget Sound	1979–2008	1.48	0.40–6.63
Harrison R	Fraser R	1984–2008	1.43	0.11–10.50
Fraser R early	Fraser R	1979–2008	1.71	0.77–5.85

Productivity

For Chinook salmon, productivity can be defined and measured in many ways (e.g., egg production, number of smolts, number of age-2 adults, etc.). Here we summarize published literature on measures that reflect the number of adult salmon that will be produced by a spawning population and on specific biological processes that may affect the future productivity of WC Chinook salmon.

We summarize estimates of recruits per spawner from the WC and adjacent regions in Table 12 (Dorner et al. 2017). Over approximately 30 years of data, WC populations appear to be as or more productive than other nearby populations (a median estimate of 3.05 recruits per spawner). For reference, a recruit/spawner ratio of 1.00 indicates that the population produced the same number of recruits as there were spawners. Note that there is substantial variability in this measure of productivity across the time series for all populations (see *Range* in Table 12). Additionally, recruits/spawner measures the observed productivity over a period of time and does not account for the potential effects of density dependence. Under density dependence, a population at low abundance will tend to have higher productivity than the same population at high abundance. However, productivity for Chinook salmon is determined by both density-dependent and density-independent processes (e.g., riverine or oceanic conditions; Sharma et al. 2013, Dorner et al. 2017, Adkison 2022) and, as a result, summaries of observed recruits/spawner are a reasonable, if imperfect, measure of productivity.

We know of only one recent stock–recruitment analysis on the WC, for the Grays Harbor fall-run Chinook stock (QDNR and WDFW 2014) that includes data through 2011. Other stock–recruitment analyses inform reference points within the PFMC process (see PFMC 2023, their Table 3-1), but those reference points are either derived from very old stock–recruitment analyses (e.g., Cooney 1984 uses data from 1982 and earlier for the Queets, Hoh, and Quillayute Rivers) or rely on a proxy derived from the average of stock–recruitment parameters along the WC (Willapa Bay fall, Hoko River fall, and all WC spring/summer stocks (Salmon Amendment Committee 2011, PFMC 2023).

Measures of productivity from Dorner et al. (2017) only include data for broodyears through 2008. To look broadly at productivity in recent years, we used available information on changes in age structure of escapement as a proxy for productivity. As older salmon and larger salmon are more fecund (Malick et al. 2023), changes to age structure should have a direct impact on the egg production in individual populations. We looked at data from five different populations: Chehalis River fall (broodyears 1972–2017, Figure 21), Chehalis River spring (1979–2017, Figure 22), Humptulips River fall (1968–2017, Figure 23), Hoh River summer (1997–2017, Figure 24), and Hoh River fall (1997–2017, Figure 25). Four of the Chinook salmon populations along the WC have seen a general decline in average spawning age over time. The Chehalis and Humptulips River populations declined from an average age of 4.75 in the early 1980s to 4.00 in 2017 (Figure 21, Figure 22, Figure 23). The Hoh River fall and spring runs declined from about 4.50 to 4.00 over a shorter time period (1997–2017; Figure 24, Figure 25). The Chehalis River spring run had a slightly smaller decline, from approximately 4.15 to 3.85 between the early 1980s and the late 2010s. The trends in age structure for WC populations reflect a broader trend toward smaller and younger-spawning Chinook salmon across the northeastern Pacific coast (Ohlberger et al. 2018; Oke et al. 2020). Changes in age and size have direct consequences for the egg production of female Chinook salmon (Malick et al. 2023), which in turn should directly affect Chinook salmon productivity.

To provide a rough estimate of the decline in productivity due to changes in egg production (Table 13), we used estimates of average size (fork length) for WC Chinook salmon from McHugh et al. (2015) and combined them with the linear relationship between fecundity and fork length developed by Malick et al. (2023). Using these data sources, a decline from age-5 to age-4 fish means a decline of approximately 7 cm on average (88.0–80.9 cm). Fecundity

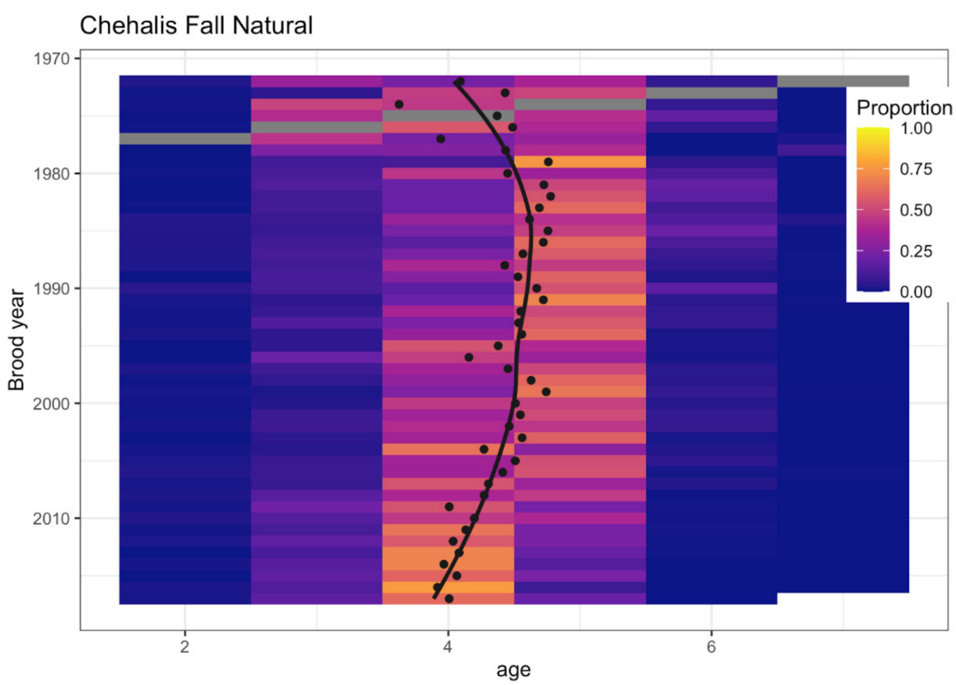

Figure 21. Proportional contribution of each age to spawning escapement for each broodyear (colors) for the Chehalis River fall run. Gray indicates missing data. Points are mean age for each broodyear; solid line is locally estimated scatterplot smoothing (LOESS), smoothed to illustrate changes in mean age.

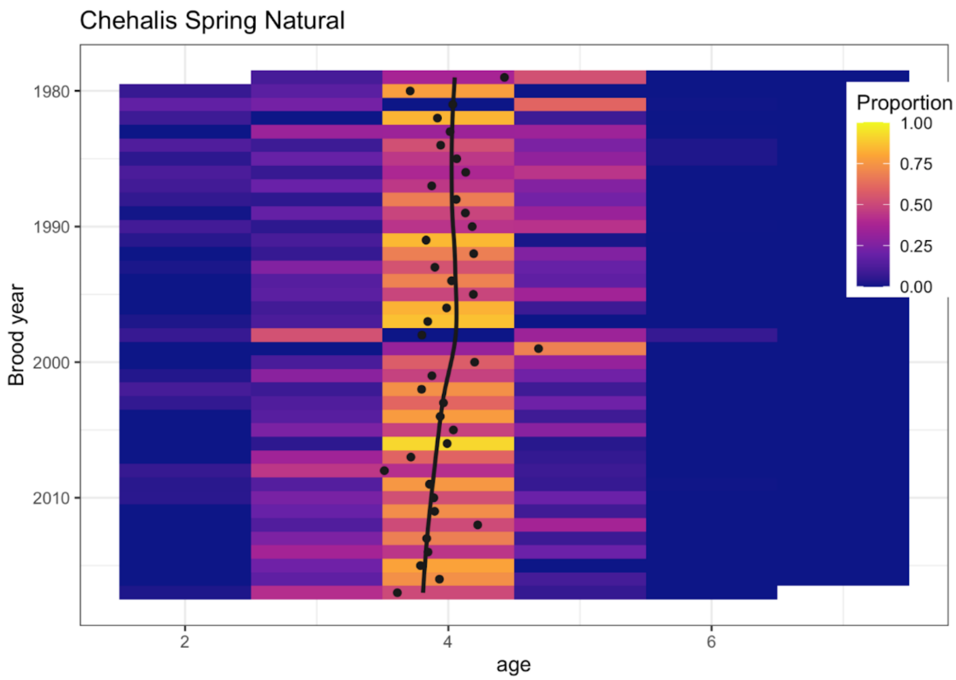

Figure 22. Proportional contribution of each age to spawning escapement for each broodyear (colors) for the Chehalis River spring run. Points are mean age for each broodyear; solid line is LOESS, smoothed to illustrate changes in mean age.

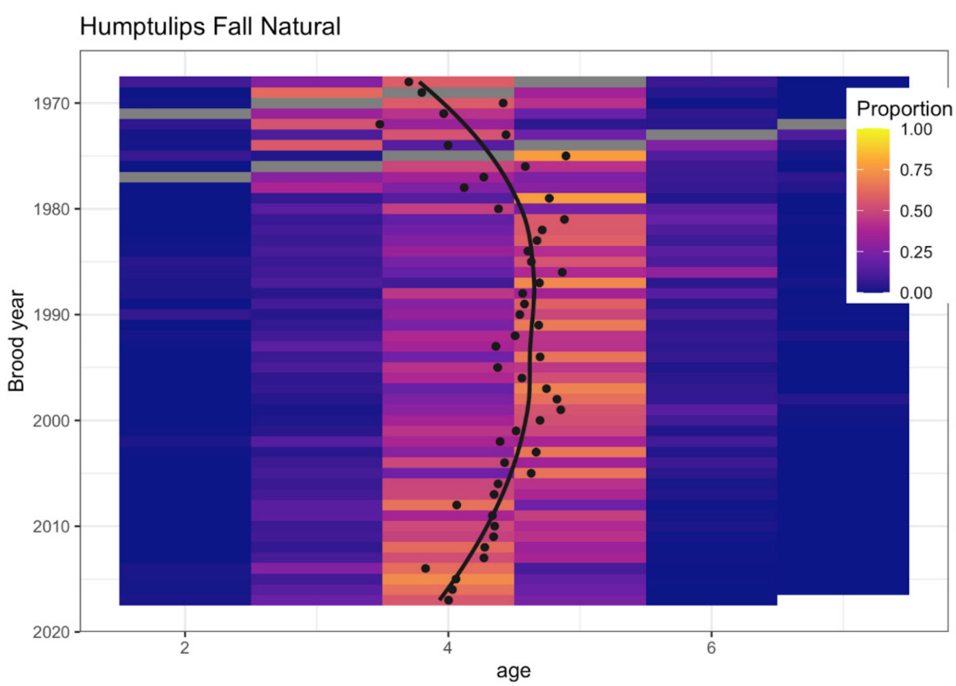

Figure 23. Proportional contribution of each age to spawning escapement for each broodyear (colors) for the Humptulips River fall run. Gray indicates missing data. Points are mean age for each broodyear; solid line is LOESS, smoothed to illustrate changes in mean age.

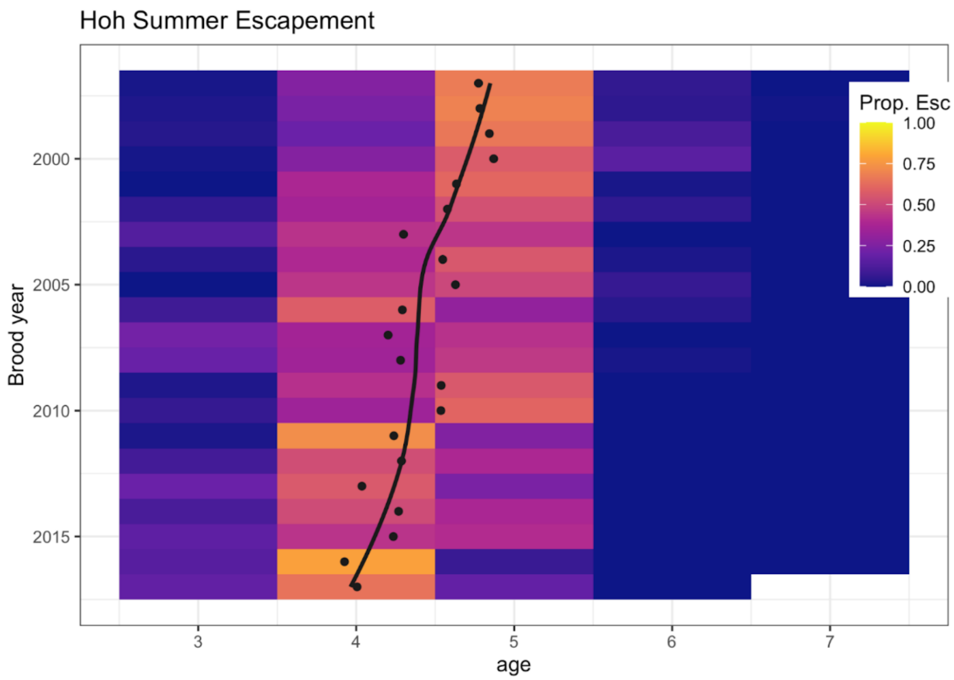

Figure 24. Proportional contribution of each age to spawning escapement for each broodyear (colors) for the Hoh River summer run. Points are mean age for each broodyear; solid line is LOESS, smoothed to illustrate changes in mean age.

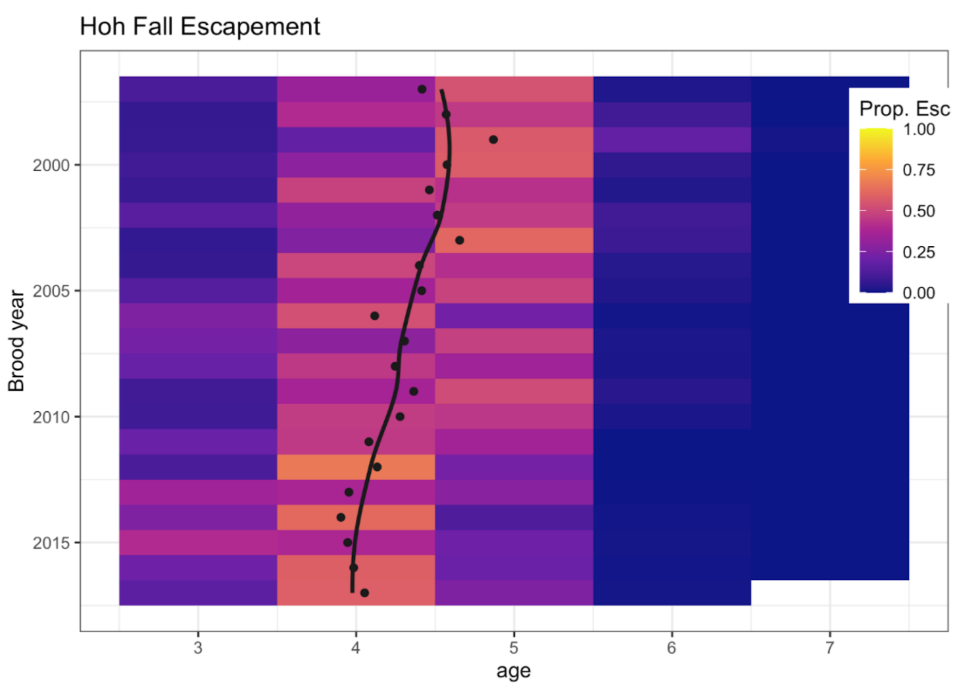

Figure 25. Proportional contribution of each age to spawning escapement for each broodyear (colors) for the Hoh River fall run. Points are mean age for each broodyear; solid line is LOESS, smoothed to illustrate changes in mean age.

Table 13. Approximate change in egg production based on estimates of length at age (fork length, FL) for fall-run hatchery Chinook salmon from WC rivers (McHugh et al. 2015, their Table A1), and fecundity–FL relation of Malick et al. (2023). Figures indicate expected percentage change in fecundity based on changes in age structure. We use the mean estimated change in fecundity (1.8% change in fecundity for a 1.0 cm change in length) and the mean FL.

Age after release (mo)	Mean predicted FL (cm)	Spawner age (yr)	Predicted % change in fecundity from age-5 (~58 mo) fish
34	65.7	3	–40%
35	69.8		
42	75.6		
44	78.6		
46	80.9	4	–13%
54	83.2		
56	85.2		
58	88.0	5	0%

declines by 1.8% per cm, meaning that a decline from age-5 to age-4 should result in ≈ 13% decline in fecundity for an individual fish (assuming constant length at age). Extending this to the population level, we can conclude that a change in average age of 4.75 to 4.00, as seen in the Chehalis River fall run, should result in roughly a 10% decline in fecundity. Changes in productivity due to age structure in other populations should be less than 10%. We emphasize that these estimates are approximate and additional work is warranted to generate more precise estimates. Coincident changes in length at age or length-specific fecundity during the time series could exaggerate or ameliorate predicted changes to productivity. There is some evidence for a decline in length at age over this period (Ohlberger et al. 2018, Malick et al. 2023), which would mean a larger decline in productivity. There is limited evidence for among-population variability in length-specific fecundity (Malick et al. 2023); we therefore expect this to be a less important factor in changing productivity.

Harvest

The Pacific Salmon Treaty (PST) defines management structures for Chinook salmon fisheries under the treaty purview (inclusive of Chinook salmon ocean fisheries from Cape Falcon, Oregon, north to Alaska). The Pacific Salmon Commission (PSC) implements the PST and the PSC's Chinook Technical Committee (CTC) uses several assessment methods to manage salmon fisheries and stocks harvested within the PST area (collectively referred to here as the "CTC process"). The CTC process covers Chinook salmon stocks from rivers in Oregon, Washington, Idaho, British Columbia, and Alaska. Broadly, the CTC process uses data on marine fishery catches, freshwater fishery catches, spawning escapements, and recoveries of CWT fish from the treaty area to provide both pre-season forecasts and post-season estimates of stock-specific abundance and fisheries exploitation rates (CTC 2023a,b). Briefly, pre-season forecasts are indices for abundance for upcoming seasons used to set annual catch limits for three major mixed-stock Chinook salmon fisheries, one in Alaska and two in British Columbia (termed Aggregate Abundance-Based Management, AABM). Other fisheries in British Columbia, Washington, and Oregon are managed to meet objectives for individual component stocks (termed Individual Stock-Based Management, ISBM). Fisheries off Washington and Oregon are planned via complicated processes involving NOAA, PFMC, the states of Washington and Oregon, tribal nations, and other interested parties. Ocean fisheries must be conducted in accordance with legal obligations under the PST, treaties, court decisions between Native American tribes and the United States, and conservation constraints of the ESA. Both AABM and ISBM fisheries are known to intercept WC Chinook salmon.

Fall-run Chinook salmon arising from the WC Chinook Salmon ESU are represented within the CTC process. Information is not available for individual river runs, but the WC group uses the Queets River exploitation indicator stock (releases from the Quinault Department of Natural Resources' Salmon River Hatchery). Indicator stocks are groups of CWT Chinook salmon recovered in marine and freshwater fisheries, at hatcheries and some spawning grounds, to determine the proportions harvested in different marine and freshwater fisheries. In addition to serving as indices of survival, exploitation, and total escapement, they are also used to determine spawning abundance and harvest rates in terminal areas (generally in-estuary or in-river harvest). For the WC, escapement indicators of fall-run stocks are the Grays Harbor, Hoh, Queets, and Quillayute River stocks. Spring-run Chinook salmon from the WC Chinook Salmon ESU do not have an indicator stock, so there are no estimates of exploitation rates.

In Figure 26, we present total mortality estimates from the CTC's Exploitation Rate Analysis. Cohort reconstruction procedures applied to tagged hatchery releases are used to estimate the exploitation rate associated with the reported catches in various fisheries, as well as the total mortality rate. Total mortality accounts for both the reported catch and incidental mortalities caused by fishing (losses due to release mortality). Below, we present only the estimates of total mortality. All stocks within the WC are assumed to have ocean mortality rates identical to the Queets River exploitation indicator, except for the two northern hatchery-dominated populations in the Hoko and Tsoo-Yess Rivers.

We can use output from the CTC process to examine fisheries mortality from 1982 to 2021 for four rivers (Figure 26). These plots show the proportion of adult-equivalent mortality caused by marine fisheries, terminal fisheries (estuary and freshwater), and total fisheries mortality (sum of ocean and terminal exploitation rates). The CTC model uses adult-equivalent mortality (AEQ) to make mortality rates that occur at different times of the life-cycle comparable.

Harvest rates for WC fall-run Chinook salmon have been consistent and relatively high for the past 40 years. Harvest reduces the total adult run size by approximately 50% for each return year. There is no clear temporal trend in total mortality in either ocean or terminal fisheries over that period.

While there are no equivalent estimates of harvest-related mortality for spring-run Chinook salmon, terminal harvest estimates are available from PFMC (2024). Figure 27 shows available estimates of terminal harvest from PFMC for fall- and spring-run populations from the Quillayute, Hoh, and Queets Rivers and the Grays Harbor complex of rivers, and from the fall-run populations from Willapa Bay. Terminal harvest rates are calculated as the proportion of the total spawner returns captured by fisheries within the estuary or river. Note that terminal harvest occurs on the proportion of the population that has survived ocean fisheries and returned to spawn. As a result, the harvest rate for terminal fisheries is not equivalent nor comparable to the units of harvest present in the CTC analyses (Figure 26).

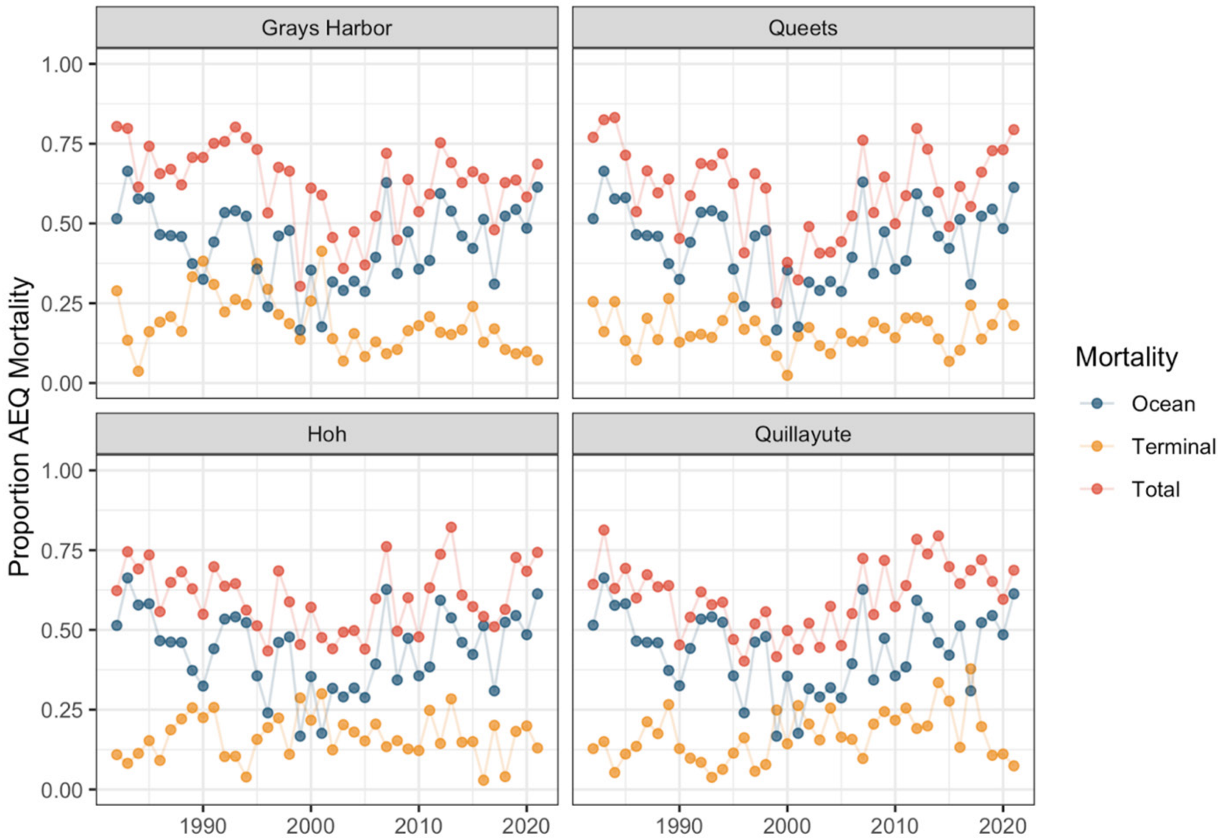

Figure 26. Estimated mortality associated with harvest expressed in terms of adult equivalencies for ocean, terminal, and total (ocean + terminal) harvest for four WC rivers from the CTC process. Ocean mortality of the Queets River fall stock applied here to all WC aggregate, but terminal harvest rates vary among populations.

Across all four rivers for which we have information, harvest rates for spring-run populations are generally lower than for fall-run populations (Figure 27). The differences are particularly notable since around 2000. Before then, terminal harvest rates in the Queets and Hoh Rivers were similar between spring and fall runs.

Information about the ocean exploitation of spring-run stocks is limited. However, based on the timing of Chinook salmon returning to freshwater to spawn, it is believed that spring-run salmon return before the majority of ocean fisheries are targeted in their spawning year. As a result, all else being equal (e.g., ocean distribution, vulnerability to fishing gears, etc.), it is hypothesized that ocean exploitation rates on spring-run stocks are lower than on fall-run stocks. Available information indicates that terminal harvest rates are also lower for spring-run than fall-run stocks (Figure 27). This is particularly true after 2010. The available information indicates the overall harvest mortality for spring runs is less than for fall runs, but the magnitude of the differences cannot be determined quantitatively with available data owing to the lack of data on marine fishery impacts on spring runs.

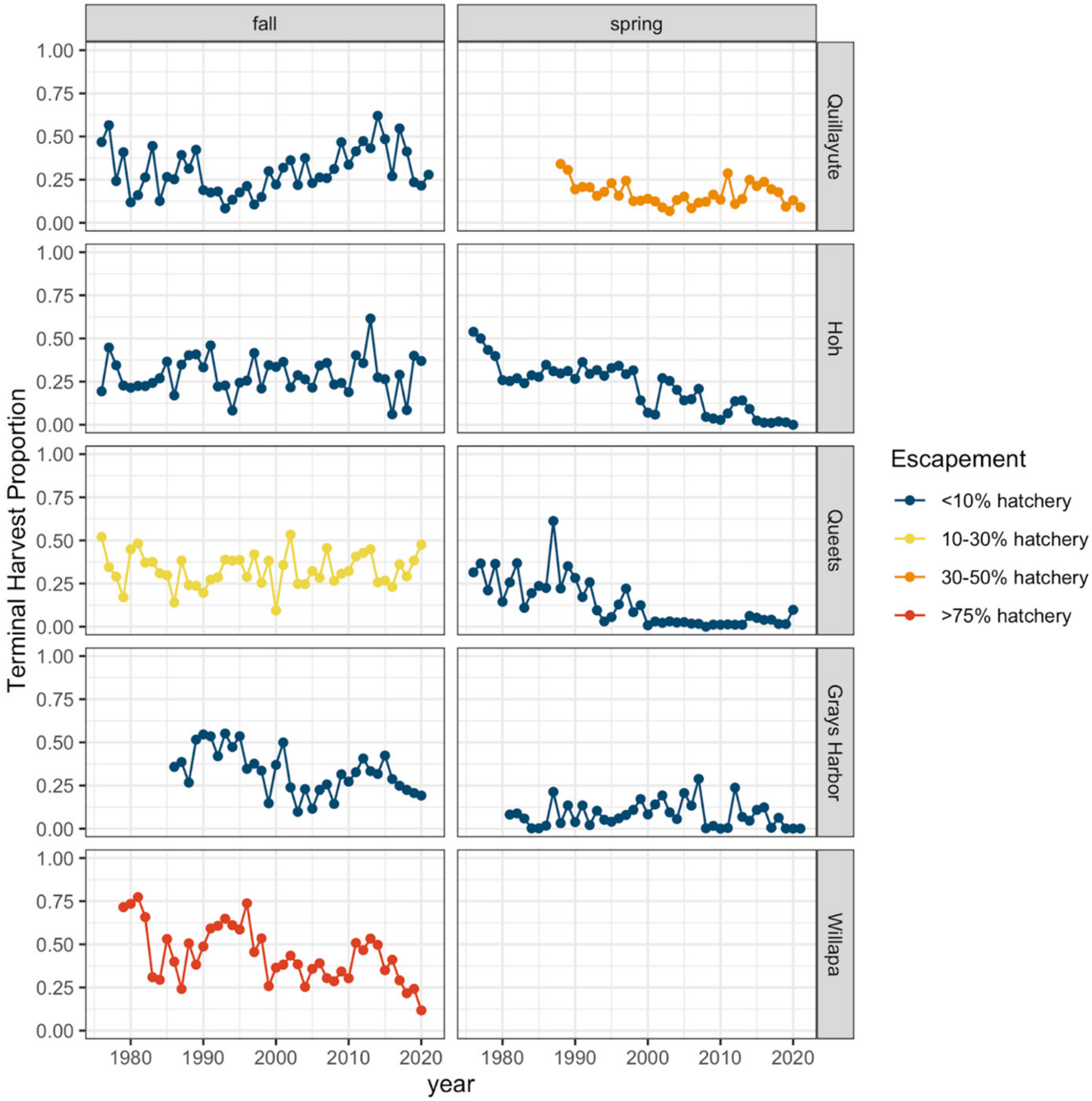

Figure 27. Terminal harvest rates collated from PFMC's blue book database. Harvest rates were calculated using estimates of natural-origin escapement and natural-origin total run size, where available (Quillayute, Hoh, Queets, and Grays Harbor spring runs), and using composite (combined natural- and hatchery-origin) escapement and total run size otherwise (Grays Harbor fall run, Willapa Bay). Colors indicate the time series' average contributions of hatchery-origin Chinook salmon to escapement in each population.

Summary of current abundance, productivity, and harvest trends

The WC Chinook Salmon ESU is composed predominantly of fall-run Chinook salmon. Spring runs contribute a smaller but potentially important number of individuals to a subset of WC rivers. Recent information on fall-run Chinook salmon abundance for the 15-year period (2007–21) showed that for 19 monitored populations, most had relatively stable abundances over the past 15 years (Figure 18, Figure 19, Figure 20, Table 10). Total returns for fall-run fish averaged 30,000–40,000 fish per year; early run timing fish returns were in the 5,000–7,000 range. WC Chinook salmon abundance trends have remained stable despite declining age at return and relatively high harvest rates.

Analysis of ESA Section 4(a)(1) Risk Factors

Section 4(a)(1) of the ESA directs NMFS to determine whether any species is threatened or endangered because of any of the following factors: 1) the present or threatened destruction, modification, or curtailment of its habitat or range; 2) overutilization for commercial, recreational, scientific, or educational purposes; 3) disease or predation; 4) the inadequacy of existing regulatory mechanisms to address identified threats; or 5) other natural or man-made factors affecting its continued existence. Section 4(b)(1)(A) requires us to make listing determinations after conducting a review of the status of the species and considering efforts to protect such species.

NMFS has previously reviewed the impacts of various factors contributing to the decline of Pacific salmon and steelhead in previous listing determinations (e.g., USOFR 1998, 2004) and supporting documentation (NMFS 1996, 1997, 1998). These Federal Register notices and technical reports concluded that all of the factors identified in ESA Section 4(a)(1) had played a role in the decline of U.S. West Coast Chinook salmon stocks.

Risk Factor 1: The Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Our previous Federal Register notices and reports (cited above), as well as numerous other reports and assessments, have reviewed in detail the effects of historical and ongoing land-management practices that altered WC Chinook Salmon ESU habitats. Chinook salmon depend on suitable freshwater, estuarine, and marine habitats, all of which influence population abundance, productivity, diversity, and spatial structure (McElhany et al. 2000). A broad range of historical and ongoing land and water management activities/practices have adversely impacted the freshwater and estuarine habitats.

Current habitat conditions along the WC are summarized in multiple documents and reports, including the State of Our Watersheds reports from Northwest Indian Fisheries Commission (NWIFC 2020) and reports on specific WRIAs. From 1998 through 2003, salmon habitat limiting factors analysis (LFA) reports were produced for each WRIA. Smith (2005) combined individual LFA ratings to form a WRIA-wide rating for each habitat parameter. Individual WRIA reports provide more information specific to the limiting factors in each WRIA: WRIA 19 (Smith 1999a), WRIA 20 (Smith 2000), WRIA 21 (Smith and Caldwell 2001), WRIAs 22 and 23 (Smith and Wenger 2001), and WRIA 24 (Smith 1999b).

Below, we start by summarizing the land-use practices (forestry, agriculture, and urbanization) that have altered, or in some cases eliminated, habitat(s) for WC Chinook salmon. We then discuss habitat-related factors that may be limiting the viability of the WC Chinook Salmon ESU.

Land-use practices: Forestry

Across the Washington coast, the most predominant land-use activities affecting WC Chinook salmon habitats are forestry practices. Figure 28 illustrates the WC watershed as percent forest and transitional forest. Historically, forestry practices allowed for harvest without stream buffers, the removal of instream wood, the use of splash dams to transport timber, high-density road construction, frequent road use, and harvesting large proportions of watersheds (Martens et al. 2019). Forestry land-use practices alter watershed processes, resulting in degradation of water quality and quantity, stream stability, and stream channel complexity (Cederholm et al. 1980, Smith 2005, NWIFC 2020). Timber harvest removes forest cover and affects watershed stability and the overall quality of habitat for salmon (NWIFC 2020). Forest cover reduces surface runoff and allows for the infiltration of precipitation into groundwater, both of which are important for stream processes, including reducing sedimentation, moderating flows, and extending the hydrologic flow duration which can increase groundwater input into lakes, streams, and wetlands. The root systems of forest vegetation help reduce mass wasting events, both in number and size, and reduce suspended sediment concentrations (Naiman et al. 1998). Forest vegetation provides shade and helps reduce water temperature increases (Naiman et al. 1998). Timber harvest in riparian areas reduces inputs of leaf litter, terrestrial insects, and large wood into streams (Thomas et al. 1993, Nakamoto 1998). Reduction of large wood from the harvest of streamside timber has resulted in the reduction of cover and shelter from turbulent high flows (Cederholm et al. 1997). Numerous studies have identified impacts, including reduced large woody debris, increased water temperature, and increased erosion and sedimentation. These impacts have been shown to impair the reproductive success of salmon due to increased turbidity, loss of interstitial spaces for use by juveniles, the smothering of eggs by fine sediments, loss of deep pools, and blockage of spawning habitat by landslides (Brown and Krygier 1971, Beschta 1978, Beschta and Taylor 1988). Large clear cuts, inadequate stream buffers, mass wasting, and poorly constructed and/or maintained forest roads have led to the degradation of salmon habitat (NWIFC 2020). Forest management activities result in high densities of forest roads for commercial timber harvest. Forest roads contribute to stream channel degradation because, if not properly constructed and maintained, roads can be a source of sediments to streams, which degrades fish habitat and water quality (Cederholm et al. 1980, Furniss et al. 1991).

Forestry by WRIA

In the Western Strait of Juan de Fuca (WRIA 19, Figure 4), most of the basin is forested and managed for timber harvest (Smith 1999a). Timber harvest began in the late 1880s, and nearly all of the basin has been harvested at least once down to the streambanks (Martin et al. 1995). Past timber harvest activities have dramatically changed the landscape, through clearcuts, road building, and log transport. Timber harvest has led to the replacement of large coniferous trees (western hemlock, Sitka spruce, western red cedar, and Douglas fir) by immature trees or deciduous trees, and changed the hydrology of most rivers, with a resulting in increased fine sediments in the streams. The harvest and salvage of large mature trees left little large woody debris in the streams, leading to the streambeds becoming progressively less stable. Gravel scouring and pool loss were observed in most of the

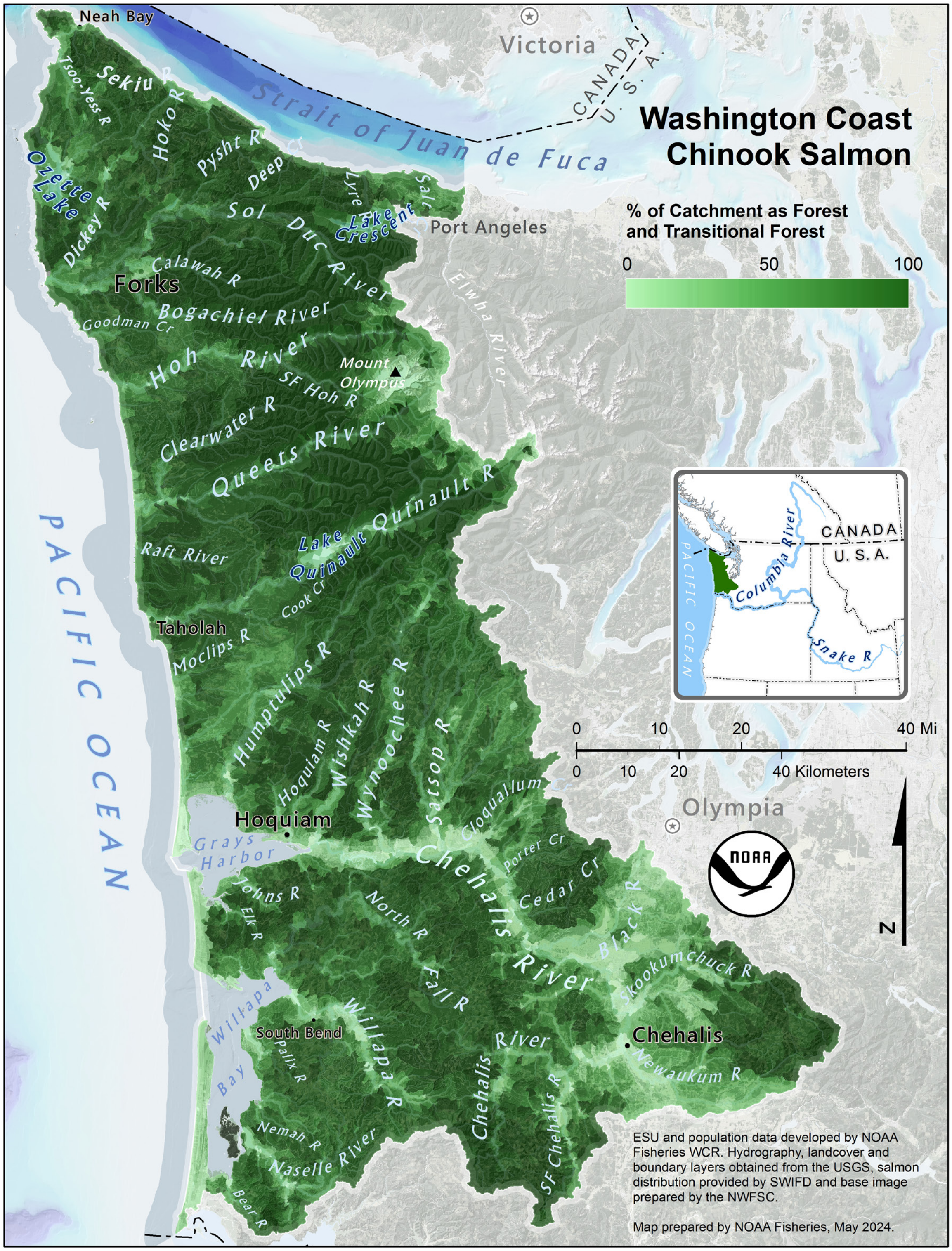

Figure 28. Percent of forest and transitional forest cover within WC Chinook Salmon ESU catchments (Yang et al. 2018).

streams in this area. Currently, private timber companies own a large percentage of the land within the basin, with 42% of the land owned by Crown Pacific and 28% by Rayonier (Martin et al. 1995). The Washington Department of Natural Resources (DNR) owns 25%, with other smaller parties owning 3%. The Washington State Parks Department and the Olympic National Forest each own about 1% of the basin land. About 95% of the old growth has been converted into commercially managed tree farms (McHenry and Kowalski-Hagaman 1996).

Unlike many other areas of Washington State, WRIA 20 has a significant portion of land located in Olympic National Park (ONP). The majority of ONP forests have never been logged and are characterized as temperate rainforest of coniferous old-growth forests dominated by Sitka spruce in the lowlands and western hemlock with silver fir in the higher elevations (Figure 29). Outside of ONP boundaries, timber harvest began in the 1920s–'30s, using rail to transport the logs (U.S. Forest Service 1995). Road construction for log trucks began in the 1940s, and early roads built on steep slopes still create sediment problems today. From the 1960s through the 1980s, extensive clearcutting and road construction occurred. However, legacy effects from past harvest practices continue to impact salmonid habitat; these include active and abandoned forest roads, undersized and failed culverts, scoured stream beds from splash dams, and removal of in-stream wood, among others (Smith 2000).

In the Quinault River basin (WRIA 21), ONP is the major landowner (> 50%). The Quinault Indian Nation owns 32% of the basin, comprising most of the area downstream of Lake Quinault (Quinault Indian Nation and USDOI 1999). The U.S. Forest Service manages 13% of the watershed, including the eastern part of the Cook Creek watershed and the southwestern half of the Lake Quinault watershed. Private landholdings comprise about 4% of the lands in the basin, and lands managed by the Washington DNR encompass about 0.1% of the drainage area. Lands within ONP are managed as wilderness, which includes the maintenance of access roads for visitors.

In the Queets River basin (WRIA 21), land ownership includes ONP, the Olympic National Forest, the Washington DNR, the Quinault Indian Nation, and privately owned lands. ONP owns 33.5 km (20.8 mi) of the Queets River and the lower reaches of many tributary streams (Phinney and Bucknell 1975). The Olympic National Forest, managed by USFWS, owns 84% of the Matheny Creek watershed, 73% of the Sams River watershed, and 30% of the Salmon River watershed, as well as some acreage north of the Queets River. All of these watersheds have established riparian reserves, while lands outside of the riparian reserves are managed as late successional reserves or adaptive management areas (Smith and Caldwell 2001, Lasorsa 2002). Washington DNR lands comprise 79% of the Clearwater River sub-basin (Smith and Caldwell 2001) and are managed as part of the Olympic Experimental Forest, which has a management objective of habitat conservation and timber production across the landscape. The riparian conservation strategy calls for interior riparian buffer zones and exterior riparian wind buffer zones (WDNR 1997). Lands in the Quinault Indian Reservation include the lower eight miles of the Queets River and estuary and 54% of the Salmon River drainage (Smith and Caldwell 2001). These lands are managed under the Quinault Forest Plan for sustainable timber harvest, maintenance or enhancement of fish and wildlife habitat, consolidation of tribal lands, and enhancement of traditional and cultural values.

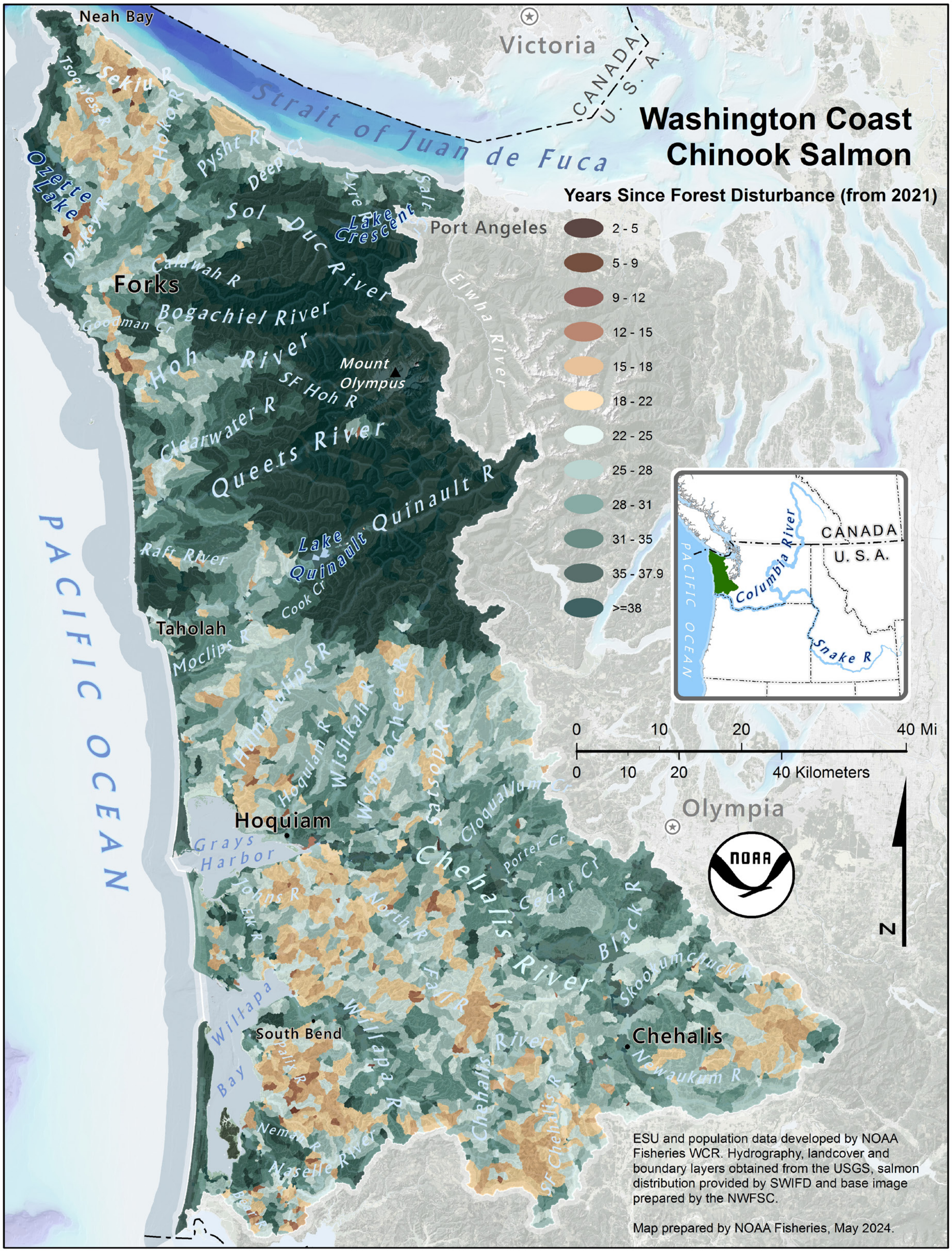

Figure 29. The average number of years since forest disturbance within WC Chinook Salmon ESU catchments (Jin et al. 2023).

WRIAs 22 and 23 include the Chehalis River basin and its tributaries and the independent watersheds that drain into Grays Harbor, including the Humptulips, Hoquiam, Johns, and Elk Rivers and several smaller streams. Forests cover 85% of the Chehalis River basin (Pickett 1992), 66% of which are privately owned (Smith and Wenger 2001). The lower Chehalis River basin (downstream of Porter) consists of 91% forest, while the upper basin is 77% forest. Agricultural activities dominate the valleys, while timber management occurs throughout most of the upland areas. Agricultural activities account for 10% of the basin's land use; the remainder is urban and industrial lands, mostly concentrated in Aberdeen, Hoquiam, Centralia, and Chehalis (Pickett 1992).

Lands within the Humptulips River basin have been highly managed for timber and are exclusively commercial timberland or National Forest, and while there is still some old-growth forest, private forest and the small state parcels in the remainder of the drainage have been completely converted to second-growth forests (Smith and Wenger 2001). In the lower part of the basin, the uplands are almost exclusively private timberlands, while the floodplain of the mainstem Humptulips River is mostly rural residential or agricultural land (Smith and Wenger 2001).

The Hoquiam River drains a 254 km^2 (98 mi^2) area, with the City of Hoquiam maintaining 7,500 acres of forested land within the West Fork Hoquiam River as a protected municipal watershed. The Middle Fork has been exclusively managed for commercial timber, while the East Fork watershed has dense residential development along the lower 1.2 km (0.75 mi), with sparse development further upstream. Most of the Little Hoquiam drainage consists of second-growth commercial timber and residential development. Outside of the developed floodplain areas and the West Fork municipal watershed, the remainder of the Hoquiam River drainage is managed for commercial timberlands of second-growth forest (Smith and Wenger 2001).

The Wishkah River drains 265 km^2 (102 mi^2) and has two forks (East and West) originating in the foothills of the southern Olympic Mountains. The uplands (above RKm 37/RM 23) have been intensely managed for commercial timber and are a patchwork of clearcuts in various successional stages of reforestation (Smith and Wenger 2001).

The South Grays Harbor region includes the two larger drainages of the Elk and Johns Rivers that have extensive estuaries which support oyster farms. The remainder of the Elk River drainage is managed as commercial timberlands.

In the Willapa Bay basin (WRIA 24), about 78% of the land is in timber production, 12% in estuary lands, 6% in agriculture, and 4% in residential use (Smith 1999b). Logging began in the mid-1800s, starting near the tidewater areas and floating the logs to sawmills. Logging, log transport, and splash damming were identified as major factors degrading salmon habitat. In the 1920s, splash dams blocked 60% of the migratory fish habitat in Willapa Bay. Smith (1999b) reported gravel mining (removal of gravel from stream beds) directly affected salmon spawning areas and produced considerable sediment, impacting salmon and aquatic species downstream.

The floodplains in the lower reaches were cleared for agriculture, and some of these areas later became urbanized. Often, these areas have little-to-none of the riparian vegetation that used to exist along these streams. Stream clearing removed much of the existing large woody debris. Less than 3% of the timberland is in permanent conservation, and only a fraction of that is old-growth timber (The Willapa Alliance 1998).

Although efforts are underway to address the legacy effects from historical logging practices, it may take decades for habitat to recover (Martens et al. 2019). Even with ~25 years of more protective timber harvest regulations related to riparian zones, important salmonid habitat components such as in-stream wood and pools have not recovered through natural recruitment of wood (Martens and Devine 2022). The estimated timeline for recovery of these remaining degradations could range from 100 to 225 years (Devine et al. 2022). The threat from current and future timber harvest will depend partly on federal, state, and tribal forest practices. This topic is explored in Risk Factor 4.

Land-use practices: Agriculture

Logging and agriculture activities result in similar impacts to salmonid habitat, though the magnitude of impact will vary because of the land conversion that typically occurs with agriculture. Agricultural lands reflect the practices that began in the late 1800s with the removal of trees and clearing of lowland forests (NWIFC 2020). Diking soon followed, with lower estuaries being diked to protect the new farmland and to increase agriculture productivity. Agricultural impacts include: loss or modification of wetland, estuaries, and floodplain habitats; channelization and loss of stream complexity; riparian removal; and reduced streamflow (associated with irrigation withdrawals). Smith (2005) noted that the major cause of unnatural low flows is water withdrawals for irrigation, industrial, and domestic use, and water transfers between basins. Removal of water, either directly from the stream channel or from wells in hydraulic continuity with stream flows, reduces the amount of in-stream flow and useable wetted area remaining for support of adult salmonid spawning and juvenile rearing (Smith 2005). Across the WC, Smith (2005) rated low-flow conditions as "poor" under several circumstances, including Clean Water Act Section 303(d) listing for low flows, known salmon mortality due to flows, stream closures due to over-appropriations when the stream produces species known to use the area during the low-flow period, and when other studies documented low-flow constraints for salmon in certain streams that do not conform to natural conditions. In the low-flow summary map by WRIA, Smith (2005) reported low-flow conditions as "unknown" for WRIAs 19, 20, and 22, "good" for WRIAs 21 and 24, and "poor" for WRIA 23. Smith (2005) also reported that data gaps for low-flow conditions are extensive. While known low-flow issues have been documented in some streams, there is a lack of linkage to salmon production, an absence of standardization regarding low-flow thresholds, and a shortage of flow monitoring.

Agriculture can also affect streams through reduced stream bank stability and sedimentation, reduction of large woody debris recruitment, elevated water temperature, and water quality problems stemming from agricultural runoff (e.g., nutrients and pesticides). The most intensive agricultural land use coincides with broad alluvial valleys and the low-lying areas (often former floodplains) of most watersheds. Because of the land clearings, agricultural practices are partially responsible for the significant decrease in large woody debris recruitment in the lower basin. Significant grazing occurs routinely on private lands and by permit on federally administered lands. Grazing may change soil infiltration rates, increase sedimentation, and can cause deleterious channel changes such as widening and shallowing of streams (Spence 1996). Riparian vegetation alteration occurs with grazing as well, affecting wood recruitment, bank stability, and stream temperatures. The salmon life stages most affected by agricultural practices are juveniles and smolts because they spend weeks to months rearing in the affected floodplain and estuarine areas, and are particularly susceptible to water contaminants and poor habitat quality.

Land-use practices: Urbanization

Urbanization refers to the concentration of human populations into discreet areas, and leads to land conversion for residential, industrial, commercial, and transportation uses. As the human population increases, additional urbanization and habitat modification will continue. The U.S. Census Bureau reports population estimates by county at the end of each decade. County boundaries do not line up with WRIA delineations, but provide an overview of population growth between 2010 and 2022 on the Washington Coast. Per USA Facts, county populations on the WC have increased from 5.8–15.5%.[8] In 2022, the population of Clallam County (WRIAs 19–21) was 77,805 (an 8.8% increase), Jefferson County (WRIAs 20 and 21) was 33,589 (a 12.3% increase), Grays Harbor County (WRIAs 21 and 22) was 77,038 (a 5.8% increase), Lewis County (WRIA 23) was 85,370 (a 13.1% increase), and Pacific County (WRIA 24) was 24,113 (a 15.5% increase).

Urbanization has led to degraded habitat through stream channelization, floodplain drainage, and riparian removal (Botkin et al. 1995). As rural populations grow, so does the demand for water, the risks of increases in peak flow, increases in sediment inputs, riparian vegetation removal, increased bank protection, and water contamination. High population densities lead to large amounts of impervious surfaces (i.e., roads, parking lots, infrastructure such as houses and buildings) that negatively impact the local watersheds and result in loss of salmon habitat. Paved roads, parking lots, rooftops, and other surfaces that do not absorb rainfall tend to send much more water to streams, elevating peak flows and contributing pollution to streams (Booth and Jackson 1997). One of the greatest concerns stemming from impervious surfaces is the stormwater runoff that occurs during rain events. When total impervious area is > 10% (a level which studies in other river systems found caused increased peak flows) it can lead to decreased base flows, simplified channel conditions, and increased non-point source stormwater pollution, resulting in loss of aquatic system function (Booth and Jackson 1997). An acute regional example of this phenomenon is toxic storm water runoff leading to high pre-spawn mortality of adult coho salmon in tributaries to Puget Sound (Booth et al. 2006, Peter et al. 2022). As the human population increases, additional urbanization and habitat modification are likely to occur.

Forestry, agriculture, and urbanization land-use practices alter the landscape through changes to riparian forests, estuary and nearshore habitat, and floodplain habitat, as described below.

[8] https://usafacts.org/data/topics/people-society/population-and-demographics/our-changing-population/

Riparian forests

Riparian areas are the connection between aquatic and terrestrial ecosystems and include the land adjacent to rivers and streams. Riparian areas play several important functions in maintaining stream processes. Tree roots and vegetation provide streambank structure (Montgomery and Buffington 2001), stabilize stream channels, decrease streambank erosion, and create undercut banks that serve as fish habitat (Bjornn and Reiser 1991). Trees provide shade (Naiman et al. 2001) and maintain the cool water temperatures required by salmon (Bjornn and Reiser 1991). Riparian vegetation also filters pollutants from soil (Knutson and Naef 1997, Welch et al. 2021) and reduces flood damage by slowing down floodwaters and dissipating energy (Naiman et al. 2001). Trees also contribute leaf litter, an important component of primary production within the stream, and provide food for macroinvertebrates which then serve as food for fish (Bisson and Bilby 2001). Large woody debris (LWD) from riparian forests stabilizes streambeds and banks, holds spawning gravels, promotes pool formation, provides resting and hiding cover for salmonids, and creates habitat for insects and other food items important to salmonids (Bisson and Bilby 2001). Loss of LWD results in a significant reduction in the complexity of stream channels, including a decline of pool habitat and low-flow rearing areas (Smith 2005).

Riparian forests are impacted by all types of land-use practices. In general, riparian forests can be completely removed, broken longitudinally by roads, and their widths can be reduced by land-use practices (Smith 2005). Additionally, riparian species composition can be altered when native, coniferous trees are replaced by exotic species, shrubs, and deciduous species (Smith 2005, NWIFC 2020). Deciduous trees are typically of smaller diameter and decompose faster than conifers, so deciduous trees do not persist as long in streams and are vulnerable to washing out from lower magnitude floods (Smith 2005). Once impacted, the recovery of a riparian zone can take many decades as the forest cover grows back and coniferous species colonize (Smith 2005). For example, in the Hoko River (WRIA 19), the riparian zone is currently 91% red alder, as compared to the historic dominance of conifer trees (western hemlock, Sitka spruce, western red cedar, and Douglas fir) with only a few patches of red alder (Martin et al. 1995). Smith (2005) rated the riparian conditions in WRIA 19 as “poor” based on the riparian forest being dominated by hardwood trees.

Similarly, the riparian vegetation in the Quinault River basin (WRIA 21) was historically dominated by coniferous trees, but logging has reduced the size and abundance of conifers as well as decreased the number of tree species (Quinault Indian Nation and USDOI 1999). Smith and Caldwell (2001) reported riparian conditions for the lower mainstem Quinault River as a mix of “good” and “fair,” and as “poor” in the upper basin. The riparian conditions in the lower reaches of the North Fork Quinault River and lower Big Creek are “fair.” Most other tributaries have “good” riparian conditions, as does the North Fork Quinault River and the upper mainstem Quinault River, which are in ONP (Smith and Caldwell 2001).

In the Queets River basin, riparian conditions differ between National Park lands and other landowners. Riparian conditions on ONP lands are described as a complex assemblage of riparian forest patches (Abbe and Montgomery 1996), considered an approximation of undisturbed conditions due to the relatively small amount of timber harvest associated with homesteading along the Queets River corridor prior to 1940, and National Park status since that time. Outside of the Queets River corridor and ONP lands, riparian areas in Queets River mainstem tributaries and in the lower Queets River mainstem show effects of past timber harvest practices (Smith and Caldwell 2001). The authors summarized the riparian conditions throughout the Queets River basin for the Clearwater River as “fair” in the lower reaches, “good” in the upper reaches, and as “fair” to “good” for the Matheny and Salmon Rivers sub-basin.

In WRIAs 22 and 23, riparian degradation is extensive throughout the sub-basins, particularly the Wynoochee, Satsop, Cloquallum, Garrard, Lincoln, Skookumchuck, Newaukum, Salzer, Bunker, and the South Fork Chehalis River sub-basins (Smith and Wenger 2001). The lower reaches of most of the other sub-basins were rated as having “poor” riparian conditions (Smith and Wenger 2001).

In the Willapa Bay basin (WRIA 24), the riparian area was described by Smith (1999b) as “impaired” to “severely impacted.” Smith (1999b) summarizes data from The Willapa Alliance (1998) on riparian forest structure (percent old growth, mid-seral stage forest, young conifer forests, hardwood dominated forests) and describes the conversion of old-growth forest to hardwood-dominated forests, resulting in low near-term LWD recruitment.

Estuarine and nearshore habitats

Estuarine habitats include areas in and around the mouths of streams extending throughout the area of tidal influence into freshwater. Nearshore habitat includes intertidal and shallow subtidal saltwater areas adjacent to land, and provides transportation and rearing habitat for adult and juvenile fish. Estuaries and nearshore areas provide many important functions, such as habitat for rearing, foraging, smoltification, migration, and refuge (Smith 2005). Chinook salmon from the WC emigrate to saltwater primarily as subyearlings and utilize the productive estuary and nearshore areas as rearing habitat. Estuaries provide a mixing zone of fresh and saltwater where juvenile and adult life stages can transition between freshwater and saltwater habitats (Smith 2005). If the habitats necessary for successful rearing and predator refuge are not available within this mixing zone, the survival of these fish can be compromised (Smith 2005). Important habitat features of nearshore areas include eelgrass, kelp beds, LWD, and the availability of prey species. Juvenile salmon use eelgrass to hide from predators as well as to feed on epibenthic and epiphytic zooplankton, including copepods and amphipods, which in turn feed on the bacteria from decaying eelgrass (Levings 1985, Webb 1991). Eelgrass also supports Pacific sand lance, surf smelt (*Hypomesus pretiosus*), and Pacific herring, all of which are important food items for salmon. Estuarine LWD serves as vital cover for juvenile salmonids (Martin and Dieu 1997) and creates firm substrates in a fine-sediment environment.

During the past century, estuary and nearshore areas have been directly impacted by human development (Smith 2005, NWIFC 2020). Shoreline modifications including armoring, dikes, dredging, and fills result in loss of habitat and habitat complexity. Shoreline modifications can also result in a loss of eelgrass and macroalgae habitats and the loss of associated prey and detritus production (Nightingale and Simenstad 2001). Development continues to occur adjacent to the estuary and fill material has reduced the size and function of estuaries. Marina development and other commercial activities in and near the estuary combine with urbanization to create a high amount of impervious area that can contribute to non-point source pollution. The habitat provided by the estuaries in western Washington continues to be degraded by the region's population growth (NWIFC 2020).

Estuary habitat is naturally limiting in WRIA 19 (Dethier 1990). Saltmarsh habitat is located in lower, tidally influenced stream reaches and around stream mouths and serves as important salmon rearing habitat (Smith 2005). Within WRIA 19, eelgrass is found in sandy, protected areas and provides important rearing habitat for juvenile salmonids (Smith 1999a). Roughly 20% of the Strait of Juan de Fuca's coastline consists of eelgrass habitat (WDF estimated 19.8% from studies conducted from 1975–89, while WDFW estimated 23.3% in 1977; Thom and Hallum 1990). Along the WRIA 19 shoreline, landslides contribute sediment to the nearshore environment (Smith 1999a). Landslides have short-term impacts on the species composition of kelp beds (Shaffer and Parks 1994) and, given the importance of kelp habitat for salmon rearing in this area, large nearshore sediment loads are considered a habitat problem (Smith 1999a).

The estuaries in WRIA 20 are small and isolated (Smith and Caldwell 2001). The largest estuary in WRIA 20 is the Quillayute River estuary (Smith and Caldwell 2001). The U.S. Army Corps of Engineers maintains a navigational channel, a protective jetty, and a boat basin in the lower Quillayute River. The jetty and dike were constructed in 1931; dredging began in 1949, and the basin requires dredging every 2–3 years (Chitwood 1981). This navigation-deep channel has increased the flow of water and reduced spawning and winter refuge habitat by disconnecting the river from its historic floodplain and increasing substrate instability (Smith and Caldwell 2001). The vast majority of land use is forestry (94%), and sedimentation has been identified as a major habitat problem throughout WRIA 20 (Smith and Caldwell 2001). Sediment loads are transported downstream, increasing sedimentation to the estuary and nearshore environment, which can reduce eelgrass and kelp habitat (Smith and Caldwell 2001).

Most estuarine areas within the Quinault and Queets River basins (WRIA 21) have no bank hardening, although there has likely been a loss of LWD compared to historic levels (Smith and Caldwell 2001). Generally, the estuarine habitat for these basins was rated as "good." Much of the nearshore habitat is part of the Copalis Rock National Wildlife Sanctuary and is rated "good." One concern is the recent decline in giant kelp, but the cause (whether natural or anthropogenic) of the decline is unknown (Smith and Caldwell 2001).

Grays Harbor (WRIA 22) is about 19 km (12 mi) wide at the widest point, and at high tide covers about 251 km^2 (97 mi^2; Smith and Wenger 2001). A two-mile-wide channel connects Grays Harbor to the Pacific Ocean. In 1996, NRC estimated that about 30% of historic estuary, amounting to 59 km^2 (23 mi^2), had been lost; by 2019, this had expanded to 45% (Brophy et al. 2019). Grays inner harbor is regularly dredged and heavily industrialized. Pulp mills, landfills, sewage treatment plants, and log storage facilities are all located within the inner harbor (Smith and Wenger 2001). Smith and Wenger (2001) summarized Grays Harbor estuarine conditions as "fair" for habitat loss but "poor" for water quality (chemical inputs and fecal coliform) and lack of large wood.

The Willapa River basin estuary (WRIA 24) consists of about 354 km^2 (137 mi^2) at mean high tide (The Willapa Alliance 1998). Smith (1999b) compared estuarine habitat to historical acres and reported ~30% loss. Smith (1999b) noted that invasive *Spartina,* a species of intertidal grass, is of concern in Willapa Bay. *Spartina* was introduced from the U.S. East Coast about a century ago, and the invasion has increased dramatically. *Spartina* grows into a "meadow," covering the mudflats, displacing native eelgrass, and raising the elevation of the flats. *Spartina's* impact on juvenile salmon rearing habitat is unknown, but the displacement of native eelgrass is a concern (Smith 1999b).

Floodplain habitat

Floodplains are found in the lowland areas of watersheds and are relatively flat areas adjacent to streams and rivers that are periodically inundated during high flows (Smith 2005). Floodplains allow for lateral movement of the main channel, thus creating braided side channels, off-channel habitat, and sloughs that provide refugia for juvenile and adult salmon during high-flow events (Benda et al. 1998). Floodplains provide storage for floodwaters, sediment, macroinvertebrate production (food), and LWD. Off-channel habitats provide an abundance of food with fewer predators, and provide habitat for juvenile salmonids (Sandercock 1991). Water seeps into the groundwater table during floods, recharging wetlands, off-channel areas, and shallow aquifers (Smith 2005). Wetlands and aquifers then release water to the stream during the summer months through a process called hydraulic continuity (Smith 2005). This process provides increased flows for salmonids during the summer months. Floods are a natural process and maintain stream function by flushing fine sediment from spawning gravel, creating pools and riffles, reshaping the streambed, depositing fine sediment on the floodplain, and recruiting large woody debris from the floodplain to the stream channel (Benda et al. 1998).

Despite the importance of floodplains for salmon habitat, floodplains continually face development pressures (NWIFC 2020). Because floodplains are dry most of the time, they are often developed, often for uses (agriculture, urbanization) that are not compatible with flooding. Floodplain habitat impacts include the direct loss of aquatic habitat from human activities (filling) and disconnection, both laterally by the construction of dikes and levees and longitudinally as a result of the construction of road crossings (Smith 2005). Riparian forests are typically reduced or eliminated as levees and dikes are constructed. The loss of large wood has contributed to the disruption of natural processes that create and sustain floodplain habitat. Floodplain disconnection also results from channel incision caused by changes in hydrology and sediment inputs (Smith 2005).

Smith (2005) provided a summary map of floodplain condition by WRIA (their p. 121), concluding that WRIAs 19–23 had "poor" floodplain habitat and rating WRIA 24 as having "fair/poor" floodplain habitat. A "poor" rating was defined as > 50% loss of floodplain connectivity of the stream and off-channel habitat due to incision, roads, dikes, or flood protection. A "fair" rating was a 10–50% connectivity loss.

In the western Strait of Juan de Fuca (WRIA 19), the majority of the floodplain problems are due to roads that border streams (Smith 1999a). Similarly, in WRIA 20, Smith (2000) summarized floodplain conditions in the Quillayute and Hoh River basins and described floodplain impacts due to the presence of riparian roads. Some of these roads closely parallel the streams and act as dikes, disconnecting off-channel habitat and increasing sediment inputs into the stream.

In the Quinault River basin (WRIA 21), Smith and Caldwell (2001) reported a lack of information regarding floodplain condition in the lower Quinault River, though noted concerns regarding bank hardening and floodplain road impacts along the Quinault River upstream of Lake Quinault. In addition, sediment problems associated with roads and undersized culverts are a problem in the timber-managed areas of the Quinault River basin. In the Queets River basin, Smith and Caldwell (2001) note that floodplain impacts such as bank hardening and roads are minimal, but that loss of off-channel habitat is a concern, especially in the Clearwater, Salmon, and Sams Rivers. Smith and Caldwell (2001) tentatively rated the Quinault River floodplain as "fair" due to the reduction of in-stream LWD from historic levels. In the Queets River basin, Smith and Caldwell (2001) note that floodplain impacts such as bank hardening and roads are minimal, but that loss of off-channel habitat is a concern, especially in the Clearwater, Salmon, and Sams Rivers.

In Grays Harbor (WRIA 22), the lower Chehalis River (between RKm 1.6–17.7/RM 1–11)—a large, fairly undeveloped floodplain complex—exists with numerous sloughs and side channels (Ralph et al. 1994). This habitat complex is under strong tidal influence, with nearby vegetation that is dominated by older conifers and hardwoods (Ralph et al. 1994). Between RKm 21.9–32.2 (RM 13–20) of the lower Chehalis River, 23 freshwater off-channel sites have been identified as potential juvenile salmon habitat (Ralph et al. 1994), though these areas are wetlands that are no longer connected to the main channel.

In the Chehalis River basin (WRIA 23), approximately 90% of floodplain habitats have been lost or degraded since the time of the historical surveys (Beechie et al. 2021, Beechie 2023). Beechie et al. (2021) reported that all of the slow-water and off-channel habitats (beaver ponds, marshes, side-channels) have declined by roughly 90%, and that only 8% of estimated historical floodplain marshes and ponds remain in relatively natural condition today. Floodplain disconnection has eliminated an estimated 241 km (150 mi) of the natural potential 266 km (165 mi) of side-channel length (–91%; Beechie et al. 2021).

The floodplains in the lower reaches of Willapa Bay (WRIA 24) were cleared for agriculture, and some of these areas later became urbanized (Smith 1999b). Often, these areas have little to none of the type of riparian vegetation that used to exist along these streams (Smith 1999b).

Land-use practices alter the landscape through changes to riparian forests, estuary, nearshore, and floodplain habitats. Below, we discuss other habitat-related factors that may be limiting the viability of the WC Chinook Salmon ESU.

Stream complexity

The loss of stream complexity has been identified as one of the key factors limiting the distribution and abundance of Chinook salmon (NWFSC and WCCSBT 1997, Myers et al. 1998, NMFS 1998, Smith 2005, Stout et al. 2012, NWIFC 2020). Stream complexity can be defined by the variety of habitats in a stream. One activity that continues to have a legacy effect on stream complexity is stream clearing and cleaning. State and federal agencies undertook major efforts to remove woody debris jams believed to be blocking fish passage, and forest-practice rules required the removal of slash (limbs and tops) from streams after timber harvest (Hicks et al. 1991, Stout et al. 2012). Debris removal can cause a decline in channel stability, a reduction in the quality and quantity of pool habitat, and an increased frequency of riffle habitat (Hicks et al. 1991). Numerous reviews of the biological role of large woody debris in streams in the Pacific Northwest (Harmon et al. 1986, Gregory et al. 1991, Bisson et al. 1997, Sergeant 2004) have concluded that LWD plays a key role in physical habitat formation, in sediment and organic-matter storage, and in maintaining habitat complexity in stream channels. Efforts to "clean" the stream channel for fish passage that began in the 1940s and continued through the 1970s resulted in the loss of salmon habitat (Botkin et al. 1995).

Another activity that continues to have a legacy effect on stream complexity is the use of splash dams and streams to transport logs, which was a common practice in Washington's coastal rivers from the late 1800s to the mid-1900s (Coast Salmon Partnership 2020). A splash dam is a temporary wooden dam historically used to control the level of water and float more logs downstream. Trees removed from the river banks were dropped into the rivers; splash dams helped transport fallen logs to mills located downstream. When ready, the logging company would open the splash dam, sending a cascade of water and logs downstream, scouring out spawning gravel and carving river beds to bedrock. Such stream cleaning depleted both wood and gravel, further degrading stream beds and disconnecting floodplains. Legacy effects from these activities are still impacting stream complexity through loss of wood or boulders that acted to hold back gravel in the channel, loss of large trees that act as key constituents of log jams, and incision of stream channels and loss of floodplain connectivity (Stout et al. 2012).

The Coast Salmon Partnership (CSP; 2022) documented over 235 splash dam and 38 stream cleanout locations, mostly in the Grays Harbor and Willapa Bay areas. CSP created an online web map (Coastal Washington Splash Dams and Stream Cleanout Inventory[9]) that provides historical information and current levels of channel incision, wood cover, and spawning gravel. The splash dam structures are mostly gone, but their legacy for rivers and salmon habitat remains. Such information helps put current and future planning restoration projects into context.

Sedimentation

One of the continuing stressors from land-use practices such as timber harvest and riparian forest removal is the residual effects of increased input of fine sediment into streams. This impact does not stop when timber harvest activities are complete. Road building and

[9] https://coastsalmon.maps.arcgis.com/apps/webappviewer/index.html?id=b79b70981d4c4b0c8f4853a10c102a27

other timber harvest activities have resulted in mass wasting and surface erosion that will continue to elevate the level of fine sediments in spawning gravels and fill the substrate interstices inhabited by invertebrates (Suttle et al. 2004).

The effects of sedimentation on salmonids are well documented and include: clogging and abrasion of gills and other respiratory surfaces; adhering to the chorion of eggs; providing conditions conducive to entry and persistence of disease-related organisms; inducing behavioral modifications; entombing different life stages; altering water chemistry by the absorption of chemicals; affecting usable habitat by scouring and filling of pools and riffles and changing bedload composition; reducing photosynthetic growth and primary production; and affecting intergravel permeability and dissolved oxygen levels (Hicks et al. 1991, Suttle et al. 2004, Jensen et al. 2009). Sediment effects can be grouped into effects of suspended sediment (turbidity), fine sediment that settles into the bed, and coarse sediment.

Turbidity from continued sediment suspension can decrease photosynthesis of aquatic plants and clog respiratory and feeding systems of animals (Bash et al. 2001). Loss of aquatic plants reduces the abundance of snails and invertebrate prey for young salmonids. Turbid water may also impact fry emergence and/or reproduction and social behaviors (Berg and Northcote 1985).

Fine sediment that settles into the stream bed affects both egg survival in the gravel and production of benthic invertebrate prey (Hicks et al. 1991). In a more recent study, egg-to-fry survival asymptotes were only 10% when fine sediment (< 0.85 mm) exceeded 25% (Jensen et al. 2009). Survival of eyed eggs was > 90% until fine percentages increased to about 20–25%, beyond which survival decreased (Jensen et al. 2009). Sediment and particles deposited as bedload sediment and unstable spawning gravels may also negatively affect survival. Increased sedimentation of gravels and pools can also increase stream temperatures (Hagans et al. 1986).

Coarse sediment (generally small gravels and larger) can fill pools with sediment and aggrade the streambed (Beechie 1998, Beechie et al. 2005), resulting in reduced flood flow capacity as well as wider and shallower streams with less structure and undercut banks. These physical habitat changes cause decreased stream stability and increased bank erosion, which exacerbates existing sedimentation problems. Erosion can also result in increased debris torrents which may decrease cover in some places but increase debris elsewhere, blocking migration and reducing survival (Hicks et al. 1991).

Water quality

Adequate water quality is important to all salmon life stages (Richter and Kolmes 2005). Salmon require cold water for spawning, rearing, and migrating. Temperature may affect salmonids directly or indirectly through interaction with other important factors. High temperatures can cause weight loss, disease, competitive displacement by species better adapted to the prevailing temperature, or death (Sullivan et al. 2000). Water temperature and dissolved oxygen requirements vary depending on life stage, but in general, a water temperature range of 10–14°C is preferred, and long-term exposure to temperatures warmer than 24°C or dissolved oxygen concentrations of ≤ 5 mg/L is fatal to salmon (Smith 2005).

Smith (2005) summarized water quality for each WRIA and rated water quality as "good" when water temperatures were below 14°C, "fair" in the range of 14–15.6°C, and "poor" when warmer than 15.6°C. Dissolved oxygen levels are considered "good" when above 8 mg/L, "fair" in the range of 6–8 mg/L, and "poor" when less than 6 mg/L. For WRIAs 20–24, (Smith 2005) provided an overall ranking of "poor" for water quality, and "poor–good" for WRIA 19.

NWIFC (2020) report mentions that in the Makah Area of Interest (WRIA 19 and a portion of WRIA 20), 40 water bodies were placed on the 303(d) list for water pollution, an increase of eight since 2016. Water temperature remains by far the most common pollutant, although the proportion of stream length impaired by temperature dropped to 79% from 86%, followed by dissolved oxygen, which saw the proportion of stream length impaired increase from 7% to 17% since the 2016 State of Our Watersheds Report (NWIFC 2016). For the Quinault Indian Nation's Area of Interest (WRIAs 21–23), 53 water bodies are currently listed in 303(d) for water pollution, an increase of 22 since 2012. Water temperature is the most common problem, although the proportion of stream length impaired by temperature decreased from 49% to 42%, followed by dissolved oxygen, which increased from 12% to 31% since 2012. The Chehalis River is the single most polluted body of water by water temperature, dissolved oxygen, and turbidity (NWIFC 2020). For Willapa Bay (WRIA 24), Smith (1999b) reported high water temperatures and low dissolved oxygen in the summer months as a major limiting factor.

Fish passage

Fish passage barriers are one of the main threats to salmon recovery. Adult salmon need access to upstream freshwater habitats to reproduce, and juvenile salmon need downstream access to the ocean to survive. Roads, culverts, tide gates, dikes, and levees present partial or complete barriers to upstream and downstream passage. Partial barriers can be temporal, based on flow or temperature, or selective, (e.g., based on fish size and/or migration direction). Barriers still persist in all watersheds, impacting a significant number of streams.

Smith (2005) rated access based on the percent of known or potential habitat blocked by artificial barriers as "poor" (> 20% habitat blocked), "fair" (10–20% blocked), or "good" (< 10% blocked). Smith (2005) reported access as "poor" for WRIA 19, unknown for WRIAs 20–23, and "good/fair" for WRIA 24. Roads reduce forest vegetation, reduce LWD recruitment, contribute to sedimentation, and can separate the river from the floodplain. NWIFC (2020; cf. Smith 2005) provided a summary map of road density ratings (their p. 141) and concluded that WRIAs 19 and 20 had "fair" road densities, WRIA 21 had "good" road densities, and WRIAs 22–24 had "poor." Smith (2005) defined "poor" as > 3 miles of roads per mi^2, "fair" as 2–3 mi/mi^2, and "good" as < 2 mi/mi^2.

Smith (2005) also considered how low flows can act as a barrier for salmon because low flows can reduce juvenile rearing habitat, block upstream migration, reduce access to spawning areas, and increase water temperatures. Additionally, if spawning has already occurred, low flows can dewater redds, killing eggs incubating in the gravel. Smith (2005) reported that the major cause of unnatural low flows is water withdrawals from irrigation, industrial and domestic use, and water transfers between basins. Removal of water, either directly from the stream channel or from wells that are in hydraulic continuity with stream flows, reduces the amount of in-stream flow and usable wetted area remaining for support of adult salmonid spawning and juvenile rearing. Smith (2005) attempted to rate low flows but noted that thresholds to define low flows have not been generally established for salmon production. The author rated low-flow conditions as "poor" under several circumstances, including 303(d) listing for low flows, known salmon mortality due to flows, stream closures due to overappropriations when the stream produces species known to use the area during the low-flow period, and when other studies have documented low-flow problems for salmon in a particular stream that are not natural conditions. In the low-flow summary map, Smith (2005) reported low-flow conditions as "unknown" for WRIAs 19, 20, and 22, "good" for WRIAs 21 and 24 and "poor" for WRIA 23. The author reported that data gaps for low-flow conditions were extensive and there was a lack of linkage to salmon production, a lack of standardization for low-flow thresholds, and a lack of trend monitoring of flows.

Since the Smith (2005) report, much work has been done to inventory and address fish passage barriers along the WC. WDFW maintains a database of fish passage information, including culverts and other potential fish passage barriers, as well as habitat data for use in planning fish passage and habitat restoration projects. WDFW also created the Washington State Fish Passage web application[10] to provide information about fish passage barriers inventoried by WDFW, other agencies, and partner organizations. WDFW cautions that the data in the web application may not represent a complete inventory of fish passage barriers, but provides a scope of known anthropogenic fish passage barriers across the Washington coast. From the web map (accessed 1 August 2024), the Hoh–Quillayute watershed (WRIAs 19 and 20) has 481 known anthropogenic barriers, the Queets–Quinault watershed (WRIA 21) has 489, Grays Harbor watershed (WRIA 22) has 661, the lower Chehalis watershed (WRIA 23) has 1,251, and the Willapa Bay watershed (WRIA 24) has 333. In addition, the Washington State Department of Transportation (WSDOT) created a passage inventory database that identifies corrected and uncorrected barriers and the level of fish blockage associated with state highways (Figure 30). WSDOT does caution that the purpose of the database is to underscore the challenges and complexities to correct fish passage barriers and that the data do not represent exhaustive inventories, but the field survey data are updated as knowledge is improved.

Many restoration actions have occurred and/or are underway to remove or replace culverts and fix fish passage and restore habitat. In CSP's (2023) annual report, they reported that, since 2010, more than 156 fish passage projects have reconnected 742 km (461 mi) of habitat for salmon and steelhead across the WC, and an additional ~1,700 fish barriers have been corrected in commercial forests as part of the state's Road Maintenance and Abandonment Plan. Additionally, various projects funded through the Washington State Recreation and Conservation Office since 2000 have led to the protection and restoration of riparian habitats for almost 133.6 km^2 (51.6 mi^2) on the WC (CSP 2022). CSP's (2022) annual report summarizes various restoration efforts by WRIA, including many efforts undertaken by the tribes. In WRIA 20, 36 fish passage barriers have been corrected, and sediment transport has improved due almost 2.1 km^2 (0.8 mi^2) of upland restoration, 5.4 km^2 (2.1 mi^2) of riparian restoration, 0.05 km^2 (0.02 mi^2) of floodplain reconnection, and 48 km (30 mi) of in-stream

[10] https://geodataservices.wdfw.wa.gov/hp/fishpassage/index.html

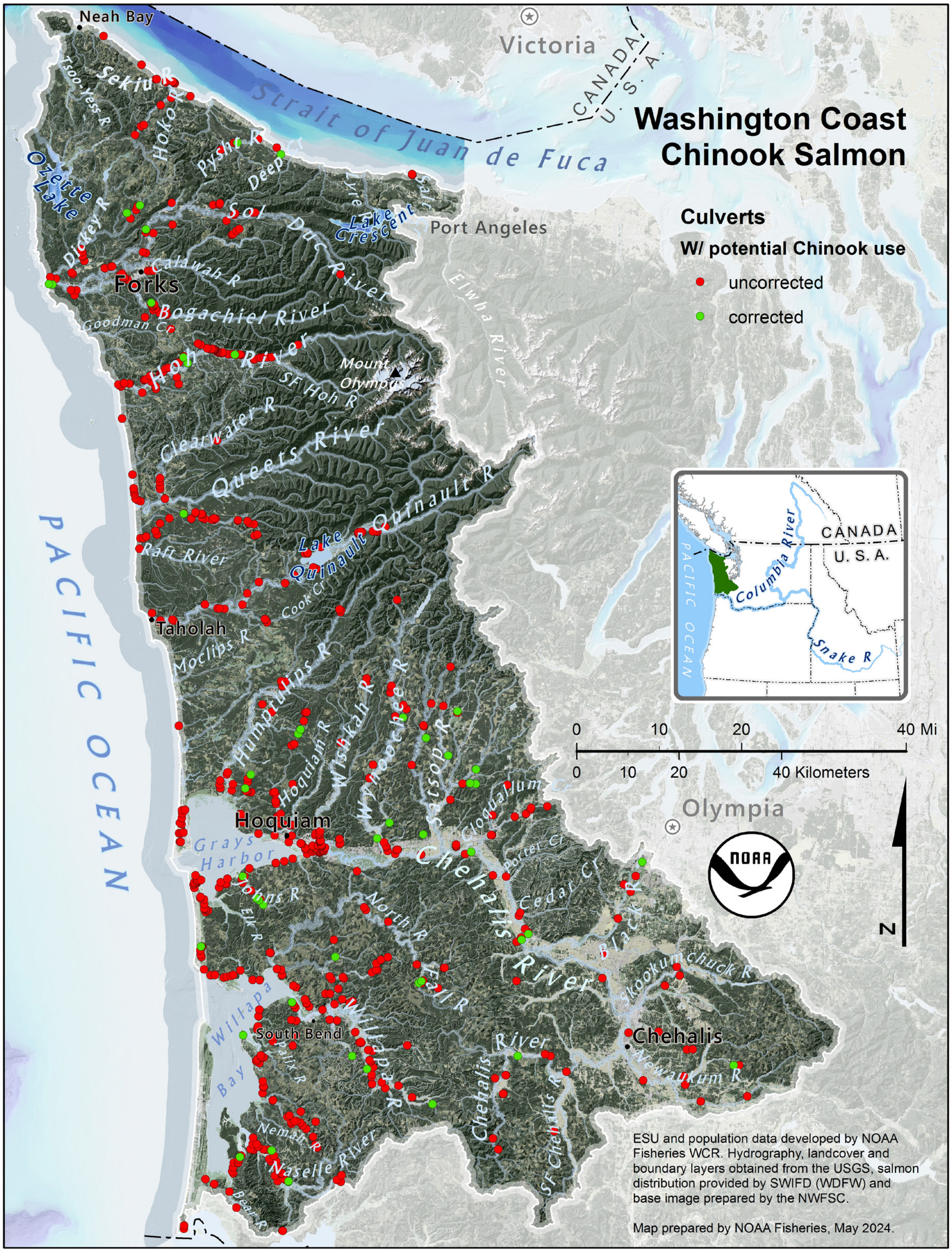

Figure 30. Fish passage barrier assessment showing corrected and uncorrected culverts with potential Chinook salmon use in the WC Chinook Salmon ESU along the coast highways (WDFW 2019).

restoration. In WRIA 21, corrections to 33 fish passage barriers have occurred, improving sediment transport due to the nearly 2.1 km^2 (0.8 mi^2) of upland area restored, 24.1 km^2 (9.3 mi^2) of riparian habitat restored, 0.05 km^2 (0.02 mi^2) of floodplain reconnected, and 9.7 km (6 mi) of in-stream restoration. In WRIAs 22 and 23, corrections to 152 fish passage barriers have occurred, improving sediment transport due to nearly 4.9 km^2 (1.9 mi^2) of upland restoration, 0.60 km^2 (0.23 mi^2) of riparian habitat restoration, 0.51 km^2 (0.19 mi^2) of floodplain habitat reconnection, and 3.2 km (2 mi) of in-stream restoration. In WRIA 24, corrections to 26 fish passage barriers have occurred, sediment transport has improved due to 0.05 km^2 (0.02 mi^2) of upland restoration, and in-stream habitat has been improved by 3.65 km^2 (1.4 mi^2) of riparian restoration and 8 km (5 mi) of in-stream restoration.

Risk Factor 2: Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Commercial, recreational, and tribal harvest

WC Chinook salmon are harvested in tribal, commercial, and recreational fisheries in the ocean and freshwater, with harvest management conducted by different authorities throughout the life of a salmon. Harvest restrictions have been used for many decades to reduce impacts and to manage the number of adults escaping to spawning grounds. In ocean fisheries, Chinook salmon populations co-mingle, and nonselective harvest tends to disproportionately impact less productive stocks. Harvest type, timing, and location can also alter size, age structure, and migration timing for both smolts and adults. Commercial, recreational, and tribal harvest is described in detail in Harvest.

Scientific and educational utilization

The utilization (take) of WC Chinook salmon for scientific and educational purposes in Washington is monitored by WDFW through its Scientific Collection Permits (SCP). In 2019–23, WDFW issued an average of 22 permits annually, with an average take of 1,365 adult Chinook salmon, with zero reported mortalities. The annual reported take for juvenile WC Chinook salmon averaged 67,496, with 45 mortalities. WDFW employees are exempt from the state's permit requirements, and we do not have information on the amount of take occurring from WDFW's research projects. Overall, the research permitted by WDFW has had minimal effects on the species' abundance.

Risk Factor 3: Disease or Predation

Disease

Infectious disease is one of many factors that can influence adult and juvenile salmon survival. Chinook salmon are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment. Specific diseases—such as amoebic gill disease (*Neoparamoeba perurans*), bacterial kidney disease (*Renibacterium salmoninarum*), bacterial cold water disease (*Flavobacterium psychrophilum*), enteronecrosis (*Ceratomyxa shasta*), columnaris (*Flavobacterium columnare*), furunculosis (*Aeromonas salmonicida*), Ich (*Ichthyophthirius multifiliis*), infectious hematopoietic necrosis (infectious hematopoietic necrosis virus), trichodiniasis (*Trichodina* spp.), enteric redmouth disease (*Yersinia ruckeri*), black spot disease (caused by digenean trematodes in the families Diplostomatidae and Heterophyidae), and viral erythrocytic inclusion body syndrome (caused by an unclassified virus)—are known, among others, to affect Chinook salmon (Rucker et al. 1954, Wood and WDFW 1979, Wertheimer and Winton 1982, Leek 1987, Foott et al. 1994). Common diseases that affect Chinook salmon on the WC include amoebic gill disease, bacterial cold water disease, bacterial kidney disease, columnaris, furunculosis, Ich, and trichodiniasis (M. Finley, WDFW, personal communication).

Naturally produced Chinook salmon may contract diseases through the water column from a number of natural vectors (i.e., waterborne pathogens; Buchanan et al. 1983); infection may also be contracted through interactions with infected fish (Fryer and Sanders 1981, Evelyn et al. 1984). Salmonids can be infected yet not show symptoms of the disease. Increased physiological stress (crowding in hatchery raceways, release from a hatchery into a riverine environment, high and low water temperatures, etc.) and physical injury in migrating juvenile salmonids may increase the susceptibility of migrating salmonids to pathogens or diminish the salmon's immune system's ability to suppress the disease (Matthews et al. 1986, Maule et al. 1988). The presence of adequate water quantity and quality during late summer is a critical factor in controlling disease epidemics. As water quantity and quality diminish, and freshwater habitat becomes more degraded, many infected salmonid populations may experience large mortalities because added stress can trigger the onset of disease (Holt et al. 1975, Wood and WDFW 1979).

Fish hatcheries are commonly associated with fish diseases, in part because of the high densities and rearing conditions they subject salmon to, but also because hatcheries, in contrast to natural systems, are actively monitored for pathogens. WDFW and NWIFC operate fish hatcheries on the WC following *The Salmonid Disease Control Policy of the Fisheries Co-Managers of Washington Sta*te (2006).[11] This policy is designed to protect free-ranging and cultured fish populations from management activities that could cause the importation, dissemination, and amplification of pathogens known to adversely affect salmonids. The management activities described in the policy include the transfer of gametes, eggs, fish, carcasses, or water between watersheds. The policy describes minimum fish health standards. Co-managers may implement additional practices or measures at their facilities at their discretion. Acknowledging that many complex fish health situations will arise, the co-managers meet annually to discuss fish health issues.

Additionally, the NWIFC member tribes created a Tribal Fish Health Program (1988) to meet the growing fish health needs of their salmon enhancement and supplementation programs.[12] The program's goal is to assist tribes in rearing and releasing healthy fish that will help to sustain tribal fisheries and/or restore wild populations. The Tribal Fish Health Program provides services to its member tribes in preventative fish health care, disease diagnostics, treatment, and fish health monitoring.

[11] https://files.nwifc.org/fish-health/FinalDiseasePolicy-July2006_Ver3.pdf
[12] https://files.nwifc.org/fish-health/TribalFishHealthManual-20070501.pdf

Predation by marine mammals

Depending on the life-history stage, salmonids are prey for other fishes—particularly during salmonid juvenile life stages—birds, and marine mammals, including Resident killer whales. Congress passed the Marine Mammal Protection Act (MMPA) in 1972 in response to increasing concerns among scientists and the public that significant declines in some species of marine mammals were caused by human activities. The MMPA's protections have stopped the decline of many marine mammal populations and have led to the recovery of several in the northeastern Pacific Ocean, such as populations of harbor seals, Steller sea lions, and California sea lions. Although the diets of seals and sea lions are diverse and salmon may be a minor part of their diet, the overall increase in abundance of these species, as well as of Resident killer whales, may have implications for the long-term status of depleted, and in some cases ESA-listed, salmonid populations.

The four common marine mammal predators of salmonids in the northeastern Pacific Ocean are harbor seals (*Phoca vitulina richardii*), fish-eating killer whales (*Orcinus orca*), California sea lions (*Zalophus californianus*), and Steller sea lions (*Eumetopias jubatus*). Salmon have also been identified as a prey item of harbor porpoises (*Phocoena phocoena vomerina*), though little is known (Elliser et al. 2020). Recent research suggests that predation pressure on salmon and steelhead from seals, sea lions, and killer whales has been increasing in the northeastern Pacific Ocean over the past few decades (Chasco et al. 2017).

Along the Washington coast, harbor seal haul-out sites are located on intertidal islands, rocks, and reefs, with a peak of abundances of harbor seals occurring during the pupping season (June–early August) and annual molt (mid-August–September). Large numbers of Steller sea lions use offshore rocks in the vicinity of Split Rock seasonally; Steller and California sea lion haul-out sites are also located at Carroll Island, Bodelteh Islands, Cape Alava, and Tatoosh Island (Jeffries et al. 2000). Smith and Wenger (2001) reported that the most abundant marine mammal in Grays Harbor was the harbor seal, with haul-outs located in North Bay and the outer Grays Harbor estuary (Schroder and Fresh 1992). Harbor seals can be found year-round in Grays Harbor, though they generally pup and breed in Grays Harbor in the summer, and feed in the Columbia River in the fall through spring months (Beach et al. 1985, Brown et al. 1995). Smith and Wenger (2001) reported that in Washington State, the numbers of harbor seals increased by 7.7% annually between 1978 and 1993, with an estimated population of 5,422 seals along the Washington coast in 1993. Small numbers of California sea lions may also be found seasonally in Grays Harbor, foraging or on docks in the Westport Marina (Jeffries et al. 2000). In Willapa Bay, numerous harbor seal haul-out sites are located on intertidal mudflats and sand bars (Jeffries et al. 2000).

Killer whales are classified as top predators in the food chain and are the world's most widely distributed marine mammal. Fish-eating killer whales in the northeastern Pacific consume at least 22 species of fish and at least one species of squid (Ford and Ellis 2006, Ford et al. 2016, Hanson et al. 2021), but salmon are their primary prey. Four populations of Resident killer whales occur in the Pacific Northwest: Southern Residents, Northern Residents, Southern Alaska Residents, and Western Alaska North Pacific Residents. WC Chinook salmon may therefore be a source of prey for multiple killer whale populations.

The diet of killer whales is the subject of ongoing research, including direct observation of feeding, scale and tissue sampling of prey remains, and fecal sampling. The diet data suggest that killer whales are consuming mostly larger (i.e., generally age-3 and up) Chinook salmon (Ford and Ellis 2006). Scale and tissue sampling from May to September in inland waters of Washington and British Columbia indicates that their diet consists of a high percentage of Chinook salmon, with monthly proportions as high as > 90% (Hanson et al. 2021). Ford et al. (2016) confirmed the importance of Chinook salmon to Resident killer whales in the summer months using DNA sequencing from whale feces. Salmon and steelhead made up to 98% of the inferred diet, of which almost 80% were Chinook salmon.

The relative impacts of marine predation on individual anadromous salmonid populations are not well understood. However, it is evident that anadromous salmonids have historically coexisted with both marine and freshwater predators. Recent analyses have shown that declining fishing mortality on WC Chinook salmon has been effectively replaced by increasing predation mortality by marine mammals as their populations rebound (Chasco et al. 2017, Couture et al. 2024). Predators play an important role in the ecosystem, culling out unfit individuals, thereby strengthening the species as a whole. Anthropogenic habitat alterations including dams, irrigation diversions, fish ladders, and artificial islands, have led to increased predation opportunities (Antolos et al. 2005, Hostetter et al. 2015, Evans et al. 2019, Moore and Berejikian 2022). Therefore, it would seem more likely that increased predation is a symptom of a much larger problem, namely, habitat modification and a decrease in water quantity and quality, changing ecosystems (including abundances of alternative prey), possibly hatchery effects, and/or climate change. For WC Chinook salmon—given there are no large dams or barriers and the relatively steady abundance trends—it seems unlikely that the level of predation would be in excess of what has historically existed, and unlikely to have increased substantially since the last review of this population.

Other marine predation

A variety of piscivorous marine predators have been identified, including sculpins (Cottoidea spp.), cod (*Gadus macrocephalus*), spiny dogfish (*Squalus suckleyi*), Pacific hake (whiting; *Merluccius productus*), Pacific jack mackerel (*Trachurus symmetricus*), and lamprey (*Entosphenus tridentatus*), although predation rates have rarely been quantified (Emmett and Krutzikowsky 2008). Beamish et al. (1992) documented predation of hatchery-reared Chinook and coho salmon by spiny dogfish. Beamish and Neville (1995) estimated that lamprey kill millions of juvenile Chinook salmon in the Fraser River plume annually. Seitz et al. (2019) noted that salmon shark predation may be a substantial source of oceanic mortality for immature and maturing Chinook salmon in the Gulf of Alaska. They also speculated that protections afforded by the Magnuson–Stevens Act (MSA) have likely contributed to increases in abundance of salmon sharks in the North Pacific. Based on the results of their study, Seitz et al. (2019) speculated that Pacific halibut (*Hippoglossus stenolepis*) or sleeper shark (*Somniosus pacificus*) may also be predators of Chinook salmon. Recent surveys in the North Pacific, however, captured few predatory species in conjunction with salmon and steelhead, suggesting that predation on the high seas may not be a major source of mortality (Weitkamp et al. 2024).

Freshwater predation

Freshwater predators on salmon include fishes, birds, and mammals, representing both native and non-native species (Sanderson et al. 2009). Of particular concern to Washington coastal rivers are warmwater fishes, which were introduced to provide recreational fishing opportunities and benefit from warming rivers. Smallmouth bass (*Micropterus dolomieu*) are an introduced species with a varied diet and are known to prey on young salmonids, especially Chinook salmon; they may also prevent salmon from effectively feeding (Fritts and Pearsons 2008, Carey et al. 2011). In the Chehalis River, Winkowski et al. (2024) reported that smallmouth bass are distributed throughout the mainstem Chehalis River and occupy tributary habitat, including the Skookumchuck, Newaukum, and South Fork Chehalis Rivers. These areas are where most spring Chinook salmon spawn and likely prey upon newly emerged fry, where an average of 79% of the spring Chinook salmon spawning has occurred over the past 20 years.

It is not clear how predation by any predator interacts with other sources of natural mortality, including disease. Limited evidence shows that avian predators of juvenile salmon are selective, targeting individuals that are unhealthy or smaller than average (Schreck et al. 2006, Miller et al. 2014, Tucker et al. 2016). In addition, Jacobson et al. (2008) showed that juvenile salmon with high parasite burdens had higher mortality during early marine residence, which decreased when ocean conditions were favorable. It is not known how widely applicable these selective predation results are throughout the life-cycle of salmon, or the degree to which they are mediated by environmental conditions. At present, our understanding of the ways in which marine predation on salmon influences their population dynamics is generally lacking due to our inability to observe both highly mobile predators and prey.

Risk Factor 4: Inadequacy of Existing Regulatory Mechanisms To Address Identified Threats

A variety of federal, state, tribal, and local laws, regulations, treaties, and measures can directly and indirectly affect the abundance and survival of WC Chinook salmon and the quality of their habitats. NMFS (1998) found that the serious depletion of Chinook salmon and other anadromous salmonids was an indication that existing regulatory mechanisms had largely failed to prevent the depletion. Since then, various agencies have worked on addressing regulatory mechanisms. Below, we summarize the current management plans/strategies for federal, state, tribal, and local agencies.

Federal management

Forestry

The Northwest Forest Plan (NWFP) guides the management of federal forest lands in the Pacific Northwest, along with the Aquatic Conservation Strategy (ACS). The ACS is a regional-scale aquatic ecosystem strategy that includes components that collectively ensure that federal land-management actions achieve a set of objectives which include salmon habitat conservation. While the NWFP covers a very large area, the overall effectiveness of the plan in conserving Chinook salmon in the WC Chinook Salmon ESU and elsewhere is limited by the amount and distribution of federal land ownership. This limits the NWFP's ability to achieve its aquatic habitat restoration objectives at the watershed and river basin levels, and highlights the need for complementary salmon habitat conservation measures on nonfederal lands.

U.S. Forest Service (1990, amended 1994): Olympic National Forest land and resource management plan

The Olympic National Forest plan guides natural resource-management activities and establishes management standards and guidelines. The three principal effects of the 1994 amendment on the management of the Olympic National Forest include: 1) the long-term maintenance of late successional forest habitat, 2) emphasizing riparian habitat, fish habitat, and water quality, and 3) reducing levels of forest-management activities such as ground disturbance and vegetation manipulation.

U.S. Forest Service (2010): Watershed restoration plan for national forest system lands within the Calawah River watershed

The U.S. Forest Service's Pacific Northwest Region Aquatic Restoration Strategy is a regionwide effort to protect and restore aquatic habitat across Washington and Oregon. The strategy relies on a collaborative approach to restoration and on focusing available resources in selected high-priority watersheds to accomplish needed restoration activities on national forest system lands as well as other ownerships. In 2010, the Olympic National Forest selected the Calawah River watershed as its "focus watershed" for the Washington coast basin. Since 2010, the U.S. Forest Service has emphasized restoration within the Calawah River watershed and worked with partners to complete priority needs to protect and restore salmon and steelhead habitat. This action plan identifies various actions needed to protect and restore watershed health, water quality, and fish habitats on national forest system lands within the Calawah River watershed.

National Park Service (2008): ONP general management plan

This general management plan for Olympic National Park (ONP) represents a commitment by the National Park Service to the public and explains how ONP will be managed for the next 15–20 years. ONP protects over 120 km (75 mi) of Pacific coast, 800 lakes, and 6,437 km (4,000 mi) of rivers and streams that support some of the most extensive runs of wild salmon, trout, and char remaining in the Pacific Northwest. The ONP plan includes the types of actions that are required for the preservation of the park's resources, and establishes management zones within ONP with goals for resource conditions (Halofsky et al. 2011).

NOAA's Office of National Marine Sanctuaries (2011): Olympic Coast National Marine Sanctuary Management Plan

The National Marine Sanctuaries Act (NMSA) authorizes the U.S. Secretary of Commerce to designate and protect areas of the marine environment with special significance due to their conservation, recreational, ecological, historical, scientific, cultural, archeological, educational, or esthetic qualities as national marine sanctuaries. The Olympic Coast National

Marine Sanctuary (OCNMS) is located within the nearshore habitat of WRIA 20 and encompasses 8,563 km^2 (2,500 nmi^2) from the U.S.–Canada border south to the southern boundary of the Copalis National Wildlife Refuge, and extends 48–65 km (30–40 mi) offshore (Smith and Caldwell 2001). The OCNMS Management Plan includes action plans to address six priority topics: 1) fulfill treaty trust responsibility, 2) achieve collaborative and coordinated management, 3) conduct collaborative research assessments and monitoring to support ecosystem-based management, 4) improve ocean literacy, 5) conserve natural resources, and 6) understand the sanctuary's cultural, historical, and socio-economic significance.

Federal Clean Water Act

The federal Clean Water Act (CWA) of 1973 addresses the development and implementation of water quality standards, the development of total maximum daily loads (TMDLs),[13] the filling of wetlands, point source permitting, the regulation of stormwater, and other provisions related to protection of U.S. waters. Some authority for clean water regulation is retained by EPA and the U.S. Army Corps of Engineers (USACE), and some authority is delegated to Washington State.

Under Section 303(d) of the CWA, states, territories, and authorized tribes are required to develop lists of impaired waters that do not meet the water-quality standards set by states. The law requires that states establish priority rankings for waters on the lists and develop TMDL thresholds for these waters; however, TMDL criteria do not cover implementation policies and action plans that describe the specific actions needed to improve impaired watersheds.

A significant number of stream reaches in the range of the WC Chinook salmon do not currently meet water-quality standards. This is largely driven by nonattainment of the temperature criteria, and suggests that the TMDLs are currently insufficient to restore water quality in impaired waters.

USACE regulates the discharge of dredged and fill material into "waters of the United States" (WOTUS), including wetlands, through permitting under the CWA Section 404 Program.[14] The CWA 404 standard is that permitted activities should not "cause or contribute to significant degradation of the WOTUS." Activities that are regulated under this program include fill for development, water resource projects (such as dams and levees), infrastructure development (such as highways and airports), and mining projects. Section 404 requires a permit before dredged or fill material may be discharged into WOTUS, unless the activity is exempt from Section 404 regulation (e.g., certain farming and forestry activities). CWA Section 404 permit exemptions, particularly those affecting agricultural and transportation activities, therefore fail to prevent the degradation of tributary and mainstem habitat conditions resulting from these activities. Further, USACE continues to lack a comprehensive and consistent process to address the cumulative effects of the continued development of waterfront, riverine, coastal, and wetland properties.

[13] A TMDL includes a calculation of the maximum amount of a pollutant that can be present in a waterbody and still meet water-quality standards.
[14] On 25 May 2023, the Supreme Court (Sackett v. Environmental Protection Agency 2023), redefined the CWA's coverage of "waters of the United States."

USACE authorizes certain floodplain fill and removal activities with nationwide permits (NWPs). In 2021, USACE finalized the reissuance of existing NWPs with modifications (USOFR 2021a,b). The modifications are likely to increase the amount of fill and destruction of floodplain habitat allowed for NWPs. The NWP authorizations will disconnect off-channel stream and floodplain areas and result in simplification of stream habitats.

National Flood Insurance Program

The National Flood Insurance Program (NFIP) is a federal benefit program that extends access to federal monies or other benefits, such as flood disaster funds and subsidized flood insurance, in exchange for communities adopting local land-use and development criteria consistent with federally established minimum standards. Under this program, development within floodplains continues to be a concern because it facilitates development without mitigation for impacts on natural habitat values.

All U.S. West Coast salmon species, including 27 of the 28 species listed under the ESA, are negatively affected by an overall loss of floodplain habitat connectivity and complex channel habitat. The reduction and degradation of habitat has progressed over decades as flood control and wetland filling occurred to support agriculture, silviculture, or conversion of natural floodplains to urbanizing uses (e.g., residential and commercial development). Loss of habitat through conversion was identified among the factors for decline for most ESA-listed salmonids. Altering and hardening stream banks and wetlands, removing riparian vegetation, constricting channels and floodplains, and regulating flows have been identified as the primary causes of anadromous fish declines (USOFR 1999a, 2000).

State management

Washington State forest management

The rules that govern forest management on nonfederal lands include the Washington State Forest Practices Act and the Washington State Forest Practices Rules (Title 222 WAC) that provide rules and guidelines for lands to be managed consistent with sound policies of natural resource protection. These rules are designed to protect public resources, such as water quality and fish habitat, while maintaining a viable timber industry.

Forest Practices Habitat Conservation Plan (FPHCP)

In Washington State, forest practices are regulated through the Washington DNR's Forest Practices Program by means of the Forest Practices Act, Chapter 76.09 RCW, and Title 222 WAC. The Forest Practices Program and rules require the maintenance and restoration of aquatic and riparian habitat.

OESF Habitat Conservation Plan Planning Unit: Forest land plan environmental impact statement (June 2010)

The DNR manages 270,000 acres of forested state trust lands within the Olympic Experimental State Forest (OESF) Habitat Conservation Plan. The focus of this Environmental Impact Statement is to provide analysis of potential impacts from the proposed management alternatives and describe proposed changes to DNR's landscape management strategies for the OESF. The key management strategy being examined in this process is the implementation of the OESF riparian conservation strategy.

Large forest owners: RMAPs (2005)

Washington State's forest management laws require most private forest landowners to prepare a forest road inventory and schedule for repair work that is needed to bring roads up to state standards. All forest roads are required to be covered under an approved Road Maintenance and Abandonment Plan (RMAP) since 31 December 2005. RMAPs are prepared by the landowners and approved by DNR. Implementation of RMAPs in the WC region has benefited salmon through replacement of blocking culverts and reduction of sediment input into streams.

Washington's Road Maintenance Program

Washington State's regional Road Maintenance Program (RMP) improves roadway safety while safely maintaining highways and following the ESA. The RMP guidelines include input from local government agencies, WSDOT, NMFS, and other interested parties. In the regional RMP biological opinion (NMFS 2003), NMFS describes how the RMP is not likely to jeopardize ESA-listed salmon or their critical habitat. The maintenance program requires ten program elements and use of best management practices to achieve environmental outcomes, including the protection of habitat and water quality. This program includes measures carried out during road maintenance to protect threatened salmon and steelhead and their habitats.

WDFW's Fish Passage Barrier Removal Board (2015)

In 2015, the Washington state legislature created the Fish Passage Barrier Removal Board (RCW 77.95.160) to establish a statewide strategy for fish barrier removal and administering grant funding. WDFW partners with others to locate, prioritize, and fund fish passage barrier repairs across the state. WDFW requires fish screens on all water diversions and pumps when water is diverted for agricultural use. WDFW also offers free training on assessing fish passage barriers and surface water diversion screening, and provides guidance on designing climate change resilient culverts and bridges.

WDOE's water quality and stream flow standards

The Washington Department of Ecology's (WDOE) mission is to improve and protect water quality, manage and conserve water resources, and effectively manage coastal and inland shorelines to assure Washington State has sufficient supplies of clean water for communities and the natural environment. In January 2018, the Washington state legislature passed the Streamflow Restoration Law (90.94 RCW) that helps restore stream flows to levels necessary to protect and preserve in-stream resources and values such as fish, wildlife, recreation, aesthetics, water quality, and navigation. One of the most effective tools for protecting stream flows is to set in-stream flows, which are flow levels adopted into rule. In-stream flows cover nearly half of the state's watersheds and the Columbia River.

Washington State has an anti-degradation standard in law (90.48 RCW). These regulations include use-based criteria for existing and designated uses to set surface-water quality standards (WAC 173-201A). The use criteria for aquatic life specifically name salmonid life-history usages such as spawning, rearing, and migration.

Hydraulic activities in Washington are regulated through RCW 77.55, specifically RCW 77.55.181, which was recently added to the Fish Habitat Enhancement Project process, referred to as the Habitat Restoration pilot program. From this, any work in saltwater (nearshore) and freshwater that changes, diverts, obstructs, or uses the water flow or bed must be sure to maintain no net loss of fish and their habitats.

Washington coast salmon recovery plans

Washington State established salmon recovery regions in response to federal requirements that endangered species recovery plans be based on ESUs. The state responded to earlier ESA listings of Pacific salmon with a bottom-up approach to restoration known as "the Washington Way." Lead entities (comprising local, watershed-based organizations) were created by the Washington state legislature in 1998 (RCW 77.85.050–0707) to develop salmon habitat recovery strategies and to recruit organizations to conduct habitat restoration and protection projects. In 2007, the Washington Coast Sustainable Salmon Partnership (WCSSP), covering five WRIAs, was formed. It was the last regional organization to form, and unlike the other regions, the organization's genesis was not in response to ESA listings but rather in an effort to prevent a listing (WCSSP 2013). WCSSP was renamed the Coast Salmon Partnership (CSP) in 2008 and consists of the four lead entities (LEs): 1) the North Pacific Coast, 2) the Quinault Indian Nation, 3) the Chehalis River basin, and 4) Willapa Bay/Pacific County. Each LE has developed a salmon recovery plan that describes the elements of a strategic action framework for guiding the development and prioritization of salmonid habitat recovery projects:

1. WRIA 20: North Pacific Coast Salmon Restoration Strategy (2023),[15] North Pacific Coast Lead Entity Group.
2. WRIA 21: Queets/Quinault Salmon Habitat Recovery Strategy (2011),[16] Quinault Indian Nation Lead Entity Group.
3. WRIAs 22 and 23: Chehalis Basin Salmon Habitat Restoration and Preservation Strategy (2011),[17] Grays Harbor County Lead Entity Group (a.k.a. Chehalis Basin Lead Entity Group).
4. WRIA 24: Pacific County Salmon Restoration Strategy (2015),[18] Pacific County Lead Entity Group.

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[15] https://www.coastsalmonpartnership.org/wp-content/uploads/2023/05/2023_NPCLE_SalmonRestorationStrategy_Final_ApprovedMay2023.pdf
[16] https://www.coastsalmonpartnership.org/wp-content/uploads/2019/02/WRIA-21-Salmon-Habitat-Restoration-Strategy-June-2011-Edition.pdf
[17] http://www.chehalisleadentity.org/wp-content/uploads/2011_CBP_strategy_update_20111.pdf
[18] https://www.coastsalmonpartnership.org/wp-content/uploads/2019/02/Pacific-County-WRIA-24-Lead-Entity-Strategy-2015.pdf

In addition to state recovery plans, many watersheds also adopted management and restoration plans; for example, the North Pacific Coast (WRIA 20) adopted a Watershed Management Plan (2009)[19] that includes a Detailed Implementation Plan (2010).[20] The Chehalis River basin (WRIAs 22 and 23) adopted a Partnership Watershed Management Plan (2007–08)[21] including a detailed implementation plan. The Chehalis River Streamflow Restoration Plan was formally adopted by the Department of Ecology in 2021. This addendum to the Chehalis Basin Watershed Management Plan was developed under the guidance of the Chehalis Basin Partnership to comply with the State Streamflow Restoration Law (RCW 90.94). The Chehalis Strategy was created by the Washington state legislature in response to the 2007 floods in the Chehalis River watershed. It is a dual-focus program that looks at reducing flood damage and recovering aquatic species. In WRIA 21, the Aquatic Species Restoration Plan (2019)[22] was developed by the Quinault Indian Nation, the Confederated Tribes of the Chehalis Reservation, and WDFW, together with farmers, foresters, conservationists, other state agencies, local governments, and local landowners.

State permits for take of aquatic species

As discussed earlier, in Washington State it is unlawful to collect fish, shellfish, or wildlife, or their nests and/or eggs, for the purpose of research or display without first obtaining a Washington State scientific collection permit (RCW 77.32.240, WAC 220-200-150, WAC 220-450-030). State scientific collection permits (SCPs) are issued to scientists, researchers, educators, educational institutions, museums, aquariums, and zoos. Approved permits and authorizations can include terms and conditions such as sampling protocols, anesthetic guidelines, and temperature restrictions, which also promote responsible and ethical treatment of animals. WDFW's permit program allows the agency to coordinate efforts so that researchers track take activities happening statewide, manage resources including the protection of listed or sensitive species or areas where sampling could have negative impacts, and do not interfere with or overlap sampling areas.

Tribal management

The Northwest Indian Fisheries Commission (NWIFC) is a natural resources management organization that provides service support for 20 treaty Indian tribes in western Washington. NWIFC was created following the 1974 Boldt Decision (see Tribal fishing) that reaffirmed the tribes' treaty-reserved fishing rights. The NWIFC member tribes with reservations within the WC ESU are: Makah, Quileute, Quinault, and Hoh Tribes. NWIFC's role is to assist member tribes as natural resources co-managers, including support in biometrics, fish health, and salmon management. NWIFC also provides a forum for tribes to address shared natural resources management issues.

[19] https://rco.wa.gov/wp-content/uploads/2019/10/GSRO-AssessWatershedCoord.pdf
[20] https://www.clallamcountywa.gov/DocumentCenter/View/6405/WRIA-20-Detailed-Implementation-Plan-PDF?bidId=
[21] https://apps.ecology.wa.gov/publications/SummaryPages/0911038.html
[22] https://officeofchehalisbasin.com/asrp-phase-i-draft-plan/

In addition to the active role that WC tribes play in CSP lead entity groups, each tribe also has a Department of Natural Resources that manages their trust resources for environmental, cultural, and economic benefits. The tribes work with tribal councils to develop tribal ordinances and regulations, carry out resource management, and design/implement habitat management, protection, and restoration.

Harvest management

Here we summarize harvest regulations, noting that the discussion in Risk Factor 2 also describes aspects of harvest regulation.

Tribal fishing

On the Washington coast, tribal governments and their jurisdictions are established either by Presidential executive orders or by treaties with the United States that preceded the establishment of the State of Washington. The treaty tribes with reservations along the coast are the Quinault Indian Nation, the Hoh Tribe, the Quileute Tribe, and the Makah Tribe. The two executive order tribes that are federally recognized with reservations are the Chehalis Tribe and the Shoalwater Bay Tribe. The Chinook Tribe, in the Lower Columbia River and Willapa Bay areas, continues to seek federal recognition.

In a 1974 federal district court decision, Judge Boldt (and later the U.S. Supreme Court) affirmed that fishing rights of the treaty tribes did extend to their usual and accustomed (U&A) areas well beyond their reservation boundaries, and that these rights were reserved by the tribes in the treaties. The tribal treaty share was 50% of the fishery, with each tribe fishing in its respective U&A areas. The treaty tribes and the State of Washington were held to be co-managers of the fisheries. Ever since the Boldt Decision in 1974, the fisheries have been jointly managed by the treaty tribes and the State of Washington (Boldt, United States et al. v Washington, 1974).

Commercial and recreational fishing

The Pacific Coast Salmon Fishery Management Plan (1999) guides management of commercial and recreational salmon fisheries off the coasts of Washington, Oregon, and California. Since 1977, salmon fisheries in the U.S. Exclusive Economic Zone (EEZ) off Washington, Oregon, and California (4.8–322 km [3–200 mi] offshore) have been managed under PFMC's fishery management plans (FMPs). The Pacific Coast Salmon FMP covers the coastwide aggregate of natural and hatchery salmon species; in addition, the plan contains requirements and recommendations with regards to essential fish habitats for managed stocks. While all species of salmon fall under the jurisdiction of this plan, it currently contains mainly fishery management objectives for Chinook, coho, pink (odd-numbered years only), and any salmon species listed under the Endangered Species Act that are measurably impacted by PFMC fisheries.

Hatchery regulations

Here we describe overall hatchery regulations and then, in Risk Factor 5, we discuss potential negative impacts of hatchery production on natural-origin WC Chinook salmon. In 2005, the WDFW Hatchery Scientific Review Group (HSRG) completed a review of coastal hatcheries that assessed hatcheries' ability to conserve naturally spawning salmon while supporting sustainable fisheries. The outcomes from this 2005 review were a series of recommendations for hatchery reform, outlined in three reports that focused on the North Coast (WRIAs 20 and 21), Grays Harbor (WRIAs 22 and 23), and Willapa Bay (WRIA 24). Hatchery and Fishery Reform Policy #C-3619 was adopted by the Washington Fish and Wildlife Commission (FWC) in 2009. The policy was intended to guide a scientific and systematic redesign of the hatchery programs operated by WDFW in order to improve hatchery effectiveness in meeting management goals, including supporting sustainable fisheries. The 2009 plan was superseded by the 2021 Anadromous Salmon and Steelhead Hatchery Policy #C-3624 ,with the intended goal of providing direction, goals, and objectives to improve hatchery effectiveness and ensure compatibility between hatchery salmon and steelhead production and wild salmon and steelhead conservation and recovery.

Agricultural regulatory mechanisms

On the WC, federal, state, and private landownership is generally characterized by forest, agricultural, and municipal land-use activities. Riparian management policies are fragmented and not well integrated. State and federal agencies' rules for riparian land management involve separate rules based on jurisdiction, resulting in a range of standards for managing riparian conditions. These varied policies create diverse protection mechanisms, which influence varying ecological outcomes.

WDOE is the primary state agency responsible for implementing the CWA and general state water-quality laws. WDOE establishes water-quality standards to protect designated and existing beneficial uses, developing TDMLs under the CWA. Water-quality standards are adopted as rules and approved by the EPA.

The mission of the Washington State Department of Agriculture (WSDA) is to support viable agriculture while protecting consumers, public health, and the environment. WSDA is charged with protecting Washington State's ground- and surface-water resources from contamination by fertilizers and pesticides. WSDA works with the agricultural community and regulators to provide technical assistance to pesticide applicators, and ensures that pesticides which pose a risk to water resources are being handled and applied according to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) label. WSDA also partners with the U.S. Geological Survey, WDOE, and the Washington State Department of Health to monitor water resources for pesticide residues and to assess the effects of pesticides on surface and groundwater. WSDA's Endangered Species Program assesses the potential impacts of pesticides on threatened and endangered species and aquatic ecosystems to ensure that pesticide use is not a limiting factor in the recovery of species in Washington State. WSDA is gathering data on site-specific pesticide use, surface-water monitoring, and crop mapping in an effort to develop geospatial analysis tools that will help identify, evaluate, and potentially mitigate impacts to ESA-listed species. If a pesticide is identified as a potential problem for ESA-listed species, WSDA will work with Washington's regulated community to develop a local plan for protecting species while minimizing the impact on the state's pesticide users.

Risk Factor 5: Other Natural or Man-Made Factors

Proposed dam

The Chehalis River basin Flood Control Zone District (FCZD) has proposed a new flood retention dam (the "Proposed Project") and temporary reservoir on the mainstem of the Chehalis River. The dam and associated reservoir are designed to reduce flows during a major flood caused by rainfall in the Willapa Hills. The Proposed Project would only store floodwater during major floods; it would then slowly release the water during certain periods. When not operated for floodwater retention, the Chehalis River would flow through the structure's low-level outlet works at its normal rate of flow and volume and allow fish to pass up- and downstream. To provide a sense of how often the Proposed Project would store floodwater, the FCZD analyzed historic floods to determine which flow events would trigger certain operations, and noted these would have been initiated 10 times over the 82 years studied, and that many 2-, 4-, and 10-year floods would not have triggered some operations.

The proposed temporary reservoir would inundate an area that overlaps with historical spring-run Chinook salmon spawning habitat (Phinney and Bucknell 1975, Weyerhaeuser Co. 1994, Lestelle et al. 2019[23]). Redds in the temporary reservoir would be subject to deep-water inundation, which, depending on the redd location, could last for extended periods (e.g., 30 days). Deep-water inundation could decrease dissolved oxygen levels at the bottom of the temporary reservoir, which could result in egg loss.

From 2013–19, WDFW conducted annual redd surveys within the upper Chehalis River mainstem and tributaries, including the reservoir inundation area. Although WDFW did not survey all reaches over all years, the 2018 surveys were representative of reach distributions within the inundation area. Results indicate spring-run Chinook salmon spawning is limited upstream of the Proposed Project site (Ronne et al. 2020). From 2013–18, between one and 24 spring-run Chinook salmon redds were observed within the temporary inundation area. Although 24 redds were observed in the 2014–15 season, only one redd was observed in the 2015–16 and 2017–18 seasons. The estimated abundance for the 2015–16 to 2018–19 seasons was three to eight spring-run Chinook salmon adults. Ronne et al. (2020) estimated the number of spring-run Chinook salmon spawning upstream of the Proposed Project represents an estimated 1.25% of the total Chehalis River basin spawner abundance, but noted this can be much less in any one year.

Under Section 404 of the CWA, USACE determined that the Proposed Project may have significant impacts on the environment and released a draft environmental impact statement (EIS) on the proposed dam project (USACE 2020). In response to the draft EIS, FCZD made numerous changes to the Proposed Project design, including site realignment, volitional fish passage during dam construction, reduction of the inundation zone, addition of a conduit design for fish passage during reservoir draw-down, and the addition of a vegetative management plan. Additionally, FCZD developed a draft mitigation plan in 2022 that identified

[23] Lestelle, L., M. Zimmerman, C. McConnaha, and J. Ferguson. 2019. Spawning Distribution of Chehalis Spring-Run Chinook Salmon and Application to Modeling. Memorandum to Aquatic Species Restoration Plan Science and Review Team, 8 April 2019.

mitigation actions to offset potential impacts to salmon and to improve currently degraded habitat conditions. FCZD is continuing to work with its design and mitigation teams to study anticipated impacts on salmonid habitat and to develop Proposed Project design measures and operational protocols that will minimize effects to Chinook salmon spawning and rearing areas.

Climate change

Major ecological realignments are already occurring in response to climate change (Crozier et al. 2019). As reviewed by Siegel and Crozier (2020), the scientific literature published in 2019 showed that long-term trends in warming have continued at global, national, and regional scales. Globally, 2014 through 2023 were the warmest years on record both on land and in the ocean. 2023 was the warmest year on record until it was surpassed by 2024. Events such as the 2013–16 marine heatwave (Jacox et al. 2018) and the 2021 terrestrial heatwave have been attributed directly to anthropogenic warming (Herring et al. 2020). Global warming and anthropogenic loss of biodiversity represent profound threats to ecosystem functionality. These two factors are often examined in isolation, but likely have interacting effects on ecosystem function (Siegel and Crozier 2020). Conservation strategies now need to account for geographical patterns in traits sensitive to climate change, as well as climate threats to species-level diversity. Recent five-year status reviews for listed species of salmonids have summarized literature on ongoing (including warming and heatwaves) and projected climate change for the U.S. West Coast and mechanisms for climate change impacts to salmonids (see Ford 2022).

Climate change has already had an impact in the WC, foremost in the loss of glaciers in the headwater areas. The loss of glacial and snowfield melt, in combination with decreased summer precipitation, has led to a significant decline in summer low flows (Figure 31; NWIFC 2020). Between 1900 and 2015, the Olympic Mountains lost approximately half of their glacial and snowfield coverage (by area), with 35 glaciers and 16 perennial snowfields disappearing completely (Fountain et al. 2022). Based on current predictions, glaciers may completely disappear from the Olympics by 2070 (Fountain et al. 2022). There has been a corresponding increase in winter peak flows (Figure 31), in part due to precipitation falling as rain instead of snow and in the increase in atmospheric river rain events (NWIFC 2020). The Hoh River could be particularly vulnerable to increased sediment supply associated with high-altitude warming (East et al. 2017). This includes new sediment resulting from glacial retreat, shrinking perennial snow fields, melting permafrost, and mass wasting of recently deglaciated valley walls, all drivers of changes to downstream channel characteristics (East et al. 2017). There is an ongoing effort to restore riparian habitats, but it is unclear if the current level of restoration can keep pace with climate change efforts and correct for the legacy degradation due to timber activities.

Climate change is projected to alter habitat conditions in freshwater, estuarine, and ocean environments, thereby impacting Pacific salmon throughout their life cycle (Wainwright and Weitkamp 2013). In particular, as stream temperatures increase, many native salmonids face increased competition with invasive species that are more warmwater-tolerant (Sanderson et al. 2009). Changes in flow regimes may alter the amount of habitat available for spawning, potentially reducing the distribution of juveniles and thus further decreasing productivity through density dependence. Along with warming stream temperatures and

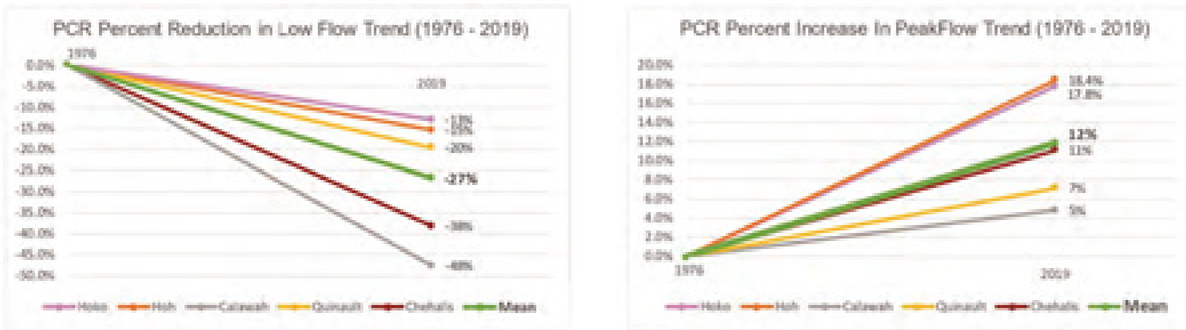


Figure 31. Changes in stream flows for select Pacific Coast Region (PCR) rivers. Reduction in low summer flows (left) and increases in winter peak flows (right). From NWIFC (2020).

concerns about sufficient groundwater to recharge streams, projections indicate nearly complete loss of existing tidal wetlands along the U.S. West Coast due to sea-level rise (Thorne et al. 2018), with 68% of Washington tidal wetlands expected to be submerged by the end of this century. Coastal development and steep topography prevent horizontal migration of most wetlands, causing the net contraction of this crucial habitat.

Increasing stream temperatures can affect salmon and steelhead—depending on species—at multiple life stages, including reproduction (egg viability; Berman 1990), incubation survival (eggs in the gravel), juvenile rearing (Bear et al. 2007, Fogel et al. 2022), smoltification, adult survival (Keefer et al. 2008), and migration timing. For example, average temperatures above 15–16°C can stop the smoltification process, while average temperatures below 12–13°C are ideal for this process. Temperatures above 21–22°C can create a migration block, and extreme temperatures (generally > 23°C) can kill fish in seconds to hours, depending on the circumstances and the degree of acclimation. Warm temperatures can also lead to greater risk of disease and parasites. Projections of maximum stream temperatures on the Washington coast for 2020, 2040, and 2080 (Isaak et al. 2018) show particularly severe warming in the Chehalis River basin and tributaries to Willapa Bay in 2020 and 2040, spreading to most low-elevation areas of the ESU by 2080 (Figure 32).

Changes in winter precipitation intensity and flood magnitudes will likely affect the incubation and/or rearing stages of most populations. Egg survival rates may decrease from more intense flooding that scours or buries redds (Goode et al. 2013, Nicol et al. 2022). Changes in hydrological regime, such as a shift from mostly snow to more rain, could drive changes in life history, potentially threatening diversity within an ESU/DPS (Beechie et al. 2006). Changes in summer temperature and flow will affect both juvenile and adult stages in some populations, especially those with extended juvenile freshwater rearing (steelhead most commonly emigrating as two-year old smolts) or summer adult return migration patterns (Figure 32, Figure 33; Beechie et al. 2006, Crozier and Zabel 2006, Quinn et al. 2007, Crozier et al. 2010).

Climate change may lead to mismatches between juvenile arrival timing and prey availability in marine environments (Siegel and Crozier 2020, Wilson et al. 2021). However, phenological diversity can contribute to metapopulation-level resilience by reducing the risk of a complete mismatch. Carr-Harris et al. (2018) explored phenological diversity of marine migration timing in relation to zooplankton prey for sockeye salmon from the

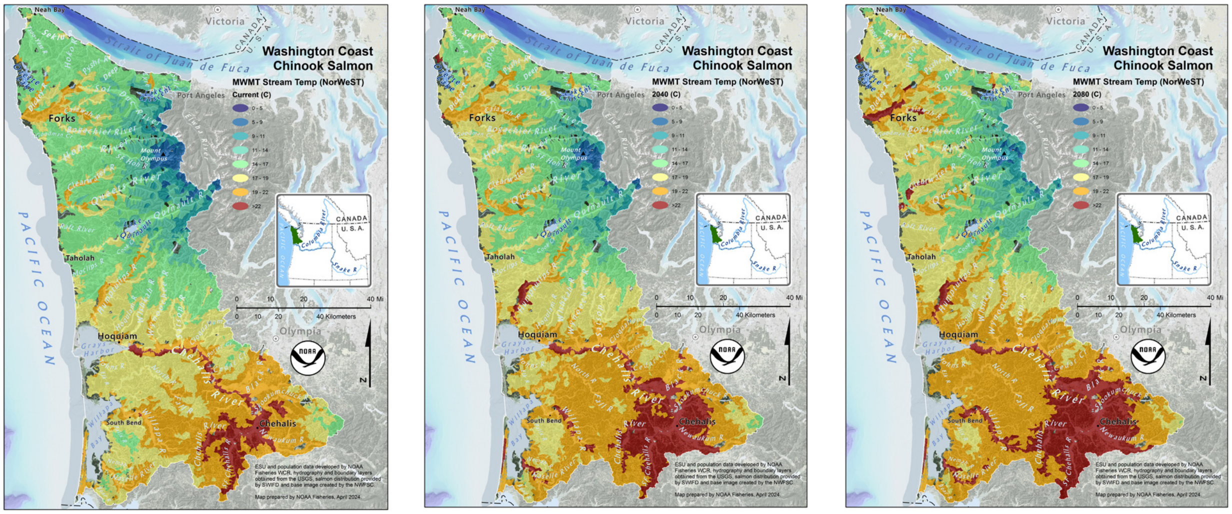


Figure 32. Maximum annual weekly mean stream temperature for the years 2020 (left), 2040 (middle), and 2080 (right). From Isaak et al. (2018).

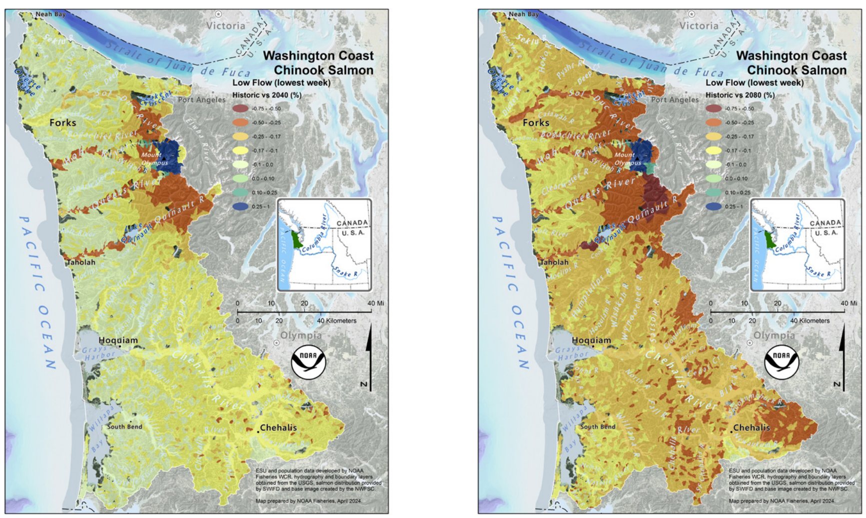


Figure 33. Projected changes in annual low flows for 2040 and 2080 vs. near-term conditions (Wenger et al. 2010).

Skeena River of Canada. They found that sockeye migrated over a period of more than 50 days. Populations from higher-elevation and further-inland streams arrived in the estuary later, and different populations encountered distinct prey fields. This underscores the importance of diversity in the SRT's evaluation of population and ESU viability.

Finally, climate change is expected to have profound impacts on the ocean environment, influencing ocean temperatures, currents, salinity, acidity, and the composition and presence of a vast array of oceanic species (Crozier et al. 2019). For Pacific salmon, warming oceans are expected to compress suitable oceanic habitats for all species (Abdul-Aziz et al. 2011, Langan et al. 2024), although the greater depths occupied by Chinook salmon make interpreting their past and future temperature preferences, and therefore ocean distributions, difficult (Langan et al. 2024). In addition to physical changes, record high abundances of pink salmon in recent decades have been shown to suppress the growth, fecundity, and survival of a wide range of species that overlap with pink salmon, including most other Pacific salmon species (Ruggerone et al. 2023, Connors et al. 2024). Washington coast Chinook salmon are not expected to be affected by pink salmon as much as northern Chinook salmon populations or other salmon species due to limited overlap, especially in the central and western Pacific Ocean (Langan et al. 2024). U.S. West Coast fall Chinook salmon populations (including those from the Washington coast) remain on the continental shelf and do use high-seas habitats (Riddell et al. 2018). They undoubtedly interact with juvenile pink salmon in this coastal habitat, but differences in diet, relative abundances, and depth distributions likely minimize negative impacts to Chinook salmon (Grimes et al. 2007, Smith et al. 2024). Less is known about spring Chinook salmon distributions and the extent to which they have greater overlap with pink salmon than fall runs (Riddell et al. 2018). Limited CWT data (Figure 8, Weitkamp 2010) suggest the marine distributions of spring-run Chinook salmon are more similar to fall-run Chinook salmon from the same areas.

Climate vulnerability assessments for WC Chinook salmon

Anadromous Pacific salmon species depend on a sequential series of freshwater, estuarine, and marine habitats as they complete their complex life cycles, all of which are being affected by climate change. Subtle differences in habitat use and life-history characteristics for U.S. West Coast salmon influence their vulnerability to climatic changes, with some being more (or less) exposed and/or able to adapt to their particular suite of climate effects. Consequently, knowing how sensitive, exposed, and therefore vulnerable a particular ESU will likely be to climate change is necessary to fully evaluate whether a species is likely to become at risk of extinction in the foreseeable future. Crozier et al. (2019) undertook a comprehensive climate vulnerability assessment for Pacific salmon and steelhead along the U.S. West Coast, focusing on ESUs that had received or were candidates for protection under the ESA. The assessment used professional judgment by experts on salmon ecology to evaluate three components of vulnerability for each ESU: 1) biological sensitivity, which is a function of individual species characteristics; 2) climate exposure, which is a function of geographical location and projected future climate conditions; and 3) adaptive capacity, which describes the ability of an ESU to adapt to rapidly changing environmental conditions.

Results of the assessment indicated that most U.S. West Coast salmon ESUs and steelhead DPSes had "high" exposure to these factors, due largely to high exposure to ocean acidification and elevated sea surface and stream temperatures. In contrast, sensitivity to these factors was highly variable, although most listed Chinook salmon ESUs fell in the "high" or "very high" categories. Chinook salmon populations with subyearling life histories produced relatively low vulnerability scores during the *early life history* and *juvenile freshwater* stages, due to limited rearing in freshwater in summer, when thermal impacts, hydrologic regime shifts, and low-flow impacts are expected to be highest.

The WC Chinook Salmon ESU was not included in the Crozier et al. (2019) assessment, so we evaluated its vulnerability using results for ESUs that had similar life histories, geographic ranges, and human land-use activities (e.g., extensive forestry, limited urban areas; NWIFC 2020). Specifically, we used the Crozier et al. (2019) assessments for the Lower Columbia River (LCR) Chinook Salmon, the LCR and Oregon Coast (OC) Coho Salmon, and the Ozette Lake Sockeye Salmon ESUs. We also included results for Oregon Coast Chinook

Table 14. Life-stage sensitivity, exposure, and overall vulnerability scores for the Lower Columbia River (LCR) and Oregon Coast (OC) Chinook Salmon, Lake Ozette Sockeye Salmon, and LCR and OC Coho Salmon ESUs, and expected scores for the WC Chinook Salmon ESU. Numerical scores: low (*L*) = 1, moderate (*M*) = 2, high (*H*) = 3, very high (*VH*) = 4. Assessment sources: *A* = Crozier et al. (2019), *B* = OC and SONCC SRT (2024).

Category	**LCR Chinook (spr & fall)**	**OC Chinook (spr & fall)**	**Ozette Lake sockeye**	**LCR coho**	**OC coho**	**WC Chinook**
Assessment source	A	B	A	A	A	*Expected*
Early life history	1.3	L	2.0	1.7	1.8	*L–M*
Juvenile freshwater	1.5	L	2.3	3.2	3.1	*H*
Estuary	2.2	M	1.3	1.8	2.3	*M*
Marine	2.8	M	2.4	3.0	3.0	*M*
Adult freshwater	1.6	L–M	1.6	1.7	1.6	*L–M*
Cumulative life-cycle effects	1.3	L	M	1.9	2.4	*L–M*
Other stressors	2.4	M	2.7	2.5	2.6	*M–H*
Hatchery influence	3.3	L	L	2.9	1.3	*L–M*
Population viability	2.5	L–M	2.8	2.9	2.2	*L–M*
Ocean acidification (OA) sensitivity	1.9	L–M	2.3	1.9	1.9	*M*
Overall sensitivity score	Moderate	Moderate	Moderate	High	High	*Moderate*
Stream temperature	3.4	H	3.3	3.3	3.2	*H*
Summer water deficit	2.3	M	2.4	2.7	2.7	*M*
Flooding	2.0	M	1.8	1.5	1.6	*M*
Hydrologic regime	2.4	L	1.3	2.2	1.3	*L–M*
Sea-level rise	2.1	M	1.3	2.0	2.0	*M*
Sea surface temperature	3.4	H	3.2	2.8	2.8	*H*
OA exposure	3.8	VH	3.9	3.9	3.9	*VH*
Upwelling	2.3	M–H	1.7	1.8	1.7	*M–H*
Ocean currents	2.0	M	1.8	1.9	1.9	*M*
Overall exposure score	High	High	High	High	High	*High*
Overall vulnerability rank	**Moderate**	**Moderate**	**Moderate**	**High**	**High**	***Moderate***

salmon using the Crozier et al. (2019) assessment framework (OC and SONCC Status Review Team 2023). We included Ozette Lake sockeye because it was the only ESU to be formally evaluated by Crozier et al. (2019) in the Washington coast Chinook salmon geographic area. Using dedicated assessment teams to generate new scores, as done by Crozier et al. (2019), would have been preferable, but was beyond the scope of the present evaluation.

Table 14 lists the life-stage and sensitivity scores for the five Chinook, coho, and sockeye salmon ESUs evaluated by Crozier et al. (2019) and the OC and SONCC Status Review Team (2023), as well as our best estimate of how the factors would affect the WC Chinook Salmon ESU. Because we relied on these other ESUs to estimate the scores, there is much overlap in individual factors and overall scores between all ESUs in the table.

For the *early life history, estuary*, and *adult freshwater* stages, all Chinook salmon ESUs had roughly overlapping adult river-entry timing (spring and fall runs), fall spawn timing, limited freshwater residency (subyearling migrants except some yearling migrants in LCR), and potentially extended estuarine residency in the larger estuaries. Consequently, we relied heavily on scores for the listed Chinook salmon ESUs and predicted "low–moderate" sensitivity for these attributes for the WC Chinook Salmon ESU. For the *marine* stage sensitivity, WC Chinook salmon marine distributions extend from local waters to southeastern Alaska (as discussed earlier in this section), similar to the LCR and OC Chinook Salmon ESUs (although fish from these ESUs are also caught farther south). In addition, their diets, length of ocean residency, and factors affecting mortality (including *ocean acidification*) are expected to be comparable, and were thus scored similarly ("low–moderate"). We also ranked *cumulative life-cycle effects*, following the Chinook and coho salmon ESUs, as "low–moderate."

Hatchery influence for the WC Chinook Salmon ESU is relatively low compared to other ESUs (e.g., LCR Chinook and coho salmon)—although some basins (Hoko River, Willapa Bay) have high proportions (> 0.5) of hatchery fish spawning naturally—and was ranked "low–moderate" overall. *Population viability* was expected to score "low–moderate," consistent with the OC Chinook Salmon ESU, which also has high productivity (see earlier sections). All four ESUs evaluated by Crozier et al. (2019) had moderate-to-high scores for *other stressors*, which include factors such as habitat loss and degradation (including high stream sediment loads for Ozette Lake sockeye salmon), pathogens, invasive species, and toxins, which we used for WC Chinook salmon. We assigned a "moderate" *overall sensitivity score* to the WC Chinook Salmon ESU, consistent with LCR and OC Chinook and Ozette Lake sockeye salmon, but lower than the two coho salmon ESUs, which were "high" due in part to extended freshwater rearing as juveniles.

For expected exposure scores, we relied on scores for all the comparative ESUs because of their shared stream, estuarine, and ocean habitats, although the details of how each species uses these habitats can differ. Specifically, expected scores for *stream temperature, summer water deficit, flooding, hydrologic regime*, and *sea-level rise* for the LCR and OC Chinook Salmon ESUs were applied directly to the WC Chinook Salmon ESU. For ocean factors (*sea surface temperature, ocean acidification exposure, upwelling*, and *ocean currents*), we relied on all five comparable ESUs, which generally had similar scores. The resulting *overall exposure scores* for all ESUs (including WC Chinook Salmon) were "high."

Our estimated *overall vulnerability rank* for the WC Chinook Salmon ESU was "moderate." This is identical to other Chinook and sockeye salmon ESUs we used for comparison, but lower than the two coho salmon ESUs, which had higher sensitivity scores due to extended freshwater rearing. The "moderate" vulnerability rating places it with many steelhead DPSes (e.g., the California Central Coast and the Southern California Coast DPSes), but below all other steelhead, coho, and Chinook salmon ESUs with the exception of the LCR Chinook Salmon ESU (Crozier et al. 2019). This rating seems appropriate given the general trends in vulnerability of listed salmonid ESUs/DPSes based on their life-history traits and geographic locations (Crozier et al. 2019).

A major difficulty in evaluating this ESU is the large heterogeneity in topography, land use, and hatchery production, and how to weigh this variation across the entire ESU. For example, the new CSP Washington Coast Climate Resilience Index[24] (Adams and Zimmerman 2024) indicates that most basins on the northern Washington coast (north of the Chehalis River basin) have

[24] https://coast-salmon-partnership.shinyapps.io/CRI_app/

high climate resilience, while those in the Chehalis River and Willapa Bay basins are much lower. At smaller spatial scales, Beechie et al. (2021) showed that, although the Chehalis River drainage had lost or degraded beaver pond, side-channel, and floodplain habits, it varied greatly by sub-basin. These differences, in turn, affected the effectiveness of restoration actions and the salmon species that most benefited from restoration actions (Fogel et al. 2022).

Recent trends in marine and terrestrial environments

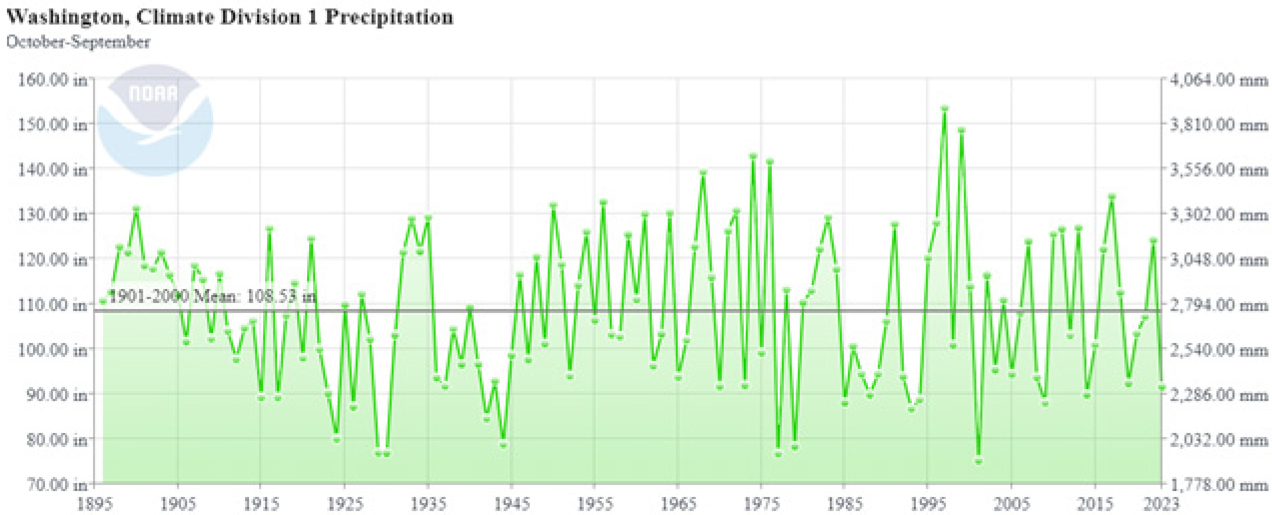
The productivity of Pacific salmon populations is influenced by numerous factors, including human activities (e.g., fishing mortality, habitat restoration and degradation, hatchery production) and environmental conditions in both freshwater and marine habitats. Because of this, changes in productivity for most populations partially reflect variation in environmental conditions.

Due to large-scale environmental variation captured by metrics such as the El Niño/Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO), population productivity undergoes periods of high and low productivity. 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 during the late 2000s and early 2010s. However, changes in ocean and freshwater conditions beginning in early 2014—due to exceptionally warm marine conditions and related terrestrial impacts, and to strong El Niño events in 2015–16 and 2023–24—led to subsequent declines in abundance in many populations. Starting with the original "blob" of 2014–16, marine heatwaves have occurred across large parts of the North Pacific every year since 2019, in addition to a record-smashing terrestrial heatwave in June 2021 that likely impacted many Pacific salmon populations. Marine and terrestrial conditions over the past 20–25 years are summarized below to provide environmental context when examining abundance and productivity trends for WC Chinook salmon.

Terrestrial conditions

Annual average temperatures and precipitation (October–September) provide a broad-brush view of terrestrial conditions across the Pacific Northwest. A strong and persistent warming trend (0.1°F/decade) and large year-to-year variations in precipitation are among the most notable features in recent decades (Figure 34). For the Pacific Northwest as a whole, 2015 stands out as the warmest year on record (Figure 34), although the year 2023 was the warmest on record globally. The combination of below-average precipitation and record high surface air temperature in 2015 brought record low springtime snowpack to much of the West, leading to low and unusually warm runoff to western watersheds in spring and early summer 2015. Also notable were record-breaking terrestrial temperatures during the 25 June–2 July 2021 "heat dome" recorded across western North America (White et al. 2023). This event resulted in some of the highest temperatures ever recorded across large parts of British Columbia, Washington, and Oregon (11–19°C [20–35°F] above normal temperatures).

Because juvenile Chinook salmon from the WC migrate after a relatively short freshwater residence (< 3 months post-hatch), juvenile Chinook salmon from the ESU likely were less impacted by these extreme terrestrial events (because they had already left freshwater)



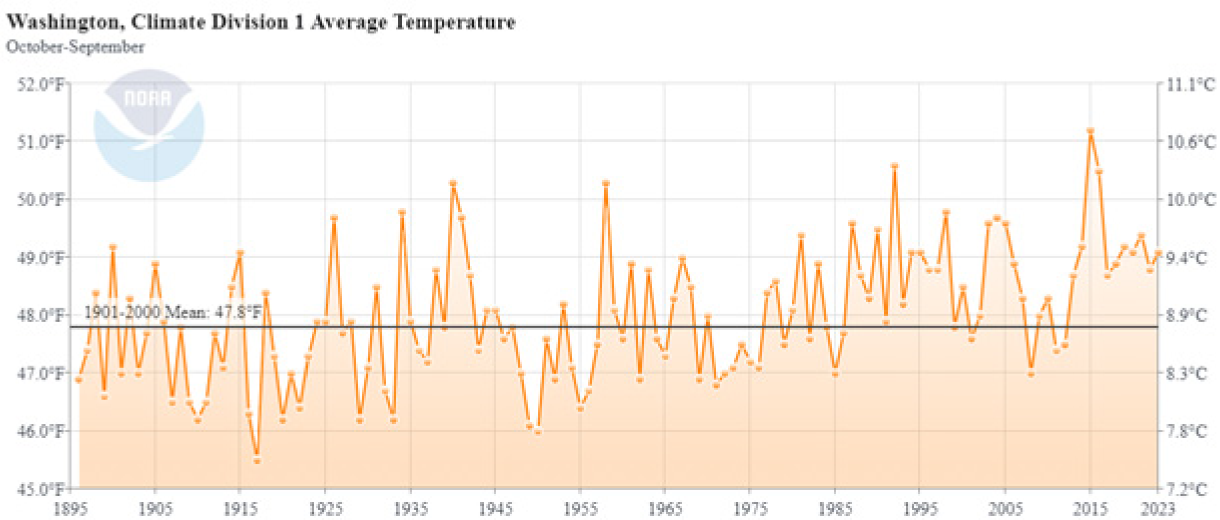


Figure 34. Water year (Oct–Sep) surface air temperature (top) and precipitation (bottom) for the Washington coast region. The historical average for 1901–2000 is shown with a black horizontal line. Data from the National Centers for Environmental Information's National Time Series, https://www.ncei.noaa.gov/access/monitoring/climate-at-a-glance/national/time-series.

than salmonids with longer freshwater residencies (e.g., coho and sockeye salmon and steelhead). By contrast, early-returning adults may have been impacted by elevated river temperatures if they were unable to find coldwater refugia.

Marine conditions

Surface temperatures in the northeastern Pacific Ocean vary on decadal time scales, with periods above or below average temperatures. For example, surface temperatures along the Washington and Oregon coasts were notably cooler than average from 1999–2002 and 2007–14, and warmer than average during 2003–05 (Figure 35). For the entire California Current region, surface temperatures in 2015 were the single warmest year in the historical record due to the marine heatwave (Bond et al. 2015, Jacox et al. 2018). Recurrent marine heatwaves across the North Pacific have resulted in alternating periods of above- and below-average water temperatures in coastal regions along Washington and Oregon (Figure 35).

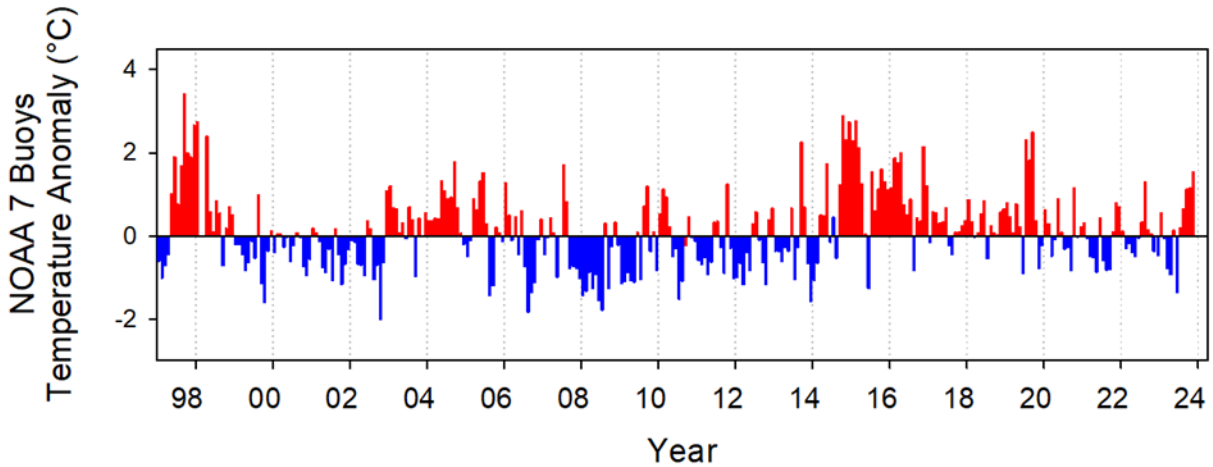


Figure 35. Time series of surface temperature anomalies along the WA and OR coasts, 1996–2023. Red bars indicate above-average temperatures, blue bars are below average. These anomalies are based on temperature records for seven NOAA buoys. Credit: C. Morgan, Oregon State University.

The biological impacts of these temperature swings and marine heatwaves are documented in a number of annual reports, for example Thompson et al. (2022) and Morgan et al. (2019), for areas of the northeastern Pacific Ocean that WC Chinook salmon occupy during their marine residence period. 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 2023. These ecosystem changes have had large effects (both positive and negative) on Pacific salmon returns around the Pacific Rim, including WC Chinook salmon.

NWFSC's "stoplight chart" of ocean indicators (Figure 36) is used to predict Columbia River salmon marine survival, although the results likely apply equally to WC Chinook salmon. The chart uses a suite of physical and biological indicators that together indicate whether coastal ocean conditions are good, fair, or poor for juvenile salmon during their first summer in marine waters, when most marine mortality is thought to occur. The stoplight chart shows periods when most indicators suggest poor survival (e.g., 2003–05, 2014–20), but also periods of good survival (e.g., 2008, 2021), consistent with observed salmon returns 1–2 years later. The long period of detrimental ocean conditions associated with marine heatwaves since 2014 indicates that recent marine heatwaves, coupled with strong El Niño events, had largely negative impacts to WC Chinook salmon via changes in marine conditions (elevated temperatures, late spring transition to upwelling, poor prey quality and quantity). The current forecast for a La Niña event in winter of 2024–25 should result in above-average snowpack (and therefore high spring and summer stream flows) and generally cooler water along the coast; both are favorable for Pacific salmon at short time scales.

Hatcheries

The effects of hatchery fish on the status of an ESU depend upon which of the four key attributes—abundance, productivity, spatial structure, and diversity—are currently limiting the ESU, and how the hatchery fish within the ESU affect each of the attributes (USOFR 2005). In general, hatchery programs can provide short-term demographic benefits

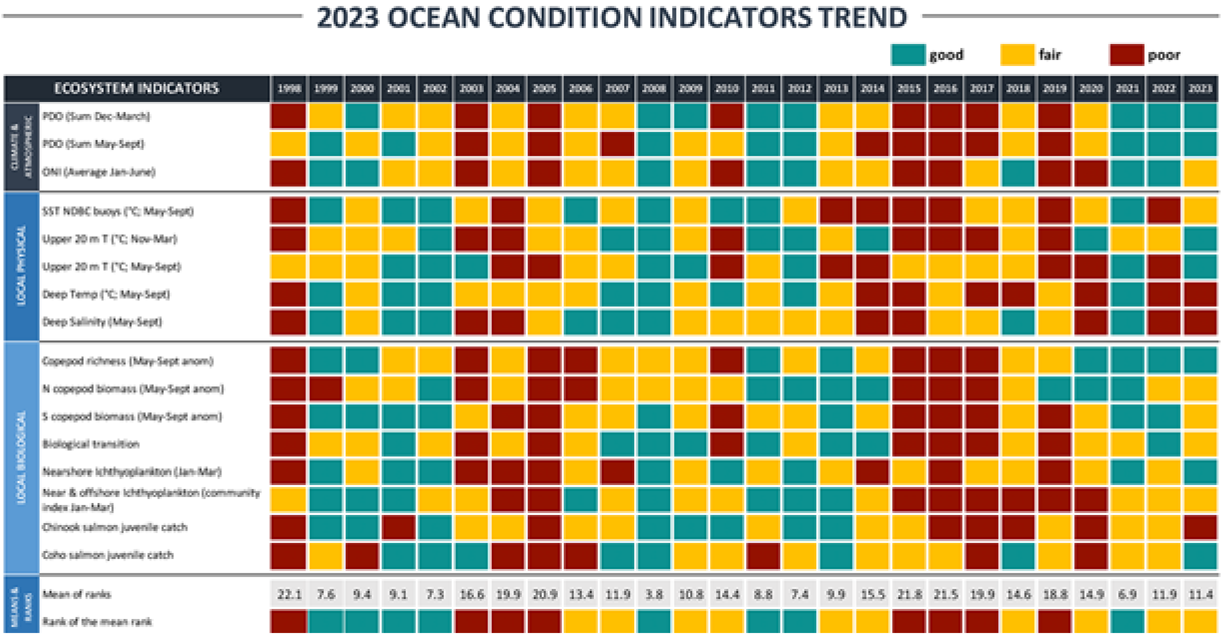


Figure 36. Stoplight chart of ocean indicators. Available: www.fisheries.noaa.gov/west-coast/science-data/ocean-conditions-indicators-trends (May 2025).

to salmon and steelhead, such as increases in abundance during periods of low natural abundance. They also can help preserve the genetic make-up of some populations until limiting factors can be reduced. However, the long-term use of artificial propagation may pose serious risks to the natural productivity and diversity of wild stocks. The magnitude and type of risk depends on the status of affected populations, the stock(s) utilized in the hatchery, and on specific practices in the hatchery program.

Within Washington State, there are two types of hatchery programs: integrated and segregated (Harbison et al. 2022). Segregated programs use eggs only from returning hatchery salmon, while integrated include natural-origin broodstock (Harbison et al. 2022). In general, segregated hatchery programs are harvest-oriented and not intended to interact (spawn) with natural-origin populations in the hatchery or on the spawning grounds. Often the broodstocks used in segregated programs were not derived from local (within-ESU) populations, or are highly domesticated, or both. Integrated programs can be conservation-oriented, harvest-oriented, or both. Integrated hatchery programs are designed to maintain a close genetic relationship with the naturally spawning population. To reduce risks from hatchery programs, WDFW's Anadromous Salmon and Steelhead Hatchery Policy and the Hatchery Scientific Review Group (HSRG, an independent scientific panel that reviews Pacific Northwest hatcheries) have thresholds of allowable levels of proportion of hatchery-origin spawners (pHOS) for segregated programs (the proportion of hatchery-origin fish spawning naturally), as well as proportion of natural influence (PNI) for integrated programs (the proportion of natural-origin fish utilized in the hatchery broodstock).

Extensive hatchery programs have been implemented throughout the range of WC Chinook salmon. While these programs may have succeeded in providing harvest opportunities and increasing the total number of naturally spawning fish, the programs also increased risks to natural populations. Hatchery programs and hatchery-produced Chinook salmon can affect naturally spawning populations of salmon, including through competition (for spawning sites and food) and increased predation, disease transmission, genetic impacts (e.g., outbreeding depression, high stray rates into wild populations, hatchery-influenced selection, broodstock removals from wild stocks, and collection effects [e.g., inadvertent selection for run timing or size, or limited numbers of broodstock]), and facility effects (e.g., water withdrawals, effluent discharge), as well as masking of trends in natural populations through straying of hatchery fish. State agencies managing aquatic natural resources have adopted or are in the process of adopting policies designed to ensure that the use of artificial propagation is conducted in a manner consistent with the conservation and recovery of natural, indigenous populations.

Status Review Team Risk Assessment

Assessment Methods

The WC Chinook Salmon SRT followed the same risk assessment methods as for prior SRTs (OC and SONCC Status Review Team 2023, OP Steelhead Status Review Team 2024). Our determination of overall risk used the categories of "high risk" of extinction, "moderate risk" of extinction, or "low risk" of extinction. The high and moderate risk levels were defined in a prior review of OC coho salmon (Stout et al. 2012), and have also been used, with minor wording changes, for recent status updates of all listed salmon and steelhead ESUs/DPSes (Ford 2022). They are defined as follows:

- **High Risk:** A species or ESU/DPS with a high risk of extinction is at or near a level of abundance, productivity, diversity, and/or spatial structure that places its continued existence in question. The demographics of a species, ESU, or DPS at such a high level of risk may be highly uncertain and strongly influenced by stochastic and/or depensatory processes. Similarly, a species, ESU, or 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; disease epidemic) that are likely to create such imminent demographic risks.
- **Moderate risk:** A species or ESU/DPS is at moderate risk of extinction if it exhibits a trajectory indicating that it is more likely than not to reach a high level of extinction risk in the foreseeable future. A species, ESU, or DPS may be at moderate risk of extinction due to projected threats and/or declining trends in abundance, productivity, spatial structure, or diversity. The appropriate time horizon for evaluating whether a species, ESU, or DPS is more likely than not to become at high risk in the future depends on various case- and species-specific factors. For example, the time horizon may reflect certain life-history characteristics (e.g., long generation time or late age-at-maturity) and may also reflect the timeframe or rate over which identified threats are likely to impact the biological status of the species, ESU, or DPS (e.g., the rate of disease spread). The appropriate time horizon is not limited to the period that status can be quantitatively modeled or predicted within predetermined limits of statistical confidence.
- **Low risk:** The species is neither at high nor moderate risk of extinction.

The overall extinction risk determination reflects the informed professional judgment of each SRT member. This assessment was guided by the results of the risk matrix analysis (see below), integrating information about demographic risks with expectations about likely interactions with threats and other factors. Following Stout et al. (2012), the WC Chinook Salmon SRT considered the "foreseeable future," as it relates to the moderate risk assessment, to be a time period of 30–80 years. Beyond the 30–80-year time horizon, the projected effects on WC Chinook salmon viability from climate change, ocean conditions, and trends in freshwater habitat become very difficult to predict with any certainty.

Risk matrix approach

In previous NMFS status reviews, SRTs have used a "risk matrix" as a method to organize and summarize the professional judgment of a panel of knowledgeable scientists. This approach has been used for over 20 years in Pacific salmonid status reviews (Myers et al. 1998, Good et al. 2005, Hard et al. 2007), as well as in reviews of other marine species (e.g., Stout et al. 2001). In this risk matrix approach, the condition of individual populations within each ESU is summarized according to four demographic risk criteria: abundance, growth rate/productivity, spatial structure/connectivity, and diversity. These viability criteria, outlined in McElhany et al. (2000), reflect concepts that are well founded in conservation biology and are generally applicable to a wide variety of species. These criteria describe demographic risks that individually and collectively provide strong indicators of extinction risk.

In addition to these four demographic criteria, the SRT also considered the impacts of the environmental threats associated with the ESA Section 4(a)(1) Risk Factors. These include habitat loss and degradation, overutilization for commercial or scientific purposes, inadequate regulatory mechanisms, disease and predation, and risks associated with hatchery operations and climate change. The summary of demographic risks and environmental risks obtained by this approach was then considered by the SRT in determining the species' overall level of extinction risk. The application of a standardized risk assessment for this ESU provides consistency and allows for direct comparisons with past and future status review conclusions.

Each of the demographic and environmental risk criteria for each population were evaluated by each team member against the following rubric:

- **Very low risk:** It is unlikely that this factor contributes significantly to risk of extinction, either by itself or in combination with other factors.
- **Low risk:** It is unlikely that this factor contributes significantly to risk of extinction by itself, but there is some concern that it may in combination with other factors.
- **Moderate risk:** This factor contributes significantly to long-term risk of extinction, but does not in itself constitute a danger of extinction in the near future.
- **High risk:** This factor contributes significantly to long-term risk of extinction and is likely to contribute to short-term risk of extinction in the foreseeable future.
- **Very high risk:** This factor by itself indicates danger of extinction in the near future.

Although this process helps to integrate and summarize a large amount of diverse information, there is no simple way to translate the risk matrix scores directly into a determination of overall extinction risk.

After population-level risks were assessed, each team member assessed the risk (low, moderate, or high) of the ESU as a whole. To allow individuals to express uncertainty in determining the overall level of extinction risk facing the species, the team adopted the "likelihood point" method, often referred to as the "FEMAT" method because it is a variation of a method used by scientific teams evaluating options under the Northwest Forest Plan (FEMAT 1993). In this approach, each SRT member distributes ten likelihood points among the three species extinction risk categories, reflecting their opinion of how likely that category is to correctly reflect the true species status. Thus, if a member is certain that a species is in the “low risk” category, that member could assign all ten points to that category. A reviewer with less certainty about the species’ status could split the points among two, or all three, categories. This method has been used in most status reviews for anadromous Pacific salmonids since 1999, excluding five-year status updates for already-listed ESUs/DPSes.

WC Chinook Salmon ESU: Population Assessment

The WC Chinook Salmon SRT carefully considered and discussed the information on demographics and threats summarized in this report. After this consideration, each team member provided their own assessment of risk, first at the individual population scale, and then at the entire-ESU scale (see Appendix C and Appendix D for scores).

Overall, the average VSP category scores indicated that nearly all of the populations were considered by the SRT to be at low risk (Table 15). Exceptions included the Wynoochee River spring-run population, which is believed to be extirpated; the Sol Duc River spring-run population, which is considered to have been established by non-native hatchery introductions; the Tsoo-Yess River fall-run population, which may not represent a historical population and largely consists of hatchery-origin fish; and the Hoko River fall-run population, which has a large hatchery-origin spawner component (Appendix C).

When assessed on a regional basis, populations in the Willapa Bay region received slightly elevated risk scores for productivity and diversity (Table 15). This reflects the relatively large fraction of hatchery fish spawning naturally in Willapa Bay rivers, a consequence of the production hatchery programs in those basins, and the higher proportion of land converted to residential, agricultural, and industrial timber uses. Similarly, in assessing the potential threats (Table 16), populations in Grays Harbor and Willapa Bay were given slightly higher risk scores for habitat (2.28/5 and 2.80/5, respectively), reflecting the higher degree of land development in those regions. The potential effects of the hatchery programs in Willapa Bay (WRIA 24) also scored higher as a threat (3.10/5) than in the rest of the ESU. Finally, the effects of climate change were viewed as a moderate threat to the entire ESU (2.10/5), especially in Grays Harbor (2.45/5), where the Chehalis River was presumed to be especially susceptible to the effects of climate change, increasing summer temperatures, and low summer flows.

WC Chinook Salmon ESU: ESU Assessment

Considering demographic status and trends, threats, and the population-level assessment, the team unanimously concluded that **the WC Chinook Salmon ESU is most likely to be at low risk of extinction in the foreseeable future**, with individual team members placing between six and ten (out of ten) likelihood points in the low-risk category. Across all members, the average assessments were: for low risk, 8.4 (1.4 SD); for moderate risk, 1.6 (1.3 SD); and for high risk, 0.0 (0.0 SD).

Table 15. WC Chinook Salmon SRT viability scores for WC Chinook salmon populations, grouped by ESU and coastal region. Key: *1* = very low risk; *5* = very high risk; *SD* = standard deviation. *ESU* refers to the average across all populations in the ESU; the remaining regions consist of smaller subsets of populations.

	Abundance		**Productivity**		**Spatial Structure**		**Diversity**		**Overall**
Region	*Avg*	*SD*	*Avg*	*SD*	*Avg*	*SD*	*Avg*	*SD*	*Avg*
ESU		0.60		0.60		0.50		0.60	
North Coast		0.52		0.57		0.42		0.58	
Queets–Quinault R		0.52		0.57		0.38		0.42	
Grays Harbor		0.69		0.70		0.66		0.80	
Willapa Bay		0.50		1.00		0.70		0.70	
Early-returning		0.43		0.53		0.46		0.58	
Late-returning		0.77		0.74		0.53		0.66	

Table 16. WC Chinook Salmon SRT scores for threats (factors for decline) for WC Chinook salmon populations, grouped by ESU and coastal region. Key: *1* = very low risk; *5* = very high risk; *SD* = standard deviation. *ESU* refers to the average across all populations in the ESU; the remaining regions consist of smaller subsets of populations.

	Habitat		**Overutilization**		**Inadequate regulation**		**Disease/predation**		**Hatchery**		**Climate**	
Region	*Avg*	*SD*	*Avg*	*SD*	*Avg*	*SD*	*Avg*	*SD*	*Avg*	*SD*	*Avg*	*SD*
ESU	1.90	0.60	1.70	0.70	1.40	0.50	1.40	0.50	1.60	0.40	2.10	0.60
North Coast	1.79	0.66	1.65	0.64	1.30	0.48	1.32	0.42	1.64	0.40	1.98	0.53
Queets–Quinault R	1.67	0.65	1.67	0.70	1.40	0.50	1.43	0.50	1.30	0.15	2.05	0.65
Grays Harbor	2.28	0.61	1.75	0.71	1.44	0.50	1.60	0.48	1.39	0.45	2.45	0.55
Willapa Bay	2.80	0.70	1.80	0.80	1.50	0.70	1.60	0.50	3.10	0.60	2.40	0.50
Early-returning	1.85	0.68	1.66	0.69	1.40	0.53	1.40	0.45	1.39	0.32	2.28	0.64
Late-returning	2.01	0.62	1.71	0.68	1.35	0.48	1.46	0.46	1.66	0.41	2.06	0.50

The primary factors leading team members to the conclusion of low extinction risk include: 1) a relatively large overall annual natural-origin escapement of > 30,000, 2) stable abundance trends in diverse geographic groupings, 3) productive populations able to sustain spawner abundances under relatively high harvest rates, and 4) moderate hatchery production using locally derived broodstock operated using improved hatchery practices since 1998. Most of the populations appear to be naturally sustaining, and are well distributed throughout the range of the ESU. Despite considerable changes to the landscape due to forestry, agriculture, and urbanization, habitat conditions are relatively good, with many populations north of Grays Harbor having their headwaters protected in Olympic National Park and in other federal and state forest lands. Habitat degradation is more common in the southern portion of the ESU (Grays Harbor and Willapa Bay), where residential and agricultural land development is more extensive. Regulatory protection of streams and riparian habitat has improved since the 1998 review, but the legacy effects of past timber harvest techniques and continued timber harvest on industrial timber lands (more extensive in the southern portion of the ESU) may still be depressing productivity.

Although generally confident that the ESU was at low risk, some team members allocated some risk points to the moderate-risk category. The rationale for these moderate-risk scores included the potential for population declines in the next half century due to climate change (including loss of snowmelt-driven flows, rising water temperatures, and an altered, less productive marine climate) and other difficult-to-predict, large-scale environmental changes. The SRT also recognized that these climate change impacts would pose a greater risk to early-returning spring- and summer-run populations (Beechie et al. 2023). Further, the legacy effects of past timber harvest techniques on streams may continue to limit stream productivity for decades to come. Additionally, the SRT identified the decreasing age, and thus smaller size, of returning adults and the potential for this to negatively affect productivity through decreased fecundity, as smaller adults are less able to construct deeper redds or redds in larger substrate, resulting in greater susceptibility to scour (Oke et al. 2020).

Most members concluded that the status of the WC Chinook Salmon ESU, in terms of overall extinction risk, remains unchanged since the 1998 review. Population declines in some areas were balanced by improvements in others. Similarly, while changes in land-use regulations and recovery actions may have stabilized or improved stream and riparian habitat in some watersheds, continued land development degraded other watersheds. Overall, however, the team was unanimous in its conclusion that this ESU is at a low risk of extinction.

WC Chinook Salmon ESU: SPR Assessment

The team approached the SPR question by first looking across the individual populations for areas with higher-than-average risk scores. Note that *higher* does not necessarily imply these are at moderate or high risk; only that they are at a higher risk level than the ESU as a whole. The team identified four such areas:

1. The northern coast and Strait of Juan de Fuca. These populations are characterized by small population sizes, spawning in small watersheds, and substantial hatchery influence—all factors that indicate that these populations are at higher risk than

the ESU as a whole. However, the Hoko and Tsoo-Yess River basins have smaller watershed areas than any other of the independent populations identified by recovery teams in other Chinook salmon ESUs. These small watersheds likely depend on a combination of hatchery production and strays from other populations to sustain their long-term abundance, and so are not independent populations.

2. Southern coastal areas of the ESU, including Willapa Bay. These areas are characterized by lower-gradient streams that are likely more susceptible to warming temperatures predicted by future climate change. They are largely in private land ownership, with greater potential for development and habitat degradation than areas protected in ONP or other public lands. Populations in the Willapa Bay area either have a high proportion of hatchery fish on the spawning grounds (> 50% in recent returns), or have an unknown hatchery contribution. Such conditions lead to greater risk scores than in other portions of the ESU. However, despite these threats, the overall fall-run Chinook salmon population abundance in this area has been relatively stable. The contribution of natural-origin spawners to the ESU from small tributaries in the Willapa Bay drainage is also quite small.
3. The upper Chehalis River basin. This area includes both spring- and fall-run populations of Chinook salmon. In contrast to other basins in the ESU, the upper Chehalis River drains the lower-elevation Willapa Hills, rather than the Olympic Mountains. As such, this basin is more vulnerable to climate change, especially the spring-run population. Further, the flood control dam proposed for the upper Chehalis River will likely have a negative impact on spring-run spawning habitat, although the effects may not be large (see ESA Section 4(a)(1) Risk Factors). The SRT also noted that there are multiple spring-run populations in the ESU, some nearly as abundant as the upper Chehalis River population.
4. Early-returning (spring- and summer-run) populations throughout the ESU were also considered to be at higher risk than the entire ESU. The abundance of early runs in each river, and collectively among all the rivers, is considerably lower than that of fall runs. Only the Hoh and Chehalis Rivers typically have more than 1,000 early-run spawners. The SRT considered that an early-returning life history exposes returning adults to increased summer temperatures and decreased summer flows during their extended over-summer holding in freshwater, especially with climatic changes observed over the last few decades and those predicted for the future. Alternatively, the early-run habitat portion is distributed among numerous watersheds throughout the ESU, many of which are both in protected federal lands and (due to higher elevations and forest cover) expected to be less vulnerable from increasing temperatures due to climate change.

A review of the risk scores for each of these portions indicated that:

1. In the northern coast and Strait of Juan de Fuca—while populations were at a somewhat higher risk of extinction that ESU overall—the average VSP risk score (1.4) was actually lower than the ESU average (1.5), and the Hoko and Tsoo-Yess River basins are geographically too small to be considered independent populations.
2. Populations in Willapa Bay exhibit a higher average VSP risk score (1.8) relative to the entire ESU (1.5). However, the overall fall-run Chinook salmon population abundance in this area has been relatively stable. The contribution of natural-origin

spawners to the overall ESU abundance from small tributaries in the Willapa Bay drainage is also relatively minor, suggesting that these populations may not be sufficiently important to the ESU's long-term viability to be considered an SPR.

3. Portions of the early-run habitat, such as the upper Chehalis River, are predicted to be especially prone to future high water temperatures and lower flows coastwide (Crozier et al. 2019, NWIFC 2020). The SRT's consensus was that Chinook salmon in the upper Chehalis River, especially the spring-run population, are at somewhat higher risk from habitat degradation and climate change, but given the abundance, productivity, and genetic information currently available, this portion does not constitute an SPR.
4. Early-run spawners are found in at least nine WC rivers (Figure 2). Recent (15-year) trends in early-run abundance are mostly stable-to-positive, and overall early-run abundance is similar to what it was before the initial ESA status review in the late 1990s. Collectively, early-run spawners make up about 15% of the annual spawning abundance of the ESU (Table 11), and the habitat preferred by early runs also makes up 15.2% of the watershed area of the ESU.[25] The overall VSP risk score for early-returning populations was 1.6, indicating "very low to low risk" (fall-run average = 1.3; Table 15)—only marginally higher than the average risk of 1.5 for the entire ESU. The team also found that the risks from threats to early runs were very similar to the ESU as a whole (Table 16).

[25] For the entire ESU, spring-run populations utilize 9.99%, summer-run populations 5.17%, and fall-run populations utilize 31.30% of the watersheds for rearing and spawning.

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Appendix A: Hatchery Profiles

Hatcheries in the Washington Coast Chinook Salmon ESU are operated by federal, tribal, and state resource agencies. Hatcheries are listed by river basin, from north to south. Production levels reflect those at the time of the SRT review.

Hoko Falls Hatchery

This hatchery is located on the Hoko River (RKm 16/RM 10). It is operated by the Makah Tribal Council. The water sources are Rights Creek (WRIA 19.0174) and Brownes Creek (WRIA 19.0170). The hatchery was constructed in 1982 and began operations in 1984. Prior to the operation of this hatchery, there were several releases into the Hoko River system, most notably from the Elwha River and Hood Canal. This stock is the only one in the Western Strait Chinook Salmon genetic diversity unit (GDU; Busack and Marshall 1995). Adults are collected with gill-nets in the river. They are spawned, incubated, and reared at Hoko River Hatchery. Genetic analysis suggests that this population is somewhat distinct from others in the ESU (Brown et al. 2017), which may be related to past introductions or being located near the boundary between the Washington Coast and Puget Sound Chinook Salmon ESUs. Current production levels for this integrated program are 500,000 (egg take). Juveniles are released in the mainstem Hoko River and Little Hoko River (WDFW 2024).

Makah National Fish Hatchery

The Makah National Fish Hatchery (NFH) is located on the Makah Indian Reservation, at the extreme northwestern tip of the Olympic Peninsula. The hatchery is situated on the Tsoo-Yess River, about three miles upstream from its confluence with the Pacific Ocean. The lower four miles of the Tsoo-Yess River watershed are on the Reservation, while the upper reaches are located on private timber lands. Deschutes River and Soos Creek (Green River, Puget Sound) stocks were released for one or two years in the 1960s; Minter Creek stock was released for one year in the 1970s; Quinault River stock was released for one year in the 1980s (there is some uncertainty about these releases). Broodstock is collected from rack returns to the Makah NFH. Adult collection, spawning, incubation, and rearing occur on-station. 100,000 fish are transferred to Educket Creek Hatchery for release, with 2.2 million planned for release in 2024 (WDFW 2024).

Bear Springs Ponds

Bear Springs is located on RKm 58 (RM 36) on the Sol Duc River, north of the Sol Duc Hatchery. It is leased from the Washington Department of Natural Resources and operated by WDFW. The property has one residence and a two-bay shop with office space. This hatchery rears summer-run Chinook salmon from the Sol Duc Hatchery, with eventual transfer back to Sol Duc Hatchery for release.

Sol Duc Hatchery

Sol Duc Hatchery is located on the Sol Duc River at RKm 48 (RM 30), north of the town of Forks, off Highway 101. It is owned and operated by WDFW. Although the summer run propagated at this hatchery was developed from native fish, there is some concern that there was some interbreeding at the hatchery with the non-native spring run that was reared beginning in 1971. Genetic studies have confirmed gene flow between the summer-run Chinook salmon population and the now-terminated spring-run Chinook salmon program (Spidle 2008, Brown et al. 2017). Chinook salmon are also incubated at Lonesome Creek Hatchery and Bear Springs Hatchery prior to release at Sol Duc Hatchery. Current production for this integrated program is 1.5 million (egg take), with 10% of the broodstock being of natural origin, and pHOS was estimated at 10% for BY 2016 (Anderson et al. 2020).

Salmon River Fish Culture Facility

The Salmon River Fish Culture Facility is owned and operated by the Quinault Indian Nation. The facility is sited at RKm 6 (RM 4)on the Salmon River (WRIA 21.0139), a lower main tributary to the Queets River (WRIA 21.0016). The Queets River is located near the middle of Washington's north coast. Since 1974, non-native stock transfers into the Salmon River hatchery stock include some fish from 1978–81 and 1985. For this integrated program, adults (60–65 pairs) are collected with gill-nets upstream of the Salmon River in the Queets and Clearwater River mainstems. For 2024, 200,000 juveniles are scheduled for release (WDFW 2024).

Lake Quinault Tribal Hatchery

Lake Quinault Tribal Hatchery is owned and operated by the Quinault Indian Nation. The facility is located on the southwestern shore of Lake Quinault (WRIA 21.0398) on the Olympic Peninsula. Lake Quinault is part of the mainstem Quinault River at RKm 53 (RM 33). Multiple stocks were used to begin the Quinault River fall Chinook salmon program, including Quinault, Queets and Hoh River stocks, as well as introductions from Puget Sound (Green, Deschutes, and Samish Rivers and Finch Creek) and Willapa Bay (Willapa and Nemah Rivers). This stock is managed as a segregated program, with planned releases of 1.3 million subyearlings (15 fish/pound) in 2024 (WDFW 2024).

Bingham Creek Hatchery

This hatchery was originally called Simpson State Salmon Hatchery. The facility sits on approximately five acres at RKm 28 (RM 17.5) on the East Fork Satsop River (a tributary of the Chehalis River, which flows into Grays Harbor). The hatchery was built in 1948, with a major upgrade conducted in 1995. There have been extensive releases of non-native hatchery fall Chinook salmon into the Satsop River basin since 1952 (including Humptulips River, Willapa Bay, Puget Sound, Columbia River, and Oregon coastal stocks). Genetic evidence from the East Fork Satsop River stock indicates a more native profile. Adults are collected from the East Fork Satsop River (within two miles of the Satsop Springs facility) by hook-and-line snagging or beach seining. Current production for the integrated program is 200,000, with 30% of the broodstock being natural-origin in 2016. Further, pHOS levels for the 2016 BY were relatively low, at 15% (Anderson et al. 2020).

Humptulips Hatchery

Humptulips Hatchery, owned by WDFW, is located on Stevens Creek at RKm 36 (RM 22.5) on the Humptulips River, approximately 1.5 miles from the town of Humptulips. The hatchery is located at the site of an old county hatchery that raised chum and possibly Chinook salmon. The current Humptulips Hatchery began operation in 1976 and achieved full production in the fall of 1977. Non-native hatchery stocks were introduced into the Humptulips River basin between the 1950s and 1980s, although genetic analysis does not indicate a high degree of hybridization between local and introduced populations. The program is run as an integrated program. 500,000 fingerlings at 70 fish/lb are released on-station (WDFW 2024). Spawning, incubation, and rearing occur on-station.

Lake Aberdeen Hatchery

The Lake Aberdeen Hatchery is located about 3 km east of the town of Aberdeen, next to the outlet of Lake Aberdeen. There have been three releases of small numbers of non-native hatchery fall Chinook salmon into the Wynoochee River basin. The Lake Aberdeen Hatchery stock was started using a combination of strays to the hatchery and wild broodstock from the Wynoochee River. Adult collection, spawning, incubation, rearing, and release occur on-station at Lake Aberdeen. This small program, with annual releases of 50,000 juveniles, incorporated natural-origin adults (18% of the broodstock) in BY 2016 (Anderson et al. 2020). Hatchery straying onto the natural spawning grounds is very low, with pHOS for BY 2016 estimated to be 0% (Anderson et al. 2020).

Satsop Springs Hatchery

Satsop Springs Hatchery is owned by WDFW and operated by the Chehalis Basin Fisheries Task Force. The facility is located on 60 acres at RKm 3.2 (RM 2) on the East Fork of the Satsop River (a tributary of the Chehalis River). Production is planned at 300,000 juveniles released on site for 2024, with an additional 200,000 transferred to Bingham Creek Hatchery for rearing and release (WDFW 2024).

Wishkah Hatchery

Wishkah Hatchery, formerly known as Mayr Brothers Ponds, is in Grays Harbor County, at RKm 40–41 (RM 25–25.5) on the mainstem Wishkah River. The property is owned by WDFW. Long Live the Kings (LLTK) operates and maintains the facility.[26]

Forks Creek Hatchery

Forks Creek Hatchery is owned, funded, and operated by WDFW. It is located at RKm 6.4 (RM 4) on Forks Creek, a tributary of the Willapa River, which flows into Willapa Bay on the southwestwern coast of Washington near Raymond. The hatchery site is at RKm 19.6 (RM 12.2) on State Highway 6. It was originally constructed in 1899 and rebuilt in 1953. Earliest records at the hatchery go back to the early 1930s, where, for the most part, Willapa River stocks were derived from the Willapa system. In the 1940s, several stocks were imported as a single stock or mixed to form a hybrid (notably, the Deschutes, Green, Satsop, Kalama, Nemah, White Salmon, Trask, and Elk Rivers, and Finch and Underwood/Spring Creeks). Transfers of hatchery stocks from out of the basin have been curtailed in the last few decades. The genetic legacy of non-native introductions has not been established, but could be considerable. Adult collection, spawning, incubation, and rearing occur at Forks Creek. For the Forks Creek, Naselle, and Nemah Hatchery programs, large numbers of hatchery-origin fall-run Chinook salmon are observed spawning naturally in the Willapa River basin; pHOS has been estimated at 81% (Anderson et al. 2020). This integrated program plans to release 400,000 subyearling juveniles in 2024. For the 2016 broodyear, 10% of the broodstock was natural-origin adults (Anderson et al. 2020).

Naselle Hatchery

This hatchery is owned by WDFW and funded by the general fund. The approximate size of the property is 5.7 hectares (14 acres). This is a mixed stock comprising native fish and introductions from the Forks Creek and Nemah Hatcheries. The majority of the naturally spawning Chinook salmon were believed to be non-native in origin (HSRG 2004). Transfers of hatchery stocks from out of the basin have been curtailed in the last few decades. The genetic legacy of non-native introductions has not been determined but could be considerable. For the program, 5 million fingerlings are planned for on-station release in 2024 at the Naselle Hatchery. Adult collection, spawning, and eyeing may occur at Naselle, Forks Creek, or Nemah Hatcheries. Hatching and rearing occur at Naselle. For the Forks Creek, Naselle, and Nemah Hatchery programs, large numbers of hatchery-origin fall-run Chinook salmon are observed spawning naturally in the Willapa River basin, with pHOS estimated at 75% for the 2016 broodyear (Anderson et al. 2020). Furthermore, for the 2016 broodyear, 20% of the broodstock were natural-origin adults for this integrated program (Anderson et al. 2020).

[26] https://lltk.org/

Nemah Hatchery

Nemah Hatchery is owned and operated by WDFW. It is located at RKm 5.6 (RM 3.5) on the North Nemah River, a tributary to Willapa Bay that flows into the Pacific Ocean between Leadbetter Point and Toke Point on the southwestern Washington coast near the town of Tokeland. The North Nemah River, including Williams Creek, drains a region of approximately 2,590 km^2 (1,000 mi^2), originating in the Willapa Hills. This is a mixed composite stock comprised of native fish and introductions from numerous sources, including Forks Creek and Naselle Hatcheries, Spring Creek, the Elochoman River (flows from the Willapa Hills into the Columbia River), Hood Canal, the Green River (Puget Sound), and the Trask and Elk Rivers (Oregon). Transfers of hatchery stocks from out of the basin have been curtailed in the last few decades. The genetic legacy of non-native introductions has not been determined but could be considerable. Adult collection and eyeing could also occur at Forks Creek Hatchery. For the Forks Creek, Naselle, and Nemah Hatchery programs, large numbers of hatchery-origin fall-run Chinook salmon are observed spawning naturally in the Willapa River basin. The current egg take goal is 3.7 million eggs and the program is run as a segregated program, with all juveniles being released as subyearlings.

Appendix B: Hatchery Releases

Table B1. Chinook salmon juveniles released from hatcheries in the WC Chinook Salmon ESU, 1899–1927. From Cobb (1930).

	Chehalis River basin	Willapa River basin	Quinault Lake	
Year	Fry	Fry	Fry	Fingerlings
1899	1,215,000			
1900	2,355,300	881,000		
1901	1,909,800	653,400		
1902				
1903		2,163,019		
1904	900,000	819,504		
1905		630,000		
1906		529,650		
1907		393,660		
1908	163,000	678,600		
1909	148,000	322,200		
1910	403,000	455,200		
1911	111,150	734,350		
1912	118,750	748,600		
1913	119,700	729,600		
1914	139,000	3,247,345		
1915	73,337	302,461	19,913	
1916	854,170	2,570,105	29,600	4,810
1917	495,350	2,178,185	160,000	
1918	2,978,288	5,411,725	220,000	
1919	279,200	1,460,206	100,000	109,400
1920	1,928,839	294,604		34,600
1921	4,376,450	6,023,500		24,800
1922	1,599,530	2,536,780		47,000
1923	826,420	5,072,605		
1924	313,519	3,784,325		
1925	172,279	6,338,790		
1926	458,700	8,989,450		13,300
1927	314,000	5,214,695		

Table B2. Chinook salmon egg take and spawning periods at Washington State hatcheries, 1927–45. From the Washington State Archives.

	Chehalis Hatchery			Humptulips Hatchery			Willapa Hatchery			Nasel/Naselle Hatchery		
Year	Egg take	Spawn start	Spawn end	Egg take	Spawn start	Spawn end	Egg take	Spawn start	Spawn end	Egg take	Spawn start	Spawn end
1927	329,000	24 Sep	4 Nov				2,280,000	1 Oct	25 Nov	4,717,000	24 Sep	1 Oct
1928	190,000	15 Sep	27 Oct				1,354,500	6 Oct	23 Nov	2,281,700	22 Sep	5 Oct
1929	4,000	26 Oct	26 Oct				738,000	28 Sep	20 Dec	6,520,800	21 Sep	25 Nov
1930	110,000	27 Sep	24 Oct				1,212,000	27 Sep	19 Dec	4,668,700	27 Sep	7 Nov
1931	223,200	24 Oct	27 Nov	1,846,000	24 Oct	27 Nov	2,503,000	26 Sep	2 Jan	8,975,500	19 Sep	21 Nov
1932	162,000	15 Oct	5 Nov	1,075,900	24 Sep	25 Nov	2,497,500	10 Sep	1 Jan	13,785,000	24 Sep	15 Oct
1933	232,000	23 Sep	4 Nov	1,140,000	23 Sep	25 Nov	1,514,000	23 Sep	16 Dec	10,575,000	23 Sep	21 Oct
1934	60,000	20 Oct	3 Nov	602,000	13 Oct	17 Nov	1,534,500	22 Sep	8 Dec	6,293,500	22 Sep	20 Oct
1935	138,000	21 Sep	23 Nov	210,000	9 Nov	30 Nov	1,467,000	5 Oct	4 Jan	5,394,400	28 Sep	23 Nov
1936	120,000	26 Sep	31 Oct				666,000	26 Sep	19 Dec	8,715,500	26 Sep	14 Nov
1937	260,000	11 Sep	23 Oct	125,000	9 Oct	13 Nov	1,533,000	2 Oct	4 Dec	1,139,000	2 Oct	8 Jan
1938												
1939				125,934	21 Oct	9 Dec	146,421	21 Oct	9 Dec	265,554	7 Oct	4 Nov
1940				123,753	12 Oct	30 Nov	490,303	12 Oct	9 Nov	554,410	28 Sep	26 Oct
1941				36,870	11 Oct	22 Nov	832,391	4 Oct	22 Nov	92,351	20 Sep	15 Nov
1942				209,192	7 Nov	28 Nov	227,759	7 Nov	21 Nov	645,747	3 Oct	31 Oct
1943				79,690	6 Nov	27 Nov	74,246	23 Oct	11 Dec	685,946	25 Sep	30 Oct
1944				108,120	7 Oct	25 Nov	29,747	4 Nov	25 Nov	352,213	23 Sep	4 Nov
1945							229,732	22 Oct	24 Nov	11,289	6 Oct	6 Oct

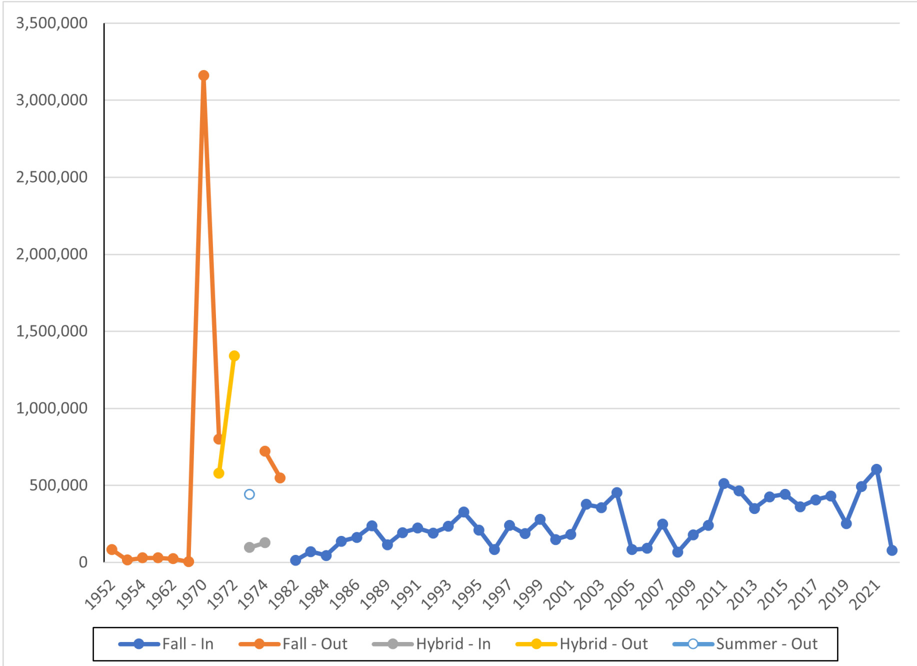

Figure B1. Chinook salmon releases into WRIA 19 watersheds, 1952–2022, by river, run timing, and ESU source. Key: *In* = from a river within the ESU, *Out* = imported from outside of the ESU, *Hybrid* = released juveniles were a cross of run timings (e.g., summer × fall run). Note: once incorporated into broodstock, progeny of out-of-ESU broodstock are considered as coming from within the ESU. Releases of < 2g fry not included. Data from the Regional Mark Information System (RMIS; January 2024), https://www.rmpc.org/data-selection/rmis-files/.

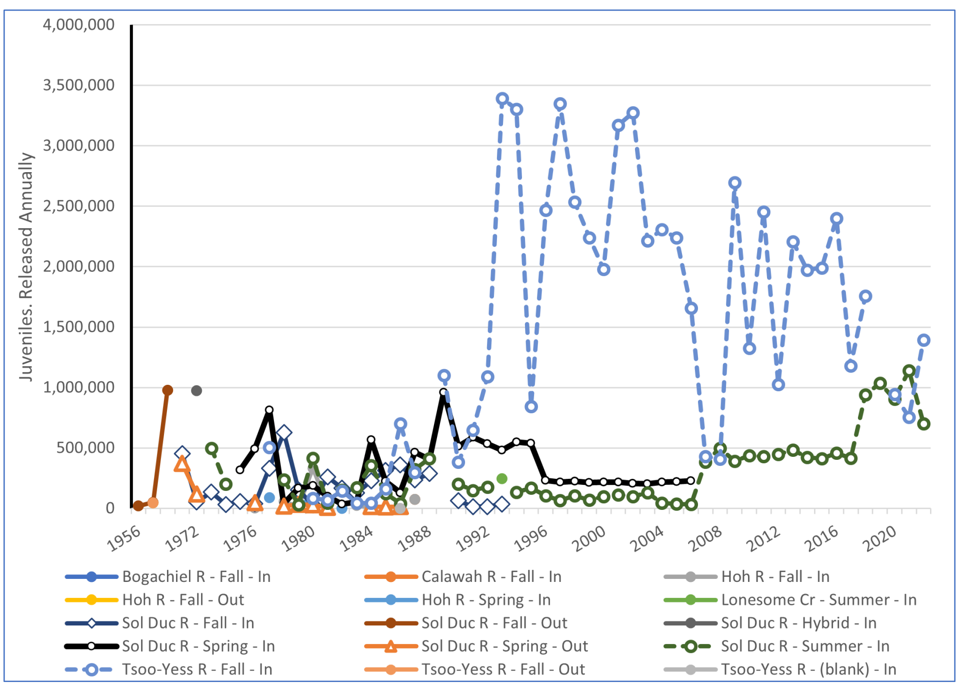

Figure B2. Chinook salmon releases into WRIA 20 watersheds, 1956–2022, by river, run timing, and ESU source. Key: *In* = from a river within the ESU, *Out* = imported from outside of the ESU, *Hybrid* = released juveniles were a cross of run timings (e.g., summer × fall run). Note: once incorporated into broodstock, progeny of out-of-ESU broodstock are considered as coming from within the ESU. Releases of < 2g fry not included. Data from RMIS (January 2024).

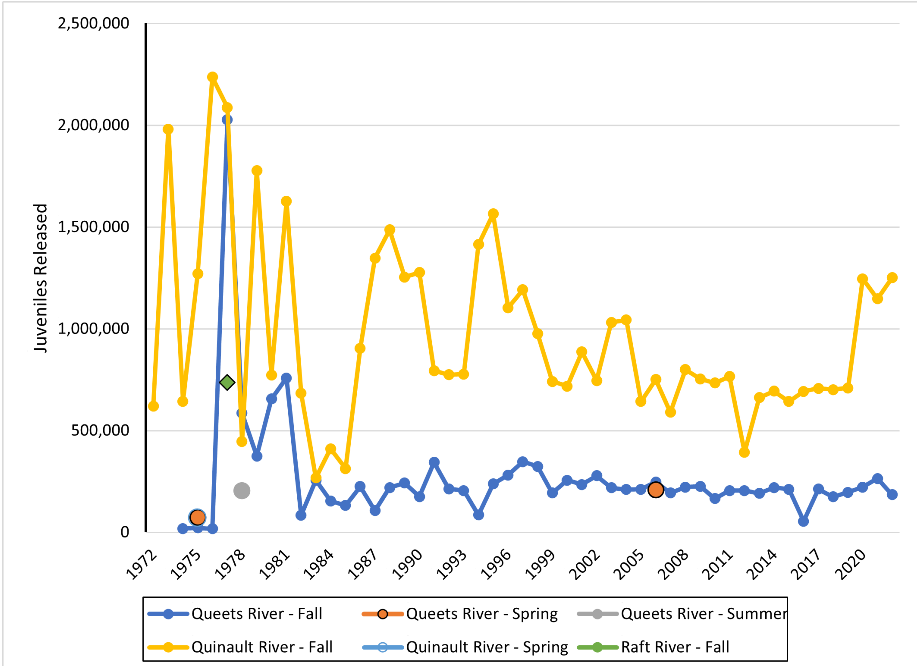

Figure B3. Chinook salmon releases into WRIA 21 watersheds, 1972–2022, by river, run timing, and ESU source. Key: *In* = from a river within the ESU, *Out* = imported from outside of the ESU, *Hybrid* = released juveniles were a cross of run timings (e.g., summer × fall run). Note: once incorporated into broodstock, progeny of out-of-ESU broodstock are considered as coming from within the ESU. Releases of < 2g fry not included. Data from RMIS (January 2024).

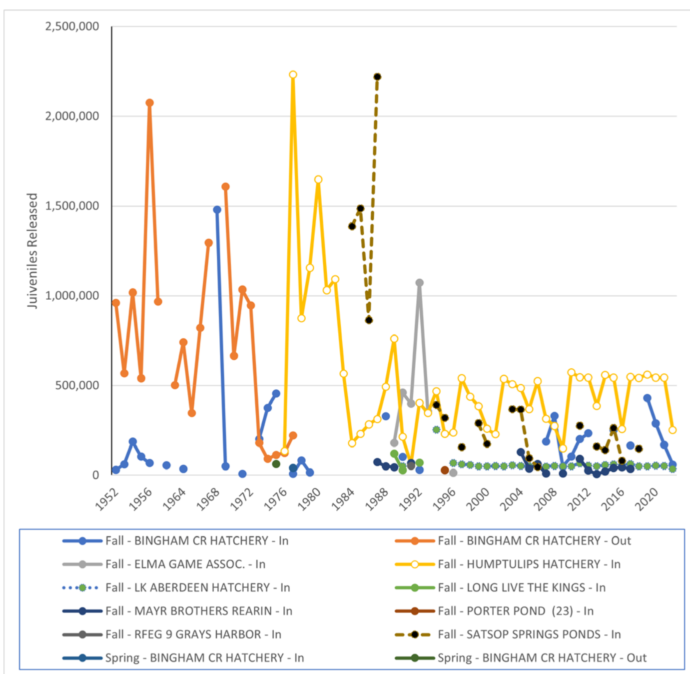

Figure B4. Chinook salmon releases into WRIA 22 and 23 watersheds, 1952–2022, by river, run timing, and ESU source. Key: *In* = from a river within the ESU, *Out* = imported from outside of the ESU, *Hybrid* = released juveniles were a cross of run timings (e.g., summer × fall run). Note: once incorporated into broodstock, progeny of out-of-ESU broodstock are considered as coming from within the ESU. Releases of < 2g fry not included. Data from RMIS (January 2024).

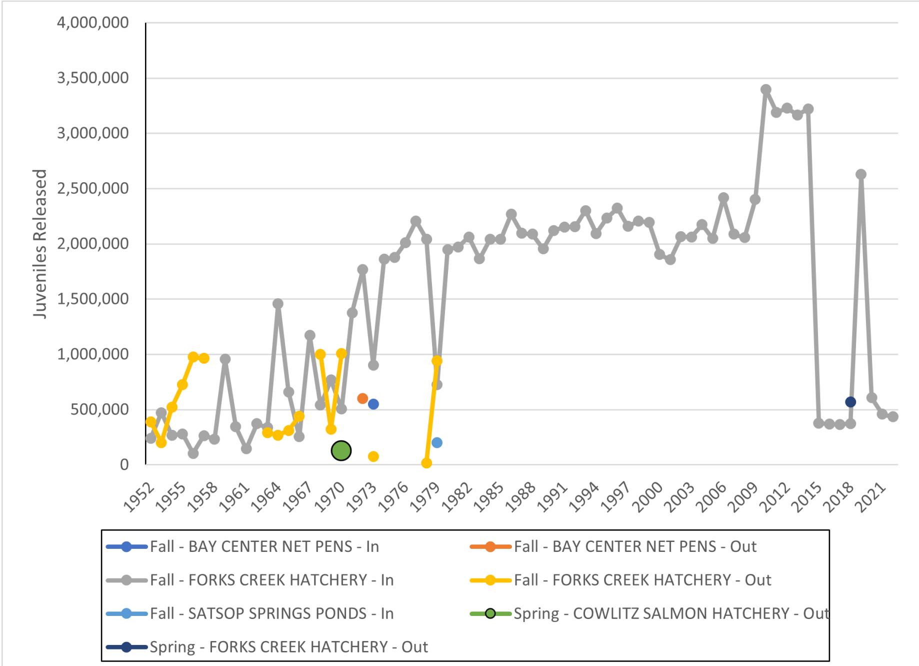

Figure B5. Chinook salmon releases into WRIA 24 (Willapa River) watersheds, 1952–2022, by river, run timing, and ESU source. Key: *In* = from a river within the ESU, *Out* = imported from outside of the ESU, *Hybrid* = released juveniles were a cross of run timings (e.g., summer × fall run). Note: once incorporated into broodstock, progeny of out-of-ESU broodstock are considered as coming from within the ESU. Releases of < 2g fry not included. Data from RMIS (January 2024).

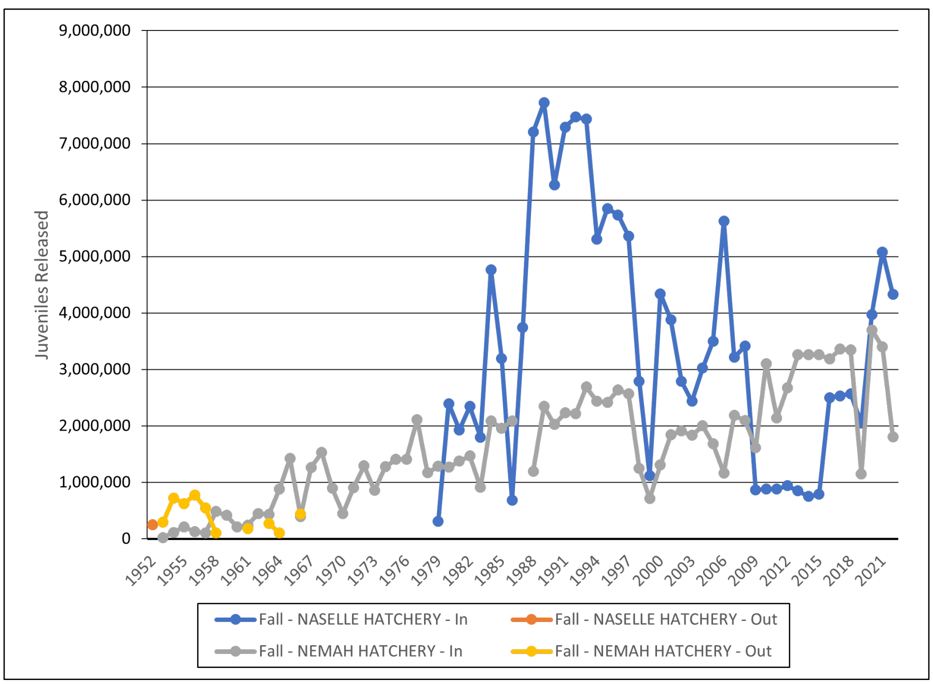

Figure B6. Chinook salmon releases into WRIA 24 (Naselle and Nemah Hatcheries) watersheds, 1952–2022 by river, run timing, and ESU source. Key: *In* = from a river within the ESU, *Out* = imported from outside of the ESU, *Hybrid* = released juveniles were a cross of run timings (e.g., summer × fall run). Note: once incorporated into broodstock, progeny of out-of-ESU broodstock are considered as coming from within the ESU. Releases of < 2g fry not included. Data from RMIS (January 2024).

Appendix C: SRT Scoring for VSP Criteria

Table C1. Status Review Team scoring for viable salmonid population (VSP) categories, averaged across team members. Key: *Su* = summer, *Spr* = spring, *SS* = spatial structure, *SD* = standard deviation.

			Viable salmonid population criteria							
			Abundance		Productivity		SS		Diversity	
Area	**Population**	**Run**	**Avg**	**SD**	**Avg**	**SD**	**Avg**	**SD**	**Avg**	**SD**
Strait	Hoko R	Fall	2.3	0.5	2.1	1.1	1.5	0.8	2.3	0.7
North Coast	Tsoo-Yess R	Fall	1.5	0.7	2.3	1.5	1.8	1.0	2.7	1.3
	Sol Duc R[a]	Spr	1.0		1.5	0.7	1.0	0.0	2.3	1.4
	Sol Duc R	Fall	1.1	0.3	1.2	0.4	1.1	0.3	1.1	0.3
	Sol Duc R	Su	1.4	0.5	1.4	0.5	1.1	0.3	1.6	0.5
	Bogachiel R	Su	1.9	0.6	1.4	0.5	1.2	0.4	1.2	0.4
	Bogachiel R	Fall	1.4	0.7	1.1	0.3	1.2	0.4	1.1	0.3
	Calawah R	Su	1.9	0.6	1.3	0.5	1.2	0.4	1.2	0.4
	Calawah R	Fall	1.1	0.3	1.1	0.3	1.2	0.4	1.1	0.3
	Dickey R	Fall	1.6	0.7	1.0	0.0	1.1	0.4	1.3	0.5
	Hoh R	Spr/Su	1.3	0.5	1.4	0.7	1.1	0.3	1.2	0.4
	Hoh R	Fall	1.1	0.3	1.1	0.3	1.1	0.3	1.2	0.4
Queets–Quinault	Queets R	Spr/Su	2.0	0.7	1.3	0.5	1.2	0.4	1.1	0.3
	Queets R	Fall	1.1	0.3	1.3	0.5	1.1	0.3	1.4	0.5
	Clearwater R	Spr/Su	2.3	0.7	1.4	0.7	1.2	0.4	1.2	0.4
	Clearwater R	Fall	1.2	0.4	1.3	0.5	1.1	0.3	1.2	0.4
	Quinault R	Spr/Su	2.3	0.7	1.5	0.8	1.3	0.5	1.1	0.4
	Quinault R	Fall	1.1	0.3	1.2	0.4	1.2	0.4	1.6	0.5
Grays Harbor	Chehalis R	Spring	1.7	0.7	1.3	0.5	1.4	0.7	1.6	0.8
	Chehalis R	Fall	1.1	0.3	1.1	0.3	1.2	0.4	1.3	0.7
	Wynoochee R[b]	Spr	3.5	1.9	3.3	1.7	3.0	2.0	2.4	1.5
	Wynoochee R	Fall	1.1	0.3	1.2	0.4	1.3	0.4	1.3	0.5
	Humptulips R	Fall	1.1	0.3	1.2	0.4	1.2	0.4	1.4	0.7
	Grays Harbor tributaries	Fall	1.8	0.7	1.4	0.7	1.4	0.5	1.5	0.7
	Satsop R	Su	2.6	1.0	2.0	1.2	1.3	0.5	1.4	0.8
	Satsop R	Fall	1.1	0.3	1.2	0.4	1.2	0.4	1.3	0.7
Willapa Bay	Willapa Bay tributaries	Fall	1.4	0.5	2.0	1.0	1.6	0.7	2.2	0.7
	WC Chinook Salmon ESU		**1.6**	**0.6**	**1.5**	**0.6**	**1.3**	**0.5**	**1.5**	**0.6**

[a] Non-native run.
[b] Extirpated.

Appendix D: SRT Risk Scores for Threats

Table D1. Status Review Team risk scores for threats, averaged across all team members. Key: *Su* = summer, *Spr* = spring, *Overutil.* = overutilization, *Inad. reg.* = inadequacy of existing regulations, *D/P* = disease/predation, *Hatch.* = hatchery influence.

			Threats											
			Habitat		Overutil.		Inad. reg.		D/P		Hatch.		Climate	
Area	**Population**	**Run**	**Avg**	**SD**	**Avg**	**SD**	**Avg**	**SD**	**Avg**	**SD**	**Avg**	**SD**	**Avg**	**SD**
Strait	Hoko R	Fall	2.4	0.9	1.7	0.5	1.4	0.5	1.4	0.5	2.6	0.8	2.0	0.0
North Coast	Tsoo-Yess R	Fall	2.0	0.8	1.7	0.4	1.5	0.5	1.5	0.5	3.0	0.7	1.9	0.2
	Sol Duc R[a]	Spr	1.4	0.5	1.4	0.5	1.9	1.2	1.3	0.4	3.1	1.5	2.1	0.5
	Sol Duc R	Fall	1.7	0.6	1.7	0.7	1.2	0.4	1.3	0.4	1.1	0.3	1.8	0.6
	Sol Duc R	Su	1.6	0.7	1.6	0.7	1.2	0.4	1.3	0.4	2.2	0.4	2.1	0.6
	Bogachiel R	Su	1.7	0.7	1.6	0.7	1.2	0.4	1.3	0.4	1.1	0.3	2.1	0.6
	Bogachiel R	Fall	1.8	0.6	1.7	0.7	1.2	0.4	1.3	0.4	1.0	0.0	1.8	0.6
	Calawah R	Su	1.7	0.7	1.6	0.7	1.2	0.4	1.3	0.4	1.1	0.3	2.1	0.6
	Calawah R	Fall	1.8	0.6	1.7	0.7	1.2	0.4	1.3	0.4	1.0	0.0	1.8	0.6
	Dickey R	Fall	1.8	0.6	1.8	0.7	1.2	0.4	1.2	0.4	1.5	0.5	1.9	0.6
	Hoh R	Spr/Su	1.8	0.6	1.6	0.7	1.2	0.4	1.3	0.4	1.0	0.0	2.3	0.8
	Hoh R	Fall	1.8	0.6	1.7	0.7	1.2	0.4	1.3	0.4	1.0	0.0	1.9	0.6
Queets–Quinault	Queets R	Spr/Su	1.7	0.6	1.6	0.7	1.4	0.5	1.4	0.5	1.0	0.0	2.2	0.7
	Queets R	Fall	1.7	0.6	1.7	0.7	1.4	0.5	1.5	0.5	1.8	0.4	1.9	0.6
	Clearwater R	Spr/Su	1.6	0.7	1.6	0.7	1.4	0.5	1.4	0.5	1.0	0.0	2.2	0.7
	Clearwater R	Fall	1.7	0.6	1.7	0.7	1.4	0.5	1.4	0.5	1.1	0.3	1.9	0.6
	Quinault R	Spr/Su	1.6	0.7	1.7	0.7	1.4	0.5	1.4	0.5	1.0	0.0	2.2	0.7
	Quinault R	Fall	1.7	0.7	1.7	0.7	1.4	0.5	1.5	0.5	1.9	0.2	1.9	0.6
Grays Harbor	Chehalis R	Spring	2.5	0.5	1.7	0.7	1.4	0.5	1.8	0.4	1.4	0.5	3.1	0.9
	Chehalis R	Fall	2.4	0.4	1.7	0.7	1.4	0.5	1.8	0.4	1.2	0.4	2.6	0.5
	Wynoochee R[b]	Spr	2.4	1.1	1.9	0.7	1.6	0.5	1.3	0.5	1.0	0.0	2.4	0.5
	Wynoochee R	Fall	2.0	0.5	1.7	0.7	1.4	0.5	1.6	0.5	1.3	0.5	2.2	0.5
	Humptulips R	Fall	2.1	0.5	1.7	0.7	1.4	0.5	1.6	0.5	1.9	0.6	2.3	0.5
	Grays Harbor tributaries	Fall	2.3	0.6	1.7	0.7	1.4	0.5	1.5	0.5	1.3	0.5	2.3	0.5
	Satsop R	Su	2.3	0.7	1.9	0.8	1.5	0.5	1.6	0.5	1.2	0.4	2.4	0.5
	Satsop R	Fall	2.2	0.6	1.7	0.7	1.4	0.5	1.6	0.5	1.8	0.7	2.3	0.5
Willapa Bay	Willapa Bay tributaries	Fall	2.8	0.7	1.8	0.8	1.5	0.7	1.6	0.5	3.1	0.6	2.4	0.5
	WC Chinook Salmon ESU		**1.9**	**0.6**	**1.7**	**0.7**	**1.4**	**0.5**	**1.4**	**0.5**	**1.6**	**0.4**	**2.1**	**0.6**

[a] Non-native run.
[b] Extirpated.

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